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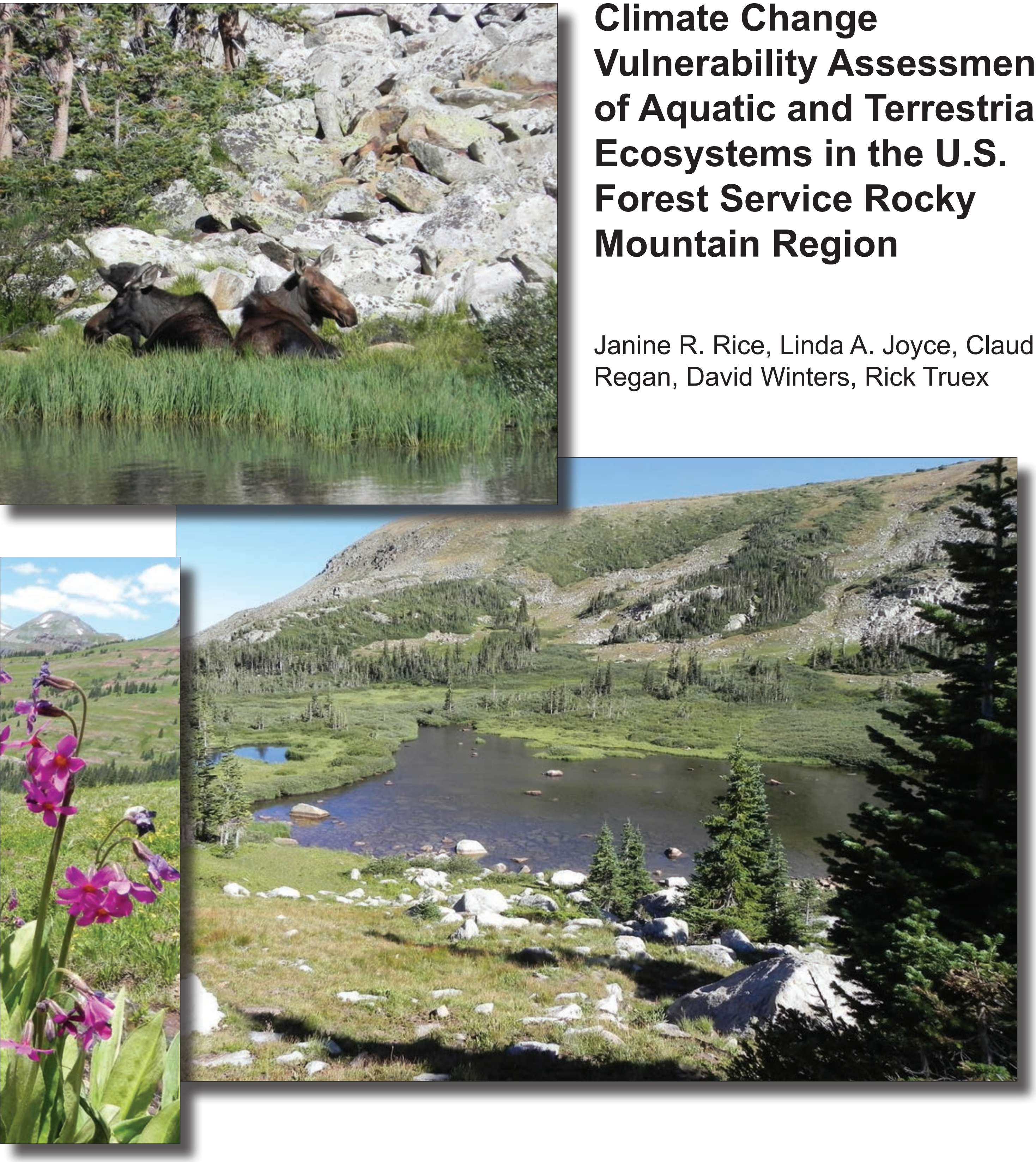
Climate Change Vulnerability Assessment of Aquatic and Terrestrial Ecosystems in the U.S. Forest Service Rocky Mountain Region

Janine R. Rice, Linda A. Joyce, Claudia Regan, David Winters, Rick Truex



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Abstract

Vulnerability assessments are important tools to assist in climate adaptation planning. Six priority ecosystems were identified in the USDA Forest Service, Rocky Mountain Region: alpine turf and dwarf-shrubland; aquatic, riparian, and wetland ecosystems in glaciated valleys; subalpine spruce-fir; low-gradient mountain stream reaches; ponderosa pine; and Great Plains streams and riparian areas. Vulnerability to nonclimate and climate stressors for these priority ecosystems is assessed. Criteria used to assess vulnerability include ecosystem traits related to the sensitivity and adaptive capacity of the ecosystem. We engaged scientists through an expert review to vet the vulnerability rankings and confidence in the assessment. Aquatic ecosystems were the most vulnerable priority ecosystem, and alpine ecosystems had higher vulnerability than lower elevation terrestrial ecosystems. The narrative for each priority ecosystem describes the nature of the vulnerability to climate change.

Keywords: low-gradient mountain stream; subalpine spruce-fir; aquatic, riparian, and wetland ecosystems in glaciated valleys; alpine turf and dwarf-shrubland; Great Plains stream reach; ponderosa pine

Cover photos by Robert Rice.

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Chapter 1. Introduction

Janine R. Rice and Linda A. Joyce

Assessing Vulnerability

One of the first steps in the process of adapting to climate change is assessing the vulnerability of natural ecosystems and species to climate change (National Research Council 2011; USDA FS 2011). Conducting vulnerability assessments and developing adaptation options are a part of the Forest Service, U.S. Department of Agriculture (USFS) Roadmap to Climate Change (USDA FS 2011). Vulnerability of a natural resource can be defined as a function of the character, magnitude, and rate of climate variation to which the natural resource is exposed, and its sensitivity, resilience, and adaptive capacity (Glick et al. 2011). Although this definition offers the conceptual components, no standard methodological approach to assessing vulnerability currently exists (USGCRP 2011).

Assessing vulnerability in a consistent and transparent manner is critical where public review of land management planning occurs and where adaptation practices may be implemented with partners across ownership boundaries or within a larger region (USDA FS 2011). Vulnerability assessments in natural resource management have focused on species, habitats, ecosystems, watersheds, or ecosystem processes, or a combination thereof (Bagne et al. 2011; Case and Lawler 2011; Furniss et al. 2013; Galbraith and Price 2011; McCarthy and Enquist 2011; Rice et al. 2012; Swanston et al. 2011, 2016; Young et al. 2012). Various biological and ecological traits and processes have been used in these assessment tools to evaluate exposure, sensitivity, and adaptive capacity. This variety of approaches can make comparisons across assessments within a geographic area difficult. The legacy of past human activities and current underlying ecological stresses, often not considered, can result in a great variation in the status of habitats, influencing adaptive capacity and vulnerability within a geographic region. Many of the methods have focused strictly on the physical or ecological vulnerabilities and have not integrated social vulnerability into the natural resource vulnerability assessment. Although the objective of vulnerability assessments is typically stated as assisting managers in developing adaptation options, many of the approaches for assessing vulnerability do not help with the difficult conversations about what should be adapted; yet conversations are needed among scientists, resource managers, and decisionmakers to identify and implement adaptation actions (Joyce and Millar 2014; Yuen et al. 2013). There is a need for a structured vulnerability assessment approach that can bring together the current literature on climate change vulnerability, engage experts in vetting the scientific information available to assess vulnerability, and initiate the dialogue with managers on what is vulnerable and how management may address these vulnerabilities.

Assessing Vulnerability of Priority Ecosystems in the U.S. Forest Service Rocky Mountain Region

Incorporating ecosystem processes as well as species and habitat considerations into vulnerability assessments can begin the process of identifying adaptation options to maintain and enhance ecosystem services under climate change. In 2008, the Manomet Center for Conservation Sciences and the Massachusetts Division of Fisheries and Wildlife, with funding from the Wildlife Conservation Society, began a cooperative effort to determine: (1) vulnerability to climate change of Massachusetts fish and wildlife habitats and species in greatest need of conservation, and (2) potential adaptations of conservation and management practices (Manomet Center for Conservation Sciences and Massachusetts Division of Fisheries and Wildlife 2010). This approach was further developed when the Manomet Center for Conservation Sciences and the National Wildlife Federation collaborated with the Northeastern Association of Fish and Wildlife Agencies and Federal agencies. Through this effort, a consistent and transparent structure was developed to identify vulnerabilities of key habitats in the northeastern United States (Galbraith 2011; Manomet Center for Conservation Sciences and National Wildlife Federation 2013). The framework also included current stressors that are a focus of resource management. In addition to these two applications in the northeastern United States, this assessment framework has been used for assessments of habitat and species in the Gunnison Basin in Colorado (Neely et al. 2011) and the Badlands National Park in South Dakota (Amberg et al. 2012).

We build on the work of this northeastern multi-institution collaboration and recent applications to extend the framework to consistently assess vulnerability of select aquatic and terrestrial ecosystems across a large geographic region. In addition, we engaged subject-matter experts to evaluate the vulnerabilities, and managers to initiate consideration of adaptation actions. This collaboration between the USFS Rocky Mountain Research Station (hereafter RMRS) and the USFS Rocky Mountain Region, as with the northeastern collaboration, has the goal of increasing the capabilities of resource agencies to manage natural resources under climate change, and to increase the dialogue among resource managers across an ecologically varied region.

Regional Focus and Priority Ecosystems of the Rocky Mountain Region

We assess ecosystem vulnerability at the geographic scale of the USFS Rocky Mountain Region (fig. 1.1). Within this Region, 17 national forests and 7 national grasslands have been administratively combined into 11 units. These 11 administrative units cover 27.8 million ac and include 91 wetland and upland terrestrial ecological systems (Comer et al. 2003). These public lands are managed for multiple resources: timber, water, range, recreation, and more (USDA FS n.d.). Elevations range from about 2,500 ft on the central mixed-grass prairie to more than 14,000 ft in the alpine tundra of the Rocky Mountains. The mountainous areas receive the most precipitation, and the driest areas are on the Great Plains (Nebraska, Kansas) and the Intermountain Basins in Wyoming. Mean annual temperatures are coolest in the highest elevations, well below freezing for some months, and warmest in the lower elevations of the Great Plains.

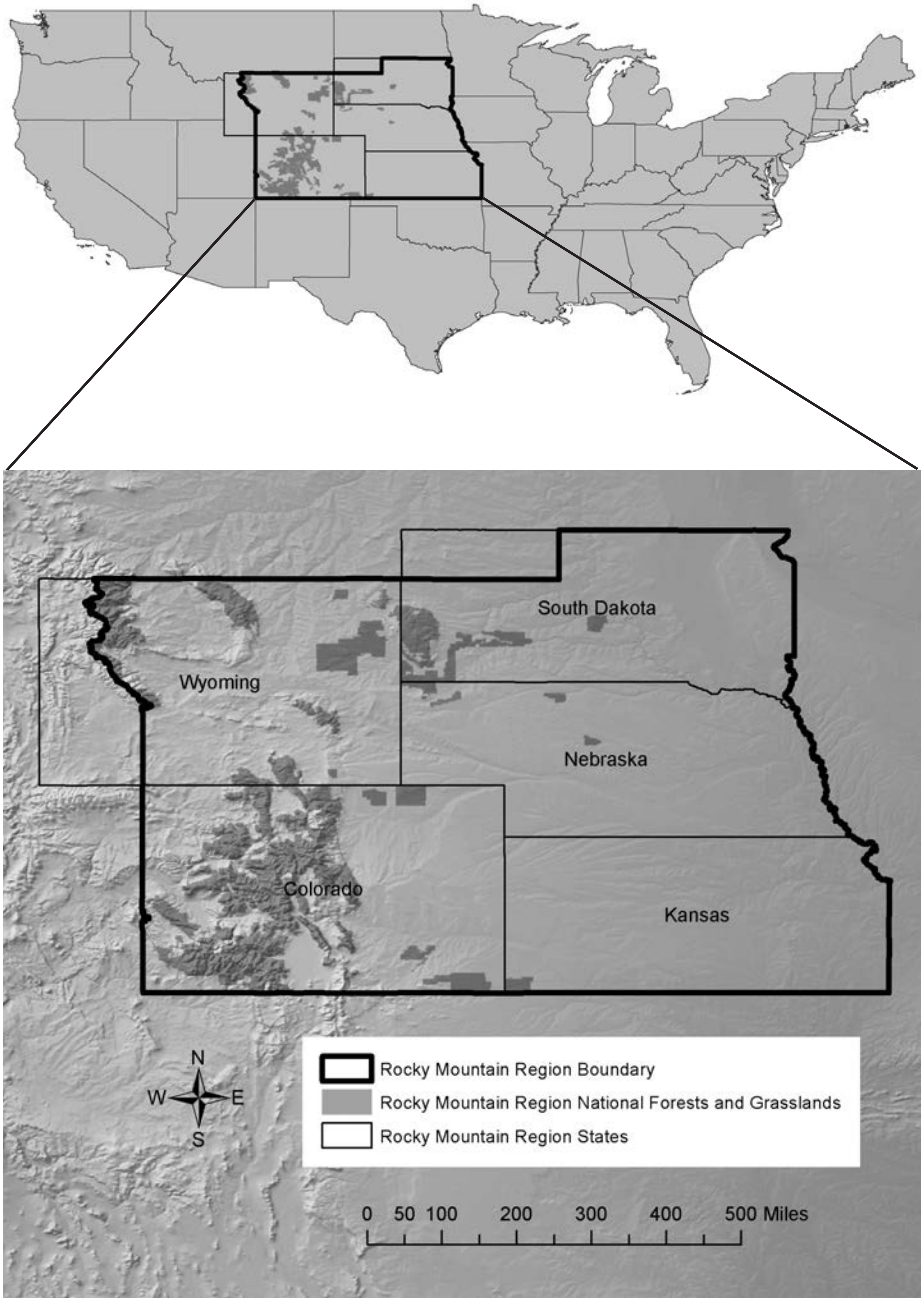

Figure 1.1—USFS Rocky Mountain Region study area.

National Forest System staff in the Rocky Mountain Region identified six ecosystems as priority ecosystems for vulnerability assessment. Three were aquatic: aquatic, riparian, and wetland ecosystems in glaciated valleys; low-gradient mountain stream reaches; and Great Plains streams and riparian areas. Three of these ecosystems were terrestrial: alpine turf and dwarf-shrubland, subalpine spruce-fir, and ponderosa pine. These six ecosystems were selected to maximize information gained from such an effort to disseminate to the 11 units in the Rocky Mountain Region (table 1.1).

Vulnerability Assessment Framework, Expert Review, Manager Workshop

The vulnerability framework has two major components, one related to nonclimate stressors and one related to climate stressors (Appendix A). Within the nonclimate

Table 1.1—Six priority ecosystems of the Rocky Mountain Climate Change Vulnerability Assessment, identified by National Forest System Regional Staff.

Priority ecosystem	Ecological types	Extent within the Rocky Mountain Region
Aquatic		
Low-gradient mountain stream reaches	Mountain stream reaches with slope of less than 2 percent in these river basins: Colorado, Rio Grande, Arkansas, Lower Missouri, Upper Missouri	3,500 miles spanning alpine to foothills in national forests
Aquatic, riparian, wetland ecosystems in glaciated valleys	Lakes, ponds, and associated riparian and wetland ecosystems	A small fraction of area in glaciated valleys of Colorado and Wyoming
Great Plains streams and riparian areas	Perennial and intermittent streams whose headwaters are in the Great Plains and their associated riparian areas	100 stream miles across the Great Plains landscape within national forests
Terrestrial		
Alpine turf and dwarf-shrubland	Alpine dry tundra; dwarf-shrubland	1 million acres on high mountain peaks in Colorado and Wyoming
Subalpine spruce-fir	Subalpine spruce-fir forest; dry subalpine spruce-fir forest and woodlands	4.5 million acres across the subalpine zone in Colorado and Wyoming; 119,000 acres of dry subalpine spruce-fir forest and woodland type primarily in northwestern Wyoming
Ponderosa pine	Ponderosa pine woodlands; ponderosa pine savannas	Woodlands - 4.8 million ac in the Rocky Mountain Region and savannas; 1.6 million ac in the Colorado Front Range and in southwestern Colorado

component, four criteria were used to define factors of vulnerability: current status of ecosystem extent, human influences on the ecosystem, intrinsic resilience of the ecosystem to nonclimate stressors, and future trends of nonclimate stressors (table 1.2). The rationale for ranking these four criteria is given in Appendix B.

Within the climate change component, seven criteria define factors of vulnerability to climate change and the interaction with nonclimate stressors (table 1.3). These seven criteria capture the vulnerability of individual species, sensitivity of ecosystem dynamics such as dependence of the ecosystem on the hydrological cycle, adaptive capacity of the ecosystem, potential for management to mitigate the effects of climate and nonclimate stressors, and the interaction between climate change and nonclimate stressors. The initial ranking of each criterion is based on the synthesis of relevant scientific literature and an understanding of the priority ecosystems within the geographic area of the Rocky Mountain Region. The literature review considers the factors of exposure, sensitivity, and adaptive capacity, and how climate change may affect hydrological function, disturbance regimes, and aquatic, riparian, and upland vegetation species. The rationale for ranking the seven criteria is given in Appendix B.

The vulnerability of each criterion is ranked as high, moderate, or low based on a rationale for scoring each criterion (Appendix B). The initial ranking of vulnerability was done by the authors and described in the narrative for that ecosystem within the draft assessment. Each narrative underwent a scientific review by experts with field research expertise in the Rocky Mountain Region to: (1) critique the narrative content and rationale for ranking each criterion, as well as the overall vulnerability ranking

Table 1.2—Four criteria are used to assess the vulnerability of these priority ecosystems to nonclimate stressors. Each criterion is used to rank the vulnerability (high, moderate, low) of the ecosystem to a particular nonclimate stressor, using scientific literature and expert elicitation (see appendices B and C for rationale).

Criterion	Component
Current status of ecosystem extent	Amount and distribution of the ecosystem
Human influences on ecosystem	Degree of ecosystem effects of past human activities and stressors associated with ongoing human influences.
Intrinsic resilience of ecosystem to nonclimate stressors	Degree of resilience to nonclimate stressors such as human influences
Future trends of nonclimate stressors	Assessment of likely trends of nonclimate stressors to determine the future degree of influence on the ecosystem

assigned by the author team; and (2) assign a confidence ranking that gauged the quality and amount of information available and whether there was sufficient agreement in the literature to assess vulnerability. Four to five experts with academic and natural resource backgrounds reviewed each of the six narratives (table 1.1). Two experts on terrestrial ecosystems reviewed all three terrestrial ecosystem narratives. Two experts on aquatic ecosystems reviewed all three aquatic ecosystem narratives. The remaining experts reviewed only one priority ecosystem narrative each. If experts disagreed with the assessment, they could change the rank, using the same system as the authors. Vulnerability to nonclimate stressors (four criteria) or to climate change (seven criteria) was derived by averaging the expert reviewers' respective scores. Mean confidence was the average of all reviewers' confidence. The overall regional vulnerability ranking for the priority ecosystem was determined by averaging the mean nonclimate and climate vulnerability scores (corresponding to a high, moderate, or low rank) into a final ranking of very high, high, moderate, low, or very low (appendices B, C).

Table 1.3—Seven criteria are used to assess the vulnerability of these priority ecosystems to climate stressors. Each criterion is used to rank the vulnerability (high, moderate, low) of the ecosystem to a particular climate stress, using scientific literature and expert elicitation (see appendices B and C for rationale for scoring).

Criterion	Component
Ecosystem shift capacity	Current range distribution; elevational ranges; connectivity of the ecosystem
Vulnerability of cold-adapted, foundation, or keystone species to climate change	Foundation species with their high biomass; keystone species that have low biomass yet exert strong effects on the structures of their communities; cold-adapted species
Sensitivity to extreme climatic events	Extreme events relevant to the ecosystem, such as drought, flooding, heat events, freeze and wind events
Intrinsic adaptive capacity	Degree of resilience to climate stressors; factors such as biotic characteristics, physical processes, and recovery times to disturbances
Dependence on specific hydrological regime	Specific and relatively narrow hydrological conditions
Likelihood of managing or alleviating climate change effects	Assessment of the feasibility of management to mitigate potential effects of climate change
Potential for climate change to exacerbate effects of nonclimate stressors, or vice versa	Potential for changes in climate to exacerbate effects on current or future nonclimate stressors, or for nonclimate stressors to exacerbate effects of climate change

To facilitate a regional dialogue about vulnerability and to begin development of adaptation options, a 1-day manager workshop was held. The workshop was assisted by additional USFS national forest regional experts. More than 20 Rocky Mountain Region resource specialists and planning, public relations, and forest land managers attended the workshop; 9 of the 11 units in the Rocky Mountain Region participated. The workshop first familiarized resource managers with the information in the six regional vulnerability narratives, scoring, and expert review. Then workshop participants were taken through exercises that (1) characterized ecosystem services within their individual national forests, (2) used the narratives to characterize vulnerability of ecosystem services, and (3) identified potential management intervention points and opportunities for adaptation actions.

After the workshop and revision of the vulnerability assessments, we asked our initial experts to review the chapters again. We incorporated their suggestions for new literature and revised the document. As per RMRS publication policy, the entire report was reviewed by three technical experts who had not previously seen the vulnerability assessment. Their comments and suggestions were included in the revision.

Intended Audience and Potential Uses

We envision that the information in these narratives will inform national forest resource managers as to which processes and components of these priority ecosystems are most sensitive to potential changes in nonclimate and climate stressors. This vulnerability assessment application used a theme-based approach to identify components of vulnerability for select aquatic and terrestrial ecosystems in a spatially nonexplicit manner over a large and spatially varied region. This framework efficiently and consistently assessed vulnerability of these priority aquatic and terrestrial ecosystems, including processes influenced by climate factors as well as by ecological and social stressors. Although these assessments looked at the regional scale, we anticipate that each forest may have more site-specific information on past management and current conditions that will help complete the picture on a local scale.

The need for the information in this vulnerability assessment is exemplified by the assessment process in the new forest planning rule, National Environmental Policy Act decisionmaking processes, project-level planning, and coordination of regionwide conservation projects. In addition, the information can be used to inform resource management strategies, conservation planning, and monitoring plans. The ecosystem focus will provide a means to link vulnerabilities to the provisioning, regulating, supporting, and cultural ecosystem services that land managers consider in their planning process. The transparent synthesis of the scientific literature also provides a platform for initiating dialogue between land managers with a variety of backgrounds and skill sets, as we saw in the manager workshop. It also may help to build support and increase understanding about vulnerability and the capacity to manage natural resources under climate change in the ecologically varied Rocky Mountain Region.

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Chapter 2. Climate Overview for the U.S. Forest Service Rocky Mountain Region

Linda A. Joyce and Janine R. Rice

Introduction

Information about weather and climate is used in natural resource planning and management. Weather is the information about conditions at a specific time and place, such as temperature, precipitation, wind, and humidity. Day-to-day implementation of on-the-ground resource management practices, such as the start of prescribed fire activities, can be made in response to weather conditions. Climate information is the set of characteristics of precipitation, temperature, wind, snowfall, and other measures of weather in a particular place over a long period of time. Management practices, such as the initiation of grazing on grassland, are based on an understanding of when green-up occurs over the long term. The actual start of grazing may be adjusted depending on the current weather conditions. Understanding the climate of an area helps managers to identify both the general characteristics and the risks associated with climate conditions, such as flooding, drought, wildfire, and extreme heat or cold events. Understanding these risks assists management decisions, such as determining culvert size to withstand expected floods. Thus, longer-term resource management strategies and plans are developed using an understanding of climate, the average conditions over time. With the potential for climate change, however, our understanding of climate as static is changing. Given the need for long-term planning and management, an understanding of how climate may change in the future is valuable.

The Forest Service, U.S. Department of Agriculture (USFS) Rocky Mountain Region (fig. 1.1) encompasses a wide range of climates, from the semiarid climate of the short-grass prairie to the cold and windy climate of the alpine tundra. The climate of the Region is influenced by the interactions among moist air masses from the Pacific Ocean and the Gulf of Mexico, and cold dry air masses from Canada. These climate forces interact with the complex terrain within the Region. The Rocky Mountains bisect Wyoming and Colorado, forcing the westerly movement of moist air upslope and resulting in higher levels of precipitation on the west side of these mountains. The eastern side of the Rockies lies in a rain shadow and annual precipitation along the Front Range of Colorado can be less than 14 in. In the Great Plains area of the Region, precipitation has a west-east gradient, gradually increasing as the eastern edges of Kansas and Nebraska are approached. In the alpine, solar radiation and wind can modify surface temperatures and snow cover along gradients in topography over short distances (Suding et al. 2015). Snowpack on the north-facing slopes under lodgepole pine (*Pinus contorta* Douglas Ex. Loudon) forests develops over the winter season whereas on the south-facing slopes under ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson) forests, snow accumulation is intermittent. Hinckley et al. (2014) found that north-facing slopes had connected moisture flow to greater soil depths than the drier south-facing forested slopes.

The variety of local climates of the Rocky Mountain Region is reflected in the number and variety of ecological systems: 91 aquatic and upland terrestrial ecological systems (Comer et al. 2003). Seventeen national forests and seven national grasslands are found in the Region. Most of the national forests are located at higher elevations in the Rocky Mountains (fig. 1.1). This chapter provides a brief discussion of the historical climate of the Region and projected changes in climate under two different futures for the Region.

Recent and Historical Climate—State-Level Summaries

We draw from several sources to review the historical climate at the State level, providing a picture of the climate at this larger spatial scale. Typically, the last 30 years of climate data is used to determine the normal climate, the likely average conditions on an annual, monthly, or daily basis. Where observations are available for a longer period of time, we can be more certain about trends in the data. As more observations are made, past records, such as the hottest year or coldest year, are broken. For example, the annual global surface temperature in 2015 exceeded the annual temperature in 2014, which was the previous record for the annual global surface temperature (Blunden and Arndt 2016). For the information that follows, the temporal period of the analysis is reported for reference.

Topography and elevation strongly influence temperature in the Rocky Mountain Region. With elevation, temperatures are generally cooler (fig. 2.1a). Most of the national forests in this Region are found in cooler areas, whereas the national grasslands in these States are found at warmer sites. The southern parts of Kansas are the warmest areas in the Region. Precipitation within the Region ranges from less than 4 in annually to greater than 50 in. Precipitation is lowest in the canyon, mesa, basin, and grassland areas of Colorado and Wyoming. The highest annual precipitation is found in the San Juan Mountains and other upper elevation areas, with much of this precipitation coming as snow.

Over the 20th century, all States in the Rocky Mountain Region observed an increase of at least 1 °F in mean annual temperature (table 2.1). All States experienced the greatest warming in winter with some States also warming in spring or at nighttime. Precipitation is highly variable and no trends were reported in annual precipitation.

Although all States experienced an increase in annual temperatures, the seasonal changes differed by State. The 1.4 °F increase in Wyoming mean annual temperature reflects changes in winter and summer temperatures. Wyoming winter temperatures have increased 1.9 °F above the historical average since 1995 (fig. 2.2). In addition, the frequency of very hot days (maximum temperature above 95 °F) has increased since 2000. Number of days with minimum temperatures above 70 °F has been above the long-term average since 2000 in Wyoming.

South Dakota mean annual temperatures have risen 2 °F over the last 100 years (table 2.1); mean temperatures in the early 21st century are exceeded only by temperatures in the Dust Bowl era of the early 1930s. Winter and spring temperatures have had the greatest influence on annual warming. The warmest winter temperatures in

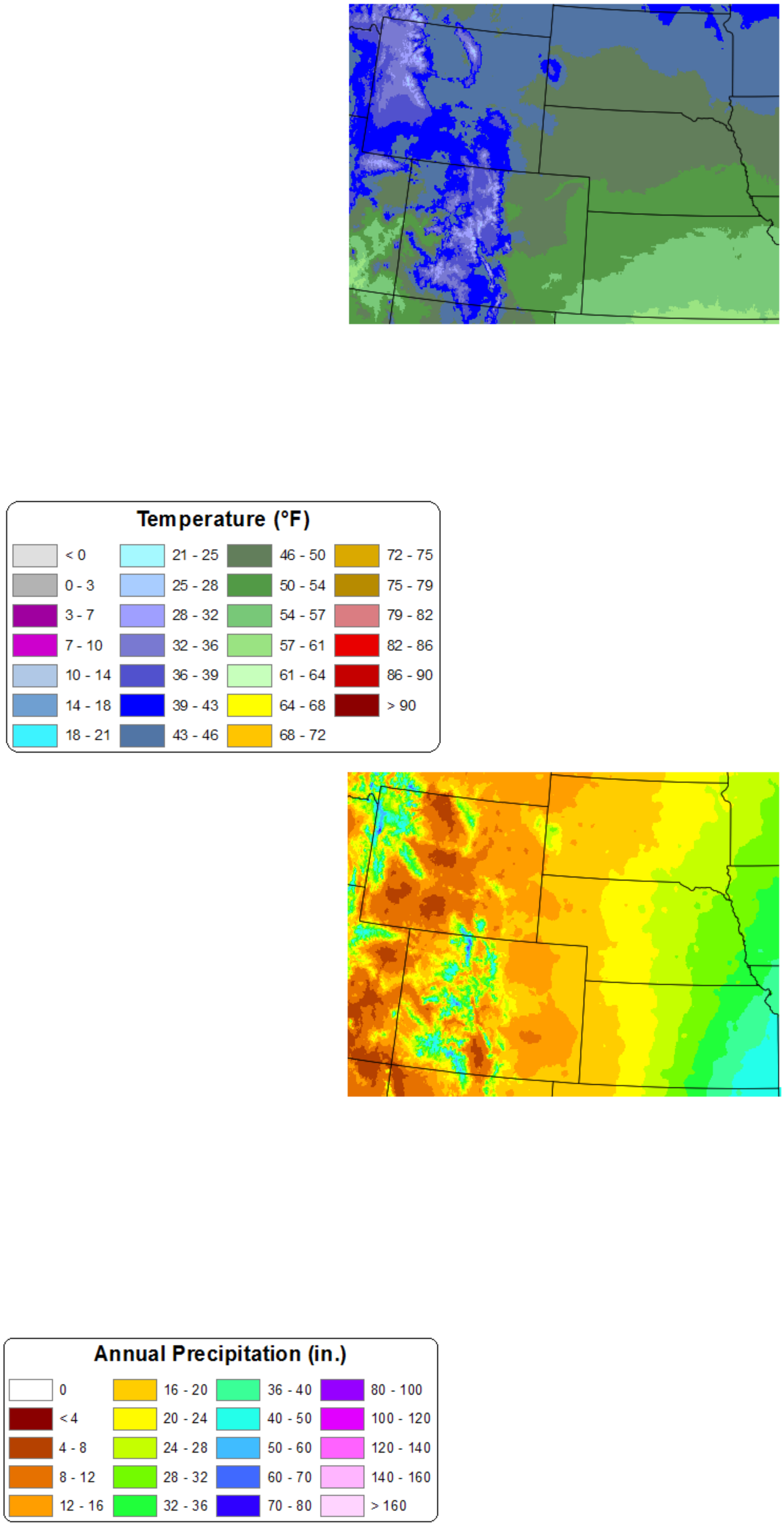


Figure 2.1—Annual mean temperature (°F) (a) and annual precipitation (inches) (b) over the 1980–2010 period. (Data source: Oregon State University 2017. Copyright © 2017.)

Table 2.1—Observed changes in temperature since the early 20th century for States within the Rocky Mountain Region (Frankson et al. 2016a,b,c,d,e).

State	Mean annual change since the early 20th century	Seasonal temperature changes
Colorado	+2°F	Winter warming has been characterized by a much below average occurrence of extremely cold days since 1990.
Kansas	+2°F	Greater warming in the winter and spring than in the summer and fall. The number of very cold nights has been much below average since 1990.
Nebraska	+1°F	Warming has been concentrated during the winter and spring. Winter warming has been characterized by a much below average occurrence of very cold nights since 1990.
South Dakota	+2°F	Warming has been concentrated during the winter and spring and nighttime minimum temperatures increasing about twice as much as daytime maximums.
Wyoming	+1.4°F	The annual temperature increase is most evident in winter warming, which has been characterized by a below average occurrence of very cold days since 2000.

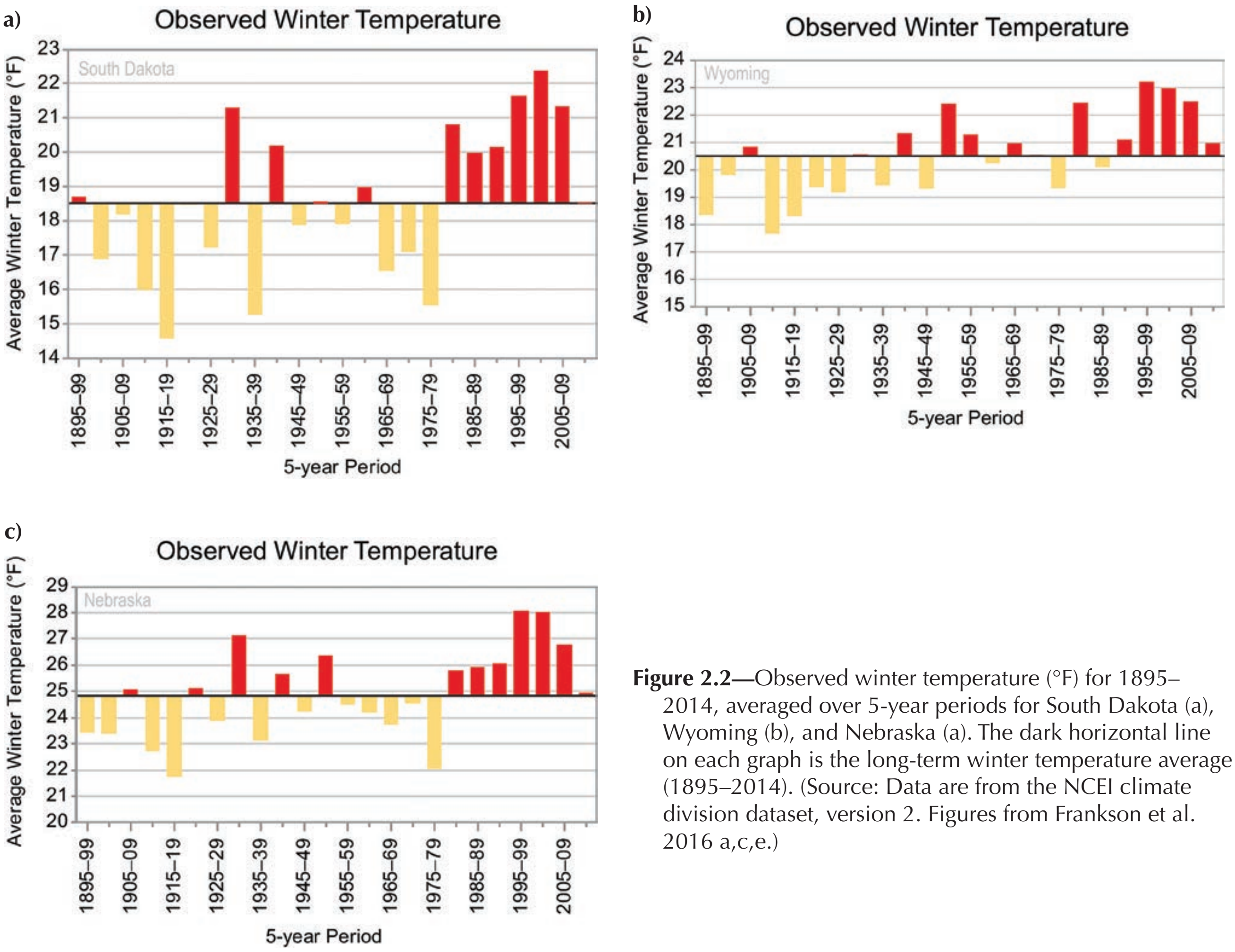


Figure 2.2—Observed winter temperature (°F) for 1895–2014, averaged over 5-year periods for South Dakota (a), Wyoming (b), and Nebraska (a). The dark horizontal line on each graph is the long-term winter temperature average (1895–2014). (Source: Data are from the NCEI climate division dataset, version 2. Figures from Frankson et al. 2016 a,c,e.)

the historical record were reported from 1995 to 2009 (fig. 2.2). The State has also experienced a below-average number of very cold nights (minimum temperature below 0 °F) since 2000. Summers in South Dakota have not warmed. Nighttime temperatures have warmed faster than daytime temperatures in South Dakota. Nebraska mean annual temperatures have risen the least of all States in the Rocky Mountain Region (table 2.1). The recent warming, since the 2000s, has been observed in winter (fig. 2.2) and spring. As with South Dakota, summers have not warmed in Nebraska and the number of days above 95 °F has been below the historical average since the 1960s.

Over the last 100 years, mean annual temperatures have risen 2 °F in both Colorado and Kansas (table 2.1). In both States, spring temperatures have warmed (fig. 2.3). The hottest year on record in Colorado was 2012, when the average temperature rose to 48.3 °F, reflecting the highest average spring and summer temperatures since 2000 (fig. 2.3). The year 2012 was also the hottest year on record in Kansas with an average temperature of 58.2 °F; record high temperatures were recorded during the summer. Spring temperatures have consistently increased in Kansas over the last three decades. No trends were seen in very warm nights (minimum temperature above 75 °F) or extremely hot days (maximum temperature above 100 °F).

No significant trends have been reported for annual precipitation. In all of the Sstates in the Rocky Mountain Region, annual precipitation is variable. Each State has seen multiyear droughts, including the dry years of the Dust Bowl era. Droughts have occurred throughout the geographic area of the Rocky Mountain Region in historical times (Kunkel et al. 2013a,b) and have been recorded in the paleo record (McWethy et al. 2010). Each State has also had multiyear periods of above-average precipitation, including significant flooding events.

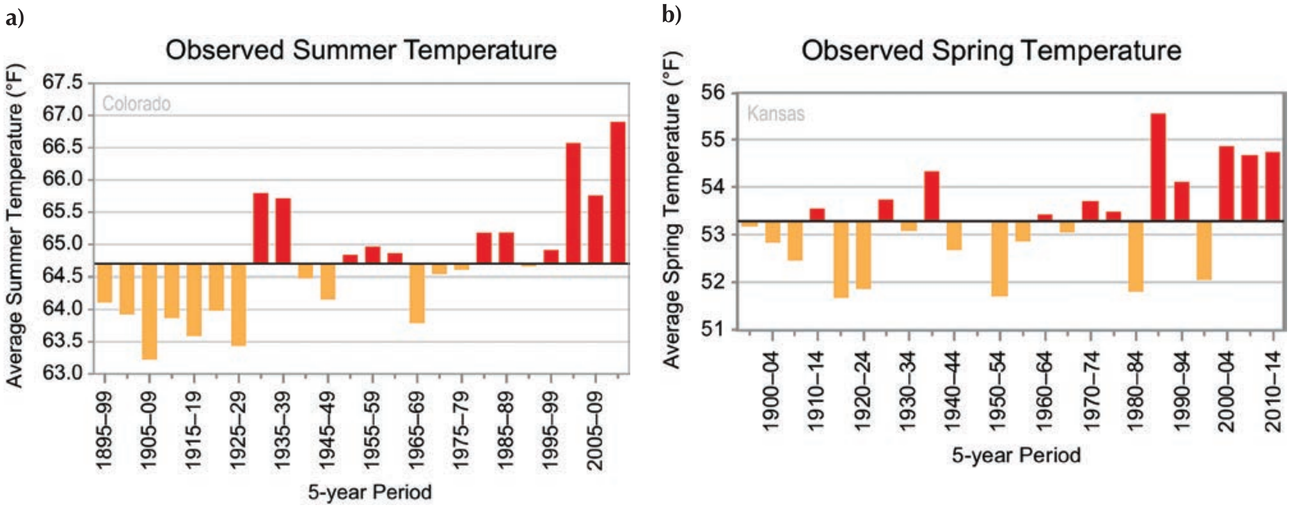


Figure 2.3—Observed spring temperature (°F) for 1895–2014, averaged over 5-year periods for Colorado (a) and Kansas (b). The dark horizontal line is the long-term average (1895–2014) of 53.3 °F (spring) for Kansas and of 42.9 °F (spring) for Colorado. (Source: Data are averages from NCEI›s version 2 climate division dataset. Figures are from Frankson et al. 2016 b,d.)

Historical and Recent Climate at the Scale of a National Forest

Climate is influenced by elevation, aspect, and topography. Consequently, temperature and precipitation patterns within a national forest will differ from the State patterns. Understanding climate depends on long-term monitoring of weather conditions. Weather stations are sparse in remote areas of the Rocky Mountain Region and particularly sparse above 10,000 ft. Most weather stations are established in valleys where people live. Weather stations may have short periods of data collection in cases where the station was established but then closed. Weather stations may also have an intermittent data collection when no data were collected over a period of time. Most weather stations will have missing values due to a variety of factors, including equipment failure. Station climatologists are the best source of information about the robustness of climate data within the State.

Projecting Climate Over the Next 100 Years

Modeling the Climate

Scientists continue to explore how climate is influenced by the physical and chemical dynamics of the atmosphere, the interaction between the atmosphere and the surface of the Earth, and the ocean currents moving heat around the globe. Climate is affected by changes in any one of these processes. Building cities (impervious surfaces) and changing land use from forests to agriculture have a local and regional effect on climate. Changes in the chemical composition of the atmosphere influence the heat absorbed in the atmosphere. Humans contribute trace gases and other components to the atmosphere through land management, energy sources, and industrial processes.

Climate models allow scientists to ask questions about how climate responds to changes in the chemical composition of the atmosphere, land use changes, and other disturbances. Global climate models have been an important part of the Intergovernmental Panel on Climate Change (IPCC) assessments since 1990. Since 1995, atmospheric scientists have been working together to coordinate climate model experiments via the Coupled Model Intercomparison Project (CMIP) (Meehl et al. 2007). This coordination allows for a rigorous comparison of results from different global climate models around the world and improves our understanding of the “range” of possible climate change. In these experiments, scientists use scenarios, a similar set of initial conditions and changes in the chemistry of the atmosphere and in land cover over time. “The goal of working with scenarios is not to predict the future but to better understand uncertainties and alternative futures, in order to consider how robust different decisions or options may be under a wide range of possible futures” (IPCC 2017).

The results of the climate experiments have also been used by hydrologists, ecologists, foresters, and other natural resource scientists to explore the potential effects of climate on their area of research (USDA Forest Service 2012; Walsh et al. 2014). This rich literature has been used to identify the vulnerabilities of the six priority ecosystems discussed in this report. As we draw from this research, it is important to keep in mind that climate scientists continue to learn about climate and to improve the ability of the climate models to capture the climate dynamics. Much of the literature consulted for this report uses climate projections from the IPCC Third and Fourth Assessments.

The types of scenarios used in the IPCC Third and Fourth Assessments differed slightly. In the Third Assessment, the scenarios involved specific assumptions about emissions to the atmosphere, population growth, economic growth, and use of fossil fuel sources and alternative energy (Solomon et al. 2007). These scenarios have been called the SRES scenarios, as they were based on the Special Report on Emissions Scenarios (Nakićenović et al. 2000). Here a suite of models was used to estimate potential future population growth, different economic pathways and their associated energy use, and different land use pathways—and from these different futures, the changes in emissions. Consequently, the projected climates ranged from low emissions (B1) to high emissions (A1FI). For the most recent assessment, the IPCC Fifth Assessment, a new set of scenarios was developed (Stocker et al. 2013). For these scenarios, scientists asked, "What would happen to the climate if different amounts of heat were added to the atmosphere?" These scenarios are called representative concentration pathways (RCPs) (van Vuuren et al. 2011). RCP 2.6 assumes an increase in total radiative forcing (heat) of 2.6 W m^{-2} (Watts per square meter) (0.82 BTU (h^{-1} ft^{-2}) (British thermal unit per hour per square foot) by 2100, whereas RCP 8.5 assumes a much larger increase in the radiative forcing, 8.5 W m^{-2} (2.7 BTU h^{-1} ft^{-2}). Intermediate scenarios include RCP 4.5 and RCP 6.0. With different amounts of energy coming into the atmosphere, the temperature projections differ by scenario. The global model comparison experiments are archived by CMIP. The SRES scenario projections are referred to as "the CMIP3 projections" and the RCP scenario projections as "the CMIP5 projections."

In reviewing the natural resource literature that used the Third Assessment versus the Fourth Assessment projections, the question arises as to how similar these future projections are. The global warming under the SRES and RCP scenarios is compared in table 2.2. This comparison takes into consideration the understanding of climate sensitivity uncertainty, synthesizes the understanding of climate system and carbon-cycle

Table 2.2—Probabilistic estimates of temperature increase above pre-industrial levels based on representative ECS distribution for six SRES scenarios and four RCP scenarios (Rogelj et al. 2012).

	Temperature increase above pre-industrial (°F)	
	2090–2099 period	
Scenario	Median	66-percent range
SRES B1	4.3	3.6–5.6
SRES A1T	5.2	4.5–6.7
SRES B2	5.2	4.3–6.3
SRES A1B	6.1	5.0–7.6
SRES A2	7.0	5.8–8.6
SRES A1FI	8.5	7.0–10.4
RCP 3-PD (2.6)	2.7	2.3–3.4
RCP 4.5	4.3	3.6–5.2
RCP 6	5.2	4.5–6.5
RCP 8.5	8.3	6.8–10.3

behavior, and is constrained by the observed historical warming (Rogelj et al. 2012). The RCP scenarios were not designed to mimic particular SRES scenarios. Temperature increase above pre-industrial temperatures (1850 to 1875 period) ranges from 4.3 to 7.0 °F in the SRES scenarios, and from 2.7 to 8.3 °F in the RCP scenarios (table 2.2). For temperatures projected by individual scenarios, Rogelj et al. (2012) identify similarities between RCP 4.5 and SRES B1, RCP 6 and SRES B2, and RCP 8.5 and SRES A1FI. One could conclude that analyses using scenario B1 might produce resource effects similar to those of RCP 4.5. Similar comparisons could be made with other scenarios (Rogelj et al. 2012). They note that the temporal patterns differ between the SRES and RCP scenarios. Broadly, the scenarios can be interpreted as a range of future climate, with the hotter scenarios being SRES A2, SRES A1FI, and RCP 8.5; a less warm scenario is RCP 3.0.

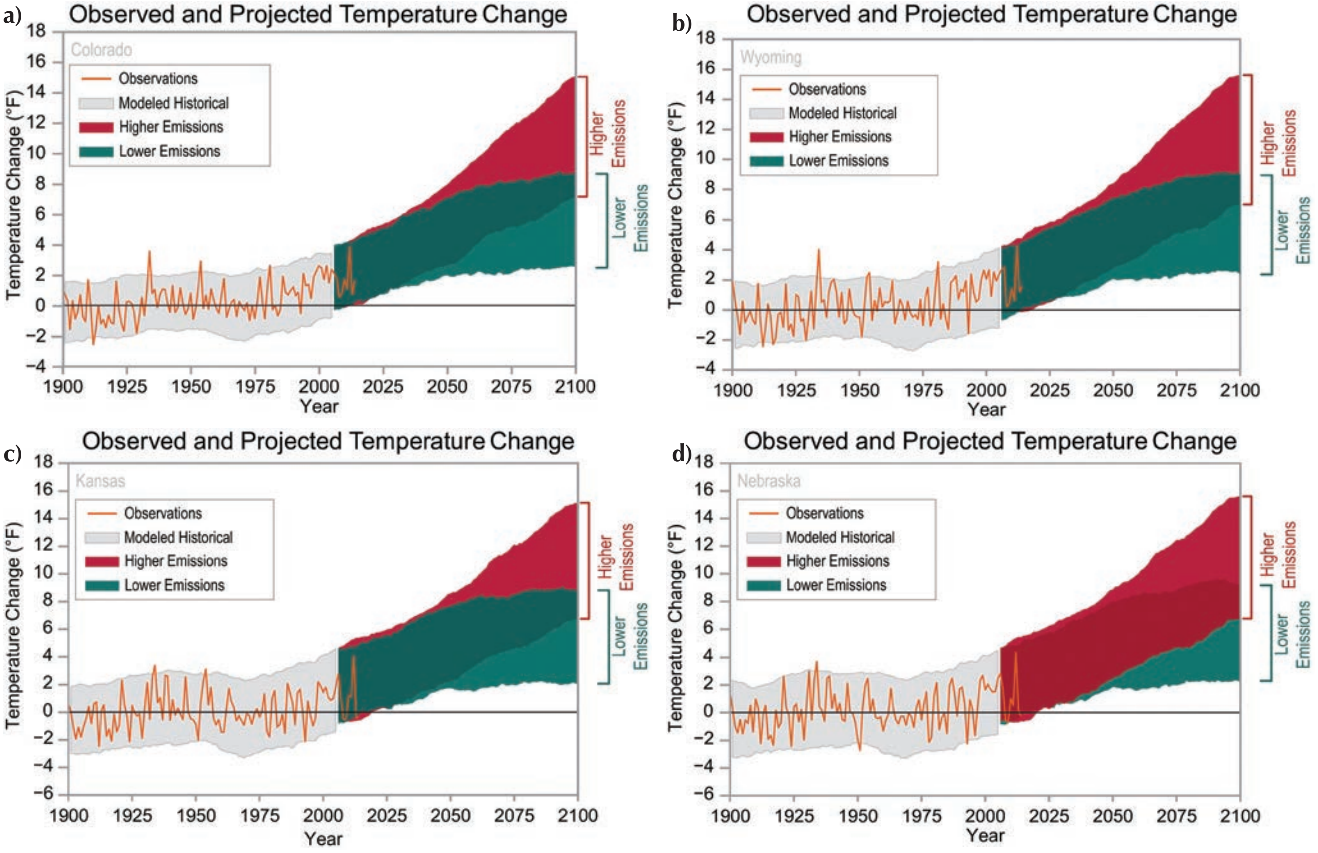
A comparison has also been made of CMIP3 and CMIP5 projections at the scale of the Northern Rocky Mountain Region (Idaho, Montana) and the Intermountain Region (Utah, southern Idaho, Nevada) (Joyce et al. 2017, 2018). Annual temperature and annual precipitation changes from projections using the A1B scenario developed by Littell et al. (2011) were compared with projections from CMIP5 models using RCP 4.5. We concluded that projected changes in temperature for the RCP 4.5 and SRES A1B scenarios used by Littell et al. (2011) were in the same range. Increase in precipitation, however, was slightly greater for the RCP 4.5 scenario than the SRES A1B scenario in each region.

Climate Projections for the Rocky Mountain Region

We draw on climate change projections for each State in the Rocky Mountain Region as summaries in Frankson et al. (2016a,b,c,d,e). These figures show the observed historical climate (yellow line) changes compared to the 1901–1960 observed historical average (fig. 2.4). The observed historical changes vary over time; the warm temperatures of the 1930s and the 1950s are apparent in all States. For most States, the changes since 1980 have exceeded the historical average. The observed historical fluctuations in all States are generally within the banded ranges of the historical modeled changes (gray band).

Two projections through 2100 are shown: a higher emissions scenario (red band) and a low emissions scenario (blue band). All States are projected to have historically unprecedented warming during the 21st century. Differences between the two scenarios are not evident until after mid-century, when the upper range of the higher emissions scenario rises above that of the low emissions scenario. Typically, the coldest years in the low emissions scenario projection are as warm as the hottest years in the historical record. For Kansas, the coldest years are about 2 °F warmer than the long-term average and the hottest years about 11 °F warmer than the hottest year in the historical record. The projections associated with the higher emissions scenario are outside of the historical range by the late 21st century as the range of the projections (red shading) is above the historical annual means.

We explore the changes in winter and spring minimum temperature for the Rocky Mountain Region using an ensemble of 20 climate models from the CMIP5 experiments and the RCP 4.5 scenario (fig. 2.5). The comparison here is between the historical



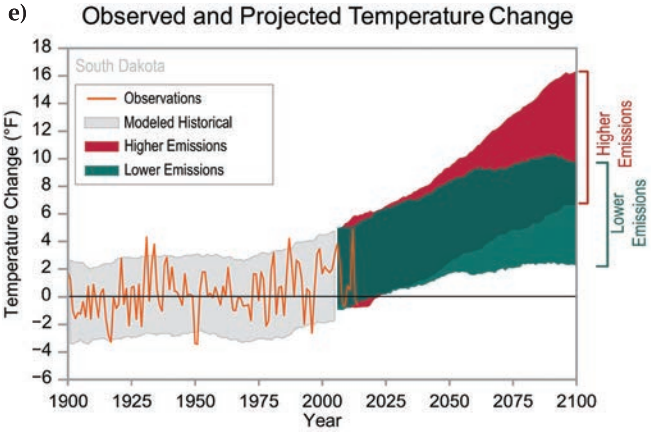


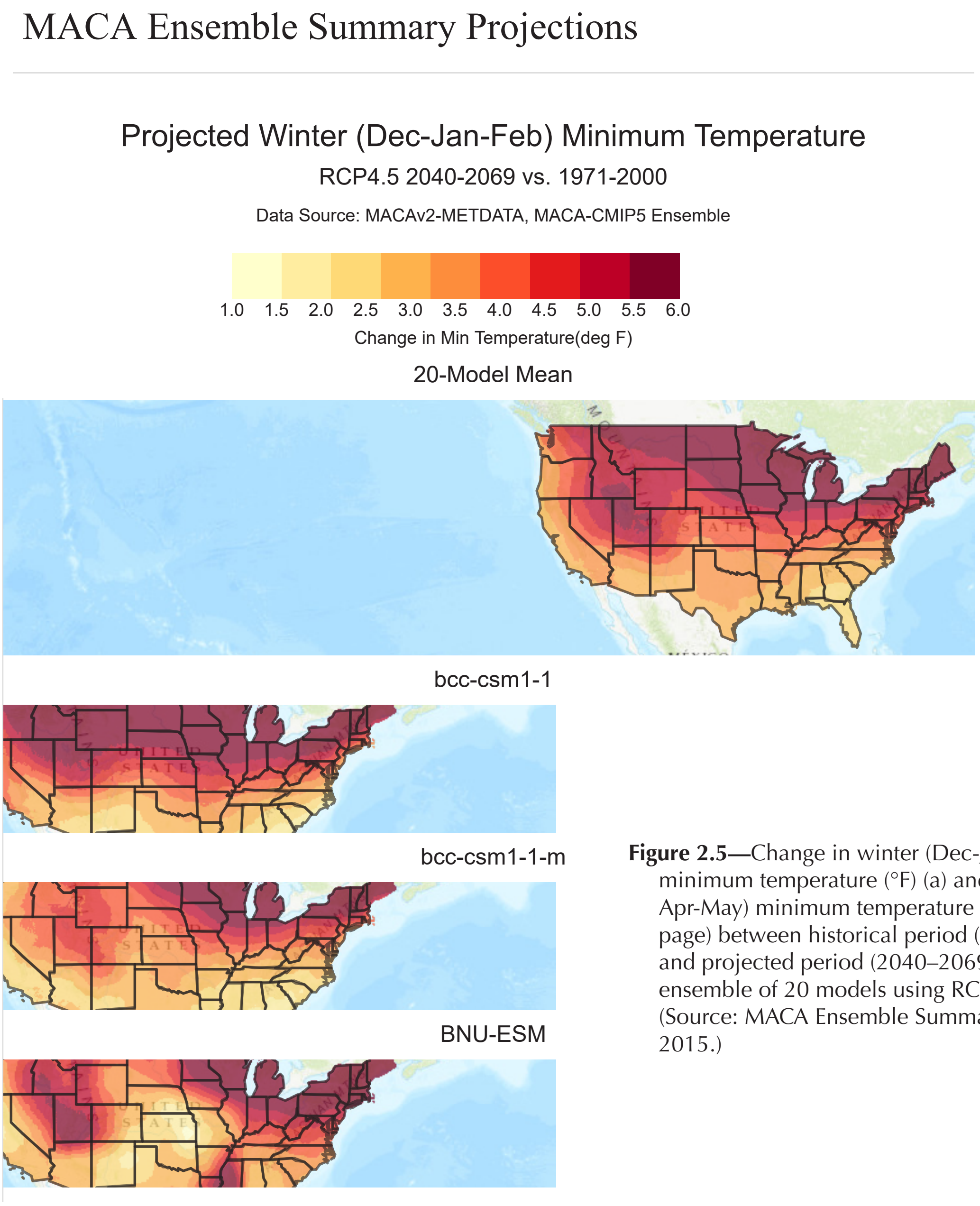
Figure 2.4—Observed and projected changes (compared to the 1901–1960 average) in near-surface air temperature (°F) for the five States in the USFS Rocky Mountain Region: Colorado (a); Wyoming (b); Kansas (c); Nebraska (d), South Dakota (e). Observed data are for 1900–2014. Projected changes for 2006–2100 are from global climate models for two possible futures: one in which greenhouse gas emissions continue to increase (higher emissions) and another in which greenhouse gas emissions increase at a slower rate (lower emissions). Shading indicates the range of annual temperatures from the set of models. (Sources: Colorado, Frankson et al. 2016d; Kansas, Frankson et al. 2016b; Nebraska, Frankson et al. 2016e; South Dakota, Frankson et al. 2016a; Wyoming, Frankson et al. 2016c.)

period 1971–2000 and the projected period 2040–2069. For winter minimum temperature, areas in the Rocky Mountain Region could see increases in winter temperatures of up to 6 °F. These temperature increases could push winter temperatures above a biologically meaningful threshold—above freezing in some areas. For spring temperatures, the projected range is 3 to 4 °F above the historical record.

Changing Climate, Changing Risks

This information about climate projections can supplement resource managers' experience-based knowledge. Within the Rocky Mountain Region, climate is highly variable, and the averages over time are used to guide expectations about, for example, the average winter temperature, the amount of annual precipitation, or the likely timing of spring green-up. This historical information is used to assess potential risks, such as drought, flooding, and fire, and to ensure temporal conditions that are appropriate for management actions, for example, the presence of frozen soil or the occurrence of green-up. What actually happens may differ from the average climate, but over the long term, the average typically is a good guide to the future. Recently, however, climatic patterns have shifted. The trend is toward warming temperature, suggesting that the long-term average to date may be an underestimate of what is likely, even in the near term. Hence, relying only on the historical climate may underestimate the potential for future risks.

a)

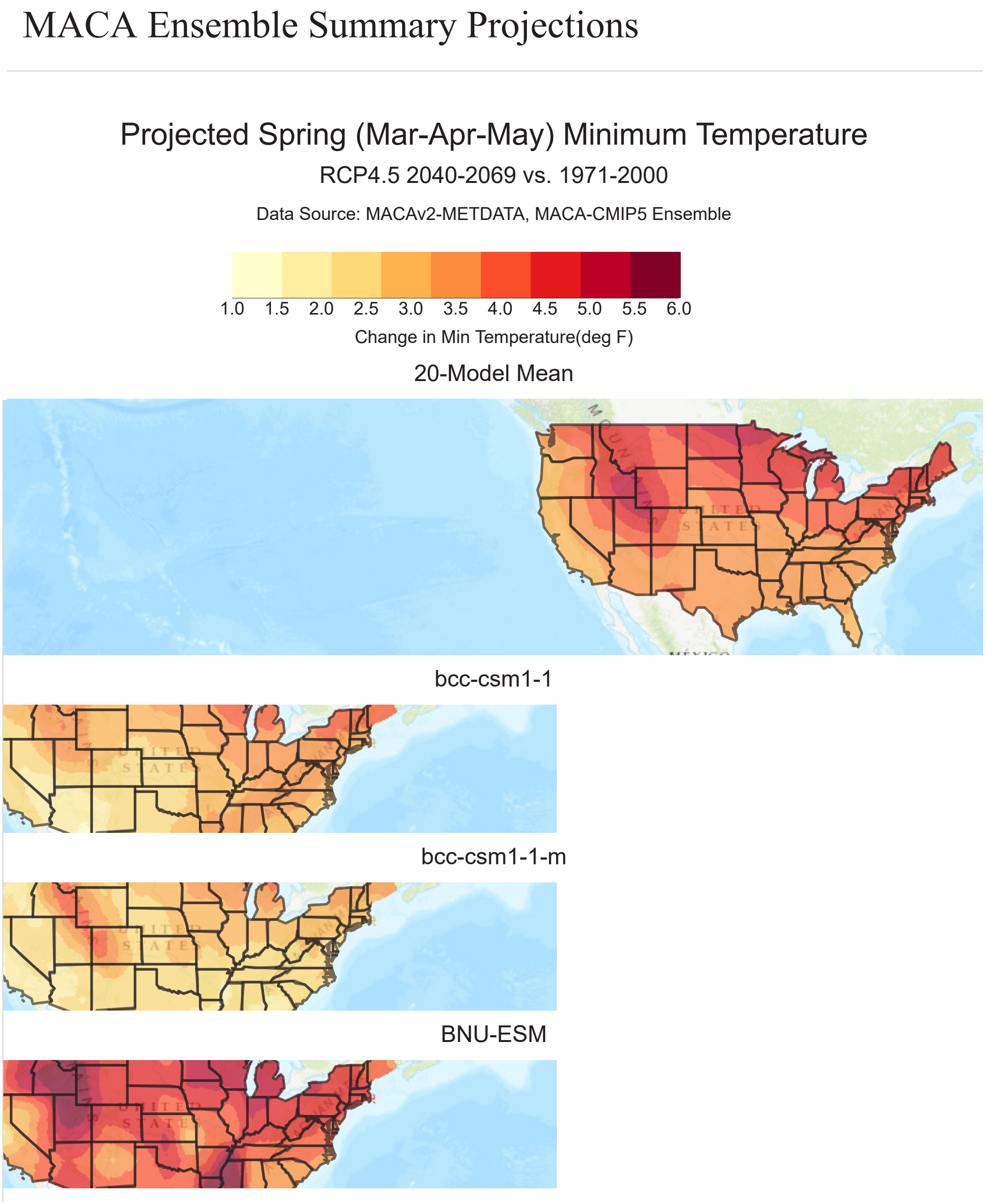


https://climate.northwestknowledge.net/MACA/tool_summarymaps3.php

Figure 2.5—Change in winter (Dec-Jan-Feb) minimum temperature (°F) (a) and spring (Mar-Apr-May) minimum temperature (°F) (b) (next page) between historical period (1971–2000) and projected period (2040–2069) based on the ensemble of 20 models using RCP 4.5 scenario. (Source: MACA Ensemble Summary Projections 2015.)

Scientists continue to learn about climate at local, regional, and global scales. The recent IPCC report reinforced this learning and confidence in the observed changes in climate, in the capability of climate models to reproduce climate features, and in the likelihood of future changes in climate (table 2.3). Greater certainty is associated with the projected temperature changes. These temperature changes will occur within the variability of the historical temperatures. In other words, individual months or seasons could warm, and yet the annual average could be close to the historical climate average. Very cold periods are also likely to be part of the near-term climate patterns.

b)



https://climate.northwestknowledge.net/MACA/tool_summarymaps3.php

Table 2.3—Key findings from the Fourth IPCC Assessment Working Group I Summary of Policymakers (IPCC 2013).

Conclusion	Statement
Observed changes in the climate	Warming of the climate system is unequivocal, and since the 1950s, many of the observed changes are unprecedented over decades to millennia. The atmosphere and ocean have warmed, the amounts of snow and ice have diminished, sea level has risen, and the concentrations of greenhouse gases have increased such that changes in the atmosphere are leading to changes in the climate.
Evaluation of climate models	Climate models have improved since the AR4. Models reproduce observed continental-scale surface temperature patterns and trends over many decades, including the more rapid warming since the mid-20th century and the cooling immediately following large volcanic eruptions (very high confidence).
Future temperature changes	Global surface temperature change for the end of the 21st century is likely to exceed 1.5 °C (2.7 °F) relative to 1850 to 1900 for all RCP scenarios except RCP 2.6. It is likely to exceed 2 °C (3.5 °F) for RCP 6.0 and RCP 8.5, and more likely than not to exceed 2 °C for RCP 4.5.

Precipitation projections are less certain, but moisture availability is likely to be lower even if precipitation remains the same, given the projected increases in temperature.

What are current biological and physical thresholds? Where will the biological and physical thresholds be crossed? The following chapters identify examples of observed changes and their observed effects, in addition to identifying potential sensitivities to moisture stress, extreme heat, and other climate features. Temperatures moving above freezing during winter are likely to have an effect on plants and animals, directly and indirectly. Warming winter temperatures have led to earlier snowmelt and an earlier onset of vegetative growth of some alpine/subalpine species; earlier flowering was then damaged by freezing events during spring (see Chapter 3). For some species, new environments may open up; areas of glacial melt have been colonized by midges (family Diamesinae) and species that are attached to submerged surfaces (see Chapter 4). Temperature and moisture availability differentially influence regeneration of tree species. An abrupt increase in regional tree establishment at the upper treeline coincided with a shift toward reduced cool-season precipitation (see Chapter 5). During the 26-month cone development phase of ponderosa pine, below-average temperatures in late spring damaged the second-year conelets, whereas high temperatures during the first year of cone production coincided with increased cone production (see Chapter 7).

Late-season moisture stress is another source of potential stress for ecosystems in the Region. Although the increased atmospheric carbon dioxide has been reported as increasing productivity, its fertilizing effect at high elevations in the Rocky Mountains was more than counterbalanced by the increased water stress due to longer growing seasons (see Chapter 5). In low-gradient streams, the response area in the stream network is more sensitive to flooding than in higher gradient streams; the large pool volumes can become filled with sediment and not be flushed out as rapidly as in higher gradient streams, thereby reducing critical habitat area (see Chapter 6). For intermittent streams on the Great Plains, local water-table fluctuations and soil moisture conditions can control hydrological responses more than local weather patterns might (see Chapter 8).

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Chapter 3. Alpine Turf and Dwarf-Shrubland Ecosystems: Vulnerability to Nonclimate and Climate Stressors in the U.S. Forest Service Rocky Mountain Region

Janine R. Rice, Claudia Regan, Dave Winters, Rick Truex, and Linda A. Joyce

Quick Look: Alpine Turf and Dwarf-Shrubland Ecosystems in the Rocky Mountain Region

Topography and climate play large roles in the distribution of plants and animals in alpine turf and dwarf-shrubland ecosystems. The alpine turf (dry tundra) ecosystem and the dwarf-shrubland ecosystem occur on moderate or gentle slopes, flatter ridges, valleys, and basins above treeline on the highest peaks of Colorado and Wyoming. These ecosystems form a mosaic along with fell-fields, wet meadows, alpine bedrock and scree, and ice fields across the alpine landscape. Temperatures year-round are cold, high winds can desiccate the soil and plants, and the growing season is short. At fine spatial scales, temperatures and moisture availability can change over short distances from an inhospitable environment to one that is suitable for some plants and animals.

The alpine turf plant community has a rich mixture of species. This diversity includes dense low-growing perennial grasses and sedges that produce new plants through rhizomes (horizontal underground rootstalks), and mat-forming forbs with thick roots or taproots. The rich diversity of the dwarf-shrubland community includes nonwoody (herbaceous) plants, and woody plants, such as dwarf shrubs of the heath family, that form a semi-continuous layer.

Alpine turf and dwarf-shrubland ecosystems provide habitat for a variety of wildlife. Voles, weasels, American pika, yellow-bellied marmots, and pocket gophers live year-round in the alpine. Large, hoofed animals, such as elk and bighorn sheep, migrate into the alpine during summer for forage. Rosy-finch, white-tailed ptarmigan, and American pipit breed above treeline. Alpine plants are pollinated by hummingbirds, and butterflies and other insects.

Quick Look: Vulnerability of Alpine Turf and Dwarf-Shrubland Ecosystems in the Rocky Mountain Region

Vulnerability to nonclimate and climate stressors: High

Confidence: High

Exposure: Warming temperatures year-round, potential early snowpack decline, high variability in timing of spring-fall freezing events.

Current extent: Isolated on high-elevation mountain peaks.

Sensitivity and adaptive capacity to climate change: Warm temperatures and loss of winter snowpack cover cause physiological stress on plants and animals; asynchronous plant and pollinator responses could lower successful plant reproduction. Life history traits of wildlife and high plant diversity aid persistence. Management practices, such as addressing air quality issues and restoring landscapes, may not completely offset the effects of warmer temperatures and snowpack loss.

Nonclimate stressors: Land use activities including domestic livestock grazing, mining, roads, and recreation, and air quality issues such as nitrogen deposition and dust on snow. Climate change may exacerbate the effects of these nonclimate stressors.

Introduction

Topography and climate greatly influence the distribution of plants and animals in the alpine ecosystem. This high elevation zone begins at treeline, the upper limit of upright tree growth, and ends where vascular plant life ceases to occur (Suding et al. 2015). The growing season is short; year-round temperatures are cold. At fine spatial scales, temperatures and moisture availability can change over short distances from an inhospitable environment to one that is suitable for specific plants and animals. For example, solar radiation and wind modify surface temperatures and snow cover along gradients in topography over short distances (Suding et al. 2015). Soil moisture can be diminished or increased by small changes in slope and aspect. Understanding how alpine plants, animals, and ecosystem processes will respond to climate change requires an awareness of how the plants and animals meet their physiological and biological needs in the harsh alpine environment (Millar et al. 2016).

Comer et al. (2003) identified six ecological systems that occur in this zone: alpine ice field, alpine bedrock and scree, alpine dwarf-shrubland, alpine fell-field, alpine dry tundra, and alpine-montane wet meadow. These ecological systems often occur as a mosaic in the alpine zone. Areas adjacent to melting snowpack are colonized by alpine wet meadows, and often alpine dwarf-shrublands are found next to the wet meadows. Comer et al. (2003) describe alpine dry tundra as the matrix of the alpine zone, intermixing with bedrock and scree, ice field, fell-field, alpine dwarf-shrubland, and alpine wet meadow.

These ecological systems are found on the highest peaks of Colorado and Wyoming; they extend down to about 11,000 ft in Colorado and about 10,500 ft in Wyoming (Comer et al. 2003). Alpine ice field, primarily snow and ice, occurs where snowfall exceeds melting. Alpine bedrock and scree is a barren or sparsely vegetated area, typically dominated by nonvascular (lichen) communities. Forbs, grasses, and low shrubs can also occur (Comer et al. 2003). Alpine fell-field systems occur on ridgetops or exposed saddles where wind keeps the area snow-free in winter (Suding et al. 2015). Unproductive shallow soils support cushion plants, with a plant cover of 15 to 50 percent. Alpine wet meadows are often found in small depressions associated with late-melting snow patches. Vegetation is often dominated by grasses or grass-like plants. Soils may be mineral or organic and show hydric soil features, including high organic content.

The two remaining ecological types in the alpine zone are the focus of this chapter: dry tundra (hereafter called alpine turf) and dwarf-shrubland. These two ecosystems span the North American mountain ranges, known as cordillera, that run from southern Mexico to Alaska (Comer et al. 2003). Within the Forest Service, U.S. Department of Agriculture (USFS) Rocky Mountain Region, alpine turf is found throughout the alpine zone; dwarf-shrubland predominantly occurs in the alpine zone of Wyoming (fig. 3.1).

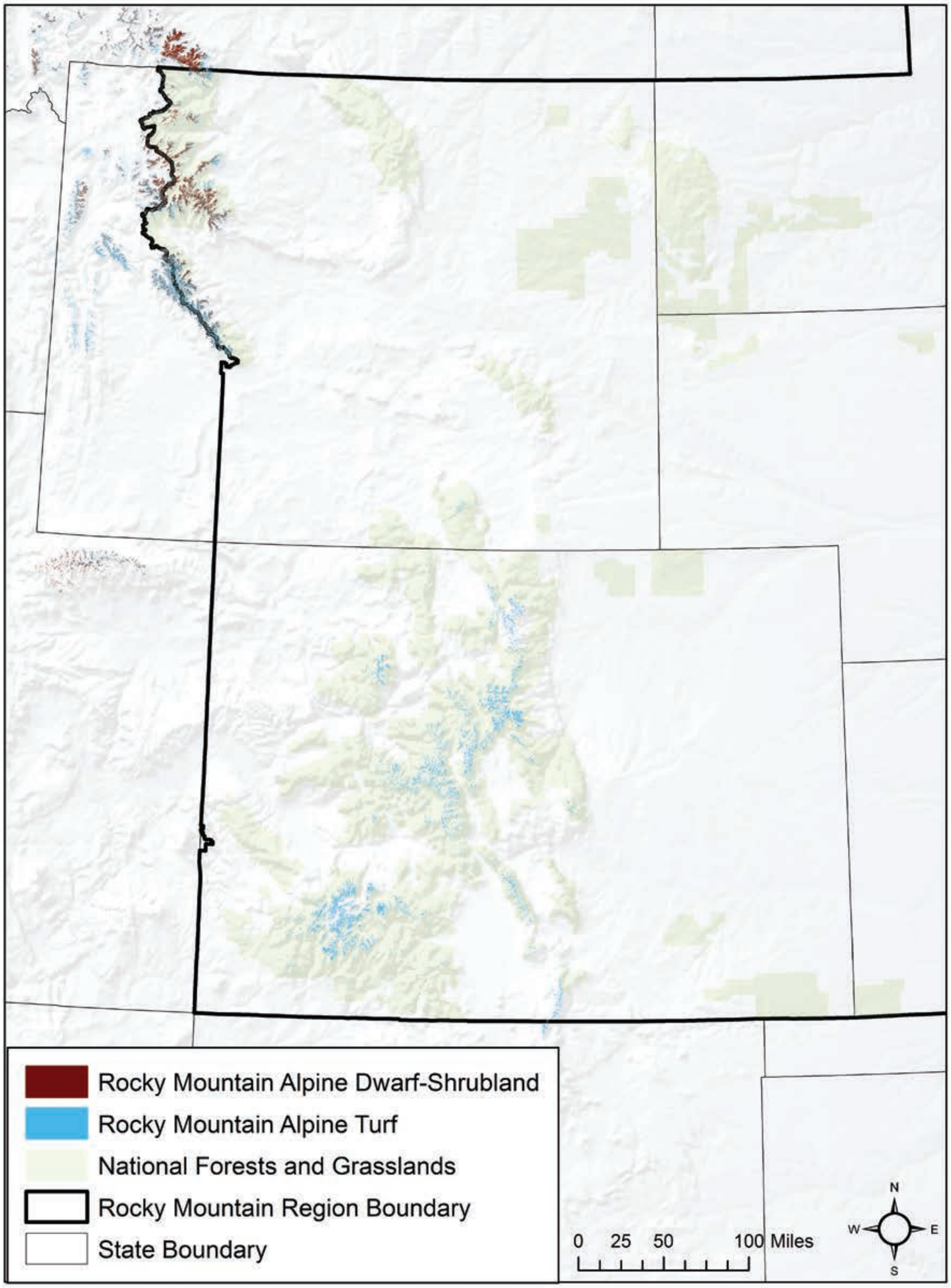

Figure 3.1—Alpine turf and alpine dwarf-shrubland distribution within USFS Rocky Mountain Region (Comer et al. 2003; Data can be found in NatureServe 2014).

Snow retention, wind desiccation, permafrost, and the short growing season strongly influence plant community composition in both of these types.

Alpine dwarf-shrubland ecosystems occupy level areas within glacial basins, where late-lying snow and sub-irrigation from surrounding slopes keep soil moist. Ericaceous dwarf shrubs or dwarf willows (*Salix* spp.) form a semicontinuous layer less than 20 in tall in which grasses, grass-like plants, and forbs are found. Dwarf-shrub species can include: pink mountain heath (*Phyllodoce empetriformis*), willows, bilberry (*Vaccinium* spp.), Western Labrador tea (*Ledum glandulosum*), yellow mountain heath (*Phyllodoce glanduliflora*), and alpine laurel (*Kalmia microphylla*). Other species found in the dwarf-shrubland communities can include: fleabane (*Erigeron* spp.), woolly pussytoes (*Antennaria lanata*), tundra aster (*Oreostemma alpigenum*), lousewort (*Pedicularis* spp.), Indian paintbrush (*Castilleja* spp.), tufted hairgrass (*Deschampsia caespitosa*), Howell's marsh marigold and white marsh marigold (*Caltha leptosepala* DC), fawnlily (*Erythronium* spp.), Parry's rush (*Juncus parryi*), Piper's woodrush (*Luzula piperi*), showy sedge (*Carex spectabilis*), black alpine sedge (*Carex nigricans*), and American bistort (*Polygonum bistortoides*) (Comer et al. 2003).

Alpine turf occurs on gentle to moderate slopes and flat ridges and in valleys, and in basins where soil moisture is relatively constant (Comer et al. 2003). This ecosystem has a rich mixture of plant species, which include dense low-growing perennial grasses and sedges that produce new plants through rhizomes (a horizontal underground rootstalk), and mat-forming forbs with thick roots or taproots. Rhizomatous sedges and grasses can include the following: blackroot sedge (*Carex elynoides*), dryspike sedge (*Carex siccata*), northern singlespike sedge or Western singlespike sedge (*Carex scirpoidea* spp.), spike sedge (*Carex nardina*), curly sedge (*Carex rupestris*), alpine fescue (*Festuca brachyphylla*), Idaho fescue (*Festuca idahoensis*), tufted hairgrass (*Deschampsia caespitosa*), and Bellardi bog sedge (*Kobresia myosuroides*). Forbs can include Ross' avens (*Geum rossii*), cushion phlox (*Phlox pulvinata*), and alpine clover (*Trifolium dasyphyllum*) (Comer et al. 2003).

Alpine turf and dwarf-shrubland ecosystems provide habitat for a variety of wildlife that are permanent residents or that migrate to utilize summer habitat resources. Small mammals occupying this ecosystem year-round can include voles (*Microtus* spp.), weasels (*Mustela* spp.), American pika (*Ochotona princeps*), yellow-bellied marmots (*Marmota flaviventris*), and pocket gophers (*Geomyidae* spp). Large, hoofed animals, such as elk (*Cervus elaphus*) and bighorn sheep (*Ovis canadensis*), migrate into the alpine for summer forage (Zeigenfuss et al. 2011). Rosy-finch (*Leucosticte australis*), white-tailed ptarmigan (*Lagopus leucura*), and American pipit (*Anthus rubescens*) breed above treeline. Alpine plants are pollinated by hummingbirds (*Selasphorus* spp.), and butterflies (Order Lepidoptera) and other insects. Alpine butterflies feed on alpine flower nectar and use host plants for reproduction (Matter et al. 2011).

Alpine turf and dwarf-shrubland ecosystems have been assessed for their vulnerability to climate change. As described in Chapter 1, the vulnerability framework has two major components, one related to nonclimate stressors and one related to climate stressors. Within the nonclimate component, four criteria were used to define factors of vulnerability: current status of ecosystem extent; human influences on the ecosystem, intrinsic resilience of the ecosystem to nonclimate stressors, and future trends of nonclimate stressors on the ecosystem. Within the climate change component, seven criteria capture the vulnerability of individual species, sensitivity of ecosystem dynamics such as dependence on a specific hydrological cycle, adaptive capacity of the ecosystem, potential for management to mitigate the effects of climate and nonclimate stressors, and interaction between climate change and nonclimate stressors. More information and the rationale for ranking ecosystem vulnerability by using these criteria are given in appendices A, B, and C.

These alpine ecosystems are increasingly being studied on national forest lands as well as in other areas throughout the Rocky Mountain Region. We draw from the literature where studies are within the Rocky Mountain Region. Even within this Region, these ecosystems vary greatly, particularly at fine spatial scale. Application of these results to alpine systems on individual national forests will require an understanding of local site characteristics and the site characteristics of the research studies reviewed here.

Vulnerability of Alpine Turf and Dwarf-Shrubland Ecosystems to Nonclimate Stressors

Summary of Key Vulnerabilities to Nonclimate Stressors

BOX 3.1

Current status of ecosystem extent

Alpine turf and dwarf-shrubland ecosystems are limited in extent, covering less than 1 percent of the entire Rocky Mountain Region. These ecosystems cover about 18 percent of the alpine zone above 10,500 ft, and the majority of alpine turf and dwarf-shrubland ecosystems in the Rocky Mountain Region occur within national forest boundaries. Consequently, their vulnerability ranking based on extent is very high.

Human influences on ecosystem

The vulnerability of alpine turf and dwarf-shrubland ecosystems is ranked as moderate to the legacy of past human influences and to the stressors associated with ongoing human influences.

Intrinsic resilience of ecosystem to nonclimate stressors

A very high vulnerability ranking is given for the intrinsic resilience of alpine ecosystems to nonclimate stressors.

- Factors that enhance the resilience of alpine turf and dwarf-shrubland ecosystems to nonclimate stressors:
    - Rich plant diversity, life history traits, and variety in community structure enable communities to persist by a shift in species composition.
    - Fire, limited in the alpine, may promote alpine plant establishment; however, postfire recovery can be slow.
- Factors that lower the resilience of alpine turf and dwarf-shrubland ecosystems to nonclimate stressors:
    - Alpine plants grow slowly.
    - Alpine plants recover slowly from disturbances, such as road and trail building, fire, trampling, wind, and water erosion. Recovery after disturbance can take more than a century.

Future trends of nonclimate stressors

Alpine turf and dwarf-shrubland ecosystems are considered to be moderately vulnerable to future nonclimate stressors associated with increased demand for recreation, invasive species, and atmospheric disturbances.

Current Status and Human Influences

Alpine turf and dwarf-shrubland ecosystems are limited in extent, covering less than 1 percent of the Rocky Mountain Region, or about 1 million ac on high mountain peaks in Colorado and Wyoming. These ecosystems cover about 18 percent of the alpine zone above about 10,500 ft in the Rocky Mountain Region, the remainder being rock, fell-fields, scree, wet meadows, and ice. About 85 percent of the area of alpine turf and dwarf-shrubland ecosystems within the boundaries of the Rocky Mountain Region occurs on national forest lands. About 55 percent of the area of alpine turf and dwarf-shrubland ecosystems is in wilderness areas.

Past human use of these ecosystems has included mining and grazing. Mining in the Rocky Mountain Region has harmed sites within some alpine areas, especially in the southern Rockies of Colorado (Bowman et al. 2002). Mines and tailings above treeline are remnants of mining activity that took place beginning in the mid-1800s. Driven by economics, mining is currently less active, but these areas still bear the marks of mining disturbance (Bowman et al. 2002).

Herbivory in alpine areas of the Region includes wild herbivores and domestic livestock. Alpine areas are grazed by permanent residents (American pika, pocket gophers) and seasonal wild herbivores (bighorn sheep, mountain goat [*Oreamnos americanus*], elk, and grizzly bear [*Ursus arctos horribilis*]). Domestic livestock grazing continues on national forest lands with high elevation allotments grazed primarily by sheep. Overgrazing coupled with drought has left a legacy of grazing impacts in the western United States; however, few studies have explored the nature of this legacy in alpine ecosystems. Generally, grazing effects, from both domestic and wild animals, vary with intensity of grazing, timing, type of grazing animal, grazing behavior of the animal, legacy of past grazing onsite, and the current grazing management system including the time available for recovery.

Recreation is increasing as a nonclimate stressor on these ecosystems. From 2011 through 2015, the Rocky Mountain Region had more than 27 million annual visits, of which 1.37 million visits were in Wilderness Areas (USDA FS 2015b). Nationally, visitors to national forests participate in a variety of activities including viewing natural features and wildlife, hiking or walking, downhill skiing, fishing, hunting, camping, backpacking, and resort use. Recreation can have direct effects on alpine plants and animals where humans come into contact with plants and animals through hiking and camping, and indirect effects, for example, soil compaction processes that are initiated when human use is high. Recreation in Rocky Mountain alpine areas has hindered plants due to trampling and erosion (Willard and Marr 1970; Willard et al. 2007). Turf-type plants face less potential damage from recreationists than do plants in wetter areas, which are more sensitive (Willard and Marr 1970). Turf-forming, matted, or rosette alpine plants are more resistant to trampling compared to upright or woody plants (Cole and Monz 2002). Lohman (2010) reported that trail and trail margins showed the greatest effect of human traffic, but the effects were not confined to the trail, suggesting overuse and congestion of the trails associated with the high mountain peak at Quandary Park, near Breckenridge, Colorado. Cross-country and downhill skiing are part of the recreational experience in alpine settings; Sato et al. (2013) in a meta-analysis describe the ecological effects of downhill skiing in the United States as negative or negligible more often than positive.

Invasive and Nonnative Species

The Rocky Mountain Region has a limited distribution of invasive plants in the alpine. Although invasive plant species in high elevation areas globally have expanded over the last decade (McDougall et al. 2011), the common dandelion (*Taraxacum officinale* F.H. Wigg.) was the only introduced species found in a survey of Rocky Mountain National Park in Colorado (Ashton 2010). Common dandelion displaces native species and may alter pollinator behavior when high densities occur (Molina-Montenegro et al. 2012; Muñoz and Cavieres 2008). The weevil *Rhinocyllus conicus* Fröelich has been found on native thistles on Niwot Ridge in Colorado, but its presence was negatively correlated with elevation. Climatic factors apparently limit its current capacity to expand above treeline; warming temperatures could facilitate an upward expansion (Hicks et al. 2013).

Atmospheric Nitrogen Deposition

Human activity is linked to an increase in atmospheric nitrogen deposition in the terrestrial and aquatic ecosystems at higher elevations of the Rocky Mountains (Baron et al. 2000; Ingersoll et al. 2006; Nanus et al. 2012). Low background rates of nutrient cycling and low biomass production of alpine ecosystems reflect the limited availability of nitrogen in alpine soils. Increased nitrogen deposition, through natural or experimental changes, has been observed to alter many community and ecosystem processes, resulting in increased alpine soil nitrogen cycling, increased soil acidification, greater nitrate leaching, shifts in plant and microbial community composition, and increased cover of nitrogen-loving grasses, such as tufted hairgrass (*Deschampsia cespitosa* (L.) P. Beauv.) (Bowman 2000; Bowman et al. 2015).

Dust on Snow

Researchers have linked more intense dust-on-snow events in the Rocky Mountains to drought and human activity in areas predominantly to the southwest but also to the west of the Rocky Mountains (Painter et al. 2007; Skiles et al. 2015). From 2010 through 2013, dust events advanced snowmelt 24 to 49 days in the San Juan Mountains and 15 to 30 days on Grand Mesa, both sites in Colorado (Skiles et al. 2015). As a result of dust-on-snow events in 2006, snow chemistry on 17 sites from southern Wyoming to central Colorado increased in pH, calcium content, and acid neutralizing capacity; the effects were more pronounced at upper elevations than on densely forested lower elevation sites (Rhoades et al. 2010). Steltzer et al. (2009) concluded that dust-on-snow events may tend to reduce the differences in snowmelt timing between north- and south-facing slopes and synchronize green-up and timing of flowering for plants across the alpine landscape. Earlier snowmelt from dust-on-snow events may temporally disconnect the availability of alpine plants for insects and animals seeking forage and pollination sources after hibernation or migration. During the growing season, less variety across the landscape in the timing of alpine plant growth and blooming may cause increased competition for pollinators (Steltzer et al. 2009).

Fire

Fire is limited and infrequent in the alpine; few studies are available to quantify the alpine fire regime in the Rocky Mountain Region. Fire return intervals were found to be a century or several centuries, based on charcoal records in high elevation lakes in Idaho (Anderson et al. 2008; Brunell and Whitlock 2003). Using fire atlases, Rollins et al. (2001) suggested that fire suppression during the 1930s through 1970s may have reduced the amount of area burned in some alpine areas in the Selway-Bitteroot Wilderness complex in Idaho and Montana. More studies are needed to determine human influence on the alpine fire regime in the Rocky Mountain Region.

Intrinsic Resilience of Ecosystem to Nonclimate Stressors

Factors that Enhance Resilience to Nonclimate Stressors

Plant species diversity, life history traits, and physical structure of alpine plants help to support the persistence of alpine plants exposed to nonclimate stressors. Plant diversity and the number of species across the landscape are high, with 150 to 300 species occurring in the Rocky Mountains (Bowman 2000; Komárková and Webber 1978). The establishment and survival of alpine plant species can be as high as for perennial species in more temperate climates of North America and Europe (Forbis 2003). Plant structure and form may provide some protection; for example, turf-forming grasses are less impeded by trampling than other plant structures. Plant traits such as being tall or prostrate also provide some resistance to grazing.

The resilience of alpine turf and dwarf-shrubland ecosystems to the limited occurrence of fire in the Rocky Mountain alpine is not well studied (see Current Status and Human Influences section). Dwarf-shrubland plants may benefit from fire occurrence; in the Washington Cascades, plant diversity eventually increased after recovery (Douglas and Ballard 1971). Fire can also promote alpine plant establishment in areas where subalpine trees previously grew (Billings 1969).

Factors That Lower Resilience to Nonclimate Stressors

Alpine turf and dwarf-shrubland ecosystems are not resilient to damage from road and trail building, mining, trampling, wind, and soil erosion; recovery can take from 30 years to more than a century (Conlin and Ebersole 2001; Ebersole 2002; Greller 1974; Willard and Marr 1970; Willard et al. 2007). Although recovery is slow, turf-forming grasses that grow in clusters are more tolerant of trampling than communities with groundcover dominated by primarily low woody shrubs or upright forbs (Cole and Monz 2002).

The alpine plant-herbivore relationship is strongly influenced by plant traits (tall, prostrate), environmental conditions (moisture, temperature, nutrients), and grazing history and use. The dominance of grasses and grass-like species as in the historically grazed alpine plant communities in the central Caucasus Mountains in Eurasia offers a contrast to the ungrazed or lightly grazed Rocky Mountain alpine systems; these differences may be, in large part, the result of a difference in human history (Bock et al. 1995). Warmer summer temperatures on grazed alpine sites in the Medicine Bow Mountains in Wyoming led to altered nitrogen cycling and carbon dioxide exchange on the grazed sites. Grazed sites were sources of carbon emissions, in contrast with the carbon capture on the ungrazed sites (Welker et al. 2004). Although alpine grazing may significantly stimulate nitrogen cycling, Martinsen et al. (2012) concluded from a controlled grazing experiment that the effect was not sufficient to shift the nutrient-poor alpine system away from nitrogen deficiency.

Future Trends of Nonclimate Stressors

Future land use in the alpine will depend on many factors. Economic factors are likely to influence mining activity and domestic livestock grazing. Cattle numbers declined in Colorado and Wyoming between 2000 and 2009 (Reeves and Mitchell 2012); however, little information is available to describe changes in livestock grazing numbers specific to the alpine zone. Nationally, winter activities projected under future scenarios are sensitive to changes in the climate and to economic growth. Downhill skiing is projected to go up, depending on income, whereas undeveloped skiing is projected to decline, influenced by changes in climate (Bowker et al. 2012). No projections are available for the alpine ecosystem alone.

Increased atmospheric nitrogen deposition at high elevations is likely to continue as human activity is expected to expand in the future. Bowman et al. (2015) suggest that dust-on-snow events, with their base cation-laden dust, may have prevented the onset of soil and surface water acidification from atmospheric nitrogen deposition; however, continued increases in atmospheric nitrogen deposition will shift that balance.

Native alpine plant communities may not be resilient to invasive or nonnative species, which disrupt ecosystem function and reduce biodiversity. Dandelion is an adaptable species that can successfully replace native species (Molina-Montenegro et al. 2012) and disrupt the role of pollinators in the ecosystem (Pauchard et al. 2009), especially when dandelions grow in high densities (Muñoz and Cavieres 2008). Although the presence of invasive species is currently limited in Rocky Mountain alpine areas, studies in Chile suggest that alpine ecosystems are not likely to be resilient to future invasions (Pauchard et al. 2009).

Vulnerability of Alpine Turf and Dwarf-Shrubland Ecosystems to Climate Stressors

Capacity for Range Shift

Box 3.2

Key Vulnerabilities

A very high vulnerability ranking is given for the capacity of alpine turf and dwarf-shrubland ecosystems for range shift.

- Plant migration potential is limited.
- Topographic variation may offer opportunities for plant redistribution within the alpine zone.
- Limited in connectivity, these ecosystems may become more fragmented with future contraction of favorable habitat.

Elevation Potential for Range Shift

Alpine turf and dwarf-shrubland ecosystems inhabit the area above subalpine forests and below any relatively inhospitable area at the highest points of mountains in Wyoming and Colorado. The lower boundary of the alpine zone may be vulnerable to treeline expansion, which would reduce the existing area of alpine (Williams et al. 2015b). Upper treeline has expanded into the alpine at rates greater than 200 ft over the last 100 years, and at greater rates in some decades in the Big Horn Mountains in Wyoming (Elliott 2012b), San Juan National Forest in Colorado (Rondeau et al. 2012), and central and southern Rocky Mountains (Elliott 2012a). Even if the area above the current alpine zone becomes climatically suitable, soils may not yet support alpine turf and dwarf-shrubland plant species (for example, in bedrock and scree, and fell-fields areas). Further, the development of suitable soil conditions may lag the rate of climate change.

As topography and climate influence plant distribution within the alpine zone, opportunities for expansion may exist within the current area via expansion into areas where glaciers have melted, and expansion into areas where moisture and temperature become suitable. Studies within the Rocky Mountain Region have quantified areal loss of glaciers as well as ice volume lost in recent periods in the Wind River Range in Wyoming (DeVisser and Fountain 2015), Arikaree Glacier on the Arapaho-Roosevelt National Forest in Colorado (Williams et al. 2015a), and glaciers in Teton National Park in Wyoming (Edmunds et al. 2011). Shrinking glaciers can potentially expose area for alpine plants to colonize if topography, soils, or microsite conditions allow.

The alpine zone is a complex terrain; landform features such as steep slopes, ridges, northerly versus southerly aspect, and depressions create a diversity of microclimates. Scherrer and Körner (2010) documented how microtopography can result in differences in temperature similar to large elevational gradients but over a short horizontal distance in the alpine. In such cases, species unable to adapt to a warming climate may need only a few yards of horizontal shift to find suitable temperatures. Alpine plants that grow in drier areas could migrate into alpine wetland areas that dry out from warming, as researchers observed in Glacier National Park in Montana (Lesica and McCune 2004).

Snow distribution and the timing of snowmelt are influenced by wind, slope, and aspect as well as air temperature and solar radiation (Suding et al. 2015). Snow distribution and snowmelt influence patterns of alpine plant community composition and plant productivity. Changes in wind and temperature could result in different spatial and temporal patterns of snow distribution and snowmelt. At Niwot Ridge, Litaor et al. (2008) concluded that decreases in the duration and depth of the snow cover in herbaceous alpine tundra ecosystems might result in a shift from dry and wet meadow communities toward less productive alpine fell-fields.

Ecosystem Connectivity

Alpine turf and dwarf-shrubland ecosystems are disconnected, existing on isolated mountain peaks in the Rocky Mountain Region. Large areas of low elevation land separate these ecosystems. Because of this lack of connectivity among alpine plant and animal communities, plants and animals in the alpine zone have been identified as possible candidates for assisted migration (Malanson et al. 2015; Wilkening et al. 2015a).

Vulnerability of Cold-adapted, Foundation, or Keystone Species to Climate Change

BOX 3.3

Key Vulnerabilities

A very high vulnerability ranking is given for cold-adapted, foundation, and keystone species in alpine turf and dwarf-shrubland ecosystems. Vulnerability to climate change by species can vary from moderate to very high.

- American pika have high vulnerability as they are sensitive to warm temperatures and to poor forage quality, a consequence of drought.
- Alpine turf and dwarf-shrubland plant communities are likely to have a very high vulnerability to warming and drying, with alpine turf communities the most vulnerable.
- These communities are also vulnerable to tree encroachment where moisture availability and soil conditions allow treeline advance.
- Pollinators are typically generalists and are mobile, characteristics that mitigate their vulnerability to climate change. However, conditions that decouple flowering phenology and pollinator emergence may contribute to moderate vulnerability to climate change.

We selected a subset of species that live in alpine turf and dwarf-shrubland ecosystems for assessment using this criterion. These species capture a range of environmental tolerances as well as a range in how their vulnerability may affect ecosystem function. We selected American pika, which live year-round in the alpine and are very sensitive to warm temperatures. We included alpine turf and dwarf-shrubland plant species as foundation species because they have substantial diversity and biomass, and provide an important food resource to species that inhabit or migrate to these ecosystems. We included pollinators as they have the qualities of keystone species because of their mutually beneficial relationship with blooming alpine plants.

American Pika

Sensitivity to warm temperatures and drier conditions, along with a low reproductive rate, poor dispersal capability, low population density, and use of scattered mountain habitat, suggests a high vulnerability to a changing climate for American pika. Recent local extirpations in California and the Great Basin (which spans most of Nevada and portions of Califronia, Idaho, Oregon, Utah, and Wyoming) have been attributed to drier conditions, poor-quality forage, and human-caused changes, such as roads and agricultural grazing (Beever et al. 2003, 2010; Grayson 2005; Stewart et al. 2015). Warmer temperatures and drier conditions associated with climate change could contribute to a decline in pika populations or local extinction in some areas, particularly at lower elevations (Calkins et al. 2012).

Concerns about this species have resulted in an increasing body of scientific research. The initial concerns about the resilience of the pika were broadly associated with temperature, but Millar et al. (2016) have shown that pika habitat—talus matrices (subsurfaces)—lagged the daily warm-up of air temperature in contrast to the talus surfaces. In addition, these matrices remained warmer than air temperature at night. The differences in air temperature in these microhabitats afford opportunities for the pika to behaviorally adapt to warming air temperatures. Millar et al. (2016) noted that warming air temperatures may become limiting for dispersal activities. Another example of an important microhabitat was identified by Wilkening et al. (2015b); pika were less stressed in matrices where subsurface ice features existed. They noted that pika inhabiting areas without subsurface ice may be vulnerable to changing climate. Erb et al. (2014) found that climatic factors were not dominant in determining pika relative density in Wyoming, Colorado, and New Mexico. Erb et al. (2014) concluded that pika populations in sites with poor-quality forage may be at the highest risk of declining.

Although this new research emphasizes the microclimatic variation in alpine habitats, this species uses habitats that are often physically isolated from each other, lowering dispersal opportunities under climate change. This limited dispersal capability and the critical dependence of pika on high-quality habitat have suggested that the species may be a candidate for assisted migration (Wilkening et al. 2015a). We conclude that pika is likely to be highly vulnerable to climate change.

Alpine Turf and Dwarf-Shrubland Vegetation

Alpine turf and dwarf-shrubland plants are highly vulnerable to warming in association with drying. Solar radiation contributes to the heating of soil surface temperatures higher than air temperatures. Although plants on warmer microsite conditions and in warmer years show an increase in heat tolerance, increases in air temperature may cause heat damage to alpine plants, especially when combined with drought stress (Buchner and Neuner 2003). Because moisture is more limited in dry alpine turf communities, plants may be more at risk from increased warming and drying.

Opedal et al. (2015) reported that small-scale topographic variability resulted in variations in microclimate (temperature, soil moisture) that benefited plants in alpine landscapes; this small-scale microclimatic heterogeneity may benefit plants under future climates. For seven community types in the alpine tundra at Niwot Ridge, most of the variability in composition over 21 years was nondirectional, both within and between community types (Spasojevic et al. 2013). Even with documented directional change in climate, nitrogen deposition, and release from grazing, finer-scale variability appears to be influencing alpine vegetation responses.

These communities may be at risk from subalpine tree encroachment under a warming climate (Williams et al. 2015a). Vegetation models project conifers will expand into higher elevations by the mid- and late-21st century (Bachelet et al. 2001; Notaro et al. 2012; Rehfeldt et al. 2012). However, projecting treeline expansion upslope may be difficult. Microsite conditions, temperature, and soil moisture aid as well to suppress tree establishment, for example by warming the alpine environment for growth as well as drying out seedlings by wind. These processes over time have led to the complex forest cover patterns on the landscape (Elliott 2012a,b; Malanson et al. 2007; Schrag et al. 2008; Weisberg and Baker 1995). Experimental treatments warming the soil with and without additional moisture have provided insights into the potential for treeline advance (Moyes et al. 2013, 2015; Winkler et al. 2016). Warming may enhance climate suitability for trees; however, summer moisture stress, along with high soil surface temperatures, was found to limit seedling success for limber pine (*Pinus flexilis* James) (Moyes et al. 2013). Encroachment of subalpine trees into alpine areas may occur with climate change, and is likely to be highly variable.

Pollinators

Pollinators in the high elevations of the Rocky Mountain Region are dominated by bumblebees (*Bombus* spp.; Bingham 1998), other bees (*Apis* spp.; Bingham 1998; Korb 2012), butterflies (e.g., Rocky Mountain Apollo butterfly [*Parnassius smintheus* Doubleday; Matter et al. 2011], beeflies [family Bombyliidae; Korb 2012], hoverflies [family Syrphidae; Korb 2012], and other flies [order Diptera; Korb 2012]). Pollinators in the alpine are likely to persist under climate change, but may be hindered by the direct effect of extreme climatic events (drought, extreme cold), and indirectly through the effects of climate change on plants in the alpine (Forrest 2014). In addition, factors other than climate may indirectly influence changes in pollinator communities. Franzén and Öckinger (2012) reported that over the last 60 years, alterations in pollinator communities were greatest at highest elevation in Sweden. This change was influenced by increased area from glacier decline, increased area of birch (*Betula* L. spp.) forest, and colonization by high elevation species previously absent from these sites.

Snowpack loss may play a role in hampering pollinators. A study of Rocky Mountain Apollo butterfly showed reduced snowfall hinders butterfly productivity, as snowpack provides insulation from freezing temperatures (Matter et al. 2011). Pollinators may be more vulnerable to bloom scarcity during dry years, or when early spring frost damages and reduces flower abundance, as researchers observed in montane meadows and subalpine forests (Aldridge et al. 2011; Inouye 2008). Warming may cause the connection between life histories of plants and pollinators to break. As plants may be more responsive to warming than pollinators and flower before pollinators emerge (Forrest and Thompson 2011; Thompson 2010), reductions in pollinator food sources could occur. Further, some plants that undergo shorter chilling periods may delay flowering beyond the time that pollinators emerge (Willmer 2012). However, researchers think the potential break in connected life histories is not a major threat (Willmer 2012). In a global study, pollinators flew distances (several hundred feet to miles depending on body size) to find other food sources (Greenleaf et al. 2007). Pollinators are typically generalists, not tied to a specific life history or plant species (Forrest and Thompson 2011; Willmer 2012).

Although there are research studies that identify potential vulnerabilities to pollinators under climate change, few of these studies have focused on identifying the major factors influencing pollinators under climate change in the alpine zone in the Rocky Mountain Region. Based on the available information, pollinators are moderately vulnerable to climate change.

Sensitivity to Extreme Climatic Events

BOX 3.4

Key Vulnerabilities

A very high vulnerability ranking is given for the sensitivity of alpine plants and animals to extreme climatic events.

- Seeding success and productivity are lower under drought stress and these effects can cascade through the trophic system. Drought also lowers the heat tolerance of alpine plants.
- Extreme heat can damage, stress, and kill plants and animals in alpine turf and dwarf-shrubland ecosystems. Although rosette, cushion, and dwarf shrub plants tolerate higher temperatures more than do nonwoody herbaceous species, they may not tolerate increases in warming and drying. Heat stress hinders the ability of animals to absorb nutrients, thus affecting their survival.
- Earlier springtime melting of the protective snowpack coupled with earlier warm air temperatures may promote emergence of plants which increases exposure of plants to springtime freezing and thus hampers plant reproduction and growth.

Sensitivity to Drought

Alpine turf and dwarf-shrubland plant and animal species experience stress, reduced reproduction, and death during drought. Seedling success was lower in drier areas, such as alpine dry tundra (Forbis 2003). Drought stress lowered the heat tolerance of alpine plants for all but the xerophytic, or highly drought tolerant, species (Buchner and Neuner 2003). Photosynthesis and productivity also declined during drought (Billings and Mooney 1968). Seedlings may be particularly sensitive to drought until their roots have reached a reliable water source (Billings and Mooney 1968). These drought effects to plants can cascade through the trophic system, limiting food for insects and animals (Aldridge et al. 2011; Calkins et al. 2012). These ecosystems are very sensitive to drought.

Sensitivity to Extreme Heat

Plants and animals in the alpine system potentially can be sensitive to extreme heat or even warming, given their life histories in the alpine environment. The paucity of long-term monitoring of alpine animal species has hampered research exploring the effects of warming temperatures or extreme heat on alpine plants and animals. Recent studies have experimentally manipulated the heat regime in plots, expanding our understanding of plants and increased heat, described next. Where available, long-term monitoring records can be analyzed in tandem with long-term records of climate to investigate the potential effects of warming and in some cases extreme heat. Recent work by Wann et al. (2014, 2016) is such an example. White-tailed ptarmigan populations have been monitored since 1968 at two locations in central Colorado: Rocky Mountain National Park and Mount Evans Wilderness. Temperatures in biologically relevant seasons (pre-nesting, nesting, and brood-rearing) warmed significantly at both sites. Median hatch dates advanced 3.7 and 1.9 days per decade for the Rocky Mountain National Park site and the Mount Evans Wilderness site, respectively; however, there were no trends in survival over the study period. Using a shorter time period, Wang et al. (2002) suggested future population declines with higher mean winter monthly minimum and maximum temperatures; however, Wann et al. (2016) found reproductive success only weakly linked to climate and concluded that the populations were invariant to fluctuations in seasonal weather. Reproductive success is likely to be influenced by many interrelated factors, and indirect consequences of warming may result from competition with other species for resources or increased predation rates under warming.

Species with a narrow range of temperature tolerance would be hindered or suffer population reductions when heat exceeds their tolerance limits; however, recent research has suggested that some alpine species exploit the complexity of the alpine terrain to escape extreme heat (e.g., pika; see Vulnerability of Cold-adapted, Foundation, or Keystone Species section). Wildlife (deer [*Odocoileus* spp.], elk) can regulate their temperatures behaviorally by migrating to cooler areas, such as the alpine zone. Parker and Robbins (1983) found that deer appeared to be more sensitive to warming temperatures than elk. Heat stress hinders the ability of animals to absorb nutrients, thus affecting their survival; examples are moose (*Alces alces*; Lenarz et al. 2009) and mountain goat (White et al. 2011).

Alpine plants can withstand high temperatures. Researchers found that rosette, cushion, and dwarf-shrub plants tolerated higher temperatures better than did nonwoody herbaceous species (Buchner and Neuner 2003). In the Swiss Alps, researchers found cushion plants died when leaf temperatures reached 135 to 136 °F, and glaucous sedge (*Carex firma*) shoots and rhizomes withstood temperatures up to 140 °F (Larcher et al. 2010; Neuner et al. 1999). Studies show that the heat tolerance of alpine plants develops over time as plants are exposed to high heat that remains below their critical thresholds (Buchner and Neuner 2003). Variety of microsite conditions over the landscape may create nearby refugia for species to move into and escape from warm temperatures (Scherrer and Körner 2010). However, these plant species may not be able to tolerate greater heat, especially when combined with drought stress, as researchers saw in heating experiments (Buchner and Neuner 2003). Because alpine turf and dwarf-shrubland plants and wildlife are damaged, stressed, or killed by extreme heat, these ecosystems are expected to be very sensitive to extreme heat.

Sensitivity to Freeze Date Variability and Earlier Snowmelt

Alpine turf and dwarf-shrubland species survive freezing temperatures, but can be sensitive to earlier snowmelt and more variability in freeze dates. Subalpine nonwoody plant growth is controlled by snowpack in the Rocky Mountains (Inouye 2008). Late flowering species may benefit from a longer growing season with earlier snowmelt. However, earlier flowering species could have an earlier onset of vegetative growth, with increased risk of frost damage, particularly with variable springtime freeze events (Aldridge et al. 2011; Inouye 2008). Munson and Sher (2015) found that the flowering date of rare species in the alpine has accelerated every decade since the late 1800s, with high spring temperatures explaining the advanced phenology. The reproduction of perennial forbs in montane and subalpine areas is hampered by frost damage that occurs when the protective snowpack cover is lost with earlier snowmelt (Miller-Rushing and Inouye 2009). Frost damage may be especially problematic for taller alpine plants, as was reported for alpine dwarf shrubs in the Swiss Alps (Rixen et al. 2012). Earlier snowmelt was found to increase the risk of lethal spring freezing exposure for less freeze-resistant species (Wheeler et al. 2014). In the Swiss Alps, freezing damage was increased for alpine dwarf shrubs grown under experimentally elevated atmospheric carbon dioxide levels compared to ambient conditions. Sensitivity to freezing damage was greater with advanced plant phenology in the spring after soil warming caused earlier snowpack melt (Rixen et al. 2012). Mondoni et al. (2012) demonstrated that fall temperature warming could result in seed germination in fall instead of the typical springtime germination, thus exposing seedlings to harsh winter environmental conditions. Alpine turf and dwarf-shrubland ecosystems are expected to be sensitive to freeze date variability and earlier snowmelt.

Intrinsic Adaptive Capacity to Climate Change

BOX 3.5

Key Vulnerabilities

A moderate vulnerability ranking is given for the intrinsic adaptive capacity of alpine ecosystems to climate change.

- Factors that strengthen adaptive capacity:
    - Species are highly tolerant to large variations in weather.
    - Microtopographic variation over alpine landscapes and rich plant species diversity support persistence by enabling transitions in community composition in response to environmental changes.
    - Varied life history strategies have aided plant and animal growth in the harsh alpine climate.
- Factors that weaken adaptive capacity:
    - Plants, especially in dry environments, have very slow growth and a long recovery time after disturbance.
    - Many alpine plants are not tolerant of early freezing events that may occur with earlier snowmelt.
    - Bird species, such as white-tailed ptarmigan, and mammal species, such as snowshoe hare, that undergo seasonal color molt may have limited capacity to respond to widely varying temporal changes in snowpack accumulation and snowmelt.

Factors That Strengthen Adaptive Capacity to Climate Change

Alpine turf and dwarf-shrubland plants and animals have acclimated and adapted to the widely varying and harsh conditions of cold, heat, and wind (Billings and Mooney 1968). Animal species have adapted by migrating to warmer environments (birds, ungulates), hibernating (yellow-bellied marmot), or staying active all year through development of life history traits to conserve energy and survive the cold alpine winters (American pika) (Beever et al. 2003). Some alpine plant species grow close to the ground to reduce wind impacts, grasses and rosette plants keep a large portion of their biomass belowground to escape cold, and plants reproduce during a short growing season (Forbis 2003). The chemical characteristics of alpine plants make them more tolerant to heat and cold (Körner 2003).

Alpine turf communities have high species diversity. Life history strategies vary; for example, Forbis and Doak (2004) contrasted cushion plants where one strategy was successful reproduction whereas the other strategy was long life spans and persistence through environmental conditions. Plant species resistance to environmental stress varies. For example, enhanced snowpack hindered Bellardi bog sedge, whereas Parry's clover (*Trifolium parryi*) increased cover (Bowman 2000). Given this diversity, some species under climate change will benefit and possibly expand, and some will decline, which will change the composition of the plant community.

Factors That Weaken Adaptive Capacity to Climate Change

The adaptive capacity of alpine turf and dwarf-shrubland ecosystems is limited by slow growth rates. Recovery after disturbance can take from 30 years to more than a century, with communities at drier sites, such as alpine turf, recovering more slowly than communities at moister sites, such as wet meadows (Ebersole 2002; Willard et al. 2007).

Species that live in alpine turf and dwarf-shrubland habitats have a low ability to adapt to changes in temperature, reduced moisture, or changes in freeze dates that could reduce food sources (see Vulnerability of Cold-adapted, Foundation, or Keystone Species section, and Sensitivity to Extreme Climatic Events section). Changes in snowpack accumulation or melt are problematic for species that undergo seasonal molts, where their coats change color to match the presence or absence of snow. Zimova et al. (2014) found that snowshoe hares (*Lepus americanus*) had limited plasticity in coat color molt phenology in the spring, and showed no capacity to change their fall molt phenology. Predation is the primary cause of mortality for this species. When snowshoe hares were mismatched against their background, Zimova et al. (2016) reported a decline in weekly survival. Based on this information and climate change projections, snowshoe hare annual survival would decrease by 23 percent in Montana at the end of this century. There is great variation at the individual level in molt phenology for snowshoe hares, but without an evolutionary response, population growth rates and population numbers of these species could decline under climate change. See also Vulnerability of Species section in Chapter 5.

Dependence on Specific Hydrological Regime

BOX 3.6

Key Vulnerabilities

A very low vulnerability ranking is given for alpine turf and dwarf-shrubland ecosystem dependence on a specific hydrological regime. These ecosystems have adapted to a wide range of variability in moisture.

Alpine turf and dwarf-shrubland ecosystems are not dependent on a specific and narrow hydrological regime. These ecosystems are not limited by forms of moisture that are largely received as snow but can also be rain. Yet wind, slope, and aspect as well as air temperature and solar radiation influence snow distribution, snowmelt, and moisture availability on the alpine landscape (Litaor et al. 2008; Suding et al. 2015). Plant phenology and growth are greatly controlled by snowpack distribution and snowmelt. Less snow or drier conditions may hinder some species, for example, alpine turf plants (Forbis 2003; Oberbauer and Billings 1981) and pika (Erb et al. 2011). But the high elevation of alpine ecosystems may buffer the near-term climate effects, such as less snowpack, on these systems compared to lower elevations (Lukas et al. 2014).

The availability of moisture in the alpine is influenced also by the advance and retreat of glaciers. Although there is a consensus that glaciers in the western United States will continue to retreat over the next 100 years, the local effect of climate change on glaciers differs according to glacier elevation, size, shape, slope or aspect position, and local monthly temperature and precipitation (Hall and Fagre 2003). Changes in glaciers can be incremental, though glacier outburst floods have the potential to alter the rate and magnitude of changes in valley bottoms. Researchers observed this effect in a flood that originated from meltwater ponded by glacial ice from the Grasshopper Glacier in the Wind River Range (Oswald and Wohl 2008).

Likelihood of Managing Climate Change Effects

BOX 3.7

Key Vulnerabilities

Management options for mitigating climate change effects in localized areas of alpine ecosystems are available, but there are few management options to offset effects from heat, earlier snowmelt, snowpack loss, variability in freezing dates, and drought stress projected with climate change. Consequently, the vulnerability ranking is very high.

Management strategies for mitigating the effects of climate change on alpine turf and dwarf-shrubland ecosystems include reducing current stressors, enhancing the resilience of the system, and, as climate continues to change, helping plants and animals adapt without substantial loss of soil, soil nutrients, and plant cover (Millar et al. 2007; Peterson et al. 2011). Alpine turf and dwarf-shrubland ecosystems have high vulnerability to nonclimate stressors; reducing these current stressors would enhance alpine resilience, potentially reducing near-term effects of climate change. Management actions include controlling or limiting invasive species, rehabilitating disturbed sites, reseeding and transplanting turf, restoring endemic communities, restoring connectivity, managing populations of key native animals, and reducing soil erosion (Bay and Ebersole 2006; Conlin and Ebersole 2001; McDowell et al. 2014; Morrison and Pickering 2013). Characteristics that enhance the intrinsic adaptive capacity may be important to conserve, such as ensuring that migration and dispersal patterns are intact to allow mobile species to move to refugia (see Intrinsic Adaptive Capacity to Climate Change section).

Adaptation strategies to address the potential for increased natural hazards under climate change have been the focus of discussion. In Europe, risk reduction strategies have been identified for increased floods, debris flow, glacial hazards, mass movement (rockfalls), and avalanches. Glacial hazards include the loss of stability in shelf glaciers, increase in glacial lakes, and risk of outburst flooding such as occurred in Wyoming (Oswald and Wohl 2008). PLANALP (2012) identified 10 recommendations for integrated climate-proof risk management in the alpine space:

1. Prepare for emergency intervention;

2. Review the climate change fitness of existing structural protection measures;
3. Set up and optimize long-term monitoring and warning;
4. Anticipate and deal with new risks;
5. Adapt hazard and risk mapping to a changing climate;
6. Enhance coordination between spatial planning and risk management;
7. Establish a risk culture and initiate risk dialogue;
8. Strengthen individual preparedness and precaution;
9. Improve the knowledge base and transfer to practice; and
10. Maintain and improve the functionality of protection forests (forests that protect people, buildings and infrastructure from natural hazards).

Climate change management options, however, may not completely offset the harm in alpine areas from heat, severe drought stress, earlier snowmelt, snowpack loss, and shifts in freezing dates. In addition, ecosystem services from the alpine ecosystems to support water supply and the tourism industry may be affected by climate change, motivating local adaptations. Morrison and Pickering (2013) reported a range of adaptation strategies in the Australian Alps by conservation managers, the tourism industry, and local communities. Some adaptation strategies complemented each other, such as efforts to control invasive species, but other adaptation strategies resulted in concern and conflicts among the stakeholders, such as year-round recreational use in the alpine and increased use of water for snowmaking.

Potential for Climate Change to Exacerbate Effects of Nonclimate Stressors, or Vice Versa

BOX 3.8

Key Vulnerabilities

Climate change may exacerbate the magnitude, intensity, and effect of nonclimate stressors. Consequently, the vulnerability ranking for alpine turf and dwarf-shrubland ecosystems is very high.

- Future increases in drought occurrence and intensity, combined with human activities to the west of the Rocky Mountain Region, may exacerbate dust-on-snow events that reduce duration of snow cover.
- Increased atmospheric nitrogen deposition attributed to human sources is projected to exceed critical levels, resulting in ecosystem damage.
- Warmer temperatures and drought may increase fire occurrence in Rocky Mountain forested areas, but it is not clear how the fire regime will change in the alpine under a changing climate.

Drought and Dust on Snow, Warmer Temperatures, Atmospheric Nitrogen Deposition

Climate change could potentially exacerbate the effects of dust on snow in alpine areas of the Rocky Mountain Region. Future dust-on-snow events coupled with regional warming could shift peak snowmelt by 2050 an additional 45 to 50 days earlier than current conditions, depending on the intensity of the dust events (Deems et al. 2013).

Climate change coupled with projected nitrogen emissions could result in critical load exceedances in the western United States. A critical load is defined as an annual deposition flux in nitrogen (kg ha^{-1} $year^{-1}$), below which significant ecosystem damage does not occur. Nitrogen gases include both nitrogen oxide and ammonia. Future emissions of nitrogen oxide are projected to decline by 52 to 73 percent, but ammonia emissions are projected to increase by 19 to 50 percent by 2050 (Ellis et al. 2013). Using these assumptions about future nitrogen emissions, Ellis et al. (2013) quantified future ecosystem-dependent critical loadings in national parks across the United States. Under these future scenarios, atmospheric deposition may exceed critical loads by mid-century for many western national parks, including Rocky Mountain National Park (Ellis et al. 2013). They conclude that returning all parks to nitrogen deposition below their ecosystem-dependent critical load by 2050 would require at least a 50-percent decrease in anthropogenic ammonia emissions relative to the projected 2050 level. See also Atmospheric Nitrogen Deposition section in Chapter 4.

Climate and Fire

Fire history records collected at treeline between subalpine forest and alpine show that fire occurrence is closely tied to drought (Buechling and Baker 2004; Sherriff et al. 2001; Sibold and Veblen 2006). Increased drought is likely in the future. Little information on fire in the alpine in the Rocky Mountain Region is available; in other alpine areas, wildfire can have significant effects, but recovery of some plant species can be fast (Takahashi 2012; Williams et al. 2008). How the alpine fire regime in the Rocky Mountain Region may change in the future is not certain.

Summary of the Regional Vulnerability of Alpine Turf and Dwarf-Shrubland Ecosystems

The overall regional vulnerability ranking for the ecosystem is a combination of the vulnerability to nonclimate stressors and the vulnerability to climate stressors. For alpine turf and dwarf-shrubland ecosystems, the mean overall vulnerability ranking was high with mean expert reviewer rankings in the high or very high category (fig. 3.2, table 3.1). Mean vulnerability rankings were high for both the nonclimate stressor vulnerability and the climate vulnerability (fig. 3.3). Expert reviewer rankings were in the high or very high category for the nonclimate stressor vulnerability and in the high category for the climate vulnerability.

The expert reviewers had high overall confidence in the ranking of the overall regional vulnerability (fig. 3.2). The five experts' individual rankings for overall confidence ranged from the moderate to the very high category (table 3.2). Mean confidence was high for the nonclimate stressor vulnerability ranking, but individual confidence rankings ranged from the low to the very high category (fig. 3.3a). Mean confidence was also high for the climate vulnerability ranking with individual rankings in the high or very high category (fig. 3.3b).

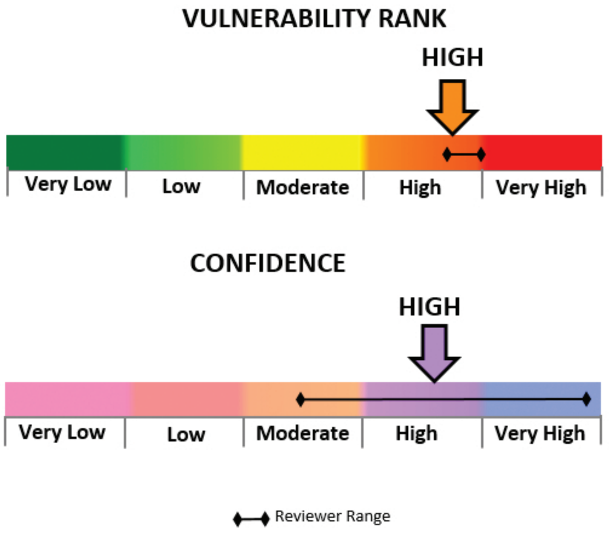

Figure 3.2—Overall regional vulnerability ranking and confidence ranking for alpine turf and dwarf-shrubland ecosystems in the Rocky Mountain Region. Large arrow points to the mean score for vulnerability and for confidence. Bar represents the range of scores.

Table 3.1—Original and reviewer rankings for the regional vulnerability of alpine turf and dwarf-shrubland ecosystems in the Rocky Mountain Region. Underline indicates a reviewer score change from the original assigned score.

Criterion	Original Score	(Reviewer 1) Score	(Reviewer 2) Score	(Reviewer 3) Score	(Reviewer 4) Score	(Reviewer 5) Score	Reviewer criterion rank mean
Nonclimate							
1. Extent	5	5	5	5	5	5	Very High
2. Human influences	3	3	3	3	3	3	Moderate
3. Resilience	5	5	5	5	5	5	Very High
4. Future trends	3	3	3	3	5	3	Moderate
Total	16 High	16 High	16 High	16 High	18 Very High	16 High	16 High
Climate							
1. Ecosystem shift	5	5	5	5	5	5	Very High
2. Species groups	5	5	5	5	5	5	Very High
3. Climatic events	5	5	5	5	5	5	Very High
4. Adaptive capacity	3	3	3	3	3	3	Moderate
5. Hydrology	1	1	1	1	1	1	Very Low
6. Management	5	5	5	5	5	5	Very High
7. Interactions	5	5	5	5	5	5	Very High
Total	29 High	29 High	29 High	29 High	29 High	29 High	29 High
Overall Vulnerability Rank	23 High	23 High	23 High	23 High	24 Very High	23 High	23 High

Table 3.2—Reviewer rankings for confidence in the vulnerability assessment of alpine turf and dwarf-shrubland ecosystems in the Rocky Mountain Region

Criterion	(Reviewer 1) Score	(Reviewer 2) Score	(Reviewer 3) Score	(Reviewer 4) Score	(Reviewer 5) Score	Criterion rank confidence mean
Nonclimate						
1. Extent	3	5	5	5	5	Very High
2. Human influences	1	3	3	5	5	Moderate
3. Resilience	3	3	5	5	5	High
4. Future trends	1	3	3	5	5	Moderate
Total	8 Low	14 High	16 High	20 Very high	20 Very high	16 High
Climate						
1. Ecosystem shift	3	4	5	5	5	High
2. Species groups	1	4	5	5	5	High
3. Climatic events	3	5	3	5	5	High
4. Adaptive capacity	5	3	3	5	3	High
5. Hydrology	5	3	1	1	5	Moderate
6. Management	5	5	5	5	5	Very high
7. Interactions	3	5	5	5	5	Very high
Total	25 High	29 High	27 High	31 Very high	33 Very high	29 High
Overall confidence rank	17 Moderate	22 High	22 High	26 Very high	27 Very high	23 High

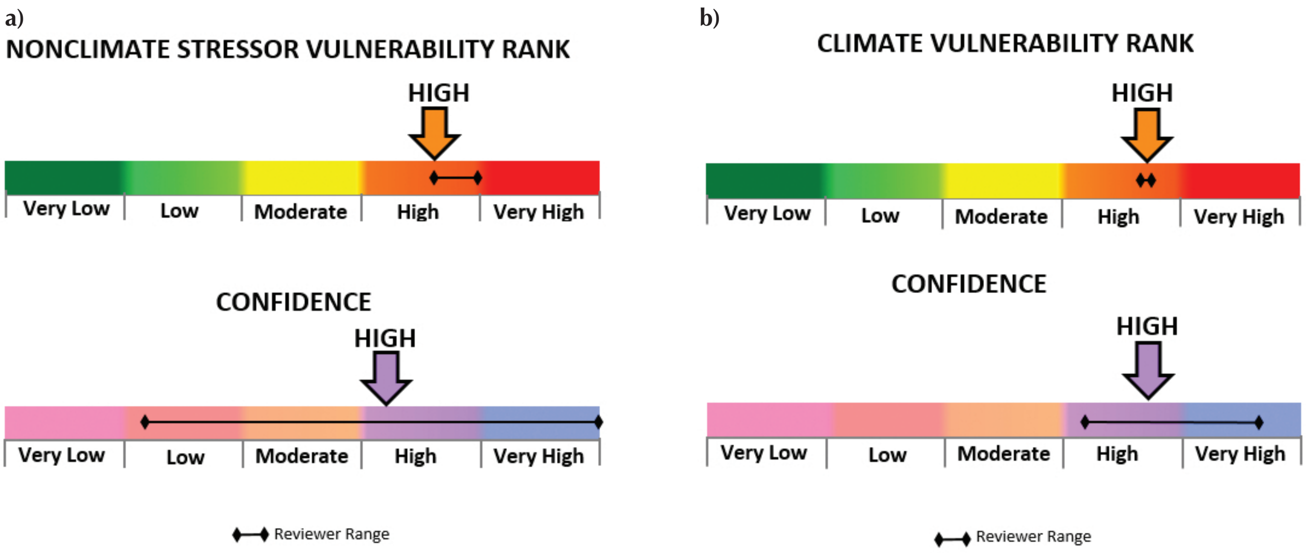

Figure 3.3—Regional vulnerability of alpine turf and dwarf-shrubland ecosystems to nonclimate (a) and climate (b) stressors in the Rocky Mountain Region. Large arrow points to the mean score for vulnerability and for confidence. Bar represents the range of scores.

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Chapter 4. Aquatic, Riparian, and Wetland Ecosystems in Glaciated Valleys: Vulnerability to Nonclimate and Climate Stressors in the U.S. Forest Service Rocky Mountain Region

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Quick Look: Aquatic, Riparian, and Wetland Ecosystems in Glaciated Valleys in the Rocky Mountain Region

Glaciated valleys in the Rocky Mountain Region have unique geological and topographic features that result in a heterogeneous landscape. Streams, kettle basins, cirques, tarns, and wetlands provide habitats for a diversity of plants, invertebrates, and animals. Valleys typically have a "U" shape, with steep side walls and a wide, low-gradient valley floor. Glaciers or permanent snowfields can be found in these valleys. The tarns, cirques, and moraines formed by glaciers retain and redirect water. Groundwater strongly influences the moisture availability in these valleys. Valley floors have riparian areas and well-developed soils in the low-gradient stretches, where gravel, sand, and silt eroded from the hillsides are deposited.

Climate is harsh: a short growing season, high snowfall, cold temperatures, and high winds that can cause drying. Aquatic, riparian, and wetland ecosystems provide important and unique habitats for cold-adapted plants and animals that live there year-round or that migrate into these areas seasonally. Cold-adapted species include benthic macroinvertebrates, cold-water fish species, and amphibians such as the boreal toad. Benthic macroinvertebrates found in these valleys provide essential ecosystem services, such as sediment mixing and the acceleration of detrital decomposition, nutrient cycling, and energy flow through food webs. Wetland communities are strongly influenced by the hydrological regime. Riparian areas adjacent to streams are sensitive to flow dynamics in streams; fens are present where water tables are high and are sensitive to groundwater availability, and wet meadows develop where water tables are not perennially high. This complex of habitats results in highly diverse plant taxa distinct from lower elevations.

Quick Look: Vulnerability of Aquatic, Riparian, and Wetland Ecosystems in Glaciated Valleys in the Rocky Mountain Region

Vulnerability to nonclimate and climate stressors: Very high

Confidence: Very high

Exposure: Warming temperatures, particularly in winter and spring. Retreat of glaciers, potential loss of annual snowpack, and potential early melt of snowpack. Higher elevations may be less affected.

Current extent: Aquatic, riparian, and wetland ecosystems in glaciated valleys account for a small fraction of area in the high elevation mountain valleys of Colorado and Wyoming.

Sensitivity and adaptive capacity to climate change: These ecosystems are highly dependent on a snow-dominated hydrological regime. They are sensitive to drought, declines in snowpack, and longer growing seasons with potential late-season drying. Insect and plant diversity is relatively high, which helps adaptive capacity to environmental changes. However, warm

> and dry conditions may exceed plant and animal tolerance levels under climate change. Early plant flowering in response to warm early spring temperatures could affect pollinators. Low streamflows, drying ponds, and lower water tables will stress aquatic and wetland plants and animals. At the highest elevations, plants and animals have little potential to migrate upstream or upslope. Additionally, climate change may exacerbate nonclimate stressors, such as dust on snow.

Nonclimate stressors: Mining, livestock grazing, and development of roads, trails, and infrastructure affect water quality as well as water flow patterns in these broad valleys. Atmospheric nitrogen deposition and dust on snow also affect snowpack and water chemistry. These stressors—particularly those that occur because of increased human influences, such as declines in air quality and increases in recreation—will continue to stress these ecosystems.

Introduction

Understanding the process of glaciation is critical for understanding where and in what form wetland and riparian ecosystems are found (Gage and Cooper 2013). No national forest or national grassland in the Forest Service, U.S. Department of Agriculture, Rocky Mountain Region was covered by the continental glaciers in the Pleistocene, but colder climates contributed to the development of mountain glaciers in the San Juan Mountains in Colorado, the Colorado Rocky Mountains, and the Medicine Bow Mountains, the Big Horn Mountains, and the Yellowstone-Teton-Highlands in Wyoming (see figure 8 in Cage and Cooper 2013; Richmond 1986; Wohl et al. 2007). Glaciation was a major factor in creating the wide variety of wetland habitats in the Rocky Mountains (McKinstry et al. 2010).

Localized ice caps and valley glaciers from the Pleistocene and the Holocene have influenced physiographic features in alpine and subalpine valleys of the Rocky Mountains (Gage and Cooper 2013; Pierce 2003; Veblen and Donnelly 2005). The most commonly understood impact of glaciers is the formation of a deep U-shaped valley, with a wide flat valley floor. Glacial retreat can result in the formation of kettle basins, small depressions resulting from the melting of buried ice masses (Molnia 2013). Cirques, formed through glacial erosion, are the half-open steep-sided amphitheater-shaped hollows at the head of a valley or on a mountainside. Tarns, which are lakes that form in the basin of a cirque, and mountain lakes are also a result of glaciation. Melt from existing glaciers or permanent snowfields is captured in the depressions of cirques, tarns, and mountain lakes. Water flow is influenced by glacial moraines.

Geomorphic and hydrological processes vary between these glacially formed valleys and the fluvially formed valleys (those formed by river dynamics). Streams within glaciated valleys have a snowmelt-dominated hydrograph; seasonal flooding is primarily from snowmelt. In contrast, lower elevation streams in fluvial valleys are subjected to rainfall and flash floods as well as snowmelt (Livers and Wohl 2015). Channel geometry characteristics are more variable in glaciated valleys than in reaches in fluvial valleys. Streams in glaciated valleys inherit the topographic conditions from glacial processes in the Holocene and Pleistocene and are not necessarily adjusted to current fluvial sediment, water, and disturbance regimes. In unglaciated valleys, the stream channel formation is likely to have co-occurred with the historical and current sediment, water, and disturbance regimes (Livers and Wohl 2015).

Glaciated valleys are characterized by a harsh climate, with short growing seasons, high snowfall and snowpack, and cold temperatures—all of which are associated with alpine and spruce-fir ecosystems. Streams, lakes, and wetland areas in glaciated valleys create a heterogeneous landscape that provides a diversity of habitats for plants, invertebrates, and animals. Invertebrates include benthic macroinvertebrates, small animals that live among the sediment and stones at the bottom of streams, ponds, and lakes (i.e., the benthos). These animals can be insects, including mayflies (order Ephemeroptera), stoneflies (order Plecoptera), caddisflies (order Trichoptera), beetles (order Coleoptera), midges (family Diamesinae), crane flies (*Tipula* spp.), and dragonflies (suborder Anisoptera). Other members of the benthic macroinvertebrate community are snails (class Gastropoda), clams (class Pelecypoda), aquatic worms (class Oligochaeta), and crayfish (*Cambarus* spp.). These benthic macroinvertebrates provide essential ecosystem services, such as sediment mixing and the acceleration of detrital decomposition, nutrient cycling, and energy flow through food webs (Covich et al. 1999). Amphibians are food sources for predators such as birds, mammals, and fish, and can include species such as boreal toad (*Anaxyrus boreas*), boreal chorus frog (*Pseudacris maculata*), barred tiger salamander (*Ambystoma mavortium*), and Columbia spotted frog (*Rana luteiventris*) (Corn et al. 2011). These high elevation areas are often too cold to support native trout populations.

A diversity of wetland habitats can be found in glaciated valleys. The valley floors may have well-developed soil in the lowest gradient reaches, where soils and detritus from erosional processes in the upper hillsides are deposited. The riparian areas are strongly influenced by the unidirectional flow of water in the streams (Cage and Cooper 2013). Groundwater-influenced fens of the Rocky Mountain Region are most abundant in glaciated valleys (Gage and Cooper 2013). Wet meadows are also found in glaciated valleys; in contrast to fens, they typically do not have perennially high water tables. This diversity of wetland habitats results in highly diverse plant taxa distinct from lower elevations (Cooper and Andrus 1994; Vance et al. 2012). On the Grand Mesa in northwestern Colorado, Austin and Cooper (2016) identified 16 different wetland habitat communities including fen, modified fen, marsh, and wet meadows. Dominant life forms across these communities ranged from semiaquatic species, small sedges, large sedges, and floating mats and bryophytes, to shrubs and trees. Peatland expanses in the Wind River Range in Wyoming were dominated by mountain sedge (*Carex scopulorum*), silvery sedge (*Carex canescens*), fewflower spikerush (*Eleocharis quinqueflora*), Boott Northwest Territory sedge (*Carex utriculata*), and water sedge (*Carex aquatilis*) (Cooper and Andrus 1994). Trees such as Engelmann spruce (*Picea engelmannii* Parry ex Engelm.), subalpine fir (*Abies lasiocarpa* (Hook.) Nutt.), and lodgepole pine (*Pinus contorta* Douglas Ex. Loudon) can be present in glaciated valleys.

Aquatic, riparian, and wetland ecosystems in glaciated valleys have been assessed for their vulnerability to climate change. As discussed in Chapter 1, the vulnerability framework has two major components, one related to nonclimate stressors and one related to climate stressors. Within the nonclimate component, four criteria were used to define factors of vulnerability: current status of ecosystem extent; intrinsic resilience of the ecosystem to nonclimate stressors, human influences on the ecosystem, and future trends of nonclimate stressors on the ecosystem. Within the climate change component, seven criteria capture the vulnerability of individual species, sensitivity of ecosystem dynamics such as dependence on a specific hydrological cycle, adaptive capacity of the ecosystem, potential for management to mitigate the effects of climate and nonclimate stressors, and interaction between climate change and nonclimate stressors. More information and the rationale for ranking ecosystem vulnerability by using these criteria are given in appendices A, B, and C.

Glaciated valleys are increasingly being studied on national forest lands as well as in other areas throughout the Rocky Mountain Region. We draw from this literature where studies are within the Rocky Mountain Region. Even within this Region, these glaciated valleys have great variation. Application of these results to glaciated valleys on individual national forests will require an understanding of local site characteristics and the site characteristics of the research studies reviewed here.

Vulnerability of Aquatic, Riparian, and Wetland Ecosystems in Glaciated Valleys to Nonclimate Stressors

Summary of Key Vulnerabilities to Nonclimate Stressors

BOX 4.1

Current status of ecosystem extent

With limited extent in the Rocky Mountain Region, the aquatic, riparian, and wetland ecosystems in glaciated valleys have very high vulnerability.

Human influences on ecosystem

Glaciated valleys are considered to have moderate vulnerability to the legacy of past human influences and to the stressors associated with ongoing human influences.

- Ecosystems have not completely recovered from early 19th-century mining and livestock grazing.
- Roads, grazing, and recreation have hindered and sometimes continue to hinder hydrological and wetland function.
- Invasive terrestrial and aquatic species appear to be limited. Pathogens such as chytrid fungus are affecting amphibian populations.
- Atmospheric nitrogen deposition has altered lakes and has potential to alter terrestrial and stream ecosystems.
- Dust-on-snow events alter snow chemistry and result in earlier snowmelt, notably in southwestern Colorado.

Intrinsic resilience of ecosystem to nonclimate stressors

A very high vulnerability ranking is given for the intrinsic resilience of aquatic, riparian, and wetland ecosystems in glaciated valleys to nonclimate stressors.

- Factors that enhance the resilience of aquatic, riparian, and wetland ecosystems in glaciated valleys to nonclimate stressors:
  - Life history strategies facilitate adaptation to fluctuating environments.
  - Functioning riparian and wetland habitats can maintain resilience to land use effects that cause erosion.
- Factors that lower the resilience of aquatic, riparian, and wetland ecosystems to nonclimate stressors:
  - Increases in atmospheric nitrogen deposition, alterations in snowmelt dynamics, and presence of aquatic and terrestrial invasive species compromise these ecosystems.
  - This ecosystem is susceptible to altered function from disturbances such as nitrogen deposition and other human-caused disturbances.

Future trends of nonclimate stressors

Glaciated valley ecosystems have very high vulnerability to future nonclimate stressors: recreation use, invasive species, and atmospheric nitrogen deposition.

Current Status and Human Influences

Aquatic, riparian, and wetland ecosystems in glaciated valleys of the Rocky Mountain Region cover a small fraction of area in Colorado and Wyoming high-elevation mountain valleys. Historical land uses in glaciated valleys included mining, livestock grazing, beaver (*Castor canadensis*) trapping, timber harvest, and log floating. Placer mining for gold and other precious metals was a common practice beginning in the mid-1800s (Wohl 2006). High elevation mining disturbed stream channels, and caused the release of toxic materials (Wohl 2006, 2011). Aquatic species and channel function have somewhat recovered from these early Euro-American activities (Wohl 2006), but researchers are still seeing elevated metal concentrations downstream of mines (Schmidt et al. 2012). Land uses in glaciated valleys share a similar history with low-gradient streams in high elevation areas (see Chapter 6).

Glaciated valleys in the Rocky Mountain Region are currently grazed by many of the same herbivores found in the montane, subalpine, and alpine (see chapters 3 and 5). Overgrazing by domestic livestock in the 19th and early 20th century has had negative effects in most Rocky Mountain Region riparian areas (Wohl 2006) by compacting soils; reducing riparian vegetation cover, leading to erosion; reducing aquatic invertebrate production; causing siltation of stream bed channels; destabilizing channels; and introducing noxious weeds (Cannon and Knopf 1984; Saunders and Fausch 2012; Winters et al. 2004a). Heavy grazing during the early 20th-century summer seasons reduced shrub density and narrowed riparian areas in Colorado glaciated valleys, and these effects lasted for decades after grazing intensity had been reduced (Cannon and Knopf 1984). Livestock grazing mostly by sheep, but also by cows and horses, continues on some high-elevation national forest allotments (USDA FS 2016). Generally, grazing effects—from both domestic and wild animals—vary with the intensity and timing of grazing activities as well as with species, land use, grazing legacy, and current grazing management system.

Roads connecting populated areas to national forest lands are widespread in the Rocky Mountain Region. Roads provide important access for a suite of economic and social uses, including recreation, hunting, camping, and skiing. However, the effects of roads include animal mortality from road development, mortality from collision with vehicles, modification of animal behavior, alteration of the physical environment, alteration of the chemical environment, spread of invasive plant species, and increased use of areas by humans (Hauer et al. 2016; Mortensen et al. 2009; Pollnac et al. 2012; Trombulak and Frissell 2000). Roads can alter drainage patterns in watersheds and increase surface runoff from soil compaction. Roads can intercept overland water flow that is important for healthy function and maintenance of wetland habitats. Road culverts can block aquatic passage, and thus fragment aquatic species habitats. Native fish species in the Upper Colorado Basin have declined because of more intense land use and greater road density (Dauwalter et al. 2011). Many roads are currently designed, managed, and maintained to reduce erosion, soil compaction, and contaminants; managers are also decommissioning roads to benefit stream and riparian areas (Winters et al. 2004a).

Recreation demands have increased over the past decade (USDA FS 2012). Recreation in high elevation ecosystems of the Rocky Mountain Region can hinder the functioning of aquatic, riparian, and wetland ecosystems. On the Big Horn National Forest, recreational activities, especially dispersed camping, were closely related to water in streams, valley bottoms, and lakes in wilderness areas (Winters et al. 2004b). Seventy percent of the mapped dispersed campsites were within 300 ft of a road on the Big Horn National Forest (Winters et al. 2004a). Visitor use can result in trampled alpine plants, cause erosion, and set back restoration efforts for decades (Willard et al. 2007). Ski areas that make artificial snow alter local hydrological function by removing water from streams, thus creating low winter streamflows for fish (Baron 2002).

Invasive and Nonnative Species

The presence of invasive species appears to be limited in aquatic, riparian, and wetland ecosystems in glaciated valleys, but more scientific documentation is needed. Although invasive plant species in high elevation areas have expanded globally over the last decade (McDougall et al. 2011), the common dandelion (*Taraxacum officinale* F.H. Wigg) was the only introduced species found in a survey of Rocky Mountain National Park (Ashton 2010). The weevil *Rhinocyllus conicus* Fröelich has been found on native thistles (*Cirsium* spp.) at Niwot Ridge in Colorado, but its presence was negatively correlated with elevation. Climatic factors apparently limited its current capacity to expand above treeline; warming temperatures could facilitate an upward expansion (Hicks et al. 2013). Chytrid fungus (*Batrachochytrium dendrobatidis*) is widespread in the Rocky Mountains and amphibians are not resistant to this disease (Muths et al. 2008). Disease detection rates were lower in amphibians inhabiting higher elevation sites (42 percent of clusters at higher elevation sites in Colorado and Wyoming vs. 84 percent of clusters at sites in lower elevations in Montana, Idaho, and Wyoming). Muths et al. (2011) suggested that when the disease is present, survival and recruitment may be compensatory in boreal toad populations. Where the pathogen was emerging, boreal toad populations had high recruitment rates, possibly compensating for low survival, and this combination resulted in a relatively slow rate of decline. In contrast, populations with no evidence of disease had high survival probability but lower recruitment rates.

Perennial invasive aquatic species are mostly absent in high-elevation glaciated valleys, and it is unclear if these snow-dominated areas would support many invasive species. Although high elevation lakes are often seasonally stocked with nonnative trout (*Salvelinus* spp.; Colorado Parks and Wildlife 2007), the environment tends to be nonsustaining due to very cold temperatures. Where native (*Onchorhynchus* spp.) and nonnative trout share the stream habitat, nonnative trout compete with, and in some cases, outcompete native fish, especially where stream conditions are degraded, angler harvest is high, temperatures are warmer, or a combination thereof (Dunham et al. 2004; Gresswell 2011).

Atmospheric Nitrogen Deposition

Researchers have linked human activity to an increase in atmospheric nitrogen deposition in high-elevation terrestrial and aquatic ecosystems (Baron et al. 2000; Benedict et al. 2013; Bowman et al. 2014; Nanus et al. 2012). From 1993 through 2012, significant trends in snowpack and wetfall chemistry mirrored changes in emissions of ammonia, nitrogen oxides, and sulfur dioxide (Ingersoll et al. 2016). Downward trends were seen in emissions of nitrogen oxides and in the regional trends in snowpack and wetfall concentrations of nitrate. However, ammonia emissions and ammonium concentrations and deposition in snowpack and wetfall all showed upward trends (Ingersoll et al. 2016).

Slow rates of nutrient cycling and low biomass production of alpine ecosystems reflect limited availability of nitrogen. Given adaptation to low nitrogen levels, these ecosystems have a low capacity to accommodate increases in nitrogen. Increased nitrogen deposition, through atmospheric deposition or by experimental changes, has been observed to increase soil nitrogen cycling, increase soil acidification, increase nitrate leaching, shift plant and microbial community composition, and increase cover of nitrogen-loving grasses in the terrestrial systems (Bowman et al. 2014). Changes in aquatic ecosystems from greater nitrogen deposition have resulted in higher nitrate concentrations in lakes (Baron et al. 2000), changes in diatom community assemblages (Wolfe et al. 2003), episodic acidification (Baron et al. 2011), and shifts in the nutrient status and productivity of lakes (Baron et al. 2011). Mast et al. (2014) examined three decades of stream nitrate export and atmospheric nitrogen deposition data from Loch Vale, a high elevation watershed in the Colorado Front Range. They documented the close relationship between the patterns of atmospheric nitrogen deposition and the temporal dynamics in stream chemistry. Similar patterns had been observed by Mast et al. (2011) in their study of sulfur deposition and stream chemistry. Effects of nitrogen deposition on aquatic systems are greatest on the eastern side of the Continental Divide in Colorado, but effects are also seen to the west (Wolfe et al. 2003).

Dust on Snow

Agricultural activities and drought to the west and southwest of the Rocky Mountains have caused increased intensity of dust-on-snow events in the Colorado Rocky Mountains (Painter et al. 2007; Toepfer et al. 2006). Dust-on-snow events have reduced duration of snow cover (Painter et al. 2007), increased rate of snowmelt associated with more extreme dust deposition, and produced earlier peak streamflow rates on the order of 1 to 3 weeks (Livneh et al. 2015; Painter et al. 2012; Steltzer et al. 2009). As a result of these dust-on-snow events, snow chemistry increased in pH, calcium content, and acid neutralizing capacity and the effects were more pronounced at upper elevations when compared to densely forested lower elevation sites (Rhoades et al. 2010) (see also Dust on Snow section in Chapter 3).

Intrinsic Resilience of Ecosystem to Nonclimate Stressors

Factors That Enhance Resilience to Nonclimate Stressors

Variation in life history strategies has helped aquatic and riparian species to persist in these high elevation habitats. Amphibians and invertebrates are adapted to fluctuating environments, through either the ability to rapidly colonize a site or the capacity to withstand very cold temperatures (Ryan et al. 2014). Functioning riparian areas serve to enhance resilience to nonclimate stressors. These moister areas are generally less at risk from fire (Dwire and Kaufmann 2003). Resilience is strengthened when functioning riparian areas are present to stabilize banks, and filter or modify water flows and matter moving through the landscape (Naiman et al. 2005).

Factors That Lower Resilience to Nonclimate Stressors

Rocky Mountain high elevation areas are not resilient to the effects of atmospheric nitrogen deposition, aerosol deposition from dust-on-snow events, introduction of nonnative fish, and human activities such as ditch construction or off-road vehicle use. Aquatic, riparian, and wetland ecosystems in glaciated valleys are sensitive to nutrient additions from natural and human-caused sources. These additions can alter ecosystem function. Bowman and Steltzer (1998) found that plants in moist meadows can use the increased nitrogen inputs from the atmosphere for growth, but this addition resulted in a change in plant dominance. Increased nitrogen deposition from human-caused sources promotes a trend toward oxygen depletion in some alpine lakes of the Colorado Front Range (Baron et al. 2000), increases acidity, and shifts species composition (Wolfe et al. 2003). The introduction of nonnative fish also weakens resilience of native species in the aquatic systems. Disruptions to the hydrological cycle affect these systems. Austin and Cooper (2016) reported that 79 percent of the fens surveyed on the Grand Mesa in western Colorado had been impacted by human activities such as ditch construction, drainage, flooding, or vehicular rutting. These disturbances, and factors such as water rights or the cost of restoration, led to the conclusion that there is little restoration potential for many fens.

Future Trends of Nonclimate Stressors

Increasing human populations in the Rocky Mountain Region will expand the urban-wildland interface, will increase the demand for ecosystem services from wildland areas, and are likely to increase several current stressors. Protected lands are altered by the human communities that surround them and by visiting recreationists' activities (Hansen et al. 2002; Wood et al. 2014). Recreational skiing may increase the demand for water to make snow, potentially reducing water in some glaciated valleys near these areas. Where roads exist in or near glaciated valleys, they may continue to alter plant and animal communities and the hydrology of these ecosystems. Although invasive species are currently rare in high-elevation glaciated valleys, some invasive species can adapt to high elevation environments (Pauchard et al. 2009). Other species introductions may occur, and travel corridors, changes in land use, and grazing improve their chances of spreading (Bradley et al. 2010; Pauchard et al. 2003). Atmospheric nitrogen deposition may reach critical levels by mid-century (Ellis et al. 2013), causing further shifts in plant community composition and changes in the nitrogen cycle. Increases in dust-on-snow events are likely to occur as agricultural activities continue and drought becomes more frequent in the southwestern United States.

Vulnerability of Aquatic, Riparian, and Wetland Ecosystems in Glaciated Valleys to Climate Stressors

Capacity for Network Shift

BOX 4.2

Key Vulnerabilities

A very high vulnerability is given for the capacity for network shift for glaciated valley ecosystems.

- Species associated with aquatic, riparian, and wetland ecosystems in glaciated valleys are unlikely to find aquatic and wetland habitats farther upslope or upstream of these high elevation ecosystems.
- Wetlands, ponds, and lakes in glaciated valleys can be connected to streams by both surface water and groundwater; alterations in these connections through changes in annual or seasonal precipitation or late season drying will affect the life history dynamics of plants and animals, especially hindering biota that have limited migration potential.
- Fragmentation of aquatic habitat could occur through changes in stream temperatures.

Elevation Potential for Habitat Shift

Glaciated valleys largely occur in the highest elevations of the Rocky Mountains in Colorado and Wyoming. Species associated with aquatic, riparian, and wetland ecosystems in these valleys are unlikely to find suitable habitats upslope or upstream; there is little potential for migration. Further, these high elevation valleys are isolated from each other by lower elevation landscapes, essentially preventing migration among the glaciated valleys.

Glacial mass is shrinking (Bell 2009; Cheesbrough et al. 2009) and retreating glaciers may provide area for colonization by plants and animals. Glaciers currently cover about 1.8 mi^2 in Colorado and about 28 mi^2 in Wyoming (Portland State University 2009). At the snout of a retreating glacier, newly formed streams develop. Insects (midges) and species that are attached to submerged surfaces (algae, cyanobacteria, microbes) rapidly grew and colonized these newly formed streams in the Wind River Range in Wyoming (Thompson et al. 2013). The local effect of climate change on glaciers varies according to elevation, size, shape, slope or aspect position, and local monthly temperature and precipitation. With climate change, researchers project that glaciers will melt entirely by 2030 in Glacier National Park in Montana (Hall and Fagre 2003).

Fragmentation

Glaciated valleys are part of a larger river network. Natural and human barriers can fragment or geographically isolate glaciated valleys. Isolation is a benefit when it serves as a barrier to downstream invasive species. It can be a detriment when stream habitat warms and organisms are unable to reach new habitat.

Wetlands, ponds, and lakes in glaciated valleys can be connected to streams by both surface water and shallow groundwater. These connections help aquatic organisms move between wetlands, ponds, and lakes in glaciated valleys. Changes in precipitation patterns or streamflows could disrupt these connections. Benthic macroinvertebrates can drift downstream and potentially out of glaciated valleys; winged adults move freely upstream as well as downstream (Graham et al. 2017). Amphibians can move between wetlands, ponds, and lakes of the ecosystem, as well as in and out of the glaciated valley. Seeds and regenerative stem segments can drift downstream. Plants have little ability to move upstream other than by wind or attached to the outside of birds and mammals or in their intestinal tracts.

Within the cold-water aquatic systems, habitat can become fragmented in these glaciated valleys. At the downstream end of valleys, terminal moraines can act as dams or create steep cascading stream segments (Gage and Cooper 2013) that block aquatic organisms from migrating upstream. Similarly, water diversion structures that occur downstream may fragment or isolate these upstream glaciated valley ecosystems. Fish movement can also be impeded during seasons in which streams have low flow, for example during fall and winter, or when ice blocks passage during late fall, winter, and early spring. This type of fragmentation is also found in low-gradient stream habitats (see Chapter 6).

Vulnerability of Cold-Adapted, Foundation, or Keystone Species to Climate Change

BOX 4.3

Key Vulnerabilities

A very high vulnerability ranking is given for cold-adapted, foundation, and keystone species in glaciated valleys, although vulnerability ranking for individual species can vary from moderate to very high.

- Given projections for increased warming and increased variability in precipitation and snowpack, amphibians are likely to have a high or very high vulnerability.
- Given increased warming and potential changes in streamflow with drier conditions, benthic macroinvertebrates are likely to have a high or very high vulnerability.
- Cold-water native fish have moderate to high vulnerability. New habitat with suitable temperature upstream may become available under a warmer climate. However, the smaller high-elevation tributaries, with more limited and disconnected habitats in glaciated valleys, may require assistance for populations to reach and survive in these areas.
- Riparian and wetland plants have moderate vulnerability. Plant communities may suffer from warming and drying.
- Pollinators have moderate vulnerability to climate change; however, under a changing climate, plant flowering may occur earlier than pollinator emergence.

We selected a subset of species that live in aquatic, riparian, and wetland communities of glaciated valleys that are foundation species (those whose high population presence substantially influences community structure), or keystone species (those exerting strong effects on the structure of their community, despite a low population presence). Amphibians, macroinvertebrates, and riparian plants were chosen as foundation species. Amphibians are important food sources for other species and are very sensitive to drying conditions. Macroinvertebrates were selected for their importance to aquatic food webs and provision of ecosystem services such as breaking down organic material. Riparian plants are a group of foundation species that substantially influence ecosystem function through biogeochemical cycling, regulate and filter runoff, and provide habitat for breeding, feeding, and sheltering of animals.

Pollinators and cold-water native fish were selected as keystone species. Pollinators have a symbiotic relationship with plants, but are a relatively small fraction of all life forms in these areas. Cold-water native fish, although rare in these ecosystems, are sensitive to warmer temperatures. Beaver, a keystone species that has been found in glaciated valleys, is not assessed here; for more information, see Chapter 6.

Cold-Adapted Amphibians

Recent declines in amphibian populations in the western United States are attributed to many factors including nonnative or invasive species, land use, overexploitation, climate change, ultraviolet radiation, contaminants, and emerging infectious diseases such as chytrid fungus (Hussain and Pandit 2012). Adams et al. (2013) analyzed the rate of change in the probability that amphibians occupy ponds and other comparable habitat features; they concluded, as have others, that amphibians are currently declining. Their analysis included species not typically considered of conservation concern, and suggested that these species may also be declining. Lack of sufficient data limits a robust conclusion on which factors, such as climate change, habitat loss, introduction of predatory nonnative fish, disease, and pollution, have directly or indirectly contributed to declining amphibian populations (Hossack et al. 2015; Hussain and Pandit 2012; Ray et al. 2016).

Amphibians are sensitive to temperature and moisture. Many species of amphibians rely on shallow or intermittently flooded wetland habitats for breeding; snowpack and runoff strongly influence the suitability of these wetland habitats (Amburgey et al. 2014; Corn 2005; Ray et al. 2016). Using climate and inventory data from Yellowstone and Grand Teton National Parks, Ray et al. (2016) concluded that models of amphibian breeding dynamics using climate variables outperformed models that were exclusively habitat based. Extinction rates were most sensitive to climate variation. Amburgey et al. (2014) reported that tadpoles of boreal chorus frogs were found more often in ponds with intermediate-length hydroperiods (length of time water remains in the pond), in contrast to ephemeral or permanent wetlands. Boreal chorus frog presence was negatively related to fish presence.

Future climate may pose a challenge for amphibians, as higher temperatures, reduced snowpack, lower soil moisture, and drought potentially decrease the occurrence of intermediate-length hydroperiods in wetland habitats (Amburgey et al. 2014; Ryan et al. 2014). Loss of these habitats may concentrate amphibians in permanent wetlands. If boreal chorus frogs shift to breeding in permanent wetlands, predation risk to tadpoles increases as fish are more likely to be found in these habitats. Ray et al. (2016) emphasized that the strength and direction of relationships between occupancy parameters and climate drivers differed across the three species they studied: boreal frog, western tiger salamander, and Columbia spotted frog. In addition, community relationships in these wetland habitats are complex. The type of habitat, type of predator, life stages of each species, and availability of other habitats will affect options for amphibians under a changing climate. Given the projections for increased warming and increased variability in precipitation and snowpack, amphibians are likely to have a very high vulnerability to climate change.

Cold-Adapted Macroinvertebrates

Different benthic macroinvertebrate species can be prey or predators. As prey, they are food for aquatic and terrestrial vertebrate consumers (fishes, birds). As predators, they may control the numbers, locations, and sizes of their prey (Covich et al. 1999). Benthic macroinvertebrates contribute to nutrient cycling and energy flow in these ecosystems. Herbaceous litter is an important source of energy to small streams. High-elevation benthic macroinvertebrates are shredders that break down organic matter. Other species also serve as collectors, further processing organic matter and thereby releasing nutrients bound up in this material (Covich et al. 1999; Vannote et al. 1980). Although such specialization might suggest the loss of these ecosystem processes if shredders or collectors are lost under climate change, few studies have explored the interaction among macroinvertebrates species in these high elevation valleys and how community structure affects these ecosystem services. Finn and Poff (2005) found that reach-scale environmental variables in four high-elevation Colorado streams were most important in defining communities; taxonomic variance and functional variance provided no greater understanding of benthic communities than did environmental variables. Finn and Poff (2005) also found that benthic communities differed more in the alpine than in the spruce-fir or lodgepole pine ecological zones and suggested this difference may be the result of increased isolation of streams in the alpine zone. Tronstad et al. (2016) also reported greater differences in stream invertebrate assemblages at higher elevations than at lower elevations in Grand Teton National Park.

Milner et al. (2016) explored statistical approaches to determine years significantly outside of reference conditions for benthic macroinvertebrate communities in three streams in the Wind River Wilderness Area in Wyoming. For most sites, diversity and community persistence trends that were not correlated with precipitation, appeared to be on trajectories away from a reference condition, suggesting further research is necessary to determine if a nonclimate stressor was acting on the stream. Trait-based responses (related to life history, including trophic level; morphology; mobility; ecology) to environmental conditions are complex and are more likely to be multivariate responses (Poff et al. 2006).

These species are sensitive to changes in temperature and changes in streamflow as well as to environmental disturbances (Pyne and Poff 2016). Macroinvertebrates in the Upper Colorado River Basin in southwestern Wyoming and western Colorado and the Great Basin are most sensitive to increases in July temperatures and reductions in snowfall, conditions projected under climate change (Poff et al. 2010). Because of the warmer temperatures, changes in streamflow, and drier conditions expected from climate change, macroinvertebrates are considered to have high vulnerability.

Cold-Water Native Fish

Many streams, lakes, and ponds of high elevation valleys do not support fish because of extremely low temperatures, barriers to migration such as terminal moraines, or high gradients that isolate them from downstream fish populations. Aquatic habitats in glaciated valleys may become more hospitable for native cold-water fish and artificially stocked fish species as temperatures warm (Cooney et al. 2005; Isaak et al. 2015; Roberts et al. 2013). At higher elevations, however, current populations are often more isolated (due to topography), tend to be smaller, and have lower genetic diversity and higher extinction risk (Hilderbrand and Kershner 2000). Such populations would most likely need human intervention to sustain their size and diversity (Cook et al. 2010; Hilderbrand and Kershner 2000). Consequently, cold-water native fish have moderate to high vulnerability to climate change (for more information, see Cold-Water Native Fish in Chapter 6).

Riparian and Wetland Vegetation

Plants associated with riparian and wetland habitats in glaciated valleys may face warming and drying that could inhibit their growth and survival in some areas. However, the high elevation of these valleys may buffer drying and warming, which may reduce plant sensitivity to climate change. Conditions are wetter in glaciated valleys, snowpack loss is buffered, and species are highly likely to survive and regenerate; therefore, the vulnerability of riparian and wetland plants to climate change is very likely moderate (for more information, see Chapter 6).

Pollinators

Pollinators are likely to persist under climate change in these glaciated valleys, but may be hindered by the direct effect of extreme climatic events (drought, extreme cold), and indirectly through the effects of climate change on plants in these valleys (Forrest 2014). Warming temperatures and the early melt of snowpack could result in plants flowering earlier than pollinators emerge, limiting food sources (Inouye 2008). However, pollinators are mobile and typically generalists; these capabilities may strengthen the adaptive capacity of pollinators to climate change. Researchers do not expect that pollinators, in general, will be severely hindered by shifts to earlier blooming of plants, or changes in the species composition of plant communities (Willmer 2012). Although there are research studies that identify potential vulnerabilities to pollinators under a changing climate, few of these studies have focused on identifying the major factors influencing pollinators under climate change at high elevations (for more information about pollinators and climate change, see Chapter 3). Based on the available information, pollinators have moderate vulnerability to climate change.

Sensitivity to Extreme Climatic Events

Sensitivity to Drought

Extreme events, such as drought, are expected to increase in frequency and magnitude under climate change (Field et al. 2012). At the level of a national forest, Sun et al. (2015) documented the impacts of the five worst droughts between 1962 and 2012. Across the Rocky Mountain Region, the most intense drought occurred in 75 percent of the national forests in 2012. Across all national forests, the top five droughts were associated with a 22-percent reduction in precipitation and a 9-percent reduction in gross primary productivity (Sun et al. 2015).

BOX 4.4

Key Vulnerabilities

A moderate ranking is given for the sensitivity of glaciated valley ecosystems to extreme climatic events:

- Aquatic, riparian, and wetland ecosystems in glaciated valleys are sensitive to drought and highly dependent on a snow-dominant hydrology; projections indicate snowpack will decline. However, snowpack loss at the highest elevations may be minimal, allowing glaciated valleys to continue to retain water throughout the year. These factors may buffer plants and organisms in glaciated valleys from drought. Additional research is needed.
- These ecosystems are only moderately sensitive to flooding given their small drainage areas in headwater areas; however, glacial outburst floods have potential to alter this ecosystem.
- High elevation ecosystems are buffered somewhat from extreme heat, but are very sensitive to warming that may exceed species tolerance limits.

Glaciated valley ecosystems are sensitive to drought and are likely to be sensitive under projected climate change. Droughts in the glaciated valleys can arise through lower snowpack, earlier melt of snowpack, or less precipitation (snow and rain). Less precipitation (snow or rain) results in declining soil moisture and runoff, and consequently lower flows in streams and lower potential to recharge groundwater. Using future climate projections, Dhungel et al. (2016) explored the potential for changes to occur in streamflow, for example, from perennial to intermittent or from snow-fed to rain-fed. Small and large perennial snow-fed streams in Colorado and Wyoming were included in their study. No changes in streamflow regimes were projected for these streams, in contrast to other parts of the United States. They acknowledged that further work exploring the subtler aspects of snow-fed flow regimes may be needed. Little quantitative information on projected drought is available for glaciated valleys in the Rocky Mountain Region.

Riparian vegetation and aquatic organisms can be stressed by less moisture. Life history traits, biotic interactions, and availability of water in ponds will influence the future success of aquatic insects. Wissinger et al. (2008) report that pond-drying can increase the risk of desiccation-induced, pre-reproductive mortality of caddisflies, as well as decrease adult body size at maturity. Life history traits, such as caddisfly species laying eggs on the water, may limit these species to permanent ponds under climate change (Wissinger et al. 2003). Species with traits such as rapid larval growth, ovarian diapause, and terrestrial oviposition of desiccation-tolerant eggs, may be able to use both permanent and temporary wetlands. Detrivorous insects, such as the larvae of caddisflies, supplement their diets with animal material, and Wissinger et al. (2004) concluded that supplementation can have strong effects on fitness, especially in temporary habitats that constrain larval development. These interactions could become more important under limiting moisture availability.

Sensitivity to Floods

If extreme precipitation and flooding increase in the Rocky Mountain Region, water retention capacity of most glaciated valleys may magnify flooding effects; however, the relatively small drainage area of these headwater basins would limit the amount of water contributing to flooding (Luce et al. 2012). Where glaciers are present, glacial outburst floods can occur and have the potential to alter valley bottoms. Such was the situation researchers observed in the Wind River Range: Meltwater, ponded by glacial ice from the Grasshopper Glacier, was suddenly released when the ice burst; the ensuing flood altered the valley bottom (Oswald and Wohl 2008). Glaciated valley aquatic and riparian habitats could have more floods with climate change but are likely to be only moderately sensitive to flooding.

Sensitivity to Extreme Heat

Extreme heat may have the most impact on stream habitats where species live at the edge of their heat tolerance levels, for example, benthic communities (Poff et al. 2010), salmonids (Isaak et al. 2015; Wenger et al. 2011), and Colorado River cutthroat trout (*Oncorhynchus clarkii pleuriticus*) (Roberts et al. 2013). Aquatic habitats in glaciated valleys are somewhat buffered from extreme heat because of their elevation, but species are likely to be sensitive to extreme heat that may happen with climate change.

Intrinsic Adaptive Capacity to Climate Change

BOX 4.5

Key Vulnerabilities

A high vulnerability ranking is given for the intrinsic adaptive capacity of glaciated valley ecosystems to climate change.

- Factors that strengthen adaptive capacity:
    - ○ Physical features and functioning riparian areas enhance moisture availability and adaptive capacity to drought.
    - ○ Biota and hydrological function recover relatively quickly following disturbance.
- Factors that weaken adaptive capacity:
    - ○ Connectivity among the wetlands, ponds, and lakes is helped by both surface water and groundwater; drier and warmer conditions may diminish these connections with implications for the diversity of habitats as well as population and community dynamics in glaciated valleys.

Factors That Strengthen Adaptive Capacity to Climate Change

Functioning riparian areas demonstrate the adaptive capacity of aquatic, riparian, and wetland areas in glaciated valleys. Riparian and wetland communities are diverse (Cooper and Andrus 1994; Gage and Cooper 2013; Vance et al. 2012), facilitating shifts in community composition in response to environmental changes (Bowman 2000). Low areas of these valleys tend to retain water in wetlands, lakes, and ponds following flooding and high flows (Naiman et al. 2005; Winters et al. 2004b), which strengthens adaptive capacity to drought. Riparian plants stabilize banks and provide food and habitat for aquatic and terrestrial species (Naiman et al. 2005). Biota and hydrological conditions recover relatively quickly after disturbances that disrupt function, such as riparian recovery after fire in the western United States (Dwire and Kauffmann 2003).

Factors That Weaken Adaptive Capacity to Climate Change

Drought and drying of aquatic, wetland, and riparian ecosystems could hinder plants and animals (Poff et al. 2010). Wetlands, ponds, lakes, and riparian areas in glaciated valleys tend to have lower productivity and population densities of plants and animals, which weaken the capacity to recover from disturbances that reduce populations. Amphibians, such as boreal toads, are sensitive to drying of ponds and lakes. In areas where riparian vegetation has been removed or damaged, adaptive capacity to floods, drought, and heat is weakened. Large woody and leaf litter inputs are necessary for insect and fish productivity. Riparian vegetation also benefits the ecosystem by shading streams and stabilizing streambanks (Naiman et al. 2005).

Dependence on Specific Hydrological Regime

BOX 4.6

Key Vulnerabilities

A high vulnerability ranking is given for dependence on the hydrological regime in glaciated valley ecosystems. Aquatic, wetland, and riparian ecosystems are dependent on a snow-dominated hydrological regime.

Aquatic, wetland, and riparian ecosystems in glaciated valleys are highly dependent on a snow-dominated hydrological regime. Regonda et al. (2005) found that snowpack at elevations above 8,200 ft stayed relatively unchanged from 1950 through 1999; in contrast, warmer temperatures at lower elevations caused snowpack loss. Klos et al. (2014) projected high-elevation precipitation regimes throughout the Rocky Mountains to remain relatively snow-dominated with lower elevations becoming more rain-dominated. Although snowpack loss is buffered in the high elevations of glaciated valleys, drying of wetlands and riparian areas could occur and hinder plants and animals. Loss of snowpack could contribute to drier conditions late in the growing season that stress plants and aquatic organisms in glaciated valleys.

Stream temperatures are projected to warm under a changing climate. Isaak and Rieman (2013) developed equations to calculate isotherm shift rates in streams that can be used to look at future warming scenarios and the associated shifts in stream temperatures. For example, for streams with an existing lapse rate of 1.4 °F per 328 ft and a projected warming of 0.2 to 0.4 °F per decade, the isotherms are projected to shift upstream at a rate of 0.08 to 0.8 mi per decade in steep streams (2–10 percent slope) and 0.80 to 15.5 mi per decade in flat streams (0.1–1 percent slope).

Glaciers contribute to the water balance of glaciated valleys. Edmunds et al. (2011) reported a volume loss of three glaciers in the Teton Range in Wyoming to be 3.20 ± 0.46 million m^{-3} (113.0 ± 16.8 million ft^{-3}) between 1967 and 2002. They identified summer (June, July, and August) temperature as the primary climatic driver in glacial melt by comparing historical temperatures from 1911 to 1967 and from 1968 to 2006. Over the period of study, no significant differences in snowpack were observed. The Wind River Range contains 269 glaciers and snowfields, including the largest glacier in the conterminous United States, the Gannet Glacier. Numerous studies have explored the dynamics of these glaciers and snowfields (Cheesbrough et al. 2009; DeVisser and Fountain 2015). DeVisser and Fountain (2015) reported that between 1900 and 2006 the ice-covered area of the 269 glaciers shrank by 47 percent. As with other studies, they identified air temperature as the primary climatic driver influencing this loss. Glaciers at lower elevations had a faster rate of shrinkage than those at higher elevations. The contribution of this glacial melt to late summer streamflow ranges from 0.4 to 10.9 percent across four different watersheds.

Most researchers agree that glaciers in the western United States will continue to retreat in the future, though the local effect of climate change on glaciers varies according to glacier elevation, size, shape, slope or aspect position, and local monthly temperature and precipitation (Hall and Fagre 2003). Changes in glaciers can be incremental, but glacier outburst floods can occur and potentially alter the rate and magnitude of change in valley bottoms (Oswald and Wohl 2008).

Likelihood of Managing Climate Change Effects

BOX 4.7

Key Vulnerabilities

Management options exist to promote resilience in aquatic, riparian, and wetland ecosystems, but these options are not always feasible or applicable in wilderness or remote areas, and may not succeed in mitigating effects from extreme heat and drought. Consequently, vulnerability is ranked as high.

Management options for mitigating the effects of climate change on aquatic, riparian, and wetland ecosystems include reducing current stressors, strengthening ecosystem resilience, and, as climate continues to change, helping plants and animals to adapt (Millar et al. 2007; Rieman and Isaak 2010; Shoo et al. 2011; Winters et al. 2004a; Wohl et al. 2005). Quantifying the water needs of riparian and wetland vegetation may be helpful in setting the basis for management options (Cooper and Merritt 2012). Management options that have been identified include:

- maintaining or restoring the natural hydrological regime;
- maintaining and restoring upslope plant cover to minimize flooding impacts;
- maintaining or restoring riparian areas, floodplains, and wetlands, and their connection to streams;
- reintroducing beaver;
- protecting or restoring critical or unique habitats that buffer survival during stressful conditions;
- disconnecting roads from the drainage network;
- limiting or stopping the introduction of nonnative species;
- eliminating or controlling pollutants or contaminants;
- removing or modifying barriers to fish movement;

- maintaining or reconnecting large networks of habitat;
- conserving or restoring diverse habitats across basins;
- with respect to amphibians, installing irrigation sprayers to manipulate water potentials at breeding sites and retaining or supplementing natural and artificial shelters (e.g., logs, cover boards) to reduce desiccation and thermal stress;
- manipulating canopy cover over ponds to reduce water temperature;
- creating hydrologically diverse wetland habitats capable of supporting larval development under variable rainfall regimes;
- targeted fish removals in wetland habitats to aid amphibians (Ryan et al. 2014);
- aiding species migration, such as transporting individuals to inaccessible habitat or places of refuge to maintain genetic diversity;
- allowing new species to colonize when the resident species can no longer survive, so that ecosystem processes continue and loss of nutrients and soil is minimized;
- reducing the impacts from livestock grazing;
- increasing cover of riparian vegetation to provide shading and maintain stream temperature;
- allowing the natural variability of stream movement across the landscapes; and
- minimizing soil compaction by limiting roads, trails, and campgrounds.

Management in these high-elevation glaciated valleys may be challenging; the remoteness of these glaciated valleys and the feasibility of reversing the effects of climate change may limit or prevent management actions. Many glaciated valleys lie within designated wilderness or identified roadless areas (Colorado Roadless Areas in Colorado and Inventories Roadless Areas in Wyoming), where interventions may not be allowed. Hotaling et al. (2017) suggested that a worthwhile goal for alpine streams is to identify and prioritize the protection of robust local populations of cold stenothermic species associated with meltwater habitat, as well as meltwater-associated habitats that are likely to be most resistant to climate change.

Potential for Climate Change to Exacerbate Effects of Nonclimate Stressors, or Vice Versa

BOX 4.8

Key Vulnerabilities

Climate change may exacerbate the magnitude, intensity, and effect of nonclimate stressors, such as atmospheric nitrogen deposition, dust-on-snow events, and introduction of invasive or undesirable native species, in glaciated valley ecosystems. Consequently, the vulnerability ranking is very high.

Drought, Dust on Snow, Warmer Temperatures, Atmospheric Deposition

Currently, drought, dust on snow, warmer temperatures, and atmospheric deposition affect glaciated valleys in different ways. Drought affecting a region far from these glaciated valleys, with the right weather conditions, can result in a dust-on-snow event in a glaciated valley. Atmospheric deposition from the eastern Plains of Colorado brings nitrogen and the risk of acidification to these historically nitrogen-limited ecosystems (Benedict et al. 2013). Warming temperatures across glaciated valleys may be speeding up the weathering processes and altering watershed chemistry (Heath and Baron 2014).

Climate change (increased drought, warmer temperatures) could potentially exacerbate the effects of dust on snow in high elevation ecosystems of the Rocky Mountain Region. Drought- affected areas of the southwestern United States have been the source of intensified dust-on-snow events in the Colorado Rockies. Temperature and drought are projected to increase, particularly in the southwestern United States. Human activities that disturb soil in dust-source regions are also likely to continue. Dust-on-snow events in the Upper Colorado River and surrounding basins of the Rocky Mountains have reduced duration of snow cover, increased the rate of snowmelt, produced earlier peak streamflow rates on the order of 1 to 3 weeks, and altered the chemistry of snow (see Current Status and Human Influences section). Deems et al. (2013) explored a future under climate change that included dust-on-snow events. While projected warming temperatures have a large impact on snowpack accumulation and melt, dust-on-snow events combined with future climate moved peak runoff earlier by 2 to 3 weeks, and peak flows at Lee's Ferry on the Colorado River in Utah decreased by 14 to 18 percent relative to the historical scenario. These results have important implications for water management, planning, and policy (Painter et al. 2012).

Climate change could exacerbate the effect of atmospheric nitrogen deposition in high-elevation aquatic systems. It has been documented that these aquatic systems are sensitive to nutrient additions from natural and human-caused sources as these additions shift aquatic community composition, increase the risk of aquatic oxygen depletion in lakes, increase alpine lake acidity, and alter water chemistry (see Current Status and Human Influences section). Flanagan et al. (2009) identified links between atmospheric deposition and the alpine lake response to drought. Response of the phytoplankton community was most strongly correlated with water quality changes associated with the drought, rather than temperature and hydraulic residence time, and the dominant post-drought phytoplankton species were found to be associated with nitrate; both findings suggest the influence of atmospheric deposition in the aquatic response to drought (Flanagan et al. 2009). Mast et al. (2014) documented that stream chemistry has responded rapidly to changes in atmospheric nitrogen deposition over the last three decades at Loch Vale in Rocky Mountain National Park. Bowman et al. (2014) suggested that these dust-on-snow events, with introduction of base cation-laden dust, may have delayed the onset of soil and surface water acidification from atmospheric nitrogen deposition in the Rocky Mountains. Continued increases in atmospheric nitrogen deposition could eventually overwhelm this delayed response and result in soil and surface water acidification.

Warming temperatures are apparently having an increasing effect on weathering dynamics at the basin scale. Heath and Baron (2014) explored historical data collected at Loch Vale watershed. They concluded that increasingly warm summer temperatures are melting what was once permanent ice; elements entrained in the ice are released, resulting in a rapid change in the flux of weathering products and inorganic nitrogen from the basin.

Warming and Invasive Species

Warming may improve habitat conditions for some invasive species that have expanded into alpine areas (Ashton 2010). Invasive plants can displace native species, alter nutrient cycling, and disrupt food webs (Pauchard et al. 2003, 2009) (see also Current Status and Human Influences and Future Trends of Nonclimate Stressors sections).

Summary of the Regional Vulnerability of Aquatic, Riparian, and Wetland Ecosystems in Glaciated Valleys

The overall regional vulnerability ranking for the ecosystem is a combination of the vulnerability to nonclimate stressors and the vulnerability to climate stressors. For aquatic, riparian, and wetland ecosystems in glaciated valleys, the mean overall vulnerability ranking was very high, and mean expert reviewer rankings were in the high or very high category (fig. 4.1, table 4.1). Mean vulnerability rankings were very high for both the nonclimate stressor vulnerability and the climate vulnerability (fig. 4.1), with expert reviewer rankings in the high or very high category.

The expert reviewers had very high overall confidence in the ranking of the overall regional vulnerability (fig. 4.2). The four experts' individual rankings for overall confidence ranged from the high to the very high category (table 4.2). Mean confidence was very high for the nonclimate stressor vulnerability ranking and high for the climate vulnerability ranking. Individual experts' confidence rankings ranged from the moderate to the very high category for both the nonclimate stressor and climate vulnerability rankings. Calculations of mean vulnerability and confidence included only the four complete reviewer rankings.

Table 4.1—Original and reviewer rankings for the regional vulnerability assessment of aquatic, riparian, and wetland ecosystems in glaciated valleys in the Rocky Mountain Region. Underline indicates a reviewer score change from the original assigned score.

Criterion	Original score	(Reviewer 1) Score	(Reviewer 2) Score	(Reviewer 3) Score*	(Reviewer 4) Score	(Reviewer 5) Score	Reviewer criterion rank mean
Nonclimate							
1. Extent	5	5	5	3	5	5	Very high
2. Human influences	3	3	3	3	3	3	Moderate
3. Resilience	5	5	5	5	5	5	Very high
4. Future trends	5	5	5	3	5	3	Very high
Total	18 Very high	18 Very high	18 Very high	14 High	18 Very high	16 High	18 Very high
Climate							
1. Ecosystem shift	5	5	5	5	5	5	Very High
2. Species groups	5	5	5	3	5	5	Very High
3. Climatic events	3	3	3	3	4	3	Moderate
4. Adaptive capacity	3	5	3	*	4	3	High
5. Hydrology	5	5	5	5	5	1	High
6. Management	3	5	3	5	4	3	High
7. Interactions	5	5	5	5	5	5	Very High
Total	29 High	33 Very high	29 High	*	32 Very high	25 High	30 Very high
Overall vulnerability rank	24 Very high	26 Very high	24 Very high	*	25 Very high	21 high	24 Very high

* Calculation of mean vulnerability and confidence (table 4.2) included only the four complete reviewer rankings.

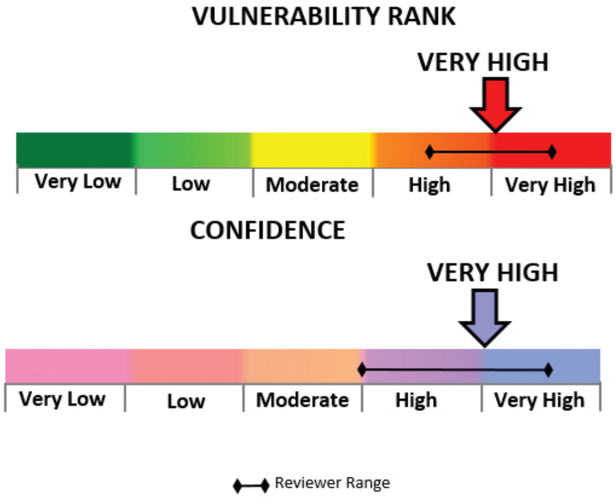

Figure 4.1—Overall regional vulnerability ranking and confidence ranking for aquatic, riparian and wetland ecosystems in glaciated valleys in the Rocky Mountain Region. Large arrow points to the mean score for vulnerability and for confidence. Bar represents the range of scores. Only four reviewer scores are included in mean calculations here and in tables 4.1 and 4.2.

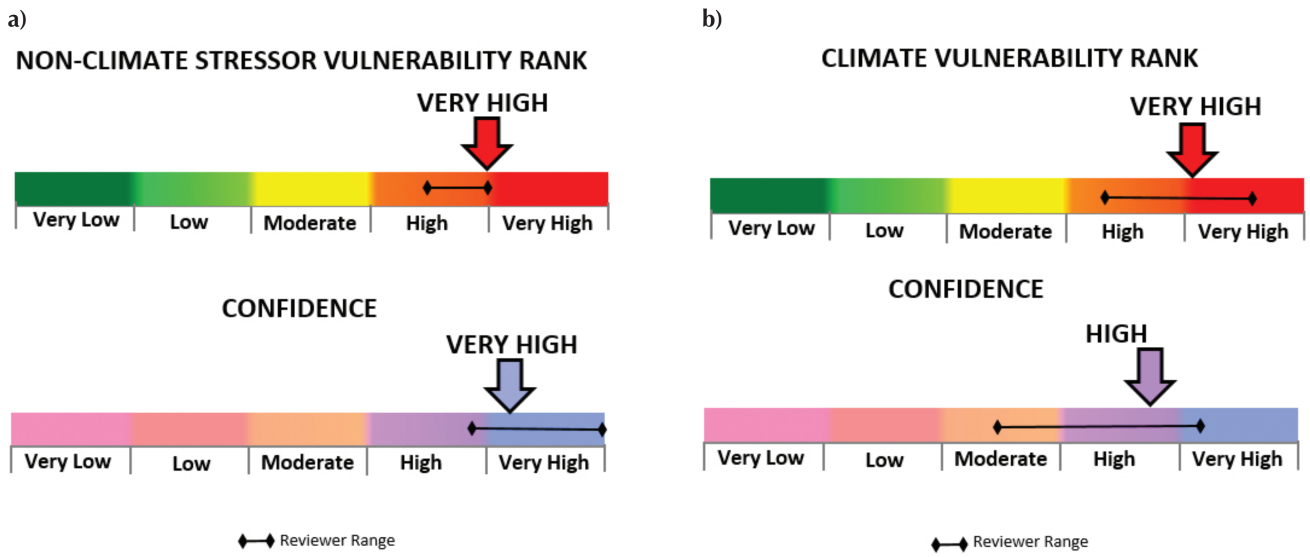

Figure 4.2—Regional vulnerability of aquatic, riparian, and wetland ecosystems in glaciated valleys to nonclimate (a) and climate (b) stressors in the Rocky Mountain region. Large arrow points to the mean ranking for vulnerability and for confidence. Bar represents the range of scores. Only four reviewer scores are included in mean calculations here and in tables 4.1 and 4.2.

Table 4.2—Reviewer rankings for confidence in the vulnerability assessment of aquatic, riparian, and wetland ecosystems in glaciated valleys in the Rocky Mountain Region.

Criterion	(Reviewer 1) Score	(Reviewer 2) Score	(Reviewer 3) Score *	(Reviewer 4) Score	(Reviewer 5) Score	Criterion rank confidence mean
Nonclimate						
1. Extent	5	5	3	5	5	Very high
2. Human influences	5	5	3	4	5	High
3. Resilience	5	5	3	4	3	Very high
4. Future trends	5	5	3	4	5	Very high
Total	20 Very high	20 Very high	12 Moderate	17 High	18 Very high	19 Very high
Climate						
1. Ecosystem shift	3	5	3	5	5	Very high
2. Species groups	5	5	3	4	3	High
3. Climatic events	3	5	3	4	3	High
4. Adaptive capacity	3	5	*	4	3	High
5. Hydrology	5	3	5	5	1	High
6. Management	5	3	3	4	3	High
7. Interactions	3	5	3	4	3	High
Total	27 High	31 Very high	*	30 Very high	21 Moderate	27 High
Overall confidence rank	24 Very high	26 Very high	*	24 Very high	20 High	24 Very high

* Calculation of mean vulnerability (table 4.1) and confidence include only the four complete reviewer rankings.

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Chapter 5. Subalpine Spruce-Fir Ecosystems: Vulnerability to Nonclimate and Climate Stressors in the U.S. Forest Service Rocky Mountain Region

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Quick Look: Subalpine Spruce-Fir Ecosystems in the Rocky Mountain Region

Subalpine spruce-fir ecosystems are moderately widespread in the subalpine zone of the Rocky Mountain Region. Climate is cold year-round with frost possible at any time. Precipitation is mainly in the form of snow. Winter snowpack and the associated slow melt-out in the spring serve as a reservoir for water in these ecosystems. The fire regime is characterized by long return intervals, and fires usually are stand-replacing. Vegetation is largely composed of mixed- or single-species stands of Engelmann spruce and subalpine fir. Lodgepole pine and quaking aspen also commonly grow in these ecosystems, especially in western Colorado. Moderately moist meadows intermix with spruce-fir ecosystems. On drier sites, spruce-fir ecosystems can grow with lodgepole pine, limber pine, Rocky Mountain bristlecone pine in Colorado, and whitebark pine in Wyoming. Moisture availability influences the shrub and herbaceous plant species associated with spruce-fir forests.

Wildlife can be year-round residents, and may include red squirrel, chipmunks, snowshoe hare, American marten, and, in Colorado, reintroduced Canada lynx. Elk, mule deer, bighorn sheep, mountain goat, black bear, and moose find habitat here in the summer. Numerous bird species use spruce-fir habitats, including the year-round resident, white-tailed ptarmigan, and the tropical migrants, flycatchers and hummingbirds.

Quick Look: Vulnerability of Subalpine Spruce-Fir Ecosystems in the Rocky Mountain Region

Vulnerability to nonclimate and climate stressors: Moderate

Confidence: High

Exposure: Warming temperatures will occur year-round. In winter and spring, warming temperatures will contribute to early snowpack melt and a longer growing season, both of which could result in late growing season moisture stress. High variability in timing of spring freezing events could also occur. Spruce-fir forest on drier sites will have greater exposure under climate change.

Current extent: Subalpine dry-mesic spruce-fir forests and woodlands occupy more than 4.5 million ac in the Rocky Mountain Region. The less common subalpine mesic-wet spruce-fir forest and woodland occupies more than 119,000 ac, mostly in northwestern Wyoming.

Sensitivity and adaptive capacity to climate change: Spruce-fir ecosystems are sensitive to warmer temperatures and drought, and to disturbances, such as insect outbreaks and fire, that could be exacerbated by warming and drying. The ability of Engelmann spruce and subalpine fir to successfully recolonize after these disturbances could be limited due to warming temperatures and moisture availability. Landscape changes in mature forest habitat under increased insect outbreaks could reduce available habitat for some wildlife. Warming can also stress wildlife species when their temperature limits are reached. Wildlife species with seasonal color coat molts will be increasingly vulnerable to predators as duration of snowpack changes. Where moisture availability and site conditions allow, spruce and fir trees potentially could move into higher elevations, but less area is available. Spruce-fir ecosystems may fragment and contract at the lower edge of their elevational range. Management options currently exist to increase the resilience of spruce-fir ecosystems to nonclimate stressors, such as insects and wildfire; however, large areas of the subalpine are inaccessible or occur in wilderness, limiting implementation of some management treatments.

Nonclimate stressors: Human activities of harvesting and road development have affected forest structure and composition, and landscape pattern. Mining impacts are localized. Future trends in recreation, atmospheric nitrogen deposition, and possibly dust on snow may impact ecosystem processes and species composition, but will not eliminate subalpine spruce-fir ecosystems in the Region. Climate change is likely to alter the dynamics and the effects of nonclimate stressors such as fire and insect outbreaks.

Introduction

Spruce-fir forests are the matrix forests of the subalpine zone, intermixing with mesic meadows and other forest types, in the Forest Service, U.S. Department of Agriculture (USFS) Rocky Mountain Region. Snow is the dominant form of precipitation and the snowpack can remain onsite for 6 to 8 months of the year. These forests are long-lived with major disturbances such as infrequent but stand-replacing wildfire, insect outbreaks, windstorms and windthrow, and avalanches. Land use has added other stressors: logging, grazing, road development, and recreation (Romme et al. 2009a).

Of the two spruce-fir ecological systems found in the Rocky Mountain Region, the subalpine dry-mesic spruce-fir forest and woodland type occupies by far the greater area in the Region: more than 4.5 million ac (Comer et al. 2003). The less common subalpine mesic-wet spruce-fir forest and woodland type covers more than 119,000 ac, mostly in northwestern Wyoming (fig. 5.1). The commonly used term “spruce-fir forest” typically focuses on the dry-mesic subalpine spruce-fir forest and woodland type that is prevalent in the Region. We henceforth group the two ecological systems and use the term “spruce-fir ecosystems” interchangeably with “spruce-fir forests.”

These forests are largely composed of mixed- or single-species stands of Engelmann spruce (*Picea engelmannii* Parry ex Engelm.) and subalpine fir (*Abies lasiocarpa* (Hook.) Nutt.) (Comer et al. 2003). Co-occurring lodgepole pine (*Pinus contorta* Douglas ex Louden) and quaking aspen (*Populus tremuloides* Michx.) are common, especially in western Colorado. On more xeric subalpine sites, lodgepole pine, limber pine (*Pinus flexilis* James), and Rocky Mountain bristlecone pine (*Pinus aristata* Engelm.) can occur within subalpine spruce-fir stands in Colorado. In Wyoming, whitebark pine (*Pinus albicaulis* Engelm.) can occur within subalpine spruce-fir stands. At the alpine-subalpine transition zone, spruce-fir forest can intermix with alpine bedrock and scree, ice field, fell-field, alpine turf, alpine dwarf-shrublands, and alpine meadows.

Common shrubs in subalpine spruce-fir ecosystems can include: blueberry, huckleberry, or whortleberry (*Vaccinium* spp.); serviceberry (*Amelanchier alnifolia* Nutt.); thimbleberry (*Rubus parviflorus* Nutt.); and willow (*Salix* spp.). The shrub Cascade azalea (*Rhododendron albiflorum* Hook.) is found in the subalpine spruce-fir in Colorado. Shrubs found only in Wyoming include western Labrador tea (*Ledum glandulosum* Nutt.) and pink mountainheath (*Phyllodoce empetriformis* (Sm.) D. Don). Herbaceous

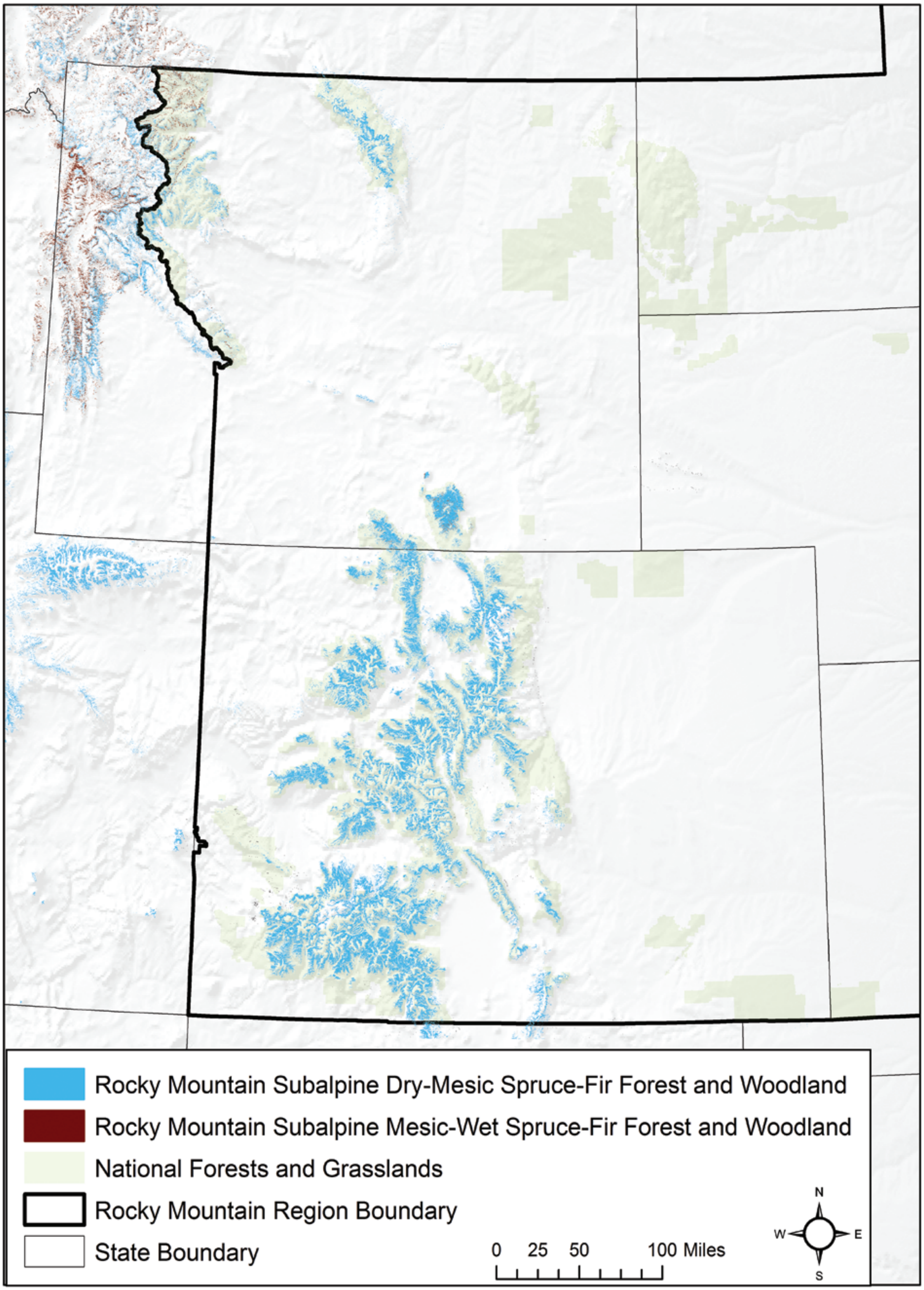

Figure 5.1—Subalpine dry-mesic (in blue) and mesic-wet (in red) spruce-fir forest and woodland distribution (Comer et al. 2003). Mesic-wet spruce-fir forests and woodlands only occur in Wyoming (northern Medicine Bow Mountains and Shoshone National Forest). (Data for Comer et al. 2003 can be found in NatureServe 2014.)

species include: red baneberry (*Actaea rubra* (Aiton) Willd.), starry false lily of the valley (*Maianthemum stellatum* (L.) Link), bunchberry dogwood (*Cornus canadensis* L.), sprucefir fleabane (*Erigeron eximius* Green), yellowdot saxifrage (*Saxifraga bronchialis* L.), and mountain bluebells (*Mertensia ciliata*). A herbaceous species found only in Wyoming is the Hitchcock's smooth woodrush (*Luzula glabrata* (Hoppe ex Rostk.) Desv. var. *hitchcockii* (Hämet-Ahti) Dorn). A common grass is bluejoint (*Calamagrostis Canadensi* (Michx.) P. Beauv.) (Comer et al. 2003). Xeric understory species that can occur in spruce-fir ecosystems include: common juniper (*Juniperus communis* L.), twinflower (*Linnaea borealis* L.), creeping barberry (*Mahonia repens* (Lindl.) G. Don.), and grouse whortleberry (*Vaccinium scoparium* Leiberg ex. Coville) (Comer et al. 2003).

Spruce-fir ecosystems provide food and shelter for permanent wildlife residents such as red squirrel (*Tamiasciurus hudsonicus*), chipmunks (*Tamias* spp.), snowshoe hare (*Lepus americanus*), and American marten (*Martes americana*), and Canada lynx (*Lynx canadensis*), which was reintroduced in Colorado. Summer habitat resources are used by large wildlife such as elk (*Cervus elaphus*), mule deer (*Odocoileus hemionus*), bighorn sheep (*Ovis canadensis*), mountain goat (*Oreamnos americanus*), black bear (*Ursus americanus*), and moose (*Alces americanus*). Numerous birds utilize spruce-fir habitats such as Clark's nutcracker (*Nucifraga columbiana*), ruby-crowned kinglet (*Regulus calendula*), nuthatches (*Sitta* spp.), juncos (*Junco* spp.), thrushes (*Catharus* spp.), mountain chickadee (*Poecile gambeli*), red crossbill (*Loxia curvirostra*), pine siskin (*Spinus pinus*), blue grouse (*Dendragapus obscurus*), white-tailed ptarmigan (*Lagopus leucura*), owls such as the boreal owl (*Aegolius funereus*), cavity nesting birds such as the three-toed woodpecker (*Picoides dorsalis*) or other woodpeckers, and migrating birds such as flycatchers (*Empidonax* spp.) and hummingbirds (*Selasphorus* spp.) (Uchytil 1991a,b).

Subalpine spruce-fir ecosystems have been assessed for their vulnerability to climate change. As described in Chapter 1, the vulnerability framework has two major components, one related to nonclimate stressors and one related to climate stressors. Within the nonclimate component, four criteria were used to define factors of vulnerability: current status of ecosystem extent, intrinsic resilience of the ecosystem to nonclimate stressors, human influences on the ecosystem, and future trends of nonclimate stressors on the ecosystem. Within the climate change component, seven criteria capture the vulnerability of individual species, sensitivity of ecosystem dynamics such as dependence of the ecosystem on the hydrological cycle, adaptive capacity of the ecosystem, potential for management to mitigate the effects of climate and nonclimate stressors, and interaction between climate change and nonclimate stressors. More information and the rationale for ranking ecosystem vulnerability by using these criteria are given in appendices A, B, and C.

Subalpine spruce-fir ecosystems are increasingly being studied on national forest lands as well as in other areas throughout the Rocky Mountain Region. We draw from this literature where studies are within the Region to assess the vulnerability of spruce-fir ecosystems to climate change at the scale of the Rocky Mountain Region. Even within this Region, these ecosystems have great variation. Application of these results to spruce-fir ecosystems on individual national forests will require an understanding of local site characteristics and the site characteristics of the research studies reviewed here.

Vulnerability of Subalpine Spruce-Fir Ecosystems to Nonclimate Stressors

Summary of Key Vulnerabilities to Nonclimate Stressors

BOX 5.1

Current extent

Subalpine spruce-fir ecosystems, though limited in area, are spread throughout the subalpine zone of the Rocky Mountain Region; their vulnerability ranking is moderate.

Human influences on ecosystem

Subalpine spruce-fir ecosystems are moderately vulnerable to the legacy of past human influences and to the stressors associated with ongoing human influences.

- Management since the mid-20th century has influenced forest structure and composition, and landscape pattern. Silvicultural activity, recreation, and fire control activities have increased road density.
- Mining activity has left a legacy of localized effects including compacted, displaced, and chemically contaminated soils.
- Currently, nonnative and invasive species occur minimally at high elevations.

Intrinsic resilience of ecosystem to nonclimate stressors

A moderate vulnerability ranking is given for the intrinsic resilience of subalpine spruce-fir ecosystems to nonclimate stressors.

- Factors that enhance subalpine spruce-fir ecosystem resilience to nonclimate stressors:
    - Plants and animals, such as understory vegetation, snowshoe hare, and red squirrel, are adapted to subalpine climate and the use of post-disturbance habitats.
- Factors that lower subalpine spruce-fir ecosystem resilience to nonclimate stressors:
    - Spruce-fir are susceptible to disturbances such as severe fires, insect outbreaks, wind events, and avalanches.
    - Tree regeneration and forest stand recovery are very slow following disturbance.

Future trends of nonclimate stressors

Subalpine spruce-fir ecosystems are considered to be moderately vulnerable to future nonclimate stressors, such as road development, increased demand for recreation, nonnative and invasive species, atmospheric deposition, and possibly dust-on-snow events.

Current Status and Human Influences

Stand-replacing fire, insect outbreaks, and windthrow events have been and continue to be the primary natural disturbances in the subalpine zone. From 1500 to 1850, the forested landscape across the northern Colorado Front Range subalpine zone would have consisted of very large patches of even-aged forests varying in composition from pure lodgepole pine or aspen to spruce-fir (Veblen and Donnegan 2005). Disturbances that promoted this pattern would have been infrequent, high-severity fires followed by successional replacement of species or recovery to the same dominant tree species according to site conditions and availability of seeds and sprouts. For south-central and southwestern Colorado, Romme et al. (2009a) described the subalpine landscape pattern as a coarse-grained mosaic of intermingling spruce-fir, aspen, cold-wet and cool-moist mixed conifer forests, bristlecone pine woodlands, meadows, grasslands, willow carrs, and sparsely vegetated rock fields. This landscape pattern was the result of stand-replacing fires at the large scale, periodic insect outbreaks at a finer scale, and, at the finest scale, windthrow events.

The past effects of Euro-American settlement in these high elevation forests have been less intense than effects at lower elevations in the Rocky Mountain Region (Romme et al. 2009a; Veblen and Donnegan 2005). Mining, logging, and road development were seen as the most significant impacts in the southern part of Colorado (Romme et al. 2009a,b) and in the Colorado Front Range (Veblen and Donnegan 2005). Although extensive grazing by domestic livestock occurred in subalpine meadows from the mid-to-late 1800s to the early 1900s, little information is available on impacts within subalpine forests. Some evidence suggests that spruce-fir tree species expanded into subalpine meadows during the mid-20th century during a period of heavy livestock grazing in the Wind River Mountains in Wyoming as lack of tree establishment coincided with cessation of grazing (Dunwiddie 1977).

Mining for gold, silver, copper, lead, molybdenum, zinc, and tungsten started in the mid-1800s. Mining activities include removal of subsurface materials and associated vegetation, and creation of tailings piles, all of which can alter hillslope stability (Wohl 2006). These effects remain as localized remnants of mines and tailings in subalpine areas of the Rocky Mountain Region. On the White River National Forest, previous mining activity compacted, displaced, and chemically contaminated soils and impaired water quality in many streams; soils in some areas are still not healthy or productive (USDA FS 2002). The long-term effects are primarily on the aquatic systems in the form of increased sediment yield to streams and pollution from heavy metals and other contaminants from mining (Schmidt et al. 2012; Wohl 2006).

In contrast to lower elevations, logging of spruce-fir forests was minimal before the 1950s. Logging was heaviest along the Front Range and southern Colorado during the 1950s to 1970s (Romme et al. 2009a; Veblen and Donnegan 2005). After 1950, foresters used clearcutting to reduce losses from windthrow and beetle kill and to lower costs (Alexander 1987). As clearcutting resulted in regeneration failure, foresters turned to partial cutting and small-patch clearcutting after the 1970s (Alexander 1987). After the 1950s, fire control and recreation interests increased development. Between 1950 and 1993, McGarigal et al. (2001) reported that the greatest impact on landscape structure on the San Juan National Forest in Colorado was a three-fold increase in road density to support timber harvest, fire control, and recreation. The harvest of trees along with road construction has continued to influence subalpine forest structure and composition (Romme et al. 2009b; Veblen and Donnegan 2005).

Invasive and Nonnative Species

Invasive and nonnative plants can influence dynamics of native plants, insects, and animals (Bradley et al. 2010). For example, fewer well-developed seeds were produced from flowers of two native species when hand-pollinated with pollen mixtures containing dandelion (*Taraxacum officinale*), suggesting an allelopathy effect (Loughnan et al. 2014). High elevations, including subalpine areas in the Rocky Mountain Region, have lower levels of nonnative plant species invasions than other areas (Averett et al. 2016; Banks and Baker 2011; Pauchard et al. 2009). Researchers have found common dandelion, Kentucky bluegrass (*Poa pratensis*), and smooth brome (*Bromus inermis*) in

Colorado Front Range subalpine areas (Coop et al. 2010). Low abundances of common dandelion, dark goosefoot (*Chenopodium atrovirens*), and many-flowered woodrush (*Luzula multiflora*) have been reported in the subalpine Medicine Bow Mountains in Wyoming (Musselman 1994). The invasive winter annual cheatgrass (*Bromus tectorum* L., also called downy brome) was found to be increasing in Rocky Mountain National Park in Colorado (Bromberg et al. 2011). Thirty-four nonnative species were recorded by Pollnac et al. (2012) in the Greater Yellowstone Ecosystem in northwestern Wyoming, spanning an elevational gradient from sagebrush to alpine ecosystems. As with other studies, they found nonnative plant richness decreased with increased elevation and increased distance from the road.

Atmospheric Nitrogen Deposition

Researchers have observed increased nitrogen deposition in high elevation areas of the Rocky Mountain Region. McDonnell et al. (2014) concluded that increased nitrogen deposition from 1900 to present has probably changed treeline vegetation. Simulating historical changes in atmospheric chemistry, they found a 25-percent increase in cover of subalpine fir saplings, mixed response in grasses, generally decreased abundance in forbs, and, specifically for the forb Ross' avens (*Geum rossii*), a reduction of 50 percent. Subalpine forests are still nitrogen limited even with these additions from human-caused sources (Baron et al. 2000; Burns 2004), but continued increases could result in less nitrogen limitation, possibly short-term increased productivity, and likely alterations in soil microbial communities leading to less soil carbon (Boot et al. 2016; Burns 2004). At low levels of nitrogen fertilization, measurable changes in the biogeochemistry of Engelmann spruce forests east of the Continental Divide were observed (Rueth and Baron 2002). If deposition continues to increase, plant species composition will continue to change (McDonnell et al. 2014), and the short-term increase in productivity may shift to a decline in productivity, as reported for other forested regions. Reaching nitrogen saturation will depend on atmospheric nitrogen deposition and concurrent changes in temperatures as well as on existing soil nitrogen levels, which vary across the subalpine zone (Boot et al. 2016; McDonnell et al. 2014; Rueth et al. 2003).

Fire

Fire occurrence in the subalpine is characterized by infrequent, high-severity crown fires that are primarily climate driven, with forest structure and topography affecting fire characteristics (Higuera et al. 2014; Romme and Knight 1981; Sibold et al. 2006). Fire suppression effects can include increases in fuel loadings and canopy cover at the stand level. However, fire return intervals in spruce-fir ecosystems are 100 to several hundred years long; hence it is difficult to detect whether fire suppression for the last 80 years has influenced the natural fire regime in the subalpine. Keane et al. (2002) concluded that fire exclusion has yet to significantly alter stand conditions and forest health in Rocky Mountain subalpine ecosystems. Buechling and Baker (2004) found longer fire intervals in Rocky Mountain National Park over the 20th century, and attributed this change to less severe drought impacts over this time compared to before the 20th century. Veblen and Donnegan (2005) found a modern pattern of less-frequent-than-expected fire along the Colorado Front Range. Conversely, a central and northern

Colorado Front Range study conducted near alpine treelines by Sherriff et al. (2001) found a slightly higher occurrence of fire, and attributed it to drought associated with La Niña events. It is possible that modern fire suppression efforts may have altered the fire return intervals at the landscape level in some subalpine forests (Kipfmueller and Baker 2000; Romme et al. 2009a; Veblen and Donnegan 2005). The naturally long fire return interval, variability of fire over landscapes, and interactions with drought make it challenging to determine if fire suppression efforts have significantly affected the fire regime. At this point, the fire regime is apparently within the range of natural variability.

Intrinsic Resilience of Ecosystem to Nonclimate Stressors

Factors That Enhance Resilience to Nonclimate Stressors

Resilience of spruce-fir ecosystems to disturbances, such as fire or beetle outbreaks, can be enhanced by variety in size and age of spruce and fir trees across the subalpine landscape. Disturbance processes, such as insect outbreaks, windthrow, and fire, affect the age and structural variety of stands (Alexander 1987; Veblen and Donnegan 2005). Variety across landscapes is also created in areas disturbed by snow avalanches, which can create fire breaks (Veblen et al. 1994). Wildfires may reduce susceptibility to spruce beetle (*Dendoctronus rufipennis*) outbreaks by reducing large Engelmann spruce trees (Bakaj et al. 2016).

These ecosystems are resilient to natural disturbance in the long term, and recover over time through changes in species that occur in plant communities. For example, pioneer species, such as lodgepole pine, or at higher elevations five-needle pines, are shade-intolerant and recover on the site quickly. Spruce and fir are shade-tolerant and seed into the understory of lodgepole trees (Minckley et al. 2012; Veblen 1986). The infrequent, large, and stand-replacing fires in subalpine Rocky Mountain forests have also been observed to increase plant understory diversity (Coop et al. 2010; Sibold et al. 2006).

Some wildlife and plant species are adapted to fire disturbance. Snowshoe hares apparently escape fire and suffer minimal mortality rates. After fire, they consume charred tree bark, and visit burned areas as plants regrow (Sullivan 1995a). Red squirrels and snowshoe hares and their predator, Canada lynx, use woody debris for dens (Ulev 2007). Red squirrels also use snags (Sullivan 1995b). Several understory plant species can resprout readily after low- or moderate-intensity fire that has killed the aboveground portion. Postfire sprouting is common for bunchberry dogwood as long as fires fail to raise soil temperatures appreciably or produce long-term soil heating (Gucker 2012a). In addition to sprouting from rhizomes, thimbleberry can emerge from seed in the soil on burned sites (Gucker 2012b). Serviceberry can sprout from the upper portion of the root crown, or if that has been killed, from rhizomes farther below the surface (Fryer 1997). Additional information about plant species and fire effects can be found at the Fire Effects Information System (USDA FS 2017).

Continued increases in atmospheric nitrogen deposition could lessen the current nitrogen limitation for subalpine plants, possibly increasing productivity (Burns 2004); however, long-term implications for subalpine forests are unclear. Dust on snow has been documented to affect the pH and raise acid neutralizing capacity of snow samples, but additional research is needed to assess the impact of atmospheric deposition as well as wind-blown deposition on snowpack chemistry, snowpack duration, plant communities, nutrient and water relations, and quality of the water exported from high elevation ecosystems (Rhoades et al. 2010).

Factors That Lower Resilience to Nonclimate Disturbances

Spruce-fir forests have historically burned in infrequent, stand-replacing fires that return about every 100 to several hundred years (Keane et al. 2002; Veblen 2000). Engelmann spruce and subalpine fir have shallow roots, and thin, less fire-resistant bark that makes them susceptible to fire (Alexander and Shepperd 1990; Alexander et al. 1990; Schoennagel et al. 2004). Recovery periods for trees are long following mortality from disturbances. Old-growth spruce-fir stands may take more than 300 years to develop after disturbance (Aplet et al. 1988; Veblen et al. 1989). Although recovery is slow, local reproduction is common and may occur by seeding, or by layering (when low branches touch the soil and take root) where forest cover is developing (Alexander and Shepperd 1990; Alexander et al. 1990). Disturbances such as severe fires, insect outbreaks, avalanches, and windthrow can reduce plant and animal habitat as well as cause mortality.

Mature spruce-fir forests have a high likelihood of insect outbreaks with associated mortality. Spruce beetle outbreaks have affected more than 1 million ac of spruce forest in Colorado and Wyoming since 1996 (Harris et al. 2013). Subalpine fir decline, a complex that includes two species of root disease fungi and western balsam bark beetle (*Dryocoetes confusus*) (McMillin et al. 2003), has affected more than 180,000 ac of this ecosystem in Colorado (Colorado State Forest Service 2012). Engelmann spruce and subalpine fir are also susceptible to other insects and pathogens, such as western spruce budworm (*Choristoneura freemani*), balsam fir bark beetle (*Dryocoetes confusus* Swaine), and broom rust (caused by the fungus *Chrysomyxa arctostaphyli*) (Alexander and Shepperd 1990; Alexander et al. 1990).

Future Trends of Nonclimate Stressors

Subalpine spruce-fir ecosystems are considered to be moderately vulnerable to future nonclimate stressors associated with human population growth and related infrastructure and road development, increased demand for recreation, atmospheric nitrogen deposition, and possibly dust-on-snow events. Although nonnative and invasive species occur minimally at high elevations in the Rocky Mountains, their spread threatens to disrupt ecosystem function, reduce biodiversity, displace native species, and disrupt pollination processes (Bradley et al. 2010; Jones 2004; Pauchard et al. 2009).

Vulnerability of Subalpine Spruce-Fir Ecosystems to Climate Stressors

Capacity for Range Shift

BOX 5.2

Key Vulnerabilities

A moderate vulnerability ranking is given for the capacity of spruce-fir ecosystems for range shift.

- Trees in southern Colorado spruce-fir forests are close to the southern limit of their current distribution and would have greater vulnerability to a changing climate.
- Overall, spruce-fir ecosystems may contract; however, expansion of tree species into higher elevation areas under a warming climate is possible if sufficient moisture is available and where microsite, geomorphic, and soil conditions allow.
- Spruce-fir forest species may expand into associated subalpine ecosystems, such as subalpine meadows.

Southern Limit Proximity

Spruce-fir ecosystems are moderately widespread throughout the subalpine areas of the Rocky Mountain Region (fig. 5.1), covering about 20 percent of the area above an elevation of 8,200 ft. The range of the spruce-fir types considered here encompasses the Cascades and Rocky Mountains, extending from British Columbia and Alberta in the north, to northern New Mexico in the south (Comer et al. 2003). Trees in southern Colorado spruce-fir forests are close to the southern limit of their current distribution and would have greater vulnerability to a changing climate.

Elevation Potential for Range Shift

Subalpine spruce-fir ecosystems cover high-elevation mountain areas of Wyoming and Colorado starting at about 9,000 ft and extending up to treeline at about 11,500 ft (Comer et al. 2003). Establishment of Engelmann spruce or subalpine fir at the interface between upright trees and alpine depends on seed dispersal by local sources or birds and wind, and sufficient moisture for seed germination in a microsite where climate and soil support establishment (Malanson et al. 2007). Treeline expansion has been observed to be limited by moisture availability during dry years (Villalba et al. 1994), topography, alpine microsite conditions, and geomorphic and soil factors (Macias-Fauria and Johnson 2013; Weisberg and Baker 1995). Elliott (2012) documented that regional climate variability (moisture and temperature) can initiate abrupt changes in tree establishment at broad regional scales (New Mexico to Montana); an abrupt increase in regional tree establishment coincided with a shift toward reduced cool-season precipitation. Under a changing climate, warmer temperatures may enhance subalpine tree expansion, but without adequate moisture availability and suitable site conditions, tree establishment will be inhibited. Although this ecosystem has the potential for upward elevational shifts, it may contract at the lower edge of its elevational range (Notaro et al. 2012; Rehfeldt et al. 2006, 2012). Because less area is available at higher elevations where this ecosystem may expand, subalpine spruce-fir ecosystems may contract overall.

Connectivity

Within the subalpine zone of the Rocky Mountain Region, spruce-fir forests are relatively connected (fig 5.1). Therefore, range shifts within the subalpine zone are not likely to be constrained, provided suitable site conditions exist. Spruce-fir forests at the southern range limit of these trees have smaller, more fragmented patches not typical of similar forests at higher latitudes, and range shifts may be the most constrained in these areas. Along with site conditions and the degree of fragmentation, climate may be a factor supporting spruce-fir expansion into subalpine meadows (Hessl and Baker 1997).

Vulnerability of Cold-Adapted, Foundation, or Keystone Species to Climate Change

BOX 5.3

Key Vulnerabilities

A moderate vulnerability ranking is given for cold-adapted, foundation, and keystone species in subalpine spruce-fir ecosystems.

- The foundation species Engelmann spruce and subalpine fir are likely to shift ranges but persist under climate change.
- Herbaceous understory plants may be hindered by earlier warming and higher spring temperatures coupled with variability in freeze dates, which could limit reproduction and productivity. However, these species are expected to persist on the landscape.
- Habitat suitability may increase for some bird and mammal species and not for others. Cavity nesting birds may find increased, albeit transient, availability of snags and food from more fire and insect outbreaks. Some species may have more reproductive success with warmer spring temperatures and earlier snowmelt. Population declines or stress on species, such as the boreal owl and snowshoe hare, is likely with warmer temperatures.

We selected a subset of species that occupy spruce-fir ecosystems to capture a range of environmental tolerances and to evaluate how their vulnerability may affect ecosystem function. Engelmann spruce and subalpine fir are selected as they are both cold-adapted and foundation species that have substantial biomass and influence within spruce-fir ecosystems. Understory shrub, forb, and grass species are also included as they are cold-adapted species, important food resources, and habitat for wildlife living in or migrating to these ecosystems. Bird and mammal species are selected as many are cold adapted. We identify groups of species, rather than specific individual species, because plant and animal species differ across spruce-fir forests from southern Colorado to northern Wyoming.

Engelmann Spruce and Subalpine Fir

Engelmann spruce and subalpine fir, alone or mixed, dominate the tree canopy of subalpine spruce-fir ecosystems. A variety of climate and species models suggest that spruce-fir ecosystems will contract overall in their extent, but continue to have a presence in the Rocky Mountain Region (Mathys et al. 2016; Notaro et al. 2012; Rehfeldt et al. 2006, 2012; Schrag et al. 2008). Differences in results are tied to how site and growth factors are modeled. When the modeling framework includes elevated carbon dioxide, researchers found reduced stresses associated with climate change and expansion of high elevation forests in the Colorado Rockies (Notaro et al. 2012). However, field research by Hu et al. (2010) showed that the fertilizing effect of increased atmospheric carbon dioxide was more than counterbalanced by the increased water stress due to longer growing seasons, suggesting that the mitigating factor of elevated carbon dioxide may not persist in the future. Using individual bioclimate models based on forest inventory plot data, Gray and Hamann (2012) reported that current populations of Engelmann spruce lag behind their optimal climate niche in the Rocky Mountains. Under climate change, they project that these lags will be even greater, delaying new tree establishment. Thus, under climate change, these tree species will continue to persist, but increasingly will occupy areas outside of their optimal climate. Given this information, Engelmann spruce and subalpine fir are moderately vulnerable to climate change.

Cold-Adapted Understory Vegetation

Little information is available on climate effects and understory plant dynamics in subalpine spruce-fir forests. We draw from literature exploring climate effects and subalpine meadow plants. Warming of early spring temperatures and early snowmelt have been documented to initiate plant growth and flowering in subalpine meadow plants (Iler et al. 2013; Inouye 2008). In an experimental treatment, subalpine meadow plants flowered 10 days earlier in plots where snow was removed (Gezon et al. 2016). Plants that survived freeze events also had more pollinator visitation and reproduction than controls, but early blooming forbs were vulnerable to variability in spring freeze dates (see Sensitivity to Extreme Climatic Events section). Iler et al. (2013) reported that early flowering species were more responsive to a shift in flowering onset, peak flowering, and flowering end days. Although many forb and shrub species can reproduce by sprouting from belowground structures, periods of sexual reproduction are necessary to produce the genetic variation on which selection can work. Further, flower blooming in subalpine meadows has been observed to shift away from the mid-growing season to an earlier or late period, negatively affecting pollinators (Aldridge et al. 2011). Loss of flowers and seeds associated with unsuccessful sexual reproduction has an impact on food sources for pollinators, herbivores, and seed predators.

Nine spruce-fir associated understory plant species were assessed using the NatureServe climate change vulnerability index (Neely et al. 2011). Six species are moderately vulnerable: reflected moonwort (*Botrychium echo*), forkleaved moonwort (*B. furcatum*), Mingan moonwort (*B. manganese*), pale moonwort (*B. pallidum*), northern moonwort (*B. pinnatum*), and Black Canyon gillia (*Gillia penstemonoides*). One species was ranked highly vulnerable, rockcress draba (*Draba globose*); and one species extremely vulnerable, peculiar moonwort (*Botrychium paradoxum*).

Growing conditions for understory subalpine spruce-fir plants may improve with the loss of tree canopy from insect outbreaks and the associated tree mortality. After bark beetle outbreaks, forbs, grasses, and shrubs increased in amount, height, and coverage (Jorgenson and Jenkins 2011). Understory vegetation of spruce-fir ecosystems is likely to persist under climate change along with their overstory components, according to studies, but reproduction could be hindered by a combination of climate change factors. Therefore, cold-adapted understory plants are likely to be moderately vulnerable to climate change.

Birds and Mammals

Warming can affect wildlife indirectly through changes in snowpack and directly through increases in air temperature. Temporal changes in snowpack accumulation or snowpack melt are problematic for species that undergo seasonal molts where their pelage or plumage changes color to match the presence or absence of snow (Mills et al. 2013; see Snowshoe Hare section in Chapter 3). Research observed that American marten population distributions are highest in areas of deep snow (>9 in) and may be hindered where snowpacks shrink (Stone 2010). Warming can stress wildlife (See Sensitivity to Extreme Heat section). White-tailed ptarmigan and its European counterpart, alpine rock ptarmigan (*Lagopus muta*), were projected to decline with warmer winter temperatures and delayed fall snowpacks in the Rocky Mountains and the Alps, respectively (Imperio et al. 2013; Wang et al. 2002). However, warmer spring temperatures may provide ptarmigan with earlier access to forage with earlier snow-free ground, and earlier breeding increases their reproductive success (Imperio et al. 2013). A suite of factors that included rising temperatures, less snow or shorter duration of snowpacks, insect infestations, periodic drought, and shorter fire return intervals contributed to a ranking of most vulnerable for boreal owl in southwestern Colorado (Rhea et al. 2013). Similar factors contributed to the highly vulnerable ranking for the owl in the Gunnison Basin in Colorado (Neely et al. 2011).

Greater loss of mature forest structure on the landscape as a result of increased disturbance, such as fire or bark beetle outbreaks, may harm some species. Declines in American marten populations have been linked to prey availability, weather conditions, and loss of habitat, particularly their preferred mature forest habitat structure (Stone 2010). In a study in northern Idaho, western Montana, and parts of the Greater Yellowstone Ecosystem, Wasserman et al. (2013) reported that projected climate warming reduces the extent of American marten dispersal habitat and increases the fragmentation of marten populations in the western and northwestern parts of the study area. In the Greater Yellowstone Ecosystem, however, climate change is not predicted to fragment dispersal habitat of marten. In southwestern Colorado, American marten was assessed as vulnerable to climate change based on these factors: long life span and small litters, decreasing abundance of prey (red-backed vole [*Myodes gapperi*]), and competition for prey by other predators (Rhea et al. 2013).

Birds that use forests after disturbance may benefit from disturbances that are likely to increase with warming (Norris et al. 2013; Saab et al. 2014). For example, cavity nesting birds and species nesting in shrubs respond positively to bark beetle outbreaks (Saab et al. 2014). The olive-sided flycatcher (*Contopus cooperi*) uses forests that have been disturbed in southwestern Colorado; however, this bird is a long-distance migrant, and may also be subjected to climate changes occurring in wintering areas outside the Rocky Mountain Region (Kotliar 2007; Rhea et al. 2013). The American three-toed woodpecker was identified as resilient to climate change in southwestern Colorado (Rhea et al. 2013). Neverthless, a high uncertainty was given for this ranking as little information is available on how climate change will affect competitive interactions, predators, diseases, and parasites, all of which influence woodpecker population dynamics. In addition, these opportunities to use forests after disturbance depend on the return of this forested type following the disturbance. Detailed species-specific information about vulnerability is currently limited. Given the available information, bird and mammal species of spruce-fir ecosystems will have differing degrees of vulnerability to climate change (Pavlacky and Sparks 2016). Vulnerability will be high for some species and low for others.

Sensitivity to Extreme Climatic Events

BOX 5.4

Key Vulnerabilities

A moderate vulnerability ranking is given for the sensitivity of spruce-fir ecosystems to extreme climatic events.

- Plants and animals in the subalpine spruce-fir ecosystems may be susceptible to increased mortality under extreme drought.
- Plants and animals may undergo heat stress, which can reduce population numbers.
- Earlier warm periods and earlier snowmelt may promote earlier plant emergence and flowering and increase plant vulnerability to greater variability in freeze dates.
- Engelmann spruce and subalpine fir are vulnerable to extreme wind events, especially on ridgetops and saddles and where wind exposure is highest.

Sensitivity to Drought

Although drought is a rare event in the moderately wet subalpine ecosystems (Schoennagel et al. 2004), drought has caused stress, population declines, and increased mortality rates for animal and plant species of spruce-fir ecosystems. Researchers reported decreased avian abundances at the ecoregional spatial scale in the western United States when extreme weather events occurred in breeding and post-fledging periods; declines were greatest when drought coincided with heat waves (Albright et al. 2010). van Mantgem et al. (2009) concluded that noncatastrophic tree mortality rates in the western United States have increased over time and are correlated with increasing temperatures and water deficits. Specifically, for the Colorado Front Range, monitoring of more than 5,000 marked trees in permanent plots since 1982 showed an accelerating rate of mortality for all the common conifers, which was attributed primarily to increased moisture stress (Smith et al. 2015). In the subalpine forest at Niwot Ridge in Colorado, Smith et al. (2015) reported an increase in the frequency of maximum daily temperatures exceeding 75 °F for the 1986–2012 period compared to the 1953–1985 period. It is noteworthy that this increase in background tree mortality rates has been occurring even in stands not affected by bark beetles or other lethal insects or pathogens.

Drought strongly affects mortality of both Engelmann spruce and subalpine fir (Bigler et al. 2007), with subalpine fir having higher drought sensitivity than Engelmann spruce (Villalba et al. 1994), and more pronounced mortality from early season drought (Bigler et al. 2007). Engelmann spruce mortality can continue for up to 5 years after a drought, and subalpine fir can be affected 11 years after a drought (Bigler et al. 2007).

Although topographic differences strongly influenced the growth of Engelmann spruce, subalpine fir, and lodgepole pine, warm falls without early snow favored lodgepole pine over the other two species (Villalba et al. 1994). Such conditions may increase in the future, favoring lodgepole pine in the subalpine zone. Other infrequent disturbances such as insect epidemics and wildfire can follow drought (Bigler et al. 2005; DeRose and Long 2012; Hart et al. 2014; Sibold and Veblen 2006). Due to drought-associated population declines and mortality in spruce-fir ecosystems, we expect these ecosystems will be moderately sensitive to drought.

Sensitivity to Extreme Heat

Wildlife and plant species associated with spruce-fir ecosystems are sensitive to extreme heat. Researchers have observed that numbers of birds of resident species decline during extended heat waves in the Rocky Mountains (Albright et al. 2011). Studies project that ptarmigan population numbers will decline with warmer temperatures associated with climate change (Wang et al. 2002). Heat may also stress other wildlife that use subalpine habitat in summer; thresholds are as follows: 77 °F for American marten (Stone 2010), 59 °F for moose (Lowe et al. 2010), and 77 °F for elk and deer (Parker and Robbins 1983). Studies link warmer temperatures to reduced wildlife populations of large mammals, as heat inhibits their ability to assimilate nutritional resources; examples are moose (Lenarz et al. 2009) and mountain goat (White et al. 2011).

Engelmann spruce and subalpine fir grow where summer temperatures are relatively cool. Extreme heat events may stress mature trees and negatively affect seedlings through drying (Alexander 1987). Buechling et al. (2016) reported increases in seed output of Engelmann spruce in the southern Rocky Mountains over the last 40 years but found that seed output declined when summer temperatures in the maturation year were more than 20 percent above the long-term mean temperature. These results suggest future declines in seed output with warming temperatures.

Sensitivity to Freeze Dates

Variability in freezing dates combined with warming temperatures and earlier snowmelt could increase plant and pollinator exposure to frost damage in spring. Heavy frosts may especially damage or kill Engelmann spruce and subalpine fir seedlings (Alexander and Shepperd 1990). Herbaceous vegetation growth in subalpine meadows is controlled by snowpack, and earlier melt of snowpack has resulted in earlier blooming, earlier emergence of seedlings, earlier onset of flowering, frost damage to flowers, and thus loss of seeds (Iler et al. 2013; Inouye 2008). Caradonna and Bain (2016) found subalpine meadow plant species in the Colorado Rocky Mountains had greater frost sensitivity of reproductive structures compared to vegetative structures and argue that this pattern may be widespread for long-lived perennial plants. Flower blooming in subalpine meadows has been observed to shift away from the mid-growing season to an earlier flower blooming state, potentially increasing frost exposure and negatively affecting pollinators, birds, and animals dependent on these plants for food (Aldridge et al. 2011). Plants and pollinators of spruce-fir ecosystems are likely to be sensitive to variability in freeze dates.

Sensitivity to Wind

Mature Engelmann spruce and subalpine fir tree mortality caused by windthrow and windstorms is historically common, and attributed to shallow root systems and shallow, poorly drained soils (Alexander 1987). The impact of wind events on subalpine forests was reported for a 1934 wind event (Kulakowski and Veblen 2003), a large blowdown event in northern Colorado (Kulakowski and Veblen 2002), and a small wind event in Rocky Mountain National Park (Veblen et al. 1989). Topography strongly influences the impact of the wind events; trees on ridgetops, ridge saddles, or moderate to steep south- and west-facing slopes (Alexander 1987), or at higher elevations and on east-facing slopes (Kulakowski and Veblen 2002) may be most sensitive to wind. Younger stands are less sensitive to wind events (Kulakowski and Veblen 2002). Advance regeneration in the understory of a lodgepole pine forest impacted by wind shifted the forest dominants to spruce and fir (Veblen et al. 1989). Although strong and frequent winds are common in the Rocky Mountain subalpine (Alexander 1987), no change, or a reduction in extreme wind compared to the late 20th century, is projected for the western United States (Pryor et al. 2012). Given this information, a moderate vulnerability ranking is assigned.

Intrinsic Adaptive Capacity to Climate Change

BOX 5.5

Key Vulnerabilities

A moderate vulnerability ranking is given for the intrinsic adaptive capacity of subalpine spruce-fir ecosystems to climate change.

- Factors that strengthen adaptive capacity:
    - Topographic positions in moister valley bottoms, and a higher percentage on north-facing slopes, enhance the ability of spruce-fir ecosystems to adapt to a warmer and drier climate.
    - Different life history strategies of Engelmann spruce and subalpine fir create flexibility for transitions in community composition under future climate variation.
    - Wildlife species have adapted to the cold- and snow-dominated subalpine climate.
- Factors that weaken adaptive capacity:
    - Dominant tree species need a long time to recover from disturbances; warming temperatures and drier conditions, and increases in wildfire, could affect successful regeneration.
    - The potential for invasive species may stress native understory plants.

Factors That Strengthen Adaptive Capacity to Climate Change

Because the subalpine zone is one of great topographic and consequently environmental variation, these forests are highly variable in composition and structure (Romme et al. 2009a). In the Rocky Mountain Region, spruce-fir forests occur in the following proportions: 34 percent on north-facing slopes, 25 percent on west-facing slopes, 22 percent on east-facing slopes, and 19 percent on south-facing slopes (derived from Comer et al. [2003] GIS data). North-facing slopes may be more buffered from warming and drying (Hinckley et al. 2014). Subalpine spruce-fir forests can occur in sheltered ravines and valley bottoms where moist conditions exist and fire is less frequent (Romme and Knight 1981; Veblen et al. 1994). In some cases, these wetter subalpine areas may mitigate negative effects from a warmer and drier climate.

Life history traits of the dominant tree species facilitate adaptive responses. Coexistence of Engelmann spruce and subalpine fir is a function of their different life history strategies: Subalpine fir has high fecundity and shorter longevity whereas Engelmann spruce has longer survival (adult survivorship) (Smith et al. 2015; Veblen 1986). Shade tolerance of spruce and fir may be advantageous in some settings. Researchers found mortality events in mixed-species stands can accelerate a shift from lodgepole pine to spruce-fir forests, as the understory in these mixed-species stands often includes a component of spruce and fir regeneration (Veblen et al. 1989).

Animals of spruce-fir ecosystems have adaptive characteristics that enhance their ability to exist in the snow-dominated subalpine ecosystem. These ecosystems are likely to remain cold and snow dominated under climate change although with some snowpack losses (Christensen and Lettenmaier 2006; Regonda et al. 2005). Squirrels cache food to sustain themselves through the winter (Sullivan 1995b). American martens, well adapted to snow, use it as a protection against cold (Stone 2010). Snowshoe hares develop white winter coats to reduce risk of predation (Sullivan 1995a).

Factors That Weaken Adaptive Capacity to Climate Change

A factor contributing to lower adaptive capacity of spruce-fir ecosystems is the long recovery times of spruce and fir trees after disturbance; it can take centuries to develop mature stands (Aplet et al. 1988; Veblen et al. 1989). Reestablishment is slow as a thick understory layer often develops before trees can establish, requiring centuries before sufficient tree regeneration forms a closed-canopy forest (Peet 1981; Turner et al. 1998). Cone production begins after trees are 20 to 40 years old, and seeds are distributed primarily by wind and gravity (Alexander and Shepperd 1990; Alexander et al. 1990).

Coop et al. (2010) also found decreased regeneration of spruce and fir in burned areas as a function of elevation and distance from unburned edges. Increases in fire extent or fire at high elevations will result in delayed recovery. Although information is available on genetic adaptation of spruce and fir to a changing climate, it is unlikely that tree adaptation will match the rate of warming in the future. Thus, trees will have a low capacity to survive in higher temperatures or moisture-limited conditions.

Although high elevation areas have minimal invasive or nonnative species, climate change coupled with human influences could provide new habitat and dispersal mechanisms (Pauchard et al. 2009). West et al. (2015) modeled future habitat for cheatgrass under an ensemble of six future climate scenarios. Habitable area for this species in the Rocky Mountain National Park is projected to expand to four times the current area by 2050.

Dependence on Specific Hydrological Regime

BOX 5.6

Key Vulnerabilities

A very low vulnerability ranking is given for dependence of subalpine spruce-fir ecosystems on a specific hydrological regime. These systems are dependent on snowpack, but their high elevation location may mitigate snowpack changes.

Subalpine spruce-fir ecosystems are not dependent on a specific and narrow hydrological regime that has little variability from year to year. Precipitation is variable, and primarily comes as snow. Snowpack in the subalpine zone acts as a reservoir for water within that zone as well as for areas downstream. Studies project that snowpack loss as a result of warmer temperatures will be less in the high-elevation Rocky Mountain subalpine compared to lower elevations (Christensen and Lettenmaier 2006; Lukas et al. 2014).

Even though the vulnerability ranking for this criterion is given as very low, subalpine spruce-fir ecosystems and the dominant trees, in particular, are dependent on snowpack. A 9-year study in Colorado found that years with longer growing seasons were correlated with shallower snowpacks and an earlier start of spring (Hu et al. 2010). For these 9 years, the longer growing season was associated with an earlier start to spring, not a lengthening of the fall growing period. The dominant subalpine tree species, lodgepole pine, Engelmann spruce, and subalpine fir, were very dependent on the snowmelt water in the soil, even late into the growing season. Most importantly, summer rains failed to compensate for years with earlier springs and shallower snowpacks. The researchers suggested that the lower reliance on summer rains is a function of the tree roots going deeper in the soil for a more consistent source of water and that summer rain events are often too small to penetrate deep into the soils (Hu et al. 2010). Projected changes in snowpack will impact spruce-fir ecosystems.

Likelihood of Managing Climate Change Effects

BOX 5.7

Key Vulnerabilities

A very high vulnerability ranking is given for mitigation of climate change effects on spruce-fir ecoystems.

- Spruce beetle management strategies exist for prevention, suppression, and restoration of beetle-impacted spruce-fir forests. The effectiveness of these treatments varies; most studies emphasize the importance of managing for regeneration as part of the treatment strategy.
- No techniques have been shown to offset negative effects from heat, snowpack variability, shifts in freezing dates, and drought stress.
- Large areas of spruce-fir forests are inaccessible, occur in complex terrain, or are in wilderness, limiting potential management options.

Management options for mitigating the effects of climate change in spruce-fir forests include reducing current stressors, enhancing ecosystem resilience and, as climate continues to change, helping plants and animals adapt without substantial loss of soil, soil nutrients, and plant cover (Millar et al. 2007). In response to the recent widespread outbreaks of spruce beetle, several research studies have evaluated silvicultural techniques in the Rocky Mountain Region as well as across the western United States. Spruce beetle management strategies include prevention to enhance stand diversity and resilience before an outbreak, suppression during an outbreak to reduce population levels and the rate of insect spread, and restoration activities to reestablish vegetation and promote the long-term resilience of forests to insects and diseases (Alexander 1987; Jacobs 2012; Jenkins et al. 2014).

In a retrospective assessment, Hansen et al. (2010) evaluated the effectiveness of partial cutting to reduce spruce beetle-caused mortality across the southern Rocky Mountains. They conclude that partial cutting apparently results in reduced losses to spruce beetle but caution that spruce beetle populations are likely to infest residual mature trees regardless of previous treatment. Windmuller-Campione and Long (2015) evaluated silvicultural treatments in a spruce-fir forest type in northern Utah, using a resistance index and a metric for resilience (minimum amount of Engelmann spruce regeneration necessary to maintain a spruce component after a spruce beetle epidemic). They argued that resilience should be considered in active management for beetles. Based on these indices, they determined that shelterwood with reserves increased short-term resistance and long-term resilience to spruce beetle outbreaks. Temperli et al. (2014) reported limited effectiveness of density reduction treatments for mitigating stand-level beetle infestation; rather than attempt to increase resistance to the spruce beetle, they recommended increasing spruce advance regeneration in the understory, thereby enhancing ecosystem resilience. The degree of protection to residual spruce following treatment is influenced by insect population pressure (Hansen et al. 2010). Large and older trees can die under such conditions whether stands were thinned or not (Black et al. 2013; Romme et al. 2006). DeRose and Long (2014) offered a framework for evaluating silvicultural options for resistance and resilience to spruce beetle (table 5.1). The importance of managing for regeneration was emphasized in most studies. We

Table 5.1—Conceptual framework of stand and landscape resistance and resilience to disturbance (DeRose and Long 2014).

	Resistance	Resilience[a]
Stand	Influence of structure and composition on disturbance severity	Influence of disturbance on subsequent structure and composition
	Wildfire: influence of structure and composition on the severity of fire behavior	Wildfire[b] : Influence of fire on subsequent structure and composition
	Spruce beetle: influence of structure and composition on the severity of spruce mortality due to high beetle population levels arising from within the stand	Spruce beetle[c] : Influence of spruce beetle infestation on subsequent structure and composition
Landscape	Influence of structure and composition on the spread of disturbance	Influence of disturbance on subsequent forest structure and composition
	Wildfire: influence of multistand structure and composition on the spread of fire	Wildfire: Influence of fire on subsequent proportion of landscape age classes and species dominance
	Spruce beetle: influence of multistand structure and composition on the severity of spruce mortality due to the transition from endemic to epidemic beetle populations	Spruce beetle: Influence of spruce beetle outbreak on proportion of landscape age classes and spruce-dominated stands

[a] Structural and compositional indicators of stand and landscape resilience are a function of management goals relating to desired conditions in a specified period after a disturbance, for example, immediately after the disturbance or longer term. Desired conditions need not be limited to live trees and may include important ecosystem attributes such as snags, coarse woody debris, or decadent crowns.

[b] Indicators of stand and landscape resilience to wildfire typically reflect specified reference conditions, for example, large, widely spaced trees of fire-tolerant species (stand) and diversity of successional stages (landscape).

[c] Indicators of stand and landscape resilience to spruce beetle might include, e.g., surviving large trees (stand) and the potential for future spruce dominance (landscape).

know of no feasible management techniques to completely mitigate the negative effects of drought; heat stress; earlier snowpack melt, which intensifies late season drought; and shifts in freezing dates, which hamper seedling and herbaceous vegetation productivity.

Jenkins et al. (2014) concluded that the spruce beetle management recommendations of the 1970s were not sufficiently applied at the landscape scale to mitigate the recent outbreaks. They note that many factors influence effective management, such as lack of timber harvesting infrastructure in some areas of the western United States and cost versus benefit realities. In the Rocky Mountain Region, spruce-fir forests encompass nearly 5 million ac. In addition, large areas of these subalpine spruce-fir forests are inaccessible, occur in complex terrain, or are in wilderness areas. These factors also limit the potential to implement silvicultural treatments.

Potential for Climate Change to Exacerbate the Effects of Nonclimate Stressors, or Vice Versa

BOX 5.8

Key Vulnerabilities

Several factors exacerbate climate and nonclimate stressors in spruce-fir ecosystems. Consequently, a very high vulnerability ranking is given.

- Warming and drought contribute to increased insect outbreaks and fire activity.

Bark Beetle Outbreaks and Warmer Temperatures

Engelmann spruce is the primary host for the spruce beetle. The insect typically completes a generation in 2 years, but high summer temperatures are correlated with a rising proportion of beetles that complete a generation in 1 year, thus increasing population growth and levels of spruce beetle-caused tree mortality (Bentz et al. 2010). Tree-ring reconstructions document widespread spruce beetle outbreaks over the past several centuries, long before forest management practices were implemented in the subalpine zone (Baker and Veblen 1990; Eisenhart and Veblen 2000; Jenkins et al. 2014); studies document regional spruce beetle outbreaks associated with warmer and drier climate (Hart et al. 2014). In Colorado, for example, large areas of relatively even-aged spruce-fir forests resulted from widespread burning in the second half of the 19th century so that by the late 1900s much of the subalpine forest had reached an age known to be susceptible to bark beetle outbreaks (Romme et al. 2006; Sibold et al. 2006). Outbreaks have affected spruce forests on more than 1 million ac in Colorado and Wyoming (Harris et al. 2013). Spruce beetle outbreaks have greatly affected stand structure and composition, causing mortality of Engelmann spruce, shifting dominance to subalpine fir, and accelerating growth of remaining trees (Veblen et al. 1991). Researchers project that, with greater probability of occurrence, future outbreaks will cause tree mortality at higher elevations, but with a large amount of variability over the landscape and over time (Bentz et al. 2010).

Fire, Drought, and Warmer Temperatures

Many studies have linked warm temperatures, drought conditions, and fire in subalpine ecosystems. Recent temperature warming and longer fire seasons have contributed to an upswing in forest fire activity in the western United States since the mid-1980s (Westerling et al. 2014). Drought and warmer temperatures are two factors associated with increased fire occurrence in paleo records (Pierce et al. 2004). Drought is well documented as a primary factor controlling historical fire occurrence in Rocky Mountain subalpine forests (Buechling and Baker 2004; DeRose and Long 2012; Sherriff et al. 2001; Sibold et al. 2006). Throughout the Rocky Mountain Region, multiple centuries of tree-ring records from subalpine forests clearly show fire occurring with warmer and drier conditions linked to cooler Pacific Ocean and warmer Atlantic Ocean sea surface temperatures. This indicates synchronized warm-drying trends on a subcontinental scale, and increased wildfire in these relatively cool, moist subalpine forests (Schoennagel et al. 2005, 2007; Sibold and Veblen 2006).

Warmer and drier climates in the future may contribute to increased fire occurrence in subalpine forests of the Greater Yellowstone Ecosystem (Westerling et al. 2011). High elevation spruce-fir stands that have not burned in the past 400 years may be more susceptible to burning (Sibold et al. 2006). Researchers found stands that burned during the 1800s had a lower likelihood of burning during recent fires (Bigler et al. 2005). In some cases, compound disturbances, such as fire and wind, may favor aspen regrowth over spruce-fir; spruce-fir is less resilient to fire than aspen, as observed in northwestern Colorado (Kulakowski et al. 2012). In contrast, Higuera et al. (2014) reported that pollen and macroscopic charcoal from high-resolution lake-sediment records in Rocky Mountain National Park suggested that fire severity may be more responsive than fire frequency to climate change in Rocky Mountain subalpine forests.

Summary of the Regional Vulnerability of Spruce-Fir Ecosystems

The overall vulnerability ranking for the ecosystem is a combination of the vulnerability to nonclimate stressors and the vulnerability to climate stressors. For spruce-fir ecosystems, the mean overall vulnerability ranking was moderate with mean expert reviewer rankings ranging from the moderate to the high category (fig. 5.2, table 5.2). Mean vulnerability rankings were moderate for both the nonclimate stressor vulnerability and the climate vulnerability (fig. 5.2). Expert reviewer rankings were in the moderate category for nonclimate vulnerability and the moderate or high category for climate vulnerability.

The expert reviewers had high overall confidence in the ranking of the overall regional vulnerability (fig. 5.3). The five experts' individual rankings of overall confidence ranged from the moderate to the very high category (table 5.3). Mean confidence was high for the nonclimate vulnerability ranking, but individual confidence rankings ranged from the moderate to the very high category (fig. 5.3a). Mean confidence was also high for the climate vulnerability ranking with individual rankings ranging from the moderate to the very high category (fig. 5.3b).

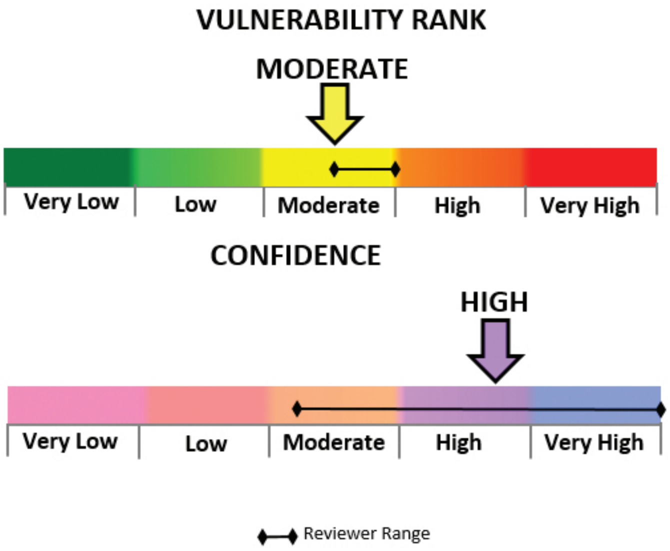

Figure 5.2—Overall regional vulnerability ranking and confidence ranking for spruce-fir ecosystems in the Rocky Mountain Region. The large arrow points to the mean score for vulnerability and for confidence. Bar represents the range of scores.

Table 5.2—Original and reviewer rankings for the regional vulnerability assessment of subalpine spruce-fir ecosystems in the Rocky Mountain Region. Underline indicates a reviewer score change from the original assigned score.

Criterion	Original score	(Reviewer 1) Score	(Reviewer 2) Score	(Reviewer 3) Score	(Reviewer 4) Score	(Reviewer 5) Score	Reviewer criterion rank mean
Nonclimate							
1. Extent	3	3	3	3	3	3	Moderate
2. Human influences	3	3	3	3	3	3	Moderate
3. Resilience	3	3	3	3	3	3	Moderate
4. Future trends	3	3	3	3	3	3	Moderate
Total	12 Moderate	12 Moderate	12 Moderate	12 Moderate	12 Moderate	12 Moderate	12 Moderate
Climate							
1. Ecosystem shift	3	3	3	3	3	3	Moderate
2. Species groups	3	4	3	3	3	3	Moderate
3. Climatic events	3	3	4	3	3	3	Moderate
4. Adaptive capacity	3	3	2	3	3	3	Moderate
5. Hydrology	1	1	1	3	1	1	Very low
6. Management	5	5	5	5	5	5	Very high
7. Interactions	3	5	5	5	5	3	Very high
Total	21 Moderate	24 High	23 Moderate	25 High	23 Moderate	21 Moderate	23 Moderate
Overall vulnerability rank	17 Moderate	18 Moderate	18 Moderate	19 High	18 Moderate	17 Moderate	18 Moderate

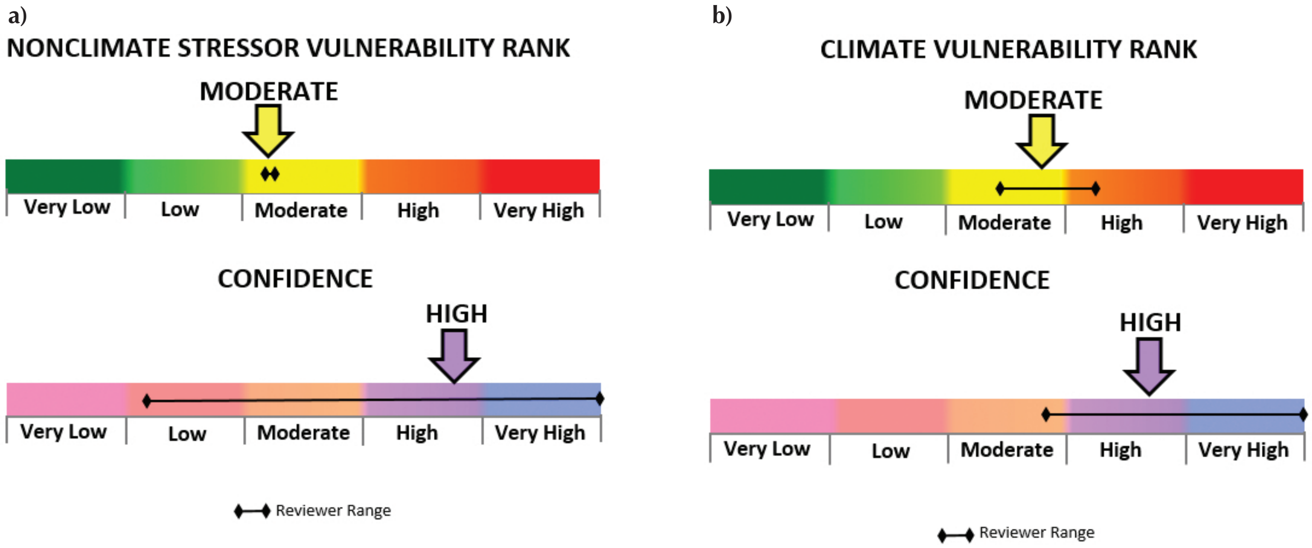

Figure 5.3—Regional vulnerability of spruce-fir ecosystems to nonclimate (a) and climate (b) stressors in the Rocky Mountain Region. The large arrow points to the mean score for vulnerability and for confidence. Bar represents the range of scores.

Table 5.3—Reviewer rankings for confidence in the vulnerability assessment of subalpine spruce-fir ecosystems in the Rocky Mountain Region.

Criterion	**(Reviewer 1) Score**	**(Reviewer 2) Score**	**(Reviewer 3) Score**	**(Reviewer 4) Score**	**(Reviewer 5) Score**	**Criterion rank confidence mean**
Nonclimate						
1. Extent	5	3	5	3	5	High
2. Human influences	5	5	5	3	5	Very high
3. Resilience	5	5	5	3	5	Very high
4. Future trends	3	3	5	1	5	Moderate
Total	18 Very high	16 High	20 Very high	10 Moderate	20 Very high	17 High
Climate						
1. Ecosystem shift	5	3	5	3	5	High
2. Species groups	1	3	5	3	5	Moderate
3. Climatic events	3	3	5	3	5	High
4. Adaptive capacity	5	2	5	1	5	High
5. Hydrology	5	5	1	3	5	High
6. Management	5	5	5	5	5	Very high
7. Interactions	1	4	1	5	5	Moderate
Total	25 High	25 High	27 High	23 Moderate	35 Very high	27 High
Overall confidence rank	22 High	21 High	24 High	17 Moderate	28 Very high	22 High

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Chapter 6. Low-Gradient Mountain Stream Reaches: Vulnerability to Nonclimate and Climate Stressors in the U.S. Forest Service Rocky Mountain Region

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Quick Look: Low-Gradient Mountain Stream Reaches in the Rocky Mountain Region

Low-gradient mountain stream reaches can be found in headwater alpine areas and downstream through subalpine, montane, and foothill zones in five river basins in the Rocky Mountain Region: Colorado, Rio Grande, Arkansas, Lower Missouri, and Upper Missouri. For this regional assessment, we include the stream reach as well as aquatic and riparian plants and animals that live there. Low-gradient mountain stream reaches typically have relatively broad valley bottoms and increased residence time of water, sediment, and organic matter as a result of the sinuous or multi-thread channels and extensive riparian zones and floodplains. Low-gradient stream reaches are also generally dominated by complex and diverse habitats, given the relative abundance and range of water and soil conditions. Cattle and wildlife prefer these areas for grazing, people enjoy recreational activities in and near them, and development tends to be focused adjacent to them.

Quick Look: Vulnerability of Low-Gradient Mountain Stream Reaches in the Rocky Mountain Region

Vulnerability to nonclimate and climate stressors: Very high

Confidence: High

Exposure: Warming stream and soil temperatures; potentially earlier local snowmelt. Greater frequency of disturbances (fire, flooding) could increase debris flow.

Current extent: Limited; 11 percent of all stream lengths on national forest lands in the Rocky Mountain Region.

Sensitivity and adaptive capacity to climate change: Low-elevation low-gradient streams may warm more than high-elevation low-gradient streams. The warming of low elevation streams may reduce the diversity of associated aquatic species. Low-gradient reaches in higher elevation streams may become more productive as stream temperatures increase. However, these high-elevation stream networks are smaller and habitats are typically more limited. Riparian plant communities may shift in elevation as temperature and streamflow conditions change, but species in high-elevation riparian areas have limited options for upstream migration. Beaver, a keystone species, can tolerate wide ranges of water and air temperature and therefore is likely to persist. However, cold-water native fish, amphibians, benthic macroinvertebrates, and riparian plants are highly vulnerable to climate change.

Nonclimate stressors: Many stream reaches have not recovered from the legacy of human activities during the mid-1800s to early 1900s. Current stressors include recreation, wildlife and agricultural grazing, roads, and other land use development. These stressors can constrict streamflows, reduce channel size, contribute to pollution, and introduce invasive species.

Introduction

We focus this assessment on low-gradient mountain stream reaches that flow through national forests in the Forest Service, U.S. Department of Agriculture (USFS) Rocky Mountain Region. We define low-gradient mountain stream reaches as those streams with slopes of less than 2 percent (Montgomery and Buffington 1997; Wohl and Merritt 2005). These stream reaches occur in the mountainous areas of the following drainage basins in the Rocky Mountain Region: Colorado, Rio Grande, Arkansas, Lower Missouri, and Upper Missouri (fig. 6.1). This assessment examines the vulnerability of the physical and hydrological characteristics of the stream reach as well as the vulnerability of the aquatic and riparian flora and fauna associated with the stream reach. Hereafter, we refer to this ecosystem as "low-gradient mountain stream reaches."

Stream gradient influences the energy available to transport sediment and modify channel and floodplain dynamics (Buffington and Montgomery 2013; Wohl and Merritt 2005). Steep-gradient stream channels are typically armored with large rocks and boulders and can rapidly transport water and sediment downstream with relatively small changes in their channels (Wohl and Merritt 2005). Segments of streams that are transport-limited sediment storage sites are referred to as "source segments." These reaches undergo intermittent debris-flow scour (i.e., colluvial channels) (Montgomery and Buffington 1997). Low-gradient stream channels, referred to as "response segments," can have small to large changes in channel morphology depending on the increase in sediment supply. Consequently, low-gradient reaches in the mountainous watersheds are more likely to exhibit changes in floodplain and channel geometry in response to upstream and within-reach activities that alter water and sediment inputs (Wohl et al. 2017).

In the Rocky Mountain Region, low-gradient stream reaches are found in alpine areas at the highest elevations (>13,000 ft) and down through the subalpine, montane, and foothills at the lowest elevations (3,000 ft). Small alluvium from upstream typically composes the streambanks. Pool-riffle and plane-bed channels occur in these low-gradient stream reaches, whereas step-pool and cascade channels occur on steeper slopes (Montgomery and Buffington 1997). Riparian vegetation affects bank cohesion, channel width, and channel pattern in these low-gradient streams (Buffington and Montgomery 2013). Root strength contributes to channel stability; when root strength is diminished, channel width can widen considerably. Habitat can be diverse with high productivity, leading to abundant terrestrial and aquatic fauna. In addition, beaver (*Castor canadensis*) activity can increase the complexity of habitat in these low-gradient stream reaches. Activities upstream can result in changes in the movement of water, sediment, and contaminants, all of which affect channel dynamics. Activities within the low-gradient stream reach also can alter channel geometry, the dynamics of water and sediment movement, contaminants in the stream, or aquatic and riparian communities (Wohl 2006), affecting the resilience of these ecosystems.

Low-gradient mountain stream reaches have been assessed for their vulnerability to climate change. The vulnerability framework has two major components, one related to nonclimate stressors and one related to climate stressors. Within the nonclimate component, four criteria were used to define factors of vulnerability: current status of ecosystem extent, human influences on the ecosystem, intrinsic resilience of the

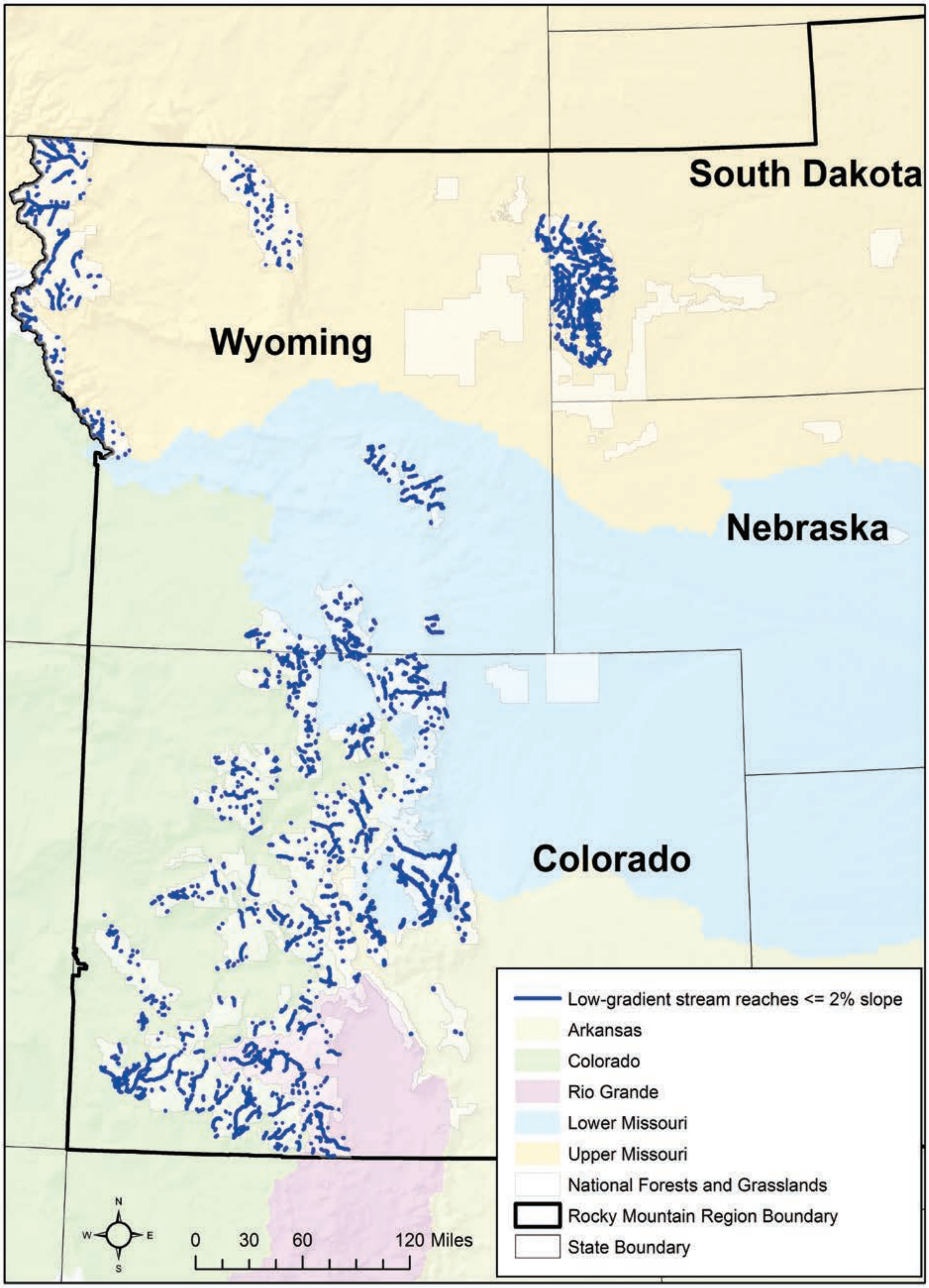


Figure 6.1—Low-gradient mountain stream reaches on national forest lands in the Rocky Mountain Region represent only about 11 percent of all stream reaches on national forests in this region. Low gradient is defined as mountain stream reaches with slope of less than or equal to 2 percent. (Data source: National Hydrography Dataset Plus (Horizon Systems Corporation n.d.); lake edges, diversions and irrigation ditches removed.)

ecosystem to nonclimate stressors, and future trends of nonclimate stressors on the ecosystem. Within the climate change component, seven criteria capture the vulnerability of individual species, sensitivity of ecosystem dynamics such as dependence of the ecosystem on the hydrological cycle, adaptive capacity of the ecosystem, potential for management to mitigate the effects of climate and nonclimate stressors, and interaction between climate change and nonclimate stressors. More information and the rationale for ranking ecosystem vulnerability by using these criteria are given in appendices A, B, and C.

These low-gradient mountain streams are increasingly being studied throughout the Rocky Mountain Region, as well as in other mountainous regions globally. We draw from this literature where studies were conducted within the Rocky Mountain Region, and then in similar settings when local information is unavailable. Even within this Region, these streams vary greatly in geology, valley formation, and land use. Assessing the vulnerability of specific low-gradient mountain stream reaches in individual national forests will require an understanding of local site characteristics and the site characteristics of the research studies reviewed here.

Vulnerability of Low-Gradient Mountain Stream Reaches to Nonclimate Stressors

Summary of Key Vulnerabilities to Nonclimate Stressors

BOX 6.1

Current status of ecosystem extent

A very high vulnerability ranking is given as low-gradient stream reaches have a limited extent in the Rocky Mountain Region.

Human influences on ecosystem

Low-gradient streams and associated habitats have a very high vulnerability to the legacy of past human activities and to the stressors associated with ongoing human activities.

- The arrival of European settlers had an almost immediate influence on low-gradient stream channels and their ecology from beaver trapping, placer mining, logging, road and railroad development, domestic livestock grazing, and water development and diversion. In many areas, reaches have not recovered from activities of the mid-1800s to early 1900s.
- Current stressors include recreation, wildlife and agricultural grazing, roads, and land use development. These stressors can lower streamflows, reduce channel size, contribute to pollution, and introduce and spread invasive species.

Intrinsic resilience of ecosystem to nonclimate stressors

A high vulnerability ranking is given for the intrinsic resilience of low-gradient mountain streams to nonclimate stressors.

- Factors that enhance low-gradient stream resilience:
    - These ecosystems have stream structure complexity, high productivity, and diversity of species, particularly at lower elevations.
    - Beavers can add habitat complexity.
    - Barriers can protect native fish from competition and hybridization with nonnative species.

- Factors that lower low-gradient stream resilience:
    - Physical and biological function of low-gradient stream reaches is hindered by a variety of historical and current activities including water management and changes in runoff timing from dust-on-snow events that reduce late season flow.
    - Low-gradient streams are also the response areas of the stream network. As such, they can be greatly affected by increased disturbances upstream, such as fire, which increases water flow or sediment deposition, or both.
    - Constructed barriers can fragment streams, preventing aquatic species movement.
    - Nonnative fish and pathogens can hinder native species.

Future trends of nonclimate stressors

Low-gradient streams are considered to be very highly vulnerable to future stressors associated with human population growth and related effects on land use, water quality, air quality, increased demand for recreation, increased fire, and the spread of invasive species.

Current Status and Human Influences

Low-gradient stream reaches are limited in extent, consisting of only 11 percent of all stream lengths on national forest lands in the Rocky Mountain Region (table 6.1). Across the Rocky Mountain Region, the Black Hills National Forest in South Dakota has the largest share of low-gradient mountain stream reaches (24 percent), and the Big Horn National Forest in Wyoming has the smallest share (2 percent). At the scale of an individual national forest, the Black Hills National Forest has the highest percentage of low-gradient stream miles as a function of all stream miles. In contrast, low-gradient stream miles in the Big Horn National Forest were only 5 percent of total stream miles. Winters et al. (2016) found that low-gradient stream systems were prevalent in only 8

Table 6.1—Low-gradient mountain stream reach and total stream miles by national forest in the Rocky Mountain Region, given as miles within each national forest, and as a percent of all stream miles within the national forest, and as percent of all low-gradient stream miles in the Rocky Mountain Region. Data Source: NHDPlus data set (lake edges, diversions and irrigation ditches removed).

National Forest	Low-gradient stream miles	Total stream miles	Low-gradient stream miles as percentage of all stream miles in National Forest	Percentage of Rocky Mountain Region low-gradient stream miles
Black Hills	851	2,826	30	24
Medicine Bow-Routt	536	4,150	13	15
San Juan	454	2,998	15	13
Shoshone	341	4,222	8	9
Grand Mesa, Uncompahgre and Gunnison	327	4,332	8	9
Pike-San Isabel	286	3,504	8	8
Rio Grande	259	3,037	9	7
Arapaho-Roosevelt	236	2,792	8	7
White River	226	3,734	6	6
Big Horn	66	1,379	5	2
Rocky Mountain Region	3,582	32,974	n.a.	11

of 162 subwatersheds on the Rio Grande National Forest in Colorado, a reflection of the dominance of steep topography in the southern Rocky Mountains landscape. At the regional scale, low-gradient streams, approximately 3,582 mi, span the elevational gradient from the alpine to the foothills zone.

The arrival of European settlers in the Rocky Mountains had an almost immediate influence on low-gradient stream channels and their ecology. Beaver trapping removed most beaver from Rocky Mountain streams by the mid-1800s (Wohl 2006), reducing the influence of beavers on low-gradient stream habitats (Smith and Tyers 2012). Placer mining for gold and other precious metals was a common practice starting in the mid-1800s. Mining increased stream sediment movement, altered stream and riparian health, and released toxic materials into stream channels (Wohl 2006). As human population increased in the late 1800s and early 1900s, logging, tie-driving, and road and railroad building altered channel shape and the supply of water and sediment, and destabilized slopes (Wohl 2006). Year-round logging practices removed trees from upland as well as riparian areas, primarily for railroad ties. Riparian trees trap sediment during floods, provide habitat for a variety of wildlife and fish species, and generally contribute to heterogeneous riparian habitats. Harvested wood was stored on floodplains and within stream channels so that high spring flows could transport wood loads downstream, a practice called tie-driving (Ruffing et al. 2015). Road and other transportation infrastructure development straightened the meanders of streams and promoted continued development in valleys, thus reducing floodplain extent. The introduction of domestic livestock grazing in these ecosystems altered the riparian communities (Wohl 2006).

Human activities from the mid-1800s to the mid-1900s have left a legacy such that aquatic species and channel functions in low-gradient streams are still recovering from their effects. In a recent study on the Medicine Bow National Forest in Wyoming, tie-driven streams had a more simplified channel morphology; narrower, shallower low cross-sectional roughness; and higher width-to-depth ratios than streams that were not tie-driven (Ruffing et al. 2015). Wood loading in these streams is still in the early stage of recovery. Fornwalt et al. (2009) concluded that settlement-era logging and grazing had a greater long-term influence on riparian understories of ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson) and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) forests than on the understories of upland ponderosa pine and Douglas-fir forests. Grazing can reduce riparian vegetation, compact soils, alter species communities, destabilize channels, and introduce noxious weeds (Kauffmann and Krueger 1984). Cannon and Knopf (1984) found that heavy grazing during summer seasons in the early 20th century reduced shrub density and narrowed the width of riparian areas in higher elevations of Colorado; such effects can last decades after grazing intensity has been reduced.

Wohl (2006) ranks the impact of human activities to all mountain streams in the Rocky Mountains (from Montana through New Mexico) based on several threats. The impact on all mountain streams was ranked as severe from activities that regulate water flow, such as construction of dams and water diversion. Stream and trans-basin diversions for agricultural, municipal, and recreational uses alter physical processes and can harm aquatic communities. Diversions reduce sediment movement, increase downstream erosion, and alter riparian plant composition (Caskey et al. 2015). Unless modified for fish passage, diversions can create migration barriers that fragment aquatic habitats but also may restrict movement upstream of nonnative species.

The Rocky Mountain Region has 910 dams with 146 occurring on national forests and grasslands, 127 of which are in Colorado (U.S. Geological Survey 2005). Reservoirs and dams regulate flow and flooding, are used for power generation, and make water available for municipal and agricultural uses throughout the year. Flow regulation by dams can harm downstream hydrology by:

- reducing meanders;
- reducing sediment transport;
- modifying flows;
- increasing erosion and downcutting of stream beds;
- altering composition and abundance of bottom-dwelling communities;
- narrowing channels;
- modifying algae presence, and populations of macroinvertebrates and fish; and
- increasing invasive species (Baker et al. 2011; Merritt and Wohl 2006; Ward and Stanford 1982; Wohl and Cenderelli 2000).

Impacts on the biotic integrity of all mountain streams in the Rocky Mountains were ranked by Wohl (2006) as moderate to limited based on the current presence of nonnative species, percentage of at-risk native species, status of endemic species, and comparisons with measures of original versus current biodiversity. Aquatic nonnative and invasive species have been intentionally or unintentionally introduced to many aquatic ecosystems in the Rocky Mountain Region. These species have caused reductions in native populations, altered streamflows and nutrient cycling, and disrupted food webs (Rahel et al. 2008; Stromberg et al. 2007). Cutthroat trout (*Oncorhynchus clarkii*) populations have declined, as rainbow trout (*O. mykiss*) breed with native species (Gresswell 2011). Nonnative brook (*Salvelinus fontinalis*), brown (*Salmo trutta*), and rainbow trout can hinder and outcompete native cutthroats (Van Kirk et al. 2009). Leafy spurge (*Euphorbia esula* L.), purple loosestrife (*Lythrum salicaria*), cheatgrass (*Bromus tectorum* L.), and Canada thistle (*Cirsium arvense*) are particularly problematic in riparian areas, where they displace native species (Al-Chokhachy et al. 2013b; Poff et al. 2011).

Though a native species, the freshwater diatom *Didymosphenia geminata* can be a nuisance during blooms in low productivity streams (Elwell et al. 2014; Spaulding and Elwell 2007). This diatom has been reported in various locations in the western United States (Spaulding and Elwell 2007) and in the Rocky Mountain Region, notably in Rapid Creek, South Dakota (James et al. 2010). Research in Rapid Creek documented the role of this diatom in shifting the invertebrate species composition (James et al. 2010) and supplementing the diet of brown trout (James and Chipps 2016). Management recommendations have focused on preventing its spread (Elwell et al. 2014).

Impacts associated with water pollution were seen as moderate to limited based on the presence of organic (human and animal) wastes, nutrients (fertilizers), organochlorine compounds (herbicides, pesticides, industrial chemicals), and heavy metals (Wohl 2006). Increased atmospheric nitrogen deposition from human activities has been observed in the snowpack and in high elevation lakes of the Rocky Mountain Region (Baron et al. 2000b; Ingersoll et al. 2008; Nanus et al. 2012; Wolfe et al. 2001).

Impacts associated with channel alteration to mountain streams were ranked as moderate to limited depending on the presence of in-channel structures, channelization, levees, and beavers, and the occurrence of log drives and placer and aggregate mining (Wohl 2006). Land use impacts on mountain streams have been locally intensive through minimal or extensive deforestation and grazing.

Roads are sources of sediment and contaminants; they can alter drainage patterns in watersheds, increase surface runoff from soil compaction, remove riparian vegetation, and facilitate the spread of invasive plant species (Mortensen et al. 2009). They can also block aquatic passage with structures such as culverts, thus fragmenting habitats (Forman et al. 2003). Populations of native fish species, some of which prefer low-gradient reaches (e.g., roundtail chub [*Gila robusta*] or flannelmouth sucker [*Catostomus latipinnis*]), have declined as a result of more-intense land use and higher road density in the Upper Colorado Basin (Dauwalter et al. 2011). Recreational demands have increased in the Rocky Mountains and such demands can affect low-gradient streams through water withdrawals for snowmaking, soil compaction and increased surface runoff from parking lots, and concentrated use of riparian areas, all of which result in reduced bank stability and degradation of riparian plant communities (Winters et al. 2004; Wohl 2006).

Intrinsic Resilience of Ecosystem to Nonclimate Stressors

Factors That Enhance Resilience to Nonclimate Stressors

Resilience of low-gradient stream reaches to nonclimate stressors is enhanced by stream structure complexity, high productivity, and diversity of species, especially at low elevations. Natural disturbances such as fire and landslides are infrequent and low-gradient streams can be resilient, in the long term, to these disturbances. Riparian areas along low-gradient reaches generally burn less severely and less frequently than upland areas (Dwire and Kaufman 2003). Harm from fire on these stream reaches is temporary, and, under natural conditions, most riparian areas recover quickly (Dwire and Kaufmann 2003), although few studies have documented postfire riparian recovery. Degraded water quality from erosion after fire eventually diminishes (see figure 1 in Goode et al. 2012). Additionally, these stream reaches can have extensive willow (*Salix* spp.) communities, and may support beavers, which add habitat complexity by building dams that keep water and sediment in these areas (Polvi and Wohl 2012). Rocky Mountain streams can have barriers that protect native fish living upstream from competition or hybridization as nonnative species are blocked from migrating upstream (Gresswell 2011).

Factors That Lower Resilience to Nonclimate Stressors

Low-gradient stream processes and aquatic and riparian species are not resilient to long-term stressors that alter hydrology of low-gradient systems. Diversions for human uses have lowered water tables and streamflow and have altered physical processes and riparian vegetation (Caskey et al. 2015; Wohl 2006). Ground-disturbing activities and drought to the west of the Rocky Mountain Region have contributed to increased incidence of dust-on-snow events; these occurrences increase snowmelt, and have shifted the timing of runoff up to 3 weeks earlier (Painter et al. 2010).

Depending on geological and watershed conditions, water quality may be harmed by greater hillslope erosion and sediment transport associated with increased fire (Goode et al. 2012; Ryan et al. 2011; Smith et al. 2011; Wondzell and King 2003). Low-gradient stream reaches in the Rocky Mountain Region that occur in areas with noncalcareous geology may be sensitive to added nutrients and disturbances that change sediment supply (Wohl et al. 2007).

Fish and other aquatic species are not resilient to many other nonclimate stressors. Natural and constructed barriers that are distributed throughout Rocky Mountain Region streams limit migration of aquatic species (Gresswell 2011). Native fish are not resilient to nonnative fish that outcompete or hybridize with them; lack of resilience contributes to the decline of native populations in Rocky Mountain Region streams (Dunham et al. 2004; Gresswell 2011). Aquatic species are at risk from pathogens and disease that kill them. Chytrid fungus (*Batrachochytrium dendrobatidis*), for example, kills boreal toads (*Anaxyrus boreas*) (Muths et al. 2008). Whirling disease (*Myxobolus cerebralis*), introduced by humans from Europe, affects fish species, causing skeletal and neurological damage (Bartholomew and Reno 2002).

Future Trends of Nonclimate Stressors

Future human population growth will increase the demand for drinking water as well as for water associated with land uses such as agriculture and energy. Using projections of population and climate from the 2010 Resources Planning Act Assessment, Foti et al. (2012) reported that increased population growth and potential changes in climate increase the vulnerability of water supply in several Rocky Mountain river basins. These analyses did not consider management to specifically address water use efficiency or any climate change mitigation actions. Expanding human populations in the Rocky Mountain Region are likely to increase the wildland-urban interface. Given the right fire weather conditions and an ignition, Haas et al. (2015) found wildfires could have high potential to impact human populations along the Front Range of Colorado. In this area, they also reported that overall, private ignitions have the potential to impact more people than Federal ignitions. Population growth, land use development and energy development, and the associated air and water quality challenges are stressors that will continue to affect low-gradient mountain streams.

Some populations of invasive plant species, such as cheatgrass, spotted knapweed (*Centaurea biebersteinii* DC), and Canada thistle, may expand (Al-Chokhachy et al. 2013b; Bradley et al. 2009). Additional introductions may occur, and the spread of invasive species may be exacerbated by roads, changes in land use, fire, grazing, and agriculture (Al-Chokhachy et al. 2013b; Bradley et al. 2010), as well as flow changes in streams (Merritt and Wohl 2006). The population of native fish species may continue to decline, and local extinction may happen due to competition and hybridization with nonnative populations (Wenger et al. 2011).

Vulnerability of Low-Gradient Mountain Stream Reaches to Climate Stressors

Capacity for Network Shift

BOX 6.2

Key Vulnerabilities

A very high vulnerability ranking is given for the capacity for network shift as low-gradient stream reaches and their associated floodplains have a constrained capacity for shifting network ranges.

- Range shift potential for species associated with low-gradient streams will be influenced by the rate and degree to which these streams warm and develop unsuitable habitat conditions.
- At higher elevations, stream reaches may become more hospitable with warming temperatures. However, these stream networks are smaller and more geomorphically restricted, providing limited habitat as climate warms.
- Warming temperatures, particularly in the mid- and late growing seasons, and reduced flows may fragment individual stream reaches, restricting movement for fish in the stream network.

Elevation Shift Potential for Stream Network Habitats

Low-gradient mountain stream reaches occur as part of a larger network within each major river system. They span elevations from 3,000 ft to more than 13,000 ft within the Rocky Mountain Region. The highest concentration of low-gradient stream miles is found between elevations of about 7,500 and 9,500 ft (fig. 6.2).

Range shift potential for species associated with low-gradient streams will be influenced by the rate at which streams develop unsuitable conditions for species, particularly at lower elevations (cold-water fish, Rahel et al. 1996; benthic macroinvertebrates, Poff et al. 2010), as well as by the rate at which high-elevation stream reaches become more hospitable. The potential for elevational or latitudinal shifts for species ranges in the low-gradient mountain stream reaches is limited. Researchers project habitat ranges will contract for native cold-water fish as well as nonnative rainbow, brook, and brown trout by the 2040s (Luce et al. 2012; Wenger et al. 2011). Higher elevation streams that are currently too cold to support fish species may warm and provide new habitats as temperatures rise (Cooney et al. 2005; Isaak et al. 2015; Roberts et al. 2013). But tributaries of these high elevation streams are less likely to support cold-water trout populations; these streams offer less stream habitat, which limits the genetic and species diversity of the fish (Cook et al. 2010; Hilderbrand 2003).

Some low-gradient streamside plants may be able to migrate to more climatically suitable upstream or upslope areas and more northern latitudes by wind or animal dispersal (Perry et al. 2012). However, aquatic organisms have limited mobility for latitudinal shifts and may be constrained in upstream migration by natural and built barriers. Range expansion by plant species in highest elevations may be constrained as available upslope area is limited (Perry et al. 2012). Downstream drift, which is the primary way in which seeds are dispersed over long distances (Merritt and Wohl 2002), would not assist upstream migration of riparian plants.

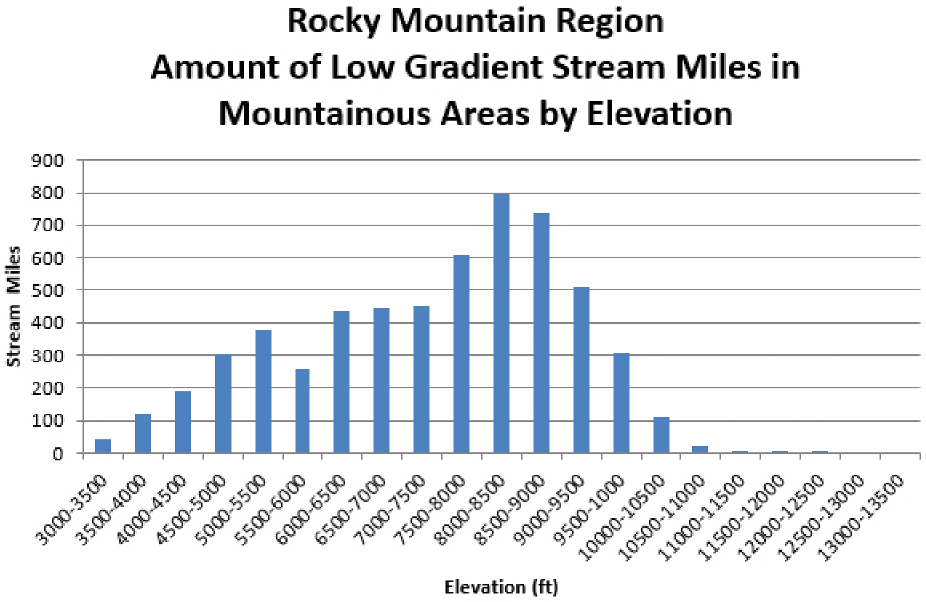


Figure 6.2—Low-gradient mountain stream reach miles by elevation category on Rocky Mountain Region National Forest lands. Low gradient is defined as mountain stream reaches with slopes of less than or equal to 2 percent. (Data source: National Hydrography Dataset Plus (Horizon Systems Corporation n.d.); lake edges, diversions and irrigation ditches removed.)

Fragmentation

Increased fragmentation of low-gradient stream reaches associated with changes in climate will limit range shifts in response to climate change. Fragmentation can be considered at the basin-wide or the reach scale. At the basin-wide scale, fragmentation can result from natural or constructed barriers, such as waterfalls, steep slopes, roads, and water diversion structures that segment the relatively flat valley bottoms and further fragment low-gradient stream habitats. At the reach scale, fragmentation can occur through disturbances (hillside erosion), but also through changes in temperature and streamflow.

The presence of nonnative species can fragment or restrict the current range of native fish species. Large, connected stream corridors allow aquatic and semiaquatic species to move and track environmental changes as well as allow for genetic exchange (Rahel et al. 2008). Fragmentation can occur within a reach through disturbances such as wildfire and flooding or high flows that transport sediment (Dunham et al. 2003). As population abundance of native trout is sensitive to stream fragment length (Roberts et al. 2013), the creation of shorter fragments reduces area for cold-water fisheries.

With climate change, low-gradient streams can become fragmented for cold-water aquatic species by warmer temperatures (Isaak et al. 2010) or changes in the timing and magnitude of streamflow (Jones et al. 2013; Lake 2000; Roberts et al. 2013). Earlier snowmelt, which lowers late season streamflow, and drought can also fragment and decrease habitat (Rahel et al. 1996; Wenger et al. 2011). Increased fragmentation may result in loss of connections between riparian and aquatic species and among their populations, reduce aquatic habitat, and restrict migration and survival of aquatic species (Fischer and Kummer 2000; Lake 2000; Leppi et al. 2012; Rahel et al. 1996; Rieman et al. 2007; Perry et al. 2012). Although higher elevation streams may become thermal refugia for some species (Isaak et al. 2015), higher summer temperatures may thermally fragment and isolate these smaller tributaries (Isaak et al. 2015; Jones et al. 2013; Rahel et al. 1996).

Disturbances (fire or flooding) are current stressors that are likely to interact with climate change effects on individual stream reaches. High-elevation stream reaches are projected to warm, but temperatures are likely to remain suitable for cold-water fish habitat. Reaches that are currently too cold for cold-water fisheries may become suitable (Isaak et al. 2015). Native trout populations in short stream fragments and at lowest elevations are projected to have the highest risk of extirpation; however, this risk is a result of disturbances such as wildfire that are likely to be a greater threat than warming (Roberts et al. 2013). Although the high elevation streams appear to be buffered from warming, the researchers emphasize that the near-term threat is from stochastic disturbances such as wildfire and that future conservation efforts for this species should focus on preventing further loss of habitat from conservation segments.

Vulnerability of Cold-Adapted, Foundation, or Keystone Species to Climate Change

BOX 6.3

Key Vulnerabilities

A very high vulnerability ranking is given for cold-adapted, keystone, or foundation species in low-gradient stream reaches. Most species are highly to very highly vulnerable to climate change. One exception is beaver, which has moderate vulnerability.

- Beaver, a keystone species, is likely to persist. It may grow more vulnerable with drier conditions and drought, changes in riparian plant communities, and increased flooding or fire.
- Cold-water native fish are highly vulnerable to warming and modified streamflows, both of which can eliminate thermally suitable habitat, especially at lower elevations.
- Cold-adapted amphibians are very highly vulnerable to the warming and drying that are likely to occur with climate change, especially at their lower elevational ranges. Reduced precipitation and stream discharge are likely to degrade habitat conditions considerably.
- Cold-adapted benthic macroinvertebrates are very highly vulnerable to warming temperatures, and altered stream dynamics may exceed their heat tolerance as well as interfere with the success of life history strategies, with implications for food chains in these ecosystems.
- Riparian plants, especially at low elevations and mid-elevations are likely to be highly vulnerable to climate change. Riparian plant species at all elevations are dependent on the moist soil conditions and periodic flooding that low-gradient reaches provide. Woody vegetation (for example, willows and cottonwoods) in particular, as well as other plants that depend on saturated and relatively shallow water tables, may be most vulnerable to warming and drying with climate change.

For this criterion, we selected species groups that demonstrate a range of environmental tolerances and have a major influence on ecosystem function. These low-gradient ecosystems provide habitat to cold-adapted species of fish, amphibians, and macroinvertebrates. Beaver is considered a keystone species in these ecosystems as it has a large effect: Dam building by beavers promotes water and sediment retention and landscape heterogeneity in low-gradient stream reaches. We included riparian plants as a group of foundation species that substantially influence ecosystem functions: They cycle organic matter and nutrients, reduce the effects of environmental disturbances, regulate and filter runoff, and offer breeding, feeding, and shelter habitat for animals.

Beaver

Beavers typically inhabit low-gradient stream reaches (Baker and Hill 2003). Their dams can increase storage of precipitation, decrease stream velocity, increase water depth, elevate the water table locally, distribute water and sediment across valley bottoms, create habitat and increase habitat diversity, decrease water turbidity, improve water quality, increase nutrient availability, and enhance resistance to disturbance (drought, fire) (Gibson and Olden 2014; McCaffery and Eby 2016; Polvi and Wohl 2012; Rosell et al. 2005; Westbrook et al. 2011). Although beaver dams can restrict fish passage in some places (Schlosser 1995; but see Baker and Hall 2003; Lokteff et al. 2013) and reduce dissolved oxygen levels (Call 1966), beaver activity creates habitat complexity and enhances ecosystem capacity to withstand streamflow variations that may increase with climate change (Gibson and Olden 2014). Beaver activity may help buffer the effects of reduced snowpack and earlier runoff due to climate change within these low-gradient ecosystems (Wohl 2000).

We draw from historical studies in North America as no studies have explored the effect of projected climate change on beavers in the Rocky Mountain Region. In eastern Canada, Jarema et al. (2009) concluded that beavers were sensitive to projected climate change, with range expansions to the north. However, population density increased more in the core of the current range than at the leading edge. Using historical climate and beaver dam deposits as proxy records, Perisco and Meyer (2012) documented the increase of beaver dams during cool and wet periods and when willow is abundant, and declines during drought in the Greater Yellowstone Ecosystem of northwestern Wyoming. Greater streamflow variation and flooding due to less snow and more rain could hamper beaver populations in the Greater Yellowstone Ecosystem (Perisco and Meyer 2012). In the dryland stream environments of North America, beaver distribution is influenced by the availability of perennial water (flowing streams; small pools in intermittent streams, lakes, and wetlands), and the availability of riparian vegetation (Gibson and Olden 2014). Though beaver population numbers and distributions fluctuate in response to environmental variations, beavers have continued to live in Rocky Mountain landscapes despite severe droughts (Perisco and Meyer 2012). Beavers are also found currently in dryland stream environments (Gibson and Olden 2014). We conclude that beavers are likely to persist under climate change as long as plant sources are available, drought does not cut off water supplies, or flooding magnitude and frequency do not hinder them.

Cold-Water Native Fish

Life history traits and populations of cold-water native fish are closely linked to the dynamics of Rocky Mountain climate and hydrological systems (Gresswell 2011). Warming stream and lake temperatures will lower habitat suitability for cold-water fish species, and unless they can move to suitable habitat in a connected stream network, these changing conditions will increase stress and metabolic rates (Al-Chokhachy et al. 2013a; Isaak et al. 2012). High summer temperatures can also prevent native fish from colonizing small headwater streams (Mullner and Hubert 2005). Changes in winter precipitation, from snow to rain, may result in altered flow regimes and increased scouring;

this scouring may impact fall-spawning species, such as brook trout and brown trout (Goode et al. 2013).

Wildfires and landslides have positively (via gravel bars) and negatively (via sedimentation) affected fish habitat. Low-gradient stream reaches are response areas in the stream network. Greater frequency of these watershed disturbances would increase habitat loss (Williams et al. 2015) and challenge the ability of fish to recover from them.

Stream and river temperatures have increased in many locations across the western United States (Isaak et al. 2012, 2013). Bioclimatic models have been used to explore the potential future effect of climate change on suitable habitat. In a large regional analysis of climate change effect on native and nonnative trout, Wenger et al. (2011) reported a 47-percent decline in suitable habitat by the end of the 21st century for all trout analyzed. Native cutthroat trout is projected to lose an additional 58 percent of its currently restricted habitat; habitat for nonnative brook trout and brown trout declines by 77 and 48 percent, respectively. Warming temperatures negatively affect nonnative rainbow trout habitat, but favorable changes in flow regime result in benefits for rainbow trout (Wenger et al. 2011).

Thermal risks from climate change were analyzed in tandem with fragmentation risks for Colorado River cutthroat trout (*Onchorhynchus clarkii pleuriticus*) in the Upper Colorado River Basin. The model study by Roberts et al. (2013) concluded that the likelihood of disturbances such as wildfire and debris flow impacting the cutthroat habitat was greater than the acute mortality associated with future higher temperatures. Past habitat loss has restricted most of the Colorado River cutthroat populations to high-elevation stream fragments that would be buffered by elevation from the potential consequence of future warming. Isaak et al. (2015) projected high probabilities of cold-water trout occupancy at new habitats that become thermally suitable at higher elevations. Cooney et al. (2005) determined that 10 streams where greenback cutthroat trout (*O. c. stomias*) had been translocated would improve as suitable habitat under several degrees' warming. As in the Colorado River cutthroat trout study, greenback cutthroat trout have been restricted to suboptimal temperature because of nonnative species expansion or introduction.

Environmental changes associated with climate change will vary across the complex terrain of the Rocky Mountain Region, as will the biotic response to these changes. As the climate changes, site condition changes will differ depending on local climate or microclimate, sensitivity of streams to climate change, availability of climate refugia in the stream network, and soil and geological factors. In addition, individual cold-water fish species will have differing responses to these environmental factors. Maximum growth temperature of Rio Grande cutthroat trout (*Onchorhynchus clarkii virginalis*) fry was similar or higher than Colorado River cutthroat trout, Yellowstone cutthroat trout (*O.c. bouvieri*), Snake River cutthroat trout (*O.c. behnkei*), and Westslope cutthroat trout (*O.c. lewisi*) (Ziegler et al. 2015). The Rio Grande cutthroat trout could be expected to be relatively more warm-adapted because it has the southernmost distribution of these cutthroat trout subspecies. However, Ziegler et al. (2015) cautioned that sub-lethal effects of temperature in wild fish populations should be explored to understand how temperatures affect other ecosystem components, such as disease, with a changing climate. Further, native populations have already suffered large range contractions that researchers attribute to introduction of nonnative trout and habitat degradation

(Gresswell 2011). Lepori et al. (2015) concluded that ecosystem function in streams with nonnative species differs in several ways from when these species are not present; for example, they reported that populations of brook trout at natural density consumed seven times more benthic invertebrates than did cutthroat trout.

Understanding the potential effects of climate change on cold-water fish is a rapidly expanding area of research, and results of this research repeatedly identify warming temperatures as a significant threat to these species. Warming of higher elevation streams may open up new habitat for fish, but this gain may be offset by losses from more fragmentation, less accessibility, and smaller tributary size. We conclude that cold-water native fish will become increasingly vulnerable to changes in climate.

Amphibians

Researchers continue to document amphibian declines globally. Adams et al. (2013) analyzed the rate of change in the probability that amphibians occupy ponds and other comparable habitat features; they concluded that amphibians, including species not typically considered of conservation concern, are declining across the conterminous United States. While reporting more declines than increases in national parks in the Rocky Mountains, Hossack et al. (2015) also observed higher colonization rates and occupancy of boreal toads and Columbia spotted frogs (*Rana luteiventris*) in beaver-influenced wetlands than in wetlands without beaver influence. Recent declines in amphibian populations in the western United States are attributed to many factors including nonnative or invasive species, land use, overexploitation, climate change, ultraviolet radiation, contaminants, and emerging infectious diseases (Hussain and Pandit 2012). Researchers have seen connections between warmer temperatures and the pathogen chytrid fungus (*Batrachochytrium dendrobatidis*), with declines in some amphibian populations (Muths et al. 2008). Lack of sufficient data limits a robust conclusion on which factors (e.g., climate change, habitat loss, introduction of predatory nonnative fish, disease, and pollution) have directly or indirectly contributed to dying amphibian populations (Hossack et al. 2015; Hussain and Pandit 2012).

Amphibians are very sensitive to temperature and moisture. Future climate may pose a larger challenge for amphibians, as higher temperatures, reduced snowpack, reduced soil moisture, and drought potentially decrease habitat availability (Corn 2005; Ryan et al. 2014). Future conditions in alpine areas could be more tolerable to amphibian populations as compared to lower montane elevations, where snowpack losses in the future could be substantial. Snowpack loss causes earlier breeding, and results in a potentially higher risk of freezing (Corn 2005). Other factors, along with changes in climate, prey species, and availability of wetland habitat, could also influence amphibians. Given the climate sensitivity and potential for population loss, amphibians have very high vulnerability to climate change.

Benthic Macroinvertebrates

Benthic macroinvertebrates include all aquatic animals exhibiting an exoskeleton (no internal backbone) and existing in or on the substrate of standing or flowing water (e.g., streams and lakes). Benthic macroinvertebrates in the low-gradient streams may include grazers, which consume plant material and serve as food for organisms such as fish. Disturbances such as seasonal flooding in high elevation catchments affect the composition of primary consumers such as these benthic macroinvertebrates; for example, Peckarsky et al. (2015) reported that as disturbances increased, the community composition of grazers switched from disturbance-intolerant taxa (caddisflies; order Trichoptera) to disturbance-tolerant taxa (mayflies; order Ephemeroptera). Disturbances also alter dispersal behavior, grazing performance, and, consequently, the benthic algae biomass.

Many of these organisms (including some insects, crustaceans, and mollusks) tolerate fairly narrow ranges of ecological conditions. At higher elevations, changes in temperature over time can influence life history traits such as emergence and mating. In a field-manipulated experiment, Harper and Peckarsky (2006) found that warming stream temperatures resulted in earlier emergence of mayflies, but reducing the streamflow did not accelerate the onset of mayfly emergence. These warming temperatures serve as a cue that enables mayflies to time their emergence during the descending limb of the hydrograph, when protruding rocks are available as sites for laying eggs. Thus, shifts in temperature and declining snowpack and streamflow could potentially lower mayfly reproductive success. At lower elevations, many stream and lake species are more tolerant to the relatively wider range of thermal conditions.

Many benthic macroinvertebrates are mobile. They drift freely in stream currents and settle onto the substrate at fairly short distances downstream. When stream insects become adults, however, they can travel upstream to deposit eggs (Graham et al. 2015). In this manner, they are not depopulated in the upper reaches of streams. However, it can be postulated that if significant warming in water temperature or greater discharge occurs, these insects would not be able to adjust their life history strategies, with consequences to population size.

Non-insect groups of benthic macroinvertebrates, such as mollusks, have little ability to move upstream or downstream to find preferable conditions. Indeed, many of these benthic macroinvertebrates are found in spring environments, where water temperature is relatively constant. Changes in the temperature regimes can result in local elimination of certain taxa.

Climate warming and subsequent drying may contribute to large shifts in the composition of benthic macroinvertebrate communities (Poff et al. 2010). These communities, with higher concentrations of cold-adapted species that survive in a narrow temperature range, are the most sensitive to projected warmer July temperatures and to less snowfall (Poff et al. 2010). Temperature-sensitive, cold-adapted species that live in higher elevations have a very high vulnerability to climate change; less temperature-sensitive, bottom-dwelling aquatic stream species that live in lower elevations will also have more stress from a warmer and drier climate (Poff et al. 2010). See also Cold-Adapted Macroinvertebrates in Chapter 4.

Riparian Plants

Dominant plant species in riparian areas of low-gradient stream reaches vary across elevations in the Rocky Mountain Region. Woody species occurring at lower elevations include cottonwoods (*Populus* spp.); at mid-elevations, species include alders (*Alnus* spp.) and birches (*Betula* spp.). Willows (*Salix* spp.) grow across a range of elevations from alpine to montane where low-gradient streams occur. Riparian plants associated with headwaters and smaller streams are primarily herbaceous species, such as blue bells (*Mertensia* spp.), mannagrass (*Glyceria* spp.), groundsel (*Senecio* spp.), and monkey flower (*Mimulus* spp.) (Cooper and Merritt 2012). At higher elevations, these plant communities can have distinctly different compositions that include sedges (*Carex* spp.), quaking aspen (*Populus tremuloides*), and Engelmann spruce (*Picea engelmannii* Parry ex Engelm.).

Riparian ecosystems are influenced by the periodic or perennial influence of flowing water, water table depths associated with stream dynamics, and flood events related to the hydrological regime (Cooper and Merritt 2012). Riparian plants are likely to respond to the projected changes in streams: earlier snowmelt, drier conditions in late summer with lowering of water tables, and fluctuations in timing and magnitude of peak flows. Earlier snowmelt that reduces late season flows and shifts the timing of runoff may cause drier conditions that stress riparian plants and inhibit them from establishing seedlings (Perry et al. 2012). Willow species, in particular, may be most vulnerable to drying as a result of more frequent and intense drought and heat stress (Perry et al. 2012). Alder population dynamics in the southern Rocky Mountains have been tied to cool climate phases, when populations expand, and to warm phases when they shrink in response to epidemics of *Cytospora* canker (Worrall et al. 2010). Decline of alder has been noted in the southern Rocky Mountains, with canker expansion and killing of branches and stems occurring during the warmest parts of the summer. Warmer phases and fewer cool phases, a distinct possibility under climate change, could result in further decline of alder in the southern Rocky Mountains.

Neely et al. (2011) rated low-elevation riparian areas as highly vulnerable, mid-elevation riparian areas as moderately vulnerable, and high-elevation riparian areas as of low to moderate vulnerability in the Gunnison Basin in Colorado. Many factors contribute to making low- and mid-elevation riparian areas more vulnerable: changes in the timing of snowmelt, flooding, and drought; decreased base flows; impacts from land use, grazing, and browsing; and increased invasive species (Neely et al. 2011). Because most low-gradient stream reaches in the Rocky Mountain Region are at mid- and low elevations (fig. 6.2), riparian plants along low-gradient mountain stream reaches are likely to be highly vulnerable to climate change.

Sensitivity to Extreme Climatic Events

BOX 6.4

Key Vulnerabilities

A high ranking is given for the overall vulnerability of low-gradient stream reaches to extreme climatic events; they are sensitive to climate extremes of drought and heat, and are moderately sensitive to flooding.

- These reaches are very sensitive to droughts, which can reduce stream habitat, limit water supply, shift and hinder aquatic organisms, reduce the flow of sediment, and increase erosion potential when streamflow returns to higher levels.
- Although flooding offers long-term benefits for stream and habitat maintenance, the increased energy associated with projected changes in annual hydrographs could destabilize streambanks and increase sedimentation, hindering aquatic habitat in the short term.
- Warming of air temperature and surface waters will result in some aquatic habitat becoming thermally unsuitable for aquatic species.

Sensitivity to Drought

Low-gradient stream ecosystems are sensitive to drought. Droughts can arise through lower snowpack, earlier melt of snowpack, or less seasonal or annual precipitation (snow and rain). Less precipitation (snow or rain) results in declining soil moisture and runoff and consequently lower flows in the stream. Groundwater may also decline. Riparian vegetation and aquatic organisms can be affected. Canton et al. (1984) studied the effects of a 1-year flow on a low-gradient stream in Colorado; total macroinvertebrate density decreased by 50 percent in the low- flow year with taxon response widely variable. Trophic group responses also varied, with shredder and predator species increasing in relative abundance during low flow, and collector-gatherer species abundant only in the normal flow year. Fish returned to the stream reaches during the normal year.

The processes of bank erosion and sediment transport in streams are altered when climate shifts rapidly (Johnson et al. 2011). Reduced flows deliver less sediment to low-gradient stream reaches, and the reduction in transported sediment increased streambank vulnerability. During subsequent higher flows and flooding, streambanks may erode more intensely (as less sediment is present to buffer the scouring effect) (Wohl et al. 2007). During the Holocene, valley filling occurred during moister periods, whereas stream incision occurred during a warmer, drier period that reduced plants and increased fire (Johnson et al. 2011).

Drought of the magnitude that is possible with climate change could also reduce instream habitat connectivity, stress riparian plants, deteriorate water quality, alter food resources and interactions between species, and reduce water supply (Lake 2003; Perry et al. 2012). Because of the potential for the increased harm that drought can bring to low-gradient mountain stream reaches, these ecosystems are likely to be very sensitive to drought.

Sensitivity to Floods

Low-gradient streams can undergo spring floods from snowmelt, and flooding from storm precipitation. Flooding can restructure channels, riparian areas, and habitats, and increase sediment delivery, but can also introduce long-term benefits to low-gradient streams. Benefits include maintaining and creating channels, and increasing the habitat complexity and diversity of streams with additions of wood and sediment (Benda et al. 2003, 2004; Miller et al. 2003). Low-gradient streams are more sensitive to flooding than higher gradient streams; the large pool volumes can become filled with sediment and not flushed out as rapidly as in higher gradient streams, causing loss of critical habitat (Wohl 2000; Wohl et al. 2007). Fire-induced flooding can increase sediment loads in the short term, as well as degrade water quality (Agnew et al. 1997; Benavides-Solorio and MacDonald 2001; Goode et al. 2012).

Flooding can result in streambed scouring and in fish embryo mortality; the risk of critical scour to fish is a function of the timing of floods relative to embryo incubation periods, location of spawning within the stream network, and egg burial depths (Goode et al. 2013). Risk of critical scour will be species-specific; for example, smaller fish bury eggs in shallow depths (Goode et al. 2013). But a modeling study by McKean and Tonina (2013) found that bank-full flows had minimal movement of sediment and gravel that would displace embryos, and overbank flows did not increase scouring or movement of sediment in the unconfined low-gradient stream they studied, a conclusion similar to that of Goode et al. (2013).

Flooding can remove riparian plants (Perry et al. 2012), and wash out beaver dams. The loss of beaver dams can reduce fish numbers, decrease water retention within the riparian area, shift riparian plant communities, and increase erosion (Westbrook et al. 2011). Although flooding may hinder species in some cases, plants and animals are likely to recover and survive in the future as the ecosystem responds. Because of this response over the long run and the long-term benefits of flooding, low-gradient stream reaches are likely to be moderately sensitive to flooding.

Sensitivity to Extreme Heat

The temperature of low-gradient streams has a strong seasonal signal, with cooler temperatures in spring, fall, and winter, and the warmest temperatures typically in summer. Stream temperature is influenced by solar radiation, heat transfer between air and water, conduction from the stream bed, and precipitation and groundwater input (Isaak et al. 2012). Climate change will affect stream temperatures by the warming of air temperature and surface waters, and potential shifts in canopy cover along riparian streams. A warming climate has already been observed to increase stream temperatures and stress fish in the Rocky Mountains (Isaak et al. 2012, 2013).

Extreme heat may most significantly impact stream habitats where species are at the edge of their tolerance to heat (Perry et al. 2012; Poff et al. 2010; Roberts et al. 2013; Wenger et al. 2011). Physiological processes for cold-water fish, such as trout, are controlled by the temperature of the ambient environment. Within the Greater Yellowstone Ecosystem in northwestern Wyoming, future climate changes include increased peak stream temperatures and protracted periods of warming from May to September. Al-Chokhachy et al. (2013a) found that these climate changes result in pronounced growth during early and late summer for Yellowstone cutthroat trout. In contrast, projected mid-21st century temperatures result in periods of increased thermal stress from July to mid-August, and low elevation streams become less suitable habitat. At high elevation, warming of cold-water temperatures was projected to extend the growing season with positive responses by fish.

Stream temperatures could lag changes in air temperature if the streams are connected to groundwater sources. Springs and seeps may reduce the sensitivity of streams to warmer conditions from climate change. Floodplains associated with low-gradient reaches historically acted as natural “sponges” for water in the drainage; the floodplain would release water slowly, helping to maintain cooler temperatures (Wohl 2000). In addition, the lack of healthy riparian vegetation along the stream may result in higher energy gains (increased temperatures) to the stream because of reduced canopy cover (Garner et al. 2014).

Isaak and Rieman (2013) modeled how fast stream temperature has shifted or will shift over future warming scenarios. Based on their analysis, stream isotherms in the last century shifted 1.5 to 43.0 mi in many streams as air temperatures increased by 1.1 °F. In the first half of the 21st century, they would shift another 3.1 to 88.8 mi if mid-range projections of an increase in air temperature of 3.6 °F occur. Slope influences the rate of change. For central Idaho, a projected 0.2 to 0.4 °F temperature increase per decade would result in isotherms shifting in steep streams (2- to 10-percent slope) at a rate of 0.08 to 0.80 mi per decade, while isotherms in flat streams (slope of <1 percent) shift at 0.80 to 15.5 mi per decade. Models are available that provide stream-specific probabilistic projections about the occurrence of juvenile bull trout (*Salvelinus confluentus*) and cutthroat trout in association with different scenarios for climate change and brook trout invasions (USDA FS n.d.). Low-gradient streams are likely to be more sensitive to temperature warming than streams on steep slopes.

Because modified temperature conditions can hinder species or cause mortality, low-gradient stream reaches have a very high sensitivity to the potential effects of extreme heat events in the Rocky Mountains.

Intrinsic Adaptive Capacity to Climate Change

BOX 6.5

Key Vulnerabilities

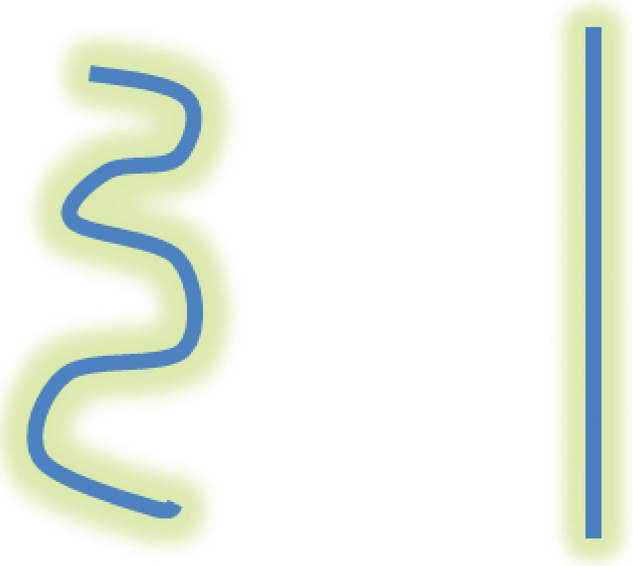
The intrinsic adaptive capacity of low-gradient streams is ranked as moderately vulnerable to climate change. The capacity of low-gradient stream reaches to withstand precipitation variability and warming can be strengthened as well as weakened by geomorphic and geological conditions, some of which can fluctuate with environmental forces.

- Factors that strengthen adaptive capacity:
    - Wider valley bottoms generally slow water flows and sediment transport; low-gradient slopes contribute to increased residence time of water in this ecosystem.
    - Functioning riparian areas stabilize streambanks and provide shade, which helps maintain cool stream temperatures.
    - Large wood, typically deposited on floodplains and in stream channels, helps sort sediment and increase habitat complexity.
    - Low-gradient stream reaches underlain by calcareous bedrock tend to have greater contributions from groundwater flow and may be better able to withstand drier and warmer conditions.
- Factors that weaken adaptive capacity:
    - Large variations in high flows and flood-related alterations of sedimentation from upstream disturbances, particularly over extended periods, can affect recovery of these systems.
    - Streams that do not have functioning riparian areas are less adapted to flooding, drought, and temperature increases.

Factors That Strengthen Adaptive Capacity to Climate Change

The capacity of low-gradient stream reaches to adapt to precipitation variability and warming can be strengthened as well as weakened by geomorphic and geological conditions, some of which can fluctuate with environmental forces. Low-gradient stream reaches generally occur in wider valley bottoms, which allow water to flow and move across wider areas compared to stream segments in narrower valleys and steeper slopes. Water flows and sediment transport occur less rapidly in these wider valley bottoms, thus helping to develop more complex habitat that features meandering streams, runs, and pools (Wohl et al. 1993) (fig. 6.3). The low-gradient slope and loose gravel and silt contribute to the likelihood that subsurface water will be retained in these streams. Watersheds underlain by calcareous bedrock also tend to have greater contributions from groundwater flow (Wohl et al. 2007). Springs, seeps, and other

Figure 6.3—Comparison between stream (blue) and riparian area (green-gray) of a meandering (left) vs. a straight (right) stream structure. A meandering stream supports a greater amount of riparian area, which in turn promotes more water retention on the landscape.



groundwater sources may also exist in many areas of the Rocky Mountain Region, potentially mitigating drier conditions from climate change (Jones and Petreman 2012). Glacial meltwater can also mitigate drier conditions. In the Wind River Range in Wyoming, glacial melting caused by warmer temperatures has contributed 4 to 10 percent of July-to-October streamflow since 1985 (Cheesbrough et al. 2009). Streamflow from glacial meltwater is temporary, however, as projections show it will disappear before mid-century (for example, Glacier National Park in Montana; Hall and Fagre 2003). Low-gradient reaches retain sediment and develop landform complexity with their meandering, thereby helping deciduous riparian communities establish, which can contribute high-quality plant matter to the channel (Gregory et al. 1991). This plant matter—leaves, bark, and branches—provides habitat and food for bottom-dwelling stream insects and macroinvertebrates. Low-gradient stream reaches often support beavers; beavers increase hydrological and ecological complexity by building dams, which help to retain water in the riparian area.

In places where functioning riparian areas border low-gradient stream reaches, adaptive capacity to flooding, drought, and heat is strengthened. Low-gradient stream reaches are commonly dominated by extensive willow communities and other plants that recover relatively quickly after a disturbance (Dwire and Kauffmann 2003). Riparian plants provide large wood to the stream, bank stability, food for aquatic and riparian species, and shade (Naiman et al. 2005).

Adaptive capacity to disturbances is strengthened by high biodiversity; high productivity and high biodiversity contribute to adaptive capacity at lower elevations and larger stream sizes (Ward 1998). The typically wider floodplains have finer grained soils, and support diverse plant and macroinvertebrate communities (Ward 1986). Kozel et al. (1989) found that these channel types are generally dominated by riffles and pools, habitat that benefits the life history of larger fish.

Factors That Weaken Adaptive Capacity to Climate Change

Low-gradient stream reaches are response areas; their large wide valleys receive high flows and disturbances from upstream. However, these systems are not adapted to large shifts in the hydrological system from drought and flooding over extended periods (see Sensitivity to Drought and Sensitivity to Floods sections). In areas where riparian plants have been removed or damaged along low-gradient streams, adaptive capacity to drought and floods is weakened. Additions of large wood and leaf litter to the stream, which are necessary for insects and fish (Naiman et al. 2005), are reduced or eliminated under low flows. Benefits that riparian plants provide—stabilizing streambanks and providing shade, which maintains cool water temperatures (Naiman et al. 2005)—can be compromised where plants are removed.

Dependence on Specific Hydrological Regime

BOX 6.6

Key Vulnerabilities

A very high vulnerability ranking is given for dependence of low-gradient stream reaches on a specific hydrological regime, as the functioning of these ecosystems and of the species living within them, is highly dependent on a snow-driven hydrological system.

Low-gradient mountain stream reaches are dependent on a snow-dominated hydrological regime. In mountainous areas of the Rocky Mountain Region, hydrological regimes range from snow-driven above an elevation of about 8,200 ft, through a mixed snow- and rain-dominated system in the montane below about 8,200 ft, to a more rain-dominated hydrological system in the foothills (Regonda et al. 2004). Spring flooding from snowmelt depends on an upstream drainage area that is snow dominated. Peak snowmelt benefits streams by delivering a bank-full flooding pulse that maintains channel size (Poff et al. 1997). Willow decline in Rocky Mountain National Park was tied to a decrease in peak streamflows and an increase in moose (*Alces alces*) populations (Kaczynski et al. 2014).

The western United States has had greater reductions in snowpack at elevations below 8,200 ft that appear to be caused by earlier snowmelt, and from precipitation shifting to more rain instead of snow (Knowles et al. 2006; Mote et al. 2005; Regonda et al. 2004; Stewart et al. 2005). Projections for the Colorado River Basin also show relatively greater reductions in snowpack over the next century at lower elevations, and moderate losses at higher elevations (Christensen and Lettenmaier 2007; Lukas et al. 2014). A warmer climate that shifts the current snow-driven hydrological system to one with more snow mixed with rain, or rain dominated, would cause greater flow variation in winter, shift peak flows earlier during spring, and lower summer late-season flows (Baron et al. 2000b; Stewart et al. 2004; Stonefelt et al. 2000). Earlier peak flows and reductions in summer low flows are likely to occur as temperatures warm (Stonefelt et al. 2000), resulting in more variation in seasonal water availability for wildlife. Hydrological changes caused by climate, such as less snowpack, could reduce channel maintenance benefits, degrade stream habitat, and increase riparian plant stress, especially during summer low flows.

Likelihood of Managing Climate Change Effects

BOX 6.7

Key Vulnerabilities

A moderate vulnerability ranking is given, as there is a likelihood of managing and mitigating climate change impacts on low-gradient stream reaches, where feasible.

- Many management options are available to mitigate the negative effects from flooding or reduced water supply, and to offset the effects of heat and drought.
- Numerous aquatic and riparian management options are known to promote resilience of aquatic habitats; however, some management actions may not always be economically, politically, socially, or ecologically feasible.

Management options for lessening the effects of climate change on aquatic habitats include reducing current stressors, strengthening ecosystem resilience and, as climate continues to change, helping the ecosystem adapt without substantial loss of soil, soil nutrients, and plant cover (Millar et al. 2007). Managers have successfully used methods to reduce current stressors to aquatic communities and enhance resilience to climate change (Luce et al. 2012; Rieman and Isaak 2010). These management practices (Rieman and Isaak 2010; Wohl et al. 2005) include:

- maintaining or restoring the natural hydrological regime;
- maintaining and restoring forest and streamside plants to minimize flooding impacts;
- maintaining or restoring riparian areas, floodplains, and wetland areas and their connection to streams;
- reintroducing beaver;
- protecting or restoring critical or unique habitats that buffer survival of the species when conditions are unfavorable;
- disconnecting roads from the drainage network;
- limiting or stopping the introduction of nonnative species;
- eliminating or controlling pollutants or contaminants;
- removing or modifying barriers to the movement of fish;
- maintaining or reconnecting large networks of habitat;
- conserving or restoring diverse habitats across basins;
- helping the ecosystem make a transition to a new state, which may require transporting fish to inaccessible habitat or places of refuge to maintain genetic variety and to allow new species to colonize when native fish species can no longer survive;
- reducing the impacts of livestock grazing;
- increasing riparian plant cover to provide shading and maintain cool stream temperatures;

- allowing the natural change of stream channels; and
- minimizing soil compaction by limiting roads, trails, campgrounds, and grazing.

Even though many management options exist to reduce stressors, they may not completely alleviate the effects of severe flooding, warming, and drought. Management actions can reduce stressors if the right technique is used at the right scale (Furniss et al. 2010). Riparian plants maintain stream temperatures by providing shade, but this maintenance of water temperature may or may not be enough to maintain or create thermally suitable habitats for aquatic species. Water diversions reduce low-gradient stream resilience; however, human water demands from agriculture and cities are expected to increase in the future. Development of management strategies also requires incorporation of new information. For example, if some stream habitats where species are targeted for restoration develop unsuitable temperatures, management strategy may have to change to address species viability (Auerbach et al. 2012).

Potential for Climate Change to Exacerbate Effects of Nonclimate Stressors, or Vice Versa

BOX 6.8

Key Vulnerabilities

A very high vulnerability ranking is given, as several factors exacerbate climate and nonclimate stressors in low-gradient mountain stream reaches.

- Climate change is likely to continue to increase fire frequency, causing erosion and subsequent harm to low-gradient stream function.
- A warmer climate is projected to reduce streamflows; human water use could considerably exacerbate the effects of drying from climate change.
- Some nonnative and invasive species may expand in range with climate change, potentially altering ecosystem function.

Climate Change and Disturbance

Climate change has exacerbated the effects of some nonclimate disturbance stressors on low-gradient stream reaches. Recent increases in fire frequency have been linked to drought, warmer temperatures, and reduced snowpack, trends that are likely to continue in the future (Westerling et al. 2011, 2014). Increased fire activity could cause more sediment to be added to streams; alter stream chemistry; create short-term, harmful consequences for local aquatic populations; and increase frequency of disturbances, such as debris flows that degrade water quality (Luce et al. 2012; Rhoades et al. 2011; Rieman and Isaak 2010). Researchers have also seen a link between warmer temperatures and increased bark beetle (*Dendroctonus* spp.) outbreaks (Bentz 2009), but there is no scientific consensus on the effects of these outbreaks on the quantity of streamflow or stream temperatures (Lukas and Gordon 2010).

In high elevation areas of Colorado and Wyoming where glaciers exist, glacial outburst floods may occur as temperatures warm. Many glaciers lie within national forests in the Rocky Mountain Region (Gage and Cooper 2013; Oswald and Wohl 2008; Veblen and Donnelly 2005; see Dependence on Hydrological Regime in Chapter 4).

Climate Change and Human Water Use

Climate warming and drying in combination with water withdrawals for human use have ecological consequences for low-gradient streams. Water is taken by diversions, sometimes across basins, and can be stored in reservoirs for use by agriculture and municipalities, or can be used for snowmaking by the ski industry. The demand for water is likely to increase as human populations grow (Foti et al. 2012), with further consequences to hydrological and ecological function. Crop water use is projected to grow in the Colorado River Basin (Colorado Water Conservation Board 2012). Rajagopalan et al. (2009) projected that increased human water use in combination with a reduced water supply from a warmer and drier climate threatens the sustainability of the water supply in the Colorado River Basin. Less available water can change the hydrological and ecological function in low-gradient streams. These changes could mean the following: a shift in the riparian plant community composition to fewer aquatic plant species, lower plant biodiversity, reduced channel sizes and sediment movement, and changes in stream chemistry and nutrient transport (Caskey et al. 2014; Wohl 2006).

Climate Change and Water Quality

Expected changes in climate could alter stream chemistry and increase channel disturbance and sediment dynamics from a possibly greater number of floods (Clow 2010; Hamlet and Lettenmaier 2007). In areas where glaciers, and especially glaciers embedded with rocks, are melting from warming temperatures, meltwater can flush nitrogen from exposed sediments. Researchers suggest that in the Colorado Rocky Mountains, water quality limits were exceeded by the addition of nitrogen from glacial meltwater (Baron et al. 2009). Historical and recent land use development has increased nutrient levels in many streams. In addition, historical mining activity has contributed pollutants as well as increased sediment to areas downstream of mine operations (Wohl 2015). A reduction in streamflows would concentrate pollutants, which potentially could reach lethal levels for aquatic and riparian taxa. Low-gradient streams may be particularly sensitive to the addition of nutrients (and the potential for nuisance algae growth) because the residence time of water is longer in those streams (Wohl et al. 2007).

Climate Change and Nonnative or Invasive Species

Some invasive plant species that occur in riparian areas, such as leafy spurge, may decrease in cover as a result of climate change, but others may expand their range, displacing native species and disrupting ecological function (cheatgrass and tamarisk [*Tamarix* spp.], Bradley et al. 2009, 2010; Canada thistle, Al-Chokhachy et al. 2013b). Nonnative fish that are more tolerant of warmer temperatures and earlier runoff flows, such as rainbow trout, may interbreed with native species, or displace native populations (Gresswell 2011; Wenger et al. 2011).

Summary of the Regional Vulnerability of Low-Gradient Mountain Stream Reaches

The overall regional vulnerability ranking for the ecosystem is a combination of the vulnerability to nonclimate stressors and the vulnerability to climate stressors. For low-gradient mountain stream reaches, the mean overall vulnerability ranking was very high with mean expert reviewer rankings in the high or very high category (fig. 6.4, table 6.2). Mean vulnerability rankings were very high for the nonclimate stressor vulnerability and high for the climate vulnerability (fig. 6.4). Expert reviewer rankings were in the high or very high category for both vulnerabilities.

The expert reviewers had high overall confidence in the ranking of the overall regional vulnerability (fig. 6.5). The four experts' individual rankings of overall confidence ranged from the high to the very high category (table 6.3). Mean confidence was very high for the nonclimate stressor vulnerability ranking, and individual experts' confidence rankings were in the high or very high category (fig. 6.5a). Mean confidence was high for the climate vulnerability ranking, with individual rankings ranging from the moderate to the very high category (fig. 6.5b). Calculations of mean vulnerability and confidence were based on the four complete reviewer rankings.

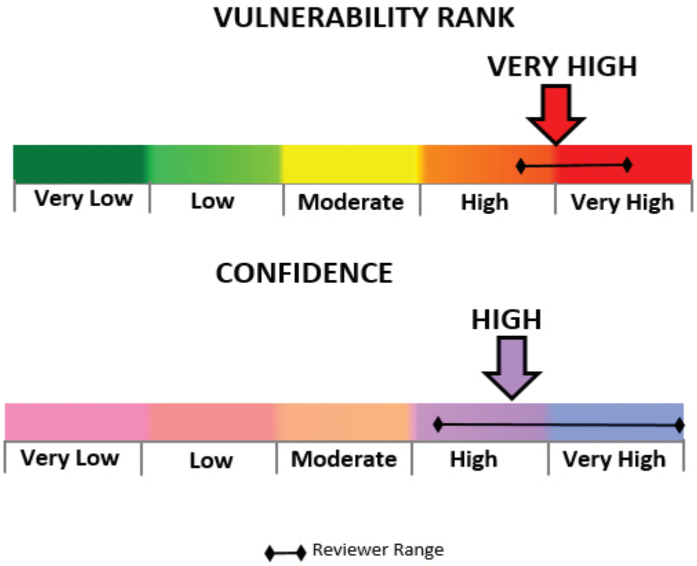


Figure 6.4—Overall regional vulnerability ranking and confidence ranking for low-gradient mountain stream reaches in the Rocky Mountain Region. The large arrow points to the mean score for vulnerability and for confidence. Bar represents the range of scores. Only four reviewer scores are included in mean calculations here and in tables 6.2 and 6.3.

Table 6.2—Original and reviewer rankings for the regional vulnerability of low-gradient mountain stream reaches of the Rocky Mountain Region. Underline indicates a reviewer score change from the original assigned score.

Criterion	Original score	(Reviewer 1) Score	(Reviewer 2) Score	(Reviewer 3) Score	(Reviewer 4) Score	(Reviewer 5) Score *	Reviewer criterion rank mean
Nonclimate							
1. Extent	5	5	5	5	5	5	Very high
2. Human influences	5	5	5	5	5	5	Very high
3. Resilience	3	3	5	3	3	3	High
4. Future trends	5	5	5	3	5	5	Very high
Total	18 Very high	18 Very high	20 Very high	16 High	18 Very high	18 Very high	18 Very high
Climate							
1. Ecosystem shift	5	3	5	5	5	5	Very high
2. Species groups	5	5	5	5	5	5	Very high
3. Climatic events	5	4	5	3	5	5	High
4. Adaptive capacity	3	3	3	3	3	3	Moderate
5. Hydrology	5	4	5	5	5	5	Very high
6. Management	1	3	3	3	1	1	Moderate
7. Interactions	5	5	5	5	5	5	Very high
Total	29 High	27 High	31 Very high	29 High	29 High	29 High	29 High
Overall vulnerability rank	24 High	23 High	26 Very high	23 High	24 Very high	24 Very high	24 Very high

* Calculation of mean vulnerability and confidence (table 6.3) included only the four complete reviewer rankings.

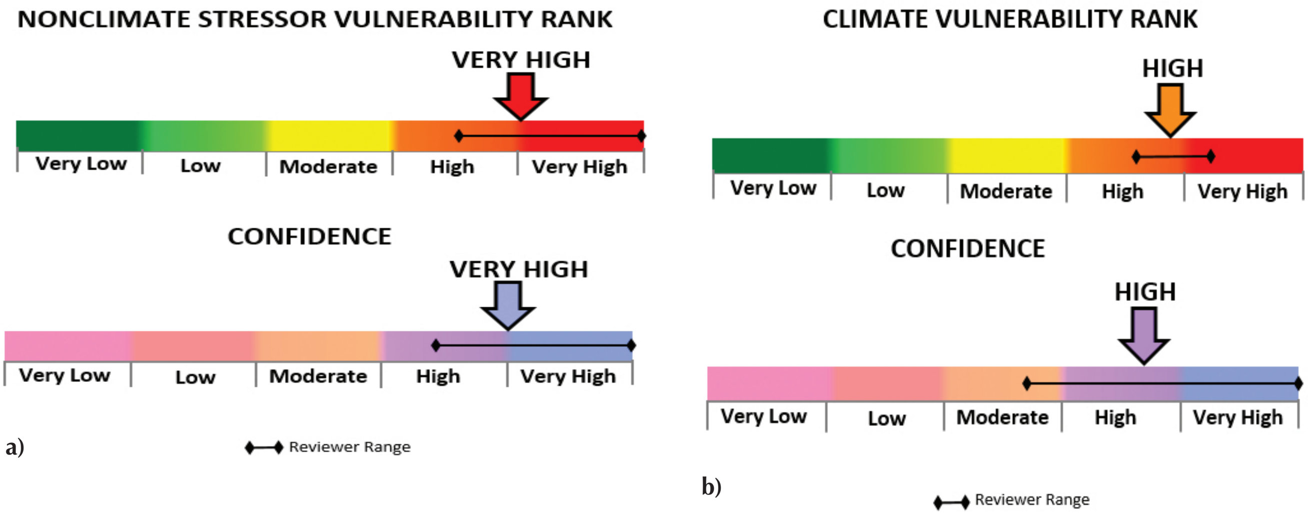


Figure 6.5—Regional vulnerability of low-gradient mountain stream reaches to nonclimate (a) and to climate (b) stressors in the Rocky Mountain Region. The large arrow points to the mean score for vulnerability and for confidence. Bar represents the range of scores. Only four reviewer scores are included in mean calculations here and in tables 6.2 and 6.3.

Table 6.3—Reviewer rankings for confidence in the vulnerability assessment of low-gradient stream reaches in the Rocky Mountain Region.

Criterion	(Reviewer 1) Score	(Reviewer 2) Score	(Reviewer 3) Score	(Reviewer 4) Score	(Reviewer 5) Score *	Criterion rank confidence mean
Nonclimate						
1. Extent	5	5	5	5	5	Very high
2. Human influences	5	5	5	5	5	Very high
3. Resilience	5	3	3	5	3	High
4. Future trends	3	3	3	5	3	High
Total	18 Very high	16 High	16 High	20 Very high	16 High	18 Very high
Climate						
1. Ecosystem shift	3	3	5	5	5	High
2. Species groups	5	5	3	5	5	Very high
3. Climatic events	3	3	3	5	3	High
4. Adaptive capacity	5	3	3	5	1	High
5. Hydrology	3	3	3	5	5	High
6. Management	3	3	5	5	*	High
7. Interactions	5	3	5	5	5	Very high
Total	27 High	23 Moderate	27 High	35 Very high	*	28 High
Overall confidence rank	23 High	20 High	22 High	28 Very high	*	23 High

* Calculation of mean vulnerability (table 6.2) and confidence included on the four complete reviewer rankings.

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Chapter 7. Ponderosa Pine Ecosystems: Vulnerability to Nonclimate and Climate Stressors in the U.S. Forest Service Rocky Mountain Region

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Quick Look: Ponderosa Pine Ecosystems in the Rocky Mountain Region

Two ecological systems of ponderosa pine forests are found in the Rocky Mountain Region. Ponderosa pine woodlands have dense stands; ponderosa pine savannas have widely spaced trees and open, park-like grassy areas. Although this narrative focuses only on these two ponderosa pine-dominated forests, ponderosa pine trees grow in other types, such as mixed-conifer. Ponderosa pine ecosystems provide habitat for a wide variety of wildlife including mule deer, wild turkey, cavity nesting birds, and avian predators.

Quick Look: Vulnerability of Ponderosa Pine Ecosystems in the Rocky Mountain Region

Vulnerability to nonclimate and climate stressors: Moderate

Confidence: High

Exposure: Variability in annual and seasonal precipitation, warming temperatures, more frequent and intense drought, longer growing season.

Current extent: Ponderosa pine savannas occur on approximately 1.6 million ac on the Colorado Front Range and in southwestern Colorado. Ponderosa pine woodlands dominate on an additional 4.8 million ac in the Rocky Mountain Region.

Sensitivity and adaptive capacity to climate change: The natural range of ponderosa pine extends well beyond the Rocky Mountain Region, to the south as well as to the north. Upslope expansion is possible in areas of Colorado and Wyoming. Ponderosa pine ecosystems in Nebraska, eastern Wyoming, and lower elevations of the Black Hills of South Dakota, the Front Range, and southwestern Colorado may be most at risk of being replaced by grassland, shrub, or other woodland types. Warming and highly variable precipitation may hinder seedling establishment and seed and cone production of ponderosa pine as well as regeneration of understory plants. Insect outbreaks and wildfires increase habitat for some wildlife, such as cavity nesting birds that depend on snags; however, these disturbances reduce habitat for species such as northern goshawks and squirrels that depend on mature forest structure. In addition, fire size may be larger and severity higher with climate change.

Nonclimate stressors: Natural and human-caused disturbances (beetle outbreaks, fire, logging, grazing) have altered the structure, composition, and function of these ecosystems. Their impacts are likely to continue, with climate change potentially exacerbating their effects.

Introduction

Ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson) ecosystems, also called forests in this report, include the two ecological systems of ponderosa pine woodlands and savannas defined by Comer et al. (2003), and cover mountainous areas of the Forest

Service, U.S. Department of Agriculture (USFS) Rocky Mountain Region and beyond in western North America. Ponderosa pine woodlands have denser stands, whereas the less common ponderosa pine savannas have widely spaced trees and open, park-like grassy areas. Ponderosa pine woodland canopy cover can range from 26 to 100 percent, whereas savannas have sparse canopy cover. Within the Rocky Mountain Region, ponderosa pine ecosystems are widespread, dominating the lower montane of Colorado and Wyoming, the Black Hills in South Dakota, and high areas in Nebraska (fig. 7.1).

Ponderosa pine ecosystems can mix with higher elevation vegetation types, and ponderosa pine trees are scattered in cliff and canyon plant communities across the western Great Plains area in the Region. Ponderosa pine woodlands and savannas can mix with Rocky Mountain montane grasslands, montane dry-mesic and mesic mixed-conifer forests and woodlands, subalpine montane limber-bristlecone pine woodlands, and aspen forests and woodlands. Tree species that can mix with ponderosa pine in the upper elevational ranges of ponderosa pine ecosystems include: white spruce (*Picea glauca* (Moench) Voss) found in the Black Hills, lodgepole pine (*Pinus contorta* Douglas ex Louden), Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco), bristlecone pine (*Pinus aristata* Engelm.), and limber pine (*Pinus flexilis* James), and in more mesic sites, Engelmann spruce (*Picea engelmannii* Parry ex Engelm.) and quaking aspen (*Populus tremuloides* Michx.). At lower elevational ranges, ponderosa pine savannas and woodlands can mix with the central mixed grass prairie and foothills grasslands, juniper woodlands, Gambel oak shrublands, lower montane woodlands and shrublands, western Great Plains canyons, and cliff and canyons semi-desert shrub steppe. Co-occurring species at lower elevational ranges include pinyon pine (*Pinus edulis*) found in Colorado, Gambel oak (*Quercus gambelii* Nutt.), and juniper (*Juniperus* spp.), as well as numerous grass species (Comer et al. 2003).

Ponderosa pine ecosystems provide habitat for a wide variety of wildlife including mule deer (*Odocoileus hemionus*) and wild turkey (*Meleagris gallopavo*); cavity nesting birds such as woodpeckers (family Picidae), northern flicker (*Colaptes auratus*), and nuthatches (*Sitta* spp.); and predators such as northern goshawk (*Accipiter gentilis*) and great horned owl (*Bubo virginianus*). Typical understory plants of ponderosa pine ecosystems include the following shrubs: antelope bitterbrush (*Purshia tridentata*) and mountain snowberry (*Symphoricarpos oreophilus*), both of which occur in much of the Rocky Mountain Region but are absent in the Black Hills; kinnickinnick (*Arctostaphylos uva-ursi*); black sagebrush (*Artemisia nova*); big sagebrush (*Artemisia tridentata*); greenleaf manzanita (*Arctostaphylos patula*) only in Colorado; mountain mahogany (*Cercocarpus* spp.); Stansbury cliffrose (*Purshia stansburiana*) in Colorado; Gambel oak, absent in the Black Hills; and chokecherry (*Prunus virginiana*), Saskatoon serviceberry (*Amelanchier alnifolia*), and rose (*Rosa* spp.). Common grasses in ponderosa pine woodlands can include bluebunch wheatgrass (*Pseudoroegneria spicata*), and species of needle and thread (*Hesperostipa* spp.), needlegrass (*Achnatherum* spp.), fescue (*Festuca* spp.), muhly (*Muhlenbergia* spp.), and grama (*Bouteloua* spp.). Grasses common in ponderosa pine savannas include: Arizona fescue (*Festuca arizonica*) in Colorado; bluebunch wheatgrass, big bluestem (*Andropogon gerardii*), little bluestem (*Schizachyrium scoparium*), fescue species (*Festuca* spp.), and blue grama (*Bouteloua gracilis*) (Comer et al. 2003).

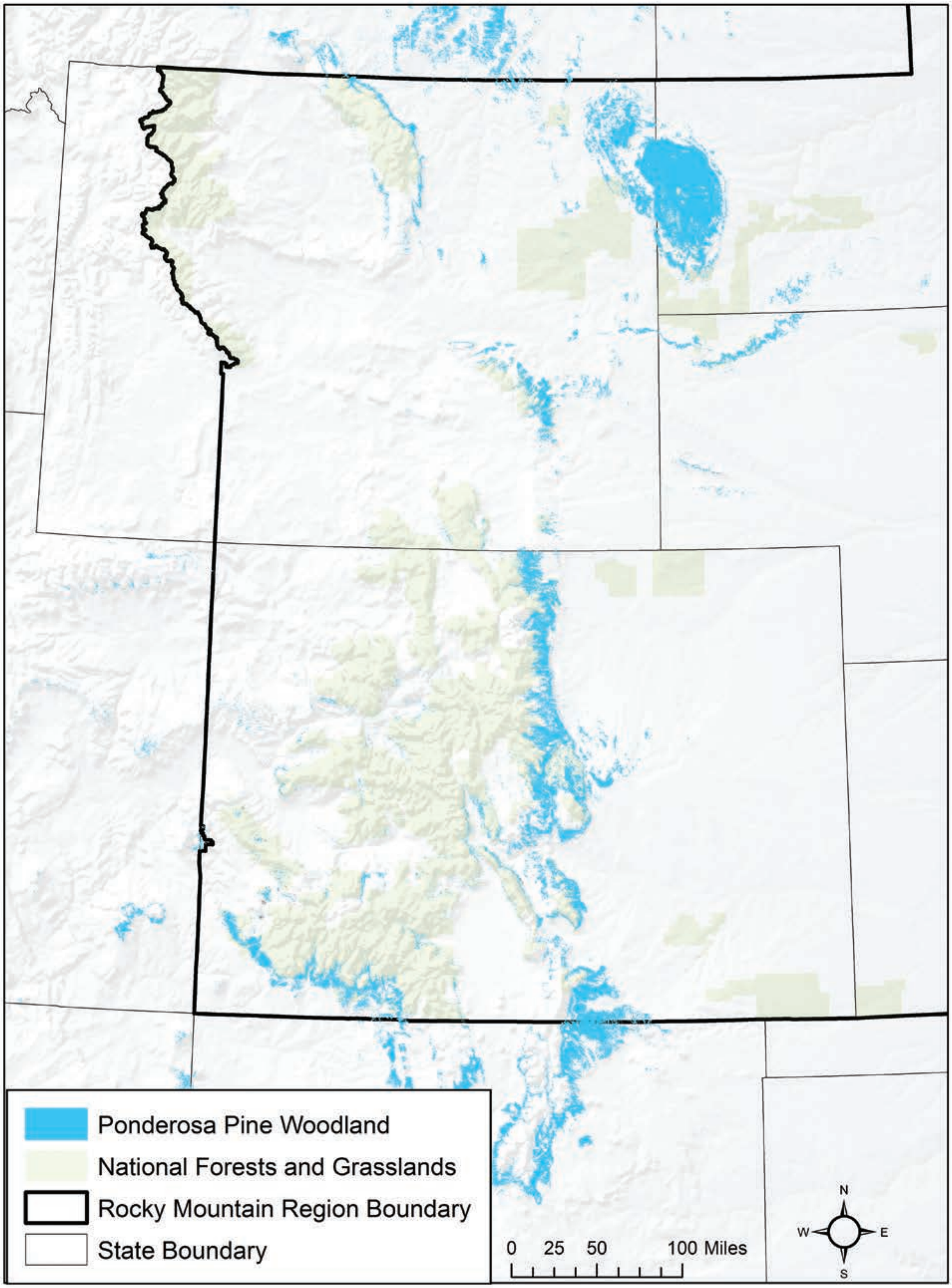

Figure 7.1—Ponderosa pine savanna and woodlands (Comer et al. 2003). (Data can be found in NatureServe 2014.)

We spatially differentiate and describe vulnerability of the ponderosa pine ecosystems in the Rocky Mountain Region by three areas based on distinct disturbance regimes and climate: southwestern Colorado, the Black Hills, and the Front Range. Southwestern Colorado ponderosa pine forests have had a historical fire regime more similar to southwestern ponderosa pine found in Arizona and New Mexico. Southwestern ponderosa pine ecosystems had a pre-Euro-American settlement fire regime of more frequent and less severe fires (Roos and Swetnam 2011), most of which occurred before the summer monsoon season (Hunter et al. 2007). The Black Hills and Colorado Front Range have historically had a low- or mixed-severity fire regime, burning less often and creating greater diversity over the landscape than in southwestern ponderosa pine forests (Ehle and Baker 2002; Sherriff et al. 2014). Some ponderosa pine forests, typically more dense woodlands and forests in the upper montane of the Colorado Front Range, have been shaped by a fire regime characterized by moderate to high severity and less frequent fire (Schoennagel et al. 2011; Sherriff et al. 2014). Lower elevation areas of the Front Range had more frequent and less severe fires (Sherriff et al. 2014; Veblen et al. 2000).

Climate plays a role in differentiating southwestern, Black Hills, and Front Range ponderosa pine ecosystem characteristics and fire regimes. The Southwest, which includes southwestern Colorado, has the driest periods during late spring and early summer; it receives the most moisture as snow during winter, and rain from the late summer monsoon. The Black Hills receive precipitation year-round, with the highest precipitation amounts falling over the summer growing season; this supports growth and prolific regeneration. The Front Range has more erratic establishment (League and Veblen 2006) and less prolific growth as this area generally receives less moisture than the Southwest and the Black Hills; the most precipitation arrives in late spring or early summer.

Ponderosa pine ecosystems have been assessed for their vulnerability to climate change. The vulnerability framework has two major components, one related to nonclimate stressors and one related to climate stressors. Within the nonclimate component, four criteria were used to define factors of vulnerability: current status of ecosystem extent, intrinsic resilience of the ecosystem to nonclimate stressors, human influences on the ecosystem, and future trends of nonclimate stressors on the ecosystem. Within the climate change component, seven criteria capture the vulnerability of individual species, sensitivity of ecosystem dynamics such as dependence of the ecosystem on the hydrological cycle, adaptive capacity of the ecosystem, potential for management to mitigate the effects of climate and nonclimate stressors, and interaction between climate change and nonclimate stressors. More information and the rationale for ranking ecosystem vulnerability by using these criteria are given in appendices A, B, and C.

Ponderosa pine ecosystems are increasingly being studied on national forest lands as well as in other areas throughout the Rocky Mountain Region. We draw from this literature where studies are within the Rocky Mountain Region. Even within this Region, these ponderosa pine ecosystems vary greatly. Application of results to ponderosa pine ecosystems on individual national forests will require an understanding of local site characteristics and the site characteristics of the research studies reviewed here.

Vulnerability of Ponderosa Pine Ecosystems to Nonclimate Stressors

Summary of Key Vulnerabilities to Nonclimate Stressors

BOX 7.1

Current status of ecosystem extent

Ponderosa pine ecosystems are widespread in the lower montane zone of the Rocky Mountains, and occur in isolated and higher areas of the Great Plains within the Rocky Mountain Region. Consequently, their vulnerability based on current extent is low.

Human influences on ecosystem

Regionally, ponderosa pine ecosystems are considered to be moderately vulnerable to the legacy of past human influences and to the stressors associated with ongoing human influences.

- Intensive livestock grazing and 20th-century fire suppression have altered the structure and function of ponderosa pine ecosystems.
- Fire exclusion has created denser forest structures and has altered the fire regime.
- Forest management, urbanization, road building, and recreational uses have fragmented ponderosa pine forests in some areas.
- Invasive species have been introduced and have altered the function of ponderosa pine ecosystems.

Intrinsic resilience of ecosystem to nonclimate stressors

A moderate vulnerability ranking is given for the intrinsic adaptive capacity of ponderosa pine ecosystems to nonclimate stressors.

- Factors that enhance the resilience of ponderosa pine ecosystems to nonclimate stressors:
  - These ecosystems occur over a wide range of soil and site conditions.
  - Plant species are generally fire-tolerant or fire adapted with understory species recovering quickly.
- Factors that lower the resilience of ponderosa pine ecosystems to nonclimate stressors:
  - Regeneration is sensitive to moisture availability and seed dispersal ability. Periodic masting is limited, delaying recovery in response to disturbance.
  - Ponderosa pine trees are susceptible to bark beetle outbreaks, which have caused widespread mortality in recent years.

Future trends of nonclimate stressors

Ponderosa pine ecosystems are considered to be very highly vulnerable to future nonclimate stressors. Current stresses associated with demand for natural resources, recreation use, and the expansion of the wildland-urban interface are likely to continue.

Current Status and Human Influences

Euro-American settlement of the late 1800s brought extensive timber harvest and large numbers of livestock to montane areas of the Rocky Mountain Region (Baron 2002; Veblen and Donnegan 2005). Research shows that domestic livestock grazing reduces tree seedling competition and can promote denser tree establishment (Milchunas 2006). Grazing also reduces understory biomass, disturbs and compacts soils, increases soil erosion, and reduces the fast-drying dead or live grass, leaves, and needles that formerly carried fires. The heavy grazing associated with 19th-century settlement very likely altered understory plant composition and productivity; however, lack of historical grazing information makes assessing the effects of grazing on specific areas challenging (Veblen and Donnegan 2005). Grazing has influenced riparian areas more than upland areas (Fornwalt et al. 2009). Grazing after the arrival of Euro-American settlers probably contributed to reduced fire occurrence (Veblen et al. 2000).

Ponderosa pine forests have had a long history of timber harvest in the Rocky Mountain Region. Logging of ponderosa pine along the Front Range of Colorado was widespread from the mid-1800s to the early 1900s (Veblen and Lorenz 1991). Before the establishment of the USFS in 1897, forests in the Black Hills were extensively clearcut (Boldt and VanDeusen 1974; Graham et al. 2015). Forests in southwestern Colorado were also harvested and were affected by domestic grazing in the early 19th century (Fulé et al. 2009). Harvest of trees continues today (USDA 2017). The most timber comes from the highly productive ponderosa pine stands in the Black Hills National Forest in South Dakota. Timber harvest and road building have fragmented landscapes, which can harm wildlife (Anderson and Crompton 2002; Montgomery et al. 2013; Reed et al. 1996). Of the 20 commonly seen bird species in the Black Hills, shelterwood cutting resulted in no effect on 6 species, greater abundance in 11 species, and less abundance in 3 bird species (Anderson and Crompton 2002). However, red-breasted nuthatch (*Sitta candensis*), brown creeper (*Certhia americana*), and ovenbird (*Seirus aurocapilla*), species associated with dense forests, were more abundant in untreated stands, suggesting a balance of treated and untreated stands is necessary to maintain interior forest bird species on the landscape (Anderson and Crompton 2002). Thinning can reduce the risk of crown fires (Hunter et al. 2007). Thinning, in some cases, has also reduced the risk from bark beetle (*Dendroctonus* spp.) outbreaks (Fettig et al. 2014). Researchers found, however, that not all silvicultural treatments are effective when high bark beetle populations exist (Schmid and Mata 2005; Six et al. 2014; see additional discussion in Likelihood of Managing Climate Change Effects).

Fire suppression has resulted in ponderosa pine forests that are more similarly aged, less patchy, and denser in Arizona (Fulé et al. 1997), the Colorado Front Range (Kaufmann et al. 2000; Veblen et al. 2000), and Montana and Idaho (Naficy et al. 2010). Of importance is understanding which habitats, prior to fire exclusion, were formed by a low- or high- or mixed-severity fire regime; this information is critical for determining where and whether fuel conditions are outside historical variability. In low-elevation Front Range forests, Sherriff et al. (2014) determined that forests historically had a low-severity fire regime. Mixed-severity fires (areas of high severity and low severity) are typical of higher elevation ponderosa pine forests in the Colorado Front Range (Schoennagel et al. 2011; Sherriff et al. 2014). Using historical observations and fire modeling, Sherriff et al. (2014) determined that less than 20 percent of the Colorado Front Range has seen a shift from frequent surface fires to less frequent, higher severity crown fires, and this shift is predominantly at elevations lower than 7,200 ft. In southwestern Colorado, fire regimes in the dry mixed-conifer forests before European settlement have been described as low severity (Brown and Wu 2005; Fulé et al. 2009). The mesic gradient of these dry mixed-conifer forests has a mixed-severity fire regime, similar to the mesic mixed-conifer forests (Tepley and Veblen 2015).

Expansion of the wildland-urban interface has occurred across lower montane areas (Theobald and Romme 2007). Colorado and Wyoming were among the 16 States with the greatest proportional expansion of wildland-urban interface from 1970 to 2000. Based on data for 2000, Colorado and Wyoming (all vegetation types) were found to have about 50 percent and 78 percent, respectively, of their wildland-urban interface within high-severity fire classes. Expansion of human communities has fragmented and reduced habitats, increased wildfires, contributed to water and air pollution, and spread invasive and nonnative species (Hansen et al. 2005). Theobald and Romme (2007) projected that Colorado wildland-urban interface would increase 21 percent by 2030. Additionally, high numbers of recreationists use Rocky Mountain Region national forests—about 28.3 million visits per year between 2011 and 2015 (USDA FS 2015b). Construction of recreation travel corridors (roads and trails) fragments ecosystems. Heavy recreational use in some areas has degraded these landscapes (Hansen et al. 2002). Along recreational trails, researchers found increased destruction of bird eggs and nestlings by predators (Miller et al. 1998). Human activity along trails can affect interactions between wildlife predators and prey, such as elk (*Cervus elaphus*) and wolves (*Canis lupus*) in Canada, in complex ways (Rogala et al. 2011).

Invasive and Nonnative Species

Many species of invasive plants have been introduced to ponderosa pine ecosystems (Brooks et al. 2016; Fornwalt et al. 2010; Symstad et al. 2014). The most frequently mentioned include cheatgrass (*Bromus tectorum* L.), leafy spurge (*Euphorbia esula* L.), Canada thistle (*Cirsium arvense*), and spotted knapweed (*Centaurea biebersteinii* DC). These and other invasive species have altered ecosystem function and nutrient cycling, displaced native plants and animals, increased fire probability and erosion, reduced biodiversity and forage, and displaced native species (Brooks et al. 2016; Kohl et al. 2012; Seig et al. 2003). One mega-wildfire introduced new invasive species to the understory of a Front Range ponderosa pine forest; however, native species accounted for more than 89 percent of cover in burn-affected communities at 10 years postfire (Abella and Fornwalt 2015; Fornwalt et al. 2010). Seventeen exotic species increased their presence postfire, including the following noxious weeds in Colorado: cheatgrass, musk thistle (*Carduus nutans*), Canada thistle, butter and eggs (*Linaria vulgaris*), and common mullein (*Verbascum thapsus*) (Abella and Fornwalt 2015).

Intrinsic Resilience of Ecosystem to Nonclimate Stressors

Factors That Enhance Resilience to Nonclimate Stressors

Plants are more resilient to nonclimate stressors when they have adapted to a wide range of environmental conditions. Ponderosa pine trees can grow in a wide variety of soil conditions, including low nutrient status and a wide range of soil pH (Oliver and Ryker 1990).

Ponderosa pine and associated understory species are generally classified as fire- and drought-tolerant. Studies show ponderosa pine of 2-in diameter at breast height survives most surface fires because of its insulating thick bark (Battaglia et al. 2009; Schoennagel et al. 2011). Grasses, forbs, and shrubs found in ponderosa pine ecosystems can regenerate or resprout quickly. In the Black Hills, Lentile et al. (2005) found regeneration densities varied by burn severity: about 1,500 and 1,100 seedlings ac^{-1} in low and moderate burn severity patches, respectively, with no regeneration in the interior of high burn severity patches. Fornwalt and Kaufmann (2014) documented no impact or a stimulatory impact to the understory vegetation in a ponderosa pine/Douglas-fir forest 5 years after the Hayman fire in Colorado. At 10 years postfire and with a decade of below-average precipitation, Abella and Fornwalt (2015) concluded that burning resulted in diverse and productive native understories in the ponderosa pine/Douglas-fir forest in the Colorado Front Range.

Among forest types, ponderosa pine forests have high biodiversity of birds and mammals (e.g., compared to lodgepole pine, Douglas-fir, pinyon-juniper, spruce-fir) (Finch and Ruggiero 1993). Taken together, the environmental tolerances of bird and mammal species contribute to strengthening the adaptive capacity of the ponderosa pine ecosystem to changes from disturbances such as fire and bark beetle outbreaks (i.e., nonclimate stressors). Some wildlife species are dependent on different successional stages of the ponderosa forest structure, such as mature ponderosa pine forests. Other species in ponderosa pine ecosystems benefit from bark beetle outbreaks and associated tree deaths. By increasing the amount of snags on a landscape, high-severity fires have benefited the black-backed woodpecker (*Picoides arcticus*), which uses postfire habitats in ponderosa pine forests.

Factors That Lower Resilience to Nonclimate Stressors

Ponderosa pine establishment is localized; the heavy cones with small winged seeds do not disperse far from seed-producing trees (Oliver and Ryker 1990). This limited ability to spread in large areas of fire-caused tree mortality can potentially slow recovery. Ponderosa pine establishment in the Southwest and Front Range is erratic, and dependent on wetter than average conditions (League and Veblen 2006; Savage et al. 1996, 2013). In the Black Hills, ponderosa pine tends to be more prolific (Shepperd and Battaglia 2002), as most moisture falls during the growing season. Seed crop periodicity varies across the Rocky Mountain Region: Colorado Front Range, 4 to 6 years or more; Southwest, 3 to 4 years; Black Hills, 3 years (Oliver and Ryker 1990). Mooney et al. (2011) found high interannual variability and synchrony in the production of both pollen and seed cones within populations along the Front Range. Synchrony among populations was evident at local scales, but was less so across elevations and at distances up to 17 mi. The authors suggested this lack of synchrony is the result of highly varied topographic conditions and variations in local climate.

Ponderosa pine forests may not be resilient to high-severity fires, particularly when far from a surviving forest. Chambers et al. (2016) studied five 11- to 18-year-old Colorado Front Range wildfires. They concluded that regeneration has occurred in high burn severity areas. However, regeneration is at densities lower than in unburned and in low-to-moderate severity burn areas. Regeneration declined with distance from surviving forest and as elevation decreased.

Ponderosa pines are at risk of mortality from outbreaks of mountain pine beetle (*Dendroctonus ponderosae* Hopkins), western pine beetle (*Dendroctonus brevicomis* LeConte), and pine engraver (*Ips* spp.) (Negrón and Fettig 2014; West et al. 2012). Bark beetle species are present at endemic levels in ponderosa pine ecosystems. With favorable conditions, outbreaks can occur, resulting in reduced habitat for some wildlife and changes in forest composition. Epidemic mountain pine beetle populations have affected approximately 1.483 million ac of ponderosa pine forests in Colorado, Wyoming, Nebraska, and South Dakota (Harris 2014). Warmer temperatures and limited moisture conditions that stress trees have exacerbated bark beetle outbreaks (Negrón and Fettig 2014; Negrón et al. 2009). Negrón and Popp (2004) reported, as have others, that infestation by mountain pine beetle increased with higher host tree stocking. In northern Colorado, they found that plots infested with mountain pine beetle exhibited higher basal area and stand density index for ponderosa pine and for all tree species combined. In addition, there were higher numbers of ponderosa pine trees per unit area. The likelihood of infestation of a stand was greater than 0.7 when the stand level basal area of ponderosa pine exceeded 75 ft^2 ac^{-1}. Likelihood of infestation of dominant or codominant individual trees was 0.77 when diameter at breast height was greater than 7.2 in.

Many other insects, pathogens, and diseases occur in ponderosa pine forests. For example, dwarf mistletoe (*Arceuthobium* M. Bieb. spp.; USDA FS 2015a) and Armillaria root disease (Klutsch et al. 2012) can damage the pine. The spatially discontinuous stand structure of ponderosa pine forests can serve to reduce mortality from bark beetle outbreaks (Chapman et al. 2012). Outbreaks can also have social and economic consequences for people: degrading aesthetic values of forests, reducing timber production, leaving hazard trees on landscapes that pose falling-tree risks for recreationists, and declining property values (Gebert et al. 2014; Price et al. 2010).

Future Trends of Nonclimate Stressors

Increased population growth, expansion of the urban-wildland interface, and increased demands for natural resources and recreational opportunities will continue to affect the ponderosa pine ecosystem (Hansen et al. 2002; Theobald 2000). Increased fragmentation of forested landscapes, reduced wildlife habitats, and increased spread of invasive plant species may result. Effects from a legacy of fire exclusion will continue to act as a nonclimate stressor. Combined with increasing human populations and urban expansion, these activities may lead to an increase in future fire occurrence. Invasive plants are expected to continue to negatively affect ponderosa pine ecosystems.

Vulnerability of Ponderosa Pine Ecosystems to Climate Stressors

Capacity for Range Shift

BOX 7.2

Key Vulnerabilities

A low vulnerability ranking is given for the capacity of ponderosa pine ecosystems for range shift.

- Ponderosa pine ecosystems in the Rocky Mountain Region are well north of their southern range limit in Mexico.
- Upslope area is available for range shifts in the mountains of Colorado and Wyoming, but eastern Wyoming, Nebraska, and the Black Hills have limited upslope area.
- Ponderosa pine ecosystems are widely distributed across the Rocky Mountain Region, but are isolated by lower elevation grasslands in the Great Plains. This may limit opportunities to migrate into new areas within the Region.

Southern Limit Proximity

Portions of ecosystems that are close to the southern limits of their geographical ranges may be more vulnerable to a warming climate than counterparts farther north of these edge zones. To assess the southern range limit, we use the native range of the Rocky Mountain ponderosa pine variety (*Pinus ponderosa* Lawson & C. Lawson var. *scopulorum Engelm*.; Oliver and Ryker 1990). It is one of three varieties of ponderosa pine in western North America. The Rocky Mountain ponderosa pine variety grows in areas east of the Continental Divide in north-central Montana to South Dakota, in Wyoming, and in north-central Nebraska. This variety also grows on both sides of the Continental Divide in Colorado and New Mexico, extending into Arizona and the eastern edge of the Great Basin in Nevada. Rocky Mountain ponderosa pine is well north of the southern limit of its species range: about 300 to 900 mi north of the southernmost extent of the Rocky Mountain ponderosa pine variety in northern New Mexico, and about 550 to 1,200 mi north of the southernmost extent of the Arizona ponderosa pine species varieties (*P. ponderosa* Lawson. & C. Lawson var. *arizonica* (Engelm.) and *P. ponderosa* Lawson. & C. Lawson ssp. *arizonica* (Engelm.) A.E. Murray) in Mexico (Oliver and Ryker 1990). As ponderosa pine are well north of their southern range limit, a low vulnerability is assigned.

Elevational Range Shift Potential

Potential upslope expansion areas for ponderosa pine forests exist in Colorado and Wyoming (Rehfeldt et al. 2006), although range contraction at lower elevations could occur (Shafer et al. 2001). Colorado ponderosa pine ecosystems currently occur at elevations ranging from about 6,000 to 9,500 ft in the Rocky Mountain Region (derived from Comer et al. 2003; NatureServe 2014). More than 9.7 million ac of land area in the Colorado Rocky Mountains are above an elevation of 9,500 ft (Cartesian area calculated from digital elevation model). Although ponderosa pine forests do not grow above an elevation of 9,000 ft on the Bighorn and Medicine Bow Mountain Ranges in Wyoming, there is potential for upslope expansion above this elevation. In South Dakota, eastern Wyoming, and Nebraska, ponderosa pine ecosystems generally occur at the highest elevations available, from about 3,200 to 7,200 ft, with limited potential area to expand upslope.

Areas upslope of where ponderosa pines currently grow could become climatically suitable to support the species (Rehfeldt et al. 2006). Migration into upslope areas would depend on a variety of conditions: future fires (frequency, severity, and size) that would not restrict ponderosa pine from growing, suitable soils, microclimate, and the effects of climate change on existing vegetation. Given the potential for upslope expansion of ponderosa pine forests in the Rocky Mountain Region, the potential for an elevational range shift is assigned a low vulnerability.

Connectivity

Ponderosa pine forests are spread throughout the Rocky Mountain Region. In addition to the lower montane in Colorado and Wyoming, they almost entirely cover the Black Hills, and higher areas of the Great Plains in eastern Wyoming and Nebraska. These forests are absent in vast expanses of low elevation shrublands and grasslands in Wyoming, Colorado, Kansas, Nebraska, and South Dakota. Ponderosa pine trees are rare in the northwestern Wyoming area of the Greater Yellowstone Ecosystem (fig. 7.1). Ponderosa pine trees are widely scattered across the western Great Plains region, and grow in isolated areas such as canyons and cliffs. However, large expanses of low elevation shrublands and grasslands do not support ponderosa pine ecosystems, which would be an inhibiting factor for ponderosa pine migration across the Great Plains part of the Rocky Mountain Region. Kaye et al. (2010) observed that ponderosa pine expanded over the past two centuries into the west-central Great Plains, but suggested the expansion would be negatively affected by drought under future climate. Given the wide distribution of ponderosa pine throughout the Rocky Mountain Region, and the large expanses of other vegetation types that could inhibit regional migration, connectivity is assigned a moderate vulnerability ranking.

Vulnerability of Cold-adapted, Foundation, or Keystone Species to Climate Change

BOX 7.3

Key Vulnerabilities

A moderate vulnerability ranking is given for cold-adapted, foundation, and keystone species of ponderosa pine ecosystems, although vulnerability to climate change by species can vary from low to high.

- At lower elevations, the range of ponderosa pine may contract or disappear, particularly in eastern Wyoming, Nebraska, and some areas of the Black Hills. At higher elevations, the ponderosa pine range may expand.
- Some bird and mammal species of ponderosa pine forests (such as cavity nesting birds that use snags and feed on bark beetles) may benefit from an increase in disturbance associated with climate change that provides more habitat resources. Other species dependent on mature forest structures such as squirrels or northern goshawk may decline. Responses of birds to changes in forest structure associated with changes in fire regimes will vary by species.
- Understory grasses and shrubs in ponderosa pine ecosystems are tolerant of drought and heat, and thus are expected to have low vulnerability to climate change. They may benefit from more disturbances that result in lower tree density and open tree canopies.

We selected a subset of species that live in ponderosa pine forests to capture a range of environmental tolerances and roles in ecosystem function. We selected ponderosa pine as it is a foundation species with high biomass, and an important habitat resource for wildlife. Mammals and birds are included as keystone species that greatly affect the function of ponderosa pine ecosystems; for example, squirrels distribute seeds, and cavity nesting birds feed on insects. We also assessed understory plants, as they are important food resources for wildlife.

Ponderosa Pine

Considering a variety of climate scenarios and modeling methods, researchers projected ponderosa pine extent will expand at higher elevations while declining at lower elevations (Bachelet et al. 2000; King et al. 2013; Notaro et al. 2012; Rehfeldt et al. 2006; Shafer et al. 2001). Dynamic vegetation models used by Bachelet et al. (2000) and King et al. (2013) more fully represent ecosystem processes such as fire, carbon dioxide enrichment, and plant interactions; climate envelope models consider the relationships between climate and current species distribution. Using a dynamic vegetation model, King et al. (2013) projected that by the end of the 21st century ponderosa pine trees will persist at the ponderosa pine and grassland ecotone in the Black Hills, but will decline in productivity, especially during drought. This finding was contrary to climate envelope models, which projected ponderosa pine to contract or to disappear from Nebraska and eastern Wyoming and some lower elevation areas of the Black Hills by mid-century, but continue to expand in the higher elevations of the Black Hills, Colorado, and Wyoming (Rehfeldt et al. 2006; Shafer et al. 2001).

Climate envelope models projected greater pine extinctions and range movement expansions compared to dynamic vegetation models (Morin and Thuiller 2009), but both model approaches projected some hindrance to ponderosa pine under climate change. Hansen and Phillips (2015) reviewed modeling studies focused on the Great Northern Landscape Conservation Cooperative in the Northern Rockies and ecosystems surrounding Glacier National Park in Montana and Yellowstone and Grand Teton National Parks in Wyoming. They found that the models suggested increased climate suitability for ponderosa pine, including in areas east of the Continental Divide where historically this pine had low occurrence. They concluded that for this area in Wyoming and in the Northern Rockies, ponderosa pine has moderate vulnerability to climate change.

Ponderosa pine in areas that remain climatically suitable under climate change are likely to persist if future intensity of drought or fire does not greatly hinder or limit this species. Ponderosa pine trees are, therefore, given a moderate ranking for vulnerability to climate change.

Birds and Mammals

Bird and mammal populations of ponderosa pine ecosystems can be closely tied with forest structure, which is influenced by climate and disturbance. Future changes in forest structure caused by warming, drought, bark beetle outbreaks, and fire may hinder some wildlife species, while benefiting others. Bird and mammal species tied to mature forest structures would be hindered by more fire and bark beetle disturbance, which reduces the amount of mature forest on the landscape. Tassel-eared squirrels (*Sciurius aberti*), for example, prefer a dense forest structure, and use mature ponderosa pine trees in northern Arizona (Kalies et al. 2012), habitat and food resources that are likely to become less abundant under climate change. Northern goshawks live in mature conifer forests in Colorado, Wyoming, and the Black Hills. These birds of prey prefer the largest trees for nesting areas (Squires and Reynolds 1997), and a loss of mature forest structure from climate change-induced fire and bark beetle disturbance could hinder reproduction. Drought and warming could also play a major role in reducing resource availability and stressing animals, such as deer (*Odocoileus* spp.) (Parker and Robbins 1983) and northern goshawk (Reynolds et al., 2017). Predator birds such as the northern goshawk are dependent on the availability of prey, and their populations are projected to decline when prey animal numbers decline during drought (Reynolds et al., 2017).

Cavity nesting birds may benefit from changes in forest structure associated with warming temperatures, and disturbances that create more snags. Snags in ponderosa pine forests provide them with nesting areas and shelter (Howard 2003). Bonnot et al. (2008) found that bark beetle outbreaks in the Black Hills benefited black-backed woodpeckers, offering nesting sites comparable in numbers to a postfire landscape. In other areas of the Rocky Mountain Region, black-backed woodpeckers did not have as positive a response to beetle outbreaks as bark-drilling specialists such as the three-toed, downy (*Picoides pubescens*), and hairy (*P. villosus*) woodpeckers, which rely on beetle larvae (Saab et al. 2014). In ponderosa pine forest across the western United States, American robin (*Turdus migratorius*), mountain bluebird (*Sialia currucoides*), chipping sparrow (*Spizella passerina*), Cassin's finch (*Haemorhous cassinii*), Swainson's thrush (*Catharus ustulatus*), and warbling vireo (*Vireo gilvus*) exhibited a positive relationship with beetle outbreaks, a larger degree of positive avian response than in lodgepole pine forests (Saab et al. 2014). Kotliar et al. (2008) found that three-toed woodpecker response to burn severity was strongly scale-dependent. Burn severity at the largest scale (home range) accounted for woodpecker use, but at the smallest scale (individual trees) tree size and beetle occurrence were important. Avian relationships with wildfire were found to be different across sites with low-severity and mixed-severity fires; researchers suggested that intensive fuels management may be ecologically less appropriate for promoting biodiversity in areas where mixed-severity wildfires and dense forest stands were historically more common (Latif et al. 2016).

Fire can also benefit wildlife by enhancing the diversity of native grass species (Griffis et al. 2001); mule deer or other grazing wildlife species that use open-canopy pine-grasslands may benefit from increased grass after fire. Generalists and mobile species, such as deer or birds, are able to move to areas where food resources exist (Amberg et al. 2012), a factor that can strengthen their ability to persist under climate change with its associated changes in disturbance.

Detailed species-specific information about vulnerability is currently limited; further study is needed. Given the current information, however, bird and mammal species of ponderosa pine ecosystems will have varying degrees of vulnerability to climate change. Vulnerability is high for some species, such as those dependent on mature forest structure, and low for others; still others may opportunistically be able to benefit from climate change impacts on the forest structure and composition.

Understory Vegetation

Amberg et al. (2012) ranked the vulnerability of grasslands in the Badlands National Park in South Dakota as "least vulnerable." Although this assessment was not in a forested area, many of the grass species that they assessed also occur in the understory of ponderosa pine ecosystems of the Rocky Mountain Region (e.g., blue grama, needle and thread, western wheatgrass [*Pascopyrum smithii* (rydb.) Á. Löve], and little bluestem). Amberg et al. (2012) gave a ranking of moderate vulnerability for the shrubs that they assessed, such as sagebrush and chokecherry, which also grow in ponderosa pine ecosystem understories.

Shrubs and grasses may become more productive and denser in areas where increased fires or bark beetle outbreaks open up dense tree canopies. The 10-year study by Abella and Fornwalt (2015) recorded increases in ponderosa pine understory graminoids, and short- and long-lived forbs after the 2002 Hayman fire in the Colorado Front Range. Researchers found grass density increased when ponderosa pine trees in the overstory were removed. Shrubs that can tolerate more shade than grasses have highest productivity when the density of ponderosa pine trees is intermediate, as in the Black Hills (Uresk and Severson 1989). Given that grasses and most shrubs are highly tolerant of warmer and drier climates and fire, and may benefit from increased disturbance associated with climate change, understory plants in ponderosa pine ecosystems are expected to have low vulnerability to climate change.

Sensitivity to Extreme Climatic Events

BOX 7.4

Key Vulnerabilities

A high vulnerability ranking is given for the sensitivity of ponderosa pine ecosystems to extreme climatic events.

- Ponderosa pine ecosystem plants are generally drought-tolerant, but mortality of understory plants and trees has been observed under historical drought. Future drought conditions are expected to exceed historical levels.
- Ponderosa pines are very sensitive to the availability of moisture during seed germination, seedling establishment, and, in mature trees, seed and cone development and production. Success of ponderosa seedlings may also be sensitive to warming temperatures. Increased temperatures and changes in the seasonal availability of moisture will increase the vulnerability of ponderosa pine regeneration success.
- Extreme heat events could exceed the environmental tolerances of some plant species, especially during drought conditions.

Sensitivity to Drought

Drought is the accumulated imbalance between the supply of water and the demand for water by plants, animals, the atmosphere, the soil column, and humans (Kunkel et al. 2013a,b). Ponderosa pine trees and shrubby or grassy understories are, for the most part, tolerant to historical drought conditions (Oliver and Ryker 1990), but have suffered mortality or reduced reproduction during drought events. Researchers observed that mature ponderosa pine tree deaths occurred in the Southwest during the drought periods of the early 2000s and 1950s, when annual moisture was as low as 4 in (Allen and Breshears 1998; Ganey and Vojta 2011). Droughts of the 1930s, 1950s, and early 2000s in Nebraska greatly reduced ponderosa pine establishment; however, the pine did not undergo the diebacks that occurred in the southwestern United States (Kaye et al. 2010). Drought in combination with other stressors, such as heat, insect outbreaks (Negrón et al. 2009), and topographic position (e.g., south-facing slopes; Hinckley et al. 2014), increases the likelihood of mortality in ponderosa pine. Future drought is expected to far exceed historical conditions (Cook et al. 2015) and to have greater effects on ponderosa pine than have been observed (Allen and Breshears 1998).

Regeneration of ponderosa pine is strongly linked with moisture availability (Mooney et al. 2011; Petrie et al. 2016). Krannitz and Duralia (2004) noted that during the 26-month cone development phase, below-average temperatures in late spring can kill the second-year conelets. High temperatures during the first year of cone production were associated with high cone production. Mooney et al. (2011) observed high interannual variation of tree pollen and cone production associated with climate, with similar production patterns at small scales, but wide variation across populations separated by 17-mi and 4,400 ft in elevation. In the Southwest, Feddema et al. (2013) projected future reductions in ponderosa pine regeneration, especially where seed producing trees were growing in the dry end of climate conditions (presumably around 8 inches a year of moisture). The erratic establishment in the Front Range is associated with annual variation in spring and fall moisture (League and Veblen 2006; Shepperd et al. 2006). Based on an extensive literature survey, Petrie et al. (2016) reported that ponderosa pine seedling success is highest at temperatures of 68 to 77 °F with higher precipitation and higher moisture availability. A warming climate with decreasing precipitation may reduce ponderosa pine seedling success. They cautioned that few studies have explored the effects of climate and environmental variables directly in an experimental framework. Greater understanding is needed here to project the potential effects of climate change on seedling emergence and establishment for ponderosa pine.

Cregg (1994) found that ponderosa pine seedlings from South Dakota and Nebraska generally survived drought conditions longer than seedlings from central Wyoming, Colorado, and especially New Mexico and Montana. Shorter needles, less needle surface area, and fewer stomata per needles in the more drought-tolerant seedlings could play a role (Cregg 1994). Another explanation may be that stomate sensitivity limits water loss under drought conditions, and maximizes growth during wet conditions (Zhang et al. 1997). However, warming temperatures with no change in precipitation could increase stress on seedlings and affect establishment success in the future. Despite the drought tolerance of these trees, future drought in combination with higher temperatures would very likely reduce ponderosa pine regeneration and raise the potential for a change to a grassland ecosystem (Kaye et al. 2010), especially in drier, low elevation areas where ponderosa pine may not recover after a disturbance.

Although plants within the ponderosa pine ecosystem may tolerate drought of historical magnitude and duration, ponderosa pine trees can die from extreme drought events. Seedling establishment and survival are sensitive to both temperature and lack of moisture. Given that drought intensity and duration are projected to exceed historical conditions, ponderosa pine ecosystems are likely to be very sensitive to drought under a warming climate.

Sensitivity to Extreme Heat

Ponderosa pine and associated plants are heat-tolerant. Understory plants common to ponderosa pine ecosystems can endure high temperatures (e.g., sagebrush: 115 °F) (Loik and Harte 1996). Mature ponderosa pine can survive extreme temperatures to 110 °F, and the pine is more successful in resisting high soil surface temperature with increasing age (Oliver and Ryker 1990). Kolb and Robberecht (1996) determined lethal temperature limits for ponderosa pine seedlings to be 145 °F at exposure for 1 minute. Though ponderosa pine seedlings are not particularly sensitive to extreme heat, they die when extreme heat is combined with drying (Oliver and Ryker 1990). Future extreme heat events could exceed the heat tolerance of ponderosa pine, grasses, and shrubs in some areas. Ponderosa pine ecosystems will be sensitive to extreme heat, especially when seedlings are also exposed to drought.

Intrinsic Adaptive Capacity to Climate Change

BOX 7.5

Key Vulnerabilities

A moderate vulnerability ranking is given for the intrinsic adaptive capacity of ponderosa pine ecosystems to climate change.

- Factors that strengthen adaptive capacity:
  - Ponderosa pine ecosystems occur across a wide range of climate conditions.
  - Understory species can recover from disturbances quickly.
  - Ponderosa pine trees have some genetic variability and have life history traits that help them adapt to heat and drought.
- Factors that weaken adaptive capacity:
  - Ponderosa pine trees require a long time for reproduction, potentially delaying regeneration and persistence under climate change.

Factors That Strengthen Adaptive Capacity to Climate Change

Ponderosa pine ecosystems occur across a wide range of climate conditions, in association with a diversity of plants and animals. Patton et al. (2014) found 753 bird, plant, and animal species associated with ponderosa pine forests in Arizona. Finch and Ruggiero (1993) identified up to 128 bird species and 57 mammals in ponderosa pine forests in Colorado, the Rocky Mountains, and the northern Great Plains. Ponderosa pine plant communities typically have 3 to 7 dominant understory types; several plant community types exist across the region: 12 in the Big Horn Mountains in Wyoming (Hoffman and Alexander 1976), 6 in the Black Hills (Thilenius 1971), and 5 in the Southwest (Hanks et al. 1983). Taken together, the high level of biodiversity widens the range of climates where these species have been found, and contributes to raising the adaptive capacity of species in ponderosa pine ecosystems.

Differences in the genetic composition of ponderosa pine trees can strengthen adaptive capacity to climate change. For example, studies show that genetic variation among ponderosa pine populations is abundant in the Southwest and Great Plains (Rehfeldt 1999; Van Haverbeke 1986). Genetic variability among ponderosa pine populations has been observed to differentiate regeneration responses to climate that varies geographically across the landscape (Mooney et al. 2010). A range of genetic composition among ponderosa pine widens the degree of potential responses and adaptation to variations in climate.

The ability of species to regenerate quickly also strengthens adaptive capacity to climate change. Grasses and shrubs found in ponderosa pine ecosystems can regenerate or resprout quickly (Griffis et al. 2001), enhancing the ability of plants to persist under climate change.

Ponderosa pine trees have physical characteristics that strengthen adaptive capacity to climate change. The long taproot system of mature ponderosa pine provides stability, enabling the trees to persist during floods better than shallow-rooted plants (Lawrence 1939). Studies show that ponderosa pine prefers coarse-textured soils of sandstone origin, which allow the trees to draw the limited moisture more easily than fine-textured soils of limestone origin permit (Oliver and Ryker 1990), thus increasing their capacity to adapt to drought.

Factors That Weaken Adaptive Capacity to Climate Change

Regeneration of ponderosa pine trees is slower than for understory plants, and can be variable. Ponderosa pine trees can take 20 or more years to start producing seeds, and viability of seeds from trees between 60 and 160 years is greater than trees outside of this age range, younger or older (Oliver and Ryker 1990). Seed crops can be smaller than for other conifers, and many birds and mammals can eat large quantities (Shepperd et al. 2006). Ponderosa pine establishment is localized; the heavy cones with small winged seeds do not disperse far from seed-producing trees (Oliver and Ryker 1990). These hindrances to ponderosa pine tree reproduction serve to delay recovery or persistence of trees undergoing climate change, thus weakening adaptive capacity.

Dependence on Specific Hydrological Regime

BOX 7.6

Key Vulnerabilities

A very low vulnerability ranking is given for ponderosa pine ecosystem dependence on a specific hydrological regime. These ecosystems have adapted to a wide range of variability in moisture.

Ponderosa pine ecosystems occur in areas that receive moisture as rain or snow, and survive in areas that have a wide range of annual precipitation. Regeneration of ponderosa pine forests is sensitive to moisture availability, which may be a critical factor in a potentially changing climate. Plants and animals in these ecosystems are particularly adapted to semiarid conditions and not dependent on a specific hydrological regime.

Likelihood of Managing Climate Change Effects

BOX 7.7

Key Vulnerabilities

Management options for mitigating some climate change effects (fire, insect outbreaks, drought) on ponderosa pine ecosystems are available. Management approaches to alleviate the direct effects of extreme temperatures and temperature variability may be limited. Vulnerability is ranked as low.

Management options for mitigating the effects of climate change include reducing current stressors, enhancing ecosystem resilience, and, as climate continues to change, helping plants and animals adapt without substantial loss of soil, soil nutrients, and plant cover (Millar et al. 2007; USDA FS 2011). Management actions are available to mitigate climate change effects from wildfire, insect attack, and drought. However, successful implementation depends on many factors. One of those factors is the capacity to implement the treatments at the appropriate scale, which for ponderosa pine ecosystems may be very large.

Specific management options exist to reduce the effects of drought. For example, where projections suggest a site will be suitable for ponderosa pine, mechanical removal of Douglas-fir trees in a mixed-conifer stand can favor the future survival and establishment of ponderosa pine (Underhill et al. 2014). Thinning or prescribed fire can reduce competition for resources and increase growth in young trees (Kolb et al. 2007). However, management treatments need to consider the possibility that drought may have negated silvicultural treatments aimed at growth stimulation, particularly for older trees (Kolb et al. 2007). Older trees may also have negative growth response to fire, as seen in north-central Idaho (Keeling and Sala 2012).

Managers have used many practices to reduce wildfire risk, and a careful examination of the treatments is critical in evaluating success or failure. Factors such as the size of the treatment, the degree of density reduction, completion of the treatment, historical fire regime in the area, and presence of invasive species all contribute to attaining the management objective of reducing wildfire risk. Restoring a more open forest structure and reducing surface fuels decrease the potential for stand-replacing fires (Fitzgerald 2005; Reynolds et al. 2013; Strom and Fulé 2007). Lezberg et al. (2008) compared prefire and postfire conditions in an experimental area that was burned in the Hayman fire in the Colorado Front Range. They reported that the treatments of soil scarification and overstory harvest influenced the severity of the fire. Other studies found that reducing fuels and burning prescriptively to reduce wildfire hazard were minimally effective because of extreme weather conditions in some instances, such as the Hayman fire (Graham 2003) and the Fourmile Canyon fire in Colorado (Graham et al. 2012). The scale and capacity of fuels treatments may not be broad enough to have an effect in some cases. Additional research to unravel treatment effects and wildfire is needed (Lezberg et al. 2008; Rocca et al. 2014), particularly as extreme fire weather is projected to be more common in the future (Tang et al. 2015) with the potential to overwhelm the effects of fuels treatments.

Researchers found that repeated moderate- and low-severity prescribed burns lowered seedling densities and surface fuel loads, and reduced crown fire potential and death in ponderosa pine forests in the Black Hills (Battaglia et al. 2008) and New Mexico (Hunter et al. 2011). Use of prescribed fire to maintain low densities is influenced by the size of seedlings and saplings and frequency of burns. The effectiveness is lost within 10 to 20 years if regeneration densities are not controlled (Battaglia et al. 2008).

Fire has been implicated in the spread of invasive and nonnative plant species. Symstad et al. (2014) suggested that postfire invasive outbreaks associated with prescribed fires may be reduced if the invasive species are noted and controlled on the landscape prior to fire. They acknowledged that finding these invasive species on large landscapes before the prescribed fire may be difficult, and suggested that moderating fire intensity or targeting areas of high severity for postfire invasive control may be more strategic in the Black Hills area. In southwestern Colorado, Korb et al. (2012) found that burn-and-thin treatments were more effective in restoring the historical reference conditions for the warm and dry mixed-conifer forests; burn-only treatments were not as effective, but they were less expensive.

Management options have successfully mitigated tree death from bark beetle infestation under particular beetle pressures and spatial scales. Direct methods to address current infestations use one or a combination of methods, such as fire, tree removal, insecticides, or semiochemicals (chemicals that mediate responses between organisms) (Fettig et al. 2014). Some studies show that indirect methods, such as thinning, increase tree resilience and change forest stand structure to make forests less favorable for beetles; several studies reported reduced death from beetle outbreaks in thinned ponderosa pine stands (Egan et al. 2010; McCambridge and Stevens 1982; Sartwell and Dolph 1976; Sartwell and Stevens 1975). Studies show higher beetle populations occurred in denser ponderosa pine forests, and lower beetle populations occurred in forests where managers used thinning and prescribed burning (Fettig et al. 2014; Zausen et al. 2005). The Schmid and Mata (2005) study in the Black Hills found growing stock levels (GSL) of 60 to 70 ft^2 ac^{-1} had 9 percent of trees attacked, whereas GSLs of 80 to 90 and 100 to 110 ft^2 ac^{-1} had 53 and 48 percent trees attacked, respectively. They questioned the effectiveness of thinning treatments when treatment areas were surrounded by unmanaged stands infested with outbreaks. Although more information is needed on the efficacy of thinning treatments, as results are not always consistent, researchers found most of the thinning treatments reduced mortality from bark beetle outbreaks (Fettig et al. 2007, 2014; Six et al. 2014). Additionally, the effects on birds and mammals may not be as expected with thinning, and guidelines for restoration are likely to differ by region (Hutto et al. 2014; Latif et al. 2016).

Despite the availability of several management options for reducing nonclimate stressors, these options are limited in their effectiveness in mitigating or completely offsetting the direct climate effects, and results may not always be consistent.

Potential for Climate Change to Exacerbate Effects of Nonclimate Stressors, or Vice Versa

BOX 7.8

Key Vulnerabilities

Climate change may exacerbate the magnitude, intensity, and effect of nonclimate stressors. Consequently, the vulnerability ranking for ponderosa pine ecosystems is very high.

- Drought exacerbates fire occurrence.
- High-severity fire could increase damage from flooding and the associated increased erosion following fire.
- Warmer temperatures and drought contribute to bark beetle population success and the spatial extent of the outbreaks.
- Increased risk from invasive plant species may also occur with climate change.

Climate and Fire

Climate change may alter fire dynamics. Recent drought has contributed to an increase in the frequency of fires larger than 988 ac in fire size, and lengthened the fire season since the mid-1980s in the western United States (Dennison et al. 2014; Westerling et al. 2014). Researchers expect fire to increase in frequency and to affect more area of the Rocky Mountains in the future (Rocca et al. 2014; Westerling et al.

2011). The paleo record shows a link between fire occurrence and drought, combined with warmer temperatures (Kitzberger et al. 2007; Pierce et al. 2004). Researchers project that drought and extreme fire weather will intensify under climate change (Cook et al. 2015; Tang et al. 2015). Drought in combination with high-severity fire was found to inhibit southwestern ponderosa pine regeneration in New Mexico more than drought alone (Savage et al. 2013). Tree death caused by drought may increase as temperatures warm, especially at lower latitudes and elevations. The potential increase in fire and drought could lower timber production and carbon storage capacity, contribute to less productive soils, reduce aesthetic values of landscapes, and create fewer, or less dependable, recreation opportunities. How these potential changes in fire will influence ponderosa pine ecosystems in the Black Hills, Front Range, and southwestern Colorado is complicated by the human-influenced changes in the current fire regimes.

Events associated with fire or the consequences of fire, such as postfire flooding, affect ponderosa pine ecosystems. Ponderosa pine forests are most susceptible to negative effects from flooding during a short-term increased erosion period following fire (e.g., 3–6 years for the Colorado Front Range; MacDonald and Larsen 2009). Paleo records show that during periods when ecosystems have an increase in fire frequency and stand-replacing fires, postfire erosion contributes greatly to long-term erosion (e.g., Idaho; Pierce et al. 2004). Areas with rugged topography and narrow canyons, which are typical of many ponderosa pine forested areas in the Front Range, are inherently at risk from postfire flash floods, and debris flows from summer thunderstorms (e.g., Waldo Canyon Fire; Stanley et al. 2015).

Climate and Bark Beetle Outbreaks

Recent warming has resulted in more frequent and severe bark beetle outbreaks (Bentz 2009; Negrón and Fettig 2014). Drought reduces the ability of southwestern ponderosa pine to resist bark beetle attacks (Negrón et al. 2009). In mixed ponderosa pine and lodgepole pine forests, West et al. (2014) found similar susceptibility to mountain pine beetle infestations for these two tree species. A high potential exists for other bark beetle species that are at their northern range limit in the southwestern United States and Mexico to expand their range northward (Bentz et al. 2010) as temperatures increase and isotherms shift northward. Evangelista et al. (2011) projected the western pine beetle population distribution will move farther north in Colorado and into Wyoming. The mountain pine beetle and pine engraver beetle will expand into areas of northwestern Colorado and the Greater Yellowstone Ecosystem.

Future beetle outbreaks will be sensitive to both climate and changes in vegetation. Using a mechanistic, phenology-based demographic model, Bentz et al. (2016) explored how climate change will affect thermal suitability for mountain pine beetle population growth in the Northern Rockies. This area is outside of the Rocky Mountain Region, yet their conclusions about host and beetle interactions may be important for future outbreaks in the Region. They reported that the best thermal habitats for mountain pine beetle will be at the lowest and the highest elevations. Changes in hosts were not simulated. Drawing on the modeling studies for ponderosa pine, which for this area suggest that ponderosa pine will move upslope, they suggested that this migration may be beneficial. Ponderosa pine would be moving out of the lower elevations where bivoltinism (two insect generations per year) is projected to be the greatest. Similar detailed analysis would be beneficial for the Rocky Mountain Region.

Climate and Invasive Species

Researchers expect that some invasive plants will expand in the Rocky Mountain Region as a result of climate change, while others may contract (Bradley et al. 2009). Leafy spurge will contract, according to projections (Bradley et al. 2009). Cheatgrass and spotted knapweed are likely to shift ranges, potentially expanding into mountain and grassland regions of Colorado and Wyoming (Bradley et al. 2010). This expansion of invasive and nonnative species could disrupt ponderosa pine ecosystem function by reducing soil moisture, increasing fire probability, reducing biodiversity and forage, displacing native species, and increasing erosion (Bradley et al. 2010).

Summary of the Regional Vulnerability of Ponderosa Pine Ecosystems__

The overall regional vulnerability ranking for the ecosystem is a combination of the vulnerability to nonclimate stressors and the vulnerability to climate stressors. For ponderosa pine ecosystems, the mean overall vulnerability ranking was moderate (fig. 7.2, table 7.1), as was the ranking by each expert reviewer. Mean vulnerability rankings were moderate for both the nonclimate stressor vulnerability and the climate vulnerability (fig. 7.2). Expert reviewer rankings ranged from the low to high category for the nonclimate stressor vulnerability and were in the moderate category for the climate vulnerability.

The expert reviewers had high overall confidence in the ranking of the overall regional vulnerability (fig. 7.3). The five experts' individual rankings were in the high or very high category (table 7.2). Mean confidence was high for the nonclimate stressor vulnerability ranking, but individual experts' confidence rankings ranged from the moderate to the very high category (fig. 7.3a). Mean confidence was also high for the climate vulnerability ranking, with consistently high confidence across all reviewers (fig. 7.3b).

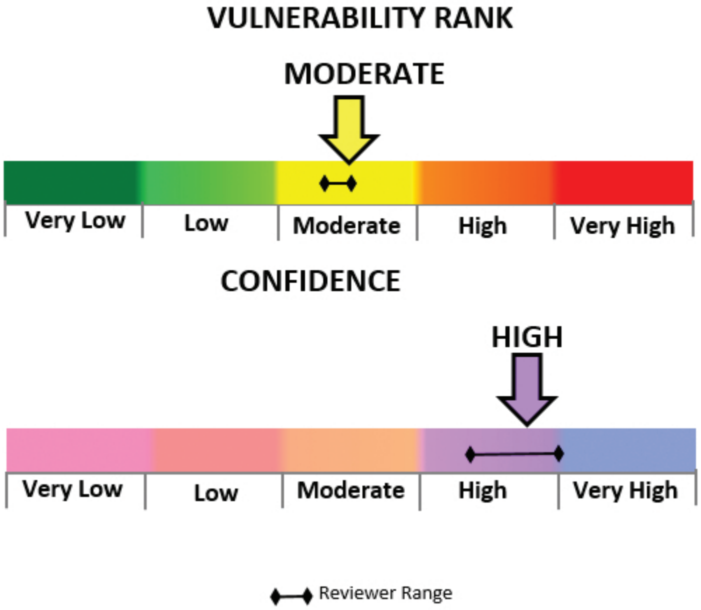

Figure 7.2—Overall regional vulnerability ranking and confidence ranking for the ponderosa pine ecosystem in the Rocky Mountain Region. The large arrow points to the mean score for vulnerability and for confidence. Bar represents the range of scores.

Table 7.1—Original and reviewer rankings for the regional vulnerability assessment of ponderosa pine ecosystems in the Rocky Mountain Region. Underline indicates a reviewer score change from the original assigned score.

Criterion	Original score	(Reviewer 1) Score	(Reviewer 2) Score	(Reviewer 3) Score	(Reviewer 4) Score	(Reviewer 5) Score	Reviewer criterion rank mean
Nonclimate							
1. Extent	3	3	3	1	3	1	Low
2. Human influences	3	3	3	3	3	3	Moderate
3. Resilience	3	3	3	1	3	3	Moderate
4. Future trends	5	5	5	5	5	5	Very high
Total	14 High	14 High	14 High	10 Low	14 High	12 Moderate	13 Moderate
Climate							
1. Ecosystem shift	3	3	1	1	3	3	Low
2. Species groups	3	3	3	1	3	3	Moderate
3. Climatic events	3	3	5	5	3	3	High
4. Adaptive capacity	3	3	3	3	3	3	Moderate
5. Hydrology	1	1	1	1	1	1	Very Low
6. Management	1	1	1	5	1	1	Low
7. Interactions	5	5	5	5	5	5	Very high
Total	19 Moderate	19 Moderate	19 Moderate	21 Moderate	19 Moderate	19 Moderate	19 Moderate
Overall vulnerability rank	17 Moderate	17 Moderate	17 Moderate	16 Moderate	17 Moderate	16 Moderate	17 Moderate

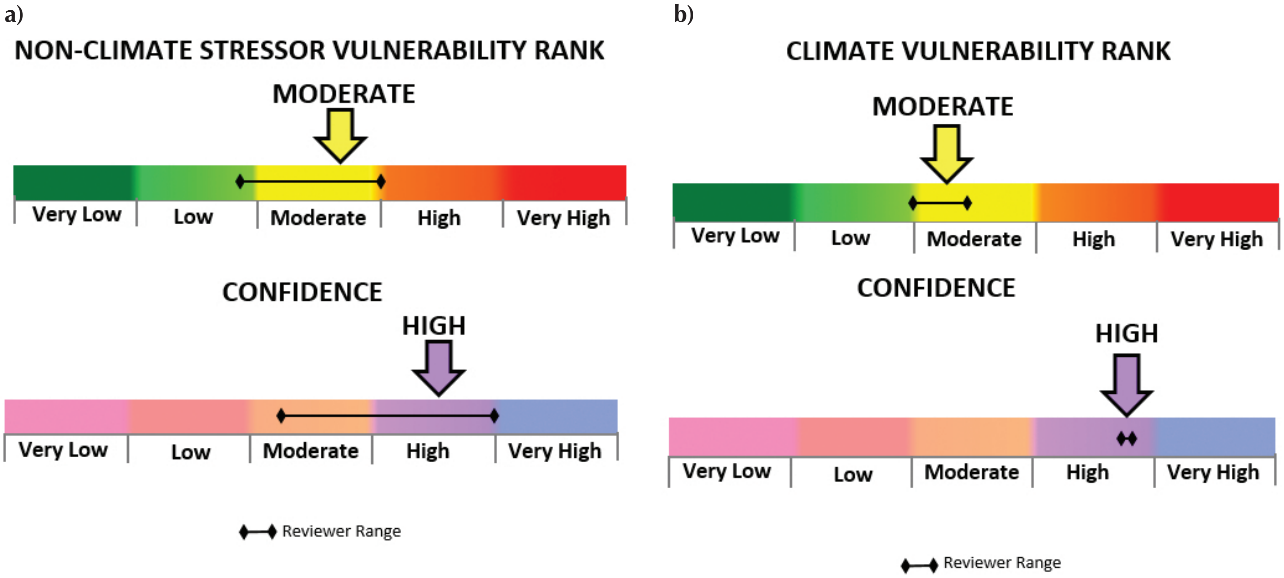

Figure 7.3—Regional vulnerability of ponderosa pine ecosystems to nonclimate (a) and to climate (b) stressors in the Rocky Mountain Region. The large arrow points to the mean score for vulnerability and for confidence. Bar represents the range of scores.

Table 7.2—Reviewer rankings for confidence in the vulnerability assessment of ponderosa pine ecosystems in the Rocky Mountain Region.

Criterion	(Reviewer 1) Score	(Reviewer 2) Score	(Reviewer 3) Score	(Reviewer 4) Score	(Reviewer 5) Score	Criterion rank confidence mean
Nonclimate						
1. Extent	5	5	5	3	5	Very high
2. Human influences	3	5	3	1	3	Moderate
3. Resilience	3	3	3	5	5	High
4. Future trends	5	5	5	3	3	High
Total	16 High	18 Very high	16 High	12 Moderate	16 High	16 High
Climate						
1. Ecosystem shift	5	1	3	3	5	Moderate
2. Species groups	3	5	5	3	3	High
3. Climatic events	3	5	5	5	3	High
4. Adaptive capacity	3	5	3	3	3	Moderate
5. Hydrology	5	5	5	5	5	Very high
6. Management	5	3	3	5	5	High
7. Interactions	5	5	5	5	5	Very high
Total	29 High	29 High	29 High	29 High	29 High	29 High
Overall confidence rank	23 High	24 Very high	23 High	21 High	23 High	23 High

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Chapter 8. Great Plains Streams and Riparian Areas: Vulnerability to Nonclimate and Climate Stressors in the U.S. Forest Service Rocky Mountain Region

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Quick Look: Great Plains Streams and Riparian Areas in the Rocky Mountain Region

Perennial and intermittent streams are important ecological features of the Great Plains landscape. Most of the Great Plains streams within the boundaries of national forests and grasslands have their headwaters on the Great Plains or east of the Rocky Mountains. They function very differently from mountain streams. Their hydrology is influenced primarily by local precipitation and the availability of groundwater; consequently, these streams tend to have highly variable streamflow. Intermittent streams are characterized by having periodic pools of standing water, remnants of streamflow, or input from springs. Great Plains intermittent and perennial streams are tributaries of larger perennial rivers on or outside of the national forests and grasslands; these rivers tend to have an east-west orientation on the Great Plains.

Plant and animal species that live in riparian areas along these streams vary with moisture availability, as well as water quality. Most of the precipitation generally falls during spring and early summer. The seasonal flooding associated with this high precipitation can create beneficial conditions by connecting habitat, enabling animal movement, and supporting reproduction. Drought hinders plants and animals by fragmenting and reducing aquatic habitat temporarily until overland flow occurs. Highly variable and extreme temperatures, as well as highly variable streamflows, create harsh conditions that are near the tolerance limits of some aquatic species. Plants and animals inhabiting these harsh, highly dynamic environments have evolved life history strategies to successfully survive and reproduce.

Quick Look: Vulnerability Assessment of Great Plains Streams and Riparian Areas in the Rocky Mountain Region

Vulnerability to nonclimate and climate stressors: Very high

Confidence: Very high

Exposure: Warming temperatures, reduced flows in streams, and potentially increased intermittent nature of streams.

Current extent: Perennial and intermittent streams and rivers on national forests and grasslands within the Great Plains area of the Rocky Mountain Region include more than 13,000 mi, with most of these streams being relatively small first-order streams.

Sensitivity and adaptive capacity to climate change: Plants and animals associated with Great Plains streams are adapted to a variable and extreme hydrological regime. However, increasing water temperatures and declining flow will fragment aquatic habitat and reduce the ability of plants and animals to move along riparian corridors. Drought and extreme temperatures increase stress on fish and aquatic insects, as well as on riparian plants and animals. Management options to mitigate the effects of climate change on aquatic systems will require a landscape-scale approach, as these Great Plains streams are interconnected. Climate change will exacerbate the stressors caused by human uses of this landscape.

Nonclimate stressors: Streams and their associated riparian areas on national forests and grasslands may occur in watersheds where a mosaic of ownerships exist; thus, agriculture, planted pasture, and urban and energy development influence the dynamics of these intermittent and perennial streams. The legacy of past land use and the current land uses have hindered aquatic life and physical processes in Great Plains streams. Invasive and nonnative species compete with native plant and aquatic animal species. Future expansion of the urban area, natural gas and petroleum drilling, and water and energy development will continue to stress this ecosystem.

Introduction

Great Plains streams are highly dynamic and important features on the plains of the Forest Service, U.S. Department of Agriculture (USFS) Rocky Mountain Region. We focus this assessment on ephemeral, intermittent, and perennial streams that flow through USFS national grasslands in the Rocky Mountain Region (fig. 8.1). We separate streams and rivers by stream order (table 8.1). Most streams on national forests and grasslands in the Great Plains are of order 1 and 2. These streams typically have their headwaters on the Great Plains (Decker 2007); thus, their hydrology is influenced primarily by local precipitation and groundwater. These streams are tributaries of larger perennial rivers on or outside of the national grasslands. These rivers tend to have an east-west orientation on the Great Plains (figs. 8.1, 8.2). Many of these rivers have headwaters at higher elevation: the Niobrara (east-central Wyoming); Purgatoire, Arkansas, and Platte (Rocky Mountains); and Cheyenne (Black Hills). The riparian ecosystems associated with these streams are included in this assessment; the streams and relevant ecosystems together are henceforth referred to collectively as "Great Plains streams."

The hydrological regimes that govern these Great Plains streams vary widely, from the relatively stable groundwater-driven flows in the Sandhills of Nebraska to the highly variable rain-driven flash floods in the intermittent and ephemeral streams on other parts of the Great Plains (Gage and Cooper 2013). Rain-driven hydrological regimes contribute to a tendency of highly variable discharge, often with intermittent flows, especially in headwater areas. Generally, Great Plains streams have highly variable and harsh conditions of temperature and highly variable discharge (Matthews 1988) with some organisms living very near lethal thermal limits (Matthews and Zimmerman 1990).

The USFS defines ephemeral, intermittent, and perennial streams as follows (USDA FS 2015):

Ephemeral stream. A stream that flows only in direct response to precipitation in the immediate locality (watershed or catchment basin), and whose channel is at all other times above the zone of saturation.

Intermittent stream. A stream or reach of stream channel that flows, in its natural condition, only during certain times of the year or in several years, and is characterized by interspersed, permanent surface water areas containing aquatic flora and fauna adapted to the relatively harsh environmental conditions found in these types of environments. Intermittent streams are identified as dashed blue lines on USGS [U.S. Geological Survey] 7 1/2-inch quadrangle maps.

Figure 8.1—Perennial rivers on USFS national forests and grasslands, in dark blue, and major tributaries, in light blue. Rivers are defined as having stream order equal to or greater than 5. (Data source: National Hydrography Dataset Plus (Horizon Systems Corporation n.d.). Note: Streams that were identified using crenulation may not be streams and these streams were removed or reduced. Intermittent streams are shown in figure 8.2.

Table 8.1—Great Plains stream and river miles by stream order on National Forests and Grasslands lands in the Rocky Mountain Region. Streams are less than 5th order; rivers that are not highly influenced by groundwater being of 5th or higher order. Data derived from NHD stream segments that intersect National Grasslands. Stream segments can extend slightly outside National Grassland boundaries and thus stream miles are overestimated by a small amount in some cases.

Stream order	Stream miles	River miles
1	8237	1*
2	2593	6*
3	1493	
4	658	
5		341
6		22
7		

*The Dismal River is a 2nd order stream, and the Middle Loup River is a 3rd order stream. Both are in Nebraska and are fed by groundwater. They are defined as Rivers in this study.

Perennial stream. A stream or reach of a channel that flows continuously or nearly so throughout the year and whose upper surface is generally lower than the top of the zone of saturation in areas adjacent to the stream. These streams are identified as solid blue on the USGS 7 1/2-inch quadrangle maps.

Most of the literature studies intermittent streams, and more often uses the term "intermittent"; however, some Great Plains publications (e.g., Friedman and Lee 2002) have focused on ephemeral streams.

Riparian cover along streams can be limited to grasses and forbs in some areas, although cottonwoods (*Populus* spp.) and shrubs occur in areas with connected floodplains, saturated soils, or nearby spring-fed streams (Decker 2007; Gage and Cooper 2013). Riparian and aquatic biota have adapted to the highly variable Great Plains hydrological regime, which receives the highest amounts of precipitation during the spring and early summer. Seasonal flooding can create beneficial conditions by connecting habitat, enabling the movement of organisms, and supporting reproduction, whereas drought hinders biota by fragmenting and reducing habitats. As a result, plants and animals inhabiting these harsh environments have evolved life history strategies to successfully survive and reproduce (Dodds et al. 2004).

Great Plains streams have been assessed for their vulnerability to climate change. As described in Chapter 1, the vulnerability framework has two major components, one related to nonclimate stressors and one related to climate stressors. Within the nonclimate component, four criteria were used to describe factors of vulnerability: current status of ecosystem extent, intrinsic resilience of the ecosystem to nonclimate stressors, human influences on the ecosystem, and future trends of nonclimate stressors on the ecosystem. Within the climate change component, seven criteria capture the vulnerability of individual species, sensitivity of ecosystem dynamics such as dependence of the ecosystem on the hydrological cycle, adaptive capacity of the ecosystem, potential for management to mitigate the effects of climate and nonclimate stressors, and interaction between climate change and nonclimate stressors. More information and the rationale for ranking ecosystem vulnerability by using these criteria are given in appendices A, B, and C.

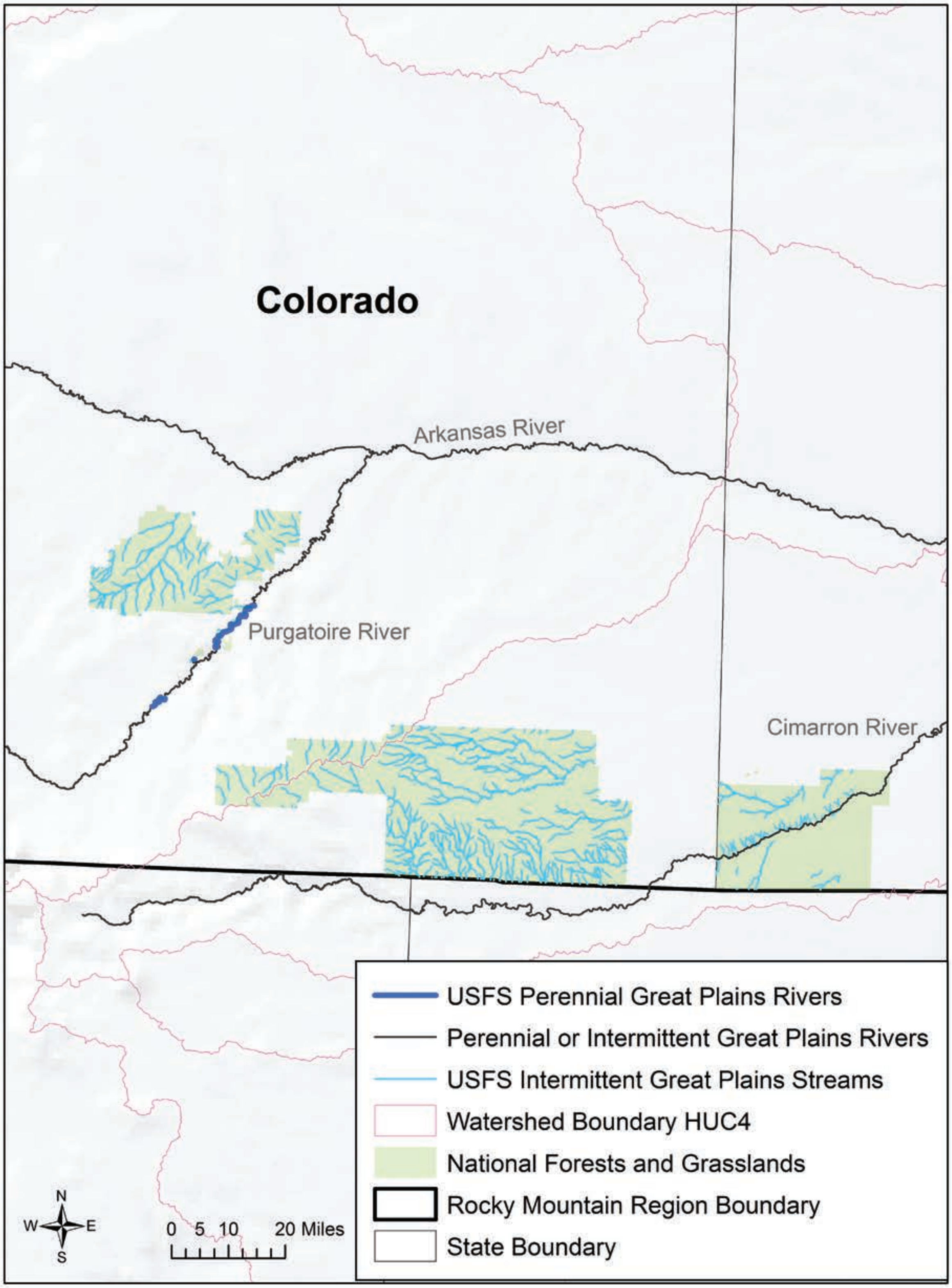

Figure 8.2—Intermittent streams (light blue) and perennial streams (dark blue) on USFS national grasslands within the watersheds of Arkansas, Purgatoire, and Cimarron Rivers (black). Watershed boundaries (pink) represent the Hydrologic Unit Code 4. Note: National forest grasslands boundaries on the Purgatoire River are small and discontinuous and are not visible on the map. (Data source: National Hydrography Dataset Plus (Horizon Systems Corporation n.d.). Streams that were identified using crenulation may not be streams and were removed or reduced. Data errors may represent perennial streams as intermittent in some cases.

In using the criteria to assess the vulnerability of Great Plains streams at the level of the Rocky Mountain Region, we drew on technical experts to evaluate our synthesis of the literature and the assigned vulnerabilities. Great Plains streams were ranked as very highly vulnerable overall; vulnerability to the nonclimate component was ranked as very high and the vulnerability to the climate component as high. The current nonclimate stresses on intermittent and perennial streams across the Great Plains have long been identified as endangering these streams (Dodds et al. 2004; Falke et al. 2011; Gido et al. 2010; Perkin et al. 2015; Worthington et al. 2014b); this assessment affirms those conclusions.

Great Plains streams are increasingly being studied on national forests and national grasslands as well as in other areas throughout the Great Plains area of the Rocky Mountain Region. We draw from this literature where studies are within the Rocky Mountain Region. Even within this Region, these streams have great variation. Application of results to individual streams on individual national grasslands will require an understanding of local site characteristics and the site characteristics of the research studies reviewed here.

Vulnerability of Great Plains Streams to Nonclimate Stressors

Summary of Key Vulnerabilities to Nonclimate Stressors

BOX 8.1

Current status of ecosystem extent

A very high vulnerability ranking is given as Great Plains streams have a limited extent in the Rocky Mountain Region.

Human influences on ecosystem

Great Plains streams are considered to have very high vulnerability to the legacy of past human influences and to the stressors associated with ongoing human influences.

- Livestock grazing, row-crop agriculture, planted pasture, urban areas, roads, dams and reservoir development, and energy development have reduced riparian areas, compacted soils, altered biotic communities, destabilized channels, introduced undesirable plants, and resulted in runoff containing fertilizers, pesticides, and herbicides.
- Water diversions and groundwater pumping have reduced or completely removed water from streams, lowered water tables, and altered natural flows.
- Invasive and nonnative aquatic and plant species have altered ecological processes in Great Plains streams.

Intrinsic resilience of ecosystem to nonclimate stressors

A moderate vulnerability ranking is given for the intrinsic resilience of Great Plains streams to nonclimate stressors.

- Factors that enhance resilience of Great Plains streams to nonclimate stressors:
  - Aquatic species have adaptive strategies to cope with short-term inhospitable conditions and tend to recover relatively fast when conditions allow.
  - Springs, seeps, and groundwater connections can lower stress associated with water withdrawals to a limited extent.
- Factors that lower resilience of Great Plains streams to nonclimate stressors:
  - Some species are not resilient to long-term water loss, fragmentation, pollution, invasive species, or land uses.

Future trends of nonclimate stressors

Great Plains streams are considered to be very highly vulnerable to future nonclimate stressors associated with human population growth, increased demand for agricultural products, water and energy development, and invasive species.

Current Status and Human Influences

Great Plains streams (ephemeral, intermittent, perennial) occur on all Great Plains national grasslands in the Rocky Mountain Region. These national grasslands are a small portion of the land area in the Great Plains and are often surrounded by a mosaic of private and other Federal or State lands.

Land uses in Great Plains watersheds include livestock grazing, row-crop agriculture, planted pasture, urban areas, roads, dams and reservoir development, and energy development (Brown and Froemke 2010; Gage and Cooper 2013; Reeves and Mitchell 2012; Smith et al. 2014). The proportion of rangeland permanently converted to other uses varies from 8 percent in Wyoming to 56 percent in Nebraska; at the county level, loss of rangeland varies from 0 to 100 percent (see figure 11 in Reeves and Mitchell 2012). Streams and their associated riparian areas on national forests and grasslands may occur in watersheds where a mosaic of ownerships exist; thus, agriculture, planted pasture, and urban and energy development may influence the dynamics of intermittent and perennial streams on these lands in national forests and grasslands.

Uplands and riparian areas are grazed by resident wildlife, such as prairie dogs (*Cynomys* spp.), antelope (*Antilocapra* spp.), and, in some areas, bison (*Bison bison*). Domestic livestock grazing (cattle, sheep, and horses) continues on national forest lands. Generally, effects of grazing, both domestic and wild, vary with intensity and timing of grazing, type of grazing animal, grazing behavior of the animal, legacy of past grazing onsite, and current grazing management systems, including the time available for some recovery. Livestock grazing can have negative effects on aquatic and riparian resources when not properly managed (Wohl 2006); these negative effects can include reductions in riparian vegetation, compacted soils, altered biotic communities, lower water quality, destabilized channels, and introduction of noxious weeds (Kauffmann and Krueger 1984). Fire and grazing in the tallgrass prairie have been important disturbances restricting the encroachment of trees under some circumstances (Veach et al. 2014).

Streams and rivers are impacted by the activities of agriculture, urban and energy development, dams and reservoir development, and roads. All of these land uses, but especially agriculture, employ herbicides, pesticides, fertilizers, and many other chemicals. In a prairie watershed dominated by intermittent streamflow in first-order higher elevation reaches, Dodds and Oakes (2006) found greater total nitrogen, nitrate, and phosphorus concentrations in tributaries occupying the lower portions of the watershed, closely mirroring the increasing density of row crop agriculture from headwaters to lower elevation areas. Increased nitrogen inputs have caused slow-moving water, stagnant water, or lakes to become oxygen-depleted, killing animal life and degrading water quality for humans as well as Great Plains stream species (Dodds et al. 2009).

Roads are sources of sediment and contaminants; they can alter drainage patterns in watersheds, increase surface runoff, and contribute to the spread of invasive plants. Road crossings can alter natural streamflow by changing stream geomorphology, function, and habitat; Bouska et al. (2010) found that transport of water, sediment, and debris during bank-full flows was more like that of natural streams when box and corrugated culvert designs were used instead of low-water crossing designs. Studying first- and second-order streams, researchers found that fish communities in stream segments isolated by perched road crossings had lower species richness (alpha diversity) relative to fish communities with connectivity to the entire stream network during summer and fall (Perkin and Gido 2012). They also found that fish communities with highly connected networks were more homogeneous than those without high connectivity. Energy development, particularly oil and gas development, across the Great Plains has increased (Reeves and Mitchell 2012), and this rapid development has raised concerns about spills and contaminants from wastewater and the associated impact on Great Plains streams and groundwater (Gallegos et al. 2015).

Water Use

Reservoirs, dams, and stream diversions, constructed to supply water for a variety of uses, regulate streamflow and flooding. Perkin et al. (2015) reported that Great Plains streams and rivers are impacted by more than 19,000 anthropogenic barriers. Flow alterations can have negative physical effects by reducing meandering loops, altering and reducing sediment transport, increasing erosion and down cutting, changing stream chemistry and nutrient transport, and narrowing channels (Merritt and Wohl 2006; Wohl and Cenderelli 2000). Flow alterations also impact plants and animals by altering composition and abundance of deepwater communities, reducing groundwater-dependent places of refuge, and modifying populations of algae, macroinvertebrates, fishes, and riparian plants (Falke et al. 2011, 2012; Friedman et al. 1998; Merritt and Wohl 2006; Perkin and Gido 2011; Strange et al. 1999; Wohl and Cenderelli 2000). Streams fragmented by water diversion structures can restrict upstream movement by fish. Altered flow patterns below dams and narrowing of river channels increase the presence of some invasive plant species (Friedman et al. 1998; Merritt and Wohl 2006). Decreased tributary flow increased salinity downstream in the Trans Pecos Region of the Rio Grande River with long-term consequences for fish assemblages (Miyazono et al. 2015). All of these process changes raise the risk of local extirpation for some aquatic species.

Across the Great Plains, the effects of water development vary; Perkin et al. (2015) found fragmentation of streams to be most affected by water development barriers in the eastern parts of the Great Plains whereas stream desiccation was more common in the western parts, where extensive groundwater extraction has occurred. Over the last 50 years, Gido et al. (2010) reported the greatest changes in fish communities across Kansas were found in basins in the western part of the State where intense groundwater withdrawals and fragmentation by reservoirs had occurred.

Invasive and Nonnative Species

Invasive plant species have expanded considerably, with 20 species affecting the Rocky Mountain Region (Pearson et al. 2011). Tamarisk (*Tamarix* spp.) and Russian olive (*Elaeagnus angustifolia*) are aggressive nonnative species that have altered ecological processes in Great Plains riparian areas (Decker 2007). Nonnative forb species are widely established in western Great Plains riparian areas and include (Anderson et al. 2006; Decker 2007):

- creeping bentgrass (*Agrostis stolonifera*),
- cheatgrass (*Bromus tectorum*),
- Canada thistle (*Cirsium arvense*),

- Mexican fireweed (*Bassia scoparia* [= *Kochia scoparia*]),
- purple loosestrife (*Lythrum salicaria*),
- sweetclover (*Melilotus* spp.),
- common dandelion (*Taraxacum officinale*), and
- yellow salsify (*Tragopogon dubius*).

Aquatic ecosystems have been invaded by:

- Asian carp (bighead carp: *Hypophthalmichthys nobilis*; silver carp: *Hypophthalmichthys molitrix*), and
- American bullfrog (*Bufo americana*).

More information on manual and mechanical treatments, biological controls, and pesticides to control invasive spread of aquatic species can be found at the National Invasive Species Center (2016).

Aquatic invasive and nonnative species have been intentionally or unintentionally introduced to many aquatic ecosystems in the Rocky Mountain Region. Nonnatives, such as largemouth bass (*Micropterus salmoides*), are placed in streams and ponds for recreational fishing and there is evidence that they can have negative effects on native species of concern (Turek et al. 2013, 2014). Invasive species have caused a decline in native populations, altered nutrient cycling, exacerbated disease spread, and disrupted food webs (Anderson et al. 2006; Lodge et al. 2000; Rahel et al. 2008; Stromberg et al. 2007).

The effects of land use, water use, and the introduction of invasive species are often interdependent and interacting. Changes in channel morphology associated with nonnative species may contribute to the success or failure of fishes as related to egg and larvae transport time in fragmented stream reaches. Another area of concern is evaluating the effects of invasive species on hydrology, especially as a function of their growth forms and evapotranspiration potentials (Hall and Rus 2013; Huddle et al. 2011).

Intrinsic Resilience of Ecosystem to Nonclimate Stressors

Factors That Enhance Resilience to Nonclimate Stressors

Great Plains streams have high physical and chemical variability, variations in flow, and wide ranges in temperature and precipitation. Species are adapted to stream conditions ranging from floods that maintain stream channels and connect habitat, to dry conditions that exacerbate stressful abiotic conditions and limit habitat (Dodds et al. 2004). Plant and animal species have adapted to these widely varying conditions and tend to recover relatively fast when conditions allow (Fritz and Dodds 2004; Matthews and Zimmerman 1990). Many species can withstand periods of poor water quality (Labbe and Fausch 2000); this ability helps increase resilience to water loss or degraded water quality caused by human land uses. Functioning riparian areas enhance aquatic species resilience as riparian plants can stabilize banks and provide organic matter to aquatic species (Dwire and Kauffmann 2003; Naiman et al. 2005). Springs, seeps, and groundwater connections in some Great Plains streams can, on a local level, mitigate drier conditions (Burk and Kennedy 2013; Dodds et al. 2004).

By building dams, beaver (*Castor canadensis*) can increase hydrological and ecological complexity: The numbers of different species increase, productivity and abundance increase, retention and distribution of water and sediment across valley bottoms increase, and floodplains can grow in complexity and size (Baker and Hill 2003; Kemp et al. 2012; Westbrook et al. 2011).

Factors That Lower Resilience to Nonclimate Stressors

Watershed function is inhibited in the Great Plains as water withdrawals exceed the historical dynamics, and transport of sediment or chemicals increases, affecting stream animals and plants (Gido et al. 2010; Perkin et al. 2014; Strange et al. 1999). Migration of aquatic species can be limited by culverts and other barriers (Perkin and Gido 2011). Stream and groundwater withdrawal for human uses alters the variation in and amount of streamflow and reduces aquatic habitat (Perkin and Gido 2011). Water management can also homogenize the flow regime; the more uniform flow along the major river corridors has led to the expansion of forested riparian corridors that facilitate species migration westward, resulting in the loss of native riparian and grassland species (Knopf 1986).

Great Plains stream species are not resilient to persistent or continuous degradations to stream habitat and water quality beyond their tolerance levels (Meador and Goldstein 2003). Tilled agricultural land that lacks adequate riparian buffers contributes increased sediment with high nutrient and pollution inputs to Great Plains streams (Dodds and Oakes 2006). Widespread grazing has increased nutrient and sediment levels, altered riparian vegetation, and, indirectly, changed temperature regimes by altering stream channel dimensions and reducing shading from riparian shrubs and trees (Belsky et al. 1999).

Great Plains stream ecosystems are not resilient to invasions by nonnative aquatic or riparian species, or diseases such as viral hemorrhagic septicemia. Invasive species may extend their range farther into Great Plains streams (Decker 2007), disrupting ecological processes and replacing native species (Dukes and Mooney 1999; Rahel 2002).

Future Trends of Nonclimate Stressors

Future population growth will increase the demand for domestic drinking water as well as water associated with land uses such as agriculture and energy. Using projections of population and climate from the 2010 Resources Planning Act Assessment, Foti et al. (2012) reported that increased population growth and potential changes in climate would increase the vulnerability of water supply in several river basins in the Great Plains. These analyses did not consider management to specifically address water use efficiency or any climate change mitigation actions. Where urbanization and other land uses have led to a modification of natural flow regimes, the associated changes in the abundance and composition of native plant and animal communities affect ecosystem services, such as water storage and nutrient cycling, that are dependent on particular species or functional groups (Strange et al. 1999). These stressors—population growth, land use development, and energy development—and the associated air and water quality challenges will continue to affect Great Plains streams.

Native fish population declines may continue, and local extirpations may occur because of competition with nonnative populations (Rahel and Olden 2008). Between 2009 and 2012, Asian carps, such as bighead carp and silver carp, invaded three prairie stream tributaries to the Missouri River in South Dakota: Big Sioux, James, and Vermillion. These carp may expand their numbers by outcompeting native species.

Population expansions of invasive plant species, such as tamarisk and cheatgrass, are likely to continue (Bradley et al. 2009). New introductions of invasive species may occur in currently uninfected areas, and these species may spread more easily because of roads, land use changes, urban areas, and agriculture (Bradley et al. 2010), as well as streamflow changes (Merritt and Poff 2010).

Vulnerability of Great Plains Streams to *Climate Stressors*

Ecosystem Capacity for Network Shift

BOX 8.2

Key Vulnerabilities

A very high vulnerability ranking is given for Great Plains stream ecosystem capacity for network shift.

- Opportunities for shifts to cooler northward latitudes are limited because of the general east-west orientation of Great Plains streams.
- Projected drying and warming may further fragment these streams, potentially making range shift responses more difficult for aquatic species.

Elevational and Latitudinal Shift Potential

The potential for elevational or latitudinal shifts for species in Great Plains streams is very limited. Most intermittent and perennial streams flow into rivers that are generally oriented east-west on the Great Plains landscape (figs. 8.1, 8.2); thus, migration to northern latitudes via stream corridors will not be possible. Fish species in these streams are not adapted to the environmental conditions of streams at higher elevations, for example, in the Rocky Mountains with steeper channel slopes (Brunger Lipsey et al. 2005). As a result, these species are unlikely to migrate to upstream habitat.

Some Great Plains riparian plants may be able to migrate to more climatically suitable upstream or upslope areas and to more northern latitudes by wind or animal dispersal (Perry et al. 2012). Downstream drift in rivers, which is the primary way in which seeds are dispersed over long distances (Merritt and Wohl 2002), would not help riparian plants migrate upslope.

Fragmentation

Increased fragmentation of Great Plains streams associated with changes in climate will limit range shift responses to climate change. Great Plains streams are fragmented because of highly variable streamflow, which fluctuates seasonally from periods of very high to low or nonexistent flow (Costigan et al. 2015; Lake 2000; Larson et al. 2013). Plants and animals have developed adaptive responses to these dynamic shifts in habitat, quickly recovering from dry periods (Falke et al. 2012; Spurgeon et al.

2014). Agriculture, roads, groundwater pumping, water diversion structures, and water withdrawal for human uses have caused more fragmentation (See Human Influences on Ecosystem section).

Large, connected areas are necessary for Great Plains watersheds to provide habitat for aquatic species (Dodds et al. 2004; Fausch et al. 2002). Flooding or bank-full runoff is an important and natural process that enhances stream connectivity and habitat complexity in Great Plains watersheds. Flooding also promotes plant and animal diversity and productivity in riparian areas (Friedman et al. 1998; Poff et al. 1997). Fish species whose eggs develop while floating downstream, including several species of minnows (family Cyprinidae), require more than 200 mi of continuous stream length to achieve population stability (Perkin and Gido 2011; Perkin et al. 2010). Eggs of pelagic broadcast spawning minnows drift in suspension during their entire development and larvae continue to drift until they become free-swimming. Worthington et al. (2014a) documented interactions between discharge, a factor in driving transport times of eggs, and higher dispersion of habitat patches, a factor in retention of eggs within the river network. At low flow velocities, eggs are likely to drop out of suspension, suggesting that low flow velocities may override habitat complexity. At high velocities, habitat complexity may help keep eggs in suspension. Worthington et al. (2014b) reported a higher probability of Arkansas River shiner (*Notropis girardi*) presence when free-flowing length was greater than 230 mi. Stream connectivity promotes greater diversity and higher population densities in fish communities (Perkin and Gido 2011).

Climate changes that reduce streamflow or reduce the amount of beneficial flooding may magnify the natural level of fragmentation common in Great Plains streams, further restricting aquatic species migration and survival (Lake 2000). Climate models project reduced streamflow and potentially greater fragmentation in the central and southern Great Plains (Perkin et al. 2010). Warmer temperatures may fragment perennial streams by creating areas of warmer water that would be unsuitable habitat for some aquatic species (Rahel et al. 1996).

Vulnerability of Foundation or Keystone Species to Climate Change

BOX 8.3

Key Vulnerabilities

A high vulnerability ranking is given for foundation and keystone species of Great Plains streams.

- Great Plains native fish have a very high vulnerability to higher temperatures, reduced streamflows, and habitat fragmentation.
- Benthic macroinvertebrates are moderately vulnerable to the expected changes in climate, as life history behaviors reduce their susceptibility to warming and drying. However, tolerance levels potentially could be exceeded.
- Foundation species of riparian plants and animals are highly vulnerable to extreme temperatures and drought, which may cause stress, hamper reproduction, or cause local extirpations.

We selected a subset of Great Plains species for assessment of vulnerability to capture a range of environmental tolerances and roles in ecosystem function. We chose native fish as foundation species; they have a substantial influence on aquatic food sources. Native or nonnative trout rarely occur in Great Plains streams, and are excluded from discussion here (see Chapter 6). We assessed benthic macroinvertebrates as keystone species. These small animals, which live in the substrates of the stream channel bed and benthos of ponds and lakes, can serve as environmental indicators and are an important food source. These animals can be insects, mayflies, stoneflies, caddisflies, beetles, midges, crane flies, and dragonflies. Other members of the benthic macroinvertebrate community are snails (class Gastropoda), clams (class Pelecypoda), aquatic worms (class Oligochaeta), and crayfish (*Cambarus* spp.). We also assessed riparian plant species, as they have properties associated with foundation species: They substantially influence ecosystem processes by cycling nutrients, dampening environmental disturbances, and regulating and filtering runoff. Animals use these riparian habitats for breeding, feeding, and shelter.

Fish

Researchers found a connection among declines and local extirpations of some native Great Plains fishes and higher temperatures, reduced streamflows, and stream fragmentation. Local extirpations occur during drier conditions, especially in streams that are not fed by groundwater (Falke et al. 2012). Researchers saw a low level of genetic diversity when streams had less water (Fitzpatrick et al. 2014). The genetic adaptation of fishes to projected warmer temperatures is not likely to keep pace with the rate of warming in the Great Plains (Matthews and Zimmerman 1990). Great Plains fish are expected to have a very high vulnerability to the warming and drying associated with climate change.

Benthic Macroinvertebrate Species

Benthic macroinvertebrate species can survive in stream habitats that may periodically have little to no flow, as well as in areas where flow is perennial. Some immature and adult benthic macroinvertebrates also burrow into the hyporheic zone under the stream bed, where adequate water and oxygen are present to sustain them when surface flow is absent. Although crayfish and many other benthic macroinvertebrates survive drought and flooding by adapting their life history behaviors and movement, Fritz and Dodds (2004) reported that recolonization from unaffected areas was a primary means of recovery in northeastern Kansas. Burk and Kennedy (2013) noted the importance of groundwater-dependent refugia for macroinvertebrate communities in the recent drought in Texas. Recovery of communities under future warming and drought will depend on the ability of species to adapt, and also the availability of habitat along the streams that species may recolonize after drying periods. Consequently, we expect many Great Plains benthic macroinvertebrates will be moderately vulnerable to the effects of climate change.

Riparian Plant Communities and Riparian-Dependent Animals

Riparian plant communities that grow in hot and sometimes dry conditions during the growing season may suffer intensified heat stress and drying related to climate change (Perry et al. 2012). Many riparian-dependent animals (such as birds, insects, reptiles, and amphibians) cannot survive in temperatures exceeding 107 to 115 °F (Perry et al. 2012), a threshold which may be exceeded with future climate warming (Kunkel et al. 2013a,b). Less water, warmer temperatures, and higher evaporation rates also increase the potential for animals (such as bats and birds) to suffer from dehydration and lower reproduction rates (Perry et al. 2012). Cottonwoods and willows (*Salix* spp.) are drought-intolerant and may also suffer considerable stress with lower or intermittent streamflows from climate change (Rood et al. 2003). We expect riparian plant and animal communities will be highly vulnerable to the warmer and drier conditions that may result from climate change.

Sensitivity to Extreme Climatic Events

BOX 8.4

Key Vulnerabilities

A high vulnerability ranking is given for the sensitivity of Great Plains streams to extreme climatic events.

- Great Plains streams are very sensitive to droughts, which can reduce aquatic habitat and limit water availability in riparian areas.
- Great Plains streams are moderately sensitive to flooding. Natural floods maintain stream habitat complexity, creating connections that enable fish to reproduce and find food resources. However, floods can restructure stream channels and riparian areas, and cause short-term disruption of plant and animal communities.
- Aquatic species in Great Plains streams are very sensitive to extreme heat that exceeds species tolerance and reduces habitat quality.

Sensitivity to Drought

Many streams on national forest lands in the Great Plains have headwaters that originate in the Great Plains. These streams may be more sensitive to drought because they are dependent on local precipitation and, possibly, connections with groundwater, in contrast to perennial streams and rivers, which receive snowmelt input to runoff. Thus, any change in precipitation will have consequences for Great Plains streams. Precipitation projections are highly variable, but with warming temperature and no change in precipitation, water demands by terrestrial plants could result in increased water stress of Great Plains streams.

Drying during summer can result in intermittent or no flows in these streams; drought can exacerbate summer drying and fragmentation of the fish habitat (Falke et al. 2011). Maintaining a base flow throughout the length of a stream may require groundwater discharge to the stream along much of its length (Winter 2007). For intermittent streams on the Konza Prairie in Kansas, Costigan et al. (2015) concluded that local water-table fluctuations and soil moisture conditions control hydrological responses more than do local weather patterns. Burk and Kennedy (2013) noted the importance of groundwater-dependent places of refuge for aquatic species. Using 12 different future climate projections, Perkin et al. (2010) reported that climate change effects on stream fragments will vary regionally; the northern Great Plains may see increases in streamflow whereas Nebraska and Kansas in the southern Great Plains may see declines. They noted the potential for increased imperilment of pelagic spawning fish because of reductions in streamflow in the central and southern Great Plains. Given the potential for the increased harm that drought can bring to Great Plains species, Great Plains streams are considered to be very sensitive to drought under climate change.

Sensitivity to Floods

Researchers observed more-intense precipitation in Kansas; in contrast, Colorado, Wyoming, South Dakota, and Nebraska did not have a statistically significant trend in precipitation intensity over the 20th century (see figure 8 in Groisman et al. 2004). Researchers project Wyoming and South Dakota will see statistically significant increases in intense precipitation by the mid-21st century (Kunkel et al. 2013a,b). Increased risk of flooding is likely in the northern Great Plains (Shafer et al. 2014), but the picture is less clear for most of the Great Plains area of the Rocky Mountain region.

Flooding of Great Plains streams is mostly driven by local storms and typically occurs in spring and early summer (Dodds et al. 2004; Matthews 1988), with detrimental or beneficial effects. Great Plains stream flooding can restructure habitats, causing increased sediment in streamflows, which may cause short-term disruption of communities; flooding also offers benefits in the long term, by maintaining stream and riparian habitat complexity (Miller et al. 2003; Poff et al. 1997). Flooding can inhibit reproduction, causing eggs to drift and larval fish to be buried under sediment and die (Durham and Wilde 2006). Flooding can increase connections and provide migrating species with opportunities to migrate off-channel, where they may find habitat with rich resources (Dodds et al. 2004; Fausch et al. 2002). Post-flooding connections also help fish reproduction, which in turn promotes greater diversity and higher population densities in fish communities (Dodd et al. 2004; Franssen et al. 2006). Post-flood conditions stimulate riparian plains cottonwood (*Populus deltoides* W. Bartram ex Marshall ssp. *monilifera* (Aiton) Eckenwalder), which establish on moist, bare, high areas when flooding subsides (Auble and Scott 1998; Friedman and Lee 2002). Great Plains streams are sensitive to flooding, but moderately so, as flooding can provide benefits.

Sensitivity to Extreme Heat

Climate warming or extreme heat events may exceed the thermal limits for species. These events may reduce growth and reproductive success of riparian plants, mammals, and birds, and in some cases result in local extirpations (Perry et al. 2012; Whitney et al. 2016). Temperature can affect riparian animals through increased heat stress, reducing midday activities, changing sex ratios, and resulting in earlier spring phenology (Perry et al. 2012).

Summer temperatures of some Great Plains streams are already close to the upper thermal limits of fishes such as minnows and darters (family Percidae) (Matthews and Zimmerman 1990). Studies of fish temperature tolerance have used the critical thermal method (CTM), where fish are acclimated to a specific temperature and then exposed to gradually rising temperatures until either loss of equilibrium or onset of muscle spasms. This endpoint temperature is the CT (critical thermal) maximum. If the fish at this point are quickly returned to their pretest acclimation temperature, survival of 100 percent is expected (Beitinger et al. 2000). This method may overestimate long-term safe

Table 8.2—Critical thermal maxima (CTmaxima) reported by Beitinger et al. 2000.

Common name (Species name)	CTmaximum
Arkansas darter (*Etheostoma cragini*)	35.0–38.4 °C (95.0–101.1 °F)
Arkansas River shiner (*Notropis girardi*)	38.6 °C (101.5 °F)
Central stoneroller (*Campostoma anomalum*)	28.8–37.2 °C (83.8–99.0 °F)
Creek chub (*Semotilus atromaculatus*)	35.7 °C (96.3 °F)
Fathead minnow (*Pimephales promelas*)	32.4–40.4 °C (90.3–104.7 °F)
Johnny darter (*Etheostoma nigrum*)	30.1–37.4 °C (86.2–99.3 °F)
Plains topminnow (*Fundulus sciadicus*)	37.0 °C (98.6 °F)
Plains minnow (*Hybognathus placitus*)	31.8–40.0 °C (89.2–104.0 °F)
Red shiner (*Cyprinella lutrensis*)	32.0–39.0 °C (89.6–102.2 °F)

temperatures; fish may suffer native effects at temperatures below these CTmaxima, especially if subjected for prolonged periods (days, weeks). We give ranges of CTmaxima, reflecting the different acclimation temperatures used in the studies (table 8.2).

Given the occurrence of high summer temperatures, fish assemblages in Great Plains streams would be negatively affected by relatively minor increases in summer water temperatures. Extreme heat events may be coupled with stream drying, reducing the opportunities for fish to move to cooler areas. Extended periods of extreme heat may also indirectly cause water quality degradation, which could exceed species tolerance levels (Murdoch et al. 2000).

High daytime maximum temperatures increase riparian plant heat stress, can reduce growth, and may also reduce germination, flowering, fruit ripening, and seed set (Perry et al. 2012). Temperatures above 113 °F damage or kill the leaf tissue in most plant species (Perry et al. 2012). The western box turtle (*Terrapene ornata* Agassiz) and mud turtles (*Kinostermon* spp.) are reptiles with temperature-dependent sex determinations, for which small increases in temperature during incubation can shift the sex ratios, with consequences to their population dynamics (Perry et al. 2012). High temperatures may also affect the ability of birds and mammals to use evaporative cooling and may increase mortality from heat stress.

Intrinsic Adaptive Capacity to Climate Change

BOX 8.5

Key Vulnerabilities

A high vulnerability ranking is given for the intrinsic adaptive capacity of Great Plains streams to climate change.

- Factors that strengthen adaptive capacity:
    - Species are adapted to the highly variable hydrological regime, and can quickly recover from disturbances such as flooding.
    - Riparian areas stabilize banks, provide organic matter to aquatic and riparian animals, and help modulate flows.
    - Some streams may receive groundwater contributions from springs and seeps, which can lessen the effects of warming and drying.
    - Beaver dams can enhance aquatic habitat.
- Factors that weaken adaptive capacity:
    - Intensified drought (warmth and drying) may exceed tolerances while reducing or degrading the habitat of Great Plains stream species.

Factors That Strengthen Adaptive Capacity to Climate Change

Great Plains stream species are adapted to the variability in seasonal hydrological patterns, where peak flow or flooding typically occurs during spring and early summer (Matthews 1988). Macroinvertebrates, fishes, birds, mammals, and reptiles have different strategies to recover from these seasonal patterns and from flooding. Recovery of microbial processes (algae) is typically within weeks, as populations are colonized by upstream or airborne propagules (Dodds et al. 2004). Macroinvertebrate species may move from places of refuge occupied during the dry period or flood. Those with slower life cycles may move 1 to 2 months after the disturbance. The diversity of benthic macroinvertebrate species, as well as population numbers, recovers from flooding twice as quickly in streams that dry out seasonally compared to perennial streams (Fritz and Dodds 2004). Riparian plants can have a relatively fast recovery period after a disturbance (Dwire and Kauffmann 2003). As they recover, they stabilize banks, provide organic matter to aquatic and riparian animal species, and maintain cool stream temperatures (Naiman et al. 2005). Riparian areas intercept sediment and retain and process nutrients, enhancing adaptive capacity for plants and animals (Naiman and Decamps 1997). Human activities can increase or decrease adaptive capacity of Great Plains streams (see Human Influences on Ecosystem section).

Great Plains fish have adapted to historical drought conditions by migrating to pools that maintain water during low flows, reproducing quickly, and withstanding periods of marginal water quality and relatively high temperatures (Dodds et al. 2004; Matthews and Zimmerman 1990). Some fish species have high genetic diversity, and are adapted to the highly variable hydrological regime (Matthews and Zimmerman 1990). For example, some Great Plains minnows, chub, and shiners spawn in open water, releasing semibuoyant eggs that disperse. This reproductive strategy is an adaptive response to the changeable streamflows that are common in the Great Plains (Durham and Wilde 2006).

Groundwater connections, springs, and seeps can be important sources of moisture during drier conditions. Spring-fed streams or groundwater, along with riparian vegetation, can help maintain suitable aquatic temperatures as air temperatures increase and warm these habitats under climate change (Burk and Kennedy 2013; Falke et al. 2012). Some Great Plains streams dominated by cottonwoods and willows can support beaver populations (Robel et al. 1993). Beavers can increase hydrological and ecological complexity by their dam building, which distributes water and sediment across valley bottoms and extends floodplains (Baker and Hill 2003; Westbrook et al. 2011). Beavers help fish populations by increasing habitat complexity, which in turn promotes species diversity and increases population sizes (Kemp et al. 2012). Beaver dams can have a negative impact on the movement of fish in some situations, such as in narrow tributary streams and under low flows (Kemp et al. 2012).

Factors That Weaken Adaptive Capacity to Climate Change

Although plant and animal species have developed traits and behaviors for surviving drought conditions common in Great Plains streams, intensified drought may exceed species tolerance levels while reducing or degrading habitat (Dodds et al. 2004; Falke et al. 2012). Researchers found that species such as minnows, which reproduce by releasing eggs in open water, did not reproduce when streamflow stopped and pools were isolated (Durham and Wilde 2006). Great Plains streams can have large channel changes during flooding (Dodds et al. 2004), which can bury and kill drifting eggs and larval fish (Durham and Wilde 2006).

Although Great Plains stream species can be highly tolerant to temperature changes, temperature limits do exist, and future climate warming may exceed species limits (see Sensitivity to Extreme Climatic Events section). Species could adapt genetically to warmer temperatures, but given the expected rapid rate of temperature increases, genetic adaption is unlikely (Matthews and Zimmerman 1990).

Dependence on Specific Hydrological Regime

BOX 8.6

Key Vulnerabilities

A high vulnerability ranking is given for Great Plains streams, as these ecosystems are dependent on a highly variable hydrological regime that may become even more variable as climate changes.

The hydrological regime for most Great Plains streams on national forests and grasslands is rain-driven. Streamflow for most streams occurs in association with precipitation events, which have high variability temporally and spatially. Flooding is flashy and challenging to predict. Streams may dry seasonally depending on connections with the groundwater table (Hansen 2001). Dodds et al. (2004) described Great Plains streams as “life on the edge”; they exist in a precarious balance between flood and drying. Species have developed adaptive strategies to tolerate these highly variable conditions: eggs that proceed through their lifecycle development while floating downstream, macroinvertebrates that survive in subsurface water in dry periods, fish that can withstand low-quality water for periods of time (Labbe and Fausch 2000; Perkin and Gido 2011; Worthington et al. 2014b).

Although aquatic species in Great Plains streams have evolved in this variable hydrological environment, there are limits or thresholds to their tolerances (Burk and Kennedy 2013; Fitzpatrick et al. 2014). Increased drought conditions could further fragment aquatic habitat, dry permanent pools, or, in extreme conditions, lower the water table such that the connection to groundwater is lost (Falke et al. 2011). Increased flooding, as well as increased drying or drought, could exceed species tolerance levels (Bouska et al. 2015). Thus, Great Plains streams are highly vulnerable to changes in hydrological regime.

Likelihood of Managing Climate Change Effects

BOX 8.7

Key Vulnerabilities

Some management options are available to lessen the impacts of climate change on Great Plains streams, but they may not always be economically, politically, socially, or ecologically feasible. Consequently, the vulnerability ranking is high.

Management options for lessening the effects of climate change on aquatic habitats include reducing current stressors, enhancing ecosystem resilience and, as climate continues to change, helping plant and animal species adapt without substantial loss of soil, soil nutrients, and plant cover (Millar et al. 2007). Managers have successfully used methods to reduce current stressors on aquatic communities and enhance resilience to climate change across the western United States (Luce et al. 2012; Rieman and Isaak 2010). These management practices include the following (Fitch et al. 2003; Palmer et al. 2009; Perkin et al. 2015; Rahel 2013, Rieman and Isaak 2010; Wohl et al. 2005):

- maintain or restore natural hydrological processes by removing stock ponds;
- maintain and restore streamside plants to reduce flooding impacts and sediment inputs;
- maintain or restore riparian, floodplain, and wetland areas, and their connection to streams;
- reintroduce beaver;
- protect or restore critical or unique habitats that buffer species survival when conditions are unfavorable;
- limit or stop the introduction of nonnative species;
- eliminate or control the input of pollutants or contaminants to surface water and groundwater;
- remove or modify barriers to the movement of native fish, and intentionally add barriers to prevent expansion of invasive species;
- maintain or reconnect large networks of habitat;

- manage domestic livestock grazing to improve condition of riparian vegetation and floodplains;
- increase riparian plant cover to provide shade and maintain stream temperature;
- allow natural changes in stream channels; and
- minimize soil compaction by limiting roads, trails, and grazing.

As conditions continue to warm and potentially dry further, it may not be possible to maintain habitat and species in their current locations. Management options may include transporting individuals to otherwise inaccessible habitat or places of refuge to maintain genetic diversity and to allow new species to colonize when native fish species can no longer survive (Rieman and Isaak 2010).

Although many management options exist to reduce stressors, they are somewhat limited and may not completely lessen the effects of severe flooding, warming, and drought. Riparian vegetation can maintain stream temperatures by providing shade, but temperature maintenance may or may not be enough to sustain suitable habitats for plants and animals. Withdrawing less water for human use can help maintain stream habitat, but can conflict with water demands from agriculture and cities, uses that are expected to increase in the future. Development of management strategies would need to incorporate new information quickly. For example, if some stream habitats where species are targeted for restoration develop unsuitably high temperatures, management strategies would have to change (Auerbach et al. 2012).

Potential for Climate Change to Exacerbate Effects of Nonclimate Stressors, or Vice Versa

BOX 8.8

Key Vulnerabilities

Climate change may exacerbate the magnitude, intensity, and effect of nonclimate stressors on Great Plains streams. Consequently, the vulnerability ranking is very high.

- Climate change may exacerbate the spread of invasive aquatic and terrestrial species that may outcompete and replace native species.
- Climate change may increase the demand for surface and subsurface water resources for human activities, with potential effects on streamflows and riparian plant communities.

Warmer Temperatures and Invasive Species

Invasive or noxious weed species grow in aquatic and riparian areas of Great Plains streams. Nonnative shrubs, forbs, and grasses are widely established in Great Plains riparian areas, where they have altered ecological processes (See Human Impacts on Ecosystem section). Carp, mollusks, and other nonnative aquatic and semiaquatic animals have invaded streams and disrupted ecosystem function. Researchers expect that many invasive plant species will expand their range as temperatures warm, displacing native species and disrupting ecological function (Bradley et al. 2009, 2010; Rahel and Olden 2008). Tamarisk and cheatgrass will expand their range, according to projections (Bradley et al. 2009). Researchers predict these invasive and nonnative plants and animals will invade more watersheds under climate change, and potentially outcompete and replace native species (Rahel and Olden 2008).

Drought, Warmer Temperatures, and Human Water Use

Great Plains streams can be sensitive to changes in groundwater dynamics. Changes in precipitation and longer periods of drought would reduce streamflows unless these systems have connections to groundwater. Groundwater pumping has lowered the water table in many places in the Great Plains in the past and continues to do so. Increased need for irrigation water or water for livestock is likely under climate change. Further groundwater pumping would exacerbate drought. Diverting Great Plains stream water for human use has various impacts: It reduces habitat connections, alters flood regimes, increases or decreases sediment transport, decreases connections with riparian floodplains, and alters aquatic and riparian food webs (see Human Influences on Ecosystem section). Increasing demand for water resources could result in less streamwater and disrupt ecological processes in Great Plains streams. There will also be increased pressure to develop reservoir storage capacity; reservoirs have many negative ecological effects on stream ecosystems (Franssen and Tobler 2013).

Summary of the Regional Vulnerability of Great Plains Streams and Riparian Areas

The overall regional vulnerability ranking for the ecosystem is a combination of the vulnerability to nonclimate stressors and the vulnerability to climate stressors. For Great Plains streams and riparian areas, the mean overall vulnerability ranking was very high with mean expert reviewer rankings in the high or very high category (fig. 8.3, table 8.3). Mean vulnerability rankings were very high for both the nonclimate stressor vulnerability and climate vulnerability. Expert reviewer rankings ranged from the high to the very high category for both vulnerabilities (fig. 8.3).

The expert reviewers had very high overall confidence in the ranking of the overall regional vulnerability (fig. 8.4). The four experts' individual rankings were in the high or very high category (table 8.4). Mean confidence was also high for the climate vulnerability ranking, but individual rankings ranged from the moderate to the very high category (fig. 8.4a). Mean confidence was very high for the nonclimate stressor vulnerability ranking, with individual confidence rankings in the high or very high category (fig. 8.4b). Calculations of mean vulnerability and confidence included only the four complete reviewer rankings.

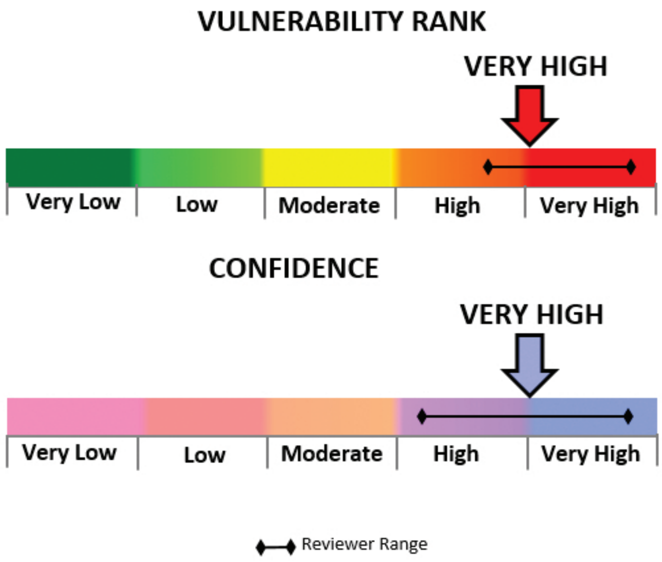

Figure 8.3—Overall regional vulnerability ranking and confidence ranking for Great Plains streams in the Rocky Mountain Region. The large arrow points to the mean score for vulnerability and for confidence. Bar represents the range of scores. Only four reviewer scores are included in mean calculations here and in tables 8.3 and 8.4.

Table 8.3—Original and reviewer rankings for the regional vulnerability assessment of Great Plains streams and riparian areas in the Rocky Mountain Region. Underline indicates a reviewer score change from the original assigned score.

Criterion	Original score	(Reviewer 1) Score	(Reviewer 2) Score	(Reviewer 3) Score	(Reviewer 4) Score *	(Reviewer 5) Score	Reviewer criterion rank mean
Nonclimate							
1. Extent	5	5	5	5	5	5	Very high
2. Human influences	5	5	5	5	5	5	Very high
3. Resilience	3	3	3	3	5	3	Moderate
4. Future trends	5	5	5	3	5	5	Very high
Total	18 Very high	18 Very high	18 Very high	16 High	20 Very high	18 Very high	18 Very high
Climate							
1. Ecosystem shift	5	5	5	5	5	5	Very high
2. Species groups	5	3	5	3	*	5	High
3. Climatic events	5	3	5	3	3	5	High
4. Adaptive capacity	3	3	3	3	3	5	High
5. Hydrology	5	3	3	5	5	5	High
6. Management	3	3	5	3	3	5	High
7. Interactions	5	5	5	5	5	5	Very high
Total	31 Very high	25 High	31 Very high	27 High	*	35 Very high	30 Very high
Overall vulnerability rank	25 Very high	22 Very high	25 Very high	22 High	*	27 Very high	24 Very high

* Calculation of mean vulnerability and confidence (table 8.4) included only the four complete reviewer rankings.

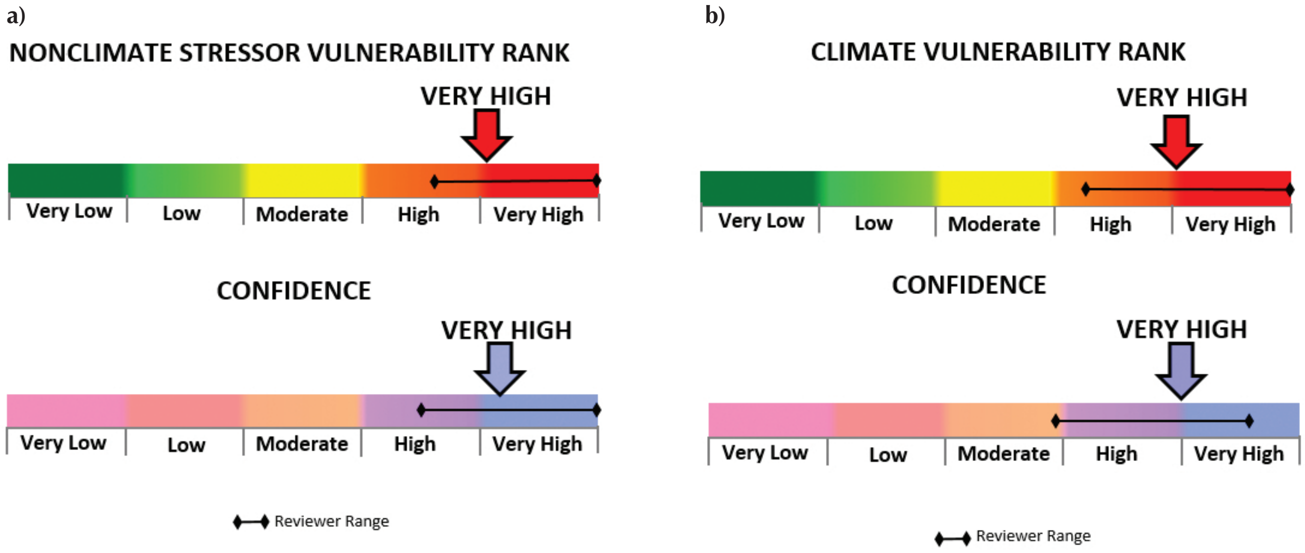

Figure 8.4—Regional vulnerability of Great Plains streams to nonclimate (a) and climate (b) stressors in the Rocky Mountain Region. The large arrow points to the mean score for vulnerability and for confidence. Bar represents the range of scores. Only four reviewer scores are included in mean calculations here and in tables 8.3 and 8.4.

Table 8.4—Reviewer rankings for confidence in the vulnerability assessment of Great Plains streams and riparian areas in the Rocky Mountain Region.

Criterion	(Reviewer 1) Score	(Reviewer 2) Score	(Reviewer 3) Score	(Reviewer 4) Score *	(Reviewer 5) Score	Criterion rank confidence mean
Nonclimate						
1. Extent	5	5	5	5	5	Very high
2. Human influences	5	5	5	5	5	Very high
3. Resilience	3	5	5	5	5	Very high
4. Future trends	3	5	3	5	5	High
Total	16 High	20 Very high	18 Very high	20 Very high	20 Very high	19 Very high
Climate						
1. Ecosystem shift	5	5	5	5	5	Very high
2. Species groups	3	5	3	*	5	High
3. Climatic events	3	5	3	5	5	High
4. Adaptive capacity	3	5	5	3	3	High
5. Hydrology	3	3	5	5	3	High
6. Management	1	5	5	1	5	High
7. Interactions	5	5	5	5	5	Very high
Total	23 Moderate	33 Very high	31 Very high	*	31 Very high	30 Very high
Overall confidence rank	20 High	27 Very high	25 Very high	*	26 Very high	25 Very high

* Calculation of mean vulnerability (table 8.3) and confidence included only the four complete reviewer rankings.

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Appendix A: Assumptions and Structure of the Rocky Mountain Climate Change Vulnerability Assessment

Ecosystem vulnerability is assessed in a theme-based approach across the Rocky Mountain Region. Factors that increase or reduce vulnerability are described in a spatially nonexplicit manner, such as along elevational gradients or areas of different disturbance regimes. Biological conditions are the primary focus of this assessment, and physical conditions, such as topography, geomorphic conditions, and soils are addressed as factors that affect biological conditions. This assessment also assumes that if a goal of management is to maintain the priority ecosystem, the identification of different components of ecosystem processes and their vulnerability can aid in adaptation planning that helps the production of ecosystem services.

This assessment determines the vulnerability to nonclimate stressors and climate change stressors separately by summarizing relevant literature for individual criteria. For example, factors for intrinsic adaptive capacity to climate are assessed separately from past human impacts and resilience to nonclimate stressors such as fire. The interactions between nonclimate and climate stressors are accounted for in one criterion. Four nonclimate criteria are used to rank the effects of nonclimate stressors on the priority ecosystem as high, moderate, or low (table A.1). Seven criteria are used to rank climate change effects on the priority ecosystem (table A.2). See Appendix B for details on the rationale for the criteria, and Appendix C for calculation of the final regional vulnerability ranking for the priority ecosystems.

Table A.1—Criteria to assess ecosystem vulnerability to nonclimate stressors.

Criterion to assess vulnerability to nonclimate stressors	Assessment
Current status of ecosystem extent	Determines the amount and distribution of the ecosystem
Human influences on ecosystem	Assesses past human activities such as land use for the degree of impacts on the ecosystem; assess stressors associated with ongoing human influences
Intrinsic resilience of ecosystem to nonclimate stressors	Assesses the degree of resilience to nonclimate stressors such as fire or anthropogenic influences
Future trends of nonclimate stressors	Assesses the likely trend of nonclimate stressors such as fire or anthropogenic influences to determine the future degree of influence on the ecosystem

Table A.2—Criteria to assess ecosystem vulnerability to climate stressors.

Criterion to assess vulnerability to climate stressors	Assessment
Capacity for network shift (aquatic ecosystems)/Capacity for range shift (terrestrial ecosystems)	Assesses availability of upstream or upslope area for a network or range shift, proximity of terrestrial ecosystem to southern range limit, and amount of fragmentation or ecosystem connectivity
Vulnerability of cold-adapted, foundation, or keystone species to climate change	Assesses the vulnerability of species that contribute to the function of the ecosystem
Sensitivity to extreme climatic events	Differentiates the sensitivity of ecosystem components to extreme climatic events such as drought, floods, extreme heat
Intrinsic adaptive capacity to climate change	Identifies factors such as biodiversity or physical characteristics that serve to strengthen or weaken adaptive capacity to climate change
Dependence on specific hydrological regime	Determines the degree of dependence on a narrow and specific hydrological regime
Likelihood of managing climate change effects	Determines the feasibility of management approaches that are available to facilitate adaptation to climate change
Potential for climate change to exacerbate effects of nonclimate stressors, or vice versa	Determines the degree of exacerbating interactions between climate and nonclimate stressors

Appendix B: Criteria and Rationale for Assessing Ecosystem Vulnerability to Nonclimate and Climate Stressors

A detailed rationale is given for each of the four criteria for nonclimate stressors (table B.1) and each of the seven criteria for climate stressors (table B.2). Each criterion was ranked as high (score = 5), moderate (score = 3), or low (score = 1). The final regional vulnerability ranking for each priority ecosystem is calculated as the mean of reviewers' numerical rankings for nonclimate and climate stressors (table B.3).

Table B.1—Ecosystem vulnerability to nonclimatic stressors: criteria and detailed rationale.

Current status of ecosystem extent	
Rationale for ranking: Ecosystems that are rarer and less widespread (e.g., alpine lakes confined to isolated mountaintops) are less likely to adapt or persist in the future when subjected to nonclimate stressors. But ecosystems that are widespread throughout an area or have a widespread and not necessarily continuous distribution throughout an area may have a higher ability to adapt.	Rank (Score)
Limited extent (<0 percent) or not widespread in the Rocky Mountain Region	High (5)
Moderate extent (10–50 percent) or moderately widespread in the Rocky Mountain Region	Moderate (3)
Widespread extent (>50 percent) or widespread in the Rocky Mountain Region	Low (1)
Human influences on ecosystem	
Rationale for ranking: Human influence and past land use can negatively or positively affect ecosystems, making them more or less susceptible to the effects of nonclimate stressors. For example, overgrazing can displace native plants or damage riparian vegetation and cause bank instability. Examples of beneficial influences are silvicultural practices that lessen negative impacts to ecosystems, or fuels treatments that reduce severe fire risk.	Rank (Score)
Human influences have had an overall negative effect on ecosystem function (restoration projects, water management, fire management, vegetation management, logging or fuels treatments, introduction or spread of invasive species, transportation, grazing, and recreation management)	High (5)
Human influences have had an overall moderately negative effect on ecosystem function (restoration projects, water management, fire management, vegetation management, logging or fuels treatments, introduction or spread of invasive species, transportation, grazing, and recreation management)	Moderate (3)
Human influences have had little effect or an overall positive effect on ecosystem function (restoration projects, water management, fire management, vegetation management, logging or fuels treatments, control of invasive species, transportation, grazing, and recreation management)	Low (1)
Intrinsic resilience of ecosystem to nonclimate stressors	
Rationale for ranking: Some ecosystems may be intrinsically more resilient to nonclimate stressors because they are adapted to a wide range of environmental conditions. This flexibility makes them better able to withstand or recover from, for example, fire disturbance or agricultural land uses such as grazing, or fragmentation from logging or road building. Ecosystems that have high connectivity and biodiversity also have a higher resilience to negative effects from nonclimate stressors.	Rank (Score)
Low ability of ecosystem to be resilient against nonclimate stressors: fire, insects and pathogens, invasive species, water use, land use (roads, urbanization, agriculture, recreation, grazing, mining)	High (5)
Moderate ability of ecosystem to be resilient against nonclimate stressors: fire, insects and pathogens, invasive species, water use, land use (roads, urbanization, agriculture, recreation, grazing, mining)	Moderate (3)
High ability of ecosystem to be resilient against nonclimate stressors: fire, insects and pathogens, invasive species, water use, land use (roads, urbanization, agriculture, recreation, grazing, mining)	Low (1)
Future trends of nonclimate stressors	
Rationale for ranking: Ecosystems that are likely to experience stabilization or a decrease in nonclimate stressor intensities in the future are likely to be able to persist better than ecosystems where such stressor trends are likely to increase. For example, extensive expansion of the wildland-urban interface, and an increased demand for water, forest products, and recreation in some areas, may increase future stresses.	Rank (Score)
Large increase in frequency or intensity (or both) of nonclimate stressors negatively affecting ecosystem: fire, invasive species, land use (inholdings, roads, agriculture, urbanization, recreation, grazing, mining, mineral extraction, water quality)	High (5)
Moderate increase in frequency or intensity (or both) of nonclimate stressors negatively affecting ecosystem: fire, invasive species, land use (inholdings, roads, agriculture, urbanization, recreation, grazing, mining, mineral extraction, water quality)	Moderate (3)
Little or no increase or lessening of frequency or intensity (or both) of nonclimate stressors negatively affecting ecosystem: fire, invasive species, land use (inholdings, roads, agriculture, urbanization, recreation, grazing, mining, mineral extraction, water quality)	Low (1)

Table B.2—Ecosystem vulnerability to climate change: Criteria and detailed rationale.

1. Capacity for network shift (aquatic ecosystems) **Capacity for range shift (terrestrial ecosystems)**	
Rationale for scoring aquatic ecosystems: Stream ecosystems that exist at high elevations are likely to be highly vulnerable to climate change (because an upslope migration is not possible). For example, many cold-adapted species in headwater streams would not be able to migrate upslope if temperatures became too warm. Upstream migration from middle elevation streams may also be constrained, but less so. Low-elevation stream migration may be least affected, as low-elevation stream ecosystems have the most potential to extend their ranges upslope. Streams that are connected are more likely to support populations and persist under a warmer climate. Stream fragmentation inhibits migration and can isolate populations. **Rationale for scoring terrestrial ecosystems:** Ecosystems that are close to the southern extremes of their biogeographic distributions in the USDA Forest Service, Rocky Mountain Region (e.g., whitebark pine) and that may be close to the southern edges of their range of climatic tolerances, may be more vulnerable to a warming climate than ecosystems that are farther north of these bioclimatic edge zones. Ecosystems closer to the northern edge of their current limit in the Rocky Mountain Region (e.g., pinyon pine) may benefit by being able to extend northward. Ecosystems that exist at high elevations (within 1,000 ft of the summits) are likely to be highly vulnerable to climate change (because they may not be able to simply migrate upslope). For example, many alpine areas in Colorado and Wyoming that exist above elevations of 10,500 to 11,500 ft may have an area of less than 3,000 ft for upslope migration. Middle elevation ecosystems may also be adversely affected, but less so, and low elevation ecosystems may be least affected, as they have the potential to extend their ranges upslope. Ecosystems that are biogeographically homogeneous are comparatively free to shift latitudinally or upslope across landscapes. This connectivity reduces vulnerability to the changing climate compared to ecosystems that are constrained by heterogeneity at a biogeographic scale. Examples of the latter might be ecosystems that are separated by mountain ranges or lowlands, large water bodies, or extensive urban areas.	Rank (Score)
Aquatic: 1) Shifts highly constrained by elevation 2) Highly constrained by fragmentation **Terrestrial:** 1) Located close to southern limit (<125 mi) 2) Located close to highest elevation (<1,000 ft) 3) Highly constrained by lack of connectivity at the biogeographic scale	High (5)
Aquatic: 1) Shifts moderately constrained by elevation 2) Moderately constrained by fragmentation **Terrestrial:** 1) Located moderately far from the southern limit of distribution (125–250 mi) 2) Located a moderate distance from highest elevation (1,000–2,000 ft) 3) Moderately constrained by lack of connectivity at the biogeographic scale	Moderate (3)
Aquatic: 1) Shifts not constrained by elevation 2) Not constrained by fragmentation **Terrestrial:** 1) Located farther from the southern limit of distribution (>250 mi) 2) Located a distance below highest elevation (>2,000 ft) 3) Not constrained by lack of connectivity at the biogeographic scale	Low (1)

2. Vulnerability of cold-adapted, keystone, or foundation species to climate change	
Rationale for ranking: Foundation species are those that have substantial influences on community structure as a consequence of high population presence. Keystone species are those that exert strong effects on the structure of their community, despite a low population presence. If there is reason to believe that either foundation or keystone species in an ecosystem are particularly vulnerable to climate change, the whole ecosystem type may be in jeopardy. The elevational range in the USFS Rocky Mountain Region is such that several ecosystems occur in colder environments and have species that are cold-adapted or require snow conditions for survival, or both. Cold-adapted species, which may or may not necessarily be a foundation or keystone species, may also be particularly vulnerable to temperature warming (e.g., ecosystem habitats that depend on snow cover in winter). Species tolerant of warmer temperatures may be less vulnerable to climate warming.	Rank (Score)
Cold-adapted, foundation, or keystone species that are likely to be particularly vulnerable to climate change	High (5)
Cold-adapted, foundation, or keystone species unlikely to be vulnerable to climate change	Low (1)
3. Sensitivity to extreme climatic events (e.g., drought, floods, extreme heat, freeze dates, windstorms)	
Rationale for ranking: Some ecosystems may be more sensitive than others to extreme climatic events. For example, prolonged droughts may eliminate wetlands, while some lakes would persist. Ice storms or windstorms may level forests. Extreme heat events may cause mortality or inhibit some biota more than others.	Rank (Score)
Highly sensitive to extreme climatic events Ecosystems that risk being greatly reduced in extent (>50-percent reduction) or eliminated entirely from an area by higher frequencies or intensities of extreme events	High (5)
Less sensitive to extreme climatic events Ecosystems that risk significant reductions in extent (20- to 50-percent reduction) due to an increased frequency or severity of extreme events, but that are unlikely to be eliminated from an area	Moderate (3)
Not sensitive to extreme climatic events Ecosystems that are unlikely to be affected to any significant extent by an increased frequency of extreme events	Low (1)
4. Intrinsic adaptive capacity to climate change	
Rationale for ranking: The diversity within which an ecosystem exists may affect its resilience and adaptive capacity. Ecosystems with diverse biotic, physical, and topographic characteristics (high biodiversity; variety in aspects, slopes, geologies and soil types, elevations) may be more able to survive climate change than ecosystems that are less varied, because the former, by existing across widely differing conditions, may be at lower risk of elimination by any future climatic conditions. Some ecosystems may be intrinsically more resistant to climate stressors because, for example, they have more rapid recovery times. Ecosystems in which the recovery period from the impacts of stressors is shorter (<20 years) may have greater intrinsic adaptive capacities than slower developing ecosystems (recovery times of >20 years). For example, riparian forests (e.g., cottonwood) may take many years to recover from a mortality event. This may render the riparian area intrinsically more vulnerable to the potential intervening effects of climate change than are ecosystems that have shorter recovery periods (e.g., grasslands or willow riparian areas).	Rank (Score)
Unlikely to be significant Climate resilience is lower, rendering it less able to withstand climate change impacts	High (5)
Moderately significant Climate resilience is moderate, enabling it to moderately withstand climate change impacts	Moderate (3)
Likely to be significant Climate resilience of ecosystem type is high, enabling it to better withstand climate change impacts.	Low (1)

5. Dependence on a specific hydrological regime	
Rationale for ranking: Some ecosystems are confined to areas with specific and relatively narrow hydrological conditions. For example, glaciated valley wetlands are confined to areas where previous glacial activity has created areas of moisture collection in valleys. Abundant winter snow cover and cold temperatures create a snowmelt- driven hydrological regime. Forested or other vegetated areas are less dependent on a specific hydrological regime and tolerate a wider variety of precipitation conditions.	Rank (Score)
Ecosystems that are dependent on specific hydrological regime	High (5)
Ecosystems less dependent on specific hydrological regime	Low (1)
6. Likelihood of managing climate change effects	
Rationale for ranking: How we are able to manage ecosystems is likely to become an important factor in conserving resources under climate change. However, some ecosystems may be less easy to manage than others. For example, managing the impacts of climate change on early seral or riverine ecosystems may be easier (e.g., through using fire, plantings, water level control) than managing ecosystems that are more intrinsically vulnerable to climate change (e.g., high elevation ecosystems). Our current understanding of adaptation or mitigation options is theoretical, and limited in application in some cases. For some ecosystem types, the management community has developed restoration techniques that may be effective management tools for adapting ecosystems to a changing climate. For example, techniques to restore the historical fire regime may facilitate adaptation to a different fire regime, provided that some understanding about potential shifts in fire regimes is available.	Rank (Score)
Not feasible No known feasible management approaches that could be employed to mitigate effects of climate change	High (5)
Moderately feasible Management approaches that could moderately mitigate the effects of changing climate already exist and have been shown to be effective	Moderate (3)
Feasible Management approaches that could mitigate effects of changing climate already exist and have been shown to be effective	Low (1)
7. Potential for climate change to exacerbate effects of nonclimate stressors, or vice versa	
Rationale for ranking: For some ecosystems and species, it is likely that significant impacts of climate change will be expressed through their exacerbating effects on current or future nonclimate stressors, or conversely, nonclimate stressors will exacerbate the effects of climate change. This indicator seeks to capture the potential effects of the interaction between climate change and nonclimate stressors. For example, warmer temperatures may increase the effects of drought, fire, or insect outbreak (e.g., warmer temperatures, greater drought, more stress to trees, greater likelihood of insect outbreak). Conversely, nonclimate stressors such as expansion of the wildland-urban interface or people's increased demand for water or recreation may exacerbate the effects of a warmer or drier climate.	Rank (Score)
Potential for large increase in stressor impacts There is a high risk (>50-percent probability) that climate change may exacerbate the impacts of nonclimate stressors, or vice versa	High (5)
Potential moderate There is a moderate risk (10- to 50-percent probability) of climate change exacerbating the effects of nonclimate stressors, or vice versa	Moderate (3)
Potential low There is only a low risk (<10-percent probability) of climate change exacerbating the effects of nonclimate stressors, or vice versa	Low (1)

Table B.3—Calculation of regional vulnerability score and confidence score for summary of vulnerability. Vulnerability is assigned to one of five categories: Very High, High, Moderate, Low, or Very Low vulnerability.

Vulnerability score:
$\frac{[\text{Nonclimate vulnerability sum} + \text{Climate vulnerability sum}]}{2}$ = Regional vulnerability
Confidence score:
$\frac{[\text{Nonclimate confidence sum} + \text{Climate confidence sum}]}{2}$ = Regional confidence

Appendix C: Vulnerability Scoring Categories

Ranges of scores and the corresponding vulnerability categories for ranking the vulnerability of the priority ecosystems are shown in table C.1.

Table C.1. Range of scores and vulnerability categories for mean reviewers' ranking of ecosystem vulnerability to nonclimate stressors, vulnerability to climate change, and overall regional vulnerability.

Nonclimate stressors	**Vulnerability category**
4–6	Very Low Vulnerability
7–10	Low Vulnerability
11–13	Moderate Vulnerability
14–17	High Vulnerability
18–20	Very High Vulnerability
Climate change	**Vulnerability category**
7–12	Very Low Vulnerability
13–18	Low Vulnerability
19–23	Moderate Vulnerability
24–29	High Vulnerability
30–35	Very High Vulnerability
Overall regional vulnerability: Mean of nonclimate stressor and climate stressor rankings	**Vulnerability category**
5–9	Very Low Vulnerability
10–14	Low Vulnerability
15–18	Moderate Vulnerability
19–23	High Vulnerability
24–28	Very High Vulnerability

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