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Climate Change Vulnerability and Adaptation in the Intermountain Region

Part 1



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Abstract

The Intermountain Adaptation Partnership (IAP) identified climate change issues relevant to resource management on Federal lands in Nevada, Utah, southern Idaho, eastern California, and western Wyoming, and developed solutions intended to minimize negative effects of climate change and facilitate transition of diverse ecosystems to a warmer climate. U.S. Department of Agriculture Forest Service scientists, Federal resource managers, and stakeholders collaborated over a 2-year period to conduct a state-of-science climate change vulnerability assessment and develop adaptation options for Federal lands. The vulnerability assessment emphasized key resource areas—water, fisheries, vegetation and disturbance, wildlife, recreation, infrastructure, cultural heritage, and ecosystem services—regarded as the most important for ecosystems and human communities.

The earliest and most profound effects of climate change are expected for water resources, the result of declining snowpacks causing higher peak winter streamflows, lower summer flows, and higher stream temperatures. These changes will in turn reduce fish habitat for cold-water fish species, negatively affect riparian vegetation and wildlife, damage roads and other infrastructure, and reduce reliable water supplies for communities. Increased frequency and magnitude of disturbances (drought, insect outbreaks, wildfire) will reduce the area of mature forest, affect wildlife populations (some positively, some negatively), damage infrastructure and cultural resources, degrade the quality of municipal water supplies, and reduce carbon sequestration. Climate change effects on recreation, a major economic driver in the IAP region, will be positive for warm-weather activities and negative for snow-based activities. IAP participants developed adaptation options that can be implemented in planning, project management, monitoring, and restoration as climate-smart responses to altered resource conditions.

Keywords: adaptation, climate change, ecological disturbance, Intermountain Adaptation Partnership, resilience, science-management partnership, vulnerability assessment

Front cover photo: Fall colors and reflection on a pond in Lamoille Canyon, Ruby Mountains District, Humboldt-Toiyabe National Forest, NV, photo by Susan Elliott, U.S. Forest Service.

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Editors

Jessica E. Halofsky is a Research Ecologist with the University of Washington, College of the Environment, School of Environmental and Forest Sciences in Seattle, Washington.

David L. Peterson was a Senior Research Biologist Scientist with the U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station in Seattle, Washington.

Joanne J. Ho is a Research Environmental Economist with the University of Washington, College of the Environment, School of Environmental and Forest Sciences in Seattle, Washington.

Natalie J. Little is the Regional Sustainability and Climate Coordinator, U.S. Department of Agriculture, Forest Service, Intermountain Region in Ogden, Utah.

Linda A. Joyce is a Research Ecologist with the U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Human Dimensions Research Program in Fort Collins, Colorado.

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Summary

The Intermountain Adaptation Partnership (IAP) is a science-management partnership with a wide variety of participants across the U.S. Department of Agriculture, Forest Service Intermountain Region, which spans Nevada, Utah, southern Idaho, eastern California, and western Wyoming. The partnership includes the Forest Service Intermountain Region, and Pacific Northwest and Rocky Mountain Research Stations; National Park Service Climate Change Response Program; North Central Climate Science Center; Desert, Great Basin, Great Northern, and Southern Rockies Landscape Conservation Cooperatives; the University of Washington; Native American tribes; and dozens of other stakeholder organizations. These organizations and other IAP participants worked together over 2 years to identify climate change issues relevant to resource management on Forest Service and National Park Service lands in the IAP region, and to find solutions that could help to minimize the negative effects of climate change and facilitate the transition of ecosystems to a warmer climate. The IAP provided education, conducted a climate change vulnerability assessment, and developed adaptation options for managing resources of the 12 national forests (Ashley, Boise, Bridger-Teton, Caribou-Targhee, Dixie, Fishlake, Humboldt-Toiyabe, Manti-La Sal, Payette, Salmon-Challis, Sawtooth, Uinta-Wasatch-Cache [plus Curlew National Grassland]) and 22 National Park Service units in the IAP region.

The IAP region is characterized by high ecological diversity. Vegetation types include mixed conifer forest, dry ponderosa pine forest, subalpine forest, sagebrush, grasslands, alpine tundra, and wetlands. Ecosystems in the IAP region produce water, fish, timber, wildlife, recreation opportunities, livestock grazing, and other ecosystem services, providing a socioeconomic foundation based on natural resources. The geographic and ecological diversity of the region, especially on Federal lands, contributes significantly to the economic sustainability of human communities, linking Federal resource management with local livelihoods.

The effects of climate change on each resource area in the IAP region are synthesized from the available scientific literature and analyses and are based on available climate change projections (Chapter 3). Highlights of the vulnerability assessment and adaptation options for each resource area are summarized next.

Water and Soil Resources

Climate Change Effects

Lower snowpack and increased drought will result in lower base flows, reduced soil moisture, wetland loss, riparian area reduction or loss, and more frequent and possibly more severe wildfire. April 1 snow water equivalent and mean snow residence time are sensitive to temperature and precipitation variations. Warmer (usually lower elevation) snowpacks are more sensitive to temperature variations, whereas colder (usually higher elevation) snowpacks are more sensitive to precipitation. Warmer locations will experience more runoff in winter months and early spring, whereas colder locations will experience more runoff in late spring and early summer. In both cases, future peakflows will be higher and more frequent.

Lower snowpacks will cause significantly lower streamflow in summer, and reduce the rate of recharge of water supply in some basins. Annual water yields, which are affected by annual precipitation totals (heavily influenced by winter and spring precipitation in the western part of the region) and summer evapotranspiration, will generally be lower. Although declining snowpacks will occur throughout the region, snowpacks at higher elevations (Uinta Mountains, Teton and Wind River Ranges, and some central Idaho ranges) may not change much through the late 21st century. Carbon content in soils will decrease in areas where decomposition rate and wildfire frequency increase, and soil erosion will be accelerated by intense fires.

Adaptation Options

Primary adaptation strategies focus on expanding water conservation; increasing water storage, managing for highly functioning riparian areas, wetlands, and groundwater-dependent ecosystems; and developing policies for water rights. Adaptation tactics include: (1) using drought-tolerant plants for landscaping, managing livestock water improvements efficiently, and educating the public about water resource issues and conservation; (2) decommissioning and improving road systems, improving grazing management practices, and promoting and establishing American beaver populations; (3) managing vegetation to reduce forest density and hazardous fuels; (4) modifying dam and reservoir operation to improve water storage, and improving streamflow and runoff forecasts; and (5) maintaining and protecting soil cover and cryptobiotic crusts, using grazing management systems that promote healthy root systems in plants, and promoting native plant species diversity.

Fish and Other Aquatic Species

Climate Change Effects

A combination of higher stream temperature, low streamflow in summer, and higher peakflow at other times of the year will create a significant stress complex for cold-water fish species. Habitats that provide the restrictive thermal requirements of juvenile bull trout are rare, and little evidence exists for flexibility in habitat use. The length of connected habitat needed to support a bull trout population varies with local conditions, but current estimates suggest a minimum of 20 to 30 miles contingent on water temperature, nonnative species presence, and local geomorphic characteristics. Juvenile cutthroat trout occupy a broader thermal and stream size niche than bull trout. They also appear to persist in smaller habitat patches. Nonetheless, they require cold-water habitat patches exceeding 3 to 6 miles. Increased frequency and extent of extreme events will be especially stressful for bull trout and cutthroat trout, except at higher elevations, where habitat will remain favorable. Both species may in some cases be able to adjust

their life histories to accommodate altered habitat, although the potential for this adaptive capacity is unknown. From the mid- to late-21st century, the vast majority of suitable cold-water fish habitat will be on Federal lands.

Rocky Mountain tailed frogs have long generation times and low fecundity, so increased summer droughts and wildfires, as well as extreme floods and postfire debris flows may threaten some populations. Sensitivities are similar for Idaho giant salamanders. Western pearlshell mussels have a broad geographic range, which reduces their vulnerability, although lower streamflow and higher stream temperatures are expected to be stressful in some locations. Springsnails are expected to be highly vulnerable because they require particular hydrological conditions, specific and stable temperature regimes, and perennial flows. Yosemite toads, already in decline, will be sensitive to reduced duration of ephemeral ponds for breeding in spring. Sierra Nevada yellow-legged frogs will be sensitive to less reliable availability of perennial water bodies needed for multiyear metamorphosis and maturation.

Adaptation Options

Primary adaptation strategies focus on increasing resilience of native fish species by restoring structure and function of streams, riparian areas, and wetlands; monitoring for invasive species and eliminating or controlling invasive populations; understanding and managing for community-level patterns and processes; and conducting biodiversity surveys to describe current baseline conditions and manage for changes in the distribution of fish and other aquatic species. Adaptation tactics include reconnecting floodplains and side channels to improve hyporheic and base flow conditions, ensuring that passage for aquatic organisms is effective, accelerating restoration in riparian areas, maintaining or restoring American beaver populations, managing livestock grazing to restore ecological function of riparian vegetation, removing nonnative fish species, maintaining or increasing habitat connectivity, and increasing the resilience of forests to wildfire.

Vegetation and Ecological Disturbances

Climate Change Effects

Increased temperature is expected to cause a gradual change in the distribution and abundance of dominant plant species. Increased ecological disturbance, driven by higher temperatures, is expected to cause near-term effects on vegetation structure and age classes, and will facilitate long-term changes in dominant vegetation. In forest ecosystems, native and non-native insects are expected to be significant stressors in a warmer climate; in fact, this appears to be already occurring. In all vegetation types, an increase in the frequency and extent of wildfire will be a significant stressor, especially where large fuel accumulations exist. Nonnative plant species will likely continue to expand in most vegetation types, especially in rangelands, potentially displacing native species and altering fire regimes. A combination of these and other stressors (stress complexes), exacerbated by climate, may accelerate the rate of change in vegetation assemblages, and reduce productivity and carbon storage in most systems. Riparian areas may be especially sensitive as a warming climate causes hydrological regimes to change, reducing the timing and amount of water available in summer. Climate change effects on specific forest types include:

- Subalpine pine forest—Most subalpine tree species will be moderately affected by a warmer climate, although bristlecone pine could undergo stress in the driest locations. Whitebark pine will be vulnerable because it is already stressed from white pine blister rust and mountain pine beetles. If wildfire increases, crown fires may quickly eliminate mature trees across the landscape.

- Subalpine spruce-fir forest—This forest type will be moderately vulnerable. Subalpine fir and Engelmann spruce may have increased growth in a longer growing season. Bark beetles will be a stressor for Engelmann spruce. If wildfire increases, crown fires may quickly eliminate mature trees across the landscape. Quaking aspen will be minimally affected by a warmer climate.

- Mesic mixed conifer forest—Late-seral forests will be susceptible to wildfire, especially where fuel loads are high. Douglas-fir, ponderosa pine, and Jeffrey pine, which have high fire tolerance, may become more common, and late-seral species less common. Growth rates of most species will decrease. Lodgepole pine and quaking aspen will persist, perhaps with increased stress from insects and pathogens.

- Dry mixed conifer forest—Most species in mixed conifer forest (ponderosa pine, Gambel oak, quaking aspen) can cope with dry soils and wildfire. Growth of less drought-tolerant species (Douglas-fir, white fir) will decrease. With increased fire frequency, early-seral species will become more common, and late-seral species less common.

- Aspen mixed conifer forest—Increased wildfire frequency and extent will determine future composition and structure of this forest type. Conifers at higher elevations (mostly not fire resistant) will become less common, confined to northern slopes and valley bottoms. Quaking aspen and Gambel oak will attain increasing dominance because of their ability to sprout vigorously after fire, outcompeting species susceptible to drought and fire.

- Persistent aspen forest—Conifers at higher elevation (mostly not fire resistant) will become less common, confined to northern slopes and valley bottoms. Quaking aspen will attain increasing dominance because of its ability to sprout vigorously after fire, outcompeting species susceptible to drought and fire. Douglas-fir will persist in locations with sufficient soil moisture. Overall productivity will probably decrease.

- Montane pine forest—Ponderosa pine will persist in this forest type because it is drought tolerant and fire tolerant, outcompeting other species following wildfire, but will grow more slowly. Limber pine and bristlecone pine will probably persist at higher elevations where fuel loads are low. If insect outbreaks are more prevalent in a warmer climate, they could increase stress in pine species, especially during drought.

- Riparian forest—This is a highly vulnerable forest type because it depends on a reliable water supply. Vegetation dominance may shift to species that are more tolerant of seasonal drought, including ponderosa pine and other deep-rooted conifers. Hardwoods could become less common. Riparian forests associated with small or transient water sources will be especially vulnerable, especially at lower elevations.

Nonforest

In nonforest ecosystems, increasing frequency and duration of drought are expected to drive direct changes on soil moisture, which will reduce the vigor of some species, causing mortality or making (mostly woody species) more susceptible to insects and pathogens. Increasing frequency and extent of wildfire will be a major stressor for species that regenerate slowly following fire, especially non-sprouting vegetation (e.g., most sagebrush species). The dominance of nonnative plant species, especially annual grasses (e.g., cheatgrass), will be enhanced by increasing disturbance and will themselves encourage more frequent fire—a significant change in the ecology of most vegetation assemblages. Although productivity may increase in some grasslands, most other nonforest ecosystems will experience lower productivity. Most native species are expected to persist if they can move to favorable portions of the landscape and are sufficiently competitive. Climate change effects on specific nonforest vegetation include:

- Pinyon-juniper shrublands and woodlands—These woodlands are sensitive to chronic low soil moisture during prolonged droughts (to which pinyon pines are more sensitive than junipers), increased insect outbreaks that follow drought stress, and increased frequency and extent of wildfire. These species will persist across the landscape, although the distribution and abundance of species may change.

- Oak-maple woodlands—Gambel oak and bigtooth maple, the dominant species in these woodlands, are widely distributed and both sprout heavily following wildfire. As a result, their vulnerability is expected to be relatively low, and Gambel oak in particular may become more dominant as wildfire frequency and extent increase across the landscape.

- Mountain mahogany woodlands—These woodlands, which are dominated by curl-leaf mountain mahogany, are expected to be moderately vulnerable. This species is slow-growing and does not sprout following wildfire, so regeneration of disturbed sites may be slow, especially where nonnative species are common. However, mountain mahogany is capable of growing on low-fertility soils, so it will continue to be competitive with other species.

- Mountain big sagebrush shrublands—Vulnerability varies from moderate to high because of the broad elevation range at which mountain big sagebrush occurs, and because of the wide range in current conditions. Livestock

grazing, expansion of pinyon pine and juniper species, altered wildfire regimes, and nonnative invasive species are significant stressors. These factors may be exacerbated by a warmer climate, especially in drier habitats.

- Dry big sagebrush shrublands—Vulnerability is high, as evidenced by significant mortality that occurred during recent drought. Conditions suitable for seedling establishment are infrequent under current climatic conditions and are likely to become less frequent in a warmer climate. Lower elevations of the Great Basin are especially vulnerable, whereas sagebrush in wetter locations may be able to persist.
- Sprouting sagebrush shrublands—Warmer, drier climate will negatively affect the vigor and abundance of sprouting sagebrush species, which are adapted to more mesic conditions. These species can sprout following wildfire, but seed viability is short and unreliability of spring soil moisture will make them susceptible to prolonged droughts. Overall vulnerability is moderate, and regeneration will be critical to long-term persistence across the landscape.
- Dwarf sagebrush shrublands—All low-growing sagebrush species are likely to be negatively affected by higher temperatures and increased periods of drought. Seed viability is short and their dependence on spring soil moisture will make them susceptible to prolonged droughts and to altered timing and amount of spring moisture. Increased wildfire frequency, coupled with drought, could inhibit regeneration on drier sites.
- Mountain, blackbrush, and salt desert shrublands—These shrublands have low to moderate vulnerability, depending on their location relative to soil moisture availability. Many of these shrublands have relatively high species diversity—some are well-adapted to periodic drought and some may be able to migrate to higher elevations. Salt desert communities at lower elevations may be vulnerable to drought and are intolerant of wildfire.
- Alpine communities—The composition and distribution of alpine ecosystems will be affected by decreasing snowpack, altering plant vigor and regeneration. Specific effects will depend on vulnerability thresholds of diverse species and the rate and magnitude of changes over time. Some species may be able to persist or migrate to suitable habitat, but the lower extent of some communities will be compromised by tree establishment.
- Mountain grasslands—The vulnerability of cool-season grass-dominated communities is moderate to high. Warm-season grasses are favored by higher temperatures, providing an opportunity for spread into mountain grasslands from lower-elevation and more southern locations. Increased wildfire frequency will facilitate more nonnative invasive species, decreasing the dominance and vigor of natives.
- Subalpine forb communities—Higher temperatures and increasing drought make this vegetation type highly vulnerable in many locations. Although some subalpine forb communities may be able to move higher in elevation, shallow soil profiles may support only lower-growing species. Tall forb communities at the highest elevations on plateaus (e.g., Wasatch Plateau) are particularly vulnerable.
- Riparian and wetland communities—Most of these communities are highly vulnerable, especially those at lower elevations where soil conditions are already affected by periodic drought. Reduced summer streamflow and groundwater will create significant stress for some dominant plant species, although high species diversity in many locations ensures some long-term persistence, perhaps with lower functionality.

Adaptation Options

Primary adaptation strategies for forest vegetation focus on promoting disturbance-resilient species, maintaining low tree densities, promoting species and genetic diversity, promoting diversity of forest structure, and increasing knowledge about climate change effects for agency land managers and stakeholders. Tactics include conducting thinning treatments, favoring disturbance-resilient species in thinnings, planting potential microsites with a mixture of species, collecting seed for postfire reforestation, and reducing density through prescribed fire and managed wildfire. Maintaining and restoring stream channels, and protecting vegetation through appropriate livestock management can be applied in riparian areas.

Primary adaptation strategies for nonforest vegetation focus on restoring resilience to and maintaining healthy and intact woodlands, shrublands, and grasslands, increasing management actions to prevent invasive species,

and maintaining and restoring natural habitat. Tactics include using mechanical treatments, prescribed fire, using integrated weed management, implementing fuels reduction projects, using ecologically based invasive plant management, implementing livestock management that reduces damage to native perennial species, and maintaining or improving native plant cover, vigor, and species richness.

Terrestrial Animals

Climate Change Effects

The effects of climate change on terrestrial animal species are expected to be highly variable, depending on habitat conditions in specific locations and on the flexibility of animal life histories to accommodate altered conditions. Flammulated owl, wolverine, and greater sage-grouse are expected to be the most vulnerable to population declines, whereas Utah prairie dog and American three-toed woodpecker will be the least vulnerable. Most species will exhibit some sensitivity to altered phenology, habitat, and physiology. Species restricted to high elevations or surface water habitats will generally be vulnerable. Following are possible climate change effects on species of conservation concern.

- Black rosy finch—An alpine specialist, this species will suffer loss of habitat associated with shrinking snowfields and glaciers and possibly encroaching tree establishment, although it does have the capacity to migrate to other locations.
- Flammulated owl—Wildfire and insects will increase early-seral forest structure over time, conditions detrimental for this species, which prefers mature, open ponderosa pine and other semiarid forests with brushy understories.
- Greater sage-grouse—Degraded habitat caused by wildfire-induced mortality of mature sagebrush, in combination with increased dominance of pinyon-juniper woodlands, invasive annual species, and possible effects of West Nile virus will be significant challenges to this species.
- White-headed woodpecker—As long as sufficient mature coniferous forest habitat with pines as a seed source and dead trees for nesting remain, this species will be relatively resilient to a warmer climate because it can move readily to more favorable locations.
- American pika—This species will be vulnerable on isolated mountaintops and at low elevations where it is near its physiological tolerance. Populations in the southern Great Basin are the most vulnerable in the IAP region, but populations in other locations may be fairly resilient.
- Bighorn sheep—Different parts of the region, and thus different subspecies, will be subject to different population dynamics. Populations in the most arid, low-elevation locations and without access to dependable springs and forage will be most vulnerable.
- Canada lynx—This species will be vulnerable to reduced snowpack and prey availability (especially snowshoe hares), although interactions among climate, wildfire, and insect outbreaks may reduce late-seral forest habitat preferred for breeding.
- Fisher—The extent, quality, and connectivity of habitat for this species will probably decrease as increasing wildfire reduces late-seral forest habitat, although fishers can readily move from unfavorable to favorable habitat.
- Fringed myotis—This species could undergo some stress if water sources become less common or more transient, although its mobility and migratory nature allow it to respond to changing conditions.
- Northern Idaho ground squirrel—Increased vegetative productivity may benefit this species, although loss of snowpack, drought, disease, and nonclimatic factors (overgrazing, land development) may be significant stressors.

- Sierra Nevada red fox—With populations that are mostly small and isolated, this species may be affected by drought, wildfire, and insects that alter vegetation, and especially by reduced snowpack, which promotes higher populations of coyotes, a competitor for limited prey.
- Townsend's big-eared bat—This species uses a variety of habitats, conferring some resilience, although increasing wildfires and nonnative grasses could degrade habitats and reduce prey availability. Declining snowpack may also reduce the number and duration of water sources.
- Utah prairie dog—This species may be fairly resilient to a warmer climate, although population declines have been observed during prolonged periods of drought, which affects food and water availability.
- Wolverine—This species, already low in numbers, could be significantly affected by declining snowpack in its preferred high-elevation forest and alpine habitats, and possibly by altered vegetation composition over time.
- Boreal toad—Subject to recent population declines, this species is sensitive to water balance, so altered timing and duration of water availability could be stressors. The harmful chytrid fungus may or may not be affected by climate change, and trampling of riparian areas by livestock is locally damaging.
- Columbia spotted frog—Historical declines of this species may be exacerbated by alteration and fragmentation of aquatic habitats. Drought, warmer temperatures, and reduced snowpack will potentially alter breeding habitat, although spotted frogs will probably be resilient in areas with reliable water sources.
- Great Basin spadefoot—This species may be fairly resilient to a warmer climate because it occurs in a variety of vegetation types, has a flexible breeding season, and has high reproductive rates. Populations in the southern portion of its range and where it relies on ephemeral ponds may be more vulnerable.
- Prairie rattlesnake—This species has low fecundity, long generation times, and low dispersal, making it vulnerable to additional climate stresses such as wildfires and flooding. It will probably be more resilient in areas with sufficient microhabitats and low habitat fragmentation.

Adaptation Options

Primary adaptation strategies focus on improving riparian habitat through restoration, encouraging healthy beaver populations, retaining mature forest structure where possible, reducing nonnative plant species, maintaining quaking aspen habitat, and maintaining connectivity of habitat patches across the landscape. Adaptation tactics include removing hazardous fuels to reduce wildfire intensities, minimizing impacts from livestock grazing, using prescribed fire and conifer removal to promote aspen stands, removing cheatgrass and other invasive species from sagebrush systems, and minimizing impacts of recreation on species sensitive to human disturbance.

Outdoor Recreation

Climate Change Effects

Summer recreation (hiking, camping, bicycling) will benefit from a longer period of suitable weather without snow, especially during the spring and fall shoulder seasons. **Snow-based recreation** (downhill skiing, cross-country skiing, snowmobiling) will be negatively affected by a warmer climate because of less snow and more transient snowpacks. Ski areas and other facilities at lower elevations will be especially vulnerable. **Hunting and fishing** may be affected somewhat by a warmer climate, depending on specific location and activity. Hunting will be sensitive to temperature during the allotted hunting season and timing and amount of snow. Fishing will be sensitive to streamflows and stream temperatures associated with target species; if summer flows are very low, some streams may be closed to fishing. **Water-based recreation** (swimming, boating, rafting) will be sensitive to lower water levels. **Gathering forest products** for recreational and personal use (e.g., huckleberries, mushrooms) will be somewhat sensitive to the climatic conditions that support the distribution and abundance of target species, and to extreme temperatures and increased occurrence of extreme events (e.g., flooding, landslides).

Adaptation Options

Recreation participants are highly adaptable to changing conditions, although Federal agencies are not very flexible in modifying management. Primary adaptation strategies focus on transitioning management to shorter winter recreation seasons, providing sustainable recreation opportunities, increasing management flexibility and facilitating transitions to meet user demand and expectations, and managing recreation sites to mitigate risks to public safety and infrastructure. Adaptation tactics include collecting data on changing use patterns and demands, maintaining current infrastructure and expanding facilities in areas where concentrated use increases, educating the public about changing resource conditions, varying the permit season for rafting to adapt to changes in peak flow and duration, and determining which recreation sites are at risk from increased hazards.

Infrastructure

Climate Change Effects

Vulnerability of infrastructure can be assessed at three levels: (1) documentation of the type and quantity of infrastructure, (2) examination of infrastructure investments at the regional level, and (3) evaluation of infrastructure at local or smaller scales. Infrastructure risk can be proactively addressed by identifying assets that have a high likelihood of being affected by future climatic conditions and significant consequences if changes do occur. Roads and other infrastructure that are near or beyond their design life are at considerable risk to damage from flooding and geomorphic disturbance (e.g., debris slides). If road damage increases as expected, it will have a profound impact on access to Federal lands and on repair costs. Trails and developed recreation sites may also be sensitive to increased flooding and chronic surface flow, especially in floodplains. Buildings and dams represent large investments, and some may be at risk to an increased frequency of extreme events (wildfire, flooding).

Adaptation Options

Primary adaptation strategies focus on maintaining an accurate inventory of at-risk infrastructure components (e.g., buildings, roads), increasing resilience of the transportation system to increased disturbances (especially flooding), and ensuring that design standards are durable under the new conditions imposed by a warmer climate. Adaptation tactics include improving roads and drainage systems to survive higher peakflows and more flooding, conducting risk assessments of vulnerable roads and infrastructure, decommissioning roads where appropriate, documenting seasonal traffic patterns, emphasizing potential increases in extreme storm events when evaluating infrastructure inventory, fireproofing of buildings, and coordinating with partners whenever possible.

Cultural Resources

Climate Change Effects

Some aspects of climate change may exacerbate damage and loss of cultural resources, which are threatened by natural biophysical factors as well as human behaviors such as vandalism and illegal artifact digging. Increasing wildfire, flooding, melting of snowfields, and erosion can quickly displace or destroy artifacts before they have been identified and examined, potentially leading to the loss of thousands of items. In addition, large disturbances can change the condition of vegetation, streams, and other landscape features valued by Native Americans.

Adaptation Options

Adaptation strategies and tactics to protect cultural resources include improving inventories of the location of cultural resources, suppressing wildfires to protect specific sites, implementing fuels treatments in dry forests to reduce wildfire intensity, implementing protection strategies (e.g., stabilization, armoring, fireproofing) in areas prone to disturbances, monitoring areas affected by flooding and debris flows in mountain canyon and foothill areas, and applying vegetation management treatments designed to protect “first food” resources.

Ecosystem Services

Climate Change Effects

Ecosystem services provided to human communities from Federal lands will be affected by climate change in several ways:

- Timber and related products and services—Reduced growth rates in primary timber species will have a minimal effect on harvestable wood volume, although increased wildfires and insect outbreaks can reduce harvestable timber supply. Economic forces and policies will continue to dominate the wood products industry and employment, regardless of climate change.
- Grazing forage for domestic livestock and wildlife—Productivity may increase in some rangelands and decrease in others, so effects will vary spatially. Increased dominance of nonnative species (e.g., cheatgrass) will reduce range quality and support more frequent wildfires. Local erosion and encroaching urbanization will reduce the amount of available forage, regardless of climate change.
- Water quantity and quality—Declining snowpack will alter hydrological regimes annually and seasonally. Water yield is expected to decrease significantly by the 2040s and considerably more by the 2080s. The most sensitive watersheds are those already impaired or at risk, based on vegetation and soil conditions. Water quality may be affected by algal blooms and by erosion following wildfires.
- Ecosystem carbon—Ecosystems will increasingly be affected by disturbances (drought, wildfires, insects) that will remove living and dead vegetation, and, in turn, reduce carbon sequestration. If fires are as frequent as expected, forests may rarely attain a mature stand structure at lower elevations, thus limiting potential carbon sequestration.
- Pollination—Altered temperature and precipitation may lead to variable flowering phenology, which could reduce pollination by native insects such as bumblebees, and reduce native plant reproduction. Increased drought and extreme temperatures may impact pollinators already under stress from insecticides and increased dominance by nonnative plants.

Adaptation Options

Adaptation strategies for ecosystem services focus on availability and quality of forage for livestock, availability and quality of water, and habitat for pollinators. Adaptation strategies for grazing focus on increasing resilience of rangeland vegetation, primarily through nonnative species control and prevention. Adaptation tactics include flexibility in timing, duration, and intensity of authorized grazing as a tactic to prevent ecosystem degradation under changing conditions, as well as a more collaborative approach to grazing management.

Adaptation strategies for water focus on timing of water availability and quality of water delivered beyond Federal lands, assessments of potential climate change effects on municipal water supplies, and identifying potential vulnerabilities to help facilitate adaptive actions. Adaptation tactics include reducing hazardous fuels in dry forests to reduce the risk of crown fires, reducing other types of disturbances (e.g., off-road vehicles, unregulated livestock grazing), and using road management practices that reduce erosion.

Adaptation strategies for pollinators focus on improving pollinator habitat by increasing native vegetation and by applying pollinator-friendly best management. Adaptation tactics include establishing a reserve of native seed mixes for pollinator-friendly plants, implementing revegetation with plants beneficial to both pollinators and wildlife, and creating guidelines that would help managers incorporate pollinator services in planning, project analysis, and decisionmaking.

Conclusions

The IAP facilitated the most comprehensive effort on climate change assessment and adaptation in the United States, including participants from stakeholder organizations interested in a broad range of resource issues. It achieved specific elements of national climate change strategies for the U.S. Forest Service and National Park Service, providing a scientific foundation for resource management, planning, and ecological restoration in the IAP region. The large number of adaptation strategies and tactics, many of which are a component of current management practice, provides a pathway for slowing the rate of deleterious change in resource conditions. Rapid implementation of adaptation as a component of sustainable resource management will help to maintain critical structure and function of terrestrial and aquatic ecosystems in the IAP region. Long-term monitoring will help to detect potential climate change effects on natural resources, and evaluate the effectiveness of adaptation options that have been implemented.

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Figure 1.1—Project area for the Intermountain Adaptation Partnership.

Chapter 1: Introduction

Joanne J. Ho

Introduction

The Intermountain Adaptation Partnership (IAP) is a science-management partnership with a wide variety of participants across the U.S. Department of Agriculture Forest Service (USFS) Intermountain Region, which spans Nevada, Utah, southern Idaho, eastern California, and western Wyoming. This USFS region is the largest in the Nation, representing nearly 17 percent of all National Forest System lands. The partnership includes the USFS Intermountain Region and the USFS Pacific Northwest and Rocky Mountain Research Stations, National Park Service (NPS) Climate Change Response Program, U.S. Fish and Wildlife Service, North Central Climate Science Center, the Desert, Great Basin, Great Northern, and Southern Rockies Landscape Conservation Cooperatives, the University of Washington, Native American tribes, and dozens of other stakeholder organizations (fig. 1.1, box 1.1). Initiated in 2015, the IAP is a collaborative project with the goals of increasing climate change awareness, assessing vulnerability, and developing science-based adaptation options to reduce adverse effects of climate change and ease the transition to new climate states and conditions (see http://adaptationpartners.org/iap). Developed in response to proactive climate change strategies of the USFS (USDA FS 2008,

Box 1.1—Intermountain Adaptation Partnership Participating Organizations

Backcountry Hunters and Anglers
Boise State University
Bureau of Land Management (BLM)
Colorado State University
Cooperative Institute for Research in Environmental Sciences
Deseret News
Desert Landscape Conservation Cooperative
Eastern Idaho Public Health
EcoAdapt
Grand Canyon Trust
Great Basin Landscape Conservation Cooperative
Great Northern Landscape Conservation Cooperative
Henry's Fork Foundation
Idaho Army National Guard
Idaho Bureau of Homeland Security
Idaho Department of Fish and Game
Idaho National Guard
Idaho Power Company
Institute of Outdoor Recreation and Tourism
Inter-Tribal Council of Nevada
McGinnis and Associates
National Oceanic and Atmospheric Administration
National Park Service
National Weather Service
Natural Resources Conservation Service
Nevada Department of Wildlife
Nez Perce Tribe
North Central Climate Science Center
Northwest Watershed Research Center
Southern Rockies Landscape Conservation Cooperative
Trout Unlimited
U.S. Fish and Wildlife Service
U.S. Forest Service
U.S. Forest Service Intermountain Region
U.S. Forest Service Pacific Northwest and Rocky Mountain Research Stations
University of Nevada, Reno
University of Utah
University of Washington
Utah Division of Forestry, Fire and State Lands
Utah State University
Utah's Hogle Zoo
Weber State University
Western Water Assessment
Wild Utah Project
Yerington Paiute Tribe
Yomba Shoshone Tribe

2010a,b), and building on previous efforts in national forests (Halofsky and Peterson 2017; Halofsky et al. 2011, 2018, in press; Littell et al. 2012; Raymond et al. 2013, 2014; Rice et al. 2012; Swanston et al. 2011, 2016), the partnership brings together resource managers, research scientists, and stakeholders to plan for climate change in the Intermountain Region.

This effort directly addressed goals identified in the USFS Intermountain Region Strategic Framework for FY 2017–2020 (USDA FS 2016) and the USFS Strategic Plan, FY 2015–2020 (USDA FS 2015). These main strategic goals are to: (1) sustain our Nation's forests and grasslands, (2) deliver benefits to the public, (3) apply knowledge globally, and (4) excel as a high-performing agency. These goals aim the USFS toward success in the agency's mission "to sustain the health, diversity, and productivity of the Nation's forests and grasslands to meet the needs of present and future generations." Finally, this assessment strives to provide options and solutions to a complex challenge in a way inspired by Gifford Pinchot, the first Chief of the USFS: for "the greatest good of the greatest number in the long run," (USDA FS 2007).

Climate Change Response in the Forest Service and National Park Service

Climate change is an agency-wide priority for the USFS, which has issued direction to administrative units for responding to climate change (USDA FS 2008) (table 1.1). In 2010, the USFS provided specific direction to the National Forest System in the form of the National Roadmap for Responding to Climate Change (USDA FS 2010a) and the Performance Scorecard for Implementing the USFS Climate Change Strategy (USDA FS 2010b). The goal of the USFS climate change strategy is to "ensure our national forests and private working lands are conserved, restored, and made more resilient to climate change, while enhancing our water resources," (USDA FS 2010a, p. 2). To achieve this goal, starting in fiscal year (FY) 2011, each national forest and grassland began using a 10-point scorecard system to report accomplishments on 10 elements in 4 dimensions: (1) increasing organizational capacity, (2) partnerships, engagement, and education, (3) adaptation, and (4) mitigation and sustainable consumption. The scorecard elements can be found in box 1.2. From FY 2011 to FY 2016, progress toward accomplishing elements of the scorecard was required to be reported annually by each national forest and national grassland. All units were expected to accomplish 7 of 10 criteria, with at least one "yes" in each dimension, and in FY 2016, all units in the Intermountain Region were successful in this endeavor. As of FY 2017, the USFS is actively developing a new reporting model.

Similarly, the NPS Climate Change Response Strategy provides direction for addressing the impacts of climate change on NPS lands (NPS 2010) (table 1.2). The strategy has four components to guide NPS actions: science, adaptation, mitigation, and communication. The science component involves conducting and synthesizing research at various scales, monitoring trends and conditions, and delivering information to resource managers and partners. It also provides the scientific basis for adaptation, mitigation, and communication. Adaptation involves developing capacity within the agency to assess climate change scenarios and risks and implementing actions to better manage natural and cultural resources and infrastructure for a changing climate. Mitigation efforts focus on reducing the agency carbon footprint and enhancing carbon sequestration. Finally, the strategy requires the NPS to take advantage of agency capacity for education and interpretation to communicate the effects of climate change to NPS employees and to the public. Park rangers and other employees are encouraged to engage visitors about climate change because national parks are visible examples of how climate change can affect natural and cultural resources. The similarity of USFS and NPS climate response strategies facilitated coordination between the two agencies.

The IAP built on previous Adaptation Partners (www.adaptationpartners.org) efforts in ecosystem-based management and ecological restoration to address climate change and put these efforts in a broader regional context in the Intermountain Region. Starting in 2008, Halofsky et al. (2011) conducted a climate change assessment for Olympic National Forest and Olympic National Park (1.55 million acres), a science-management collaboration initiated to develop climate adaptation strategies. In 2010, the North Cascadia Adaptation Partnership (Raymond et al. 2014) began a similar effort with an expanded geographic scope of two national forests and two national parks. These organizations worked with stakeholders over 2 years to identify climate change issues relevant to resource management in the North Cascades to assist in the transition of diverse ecosystems of the region to a warmer climate. The North Cascadia Adaptation Partnership provided education, conducted a climate change vulnerability assessment, and developed adaptation options for the Federal agencies that manage nearly 6 million acres in north-central Washington. In 2013, the USFS Pacific Northwest Research Station, Pacific Northwest Region, and Malheur, Umatilla, and Wallowa-Whitman National Forests (5.29 million acres in Oregon and Washington) initiated the Blue Mountains Adaptation Partnership (Halofsky and Peterson 2017). Formed in 2015, the South Central Oregon Adaptation Partnership (Halofsky et al. in press) brought together the Deschutes National Forest, Fremont-Winema National Forest, Ochoco National Forest, Crooked River National Grassland, the USFS Pacific Northwest Region, USFS Pacific Northwest and Rocky Mountain Research Stations, Crater Lake National Park, and the University of Washington to conduct a similar climate change assessment, covering 5 million acres. In the largest effort to date in the western United States, the Northern Rockies Adaptation

Table 1.1—U.S. Forest Service policies related to climate change.

Policy	Description
Forest Service Strategic Framework for Responding to Climate Change (USDA FS 2008)	Developed in 2008, the Strategic Framework is based on seven strategic goals in three broad categories: foundational, structural, and action. The seven goals are science, education, policy, alliances, adaptation, mitigation, and sustainable operations. Like the challenges themselves, the goals are interconnected; actions that achieve one goal tend to help meet other goals. The key is to coordinate approaches to each goal as complementary parts of a coherent response to climate change. All seven goals are ultimately designed to achieve the same end (the USFS mission): to ensure that Americans continue to benefit from ecosystem services from national forests and grasslands.
USDA 2010-2015 Strategic Plan (USDA FS 2010c)	In June 2010, the U.S. Department of Agriculture released the Strategic Plan that guides its agencies toward achieving several goals including Strategic Goal 2—Ensure our national forests and private working lands are conserved, restored, and made more resilient to climate change, while enhancing our water resources. This goal has several objectives. Objective 2.2 is to lead efforts to mitigate and adapt to climate change. The performance measures under this objective seek to reduce greenhouse gas emissions by the U.S. agricultural sector, increase the amount of carbon sequestered on U.S. lands, and bring all national forests into compliance with a climate change adaptation and mitigation strategy. The USFS response to this goal includes the National Roadmap for Responding to Climate Change and Performance Scorecard (Roadmap).
National Roadmap for Responding to Climate Change (USDA FS 2010a)	Developed in 2011, the Roadmap integrates land management, outreach, and sustainable operations accounting. It focuses on three kinds of activities: assessing current risks, vulnerabilities, policies, and gaps in knowledge; engaging partners in seeking solutions and learning from as well as educating the public and employees on climate change issues; and managing for resilience in ecosystems and human communities through adaptation, mitigation, and sustainable consumption strategies.
Climate Change Performance Scorecard (USDA FS 2010b)	To implement the Roadmap, starting in 2011, each national forest and grassland began using a 10-point scorecard to report accomplishments and plans for improvement on 10 questions in four dimensions: organizational capacity, engagement, adaptation, and mitigation. By 2015, each is expected to answer "yes" to at least seven of the scorecard questions, with at least one "yes" in each dimension. The goal is to create a balanced approach to climate change that includes managing forests and grasslands to adapt to changing conditions, mitigating climate change, building partnerships across boundaries, and preparing employees to understand and apply emerging science.
2012 Planning Rule (USDA FS 2012)	The 2012 Planning Rule is based on a planning framework that will facilitate adaptation to changing conditions and improvement in management based on new information and monitoring. There are specific requirements for addressing climate change in each phase of the planning framework, including in the assessment and monitoring phases, and in developing, revising, or amending plans. The 2012 Planning Rule emphasizes restoring the function, structure, composition, and connectivity of ecosystems and watersheds to adapt to the effects of a changing climate and other ecosystem drivers and stressors, such as wildfire and insect outbreaks. A baseline assessment of carbon stocks required in assessment and monitoring will check for measureable changes in the plan area related to climate change and other stressors. Requirements of the Roadmap and Scorecard and requirements of the 2012 Planning Rule are mutually supportive and provide a framework for responding to changing conditions over time.

Box 1.2—The Forest Service Climate Change Performance Scorecard, 2011

1. **Employee Education.** Are all employees provided with training on the basics of climate change, impacts on forests and grasslands, and the Forest Service response? Are resource specialists made aware of the potential contribution of their own work to climate change response?
2. **Designated Climate Change Coordinators.** Is at least one employee assigned to coordinate climate change activities and be a resource for climate change questions and issues? Is this employee provided with the training, time, and resources to make his/her assignment successful?
3. **Program Guidance.** Does the Unit have written guidance for progressively integrating climate change considerations and activities into Unit-level operations?
4. **Science and Management Partnerships.** Does the Unit actively engage with scientists and scientific organizations to improve its ability to respond to climate change?
5. **Other Partnerships.** Have climate change related considerations and activities been incorporated into existing or new partnerships (other than science partnerships)?
6. **Assessing Vulnerability.** Has the Unit engaged in developing relevant information about the vulnerability of key resources, such as human communities and ecosystem elements, to the impacts of climate change?
7. **Adaptation Actions.** Does the Unit conduct management actions that reduce the vulnerability of resources and places to climate change?
8. **Monitoring.** Is monitoring being conducted to track climate change impacts and the effectiveness of adaptation activities?
9. **Carbon Assessment and Stewardship.** Does the Unit have a baseline assessment of carbon stocks and an assessment of the influence of disturbance and management activities on these stocks? Is the Unit integrating carbon stewardship with the management of other benefits being provided by the Unit?
10. **Sustainable Operations.** Is progress being made toward achieving sustainable operations requirements to reduce the environmental footprint and increase the resilience of agency operations and assets?

Partnership developed a vulnerability assessment and adaptation options for 15 national forests and 3 national parks in Montana, northern Idaho, North Dakota, and parts of South Dakota and Wyoming, covering a total of 183 million acres (Halofsky et al. 2018). The IAP continues these efforts to develop science-based adaptation strategies.

Other efforts have also demonstrated the success of science-management partnerships for increasing climate change awareness among resource managers and adaptation planning on Federal lands. In addition to the Adaptation Partners assessments described earlier, Tahoe National Forest, Inyo National Forest, and Devils Postpile National Monument worked with the USFS Pacific Southwest Research Station to develop climate change vulnerability assessments (Littell et al. 2012) and the Climate Project Screening Tool (Morelli et al. 2012) in order to incorporate adaptation into project planning. In response to requests from Shoshone National Forest in northern Wyoming, the Rocky Mountain Research Station synthesized information on past climate, future climate projections, and potential effects of climate change on the multiple ecosystems within the forest (Rice et al. 2012).

In the largest effort to date in the eastern United States, the USFS Northern Research Station, in collaboration with Chequamegon-Nicolet National Forest in northern Wisconsin and numerous other partners, conducted a vulnerability assessment for natural resources (Swanston et al. 2011) and developed adaptation options (Swanston et al. 2016). Another USFS science-management partnership assessed the vulnerability of watersheds to climate change (Furniss et al. 2013). These watershed vulnerability assessments, conducted on 11 national forests throughout the United States, were locally focused (at a national forest scale) and included water resource values, hydrological reaction to climate change, watershed condition, and landscape sensitivity. The assessments were intended to help national forest managers identify where limited resources could be best invested to increase watershed resilience to climate change. More recently, Butler et al. (2015) conducted a climate change vulnerability assessment and synthesis for forest ecosystems of the Central Appalachians region.

The processes, products, and techniques used for several studies and other climate change efforts on national forests have been compiled in a guidebook for developing adaptation options for national forests (Peterson et al. 2011). The guidebook outlines four key steps to facilitate adaptation in national forests: (1) Become aware of basic climate change science and integrate that understanding with knowledge of local conditions and issues (review), (2) evaluate sensitivity of natural resources to climate change (rank), (3) develop and implement options for adapting resources to climate change (resolve), and (4) monitor the effectiveness of

Table 1.2—National Park Service policies related to climate change.

Policy	Description
National Park Service Climate Change Response Strategy (NPS 2010)	Developed in 2010, the Climate Change Response Strategy is designed to guide management actions and collaboration, from the national to park levels, to address the effects of climate change. The Response Strategy is based on four components: science, mitigation, adaptation, and communication. These components provide a framework for consistent, legal, and appropriate management decisions. The Response Strategy calls for a scientific approach to updating interpretations of previous policy and mandates in order to uphold the mission of the NPS in the face of new conditions created by climate change.
A Call to Action: Preparing for a Second Century of Stewardship and Engagement (NPS 2011)	The Call to Action outlines themes and goals for the second century of stewardship and engagement of the NPS. The plan provides actions for the achievement of each goal before the NPS centennial in 2016. Under the theme of preserving America's special places, the plan sets the goal for management of resources to increase resilience to climate change stressors. Specific actions include revised management objectives, increases in sustainability, and changes in investments.
Green Parks Plan (NPS 2012b)	The Green Parks Plan (GPP) outlines how the NPS will achieve the commitment set in A Call to Action, to "Go Green." An overarching vision and strategy for sustainable management in the future, the GPP is based on nine strategic goals that focus on the effects of park operations on the environment and human welfare. The goals are to continually improve environmental performance; be climate friendly and climate ready; be energy smart; be water wise; develop a green NPS transportation system, buy green and reduce, reuse, and recycle; preserve outdoor values; adopt best practices; and foster sustainability beyond NPS boundaries.
Revisiting Leopold: Resource Stewardship in the National Parks (NPS 2012c)	In August 2012, the NPS released the Revisiting Leopold, intended as an updated interpretation of the guiding document, The Leopold Report (Leopold et al. 1963). Members of the current NPS Science Committee were tasked with revisiting three questions: (1) What should be the goals of resource management in the National Parks? (2) Which policies for resource management are necessary to achieve these goals? (3) Which actions are required to implement these policies? The interpretation presents general principles and guidance for the enlarged scope of all natural and cultural resources of the NPS. The committee stresses that the NPS needs to act quickly on structural changes and long-term investments in management in order to preserve resources through the uncertainties of environmental change.
Climate Change Action Plan 2012-2014 (NPS 2012a)	The 2012 Climate Change Action Plan builds on the 2010 NPS Climate Change Response Strategy to communicate how the NPS can respond to climate change at different geographic scales. The plan outlines parameters for introducing science, adaptation, mitigation, and communication actions to address climate change. The plan also identifies high-priority actions for addressing climate change in NPS operations, and describes how to anticipate and prepare for future changes.

on-the-ground management (observe) and adjust as needed. The IAP is focused on implementation of the principles and practices in the guidebook.

The Intermountain Adaptation Partnership Process

The IAP geographic area includes 12 national forests (table 1.3) and 22 NPS units across an ecologically and geographically complex area. The IAP process includes:

- A vulnerability assessment of the effects of climate change on water resources, fisheries, forest and nonforest vegetation, disturbance, wildlife, recreation, infrastructure, cultural resources, and ecosystem services. These resource sectors were selected based on their importance in the region and current management concerns and challenges.
- Development of adaptation options that will help to reduce negative effects of climate change and assist the transition of biological systems and management to a warmer climate.
- Development of an enduring science-management partnership to facilitate ongoing dialogue and activities related to climate change and to other natural resource management challenges and actions.
- Vulnerability assessments typically involve exposure, sensitivity, and adaptive capacity (Parry et al. 2007), where exposure is the degree to which the

Table 1.3—Area of U.S. Forest Service units in the Forest Service Intermountain Region (from USDA FS 2016). The national forests and grassland in the Intermountain Region are organized into 12 administrative units discussed throughout this assessment. The Forest Service Rocky Mountain Research Station manages the Desert Range Experimental Station. The "Other" category refers to areas located within National Forest System boundaries that are not Federally owned or administered by the Forest Service.

	Number of units	National Forest System	Other	Total
		Acres		
National forest	18	31,784,550	2,330,896	34,115,446
National grassland	1	47,544	27,240	74,784
Research and experimental area	1	55,510	0	55,510
Regional totals	20	31,887,604	2,358,136	34,245,740

system is exposed to changes in climate, sensitivity is an inherent quality of the system that indicates the degree to which it could be affected by climate change, and adaptive capacity is the ability of a system to respond and adjust to the exogenous influence of climate. Vulnerability assessments can be both qualitative and quantitative and focus on whole systems or individual species or resources (Glick et al. 2011). Several tools and databases are available for systematically assessing sensitivity of species (e.g., Case and Lawler 2016; Luce et al. 2014; Potter and Crane 2010).

We used scientific literature and expert knowledge to assess exposure, sensitivity, and adaptive capacity to identify key vulnerabilities for the selected resource areas. The assessment process took place over 2 years and involved monthly phone meetings for each of the resource-specific assessment teams. Each assessment team refined key questions that the assessment needed to address, chose values and key ecosystem attributes to assess, and determined which climate change effects models best informed the assessment. In some cases, assessment teams conducted spatial analyses or ran and interpreted models, selected criteria by which to evaluate model outputs, and developed maps of model output and resource sensitivities. To the greatest extent possible, teams focused on effects and projections specific to the region and used the finest scale projections that are scientifically valid.

By working collaboratively with scientists and resource managers and focusing on a specific region, the goal of IAP was to provide the scientific foundation for integrating climate change in planning, ecological restoration, and project management (Peterson et al. 2011; Raymond et al. 2013, 2014; Swanston et al. 2016). After identifying key vulnerabilities for each resource sector, scientists, land managers, and other stakeholders (box 1.1) convened five 2-day workshops in May and early June 2016 in Ogden, Utah (Uinta and Wasatch Front subregion), Boise, Idaho (Middle Rockies subregion), Salt Lake City, Utah (Plateaus subregion), Reno, Nevada (Great Basin region), and Idaho Falls, Idaho (Southern Greater Yellowstone subregion) to present and discuss the vulnerability assessment, and to elicit adaptation options from resource managers.

During these workshops, scientists and resource specialists presented information on climate change effects and current management practices for each of the resources. Facilitated dialogue was used to identify key sensitivities and adaptation options. Participants identified strategies (general approaches) and tactics (on-the-ground actions) for adapting resources and management practices to climate change, as well as opportunities for implementing these adaptation actions in projects, management plans, partnerships, and policies. Participants generally focused on adaptation options that can be implemented given our current scientific understanding of climate change effects, but they also identified research and monitoring that would benefit future efforts to assess vulnerability and guide management practices. Information from the assessment was also downscaled to identify the most significant vulnerabilities to climate change for priority resources in each subregion where appropriate. Facilitators captured information generated during the workshops with worksheets adapted from Swanston et al. (2016). Initial results from the workshops were augmented with continued dialogue with Federal agency resource specialists.

This publication contains a chapter on expected climatological changes in the IAP region, and one chapter for each of the resource sectors covered in the vulnerability assessment (water resources, fisheries, forest and nonforest vegetation, disturbance, wildlife, recreation, infrastructure, cultural resources, and ecosystem services). Each of the resource chapters includes a review of climate change effects, sensitivities, and current management practices. An additional chapter summarizes adaptation strategies and tactics that were compiled at the workshops (see Appendix 1 for author affiliations).

Resource managers and other decisionmakers can use this publication in several ways. First, the vulnerability assessment will provide information on climate change effects needed for national forest and national park planning,

projects, conservation strategies, restoration, monitoring, and environmental effects analyses. Second, climate change sensitivities and adaptation options developed at the broad scale provide the scientific foundation for finer scale assessments, adaptation planning, and resource monitoring. We expect that over time, and as needs and funding align, appropriate adaptation options will be incorporated into plans and programs of Federal management units. Third, we anticipate that resource specialists will apply this assessment to incorporate climate-smart resource management and planning in land management throughout the region.

Adaptation planning is an ongoing and iterative process. Implementation may occur at critical times in the planning process, such as when managers revise USFS land management plans and other planning documents, or after the occurrence of extreme events and ecological disturbances (e.g., wildfire). We focus on adaptation options for the USFS and NPS, but information in this publication can be used by other land management agencies as well. Just as the IAP process has been adapted from previous vulnerability assessments and adaptation planning, it can be further adapted by other national forests and organizations, thus propagating climate-smart management across larger landscapes.

Toward an All-Lands Approach to Climate Change Adaptation

The USFS and NPS climate change strategies identify the need to build partnerships and work across jurisdictional boundaries when planning for adaptation. This concept of responding to the challenge of climate change with an "all-lands" approach is frequently mentioned, but a process for doing so is rarely defined. In addition to representatives from the USFS and NPS, several other agencies and organizations participated in the resource sector workshops. This type of partnership enables a coordinated and complementary approach to adaptation that crosses jurisdictional boundaries. The IAP also provides a venue for agencies to learn from the practices of others so that the most effective adaptation strategies can be identified.

Risks and vulnerabilities resulting from climate change and gaps in scientific knowledge and policy need to be assessed on a continual basis. Adaptation is a prominent focus of the IAP, with emphasis on creating resilience in human and natural systems. Communicating climate change information and engaging employees, partners, and the public in productive discussions is also an integral part of successfully responding to climate change. The need for partnerships and collaborations on climate change issues was clearly identified in the IAP. Sharing climate change information, vulnerability assessments, and adaptation strategies across administrative boundaries will contribute to the success of climate change responses throughout the Intermountain West.

References

Butler, P.R.; Iverson, L.; Thompson, F.R., III; [et al.]. 2015. Central Appalachians forest ecosystem vulnerability assessment and synthesis: A report from the Central Appalachians Climate Change Response Framework project. Gen. Tech. Rep. NRS-146. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station.

Case, M.J.; Lawler, J.J. 2016. Relative vulnerability to climate change of trees in western North America. Climatic Change. 136: 367–379.

Furniss, M.J.; Roby, K.B.; Cenderelli, D.; [et al.]. 2013. Assessing the vulnerability of watersheds to climate change: Results of national forest watershed vulnerability pilot assessments. Gen. Tech. Rep. PNW-GTR-884. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 32 p.

Glick, P.; Stein, B.A.; Edelson, N.A., eds. 2011. Scanning the conservation horizon: A guide to climate change vulnerability assessment. Washington, DC: National Wildlife Federation. 5 p.

Halofsky, J.E.; Peterson, D.L., eds. 2017. Climate change vulnerability and adaptation in the Blue Mountains. Gen. Tech. Rep. PNW-GTR-939. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 331 p.

Halofsky, J.E.; Peterson, D.L.; Dante-Wood, S.K.; [et al.]. 2018. Climate change vulnerability and adaptation in the Northern Rocky Mountains. Gen. Tech. Rep. RMRS-GTR-374. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Halofsky, J.E.; Peterson, D.L.; Ho, J.J., eds. In press. Vulnerability of natural resources to climate change in the South Central Oregon region. PNW-GTR-XXX. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

Halofsky, J.E.; Peterson, D.L.; O'Halloran, K.A.; Hoffman, C.H. 2011. Adapting to climate change at Olympic National Forest and Olympic National Park. Gen. Tech. Rep. PNW-GTR-844. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 130 p.

Leopold, A.S.; Cain, S.A.; Cottam, C.M.; Gabrielson, I.N.; Kimball, T.L. 1963. Wildlife management in the National Parks: The Leopold Report. Washington, DC: U.S. National Park Service. 14 p.

Littell, J.S.; Peterson, D.L.; Millar, C.I.; O'Halloran, K.A. 2012. U.S. national forests adapt to climate change through science-management partnerships. Climatic Change. 110: 269–296.

Luce, C.; Lopez-Burgos, V.; Holden, Z. 2014. Sensitivity of snowpack storage to precipitation and temperature using spatial and temporal analog models. Water Resources Research. 50: 9447–9462.

Morelli, T.L.; Yeh, S.; Smith, N.M.; [et al.]. 2012. Climate project screening tool: An aid for climate change adaptation. Res. Pap. PSW-RP-263. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station.

National Park Service [NPS]. 2010. National Park Service climate change response strategy. Fort Collins, CO: National Park Service Climate Change Response Program.

National Park Service [NPS]. 2011. A call to action: Preparing for a second century of stewardship and engagement. http://www.nps.gov/calltoaction/PDF/Directors_Call_to_Action_Report.pdf [Accessed February 9, 2017].

National Park Service [NPS]. 2012a. Climate Change Action Plan. 2012–2014. National Park Service Climate Change Response Program, Fort Collins, Colorado. 36 p. http://www.nps.gov/orgs/ccrp/upload/NPS_CCActionPlan.pdf. (21 July 2015).

National Park Service [NPS]. 2012b. Green parks plan. 15 p. http://www.nps.gov/greenparksplan/downloads/NPS_2012_Green_Parks_Plan.pdf [Accessed February 9, 2017].

National Park Service [NPS]. 2012c. Revisiting Leopold: Resource stewardship in the National Parks. National Park System Advisory Board Science Committee. 18 p. http://www.nps.gov/calltoaction/PDF/LeopoldReport_2012.pdf [Accessed February 9, 2017].

Parry, M.L.; Canzianai, O.F.; Palutikof, J.P.; [et al.], eds. 2007. Climate change 2007: Impacts, adaptation and vulnerability: A contribution of working group II to the fourth assessment report of the Intergovernmental Panel on Climate Change. Cambridge, United Kingdom: Cambridge University Press. 976 p.

Peterson, D.L.; Millar, C.I.; Joyce, L.A.; [et al.]. 2011. Responding to climate change in national forests: A guidebook for developing adaptation options. Gen. Tech. Rep. PNW-GTR-855. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 109 p.

Potter, K.M.; Crane, B.S. 2010. Forest tree genetic risk assessment system: A tool for conservation decision-making in changing times. Version 1.2. Asheville, NC: U.S. Department of Agriculture, Forest Service, Eastern Forest Environmental Threat Assessment Center. http://www.forestthreats.org/current-projects/project-summaries/genetic-risk-assessment-system [Accessed February 9, 2017].

Raymond, C.L.; Peterson, D.L.; Rochefort, R.M. 2013. The North Cascadia Adaptation Partnership: A science-management collaboration for responding to climate change. Sustainability. 5: 136–159.

Raymond, C.L.; Peterson, D.L.; Rochefort, R.M., eds. 2014. Climate change vulnerability and adaptation in the North Cascades region, Washington. Gen. Tech. Rep. PNW-GTR-892. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 279 p.

Rice, J.; Tredennick, A.; Joyce, L.A. 2012. Climate change on the Shoshone National Forest, Wyoming: A synthesis of past climate, climate projections, and ecosystem implications. Gen. Tech. Rep. RMRS-GTR-264. Fort Collins, CO; U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 60 p.

Swanston, C.W.; Janowiak, M.K.; Brandt, L.A.; [et al.], eds. 2016. Forest Adaptation Resources: Climate change tools and approaches for land managers, 2nd edition. Gen. Tech. Rep. NRS-GTR-87-2. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 161 p.

Swanston, C.; Janowiak, M.; Iverson, L.; [et al.]. 2011. Ecosystem vulnerability assessment and synthesis: A report from the Climate Change Response Framework project in northern Wisconsin. Gen. Tech. Rep. NRS-82. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 142 p.

U.S. Department of Agriculture, Forest Service [USDA FS]. 2007. The greatest good: A Forest Service centennial film. https://www.fs.fed.us/greatestgood/press/mediakit/facts/pinchot.shtml# [Accessed June 26, 2017].

U.S. Department of Agriculture, Forest Service [USDA FS]. 2008. Forest Service strategic framework for responding to climate change. Version 1.0. Washington, DC: U.S. Department of Agriculture, Forest Service, Climate Change Advisor's Office. http://www.fs.fed.us/climatechange/documents/strategic-framework-climate-change-1-0.pdf [Accessed February 9, 2017].

U.S. Department of Agriculture, Forest Service [USDA FS]. 2010a. National roadmap for responding to climate change. Washington, DC: U.S. Department of Agriculture, Forest Service, Climate Change Advisor's Office. http://www.fs.fed.us/climatechange/pdf/roadmap.pdf [Accessed June 6, 2015].

U.S. Department of Agriculture, Forest Service [USDA FS]. 2010b. A performance scorecard for implementing the Forest Service climate change strategy. Washington, DC: U.S. Department of Agriculture, Forest Service, Climate Change Advisor's Office. http://www.fs.fed.us/climatechange/pdf/performance_scorecard_final.pdf [Accessed February 9, 2017].

U.S. Department of Agriculture, Forest Service [USDA FS]. 2010c. Strategic plan 2010–2015 FY. Washington, DC: U.S. Department of Agriculture, Forest Service. http://www.ocfo.usda.gov/usdasp/sp2010/sp2010.pdf [Accessed February 9, 2017].

U.S. Department of Agriculture, Forest Service [USDA FS]. 2012. 2012 planning rule. Washington, DC: U.S. Department of Agriculture, Forest Service. Federal Register. 77(68). 115 p. http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5362536.pdf [Accessed February 9, 2017].

U.S Department of Agriculture, Forest Service [USDA FS]. 2015. USDA Forest Service Strategic Plan: FY 2015–2020. Washington, DC: U.S. Department of Agriculture, Forest Service. 53 p. https://www.fs.fed.us/sites/default/files/strategic-plan%5B2%5D-6_17_15_revised.pdf?utm_medium=email&utm_source=govdelivery [Accessed June 26, 2017].

U.S. Department of Agriculture, Forest Service [USDA FS]. 2016. USDA Forest Service Intermountain Region Strategic Framework: FY 2017-2020. Washington, D.C.: U.S. Department of Agriculture, Forest Service. 18 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fseprd530247.pdf?utm_medium=email&utm_source=govdelivery [Accessed June 26, 2017].

Appendix 1—Author Affiliations

Chapter 1

Joanne J. Ho is a Research Environmental Economist with the University of Washington, College of the Environment, School of Environmental and Forest Sciences in Seattle, Washington.

Chapter 2

Hanna K. Olson is an undergraduate student at the University of Washington in Seattle, Washington.

Don W. Fallon is the Regional Resource Information and Monitoring Coordinator with the U.S. Department of Agriculture, Forest Service, Intermountain Region in Ogden, Utah.

Chapter 3

Linda A. Joyce is a Research Ecologist with the U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Human Dimensions Research Program in Fort Collins, Colorado.

Marian Talbert is a Biostatistician with the U.S. Department of the Interior, North Central Climate Science Center in Fort Collins, Colorado.

Chapter 4

Mark J. Muir is a Hydrologist with the U.S. Department of Agriculture, Forest Service, Intermountain Region in Ogden, Utah.

Charles H. Luce is a Research Hydrologist with the U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station in Boise, Idaho.

Joseph T. Gurrieri is a Hydrogeologist with the U.S. Department of Agriculture, Forest Service, National Forest System in Golden, Colorado.

Marek Matyjasik is an Advisor in Applied Environmental Geosciences with Weber State University, College of Science in Ogden, Utah.

Jeffrey L. Bruggink is the Regional Soil Scientist with the U.S. Department of Agriculture, Forest Service, Intermountain Region in Ogden, Utah.

Stacey L. Weems is a Soil Scientist with the U.S. Department of Agriculture, Forest Service, Wasatch-Cache National Forest in South Jordan, Utah.

James C. Hurja is a Soil Scientist with the U.S. Department of Agriculture, Forest Service, Humboldt-Toiyabe National Forest and Spring Mountains National Recreation Area in Sparks, Nevada.

David B. Marr is a Forest Soil Scientist and Soil and Air Program Manager with the U.S. Department of Agriculture, Forest Service, Caribou-Targhee National Forest in Idaho Falls, Idaho.

Sarah D. Leahy is a Forest Soil Scientist with the U.S. Department of Agriculture, Forest Service, Ashley National Forest in Vernal, Utah.

Chapter 5

Daniel J. Isaak is a Research Fish Biologist with the U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station in Boise, Idaho.

Michael K. Young is a Research Fisheries Biologist with the U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station in Missoula, Montana.

Cynthia Tait is the Regional Aquatic Ecologist with the U.S. Department of Agriculture, Forest Service, Intermountain Region in Ogden, Utah.

Daniel Duffield is a Fisheries Program Manager with the U.S. Department of Agriculture, Forest Service, Intermountain Region in Ogden, Utah.

Dona L. Horan is a Fisheries Biologist with the U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station in Boise, Idaho.

David E. Nagel is a Geographic Information Systems (GIS) Analyst with the U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station in Boise, Idaho.

Matthew C. Groce is a GIS Analyst with the U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station in Boise, Idaho.

Chapter 6

Patrick N. Behrens is a Silviculturist with the U.S. Department of Agriculture, Forest Service, Intermountain Region in Ogden, Utah.

Robert E. Keane is a Research Ecologist with the U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Missoula Fire Sciences Laboratory in Missoula, Montana.

David L. Peterson was a Senior Research Biologist Scientist with the U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station in Seattle, Washington.

Joanne J. Ho is a Research Environmental Economist with the University of Washington, College of the Environment, School of Environmental and Forest Sciences in Seattle, Washington.

Chapter 7

Wayne G. Padgett is a retired Ecologist, formerly with the U.S. Department of Agriculture, Forest Service, Washington Office in Washington, DC.

Matthew C. Reeves is a Research Ecologist with the U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station in Missoula, Montana.

Stanley G. Kitchen is a Research Botanist with the U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station in Provo, Utah.
David L. Tart is a Regional Ecologist with the U.S. Department of Agriculture, Forest Service, Intermountain Region in Ogden, Utah.
Jeanne C. Chambers is a Research Ecologist with the U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station in Reno, Nevada.
Cheri Howell is an Ecologist with the U.S. Department of Agriculture, Forest Service, Humboldt-Toiyabe National Forest in Sparks, Nevada.
Mary E. Manning is a Regional Vegetation Ecologist with the U.S. Department of Agriculture, Forest Service, Northern Region in Missoula, Montana.
John G. Proctor is a Regional Botanist with the U.S. Department of Agriculture, Forest Service, Intermountain Region in Ogden, Utah.

Chapter 8

Danielle M. Malesky is an Entomologist with the U.S. Department of Agriculture, Forest Service, Intermountain Region in Ogden, Utah.
Barbara J. Bentz is a Research Entomologist with the U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station in Logan, Utah.
Gary R. Brown is a Regional Fire Specialist with the U.S. Department of Agriculture, Forest Service, Intermountain Region in Ogden, Utah.
Andrea R. Brunelle is a Professor with the University of Utah, Department of Geography in Salt Lake City, Utah.
John M. Buffington is a Research Geomorphologist with the U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station in Boise, Idaho.
Linda M. Chappell is a Fire Ecologist with the U.S. Department of Agriculture, Forest Service, Dixie and Fishlake National Forests in Richfield, Utah.
R. Justin DeRose is a Research Ecologist with the U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station in Ogden, Utah.
John C. Guyon, II, is a Plant Pathologist with the U.S. Department of Agriculture, Forest Service, Intermountain Region in Ogden, Utah.
Carl L. Jorgensen is an Entomologist with the U.S. Department of Agriculture, Forest Service, Intermountain Region in Ogden, Utah.
Rachel A. Loehman is a Research Landscape Ecologist, U.S. Geological Survey, Alaska Science Center in Anchorage, Alaska.
Laura L. Lowrey is a Forest Entomologist with the U.S. Department of Agriculture, Forest Service, Intermountain Region in Ogden, Utah.
Ann M. Lynch is a Research Entomologist with the U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station in Tucson, Arizona.
Marek Matyjasik is an Advisor in Applied Environmental Geosciences with Weber State University, College of Science in Ogden, Utah.
Joel D. McMillin is a Supervisory Entomologist with the U.S. Department of Agriculture, Forest Service, Intermountain Region in Ogden, Utah.
Javier E. Mercado is an Entomologist with the U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station in Fort Collins, Colorado.
Jesse L. Morris is a Research Assistant Professor with the University of Utah, Department of Geography in Salt Lake City, Utah.
Jose F. Negrón is a Research Entomologist with the U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station in Fort Collins, Colorado.
Wayne G. Padgett is a retired Ecologist, formerly with the U.S. Department of Agriculture, Forest Service, Washington Office in Washington, DC.
Robert A. Progar is a Research Entomologist with the U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station in La Grande, Oregon.
Carol B. Randall is a Forest Entomologist with the U.S. Department of Agriculture, Forest Service, Northern Region in Boise, Idaho.

Chapter 9

Megan M. Friggens is a Research Ecologist with the U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station in Albuquerque, New Mexico.
Mary I. Williams is a Postdoctoral Researcher with the U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station in Boise, Idaho.
Karen E. Bagne is a Wildlife Biologist with the U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station in Albuquerque, New Mexico.
Tosha T. Wixom is a Regional Sustainability and Climate Intern with the U.S. Department of Agriculture, Forest Service, Intermountain Region in Ogden, Utah.
Samuel A. Cushman is a Research Landscape Ecologist with the U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station in Flagstaff, Arizona.

Chapter 10

Michael S. Hand is a Research Economist with the U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station in Missoula, Montana.
Jordan W. Smith is the Director of the Institute of Outdoor Recreation and Tourism and an Assistant Professor with Utah State University, College of Natural Resources, Department of Environment and Society in Logan, Utah.
David L. Peterson was a Senior Research Biologist Scientist with the U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station in Seattle, Washington.

Nancy A. Brunswick is a Regional Landscape Architect with the U.S. Department of Agriculture, Intermountain Region in Ogden, Utah.
Carol P. Brown is the Forest Environmental Coordinator with the U.S. Department of Agriculture, Sawtooth National Forest in Twin Falls, Idaho.

Chapter 11

Michael J. Furniss is a retired Hydrologist, formerly with the U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Redwood Sciences Lab in Arcata, California.
Natalie J. Little is the Regional Sustainability and Climate Coordinator with the U.S. Department of Agriculture, Intermountain Region in Ogden, Utah.
David L. Peterson was a Senior Research Biologist Scientist with the U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station in Seattle, Washington.

Chapter 12

Tom H. Flanigan is an Archaeologist with the U.S. Department of Agriculture, Forest Service, Wasatch-Cache National Forest in South Jordan, Utah.
Charmaine Thompson is the Heritage Program Leader with the U.S. Department of Agriculture, Forest Service, Uinta National Forest in Provo, Utah.
William G. Reed is the Regional Heritage Program Leader with the U.S. Department of Agriculture, Intermountain Region in Ogden, Utah.

Chapter 13

Travis W. Warziniack is a Research Economist with the U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station Human Dimensions Research Program in Fort Collins, Colorado.
Matthew J. Elmer is a graduate student at Colorado State University in Fort Collins, Colorado, where he is currently working on his Ph.D. in economics.
Chris J. Miller is an Economist with the U.S. Department of Agriculture, Forest Service, Washington Office Ecosystem Management Coordination in Ogden, Utah.
S. Karen Dante-Wood is a Natural Resource Specialist with the U.S. Department of Agriculture, Forest Service, Office of Sustainability and Climate in Washington, DC.
Christopher W. Woodall is a Project Leader for the U.S. Department of Agriculture, Forest Service, Northern Research Station in Durham, New Hampshire.
Michael C. Nichols is a Computer Systems Analyst for the U.S. Department of Agriculture, Forest Service, Northern Research Station in Durham, New Hampshire.
Grant M. Domke is a Team Leader and Research Forester for the U.S. Department of Agriculture, Forest Service, Northern Research Station in St. Paul, Minnesota.
Keith D. Stockmann is an Economist with the U.S. Department of Agriculture, Forest Service, Missoula Technology and Development Center in Missoula, Montana.
John G. Proctor is the Regional Botanist for the U.S. Department of Agriculture, Forest Service, Intermountain Region in Ogden, Utah.
Allison M. Borchers is an Economist with the U.S. Department of Agriculture, Forest Service, Washington Office Enterprise Program, TEAMS in Washington, DC.

Chapter 14

Jessica E. Halofsky is a Research Ecologist with the University of Washington, College of the Environment, School of Environmental and Forest Sciences in Seattle, Washington.

Chapter 15

Joanne J. Ho is a Research Environmental Economist with the University of Washington, College of the Environment, School of Environmental and Forest Sciences in Seattle, Washington.
David L. Peterson was a Senior Research Biologist Scientist with the U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station in Seattle, Washington.
Natalie J. Little is the Regional Sustainability and Climate Coordinator with the U.S. Department of Agriculture, Intermountain Region in Ogden, Utah.

Chapter 2: Biogeographic, Cultural, and Historical Setting

Hanna K. Olson and Don W. Fallon

Introduction

The Intermountain Adaptation Partnership (IAP) encompasses unique landscapes within the Intermountain Region of the U.S. Forest Service (USFS), from rugged mountains to deep canyons, from alpine snowfields to wild and scenic rivers (fig. 1.1). The area defined by the boundaries of the Intermountain Region contains both private and Federally owned lands, including 12 national forests and 22 national parks. Before Euro-Americans settled this area, Native American tribes occupied the land for thousands of years. With Euro-American settlement came timber extraction, mining, grazing, water extraction, and increased recreation to the region. Urban growth has increased significantly during recent decades, bringing new businesses and development that affect socioeconomic and natural environments.

Climate, biogeography, natural resource conditions, and management issues differ considerably from Idaho to Nevada, and from western Wyoming to the southern border of Utah. To capture how these differences influence potential climate change effects and adaptation strategies, the IAP region was divided into six subregions that are detailed in this assessment: the Middle Rockies, Southern Greater Yellowstone, Uintas and Wasatch Front, Plateaus Great Basin and Semi Desert, and Intermountain Semi Desert subregions (fig. 1.1, table 2.1). The Intermountain Semi Desert contains no national forests, but is identified as a discrete area that may be of interest to those outside the USFS. Each subregion is briefly characterized in the next section.

Table 2.1—Subregions within national forests and national forests within subregions of the Intermountain Adaptation Partnership region.

National forest	Subregions	Number of subregions in a national forest
Ashley	Uintas and Wasatch Front, Plateaus	2
Boise	Middle Rockies	1
Bridger-Teton	Southern Greater Yellowstone	1
Caribou-Targhee	Middle Rockies, Southern Greater Yellowstone, Uintas and Wasatch Front	3
Dixie	Plateaus, Great Basin and Semi Desert	2
Fishlake	Plateaus, Great Basin and Semi Desert	2
Humboldt-Toiyabe	Great Basin and Semi Desert	1
Manti-La Sal	Uintas and Wasatch Front, Plateaus	2
Payette	Middle Rockies	1
Salmon-Challis	Middle Rockies	1
Sawtooth	Middle Rockies, Uintas and Wasatch Front	2
Uinta-Wasatch-Cache	Uintas and Wasatch Front, Plateaus, Great Basin and Semi Desert	3

Subregion	National forest	Number of national forests in a subregion
Middle Rockies	Boise, Caribou-Targhee, Payette, Salmon-Challis, Sawtooth	5
Southern Greater Yellowstone	Bridger-Teton, Caribou-Targhee	2
Uintas and Wasatch Front	Ashley, Caribou-Targhee, Manti-La Sal, Uinta-Wasatch-Cache, Sawtooth	5
Plateaus	Ashley, Dixie, Fishlake, Manti-La Sal, Uinta-Wasatch-Cache	5
Great Basin and Semi Desert	Dixie, Fishlake, Humboldt-Toiyabe, Uinta-Wasatch-Cache	4
Intermountain Semi Desert	None	0

Descriptions of Subregions

Middle Rockies Subregion

The Middle Rockies subregion is part of the Rocky Mountains and extends over 16 million acres of central Idaho (fig. 2.1). The subregion is characterized by rugged mountain ranges and intermontane valleys vegetated with coniferous forests, as well as sagebrush-steppe ecosystems in the lower elevations, particularly in the southern and eastern portions (fig. 2.2). The subregion includes the Payette, Salmon-Challis, Boise, and Sawtooth National Forests, and a small portion of the Caribou-Targhee National Forest. This area is bordered by the Centennial Mountains near the Montana-Idaho border, the Lemhi Mountains along the Continental Divide of the western Montana-Idaho border, the northeastern Beaverhead Mountains, and the Salmon River Mountains in the northern section of the subregion. The area extends southward to the Intermountain Semi Desert subregion, which is dominated by geographic features associated with the Snake River Plains volcanic fields. Designated wilderness areas encompass almost 1.5 million acres of this subregion.

Geologically, the area is relatively young with emplacement of Cretaceous igneous intrusions (batholiths) roughly 120 million years BP, and younger Columbia basalts on the western boundary. During the Pleistocene (roughly 10,000 to 130,000 years BP), mountain glaciers carved and gouged the bedrock while depositing glacial till and associated river deposits in the intermontane valleys. The modern mountain ranges are characterized by high elevation ridges and deeply incised river valleys, such as those associated with the Salmon River in the Frank Church River of No Return Wilderness (Demarchi 1994).

Numerous rivers run through the Middle Rockies subregion. The Salmon River, flowing westward and spanning

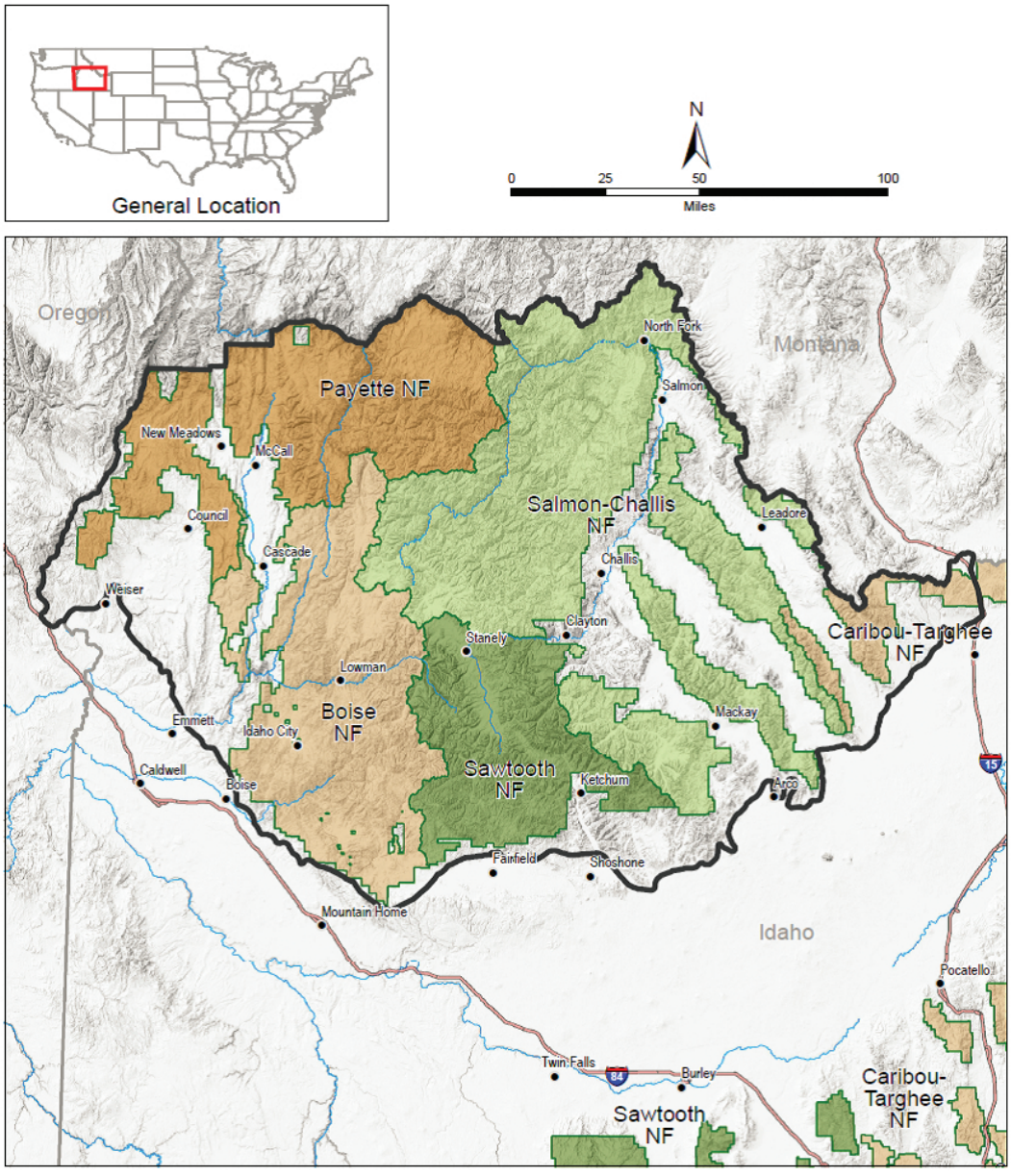


Figure 2.1—National forests within the Middle Rockies subregion of the Intermountain Adaptation Partnership region.



Figure 2.2—Rugged mountain topography in the Salmon-Challis National Forest. Subalpine forest and montane shrubs characterize the higher elevations (photo: U.S. Forest Service).

425 mi, divides northern Idaho from the remainder of the State. Major subbasins include the Little Salmon River, the Lemhi River, and the Big and Little Lost Rivers in east-central Idaho. Dry land farming occurs in valleys within the southeastern portion of central Idaho. The 1,000-mile-long Snake River enters the eastern edge of the subregion and is joined by the Salmon River near Riggins, Idaho, where river incision has created Hells Canyon, the deepest river gorge in North America. High flows of the merged rivers exceed those of the Colorado River (Idaho Department of Health and Welfare 1999). In the northwestern portion of the Middle Rockies subregion, the Boise River and Payette River are major tributaries of the Snake River. The Payette River leads to the popular recreation areas of Lake Cascade and Lake Payette. The Lemhi River is fed by the Bitterroot Range and the Big Lost River Range, and is a critical spawning habitat for Federally protected steelhead trout (*Oncorhynchus mykiss*) and Chinook salmon (*Oncorhynchus tshawytscha*) (Idaho Department of Health and Welfare 1999). The Big Lost River and the Little Lost River Range are named for their disappearance underground as they leave their valleys and flow into the Intermountain Semi Desert subregion.

Climate varies along a gradient between the western and eastern areas of central Idaho. Maritime atmospheric patterns are prevalent near the northwestern border of the subregion, delivering high precipitation that allows coniferous forests to thrive. Toward the southeastern border, the climate creates an arid environment that supports sagebrush and grassland ecosystems at lower elevations. Eastern Idaho experiences large temperature variation during the year, and precipitation differs considerably from the western portions of the subregion, with monsoonal summer moisture and snowier winters. The abrupt elevation gradients associated with the steep mountain ranges of central Idaho creates an orographic (rain-shadow) effect in which condensed moisture is deposited on the western or windward mountain slopes, and reduced moisture falls on leeward valley sides of the mountains.

Mixed conifer and subalpine forests are the dominant vegetation in the Middle Rockies. Heavily forested areas are most common in the northern portion of the subregion, progressing to more arid lowlands in the southeast. Dominant species include Douglas-fir (*Pseudotsuga menziesii*), ponderosa pine (*Pinus ponderosa*), lodgepole pine (*Pinus contorta* var. *latifolia*), grand fir (*Abies grandis*), subalpine fir (*A. lasiocarpa*), Engelmann spruce (*Picea engelmannii*), and quaking aspen (*Populus tremuloides*) (Brown and Chojnacky 1991). Of these species, Douglas-fir, ponderosa pine, lodgepole pine, grand fir, subalpine fir, and Engelmann spruce are commercially harvested (USDA FS 2016). Since 1952, ponderosa pine cover has decreased by 40 percent because of its high timber value (O'Laughlin et al. 1993). Conifer species with high ecological value but no commercial value include whitebark pine (*Pinus albicaulis* Engelm.), limber pine (*P. flexilis*), alpine larch (*Larix lyallii*), and western juniper (*Juniperus occidentalis*) (Brown and Chojnacky 1991).

Among the abundant shrub species are serviceberry (*Amelanchier alnifolia*), Rocky Mountain maple (*Acer glabrum*), tall Oregon-grape (*Berberis aquifolium*), snowbrush ceanothus (*Ceanothus velutinus*), and multiple species of sagebrush (*Artemisia* spp.) (Pyke et al. 2015; Robson and

Kingery 2006). Idaho fescue (*Festuca idahoensis*) and bluebunch wheatgrass (*Pseudoroegneria spicata*) are common grassland species. Conservation of rare plant species in the subregion is an important management responsibility (Tilley et al. 2013).

Over 300 animal species live in the Middle Rockies. Large mammals include Canada lynx (*Lynx canadensis*), grizzly bear (*Ursus arctos*), elk (*Cervus elaphus*), gray wolf (*Canis lupus*), bighorn sheep (*Ovis canadensis*), and moose (*Alces alces*). Smaller vertebrates include American beaver (*Castor canadensis*), American pika (*Ochotona princeps*), snowshoe hare (*Lepus americanus*), yellow-bellied marmot (*Marmota flaviventris*), and Rocky Mountain tailed frog (*Ascaphus montanus*) (Link et al. 2000). The Middle Rockies are home to over 400 bird species, including bald eagle (*Haliaeetus leucocephalus*), golden eagle (*Aquila chrysaetos*), osprey (*Pandion haliaetus*), multiple owl species, trumpeter swan (*Cygnus buccinator*), greater sage-grouse (*Centrocercus urophasianus*), and white-headed woodpecker (*Picoides albolarvatus*).

Fish rearing and spawning sites of both native and nonnative fish species are common in the rivers, lakes, and ponds of the Middle Rockies. Steelhead trout, cutthroat trout (*O. clarkii*), and bull trout (*Salvelinus confluentus*) are sensitive species found throughout the subregion. Nonnative lake trout (*S. namaycush*) and brook trout (*S. fontinalis*) were introduced into Idaho's lakes and streams during the 1900s and continue to thrive (Dillon and Grunder 2008). Since the 1880s, dams and reservoirs have altered water flows and decreased water quality in some rivers, leading to declines in the white sturgeon (*Acipenser transmontanus*) population (Dillon and Grunder 2008).

Before the 1900s, wildfires in the Middle Rockies subregion maintained grasslands and open forests in lower elevation and drier portions of the landscape. Historical fire frequencies were 10 to 20 years for ponderosa pine and dry Douglas-fir forests in Boise National Forest (Crane and Fischer 1986). In recent decades, grand fir and Douglas-fir have increased in ponderosa pine stands in response to decreased fire frequency. The resulting dense, multistoried stands are more susceptible to crown fires, insects, and fungal pathogens. Sagebrush communities generally have lower wildfire frequencies, but are increasingly affected by cheatgrass (*Bromus tectorum*) and other nonnative species. These annual species create fine fuels that facilitate more frequent fire, mostly to the detriment of native species.

Lodgepole pine, whitebark pine, and limber pine have undergone high mortality from mountain pine beetle (*Dendroctonus ponderosae*) outbreaks in much of central Idaho. A warmer climate has encouraged higher reproductive rates in the beetles and allowed them to survive at higher elevations (Gibson et al. 2008). Western spruce budworm (*Choristoneura occidentalis*) attacks Douglas-fir, grand fir, and pine, causing mortality and low vigor in host stands (Fellin and Dewey 1982). Balsam woolly adelgid (*Adelges piceae*) causes subalpine fir mortality and lower growth in grand fir (Lowrey 2015).

Forest diseases are especially prominent in conifer forests in the Middle Rockies. The nonnative fungus white pine blister rust (*Cronartium ribicola*) has caused extensive mortality in western white pine (*Pinus monticola*), whitebark pine, and limber pine (Cairns 2015). The formerly abundant western white pine has been mostly replaced by Douglas-fir and grand fir, driving forest succession toward more susceptible stand composition. Although efforts to eradicate blister rust have been unsuccessful, decades of natural selection have increased rust resistance in remnant western white pines, providing a foundation for regeneration of this species in the future (Schwandt et al. 2013). Parasitic dwarf mistletoe (*Arceuthobium* spp.) is also fairly common in coniferous forests. The spread rate of mistletoe has accelerated in dense, multistoried Douglas-fir stands because of the abundance of susceptible hosts and ease of mistletoe dispersal and regeneration. Long-term effects include growth reduction of the host and reduced forest productivity (Giunta et al. 2016).

No Native American communities or reservations reside within the Middle Rockies, although portions of national forests are traditional use areas. Historically, riverbanks in the Salmon-Challis National Forest provided fishing and foraging grounds for the Sheepeater Indians, the Nez Perce Tribe, and the Flathead Tribe. After the Lewis and Clark expedition during the early 19th century, assistance from Indian tribes opened the way for fur trappers, explorers, miners, ranchers, and settlers to populate what is now Lemhi County. Previously, the Whitebird band of the Nez Perce Tribe occupied portions of Payette National Forest, and utilized summer hunting and fishing grounds along the Little Salmon River. Other bands camped along the Little Salmon and Salmon Rivers as a primary source for salmon throughout the harvesting season. A wide variety of plant species provided materials for food and construction (Reddy 1993). In the mid-1800s, the Federal Government gained control of the land in Idaho, restricting the Nez Perce to the northwestern part of the State.

Resource-dependent economic activities in the Middle Rockies include livestock grazing, logging, recreation, and tourism. The gold rush of the mid-19th century brought settlers to the mountainous areas of the IAP region. The initial boom quickly died out, leaving only ghost towns to mark the once-promising gold and copper mines. Several mineral exploration projects are still operating and more have been recently proposed. Between the 1860s and 1870s, the livestock industry grew within the South Boise, Atlanta, and Deadwood mining districts (Jones 1990). After the gold rush, summer grazing activity extended beyond the outskirts of mining towns, and became established along the Boise and Payette river valleys (Jones 1990). The economic viability of livestock grazing has declined in recent decades, although large areas of national forest and other Federal lands still have grazing allotments. Exposure of bighorn sheep to cattle-borne diseases is a concern for wildlife managers, and programs are aimed at renewing the viability of bighorn sheep populations on Federal lands (USDA FS 2011).

Timber operations came to the IAP region during the "great buying rush" between 1899 and 1908. These

operations supported the expanding railroad system with railroad ties, as well as traditional building products. By 1902, the Payette Lumber and Manufacturing Company had been established, with ponderosa pine as the main commodity. The company continued to expand throughout the early 20th century, resulting in more sawmills, higher lumber production, and a growing economy. However, timber production has declined significantly in the past 30 years. Counties south of the Salmon River currently contribute up to 11 percent of Idaho's overall timber harvest and 45 percent of the State's timber exports. Valley County provides the largest annual timber harvest in the State: 65 million board feet (Brandt et al. 2012).

Travel and tourism support a significant portion of the subregional economy. A study conducted in the Sawtooth National Recreation Area found that 77 percent of visitors traveled to the forest for recreational purposes such as viewing natural features, hiking, and driving for pleasure (Headwaters Economics 2014). In 2012, 6.5 million visitors to Idaho national forests (including outside the subregion) spent over $300 million on recreation activities.

Southern Greater Yellowstone Subregion

The Southern Greater Yellowstone subregion covers 6.5 million acres of eastern Idaho, and western Wyoming (fig. 2.3). Bridger-Teton National Forest and Caribou-Targhee National Forests are the only national forests in this subregion. The Bridger-Teton spans the eastern portion of the Idaho-Wyoming border from Jackson Hole, Wyoming, eastward toward west-central Wyoming and contains 3.4 million acres of watersheds and wildlands. Approximately 2 million acres of wilderness land, as well as Grand Teton National Park, are contained within the subregion. There are several designated wilderness areas located on USFS lands in the subregion, including the Teton Wilderness and Wind

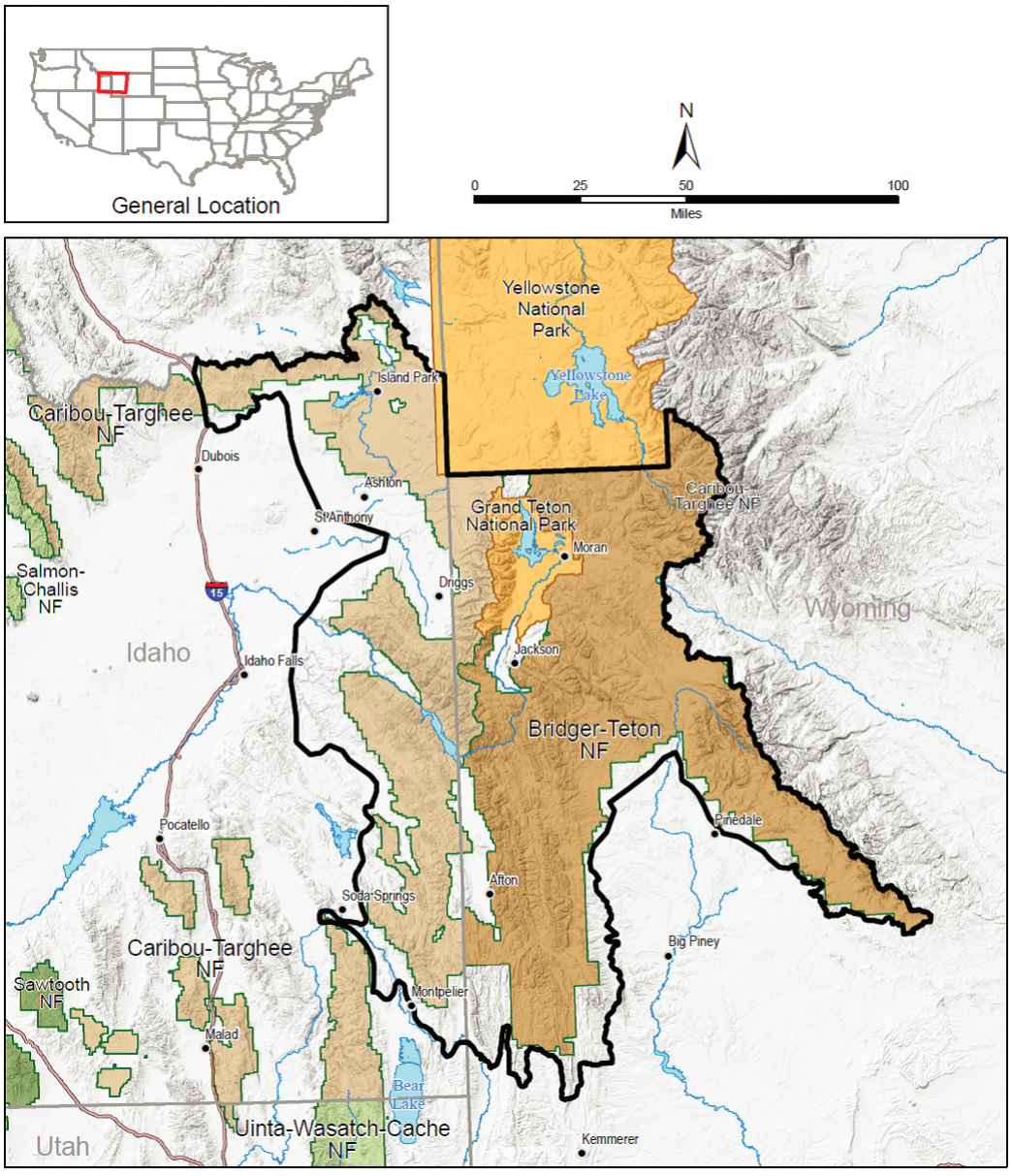


Figure 2.3—National forests and national parks within the Southern Greater Yellowstone subregion of the Intermountain Adaptation Partnership region.

River Range Wilderness in Wyoming, and two smaller wildernesses in Idaho: the Jedediah Smith Wilderness, known for karst limestone formations and caves, and the Wineger Hole Wilderness, which was set aside to protect grizzly bear habitat.

Physiography of this area reflects a wide variety of geologic mountain-building events. The subregion is part of the Greater Yellowstone Area, which extends south from the Centennial Mountains and Yellowstone National Park, through Grand Teton National Park, and southward to include the Wind River Range and Wyoming Range in Wyoming. The Centennial Mountains are a result of uplift and faulting that form steep, high-relief mountain ranges with sharp alpine ridges and cirques at high elevation. Faulting in the Centennial Mountains is fairly recent, and young faults are also present along the western border of Wyoming in the Teton Range, where active block faulting continues today. In the southeastern portions of the subregion, deformation and faulting produced the Wind River Range, with over 2,500 lakes in alpine and piedmont settings. Intermontane valleys include alluvial terraces and floodplains on the valley floor, and glacial till and moraines along the mountain flanks. The Wyoming Range, Caribou Range, and other smaller mountains of southeastern Idaho and western Wyoming were formed from low-angle thrust faulting associated with overthrust belt mountain building. Some smaller areas of volcanic activity produced lava flows and basalt in southeastern Idaho. The volcanic hotspot that underlies present-day Yellowstone National Park created volcanic tuff and tephra deposits in the Absaroka Range, and giant volcanic craters in the Island Park area. Glaciers covered most mountain ranges at times during the Pleistocene, carrying till and debris from the mouths of drainages to produce moraines that were dissected and buried by subsequent erosion and sedimentation from rivers and streams.

As elevation increases in the Southern Greater Yellowstone subregion, average temperatures decrease, and annual precipitation increases (Knight et al. 2014). Mountain ecosystems tend to be cool and moist, although mountain slopes facing south may experience dry environments similar to lower elevations. Direct solar radiation and increased rates of evaporation allow higher elevations to support more arid vegetation, such as mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*) and grassland species (Knight et al. 2014). Alpine ecosystems are found on the uppermost ridges and peaks, dominated by alpine vegetation and stunted conifers. Subalpine ecosystems are typically coniferous forests with wildflower meadows, lush wetlands, and arid shrublands along mountain flanks (fig. 2.4). Overall, the steep and rugged topography of the mountain ranges provides conditions with 60 to 80 inches of annual precipitation in higher elevations and as little as 15 inches of precipitation at lower elevations. This translates to heavy snowfall at high elevations throughout the subregion, with prevailing winds dispersing snow accumulations on exposed ridges and slopes.

The largest river system in the subregion is the Snake River, including the North, South, and Henry's forks. This system drains to the Columbia River through the Intermountain Semi Desert subregion. To the southeast, the Green River, Sandy River, and Newfork River drain southward to the Colorado River system and the Pacific Ocean, while a small area in the northeastern portion of the subregion drains to the Gulf of Mexico. Smaller rivers, such as the Greys River, Blackfoot River, Portneuf River, and Hoback River, emanate from surrounding mountains and join the larger rivers in the valleys. Numerous reservoirs and piedmont lakes are located along the river network, including Island Park Reservoir in eastern Idaho, Palisades Reservoir near the Idaho-Wyoming border, and several lakes at the base of the Wind River Range.

Vegetation in the Southern Greater Yellowstone subregion varies from high-elevation alpine grasslands and shrublands, through vast coniferous subalpine forests, to sagebrush shrublands and agricultural lands at the lowest



Figure 2.4—Complex mountain topography, alpine meadows, and dense coniferous forests, typical of the Southern Greater Yellowstone subregion of the Intermountain Adaptation Partnership region (photo: U.S. Forest Service).

elevations. At the uppermost elevations, high precipitation (mostly as snow), short growing seasons, and cold temperatures create a harsh environment where only stunted trees and hardy plants survive. Middle elevations are dominated by coniferous forests, typically Douglas-fir, lodgepole pine, Engelmann spruce, and subalpine fir. These forests were historically logged to provide railroad ties for the expanding railroad system to the south. Low elevations are dominated by several species of sagebrush. These areas are used as rangelands for small communities in the winter, as cold weather moves livestock to their lower elevation winter ranges (Blackwell and Reese 2001). In some cases, hay from surrounding agricultural areas is delivered to feed grounds in winter to reduce wildlife mortality. Stands of quaking aspen are found throughout middle elevations, providing important habitat for wildlife and subalpine herbaceous species.

Wildfire plays an important role in maintaining the biodiversity of terrestrial ecosystems. Since the big northern Idaho wildfires in 1910, fire suppression has been implemented to protect urban communities and other resources (Caribou-Targhee National Forest 2005). As a result, many conifer forests are relatively mature, in some locations blending with aspen woodlands, riparian zones, mountain meadows, and sagebrush-grassland habitats. Biodiversity in these ecosystems has decreased, promoting high-severity fires, loss of habitat for bighorn sheep and greater sage-grouse, and western balsam bark beetle (*Dryocoetes confusus*) outbreaks in subalpine fir. Prescribed fire has been implemented in some locations to reduce fuels, improve wildlife habitat, and increase habitat heterogeneity (Caribou-Targhee National Forest 2005).

White pine blister rust and mountain pine beetle have reduced whitebark pine populations near the northern border of the subregion. Whitebark pine provides forage for grizzly bear and black bear (*Ursus americanus*) during the winter. Mountain pine beetle has affected lodgepole pine populations, causing increased mortality throughout the Intermountain West. Douglas-fir and Engelmann spruce stands have suffered stunted growth and mortality from the spread of western spruce budworm.

About 100 species of mammals live within the subregion. Common rangeland and forest species include black bear, Rocky Mountain elk (*Cervus canadensis*), moose, mule deer (*Odocoileus hemionus*), pronghorn (*Antilocapra americana*), and bighorn sheep. Areas near Yellowstone National Park provide protection for grizzly bear and gray wolf. Small vertebrates include American pika, American beaver, wolverine (*Gulo gulo*), and numerous amphibians. Grassland ecosystems support large areas for livestock grazing. The National Elk Refuge was established in 1912 to provide feed, sanctuary, and habitat for one of the largest elk herds on Earth. In early fall, elk migrate from the surrounding uplands down to Jackson Hole, where hay from the surrounding communities is used as forage for the surviving elk.

Over 350 avian species reside within the subregion. Riparian and forest habitats provide nesting sites for bald eagle, golden eagle, peregrine falcon (*Falco peregrinus*), and other raptors. Greater sage-grouse occupies sagebrush-grassland ecosystems, and is a high-priority species for conservation. Trumpeter swan, sandhill crane (*Grus canadensis*), Canada geese (*Branta canadensis*), and many species of ducks are common in the wetlands of Bridger-Teton National Forest.

World-class fishing is supported along all rivers in the Southern Greater Yellowstone subregion. Cutthroat trout and mountain whitefish (*Prosopium williamsoni*) are indigenous species, whereas brook trout, brown trout (*Salmo trutta*), lake trout, and golden trout (*Oncorhynchus mykiss aguabonita*) have been introduced in many lakes throughout the river system.

Beginning around 11,000 years BP, bands of Shoshone, Blackfoot, and Bannock Indians occupied western portions of the subregion (State of Wyoming 2013). Farther east, nomadic hunters and gatherers migrated throughout the valleys and mountains of the Wind River and Absaroka Ranges. At the beginning of the 19th century, trappers and explorers began to frequent western Wyoming and started trading with the Indian tribes of the area. Over time, Native American tribes were confined to smaller portions of the subregion.

After the Lewis and Clark expedition, John Colter traversed the Continental Divide and descended into Jackson Hole during the winter of 1807–1808 (State of Wyoming 2013). Other settlers soon followed, establishing trading posts along the Wind River, Gros Ventre, Teton, and Wyoming Ranges. Wyoming became the focus for Governmental expeditions, fur trapping, and hunting for the next 30 years. Independent trappers began to appear in the Jackson Hole Valley, changing the socioeconomic trajectory of the area. With increasing settlement of western Wyoming, forest reserves were established to protect water, wood, wildlife, recreation, and forage. These reserves would eventually become the Caribou-Targhee and Bridger-Teton National Forests. The Civilian Conservation Corps contributed to an increase in national forest infrastructure and expansion of the timber industry. Timber has remained an important component of local economies, although the volume of harvests has decreased greatly in the past 30 years. Thinning is now used to reduce stand densities and improve wildlife habitat.

The gold rush of 1870, when gold was found near Caribou Mountain, sparked a 20-year run of gold mining in eastern Idaho and western Wyoming, after which mining of phosphate deposits for fertilizers was conducted in the Caribou-Targhee National Forest for 50 years. Recently, exploration for oil and gas reserves has been conducted along the southern portion of the subregion, mostly in the surrounding basins.

Tourism is a vital industry in the area, with Jackson Hole serving as a major entrance for millions of visitors each year to Grand Teton and Yellowstone National Parks. The Grand Targhee and Jackson Hole resorts attract skiers from around

the world. Other activities, such as camping, hunting, and fishing, are popular summer activities for visitors to the subregion.

Uintas and Wasatch Front Subregion

The northern portion of the Uintas and Wasatch Front subregion includes the Albion, Black Pine, Raft River, and Sublett Mountains along the Idaho-Utah border. Sagebrush and grasslands dominate the low to middle elevations that surround islands of higher elevation coniferous forests. Farther south, the rugged Wasatch Front is the western border of the subregion, with high ridges and an abrupt transition from high desert to alpine ecosystems (fig. 2.5). National forests in the subregion are the Uinta-Wasatch-Cache National Forest, Ashley National Forest, and southern portions of the Sawtooth and Caribou-Targhee National Forests (fig. 2.6). Over 600,000 acres are dedicated to wilderness, including the Wellsville Mountain Wilderness. The Curlew National Grassland encompasses over 47,000 acres.

The complex topography of the subregion has resulted from millions of years of mountain building, sedimentation, and erosion. Warm, shallow water once covered the area of the Wasatch and Uinta Mountains, depositing thick layers of fossil-bearing gray sediment. Once compression of the North American Plate began, thrust faulting and buckling on the eastern half of the Wasatch Front created highly deformed mountain ranges that shed sediment into the lower valleys to create the sandstone and limestone valleys of the present day (Atwood 2012). Local and regional uplift brought the landscape to its current position, allowing rivers and glaciers to carve steep slopes and deposit high volumes of sediment within the surrounding basins and ranges. The Wasatch fault displays faulting along the base of the steep Wasatch Front where fault scarps have displaced beach deposits of ancient Lake Bonneville. This lake was the larger precursor to the present-day Great Salt Lake, forming from meltwaters during warming periods of the Pleistocene.

Near the southeastern corner of the subregion, the Green River flows into Flaming Gorge Reservoir, continuing southeastward and emerging near Vernal, Utah. Numerous tributaries that run through the Ashley and Uinta-Wasatch-Cache National Forests provide water for critical watersheds, including Strawberry Reservoir, where diversions and tunnels carry the water from the Uinta Basin and the Colorado River drainage to the Wasatch Front for agricultural and municipal use.

The northern portion of the subregion has hot summers with little precipitation and winters that receive high amounts of precipitation at higher elevations, but less at lower elevations. The southern portion experiences both dry summer and wet winter patterns, as well as wet summer and dry winter patterns (Shaw and Long 2007). The western portion of the subregion has average summer temperatures above 90 °F and average winter temperatures below 20 °F. The eastern portion typically experiences wet summers and dry winters, with annual precipitation as low as 7 to 10 inches at lower elevations.

Coniferous forests are prominent in the southern portions of the subregion, especially in the Wasatch and Uinta Mountains. Scattered patches of coniferous forests occur along the middle to high slopes of the northern mountain ranges, which are surrounded by sage-steppe ecosystems in valleys. Dominant conifer species, including subalpine fir, Engelmann spruce, lodegepole pine, and Douglas-fir, account for a large proportion of the forest types (Heyerdahl et



Figure 2.5—U-shaped valleys, rugged mountain ranges, scattered alpine lakes, and dense coniferous forests, which are common in the Uintas and Wasatch Front subregion of the Intermountain Adaptation Partnership region (photo: U.S. Forest Service).

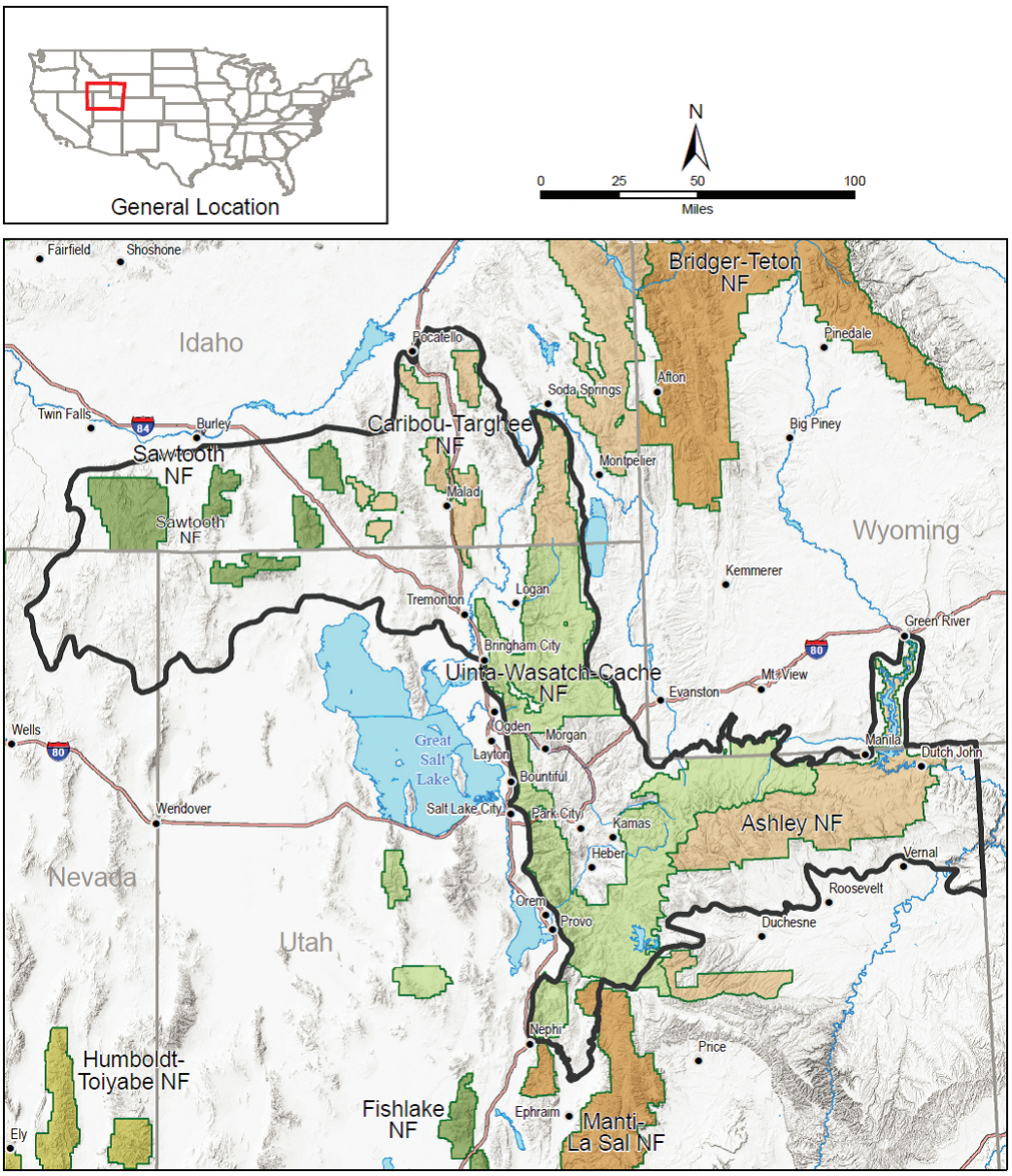


Figure 2.6—National forests within the Uintas and Wasatch Front subregion of the Intermountain Adaptation Partnership region.

al. 2011). Utah juniper (*Juniperus osteosperma*) and pinyon pine (*Pinus edulis*) occupy arid, shallow, and rocky soils at lower elevation. Lower elevations also provide habitat for Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) and mountain big sagebrush, as well as bluebunch wheatgrass, bottlebrush squirreltail (*Elymus elymoides*), Sandberg bluegrass (*Poa sandbergii*), and Thurber's needlegrass (*Achnatherum thurberuanum*). These shrub-steppe systems provide habitat for small fauna, ungulates, and local populations of greater sage-grouse.

Wildfire regimes influence the overall health and biodiversity of all ecosystems, with most fires occurring during July through October (Morris 2006). Fire exclusion in the Wasatch Range and Uinta Mountains has mixed conifer forests that are relatively mature and dense with high fuel loadings, making them susceptible to crown fire. Low-vigor stands in Uinta-Wasatch-Cache National Forest provide suitable conditions for bark beetle outbreaks, such as the outbreak in 1980 when beetle populations were extremely high and conditions for beetle survival were optimum (Shaw and Long 2007). These forests have been subject to outbreaks by mountain pine beetle and other beetle species during the past 15 years.

Moose are common in wetlands of most intermontane valleys. Mountainous topography provides habitat for elk, black bear, bighorn sheep, mule deer, and small populations of mountain goats (*Oreamnos americanus*). Sensitive species within the Albion Range include mountain lion (*Felis concolor*) and Canada lynx. Small vertebrates include lizards, rattlesnakes, sagebrush vole (*Lemmiscus curtatus*), and muskrat (*Ondatra zibethicus*).

Rivers, streams, creeks, and reservoirs throughout the Uintas and Wasatch Front subregion support both native and nonnative fish. Colorado River cutthroat trout (*O. clarkii pleuriticus*) is native to the Green River watershed, and the Bonneville cutthroat trout (*O. c. utah*) is native to

other drainages. These species attract anglers from around the world for recreational fishing. Nonnative fish include numerous trout and bass species, kokanee salmon (*O. nerka*), walleye (*Sander vitreus*), and yellow perch (*Perca flavescens*).

Numerous avian species inhabit various ecosystems of the subregion. Olive-sided flycatcher (*Contopus cooperi*), chipping sparrow (*Spizella passerina*), and chickadees (*Poecile* spp.) are common in subalpine forests. Raptors, including red-tailed hawk (*Buteo jamaicensis*), golden eagle, and bald eagle, populate low rangelands that border the mountains. Along the Wasatch Front, hermit thrush (*Catharus guttatus*), fox sparrow (*Passerella iliaca*), mountain bluebird (*Sialia currucoides*), and sandhill crane are common near small reservoirs and streams. In the Uinta Mountains, alpine habitat is occupied by pine grosbeak (*Pinicola enucleator*), three-toed woodpecker (*Picoides dorsalis*), and gray jay (*Perisoreus canadensis*). Many species of shorebirds and waterfowl populate reservoirs across the subregion.

Native American tribes once lived throughout the area. In 1500, the Northern Utes occupied central Utah, western Colorado, southern Wyoming, and northern New Mexico. Utah Valley with its abundant supply of fish and other natural resources supported most of the population (Simmons 2000). With the acquisition of horses in 1600, the Utes were able to extend their range and travel to the Great Plains to hunt American bison (*Bison bison*). The Uintah and Ouray Indian Reservation is now home to the Ute Tribe and is situated south of the Ashley National Forest.

Agriculture accounts for a large portion of the economy. The animal industry is the single largest sector of farm income in Utah. River valleys and cultivated fields provide fertile soils for the production of hay, corn, barley, and a variety of fruit. Orchards of apples, apricots, and peaches are scattered throughout the subregion and concentrated along the Wasatch Front. In the southeastern portion of the subregion, melons and other fruits provide economic benefits to smaller communities. Specialty products such as soap and honey also contribute agricultural income in some communities.

Watersheds throughout the subregion are an important source of water for municipal, industrial, and recreational use, and for hydroelectric power. Flaming Gorge National Recreation Area, Strawberry Reservoir, and other smaller reservoirs provide these water resources, as well as popular destinations for recreation. Watershed and groundwater management are a component of resource management for urban populations along the Wasatch Front; water supplies are decreasing with increasing urban populations, making water conservation a significant concern. The Bear River, Jordan River, Weber River, and Ogden River are sources for this water, relying on dams and diversions to create power and allocate water resources. Regulations restricting construction, wastewater, and livestock operations help ensure adequate water supplies and water quality, although overdraft has occurred in aquifers in northern Utah since 1985 (Burden 2015).

Mining projects have been proposed throughout the subregion, including sites that are focused on hard rock such as gold or copper mining, and soft rock such as phosphate mining. Drilling for oil and gas has expanded, including proposed drilling of up to 400 new wells over a 20-year period in the Vernal Basin area. About 50 active oil and gas wells are currently in national forest lands within the subregion. Coal mining is a major industry, with coal fields scattered throughout the subregion. Tourism continues to provide economic stimulus to many mountainous areas along the Wasatch Front. Several world-class ski resorts such as Snowbird, Alta, and Park City serve local and international enthusiasts. Snowbasin and Park City hosted the 2002 Olympic Games.

Plateaus Subregion

The Plateaus subregion covers the southern half of Utah and a small portion of western Colorado, including Ashley, Manti-La Sal, Fishlake, and Uinta-Wasatch-Cache, Dixie National Forests, as well as Zion, Bryce Canyon, Capitol Reef, Arches, and Canyonlands National Parks (fig. 2.7). Rugged mountain landscapes contrast with the red-rock desert of Utah badlands, where millions of years of erosion have carved steep canyons and intricate rock formations through the sandstone and carbonate composition of the Colorado Plateau. About 150,000 acres are devoted to wilderness. Mountain ranges, plateaus, cliffs, and canyons characterize the rugged landscape (fig. 2.8), providing recreational and economic benefits for surrounding communities.

In the northwestern portion of the Plateaus subregion, Manti-La Sal National Forest contains the La Sal, Manti, North Horn, and Abajo Mountains. Rugged topography, alpine meadows, and forests cover these north-south trending ranges. The Tushar Mountain Range in Fishlake National Forest provides the basin for Fish Lake, which is a popular location for summer activities. Dixie National Forest contains white cliffs, red canyons, and several mountain peaks covered by dense forests and desert shrublands. Sedimentary rocks are pronounced at Cedar Breaks National Monument and at adjacent national parks, Glenwood Canyon National Recreation Area, and Grand Staircase-Escalante National Monument.

The major river network in the Plateaus subregion belongs to the Green and Colorado River system. Major tributaries flow through the surrounding lowlands and empty into Lake Powell. The Dirty Devil, Virgin, and San Juan Rivers are smaller tributaries that are locally important. The north-to-south running Green River joins the Colorado River in Canyonlands National Park. The Colorado River then flows to the southwest, carving steep cliffs through sandstone bedrock and continuing into Arizona. Glen Canyon National Recreation Area contains the second largest reservoir in the United States, providing hydroelectric power, water resources, and recreation. The westerly

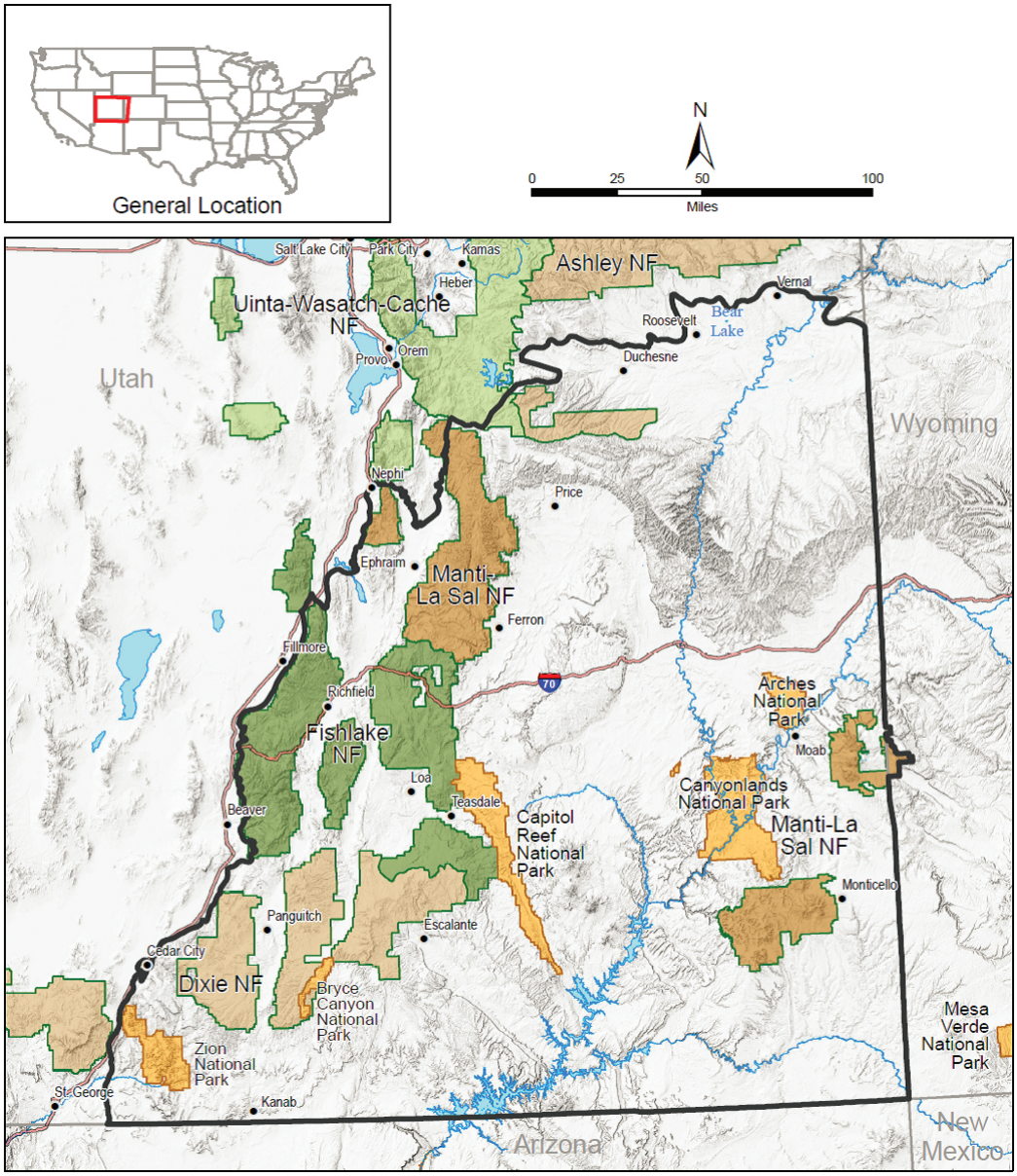


Figure 2.7—National forests and national parks within the Plateaus subregion of the Intermountain Adaptation Partnership region.



Figure 2.8—A red-rock landscape in the Plateaus subregion of the Intermountain Adaptation Partnership region. Steep cliffs line canyons that have been powerfully eroded throughout geologic history (photo: U.S. Forest Service).

flowing San Juan River begins in southwestern Colorado, flowing southeast until it meets the Colorado River near the mouth of Glen Canyon.

The Colorado Plateau is the dominant physiographic feature of this area, characterized by isolated buttes, mesas, plateaus, steep escarpments and cliffs, and gently folded sedimentary rocks. Several small mountain ranges such as the La Sal Mountains and Henry Mountains provide higher elevations from the surrounding lowlands, supplying water to the tributaries. Millions of years of evaporate deposition, folding, wind and water erosion, and salt deformation formed the diverse landscape. Between 70 and 265 million years BP, a shallow marine depositional environment covered the Colorado Plateau, forming limestone, sandstone, shales, marine fossils, and salt deposits (Harris et al. 1997).

Erosion processes, such as by wind and rivers, shaped the bedrock and topography of this area. Stream incisions dissect this relatively flat area through down-cutting and headward erosion processes. Melting of Pleistocene glaciers provided large volumes of water to be carried throughout the Colorado River network, and incised drainages. As uplift continued, river incision further carved deep canyons throughout the flat landscape, creating landmarks that can be observed today (Harris et al. 1997). Strong winds have eroded the smooth arches and other geologic formations in Arches National Park and Natural Bridges National Monument (Harris et al. 1997).

Dominant ecosystems of the Plateaus subregion include coniferous and deciduous forests at higher elevations, and woodlands, mountain shrub communities, sagebrush shrublands, grasslands, and desert ecosystems at lower elevations. Other more isolated ecosystems are also important including riparian wetlands, and unique niches within the associated canyons, cliffs, and talus slopes. High mountain ranges support subalpine ecosystems of coniferous and deciduous forests throughout the western portion of the subregion and in the La Sal and Henry Mountains. Coniferous forests of ponderosa pine, Douglas-fir, white fir (*Abies concolor*), Engelmann spruce, and limber pine occupy mid- to upper slopes. Mountain shrub communities intermix with forests, and are dominated by understory layers of mountain big sagebrush, Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*), common snowberry (*Symphoricarpos albus*), and serviceberry (*Amelanchier alnifolia*). Deciduous woodlands of pinyon pine, Utah juniper, quaking aspen, and Gambel oak (*Quercus gambelii*) exist throughout the subregion and mix with the sagebrush shrublands and agricultural grassland ecosystems at lower elevations. These stands contain understory layers of black sagebrush (*Artemisia nova*), mountain big sagebrush, and curl-leaf mountain mahogany (*Cercocarpus ledifolius*) (Mohlenbrock 2006). Utah juniper has encroached into sagebrush areas, depleting habitat of greater sage-grouse. Riparian areas create hotspots of biodiversity for flora and fauna, and help moderate flooding, sediment deposition, and water temperature (Knight et al. 2014). Riparian ecosystems are vulnerable to damage from livestock grazing and construction activities for roads and other infrastructure (Utah Division of Wildlife Resources 1991).

Some 480 plant species are found in the subregion. Common shrubs include mountain big sagebrush, black sagebrush, greenleaf manzanita (*Arctostaphylos patula*), fourwing saltbush (*Atriplex canescens*), rubber rabbitbrush (*Ericameria nauseosa*), antelope bitterbrush (*Purshia tridentata*), Mormon tea (*Ephedra viridis*), and greasewood (*Sarcobatus vermiculatus*) (Mohlenbrock 2006). Grasses include Salina wildrye (*Leymus salinus*), galleta (*Hilaria jamesii*), and purple three-awn (*Aristida purpurea*) (West 1983). Many of these species are tolerant of high salt concentration in low-elevation soils, especially ancient lake beds.

A particularly fragile feature of desert ecosystems is known as a "cryptobiotic crust." Cyanobacteria and other organisms create these crusts on the soil surface where they live and retain carbon, fix nitrogen, and stabilize soil. These crusts thicken through time, providing nutrients to the dry soil below. The crust is vulnerable to disturbances (e.g., human footprints, vehicles) that disrupt its structure and reduce plant growth.

Large mammal species include mule deer, bighorn sheep, elk, mountain lion, black bear, and coyote (*Canis latrans*). Smaller vertebrates include red squirrel (*Tamiasciurus hudsonicus*), hoary bat (*Lasiurus cinereus*), pinyon mouse (*Peromyscus truei*), and desert cottontail (*Sylvilagus audubonii*). Bird species include red-tailed hawk, northern goshawk (*Accipiter gentilis*), Cooper's hawk (*A. cooperi*), wild turkey (*Meleagris gallopavo*), boreal owl (*Aegolius funereus*), and numerous warblers. Amphibians and reptiles include tiger salamander (*Ambystoma tigrinum*) and milk snake (*Lempropeltis triangulum*) (West 1983).

The Plateaus subregion contains diverse fisheries in streams, lakes, and reservoirs. Common species of fish include rainbow trout (*Oncorhynchus mykiss*), cutthroat trout, lake trout, brook trout, speckled dace (*Rhinichthys osculus*), desert sucker (*Catostomus clarki*), and yellow perch. The Virgin River system provides habitat for the rare Virgin River chub (*Gila seminude*), Virgin River spinedace (*Lepidomeda mollispinis*), and woundfin (*Plagopterus argentissimus*). Nonnative fish, stream diversions, and poor water quality have reduced the abundance of these species.

Native Americans arrived in the Plateaus subregion around 12,500 years BP. Hunting of mammoths, giant sloths, and American bison sustained these peoples until the megafauna became extinct around 8,000 years BP, when the Puebloan people emerged and populated the entire region (Hatt 2014). Native Americans had diverse lifeways focused on hunting, fishing, food gathering, and eventually agriculture. Near the end of the 19th century, pioneers began to settle southeastern Utah for mineral extraction, farming, and ranching (Hatt 2014). Native American populations were greatly reduced following Euro-American contact. Currently, the Ute Indians and Navajo Nation have large reservations within the subregion.

Geologic evolution of the subregion created extensive deposits of coal, oil, gas, tar sands, oil shale, uranium, and

potash (Hatt 2014). Coal beds have allowed the subregion to become a major contributor to U.S. energy production. On a local scale, coal, oil, and gas industries support economic development. Renewable energy sources include wind, hydroelectric, geothermal, and solar (Hatt 2014). In 2010, Ashley National Forest sold more than 1.3 million cubic feet of timber (Ashley National Forest 2010), representing a significant source of income for local communities.

National parks, national forests, and other lands contribute significantly to the tourism economy of the Plateaus subregion, attracting millions of visitors, of which two-thirds are from out of state. Diverse landscapes from alpine forests to the red-rock deserts provide a wide range of attractions for recreation and tourism. Summer offers hunting, fishing, hiking, mountain biking, and off-road travel, and winter attracts skiers, snowmobilers, and other winter sport enthusiasts.

Great Basin and Semi Desert Subregion

The Great Basin and Semi Desert subregion is the largest subregion, occupying over 100 million acres, and covering most of Nevada, the western half of Utah, small sections of southern Idaho, and portions of east-central California (fig. 2.9). Sections of Humboldt-Toiyabe National Forest are distributed throughout Nevada and parts of east-central California. The western half of the Dixie National Forest and the northern tip of the Fishlake National Forest are included near the eastern border of the subregion.

Between 300 and 800 million years BP, deep oceans covered western Utah, depositing limestone and dolomite sediments. These deposits were buried by thick layers of shale, forming large coal beds that underlie much of the subregion. Between 18 and 40 million years BP, volcanism formed large calderas, cinder cones, and lava flows throughout Nevada and western Utah. Large areas of mineral deposits (silver, gold, molybdenum, zinc, beryllium, iron, and copper) were formed in hydrothermal veins and host rock. Around 65 million years BP, Utah began to

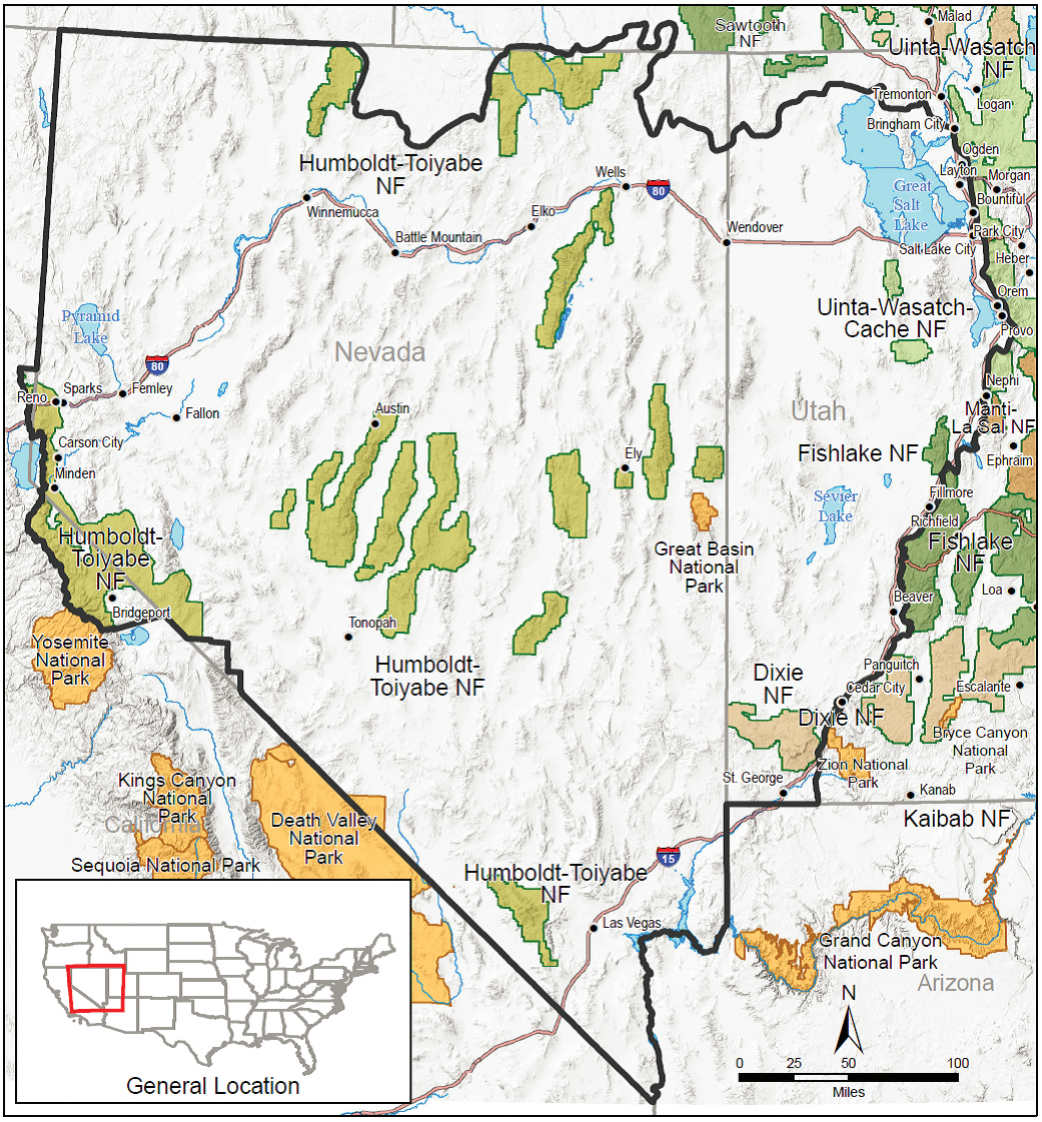


Figure 2.9—National forests and national parks within the Great Basin and Semi Desert subregion of the Intermountain Adaptation Partnership region.

sink and extensional faulting produced the basin-and-range topography of the Great Basin (Black 2011). More than 150 different mountain ranges are scattered throughout Nevada and Utah, including the Schell Creek Range, Toiyabe Range, Ruby Mountains, Shoshone Mountains, and Snake Range. These mountains are typically isolated and surrounded by intervening lowlands.

Around 10,000 years BP, melting glaciers of the Rocky Mountains and Sierra Nevada filled valleys of the Great Basin (Black 2011). Lake Bonneville and Lake Lahontan formed, covering most of western Utah. As climate warmed, both lakes evaporated and left remnant lakes (e.g., Sevier Lake and the Great Salt Lake from Lake Bonneville), as well as numerous dry playas.

Climate in the Great Basin and Semi Desert subregion is mostly desert and semiarid. The high Sierra Nevada traps moisture coming from the Pacific Ocean, creating a significant rain-shadow effect that makes Nevada the driest State in the United States. Average temperatures in northwestern Nevada are about 50 °F, with hot, dry summers and moderately cool winters (WRCC 2016). Farther south, average temperatures are about 65 °F; summers are long and hot, and winters are short and mild. Little precipitation falls in the southern part of the subregion, whereas the northwestern portion receives more precipitation because of proximity to the Sierra Nevada.

The Humboldt, Colorado, Sevier, and Truckee Rivers are the major river networks in the subregion. Smaller river networks include the Carson and Walker Rivers. Supplied with water from the Rocky Mountains to the north and east, the Colorado River flows in the southeastern portion of the subregion. The Humboldt River runs 290 miles through northern Nevada (the longest U.S. river that exists entirely within one State), and drains into the Humboldt Sink, a playa in western Nevada. The headwaters of the Humboldt River drain the East Humboldt and Jarbidge Ranges, and the river continues westward, gathering water from scattered tributaries along the way during the spring and losing surface water to groundwater during much of the rest of the year. The Sevier River is Utah's longest river, flowing 279 miles through western Utah and emptying into the nearly dry Sevier Lake (Seligman et al. 2008).

With elevations ranging from 4,100 to 13,065 feet, the Great Basin and Semi Desert subregion exhibits a broad spectrum of ecosystems. High elevations contain a mixture of alpine, subalpine, woodland, riparian, and shrub habitats. Subalpine ecosystems include quaking aspen woodlands, coniferous forests, pinyon-juniper woodlands, and mountain shrublands. Aspen woodlands dominate more than 250,000 acres and are commonly found between 5,200 and 10,500 feet, providing wildlife habitat, areas for livestock grazing, and recreation opportunities. Lower elevations are dominated by sagebrush-steppe ecosystems (fig. 2.10), which encompass most of the subregion, and include numerous small spring-fed riparian and wetland areas.

Mountain shrub communities of mountain big sagebrush, serviceberry, Gambel oak, and many other species occupy mountain slopes at 5,000 to 9,000 feet elevation. These are the most widespread communities in the subregion, providing nesting, food resources, and shelter for many vertebrate and invertebrate species. Riparian zones are common along the lower portions of the mountain ranges in Humboldt-Toiyabe National Forest. Springs, rivers, creeks, and lakes offer moist conditions for grasses and forbs and habitat for mammals, migratory songbirds, and wildflowers.

Coniferous forests, including bristlecone pine (*Pinus longaeva*), limber pine, whitebark pine, and ponderosa pine, occur at mid- to high elevations. In the eastern Sierra Nevada, coniferous forests also include Jeffrey pine (*Pinus jeffreyi*), lodgepole pine, western white pine, Douglas-fir, white fir, and Shasta red fir (*Abies magnifica*). Pinyon-juniper woodlands exist at middle elevations, dominated



Figure 2.10—Sagebrush and desert shrub habitats characteristic of the Great Basin and Semi Desert subregion of the Intermountain Adaptation Partnership region (photo: U.S. Geological Survey).

by pinyon pine and Utah juniper. These woodlands provide habitat for mule deer, desert cottontail, pinyon jay (*Gymnorhinus cyanocephalus*), and Clark's nutcracker (*Nucifraga columbiana*). Expansion of pinyon-juniper woodlands has displaced some shrub-steppe habitat, and cutting and mechanical treatments are used to control the extent of woodlands in some locations.

Sagebrush is common throughout the Great Basin and Semi Desert subregion, providing habitat for sage sparrow (*Artemisiospiza belli*), sage thrasher (*Oreoscoptes montanus*), loggerhead shrike (*Lanius ludovicianus*), burrowing owl (*Athene cunicularia*), and greater sage-grouse. Both large and small vertebrates, including pronghorn and mule deer, inhabit sagebrush ecosystems.

Other large mammals in this subregion include desert (*Ovis canadensis nelsoni*), California (*O.c. californiana*), Rocky Mountain (*O.c. canadensis*), and Sierra Nevada (*O.c. sierra*) bighorn sheep; elk; mountain goat; black bear; and mountain lion. Small mammals include several species of rabbits, Sierra Nevada red fox (*Vulpes vulpes necator*), coyote, bobcat (*Lynx rufus*), Palmer's chipmunk (*Tamias palmeri*), American marten (*Martes americana*), and numerous bat species. Wild horses and burros are common across the entire State, competing with wildlife and livestock for scarce forage. Birds in the subregion include greater sage-grouse, goshawk, California spotted owl (*Strix occidentalis occidentalis*), great gray owl (*Strix nebulosi*), flammulated owl (*Psiloscops flammeolus*), burrowing owl (*Athene cunicularia*), red-tailed hawk, golden eagle, prairie falcon (*Falco mexicanus*), and Cooper's hawk.

Many fish species occupy streams and major rivers in the subregion. Rivers and streams of eastern Nevada contain Bonneville cutthroat trout, interior redband trout (*Oncohynchus mykiss gairdneri*), and bull trout. The Sevier River contains primarily brown trout, rainbow trout, and cutthroat trout. Lahontan cutthroat trout (*O. clarkii henshawi*), a Federally listed threatened species, occurs in isolated mountain streams throughout much of Nevada. Silver King Creek in California contains the only population of Paiute cutthroat trout (*O. clarkii seleniris*).

Desert ecosystems support Pacific tree frog (*Hyla regilla*), Columbia spotted frog (*Rana luteiventris*), and red-spotted toad (*Bufo punctatus*). The mountain yellow-legged frog (*R. muscosa*) and Yosemite toad (*B. canorus*) occupy the higher elevations in the far western part of the subregion. Reptiles include side-blotched lizard (*Uta stansburiana*), zebra-tailed lizard (*Callisaurus draconoides*), Great Basin collared lizard (*Crotaphytus bicinctores*), coachwhip (*Masticophis flagellum*), speckled rattlesnake (*Crotalus mitchellii*), and Gilbert's skink (*Plestiodon gilberti*) (Knight et al. 2014). Many species of hummingbirds and small songbirds populate both desert and mountain ecosystems (Knight et al. 2014).

Wildfire regimes in the subregion have been altered where nonnative species have proliferated following fire. At the end of the 19th century, cheatgrass was introduced to the shrublands. This species is highly flammable, increases the spread rate of wildfires, and regenerates quickly after fire, outcompeting sagebrush and other perennial species (Blank et al. 2008). Public policy has encouraged fire suppression throughout the subregion, which has resulted in single-age sagebrush habitats that lack herbaceous and perennial grass species (Blank et al. 2008). Some shrub cover is so dense that biodiversity is low within these habitats.

Before Euro-American settlement, Native American tribes claimed lands that are now in Humboldt-Toiyabe and Dixie National Forests. In the Nevada portion of the area, the ancestral land of the Southern Paiute, Northern Paiute, Western Shoshone, and Washoe Indians has been inhabited for over 4,500 years. The Fremont, Anasazi, and Paiute Indians inhabited what is now Dixie National Forest. Several Indian reservations occupy the area, including those for the Summit Lake Paiute Tribe, Pyramid Lake Paiute Tribe, Walker River Paiute Tribe, Moapa Band of Paiute Indians, Goshute Confederated Tribes, Te-Moak Tribe of Western Shoshone Indians-South Fork band, Skull Valley band of Goshute Indians, Paiute Indian Tribes-Shivwits band, and a portion of the Shoshone-Paiute Tribe.

The 19th-century California gold rush brought Euro-American settlers to the area. Emigrant roads and trails connected mining towns, logging sites, and stagecoach stops throughout the region. Industries and economic pressure grew with continued settlement of the American West throughout the 19th century. Nevada leads the country in production of gold, barite, diatomite, and mercury, and is the only State that produces magnesite, lithium, and specialty clays (Price 2004).

Oil and gas reserves lie below the eastern part of the subregion. The National Forest System evaluates lands for drilling potential in accordance with the Federal Onshore Oil and Gas Leasing Reform Act of 1987 (Dixie National Forest 2011). Drilling is allowed within some areas of Dixie National Forest and Humboldt-Toiyabe National Forest, excluding designated wilderness, although no drilling has been conducted on Dixie National Forest since 1987. Most expressions of interest in drilling have been processed by the Bureau of Land Management for its lands adjacent to national forests.

Several prominent conservation issues influence resource management in this subregion. Humboldt-Toiyabe National Forest is home to over 800 wild horses and burros. The Wild Free-Roaming Horses and Burros Act of 1971 (Public Law 92-195) protects wild horses and burros of the American West, but competition for forage and water resources with domestic livestock is controversial. Fragmentation and loss of sagebrush habitat have made the Columbia spotted frog a candidate endangered species, susceptible to extirpation in some areas. Protection of the species has been implemented in the greater Humboldt River watershed of central and northeastern Nevada. Conservation of greater sage-grouse, one of the highest profile natural resource issues in the West, is influenced by the loss of sagebrush ecosystems, with declining populations of sage-grouse in Nevada and Utah. Methods to sustain both the habitat and population of

sage-grouse include restoration of shrub-steppe ecosystems and control of nonnative species (Nevada Department of Wildlife 2013).

Intermountain Semi Desert Subregion

The Intermountain Semi Desert subregion covers 5.5 million acres. No national forests are located here, although 500,000 acres of wilderness area have been reserved, along with Craters of the Moon National Monument and Idaho National Laboratory in the north-central part of the subregion (fig. 2.11).

The western half of the subregion contains the Snake River Plain and Camas Plain. Uplands consist of the Owyhee Plateau located along the Idaho-Nevada border, and the Blackfoot Mountains. Flat plateaus and volcanic plains dotted with cinder cones and other volcanic remnants characterize the landscape. Deep river canyons, granite domes, and rugged peaks distinguish the Owyhee Mountains from the low-lying areas along the Snake River to the north, where basalt sheets underlie the irrigated plains. The eastern half of the subregion begins just north of the junction with Idaho, Utah, and Wyoming, extending west into the Great Basin. Desert habitats are uniformly distributed throughout the plains and plateaus.

The northwestern corner of the subregion typically has the same maritime climate trends as northern and central Idaho. Within the Snake River Plain, the climate is influenced by warm, dry Pacific air masses and cold, moist Arctic air masses that converge from western and northern directions, respectively. This interaction allows for freezing winter temperatures, summer thunderstorms, and hot, humid summers. The Snake River Plain and the area south to the Nevada/Idaho border receive an average of 10 to 15 inches of precipitation annually (Chandler 2003). However, the higher elevation Owyhee Mountains receive more precipitation and prolonged freezing temperatures during the winter. The Blackfoot Mountains have cool, dry winters and hot, dry summers.

The South Fork of the Boise River runs westward across the northwestern corner of the Intermountain Semi Desert subregion, joining the North and Central Forks of the Boise River at Arrow Rock Reservoir. Originating from the northeastern corner of the subregion, the Snake River cuts across the middle of the Snake River Plain from the Southern Greater Yellowstone subregion to the Oregon-Idaho border.

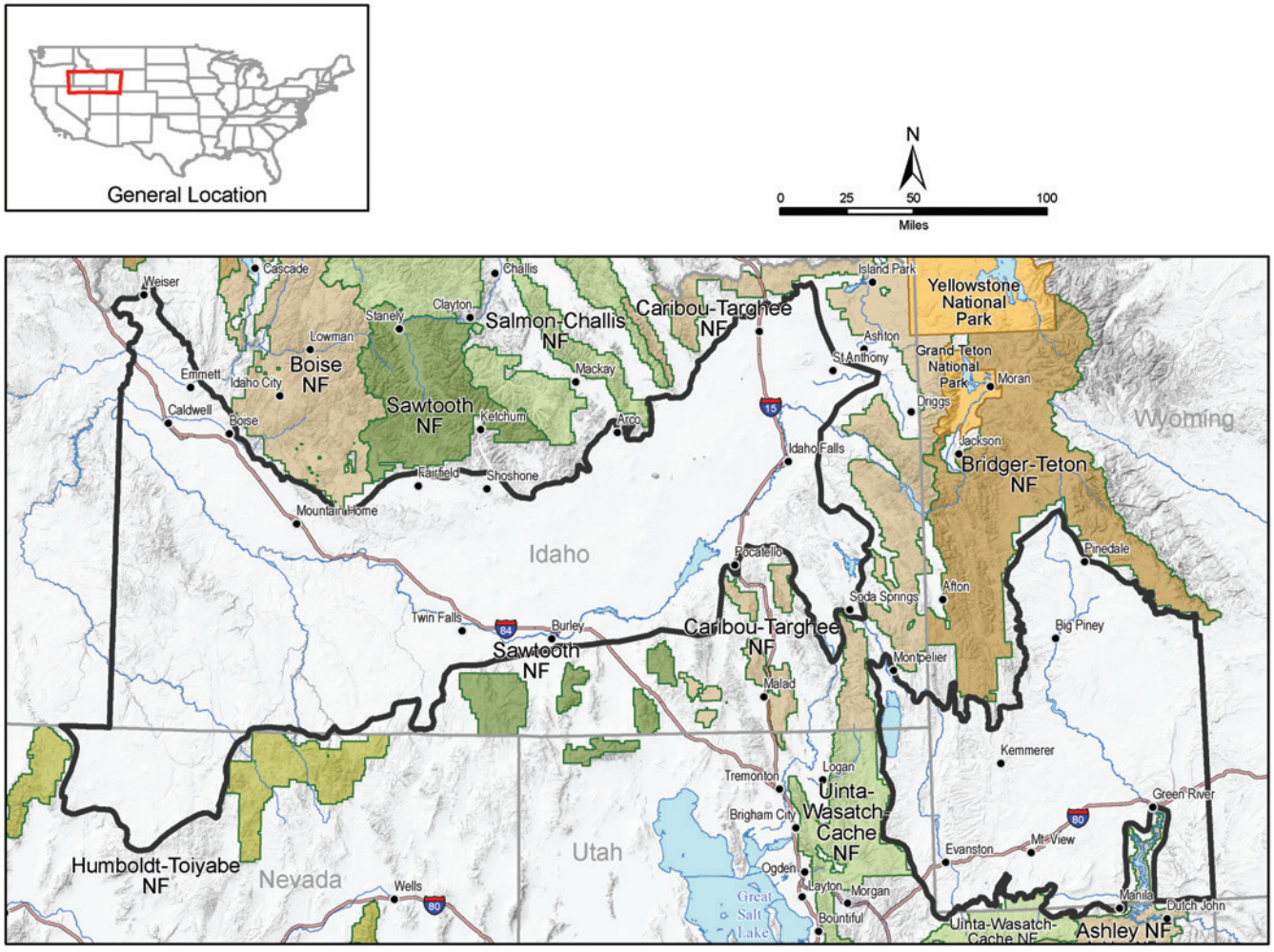


Figure 2.11—Location of the Intermountain Semi Desert subregion of the Intermountain Adaptation Partnership region.

Impoundments to the Snake River, such as those at Lake Walcott and American Falls Reservoir, provide hydroelectricity and irrigation water for nearby agriculture. In the northeastern areas, the Big Lost River disappears from its central Idaho source, drains to the Snake River aquifer below the surface of the volcanic rocks, and re-emerges near Thousand Springs Creek. Incising through canyons of the Owyhee Mountains, the Owyhee River flows generally northward to the Snake River near the Oregon-Idaho border, forming one of the largest subbasins of the Columbia River system.

The Snake River Plain is believed to have originated 16 to 17 million years BP as a hotspot of the Earth's crust moved from west to east, producing lava flows and basalt throughout the subregion. Large basalt flows erupted onto the surface in volcanic rift zones. Between 12,000 and 15,000 years BP, powerful floods cut through the Snake River basin from Lake Bonneville to the south, depositing alluvial sediments derived from adjacent mountain ranges. The Owyhee Plateau marks the highland portions of the subregion, characterized by warped volcanic deposits that eventually converge with the lower plains (Chandler 2003).

Sagebrush-steppe ecosystems are distributed throughout lower elevations in the subregion, giving way to mountain big sagebrush and other subalpine systems at higher elevations (fig. 2.12). The northern border of the western section of the subregion is dominated by subalpine ecosystems similar to those of the Middle Rockies subregion. Farther to the south, the high desert habitats of the Great Basin become more prevalent with more arid vegetation.

High desert habitats are dominated by open desert with low-lying sagebrush and grasslands. As the Snake River flows along the natural arc-shape of the plain, riparian zones are plentiful and contain habitat for willows (*Salix* spp.), quaking aspen, narrowleaf cottonwood (*Populus angustifolia*), and chokecherry (*Prunus virginiana*). Sagebrush shrublands, including mountain big sagebrush, greasewood, and shadscale (*Atriplex confertifolia*), cover most of the central and southern portions of lower elevations (Chandler 2003). Nonnative Russian thistle (*Salsola kali*), cheatgrass, and other species have degraded the once productive grazing land that lies between Twin Falls and the Boise River.

Wildfire frequency in the subregion has increased since the early 1900s. Before European settlement, fire intervals were 60 to 110 years, allowing for productive native grasslands to thrive (Manier et al. 2011). Introduction of cheatgrass has reduced intervals to less than 5 years in some places. After fires occur, cheatgrass regenerates quickly, limiting the growth of native vegetation and providing fuel for subsequent fires. Addressing this change in ecology and disturbance regimes is a major challenge for resource management in the Snake River Plain. In addition, juniper woodlands have expanded into sagebrush and grasslands of the lower plains, reducing wildlife habitat, water quality, and forage for grazing.

The subregion is home to a wide range of mammals, including mule deer, whitetail deer (*Odocoileus virginianus*), black bear, Rocky Mountain elk, bighorn sheep, pronghorn, and American bison. Smaller mammals include badger (*Taxidea taxus*), coyote, yellow pine chipmunk (*Tamias amoenus*), Great Basin pocket mouse (*Perognathus parvus*), and bobcat (Chandler 2003).

Greater sage-grouse can be found throughout the sagebrush-steppe ecosystem, especially in the Owyhee Canyonlands. Steeper cliffs of the Canyonlands provide important nesting and foraging habitat for raptor species, such as red-tailed hawk, golden eagle, and peregrine falcon. These species also populate the eastern and western plains, which are home to osprey, American kestrel (*Falco*



Figure 2.12—Highly incised river canyons, flat plateaus, and sagebrush-grasslands characteristic of the Intermountain Semi Desert subregion of the Intermountain Adaptation Partnership region, shown here in Birds of Prey National Conservation Area (photo: Natural Resources Conservation Service).

sparverius), bald eagle, and prairie falcon as well. In the juniper highlands and Blackfoot Mountains, iconic birds include common poorwill (*Phalaenoptilus nuttallii*), broad-tailed hummingbird (*Selasphorus platycercus)*, bushtit (*Psaltriparus minimus*), and green-tailed towhee (*Pipilo chlorurus*).

Within the Snake River system, anadromous fish, including steelhead trout and Chinook salmon, arrive in the spring. Interior redband trout, bull trout, mountain whitefish, and other fish species inhabit the Owyhee River and Bruneau River. The Owyhee River contains large populations of non-native brown trout and rainbow trout.

The Fort Hall Reservation, located in southeastern Idaho, is home to the Shoshone-Bannock Indian Tribes. The Fort Bridger Treaty of 1868 and the Indian Reorganization Act of 1934 granted self-governing rights to the tribes (Shoshone-Bannock Tribes 2015). Historically, the tribes consisted of hunters and gatherers who traveled in spring and summer collecting winter supplies. Riparian zones along the Snake River provided a diversity of plant food sources. Abundant salmon in the river were harvested year round, along with mammals that inhabited the river banks. American bison was valued as a primary food and raw material for tribal communities. By 1864, bison had vanished from the area, and land ownership changed from ancestral tribal lands to the reservation system. Resource management and land ownership policies in the surrounding area of the current-day reservation are administered by U.S. and Tribal Governments.

References

Ashley National Forest. 2010. Ashley National Forest: 2012 year in review. Vernal, UT: U.S. Department of Agriculture, Forest Service, Intermountain Region, Ashley National Forest.

Atwood, G. 2012. Geology of Utah. In: Powell, A.K., ed. Utah History Encyclopedia. Utah Education Network. http://www.uen.org/utah_history_encyclopedia/uheindex.html [Accessed June 19, 2016].

Black, D.V. 2011. The geologic history of Utah. Provo, UT: Walden School of Liberal Arts. 26 p. https://elementsunearthed.files.wordpress.com/2010/12/utah_geology.pdf [Accessed July 12, 2016].

Blackwell, J.; Reese, J. 2001. Draft environmental impact statement for the Caribou National Forest: Draft revised forest plan. Idaho Falls, ID: U.S. Department of Agriculture, Forest Service, Intermountain Region.

Blank, R.; Burkhardt, W.; Chatterton, W.; [et al.]. 2008. Great Basin wildfire forum: The search for solutions. Wildfire Forum for the Great Basin. Reno, NV: University of Nevada, Nevada Agricultural Experiment Station.

Brandt, J.P.; Morgan, T.A.; Keegan, C.E.; [et al.]. 2012. Idaho's forest products industry and timber harvest, 2006. Res. Bull. RMRS-RB-12. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Brown, M.J.; Chojnacky, D.C. 1991. Idaho's Forests, 1991. Res. Bull. INT-RB-88. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station.

Burden, C.B. 2015. Groundwater conditions in Utah, spring of 2015. Coop. Invest. Rep. 56. Salt Lake City, UT: U.S. Department of the Interior, U.S. Geological Survey; Utah Department of Natural Resources, Utah Department of Environmental Quality.

Cairns, C., ed. 2015. Rocky Mountain Research Station: Looking back on 2014. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 4 p.

Caribou-Targhee National Forest. 2005. Fire management plan, 2005. FMP-2005. Ashton, ID: U.S. Department of Agriculture, Forest Service, Caribou-Targhee National Forest. http://www.fs.usda.gov/Internet/FSE_DOCUMENTS /stelprdb5238562.pdf. [Accessed June 23, 2016].

Chandler, J.A., ed. 2003. Feasibility of reintroduction of anadromus fish above or within the Hells Canyon Complex. Gen. Tech. Rep. FERC-GTR-1971: E.3.1-2. Boise, ID: Idaho Power Company. https://www.idahopower.com/pdfs/Relicensing/hellscanyon/hellspdfs/techappendices/Aquatic/e31_02_ch03.pdf. [Accessed June 15, 2016].

Crane, M.F.; Fischer, W.C. 1986. Fire ecology of the forest habitat types of central Idaho. Gen. Tech. Rep. INT-GTR-218. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station.

Demarchi, D.A. 1994. Ecoprovinces of the central North American cordillera and adjacent plains. Gen. Tech. Rep. RM-GTR-254. Fort Collins, CO: U.S. Department of Agriculture, Rocky Mountain Forest and Range Experiment Station.

Dillon, J.; Grunder, S.A. 2008. Management plan for the conservation of Snake River white sturgeon in Idaho. Boise, ID: Idaho Department of Fish and Game. https://fishandgame.idaho.gov/public/fish/planSnakeWhiteSturgeon.pdf. [Accessed June 15, 2016].

Dixie National Forest. 2011. Oil and gas leasing on the Dixie National Forest: Final Environmental Impact Statement. Cedar City, UT: U.S. Department of Agriculture, Forest Service, Intermountain Region, Dixie National Forest. http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5325181.pdf [Accessed on June 20, 2016].

Fellin, D.G.; Dewey, J.F. 1982. Western spruce budworm. For. Insect Dis. Leafl. 53. Missoula, MT: U.S Department of Agriculture, Forest Service, Forest Health Protection.

Gibson, K.; Skov, K.; Kegley, S.; [et al.]. 2008. Mountain pine beetle impacts in high-elevation five-needle pines: current trends and challenges. R1-08-020. Missoula, MT: U.S. Department of Agriculture, Forest Service, Forest Health Protection.

Giunta, A.D.; Jenkins, M.J.; Hebertson, E.G.; [et al.]. 2016. Disturbance agents and their associated effects on the health of interior Douglas-fir forests in the Central Rocky Mountains. Forests. 7: 80.

Harris, A.G.; Tuttle, E.; Tuttle, S.D. 1997. Geology of national parks, 6th ed. Dubuque, ID: Kendall/Hunt Publishing Company.

Hatt, D.L. 2014. 2015–2019 Southeastern Utah economic development district comprehensive economic development strategy. Report prepared for the Southeastern Utah Economic Development District. Price, UT: U.S. Department of Commerce, Economic Development Administration. 150 p.

Headwaters Economics. 2014. High Divide region: Summary of recreation economy. Report written for the High Divide region. Bozeman, MT: Headwaters Economics. http://headwaterseconomics.org/wphw/wp-content/uploads/High_Divide_ Outdoor_Rec_Economy.pdf [Accessed June 20, 2016].

Heyerdahl, E.K.; Brown, P.M.; Kitchen, S.G.; [et al.]. 2011. Multicentury fire and forest histories at 19 sites in Utah and Eastern Nevada. Gen. Tech. Rep. RMRS-GTR-261WWW. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 192 p.

Idaho Department of Health and Welfare. 1999. Lemhi River Watershed TMDL: An allocation of nonpoint source pollutants in the water quality limited watersheds of the Lemhi River Valley. Boise, ID: Idaho Department of Health and Welfare, Division of Environmental Quality. http://www.deq.idaho.gov/media/454821_water_data_reports_surface_water_tmdls_lemhi_river_lelem_river_entire.pdf [Accessed June 14, 2016].

Jones, M. 1990. History of settlement and early livestock grazing in the area of the Boise National Forest. R4-1680-95-0001-14-8. U.S. Department of Agriculture, Forest Service. http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev3_042215.pdf [Accessed June 22, 2016].

Knight, D.H.; Jones, G.P.; Reiners, W.A.; [et al.]. 2014. Mountains and plains: The ecology of Wyoming landscapes, 2nd ed. Laramie, WY: University of Wyoming Biodiversity Institute.

Link, P.; Welhan, J.; Peterson, C.; [et al.]. 2000. Digital atlas of Idaho: Idaho's natural history online. Boise, ID: State of Idaho, Board of Education. http://imnh.isu.edu/digitalatlas/aboutus/about.htm. [Accessed June 15, 2016].

Lowrey, L.L. 2015. Evaluation of the extent of the non-native insect, balsam wooly adelgid, south of the Salmon River in Idaho from 2006–2014 by ground delimitation and aerial detection surveys. BFO-PR-2015-01. Boise, ID: U.S. Department of Agriculture, Forest Service, Forest Health Protection. http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprd3834265.pdf. [Accessed June 9, 2016].

Manier, D.J.; Aldridge, C.; Anderson, P.; [et al.]. 2011. Land use and habitat conditions across the southwestern Wyoming sagebrush steppe: Development impacts, management effectiveness and the distribution of invasive plants. Natural Resources and Environmental Issues. 17: Article 4. http://digitalcommons.usu.edu/cgi/viewcontent.cgi?article=1749&context=nrei. [Accessed July 13, 2016].

Mohlenbrock, R.H. 2006. This land: A guide to Western national forests. Berkeley, CA: University of California Press.

Morris, L.A. 2006. Fire history of the City of Rocks National Reserve from 1926 to 2005. Cooperative Agreement H8R07010001. Boise, ID: Idaho State Parks and Recreation.

Nevada Department of Wildlife. 2013. Sage-grouse conservation. Reno, NV: Nevada Department of Wildlife. http://www.ndow.org/Nevada_Wildlife/Sage_Grouse [Accessed June 20, 2016].

O'Laughlin, J.; Adams, D.L.; Bunting, S.C.; [et al.]. 1993. Forest health conditions in Idaho. Rep. RP-11. Boise, ID: Idaho Forest, Wildlife and Range Policy Analysis Group. http://www.idahoforests.org/health3.htm [Accessed June 17, 2016].

Price, J.G. 2004. Geology of Nevada. Proceedings, 39th forum on the geology of industrial minerals; 2003 May 19–21; Reno, NV. Proc. NBMG-P-33. Reno, NV: Nevada Bureau of Mines and Geology.

Pyke, D.A.; Chambers, J.C.; Pellant, M.; [et al.]. 2015. Restoration handbook for sagebrush steppe ecosystems with emphasis on greater sage-grouse habitat: Part 1, concepts for understanding and applying restoration. Circ. 1416: 44. Reston, VA: U.S Department of the Interior, U.S. Geological Survey.

Reddy, S.D. 1993. The empty land: The search for the Nez Perce on the Payette National Forest. McCall, ID: U.S. Department of Agriculture, Forest Service, Heritage Program. http:// http://www.fs.usda.gov/detailfull/payette/learning/history-culture/?cid=fsm9_031023&width=full [Accessed June 18, 2016].

Robson, S.; Kingery, J. 2006. Idaho native plants for roadside restoration and revegetation programs: A report for the Idaho Transportation Department. Moscow, ID: University of Idaho, Idaho Transportation Department. http://itd.idaho.gov/highways/ops/maintenance/roadside/ documents/RP171Roadside_RevegReveget.pdf [Accessed June 13, 2016].

Schwandt, J.; Kearns, H.; Byler, J. 2013. White pine blister rust: General ecology and management. Insect Dis. Manage. Ser. 14.2. Boise, ID: U.S Department of Agriculture, Forest Service, Forest Health Protection. http://www.fs.usda.gov/Internet/FSE_ DOCUMENTS /stelprdb5415080.pdf [Accessed June 18, 2016].

Seligman, D.; Peterson, M.; Price, S.; [et al.]. 2008. World's major rivers: An introduction to international water law with case studies. Las Vegas, NV: Colorado River Commission of Nevada. http://crc.nv.gov/docs/Worlds_Major_Rivers.pdf [Accessed June 18, 2016].

Shaw, J.D.; Long, J.N. 2007. Forest ecology and biogeography of the Uinta Mountains, U.S.A. Arctic, Antarctic, and Alpine Research. 39: 614–628.

Shoshone-Bannock Tribes. 2015. History of the Shoshone-Bannock Tribes. Fort Hall, ID: Shoshone-Bannock Tribes of the Fort Hall Reservation. http://www.shoshonebannocktribes.com/shoshone-bannock-history.html [Accessed June 18, 2016].

Simmons, V.M. 2000. The Ute Indians of Utah, Colorado, and New Mexico. Boulder, CO: University Press of Colorado.

State of Wyoming. 2013. Wyoming history. Cheyenne, WY: State of Wyoming. http://www.wyo.gov/about-wyoming/wyoming-history [Accessed June 24, 2016].

Tilley, D.; Ogle, D.; St. John, L.; [et al.]. 2013. Threatened, endangered, and candidate plant species of Idaho. Tech. Note TN-51. Boise, ID: U.S. Department of Agriculture, Natural Resources Conservation Service.

U.S. Department of Agriculture, Forest Service [USDA FS]. 2011. Payette National Forest land and resource and management plan: Five-year evaluation report 2004–2008. McCall, ID: U.S. Department of Agriculture, Forest Service, Intermountain Region. http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5292644.pdf [Accessed June 22, 2016].

U.S. Department of Agriculture, Forest Service [USDA FS]. 2016. Cut and sold (new). CUTS203F. Quarterly report prepared for the Forest Service. Quart. Rep., FY 2016, Q1. Washington, DC: U.S. Department of Agriculture, Forest Service.

Utah Division of Wildlife Resources. 1991. Utah's wildlife habitats. Utah Nongame Leaflet 2(1). Salt Lake City, UT: Utah Department of Natural Resources. http://digitallibrary.utah.gov/awweb/awarchive?type=file&item=29617 [Accessed June 22, 2016].

West, N.E. 1983. Southeastern Utah galleta-threeawn shrub steppe. In: West, N.E., ed. Ecosystems of the world: Temperate deserts and semi-deserts. Amsterdam, The Netherlands: Elsevier: 413–421.

Western Regional Climate Center [WRCC]. 2016. Climate of Nevada. Reno, NV: Western Regional Climate Center. http://www.wrcc.dri.edu/narratives/NEVADA.htm [Accessed November 19, 2016].

Chapter 3: Historical and Projected Climate

Linda A. Joyce and Marian Talbert

Introduction

The Intermountain Adaptation Partnership (IAP) region is characterized by extreme temperatures and precipitation; it is home to some of the driest, hottest, and coldest locations in the conterminous United States. The region has numerous mountain ranges, high-elevation basins and valleys, and low-elevation mesas and canyons. Climate is influenced by this diverse and complex terrain. The IAP region is bounded on the eastern side by the Rocky Mountains and on the western side by the Sierra Nevada. For this region, moisture comes predominantly from the Pacific Ocean, and consequently much of the moisture occurs as orographic precipitation in the mountains.

This chapter provides a brief discussion of the current climate in each of the six subregions to set the context for projected changes in climate under two future scenarios. The Great Basin and Semi Desert, and the Intermountain Semi Desert subregions are warmer and drier than the Southern Greater Yellowstone and Middle Rockies subregions. The Plateaus, and Uintas and Wasatch Front subregions have intermediate amounts of precipitation. Within each subregion, climate influences the ecosystem services that forests and rangelands provide. Thus, an understanding of how climate may change in the future is vital for long-term planning and management.

This chapter focuses on the historical record of climate, primarily temperature and precipitation, and the projected changes in temperature and precipitation. Chapter 4 describes the effects of climate on hydrological processes, snowpack and glaciers, streamflow, sediment yield, and drought. The impacts of climate change on wildfire and geologic processes, such as mass wasting, are discussed in Chapter 8. Other chapters review literature on the effects of climate change on particular resources, such as individual species in Chapter 9.

Assessing Climate Futures for Natural Resource Management

Information on weather and climate are used to inform the decisionmaking process in natural resource management. Day-to-day resource management practices are implemented with information on real-time conditions of temperature, precipitation, wind, humidity, and other meteorological factors. These specific conditions over a relatively short time period and in a particular place are what we call weather. This information is used to make on-the-ground real-time decisions, such as the start of prescribed fire activities.

Long-term resource management strategies and plans are informed by the observed dynamics of precipitation, temperature, wind, snowfall, and other measures of weather over a long time period in a particular place. Climate is defined by this long-term meteorological information. Understanding the climate of an area assists managers in identifying both the general characteristics of the area and the risks associated with potential extremes of weather conditions, such as flooding, drought, wildfire, and extreme heat or cold events. Typically, the most recent three full decades (e.g., 1981–2010) are used to determine the normal climate, or the average conditions on an annual, monthly, or daily basis. Long-term records (>30 years) are also used to assess the risk of specific weather events and thresholds that have relevance to management, such as in the sizing of culverts and bridges.

Although long-term records of climate can help to establish the characteristics of the average climate or the probability of certain extreme events, future climate may be different than the past 30 to 60 years (Walsh et al. 2014). To understand how climate might change, scientists use global climate models (GCMs), which are supercomputer-based simulations that represent the key components, interactions, and feedback processes of the climate system, based on fundamental physical laws and decades of observations of the atmosphere, oceans, ice sheets, and biosphere. These models have been used to study the physical dynamics of the atmosphere and the interaction between the atmosphere and the surface of the Earth, as well as interactions with ocean currents moving heat around the globe. Information on future changes in climate provided by GCMs can be helpful in understanding how the environmental settings of plants, animals, and habitats may change in the future; how runoff and seasonal flows may vary with precipitation and timing of snowmelt; and how wildfire, insects and disease patterns may change.

Scenarios

Climate models have been an important part of the Intergovernmental Panel on Climate Change (IPCC) assessments since 1990. In 1995, atmospheric scientists came together to coordinate a standardized set of model runs (also called experiments) for evaluating changes to past and future global climate: the Coupled Model Intercomparison Project (CMIP) (Meehl et al. 2007). This project also developed a web portal where these GCM results could be archived and made available for use by other scientists. This

approach allows for a rigorous comparison of results across the models because the models have similar initial conditions and similar changes in atmospheric chemistry and land use cover over time. This approach improves our understanding of the range of possible future climate change.

Many of the model experiments focus on improving the ability of the model to describe climate; however, these models can also be used to look at the evolution of climate over time when a description of how the future may develop is constructed. A climate scenario is a simplified representation of future climate, based on climatological relationships constructed for investigating the potential consequences of human-caused climate change (Stocker et al. 2013). Climate dynamics are influenced by land surface changes, such as building cities or changing land use from forest to agriculture, by the chemical composition of the atmosphere, and by human contributions to the atmosphere through land management, energy sources, and industrial processes. Constructing a scenario requires describing how these forces may develop over time. The goal of working with scenarios is to understand uncertainties and alternative conditions associated with climate change, thus informing decisions or options for a range of possible futures (IPCC Data Distribution Center 2016).

Climate scenarios in the last three IPCC assessments have been constructed in two ways (table 3.1). In the third and fourth IPCC assessments, scenario development started with specific assumptions about population growth, economic growth, and policies related to alternative energy and conventional fossil fuel sources; then the resultant greenhouse gas emissions to the atmosphere were projected (Solomon et al. 2007). These scenarios were called SRES scenarios, named for the report that developed them, the Special Report on Emission Scenarios (Nakićenović et al. 2000). These different combinations of population, economic growth, and energy policy resulted in scenarios that ranged from low emissions (B1) to high emissions (A1FI), with a range of scenarios in between.

For the IPCC Fifth Assessment Report (Stocker et al. 2013), scenario development focused on radiative forcing in the atmosphere. Here the emphasis was on adding different amounts of energy to the climate system over time. Scientists reviewed current estimates on radiative forcing, the total amount of extra energy entering the climate system throughout the 21st century and beyond. They used this information to construct a set of scenarios that would bound these estimates, from lesser amounts of energy entering the climate system to greater amounts of energy. These

Table 3.1—Scenarios used by climate models in the Special Report on Emission Scenarios (SRES) and representative concentration pathway (RCP) scenarios in the Intergovernmental Panel on Climate Change (IPCC) assessment reports.

	SRES scenarios	RCP scenarios
Definition	A plausible representation of the future development of emissions of substances that are potentially radiatively active (e.g., greenhouse gases, aerosols) based on a coherent and internally consistent set of assumptions about driving forces (such as demographic and socioeconomic development, technological change) and their key relationships. Concentration scenarios, derived from emissions scenarios, are used as input to a climate model to compute climate projections.	Scenarios that include time series of emissions and concentrations of the full suite of greenhouse gases and aerosols and chemically active gases, as well as land use/land cover (Moss et al. 2008). The word "representative" signifies that each RCP provides only one of many possible scenarios that would lead to the specific radiative forcing characteristics. The term "pathway" emphasizes that not only the long-term concentration levels, but also the trajectory taken over time to reach that outcome, are of interest (Moss et al. 2010).
Different scenarios	Scenario family is a set of scenarios that have a similar demographic, societal, economic, and technical change storyline. Four scenario families compose the SRES scenario set: A1, A2, B1, and B2.	RCP 2.6: One pathway where radiative forcing peaks at approximately 3 W m^{-2} before 2100 and then declines. RCP 4.5 and RCP 6.0: Two intermediate stabilization pathways in which radiative forcing is stabilized at approximately 4.5 W m^{-2} and 6.0 W m^{-2} after 2100. RCP 8.5: One high pathway for which radiative forcing reaches greater than 8.5 W m^{-2} by 2100 and continues to rise for some amount of time.
Use in models	Used to drive climate models in CMIP3[a]	Used to drive climate models in CMIP5.
Use in IPCC assessments	Houghton et al. (2001), Solomon et al. (2007).	Stocker et al. (2013).

[a] CMIP = Coupled Model Intercomparison Project; versions 3 and 5 are cited here.

scenarios are called representative concentration pathways (RCPs) (van Vuuren et al. 2011) because these scenarios represent one of many possible ways in which population, economic growth, and energy policy would lead to the specific radiative forcing characteristics. Consequently, the scenarios are defined by the amount of energy that is added to the atmosphere. For example, RCP 2.6 assumes a total radiative forcing increase of 2.6 Watts per square meter (0.82 British thermal units per square foot) by 2100, whereas RCP 8.5 assumes a much larger increase in the radiative forcing (8.5 Watts per square meter [2.7 British thermal units per square foot]). Intermediate scenarios include RCP 4.5 and RCP 6.0.

The RCPs focused on the implications of energy added to the atmosphere and consequently were not designed to mimic particular SRES scenarios. But because the SRES scenarios have been used for many analyses of climate change effects on natural resources (Walsh et al. 2014), an understanding of how the RCP and the SRES scenarios compare is helpful in interpreting past analyses. The comparison by Rogelj et al. (2012) considers climate sensitivity uncertainty, synthesizes the understanding of climate system and carbon cycle behavior, and is constrained by observed historical warming. Rogelj et al. (2012) identify analogs between RCP 4.5 and SRES B1, RCP 6.0 and SRES B2, and RCP 8.5 and SRES A1FI (table 3.2). The SRES A1B has a greater range in temperature changes than RCP 6.0, and a warmer upper range. Rogelj et al. (2012) note that temporal patterns differ between the SRES and RCP scenarios.

Climate Projections for the Region

This report uses scenarios RCP 4.5 and RCP 8.5 to explore future climate in the IAP region. These scenarios capture a moderate and a high future warming. In addition, more GCMs have used these scenarios than RCP 2.6 or RCP 6.0. The breadth of these scenarios and the availability of a larger set of projections enhance our understanding of the possible ranges in future climate. The term "climate projection" is used to describe the results of a climate model when forced by a particular scenario.

For an overview of projected future climate in the IAP region, we use the most recent climate change projections based on RCP 4.5 and RCP 8.5 scenarios (Stocker et al. 2013). Output from GCMs is at a scale too coarse to represent climate patterns in subregions and management areas relevant for the IAP. Therefore, we drew on climate projections that had been downscaled using the bias-correction and spatial disaggregation (BCSD) method (Maurer et al. 2007). We used projections from 36 climate models for RCP 4.5 and 34 climate models for RCP 8.5 (see Appendix 2). The variables available for each BCSD climate projection include monthly precipitation and monthly surface air temperature for the 1950–2099 period. Spatial resolution of the data is 1/8 degree latitude-longitude (about 7 square miles), and data cover the entire IAP region. Climate projections were archived by CMIP; hereafter climate projections that used SRES scenarios are referred to as "CMIP3" and climate projections that used the RCP scenarios as "CMIP5."

Historical mean annual temperature and precipitation vary across the IAP region (figs. 3.1, 3.2). We use a base period of 1979–2009 for the mean historical climate, and show mean temperatures and precipitation projected for two periods: 2030–2059 and 2070–2099. These time periods were selected to summarize climate that has influenced the current environmental conditions (base period) and two future periods that will be relevant to long-term management action (e.g., road construction, management of hydrological infrastructure, or vegetation planting). Historical mean

Table 3.2—Probabilistic estimates of temperature increase above preindustrial levels based on representative equilibrium climate sensitivity distribution for six Special Report on Emission Scenarios (SRES) market scenarios and four representative concentration pathway (RCP) scenarios (Rogelj et al. 2012).

	2090–2099		2100	
Scenario	Median	66% range	Median	66% range
	---------- °F ----------			
SRES B1	4.3	3.6–5.6	4.5	3.6–5.8
SRES A1T	5.2	4.5–6.7	5.4	4.5–6.8
SRES B2	5.2	4.3–6.3	5.4	4.7–6.7
SRES A1B	6.1	5.0–7.6	6.3	5.2–7.9
SRES A2	7.0	5.8–8.6	7.6	6.3–9.4
SRES A1F1	8.5	7.0–10.4	9.0	7.4–11.2
RCP 3-PD (2.6)	2.7	2.3–3.4	2.7	2.3–3.4
RCP 4.5	4.3	3.6–5.2	4.3	3.6–5.4
RCP 6.0	5.2	4.5–6.5	5.4	4.7–6.7
RCP 8.5	8.3	6.8–10.3	8.8	7.2–11.0

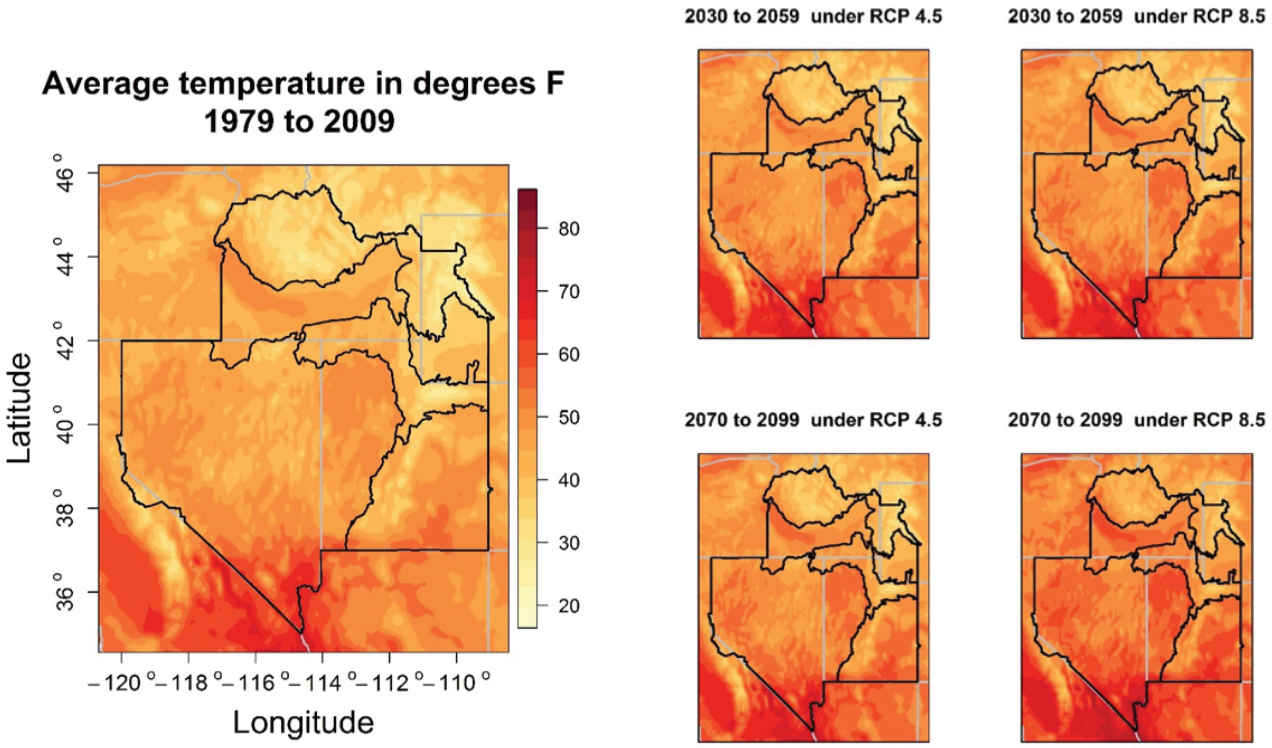


Figure 3.1—(a) Historical (1979–2009) and (b–e) projected (2030–2059 and 2070–2099) mean annual monthly temperature (°F) under RCP 4.5 and RCP 8.5 scenarios for the entire Intermountain Adaptation Partnership region. Projected climate is the 36-model mean for RCP 4.5 and the 34-model mean for RCP 8.5. Spatial resolution of the data is 1/8th degree latitude-longitude.

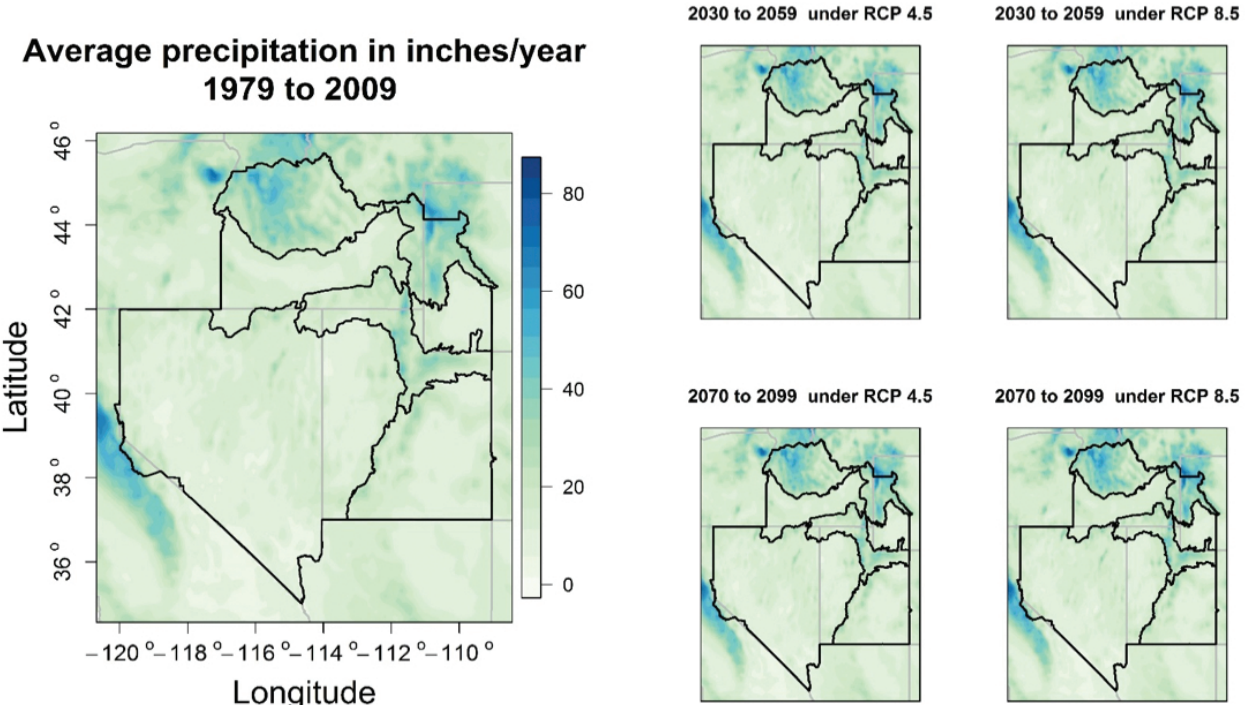


Figure 3.2—(a) Historical (1979–2009) and (b–e) projected (2030–2059 and 2070–2099) total annual precipitation (inches) under RCP 4.5 and RCP 8.5 scenarios for the entire Intermountain Adaptation Partnership region. Projected climate is the 36-model mean for RCP 4.5 and the 34-model mean for RCP 8.5. Spatial resolution of the data is 1/8th degree latitude-longitude.

annual temperatures can vary from nearly 20 °F to 75 °F within the IAP region. Historical total annual precipitation ranges from less than 3 inches to above 70 inches. All subregions of the IAP are expected to see increases in mean annual temperature. Temperatures are projected to increase more by the end-of-century period (2070–2099) and under the warmer scenario, RCP 8.5. Precipitation changes are less consistent; the northern parts of the IAP region may have precipitation increases.

Several resource chapters in this report have drawn on published literature about the effects of climate change on natural resources which used SRES scenarios and CMIP3 models. The question arises as to the regional differences between the CMIP3 projections under SRES scenarios and the CMIP5 projections under RCP scenarios. Understanding these differences may be helpful in interpreting the different projections used across these studies. Here, we compare CMIP5 climate projections under RCP 4.5 and RCP 8.5 scenarios from this study with CMIP3 models under the SRES scenario A1B, which has been used in several natural resource assessments, including the cold-water fisheries analysis in chapter 5.

To identify differences between these two sets of scenarios, we compare the projected changes in mean temperature with change in total annual precipitation over a common period for the IAP region. We obtained 10 CMIP3 global climate projections for the A1B scenario (moderate emissions) from Littell's group. We estimate the change in temperature and percentage change in precipitation between the future period (2040–2060) and the historical period (1979–2009) for the CMIP3 models that Littell et al. (2011) used and the CMIP5 models used in this study. In figure 3.3, the projected change in mean annual temperature is shown on the horizontal axis, and the percentage change in precipitation is shown on the vertical axis. Change is described as the difference in temperature and percentage change in precipitation between historical and projected future values. We show all CMIP5 models used in this study. We show only the mean (ensemble) of all 10 CMIP3 models and two individual models (pcm1 and miroc_3.2), as these projections from Littell et al. (2011) are used most often.

Across all CMIP5 models, projected change in temperature by the 2040–2060 period ranges from just under 2 °F to nearly 8 °F (fig. 3.3). Generally, the projected change for CMIP5 models under the RCP 8.5 scenario (shown in red) is greater than the change projected under the RCP 4.5 scenario (shown in yellow), but not always. For example, the RCP 8.5 projections for models FIO-ESM (number 15 in fig. 3.3) and MRI-CGCM3 (number 34 in fig. 3.3) show temperature changes that are less than the median changes for all RCP 4.5 models. The projected change in precipitation ranges from a decrease of about 12 percent to an increase of nearly 30 percent. However, 28 of the 36 projections under RCP 4.5 and 29 of the 34 projections under RCP 8.5 indicate an increase in precipitation. Across the IAP region, mean temperatures are projected to warm by 3.5 °F to almost 5 °F over the next 50 years, with precipitation projected to increase slightly, by 5 to 8 percent.

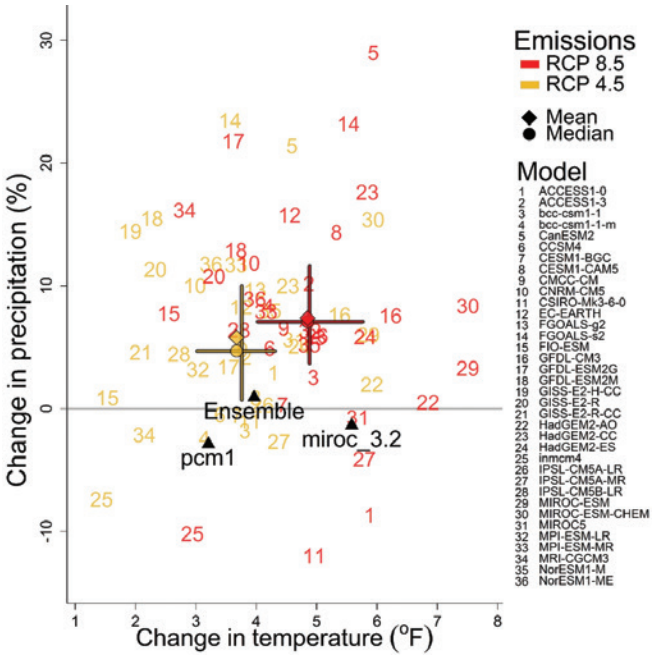


Figure 3.3—For the entire Intermountain Adaptation Partnership region, change in mean annual temperature (°F) and total annual precipitation (%) from the simulated historical climate (1979–2009) and the projected climate (2040–2060) using the CMIP5 RCP 4.5 and RCP 8.5 scenarios and the CMIP3 A1B scenario. Each CMIP5 model result is labeled by a number with a key in the legend (e.g., 29 is MIROC-ESM) in colors to indicate RCP 4.5 (yellow) and RCP 8.5 (red) (see table 3.2). The crosses in the middle represent the median and 25 to 75 percent of the RCP 4.5 and the RCP 8.5 projections used in this study. The mean values for the CMIP5 changes are shown on the figure as colored diamonds. The CMIP3 results are labeled in black triangles (see table 3.2 and Littell et al. [2011]).

The CMIP3 projections for the A1B scenario are within the mean temperature range of the CMIP5 projections (fig. 3.3). The A1B ensemble projection of a nearly 4 °F increase in temperature is similar to the mean increase for the RCP 4.5 scenario; both of these scenarios are considered moderate in terms of future warming. The two individual model projections under the A1B scenario span the temperature range of the individual model projections under RCP 4.5. For precipitation, the CMIP3 models under the A1B scenario project a slight increase (ensemble) and decreases in precipitation (pcm1 and miroc_3.2). We conclude that when this set of CMIP3 results are compared with CMIP5 results for the IAP region, future temperatures are projected to be similar. However, CMIP5 precipitation projections collectively show a greater likelihood of increases in precipitation than the CMIP3 projections (fig. 3.3). This slight increase in projected precipitation with the CMIP5 models might be considered when evaluating the impact of analyses using CMIP3 climate projections.

Climatic Variability and Change in the Subregions

Historical Climate

To understand historical climatic variability and trends at the subregional scale in the IAP region, we compared three gridded datasets: Parameter-elevation Regressions on Independent Slopes Model (PRISM) (PRISM Climate Group 2014), Maurer (Maurer et al. 2002), and TopoWx (Oyler et al. 2015a). These three datasets used observed point measurements of precipitation, temperature, and other climatic factors to produce continuous, digital grid estimates of monthly, yearly, and event-based climatic parameters. Because of differences in the station data used by these gridded products as well as the models and assumptions used to interpolate to a grid, these models do not always agree on the historical climate or trend for a region. For example, in the western mountains, PRISM has been shown to have an artificial amplification in warming trend (Oyler et al. 2015b).

The Middle Rockies subregion encompasses central Idaho, an area known as the "Idaho Batholith," and the Salmon River Mountains. Climate is strongly influenced by interactions with topography, elevation, and aspect. Westerly winds bring in moisture from the Pacific Ocean, but moisture is precipitated over the western blocking mountains. Elevations range from 3,000 to 10,000 feet, with the highest peaks in the Salmon River Mountains. The deep dissections of this subregion can be seen in the ridge patterns in temperature (fig. 3.1a) and large gradients in precipitation (fig. 3.2a).

Climate in the Middle Rockies is characterized by cold winters with moderate to heavy snow accumulations at higher elevations. Throughout the 1940–2009 period, mean minimum temperatures showed a distinct increasing trend (fig. 3.4). Mean maximum temperature ranged from 48 to 54 °F over this period (fig. 3.5). As with other subregions, no warming trend is evident across the entire time period. In the last 25 years, however, the Middle Rockies maximum temperatures showed a slight increasing trend (fig. 3.5); similar increasing trends in summer temperatures were noted by Isaak et al. (2010) for the Upper Boise River watershed. Annual precipitation over the historical period ranged from 20 inches to greater than 35 inches per year. Precipitation patterns were highly variable, with a slight downward trend in the last 25 years (fig. 3.6).

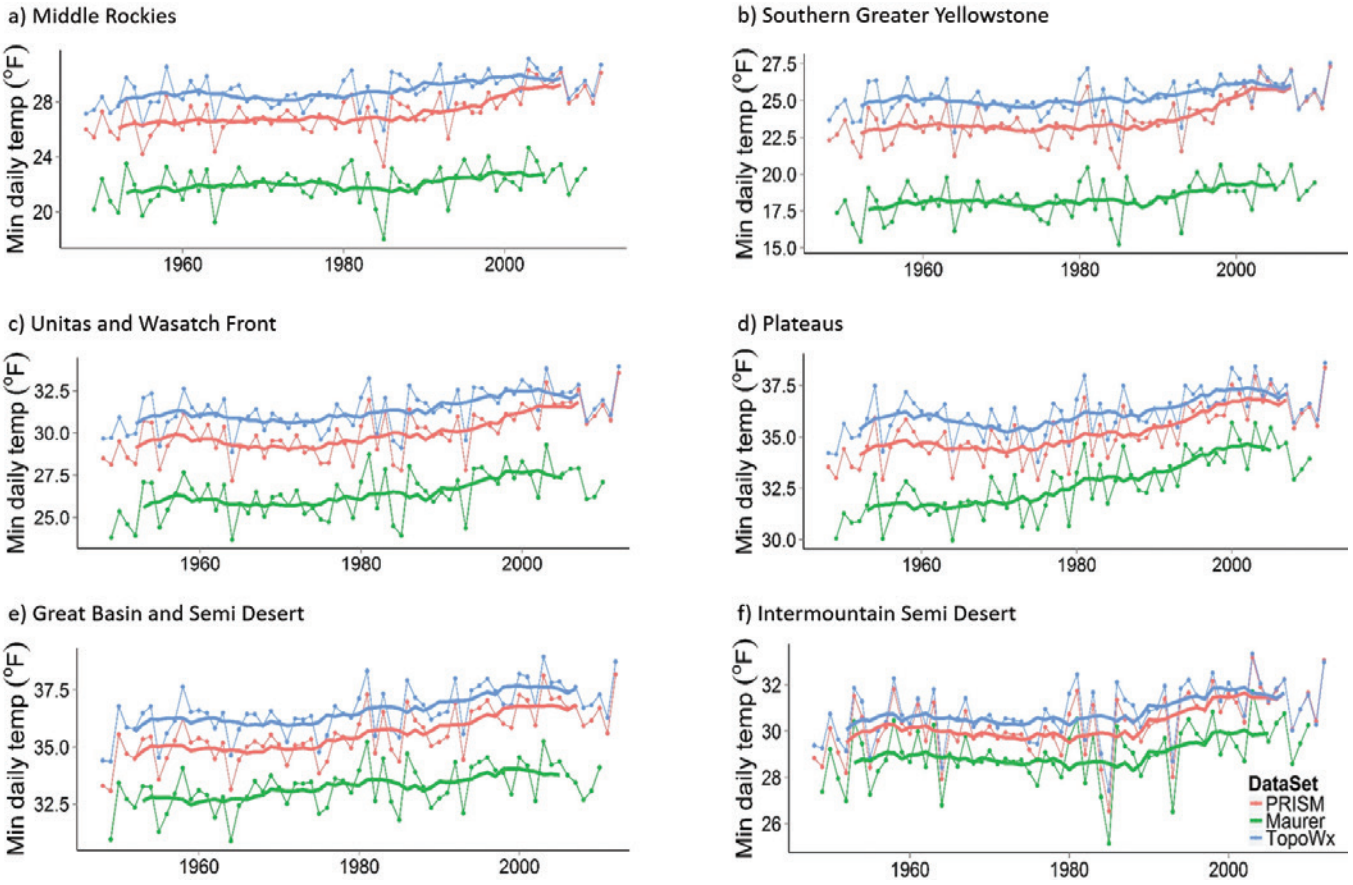


Figure 3.4—Annual historical mean monthly minimum temperature from the monthly gridded PRISM, Maurer, and TopoWx datasets for 1949–2010 for all six subregions of the Intermountain Adaptation Partnership region. The heavy lines are the 10-year rolling average from each dataset to show short-term trends.

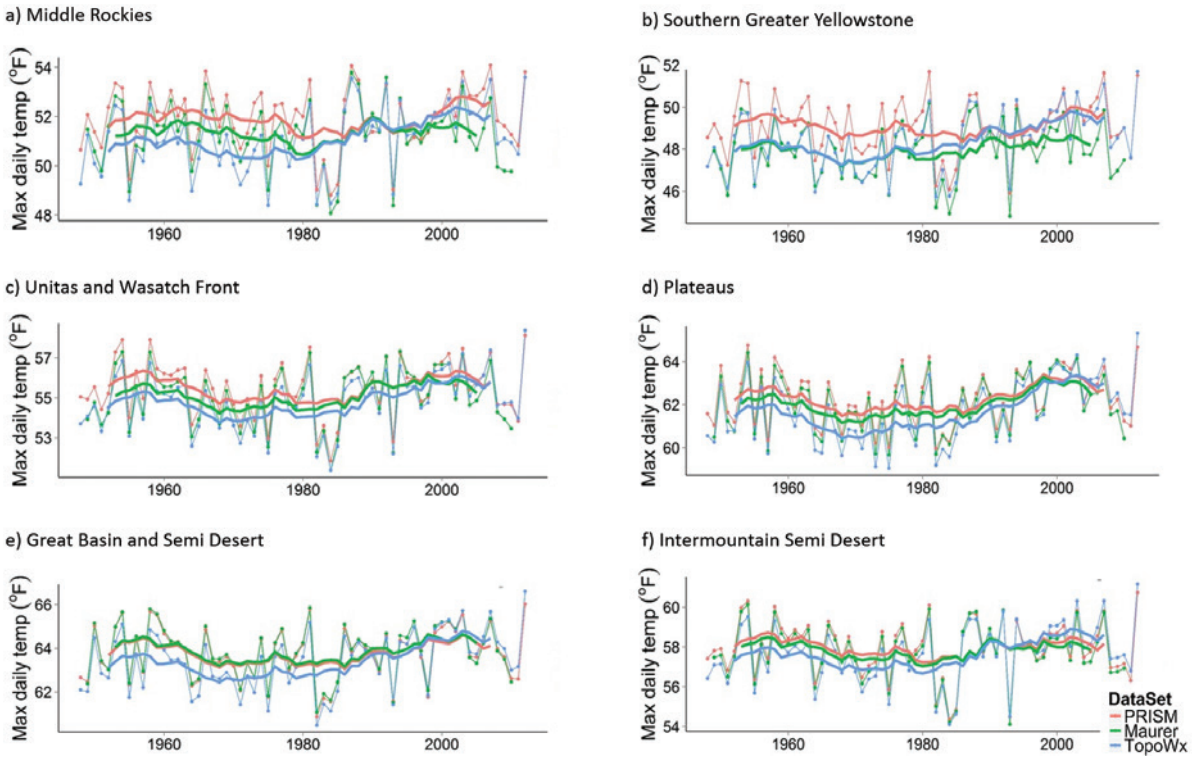


Figure 3.5—Annual historical mean monthly maximum temperature from the monthly gridded PRISM, Maurer, and TopoWx datasets for 1949–2010 for all six subregions of the Intermountain Adaptation Partnership region. The heavy lines are the 10-year rolling average from each dataset to show short-term trends.

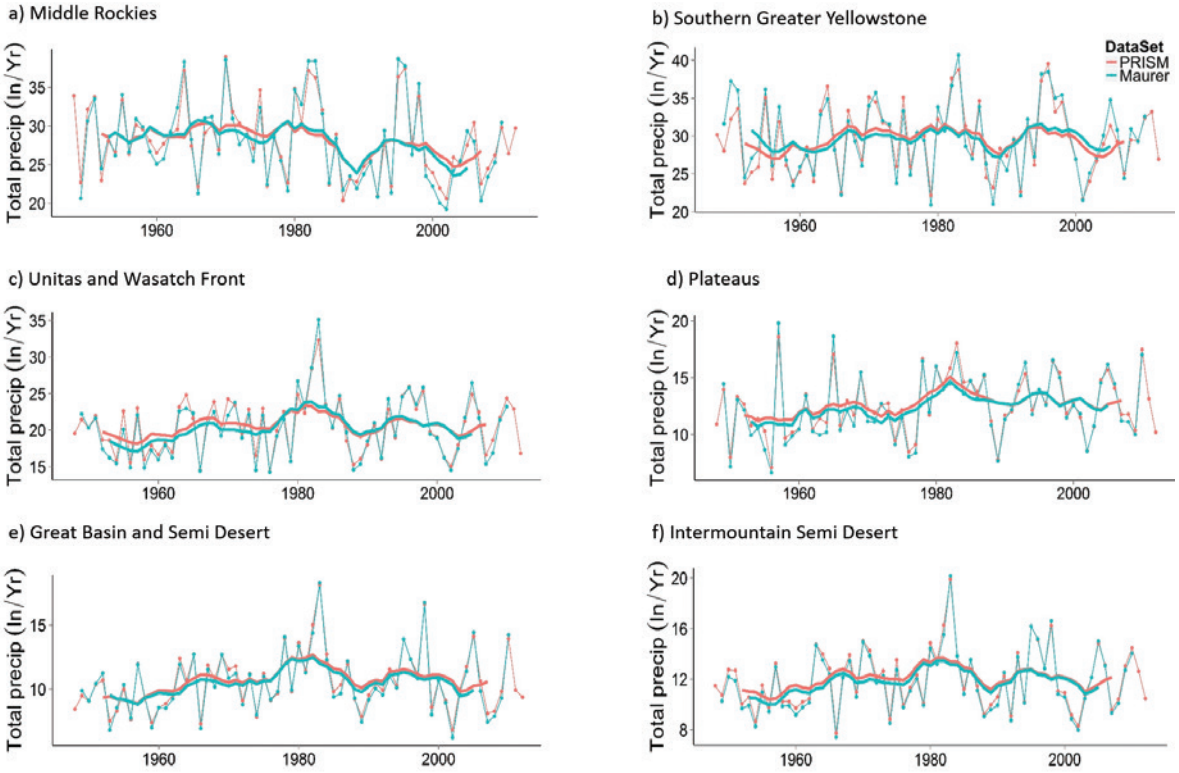


Figure 3.6—Historical total annual precipitation from the monthly gridded PRISM and Maurer et al. (2007) datasets for 1949–2010 for all six subregions of the Intermountain Adaptation Partnership region. The heavy lines are the 10-year rolling average from each dataset to show short-term trends.

The Southern Greater Yellowstone subregion includes Grand Teton National Park, and the Targhee and Bridger-Teton National Forests. This subregion and the Middle Rockies are the coldest subregions (figs. 3.4, 3.5). Mean minimum temperatures in the Southern Greater Yellowstone subregion showed an increasing trend over the last 50 years of the historical period (fig. 3.4). Monahan and Fisichelli (2014) reported that recent annual mean temperatures (last 10-, 20-, or 30-year periods) for Grand Teton National Park were higher than 90 percent of the historical temperatures over the 1901–2012 period. They also found that recent minimum temperatures in the coldest month (last 10-, 20-, or 30-year periods) were greater than 95 percent of the historical record. In contrast, maximum temperatures showed an increasing trend only during the last 25 years (fig. 3.5).

Climate in the Southern Greater Yellowstone subregion, as in the Middle Rockies, is strongly influenced by the mountains and interactions among topography, elevation, and aspect. Few mountains block the western passage of moist air from the Pacific Ocean to the Teton Range; consequently, the Teton Range, along with portions of southwestern Yellowstone National Park, are among the wettest areas in the larger Greater Yellowstone Area. Annual precipitation on the west side of the Teton Range can exceed 70 inches, with most of this precipitation falling as snow. In contrast, precipitation on the east side of the Teton Range can be as little as 19 inches (Davey et al. 2006). The Southern Greater Yellowstone subregion is the wettest in the IAP region (fig. 3.6). No trend is found in annual precipitation over the last 50 years although distinct wet and dry periods occurred. Davey et al. (2006) describe a 6-year drought (1999–2005) as the longest drought in the Grand Teton National Park since the Dust Bowl drought of the 1930s.

Climate in the Uintas and Wasatch Front can be characterized as humid continental, with warm to hot summers, cold winters with abundant snowfall, and no distinct dry period (Gillies and Ramsey 2009). As with the Middle Rockies and the Southern Greater Yellowstone subregions, climate in this subregion is strongly influenced by the terrain and interactions among topography, elevation, and aspect. The Wasatch Range generally runs north to south, and the Uinta Mountains extend east and west. These ranges crest above 10,000 feet, with the highest point of 13,498 feet at Kings Peak in the Uinta Mountains. As in most of the subregions, minimum temperatures trended upward over the last 50 years (fig. 3.4). Mean maximum temperatures were between about 53 and 57 °F, with no clear trends over the historical period (fig. 3.5). Gillies et al. (2012) concluded that during the last half-century, the proportion of winter (January–March) precipitation falling as snow across Utah has decreased by 9 percent, the result of a significant increase in rainfall combined with a minor decrease in snowfall. Although warming temperatures play a role, climate features such as circulation patterns also contribute to these changes.

Mean annual precipitation within the Uintas and Wasatch Front subregion ranged from 10 inches to more than 40 inches (fig. 3.6). Over the last 50 years, precipitation exhibited no annual trend, although dry or wet periods are evident. Morrisette (1988) reported that the 1982–1986 precipitation reached 134 percent of normal in the Salt Lake drainage area, resulting in historically high levels in the Great Salt Lake. Using tree-ring analyses, DeRose et al. (2015) concluded that 1986 was the fourth wettest year over the last 1,200 years in the Bear River watershed. They also reported that although the later part of the 20th century was the second wettest period (40 years), the first half was the fourth driest period in the past 1,200 years. Using tree rings to reconstruct streamflow for the Weber River, Bekker et al. (2014) reported that the 20th-century instrumental record includes the fewest extreme dry years in the 576-year, tree-ring-based reconstruction. In the Uinta Mountains, severe droughts have occurred, on average, two to five times per century (MacDonald and Tingstad 2007). Strong decreases in winter precipitation characterized the major droughts in the Uinta Mountains: the 1930s Dust Bowl event, the 1976–1977 event, and the 1987–1989 event. These later two droughts were related to decreases in eastern Pacific Ocean sea surface temperatures (MacDonald and Tingstad 2007). Droughts in the Wasatch Range occurred during the 1400s and 1500s; even though droughts were fewer in the 1700s and 1800s, they had longer duration (Bekker et al. 2014). The tree-ring studies consistently emphasize the importance of local conditions in understanding climate-vegetation relationships (Louderback et al. 2015).

The Plateaus subregion encompasses three ecological provinces of the Colorado Plateau. The Uinta Basin is a gently rolling plateau that the Green River and its tributaries have eroded into many spectacular canyons (Leydsman McGinty and McGinty 2009). In the High Plateaus province, north-south trending faults and valleys separate individual plateaus, such as the Awapa, Aquarius, and Paunsaugunt Plateaus (Leydsman McGinty and McGinty 2009). The Uinta Basin and the High Plateaus are cooler than the Canyonlands province. The Colorado River has eroded the Canyonlands into many deep, sheer-walled canyons, plateaus, mesas, buttes, and badlands. In addition, the La Sal and Abajo Mountains rise above the Colorado Plateau in southeastern Utah. Precipitation in this subregion ranges from 14 to 35 inches, with the highest precipitation in the High Plateaus province.

Minimum temperatures in the Plateaus subregion showed an upward trend over the entire historical period (fig. 3.4). Mean maximum temperature in the subregion ranged between 59 and 65 °F over the last 50 years, showing a slight upward trend after 1980 (fig. 3.5). No trend is seen in precipitation over the last 50 years, although higher precipitation occurred in the 1980s (fig. 3.6). Tree-ring reconstruction of precipitation from 1200–2001 in the Uinta Basin indicates significant precipitation variability at interannual to decadal scales, with more severe dry events prior to 1900 than after (Gray et al. 2004).

The Great Basin and Semi Desert subregion has great climatic diversity (Svejcar et al. 2016). The surrounding

mountains, as well as the mountains within, strongly influence the climate of this subregion. A series of north-south mountain ranges within the subregion are interspersed among low-elevation basins, resulting in wide local variations in temperature and precipitation (fig. 3.1). Elevation ranges from less than 1,500 feet to more than 10,000 feet. Extreme temperatures can range from -15 to 120 °F (WRCC 2016b,c).

The Great Basin and Semi Desert subregion is the warmest of the subregions. For the larger Southwest region in the United States, Kunkel et al. (2013) reported an increase in mean annual temperature, with greater warming for minimum temperature than maximum temperature, over the last 115 years. That pattern for minimum and maximum monthly temperature was also seen in this subregion; mean minimum temperatures ranged from 30 to 38 °F (fig. 3.4) and mean maximum temperatures from 60 to 66 °F (fig. 3.5).

The primary source of moisture for the Great Basin and Semi Desert subregion is the Pacific Ocean, with seasonal monsoonal influences from the Gulf of Mexico. Much of the moisture from the Pacific Ocean is lost through orographic precipitation as the moisture flows upward and then over the bordering Sierra Nevada and Cascade Range. Mean annual precipitation varied from less than 8 inches to more than 15 inches over the last 50 years (fig. 3.6), with wet and dry years evident. Annual snowfall ranges from less than 1 inch to 124 inches (WRCC 2016b,c).

The Intermountain Semi Desert subregion includes two distinct areas: the large valley encompassing the Snake River plains in Idaho, and the southwestern area in Wyoming (Chapter 2). The long, large valley in Idaho gradually rises in elevation from 3,300 feet in the Magic Valley, through the Lower Snake River Plain, to the eastern end of the Upper Snake River Plain at 5,600 feet (Andretta and Geerts 2010). Swan Falls, on the Snake River in Idaho, has a mean annual temperature of 55 °F (WRCC 2016a). The Central Mountains, where the Boise, Payette, Salmon-Challis, and Sawtooth National Forests are located, bound this valley to the north. To the south and east of this subregion lies the Southern Highlands at elevations from 6,500 to 8,200 feet, Eastern Highlands, and Upper Snake Highlands of Idaho. As with the Great Basin and Semi Desert subregion, topography plays a role in the climate; however, with fewer barriers to the west, this area of the subregion receives greater moisture (fig. 3.2).

The Wyoming portion of this subregion encompasses the Green River Basin, a valley bounded on the northern edges by the Gros Ventre Mountains to the west and the Wind River Mountains to the east. The nearly 4,000-square-mile area is high, dry sagebrush-steppe, basically a desert where winter brings wind and cold, and little snow (Ostlind 2011). Temperatures are cooler in the Wyoming section than in the Idaho valley (fig. 3.1); mean annual temperature for Big Piney in the northern part of Green River Basin is 36 °F, making for a short growing season. Winter extremes can reach -50 °F as cold air settles in the Green River Basin.

Mean minimum temperatures for this subregion showed a slight warming trend over the 1949–2010 period (fig. 3.4). Mean maximum temperatures during this time ranged from about 55 to 60 °F, with little trend (fig. 3.5). Hoekema and Sridhar (2011) reported regional warming in the Snake River Basin, particularly in spring, over the last 35 years. This subregion and the Great Basin and Semi Desert are the driest subregions. Annual precipitation in the Intermountain Semi Desert subregion is generally between 8 and 12 inches (fig. 3.6). Precipitation showed no trend over the historical period.

Projected Climate

Changes in annual and seasonal temperature and precipitation are summarized by subregion in table 3.3. Trends in annual and seasonal temperature and precipitation projections are shown as figures for each subregion (see box 3.1 for more information on the format of these subregional figures): Middle Rockies (figs. 3.7–3.9), Southern Greater Yellowstone (figs. 3.10–3.12), Uintas and Wasatch Front (figs. 3.13–3.15), Plateaus (figs. 3.16–3.18), Great Basin and Semi Desert (figs. 3.19–3.21), and Intermountain Semi Desert (figs. 3.22–3.24).

In the future, all subregions in the IAP are projected to see increases in annual and seasonal minimum and maximum temperatures, with greater changes under the RCP 8.5 scenario by 2100 than under the RCP 4.5 scenario (table 3.3). Within each subregion, temperatures vary greatly across landscapes, the result of topography and aspect. In the Middle Rockies, the projected increase in minimum temperature under the RCP 8.5 scenario will bring the subregional median temperature above freezing (fig. 3.7a); this projected increase suggests that for some areas in the IAP region, a biologically meaningful threshold could be crossed. Similar patterns for an increase in median minimum temperature above freezing occur in the Southern Greater Yellowstone subregion under the RCP 8.5 scenario only (fig. 3.10), and Uintas and Wasatch Front (fig. 3.13), the Plateaus (fig. 3.16), and the Intermountain Semi Desert subregions (fig. 3.22) under both scenarios. For most subregions, annual precipitation projections are highly variable with no discernible trend over time or between the two scenarios. However, slight increases in annual precipitation are projected under the RCP 8.5 scenario for the Middle Rockies, the Southern Greater Yellowstone, and the Intermountain Semi Desert subregions (figs. 3.7, 3.10, 3.22), which lie in the northern part of the IAP region.

Seasonal temperatures across the subregions are projected to increase and may cross biologically meaningful thresholds in particular seasons (table 3.3). Minimum seasonal temperatures are projected to rise in all seasons across all subregions under both the RCP 4.5 and RCP 8.5 scenarios (figs. 3.8, 3.11, 3.14, 3.17, 3.20, 3.23). Maximum seasonal temperatures are also projected to rise in all seasons under both RCP 4.5 and RCP 8.5 across all

Table 3.3—Summary of projected changes in annual and seasonal temperature, and precipitation for each Intermountain Adaptation Partnership subregion.

Subregion	Temperature	Precipitation	Seasonality
Middle Rockies	By 2100, median maximum temperature is projected to rise about 6 °F under RCP 4.5 and about 10 °F under RCP 8.5; the two ensemble projections begin to diverge after 2045. Median minimum temperature is projected to rise about 5 °F under RCP 4.5 and about 10 °F under RCP 8.5.	Annual precipitation projections are highly variable with no discernible trend under RCP 4.5 and a slight increasing trend under RCP 8.5.	Maximum temperatures are projected to increase by 5 to 10 °F across the seasons. The greatest departure from historical seasonal minimum temperatures occurs in summer. Over the historical period, summer minimum temperatures ranged around the mid- to upper 30s. By 2040, the projected median is 40 °F and rises to nearly 50 °F under the RCP 8.5 scenario by 2100 (fig. 3.9). By 2100, the median projections for spring, summer, and fall under the RCP 8.5 scenario are outside of historical ranges.
Southern Greater Yellowstone	By 2100, median maximum temperature is projected to rise about 5 °F under RCP 4.5 and about 11 °F under RCP 8.5; projections for the two RCPs begin to diverge around 2040. By 2100, median minimum temperature is projected to increase about 6 °F under RCP 4.5 and about 12 °F under RCP 8.5. Median minimum temperatures are projected to remain below freezing under RCP 4.5. However, minimum temperatures are likely to rise to just under freezing by 2100 under RCP 8.5.	Annual precipitation projections are highly variable with no discernible trend under RCP 4.5 and a slight increasing trend under RCP 8.5.	Maximum temperature is projected to increase in all seasons, with winter temperatures rising about 3 °F and all other seasons rising about 5 °F under RCP 4.5 by the end of the 21st century. Under the warmest scenario, seasonal temperatures increase about 10 °F in winter, spring, and fall, but by more than 12 °F in summer by the end of the 21st century. Median minimum temperatures for all seasons by the 2080s are projected to be outside of historical ranges in the warmest scenario. Median minimum spring and fall temperatures are projected to increase, such that some projections rise above freezing by the end of the 21st century under the RCP 8.5 scenario.
Uintas and Wasatch Front	By 2100, median maximum temperature is projected to rise about 5 °F under RCP 4.5 and about 11 °F under RCP 8.5. Projections for the two scenarios begin to diverge around 2040. By 2100, median minimum temperature is projected to rise about 5 °F under RCP 4.5 and about 12 °F under RCP 8.5. Median minimum temperatures are projected to rise above freezing in both scenarios (by 2050 under RCP 8.5 and by 2075 under RCP 4.5).	Annual precipitation projections are highly variable with no discernible trends.	Maximum temperatures are projected to increase in all seasons, with winter temperatures rising about 4 °F and all other seasons rising about 6 °F under RCP 4.5 by the end of the 21st century. Maximum median temperatures by the 2080s are outside of the historical range of values for all seasons in the warmest scenario. Median minimum spring and fall temperatures are projected to rise above freezing by the 2080s in both scenarios. The greatest departure from historical seasonal minimum temperatures occurs in summer.

Table 3.3—*Continued.*

Subregion	Temperature	Precipitation	Seasonality
Plateaus	By 2100, median maximum temperature is projected to rise about 5 °F under RCP 4.5 and about 10 °F under RCP 8.5. By 2100, median minimum temperature is projected to rise about 5 °F under RCP 4.5 and about 12 °F under RCP 8.5; the two ensemble projections begin to diverge after 2050. By 2050, median minimum temperature is projected to rise above freezing in both scenarios.	Precipitation projections are highly variable with no discernible trend over time or between the two scenarios.	Maximum temperature is projected to increase in all seasons by about 5 °F under RCP 4.5 and by about 10 °F under RCP 8.5 by the end of the 21st century. The greatest departure from historical temperatures by 2100 is projected to occur in summer, where median temperatures rise above 95 °F under the RCP 8.5 scenario. Projected median maximum temperatures for winter, spring and autumn are also outside of historical ranges by end of the 21st century. The greatest departure from historical seasonal minimum temperatures is projected to occur in summer. Minimum temperatures in summer are projected to rise about 6 °F by 2100 under RCP 4.5 and over 10 °F under RCP 8.5, with the variation well outside of the historical ranges.
Great Basin and Semi Desert	By 2100, median maximum temperature is projected to rise about 5 °F under the RCP 4.5 scenario and about 10 °F under the RCP 8.5 scenario. Historically median minimum temperature has ranged around freezing; in the near future, it is projected to rise above freezing, and by end of century, is projected to increase by 6 to 10 °F. The two ensemble projections begin to diverge after 2050.	Precipitation projections are highly variable with no discernible trend over time or between the two scenarios.	Maximum temperature is projected to increase by 5 °F in winter and spring under the RCP 4.5 scenario and by 10 °F under the RCP 8.5 scenario by the end of the 21st century. Summer and fall temperatures are projected to increase by 12 °F by the end of the 21st century under the RCP 8.5 scenario. Median minimum spring temperatures rise above freezing for both scenarios by 2100 and for the RCP 8.5 scenario, temperatures approach 40 °F. Median minimum and maximum projections for the both scenarios in all seasons by 2100 are outside of the historical range.
Intermountain Semi Desert	By the mid-21st century, median maximum temperature is projected to rise about 5 °F under RCP 4.5 and about 10 °F under the RCP 8.5. Median minimum temperature is projected to rise about 5 °F under RCP 4.5 and about 11 °F under RCP 8.5. By 2100, the projected changes for minimum temperature rise above freezing.	The highly variable precipitation projections show no discernible trend over time under the RCP 4.5 and suggest a slight increase under the RCP 8.5 scenario.	Maximum temperature increases in all seasons to the end of the 21st century in both climate scenarios. The greatest departure from historical temperatures by 2100 occurs in summer under RCP 8.5, when projected mean temperatures approach 95 °F, nearly 15 °F above historical temperature. Median minimum seasonal temperatures are projected to rise in all seasons. Median minimum and maximum values for all seasons under the RCP 8.5 scenario are outside of the historical ranges by 2100.

subregions (figs. 3.9, 3.12, 3.15, 3.18, 3.21, 3.24). Thus, the frequency of days with extreme heat in summer is likely to increase. Winter precipitation is an important reservoir for mountain and surrounding lower elevation communities. Few subregional or site-specific projections for snowpack have been made for the IAP region. For the Wasatch Range and Uinta Mountains, Klos et al. (2014) used only RCP 8.5 and 20 climate models (similar to the projections used in this chapter), and they report that these ranges will have fall-through-spring temperatures such that wintertime precipitation will begin to shift from strongly snow dominated to a mixed rain-snow regime by the mid-21st century. The shoulder months of November and March will shift to rain, with December through February retaining a snow-dominated system longer. For additional discussion of snowpack and climate change, see Chapter 4. Also see Rice et al. (2017) for a review of climate change literature for the Uintas and Wasatch Front area.

Box 3.1—Template for Projected Climate Change Figures

For each of the six subregions, a common template for figures has been used to describe the projected climate. The first figure for each subregion (figs. 3.7, 3.10, 3.13, 3.16, 3.19, 3.22) shows the historical simulations and projections for annual daily minimum and maximum mean temperatures (°F), and total annual precipitation (inches) under the RCP 4.5 and RCP 8.5 scenarios based on the CMIP5 BCSD data. These figures show the historical climate simulations by the CMIP5 models, which reflect the pre-2010 climate forcings. These historical simulations are bias corrected and downscaled in the same manner as model future projections. In these figures, we overlaid the gridded historical observation data (blue line) from Maurer et al. (2002). In most regions, the historical simulated minimum and maximum temperatures and annual precipitation are less variable than the historical observed gridded climate. The future projections are shown in colors: red for RCP 8.5 and yellow for RCP 4.5. The ensemble median from all 20 models for each scenario is shown in the heavy line; the 5th- and 95th-percent quantiles for all models are shown by the shaded area. Typically, climate projections under the higher emissions scenario (RCP 8.5) will indicate a higher temperature by 2100 than climate projections under the RCP 4.5 scenario.

The second figure for each subregion (figs. 3.8, 3.11, 3.14, 3.17, 3.20, 3.23) shows the seasonal daily minimum temperature (°F) for the historical and projected period 1950–2100. Winter is defined as the months of December, January, and February. Spring is defined as March, April, and May; summer as June, July, and August; and fall as September, October, and November. We use box plots here, where each box is an aggregation of 20 years of modeled historical or projected seasonal data. For example, the box labeled "1960" represents the seasonal average of 1950 to 1969. We used the 20-year period here to explore the temporal changes over this century. The modeled historical boxes are gray, and boxes for projections use the same colors as in other figures: yellow for RCP 4.5 and red for RCP 8.5. The central line in each box is the median, indicating the same number of modeled historical or projections above and below this line. The hinges or edges of the boxes are the first and third quartiles. Whiskers extend past the first and third quartile by 1.5 times the interquartile range.

The third figure for each subregion (figs. 3.9, 3.12 3.15, 3.18, 3.21, 3.24) shows the seasonal daily maximum temperature (°F) for the historical and projected period. These figures are set up in the same way as the second figures. There is large variability and no discernible trend in the seasonal precipitation projections, and hence, less confidence overall in the finer scale precipitation projections; these figures are not shown here.

Summary and Conclusions

For this overview, the projected climate was derived from climate models in the CMIP5 database, which was used in the most recent IPCC reports. We quantified changes in temperature and precipitation by the 2040 period (2030–2059) and 2080 period (2070–2099). Over the next 100 years, annual minimum and maximum temperatures are projected to rise by as much as 10 °F in the IAP region. Projections for annual precipitation are highly variable. For most subregions in the IAP area, precipitation remains variable; slight increases in total annual precipitation are projected for the Middle Rockies, Southern Greater Yellowstone, and Intermountain Semi Desert subregions under the RCP 8.5 scenario. As with annual temperature, winter, spring, summer, and fall temperatures are projected to increase, with summer temperatures showing the greatest increases in several subregions. For many subregions, the seasonal temperatures by end of century are outside of the historical observed ranges. Many of the resource chapters draw from existing scientific literature that used climate projections from the 2007 IPCC reports (the CMIP3 database). In mid-century (2040–2060), CMIP3 and CMIP5 temperature projections are similar. However, CMIP5 precipitation projections appear to be slightly wetter than those in CMIP3.

Acknowledgments

We acknowledge the World Climate Research Programme's Working Group on Coupled Modelling, which is responsible for CMIP, and we thank the climate modeling groups (listed in Appendix 2) for producing and making available their model output. For CMIP, the U.S. Department of Energy's Program for Climate Model Diagnosis and Intercomparison provides coordinating support and led development of software infrastructure in partnership with the Global Organization for Earth System Science Portals. Contributions from Marian Talbert were supported by the Department of the Interior's North Central Climate Science Center. We appreciate the contributions of Jeff Morisette to the development of the chapter and the workshop presentations.

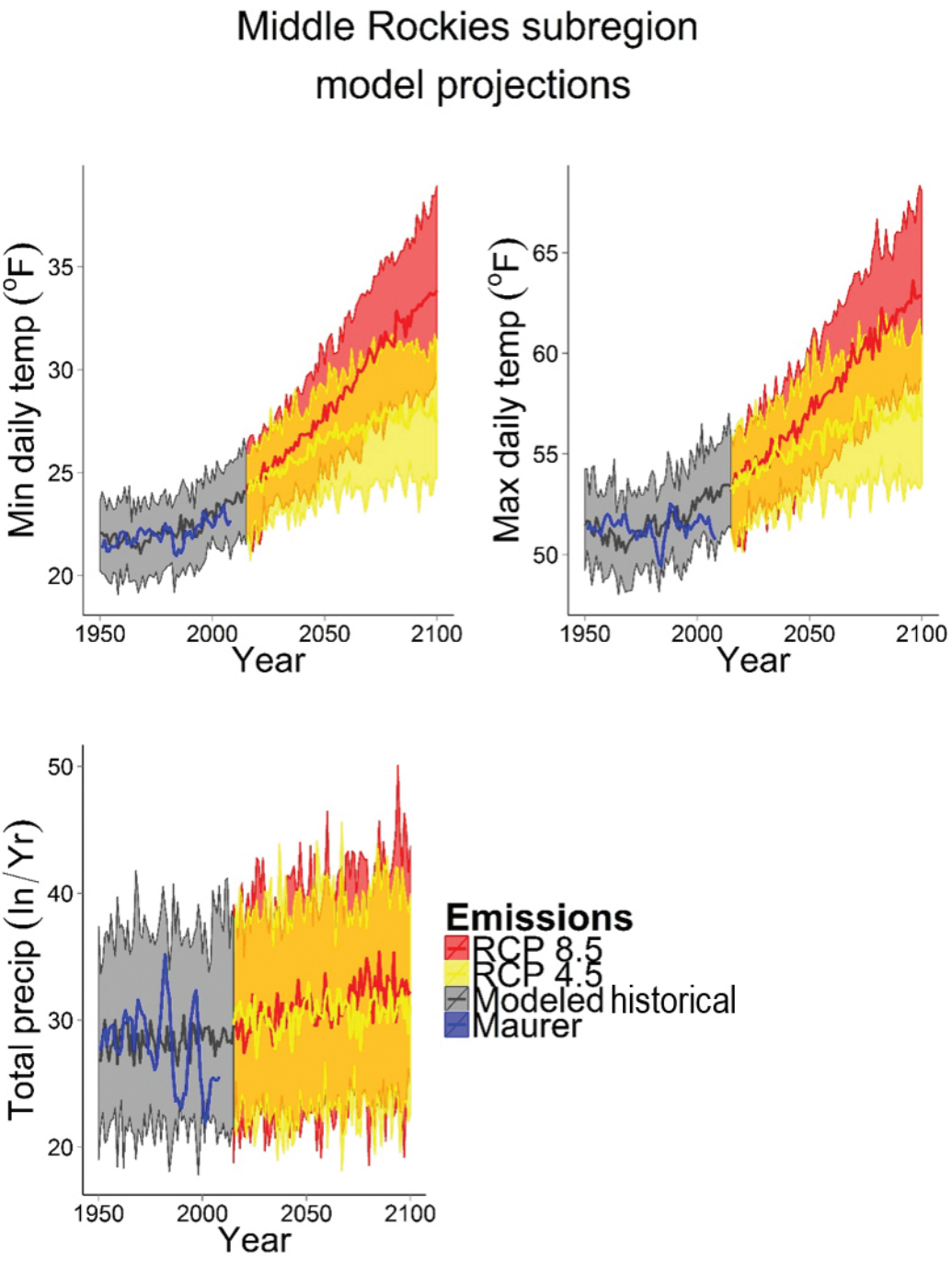


Figure 3.7—Historical modeled and projected annual mean monthly minimum temperature, annual mean monthly maximum temperature, and total annual precipitation for 1950–2100 under RCP 4.5 and RCP 8.5 emissions scenarios based on CMIP5 data for the Middle Rockies subregion of the Intermountain Adaptation Partnership region. Historical annual data from the gridded Maurer et al. (2007) dataset are also shown. See box 3.1 for further explanation.

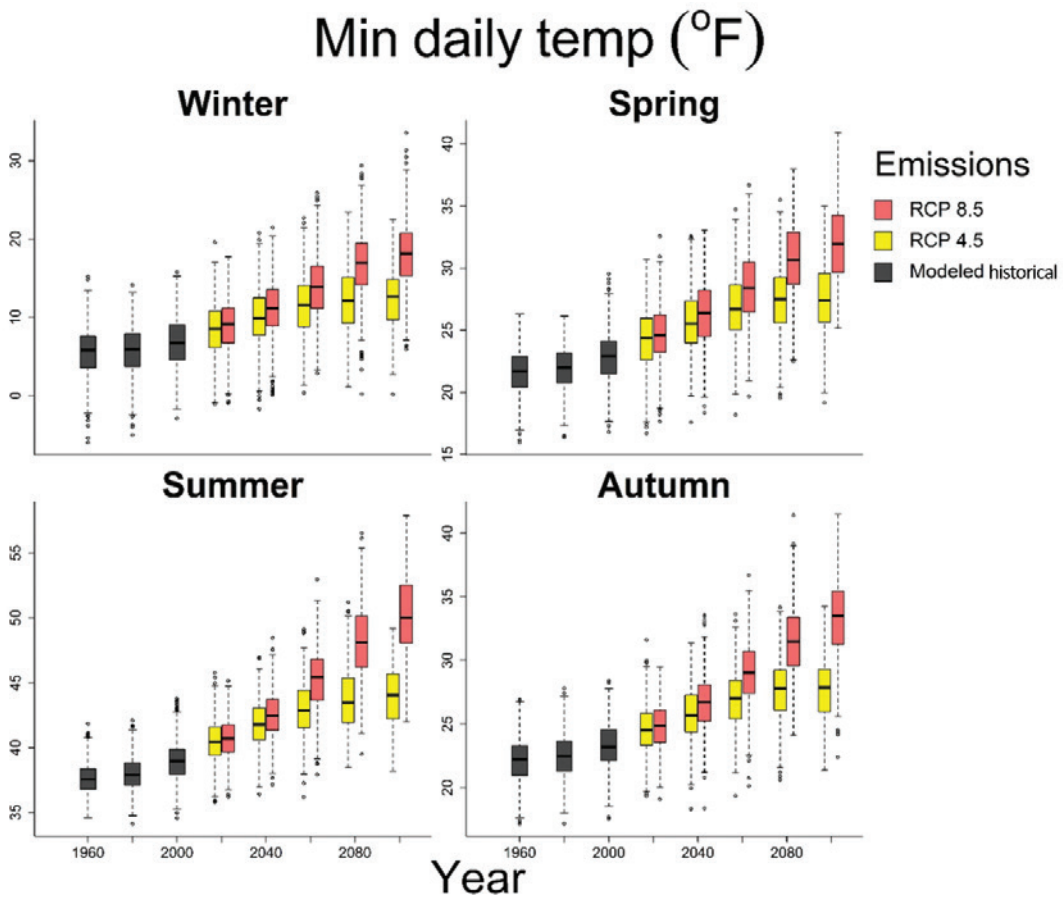


Figure 3.8—Historical modeled and projected seasonal mean monthly minimum temperature for 1950–2100 under RCP 4.5 and RCP 8.5 emissions scenarios for the Middle Rockies subregion of the Intermountain Adaptation Partnership region. See box 3.1 for further explanation.

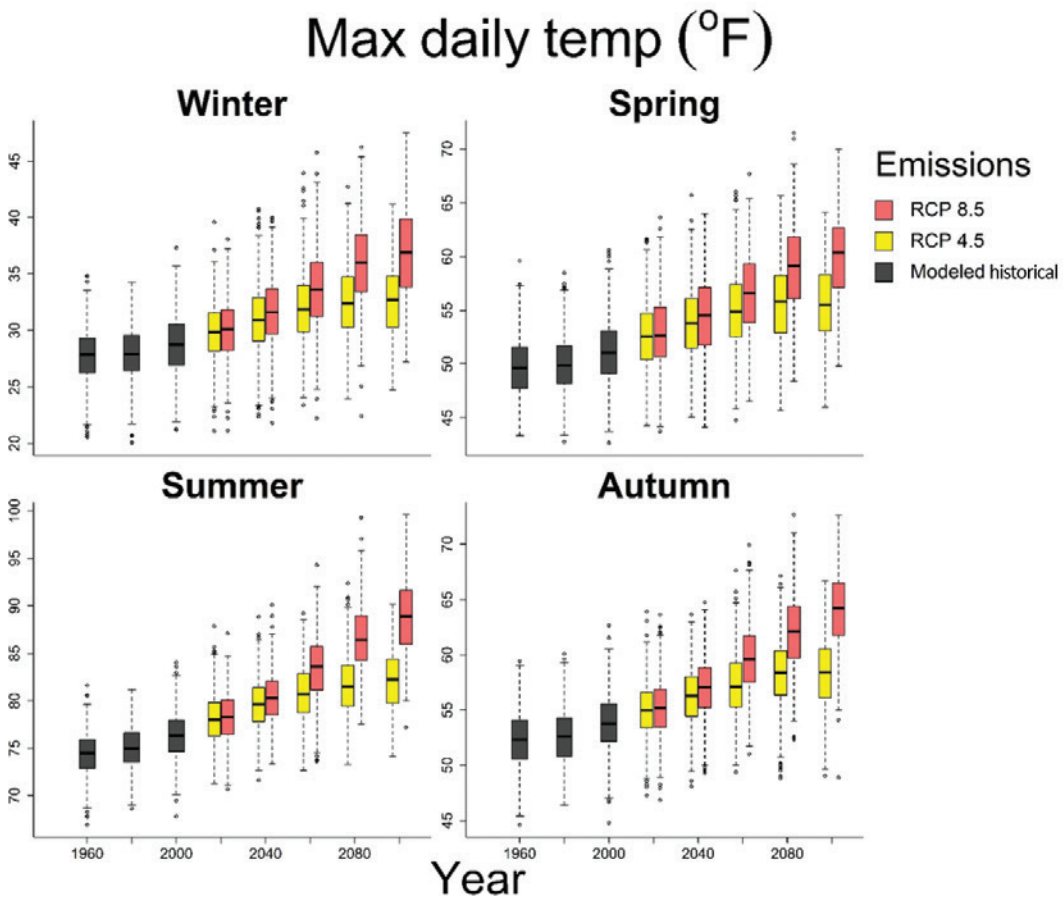


Figure 3.9—Historical modeled and projected seasonal mean monthly maximum temperature for 1950–2100 under RCP 4.5 and RCP 8.5 emissions scenarios for the Middle Rockies subregion of the Intermountain Adaptation Partnership region. See box 3.1 for further explanation.

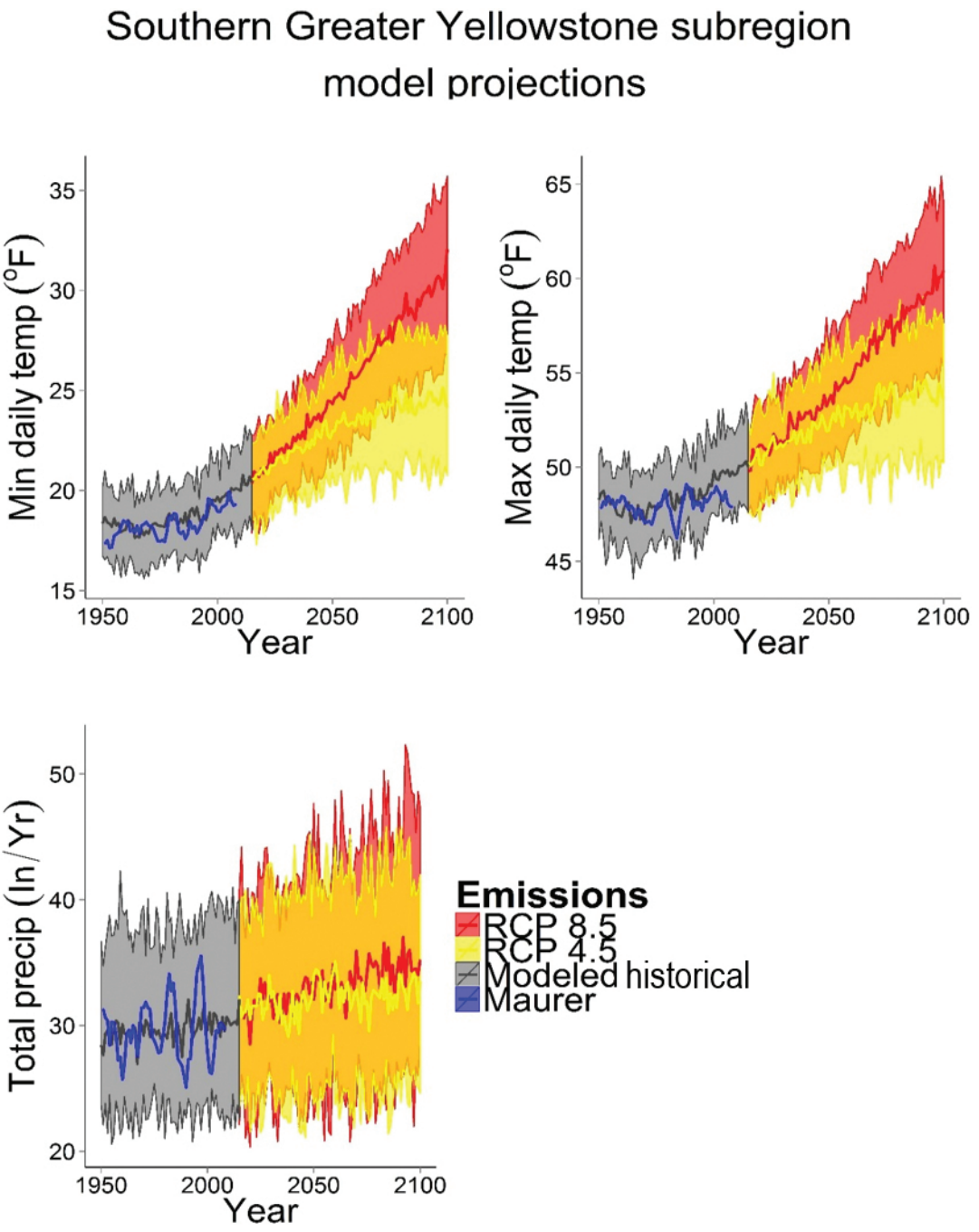


Figure 3.10—Historical modeled and projected annual mean monthly minimum temperature, annual mean monthly maximum temperature, and total annual precipitation for 1950–2100 under RCP 4.5 and RCP 8.5 emissions scenarios based on CMIP5 data for the Southern Greater Yellowstone subregion of the Intermountain Adaptation Partnership region. Historical annual data from the gridded Maurer et al. (2007) dataset are also shown. See box 3.1 for further explanation.

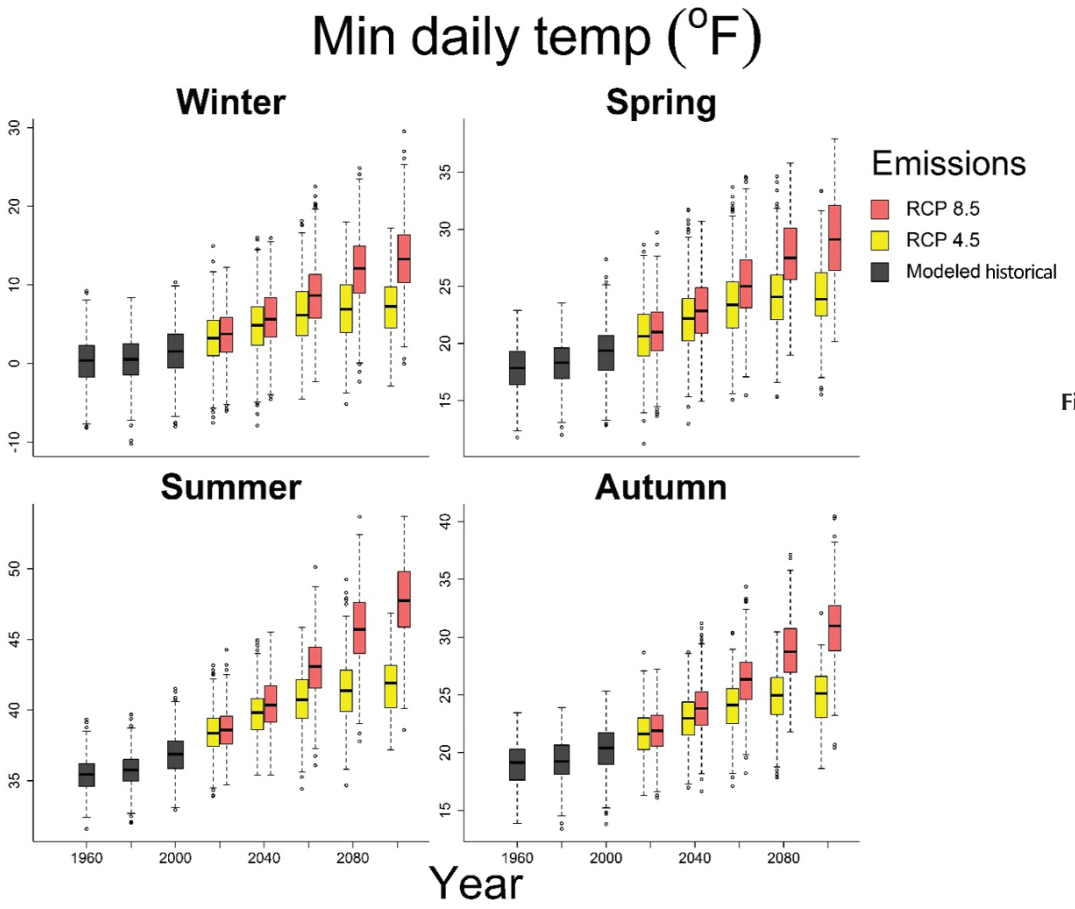


Figure 3.11— Historical modeled and projected seasonal mean monthly minimum temperature for 1950–2100 under RCP 4.5 and RCP 8.5 emissions scenarios for the Southern Greater Yellowstone subregion of the Intermountain Adaptation Partnership region. See box 3.1 for further explanation.

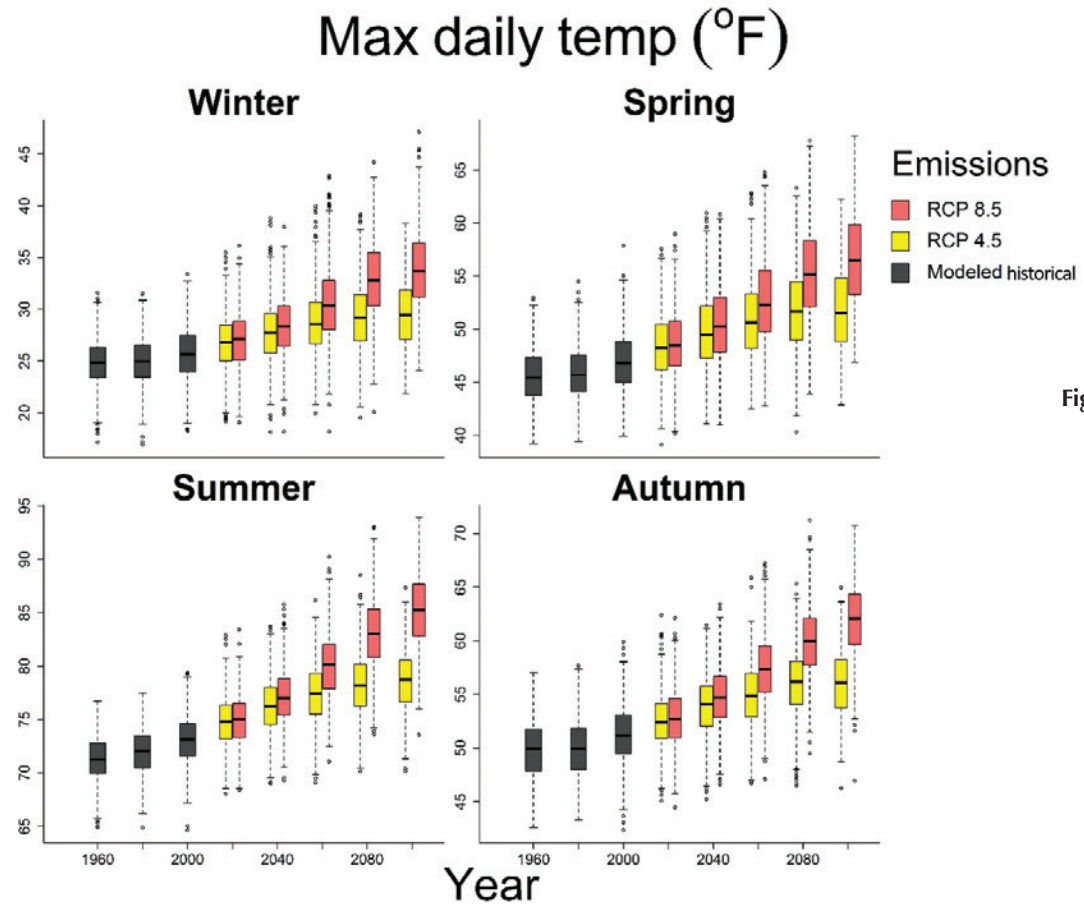


Figure 3.12—Historical modeled and projected seasonal mean monthly maximum temperature for 1950–2100 under RCP 4.5 and RCP 8.5 emissions scenarios for the Southern Greater Yellowstone subregion of the Intermountain Adaptation Partnership region. See box 3.1 for further explanation.

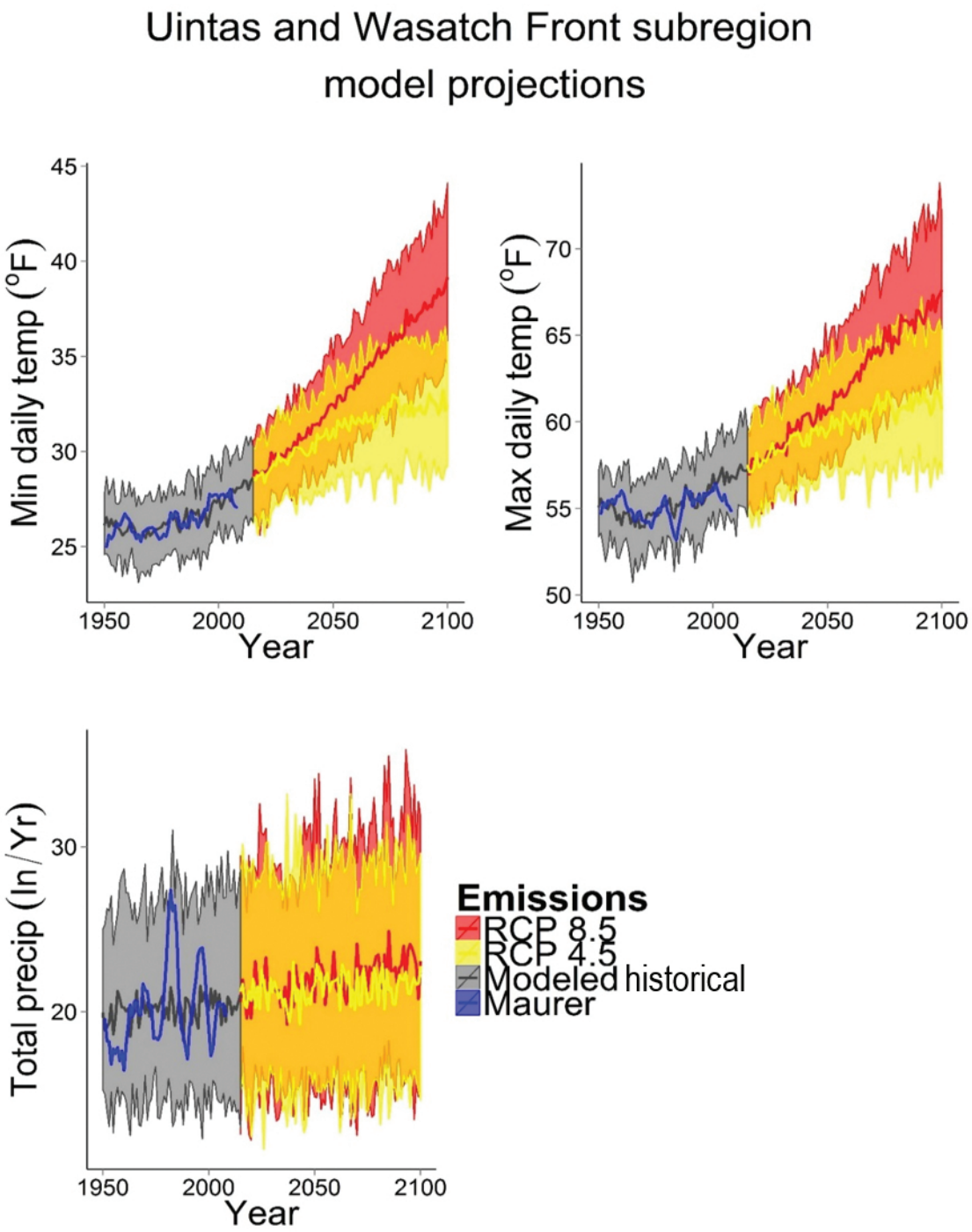


Figure 3.13—Historical modeled and projected annual mean monthly minimum temperature, annual mean monthly maximum temperature, and total annual precipitation for 1950–2100 under RCP 4.5 and RCP 8.5 emissions scenarios based on CMIP5 data for the Uintas and Wasatch Front subregion of the Intermountain Adaptation Partnership region. Historical annual data from the gridded Maurer et al. (2007) dataset are also shown. See box 3.1 for further explanation.

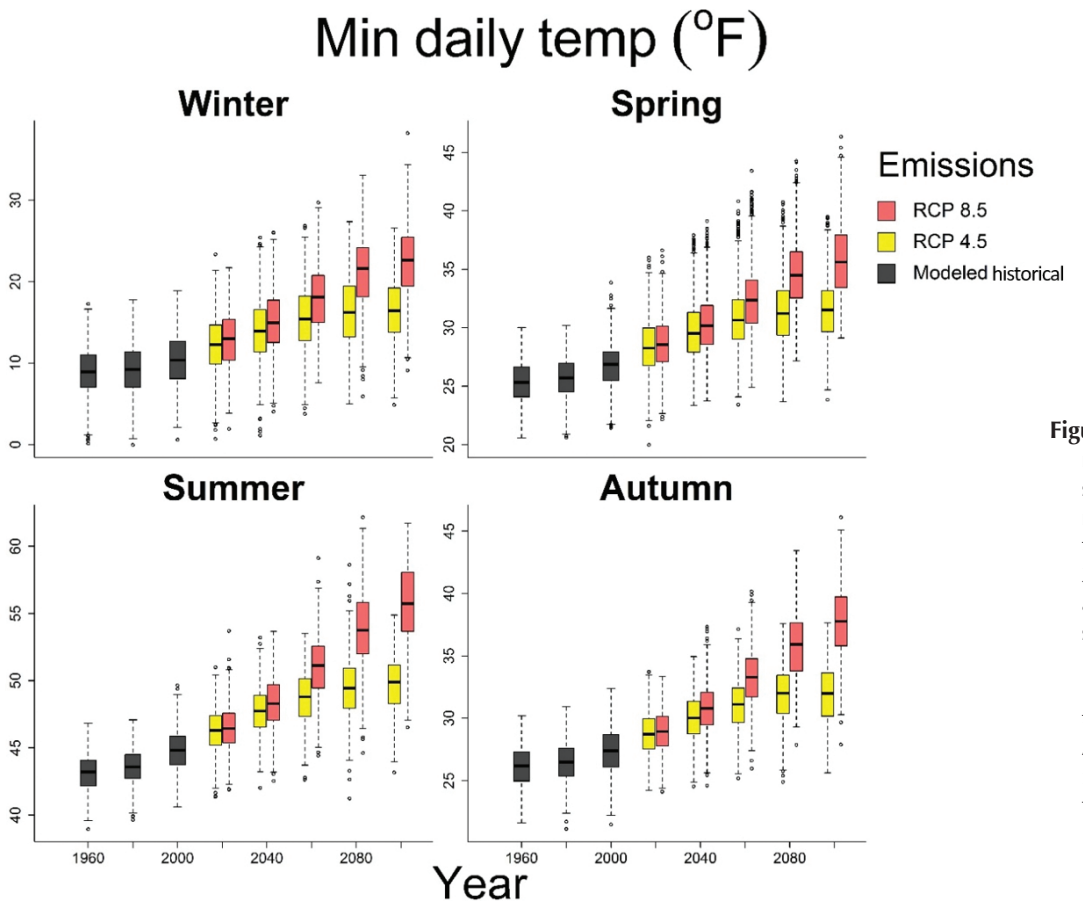


Figure 3.14— Historical modeled and projected seasonal mean monthly minimum temperature for 1950–2100 under RCP 4.5 and RCP 8.5 emissions scenarios for the Uintas and Wasatch Front subregion of the Intermountain Adaptation Partnership region. See box 3.1 for further explanation.

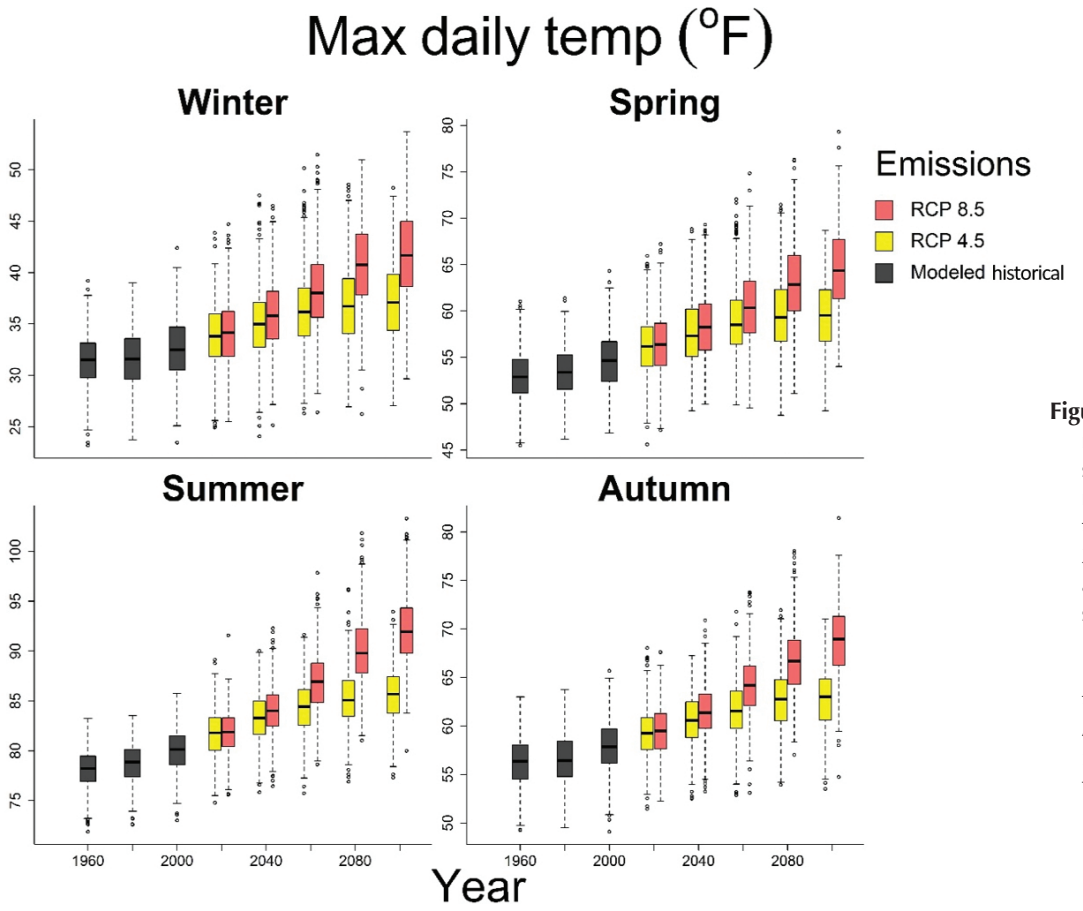


Figure 3.15— Historical modeled and projected seasonal mean monthly maximum temperature for 1950–2100 under RCP 4.5 and RCP 8.5 emissions scenarios for the Uintas and Wasatch Front subregion of the Intermountain Adaptation Partnership region. See box 3.1 for further explanation.

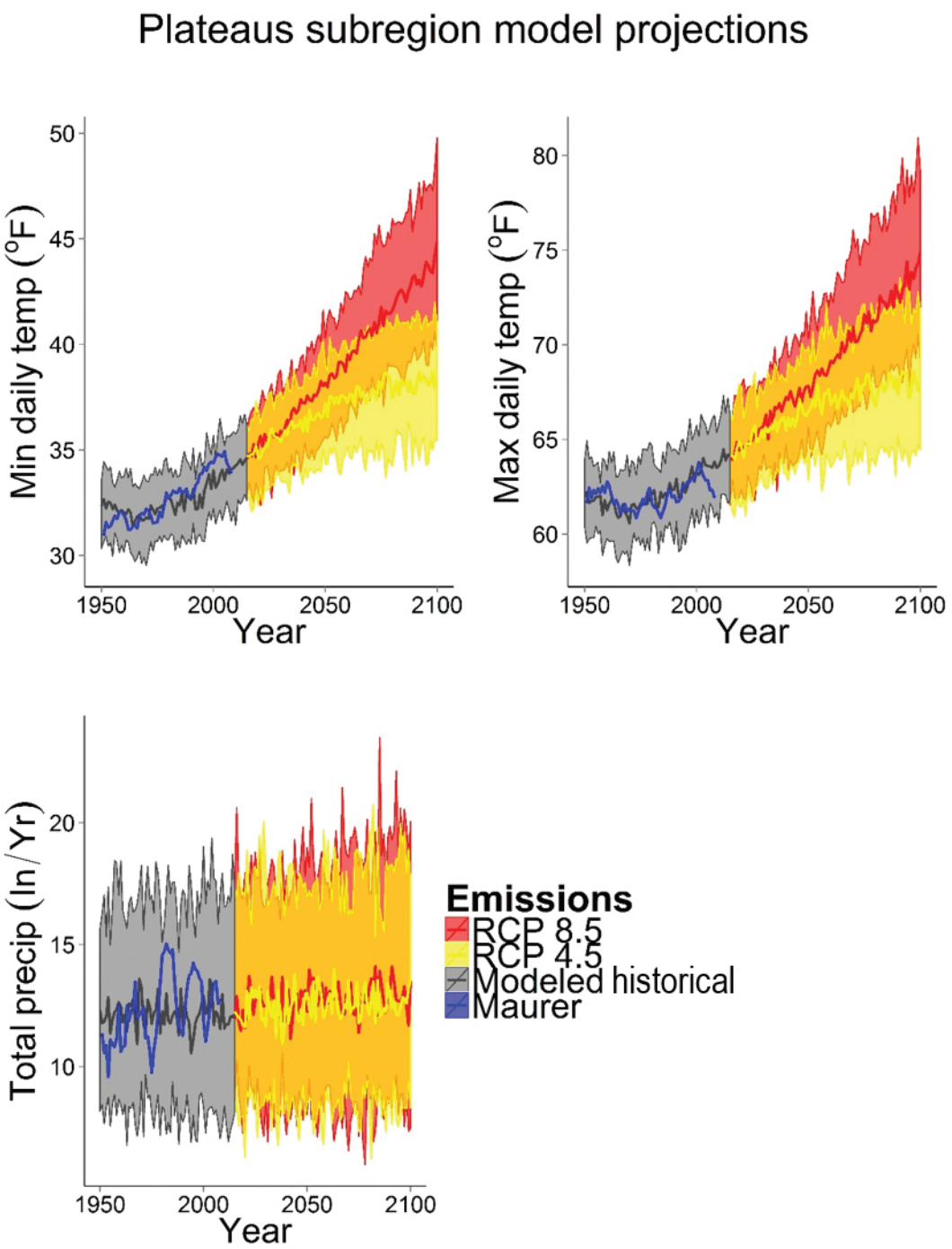


Figure 3.16—Historical modeled and projected annual mean monthly maximum temperature, annual mean monthly minimum temperature, and total annual precipitation for 1950–2100 under RCP 4.5 and RCP 8.5 emissions scenarios based on CMIP5 data for the Plateaus subregion of the Intermountain Adaptation Partnership region. Historical annual data from the gridded Maurer et al. (2007) dataset are also shown. See box 3.1 for further explanation.

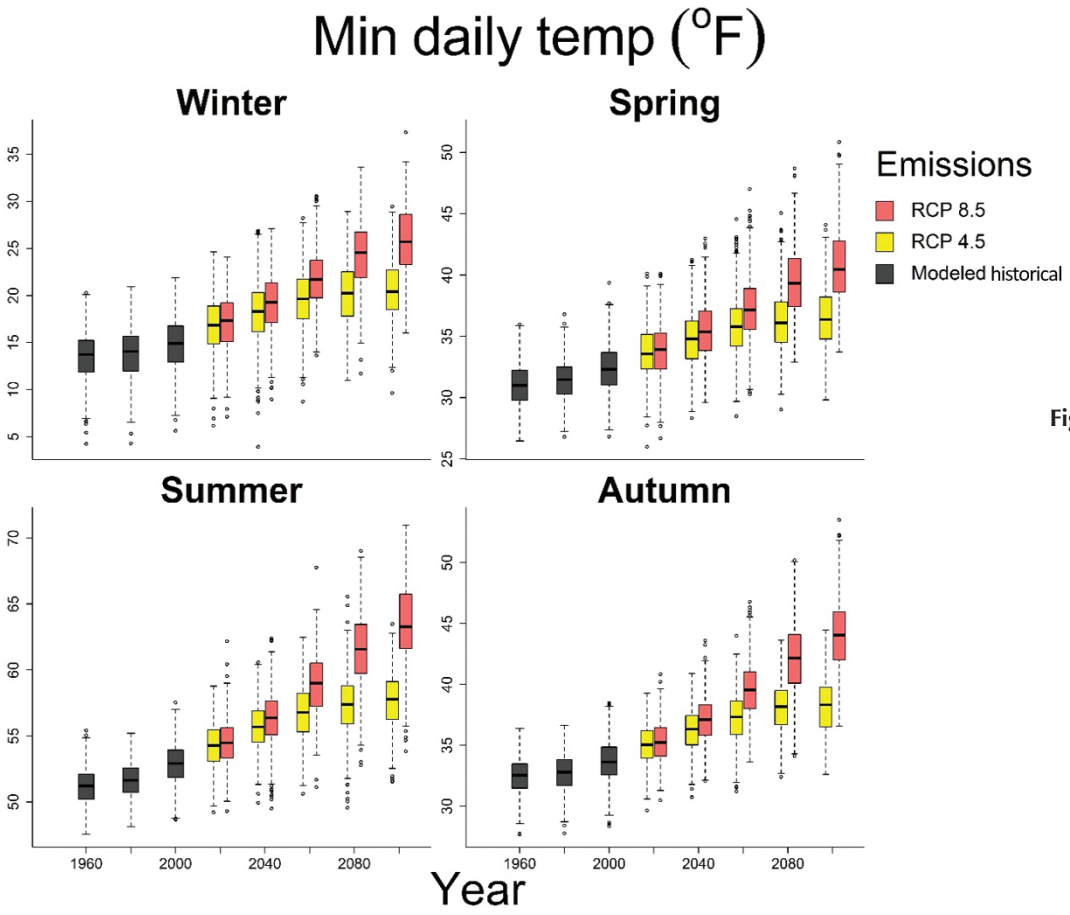


Figure 3.17— Historical modeled and projected seasonal mean monthly minimum temperature for 1950–2100 under RCP 4.5 and RCP 8.5 emissions scenarios for the Plateaus subregion of the Intermountain Adaptation Partnership region. See box 3.1 for further explanation.

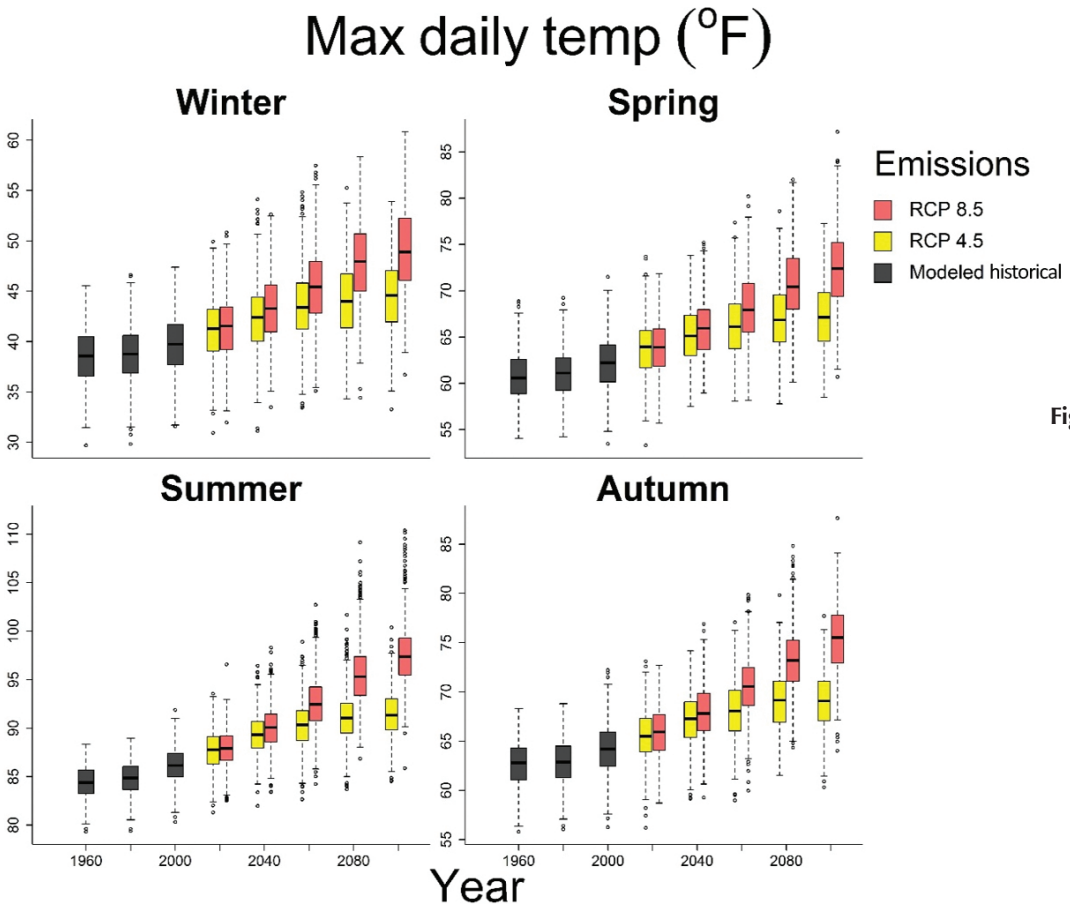


Figure 3.18— Historical modeled and projected seasonal mean monthly maximum temperature for 1950–2100 under RCP 4.5 and RCP 8.5 emissions scenarios for the Plateaus subregion of the Intermountain Adaptation Partnership region. See box 3.1 for further explanation.

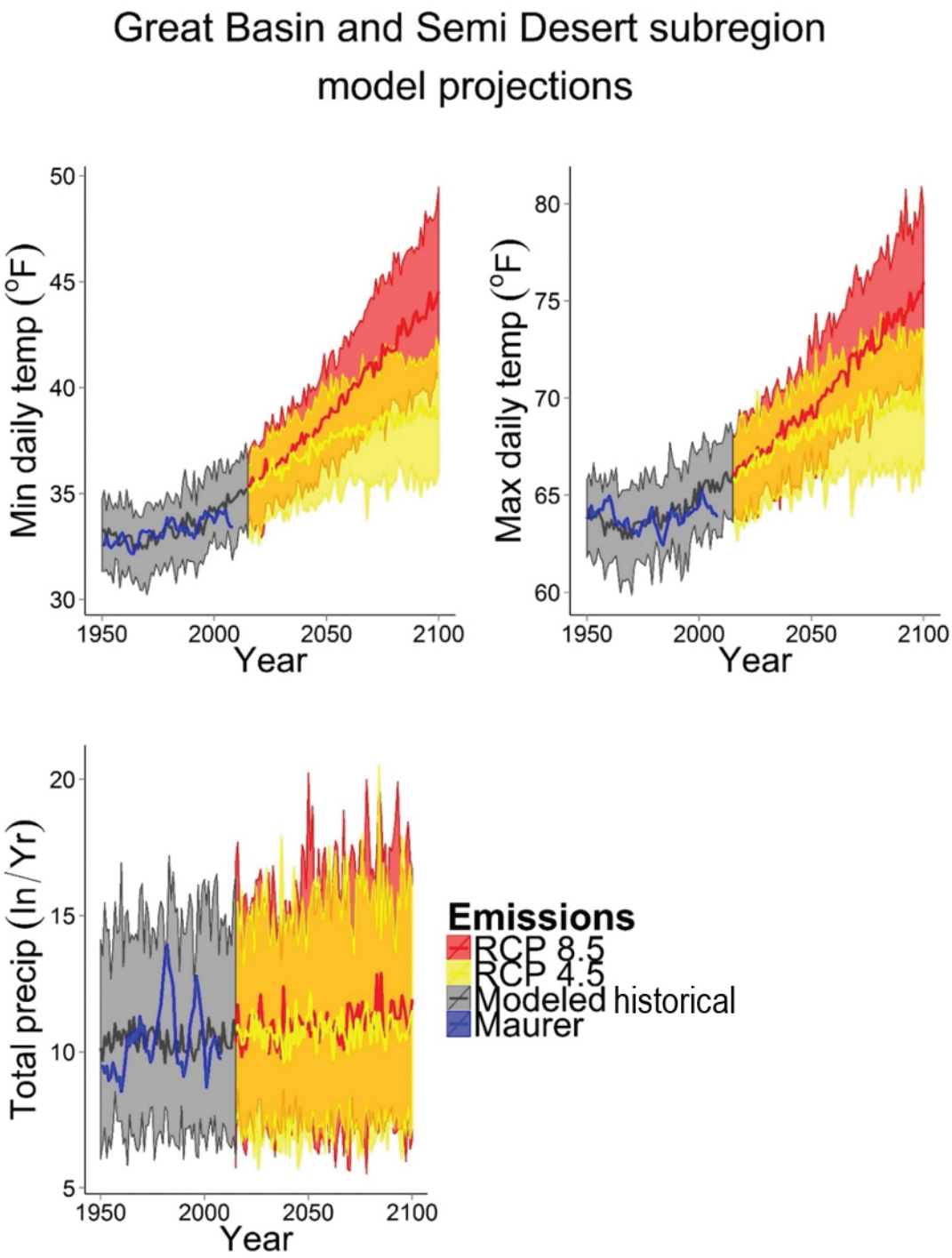


Figure 3.19—Historical modeled and projected annual mean monthly minimum temperature, annual mean monthly maximum temperature, and total annual precipitation for 1950–2100 under RCP 4.5 and RCP 8.5 emissions scenarios based on CMIP5 data for the Great Basin and Semi Desert subregion of the Intermountain Adaptation Partnership region. Historical annual data from the gridded Maurer et al. (2007) dataset are also shown. See box 3.1 for further explanation.

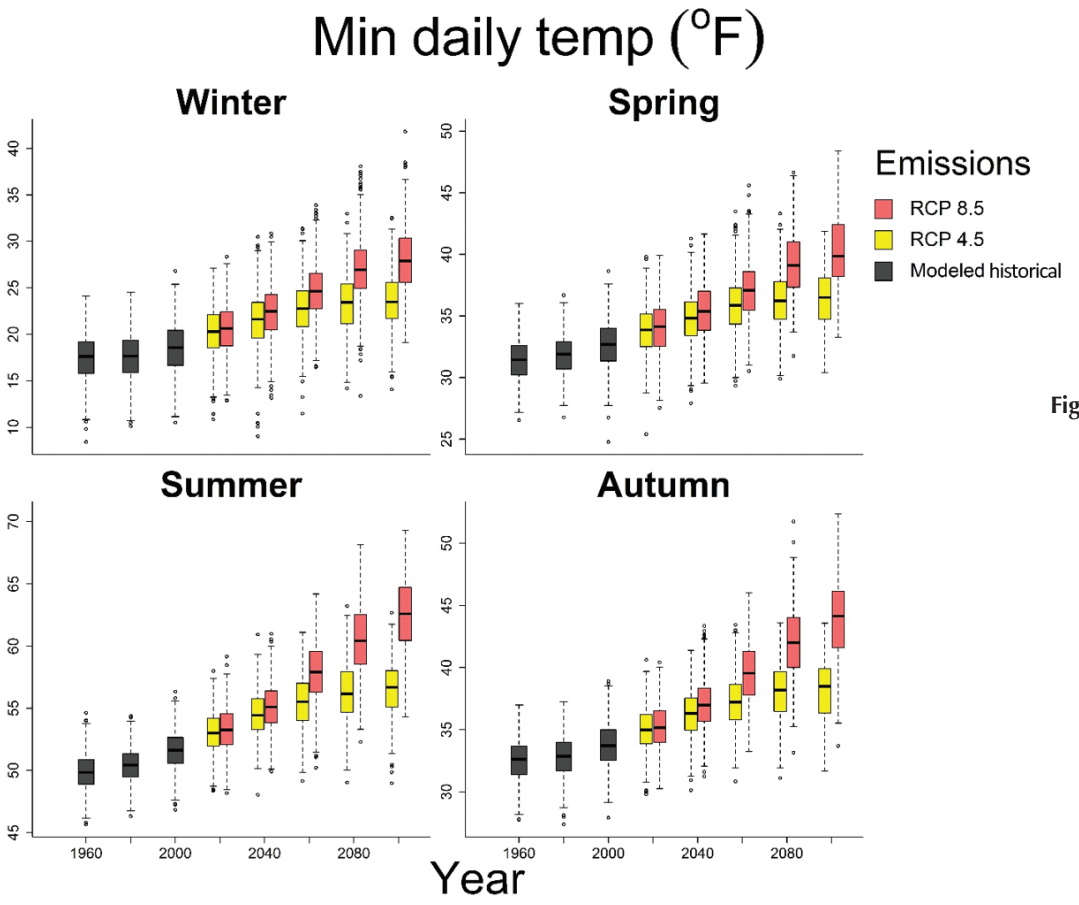


Figure 3.20—Historical modeled and projected seasonal mean monthly minimum temperature for 1950–2100 under RCP 4.5 and RCP 8.5 emissions scenarios for the Great Basin and Semi Desert region of the Intermountain Adaptation Partnership region. See box 3.1 for further explanation.

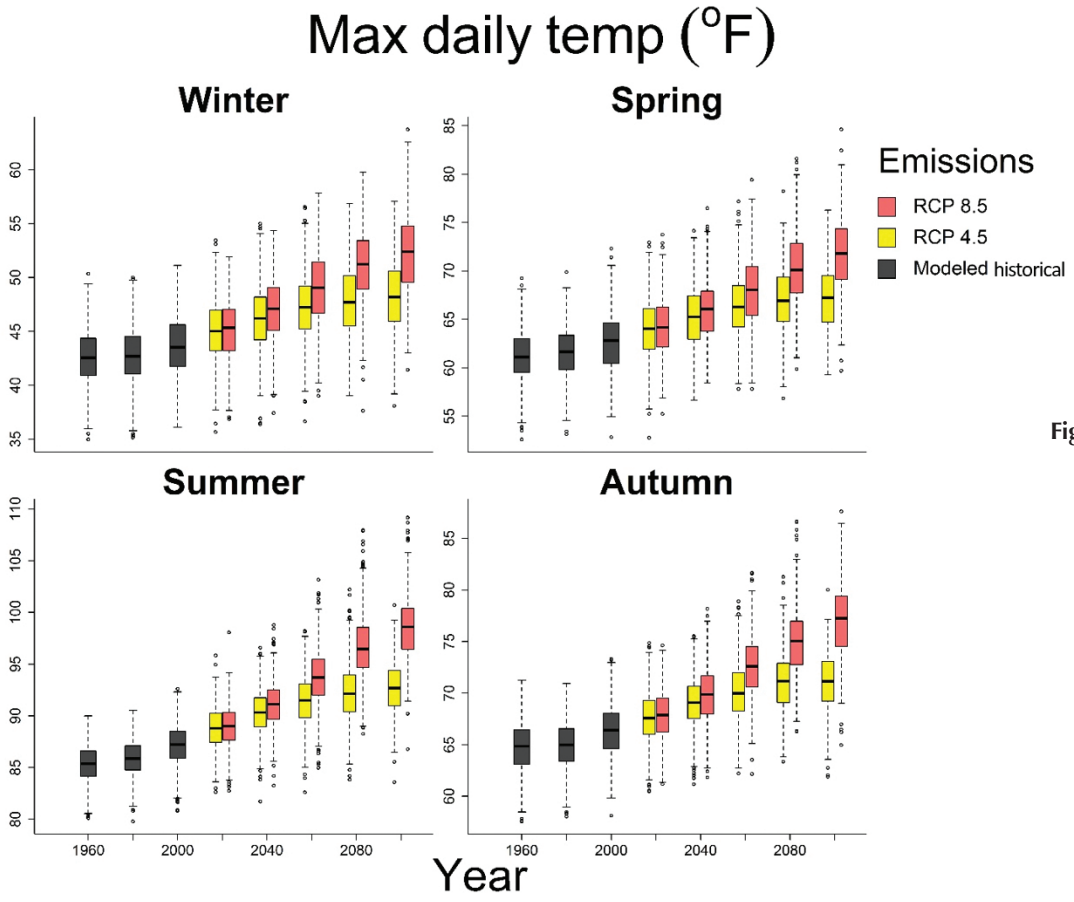


Figure 3.21—Historical modeled and projected seasonal mean monthly maximum temperature for 1950–2100 under RCP 4.5 and RCP 8.5 emissions scenarios for the Great Basin and Semi Desert subregion of the Intermountain Adaptation Partnership region. See box 3.1 for further explanation.

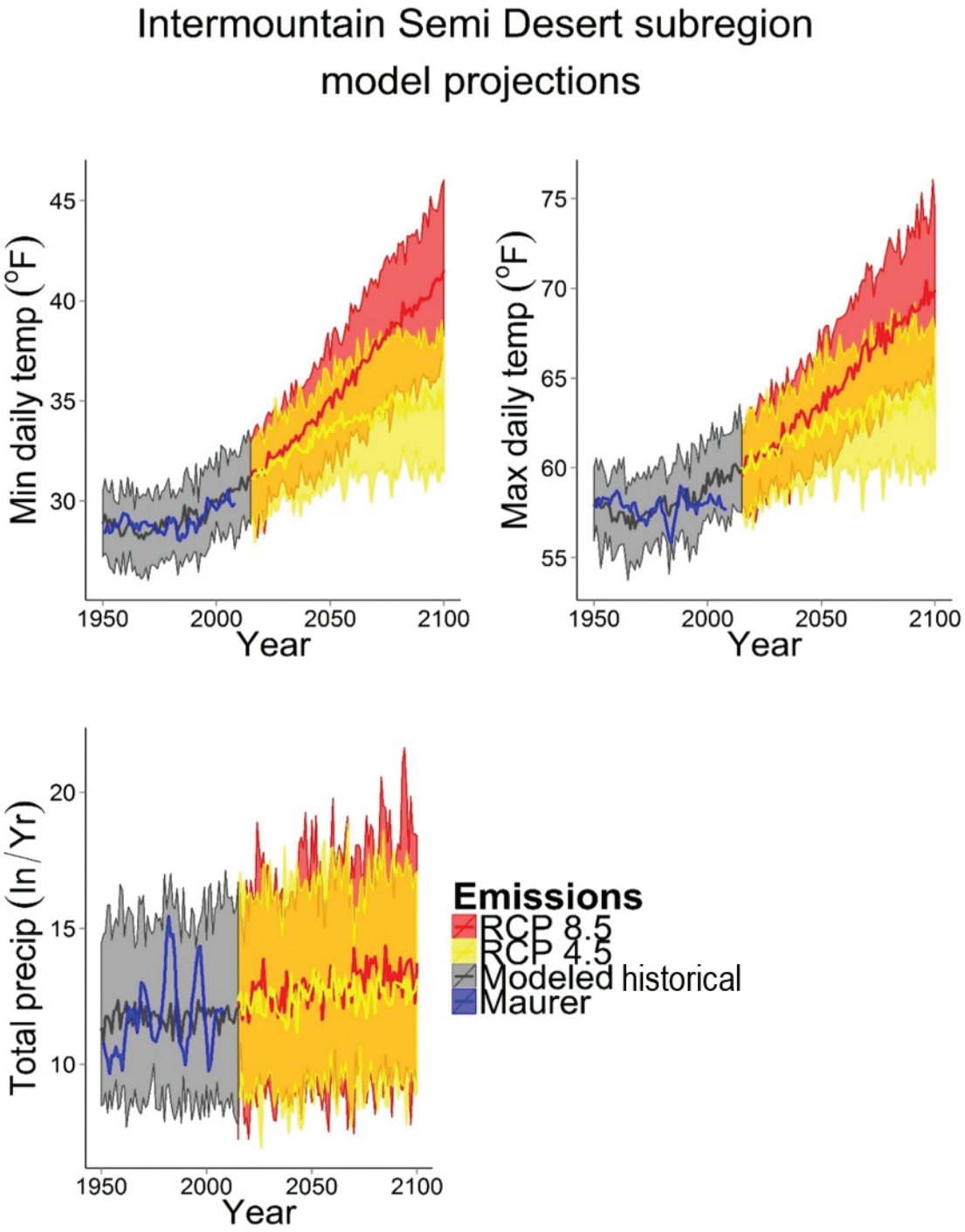


Figure 3.22—Historical modeled and projected annual mean monthly minimum temperature, annual mean monthly maximum temperature, and total annual precipitation for 1950–2100 under RCP 4.5 and RCP 8.5 emissions scenarios based on CMIP5 data for the Intermountain Semi Desert subregion of the Intermountain Adaptation Partnership region. Historical annual data from the gridded Maurer et al. (2007) dataset are also shown. See box 3.1 for further explanation.

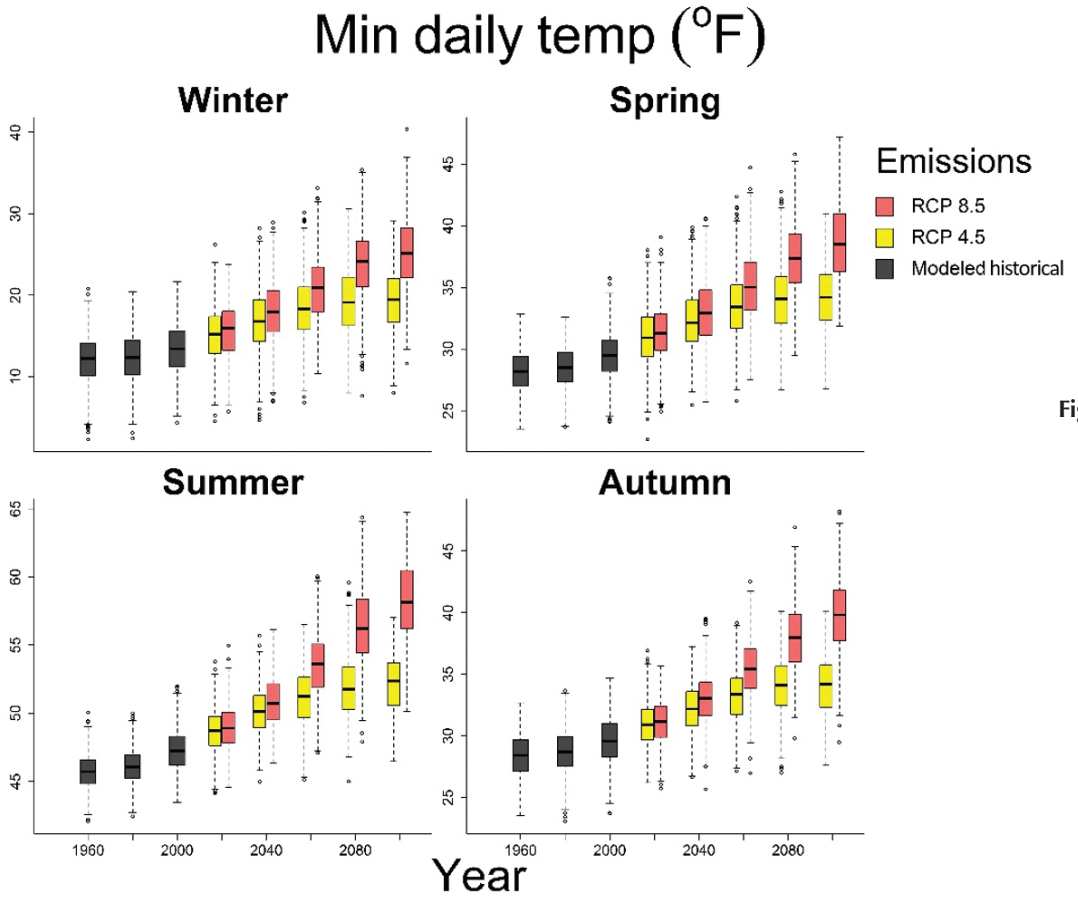


Figure 3.23—Historical modeled and projected seasonal mean monthly minimum temperature for 1950–2100 under RCP 4.5 and RCP 8.5 emissions scenarios for the Intermountain Semi Desert subregion of the Intermountain Adaptation Partnership region. See box 3.1 for further explanation.

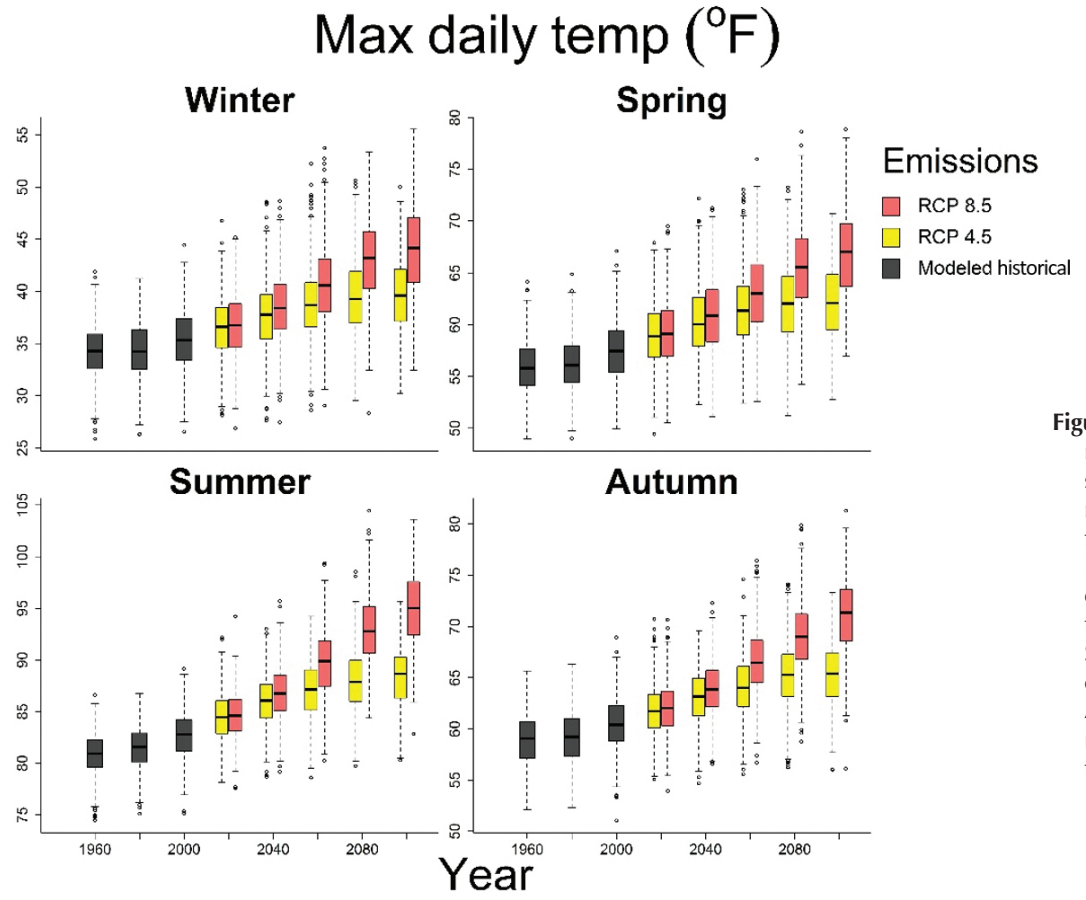


Figure 3.24— Historical modeled and projected seasonal mean monthly maximum temperature for 1950–2100 under RCP 4.5 and RCP 8.5 emissions scenarios for the Intermountain Semi Desert subregion of the Intermountain Adaptation Partnership region. See box 3.1 for further explanation.

References

Andretta, T.A.; Geerts, B. 2010. Heavy snowfall produced by topographically induced winds in the Snake River Plain of Eastern Idaho. Part I: Observational analysis. E-Journal of Severe Storms Meteorology. 5. http://www.ejssm.org/ojs/index.php/ejssm/article/viewArticle/56/74 [Accessed July 13, 2017].

Bekker, M.F.; DeRose, R.J.; Buckley, B.M.; [et al.]. 2014. A 576-year Weber River streamflow reconstruction from tree rings for water resource risk assessment in the Wasatch Front, Utah. Journal of the American Water Resources Association. 50: 1338–1348.

Davey, C.A.; Redmond, K.T.; Simeral, D.B. 2006. Weather and climate inventory, National Park Service, Rocky Mountain Network. Natural Resource Technical Report NPS/GRYN/NRTR—2006/001. Fort Collins, CO: National Park Service. 83 p.

DeRose, R.J.; Bekker, J.F.; Wang, S.-Y.; [et al.]. 2015. A millennium-length reconstruction of Bear River stream flow, Utah. Journal of Hydrology. 529: 524–534.

Gillies, R.R.; Ramsey, R.D. 2009. Climate of Utah. In: Banner, R.E., Baldwin, B.D., Leydsman McGinty, E.K., project coordinators. Rangeland resources of Utah. Logan, UT: Utah State University, Cooperative Extension Service. http://extension.usu.edu/rangelands/ou-files/RRU_Introduction.pdf [Accessed December 5, 2016].

Gillies, R.R.; Wang, S-Y.; Booth, M.R. 2012. Observational and synoptic analyses of the winter precipitation regime change over Utah. Journal of Climate. 25: 4679–4698.

Gray, S.T.; Jackson, S.T.; Betancourt, J.L. 2004. Tree-ring based reconstructions of interannual to decadal scale precipitation variability for northeastern Utah since 1226 A.D. Journal of American Water Resources Association. 40: 947–960.

Hoekema, D.J.; Sridhar, V. 2011. Relating climatic attributes and water resources allocation: A study using surface water supply and soil moisture indices in the Snake River Basin, Idaho. Water Resources Research. 47: W07536. doi:10.1029/2010WR009697.

Houghton, J.T.; Ding, Y.; Griggs, D.J.; [et al.], eds. 2001. Climate change 2001: The scientific basis. Contribution of working group I to the third assessment report of the Intergovernmental Panel on Climate Change. Cambridge, United Kingdom and New York, NY: Cambridge University Press. 881 p.

Intergovernmental Panel on Climate Change Data Distribution Center [IPCC]. 2016. Scenario process of AR5. http://sedac.ipcc-data.org/ddc/ar5_scenario_process/index.html [Accessed March 22, 2017].

Isaak, D.J.; Luce, C.H.; Rieman, B.E.; [et al.]. 2010. Effects of climate change and wildfire on stream temperatures and salmonid thermal habitat in a mountain river network. Ecological Applications. 20: 1350–1371.

Klos, P.Z.; Link, T.E.; Abatzoglou, J.T. 2014. Extent of the rain-snow transition zone in the western U.S. under historic and projected climate. Geophysical Research Letters. 41: 4560–4568.

Kunkel, K.E.; Stevens, L.E.; Stevens, S.E.; [et al.]. 2013. Regional climate trends and scenarios for the U.S. National Climate Assessment: Part 5. Climate of the Southwest U.S. NOAA Technical Report NESDIS 142-5. Washington, DC: National Oceanic and Atmospheric Administration, National Environmental Satellite, Data, and Information Service. 87 p.

Leydsman McGinty, E.I.; McGinty, C.M. 2009. The physiography of Utah. In: Banner, R.E.; Baldwin, B.D.; Leydsman McGinty, E.K., project coordinators. Rangeland resources of Utah. Logan, Utah: Logan, UT: Utah State University, Cooperative Extension Service. http://extension.usu.edu/rangelands/ou-files/RRU_Introduction.pdf [Accessed December 5, 2016].

Littell, J.S.; Elsner, M.M.; Mauger, G.S.; [et al.]. 2011. Regional climate and hydrologic change in the northern US Rockies and Pacific Northwest: internally consistent projections of future climate for resource management. Unpublished project report. http://cses.washington.edu/picea/USFS/pub/Littell_etal_2010 [Accessed December 5, 2016].

Louderback, L.A.; Rhode, D.; Madsen, D.B.; Metcalf, M. 2015. Rapid vegetation shifts in the Uinta Mountains (Utah and Wyoming, USA) during the Late Pleistocene and Holocene. Palaeogeography, Palaeoclimatology, Palaeoecology. 438: 327–343.

MacDonald, G.M.; Tingstad, A.H. 2007. Recent and multicentennial precipitation variability and drought occurrence in the Uinta Mountains region, Utah. Arctic, Antarctic, and Alpine Research. 39: 549–555.

Maurer, E.P.; Wood, A.W.; Adam, J.C.; [et al.]. 2002. A long-term hydrologically based dataset of land surface fluxes and states for the conterminous United States. Journal of Climate. 15: 3237–3251.

Maurer, E.P.; Brekke, L.; Pruitt, T.; Duffy, P.B. 2007. Fine-resolution climate projections enhance regional climate change impact studies. Eos Transactions of the American Geophysical Union. 88: 504.

Meehl, G.A.; Covey, C.; Delworth, T.; [et al.]. 2007. The WCRP CMIP3 multimodel data set. Bulletin of the American Meteorological Society. 88: 1383–1394.

Monahan, W.B.; Fisichelli, N.A. 2014. Climate exposure of US national parks in a new era of change. PloS One. 9: e101302. https://doi.org/10.1371/journal.pone.0101302

Moss, R.; Babiker, W.; Brinkman, S.; [et al.]. 2008. Towards new scenarios for the analysis of emissions, climate change, impacts and response strategies. IPCC Expert Meeting Report. Technical Summary. Geneva, Switzerland: Intergovernmental Panel on Climate Change. 25 p.

Moss, R.H.; Edmonds, J.A.; Hibbard, K.A.; [et al.]. 2010. The next generation of scenarios for climate change research and assessment. Nature. 463: 747.

Morrisette, P.M. 1988. The rising level of the Great Salt Lake: Impacts and adjustments. Bulletin of the American Meteorological Society. 69: 1034–1040.

Nakićenović, N.; Davidson, O.; Davis, G.; [et al.]. 2000. Special report on emissions scenarios: A special report of Working Group III of the Intergovernmental Panel on Climate Change. Cambridge, United Kingdom: Cambridge University Press. 599 p.

Ostlind, E. 2011. The Green River Basin: A natural history. Wheatland, WY: Wyoming State Historical Society. http://www.wyohistory.org/encyclopedia/green-river-basin-natural-history [Accessed December 5, 2016].

Oyler, J.W.; Ballantyne, A.; Jencso, K.; [et al.]. 2015a. Creating a topoclimatic daily air temperature dataset for the conterminous United States using homogenized station data and remotely sensed land skin temperature. International Journal of Climatology. 35: 2258–2279.

Oyler, J.W.; Dobrowski, S.Z.; Ballantyne, A.P.; [et al.]. 2015b. Artificial amplification of warming trends across the mountains of the western United States. Geophysical Research Letters. 42: 153–161.

PRISM Climate Group. 2014. Oregon State University. PRISM 4 km climate data, 2014. http://prism.oregonstate.edu/ [Accessed March 31, 2016].

Rice, J.; Bardsley, T.; Gomben, P.; [et al.]. 2017. Assessment of watershed vulnerability to climate change for the Uinta-Wasatch-Cache and Ashley National Forests, Utah. Gen. Tech. Rep. RMRS-GTR-362. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 111 p.

Rogelj, J.; Meinshausen, M.; Knutti, R. 2012. Global warming under old and new scenarios using IPCC climate sensitivity range estimates. Nature Climate Change. 2: 248–253.

Solomon, S.; Qin, D.; Manning, M.; [et al.], eds. 2007. Climate change 2007: The physical science basis. Contribution of Working Group I to the fourth assessment report of the Intergovernmental Panel on Climate Change. Cambridge, UK and New York, NY: Cambridge University Press. 996 p.

Stocker, T.F.; Qin, D.; Plattner, G.-K.; [et al.], eds. 2013. Climate change 2013: The physical science basis. Contribution of Working Group I to the fifth assessment report of the Intergovernmental Panel on Climate Change. Cambridge, UK and New York, NY: Cambridge University Press. 1,535 p.

Svejcar, T.; Angell, R.; James, J. 2016. Spatial and temporal variability in minimum temperature trends in the western U.S. sagebrush steppe. Journal of Arid Environments. 133: 125–133.

van Vuuren, D.P.; Edmonds, J.; Kainuma, M.; [et al.]. 2011. The representative concentration pathways: An overview. Climatic Change. 109: 5–31.

Walsh, J.; Wuebbles, D.; Hayhoe, K.; [et al.]. 2014. Our changing climate. In: Melillo, J.M.; Richmond, T.C.; Yohe, G.W., eds. Climate change impacts in the United States: The Third National Climate Assessment. Washington, DC: U.S. Global Change Research Program. p. 19–67.

Western Regional Climate Center [WRCC]. 2016a. Climate of Idaho. https://wrcc.dri.edu/Climate/narrative_id.php [Accessed November 19, 2016].

Western Regional Climate Center [WRCC]. 2016b. Climate of Nevada. https://wrcc.dri.edu/Climate/narrative_wy.php [Accessed November 19, 2016].

Western Regional Climate Center [WRCC]. 2016c. Climate of Utah. https://wrcc.dri.edu/Climate/narrative_ut.php [Accessed November 19, 2016].

Appendix 2—Models Included in the Climate Analysis for the Intermountain Adaptation Partnership Region

Following are the climate models we used in the Intermountain Adaptation Partnership Region climate analysis, and the institutions that developed them. CMIP5 climate projections for RCP 4.5 and RCP 8.5 scenarios were obtained for these models using the downscaled CMIP3 and CMIP5 climate and hydrology projections archive at: http://gdo-dcp.ucllnl.org/downscaled_cmip_projections. The first model run was selected for this analysis. Model runs were available for both RCP 4.5 and RCP 8.5 scenarios, except for the NASA Goddard Institute for Space Studies models GISS-E2-H-CC and GISS-E2-R-CC.

Institution	Climate model
Commonwealth Scientific and Industrial Research Organization and Bureau of Meteorology, Australia	ACCESS1-0
	ACCESS1-3
Beijing Climate Center, China Meteorological Administration	bcc-csm1-1
Beijing Climate Center, China Meteorological Administration	bcc-csm1-1-m
Canadian Centre for Climate Modelling and Analysis	CanESM2
National Center for Atmospheric Research	CCSM4
Community Earth System Model Contributors	CESM1-BGC
	CESM1-CAM5
Centro Euro-Mediterraneo per i Cambiamenti Climatici	CMCC-CM
Centre National de Recherches Météorologiques/Centre Européen de Recherche et Formation Avancée en Calcul Scientifique	CNRM-CM5
Commonwealth Scientific and Industrial Research Organization, Queensland Climate Change Centre of Excellence	CSIRO-Mk3-6-0
EC-EARTH consortium	EC-EARTH
Laboratory of Numerical Modeling for Atmospheric Sciences and Geophysical Fluid Dynamics, Institute of Atmospheric Physics, Chinese Academy of Sciences, and Center for Earth System Science, Tsinghua University	FGOALS-g2
Laboratory of Numerical Modeling for Atmospheric Sciences and Geophysical Fluid Dynamics, Institute of Atmospheric Physics, Chinese Academy of Sciences, and Center for Earth System Science, Tsinghua University	FGOALS-s2
The First Institute of Oceanography, State Oceanic Administration, China	FIO-ESM
NOAA Geophysical Fluid Dynamics Laboratory	GFDL-CM3
	GFDL-ESM2G
	GFDL-ESM2M
NASA Goddard Institute for Space Studies	GISS-E2-H-CC
	GISS-E2-R
	GISS-E2-R-CC
Met Office Hadley Centre (additional HadGEM2-ES realizations contributed by Instituto Nacional de Pesquisas Espaciais)	HADGEM2-AO
Met Office Hadley Centre (additional HadGEM2-ES realizations contributed by Instituto Nacional de Pesquisas Espaciais)	HADGEM2-CC
Met Office Hadley Centre (additional HadGEM2-ES realizations contributed by Instituto Nacional de Pesquisas Espaciais)	HADGEM2-ES

Institute for Numerical Mathematics	INM-CM4
Institut Pierre-Simon Laplace	IPSL-CM5A-LR
	IPSL-CM5A-MR
	IPSL-CM5B-LR
Japan Agency for Marine-Earth Science and Technology, Atmosphere and Ocean Research Institute (The University of Tokyo), and National Institute for Environmental Studies	MIROC-ESM
	MIROC-ESM-CHEM
Atmosphere and Ocean Research Institute (The University of Tokyo), National Institute for Environmental Studies, and Japan Agency for Marine-Earth Science and Technology	MIROC5
Max Planck Institute for Meteorology (MPI-M)	MPI-ESM-LR
Max Planck Institute for Meteorology (MPI-M)	MPI-ESM-MR
Meteorological Research Institute	MRI-CGCM3
Norwegian Climate Centre	NorESM1-M
Norwegian Climate Centre	MorESM1-ME

Chapter 4: Effects of Climate Change on Hydrology, Water Resources, and Soil

Mark J. Muir, Charles H. Luce, Joseph T. Gurrieri, Marek Matyjasik, Jeffrey L. Bruggink, Stacey L. Weems, James C. Hurja, David B. Marr, and Sarah D. Leahy

Introduction

Water is critical to life, and many of the effects of climate change on ecosystems are mediated through altered hydrology. Snow accumulation and melt are consistently cited as the most important changes to water in the western United States (Barnett et al. 2005; Service 2004), affecting when water will be available for forests, fish, and people. Changes in summer atmospheric circulation patterns may alter the ability of summer precipitation to provide a midsummer respite from seasonal drought and dampening of wildfire spread (IPCC 2013) (Chapter 8). Declining summer water contributions will challenge municipal and agricultural water supplies (Barnett and Pierce 2009; Dawadi and Ahmad 2012). Aquatic and terrestrial ecosystems—including riparian areas, wetlands, and groundwater-dependent ecosystems—will be affected by lower base flows (Kormos et al. 2016; Rood et al. 2008), earlier snowmelt (Luce et al. 2014), increased periods of drought (Cayan et al. 2009), increased sediment delivery (Goode et al. 2012), and higher midwinter floods (Goode et al. 2013). Soils will likewise be affected by increased temperatures and shifts in precipitation and hydrological processes, with effects on physical and biological processes and attributes of soils.

This chapter describes potential changes to hydrological processes, groundwater resources, and soil attributes and processes in the Intermountain Adaptation Partnership (IAP) region with a changing climate. We specifically discuss potential changes in snowpack and glaciers, streamflow, drought, sediment yield, and groundwater recharge, and in soil temperature, moisture, carbon, nitrogen, biological activity, and chemical properties. The *Soil Resources* section concludes with an example vulnerability assessment method and application.

Hydrological Processes

Climate and Hydrological Processes

Warming temperatures are the most certain consequence of increased carbon dioxide in the atmosphere. The hydrological consequences of warmer temperatures include less snowpack and greater evaporative demand from the atmosphere. In general, snowpack depth, extent, and duration are expected to decrease, particularly at lower and middle elevations, because of a combination of less precipitation falling as snow (Pierce et al. 2008) and slightly earlier melt (Luce et al. 2014). The degree of change expected as a result of warming varies considerably over the landscape as a function of temperature (Luce et al. 2014). Places that are warm (near the melting point of snow) are expected to be more sensitive than places where temperatures remain subfreezing throughout much of the winter despite warming (Woods 2009).

The relationship of evapotranspiration (ET) to a warming climate is more complicated (Roderick et al. 2014). Warmer air can hold more water, which means that even if relative humidity stays constant, vapor pressure deficit—the difference between actual water content of the air and water content at saturation—increases. That difference between actual and saturation drives a water vapor gradient between leaves and the atmosphere that can draw more moisture out of the leaves. This is likely to cause more evaporation in a warmer climate (Cook et al. 2014; Dai 2013).

However, evaporation is an energy-intensive process, and there is only so much additional energy that will be available for evaporation. In addition, one needs to consider both the water balance and the energy balance when considering future warming (Roderick et al. 2015). The observation that temperatures are warmer during drought is related more generally to the lack of water to evaporate, leading to warmer temperatures, than to warmer temperatures causing faster evaporation (Yin et al. 2014). Unfortunately, when potential ET models based on air temperature (including Penman-Monteith) are applied as postprocessing to global climate model (GCM) calculations, an overestimate of increased ET is likely, because the energy balance is no longer tracked (Milly 1992; Milly and Dunne 2011). The reality is that most of the increased energy from increased longwave radiation will result in warming rather than increased evaporation (Roderick et al. 2015).

Precipitation has a direct effect on hydrological processes, although precipitation is less commonly discussed because climate change projections are uncertain (Blöschl and Montanari 2010; IPCC 2013). Figure 4.1 helps to illustrate how the IAP region is located in an area of high

uncertainty in regard to precipitation projections, slightly overlapping with projected increases to the north and drying to the south (Walsh et al. 2014). The bounds of uncertainty (-20 to +30 percent) are large, making it difficult to accurately project the effects of precipitation on many hydrological processes (e.g., floods, hydrological drought, snow accumulation, groundwater recharge). As a consequence, we use an ensemble average precipitation for streamflow projections here. In this assessment, we also discuss uncertainty surrounding the mean estimate to illustrate which processes or hydrological outcomes are most uncertain and where. Not all processes are sensitive to precipitation, and uncertainty in outcomes caused by uncertainty in precipitation is not the same everywhere for a given process.

Background information can help to clarify where and when some precipitation estimates may be more reliable than others. Two primary concepts are applied for precipitation change: dynamic (referring to changes in wind and atmospheric circulation) and thermodynamic (referring to how much water the air can hold) (Seager et al. 2010). Dynamic drivers of precipitation change include changes in global circulation patterns (e.g., the Hadley cell extent) and changes in mid-latitude storm tracks. Changes in teleconnection patterns (e.g., the North American Monsoon System [NAMS]) fall into this category and are very important for this region. Thermodynamic changes reflect the fact that the atmosphere can hold more water (Held and Soden 2006), leading to an expectation on the order of a 3.9 percent increase in precipitation per 1 °F of temperature change. There are, however, other physical limits on the disposition of energy driving the cycling of water in the atmosphere. These lead to estimates on the order of less than 1 percent per 1 °F of temperature change at the global scale, with individual grid cells being less or potentially negative, particularly over land (Roderick et al. 2014). Different approaches to scaling the thermodynamic contribution are a reason for differences among models, although the dynamic process modeling differences can be great as well.

One outcome of thermodynamically driven changes is that when precipitation occurs, the same total volume is expected to fall with greater intensity, leading to shorter events and longer dry periods between events. The number of consecutive dry days is projected to increase across the western United States, which will affect portions of the IAP region (fig. 4.1). In the Pacific Northwest, this projection is connected to an expected decrease in summer precipitation, but for the Southwest, it is more likely connected to a decrease in monsoonal moisture during the late spring (fig. 4.2). In general, the NAMS is expected to weaken (IPCC 2013), particularly in the early portion of the monsoon season (Cook and Seager 2013), which could have substantial effects across the southern portion of the IAP region. Late spring and early summer precipitation contributions can be an important determinant of the severity of summer drought and length of fire seasons (Abatzoglou and Kolden 2013). Pairing longer periods of precipitation deficit in summer with decreasing snowpack may be particularly challenging for vegetation and aquatic ecosystems.

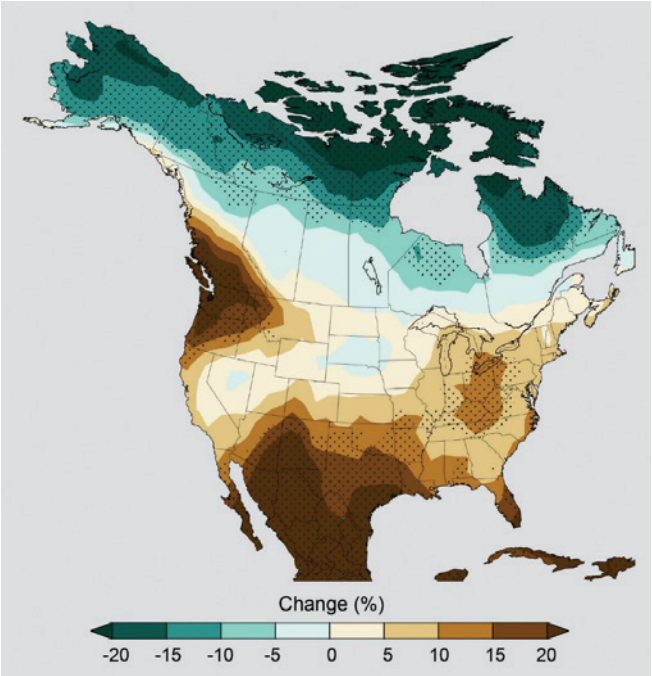


Figure 4.1—Projected change in the number of consecutive dry days for 2071–2099 (compared to 1970–1999) for the RCP8.5 emissions scenario (from Walsh et al. [2014]).

Changes in orographic enhancement of precipitation over mountainous areas also have dynamic effects. Historical changes in westerly windflows have led to a decrease in the enhancement of winter precipitation by orographic lifting over mountain ranges (Luce et al. 2013), raising the question of whether such a pattern may continue into the future. There is general agreement among GCMs projecting further decreases in windspeed into the 21st century, but the correlation is applicable only to the northern portion of the IAP region. Westerly winds are strongly correlated with precipitation in mountainous areas (fig. 4.3), but valley precipitation is not, nor is precipitation in much of southern Idaho. The historical trend in westerlies was driven by pressure and temperature changes spatially consistent with those expected under a changing climate; however, the rapidity of the changes in the last 60 years may have been partly enhanced by normal climatic variability.

Dynamic downscaling using a regional climate model (RCM) with small (~8 mile) cells provides a means to estimate orographically induced precipitation, which cannot be simulated with the large cell size of GCMs. Although the GCM shows general moistening over most of the area, the RCM shows a pattern of drying or no change on the upwind side of major mountain ranges, with moistening limited to valleys in the lee. Because mountainous areas are where most of the precipitation falls (and streamflow originates), this is a potentially important aspect of future changes. The Variable Infiltration Capacity (VIC) model simulations

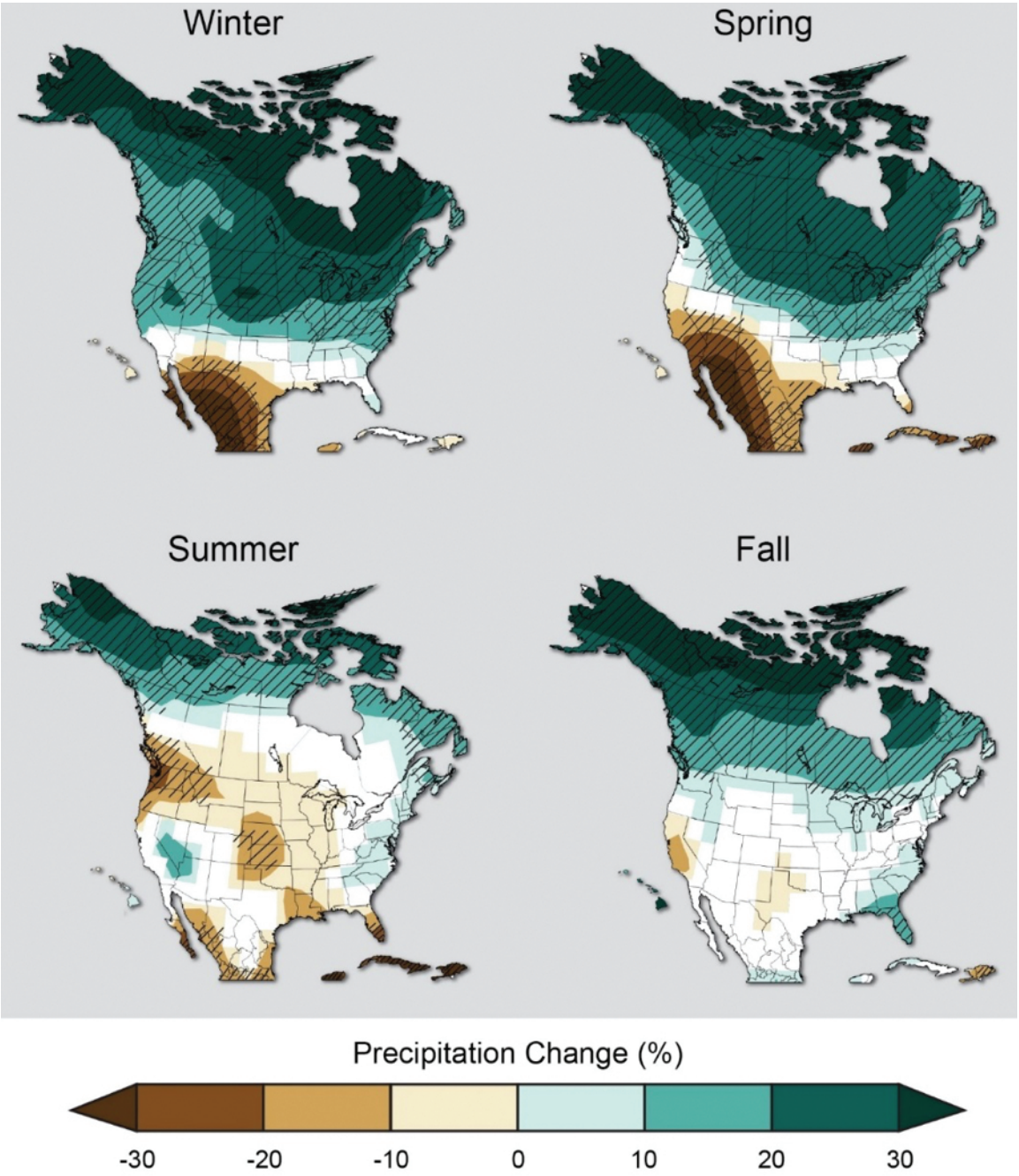


Figure 4.2—Projected change in seasonal precipitation for 2071–2099 (compared to 1970–1999) under the RCP 8.5 emissions scenario (Walsh et al. 2014). Hatched lines indicate model agreement and nonhatched areas have the highest uncertainty. The Intermountain Adaptation Partnership region sits in an area of high uncertainty, between projected moistening to the north and drying to the south.

discussed later in this chapter do not include this effect, so for purposes of general discussion, it can be considered an additional source of uncertainty for precipitation.

The range of potential changes to climate is complex, particularly for such a varied landscape as the IAP region (fig. 4.2), and current climatological settings vary over the landscape at both large and small spatial scales. Precipitation seasonality and amount differ between mountain and valley locations. Trends and drivers for climatic variation will also differ from the northern to southern portions of the region. Fundamentally, topography is an important factor affecting seasonality, precipitation amount, and potential trends. Because most forests and generation of water supply are generally in mountainous areas, it is important to recognize how topography affects climate. Specific hydrological outcomes of interest are discussed in the following sections.

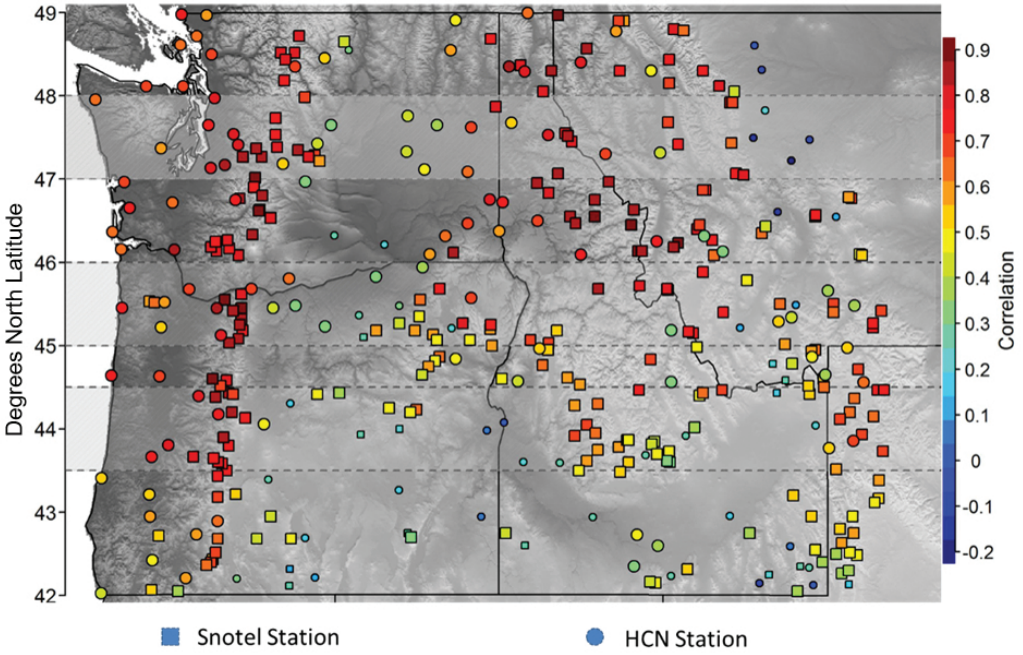


Figure 4.3—Correlation of winter precipitation to winter westerly windspeed across the Pacific Northwest, showing snowpack telemetry (SNOTEL) and Historical Climate Network (HCN) stations (from Luce et al. [2013]).

Snowpack and Glaciers

Snowpack

Snowpack declines are among the most widely cited changes occurring with climate change, through the effect of warmer temperatures on the fraction of precipitation falling as snow (Barnett et al. 2008). About 70 percent of the water supply in the western United States is tied to mountain snowpacks (Service 2004), so changes in snowpack are highly relevant to municipal and agricultural water supplies and timing (Stewart et al. 2005).

Historical trends in snowpack accumulation have been negative across most of the western United States (Mote et al. 2005; Regonda et al. 2005). However, care must be taken when looking at individual sites, which can be influenced by site-specific effects such as vegetation changes, physical site changes, sensor changes, and measurement technique (Clayton and Julander 2015). Temperature sensitivity of the snowpack is highest in places that are already relatively warm (warm snowpacks), and warm snowpacks with high precipitation are likely to undergo some of the largest changes in snow storage as the climate warms (Luce et al. 2014; Nolin and Daly 2006).

The most sensitive locations within the IAP region include the eastern Sierra Nevada and mid- to lower-elevation sites across Idaho, Utah, and Nevada (figs. 4.4–4.7). In contrast, many interior portions of the IAP region are cold enough to be relatively insensitive to warming and strongly sensitive to precipitation variation (Luce et al. 2014; Mote 2006). At the coldest and highest elevations, in the Uinta, Teton, Wind River, and some central Idaho ranges, for instance, there could be increases in snow water equivalent (SWE) if precipitation increases (Rice et al. 2017). Despite warming temperatures, a large proportion of precipitation would still fall as snow in these areas. This means that the future of snow, and consequently hydrology in these regions, depends on one of the more uncertain parts of GCM projections: the precipitation.

Precipitation uncertainty can be substantial, but it does not translate into equal uncertainty in snowpack changes everywhere (fig. 4.8). We estimated sensitivity of April 1 SWE using data from 524 snowpack telemetry (SNOTEL) stations across the western United States in a space-for-time model (Luce et al. 2014). This allowed us to determine where in the western United States snowpack was more sensitive to variability in precipitation or variability in temperature. We computed an index of uncertainty as the ratio (R_u) of the effects on snow (ΔS) from the likely range of precipitation values (about ±7.5 percent for 1 standard deviation across models) in the numerator, to ΔS from the relatively certain temperature change in the denominator:

$$R_u = \frac{\Delta S \text{ across precipitation uncertainty } (+7.5\%)}{\Delta S \text{ expected from warming}}$$

We found strong certainty of large changes in April 1 SWE for the Cascade Range, Sierra Nevada, and the Southwest ($R_u < 0.2$). But we found substantial uncertainty ($R_u > 0.6$) in outcomes for interior locations such as the Greater Yellowstone Area and higher elevations in Idaho and Utah, where cold temperatures leave the snowpack more sensitive to precipitation than to temperature changes (fig. 4.8). The uncertainty ratio in these colder areas suggests that relatively large increases in precipitation could help counter the effects of warming on snowpack loss. These results are similar to those seen using a physically based model across the western United States (Gergel et al. 2017).

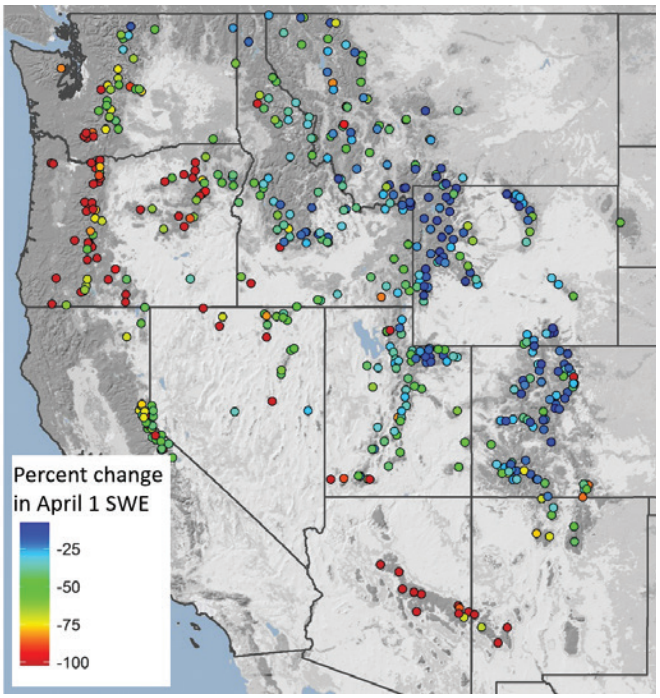


Figure 4.4—Estimated April 1 snow water equivalent (SWE) sensitivity (percentage change) for a 5.5-°F increase in winter average temperature at each snowpack telemetry station (modified from Luce et al. [2014]).

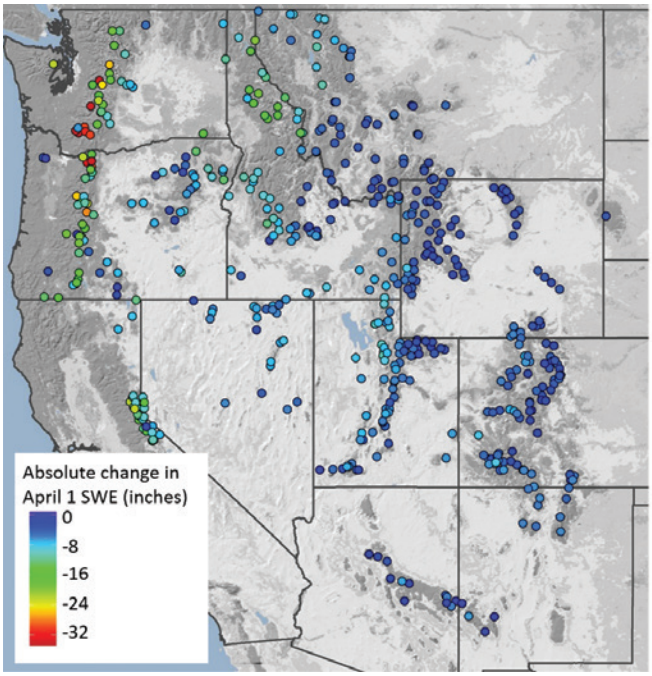


Figure 4.5—Estimated April 1 snow water equivalent (SWE) sensitivity (absolute change in inches) for a 5.5-°F increase in winter average temperature at each snowpack telemetry station (modified from Luce et al. [2014]).

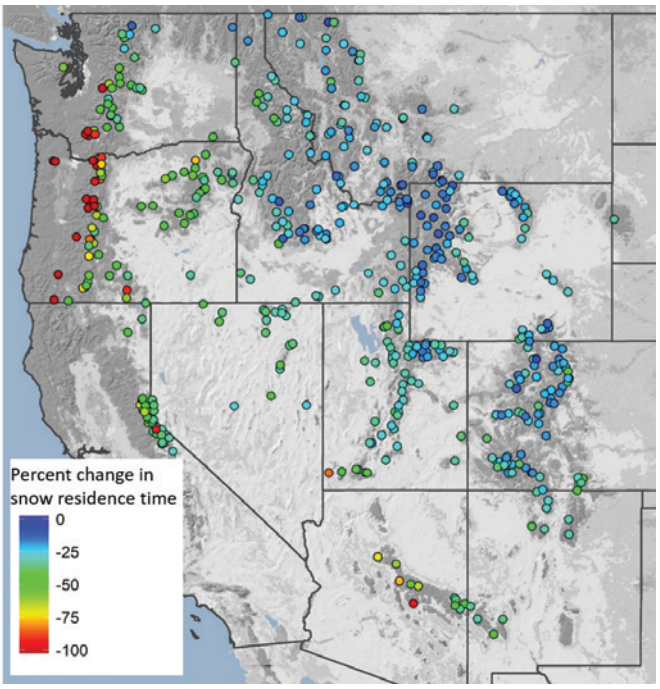


Figure 4.6—Estimated mean snow residence time sensitivity (percentage change) for a 5.5-°F increase in winter average temperature at each snowpack telemetry station (modified from Luce et al. [2014]).

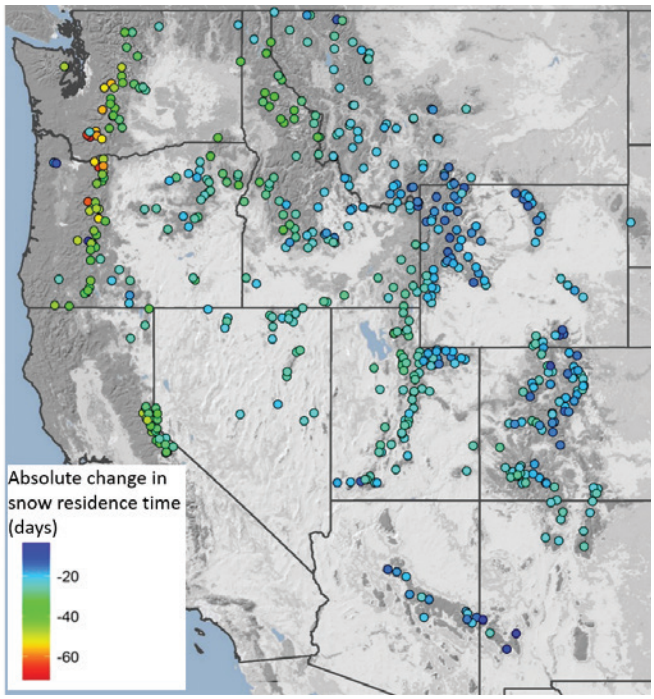


Figure 4.7—Estimated mean snow residence time sensitivity (absolute change in days), for a 5.5-°F increase in winter average temperature at each snowpack telemetry station (modified from Luce et al. [2014]).

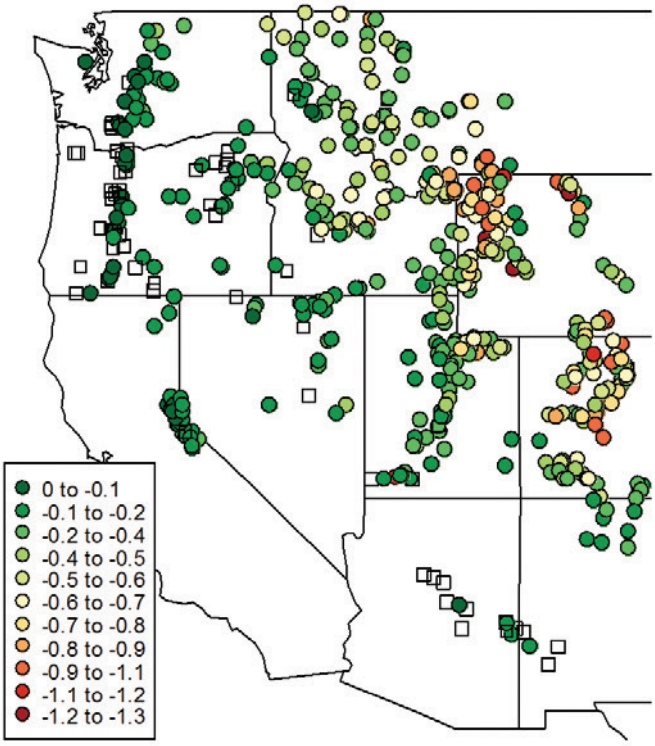


Figure 4.8—Uncertainty ratio for April 1 snow water equivalent. Orange to dark red sites are strongly influenced by precipitation in contrast to temperature. Thus, temperature-based projections for snow water equivalent in those sites may be inaccurate if precipitation changes are large. At dark green (and white) sites, temperature effects will predominate, and precipitation changes in either direction are inconsequential.

Glaciers

Glaciers are limited throughout the IAP region but do occur in central Idaho, in western Wyoming, and in an isolated location at Wheeler Peak in Great Basin National Park (see maps at Portland State University 2009). Declines in the extent of glaciers in the Wind River Range have been observed over the 20th century (Marston et al. 1991).

Estimating future changes in glaciers is complex (Hall and Fagre 2003), but empirical relationships derived for glaciers indicate a brief future for them, with many glaciers becoming fragmented or disappearing by the 2030s. Increasing temperatures yield a rising equilibrium line altitude (ELA), decreasing the effective contributing area for each glacier as warming progresses. Warming of 5.5 °F can translate to an elevation rise of 1,000 to 1,600 feet in snow-rain partitioning and summer temperatures. Those changes do not directly equate to a shift in ELA, which depends on the geometry and topography of the contributing cirque.

Temperate alpine glaciers are well known for being as, or more, sensitive to precipitation variations as they are to temperature variations (McCabe and Fountain 1995), which has very likely contributed to changes in glacial dynamics across the Pacific Northwest. Westerlies and their contribution to winter precipitation have changed over the northern part of the region since the 1940s (Luce et al. 2013), and April 1 SWE at these elevations and latitudes is relatively insensitive to temperature. However, summer temperature is a strong predictor of glacial behavior, and changes in summer temperatures could affect the melt rate and additional snow contributions in glaciers because this area receives significant spring and summer precipitation (Hall and Fagre 2003).

Streamflow

Streamflow changes of significance for aquatic species, water supply, and infrastructure include annual yield, summer low flows (average, extreme), peakflows (scouring floods), peakflow seasonality, and center of runoff timing. Irrigation water for crops and urban landscapes is typically needed in summer months. Annual yield, summer low flows, and center of runoff timing are important metrics with respect to water supply, but they are most relevant to surface water supplies rather than groundwater supplies, although changes in long-term annual means could be informative for the latter. The mean summer yield (June through September) is used for summer low flows. Center of runoff timing is the date on which 50 percent of the annual runoff has flowed out of a basin and is an effective index for the timing of water availability in snowmelt-driven basins. Shifts to earlier runoff in the winter or spring disconnect streamflow timing from water supply needs such as agricultural irrigation. Center of timing can be redundant with other metrics that measure impact more directly, but with care in interpretation, it can help clarify different potential causal mechanisms, such as changing precipitation versus changing temperature.

Peakflows are important to fish and infrastructure. Scouring flows can damage eggs in fish redds if they occur while the eggs are in the gravel or alevins are emerging (Goode et al. 2013; Tonina et al. 2008). Winter peakflows can affect fall-spawning fish (chinook [*Oncorhynchus tshawytscha*], bull trout [*Salvelinus confluentus*], and brook trout [*S. fontinalis*]), whereas spring peakflows affect spring-spawning cutthroat trout (*O. clarkii*), resident rainbow trout (*O. mykiss*), and steelhead (*O. m. gairdneri*) (Wenger et al. 2011a,b). Spring peakflows associated with the annual snowmelt pulse are typically muted in magnitude compared to winter rain-on-snow events for two reasons. The rain-on-snow events can generate larger water input rates (rainfall precipitation plus high melt rate), and they tend to affect much larger fractions of a basin at a time, so scouring is less of a risk to spring-spawning fishes. Consequently, a shift to more midwinter events can yield higher peakflow magnitudes, which can also threaten infrastructure such as roads, recreation sites, and water management facilities (e.g., diversions, dams).

Historical changes in some of these streamflow metrics have been examined in northern portions of the IAP region, specifically earlier runoff timing (Cayan et al. 2001; Stewart

et al. 2005) and declining annual streamflows (Clark 2010; Luce and Holden 2009). Declining low flows (7Q10) have also been observed in the western half of the Northern Rockies (Kormos et al. 2016), associated more with declining precipitation than warming temperature effects for the historical period. Low-flow changes and timing changes in projections are generally associated with expected changes in snowpack related to temperature (e.g., more melt or precipitation as rain in winter, yielding a longer summer dry period). Low-flow changes driven by these timing changes are strongly dependent on groundwater conditions in the basin (Tague and Grant 2009), which vary considerably across the IAP region as discussed later in the Groundwater Resources section.

Streamflow Projections

Streamflow projections for an ensemble of Coupled Model Intercomparison Project Phase 3 (CMIP3) models under the A1B scenario (Littell et al. 2011) were produced from the VIC model (Liang et al. 1994) for the western United States (University of Washington, Climate Impacts Group 2017)) (figs. 4.9–4.13). Differences between the climate described by CMIP3 projections and the more recently developed CMIP5 projections are minimal with respect to temperature (Chapter 3). The gridded VIC data were used to estimate streamflow by using area-weighted averages of runoff from each VIC grid cell within a given basin, following the methods of Wenger et al. (2010), to accumulate flow and validate. Streamflow metrics were calculated for stream segments in the National Hydrography Dataset Plus (version 2) stream segments (USDA FS n.d.).

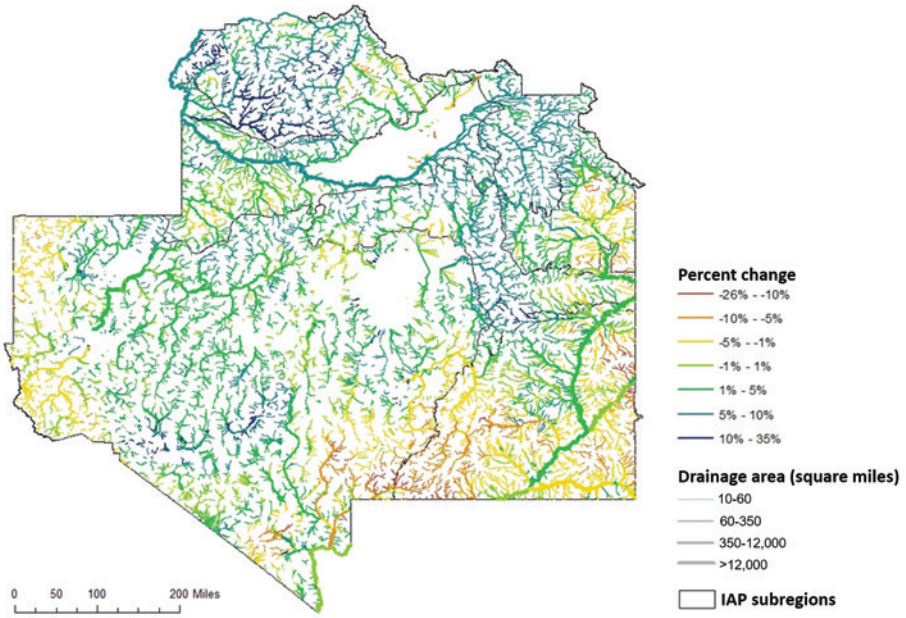


Figure 4.9—Percentage change in mean annual flow projections in the Intermountain Adaptation Partnership (IAP) region between a historical period (1970–1999) and the 2040s. Projections are from the Variable Infiltration Capacity model, following the methods of Wenger et al. (2010).

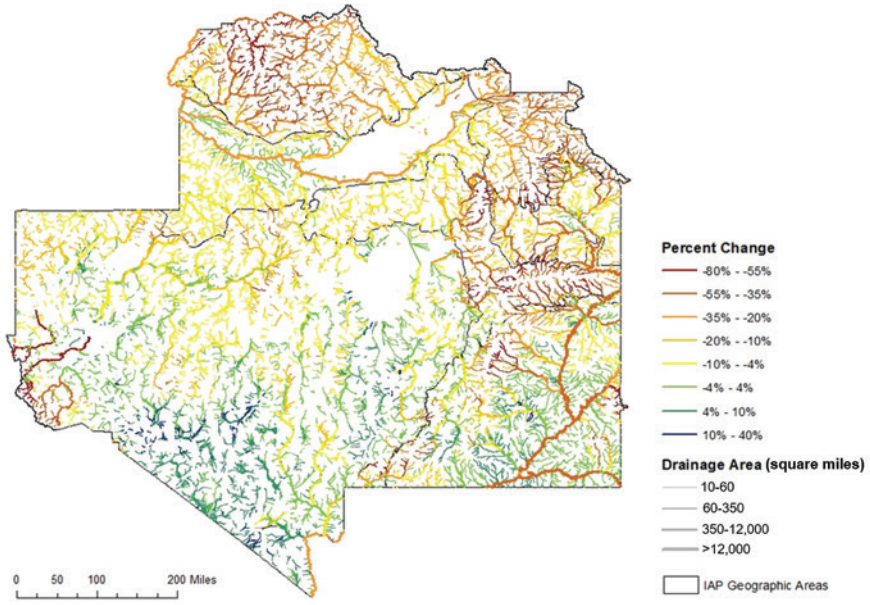


Figure 4.10—Percentage change in mean summer flow projections in the Intermountain Adaptation Partnership region between a historical period (1970–1999) and the 2040s. Projections are from the Variable Infiltration Capacity model, following the methods of Wenger et al. (2010).

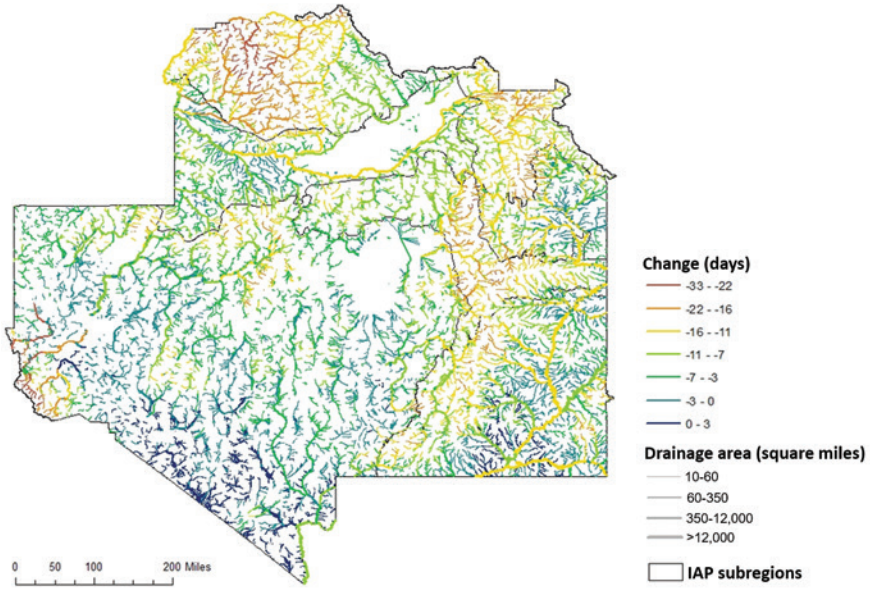


Figure 4.11—Change (days) in the center of flow mass projections in the Intermountain Adaptation Partnership region between a historical period (1970–1999) and the 2040s. Projections are from the Variable Infiltration Capacity model, following the methods of Wenger et al. (2010).

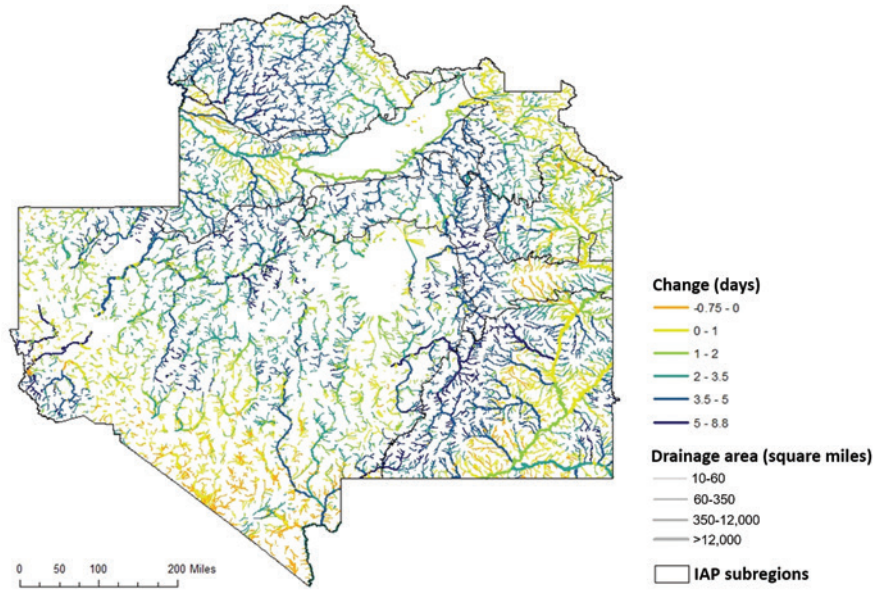


Figure 4.12—Projections of change (days) in the number of mid-winter floods (95th-percentile flow) in the Intermountain Adaptation Partnership region between a historical period (1970–1999) and the 2040s. Projections are from the Variable Infiltration Capacity model, following the methods of Wenger et al. (2010).

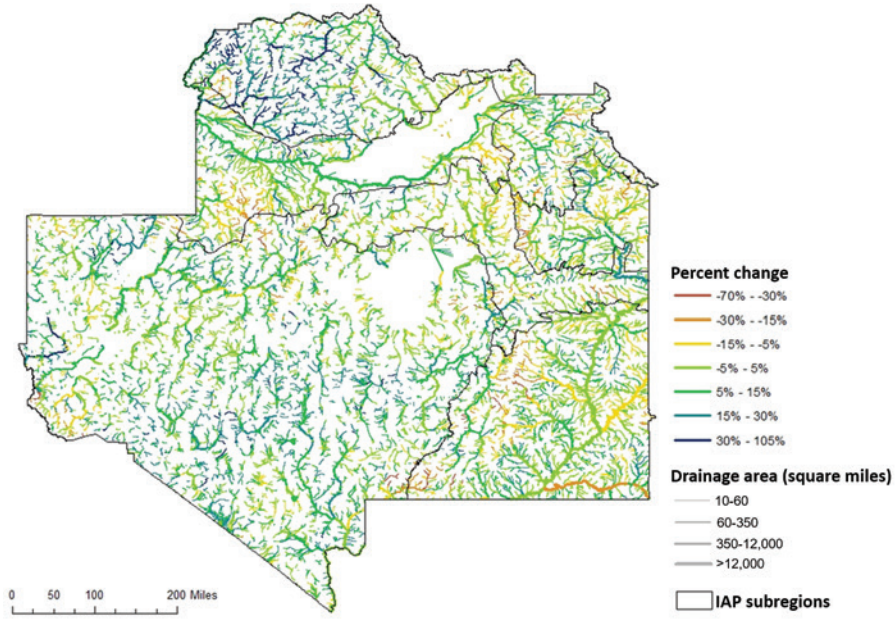


Figure 4.13—Percentage change in 1.5-year flood magnitude (approximate "bankfull" flow) in the Intermountain Adaptation Partnership region between a historical period (1970–1999) and the 2040s. Projections are from the Variable Infiltration Capacity model, following the methods of Wenger et al. (2010).

Uncertainty in climate model inputs can be a significant factor in uncertainty for outcomes related to natural resources (Wenger et al. 2013). Downscaling for these runs was done statistically, not dynamically, using an RCM to account for orographic enhancement changes, so GCM expectations for precipitation are implicit in the streamflow estimates. No effects of change in orographic enhancement are inherent in these images; thus, uncertainty may be higher (in a drier direction) on the windward side of mountain ranges.

Mean annual flow projections (fig. 4.9) suggest a slight increase across the northern portion of the IAP region, which ties back to the general moistening predicted by CMIP3 GCM runs (also illustrated in fig. 4.2). Minor changes are displayed through the central part of the region. The decreases shown in the southern part of the region are associated with changes in the Hadley cell circulation, which has also been described as an expansion of the mid-latitude deserts.

Despite projections of slightly increased annual flow across much of the region, summer low flows are expected to decline (fig. 4.10), with relatively uniform changes in mountainous areas, particularly in wetter ranges. The primary mechanism expected to drive lower summer base flows is reduced snowpack in winter, leading to less stored water. The VIC model simulations do not include the effects of large groundwater reserves; thus, this effect could be moderated in systems where groundwater flow contributes a substantial volume of water to late summer flows (see more discussion in the *Groundwater Resources* section on where this may be important). Although such groundwater support could moderate the percentage declines as shown in figure 4.10, actual low-flow runoff rates could have greater declines in such places because the fractional decline is applied to a larger pre-change low-flow rate (e.g., Tague and Grant 2009). This is an important consideration when dealing with water rights, in which actual volumes or flows, rather than percentages, are allocated to individual rights.

Places where summer precipitation plays an important role, particularly the southern portion of the region, are more likely to see low flows affected by summer precipitation patterns. Shifts in circulation that affect how moisture flows from the Gulf of Mexico in summer are expected to negatively affect precipitation. Increased spacing between precipitation events (IPCC 2013; Luce et al. 2016) and decreased moisture in the early portion of the monsoon season (Cook and Seager 2013) are other likely occurrences. These summer wet areas are also more likely to have greater losses of precipitation with increased evaporation, but it is important to recognize energy balance constraints when estimating the degree of loss (Roderick et al. 2014). This is not done in the VIC modeling, which uses only the temperature outputs from GCMs without reevaluating the change in energy balance from a different hydrological formulation; loss by evapotranspiration may thus be overestimated (Milly and Dunne 2011).

Generally, areas showing a change in summer low flows also show a shift to earlier center of flow mass timing (2–4 weeks) (fig. 4.11) and a shift to stronger changes in mountains dominated by snowmelt runoff. Changes in timing are related to snow residence time and earlier snowmelt runoff (figs. 4.6, 4.7).

Projected changes in the number of winter floods (95th percentile flow) show more of an effect in mid- and lower-elevation mountain ranges. Higher elevation and colder ranges, which will preserve more snowpack, show less change (fig. 4.12). The shift to more midwinter rain and more rain-on-snow flooding depends strongly on the elevational range of each basin. At middle elevations, temperatures are projected to increase enough that rain is likely on snowpacks, even in midwinter. Consequently, projected peakflow increases are generally stronger in these mid-elevation mountainous areas (fig. 4.13). Greater midwinter flooding could increase both the occurrence and magnitude of peakflows (fig. 4.14), as well as the potential for scour in gravel riverbeds (Goode et al. 2013).

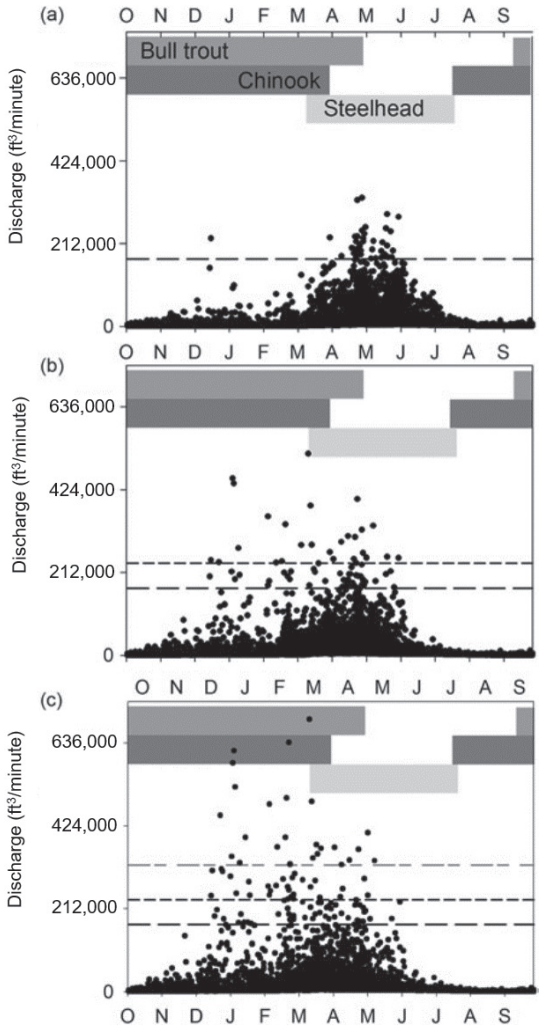


Figure 4.14—Illustration of increased mid-winter flooding potential. Projected streamflows are from Variable Infiltration Capacity modeling for (a) current conditions, (b) 2040s, and (c) 2080s. The long, short, and gray dashed lines indicate the 2-year flood for each period (current, 2040s, and 2080s, respectively) (from Goode et al. [2013]).

Drought

Several studies help to provide a paleoclimatic context for evaluating drought in the IAP region. For example, both an early and an updated reconstruction of streamflow in the Colorado River Basin indicate that water allocation agreements were developed during one of the wettest periods in the last 500 years (Stockton and Jacoby 1976; Woodhouse et al. 2006), and that droughts were more severe before the 20th century (Woodhouse et al. 2006). Similarly, DeRose et al. (2015) found that in the Bear River of the Great Basin, the latter half of the 20th century was the second wettest period in the last 1,200 years. Other studies have also demonstrated high variability and severe droughts in the Uinta Mountains (MacDonald and Tingstad 2007), Weber River (Bekker et al. 2014), Logan River (Allen et al. 2013), and Great Salt Lake (Wang et al. 2012). Figure 4.15 illustrates a general correlation in wet and dry cycles between these basins over time, but also some unique differences based on onsite-specific factors (DeRose et al. 2015).

Understanding long-term climate dynamics is critical for sustainable management of environmental resources. In combination with projections for climate change, knowledge of past climatic conditions can help inform water and land management decisions. For a more extensive discussion of drought, paleoclimatic history, and effects on forests and streams, see Luce et al. 2016.

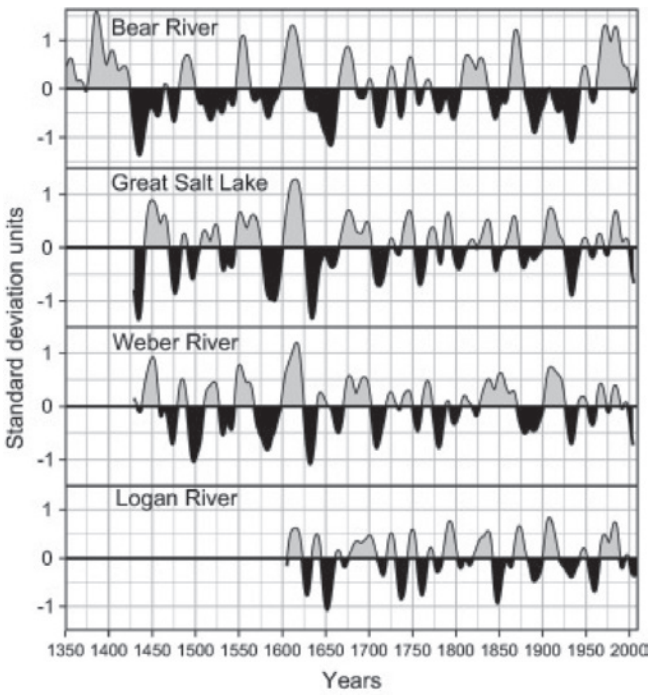


Figure 4.15—Comparison between the Bear River and other Wasatch Front hydroclimate reconstructions, illustrating the cyclical nature of wet and dry periods in the Intermountain Adaptation Partnership region (from DeRose et al. [2015]).

Sediment Yield

The delivery and transport of sediment through mountain rivers affect aquatic habitat and water resource infrastructure. Although climate change is expected to produce significant changes in hydrology and stream temperature, the effects of climate change on sediment yield have received less attention. Climate change is expected to increase sediment yield primarily through the effects of temperature and hydrology on vegetation disturbances (wildfire, insects, drought-related mortality) (Goode et al. 2012).

A conceptual model (fig. 4.16) of sediment yield (solid black line) relative to climate can help to illustrate the regulating role of vegetation. The dashed lines indicate the relative increase in resistance to erosion that vegetation provides as the driving force of precipitation increases. The biggest divergence in the lines occurs in semiarid climates where sufficient precipitation is available to drive erosion, but there is a limited amount of vegetation to stabilize hillslopes from erosion. The result is higher sediment yield in semiarid climates. The red arrow and circle on the plot depict the potential shift of current temperate forest climates to more semiarid climates, increasing overall erosion potential and sediment yield (Goode et al. 2012) (fig. 4.16).

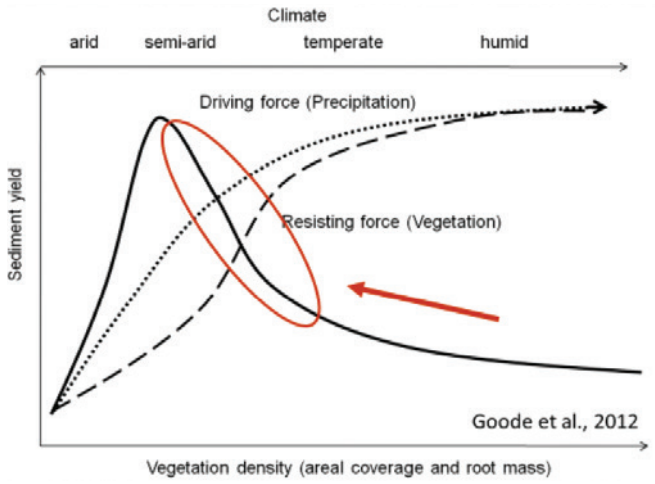


Figure 4.16—Conceptual plot of sediment yield (solid black line) relative to hydroclimate and the regulating role of vegetation. The red arrow and circle illustrate the potential shift of current temperate forest climates, which under a warming climate could become more like semiarid climates, increasing erosion potential and sediment yield (from Goode et al. [2012]).

Groundwater Resources

Climate change is likely to have significant, long-term implications for groundwater resources in the IAP region. Climate change is expected to cause a transition from snow to rain, resulting in diminished snowpack, shifts in streamflow to earlier in the season (Leibowitz et al. 2014; Luce et

al. 2012), and changes in groundwater recharge to aquifers and groundwater discharge to groundwater-dependent ecosystems (GDEs). In this section, we synthesize existing information about occurrence of groundwater resources in five of the six subregions, describe potential effects of climate change, and describe how climate change can affect GDEs, including aquifers, streams, wetlands, and springs.

Groundwater is broadly defined as "all water below the ground surface, including water in the saturated and unsaturated zones" (USDA FS 2012). Groundwater resources include water residing in the subsurface, as well as ecosystems that depend on the presence or discharge of groundwater.

Groundwater-dependent ecosystems are "communities of plants, animals and other organisms whose extent and life processes are dependent on access to or discharge of groundwater" (USDA FS 2012). In the IAP region, GDEs include springs, springbrooks, groundwater-supported lakes, fens, streams, and rivers with base flow and riparian wetlands or phreatophytic vegetation along segments known as "gaining river reaches." **Fens** are wetlands supported primarily by groundwater with a minimum depth (usually 12–16 inches) of accumulated peat (Chadde et al. 1998; USDA FS 2012a). **Springs** are entirely supported by groundwater. These GDEs contribute significantly to local and regional biodiversity (Murray et al. 2006).

The fundamental hydrological processes that influence GDEs are: (1) amount, timing, and type of precipitation (rain or snow); (2) groundwater recharge; (3) groundwater quality; (4) groundwater discharge; and (5) evapotranspiration (Lins 1997). Along stream segments referred to as "gaining reaches," groundwater enters the stream from the banks or the channel bed, and the volume of downstream streamflow is subsequently increased (Winter 2007; Winter et al. 1996). Groundwater can contribute substantially to late summer streamflow (Gannett 1984) and is the source for cool-water upwellings that serve as refugia for cold-water aquatic species (Lawrence et al. 2014; Torgersen et al. 1999, 2012).

Hydrogeologic Setting

Hydrogeologic setting provides a context for assessing potential climate-induced changes to groundwater resources. Geologic units respond differently to changes in precipitation because of differences in hydraulic conductivity, transmissivity, primary versus secondary porosity, and fracture patterns. In a study that combined aerial photography (over 50–80 years) and climate analysis, Drexler et al. (2013) showed that five fens in the Sierra Nevada (California) decreased 10 to 16 percent in area. This decrease in area occurred over decades with high mean minimum air temperature and low SWE and snowpack longevity. However, two fens in the southern Cascade Range, underlain by different geology than the Sierra Nevada, did not change in area, suggesting that the hydrogeologic setting plays an important role in mediating GDE functionality.

Several different hydrogeologic settings have been delineated in the IAP region, including igneous/metamorphic, sedimentary, karst, and unconsolidated sediments. Igneous and metamorphic rocks with low permeability and porosity, with low-volume groundwater discharges, and that are recharged only during large infrequent precipitation or snowmelt events may not be vulnerable to changes in temperature and precipitation. However, aquifers in sedimentary formations, karst formations (fig. 4.17), and unconsolidated sediments may be more sensitive to climate change because they have high permeability, high porosity, and larger volume discharges to GDEs.

Groundwater Systems in the Intermountain Adaptation Partnership Region

Middle Rockies Subregion

Located in central Idaho, the Middle Rockies subregion is underlain with predominantly igneous and volcanic rocks with carbonate (fig. 4.17) and other sedimentary rocks in the southeast. Groundwater occurs in fractured and weathered crystalline rocks and sedimentary rocks (USGS 2000). Sand and gravel aquifers are found in floodplains and terraces in the valleys. Bedrock aquifers are the only source of groundwater across much of the subregion. Igneous, metamorphic, and sedimentary rocks that underlie the mountains generally yield little water to wells. Recharge to the basin aquifer system is by precipitation that falls directly on basin floors and by snowmelt that runs off the surrounding mountains and is transported into the basins by tributary streams.

Southern Greater Yellowstone Subregion

The Southern Greater Yellowstone subregion in western Wyoming and eastern Idaho consists primarily of sedimentary rocks but also contains igneous and metamorphic rocks (USGS 2000). Aquifers include sedimentary rocks, sand, and gravel along streams and basin-fill aquifers adjacent to the mountain blocks. Groundwater occurs in pore spaces, joints, fractures, faults, and solution openings in carbonate rocks. Many basins are bounded by mountain front faults. The most important aquifers are basin-fill aquifers, but they are recharged mainly from the mountain blocks. Deposits that fill the basins are mostly alluvium derived from the weathering and erosion of consolidated rocks that underlie the mountains bordering the basins. Primary recharge areas are generally located along the mountain fronts and extend into some mountain valleys. Groundwater is obtained principally from the basin-fill deposits. Basin margin faults are likely to influence flow from the bedrock aquifers to the basin-fill aquifers by forming barriers and highly permeable pathways, depending on the fault-zone geometry.

Uintas and Wasatch Front Subregion

Consolidated rocks of Precambrian to Tertiary age, which form the Wasatch Range and other mountain ranges in the Uintas and Wasatch Front subregion, yield water chiefly through complex systems of fractures, joints, solution

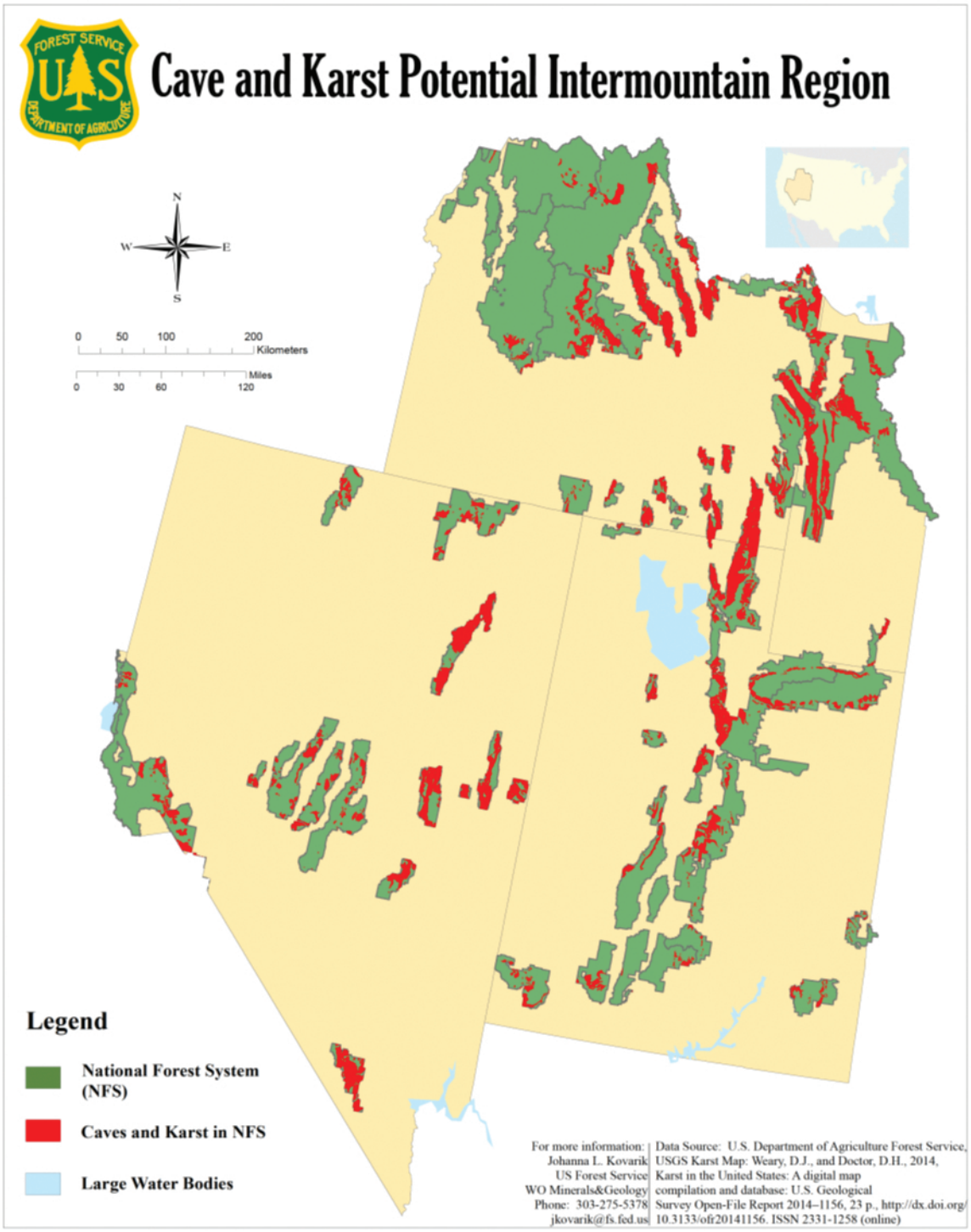


Figure 4.17—Areas with cave and karst potential composed of carbonate and minor volcanic rocks in the U.S. Forest Service Intermountain Region (from Weary and Doctor [2014]).

cavities, fault zones, and vesicles (Price 1985). These water-bearing zones, which are not present at all locations, are difficult to find and delineate. Wells in consolidated rocks also have small yields, and the depth to the saturated zone can be great. Consequently, the consolidated rocks in the Wasatch Front area are not considered to be favorable sources of water for withdrawal from wells. As a unit, however, they do absorb, store, and transmit large volumes of water to downstream aquifers. This is particularly true for carbonate units (fig. 4.17). In Utah, the aquifers in Cache Valley, the lower Bear River area, and along the Wasatch Front provide water to 84 percent of the population of Utah.

Geologic conditions vary considerably throughout the Wasatch Front area, and thus, groundwater occurrence,

movement, quality, and availability also vary. Most of the wells that obtain water from the consolidated rocks are used for domestic supply and produce only a few gallons of water per minute. However, some of the springs that discharge from these rocks (especially carbonate rocks) produce several hundred to more than 1,000 gallons per minute. Alluvial fans make up much of the valley fill near mountain fronts.

Plateaus Subregion

Colorado Plateau aquifers underlie the Plateaus subregion of eastern Utah. Principal aquifers in the Colorado Plateau include the Uinta-Animas, Mesaverde, Dakota-Glen Canyon, and Coconino-De Chelly (USGS 2000). Distribution of aquifers on the Colorado Plateau is controlled in part by the structural deformation and erosion that have occurred since deposition of the sediments composing the aquifers. Much of the land is underlain by rocks that contain aquifers capable of yielding usable quantities of water of quality suitable for most agricultural or domestic use. In general, the aquifers in the Colorado Plateau area are composed of permeable, moderately to well-consolidated sedimentary rocks. These rocks range in age from Permian to Tertiary and vary greatly in thickness, lithology, and hydraulic characteristics.

Relatively impermeable confining units separate each of the four principal aquifers in the Colorado Plateau. The two thickest units are the Mancos (underlying the Mesaverde aquifer) and Chinle-Moenkopi (underlying the Dakota-Glen Canyon aquifer). Groundwater recharge to the Uinta-Animas aquifer generally occurs at higher elevations along the margins of each basin. The Mesaverde aquifer is at or near the land surface in extensive areas of the Colorado Plateau.

Great Basin and Semi Desert Subregion

The Great Basin and Semi Desert Subregion lies within the Basin and Range Physiographic Province, which contains three principal aquifer types: (1) volcanic-rock aquifers, which are primarily tuff, rhyolite, or basalt of Tertiary age; (2) carbonate-rock aquifers (fig. 4.17), which are primarily limestones and dolomites of Mesozoic and Paleozoic age; and (3) basin-fill aquifers, which are primarily unconsolidated sand and gravel of Quaternary and Tertiary age (USGS 2000). These aquifers underlie most of Nevada, western Utah, and southern Idaho. All the precipitation that falls in the area is returned to the atmosphere by ET, and streams do not carry water to the oceans.

Fracturing in carbonate rocks (limestone and dolomite) may enable groundwater to circulate through the fractures where the water can dissolve the slightly soluble rock and enlarge and increase the size and number of pathways for water movement through the rock. Such dissolution eventually can change a relatively impermeable carbonate rock into a permeable water-yielding unit. Carbonate rocks predominate in a 20,000- to 30,000-foot thick sequence of Paleozoic and Lower Mesozoic rocks in an extensive area of southern and eastern Nevada and are present on all National Forests in the region. The location of solution-altered zones of enhanced permeability within these carbonate rocks is poorly known. Although extrusive igneous rocks (primarily basalt) can be permeable in local areas, most other types of consolidated rock are not sufficiently permeable to transmit large volumes of water, and bedrock generally forms a relatively impermeable boundary to the Basin and Range aquifers.

The groundwater flow systems of the Basin and Range area are in individual isolated basins or in two or more hydraulically connected basins. The impermeable rocks are boundaries to the flow system, and most of the groundwater flows through basin-fill deposits. If carbonate rocks underlie the basins, substantial quantities of water can flow between basins through the carbonate rocks and into the basin-fill deposits. Most recharge to the basin-fill deposits originates in the mountains as snowmelt. Major faults that cut the alluvial deposits can act as partial barriers to the movement of groundwater.

Dependence of Special Habitats on Different Water Sources

Groundwater-dependent ecosystems occur in locations with abundant growing-season water. Because precipitation is the ultimate source of water and directly influences streamflow characteristics and groundwater dynamics, it is expected that climate-induced changes in precipitation will affect riparian areas, wetlands, and GDEs. Availability of water is also influenced by physical watershed characteristics that affect infiltration and surface and hillslope runoff, including lithology, soil depth, and topography (Jencso et al. 2009).

Groundwater Recharge in Mountain Aquifers in the Western United States

Most aquifers in Western mountains are small compared to the major aquifer systems in the basins. Despite being small, these aquifers are essential in storing and transmitting groundwater that becomes recharge to the adjacent major aquifer systems. Altered recharge caused by climate change will translate into altered mountain aquifer storage and discharge, which will, in turn, directly influence recharge to downgradient aquifers and stream base flows. Between 61 and 93 percent of diffuse mountain catchment recharge becomes streamflow available for downstream aquifer recharge by stream loss (Meixner et al. 2016).

Snowmelt is likely to contribute the majority of recharge in most mountainous regions of the western United States, either because most of the precipitation falls as snow, or snowmelt infiltrates below the root zone more effectively than rainwater (Earman et al. 2006). Snowmelt can compose a large fraction of recharge because much of the water released from the snowpack occurs over a prolonged period in early spring when ET is low (Ajami et al. 2012; Earman et al. 2006). Consequently, mountain recharge is sensitive

to the climatic shifts that result in changes in SWE noted earlier in the chapter.

Recharge in many mountainous areas is permeability limited rather than recharge limited where thin soils overlie low-permeability crystalline bedrock (Flint et al. 2008). Lower maximum annual SWE in these areas may decrease overland flow of snowmelt to streams, but has little influence on recharge because spring snowmelt substantially exceeds the unsaturated zone storage capacity (Blankinship et al. 2014). Conversely, bedrock permeability in karst areas is so high that most snowmelt and rainfall infiltrates into the porous rock, flows in conduits, and is discharged to the surface as springs or discharges directly to fill aquifers.

Higher minimum temperatures will reduce the longevity of snowpack, and decrease the length of time aquifer recharge can occur, potentially leading to less groundwater recharge. Some watersheds will be shifting from snow-dominated to rain-dominated, which may result in declines in groundwater recharge (Earman and Dettinger 2011; Safeeq et al. 2013, 2014). Recharge could also increase in these areas as a result of a more gradual release of water from snowpack from enhanced winter melting (Byrne et al. 2014; Musselman et al. 2017). Projecting future mountain recharge requires knowledge of groundwater flow systems that is generally unavailable.

In summary, recharge is likely to decrease in the southern IAP region, but changes in other parts of the region are uncertain because of limited understanding of mountain recharge processes and groundwater flow in mountains (Meixner et al. 2016). However, there are existing approaches (e.g., Safeeq et al. 2014) that can be used to develop sensitivity maps from available information about geology, stream recession behavior, and other factors. These approaches could be used to evaluate sensitivities for future mountain recharge in the IAP.

Current Resource Conditions

Steep elevation gradients, varied bedrock, and glacial landforms influence the distribution, characteristics, and water chemistry of groundwater-dependent features. Existing information on the condition and distribution of GDEs in National Forests of the IAP region is limited. Here, we rely on data compiled by the Spring Stewardship Institute, the National Hydrography Dataset (USGS 2017), and the National Wetlands Inventory (NWI) (USFWS n.d.) to assess the current condition of springs, wetlands, and GDEs in the region.

Springs are usually small, averaging less than 0.5 acre, with few spring habitat patches larger than 2 acres (Kreamer et al. 2015). Thus, springs fall below the scale of most landscape mapping efforts and are therefore neglected in remote sensing, soil, and floristic mapping. Recently, National Forests in the U.S. Forest Service Intermountain Region and the Spring Stewardship Institute have begun to map springs and other GDEs, but the known occurrence of these are limited, and many more springs certainly exist (table 4.1).

Table 4.1—Area of palustrine emergent wetlands with saturated water regime and number of mapped springs in the U.S. Forest Service Intermountain Region. Differences in wetland area reflect different mapping accuracy and limitation among national forests; wetlands less than 1 to 3 acres are generally not included (USFWS n.d.).

National Forest	Wetland area	Springs
	Acres	*Number*
Ashley	18,388	426
Boise	92	442
Bridger-Teton	3,397	140
Caribou-Targhee	262	1,467
Dixie	1,048	652
Fishlake	1,259	622
Humboldt-Toiyabe	1,275	4,286
Manti-La Sal	1,894	481
Payette	218	155
Salmon-Challis	205	692
Sawtooth	224	1,211
Uinta-Wasatch-Cache	19,269	917
Total	47,530	11,491

Springs play a key role as groundwater discharge zones that deliver cool water to warming streams, support late-season streamflows in summer, and deliver relatively warm water during winter months (Lawrence et al. 2014; Winter 2007). Most streams and rivers in the region are at least partially groundwater dependent (Santhi et al. 2008). Locations of groundwater discharge to streams have been identified by using remote sensing (Torgersen et al. 1999) and field techniques (Torgersen et al. 2012), but have not been systematically mapped.

Wetlands can be identified by targeting palustrine/emergent wetlands with a saturated water regime (Cowardin et al. 1979) in the NWI database (table 4.1). To ascertain whether these wetlands are indeed fens, each wetland would require a field visit to determine if it is supported (at least in part) by groundwater and is peat-forming (Chadde et al. 1998; USDA FS 2012a, b). Fens occupy a small portion of the landscape, but contribute substantially to biodiversity of plants and animals (Blevins and Aldous 2011). In an otherwise arid region, perennially saturated fens are critical habitat for invertebrate and amphibian species. Although not explicitly differentiated as fen vegetation, several herbaceous-dominated plant associations frequently occur in fens.

Since 2008, GDEs, mostly springs and fens, have been inventoried and documented in National Forests by using draft and final versions of the Groundwater-Dependent Ecosystems Level I and Level II inventory methods

(USDA FS 2012a, b). The Level I guide (USDA FS 2012a) describes basic methods for assessment of GDEs within a given area (e.g., National Forest, ranger district, specific project area) and is intended to qualitatively document the location, size, and basic characteristics of each GDE site. The Level II guide presents more detailed inventory (USDA FS 2012b) in addition to protocols for more comprehensive characterization of the vegetation, hydrology, geology, and soils at a given site.

Inventories have been conducted in Ashley, Caribou-Targhee, Dixie, Humboldt-Toiyabe, Manti-La Sal, and Sawtooth National Forests. Inventories in Humboldt-Toiyabe National Forest targeted springs with terrestrial and aquatic flora and fauna values. In the GDE Level I protocol (USDA FS 2012a), a series of 25 management indicator statements assist in identifying potential concerns and needs for management action based on observations recorded during field inventories. Three of the most important indicators are aquifer functionality, soil integrity, and vegetation composition. Most inventories targeted sites where proposals for water development could be damaging, or portions of grazing allotments and watersheds with specific management concerns. Findings of inventories commonly show notable resource impacts through water diversion, soil disturbance, and effects of livestock on vegetation.

Potential Climate Change Effects on Groundwater and Groundwater-Dependent Ecosystems

Groundwater recharge has been examined in only a few locations (Tague and Grant 2009), and little is known about groundwater recharge processes in many watersheds, including those that may be shifting from snow-dominated to more rain-dominated hydrological regimes (Safeeq et al. 2013, 2014). Depending on elevation and hydrogeologic setting, slowly infiltrating precipitation that includes both rain and snow may recharge some groundwater aquifers as effectively as rapid, seasonal snowmelt runoff. Although rain-on-snow zones are expected to shift upwards in elevation, the influence of these shifts on groundwater recharge is unknown.

Small, unconfined aquifers (especially surficial and shallow aquifers) are more likely to have renewable groundwater on shorter time scales and may respond rapidly to changes in climate (Healy and Cook 2002; Lee et al. 2006; Sophocleous 2002; Winter 1999). Larger, deeper, and confined aquifers are more likely to have nonrenewable groundwater, may be less sensitive to the direct effects of climate change, and are projected to have a slower response (Wada et al. 2012).

Groundwater storage can moderate surface water response to precipitation changes (Maxwell and Kollet 2008), and changes to groundwater levels can alter the interaction between groundwater and surface water systems (Hanson et al. 2012). Climate-induced changes in connectivity between groundwater and surface water could directly affect stream base flows and associated wetlands and other GDEs (Earman and Dettinger 2011; Kløve et al. 2012; Tujchneider et al. 2012). Short flow-path groundwater systems, including many that provide base flow to headwater streams, could change substantially in timing of discharge in response to changes in seasonality of recharge (Waibel et al. 2013). In contrast, regional-scale aquifer systems, with flow paths on the order of tens of miles, are much less affected by shifts in seasonality of recharge. These effects may be highly variable, depending on local hydrogeology.

Altered groundwater levels in wetlands can reduce groundwater inflow, leading to lower water table levels and altered wetland water balances. For local and intermediate-scale systems, the spatial extent of some GDEs is likely to contract in response to decreasing surface water and groundwater and increasing temperatures. Changes in groundwater and surface water will also vary depending on location within the watershed, as well as future land use.

Effects of changing climate on the ecology of GDEs will depend on changes in groundwater levels and recharge rates, as influenced by the size and position of groundwater aquifers (Aldous et al. 2015). GDEs supported by small, local groundwater systems tend to exhibit more variation in temperature and nutrient concentrations than regional systems (Bertrand et al. 2012). It is likely that larger systems will be more resilient to climate change.

Freshwater springs depend on continuous discharge of groundwater, forming ecotones between subsurface-surface water and aquatic-terrestrial environments (Ward and Tockner 2001). Springs and springbrooks support locally unique biological communities (Barquin and Death 2006). However, climate-induced changes in recharge may cause decreased summer flows with possible drying, as well as increased winter flow and inundation of biological communities (Green et al. 2011).

Many biogeochemical processes are temperature dependent, so climate-induced changes in groundwater temperature may negatively affect the quality of groundwater, and, in turn, influence aquatic communities (Figura et al. 2011). However, because the thermal regime of groundwater systems is less dependent on air temperature patterns than surface waters, the effects of rising air temperatures are likely to be less pronounced in springs and other GDEs.

Peat-accumulating processes in fens will be influenced by increasing temperatures and local and regional changes in hydrological regime. Reduced groundwater levels tend to promote soil aeration and organic matter oxidation. Generation and maintenance of peat soils over time depend on stable hydrological conditions. In recent studies of peatlands exposed to groundwater lowering, responses such as soil cracking, peat subsidence, and secondary changes in water flow and storage patterns have been observed (Kværner and Snilsberg 2011). Wetland plant species can respond to even slight changes in water table elevation (Magee and Kentula 2005; Shipley et al. 1991; Vitt and Slack 1984), and shifts in composition of vascular and bryophyte species could occur with lowered water tables.

Some riparian ecosystems depend on the presence of flowing water, although streamflow may not be perennial along all stream segments and can vary considerably with season, physical features of the watershed, and water source. Depending on physical characteristics of a given stream segment, the volume of streamflow can also drive seasonal changes in water table elevation of the adjacent riparian area (Jencso et al. 2011). These hydrological and fluvial processes and resulting geomorphic surfaces are essential for the persistence of riparian vegetation (Naiman et al. 2005). According to long-term daily flow data, different streams in the region are supported by perennial runoff, snow plus rain, and stable groundwater levels (Poff 1996).

Changes in water table elevations and streamflow volumes may affect riparian areas and their plant communities (Jencso et al. 2009; Naiman et al. 2005). Examples of changes in flow systems are decreased summer base flows (see earlier *Hydrological Processes* section), lower riparian water table elevations, and reduced hydrological connectivity between uplands and riparian areas (Jencso et al. 2009, 2011). Streamflow volume along gaining reaches increases with the inflow of groundwater to the channel. Stream water can also drain from the channel bed and banks to the groundwater system (losing reaches), resulting in a loss of downstream surface flow volume (Winter et al. 1996). Gaining and losing stream reaches result in different aquatic communities in the channels and different riparian plant communities on the floodplains. The extent to which gaining or losing characteristics of specific reaches may change in response to climate-induced changes in precipitation, streamflow characteristics, and groundwater discharge is unknown.

In wetlands and riparian ecosystems, hydrological variables are consistently the strongest predictors of plant species distributions (e.g., Cooper and Merritt 2012). Current understanding of the water sources used by riparian and wetland plants is limited to a few highly valued or highly invasive species (mostly woody). However, riparian and wetland plant species use water from multiple sources (surface water, soil water, groundwater), depending on life stage and season (Busch and Smith 1995; Cooper et al. 1999; Goslee et al. 1997). In assessing the vulnerability of riparian and wetland species to climate-induced changes in streamflow or groundwater, the availability of water at all life stages must be considered, from plant recruitment and establishment, to reproducing adults, to persistence at later life stages (Cooper and Merritt 2012).

Climate-induced changes in precipitation, drought, and streamflow will influence the distribution of riparian vegetation via changes in local hydrological regimes, especially if summer base flows decrease. If water table elevation can be assumed to be in equilibrium with water levels in the stream, reduced base flows could result in lower riparian water table elevations and subsequent drying of streamside areas, particularly in wide valley bottoms. Wetland and riparian plant communities will respond to climate-induced changes in hydrological variables differently as a function of species composition (Merritt et al. 2010; Weltzin et al. 2000).

Although ET is not expected to increase substantially from the landscape generally as outlined earlier in the chapter, water supplies around riparian areas and GDEs are consistently high. Riparian areas and GDEs compose a very small fraction of the landscape, so they affect the land-surface energy balance only very slightly. Consequently, the increased net radiation and atmospheric demand will induce higher ET rates in riparian areas and GDEs, and this higher ET rate will not substantially feed back into the regional energy balance. Higher ET will result in drying in these ecosystems, potentially stressing characteristic plant species and resulting in compositional shifts in vegetation. If plant cover is reduced in riparian areas, erosion may increase.

Soil Resources

The potential effects of climate change on soils are multifaceted; changes in soil physical, biological, and chemical processes can occur with changing climate, which may in turn affect other processes such as carbon cycling and vegetation growth. Soil responses to climate change will vary by geographic location and are determined by the interactions of soil, vegetation, and the degree of management intervention. The following sections provide a summary of potential effects of climate change on soils in the IAP region.

Soil Temperature and Moisture

Soil temperature and moisture are the primary drivers of change for all soil processes. Potential changes to these soil properties with climate change have been well studied, but where and when the changes may occur is difficult to predict. The magnitude of projected change is variable depending on existing soil resources and existing climate. The properties and processes of soils are not independent, and a change to one soil property will affect other soil properties and processes. For example, changes to soil temperature and moisture will affect carbon and nitrogen cycles, which can in turn affect soil properties such as water holding capacity, cation exchange capacity, soil nutrient content, and aggregate stability (Brevik 2013).

An increase in soil temperature will generally produce an increase in soil biological activity and soil respiration. However, the rate and magnitude of change are dependent on soil moisture (Kardol et al. 2010). In the current semiarid soils of the IAP region, an increase in soil temperature without an increase in soil moisture is likely to create soils that have reduced biological activity and less potential to store carbon. If soil moisture is limiting with increased soil temperatures, the soils may become a net source of carbon until equilibrium is reached. However, an increase in soil temperature could be offset by an increase in water available for biological activity and vegetation production, resulting in little change or a possible increase in carbon storage. In

the colder and wetter areas of the IAP region, an increase in soil temperature may lead to longer growing seasons if soil moisture is not limiting (Kurylyk et al. 2014). Thus, the timing and type of moisture will determine soil biological activity, respiration, and ultimately vegetation supported by the soil.

Changes to soils with climate change will not be uniform across the IAP region. Soil responses to temperature and moisture are highly dependent on the soil parent material. Soils derived from coarse-textured granitic soils will transfer heat more efficiently downward into the soil profile than fine-textured limestone soils. The heat transfer downward can affect soil processes and even groundwater temperatures, and it could ultimately affect surface water temperatures where groundwater is the source for surface water. Fine-textured soils, which are capable of storing water longer in the soil profile, will generally have a higher buffering capability to changes in soil temperature and moisture.

The timing of soil moisture can also affect soil erosional processes. A shift away from winter precipitation as snow to greater amounts of rain and more intense rain storms can generate a higher frequency of runoff and erosional process from disturbance events such as fire (Litschert et al. 2014). Runoff from extreme rain events could increase for shallow soils with little capacity to store water.

Soil Carbon and Nitrogen

Soils are a major component of carbon and nitrogen cycles. Changes in soil temperature and moisture will affect carbon and nitrogen cycles. Management of soil organic matter can affect both of these cycles at local and global scales. The greenhouse gases carbon dioxide, methane, and nitrous oxide are regulated to some extent by the soil organic matter. Soils provide both a source and sink for carbon and nitrogen and the greenhouse gases associated with carbon and nitrogen. Changes to the carbon and nitrogen cycles may include an increase or reduction in cycling rates or storage of carbon and nitrogen. Soil temperature and moisture drive the type of change that will occur as they affect microbial activity and plant composition.

Soil Carbon Pool

Soil organic carbon (SOC) is derived from soil organic matter (SOM). The SOM is composed of plant and animal residues, cells and tissues of soil organisms, and substances produced by decomposing organisms. The SOC is the carbon component of SOM. Generally, about 58 percent of SOM is SOC by weight. Most soil orders within the IAP region have a near-surface SOC content (by mass) of 0.5 percent (for the hotter and drier areas) to 8 percent (for the cooler and wetter areas; Histosols excluded) (Brady 1999). Hereafter, we use SOC to represent both SOC and SOM, as these properties are likely to have similar response to climate change.

Soil organic carbon may be the best indicator and contributor to soil health because SOC supports many soil processes and functions. These include providing nutrients for plants, binding soil particles together and thereby maintaining structure, providing an energy source for microbes, increasing water infiltration and retention, and providing cation-anion exchange for retention of ions and nutrients. Climate change will affect SOC and ultimately the functions and processes supported by SOC.

Globally, SOC may contain more than three times as much carbon as is found in the atmosphere or terrestrial vegetation. In forest ecosystems, SOC may be as much as 80 percent of the total terrestrial carbon pool, and in nonforest ecosystems, SOC may be as much as 95 percent of the total terrestrial carbon pool (Meyer 2012). Soils can store and release carbon at the same time. If soils store more carbon than they release back to the atmosphere, the soil is a carbon sink. If soils are releasing more carbon than what is being stored, they become a carbon source. Therefore, the management of SOC is critical to the management of atmospheric carbon concentrations (Woodall et al. 2015).

Carbon is stored in soils in organic or inorganic forms. Soil organic carbon originates from carbon fixation during photosynthesis and microbial decomposition (Thomey et al. 2014). Inorganic carbon (IC) is in rocks and minerals. An example of IC is limestone, or calcium carbonate. The stable IC is released slowly through weathering or anthropogenic manipulation, such as mining and conversion to other chemical compounds. Although IC is slow to change, it represents a large portion of stored carbon in many ecosystems, such as drier shrublands and grasslands. Many of the drier rangeland soils include carbonate horizons within the soil profile. Climatic changes may affect the release of IC. Higher atmospheric carbon dioxide concentrations and a warmer and wetter climate will increase weathering of rocks and acidification in the carbonate layers in the soil, which will release greater amounts of carbon into the active carbon cycle (USDOE 2014).

Different soils have different capacity to store carbon. The differences are related to parent material, existing climate, and terrestrial ecological community types. Shrublands have a higher percentage of SOC stored lower in the soil profile (below 3 feet). Forests have most of their SOC in the first 3 feet of the soil. Changes to vegetative composition can affect long-term carbon storage. A shift from shrublands to annual grasslands will eventually move the bulk of carbon from deep in the soil profile to the upper 8 inches (USDOE 2014). This may be happening with conversions of shrublands to cheatgrass (*Bromus tectorum*) in the IAP region. This process is very slow, however, and respiration of carbon deep in the soil profile is much slower than near the soil surface. A shift from shrublands to forest will increase near-surface carbon pools as a result of litter addition and deeper reserves being tapped by roots for the production of biomass (Nave et al. 2013).

Soils have SOC storage limits set by soil physical and chemical composition as well as microbial and plant

community types, all of which are determined by soil moisture and temperature. Most soils in the IAP region are at SOC saturation for the existing climate (Woodall et al. 2015). Soils that are degraded or furthest from potential SOC saturation have the greatest ability to sequester additional carbon. These are generally areas with vegetation that has been altered for long periods of time, such as agricultural fields. Most of the soils in the IAP region could sequester additional carbon if soil temperatures decrease and soil moisture increases. This is particularly true with the lower-elevation soils. However, soil moisture may not increase in a warming climate.

Soil Physical Properties Related to Carbon and Climate Change

Changes to SOC can alter several soil properties, including soil structure, bulk density, and soil porosity (Pal Singh et al. 2011). These soil properties affect water infiltration, rooting depth, soil erosional losses, and water holding capacity. A reduction in SOC will change soil structure through reducing the bonds between soil particles and the microbial "glue" that helps hold soil particles together. This can lead to less resistance of the soil to erosive forces of wind and water. A change in soil structure can also lead to changes in soil porosity and bulk density. Soil porosity and pore size distribution are important for soil water management and maintaining release rates of greenhouse gases (carbon dioxide, methane, and nitrous oxide) within the soil. A loss of macropores with reduction in SOC could lead to slower water infiltration rates, increased runoff, decreased nutrient cycling, reduced plant growth (above and below ground), and poor aeration of the soil, resulting in a decreased capacity to oxidize greenhouse gases, specifically methane. A reduction in SOC also leads to increased risk of surface compaction with management activities through an increase in surface bulk density of soils. Surface compaction restricts water infiltration and increases surface runoff.

Although the changes to soil physical properties with loss of SOC are highly variable across the landscape, they do provide potential indicators that can be used to prioritize management in a changing climate; the soil types where current management is already having negative effects on soil physical properties could be the soil types that are prioritized for climate change adaptation actions. For example, areas where excessive runoff and soil loss have occurred because of grazing management may be a priority. These areas would be expected to have a higher risk of soil quality loss under a warmer and drier climate with reduction in plant growth and SOC development.

Soil Nitrogen

SOM typically contains about 5 percent nitrogen; therefore, the distribution of soil nitrogen closely parallels that of SOM (Brady 1999). Nitrogen cycles are closely tied to carbon cycles, although they may respond differently to changes in climate. On average, nitrogen fixation occurs at a rate of about 9 pounds per acre for forested sites and 13 pounds per acre for grasslands (Brady 1999). Forest soils may contain 15 times as much nitrogen as the standing vegetation, including roots (Brady 1999). About 29 to 56 percent of the soil nitrogen pool is found in the upper 4 inches (Page-Dumroese and Jurgensen 2006), making the soil nitrogen pool highly susceptible to loss from erosion.

Although most of the nitrogen in terrestrial systems is found within the soil, the mineralization of nitrogen is required to provide a form of nitrogen that plants can utilize. Nitrogen mineralization occurs through decomposition of organic material by soil micro-organisms. Warmer soil temperatures increase decomposition by microbial activity, thus increasing nitrogen mineralization. However, soil moisture may have a greater effect on net nitrogen mineralization through changes in the form of nitrogen (Emmett et al. 2004).

Nitrogen could be limiting to plants in some soils of the IAP region even if conditions for plant growth improve with changes in climate. If plant growth increases with increased atmospheric carbon dioxide, then organisms that decompose residual plant material will need more nitrogen to survive. If nitrogen is tied up by soil microflora and microfauna, the nitrogen would be unavailable to plants. Thus, any positive effects of increased atmospheric carbon dioxide may be offset by the reduction in available soil nitrogen (Brevik 2013), particularly on nitrogen-limited soils. In the IAP region, those areas most susceptible to reduction in nitrogen are forested areas on soils with coarse-textured parent material.

Soil Biological Activity

Soil organisms perform many functions in the soil, including decomposition and nutrient cycling. As with other soil processes, the soil biology is affected by other soil processes and the inherent soil composition and climate. Thus, the effects of climate change on soil biology will be variable. Some models project that a warming of the soil will create greater microbial activity, resulting in more carbon being released to the atmosphere because of increased decomposition (Kardol et al. 2010). Other models suggest that a warming of the soil will result in lower microbial growth and less carbon released through respiration (Wieder et al. 2013). In the warm and dry desert ecosystems, such as those in the Great Basin and Semi Desert and Intermountain Semi Desert subregions, the effects of soil warming are expected to increase microbial activity and carbon dioxide released to the atmosphere (Thomey et al. 2014). In cooler and wetter ecosystems, projections are mixed (Steinweg et al. 2013).

Vegetation composition can affect soil biology, soil processes, and soil response to climate change. Some soil organisms prefer specific plant types, and plant diversity increases soil biological diversity. Greater biodiversity in soils is likely to increase soil resilience to climate change (Kardol et al. 2010).

Soil Chemical Properties

Potential effects of climate change on soil chemical properties are linked to other biological and physical changes in the soil, all of which are driven by soil temperature and moisture. Soil pH is closely tied to organic matter, parent material, and soil moisture. A warming climate with additional or similar precipitation will lower soil pH (Pal Singh et al. 2011). A warming climate with less precipitation may increase pH on some higher-elevation soils and have little effect on existing lower-elevation high-pH soils (Pal Singh et al. 2011).

In areas that are expected to experience increased drought, such as drier shrubland and grassland systems, an increase in the accumulation of carbonates and salts in the soil profile is expected. This would result in a salinization of the soil and have significant effects on vegetation composition. In wetter areas of the IAP region, an increase in soil temperatures may cause an increase in acidification from the decomposition of organic matter. This could change species composition and diversity to favor species more adapted to acidic soils.

Cation exchange capacity (CEC) is the ability of the soil to retain nutrients such as calcium, potassium, and magnesium and make them available to the soil solution and plants. It also provides the capacity to retain and immobilize some cations that may be toxic to soil microbes and plants in high amounts, such as aluminum and manganese. The CEC is generally low in coarse-textured soils or soils with low amounts of SOC. Soils with a subsurface argillic horizon (higher CEC) are likely to be able to moderate a shift in nutrient exchange, specifically a loss of SOC and other major soil nutrients requiring cation exchange sites. An increase in soil temperatures could lead to a reduction of SOC and the CEC of soils. A reduction in CEC would result in loss of base cations in the soil that are released to groundwater and surface water (Pal Singh et al. 2011).

Managing soil resources for optimum SOC will limit the effects of climate change to the CEC. Areas with low SOC are the most susceptible to CEC changes in soils due to climate change, as they have poor buffering capabilities. In the IAP region, these are primarily drier rangeland soils, particularly those that have been vegetated for many decades with annual shallow-rooted grasses.

Weathering of rock and erosion of soil is a continuous process. Changes to rainfall and wind as well as changes to chemical, physical, and biological properties of soils can affect the type, amount, and rates of runoff and erosion of soil. More precipitation may not have any long-term effects on erosion and runoff, because vegetation will respond positively to increases in available soil moisture. However, an increase in the number of intense rain events may result in an increasing rate of erosion. A reduction in the amount of precipitation generally reduces the rate of erosion. However, lower vegetation cover in response to low soil moisture may result in increased rates of erosion from wind and water. Areas of the IAP region that are most susceptible to increased erosion are the lower-elevation shrublands and grasslands, where a warmer and drier climate will reduce the potential for vegetative cover.

Regional-Scale Soil Vulnerability Ratings

The potential magnitude of change to soil resources in a changing climate is extremely variable because of the heterogeneity of soil types and their potential response across the landscape. Identifying the degree of potential change and risk to soils spatially across a landscape is difficult, even with detailed soils data. However, general projections can be made across large landscapes by using different soil and landscape attributes, particularly vegetation composition and productivity. The following section provides a general rating of soil vulnerability to climate change across the IAP region. The rating is based on general soil characteristics and data from Natural Resources Conservation Service STATSGO datasets (NRCS 2017).

Several assumptions were used to develop a regionwide rating of soil vulnerability to climate change. These include:

- The existing climate will generally be warmer and drier in the future across all subregions in the IAP.
- Soils that are currently capable of holding water longer and deeper within the soil profile have a greater buffering capacity to change.
- Soils that are currently cooler and wetter, or have higher SOC, are less susceptible to climate change.
- Many soil properties will change in a changing climate.
- The ratings of soil vulnerability to climate change are based on general surrogates of landscape and soil conditions. Data on detailed soil properties, such as available water holding capacity, were not available across the region to make predictions.
- Soil properties used to derive a vulnerability map were soil temperature and moisture regimes (combined into subclasses), and SOC from depths of 0 to 12 inches.
- Soil polygons contain many soil components. The components were combined by dominant condition for soil temperature and moisture subclasses and by weighted averages for SOC.

A combined STATSGO soil map was made for all lands within the IAP region. Soil temperature and moisture classes were determined for each polygon based on the dominant condition within the polygon. The soil temperature and moisture classes were further combined according to soil taxonomy to create 44 different subclasses. These subclasses were combined qualitatively based on common soil temperature and moisture breaks to create four class ratings of low, moderate, high, and very high susceptibility to a warming and drying climate. The same STATSGO soil map was used to create a SOC map for the 0 to 4 inches soil depth (the database was poorly populated for depths deeper than 4 inches). Each polygon received a value for SOC. The

Table 4.2—Final rating classes for soil vulnerability to climate change, based on a combination of soil temperature/moisture rating and soil organic carbon rating.

	Organic carbon rating = Low	Organic carbon rating = Moderate	Organic carbon rating = High	Organic carbon rating = Very high
Temperature/moisture rating = Low	Low	Low	Moderate	High
Temperature/moisture rating = Moderate	Low	Moderate	Moderate	High
Temperature/moisture rating = High	Moderate	Moderate	High	Very High
Temperature/moisture rating = Very high	High	High	Very High	Very High

polygon values were assigned to one of four classes (low, moderate, high, very high) of SOC such that all classes contained the same number of polygons in each class. This method of creating general SOC classes was chosen because threshold class values for SOC are not available.

The ratings for soil temperature and moisture were then combined with the ratings for SOC into a four-class rating of soil vulnerability to climate change. A simple matrix was used to determine the final rating (table 4.2). The final soil vulnerability rating was applied to each polygon, and a general soil vulnerability map to climate change was developed (fig. 4.18). The map represents soils that may or may not be capable of sustaining existing ecosystems in a changing climate. The map suggests that soils at higher elevations and deeper soils are not as susceptible to climate change as the soils in warmer and drier areas. But this does not mean soils will not change in wetter or cooler climates or in locations high in SOC.

The regional-scale soil vulnerability map is a coarse estimation of potential change to soils with climate change. Local data and information are needed to estimate vulnerability at a local scale. Other soil properties that should be considered in creating a local map include: available water holding capacity, soil depth, hydrological soil group, erodibility (K) factor, soil texture, parent material, SOC, and calcium carbonate content (inorganic carbon), as well as vegetation type, geology, slope, aspect, and elevation. An example of how to create a soil vulnerability rating at a forest-project scale is given next.

Example Project-Scale Assessment of Soil Vulnerability to Climate Change

At the individual forest level, local soils information could be utilized to create maps of soil vulnerability to climate change at the project, watershed, or landscape scale. The Uinta-Wasatch-Cache National Forest used available soil data and applied a soil vulnerability assessment to a watershed-scale project during the project planning stage. The assessment was used initially to identify potential projects that would help create more resilient ecosystems and then to stratify fieldwork. Vegetation manipulation is one example of a specific adaptive management strategy to maintain or enhance soil health. Quaking aspen (*Populus tremuloides*) and pinyon-juniper vegetative communities were examined for this example.

A soil map was created for two watersheds within a project boundary. Vegetation and geology layers were added, as well as a digital elevation model to create slope and aspect. Soil available water holding capacity, soil depth to a restrictive layer, hydrological group, parent material, and soil temperature and moisture regimes were included in a matrix and rated for each soil type to derive vulnerability to climate change. Using soil indicators, along with vegetation and geology layers, we can estimate nutrient content, SOM, and how well a soil retains moisture. These estimates were used to assign a rating for soil vulnerability to climate change (table 4.3). The higher the point value rating, the more resilient and resistant the soil resources are to the effects of a warmer and drier climate. This information was added to the soil attribute table in a geographic information system (GIS). Potential focus areas for fieldwork verification and for recommended vegetation projects to meet desired conditions were identified by creating intersects for vegetation attributes of interest. An example of an interpretive map is shown in figure 4.19.

The value ranges or ratings will be used to focus attention on soil resources that best meet vegetation management objectives. Soils that are more suitable or resistant to change (less vulnerable to climate change) are those expected to better maintain soil moisture and nutrient conditions favorable for the vegetative communities present. These soils will be areas of opportunity for vegetation management designed to sustain existing vegetation community types.

Conclusions from the soil vulnerability analysis include:

1. Sustaining aspen vegetative communities within the project area will be one of the objectives of the project. Vegetative treatments will be implemented to promote aspen retention and increase diversity of aspen age classes. The climate change soil vulnerability rating was determined for each of the aspen polygons in the treatment area. An examination of the rating criteria showed some aspen stands with

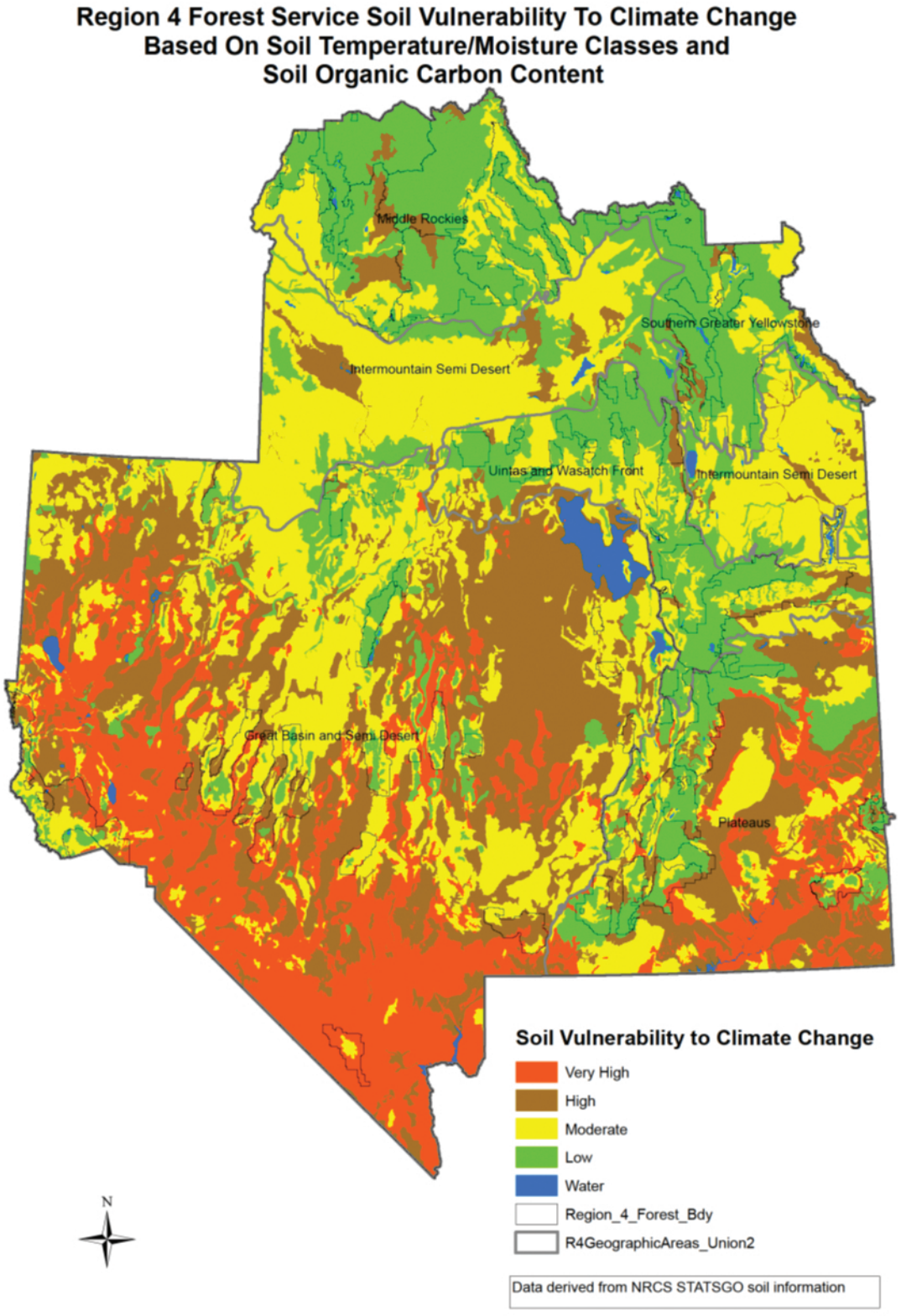


Figure 4.18—Soil vulnerability to climate change in the Intermountain Adaptation Partnership region, based on ratings in table 4.2 (combination of soil temperature/moisture and soil organic carbon).

Table 4.3—Forest-level soil vulnerability indicators and ratings.

Water holding capacity		Soil depth		Hydrological group		Aspect		Parent material		Soil temperature/ moisture regime		Point value range	Suitability rating	Vulnerability rating
Value[a]	*Pts*[b]	*Inches*	*Pts*	*Group*	*Pts*	*Group*	*Pts*	*Texture*	*Pts*	*Value*[a]	*Pts*			
VH	12	60+	12	A	9	N	9	Fine	9	L	9	46–60	VH	VL
H	9	40–60	9	B	6	E	6	Medium	6	M	6	33–45	H	L
M	6	20–40	6	C	3	W	3	Coarse	3	H	3	20–32	M	M
L	3	11–20	3	D	0	S	0			VH	0	7–19	L	H
VL	0	0–10	0									0–6	VL	VH

[a] VH = very high, H = high, M = moderate, L = low, VL = very low.
[b] Pts = points.

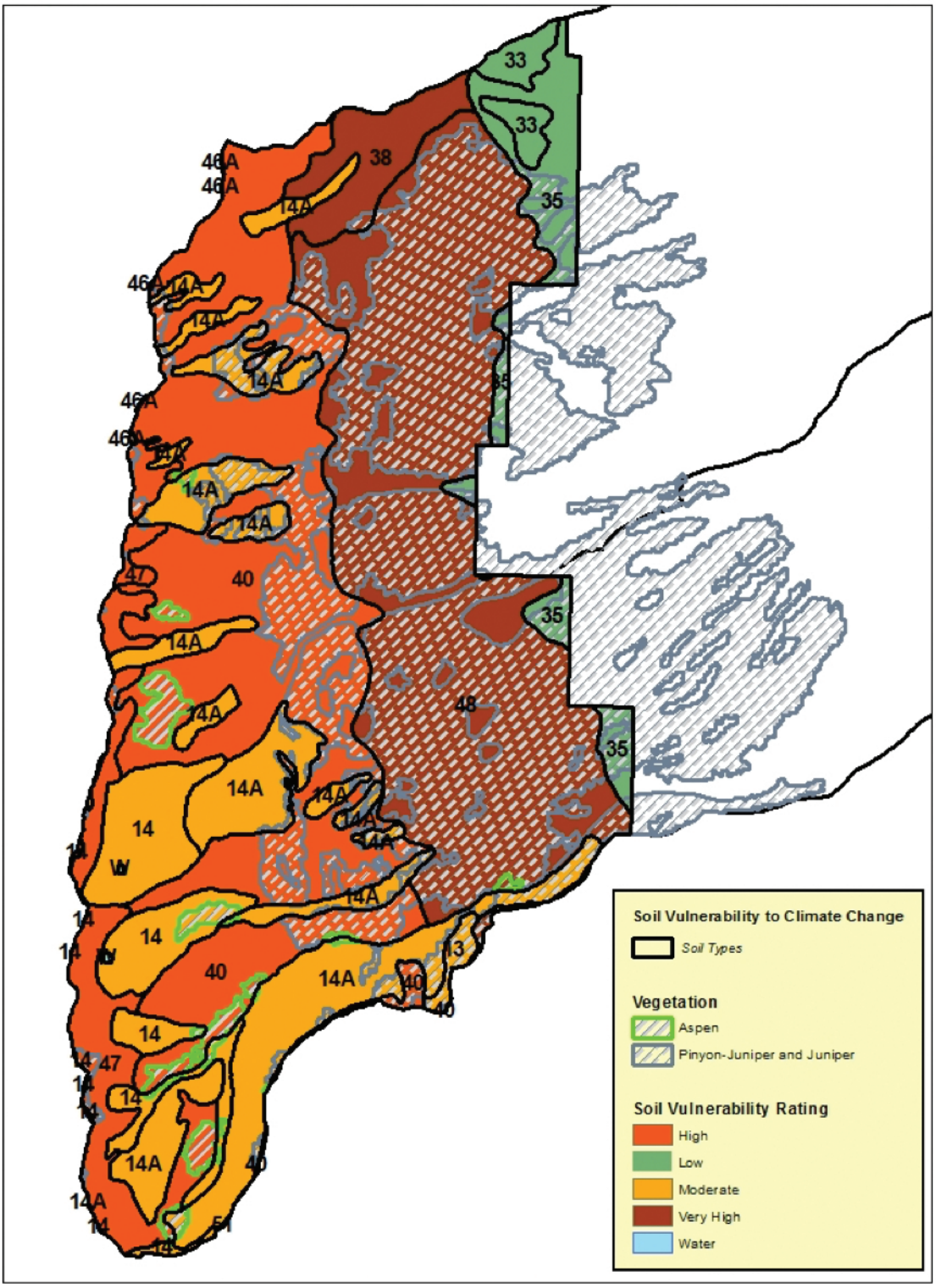


Figure 4.19—Example of soil vulnerability to climate change at a small spatial scale, based on ratings in table 4.2 (combination of soil temperature/moisture and soil organic carbon). Numbers indicate soil series mapping units (not discussed here).

a very high to high vulnerability rating and some with a moderate vulnerability rating. Aspen stands with a moderate vulnerability rating will be areas for recommended project work to maintain aspen community types. The aspen stands located within the high and very high vulnerability areas will not be recommended for aspen regeneration treatments unless a site visit indicates available water could be sustained in a warmer, drier climate.

2. Pinyon-juniper encroachment into shrublands and grasslands is occurring because of a lack of fire. Removal of the pinyon and juniper is recommended in areas that currently have mountain big sagebrush in the understory, with the objective of increasing sagebrush cover. Removal would be accomplished through mechanical treatments (mastication) and lop-and-scatter. Very high to low soil vulnerability ratings were found across the project area where pinyon-juniper exists. Soils that were rated as very high to high in vulnerability are areas with shallow soils and low available water holding capacity. Pinyon-juniper may decrease in dominance in these areas with a reduction in available soil water due to climate change. Removal of pinyon and juniper may make these areas more susceptible to invasion by nonnative species such as cheatgrass. In areas rated as having low to moderate vulnerability to climate change, the ability to manage for restoration of shrublands may be higher as soils are expected to retain soil moisture longer. Site visits will be used to verify potential for management.

Summary and Conclusions

Climate change will alter fundamental physical processes in the IAP region, including hydrological processes and soil processes. Changes in physical processes will in turn affect biological processes, including soil microbial processes and vegetation growth and development. These physical and biological effects of climate change are complex and will be highly variable across the IAP region.

Warming temperatures will reduce snowpack accumulation and advance snowmelt timing in the region. Despite mixed signals from precipitation and temperature changes in the historical record, future temperature changes are expected to be higher than historical temperature trends, and future precipitation declines are expected to be less pronounced—and increased precipitation is possible. Earlier streamflow center of timing is expected over much of the region, and summer low flows are expected to be lower. Total water yields may decrease due to increased ET, but precipitation amounts are uncertain. Increasing precipitation could outweigh ET effects on total water yields, but decreasing precipitation could substantially exacerbate declines in annual water yield and low flows. The frequency and extent of midwinter flooding are expected to increase. Flood magnitudes are also expected to increase because rain-on-snow-driven peakflows will become more common.

Places with seasonally intermittent snowpacks are likely to see snow more rarely. Some mid- to low-elevation seasonal snowpacks are likely to become intermittent. Higher-elevation snowpacks may or may not undergo substantial changes in April 1 SWE, snow residence time, or center of melt timing, depending on precipitation outcomes. In warmer locations, temperature-dependent changes are relatively robust even if precipitation increases. In colder locations, a precipitation increase within the range of projected possibilities could cancel or overwhelm the effects of even a relatively large temperature change. Alternatively, a precipitation decrease could exacerbate projected temperature-related declines.

Glacier accumulation zones are at some of the highest elevations of the region, so they may respond positively if precipitation increases. Annual dynamics of mass balance with respect to input and output suggest that the equilibrium line (demarcating places where annual snow does not completely melt each summer) will increase in elevation, regardless of precipitation; where that elevation falls on each glacier will influence glacier response. Most glaciers will be reduced in volume and area and may become small enough to prevent movement. If climate at higher elevations becomes both warmer and drier, glaciers are unlikely to persist.

Groundwater recharge is likely to decrease in the southern portion of the IAP region, but changes in recharge remain uncertain throughout the region given limited understanding of mountain recharge processes and groundwater flow in mountain blocks. Groundwater recharge has been examined in only a few locations, and little is known about groundwater recharge processes in many watersheds. Higher minimum temperatures will reduce the longevity of snowpack, and decrease the length of time aquifer recharge can occur, potentially leading to faster runoff and less groundwater recharge. Some watersheds will be shifting from snow-dominated to rain-dominated, which may result in declines in groundwater recharge. Because many biological processes are temperature dependent, climate-induced changes in groundwater temperature may negatively affect aquatic communities. But because the thermal regime of groundwater systems is less dependent on air temperature patterns than on surface waters, the effects of rising air temperatures are likely to be less pronounced in groundwater discharges. Plant species in GDEs can respond to even slight changes in water table elevation, and shifts in composition of both vascular and bryophyte species could occur with lowered water tables.

Soil temperature and moisture are the primary drivers of change for all soil processes. The magnitude of projected change in soils with climate change is variable and depends on existing soil resources and existing climate. An increase in soil temperature will generally produce an increase in soil biological activity and soil respiration. In the current semi-arid soils of the IAP region, an increase in soil temperature

without an increase in soil moisture is likely to result in reduced biological activity, increased respiration, and decreased potential to store carbon. In the colder and wetter areas of the IAP region, an increase in soil temperature may lead to greater biological activity and to longer growing seasons if soil moisture is not limiting. Soils derived from coarse-textured parent material will transfer heat more efficiently down into the soil profile than will fine-textured soils. The heat transfer downward can affect soil processes and even groundwater temperatures. Fine-textured soils are capable of storing water longer in the soil profile, providing a buffer to warming and higher water demands by plants.

Changes in soil temperature and moisture will affect carbon and nitrogen cycles. Changes to carbon and nitrogen cycles may include an increase or reduction in cycling rates or storage of carbon and nitrogen. Soil organic carbon may contain more than three times as much carbon as is found in the atmosphere or terrestrial vegetation, and it supports many soil processes and functions. These include providing nutrients for plants, binding soil particles together and thereby maintaining structure, providing an energy source for microbes, increasing water infiltration and retention, and providing cation/anion exchange for retention of ions and nutrients. Climate change will affect SOC and ultimately the functions and processes supported by SOC. Most of the soils in the IAP region can sequester additional carbon if soil temperatures decrease and soil moisture increases. However, most climate models suggest warmer soil temperatures and various soil moisture changes. The warming temperatures without additional moisture may reduce SOC and capability of soils to store carbon.

Changes to SOC with climate change can cause changes to several soil properties that are directly tied to the amount of SOC. These include soil structure, bulk density, and soil porosity. These soil properties affect water infiltration, rooting depth, soil erosional losses, and water holding capacity. These properties are potential indicators that can be used to determine the effects of climate change and where management changes may be needed to adapt to a changing soil environment.

Soil organisms perform many functions including decomposition and nutrient cycling. The effects of climate change on soil biology are mixed. Warming of the soil may result in greater microbial activity, releasing more carbon to the atmosphere through increased decomposition. Warming of the soil may also result in slowed microbial growth and less carbon being released through respiration. Greater soil biodiversity is expected to increase soil resilience to changing climate.

Potential effects to soil chemical properties with climate change are linked to other biological and physical changes in the soil, all of which are driven by the soil temperature and moisture inputs. Salinization, acidification, pH, and cation exchange capacity are soil processes and properties that will change with changes to climate. In general, the lower-elevation, drier shrubland and grassland soils are more vulnerable to changes in soil chemical processes and properties.

There are many potential management actions to decrease vulnerabilities to climate change. The information in this chapter was used as the basis for development of climate change adaptation strategies and tactics for water use, GDEs, and soils (Appendix 4, Chapter 14).

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References

Abatzoglou, J.T.; Kolden, C.A. 2013. Relationships between climate and macroscale area burned in the western United States. International Journal of Wildland Fire. 22: 1003–1020.

Ajami, H.; Meixner, T.; Dominguez, F.; [et al.]. 2012. Seasonalizing mountain system recharge in semi-arid basins—climate change impacts. Groundwater. 50: 585–597.

Aldous, A.R.; Gannett, M.W.; Keith, M.; O'Connor, J. 2015. Geologic and geomorphic controls on the occurrence of fens in the Oregon Cascades and implications for vulnerability and conservation. Wetlands. 35: 757–767.

Allen, E.F.; Rittenour, T.M.; DeRose, R.J.; [et al.]. 2013. A tree-ring based reconstruction of Logan River streamflow in northern Utah. Water Resources Research. 49: 1–10.

Barnett, T.P.; Adam, J.C.; Lettenmaier, D.P. 2005. Potential impacts of a warming climate on water availability in snow-dominated regions. Nature. 438: 303–309.

Barnett, T.P.; Pierce, D.W. 2009. Sustainable water deliveries from the Colorado River in a changing climate. Proceedings of the National Academy of Sciences. 106: 7334–7338.

Barnett, T.P.; Pierce, D.W.; Hidalgo, H.G.; [et al.]. 2008. Human-induced changes in the hydrology of the western United States. Science. 319: 1080–1083.

Barquin, J.; Death, R.G. 2006. Spatial patterns of diversity in New Zealand springs and rhithral streams. Journal of the North American Benthological Society. 25: 768–786.

Bekker, M.F.; DeRose, R.J.; Buckley, B.M.; [et al.]. 2014. A 576-year Weber River streamflow reconstruction from tree rings for water resource risk assessment in the Wasatch Front, Utah. Journal of the American Water Resources Association. 50: 1338–1348.

Bertrand, G.; Goldscheider, N.; Gobat, J.M.; Hunkeler, D. 2012. Review: from multiscale conceptualization of groundwater-dependent ecosystems to a classification system for management purposes. Hydrogeology Journal. 20: 5–25.

Blankinship, J.C.; Meadows, M.W.; Lucas, R.G.; Hart, S.C. 2014. Snowmelt timing alters shallow but not deep soil moisture in the Sierra Nevada. Water Resources Research. 50: 1448–1456.

Blevins, E.; Aldous, A. 2011. Biodiversity value of groundwater-dependent ecosystems. Wetland Science and Practice. 1: 18–24.

Blöschl, G.; Montanari, A. 2010. Climate change impacts—Throwing the dice? Hydrological Processes. 24: 374–381.

Brady, N.C. 1999. Elements of the nature and properties of soils, 12th edition. Upper Saddle River, NJ: Prentice-Hall.

Brevik, E.C. 2013. The potential impact of climate change on soil properties and processes and corresponding influence on food security. Agriculture. 3: 398–417.

Busch, D.E.; Smith, S.D. 1995. Mechanisms associated with decline of woody species in riparian ecosystems of the southwestern U.S. Ecological Monographs. 65: 347–370.

Byrne, J.M.; Fagre, D.; MacDonald, R. 2014. Climate change and the Rocky Mountains. In: Grover, V.I.; Borsdorf, A.; Breuste, J.; [et al.], eds. Impact of global changes on mountains: Responses and adaptation. Boca Raton: CRC Press: 432–463.

Cayan, D.R.; Das, T.; Pierce, D.W.; [et al.]. 2009. Future dryness in the southwest US and the hydrology of the early 21st century drought. Proceedings of the National Academy of Sciences 107: 21271–21276.

Cayan, D.R.; Dettinger, M.D.; Kammerdiener, S.A.; [et al.]. 2001. Changes in the onset of spring in the Western United States. American Meteorological Society. 82: 399–415.

Chadde, S.W.; Shelly, J.S.; Bursik, R.J.; [et al.]. 1998. Peatlands on National Forests of the northern Rocky Mountains: Ecology and conservation. Gen. Tech. Rep. RMRS-GTR-11. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 75 p.

Clark, G.M. 2010. Changes in patterns of streamflow from unregulated watersheds in Idaho, Western Wyoming, and Northern Nevada. Journal of the American Water Resources Association. 46: 486–497.

Clayton, J.A.; Julander, R.P. 2015. Discussion of influences on snow water equivalent at Utah snow courses. Journal of Earth Science and Engineering. 5: 147–172.

Cook, B.I.; Seager, R. 2013. The response of the North American monsoon to increased greenhouse gas forcing. Journal of Geophysical Research, Atmospheres. 118: 1690–1699.

Cook, B.I.; Smerdon, J.E.; Seager, R.; Coats, S. 2014. Global warming and 21st century drying. Climate Dynamics. 43: 2607–2627.

Cooper, D.J.; Merritt, D.M. 2012. Assessing the water needs of riparian and wetland vegetation in the western United States. Gen. Tech. Rep. RMRS-GTR-282. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 125 p.

Cooper, D.J; Merritt, D.M.; Anderson, D.C.; Chimner, R.A. 1999. Factors controlling the establishment of Fremont cottonwood seedlings on the upper Green River, U.S.A. Regulated Rivers: Research and Management. 15: 419–440.

Cowardin, L.M.; Carter, V.; Golet, F.C.; LaRoe, E.T. 1979. Classification of wetlands and deepwater habitats of the United States. FWS/OBS-79/31. Washington, DC: U.S. Department of the Interior, U.S. Fish and Wildlife Service. 132 p.

Dai, A. 2013. Increasing drought under global warming in observations and models. Nature Climate Change. 3: 52–58.

Dawadi, S.; Ahmad, S. 2012. Changing climatic conditions in the Colorado River Basin: Implications for water resources management. Journal of Hydrology. 430–431: 127–141.

DeRose, R.J.; Bekker, J.F.; Wang, S.-Y.; [et al.]. 2015. A millennium-length reconstruction of Bear River stream flow, Utah. Journal of Hydrology. 529: 524–534.

Drexler, J.A.; Knifong, D.; Tuil, J.; [et al.]. 2013. Fens as whole-ecosystem gauges of groundwater recharge under climate change. Journal of Hydrology. 481: 22–34.

Earman, S.; Campbell, A.R.; Phillips, F.M.; Newman, B.D. 2006. Isotopic exchange between snow and atmospheric water vapor: Estimation of the snowmelt component of groundwater recharge in the southwestern United States. Journal of Geophysical Research Atmospheres. 111: D09302.

Earman, S.; Dettinger, M. 2011. Potential impacts of climate change on groundwater resources—A global review. Journal of Water and Climate Change. 2: 213–229.

Emmett, B.A.; Beier, C.; Estiarte, M.; [et al.]. 2004. The response of soil processes to climate change: results from manipulation studies of shrublands across an environmental gradient. Ecosystems. 7: 625–637.

Figura, S.; Livingstone, D.M.; Hoehn, E.; Kipfer, R. 2011. Regime shift in groundwater temperature triggered by the Arctic Oscillation. Geophysical Research Letters. 38: 1–5.

Flint, A.L.; Flint, L.E.; Dettinger, M.D. 2008. Modeling soil moisture processes and recharge under a melting snowpack. Vadose Zone Journal. 7: 350–357.

Gannett, M. 1984. Groundwater assessment of the John Day basin. Salem, OR: Oregon Water Resources Department. 99 p.

Gergel, D.R.; Nijssen, B.; Abatzoglou, J.T.; [et al.]. 2017. Effects of climate change on snowpack and fire potential in the western United States. Climatic Change. 141: 287–299.

Goode, J.R.; Buffington, J.M.; Tonina, D.; [et al.]. 2013. Potential effects of climate change on streambed scour and risks to salmonid survival in snow-dominated mountain basins. Hydrological Processes. 27: 750–765.

Goode, J.R.; Luce, C.H.; Buffington, J.M. 2012. Enhanced sediment delivery in a changing climate in semi-arid mountain basins: Implications for water resource management and aquatic habitat in the northern Rocky Mountains. Geomorphology. 139–140: 1–15.

Goslee, S.C.; Brooks, R.P.; Cole, C.A. 1997. Plants as indicators of wetland water source. Plant Ecology. 131: 199–206.

Green, T.R.; Taniguchi, M.; Kooi, H.; [et al.]. 2011. Beneath the surface: Impacts of climate change on groundwater. Journal of Hydrology. 405: 532–560.

Hall, M.H.; Fagre, D.B. 2003. Modeled climate-induced glacier change in Glacier National Park, 1850–2100. BioScience. 53: 131–140.

Hanson, R.T.; Flint, L.E.; Flint, A.L.; [et al.]. 2012. A method for physically based model analysis of conjunctive use in response to potential climate changes. Water Resources Research. 48: W00L08.

Healy, R.W.; Cook, P.G. 2002. Using groundwater levels to estimate recharge. Hydrogeology Journal. 10: 91–109.

Held, I.M.; Soden, B.J. 2006. Robust responses of the hydrological cycle to global warming. Journal of Climate. 19: 5686–5699.

Intergovernmental Panel on Climate Change [IPCC]. 2013. Climate change 2013: The physical science basis. Contribution of working group I to the fifth assessment report of the Intergovernmental Panel on Climate Change. Cambridge, United Kingdom and New York, NY, USA: Cambridge University Press. 1535 p.

Jencso, K.G.; McGlynn, B.L. 2011. Hierarchical controls on runoff generation: Topographically driven hydrologic connectivity, geology, and vegetation. Water Resources Research. 47: W11527.

Jencso, K.G.; McGlynn, B.L.; Gooseff, M.N.; [et al.]. 2009. Hydrologic connectivity between landscapes and streams: Transferring reach-and-plot-scale understanding to the catchment scale. Water Resources Research. 45: W04428.

Kardol, P.; Cregger, M.A.; Campany, C.E.; Classen, A.T. 2010. Soil ecosystem functioning under climate change: Plant species and community effects. Ecology. 91: 767–781.

Kløve, B.; Ala-aho, P.; Okkonen, J.; Rossi, P. 2012. Possible effects of climate change on hydrogeological systems: Results from research on esker aquifers in northern Finland. In: Treidel, H.; Martin-Bordes, J.J.; Gurdak, J.J., eds. Climate change effects on groundwater resources: A global synthesis of findings and recommendations. New York, NY: Taylor & Francis Publishing, International Association of Hydrogeologists (IAH)–International Contributions to Hydrogeology: 305–322.

Kormos, P.; Luce, C.; Wenger, S.J.; Berghuijs, W.R. 2016. Trends and sensitivities of low streamflow extremes to discharge timing and magnitude in Pacific Northwest mountain streams. Water Resources Research. 52: 4990–5007.

Kreamer, D.K.; Stevens, L.E.; Ledbetter, J.D. 2015. Groundwater dependent ecosystems—Science, challenges, and policy. In: Adelana, S.M., ed. Groundwater. Hauppauge, NY: Nova Science Publishers, Inc.: 205–230.

Kurylyk, B.L.; MacQuarrie, K.T.B; McKenzie, J.M. 2014. Climate change impacts on groundwater and soil temperatures in cold and temperate regions: Implications, mathematical theory, and emerging simulation tools. Earth-Science Reviews. 138: 313–334.

Kværner, J.; Snilsberg, P. 2011. Groundwater hydrology of boreal peatlands above a bedrock tunnel—Drainage impacts and surface water groundwater interactions. Journal of Hydrology. 403: 278–291.

Lawrence, D.J.; Stewart-Koster, B.; Olden, J.D.; [et al.]. 2014. The interactive effects of climate change, riparian management, and a nonnative predator on stream-rearing salmon. Ecological Applications. 24: 895–912.

Lee, L.J.E.; Lawrence, D.S.L.; Price, M. 2006. Analysis of water level response to rainfall and implications for recharge pathways in the Chalk aquifer, SE England. Journal of Hydrology. 330: 604–620.

Leibowitz, S.G.; Comoleo, R.L.; Wigington, P.J.; [et al.]. 2014. Hydrologic landscape classification assesses streamflow vulnerability to climate change in Oregon, USA. Hydrology and Earth System Sciences Discussions. 11: 2875–2931.

Liang, X.; Lettenmaier, D.P.; Wood, E.F.; Burges, S.J. 1994. A simple hydrologically based model of land surface water and energy fluxes for general circulation models. Journal of Geophysical Research. 99: 14415–14428.

Lins, H.F. 1997. Regional streamflow regimes and hydroclimatology. Water Resources Research. 33: 1655–1667.

Litschert, S.E.; Theobald, D.M.; Brown, T.C. 2014. Effects of climate change and wildfire on soil loss in the Southern Rockies ecoregion. Catena. 118: 206–219.

Littell, J.S.; Elsner, M.M.; Mauger, G.S.; [et al.]. 2011. Regional climate and hydrologic change in the northern US Rockies and Pacific Northwest: Internally consistent projections of future climate for resource management. Unpublished project report. http://cses.washington.edu/picea/USFS/pub/Littell_etal_2010 [Accessed December 5, 2016].

Luce, C.H.; Abatzoglou, J.T.; Holden, Z.A. 2013. The missing mountain water: Slower westerlies decrease orographic enhancement in the Pacific Northwest USA. Science. 342: 1360–1364.

Luce, C.H.; Holden, Z.A. 2009. Declining annual streamflow distributions in the Pacific Northwest United States, 1948-2006. Geophysical Research Letters 36: L16401.

Luce, C.H.; Lopez-Burgos, V.; Holden, Z. 2014. Sensitivity of snowpack storage to precipitation and temperature using spatial and temporal analog models. Water Resources Research. 50: 9447–9462.

Luce, C.H.; Morgan, P.; Dwire, K.A.; [et al.]. 2012. Climate change, forests, fire, water, and fish: Building resilient landscapes, streams and managers. Gen. Tech. Rep. RMRS-GTR-290. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 207 p.

Luce, C.H.; Pederson, N.; Campbell, J.; [et al.]. 2016. Characterizing drought for forested landscapes and streams. In: Vose, J.M.; Clark, J.S.; Luce, C.H.; Patel-Weynand, T., eds. Effects of drought on forests and rangelands in the United States: A comprehensive science synthesis. Gen. Tech. Rep. WO-93b. Washington, DC: U.S. Department of Agriculture, Forest Service, Washington Office: 13–48.

MacDonald, G.M.; Tingstad, A.H. 2007. Recent and multicentennial precipitation variability and drought occurrence in the Uinta Mountains region, Utah. Arctic, Antarctic, and Alpine Research. 39: 549–555.

Magee, T.K; Kentula, M.E. 2005. Response of wetland plant species to hydrologic conditions. Wetlands Ecology and Management. 13: 163–181.

Marston, R.A.; Pochop, L.O.; Kerr, G.L.; [et al.]. 1991. Recent glacier changes in the Wind River Range, Wyoming. Physical Geography. 12: 115–123.

Maxwell, R.M.; Kollet, S.J. 2008. Interdependence of groundwater dynamics and land-energy feedbacks under climate change. Nature Geoscience. 1: 665–669.

McCabe, G.J.; Fountain, A.G. 1995. Relations between atmospheric circulation and mass balance of South Cascade Glacier, Washington, USA. Arctic and Alpine Research. 27: 226–233.

Meixner, T., Manning, A.H., Stonestrom, D.A.; [et al.]. 2016. Implications of projected climate change for groundwater recharge in the western United States. Journal of Hydrology. 534: 124–138.

Merritt, D.M.; Scott, M.L.; Poff, N.L.; [et al.]. 2010. Theory, methods, and tools for determining environmental flows for riparian vegetation: Riparian-flow response guilds. Freshwater Biology. 55: 206–225.

Meyer, S.E. 2012. Restoring and managing cold desert shrublands for climate change mitigation. In: Finch, D.M., ed. Climate change in grasslands, shrublands, and deserts of the interior American West: A review and needs assessment. Gen. Tech. Rep. RMRS-GTR-285. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 21–34.

Milly, P.; Dunne, K.A. 2011. On the hydrologic adjustment of climate-model projections: The potential pitfall of potential evapotranspiration. Earth Interactions. 15: 1–14.

Milly, P.C.D. 1992. Potential evaporation and soil moisture in general circulation models. Journal of Climate. 5: 209–226.

Mote, P.W. 2006. Climate-driven variability and trends in mountain snowpack in western North America. Journal of Climate. 19: 6209–6220.

Mote, P.W.; Hamlet, A.F.; Clark, M.P.; Lettenmaier, D.P. 2005. Declining mountain snowpack in western North America. Bulletin of the American Meteorological Society. 86: 39–49.

Murray, B.R.; Hose, G.C.; Eamus D.; Licari, D. 2006. Valuation of groundwater-dependent ecosystems: A functional methodology incorporating ecosystem services. Australian Journal of Botany. 54: 221–229.

Musselman, K.N.; Clark, M.P.; Liu, C.; [et al.]. 2017. Slower snowmelt in a warmer world. Nature Climate Change. 7: 214–219.

Naiman, R.J.; Dècamps, H.; McCalin, M.E. 2005. Riparia: Ecology, conservation, and management of streamside communities. Burlington, MA: Elsevier Academic Press. 430 p.

Natural Resources Conservation Service [NRCS]. 2017. Web Soil Survey. http://websoilsurvey.nrcs.usda.gov/ [Accessed March 15, 2016].

Nave, L.E.; Swanston, C.W.; Mishra, U.; Nadelhoffer, K.J. 2013. Afforestation effects on soil carbon storage in the United States: A synthesis. Soil Science Society of America Journal. 77: 1035–1047.

Nolin, A.W.; Daly, C. 2006. Mapping "at risk" snow in the Pacific Northwest. Journal of Hydrometeorology. 7: 1164–1171.

Page-Dumroese, D.S.; Jurgensen, M.F. 2006. Soil carbon and nitrogen pools in mid- to late-successional forest stands of the northwestern United States: Potential impact of fire. Canadian Journal of Forest Research. 36: 2270–2284.

Pal Singh, B.; Cowie, A.L.; Chan, K.Y. 2011. Soil health and climate change. Berlin: Springer-Verlag.

Pierce, D.W.; Barnett, T.P.; Hidalgo, H.G.; [et al.]. 2008. Attribution of declining western U.S. snowpack to human effects. Journal of Climate. 21: 6425–6444.

Poff, N.L. 1996. A hydrogeography of unregulated streams in the United States and an examination of scale-dependence in some hydrological descriptors. Freshwater Biology. 36: 71–91.

Portland State University. 2009. Glaciers of the American West. http://glaciers.research.pdx.edu/states- [Accessed April 21, 2017].

Price, D. 1985. Ground water in Utah's densely populated Wasatch Front area—The challenge and the choices. U.S. Geological Survey Water Supply Paper 2232. Washington, DC: U.S. Government Printing Office.

Regonda, S.; Rajagopalan, B.; Clark, M.; Pitlick, J. 2005. Seasonal cycle shifts in hydroclimatology over the western United States. Journal of Climate. 18: 372–384.

Rice, J.; Bardsley, T.; Gomben, P.; [et al.]. 2017. Assessment of watershed vulnerability to climate change for the Uinta-Wasatch-Cache and Ashley National Forests, Utah. Gen. Tech. Rep. RMRS-GTR-362. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 111 p.

Roderick, M.L.; Greve, P.; Farquhar, G.D. 2015. On the assessment of aridity with changes in atmospheric CO_2. Water Resources Research. 51: 5450–5463.

Roderick, M.L.; Sun, F.; Lim, W.H.; Farquhar, G.D. 2014. A general framework for understanding the response of the water cycle to global warming over land and ocean. Hydrology and Earth System Sciences. 18: 1575–1589.

Rood, S.B.; Pan, J.; Gill, K.M.; [et al.]. 2008. Declining summer flows of Rocky Mountain rivers: Changing seasonal hydrology and probable impacts on floodplain forests. Journal of Hydrology. 349: 397–410.

Safeeq, M.; Grant, G.E.; Lewis, S.L.; [et al.]. 2014. A hydrogeologic framework for characterizing summer streamflow sensitivity to climate warming in the Pacific Northwest, USA. Hydrology and Earth System Sciences. 18: 3693–3710.

Safeeq, M.; Grant, G.E.; Lewis, S.L.; Tague, C.L. 2013. Coupling snowpack and groundwater dynamics to interpret historical streamflow trends in the western United States. Hydrological Processes. 27: 655–668.

Santhi, C.; Allen, P.M.; Muttiah, R.S.; [et al.]. 2008. Regional estimation of base flow for the conterminous United States by hydrologic landscape regions. Journal of Hydrology. 351: 139–153.

Seager, R.; Naik, N.; Vecchi, G.A. 2010. Thermodynamic and dynamic mechanisms for large-scale changes in the hydrological cycle in response to global warming. Journal of Climate. 23: 4651–4668.

Service, R.F. 2004. As the West goes dry. Science. 303: 1124–1127.

Shipley, B.; Keddy, P.A.; Lefkovich, L.P. 1991. Mechanisms producing plant zonation along a water depth gradient: A comparison with the exposure gradient. Canadian Journal of Botany. 69: 1420–1424.

Sophocleous, M. 2002. Interaction between groundwater and surface water: the state of the science. Hydrogeology Journal. 10: 52–67.

Steinweg, J.M.; Dukes, J.S.; Paul, E.A.; Wallenstein, M.D. 2014. Microbial responses to multi-factor climate change: Effects on soil enzymes. The Microbial Regulation of Global Biogeochemical Cycles. 4: 105–118.

Stewart, I.T.; Cayan, D.R.; Dettinger, M.D. 2005. Changes toward earlier streamflow timing across western North America. Journal of Climate. 18: 1136–1155.

Stockton, C.W.; Jacoby, G.C. 1976. Long-term surface-water supply and streamflow trends in the Upper Colorado River Basin. Lake Powell Research Project Bulletin No. 18. Arlington, VA: National Science Foundation. 70 p.

Tague, C.; Grant, G.E. 2009. Groundwater dynamics mediate low-flow response to global warming in snow-dominated alpine regions. Water Resources Research. 45: W07421.

Thomey, M.L.; Ford, P.L.; Reeves, M.C.; [et al.]. 2014. Review of climate change impacts on future carbon stores and management of warm deserts of the United States. Gen. Tech. Rep. RMRS-GTR-316. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 26 p.

Tonina, D.; Luce, C.H.; Rieman, B.; [et al.]. 2008. Hydrological response to timber harvest in northern Idaho: Implications for channel scour and persistence of salmonids. Hydrological Processes. 22: 3223–3235.

Torgersen, C.E.; Ebersole, J.L.; Keenan, D.M. 2012. Primer for identifying cold-water refuges to protect and restore thermal diversity in riverine landscapes. U.S. Environmental Protection Agency Report EPA 910-C-12-001. Seattle, WA: U.S. Environmental Protection Agency, Region 10. 78 p.

Torgersen, C.E.; Price, D.M.; Li, H.W.; McIntosh, B.A. 1999. Multiscale thermal refugia and stream habitat associations of Chinook salmon in northwestern Oregon. Ecological Applications. 9: 301–319.

Tujchneider, O.; Paris, M.; Perez, M.; D'Elia, M. 2012. Possible effects of climate change on groundwater resources in the central region of Santa Fe province, Argentina. In: Treidel, H.; Martin-Bordes, J.J.; Gurdak, J.J., eds. Climate change effects on groundwater resources: A global synthesis of findings and recommendations. New York, NY: Taylor & Francis Publishing, International Association of Hydrogeologists (IAH)–International Contributions to Hydrogeology: 265–280.

U.S. Department of Agriculture, Forest Service [USDA FS]. 2012a. Groundwater-dependent ecosystems: Level I inventory field guide; inventory methods for assessment and planning. Gen. Tech. Rep. WO-86a. Washington, DC: U.S. Department of Agriculture, Forest Service. 191 p.

U.S. Department of Agriculture, Forest Service [USDA FS]. 2012b. Groundwater-dependent ecosystems: Level II inventory field guide; inventory methods for assessment and planning. Gen. Tech. Rep. WO-86b. Washington, DC: U.S. Department of Agriculture, Forest Service. 124 p.

U.S. Department of Agriculture, Forest Service [USDA FS]. [N.d.]. Western U.S. stream flow metrics. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Air, Water, and Aquatic Environments Program. http://www.fs.fed.us/rm/boise/AWAE/projects/modeled_stream_flow_metrics.shtml [Accessed April 26, 2017].

U.S. Department of Energy [USDOE]. 2014. Overview of different aspects of climate change effects on soils. Richland, WA: U.S. Department of Energy, Pacific Northwest Laboratory. http://www.pnnl.gov/main/publications/external/technical_reports/PNNL-23483.pdf [Accessed January 24, 2017].

U.S. Fish and Wildlife Service [USFWS]. [n.d.]. National Wetlands Inventory. http://www.fws.gov/wetlands [Accessed September 22, 2017].

U.S. Geological Survey [USGS]. 2000. Groundwater atlas of the United States. http://pubs.usgs.gov/ha/ha730/gwa.html [Accessed March 15, 2016].

U.S. Geological Survey [USGS]. 2017. Hydrography http://nhd.usgs.gov [Accessed September 22, 2017].

University of Washington, Climate Impacts Group. 2017. CIG datasets. Western U.S. hydroclimate scenarios project. Seattle, WA: University of Washington, College of the Environment, Climate Impacts Group. https://cig.uw.edu/datasets/wus [Accessed September 22, 2017].

Vitt, D.H.; Slack, N.G. 1984. Niche diversification of Sphagnum relative to environmental factors in northern Minnesota peatlands. Canadian Journal of Botany. 62: 1409–1430.

Wada, Y.; van Beek, L.P.H.; Bierkens, M.F.P. 2012. Nonsustainable groundwater sustaining irrigation: A global assessment. Water Resources Research. 48: W00L06.

Waibel, M.S.; Gannett, M.W.; Chang, H.; Hulbe, C.L. 2013. Spatial variability of the response to climate change in regional groundwater systems—Examples from simulations in the Deschutes Basin, Oregon. Journal of Hydrology. 486: 187–201.

Walsh, J., D. Wuebbles, K. Hayhoe,; [et al.]. 2014. Our changing climate. Climate change impacts in the United States: The third National Climate Assessment. In: Melillo, J. M.; Richmond, T.; Yohe, G.W., eds. Washington, DC: U.S. Global Change Research Program: 19–67.

Wang, S.Y.; Gillies, R.R.; Reichler, T. 2012. Multi-decadal drought cycles in the Great Basin recorded by the Great Salt Lake: Modulation from a transition-phase teleconnection. Journal of Climate. 25: 1711–1721.

Ward, J.V.; Tockner, K. 2001. Biodiversity: Towards a unifying theme for river ecology. Freshwater Biology. 46: 807–819.

Weary, D.J.; Doctor, D.H. 2014. Karst in the United States: A digital map compilation and database. U.S. Geological Survey Open-File Report 2014–1156. Reston, VA: U.S Geological Survey.

Weltzin, J.F.; Pastor, J.; Harth, C.; [et al.]. 2000. Response of bog and fen plant communities to warming and water-table manipulations. Ecology. 81: 3464–3478.

Wenger, S.J.; Isaak, D.J.; Dunham, J.B.; [et al.]. 2011a. Role of climate and invasive species in structuring trout distributions in the interior Columbia River Basin, USA. Canadian Journal of Fisheries and Aquatic Sciences. 68: 988–1008.

Wenger, S.J.; Isaak, D.J.; Luce, C.H.; [et al.]. 2011b. Flow regime, temperature, and biotic interactions drive differential declines of trout species under climate change. Proceedings of the National Academy of Sciences, USA. 108: 14175–14180.

Wenger, S.J.; Luce, C.H.; Hamlet, A.F.; [et al.]. 2010. Macroscale hydrologic modeling of ecologically relevant flow metrics. Water Resources Research. 46: W09513.

Wenger, S.J.; Som, N.A.; Dauwalter, D.C.; [et al.]. 2013. Probabilistic accounting of uncertainty in forecasts of species distributions under climate change. Global Change Biology. 19: 3343–3354.

Wieder, W.R.; Bonan, G.B.; Allison, S.D. 2013. Global soil carbon projections are improved by modelling microbial processes. Nature Climate Change. 3: 909-912.

Winter, T.C. 1999. Relation of streams, lakes, and wetlands to groundwater flow systems. Hydrogeology Journal. 7: 28–45.

Winter, T.C. 2007. The role of groundwater in generating streamflow in headwater areas and in maintaining baseflow. Journal of the American Water Resources Association. 43: 15–25.

Winter, T.C.; Harvey, J.W.; Franke, O.L.: Alley, W.M. 1996. Ground water and surface water, a simple resource. USGS Circular 1139. Reston, VA: U.S. Geological Survey.

Woodall, C.W.; Coulston, J.W.; Domke, G.M.; [et al.]. 2015. The U.S. forest carbon accounting framework: stocks and stock change, 1990-2016. Gen. Tech. Rep. NRS-154. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 49 p.

Woodhouse, C.A.; Gray, S.T.; Meko, D.M. 2006. Updated streamflow reconstructions for the Upper Colorado River Basin. Water Resources Research. 42: W05415.

Woods, R.A. 2009. Analytical model of seasonal climate impacts on snow hydrology: Continuous snowpacks. Advances in Water Resources. 32: 1465–1481.

Yin, D.; Roderick, M.L.; Leech, G.; [et al.]. 2014. The contribution of reduction in evaporative cooling to higher surface air temperatures during drought. Geophysical Research Letters. 41: 7891–7897.

Chapter 5: Effects of Climate Change on Native Fish and Other Aquatic Species

Daniel J. Isaak, Michael K. Young, Cynthia Tait, Daniel Duffield, Dona L. Horan, David E. Nagel, and Matthew C. Groce

Introduction

The diverse landscapes of the Intermountain Adaptation Partnership (IAP) region contain a broad range of aquatic habitats and biological communities. A number of aquatic species are regional endemics, several are threatened or endangered under the U.S. Endangered Species Act (ESA), and many have declined because of the introduction of nonnative aquatic species, habitat fragmentation, and human development. Environmental trends associated with human-caused climate change have been altering the habitats of these species for several decades (Barnett et al. 2008; Hamlet and Lettenmaier 2007; Luce and Holden 2009; Mote et al. 2005), and more significant changes are expected during the 21st century (chapters 3, 4). For animals that live in or near aquatic environments such as fishes, amphibians, crayfish, mussels, and aquatic macroinvertebrates, changes in habitat and hydrological regimes are expected to shift their abundance and distribution (Ficke et al. 2007; Hauer et al. 1997; Poff et al. 2002; Rieman and Isaak 2010; Schindler et al. 2008). This is primarily because many of these species are ectothermic; thus, thermal conditions dictate their metabolic rates and most aspects of their life stages—how fast they grow and mature, whether and when they migrate, when and how often they reproduce, and when they die (Magnuson et al. 1979). Buffering these changes are the topographic diversity and steep environmental gradients of many landscapes throughout the IAP region, which contribute to slow climate velocities (sensu Loarie et al. 2009) and often create climate refugia where populations of many species can persist under all but the most extreme climatic changes (Isaak et al. 2016a; Morelli et al. 2016).

A large literature exists that describes the many interactions among climate change, aquatic environments, and cold-water fishes such as trout, salmon, and char (Hauer et al. 1997; Isaak et al. 2012a,b; ISAB 2007; Mantua and Raymond 2014; Mantua et al. 2010; Mote et al. 2003; Rieman and Isaak 2010; Young et al. 2018). Rather than revisiting those sources, we focus on providing information specific to the IAP region. First, we describe the ecology, status, and climate vulnerabilities of species of concern. Through discussions with scientists and U.S. Department of Agriculture Forest Service (USFS) Intermountain Region staff and national forest resource managers, species were chosen for their perceived vulnerability to climate change and their societal prominence as ESA-listed threatened and endangered species. The species are Rocky Mountain tailed frog (*Ascaphus montanus*), Idaho giant salamander (*Dicamptodon aterrimus*), western pearlshell mussel (*Margaritifera falcata*), springsnails (*Pyrgulopsis* spp.), Yosemite toad (*Anaxyrus canorus*), Sierra Nevada yellow-legged frog (*Rana sierrae*), bull trout (*Salvelinus confluentus*), and cutthroat trout (*Oncorhynchus clarkii*). Second, we develop spatially explicit model projections and geospatial datasets showing where bull trout and cutthroat trout are most likely to occur in current and future climates. These projections are used to assess the potential and future distribution of suitable habitats for these species, but could also be used to design and implement long-term conservation strategies or monitoring programs. Although the availability of biological datasets and models for stream networks restricted our projections to trout in streams, the approach used here is easily extended to other aquatic species as more geographic data on these taxa are gathered and models are extended to standing waters. The preceding information was used to develop climate adaptation options for species of concern, including how new technologies and ongoing development of better information could enable strategic implementation of those options to maximize their effectiveness.

Analysis Area Network and Stream Climate Scenarios

This assessment encompasses all streams in the USFS Intermountain Region that flow through its 12 national forests and lands downstream of those forests. To delineate a network that represented streams within this area, geospatial data for the 1:100,000-scale National Hydrography Dataset (NHD)-Plus Version 2 were downloaded from the Horizons Systems website (Horizon Systems Corp. http://ww.horizon-systems.com/nhdplus/; McKay et al. 2012) and filtered by minimum flow and maximum stream slope criteria. Summer flow values predicted by the Variable Infiltration Capacity (VIC) hydrological model (Wenger et al. 2010) were obtained from the Western U.S. Flow Metrics website (USDA FS https://www.fs.fed.us/rm/boise/AWAE/projects/modeled_stream_flow_metrics.shtml) and

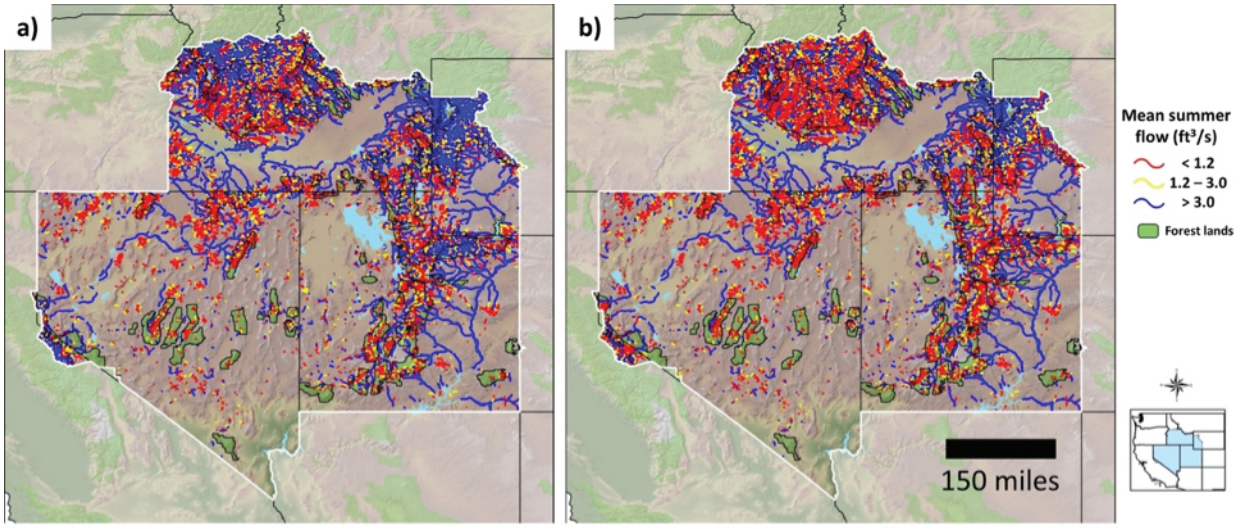


Figure 5.1—Stream network showing channels with perennial flows for (a) the baseline period and (b) 2080s based on the A1B emissions trajectory. Increasing prevalence of red stream reaches late in the century indicates a trend towards lower summer flows as winter snow accumulations decrease and melt earlier in the spring.

linked to NHDPlus stream reaches. Reaches with summer flows less than 0.2 cubic feet per second, approximating a wetted width of 3 feet (based on an empirical relationship developed in Peterson et al. 2013b), or with slopes greater than 15 percent were trimmed from the network because they are rarely occupied by fish or other aquatic vertebrates (Isaak et al. 2017b). The steepest headwater reaches are also prone to frequent large disturbances (e.g., postfire debris torrents) that may cause local extirpations of fish populations (Bozek and Young 1994; Miller et al. 2003). To exclude dry channels throughout much of the Great Basin, reaches that were coded as intermittent in the NHDPlus network were also trimmed. Application of these criteria resulted in a final network extent of 55,700 miles (fig. 5.1), which was almost evenly split between USFS (48 percent) and non-USFS (52 percent) lands.

Scenarios representing mean August stream temperature were downloaded from the NorWeST website (USDA FS https://www.fs.fed.us/rm/boise/AWAE/projects/NorWeST.html) and linked to reaches in the analysis network. NorWeST scenarios have a 0.6-mile resolution and were developed by applying spatial-stream network models (Isaak et al. 2016b, 2017b; Ver Hoef et al. 2006) to temperature records for 8,726 summers of measurement at 4,277 unique stream sites within the IAP region. The predictive accuracy of the NorWeST model (cross-validated $r^2 = 0.91$; cross-validated root mean square prediction error = 2.0 °F), combined with substantial empirical support, provided a consistent and spatially balanced rendering of temperature patterns and thermal habitat for all streams. To depict temperatures during a baseline period, we used the S1 scenario, which represented average conditions for 1993–2011 (hereafter 2000s). The mean August stream temperature during this period was 57 °F for all streams, although temperatures were significantly colder in streams flowing through national forests, where the average was 52 °F (table 5.1, fig. 5.2).

Future stream temperature scenarios were also downloaded from the NorWeST website (USDA FS n.d.b) and chosen for the same climate periods (2030–2059, hereafter 2040s; 2070–2099, hereafter 2080s) and emissions scenario (A1B) as those used for the streamflow analysis in the IAP hydrological assessment (Chapter 4). With respect to scenarios used in other chapters of the IAP assessment, the A1B scenario is similar to the RCP 6.0 scenario associated with CMIP5 simulations (Chapter 3). The future NorWeST scenarios used were S30 (2040s) and S32 (2080s), which accounted for differential sensitivity and slower warming rates of the coldest streams (Luce et al. 2014). Future stream temperature increases were projected to range from 1.4 to 2.3 °F by mid-21st century and from 2.3 to 4.0 °F by late century, with variation occurring within and among river basins (table 5.2, fig. 5.2). Future temperature increases imply warming rates similar to those observed in recent decades (Isaak et al. 2012b, 2017a) and shifts of stream temperature isotherms upstream at 1,000 to 1,600 feet per decade (Isaak and Rieman 2013; Isaak et al. 2016a).

Changes in several ecologically relevant streamflow characteristics were discussed in the hydrological assessment (Chapter 3), indicating that future snowmelt and spring runoff will occur earlier, summer flows will decrease, stream intermittency may increase in marginal areas, and more high-flow events will occur during the winter in transitional watersheds where air temperatures are near freezing (Hamlet and Lettenmaier 2007). Those projections concur with historical trends that show streams now run off 1 to 3 weeks earlier in the spring (Stewart et al. 2005), and that summer flows have decreased 10 to 30 percent in the last 50 years (Leppi et al. 2012; Luce and Holden 2009). Hydrological changes make it likely that mountain wetlands

Table 5.1—Projected changes in mean August air temperatures, streamflow, and stream temperatures for major river basins in the Intermountain Adaptation Partnership region. Projections are based on the A1B emissions scenario represented by an ensemble of 10 global climate models that best predicted historical climate conditions during the 20th century in the northwestern United States (Hamlet et al. 2013; Mote and Salathé 2010). Additional details about scenarios are provided elsewhere (Hamlet et al. 2013; Wenger et al. 2010). For more information on flow, see the western United States flow metrics website (USDA FS n.d.c) and the Pacific Northwest Hydroclimate Scenarios Project website (University of Washington, Climate Impacts Group 2010). For more information on stream temperatures, see Isaak et al. (2017b), Luce et al. (2014), and the NorWeST website (USDA FS n.d.b).

	2040s (2030–2059)			2080s (2070–2099)		
NorWeST unit[a]	Air temperature change[b]	Streamflow change	Stream temperature change[c]	Air temperature change	Streamflow change	Stream temperature change
	°F	*Percent*	*°F*	*°F*	*Percent*	*°F*
Salmon River basin	5.9	-22.3	2.3	9.9	-31.4	3.7
Upper Snake River, Bear River basins	5.8	-7.6	1.5	9.5	-9.5	2.4
Middle Snake River	5.8	-19.5	2.2	9.8	-26.7	3.7
Utah basins	4.7	+2.3	2.2	10.4	12.6	4.0
Lahontan basin	4.8	+2.6	1.4	10.7	+6.5	2.4

[a] Boundaries of NorWeST production units as described in USDA FS (n.d.c).
[b] Changes in air temperature and stream flow are expressed relative to the 1980s (1970–1999) baseline climate period.
[c] Changes in stream temperatures account for differential sensitivity to climate forcing within and among river basins as described in Luce et al. (2014) and USDA FS (n.d.c).

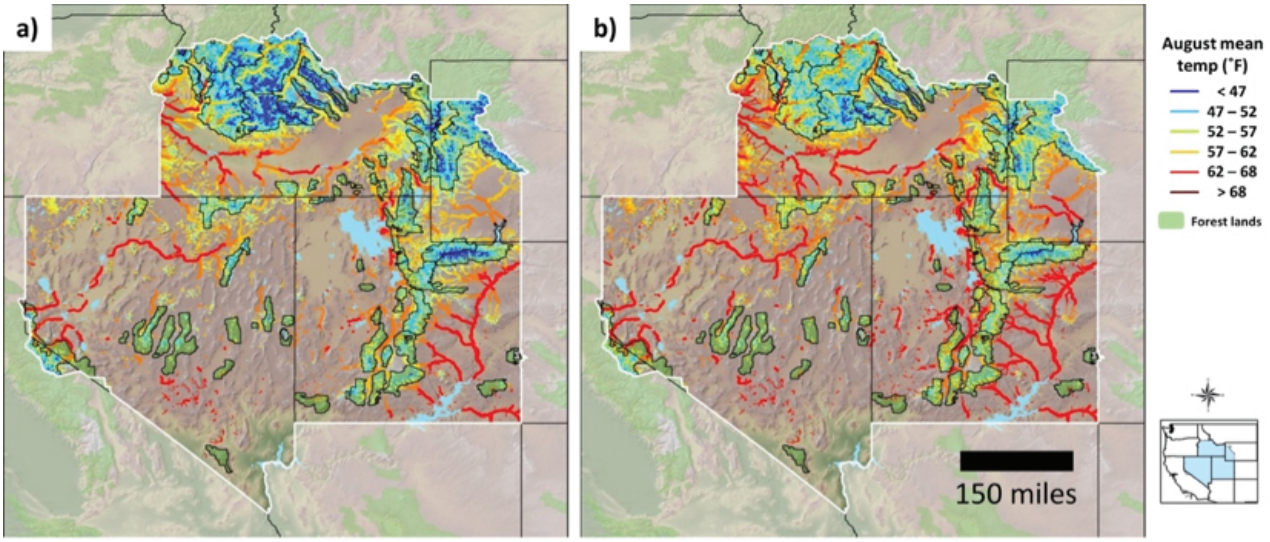


Figure 5.2—NorWeST August mean stream temperature maps interpolated from 8,726 summers of monitoring data at 4,277 unique stream sites across the 55,700 miles of streams in the analysis area. Map panels show conditions for (a) the baseline period (2000s) and (b) late-century scenario (2080s). Networks were trimmed to represent potential fish-bearing streams by excluding intermittent reaches and those with slopes greater than 15 percent and summer flows less than 0.2 cubic feet per second. High-resolution digital images of these maps and ArcGIS databases with reach-scale predictions are available at the NorWeST website (USDA FS n.d.b).

Table 5.2—Lengths of streams in the Intermountain Adaptation Partnership region (slope less than 15 percent and Variable Infiltration Capacity model-predicted summer flows greater than 0.2 cubic feet per second) categorized by mean August stream temperatures during the baseline and two future climate periods and by land administrative status. Values in parentheses are percentages of the total in the last column.

	<46 °F	46–52 °F	52–57 °F	57–63 °F	63–68 °F	>68 °F	Total[a]
Forest Service lands	*Miles (%)*	*Miles (%)*	*Miles (%)*	*Miles (%)*	*Miles (%)*	*Miles (%)*	*Miles (%)*
2000s	3,872 (14.4)	11,061 (39.5)	8,799 (32.7)	3,014 (11.2)	559 (2.1)	37 (0.1)	27,305
2040s	1,644 (6.3)	8,692 (33.1)	9,994 (38.0)	4,790 (18.2)	1,016 (3.9)	141 (0.5)	26,277
2080s	835 (3.2)	6,752 (26.2)	10,371 (40.2)	6,058 (23.5)	1,424 (5.5)	343 (1.3)	25,783
Non-Forest Service lands							
2000s	48 (0.2)	924 (3.2)	4,655 (16.2)	11,490 (39.9)	8,027 (27.9)	3,639 (12.6)	28,783
2040s	6 (0.0)	348 (1.2)	2,896 (10.2)	9,242 (32.5)	9,767 (34.3)	6,185 (21.7)	28,444
2080s	3 (0.0)	173 (0.6)	1,990 (7.0)	7,853 (26.8)	10,552 (37.3)	7,966 (28.2)	28,537

[a] Reductions in network extent in future scenarios result from projected decreases in summer flows as described in Chapter 4.

and moist areas near streams will become drier during future summer periods (Lee et al. 2015). Increased frequency or severity of wildfires in portions of the IAP region could also cause more extensive debris flows and channel disturbances in headwater streams with steep channels (Luce et al. 2012) (Chapter 3). Those combined changes suggest that stream environments and habitats for aquatic species will become more variable, subject to more disturbances, and gradually warmer throughout the rest of the 21st century.

Focal Species Ecology and Climate Vulnerability

Rocky Mountain Tailed Frog

The Rocky Mountain tailed frog occurs throughout central and northern Idaho, western Montana, and northeastern Oregon, but occurs within the IAP region only in Boise, Payette, Salmon-Challis, and Sawtooth National Forests. Populations inhabit steep, cold headwater streams, their distributions often extending upstream past waterfalls and cascades that limit fish distributions (Dunham et al. 2007; Isaak et al. 2017b). After eggs hatch in late summer, tadpoles grow for 1 to 4 years before metamorphosing into adults, which reach sexual maturity after another 4 to 5 years; local densities may be high (Hayes and Quinn 2015; Pilliod et al. 2013). Larval frogs are strictly aquatic, but adults often exploit cool, moist riparian zones to forage. Adult body size is 1 to 2 inches, and dispersal is limited, so floods and fire-related channel disturbances may suppress populations for some time after an event (Hossack and Pilliod 2011; Hossack et al. 2006). Populations are patchily distributed among headwater streams and show evidence of genetic divergence (Metzger et al. 2015).

The Rocky Mountain tailed frog is of conservation concern but does not appear on the sensitive species list of the Intermountain Region. Land use practices that warm streams, increase sedimentation, and reduce interstitial spaces in substrates or reduce habitat moisture (through loss of stream and terrestrial canopy cover) are thought to reduce habitat quality (Hayes and Quinn 2015). Nonnative fish predators may increase mortality where distributions overlap, as has been documented for other amphibians (Pilliod and Peterson 2001; Pilliod et al. 2010). Although existing data for tailed frogs suggest that the species occurs in many streams throughout its range (Isaak et al. 2017b), monitoring data are not available to describe temporal trends in abundance.

Rocky Mountain tailed frogs require cold water, so increasing temperatures may decrease the suitability of warmer downstream habitats (Isaak et al. 2017b). Their long generation times and relatively low fecundity cause populations to rebound slowly from disturbances; thus, extreme floods or postfire debris flows in steep channels may threaten persistence of some populations as climate change causes these events to become more common (Hossack and Pilliod 2011; Hossack et al. 2006). Tailed frog populations may also be negatively affected by more extreme summer droughts or wildfires that open riparian canopies, making areas adjacent to streams warmer and drier (Hossack and Pilliod 2011; Hossack et al. 2006).

Idaho Giant Salamander

The Idaho giant salamander occurs in northern and west-central Idaho and a small portion of west-central Montana, but is found within the IAP region only in Boise and Payette National Forests. Populations are patchily distributed and often co-occur with native salmonids in headwater streams, although salamanders also occupy reaches farther upstream from which fish are excluded (Sepulveda and Lowe 2009).

Giant salamander may also use lakes and ponds. Neotony (maturation as a strictly aquatic form with larval characteristics) is common (Blaustein et al. 1995). Uncertainty exists about timing of reproduction, although some literature suggests both spring and fall spawning (Nussbaum 1969). Females guard egg masses until hatching occurs and larval stages last several years before metamorphosing into adults. Adults reach body sizes of 7 to 12 inches and prey on a variety of aquatic and terrestrial species, including tailed frog tadpoles, in and near streams (Blaustein et al. 1995). Dispersal is limited, and population genetic structure varies among basins (Mullen et al. 2010).

Idaho giant salamanders are of conservation concern but do not appear on the Intermoutain Region sensitive species list. There is some indication that land use practices may affect their occurrence, but their patchy distribution and sparse datasets limit inferences about habitat requirements. They are prey for both native and nonnative fish species, but fish species presence is not known to affect their population status (Sepulveda and Lowe 2011). Overall, their distribution is poorly described, and monitoring data are not available to evaluate temporal trends. Idaho giant salamander sensitivities are presumed to be similar to Rocky Mountain tailed frogs, although the salamander may be even more susceptible to disturbance of headwater natal areas, given nest guarding behavior by females and multiyear development of larval stages before maturity (Blaustein et al. 1995; Nussbaum 1969).

Western Pearlshell Mussel

The western pearlshell mussel is found throughout the Columbia River Basin, in a portion of the Missouri River headwaters in Montana, and in internally draining basins such as the Humboldt, Truckee, and Provo Rivers. It has been recorded in all national forests in the IAP region except the three southern Utah forests. This sedentary filter-feeder inhabits cool streams and rivers at depths of 1.5 to 3.0 feet, and tends to congregate in stable substrates amid boulders, gravel, and some sand, silt, and clay (Roscoe and Redelings 1964). The species has limited mobility and will not tolerate accumulation of fine sediment. Western pearlshell larvae are obligate parasites of an array of salmonid species (Chinook salmon [*O. tshawytscha*], cutthroat trout, and rainbow trout [*O. mykiss*]; utilization of bull trout is unknown) and rely on these hosts for recruitment and dispersal (Karna and Milleman 1978; Meyers and Milleman 1977; Murphy 1942). Female mussels generally release larvae (or glochidia) in spring or early summer, depending on water temperature. Glochidia attach to fish gills and develop for a period of weeks to months. Once metamorphosed, juvenile mussels drop from their host fishes and burrow into the substrate (Murphy 1942).

The western pearlshell mussel ranges from Alaska and British Columbia south to California and east to Nevada, Wyoming, Utah, and Montana. Many examples exist of pearlshell decline or extirpation from streams and rivers across its range, especially in arid areas (Hovingh 2004; Stone et al. 2004). Threats include impoundments, loss of host fishes, channel modification, dredging and mining, pollution, sedimentation, nutrient enrichment, water diversion, degradation of native riparian vegetation, and introduction of nonnative fishes that outcompete host species. Many of these impacts, especially reduction in streamflow and increased stream temperatures, can be exacerbated by climate change.

The western pearlshell mussel occupies streams with broad ranges of thermal regimes, but nevertheless prefers cold water and perennial flows. Its habitat must also be suitable for its trout and salmon hosts, and mussel sensitivity to climatic variability will closely parallel that of salmonids. Although potentially vulnerable to climate change, the western pearlshell mussel does not face an immediate risk of extinction because it occupies such a broad geographic range.

Springsnails

Springsnails are hydrobiid snails that occur in freshwater habitats throughout much of western North America. About 100 species inhabit the IAP region (Hershler et al. 2014). These tiny mollusks (shell length 0.04–0.31 inches) are widespread and locally abundant (often greater than 100 per square foot) in perennial, groundwater-dependent springs and brooks. Spring habitats may be either ambient temperature or thermal, and springsnails are often concentrated near sources of groundwater discharge with stable water chemistry (Mladenka and Minshall 2001). They typically live on emergent plants and hard substrates, grazing on attached algae and fungi (Hershler 1998; Mladenka and Minshall 2001). They are gill breathers and do not tolerate desiccation.

Distributed from southern Canada to northern Mexico, the springsnail exhibits habitat specificity and low dispersal ability, which contribute to a high degree of endemism; many species occurr only within a single spring or seep (Hershler et al. 2014). Springsnails have life history traits that make them vulnerable to extinction. First, they have specialized habitat requirements, typically occurring in pristine, cold-water or thermal springs close to the spring source, where dissolved carbon dioxide and calcium concentrations are high (Mladenka and Minshall 2001; O'Brien and Blinn 1999). Slight changes in water chemistry or warming temperatures could negatively affect local populations (Jyväsjärvi et al. 2015). Second, springsnails are poor dispersers, and suitable habitats are generally isolated from each other by arid uplands. Once a springsnail population has been extirpated, it is unlikely to be naturally refounded. Threats to springsnails include groundwater pumping and aquifer drawdown, surface flow diversion for agriculture, impoundments, channelization of outflows, springhead development, physical alteration of thermal springs for bathing, overgrazing, and nonnative species (e.g., New Zealand mudsnail [*Potamopyrgus antipodarum*]).

The limited ability of springsnails to disperse and their narrow environmental tolerances make them vulnerable to emerging threats associated with climate change. Because they require particular hydrological conditions, specific and stable temperature regimes, and perennial flows, some taxa (e.g., eight Nevada springsnail species) have been rated as "extremely vulnerable" using the Climate Change Vulnerability Index (Young et al. 2012).

Yosemite Toad

The Yosemite toad occurs in the Sierra Nevada, restricted primarily to publicly managed lands at high elevations (3,000–12,000 feet) (Brown et al. 2015). It inhabits ponds and wet meadows as well as drier upland sites. Adult toads emerge from their overwintering refuges in rodent burrows or underground cover from late April to late June, depending on elevation and year (Brown et al. 2012). Females lay approximately 1,000 eggs per clutch in shallow standing water amid emergent vegetation. After hatching, tadpoles require 4 to 6 weeks to reach metamorphosis and are not known to overwinter (Jennings and Hayes 1994; Kagarise Sherman 1980; Karlstrom 1962). Adults spend most of their lives in upland habitats adjacent to breeding sites, and are capable of moving and dispersing several hundred feet through dry forests.

Yosemite toad populations are in decline. The Yosemite toad, once common in high-elevation aquatic ecosystems of the Sierra Nevada, had disappeared from half its historical range by the 1990s (Jennings 1996). More recent surveys indicate an 87-percent decline from watersheds occupied before 1990, with scattered remaining populations and fewer individuals per site (Brown et al. 2015). Toads were most recently recorded at 470 sites in 5 national forests (17 sites in Humboldt-Toiyabe National Forest) and about 100 more sites in national parks (Brown et al. 2015). Several factors, such as disease, drought, airborne contaminants, habitat alteration, water diversions, nonnative fishes, and wildfire, may have contributed to the declines, but there is no clear evidence targeting any particular threat (Brown et al. 2015).

The dependence of Yosemite toad on shallow, ephemeral breeding ponds filled by melting snow makes the species susceptible to risks of climate change (Kagarise Sherman and Morton 1993). Models project that climate change will lead to higher average temperatures in all seasons, higher precipitation, and decreased spring and summer runoff due to decreased snowpack (Smith and Tirpak 1989) (Chapter 4). Less runoff could affect egg and tadpole survival by premature drying of breeding sites. Earlier snowmelt could lead to earlier breeding with possible positive effects on developmental time, but negative effects and mortality from uncertain weather patterns. For example, toads that emerge early risk starvation or death in late-spring snowstorms (Kagarise Sherman and Morton 1993).

Sierra Nevada Yellow-Legged Frog

The Sierra Nevada yellow-legged frog currently inhabits the Sierra Nevada, restricted primarily to publicly managed lands at high elevations (4,500–12,000 feet), including streams, lakes, ponds, and meadow wetlands in Humboldt-Toiyabe National Forest (CDFG 2011). The species is highly aquatic during all times of the year (Mullally and Cunningham 1956). At high elevation, both frogs and tadpoles overwinter under ice in lakes and streams, and because tadpoles require 1 to 4 years to metamorphose, successful breeding sites cannot dry out in summer and need to be deep enough to preclude complete freezing or deoxygenation (Bradford 1983). Although almost always found in or near water, the frog moves seasonally between breeding ponds, foraging, and overwintering habitats, usually along watercourses. However, individuals are capable of moving several hundred feet over dry land, which facilitates recolonization of sites that have lost populations (Pope and Matthews 2001).

The Sierra Nevada yellow-legged frog was listed as endangered under the ESA after populations declined severely during the 20th century due to chytridiomycosis disease (caused by the chytrid fungus [*Batrachochytrium dendrobatidis*]), predation from nonnative fishes, livestock grazing, habitat loss and fragmentation, and perhaps airborne contaminants from the Central Valley (CDFG 2011). The species was estimated to be extirpated from 220 of 318 historical occurrence localities and most remaining populations were thought to have fewer than 100 post-metamorphic individuals (CDFG 2011). During the last 20 years, however, yellow-legged frog populations have increased significantly in Yosemite (Knapp et al. 2016). The disappearance of nonnative fish from numerous water bodies after cessation of stocking, combined with reduced susceptibility to chytridiomycosis, are thought to have stimulated the recovery (Knapp et al. 2016).

Sierra Nevada yellow-legged frogs may be vulnerable to climate change because the species relies on perennial waterbodies and needs several years to metamorphose and mature (Mullally and Cunningham 1956). Climate change could result in greater interannual climatic variability or drier summers and cause lakes, ponds, and other standing waters fed by snowmelt or streams to dry more frequently, which would reduce available breeding habitat and lead to more frequent stranding and death of tadpoles (Lacan et al. 2008). On the other hand, projected earlier snowmelt is expected to cue breeding earlier in the year, which could allow additional time for tadpole growth and development. However, earlier breeding may also expose young tadpoles and eggs to mortality from early spring frosts (Corn 2005).

Bull Trout

Bull trout are broadly distributed across the northwestern United States but are restricted to the northwestern portion of the IAP region in Boise, Humboldt-Toiyabe, Payette, Salmon-Challis, and Sawtooth National Forests (Rieman et

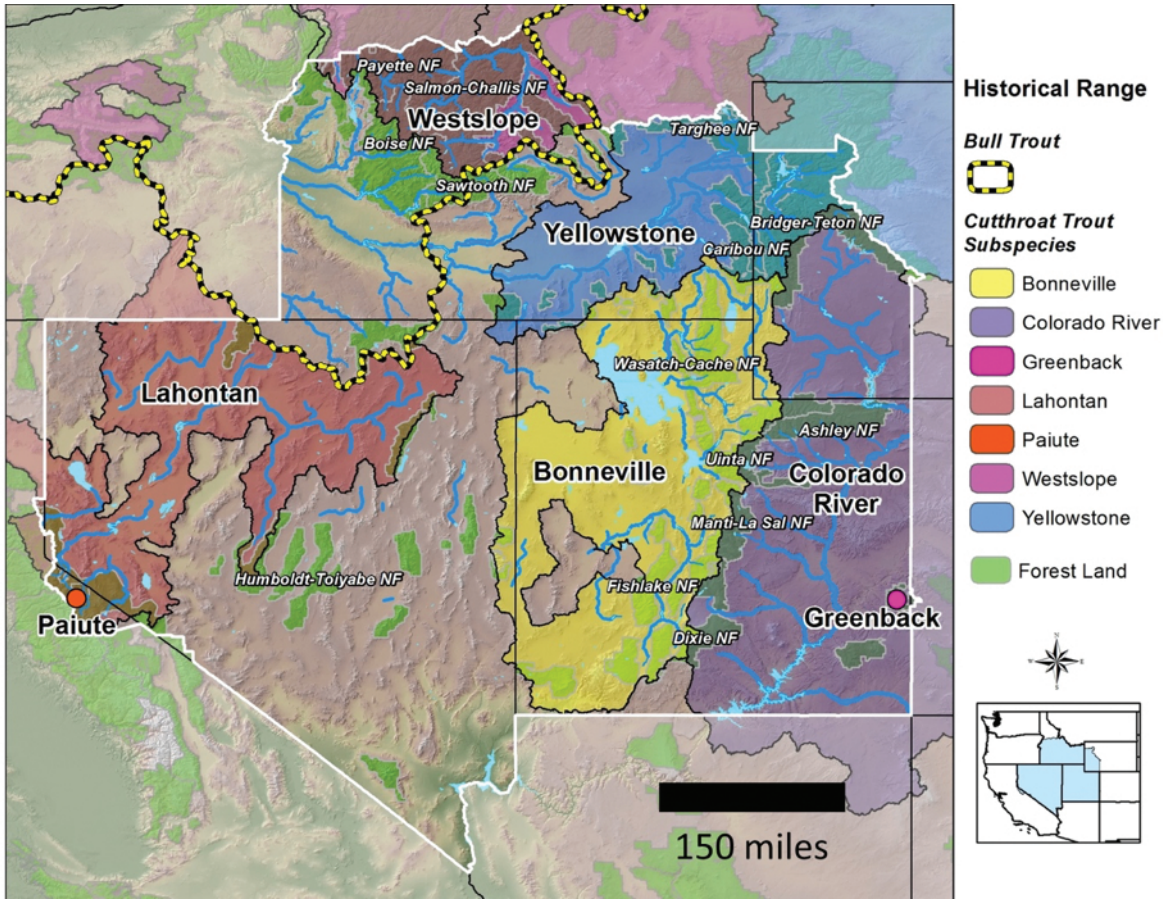


Figure 5.3—Native range distributions of (a) cutthroat trout and (b) bull trout in the Intermountain Adaptation Partnership region. The ranges of six cutthroat trout subspecies occur partly or wholly in this area.

al. 1997) (fig. 5.3). Populations may exhibit migratory or resident life histories. Migratory fish travel long distances as subadults to more productive habitats and achieve larger sizes and greater fecundity as adults before returning to cold natal headwater habitats to spawn (Howell et al. 2010; Rieman and McIntyre 1993). Fish exhibiting resident life histories remain in natal habitats and mature at smaller sizes, though often at the same age as migratory adults. Adults spawn primarily in September, and eggs incubate throughout the winter before juveniles hatch and emerge from stream substrates in late winter or early spring (Dunham et al. 2008). Reproduction and juvenile growth for the first 2 to 3 years is almost exclusively in streams with average August water temperatures less than 54 °F and flows greater than 1.2 cubic feet per second (Isaak et al. 2015). Bull trout populations are typically low density, even among strong populations in the best habitats (Isaak et al. 2017b; Rieman et al. 2006).

The bull trout is a sensitive species in the Intermountain Region and is ESA-listed as threatened (USFWS 2014). Their historical distribution has declined because of water development and habitat degradation (e.g., simplification of in-channel habitat complexity and fragmentation of some habitats), temperature increases, elimination of migratory life histories by anthropogenic barriers, harvest by anglers, and interactions with introduced nonnative fishes (Al-Chokhachy et al. 2010; Rieman et al. 1997, 2007). Encounters with nonnative fishes may involve wasted reproductive opportunities (e.g., interbreeding with brook trout [*Salvelinus fontinalis*]) (Kanda et al. 2002), competition, and predation (in streams, perhaps with brown trout [*Salmo trutta*]; in lakes, with lake trout [*Salvelinus namaycush*]) (Al-Chokhachy et al. 2016b; Martinez et al. 2009).

Bull trout evolved in western North America in interior and coastal streams that exhibit a wide array of flow characteristics and natural disturbance at scales from reaches to riverscapes (Dunham et al. 2003, 2008). Nevertheless, large habitats satisfying their restrictively cold thermal requirements for spawning and early juvenile rearing are relatively rare, and little evidence exists for flexibility in habitat use (Rieman et al. 2007). The length of connected cold-water habitats needed to support a bull trout population varies with local conditions, but current estimates suggest 10 to 30 miles are needed to ensure a high probability (>0.9) of habitat occupancy, with specifics contingent on water temperature, prevalence of brook trout, and stream slope (Isaak et al. 2015). Migratory life histories probably conferred greater resistance to extirpation under historical conditions

(Dunham et al. 2008; Rieman and Dunham 2000), but may no longer do so in some areas. Bull trout may also be sensitive to larger or more frequent winter high flows because eggs incubate in stream substrates throughout the winter where they are susceptible to bed-scour (Goode et al. 2013; Wenger et al. 2011a).

Cutthroat Trout

Cutthroat trout are represented by six subspecies in the IAP region: westslope cutthroat trout (*O. clarkii lewisi*), Yellowstone cutthroat trout (*O. c. bouvieri*), Lahontan cutthroat trout (*O. c. henshawi*), Paiute cutthroat trout (*O. c. seleniris*), Bonneville cutthroat trout (*O. c. utah*), and Colorado River cutthroat trout (*O. c. pleuriticus*) (Behnke 2002) (fig. 5.3). Although there was no historical overlap in the distribution of these subspecies, one or more were distributed throughout all national forests in the IAP region except where perennial streams are lacking (e.g., southern Nevada). These subspecies have a complex evolutionary history with two major clades (Crête-Lafrenière et al. 2012; Loxterman and Keeley 2012). One consists of westslope, Lahontan (including Paiute), and coastal cutthroat trout, and the other includes the rest of the interior subspecies. Phylogeographic structure in the latter group suggests that another one to four taxa may be present (Loxterman and Keeley 2012; Metcalf et al. 2012), but we confine our discussion to the prevailing taxonomy (Behnke 2002).

Cutthroat trout exhibit resident and migratory life history strategies similar to bull trout, but are spring spawners that reproduce in stream reaches where August temperatures are slightly warmer (up to 57 °F) (Isaak et al. 2015). Cold stream reaches where average August temperatures are less than 48 °F are suboptimal for cutthroat trout because of frequent recruitment failures associated with small numbers of growing degree days (Coleman and Fausch 2007). Cutthroat trout populations are generally found at higher densities than are bull trout (Isaak et al. 2017b).

Among cutthroat trout, all subspecies are either ESA-listed as threatened (Lahontan cutthroat trout [USFWS 1995] and Paiute cutthroat trout [USFWS 1985]) or have been petitioned for listing and found not warranted. Those not listed are on the Intermountain Region sensitive species list (Bonneville, Colorado River, westslope, and Yellowstone cutthroat trout). Distributions of these subspecies have declined more than 50 percent in response to the same stressors affecting bull trout (Behnke 2002; Shepard et al. 2005). Declines in response to nonnative species can be more severe than in bull trout, probably because cutthroat trout historically used slightly warmer habitats and overlap with more nonnative species. Brook trout have replaced cutthroat trout in many waters in the IAP region (Benjamin and Baxter 2012; Dunham et al. 2002a). These invasions seem to be influenced by the distribution of low-gradient alluvial valleys that may serve as nurseries for brook trout (Benjamin et al. 2007; Wenger et al. 2011a). Introduced rainbow trout have introgressively hybridized with cutthroat trout at lower elevations and in warmer streams (>50 °F) across their historical ranges (McKelvey et al. 2016), although genetically pure populations often persist in cold headwaters where climatic conditions limit the expansion of hybrid zones from downstream areas (Young et al. 2016, 2017).

Cutthroat trout occupy a broader thermal and stream size niche than do bull trout and can persist in small habitat patches for extended periods (Peterson et al. 2013a; Whitely et al. 2010). However, they still require cold-water natal habitat patches exceeding 2 to 6 miles to have a high probability of persistence (Isaak et al. 2015), with the habitat size depending on the prevalence of brook trout, water temperatures, and stream slope. Temperatures at the upstream extent of cutthroat trout populations in extremely cold streams will become more suitable from climate warming, but that trend may be countered by decreasing flows as snowmelt and runoff occur earlier (Chapter 3).

Climate Vulnerability and Adaptive Capacity of Focal Species

Warmer temperatures and declining summer streamflows will have broadly similar effects on aquatic species in the IAP region by reducing habitat volumes in perennial streams, fragmenting large habitat patches into smaller areas of suitable habitats (Isaak et al. 2012a; Rieman and Isaak 2010; Rieman et al. 2007), and shifting thermally suitable habitats upstream (Isaak et al. 2016a). Nonnative trout species more tolerant of warmer temperatures—brook trout, rainbow trout, and brown trout—will expand their distributions upstream and further constrain, replace, or prey on native trout and amphibians in some stream reaches. The relatively warm thermal niches of most nonnative species other than brook trout will restrict them from colonizing the coldest headwater streams, so refugial habitats, mostly at higher elevations, will continue to persist in some mountainous areas for the foreseeable future.

Wildfires may cause more extensive geomorphic disturbances and debris flows into streams, especially the smallest and steepest channels at the upstream extent of the drainage network (Miller et al. 2003; Sedell et al. 2015). Less water, more variable environments, and declining fluvial connectivity (e.g., from water development or interactions with road culverts) may favor resident life histories, as would greater separation between spawning and adult growth habitats. Smaller, more isolated populations will be more susceptible to extirpation from local environmental disturbances and during years of extreme drought and high summer water temperatures.

Species-specific vulnerabilities to these changes depend on the nexus among species ecological attributes, rates at which habitats are changing, and extent of current distributions (table 5.3). Bull trout and some cutthroat trout subspecies (e.g., westslope cutthroat trout, Yellowstone cutthroat trout) are moderately vulnerable because they are widely distributed, have good dispersal abilities, and

Table 5.3—Summary of anticipated vulnerability of selected aquatic species to climate change in the Intermountain Adaptation Partnership (IAP) region.

Species or subspecies	Taxonomy and phylogeography	Range extent[a]	Population trend	Climate vulnerability	Comment
Bull trout	Resolved	Locally common in north and elsewhere	Stable	Moderate	ESA listed[b]
Cutthroat trout subspecies					
Paiute	Resolved	Narrow endemic in west	Stable	High	ESA listed
Yellowstone	Resolved	Widespread in northeast and elsewhere	Stable	Moderate	
Westslope	Resolved	Widespread in north and elsewhere	Stable	Moderate	
Colorado River	Pending	Widespread in east and elsewhere	Stable	Moderate	
Bonneville	Pending	Restricted distribution in east	Stable	Moderate	
Lahontan	Resolved	Restricted distribution in west	Stable	High	ESA listed
Western pearlshell mussel	Resolved	Widespread in north and elsewhere	Unknown	Moderate	
Springsnails	Partially resolved	Widespread	Unknown	High	
Idaho giant salamander	Resolved	Restricted distribution in north and northern Idaho	Unknown	Moderate	
Rocky Mountain tailed frog	Resolved	Restricted distribution in north but more common elsewhere	Unknown	Moderate	
Yosemite toad	Resolved	Restricted distribution in west	Declining	High	Warranted but precluded
Sierra Nevada yellow-legged frog	Resolved	Restricted distribution in west	Increasing	High	ESA listed

[a] Geographic location (north, west, etc.) refers to IAP region only.
[b] ESA refers to U.S. Endangered Species Act.

occupy headwater habitats that are relatively resistant to thermal changes. Other subspecies of cutthroat (Bonneville, Lahontan, and Paiute cutthroat trout) are more vulnerable because distributions are limited to small numbers of isolated stream habitats.

Vulnerability of western pearlshell mussel will track that of their native trout hosts, but summer flow declines may be especially problematic because adult mollusks are immobile. Debris flows triggered by increased wildfire frequency could extirpate local mussel populations, although the species usually inhabits low-gradient stream reaches, where those events are rare (Stagliano 2005). However, fine sediments from debris flows could propagate downstream and smother mussel beds. Rocky Mountain tailed frog and Idaho giant salamanders are poor dispersers occupying headwater habitats and could be vulnerable to debris flows and more frequent disturbances.

Reduced aquifer recharge caused by altered seasonal precipitation and runoff could adversely affect groundwater-dependent and lake ecosystems that support endemic taxa such as springsnails, Yosemite toad, and Sierra Nevada yellow-legged frog (Burns et al. 2017; Hershler et al. 2014; Jyväsjärvi et al. 2015). The extreme endemism of springsnails means that the drying of individual springs could result in extirpation or extinction of a species. Arid land springs, which provide habitat for most springsnail species, are usually isolated, and dispersal of snails may be impossible without assistance by humans or other animal vectors. Recent and abrupt declines in the number and extent of Yosemite toad puts this species at high risk regardless of climate-induced environmental change. Altered aquatic habitat conditions, especially the greater environmental stochasticity that is expected, are predicted to exacerbate existing stressors and further degrade the resilience of remaining populations.

Niche conservatism suggests there is little capacity for rapid evolutionary or physiological adaptations to warmer water temperatures or desiccation within the aquatic species considered here (McCullough et al. 2009; Narum et al. 2013; Wiens et al. 2010). However, species with good dispersal abilities may track shifting habitats or recolonize previously disturbed habitats or those that have been

recently restored as long as artificial barriers do not impede their movement (Fausch et al. 2009). Some species exhibit both migratory and resident life history strategies, and the relative frequencies of these strategies may evolve based on how climate change affects metabolic rates, water temperature, stream productivity, and connectivity. Development of disease resistance or other adaptive responses associated with phenology may also bolster population resilience in ways that allow species to persist in dynamic environments (Knapp et al. 2016; Kovach et al. 2012).

Climate Refugia for Native Trout

Trout Distribution Model and Scenarios

Species distribution models for bull trout and cutthroat trout were developed previously in the Coldwater Climate Shield (CS) project by compiling large species occurrence datasets (more than 4,000 sites in over 500 streams) from field biologists, peer-reviewed literature, and State and Federal agency reports (Isaak et al. 2015). The CS models use the high-resolution stream temperature and flow scenarios described earlier with stream slope and the prevalence of brook trout as predictor variables in logistic regression models that predict occurrence probabilities of juvenile bull trout and cutthroat trout in cold-water habitat (CWH) patches (fig. 5.4). Juvenile occurrence is used as an indicator of important natal habitat locations and serves as evidence of locally reproducing populations for salmonid fishes (Dunham et al. 2002b; Rieman and McIntyre 1995). The CS models are also designed to identify CWHs that are too cold (<52 °F mean August temperature) for invasions by most nonnative species other than brook trout and thus should require limited management interventions to support native trout populations.

Previously, Young et al. (2018) applied the CS models to describe bull trout and cutthroat trout distributions and refugia in the Northern Rockies Adaptation Partnership region (Halofsky et al. 2018). Here, we repeat that exercise and summarize CS model predictions for native trout populations across the IAP region. Information for these summaries was downloaded from the CS website (USDA FS https://www.fs.fed.us/rm/boise/AWAE/projects/

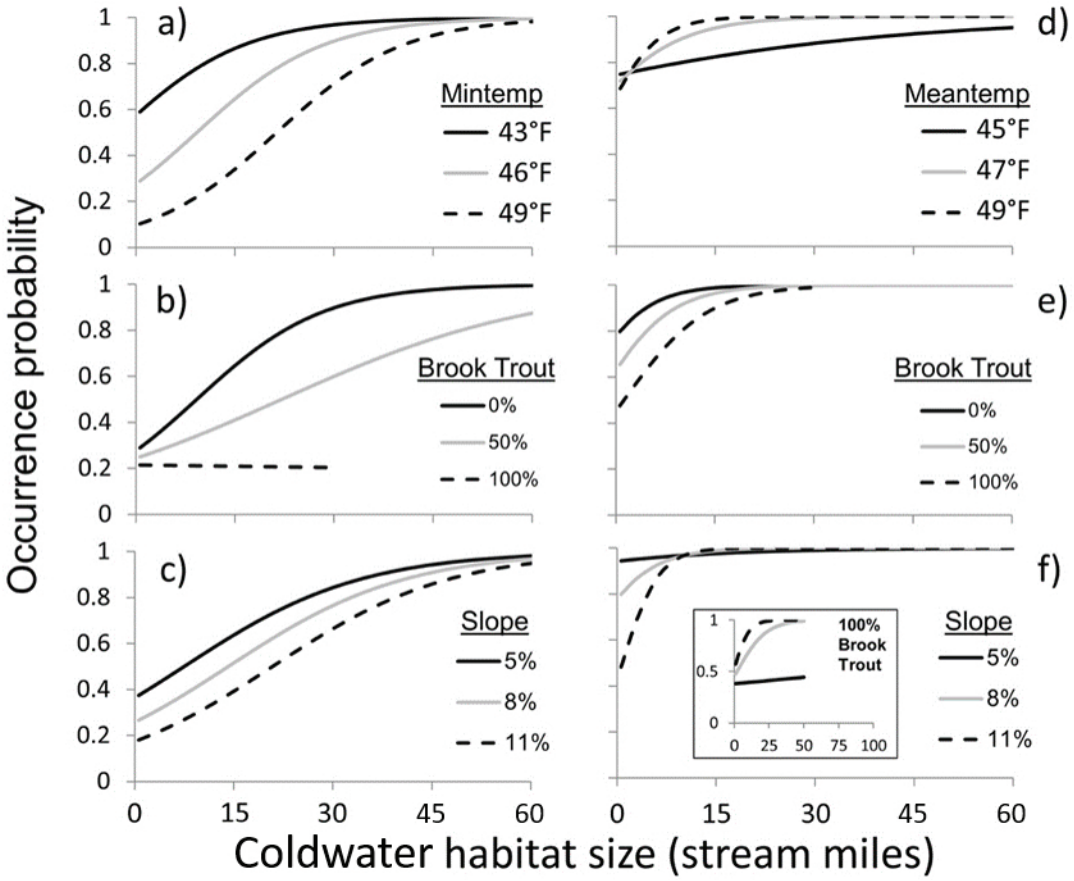


Figure 5.4—Relationships between predictor variables in the Climate Shield models and the probability of occupancy by juvenile native trout for (a–c) 512 bull trout and (d–f) 566 cutthroat trout streams characterized by cold-water habitats less than 52 °F (from Isaak et al. 2015). Relationships shown are conditioned on mean values of the two predictors not shown in a panel. An exception occurs for cutthroat trout with regard to stream slope (panel f), where brook trout values of 0 and 100 percent were used to highlight the interaction between these covariates.

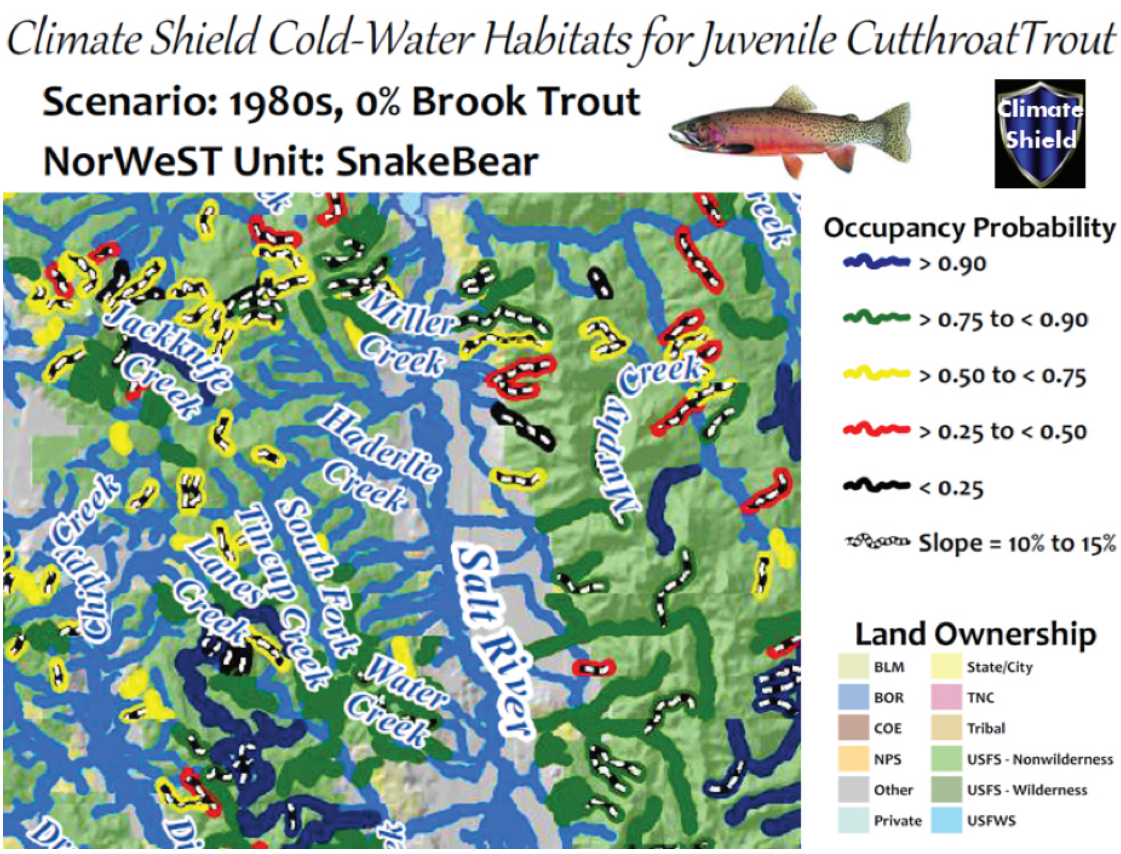


Figure 5.5—Example of a detailed Climate Shield map that shows probabilities of juvenile cutthroat trout occupancy in cold-water stream habitats. Information is available for three climate periods and five brook trout invasion scenarios for bull trout and cutthroat trout streams in the Intermountain Adaptation Partnership region at the Climate Shield website (USDA FS n.d.a).

ClimateShield.html), which also provides extensive metadata descriptions, a user-friendly digital map archive (fig. 5.5), and geospatial databases showing stream-specific model predictions for all streams in the USFS Northern, Rocky Mountain, Intermountain, and Pacific Northwest Regions in the western United States.

Scenarios used to assess native trout distributions were based on the same climate periods (2040s and 2080s) and A1B trajectory already described earlier for the NorWeST stream temperature and VIC scenarios. To account for uncertainties in brook trout distributions across the IAP region, native trout occupancy probabilities were also summarized for a pristine scenario (no brook trout) and a broad invasion scenario that assumed brook trout would be present at half the sites within each CWH (50-percent brook trout). For the IAP region, we did not summarize a scenario in which brook trout were present at all sites because their prevalence rarely exceeded 50 percent in the largest cold-water habitats (>25 miles), which are most likely to serve as strongholds for native trout (see Appendices S2 and S3 in Isaak et al. 2015), and because not all stream locations are suitable for brook trout (Isaak et al. 2017b; Wenger et al. 2011a). Brook trout prevalence may reach 100 percent in small, low-gradient streams, so native trout probabilities for a full range of invasion scenarios (0-, 25-, 50-, 75-, and 100-percent brook trout prevalence) have been calculated and integrated into the CS geospatial databases to facilitate stream-specific assessments (if local information is available on brook trout prevalence).

Cutthroat Trout Status and Projected Trends

Invasion-resistant streams with mean August temperatures less than 52 °F encompassed 15,000 miles of the 56,000-mile network draining the IAP region during the baseline period; 94 percent of those cold streams are on national forests (table 5.2). The number of discrete CWHs that were potentially occupied by juvenile cutthroat trout during the baseline period was estimated to be 2,600 and encompassed nearly 13,000 stream miles across all subspecies (table 5.4, fig. 5.6). Three subspecies—Lahontan cutthroat trout, Bonneville cutthroat trout, and Paiute cutthroat trout—had restricted ranges and are summarized separately (table 5.5). Bonneville and Lahontan cutthroat had about 620 miles of CWHs during the baseline period, whereas habitat for the Paiute cutthroat trout was less than 12 miles because it occurred in only a few streams.

Across all cutthroat trout subspecies, 89 percent of CWHs were predicted to have probabilities of occupancy exceeding 50 percent in the current period if brook trout

Table 5.4—Number and length of cold-water habitats for all subspecies of juvenile cutthroat trout by probability of occurrence for three climate periods and two brook trout invasion scenarios in the Intermountain Adaptation Partnership region.

		Probability of occurrence (percent)					Total cold-water habitats
	Period	<25	25–50	50–75	75–90	>90	
Cold-water habitat number				*Number*			
0% brook trout prevalence	2000s	73	206	540	872	909	2,600
	2040s	49	170	544	791	680	2,234
	2080s	66	252	479	572	476	1,845
50% brook trout prevalence	2000s	80	261	1,296	736	227	2,600
	2040s	50	193	1,152	606	233	2,234
	2080s	66	260	975	440	104	1,845
Cold-water habitat length				*Miles*			
0% brook trout prevalence	2000s	215	388	1,272	2,322	8,794	12,776
	2040s	68	288	951	1,858	5,871	9,036
	2080s	93	398	931	1,402	3,665	6,489
50% brook trout prevalence	2000s	578	864	2,894	4,447	4,208	12,991
	2040s	140	386	2,202	3,047	3,260	9,035
	2080s	93	454	2,008	2,290	1,643	6,488

were absent (table 5.4), largely because of the relatively small stream networks that a cutthroat trout population requires for persistence (e.g., 6 miles is associated with a 90-percent probability of occupancy) (fig. 5.4). Nonetheless, the largest CWHs accounted for a disproportionate amount of the habitat most likely to be occupied; 35 percent of CWHs had probabilities greater than 90 percent, but these accounted for 68 percent of the length of CWHs. As expected, the number and extent of CWHs were predicted to decrease substantially (14–50 percent) in future periods, but 1,845 potential habitats encompassing almost 6,500 miles were projected to remain in the extreme 2080 scenario.

Where headwater stream reaches are currently too cold for cutthroat trout, future warming may increase habitat suitability and the probability of occupancy in portions of the Uinta Mountains and other cold streams draining the upper Green, Salmon, and Snake Rivers. As expected, the brook trout invasion scenario did not affect the number or amount of CWHs because the habitats remained potentially suitable for cutthroat trout, but occupancy probabilities declined (table 5.4). The sensitivity of streams to brook trout invasions varies with local conditions, but impacts were most pronounced in small CWHs with low slopes (fig. 5.4).

Bull Trout Status and Projected Trends

The historical range of bull trout occupies a smaller portion of the IAP region than cutthroat trout, so the number of discrete CWHs for bull trout during the baseline climate period was estimated to be 984, encompassing 7,700 miles (table 5.6, fig. 5.7). In contrast to cutthroat trout, most CWHs for bull trout had probabilities of occupancy less than 50 percent because of the relatively large stream networks that bull trout require (30 miles is associated with a 90-percent probability of occupancy) (fig. 5.4). Although only 8 percent of CWHs had probabilities greater than 90 percent, they provided 36 percent of the total length of CWH, emphasizing the contribution of large CWHs to the amount of habitat predicted to be occupied. The requirement for larger CWHs caused projected decreases in the number (9–28 percent) and network extent (35–57 percent) of bull trout CWHs to be more substantial than those for cutthroat trout, particularly for the CWHs with the highest probabilities of occupancy. However, more than 700 CWHs representing 3,330 miles were projected to remain even in the extreme 2080 scenario.

Similar to the effect on cutthroat trout, the brook trout invasion scenario showed reduced bull trout occupancy rates (especially in CWHs with greater than 50-probability of occupancy), and as few as three to four CWHs with probabilities greater than 90 percent remained under the extreme warming scenario with a ubiquitous brook trout presence. However, many of the large bull trout habitats are less susceptible to broad-scale brook trout invasions given the preference by the latter species for small low-gradient streams (Dunham et al. 2002a; Isaak et al. 2015). Not surprisingly, CWHs with the highest bull trout occupancy probabilities in all scenarios coincided with river networks containing the largest number of cold streams (headwater portions of the Boise, Middle Fork Salmon, and Upper Salmon Rivers) (fig. 5.7).

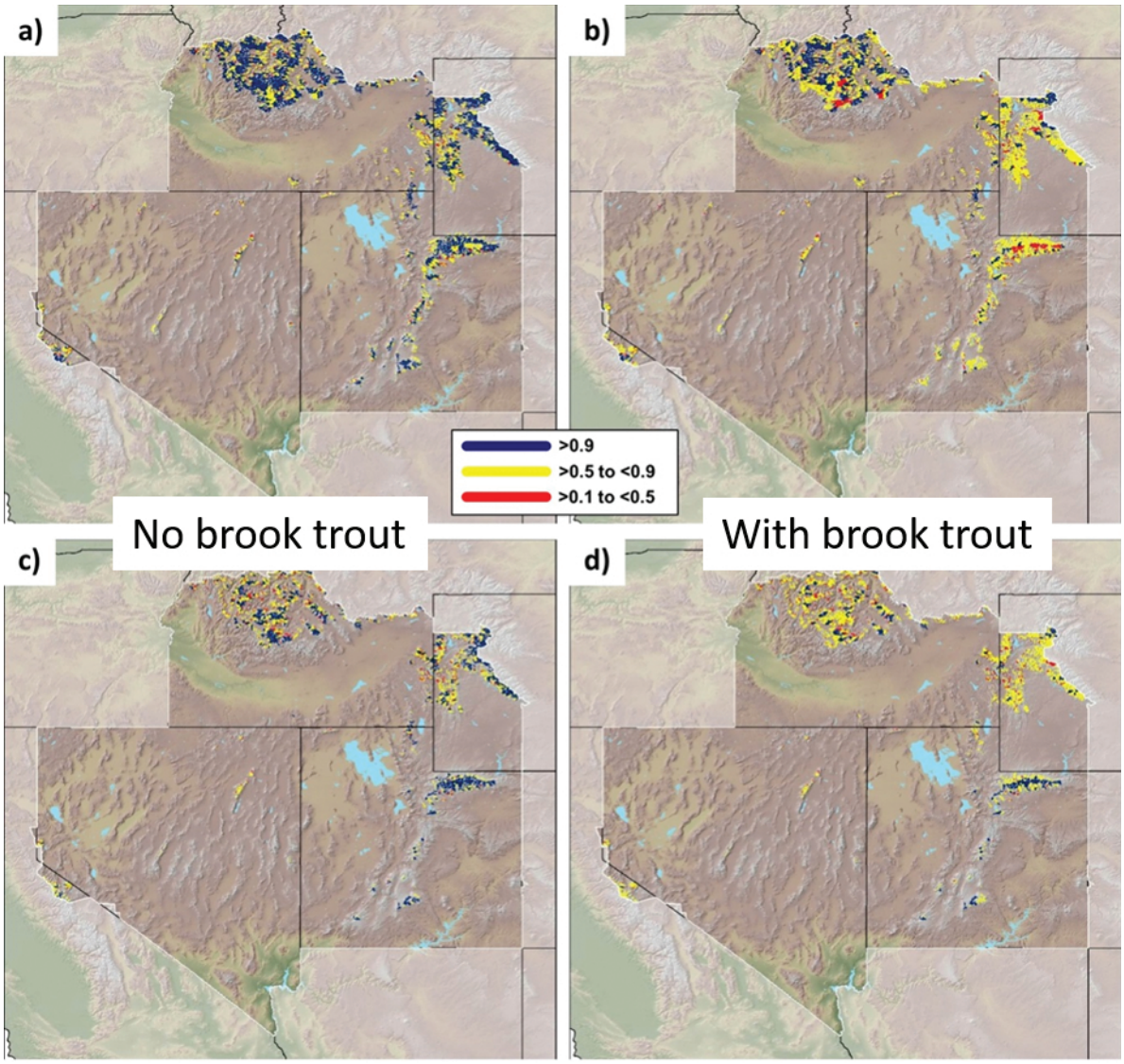


Figure 5.6—Distribution of cold-water habitats with probabilities of occupancy greater than 0.1 for juvenile cutthroat trout for (a–b) the baseline period (2000s) and (c–d) late-century scenario (2080s). Panels a and c illustrate occupancy when brook trout are absent; panels b and d illustrate occupancy when brook trout prevalence is 50 percent.

Conclusions and Implications

A changing climate has significant implications for distributions of aquatic habitats and species dependent on them in the IAP region. Vulnerability to habitat shifts or losses this century may be high, especially for taxa that have either restricted habitats or limited dispersal abilities. Yet the symptoms of rapid climate change have been manifest throughout the western United States for several decades (Barnett et al. 2008; Hamlet and Lettenmaier 2007; Luce and Holden 2009; Mote et al. 2005) without widespread losses of aquatic populations or species. Three factors help explain this apparent paradox. First, the steep topography in parts of the IAP region translates to slow climate velocities, which may enable species to track gradual shifts in their habitats (Isaak et al. 2016a). Second, the thermal and hydrological changes that have accumulated to date are relatively small compared to changes expected throughout the remainder of the 21st century (Chapter 3). Third, existing monitoring programs and analyses of available datasets may be inadequate for detecting the subtle distribution shifts or extirpation of small populations that are expected with climate change (Isaak and Rieman 2013).

In one of the few attempts to rigorously measure how distributions of stream organisms are responding to climate change, site revisits over a 20-year period revealed that juvenile bull trout distributions were contracting upstream within the Bitterroot River basin of Montana (Eby et al. 2014). In a larger study across western Montana, long-term monitoring indicated that brown trout populations were increasing in abundance and gradually expanding into streams

Table 5.5—Number and length of cold-water habitats for three cutthroat trout subspecies by probability of occurrence for three climate periods, assuming the absence of brook trout. Reduced occurrence probabilities associated with brook trout invasions would be similar to declines described in table 5.4.

Cutthroat trout subspecies	Period	Probability of occurrence (percent)					Total cold-water habitats
		<25	25–50	50–75	75–90	>90	
Bonneville				*Number*			
Cold-water habitat number	2000s	1	24	70	90	96	281
	2040s	0	12	34	62	54	162
	2080s	0	8	23	22	44	97
				Miles			
Cold-water habitat length	2000s	0	33	100	176	530	839
	2040s	0	12	39	124	310	485
	2080s	0	9	26	48	228	311
Lahontan				*Number*			
Cold-water habitat number	2000s	3	25	39	63	29	159
	2040s	2	16	40	50	25	133
	2080s	1	10	40	31	11	93
				Miles			
Cold-water habitat length	2000s	5	48	86	175	227	541
	2040s	3	31	78	137	150	399
	2080s	2	17	85	93	58	255
Paiute				*Number*			
Cold-water habitat number	2000s	0	0	0	1	2	3
	2040s	1	0	1	3	0	5
	2080s	1	0	0	2	0	3
				Miles			
Cold-water habitat length	2000s	0	0	0	4	8	12
	2040s	1	0	1	6	0	7
	2080s	1	0	0	5	0	6

that were previously too cold (Al-Chokhachy et al. 2016b). Similar monitoring efforts and site resurveys are needed at broad scales for many species to document the effects of climate change on aquatic life and to provide the basis for strategic planning and proactive conservation actions (Comte and Grenouillet 2013; Craine et al. 2007).

The first step in promulgating an informative aquatic biodiversity census is the aggregation and organization of historical datasets into functional databases, allowing species occurrence information to be linked with environmental covariates, then modeled and analyzed for trends. Development of the CS models involved organizing a small fraction of existing datasets for trout species (Isaak et al. 2015), but even this initial effort yielded significant improvements in the ability to predict where trout populations are most likely to persist, has assisted monitoring efforts, and is setting the stage for developing more precise models in the near future. For example, the spatially explicit CS model predictions are being used to guide an interagency crowd-sourcing campaign to collect environmental DNA (eDNA) samples from 7,000 locations throughout the native range of bull trout, which includes the northern portion of the IAP region (Young et al. 2017). Those samples will be paired with new spatial stream-network models (Isaak et al. 2014; Ver Hoef et al. 2006) to model bull trout occurrence at high resolution (<1 mile) across broad areas to provide accurate predictions of distribution and abundance and a better understanding of environmental constraints. This new generation of eDNA samples could be compared to historical occurrence datasets to provide broad-scale climate trend assessments. The bull trout eDNA samples also contain the DNA of many

Table 5.6—Number and length of cold-water habitats for juvenile bull trout by probability of occurrence for three climate periods and two brook trout invasion scenarios in the Intermountain Adaptation Partnership region.

		Probability of occurrence (percent)					
	Period	<25	25–50	50–75	75–90	>90	Total
Cold-water habitat number		----------*Number*----------					
0% brook trout prevalence	2000s	406	289	141	70	78	984
	2040s	538	216	90	31	23	898
	2080s	387	215	76	22	12	712
50% brook trout prevalence	2000s	474	301	132	52	25	984
	2040s	608	211	56	19	4	898
	2080s	456	197	43	13	3	712
Cold-water habitat length		----------*Miles*----------					
0% brook trout prevalence	2000s	1,158	1,480	1,245	1,003	2,806	7,692
	2040s	1,630	1,232	874	464	766	4,967
	2080s	1,059	999	645	289	344	3,336
50% brook trout prevalence	2000s	1,452	1,950	1,651	1,097	1,542	7,692
	2040s	1,994	1,531	723	529	190	4,967
	2080s	1,337	1,180	463	216	140	3,336

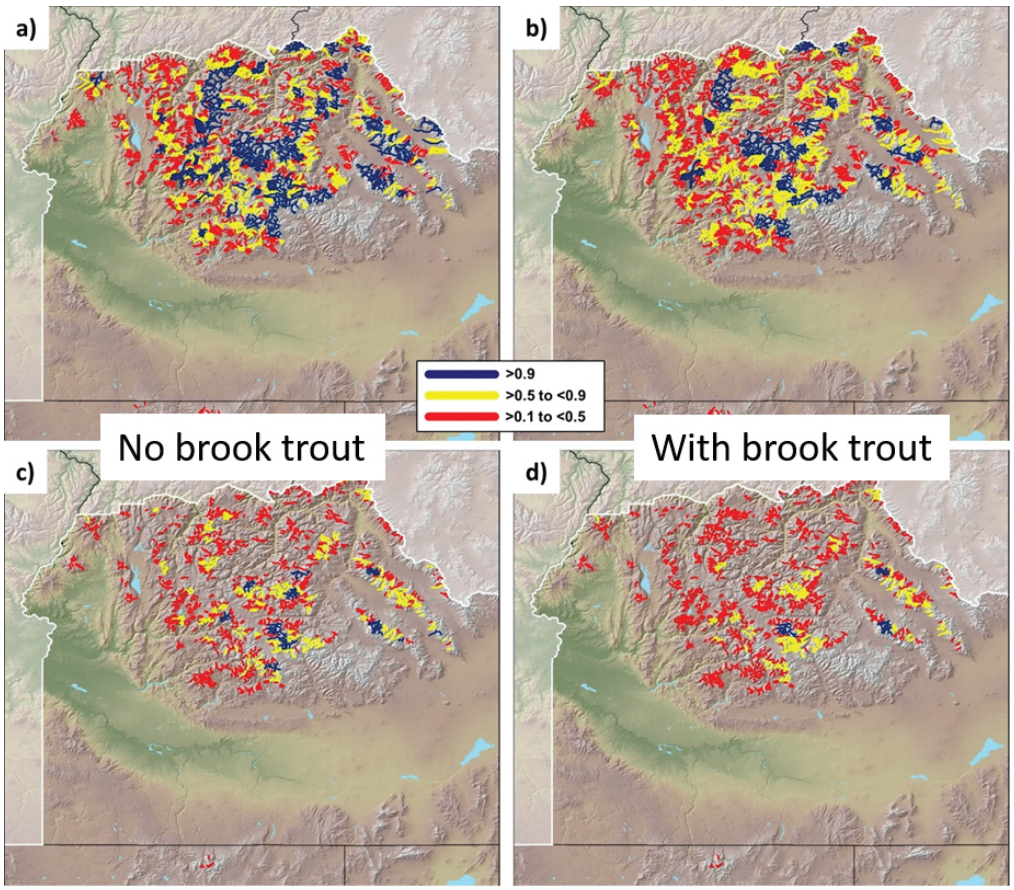


Figure 5.7—Distribution of cold-water habitats with probabilities of occupancy greater than 0.1 for juvenile bull trout for (a–b) the baseline period (2000s) and (c–d) late-century scenario (2080s). Panels a and c illustrate occupancy when brook trout are absent; panels b and d illustrate occupancy when brook trout prevalence is 50 percent.

other aquatic taxa, so will help establish baseline conditions for many species that lack data. Moreover, thousands of additional eDNA samples are now being collected annually across the western United States through partnerships with the National Genomics Center for Wildlife and Fish Conservation (http://www.fs.fed.us/research/genomics-center), so a taxonomically diverse and geographically comprehensive monitoring system for aquatic species is emerging.

For native bull trout and cutthroat trout, the CS models provided accurate, spatially explicit predictions of habitat occupancy throughout the IAP region with respect to current conditions. Assuming that species responses are related to the effects of climate on stream ecosystems—and the accuracy of the models supports this contention—the models also provide reasonably robust projections of habitat occupancy in light of anticipated climate change. Although both native trout species are regarded as cold-water taxa, their exact responses to a changing climate are expected to differ. Bull trout, and most members of the genus *Salvelinus*, are adapted to some of the coldest freshwater environments in the Northern Hemisphere (Klemetsen et al. 2003). These species also tend to inhabit highly variable environments, often with strong gradients in productivity that appear to favor migration as a life history tactic (Klemetsen 2010). It is unsurprising, therefore, that a species with those attributes near the southern end of its distribution would be susceptible to range contractions as temperatures warm. In the IAP region, we anticipate declines in bull trout distributions, but the species is unlikely to be extirpated from the region because many climate refuge habitats will persist. As climate change proceeds this century, it is possible that conditions favoring migratory or resident life histories will change, although it is unclear how these conditions would be accommodated or exploited by bull trout. As we learn more about the extent and prevalence of populations occupying CWHs with varying probabilities of occupancy, an understanding of the environmental drivers of bull trout life history may emerge.

Cutthroat trout, in contrast, can accommodate a wider range of thermal environments, consistent with their broad latitudinal distribution and an evolutionary history. Since the late Miocene or early Pliocene, this species was exposed to intervals cycling between warm/dry and cool/moist periods in western North America (McPhail and Lindsey 1986; Minckley et al. 1986). Cutthroat trout are relatively plastic with respect to life history strategies, ranging from highly migratory populations dependent on large rivers or lakes to promote growth and fecundity, to resident populations that move little and have been isolated for decades to centuries (Northcote 1992; Peterson et al. 2013b). Although we anticipate net losses in cutthroat distribution in the IAP region, they are not expected to be as severe as for bull trout, and some basins that are currently too cold to support cutthroat trout may become high-quality climate refugia in the future (Al-Chokhachy et al. 2013; Coleman and Fausch 2007; Cooney et al. 2005). Of greater importance may be how nonnative salmonids, which often displace or replace cutthroat trout, respond to warming conditions (Wenger et al. 2011a). These factors do not represent similar risks to the six cutthroat trout subspecies, primarily because of the large differences in quality and quantity of habitats currently occupied. For example, Yellowstone and westslope cutthroat trout are widely distributed, occupy many streams throughout their ranges, and exhibit a broad array of life histories. However, remaining subspecies often persist in small, isolated headwater habitats that may be especially vulnerable to brook trout invasions (Benjamin and Baxter 2012; Dunham et al. 2002a).

The presence of brook trout is problematic for both bull trout and cutthroat trout. The tolerance of brook trout to cold temperature is nearly equivalent to that of cutthroat trout, and brook trout favor the low-gradient environments preferred by cutthroat trout and bull trout (Wenger et al. 2011a). However, very large habitats appear less likely to be invaded by brook trout, possibly because this species prefers small streams but also because large systems usually contain other salmonid species, such as rainbow trout or brown trout, that compete with brook trout (Fausch et al. 2009). Rainbow trout and brown trout are expected to shift their distribution upstream as temperature isotherms optimal for these species move in that direction (Isaak and Rieman 2013; Wenger et al. 2011b), but may be constrained by unsuitable stream steepness because these species are rare where slopes exceed 4 percent (Isaak et al. 2017b). Where overlap occurs, both species appear to have negative effects on cutthroat trout, but cold, steep headwater streams that resist their invasions are expected to remain widespread.

Most current and future CWHs occur on public lands, mostly national forests, emphasizing the critical role of the USFS in conservation of native fish. Exploring an array of conservation strategies will be important because most of the CWHs are outside designated wilderness areas or national parks so are subject to various land management activities. Conservation options will vary by location because current and future CWHs are expected to be more abundant and persistent in some river basins than others across the IAP region. Where CWHs are abundant, maintaining those conditions and avoiding significant new impairments may be all that is necessary to ensure the persistence of native fish populations. In contrast, very few habitats that function as climate refugia may occur in other basins or where current habitats for some species or subspecies are very limited. Those circumstances favor strategic, active management to promote population persistence, whether by manipulating habitat or fish populations, or both. And because many CWHs are in landscapes that have multiple resource values, balancing among competing interests will remain an underlying theme of public land management (Rieman et al. 2010). Retaining native populations of aquatic organisms in many of these areas may require conservation investments that are unsustainable or

could prove ineffective if climate warming accelerates appreciably this century. In these circumstances, reallocating investment resources to areas where native populations are more likely to persist may be preferable.

Climate Adaptation Options

Many things can be done to adapt to climate change and improve the resilience of aquatic species in the IAP region (Chapter 14). Climate change adaptation options for aquatic species have been the subject of a number of comprehensive reviews for the IAP region (Rieman and Isaak 2010) and the Pacific Northwest (Beechie et al. 2013; Isaak et al. 2012a; ISAB 2007; Luce et al. 2012; Williams et al. 2007, 2015; Young et al., in press b). Several key themes emerge from these reviews: (1) be strategic; (2) implement monitoring programs; (3) restore and protect natural flow, sediment, and temperature regimes; (4) manage fluvial connectivity; (5) remove or suppress nonnative species, and (6) consider assisted migration.

Be strategic—Prioritize watershed restoration such that the most important work is done in the most important places because the funds, labor, and time available for management of native fish populations are limited (Peterson et al. 2013a). Efforts are directed at only a few of the locations and problems that could be addressed. For example, climate refugia for native trout that are in wilderness areas may neither require nor be amenable to habitat modification to ensure the persistence of those populations. Similar refugia outside protected areas could be targeted to improve habitat conditions or remove or reduce nonnative species presence, particularly if doing so increases the probability of occupancy of such habitats by native species.

Implement monitoring programs—Being strategic means reducing current and future uncertainties for decisionmaking. More data are needed for streamflow (more sites), stream temperature (annual data from sensors maintained over many years), and species distributions. These data could be used for better status and trend descriptions, and to develop robust (or more accurate) models for species to understand their response to climate change, natural variation, and land management. The feasibility of monitoring at small to broad scales is increasing with the advent of rapid, reliable eDNA inventories of aquatic organisms (McKelvey et al. 2016; Thomsen et al. 2012) and the availability of inexpensive, reliable temperature and flow sensors (USEPA 2014).

Restore and maintain natural flow, sediment, and thermal regimes—Persistence of native species can be enhanced using a variety of habitat techniques to improve stream shade, narrow unnaturally widened channels, minimize flow diversions, and improve stream substrate conditions. Actions may include decommissioning or relocating roads away from streams (Al-Chokhachy et al. 2016a; Zurstadt 2015), limiting seasonal grazing in some areas, and managing streamside riparian forest buffer zones to maintain effective shade and cool, moist riparian microclimates (Nusslé et al. 2015). Tactics may also involve directly managing water, such as increasing water storage in headwater reservoirs, restoring populations of American beaver (*Castor canadensis*) (Bouwes et al. 2016; Pollock et al. 2014), or acquiring water rights to maintain or enhance summer streamflows (Elmore et al. 2015). Such actions obviously have implications and consequences far beyond enhancing the persistence of native fish populations, but being open to opportunities is part of strategic thinking.

Manage connectivity—Obstacles to fish migration may be removed in hopes of enhancing the success of migratory life history forms, or permitting native species to reoccupy former habitat or supplement existing populations. This also presents a dilemma: Accessible waters can be invaded by nonnative species that sometimes replace native species (Fausch et al. 2009). The alternative is the installation of barriers to prevent these invasions. Native populations above barriers may be secure if they can adopt resident life histories, but could be susceptible to loss from catastrophic events in small habitats and will require human intervention for refounding or supplementation. Barriers are also temporary, and eventually will require reconstruction if nonnative species still remain downstream. Barriers may also be associated with small headwater diversion structures that sometimes route fish out of streams, where they are susceptible to stranding when water ditches are seasonally dewatered (Roberts and Rahel 2008; Walters et al. 2013). Headwater diversions are numerous and may be difficult to locate, so tools for locating them may be useful. For example, the Trout Unlimited Water Transaction Tool (McFall 2017) shows the locations of all diversion points in the Idaho Department of Water Resources database relative to the CS native trout refuge streams within Idaho. The tool enables users to identify and visit points of diversions in critical trout habitats to determine their potential impact to fish populations and design mitigation strategies.

Remove nonnative species—Removal of nonnative fish species will be essential to maintain or restore some populations. These efforts typically consist of chemical treatments or electrofishing, and both tend to be feasible only in smaller, simpler habitats (Shepard et al. 2002). Both are also costly, in part because they need to be conducted on multiple occasions to be effective (Peterson et al. 2008). Chemical treatments are controversial because of their perceived effects on water quality. Furthermore, success with either method is obtained only if the source of nonnative species is removed, often by the installation of a migration barrier. Unauthorized introductions are also common, and can undermine conservation efforts. Finally, using control measures to manage the abundance of nonnative species rather than removing them has been applied in some areas (e.g., the removal of lake trout to promote bull trout persistence or regular electrofishing to depress brook trout and favor cutthroat trout). Such activities are likely to be successful only if conducted at regular intervals for the

foreseeable future, which assumes funding and enthusiasm for such ventures will be available indefinitely.

Consider assisted migration—Moving native fish species from one location to another, a historically common activity in fish management, has typically been used to found populations in previously fishless waters. This tactic may be further pursued in the IAP region where a few basins are currently fishless (or have only limited populations of nonnative species) because of natural barriers such as waterfalls, and may create high-quality climate refugia in the near future. Moving native fish to such areas is feasible but controversial because other at-risk native taxa may be vulnerable to predation or competition with native fish species. Reintroductions of native species (Dunham et al. 2011, 2016) may also be performed when natural refounding is not an option, such as where populations are isolated and periodically fail, or suffer population bottlenecks. Management intervention at this level will require an understanding of genetic principles and broodstock establishment.

In conclusion, responding to the environmental trends associated with climate change will require a diverse portfolio that includes many of the actions described earlier. We need to adapt our mindsets and administrative processes to a new paradigm of dynamic disequilibrium. Under this paradigm, stream habitats will become more variable, undergo gradual shifts through time, and sometimes decline. Many populations are resilient enough to persist in, or track, suitable habitats, but others could be overwhelmed by future changes. It is unlikely that we will be able to preserve all populations of aquatic species as they currently exist this century. But as better information continues to be developed in the future, managers will have ever more precise tools at their disposal to know when and where resource commitments are best made to enhance the resilience of existing populations or to benefit other species for which management was previously not a priority. There is much to do as climate change adaptation continues in future years (Chapter 14), and USFS lands will play an increasingly important role in providing refuge habitats for aquatic biodiversity.

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References

Al-Chokhachy, R.; Alder, J.; Hostetler, S.; [et al.]. 2013. Thermal controls of Yellowstone cutthroat trout and invasive fishes under climate change. Global Change Biology. 19: 3069–3081.

Al-Chokhachy, R.; Black, T.A.; Thomas, C.; [et al.]. 2016a. Linkages between unpaved forest roads and streambed sediment: Why context matters in directing road restoration. Restoration Ecology. 24: 589–598.

Al-Chokhachy, R.; Roper, B.B.; Bowerman, T.; [et al.]. 2010. A review of bull trout habitat associations and exploratory analyses of patterns across the interior Columbia River Basin. North American Journal of Fisheries Management. 30: 464–480.

Al-Chokhachy, R.; Schmetterling, D.; Clancy, C.; [et al.]. 2016b. Are brown trout replacing or displacing bull trout populations in a changing climate? Canadian Journal of Fisheries and Aquatic Sciences. 73: 1395–1404

Barnett, T.P.; Pierce, D.W.; Hidalgo, H.G.; [et al.]. 2008. Human-induced changes in the hydrology of the western United States. Science. 319: 1080–1083.

Beechie, T.; Imaki, H.; Greene, J.; [et al.]. 2013. Restoring salmon habitat for a changing climate. River Research and Applications. 29: 939–960.

Behnke, R. 2002. Trout and salmon of North America. New York, NY: Simon and Schuster. 384 p.

Benjamin, J.R.; Baxter, C.V. 2012. Is a trout a trout? A range-wide comparison shows nonnative brook trout exhibit greater density, biomass, and production than native inland cutthroat trout. Biological Invasions. 14: 1865–1879.

Benjamin, J.R.; Dunham, J.B.; Dare, M.R. 2007. Invasion by nonnative brook trout in Panther Creek, Idaho: Roles of local habitat quality, biotic resistance, and connectivity to source habitats. Transactions of the American Fisheries Society. 136: 875–888.

Blaustein, A.P.; Beatty, J.J.; Olson, D.H.; [et al.]. 1995. The biology of amphibians and reptiles in old-growth forests in the Pacific Northwest. Gen. Tech. Rep. PNW-GTR-337. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 98 p.

Bouwes, N.; Weber, N.; Jordan, C.E.; [et al.]. 2016. Ecosystem experiment reveals benefits of natural and simulated beaver dams to a threatened population of steelhead (*Oncorhynchus mykiss*). Scientific Reports. 6: doi:10.1038/srep28581.

Bozek, M.A.; Young, M.K. 1994. Fish mortality resulting from delayed effects of fire in the Greater Yellowstone Ecosystem. Great Basin Naturalist. 54: 91–95.

Bradford D.F. 1983. Winterkill, oxygen relations, and energy metabolism of a submerged dormant amphibian, *Rana muscosa*. Ecology. 64: 1171–1183.

Brown, C.; Hayes, M.P.; Green, G.A.; [et al.]. 2015. Yosemite toad conservation assessment. R5-TP-040. Vallejo, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Region.

Brown, C.; Kiehl, K.; Wilkinson, L. 2012. Advantages of long-term, multi-scale monitoring: Assessing the current status of the Yosemite toad (*Anaxyrus* [*Bufo*] *canorus*) in the Sierra Nevada, California, USA. Herpetological Conservation and Biology. 7: 115–131.

Burns, E.R.; Zhu, Y.; Zhan, H.; [et al.]. 2017. Thermal effect of climate change on groundwater-fed ecosystems. Water Resources Research. 53: 3341–3351.

California Department of Fish and Game [CDFG]. 2011. A status review of the mountain yellow-legged frog (*Rana sierrae* and *Rana muscosa*). Report to the Fish and Game Commission. Sacramento, CA: State of California, Natural Resources Agency, Department of Fish and Game. 51 p.

Coleman, M.A.; Fausch, K.D. 2007. Cold summer temperature limits recruitment of age-0 cutthroat trout in high-elevation Colorado streams. Transactions of the American Fisheries Society. 136: 1231–1244.

Comte, L.; Grenouillet, G. 2013. Do stream fish track climate change? Assessing distribution shifts in recent decades. Ecography. 36: 1236–1246.

Cooney, S.J.; Covich, A.P.; Lukacs, P.M.; [et al.]. 2005. Modeling global warming scenarios in greenback cutthroat trout (*Oncorhynchus clarki stomias*) streams: Implications for species recovery. Western North American Naturalist. 65: 371–381.

Corn, P.S. 2005. Climate change and amphibians. Animal Biodiversity and Conservation. 28: 59–67.

Craine, J.M.; Battersby, J.; Elmore, A.J.; [et al.]. 2007. Building EDENs: The rise of environmentally distributed ecological networks. BioScience. 57: 45–54.

Crête-Lafrenière, A.; Weir, L.K.; Bernatchez, L. 2012. Framing the Salmonidae family phylogenetic portrait: A more complete picture from increased taxon sampling. PLoS ONE. 7: e46662.

Dunham, J.B.; Adams, S.B.; Schroeter, R.E.; [et al.]. 2002a. Alien invasions in aquatic ecosystems: Toward an understanding of brook trout invasions and potential impacts on inland cutthroat trout in western North America. Reviews in Fish Biology and Fisheries. 12: 373–391.

Dunham, J.B.; Baxter, C; Fausch, K.D.; [et al.]. 2008. Evolution, ecology, and conservation of Dolly Varden, white spotted char, and bull trout. Fisheries 33: 537–550.

Dunham, J.B.; Gallo, K.; Shively, D.; [et al.]. 2011. Assessing the feasibility of native fish reintroductions: A framework applied to threatened bull trout. North American Journal of Fisheries Management. 31: 106–115.

Dunham, J.B.; Rieman, B.E.; Chandler, G.L. 2003. Influences of temperature and environmental variables on the distribution of bull trout within streams at the southern margin of its range. North American Journal of Fisheries Management. 23: 894–904.

Dunham, J.B.; Rieman, B.E.; Peterson, J.T. 2002b. Patch-based models to predict species occurrence: Lessons from salmonid fishes in streams. In: Scott, J.M.; Heglund, P.J.; Morrison, M.; [et al.], eds. Predicting species occurrences: issues of scale and accuracy. Covelo, CA: Island Press: 327–334.

Dunham, J.B.; Rosenberger, A.E.; Luce, C.H.; [et al.]. 2007. Influences of wildfire and channel reorganization on spatial and temporal variation in stream temperature and the distribution of fish and amphibians. Ecosystems. 10: 335–346.

Dunham, J.B.; White, R.; Allen, C.S.; [et al.]. 2016. The reintroduction landscape. In: Jachowski, D.S.; Millspaugh, J.J.; Angermeier, P.L.; [et al.], eds. Reintroduction of fish and wildlife populations. Berkeley, CA: University of California Press: 79–103.

Dunham, J.B.; Young, M.K.; Gresswell, R.E.; [et al.]. 2003. Effects of fire on fish populations: Landscape perspectives on persistence of native fishes and non-native fish invasions. Forest Ecology and Management. 178: 183–196.

Eby, L.A.; Helmy, O.; Holsinger, L.M.; [et al.]. 2014. Evidence of climate-induced range contractions for bull trout in a Rocky Mountain watershed, U.S.A. PLoS ONE 9: e98812.

Elmore, L.R.; Null, S.E.; Mouzon, N.R. 2015. Effects of environmental water transfers on stream temperatures. River Research and Applications. 32: 1415–1427.

Fausch, K.D.; Rieman, B.E.; Dunham, J.B.; [et al.]. 2009. Invasion versus Isolation: Trade-offs in managing native salmonids with barriers to upstream movement. Conservation Biology. 23: 859–870.

Ficke, A.D.; Myrick, C.A.; Hansen, L.J. 2007. Potential impacts of global climate change on freshwater fisheries. Reviews in Fish Biology and Fisheries. 17: 581–613.

Goode, J.R.; Buffington, J.M.; Tonina, D.; [et al.]. 2013. Potential effects of climate change on streambed scour and risks to salmonid survival in snow-dominated mountain basins. Hydrological Processes. 27: 750–765.

Halofsky, J.E.; Peterson, D.L.; Dante-Wood, S.K.; [et al.], eds. 2018. Climate change vulnerability and adaptation in the Northern Rocky Mountains. Gen. Tech. Rep. RMRS-GTR-374. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Hamlet, A.F.; Lettenmaier, D.P. 2007. Effects of 20th century warming and climate variability on flood risk in the western U.S. Water Resources Research. 43: W06427

Hamlet, A.F.; Elsner, M.M.; Mauger, G.S.; [et al.]. 2013. An overview of the Columbia Basin climate change scenarios project: Approach, methods, and summary of key results. Atmosphere-Ocean. 51: 392–415.

Hauer, F.R.; Baron, J.S.; Campbell, D.H.; [et al.]. 1997. Assessment of climate change and freshwater ecosystems of the Rocky Mountains, USA and Canada. Hydrological Processes. 11: 903–924.

Hayes, M.P.; Quinn, T., eds. 2015. Review and synthesis of the literature on tailed frogs (genus *Ascaphus*) with special reference to managed landscapes. Coop. Monitor. Eval. Res. Rep. CMER 01-107. Olympia, WA: Washington Department of Natural Resources, Washington State Forest Practices Adaptive Management Program.

Hershler, R. 1998. A systematic review of the hydrobiid snails (Gastropoda:Rissooidea) of the Great Basin, western United States: Part I. Genus *Pyrgulopsis*. Veliger. 41: 1–132.

Hershler, R.; Liu, H.P; Howard, J. 2014. Springsnails: A new conservation focus in western North America. BioScience. 64: 693–700.

Horizon Systems Corp. [n.d.] NHDPlus home. http://www.horizon-systems.com/NHDPlus/index.php, ver. 2 [Accessed April 26, 2017].

Hossack, B.R.; Pilliod, D.S. 2011. Amphibian responses to wildfire in the western United States: Emerging patterns from short-term studies. Fire Ecology. 7: 129–144.

Hossack, B.R.; Corn, P.S.; Fagre D.B. 2006. Divergent patterns of abundance and age-class structure of headwater stream tadpoles in burned and unburned watersheds. Canadian Journal of Zoology. 84: 1482–1488.

Hovingh, P. 2004. Intermountain freshwater mollusks, USA (*Margaritifera, Anodonta, Gonidea, Valvata, Ferrissia*): Geography, conservation, and fish management implications. Monographs of the Western North American Naturalist. 2: 109–135.

Howell, P.J.; Dunham, J.B.; Sankovich, P.M. 2010. Relationships between water temperatures and upstream migration, cold water refuge use, and spawning of adult bull trout from the Lostine River, Oregon, USA. Ecology of Freshwater Fish. 19: 96–106.

Isaak, D.J.; Rieman, B.E. 2013. Stream isotherm shifts from climate change and implications for distributions of ectothermic organisms. Global Change Biology. 19: 742–751.

Isaak, D.J.; Luce, C.; Rieman, B.; [et al.]. 2010. Effects of climate change and recent wildfires on stream temperature and thermal habitat for two salmonids in a mountain river network. Ecological Applications. 20: 1350–1371.

Isaak, D.J.; Muhlfeld, C.C.; Todd, A.S.; [et al.]. 2012a. The past as prelude to the future for understanding 21st-century climate effects on Rocky Mountain trout. Fisheries. 37: 542–556.

Isaak, D.J.; Peterson, E.; Ver Hoef, J.; [et al.]. 2014. Applications of spatial statistical network models to stream data. Wiley Interdisciplinary Reviews–Water. 1: 27–294.

Isaak, D.J.; Wenger, S.J., Peterson, E.E.; [et al.]. 2016a. NorWeST modeled summer stream temperature scenarios for the western U.S. Data archive. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Isaak, D.J.; Wenger, S.J., Peterson, E.E.; [et al.]. 2017a. The NorWeST database and summer temperature scenarios: A crowd-sourced database and new geospatial tools foster a user-community and predict broad climate warming of rivers and streams in the western U.S. Water Resources Research. 53: 9181–9205.

Isaak, D.J.; Wenger, S.J.; Young, M.K. 2017b. Big biology meets microclimatology: Defining thermal niches of ectotherms at landscape scales for conservation planning. Ecological Applications. 27: 977–990.

Isaak, D.J.; Wollrab, S.; Horan, D.; [et al.]. 2012b. Climate change effects on stream and river temperatures across the northwest U.S. from 1980–2009 and implications for salmonid fishes. Climatic Change. 113: 499–524.

Isaak, D.; Young, M.K.; Luce, C.; [et al.]. 2016b. Slow climate velocities of mountain streams portend their role as refugia for cold-water biodiversity. Proceedings of the National Academy of Sciences, USA. 113: 4374–4379.

Isaak, D.J.; Young, M.K.; Nagel, D.; [et al.]. 2015. The cold-water climate shield: Delineating refugia to preserve salmonid fishes through the 21st century. Global Change Biology. 21: 2540–2553.

Independent Science Advisory Board [ISAB]. 2007. Climate change impacts on Columbia River basin fish and wildlife. ISAB Clim. Change Rep. ISAB 2007-2. Portland, OR: Northwest Power and Conservation Council.

Jennings, M.R. 1996. Status of amphibians. In: Sierra Nevada ecosystem project: Final report to Congress, Vol. 2, Assessments and scientific basis for management options. Rep. 37. Davis, CA: University of California, Davis, Centers for Water and Wildland Resources: 921–944.

Jennings, M.R.; Hayes, M.P. 1994. Amphibian and reptile species of special concern status in California. Unpublished report on file with: California Department of Fish and Game, Rancho Cordova, CA. 255 p.

Jyväsjärvi, J.; Marttila, H.; Rossi, P.M.; [et al.]. 2015. Climate-induced warming imposes a threat to north European spring ecosystems. Global Change Biology. 21: 4561–4569.

Kagarise Sherman, C. 1980. A comparison of the natural history and mating system of two anurans: Yosemite toads (*Bufo canorus*) and black toads (*Bufo exsul*). PhD dissertation. Ann Arbor, MI: University of Michigan. 394 p.

Kagarise Sherman, C.; Morton, M.L. 1984. The toad that stays on its toes. Natural History. 93: 72–78.

Kagarise Sherman, C.; Morton, M.L. 1993. Population declines of Yosemite toads in the eastern Sierra Nevada of California. Journal of Herpetology. 27: 186–198.

Kanda, N.; Leary, R.F.; Allendorf, F.W. 2002. Evidence of introgressive hybridization between bull trout and brook trout. Transactions of the American Fisheries Society. 131: 772–782.

Karlstrom, E.L. 1962. The toad genus *Bufo* in the Sierra Nevada of California: Ecological and systematic relationships. University of California Publications in Zoology. 62: 1–104.

Karna, D.W.; Millemann, R.E. 1978. Glochidiosis of salmonid fishes. III. Comparative susceptibility to natural infection with *Margaritifera margaritifera* (L.) (Pelecypoda: Margaritanidae) and associated histopathology. Journal of Parasitology. 64: 528–537.

Klemetsen, A. 2010. The charr problem revisited: Exceptional phenotypic plasticity promotes ecological speciation in postglacial lakes. Freshwater Reviews. 3: 49–74.

Klemetsen, A.; Amundsen, P.A.; Dempson, J.B.; [et al.]. 2003. Atlantic salmon *Salmo salar* L., brown trout *Salmo trutta* L. and Arctic charr *Salvelinus alpinus* (L.): A review of aspects of their life histories. Ecology of Freshwater Fish. 12: 1–59.

Knapp, R.A.; Fellers, G.M.; Kleeman, P.M.; [et al.]. 2016. Large-scale recovery of an endangered amphibian despite ongoing exposure to multiple stressors. Proceedings of the National Academy of Sciences, USA. 113: 11889–11894.

Kovach, R.P.; Al-Chokhachy, R.; Whited, D.C.; [et al.]. 2017. Climate, invasive species and land use drive population dynamics of a cold-water specialist. Journal of Applied Ecology. 54: 638–647.

Kovach, R.P.; Gharrett, A.J.; Tallmon, D.A. 2012. Genetic change for earlier migration timing in a pink salmon population. Proceedings of the Royal Society of London B: Biological Sciences. 279: 3870–3878.

Lacan, I.; Matthews, K.; Feldman, K. 2008. Interaction of an introduced predator with future effects of climate change in the recruitment dynamics of the imperiled Sierra Nevada yellowlegged frog (*Rana sierrae*). Herpetological Conservation and Biology. 3: 211–223.

Lee, S-Y.; Ryan, M.E.; Hamlet, A.F. 2015. Projecting the hydrologic impacts of climate change on montane wetlands. PLoS ONE. 10: e0136385.

Leppi, J.C.; DeLuca, T.H.; Harrar, S.W.; [et al.]. 2012. Impacts of climate change on August stream discharge in the Central-Rocky Mountains. Climatic Change. 112: 997–1014.

Loarie, S.R.; Duffy, P.B.; Hamilton, H.; [et al.]. 2009. The velocity of climate change. Nature. 462: 1052–1055.

Loxterman, J.L.; Keeley, E.R. 2012. Watershed boundaries and geographic isolation: Patterns of diversification in cutthroat trout from western North America. BMC Evolutionary Biology. 12: 38.

Luce, C.H.; Holden, Z.A. 2009. Declining annual streamflow distributions in the Pacific Northwest United States, 1948–2006. Geophysical Research Letters. 36: L16401.

Luce, C.; Morgan, P.; Dwire, K.; [et al.]. 2012. Climate change, forests, fire, water, and fish: Building resilient landscapes, streams, and managers. Gen. Tech. Rep. RMRS-GTR-290. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Luce, C.H.; Staab, B.P.; Kramer, M.G.; [et al.]. 2014. Sensitivity of summer stream temperatures to climate variability in the Pacific Northwest. Water Resources Research. 50: 1–16.

Magnuson, J.J.; Crowder, L.B.; Medvick, P.A. 1979. Temperature as an ecological resource. American Zoologist. 19: 331–343.

Mantua, N.J.; Raymond, C.L. 2014. Climate change, fish, and aquatic habitat in the North Cascade Range. In: Raymond, C.L.; Peterson, D.L.; Rochefort, R.M., eds. Climate change vulnerability and adaptation in the North Cascades region, Washington. Gen. Tech. Rep. PNW-GTR-892. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 235–270.

Mantua, N.J.; Tohver, I.; Hamlet, A. 2010. Climate change impacts on streamflow extremes and summertime stream temperature and their possible consequences for freshwater salmon habitat in Washington State. Climatic Change. 102: 187–223.

Martinez, P.J.; Bigelow, P.E.; Deleray, M.A.; [et al.]. 2009. Western lake trout woes. Fisheries. 34: 424–442.

McCullough, D.A.; Bartholow, J.M.; Jager, H.I.; [et al.]. 2009. Research in thermal biology: Burning questions for coldwater stream fishes. Reviews in Fisheries Science. 17: 90–115.

McFall, Sean. 2017. Water Transaction Tool, Idaho. Trout Unlimited. https://public.tableau.com/profile/sean.mcfall#!/vizhome/ClimateShieldIdaho/Story1 [Accessed October 25, 2017].

McKay, L.; Bondelid, T.; Dewald, T.; [et al.]. 2012. NHDPlus version 2: User guide. ftp://ftp.horizon-systems.com/NHDPlus/NHDPlusV21/Documentation/NHDPlusV2_User_Guide.pdf [Accessed September 16, 2016].

McKelvey, K.S.; Young, M.K.; Knotek, W.L.; [et al.]. 2016. Sampling large geographic areas for rare species using environmental DNA: A study of bull trout occupancy in western Montana. Journal of Fish Biology. 88: 1215–1222.

McKelvey, K.S.; Young, M.K.; Wilcox, T.M.; [et al.]. 2016. Patterns of hybridization among cutthroat and rainbow trout in northern Rocky Mountain streams. Ecology and Evolution. 6: 688–706.

McPhail, J.D.; Lindsey, C.C. 1986. Zoogeography of the freshwater fishes of Cascadia (the Columbia system and rivers north to the Stikine). In: Hocutt, C.H.; Wiley, E.O., eds. Zoogeography of North American freshwater fishes. New York, NY: Wiley: 615–637.

Metcalf, J.L.; Love Stowell, S. Kennedy, C.M.; [et al.]. 2012. Historical stocking data and 19th century DNA reveal human-induced changes to native diversity and distribution of cutthroat trout. Molecular Ecology. 21: 5194–5207.

Metzger, G.; Espindola, A.; Waits, L.P.; Sullivan, J. 2015. Genetic structure across broad spatial and temporal scales: Rocky Mountain tailed frogs (*Ascaphus montanus*; Anura: Ascaphidae) in the inland temperate rainforest. Journal of Heredity. 106: 700–710.

Meyers, T.R.; R.E. Millemann. 1977. Glochidiosis of salmonid fishes. I. Comparative susceptibility to experimental infection with *Margaritifera margaritifera* (L.) (Pelecypoda: Margaritanidae). Journal of Parasitology 63: 728–733.

Miller, D.; Luce, C.; Benda, L. 2003. Time, space, and episodicity of physical disturbance in streams. Forest Ecology and Management. 178: 121–140.

Minckley, W.L.; Hendrickson, D.A.; Bond, C.E. 1986. Geography of western North American freshwater fishes: Description and relationships to intracontinental tectonism. In: Hocutt, C.H.; Wiley, E.O., eds. Zoogeography of North American freshwater fishes. New York, NY: Wiley: 519–614.

Mladenka, G.C.; Minshall, G.W. 2001. Variation in the life history and abundance of three populations of Bruneau hot springsnails (*Pyrgulopsis bruneauensis*). Western North American Naturalist. 61: 204–212.

Morelli, T.L.; Daly C.; Dobrowski, S.Z.; [et al.]. 2016. Managing climate change refugia for climate adaptation. PLoS ONE. 11: e0159909.

Mote, P.W.; Salathé, E.P. 2010. Future climate in the Pacific Northwest. Climatic Change. 102: 29–50.

Mote, P.W.; Hamlet, A.F.; Clark, M.P.; [et al.]. 2005. Declining mountain snowpack in western North America. Bulletin of the American Meteorological Society. 86: 39–49.

Mote, P.W.; Parson, E.A.; Hamlet, A.F.; [et al.]. 2003. Preparing for climatic change: The water, salmon, and forests of the Pacific Northwest. Climatic Change. 61: 45–88.

Mullally, D.P.; Cunningham, J.D. 1956. Ecological relations of *Rana muscosa* at high elevations in the Sierra Nevada. Herpetologica. 12: 189–198.

Mullen, L.B.; Woods, H.A.; Schwartz, M.K.; [et al.]. 2010. Scale-dependent genetic structure of the Idaho giant salamander (*Dicamptodon aterrimus*) in stream networks. Molecular Ecology. 19: 898–909.

Murphy, G. 1942. Relationship of the fresh water mussel to trout in the Truckee River. California Fish and Game 28: 89–102.

Narum, S.R.; Campbell, N.R.; Meyer, K.A.; [et al.]. 2013. Thermal adaptation and acclimation of ectotherms from differing aquatic climates. Molecular Ecology. 22: 3090–3097.

Northcote, T.G. 1992. Migration and residency in stream salmonids: Some ecological considerations and evolutionary consequences. Nordic Journal of Freshwater Research. 67: 5–17.

Nussbaum, R.A. 1969. Nests and eggs of the Pacific giant salamander, *Dicamptodon ensatus* (Eschscholtz). Herpetologica. 25: 257–262.

Nusslé, S.; Matthews, K.R.; Carlson, S.M. 2015. Mediating water temperature increases due to livestock and global change in high elevation meadow streams of the Golden Trout Wilderness. PLoS ONE. 10: e0142426.

O'Brien, C.; Blinn, D.W. 1999. The endemic spring snail *Pyrgulopsis montezumensis* in a high CO_2 environment: Importance of extreme chemical habitats as refugia. Freshwater Biology. 42: 225–234.

Peterson, D.P.; Fausch, K.D.; Watmough, J.; [et al.]. 2008. When eradication is not an option: Modeling strategies for electrofishing suppression of nonnative brook trout to foster persistence of sympatric native cutthroat trout in small streams. North American Journal of Fisheries Management. 28: 1847–1867.

Peterson, D.P.; Rieman, B.E.; Horan, D.L.; [et al.]. 2013a. Patch size but not short-term isolation influences occurrence of westslope cutthroat trout above human-made barriers. Ecology of Freshwater Fish. 23: 556–571.

Peterson, D.P.; Wenger, S.J.; Rieman, B.E.; [et al.]. 2013b. Linking climate change and fish conservation efforts using spatially explicit decision support models. Fisheries. 38: 111–125.

Pilliod, D.S.; Peterson, C.R. 2001. Local and landscape effects of introduced trout on amphibians in historically fishless watersheds. Ecosystems. 4: 322–333.

Pilliod, D.S.; Goldberg, C.S.; Arkle, R.S.; [et al.]. 2013. Estimating occupancy and abundance of stream amphibians using environmental DNA from filtered water samples. Canadian Journal of Fisheries and Aquatic Sciences. 70: 1123–1130.

Pilliod, D.S.; Hossack, B.R.; Bahls, P.F.; [et al.]. 2010. Non-native salmonids affect amphibian occupancy at multiple spatial scales. Diversity and Distributions. 16: 959–974.

Poff, N.L.; Brinson, M.M.; Day, J.W.J. 2002. Aquatic ecosystems and global climate change: Potential impacts on inland freshwater and coastal wetland ecosystems in the United States. Online report. Washington, DC: Pew Center on Global Climate Change. http:www.pewclimate.org/projects/aquatic [Accessed September 16, 2016].

Pollock, M.M.; Beechie, T.J.; Wheaton, J.M.; [et al.]. 2014. Using beaver dams to restore incised stream ecosystems. BioScience. 64: 279–290.

Pope, K.L.; Matthews, K.R. 2001. Movement ecology and seasonal distribution of mountain yellow-legged frogs, *Rana muscosa*, in a high-elevation Sierra Nevada basin. Copeia. 2001(3): 787–793.

Rieman, B.E.; Dunham, J.B. 2000. Metapopulations and salmonids: A synthesis of life history patterns and empirical observations. Ecology of Freshwater Fish. 9: 51–64.

Rieman, B.E.; Isaak, D.J. 2010. Climate change, aquatic ecosystems, and fishes in the Rocky Mountain West: Implications and alternatives for management. Gen. Tech. Rep. RMRS-GTR-250. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Rieman, B.E.; McIntyre, J.D. 1993. Demographic and habitat requirements for conservation of bull trout. Gen. Tech. Rep. INT-302. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station.

Rieman, B.E.; McIntyre, J.D. 1995. Occurrence of bull trout in naturally fragmented habitat patches of varied size. Transactions of the American Fisheries Society. 124: 285–296.

Rieman, B.E.; Hessburg, P.F.; Luce, C.; [et al.]. 2010. Wildfire and management of forests and native fishes: Conflict or opportunity for convergent solutions? BioScience. 60: 460–468.

Rieman, B.E.; Isaak, D.; Adams, S.; [et al.]. 2007. Anticipated climate warming effects on bull trout habitats and populations across the interior Columbia River basin. Transactions of the American Fisheries Society. 136: 1552–1565.

Rieman, B.E.; Lee, D.C.; Thurow, R.F. 1997. Distribution, status, and likely future trends of bull trout within the Columbia River and Klamath River basins. North American Journal of Fisheries Management. 17: 1111–1125.

Rieman, B.E.; Peterson, J.T.; Myers, D.L. 2006. Have brook trout (*Salvelinus fontinalis*) displaced bull trout (*Salvelinus confluentus*) along longitudinal gradients in central Idaho streams? Canadian Journal of Fisheries and Aquatic Sciences. 63: 63–78.

Roberts, J.J.; Rahel, F.J. 2008. Irrigation canals as sink habitat for trout and other fishes in a Wyoming drainage. Transactions of the American Fisheries Society. 137: 951–961.

Roscoe, E.J.; Redelings, S. 1964. The ecology of the freshwater pearl mussel *Margaritifera margaritifera* (L.). Sterkiana. 16: 19–32.

Schindler, D.E.; Augerot, X.; Fleishman, E.; [et al.]. 2008. Climate change, ecosystem impacts, and management for Pacific salmon. Fisheries. 33: 502–506.

Sedell, E.R.; Gresswell, R.E.; McMahon, T.E. 2015. Predicting spatial distribution of postfire debris flows and potential consequences for native trout in headwater streams. Freshwater Science. 34: 1558–1570.

Sepulveda, A.J.; Lowe, W.H. 2009. Local and landscape-scale influences on the occurrence and density of *Dicamptodon aterrimus*, the Idaho giant salamander. Journal of Herpetology. 43: 469–484.

Sepulveda, A.J.; Lowe, W.H. 2011. Coexistence in streams: Do source–sink dynamics allow salamanders to persist with fish predators? Oecologia. 166: 1043–1054.

Shepard, B.B.; May, B.E.; Urie, W. 2005. Status and conservation of westslope cutthroat trout within the western United States. North American Journal of Fisheries Management. 25: 1426–1440.

Shepard, B.B.; Spoon, R.; Nelson, L. 2002. A native westslope cutthroat trout population responds positively after brook trout removal and habitat restoration. Intermountain Journal of Sciences. 8: 193–214.

Smith, G.R.; Dowling, T.E.; Gobalet, K.W.; [et al.]. 2002. Biogeography and timing of evolutionary events among Great Basin fishes. Great Basin aquatic systems history. Smithsonian Contributions to the Earth Sciences. 33: 175–234.

Smith, J.; Tirpak, D., eds. 1989. Potential impacts of global climate change on the United States. Report to Congress, PM-221. Washington, DC: U.S. Environmental Protection Agency, Office of Policy, Planning and Evaluation, Office of Research and Development. 689 p.

Stagliano, D.M. 2005. Freshwater mussels in Montana: Comprehensive results from 3 years of SWG funded surveys. Report to the Montana Department of Fish, Wildlife and Parks. Helena, MT: Montana Natural Heritage Program. 74 p.

Stewart, I.T.; Cayan, D.R.; Dettinger, M.D. 2005. Changes toward earlier streamflow timing across western North America. Journal of Climate. 18: 1136–1155.

Stone, J.; Barndt, S.; Gangloff, M. 2004. Spatial distribution and habitat use of the western pearlshell mussel (*Margaritifera falcata*) in a western Washington stream. Journal of Freshwater Ecology. 19: 341–352.

Thomsen, P.; Kielgast, J.O.S.; Iversen, L.L.; [et al.]. 2012. Monitoring endangered freshwater biodiversity using environmental DNA. Molecular Ecology. 21: 2565–2573.

University of Washington, Climate Impacts Group. 2010. Pacific Northwest (PNW) Hydroclimate Scenarios Project (2860). http://warm.atmos.washington.edu/2860 [Accessed September 29, 2017].

USDA Forest Service [USDA FS]. [n.d.a]. Climate Shield cold-water refuge streams for native trout. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Air, Water, & Aquatic Environments Program. http://www.fs.fed.us/rm/boise/AWAE/projects/ClimateShield.html [Accessed April 26, 2017].

USDA Forest Service [USDA FS]. [n.d.b]. NorWeST regional database and modeled stream temperatures. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Air, Water, & Aquatic Environments Program. www.fs.fed.us/rm/boise/AWAE/projects/NorWeST.html [Accessed April 21, 2017].

USDA Forest Service [USDA FS]. [n.d.c]. Western U.S. stream flow metrics. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Air, Water, & Aquatic Environments Program. http://www.fs.fed.us/rm/boise/AWAE/projects/modeled_stream_flow_metrics.shtml [Accessed April 26, 2017].

USDOI Fish and Wildlife Service [USFWS]. 1985. Paiute cutthroat trout (*Oncorhynchus clarki seleniris*) recovery plan. Portland, OR: U.S. Department of the Interior, Fish and Wildlife Service. 117 p.

USDOI Fish and Wildlife Service [USFWS]. 1995. Lahontan cutthroat trout (*Oncorhynchus clarki henshawi*) recovery plan. Portland, OR: U.S. Department of the Interior, Fish and Wildlife Service. 157 p.

USDOI Fish and Wildlife Service [USFWS]. 2014. Revised draft recovery plan for the coterminous United States population of bull trout (*Salvelinus confluentus*). Portland, OR: U.S. Department of the Interior, Fish and Wildlife Service. 151 p.

U.S. Environmental Protection Agency [USEPA]. 2014. Best practices for continuous monitoring of temperature and flow in wadeable streams. EPA/600/R-13/170F. Washington, DC: U.S. Environmental Protection Agency, Global Change Research Program, National Center for Environmental Assessment.

Ver Hoef, J.M.; Peterson, E.E.; Theobald, D.M. 2006. Spatial statistical models that use flow and stream distance. Environmental and Ecological Statistics. 13: 449–464.

Walters, A.W.; Bartz, K.K.; McClure, M.M. 2013. Interactive effects of water diversion and climate change for juvenile Chinook salmon in the Lemhi River Basin (USA). Conservation Biology. 27: 1179–1189.

Wenger, S.J.; Isaak, D.J.; Dunham, J.B.; [et al.]. 2011a. Role of climate and invasive species in structuring trout distributions in the interior Columbia River Basin, USA. Canadian Journal of Fisheries and Aquatic Sciences. 68: 988–1008.

Wenger, S.J.; Isaak, D.J.; Luce, C.H.; [et al.]. 2011b. Flow regime, temperature, and biotic interactions drive differential declines of Rocky Mountain trout species under climate change. Proceedings of the National Academy of Sciences, USA. 108: 14175–14180.

Wenger, S.J.; Luce, C.H.; Hamlet, A.F.; [et al.]. 2010. Macroscale hydrologic modeling of ecologically relevant flow metrics. Water Resources Research. 46: W09513.

Whiteley, A.R.; Hastings, K.; Wenburg, J.K.; [et al.]. 2010. Genetic variation and effective population size in isolated populations of coastal cutthroat trout. Conservation Genetics. 11: 1929–1943.

Wiens, J.J., Ackerly, D.D., Allen, A.P.; [et al.]. 2010. Niche conservatism as an emerging principle in ecology and conservation biology. Ecology Letters. 13: 1310–1324.

Williams, J.E.; Haak, A.L.; Gillespie, N.G.; [et al.]. 2007. The Conservation Success Index: Synthesizing and communicating salmonid condition and management needs. Fisheries. 32: 477–493.

Williams, J.E.; Neville, H.M.; Haak, A.L.; [et al.]. 2015. Climate change adaptation and restoration of western trout streams: Opportunities and strategies. Fisheries. 40: 304–317.

Young, B.E.; Hall, K.R.; Byers, E.; [et al.]. 2012. Rapid assessment of plant and animal vulnerability to climate change. In: Brodie, J.F.; Post, E.S.; Doak, D.F., eds. Wildlife conservation in a changing climate. Chicago, IL: University of Chicago Press: 129–152.

Young, M.K.; Isaak, D.J.; McKelvey, K.S.; [et al.]. 2016. Climate, demography, and zoogeography predict introgression thresholds in salmonid hybrid zones in Rocky Mountain streams. PLoS One. 11: e0163563.

Young, M.K.; Isaak, D.J.; McKelvey, K.S.; [et al.]. 2017. Ecological segregation moderates a climactic conclusion to trout hybridization. Global Change Biology. 12: 521–523.

Young, M.K.; Isaak, D.J.; Spaulding, S.; [et al.]. 2018. Climate vulnerability of native salmonids in the Northern Rockies. In: Halofsky, J.E.; Peterson, D.L.; Dante-Wood, S.K. [et al.], eds. Climate change vulnerability and adaptation in the Northern Rocky Mountains. Gen. Tech. Rep. RMRS-GTR-374. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Young, M.K.; McKelvey, K.S.; Pilgrim, K.L.; [et al.]. 2013. DNA barcoding at riverscape scales: Assessing biodiversity among fishes of the genus *Cottus* (Teleostei) in northern Rocky Mountain streams. Molecular Ecology Resources. 13: 583–595.

Young, Michael K.; Isaak, Daniel J.; McKelvey, Kevin S.; [et al.]. 2017. Species occurrence data from the Rangewide Bull Trout eDNA Project. Fort Collins, CO: U.S. Department of Agriculture, Forest Service Research Data Archive. https://doi.org/10.2737/RDS-2017-0038. http://www.fs.fed.us/rm/boise/AWAE/projects/BullTrout_eDNA.html [Accessed October 25, 2017].

Zurstadt, C. 2015. Deposition of fine sediment in the South Fork Salmon River watershed, Payette and Boise National Forests, Idaho. Unpublished report on file with: Payette National Forest, McCall, ID.

Chapter 6: Effects of Climate Change on Forest Vegetation

Patrick N. Behrens, Robert E. Keane, David L. Peterson, and Joanne J. Ho

Introduction

Projected rapid changes in climate will affect vegetation assemblages in the Intermountain Adaptation Partnership (IAP) region directly and indirectly. Direct effects include altered vegetation growth, mortality, and regeneration, and indirect effects include changes in disturbance regimes (Chapter 8) and interactions with altered ecosystem processes (e.g., hydrology, snow dynamics, nonnative species) (Bonan 2008; Hansen and Phillips 2015; Hansen et al. 2001, 2016; Notaro et al. 2007). Some species may have decreased abundance, whereas others may expand their range (Landhäusser et al. 2010). New vegetation communities may form, and historical vegetation complexes may shift to other areas of the landscape or become rare. The consequences of land management policies and activities, including fire exclusion, fuels treatments, and grazing, interact with potential climate change effects.

Here we assess the effects of climate change on vegetation in the IAP region, based on species autecology, disturbance regimes, current conditions, and modeling results (table 6.1). We summarize how climate change affects vulnerability of important tree species, vegetation types, and resources of concern (box 6.1). We have integrated modeling results with a detailed synthesis of climate change literature for western North America.

This assessment is focused on vegetation types listed in table 6.1, where the vulnerability of each vegetation type is inferred from the aggregate vulnerability of its dominant species (table 6.2). Vulnerability is also considered with respect to heterogeneous landscapes, including both vegetation disturbance and land use history.

All projections of future conditions contain uncertainty (box 6.2). Uncertainty can result from a lack of information or from a disagreement about what is known or predictable. Uncertainty can also result from known and unknown errors. It may have many sources, including quantifiable errors in data, ambiguously defined concepts or terminology, and uncertain projections of human behavior. Uncertainty can be represented by quantitative measures (e.g., a range of values) or by qualitative statements (e.g., judgment of a team of experts).

Climate Change Assessment Techniques

Ecologists have invested considerable effort to project the effects of climate change on ecosystem processes across various scales (Clark et al. 2001; Joyce et al. 2014; Schumacher et al. 2006). Using traditional field methods to explore climate change response is difficult because of the complex interactions between ecological processes, disturbance, and climate at multiple temporal and spatial scales (McKenzie et al. 2014).

Four techniques exist to assess and project the effects of climate change on vegetation and related resource concerns. First, **expert opinion** involves experts in the fields of climate change, ecology, and vegetation dynamics qualitatively assessing what will happen to vegetation under various climate change scenarios. Second, **field assessment** involves monitoring or remote sensing to monitor vegetation change as the climate warms. Field sampling involves establishing plots across the landscape, detecting change between plot measurements, and correlating these changes to climate data. Demographic studies track individuals over time, rather than using plot-scale inventories, to understand the role of climate relative to other factors. The U.S. Department of Agriculture Forest Service (USFS) Forest Inventory and Analysis database, the only demographic dataset in the IAP region, has not been analyzed for the interaction of vegetation and climate. Although field assessment techniques are the most reliable and useful, they are often intractable because of the large areas and long time periods for which sampling is needed to detect changes.

Third, **statistical analysis can be used to create empirical models** that project climate change response. Many studies that project habitat, range, or occupational shifts of tree species from climate warming use species distribution models (SDMs; also called bioclimatic envelope models or niche models) to project future geographic ranges (Hansen and Phillips 2015; Iverson and Prasad 2002; Warwell et al. 2007). However, SDMs are inherently flawed for projecting future species distributions because they rely on recent or historical climate-species relationships, resulting in predictions of potential species habitat, not actual species distribution (Iverson and McKenzie 2013). One of the biggest limitations of this approach is that most species distributions are not in equilibrium with climate, thereby causing SDMs to miss areas favorable for a species but where the species is currently absent. In addition, SDMs do not include critical ecological processes (e.g., reproduction,

Table 6.1—Vegetation types included in the IAP vulnerability assessment.

Vegetation type[a]	Description
Subalpine pine forest	Forest communities dominated or co-dominated by bristlecone, limber, and/or whitebark pine for long periods of time. Other co-dominant trees may include subalpine fir, Engelmann spruce, white fir, and aspen.
Subalpine spruce-fir forest	Upland forest communities in which the most shade-tolerant tree capable of occupying the site is subalpine fir, Engelmann spruce, and/or blue spruce. Major seral species include lodgepole pine, aspen, and Douglas-fir.
Mesic mixed conifer forest	Upland forest communities where the most shade-tolerant tree capable of occupying the site is grand fir, white fir, Shasta red fir, mountain hemlock, or Sierra lodgepole pine. Major seral species include lodgepole pine, Douglas-fir, ponderosa pine, Jeffery pine, and aspen.
Dry mixed conifer forest	Upland, lower montane, forest communities where the most shade-tolerant tree capable of occupying the site is Douglas-fir, white fir, or limber pine; and woodland species such as curl-leaf mountain mahogany, Gambel oak, bigtooth maple, pinyons, and junipers are usually present. Ponderosa pine is a major seral species. Lodgepole pine is absent. Aspen is sometimes an important seral species.
Aspen-mixed conifer forest	Upland forest communities where the most shade-tolerant tree capable of occupying the site is a conifer species but aspen is (or was) an important component due to periodic disturbances. Following a disturbance, conifers can return to dominance in less than 150 years.
Persistent aspen forest	Upland forest communities dominated by aspen in which succession to conifer dominance is not possible or takes longer than 150 years.
Ponderosa pine forest	Upland forest communities where ponderosa pine is the only forest tree species capable of occupying the site, or where natural under-burning periodically eliminates other conifers and maintains ponderosa pine dominance.
Riparian forest	Forest communities occurring adjacent to water bodies or around seeps and springs. They may be dominated by any of the species listed above in addition to cottonwoods, willows, alders, birch, or nonnative trees such as saltcedar and Russian olive.

[a]Vegetation types are those used by the U.S. Forest Service Intermountain Region.

tree growth, competitive interactions, disturbance) (Iverson and McKenzie 2013; Watling et al. 2012).

Finally, the most effective technique uses **modeling to assess climate-mediated vegetation responses** (Gustafson 2013; Iverson and McKenzie 2013; McKenzie et al. 2014), incorporating projected future climate into ecological models to simulate climate change effects (Baker 1989; He et al. 2008; Keane et al. 2004; Merriam et al. 1992; Perry and Millington 2008). Many existing models simulate ecological change at broad (global, regional) and fine (point, ecosystem, stand) scales (Bugmann 2001; Cramer et al. 2001). However, models focused on large spatial scales (50–500 square miles) are best suited for projecting climate change effects because most ecosystem processes operate and most management decisions are made at large scales (Cushman et al. 2007; Littell et al. 2011; McKenzie et al. 2014).

To realistically model species composition changes, a mechanistic, process-driven simulation approach is needed to emphasize physical drivers of vegetation dynamics that are directly related to climate (Falk et al. 2007; Gustafson 2013; McKenzie et al. 2014). However, mechanistic model design is complex, containing detailed parameterization of species life histories and physiologies, interacting disturbance factors, and high-resolution modeling over large areas (Lawler et al. 2006). Dynamic global vegetation models operate at scales from regional (hundreds of miles) to global (degrees of latitude and longitude), projecting aggregates of species as life forms or plant functional types, which may not be directly relevant for resource managers (Bachelet et al. 2003; Bonan 2008; Neilson et al. 2005). Most of these models project shifts to more drought-tolerant and disturbance-tolerant species in a warmer climate. In some models, increased water-use efficiency in trees, induced by elevated carbon dioxide (carbon dioxide fertilization), may offset this general shift in vegetation as forests expand into areas where the climate is currently too dry (Bachelet et al. 2003).

Ecosystem models that accurately project climate change effects must simulate disturbances, vegetation, climate, and their interactions across multiple spatial scales (Purves and Pacala 2008), but few models simulate ecosystem processes with the mechanistic detail needed to realistically represent important interactions (Keane et al. 2015b; Riggs et al. 2015) (table 6.2). For example, direct interactions between climate and vegetation may be more realistically represented by simulating the daily dynamics of carbon (photosynthesis, respiration), water (evapotranspiration), and nutrients at the plant level than by simulating vegetation development

Box 6.1—Summary of the Primary Effects of Climate Change on Vegetation Types in the IAP Region

Syntheses of autecological information, empirical data, and modeling were used to identify expected responses of forest vegetation in the IAP region through the end of the 21st century, summarized here for vegetation types (table 6.1)

Subalpine Pine Forest

Highly vulnerable

- Whitebark pine will be especially vulnerable, because warming is expected to exacerbate existing stressors (white pine blister rust, mountain pine beetle, fire exclusion).
- Limber pine, Engelmann spruce, and white fir may grow faster with less snowpack (longer growing season), although limber pine could be stressed by more bark beetles.
- Great Basin and Rocky Mountain bristlecone pines growth may decrease but with high variability among locations.
- Quaking aspen will be minimally affected by a warmer climate, especially compared to aspen at lower elevations.

Subalpine Spruce-Fir Forest

Moderately vulnerable

- Subalpine fir, Engelmann spruce, and blue spruce may grow faster in the upper subalpine zone because of less snowpack (longer growing season).
- Lodgepole pine will be more susceptible to mountain pine beetle.
- Quaking aspen will be minimally affected by a warmer climate, especially compared to aspen at lower elevations.
- Douglas-fir could increase at the lower end of the subalpine zone.
- Increased wildfire could reduce the distribution of all subalpine species except aspen.

Mesic Mixed Conifer Forest

Moderately vulnerable; some winners, some losers

- Douglas-fir, ponderosa pine, and Jeffrey pine (early seral, fire tolerant) may become relatively more common than other (late seral) species that are less fire tolerant, but they will probably grow slower.
- Shasta red fir will grow slower, and distribution may decrease because of increased wildfire.
- Lodgepole pine and quaking aspen, which regenerate rapidly after wildfire, will persist across the landscape, possibly with increased stress from insects and pathogens.

Dry Mixed Conifer Forest

Moderately vulnerable; some winners, some losers

- Curl-leaf mountain-mahogany, Gambel oak, and bigtooth maple can cope with both drier soils (drought tolerant) and increased wildfire (vigorous sprouting), and they may become more abundant in some locations.
- Two-needle pinyon and singleleaf pinyon are sensitive to long periods of drought combined with insects, and they may have reduced growth and some mortality; frequent wildfire may reduce abundance.
- Limber pine may be challenged by a combination of mountain pine beetles, white pine blister rust, and increasing wildfire.
- Douglas-fir and white fir growth will decrease; white fir will be less abundant if wildfire frequency increases.

Aspen-Mixed Conifer Forests

- *Moderately vulnerable, depending on vegetation*
- Mature spruce-fir forest will become less common if wildfire frequency increases.
- At higher elevations, early-seral species such as quaking aspen will become more abundant and possibly more widely distributed.
- At lower elevations, ponderosa pine will persist, and quaking aspen and Gambel oak will become more abundant.

Box 6.1—*continued*.

- Changes in species distribution and abundance will depend on topography (north vs. south aspect, canyons vs. side slopes, etc.).

Persistent Aspen Forests

Moderately vulnerable, depending on vegetation

- Mature spruce-fir forest will become less common if wildfire frequency increases.
- Aspen will maintain dominance because of its ability to sprout after wildfire.
- At lower elevations, ponderosa pine will persist, and quaking aspen and Gambel oak will become more abundant.
- Douglas-fir will probably persist because it has relatively high drought and fire tolerance, but will grow slower.

Montane Pine Forests

Moderately vulnerable

- Ponderosa pine will maintain and probably increase dominance over associated species that are less tolerant of drought and wildfire, but it may grow slower.
- Limber pine and bristlecone pine will probably persist at higher elevations where fuel loads are low.
- If bark beetles become more prevalent, they could increase stress and mortality in pine species, especially during drought periods.

Riparian Forests

Highly vulnerable

- Vegetation dominance will transition to species that are more tolerant of seasonal drought, including ponderosa pine and other deep-rooted conifers.
- Hardwood species that rely on periodic high water levels for regeneration will become less common.
- Riparian forests associated with small or transient water sources (e.g., springs) will be more susceptible than forests near large water sources (e.g., rivers). Low-elevation riparian forests near small water sources will be more susceptible than high-elevation forests with persistent snowpack.
- Saltcedar will persist in riparian areas because it is more drought tolerant than native vegetation, but tamarisk beetle is a promising biocontrol.

annually using state-and-transition modeling approaches (Keane et al. 2015a).

Forest Vegetation Responses to Climate

The effects of climate change on forest vegetation are likely to be driven primarily by vegetation responses to altered disturbance regimes, and secondarily through direct effects on vegetation through shifts in regeneration, growth, and mortality (Dale et al. 2001; Flannigan et al. 2009; Temperli et al. 2013) (box 6.3). Effects on vegetation caused by a changing climate (Chapter 3) will vary over different spatial and temporal scales. Trees will respond to reduced water availability, higher temperatures, and changes in growing season in different ways, but because trees are stationary organisms, altered vegetation composition and structure will be the result of changes in plant processes and responses to disturbance. This section discusses responses of trees and other forest vegetation to projected climate.

Individual Plant Effects

There are several important modes of response of plants to changing climates (Joyce and Birdsey 2000). The first is changes in **productivity**, which could increase in some locations because of increasing temperatures, longer growing seasons, and improved water-use efficiency (Aston 2010; Joyce 1995). The window of successful **seedling establishment** will change (Ibáñez et al. 2007), and increasing drought and high temperatures may narrow the time for effective regeneration in low-elevation forests and widen the window in high-elevation forests. Climate may directly cause **tree mortality** through the effects of increased temperature on moisture stress in trees. Extreme climatic events, such as late growing-season frosts and high winds causing breakage and blowdowns, may increase because of projected increases in climatic variability (Notaro 2008), and these events may cause mortality (Joyce et al. 2014; Vanoni

Table 6.2—Dominant tree species in each vegetation type (see table 6.1) in each IAP subregion. Indicator species are shown in bold text.

Vegetation type[a]	Middle Rockies	Greater Yellowstone	Uintas and Wasatch Front	Plateaus	Great Basin	Intermountain Semi Desert
Subalpine pine forest	PNV[b] **Whitebark pine**	PNV **Whitebark pine**	PNV **Whitebark pine** **Limber pine**	PNV **Limber pine** **GB bristlecone pine**[c]	PNV **Limber pine** **GB bristlecone pine**	
Subalpine spruce-fir forest	PNV **Subalpine fir** **Engelmann spruce** Seral Lodgepole pine Douglas-fir Western larch	PNV **Subalpine fir** **Engelmann spruce** Seral Lodgepole pine Douglas-fir	PNV **Subalpine fir** **Engelmann spruce** Seral Lodgepole pine Douglas-fir	PNV **Subalpine fir** **Engelmann spruce** **Blue spruce** Seral Douglas-fir	PNV **Subalpine fir** **Engelmann spruce** **Blue spruce** Seral Lodgepole pine Douglas-fir	
Mesic mixed conifer forest	PNV **Grand fir** **Douglas-fir** Seral **Lodgepole pine** Douglas-fir Ponderosa pine **Western larch**	PNV **Douglas-fir** Seral **Lodgepole pine**	PNV **White fir** **Douglas-fir** Seral **Lodgepole pine** Douglas-fir	PNV **White fir** **Douglas-fir** Seral Douglas-fir Ponderosa pine	PNV **White fir** **Sierra white fir** **Shasta red fir** **Mountain hemlock** **Sierra lodgepole pine** Seral **Sierra lodgepole pine** **Western white pine** Jeffrey pine Ponderosa pine	
Dry mixed conifer forest	PNV **Douglas-fir** **Grand fir** Seral Ponderosa pine Douglas-fir	PNV **Douglas-fir** **Limber pine** Seral Limber pine	PNV **White fir** **Douglas-fir** Seral **Ponderosa pine** **Gambel oak** **Curl-leaf mtn.-mahogany** **Bigtooth maple** **Rocky Mtn. juniper**	PNV **White fir** **Douglas-fir** Seral **Ponderosa pine** **Gambel oak** **Curl-leaf mtn.-mahogany** **Bigtooth maple** **Utah juniper** **Two-needle pinyon**	PNV **White fir** **Sierra white fir** **Douglas-fir** Seral **Ponderosa pine** **Jeffrey pine** **Gambel oak** **Curl-leaf mtn.-mahogany** **Bigtooth maple** **Utah juniper** **Singleleaf pinyon**	

Table 6.2—*Continued*.

Vegetation type[a]	Middle Rockies	Greater Yellowstone	Uintas and Wasatch Front	Plateaus	Great Basin	Intermountain Semi Desert
Aspen-mixed conifer forest	PNV Subalpine fir Engelmann spruce Grand fir Douglas-fir Seral **Aspen** Lodgepole pine Douglas-fir Ponderosa pine	PNV Subalpine fir Engelmann spruce Douglas-fir Seral **Aspen** Lodgepole pine Douglas-fir	PNV Subalpine fir Engelmann spruce White fir Douglas-fir Seral **Aspen** Lodgepole pine Douglas-fir Ponderosa pine Gambel oak Curl-leaf mtn.-mahogany Bigtooth maple	PNV Subalpine fir Engelmann spruce Blue spruce White fir Douglas-fir Seral **Aspen** Douglas-fir Ponderosa pine Gambel oak Curl-leaf mtn.-mahogany Bigtooth maple	PNV White fir Sierra white fir Shasta red fir Seral **Aspen** Sierra lodgepole pine Jeffrey pine Western juniper	
Persistent aspen forest	None	PNV **Aspen** Subalpine fir Engelmann spruce Douglas-fir Seral **Aspen** Lodgepole pine	PNV **Aspen** Subalpine fir Engelmann spruce White fir Douglas-fir Seral **Aspen** Lodgepole pine	PNV **Aspen** Subalpine fir Engelmann spruce Blue spruce White fir Douglas-fir Seral **Aspen** Douglas-fir Ponderosa pine Gambel oak Curl-leaf mtn. mahogany Bigtooth maple	PNV **Aspen** White fir Sierra white fir Shasta red fir Seral **Aspen** Sierra lodgepole pine Jeffrey pine Western juniper	PNV **Aspen** (snow pockets)
Montane pine forest	PNV Douglas-fir **Ponderosa pine** Grand fir **Limber pine** Seral **Ponderosa pine** Douglas-fir **Limber pine**	PNV **Limber pine** Douglas-fir Seral **Limber pine**	PNV **Limber pine** **Ponderosa pine** Douglas-fir White fir **GB bristlecone pine** Seral **Limber pine** **GB bristlecone pine** **Ponderosa pine**	PNV **Limber pine** **GB bristlecone pine** **Ponderosa pine** Douglas-fir White fir Seral **Limber pine** **GB bristlecone pine** **Ponderosa pine**	PNV **Limber pine** **GB bristlecone pine** **Ponderosa pine** **Jeffrey pine** Douglas-fir White fir Shasta red fir Seral **Limber pine** **GB bristlecone pine** **Ponderosa pine** **Jeffrey pine** Douglas-fir	

Table 6.2—*Continued*.

Vegetation type[a]	Middle Rockies	Greater Yellowstone	Uintas and Wasatch Front	Plateaus	Great Basin	Intermountain Semi Desert
Riparian forest	PNV Subalpine fir Engelmann spruce Grand fir Douglas-fir Ponderosa pine Aspen Black cottonwood White alder Sitka alder Thinleaf alder Water birch	PNV Subalpine fir Engelmann spruce Blue spruce Douglas-fir Aspen Lodgepole pine Narrowleaf cottonwood Black cottonwood Balsam cottonwood Water birch Thinleaf alder Boxelder Crack willow	PNV Subalpine fir Engelmann spruce Blue spruce Douglas-fir Ponderosa pine Aspen Lodgepole pine Narrowleaf cottonwood Fremont cottonwood Water birch Thinleaf alder Boxelder Velvet ash Crack willow Salt cedar	PNV Subalpine fir Engelmann spruce Blue spruce White fir Douglas-fir Ponderosa pine Aspen Narrowleaf cottonwood Fremont cottonwood Water birch Thinleaf alder Boxelder Velvet ash Crack willow Salt cedar	PNV Subalpine fir Engelmann spruce Blue spruce White fir Sierra white fir Shasta red fir Sierra lodgepole pine Douglas-fir Jeffrey pine Ponderosa pine Aspen Lodgepole pine Narrowleaf cottonwood Black cottonwood Lanceleaf cottonwood Water birch Thinleaf alder	

[a] Vegetation types are those used by the U.S. Forest Service Intermountain Region.
[b] PNV indicates potential natural vegetation.
[c] "GB bristlecone pine" indicates Great Basin bristlecone pine.

et al. 2016). There will also be disruptions in **phenology** in a warmer climate, with some plants suffering damage or mortality when phenological cues and events are mistimed with new climates (e.g., flowering during dry portions of the growing season) (Cayan et al. 2001). In addition, the **genetic limitation** of species or trees to respond to climate change will vary greatly among species and populations (Hamrick 2004). For example, species restricted to a narrow range of habitat conditions may become maladapted to new climates (St. Clair and Howe 2007).

Plants can respond to climate-mediated changes in different ways (Aitken et al. 2008). Direct effects of temperature at the cellular level may increase photosynthesis and respiration (Waring and Running 1998). Photosynthesis rates increase with temperature up to an optimum and decline thereafter, although potential effects on tree growth vary by species and local soil and moisture conditions. In the IAP region, any decrease in tree growth would be expected to occur at low elevations, whereas some trees at high elevations may have increased growth. Respiration increases with temperature, and respiration occurs even when stomata are closed, so high temperatures coupled with low water availability may result in high respirational losses with few photosynthetic gains (Ryan et al. 1995).

Increased atmospheric carbon dioxide levels may also directly modify physiological growth processes at the cellular level. Water-use efficiency may increase for some conifer species, potentially compensating for lower water availability (Waring and Running 1998). Leaf biomass is usually the first to increase as plants attempt to optimize photosynthesis by growing more leaf tissue (i.e., leaf area), although increased leaf area can be transitory depending on available water and nitrogen. Higher atmospheric carbon dioxide levels and temperatures can also interact to increase growth, especially if warmer temperatures are closer to temperature optima for photosynthesis.

Another direct effect of warming temperatures is longer growing seasons (Cayan et al. 2001; McKenzie et al. 2009). In addition, future climate may be more variable, affecting dormancy regulation, bud burst, and early growth (Hanninen 1995; Harrington et al. 2010). Plant phenological cues may be disrupted or triggered inappropriately because of high weather variability, a response that may be fatal for seedlings. Warmer temperatures may reduce growing-season frosts in mountain valleys, thereby allowing more frost-susceptible species, such as ponderosa pine (*Pinus ponderosa*) to exist in habitats currently occupied by lodgepole pine (*Pinus contorta* var. *latifolia*), subalpine fir (*Abies lasiocarpa*), and Engelmann spruce (*Picea engelmannii*). Increased temperatures may result in decreased winter chilling, which could result in delayed bud burst, reduced flowering, and reduced seed germination (Chmura et al. 2011).

Box 6.2—Uncertainty and Climate Change Effects on Vegetation

Global Climate Models (GCMs) that project rapidly warming climates have a high degree of uncertainty. Although it is clear that increasing atmospheric carbon dioxide will cause a significant increase in temperature (IPCC 2007), uncertainty exists about the magnitude and rate of climate change (Roe and Baker 2007; Stainforth et al. 2005). This uncertainty is generally higher for climate projections made at fine resolutions and for longer time periods (Knutti and Sedlacek 2013). The range of possible projections of future climate from GCMs (anywhere from a 1.6 to 8 °C increase in global average annual temperature) is much greater than the variability of climate over the past 3 centuries (Stainforth et al. 2005), and the variability across GCMs is greater than the variability in each model's climate projections.

Because it is impossible to know whether society will respond to climate change by employing technological innovations to minimize carbon dioxide emissions or to mitigate its effects, most GCMs also simulate a range of scenarios that capture different strategies and socioeconomic policies to deal with climate change, introducing yet another source of uncertainty. Moreover, it is the high variability of climate extremes, not the gradual change of average climate, that will drive most ecosystem responses to disturbance and plant dynamics—and these rare, extreme events are the most difficult to project (Easterling et al. 2000).

Yet another source of uncertainty is introduced when we try to project how the Earth's vegetation and ecosystems will respond to climate change (Araujo et al. 2005). Mechanistic ecological simulation of climate, vegetation, and disturbance dynamics across landscapes is still evolving (Keane and Finney 2003; Sklar and Costanza 1991; Walker 1994). Many current ecosystem simulation models are missing important direct interactions of disturbance, hydrology, and land use with climate that will affect plant distributions (Notaro et al. 2007). Little is known about the interactions among climate, vegetation, and disturbance, and interactions among different disturbance regimes (e.g., fire and beetles) could create novel landscape behaviors. It is also difficult to determine how plant and animal reproduction, growth, and mortality will respond to changing climate (Gworek et al. 2007; Ibanez et al. 2007; Keane et al. 2001; Lambrecht et al. 2007). These modeling uncertainties greatly increase as projections are made further into the future and at finer spatial scales (Xu et al. 2009).

Uncertainties need to be considered when using this assessment for analysis, planning, and project management. Sometimes there is less uncertainty in implementing conventional restoration designs than in designing restoration or treatment plans that attempt to account for climate change effects. For example, including climate change in restoration of western larch ecosystems may be more straightforward than for ponderosa pine ecosystems. Because all climate effects will be manifest in different ways on different landscapes, there is no "one-size-fits-all" prescription that can be adopted everywhere.

Much of the water used by trees in mountain forests comes from snowmelt, so amount and duration of snowpack influence regeneration and growth patterns of tree species and forest communities. Warmer temperatures will cause earlier snowmelt, leading to an earlier start of the growing season, and longer periods of low soil moisture during the rest of the growing season. In contrast, less snowpack will create longer growing seasons in subalpine communities where cold and snowpack duration limit tree regeneration and growth, potentially facilitating increased productivity (Peterson and Peterson 2001) and regeneration (Woodward et al. 1994).

Climate change can indirectly affect vegetation by altering mycorrhizal dynamics (Amaranthus et al. 1999). Many trees, particularly in the seedling and sapling stages, need mycorrhizae to survive, especially in areas with chronic water shortage (Mohatt et al. 2008; Walker et al. 1995). Migration of tree species to more favorable sites in future climates may be governed by the ability of mycorrhizae to also populate these areas (Lankau et al. 2015). Mycorrhizal responses following wildfire are important because fire is expected to increase significantly in a warmer climate (Chapter 8). Establishment of trees in burned areas can be delayed for decades or even centuries (Little et al. 1994), as both mycorrhizae and trees revegetate the area (Schowalter et al. 1997).

Migrating to a new site has historically been the main response of plants to climate change (Huntley 1991), requiring that species have the ecological ability to quickly occupy available sites and the genetic capacity to survive and reproduce successfully (Davis et al. 2005). Most tree species in the IAP region are long lived and genetically diverse, so they can survive wide fluctuations of weather, but the interaction of increasing drought and modified disturbance regimes will play a role in the future distribution and abundance of forest species (Allen et al. 2010) (Chapter 8).

A warmer climate is expected to facilitate upward shifts in the elevation distribution of plant species. For example, Lenoir et al. (2008) found that some plant species have moved upward in elevation at a rate of 100 feet per decade, but it is unclear whether such shifts will drive long-term changes in forest communities. For example, wildfire plays a dominant role in most ecosystems in western North America, and increasing wildfire frequency and extent may overwhelm potential shifts in forest species distribution. The potential for tree populations to migrate may vary among diverse mountain ranges, depending on local biophysical conditions.

Box 6.3—How Do Climate Change Vulnerability Assessments and Adaptation Inform Ecological Restoration?

In an ideal sense, *ecological restoration* is defined as the practice of reestablishing historical plant and animal communities in a given area and the renewal of ecosystem and cultural functions necessary to maintain these communities now and into the future (Egan and Howell 2001). However, this ideal may be difficult to manage because: (1) little is known about historic conditions, (2) many key species may already be lost, (3) some efforts may be prohibitively expensive, and (4) future climates will create novel ecosystems. As a result, The Society for Ecological Restoration has opted for a definition that states that ecological restoration is "the process of renewing and maintaining ecosystem health."

The U.S. Forest Service manual direction (FSM 2020) includes objectives and a policy for restoration:

- *Restore and maintain ecosystems* that have been damaged, degraded, or destroyed by reestablishing the composition, structure, pattern, and ecological processes.
- *Manage for resilient ecosystems* that have a greater capacity to withstand stressors, absorb and recover from disturbances, and reorganize and renew themselves, especially under changing and uncertain environmental conditions.
- *Achieve long-term ecological sustainability* and provide a broad range of ecosystem services to society.

The Forest Service emphasizes ecosystem restoration across all National Forest System lands with the goal of attaining resilient ecosystems. All strategic plans, including the Forest Service Strategic Plan and land management plans, must include goals and objectives to sustain the resilience and adaptive capacity of aquatic and terrestrial ecosystems by reestablishing, maintaining, or modifying their composition, structure, function, and connectivity. The goals and objectives must be established within this framework as defined by laws, Indian treaties and Tribal values and desires, and regulations. The goals and objectives must also consider public values and desires, social concerns, economic sustainability, the historical range of variability, ecological integrity, current and likely future ecological capabilities, a range of climate and other environmental change projections, the best available scientific information, and technical and economic feasibility to achieve desired conditions for National Forest System lands.

A primary element of an integrated approach is to identify and eliminate or reduce stressors that degrade or impair ecosystems. Restoration activities should also take into account social and ecological influences at multiple scales and incorporate the concept of a dynamic system and ecological trajectory. Some ecosystems may have been altered to such an extent that reestablishing components of the historical range of variability may not be ecologically or economically possible. Therefore, goals and activities can focus on restoring the underlying processes that create functioning ecosystems.

Functional restoration, an alternative concept used in the Forest Service, is defined as the "restoration of abiotic and biotic processes in degraded ecosystems." Functional restoration focuses on underlying processes that may be degraded, regardless of the structural condition of the ecosystem. As contrasted with ecological restoration that tends to seek a historical reference condition, functional restoration focuses on dynamic processes that drive structural and compositional patterns. Functional restoration aims to restore functions and improve structures with a long-term goal of restoring interactions between function and structure. However, a functionally restored system may look quite different than the historical reference condition in terms of structure and composition. In this case, disparities cannot be easily resolved, because a threshold of degradation has been crossed, or environmental drivers (e.g., climate) that influenced structural and compositional development have changed.

Reproduction

Cone and seed crops of some tree species could be affected by climate change (Ibáñez et al. 2007; LaDeau and Clark 2001). Low-elevation, xeric forests may have fewer and smaller cone crops because of increased stand density and water stress. Cone crops may also have a lower percentage of viable seed because of increased stress. Infrequent cone crops coupled with low seed production may cause reduced regeneration in recently burned areas, in some cases resulting in dominance of nonforest vegetation. The opposite may be true in higher, colder environments, where increased temperatures will increase growing season length and thereby increase potential for more cone crops with more seeds. Spruce-fir communities may produce so much seed that they overwhelm regeneration of other conifers, especially after mixed-severity fires. Species such as whitebark pine (*Pinus albicaulis*) and lodgepole pine have unique cone characteristics; whitebark pine cones require birds for seed dispersal, and lodgepole pine cones may be serotinous and opened only by fire.

Growth and Mortality

Climate adversely affects growth and mortality through decreased water availability, resulting in shorter effective growing seasons (Bugmann and Cramer 1998; Chmura et al. 2011; Keane et al. 2001; Williams et al. 2010). Extended droughts require conifers to close stomata longer to conserve water. Ponderosa pine and limber pine (*Pinus flexilis*)

have excellent stomatal control, and stomata can remain closed for long periods of time; Douglas-fir (*Pseudotsuga menziesii*) has poor stomatal control, which can drive leaf water potentials to low values and contribute to physiological damage (Sala et al. 2005). If photosynthetic production cannot exceed respiration demands, then plants become stressed.

If physiological damage is high enough, carbon storage in plant cells may decline as a result of stomatal closure and insufficient carbon assimilation to meet demands for tissue maintenance. In addition, lack of water for uptake, especially while stomata are open, can greatly reduce hydraulic conductance (McDowell et al. 2008; Sevanto et al. 2014). Both of these physiological responses to low water supply, which typically occur during prolonged drought, can substantially reduce vigor, making weakened trees more susceptible to other stresses. In the most extreme cases, the ultimate outcome is tree mortality, often facilitated by bark beetles or other insects.

In mesic ecosystems in the IAP region, a warmer climate may enhance growth and decrease mortality (Wu et al. 2011). Earlier growing seasons with ample moisture, as projected for some forests, may promote increased productivity. This will be especially true at higher elevations where cold temperatures, not moisture, limit tree growth. Increased biomass will also amplify competition between species, thereby favoring shade-tolerant individuals in the absence of disturbance. Increased biomass could also reduce resistance to forest insect and diseases (Chapter 8).

Regeneration

Microsite conditions required for successful establishment of tree species are typically rare, so seed germination and survival, especially for seeds that are wind dispersed, are rarely successful (Anderson and Winterton 1996). Suitable moisture conditions must persist for long periods of time for seed germination and early seedling growth. In dry forests, most of the successful regeneration occurs in years when soils are moist for an adequate time and heating at the soil level does not kill developing leaves and stems. A warmer climate may decrease the frequency of high-regeneration years, and regeneration may become rare on the driest sites. In contrast, regeneration may be enhanced by warming at high elevation because earlier snowmelt will provide more time for seedlings to survive and grow (Butler 1986).

During mild winters, seed chilling requirements may not be met for some species, thereby reducing germination. In addition, germination may be delayed to drier times during the growing season. For example, Nitschke and Innes (2008) found that in a warmer climate, chilling requirements were not met for most low-elevation tree species in British Columbia. High soil temperatures can stress both germinants and established seedlings (Rochefort et al. 1994). Climate change may also affect the dispersal properties of seeds. For example, rodent and bird species that disperse seeds may shift habitats because of climate-mediated changes (Tomback 1998). Longer and drier summers and autumns suggest that seed dispersal may occur when the ground and litter are dry and unsuitable for seed germination and establishment (Neilson et al. 2005).

Genetics and Species Adaptation

Climate affects plant phenotypes and is an agent of natural selection. Plant adaptations to local environments have often developed a **clinal** (or continuous) response to abiotic and biotic factors. In addition, **ecotypic** (or discontinuous) response to environmental gradients may play an important role, depending on local soils and topography. Therefore, a combination of clinal and ecotypic environmental gradients determines long-term plant survival and persistence across the landscape.

Natural selection, migration, genetic drift, and mating system determine species genetic composition. Thus, the ability of plant populations to respond to climate change is influenced by underlying patterns of genetic variation. Molecular markers can reveal significant genetic diversity and divergence among populations. Populations may diverge because of fire, volcanic activity (Hansen 1949), glaciation (Hamrick 2004), seed dispersal agents (Lorenz and Sullivan 2009), and pollinator history. Plants that are pollinated by insects or rely on animals to disperse seed are more vulnerable to climate change than plants with wind-dispersed seed, because of the requirement for interaction with another organism.

Genetic diversity allows species to adapt to changing environments, colonize new areas, occupy new ecological niches, and produce substantial and robust progeny that persist in the long term (Ledig and Kitzmiller 1992). Populations within a species adapt to environmental change over time. Species and populations of plants most vulnerable to climate change are typically (1) rare species or genetic specialists, (2) species with limited phenotypic plasticity, (3) species or populations with low genetic variation, (4) populations with low dispersal or colonization potential, (5) populations at the trailing edge of a species range, (6) populations at the lower-elevation limit of their distribution, and (7) populations threatened by habitat loss, fire, insects, or disease (Spittlehouse and Stewart 2003; St. Clair and Howe 2011). The ability of a species to respond to environmental change is closely tied to its adaptive strategy (e.g., specialist or generalist) (table 6.3), mechanisms that shape its genetic structure, and the rate of environmental change.

Fragmentation is a critical issue for plant populations because isolation and small populations promote inbreeding and loss of genetic diversity (Broadhurst et al. 2008; Potter et al. 2015). Gene flow from adjacent populations can increase the rate of adaptation by introducing genetic variation that is preadapted to warmer or drier climates (Aitken et al. 2008). This knowledge allows resource managers to select an appropriate population or seed source to increase the likelihood of desired revegetation or restoration (box 6.4).

Table 6.3—Summary of attributes characterizing plant species' adaptive strategies.[a]

	Adaptive strategy	
Attributes	**Specialist**	**Generalist**
Factor controlling phenotypic expression of adaptive traits	Genotype	Environment
Mechanisms for accommodating environmental heterogeneity	Genetic variation	Phenotypic plasticity
Range of environments where physiological processes function optimally	Small	Large
Slope of clines for adaptive traits	Steep	Flat
Partitioning of genetic variation in adaptive traits	Mostly among populations	Mostly within populations

[a]Modified from Rehfeldt (1994).

Box 6.4—Using Historical Range and Variability to Assess and Adapt to Climate Change

To effectively implement ecosystem-based management, land managers often find it necessary to obtain a reference, or benchmark, to represent the conditions that describe fully functional ecosystems (Cissel et al. 1994; Laughlin et al. 2004). Contemporary conditions can be evaluated against this reference to determine status, trend, and magnitude of change, and to design treatments that provide society with valuable ecosystem services while returning declining ecosystems to a more sustainable condition (Hessburg et al. 1999; Swetnam et al. 1999). Reference conditions are assumed to represent the dynamic character of ecosystems and landscapes, varying across time and space (Swanson et al. 1994; Watt 1947).

The concept of *historical range and variability* (HRV) was introduced in the 1990s to describe past spatial and temporal variability of ecosystems (Landres et al. 1999), providing a spatial and temporal foundation for planning and management. HRV has sometimes been equated with "target" conditions (Harrod et al. 1999), although targets can be subjective and somewhat arbitrary, representing only one possible situation from a range of potential conditions (Keane et al. 2009). HRV encompasses a full range of conditions that have occurred across multiple spatial and temporal scales.

HRV represents a broad historical envelope of possible ecosystem conditions—burned area, vegetation cover type area, patch size distribution—that can provide a time series of reference conditions. This assumes that:

- Ecosystems are dynamic, not static, and their responses to changing processes are represented by past variability
- Ecosystems are complex and have a range of conditions within which they are self-sustaining, and beyond this range they transition to disequilibrium (Egan and Howell 2001)
- Historical conditions can serve as a proxy for ecosystem health
- Time and space domains that define HRV are sufficient to quantify observed variation
- Ecological characteristics being assessed for an ecosystem or landscapes match the management objective (Keane et al. 2009).

The use of HRV has been challenged because a warmer climate may permanently alter the environment of ecosystems beyond what was observed under historical conditions (Millar et al. 2007a), particularly altered disturbance processes, shifts in plant species distribution, and hydrologic dynamics (Notaro et al. 2007). However, a critical evaluation of possible alternatives suggests that HRV is still a viable approach in the near term because it has relatively low uncertainty.

An alternative to HRV is projecting future landscape characteristics in a changing climate using complex empirical and mechanistic models. However, the range of projections for future climate from commonly used GCMs is quite broad (Chapter 3; Stainforth et al. 2005). Additional uncertainty accrues from unknown technological advances, behavioral adaptations, and human population growth (Schneider et al. 2007). Moreover, variability of climate extremes, not the gradual change of average climate, will drive most ecosystem response to climate-mediated disturbance and plant dynamics (Smith 2011). Despite these uncertainties, it will be useful to quantify *future range and variability* (FRV) for landscapes where it is feasible and appropriate (Araujo et al. 2005; Keane et al. 2009).

Given cumulative uncertainties, time series of HRV may have lower uncertainty than simulated projections of future conditions, especially because large variations in past climates are already captured in the time series. It may be prudent to wait until simulation technology has improved enough to create credible FRV landscape pattern and composition. In the meantime, attaining HRV would be a significant improvement in the functionality of most ecosystems in the IAP region, and would be unlikely to result in negative outcomes from a management perspective. As with any approach to reference conditions, HRV is useful as a guide, not a target, for restoration and other management activities.

Some species may not be able to migrate quickly enough to keep pace with projected rates of climate change (30–300 feet per year) (Davis 1989; Malcolm et al. 2002). Slow rates of migration may be further impeded by landscape fragmentation (Davis and Shaw 2001; Davis et al. 2005). Therefore, adaptation may be a more important response to climate change than migration. Some authors suggest that long-lived species with high levels of genetic variation can respond favorably to climate change (Hamrick 2004; Hamrick et al. 1992). However, others dispute the ability of forest trees to adapt or migrate and suggest trees may be restricted by their long lifespans, generation intervals, and juvenile phases (Etterson and Shaw 2001; Jump and Peñuelas 2005; Parmesan 2006). Because plant populations are genetically adapted to local climates, the climatic tolerance of individual populations will be critical.

Adaptive strategies for conifers in the IAP region are well documented (Rehfeldt 1994). Differences in adaptive strategy can be characterized by varietal modifications (e.g., *Pinus ponderosa* var. *ponderosa* versus var. *scopulorum*), different elevations, and variable geography. For example, *P. ponderosa* var. *ponderosa* is characterized as having an intermediate (neither generalist nor specialist) adaptive strategy, but at high elevation it has a specialist strategy (genetic variation is organized into numerous local populations, finely tuned to site-specific gradients). Rocky Mountain Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) is characterized as having a specialist adaptive strategy, but at high elevation it has a generalist adaptive strategy (genetic variation is organized into one or a few populations capable of surviving, growing, and reproducing over a broad range of environments) (Rehfeldt 1989). A generalist adaptive strategy is considered more beneficial for responding to climate change (table 6.3).

Patterns of adaptive variation in other native plants (shrubs, forbs, grasses, sedges) are both clinal and ecotypic. These patterns involve multiple life forms (annual, biennials, perennials) and different ploidy levels (multiple copies of DNA, such as 4X, 6X, or 8X), where 2X is the base level, in which one copy of DNA is inherited on both the maternal and paternal sides. Grasses are largely generalists and less vulnerable to climate change, although ecotypic variation can overlie the generalist adaptive strategy. Forbs, which are mostly insect pollinated and coupled with longer growing seasons and changes in phenology, are considered more vulnerable to climate change than trees and grasses.

Stressors: Biotic and Abiotic Disturbances

A warming climate will rarely be the direct agent of change for tree species and communities. Most changes in vegetation will occur in response to disturbance or some combination of other stressors to climate change (Keane et al. 2015a; McKenzie et al. 2009; Peterson et al. 2014a, b). The biggest changes across the IAP region are likely to be altered water balance and increasing disturbances such as wildfire, insects, and nonnative species (Chapter 8). Disturbances in combination with other stressors (e.g., drought) will create disturbance regimes in which multiple factors interact to modify ecosystem structure and function (Iverson and McKenzie 2013; McKenzie et al. 2009).

Wildfire is pervasive throughout forest ecosystems in western North America and was historically a dominant landscape disturbance agent in the IAP region. Fire exclusion since the 1920s has disrupted annual occurrence, spatial extent, and cumulative area of wildfires, resulting in increased surface fuel loads, tree densities, and ladder fuels, especially in low-elevation, dry conifer forests. Wildfire regimes, defined by fire frequency, annual area burned, severity, and pattern, are greatly influenced by variability in landscape environmental conditions including vegetation distribution, climate, weather, and topography (McKenzie et al. 2011). Regionally, years with high area burned are correlated with drought, so if drought increases as expected, area burned is expected to increase significantly (McKenzie et al. 2004; Peterson et al. 2014a).

Fire history determines composition and structure of most forests in the IAP region. At the lowest and driest elevations, frequent surface fires historically consumed litter and dead wood and killed seedlings and smaller trees. Fuel accumulations over several decades indicate that future fires may be larger and more intense and may cause higher rates of tree mortality than historical fire (box 6.3). Fire exclusion has not affected fire regimes as much where fires were historically infrequent because of relatively cool, wet conditions (e.g., high elevation) (Romme and Despain 1989; Schoennagel et al. 2004). However, earlier onset of snowmelt, predicted to occur with changing regional climate, will reduce fuel moisture, making these systems flammable for longer periods of time and potentially leading to increased area burned (Miller et al. 2009).

Fire exclusion has resulted in increased tree regeneration and denser forest canopies, coupled with accumulation of understory and canopy fuels in dry forests (Ferry et al. 1995; Keane et al. 2002) (fig. 6.1). These conditions create competition for water, light, and nutrients, making trees in fire-excluded forests susceptible to mortality from biotic and abiotic stressors, such as insects (Anderegg et al. 2012; Wikars and Schimmel 2001), drought (Allen et al. 2010), and fire (Hood et al. 2007).

Native insects and diseases naturally occur throughout forest cover types of the IAP region (Chapter 8). The level of insect and disease activity fluctuates with the availability of host material, stand conditions, environmental factors, and abundance of parasites and predators. These agents typically occur at endemic levels within forest ecosystems and affect mature and weakened trees.

Climate and forest composition and structure influence insect activity and outbreaks. Mountain pine beetle (*Dendroctonus ponderosae*) is an integral component of forest ecosystem processes because of its role in stand



Figure 6.1—Area where fire has been excluded for many decades. Dense stands of ponderosa pine and other species create fuel ladders that can facilitate crown fires (photo: U.S. Forest Service).

thinning and redistribution of resources and nutrients. It is responsible for tree mortality across large areas (Logan et al. 2003), causing significant ecological and economic impacts. Many bark beetle life history traits that influence population success are temperature dependent (Bentz and Jönsson 2015); warming temperatures have directly increased bark beetle-caused tree mortality in some areas of western North America (Safranyik et al. 2010; Weed et al. 2015) (fig. 6.2). Temperature increases will affect tree distribution and tree vigor (Chapman et al. 2012; Hart et al. 2013). Therefore, future bark beetle-caused tree mortality will depend not only on the spatial distribution of live host trees and heterogeneity of future landscapes but also on the ability of beetle populations to adapt to changing conditions.

Fungal diseases, dwarf mistletoes (*Arceuthobium* spp.), root diseases, needle casts and blights, and abiotic diseases affect forest ecosystems, although the effects of climate change on forest diseases are difficult to project. The effects of climate change on root disease contribute significantly to mortality and loss of tree vigor, although little is known about climate-disease relationships. Climate-mediated changes to forest tree diseases will be dictated by



Figure 6.2—Stand containing lodgepole pine killed by mountain pine beetle. This insect has killed lodgepole pine across large areas of western North America, including the Intermountain Adaptation Partnership region, during the past 20 years. Chronic damage from the beetle may become more common in a warmer climate (photo: U.S. Forest Service).

disease and host tree responses to new climates and their interactions (Sturrock et al. 2010). Interactions among biotic diseases, abiotic stressors, and host species will drive future pathogen outbreaks.

Soil characteristics, aspect, elevation, and forest stand structure contribute to effective moisture availability for tree establishment and growth, helping to shape spatial patterns of forests. Global climate models (GCMs) indicate that the IAP region will have longer, warmer summers (Chapter 3). Seral species such as ponderosa pine, which can establish on bare soil where high surface temperatures (up to 150 °F) exclude other species, have deep roots that can reach water and avoid competition with shallow-rooted species. In the absence of disturbance, shade-tolerant tree species can establish and grow in the understory, allowing them to take up water from the nutrient-rich soil surface. Leaf surface area increases over time, with leaf areas in excess of 6 square feet per square foot of soil surface area in some forests. Transpiration also increases over time, with the potential to deplete soil water needed to keep trees hydrated throughout the summer.

Climate Change Assessment for Tree Species

Here we assess vulnerability for tree species, vegetation types, and resources of concern in the IAP region, based on (1) ecological characteristics, (2) disturbance interactions, (3) current and historical conditions, and (4) potential climate change responses (table 6.4). Most of the material in this section was derived from published literature, although observational information is included for context. Scientific literature on climate change effects is limited for some species and forest types, making it necessary in some cases to augment the literature with expert knowledge to develop inferences.

Tree Species

Tree species in the IAP region will respond to climate change through modification, contraction, and expansion. First, a species could increase or decrease in productivity in situ within its current range because of increasing temperatures and adequate precipitation (**modification**, or acclimatization). Second, a species may diminish or be extirpated, if conditions change enough to become inhospitable to that species (Allen et al. 2010) (**contraction**). Finally, a species could migrate to areas that are more conducive to establishment and growth (Johnstone and Chapin 2003) (**expansion**). Any species can have multiple modes of response to climate change, and most species will respond to future climates via all three modes.

Application of these three modes to determine future species dynamics requires integration of variability and scale. For example, assessment of species migration requires a long temporal scope to evaluate species range shifts (Prentice et al. 1991). A tree species could become

Table 6.4—Categories used to assess vulnerability of species and vegetation types.

Evaluation category	Description	Example
Habitat, ecosystem function, or species	Specific biophysical or social entity of interest	Whitebark pine
Broad-scale climate change effect	Overarching change in climate that is expected to affect a resource	Warming temperatures
Current condition, existing stressors	Current status of resource relative to desired conditions, including factors that are reducing the quality or quantity of the resource	Reduced abundance, wildfire, mountain pine beetle, white pine blister rust
Sensitivity to climatic variability and change	Specific sensitivity of a habitat, species, or ecosystem function that responds to climate	Low ability to compete with encroaching conifers
Expected effects of climate change	How specific habitat, species, or ecosystem function is expected to respond to climate change (develop inferences from model projections and known responses to climatic variability)	Regeneration may be reduced by combination of warming and low seed availability
Adaptive capacity	Ability to adjust to climate change, to moderate potential damages, or to cope with the consequences; usually more appropriate for species than for systems and processes	Variable: unable to compete with other tree species, but bird-mediated seed dispersal allows rapid colonization of burned areas
Exposure	The extent to which each species' physical environment will change	High

established in a "new" environment made suitable by climate change, such as subalpine tree expansion, but variability in climate may prevent long-term establishment. In addition, shifts in species distribution and abundance will be governed primarily by disturbance, not competition, so disturbance adaptations will be more important than climatic niches.

Most of the information on vulnerability of tree species to climate change was derived from recent summaries on projected climate change effects (Bollenbacher 2012; Devine et al. 2012; Keane et al. 2015a) and older literature on autecology and silviculture (Burns and Honkala 1990; Minore 1979). The following summaries integrate genetic, morphological, ecological, and disturbance characteristics to project how a tree species will respond to a warmer climate.

In general, the literature is inconsistent on the response of tree species to climate change. Results from SDMs often differ from other sources that include gap modeling, mechanistic ecosystem simulation, and field data summaries. As a result, we do not emphasize SDM results in assessment evaluations. Most climate change studies project few species changes after moderate warming (e.g., B1, B2, A1B, RCP 4.5 scenarios) but major species shifts under the most extreme emissions scenarios (e.g., A1, RCP 8.5). Timeframe also affects inferences about vulnerability. Management timeframes of 10 to 50 years are not long enough to effectively evaluate changes in wildfire, native insects, and tree growth because ecosystem response to disturbance may require two to five times the disturbance return interval. Finally, projections by GCMs vary, so the magnitude and rate of climate change, especially by the end of the 21st century, are uncertain (but are always considerably warmer). We have confidence in these projections at broad spatial scales, but less confidence for specific locations.

Douglas-fir (Pseudotsuga menziesii)

(Middle Rockies, Southern Greater Yellowstone, Uintas and Wasatch Front, Plateaus, Great Basin and Semi Desert subregions)

Autecology

Douglas-fir (fig. 6.3) is found throughout the IAP region, growing in pure and mixed conifer stands (Hermann and Lavender 1990), often associated with ponderosa pine, Jeffrey pine (*Pinus jeffreyi*), grand fir (*Abies grandis*), subalpine fir, and quaking aspen (*Populus tremuloides*). Regeneration is most successful where Douglas-fir is seral (Ryker and Losensky 1983), and seedling growth is strongly limited by moisture and competing vegetation. Douglas-fir is intermediate in shade tolerance, tolerating drought better than most competitors (except for ponderosa pine and Jeffrey pine) by keeping stomata open to extract soil water at low soil water potentials (Sala et al. 2005; Stout and Sala 2003). The species exhibits high genetic differentiation, which is strongly associated with geographic or topographic features (Rehfeldt 1978). Seed sources on south aspects have adaptive characteristics for a shorter growing season and drier soils and may survive under drought stress better than seedlings from north aspects.

Disturbance Interactions

Mature Douglas-fir is resistant to fire injury because of its thick bark, deep main roots, and high crowns (Ryan and Reinhardt 1988). Ponderosa pine and western larch can survive fire across all life stages, so on sites with frequent fires where Douglas-fir is associated with these other species, its cover is usually kept low by fire (Agee 1991). Douglas-fir is subject to damage from a variety of agents that may increase under future climates (Hermann and Lavender 1990), including Douglas-fir beetle



Figure 6.3—Douglas-fir. Growth of Douglas-fir in the Intermountain Adaptation Partnership region is expected to decrease in a warmer climate (photo: C. Restaino, used with permission).

(*Dendroctonus pseudotsugae*), western spruce budworm (*Choristoneura occidentalis*), and Douglas-fir tussock moth (*Orgyia pseudotsugata*). The latter two insects attack trees of all ages at periodic outbreak intervals, often resulting in severe defoliation during outbreak years. Armillaria (*Armillaria solidipes*) and annosus (*Heterobasidion annosum*) root diseases may intensify in infection and widen in distribution to cause high tree mortality. Annosus root disease is particularly lethal in Douglas-fir (Hagle 2003). Of the many heart rot fungi (more than 300) attacking Douglas-fir, the most damaging and widespread is red ring rot (*Porodaedalea pini*).

Historical and Current Conditions

Historical frequent wildfire kept Douglas-fir from becoming established on some dry sites where it was associated with ponderosa pine. The cumulative effects of fire exclusion and logging have allowed Douglas-fir to become more dominant across some portions of the IAP region, often with high stem densities in fire-excluded stands. This has created areas where both canopy and surface fuels are high (Keane et al. 2002), predisposing Douglas-fir forests to future crown fires. In addition, these dense stand conditions have contributed to decreased vigor, which makes species susceptible to western spruce budworm and Douglas-fir beetle outbreaks.

Climate Change Responses

Some studies suggest that Douglas-fir distribution will increase in a warmer climate (Morales et al. 2015) and that growth will increase (Soulé and Knapp 2013), although a recent study provides convincing evidence that growth will decrease throughout its range (Restaino et al. 2016). It is likely that multiple factors will contribute to reduced distribution and vigor of Douglas-fir forests in some locations. Increased heat loading following severe wildfires is expected to be more common in the future, and may reduce Douglas-fir regeneration at lower-elevation sites and on south aspects (Kemp 2015). Douglas-fir may also face increasing competition from ponderosa pine, which is more drought tolerant (Stout and Sala 2003), and it may not have the genetic potential to rapidly migrate to more conducive sites (Aitken et al. 2008). In addition, Douglas-fir could have less resistance to the native insects previously mentioned if it is chronically stressed by low soil moisture. Increased wildfires, coupled with adverse effects of fire exclusion, could reduce tree survival in the future and make trees more susceptible to Douglas-fir beetle (Hood and Bentz 2007; Hood et al. 2007). Klopfenstein et al. (2009) projected that the range of Armillaria root rot will remain constant in a warmer climate, and if areas where Douglas-fir is maladapted increase, susceptibility to root rot could also increase. With limited genetic diversity at low to middle elevations and a more generalist strategy at higher elevations (St. Clair and Howe 2007), Douglas-fir may retract from the driest portions of its range.

Grand Fir (Abies grandis)

(Middle Rockies subregion)

Autecology

Grand fir is found on a wide variety of sites, including stream bottoms, valleys, and mountain slopes in the Middle Rockies of the IAP region (Foiles et al. 1990), typically in association with other conifer species. Grand fir grows best on rich soils of valley bottoms but also grows well on shallow exposed soils of mountain ridges, if moisture is adequate (Antos and Shearer 1980). Grand fir is either an early- or late-seral species, depending on site moisture (Ferguson and Johnson 1996). On productive mesic sites, it grows rapidly to compete with other seral species in the overstory, but other conifer species can outcompete it. On drier sites, it is the most shade-tolerant species and can dominate the understory. Grand fir can also share dominance with subalpine fir, especially in narrow valley bottoms, where it can exert dominance in lower elevational zones (Antos and Shearer 1980). Grand fir has high shade tolerance but low drought tolerance. It forms associations with ectomycorrhizae and arbuscular mycorrhizae, which may allow it to outcompete some shade-tolerant conifers. It has low frost tolerance but can tolerate fluctuating water tables.

Disturbance Interactions

Grand fir is susceptible to fire damage in moist creek bottoms but is more resistant on dry hillsides where roots are deeper and bark is thicker (Ryan and Reinhardt 1988). Wildfires that burn grand fir stands are stand replacing or mixed severity, generating sufficient heat to kill even mature trees (Arno 1980; Arno et al. 2000). Grand fir is susceptible to Armillaria and annosus root diseases, which can cause high levels of tree mortality (Hagle et al. 2003). Numerous insects attack grand fir, including western spruce budworm and Douglas-fir tussock moth, which cause widespread defoliation, top kill, and mortality. The western balsam bark beetle (*Dryocoetes confusus*) and fir engraver (*Scolytus ventralis*) are the principal bark beetles that attack grand fir (Foiles et al. 1990).

Historical and Current Conditions

Fire exclusion has increased grand fir on both dry and mesic sites, and higher tree densities have stressed grand fir, making it more susceptible to root rot and insect attacks. Therefore, the condition of most grand fir stands depends on the last severe fire; if fire exclusion has caused grand fir to dominate in both the overstory and understory, then these stands are usually stressed because of high densities and increased root rot and insects. However, in early-seral stands where high grand fir regeneration has not yet occurred, an increase in fir is likely with continued fire exclusion.

Climate Change Responses

On dry sites, increased drought and longer growing seasons will exacerbate stress caused by competition, resulting in high mortality of grand fir, mainly from insects and disease. Nitschke and Innes (2008) used a gap model to project major declines in grand fir, and Coops and Waring (2011) used a mechanistic model to simulate a nearly 50 percent decrease in the range of grand fir compared to historical distributions. However, increased productivity may lead to increased grand fir populations in locations with higher soil moisture (Aston 2010; Urban et al. 1993). As noted earlier, increased densities may also lead to increased stress. Longer fire seasons and high fuel loadings from fire exclusion will probably lead to large, severe fires that may reduce grand fir in drier locations. In summary, although grand fir is often stressed by high stem densities, the species is likely to tolerate changes in climate and remain on the landscape at levels that are closer to historical conditions rather than its current abundance.

Shasta Red Fir (Abies magnifica)

(Great Basin and Semi Desert subregion)

Autecology

Shasta red fir grows best in areas with cold, wet winters and warm, dry summers (Lanner 1983; Oosting and Billings 1943; Rundel et al. 1977). The growing season in these areas is short, with snow often on the ground in July (Barbour 1988; Barbour et al. 1991; Holland 1986; Mitchell and Moir 1976). Red fir can be found growing at lower elevations in canyons and other protected places where significant cold air drainage keeps soil and air temperatures low (Parker 1984). The species also occurs at high elevation on mountain ranges that continue in active formation, where it thrives on young, xeric soils. Red fir has a high frost tolerance and low drought tolerance. It is a late-seral species nearly everywhere it is found. Although red fir grows best in full sunlight, it can survive and grow for long periods in relatively dense shade.

Disturbance Interactions

Shasta red fir sustains moderate damage from low-severity fires but is often killed by mixed-severity fires (Atzet and Wheeler 1982). Openings created in mixed red fir and white fir (*Abies concolor*) stands in the Sierra Nevada tend to regenerate more readily to red fir (Parker 1986). Red fir is susceptible to windthrow after partial cutting, especially when marginal codominant and lower crown classes are left as the residual stand (Gordon 1973). Root diseases such as annosus root rot contribute significantly to lack of wind firmness. Other diseases that reduce tree vigor include dwarf mistletoe and cytospora (*Cytospora* spp.) canker, which, in turn, make trees susceptible to fir engraver attack.

Historical and Current Conditions

Native Americans used Shasta red fir forests for hunting mule deer (*Odocoileus hemionus*) and for other sources of food and materials during summer. Mining, logging, water diversions, railroad development, and sheep grazing altered some lower-elevation fir forests during the late 19th and early 20th centuries (Meyer n.d.). Burning was used to promote growth of grasses and forbs and to remove fuel and young trees from the understory (McKelvey and Johnston 1992), thus reducing fir regeneration. Starting in the 1950s, timber harvest and extensive road infrastructure began in portions of red fir forest, with silvicultural techniques that create even-aged stands being implemented (Potter 1998). By the 1990s, silvicultural practices emphasized shelterwood cutting and uneven-aged silvicultural systems (Laacke and Tappeiner 1996). Despite this history of resource use, red fir is largely undisturbed in many higher-elevation and isolated locations.

Climate Change Responses

Shasta red fir is expected to sustain moderate effects from a warmer climate. If snowpack decreases as expected, a longer growing season may increase growth at higher elevations. Regeneration could also improve under these conditions. Lower-elevation populations may grow more slowly where soil moisture is limited in summer. Red fir is typically found in forests with mixed-severity fire regimes, so if wildfire becomes more frequent than historical records indicate, especially where fuel loadings are elevated, fire severity could cause crown fires with high mortality in younger trees (older trees have thick bark and high crowns). Increased fire could produce a more open forest structure over decades to centuries.

Subalpine Fir (Abies lasiocarpa)

(Middle Rockies, Southern Greater Yellowstone, Uintas and Wasatch Front, Plateaus, Great Basin and Semi Desert subregions)

Autecology

Although widely distributed, subalpine fir grows within a narrow range of mean temperatures of 25 to 40 °F, with average January temperatures of 5 to 25 °F. Cool summers, cold winters, and deep winter snowpack are more important than precipitation in determining where it grows. Subalpine fir ranges from lower valleys to the upper subalpine zone in the IAP region, typically mixed with other species, most notably Douglas-fir and Engelmann spruce. Subalpine fir is shade tolerant; partial shade usually favors seedling establishment and early survival (Knapp and Smith 1982). It is relatively intolerant of drought, and seedlings can be killed by lengthy droughts. It is a prolific seeder, often having large cone crops every

2 to 3 years (Alexander et al. 1990), and although dense mats of seedlings can occur, they are also susceptible to many herbivores and pathogens.

Disturbance Interactions

Subalpine fir is highly susceptible to fire damage because of thin bark, shallow roots, and low, dense crowns (Ryan and Reinhardt 1988). Even low-intensity fire can cause mortality, and frequent fires can eliminate subalpine fir from both the overstory and understory, thereby maintaining more fire-adapted species such as lodgepole pine (Little et al. 1994; Murray et al. 1995; Wadleigh and Jenkins 1996). In spruce-fir forests, the most important insects are western spruce budworm and western balsam bark beetle (*Drycoetes confusus*). Fir broom rust (*Melampsorella caryophyllacearum*) and wood rotting fungi are responsible for most disease losses, but root and butt rots may be important locally. Decades of intense competition, coupled with a period of moderate to severe drought, can cause mortality in subalpine fir stands.

Historical and Current Conditions

Effects of fire exclusion have not yet become manifest in most subalpine fir ecosystems because of historically infrequent fire and slow successional advancement. However, abundance of subalpine fir has increased in some landscapes (Keane et al. 1994). These dense stands have become stressed from competitive interactions, resulting in susceptibility to disturbances and drought. If these stands continue to escape fire, the seed sources of co-located, fire-adapted species may be eliminated, and high-elevation sites could be converted to grass and shrublands (Keane 2001). In addition, if fire is excluded from these dense forests, fuels will accumulate, inevitably leading to high-severity fires (Keane 2001; Morgan et al. 1994b). Recent USFS Forest Health Monitoring data in the IAP region indicate that dieback of subalpine fir is occurring in some locations, attributed to a complex of drought, insects, and pathogens.

Climate Change Responses

Because subalpine fir is adapted to moist growing conditions, it is likely to respond poorly to increasing temperatures and drought (Alexander et al. 1990; Brunelle et al. 2005; Whitlock and Bartlein 1993). However, it is a good competitor and may be able to expand its range at treeline (Little et al. 1994; Rochefort et al. 1994; Villalba et al. 1994) and increase growth in a longer growing season (Peterson et al. 2002). Seedling establishment may be the bottleneck for subalpine fir establishment in the future because the species needs long periods of high moisture for germination and seedling establishment (Urban et al. 1993), and years that meet these conditions may be less frequent in the future. If stand densities increase, competitive stress will increase, making fir more vulnerable to insects, disease, and abiotic factors. If wildfire increases where subalpine fir is dominant, abundance would decrease from the direct effects of higher temperature. Subalpine fir is likely to shift across the high mountain landscape, with expansion balancing retraction, although fire, disease, and insects may limit abundance.

White Fir (Abies concolor var. concolor)

(Uintas and Wasatch Front, Plateaus, Great Basin and Semi Desert subregions)

Autecology

White fir is distributed throughout most of the American Southwest, from canyon bottoms and ravines up to ridgetops. It is a dominant, late-seral component of some habitat types in Utah and develops best on gentle slopes (Laacke 1990), although the rooting habit is adaptable to depth of the soil profile. It can survive for long periods as a suppressed tree in the understory, then respond with rapid growth if light becomes available. Within mixed conifer forests, white fir tends to achieve dominance on moist sites, especially if long fire return intervals provide the opportunity for it to mature to a point at which it is moderately fire tolerant. White fir is sensitive to frost damage (Laacke 1990), and is susceptible to windthrow following partial cutting.

Disturbance Interactions

In mixed conifer forests with an intact low-severity fire regime, white fir rarely attains dominance because it is more fire sensitive than its associates (Agee 1982; Alexander et al. 1984). Thus, many white fir habitat types are in mid-seral stages, with various species dominating the overstory and white fir dominating the reproductive size classes. White fir mistletoe (*Phoradendron bolleanum* subsp. *pauciflorum*) and white fir dwarf mistletoe (*Arceuthobium abietinum* f. sp. *concoloris*) damage white fir, causing spike tops, loss of vigor, and increased susceptibility to bark beetles (Bega 1978). White fir is susceptible to a number of decay fungi including annosus root disease, Armillaria root disease, laminated root rot (*Phelllinus weirii*), yellow cap fungus (*Pholiota limonella*), Indian paint fungus (*Echinodontium tinctorium*), and white pocket rot (*Phellinus mini*). Fir engraver beetle causes major losses throughout the range of white fir (Wilson and Tkacz 1996).

Historical and Current Conditions

White fir, which has historically been dominant on wetter sites and codominant in drier mixed conifer forests, has increased in abundance in areas where fire has been excluded. In some cases, the understory in fire-excluded stands is dense, and surface fuels are high, conditions conducive to a crown fire (Dahms and Geils 1997; McKelvey and Johnston 1992). If dense stands escape fire, the seed sources of other fire-adapted species may be eliminated, and some sites may have increased dominance of grass and shrublands (Keane 2001). White fir mortality following wildfire is often

100 percent, although associated species such as ponderosa pine and Douglas-fir often survive. White fir has never been a primary timber species, although it has been logged in some places. It was often left uncut where more valued species were removed, becoming the residual dominant.

Climate Change Responses

White fir has high shade tolerance but low drought tolerance, so low soil moisture will have the greatest effects in well-drained soils and on south aspects. Sudden temperature increases during May and June can cause damage nearly identical to that of spring frosts, which may be an issue for some fir populations. A modeling study in California suggested that effects of climate change on white fir will be moderate (Battles et al. 2008), and although this may be true in the IAP region, wildfire will play a major role in its future distribution and abundance. White fir is typically found in forests with low-severity and sometimes mixed-severity fire regimes, so if fire becomes more frequent than historically, especially where fuel loadings are elevated, fire severity could cause crown fires with high mortality rates. Over decades to centuries, increased fire could produce a more open forest structure with fewer white fir in both the canopy and understory.

Rocky Mountain Juniper (Juniperus scopulorum)

(Uintas and Wasatch Front subregion)

Autecology

Rocky Mountain juniper grows in dry, sub-humid climates. It is a drought-enduring species with a shallow but fairly extensive lateral root system. Rocky Mountain juniper is normally a component of early-seral or near late-seral vegetation. It is relatively shade tolerant during the seedling and sapling stages, but it later becomes more intolerant and is unable to endure excessive shade. In Utah, junipers have been observed occupying sagebrush stands under certain conditions; twoneedle pinyon (*Pinus edulis*) generally follows and tends to replace juniper. Pinyon-juniper communities may encroach into grasslands that have been overgrazed or disturbed. Once established, Rocky Mountain juniper competes well with understory vegetation for water and nutrients.

Recent paleobotanical studies indicate the macroclimate covering much of the Rocky Mountain juniper range has changed from mesic to more xeric conditions. Juniper is generally less drought resistant than other juniper species, and high temperatures are not favorable for regeneration or growth. Rocky Mountain juniper was present in western Nebraska and the Laramie Basin of Wyoming as recently as 1,000 years BP, with some trees over 50 inches in diameter (Tauer et al. 1987; Van Devender 1987).

Disturbance Interactions

Rocky Mountain juniper is susceptible to loss from erosion simply because it is often established on exposed sites where soils are readily eroded. It is susceptible to death or injury from fire, primarily because the bark is thin, and the lower branches contain volatile oils and normally extend to the ground (Hepting 1971; Noble 1990; Sieg 1997). Rocky Mountain juniper has a compact crown when young, and because it grows slowly, is susceptible to fire for the first 20 years or more (Crane 1982; Fischer and Clayton 1983; Hansen and Hoffman 1988; Mitchell 1984; Mueggler 1976; Stanton 1974). As trees mature, they develop thicker bark and a more open crown, allowing them to potentially survive surface fires. Large-diameter junipers have been documented to survive four to six fires.

Postfire reestablishment is solely by seed (Floyd et al. 2000), and animal transport of seeds is an important factor (Paysen et al. 2000). Regeneration is often high after old trees burn (Stanton 1974; Wright 1972). Frequent fires in pinyon-juniper habitat can maintain a grassland setting, and the absence of fire will allow conversion to woodlands (Gruell 1986). After fire in pinyon-juniper, junipers usually establish first, followed by pinyon pine, which may eventually replace juniper on higher-elevation sites (Holland 1990). The nonnative annual cheatgrass (*Bromus tectorum*) has become increasingly common in the understory over the past few decades, providing abundant fine surface fuels and increasing the potential for more frequent wildfires (Shinneman and Baker 2009).

Historical and Current Conditions

Rocky Mountain juniper was a source of fuel (charcoal) for the mining industry between the 1860s and 1920s (Young and Budy 1979), and it has been used extensively for firewood, fence posts, and other needs, with local deforestation occurring in some locations. In some lower-elevation sites, juniper has been cut or removed from the landscape through chaining and herbicides to encourage the growth of grasses and forage for livestock grazing. Persistent woodlands of Rocky Mountain juniper, pinyon pine, or a mixture of both are found where local soils and climate are favorable, and wildfire has been infrequent (Romme et al. 2009). Pinyon-juniper savannas are found where local soils and climate are suitable for both trees and grasses, and low-severity fires have been relatively frequent. Wooded shrublands are found where local soils and climate support a shrub community, but trees have increased during moist climatic conditions and periods without wildfire. Large increases in juniper density have occurred in portions of all types of pinyon-juniper vegetation, which may have been driven by factors such as natural range expansion, livestock grazing since the 1880s (which reduced fuels and probably decreased fire frequency), fire exclusion, climatic variability, and carbon dioxide fertilization (Romme et al. 2009).

Climate Change Responses

Rocky Mountain juniper is drought tolerant, and reduced soil moisture is not expected to have a significant effect on its abundance and distribution, although its growth and expansion into adjacent shrub-steppe systems could be slowed. The future of Rocky Mountain juniper will largely depend on spatial and temporal patterns of wildfire, which is expected to increase in frequency (Floyd et al. 2004). Junipers can generally survive low-severity fire if they are at least 20 years old, so if fires occur more frequently than that, tree mortality will be high. After an initial fire, accumulation of surface fuels and tree regeneration could be slow because of moisture limitations, resulting in a sparse canopy and disconnected fuels (Rocca et al. 2014). The long-term condition of juniper is complicated by nonnative annual grasses, especially cheatgrass, which increases surface fuels and fire frequency.

Utah Juniper (Juniperus osteosperma)

(Plateaus, Great Basin and Semi Desert subregions)

Autecology

Utah juniper is a late-seral species in several pinyon-juniper, sagebrush (*Artemisia* spp.)-grassland, and shrub-steppe habitats. Utah juniper tolerates dry soils (Hickman 1993; Lanner 1983; Meeuwig and Bassett 1983), commonly growing on alluvial fans and dry, rocky hillsides (Barney and Frischknecht 1974; Hitchcock and Cronquist 1973; Pieper 1977; Shantz and Piemeisel 1940) with shallow, alkaline soils (Bunderson et al. 1985). Utah juniper is shade intolerant (Meeuwig and Bassett 1983); it is a late-seral species in areas where stands are open and regeneration can occur without competition for light. Utah juniper has a taproot that extends deep into the soil, responding to low nutrient levels in the soil by developing extensive fine roots. Juniper competes more efficiently for soil moisture than do herbaceous understory plants, and is more likely to maintain a stable population as understory plants decrease (Austin 1987; Everett et al. 1983; Springfield 1976).

Disturbance Interactions

Utah juniper is generally not fire tolerant, although trees more than 4 feet tall are capable of surviving low-intensity fires (Bradley et al. 1992; Springfield 1976). Cheatgrass has become increasingly common in the understory over the past few decades, continually providing abundant fine surface fuels on and increasing the potential for more frequent wildfires (Shinneman and Baker 2009).

Historical and Current Conditions

Utah juniper was a source of fuel (charcoal) for the mining industry between the 1860s and 1920s (Young and Budy 1979), and has been used extensively for firewood, fence posts, and other needs, with local deforestation in some locations. In some lower-elevation locations, juniper has been removed from the landscape through chaining and herbicides to encourage growth of grasses and forage for livestock grazing. Persistent woodlands of juniper or pinyon pines, or a mixture of both, are found where local soils and climate are favorable, and wildfire has been infrequent (Romme et al. 2009). Pinyon-juniper savannas are found where local soils and climate are suitable for both trees and grasses, and low-intensity fires are relatively frequent. Wooded shrublands are found where local soils and climate support a shrub community, but trees have increased during moist climatic conditions and periods without wildfire. Large increases in juniper density have occurred in portions of all types of pinyon-juniper vegetation, which may have been driven by factors such as natural range expansion, livestock grazing (since the 1880s, which reduced fuels and probably decreased fire frequency), fire exclusion, climatic variability, and carbon dioxide fertilization (Romme et al. 2009).

Climate Change Responses

Utah juniper is drought tolerant, and reduced soil moisture is not expected to have a significant effect on its abundance and distribution, although growth may decline even as it spreads into adjacent shrub-steppe systems. The future of Utah juniper will largely depend on spatial and temporal patterns of wildfire, which is expected to increase in frequency (Floyd et al. 2004). Junipers can generally survive fire only if they are tall enough for the crown to escape flames. Following an initial fire, accumulation of surface fuels and tree regeneration will probably be slow because of moisture limitations, resulting in a sparse canopy and disconnected fuels (Rocca et al. 2014). The long-term condition of juniper is complicated by nonnative annual grasses, especially cheatgrass, which increases surface fuels and fire frequency.

Western Larch (Larix laricina)

(Middle Rockies subregion)

Autecology

Western larch grows in relatively cool, moist forests in the Middle Rockies portion of the IAP region (Habeck 1990; Schmidt and Shearer 1990), typically associated with several other conifer species. It is often found in locations that have relatively high snowfall, and is rarely found in xeric sites (Gower et al. 1995). Cone and seed production is abundant when trees are older than 30 years, with good seed crops occurring every 10 to 14 years (Owens 2008). Seed germinates best on seedbeds exposed by burning or mechanical scarification (Antos and Shearer 1980; Beaufait et al. 1977; Schmidt 1969; Shearer 1976), and young seedlings grow fast on suitable sites, although drought reduces seedling survival (Schmidt and Shearer 1995). Shade intolerant, larch grows fast with tall, open crowns, allowing it to outcompete other

species on mesic sites (Milner 1992). It is moderately drought tolerant and can survive seasonal drought, but performs poorly when droughts last more than 2 years.

Disturbance Interactions

Western larch depends on open-canopy, high-light environments and mineral soil seedbeds created by fire for successful regeneration (Schmidt et al. 1976). It can survive intense fire because of thick bark (Ryan and Reinhardt 1988), high crowns, deep roots, and epicormic branch production (Fiedler and Lloyd 1995; Harrington 2012; Schmidt and Shearer 1995; Schmidt et al. 1976), often surviving crown fires that kill other species (Marcoux et al. 2015). Seeds are wind dispersed across large burns, and if mature lodgepole pine occurs with larch, regeneration may be dominated by both species (Hopkins et al. 2013).

Western dwarf mistletoe (*Arceuthobium campylopodum*) is a damaging, parasitic plant of larch (Schmidt and Shearer 1990). It infects seedlings and persists throughout the life of the tree, causing reduced growth, water loss, and deformities. Cool, wet springs favor foliar diseases such as larch needle cast (*Meria larisis*), which, in turn, can reduce cone production. Larch needle blight (*Hypodermella laricis*), brown trunk rot (*Fomitopsis officinalis*), and red ring rot (*Phellinus pini*) are also important pathogens. Western spruce budworm and the nonnative larch casebearer (*Coleophora laricella*) are the two most serious insect pests (DeNitto 2013; Schmidt and Fellin 1973). Although neither insect causes substantial mortality, episodic outbreaks can cause severe defoliation and reduce growth and cone production (Schmidt et al. 1976).

Historical and Current Conditions

Western larch was formerly an important timber species, but extensive logging during the 20th century removed many of the large larches, reducing its dominance on the landscape (Arno 2010). Reduced seed sources for regeneration and fire exclusion have reduced burned mineral soil seedbeds where larch can regenerate. Continued fire exclusion has increased stand densities and increased surface fuel loads, which will make future fires more intense than they have been historically. Considerable effort is underway to increase the distribution and abundance of western larch in locations where it was previously more common.

Climate Change Responses

Western larch may be susceptible to a warmer climate because of its narrow geographic and elevation distribution and its uncertain association with wildfire. If fire increases, larch may have a colonization advantage, as long as fire mortality is moderate and mature trees remain to serve as seed sources. Without seed sources, regeneration may require assistance from management through planting. If fire exclusion continues, stand densities will increase and larch may be outcompeted by shade-tolerant competitors, making it more susceptible to insects and disease. When dense stands burn, crown fires may kill older, seed-producing trees (Hopkins et al. 2013). Keane et al. (1996) simulated major declines for western larch under fire exclusion and moderate climate change, but found it increased as more fire was allowed to burn over many decades. Larch can take advantage of changes in productivity in colder sites, as long as these areas burn with low intensity and larch survives the fires to provide seed for regeneration.

Great Basin Bristlecone Pine (Pinus longaeva) and Rocky Mountain Bristlecone Pine (P. aristata)

(Uintas and Wasatch Front, Plateaus, Great Basin and Semi Desert subregions)

Autecology

Great Basin bristlecone pine occurs in montane, subalpine, and treeline communities from 7,200 to 12,000 feet elevation (Hickman 1993; Lanner 1999), typically in multi-aged stands (Bradley et al. 1992). It grows in pure stands at treeline and in the upper subalpine zone, and is codominant with limber pine at lower elevations (Critchfield and Allenbaugh 1969; Vasek and Thorne 1977). Great Basin bristlecone pine is drought tolerant (Bare 1982; Tang et al. 1999), occurring in climates that are cold in winter and dry in summer. It establishes quickly in open mesic sites (Hawksworth and Bailey 1980), but competes poorly for water and nutrients, and is usually excluded from productive sites (Beasley and Klemmedson 1973; Hiebert 1977).

Rocky Mountain bristlecone pine occurs from 8,200 to 11,000 feet elevation (FNAA 2009) and is common on steep, dry, south- or west-facing slopes. It grows in cold, continental climates, with precipitation patterns influenced by summer monsoons bringing afternoon rain. Rocky Mountain bristlecone pine is commonly found on unproductive sites with nutrient-poor, acidic soils. This species often occurs in pure stands or mixed with limber pine.

Disturbance Interactions

As a thin-barked species (Zavarin and Snajberk 1973), Great Basin bristlecone pine is adapted to survive only low-intensity surface wildfires, although fire is infrequent at high-elevation sites (Bradley et al. 1992). White pine blister rust (*Cronartium ribicola*) is present in some stands, but it rarely has a significant effect on populations. Most high-elevation pines eventually die from root rot decay or soil erosion, which exposes and kills roots (Lanner 1999). Small wildfires may kill a few trees.

Wildfires are common in Rocky Mountain bristlecone pine sites, but are usually small in extent and cause minimal damage because patchy stand structure and low fuel loadings limit fire spread (Crane 1982). Although fire is not a major disturbance factor, Rocky Mountain bristlecone pine is favored in early postfire succession because it is a shade-intolerant seral species (Baker 1992; Schoettle 2003). Blister rust has been

recorded in this species only recently (Blodgett and Sullivan 2004) and is rarely observed in the IAP region.

Historical and Current Conditions

Great Basin bristlecone pine and Rocky Mountain bristlecone pine are located at high elevation in relatively inaccessible locations. Because these species have no commercial value, they generally remain undisturbed by human activity and exist in intact subalpine forests and woodlands.

Climate Change Responses

Great Basin bristlecone pine and Rocky Mountain bristlecone pine are tolerant of cold temperatures and deep snowpack in winter, low soil moisture in summer, and high winds. Therefore, they are expected to be moderately vulnerable to climate change, with considerable variation among sites. A recent study showed that Great Basin bristlecone has a threshold at 60 to 250 vertical feet below treeline, above which trees have a positive growth response to temperature (Salzer et al. 2014). Growth chronologies from 250 feet or more below treeline had a change in climate response and did not correlate strongly with temperature-sensitive chronologies developed from trees growing at upper treeline. At the highest sites, trees on south-facing slopes grew faster than trees on north-facing slopes. High growth rates on the south aspect have declined since the mid–1990s, suggesting that temperature may no longer be as limiting to growth. Therefore, increasing warmth may lead to a divergence between growth and temperature at previously temperature-limited sites. Neither species of bristlecone pine is expected to change in distribution and abundance significantly during the 21st century. Increased wildfire could affect Rocky Mountain bristlecone pine in mixed-species stands with high surface fuels, but not in higher-elevation locations where trees are scattered and fuels are low.

Jeffrey Pine (Pinus jeffreyi)

(Great Basin and Semi Desert subregion)

Autecology

Jeffrey pine is shade intolerant and drought adapted, occupying low- to mid-elevation, dry forests (Minore 1979). It is both an early- and late-seral species, often associated with other conifer species on moist sites. Moisture typically limits growth, especially in summer, and distribution of Jeffrey pine on drier sites is limited by available soil moisture, which, in turn, is affected by soil texture and depth. Jeffrey pine tolerates dry soil conditions by efficiently closing stomata to avoid water loss and xylem cavitation (Sala et al. 2005), tolerating intense drought, and drawing groundwater at low soil water potentials. Jeffrey pine is associated with several species of ectomycorrhizae, giving it the capacity to survive in dry environments. Soil texture, plant competition, and seedbed conditions reduce seed germination and limit seedling survival and growth, although it can often germinate under moisture stress (Oliver and Ryker 1990).

Disturbance Interactions

Disturbance plays a major role in Jeffrey pine forests. The most damaging of the tree-killing insects are several species of *Dendroctonus* (Oliver and Ryker 1990), followed by ips beetles, all of which are native and present naturally in many stands. Dwarf mistletoe is widespread in Jeffrey pine stands but rarely fatal. Bark beetles can cause extensive mortality given availability of preferred host stand conditions. Jeffrey pine has a high capacity to survive fire (Minore 1979; Ryan and Reinhardt 1988), and wildfire favors the growth of large (thicker bark) Jeffrey pine by killing its primary competitors and small-diameter Jeffrey pines (Arno 1988; Steele et al. 1986).

Historical and Current Conditions

Fire exclusion, mining, timber harvest, and livestock grazing have contributed to reductions in distribution and abundance of Jeffrey pine. Changes in fire regime have altered the composition and structure of many dry forests, with area burned by surface fires decreasing and crown fires increasing in many areas (Hann et al. 1997). Landscapes where fire has been excluded for many decades typically have high stand densities and surface fuel loadings, setting the stage for future crown fires.

Climate Change Responses

Jeffrey pine is expected to be relatively tolerant of increasing temperatures and longer droughts. This species has high phenotypic plasticity and is therefore adapted to drought, although regeneration may decrease if precipitation decreases or becomes more variable. Some studies project an increase in distribution for ponderosa pine in western North America (Hansen et al. 2001; Morales et al. 2015; Nitschke and Innes 2008) that may be true for Jeffrey pine as well. Advancing competition resulting from fire exclusion, increased wildfire extent and intensity, and potential increases in mountain pine beetle, western pine beetle (*Dendroctonus brevicomis*), several *Ips* species, and western pine shoot borer (*Eucosma sonomana*) will dictate the future of Jeffrey pine. If fires are too frequent, established regeneration will not grow above the lethal scorch height. Increasing wildfire extent and severity (crown fires) could also eliminate the mature Jeffrey pine trees that provide seed sources for populating future burns.

Limber Pine (Pinus flexilis)

(Middle Rockies, Southern Greater Yellowstone, Uintas and Wasatch Front, Plateaus, Great Basin and Semi Desert subregions)

Autecology

Limber pine is a shade-intolerant, early-seral species (Steele 1990) that is slow growing but long lived. It occupies xeric sites across a wide range of elevations (Jackson et al. 2010). Because it is easily killed by fire, the species

is found in fire-protected sites (e.g., rocky outcrops) with infrequent fires of low severity (Steele 1990). It can be associated with a wide range of other conifer species and quaking aspen (Langor 2007; Steele 1990). It is associated with both ectomycorrhizae and arbuscular mycorrhizae that facilitate its ability to exist in extremely dry environments. Limber pine seedlings are poor competitors with grass, but grow reasonably well on rocky substrates and with shrubs. Limber pine has difficulty competing with encroaching species on productive mesic sites and is often succeeded by Douglas-fir and subalpine fir. Its seeds are dispersed by rodents and by Clark's nutcracker (*Nucifraga columbiana*), which relies on pine seeds as a food source and caches them throughout the subalpine zone (Lanner 1980; Lanner and Vander Wall 1980).

Disturbance Interactions

Thin bark and low crowns make limber pine susceptible to damage from wildfire. It is also susceptible to white pine blister rust, and some stands in newly infected areas are currently undergoing high mortality (Smith et al. 2013). Limber pine also facilitates the expansion of currant (*Ribes* spp.) into traditional grasslands (Baumeister and Callaway 2006), thus increasing rust infections and subsequent mortality. Mountain pine beetle (Jackson et al. 2010) and severe dwarf mistletoe infections can cause mortality. Porcupine (*Erethizon dorsatum*) damage is prevalent in some areas.

Historical and Current Conditions

Fire exclusion has allowed limber pine to expand its range from fire-protected sites into areas where frequent fires historically restricted it (Arno and Gruell 1983; Brown and Schoettle 2008). Expansion into some grass and shrub rangelands has facilitated expansion of other species as well (e.g., Douglas-fir) (Baumeister and Callaway 2006; Jackson et al. 2010). Some of the newly established limber pine forests have suffered recent mortality from white pine blister rust, mountain pine beetle, and red belt (*Fomitopsis pinicola*) (Jackson et al. 2010; Langor 2007; Taylor and Sturdevant 1998). Increasing wildfire extent has also affected some stands.

Climate Change Responses

Limber pine has a generalist adaptive strategy with broad phenotypic plasticity (Devine et al. 2012; Feldman et al. 1999), so it is expected to be moderately vulnerable to climate change. The ability of limber pine to occupy shallow, infertile soils and tolerate periods of drought will confer resistance to warmer temperatures and drought. Reduced snowpack could increase growth of limber pine at higher elevations by lengthening the growing season (Aston 2010). However, warmer temperatures could also reduce soil moisture for seed germination and seedling growth, and lack of ectomycorrhizal associations could inhibit establishment in some locations (Coop and Schoettle 2009). Increasing wildfire extent and intensity may impact some limber pine stands in the future, causing higher mortality and reducing encroachment into grasslands.

Lodgepole Pine (Pinus contorta var. latifolia, P.c. var. murrayana)

(Middle Rockies, Southern Greater Yellowstone, Uintas and Wasatch Front, Great Basin and Semi Desert subregions)

Autecology

Lodgepole pine has broad ecological amplitude and environmental tolerance, including both the *murrayana* variety in the western portion of the IAP region and the *latifolia* variety elsewhere in the region (Lotan and Critchfield 1990). It grows well on gentle slopes and in basins but is also found on steep slopes and shallow soils. It is shade intolerant but highly tolerant of frost and drought. Lodgepole pine grows in pure stands and in association with many other conifer species, including subalpine fir, Engelmann spruce, Douglas-fir, and western larch (Steele et al. 1983). It can be either early or late seral, depending on location. Its ability to remain on xeric landscapes is enhanced by its association with many types of mycorrhizae. Lodgepole pine is generally a prolific seed producer, and the prevalence of cone serotiny in most individuals of the *latifolia* variety promotes rapid regeneration following wildfire (Hardy et al. 2000).

Disturbance Interactions

Fire plays a critical role in lodgepole pine forest succession (Brown 1973; Lotan et al. 1984). Mature lodgepole pine appears to be able to survive low-intensity fire, despite having thin bark (Ryan and Reinhardt 1988). In many cases, natural regeneration is prolific via abundant seed from serotinous cones (Lotan and Perry 1983; Nyland 1998), although drought is a common cause of mortality in first-year seedlings. Mountain pine beetle has played a significant role in the dynamics of lodgepole pine ecosystems. Beetles and wildfire create an important stress complex for lodgepole pine in some locations (Brown 1973; Geiszler et al. 1980), but can also act independently (Axelson et al. 2009; Moran and Corcoran 2012).

Historical and Current Conditions

Advancing succession associated with fire exclusion is contributing to replacement of lodgepole pine by subalpine fir in some areas of the IAP region. Concurrent increases in recently burned areas are creating new lodgepole stands, some of which may become very dense. Increased drought in these dense stands may exacerbate stress from other factors, including competition and insects. Warming temperatures have contributed to unprecedented mountain pine beetle outbreaks in lodgepole pine in western North America, including in the IAP region, causing 100 percent mortality in many mature lodgepole pine stands (Carroll et al. 2003; Jenkins et al. 2008; Page and Jenkins 2007).

Climate Change Responses

Longer drought periods and warmer temperatures in drier lodgepole pine forests may cause decreased growth and regeneration, perhaps resulting in a transition to more xeric tree species. Chhin et al. (2008) and Nigh (2014) projected that growth will decrease in moderate future warming, but the species probably has sufficient genetic capacity to compensate for this loss. Given that lodgepole pine is a generalist and capable of regenerating and growing in a wide range of environments, it is likely that any reduction in lodgepole pine dominance in dry sites would occur only under extreme warming scenarios over many decades to centuries.

In high-elevation subalpine systems where seasonal drought is not a problem, a warmer climate may increase productivity (Aston 2010; Johnstone and Chapin 2003). Wang et al. (2006) found major increases in lodgepole pine productivity under future climates with moderate warming, but decreased productivity and perhaps local extinctions were associated with extreme warming. Romme and Turner (1991) projected increases in the lodgepole pine zone in the Greater Yellowstone Area under moderate warming. Lodgepole pine could migrate into upper subalpine areas where it is currently excluded by cold, windy conditions (Hamann and Wang 2006; Romme and Turner 1991). The *latifolia* variety is well adapted to increased fire occurrence, depending on level of serotiny (Turner et al. 1999), although if fire is too frequent, it could be eliminated from sites where fire returns before established seedlings and saplings become reproductively mature (Larson et al. 2013). Projected increases in climatic conditions that facilitate mountain pine beetle outbreaks (higher reproductive rates) (Bentz et al. 2010) could reduce the abundance of lodgepole pine in some landscapes (Creeden et al. 2014; Gillette et al. 2014) (Chapter 8), especially where fire has been excluded (Temperli et al. 2013).

Ponderosa Pine (Pinus ponderosa var. ponderosa, P.p. var. scopulorum)

(Middle Rockies, Uintas and Wasatch Front, Plateaus, Great Basin and Semi Desert subregions)

Autecology

Ponderosa pine is shade intolerant and drought adapted, occupying low-elevation, dry forests (Minore 1979), and is both an early- and late-seral species, often associated with Douglas-fir and grand fir on moister sites. In the IAP region, Pacific ponderosa pine (var. *ponderosa*) extends from the central mountains of Idaho to the east side of the Sierra Nevada in Nevada. The Rocky Mountain variety (var. *scopulorum*) extends from the eastern mountains of Nevada to the central and northern mountains of Utah. (Note that Washoe pine [*P. p.* subsp. *washoensis*], which is found in a few locations in the Great Basin and northeastern slope of the Sierra Nevada, is no longer considered a discrete subspecies and is not included in the assessment.) For both the Pacific and Rocky Mountain varieties, moisture typically limits growth, especially in summer, and distribution of ponderosa pine on drier sites is limited by available soil moisture, which, in turn, is affected by soil texture and depth. Ponderosa pine seedlings are susceptible to frost damage, which can exclude this species from low valleys (Shearer and Schimidt 1970). Ponderosa pine tolerates dry soil conditions by efficiently closing stomata to avoid water loss and xylem cavitation (Sala et al. 2005), tolerating intense drought, and drawing groundwater at low soil water potentials.

Cone crop periodicity in ponderosa pine varies greatly, but it is a poor seed producer in some areas. Natural regeneration is sporadic and is best when a heavy seed crop is followed by favorable weather in the next growing season (Heidmann 1983; Shearer and Schmidt 1970). The Rocky Mountain variety is highly inbred, and its vulnerability could be further compromised with limited gene flow between populations (Potter et al. 2015). Soil texture, plant competition, and seedbed conditions reduce seed germination and limit seedling survival and growth, although ponderosa pine can often germinate under moisture stress (Oliver and Ryker 1990). Young seedlings are susceptible to cold night temperatures and deep frosts, and trees occasionally suffer winter desiccation in drying winds.

Disturbance Interactions

Disturbance plays a major role in sustaining ponderosa pine forests. Over 100 species of insects attack the Pacific variety, and over 50 species attack the Rocky Mountain variety. The most damaging of the tree-killing insects are several species of *Dendroctonus* (Oliver and Ryker 1990), followed by ips beetles, both of which are present naturally in all stands. Dwarf mistletoe is widespread but rarely fatal. In the absence of fire or another disturbance that reduces stem density, bark beetles can cause extensive tree mortality. Ponderosa pine has a high capacity to survive fire, better than all of its competitors except western larch (Minore 1979; Ryan and Reinhardt 1988). Thus, wildfire favors the growth of large-diameter ponderosa pine by killing its primary competitors and small-diameter ponderosa pines (Arno 1988; Steele et al. 1986).

Historical and Current Conditions

Wildfire historically promoted ponderosa pine dominance across most low-elevation savannas because of its high resistance to fire, including high-intensity fire. Fire exclusion, mining, timber harvest, and livestock grazing caused major reductions in ponderosa pine forests (Jain and Graham 2005). Changes in fire regime altered the composition and structure of the remaining dry forests (Hann et al. 1997), with area burned by surface fires decreasing (Page and Jenkins 2007), mean fire return interval increasing, and crown fires increasing (Hann et al. 1997). Mid-seral structures have increased, often containing dense stands of small ponderosa pine, Douglas-fir, and grand fir. The proportion of

dry forests occupied by late-seral, single-storied ponderosa pine has declined significantly, and Douglas-fir or grand fir is often common in the understory.

Climate Change Responses

Ponderosa pine is expected to be relatively tolerant of increasing temperatures and longer droughts. The Rocky Mountain variety has relatively high phenotypic plasticity and is therefore better adapted to drought, although regeneration may decrease if precipitation decreases or becomes more variable. Morales et al. (2015) projected an 11 percent increase in the range of ponderosa pine in the western United States, and Nitschke and Innes (2008) used gap modeling to project replacement of dry Douglas-fir with ponderosa pine in British Columbia. Hansen et al. (2001) projected that the range of ponderosa pine will expand in the western United States, whereas most other tree species ranges will decrease. Although species distribution models suggest that the range of ponderosa pine may decrease (Bell et al. 2014; Franklin et al. 1991; Gray and Hamann 2013) and rise in elevation (Crimmins et al. 2011) in a warmer climate, these projections are questionable because they do not consider on-the-ground growth processes and competition.

Advancing competition resulting from fire exclusion, increased wildfire extent and severity, and the potential for increased susceptibility to insects in warmer, drier conditions will dictate the future of ponderosa pine in the IAP region (Hann et al. 1997; Miller and Keen 1960; Negrón and Fettig 2014). If fires are too frequent, regenerating trees will not grow above the lethal scorch height and will not reach maturity. Increasing wildfire severity could also eliminate mature ponderosa pine trees that provide seed sources for populating future burns.

Singleleaf Pinyon (Pinus monophylla)

(Great Basin and Semi Desert subregion)

Autecology

Singleleaf pinyon is adapted to a wide variety of sites. It usually grows on pediments; dry, rocky slopes; ridges; and alluvial fans between 4,500 and 7,500 feet elevation (Lanner 1999; Stephenson and Calcarone 1999). It is frost resistant, drought tolerant, and shade intolerant (Lanner 1983), typically growing on shallow, well-drained, low-fertility soils, although it has been found on more productive soils as well (Evans 1988; Gottfried and Severson 1993; Gottfried et al. 1995). Pinyon pine typically grows in association with juniper species, where juniper dominates lower elevations of their range and pinyon the upper. Pinyon-juniper woodlands typically progress toward increased tree density and canopy cover over time (Everett 1985; Meeuwig et al. 1990; Short and McCulloch 1977; West et al. 1975), often expanding into adjacent grass and shrublands (Burwell 1998; West et al. 1975). Understory species make up a small portion of the total biomass in denser stands, although they may be important forage species and typically persist following disturbance (Everett and Koniak 1981).

Disturbance Interactions

In the Great Basin, there is evidence of both frequent, low-intensity fires carried by once-abundant perennial grasses, and less frequent, local stand-replacement fires during extreme conditions. Fires burned in irregular patterns, producing a mosaic of burned and unburned landscape. On high-productivity sites where sufficient fine fuels existed, fires burned every 15 to 20 years, and on less productive sites with patchy fuels, fire intervals were 50 to 100 years or longer. Fire frequency in singleleaf pinyon communities varies with fuel loads and ignition source, which, in turn, vary with habitat type, aspect, topography, stand history, and climatic conditions (Gruell 1999; Paysen et al. 2000). Cheatgrass has become increasingly common in the understory over the past few decades, continually providing abundant fine surface fuels and increasing the potential for more frequent wildfires (Shinneman and Baker 2009).

Pinyon dwarf mistletoe (*Arceuthobium divaricatum*) can cause extensive damage (Hawksworth and Wiens 1972), leaving trees susceptible to insect attack. Pinyon blister rust (*Cronartium occidentale*) occurs extensively on *Ribes* species in most western States but infects singleleaf pinyon only sporadically (Stillinger 1944), occasionally girdling small trees. Black stain root disease (*Ophiostoma wageneri*) occasionally kills singleleaf pinyon (Smith 1967b; Wagener and Mielke 1961). The disease spreads by root contact, and infection is confined to xylem in the roots and lower trunk. Pinyon ips (*Ips confusus*) is commonly found in pinyon woodlands, with outbreaks occurring when trees are stressed (Chapter 8).

Historical and Current Conditions

Singleleaf pinyon was a source of fuel (charcoal) for the mining industry between the 1860s and 1920s (Young and Budy 1979), and it has been used extensively for firewood and other uses, with local deforestation in some locations. In some lower-elevation locations, pinyon has been removed from the landscape to encourage the growth of grasses and forage for livestock grazing. Persistent woodlands of pinyon pine or juniper species, or a mixture of both, are found where local soils and climate are favorable, and wildfire has been infrequent (Romme et al. 2009). Pinyon-juniper savannas are found where local soils and climate are suitable for both trees and grasses, and low-intensity fires have been relatively frequent. Large increases in junipers have occurred in portions of pinyon-juniper woodlands (Romme et al. 2009). Damage to cryptobiotic crusts has caused erosion in some pinyon-juniper woodlands.

Climate Change Responses

Singleleaf pinyon is drought tolerant, and reduced soil moisture is not expected to have a significant effect on its abundance and distribution, although its growth may decrease over time. It is not as drought tolerant as the juniper

species with which it is associated, and may decrease in abundance where the species compete. However, it may be able to outcompete ponderosa pine at higher elevations.

The future of singleleaf pinyon will largely depend on spatial and temporal patterns of wildfire, which is expected to increase in frequency (Floyd et al. 2004). Pinyon is only moderately fire tolerant and is easily engaged in crown fires because of low crowns and high concentrations of volatile chemicals. After an initial fire, accumulation of surface fuels and tree regeneration is likely to be slow (unless Gambel oak [*Pinus gambelii*] is present) because of moisture limitations, resulting in a sparse canopy and disconnected fuels (Rocca et al. 2014). If fire frequency is high in areas where pinyon is codominant with ponderosa pine, the latter species will become more common and pinyon will become less common. The long-term condition of juniper is complicated by nonnative annual grasses, which increase surface fuels and fire frequency. Insects, especially pinyon ips, will also be an important stressor, especially during extended droughts.

Twoneedle Pinyon (Pinus edulis)

(Plateaus subregion)

Autecology

Pinyon-juniper woodlands are found between the low plains covered by grassland, desert shrub, or chaparral vegetation and the high mountains just below the zone dominated by either submontane shrubs or ponderosa pine. They grow best on higher, wetter sites of the woodland zone, just below ponderosa pine (Fowells 1965; Jameson et al. 1962). Twoneedle pinyon grows in semiarid to arid climates, often associated with oneseed juniper (*Juniperus monosperma*) and Utah juniper. Pinyon is drought tolerant and shade intolerant, and seedlings require extra moisture or shade until their elongating taproots reach deeper substrates (Mitchell 1984). The extensive root system of established pinyons and relatively rapid rate of root elongation, especially of young seedlings, enhance the ability of pinyon to survive in dry environments.

Disturbance Interactions

Small pinyon pines are sensitive to fire and may be killed by low-intensity fire (Floyd et al. 2000; McCulloch 1969), whereas larger trees tend to be somewhat resistant to surface fire because foliage is high enough above the ground to avoid crown scorch or other damage (Dwyer and Pieper 1967; Wittie and McDaniel 1990). Cheatgrass has become increasingly common in the understory over the past few decades, continually providing abundant fine surface fuels and increasing the potential for more frequent wildfires (Shinneman and Baker 2009). Foliage diseases include needle casts (*Elytroderma deformans*, *Bifusella saccata*) and needle rusts (*Coleosporium jonesii*, *C. crowellii*) (Fowells 1965; Hepting 1971). Pinyon blister rust and pinyon dwarf mistletoe damage stems; the latter species is considered the major pathogen of pinyon.

Historical and Current Conditions

Pinyon-juniper woodland expansion since the time of settlement has been attributed to several factors, including a warming climate, fire exclusion, increased populations of seed-dispersing birds and mammals, and reduced competition from grasses resulting from overgrazing by livestock (Everett 1987; Jameson 1970). In the absence of wildfire, fuels have accumulated in some stands, especially in more mesic sites, increasing the possibility of crown fire. Hazardous fuels reduction, including prescribed burning, has been used in some locations.

Climate Change Responses

Twoneedle pinyon is drought tolerant, and reduced soil moisture is not expected to have a significant effect on its abundance and distribution, although its growth may decrease over time. It is not as drought tolerant as juniper, and may decrease in abundance where the species co-occur. However, it may outcompete ponderosa pine at higher elevations. Since 2000, twoneedle pinyon at low-elevation sites in northern New Mexico has suffered significant mortality associated with extended drought and pinyon *Ips* (Breshears et al. 2009) (Chapter 8), and although similar mortality has not been widespread in Utah, it may be possible during long droughts. If pinyon mortality increases in the future, juniper would probably become more dominant.

The future of twoneedle pinyon will largely depend on spatial and temporal patterns of wildfire, which is expected to increase in frequency (Floyd et al. 2004). Pinyon pine is only moderately fire tolerant, and it is easily engaged in crown fires because of low crowns and high concentrations of volatile chemicals. After an initial fire, accumulation of surface fuels and tree regeneration will probably be slow (unless Gambel oak is present) because of moisture limitations, resulting in a sparse canopy and disconnected fuels (Rocca et al. 2014). If fire frequency is high in areas where pinyon pine is codominant with ponderosa pine, the latter species will become more common and pinyon pine will become less common. The long-term condition of juniper is complicated by nonnative annual grasses, which increase surface fuels and fire frequency. Insects, especially pinyon *Ips*, will also be an important stressor, especially during extended droughts.

Western White Pine (Pinus monticola)

(Great Basin and Semi Desert subregion)

Autecology

Western white pine occupies the extreme western Great Basin portion of the IAP region and is typically associated with other conifer species. It is limited by moisture at lower elevations and by temperature at higher elevations. Western white pine grows on a variety of sites, but is more common along moist creek bottoms, lower benches, and northerly slopes. Seedling establishment is favored by

partial shade in severe sites (Graham 1990) but minimal shade on northern slopes. Once established, it grows best in full sun. Seedlings have low drought tolerance, and first-year seedlings are subject to mortality from high surface temperatures on exposed sites. White pine attains dominance only after wildfire or in silvicultural systems that favor it. A generalist species with broad climate and environmental tolerances (Devine et al. 2012), western white pine adapts to different conditions through phenotypic plasticity and selective genetic differences.

Disturbance Interactions

Historically, white pine forests originated from wildfires, especially stand-replacing burns, but they were also maintained by frequent low-intensity fires (Barrett et al. 1991). When mature, white pine has thick bark and a high crown, which make it tolerant of fire. White pine blister rust has greatly decreased survival and vigor of white pine (Fins et al. 2002; Harvey et al. 2008), virtually eliminating this species in some locations. Armillaria root rot causes foliar chlorisis and root mortality, as well as reduced growth. Annosus root disease and laminated root rot also cause reduced vigor and some mortality. Bark beetles attack western white pine, killing groups of trees in mature forests, especially those weakened by blister rust (Chapter 8).

Historical and Current Conditions

Western white pine stands were previously more dominant in western North America (Harvey et al. 2008). It is much less abundant in mixed conifer forests as a result of logging, fire exclusion, and blister rust (Fins et al. 2002). This decline will probably continue to reduce abundance, and in some cases, cause local extirpation in the absence of assertive restoration.

Climate Change Responses

Western white pine may be well adapted to a warmer climate in some portions of its range (Loehman et al. 2011). It can disperse seeds into burned areas, which are likely to increase in the future, and a warmer climate may increase its productivity in some locations. However, the prevalence of white pine blister rust will make it difficult for white pine to persist in most forests (Fins et al. 2002; Harvey et al. 2008), and it is expected to continue to decline throughout much of its range.

Whitebark Pine (Pinus albicaulis)

(Middle Rockies, Southern Greater Yellowstone, Uintas and Wasatch Front subregions)

Autecology

Whitebark pine is an important component of upper subalpine forests in the IAP region (Arno and Hoff 1990). It supports unique components of floral and faunal diversity and promotes community development and stability (Tomback and Achuff 2010; Tomback et al. 2001). It is a long-lived tree, tolerates moderate shade (Minore 1979), grows slowly, and tolerates long periods of drought (Callaway et al. 1998). In the absence of wildfire, whitebark pine is replaced by subalpine fir and Engelmann spruce in some locations (Arno and Hoff 1990). Whitebark pine has a mutualistic relationship with Clark's nutcracker, which caches and disperses seeds (Tomback 1982, 1983). Whitebark pine is genetically diverse (Keane et al. 2012), allowing it to exist across many environments.

Disturbance Interactions

Whitebark pine fire regimes are complex and variable in space and time (Morgan et al. 1994b). Most fires in the upper subalpine zone burn in mixed-severity patterns that facilitate long-term survival of the species (Keane et al. 1994). Mountain pine beetle can damage mature stands, often causing high mortality. White pine blister rust is also damaging, preventing tree development and often causing mortality. Whitebark pine has some resistance to white pine blister rust, and although efforts at developing rust-resistant seed for regenerating burned and treated areas hold promise, restoration will need to occur at large spatial scales to be effective.

Historical and Current Conditions

Whitebark pine, a candidate species for listing under the U.S. Endangered Species Act (USFWS 2011), has been declining since the early 20th century from the combined effects of mountain pine beetle-caused mortality, fire exclusion, and spread of white pine blister rust (Schwandt 2006; Tomback and Achuff 2010) (fig. 6.4). Within the last decade, outbreaks of mountain pine beetle and increasing damage and mortality from blister rust have resulted in cumulative whitebark pine losses that have altered high-elevation community composition and ecosystem processes throughout much of western North America.

Climate Change Responses

Although whitebark pine was able to persevere through climatic variability in the past, it will be highly exposed to future climate change because of its confined distribution to upper subalpine environments. It is expected to continue to decline in abundance and vigor in a warmer climate, not because it is poorly adapted to an altered climate, but because it is experiencing so much stress from blister rust and mountain pine beetle that regeneration capability is greatly reduced (Bartlein et al. 1997; Bentz et al. 2016; Devine et al. 2012). In some cases, whitebark pine populations are so low that Clark's nutcracker is acting more as a seed predator than a seed disperser (Keane and Parsons 2010; Leirfallom et al. 2015). A warmer climate is expected to exacerbate this decline in most locations.



Figure 6.4—Whitebark pine. This species, which has been subjected to mortality from white pine blister rust for decades, may be more susceptible to mountain pine beetles in a warmer climate (photo: J. Beck, National Park Service).

Blue Spruce (Picea pungens)

(Middle Rockies, Southern Greater Yellowstone, Uintas and Wasatch Front, Plateaus, Great Basin and Semi Desert subregions)

Autecology

The shallow roots of blue spruce restrict it to moist sites where water is close to the surface (Lanner 1983). Blue spruce occurs at middle elevations on montane streambanks, well-drained floodplains, first-level terraces, ravines, intermittent streams, and gentle slopes (Fechner 1985; Hess and Alexander 1986; Lanner 1983). Spruce grows in cool climates that are sub-humid to humid and characterized by low summer temperatures and low winter precipitation. It is a pioneer species in riparian communities that are subject to periodic disturbances, such as scouring and flooding (Baker 1990; Fechner 1990; Szaro 1990). It is a shade-tolerant, mid- to late-seral species in montane and subalpine zones (Baker 1988; Schmidt and Larson 1989).

Disturbance Interactions

Blue spruce is easily killed by fire (Jones 1974; Wright and Bailey 1982). Insects and disease reduce growth, viability, and vigor of spruce (Fechner 1985; Walters 1978) (Chapter 8). Heart and root rots, cone rusts, nematodes, snow molds, canker, and tip blight can reduce the vigor of spruce (Fechner 1990; Nelson and Krebill 1982) and can cause mortality in older, low-vigor trees.

Historical and Current Conditions

Because blue spruce is located at high elevation and has no commercial value for timber, it has been relatively free of human influence, except in stands where it may have been associated with harvest of other species, such as Engelmann spruce. Western spruce budworm has killed patches of spruce and often other species in some locations, but this appears to be a normal occurrence in older, low-vigor stands.

Climate Change Responses

Climate change may reduce the functionality of riparian and wet meadow locations where blue spruce is commonly found. Therefore, its distribution and abundance could decrease locally if growth and vigor decline over time. Wildfire is currently uncommon in blue spruce communities, but if it becomes more frequent in a warmer climate, blue spruce will decrease in abundance because of fire. If fire frequency is high enough, spruce may not achieve dominance in the overstory.

Engelmann Spruce (Picea engelmannii)

(Middle Rockies, Southern Greater Yellowstone, Uintas and Wasatch Front, Plateaus, Great Basin and Semi Desert subregions)

Autecology

Engelmann spruce is widely distributed and is a major component of high-elevation forests in the IAP region (Alexander and Shepperd 1990) It occupies very cold environments in some locations. It is shade tolerant, although not as much as its common associate subalpine fir. This species is not drought tolerant, especially as a seedling (Alexander and Shepperd 1990), but tolerates frost and seasonal standing water. Pure Engelmann spruce is found in wet areas, but the species is usually mixed with other conifer species in upland locations. Seeds germinate in a variety of substrates, including litter and decomposed humus. Following establishment, survival is favored by adequate soil moisture, cool temperatures, and some shade.

Disturbance Interactions

Engelmann spruce is highly susceptible to fire injury and death, but some large spruce can survive severe burns (Bigler et al. 2005; Wadleigh and Jenkins 1996). It survives fire better than its primary associate, subalpine fir (Ryan and Reinhardt 1988). Surviving spruce can provide abundant seed in burned areas, although the subsequent forest may or may not be dominated by both spruce and

other species (e.g., subalpine fir) (Pfister et al. 1977). Engelmann spruce is susceptible to windthrow, especially after timber harvest and thinning. Several insect species are associated with Engelmann spruce (Chapter 8).

Historical and Current Conditions

Recent trends in Engelmann spruce forests across the IAP region are unclear. Advancing succession during many decades of fire exclusion has probably increased spruce abundance in subalpine and upper subalpine landscapes. But logging and fire have reduced spruce at lower elevations, where it occurs in seasonally wet areas and frost pockets. Several locations throughout the IAP region with mature Engelmann spruce have sustained extensive mortality from spruce beetle (*Dendroctonus rufipennis*) outbreaks.

Climate Change Responses

Some losses of Engelmann spruce are likely in drier portions of its range, especially in seasonally moist sites that will be drier in the future. Some mortality may have already occurred from recent drought (Liang et al. 2015). Higher temperatures can increase growth in some locations (Luckman et al. 1984) and reduce growth in other locations (Alberto et al. 2013). If wildfire frequency increases, it will probably reduce the extent of mature spruce, although it readily establishes following wildfire. Spruce beetle can cause greater stress in a warmer climate, especially in mature stands (Bentz et al. 2010). Although Engelmann spruce is sensitive to climate, it will probably persist in high-elevation landscapes, because of its genetic capacity to adapt to climatic variability by taking advantage of suitable microsites (Jump and Peñuelas 2005).

Quaking Aspen (Populus tremuloides)

(Middle Rockies, Southern Greater Yellowstone, Uintas and Wasatch Front, Plateaus, Great Basin and Semi Desert subregions)

Autecology

Quaking aspen is abundant in the IAP region, with its distribution limited by water availability and growing-season temperature. Aspen stems are relatively short lived and maintained by disturbance (Mueggler 1985; Rogers 2002), although belowground genets of aspen clones can survive for millennia. It is shade intolerant and sprouts aggressively following disturbance (usually fire), which kills most of the live stems, thus stimulating vegetative propagation (suckering) (Bartos 1978) and facilitating rapid reoccupation of the site. This species has substantial phenotypic variation, as evidenced by varied foliar morphology, stem morphology, and phenology among different clones.

Disturbance Interactions

Browsing of post-disturbance suckers by ungulates—including elk (*Cervus elaphus*), moose (*Alces alces*), and cattle—frequently damages seedlings and sprouted stems (Eisenberg et al. 2013; White et al. 1998), and in some cases mature trees, thus increasing susceptibility to insects and pathogens. Wildfire can kill aboveground stems (Bartos 1998) but also promotes new seedlings and suckering by eliminating conifers (Campbell and Bartos 2001; Shepperd et al. 2001). Following disturbance, aspen dominates a site for 40 to 80 years; thinning from insects and disease and succession (shading) by conifers eventually reduce aspen abundance (Mueggler 1985; Rogers 2002). Hypoxylon canker (*Hypoxylon mammatum*) causes significant damage in some locations (Perala 1990). Young trees can be killed by small rodents and mammals (Eisenberg et al. 2013).

Historical and Current Conditions

Since around 1970, aspen has been in a period of general decline that may be at least partly attributed to wildfire exclusion, allowing plant succession to proceed toward conditions that ordinarily exclude aspen (Campbell and Bartos 2001; Frey et al. 2004). Recent episodes of aspen dieback ("sudden aspen decline") have been superimposed on this general decline; the epidemiology begins with death of branch tips and progresses to death of mature trees and eventually death of entire clones (Frey et al. 2004). Dieback is suspected to be caused by periods of drought (Worrall et al. 2013). The worst symptoms are generally found at lower elevations.

Climate Change Responses

Seral aspen communities will respond to a warmer climate differently than mature aspen communities (Rice et al. 2017). Aspen on warmer sites could suffer high mortality because of increasing water deficit (Hogg and Hurdle 1995; Ireland et al. 2014). Extreme droughts (Frey et al. 2004) and high temperatures (Perala 1983) are of special concern, especially at the margins of aspen distribution at low elevation, and may weaken trees enough that insects and pathogens can cause tree mortality (Rice et al. 2017). Increased wildfire frequency, particularly on moist sites, is likely to favor aspen regeneration in the future by removing conifers. If future wildfires are severe, however, they may kill shallow root systems and locally extirpate aspen. In some locations, declining stands may have little regeneration because of ungulate herbivory (Rogers et al. 2013).

Fremont Cottonwood (Populus fremontii)

(Southern Greater Yellowstone, Uintas and Wasatch Front subregions)

Lanceleaf Cottonwood (Populus × acuminata)

(Great Basin and Semi Desert subregion)

Narrowleaf Cottonwood (Populus angustifolia)

(Southern Greater Yellowstone, Uintas and Wasatch Front, Plateaus, Great Basin and Semi Desert subregions)

Balsam Cottonwood (Populus balsamifera)

(Middle Rockies subregion)

Autecology

The four cottonwood species in the IAP region—Fremont cottonwood, lanceleaf cottonwood, narrowleaf cottonwood, and balsam cottonwood—grow primarily in seasonally wet to moist open-canopy sites, typically along streams and rivers. Cottonwood often dominates riparian communities on alluvial sites from 4,000 to 6,000 feet elevation, with other hardwood species, shrubs, and grasses in the understory. Cottonwood is shade and drought intolerant, requiring access to the water table during most of the growing season (Rood et al. 2003). High streamflows facilitate seedling establishment through scouring and deposition of alluvial sediments for germination of windborne seeds. High numbers of seedlings often become established after a flood, but thin over time. Seedlings and saplings are frequently injured and sometimes killed by early or late frosts (DeBell 1990).

Disturbance Interactions

Cottonwood is somewhat fire tolerant owing to its thick bark and high branches. It is a weak stump sprouter, but rarely regenerates from suckers (Brown 1996). Cottonwood can resprout and survive low-intensity fires in the short term (Gom and Rood 1999), but fire injuries can introduce diseases that weaken and sometimes kill trees (Borman and Larson 2002). Several insects attack cottonwood. Many fungal species cause decay in cottonwood, but only brown stringy heart rot (*Spongipellis delectans*) and yellow laminated butt rot (*Pholiota populnea*) cause significant damage. Russian olive (*Elaeagnus angustifolia*) and saltcedar (*Tamarix* spp.) are aggressive nonnative trees that can outcompete cottonwood in some locations, particularly during or after drought (Shafroth et al. 2002).

Historical and Current Conditions

Cottonwood species are well distributed within their respective habitats in the IAP region, although degradation of riparian areas by grazing and other land uses have damaged some trees and the functional integrity of the riparian system. Russian olive and saltcedar have displaced cottonwoods in many locations, thus altering local hydrological function, because the nonnative species take up more water than native species. Biological control releases of the nonnative northern tamarisk beetle (*Diorhabda carinulata*) in the Southwest, starting in the 1990s, have caused rapid mortality and decline in vigor in many saltcedar populations (Sher and Quigley 2013; Tracy and Robbins 2009).

Climate Change Responses

As snowpack declines and melts earlier with warming temperatures, there will be reduced, attenuated river flows, along with a possible shift in timing of peakflows. These shifts may decrease germination and establishment of young cottonwoods, depending on the relative timing of floods and seed production (Whited et al. 2007). Altered hydrological flow, which can also be caused by withdrawal for human use, will affect both floodplain interaction and water available to cottonwoods, which, in turn, can affect recruitment and establishment of seedlings (Auble and Scott 1998; Beschta and Ripple 2005). Upland conifers can potentially establish in the riparian zone if the local water table has dropped, increasing competition with cottonwoods. Long-term transport of seeds provides cottonwood with an effective mechanism for regeneration across large landscapes, conferring some resilience to future climate.

Sitka Alder (Alnus viridis subsp. sinuata)

(Middle Rockies subregion)

Thinleaf Alder (Alnus incana subsp. tenuifolia)

(Middle Rockies, Southern Greater Yellowstone, Uintas and Wasatch Front, Plateaus, Great Basin and Semi Desert subregions)

White Alder (Alnus rhombifolia)

(Middle Rockies subregion)

Autecology

Sitka alder and thinleaf alder are small, deciduous trees or shrubs found on a wide range of soils and wide range of elevations. Sitka alder is usually multistemmed and bushy, forming dense thickets. White alder is a deciduous, medium to large tree found on a variety of soils typically near permanent streams at low to mid-elevations. All species are found on moist, cool sites, typically riparian areas or other locations where a reliable water

source is available; for example, Sitka alder is often located in avalanche tracks. These species are associated with other hardwood and shrub species, mixed with or intermittent with overstory conifer species. All species are moderately shade tolerant (Haeussler and Coates 1986). Seeds require a moist mineral soil for germination, which normally takes place in the spring. Thinleaf alder can also propagate by cloning (Hall 1973). All alder species fix nitrogen through their association with actinomycetes.

Disturbance Interactions

Many sites occupied by alder species are subject to periodic flooding. Although tops may be damaged or killed, all species can sprout from root crowns. Wind-dispersed seeds readily germinate on alluvial soils exposed by floods or covered by sediment, and on bare soil created by wildfire, avalanches, and soil slumps. Alders have thin bark, and stems are easily killed by fire. Although alder can be killed by severe fire (Barro et al. 1989), it can also sprout following top kill by fire (Fischer and Bradley 1987). Although alder wood is resilient and somewhat limber, avalanches can damage Sitka alder and thinleaf alder, which often reproduce by sprouting (Oliver et al. 1985).

Historical and Current Conditions

Alders have rarely been disturbed by human activity because they have no timber value. They have been subjected to some stress in riparian areas that have been disturbed by water withdrawals or livestock grazing.

Climate Change Responses

In general, higher temperatures are not expected to have significant direct effects on alder species because alders are usually located in riparian areas that are buffered from temperature increases. However, smaller riparian areas may become drier in a warmer climate, especially if they rely on adjacent snowpack. Lower levels of soil moisture could reduce the vigor of alder and other species. Increased frequency of wildfire may be a significant stressor for white alder because it may not sprout vigorously after fire (Fryer 2014), possibly making associated species more competitive. Sitka alder and thinleaf alder can sprout after wildfire, so increased disturbance will probably not affect their distribution and abundance.

Velvet Ash (Fraxinus velutina)

(Uintas and Wasatch Front, Plateaus subregions)

Autecology

Velvet ash (also called desert ash or Arizona ash) is a deciduous tree with spreading branches and a rounded crown (if it has sufficient sunlight) that grows up to 30 feet tall when mature. It is found in riparian areas in canyons and along streambanks in desert mountains of southern Utah and southern Nevada above 3,000 feet elevation. Velvet ash grows in a variety of substrates, including alkaline soils. The presence of this species in the desert generally indicates a permanent underground water supply. It is shade intolerant, regenerates through wind dispersal of winged seeds, and can sprout from the base when damaged.

Disturbance Interactions

Velvet ash is easily top-killed by fire, but stumps can sprout vigorously following fire and mechanical damage and can attain prefire heights in 8 years (Winkel and Syzdek 2015). North American ash populations are at substantial risk from the introduction of emerald ash borer (*Agrilus planipennis*), which has now reached as far west as Colorado. Large-scale mortality of ash trees, as a result of borer infestations, would probably result in significant modifications in the composition and successional dynamics of many natural forests (MacFarlane and Meyer 2005) (Chapter 8).

Historical and Current Conditions

Little is known about the historical distribution or uses of velvet ash. It has no commercial value for timber but may have been used for firewood in some locations. Its populations are probably mostly intact, except where riparian areas have been modified.

Climate Change Responses

As soil moisture declines in a warmer climate, marginal riparian sites for velvet ash may become less favorable for regeneration and survival of young trees. With increases in fire frequency, there are likely to be increased vegetative regeneration and decreased production of seedlings following fire; fire would probably kill seeds on or near the soil surface, restricting seedling recruitment to surviving seed-producing trees. Low-intensity fires may promote regeneration by thinning stands and stimulating sprouting. Increased temperatures may promote ash seedling and mature tree growth by increasing soil temperatures. Browsing pressure on ash may increase with increased drought, as upland grasses and forbs desiccate and senesce earlier, or are replaced by less palatable species.

Water Birch (Betula occidentalis)

(Middle Rockies, Southern Greater Yellowstone, Uintas and Wasatch Front, Plateaus, Great Basin and Semi Desert subregions)

Autecology

Water birch is primarily a riparian species, occurring near waterways, wet swales, marshes, ravines, bogs, and moist woodlands (Arno and Hammerly 1977; Welsh et al. 1987). Water birch is common along streams in steep

areas of the IAP region, especially in coarse-textured, moist to wet soils (Sutton and Johnson 1974). Although common in semiarid climates (Arno and Hammerly 1977), water birch is not particularly drought tolerant (Merigliano 1996) but is moderately shade tolerant and flood tolerant. In the Great Basin, riparian habitats with water birch are found in upland habitats ranging from mountain big sagebrush (*Artemisia tridentata* subsp. *vaseyana*) shrublands (Manning and Padgett 1989) to fir forests (Lanner 1983).

Disturbance Interactions

Water birch often forms clumps by sprouting from the base of the trunk (Harrington 1964). Sprouts can develop after top kill by flooding or other physical injury (Hansen et al. 1995; Skinner et al. 2006) or without aboveground damage (Lanner 1983). This species can regenerate quickly following damage and disturbance.

Historical and Current Conditions

Little is known about the historical distribution and use of water birch. It has little commercial value but is sometimes used for firewood and fence posts. Birch is used as browse by ungulates, including livestock, to some extent. Birch stems may have increased in some areas where American beaver (*Castor canadensis*) populations were reduced or extirpated.

Climate Change Responses

Water birch adapts well to a wide range of climate and water availability (Disalvo and Hart 2002). As soil moisture declines with a warmer, drier climate, marginal riparian sites for birch may become less favorable for regeneration and survival of young trees. With increased fire frequency, there are likely to be better vegetative regeneration and decreased production of seedlings following fire events. Fire would probably kill seeds on or near the soil surface, restricting seedling recruitment to surviving seed-producing trees. Low-intensity fires may promote regeneration by thinning stands and stimulating sprouting. Birch productivity may benefit from increased temperatures because seedling and mature tree growth may increase with increasing soil temperatures. Browsing pressure may increase with increased drought, as upland grasses and forbs desiccate and senesce earlier, or are replaced by less palatable species.

Boxelder (*Acer negundo*)

(Southern Greater Yellowstone, Uintas and Wasatch Front, Plateaus subregions)

Autecology

Boxelder is a fast-growing and fairly short-lived hardwood that grows in riparian and palustrine communities. It generally grows on moist sites along lakes and streams, on floodplains, and in low-lying wet places where its shallow root system can find abundant moisture (Lanner 1983). Resilient to climate extremes (Preston 1948), boxelder is drought tolerant once established and can withstand short periods of flooding (Sutton and Johnson 1974). It is moderately shade tolerant but does not reproduce in its own shade. Boxelder roots are shallow and spreading, except in deep soils (Preston 1948; Sutton and Johnson 1974). It tolerates a wide range of soils but grows more vigorously in well-drained soils (Medina 1986).

Disturbance Interactions

Boxelder grows on moist bottomland sites, which are seldom subject to burning. This thin-barked species is easily injured by fire (Van Dersal 1938), regenerating via sprouting and seeds. It produces large annual crops of wind-dispersed seeds that germinate on a wide variety of soils. It also sprouts from the root crown, stump, or exposed roots following top kill by mechanical damage (Hansen and Hoffman 1988; Nix and Cox 1987). Verticillium wilt (*Verticillium albo-atrum*) is the only notable disease that kills boxelder, although it is also susceptible to stem canker caused by eutypella canker (*Eutypella parasitica*). Boxelder bugs (*Boisea trivittata*) infest boxelder trees and other maples, but do not cause significant damage.

Historical and Current Conditions

Boxelder was used for windbreaks and erosion control in many parts of the West. It has no commercial value but is sometimes used for firewood. It is used as browse by ungulates, and although it is unpalatable to livestock, the animals may damage stems while seeking shade. It is possible that boxelder stems have increased in some areas where American beaver populations were reduced or extirpated (Dieter and McCabe 1989). It is probably mostly intact from an ecological perspective.

Climate Change Responses

Boxelder exists across a broad range of soils and topographic locations, but as soil moisture declines in a warmer climate, marginal riparian sites may become less favorable for regeneration and survival of young trees. With increased fire frequency, there are likely to be increased vegetative regeneration and decreased production of seedlings following fire. Fire is likely to kill seeds on or near the soil surface, restricting seedling recruitment to surviving seed-producing trees. Low-intensity fires may promote regeneration by thinning stands and stimulating sprouting. Boxelder productivity may benefit from increased temperatures because seedling and mature tree growth may increase with increasing soil temperatures. Browsing pressure on boxelder may increase with increased drought, as upland grasses and forbs desiccate and senesce earlier, or are replaced by less palatable species.



Figure 6.5—Gambel oak sprouts vigorously following wildfire, as shown in both the foreground and background. The distribution and abundance of this species may increase in a warmer climate, replacing conifers as a dominant species in some locations (photo by Heath Haussamen, used with permission).

Gambel Oak (Quercus gambelii)

(Uintas and Wasatch Front, Plateaus, Great Basin and Semi Desert subregions)

Autecology

Gambel oak is a small deciduous tree or large shrub that is widespread in foothills and lower mountain locations of the IAP region. The tree typically grows at between 3,000 and 10,000 feet elevation, where average annual precipitation is 10 to 24 inches. Oak height is typically 10 to 30 feet depending on soil type and water availability. Branches are irregular and crooked, making them flexible enough to bend without breaking when covered with snow. Deep roots, xeromorphic leaves, and efficient water transport contribute to the high drought tolerance of Gambel oak (Kolb and Stone 2000), which grows in both pure stands and associated with ponderosa pine, pinyon pine, and other hardwood tree and shrub species. In most of its range, Gambel oak regeneration depends more on sprouting than establishment from seed (Clary and Tiedemann 1986; Larsen and Johnson 1998).

Disturbance Interactions

Gambel oak responds to fire by vegetative sprouting from its lignotuber and rhizomes, and even the stems can survive low-intensity fires (Harper et al. 1985) (fig. 6.5). Fire frequency in oak stands depends on accumulation of fuels by both oak and associated species (Mitchell 1984). Following wildfire, sprouts continue to grow, and natural thinning occurs, adding dead stems to the fuel bed. In the absence of fire, sprouts form young poles. At the pole stage, fires cause 100 percent stem mortality, either creating openings within stands for resprouting or cycling back to a grass-forb stage. In the absence of fire, Gambel oak stands reach maturity in 60 to 80 years. Dense understories of oak may serve as ladder fuels that carry fire to overstory conifers, increasing fire risk to adjacent species. Fire in some ponderosa pine stands can convert to thickets of Gambel oak, initiating a Gambel oak successional stage after the competing ponderosa pine overstory is removed (Dick-Peddie and Moir 1970). Late-spring frosts that kill oak leaves can cause extreme fire behavior later in the summer; the dead leaves tend to cling to the stem and act as dry aerial fuels (Jester et al. 2012). Many insects and diseases are associated with Gambel oak.

Historical and Current Conditions

Gambel oak acorns have been an important food for Native Americans for thousands of years, and the species is widely used for firewood. Oak density has been reduced in some areas with herbicides, mechanical treatments, and prescribed burning, typically to reduce fire hazard and protect overstory species such as ponderosa pine. In some areas where multiple wildfires have occurred in the past 30 years, oak appears to be increasing in dominance through sprouting and mortality of conifers (e.g., Adams and Dockter n.d.) combined with slow regeneration of the overstory.

Climate Change Responses

Gambel oak is very tolerant of low soil moisture, providing an advantage in a warmer climate with more droughts. Oaks sprout readily following wildfire, and with an expected increase in fire in the future, oaks may retain dominance or codominance in most locations. Being adapted to both drought and fire will improve the competitive status of oak with co-occurring tree species such as ponderosa pine and pinyon pines and probably with other shrub species, except at the lowest elevations where shrub-steppe systems dominate. Therefore, it is likely that Gambel oak will increase in abundance and possibly distribution in a warmer climate with more fire.

Curl-Leaf Mountain Mahogany (Cercocarpus ledifolius)

(Uintas and Wasatch Front, Plateaus, Great Basin and Semi Desert subregions)

Autecology

Curl-leaf mountain mahogany is a drought-tolerant, somewhat shade-tolerant, slow-growing tree or tall shrub (Lacey and Mosley 2002; Lanner 1983) found at 4,000 to 8,000 feet elevation in the IAP region (Brotherson 1990), typically in scattered patches and in extensive pure stands on dry, rocky slopes between conifer and desert steppe communities (Munz 1973; Stubbendieck et al. 1992). The root system is shallow and spreads widely (Sutton and Johnson 1974), typically in shallow to deep, well-drained, low-fertility sandy loam soils (Davis 1990; Hickman 1993).

Disturbance Interactions

Wildfires usually cause mortality of curl-leaf mountain mahogany, although older plants with thick bark may survive low-intensity fires (Gruell et al 1985; Martin and Johnson 1979). Postfire regeneration is primarily by seedling establishment (Gruell et al. 1985), and sprouting after fire is rare (Tisdale and Hironaka 1981). Several species of insects, including mountain-mahogany looper (*Iridopsis clivinaria*), feed on mountain mahogany, but do not generally cause significant damage.

Historical and Current Conditions

Curl-leaf mountain mahogany was used by Native Americans to make bows and other small implements. Euro-Americans first used it as fuel for mining in the 1860s. The species provides forage for ungulates and wildlife. Livestock can damage productivity where grazing is heavy (Smith 1967a). Fire exclusion has facilitated increased mountain mahogany abundance and successful regeneration in some locations (Gruell 1982; Kay 2003), allowing it to compete with more fire-adapted species. Mountain mahogany is occasionally killed with herbicides, mechanical removal, or prescribed fire to improve range quality or reduce fuel bed continuity.

Climate Change Responses

Curl-leaf mountain mahogany is drought tolerant, so it should continue to be productive in a warmer climate with more droughts, especially compared to other species. However, if wildfire frequency and extent increase as expected, it will be at a disadvantage because it is not fire tolerant and does not regenerate by sprouting. If wildfire is sufficiently frequent, new seedlings may be successively killed, thus reducing the abundance of mountain mahogany across fire-prone landscapes.

Bigtooth Maple (Acer grandidentatum)

(Uintas and Wasatch Front, Plateaus, Great Basin and Semi Desert subregions)

Autecology

Bigtooth maple is an important component of montane riparian communities in the IAP region (Fitzhugh et al. 1987; Moir 1982), typically located in canyons, in ravines, along mountain streams, and on lower slopes (Alexander et al. 1984; Cronquist et al. 1997) at 4,000 to 7,000 feet elevation. It is more abundant in the bottom than in the top of snowmelt drainages. In Idaho, it grows on hillsides, below springs and seeps, and on secondary floodplains of narrow canyon drainages (Hall and Hansen 1997). It is found on upper slopes in the Wasatch Mountains, although it is more common on mesic, north-facing slopes than on drier south-facing slopes (Dina and Klikoff 1973; Ehleringer et al. 1992). Bigtooth maple is drought tolerant (Sorenson et al. 1984; Sutton and Johnson 1974) and cold hardy, and tolerates summer temperatures above 100 °F (Sorenson et al. 1984).

Disturbance Interactions

Although bigtooth maple can be top-killed by fire, plants can survive by sprouting from the root crown (Bradley et al. 1992; Harper et al. 1992). In a severe burn, this species is likely to be killed (Harper et al. 1992). Smaller stems are more likely to be killed by fire, clearing areas for new tree seedlings and sprouts from surviving larger trees.

Historical and Current Conditions

Little is known about the historical distribution and use of bigtooth maple. It has no economic value except as firewood. The species is used for forage and cover by native ungulates and for cover by livestock (Hall and Hansen 1997). Bigtooth maple is useful for restoration of sites where vegetation has been denuded because it establishes deep roots, even in infertile soils (Barker 1977).

Climate Change Responses

Because bigtooth maple is drought tolerant, its productivity may be relatively stable in a warmer climate, especially because it is located in canyons and other places where water is typically present. As soil moisture declines in a warmer climate, marginal riparian sites for maple may become less favorable for regeneration and survival of young trees. With increased fire frequency, there are likely to be increased vegetative regeneration and decreased production of seedlings in some locations. Low-intensity fires may promote regeneration by thinning stands and stimulating sprouting. Browsing pressure on maple could increase with increased drought, as grasses and forbs desiccate and senesce earlier, or are replaced by less palatable species.

Crack Willow (Salix fragilis)

(Southern Greater Yellowstone, Uintas and Wasatch Front, Plateaus subregions)

Autecology

Crack willow is native to Europe and western Asia, where it is found in riparian habitats, usually growing beside rivers and streams and in marshes and wet meadows. It grows in similar habitats in the IAP region. The tree grows rapidly to a height of 30 to 60 feet, occasionally reaching 90 feet. Stem fragments are spread by water, and suckers spread locally. Crack willow outcompetes native species in riparian sites and forms dense, often pure stands along channels (Czarapata 2005), in some cases causing blockages, flooding, and structural changes in waterways (Weedbusters n.d.). This species is susceptible to wind, ice, and snow damage.

Disturbance Interactions

Crack willow responds favorably to periodic flooding. Broken twigs and branches can take root readily, enabling the species to colonize new areas as broken twigs fall into waterways and can be carried some distance downstream. Crack willow is assumed to respond to wildfire like most willow species, by sprouting from the root crown following top kill.

Historical and Current Conditions

Crack willow is now well established in many riparian areas in the IAP region. It was planted for erosion control and water uptake in some locations where rapid plant growth was desired. Although a nonnative species, crack willow is not listed as a noxious weed in any State in the IAP region, and it provides habitat for native bird species.

Climate Change Responses

As soil moisture declines in a warmer climate, marginal riparian sites for crack willow may become less favorable for regeneration and survival of young trees. With increased fire frequency, there are likely be increased vegetative regeneration and decreased production of seedlings in some locations. Even if this species is inhibited somewhat by a warmer climate, it is unclear whether native species could outcompete it.

Saltcedar (Tamarix spp.)

(Uintas and Wasatch Front, Plateaus subregion)

Autecology

Saltcedar (five species) is a nonnative, shrub-like tree with numerous large basal branches and a deep, extensive



Figure 6.6—Saltcedar. This species has caused widespread damage in riparian areas, although the recently introduced tamarisk beetle is a promising biocontrol (photo: M. Mejia, Bureau of Land Management).

root system that extends to the water table and is capable of extracting water from unsaturated soil layers. Saltcedar is found on lakeshores, in riparian floodplain habitats, on seasonally submerged sites, and in fine fluvial substrates (Diggs et al. 1999; Welsh et al. 1987). It is competitively superior to most native species under dry, saline conditions (Stromberg 1998; Vandersande et al. 2001), and few species can tolerate the understory environment (Brotherson and Winkel 1986) (fig. 6.6). Saltcedar is less sensitive to changes in groundwater availability than native riparian trees with which it is commonly associated. Greater tolerance of water stress can lead to saltcedar dominance on relatively dry riparian sites (Horton et al 2001; Smith et al. 1998; Stromberg 1998). Seedlings establish as flood waters recede, leaving moist deposits of bare soil along riparian corridors. Its small, wind- and water-dispersed seeds make it ideally suited as a pioneer species on these sites. Saltcedar is also early seral after fire (Busch and Smith 1993; Stuever et al. 1997). It is listed as a noxious weed in Nevada.

Disturbance Interactions

Evidence for specialized adaptation to wildfire in saltcedar remains unclear, despite its efficient postfire recovery—it is usually top-killed by fire but sprouts readily from the root crown (Busch 1995). Flammability increases with accumulation of dead and senescent woody material within the plant. When plants burn under high fuel loads, fire tends to be more severe, increasing the likelihood of killing the root crown of some individuals (Hohlt et al. 2002). Saltcedar plants can have many stems and high rates of stem mortality, resulting in a dense accumulation of dead, dry branches. Fire hazard peaks in tamarisk stands at 10 to 20 years of age (Ohmart and Anderson 1982). With a combination of flood suppression, water stress, and invasion by saltcedar, wildfires have replaced floods as the primary disturbance factor in many southwestern riparian systems.

Historical and Current Conditions

Saltcedar was introduced in North America in the 1800s, spreading rapidly in the southwestern and intermountain western United States in the 1920s, and altering the ecology and hydrology of riparian areas in this broad region. Control of saltcedar has proven to be challenging. Cutting, burning, and herbicides have been used in various combinations to reduce saltcedar populations, but treatments need to be conducted at large spatial scales to make a significant difference (Racher and Mitchell 1999). Biological control releases of the nonnative northern tamarisk beetle in the Southwest, starting in the 1990s, have caused rapid mortality and decline in vigor in many saltcedar populations (Sher and Quigley 2013; Tracy and Robbins 2009). The success of the beetle as a biological control agent is aiding the recovery of previously suppressed native riparian species. However, because the endangered southwestern willow flycatcher (*Empidonax traillii extimus*) now uses saltcedar as habitat, concern exists about beetles causing the loss of flycatcher habitat, and introduction of beetles has been restricted in some locations.

Climate Change Responses

Saltcedar is more drought tolerant and more efficient at obtaining water than most native species with which it is associated in riparian areas, so it is not expected to be vulnerable to a warmer climate. It also sprouts readily after wildfire, so it will be able to persist in a warmer climate with more fire. Despite efforts to control saltcedar, it would appear to be a permanent fixture in many riparian systems regardless of climate change.

Climate Change Assessment for Forest Vegetation Types

Vegetation types are broad species assemblages used to identify the geographic distribution of vegetation in the USFS Intermountain Region (table 6.1). These types are used to characterize broad landscapes for mapping, planning, and various aspects of vegetation management, but do not have specific spatial definitions. Here we describe the likely response of forest vegetation types to climate change, based on the preceding species descriptions (box 6.3).

Subalpine Pine Forest

Subalpine forests dominated by whitebark pine will be highly vulnerable in a warmer climate, primarily because this species is already subjected to considerable stress from white pine blister rust and mountain pine beetle (Chapter 8). As a result, populations are in decline and reproductive capacity is limited, even when germination conditions are suitable. In areas where wildfire has been excluded for many decades, subsequent fuel loading may create intense future fires that lead to mortality of mature trees. Decline in whitebark pine would have cascading effects on other species that eat its seeds, especially Clark's nutcracker. Subalpine forests in which bristlecone pine is a major component are mostly in dry locations that could become increasingly stressed by low soil moisture, which would reduce growth.

Other subalpine forests are expected to be moderately affected by a warmer climate. Limber pine, subalpine fir, Engelmann spruce, and white fir may all have increased growth in the upper subalpine zone because of a longer, snow-free growing season. These species may migrate to higher elevations where conditions are suitable, although this would be a slow process over many decades. If wildfire increases in the subalpine zone, especially where it has been excluded in the past, crown fires may be prevalent, quickly eliminating mature trees across the landscape. Limber pine is challenged by mountain pine beetle and white pine blister rust (Chapter 8). Quaking aspen found in subalpine forests will be minimally affected by a warmer climate, especially compared to aspen at lower elevations.

Subalpine Spruce-Fir Forest

Spruce-fir forest will be moderately vulnerable to a warmer climate. Subalpine fir, Engelmann spruce, and blue spruce may all have increased growth in the upper subalpine zone because of a longer, snow-free growing season, so overall productivity could increase. These species may migrate to higher elevations where conditions are suitable, although this would be a slow process over decades to centuries. If wildfire increases in the subalpine zone, especially where it has been excluded in the past, crown fires may be prevalent, quickly eliminating mature trees across the landscape.

Often a seral species in spruce-fir forests, lodgepole pine is a host of mountain pine beetle. Bark beetle outbreaks in Engelmann spruce and lodgepole pine are often severe and can accelerate succession in areas of high tree mortality (Chapter 8). Most subalpine species are fire intolerant, but because most lodgepole pine populations have serotinous cones and the potential for rapid, dense regeneration, it is likely to persist in high-elevation landscapes. Quaking aspen in subalpine forests will be minimally affected by a warmer climate, especially compared to aspen at lower elevations. Where Douglas-fir is a seral species, it could increase in distribution and abundance where sufficient soil water is available. In addition, Douglas-fir is more fire tolerant than any of its associates, so it may become more common if fire increases.

Mesic Mixed Conifer Forest

The composition of mesic mixed conifer forest varies greatly across the IAP region, with site conditions and species assemblages determining vulnerability to climate change. In general, late-seral forests may become increasingly susceptible to wildfire, especially where fire has been excluded for many decades and fuel loads are elevated. Shasta red fir has some fire tolerance, but other firs and lodgepole pine are subject to high mortality from intense fires. The firs are intolerant of low soil moisture, so as snowpack declines and summer temperature increases, growth and productivity will probably decrease, except on north aspects.

Douglas-fir, ponderosa pine, and Jeffrey pine have high tolerance to fire and can survive mixed- severity fires. Therefore, if wildfire extent and intensity increase in the future (Chapter 8), these species may become relatively more common, and late-seral species may become less common. Douglas-fir, ponderosa pine, and Jeffrey pine are all tolerant of dry soils, so they are likely to persist across the landscape, but their growth rates will probably decrease. Lodgepole pine and quaking aspen, which are also common in this forest type, both respond to wildfire with rapid, abundant regeneration and are expected to persist across the landscape, possibly with increased stress from insects and pathogens (Chapter 8).

Dry Mixed Conifer Forest

The composition of dry mixed conifer forest varies across the IAP region, with site conditions and species assemblages determining vulnerability to climate change. Located in lower-elevation montane sites, often on steep slopes and shallow soils, this forest type contains some of the most drought-tolerant species in the region. Common seral species include ponderosa pine, which is fire tolerant and regenerates well after fire, and quaking aspen, which sprouts heavily and reproduces after fire. The woodland species curl-leaf mountain mahogany, Gambel oak, and bigtooth maple are drought tolerant, and the latter two sprout vigorously after fire. Therefore, a major component of mixed conifer forest is expected to be able to cope with both drier soils and increased wildfire.

Twoneedle pinyon and singleleaf pinyon are drought tolerant, and although intense fire typically kills them, they can usually regenerate successfully if competition is minimal. Singleleaf pinyon at its lowest elevational extent in northern New Mexico has undergone significant mortality from prolonged drought and pinyon engraver beetles during the past 15 years (Floyd et al. 2009) (Chapter 8), so this species may be susceptible to increasing drought in the future. Limber pine, which is considered late seral in these forests, is drought tolerant, but may be challenged by mountain pine beetle, white pine blister rust, and increasing (usually fatal) wildfire (Chapter 8).

Other species such as Douglas-fir and white fir are not nearly as drought tolerant as other mixed conifer species. Their growth will probably decrease in a warmer climate, and although Douglas-fir has relatively high fire tolerance, white fir tolerates fire only when it has large-diameter and thick bark. In a warmer climate with more fire, it will be increasingly difficult for these conifer species to compete with early-seral and woodland species that are more tolerant of both drought and fire. Therefore, it is likely that early-seral species will become more dominant in the future, and late-seral species will become less common and perhaps confined to north aspects and valley bottoms.

Aspen-Mixed Conifer Forest

The composition of this forest type is diverse, distinguished by the prominent role of quaking aspen as an early-seral species, often in combination with other conifer and woodland species. Response to climate change will depend on associated species, ranging from high to low elevation, and from north to south aspects. Increased wildfire frequency and extent will be the primary factor determining future composition and structure of aspen-mixed conifer forests.

Most of the higher-elevation, late-seral species in this forest type (firs, Engelmann spruce) are readily killed by fire, especially when immature. If wildfire reaches into the subalpine zone, it is likely that mature spruce-fir forests will become less common, or will persist only on northern slopes and in valley bottoms. Therefore, early-seral species,

especially aspen, will attain increasing dominance because of their ability to resist fire or regenerate after it occurs. This will also be true at lower elevations in this forest type, where species such as ponderosa pine can readily survive intense fires, and other species such as aspen and Gambel oak sprout aggressively after fire. Productivity in these systems will probably be lower in a warmer climate with more fire. But the more fire-tolerant species will persist, especially in drier locations, where they can outcompete species that are susceptible to drought and fire.

Persistent Aspen Forest

Quaking aspen can persist for many decades in some forests in the IAP region, where productivity is relatively low and conifer species do not compete well. Succession proceeds slowly in persistent aspen forests, even in the absence of wildfire, especially at the higher elevations of the subalpine zone. The late-seral species in this forest type (firs, spruces) are readily killed by fire, especially when immature. Consequently, if wildfire reaches into the subalpine zone, it is likely that mature spruce-fir forests will become less common, or will persist only on northern slopes and in valley bottoms. Therefore, aspen will maintain dominance because of its ability to sprout aggressively after fire.

This will also be true at lower elevations in this forest type, where species such as ponderosa pine can readily survive intense fires, and other species such as aspen and Gambel oak sprout aggressively after fire. Douglas-fir will probably persist at some locations on the landscape because it has relatively high drought tolerance and fire tolerance. Productivity in these systems will probably be lower in a warmer climate with more fire. But the more fire-tolerant species will persist, especially in drier locations, where they can outcompete species that are susceptible to drought and fire.

Montane Pine Forest

Ponderosa pine is a dominant species in drier montane locations throughout much of the IAP region. Several other conifer species are included in this forest type, but are rarely as abundant as ponderosa pine, except in wetter locations (north aspects, valley bottoms). Ponderosa pine is persistent in these systems because it is tolerant of drought and very tolerant of fire. Consistently drier soils will cause this species to grow slower, but mortality will be rare unless drought lasts for several consecutive years and biotic agents cause additional stress (Chapter 8).

The expected increase in frequency and extent in a warmer climate will favor ponderosa pine over its less fire-tolerant competitors, thus ensuring dominance in most forests. But limber pine and bristlecone pine will probably persist at higher elevations, where fuel loads are typically low. An exception might be in areas where fire exclusion has increased stand density and fuel loads conducive to crown fires, but even then, regeneration of ponderosa pine will probably be sufficient to maintain dominance after fire. If insects become more prevalent in a warmer climate (Chapter 8), they could increase stress and mortality in pine species, especially during drought periods.

Riparian Forest

Riparian forests are distributed throughout the IAP region, adjacent to lakes, streams, seeps, springs, and high water tables. Vegetation is extremely diverse, including a broad range of conifer and hardwood species. Many of these species occur only in riparian systems, providing habitat for numerous wildlife species. In some lower-elevation, drier locations, nonnative saltcedar and Russian olive have been present, and, in some cases, dominant for many decades, displacing native species and reducing available groundwater.

Riparian systems will be one of the most vulnerable vegetation types in a warmer climate because they depend on reliable water supply. Higher temperatures will accelerate evapotranspiration as soils dry faster and as vegetation takes up water earlier and faster during the growing season. Both surface and subsurface water flows will decrease if snowpack decreases and melts earlier, precluding recharge during dry summers (Chapter 4). At a minimum, this will alter vegetation dominance to species that are more tolerant of seasonal drought, including ponderosa pine and other deep-rooted conifers. Hardwood species that rely on periodic high water levels for regeneration could become less common. Riparian forests associated with small or transient water sources (e.g., springs) will be more susceptible than forests near large water sources (e.g., rivers). Low-elevation riparian forests near small water sources will be more susceptible than high-elevation forests that have long duration of snowpack.

References

Adams, K.R.; Dockter, A.R. [n.d.]. Five years of fire recovery in Mesa Verde National Park, USA. https://www.nps.gov/meve/learn/management/upload/mesa_verde_fire_ecology_adams_dockter.pdf [Accessed October 30, 2016].

Agee, James K. 1982. True fir management for wilderness, water, recreation and wildlife values. In: Oliver, C.D.; Kenady, R.M., eds. Proceedings of the biology and management of true fir in the Pacific Northwest symposium; 1981 February 24-26; Seattle-Tacoma, WA. Contribution 45. Seattle, WA: University of Washington, College of Forest Resources: 227–237.

Agee, J.K. 1991. Fire history of Douglas-fir forests in the Pacific Northwest. In: Ruggiero, L.F.; Aubry, K.B.; Carey, A.B.; Huff, M.H., eds. Wildlife and vegetation of unmanaged Douglas-fir forests. Gen. Tech. Rept. PNW-GTR-285. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 25–33.

Aitken, S.N.; Yeaman, S.; Holliday, J.A.; [et al.]. 2008. Adaptation, migration or extirpation: Climate change outcomes for tree populations. Evolutionary Applications. 1: 95–111.

Alberto, F.J.; Aitken, S.N.; Alía, R.; [et al.]. 2013. Potential for evolutionary responses to climate change—evidence from tree populations. Global Change Biology. 19: 1645–1661.

Alexander, B.G.; Ronco, F.; Fitzhugh, E.L.; Ludwig, J.A. 1984. A classification of forest habitat types of the Lincoln National Forest, New Mexico. Gen. Tech. Rep. RM-GTR-104. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 29 p.

Alexander, R.R.; Shearer, R.C.; Shepperd, W.D. 1990. *Abies lasiocarpa* (Hook.) Nutt. subalpine fir. In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America: Volume 1, conifers. Washington, DC: U.S. Department of Agriculture, Forest Service. 149–166.

Alexander, R.R.; Shepperd, W.D. 1990. *Picea engelmannii Parry* ex Engelm. Engelmann spruce. In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America: Volume 1, conifers. Washington, DC: U.S. Department of Agriculture, Forest Service: 403–444.

Allen, C.D.; Macalady, A.K.; Chenchouni, H.; [et al.]. 2010. A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. Forest Ecology and Management. 259: 660–684.

Amaranthus, M.P.; Cazares, E.; Perry, D.A. 1999. The role of soil organisms in restoration. In: Meurisse, R.T.; Ypsilantis, W.G.; Seybold, C., eds. Proceedings: Pacific Northwest forest and rangeland soil organism symposium: Organism functions and processes, management effects on organisms. Gen. Tech. Rep. PNW-GTR-461. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 179–189.

Anderegg, W.R.L.; Berry, J.A.; Smith, D.D.; [et al.]. 2012. The roles of hydraulic and carbon stress in a widespread climate-induced forest die-off. Proceedings of the National Academy of Sciences, USA. 109: 233–237.

Anderson, L.J.; Winterton, A.J. 1996. Germination as a determinant of seedling distributions among natural substrates in *Picea engelmanni* (Pinaceae) and *Abies lasiocarpa* (Pinaceae). American Journal of Botany. 83: 112–117.

Antos, J.A.; Shearer, R.C. 1980. Vegetation development on disturbed grand fir sites, Swan Valley, northwestern Montana. Res. Pap. INT-RP-251. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station.

Araújo, M.B.; Whittaker, R.J.; Ladle, R.J.; Erhard, M. 2005. Reducing uncertainty in projections of extinction risk from climate change. Global Ecology and Biogeography Letters. 14: 529–538.

Arno, S.F. 1980. Forest fire history of the northern Rockies. Journal of Forestry. 78: 460–465.

Arno, S.F. 1988. Fire ecology and its management implications in ponderosa pine forests. In: Ponderosa pine: The species and its management, September 1987, Spokane, WA. Pullman, WA Washington State University, Extension Service: 133–139.

Arno, S.F. 2010. The Seeley Lake larch: Living link to Indian and frontier history. Forest History Today. Spring/Fall 2010: 13–19.

Arno, S.F.; Gruell, G.E. 1983. Fire history at the forest-grassland ecotone in southwestern Montana. Journal of Range Management. 36: 332–336.

Arno, S.F.; Hammerly, R.P. 1977. Northwest trees. Seattle, WA: The Mountaineers. 222 p.

Arno, S.F.; Hoff, R. 1990. *Pinus albicaulis* Engelm. whitebark pine. In: Silvics of North America. Volume 1, conifers. Agric. Handbk. Washington, DC: U.S. Department of Agriculture, Forest Service: 268–279.

Arno, S.F.; Parsons, D.J.; Keane, R.E. 2000. Mixed-severity fire regimes in the northern Rocky Mountains: Consequences of fire exclusion and options for the future. In: Wilderness science in a time of change conference, volume 5: Wilderness ecosystems, threat and management, Missoula, MT, May 23-27, 1999. Proceedings RMRS-P-15-Vol 5. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 225–232.

Aston, I.W. 2010. Observed and projected ecological response to climate change in the Rocky Mountains and Upper Columbia Basin: A synthesis of current scientific literature. Nat. Res. Rep. NPS/ROMN/NPR-2010/220. Fort Collins, CO: National Park Service. 98 p.

Atzet, T.; Wheeler, D.L. 1982. Historical and ecological perspectives on fire activity in the Klamath geological province of the Rogue River and Siskiyou National Forests. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region. 16 p.

Auble, G.T.; Scott, M.L. 1998. Fluvial disturbance patches and cottonwood recruitment along the upper Missouri River, Montana. Wetlands. 18: 546–556.

Austin, Dennis D. 1987. Plant community changes within a mature pinyon-juniper woodland. The Great Basin Naturalist. 47: 96–99.

Axelson, J.N.; Alfaro, R.I.; Hawkes, B.C. 2009. Influence of fire and mountain pine beetle on the dynamics of lodgepole pine stands in British Columbia, Canada. Forest Ecology and Management. 257: 1874–1882.

Bachelet, D.; Neilson, R.P.; Hickler, T.; [et al.]. 2003. Simulating past and future dynamics of natural ecosystems in the United States. Global Biogeochemical Cycles. 17: 1045.

Baker, W.L. 1988. Size-class structure of contiguous riparian woodlands along a Rocky Mountain river. Physical Geography. 9: 1–14.

Baker, W.L. 1989. A review of models of landscape change. Landscape Ecology. 2: 111–133.

Baker, W.L. 1990. Climatic and hydrologic effects on the regeneration of *Populus angustifoli*a James along the Animas River, Colorado. Journal of Biogeography. 17: 59–73.

Baker, W.L. 1992. Structure, disturbance, and change in the bristlecone pine forests of Colorado, U.S.A. Arctic and Alpine Research. 24: 17–26.

Barbour, M.G. 1988. Californian upland forests and woodlands. In: Barbour, M.G.; Billings, W.D., eds. North American terrestrial vegetation. New York, NY: Cambridge University Press: 131–164.

Barbour, M.G.; Berg, N.H.; Kittel, G.F.; Kunz, M.E. 1991. Snowpack and the distribution of a major vegetation ecotone in the Sierra Nevada of California. Journal of Biogeography. 18: 141–149.

Bare, B.B. 1982. The economics of true fir management. In: Oliver, C.D.; Kenady, R.M., eds. Proceedings of the biology and management of true fir in the Pacific Northwest symposium; 1981 February 24-26; Seattle-Tacoma. Contrib. 45. Seattle, WA: University of Washington, College of Forest Resources: 9–14.

Barker, P.A. 1977. Canyon maple—A colorful mountaineer. American Forests. 83: 22–25.

Barney, M.A.; Frischknecht, N.C. 1974. Vegetation changes following fire in the pinyon-juniper type of west-central Utah. Journal of Range Management. 27: 91–96.

Barrett, S.W.; Arno, S.F. 1991. Classifying fire regimes and defining their topographic controls in the Selway-Bitterroot Wilderness. In: Andrews, P.L.; Potts, D.F., eds. Proceedings of the 11th conference on fire and forest meteorology. Missoula, MT: Society of American Foresters: 299–307.

Barrett, S.W.; Arno, S.F.; Key, C.H. 1991. Fire regimes of western larch-lodgepole pine forests in Glacier National Park, Montana. Canadian Journal of Forest Research. 21: 1711–1720.

Barro, S.C.; Wohlgemuth, P.M.; Campbell, A.G. 1989. Post-fire interactions between riparian vegetation and channel morphology and the implications for stream channel rehabilitation choices. In: Abell, D.L., tech. coord. Proceedings of the California riparian systems conference: Protection, management, and restoration for the 1990's. Gen. Tech. Rep. PSW-GTR-110. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station: 51–53.

Bartlein, P.J.; Whitlock, C.; Shafer, S.L. 1997. Future climate in the Yellowstone National Park region and its potential impact on vegetation. Conservation Biology. 11: 782–792.

Bartos, D. 1998. Aspen, fire and wildlife. Spokane, WA: The Wildlife Society, Northwest Section, Oregon and Washington Chapters: 44–48.

Bartos, D.L. 1978. Modeling plant succession in aspen ecosystems. In: Hyder, D.N., ed. Proceedings of the first international rangeland congress. Denver, CO: Society for Range Management: 208–211.

Battles, J.J.; Robards, T.; Das, A.; [et al.]. 2008. Climate change impacts on forest growth and tree mortality: A data-driven modeling study in the mixed conifer forest of the Sierra Nevada, California. Climatic Change. 87 (Suppl. 1): S193–S213.

Baumeister, D.; Callaway, R.M. 2006. Facilitation by *Pinus flexilis* during succession: A hierarchy of mechanisms benefits other plant species. Ecology. 87: 1816–1830.

Beasley, R.S.; Klemmedson, J.O. 1973. Recognizing site adversity and drought-sensitive trees in stands of bristlecone pine (*Pinus longaeva*). Economic Botany 27: 141–146.

Beaufait, W.R.; Hardy, C.E.; Fischer, W.C. 1977. Broadcast burning in larch-fir clearcuts: the Miller Creek-Newman Ridge study. Res. Pap. INT-RP-175. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 53 p.

Bega, Robert V., tech. coord. 1978. Diseases of Pacific Coast conifers. Agric. Handbk 521. Washington, DC: U.S. Department of Agriculture. 206 p.

Bell, D.M.; Bradford, J.B.; Lauenroth, W.K. 2014. Early indicators of change: Divergent climate envelopes between tree life stages imply range shifts in the western United States. Global Ecology and Biogeography. 23: 168–180.

Bentz, B.J.; Duncan, J.P.; Powell, J.A. 2016. Elevational shifts in thermal suitability for mountain pine beetle population growth in a changing climate. Forestry. 89: 271–283.

Bentz, B.J.; Jönsson, A.M. 2015. Modeling bark beetle responses to climate change. In: Vega, F.; Hofstetter, R., eds. Bark beetles, biology and ecology of native and invasive species. San Diego, CA: Academic Press: 533–554.

Bentz, B.J.; Régnière, J.; Fettig, C.J.; [et al.]. 2010. Climate change and bark beetles of the western United States and Canada: direct and indirect effects. BioScience. 60: 602–613.

Beschta, R.L.; Ripple, W.J. 2005. Rapid assessment of riparian cottonwood recruitment: Middle Fork John Day River, northeastern Oregon. Ecological Restoration. 23: 150–156.

Bigler, C.; Kulakowski, D.; Veblen, T.T. 2005. Multiple disturbance interactions and drought influence fire severity in Rocky Mountain subalpine forests. Ecology. 86: 3018–3029.

Blodgett, J.T.; Sullivan, K.F. 2004. First report of white pine blister rust on Rocky Mountain bristlecone pine. Plant Disease. 88: 311.

Bollenbacher, B. 2012. Characteristics of primary tree species in the Northern Region. Missoula, MT: U.S. Department of Agriculture, Forest Service. Northern Region. Various pagings.

Bonan, G.B. 2008. Forests and climate change: forcings, feedbacks, and the climate benefits of forests. Science. 320: 1444–1449.

Borman, M.M.; Larson, L.L. 2002. Cottonwood: Establishment, survival, and stand characteristics. Corvallis, OR: Oregon State University, Extension Service. 12 p.

Bradley, A.F.; Noste, N.V.; Fischer, W.C. 1992. Fire ecology of forests and woodlands in Utah. Gen. Tech. Rep. INT-287. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 128 p.

Breshears, D.D.; Myers, O.B.; Meyer, C.W.; [et al.]. 2009. Tree die-off in response to global change-type drought: mortality insights from a decade of plant water potential measurements. Frontiers in Ecology and the Environment. 7: 185–189.

Broadhurst, L.M.; Lowe, A.; Coates, D.J.; [et al.]. 2008. Seed supply for broadscale restoration: maximizing evolutionary potential. Evolutionary Applications. 1: 587–597.

Brotherson, J.D. 1990. Mineral-nutrient relationships of the mountain mahoganies (*Cercocarpus*). In: Johnson, K.L., ed. Proceedings, 5th Utah shrub ecology workshop: The genus *Cercocarpus*; 1988 July 13-14; Logan, UT. Logan, UT: Utah State University, College of Natural Resources: 43–60.

Brotherson, J.D.; Winkel, V. 1986. Habitat relationships of saltcedar (*Tamarix ramosissima*). The Great Basin Naturalist. 46: 535–541.

Brown, J.K. 1973. Fire cycles and community dynamics of lodgepole pine forests. In: Management of lodgepole pine ecosystems. Pullman, WA: Washington State University Press: 23–55.

Brown, J.K. 1996. Fire effects on aspen and cottonwood. In: Aspen and cottonwood in the Blue Mountains workshop. 1996, April 2–4. La Grande, OR: Blue Mountains Natural Resources Institute. 11 p.

Brown, P.M.; Schoettle, A.W. 2008. Fire and stand history in two limber pine (*Pinus flexilis*) and Rocky Mountain bristlecone pine (*Pinus aristata*) stands in Colorado. International Journal of Wildland Fire. 17: 339–347.

Brunelle, A.; Whitlock, C.; Bartlein, P.; Kipfmueller, K. 2005. Holocene fire and vegetation along environmental gradients in the Northern Rocky Mountains. Quaternary Science Reviews. 24: 2281–2300.

Bugmann, H. 2001. A review of forest gap models. Climatic Change. 51: 259–305.

Bugmann, H.; Cramer, W. 1998. Improving the behavior of forest gap models along drought gradients. Forest Ecology and Management. 103: 247–263.

Bunderson, E.D.; Weber, D.J.; Davis, J.N. 1985. Soil mineral composition and nutrient uptake in *Juniperus osteosperma* in 17 Utah sites. Soil Science. 139: 139–148.

Burns, R.M.; Honkala, B.H. 1990. Silvics of North America: Volume 1, conifers. Washington, DC: U.S. Department of Agriculture, Forest Service. 675 p.

Burwell, T. 1998. Successional patterns of the lower montane treeline, eastern California. Madroño. 45: 12–16.

Busch, D.E. 1995. Effects of fire on southwestern riparian plant community structure. The Southwestern Naturalist. 40: 259–267.

Busch, D.E.; Smith, S.D. 1993. Effects of fire on water salinity relations of riparian woody taxa. Oecologia. 94: 186–194.

Butler, D.R. 1986. Conifer invasion of subalpine meadows, central Lemhi Mountains, Idaho. Northwest Science. 60: 166–173.

Callaway, R.; Sala, A.; Carey, E.; Keane, R.E. 1998. Replacement of whitebark pine by subalpine fir: The consequences for stand carbon, water, and nitrogen cycles. Contract Completion Rep. Res. Joint Venture Agree. INT-95086-RJVA. Missoula, MT: U.S. Department of Agriculture, Forest Service, Fire Sciences Laboratory. Various paging.

Campbell, R.B.; Bartos, D. 2001. Aspen ecosystems—Objectives for sustaining biodiversity. In: Sustaining aspen ecosystems in western landscapes, proceedings. RMRS-P-18. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 299–306.

Carroll, A.L.; Taylor, S.W.; Régnière, J.; Safranyik, L. 2003. Effects of climate change on range expansion by the mountain pine beetle in British Columbia. In: Shore, T.L.; Brooks, J.E.; Stone, J.E., eds. Mountain pine beetle symposium: Challenges and solutions. Victoria, British Columbia, Canada: Natural Resources Canada, Canadian Forest Service: 223–231.

Cayan, D.R.; Dettinger, M.D.; Kammerdiener, S.A.; [et al.]. 2001. Changes in the onset of spring in the western United States. Bulletin of the American Meteorological Society. 82: 399–415.

Chapman, T.B.; Veblen, T.T.; Schoennagel, T. 2012. Spatiotemporal patterns of mountain pine beetle activity in the southern Rocky Mountains. Ecology. 93: 2175–2185.

Chhin, S.; Hogg, E.T.; Lieffers, V J.; Huang, S. 2008. Potential effects of climate change on the growth of lodgepole pine across diameter size classes and ecological regions. Forest Ecology and Management. 256: 1692–1703.

Chmura, D.J.; Anderson, P.D.; Howe, G.T.; [et al.]. 2011. Forest responses to climate change in the northwestern United States: Ecophysiological foundations for adaptive management. Forest Ecology and Management. 261: 1121–1142.

Cissel, J.H.; Swanson, F.J.; McKee, W.A.; Burditt, A.L. 1994. Using the past to plan the future in the Pacific Northwest. Journal of Forestry. 92: 30–31, 46.

Clark, J.S.; Carpenter, S.R.; Barber, M.; [et al.]. 2001. Ecological forecasts: An emerging imperative. Science. 293: 657–660.

Clary, W.P.; Tiedemann, A.R. 1986. Distribution of biomass within small tree and shrub form *Quercus gambelii* stands. Forest Science. 32: 234–242.

Coop, J.D.; Schoettle, A.W. 2009. Regeneration of Rocky Mountain bristlecone pine (*Pinus aristata*) and limber pine (*Pinus flexilis*) three decades after stand-replacing fires. Forest Ecology and Management. 257: 893–903.

Coops, N.C.; Waring, R.H. 2011. A process-based approach to estimate lodgepole pine (*Pinus contorta* Dougl.) distribution in the Pacific Northwest under climate change. Climatic Change. 105: 313–328.

Cramer, W.; Bondeau, A.; Woodward, F.I.; [et al.]. 2001. Global response of terrestrial ecosystem structure and function to CO_2 and climate change: results from six dynamic global vegetation models. Global Change Biology. 7: 357–373.

Crane, M.F. 1982. Fire ecology of Rocky Mountain Region forest habitat types. Final rep. contract 43-83X9-1-884. Missoula, MT: U.S. Department of Agriculture, Forest Service, Northern Region. 272 p.

Creeden, E.P.; Hicke, J.A.; Buotte, P.C. 2014. Climate, weather, and recent mountain pine beetle outbreaks in the western United States. Forest Ecology and Management. 312: 239–251.

Crimmins, S.M.; Dobrowski, S.Z.; Greenberg, J.A.; [et al.]. 2011. Changes in climatic water balance drive downhill shifts in plant species' optimum elevations. Science. 331: 324–327.

Critchfield, W.B.; Allenbaugh, G.L. 1969. The distribution of Pinaceae in and near northern Nevada. Madroño. 20: 12–25.

Cronquist, A.; Holmgren, N.H.; Holmgren, P.K. 1997. Intermountain flora: Vascular plants of the Intermountain West, U.S.A. volume 3, part A: Subclass Rosidae (except Fabales). New York. NY: New York Botanical Garden. 446 p.

Cushman, S.A.; McKenzie, D.; Peterson, D.L.; [et al.]. 2007. Research agenda for integrated landscape modeling. Gen. Tech. Rep. RMRS-GTR-194. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 50 p.

Czarapata, E.J. 2005. Invasive plants of the upper Midwest: An illustrated guide to their identification and control. Madison, WI: University of Wisconsin Press. 236 p.

Dahms, C.W.; Geils, B.W., tech. eds. 1997. An assessment of forest ecosystem health in the Southwest. Gen. Tech. Rep. RM-GTR-295. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 97 p.

Dale, V. H.; Joyce, L. A.; McNulty, S.; Neilson, R.P.; [et al.]. 2001. Climate change and forest disturbances. BioScience 51: 723–734.

Davis, J.N. 1990. General ecology, wildlife use, and management of the mountain mahoganies in the Intermountain West. In: Johnson, K.L., ed. Proceedings, 5th Utah shrub ecology workshop: The genus *Cercocarpus*; 1988 July 13-14; Logan, UT. Logan, UT: Utah State University, College of Natural Resources: 1-13.

Davis, M.B. 1989. Lags in vegetation response to greenhouse warming. Climatic Change. 15: 75–82.

Davis, M.B.; Shaw, R.G. 2001. Range shifts and adaptive responses to Quaternary climate change. Science. 292: 673–679.

Davis, M.B.; Shaw, R.G.; Etterson, J.R. 2005. Evolutionary responses to changing climates. Ecology. 86: 1704–1714.

DeBell, D.S. 1990. *Populus trichocarpa* Torr. and Gray black cottonwood. In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America: Volume 2 hardwoods. Washington, DC: U.S. Department of Agriculture, Forest Service: 1322–1355.

DeNitto, G. 2013. FHP annual report for the Northern Region. Rep. R1-FHP-2013. Missoula, MT: U.S. Department of Agriculture, Forest Service, Northern Region, Forest Health and Protection. Various paging.

Devine, W.; Aubry, C.; Bower, A.; [et al.]. 2012. Climate change and forest trees in the Pacific Northwest: A vulnerability assessment and recommended actions for National Forests. Olympia, WA: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region. 102 p.

Dick-Peddie, W.A.; Moir, W.H. 1970. Vegetation of the Organ Mountains, New Mexico. Sci. Ser. 4. Fort Collins, CO: Colorado State University, Range Science Department. 28 p.

Dieter, C.D.; McCabe, T.R. 1989. Habitat use by beaver along the Big Sioux River in eastern South Dakota. In: Gresswell, R.E.; Barton, B.A.; Kershner, J.L., eds. Practical approaches to riparian management: An educational workshop. BLM-MIT-PT-89-0001-4351. Billings, MT: U.S. Department of the Interior, Bureau of Land Management, Montana State Office: 135–140.

Diggs, G.M.; Lipscomb, B.L.; O'Kennon, R.J. 1999. Illustrated flora of north-central Texas. Sida Bot. Misc. 16. Fort Worth, TX: Botanical Research Institute of Texas. 1640 p.

Dina, S.J.; Klikoff, L.G. 1973. Carbon dioxide exchange by several streamside and scrub oak community species of Red Butte. The American Midland Naturalist. 89: 70–80.

Disalvo, A.C.; Hart, S.C. 2002. Climatic and stream-flow controls on tree growth in a western montane riparian forest. Environmental Management. 30: 678–691.

Dwyer, Don D.; Pieper, Rex D. 1967. Fire effects on blue grama-pinyon-juniper rangeland in New Mexico. Journal of Range Management. 20: 359–362.

Easterling, D.R.; Meehl, G.A.; Parmesan, C.; [et al.]. 2000. Climate extremes: Observations, modeling, and impacts. Science. 289: 2068–2074.

Egan, D.; Howell, E.A., eds. 2001. The historical ecology handbook. Washington, DC: Island Press. 469 p.

Ehleringer, J.R.; Arnow, L.A.; Arnow, T.; [et al.]. 1992. Red Butte Canyon Research Natural Area: History, flora, geology, climate, and ecology. The Great Basin Naturalist. 52: 95–121.

Eisenberg, C.; Seager, S.T.; Hibbs, D.E. 2013. Wolf, elk, and aspen food web relationships: context and complexity. Forest Ecology and Management. 299: 70–80.

Etterson, J.R.; Shaw, R.G. 2001. Constraint to adaptive evolution in response to global warming. Science. 294: 151–154.

Evans, R.A. 1988. Management of pinyon-juniper woodlands. Gen. Tech. Rep. INT-GTR-249. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 218 p.

Everett, R.L. 1985. Great Basin pinyon and juniper communities and their response to management. In: Symposium on the cultural, physical and biological characteristics of range livestock industry in the Great Basin: Proceedings, 38th annual meeting of the Society for Range Management; 1985 February 11-14; Salt Lake City, UT. Denver, CO: Society for Range Management: 53–62.

Everett, R.L. 1987. Allelopathic effects of pinyon and juniper litter on emergence and growth of herbaceous species. In: Frasier, G.W.; Evans, R.S., comps. Proceedings, symposium on seed and seedbed ecology of rangelands. Washington, DC: U.S. Department of Agriculture, Agricultural Research Service: 62–67.

Everett, R.L.; Koniak, S. 1981. Understory vegetation in fully stocked pinyon-juniper stands. The Great Basin Naturalist. 41: 467–475.

Everett, R.L.; Sharrow, S.H.; Meeuwig, R.O. 1983. Pinyon-juniper woodland understory distribution patterns and species associations. Bulletin of the Torrey Botanical Club. 110: 454–463.

Falk, D.A.; Miller, C.; McKenzie, D.; Black, A.E. 2007. Cross-scale analysis of fire regimes. Ecosystems. 10: 809–823.

Fechner, G.H. 1985. Silvical characteristics of blue spruce. Gen. Tech. Rep. RM-GTR-117. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 117 p.

Fechner, G.H. 1990. *Picea pungens* Engelm. blue spruce. In: Burns, R.M.; Honkala, B.H., tech. coords. Silvics of North America: Volume 1, conifers. Agric. Handbk. 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 238–249.

Feldman, R.; Tomback, D.F.; Koehler, J. 1999. Cost of mutualism: Competition, tree morphology, and pollen production in limber pine clusters. Ecology. 80: 324–329.

Ferguson, D.E.; Johnson, F.D. 1996. Classification of grand fir mosaic habitats. Gen. Tech. Rep. INT-GTR-337. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station.

Ferry, G.W.; Clark, R.G.; Montgomery, R.E.; [et al.]. 1995. Altered fire regimes within fire-adapted ecosystems. In: LaRoe, E.T.; Farris, G.S.; Puckett, C.E. Our living resources: A report to the nation on the distribution, abundance, and health of U.S. plants, animals, and ecosystems. Washington, DC: U.S Department of the Interior, National Biological Service: 222–224.

Fiedler, C.E.; Lloyd, D.A. 1995. Autecology and synecology of western larch. In: Schmidt, W.C.; McDonald K.J., comps. Proceedings, symposium on ecology and management of *Larix* forests: A look ahead. Gen. Tech. Rep. GTR-INT-319. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 118–122.

Fins, L.; Byler, J.; Ferguson, D.; [et al.]. 2002. Return of the giants: restoring white pine ecosystems by breeding and aggressive planting of blister rust-resistant white pines. Journal of Forestry. 100: 20–26.

Fischer, W.C.; Bradley, A.F. 1987. Fire ecology of western Montana forest habitat types. Gen. Tech. Rep. INT-GTR-223. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station.

Fischer, W.C.; Clayton, B.D. 1983. Fire ecology of Montana forest habitat types east of the Continental Divide. Gen. Tech. Rep. INT-GTR-141. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station.

Fitzhugh, E.L.; Moir, W.H.; Ludwig, J.A.; Ronco, F. 1987. Forest habitat types in the Apache, Gila, and part of the Cibola National Forests, Arizona and New Mexico. Gen. Tech. Rep. RM-GTR-145. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station.

Flannigan, M.D.; Krawchuk, M.A.; de Groot, W.J.; [et al.]. 2009. Implications of changing climate for global wildland fire. International Journal of Wildland Fire. 18: 483–507.

Flora of North America Association [FNAA]. 2009. Flora of North America: The flora. http://www.fna.org/FNA [Accessed October 26, 2016].

Floyd, M.L.; Hanna, D.D.; Romme, W.H. 2004. Historical and recent fire regimes in piñon-juniper woodlands on Mesa Verde, Colorado, USA. Forest Ecology and Management. 198: 269–289.

Floyd, M.L.; Romme, W.H.; Hanna, D.D. 2000. Fire history and vegetation pattern in Mesa Verde National Park, Colorado, USA. Ecological Applications. 10: 1666–1680.

Floyd, M.L.; Clifford, M.; Cobb, N.S.; [et al.]. 2009. Relationship of stand characteristics to drought-induced mortality in three Southwestern piñon-juniper woodlands. Ecological Applications. 19: 1223–1230.

Foiles, M.W.; Graham, R.T.; Olson, D.F. 1990. *Abies grandis* (Dougl. ex D. Don) Lindl. grand fir. In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America: Volume 1, conifers. Washington, DC: U.S. Department of Agriculture, Forest Service: 132–155.

Fowells, H.A., comp. 1965. Silvics of forest trees of the United States. Agric. Handbk. 271. Washington, DC: U.S. Department of Agriculture.

Franklin, J.F.; Swanson, F.J.; Harmon, M.E.; [et al.]. 1991. Effects of global climatic change on forests in northwestern North America. The Northwest Environmental Journal. 7: 233–254.

Frey, B.R.; Lieffers, V.J.; Hogg, E.H.; Landhäusser, S.M. 2004. Predicting landscape patterns of aspen dieback: Mechanisms and knowledge gaps. Canadian Journal of Forest Research. 34: 1379–1390.

Fryer, J.L. 2014. *Alnus rhombifolia*. In: Fire Effects Information System. Missoula, MT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Missoula Fire Sciences Laboratory. http://www.fs.fed.us/database/feis/plants/tree/alnrho/all.html [Accessed September 6, 2017].

Geiszler, D.R.; Gara, R.I.; Driver, C.H.; [et al.]. 1980. Fire, fungi, and beetle influences on a lodgepole pine ecosystem of south-central Oregon. Oecologia. 46: 239–243.

Gillette, N.E.; Wood, D.L.; Hines, S.J.; [et al.]. 2014. The once and future forest: Consequences of mountain pine beetle treatment decisions. Forest Science. 60: 527–538.

Gom, L.A.; Rood, S.B. 1999. Fire induces clonal sprouting of riparian cottonwoods. Canadian Journal of Botany. 77: 1604–1616.

Gordon, D.T. 1973. Damage from wind and other causes in mixed white fir-red fir stands adjacent to clearcuttings. USDA Forest Service, Res. Pap. PSW-RP-90. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station.

Gottfried, G.J.; Folliott, P.F.; DeBano, L.F. 1995. Forests and woodlands of the sky islands: Stand characteristics and silvicultural prescriptions. In: DeBano, L.F.; Ffolliott, P.F.; Ortega-Rubio, A.; [et al.], tech. coords. Biodiversity and management of the Madrean Archipelago: The sky islands of southwestern United States and northwestern Mexico: Proceedings; September 19-23, 1994; Tucson, AZ. Gen. Tech. Rep. RM-GTR-264. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 152–164.

Gottfried, G.J.; Severson, K.E. 1993. Distribution and multiresource management of piñon-juniper woodlands in the southwestern United States. In: Aldon, E.F.; Shaw, D.W., tech. coords. Managing piñon-juniper ecosystems for sustainability and social needs: Proceedings; April 26-30, 1993; Santa Fe, NM. Gen. Tech. Rep. RM-GTR-236. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 108–116.

Gower, S.T.; Kloeppel, B.D.; Reich, P.B. 1995. Carbon, nitrogen, and water use by larches and co-occurring evergreen conifers. In: Schmidt, W.C.; McDonald, K.J., eds. Ecology and management of *Larix* forests: A look ahead. October 5–9, 1992, Whitefish, MT. Gen. Tech. Rep. INT-GTR-319. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Region: 110–117.

Graham, R.T. 1990. *Pinus monticola* Dougl. ex D. Don western white pine. In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America: Volume 1, conifers. Washington DC: U.S. Department of Agriculture, Forest Service: 348–353.

Gray, L.K.; Hamann, A. 2013. Tracking suitable habitat for tree populations under climate change in western North America. Climatic Change. 117: 289–303.

Gruell, G.E. 1982. Fire's influence on vegetative succession—wildlife habitat implications and management opportunities. In: Eustace, C.D., comp. Proceedings, Montana Chapter of the Wildlife Society. Billings, MT: The Wildlife Society: 43–50.

Gruell, G.E. 1986. Post-1900 mule deer irruptions in the Intermountain West: Principle cause and influences. Gen. Tech. Rep. INT-GTR-206. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station.

Gruell, George E. 1999. Historical and modern roles of fire in pinyon-juniper. In: Monsen, S.B.; Stevens, R., comps. Proceedings: Ecology and management of pinyon-juniper communities within the Interior West: Sustaining and restoring a diverse ecosystem; September 15-18, 1997; Provo, UT. Proc. RMRS-P-9. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 24–28.

Gruell, G.; Bunting, S.; Neuenschwander, L. 1985. Influence of fire on curlleaf mountain-mahogany in the Intermountain West. In: Lotan, J.E.; Brown, J.K., comps. Fire's effects on wildlife habitat, symposium proceedings; 1984 March 21; Missoula, MT. Gen. Tech. Rep. INT-GTR-186. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 58–72.

Gustafson, E. 2013. When relationships estimated in the past cannot be used to predict the future: Using mechanistic models to predict landscape ecological dynamics in a changing world. Landscape Ecology. 28: 1429–1437.

Gworek, J.R.; Vander Wall, S.B.; Brussard, P.F. 2007. Changes in biotic interactions and climate determine recruitment of Jeffrey pine along an elevation gradient. Forest Ecology and Management. 239: 57–68.

Habeck, J.R. 1990. Old-growth ponderosa pine-western larch forests in western Montana: Ecology and management. The Northwest Environmental Journal. 6: 271–292.

Haeussler, S.; Coates, D. 1986. Autecological characteristics of selected species that compete with conifers in British Columbia: A literature review. Land Manage. Rep. 33. Victoria, BC, Canada: Ministry of Forests, Information Services Branch.

Hagle, S.K.; Gibson, K.; Tunnock, S. 2003. Field guide to diseases and insect pests of northern and central Rocky Mountain conifers. Rep. R1-03-08. Missoula, MT: U.S. Department of Agriculture, Forest Service, State and Private Forestry, Northern Region and Intermountain Region.

Hall, F.C. 1973. Plant communities of the Blue Mountains in eastern Oregon and southeastern Washington. R6 Area Guide 3-1. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region.

Hall, J.B.; Hansen, P.L. 1997. A preliminary riparian habitat type classification system for the Bureau of Land Management districts in southern and eastern Idaho. Tech. Bull. 97-11. Boise, ID: U.S. Department of the Interior, Bureau of Land Management.

Hamann, A.; Wang, T. 2006. Potential effects of climate change on ecosystem and tree species distribution in British Columbia. Ecology. 87: 2733–2786.

Hamrick, J.L. 2004. Response of forest trees to global environmental changes. Forest Ecology and Management. 197: 323–335.

Hamrick, J.L.; Godt, M.J.; Sherman-Broyles, S.L. 1992. Factors influencing levels of genetic diversity in woody plant species. New Forests 6: 95–124.

Hann, W.J.; Jones, J.L.; Karl, M.G.; [et al.]. 1997. Landscape dynamics of the basin. In: Quigley, T.M.; Arbelbide, S.J., eds. An assessment of ecosystem components in the Interior Columbia Basin and portions of the Klamath and Great Basins: Volume II. Gen. Tech. Rep. PNW-GTR-405. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 338–1055.

Hanninen, H. 1995. Effects of climatic change on trees from cool and temperate regions: An ecophysiological approach to modelling of bud burst phenology. Canadian Journal of Botany. 73: 183–199.

Hansen, A.J.; Neilson, R.P.; Dale, V.H.; [et al.]. 2001. Global change in forests: Responses of species, communities, and biomes. BioScience. 51: 765–779.

Hansen, A.J.; Phillips, L.B. 2015. Which tree species and biome types are most vulnerable to climate change in the US Northern Rocky Mountains? Forest Ecology and Management. 338: 68–83.

Hansen, H.P. 1949. Post-glacial forests in west central Alberta, Canada. Torrey Botanical Club Bulletin. 76: 278–289.

Hansen, P.L.; Hoffman, G.R. 1988. The vegetation of the Grand River/Cedar River, Sioux, and Ashland Districts of the Custer National Forest: A habitat type classification. Gen. Tech. Rep. RM-GTR-157. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station.

Hansen, P.L.; Pfister, R.D.; Boggs, K.; Cook, B.J.; Joy, J.; Hinckley, D.K. 1995. Classification and management of Montana's riparian and wetland sites. Misc. Pub. 54. Missoula, MT: University of Montana, School of Forestry, Montana Forest and Conservation Experiment Station.

Hardy, C.C.; Keane, R.E.; Stewart, C.A. 2000. Ecosystem-based management in the lodgepole pine zone. In: Smith, H.Y., ed. The Bitterroot ecosystem management research project: What we have learned. Missoula, MT, May 18–20, 1999. Proceedings RMRS-P-17. Fort Collins CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 31–35.

Harper, K. T.; Sanderson, S. C.; McArthur, E. D. 1992. Riparian ecology in Zion National Park, Utah. In: Clary, W.P.; McArthur, E.D.; Bedunah, D.; Wambolt, C.L., comps. Proceedings, symposium on ecology and management of riparian shrub communities; May 29-31, 1991; Sun Valley, ID. Gen. Tech. Rep. INT-GTR-289. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 32–42.

Harper, K.T.; Wagstaff, F.J.; Kunzler, L.M. 1985. Biology management of the Gambel oak vegetative type: A literature review. Gen. Tech. Rep. INT-GTR-179. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station.

Harrington, C.A.; Gould, P.J.; St. Clair, J.B. 2010. Modeling the effects of winter environment on dormancy release of Douglas-fir. Forest Ecology and Management. 259: 798–808.

Harrington, H.D. 1964. Manual of the plants of Colorado. 2nd ed. Chicago, IL: The Swallow Press.

Harrington, M.G. 2012. Duff mound consumption and cambium injury for centuries-old western larch from prescribed burning in western Montana. International Journal of Wildland Fire. 22: 359–367.

Harrod, R.J.; McRae, B.H.; Hartl, W.E. 1999. Historical stand reconstruction in ponderosa pine forests to guide silvicultural prescriptions. Forest Ecology and Management. 114: 433–446.

Hart, S.J.; Veblen, T.T.; Eisenhart, K.S.; [et al.]. 2013. Drought induces spruce beetle (*Dendroctonus rufipennis*) outbreaks across northwestern Colorado. Ecology. 95: 930–939.

Harvey, A.E.; Byler, J.W.; McDonald, G.I.; [et al.]. 2008. Death of an ecosystem: perspectives on western white pine ecosystems of North America at the end of the twentieth century. Gen. Tech. Rep. RMRS-GTR-208. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Hawksworth, F.G.; Bailey, D.K. 1980. Bristlecone pine. In: Eyre, F. H., ed. Forest cover types of the United States and Canada. Washington, DC: Society of American Foresters: 89–90.

Hawksworth, F.G.; Wiens, D. 1972. Biology and classification of dwarf mistletoes (*Arceuthobium*). Agric. Hndbk. 401. Washington, DC: U.S. Department of Agriculture.

He, H.; Keane, R.E.; Iverson, L. 2008. Forest landscape models, a tool for understanding the effect of the large-scale and long-term landscape processes. Forest Ecology and Management. 254: 371–374.

Heidmann, L.J. 1983. Seed production in southwestern ponderosa pine on a sedimentary soil. Res. Note. RM-RN-434. Flagstaff, AZ: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 2 p.

Hepting, George H. 1971. Diseases of forest and shade trees of the United States. Agric. Handbk. 386. Washington, DC: U.S. Department of Agriculture.

Hermann, R.K.; Lavender, D.P. 1990. *Pseudotsuga menziesii* (Mirb.) Franco Douglas-fir. In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America: Volume 1, conifers. Washington, DC: U.S. Department of Agriculture, Forest Service: 1119–1133.

Hess, K.; Alexander, R.R. 1986. Forest vegetation of the Arapaho and Roosevelt National Forests in central Colorado: A habitat type classification. Res. Pap. RM-RP-266. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station.

Hessburg, P.F.; Smith, B.G.; Salter, R.B. 1999. Detecting change in forest spatial patterns from reference conditions. Ecological Applications. 9: 1232–1252.

Hiebert, R.D. 1977. The population biology of bristlecone pine (*Pinus longaeva*) in the eastern Great Basin. Doctoral dissertation. Lawrence, KS: University of Kansas.

Hickman, J.C., ed. 1993. The Jepson manual: Higher plants of California. Berkeley, CA: University of California Press.

Hitchcock, C.L.; Cronquist, A. 1973. Flora of the Pacific Northwest. Seattle, WA: University of Washington Press.

Hogg, E.H.; Hurdle, P.A. 1995. The aspen parkland in western Canada: A dry-climate analogue for the future boreal forest? Water Air and Soil Pollution. 82: 391–400.

Hohlt, J.C.; Racher, B.J.; Bryan, J.B.; [et al.]. 2002. Saltcedar response to prescribed burning in New Mexico. In: Wilde, G.R.; Smith, L.M., eds. Research highlights—2002: Range, wildlife, and fisheries management. Vol. 33. Lubbock, TX: Texas Tech University, College of Agricultural Sciences and Natural Resources: 25.

Holland, Robert F. 1986. Preliminary descriptions of the terrestrial natural communities of California. Sacramento, CA: California Department of Fish and Game.

Holland, Carol J. 1990. Pinyon-juniper management in Region 3. In: Loftis, D.L., ed. Silvicultural challenges and opportunities in the 1990's: Proceedings of the national silvicultural workshop; July 10-13, 1989; Petersburg, AK. Washington, DC: U.S. Department of Agriculture, Forest Service, Timber Management: 206–216.

Hood, S.; Bentz, B.J. 2007. Predicting post-fire Douglas-fir beetle attacks and tree mortality in the northern Rocky Mountains. Canadian Journal of Forestry. 37: 1058–1069.

Hood, S.; Bentz, B.; Gibson, K.; [et al.]. 2007. Assessing post-fire Douglas-fir mortality and Douglas-fir beetle attacks in the northern Rocky Mountains. Gen. Tech. Rep. RMRS-GTR-199. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Hopkins, T.A.; Larson, J.; Belote, R.T. 2013. Contrasting effects of wildfire and ecological restoration in old-growth western larch forests. Forest Science. 60: 1005–1013.

Horton, J.L.; Kolb, T.E.; Hart, S.C. 2001. Physiological response to groundwater depth varies among species and with river flow regulation. Ecological Applications. 11: 1046–1059.

Huntley, B. 1991. How plants respond to climate change: Migration rates, individualism and the consequences for plant communities. Annals of Botany. 67: 15–22.

Ibáñez, I.; Clark, J.S.; LaDeau, S.; Hille Ris Lambers, J. 2007. Exploiting temporal variability to understand tree recruitment response to climate change. Ecological Monographs. 77: 163–177.

Intergovernmental Panel on Climate Change [IPCC]. 2007. Climate change 2007: The physical science basis. Climate change 2007 working group I contribution to the fourth assessment report of the IPCC. Cambridge, United Kingdom and New York, NY: Cambridge University Press.

Ireland, K.B.; Moore, M.M.; Fulé, P.Z.; [et al.]. 2014. Slow lifelong growth predisposes *Populus tremuloides* trees to mortality. Oecologia. 175: 847-859.

Iverson, L.R.; McKenzie, D. 2013. Tree-species range shifts in a changing climate: Detecting, modeling, assisting. Landscape Ecology. 28: 879–889.

Iverson, L.R.; Prasad, A.M. 2002. Potential redistribution of tree species habitat under five climate change scenarios in the eastern US. Forest Ecology and Management. 155: 205–222.

Jackson, M.; Gannon, A.; Kearns, H.; Kendall, K.C. 2010. Current status of limber pine in Montana. Num. Rep. 10-06. Missoula, MT: U.S. Department of Agriculture, Forest Service, Northern Region, Forest Health and Protection.

Jain, T.B.; Graham, R.T. 2005. Restoring dry and moist forests of the inland northwestern U.S. In: Stranhauf, J.A.; Madsen, P., eds. Restoration of boreal and temperate forests. New York, NY: CRC Press: 463–480.

Jameson, D. A. 1970. Degradation and accumulation of inhibitory substances from *Juniperus osteosperma* (Torr.) Little. Plant and Soil 33: 213–224.

Jameson, D.A.; Williams, J.A.; Wilton, E.W. 1962. Vegetation and soils of Fishtail Mesa, Arizona. Ecology 43: 403–410.

Jenkins, M.J.; Hebertson, E.; Page, W.; Jorgensen, A. 2008. Bark beetles, fuels, fires and implications for forest management in the Intermountain West. Forest Ecology and Management. 254: 16–34.

Jester, N.; Rogers, K.; Dennis, F.C. 2012. Gambel oak management. Fact Sheet 6.311. Fort Collins, CO: Colorado State University, Extension.

Johnstone, J.F.; Chapin, F.S. 2003. Non-equilibrium succession dynamics indicate continued northern migration of lodgepole pine. Global Change Biology. 9: 1401–1409.

Jones, J.R. 1974. Silviculture of southwestern mixed conifers and aspen: The status of our knowledge. Res. Pap. RM-RP-122. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station.

Joyce, L.A. 1995. Productivity of America's forests and climate change. Gen. Tech. Rep. RMRS-GTR-271. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Joyce, L.A.; Birdsey, R A. 2000. The impact of climate change on America's forests: A technical document supporting the 2000 USDA Forest Service RPA assessment. Gen. Tech. Rep. RMRS-GTR-59. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Joyce, L.A.; Running, S.W.; Breshears, D.D.; [et al.]. 2014. Forests. In: Melillo, J.M.; Richmond, T.C.; Yohe, G.W., eds. Climate change impacts in the United States: third National Climate Assessment. Washington, DC: U.S. Global Change Research Program: 175–194.

Jump, A.S.; Peñuelas, J. 2005. Running to stand still: Adaptation and the response of plants to rapid climate change. Ecology Letters. 8: 1010–1020.

Kay, C.E. 2003. Long-term vegetation change on Utah's Fishlake National Forest: A study in repeat photography. Logan, UT: Utah State University, Extension.

Keane, R.E. 2001. Successional dynamics: Modeling an anthropogenic threat. In: Tomback, D.; Arno, S.; Keane, R., eds. Whitebark pine communities: Ecology and restoration. Washington, DC: Island Press: 159–192.

Keane, R.E.; Austin, M.; Dalman, R.; [et al.]. 2001. Tree mortality in gap models: Application to climate change. Climatic Change. 51: 509–540.

Keane, R.E.; Cary, G.; Davies, I.D.; [et al.]. 2004. A classification of landscape fire succession models: spatially explicit models of fire and vegetation dynamic. Ecological Modelling. 256: 3–27.

Keane, R.E.; Finney, M.A. 2003. The simulation of landscape fire, climate, and ecosystem dynamics. In: Veblen, T.T.; Baker, W.L.; Montenegro, G.; Swetnam, T.W., eds. Fire and global change in temperate ecosystems of the western Americas. New York, NY: Springer-Verlag: 32–68.

Keane, R.E.; Hessburg, P.F.; Landres, P.B.; Swanson, F.J. 2009. A review of the use of historical range and variation (HRV) in landscape management. Forest Ecology and Management. 258: 1025–1037.

Keane, R.E.; Loehman, R.; Clark, J.; [et al.]. 2015a. Exploring interactions among multiple disturbance agents in forest landscapes: Simulating effects of fire, beetles, and disease under climate change. In: Perera, A.H.; Remmel, T.K.; Buse, L.J., eds. Modeling and mapping forest landscape patterns. New York: Springer: 201–231.

Keane, R.E.; McKenzie, D.; Falk, D.A.; [et al.]. 2015b. Representing climate, disturbance, and vegetation interactions in landscape models. Ecological Modelling. 309–310: 33–47.

Keane, R.E.; Morgan, P.; Menakis, J.P. 1994. Landscape assessment of the decline of whitebark pine (*Pinus albicaulis*) in the Bob Marshall Wilderness Complex, Montana, USA. Northwest Science. 68: 213–229.

Keane, R.E.; Parsons, R. 2010. Restoring whitebark pine forests of the northern Rocky Mountains, USA. Ecological Restoration. 28: 56–70.

Keane, R.E.; Ryan, K.C.; Running, S.W. 1996. Simulating effects of fire on northern Rocky Mountain landscapes with the ecological process model Fire-BGC. Tree Physiology. 16: 319–331.

Keane, R.E.; Tomback, D.F.; Aubry, C.A.; [et al.]. 2012. A range-wide restoration strategy for whitebark pine forests. Gen. Tech. Rep. RMRS-GTR-279. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Keane, R.E.; Veblen, T.; Ryan, K.C.; [et al.]. 2002. The cascading effects of fire exclusion in the Rocky Mountains. In: Baron, J., ed. Rocky Mountain futures: An ecological perspective. Washington, DC: Island Press: 133–153.

Kemp, K. 2015. Resilience and regeneration after wildfire in dry mixed-conifer forests of the U.S. northern Rockies http://figshare.com/articles/Kemp_NRFSNPresentation_PDF_pdf/1333592 [Accessed October 25, 2016].

Klopfenstein, N.B.; Kim, M.; Hanna, J.W.; [et al.]. 2009. Approaches to predicting potential impacts of climate change on forest disease: An example with Armillaria root disease. Res. Pap. RMRS-RP-76. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Knapp, A.K.; Smith, W.K. 1982. Factors influencing understory seedling establishment of Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*) in southeast Wyoming. Canadian Journal of Botany. 60: 2753–2761.

Knutti, R.; Sedlacek, J. 2013. Robustness and uncertainties in the new CMIP5 climate model projections. Nature Climate Change. 3: 369–373.

Kolb, T. E.; Stone, J. E. 2000. Differences in leaf gas exchange and water relations among species and tree sizes in an Arizona pine-oak forest. Tree Physiology. 20: 1–12.

Laacke, R.J. 1990. *Abies concolor* (Gord. and Glend.) Lindl. ex Hildebr. white fir. In: Burns, R.M.; Honkala, B.H., tech. coords. Silvics of North America: Volume 1, conifers. Agric. Handbk. 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 36–46.

Laacke, R.J.; Tappeiner, J.C. 1996. Red fir ecology and management. In: Erman, D.C., ed. Sierra Nevada Ecosystem Project: Final report to Congress, Volume III. Davis, CA: University of California, Wildland Resources Center Report: chapter 10.

Lacey, J.; Mosley, J. 2002. 250 plants for range contests in Montana. MONTGUIDE MT198402 AG 6/2002. Range E-2 (Misc.). Bozeman, MT: Montana State University, Extension Service.

LaDeau, S.L.; Clark, J.S. 2001. Rising CO_2 levels and the fecundity of forest trees. Science. 292: 95–98.

Lambrecht, S.C.; Loik, M.E.; Inouye, D.W.; Harte, J. 2007. Reproductive and physiological responses to simulated climate warming for four subalpine species. New Phytologist. 173: 121–134.

Landhäusser, S. M.; Deshaies, D.; Lieffers, V.J. 2010. Disturbance facilitates rapid range expansion of aspen into higher elevations of the Rocky Mountains under a warming climate. Journal of Biogeography 37:68–76.

Landres, P.B.; Morgan, P.; Swanson, F.J. 1999. Overview and use of natural variability concepts in managing ecological systems. Ecological Applications. 9: 1179–1188.

Langor, D.W. 2007. Status of limber pine (*Pinus flexilis*) in Alberta. In: Peters, S.; Gutsell, R.; Sharp, N.; Matthais, L., eds. Edmonton, Alberta, Canada: Alberta Sustainable Resource Development and Alberta Conservation Association.

Lankau, R.A.; Zhu, K.; Ordonez, A. 2015. Mycorrhizal strategies of tree species correlate with trailing range edge responses to current and past climate change. Ecology. 96: 1451–1458.

Lanner, R.M. 1980. Avian seed dispersal as a factor in the ecology and evolution of limber and whitebark pines. In: Dancik, B.P.; Higginbotham, K.O., eds. Sixth North American Forest Biology Workshop. Edmonton, Alberta, Canada: University of Alberta: 15–47.

Lanner, R.M. 1983. Trees of the Great Basin: A natural history. Reno, NV: University of Nevada Press. 215 p.

Lanner, R.M. 1999. Conifers of California. Los Olivos, CA: Cachuma Press.

Lanner, R.M.; Vander Wall, S.B. 1980. Dispersal of limber pine seed by Clark‘s nutcracker. Journal of Forestry. 78: 637–639.

Larsen, D.R.; Johnson, P.S. 1998. Linking the ecology of natural oak regeneration to silviculture. Forest Ecology and Management. 106: 1–7.

Larson, A.J.; Belote, R.T.; Cansler, C.A.; [et al.]. 2013. Latent resilience in ponderosa pine forest: Effects of resumed frequent fire. Ecological Applications. 23: 1243–1249.

Laughlin, D.C.; Bakker, J.D.; Stoddard, M.T.; [et al.]. 2004. Toward reference conditions: Wildfire effects on flora in an old growth ponderosa pine forest. Forest Ecology and Management. 199: 137–152.

Lawler, J.J.; White, D.; Neilson, R.P.; Blaustein, A.R. 2006. Predicting climate-induced range shifts: model differences and model reliability. Global Change Biology. 12: 1568–1584.

Ledig, F.T; Kitzmiller, J.H. 1992. Genetic strategies for reforestation in the face of global climate change. Forest Ecology and Management. 50: 153–169.

Leirfallom, S.; Keane, R.E.; Tomback, D.; Dobrowski, S. 2015 The effects of seed source mortality on whitebark pine (*Pinus albicaulis*) regeneration dynamics after wildfire. Canadian Journal Forest Research. 45: 1597–1606.

Lenoir, J.; Gégout, J.C.; Marquet, P.A.; [et al.]. 2008. A significant upward shift in plant species optimum elevation during the 20th century. Science. 320: 1768–1771.

Liang, Y.; He, H.S.; Wang, W.J.; [et al.]. 2015. The site-scale processes affect species distribution predictions of forest landscape models. Ecological Modelling. 300: 89–101.

Littell, J.S.; McKenzie, D.; Kerns, B.K.; [et al.]. 2011. Managing uncertainty in climate-driven ecological models to inform adaptation to climate change. Ecosphere. 2: 102–111.

Little, R.L.; Peterson, D.L.; Conquest, L.L. 1994. Regeneration of subalpine fir (*Abies lasiocarpa*) following fire: Effects of climate and other factors. Canadian Journal of Forest Research. 24: 934–944.

Loehman, R.A.; Clark, J.A.; Keane, R.E. 2011a. Modeling effects of climate change and fire management on western white pine (*Pinus monticola)* in the Northern Rocky Mountains, USA. Forests. 2: 832–860.

Logan, J.A.; Régnière, J.; Powell, J.A. 2003. Assessing the impacts of global warming on forest pest dynamics. Frontiers in Ecology and the Environment. 1: 130–137.

Lorenz, T.J.; Sullivan, K.A. 2009. Seasonal differences in space use by Clark's nutcrackers in the Cascade Range. The Condor. 11: 326–340.

Lotan, J.E.; Brown, J.K.; Neuenschwander, L.F. 1984. Role of fire in lodgepole pine forests. In: Baumgartner, D.M.; Krebill, R.G.; Arnott, J.T.; Weetman, G.F., eds. Lodgepole pine: The species and its management. May 8-10, 1984; Spokane, WA. Pullman, WA: Washington State University, Cooperative Extension: 133–152.

Lotan, J.E.; Critchfield, W.B. 1990. *Pinus contorta* Dougl. ex. Loud. lodgepole pine. In: Burns, R.M.; Honkala, B.H., tech. eds. Silvics of North America: Volume 1, conifers. Agric. Handbk. 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 648–666.

Lotan, J.E.; Perry, D.A. 1983. Ecology and regeneration of lodgepole pine. Agric. Handbk. 606. Washington, DC: U.S. Department of Agriculture, Forest Service.

Luckman, B.H.; Jozsa, L.A.; Murphy, P.J. 1984. Living seven-hundred-year-old *Picea engelmannii* and *Pinus albicaulis* in the Canadian Rockies. Arctic and Alpine Research. 16: 419–422.

MacFarlane, D.; Meyer, S.P. 2005. Characteristics and distribution of potential ash tree hosts for emerald ash borer. Forest Ecology and Management. 213: 15–24.

Malcolm, J.R.; Markham, A.; Neilson, R.P.; Garaci, M. 2002. Estimated migration rates under scenarios of global climate change. Journal of Biogeography. 28: 835–849.

Manning, M.E.; Padgett, W.G. 1989. Preliminary riparian community type classification for Nevada. Unpub. rep. Missoula, MT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory.

Marcoux, H.M.; Daniels, L.D.; Gergel, S.E.; [et al.]. 2015. Differentiating mixed- and high-severity fire regimes in mixed-conifer forests of the Canadian Cordillera. Forest Ecology and Management. 341: 45–58.

Martin, R.E.; Johnson, A.H. 1979. Fire management of Lava Beds National Monument. In: Linn, Robert M., ed. Proceedings, 1st conference on scientific research in the national parks: Volume 2; November 9-12, 1976; New Orleans, LA. NPS Trans. and Proc. Ser. 5. Washington, DC: U.S. Department of the Interior, National Park Service: 1209–1217.

McCulloch, C.Y. 1969. Some effects of wildfire on deer habitat in pinyon-juniper woodland. Journal of Wildlife Management. 33: 778–784.

McDowell, N.; Pockman, W.T.; Allen, C.D.; [et al.]. 2008. Mechanisms of plant survival and mortality during drought: Why do some plants survive while others succumb to drought? New Phytologist. 178: 719–739.

McKelvey, K.S.; Johnston, J.D. 1992. Historical perspectives on forests of the Sierra Nevada and the Transverse Ranges of southern California: Forests condition at the turn of the century. In: Verner, J.; McKelvey, K.S.; Noon, B.R., tech. coords. The California spotted owl: A technical assessment of its current status. Gen. Tech. Rep. PSW-GTR-133. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 225–246.

McKenzie, D., Gedalof, Z.; Peterson, D.L.; Mote, P. 2004. Climatic change, wildfire, and conservation. Conservation Biology. 18: 890–902.

McKenzie, D.; Miller, C.; Falk, D.A., eds. 2011. The landscape ecology of fire. Dordrecht, The Netherlands: Springer Ltd.

McKenzie, D.; Peterson, D.L.; Littell, J.S. 2009. Global warming and stress complexes in forests of western North America. In: Bytnerowicz, A.; Arbaugh, M.J.; Riebau, A.R.; Andersen, C., eds. Wildland fires and air pollution. Dordrecht, The Netherlands: Elsevier: 317–337.

McKenzie, D.; Shankar, U.; Keane, R.E.; [et al.]. 2014. Smoke consequences of new wildfire regimes driven by climate change. Earth's Future. 2: 35–59.

Medina, A.L. 1986. Riparian plant communities of the Fort Bayard watershed in southwestern New Mexico. Southwestern Naturalist. 31: 345–359.

Meeuwig, R.O.; Bassett, R.L. 1983. Pinyon-juniper. In: Burns, R.M., comp. Silvicultural systems for the major forest types of the United States. Agric. Handbk. 445. Washington, DC: U.S. Department of Agriculture, Forest Service: 84–86.

Meeuwig, R.O.; Budy, J.D.; Everett, R.L. 1990. *Pinus monophylla* Torr. and Frem. singleleaf pinyon. In: Burns, R.M.; Honkala, B.H., tech. coords. Silvics of North America: Volume 1, conifers. Agric. Handbk. 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 380–384.

Meyer, M.D. [n.d.]. Natural range of variation of red fir forests in the bioregional assessment area. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Region. http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5434332.pdf [Accessed October 29, 2016].

Merigliano, M.F. 1996. Ecology and management of the South Fork Snake River cottonwood forest. Tech. Bull. 96-9. Boise, ID: U.S. Department of the Interior, Bureau of Land Management, Idaho State Office.

Merriam, G.; Henein, K.; Stuart-Smith, K. 1992. Landscape dynamics models. In: Turner, M.G.; Gardner, R.H., eds. Quantitative methods in landscape ecology: The analysis and interpretation of landscape heterogeneity. New York, NY: Springer-Verlag: 399–416.

Millar, C.I.; Stephenson, N.L.; Stephens, S.L. 2007. Climate change and forests of the future: Managing in the face of uncertainty. Ecological Applications. 17: 2145–2151.

Miller, J.M.; Keen, F.P. 1960. Biology and control of the western pine beetle: A summary of the first fifty years of research. Misc. Pub. 800. Washington, DC: U.S. Department of Agriculture.

Miller, J.D.; Safford, H.D.; Crimmins, M.; Thode, A.E. 2009. Quantitative evidence for increasing forest fire severity in the Sierra Nevada and southern Cascade Mountains, California and Nevada, USA. Ecosystems. 12: 16–32.

Milner, K.S. 1992. Site index and height growth curves for ponderosa pine, western larch, lodgepole pine, and Douglas-fir in western Montana. Western Journal of Applied Forestry. 7: 9–14.

Minore, D. 1979. Comparative autecological characteristics of northwestern tree species: A literature review. Gen. Tech. Rep. PNW-GTR-087. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station.

Mitchell, J.M. 1984. Fire management action plan: Zion National Park, Utah. Record of Decision. Springdale, UT: U.S. Department of the Interior, National Park Service, Zion National Park.

Mitchell, R.; Moir, W. 1976. Vegetation of the Abbott Creek Research Natural Area, Oregon. Northwest Science. 50: 42–58.

Mohatt, K.; Cripps, C.L.; Lavin, M. 2008. Ectomycorrhizal fungi of whitebark pine (a tree in peril) revealed by sporocarps and molecular analysis of mycorrhizae from treeline forests in the Greater Yellowstone Ecosystem. Botany. 86: 14–15.

Moir, W.H. 1982. A fire history of the High Chisos, Big Bend National Park, Texas. The Southwestern Naturalist. 27: 87–98.

Morales, J.M.; Mermoz, M.; Gowda, J.H.; Kitzberger, T. 2015. A stochastic fire spread model for north Patagonia based on fire occurrence maps. Ecological Modelling. 300: 73–80.

Moran, C.J.; Corcoran, M.A. 2012. Do mountain pine beetle outbreaks change the probability of active crown fire in lodgepole pine forests? Comment. Ecology. 93: 939–950.

Morgan, P.; Bunting, S.C.; Keane, R.E.; Arno, S.F. 1994b. Fire ecology of whitebark pine (*Pinus albicaulis*) forests in the Rocky Mountains, USA. In: Schmidt, W.C.; Holtmeier, F.K., eds. Proceedings of the international symposium on subalpine stone pines and their environment: The status of our knowledge; September 5–11, 1992; St. Moritz, Switzerland. Gen. Tech. Rep. INT-GTR-309. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station: 136–142.

Mueggler, W.F. 1976. Ecological role of fire in western woodland and range ecosystems. In: Use of prescribed burning in western woodland and range ecosystems: proceedings of the symposium; March 18-19, 1976; Logan, UT. Logan, UT: Utah State University, Agricultural Experiment Station: 1–9.

Mueggler, W.F. 1985. Vegetation associations. In: DeByle, N.V.; Winokur, R.P. eds., Aspen: Ecology and management in the western United States. Gen. Tech. Rep. GTR-RMRS-119. Fort Collins, CO: U.S. Department of Agriculture, Forest Service: 45–55.

Munz, P.A. 1973. A California flora and supplement. Berkeley, CA: University of California Press.

Murray, M.P.; Bunting, S.C.; Morgan, P. 1995. Subalpine ecosystems: the roles of whitebark pine and fire. In: Greenlee, J.M., ed. Fire effects on rare and endangered species and habitats conference. Coeur d'Alene, ID. Fairfield, WA: International Association of Wildland Fire: 295–299.

Negrón, J.F.; Fettig, C.J. 2014. Mountain pine beetle, a major disturbance agent in US western coniferous forests: A synthesis of the state of knowledge. Forest Science. 60: 409–413.

Neilson, R.P.; Pitelka, L.F.; Solomon, A.M.; [et al.]. 2005. Forecasting regional to global plant migration in response to climate change. BioScience. 55: 749–760.

Nelson, D.L.; Krebill, R.G. 1982. Occurrence and effect of *Chrysomyxa pirolata* cone rust on *Picea pungens* in Utah. The Great Basin Naturalist. 42: 262–272.

Nigh, G. 2014. Mitigating the effects of climate change on lodgepole pine site height in British Columbia, Canada, with a transfer function. Forestry. 87: 377–387.

Nitschke, C.R.; Innes, J.L. 2008. A tree and climate assessment tool for modelling ecosystem response to climate change. Ecological Modelling. 210: 263–277.

Nix, L.E.; Cox, S.K. 1987. Cherrybark oak enrichment plantings appear successful after seven years in South Carolina bottomlands. In: Phillips, D.R., comp. Proceedings of the fourth biennial southern silvicultural research conference; November 4-6, 1986; Atlanta, GA. Gen. Tech. Rep. SE-GTR-42. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station: 129–132.

Noble, D.L. 1990. *Juniperus scopulorum* Sarg. Rocky Mountain juniper. In: Burns, Russell M.; Honkala, Barbara H., tech. coords. Silvics of North America: Volume 1, conifers. Agric. Handbk. 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 116–126.

Notaro, M. 2008. Response of the mean global vegetation distribution to interannual climate variability. Climate Dynamics. 30: 845–854.

Notaro, M.; Vavrus, S.; Liu, Z. 2007. Global vegetation and climate change due to future increases in CO_2 as projected by a fully coupled model with dynamic vegetation. Journal of Climate. 20: 70–88.

Nyland, R.D. 1998. Patterns of lodgepole pine regeneration following the 1988 Yellowstone fires. Forest Ecology and Management. 111: 23–33.

Ohmart, R.D.; Anderson, B.W. 1982. North American desert riparian ecosystems. In: Bender, G.L., ed. Reference handbook on the deserts of North America. Westport, CT: Greenwood Press: 433–479.

Oliver, C.D.; Adams, A.B.; Zasoski, R.J. 1985. Disturbance patterns and forest development in a recently deglaciated valley in the northwestern Cascade Range of Washington, U.S.A. Canadian Journal of Forest Research. 15: 221–232.

Oliver, W.W.; Ryker, R.A. 1990. *Pinus ponderosa* Dougl. ex Laws ponderosa pine. In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America: Volume 1, conifers. Agric. Handbk. 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 348–353.

Oosting, H.J.; Billings, W.D. 1943. The red fir forest of the Sierra Nevada: *Abietum magnificae*. Ecological Monographs. 13: 260–273.

Owens, J.N. 2008. The reproductive biology of western larch. Moscow, ID: University of Idaho, Inland Empire Tree Improvement Cooperative, and the Forest Genetics Council of British Columbia.

Page, W.G.; Jenkins, M.J. 2007. Mountain pine beetle-induced changes to selected lodgepole pine fuel complexes within the intermountain region. Forest Science. 53: 507–518.

Parmesan, C. 2006. Ecological and evolutionary responses to recent climate change. Annual Review of Ecology, Evolution, and Systematics. 37: 637–669.

Parker, A.J. 1984. Mixed forests of red and white fir in Yosemite National Park, California. The American Midland Naturalist. 112: 15–23.

Parker, A.J. 1986. Environmental and historical factors affecting red and white fir regeneration in ecotonal forests. Forest Science. 32: 339–347.

Paysen, T.E.; Ansley, R. J.; Brown, J.K.; [et al.]. 2000. Fire in western shrubland, woodland, and grassland ecosystems. In: Brown, J.K.; Smith, J.K., eds. Wildland fire in ecosystems: Effects of fire on flora. Gen. Tech. Rep. RMRS-GTR-42-volume 2. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 121–159.

Perala, D.A. 1983. Modeling aspen and red pine shoot growth to daily weather variations. Res. Pap. NC-RP-236. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station.

Perala, D.A. 1990. *Populus tremuloides* Michx. quaking aspen. In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America: Volume 2, hardwoods. Washington DC: U.S. Department of Agriculture, Forest Service: 348–353.

Perry, G.L.W.; Millington, J.D.A. 2008. Spatial modelling of succession-disturbance dynamics in forest ecosystems: Concepts and examples. Perspectives in Plant Ecology, Evolution and Systematics. 9: 191–210.

Peterson, D.L.; Vose, J.M.; Patel-Weynand, T. 2014a. Climate change and United States forests. Dordrecht, The Netherlands: Springer.

Peterson, D.W.; Kerns, B.K.; Donson, E.K. 2014b. Climate change effects on vegetation in the Pacific Northwest: A review and synthesis of the scientific literature and simulation model projections. Gen. Tech. Rep. PNW-GTR-900. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

Peterson, D.W.; Peterson, D.L. 2001. Mountain hemlock growth responds to climatic variability at annual and decadal scales. Ecology. 82: 3330–3345.

Peterson, D.W., D.L. Peterson, and G.J. Ettl. 2002. Growth responses of subalpine fir (*Abies lasiocarpa*) to climatic variability in the Pacific Northwest. Canadian Journal of Forest Research 32: 1503–1517.

Pfister, R.D.; Kovalchik, B.L.; Arno, S.F.; Presby, R.C. 1977. Forest habitat types of Montana. Gen. Tech. Rep. INT-GTR-34. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station.

Pieper, R.D. 1977. The southwestern pinyon-juniper ecosystem. In: Aldon, E.F.; Loring, T.J., tech. coords. Ecology, uses, and management of pinyon-juniper woodlands: Proceedings of the workshop; March 24-25, 1977; Albuquerque, NM. Gen. Tech. Rep. RM-GTR-39. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 1–6.

Potter, D.A. 1998. Forested communities of the upper montane in the central and southern Sierra Nevada. Gen. Tech. Rep. PSW-GTR-169. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station.

Potter, K.M.; Hipkins, V.D.; Mahalovich, M.F.; Means, R.E. 2015. Nuclear genetic variation across the range of ponderosa pine (*Pinus ponderosa*): Phylogeographic, taxonomic and conservation implications. Tree Genetics and Genomes. 11: 1–23.

Prentice, I.C.; Bartlein, P.J.; Webb, T. 1991. Vegetation and climate change in eastern North America since the last glacial maximum. Ecology. 72: 2038–2056.

Preston, R.J. 1948. North American trees. Ames, IA: Iowa State College Press.

Purves, D.; Pacala, S. 2008. Predictive models of forest dynamics. Science. 320: 1452–1453.

Racher, B.J.; Mitchell, R.B. 1999. Management of saltcedar in eastern New Mexico and Texas. In: Wester, D.B.; Britton, C.M., eds. Research highlights—1999: Noxious brush and weed control: range, wildlife, and fisheries management. Vol. 30. Lubbock, TX: Texas Tech University, College of Agricultural Sciences and Natural Resources: 14–15.

Rehfeldt, G.E. 1978. Genetic differentiation of Douglas-fir populations from the northern Rocky Mountains. Ecology. 1264–1270.

Rehfeldt, G.E. 1989. Ecological adaptations in Douglas-fir (*Pseudotsuga menziesii* var. *glauca*): A synthesis. Forest Ecology and Management. 28: 203–215.

Rehfeldt, G.E. 1994. Evolutionary genetics, the biological species, and the ecology of the Cedar-Hemlock forests. In: Baumgartner, D.M.; Lotan, J.E.; Tonn, J.R., eds. Interior cedar-hemlock-white pine forests: Ecology and management. March 2-4, 1993, Spokane, WA. Pullman, WA: Washington State University, Cooperative Extension: 91–100.

Restaino, C.M., Peterson, D.L.; Littell, J.S. 2016. Increased water deficit decreases Douglas-fir growth throughout western US forests. Proceedings of the National Academy of Sciences, USA. 113: 9557–9562.

Rice, J.; Bardsley, T.; Gomben, P.; [et al.]. 2017. Assessment of aspen ecosystem vulnerability to climate change for the Uinta-Wasatch-Cache and Ashley National Forests, Utah. Gen. Tech. Rep. RMRS-GTR-366. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 67 p.

Riggs, R.A.; Keane, R.E.; Cimon, N.; [et al.]. 2015. Biomass and fire dynamics in a temperate forest-grassland mosaic: Integrating multi-species herbivory, climate, and fire with the FireBGCv2/GrazeBGC system. Ecological Modelling. 296: 57–78.

Rocca, M.E.; Brown, P.M.; MacDonald, L.H; Carrico, C.M. 2014. Climate change impacts on fire regimes and key ecosystem services in Rocky Mountain forests. Forest Ecology and Management. 327: 290–305.

Rochefort, R.M.; Little, R.L.; Woodward, A.; Peterson, D.L. 1994. Changes in subalpine tree distribution in western North America: A review of climatic and other causal factors. The Holocene. 4: 89–100.

Roe, G.H.; Baker, M.B. 2007. Why is climate sensitivity so unpredictable? Science. 318: 629–632.

Rogers, P. 2002. Using forest health monitoring to assess aspen forest cover change in the southern Rockies ecoregion. Forest Ecology and Management. 155: 223–236.

Rogers, P.C.; Eisenberg, C.; St. Clair, S.B. 2013. Resilience in quaking aspen: Recent advances and future needs. Forest Ecology and Management. 299: 1–5.

Romme, W.H.; Allen, C.D.; Bailey, J.D. 2009. Historical and modern disturbance regimes, stand structures, and landscape dynamics in piñon-juniper vegetation of the western United States. Rangeland Ecology and Management. 62: 203–222.

Romme, W.H.; Despain, D G. 1989. Historical perspective on the Yellowstone fires of 1988. BioScience. 39: 695–699.

Romme, W.H.; Turner, M.G. 1991. Implications of global climate change for biogeographic patterns in the greater Yellowstone ecosystem. Conservation Biology. 5: 373–386.

Rood, S.B.; Braatne, J.H.; Hughes, F.M. 2003. Ecophysiology of riparian cottonwoods: Stream flow dependency, water relations and restoration. Tree Physiology. 23: 1113–1124.

Rundel, P.W.; Parsons, D.J.; Gordon, D.T. 1977. Montane and subalpine vegetation of the Sierra Nevada and Cascade Ranges. In: Barbour, M.G.; Major, J., eds. Terrestrial vegetation of California. New York, NY: John Wiley and Sons: 559–599.

Ryan, K.C.; Reinhardt, E.D. 1988. Predicting postfire mortality of seven western conifers. Canadian Journal of Forest Research. 18: 1291–1297.

Ryan, M.G.; Gower, S.T.; Hubbard, R.M.; [et al.]. 1995. Woody tissue maintenance respiration of four conifers in contrasting climates. Oecologia. 101: 133–140.

Ryker, R.A.; Losensky, J. 1983. Ponderosa pine and Rocky Mountain Douglas-fir. Silvicultural systems for the major forest types of the United States. Agric. Handbk. 445. Washington, DC: U.S. Department of Agriculture: 53–55.

Safranyik, L.; Carroll, A.; Régnière, J.; [et al.]. 2010. Potential for range expansion of mountain pine beetle into the boreal forest of North America. The Canadian Entomologist. 142: 415–442.

Sala, A.; Peters, G.D.; McIntyre, L.R.; Harrington, M.G. 2005. Physiological responses of ponderosa pine in western Montana to thinning, prescribed fire and burning season. Tree Physiology. 25: 339–348.

Salzer, M.W; Larson, E.R.; Bunn, A.G.; Hughes, M.K. 2014. Changing climate response in near-treeline bristlecone pine with elevation and aspect. Environmental Research Letters. 9.11: 114007.

Schmidt, W.C. 1969. Seedbed treatments influence seedling development in western larch forests. Res. Note INT-RN-93. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station.

Schmidt, W.C.; Fellin, D.G. 1973. Western spruce budworm damage affects form and height growth of western larch. Canadian Journal of Forest Research. 3: 17–26.

Schmidt, W.C.; Larson, M. 1989. Silviculture of western inland conifers. In: Burns, R.M., comp. The scientific basis for silvicultural and management decisions in the National Forest System. Gen. Tech. Rep. WO-GTR-55. Washington, DC: U.S. Department of Agriculture, Forest Service: 40–58.

Schmidt, W.C.; Shearer, R.C. 1990. *Larix occidentalis* Nutt. western larch. In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America: Volume 1, conifers. Agric. Handbk. 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 348–353.

Schmidt, W.C.; Shearer, R.C. 1995. *Larix occidentalis*: A pioneer in the North American West. In: Schmidt, W.C.; McDonald, K.J., eds. Proceedings of an international symposium: Ecology and management of *Larix* forests: A look ahead; 1992; October 5-9, 1992; Gen. Tech. Rep. INT-GTR-319 Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 33–37.

Schmidt, W.C.; Shearer, R.C.; Roe, A.L. 1976. Ecology and silviculture of western larch forests. Tech. Bull. 1520. Washington, DC: U.S. Department of Agriculture, Forest Service.

Schneider, S.H.; Semenov, S.; Patwardhan, A.; [et al.]. 2007. Assessing key vulnerabilities and the risk from climate change. In: Parry, M.L.; Canziani, O.F.; Palutikof, P.J.; [et al.], eds. Climate change 2007: Impacts, adaptation and vulnerability. Contribution of working group II to the fourth assessment report of the Intergovernmental Panel on Climate Change. Cambridge, United Kingdom and New York, NY: Cambridge University Press: 779–810.

Schoennagel, T.L.; Veblen, T.T.; Romme, W.H. 2004. The interaction of fire, fuels, and climate across Rocky Mountain landscapes. BioScience. 54: 651–672.

Schoettle, A.W. 2003. Patterns of white pine regeneration after fire and its implications for forest establishment in the presence of white pine blister rust—A research program within the U.S. National Fire Plan. In: Parks Canada whitebark and limber pine workshop: workshop proceedings; February 18-19, 2003; Calgary, Alberta. Ottawa, Ontario, Canada: Parks Canada: 14–15.

Schowalter, T.; Hansen, E.; Molina, R.; Zhang, Y. 1997. Integrating the ecological roles of phytophagous insects, plant pathogens, and mycorrhizae in managed forests. In: Kohm, K.A.; Franklin, J.F., eds. Creating a forestry for the 21st century: the science of ecosystem management. Washington, DC: Island Press: 171–189.

Schumacher, S.; Reineking, B.; Sibold, J.; Bugmann, H. 2006. Modeling the impact of climate and vegetation on fire regimes in mountain landscapes. Landscape Ecology. 21: 539–554.

Schwandt, J.W. 2006. Whitebark pine in peril: A case for restoration. Rep. R1-06-28. Missoula, MT: U.S. Department of Agriculture, Forest Service, Northern Region. 20 p.

Sevanto, S.; McDowell, N.G.; Dickman, L.T.; [et al.]. 2014. How do trees die? A test of the hydraulic failure and carbon starvation hypotheses. Plant, Cell and Environment. 37: 153–161.

Shafroth, P.B.; Stromberg, J.C.; Patten, D.T. 2002. Riparian vegetation response to altered disturbance and stress regimes. Ecological Applications. 12: 107–123.

Shantz, H.L.; Piemeisel, R.L. 1940. Types of vegetation in Escalante Valley, Utah, as indicators of soil conditions. Tech. Bull. 713. Washington, DC: U.S. Department of Agriculture.

Shearer, R.C. 1976. Early establishment of conifers following prescribed broadcast burning in western larch/Douglas-fir forests. In: Proceedings from the Tall Timbers Fire Ecology Conf. 14 and Intermountain Fire Research Council Fire and Land Management Symposium. Tallahassee, FL: Tall Timbers Research Station: 481–500.

Shearer, R.C.; Schmidt, W.C. 1970. Natural regeneration in ponderosa pine forests of western Montana. Res. Pap. INT-RP-86. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station.

Shepperd, W.D.; Bartos, D.L.; Mata, S.A. 2001. Above- and below-ground effects of aspen clonal regeneration and succession to conifers. Canadian Journal of Forest Research. 31: 739–745.

Sher, A.; Quigley, M. 2013. Tamarix: A case study of ecological change in the American West. Oxford, United Kingdom: Oxford University Press.

Shinneman, D.J.; Baker, W.L. 2009. Environmental and climatic variables as potential drivers of post-fire cover of cheatgrass (*Bromus tectorum*) in seeded and unseeded semiarid ecosystems. International Journal of Wildland Fire. 18: 191–202.

Short, H.L.; McCulloch, C.Y. 1977. Managing pinyon-juniper ranges for wildlife. Gen. Tech. Rep. RM-47. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 10 p.

Sieg, C.H. 1997. The role of fire in managing for biological diversity on native rangelands of the Northern Great Plains. In: Uresk, D.W.; Schenbeck, G.L.; O'Rourke, J.T., tech. coords. Conserving biodiversity on native rangelands: Symposium proceedings; August 17, 1995; Fort Robinson State Park, NE. Gen. Tech. Rep. RM-GTR-298. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 31–38.

Skinner, C.N.; Taylor, A.H.; Agee, J.K. 2006. Klamath Mountains bioregion. In: Sugihara, N.G.; van Wagtendonk, J.W.; Shaffer, K.E., eds. Fire in California's ecosystems. Berkeley, CA: University of California Press: 170–194.

Sklar, F.H.; Costanza, R. 1991. The development of dynamic spatial models for landscape ecology: A review and prognosis. In: Turner, M.G.; Gardner, R.H., eds. Quantitative methods in landscape ecology. New York, NY: Springer-Verlag. 82: 239–288.

Smith, C.M.; Langor, D.W.; Myrholm, C.; [et al.]. 2013. Changes in white pine blister rust infection and mortality in limber pine over time. Canadian Journal of Forest Research 43: 919–928.

Smith, D.R. 1967a. Effects of cattle grazing on a ponderosa pine-bunchgrass range in Colorado. Tech. Bull. 1371. Washington, DC: U.S. Department of Agriculture, Forest Service.

Smith, M.D. 2011. The ecological role of climate extremes: Current understanding and future prospects. Journal of Ecology. 99: 651–655.

Smith, R.S. 1967b. Verticicladiella root disease of pines. Phytopathology 57:935–938.

Smith, S.D.; Devitt, D.A.; Sala, A.; [et al.]. 1998. Water relations of riparian plants from warm desert regions. Wetlands. 18: 687–696.

Sorenson, E.; Williams, C. F.; Walser, R. H.; Davis, T. D.; Barker, P. 1984. Growth response of *Acer grandidentatum* Nutt. to chilling treatments. Journal of Environmental Horticulture. 2: 128–130.

Soulé, P.; Knapp, P. 2013. Radial growth rates of two co-occurring coniferous trees in the Northern Rockies during the past century. Journal of Arid Environments. 94: 87–95.

Spittlehouse, D.L.; Stewart, R.B. 2003. Adaptation to climate change in forest management. British Columbia Journal of Ecosystems and Management. 4: 1–11.

Springfield, H.W. 1976. Characteristics and management of Southwestern pinyon-juniper ranges: the status of our knowledge. Res. Pap. RM-RP-160. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station.

St. Clair, J.B.; Howe, G.T. 2007. Genetic maladaptation of coastal Douglas-fir seedlings to future climates. Global Change Biology. 13: 1441–1454.

St. Clair, J.B., Howe, G.T. 2011. Strategies for conserving forest genetic resources in the face of climate change. Turkish Journal of Botany 35: 403–409.

Stainforth, D.A.; Aina, T.; Christensen, C.; [et al.]. 2005. Uncertainty in predictions of the climate response to rising levels of greenhouse gases. Nature. 433: 403–406.

Stanton, F. 1974. Wildlife guidelines for range fire rehabilitation. Tech. Note 6712. Denver, CO: U.S. Department of the Interior, Bureau of Land Management.

Steele, R. 1990. *Pinus flexilis* James limber pine. In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America: Volume 1, conifers. Agric. Handbk. 654. Washington DC: U.S. Department of Agriculture, Forest Service: 348–353.

Steele, R.; Cooper, S.V.; Ondov, D.M.; [et al.]. 1983. Forest habitat types of eastern Idaho-western Wyoming. Gen. Tech. Rep. INT-GTR-144. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 122 p.

Steele, R.; Arno, S.F; Geier-Hayes, K. 1986. Wildfire patterns change in central Idaho's ponderosa pine-Douglas-fir forest. Western Journal of Applied Forestry. 1: 16–19.

Stephenson, J.R.; Calcarone, G.M. 1999. Mountain and foothills ecosystems: Habitat and species conservation issues. In: Stephenson, J.R.; Calcarone, G.M. Southern California mountains and foothills assessment. Gen. Tech. Rep. PSW-GTR-172. Albany, CA: U.S. Department of Agriculature, Forest Service, Pacific Southwest Research Station: 15–60.

Stillinger, C.R. 1944. Notes on *Cronartium occidentale.* Northwest Science 28:11–16.

Stout, D.L.; Sala, A. 2003. Xylem vulnerability to cavitation in *Pseudotsuga menziesii* and *Pinus ponderosa* from contrasting habitats. Tree Physiology. 23: 43–50.

Stromberg, J. 1998. Dynamics of Fremont cottonwood (*Populus fremontii*) and saltcedar (*Tamarix chinensis*) populations along the San Pedro River, Arizona. Journal of Arid Environments. 40: 133-155.

Stubbendieck, J.; Hatch, S.L.; Butterfield, C.H. 1992. North American range plants. 4th ed. Lincoln, NE: University of Nebraska Press.

Stuever, M.C.; Crawford, C.S.; Molles, M.C.; [et al]. 1997. Initial assessment of the role of fire in the Middle Rio Grande bosque. In: Greenlee, J.M., ed. Fire effects on rare and endangered species and habitats conference. Coeur d'Alene, ID. Fairfield, WA: International Association of Wildland Fire: 275–283.

Sturrock, R.N.; Frankel, S.J.; Brown, A.V.; [et al.]. 2010. Climate change and forest diseases. Plant Pathology. 60: 133–149.

Sutton, Richard F.; Johnson, Craig W. 1974. Landscape plants from Utah's mountains. EC-368. Logan, UT: Utah State University, Cooperative Extension.

Swanson, F.J.; Jones, J.A.; Wallin, D.O.; Cissel, J.H. 1994. Natural variability: Implications for ecosystem management. Volume II: Ecosystem management principles and applications. In: Eastside forest ecosystem health assessment. Gen. Tech. Rep. PNW-GTR-318. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 80–94.

Swetnam, T.W.; Allen, C.D.; Betancourt, J.L. 1999. Applied historical ecology: Using the past to manage for the future. Ecological Applications. 9: 1189–1206.

Szaro, R.C. 1990. Southwestern riparian plant communities: Site characteristics, tree species distributions, and size-class structures. Forest Ecology and Management. 33/34: 315–334.

Tang, K.; Feng, X.; Funkhouser, G. 1999. The δ13C of tree rings in full-bark and strip-bark bristlecone pine trees in the White Mountains of California. Global Change Biology. 5: 33–40.

Tauer, C.G.; Harris, K.D.; Van Haverbeke, D.F. 1987. Seed source influences juniper seedling survival under severe drought stress. Res. Note RM-RN-470. Fort Collins, CO: U.S. Department of Agriculture, Rocky Mountain Forest and Range Experiment Station.

Taylor, J.; Sturdevant, N. 1998. Limber pine mortality on the Lewis and Clark National Forest, Montana. For. Health Protect. Rep. 98-5. Missoula, MT: U.S. Department of Agriculture, Forest Service, Northern Region: 1–7.

Temperli, C.; Bugmann, H.; Elkin, C. 2013. Cross-scale interactions among bark beetles, climate change, and wind disturbances: A landscape modeling approach. Ecological Monographs. 83: 383–402.

Tisdale, E.W.; Hironaka, M. 1981. The sagebrush-grass region: A review of the ecological literature. Bull. 33. Moscow, ID: University of Idaho, Forest, Wildlife and Range Experiment Station.

Tomback, D.F. 1982. Dispersal of whitebark pine seeds by Clark's nutcracker: A mutualism hypothesis. Journal of Animal Ecology. 51: 451–467.

Tomback, D.F. 1983. Nutcrackers and pines: coevolution or coadaptation? In: Nitecki, M.H., ed. Coevolution. Chicago, IL: University of Chicago Press: 179–223.

Tomback, D.F. 1998. Clark's nutcracker (*Nucifraga columbiana*). The Birds of North America. 331: 1–23.

Tomback, D.F.; Achuff, P. 2010. Blister rust and western forest biodiversity: Ecology, values and outlook for white pines. Forest Pathology. 40: 186–225.

Tomback, D.; Arno, S.F.; Keane, R.E. 2001. Whitebark pine communities: ecology and restoration. Washington, DC: Island Press.

Tracy, J.L.; Robbins, T.O. 2009. Taxonomic revision and biogeography of the *Tamarix*-feeding *Diorhabda elongata* (Brullé, 1832) species group (Coleoptera: Chrysomelidae: Galerucinae: Galerucini) and analysis of their potential in biological control of *Tamarisk*. Auckland, New Zealand: Magnolia Press.

Turner, M.G.; Romme, W.H.; Gardner, R.H. 1999. Prefire heterogeneity, fire severity, and early postfire plant reestablishment in subalpine forests of Yellowstone National Park, Wyoming. International Journal of Wildland Fire. 9: 21–36.

Urban, D.L.; Harmon, M.E.; Halpern, C.B. 1993. Potential response of Pacific Northwestern forests to climatic change, effects of stand age and initial composition. Climatic Change. 23: 247–266.

U.S. Fish and Wildlife Service [USFWS]. 2011. Endangered and threatened wildlife and plants: 12-month finding on a petition to list *Pinus albicaulis* as endangered or threatened with critical habitat. Federal Register. 76: 42631-42654.

Van Dersal, W.R. 1938. Native woody plants of the United States, their erosion-control and wildlife values. Washington, DC: U.S. Department of Agriculture.

Vandersande, M.W.; Glenn, E.P.; Walworth, J.L. 2001. Tolerance of five riparian plants from the lower Colorado River to salinity drought and inundation. Journal of Arid Environments. 49: 147–159.

Van Devender, T.R. 1987. Late quaternary history of pinyon-juniper-oak woodlands dominated by *Pinus remota* and *Pinus edulis*. In: Everett, R.L., comp. Proceedings, pinyon-juniper conference; January 13-16, 1986; Reno, NV, Gen. Tech. Rep. INT-GTR-215. U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station, Ogden, UT: 99–103.

Vanoni, M.; Bugmann, H.; Nötzli, M.; Bigler, C. 2016. Drought and frost contribute to abrupt growth decreases before tree mortality in nine temperate tree species. Forest Ecology and Management. 382: 51–63.

Vasek, F.C.; Thorne, R.F. 1977. Transmontane coniferous vegetation. In: Barbour, M.G.; Major, Jack, eds. Terrestrial vegetation of California. New York: John Wiley and Sons: 797–832.

Villalba, R.; Veblen, T.T.; Ogden, J. 1994. Climatic influences on the growth of subalpine trees in the Colorado Front Range. Ecology. 75: 1450–1462.

Wadleigh, L.; Jenkins, M.J. 1996. Fire frequency and the vegetative mosaic of a spruce-fir forest in northern Utah. Great Basin Naturalist. 56: 28–37.

Wagener, W.W.; Mielke, J.L. 1961. A staining-fungus root disease of ponderosa, Jeffrey, and pinyon pine. Plant Disease Reporter. 45: 831–835.

Walker, R.F.; Geisinger, D.R.; Johnson, D.W.; Ball, J.T. 1995. Interactive effects of atmospheric CO_2 enrichment and soil N on growth and ectomycorrhizal colonization of ponderosa pine seedlings. Forest Science. 41: 491–500.

Walters, J.W. 1978. A guide to forest diseases of southwestern conifers. R3 78-9. Albuquerque, NM: U.S. Department of Agriculture, Forest Service, Southwest Region, Forest Insect and Disease Management.

Wang, T.; Hamann, A.; Yanchuk, A.; [et al.]. 2006. Use of response functions in selecting lodgepole pine populations for future climates. Global Change Biology. 12: 2404–2416.

Waring, R.H.; Running, S.W. 1998. Forest ecosystems: Analysis at multiple scales. 2nd ed. San Diego, CA: Academic Press.

Warwell, M.V.; Rehfeldt, G.E.; Crookston, N.L. 2007. Modeling contemporary climate profiles of whitebark pine (*Pinus albicaulis*) and predicting responses to global warming. In: Goheen, E.M.; Sniezko, tech. coords. Proceedings of the conference on whitebark pine: A Pacific Coast perspective. August 27-31, 2006; Ashland, OR. R6-NR-FHP-2007-01. Ashland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region: 139–142.

Watling, J.I.; Romañach, S.S.; Bucklin, D.N.; [et al.]. 2012. Do bioclimate variables improve performance of climate envelope models? Ecological Modelling. 246: 79–85.

Watt, A.S. 1947. Pattern and process in the plant community. Journal of Ecology. 35: 1–22.

Weed, A.S.; Bentz, B.J.; Ayres, M.P.; Holmes, T.P. 2015. Geographically variable response of *Dendroctonus ponderosae* to winter warming in the western United States. Landscape Ecology. 30: 1075–1099.

Weedbusters. n.d. Crack willow. http://www.weedbusters.org.nz/weed-information/salix-fragilis/59 [Accessed October 30, 2016].

Welsh, S.L.; Atwood, N.D.; Goodrich, S.; Higgins, L.C., eds. 1987. A Utah flora. The Great Basin Naturalist Memoir 9. Provo, UT: Brigham Young University.

West, N.E.; Rea, K.H.; Tausch, R.J. 1975. Basic synecological relationships in pinyon-juniper woodland understory vegetation related to climate. In: Gifford, G.F.; Busby, F.E., eds. The pinyon-juniper ecosystem: A symposium, May 1975, Logan, UT. Logan, UT: Utah State University, Agricultural Experiment Station: 41–53.

White, C.A.; Olmsted, C.E.; Kay, C.E. 1998. Aspen, elk, and fire in the Rocky Mountain National Parks of North America. Wildlife Society Bulletin. 26: 449–462.

Whited, D.C.; Lorang, M.S.; Harner, M.J.; [et al.]. 2007. Climate, hydrologic disturbance, and succession: drivers of floodplain pattern. Ecology. 88: 940–953.

Whitlock, C.; Bartlein, P.J. 1993. Spatial variations of Holocene climatic change in the Yellowstone region. Quaternary Research. 39: 231–238.

Wikars, L.O.; Schimmel, J. 2001. Immediate effects of fire-severity on soil invertebrates in cut and uncut pine forests. Forest Ecology and Management. 141: 189–200.

Williams, A.P.; Allen, C.D.; Millar, C.I.; [et al.]. 2010. Forest responses to increasing aridity and warmth in the southwestern United States. Proceedings of the National Academy of Sciences, USA. 107: 21289–21294.

Wilson, J.L.; Tkacz, B.M. 1996. Historical perspectives on forest insects and pathogens in the Southwest: Implications for restoration of ponderosa pine and mixed conifer forests. In: Covington, W.; Wagner, P.K., tech. coords. Conference on adaptive ecosystem restoration and management: Restoration of Cordilleran conifer landscapes of North America: Proceedings; June 6-8, 1996; Flagstaff, AZ. Gen. Tech. Rep. RM-GTR-278. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 26–31.

Winkel, V.K.; Syzdek, D.J. 2015. Response of vegetation after wildfire on the Warm Springs Natural Area in Moapa, Nevada. Presentation at the annual meeting of the Society of Ecological Restoration Southwest chapter, November 21, 2015. http://chapter.ser.org/southwest/files/2016/03/Winkel_WSNA-Fire-Presentation-November-2015D_2.pdf [Accessed October 30, 2016].

Wittie, Roger D.; McDaniel, Kirk C. 1990. Effects of tebuthiuron and fire on pinyon-juniper woodlands in southcentral New Mexico. In: Krammes, J. S., tech. coord. Effects of fire management on southwestern natural resources: Proceedings of the symposium; November 15-17, 1988; Tucson, AZ. Gen. Tech. Rep. RM-GTR-191. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 174–179.

Woodward, A.; Silsbee, D.G.; Schreiner, E.G.; Means, J.E. 1994. Influence of climate on radial growth and cone production in subalpine fir (*Abies lasiocarpa*) and mountain hemlock (*Tsuga mertensiana)*. Canadian Journal of Forest Research. 24: 1133–1143.

Worrall, J.J.; Rehfeldt, G.E.; Hamann, A.; [et al.]. 2013. Recent declines of *Populus tremuloides* in North America linked to climate. Forest Ecology and Management. 299: 35–51.

Wright, H.A. 1972. Shrub response to fire. In: McKell, C.M.; Blaisdell, J.P.; Goodin, J.R., eds. Wildland shrubs—Their biology and utilization: proceedings of a symposium; July 1971; Logan, UT. Gen. Tech. Rep. INT-GTR-1. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station: 204–217.

Wright, H.A.; Bailey, A.W. 1982. Fire ecology: United States and southern Canada. New York, NY: John Wiley and Sons.

Wu, Z.; Dijkstra, P.; Koch, G.W.; [et al.]. 2011. Responses of terrestrial ecosystems to temperature and precipitation change: A meta-analysis of experimental manipulation. Global Change Biology. 17: 927–942.

Xu, C.; Gertner, G.Z.; Scheller, R.M. 2009. Uncertainties in the response of a forest landscape to global climatic change. Global Change Biology. 15: 116–131.

Young, J.A.; Budy, J.D. 1979. Historical use of Nevada's pinyon-juniper woodlands. Journal of Forest History. 23: 112–121.

Zavarin, E.; Snajberk, K. 1973. Variability of the wood monoterpenoids from *Pinus aristata*. Biochemical Systematics. 1: 39–44.

Chapter 7: Effects of Climate Change on Nonforest Vegetation

Wayne G. Padgett, Matthew C. Reeves, Stanley G. Kitchen, David L. Tart, Jeanne C. Chambers, Cheri Howell, Mary E. Manning, and John G. Proctor

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.

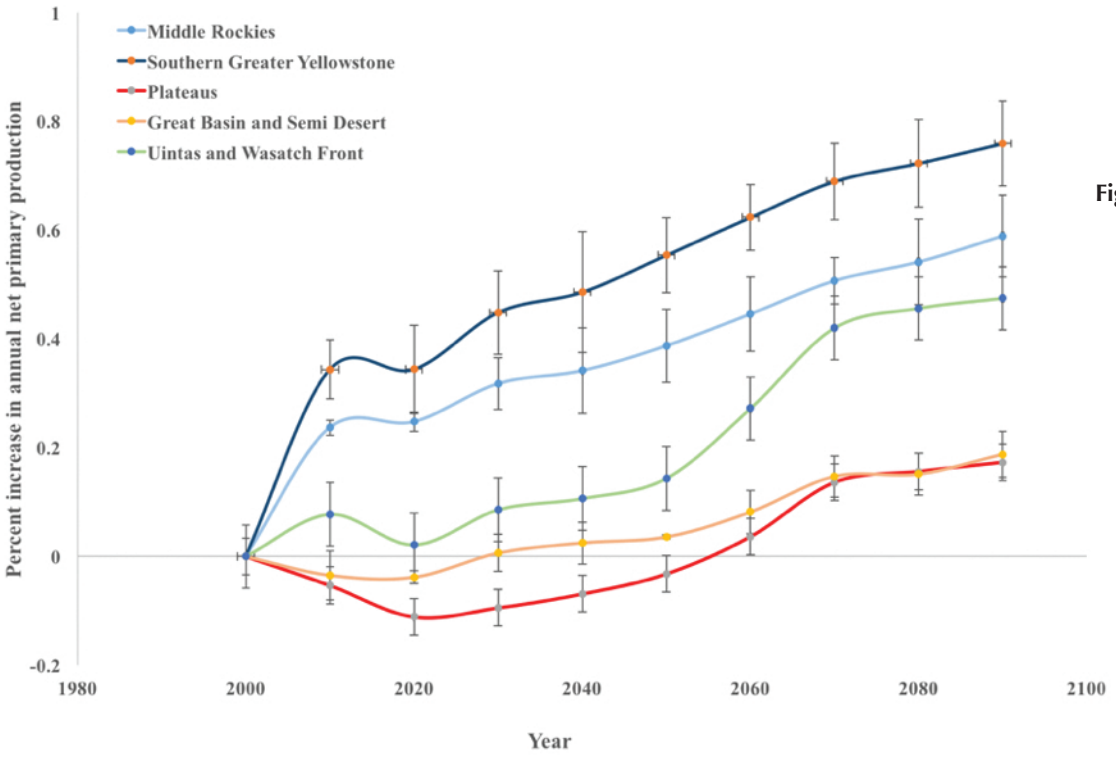


Figure 7.1—Average and standard deviation of net primary production under the A1B, A2, and B2 climate change scenarios for five subregions in the Intermountain Adaptation Partnership region (data from Reeves et al. [2014]).

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	

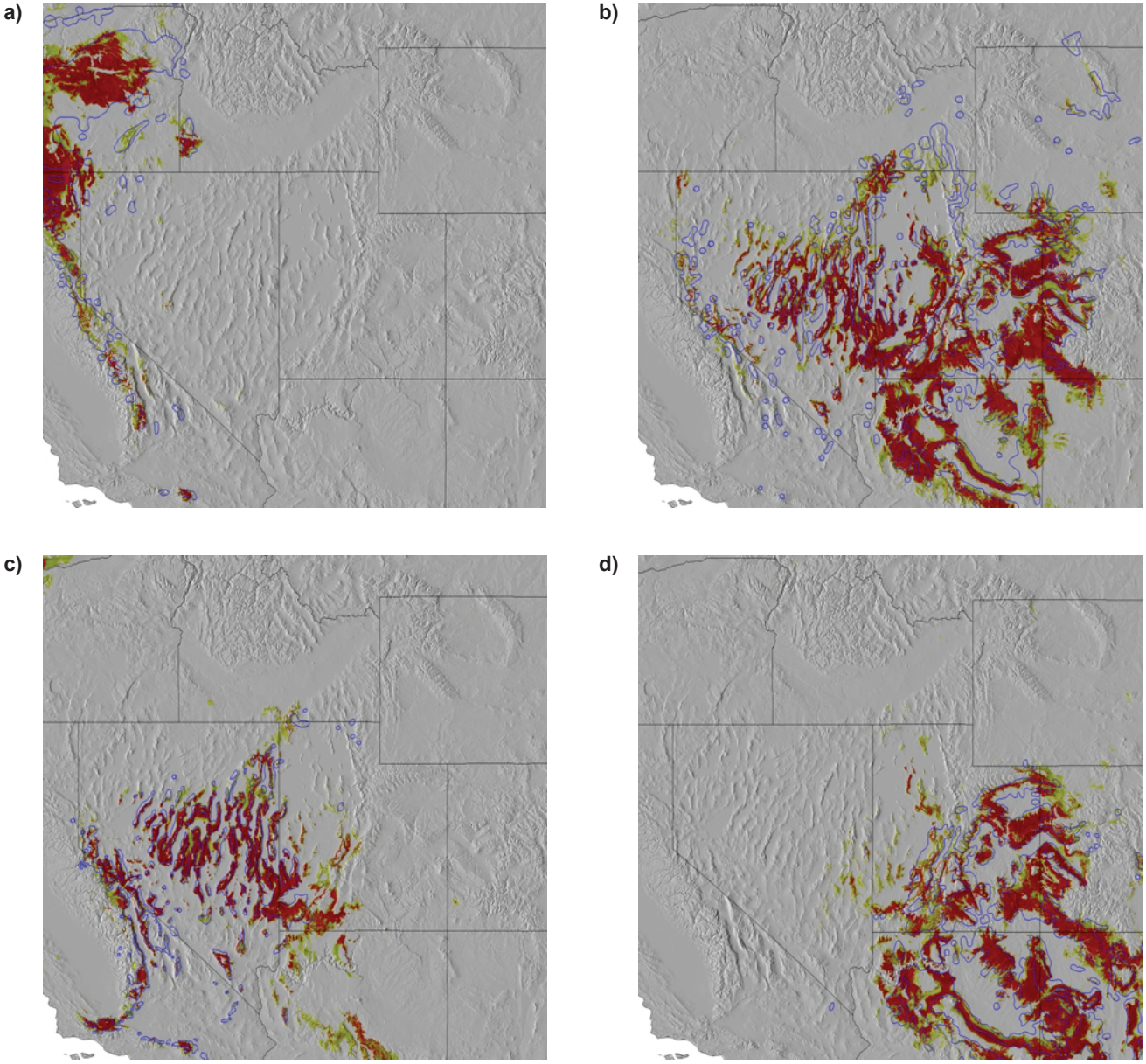


Figure 7.2—Modeled distribution of juniper and pinyon pine species in the Intermountain Adaptation Partnership region: (a) western juniper, (b) Utah juniper, (c) singleleaf pinyon, and (d) twoneedle pinyon. Data are from USDA FS (2017). Model projections to 2090 are based on two global circulation models (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

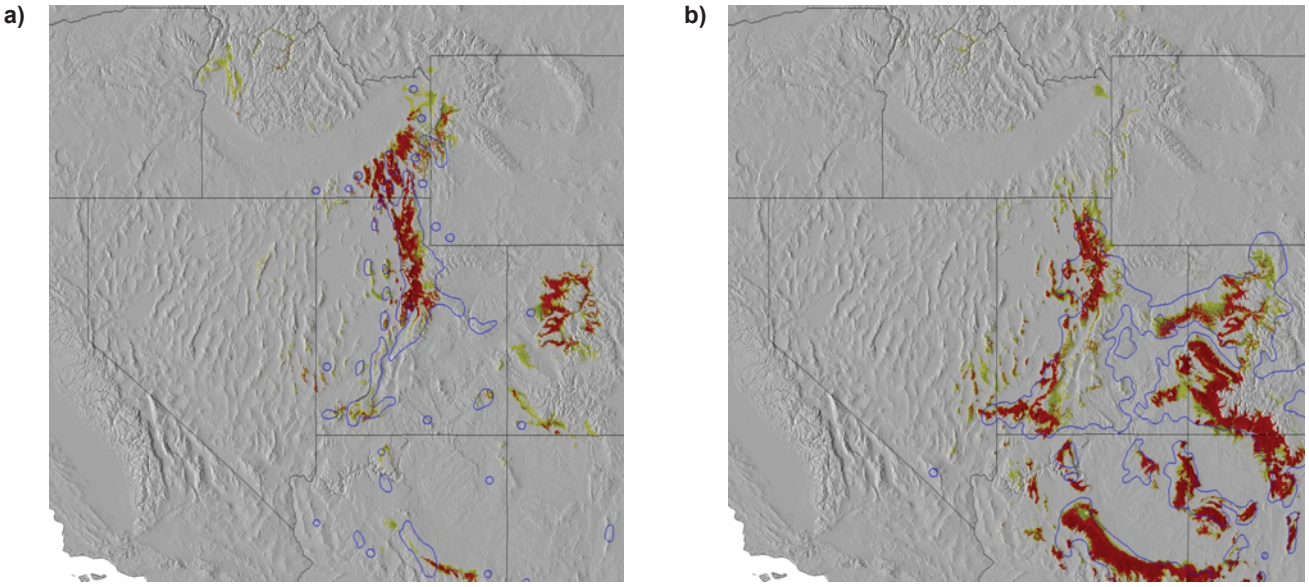


Figure 7.4—Modeled distribution of (a) bigtooth maple and (b) Gambel oak in the Intermountain Adaptation Partnership region. Data are from USDA FS (2017). Model projections to 2090 are based on two global circulation models (HadCM3GGa1, CGCM2_ghga), assuming an increase of greenhouse gas emissions of 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 co-dominant. 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.

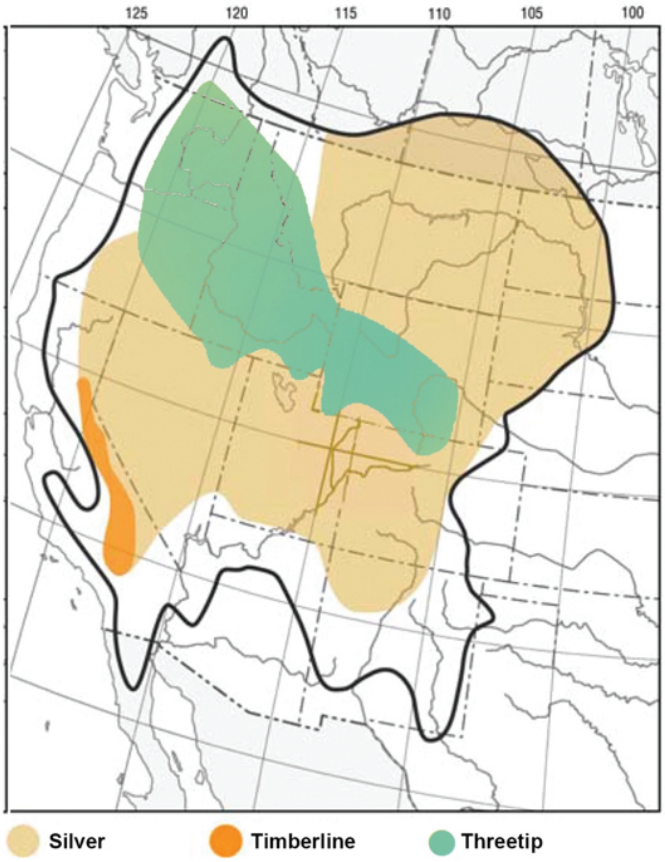


Figure 7.5—Distribution of silver sagebrush, timberline sagebrush, and threetip sagebrush in the western United States (from Mahalovich and McArthur [2004]).

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

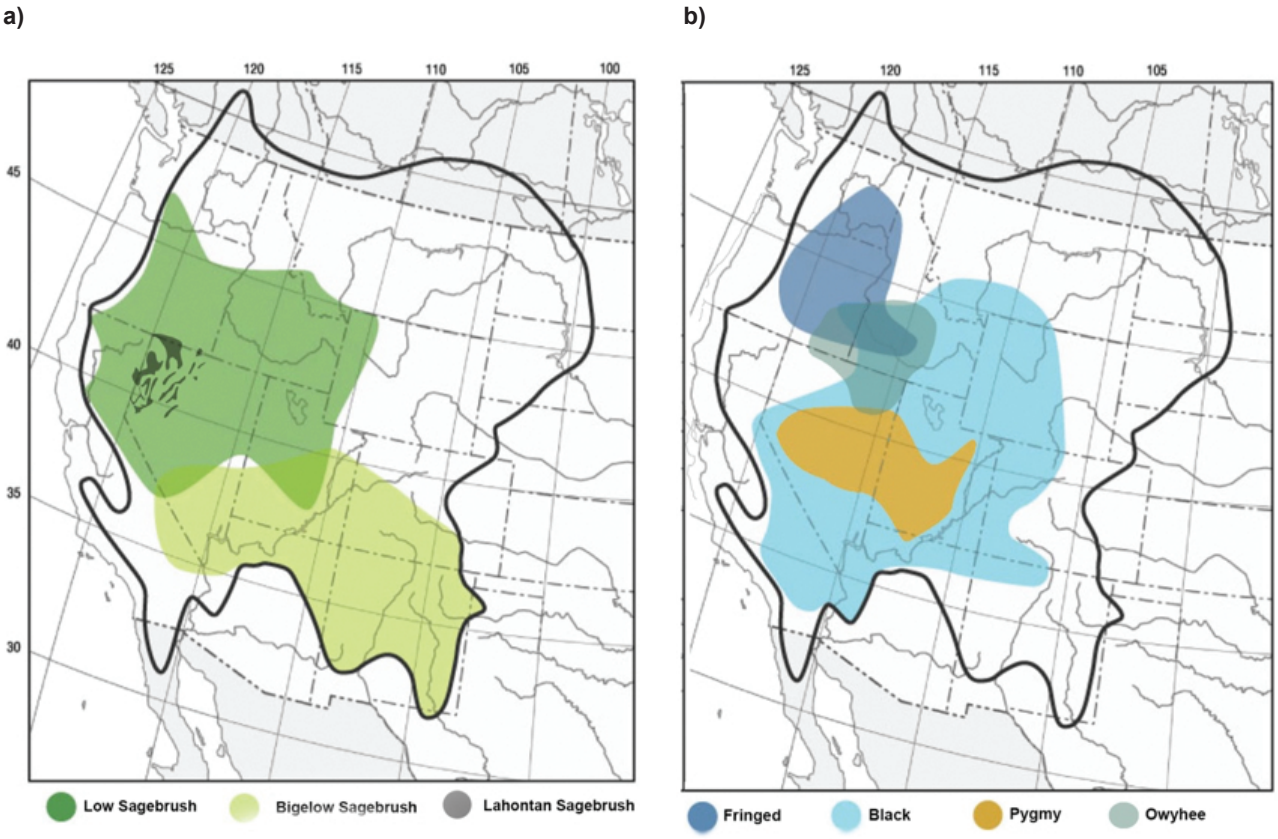


Figure 7.6—Distribution of low sagebrush species in the western United States: (a) low, Bigelow, and Lahontan sagebrush; and (b) fringed, black, pygmy, and Owyhee sagebrush (from Mahalovich and McArthur [2004]).

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).

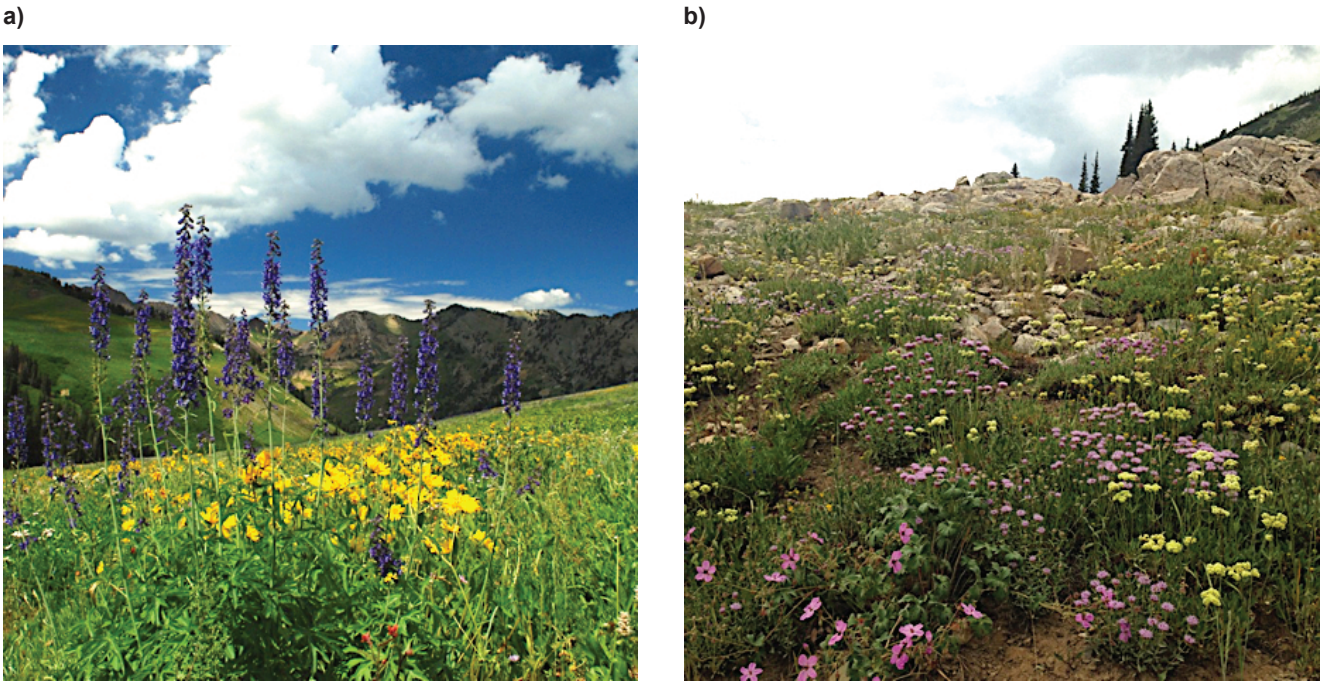


Figure 7.9—Subalpine forb communities in the Intermountain Adaptation Partnership region: (a) a subalpine tall forb community occurring on deep, productive soils, and (b) a subalpine low forb community occurring on shallow, well-drained soils (photos: W. Padgett, U.S. Forest Service).

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).

References

Abatzoglou, J.T.; Kolden, C.A. 2013. Relationships between climate and macroscale area burned in the western United States. International Journal of Wildland Fire. 22: 1003–1020.

Aleksoff, K.C. 1999. *Symphoricarpos oreophilus*. In: Fire Effects Information System. Missoula, MT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. http://www.feis-crs.org/feis/faces/index.xhtml [Accessed March 30, 2016].

Alexander, J.; Diez, J.; Levine, J. 2015. Novel competitors shape species' responses to climate change. Nature. 525: 515–518.

Alo, C.A.; Wang, G. 2008. Hydrological impact of the potential future vegetation response to climate changes projected by 8 GCMs. Journal of Geophysical Research: Biogeosciences. 113: G3.

Anderson, J.E.; Inouye, R.S. 2001. Landscape-scale changes in plant species abundance and biodiversity of a sagebrush steppe over 45 years. Ecological Monographs. 71: 531–556.

Anderson, M.D. 2001. *Salix scouleriana*. In: Fire Effects Information System. Missoula, MT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. http://www.feis-crs.org/feis/faces/index.xhtml [Accessed March 30, 2016].

Auble, G.T.; Friedman, J.M.; Scott, M.L. 1994. Relating riparian vegetation to present and future stream-flows. Ecological Applications. 4: 544–554.

Auble, G.T.; Scott, M.L. 1998. Fluvial disturbance patches and cottonwood recruitment along the upper Missouri River, Montana. Wetlands. 18: 546–556.

Baker, W.L. 2006. Fire and restoration of sagebrush ecosystems. Wildlife Society Bulletin. 34: 177–185.

Baker, W.L. 2011. Pre-Euro-American and recent fire in sagebrush ecosystems. In: Knick, S.T.; Connelly, J.W., eds. Greater Sage-grouse: Ecology and conservation of a landscape species and its habitats. Berkeley, CA: University of California Press: 185–201.

Baker, W.L.; Shinneman, D.J. 2004. Fire and restoration of piñon-juniper woodlands in the western United States: A review. Forest Ecology and Management. 189: 1–21.

Balch, J.K.; Bradley, B.A.; D'Antonio, C.; Gomez-Dans, J. 2013. Introduced annual grass increases regional fire activity across the arid western USA (1980-2009). Global Change Biology. 19: 173–183.

Balzotti, C.S.; Kitchen, S.G.; McCarthy, C. 2016. Beyond the single species climate envelope: A multifaceted approach to mapping climate change vulnerability. Ecosphere. 7: e01444.

Bauer, J.M.; Weisberg, P.J. 2009. Fire history of a central Nevada pinyon-juniper woodland. Canadian Journal of Forest Research. 39: 1589–1599.

Beniston, M. 2003. Climatic change in mountain regions: A review of possible impacts. Climatic Change. 59: 5–31.

Beaubien, E.; Hamann, A. 2011. Spring flowering response to climate change between 1936 and 2006 in Alberta, Canada. BioScience. 61: 514–524.

Blaisdell, J.P.; Holmgren, R.C. 1984. Managing Intermountain rangelands–salt-desert shrub ranges. Gen. Tech. Rep. INT-163. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 52 p.

Blaisdell, J.P.; Murray, R.B.; McArthur, E.D. 1982. Managing Intermountain rangelands–sagebrush-grass ranges. Gen. Tech. Rep. INT-134. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 41 p.

Brabec, M.M.; Germino, M.J.; Shinneman, D.J. [et al.]. 2015. Challenges of establishing big sagebrush (*Artemisia tridentata*) in rangeland restoration: Effects of herbicide, mowing, whole-community seeding, and sagebrush seed sources. Rangeland Ecology and Management. 68: 432–435.

Bradley, B.A. 2009. Regional analysis of climate change on cheatgrass invasion shows potential risk and opportunity. Global Change Biology. 15: 196–208.

Bradley, B.A. 2010. Assessing ecosystem threats from global and regional change: Hierarchical modeling of risk to sagebrush systems from climate change and invasive species in Nevada, USA. Ecography. 33: 198–208.

Bradley, B.A.; Curtis, C.A.; Chambers, J.C. 2016. *Bromus* response to climate and projected changes with climate change. In: Germino, M.J.; Chambers, J.C.; Brown, C.S, eds. Exotic brome-grasses in arid and semiarid ecosystems of the western US: Causes, consequences, and management implications. Cham, Switzerland: Springer: 257–274.

Brooks, M.L.; Esque, T.C.; Duck, T. 2007. Creosotebush, blackbrush, and interior chaparral shrublands [Chapter 6]. In: Hood, S.M.; Miller, M., eds. Fire ecology and management of the major ecosystems of southern Utah. Gen. Tech. Rep. RMRS-GTR-202. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 97–110.

Burkhardt, J.W.; Tisdale, E.W. 1969. Nature and successional status of western juniper vegetation in Idaho. Journal of Range Management. 22: 264–270.

Chambers, J.C. 2008. Climate change and the Great Basin. In: Chambers, J.C.; Devoe, N.; Evenden, A. eds. Collaborative management and research in the Great Basin: Examining the issues and developing a framework for action. Gen. Tech. Rep. RMRS-GTR-204. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 29–32.

Chambers, J.C.; Bradley, B.A.; Brown, C.S.; [et al.]. 2013. Resilience to stress and disturbance, and resistance to *Bromus tectorum* L. invasion in cold desert shrublands of western North America. Ecosystems 17: 360–375.

Chambers, J.C.; Miller, R.F.; Board, D.I.; [et al.]. 2014. Resilience and resistance of sagebrush ecosystems: Implications for state and transition models and management treatments. Rangeland Ecology and Management. 67: 440–454.

Chambers, J.C.; Roundy, B.A.; Blank, R.R.; [et al.]. 2007. What makes Great Basin sagebrush ecosystems invasible by *Bromus tectorum*? Ecological Monographs. 77: 117–145.

Chaney, L.; Richardson, B.A.; Germino, M.J. 2017. Climate drives adaptive genetic responses associated with survival in big sagebrush (*Artemisia tridentata*). Evolutionary Applications. 22: 699–710.

Collen, P.; Gibson, R.J. 2001. The general ecology of beavers (*Castor* spp.) as related to their influence on stream ecosystems and riparian habitats, and the subsequent effects on fish—A review. Reviews in Fish Biology and Fisheries. 10: 439–461.

Comer, P.; Faber-Langendoen, D.; Evans, R.; [et al.]. 2003. Ecological systems of the United States: A working classification of U.S. terrestrial systems. Arlington, VA: NatureServe.

Comer, P. J.; Young, B.; Schulz, K.; [et al.]. 2012. Climate change vulnerability and adaptation strategies for natural communities: piloting methods in the Mojave and Sonoran deserts. Report to the U.S. Fish and Wildlife Service. Arlington, VA: NatureServe.

Cooper, D.J.; Andersen, D.C.; Chimner, R.A. 2003. Multiple pathways for woody plant establishment on floodplains at local to regional scales. Journal of Ecology. 91: 182–196.

Crawford, J.A.; Olson, R.A.; West, N.E.; [et al.]. 2004. Ecology and management of sage-grouse and sage-grouse habitat. Journal of Range Management. 57: 2–19.

Cronin, E.H.; Williams, M.C. 1966. Principles for managing ranges infested with halogeton. Journal of Range Management. 19: 226–227.

Cronquist, A.; Holmgren, N.H.; Holmgren, P.K. 1997. Intermountain flora: Vascular plants of the Intermountain West, USA. Volume 3, Part A: Subclass Rosidae (except Fabales). New York, NY: The New York Botanical Garden. 446 p.

D'Antonio, C.M.; Vitousek, P.M. 1992. Biological invasions by exotic grasses, the grass/fire cycle, and global change. Annual Review of Ecology and Systematics. 23: 63–87.

Davies, K.W.; Boyd, C.S., Nafus, A.M. 2013. Restoring the sagebrush component in crested wheatgrass–dominated communities. Rangeland Ecology and Management. 66: 472–478.

Davies, K.W.; Sheley, R.L.; Bates, J.D. 2008. Does prescribed fall burning *Artemisia tridentata* steppe promote invasion or resistance to invasion after a recovery period? Journal of Arid Environments. 72: 1076−1085.

Davis, J.N.; Brotherson, J.D. 1991. Ecological relationships of curlleaf mountain-mahogany (*Cercocarpus ledifolius* Nutt.) communities in Utah and implications for management. The Great Basin Naturalist. 51: 153–166.

Dealy, J.E. 1975. Ecology of curl-leaf mahogany (*Cercocarpus ledifolius* Nutt.) in Oregon and adjacent areas. Doctoral dissertation. Corvallis, OR: Oregon State University. 168 p.

DeBano, L.F. 1981. Water repellent soils: A state-of-the-art. Gen. Tech. Rep. PSW-46. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station. 21 p.

Delcourt, P.A.; Delcourt, H.R. 1981. Vegetation maps for eastern North America: 40,000 yr B.P. to the present. In: Romans, R.C., ed. Geobotany II. New York, NY: Plenum: 123–165.

Dwire, K.A.; Kauffman, J.B. 2003. Fire and riparian ecosystems in landscapes of the western USA. Forest Ecology and Management. 178: 61–74.

Eckert, R.E., Jr. 1954. A study of competition between whitesage and halogeton in Nevada. Journal of Range Management. 7: 223–225.

Ellison, L. 1954. Subalpine vegetation of the Wasatch Plateau, Utah. Ecological Monographs. 24: 89–184.

Engel, D.M.; Bonham, C.D.; Bartel, L.E. 1983. Ecological characteristics and control of Gambel oak. Journal of Range Management. 36: 363–365.

Engelhardt, B.M.; Chambers, J.C.; Weisberg, P.J. 2015. Geomorphic predictors of riparian vegetation in small mountain watersheds. Journal of Plant Ecology. 8: 593–604.

Fernandez, D.P.; Neff, J.C.; Cho-ying, H.; [et al.]. 2013. Twentieth century carbon stock changes related to piñon-juniper expansion into a black sagebrush community. Carbon Balance and Management. 8: 8.

Finch, D.M., ed. 2012. Climate change in grasslands, shrublands, and deserts of the interior American West: A review and needs assessment. Gen. Tech. Rep. RMRS-GTR-285. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 139 p.

Fischer, W.C.; Clayton, B.D. 1983. Fire ecology of Montana forest habitat types east of the Continental Divide. Gen. Tech. Rep. INT-GTR-141. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 83 p.

Floyd, M.L.; Romme, W.H.; Rocca, M.E.; [et al.]. 2015. Structural and regenerative changes in old-growth piñon-juniper woodlands following drought-induced mortality. Forest Ecology and Management. 341: 18–29.

Fryer, J.L. 2009. *Artemisia nova*. In: Fire Effects Information System. Missoula, MT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. http://www.fs.fed.us/database/feis/ [Accessed March 30, 2016].

Garrison, H. 2006. Study of a putative hybrid taxon in the *Artemisia tridentata* complex. Master of Science Thesis. Logan, UT: Utah State University.

Garrison, H.D.; Shultz, L.M.; McArthur, E.D. 2013. Studies of a new hybrid taxon in the *Artemisia tridentata* (Asteraceae: Anthemideae) complex. Western North American Naturalist. 73: 1–19.

Glenn, E.P.; Hodges, C.N.; Lieth, H.; [et al.]. 1992. Climate: Growing halophytes to remove carbon from the atmosphere. Environment. 34: 40–43.

Gonella, M.P.; Neel, M.C. 1995. Characterizing rare plant habitat for restoration in the San Bernardino National Forest. In: Roundy, B.A.; McArthur, E.D.; Haley, J.S.; Mann, D.K., comps. Proceedings: wildland shrub and arid land restoration symposium; 1993 October 19-21; Las Vegas, NV. Gen. Tech. Rep. INT-GTR-315. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 81–93.

Goodrich, S.K.; Zobell, A. 2011. Trend of Gardner saltbush and halogeton in the lower Green River Basin, Wyoming. In: Monaco, T.A.; Schupp, E.W.; Pendleton, R.L.; [et al.], comps. Proceedings—Threats to shrubland ecosystem integrity; 2010 May 18-20; Logan, UT. Natural Resources and Environmental Issues, Volume 17. Logan, UT: S.J. and Jessie E. Quinney Natural Resources Research Library: 291–297.

Gori, D.; Cooper, M.S.; Soles, E.S.; [et al.]. 2014. Gila River flow needs assessment. The Nature Conservancy. http://nmconservation.org/Gila/GilaFlowNeedsAssessment.pdf [Accessed January 13, 2017].

Gottfried, M.; Pauli, H.; Futschik, A.; [et al.]. 2012. Continent-wide response of mountain vegetation to climate change. Nature Climate Change. 2: 111–115.

Gregory, S. 1982. Subalpine forb community types of the Bridger-Teton National Forest, Wyoming. Ogden, Utah: U.S. Department of Agriculture, Forest Service, Intermountain Region.

Gruell, G.; Bunting, S.; Neuenschwander, L. 1985. Influence of fire on curlleaf mountain-mahogany in the Intermountain West. In: Lotan, J.E.; Brown, J.K., comps. Fire's effects on wildlife habitat—Symposium proceedings; 1984 March 21; Missoula, MT. Gen. Tech. Rep. INT-186. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 58–72.

Gucker, C.L. 2006. *Cercocarpus ledifolius*. In: Fire Effects Information System. Missoula, MT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. http://www.fs.fed.us/database/feis/ [Accessed March 30, 2016].

Gucker, C. 2012. *Rubus parviflorus*. In: Fire Effects Information System. Missoula, MT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. http://www.fs.fed.us/database/feis/ [Accessed March 30, 2016].

Harper, K.T.; Van Buren, R.; Kitchen, S.G. 1996. Invasion of alien annuals and ecological consequences in salt desert shrublands of western Utah. In: Barrow, J.R.; McArthur, E.D.; Sosebee, R.E.; Tausch, J.R., comps. Proceedings—Symposium on shrubland ecosystem dynamics in a changing environment; 1995 May 23–25; Las Cruces, NM. Gen. Tech. Rep. INT-GTR-338. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 58–65.

Hayward, C.L. 1952. Alpine biotic communities of the Uinta Mountains, Utah. Ecological Monographs. 22: 93–120.

Heyerdahl, E.K.; Miller, R.F.; Parsons, R.A. 2006. History of fire and Douglas-fir establishment in a savanna and sagebrush–grassland mosaic, southwestern Montana, USA. Forest Ecology and Management. 230: 107–118.

Hoffmann, A.A.; Sgrò, C.M. 2011. Climate change and evolutionary adaptation. Nature. 470: 479–485.

Hooker, T.D.; Stark, J.M.; Norton, U.; [et al.]. 2008. Distribution of ecosystem C and N within contrasting vegetation types in a semiarid rangeland, in the Great Basin, USA. Biogeochemistry. 90: 291–308.

Howard, J.L. 2003. *Artemisia bigelovii*. In: Fire Effects Information System. Missoula, MT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. http://www.fs.fed.us/database/feis/plants/shrub/artbig/all.html [Accessed March 30, 2016].

Hull, A.C., Jr.; Hull, M.K. 1974. Presettlement vegetation in Cache Valley, Utah and Idaho. Journal of Range Management. 27: 27–29.

Intergovernmental Panel on Climate Change [IPCC]. 2007. Climate change 2007: Impacts, adaptation, and vulnerability; contribution of working group II to the fourth assessment report of the Intergovernmental Panel on Climate Change, (ed.) M.L. Parry, O.F. Canziani, J.P. Palutikof, P.J. van der Linden, and C.E. Hanson. Cambridge, UK: Cambridge University Press. 976 p.

Jakubos, B.; Romme, W.H. 1993. Invasion of subalpine meadows by lodgepole pine in Yellowstone National Park, Wyoming, USA. Arctic and Alpine Research. 25: 382–390.

Jenkins, S.H.; Busher, P.E. 1979. *Castor canadensis*. Mammalian Species. 120: 1–8.

Jepson Flora Project. 2016. *Artemisia rothrockii* (Rothrock sagebrush). The Jepson Herbarium. http://ucjeps.berkeley.edu/eflora/eflora_display.php?tid=1238 [Accessed March 30, 2016].

Johnson, K.A. 2000. *Prunus virginiana*. In: Fire Effects Information System. Missoula, MT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. http://www.fs.fed.us/database/feis/plants/tree/pruvir/all.html [Accessed March 30, 2016].

Kauffman, J.B. 2001. Workshop on the multiple influences of riparian/stream ecosystems on fires in western forest landscapes: Summary report to Rocky Mountain Forest and Range Experiment Station, Stream Systems Technical Center, Fort Collins, CO. Corvallis, OR: Oregon State University, Department of Fisheries and Wildlife. 209 p.

Kaufmann, M.R.; Huisjen, D.W.; Kitchen, S.; [et al.]. 2016. Gambel oak ecology and management in the southern Rockies: The status of our knowledge. SRFSN Publication 2016-1. Fort Collins, CO: Colorado State University, Southern Rockies Fire Sciences Network. 16 p.

Kay, C. 2015. Long-term vegetation change in Utah's Henry Mountains: A study in repeat photography. Publication Number BLM-UT-GI-14001-8002. Providence, UT: U.S. Department of the Interior, Bureau of Land Management. https://www.blm.gov/style/medialib/blm/wo/blm_library/misc_pubs.Par.74891.File.dat/Kay2015_HenryMtns.pdf [Accessed February 13, 2017].

Kitchen, S.G.; Jorgensen, G.L. 1999. Annualization of rodent burrow clusters and winterfat decline in a salt-desert community. In: McArthur E.D.; Ostler, W.K.; Wambolt, C.L. comps. Proceedings: shrubland ecotones; 1998 August 12-14; Ephraim, UT. Proc. RMRS-P-11. Odgen, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 175–180.

Kitchen, S.G.; Jorgensen, G.L. 2001. Winterfat decline and halogeton spread in the Great Basin. In: McArthur, E.D.; Fairbanks, D.J., comps. Shrubland ecosystem genetics and biodiversity: Proceedings; 2000 June 13-15; Provo, UT. Proceedings RMRS-P-21. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 200–203.

Kitchen, S.G.; McArthur, E.D. 2007. Big and black sagebrush landscapes. In: Hood, S.M.; Miller, M., eds. Fire ecology and management of the major ecosystems of southern Utah. Gen. Tech. Rep. RMRS-GTR-202. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 73–95.

Kitchen, S.G.; Meyer, S.E.; Carlson, S.L. 2015. Mechanisms for maintenance of dominance in a nonclonal desert shrub. Ecosphere. 6: 252.

Kropp, B.R.; Hutchison, L.J. 1996. *Gigasperma americanum*, a new hypogeous member of the Basidiomycota associated with *Cercocarpus ledifolius* in Utah. Mycologia. 88: 662–665.

LANDFIRE. 2008. Biophysical settings layers, LANDFIRE 1.1.0. U.S. Department of the Interior, Geological Survey. http://landfire.cr.usgs.gov/viewer/ [Accessed March 27, 2016].

LANDFIRE. 2012. Existing vegetation type layers, LANDFIRE 1.1.0. U.S. Department of the Interior, Geological Survey. http://landfire.cr.usgs.gov/viewer/ [Accessed March 27, 2016].

Lanner, R.M. 1983. Trees of the Great Basin: A natural history. Reno, NV: University of Nevada Press. 215 p.

Lesica, P.; Cooper, S.V.; Kudray, G. 2007. Recovery of big sagebrush following fire in southwestern Montana. Rangeland Ecology and Management. 60: 261–269.

Lesica, P.; DeLuca, T.H. 1996. Long-term harmful effects of crested wheatgrass on Great Plains grassland ecosystems. Journal of Soil and Water Conservation. 51: 408–409.

Lewis, M.E. 1971. Flora and major plant communities of the Ruby-East Humboldt Mountains; with special emphasis on Lamoille Canyon. Ogden, UT: U.S. Department of Agriculture, Forest Service, Humboldt National Forest.

Lewis, M.E. 1975. Plant communities of the Jarbridge Mountain complex; Humboldt National Forest. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Region.

Lewis, M.E. 1993. Successional changes in the tall forb type: Wasatch Plateau. Report. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Region. 75 p.

Loope, L.L. 1970. Subalpine and alpine vegetation of northeastern Nevada. Ph.D. dissertation. Durham, NC: Duke University. 292 p.

Mahalovich, M.F.; McArthur, E.D. 2004. Sagebrush (*Artemisia* spp.) seed and plant transfer guidelines. Native Plants Journal. 5: 141–148.

Mahoney, J.M.; Rood, S. 1998. Streamflow requirements for cottonwood seedling recruitment—An integrative model. Wetlands. 18: 634–645.

McArthur, E.D. 1994. Ecology, distribution, and values of sagebrush within the Intermountain Region. In: Monsen, S.B.; Kitchen, S.G., comps. Proceedings—ecology and management of annual rangelands; 1992 May 18-22; Boise, ID. Gen. Tech. Rep. INT-GTR-313. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 347–351.

McArthur, E.D. 2000. Sagebrush systematics and distribution. In: Entwistle, P.G.; DeBolt, A.M.; Kaltenecker, J.H.; Steenhof, K., comps. Proceedings: Sagebrush steppe ecosystems symposium. Bureau of Land Management Publication No. BLM/ID/PT-001001+1150. Boise, ID: Bureau of Land Managment: 9–14.

McArthur, E.D.; Richardson, B.A.; Kitchen, S.G. 2013. Great Basin Experimental Range: Annotated bibliography. Gen. Tech. Rep. RMRS-GTR-305WWW. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 97 p.

McArthur, E.D.; Sanderson, S.C. 1999. Ecotone: Introduction, scale, and big sagebrush example. In: McArthur, E.D.; Ostler, K.W.; Wambolt, C.L., comps. Proceedings: Shrubland ecotones. Proceedings RMRS-P-11. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 3–8.

McKell, C.M. 1989. Shrub palatability. In: McKell, C.M., ed. The biology and utilization of shrubs. San Diego, CA: Academic Press, Inc.: 267–281.

McKenzie, D.; Gedalof, Z.E.; Peterson, D.L.; Mote, P. 2004. Climatic change, wildfire, and conservation. Conservation Biology. 18: 890–902.

McLachlan, J.S.; Clark, J.S.; Manos, P.S. 2005. Molecular indicators of tree migration capacity under rapid climate change. Ecology. 86: 2088–2098.

McWilliams, J. 2003. *Artemisia rigida*. In: Fire Effects Information System. Missoula, MT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. http://www.fs.fed.us/database/feis/ [Accessed March 30, 2016].

Mensing, S.; Livingston, S.; Barker, P. 2006. Long-term fire history in Great Basin sagebrush reconstructed from macroscopic charcoal in spring sediments, Newark Valley, Nevada. Western North American Naturalist. 66: 64–77.

Merritt, D.M.; Nilsson, C.; Jansson, R. 2010. Consequences of propagule dispersal and river fragmentation for riparian plant community diversity and turnover. Ecological Monographs. 80: 837–858.

Meyer, S.E. 2003. *Atriplex* L. saltbush. In: Bonner, F.T., tech. coord. Woody plant seed manual. Washington, DC: U.S. Department of Agriculture, Forest Service. http://www.nsl.fs.fed.us/A%20genera.pdf [Accessed March 30, 2016].

Meyer, S.E.; Garvin, S.C.; Beckstead, J. 2001. Factors mediating cheatgrass invasion of intact salt desert shrubland. In: McArthur, E.D.; Fairbanks, D.J., comps. Shrubland ecosystem genetics and biodiversity: Proceedings; 2000 June 13-15; Provo, UT. Proceedings RMRS-P-21. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 224–232.

Meyer, S.E.; Warren, T.W. 2015. Seeding big sagebrush successfully on Intermountain rangelands. Great Basin Factsheet No. 10. Sage Grouse Initiative. http://www.sagegrouseinitiative.com/seeding-big-sagebrush-successfully-on-intermountain-rangelands/ [Accessed November 10, 2016].

Millar, C.I.; Stephenson, N.L. 2015. Temperate forest health in an era of emerging megadisturbance. Science. 349: 823–826.

Miller, R.F.; Baisan, C.; Rose, J.; Pacioretty, D. 2001. Pre- and post-settlement fire regimes in mountain big sagebrush steppe and aspen: The Northwestern Great Basin. Final Report to the National Interagency Fire Center. http://fresc.usgs.gov/products/papers/fr-nifc.pdf [Accessed November 10, 2016].

Miller, R.F., Knick, S.T.; Pyke, D.A.; [et al.]. 2011. Characteristics of sagebrush habitats and limitations to long-term conservation. In: Knick, S.T.; Connelly, J.W., eds. Greater sage-grouse: Ecology and conservation of a landscape species and its habitats. Studies in Avian Biology, Volume 38. Berkeley, CA: University of California Press: 145–184.

Miller R.F.; Rose, J.A. 1999. Fire history and western juniper encroachment in sagebrush steppe. Journal of Range Management. 52: 550–559.

Miller, R.F.; Tausch, R.J. 2001. Role of fire in juniper and pinyon woodlands: A descriptive analysis. In: Proceedings of the invasive species workshop: Tall Timbers Research Station Miscellaneous Publication, No. 11: 15–30.

Miller, R.F.; Tausch, R.J.; McArthur, E.D.; [et al.]. 2008. Age structure and expansion of piñon–juniper woodlands: A regional perspective in the Intermountain West. Research Paper RMRS-RP-69. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 15 p.

Miller, R.F.; Tausch, R.; Waichler, W. 1999. Old-growth juniper and pinyon woodlands. In: Monsen, S.B.; Stevens, R., comps. Proceedings: Ecology and management of pinyon-juniper communities within the Interior West; September 15-18, 1997; Provo, UT. Proceedings RMRS-P-9. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 375–384.

Morgan, J.A.; Milchunas, D.G.; LeCain, D.R.; [et al.]. 2007. Carbon dioxide enrichment alters plant community structure and accelerates shrub growth in the shortgrass steppe. Proceedings of the National Academy of Sciences, USA. 104: 14724–14729.

Morgan, J.A.; Pataki, D.E.; Korner, C.; [et al.]. 2004. Water relations in grassland and desert ecosystems exposed to elevated atmospheric CO_2. Oecologia. 140: 11–25.

Mozingo, H.N. 1987. Shrubs of the Great Basin: A natural history. Reno, NV: University of Nevada Press.

Munson, S.M., Belnap, J.; Schelz, C.D.; [et al.]. 2011. On the brink of change: Plant responses to climate on the Colorado Plateau. Ecosphere. 2: 68.

Naiman, R.J.; Dècamps, H. 1997. The ecology of interfaces: Riparian zones. Annual Review of Ecology and Systematics. 28: 621–658.

Nakamura, F.; Swanson, F.J.; Wondzell, S.M. 2000. Disturbance regimes of stream and riparian systems—A disturbance-cascade perspective. Hydrological Processes. 14: 2849–2860.

Natural Resources Conservation Service [NRCS]. 2017. The PLANTS Database. Greensboro, NC: National Plant Data Team. http://plants.usda.gov [Accessed July 12, 2017].

NatureServe. 2013. International ecological classification standard: Terrestrial ecological classifications. NatureServe central databases. Arlington, VA: NatureServe.

Neff, J.C.; Barger, T.N.N.; Baisden, W.T.; [et al.]. 2009. Soil carbon storage responses to expanding pinyon-juniper populations in southern Utah. Ecological Applications. 19: 1405–1416.

Nelson, Z.J.; Weisberg, P.J.; Kitchen, S.G. 2014. Influence of climate and environment on post-fire recovery of mountain big sagebrush. International Journal of Wildland Fire. 23: 131–142.

Neilson, R.P.; Lenihan, J.M.; Bachelet, D.; Drapek, R.J. 2005. Climate change implications for sagebrush ecosytems. Transactions of the North American Wildlife and Natural Resources Conference: 145–159.

Nilsson, C.; Berggren, K. 2000. Alterations of riparian ecosystems caused by river regulation. Bioscience. 50: 783–792.

Noss, R.F.; LaRoe, E.T., III; Scott, J.M. 1995. Endangered ecosystems of the United States: A preliminary assessment of loss and degradation. Biological Report 28. Washington, DC: U.S. Department of the Interior, National Biological Service.

Nowak R.S.; Ellsworth, D.S.; Smith, S.D. 2004. Functional responses of plants to elevated atmospheric CO_2—Do photosynthetic and productivity data from FACE experiments support early predictions? New Phytologist. 162: 253–280.

Otsyina, R.M.; McKell, C.M.; Malecheck, J.M.; Van Epps, G.A. 1984. Potential of *Atriplex* and other chenopod shrubs for increasing range productivity and fall and winter grazing. In: Tiedemann, A.R.; McArthur, E.D.; Stutz, H.C.; [et al.], comps. Proceedings—Symposium on the biology of *Atriplex* and related chenopods; 1983 May 2-6; Provo, UT. Gen. Tech. Rep. INT-172. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station: 215–219.

Parmesan, C. 2006. Ecological and evolutionary responses to recent climate change. Annual Review of Ecology, Evolution, and Systematics. 37: 637–669.

Parry, O.F.; Canziani, J.P.; Palutikof, P.J.; [et al.], eds. 2007. Climate change 2007: Impacts, adaptation, and vulnerability; contribution of working group II to the fourth assessment report of the Intergovernmental Panel on Climate Change. Cambridge, UK: Cambridge University Press.

Paruelo, J.M.; Lauenroth, W.K. 1996. Relative abundance of plant functional types in grasslands and shrublands of North America. Ecological Applications. 6: 1212–1224.

Patten, D.T. 1998. Riparian ecosystems of semi-arid North America: Diversity and human impacts. Wetlands. 18: 498–512.

Pauli, H.; Gottfried, M.; Grabherr, G. 2003. Effects of climate change on the alpine and nival vegetation of the Alps. Journal of Mountain Ecology. 7: 9–12.

Pendleton, R.L.; Pendleton, B.K.; Harper, K.T. 1989. Breeding systems of woody plant species in Utah. In: Wallace, A.; McArthur, E.D.; Haferkamp, M.R., comps. Proceedings—Symposium on shrub ecophysiology and biotechnology; 1987 June 30-July 2; Logan, UT. Gen. Tech. Rep. INT-256. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 5–22.

Pendleton, R.L.; Pendleton, B.K.; Meyer, S.E.; [et al.]. 2015. Blackbrush (*Coleogyne ramosissima* Torr.): State of our knowledge and future challenges. In: Huenneke, L.F.; van Riper, C., III; Hays-Gilpin, K.A., eds. The Colorado Plateau VI: science and management at the landscape scale. Tucson, AZ: University of Arizona Press: 142–155.

Perry, L.G.; Andersen, D.C.; Reynolds, L.V.; [et al.]. 2012. Vulnerability of riparian ecosystems to elevated CO_2 and climate change in arid and semiarid western North America. Global Change Biology. 18: 821–842.

Pickford, G.D. 1932. The influence of continued heavy grazing and promiscuous burning on spring–fall ranges in Utah. Ecology. 13: 159–171.

Piemeisel, R.L. 1951. Causes affecting change and rate of change in a vegetation of annuals in Idaho. Ecology. 32: 53–72.

Poff, N.L.; Allan, J.D.; Bain, M.B.; [et al.]. 1997. The natural flow regime: A paradigm for river conservation and restoration. Bioscience. 47: 769–784.

Pollock, M.M.; Heim, M.; Werner, D. 2003. Hydrologic and geomorphic effects of beaver dams and their influence on fishes. American Fisheries Society Symposium. 37: 1–21.

Poore, R.E.; Lamanna, C.A.; Ebersole, J.J.; Enquist, B.J. 2009. Controls on radial growth of mountain big sagebrush and implications for climate change. Western North American Naturalist. 69: 556–562.

Potter, A.F. 1902. Diary of Albert F. Potter (Former Associate Chief of Forest Service) July, 1902 to November 22, 1902. 58 p.

Prevedel, D.A.; Johnson, C.M. 2005. Beginnings of range management: Albert F. Potter, first Chief of Grazing, U.S. Forest Service, and a photographic comparison of his 1902 Forest Reserve Survey in Utah with conditions 100 years later. R4-VM 2005-01. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Region. 94 p.

Rau, B.M.; Johnson, D.W.; Blank, R.R.; [et al.]. 2011. Transition from sagebrush steppe to annual grass (*Bromus tectorum*): Influence on belowground carbon and nitrogen. Rangeland Ecology and Management. 64: 139–147.

Reeves, M.C.; Moreno, A.L.; Bagne, K.E.; Running, S.W. 2014. Estimating climate change effects on net primary production of rangelands in the United States. Climatic Change. 126: 429–442.

Rehfeldt, G.E.; Crookston, N.L.; Saenz-Romero, C.; Campbell, E.M. 2012. North American vegetation model for land-use planning in a changing climate: A solution to large classification problems. Ecological Applications. 22: 119–141.

Rehfeldt, G.E.; Crookston, N.L.; Warwell, M.V.; Evans, J.S. 2006. Empirical analyses of plant-climate relationships for the western United States. International Journal of Plant Sciences. 167: 1123–1150.

Richardson, B.A.; Meyer, S.E. 2012. Paleoclimate effects and geographic barriers shape regional population genetic structure of blackbrush (*Coleogyne ramosissima*: Rosaceae). Botany. 90: 293–299.

Richardson, B.A.; Kitchen, S.G.; Pendleton, R.L.; [et al.]. 2014. Adaptive responses reveal contemporary and future ecotypes in a desert shrub. Ecological Applications. 24: 413–427.

Riegel, G.M.; Miller, R.F.; Skinner, C.N.; Smith, S.E. 2006. Northeastern Plateaus Bioregion. In: Sugihara, N.G.; van Wagtendonk, J.W.; Fites-Kaufmann, J.; [et al.], eds. Fire in California's ecosystems. Berkeley, CA: University of California Press: 225–263.

Rivera, S.; Shultz, L; Hernandez, A.J.; Ramsey, R.D. 2011. GIS ordination approach to model distribution of shrub species in northern Utah. Natural Resources and Environmental Issues. 17: 1–12.

Robertson, J.H.; Kennedy, P.B. 1954. Half-century changes on northern Nevada ranges. Journal of Range Management. 7: 117–121.

Romme, W. H.; Allen, C.D.; Bailey, J.D.; [et al.]. 2009. Historical and modern disturbance regimes, stand structures, and landscape dynamics in piñon-juniper vegetation of the western United States. Rangeland Ecology and Management. 62: 203–222.

Romo, J.T.; Young, J.A. 2002. Temperature profiles and the effects of field environments on germination of silver sagebrush. Native Plants Journal. 3: 5–13.

Rosell, F.; Bozser, O.; Collen, P.; Parker, H. 2005. Ecological impact of beavers *Castor fiber* and *Castor canadensis* and their ability to modify ecosystems. Mammal Review. 35: 248–276.

Rowland, M.M.; Suring, L.H.; Leu, M.; [et al.]. 2011. Sagebrush-associated species of conservation concern. In: Hanser, S.E.; Leu, M.; Knick, S.T.; Aldridge, C.L., eds. Sagebrush ecosystem conservation and management: ecoregional assessment tools and models for the Wyoming Basins. Lawrence, KS: Allen Press: 46–68.

Sada, D. 2008. Great Basin riparian and aquatic ecosystems. In: Chambers, J.C.; Devoe, N.; Evenden, A., eds. Collaborative management and research in the Great Basin—Examining the issues and developing a framework for action. Gen. Tech. Rep. RMRS-GTR-204. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 49–54.

Salo, C. 2005. Red brome (*Bromus rubens* subsp. *madritensis*) in North America: Possible modes for early introductions, subsequent spread. Biological Invasions. 7: 165–180.

Sanderson, S.C.; McArthur, E.D.; Stutz, H.C. 1989. A relationship between polyploidy and habitat in western shrub species. In: Wallace, A.; McArthur, E.D.; Haferkamp, M.R., comps. Proceedings on shrub ecophysiology and biotechnology. Gen. Tech. Rep. INT-256. Ogden, UT: U.S. Department of Agriculture, Forest Service: 23–30.

Schlaepfer, D.R.; Lauenroth, W.K; Bradford, J.B. 2012. Effects of ecohydrological variables on current and future ranges, local suitability patterns, and model accuracy in big sagebrush. Ecohydrology. 35: 374–384.

Schlaepfer, D.R.; Lauenroth, W.K.; Bradford, J.B. 2014. Modeling regeneration responses of big sagebrush (*Artemisia tridentata*) to abiotic conditions. Ecological Modeling. 286: 66–77.

Schultz, B.; McAdoo, K. 2002. Common sagebrush in Nevada. Special Publication SP-02-02. Reno, NV: University of Nevada, Cooperative Extension. 9 p.

Schwinning, S.; Belnap, J.; Bowling, D.R.; Ehleringer, J.R. 2008. Sensitivity of the Colorado Plateau to change: Climate, ecosystems, and society. Ecology and Society. 13: 28.

Schwinning, S., Starr, B.I.; Wojcik, N.J.; [et al.]. Sanford. 2005. Effects of nitrogen deposition on an arid grassland in the Colorado plateau cold desert. Rangeland Ecology and Management. 58: 565–574.

Scott, M.L.; Auble, G.T.; Friedman, J.M. 1997. Flood dependency of cottonwood establishment along the Missouri River, Montana, USA. Ecological Applications. 7: 677–690.

Scott M.L.; Friedman, J.M.; Auble, G.T. 1996. Fluvial process and the establishment of bottomland trees. Geomorphology. 14: 327–339.

Shiflet, T.N., ed. 1994. Rangeland cover types of the United States. Denver, CO: Society for Range Management.

Shultz, L.M. 2006. *Artemisia tripartita* subsp. *tripartita*. In: Flora of North America. Vol. 19, 20, and 21. New York, NY: Flora of North America.

Siegel, R.S.; Brock, J.H. 1990. Germination requirements of key Southwestern woody riparian species. Desert Plants. 10: 3–8.

Simonin, K.A. 2000. *Quercus gambelii*. In: Fire Effects Information System. Missoula, MT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. http://www.fs.fed.us/database/feis/plants/tree/quegam/all.html [Accessed March 30, 2016].

Smith, A.D.; Beale, D.M. 1980. Pronghorn antelope in Utah: Some research and observations. Publication No. 80-13. Salt Lake City, UT: Utah Division of Wildlife Resources. 88 p.

Smith, D.M.; Finch, D.M. 2016. Riparian trees and aridland streams of the southwestern United States: An assessment of the past, present, and future. Journal of Arid Environments. 135: 120–131.

Smith, J.K.; Fischer, W.C. 1997. Fire ecology of the forest habitat types of northern Idaho. Gen. Tech. Rep. INT-363. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Region. 142 p.

Smith, S.D.; Strain, B.R.; Sharkey, T.D. 1987. Effects of CO_2 enrichment on four Great Basin grasses. Functional Ecology. 1: 139–143.

Steinberg, P.D. 2002. *Artemisia arbuscula*. In: Fire Effects Information System. Missoula, MT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. http://www.fs.fed.us/database/feis/ [Accessed March 30, 2016].

Stella, J.C.; Hayden, M.K.; Battles, J.J.; [et al.]. 2011. The role of abandoned channels as refugia for sustaining pioneer riparian forest ecosystems. Ecosystems. 14: 776–790.

Still, S.M.; Richardson, B.A. 2015. Protections of contemporary and future climate niche for Wyoming big sagebrush (*Artemisia tridentata* subsp. *wyomingensis*): A guide for restoration. Natural Areas Journal. 35: 30–43.

Stromberg, J. 1993. Instream flow models for mixed deciduous riparian vegetation within a semiarid region. Regulated Rivers. 8: 225–235.

Stromberg, J.C.; Lite, S.J.; Dixon, M.D. 2010. Effects of stream flow patterns on riparian vegetation of a semiarid river: Implications for a changing climate. River Research and Applications. 26: 712–729.

Stromberg, J.C.; Patten, D.T. 1995. Instream flow and cottonwood growth in the eastern Sierra Nevada of California USA. Regulated Rivers. 12: 1–12.

Summers, H.A.; Smith, B.N.; Hansen, L.D. 2009. Comparison of respiratory and growth characteristics of two co-occurring shrubs from a cold desert, *Coleogyne ramosissima* (blackbrush) and *Atriplex confertifolia* (shadscale). Journal of Arid Environments. 73: 1–6.

Tausch, R.J.; Hood, S. 2007. Pinyon/juniper woodlands. In: Hood, S.M.; Miller, M., eds. Fire ecology and management of the major ecosystems of southern Utah. Gen. Tech. Rep. RMRS-GTR-202. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 57–71.

Tollefson, J.E. 2006. *Acer grandidentatum*. In: Fire Effects Information System. Missoula, MT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. http://www.fs.fed.us/database/feis/plants/tree/acegra/all.html [Accessed March 30, 2016].

Tueller, P.T.; Eckert, R.E., Jr. 1987. Big sagebrush (*Artemisia tridentata vaseyana*) and longleaf snowberry (*Symphoricarpos oreophilus*) plant associations in northeastern Nevada. The Great Basin Naturalist. 47: 117–131.

Ulev, E.D. 2005. *Artemisia pygmaea*. In: Fire Effects Information System. Missoula, MT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. http://www.fs.fed.us/database/feis/ [Accessed March 30, 2016].

U.S. Environmental Protection Agency [EPA]. 1998. Climate change and Idaho. EPA 236-F-98-007f. Washington, DC: Environmental Protection Agency. 4 p.

U.S. Department of Agriculture, Forest Service [USDA FS]. 2013. Inyo National Forest assessment. Bishop, CA: U.S. Department of Agriculture, Forest Service, Inyo National Forest. 229 p.

U.S. Department of Agriculture, Forest Service [USDA FS]. 2017. Plant species and climate profile predictions. Rocky Mountain Research Station Moscow Forestry Sciences Laboratory. http://forest.moscowfsl.wsu.edu/climate/species/index.php [Accessed October 26, 2017].

U.S. Department of Agriculture, Forest Service [USDA FS]. [n.d.]. What is a tall forb community? Rangeland Management and Vegetation Ecology—Botany Program. http://www.fs.fed.us/wildflowers/beauty/Tall_Forb/what.shtml [Accessed October 26, 2017].

Utah State University [USU] Extension. 2016. Range plants of Utah. http://extension.usu.edu/rangeplants/ [Accessed March 14, 2016].

Vale, T.R. 1975. Presettlement vegetation in the sagebrush-grass area of the Intermountain West. Journal of Range Management. 28: 22–36.

Welch, B.L. 1989. Nutritive value of shrubs. In: McKell, C.M., ed. The biology and utilization of shrubs. San Diego, CA: Academic Press, Inc.: 405–424.

Wellner, C.A. 1970. Fire history in the Northern Rocky Mountains. In: Symposium, the role of fire in the Intermountain West. October 27-29; Missoula, MT. Missoula, MT: University of Montana, School of Forestry: 42–64.

Welsh, S.L.; Atwood, N.D.; Goodrich, S.; Higgins, L.C., eds. 2008. A Utah flora, fourth edition, revised. 2004-2008 summary monograph. Provo, UT: Brigham Young University. 1019 p.

West, N.E. 1983. Intermountain salt desert shrubland. In: West, N.E., ed. Temperate deserts and semi-deserts. Amsterdam: Elsevier Scientific Publishing Company.

West, N.E. 1994. Effects of fire on salt-desert shrub rangelands. In: Monsen, S.B.; Kitchen, S.G., comps. Proceedings—Ecology and management of annual rangelands; 1992 May 18-22; Boise, ID. Gen. Tech. Rep. INT-GTR-313. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 71–74.

West, N.E.; Tausch, R.J.; Rea, K.H.; Tueller, P.T. 1978. Taxonomic determination, distribution, and ecological indicator values of sagebrush within the pinyon-juniper woodlands of the Great Basin. Journal of Range Management. 31: 87–92.

West, N.E.; Yorks, T.P. 2002. Vegetation responses following wildfire on grazed and ungrazed sagebrush semi-desert. Journal of Range Management. 55: 171–181.

Williams, J.W.; Jackson, S.T. 2007. Novel climates, no-analog communities, and ecological surprises. Frontiers in Ecology and the Environment. 5: 475–482.

Winward, A.H. 1980. Taxonomy and ecology of sagebrush in Oregon. Station Bulletin 642. Corvallis, OR: Oregon State University, Agricultural Experiment Station. 15 p.

Winward, A.H. 2004. Sagebrush of Colorado: Taxonomy, distribution, ecology, and management. Denver, CO: Colorado Division of Wildlife, Department of Natural Resources. 46 p.

Winward, A.H.; Tisdale, E.W. 1977. Taxonomy of the *Artemisia tridentata* complex in Idaho. Bulletin Number 19. Moscow, ID: University of Idaho, College of Forestry, Wildlife and Range Sciences, Forest, Wildlife and Range Experiment Station. 15 p.

Wisniewski, J.; Dixon, R.K.; Kinsman, J.D.; [et al.]. 1993. Carbon dioxide sequestration in terrestrial ecosystems. Climate Research. 3: 1–5.

Yansa, C.H. 2006. The timing and nature of late Quaternary vegetation changes in the northern Great Plains, USA and Canada: A reassessment of spruce phase. Quaternary Science Reviews. 25: 263–281.

Young, R.P. 1983. Fire as a vegetation management tool in rangelands of the Intermountain region. In: Monsen, S.B., Shaw, N., comps. Managing Intermountain rangelands—Improvement of range and wildlife habitats: Proceedings of symposia; 1981 September 15-17; Twin Falls, ID; 1982 June 22-24; Elko, NV. Gen. Tech. Rep. INT-157. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station: 18–31.

Young, J.A.; Clements, C.D. 2003. Germination of seeds of Fremont cottonwood. Journal of Range Management. 56: 660–664.

Youngblood, A.P. 1980. Vegetation analysis for the Cache Creek EIS. Report to the Bridger-Teton National Forest. 30 p.

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