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Colorado Front Range Climate Change Vulnerability Assessment for National Forests



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Abstract

This climate change vulnerability assessment was conducted in support of land management planning for the Pike-San Isabel (PSI) National Forests and Arapaho-Roosevelt (AR) National Forests in the Front Range region of Colorado, USA. Chapters describe climate and climate change and climate change issues relevant to management of water resources and fisheries, forests, wildlife and birds, community engagement, insects and disease, fire risk, infrastructure and flooding, ecosystem services, and recreation. A concluding chapter explores implications and potential adaptation actions for land managers. Managers face future ecosystems that are likely to depart from past and current conditions, but resource management will retain the core principle of restoring and maintaining ecosystems. This vulnerability assessment is intended to serve as a science-based foundation to inform and be incorporated into upcoming project and programmatic planning.

Keywords: adaptation, Arapaho-Roosevelt National Forests, disturbances, ecosystem services, fire, flooding, forest planning, forest resources, infrastructure, insects, Pike-San Isabel National Forests, socioeconomic, resource management, water resources, wildfire, wildlife

Cover: Mountain fen surrounded by a ponderosa pine forest near Boulder, Colorado. USDA Forest Service photo by D. Max Smith.

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Summary

The *Colorado Front Range Climate Change Vulnerability Assessment for National Forests* contains 13 chapters to inform land management planning by the Pike-San Isabel (PSI) National Forests and Arapaho-Roosevelt (AR) National Forests. This assessment first provides background information about vulnerability assessments, climate change effects on resource areas, and changes in climate specific to the Front Range. Projected climate change was quantified for a "2050" time horizon, which was defined as the change between historical (1985–2015) and future (2035–2065) time periods, encompassing four fossil fuel emissions scenarios: Very Hot and Dry (general circulation model IPSL-CM5A-MR, Representative Concentration Pathway [RCP] 8.5, or high emissions), Very Hot and Wet (CanESM2, RCP 8.5), Hot (CNRM-CM5, RCP 8.5), and Warm and Wet (GFDL-ESM2M, RCP 8.5). Climate scenarios used for this assessment projected substantial changes in the Colorado Front Range, such as snowpack decrease of 15–30 percent, more frequent severe droughts, and increases of very high and severe fire hazard conditions. Climate change effects, based on best available science, on resource areas are detailed. The conclusion summarizes the 2-day science-management workshop and scenario planning approach, identifies themes in adaptation actions apparent across the different resource areas, and applies climate change vulnerabilities to decision making processes and current priorities. The overall objective of this assessment is to provide a science-based foundation for planning on these two national forests.

Chapter 1, *Scope, Setting, and Purpose of the Colorado Front Range Climate Change Vulnerability Assessment for National Forests*, describes the Front Range setting and the Climate Change Vulnerability Assessment process.

- We defined the Colorado Front Range as encompassing the entire extent of Pike-San Isabel (PSI) National Forests and Arapaho-Roosevelt (AR) National Forests, and surrounding lands.
- Vulnerability assessments document effects of climate change and synthesize best available science for designing adaptation strategies.
- Future ecosystems are likely to be outside of familiar conditions, but land management will continue to restore and maintain resources.

Chapter 2, *Summarizing Concerns for Natural Resources Due to Climate Change and Interactions With Non-Climate Stressors in the Rocky Mountains*, supplies a broad overview of the various resources addressed in following chapters and how a changing climate interacts with those resources. Climate change poses many challenges to ecosystems, biodiversity, and ecosystem services, particularly exacerbating natural resource issues and interacting with non-climate stressors of natural resources. Climate change, including increased temperatures, variable precipitation, and extreme events, is expected to disrupt the composition of plant and animal communities and destabilize existing ecosystems.

- Climate change affects natural resources both directly and indirectly and may help or harm species or populations over time by modifying survival, growth, and reproduction.
- Ranges of plant and wildlife species are predicted to move poleward in latitude or upward in elevation at species-specific rates and trajectories, resulting in disrupted ecosystem assemblages.
- Climate change can impact resources, species, and productivity by enhancing the growth and reproduction of insects and pathogens and increasing occurrence of severe, stand-replacing fires.
- Disturbances of drought, severe wildfires, and insect outbreaks can occur in a rapid sequence of compound events, resulting in cumulative potential effects on forests outside of the effects from a single disturbance.
- Infrastructure also is vulnerable to extreme weather events and stress from changing climate through flooding, drought, wildfire, extreme heat, ice, strong winds, freeze-thaw cycles, and snowmelt.

Chapter 3, *Future Climate Scenarios for Colorado's Front Range Forests*, documents current and future climate change in the Front Range. Four divergent future climate scenarios were selected to account for the large range of potential scenarios and to quantify changes under plausible future climate scenarios: Very Hot and Dry, Very Hot and Wet, Hot, and Warm and Wet. These four scenarios included the following potential conditions for the two Colorado Front Range national forests:

- Temperature increases by 2.5 to 5.5 ^{o}F
- Precipitation changes between -10 percent and +10 percent
- Snowpack decreases in all scenarios between -15 percent and -40 percent
- Substantial spring runoff increases in all scenarios between +20 percent and +80 percent
- Summer and fall runoff decrease in all scenarios; summer decreases between -10 percent and -50 percent
- Potential evapotranspiration (a measure of drought and heat stress from a drier atmosphere) increases between 5 percent and 10 percent
- Coldest winter day warming by 3 to 10 ^{o}F
- Hottest summer day warming by 4 to 8 ^{o}F
- First fall freeze occurs later by 3 to 5 weeks

- Last spring freeze occurs earlier by 2 to 3 weeks; except for the Warm and Wet scenario where no change occurs
- Growing season increases by 3 to 8 weeks
- Freezing level (mountain snowline) shifts uphill by 1,000 to 2,000 ft
- Severe droughts, such as the droughts observed in 2002 or 2012, occur every 2 to 3 years; except for the Warm and Wet scenario where they do not occur
- "High" to "extreme" fire hazard days increase by several days and weeks, particularly in the Very Hot and Dry scenario; for the Warm and Wet they either decrease or do not change
- Extreme rainfall increases by 10 to 20 percent with higher frequency of those events in the wetter scenarios

Chapter 4, *Water Resources of the Front Range National Forests*, describes the underlying components of three key water resources (cutthroat trout habitat; riparian areas, wetlands, and groundwater-dependent ecosystems; and anthropogenic water uses). This information is incorporated into an assessment of vulnerability under four climate change scenarios. Exposure, sensitivity, and adaptive capacity are discussed, along with levels of uncertainty.

- Water resources are vulnerable to changes in volume and timing of stream flows, increases in water temperatures, decreases in groundwater recharge, and increases in disturbances such as wildfire.
- Habitat for greenback cutthroat trout is likely the most vulnerable water resource, given the small size and fragmented nature of suitable stream segments for this species.
- Managers can protect cutthroat trout habitat and other water resources by updating water storage infrastructure and using water rights to their maximum ecological benefit. Components of the natural infrastructure, such as beaver dams, aquifers, and fens, can also be managed for the protection of water resources in a warming and drying climate.

Chapter 5, *Vulnerability of Major Colorado Front Range Tree Species to Climate Change*, evaluates the vulnerability of nine major tree species in the Colorado Front Range to climate change. Detailed literature-based narratives describe the habitat and ecology of each species, explore how the species may be directly and indirectly impacted by climate change, and examine the mechanisms by which the species may be able to adapt to climate change. The narratives are then used to produce an overall vulnerability rating and an accompanying confidence rating for each species.

- The vulnerability of all nine major tree species to climate change is quite uncertain due to unknown patterns and rates of future climate change, as well as due to key research gaps and conflicting research findings.
- Generally speaking, lower-elevation tree species such as twoneedle pinyon, ponderosa pine, and Douglas-fir may be somewhat less vulnerable to climate change than higher-elevation species like limber pine, Rocky Mountain bristlecone pine, and Engelmann spruce.
- Climate change is likely to more greatly impact tree species through indirect mechanisms than direct ones. In particular, it is expected to drive significant increases in wildfire activity, which in turn is expected to cause considerable mortality of existing trees for all species and limit establishment of new trees for most species.

Chapter 6, *Future-Oriented Wildlife Management in the Pike-San Isabel and Arapaho-Roosevelt National Forests Under a Warming Climate: Anticipating Change and Facing Uncertainties With a Focus on Avian Species*, illustrates potential changes in wildlife by highlighting effects on the avian community due to climate change. Relative to other taxa, birds are predicted to be less vulnerable to climate change due to their vagility and ability to shift their range and phenology in response to environmental changes. However, these responses can have cascading impacts including novel species assemblages and changes in ecosystem function and stability. This chapter builds on existing assessments and applies future-oriented conservation planning strategies that are designed around anticipating changes, recognizing societal values, shared governance, and ongoing learning.

- The application of adaptive management approaches is not only complicated by the uncertainties in species' responses to climate change, but also differences in climate projections that predict different rates and magnitude of change.
- Climate-induced changes are predicted to impact habitat structure, the distribution of species, and the timing of their life history strategies (e.g., breeding and migration), however impacts are not fully understood and investment in learning should be ongoing.
- Most species do not exist within the confines of social-political boundaries in which management actions are implemented. Cross-boundary management is needed to protect ecosystems and associated species.
- Identification of societal values, economic constraints, and political feasibility as part of a conservation strategy is needed.

Chapter 7, *Community Involvement to Increase Monitoring Capacity for Climate-Sensitive Resources*, offers a way to increase capacity under climate change for completing desired or routine work while serving in the role of connecting local residents and visitors to past, present, and future of the forests, through additional opportunities such as volunteering, education, and scientific learning. Ecological, cultural, and historical features can provide community engagement through monitoring and recreation opportunities, education, and interpretation. High volunteer rates occur for these two national forests, which will likely increase with population growth in the Front Range.

- Community science has been in existence for decades and successful programs have capitalized on the interests of nature enthusiasts.
- Mobile computing has given rise to several platforms capable of streamlining data workflows and supporting community science initiatives that inform both local and global data compilations.
- One ongoing community science project occurring on the Arapaho-Roosevelt National Forests is part of a global monitoring effort on ungauged streams.
- Although initiating and overseeing a community science project does take time, engagement can be mutually beneficial.

Chapter 8, *Insects and Diseases and Climate Change in Colorado Front Range Forests*, synthesizes what is known about the general effects of a warming climate on pathogens and insects, followed by a general description of the most relevant insects and pathogens in the Colorado Front Range. Additionally, comments on how a warming climate may affect these organisms are offered.

- Warming temperatures and changes in precipitation patterns associated with climate change will influence the biology and ecology of forest insects and diseases.
- The literature suggests an increase in frequency and intensity of outbreaks, range expansions, and increases in population numbers.
- In the long term, effects will also be governed by how host trees respond to climate change as their distributions will change and species are expected to migrate.
- Increased stress in hosts associated with drought may make trees more susceptible to some insects and diseases. The effects of climate change on associated organisms of insects and diseases, such as natural enemies and vectors remain largely unknown.
- Some insect and diseases will exhibit positive responses such as faster growth, larger insects, range expansion and increased capability to successfully infest

their hosts while others may exhibit negative responses such as decoupling host and insect phenology and fostering maladaptive life cycles.

Chapter 9, *Fire Risk and Climate Change in the Colorado Front Range*, evaluates the role of fire in fire-adapted ecosystems and human development in the Front Range. We define wildfire risk, examine the factors contributing to each component of risk, and evaluate how each component may change in the future. We then summarize current wildfire exposure and risk using several available spatial data resources. Finally, we discuss mitigation strategies that target different components of wildfire risk.

- Climate change, insects and disease, human ignitions, and wildland-urban interface expansion coalesce to exacerbate wildfire risk on the Colorado Front Range.
- Quantifying and mapping wildfire risk in the wildland-urban interface can help managers, local leaders, and the public understand potential wildfire scenarios and take actions to avoid detrimental outcomes.
- Front Range land management agencies, organizations, communities, and individuals must work collaboratively and share the investment responsibility for preventive wildfire mitigation and planning.

Chapter 10, *Infrastructure, Flood Risk, and Sustainable Operations in Front Range National Forests*, describes and analyzes flood risk, with a brief summary of best management practices for adaptation to climate change and sustainable operations to mitigate climate change.

- Of approximately 200 camping areas, 62 were within 300 feet of 100-year flood zones. Seventeen of these recreational areas were in the Arapaho-Roosevelt National Forests and 45 were in the Pike-San Isabel National Forests.
- Installing early warning systems at strategic locations along river corridors will help broadcast flood risks.
- Stream flows that can lead to larger floods may increase in over one-third of Arapaho-Roosevelt and Pike-San Isabel National Forests streams, with greatest increases (> 25 percent change in 25-year floods) projected to occur within tributaries of large Front Range canyons and the Upper Arkansas River.
- Given that current flood risks are not well-established, future risks are more uncertain. Continued assessments are critical to establish risk from a variety of natural hazards.
- Best management practices and sustainable operations will contribute to adaptation and mitigation.

Chapter 11, *Vulnerability of Ecosystem Services Along the Colorado Front Range*, examines the socioeconomic benefits provided by ecosystems and societal inputs into resource management. The 2012 U.S. Forest Service Planning Rule further directed national forests to manage for key ecosystem services beyond multiple uses of outdoor recreation, range, timber, watershed, wildlife, and fish. The first part of this chapter describes some of the potential hazards to ecosystem services from climate change and the social forces on the Colorado Front Range that are likely to impact the condition and demand for ecosystem services. Other sections focus on specific ecosystem services, including water and watershed health, mining, forage for wildlife and livestock, and carbon sequestration. The chapter concludes with potential uses of this assessment for management.

- There is little difference between multiple-use and ecosystem services.
- Climate impacts include increased frequency of fire, flood, and drought, insects and disease, and decreased snowpack; social forces include population growth and changing demographics.
- Ecosystem services are likely to be either affected by climate change or by socioeconomic forces in the region (like large changes in the region's population). They will play an important role in climate change policy and mitigation.

Chapter 12, *Effects of Climate Change on Outdoor Recreation Benefits in the Pike-San Isabel and Arapaho-Roosevelt National Forests*, investigates, from an economic perspective, current outdoor recreation use on the Arapaho, Roosevelt, Pike, and San Isabel National Forests, and considers what future outdoor recreation use may look like under the conditions of changing climate and a growing state population.

- Recreation days for warm-weather activities on both national forests are expected to double by 2060.
- While climate change will result in some recreation activities seeing larger increases than other recreation activities, the biggest challenge is to design plan alternatives to deal with the total expected increase in visitor use.
- Options include building additional recreation infrastructure or imposing rationing mechanisms to balance the demand for recreation.
- Winter activity recreation days are expected to decrease for both national forests based on climate change scenarios.
- Recreationists may need to adjust the timing, location, and type of activity with a changing climate and environmental conditions.

- With the increased demand and limited supply of recreation opportunities, land managers may need to consider planning for substitution effects between recreation activities.

Chapter 13, *Conclusions*, summarizes the 2-day science-management workshop during 2019 and the scenario planning approach employed at this workshop; discusses themes in adaptation actions apparent across the different resource chapters in this report; and concludes with how this information on climate change vulnerabilities fits with decision making processes and current priorities.

- Several recent policies have increased expectations for national forests to consider climate change in their work, and this report helps the Arapaho-Roosevelt and Pike-San Isabel National Forests meet these expectations.
- Concepts like resistance, resilience, and transition, and processes like scenario planning, can support application of this assessment in management decisions.
- Planning teams can use this report to strengthen considerations of climate change in land management plan revision under the National Forest Management Act 2012 Planning Rule and project-level analyses under the National Environmental Policy Act.

Scope, Setting, and Purpose of the Colorado Front Range Climate Change Vulnerability Assessment for National Forests

Brice Hanberry, Thomas Timberlake, Nehalem Clark, Brian W. Miller, Courtney Peterson

Scope

The Rocky Mountain Region of the U.S. Department of Agriculture, Forest Service requested a climate change vulnerability assessment from Rocky Mountain Research Station in support of future forest plan revision for the Pike-San Isabel National Forests (PSI) and Arapaho-Roosevelt National Forests (AR) in the Rocky Mountain Region. Iterative conversations among the Rocky Mountain Research Station, the PSI and AR National Forests, and the Rocky Mountain Region identified management concerns for high-priority resources that were used to frame a scientist-manager workshop and this climate change vulnerability assessment. A workshop held during November 13–14, 2019, was structured around these priority resource areas and participants included PSI and AR specialists, Rocky Mountain Regional Office program managers, and Rocky Mountain Research Station staff. Workshop presentations included climate change scenarios, discussion of consequences of changes for priority natural and socioeconomic resource areas, and development of corresponding adaptation strategies (Timberlake et al. 2024). Results from the workshop and this climate change vulnerability assessment will help inform upcoming land management plan revisions, as well as project-level work being completed on the PSI and AR National Forests.

This climate change vulnerability assessment summarizes effects of climate change on the PSI and AR and surrounding extent of the Front Range in the southern Rocky Mountains. Here, for this introductory chapter and background for this publication, we provide context for the Front Range region and the role of vulnerability assessments in supporting climate change adaptation. The following overview chapter integrates general issues addressed in all the chapters (Hanberry 2024). Chapters encompass climate and climate change (Rangwala 2024), water resources and fisheries (Smith 2024), trees (Fornwalt et al. 2024), birds (White 2024), citizen involvement in monitoring (Fairchild and Hanberry 2024), insects and disease (Negrón 2024), fire risk (Karau et al. 2024), infrastructure and flooding (Hanberry et al. 2024), ecosystem services (Warziniack et al. 2024), recreation (Sánchez et al. 2024), and a summary of the applications for this assessment (Timberlake et al. 2024).

As subsequent chapters describe, climate change effects on hydrology, ecology, and wildland fire contribute to a complex and rapidly changing management environment. Climate change also will interact with social and environmental drivers, causing either compounding or conflicting changes to the ecosystems and communities of the Front Range (Mora et al. 2018). These interacting influences will shape how climate change impacts manifest and what adaptation activities can be implemented (Peterson et al. 2011). In some instances, long-standing and management practices that maintain or restore resources will help sustain species and ecosystems in the face of climate change (Keane et al. 2018). Other situations will require new or modified management and monitoring strategies to ensure success and improved resilience of ecosystems

and communities (Fairchild and Hanberry 2024; Keane et al. 2018). Adoption of new technologies such as low-carbon energy and energy efficiency for Forest Service infrastructure, such as rooftop solar, electric vehicles, and heat pumps help mitigate climate change (Hanberry et al. 2024).

Setting

The Front Range region is located in central Colorado, where the Great Plains, at 5,000 feet (1,550 m) in elevation, meet the Rocky Mountains, rising over 14,000 feet (4,300 m) in elevation (fig. 1). As elevation increases, ecosystems transition from grasslands and shrublands associated with the Great Plains to montane and subalpine forests, with alpine tundra occupying areas above tree line. Prevailing winds originating from the west release the majority of Colorado's precipitation on the mountains and western-facing slopes during winter months (Veblen and Donnegan 2005). Moisture from the Gulf of Mexico can produce heavy precipitation events on the eastern slope of the Continental Divide, especially during the growing season (Veblen and Donnegan 2005). Much of the State's population, and water demand for municipal purposes, resides east of the Continental Divide in an urban corridor of flat land along the Front Range of the Rocky Mountains (Smith 2024).

For the purposes of this assessment, the Colorado Front Range encompasses the entire extent of our focus area, the AR and PSI, including the area west of the Continental Divide where most of Colorado's water flows. The PSI covers approximately 2.2 million acres (890,000 ha) and the AR another 1.3 million acres (525,000 ha). The PSI and AR span key watersheds that provide water for eastern Colorado's communities and agricultural producers (Smith 2024; Warziniack et al. 2024). The PSI and AR also are highly valued for the recreational opportunities that they provide to a rapidly growing Colorado Front Range population and out-of-state and international visitors (Sánchez et al. 2024; Warziniack et al. 2024). The forests generally are adjacent to other public lands and protected areas, including the Rocky Mountain National Park, and privately owned land. Overall, dry climate and shallow, undeveloped montane soils produce low site quality and vegetation growing conditions (Addington et al. 2018). More detailed descriptions are available in land and resource management plans (Arapaho-Roosevelt National Forests and Pawnee Grasslands 1997; Pike-San Isabel National Forests and Cimarron and Comanche National Grasslands 1984).

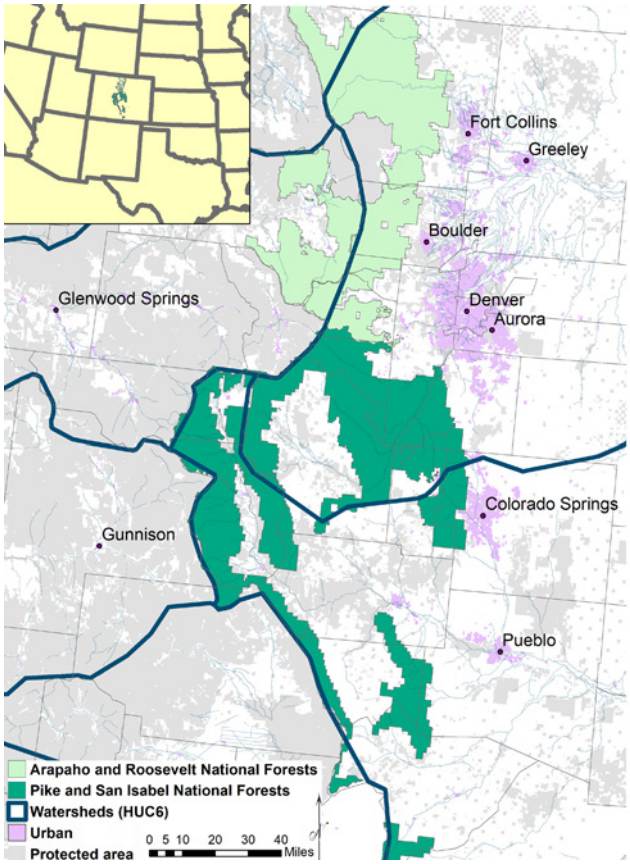


Figure 1—The setting for the Front Range region in central Colorado, including the Arapaho and Roosevelt National Forests and Pike and San Isabel National Forests; watersheds (at the scale of small river basins, HUC6); protected areas; and urban population densities.

The national forests of the Front Range have a complex history of past land-use patterns. Native Americans managed forests with fire as a tool, which controls tree densities, resulting in largely timberless landscapes with limited forests on the eastern side of the Continental Divide comprised predominantly of ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*) at lower elevations and Engelmann spruce (*Picea engelmannii*) at higher elevations, according to the first published assessment (Jack 1900). Benefits of this structure include reduced risk of extensive high-severity fires and support of biodiversity of open forest ecosystems, such as herbaceous plants and pollinators (Hanberry and Dumroese 2020; Keane et al. 2018). Euro-American colonization of the Front Range during the second half of the 19th century resulted in logging and grazing, followed by fire exclusion in the 20th century (Veblen and Donnegan 2005). Fire suppression is no longer the exclusive fire management strategy and there is growing recognition of the role of fire, including prescribed burning and management of natural ignitions by managers and communities (Karau et al. 2024; Urza et al. 2023). These forests, however, continue to face challenges associated with the increasing amount of fire along the Front Range, like elsewhere in the West,

due to the large human population and high intermix of homes within and close to national forests (Addington et al. 2018; Schultz et al. 2019).

In the drier forest types found at lower elevations in the Front Range, the product of these legacy effects is substantially denser stands than those that occurred before Euro-American influence, altering fire behavior and increasing risk of fire severity (Addington et al. 2018; Battaglia et al. 2018; Keane et al. 2002). The mountain pine beetle outbreak that occurred during the 2000s also affected the forests of Colorado (Abrams et al. 2017; Negrón 2024; Negrón and Cain 2019). Among the drivers contributing to this epidemic were greater basal area, as well as the effects of climate change, including drought that weakened trees' defenses and warmer winters that enabled rapid beetle reproduction (Abrams et al. 2017). Many areas impacted by pine beetle mortality have since burned in recent fires, and ecologists have highlighted complex interactions between beetle mortality and fire behavior and also postfire and postbeetle recovery (Jenkins et al. 2014).

During the past few decades, fires have increased in residential areas, which also have expanded to intermix with wildlands (Karau et al. 2024). In 1989, the Black Tiger Fire burned 44 homes and other structures in Boulder County; this fire was one of the first fires in the State to occur in the wildland-urban interface (Mockrin et al. 2015). In the Upper South Platte River watershed on the PSI, the Buffalo Creek Fire in 1996, the Hayman Fire in 2002, and the rainstorms following both of these events collectively highlighted the challenges that postfire erosion presents for the delivery of clean water to the downstream city of Denver (Huber-Stearns et al. 2019). Long-term ecosystem and watershed impacts from these fires are related to the vast areas that burned at high severities as well as warmer and drier climatic conditions unconducive to forest regeneration and reestablishment (Chambers et al. 2016; Rhoades et al. 2019; Stevens-Rumann et al. 2018). In June of 2012, lightning sparked the High Park Fire, which eventually would burn nearly 90,000 acres and over 250 buildings in areas on and around the AR west of Fort Collins (Miller et al. 2017). As the High Park Fire burned to the north, the Waldo Canyon Fire started a few miles outside of Colorado Springs, eventually burning into the city and destroying over 300 homes (Stratton 2012). The following year, 2013, brought record rainfall in northern Colorado, including in areas recently burned by the High Park Fire. All told, flooding in September 2013 accounted for over $2 billion of damages in Colorado (Gochis et al. 2015). The 2020 fire season further highlighted the stark challenges of managing forests in light of climate change and its effects on fire. In late summer and well into the fall, concurrent fires burned in northern Colorado on the AR, including the Cameron Peak and East Troublesome Fires. These two fires, along with the Calwood Fire in Boulder County, resulted in evacuation of residents, destroyed homes, and reduced air quality (Abbe et al. 2022). Population growth contributes to increasing development in the wildland-urban interface, which increases risks associated with fire and complicates efforts to mitigate these risks and manage fire (Hammer et al. 2009; Karau et al. 2024).

Colorado has experienced substantial population growth during recent decades, especially in urban communities along the Front Range, which is expected to continue (fig. 2; Warziniack et al. 2024). Population growth increases demand for water originating on national forests. A growing population likely will increase demand for recreational opportunities on public lands, which can lead to resource damage from overuse and can increase risks of wildfires ignited by humans (Karau et al. 2024; Sánchez et al. 2024). Land-use patterns, both historic and contemporary, also influence social vulnerability to natural disturbances and effects of climate change (Warziniack et al. 2024). The wildland-urban interface along the Front Range occurs both along the boundaries of National Forest System land and private land inholdings; many of these inholdings were historic mining claims (Calkin et al. 2014; Riebsame et al. 1996). Land use in more distant areas, including the urban corridor and agricultural lands farther east, also influence national forest management through their interactions with demands for water, recreation, air quality, and other ecosystem services offered by the Front Range national forests (Sánchez et al. 2024; Smith et al. 2024; Warziniack et al. 2024).

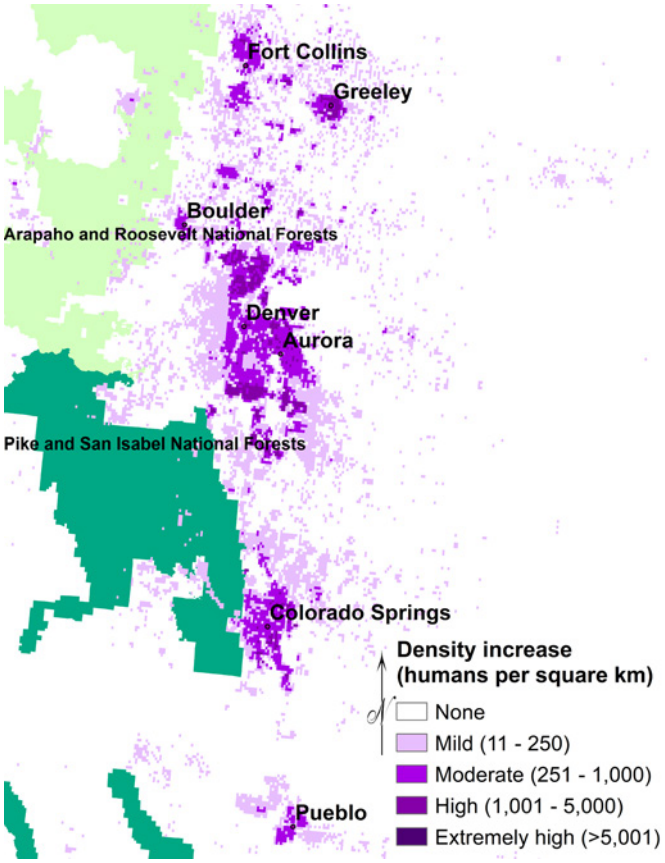


Figure 2—Location of population changes between years 2000 and 2020 (Data from WorldPop 2020).

Purpose of Climate Change Vulnerability Assessments

Climate is the long-term average of atmospheric conditions as opposed to short-term weather. Climate change is the departure of average climate conditions, such as mean temperature or average annual precipitation, that is statistically distinguishable from past climate (Keane et al. 2018). Climate change, as driven by anthropogenic greenhouse gas emissions, is already observable, and general circulation models (GCMs) are used to estimate future change (Keane et al. 2018; Schwalm et al. 2020). The GCMs project continued warming, but precipitation trends are more uncertain in magnitude and direction (increase, decrease, stable) of change (Foote 1856; IPCC 2021; Rangwala 2024). GCMs and downscaled climate data and models generate different spatial and temporal estimates, such as the number and location of extreme rainfall events (Rangwala 2024). Projected climate change also depends upon future socioeconomic changes and, thus, greenhouse gas emissions (Rangwala 2024). Uncertainty in projected climate change, similar to uncertainty in short-term weather forecasts, is not equivalent to a lack of knowledge but represents the best available science. Projections from different GCMs and emissions scenarios offer a way to evaluate and plan for multiple potential futures.

Vulnerability assessments, such as this one, document recent climate and projected climate change and help inform how changes might affect natural resources, and thus serve as the basis for designing strategies for adapting to climate change. Vulnerability, or a version of risk, is a function of exposure to change in climatic conditions, sensitivity to that change, and adaptive capacity (Glick et al. 2011). There are a variety of vulnerability assessment approaches, which can include confidence assignments based on qualitative synthesis of the literature and expert opinion, for example, ranging from high (trends seem clear, evidence supports conclusions), through medium (some evidence supports conclusions), to low (trends are unclear, evidence is lacking) confidence (Colorado Natural Heritage Program 2015).

Vulnerabilities for resources result from direct exposure to climate change or altered timing of annual events and indirect sensitivities to changed interactions or disturbances (Fornwalt et al. 2024; Smith 2024; White 2024). Increased vulnerability occurs in species at the edge of their climatic range in latitude or elevation (Fornwalt et al. 2024; Keane et al. 2018). Altered timing of cyclical biological events, such as spring bud burst, overwintering emergence, and annual migration, may result in a mismatch between needs of species and available resources; for example, food sources may not be abundant during annual migration (White 2024). Changed conditions may favor competitors, damaging insects, or diseases while reducing water, forage, cover, and breeding opportunities (Fornwalt et al. 2024; Keane et al. 2018; White 2024). In addition, climate change amplifies effects from natural disturbances of fire, insects, and other stressors such as extreme weather events and land and water use, which may increase species and ecosystem vulnerabilities to climate change (Fornwalt et al. 2024; Keane et al. 2018; Negrón 2024; Rangwala 2024; Smith 2024; White 2024). Specific sensitivities may include poor dispersal ability or barriers to dispersal, such as low

connectivity of suitable habitat (Fornwalt et al. 2024; Smith 2024; White 2024). Climate change may allow expansion of generalist and nonnative species, with traits of strong dispersal ability, rapid growth rates with short generation times, and great capacity to tolerate broad and changing environmental conditions (Glick et al. 2011; Keane et al. 2018). Species may respond to habitat changes by shifting with climate space or adapting within their current range, with some change in abundance, or by decreasing in isolated and increasingly contracting refugia, such as alpine meadows (Fornwalt et al. 2024; Hanberry and Fraser 2019; Smith 2024; White 2024). Over longer time periods, vegetation shifts may disrupt current ecosystems as species respond uniquely in distribution and abundance (fig. 3; Davis and Shaw 2001).



Figure 3—Floral resources may be reduced or not available at the correct time due to climate change (Courtesy photo by P. Hanberry).

Vulnerability assessment and adaptation workshops are useful for exploring implications of climate change to determine climate-informed management actions. During our Colorado Front Range workshop, scientists presented climate futures and participants described the potential implications for resources based on contemporary science and subject-matter expertise, focusing on management concerns in different resource disciplines (e.g., vegetation, wildlife, water) or ecosystem types. Later, managers and scientists discussed potential adaptation options to meet management goals, objectives, and planning needs under these climate-resource scenarios. In general, intentional integration of climate change considerations into management

strategies is more likely to be successful compared to assuming an unchanging climate at achieving desired outcomes and meeting management goals (Keane et al. 2018; Timberlake et al. 2024).

Climate change adaptation strategies can be integrated into management and planning, and following an adaptive management process, be updated as needed based on monitoring or advances in science and practice. Adaptation is the adjustment of systems in response to climate change and means adapting to today's changed climate conditions and predicted future conditions, rather than relying on past experience (Keane et al. 2018). Nonetheless, continuing to manage resources and mitigate non-climate stressors will support natural resources under climate change (Keane et al. 2018). Adaptation resources are available to help land managers integrate climate change information into land management planning and decisionmaking, including, for example, the Climate Change Resource Center (USDA FS 2022), Climate Change Response Framework (Swanston et al. 2016), and climate change vulnerability and adaptation assessments (for example, Halofsky et al. 2018).

Land management plan revisions are a key venue for addressing climate change, but climate change considerations are also relevant to other types of strategic planning and project-level planning and implementation (Timberlake and Schultz 2019). The 2012 Planning Rule requires planning teams to consider climate change as a system stressor and driver when developing plan components for ecological integrity and for providing multiple uses (USDA FS 2012). Climate change vulnerability assessments synthesize best available science that can directly inform the assessment phase and other phases of planning of planning and provide useful information for project planning. Managers can use information about climate vulnerability of key resources to assess how planned management actions may help or hinder resource adaptation to climate change. Vulnerability information and subsequent adaptation strategies also can help managers plan activities to better support resources, species, and ecosystems in the face of climate stress.

References

Abbe, E.; Turett, E.; Patel, H.; [et al.]. 2022. Supporting community engagement in wildfire resilience planning in Boulder. https://digital.wpi.edu/downloads/n583xz273?locale=en [Accessed 3 July 2023]

Abrams, J.B.; Huber-Stearns, H.R.; Bone, C.; [et al.]. 2017. Adaptation to landscape-scale mountain pine beetle epidemic in the era of networked governance: The enduring importance of bureaucratic institutions. Ecology and Society. 22: 22–35. https://doi.org/10.5751/ES-09717-220422

Addington, R.N.; Aplet, G.H.; Battaglia, M.A.; [et al.]. 2018. Principles and practices for the restoration of ponderosa pine and dry mixed-conifer forests of the Colorado Front Range. Gen. Tech. Rep. RMRS-GTR-373. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 121 p. https://doi.org/10.2737/RMRS-GTR-373

Arapaho and Roosevelt National Forests and Pawnee Grasslands. 1997. 1997 Revision of the Land and Resource Management Plan. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fseprd641737.pdf [Accessed 16 February 2022]

Battaglia, M.A.; Gannon, B.; Brown, P.M.; [et al.]. 2018. Changes in forest structure since 1860 in ponderosa pine dominated forests in the Colorado and Wyoming Front Range, USA. Forest Ecology and Management. 422: 147–160. https://doi.org/10.1016/j.foreco.2018.04.010

Calkin, D.E.; Cohen, J.D.; Finney, M.A.; [et al.]. 2014. How risk management can prevent future wildfire disasters in the wildland-urban interface. Proceedings of the National Academy of Sciences of the United States of America. 111: 746–751. https://doi.org/10.1073/pnas.1315088111

Chambers, M.E.; Fornwalt, P.J.; Malone, S.L.; [et al.]. 2016. Patterns of conifer regeneration following high severity wildfire in ponderosa pine-dominated forests of the Colorado Front Range. Forest Ecology and Management. 378: 57–67. https://doi.org/10.1016/j.foreco.2016.07.001

Colorado Natural Heritage Program. 2015. Climate change vulnerability assessment for Colorado Bureau of Land Management. Decker, K.; Grunau, L., Handwerk, J.; [et al.]. eds. Fort Collins, Colorado: Colorado Natural Heritage Program, Colorado State University. 565 p.

Davis, M.B.; Shaw, R.G. 2001. Range shifts and adaptive responses to Quaternary climate change. Science. 292: 673–679. https://doi.org/10.1126/science.292.5517.673

Decker, K.; Grunau, L., Handwerk, J.; [et al.]. eds. 2015. Climate change vulnerability assessment for Colorado Bureau of Land Management. Fort Collins, Colorado: Colorado Natural Heritage Program, Colorado State University. 565 p. https://www.sciencebase.gov/catalog/item/5a0b0ccee4b09af898cb6e64 [Accessed 15 April 2024]

Fairchild, M.; Hanberry, B.B. 2024. Chapter 7: Community involvement to increase monitoring capacity for climate-sensitive resources. In: Hanberry, B.B., ed. Colorado Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-GTR-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 281-292. https://doi.org/10.2737/RMRS-GTR-438

Foote, E. 1856. Circumstances affecting the heat of the sun's rays. American Journal of Science and Arts. 22: 382–383.

Fornwalt, P.J; Wion, A.P.; Schoettle, A.W. [et al.]. 2024. Chapter 5: Vulnerability of major Colorado Front Range tree species to climate change. In: Hanberry, B.B., ed. Colorado Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-GTR-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 123-244. https://doi.org/10.2737/RMRS-GTR-438

Glick, P.; Stein, B.A.; Edelson, N.A. 2011. Scanning the conservation horizon: A guide to climate change vulnerability assessment. Washington, DC: National Wildlife Federation. 168 p. https://www.nwf.org/~/media/PDFs/Global-Warming/Climate-Smart-Conservation/NWFScanningtheConservationHorizonFINAL92311.ashx

Gochis, D.; Schumacher, R.; Friedrich, K. [et al.]. 2015. The Great Colorado Flood of September 2013. BAMS. 1461–1487. https://doi.org/10.1175/BAMS-D-13-00241.1

Halofsky, J.E.; Peterson, D.L.; Dante-Wood, S.K.; [et al.], eds. 2018. Climate change vulnerability and adaptation in the Northern Rocky Mountains. Gen. Tech. Rep. RMRS-GTR-374. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. Part 1. 273 p. https://doi.org/10.2737/RMRS-GTR-374PART1

Hammer, R.B.; Stewart, S.I.; Radeloff, V.C. 2009. Demographic trends, the wildland-urban interface, and wildfire management. Society and Natural Resources. 22: 777–782. https://doi.org/10.1080/08941920802714042

Hanberry, B.B. 2024. Chapter 2: Summarizing concerns for natural resources due to climate change and interactions with non-climate stressors in the Rocky Mountains. In: Hanberry, B.B., ed. Colorado Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-GTR-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 15-38. https://doi.org/10.2737/RMRS-GTR-438

Hanberry, B.B.; Dumroese., R.K. 2020. Biodiversity and representative species in dry pine forests. In: Dumroese, R. K.; Moser, W. K., eds. Northeastern California plateaus bioregion science synthesis. Gen. Tech. Rep. RMRS-GTR-409. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 85-100. https://doi.org/10.2737/RMRS-GTR-438

Hanberry, B.B.; Fraser, J.S. 2019. Visualizing current and future climate boundaries of the conterminous United States: Implications for forests. Forests. 10(3): 280. https://doi.org/10.3390/f10030280

Hanberry, B.B; Zheng, X.; Johnson, E.; [et al.]. 2024. Chapter 10: Infrastructure, flood risk, and sustainable operations in Front Range National Forests. In: Hanberry, B.B., ed. Colorado Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-GTR-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 361-386. https://doi.org/10.2737/RMRS-GTR-438

Huber-Stearns, H.R.; Schultz, C.; Cheng, A.S. 2019. A multiple streams analysis of institutional innovation in forest watershed governance. Review of Policy Research. 36: 781–804. https://doi.org/10.1111/ropr.12359

Intergovernmental Panel on Climate Change [IPCC]. 2021: Summary for policymakers. In: Masson-Delmotte, V.; Zhai P.; A. Pirani A.; [et al.], eds. Climate change 2021: The physical science basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change Switzerland Intergovernmental Panel on Climate Change. https://www.ipcc.ch/report/ar6/wg1/downloads/report/IPCC_AR6_WGI_SPM_final.pdf [Accessed 11 February 2022]

Jack, J.G. 1900. Pikes Peak, Plum Creek, and South Platte Reserves. Twentieth annual report of the United States Geological Survey to the Secretary of the Interior, 1898–1899. Washington, DC: United States Government Printing Office. https://pubs.usgs.gov/ar/20-5/report.pdf [Accessed 16 April 2022]

Jenkins, M.J.; Runyon, J.B.; Fettig, C.J.; [et al.]. 2014. Interactions among the mountain pine beetle, fires, and fuels. Forest Science. 60: 489–501. https://doi.org/10.5849/forsci.13-017

Karau, E.C.; Hanberry, B.B.; Dillon, G.K. 2024. Chapter 9: Fire risk and climate change in the Colorado Front Range. In: Hanberry, B.B., ed. Colorado Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-GTR-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 323-360. https://doi.org/10.2737/RMRS-GTR-438

Keane, R.E.; Ryan, K.C.; Veblen, T.T; [et al.]. 2002. Cascading effects of fire exclusion in the Rocky Mountain ecosystems: A literature review. Gen. Tech. Rep. RMRS-GTR-91. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 24 p.

Keane, R.E.; Mahalovich, M.F.; Bollenbacher, B.L.; [et al.]. 2018. Effects of climate change on forest vegetation in the Northern Rockies Region. In: Halofsky, J.E.; Peterson, D.L.; Dante-Wood, S.K.; [et al.], eds. Climate change vulnerability and adaptation in the northern Rocky Mountains. Gen. Tech. Rep. RMRS-GTR-374. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. Chapter 6: 128–173.

Miller, S. 2017. Learn from the burn: The High Park Fire 5 years later. Science You Can Use Bulletin. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 25.

Mockrin, M.H.; Stewart, S.I.; Radeloff, V.C.; [et al.]. 2015. Adapting to wildfire: Rebuilding after home loss. Society and Natural Resources. 28: 839–856. https://doi.org/10.1080/08941920.2015.1014596

Mora, C.; Spirandelli, D.; Franklin, E.C.; [et al.]. 2018. Broad threat to humanity from cumulative climate hazards intensified by greenhouse gas emissions. Nature Climate Change. 8:1062–1071. https://doi.org/10.1038/s41558-018-0315-6

Negrón, J.F. 2024. Chapter 8: Insects and diseases and climate change in Colorado Front Range forests. In: Hanberry, B.B., ed. Colorado Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-GTR-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 293-322. https://doi.org/10.2737/RMRS-GTR-438

Negrón, J.F.; Cain, B. 2019. Mountain pine beetle in Colorado: A story of changing forests. Journal of Forestry. 117: 144–151. https://doi.org/10.1093/jofore/fvy032

Peterson, D.L.; Millar, C.I.; Joyce, L.A.; [et al.]. 2011. Responding to Climate change in National Forests: A guidebook for developing adaptation options. Gen. Tech. Rep. PNW-GTR-855. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 1–109. https://doi.org/10.2737/PNW-GTR-855

Pike-San Isabel National Forests and Cimarron and Comanche National Grasslands. 1984. Land and resource management plan. https://www.fs.usda.gov/detail/psicc/landmanagement/planning/?cid=fsm9_032609 [Accessed 16 February 2022]

Rangwala, I. 2024. Chapter 3: Future climate scenarios for Colorado's Front Range forests. In: Hanberry, B.B., ed. Colorado Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-GTR-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 39-56. https://doi.org/10.2737/RMRS-GTR-438

Rhoades, C.C.; Chow, A.T.; Covino, T.P.; [et al.]. 2019. The legacy of a severe wildfire on stream nitrogen and carbon in headwater catchments. Ecosystems. 22: 643–657. https://doi.org/10.1007/s10021-018-0293-6

Riebsame, W.E.; Gosnell, H.; Theobald, D.M. 1996. Land use and landscape change in Colorado Mountains I: Theory, scale, and patterns. Mountain Research and Development. 16: 395–405. https://doi.org/10.2307/3673989

Sánchez, J.J.; Huber, C.; Loomis, J. 2024. Chapter 12: Effects of climate change on outdoor recreation benefits in the Pike-San Isabel and Arapaho-Roosevelt National Forests. In: Hanberry, B.B., ed. Colorado Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-GTR-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 413-452. https://doi.org/10.2737/RMRS-GTR-438

Schultz, C.A.; McCaffrey, S.M.; Huber-Stearns, H.R. 2019. Policy barriers and opportunities for prescribed fire application in the western United States. International Journal of Wildland Fire. 28: 874–884. https://doi.org/10.1071/WF19040

Schwalm, C.R.; Glendon, S.; Duffy, P.B. 2020. RCP8.5 tracks cumulative CO2 emissions. Proceedings of the National Academy of Sciences of the United States of America. 117: 19656–19657. https://doi.org/10.1073/pnas.2007117117

Smith, D.M. 2024. Chapter 4: Water resources of the Front Range National Forests. In: Hanberry, B.B., ed. Colorado Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-GTR-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 57-122. https://doi.org/10.2737/RMRS-GTR-438

Stevens-Rumann, C.S.; Kemp, K.B.; Higuera, P.E.; [et al.]. 2018. Evidence for declining forest resilience to wildfires under climate change. Ecology Letters. 21: 243–252. https://doi.org/10.1111/ele.12889

Stratton, R. 2012. The Waldo Canyon Fire: Fires on the Colorado Front Range and home destruction. A report to the Pike and San Isabel National Forests. USDA Forest Service. https://www.frames.gov/documents/catalog/stratton_2012_waldo_canyon_fire.pdf [Accessed 12 November 2023]

Swanston, C.W.; Janowiak, M.K.; Brandt, L.A.; [et al.]. 2016. Forest adaptation resources: Climate change tools and approaches for land managers, 2nd ed. Gen. Tech. Rep. NRS-GTR-87-2. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 161 p. http://dx.doi.org/10.2737/NRS-GTR-87-2

Timberlake, T.J.; Schultz, C.A. 2019. Climate change vulnerability assessment for forest management: The case of the U.S. Forest Service. Forests. 10: 1–22. https://doi.org/10.3390/f10111030

Timberlake, T.J.; Shorrock, D.E.; Jaworski, D. 2024. Chapter 13: Conclusions. In: Hanberry, B.B., ed. Colorado Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-GTR-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 453-460. https://doi.org/10.2737/RMRS-GTR-438

Urza, A.K.; Hanberry B.B.; Jain, T.B. 2023. Landscape-scale fuel treatment effectiveness: Lessons learned from wildland fire case studies in forests of the western United States and Great Lakes region. Fire Ecology. 19:1. https://doi.org/10.1186/s42408-022-00159-y

USDA Forest Service [USDA FS]. 2012. National Forest System Land Management Planning. Washington, DC: U.S. Department of Agriculture, Forest Service. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5362536.pdf [Accessed 16 February 2022]

USDA Forest Service [USDA FS]. 2022. Climate change resource center. https://www.fs.usda.gov/ccrc/ [Accessed 16 February 2022]

Veblen, T.T.; Donnegan, J.A. 2005. Historical range of variability for forest vegetation of the National Forests of the Colorado Front Range. Golden, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Region. 151 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5386430.pdf

Warziniack, T.; Cox, E.; Froemke, P.; [et al.]. 2024. Chapter 11: Vulnerability of ecosystem services along the Colorado Front Range. In: Hanberry, B.B., ed. Colorado Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-GTR-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 387-412. https://doi.org/10.2737/RMRS-GTR-438

White, A.M. 2024. Chapter 6: Future-oriented wildlife management in the Pike-San Isabel and Arapaho-Roosevelt National Forests under a warming climate: Anticipating change and facing uncertainties with a focus on avian species . In: Hanberry, B.B., ed. Colorado Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-GTR-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 245-279. https://doi.org/10.2737/RMRS-GTR-438

WorldPop. 2020. Home page. School of Geography and Environmental Science, University of Southampton, UK. https://www.worldpop.org [Accessed 15 April 2024]

Summarizing Concerns for Natural Resources Due to Climate Change and Interactions With Non-Climate Stressors in the Rocky Mountains

Brice Hanberry

Overview of Climate Change Effects on Plant and Animal Species in the Rocky Mountains

Climate change presents many challenges for ecosystems, biodiversity, and ecosystem services, particularly in combination with non-climate stressors. Progressive warming, variable precipitation, and more frequent extreme weather events produce changes in ecosystem structure and function, species distribution and abundance, and productivity (IPCC 2021; Lipton et al. 2018). Climate change affects natural resources and interacts with disturbances, such as fire, and non-climate stressors, such as habitat loss, to sometimes generate synergistic effects (Coristine and Kerr 2011). However, disturbance and land-use change also may have antagonistic interactions with climate change, for example, creating lags in species response to climate change or even species shifts to warming locations (Song et al. 2021). Vulnerability of plant and animal species intensifies with exposure to a greater magnitude of climate change, posing substantial risks to ecosystems and their services (Glick et al. 2011). The ability of wildlands to continue to provide goods and services may be disrupted when ecosystems and processes operate outside of historical climate conditions (Warziniack et al. 2024).

Climate has warmed during recent decades due to radiative forcing by emissions of radiatively-active greenhouse gases (Foote 1856; IPCC 2021; Rangwala 2024). General circulation models unequivocally predict progressive warming (fig. 1), but projected precipitation is less certain in magnitude, direction, and spatial and temporal variation (Rangwala 2024). The northern Rocky Mountains are projected to experience increased precipitation, based on agreement among most general circulation models (fig. 2; Hayhoe et al. 2018). Regardless of total precipitation, precipitation may occur more frequently in heavy precipitation events, resulting in surface runoff, flooding, erosion, and fallen trees (Hanberry et al. 2024; Rangwala 2024). Heavy rains, with water lost to overland flow, and warming trends, with greater evapotranspiration rates, will cause effectively drier conditions despite the same or even greater total precipitation, influencing resources and disturbance events such as wildfires (Rangwala 2024). Moreover, warmer temperatures will lead to more precipitation in the form of rain instead of snow, decreased snowpack, earlier snowmelt, and earlier dry conditions in soils and streams during summer, resulting in water limitations and longer fire seasons (Rangwala 2024). Nonetheless, cold-limited mountainous elevations may resist some of the impacts of warming temperatures, whereas water limitations that reduce biomass production may slow fuel accumulation and potentially reduce fire frequency or severity (Addington et al. 2018).

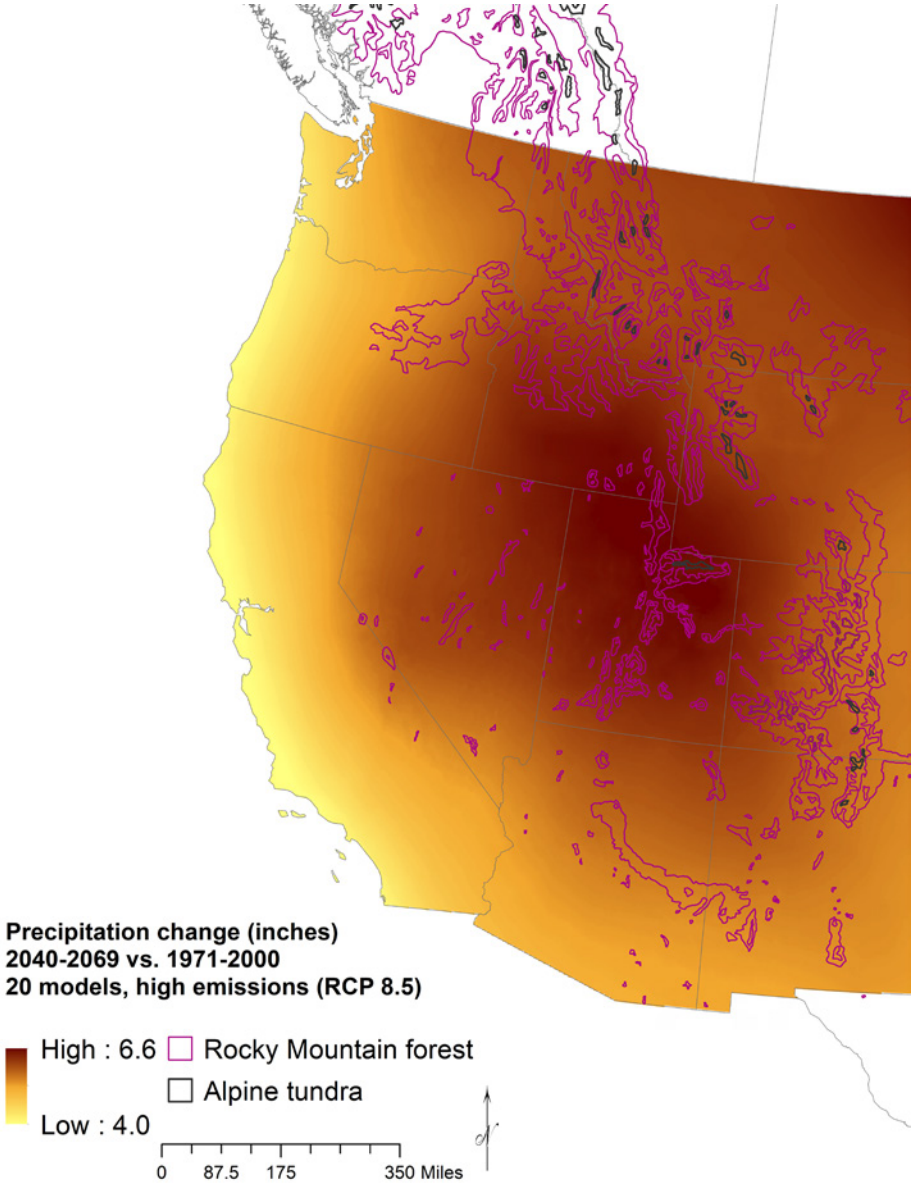


Figure 1—Projected change in mean annual temperature (°F) during 2040–2069 as compared to the baseline average during 1971-2000 for an ensemble of 20 general circulation models under the fossil-fueled representative concentration pathway (RCP8.5; Abatzoglou 2013; modified from Climate Toolbox 2022).

Climate change affects natural resources both directly and indirectly and may help or hurt species or populations over time by modifying survival, growth, and reproduction (Fornwalt et al. 2024; Smith 2024; White 2024). Direct impacts include stress or mortality from heat, drought, flooding, and cold, due to less insulative snow cover, and mistimed migration, reproduction, and hibernation activities (White 2024). Warming likely will affect growing season length, dormancy, and productivity (Fornwalt et al. 2024; Rangwala 2024). Longer growing seasons may allow for greater overall growth rates, especially for cold-limited species, creating compensation for metabolically stressful

periods (Fornwalt et al. 2024; Smith 2024; White 2024). A warmer climate generally may increase growth of species in energy-limited high-elevation or high-latitude forests with short growing seasons, until exceeding species heat tolerances, and conversely, may decrease growth, particularly of trees, in water-limited forests at low elevations in the western United States (figs. 1 and 2; Fornwalt et al. 2024; Smith 2024; White 2024).

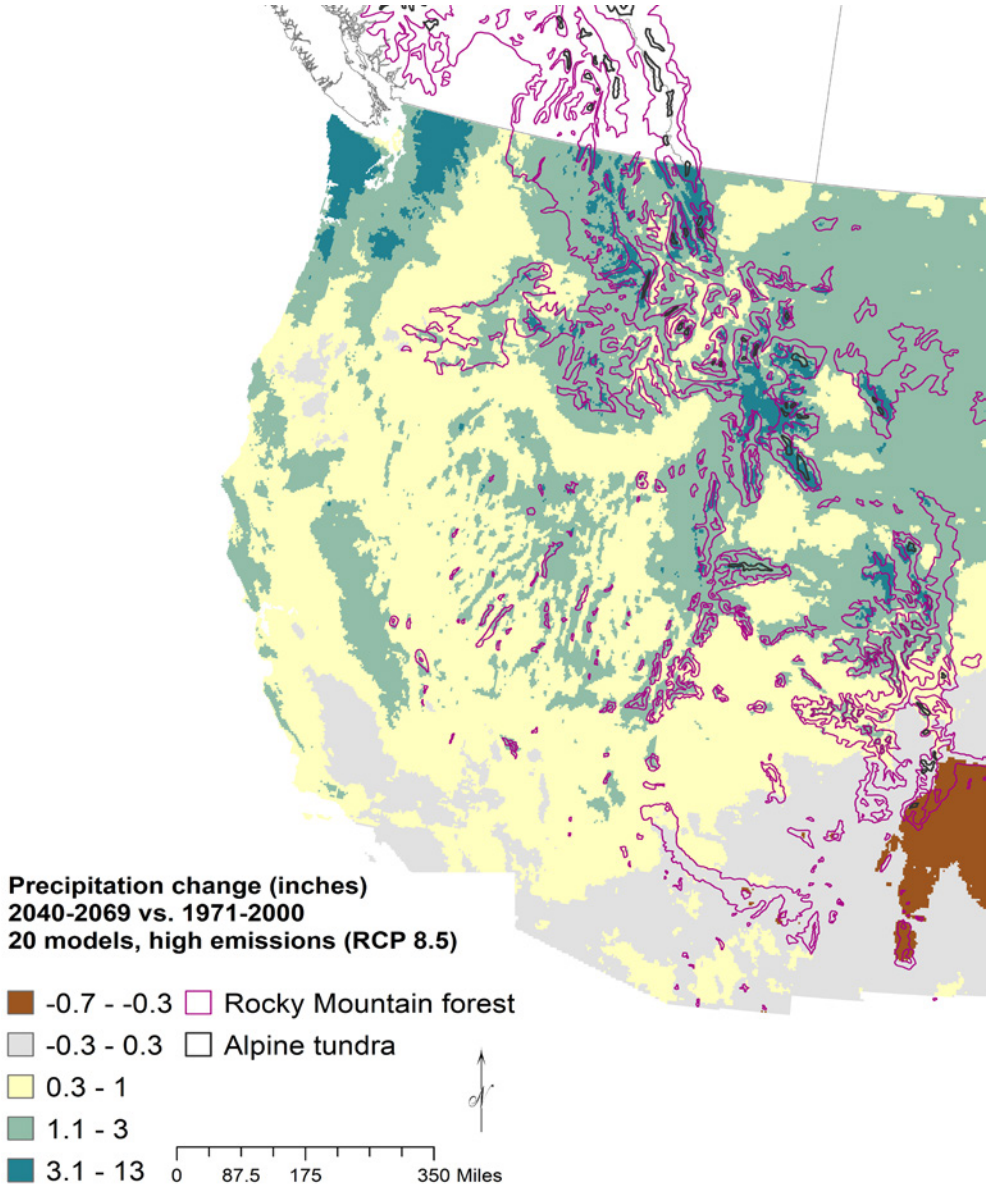


Figure 2—Projected change in mean annual total precipitation change (inches) during 2040–2069 as compared to the baseline mean annual average during 1971–2000 for an ensemble of 20 general circulation models under the fossil-fueled representative concentration pathway (RCP 8.5; Abatzoglou 2013; modified from Climate Toolbox 2022).

Climate change may have a greater influence on plant and animal species through influences on resources and biological interactions rather than direct physiological stress, excluding mortal thresholds (Keane et al. 2018). Extreme weather events such as flooding and drought can disrupt foraging and nesting resources by altering the quantity, timing, and location of resources leading to reduced survival, growth, and breeding success (Keane et al. 2022). Dietary, pollinator, or propagule dispersal specialists (i.e., dependent on a single or small number of species for food, pollination, or propagule dispersal) are more vulnerable to climate-related changes in resources than more generalist species (Glick et al. 2011). Faster-growing species that dedicate fewer resources to cold tolerance typically are able to outcompete cold-adapted species under warmer climates (Fornwalt et al. 2024; Smith 2024). Species with a greater capacity to tolerate biological stressors (such as insect outbreaks beyond endemic levels or invasive species) and increased frequency or intensity of extreme natural disturbances also may have a competitive advantage and experience relatively greater success in survival, growth, and reproduction than competitors (Decker and Fink 2015).

Species, populations, and individuals respond uniquely to climate change; therefore, climate-related changes are likely to disrupt ecosystems (Keane et al. 2018; Fornwalt et al. 2024). The ranges of plant and wildlife species are predicted to move in response to shifts in climate suitability at species-specific rates and trajectories, potentially disordering current ecosystem assemblages (Davis and Shaw 2001). Typically, species are composed of populations genetically suited to the climate of their location (Friggens et al. 2013). Populations at the southernmost or lowest elevation range edges are predicted to have greater exposure to temperature increases beyond optimal for persistence, compared to populations in the center or leading edge of the range (fig. 3a-d; Decker and Fink 2015). The degree to which individual species are able to adjust through behavioral or physiological plasticity to changing climate spaces within their occupied range is partially driven by genetic variation (Lipton et al. 2018). Some individuals and species can alter their behavior of foraging or habitat use in response to new environmental conditions (Lipton et al. 2018). Long-term strategies to adjust to climate change include shifting ranges poleward or up in elevation for montane species and selection to optimize fitness by matching genotypes with shifting climate (Friggens et al. 2013). Species with greater genetic variation and wide-ranging species that thrive under diverse conditions are likely to be less sensitive to climate change than species with limited genetic variation and narrow environmental tolerances (Decker and Fink 2015). Species with substantial dispersal ability are better able to track shifting climate envelopes than species with limited dispersal or with barriers of unsuitable habitat to dispersal (Decker and Fink 2015). Climate change may favor generalist and nonnative species with strong dispersal abilities, rapid growth rates, short generation times, and tolerance of broad and changing environmental conditions (Glick et al. 2011).

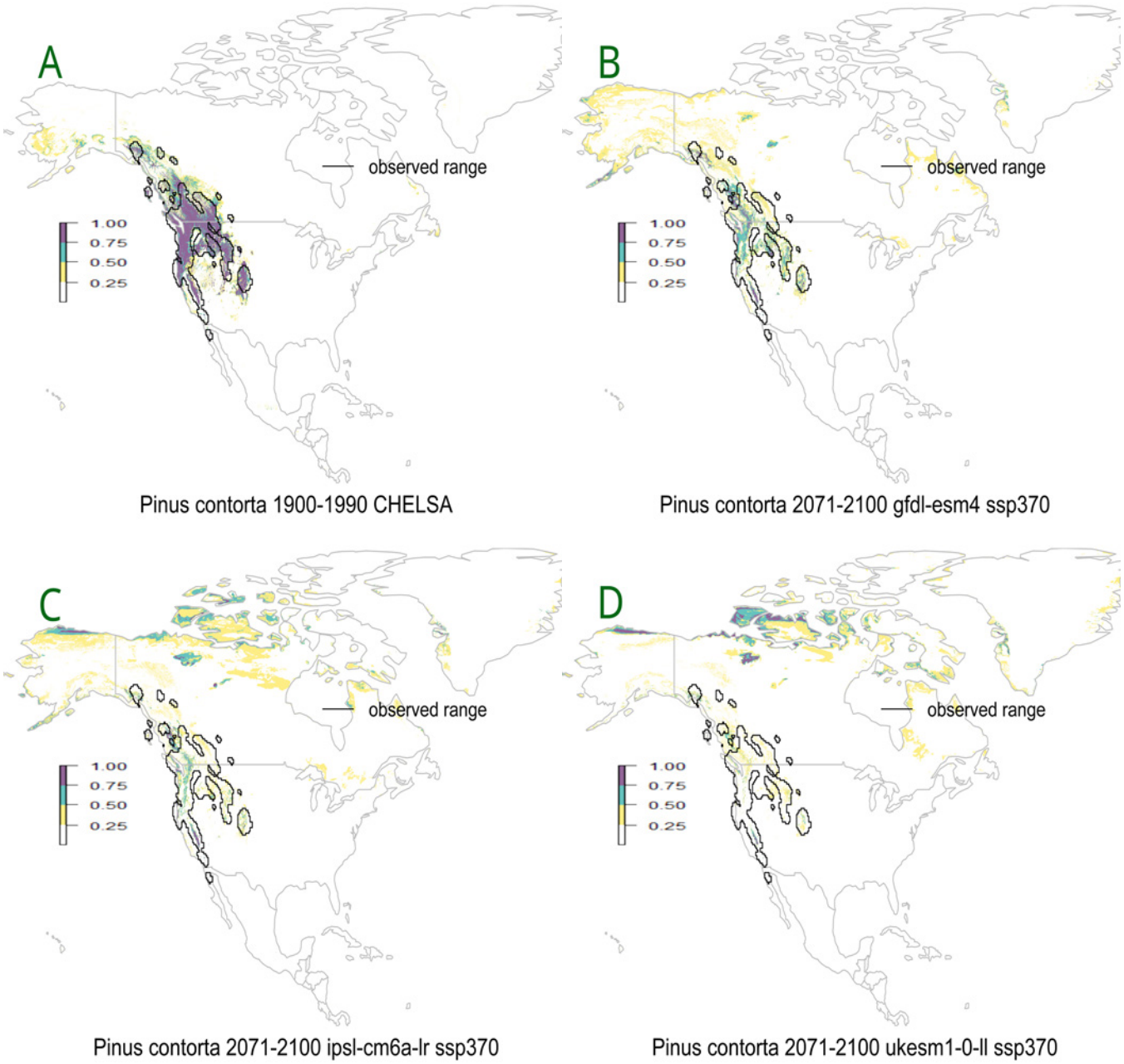


Figure 3a-d—Bioclimatic envelope, or modeled species distribution, of lodgepole pine (*Pinus contorta*) that shows observed range (black outline) from known locations and the climate envelope (temperature and precipitation of occurrences) during 1900–1990, and predicted future climate envelopes, 2071–2100, under three general circulation models and moderate emissions. Based on this model, lodgepole pine has great sensitivity to climate change.

Bioclimatic envelopes are modeled distributions of species observations based on temperature, precipitation, and other climate metrics (fig. 3, B. Hanberry 2024). However, species locations are constrained by many factors, such as disturbances, competitive interactions, and barriers to dispersal. Climate limitations are not known and indeed, climate interactions with the ecological setting and the life stage of the organism may generate a range of outcomes, from lethal to optimal for survival, growth, and reproduction (Fornwalt et al. 2024; Keane et al. 2018). Due to warmer temperatures, bioclimatic envelopes will be pushed upslope in elevation or northward in latitude (Fornwalt et al. 2024; Keane et al. 2018). In general, higher elevations have cooler temperatures and receive more precipitation and the wide range in elevations available in mountains will be to the advantage of species to readily locate to cooler conditions (Fornwalt et al. 2024; Keane et al. 2018).

Many species use seasonal cycles of climate, such as heat and chilling accumulation and precipitation patterns, as cues for blooming, reproduction, migration, or hibernation (Wolkovich and Cleland 2014). Phenology is the timing of periodic events (Lipton et al. 2018). Factors that cause or change timing and relationships among phenological events can affect species, communities, productivity, and ecosystem function (Lipton et al. 2018). Altered life cycle timing or weather patterns, such as earlier onset of bud burst and flowering, precipitation and flood events, fire seasons, emergence from overwintering, nesting, annual migration, and water availability, may cause seasonal mismatches between species and necessary food sources, or other resources, such as water, cover, and nesting opportunities (Lipton et al. 2018). Decoupling of temperature and photoperiod cues may be maladaptive for plants and animals, resulting in lessened growth, survival, and reproduction, and overall competitive disadvantage for resources (Keane et al. 2018; White 2024). Interdependencies occur among plants, insects, and birds for pollination and nesting and foraging resources (Keane et al. 2018; White 2024). Differential phenologies may disrupt the synchronicity of plant-pollinator or insect-breeding bird relationships in space or time where one partner relies more heavily on different cues; for example, temperature, day length, or snowmelt (fig. 4; Keane et al. 2018; White 2024). In addition, invasive species generally appear to be tracking climate change phenologically more closely than native species (Wolkovich and Cleland 2014).

Climate change can impact resources, species, and productivity by enhancing the growth and reproduction of insects and pathogens and increasing severe, stand-replacing fires. Recent insect outbreaks and insect-caused tree mortality may be outside of historical bounds, in that historically insects occurred in low numbers and killed weak trees with rare outbreaks (Negrón 2024; Romme et al. 2006). For example, warming has allowed mountain pine beetles (*Dendroctonus ponderosae*) to erupt at higher elevations and latitudes where winters historically were cold enough to reduce overwintering survival (Negrón 2024). A century of fire exclusion and overstory disturbance has generated dense forests with accumulated surface and canopy tree fuels (Romme et al. 2006). Warmer temperatures and more frequent drought in

higher-elevation forests with abundant fuels may increase fire frequency, while lower-elevation woodlands may become fuel-limited due to reduced production (Keane et al. 2018).



Figure 4—Climate change may alter biotic relations and processes, such as mutualisms that exist between plants and pollinators, including bees, beetles, flies, wasps, ants, butterflies, and moths (Courtesy photo by B. Hanberry).

Disturbances of drought, severe wildfires, and insect outbreaks can occur in a rapid sequence (compounding), modulating both the severity of the next disturbance and effects on species (Bigler et al. 2005; Kulakowski et al. 2013; Page et al. 2014). For example, wildfires that reduce stand basal area may aid in protecting pine from future fires, beetle outbreaks, and drought, perhaps for decades or longer by reducing stress related to competition (Bigler et al. 2005). However, effects may be complex to partition. Page et al. (2014) reported that the likelihood of severe fires increased in lodgepole pine (*Pinus contorta*) stands attacked by mountain pine beetles, though typically this effect lasted less than 10 years. In contrast, mountain pine beetles did not appear to influence 57 large (> 1,140 ha) wildfire events that burned for greater than 10 days (Hart and Preston 2020). Multiple severe disturbances, from insects, wildfire, blowdowns, and logging, do not appear to favor lodgepole pine or conifers in general relative to quaking aspen (*Populus tremuloides*); conversely, aspen is not tolerant of drought (Bretfeld et al. 2016; Kulakowski et al. 2013; Rhoades et al. 2018).

Species are often less tolerant of additional stresses when already under stress from one factor (Kolb et al. 2019; Sturrock et al. 2011; Vanbergen et al. 2013). That is, cumulative stresses are likely to have synergistic interactions, resulting in greater negative impact on species (Kolb et al. 2019; Vanbergen et al. 2013). Stress can suppress immune function in animals and stressed trees generate weaker resin flow, amplifying the negative effects of disease or damaging insect exposure in populations (Goulson et al. 2015; Kolb et al. 2019). Non-climate human activities that degrade and fragment wildlands, such as agricultural and residential conversion or infrastructure development, increase species sensitivity to climate change by reducing resources that sustain plant vigor and animal health against insect or disease infection (Vanbergen et al. 2013). Many predisposing long-term factors for disease and mortality, such as previous disturbance, fire exclusion, stand structure, tree age, and site location (elevation, aspect, site quality), influence disturbance severity by altering the capacity of trees to resist stress (Sturrock et al. 2011). Inciting factors for disease and mortality are short-term conditions such as drought that weakens trees, particularly in high-density stands (Sturrock et al. 2011). Contributing factors for disease and mortality consist of attacks by insects or fungi, which are otherwise unlikely to attack and kill healthy, vigorous trees (Sturrock et al. 2011). Recent large-scale insect outbreaks may be due to a combination of climate change (drought-stressed trees, warm summers that accelerate insect growth and development, warm winters that allow insect larvae to survive) and land-use change, which reduce tree vigor, particularly in dense forest stands (Romme et al. 2006).

Aquatic Ecosystems in the Rocky Mountains

Physical (water temperature) and biological (vegetation) conditions in aquatic ecosystems provide the habitat required by native aquatic and semiaquatic species, including amphibians, fishes, macroinvertebrates, and plant and periphyton communities (fig. 5; Smith 2024). Climate change is likely to warm water temperatures, change timing and magnitude of flows, and reduce water levels, due to greater evapotranspiration and longer growing seasons (Lall et al. 2018; Warziniack et al. 2024). Water level reductions and warming, with changed flow patterns, will interact with stresses arising from poorer water quality due to sedimentation and pollutants, fragmentation and loss of shading vegetation cover, water regulation, and human water use (e.g., irrigation, diversions, reservoirs; Young et al. 2018).

Due to likely loss of cold and transitional stream reaches under warmer and drier conditions, vulnerability of freshwater ecosystems overall is greater than for terrestrial vegetation types (Decker and Fink 2015). As water temperatures change, cool- or coldwater aquatic ecosystems are likely to transition to warmwater aquatic ecosystems, with only high-elevation streams potentially resistant to these effects. Likewise, warmer water temperatures will favor warmwater fish over coldwater fish (e.g., bull trout, *Salvelinus confluentus*; Young et al. 2018). Warmer stream temperatures may allow increased invasion by nonnative fish, supplanting native fish or increasing

hybridization risk (Smith et al. 2024; Young et al. 2018). Lower base flows could lead to reduced fish abundance and reduced recruitment (Smith et al. 2024; Young et al. 2018). Hot days could be a potential trigger point for fish mortality, especially if combined with low flow and low oxygen levels or of a long duration (Smith et al. 2024).



Figure 5—Climate change is likely to warm water temperatures and change timing and magnitude of flows in aquatic and wetland ecosystems. Muskrats (*Ondatra zibethicus*) are an ecohydrological indicator species and may serve as a bellwether for drying ecosystems (Ward et al. 2021; Courtesy photo by B. Hanberry).

Higher high flows and lower low flows could lead to increased erosion, which reduces water quality and affects reproduction by aquatic species, and leads to channel adjustments or breaks in stream connectivity (Young et al. 2018). Some aquatic species require floodplains created by peak flows for spawning or larval/juvenile development (Wydoski and Wick 1998). Fish may need access to thermal refugia and connectivity to access those refugia (e.g., cooler tributaries, streams with spring inputs, deeper pools, more shade; Young et al. 2018). For most species, intact connectivity within the hydrologic network will support movement and long-term migration to appropriate conditions as existing reaches become increasingly dry and warm (Young et al. 2018). Species assemblages likely will change as species migrate and as less mobile and more vulnerable community members are eliminated (Young et al. 2018).

Forests in the Rocky Mountains

Forests of the Rocky Mountains are primarily dominated by conifers and can be loosely grouped into (1) pinyon pine (*Pinus edulis*)-juniper (*Juniperus scopulorum, J. osteosperma*), (2) ponderosa pine (*Pinus ponderosa*), (3) mixed-conifer forests including ponderosa pine, Douglas-fir (*Pseudotsuga menziesii*), white fir (*Abies concolor*), and aspen (*Populus tremuloides*), (4) lodgepole pine (*Pinus contorta*), (5) aspen, and (6) subalpine forests including Engelmann spruce (*Picea engelmannii*), subalpine fir (*Abies lasiocarpa*), and lodgepole pine (Fornwalt et al. 2024). Disturbance regimes, which are modulated by climate and site-specific context, favor different tree species and drive plant community assemblages (Keane et al. 2018). For example, surface-fire tolerant ponderosa pine is a long-lived species; longevity is indicative of a species that has evolved in a system with infrequent stand-replacing disturbance events (Hanberry et al. 2020). Conversely, aspen, lodgepole pine, and many firs and spruces are short-lived species that typically respond well to stand-replacing disturbances (Fornwalt et al. 2024; Keane et al. 2018).

Fires and vegetation have synergistic and antagonistic interactions. Pinyon and juniper typically occur in warm, moisture- and fuel-limited low-elevation sites with discontinuous vegetation, resulting in infrequent fires (> 50 years between events) and fire severity determined by amount of available tree and shrub fuels (Veblen and Donnegan 2005). Generally, albeit varying in space and time, ponderosa pine forests, in particular at lower montane elevations (that is, warmer locations) with enough moisture to grow continuous fine surface fuels, had frequent (< 35 years) low-severity fire regimes, given discontinuous or at least clumped tree densities (Addington et al. 2018). Conversely, spruce and fir forests, with continuous understory to canopy and canopy to canopy fuels, at higher elevations (that is, colder locations) experienced high-severity fires, perhaps every century or so due to ignition limitations and also lack of fine surface fuels that ignite easily (Addington et al. 2018). However, fire spread is contagious and higher elevation forests experienced fires that spread upslope from grasslands and woodlands (Jack 1900). For mixed forests at intermediate elevations and temperatures or wet conditions, historical fire probabilities also were intermediate in range with mixed-severity burns occurring every 35 to 100 years (Arno 2000).

Fires, which are whole-system disturbances, have different regimes dependent on the amount, continuity, and type (that is, fine herbaceous or coarse tree) of fuels and probability of ignition. Fires can be patchy in distribution, though mixed-severity burns may remove 20 to 80 percent of the dominant trees (Addington et al. 2018). High-severity fires, with long flame lengths, occur when midstory trees are dense enough to allow fire to spread to the overstory (Urza et al. 2023). If canopies are not connected, torching of individual or grouped trees may occur, reducing overstory tree density without necessarily terminating most of the stand (Urza et al. 2023). Faster fuel accumulation due to greater productivity, which may occur under wetter conditions in water-limited forests or under warmer conditions in cold-limited forests, increases

frequency of fires, whereas fuel connectivity to and among tree canopies increases fire severity; extended warm and dry conditions and abundance of fine, cured fuels increase the chance of ignition (Urza et al. 2023).

During the early 1900s, fire exclusion became standard management to prevent forest destruction and loss of valuable board feet per acre, effectively disrupting the natural fire cycle of many ecosystems (Addington et al. 2018; Keane et al. 2002; Veblen and Donnegan 2005). Euro-American settlement resulted in extensive logging and grazing by domestic livestock during the mid- to late 1800s, modifying stand structure, fuels, and fire regimes, although during initial settlement before extensive land cover change, fire frequencies typically increased (Veblen and Donnegan 2005). In general, forest changes during the past century have included increased tree density, reduced tree diameters, increased abundance and range of fire-sensitive species, and increased fuel accumulation, in particular surface and ladder tree fuels (Addington et al. 2018; Hanberry et al. 2024; Keane et al. 2002).

Reduced water availability and hotter or more extreme temperatures directly may change community composition, favoring species with drought and heat tolerance traits (Fornwalt et al. 2024; Keane et al. 2018). As climate shifts, trees are likely to migrate upslope in elevation or northward in latitude (fig. 6; Fornwalt et al. 2024; Keane et al. 2018). Growth and productivity also may shift, as soils in warmer locations become water-limited during the growing season whereas cooler locations will experience an increase in growing season length (Keane et al. 2018). The influence of drought, in combination with variable heat tolerance of individual tree species, likely will drive overall forest response to climate change and, ultimately, resistance to damage from insects and fire (Keane et al. 2018). However, climate change effects may be obscured by the slow response times of long-lived tree species, with a lag in species response relative to the rate of change (Davis and Shaw 2001). Non-climate influences such as changing land use and inherent variability among species can make it difficult to attribute changes in abundance or range to climate alone (Coristine and Kerr 2011; Song et al. 2021).

Furthermore, drought may interact with severe disturbances, particularly wildfire and insects, creating different outcomes (Davis et al. 2019). While some tree species auto-replace under severe disturbances, such as drought-tolerant lodgepole pine, compounding disturbance events that include insects may prevent lodgepole pine regeneration and allow stand replacement by quaking aspen and southern tree species (Rhoades et al. 2018). Aspen is another species that thrives on severe fires and other severe disturbances, auto-replacing typically every century, but aspen is sensitive to drought and heat (Anderegg et al. 2013; Carroll et al. 2019; Kulakowski et al. 2013). Aspen has been successful at establishing following recent high-severity wildfires upslope of its previous range within burns, but not in unburned areas, probably due to seedbed requirements (Nigro et al. 2022).



Figure 6—Spruces, firs, and Douglas-fir (*Pseudotsuga menziesii*) may increase in alpine meadows as temperatures warm. These species currently are expanding due to fire exclusion (Hanberry et al. 2020; Courtesy photo by B. Hanberry).

Extreme Events and Infrastructure Development in the Rocky Mountains

Residential development has increased adjacent to public lands (Karau et al. 2024). Wildlands are desirable amenities that supply ecosystem services of privacy, aesthetics, well-being, and recreational access (Warziniack et al. 2024; Wear and Joyce 2012). Furthermore, national forests have protected area status as an added attraction, guaranteeing reduced development and changes to the surroundings, and the benefit of investment in land management services, including wildlife support, invasive species control, and fuel treatments (Karau et al. 2024; Wear and Joyce 2012).

At the same time, the risk of wildland fire affecting homes and other structures is greatest in wildlands (Hanberry 2020). The length of wildfire season has increased due to warming temperatures, with increased number of extreme fire weather days, and fire severity has increased with fuel accumulation (fig. 7; Karau et al. 2024; Rangwala 2024). Severe wildfires are difficult to contain, particularly in areas with widespread low-density exurban housing, driving up the cost of fire containment (Karau et al. 2024). Awareness and preparation for wildland fire by communities and homeowners is a challenge near public lands, where homeowners tend to rely on public agencies to provide fire containment rather than act themselves (Busby and Albers 2010). Ideally, homeowners would limit fuel accumulation surrounding homes and fires could spread

without causing extensive damage (Karau et al. 2024; Wear and Joyce 2012). Defensible perimeters protect buildings from forest fire and inversely can prevent fire spread from houses to forests (Karau et al. 2024).

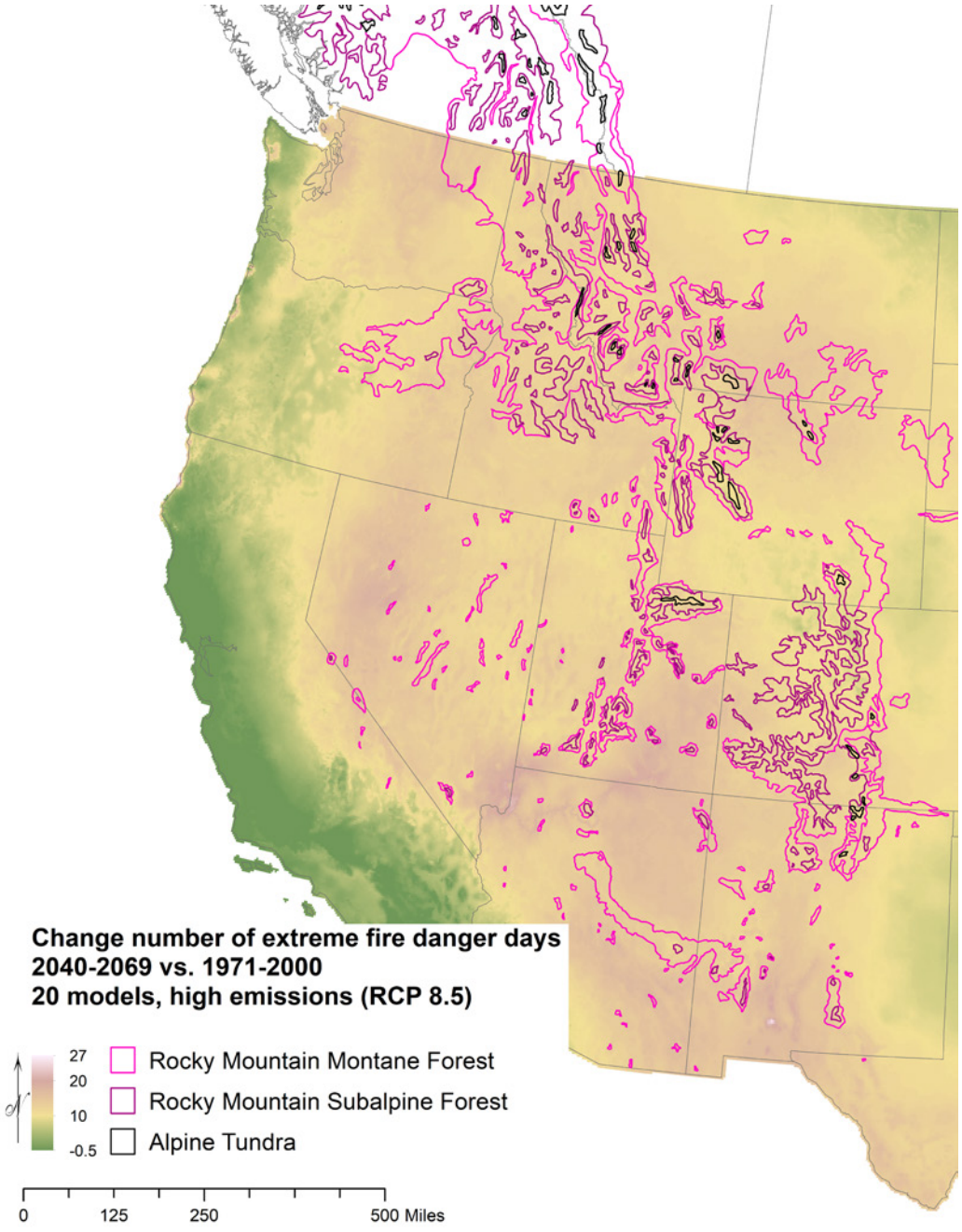


Figure 7—Projected change in the mean number of extreme fire danger days (100-hour fuel moisture below 3 percentile) during 2040–2069 as compared to the baseline mean number of extreme fire danger days during 1971–2000 for an ensemble of 20 general circulation models under the fossil-fueled representative concentration pathway (RCP 8.5; Abatzoglou 2013; modified from Climate Toolbox 2022).

As the spatial extent of recently burned wildfires increases, previously burned areas may provide fuel breaks that influence the pattern, extent, and severity of future fires (Urza et al. 2023). Fuel treatments can protect bordering infrastructure and treatment placement can take advantage of adjacency to land uses, such as roads and fields, that act as firebreaks, and leverage locations of prior treatments or fires (Urza et al. 2023). Large fires can easily overwhelm small, unconnected fuel treatments, and extreme weather conditions will overwhelm most preventive and suppression actions (Prichard et al. 2020).

Infrastructure is vulnerable to extreme weather events and stress from changing climate through flooding, drought, wildfire, extreme heat, ice, strong winds, freeze-thaw cycles, and snowmelt (Hanberry et al. 2024). The U.S. Department of Agriculture, Forest Service infrastructure portfolio consists of structures and facilities for administrative, transportation, recreation, or special uses, such as dams and camping concessions (Hanberry et al. 2024). Extreme events may cause injury and mortality and disrupt livelihoods and infrastructure networks, such as roads, water and power supplies, and health and emergency services (Mora et al. 2018). Extreme events often interact because they are spatially and temporally dependent, leading to underestimation of risk (Mora et al. 2018; Zscheischler et al. 2018). Fire may lead to postfire flooding due to loss of vegetation and debris flow due to soil erosion, intensifying vulnerability (Keller et al. 2021). On top of heat, fire, and postfire events, climate change likely will increase heavy precipitation and flooding, resulting in a cascade in which greater temperatures may produce more fires that will interact with floods (Rangwala 2024). Flooding is the most costly natural disaster in the United States, through direct rainfall, overbank flows, soil erosion, and landslides (Keller et al. 2021). Simple designs to reduce flooding hazard, by reducing susceptibility to flooding, include regularly cleaning out culverts, maintaining roads and ensuring surface drainage measures are functional, modifying bridge and culvert designs to accommodate larger flows, avoiding potential of stream diversion at road-stream crossings, and correcting scour problems at bridges (Keller et al. 2021).

Social and Economic Considerations in the Rocky Mountains

Forests on public and private lands benefit human populations through ecosystem services that encompass four primary functions: provisioning materials, such as forest products, forage, and water; regulating services, such as clean water, erosion control, climate and natural hazard control; cultural services, such as scenic and recreational opportunities, spiritual enrichment, and physical and mental health; and providing supporting services, such as nutrient cycling, pollination, and soil formation (Warziniack et al. 2024; Wear and Joyce 2012). Public land management aims to provide both market and nonmarket ecosystem services, maximizing public welfare derived from forests by maintaining and considering resources, recreation, infrastructure, natural resource extraction, scenic appeal, Tribal areas, heritage status, land status, designated areas, and socioeconomics (Arapaho-Roosevelt National Forests and Pawnee Grasslands 1997). National forests connect local residents and visitors to the land and

one another, through additional opportunities such as volunteering, education, and scientific learning, contributing to the quality of life, sense of place, and scientific monitoring (Fairchild and Hanberry 2024).

Natural resource-based communities derive economic benefits from surrounding natural resources and withstand associated natural disturbances. Tourism and recreation are important for economies and for human well-being, with winter and water sports likely to be affected by climate change (Sánchez et. 2024). Demand for recreation is projected to increase with population growth; however, recreation and tourism are relatively adaptive (Sánchez et. 2024). Vulnerable populations may have relatively greater exposure to climate change effects or have sensitive characteristics, resulting in reduced adaptive capacity (Warziniack et al. 2024; Wear and Joyce 2012).

Climate change likely will exacerbate long-standing issues about sharing natural resources, including water rights (Prestemon et al. 2016; Smith 2024). Currently, water use is highly subsidized so that prices do not reflect the full cost of water, resulting in the majority of water consumption for agricultural irrigation (Prestemon et al. 2016). With increased drought and demand for water from other competitive sectors (domestic, public, commercial, and industrial water use), transfer of water rights may occur from agriculture to urban centers or for multiple objectives, such as maintenance of minimum stream flows for protection of aquatic species, recreation, maintenance of water quality, and riparian land restoration (Prestemon et al. 2016).

Management in the Rocky Mountains

Climate change is projected to significantly damage human health, the economy, and the environment (Mora et al. 2018). A number of approaches have been developed to mitigate climate change through low-carbon energy technologies and energy efficiency (Griffith et al. 2021). At the Federal level, recommendations for sustainable operations include reducing water and energy consumption and increasing water and energy efficiency, purchasing sustainable goods and services, reducing waste generation, and reusing and recycling materials. Sustainable operations include a shift toward renewable energy, with installation of solar power systems on buildings as an option to consider (USDA FS 2014). It is possible to lessen exposure of infrastructure to extreme events, such as by not building near floodplains (Hanberry et al. 2024), and diminish susceptibility by building for tolerance to extreme events.

Climate changes are likely to disrupt natural resources in ways that challenge management. Public forest management can be slow to adjust management plans by design, due to the public process of resource planning (Wear and Joyce 2012). Earlier action compared to delay or inaction will allow development of experience in incorporating climate adaptation planning into natural resource management and testing effectiveness of actions (Keane et al. 2018). Decisionmakers may wish to take active steps to anticipate and steer change in desired directions and to avoid

consequences of unplanned, crisis-based responses (Timberlake et al. 2024). Integrating climate risk management into existing planning, decision, and operations workflows, in contrast to adding novel decision processes for climate adaptation alone, can make adaptation more likely to succeed by strengthening already familiar processes with new information and tools (Keane et al. 2018). Known climate change vulnerabilities can inform prioritization of projects to receive funding, capacity, and other resources (Keane et al. 2018). The feasibility of project goals may be dependent on decisions to resist, accept, or direct climate-driven changes (Timberlake et al. 2024).

Integration of climate change into existing management workflows coherently addresses adaptation along with other objectives that are regularly managed, but in some cases may prove insufficient to address the full range and magnitude of climate risk (Essen et al. 2022). Even if potential magnitude and effects of climate change are assessed and adaptive management responses are developed, adaptation measures may not occur unless management organizations have sufficient people and financial resources (Essen et al. 2022). Linking across resource sectors and agency boundaries may generate additional capacity, for example, to focus on water rights (Essen et al. 2022). Addressing the full range of future climate change may entail changes in organizational practices and procedures (Tedim et al. 2021).

Effective management for climate change retains the basic management objective to restore and maintain resources against non-climate anthropogenic stresses. The 2012 Forest Service Planning Rule codifies management principles for National Forest System lands (USDA FS 2012). Systems that are already degraded from non-climate stressors likely have lower adaptive capacity for climate change and poorly nourished animals are less resistant to drought, temperature fluctuations, fire, land-use pressure, and disease (Goulson et al. 2015; Keane et al. 2018; Kolb et al. 2019; Wear and Joyce 2012). Both conventional and climate change management strategies for aquatic systems incorporate enhancing hydrologic connectivity and stream flows and guarding streams from sedimentation and erosion (Smith et al. 2024; Young et al. 2018). However, connectivity projects will be more proactive if designed for ecosystems likely to remain, for instance, perennial streams or deep pools, or future ecosystems with anticipation of higher high flows, lower low flows, and more frequent extreme flood events (Smith et al. 2024; Young et al. 2018). Management additionally includes restoration of natural disturbance regimes, including wildland fire where practical, to enhance resources (Keane et al. 2018; Veblen and Donnegan 2005). Other benefits include reduced fire severity, reduced outbreaks of native insect and disease populations, and reduced drought stress (Keane et al. 2018; Romme et al. 2016). For example, fires that are contained could lead to more severe fire or insect disturbances but areas treated with thinning or prescribed burns may experience mostly surface fires rather than crown fires, and equally, insect outbreaks and drought stress may be limited by reduced basal area (Keane et al. 2018; Romme et al. 2016). Management that considers interacting impacts of climate and other stressors on resources, coordinated across landscape scales, can greatly support ecosystems in the future.

References

Abatzoglou, J.T. 2013. Development of gridded surface meteorological data for ecological applications and modelling. International Journal of Climatology. 33: 121–131. https://doi.org/10.1002/joc.3413

Addington, R.N.; Aplet, G.H.; Battaglia, M.A.; [et al.]. 2018. Principles and practices for the restoration of ponderosa pine and dry mixed-conifer forests of the Colorado Front Range. RMRS-GTR-373. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 121 p. https://doi.org/10.2737/RMRS-GTR-373

Anderegg, L.D.; Anderegg, W.R.; Abatzoglou, J.; Hausladen, A.M.; Berry, J.A., 2013. Drought characteristics' role in widespread aspen forest mortality across Colorado, USA. Global Change Biology. 19: 1526–1537. https://doi.org/10.1111/gcb.12146

Arapaho and Roosevelt National Forests and Pawnee Grasslands. 1997. 1997 Revision of the Land and Resource Management Plan. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fseprd641737.pdf [Accessed 16 February 2022]

Arno, S. 2000. Fire in western forest ecosystems. In: Brown, J.K; Smith, J.K., eds. Wildland fire in ecosystems: Effects of fire on flora. Gen. Tech. Rep. RMRS-GTR-42-vol. 2. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 257 p. https://doi.org/10.2737/RMRS-GTR-42-V2

Bigler, C.; Kulakowski, D.; Veblen, T.T. 2005. Multiple disturbance interactions and drought influence fire severity in Rocky Mountain subalpine forests. Ecology. 86: 3018–3029. https://doi.org/10.1890/05-0011

Bretfeld, M.; Franklin, S.B.; Peet, R.K. 2016. A multiple-scale assessment of long-term aspen persistence and elevational range shifts in the Colorado Front Range. Ecological Monographs. 86: 244–260. https://doi.org/10.1890/15-1195.1

Busby, G.; Albers, H. J. 2010. Wildfire risk management on a landscape with public and private ownership: Who pays for protection? Environmental Management. 45:296–310. https://doi.org/10.1007/s00267-009-9381-x

Carroll, C.J.; Martin, P.H.; Knapp, A.K.; [et al.]. 2019. Temperature induced shifts in leaf water relations and growth efficiency indicate climate change may limit aspen growth in the Colorado Rockies. Environmental and Experimental Botany. 159: 132–137. https://doi.org/10.1016/j.envexpbot.2018.12.014

Climate Toolbox. 2022. Climate mapper. https://climatetoolbox.org/tool/climate-mapper [Accessed 7 March 2022]

Coristine, L.E.; Kerr, J.T. 2011. Habitat loss, climate change, and emerging conservation challenges in Canada. Canadian Journal of Zoology. 89:435–451. https://doi.org/10.1139/z11-023

Davis, M.B.; Shaw, R.G. 2001. Range shifts and adaptive responses to Quaternary climate change. Science. 292(5517). 673–679. https://doi.org/10.1126/science.292.5517.673

Davis, K.T.; Dobrowski, S.Z.; Higuera, P.E.; [et al.]. 2019. Wildfires and climate change push low-elevation forests across a critical climate threshold for tree regeneration. Proceedings of the National Academy of Sciences of the United States of America. 116:6193–6198. https://doi.org/10.1073/pnas.1815107116

Decker, K.; Fink M. 2015. Ecosystems. In: Decker, K.; Grunau, L., Handwerk, J.; Siemers, J., eds. Climate change vulnerability assessment for Colorado Bureau of Land Management. Fort Collins, Colorado: Colorado Natural Heritage Program, Colorado State University. https://cnhp.colostate.edu/download/documents/2015/CCVA_for_Colorado_BLM_final.pdf

Essen, M.; McCaffrey, S.; Abrams, J. [et al.]. 2022. Improving wildfire management outcomes: Shifting the paradigm of wildfire from simple to complex risk. Journal of Environmental Planning and Management. 1–19. https://doi.org/10.1080/09640568.2021.2007861

Fairchild, M.; Hanberry, B.B. 2024. Chapter 7: Community involvement to increase monitoring capacity for climate-sensitive resources. In: Hanberry, B.B., ed. Colorado Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-GTR-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 281-292. https://doi.org/10.2737/RMRS-GTR-438

Foote, E. 1856. Circumstances affecting the heat of the sun's rays. American Journal of Science and Arts. 22: 382–383.

Fornwalt, P.J; Wion, A.P.; Schoettle, A.W.; [et al.]. 2024. Chapter 5: Vulnerability of major Colorado Front Range tree species to climate change. In: Hanberry, B.B., ed. Colorado Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-GTR-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 123-244. https://doi.org/10.2737/RMRS-GTR-438

Friggens, M.; Bagne, K.; Finch, D.; [et al.]. 2013. Review and recommendations for climate change vulnerability assessment approaches with examples from the Southwest. Gen. Tech. Rep. RMRS-GTR-309. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 106 p. https://doi.org/10.2737/RMRS-GTR-309

Glick, P.; Stein, B.A.; Edelson, N.A. 2011. Scanning the conservation horizon: A guide to climate change vulnerability assessment. Washington, DC: National Wildlife Federation. 168 p. https://www.fs.usda.gov/research/treesearch/37406

Goulson, D.; Nicholls, E.; Botías, C.; [et al.]. 2015. Bee declines driven by combined stress from parasites, pesticides, and lack of flowers. Science. 347(6229), 1255957. https://doi.org/10.1126/science.1255957

Griffith, S.; Calisch, S.; Fraser, L. 2020. Rewiring America, a handbook to winning the climate fight. https://www.rewiringamerica.org/handbook 2020 [Accessed 31 July 2021].

Hanberry, B.B. 2020. Reclassifying the wildland-urban interface using fire occurrences for the United States. Land. 9: 225. https://doi.org/10.3390/land9070225

Hanberry, Brice B. 2024. Tree species distributions of southern North America under current and future climate to year 2100. Fort Collins, CO: Forest Service Research Data Archive. https://doi.org/10.2737/RDS-2024-0016

Hanberry, B.B.; Justice, D.C.; Powell, D.C. 2020. Discovering Douglas-fir woodlands in the historical forests of Umatilla National Forest, Eastern Oregon and Washington. Forests. 11(10): 1122. https://doi.org/10.3390/f11101122

Hanberry, B. B.; Seidel, J. M.; DeLeon, P. 2024. Exploring tree density Increases after fire exclusion in the northern Front Range and Great Plains, Colorado, USA. Fire. 7 :103.

Hanberry, B.B; Zheng, X.; Johnson, E.; [et al.]. Chapter 10: Infrastructure, flood risk, and sustainable operations in Front Range National Forests. In: Hanberry, B.B., ed. Colorado Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-GTR-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 361-386. https://doi.org/10.2737/RMRS-GTR-438

Hart, S.J.; Preston, D.L. 2020. Fire weather drives daily area burned and observations of fire behavior in mountain pine beetle affected landscapes. Environmental Research Letters. 15(5): 054007. https://doi.org/10.1088/1748-9326/ab7953

Hayhoe, K.; Wuebbles, D.J.; Easterling, D.R.; [et al.]. 2018: Our changing climate. In: Reidmiller, D.R.; Avery, C.W.; Easterling, D.R.; [et al.]., eds. Impacts, risks, and adaptation in the United States: Fourth national climate assessment, Volume II. Washington, DC: U.S. Global Change Research Program: 72–144. https://doi.org/10.7930/NCA4.2018 and https://nca2018.globalchange.gov/downloads/NCA4_Ch02_Changing-Climate_Full.pdf

Intergovernmental Panel on Climate Change [IPCC]. 2021: Summary for policymakers. In: Masson- Delmotte, V.; Zhai P.; A. Pirani A.; [et al.], eds. Climate change 2021: The physical science basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change Switzerland Intergovernmental Panel on Climate Change. https://www.ipcc.ch/report/ar6/wg1/downloads/report/IPCC_AR6_WGI_SPM_final.pdf [Accessed 11 February 11, 2022]

Jack, J.G. 1900. Pikes Peak, Plum Creek, and South Platte Reserves. Twentieth annual report of the United States Geological Survey to the Secretary of the Interior, 1898–1899. Washington, DC: United States Government Printing Office. https://pubs.usgs.gov/ar/20-5/report.pdf [Accessed 16 April 2022]

Karau, E.C.; Hanberry, B.B.; Dillon, G.K. Chapter 9: Fire risk and climate change in the Colorado Front Range. In: Hanberry, B.B., ed. Colorado Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-GTR-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: **XX-XX**. https://doi.org/10.2737/RMRS-GTR-438

Keane, R.E.; Ryan, K.C.; Veblen, T.T; [et al.]. 2002. Cascading effects of fire exclusion in the Rocky Mountain ecosystems: A literature review. Gen. Tech. Rep. RMRS-GTR-91. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 24 p. https://doi.org/10.2737/RMRS-GTR-91

Keane, R.E.; Mahalovich, M.F.; Bollenbacher, B.L.; [et al.]. 2018. Effects of climate change on forest vegetation in the Northern Rockies Region. In: Halofsky, J.E.; Peterson, D.L.; Dante-Wood, S.K.; [et al.], eds. Climate change vulnerability and adaptation in the northern Rocky Mountains. Gen. Tech. Rep. RMRS-GTR-374. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. Chapter 6: 128–173. https://www.fs.usda.gov/research/treesearch/55991

Keane, R.E.; Schoettle, A.W.; Tomback, D.F. 2022. Effective actions for managing resilient high elevation five-needle white pine forests in western North America at multiple scales under changing climates. Forest Ecology and Management. 505: 119939. https://doi.org/10.1016/j.foreco.2021.119939

Keller, G.R.; Boak, L.J.; Furniss, M.J. 2021. Climate change and infrastructure in the Sierra Nevada. In: Halofsky, J.E.; Peterson, D.L.; Buluç, L.Y; [et al.], eds. Climate change vulnerability and adaptation for infrastructure and recreation in the Sierra Nevada. Gen. Tech. Rep. PSW-GTR-272. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 49–179. Chapter 4. https://www.fs.usda.gov/psw/publications/documents/psw_gtr272/psw_gtr272.pdf

Kolb, T.; Keefover-Ring, K.; Burr, S.J.; [et al.]. 2019. Drought-mediated changes in tree physiological processes weaken tree defenses to bark beetle attack. Journal of Chemical Ecology. 45: 888–900. https://doi.org/10.1007/s10886-019-01105-0

Kulakowski, D.; Matthews, C.; Jarvis, D.; [et al.]. 2013. Compounded disturbances in sub-alpine forests in western Colorado favour future dominance by quaking aspen (*Populus tremuloides*). Journal of Vegetation Science. 24: 168–176. https://doi.org/10.1111/j.1654-1103.2012.01437.x

Lall, U.; Johnson, T.; Colohan, P.; [et al.]. 2018: Water. In: Reidmiller, D.R.; Avery, C.W.; Easterling, D.R.; [et al.]., eds. Impacts, risks, and adaptation in the United States: Fourth national climate assessment, Volume II. Washington, DC: U.S. Global Change Research Program: 145–173. https://nca2018.globalchange.gov/downloads/NCA4_Ch03_Water_Full.pdf

Lipton, D.; Rubenstein, M.A.; Weiskopf, S.R.; [et al.]. 2018. Ecosystems, ecosystem services, and biodiversity. In: Reidmiller, D.R.; Avery, C.W.; Easterling, D.R.; [et al.]., eds. Impacts, risks, and adaptation in the United States: Fourth national climate assessment, Volume II. Washington, DC: U.S. Global Change Research Program: 268–321. https://nca2018.globalchange.gov/downloads/NCA4_Ch07_Ecosystems_Full.pdf

Mora, C.; Spirandelli, D.; Franklin, E.C.; [et al.]. 2018. Broad threat to humanity from cumulative climate hazards intensified by greenhouse gas emissions. Nature Climate Change. 8:1062–1071. https://doi.org/10.1038/s41558-018-0315-6

Negrón, J.F. 2024. Chapter 8: Insects and diseases and climate change in Colorado Front Range forests. In: Hanberry, B.B., ed. Colorado Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-GTR-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 323-360. https://doi.org/10.2737/RMRS-GTR-438

Nigro, K.M.; Rocca, M.E.; Battaglia, M.A.; [et al.]. 2022. Wildfire catalyzes upward range expansion of trembling aspen in southern Rocky Mountain beetle-killed forests. Journal of Biogeography. 49(1): 201–214. https://doi.org/10.1111/jbi.14302

Page, W.G.; Jenkins, M.J.; Alexander, M.E. 2014. Crown fire potential in lodgepole pine forests during the red stage of mountain pine beetle attack. Forestry: An International Journal of Forest Research. 87: 347–361. https://doi.org/10.1093/forestry/cpu003

Prichard, S.J.; Povak, N.A.; Kennedy, M.C.; [et al.]. 2020. Fuel treatment effectiveness in the context of landform, vegetation, and large, wind-driven wildfires. Ecological Applications. 30: p.e02104. https://doi.org/10.1002/eap.2104

Prestemon, J.P.; Kruger, L.; Abt, K.L. [et al.]. 2016. Economics and societal considerations of drought. In: Vose, J.; Clark, J.; Luce, C.; Patel-Weynand, T., eds. Effects of drought on forests and rangelands in the United States: A comprehensive science synthesis. Gen. Tech. Rep. WO-93b. Washington, DC: U.S. Department of Agriculture, Forest Service, Washington Office: 253–281. https://www.fs.usda.gov/research/treesearch/52339

Rangwala, I. 2024. Chapter 3: Future climate scenarios for Colorado's Front Range forests. In: Hanberry, B.B., ed. Colorado Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-GTR-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 39-56. https://doi.org/10.2737/RMRS-GTR-438

Rhoades, C.C.; Pelz, K.A.; Fornwalt, P.J.; [et al.]. 2018. Overlapping bark beetle outbreaks, salvage logging and wildfire restructure a lodgepole pine ecosystem. Forests. 9: 101. https://doi.org/10.3390/f9030101

Romme, W.H.; Clement, J.; Hicke, J.; [et al.]. 2006. Recent forest insect outbreaks and fire risk in Colorado forests: A brief synthesis of relevant research. Fort Collins, CO: Colorado Forest Restoration Institute. https://cfri.colostate.edu/wp-content/uploads/sites/22/2021/12/Romme.et_.al_.2006.pdf

Sánchez, J.J.; Huber, C.; Loomis, J. 2024. Chapter 12: Effects of climate change on outdoor recreation benefits in the Pike-San Isabel and Arapaho-Roosevelt National Forests. In: Hanberry, B.B., ed. Colorado Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-GTR-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 413-452. https://doi.org/10.2737/RMRS-GTR-438

Smith, D.M. 2024. Chapter 4: Water resources of the Front Range National Forests. In: Hanberry, B.B., ed. Colorado Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-GTR-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 57-122. https://doi.org/10.2737/RMRS-GTR-438

Song, Y.; Zajic, C.J.; Hwang, T.; [et al.]. 2021. Widespread mismatch between phenology and climate in human-dominated landscapes. AGU Advances. 2: p.e2021AV000431. https://doi.org/10.1029/2021AV000431

Sturrock, R.N.; Frankel, S.J.; Brown, A.V.; [et al.]. 2011. Climate change and forest diseases. Plant Pathology. 60(1): 133–149. https://doi.org/10.1111/j.1365-3059.2010.02406.x

Tedim, F.; McCaffrey, S.; Leone, V.; [et al.]. 2021. Supporting a shift in wildfire management from fighting fires to thriving with fires: The need for translational wildfire science. Forest Policy and Economics. 131: 102565. https://doi.org/10.1016/j.forpol.2021.102565

Timberlake, T.J.; Shorrock, D.E.; Jaworski, D. 2024. Chapter 13: Conclusions. In: Hanberry, B.B., ed. Colorado Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-GTR-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 453-460. https://doi.org/10.2737/RMRS-GTR-438

Urza, A.K.; Hanberry B.B.; Jain, T.B. 2023. Landscape-scale fuel treatment effectiveness: Lessons learned from wildland fire case studies in forests of the western United States and Great Lakes region. Fire Ecology. 19:1. https://doi.org/10.1186/s42408-022-00159-y

USDA Forest Service [USDA FS]. 2012. National Forest System Land Management Planning. Washington, DC: U.S. Department of Agriculture, Forest Service. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5362536.pdf [Accessed 16 February 2022]

USDA Forest Service [USDA FS]. 2014. Sustainable operations. https://www.fs.usda.gov/managing-land/sc/susops [Accessed 21 February 2022]

Vanbergen, A.J.; Insect Pollinators Initiative. 2013. Threats to an ecosystem service: Pressures on pollinators. Frontiers in Ecology and the Environment. 11: 251–259. https://doi.org/10.1890/120126

Veblen, T.T.; Donnegan, J.A. 2005. Historical range of variability for forest vegetation of the National Forests of the Colorado Front Range. Agreement No. 1102-0001-99-033. Golden, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Region. 151 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5386430.pdf

Ward, E.M.; Solari, K.A.; Varudkar, A.; [et al.]. 2021. Muskrats as a bellwether of a drying delta. Communications Biology. 4:1–9. https://doi.org/10.1038/s42003-021-02288-7

Warziniack, T.; Cox, E.; Froemke, P.; [et al.]. 2024. Chapter 11: Vulnerability of ecosystem services along the Colorado Front Range. In: Hanberry, B.B., ed. Colorado Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-GTR-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 387-412. https://doi.org/10.2737/RMRS-GTR-438

Wear, D.N.; Joyce, L.A. 2012. Climate change, human communities, and forests in rural, urban, and wildland-urban interface environments. In: Vose, J. M.; Peterson, D. L.; Patel-Weynand, T., eds. Effects of climatic variability and change on forest ecosystems: A comprehensive science synthesis for the U.S. forest sector. Gen. Tech. Rep. PNW-GTR-870. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 97–123. Chapter 3. https://www.fs.usda.gov/research/treesearch/42652

White, A.M. 2024. Chapter 6: Future-oriented wildlife management in the Pike-San Isabel and Arapaho-Roosevelt National Forests under a warming climate: Anticipating change and facing uncertainties with a focus on avian species . In: Hanberry, B.B., ed. Colorado Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-GTR-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 245-280. https://doi.org/10.2737/RMRS-GTR-438

Wolkovich, E.M.; Cleland, E.E. 2014. Phenological niches and the future of invaded ecosystems with climate change. AoB Plants. 6: plu013. https://doi.org/10.1093/aobpla/plu013

Wydoski, R.S.; Wick, E.J. 1998. Ecological value of floodplain habitats to razorback suckers in the upper Colorado River basin. Denver, CO: Upper Colorado River Basin Recovery Program. https://www.nativefishlab.net/library/textpdf/20871.pdf [Accessed 7 July 2023]

Young, M.K.; Isaak, D.J.; Spaulding, S.; [et al.]. 2018. Climate vulnerability of native cold-water salmonids in the Northern Rockies Region. In: Halofsky, J.E.; Peterson, D.L.; Dante-Wood, S.K.; [et al.], eds. Climate change vulnerability and adaptation in the northern Rocky Mountains. Gen. Tech. Rep. RMRS-GTR-374. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. Chapter 5: 87-127. https://www.fs.usda.gov/research/treesearch/55990

Zscheischler, J.; Westra, S.; Van Den Hurk, B.J.; [et al.]. 2018. Future climate risk from compound events. Nature Climate Change. 8(6): 469–477. https://doi.org/10.1038/s41558-018-0156-3

Future Climate Scenarios for Colorado's Front Range Forests

Imtiaz Rangwala

Introduction

This chapter provides a summary of the climate change analysis done for Colorado's Front Range forests that include the Pike-San Isabel and Arapaho-Roosevelt (PSI-AR) National Forests. This analysis includes assessment of historical climate trends and future climate projections by 2050. Because of the wide range of future climate projections simulated by global climate models, four divergent future climate scenarios were selected to incorporate that large range and to quantify plausible future climate scenarios for PSI-AR. Following sections provide information on the (1) data and approaches used for the analysis and (2) the main findings from it. Not all of the results and output from the analysis are shown here, and they can be accessed elsewhere as described below.

Dataset and Methods

Table 1 describes the different climate datasets employed for the analysis of historical climate trends and future climate projections. This analysis was also done using different spatial boundary considerations as shown in figure 1, including a point location which allowed for inclusion of several other ecologically relevant climate metrics. These considerations were made to derive scale-appropriate quantification of climate and hydrological projections. For example, the rectangular boundary that encompasses all of the Colorado Front Range forests (fig. 1a) was considered for the selection of future climate change scenarios because a larger regional consideration is generally more appropriate for this purpose. Forest boundaries specific shapefiles were considered to quantify projections in hydroclimate variables that were specific to either Pike-San Isabel (PSI) or Arapaho-Roosevelt (AR) Forests. Finally, a point location along the middle of the north-south extent of the PSI-AR Forest, that is largely representative of the Front Range forests, was identified to derive specific ecologically sensitive metrics that require a very localized spatial consideration.

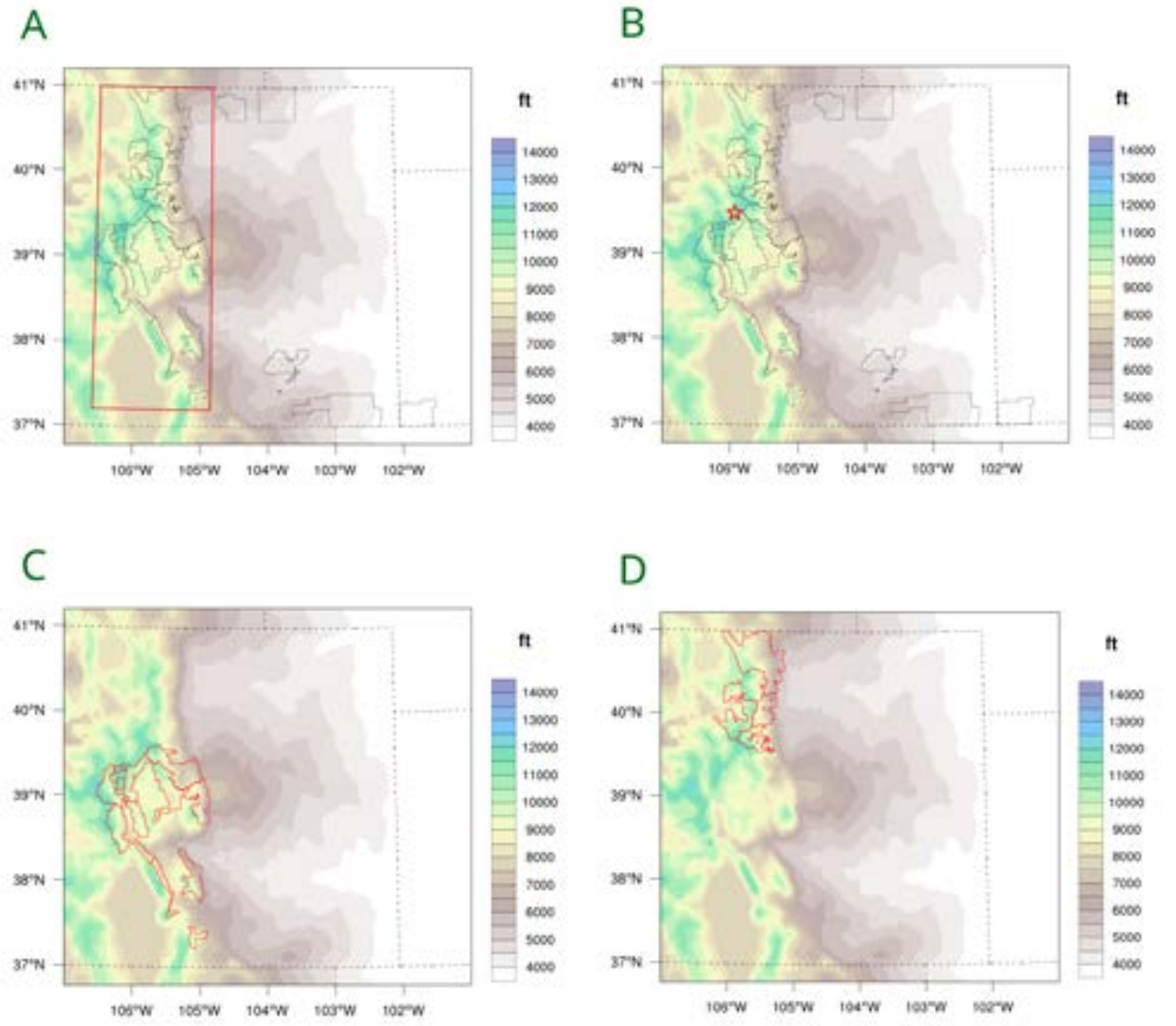


Figure 1a-d—Spatial consideration for climate analysis for the Pike, San Isabel, Arapaho, and Roosevelt (PSI-AR) National Forests. Four different considerations were made: (a) rectangular region between 37.2-41°N latitude and 104.85-106.6°W longitude, (b) point location at 39.5°N and 106°W, (c) PSI, and (d) AR forests boundaries. Colored contours in the background show elevation in feet.

Different datasets were considered for the analysis because no single dataset had all the attributes that were desired. Nonetheless, these different datasets are largely consistent with each other in terms of describing long-term historical trends or interannual variability. For assessing long-term (1895–2018) historical trends in temperature and precipitation, the 5 km monthly NOAA's NCEI nClimDiv data (Vose et al. 2014) are analyzed. Monthly precipitation and vapor pressure deficit data between 1979 and 2018 from the 4 km gridMET data (Abatzoglou 2013) are used to estimate Forest Drought Stress Index (FDSI) based on the relationship described in Williams et al. (2013). Trends

in the atmospheric freezing level during the cold season (October–May) between 1949 and 2018 are obtained from North American Freezing Level Tracker (Abatzoglou et al. 2017). A linear regression is performed between this freezing level height data and high resolution (800 m) temperature data (Topo Wx; Oyler et al. 2014) at a point location in PSI-AR to assess the relationship between the two variables. This relationship is then used to estimate changes in future freezing level heights.

Projected climate change is quantified for the nominal "2050" time horizon, which is defined as the change between historical (1985–2015) and future (2035–2065) time periods. The selection of these time periods was guided by the larger project team. For quantifying future climate projections, two statistically downscaled global climate model datasets were considered. For selecting future climate scenarios and quantifying projected change in several primary hydroclimate variables, US DOI Bureau of Reclamation's statistically downscaled (bias correction spatially disaggregated; BCSD) CMIP5 Climate and Hydrology Projection dataset available at both 1° (~100 km; for temperature and precipitation only) and 1/8° (~12 km; for all other variables) grid cell resolutions are considered (Reclamation 2014). Future climate scenarios are selected based on changes in annual temperature and precipitation from 34 Global Climate Models (GCMs) and two emissions scenarios—Representative Concentration Pathway (RCP) 4.5 (moderate emissions) and RCP 8.5 (high emissions). These GCMs are identified in table A1 in Appendix A of this chapter. Projected change in several ecologically relevant metrics were quantified for a point location (39.5°N, 106°W; mean grid cell elevation = 10,895 ft) based on the Multivariate Adaptive Climate Analog (MACA) downscaled climate projections ((Abatzoglou and Brown 2012; see table 1 for specific metrics).

Rasters (as geotiff) and datafile (as netcdf) of seasonal and annual mean values of nine variables—Mean, Maximum and Minimum Temperatures, Precipitation, Soil Moisture, Potential Evapotranspiration, Actual Evapotranspiration, Runoff, and Snow Water Equivalent—for the historical and future (all four scenarios) periods are produced for a larger domain surrounding the Front Range forests based on the Bureau of Reclamation's BCSD CMIP5 Climate and Hydrology Projection dataset. These datasets can be accessed at http://tinyurl.com/rasters-all. Nine variables between the future and historical period across the four scenarios can be found at http://tinyurl.com/spatial-plots (figs. A1–A7).

Table 1—Observed historical and future climate projections data used in the analysis.

	Dataset	Variables	Time period	Spatial resolution	Source
Historical climate observations	NOAA NCEI nClimDiv Dataset	Temperature, Precipitation	1895–2018	5 km	https://www.ncei.noaa.gov/pub/data/cirs/climdiv/
	gridMET	Precipitation, Vapor Pressure Deficit	1979–2018	4 km	https://www.climatologylab.org/gridmet.html
	North American Freezing Level Tracker	Atmospheric Freezing Level	1949–2018	2.5°	https://wrcc.dri.edu/cwd/products/
	Topo Wx	Temperature	1948–2016	800 m	http://www.scrimhub.org/resources/topowx/
Future climate projections	Reclamation BCSD CMIP5 Climate and Hydrology Projection	Temperature, Precipitation, Soil Moisture, Potential Evapotranspiration, Actual Evapotranspiration, Runoff, Snow Water Equivalent	1950–2099	12 km, 1°	https://gdo-dcp.ucllnl.org
	MACA Downscaled Climate Projections	Temperature, Precipitation, Vapor Pressure Deficit, Coldest Winter Temperature, Hottest Summer Temperature, Date of First Fall Freeze, Date of Last Spring Freeze, Growing Season Length, Wildfire Metrics	1950–2099	4 km	http://www.climatologylab.org/maca.html

Results

Historical Climate Trends

Long-term observational record (since late 19th century) shows that temperatures in PSI-AR have increased by at least 2 °F (fig. 2a) when comparing the mean of the last two decades (1995–2017) with the 20th century period mean (1895–2000). The largest portion of this warming has occurred since the 1990s. Since 1994, only 1 year (2008) has been cooler than the long-term average, albeit only marginally. The period between 2001 and 2017 has been 1.5 °F warmer than the 20th century. The precipitation record, however, does not show any long-term trend (fig. 2b). Recent decades have been slightly on the wetter side despite severe droughts in the early 2000s, 2012 and 2018. The precipitation record also shows substantial interannual variability—varying between −20 percent to +20 percent about the long-term average. Examination of drought indices such as the Palmer Drought Severity Index (PDSI) and the Standardized Precipitation Evapotranspiration Index (SPEI) between 1900 and 2019 (fig. 3) shows that comparably extreme and even longer-lasting droughts have occurred in the past (e.g., 1930s Dust

Bowl era); however, there are indications that intensity of recent droughts, e.g., 2002, 2012, or 2018, have been greater. Consistent with increasing trends in temperature, the atmospheric freezing level (a proxy for mountain snowline) has shifted higher in elevation by 700 ft between 1948 and 2018. Atmospheric freezing level controls whether precipitation falls as snow or rain. Therefore, as it shifts higher, we expect an increase in the proportion of precipitation falling as rain rather than snow; elevation and seasonality will influence the extent to which that will occur at a given location.

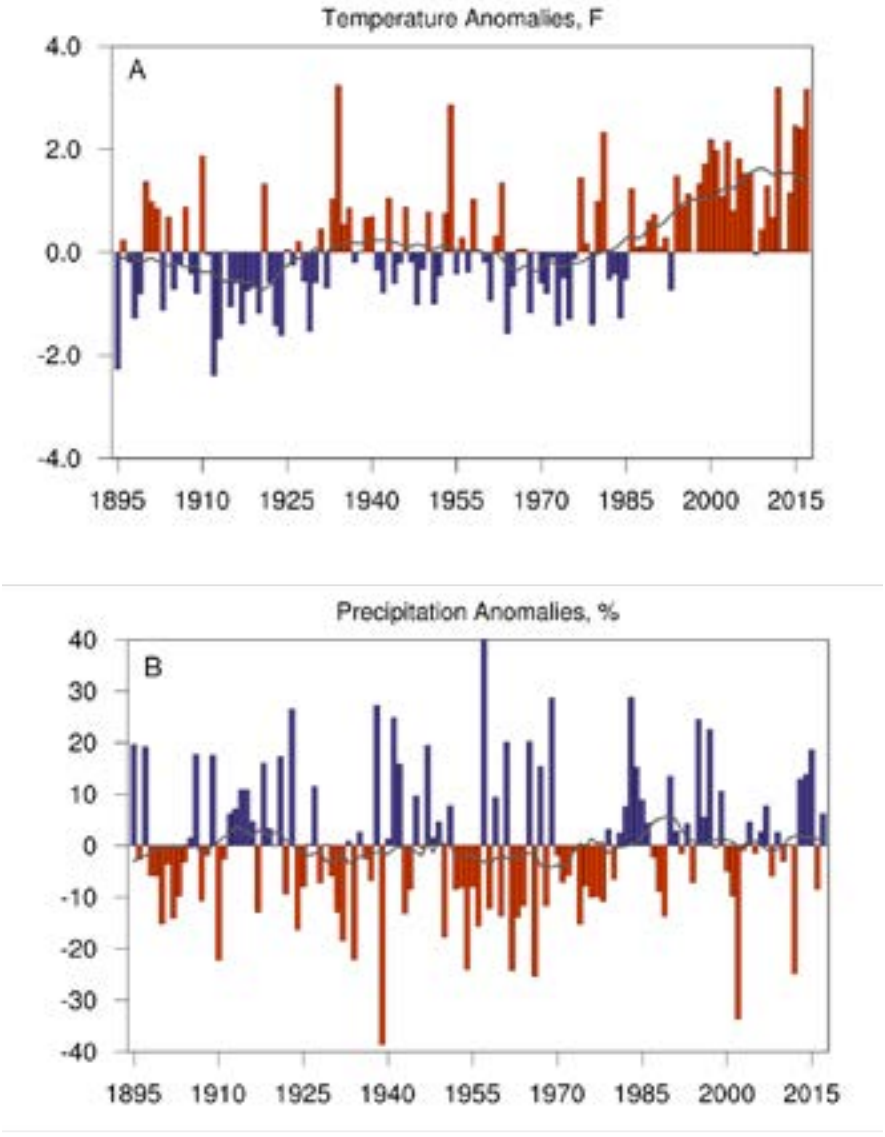


Figure 2a-b—Historical trends in (A) temperature (°F) and (B) precipitation (%) between 1895 and 2017 for the PSI-AR region bounded by 37.2-41°N latitude and 104.85-106.6°W longitude (fig. 1a). The Y-axis shows anomalies in the two variables relative to the 1901–2000 period. Bars show annual values, and the gray line depicts 10-year moving average. Dataset: 5 km monthly NOAA's NCEI nClimDiv data.

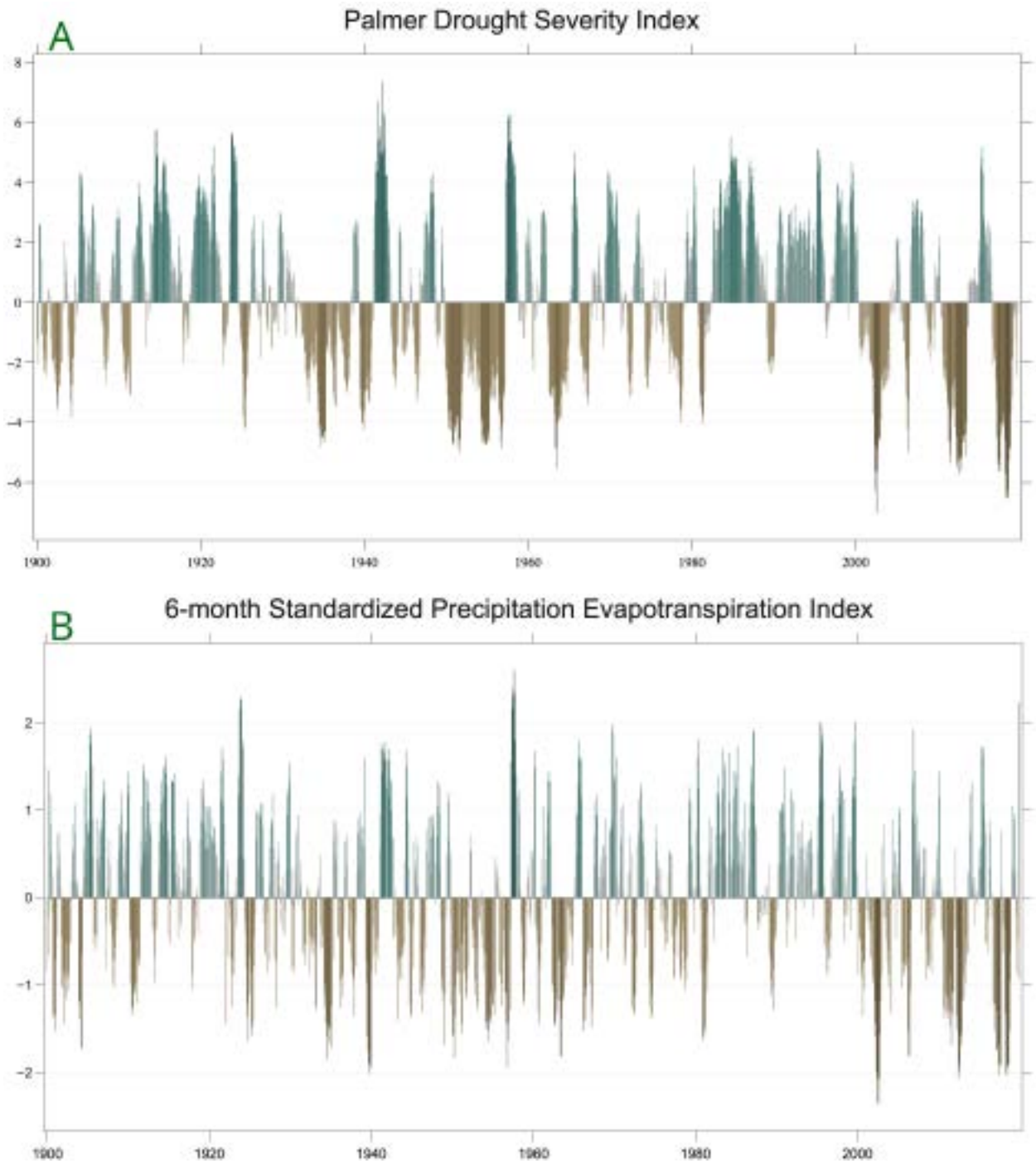


Figure 3a-b—Time series of the Palmer Drought Severity Index (PDSI) and 6-month integrated Standardized Precipitation Evapotranspiration Index (6 month-SPEI) between 1900 and 2019 for the PSI-AR region bounded by 37.2-41°N latitude and 104.85-106.6°W longitude (fig. 1a). Plots are produced using the University of Colorado's Drought Index Portal (DriP; https://droughtindexportal.colorado.edu/). Dataset: The West Wide Drought Tracker (Abatzoglou et al. 2017).

Future Climate Projections

Four future climate scenarios by 2050 were selected from a range of projected changes in annual temperature and precipitation from 34 GCMs and two emission scenarios—totaling 68 climate projections—as shown in figure 4. The primary selection criterion was to pick for divergence in the range of projections. Additional, although secondary, consideration included how a model ranks in representing historical climate (Joyce and Coulson 2020; Rupp et al. 2013). Only projections from RCP 8.5 were considered for the final selection, and because there is a high overlap between the projected changes from the two emissions scenarios by 2050 (fig. 4), this selection choice is expected to have limited influence on the findings. The four future climate scenarios identified are named:

1. **Very Hot and Dry** (model = IPSL-CM5A-MR.rcp85)
2. **Very Hot and Wet** (model = CanESM2.rcp85)
3. **Hot** (model = CNRM-CM5.rcp85)
4. **Warm and Wet** (model = GFDL-ESM2M.rcp85)

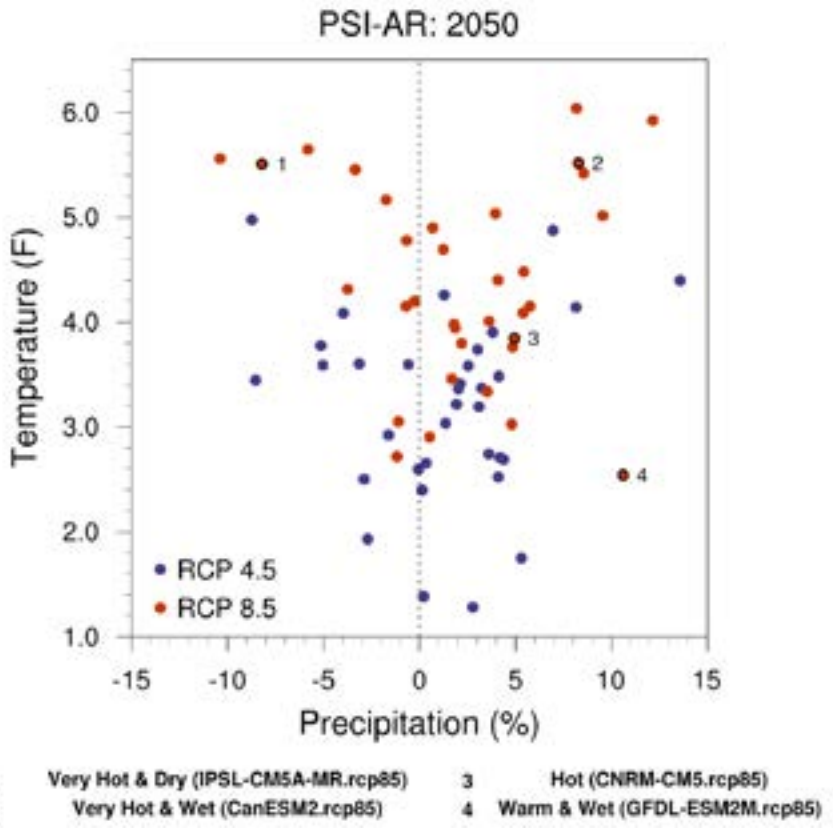


Figure 4—Selection of four future climate scenarios by 2050 based on changes in annual temperature and precipitation from 34 Global Climate Models (GCMs) and two emissions scenarios: Representative Concentration Pathway (RCP) 4.5 (moderate emissions) and RCP 8.5 (high emissions) for the PSI-AR region bounded by 37.2-41°N latitude and 104.85-106.6°W longitude (fig. 1a). Dataset: Reclamation BCSD CMIP5 1o Regridded Projection.

Table 2 provides changes in several primary climate and hydrological variables by 2050 (relative to 1985–2015) across these four climate future scenarios for the PSI-AR region, a region bounded by 37.2 °N – 41 °N latitude and 104.85 °W – 106.6 °W longitude. A summary of these changes includes:

1. Temperature increases by 2.5 to 5.5°F
2. Precipitation changes between -10 percent and +10 percent
3. Summer soil moisture decreases in all scenarios, except Warm and Wet for which it increases slightly; for the two Very Hot scenarios, it decreases by 10 percent
4. Snowpack decreases in all scenarios; for three scenarios the decreases are between -15 percent and -40 percent
5. Annual runoff changes between +10 percent and -30 percent
6. Spring runoff increases substantially in all scenarios between +20 percent and +80 percent
7. Summer and fall runoff decrease in all scenarios; summer decreases are between -10 percent and -50 percent
8. Potential evapotranspiration (a measure of drought and heat stress from a drier atmosphere) increases between 5 percent and 10 percent

These changes are also quantified separately for the region under PSI and AR boundaries in tables A2 and A3 in the Appendix for this chapter.

Table 3 describes changes in several ecologically sensitive climate variables by 2050 (relative to 1971–2000 period) across the four climate scenarios at a point location (39.5 °N, 106 °W, figure 1b; mean grid cell elevation = 10,895 ft) in the PSI-AR region. Main summary of these changes includes:

1. Coldest winter day warms by 3 to 10 °F
2. Hottest summer day warms by 4 to 8 °F
3. First fall freeze occurs later by 3 to 5 weeks
4. Last spring freeze occurs earlier by 2 to 3 weeks, except for the Warm and Wet scenario, where no change occurs
5. Growing season increases by 3 to 8 weeks
6. Freezing level (mountain snowline) shifts uphill by 1,000 to 2,000 ft
7. Severe droughts, such as in 2002 or 2012, occur every 2 to 3 years, except for the Warm and Wet scenario, where they do not occur
8. “High” to “extreme” fire danger days increase by several days and weeks, particularly in the Very Hot and Dry scenario; for the Warm and Wet they either decrease or do not change
9. Extreme rainfall increases by 10 to 20 percent with higher frequency of those events in the wetter scenarios

Table 2—Climate change in PSI-AR by 2050 under the four climate scenarios. These changes are estimated for a rectangular region surrounding PSI-AR and bounded by 37.2-41oN latitude and 104.85-106.6oW longitude (see fig. 1a). Dataset: Reclamation BCSD CMIP5 Climate and Hydrology Projection. (DJF = December, January, February; MAM = March, April, May; JJA = June, July, August; and SON = September, October, November.)

Climate metric	Annual/ Seasonal	Scenarios (GCMs) Very Hot and Wet (CanESM2.rcp85)	Hot (CNRM-CM5.rcp85)	Warm and Wet (GFDL-ESM2M.rcp85)	Very Hot and Dry (IPSL-CM5A_MR.rcp85)
Average temperature	Annual, °F	5.5	3.8	2.5	5.5
	Winter, DJF, °F	5.8	4.6	2.7	5.3
	Spring, MAM, °F	6.6	3.0	3.1	5.0
	Summer, JJA, °F	4.8	4.0	1.3	6.0
	Fall, SON, °F	4.8	3.8	3.1	5.7
Precipitation	Annual, %	8.3	5.0	10.6	-8.2
	Winter, DJF, %	12.8	2.8	5.9	0.2
	Spring, MAM, %	0.7	3.8	10.0	-13.8
	Summer, JJA, %	13.5	6.6	16.7	-16.3
	Fall, SON, %	7.3	5.0	1.4	-2.8
Minimum temperature	Annual, °F	5.8	3.9	2.3	6.0
	Winter, DJF, °F	6.9	5.0	3.0	6.4
	Spring, MAM, °F	6.5	3.3	3.1	5.4
	Summer, JJA, °F	4.9	3.8	0.8	6.3
	Fall, SON, °F	4.8	3.7	2.4	6.0
Maximum temperature	Annual, °F	5.4	3.6	2.9	6.0
	Winter, DJF, °F	4.7	3.8	2.4	7.5
	Spring, MAM, °F	7.0	2.8	3.2	5.2
	Summer, JJA, °F	5.0	3.8	2.2	5.8
	Fall, SON, °F	5.1	4.1	3.9	5.7
Soil moisture	Annual, %	-1.0	1.2	0.4	-7.9
	Winter, DJF, %	0.4	2.2	-0.3	-4.6
	Spring, MAM, %	6.5	5.8	2.6	-1.9
	Summer, JJA, %	-8.1	-1.0	-0.2	-13.5
	Fall, SON, %	-2.6	-2.3	-0.9	-11.5
Snow water equivalent (SWE)	Annual, %	-26.5	-17.3	-6.5	-38.8
	Winter, DJF, %	-14.8	-22.0	-3.6	-29.3
	Spring, MAM, %	-24.5	-11.0	-3.2	-39.3
	Summer, JJA, %	-81.5	-45.0	-40.6	-78.6
	Fall, SON, %	-63.5	-35.2	-43.7	-31.9

Table 2 continued.

Climate metric	Annual/ Seasonal	Scenarios (GCMs) Very Hot and Wet (CanESM2.rcp85)	Hot (CNRM-CM5.rcp85)	Warm and Wet (GFDL-ESM2M.rcp85)	Very Hot and Dry (IPSL-CM5A_MR.rcp85)
Runoff	Annual, %	1.1	4.5	9.9	-27.2
	Winter, DJF, %	12.3	7.3	3.0	-10.8
	Spring, MAM, %	80.4	42.4	50.1	14.3
	Summer, JJA, %	-45.6	-15.4	-12.0	-52.4
	Fall, SON, %	-14.6	-13.3	-1.4	-43.1
Evapotranspiration	Annual, %	14.4	7.3	11.6	-3.2
	Winter, DJF, %	5.8	1.6	1.1	5.1
	Spring, MAM, %	27.1	12.4	16.3	8.9
	Summer, JJA, %	7.5	6.5	11.3	-10.6
	Fall, SON, %	22.7	4.4	10.7	-1.2
Potential evapotranspiration	Annual, %	9.4	5.8	4.2	12.5
	Winter, DJF, %	11.1	9.7	5.2	22.3
	Spring, MAM, %	13.4	4.1	5.7	11.9
	Summer, JJA, %	7.4	5.0	1.5	11.0
	Fall, SON, %	6.9	7.8	7.1	11.3

Table 3—Changes in selective climate extremes under the four climate scenarios by 2050 for a point location in the Colorado Front Range forests at 39.5°N, 106°W and elevation of 10,895 ft (see fig. 1b).

Climate Metric	Very Hot and Wet	Hot	Warm and Wet	Very Hot and Dry	Historical
Coldest Winter Day (°F)	-7	-9	-12	-5	-15
Hottest Summer Day (°F)	80	77	77	81	73
First Fall Freeze	Sep 28	Sep 20	Sep 16	Sep 25	Aug 25
Last Spring Freeze	May 31	Jun 3	Jun 16	May 29	Jun 17
Growing Season Length (# days)	116	109	92	119	69
Growing Degree Days (°F; 32°F base)	4,190	3,690	3,537	4,267	2,643
Freezing Level or Snowline, ft	10,400	9,600	9,360	10,250	8,400
Frequency of Severe Droughts like 2012	every 3 years	every 5 years	do not occur	every 2 years	—
“High” Fire Danger Days	76	87	61	121	73
“Very High” Fire Danger Days	45	43	32	74	36
“Extreme” Fire Danger Days	17	17	11	36	11
Extreme Rainfall, i.e., % increase in short duration extreme value	18	14	7	21	—

(—) indicates no data.

Above values are based on MACA metdata v2 (4km downscaled climate projections) and gridMET (4km observed climate data for Historical, which is the 1971-2000 time period). Drought severity is estimated using Forest Drought Stress Index (FDSI). Freezing level (FL) changes are estimated based on historical relationship between Oct-May FL and temperature. Extreme rainfall risk is based using a scaling factor of 3.5% increase with every 1°F increase in temperature. Growing Degree Days are an estimation of heat accumulation during the growing season calculated by adding daily values of diurnal-average temperature exceedance above a certain threshold (here 32°F) over a year.

Caveats

Consideration of results from multiple climate scenarios and for different geographical boundaries (as done in fig. 1) provides us with a range of future climate and hydrological conditions that differ based on the combination of those choices. Consideration of multiple scenarios allows us to incorporate the large range of plausible climate change and impacts in our study but they are not an exhaustive consideration of all possible outcomes. Specific consideration of results related to a specific geographical choice is also relevant. For example, if one is primarily interested in understanding impacts for the AR region then the consideration of results described in table A3 are more appropriate than those in table 2, which are for a larger region. Furthermore, the results associated with a point location (as shown in table 3) are also appropriate to extrapolate to a different region if one ignores the absolute values but applies the change factors (changes between historical and future periods) quantified for this location to the historical values of another region. With the exception of "growing degree days," change factors from all other variables in table 3 could be extrapolated to another location in the Front Range forests to understand the nature of regional-scale change. And lastly, because of large topographical differences across the Front Range forests, it is also important to recognize how hydrological response such as soil moisture and runoff will vary across the landscape at any given time as shown in figures A1–A7.

References

Abatzoglou, J. T. 2013. Development of gridded surface meteorological data for ecological applications and modelling. International Journal of Climatology. 33: 121–131. https://doi.org/10.1002/joc.3413

Abatzoglou J.T.; Brown T.J. 2012. A comparison of statistical downscaling methods suited for wildfire applications, International Journal of Climatology. 32: 772-780. https://doi.org/10.1002/joc.2312

Abatzoglou, J.T.; McEvoy, D.J.; Redmond, K.T. 2017. The west wide drought tracker: Drought monitoring at fine spatial scales. Bulletin of the American Meteorological Society. 98(9): 1815–1820. https://doi.org/10.1175/BAMS-D-16-0193.1

Joyce, L.A.; Coulson, D. 2020. Climate scenarios and projections: A technical document supporting the USDA Forest Service 2020 RPA Assessment. Gen. Tech. Rep. RMRS-GTR-413. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 85 p. https://doi.org/10.2737/RMRS-GTR-413

Oyler, J.W.; Ballantyne A.; Jencso K.; Sweet M.; Running S.W. 2014. Creating a topoclimatic daily air temperature dataset for the conterminous United States using homogenized station data and remotely sensed land skin temperature. International Journal of Climatology. 35:2258-2279. https://doi.org/10.1002/joc.4127

Rupp, D.E.; Abatzoglou, J.T.; Hegewisch, K.C. [et al.]. 2013. Evaluation of CMIP5 20th century climate simulations for the Pacific Northwest USA. Journal of Geophysical Research: Atmospheres. 118(19): 10–884. https://doi.org/10.1002/jgrd.50843

U.S. Department of the Interior, Bureau of Reclamation [Reclamation]. 2014. Downscaled CMIP3 and CMIP5 climate and hydrology projections: Release of hydrology projections, comparison with preceding information, and summary of user needs. Denver, CO: U.S. Department of the Interior, Bureau of Reclamation, Technical Services Center. 110 pp. https://gdo-dcp.ucllnl.org/downscaled_cmip_projections/techmemo/BCSD5HydrologyMemo.pdf

Vose, Russell S.; Applequist, S.; Squires, M.; Durre, I.; Menne, M. J.; Williams, C. N., Jr.; Fenimore, C.; Gleason, K.; Arndt, D. 2014. NOAA Monthly U.S. Climate Divisional Database (NClimDiv). NOAA National Climatic Data Center. https://doi.org/10.7289/V5M32STR

Williams, A.P.; Allen, C.D.; Macalady, A.K.; [et al.]. 2013. Temperature as a potent driver of regional forest drought stress and tree mortality. Nature Climate Change. 3(3): 292. https://doi.org/10.1038/nclimate1693

Appendix A

Table A1—List of the 68 Global Climate Model (GCM) projections considered in the study to select the future climate scenarios.

1. access1-0.1.rcp45
2. access1-0.1.rcp85
3. access1-3.1.rcp45
4. access1-3.1.rcp85
5. bcc-csm1-1.1.rcp45
6. bcc-csm1-1.1.rcp85
7. bcc-csm1-1-m.1.rcp45
8. bcc-csm1-1-m.1.rcp85
9. canesm2.1.rcp45
10. canesm2.1.rcp85
11. ccsm4.1.rcp45
12. ccsm4.1.rcp85
13. cesm1-bgc.1.rcp45
14. cesm1-bgc.1.rcp85
15. cesm1-cam5.1.rcp45
16. cesm1-cam5.1.rcp85
17. cmcc-cm.1.rcp45
18. cmcc-cm.1.rcp85
19. cnrm-cm5.1.rcp45
20. cnrm-cm5.1.rcp85
21. csiro-mk3-6-0.1.rcp45
22. csiro-mk3-6-0.1.rcp85
23. ec-earth.2.rcp45
24. ec-earth.6.rcp85
25. fgoals-g2.1.rcp45
26. fgoals-g2.1.rcp85
27. fgoals-s2.2.rcp45
28. fgoals-s2.2.rcp85
29. fio-esm.2.rcp45
30. fio-esm.2.rcp85
31. gfdl-cm3.1.rcp45
32. gfdl-cm3.1.rcp85
33. gfdl-esm2g.1.rcp45
34. gfdl-esm2g.1.rcp85
35. gfdl-esm2m.1.rcp45
36. gfdl-esm2m.1.rcp85
37. giss-e2-r.1.rcp45
38. giss-e2-r.1.rcp85
39. hadgem2-ao.1.rcp45
40. hadgem2-ao.1.rcp85
41. hadgem2-cc.1.rcp45
42. hadgem2-cc.1.rcp85
43. hadgem2-es.1.rcp45
44. hadgem2-es.1.rcp85
45. inmcm4.1.rcp45
46. inmcm4.1.rcp85
47. ipsl-cm5a-lr.1.rcp45
48. ipsl-cm5a-lr.1.rcp85
49. ipsl-cm5a-mr.1.rcp45
50. ipsl-cm5a-mr.1.rcp85
51. ipsl-cm5b-lr.1.rcp45
52. ipsl-cm5b-lr.1.rcp85
53. miroc-esm.1.rcp45
54. miroc-esm.1.rcp85
55. miroc-esm-chem.1.rcp45
56. miroc-esm-chem.1.rcp85
57. miroc5.1.rcp45
58. miroc5.1.rcp85
59. mpi-esm-lr.1.rcp45
60. mpi-esm-lr.1.rcp85
61. mpi-esm-mr.1.rcp45
62. mpi-esm-mr.1.rcp85
63. mri-cgcm3.1.rcp45
64. mri-cgcm3.1.rcp85
65. noresm1-m.1.rcp45
66. noresm1-m.1.rcp85
67. noresm1-me.1.rcp45
68. noresm1-me.1.rcp85

Table A2—Climate change in Pike and San Isabel (PSI) National Forests by 2050 under the four climate scenarios. These changes are estimated for the region under the PSI National Forest boundaries. Dataset: Reclamation BCSD CMIP5 Climate and Hydrology Projection. (DJF = December, January, February; MAM = March, April, May; JJA = June, July, August; and SON = September, October, November.)

Climate metric	Annual/ Seasonal	Scenarios (GCMs) Very Hot and Wet (CanESM2.rcp85)	Hot (CNRM-CM5.rcp85)	Warm and Wet (GFDL-ESM2M.rcp85)	Very Hot and Dry (IPSL-CM5A_MR.rcp85)
Average temperature	Annual, °F	5.6	3.8	2.6	5.5
	Winter, DJF, °F	5.9	4.6	2.8	5.3
	Spring, MAM, °F	6.7	3.0	3.3	5.0
	Summer, JJA, °F	4.8	3.8	1.1	6.0
	Fall, SON, °F	4.8	3.7	3.0	5.7
Precipitation	Annual, %	6.8	6.0	10.5	-10.1
	Winter, DJF, %	11.9	3.3	5.8	-0.3
	Spring, MAM, %	-2.3	3.2	8.9	-13.7
	Summer, JJA, %	13.6	8.8	17.3	-19.7
	Fall, SON, %	5.2	6.3	1.5	-6.9
Minimum temperature	Annual, °F	5.4	4.5	2.4	6.1
	Winter, DJF, °F	6.1	5.7	2.7	6.7
	Spring, MAM, °F	5.5	3.8	2.4	5.5
	Summer, JJA, °F	5.1	4.4	1.7	6.5
	Fall, SON, °F	4.8	4.1	2.7	5.7
Maximum temperature	Annual, °F	5.3	4.4	2.8	5.8
	Winter, DJF, °F	4.9	4.6	2.4	6.9
	Spring, MAM, °F	6.1	3.4	2.4	5.3
	Summer, JJA, °F	5.4	4.8	2.3	5.9
	Fall, SON, °F	4.8	4.5	4.0	5.3
Soil moisture	Annual, %	-0.4	0.2	1.2	-10.4
	Winter, DJF, %	1.2	1.0	0.6	-8.2
	Spring, MAM, %	5.4	3.7	2.4	-5.6
	Summer, JJA, %	-7.5	-1.3	0.9	-13.4
	Fall, SON, %	-0.4	-2.7	0.9	-14.4
Snow water equivalent (SWE)	Annual, %	-20.3	-10.9	-2.2	-28.0
	Winter, DJF, %	-10.2	-15.7	-1.2	-18.7
	Spring, MAM, %	-13.5	-2.9	2.2	-26.0
	Summer, JJA, %	-76.9	-38.7	-28.2	-69.6
	Fall, SON, %	-53.8	-19.8	-33.7	-18.9

Table A2 continued.

Climate metric	Annual/ Seasonal	Scenarios (GCMs) Very Hot and Wet (CanESM2.rcp85)	Hot (CNRM-CM5.rcp85)	Warm and Wet (GFDL-ESM2M.rcp85)	Very Hot and Dry (IPSL-CM5A_MR.rcp85)
Runoff	Annual, %	-1.8	4.8	10.9	-30.6
	Winter, DJF, %	16.5	9.7	8.8	-27.9
	Spring, MAM, %	109.3	50.0	49.0	17.5
	Summer, JJA, %	-51.6	-14.9	-10.3	-52.8
	Fall, SON, %	-5.9	-7.5	9.8	-57.0
Evapotranspiration	Annual, %	15.4	8.0	11.9	-2.2
	Winter, DJF, %	11.0	4.1	3.1	8.2
	Spring, MAM, %	22.1	8.8	15.0	4.5
	Summer, JJA, %	10.8	8.5	11.4	-6.5
	Fall, SON, %	24.4	6.7	13.1	-1.4
Potential evapotranspiration	Annual, %	9.6	6.1	4.5	13.1
	Winter, DJF, %	12.1	11.0	6.2	22.5
	Spring, MAM, %	14.3	4.7	6.0	12.8
	Summer, JJA, %	7.2	4.8	1.9	11.5
	Fall, SON, %	6.8	8.0	7.0	11.6

Table A3—Climate change in Arapaho and Roosevelt (AR) National Forests by 2050 under the four climate scenarios. These changes are estimated for the region under the AR boundaries. Dataset: Reclamation BCSD CMIP5 Climate and Hydrology Projection. (DJF = December, January, February; MAM = March, April, May; JJA = June, July, August; and SON = September, October, November.)

Climate metric	Annual/ Seasonal	Scenarios (GCMs)			
		Very Hot and Wet (CanESM2.rcp85)	Hot (CNRM-CM5.rcp85)	Warm and Wet (GFDL-ESM2M.rcp85)	Very Hot and Dry (IPSL-CM5A_MR.rcp85)
Average temperature	Annual, °F	5.3	4.1	2.5	5.5
	Winter, DJF, °F	5.5	4.8	2.2	5.3
	Spring, MAM, °F	6.4	3.0	2.4	4.9
	Summer, JJA, °F	4.7	4.6	2.0	6.0
	Fall, SON, °F	4.7	4.1	3.3	5.6
Precipitation	Annual, %	12.8	0.3	9.7	-1.5
	Winter, DJF, %	18.4	3.6	6.7	2.7
	Spring, MAM, %	9.3	5.8	13.5	-12.7
	Summer, JJA, %	12.2	-5.3	11.8	-6.5
	Fall, SON, %	15.3	1.1	0.0	13.6
Minimum temperature	Annual, °F	5.6	4.1	2.8	6.0
	Winter, DJF, °F	6.4	5.3	2.7	6.2
	Spring, MAM, °F	6.3	3.3	2.9	5.9
	Summer, JJA, °F	4.7	4.1	2.4	6.2
	Fall, SON, °F	4.8	3.8	3.1	5.8
Maximum temperature	Annual, °F	5.2	4.0	2.2	5.7
	Winter, DJF, °F	4.5	3.9	1.6	7.2
	Spring, MAM, °F	6.7	2.8	1.9	4.2
	Summer, JJA, °F	4.9	4.9	1.7	5.6
	Fall, SON, °F	4.8	4.5	3.7	5.6
Soil moisture	Annual, %	1.5	-1.1	1.1	-5.7
	Winter, DJF, %	3.3	-0.5	-0.7	0.9
	Spring, MAM, %	15.9	7.7	4.7	6.7
	Summer, JJA, %	-8.5	-3.6	1.5	-16.1
	Fall, SON, %	-3.6	-8.7	-1.9	-12.4
Snow water equivalent (SWE)	Annual, %	-23.8	-16.7	-3.9	-39.2
	Winter, DJF, %	-9.2	-20.0	1.6	-28.6
	Spring, MAM, %	-20.4	-10.4	-0.1	-38.1
	Summer, JJA, %	-80.3	-44.9	-43.2	-80.8
	Fall, SON, %	-64.9	-41.2	-43.9	-39.0

Table A3 continued.

Climate metric	Annual/ Seasonal	Scenarios (GCMs) Very Hot and Wet (CanESM2.rcp85)	Hot (CNRM-CM5.rcp85)	Warm and Wet (GFDL-ESM2M.rcp85)	Very Hot and Dry (IPSL-CM5A_MR.rcp85)
Runoff	Annual, %	7.3	0.7	11.7	-22.6
	Winter, DJF, %	29.5	2.9	0.4	13.0
	Spring, MAM, %	108.6	46.3	64.4	33.1
	Summer, JJA, %	-44.6	-18.5	-12.6	-53.9
	Fall, SON, %	-20.1	-28.8	-5.1	-41.0
Evapotranspiration	Annual, %	17.1	5.8	10.9	3.5
	Winter, DJF, %	7.5	3.0	-0.7	9.6
	Spring, MAM, %	33.7	13.5	15.5	19.0
	Summer, JJA, %	9.5	4.4	11.2	-4.7
	Fall, SON, %	23.9	0.6	8.5	6.5
Potential evapotranspiration	Annual, %	8.5	6.2	2.7	10.7
	Winter, DJF, %	8.2	8.2	2.0	19.9
	Spring, MAM, %	12.0	3.2	3.5	9.2
	Summer, JJA, %	7.2	6.0	0.2	9.6
	Fall, SON, %	6.5	9.3	7.1	10.1

Water Resources of the Front Range National Forests

D. Max Smith

Introduction

The Colorado Front Range is the meeting of the Rocky Mountains and the Great Plains, a nexus of great topographic, geologic, and atmospheric forces. As water collects in the mountains and moves to the deserts and plains, it fosters unique ecosystems, some of which support rare and sensitive species. This water also creates opportunities and challenges for a growing metro area, fuels the recreational activities of residents and visitors, and supports an expansive agricultural industry. A complex system of ditches, pipelines, reservoirs, and transbasin diversions has been constructed to meet these myriad uses. The management challenges of the Front Range national forests are exacerbated by a warming climate and associated changes in hydrological patterns (Brown et al. 2019). In particular, the water resources provided by the Rocky Mountains in Colorado are vulnerable to the interacting effects of rising temperatures and increasing demands (Imerzeel et al. 2019). A detailed understanding of this vulnerability is necessary to plan management activities that will benefit natural and human communities through the end of the 21st century.

For this chapter, I reviewed the literature to describe the underlying components of Front Range water resources. This information was then incorporated into an assessment of the vulnerability of key water resources under four climate change scenarios. This was completed by selecting resources for analysis and discussing exposure, sensitivity, and adaptive capacity to the four scenarios. Finally, I provided a ranking of uncertainty based on the volume of empirical evidence supporting similar effects and responses. Datasets used in this assessment are listed in the appendix.

Environmental Setting

The natural hydrology of the Front Range was shaped by geological events and long-term climatic changes. The dynamics of surface water and groundwater are also influenced by the modern climate, plant and animal communities, and the activities of an expanding human population.

Geography

The region now known as the Front Range is located on the traditional territories and homelands of the Arapaho, Apache, Cheyenne, Comanche, Ute, and many other Indigenous nations (https://native-land.ca/). Front Range lands are now managed by a number of entities including the U.S. Department of Agriculture, Forest Service. For the purposes of this report, I focus on the resources provided by the Arapaho-Roosevelt National Forests (AR) and the Pike-San Isabel National Forests (PSI). In the AR National Forests, streams flow east and west of the Continental Divide, feeding into three river basins. Streams drain into the South Platte and North Platte Basins east of the divide, and into the Upper Colorado Basin west of the divide (fig. 1). Greater than 50 percent

of the stream miles of the AR National Forests are in the South Platte Basin. In the PSI National Forests, streams drain east of the Continental Divide into the Arkansas and South Platte Basins. Greater than 50 percent of PSI National Forests stream miles are in the Arkansas Basin.

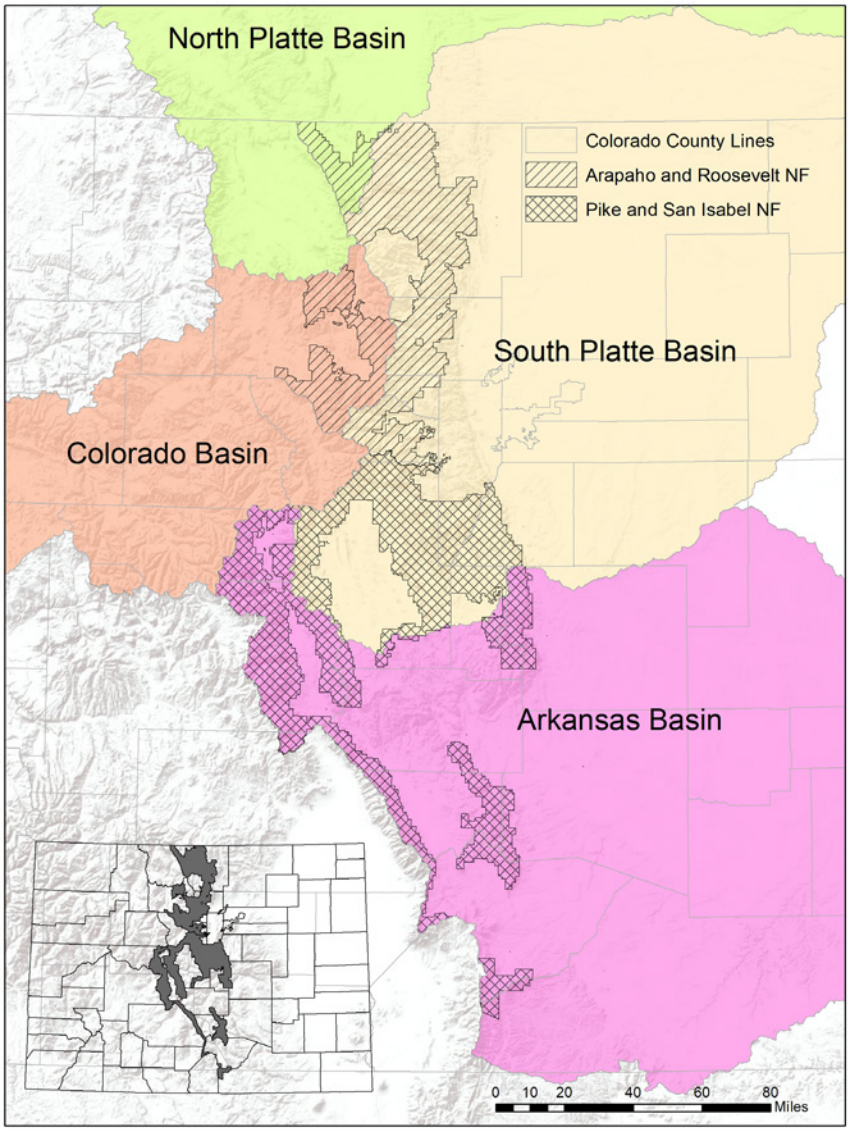


Figure 1—Location of the Front Range National Forests and major river basins.

The South Platte Basin contains 85 percent of Colorado's human population and the State's greatest concentration of irrigated cropland. Population centers in the Arkansas Basin include the cities of Colorado Springs and Pueblo. Ranching and livestock production are the dominant land uses in the portions of the North Platte and Upper Colorado Basins that are near the Front Range national forests (CWCB 2019).

For analysis purposes, the river basins are subdivided into 8-digit Hydrologic Unit Code (HUC8) watersheds that overlap the AR and PSI National Forests (Jones et al. 2022). Seven of these watersheds are in the South Platte Basin, six are in the Arkansas Basin, and there is one each in the North Platte and Upper Colorado Basins (table 1, fig. 2).

Table 1—Physical and administrative characteristics of 8-digit Hydrologic Unit Code (HUC8) watersheds overlapping the Arapaho and Roosevelt (AR) and Pike and San Isabel (PSI) National Forests. Data fields are described in the appendix.

Basin	HUC8 watershed	National Forest	Acres	Elevation range (ft)	Percent of HUC8 within Front Range NF Boundaries (%)	Private land (%)	Decreed instream flow miles
North Platte	Upper Laramie	AR	1,453,543	7,550–12,954*	65*	22*	92*
South Platte	Cache La Poudre	AR	1,209,013	4,593–13,574*	32*	52*	395*
	Big Thompson	AR	532,352	4,670–14,258	23	52	58
	St. Vrain	AR	626,865	4,736–14,257	26	51	341
	Clear	AR	364,755	5,101–14,270	38	48	127
	South Platte Headwaters	AR, PSI	1,025,992	7,217–14,287	39	44	318
	Upper South Platte	PSI	1,182,910	5,169–14,253	48	43	370
Upper Colorado	Colorado Headwaters	AR	1,856,423	5,716–13,547	22	25	822
Arkansas	Fountain	PSI	593,708	4,634–14,114	18	61	53
	Arkansas Headwaters	PSI	1,960,346	5,359–14,421	42	36	662
	Upper Arkansas	PSI	1,474,119	4,515–14,113	11	67	182
	Huerfano	PSI	1,189,513	4,440–14,330	12	79	53
	Apishapa	PSI	688,494	4,261–13,582	1	84	0
	Purgatoire	PSI	2,206,211	3,860–14,045	1	76	21

An asterisk (*) indicates that these data only apply to the portion of these watersheds within the State of Colorado.

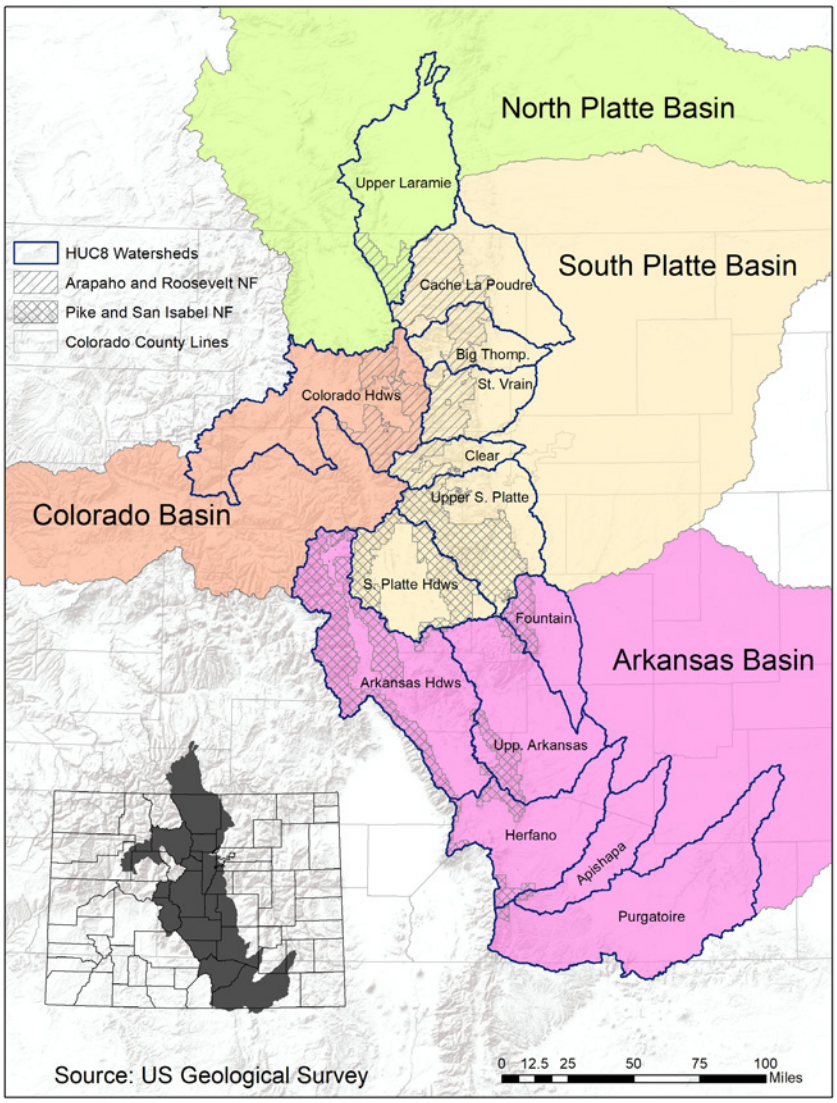


Figure 2—HUC 8 watersheds of the Front Range National Forests.

Physical Features

The AR and PSI National Forests were established on several mountain ranges that generally trend north to south. These ranges include the Medicine Bow Mountains and Front Range in northern Colorado, the Sawatch and Mosquito Ranges in the central portion of the State, and the Sangre de Cristo Mountains, Wet Mountains, and Spanish Peaks in the south. The cores and high peaks of these ranges are primarily composed of Precambrian intrusive igneous rocks and metamorphic rocks. These crystalline core complexes are flanked or overlaid by younger sedimentary rocks, including limestones, sandstones, and shales (Fields and Dethier 2019; Livers and Wohl 2015).

The upper elevations (approximately 8,000 ft and above) of the ranges were shaped by Pleistocene glaciers, which created mountain lakes, cirques, and broad valley bottoms (Caine 1995; Livers and Wohl 2015). Rock glaciers continue to influence the form and hydrology of mountains in some locations (Giardino et al. 1984; Jones et al. 2019). The lower elevations are dissected by stream-carved canyons, many of which are narrowly confined by crystalline bedrock (Clarey et al. 1999).

Climate

Snowfall typically occurs as a result of northwesterly airmass flow in the fall, winter, and early spring (Lukas et al. 2014). Storms from the south and east deliver moisture from the Gulf of Mexico to the Front Range via thunderstorms in late spring and summer. Precipitation ranges from less than 10 inches in lowland areas to greater than 40 inches in the highest mountains (Abbott 1985). Sixty percent of runoff in the Arkansas Basin occurs from April to July (Elgaali and Garcia 2007). Intense thunderstorms and multi-day precipitation events can occur from spring to fall. These storms are responsible for the largest floods on record on the eastern slope of the Front Range (Lukas et al. 2014).

Aquatic Ecosystems

Within the AR and PSI National Forests, aquatic ecosystems occur in natural settings, such as streams, ponds, and mountain lakes. Aquatic ecosystems also occur in anthropogenic features such as reservoirs, stock ponds, and irrigation ditches. After leaving the forests, surface flows sustain aquatic ecosystems in streams and reservoirs. Aquatic ecosystems contain plant and animal species that spend their entire lives in water, as well as animals that spend only certain stages of their life cycles in water. Specialized aquatic communities include amphibian larvae and benthic macroinvertebrates such as insects, snails, worms, and crayfish (Rice et al. 2018). Coldwater fish communities include the federally threatened greenback cutthroat trout (*Oncorhynchus clarkii stomias*) and the State-listed Colorado River cutthroat trout (*Oncorhynchus clarkii pleuriticus*; box 1). Introduced sportfish such as brook trout (*Salvelinus fontinalis*), rainbow trout (*Oncorhynchus mykiss*), and brown trout (*Salmo trutta*) are abundant in streams throughout the forests (Bestgen et al. 2019a; Naesser and Smith 1995; Richer et al. 2019). Lake trout (*Salvelinus namaycush*), kokanee salmon (*Oncorhynchus nerka*), and other nonnative sportfish are managed in lakes and reservoirs to provide recreational opportunities (Nesler 1981; Walch 1980). Downstream of the forests, surface flows support habitat for warm water fishes, including the Colorado pikeminnow (*Ptychocheilus lucius*), humpback chub (*Gila cypha*), and razorback sucker (*Xyrauchen texanus*), all of which are federally endangered (ARP 1997).

Box 1

Status of Cutthroat Trout Populations

South Platte Basin

The greenback cutthroat trout is a federally threatened salmonid native only to the South Platte Basin. The only known, genetically confirmed, strain was recently discovered in the Bear Creek drainage of the Arkansas Basin, where they were stocked in the late 1800s (Bestgen et al. 2019a). Since 2014, this strain has been reintroduced to stream segments in the South Platte Basin that are protected by barriers to invasion by nonnative salmonids. Populations in these segments are heavily managed and protected.

Arkansas Basin

There are currently no subspecies of cutthroat trout in Arkansas Basin that are considered native. However, at least one subspecies naturally occurred in the basin prior to Euro-American settlement and there are several extant populations representing unique genetic lineages (Janelle Valladares, USDA Forest Service, personal communication). The yellowfin cutthroat trout was a large subspecies that was extirpated from its range in the Twin Lakes area. The Bear Creek population of greenback cutthroat trout originated with fish stocked from the South Platte Basin in the late 1800s. Recent molecular analyses suggest that populations of the green lineage of Colorado River cutthroat are present east of the Continental Divide and may be native to the Arkansas Basin (Bestgen et al. 2019a). Another population, genetically similar to the yellowfin but classified as the Colorado River subspecies, has been found in Hayden Creek and is being reintroduced into suitable streams. Federal status of cutthroat trout in Arkansas Basin will likely be reexamined to incorporate results from recent molecular analyses (Rogers 2012).

Upper Colorado Basin

Populations of green lineage Colorado River cutthroat trout are native to streams in the basin, including a section of the AR that lies west of the Continental Divide. This subspecies is listed as a sensitive species by the Forest Service and the State of Colorado. A multi-state conservation agreement is in place to protect this subspecies (CRCT Coordination Team 2006).

Riparian Areas, Wetlands, and Groundwater-Dependent Ecosystems

In addition to aquatic habitats, Front Range water sustains riparian areas, wetlands, and groundwater-dependent ecosystems (box 2). Riparian ecosystems that occur along streams are influenced by geological setting, watershed hydrology, and biotic communities (Castro and Thorne 2019). Low-gradient, unconfined valley bottoms contain extensive riparian ecosystems, which can develop into mosaics of vegetation in various seral stages and growth forms (Livers and Wohl 2015). Confined stream reaches, often found in bedrock canyons, support smaller riparian areas, which may be difficult to distinguish from adjacent uplands (Polvi et al. 2011). Partially confined reaches contain features of both canyons and low-gradient, unconfined valley bottoms. Groundwater-dependent ecosystems (GDEs) occur where groundwater discharges to the surface or near the surface, supporting unique communities of plants and animals. GDEs in the AR and PSI National Forests include springbrooks, spring pools, fens, and other wetlands.

Box 2

Definitions of Riparian and Wetland Ecosystems

Authors use various criteria to describe riparian ecosystems, but their definitions typically contain at least one of these components: (1) an interface of terrestrial and aquatic communities at a body of water; (2) landforms and organisms that are influenced by surface water and groundwater; and/or (3) plant communities near water that differ in composition and/or structure from those in adjacent upland ecosystems (Dall et al. 1997; Gregory et al. 1991).

The Federal regulatory definition of wetlands is "ecosystems that are inundated or saturated by surface or groundwater at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions." This and other definitions of wetlands allows for some overlap with riparian ecosystems.

Groundwater-dependent ecosystems are defined by the Forest Service as "communities of plants, animals, and other organisms whose extent and life processes are dependent on access to or discharge of groundwater." Springs, fens, and certain riparian areas are examples of groundwater-dependent ecosystems. Springs are sites where groundwater is discharged or exposed at the soil surface. Unlike riparian ecosystems, spring ecosystems are entirely supported by groundwater. There are several types of springs in the Front Range forests, each of which can support unique flora and fauna (Springer and Stevens 2009; Stevens and Meretsky 2008).

Fens are groundwater-dependent wetlands with saturated organic soils that have accumulated a minimum of 30 to 40 cm of peat. These conditions support unique plant communities that include many species typically associated with higher latitudes (Bedford and Godwin 2003; Chadde et al. 1998; USDA FS 2007; 2012).

Water Uses and Infrastructure

Humans have long shaped the hydrology of the Front Range watersheds, with influences accelerating following Euro-American settlement and culminating in today's complex system of reservoirs, wells, ditches, and pipelines. It is often stated that most of Colorado's population lives east of the Rocky Mountains, yet most of the State's water flows west of the Continental Divide (Jones and Cech 2009). This distribution of people and water has resulted in the alteration of the hydrology of streams and lakes throughout the AR and PSI National Forests. The Cache La Poudre River, Big Thompson River, and other streams in the South Platte Basin are augmented by transbasin diversions, but these streams are heavily depleted by agricultural and municipal uses before reaching their confluences (Bestgen et al. 2019b). The Arkansas River is augmented by transbasin diversions as well, with surface flows heavily manipulated by dams and diversions (Abbott 1985).

Irrigation canals have been used to divert water from streams to farms since the 1850s or earlier (Abbott 1985). Average-year surface flows were fully appropriated on the eastern slope of the Front Range prior to surface flows on the western slope being fully appropriated, creating the need and opportunity for transbasin diversions from the west to the east. These include the Colorado–Big Thompson Project, the Frying Pan–

Arkansas Project, The Twin Lakes Project, and several smaller tunnels and ditches that cross the Continental Divide. These projects provide water for municipal, agricultural, and other consumptive uses on the eastern slope, as well as water for energy generation and recreational activities (Naesser and Smith 1995; Nessler 1981; Walch 1980). As a result of water releases timed to meet these needs, the natural flow regime (timing and volume) of many streams has been altered. In general, west side water is stored in reservoirs, transported to the east side through tunnels, stored in east side reservoirs, and transported through natural stream channels, augmenting native flows in the process.

Water Resource Components

Based on the above literature review (conducted in 2020) and feedback from managers at the November 2019 vulnerability workshop, I selected seven water resource components for further analysis. These components are described below, along with climate change implications specific to the Front Range national forests (table 2).

Table 2—Climate change implications for water resource components, obtained from literature review.

Water Resource Component	Implication
1. Streamflow	• Atmospheric warming decreases the contribution of snowmelt to surface flows (Ficklin et al. 2013; Johnson et al. 2015; Kampf and Lefsky 2016; Perry et al. 2019; Rasmussen et al. 2014; Zhang et al. 2018). • In the snow/rain transition zones, peak flows result from rain instead of snowmelt (Kampf and Lefsky 2016; Zhang et al. 2018). • Increasing temperatures and decreasing winter precipitation lead to earlier peak runoff in snowmelt-dominated streams (Clow 2010). • Atmospheric warming decreases the amount of water leaving the forests in streamflows as a result of increased rates of evapotranspiration (Ficklin et al. 2013; Foster et al. 2016; Rasmussen et al. 2014). • Combined effects of drought, bark beetles, and dust on snow result in earlier snowmelt and peak discharge in mountain streams (Deems et al. 2013; Livneh et al. 2015).
2. Snowpack	• Change in precipitation type at middle elevations (6,500–8,000 feet) will shift the transition zone upwards causing the lower elevation of the snowpack zone to shift upwards (Kampf and Lefsky 2016; Rasmussen et al. 2011, 2014; Zhang et al. 2018). • Change in vegetation cover following drought, bark beetles, and fire influences snowpack characteristics and infiltration rates below canopies (Harpold et al. 2015; Livneh et al. 2015).

Table 2 continued.

Water Resource Component	Implication
	• Change in lowland vegetation cover following drought increases atmospheric dust, which influences snowpack behavior (Deems et al. 2013). • Dust on snow interacts with climatic warming to decrease the ability for snowpack to act as water storage throughout the growing season (Painter et al. 2018). • Decreasing winter precipitation and increasing temperatures result in a trend of decreasing snow water equivalent (SWE) in Colorado (Clow 2010).
3. Groundwater	• Change from snow to rain at upper elevations likely decreases groundwater recharge, altering transit times and chemistry of water discharged at springs (Earman and Dettinger 2011; Fields and Dethier 2019). • Growing demand for groundwater may decrease its availability for aquatic, riparian, and wetland ecosystems (Kundzewicz and Doll 2009).
4. Drought	• Higher temperatures increase the likelihood and severity of drought (Breshears et al. 2005; Udall and Overpeck 2017). • Periods of drought result in water shortages in Front Range basins. Future long-term drought may tax storage and diversion infrastructure (Kenney et al. 2004; Salas et al. 2005).
5. Extreme Rainfall Events	• Atmospheric moisture increases with atmospheric warming, but there is uncertainty about projected changes in frequency and intensity of rain events (Mahoney et al. 2013). • Post-wildfire sites are prone to erosion, heavy flooding, and debris flows during extreme rainfall events (Lukas et al. 2014).
6. Water Quality	• Atmospheric warming results in increased stream temperatures, with consequences for aquatic ecosystems (Isaak et al. 2015; Roberts et al. 2017a). • Drought decreases the dilution of pollutants in streams, lakes, reservoirs, and groundwater (Johnson et al. 2015; Saunders and Lewis 2003; Sprague et al. 2005). • Atmospheric warming increases the risk of wildfire, which reduces vegetation cover, increasing the risk of debris flows and sedimentation (Rice et al. 2018).
7. Water Use Infrastructure	• Evaporative losses from reservoirs reduce the availability of surface flows and water uses (Naeser and Smith 1991). • Aging infrastructure is at risk of damage from floods and fires (Lukas et al. 2014). • Aging infrastructure is not capable of meeting challenges from increasing demands and a changing climate (Dettinger et al. 2015).

Streamflows

Surface water in streams is habitat for native fish and other aquatic organisms. The movement of this water influences streambed, streambank, and floodplain habitat features. Surface flows also contribute to groundwater and soil moisture, helping to sustain riparian and wetland ecosystems (Bestgen et al. 2019b). Aquatic and riparian organisms have traits that are adapted to the natural flow regimes of Front Range streams. Therefore, changes to the timing, magnitude, and frequency of high flows and low flows have the potential to alter ecosystem structure (Poff et al. 1997). Under typical conditions, surface flows originate as runoff from snowmelt, runoff from rain, or from groundwater discharge (Williams et al. 2015a). Runoff in large streams is usually snowmelt-dominated, with peak flows occurring in May or June (table 3; fig. 3) (Bartholow 2010; Lukas et al. 2014). Groundwater and rain have greater contributions to streamflows at lower elevations (Kampf and Lefsky 2016; Livers and Wohl 2015). Peak flows from rain-produced floods are typically more severe than those arising from snowmelt (fig. 3) (Jarrett and Costa 1988). Variation in magnitude and timing of peak flows is generally greater at lower elevations, as a result of these rainfall events (table 2). By decreasing interception and infiltration of precipitation, wildfire increases the amount of runoff from storms, magnifying peak flows for several years postburn (Moody and Martin 2001).

Low flows typically occur in late summer through winter and are sustained by persistent snowmelt, groundwater discharge, and in some areas by melting ice glaciers and rock glaciers (Williams et al. 2006). Because numerous factors determine the amount of precipitation that enters streams, year-to-year variation in streamflow is greater than that of precipitation (Lukas et al. 2014). Within and downstream of the AR and PSI National Forests, streamflows are diverted for a variety of anthropogenic uses, including municipal, agricultural, and industrial.

Table 3—Streamflow characteristics at select gages in 8-digit Hydrologic Unit Code (HUC8) watersheds of the Arapaho and Roosevelt (AR) and Pike and San Isabel (PSI) National Forests. Data fields are described in the appendix.

HUC8 watershed	Gage name	National Forest	Gage number	Elevation (ft)	Period examined	Mean discharge (cfs)	Max. discharge (cfs)	Mean peak date	Mean center of flow mass date	Peak discharge (cv)	Peak date (cv)	Center of flow mass date (cv)
Cache La Poudre	Joe Wright Creek above Joe Wright Reservoir	AR	06746095	9,990	1979 –2019	9.9	150	15-Jun	17-Jun	32.3	4.6	3.5
	Cache La Poudre at Ft Collins	AR	06752260	4,940	1976– 2019	175.8	7,150	6-Jun	8-Jun	75.6	15.9	7.2
Big Thompson	Big Thompson River below Moraine Park near Estes Park	AR	402114105 350101	8,005	2002–2019	53.7	761	17-Jun	15-Jun	43.0	10.0	2.9
	Big Thompson River at Loveland	AR	06741510	4,906	1980–2019	70.1	7,200	5-Jun	20-Jun	146.4	25.7	9.8
St. Vrain	Boulder Creek at North 75th	AR	06730200	5,106	1987–2019	95.6	4,770	20-Jun	12-Jun	97.6	10.7	7.9
Clear	Clear Creek near Lawson	AR	06716500	8,070	1995–2019	144.2	1440	14-Jun	16-Jun	38.1	4.2	3.5
	Clear Creek at Golden	AR	06719505	5,695	1975–2019	191.0	2,300	17-Jun	14-Jun	40.3	6.6	3.4
Upper South Platte	South Platte River Below Union Ave	AR, PSI	06710247	5,290	1997–2019	166.2	2,750	8-Jun	29-May	63.1	12.7	11.0
Colorado Headwaters	Colorado River below Baker Gulch near Grand Lake	AR	09010500	8,750	1954–2018	65.4	1,770	9-Jun	8-Jun	48.4	5.1	3.8
Fountain	Fountain Creek near Colorado Springs	PSI	07103700	6,110	1959–2019	16.4	813	2-Jul	6-May	126.9	14.5	17.3
Arkansas Headwaters	Arkansas River near Leadville	PSI	07081200	9,665	1991–2019	75.1	1,120	5-Jun	6-Jun	45.2	4.2	4.1
	Arkansas River at Canon City	PSI	07096000	5,342	1896–2017	717.8	9,480	14-Jun	6-Jun	40.8	8.9	8.0

Table 3 continued.

HUC8 watershed	Gage name	National Forest	Gage number	Elevation (ft)	Period examined	Mean discharge (cfs)	Max. discharge (cfs)	Mean peak date	Mean center of flow mass date	Peak discharge (cv)	Peak date (cv)	Center of flow mass date (cv)
Upper Arkansas	Arkansas River at Avondale	PSI	07109500	4,509	1966–2019	884.8	12,300	24-Jun	3-Jun	43.0	10.8	7.4
Huerfano	Huerfano River Near Boone	PSI	07116500	4,443	1980–2018	31.6	2,900	17-Jun	6-May	99.4	21.0	27.3
Apishapa	Apishapa River near Fowler	PSI	07119500	4,290	1940–2019	23.4	10,100	6-Jul	4-Jun	129.4	20.5	19.6
Purgatoire	Purgatoire River near Madrid	PSI	07124200	6,261	1973–2019	71.7	3,910	30-Jun	5-Jun	100.2	20.4	13.5

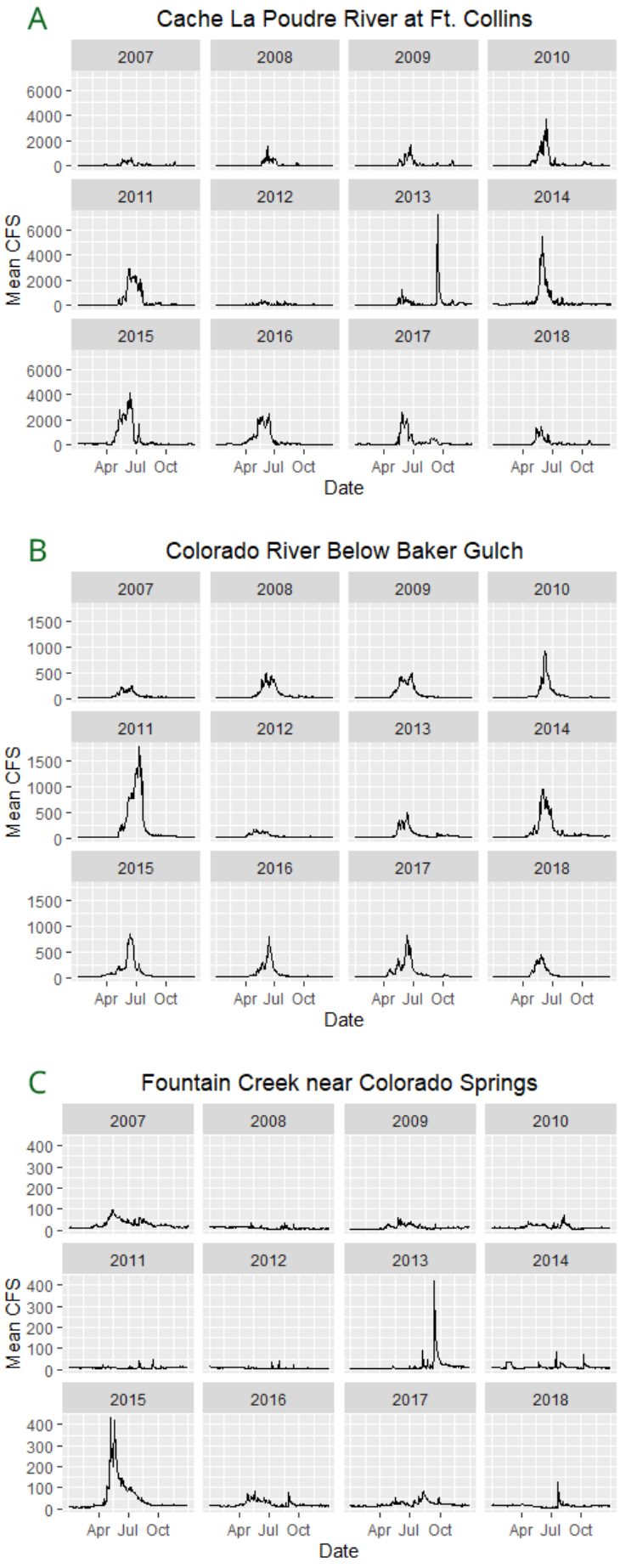


Figure 3—Hydrographs for three Front Range streams. Peak flows in the Cache la Poudre River are typically snowmelt-driven, with exceptions including the late summer rainstorm in 2013, which produced the largest discharge on record. Peak flows are snowmelt-driven in the Colorado River as well, with year-to-year variation in volume and timing of snowmelt runoff. In Fountain Creek, peak flows result from snowmelt runoff (as in 2007 and 2015), summer monsoon rains (2010, 2017, 2018), and the rainstorm of 2013.

Snowpack

Front Range streams with headwaters above 8,000 ft in elevation originate as melting snowpack (Lukas et al. 2014). Snowmelt is also the primary source of recharge for groundwater in upper elevations (Earman and Dettinger 2011; Fields and Dethier 2019). Movement of snow and snowmelt runoff influences sedimentation in streams and valley bottoms (Caine 1995). For these reasons, snowpack is important to a variety of aquatic, riparian, and anthropogenic resources.

Snowpack accumulates during the fall, winter, and spring months, with maximum snow water equivalent typically occurring in the spring (Lukas et al. 2014). Front Range snowmelt volume and runoff timing vary yearly and are influenced by numerous physical and biotic factors. Snow accumulation is influenced by complex interactions among topography, wind, temperature, and precipitation (Rasmussen et al. 2011). Snow accumulates at greater depths in open sites than under canopy (Harpold et al. 2015; Livneh et al. 2015). Winter temperatures determine the amount of energy required to melt snowpacks (Fyfe et al. 2017). Snowmelt rate is also influenced by snowpack depth, elevation, and solar radiation, factors that vary spatially and temporally (Caine 1995; Foster et al. 2016). Accumulation of dust and carbon on snow accelerates the warming and melting of the snowpack (Painter et al. 2018).

Groundwater

The Precambrian crystalline rock aquifers of the Front Range are recharged during large, infrequent precipitation or snowmelt events, with only a small fraction of precipitation and snowmelt becoming groundwater (Dwire et al. 2019). Recharge, storage, and transmission are low relative to sedimentary rocks and alluvial aquifers but remain critical components of the mountain hydrological cycle (Clarey et al. 1999; Williams et al. 2015a). The amount of recharge is dependent on the type of precipitation falling, snowpack depth, snow moisture content, rate of snowmelt, surface characteristics, and transpiration rate (Earman and Dettinger 2011; Fields and Dethier 2019). For the most part, wells drilled into crystalline bedrock can only accommodate domestic use and limited public water supply, whereas wells drilled into sedimentary aquifers can support agricultural, industrial, and municipal use (Topper et al. 2003).

Extreme Rainfall Events

Extreme rainfall events have produced damaging floods in many Front Range streams. Areas affected by extreme rainfall floods are typically below 7,500 feet in elevation. Above this elevation, most floods are caused by snowmelt and are not as destructive (Jarrett and Costa 1988). Extreme rainfall events arise from several atmospheric scenarios. In one, air masses arrive from the south or east and produce convective thunderstorms over mountain ranges. Such events occurred in 1976 and 1997, resulting in the Big Thompson and Fort Collins floods (Jarrett and Costa 2006; Weaver et al. 2000). In another scenario, persistent low-pressure systems keep low-intensity storms in place for several days, resulting in steady, widespread rainfall and flooding. Examples include the Front Range floods of 1938 and 2013 (Lukas et al. 2014).

Drought

It is difficult to define and evaluate droughts because of the complexities associated with measuring the temporal and spatial dynamics of the Colorado climate (Pielke et al. 2005). Factors contributing to droughts on the AR and PSI National Forests include sea surface temperature anomalies; antecedent conditions; above-average temperatures; precipitation-rainfall deficit; deficient atmospheric moisture; and absence of precipitation events such as winter frontal storms, spring low-pressure systems, and summer thunderstorms (Hoerling et al. 2014; Ryan et al. 2013).

Exceptionally dry years occurred in 2002 and 2012, providing lessons for drought effects on water resources. Increased rates of evapotranspiration reduce the amount of water that enters streams and aquifers (Udall and Overpeck 2017). During the 2002 drought, for example, streamflows at the mouth of the Poudre River Canyon were the lowest in recorded history (Salas et al. 2005). During the last 20 years, high temperatures have combined with dry conditions to shift modest droughts to extreme droughts (Udall and Overpeck 2017; Williams et al. 2020). As a result, Colorado has experienced some of its most severe plant die-offs and wildfire seasons in recent memory (Breshears et al. 2005; Pielke et al. 2005; Ryan et al. 2013). Water supplies were exhausted earlier than usual in the driest years, and water quality was affected (Pielke et al. 2005; Saunders and Lewis 2003; Sprague et al. 2005). A reduction in outdoor recreation also occurs during periods with low water levels and high fire danger (Ryan et al. 2013).

Water Quality

Changes in the chemical composition of streamflows have long-term potential to impact ecosystems and water uses on the Colorado Front Range (Caine 1995). Front Range water quality is particularly important to municipal users because 90 percent of Colorado's water supply comes from surface water in forested watersheds (Rhoades et al. 2011). Measures of water quality include stream and lake temperature, water chemistry, nitrification, dissolved oxygen, acidification, suspended sediment, and bacterial content. Several factors, directly and indirectly affected by climate, impact these variables.

Atmospheric warming has a direct influence on stream and lake temperature (Isaak et al. 2015; Roberts et al. 2017b). Higher temperatures contribute to lowered dissolved oxygen concentration of streams, with implications for aquatic communities (Sprague 2005). Heavy rainfall events elevate levels of suspended sediments and deliver various pollutants to streams and lakes (Caine 1995). Wildfire presents numerous challenges to maintaining water quality. Water temperatures immediately increase during fires, and stream temperatures remain high for several years because of increased exposure to solar radiation (Rhoades et al. 2011). Postfire increases in stream temperature add to the percentage of salmonid habitat lost by warming as a result of climate change (Rhoades et al. 2011). Smoke and ash deposits alter water chemistry, but conditions typically return to prefire levels several weeks postfire. Atmospherically deposited nitrogen is mobilized following wildfire and enters streams during runoff events (Riggan et al.

1994). Summer convective storms cause heavy runoff events in postfire watersheds, potentially raising nitrate loads above quality standards for drinking water. Suspended sediments increase during periods of runoff in postfire sites as well (Rhoades et al. 2011; Writer et al. 2012). Postfire sedimentation is thought to increase with the percentage of watershed that is burned with high severity (Mast et al. 2016). Elevated levels of nitrate, sedimentation, and other forms of pollution can persist for several years postfire and are recharged following heavy runoff events (Mast et al. 2016; Rhoades et al. 2011; Writer et al. 2012). These changes exacerbate preexisting water quality concerns in Front Range streams, especially following fires in the lower-elevation montane zones. Wildfire creates problems for municipal water supplies because nitrate levels can impact human health, while suspended sediment increases water treatment costs (Mast et al. 2016).

Additional water quality issues include livestock effects and legacy mining impacts. Most meadows along streams have been grazed by cattle during the last century, introducing pollutants that include nitrate, ammonia, bacteria, and sediment (Gary et al. 1983). High streamflows and lake levels protect ecosystem and human health by diluting these pollutants (Sprague 2005). In watersheds with a history of hard rock mining, abandoned tailings piles and polluted sediments remain in some floodplains. During periods of runoff, heavy metals and other pollutants are leached from these deposits into streams and lakes, with effects that cascade through aquatic and terrestrial ecosystems (Clements et al. 2010; Wohl 2001). Discharge from mine workings can also be a source of dissolved metals pollution, but this discharge is often only noticeable during periods of low flow (Steve Sanchez, PSI National Forests, personal communication).

Water Use Infrastructure

In addition to the naturally occurring streams, lakes, and groundwater systems, the AR and PSI National Forests contain diversions, ditches, pipelines, dams, reservoirs, stock tanks, catchment basins, and other engineered structures. This infrastructure currently influences the distribution, composition, and structure of aquatic, riparian, and wetland communities throughout the Forests (Decker et al. 2015). Reservoirs in the Front Range forests are critical to maintaining availability of water that is allocated for downstream uses (box 3). Reservoirs store water originating within their watersheds, as well as water delivered via transbasin diversions. Ditches and pipelines transport water to points of use both inside and outside of the Forests.

Box 3

Front Range Water Rights

Use of water on the AR and PSI National Forests and of water leaving the forests is governed by Colorado water laws, interstate compacts, and international treaties. Per the Colorado River Compact, water originating in the upper Colorado Basin can only be used within the basin after delivery quotas to the lower Colorado Basin are met (Udall and Overpeck 2017). This obligation also includes the amount of water allocated for use in Mexico.

The Forests own numerous water rights, which allow the storage of water, diversion of surface flows, or pumping of groundwater for beneficial use by the AR and PSI National Forests or leaseholders on the forests. Municipal and private interests own water rights on the Forests as well, allowing them to divert or store water from the forests for beneficial use. In the State of Colorado, water rights can be used to maintain instream flows for aquatic and riparian ecosystems, but such rights can only be held by the Colorado Water Conservation Board (Jones and Cech 2009). Decreed instream flow reaches are found throughout the Front Range national forests (fig. 4).

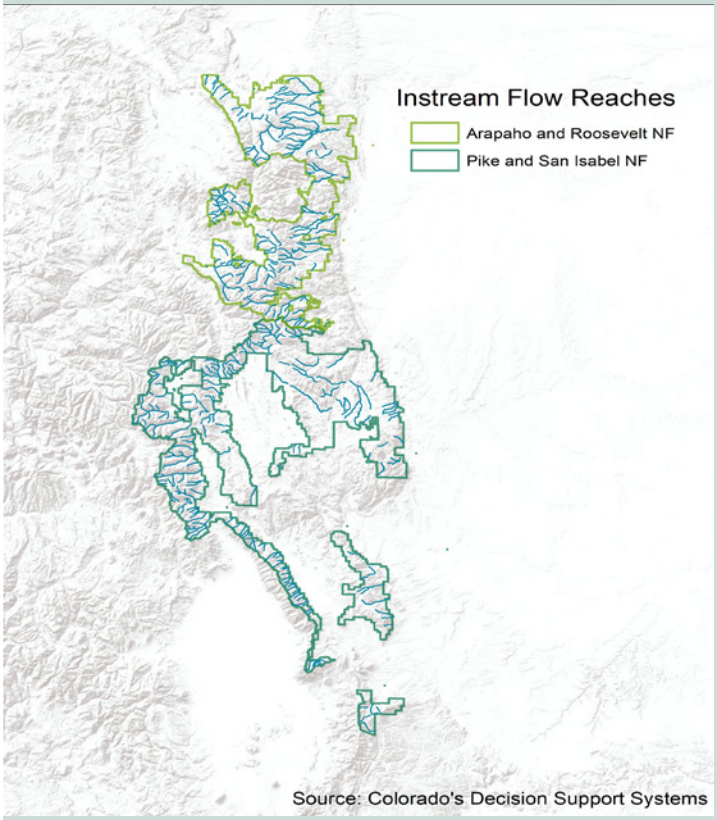


Figure 4—Location of stream segments with decreed instream flow protections in the Front Range National Forests.

Under Colorado's system of prior appropriation, water rights are adjudicated based on the date that water was first put to beneficial use. As a result, many Front Range water rights predate the establishment of the national forests (Hobbs 1997; Jones and Cech 2009). Junior water rights holders, including the AR and PSI National Forests, cannot divert water from a stream unless water will be available for decreed uses of downstream senior rights holders. Given the overallocation of streams in most watersheds, many junior water rights remain out of priority except during years with above-average surface flows. During periods of water scarcity, a call on water rights from downstream users will curtail the use of junior water rights on the national forests.

Vulnerability of Water Resources

In the Colorado Front Range, water creates resources that are valued for their ecological importance and for the services they provide to those that live near the forest, work in the forest, and visit the forests. In this section, I assess the vulnerability of cutthroat trout habitat; riparian areas, wetlands, and groundwater-dependent ecosystems; and anthropogenic water uses. I selected these key resources because (1) they are high management priorities for the AR and PSI National Forests and (2) they encompass many of the aquatic, riparian, and wetland ecosystems on the forests, along with numerous societal uses. Below are descriptions of each resource.

Cutthroat Trout Habitat

The AR and PSI National Forests contain aquatic habitats occupied by two known subspecies of native cutthroat trout (box 2). To prevent the extinction of these subspecies, managers must maintain water quality, habitat heterogeneity, and protection from nonnative fishes (fig. 4). Native cutthroat trout were once found in perennial streams throughout the Arkansas, South Platte, and Colorado Basins. Today, most of this habitat is occupied by nonnative salmonids (fig. 5; Wohl 2001). Front Range populations of cutthroat trout are now mostly limited to stream segments and lakes above 8,000 feet in elevation (Young 2009) According to data obtained from the inland cutthroat trout database in 2019, cutthroat trout populations are managed in nine of the 14 HUC8 watersheds that overlap the AR and PSI National Forests (table 4). The Colorado River Headwaters has the largest number, with 83 populations. The greatest numbers on the eastern slope are found in the Big Thompson, Upper Arkansas River, Cache La Poudre, and St. Vrain watersheds, with the remaining watersheds having fewer than 10 populations.

Water quality is a key aspect of cutthroat trout habitat. Maximum stream temperatures (hottest 7-day average) of less than 26 °C are required for survival. Mean monthly summer water temperatures between 9 °C and 18.0 °C provide ideal conditions for growth and recruitment but recruitment can occur at temperatures as low as 8.0 °C (Harig and Fausch 2002; Roberts et al. 2013). Declining growth occurs between 18.1 °C and 19.9 °C. Low to no growth occurs at temperatures 20 °C or greater. Suspended sediment reduces the amount of dissolved oxygen reaching eggs (Wohl 2001). Pollutants such as acids and heavy metals threaten the health of trout and aquatic food webs (Naeser and Smith 1995; Wohl 2001; Woodward et al. 1991).



Figure 5—Cutthroat trout were historically present across elevations in the Front Range National Forests, but, because of nonnative trout and other stressors, occupied streams are limited to high elevation stream reaches such as the one above. USDA Forest Service photo by Max Smith.

Instream features and riparian vegetation are important habitat components as well. Deep pools are needed for predation refugia and overwinter survival (Harig and Fausch 2002). Gravel beds provide incubation sites for eggs; logs and boulders provide protective cover for young (Wohl 2001). Annual high flows and periodic floods are necessary to create a substrate for spawning, induce spawning behavior, and scour pools used for overwintering habitat (Rathburn et al. 2009). Willows (*Salix* spp.), sedges (*Carex* spp.), and other native riparian plants stabilize soils to create overhanging banks, which are used for cover and thermal protection (Horan et al. 2000; Winward 2000). Willows, alders (*Alnus* spp.), cottonwoods (*Populus* spp.), and other woody plants provide shade that helps to maintain cool summer stream temperatures (Beschta 1997).

Table 4—Water resources mapped in 8-digit Hydrologic Unit Code (HUC8) watersheds.

Basin	HUC8 watershed	Cutthroat trout reaches	Perennial stream miles	Intermittent stream miles	Acres mapped wetlands (% of watershed)	Acres mapped fens	Mapped springs	Forest Service fishing and boating sites	Active ditches and pipelines	Active reservoirs	Active wells
North Platte	Upper Laramie	0	1,387	3,730	13,321 (5%)*	382*	10*	0*	73*	9*	8*
South Platte	Cache La Poudre	14	1,258	2,866	40,311 (4%)*	197*	81*	20*	377*	367*	2,240*
	Big Thompson	27	655	1,086	18,549 (3%)	0	44	2	217	251	699
	St Vrain	12	815	842	23,819 (4%)	7	36	7	564	535	1,894
	Clear	6	574	465	7,393 (2%)	549	50	0	184	171	1,209
	South Platte Headwaters	2	876	2,776	60,336 (6%)	3,430	62	3	191	209	1,936
	Upper South Platte	0	1,350	3,875	30,326 (3%)	646	66	1	581	470	4,783
Upper Colorado	Colorado Headwaters	83	2,751	3,506	82,694 (4%)	2,404	9	10	1,310	571	990
Arkansas	Fountain	0	502	1,987	9,454 (2%)	0	65	3	136	162	1,662
	Arkansas Headwaters	25	2,019	5,820	76,644(4%)	2,334	287	20	1,104	465	1,212
	Upper Arkansas	5	869	5,367	24,995 (2%)	0	113	3	335	234	1,728
	Huerfano	3	665	4,134	32,242 (3%)	0	164	2	368	198	802
	Apishapa	0	140	2,070	4,087 (1%)	0	81	0	38	22	517
	Purgatoire	0	790	6144	21,202 (1%)	0	276	0	221	133	3,026

*These data only apply to the portions of these watershed within the State of Colorado. See figures 10–13.

In addition to streams, mountain lakes provide important habitat for cutthroat trout and other native fishes (Young 2009). Though surface temperatures of lakes have increased as a result of atmospheric warming (Roberts et al. 2017b), lake-stream networks may increase watershed resistance to climate impacts because lakes and reservoirs have greater thermal stability than some streams and are less sensitive to disturbances such as fire, floods, debris flows, and ice (Roberts and Fausch 2015; Young 2009).

Cutthroat trout also require protection from nonnative salmonids, which threaten populations through hybridization, predation, displacement, and competition (McGrath and Lewis 2007; Young 2009). As a result of downstream habitat loss, Colorado River cutthroat trout are now limited to stream segments above 5,000 feet in elevation and greenback cutthroat trout are limited to segments above 8,000 feet (Roberts et al. 2013; Young 2009). At these elevations, stream temperatures act as a thermal barrier to rainbow trout and brown trout, which are less tolerant of cold stream temperatures. Brook trout, however, are tolerant of cold temperatures at higher elevations and must be excluded from cutthroat trout reaches via physical barriers (Roberts et al. 2017a).

Protection of stream reaches currently occupied by cutthroat trout is necessary to preserve the genetic diversity of greenback and Colorado River cutthroat trout populations (Bestgen et al. 2019a). Protecting potentially suitable stream reaches and expanding the range of greenback cutthroat trout is necessary to reduce extinction risk (Young and Harig 2001). Sportfishing for trout is a culturally and economically important form of recreation in the Colorado Rockies (Naeser and Smith 1995). Brook trout are found in many of the upper tributaries of Front Range streams, whereas rainbow trout and brown trout occupy streams in the lower portions of watersheds (Wohl 2001). Because there are many overlapping habitat requirements, actions that improve or harm habitat for native cutthroat trout will have similar impacts on nonnative, yet popular sport fisheries (Richer et al. 2019).

Riparian Areas, Wetlands, and Groundwater-Dependent Ecosystems

In the AR and PSI National Forests, valley bottom characteristics influence the structure and composition of many riparian and wetland ecosystems. Low-gradient streams meander through unconfined valley bottoms, creating extensive riparian and wetland habitats on floodplains and terraces (fig. 6) (Friedman et al. 1995). Low-gradient valley bottoms have been carved and filled by glaciers, incised by streams, and filled by beaver (*Castor canadensis*) pond sedimentation (Kramer et al. 2012). High-gradient streams in bedrock canyons typically have limited development of riparian vegetation (fig. 7) (O'Neil et al. 1997). Along partially confined streams, riparian ecosystems have characteristics of both unconfined valley bottoms and bedrock-restricted canyons. These settings contain a range of vegetation types that differ from one another in structure and composition. Depending on the setting, riparian ecosystems are dominated by coniferous trees, deciduous trees, willows, or herbaceous vegetation (Baker 1989).



Figure 6—Low-gradient riparian ecosystems contain riparian vegetation types dominated by deciduous trees, willows, and herbaceous plants. (USDA Forest Service photo by Max Smith.)



Figure 7—Riparian vegetation is limited in extent in the Big Thompson Canyon and other confined reaches. (USDA Forest Service photo by Max Smith.)

Within valley bottom types, the composition, structure, and arrangement of riparian communities are determined by the frequency and magnitude of floods, base flow levels, groundwater levels, groundwater discharge, and beaver activity (Polvi and Wohl 2011). Maintenance of diverse riparian vegetation types requires periodic floods of varying magnitudes along Front Range streams (Rathburn et al. 2009). Anthropogenic features such as dams, transbasin augmentation, diversion withdrawals, livestock, and roads influence riparian vegetation as well (Friedman et al. 1995; O'Neil et al. 1997; Rathburn et al. 2009).

Wetland types include wet meadows, emergent marshes, fens, and springs (Decker et al. 2015). Springs support components of aquatic and wetland ecosystems (Kreamer et al. 2015). Fens, which are unique, groundwater-dependent wetlands (fig. 8; box 2), tend to be dominated by sedges and other herbaceous plants, but shrubs and small trees may be present as well (Decker et al. 2015). Rare plants, including the threatened Penland's eutrema (*Eutrema penlandii*) and sensitive species such as livid sedge (*Carex livida*), Altai cottongrass (*Eriophorum altaicum*), and blueberry willow (*Salix myrtillifolia*) are found only in fens (ARP 1997; Wrigley et al. 2012). Environmental conditions, such as groundwater chemistry, drive plant composition, resulting in variable and rare communities among Front Range fens (Johnson and Steingraeber 2003).



Figure 8—The wetland vegetation in this montane fen is dominated by sedges, with buckbean (*Menyanthes trifolia*) and other forbs present as well. (USDA Forest Service photo by Max Smith.)

Riparian and wetland ecosystems provide a range of services, including sediment storage, carbon storage, and water quality maintenance (Beckman and Wohl 2014; Wohl et al. 2012). Some of the most ecologically important riparian ecosystems are dominated by cottonwood trees and willow shrubs (Hansen et al. 2002; Hansen 2009; Smith and Finch 2014). Riparian ecosystems are critical for habitat requirements of many wildlife species and serve as migration corridors and rare species habitat (Polvi et al. 2011). For example, the federally threatened Preble's meadow jumping mouse (*Zapus hudonius preblei*) is found in foothills riparian ecosystems that are heavily vegetated and near open, flowing water (Smith et al. 2004). Occupied sites are composed of dense herbaceous vegetation, such as grasses and sedges. Sites often contain an overstory of willows or other woody plants (Smith et al. 2004). These habitats are created by the expansion and contraction of beaver ponds (Frey and Malaney 2009). Amphibians, including tiger salamander (*Ambystoma mavortium*), boreal toad (*Anaxyrus boreas boreas*), northern leopard frog (*Rana pipiens*), and plains leopard frog (*Rana blairi*), use riparian areas, wetlands, and adjacent water bodies for foraging and reproduction (ARP 1997; Decker et al. 2015; Wrigley et al. 2012). Streamside riparian vegetation is also an important habitat component for cutthroat trout because it provides bank stability, stream shading, and instream wood (Dunaway et al. 1994; Hough-Snee et al. 2013; Wondzell et al. 2019).

Springs are found in a variety of settings, often at the contacts of permeable and nonpermeable bedrock layers (Kreamer and Springer 2008). Springs are mapped by the National Hydrography Dataset and the Spring Stewardship Institute (fig. 9). These datasets contain similar numbers of springs for the AR and PSI, but likely underestimate the actual abundance. Mapped springs are most abundant in the Upper Arkansas and Puragtoire watersheds (table 4). Fens have been delineated in some sections of the Front Range national forests, but a systematic mapping project has not been completed (Gabrielle Smith, Colorado Natural Heritage Program, personal communication). The area of wetlands, including fens, has been measured in the Upper Arkansas Basin, where 4,000 acres of seasonally flooded wetlands were classified as fens (Lemly et al. 2016). Large acreages of fens have also been mapped in the Colorado Headwaters and South Platte Headwaters (table 4). Fens are typically found above 7,000 ft, having formed where groundwater discharge supports perennially saturated soils (Decker et al. 2015).

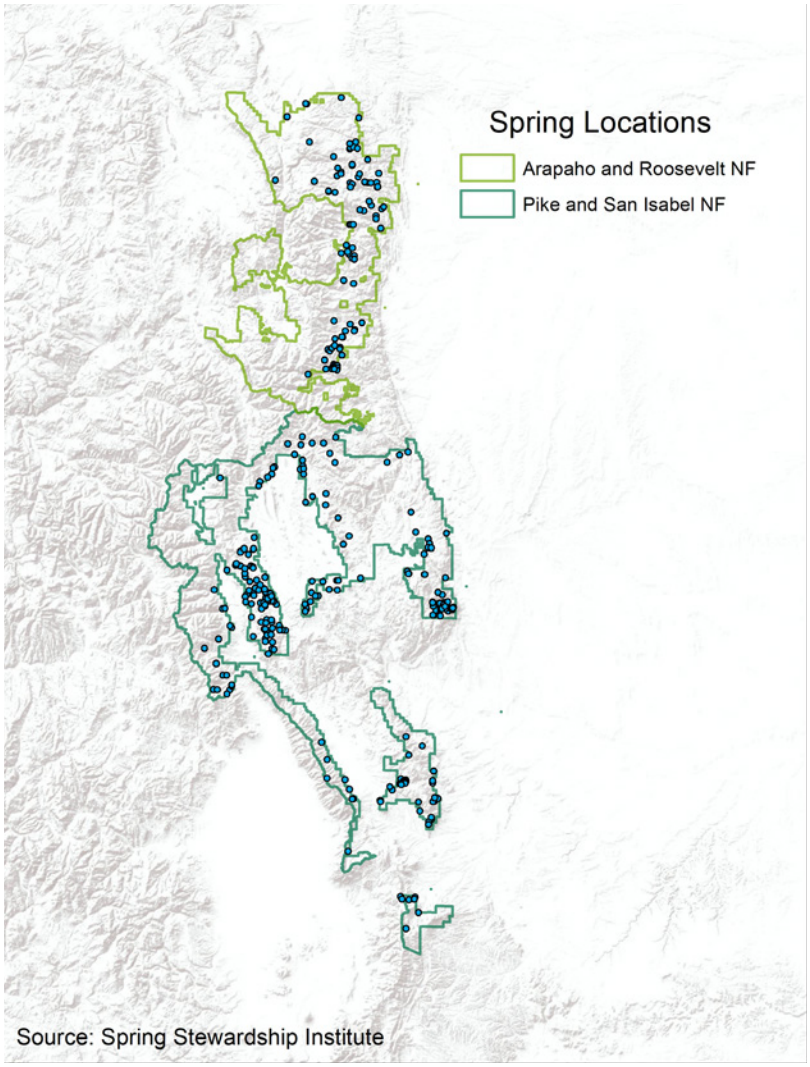


Figure 9—Location of springs mapped in the Front Range National Forests.

Anthropogenic Water Uses

There are numerous uses of surface and groundwater in the AR and PSI National Forests, both consumptive, and nonconsumptive, such as for recreation (figs. 10–15). Relatively small amounts of water are obtained from streams, springs, and wells for on-forest uses at administrative facilities, campgrounds, and grazing allotments (Jones and Cech 2009). Whitewater rafting and kayaking, both commercial and noncommercial, are economically important activities (Loomis and McTernan 2014; Naesser and Smith 1995). The Arkansas River, portions of which pass through the PSI National Forests, contains the most heavily used whitewater in the country. Upper portions of the Cache La Poudre River are designated as wild and scenic. On rivers, including the Arkansas and Cache La Poudre, fishing is popular prior to and after peak flows, whereas whitewater rafting is popular during the peak flow season (Daubert and Young 1981; Naesser and Smith 1995).

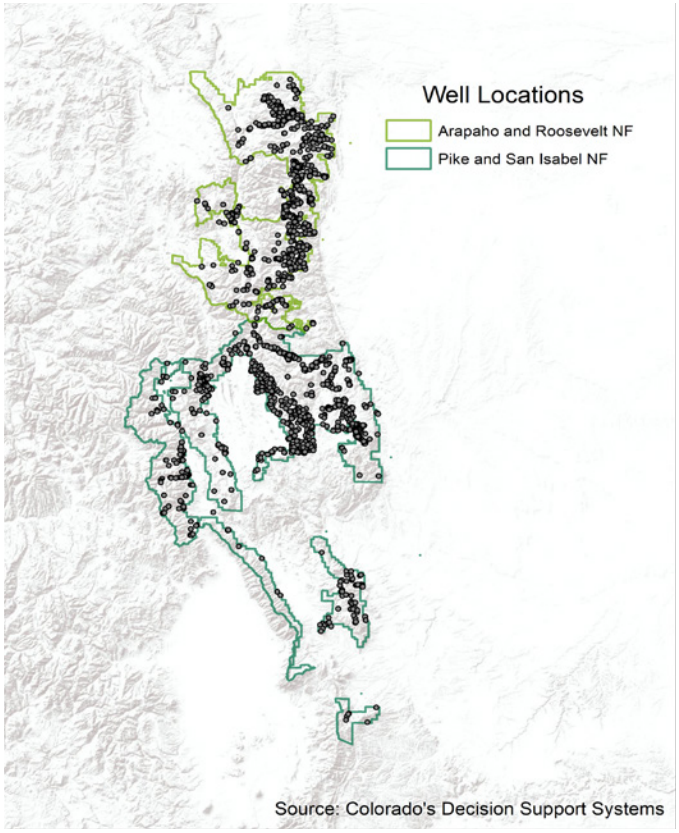


Figure 10—Location of active wells in the Front Range National Forests.

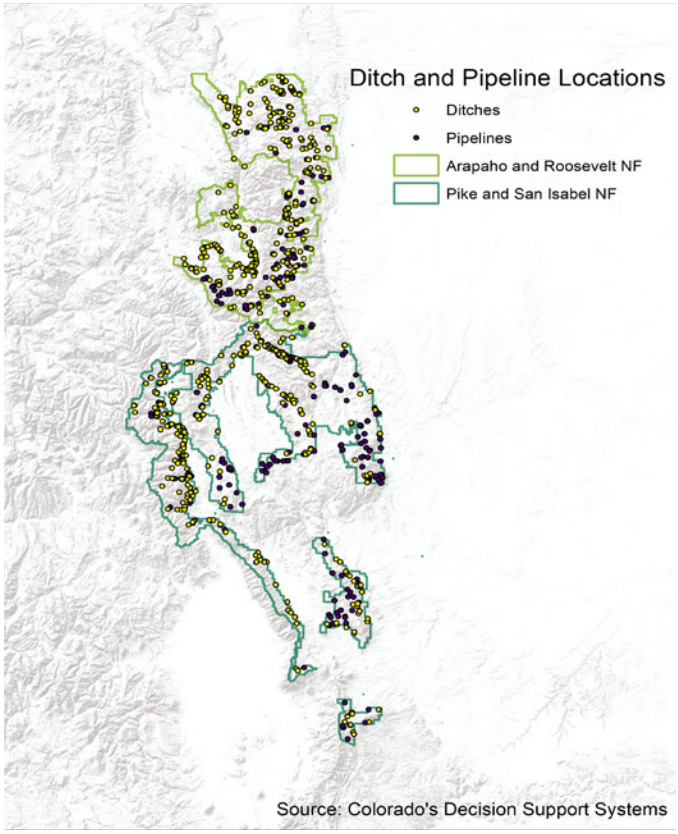


Figure 11—Location of active ditches and pipelines in the Front Range National Forests.

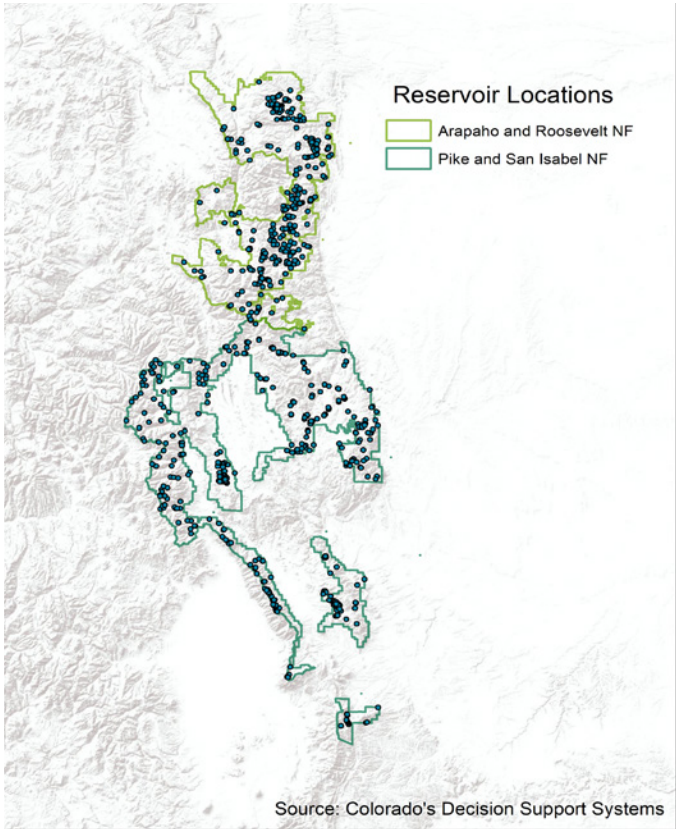


Figure 12—Location of reservoirs in the Front Range National Forests.



Figure 13—Corn and other crops are irrigated with water diverted from the South Platte River, whose water originates in the Front Range National Forests and is augmented by transbasin diversions. (USDA Forest Service photo by Max Smith.)



Figure 14—Water is diverted from the Cache La Poudre River for use by a growing municipal population. (USDA Forest Service photo by Max Smith.)



Figure 15—Twin Lakes, in the Arkansas Basin, is managed for many uses, including water storage, power generation, and a recreational fishery. (USDA Forest Service photo by Max Smith.)

Water originating in or passing through the AR and PSI National Forests is diverted for a variety of uses beyond the Forests' boundaries in Colorado and other States. The State of Colorado provides water to 18 downstream States in both the Colorado and Mississippi River drainages and is a party to nine interstate water compacts (Loomis 1997). Provision of clean water for downstream uses is one of the primary management goals of the AR and PSI National Forests (ARP 1997). Ditch and pipeline diversions are most abundant in the Arkansas headwaters, Cache La Poudre, Colorado headwaters, St. Vrain, and Upper South Platte watersheds. Active wells are widespread but are most abundant in the Cache La Poudre, Upper South Platte, and Purgatoire watersheds (table 4).

Agriculture has long been the primary use of water in the South Platte and Arkansas Basins (fig. 13; Abbott 1985; Watson and Davies 2011). Agricultural uses of water diverted from the Arkansas River include a variety of irrigated crops (CWCB 2019). The South Platte Basin, which includes irrigated crops and livestock production, is the highest-producing agricultural region in the state (CWCB 2019; Watson and Davies 2011). In the Colorado Basin, livestock production dominates agriculture in the upper reaches, with irrigated crops more prevalent near the western state line. As cities expand and the population grows, agriculture to urban water transfer is expected to increasingly occur, especially in the South Platte Basin (CWCB 2019; Howe and Geomans 2003).

The municipal and industrial sectors are second to agriculture as the primary users of water from the AR and PSI National Forests (CWCB 2019). The South Platte Basin is the most populous and rapidly growing basin in the State (Watson and Davies 2011). The population is projected to grow by 42 to 70 percent by 2050. This growth is dependent on water originating in or being transported through the Forests (fig. 14). Population growth is projected for the Arkansas Basin as well, by 45 to 61 percent by 2050 (CWCB 2019). The Upper Colorado Basin includes only 6 percent of the statewide population but is expected to grow by 48 to 88 percent by 2050. Water from several tributaries to the Colorado River is delivered to Front Range communities via transbasin diversions. Colorado water law allows for transbasin water to be fully consumed, thereby providing fewer ecological benefits than uses of native flows, which require the return of excess water to streams or aquifers (Jones and Cech 2009). Water from native flows and transbasin diversions is stored in a series of reservoirs in each basin (figs. 12 and 15). Primary sources of drinking water for Front Range cities include the South Platte River, the Roberts Tunnel transbasin diversion, the Colorado-Big Thompson Project transbasin diversion, the Homestake Project transbasin diversion, the Frying Pan–Arkansas Project transbasin diversion, the Pike's Peak watershed, and the Cache La Poudre River (Folk-Williams et al. 1985; Mast et al. 2016).

Climate Change Vulnerability
Exposure

I evaluated the projected climate change effects on water resources using the four climate change scenarios selected for this assessment. I conducted spatial analyses with Bureau of Reclamation's BCSD CMIP5 Climate and Hydrology Projection dataset. Using rasters covering the Front Range region, I evaluated magnitude of climatic changes among watersheds. I calculated spatial means for these variables, but it is important to note that I did not capture the small-scale variability within the watersheds caused by topographic and elevational differences. A finer-scale approach will be necessary to evaluate exposure for individual streams and sub-watersheds.

Results for each climate scenario are presented in table 5 and figures 16 to 20. Using these data, which were accessed in 2020, I evaluated exposure (magnitude and nature of hydrological change) of the study area. I used three exposure classes: 1 = positive or minimal negative effects; 2 = moderately negative effects or limited negative effects; and 3 = widespread, negative effects from multiple (>3) sources of climate exposure.

Table 5—Sources of water resource exposure under four climate change scenarios for the Colorado Front Range.

Exposure Source	Climate Scenario			
	Very Hot and Wet	Hot	Warm and Wet	Very Hot and Dry
Increases in maximum summer air temperature 2° C or greater are projected in all watersheds.	X	X	—	X
Spring snow water equivalent is projected to decrease in all watersheds.	X	X	X	X
Increases in spring runoff are projected in most watersheds.	X	X	X	—
Decreases in summer flows are projected in most watersheds.	X	X	X	X
Increases in annual runoff are projected in most watersheds.	X	—	—	—
Decreases in annual runoff are projected in most watersheds.	—	—	—	X
Increases in fire danger, drought severity, and/or extreme rainfall events are projected in the Front Range (Cook et al. 2022; Coop et al. 2022; Lukas et al. 2014; Mahoney et al. 2013).	X	X	X	X
Sum	6	5	4	5
Final Exposure Score	3	3	2	3

— indicates no data.

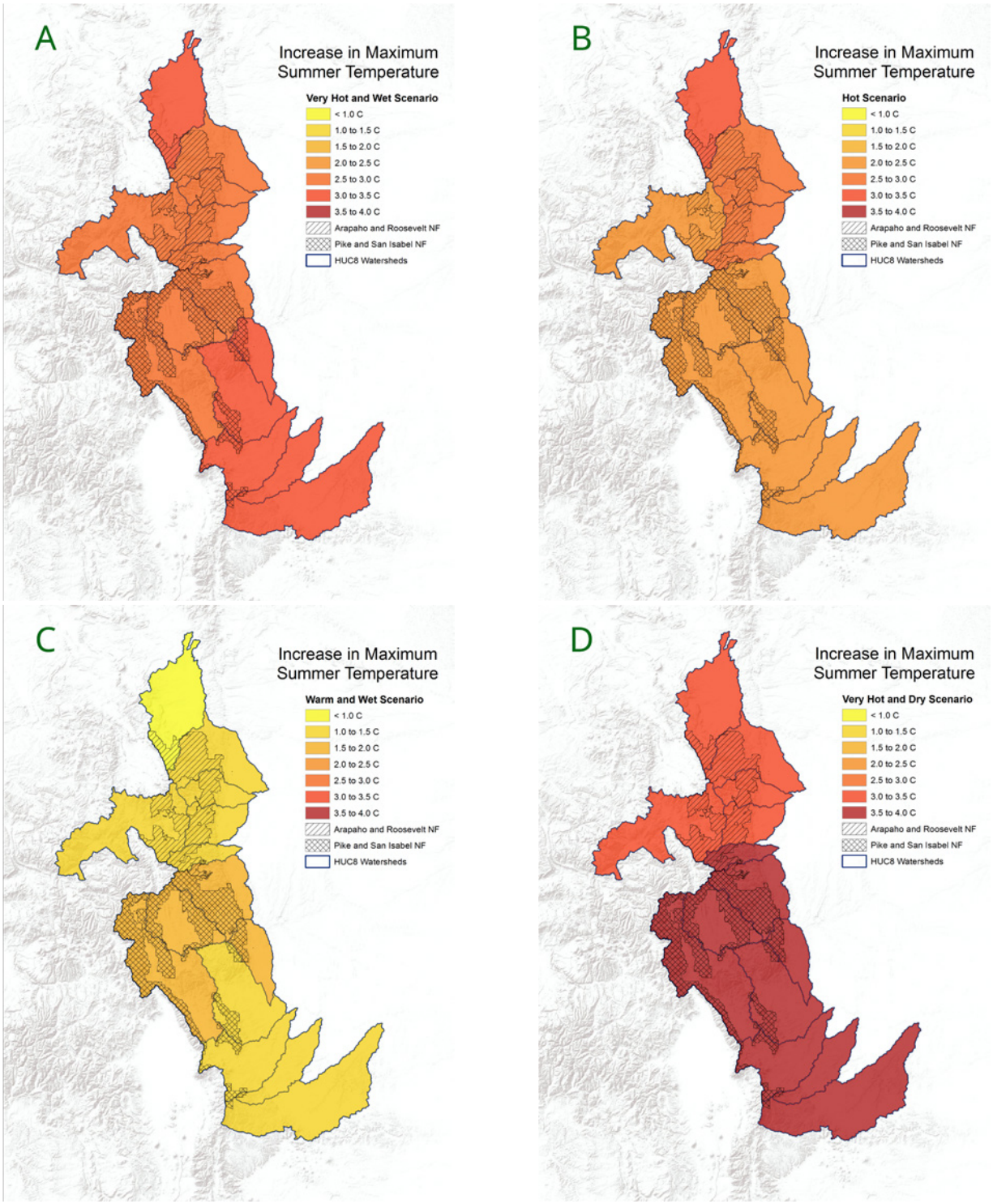


Figure 16a-d—Projected changes in maximum summer air temperatures under four climate scenarios. Values for watersheds depict change in conditions from 1985–2015 (historical) to 2035–2065 (future climate). Data were obtained from the Reclamation BCSD CMIP5 Climate and Hydrology Projection Project.

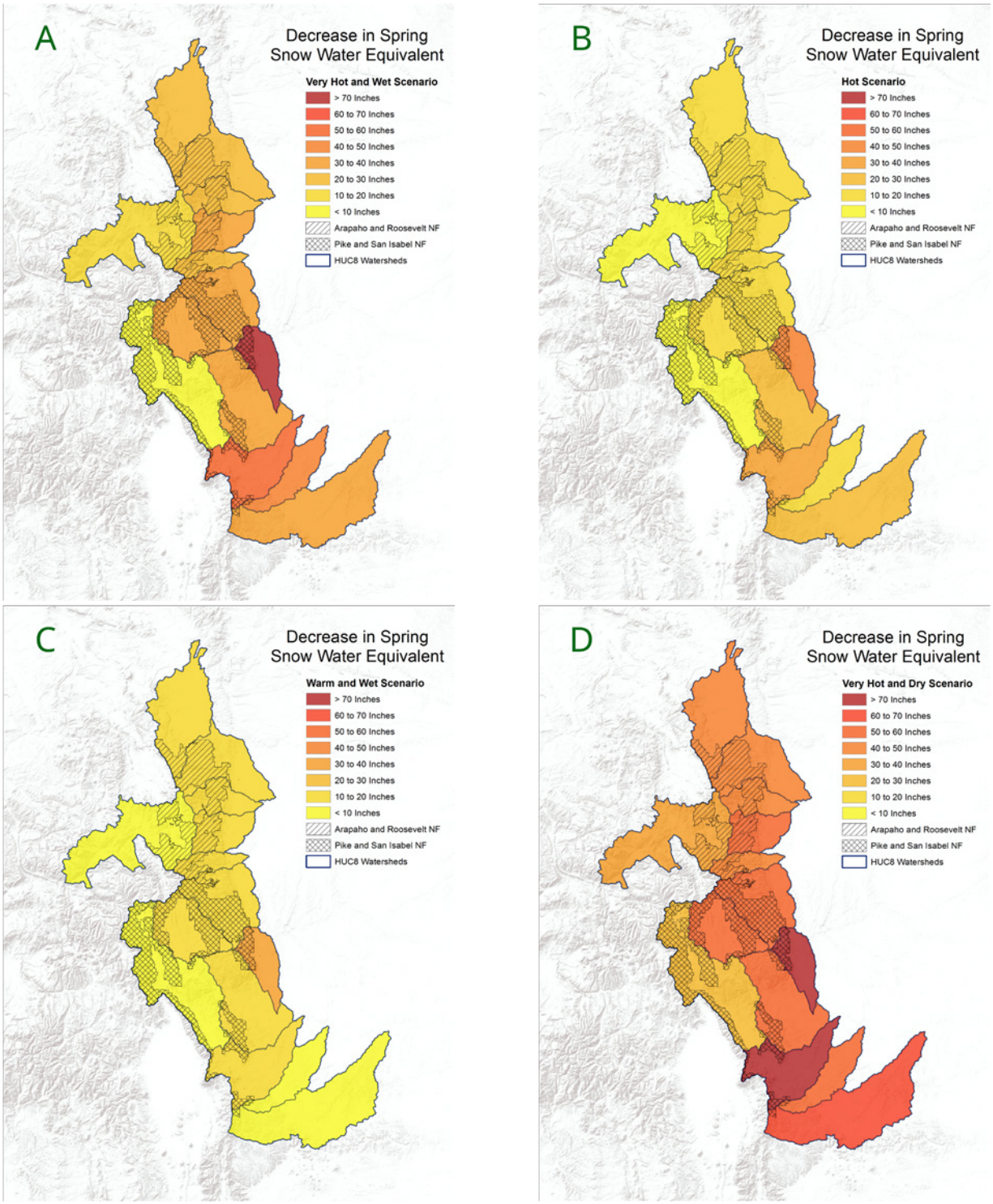


Figure 17a-d—Projected changes in spring snow water equivalent under four climate scenarios. Values for watersheds depict change in conditions from 1985–2015 (historical) to 2035–2065 (future climate). Data were obtained from the Reclamation BCSD CMIP5 Climate and Hydrology Projection Project.

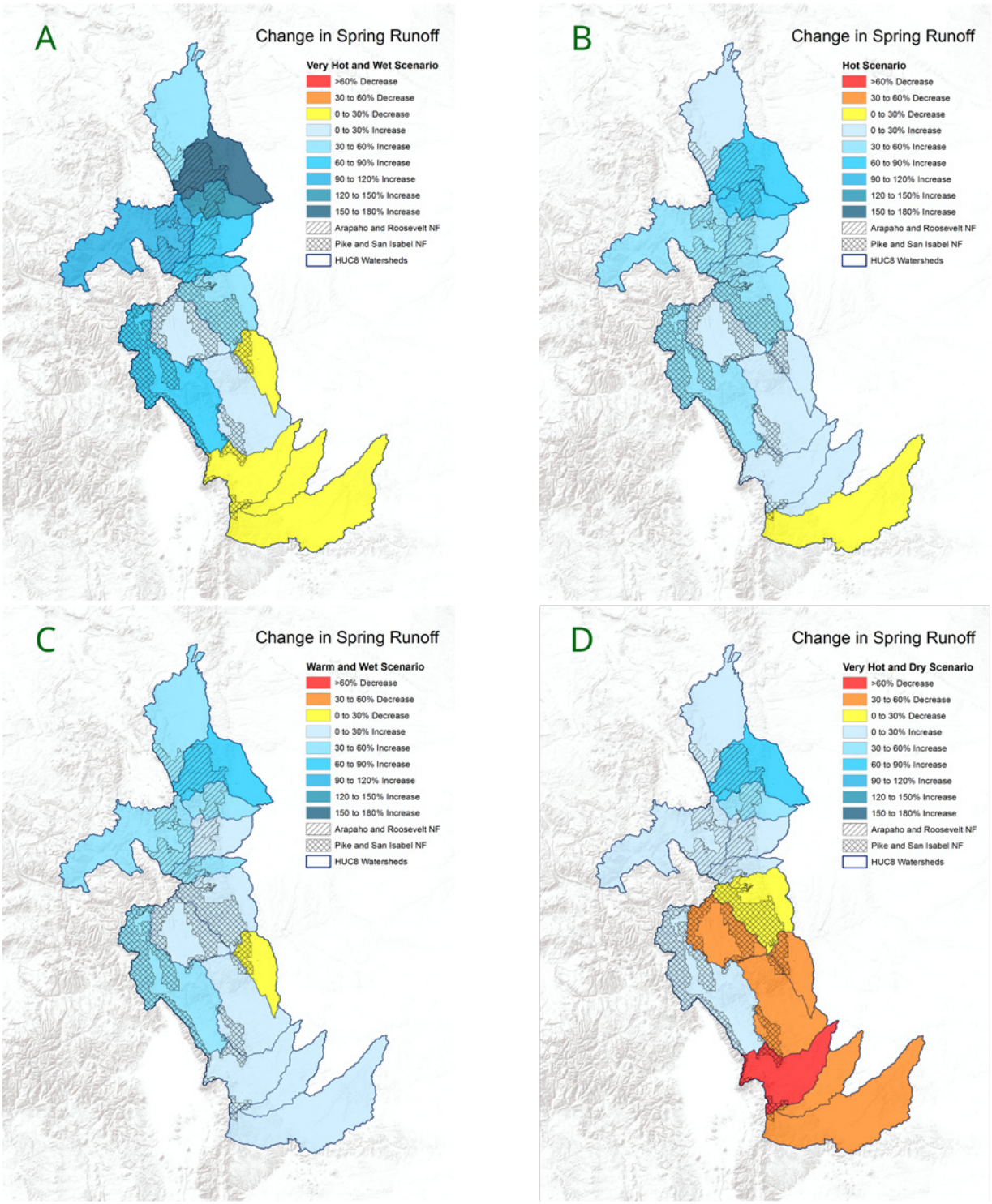


Figure 18a-d—Projected changes in spring runoff under four climate scenarios. Values for watersheds depict change in conditions from 1985–2015 (historical) to 2035–2065 (future climate). Data were obtained from the Reclamation BCSD CMIP5 Climate and Hydrology Projection Project.

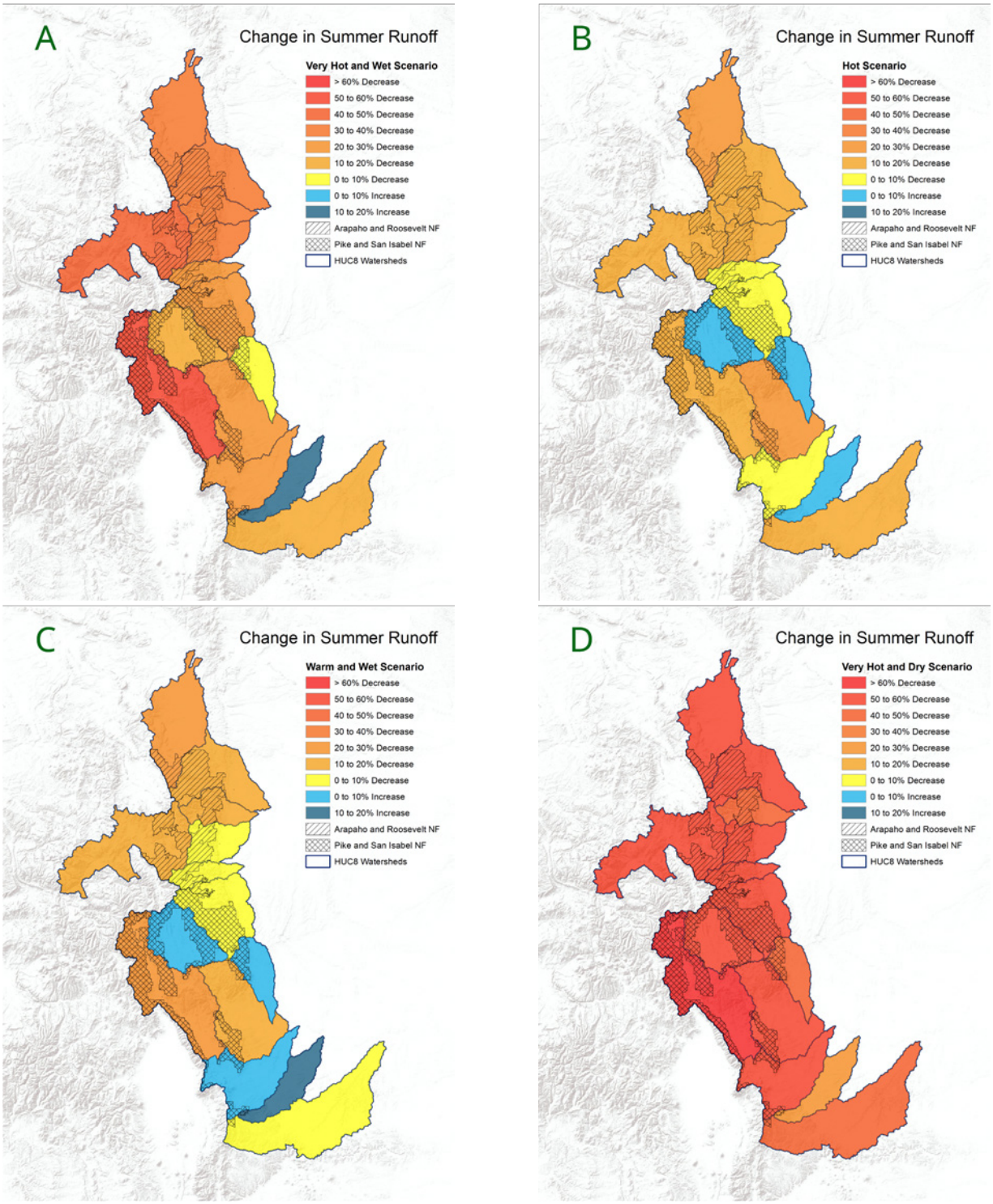


Figure 19a-d—Projected changes in summer runoff under four climate scenarios. Values for watersheds depict change in conditions from 1985–2015 (historical) to 2035–2065 (future climate). Data were obtained from the Reclamation BCSD CMIP5 Climate and Hydrology Projection Project.

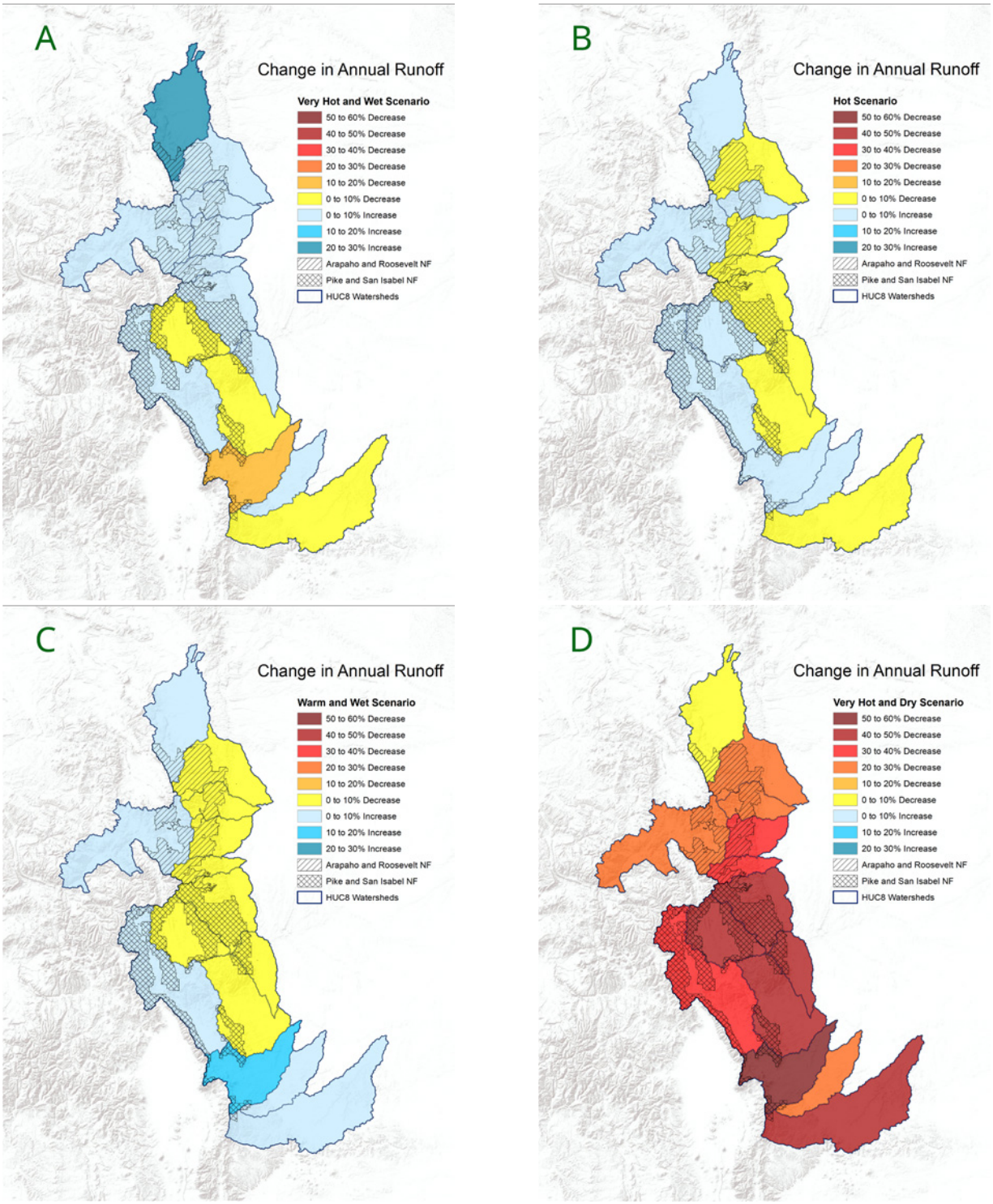


Figure 20a-d—Projected changes in annual runoff under four climate scenarios. Values for watersheds depict change in conditions from 1985–2015 (historical) to 2035–2065 (future climate). Data were obtained from the Reclamation BCSD CMIP5 Climate and Hydrology Projection Project.

On the AR, watersheds with the highest levels of exposure in the South Platte Basin are the Cache La Poudre (based on multiple models projecting increases in spring runoff), Big Thompson (based on multiple models projecting increases in spring runoff), and South Platte headwaters (based on multiple models projecting increases in summer runoff).

Two watersheds are partially within the AR. The Upper Laramie watershed has high levels of exposure based on multiple models projecting increases in maximum summer temperature, decreases in summer runoff, and increases in annual runoff. The Colorado headwaters watershed has high exposure based on multiple models projecting decreases in annual runoff (figs. 16 to 20).

On the PSI, the Arkansas Basin watersheds with the highest levels of exposure are the Fountain (based on multiple models projecting increases in maximum summer temperature, decreases in spring snow water equivalent, and decreases in annual runoff), Arkansas headwaters (based on multiple models projecting decreases in summer runoff), Upper Arkansas (based on multiple models projecting increases in maximum summer temperature and decreases in summer runoff), Huerfano (based on multiple models projecting increases in maximum summer temperature and increases in annual runoff), Apishapa (based on multiple models projecting increases in maximum summer temperature and increases in summer runoff), and Purgatoire (based on multiple models projecting increases in summer temperature and decreases in annual runoff).

Sensitivity

I used spatial data analysis, literature review, and comments from the November 2019 adaptation workshop to identify potential ways that species, ecosystems, and society will be affected under climate change scenarios. I ranked sensitivity as 1 = no or few negative responses; 2 = several negative responses; and 3 = numerous (>6) negative responses. Tables 6–8 show the results of the three resources under each scenario. Sensitivity of streams to climate change is relatively high in the Upper Laramie, Fountain, and Upper Arkansas watersheds (figs. 21 to 24). In the Upper Laramie watershed, projected decreases in summer stream volume and earlier center of flow mass timing are high. In the Fountain and Upper Arkansas watersheds, decreases in coldwater habitat and annual stream volume are relatively high.

Table 6—Sensitivity of cutthroat trout habitat to exposure under four climate change scenarios for the Colorado Front Range. For each row, an "X" indicates that there is potential for a negative response under a given scenario. A "+" indicates that there is potential for a positive response.

Sensitivity Source	Climate Scenario			
	Very Hot and Wet	Hot	Warm and Wet	Very Hot and Dry
Water temperatures increase, reducing the total miles of cold-water habitat in each watershed (fig. 22) (Decker et al. 2015; Isaak et al. 2015).	X	X	—	X
Water temperatures increase in lakes (Decker et al. 2015; Roberts et al. 2017a)	X	X	—	X
Increase in air temperatures results in decreases in summer streamflows in each watershed (fig. 23) (Decker et al. 2015; Lukas et al. 2014; Woodbury et al. 2012)	X	X	—	X
Springtime increases in streamflow volume benefit the growth and survival of young (2019 Workshop)	+	+	+	—
Barriers to migration and dispersal prevent natural adaptation (Decker et al. 2015; Young 2009).	X	X	X	X
Increasingly small, fragmented trout populations are at greater risk of stochastic disturbance events (Rieman and Isaak 2010; Rice et al. 2018; Roberts et al. 2013).	X	X	X	X
Native trout populations become more vulnerable to non-native species, livestock grazing, and pollution (Rice et al. 2018; Roberts et al. 2017b).	X	X	X	X
Water quality is affected by sedimentation, nutrient loading, increased runoff, and post-fire erosion (Decker et al. 2015; Rieman and Isaak 2010).	X	X	X	X
Groundwater recharge decreases at high elevations, which impacts instream flows (Earman and Dettinger 2011; Fields and Dethier 2019; Rice et al. 2018).	X	X	X	X
Sum Negative Responses	8	8	5	8
Sum Positive Responses	1	1	1	0
Final Sensitivity Score	3	3	2	3

Table 7—Sensitivity of riparian areas, wetlands, and groundwater-dependent ecosystems to exposure under four climate change scenarios for the Colorado Front Range. For each row, an "X" indicates that there is potential for a negative response under a given scenario. A "+" indicates that there is potential for a positive response.

Sensitivity Source	Climate Scenario			
	Very Hot and Wet	Hot	Warm and Wet	Very Hot and Dry
Changes in snowpack and precipitation result in larger, earlier peaks in runoff and decreased summer flows (figs. 23 and 24) (Decker et al. 2015; Ficklin et al. 2013; Rasmussen et al. 2014; Woodbury et al. 2012).	X	X	X	X
Decrease in snowpack and late-season flows causes stream intermittency or year-round drying (Decker et al. 2015).	—	—	—	X
Increasing temperatures lead to evaporative losses, increased atmospheric demand, and plant stress and mortality (Breshears et al. 2005).	X	X	—	X
Conifer encroachment is encouraged by a reduction in groundwater and soil moisture availability (Dwire et al. 2018).	X	X	—	X
Increases in wildfire and beetle kill occurrence results in long-term loss of floodplain carbon storage (Beckman and Wohl 2014).	X	X	—	X
Rates of erosion and sedimentation increase following loss of stabilizing vegetation and increasing severity of droughts, floods, and wildfires (Luce 2018).	X	X	X	X
Riparian and wetland ecosystems are at greater risk from non-native species, livestock grazing, and drainage via ditching (Rice et al. 2018).	X	X	X	X
Lower elevation systems are exposed to greater drought, flood, or fire disturbance, which degrades Preble's meadow jumping mouse habitat (2019 Workshop).	X	X	X	X
High spring flows or heavy precipitation events can sustain natural channel processes if stream channels are healthy, improving working relationships with engineering staff, while allowing riparian and aquatic animals to move among habitats (2019 workshop).	+	+	+	—
Decreases in groundwater recharge at high elevations impact downslope riparian communities (Luce 2018).	X	X	X	X
Peat soils and springbrook communities are affected by decreases in groundwater recharge (Dwire et al. 2018).	X	X	X	X
Sum Negative Responses	9	9	6	10
Sum Positive Responses	1	1	1	0
Final Sensitivity Score	3	3	2	3

Table 8—Sensitivity of anthropogenic water uses to exposure under four climate change scenarios for the Colorado Front Range. For each row, an "X" indicates that there is potential for a negative response under a given scenario. A "+" indicates that there is potential for a positive response.

Sensitivity Source	Climate Scenario			
	Very Hot and Wet	Hot	Warm and Wet	Very Hot and Dry
Reduction in stream flows reduces water available for junior water rights holders (fig. 24) (CWCB 2019).	X	X	X	X
Groundwater recharge decreases at high elevations, reducing water available from springs, streams, and wells (Earman and Dettinger 2011).	X	X	X	X
Reservoir storage capacity is reduced because of low surface flows, reduced rates of groundwater recharge, and high rates of evaporation (CWCB 2019; Lukas et al. 2014).	—	—	—	X
Population growth increases demand for a dwindling water supply, causing conflicts between storage and on-forest uses (CWCB 2019; Naeser and Smith 1995).	X	X	X	X
Commitments to interstate and international compacts further complicate water management in the forests (CWCB 2019).	X	X	X	X
Water quality is affected by heavy runoff and wildfires (Rhoades et al. 2011; Hohner et al. 2016).	X	X	X	X
High spring flows and runoff from heavy precipitation events threaten the function of outdated water storage and delivery infrastructure (Lukas et al. 2014).	X	X	X	X
Outdated infrastructure makes it difficult for agencies to meet future water demands (CWCB 2019).	X	X	X	X
Sum Negative Responses	7	7	7	8
Sum Positive Responses	0	0	0	0
Final Sensitivity Score	3	3	3	3

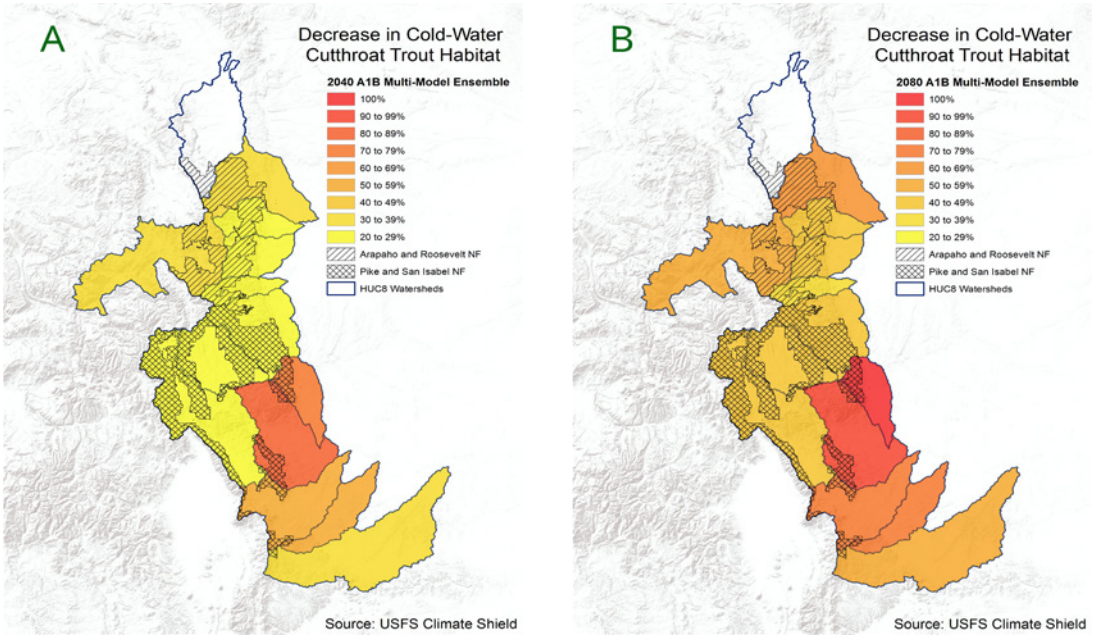


Figure 21a-b—Change in amount of cold-water cutthroat trout habitat based on stream temperatures ideal for reproduction. Projections were made under historical conditions and climate change scenarios. Projections for two future periods (1970–1999 and 2030–2059) were generated using the A1B emissions scenario applied to 10 global climate models. Percent change in miles of suitable stream habitat from the baseline period of 1970–1999 to the future period of 2030–2059 correspond to a moderate climate change scenario. Percent change from the baseline period to the period of 2070–2099 corresponds an extreme climate change scenario.

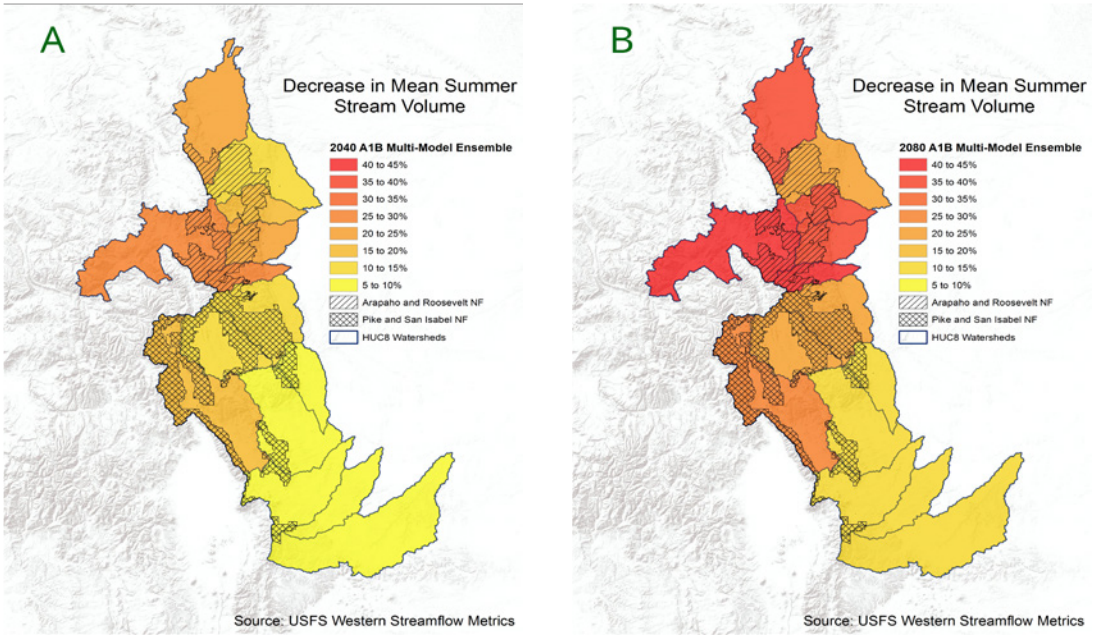


Figure 22a-b—Change in mean daily stream flow volume during the summer months. Streamflow projections were made under historical conditions and climate change scenarios. Projections for two future periods (2030–2059 and 2070–2099) were generated using the A1B emissions scenario applied to 10 global climate models. The VIC method was used to translate climate data into streamflow (USDA FS 2015). Percent change in mean summer stream volume from the baseline period of 1970–1999 to the future period of 2030–2059 corresponds to a moderate climate change scenario. Percent change from the baseline period to the period of 2070–2099 corresponds an extreme climate change scenario.

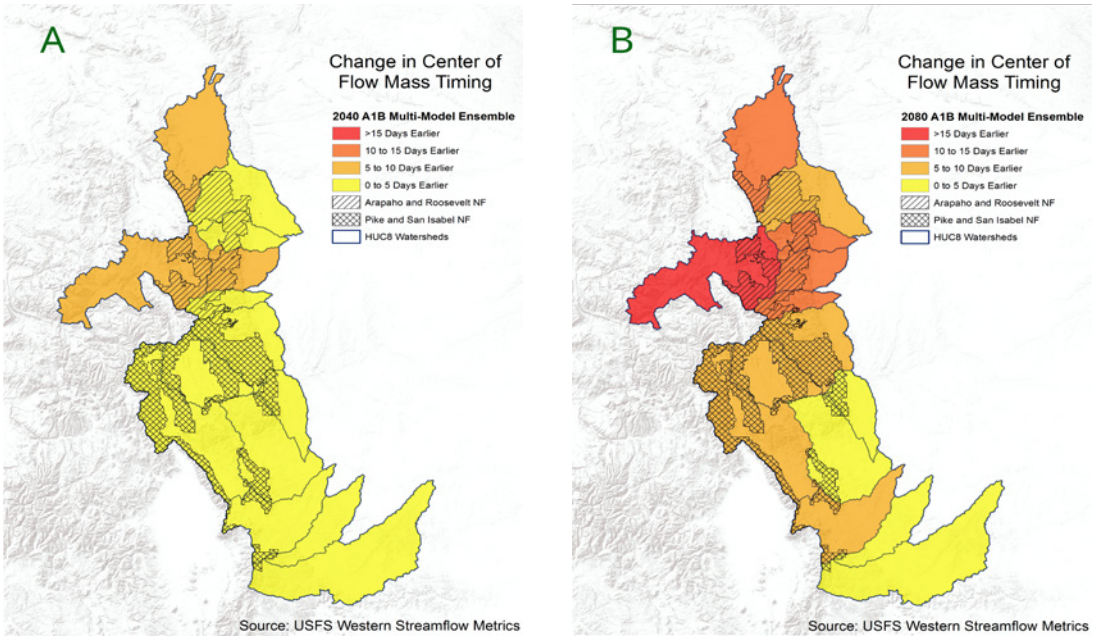


Figure 23a-b—Change, in number of days, of the center of flow mass timing, a measure of runoff timing. Streamflow projections were made under historical conditions and climate change scenarios. Projections for two future periods (2030–2059 and 2070–2099) were generated using the A1B emissions scenario applied to 10 global climate models. The VIC method was used to translate climate data into streamflow (USDA FS 2015). Change in flow mass timing from the baseline period of 197–1999 to the future period of 2030–2059 corresponds to a moderate climate change scenario. Change from the baseline period to the period of 2070–2099 corresponds to an extreme climate change scenario.

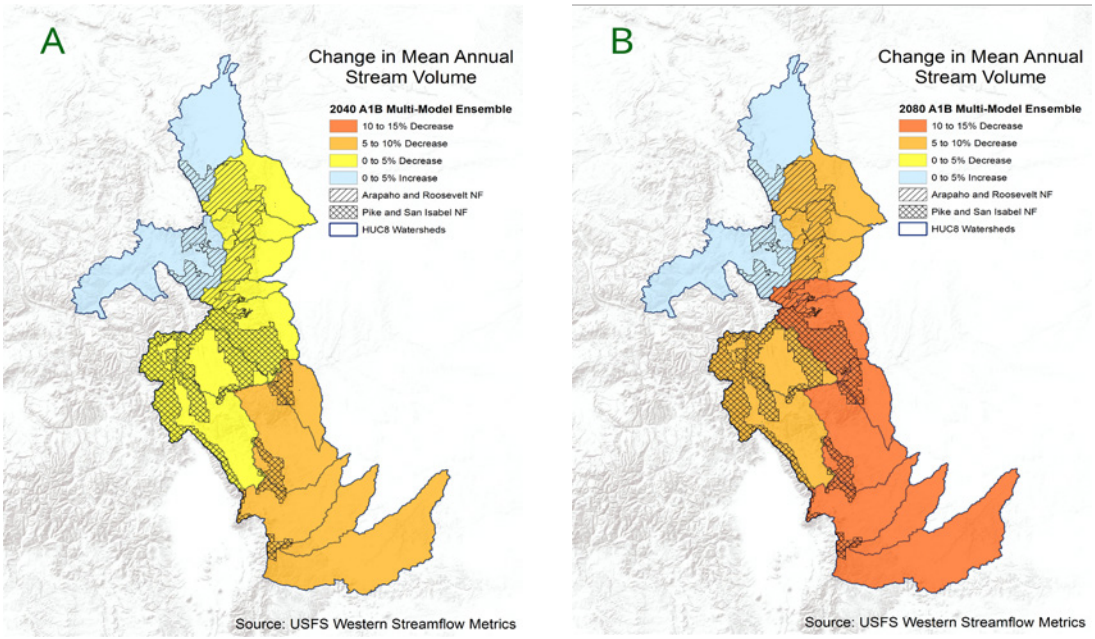


Figure 24a-b—Change in mean annual stream flow. Streamflow projections were made under historical conditions and climate change scenarios. Projections for two future periods (2030–2059 and 2070–2099) were generated using the A1B emissions scenario applied to 10 global climate models. The VIC method was used to translate climate data into streamflow (USDA FS 2015). Percent change in mean annual stream volume from the baseline period of 1970–1999 to the future period of 2030–2059 corresponds to a moderate climate change scenario. Percent change from the baseline period to the period of 2070–2099 corresponds to an extreme climate change scenario.

Adaptive Capacity

I used literature review and comments from the November 2019 workshop to identify the capacity of species, ecosystems, and society to cope with climate change scenarios with minimal disruption (Brandt et al. 2017). I ranked adaptive capacity as 1 = no or few opportunities to cope with changes; 2 = several opportunities to cope with changes (>2); and 3 = numerous opportunities to cope with changes. Results are presented in tables 9–11.

Table 9—Adaptive capacity of cutthroat trout habitat to climate impacts under four climate change scenarios for the Colorado Front Range. For each row, an X indicates that there is potential for adaption under a given climate scenario.

Adaptive Capacity Source	Climate Scenario			
	Very Hot and Wet	Hot	Warm and Wet	Very Hot and Dry
Reestablished beaver populations can create dams, canals, and ponds, which store water throughout valley bottoms, maintaining downstream perennial flows and providing cover in ponds (Grudzinski et al. 2020; Pollock et al. 2014).	X	X	X	X
Managers can move trout to cold stream segments and lakes as they become more thermally suitable in a warming climate (2019 workshop).	X	X	X	—
The AR, PSI, and other entities can change their water rights from consumptive uses to instream flows, which would sustain cold-water habitats during periods of drought (2019 workshop).	X	X	X	X
Sum Positive Responses	3	3	3	2
Final Adaptive Capacity Score	2	2	2	1

Table 10—Adaptive capacity of riparian areas, wetlands, and groundwater-dependent ecosystems to climate impacts under four climate change scenarios for the Colorado Front Range. For each row, an X indicates that there is potential for adaption under a given climate scenario.

Adaptive Capacity Source	Climate Scenario			
	Very Hot and Wet	Hot	Warm and Wet	Very Hot and Dry
Reestablished beaver populations can use dams, canals, and ponds to spread water throughout valley bottoms, protecting riparian and wetland ecosystems from drought and wildfire (Grudzinski et al. 2020; Fairfax and Whittle 2020).	X	X	X	X
Floodplains and vegetation adjust to reduction in streamflow volume, maintaining riparian habitats if surface flows and groundwater remain available (Stromberg et al. 2010).	X	X	X	—
If restored or kept in functioning condition, fens supported by large mountain aquifers may be resilient to droughts and warming (Drexler et al. 2013).	X	X	X	X
The AR, PSI, and other entities can change their water rights from consumptive uses to instream flows, which would maintain minimum flows and sustain riparian and wetland ecosystems during periods of drought (2019 workshop).	X	X	X	X
Reservoirs and transbasin diversion operations can be adjusted to maintain summer flows in large streams and peak flows could be timed to coincide with riparian plant seed release (Duan et al. 2023; Gill et al. 2018).	X	X	X	X
Sum Positive Responses	5	5	5	4
Final Adaptive Capacity Score	3	3	3	2

Table 11—Adaptive capacity of anthropogenic water uses to climate impacts under four climate change scenarios for the Colorado Front Range. For each row, an X indicate that there is potential for adaption under a given climate scenario.

Adaptive Capacity Source	Climate Scenario			
	Very Hot and Wet	Hot	Warm and Wet	Very Hot and Dry
Reestablished beaver populations can use dams, canals, and ponds to store water in valley bottoms, allowing for more on-forest and downstream uses (Grudzinski et al. 2020; Pollock et al. 2014).	X	X	X	X
Restoration of fens and other wetlands can increase water storage capacity in watersheds, allowing for more on-forest and downstream uses (Cooper et al. 1998; Hunt et al. 2018).	X	X	X	X
Infrastructure upgrades can improve efficiency of water transport and use (2019 workshop).	X	X	X	X
The AR, PSI, and other entities can change their water rights from consumptive uses to instream flows, which would sustain some water uses during periods of drought (2019 workshop).	X	X	X	X
Reservoirs and transbasin diversion operations can be adjusted to maximize storage of water from extreme precipitation events for drier seasons (Duan et al. 2023; 2019 workshop).	X	X	X	X
Sum Positive Responses	5	5	5	5
Final Adaptive Capacity Score	3	3	3	3

Vulnerability

Following the methods of Glick et al. (2011) and Brandt et al. (2017), I combined exposure and sensitivity scores into a single impact score, which represents direct and indirect consequences for resources. I compared this score with adaptive capacity in a vulnerability matrix (table 12) to rank vulnerability as low, moderate, high, or very high.

Table 12—Vulnerability matrix used to incorporate exposure, sensitivity, and adaptive capacity scores into a vulnerability ranking.

Adaptive capacity score	Impact score (exposure + sensitivity) 2	3	4	5	6
1	Low	Moderate	Moderate	High	Very High
2	Low	Low	Moderate	High	High
3	Very Low	Low	Moderate	Moderate	High

The final rankings of vulnerability differed between climate scenarios (table 13). The most vulnerable resource is cutthroat trout habitat under the Very Hot and Dry scenario. Riparian areas, wetlands, groundwater-dependent ecosystems, and water uses are also highly vulnerable under this scenario. All three resources were ranked as highly vulnerable under the Hot and the Very Hot and Wet scenarios. All three were ranked as moderately vulnerable under the Warm and Wet scenario.

Table 13—Final vulnerability rankings for water resources under four climate scenarios.

	Very Hot and Wet	Hot	Warm and Wet	Very Hot and Dry
Cutthroat trout habitat	High	High	Moderate	Very High
Riparian and wetland ecosystems	High	High	Moderate	High
Water uses	High	High	Moderate	High

Exposure to climate change is high in three of the four future scenarios. This exposure is driven by multiple indirect effects of increasing temperatures, which cancel out potential positive effects of increasing precipitation in the Very Hot and Wet scenario and the Hot scenario. Sensitivity of cutthroat trout is high in three of the four scenarios because of multiple threats to water quality and the confinement of populations to small stream segments. Drought, wildfire, extreme rainfall, and flood contribute to high sensitivity of each of the key resources (fig. 25). Changes in the timing of peak flows contribute to high sensitivity of riparian and wetland ecosystems. For water uses, water rights and use conflicts are major contributors to sensitivity. In summary, the results demonstrate substantial negative impacts of climate change for water resources under all but the Warm and Wet climate scenario.



Figure 25— Beaver dam complexes slow the movement of water through valley bottoms, providing adaptive capacity to cope with decreased summer flows and frequent droughts. (USDA Forest Service photo by Max Smith)

Riparian ecosystems, wetlands, and water uses have relatively high adaptive capacity because managers can use infrastructure for water storage and delivery to cope with droughts and periods of low flows. In addition, beaver dams can increase the adaptive capacity for each of the resources (fig. 26). Adaptive capacity is lowest for cutthroat trout because of current limitations on dispersal and assisted migration.



Figure 26—Post-wildfire floods and debris flows are an indirect effect of climate change on Front Range streams. These disturbances deliver large amounts of sediment to streams and floodplains. (USDA Forest Service photo by Max Smith)

Uncertainty

For each resource, I examined sources of uncertainty in the vulnerability determinations based on evidence for, and agreement with, the descriptions of exposure, sensitivity, and adaptive capacity (Brandt et al. 2017). I ranked uncertainty as low if there are multiple studies and assessments that report similar effects of climate change and resource responses in the region. If there are no studies or assessments reporting similar effects and responses, I ranked uncertainty as high. If there are few studies or assessments reporting similar effects and responses, I ranked uncertainty as moderate. Results are presented in tables 14a, 14b, and 14c.

Table 14a—Sources of low, moderate, and high uncertainty for cutthroat trout habitat vulnerability in this assessment of key water resources.

a. Low Uncertainty
   - The four climate scenarios agree on increasing temperatures, decreasing SWE, increasing spring runoff, decreasing summer runoff, and increases in extreme rainfall.
   - Multiple assessments project increases in stream temperatures, decreases in summer flows, and increases in spring flows of Front Range streams.
   - Multiple studies project increases in drought and wildfire effects.
   - Multiple assessments and studies report high vulnerability of cutthroat trout populations due to small habitat fragments, barriers to movement, and threats from non-native species.
   - Multiple assessments and studies report risks to water quality, including sedimentation, post-wildfire debris flows, and increased concentration of pollutants.

b. Moderate Uncertainty
   - Three of the four climate models agree on increases in drought severity and fire danger.
   - Assessments project decreases in groundwater discharge following a shift from rain to snow, but studies are needed to confirm this response.

c. High Uncertainty
   - Projected changes in precipitation totals and extreme precipitation effects vary among the four climate scenarios and models used in other Front Range assessments.
   - There are no studies reporting the effects of increased spring runoff on cutthroat trout reproduction.

Table 14b—Sources of low, moderate, and high uncertainty for riparian and wetland vulnerabilaity in this assessment of key water resources.

a. Low Uncertainty
   - The four climate scenarios agree on increasing temperatures, decreasing SWE, increasing spring runoff, decreasing summer runoff, and increases in extreme rainfall.
   - Multiple assessments project decreases in summer flows and earlier peak flows of Front Range streams.
   - Multiple studies show the response of riparian vegetation to changes in groundwater levels, peak flow timing, and soil moisture.
   - Several studies have examined the dynamics of carbon storage along Front Range streams.
   - Multiple studies show the contraction of riparian ecosystems, fens, and other wetlands in response to a reduction in groundwater and surface flows.
   - Multiple studies demonstrate that drought, fire, and extreme rainfall will increase rates of erosion and sedimentation.

b. Moderate Uncertainty

- Three of the four climate models agree on increases in drought severity and fire danger.
- Projected changes in precipitation totals and extreme precipitation effects vary among models used in Front Range assessments.
- Assessments project decreases in groundwater discharge following a shift from rain to snow, but studies are needed to confirm this response.
- Studies are needed to examine evaporative losses, increased atmospheric demand, plant stress, and mortality in Front Range riparian and wetland ecosystems.

c. High Uncertainty

- Little is known about climate impacts on Preble's meadow jumping mouse populations.
- Little is known about the resilience of Front Range fens and springs to drought and other disturbances.

Table 14c—Sources of low, moderate, and high uncertainty for water uses vulnerability in this assessment of key water resources.

a. Low Uncertainty

- The four climate scenarios agree on increasing temperatures, decreasing SWE, increasing spring runoff, decreasing summer runoff, and increases in extreme rainfall.
- Multiple assessments project decreases in streamflows and inflows into reservoirs.
- Multiple studies project increases in flood, drought, and wildfire effects on infrastructure.
- Multiple studies project increasing Front Range population growth and recreation use.
- Multiple assessments and studies report risks to water quality, including sedimentation, post-wildfire debris flows, and increased concentration of pollutants.
- Multiple studies project increasing water rights conflicts.

b. Moderate Uncertainty

- Three of the four climate models agree on increases in drought severity and fire danger.
- Projections of water availability and demand differ among models used in other assessments.

c. High Uncertainty

- The four climate models differ in projections of precipitation totals.

There is high certainty of future temperature increases and increasing risk of drought and wildfire effects. There is high uncertainty, however, on the effects of climate change on precipitation totals and the future frequency of extreme rainfall events in the region. There is also uncertainty concerning the response of cutthroat trout to increases in spring runoff and decreases in groundwater discharge. There is high certainty that prolonged decreases in streamflow and groundwater levels will cause the contraction of riparian and wetland ecosystems in the region. Still, additional information is needed to evaluate better the vulnerability of sensitive species associated with springs, fens, and low-gradient riparian ecosystems. Multiple studies report high vulnerability of water uses to climate change due to aging infrastructure, continued population growth, and decreasing stream flows.

Based on these results, key water resources are highly vulnerable to climate change. Uncertainty is low for cutthroat trout habitat, moderate for riparian and wetland ecosystems, and low for water uses.

Conclusions

Water resources of the AR and PSI National Forests are put to important beneficial ecological and societal uses, both inside and outside of the Forest boundaries. The demand for these resources will only increase in response to a warming climate and a growing human population. The specific climate scenario that will occur in the Front Range is unknown, but increasingly dry summer conditions are projected under all scenarios, necessitating adaptation measures (Cook et al. 2020). This chapter's vulnerability ranking system will give managers a starting point for creating adaptation strategies. Results from stream restoration projects have shown that managers can implement climate change adaptation by repairing habitat, updating water use policies, and allowing natural processes to enhance the resistance and resilience of water resources (Williams et al. 2015b).

For this chapter, key water resources were selected for analysis and discussion. Next, vulnerability to four climate change scenarios was assessed. Finally, a ranking of uncertainty was provided based on the available evidence for similar effects and responses. To summarize: The habitat for cutthroat trout, particularly greenback cutthroat trout, is likely the most vulnerable water resource, given the small size and fragmented nature of suitable stream segments. Without management to enhance climate adaptation, the establishment of self-sustaining populations will be increasingly unlikely under the drier climate scenarios. In areas with cutthroat trout habitat, restoration to reconstruct historical stream channels, enhance riparian cover, and reconnect mainstems and tributaries can decrease the sensitivity of water resources to the effects of climate change.

Forest Service managers can also maintain or enhance water resources by working with partners to establish priorities; for example, updating water storage infrastructure, developing conditional water storage rights, obtaining additional water rights, or changing the administration of existing rights (CWCB 2019). Well-designed and implemented plans to maintain or restore natural infrastructure such as beaver dams, aquifers, and fens can maintain surface flows and maximize groundwater recharge while storing and delivering water supplies to downstream users. The success of these efforts may depend on them being implemented at the watershed scale, involving multiple stakeholders from multiple professions, disciplines, and socioeconomic backgrounds (Lukas et al. 2014; Williams et al. 2015b). As detailed in this assessment, the prioritization of watersheds for restoration and protection should be based on the importance of their water resources, their levels of climate exposure and sensitivity, and their relevance to basin-wide management issues.

References

Abatzoglou, J. T. 2013. Development of gridded surface meteorological data for ecological applications and modelling. International Journal of Climatology. 33: 121–131. https://doi.org/10.1002/joc.3413

Abatzoglou J.T.; Brown T.J. 2012. A comparison of statistical downscaling methods suited for wildfire applications, International Journal of Climatology. 32: 772-780. https://doi.org/10.1002/joc.3413

Abatzoglou, J.T.; McEvoy, D.J.; Redmond, K.T. 2017. The west wide drought tracker: Drought monitoring at fine spatial scales. Bulletin of the American Meteorological Society. 98(9): 1815–1820. https://doi.org/10.1175/BAMS-D-16-0193.1

Arapaho and Roosevelt National Forests and Pawnee Grasslands (ARP). 1997. Revised Land and Resource Management Plan. 427 p. https://www.fs.usda.gov/detail/arp/landmanagement/planning/?cid=fsm91_058285

Clements, W.H.; Vieira, N.K.; Church, S.E. 2010. Quantifying restoration success and recovery in a metal-polluted stream: A 17-year assessment of physicochemical and biological responses. Journal of Applied Ecology. 47: 899–910. https://doi.org/10.1111/j.1365-2664.2010.01838.x

Clow, D.W. 2010. Changes in the timing of snowmelt and streamflow in Colorado: A response to recent warming. Journal of Climate. 23: 2293–2306. https://doi.org/10.1175/2009JCLI2951.1

Colorado Water Conservation Board [CWCB]. 2019. Analysis and technical update to the Colorado Water Plan. Denver: Colorado Water Conservation Board. https://www.colorado.gov/pacific/cowaterplan/analysis-and-technical-update

Cook, B.I.; Mankin; J.S.; Williams. A.P.; [et al.]. 2020. Twenty-first century drought projections in the CMIP6 forcing scenarios. Earth's Future. 8: p.e2019EF001461. https://doi.org/10.1029/2019EF001461

Cook, B.I.; Williams, A.P.; Marvel, K. 2022. Projected changes in early summer ridging and drought over the Central Plains. Environmental Research Letters. 17: 104020. https://doi.org/10.1088/1748-9326/ac8e1a

Coop, J.D.; Parks, S.A.; Stevens-Rumann, C.S.; [et al.]. Extreme fire spread events and area burned under recent and future climate in the Western USA. Global Ecology and Biogeography. 31: 1949–59. https://doi.org/10.1111/geb.13496

Colorado River Cutthroat Trout Coordination Team [CRCT Coordination Team]. 2006. Conservation strategy for Colorado River cutthroat trout (*Oncorhynchus clarkii pleuriticus*) in the States of Colorado, Utah, and Wyoming. Colorado Division of Wildlife, Fort Collins. 24 p. https://cpw.state.co.us/Documents/Research/Aquatic/pdf/CRCT_Conservation_Strategy_Jun06.pdf

Dall, D.; Elliott, C.; Peters, D. 1997. A system for mapping riparian areas in the western United States. Lakewood, CO: USDI Fish and Wildlife Service, National Wetlands Inventory. 43 p.

Daubert, J.T.; Young, R.A. 1981. Recreational demands for maintaining instream flows: A contingent valuation approach. American Journal of Agricultural Economics. 63: 666–676. https://doi.org/10.2307/1241209

Decker, K.; Grunau, L.; Handwerk, J.; [et al.]., eds. 2015. Colorado Natural Heritage Program climate change vulnerability assessment for Colorado Bureau of Land Management. Fort Collins, CO: Colorado Natural Heritage Program, Colorado State University. 573 p.

Deems, J.S.; Painter, T.H.; Barsugli, J.J. 2013. Combined impacts of current and future dust deposition and regional warming on Colorado River Basin snow dynamics and hydrology. Hydrology and Earth System Sciences. 17: 4401–4413. https://doi.org/10.5194/hess-17-4401-2013

Dettinger, M.; Udall, B.; Georgakakos, A. 2015. Western water and climate change. Ecological Applications. 25: 2069–2093. https://doi.org/10.1890/15-0938.1

Drexler, J.Z.; Knifong, D.; Tuil, J.; [et al.]. 2013. Fens as whole-ecosystem gauges of groundwater recharge under climate change. Journal of Hydrology. 481: 22–34. https://doi.org/10.1016/j.jhydrol.2012.11.056

Duan, K.; Qu, S.; Liu, N.; [et al.]. 2023. Evolving efficiency of inter-basin water transfers in regional water stress alleviation. Resources, Conservation and Recycling. 191: 106878. https://doi.org/10.1016/j.resconrec.2023.106878

Dunaway, D.; Swanson, S.R.; Wendel, J.; [et al.]. 1994. The effect of herbaceous plant communities and soil textures on particle erosion of alluvial streambanks. Geomorphology. 9: 47–56. https://doi.org/10.1016/0169-555X(94)90030-2

Dwire, K.A.; Mellmann-Brown, S.; Gurrieri, J.T. 2018. Potential effects of climate change on riparian areas, wetlands, and groundwater-dependent ecosystems in the Blue Mountains, Oregon, USA. Climate Services. 10: 44–52. https://doi.org/10.1016/j.cliser.2017.10.002

Earman, S.; Dettinger, M. 2011. Potential impacts of climate change on groundwater resources—a global review. Journal of Water and Climate Change. 2: 213–229. https://doi.org/10.2166/wcc.2011.034

Elgaali, E.; Garcia, L.A. 2007. Using neural networks to model the impacts of climate change on water supplies. Journal of Water Resources Planning and Management. 133: 230–243. https://doi.org/10.1061/(ASCE)0733-9496(2007)133:3(230)

Ficklin, D.L.; Stewart, I.T.; Maurer, E.P. 2013. Climate change impacts on streamflow and subbasin-scale hydrology in the Upper Colorado River Basin. PLoS One. 8. https://doi.org/10.1371/journal.pone.0071297

Fields, J.F.; Dethier, D.P. 2019. From on high: Geochemistry of alpine springs, Niwot Ridge, Colorado Front Range, USA. Hydrological Processes. 33: 1756–1774. https://doi.org/10.1002/hyp.13436

Folk-Williams, J.A.; Fry, S.C.; Hilgendorf, L. 1985. Western water flows to the cities. Washington, DC: Island Press. 217 pp.

Foster, L.M.; Bearup, L.A.; Molotch, N.P.; [et al.]. 2016. Energy budget increases reduce mean streamflow more than snow–rain transitions: Using integrated modeling to isolate climate change impacts on Rocky Mountain hydrology. Environmental Research Letters. 11: 044015. https://doi.org/10.1088/1748-9326/11/4/044015

Frey, J.K.; Malaney, J.L. 2009. Decline of the meadow jumping mouse (*Zapus hudsonius luteus*) in two mountain ranges in New Mexico. The Southwestern Naturalist. 54: 31–44. https://www.jstor.org/stable/40263664

Friedman, J.M.; Scott, M.L.; Lewis, W.M. 1995. Restoration of riparian forest using irrigation, artificial disturbance, and natural seedfall. Environmental Management. 19: 547–557. https://doi.org/10.1007/BF02471966

Fyfe, J.C.; Derksen, C.; Mudryk, L.; [et al.]. 2017. Large near-term projected snowpack loss over the western United States. Nature Communications. 8: 14996. https://doi.org/10.1038/ncomms14996

Gary, H.L.; Johnson, S.R.; Ponce, S.L. 1983. Cattle grazing impact on surface water quality in a Colorado Front Range stream. Journal of Soil and Water Conservation. 38: 124–128. https://www.jswconline.org/content/38/2/124

Giardino, J.R.; Shroder Jr, J.F.; Lawson, M.P. 1984. Tree-ring analysis of movement of a rock-glacier complex on Mount Mestas, Colorado, USA. Arctic and Alpine Research. 16: 299–309. https://doi.org/10.2307/1550939

Gill, K.M.; Goater, L.A.; Braatne, J.H.; [et al.]. 2018. The irrigation effect: How river regulation can promote some riparian vegetation. Environmental Management. 61: 650–60. https://doi.org/10.1007/s00267-017-0991-4

Glick, P.; Stein, B.A.; Edelson, N.A. 2011. Scanning the conservation horizon: A guide to climate change vulnerability assessment. Washington, DC: National Wildlife Federation. Available: https://www.nwf.org/vulnerabilityguide

Gregory, S.V.; Swanson, F.J.; McKee, W.A.; [et al.]. 1991. An ecosystem perspective of riparian zones. BioScience. 41: 540–551. https://doi.org/10.2307/1311607

Grudzinski, B.P; Cummins, H.; Vang, T.K. 2020. Beaver canals and their environmental effects. Progress in Physical Geography: Earth and Environment. 44: 189–211. https://doi.org/10.1177/0309133319873116

Hansen, A.J. 2009. Species and habitats most at risk in Greater Yellowstone. Yellowstone Science. 17: 27–36. https://www.montana.edu/hansenlab/documents/bio515_13/Hansen12072009yellscience.pdf [Accessed 6 June 2024]

Hansen, A.J.; Rasker, R.; Maxwell, B.; [et al.]. 2002. Ecology and socioeconomics in the New West: A case study from Greater Yellowstone. BioScience. 52: 151–168. https://doi.org/10.1641/0006-3568(2002)052[0151:ECACOD]2.0.CO;2 [Accessed 6 June 2024]

Harig, A.L.; Fausch, K.D. 2002. Minimum habitat requirements for establishing translocated cutthroat trout populations. Ecological Applications. 12: 535–551. https://doi.org/10.1890/1051-0761(2002)012[0535:MHRFET]2.0.CO;2

Harpold, A.A.; Molotch, N.P.; Musselman, K.N.; [et al.]. 2015. Soil moisture response to snowmelt timing in mixed-conifer subalpine forests. Hydrological Processes. 29: 2782–2798. https://doi.org/10.1002/hyp.10400

Hobbs Jr., G.J. 1997. Colorado water law: An historical overview. University of Denver Water Law Review. 1: 1. https://www.judges.org/wp-content/uploads/2020/03/Colorado-Water-Law-An-Historical-Overview-through-8th-update.pdf [Accessed 6 June 2024]

Hoerling, M.; Eischeid, J.; Kumar, A.; [et al.]. 2014. Causes and predictability of the 2012 Great Plains drought. Bulletin of the American Meteorological Society. 95: 269–282. https://doi.org/10.1175/BAMS-D-13-00055.1

Hohner, A.K.; Cawley, K.; Oropeza, J.; [et al.]. 2016. Drinking water treatment response following a Colorado wildfire. Water Research. 105: 187–98. https://doi.org/10.1016/j.watres.2016.08.034

Horan, D.L.; Kershner, J.L.; Hawkins, C.P.; [et al.]. 2000. Effects of habitat area and complexity on Colorado River cutthroat trout density in Uinta Mountain streams. Transactions of the American Fisheries Society. 129: 1250–1263. https://doi.org/10.1577/1548-8659(2000)129<1250:EOHAAC>2.0.CO;2

Hough-Snee, N.; Roper, B.B.; Wheaton, J.M.; [et al.]. 2013. Riparian vegetation communities change rapidly following passive restoration at a northern Utah stream. Ecological Engineering. 58: 371–377. https://doi.org/10.1016/j.ecoleng.2013.07.042

Howe, C.W.; Goemans, C. 2003. Water transfers and their impacts: Lessons from three Colorado water markets 1. JAWRA, Journal of the American Water Resources Association. 39: 1055–1065. https://doi.org/10.1111/j.1752-1688.2003.tb03692.x

Hunt, L.J.H.; Fair, J.; Odland, M. 2018. Meadow restoration increases baseflow and groundwater storage in the Sierra Nevada Mountains of California. JAWRA, Journal of the American Water Resources Association. 54: 1127–1136. https://doi.org/10.1111/1752-1688.12675

Immerzeel, W.W.; Lutz, A.F.; Andrade, M.; [et al.]. 2019. Importance and vulnerability of the world's water towers. Nature. 577: 364–369. https://doi.org/10.1038/s41586-019-1822-y

Isaak, D.J.; Young, M.K.; Nagel, D.E.; [et al.]. 2015. The cold-water climate shield: Delineating refugia for preserving salmonid fishes through the 21st century. Global Change Biology. 21: 2540–2553. https://doi.org/10.1111/gcb.12879

Jarrett, R.D.; Costa, J.E. 1988. Evaluation of the flood hydrology in the Colorado Front Range using precipitation, streamflow, and paleoflood data for the Big Thompson River basin. Water Resources Investigations Report 87-4117. Denver, CO: Department of the Interior, U.S. Geological Survey. https://pubs.usgs.gov/wri/1987/4117/report.pdf [Accessed 6 June 2024]

Jarrett, R.D.; Costa, J.E. 2006. Big Thompson Flood, Colorado: Thirty years later. USGS Fact Sheet 2006-3095. Denver, CO: Department of the Interior, U.S. Geological Survey. 5 p. https://pubs.usgs.gov/fs/2006/3095/pdf/FS06-3095_508.pdf

Johnson, T.; Butcher, J.; Deb, D.; [et al.]. 2015. Modeling streamflow and water quality sensitivity to climate change and urban development in 20 U.S. watersheds. JAWRA, Journal of the American Water Resources Association. 51: 1321–1341. https://doi.org/10.1111/1752-1688.12308

Johnson, J.B.; Steingraeber, D.A. 2003. The vegetation and ecological gradients of calcareous mires in the South Park Valley, Colorado. Canadian Journal of Botany. 81: 201–219. https://doi.org/10.1139/b03-017

Jones, D.B.; Harrison, S.; Anderson, K; [et al.]. 2019. Rock glaciers and mountain hydrology: A review. Earth-Science Reviews. 193: 66–90. https://doi.org/10.1016/j.earscirev.2019.04.001

Jones, K.A.; Niknami, L.S.; Buto, S.G.; [et al.]. 2022. Federal standards and procedures for the national Watershed Boundary Dataset (WBD), 5th ed. U.S. Geological Survey Techniques and Methods 11-A3. Reston, VA: Department of the Interior, U.S. Geological Survey. 54 p. https://pubs.usgs.gov/tm/11/a3/

Jones, P.A.; Cech, T. 2009. Colorado water law for non-lawyers. Denver, CO: University Press of Colorado. 276 p.

Joyce, L.A.; Coulson, D. 2020. Climate scenarios and projections: A technical document supporting the USDA Forest Service 2020 RPA Assessment. Gen. Tech. Rep. RMRS-GTR-413. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 85 p. https://doi.org/10.2737/RMRS-GTR-413

Kampf, S.K.; Lefsky, M.A. 2016. Transition of dominant peak flow source from snowmelt to rainfall along the Colorado Front Range: Historical patterns, trends, and lessons from the 2013 Colorado Front Range floods. Water Resources Research. 52: 407–422. https://doi.org/10.1002/2015WR017784

Kenney, D.S.; Klein, R.A.; Clark, M.P. 2004. Use and effectiveness of municipal waters restrictions during drought in Colorado 1. JAWRA, Journal of the American Water Resources Association. 40: 77–87. https://doi.org/10.1111/j.1752-1688.2004.tb01011.x

Kramer, N.; Wohl, E.E.; Harry, D.L. 2012. Using ground penetrating radar to "unearth" buried beaver dams. Geology. 40: 43–46. https://doi.org/10.1130/G32682.1

Kreamer, D.K.; Springer, A.E. 2008. The hydrology of desert springs in North America. In: Stevens, LE: Meretsky, V.J., eds. 2008. Aridland springs in North America: Ecology and conservation. Tucson, AZ: University of Arizona Press. pp. 35-48.

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

Kundzewicz, Z.W.; Doll, P. 2009. Will groundwater ease freshwater stress under climate change? Hydrological Sciences Journal. 54: 665–675. https://doi.org/10.1623/hysj.54.4.665

Lemly, J. L.; Long, G.; Smith, G.; [et al.]. 2016. National wetland inventory mapping of the Arkansas headwaters subbasin. Fort Collins, CO: Colorado Natural Heritage Program, Colorado State University. 58 p. https://cnhp.colostate.edu/download/documents/2016/Arkansas_Headwaters_NWI_Mapping_Report_FINAL.pdf [Accessed 6 June 2024]

Livers, B.; Wohl, E. 2015. An evaluation of stream characteristics in glacial versus fluvial process domains in the Colorado Front Range. Geomorphology. 231: 72–82. https://doi.org/10.1016/j.geomorph.2014.12.003

Livneh, B.; Deems, J.S.; Buma, B.; [et al.]. 2015. Catchment response to bark beetle outbreak and dust-on-snow in the Colorado Rocky Mountains. Journal of Hydrology. 523: 196–210. https://doi.org/10.1016/j.jhydrol.2015.01.039

Loomis, J.; McTernan, J. 2014. Economic value of instream flow for non-commercial whitewater boating using recreation demand and contingent valuation methods. Environmental Management. 53: 510–519. https://doi.org/10.1007/s00267-014-0232-z

Luce, C.H. 2018. Effects of climate change on snowpack, glaciers, and water resources in the Northern Rockies region. In: Halofsky, J.E.; Peterson, D.L.; Dante-Wood, SK.; [et al.], eds. 2018. Climate change vulnerability and adaptation in the Northern Rocky Mountains. Gen. Tech. Rep. RMRS-GTR-374. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. Part 1: 1–273. https://www.fs.usda.gov/research/treesearch/55989

Lukas, J.; Barsugli, J.; Doesken, N.; [et al.]. K. 2014. Climate change in Colorado: A synthesis to support water resources management and adaptation. Boulder, CO: University of Colorado. https://www.cakex.org/sites/default/files/documents/Climate_Change_CO_Report_2014_FINAL_0.pdf [Accessed 6 June 2024]

Mahoney, K.; Alexander, M.; Scott, J.D; [et al.]. 2013. High-resolution downscaled simulations of warm-season extreme precipitation events in the Colorado Front Range under past and future climates. Journal of Climate. 26: 8671–8689. https://doi.org/10.1175/JCLI-D-12-00744.1

Mast, M.A.; Murphy, S.F.; Clow, D.W.; [et al.]. 2016. Water-quality response to a high-elevation wildfire in the Colorado Front Range. Hydrological Processes. 30: 1811–1823. https://doi.org/10.1002/hyp.10755

McGrath, C.C.; Lewis Jr., W.M. 2007. Competition and predation as mechanisms for displacement of greenback cutthroat trout by brook trout. Transactions of the American Fisheries Society. 136: 1381–1392. https://doi.org/10.1577/T07-017.1

Moody, J.A.; Martin, D.A. 2001. Initial hydrologic and geomorphic response following a wildfire in the Colorado Front Range. Earth Surface Processes and Landforms: The Journal of the British Geomorphological Research Group. 26: 1049–1070. https://doi.org/10.1002/esp.253

Naeser, R.B.; Smith, M.G. 1995. Playing with borrowed water: Conflicts over instream flows on the Upper Arkansas river. Natural Resources Journal. 35 (1): 93-110). https://www.jstor.org/stable/24885635

Nesler, T.P. 1981. Studies of the limnology, fish populations, and fishery of Turquoise Lake, Colorado—1979–80. Report no. REC-ERC-82-5. Denver, CO: U.S. Department of Interior, Bureau of Reclamation. 31 p.

O'Neill, M.P.; Schmidt, J.C.; Dobrowolski, J.P.; [et al.]. 1997. Identifying sites for riparian wetland restoration: Application of a model to the Upper Arkansas River Basin. Restoration Ecology. 5: 85–102. https://doi.org/10.1111/j.1526-100X.1997.tb00208.x

Oyler, J.W.; Ballantyne A.; Jencso K.; Sweet M.; Running S.W. 2014. Creating a topoclimatic daily air temperature dataset for the conterminous United States using homogenized station data and remotely sensed land skin temperature. International Journal of Climatology. 35:2258-2279. http://dx.doi.org/10.1002/joc.4127. https://doi.org/10.1002/joc.4127

Painter, T.H.; Skiles, S.M.; Deems, J.S.; [et al.]. 2018. Variation in rising limb of Colorado River snowmelt runoff hydrograph controlled by dust radiative forcing in snow. Geophysical Research Letters. 45: 797–808. https://doi.org/10.1002/2017GL075826

Perry, L.G.; Shafroth, P.B.; Hay, L.E.; [et al.]. 2019. Projected warming disrupts the synchrony of riparian seed release and snowmelt streamflow. New Phytologist. 225: 693–712. https://doi.org/10.1111/nph.16191

Pielke, R.A.; Doesken, N.; Bliss, O.; [et al.]. 2005. Drought 2002 in Colorado: An unprecedented drought or a routine drought? Pure and Applied Geophysics. 162: 1455–1479. https://doi.org/10.1007/s00024-005-2679-6

Poff, N.L.; Allan, J.D.; Bain, M.B.; [et al.]. 1997. The natural flow regime. BioScience. 47: 769-784. https://doi.org/10.2307/1313099

Pollock, M.M.; Beechie, T.J.; Wheaton, J.M.; [et al.]. 2014. Using beaver dams to restore incised stream ecosystems. BioScience. 64: 279–90. https://doi.org/10.1093/biosci/biu036

Polvi, L.E.; Wohl, E.E.; Merritt, D.M. 2011. Geomorphic and process domain controls on riparian zones in the Colorado Front Range. Geomorphology. 125: 504–516. https://doi.org/10.1016/j.geomorph.2010.10.012

Rasmussen, R.; Ikeda, K.; Liu, C.; [et al.]. 2014. Climate change impacts on the water balance of the Colorado headwaters: High-resolution regional climate model simulations. Journal of Hydrometeorology. 15: 1091–1116. https://doi.org/10.1175/JHM-D-13-0118.1

Rasmussen, R.; Liu, C.; Ikeda, K.; [et al.]. 2011. High-resolution coupled climate runoff simulations of seasonal snowfall over Colorado: A process study of current and warmer climate. Journal of Climate. 24: 3015–3048. https://doi.org/10.1175/JHM-D-13-0118.1

Rathburn, S.L.; Merritt, D.M.; Wohl, E.E.; [et al.]. 2009. Characterizing environmental flows for maintenance of river ecosystems: North Fork Cache la Poudre River, Colorado. Management and restoration of fluvial systems with broad historical changes and human impacts. 451: 143. https://doi.org/10.1130/2009.2451(10)

Rhoades, C.C.; Entwistle, D.; Butler, D. 2011. The influence of wildfire extent and severity on streamwater chemistry, sediment, and temperature following the Hayman Fire, Colorado. International Journal of Wildland Fire. 20: 430–442. https://doi.org/10.1071/WF09086

Rice, J.R.; Winters, D.; Regan, C.; [et al.]. 2018. Aquatic, riparian, and wetland ecosystems in glaciated valleys: Vulnerability to nonclimate and climate stressors in the U.S. Forest Service Rocky Mountain Region. In: Rice, J.R.; Joyce, L.A.; Regan, C.; Winters, D.; Truex, R. Climate change vulnerability assessment of aquatic and terrestrial ecosystems in the U.S. Forest Service Rocky Mountain Region. Gen. Tech. Rep. RMRS-GTR-376. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 216 p. https://www.fs.usda.gov/research/treesearch/56392

Richer, E.E.; Gates, E.A.; Kondratieff, M.C.; [et al.]. 2019 Modelling changes in trout habitat following stream restoration. River Research and Applications. 35: 680–691. https://doi.org/10.1002/rra.3444

Rieman, B.E.; Isaak, D.J. 2010. Climate change, aquatic ecosystems, and fishes in the Rocky Mountain West: Implications and alternatives for management. Gen. Tech. Rep. RMRS-GTR-250. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 46 p. https://doi.org/10.2737/RMRS-GTR-250

Riggan, P.J.; Lockwood, R.N.; Jacks, P.M.; [et al.]. 1994. Effects of fire severity on nitrate mobilization in watersheds subject to chronic atmospheric deposition. Environmental Science & Technology. 28: 369–375. https://doi.org/10.1021/es00052a005

Roberts, J.J.; Fausch, K.; 2015. Consequences of climate change for mountain lakes and native cutthroat trout. Denver, CO: Colorado Parks and Wildlife. 29 p. https://cpw.state.co.us/Documents/Research/Aquatic/CutthroatTrout/Roberts2015FinalReport.pdf [Accessed 6 June 2024]

Roberts, J.J.; Fausch, K.D.; Peterson, D.P.; [et al.]. 2013. Fragmentation and thermal risks from climate change interact to affect persistence of native trout in the Colorado River basin. Global Change Biology. 19: 1383–1398. https://doi.org/10.1111/gcb.12136

Roberts, J.J.; Fausch, K.D.; Hooten, M.B.; [et al.]. 2017a. Non-native trout invasions combined with climate change threaten persistence of isolated cutthroat trout populations in the southern Rocky Mountains. North American Journal of Fisheries Management. 37: 314–325. https://doi.org/10.1080/02755947.2016.1264507

Roberts, J.J.; Fausch, K.D.; Schmidt, T.S.; [et al.]. 2017b. Thermal regimes of Rocky Mountain lakes warm with climate change. PLoS One. 12: p.e0179498. https://doi.org/10.1371/journal.pone.0179498

Rogers, K. 2012. Piecing together the past: Using DNA to resolve the heritage of our state fish. Colorado Outdoors. September/October: 28-32. https://cpw.state.co.us/Documents/Research/Aquatic/CutthroatTrout/Rogers2012CoOutdoors.pdf [Accessed 6 June 2024]

Rupp, D.E.; Abatzoglou, J.T.; Hegewisch, K.C. [et al.]. 2013. Evaluation of CMIP5 20th century climate simulations for the Pacific Northwest USA. Journal of Geophysical Research: Atmospheres. 118(19): 10–884. https://doi.org/10.1002/jgrd.50843

Ryan, W.; Doesken, N.; Center, C.C. 2013. Drought of 2012 in Colorado. Colorado Climate Center. 7 p. https://climate.colostate.edu/pdfs/drought_2012.pdf [Accessed 6 June 2024]

Salas, J.D.; Fu, C.; Cancelliere, A.; [et al.]. 2005. Characterizing the severity and risk of drought in the Poudre River, Colorado. Journal of Water Resources Planning and Management. 131: 383–393. https://doi.org/10.1061/(ASCE)0733-9496(2005)131:5(383)

Saunders III, J.F.; Lewis Jr., W.M. 2003. Implications of climatic variability for regulatory low flows in the South Platte River Basin, Colorado. JAWRA, Journal of the American Water Resources Association. 39: 33–45. https://doi.org/10.1111/j.1752-1688.2003.tb01559.x

Smith, H.; Beauvais, D.G.P.; Keinath, D.A. 2004. Species assessment for Preble's meadow jumping mouse (*Zapus hudsonius preblei*) in Wyoming. Laramie, WY: Wyoming Natural Diversity Database, University of Wyoming. https://www.uwyo.edu/wyndd/_files/docs/reports/speciesassessments/preblesmeadowjumpingmouse-oct2004.pdf [Accessed 6 June 2024]

Smith, D.M.; Finch, D.M. 2014. Use of native and non-native nest plants by riparian-nesting birds along two streams in New Mexico. River Research and Applications. 30: 1134–1145. https://doi.org/10.1002/rra.2713

Sprague, L.A. 2005. Drought effects on water quality in the South Platte River Basin, Colorado. JAWRA, Journal of the American Water Resources Association. 41: 11–24. https://doi.org/10.1111/j.1752-1688.2005.tb03713.x

Springer, A.E.; Stevens, L.E. 2009. Spheres of discharge of springs. Hydrogeology Journal. 17: 83–94. https://doi.org/10.1007/s10040-008-0341-y

Stevens L.E.; Meretsky, V.J., eds. 2008. Aridland springs in North America: Ecology and conservation. Tucson, AZ: University of Arizona Press.

Stromberg, J. C.; Lite, S.J.; Dixon, M.D. 2010. Effects of stream flow patterns on riparian vegetation of a semiarid river: Implications for a changing climate. River Research and Applications. 26: 712–29. https://doi.org/10.1002/rra.1272

Topper, R.; Spray, K.L.; Bellis, W.H.; [et al.]. 2003. SP-53 Ground water atlas of Colorado. Special Publications, SP-53. Denver, CO: Colorado Geological Survey, Division of Minerals and Geology, Department of Natural Resources.

Udall, B.; Overpeck, J. 2017. The 21st century Colorado River hot drought and implications for the future. Water Resources Research. 53: 2404–2418. https://doi.org/10.1002/2016WR019638

U.S. Department of the Interior, Bureau of Reclamation (Reclamation). 2014. Downscaled CMIP3 and CMIP5 climate and hydrology projections: Release of hydrology projections, comparison with preceding information, and summary of user needs. Denver, CO: U.S. Department of the Interior, Bureau of Reclamation, Technical Services Center. 110 pp. https://gdo-dcp.ucllnl.org/downscaled_cmip_projections/techmemo/BCSD5HydrologyMemo.pdf [Accessed 6 June 2024]

Vose, Russell S.; Applequist, S.; Squires, M.; Durre, I.; Menne, M. J.; Williams, C. N., Jr.; Fenimore, C.; Gleason, K.; Arndt, D. 2014. NOAA Monthly U.S. Climate Divisional Database (NClimDiv). NOAA National Climatic Data Center. https://doi.org/10.7289/V5M32STR

Walch, L.A. 1980. Movements of lake trout in Twin Lakes, Colorado, in relation to the Mt. Elbert pumped-storage powerplant. Applied Sciences Branch, Division of Research, Engineering and Research Center, U.S. Department of the Interior, Water and Resources Service. 82 p.

Watson, P.S.; Davies, S. 2011. Modeling the effects of population growth on water resources: A CGE analysis of the South Platte River Basin in Colorado. The Annals of Regional Science. 46: 331–348. https://doi.org/10.1007/s00168-009-0326-3

Weaver, J.F.; Gruntfest, E.; Levy, G.M. 2000. Two floods in Fort Collins, Colorado: Learning from a natural disaster. Bulletin of the American Meteorological Society. 81: 2359–2366. https://doi.org/10.1175/1520-0477(2000)081<2359:TFIFCC>2.3.CO;2

Williams, A.P.; Allen, C.D.; Macalady, A.K.; [et al.]. 2013. Temperature as a potent driver of regional forest drought stress and tree mortality. Nature Climate Change. 3(3): 292. https://doi.org/10.1038/nclimate1693

Williams, A.P.; Cook, E.R.; Smerdon, J.E.; [et al.]. 2020. Large contribution from anthropogenic warming to an emerging North American megadrought. Science. 368: 314–318. https://doi.org/10.1126/science.aaz9600

Williams, M.W.; Hood, E.; Molotch, N.P.; [et al.]. 2015a. The “Teflon basin” myth: Hydrology and hydrochemistry of a seasonally snow-covered catchment. Plant Ecology & Diversity. 8: 639–661. https://doi.org/10.1080/17550874.2015.1123318

Williams, M.W.; Knauf, M.; Caine, N.; [et al.]. 2006. Geochemistry and source waters of rock glacier outflow, Colorado Front Range. Permafrost and Periglacial Processes. 17: 13–33. https://doi.org/10.1002/ppp.535

Williams, J.E.; Neville, H.M.; Haak, A.L.; [et al.]. 2015b. Climate change adaptation and restoration of western trout streams: Opportunities and strategies. Fisheries. 40: 304–317. https://doi.org/10.1080/03632415.2015.1049692

Winward, A.H. 2000. Monitoring the vegetation resources in riparian areas. Gen. Tech. Rep. RMRS-GTR-47. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 49 p. https://doi.org/10.2737/RMRS-GTR-47

Wohl, E.E. 2001. Virtual rivers: Lessons from the mountain rivers of the Colorado Front Range. New Haven, CT: Yale University Press.

Wohl, E.; Dwire, K.; Sutfin, N.; [et al.]. 2012. Mechanisms of carbon storage in mountainous headwater rivers. Nature Communications. 3: 1263. https://doi.org/10.1038/ncomms2274

Wondzell, S.M.; Diabat, M.; Haggerty, R.. 2019. What matters most: Are future stream temperatures more sensitive to changing air temperatures, discharge, or riparian vegetation? JAWRA, Journal of the American Water Resources Association. 55: 116–132. https://doi.org/10.1111/1752-1688.12707

Woodbury, M.; Baldo, M.; Yates, D.; [et al.]. 2012. Joint Front Range climate change vulnerability study. Denver, CO: Water Research Foundation. 148 p. https://climatewise.org/images/stories/pdfs/Publications/ClimateWise/Joint_Front_Range_Climate_Vulnerability_Study.pdf [Accessed 6 June 2024]

Woodward, D.F.; Farag, A.M.; Little, E.E.; [et al.]. 1991. Sensitivity of greenback cutthroat trout to acidic pH and elevated aluminum. Transactions of the American Fisheries Society. 120: 34–42. https://doi.org/10.1577/1548-8659(1991)120<0034:SOGCTT>2.3.CO;2

Wrigley, M.J.; White, M.; Elliott, B.; [et al.]. 2012. Threatened, endangered, and Forest Service sensitive species on the Pike and San Isabel National Forests (updated June 2012). Unpublished Report on file with: U.S. Department of Agriculture, Forest Service, Pike and San Isabel National Forests and Comanche and Cimarron National Grasslands, Salida, Colorado. 115 p.

Writer, J.H.; McCleskey, R.B.; Murphy, S.F.; [et al.]. 2012. Effects of wildfire on source-water quality and aquatic ecosystems, Colorado Front Range. Wildfire and water quality: Processes, impacts and challenges. 354: 117–122. https://iahs.info/uploads/dms/16029.18-117-122-354-09-Writer-et-al-CORR.pdf [Accessed 6 June 2024]

Young, M.K. 2009. Greenback cutthroat trout (*Oncorhynchus clarkii stomias*): A technical conservation assessment. Golden, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Region. 100 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5206801.pdf [Accessed 6 June 2024]

Young, M.K.; Harig, A.L. 2001. A critique of the recovery of greenback cutthroat trout. Conservation Biology. 15: 1575–1584. https://www.jstor.org/stable/3061259

Zhang, Q.; Knowles, J.F.; Barnes, R.T.; [et al.]. 2018. Surface and subsurface water contributions to streamflow from a mesoscale watershed in complex mountain terrain. Hydrological Processes. 32: 954–967. https://doi.org/10.1002/hyp.11469

Appendix

Datasets used in water resources vulnerability analysis.

1. Watershed Elevation Range—Maximum and minimum elevation of HUC-8 watershed. Source: Colorado Natural Heritage Program (https://cnhp.colostate.edu)
2. Percent National Forest Land—Percentage of HUC8 watershed with AR or PSI National Forest land. Source: Colorado Natural Heritage Program (https://cnhp.colostate.edu)
3. Percent Private Land—Percentage of HUC8 watershed with privately owned land. Source: Colorado Natural Heritage Program (https://cnhp.colostate.edu)
4. Decreed Instream Flow Miles—Total length of stream miles with instream flow water rights managed by the Colorado Water Conservation Board. Source: Colorado's Decision Support System (https://dwr.state.co.us/Tools)
5. Mean Discharge—Mean daily discharge volume, in cubic feet per second, measured at stream gages across the period of record. Source: U.S. Geological Survey National Water Information System (https://waterdata.usgs.gov/nwis)
6. Max. Discharge—Maximum daily discharge volume, in cubic feet per second, measured at stream gages across the period of record. Source: U.S. Geological Survey National Water Information System (https://waterdata.usgs.gov/nwis)
7. Mean Peak Date—Mean calendar date, across years, when the annual peak daily discharge occurred at gage sites. Source: U.S. Geological Survey National Water Information System (https://waterdata.usgs.gov/nwis)
8. Mean Center of Flow Mass Date—Mean date that 50% of cumulative water year flow was recorded at stream gages. Source: U.S. Geological Survey National Water Information System (https://waterdata.usgs.gov/nwis)
9. Peak Discharge CV—Coefficient of variation, representing variation in peak daily discharge among years. Source: U.S. Geological Survey National Water Information System (https://waterdata.usgs.gov/nwis)
10. Peak Date CV—Coefficient of variation, representing variation in peak discharge date among years. Source: U.S. Geological Survey National Water Information System (https://waterdata.usgs.gov/nwis)

11. Center of Flow Mass CV—Coefficient of variation, representing variation in center of flow mass timing among years. Source: U.S. Geological Survey National Water Information System (https://waterdata.usgs.gov/nwis)
12. Cutthroat Trout Reaches—Number of stream reaches with managed populations of cutthroat trout. Source: Inland Cutthroat Trout Program (https://icp.wygisc.org)
13. Perennial Stream Miles—Total length of perennially flowing stream reaches. Source: U.S. Geological Survey National Hydrography Dataset (https://www.usgs.gov/core-science-systems/ngp/national-hydrography/nhdplus-high-resolution)
14. Intermittent Stream Miles—Total length of stream reaches with surface flows occurring only during periods of runoff. Source: U.S. Geological Survey National Hydrography Dataset (https://www.usgs.gov/core-science-systems/ngp/national-hydrography/nhdplus-high-resolution)
15. Mapped Wetlands—Acres of wetlands mapped and classified by the National Wetland Inventory and the Colorado Natural Heritage Program. Source: Colorado Natural Heritage Program (https://cnhp.colostate.edu/cwic)
16. Mapped Fens—Acres of fens calculated using numerous datasets compiled by the Colorado Natural Heritage Program. Source: Colorado Natural Heritage Program (https://cnhp.colostate.edu/cwic)
17. Mapped Springs—Number of known springs. Source: Spring Stewardship Institute (https://springstewardshipinstitute.org/)
18. Forest Service Fishing and Boating Sites—Recreation sites primarily used for access to streams and lakes. Source: U.S. Forest Service National Dataset (https://data.fs.usda.gov/geodata/)
19. Active Ditches and Pipelines—Structures used to divert water from streams and springs. Source: Colorado's Decision Support System (https://dwr.state.co.us/Tools)
20. Active Reservoirs—Impoundments created for water storage. Source: Colorado's Decision Support System (https://dwr.state.co.us/Tools)
21. Active Wells—Sites developed for groundwater use. Source: Colorado's Decision Support System (https://dwr.state.co.us/Tools)

22. Change in Maximum Summer Air Temperature—Projected change in maximum air temperature during June, July, and August between the historical period (1985-2015) and a future period (2035-2050) under four climate scenarios. Source: U.S. Bureau of Reclamation Climate and Hydrology Projections (https://gdo-dcp.ucllnl.org/downscaled_cmip_projections/)

23. Change in Spring Snow Water Equivalent—Projected change in snow water equivalent during March, April, and May between the historical period (1985-2015) and a future period (2035-2050) under four climate scenarios. Source: U.S. Bureau of Reclamation Climate and Hydrology Projections (https://toolkit.climate.gov/tool/downscaled-cmip3-and-cmip5-climate-and-hydrology-projections)

24. Change in Spring Runoff—Projected change in runoff during March, April, and May between the historical period (1985-2015) and a future period (2035-2050) under four climate scenarios. Source: U.S. Bureau of Reclamation Climate and Hydrology Projections (https://toolkit.climate.gov/tool/downscaled-cmip3-and-cmip5-climate-and-hydrology-projections)

25. Change in Summer Runoff—Projected change in runoff during June, July, and August between the historical period (1985-2015) and a future period (2035-2050) under four climate scenarios. Source: U.S. Bureau of Reclamation Climate and Hydrology Projections (https://toolkit.climate.gov/tool/downscaled-cmip3-and-cmip5-climate-and-hydrology-projections)

26. Change in Annual Runoff—Projected change in runoff between the historical period (1985-2015) and a future period (2035-2050) under four climate scenarios. Source: U.S. Bureau of Reclamation Climate and Hydrology Projections (https://toolkit.climate.gov/tool/downscaled-cmip3-and-cmip5-climate-and-hydrology-projections)

27. Change in Fire Danger—Projected change in number of fires between the historical period (1985-2015) and a future period (2035-2050) under four climate scenarios. Source: Multivariate Adaptive Constructed Analogs Climate Projections (http://www.climatologylab.org/maca.html)

28. Change in Drought Frequency—Projected change in frequency of severe droughts like 2012 between the historical period (1985-2015) and a future period (2035-2050) under four climate scenarios. Source: Multivariate Adaptive Constructed Analogs Climate Projections (http://www.climatologylab.org/maca.html)

29. Change in Extreme Rainfall Events—Projected percent change in extreme rainfall events between the historical period (1985-2015) and a future period (2035-2050) under four climate scenarios. Source: Multivariate Adaptive Constructed Analogs Climate Projections (http://www.climatologylab.org/maca.html)

30. Change in Cold-water Cutthroat Trout Habitat—Projected change in miles of thermally suitable cutthroat trout streams between the historical period of 1970-1999 and two projection periods (2030-2050 and 2070-2099) under the A1B climate scenario. Source: U.S. Forest Service Climate Shield (https://www.fs.usda.gov/rm/boise/AWAE/projects/ClimateShield.html)

31. Change in Mean Summer Stream Volume—Projected change in mean daily stream flow volume during the summer months between the historical period of 1970-1999 and two projection periods (2030-2050 and 2070-2099) under the A1B climate scenario. Source: U.S. Forest Service Climate Western Streamflow Metrics (https://www.fs.usda.gov/rm/boise/AWAE/projects/modeled_stream_flow_metrics.shtml)

32. Change in Center of Flow Mass Timing—Projected change, in number of days, of the center of flow mass timing between the historical period of 1970-1999 and two projection periods (2030-2050 and 2070-2099) under the A1B climate scenario. Source: U.S. Forest Service Climate Western Streamflow Metrics (https://www.fs.usda.gov/rm/boise/AWAE/projects/modeled_stream_flow_metrics.shtml)

33. Change in Mean Annual Stream Volume—Change in mean annual stream flow volume between the historical period of 1970-1999 and two projection periods (2030-2050 and 2070-2099) under the A1B climate scenario. Source: U.S. Forest Service Climate Western Streamflow Metrics (https://www.fs.usda.gov/rm/boise/AWAE/projects/modeled_stream_flow_metrics.shtml)

Vulnerability of Major Colorado Front Range Tree Species to Climate Change

Paula J. Fornwalt, Andreas P. Wion, Anna W. Schoettle, Miranda D. Redmond, Steven K. Alton, Javier E. Mercado, Brice B. Hanberry, José F. Negrón

Introduction

Forests and woodlands are a vital component of the Colorado Front Range landscape. Occupying millions of acres, Front Range forests and woodlands support myriad species (Latif et al. 2020; Peet 1981; Simonson et al. 2001) and regulate geological, hydrological, and chemical processes (Bradford et al. 2008; Burns 2004; Troendle and King 1985). They also provide the people of the Front Range and beyond with ecosystem services ranging from timber, food, and water production to recreational opportunities (Clement and Cheng 2011).

Climate change over the next several decades is expected to profoundly impact Front Range forests and woodlands through both direct and indirect mechanisms. Indeed, climate change and its impacts have already been observed. Mean annual temperatures during the last two decades averaged 1.5 °F (0.8 °C) warmer than those of the 20th century (Rangwala 2024, this report). While annual precipitation totals in the last two decades do not show a clear deviation from the long-term trend, warmer temperatures have led to an increase in measures of water stress, like climatic water deficit and vapor pressure deficit (Andrus et al. 2021; Rangwala 2024; Rodman et al. 2020a). These climatic changes have directly contributed to higher mature tree mortality rates and to lower tree seedling establishment rates (Andrus et al. 2018, 2021; Worrall et al. 2008). Mature tree mortality due to wildfires and insect and disease outbreaks, disturbances which themselves bear the marks of recent climatic changes, has also increased, and postdisturbance tree establishment has declined (Chapman et al. 2012; Lalande et al. 2020; Rodman et al. 2020a). Although precise details about the nature of future climate are unknown, additional climatic changes are nonetheless predicted to bring even more direct and indirect impacts to Front Range forests and woodlands (Crookston 2009; Malone et al. 2018b; Monahan et al. 2013; Rodman et al. 2020a; Temperli et al. 2015).

Detailed assessments of species' vulnerabilities to climate change—that is, the degree to which species are susceptible to, and unable to cope with, climate change effects (IPCC 2007)—can help land managers incorporate the anticipated effects of climate change into their planning and decision documents. In this chapter, we used the best available scientific information to assess the vulnerability of nine major tree species in the Colorado Front Range to climate change. We broke the chapter into four main sections, with the first two describing our vulnerability assessment area and assessment approach, and the last two containing our assessments and conclusions.

Vulnerability Assessment Area

Our assessment area was the forested portion of the Colorado Front Range. As in the other chapters in this General Technical Report, we delineated the Front Range as a rectangular area surrounding the Pike-San Isabel (PSI) and the Arapaho-Roosevelt (AR) National Forests (fig. 1). A large proportion of the forested lands in the Front Range is under the management of these National Forests. Other forested public lands include those managed by the National Park Service, the Bureau of Land Management, and the State of Colorado. A small but noteworthy proportion of the forested lands in the Front Range is managed by local government agencies or is privately owned.

The Front Range encompasses three broadly defined major forest and woodland vegetation types, or macrogroups (table 1; fig. 1) (USNVC 2020). The Southern Rocky Mountain Twoneedle Pinyon–Juniper Woodland Macrogroup (hereafter pinyon–juniper woodland) is primarily found in the foothills and lower montane zones of the southern Front Range. This vegetation type occupies only about 1 percent of the PSI-AR, or about 0.1 million ac (< 0.1 million ha), almost all of which is on the PSI (ARP 2019; PSICC 2019). However, it is quite common on lower-elevation neighboring lands. The most ubiquitous tree species is twoneedle pinyon (*Pinus edulis*) (fig. 2). The Southern Rocky Mountain Lower Montane Forest Macrogroup (hereafter montane forest) occupies the lower montane zone of the Front Range. This vegetation type covers approximately 0.9 million ac (0.4 million ha), or 20 percent, of the PSI-AR (ARP 2019; PSICC 2019). The major tree species are ponderosa pine (*Pinus ponderosa* var. *scopulorum*) and Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) (fig. 3a-b). Finally, the Rocky Mountain Subalpine–High Montane Forest Macrogroup (hereafter subalpine forest) occurs from the upper portions of the montane zone to the subalpine zone. It occurs on about 41 percent of the PSI-AR, or on about 1.9 million ac (0.8 million ha) (ARP 2019; PSICC 2019). Quaking aspen (*Populus tremuloides*), lodgepole pine (*Pinus contorta* var. *latifolia*), limber pine (*Pinus flexilis*), Rocky Mountain bristlecone pine (*Pinus aristata*), subalpine fir (*Abies lasiocarpa* var. *lasiocarpa*), and Engelmann spruce (*Picea engelmannii* var. *engelmannii*) are the most common tree species (fig. 4a-e). The remaining 38 percent of the PSI-AR, encompassing 1.7 million ac (0.7 million ha), is primarily composed of vegetation types dominated by graminoids, forbs, and/or shrubs; it also includes minor tree-dominated vegetation types (e.g., riparian forests dominated by narrowleaf cottonwood [*Populus angustifolia*]), undefined vegetation types, and water.

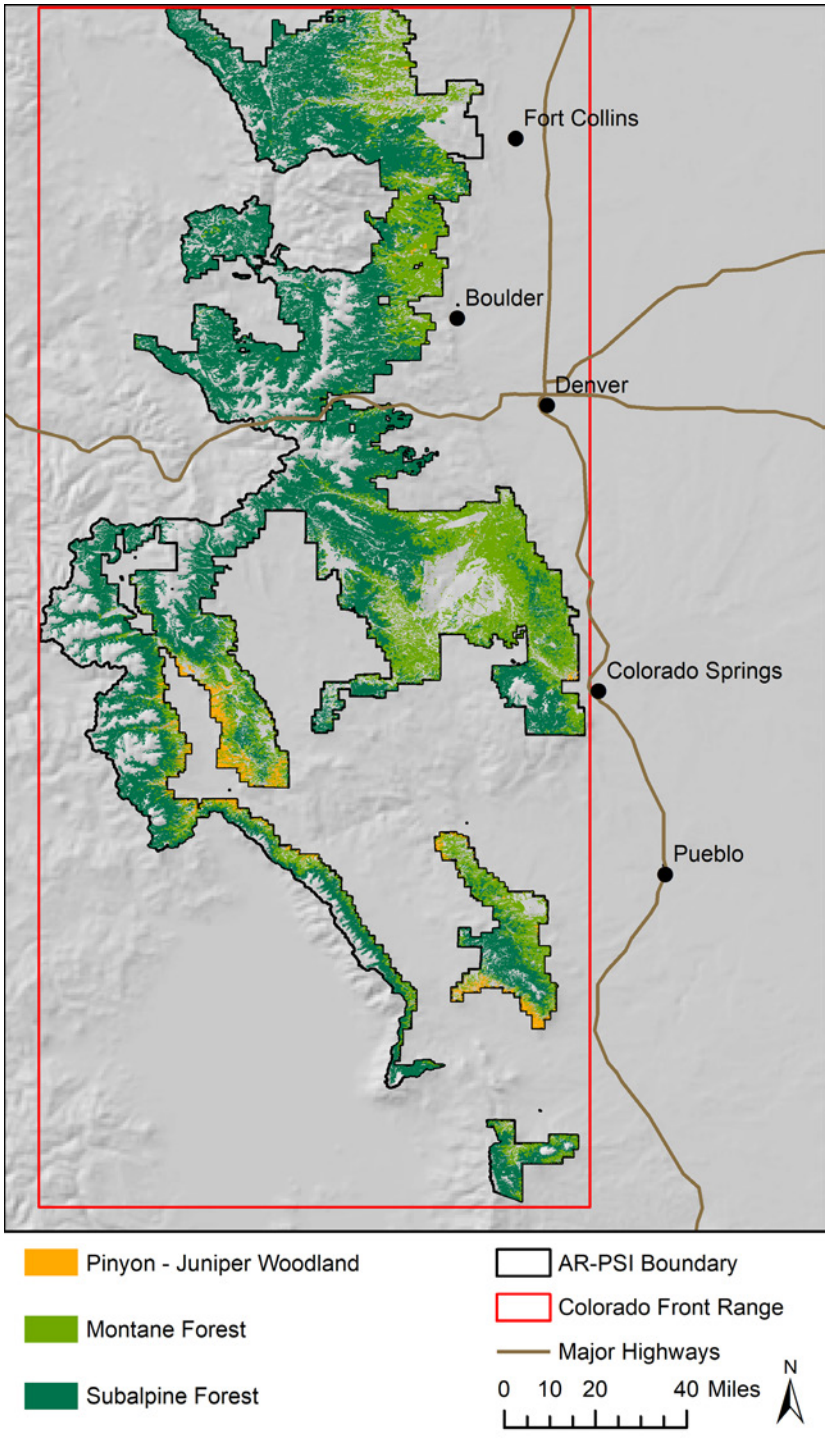


Figure 1—Map of the current distribution of the three major forest and woodland vegetation types on the Pike-San Isabel (PSI) and Arapaho-Roosevelt (AR) National Forests in the Colorado Front Range (ARP 2019; PSICC 2019). The vegetation types were derived by crosswalking USDA Forest Service Region 2 cover types to U.S. National Vegetation Classification macrogroups as described in table 1 (ARP 2019; PSICC 2019; USNVC 2020). Note that in some places the map reflects changes stemming from recent disturbances, such as the conversion of forest to nonforest by stand-replacing wildfire (e.g., the Hayman Fire northwest of Colorado Springs).

Table 1—The three major forest and woodland vegetation types of the Colorado Front Range. Vegetation types were defined by U.S. National Vegetation Classification (USNVC) macrogroups and Forest Service Region 2 (USFS R2) cover types (ARP 2019; PSICC 2019; USNVC 2020). The Front Range was defined as a box bounding the Arapaho-Roosevelt (AR) and Pike-San Isabel (PSI) National Forests.

Vegetation type	Description
Pinyon-juniper woodland	The Southern Rocky Mountain Twoneedle Pinyon–Juniper Woodland Macrogroup (M897; hereafter pinyon-juniper woodland) occurs in the foothill and lower montane zones of the southern Front Range. It is present on about 0.1 million ac (< 0.1 million ha) of the PSI-AR National Forests; almost all of this is on the PSI (fig. 1). Twoneedle pinyon is the major tree species. Oneseed juniper and Rocky Mountain juniper are also common, with the former tending to be more prevalent on lower-elevation lands surrounding the PSI and the latter tending to be more prevalent on the PSI itself. Other species such as ponderosa pine and Douglas-fir are sometimes also present. We crosswalked this vegetation type to the USFS R2 pinyon-juniper cover type.
Montane forest	The Southern Rocky Mountain Lower Montane Forest Macrogroup (M022; hereafter montane forest) occurs throughout the lower montane zone of the Front Range. It covers about 0.9 million ac (0.4 million ha) of the PSI-AR (fig. 1). The major tree species are ponderosa pine and Douglas-fir. Occasionally other species, such as Rocky Mountain juniper, quaking aspen, white fir, and blue spruce, mix into the canopy or are dominant. The vegetation type was crosswalked to four USFS R2 cover types: ponderosa pine; Douglas-fir; white fir; and blue spruce.
Subalpine forest	The Rocky Mountain Subalpine–High Montane Forest Macrogroup (M020; hereafter subalpine forest) captures the upper montane and subalpine zones of the Front Range. It occurs on about 1.9 million ac (0.8 million ha) of the PSI-AR (fig. 1). The major tree species in this macrogroup are quaking aspen, lodgepole pine, limber pine, bristlecone pine, subalpine fir, and Engelmann spruce. Douglas-fir and other species can sometimes also occur. We crosswalked this vegetation type to five USFS R2 cover types: aspen; lodgepole pine; limber pine; bristlecone pine; and spruce-fir.



Figure 2—Twoneedle pinyon, a major tree species of the pinyon-juniper woodland vegetation type, is abundant at this Bureau of Land Management site near Salida, Colorado. (USDA Forest Service photo by Mike Battaglia)



Figure 3a-b—Major tree species of the montane forest vegetation type. (a) Ponderosa pine dominates the landscape at the Manitou Experimental Forest on the Pike National Forest. (b) This Douglas-fir survived the 2002 Hayman Fire on the Pike National Forest. (USDA Forest Service photos by Paula Fornwalt.)

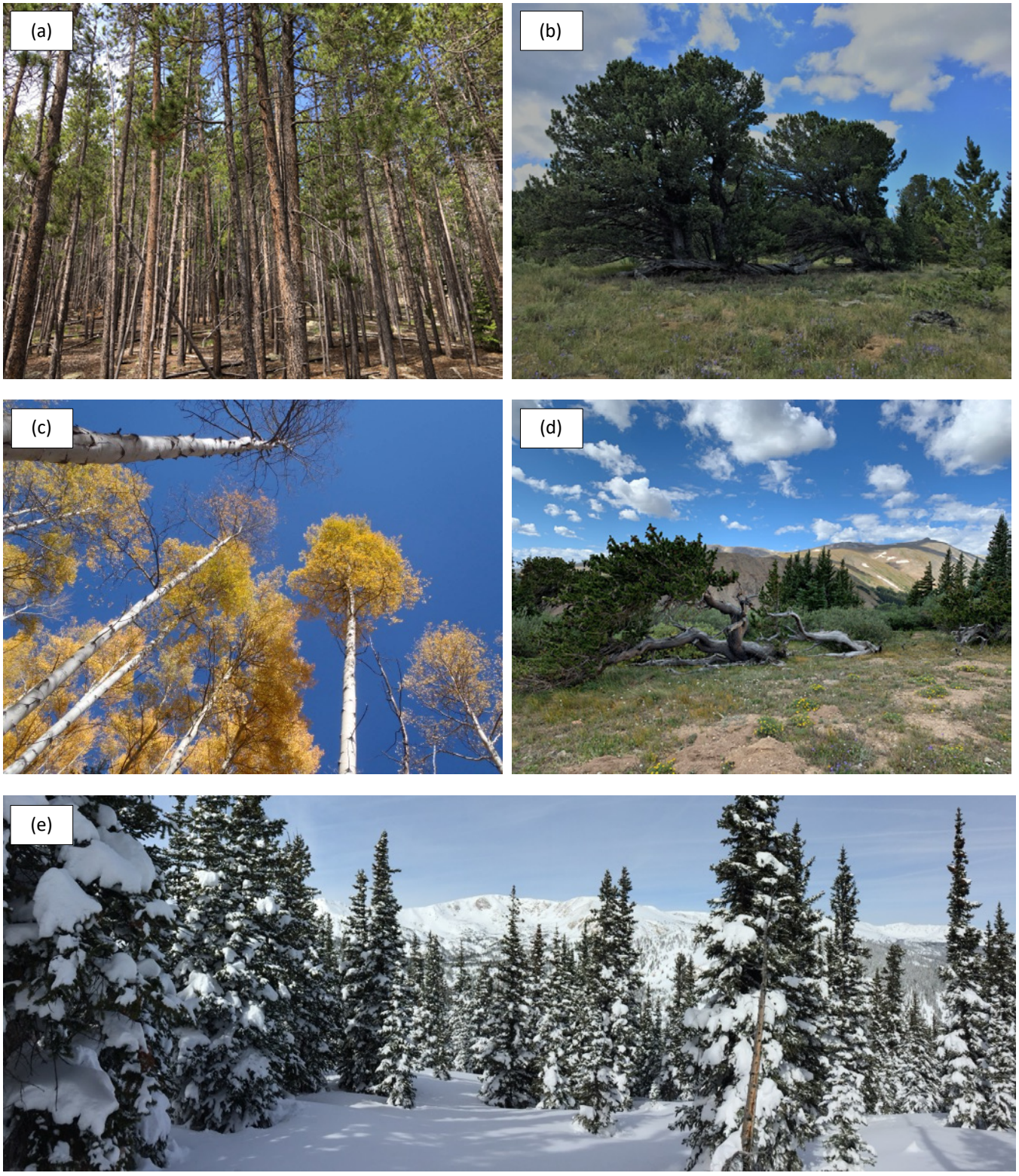


Figure 4a-e—Major tree species of the subalpine forest vegetation type. (a) Lodgepole pine occupies the upper portions of Crosier Mountain, Roosevelt National Forest. (b) Limber pine grows along the Brown's Lake Trail, Roosevelt National Forest. (c) Quaking aspen displays its fall colors at the Fraser Experimental Forest, Arapaho National Forest. (d) This twisted, old Rocky Mountain bristlecone pine helps form treeline on Pennsylvania Mountain, Pike National Forest. (e) Engelmann spruce and subalpine fir intermix near Uncle Bud's Hut, San Isabel National Forest. (USDA Forest Service photos by Paula Fornwalt (a, b, c, e) and John Frank (d))

Vulnerability Assessment Approach

We assessed the vulnerability of the above nine major Front Range tree species—twoneedle pinyon, ponderosa pine, Douglas-fir, quaking aspen, lodgepole pine, limber pine, Rocky Mountain bristlecone pine, subalpine fir, and Engelmann spruce—using the general framework described by the Intergovernmental Panel on Climate Change (IPCC) and others (Glick et al. 2011; IPCC 2001, 2007; Swanston et al. 2016). This framework examines vulnerability by evaluating the potential impacts of climate change on a natural resource, such as a single species, a community of species, or an ecosystem service, and the capacity of the resource to adapt to it (fig. 5).

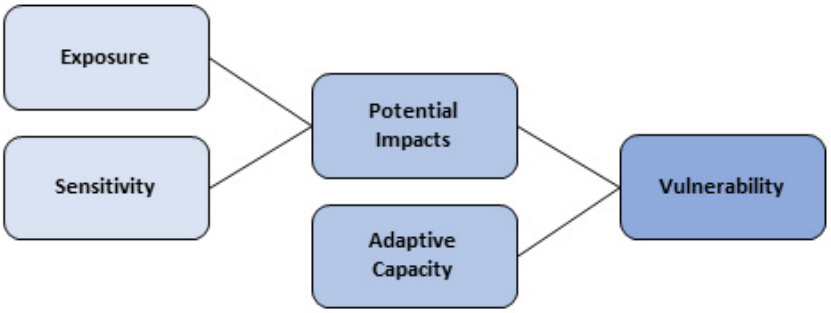


Figure 5—The components of climate change vulnerability. (Adapted from Glick et al. 2011.)

Potential impacts describe the consequences of climate change for a natural resource, and are a function of exposure, or how the climatic environment is expected to change, and sensitivity, or how the resource is expected to be affected by that change (fig. 5; Glick et al. 2011). Potential impacts can be either direct or indirect. Direct potential impacts are those of climate change per se, such as tree death due to warmer temperatures and more extreme weather events. Indirect potential impacts are those that are mediated through climate change's influence on other factors. Examples of indirect potential impacts are those due to a climate change-driven increase of the frequency, severity, or size of wildfires or insect and disease outbreaks.

Adaptive capacity reflects the mechanisms that a biological natural resource has to cope with climate change by enabling it to either persist in its current location or migrate to a more suitable location (Glick et al. 2011; Jump and Peñuelas 2005). One factor that can influence the adaptive capacity of a resource is its phenotypic plasticity, or its ability to maintain high fitness in the face of environmental change by altering its morphology, phenology, or physiology (Nicotra et al. 2010). A resource with high phenotypic plasticity is therefore expected to be better able to tolerate changes in climate. Another factor that can influence a resource's adaptive capacity is its genetic diversity (Jump et al. 2009). Genetic diversity reflects the potential to adapt to climate change through evolution; a resource with high genetic diversity is more likely to contain individuals with heritable traits that increase its tolerance of climate change. A resource that can readily shift its

range—for example, because it can disperse long distances or because its distribution is broad and connected—is expected to be better able to adapt to climate change by migrating to an area with a more favorable climate (Corlett and Westcott 2013).

Even within the context of the IPCC framework, vulnerability can be assessed through a variety of approaches. These range from qualitative approaches that are based on expert opinions to quantitative approaches that use vulnerability assessment tools to integrate input data. Our approach took inspiration from that of Keane et al. (2018) and drew from the relevant available scientific literature to produce vulnerability assessment narratives. We particularly relied on literature describing studies conducted within the Front Range, supplementing it as appropriate with literature describing studies conducted in comparable areas elsewhere. We then used these narratives to produce overall vulnerability ratings and accompanying confidence ratings, as Brandt et al. (2014), Butler et al. (2015), Janowiak et al. (2018), and others have done.

Vulnerability Assessment Narrative

Habitat

We began our vulnerability assessment narrative for each tree species with a basic discussion of its habitat. Its current distribution was described, both in the western United States and in the Front Range. Its preferred environmental conditions in the Front Range were also described, as were its common tree, graminoid, forb, and shrub associates. We acquired much of this information from classic references like *Silvics of North America* (Burns and Honkala 1990a,b). We also routinely took information from the Field Sampled Vegetation Database, also known as FSVeg (ARP 2019; PSICC 2019).

Ecology

Next, we synthesized the scientific literature to provide a foundational description of each species' basic ecology. Included in this portion of the vulnerability assessment narrative was information on its reproductive ecology, as well as information on its tolerance to drought, fire, insects, diseases, and other environmental stressors at seedling, sapling, and mature life stages.

Potential Direct and Indirect Impacts of Climate Change

We broke our discussion of potential impacts into one section on direct impacts and another section on indirect impacts. Observational, experimental, and modeling studies were all used as input in these sections. In particular, we made extensive use of the west-wide bioclimatic envelope models developed by Crookston (2009) for our nine species. Bioclimatic envelope models relate the current geographic distribution of a species (or a community of species) to current climate variables to identify the climatic envelope within which it occurs, and then project the geographic distribution of the climatic envelope under various future climate scenarios. Please see box 1 for more information about bioclimatic envelope models in general, and about our use of Crookston's (2009) models specifically.

Box 1

Bioclimatic envelope models

Bioclimatic envelope models are commonly used to assess the potential impacts of climate change on vegetation (Araújo and Peterson 2012). Bioclimatic envelope models relate the current geographic distribution of a species or a species assemblage to current climate variables to identify the climatic envelope under which it occurs. Potential climate change impacts are then explored by projecting the geographic distribution of the climatic envelope under future climate scenarios.

We drew from numerous studies that developed bioclimatic envelope models to investigate potential climate change impacts on Front Range tree species (e.g., Crookston 2009; Gray and Hamann 2013; Malone et al. 2018b). In particular, we made extensive use of a set of west-wide bioclimatic envelope maps developed by Crookston (2009) for more than 70 tree and shrub species. These maps depict the likelihood that climate at a given location and time period is (or is predicted to be) suitable for the species; values approaching 0 indicate that climate is very unsuitable and values approaching 1 indicate that climate is very suitable. West-wide maps for each of the 70 species were produced for the "current" (1990) climate and for seven potential future climates in 2030, 2060, and 2090.

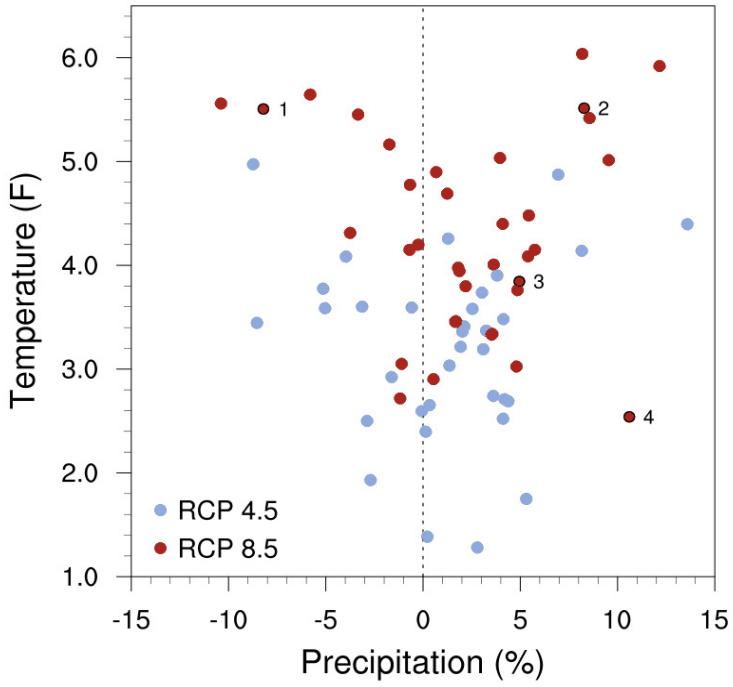
Unfortunately, none of the GCMs utilized by Crookston (2009) to represent the seven potential future climates were the same as those used in this GTR to represent the four potential future climate scenarios (Rangwala this report). However, HADCM3-A2, one of the GCMs used by Crookston (2009), seemed to crosswalk relatively well to CNRM-CM5.RCP85, the GCM representing the middle-of-the-road Hot scenario. In the Front Range, HADCM3-A2 predicts a ~4.5 °F (2.5 °C) increase in mean annual temperature and a 2 percent decrease in mean annual precipitation between the 2000 (1985–2015) and the 2050 (2035–2065) time periods. In comparison, CNRM-CM5.RCP85 predicts a ~3.8 °F (2.1 °C) increase in mean annual temperature and a 5 percent increase in mean annual precipitation between the 2000 and the 2050 time periods. We therefore chose to feature Crookston's (2009) bioclimate envelope maps under HADCM3-A2 in our chapter.

We integrated Crookston's (2009) current bioclimate envelope maps and the 2060 and 2090 bioclimate envelope maps under HADCM3-A2 to create Colorado-wide maps depicting potential future changes in climatic suitability for the major Front Range tree species (e.g., fig. 8). These changes were represented as four classes, as was done by Rehfeldt et al. (2015):

- Suitable climate lost (current suitability likelihood ≥ 0.5 and future suitability likelihood < 0.3);
- Suitable climate threatened (current suitability likelihood ≥ 0.5 and future suitability likelihood < 0.5 and ≥ 0.3);
- Suitable climate persistent (current and future suitability likelihood ≥ 0.5); and
- Suitable climate gained (current suitability likelihood < 0.5 and future suitability likelihood ≥ 0.5).

Bioclimatic envelope modeling is predicated on several assumptions (Araújo and Peterson 2012). Perhaps the most important assumptions are that distributions of species or species assemblages are governed first and foremost by climate, and that species or species assemblages inhabit all areas that are climatically suitable. While climate does in fact govern distributions at large spatial scales, non-climate factors such as species interactions and dispersal limitations may constrain them, particularly at small scales. For this reason, bioclimatic envelope models are best used at broad scales, such as across the Front Range.

To the extent the literature allowed, we examined potential impacts through the lens of four future climate scenarios, or exposures, for the 2035 to 2065 (hereafter 2050) time period. The scenarios, which are described in detail in Chapter 3, bracket the wide range of potential climates that have been projected for the Front Range by various CMIP5 general circulation models (GCMs; fig. 6; Rangwala 2024, this report). The first climate scenario, Hot (CNRM-CM5.rcp85), projects a future climate in which mean annual temperature is 3.8 °F (2.1 °C) warmer and mean annual precipitation is 5.0 percent greater than the 1985 to 2015 (hereafter 2000) baseline climate. Relative to the other three scenarios, the Hot scenario has the most middle-of-the-road 2050 temperature and precipitation anomalies, and so it may best represent a middle-of-the-road future climate scenario. The Very Hot and Wet (CanESM2.rcp85) climate scenario projects a 5.5 °F (3.1 °C) increase in mean annual temperature and an 8.3 percent increase in mean annual precipitation, while the Warm and Wet (GFDL-ESM2M.rcp85) climate scenario projects a 2.5 °F (1.4 °C) increase in mean annual temperature and a 10.6 percent increase in mean annual precipitation. The final climate scenario, Very Hot and Dry (IPSL-CM5A-MR.rcp85), projects a 5.5 °F (3.1 °C) increase and an 8.2 percent decrease in mean annual temperature and precipitation, respectively.



1 Very Hot & Dry (IPSL-CM5A-MR.rcp85)
2 Very Hot & Wet (CanESM2.rcp85)
3 Hot (CNRM-CM5.rcp85)
4 Warm & Wet (GFDL-ESM2M.rcp85)

Figure 6—Predicted changes in future (2050) precipitation and temperature on the Colorado Front Range based on the output from 34 CMIP5 general circulation models and two emissions scenarios (RCPs). The four future climate scenarios identified by Rangwala (this report), which approximately bracket the range of predicted changes, are labeled. The Front Range was defined as a box bounding the Pike-San Isabel (PSI) and Arapaho-Roosevelt (AR) National Forests. (Adapted from Rangwala, this report.)

Capacity to Adapt to Climate Change

We assessed adaptive capacity for each species by reviewing the scientific literature to gather available information on its phenotypic plasticity, its genetic diversity, and its ability to migrate. As with potential climate change impacts, we incorporated results from observational, experimental, and modeling studies.

Overall Vulnerability Rating

We assigned an overall climate change vulnerability rating of low, moderate, or high for each tree species by subjectively integrating the information on potential impacts and adaptive capacity (fig. 7a; Brandt et al. 2014; Butler et al. 2015; Janowiak et al. 2018). Vulnerability to climate change increased as potential impacts increased and as adaptive capacity decreased. We then qualified each vulnerability rating with a confidence rating (fig. 7b; Brandt et al. 2014; Butler et al. 2015; Janowiak et al. 2018). The confidence rating characterizes both the amount of information supporting the vulnerability rating (limited, moderate, abundant) and the level of agreement among that information (low, moderate, high). The amount of information was considered limited when the vulnerability rating was supported by a small number of studies, and abundant when it was supported by numerous studies that collectively employed a range of observational, experimental, and modeling approaches. Agreement among the information was considered low when individual studies generally led to different vulnerability ratings, and high when they led to similar vulnerability ratings. Finally, we supported our overall vulnerability rating for each species with a brief paragraph that captured the key points.

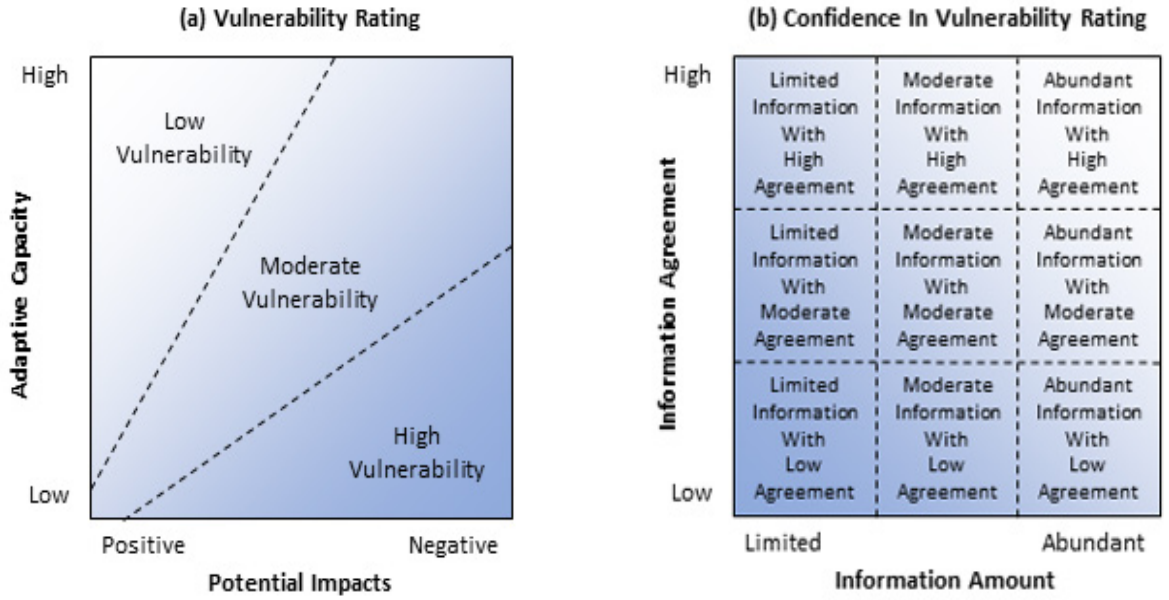


Figure 7a-b—The approach used to determine (a) the climate change vulnerability rating for a tree species and (b) the confidence in the climate change vulnerability rating. (Adapted from Brandt et al. 2014; Butler et al. 2015; Janowiak et al. 2018; and Mastrandrea et al. 2010.)

Twoneedle Pinyon (*Pinus edulis*)

Climate change vulnerability rating: moderate

Confidence in climate change vulnerability rating: limited information with moderate agreement

Our ability to confidently assess the vulnerability of Front Range twoneedle pinyon populations to climate change is constrained by a lack of local research. There is already considerable evidence of high tree mortality and low regeneration due to climate-change driven drought events, wildfires, and pinyon ips outbreaks across much of twoneedle pinyon's range, and such trends are expected to be exacerbated under a changing climate. However, it is unclear how well these range-wide trends transfer to the Front Range specifically. Bioclimatic envelope modeling suggests that twoneedle pinyon in the Front Range could potentially benefit under a warmer future climate through a net gain of suitable climate space, although capitalizing on it seems unlikely without human intervention (i.e., assisted migration) because the species is slow to regenerate naturally. Moreover, a warmer climate would likely increase the severity, frequency, and extent of drought, wildfires, and pinyon ips outbreaks, to twoneedle pinyon's detriment.

Habitat

Twoneedle pinyon is a short-statured tree that occupies dryland habitats across the southwestern United States, including Arizona, Colorado, New Mexico, and Utah, with isolated populations in southern California, eastern Oklahoma, northern Texas, and southern Wyoming (fig. 2; Little 1971; NRCS 2020; Ronco 1990). Its distribution in the Front Range is largely restricted to the southern counties; very little twoneedle pinyon occurs in the northern counties (ARP 2019; Little 1971; NRCS 2020; PSICC 2019). Indeed, it is not mapped as a major component of forests and woodlands on the AR at all (ARP 2019). On the PSI, it is a major component on 0.1 million ac (< 0.1 million ha) (PSICC 2019). It is also a major component of forests and woodlands adjacent to the PSI, such as Bureau of Land Management lands.

Twoneedle pinyon is typically found in areas with hot, dry conditions. On the PSI, long-term (1981 to 2010) mean annual precipitation in the twoneedle pinyon zone generally ranges from 11 to 27 in (29 to 68 cm) and mean annual temperature generally ranges from 40 to 48 °F (4 to 9 °C) (PRISM 2020; PSICC 2019); however, across the full distribution of the species, mean annual precipitation can be as low as 10 in (25 cm) and mean annual temperature can be as high as 61 °F (16 °C) (Ronco 1990). The elevational range of twoneedle pinyon on the PSI generally spans from 7,100 to 9,500 ft (2,100 to 2,900 m), but it can extend down to around 5,200 ft (1,600 m) elsewhere (PSICC 2019; Ronco 1990; USGS 2020). Twoneedle pinyon occurs across a variety of topographic conditions, including mountain slopes, rocky plateaus, mesas, foothill terraces, and, to a

lesser extent, intervening valleys (Ronco 1990). It also occurs on a variety of soil types, ranging from shallow soils with exposed bedrock to deep soils (Romme et al. 2009).

Twoneedle pinyon is a foundational species of pinyon–juniper woodlands, where it often co-occurs with various species of juniper (table 1; figs. 1, 2). On the PSI, it is commonly found with Rocky Mountain juniper (*Juniperus scopulorum*), but rarely with oneseed juniper (*Juniperus monosperma*) (PSICC 2019). However, oneseed juniper is a common associate on Bureau of Land Management and other surrounding lands in drier areas. Stand structure can vary across the broad set of environmental conditions that support twoneedle pinyon. A sparse, savanna-like structure with a strong grass component commonly occurs in hotter and drier areas with greater monsoonal precipitation, while a closed-canopy structure commonly occurs in cooler and wetter locations (Romme et al. 2009). Twoneedle pinyon is also occasionally found in montane forests, co-occurring with ponderosa pine and Douglas-fir (PSICC 2019); in these forests, pinyon generally achieves a taller stature.

Ecology

Reproductive Ecology

Twoneedle pinyon regeneration occurs exclusively from seed. It is a mast-seeding species, with mast events (i.e., large seed production years) typically occurring every 3 to 7 years (Redmond et al. 2012; Wion et al. 2020). Twoneedle pinyon seeds take over 2 years to mature following seed cone initiation, which occurs in late summer (Little 1938; Mirov 1967). Large mast years are associated with cool and wet climatic conditions during seed cone initiation and seed cone fertilization, which occur in summer two years prior and spring one year prior to mature seed formation, respectively (Forcella 1981; Redmond et al. 2012; Wion et al. 2020). Twoneedle pinyon seeds disperse from seed cones following maturation. Seeds are heavy and lack wings, and therefore they disperse poorly via wind (Vander Wall and Balda 1977). However, seeds are highly nutritious and commonly collected by a wide array of wildlife species, including various corvids (e.g., Clark's nutcracker [*Nucifraga columbiana*]) and rodents (e.g., pinyon mouse [*Peromyscus truei*]), which can disperse these seeds over 13 mi (22 km) (Vander Wall and Balda 1977). Following dispersal, seeds typically remain viable for about a year (Meeuwig and Bassett 1983).

Ecology of Seedlings, Saplings, and Mature Trees

Twoneedle pinyon is considered highly drought-tolerant (drought tolerance score of 4.9 out of 5.0; Niinemets and Valladares 2006). Yet, because it occurs in hot, dry climates, seedling establishment generally occurs during consecutive cool, wet years and in cool microsites beneath nurse trees and shrubs (Barger et al. 2009; Redmond et al. 2018; Urza et al. 2019). Similarly, twoneedle pinyon growth is greatest during years with high winter precipitation and low summer vapor pressure deficits (Adams and Kolb 2005; Macalady and Bugmann 2014; Redmond et al. 2017).

Twoneedle pinyon ranks as one of the most shade-intolerant tree species in the Front Range, receiving a shade tolerance score of 1.4 out of 5.0 (Niinemets and Valladares 2006; Ronco 1990). However, its shade tolerance is dependent on life stage. It strongly relies on shading from nurse plants to facilitate seedling establishment (Redmond et al. 2015, 2018; Sthultz et al. 2007). As seedlings age, the effect of nurse plants shifts from facilitative to competitive (Redmond et al. 2018).

Twoneedle pinyon experienced considerable variability in historical fire regimes across its range, with infrequent high-severity fire regimes being the most prevalent (Romme et al. 2009). Fire regimes of twoneedle pinyon have yet to be studied in the Front Range, but studies from relatively nearby areas suggest that infrequent high-severity fire regimes were common (Floyd et al. 2004, 2017; Huffman et al. 2008). Fire regimes of Mesa Verde National Park in southwest Colorado, for example, were characterized by high-severity events that burned with a rotation of 400 years or longer; fire years were coincident with periods of severe drought (Floyd et al. 2004). The tendency of twoneedle pinyon to have historically experienced a high degree of mortality due to wildfire is a product of its relatively thin bark, short stature, and low-growing branches (Stevens et al. 2020). Postfire recovery processes were often protracted historically because of climatic and microclimatic conditions and seed dispersal limitations (Floyd et al. 2017; Huffman et al. 2012; Vander Wall and Balda 1977).

Twoneedle pinyon can be affected by a variety of insects and diseases, of which the most damaging is the pinyon ips beetle (*Ips confusus*) (Eager 1999). Pinyon ips beetles are native beetles that attack damaged or stressed adult twoneedle pinyon pine trees. Extensive and severe mortality events have been caused by these insects across pinyon's range, often in combination with drought, dwarf mistletoe, or high stand densities (Breshears et al. 2005; Negrón and Wilson 2003). Pinyon ips beetles preferentially attack larger-diameter trees (Negrón and Wilson 2003). As with fire, the establishment of twoneedle pinyon seedlings following pinyon ips outbreaks can be limited by climate and microclimate (Redmond and Barger 2013).

Potential Direct Impacts of Climate Change

While twoneedle pinyon is considered highly drought-tolerant relative to other Front Range tree species, within its range, it still appears to be vulnerable to recent hotter droughts. Mature twoneedle pinyon trees have already been found to be highly sensitive to recent increases in aridity associated with climate change. Extensive twoneedle pinyon mortality has occurred in many portions of the Four Corners States in the past two decades due to a combination of hotter droughts and drought-driven pinyon ips outbreaks (Breshears et al. 2005; Floyd et al. 2009; Shaw et al. 2005). As a result, a suite of metrics has been developed to forecast future twoneedle pinyon mortality (Breshears et al. 2018), which suggest projected increases in the frequency and severity of drought events in the region will drive additional twoneedle pinyon

mortality events over upcoming decades. In fact, twoneedle pinyon radial growth is tightly linked to cool and wet climate conditions and growth declines have also already been observed in southwestern Colorado and in northern Arizona (Adams and Kolb 2004; Redmond et al. 2017).

Twoneedle pinyon recruitment also appears to be highly sensitive to recent increases in aridity. Cool and wet climate conditions during seed cone initiation and fertilization are associated with years of higher seed cone production (Forcella 1981; Parmenter et al. 2018; Redmond et al. 2012; Wion et al. 2020) and recent declines in twoneedle pinyon cone production have occurred in New Mexico and Oklahoma in association with recent warming (Redmond et al. 2012). Additionally, twoneedle pinyon seedling establishment is often limited to cool and wet climate periods (Barger et al. 2009; Shinneman and Baker 2009) and to microsites beneath nurse trees and shrubs that provide cooler microenvironments (Redmond et al. 2018; Urza et al. 2019). This apparent reliance on cool and wet conditions for seed production and seedling establishment is likely why recovery following recent tree mortality events has been limited in many areas, including in western Colorado, northern Arizona, and New Mexico (Redmond et al. 2015, 2018).

However, while this body of research shows that twoneedle pinyon has been adversely affected by ongoing climate change already, to our knowledge only one study included data from the Front Range. This study utilized a spatially extensive dataset collected by the U.S. Department of Agriculture, Forest Service's Forest Inventory and Analysis program to examine recent climate-driven twoneedle pinyon mortality (Shaw et al. 2005). It revealed that mortality in the Front Range has been fairly low in comparison to other portions of twoneedle pinyon's range.

In fact, bioclimatic modeling suggests twoneedle pinyon in the Front Range could potentially benefit from a warmer climate through a net gain of suitable climate space (Cole et al. 2008; Crookston 2009). The results of Crookston (2009), for example, indicated that 1.4 million ac (0.6 million ha) in the Front Range are currently climatically suitable for this species (box 1; table 2; fig. 8). By 2060, results predicted that 2.0 million ac (0.8 million ha) of suitable climate space will be gained, while 0.4 million ac (0.2 million ha) will be lost. By 2090, gains were predicted to be 2.2 million ac (0.9 million ha) and losses were predicted to be 0.8 million ac (0.3 million ha). Crookston (2009) utilized a GCM that is fairly analogous to the one representing the Hot future climate scenario.

Table 2—Characteristics of the wildfires > 1,000 ac (400 ha) affecting Colorado Front Range forests and woodlands, 1984 to 2020. Areas are from the Monitoring Trends in Burn Severity program (MTBS 2022). The Front Range was defined as a box bounding the Pike-San Isabel (PSI) and Arapaho-Roosevelt (AR) National Forests.

Wildfire	On PSI-AR?	Year	Area (ac)	Major forest and woodland vegetation type(s)
Canyon	Yes	1988	1,092	Montane forest
Grace Creek	Yes	1988	1,455	Montane forest; subalpine forest
#6	Yes	1989	1,547	Montane forest
Sunnyside	Yes	1989	1,416	Montane forest
Unnamed	Yes	1989	1,305	Montane forest
Buffalo Creek	Yes	1996	11,690	Montane forest
Bobcat	Yes	2000	10,444	Montane forest; subalpine forest
Eldorado	No	2000	1,001	Montane forest
High Meadows	Yes	2000	9,607	Montane forest
Big Elk	Yes	2002	4,348	Montane forest; subalpine forest
Hayman	Yes	2002	129,417	Montane forest; subalpine forest
Million	No	2002	7,424	Montane forest; subalpine forest
Mt. Zirkel Complex	No	2002	15,478	Subalpine forest
Schoonover	Yes	2002	2,821	Montane forest
Snaking	Yes	2002	2,080	Montane forest
Unnamed	Yes	2002	1,104	Montane forest
Overland	Yes	2003	3,232	Montane forest
Picnic Rock	Yes	2004	9,014	Montane forest
Mason	Yes	2005	11,023	Pinyon-juniper woodland; montane forest
Mato Vega	No	2006	13,126	Subalpine forest
Mauricio Canyon	No	2006	4,370	Montane forest
Nash Ranch	No	2008	1,107	Montane forest
Fourmile Canyon	Yes	2010	5,865	Montane forest
Medano	Yes	2010	4,887	Montane forest; subalpine forest
Crystal	Yes	2011	2,855	Montane forest; subalpine forest
Duckett	Yes	2011	4,705	Montane forest; subalpine forest
Fern Lake	No	2012	2,549	Subalpine forest
High Park	Yes	2012	90,769	Montane forest; subalpine forest

Table 2 continued.

Wildfire	On PSI-AR?	Year	Area (ac)	Major forest and woodland vegetation type(s)
Lower North Fork	Yes	2012	3,436	Montane forest
Springer	Yes	2012	1,665	Montane forest
Waldo Canyon	Yes	2012	20,112	Montane forest; subalpine forest
Wetmore	Yes	2012	2,293	Pinyon-juniper woodland
East Peak	Yes	2013	10,082	Montane forest; subalpine forest
Galena Fire	No	2013	1,319	Montane forest
Beaver Creek	No	2016	44,221	Subalpine forest
Beulah Hill	No	2016	5,769	Pinyon-juniper woodland; montane forest
Hayden Pass	Yes	2016	17,977	Pinyon-juniper woodland; montane forest; subalpine forest
Junkins	Yes	2016	19,023	Montane forest; subalpine forest
Chateau	No	2018	1,414	Montane forest
Silver Creek	No	2018	20,725	Subalpine forest
Spring Creek	Yes	2018	10,7108	Pinyon-juniper woodland; montane forest; subalpine forest
Sugarloaf	Yes	2018	1,216	Subalpine forest
Weston Pass	Yes	2018	13,169	Montane forest; subalpine forest
Decker	Yes	2019	9,875	Pinyon-juniper woodland; montane forest; subalpine forest
Calwood	Yes	2020	9,836	Montane forest
Cameron Peak	Yes	2020	208,910	Montane forest; subalpine forest
East Troublesome	Yes	2020	188,924	Montane forest; subalpine forest
Mullen	No	2020	179,894	Montane forest; subalpine forest
Williams Fork	Yes	2020	14,679	Subalpine forest

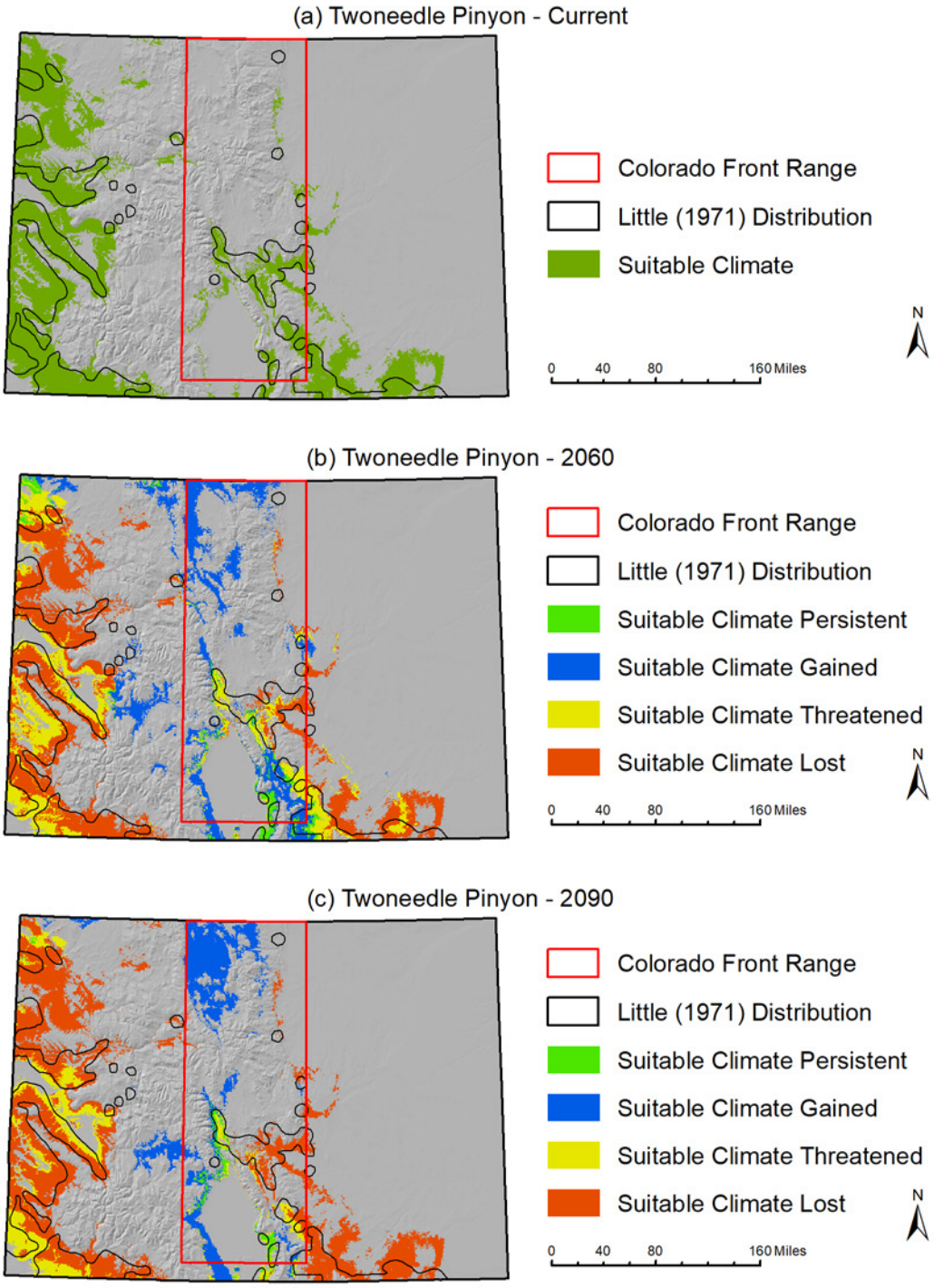


Figure 8a-c—The current and potential future suitable climate space of twoneedle pinyon (Crookston 2009). Panel (a) depicts areas in Colorado where the current climate is suitable for the species. Panels (b) and (c) depict areas in Colorado where suitable climate in the future is predicted to be either lost, threatened, persistent, or gained under the HADCM3-A2 general circulation model (GCM), a GCM that crosswalks relatively well to the one representing our Hot future climate scenario. See box 1 for more information about how current and potential future climate suitability were determined. For all panels, the map of Little (1971) is used to represent the general current distribution of the species.

Potential Indirect Impacts of Climate Change

Wildfire

Recent hotter drought events across the range of twoneedle pinyon have not only caused widespread mortality due to increased tree water stress, but have also caused mortality by driving an increase in spatially extensive stand-replacing wildfires (Floyd et al. 2004, 2017; Kennard and Moore 2013). An uptick in wildfires has also occurred in recent decades in the Front Range twoneedle pinyon zone, and while the extent and amount of mortality is unknown, it is likely that this climate change interaction is already underway here (table 3). A warmer, drier future would portend more wildfires and more pinyon mortality in the Front Range (Rangwala 2024, this report; Rocca et al. 2014).

Table 3—The area of current and potential future climate space for nine major Colorado Front Range tree species, per Crookston (2009). Current climate space is categorized as either suitable (suit) or unsuitable (unsuit), while future climate spaces is categorized as either lost, threatened (threat), persistent (pers), or gained (gain). Area values (ac, millions) are for the Front Range, defined as a box bounding the PSI-AR. See box 1 for more information about how current and potential future climate suitability were calculated.

	Current		2060				2090			
Species	Suit	Unsuit	Lost	Threat	Pers	Gain	Lost	Threat	Pers	Gain
Twoneedle pinyon	1.4	14.4	0.4	0.7	0.3	2.0	0.8	0.3	0.3	2.2
Ponderosa pine	3.0	12.8	1.3	0.8	0.9	0.9	2.3	0.6	0.1	0.6
Douglas-fir	3.0	12.7	1.7	0.9	0.5	1.1	2.5	0.5	< 0.1	0.3
Lodgepole pine	5.6	10.1	4.4	0.7	0.6	0.6	5.6	< 0.1	< 0.1	0.3
Quaking aspen	7.4	8.4	2.7	1.3	3.4	0.9	5.1	1.2	1.1	1.3
Limber pine	8.0	7.8	3.8	2.1	2.0	0.4	6.6	1.0	0.4	0.3
Rocky Mountain bristlecone pine[1]	3.1	12.6	2.7	0.2	0.2	< 0.1	NA	NA	NA	NA
Subalpine fir	4.5	11.3	1.5	0.8	2.3	0.3	3.5	0.9	0.1	0.2
Engelmann spruce	5.8	10.0	2.9	0.6	2.3	0.2	4.2	1.4	0.2	0.1

[1] 2090 data are unavailable for this species.

Climate change-driven wildfires may already be leading to contractions in twoneedle pinyon across its range due to limited postfire recovery, and further contractions are likely (Floyd et al. 2021). Twoneedle pinyon is strongly seed limited following disturbances due to its poor dispersal ability in the absence of animals (Vander Wall and Balda 1977) and, as a result, new seedling establishment following disturbances is strongly limited to areas with remaining adult live pinyon seed sources (Redmond et al. 2015). Further, twoneedle pinyon is strongly reliant on nurse trees and shrubs

for seedling establishment and survival (Redmond and Barger 2013; Redmond et al. 2015; Sthultz et al. 2007), and thus losses in tree and shrub canopy cover following disturbances further negatively impact twoneedle pinyon recovery.

Insects and Diseases

Recent hotter droughts have made twoneedle pinyon more susceptible to insect attacks and mortality by pinyon ips beetles (Gaylord et al. 2013). The widespread and severe twoneedle pinyon mortality event that occurred following the 2002 to 2003 drought coincided with a pinyon ips outbreak (Breshears et al. 2005). In a study spanning New Mexico, Colorado, and Arizona, all adult trees that died during that mortality event had evidence of pinyon ips beetle attack, highlighting how drought and insect infestations can—and will likely continue to—interact to lead to widespread dieoff events in twoneedle pinyon (McDowell et al. 2019).

Capacity to Adapt to Climate Change

While twoneedle pinyon as a species appears vulnerable to climate change, certain populations are likely better adapted to handle projected future conditions and may provide refugia as well as seed sources for assisted migration. For instance, radial growth among populations in the warmer and drier portions of the species' range appear to be more sensitive to annual fluctuations in climate (Barger and Woodhouse 2015; Redmond et al. 2017), suggesting that populations located in more mesic habitats may serve as refugia to future drought. In addition, variation in stomata shape and density can influence a tree's water use efficiency and thereby influence the effects of drought and warmer temperature (Lawson and Blatt 2014). Previous research on twoneedle pinyon found heritable differences in stomata shape between populations that were associated with differences in soil water availability (Mitton and Duran 2004; Mitton et al. 1998), suggesting that some populations, and even individuals within a population, are likely better able to persist and recover following projected hotter droughts.

Despite the projected future increase in suitable climate space in the Front Range due to climate change, in the absence of human intervention (i.e., assisted migration), twoneedle pinyon will likely be unable to take advantage of this space by rapidly shifting its range. Its ability to migrate at the pace of changing climatic suitability is low due to its low seed viability after one year, its poor seed dispersal ability, and its slow rates of recovery following disturbances (Meeuwig and Bassett 1983; Redmond et al. 2013, 2015). Indeed, Minott and Kolb (2020) found no evidence of recent upward twoneedle pinyon migration into adjacent ponderosa pine forests in northern Arizona.

Ponderosa Pine (*Pinus ponderosa* var. *scopulorum*)

Climate change vulnerability rating: moderate

Confidence in climate change vulnerability rating: moderate information with high agreement

Ponderosa pine in the Front Range is expected to be moderately vulnerable to climate change. All four future climate scenarios indicate that temperatures will increase, which will likely decrease ponderosa pine reproduction, survival, and growth across its current range, particularly in lower-elevation areas. Reproduction, survival, and growth of ponderosa pine will likely be most compromised for the Hot and Very Hot and Dry scenarios, where increases in temperature are not coupled with notable increases in precipitation. Moreover, all four scenarios are likely to interact with disturbances, including past land management, wildfire, and mountain pine beetle outbreaks, to exacerbate the adverse effects of climate change on ponderosa pine. This is particularly true for wildfire, the extent and severity of which already bear a clear mark of a changing climate. That said, several factors may help ameliorate adverse climate change effects. Ponderosa pine can modify some aspects of its morphology, phenology, and physiology in response to climatic changes. Populations within the Front Range are broadly distributed and well-connected, which may create some opportunity for latitudinal and upslope range shifts, particularly where climate change and climate change × disturbance interactions reduce the abundance of other preexisting species. Populations are also fairly genetically diverse, which may allow for evolutionary adaptation.

Habitat

The Rocky Mountain variety of ponderosa pine is a prominent tree species in forests of the interior western United States, including in the Front Range. The core of its range extends from Idaho and Montana, through Wyoming, Colorado, Utah, and Nevada, and into Arizona and New Mexico (Little 1971; NRCS 2020; Oliver and Ryker 1990). Ponderosa pine populations also grow outside this core range, in South Dakota, Nebraska, and elsewhere (Little 1971; NRCS 2020; Oliver and Ryker 1990). Ponderosa pine occurs along the entire length of the Front Range from north to south, but it is largely restricted to the eastern side of the Continental Divide from west to east (ARP 2019; Little 1971; NRCS 2020; PSICC 2019). It dominates or co-dominates 0.9 million ac (0.4 million ha) of forest managed by the PSI-AR, as well as other forested acres in the Front Range that are managed by private, county, state, and other Federal entities (ARP 2019; PSICC 2019).

The ideal ponderosa pine habitat is dry and warm relative to that of other tree species in the region. Long-term (1981 to 2010) mean annual precipitation and temperature for the PSI-AR ponderosa pine zone generally range from around 13 to 27 in (34 to 68 cm) and from around 38 to 48 °F (3 to 9 °C), respectively (ARP 2019; PRISM 2020; PSICC

2019); elevations in this zone range from around 6,700 to 9,600 ft (2,000 to 2,900 m) (ARP 2019; PSICC 2019; USGS 2020). Ponderosa pine also extends into drier, warmer, and lower-elevation locales on other lands closer to the Front Range urban corridor, such as Lory State Park just west of Fort Collins (Peet 1981). Ponderosa pine occurs on a variety of soil types in the Front Range, including those that are poorly developed with limited organic matter as well as those that are well-developed with moderate amounts of organic matter (Veblen and Donnegan 2005).

In the Front Range, ponderosa pine is one of the most common tree species in montane forests (table 1; figs. 1, 3). In xeric areas, such as at low elevations or on south-facing slopes, ponderosa pine commonly occurs either in pure stands or in stands mixed with Rocky Mountain juniper and/or Douglas-fir (ARP 2019; Peet 1981; PSICC 2019). Understory communities in these stands are typically diverse and frequently include prairie sagewort (*Artemisia frigida*), blue grama (*Bouteloua gracilis*), and alderleaf mountain mahogany (*Cercocarpus montanus*) (Fornwalt et al. 2009; Keith et al. 2010; Peet 1981). In mesic areas, such as at high elevations or on north-facing slopes, ponderosa pine commonly grows in stands with Douglas-fir, lodgepole pine, quaking aspen, blue spruce (*Picea pungens*), and (on the PSI) white fir (*Abies concolor*) (ARP 2019; Peet 1981; PSICC 2019). Common understory associates in mesic areas include kinnikinnick (*Arctostaphylos uva-ursi*), Virginia strawberry (*Fragaria virginiana*), and common juniper (*Juniperus communis*), among many others (Fornwalt et al. 2009; Keith et al. 2010; Peet 1981). Ponderosa pine tends to be a climax species in xeric areas and a seral species in mesic areas (Oliver and Ryker 1990).

Ecology

Reproductive Ecology

Ponderosa pine is a seed-obligate species. Trees in the southern Rockies can produce seed cones by as early as 28 years of age, and by 52 years of age on average (Rodman et al. 2021). Seed production takes more than 2 full years, beginning with seed cone initiation in the summer of the first year, continuing with pollination in the spring of the second year, and ending with seed ripening and dispersal in the fall of the third year (Owens and Blake 1985). Seed production is highly episodic. In the Front Range, trees produce good crops of viable seeds one to several times a decade; few viable seeds are produced in intervening years (Rodman et al. 2020b; Shepperd et al. 2006). A clear and consistent relationship between seed production and climate has not been found for ponderosa pine populations across the Front Range, although above-average moisture availability during the year of seed cone initiation appears to be important (Mooney et al. 2011). Seeds mostly disperse within two tree heights of parent trees (Barrett 1966; McDonald 1980; Oliver and Ryker 1990). Seed predators such as birds and small mammals can greatly reduce the number of viable seeds reaching the soil seedbank (Krannitz and Duralia 2004; Shepperd et al. 2006). Seeds in the soil seedbank remain viable for only a couple years (Bai et al. 2004).

Ecology of Seedlings, Saplings, and Mature Trees

Ponderosa pine is very drought tolerant. Niinemets and Valladares (2006) assigned ponderosa pine a mean drought tolerance score of 4.3 out of 5.0, a score higher than the one assigned to all of its common tree associates except Rocky Mountain juniper and limber pine. Nonetheless, ponderosa pine can be affected by drought. Germination studies show that ponderosa pine seedling emergence is negatively impacted by high temperatures and low moisture availability (Petrie et al. 2016). Dendrochronological studies likewise show that fewer seedlings emerge during drought years than during cool, wet years (League and Veblen 2006; Rother and Veblen 2017). Additionally, mature trees tend to produce less woody growth in drought years than in cool, wet years (McCullough et al. 2017). Indeed, in severe drought years, such as 1851, 1880, and 2002 in the Front Range, mature trees may not produce woody growth at all. Multiple consecutive years of drought can exacerbate growth declines and even cause mortality (Peltier and Ogle 2019).

In the Front Range, ponderosa pine historically occurred in spatially heterogeneous, yet generally open forests, and correspondingly, it tends to establish and grow best in open forest environments (Battaglia et al. 2018; Malone et al. 2018a; Schubert 1971). While ponderosa pine can benefit from some shade, particularly in the seedling stage, it does not tolerate heavy shade well. Niinemets and Valladares (2006) gave ponderosa pine a shade tolerance score of 1.6 out of 5.0; they consider Rocky Mountain juniper, limber pine, lodgepole pine, and aspen to be less shade-tolerant associates, and Douglas-fir to be more shade-tolerant.

Ponderosa pine is fairly fire-tolerant, making it well-adapted to survive the relatively frequent, low- to mixed-severity wildfires that historically regulated montane Front Range forests (Brown et al. 1999, 2015; Sherriff and Veblen 2007). That said, its fire tolerance is dependent on life stage. Even low-severity fire can result in high levels of mortality for seedlings; flame lengths of only 2 in (5 cm) were sufficient to kill most seedlings under 24 in (60 cm) tall during a prescribed fire in South Dakota, for example (Battaglia et al. 2009). As trees mature, their lower branches self-prune and they develop thick bark, an open crown architecture, thick bud scales, and deep roots, which together increase their ability to survive fire (Fornwalt et al. 2018; Stevens et al. 2020). Ponderosa pine can survive a considerable amount of crown loss due to fire, although it can rarely survive complete crown loss such as occurs from high-severity fire (Sieg et al. 2006).

Ponderosa pine is also well-adapted to regenerate after low- to mixed-severity wildfires. Such fires leave behind some surviving mature ponderosa pine trees to serve as necessary seed sources (Fornwalt et al. 2018; Malone et al. 2018a; Rodman et al. 2020b). But by killing smaller ponderosa pine trees and/or trees of less fire-tolerant associates, these fires also foster the open forest environment that promotes seedling establishment and growth (Fornwalt et al. 2018; Malone et al. 2018a; Rodman et al. 2020b).

Ponderosa pine can be affected by a variety of insects and diseases, of which the most damaging in the Front Range is the mountain pine beetle (*Dendroctonus ponderosae*). This native beetle can kill trees by mass attack, with large and/or stressed trees preferentially targeted (McCambridge et al. 1982; Negrón and Popp 2004). It can cause considerable mortality during epidemics, which can be tied to periods of drought (Bentz et al. 2010). For more information on this and other insects and diseases of ponderosa pine, please see Chapter 8 (Negrón 2024, this report).

Potential Direct Impacts of Climate Change

Bioclimatic envelope modeling done by Crookston (2009) suggests that ponderosa pine will probably experience a net loss of suitable climate space in the Front Range under a changing climate. Model results indicated that 3.0 million ac (1.2 million ha) are currently climatically suitable for ponderosa pine (box 1; table 2; fig. 9). However, under a future climate that is fairly analogous to the Hot climate scenario, model results predicted that 1.3 million of these acres (0.5 million ha) will no longer be suitable by 2060, and that suitability will be threatened for an additional 0.8 million acres (0.3 million ha). Model results also predicted that 0.9 million previously unsuitable acres (0.4 million ha) will become suitable. Areas where suitable climate was predicted to be lost generally occurred at lower elevations, while areas where suitable climate was predicted to be gained generally occurred at higher elevations.

Bioclimatic envelope modeling done by Gray and Hamann (2013) and Rehfeldt et al. (2014) also indicates that the suitable climate space of ponderosa pine will shift to higher elevations due to climate change. For example, Gray and Hamann (2013) predicted that in the U.S. Rockies, the suitable climate space for ponderosa pine in 2050 will shift upward in elevation by nearly 1,000 ft (294 m) relative to that of the 1961 to 1990 reference period. Because the authors of both studies used GCM ensembles, their future climate may best approximate our most middle-of-the-road future climate scenario, the Hot scenario.

A field-based experiment by Rother et al. (2015) suggests that the Hot future climate scenario will negatively impact the survival and height growth of ponderosa pine seedlings, at least in open areas near lower treeline. Working in Boulder County in 2012 and 2013, the authors used open-top chambers to warm the growing season air temperature around planted seedlings by about 2.7 °F (1.5 °C). They found that the warming treatment decreased both survival and height growth. For example, in 2012, median survival was around 40 percent for the warming treatment and around 90 percent for the control treatment, and in 2013, median survival was around 70 percent for the warming treatment and 100 percent for the control treatment.

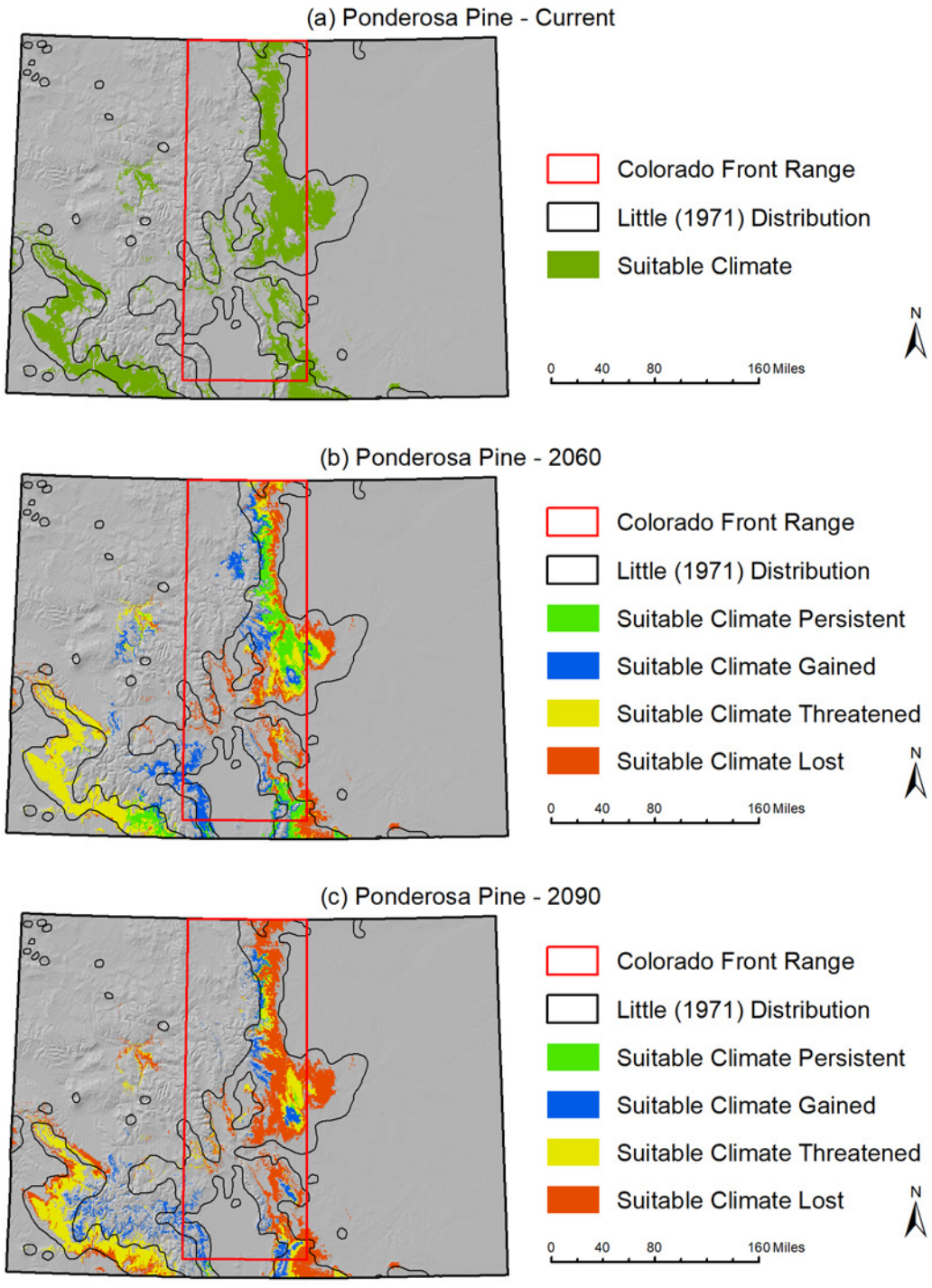


Figure 9a-c—The current and potential future suitable climate space of ponderosa pine (Crookston 2009). Panel (a) depicts areas in Colorado where the current climate is suitable for the species. Panels (b) and (c) depict areas in Colorado where suitable climate in the future is predicted to be either lost, threatened, persistent, or gained under the HADCM3-A2 general circulation model (GCM), a GCM that crosswalks relatively well to the one representing our Hot future climate scenario. See box 1 for more information about how current and potential future climate suitability were determined. For all panels, the map of Little (1971) is used to represent the general current distribution of the species.

Relative to the Hot future climate scenario, the Warm and Wet scenario will probably have less of an adverse impact on ponderosa pine owing to cooler, wetter conditions (Rangwala 2024, this report). The experiment by Rother et al. (2015) also supports this notion. When the authors coupled the warming treatment with a watering treatment, they found that the declines in survival and height growth were less severe than the declines for the warming treatment alone. Meanwhile, the Very Hot and Dry scenario will probably have a greater adverse impact on ponderosa pine than the Hot scenario. The Very Hot and Wet scenario may have an impact comparable to the Hot scenario if the effects of higher temperatures are offset by the effects of greater precipitation.

Potential Indirect Impacts of Climate Change

Past Land Management

Overly dense stands are currently widespread throughout the Front Range ponderosa pine zone due to past land management, particularly unregulated logging and livestock grazing from the 1850s to the 1900s and fire suppression from the 1900s to present (fig. 10a-b; Battaglia et al. 2018; Brown et al. 2015; Kaufmann et al. 2001). These activities allowed for substantial tree recruitment, particularly of ponderosa pine's more shade-tolerant and fire-intolerant associate, Douglas-fir (Battaglia et al. 2018; Brown et al. 2015). Lower-elevation stands now have basal areas that are 179 percent higher than they were in the mid-1800s, while higher-elevation stands now have basal areas that are 139 percent higher (Battaglia et al. 2018). Overly dense stand conditions can decrease the vigor of ponderosa pine (Bottero et al. 2017; Bradford et al. 2022); decreased vigor may in turn predispose it to higher mortality under warmer and drier conditions, such as would be encountered under the Hot and Very Hot and Dry future climate scenarios, and probably under the Very Hot and Wet scenario as well. For example, a west-wide study found that as climatic conditions became warmer and drier, the rate of ponderosa pine mortality increased more dramatically for high basal area stands than for low basal area stands (Bradford et al. 2022). Front Range stands were particularly sensitive to this phenomenon (Bradford et al. 2022).

Wildfire

Climate change will likely interact with wildfire through multiple mechanisms to reduce the abundance and distribution of ponderosa pine in the Front Range. One mechanism is by driving increases in the total area experiencing tree-killing, high-severity burning (fig. 11). Indeed, dramatic increases in total high-severity area have occurred since the 1990s, and while uncharacteristically dense forest conditions are thought to be contributing to increases, so too is a warmer, drier climate (Chapman et al. 2020; Coop et al. 2022; Parks et al. 2018; Picotte et al. 2016). For example, the Hayman Fire burned with high severity across approximately 77,000 ac (31,000 ha) of predominantly ponderosa pine forest, and occurred during the most severe drought in decades (Chapman et al. 2020; Graham 2003). Continued climate change in upcoming decades will likely further exacerbate total high-severity area, with the area likely greatest under the Very Hot and Dry scenario (Coop et al. 2022; Heidari et al. 2021; Rangwala 2024, this report; Rocca et al. 2014).

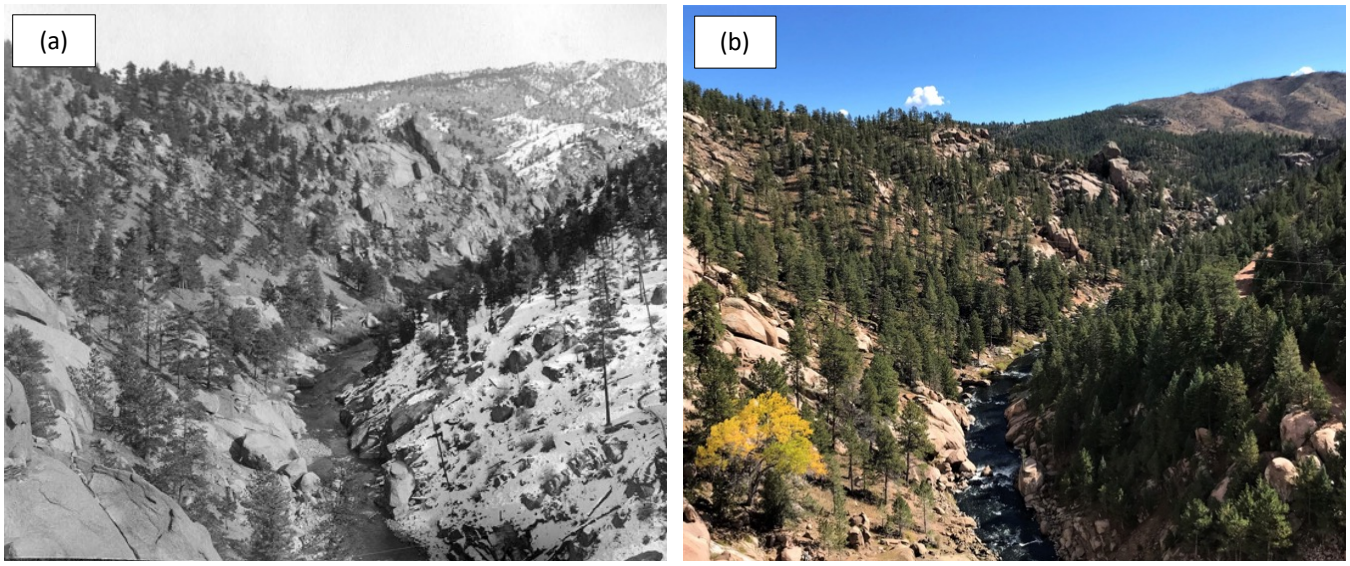


Figure 10a-b—Paired photographs demonstrating montane forest conditions along the South Platte River on the Pike National Forest in (a) 1903 and (b) 2022. In general, historical montane forests across the Front Range were considerably less dense and had proportionately more ponderosa pine and less Douglas-fir than current forests (Battaglia et al. 2018; Brown et al. 2015; Kaufmann et al. 2001). Uncharacteristically dense forest conditions can predispose trees to mortality under a changing climate through several mechanisms, including wildfire, as seen in the background of (b) (Coop et al. 2022; Parks et al. 2018; Rocca et al. 2014). Courtesy photo (a) by Denver Water; USDA Forest Service photo (b) by Paula Fornwalt.



Figure 11—A portion of a large treeless patch created by the 2002 Hayman Fire on the Pike National Forest. Driven by severe drought and dense stand conditions, this fire burned more than 129,000 ac (52,000 ha) of montane forest dominated by ponderosa pine and Douglas-fir (Chapman et al. 2020; Graham 2003). Courtesy photo by Matt Cooney, Colorado College, used with permission.

Climate-driven increases in the contiguous area of high-severity burning, or high-severity patch area, will also likely negatively impact ponderosa pine by constraining its ability to regenerate. High-severity patch area is an important metric because ponderosa pine is dependent on seeds dispersed from surviving trees to regenerate, and these seeds generally do not disperse long distances (Barrett 1966; McDonald 1980; Oliver and Ryker 1990). Chambers et al. (2016) identified 164 ft (50 m) from surviving trees as a critical threshold distance for Front Range ponderosa pine, with areas beyond it generally having very little postfire regeneration. Rother and Veblen (2016) and Rodman et al. (2020b) found similar patterns. While high-severity burning was a component of the historical fire regime in Front Range forests dominated or co-dominated by ponderosa pine, high-severity patches were not thought to be greater than 124 to 247 ac (50 to 100 ha), and so surviving trees were generally in close proximity (Fornwalt et al. 2016; Graham 2003; Sherriff et al. 2014). However, as with total high-severity area, high-severity patch area appears to have already increased in recent decades, and patches larger than 247 ac are now commonly created (Chapman et al. 2020; Coop et al. 2022). Referencing back to the Hayman Fire, for example, more than 60 percent of the total high-severity area was in patches over 247 ac in size (Chambers et al. 2016). Uncharacteristically large high-severity patches will probably continue to be created under all future climate scenarios, but especially under the Very Hot and Dry scenario (Heidari et al. 2021; Rangwala 2024, this report).

Assuming that ponderosa pine's ability to regenerate is not constrained by seed dispersal, a third mechanism through which climate change × wildfire interactions will likely adversely impact this species is by creating climatic conditions that are unfavorable for seedling establishment. Chambers et al. (2016) found that elevation was a strong driver of postfire ponderosa pine regeneration density, with higher elevations (where climate tends to be cooler and wetter) having higher densities than lower elevations (where climate tends to be warmer and drier). These results suggest that a warmer and drier future climate will likely constrain tree establishment across the current range of ponderosa pine, although upslope opportunities outside the current range may be created. Meanwhile, Rodman et al. (2020a) found that climatic water deficit, a metric that integrates climate and topography to provide an indication of water stress, was a strong predictor of postfire ponderosa pine regeneration density in recent wildfires across the southern Rocky Mountains. Moreover, through the use of an ensemble of GCMs, they found that future increases in climatic water deficit would drive declines in postfire regeneration density across much of the species' current range, and that declines would be greater under more extreme emissions scenarios than under less extreme emissions scenarios.

Insects and Diseases

Interactions between climate change and insects, particularly mountain pine beetle, will likely negatively impact ponderosa pine's abundance. As with other climate change × disturbance interactions, negative impacts are already thought to have occurred. A recent epidemic in the northern Front Range killed ponderosa pine trees across thousands of acres (Chapman et al. 2012; Meddens and Hicke 2014; West et al. 2014). The epidemic began in higher-elevation lodgepole pine forests in the late 1990s and grew rapidly due, at least in part, to a severe regional drought that stressed trees and reduced the mortality of overwintering beetles (Chapman et al. 2012; Negrón and Cain 2019). The epidemic subsequently transitioned into lower-elevation ponderosa pine forests (Chapman et al. 2012; Mitton and Ferrenberg 2012; West et al. 2014). It is likely that mountain pine beetle epidemics will continue to impact ponderosa pine under all four future climate scenarios in the near-term; however, over longer time frames, climate change may cause conditions to be outside the beetle's climatic niche (Sidder et al. 2016). Moreover, while beetle-caused canopy openings may create opportunities for seedling establishment, these opportunities may be increasingly difficult to capitalize on under the future climate scenarios (Rother et al. 2015).

Capacity to Adapt to Climate Change

The work of Carroll et al. (2017) suggests that ponderosa pine trees at mid elevations may be able to tolerate, or even benefit from, a warmer climate in situ by modifying some aspects of their morphology, phenology, and physiology. These researchers implemented an experiment in the northern Front Range, wherein ponderosa pine seedlings were grown from seeds collected at an intermediate-temperature site in the middle of the species' local elevational range and then planted at a low-temperature, an intermediate-temperature, and a high-temperature site. They found that ponderosa pine seedlings put on larger needles at the high-temperature site than at the intermediate- and low-temperature site, and that they burst their buds earlier at the high- and intermediate-temperature site than at the low-temperature site; together these traits may enable seedlings to maintain or even increase growth under warmer conditions. Additionally, these seedlings maintained a constant photosynthetic rate across the sites, suggesting that they may not need to close stomates to reduce water loss due to warming.

Genetics research suggests that ponderosa pine may have some ability to adapt evolutionarily to climate change. Relative to the isolated, disjunct populations of ponderosa pine found elsewhere in the west, the fairly well-connected Front Range populations have high genetic diversity within them (Potter et al. 2015). This high genetic diversity should enhance the capacity of ponderosa pine to adapt to a changing climate. However, Front Range populations are somewhat inbred, suggesting somewhat limited gene flow between them despite their connectedness, which may constrain

adaptive capacity (Potter et al. 2015). Moreover, the capacity of ponderosa pine to adapt may be constrained by its irregular seed production and by its relatively long generation time (Oliver and Ryker 1990; Rodman et al. 2020b, 2021; Shepperd et al. 2006).

Several aspects of ponderosa pine's current distribution could potentially enable it to migrate in response to climate change. Its range currently extends well beyond the Front Range latitudinally, not only to the south but also to the north, suggesting that any latitudinal migration the species undergoes will probably not result in range contractions in the Front Range in the foreseeable future (Oliver and Ryker 1990). Opportunities for upslope migration, into areas where it is not currently found, also exist in the Front Range. For example, while 1.3 million ac (0.5 million ha) that are currently climatically suitable for ponderosa pine are predicted to be lost by 2060 under a future climate similar to the Hot scenario, 0.9 million ac (0.4 million ha) are predicted to be gained (box 1; table 3; fig. 9; Crookston 2009). These predicted gains typically occurred in higher-elevation locales where subalpine tree species dominate, at least at the moment. Moreover, because forests currently dominated or co-dominated by ponderosa pine are fairly well-connected in the Front Range, there are few geographic barriers that would impede latitudinal or upslope migration. However, migration may not occur rapidly enough to keep pace with climate change because good seed crops are somewhat infrequently produced, seeds do not disperse long distances, and seedlings require two or more decades to grow into reproductively mature trees (Barrett 1966; McDonald 1980; Rodman et al. 2021; Shepperd et al. 2006).

Douglas-fir (*Pseudotsuga menziesii* var. *glauca*)

Climate change vulnerability rating: moderate

Confidence in climate change vulnerability rating: moderate information with high agreement

Climate change is expected to have a negative impact on Douglas-fir in the Front Range, through both direct and indirect mechanisms. Douglas-fir reproduction, survival, and growth can be adversely affected by increases in temperature and decreases in available moisture, making the species vulnerable to all four future climate scenarios, and especially vulnerable to the Hot and Very Hot and Dry scenarios. Douglas-fir's primary disturbance agents—wildfire, Douglas-fir beetle, western spruce budworm, and Douglas-fir tussock moth—can also be promoted by warmer and drier conditions, suggesting that they will drive further increases in Douglas-fir vulnerability. That said, Douglas-fir on the Front Range appears to have some opportunity for upslope migration and evolutionary adaptation, which may buffer some populations against future changes in climate.

Habitat

The Rocky Mountain variety of Douglas-fir is widely distributed throughout interior western North America. Its core range extends from British Columbia and Alberta south to Arizona and New Mexico, and from Oregon east to Wyoming, Colorado, and New Mexico (Hermann and Lavender 1990; Little 1971; NRCS 2020). Small, often discontinuous, populations also occur outside this core range, including in Washington, Montana, Texas, and northern Mexico (Hermann and Lavender 1990; Little 1971; NRCS 2020). Douglas-fir is found throughout the Front Range, although it is much more prevalent east of the Continental Divide than west of it (ARP 2019; Little 1971; NRCS 2020; PSICC 2019). It is a major constituent of 1.0 million ac (0.4 million ha) of forest on the PSI-AR alone (ARP 2019; PSICC 2019).

In the Front Range, Douglas-fir is most commonly found in montane forests, but it also occurs in some subalpine forests (table 1; figs. 1, 3). Douglas-fir rarely grows in pure stands in the Front Range (ARP 2019; PSICC 2019). It typically grows in mixed stands alongside ponderosa pine in more xeric environments, and alongside lodgepole pine, limber pine, aspen, subalpine fir, and/or Engelmann spruce in more mesic environments (ARP 2019; Peet 1981; PSICC 2019). Herbaceous and shrubby understory associates are numerous and often include Ross' sedge (*Carex rossii*), alderleaf mountain mahogany (*Cercocarpus montanus*), and mountain muhly (*Muhlenbergia montana*) in xeric areas and heartleaf arnica (*Arnica cordifolia*), common juniper (*Juniperus communis*), and kinnikinnick (*Arctostaphylos uva-ursi*) in mesic areas (Johnston 1987; Peet 1981). Douglas-fir is best characterized as a climax species in montane forests and a seral species in subalpine forests (Hermann and Lavender 1990).

As suggested by its broad geographic extent, Douglas-fir grows under a wide variety of environmental conditions (Hermann and Lavender 1990). On the PSI-AR National Forests, long-term (1981 to 2010) mean annual precipitation in the Douglas-fir zone ranges from ca. 13 to 29 in (34 to 73 cm), while long-term mean annual temperature ranges from ca. 36 to 46 °F (2 to 8 °C) (ARP 2019; PRISM 2020; PSICC 2019). Elevations in the PSI-AR Douglas-fir zone range from ca. 7,000 to 10,200 ft (2,100 to 3,100 m) (ARP 2019; PSICC 2019; USGS 2020). Closer to the Front Range urban corridor, Douglas-fir extends even lower in elevation, to around 5,500 ft (1,700 m) (Peet 1981). Douglas-fir grows on a variety of soils, but grows best in those that are deep and well-aerated (Hermann and Lavender 1990).

Ecology

Reproductive Ecology

Douglas-fir reproduces solely from seeds. Trees can begin producing seeds as early as 12 to 15 years after establishment (Hermann and Lavender 1990). Seed production is irregular, with heavy crops occurring about every 7 years (Hermann and Lavender 1990). Climatic controls on seed production are unclear in the Front Range, but at the continental scale, larger trees in warmer, drier locations appear to be the most fecund (Clark et al. 2021). The small seeds are wind-dispersed, with most falling within 330 ft (100 m) of the source tree, although a proportion of seeds may travel farther (Hermann and Lavender 1990). Seeds entering the soil seedbank probably remain viable for only about a year.

Ecology of Seedlings, Saplings, and Mature Trees

Douglas-fir is moderately drought-tolerant, with a drought tolerance score of 2.6 out of 5.0 (Niinemets and Valladares 2006). Seedlings are especially sensitive to drought, with warm, dry conditions yielding lower levels of germination and establishment than cool, wet conditions (Davis et al. 2019; Foster et al. 2020; Rother and Veblen 2017). Mature trees also grow more slowly under warm, dry conditions than under cool, wet conditions (Restaino et al. 2016).

Douglas-fir tends to grow best in environments with some shade. Seedling establishment, in particular, benefits from shade, especially on warm, dry sites like south-facing slopes (Ryker and Potter 1970). Niinemets and Valladares (2006) assigned Douglas-fir a shade tolerance score of 2.8 out of 5.0, higher than all of the other major Front Range tree species except subalpine fir and Engelmann spruce.

Prior to the 1900s, forests supporting Douglas-fir in the Front Range primarily burned with a mixed-severity wildfire regime, defined by wildfire events that burned with a mosaic of low-, moderate-, and high-severity fire effects at multi-decadal intervals (Brown et al. 1999; Schoennagel et al. 2011; Sherriff and Veblen 2007). Douglas-fir is only moderately tolerant of this regime (Giunta et al. 2016). Douglas-fir seedlings and saplings are commonly killed by even low-severity wildfires due to their low-growing branches, dense crown architecture, and thin bark (Fornwalt et al. 2018; Steinberg

2002). The bark of Douglas-fir becomes thicker with age, rendering mature trees less susceptible to wildfire-caused mortality (Steinberg 2002; Stevens et al. 2020). However, mature Douglas-fir trees are still more susceptible to mortality than those of its most common associate, ponderosa pine (Fornwalt et al. 2018; Steinberg 2002; Stevens et al. 2020). Douglas-fir can regenerate readily after wildfire, including high-severity wildfire, so long as living trees are nearby to serve as seed sources (Malone et al. 2018a; Rodman et al. 2020a,b; Rother and Veblen 2016). Regeneration after wildfire is also enhanced by relatively cool, wet conditions (Chambers et al. 2016; Rodman et al. 2020a,b).

One of the principal biotic disturbance agents of Douglas-fir in the Front Range is the western spruce budworm (*Choristoneura occidentalis*), a moth defoliator that can cause a considerable amount of mortality and decline, especially to seedlings, saplings and other subcanopy trees (Giunta et al. 2016; Hadley and Veblen 1993). The Douglas-fir tussock moth (*Orgyia pseudotsugata*), another moth defoliator, and the Douglas-fir beetle (*Dendroctonus pseudotsugae*), a bark beetle, can also cause damage to this species in the Front Range (Giunta et al. 2016). Western spruce budworm and Douglas-fir tussock moth consume new shoot growth, seeds, and cones; though they rarely cause mortality in large-diameter, canopy dominant individuals, they will often weaken them, leading to slower growth rates and predisposing them to attacks from Douglas-fir beetles (Cole et al. 2022; Hadley and Veblen 1993; Negrón et al. 2014). Douglas-fir beetles also attack trees weakened by drought, competition, fire, or other stressors (Gibson and Negrón 2009; Negrón 1998). For more information on these and other insects and diseases of Douglas-fir, please see Chapter 8 (Negrón 2024, this report).

Potential Direct Impacts of Climate Change

A field study by Restaino et al. (2016) highlights how future increases in vapor pressure deficit, a measure of water stress that integrates temperature and relative humidity, will negatively affect mature Douglas-fir growth in the northern Front Range and across much of the western United States. Historically (1916 to 2006) at the northern Front Range site, vapor pressure deficit exceeded values associated with extremely low growth rates only 10 percent of the time. However, under a changing climate, vapor pressure deficit was predicted to exceed this threshold about 60 percent of the time by the 2040s and about 90 percent of the time by the 2080s. The study used an ensemble of GCMs, making the findings most relatable to the middle-of-the-road Hot future climate scenario.

Two other field studies provide insight into the potential effects of climate change on Douglas-fir seedling survival and growth. Rother et al. (2015) used open-top chambers at a lower-elevation northern Colorado site to increase growing season air temperature around a subset of the planted seedlings by about 2.7 °F (1.5 °C); they also used weekly watering treatments to approximate an upper quartile growing season precipitation regime for a subset of the seedlings. They found that seedling survival was highest for the control (unwarmed and unwatered), intermediate for the warmed and watered seedlings, and lowest for the warmed and unwatered seedlings. They also found that

seedling growth was higher for the control and the warmed and watered seedlings than for the warmed and unwatered seedlings, though differences between the former two groups of seedlings were less apparent. Taken together, these results suggest that the higher temperatures for all four future climate scenarios will result in decreased Douglas-fir seedling survival and growth, particularly for scenarios where higher temperatures are not accompanied by increased precipitation. In a water manipulation field study, Foster et al. (2020) found that soil moisture and warmer temperatures strongly limited Douglas-fir seedling establishment at the upper elevation limit of Douglas-fir's range in northern Colorado, highlighting how increasingly warm and dry conditions may lead to recruitment failure, even at the leading edge of the species' distribution.

Bioclimatic envelope models have predicted a notable change in the suitable climate space of Douglas-fir due to future climate change. Crookston (2009) found that 3.0 million ac (1.2 million ha) were currently considered climatically suitable for Douglas-fir in the Front Range, but that under a climate change scenario akin to the Hot scenario, only 2.5 million ac (1.0 million ha) were predicted to be suitable in 2060 and 0.9 million ac (0.4 million ha) were predicted to be suitable in 2090 (box 1; table 3; fig. 12). Furthermore, 35 percent (2060) and 59 percent (2090) of these future suitable acres were threatened, meaning that their climatic suitability was predicted to be marginal. Areas where suitable climate was predicted to be lost generally occurred at lower elevations, while areas where suitable climate was predicted to be gained generally occurred at higher elevations. Gray and Hamann (2013) and Rehfeldt et al. (2014) also found similar trends with elevation, with the former predicting that by 2050, Douglas-fir populations across the U.S. Rocky Mountains will lag behind their baseline (1961 to 1990) climatic niche by approximately 587 ft (179 m).

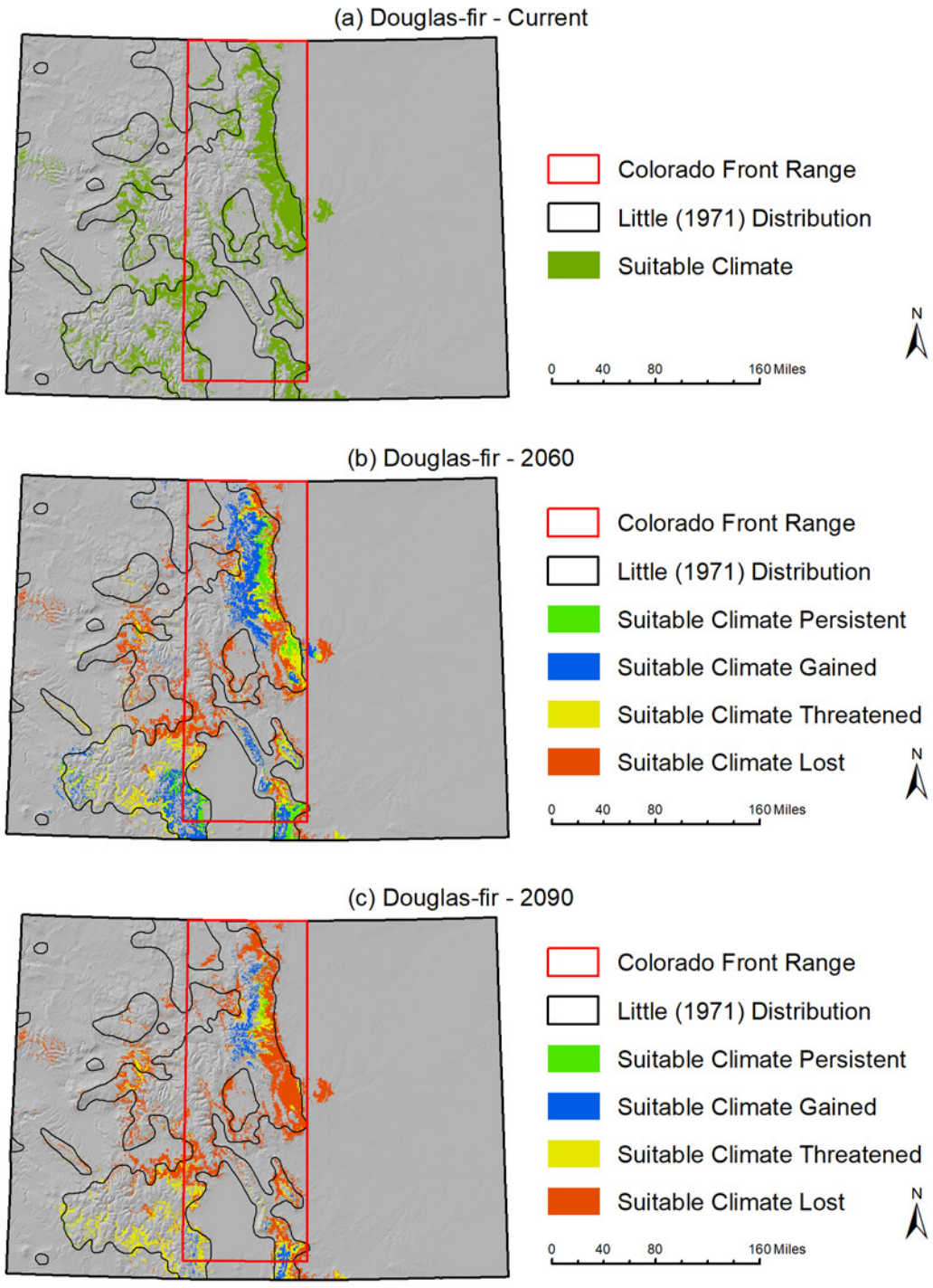


Figure 12a-c—The current and potential future suitable climate space of Douglas-fir (Crookston 2009). Panel (a) depicts areas in Colorado where the current climate is suitable for the species. Panels (b) and (c) depict areas in Colorado where suitable climate in the future is predicted to be either lost, threatened, persistent, or gained under the HADCM3-A2 general circulation model (GCM), a GCM that crosswalks relatively well to the one representing our Hot future climate scenario. See box 1 for more information about how current and potential future climate suitability were determined. For all panels, the map of Little (1971) is used to represent the general current distribution of the species.

Potential Indirect Impacts of Climate Change

Past Land Management

Lower-elevation montane forests supporting Douglas-fir have experienced dramatic increases in tree density since the mid to late 19th century due to past management (fig. 10). Fire exclusion, logging, livestock grazing, and other management activities since Euro-American settlement have disrupted the historical wildfire regime, wherein predominately mixed-severity events occurred every few decades (Brown et al. 1999, 2015; Sherriff and Veblen 2007). This disruption allowed for substantial tree recruitment, particularly of relatively shade-tolerant and fire-intolerant species like Douglas-fir (Battaglia et al. 2018; Brown et al. 2015). Uncharacteristically dense stand conditions may contribute to decreased vigor, which may in turn predispose the species to increased mortality under a warmer future climate (Restaino et al. 2016).

The effects of past management are less clear at higher elevations, such as in the upper montane and lower subalpine zones. Mixed-severity wildfire regimes were also in effect historically, but fire events tended to be less frequent (Schoennagel et al. 2011; Sherriff and Veblen 2007). While many of these higher-elevation forests, particularly those that have not burned for over a century, appear to be uncharacteristically dense and therefore vulnerable to mortality from climate change, the density increases are less dramatic compared to lower montane forests (Battaglia et al. 2018; Brown et al. 2015). For example, Battaglia et al. (2018) reconstructed a 350 percent increase in density for lower montane forests of the Front Range since Euro-American settlement, but only an 82 percent increase in density in upper montane forests.

Wildfire

Climate change, together with overly dense forest conditions, has driven large increases in the annual area burned in the Front Range montane zone (table 2; fig. 11; Chapman et al. 2020; Coop et al. 2022; Parks et al. 2018; Picotte et al. 2016). This has, in turn, driven large increases in the amount of Douglas-fir mortality. Douglas-fir was a major component of forests impacted by the 2002 Hayman Fire, for example, which burned approximately 129,000 ac (52,000 ha) during a period of extreme drought (Chapman et al. 2020; Fornwalt et al. 2018; Graham 2003). Douglas-fir was also a major component of the forests that were burned by the drought-driven 2012 High Park and 2020 Cameron Peak Fires, among others (Chapman et al. 2020; Coop et al. 2022). Continued increases in the annual area burned and resultant Douglas-fir mortality are expected under all four future climate scenarios, particularly the Very Hot and Dry scenario (Heidari et al. 2021; Rangwala 2024, this report; Rocca et al. 2014).

Climate change has also exacerbated the risk of postfire Douglas-fir regeneration failures, and this risk will likely continue to grow in the future. Main contributors to this risk are not only the amount of high-severity area, but also the contiguity of this area (Davis et al. 2020). Douglas-fir regenerates primarily from seeds produced by nearby

living trees (Hermann and Lavender 1990); however, for recent fires, a considerable proportion of high-severity area is too distant from seed-bearing trees to have a reliable seed source (Chapman et al. 2020; Rodman et al. 2020a). A trend of greater and more contiguous high-severity area is expected to continue under all four future climate change scenarios (Heidari et al 2021; Rangwala 2024, this report). Another contributor to the risk of Douglas-fir regeneration failure after wildfire is the occurrence of suitable climatic conditions. For example, Rodman et al. (2020a) forecasted future climatic suitability for postfire regeneration across the southern Rockies using an ensemble GCM approach (likely corresponding best to the Hot future climate scenario), finding strong declines in Douglas-fir climatic suitability through the mid and end of the 21st century. Similar analyses using the other three future climate scenarios would likely also show declines in climatic suitability (Rangwala 2024, this report).

Insects and Diseases

Stands with abundant Douglas-fir can sustain irruptive populations of insects such as Douglas-fir beetle, western spruce budworm, and Douglas-fir tussock moth. All four future climate scenarios will result in warmer temperatures, which will probably allow Douglas-fir beetle to expand. In the Colorado Front Range, the beetle's activity builds throughout the early growing season, peaking in mid to late June (Negrón et al. 2011). Temperatures reaching to 60 °F (15.5 °C) earlier in spring and lasting longer into the fall would extend its activity period and potentially drive spikes in its population (Bentz et al. 2010). The defoliator western spruce budworm is also expected to increase under the warmer conditions predicted by the future climate scenarios (Régnière and Nealis 2019). Moreover, warmer and drier summers, as well as dense stand conditions and wildland fires, stress Douglas-fir, increasing its susceptibility to the three insects alone and in combination (Gibson and Negrón 2009; Negrón 1998, 2014). For example, Douglas-fir beetle populations commonly increase in stressed trees during defoliation events; eventually defoliators can kill smaller-diameter and understory trees while Douglas-fir beetles can kill the larger-diameter trees (Negrón et al. 2014).

However, mountain pine beetle outbreaks that kill co-occurring lodgepole pine trees may provide Douglas-fir with opportunities for growth and expansion. Renwick et al. (2016) conducted an observational study in beetle-impacted lodgepole pine stands of Rocky Mountain National Park, including in some lower-elevation stands that also contained Douglas-fir. They found that in these stands, the decline in lodgepole pine basal area was partially offset by an increase in Douglas-fir basal area. They also found that Douglas-fir seedling densities increased, with greater increases in areas of higher mountain pine beetle mortality. Because these increases in basal area and seedling establishment occurred at Douglas-fir's leading edge, the species should be able to maintain them, and possibly even expand them, under a changing climate.

Capacity to Adapt to Climate Change

Despite extensive logging, centuries-old Douglas-fir trees can be found across the Front Range (Fornwalt et al. 2016; Huckaby et al. 2001, 2003; Kaufmann et al. 2000). These trees have persisted through a wide range of historical climate conditions, and may indicate that Douglas-fir has some ability to plastically modify aspects of its morphology, phenology, and/or physiology to tolerate climate variation.

Southern populations of Douglas-fir in the Interior West, such as those in the Front Range, have fairly high genetic diversity among them (Li and Adams 1989). This variation may allow for gene exchange and in turn promote evolutionary adaptation to climate change. That said, southern populations also have relatively limited genetic diversity within populations, which may limit adaptive capacity (Li and Adams 1989). Adaptive capacity may also be limited by Douglas-fir's reproductive ecology, including its irregular seed production, its relatively short seed dispersal distances, and its relatively long generation time (Hermann and Lavender 1990).

Douglas-fir has some potential opportunity to migrate in the Front Range with a changing climate. For example, under a future climate scenario comparable to the Hot scenario, 1.1 million ac (0.4 million ha) of climatically suitable area may be gained by 2060, most of which is above Douglas-fir's current elevational range (Crookston 2009). Because low moisture availability limits Douglas-fir establishment, the Warm and Wet scenario would likely provide Douglas-fir with even more opportunity to migrate, while the Very Hot and Dry scenario would likely provide it with less opportunity (Foster et al. 2020). However, regardless of the future climate scenario, the reproductive ecology of Douglas-fir may not allow migration to keep pace with climatic change (Hermann and Lavender 1990).

Lodgepole Pine (*Pinus contorta* var. *latifolia*)

Climate change vulnerability rating: moderate to high

Confidence in climate change vulnerability rating: moderate information with moderate agreement

Lodgepole pine will be moderately to highly vulnerable to climate change by mid-century. Under a middle-of-the-road Hot climate change scenario, the vast majority of the area that is currently climatically suitable for lodgepole pine will no longer be so, with losses concentrated at mid to lower elevations; moreover, gains in climatically suitable area at higher elevations will be small relative to losses. High-severity wildfires have long played an important role in maintaining lodgepole pine in the Front Range, particularly where serotinous trees are present. Climate change driven increases in the frequency of high-severity wildfires may favor lodgepole pine establishment over other tree species, including into upslope areas that have become climatically suitable; however, establishment potential may be diminished in areas where climatic suitability has been lowered. The 1990s to 2000s mountain pine beetle outbreak further complicates predictions of postfire lodgepole establishment, but it is possible that this prior disturbance could restrict establishment by reducing the availability of viable seeds.

Habitat

The Rocky Mountain variety of lodgepole pine is the inland form of the species. It occurs from British Columbia, Alberta, and Saskatchewan in the north to the Utah and Colorado mountains in the south (Little 1971; Lotan and Critchfield 1990; NRCS 2020). Lodgepole pine is widespread in Front Range forests, particularly in the northern counties (ARP 2019; Little 1971; NRCS 2020; PSICC 2019). It is a major component of forests on 1.1 million ac (0.5 million ha) on the PSI-AR National Forests alone (ARP 2019; PSICC 2019).

Lodgepole pine occupies a wide range of environments. In the PSI-AR lodgepole pine zone, long-term (1981 to 2010) mean annual precipitation largely ranges from 16 to 35 in (41 to 88 cm), while long-term mean annual temperature largely ranges from 32 to 42 °F (0 to 6 °C) (ARP 2019; PRISM 2020; PSICC 2019). Precipitation mostly falls as snow. Elevations supporting lodgepole pine on the PSI-AR are typically between 8,100 and 11,100 ft (2,400 to 3,400 m) (ARP 2019; PSICC 2019; USGS 2020). Lodgepole pine grows on a wide variety of soil types, including those that are poorly developed, shallow, and infertile (Lotan and Critchfield 1990).

Lodgepole pine in the Front Range is a keystone member of the subalpine forest vegetation type (table 1; figs. 1, 4). It commonly grows in pure stands (ARP 2019; Peet 1981; PSICC 2019). It also grows alongside ponderosa pine and Douglas-fir in warmer, lower-elevation areas of the upper montane zone, and alongside quaking aspen,

subalpine fir, and Engelmann spruce in cooler, higher-elevation areas of the subalpine zone (ARP 2019; Peet 1981; PSICC 2019). Understory communities can be relatively depauperate in stands containing lodgepole pine, with the shrubs kinnikinnick (*Arctostaphylos uva-ursi*), common juniper (*Juniperus communis*), and blueberry (*Vaccinium* spp.) being some of the most common understory community members (Peet 1981). Lodgepole pine is described as a shade-intolerant seral species that establishes right after stand-replacing disturbances, particularly wildfire; however, it can be a climax or near-climax species where such disturbances are relatively frequent or where other, more shade-tolerant species do not co-occur (Alexander 1974; Huckaby and Moir 1998; Peet 1981).

Ecology

Reproductive Ecology

Lodgepole pine reproduction is dependent on seeds. Trees can begin producing annual seed crops when they are as young as 5 to 10 years of age in open stands, and 15 to 20 years of age in closed stands (Alexander 1974). Large seed crops are generally produced at 1- to 3-year intervals thereafter (Lotan and Critchfield 1990). Individual trees have predominantly either serotinous cones that remain closed after seeds mature and open following exposure to intense sunlight or fire, or nonserotinous cones that open at seed maturity; however, serotiny is not expressed until trees are 30 to 60 years old (Alexander 1974). The proportion of trees with serotinous cones is highly variable across stands in the Rocky Mountains, with greater levels of serotiny potentially favored where lodgepole pine's native stand-replacing fire regime is more frequent (Alexander 1974; Decker and Fink 2015). Serotinous cones are persistent, storing viable seeds in tree crowns for many decades (Alexander 1974). Seeds from both serotinous and nonserotinous cones are primarily dispersed by wind, but some seeds may be dispersed by rodents or birds. Seeds are usually dispersed less than 200 ft (60 m), although strong winds may carry seeds much farther (Alexander 1974). Once seeds are dispersed and enter the soil seedbank, they likely remain viable for only a few years.

Ecology of Seedlings, Saplings and Mature Trees

Lodgepole pine is highly drought-tolerant; with a drought tolerance score of 4.2 out of 5.0, it needs less water than its major tree associates Douglas-fir, quaking aspen, Engelmann spruce and subalpine fir (Niinemets and Valladares 2006). Even so, it can be adversely affected by drought. Young seedlings are shallow-rooted and can be particularly drought-sensitive (Kaufmann and Eckard 1977; Petrie et al. 2016; Shepperd and Noble 1976). Mature trees can also be sensitive (Andrus et al. 2021).

Lodgepole pine is a shade-intolerant species (shade tolerance score of 1.5 out of 5.0; Niinemets and Valladares 2006). Optimal germination and seedling establishment occur in full sunlight, although shading can be beneficial on warm, dry sites (Alexander 1974). Once established, trees grow quickly, with the greatest growth in full sunlight (Alexander 1974).

As previously alluded to, high-severity, stand-replacing wildfires are the major disturbance agent in the Front Range lodgepole pine zone. Such wildfire events, which historically occurred roughly every 150 to 350 years, are often associated with infrequent, extreme regional droughts (Kulakowski and Jarvis 2011; Schoennagel et al. 2011; Sibold and Veblen 2006; Sibold et al. 2006). Stand-replacing wildfires generally spread rapidly through the forest canopy and result in the complete or near-complete mortality of lodgepole pine across large areas (Schoennagel et al. 2004). Although less severe wildfires that burn on the forest floor can also sometimes occur in the lodgepole pine zone, particularly at lower elevations, these fires can also result in significant mortality because lodgepole pine's bark is relatively thin (Schoennagel et al. 2011; Sibold et al. 2006; Stevens et al. 2020). A dense pulse of lodgepole pine trees often establishes and grows rapidly following wildfires, particularly where prefire stands exhibited a high degree of serotiny (Guz et al. 2021; Schoennagel et al. 2011; Sibold et al. 2006).

Lodgepole pine is highly susceptible to infestations by insects and diseases. The most common and damaging insect and disease agent of lodgepole pine is the mountain pine beetle, a native species. Epidemic outbreaks of these beetles, which are commonly driven by drought and warmer-than-average winters, have long caused extensive mortality events in the Colorado Front Range and across the southern Rocky Mountains (Chapman et al. 2012; Jarvis and Kulakowski 2015; Negrón and Huckaby 2020). Trees that are large, old, or stressed are more likely to be killed (Klutsch et al. 2009; Kulakowski and Jarvis 2013; Negrón and Huckaby 2020). While mountain pine beetle outbreaks can trigger a pulse of regeneration, they tend not to trigger as dense of a pulse as fire or harvesting do because they do not dramatically increase light availability in the understory, expose mineral soil, or open most serotinous cones (Pelz et al. 2018; Rhoades et al. 2020a). For more information about insects and diseases affecting lodgepole pine, please see Negrón 2024, this report.

Potential Direct Impacts of Climate Change

Climate change may directly lead to increased mortality of mature lodgepole pine trees in the Front Range (Andrus et al. 2021; Smith et al. 2015). Indeed, recent (1982 to 2019) increases in annualized mortality rates of lodgepole pine have already been documented at Niwot Ridge in the northern Front Range, albeit nonsignificantly (Andrus et al. 2021). Higher mortality rates were correlated with higher frequencies of extremely warm (90th percentile) summer days, higher summer climatic water deficits, and cooler phases (i.e., La Niña events) of the El Niño–Southern Oscillation (Andrus et al. 2021). Looking forward, it is likely that all four future climate change scenarios would spur additional increases in mortality rates, but also that rates would be lowest under the Warm and Wet scenario and highest under the Very Hot and Dry scenario.

Bioclimatic envelope models provide insight into how shifts in suitable climate space may affect lodgepole pine's distribution. Bioclimatic envelope models developed by Crookston (2009) suggest that lodgepole pine will likely experience a considerable loss of suitable climate space by mid-century in the Front Range (box 1; table 3; fig. 13). Results indicated that 5.6 million ac (2.3 million ha) are currently climatically suitable for lodgepole pine. However, under a future climate that is fairly analogous to the Hot climate scenario, model results predicted that around 79 percent of these acres will no longer be suitable in 2060, and that suitability will be threatened for an additional 18 percent. Model results also predicted that 0.6 million previously unsuitable acres (0.2 million ha) will become suitable. Additionally, bioclimatic envelope models developed by Gray and Hamann (2013) estimated that today's lodgepole pine populations across the U.S. Rocky Mountains already lag behind their baseline (1961 to 1990) suitable climate space by approximately 105 mi (168 km) in latitude and 245 feet (74 m) in elevation. The models further predicted that, relative to the baseline suitable climate space, the suitable climate space for lodgepole pine in 2050 will shift upward by 335 mi (542 km) in latitude or 780 ft (238 m) in elevation.

A warming and watering experiment conducted by Conlisk et al. (2018) at Niwot Ridge showed that temperature increases negatively impacted first-year lodgepole pine seedling establishment, but that this impact was partially to fully alleviated by moisture increases that compensated for warming-induced evaporative losses. Interestingly, this pattern held for all sites (forest, treeline, and alpine), as well as for all seed provenances (low-elevation and high-elevation). This pattern also held through time, although seedlings became less sensitive to the effects of warmer and/or wetter conditions as they approached 3 years of age.

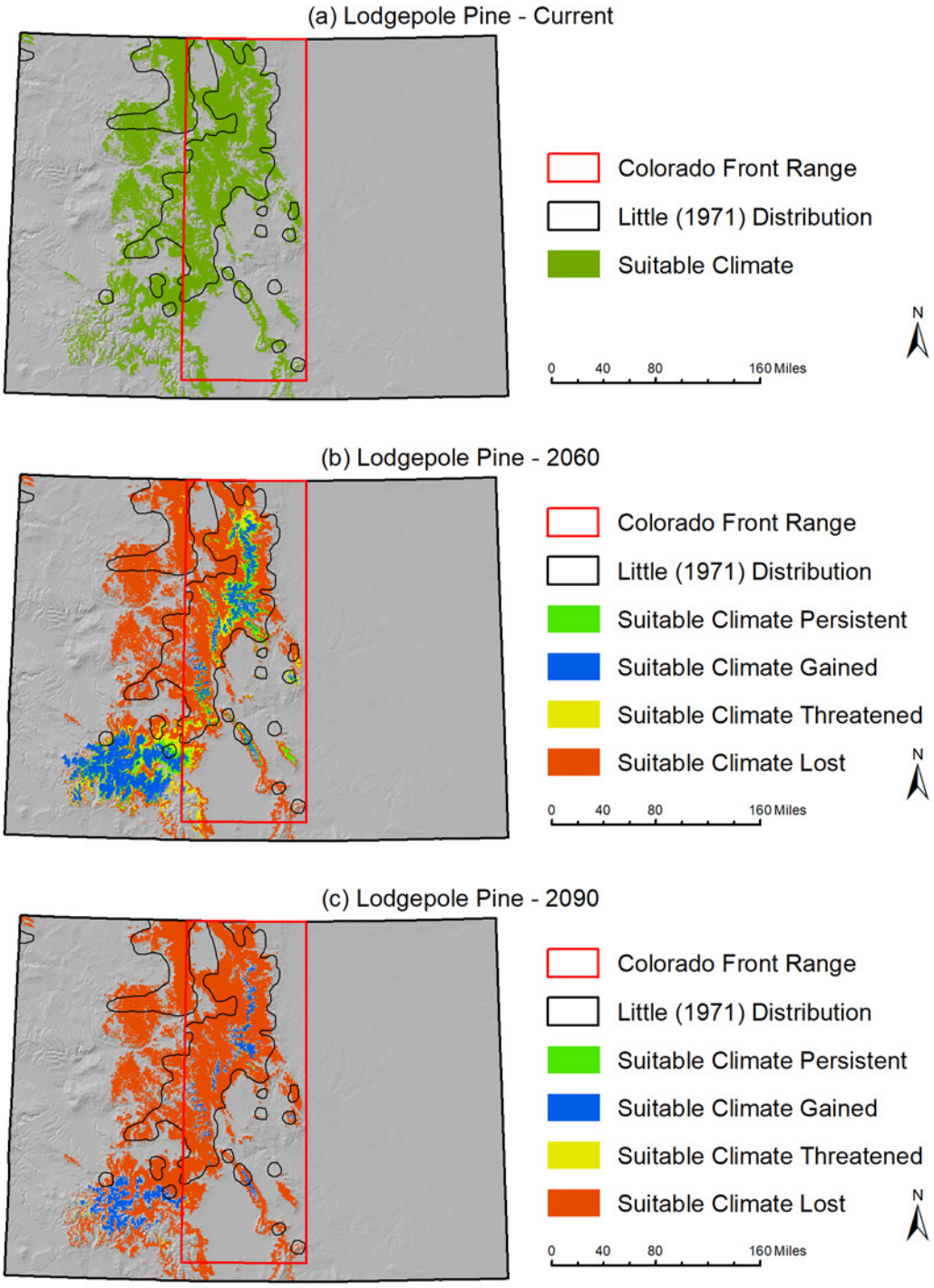


Figure 13a-c—The current and potential future suitable climate space of lodgepole pine (Crookston 2009). Panel (a) depicts areas in Colorado where the current climate is suitable for the species. Panels (b) and (c) depict areas in Colorado where suitable climate in the future is predicted to be either lost, threatened, persistent, or gained under the HADCM3-A2 general circulation model (GCM), a GCM that crosswalks relatively well to the one representing our Hot future climate scenario. See box 1 for more information about how current and potential future climate suitability were determined. For all panels, the map of Little (1971) is used to represent the general current distribution of the species.

Potential Indirect Impacts of Climate Change
Insects and Diseases

The mountain pine beetle outbreak that unfolded across Colorado from the 1990s to 2000s affected millions of acres of forest in the lodgepole pine zone, killing hundreds of millions of trees (fig. 14; Chapman et al. 2012; Meddens and Hicke 2014). Climate change played a key role in triggering and sustaining the outbreak: ongoing increases in temperature and decreases in precipitation bolstered beetle survival and reproduction, and also stressed trees and reduced their capacity to defend against attacks (Bentz et al. 2010; Chapman et al. 2012; Creeden et al. 2014; Mitton and Ferrenberg 2012; Sidder et al. 2016). An abundance of suitable high-density stands with large-diameter trees also contributed to the outbreak (Negrón and Cain 2019). Assuming suitable stands and trees remain available, additional warming and drying are expected to exacerbate future outbreaks, at least at higher elevations; however, at lower elevations, additional warming and drying may result in conditions that are outside the climatic niche of the beetle (Hicke et al. 2006; Sidder et al. 2016).



Figure 14—The recent mountain pine beetle outbreak resulted in a considerable amount of lodgepole pine mortality across the northern Front Range and elsewhere in Colorado. Ongoing climate change played a key role in triggering and sustaining the outbreak (Bentz et al. 2010; Chapman et al. 2012; Creeden et al. 2014; Mitton and Ferrenberg 2012; Sidder et al. 2016). USDA Forest Service photo by Paula Fornwalt.

Lodgepole pine regeneration events triggered by mountain pine beetle outbreaks may be compromised under a changing climate, particularly at the lower-elevation margins of lodgepole's range. In fact, such reductions may already be occurring. Renwick et al. (2016) found that at three low-elevation lodgepole pine sites in Rocky Mountain

National Park that experienced recent beetle disturbance, lodgepole seedling densities were very low, ranging from only 17 to 36 stems ac^{-1} (42 to 90 stems ha^{-1}). Moreover, lodgepole pine seedlings were completely absent at the lowest-elevation portions of the sites. Meanwhile, at high-elevation lodgepole pine sites that experienced beetle disturbance, seedling densities were considerably higher, ranging from 55 to 115 stems ac^{-1} (136 to 284 stems ha^{-1}). The authors suggested that the differences in regeneration rates between low- and high-elevation sites provide insight into how future mountain pine beetle outbreaks and climatic changes could interact to drive trailing-edge contractions in lodgepole pine's range.

Wildfire

In their review on climate change × fire regime interactions in the southern Rocky Mountains, Rocca et al. (2014) concluded that subalpine forests will continue to burn with high severity as they did historically, but will burn with greater frequency. Increases in fire frequency already seem to be unfolding: largely because of the record-setting, drought-driven fire season of 2020, the 21st century fire rotation interval for subalpine forests in and adjacent to the Front Range is now 117 years, a rate that is nearly double that of the past 2,000 years (Higuera et al. 2021). Because lodgepole pine is poorly adapted to survive even low-severity burning, future increases in fire frequency can be expected to drive considerable increases in mortality (Stevens et al. 2020). But, if prefire stands contain live trees with serotinous cones, future fires could encourage lodgepole pine to regenerate, so long as climate is favorable (Guz et al. 2021; Schoennagel et al. 2011; Sibold et al. 2006). Unfortunately, ongoing climate change has already reduced the probability of postfire seedling establishment in the southern Rockies and elsewhere in the West, and future changes in climate over the next 10 to 30 years are expected to lead to further reductions (Davis et al. 2023).

Stands that were impacted by mountain pine beetle outbreaks prior to burning could potentially experience additional obstacles to regeneration under a warmer, drier climate. This is primarily because the viability of seeds stored in the serotinous cones of beetle-killed trees diminishes as time since outbreak passes. Research conducted shortly after the onset of the recent beetle outbreak in Rocky Mountain National Park reported that on average, the germination rate of seeds stored in serotinous cones on recently beetle-killed trees exceeded 50 percent (Aoki et al. 2011). However, a subsequent study conducted on the AR and the adjacent Medicine Bow-Routt National Forests found that the germination rate of seeds in serotinous cones on long-dead trees averaged only 34 percent (Rhoades et al. 2022). The regeneration challenges created by declining seed viability could be exacerbated by the low fuel moistures of dead trees and by dead tree fall rates, increasing the likelihood of cone consumption during fire (Rhoades et al. 2018, 2020b). This said, Davis et al. (2023) did not find an effect of prior bark beetle outbreaks on postfire lodgepole pine regeneration, highlighting the uncertainty surrounding the interactive effects of climate change, mountain pine beetle outbreaks, and wildfire.

Capacity to Adapt to Climate Change

Lodgepole pine's phenological plasticity may help it adapt to climate change in situ, and possibly even benefit from it. Carroll et al. (2017) conducted a common garden experiment in the northern Front Range, where lodgepole pine seedlings were grown from seeds collected at an intermediate-temperature site in the middle of the species' local elevational range and then planted at a low-temperature, an intermediate-temperature, and a high-temperature site. They found that seedlings burst their buds earlier at the high- and intermediate-temperature site than at the low-temperature site, extending the growing season; longer growing seasons under a changing climate could result in greater growth. Moreover, lodgepole may not need to close stomates to reduce water loss due to warming, as seedlings maintained a constant photosynthetic rate across the sites; this may also enable greater growth.

Lodgepole pine appears to have some potential to adapt evolutionarily to a changing climate. West-wide studies indicate that populations are highly adapted to their local climate, resulting in considerable genetic diversity across the species' range (Rehfeldt et al. 1999; Yang and Yeh 1993, 1995). Genetic diversity within populations is also high due to high gene flow rates (Yang and Yeh 1993, 1995). The adaptive capacity provided by these diversity patterns is likely augmented by lodgepole's short (relative to other Front Range conifers) generation time and its ability to produce an abundance of seedlings.

Lodgepole pine's distribution in the Front Range may both hurt and help its ability to migrate as climate changes. It reaches the southern extent of its range in the southern Front Range, thus limiting latitudinal migration opportunities. The topography in the Front Range suggests that ample opportunities for upslope migration exist, although bioclimatic envelope modeling indicates that this topographic variation does not fully translate into suitable climate space under a future climate akin to that of the Hot scenario (Crookston 2009).

Quaking Aspen (*Populus tremuloides*)

Climate change vulnerability rating: moderate

Confidence in climate change vulnerability rating: moderate information with moderate agreement

Quaking aspen has a low tolerance of warm, dry conditions. Such conditions reduce survival and growth, as well as reproduction from both sprouts and seeds. Bioclimatic envelope models indicate that, under a warmer climate similar to the Hot future climate scenario, quaking aspen in the Front Range may experience 24 to 42 percent net loss of suitable climate space by the middle of the century. Consistent with drought-driven dieback events already observed across Colorado, the suitable climate space that was predicted to be either lost or threatened tended to occur at lower elevations. That said, quaking aspen has the potential to benefit from climate-change driven increases in disturbances like wildfire and mountain pine beetle and spruce beetle outbreaks. By killing aspen and/or co-occurring conifers, wildfires and outbreaks may create opportunities for this disturbance-stimulated species to regenerate not only within climatically suitable portions of its current range via sprouting, but also above its current range via seedling establishment as climate there becomes suitable.

Habitat

Quaking aspen is the most broadly distributed native tree species in North America. In the northern portion of its range, it extends west to east from Alaska and British Columbia to Newfoundland and Labrador and Nova Scotia (Little 1971; NRCS 2020; Perala 1990). It then extends down through mountainous regions in the west to Arizona and New Mexico, while in the east it extends to Iowa, Illinois, and over to New Jersey (Little 1971; NRCS 2020; Perala 1990). Very small, disjunct populations also occur farther south in both the west and the east (Little 1971; NRCS 2020; Perala 1990). In the Front Range, quaking aspen is a major component of about 1.1 million forested acres (0.4 million ha) on the PSI-AR alone, and likely occurs on tens of thousands of acres on adjacent lands (ARP 2019; Little 1971; NRCS 2020; PSICC 2019).

Given its broad distribution, it follows that quaking aspen tolerates a wide variety of environmental conditions. On the PSI-AR, areas where quaking aspen is a major forest component predominately have long-term (1981 to 2010) mean annual temperature values between 32 and 44 °F (0 and 7 °C) and long-term mean annual precipitation values between 14 and 33 in (36 and 83 cm) (ARP 2019; PRISM 2020; PSICC 2019). Elevations in these areas mostly fall between 7,700 and 11,000 ft (2,300 and 3,400 m) (ARP 2019; PSICC 2019; USGS 2020). Aspen grows in a wide range of soil types in the Front Range, so long as they are moist (Peet 1981); where soils are on the drier end of aspen's soil moisture spectrum, they are typically well-developed and fertile (Peet 1981).

In the Front Range, quaking aspen is an important member of subalpine forests, and to a lesser extent, montane forests (table 1; figs. 1, 4). Quaking aspen can sometimes grow in pure stands in the Front Range, but it most commonly grows in mixed-species stands (ARP 2019; PSICC 2019). Its most common associates in mixed stands are ponderosa pine, Douglas-fir, lodgepole pine, subalpine fir, and Engelmann spruce (ARP 2019; PSICC 2019). It is also commonly associated with limber pine and Rocky Mountain bristlecone pine in mixed stands (ARP 2019; PSICC 2019). Stands containing aspen tend to support understories that are species-rich and productive relative to adjacent, purely coniferous, stands, although composition is highly variable (Peet 1981).

Ecology

Reproductive Ecology

Quaking aspen most commonly reproduces vegetatively, by suckering along a shallow, extensive lateral root system (Perala 1990). Suckering in quaking aspen is suppressed by aboveground stems, and released when stems are killed or removed by disturbances such as wildfire or logging (Perala 1990). That said, aspen can also reproduce from seed; indeed, recent genetics work suggests that, while rare, seed reproduction is more common than previously thought (Landhäusser et al. 2019; Long and Mock 2012; Mock et al. 2008). Aspen begins to flower at 2 to 3 years of age and begins to produce large seed crops at 10 to 20 years of age (Perala 1990; Schopmeyer 1974). The seeds are attached to a pappus of long, silky hairs that facilitates long-distance dispersal by wind (Perala 1990). Dispersed seeds tend to have high viability, although this viability lasts for only a few weeks at best (Maini and Cayford 1968).

Ecology of Seedlings, Saplings, and Mature Trees

Quaking aspen can have a low tolerance of warm, dry conditions. Its drought tolerance score is 1.8 out of 5.0, the lowest of all the major Front Range tree species (Niinemets and Valladares 2006). Mature stems are shallowly rooted, and warm, dry conditions can diminish their vigor or even cause mortality (Perala 1990). For example, in USDA Forest Service Region 2, a mid-2000s rapid dieback event (sometimes referred to as SAD, or sudden aspen decline) was triggered by the extreme temperature and precipitation conditions that occurred in the years prior to the event (Rehfeldt et al. 2009). Warm, dry conditions can also negatively impact regeneration, especially from seeds, which require water-saturated soils to germinate (Perala 1990).

As a disturbance-stimulated species, aspen does not tolerate shade well. It has a shade tolerance score of 1.2 out of 5.0, the lowest of all tree species in the Front Range (Niinemets and Valladares 2006).

Wildfires play a key role in quaking aspen's persistence in the Front Range. While aspen stands can be stable in the absence of wildfire or other disturbance, many experience decline due to mortality of the relatively short-lived stems coupled with low regeneration rates and invasion by longer-lived and more shade-tolerant conifers (Bretfield et al. 2016; Kashian et al. 2007; Zier and Baker 2006). When wildfire burns

through an aspen stand, aspen stems are typically killed; the stems' very thin bark makes them highly susceptible to mortality by even low-intensity wildfires (Rodman et al. 2020b; Shinneman et al. 2013; Wooten et al. 2022). Yet this mortality often triggers rapid, prolific suckering from the root system (Rodman et al. 2020b; Shinneman et al. 2013; Wooten et al. 2022). Because wildfires tend to reduce competition and expose a mineral soil seedbed, aspen establishment from seed can sometimes also be promoted by them (Kreider and Yocom 2021; Nigro et al. 2022; Shinneman et al. 2013).

Quaking aspen stems are highly susceptible to mortality from a diversity of insects and diseases. Among the most damaging insects are various species of wood borers, bark beetles, and defoliators (Dudley et al. 2015; Perala 1990). A host of pathogenic fungi cause cankers and butt, root, and trunk rots (Dudley et al. 2015; Hinds and Wengert 1977; Perala 1990). For more information on insects and diseases of aspen, please see Negrón 2024, this report). Aspen also is an important forage for wildlife and livestock, which may result in mortality through heavy browsing, trampling, and rubbing (Dudley et al. 2015; Perala 1990).

Potential Direct Impacts of Climate Change

The bioclimatic envelope models of Crookston (2009) revealed that quaking aspen could experience a net loss of 24 to 42 percent of suitable climate space in the Front Range under the Hot future climate scenario by mid-century (box 1; table 3; fig. 15). Model results indicated that 7.4 million ac (3.0 million ha) are currently climatically suitable for quaking aspen. However, model results predicted that 2.7 million of these acres (1.1 million ha) will no longer be suitable in 2060, and that suitability will be threatened for an additional 1.3 million of these acres (0.5 million ha). While model results also predicted that 0.9 million previously unsuitable acres (0.2 million ha) will become suitable, these gains are insufficient to balance losses. Suitable climate space was generally predicted to be either lost or threatened at lower elevations, while suitable climate was generally predicted to be gained at higher elevations.

Similarly, bioclimatic envelope models developed by Gray and Hamann (2013) for the U.S. Rocky Mountains estimated that the current (1997 to 2006) suitable climate space for quaking aspen already exceeds the baseline (1961 to 1990) suitable climate space by approximately 130 ft (40 m) in elevation. Moreover, relative to the baseline suitable climate space, the suitable climate space for quaking aspen in 2050 will potentially shift upward by 540 ft (164 m) in elevation. Bioclimatic envelope models developed by Rehfeldt et al. (2009) for U.S. Forest Service Region 2 predicted even greater elevational shifts in suitable climate space for quaking aspen by 2090 (as high as 3,280 ft (1,000 m)).

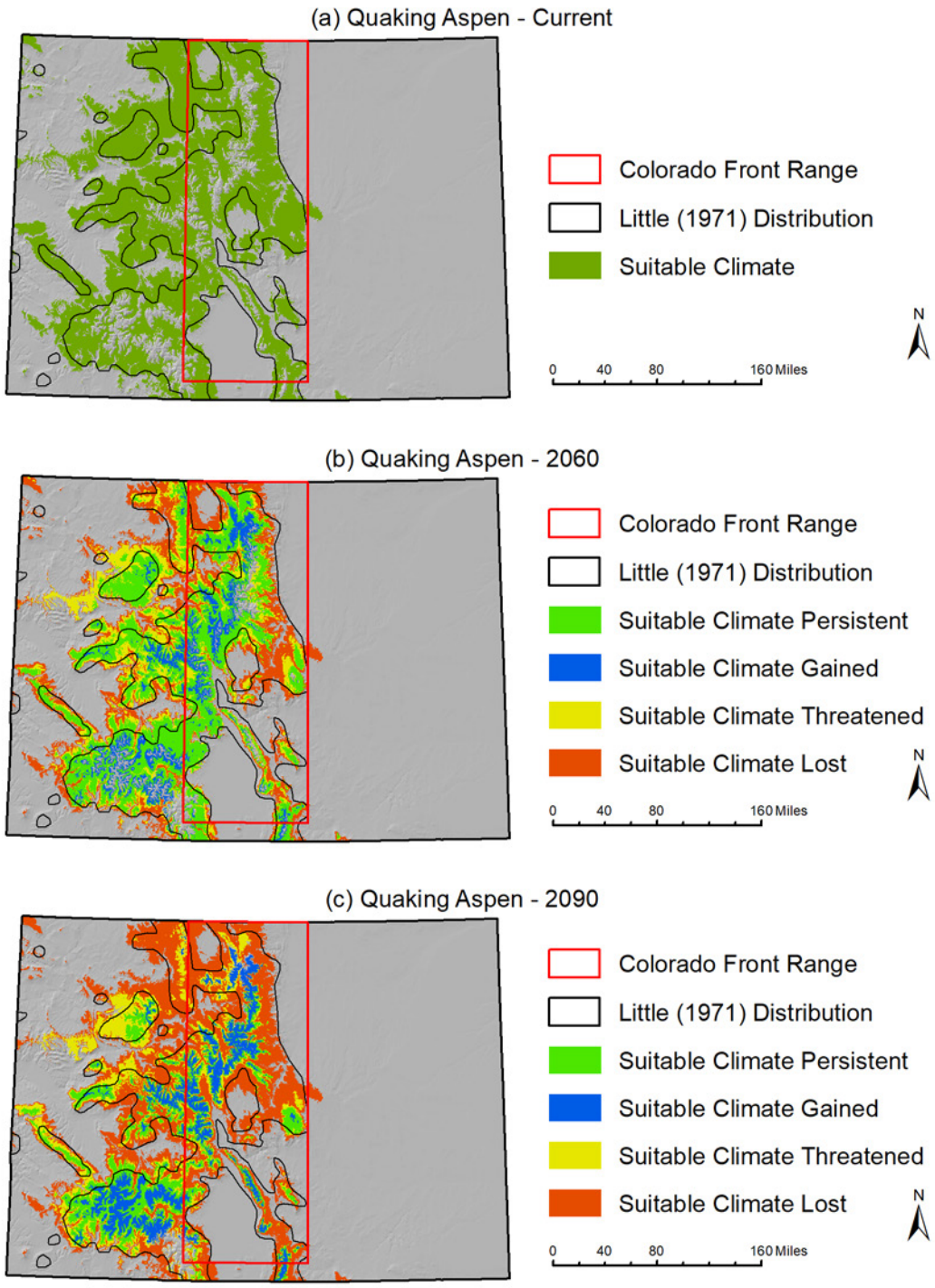


Figure 15a-c—The current and potential future suitable climate space of quaking aspen (Crookston 2009). Panel (a) depicts areas in Colorado where the current climate is suitable for the species. Panels (b) and (c) depict areas in Colorado where suitable climate in the future is predicted to be either lost, threatened, persistent, or gained under the HADCM3-A2 general circulation model (GCM), a GCM that crosswalks relatively well to the one representing our Hot future climate scenario. See box 1 for more information about how current and potential future climate suitability were determined. For all panels, the map of Little (1971) is used to represent the general current distribution of the species.

Rehfeldt et al. (2009) also used a bioclimatic envelope modeling approach to examine the climatic factors that triggered a rapid quaking aspen dieback event in U.S. Forest Service Region 2 during the mid-2000s, and to predict where dieback events may be likely to occur in the future. Analyses suggested that the dieback event was triggered by the extremely warm, dry conditions that occurred from 2000 to 2003, and that dieback was largely concentrated within lower-elevation portions of aspen's current suitable climate space. Analyses using an ensemble of GCMs further indicated that aspen's suitable climate space could potentially shift upslope so as to exclude 58 percent of the dieback area by 2030 and 76 percent of the dieback area by 2060. These findings highlight the extent to which dieback events could play a major role in adjusting aspen's lower-elevation distribution under a changing climate.

Potential Indirect Impacts of Climate Change

Wildfire

Continued climate change in the upcoming decades is expected to further exacerbate wildfire activity by increasing the number of high, very high, and extreme fire danger days (Rangwala 2024, this report). While greater wildfire activity will drive increases in the mortality of aspen stems, it could also promote ample regeneration from suckers, so long as severity is not high enough to damage aspen's shallow root system (Rodman et al. 2020b; Shinneman et al. 2013; Wooten et al. 2022). Greater wildfire activity could also promote aspen regeneration from seed by reducing competition and exposing a mineral soil seedbed (Kreider and Yocom 2021; Shinneman et al. 2013). Indeed, Nigro et al. (2022) documented aspen seedling establishment following recent wildfires on the Rio Grande National Forest, Colorado, including establishment 239 to 1,296 ft (73 to 395 m) above prefire aspen stands, suggesting that postfire upslope establishment could help aspen successfully shift its range in response to climate change. However, postfire regeneration from both suckers and seeds would likely be compromised by future warming and drying, particularly at lower elevations (Perala 1990).

Insects and Diseases

Future deviations from normal temperature and precipitation regimes may lead to conditions that stress quaking aspen, increasing its susceptibility to damage from insects and diseases. The recent aspen dieback events that occurred across Colorado and the Interior West serve as a case in point. While extremely warm, dry conditions triggered the events, insects and diseases then commonly infiltrated the stressed trees to serve as the proximate agents of mortality. Working in five National Forests of central Colorado and Wyoming (including the PSI), Dudley et al. (2015) found that aspen stands experiencing dieback were not only on warmer, drier sites than healthy stands, but also had higher incidence of damage from wood borers, bark beetles, and Cytospora (*Cytospora* spp.) canker. These findings largely track those documented elsewhere in Colorado (Marchetti et al. 2011; Worrall et al. 2008).

In some areas, aspen has been a beneficiary of recent bark beetle epidemics that killed co-occurring conifers. Rodman et al. (2022a) synthesized field plot data from 16 studies conducted across subalpine forests that were recently impacted by bark beetles in the southern Rocky Mountains. They found that aspen dominated only 2 percent of the sampled plots prior to the outbreaks. However, aspen dominated 6 percent of the plots after the outbreaks, and simulations using the Forest Vegetation Simulator indicated that aspen would dominate 10 percent of the plots after 30 years. Aspen may be able to maintain dominance under changing climate in some of these expanded areas, particularly those that are in cooler, wetter environments.

Capacity to Adapt to Climate Change

An experiment implemented in the northern Front Range by Carroll et al. (2017, 2019) suggests that quaking aspen may have only a limited ability to withstand climate change by plastically modifying its morphology, phenology, and physiology. The authors grew aspen seedlings from seeds collected at an intermediate-temperature site in the middle of the species' local elevational range and then planted the seedlings at a low-temperature, an intermediate-temperature, and a high-temperature site. They found that aspen displayed a strong degree of temperature sensitivity in photosynthesis, leaf size, and growth efficiency, with traits achieving their greatest values at the intermediate elevation site closest to the seed source location. Together, these findings suggest that warmer temperatures will constrain aspen growth. While they also found that warmer temperatures lengthened the growing season by triggering earlier bud break, they predicted that any resultant growth benefits will be insufficient to overcome the observed other constraints on growth.

The capacity of quaking aspen to evolve or migrate in response to climate change will be highly dependent on seed-based reproduction. While random mutations within clones contribute to aspen's high level of genetic diversity, given the pace of climatic changes expected, reproduction from seeds will be needed to support the rapid evolution of favorable traits (Mitton and Grant 1996; Mock et al. 2008). Similarly, while clonal expansion enables some migration, reproduction from seeds will be crucial if aspen is to move along with its suitable climate space (Elliott and Baker 2004; Mock et al. 2008; Nigro et al. 2022). However, aspen's ability to reproduce from seeds is constrained. Aspen produces large quantities of seeds that readily disperse long distances, but seeds rarely grow into seedlings due to very short seed viability windows and highly exacting moisture requirements for seed germination and seedling establishment (Landhäusser et al. 2019; Long and Mock 2012; Maini and Cayford 1968; Perala 1990; Schopmeyer 1974).

Limber Pine (*Pinus flexilis*)

Climate change vulnerability rating: high

Confidence in climate change vulnerability rating: moderate information with moderate agreement

If climate change was the only factor challenging limber pine, we may hypothesize that the species could fare better than other less drought-tolerant species. It is a generalist and has broad climatic tolerances. However, limber pine is highly susceptible to disturbances, especially white pine blister rust but also mountain pine beetle and wildfire. These disturbances greatly exacerbate the vulnerability of limber pine to climate change. A key limitation to projecting limber pine's population trajectories under a changing climate is the paucity of data on the frequency of quantitative resistance to blister rust.

Habitat

Limber pine, a five-needle white pine, has a very broad distribution in North America[1]. Its core range extends from the Canadian Rockies (British Columbia, Alberta) south through the central (Montana, Idaho, Wyoming) and southern Rockies (Colorado, New Mexico) and west into the Great Basin (Utah, Nevada) and southern Sierra Nevada (California). It occurs in small populations across much of this range; the largest, most connected populations occur in Montana, Wyoming, Utah, and Colorado (Little 1971; NRCS 2020; Steele 1990). Limber pine occurs throughout the Front Range (ARP 2019; Little 1971; NRCS 2020; PSICC 2019). It is a dominant or co-dominant tree species on approximately 0.3 million ac (0.1 million ha) of forest on the PSI-AR (ARP 2019; PSICC 2019). Limber pine also occurs on other forested lands in the Front Range, such as in Rocky Mountain National Park and Great Sand Dunes National Park and Preserve.

Limber pine tends to be most abundant on xeric sites, growing in harsh habitats where other tree species cannot (Schoettle 2004; Tomback et al. 2011). However, it can establish on more mesic sites if competition is low. On the PSI-AR, long-term (1981 to 2010) mean annual precipitation for sites where limber pine is dominant or co-dominant ranges from about 15 to 34 in (39 to 86 cm) (ARP 2019; PRISM 2020; PSICC 2019). Long-term mean annual temperature for these same sites ranges from about 32 to 44 °F (0 to 7 °C). Limber pine has a very broad elevational range across its distribution and especially in the Front Range, where it extends from the grassland to

[1]In the 1970s, geneticists reported hybridization between limber pine and southwestern white pine (*Pinus strobiformis*) (Steinhoff and Andresen 1971). Recently, the broad hybrid zone has been confirmed using genomic tools to span the U.S. portion of the southwestern white pine's range and into southern Colorado (Menon et al. 2018). For this discussion, we refer to the full distribution of limber pine as mapped by Little (1971) as being limber pine, although ongoing research may lead to changes in the taxonomic designation of some populations.

the alpine ecotones. On the PSI-AR, it is predominately found from about 8,100 to 11,100 ft (2,400 to 3,400 m) (ARP 2019; PRISM 2020; PSICC 2019; USGS 2020); it also occurs on the Pawnee National Grassland at about 5,200 ft (1,600 m) (Schoettle and Rochelle 2000). Limber pine grows on well-drained soils derived from a wide range of parent materials (Steele 1990).

Limber pine is most prevalent in subalpine forests of the Front Range, but it can be common in montane forests as well (table 1; figs. 1, 4). Limber pine rarely grows in pure stands (ARP 2019; PSICC 2019). It is most commonly found growing alongside ponderosa pine, Douglas-fir, lodgepole pine, quaking aspen, Rocky Mountain bristlecone pine, and Engelmann spruce (ARP 2019; PSICC 2019). Plant communities growing underneath limber pine tend to be sparse and dominated by shrubs, such as kinnikinnick (*Arctostaphylos uva-ursi*), common juniper (*Juniperus communis*), and currant (*Ribes* spp.) (Peet 1981). Limber pine is a long-lived seral species but can be a climax species on the most xeric of sites (Schoettle 2004; Steele 1990).

Ecology

Reproductive Ecology

Limber pine is a slow-growing species with delayed reproductive maturation (Schoettle 2004; Tomback et al. 2011). Forest-grown trees can produce their first seed-bearing cones at around 30 years of age but do not produce full cone crops until they are approximately 60 to 80 years old, or older on harsh sites (Schoettle, personal observation). Limber pine is a masting species with good cone crops occurring every 4 to 5 years, although some trees may produce some cones at other times. Along the Front Range, cones mature in the first half of September (Schoettle et al. 2019b). Hot, dry summers can accelerate cone maturation by up to a few weeks. Seeds lost to cone and seed insects can be extensive for limber pine in Colorado, especially for those seeds that mature following a mast year (Schoettle and Negrón 2001; Williams et al. 2020). We hypothesize that the elevated risk of insect damage to seeds following a masting event is due to an increase in predator populations.

Throughout the core range of limber pine, including the Front Range, Clark's nutcracker (*Nucifraga columbiana*) is the primary seed disperser, influencing population structure (Carsey and Tomback 1994; Schuster et al. 1989). Limber pine seeds are large and commonly wingless. The long-distance dispersal by nutcrackers facilitates the mixing of diverse seed sources for the colonization of sites following disturbance, resulting in low levels of genetic differentiation (i.e., structure) among populations across the landscape (Bower et al. 2011; Tomback et al. 2011). Nutcrackers are rare visitors to the Dave's Draw limber pine population on the Pawnee National Grassland and it is hypothesized that seed-caching rodents are the primary seed dispersers at grassland sites (Tomback et al. 2005). Once dispersed, seeds are readily consumed by not only their bird and rodent dispersers but by numerous other species as well (Tomback and Achuff 2010).

Ecology of Seedlings, Saplings, and Mature Trees

Limber pine has many traits that confer drought tolerance (drought tolerance score of 4.7 out of 5.0; Niinemets and Valladares 2006). Limber pine has a strong stomatal response to vapor pressure deficit, which limits water loss under highly evaporative conditions and thereby conserves hydraulic function (Pataki et al. 2000; Reinhardt et al. 2011, 2015). The species also tends to have a high root-to-shoot ratio, a characteristic of drought-tolerant species (Borgman et al. 2014, 2015). In some habitats, it can access groundwater via a deep taproot, thereby decoupling its water supply from local precipitation patterns (Roberts et al. 2004).

Limber pine is considered shade-intolerant, receiving a shade tolerance score of 1.6 out of 5.0 (Niinemets and Valladares 2006; Steele 1990). However, sparse overstory canopy cover and nurse objects can improve seedling establishment and performance (Casper et al. 2016; Coop and Schoettle 2009); with climate warming, these conditions may be increasingly important (Schoettle et al. 2022a).

Fire has played a historical role in the metapopulation dynamics of limber pine in and surrounding the Front Range (Brown and Schoettle 2008) and elsewhere (Webster and Johnson 2000). Because of limber pine's broad elevational distribution in the Front Range, it likely experienced the full spectrum of fire regimes prior to the 1900s (Brown and Schoettle 2008; Veblen 1986). However, limber pine has bark that is thin, branches that do not readily self-prune, and a stature that is generally short, making it poorly adapted to survive even low-severity fire (Stevens et al. 2020). Although limber pine is an early seral species, postfire seedling establishment is typically slow. Individuals accumulate over decades, generating an uneven age stand structure (Brown and Schoettle 2008; Cleaver et al. 2016; Goeking and Windmuller-Campione 2021; Windmuller-Campione and Long 2016). Postfire seedling densities can vary across the landscape, with successful establishment in some areas of the Front Range and only sparse establishment in others (Coop and Schoettle 2009, 2011); this suggests that numerous factors, including climatic, seedbed, overstory, and understory conditions, all influence seedling establishment (Kueppers et al. 2017; Schoettle et al. 2022a).

Limber pine is highly susceptible to mortality from biotic agents such as white pine blister rust and mountain pine beetle. White pine blister rust is caused by the nonnative fungal pathogen *Cronartium ribicola* (fig. 16). The pathogen can infect all five-needle pines in North America, but to complete its life cycle it must use currant (*Ribes* spp.) or select other species as alternate hosts (Geils et al. 2010). Infections in mature trees build over a period of several years, initially causing reduced vigor and branch death, and if the infection reaches the tree bole, tree death (Burns et al. 2008). Infections in seedlings tend to progress more rapidly (Schoettle and Sniezko 2007). Mountain pine beetle is a native insect that attacks numerous North American pine species. Limber pine has very low constitutive chemical defenses against mountain pine beetle, making it highly susceptible to attack and mortality (Bentz et al. 2016; Gray et al. 2015). Mountain pine beetle kills limber pine trees by mass attack, with large trees preferentially affected.



Figure 16a-b—Fruiting canker of *Cronartium ribicola* on (a) a limber pine branch and (b) a bole of a Rocky Mountain bristlecone pine (*Pinus aristata*), is the nonnative fungal pathogen that causes the lethal disease white pine blister rust. The high level of mortality by white pine blister rust will reduce population size and therefore may reduce the adaptive capacity of the species to climate change (Schoettle et al 2022a.) USDA Forest Service photos by Anna Schoettle.

The sensitivity of limber pine to mountain pine beetle and other stressors is increased by limber pine dwarf mistletoe (*Arceuthobium cyanocarpum*), making it another forest health concern. Trees infested with dwarf mistletoe use more water and experience greater water stress and mortality than uninfected trees (Glatzel and Geils 2009; Kolb et al. 2016; Robinson and Geils 2006; Zweifel et al. 2012). Until the recent beetle epidemic, dwarf mistletoe was the second most damaging agent of limber pine behind blister rust in the central and southern Rocky Mountains (Cleaver et al. 2015; Kearns and Jacobi 2007; Taylor and Mathiasen 2002). Chapter 8 (Negrón 2024, this report) presents more information on these and other insects and diseases of limber pine.

Potential Direct Impacts of Climate Change

Evidence and characterization of the impacts of climatic warming on limber pine suggest that mortality of mature trees is associated with long-term cumulative, rather than short-term acute, climatic stress (Kane and Kolb 2014; Millar et al. 2015). Sustained climatic stress is already increasing along the Front Range and contributing to increased limber pine mortality (Andrus et al. 2021; Smith et al. 2015). For example, at

Niwot Ridge in the northern Front Range, limber pine mean annualized mortality rates increased over a 31-year period of observation, coincident with increases in warmer and drier conditions (Smith et al. 2015). However, climate-associated mortality has been less for limber pine than some other co-occurring species (Andrus et al. 2021; Kane et al. 2014; Smith et al. 2015) and could portend an increased role of limber pine in a drier future (Schoettle et al. 2022a; Windmuller-Campione and Long 2016).

The bioclimatic envelope models of Crookston (2009) suggest a future overall loss in climatically suitable area for limber pine in the Front Range. Under a climate change scenario similar to the Hot scenario, climatically suitable area was predicted to decrease from 8.0 million ac (3.2 million ha) currently to 4.5 million ac (1.8 million ha) by 2060 and 1.7 million ac (0.7 million ha) by 2090 (box 1; table 3; fig. 17). Crookston's (2009) models also suggest a future shift in climatically suitable area to higher elevations.

While bioclimatic envelope models developed by Monahan et al. (2013) for limber pine in Rocky Mountain National Park did not show a clear trend in the amount of climatically suitable area through 2100, they did show a clear future upslope shift. These authors developed their models using an ensemble of 31 GCMs and two distinct training datasets: one collected locally in the Park, and one collected across limber pine's range.

A more suitable climate at higher elevations may promote the elevational advancement of limber pine seedling establishment and performance above the current alpine treeline (Moyes et al. 2013, 2015). Although restricted, limber pine regeneration above the current alpine treeline appears to be greater than for other species due to long-distance seed dispersal by Clark's nutcracker (e.g., Millar et al. 2015; Smithers et al. 2018), suggesting that its role in treeline dynamics may increase in the absence of insect, disease, and wildfire pressures (Sindewald et al. 2020).

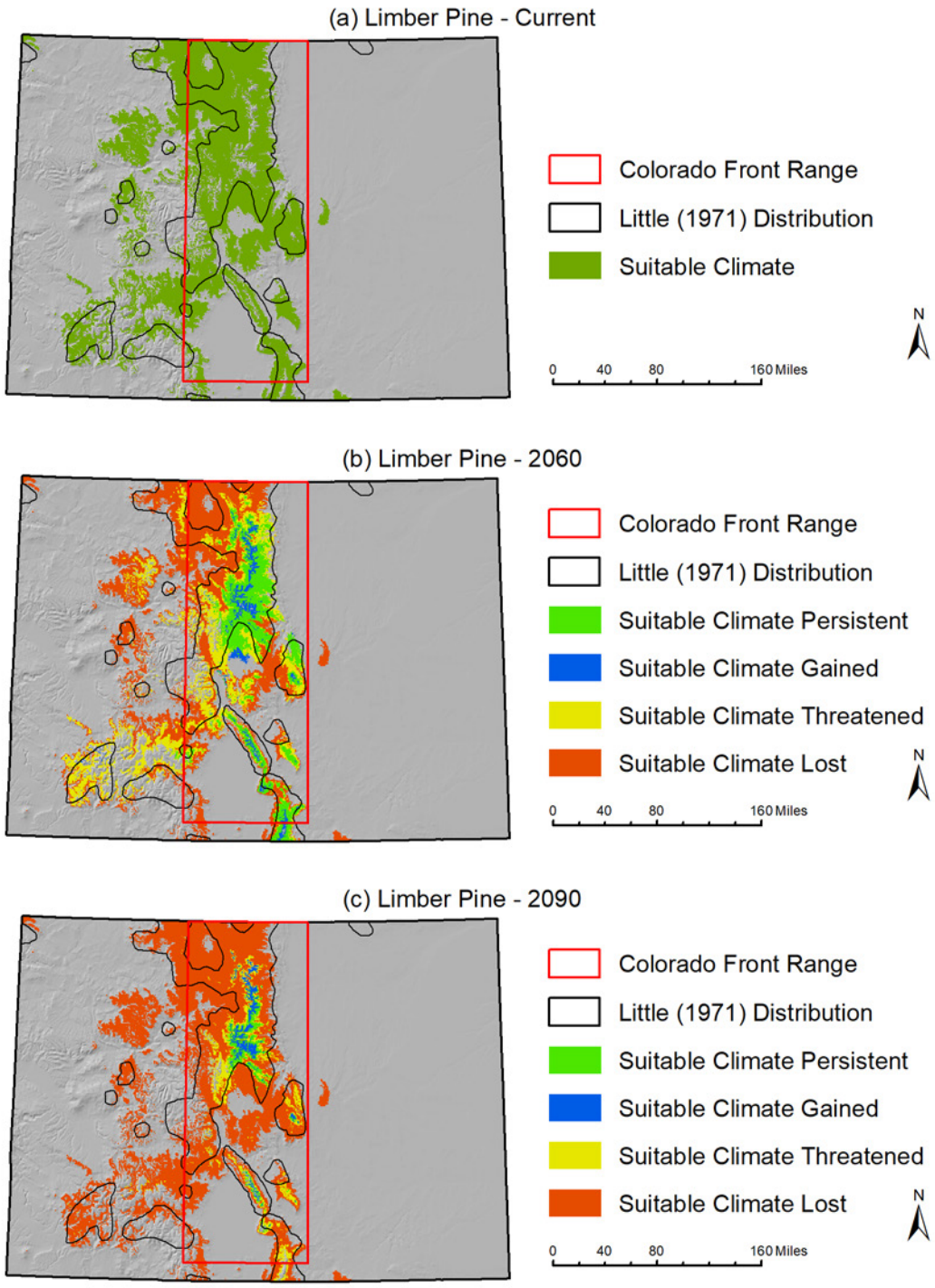


Figure 17a-c—The current and potential future suitable climate space of limber pine (Crookston 2009). Panel (a) depicts areas in Colorado where the current climate is suitable for the species. Panels (b) and (c) depict areas in Colorado where suitable climate in the future is predicted to be either lost, threatened, persistent, or gained under the HADCM3-A2 general circulation model (GCM), a GCM that crosswalks relatively well to the one representing our Hot future climate scenario. See box 1 for more information about how current and potential future climate suitability were determined. For all panels, the map of Little (1971) is used to represent the general current distribution of the species.

Potential Indirect Impacts of Climate Change

Wildfire

Climate-driven increases in the size and severity of wildfires have been documented for many regions in the western United States and are predicted to continue (Abatzoglou and Williams 2016; Gergel et al. 2017; Higuera et al. 2021). In the central and southern Rocky Mountains, these current and future trends hold for both the montane and the subalpine forests where limber pine resides (Gergel et al. 2017; Higuera et al. 2021; Picotte et al. 2016; Wright and Roy 2022; Yue et al. 2013). Because limber pine is considered an early seral species, some fire is likely to benefit the species; however, increases in fire severity and frequency may tip the balance toward risking population extirpation from too much fire (Coop and Schoettle 2011; Keane et al. 2021).

Insects and Diseases

Repeated forest health monitoring assessments provide valuable condition trends for limber pine in the Rockies, revealing high mortality from white pine blister rust (Burns et al. 2023; Cleaver et al. 2015). In northern Colorado, Wyoming, and southeastern Montana, 73 percent of the stands have been invaded by blister rust with an average disease incidence of 26 percent, an increase of 6 percent over 8 to 9 years (Cleaver et al. 2015). Blister rust was first detected in northern Colorado in 1998 (Johnson and Jacobi 2000) and continues to spread southward (Schoettle et al. 2022a). A disjunct infection center in the Sangre de Cristo Mountains was found in 2003 (Blodgett and Sullivan 2004) and the incidence of blister rust on limber pine is currently over 50 percent in the area and has spread to the Wet Mountains (Burns 2006; Schoettle et al. 2022a). Disease severity continues to increase on limber pine (Burns et al. 2023). Risk analysis modeling using climate variables projected that 41 to 53 percent of limber pine habitat is at risk for blister rust in Colorado (Kearns et al. 2014). By 2030, limber pine is expected to experience a 40 percent reduction in basal area in the United States (Krist et al. 2014). In Alberta, Canada where blister rust has been impacting limber pine for more than half a century, limber pine is listed as endangered (Alberta 2014), and it is being considered for national listing as such under the Species at Risk Act.

The recent epidemic of mountain pine beetle has taken its toll on limber pine. This is likely the result of limber pine being a good host tree that fosters brood development (Cerezke 1995). High beetle-caused mortality has recently been observed on 75 percent of limber pine plots distributed across southern Montana, Wyoming, and northern Colorado, and on 18 percent of trees (Cleaver et al 2015). As of 2016–2017, mountain pine beetle continued to cause mortality of limber pine in northern Colorado though it has caused only minimal mortality in the Sangre de Cristo Mountains of southern Colorado (Burns et al. 2023). Climate warming is likely to expand the threat of mountain pine beetle to the higher-elevation five-needle pines, including limber pine (Fettig et al. 2022; Howe et al. 2021; Logan et al. 2010).

Capacity to Adapt to Climate Change

Limber pine is facing multiple compounding factors that threaten the sustainability of populations and their evolutionary potential. The strong selection pressure of white pine blister rust alone is formidable, and adaptation to the disease is imperative to sustain or restore limber pine populations as it continues to spread. How white pine blister rust epidemiology will be affected by climate change is likely to be context-dependent and is therefore difficult to predict (Hennon et al. 2020; Schoettle et al. 2022a). This challenge is further complicated by pressures from other mortality factors that are each expected to further increase in a warming climate, most notably drought, mortality from mountain pine beetle, and wildfires of increasing frequency, size, and intensity (Schoettle et al. 2022a).

Because limber pine has not co-evolved with the nonnative fungal pathogen that causes white pine blister rust, it is highly susceptible to it. Fortunately, limber pine has some naturally occurring genetic resistance to the pathogen, providing the opportunity for adaptation. However, although limber pine has both heritable major gene resistance and quantitative resistance to white pine blister rust, the frequency of the more durable quantitative resistance appears to be very low (Schoettle et al. 2014, 2022b). The frequency of quantitative resistance traits may be too low to sustain viable limber pine populations when under high natural selection pressures without active management. A key limitation to projecting limber pine's population trajectories is the paucity of data on the frequency of quantitative resistance to white pine blister rust (Schoettle et al. 2022a). The high level of mortality by the rust will reduce population size and therefore may reduce the adaptive capacity of the species to climate change and other stressors (Schoettle et al. 2022a).

Rocky Mountain Bristlecone Pine (*Pinus aristata*)

Climate change vulnerability rating: moderate to high

Confidence in climate change vulnerability rating: moderate information with moderate agreement

Drought is already a feature of Rocky Mountain bristlecone pine's environment, and the forecasted changes in climate may further intensify drought as a primary stressor. The relationships between cone production, seedling establishment, and climate suggest particular vulnerabilities to climate change. Rocky Mountain bristlecone pine currently shows only minor impacts from insects and diseases. However, the nonnative disease white pine blister rust is just beginning to spread into Rocky Mountain bristlecone pine's distribution; the species is susceptible to the disease and extensive mortality can be expected as the disease expands in area. The combination of low genetic diversity, moderate population isolation, a protracted and vulnerable regeneration dynamic, and added mortality by white pine blister rust will limit the adaptive capacity of this species to climate change.

Habitat

Rocky Mountain bristlecone pine (hereafter RM bristlecone pine) is a long-lived five-needle pine with a very narrow distribution. Its disjunct populations grow in just three states: Colorado, Arizona, and New Mexico (NRCS 2020). Its core range is located in Colorado, and it achieves its greatest distribution in the Front Range, where it extends from Boulder County south to Las Animas County (ARP 2019; NRCS 2020; PSICC 2019). RM bristlecone pine is a major component of about 0.3 million ac (0.1 million ha) of forest on the PSI-AR (ARP 2019; PSICC 2019); these acres likely capture the bulk of the RM bristlecone pine in the Front Range. Great Basin bristlecone pine (*Pinus longaeva*), which grows in Utah, Nevada, and California, was distinguished from RM bristlecone pine in 1970 and the distributions of the two bristlecone species do not overlap.

RM bristlecone pine grows in cold, dry, and high-elevation environments. In PSI-AR Forests with a major component of RM bristlecone pine, long-term (1981 to 2010) mean annual temperature ranges from around 31 to 39 °F (-1 to 4 °C), while long-term mean annual precipitation ranges from around 15 to 38 in (39 to 96 cm) (ARP 2019; PRISM 2020; PSICC 2019). Precipitation peaks in late summer with the North American Monsoon (Schoettle et al. 2022a). Elevations range from around 9,400 to 11,800 ft (2,800 to 3,600 m) (ARP 2019; PSICC 2019; USGS 2020). Slopes on which the species is found tend to face southwest.

RM bristlecone pine in the Front Range is almost exclusively restricted to the subalpine forest vegetation type (table 1; figs. 1, 4). Its most common tree associate is Engelmann spruce, but it also regularly grows with lodgepole pine, limber pine, aspen, and subalpine fir (ARP 2019; PSICC 2019). It can sometimes form pure stands, particularly

where harsh conditions put the site outside of the physiological tolerance of other tree species (ARP 2019; PSICC 2019). Ground cover is commonly dominated by rock, litter, and bare soil, with understory associates such as fescue (*Festuca* spp.), alpine clover (*Trifolium dasyphyllum*), and common juniper (*Juniperus communis*) generally of much lesser abundance (Johnston 1987; Schoettle and Coop 2017).

Ecology

Reproductive Ecology

RM bristlecone pine reproduces from seeds, which are relatively small and winged, as is typical of wind-dispersed conifers. Clark's nutcrackers, squirrels, and other animals cache seeds, but animals may predate on seeds to a greater degree than serve as agents of seed dispersal (Coop and Schoettle 2009). Cone production can occur by 40 years of age, but production of a full cone crop takes far longer; maturation rates likely vary with growing conditions. Good seed crops are produced every 2 to 3 years, with reproductive output greatest in high-elevation and low-density stands (Schoettle and Coop 2017). Potential climate drivers for cone production include negative associations with growing degree days and maximum summer temperatures, as well as the mean temperature of the coldest month (Schoettle and Coop 2017). At the level of individual trees, cone production is weakly related to tree basal area.

Ecology of Seedlings, Saplings, and Mature Trees

RM bristlecone pine is limited by available habitat at the southern edge of its distribution, but it is unknown what limits it farther north into the abundant suitable habitat occupied by limber pine. Its northern range limit coincides with the reliable extent of the North American Monsoon, suggesting that seedling establishment may require late summer precipitation (Schoettle 2004). This hypothesis is further supported by the observation that RM bristlecone pine seedlings have a lower root:shoot ratio than limber pine, and prioritizing root growth provides limber pine with greater tolerance to inconsistent moisture availability during seedling establishment (Borgman et al. 2014).

While mature RM bristlecone pine trees are highly drought-tolerant (drought tolerance score of 4.9 out of 5.0; Niinemets and Valladares 2006), drought can still affect tree growth. Early studies have related antecedent moisture and temperature to growth in high-elevation stands of RM bristlecone pine. In recent decades, the severity of droughts in the southern Rocky Mountains and southwestern United States has increased (Allen et al. 2010; Williams et al. 2022) and this is evident in the radial growth patterns of RM bristlecone pine (Tintor and Woodhouse 2021). A large portion of interannual variability (52 percent) in annual ring growth of high-elevation RM bristlecone pines was attributed to June water stress (Tintor and Woodhouse 2021).

RM bristlecone pine is relatively shade-intolerant (shade tolerance score of 1.3 out of 5.0; Niinemets and Valladares 2006), and establishment is favored on exposed, rocky sites with relatively acidic, bare mineral soil, presumably due to competitive exclusion

on better sites (Baker 1992). However, young seedlings can also be found underneath nurse canopies of mature trees and beside other objects (Coop and Schoettle 2009).

RM bristlecone pine fire regimes can vary considerably with topography, elevation, and other environmental conditions. While a stand-replacing fire regime may be prevalent in some settings (Baker 1992), at least some stands may be best characterized by a low-severity fire regime. Fire-scarred RM bristlecone pines in open, low-density stands with grassy understories occur nearly throughout the range of the species, particularly where stands are adjacent to montane and subalpine grasslands (Coop and Schoettle 2011). Numerous fire histories have been reconstructed from RM bristlecone pines scarred by low-severity fires with mean fire intervals ranging from 7 to 103 years (mean of 40 years; Coop and Schoettle 2011). Stand density tends to increase with elevation, but fire behavior can be moderated in RM bristlecone pine stands by the lack of surface fuel continuity and widely spaced crowns. The regeneration response of RM bristlecone pine to fire appears to be the subject of some uncertainty. Baker (1992) concluded that bristlecone pine is a "long-lived pioneer that regenerates primarily after fire," while Brown and Schoettle (2008) found regeneration to be associated with long fire-free intervals, and Cocke and others (2005) reported nearly continuous regeneration in the Arizona populations. Consideration of the temporal and spatial scale of recovery can help reconcile these apparent contradictions. Open conditions that favor RM bristlecone pine seedling establishment may persist for decades to centuries after disturbance in these dry southern Rockies settings (Coop et al. 2010; Shankman and Daly 1988). Coop and Schoettle (2009) observed poor postfire regeneration three decades after two stand-replacing fires and RM bristlecone pine populations were low in burn interiors compared to unburned stands. However, seedling numbers were elevated near or beneath surviving trees, and total density was increased in partially burned patches. Seed availability and microsites that provide shelter strongly influence postfire RM bristlecone pine regeneration. On more favorable sites, regeneration densities of other species can be orders of magnitude greater than RM bristlecone pine due to the latter's relatively low fecundity, likely leading to its competitive exclusion over time (Moir and Ludwig 1979).

Both white pine blister rust and mountain pine beetle can cause significant mortality in RM bristlecone pine. White pine blister rust was first discovered on RM bristlecone pine in 2003 in the Sangre de Cristo Mountains of southern Colorado and is still spreading in the southern Rockies (fig. 16; Blodgett and Sullivan 2004; Burns 2006). The first detected RM bristlecone pine trees infected with blister rust were predominantly in moist, riparian areas, where the primary alternate host (*Ribes* spp.) is common and conditions are conducive to C. ribicola spore production. Infections are currently less common on the upland drier slopes (Burns 2006). RM bristlecone pine is susceptible to white pine blister rust, but early studies suggest that RM bristlecone pine has modest levels

of genetic resistance (Schoettle et al. 2022b); more studies are needed to understand the frequency and distribution of resistance traits. RM bristlecone pine has moderate constitutive defenses against mountain pine beetle, a native insect (Bentz et al. 2021). As with other pines, mountain pine beetle kills RM bristlecone by mass attack. More information on these and other insects and diseases is included in Chapter 8 (Negrón 2024, this report).

Potential Direct Impacts of Climate Change

Drought is already a feature of RM bristlecone's environment. Drought and temperature-related mortality has not been widely documented for RM bristlecone pine. However, drought may be contributing to the substantial recent crown and branch dieback in individuals observed across much of the species' range (Schoettle and Coop 2017), and the forecasted changes in climate may intensify drought as a primary stressor in upcoming decades.

The bioclimatic envelope models of Crookston (2009) predicted that RM bristlecone pine's climatic niche may nearly vanish from the Front Range and the rest of Colorado by 2060 (table 3; fig. 18). Results indicated that 3.1 million ac (~1.3 million ha) in the Front Range are currently climatically suitable for bristlecone. However, under a future climate similar to the Hot climate scenario, approximately 87 percent of this area was predicted to be unsuitable by 2060, and an additional 6 percent was predicted to be only marginally suitable. Moreover, a negligible amount (< 0.1 million ac (< 0.1 million ha)) of currently unsuitable area was predicted to become suitable by 2060. However, when Malone et al. (2018b) developed range-wide bioclimatic envelope models that incorporated distinct RM bristlecone pine genetic lineages, the results presented a slightly more positive forecast; they projected a 74 percent decrease in the bristlecone pine suitable climate space by 2090.

The relationships of cone production to climate factors are suggestive of potential sensitivity to climate warming (Schoettle and Coop 2017). Likewise, the importance of the amount and timing of moisture to seedling establishment also suggests a vulnerability to increases in climate variability. A study in northern Arizona, well within the area of influence of the North American Monsoon, showed continuous recruitment of RM bristlecone pine (Cocke et al. 2005). These observations suggest that changes to the intensity and regularity of monsoonal precipitation may impact future RM bristlecone pine recruitment (Schoettle et al. 2022a). A lack of regeneration in some stands indicates that older trees can persist in suboptimal environments that are currently outside the conditions suitable for seedling establishment (Schoettle and Coop 2017).

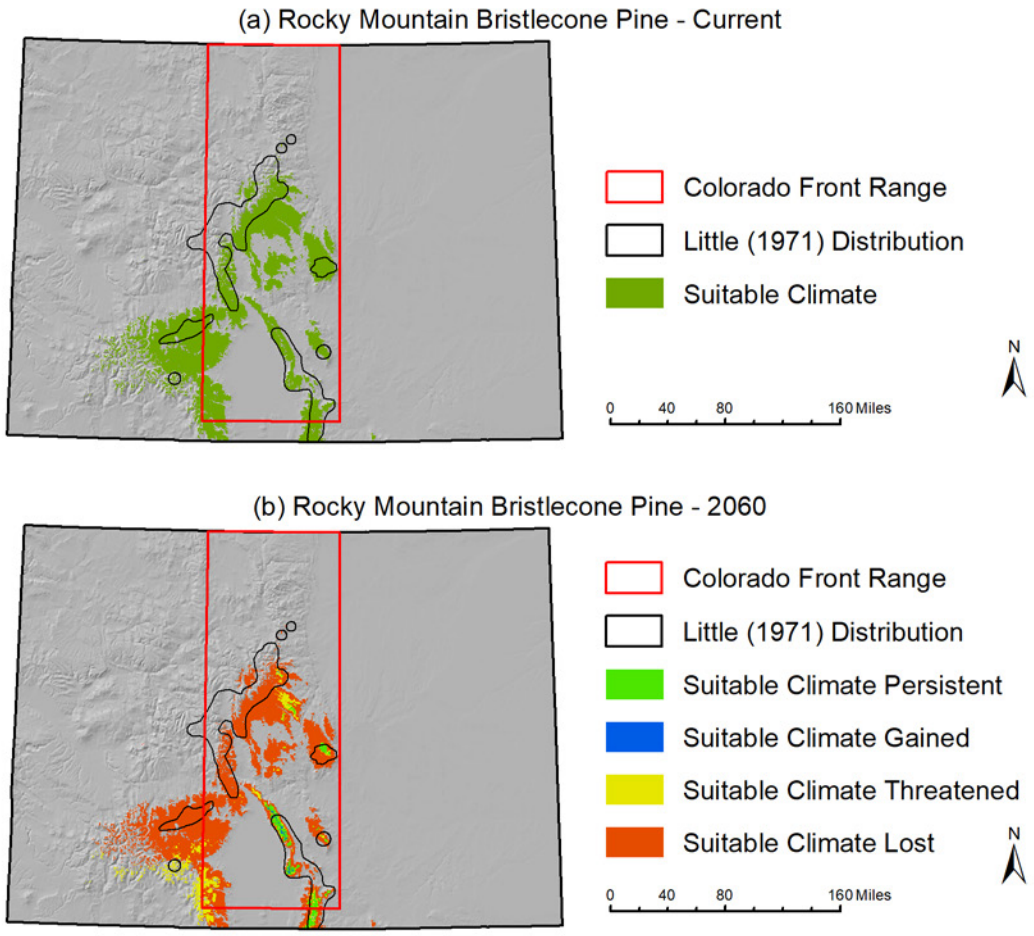


Figure 18a-b—The current and potential future suitable climate space of Rocky Mountain bristlecone pine (Crookston 2009). Panel (a) depicts areas in Colorado where the current climate is suitable for the species. Panel (b) depicts areas in Colorado where suitable climate in the future is predicted to be either lost, threatened, persistent, or gained in 2060 under the HADCM3-A2 general circulation model (GCM), a GCM that crosswalks relatively well to the one representing our Hot future climate scenario (2090 data are unavailable for this species). See box 1 for more information about how current and potential future climate suitability were determined. For all panels, the map of Little (1971) is used to represent the general current distribution of the species.

Potential Indirect Impacts of Climate Change

Insects and Diseases

During the recent epidemic, mountain pine beetle populations were far greater in northern than southern Colorado and therefore did not overlap with much of RM bristlecone pine's distribution (Bentz et al. 2021). Severe drought conditions on Telescope Peak in eastern California have enabled mountain pine beetle to overcome defenses in Great Basin bristlecone pine, resulting in high tree mortality (Bentz et al. 2022); however, it is unknown if a future drought in combination with a beetle epidemic will lead to a similar outcome for RM bristlecone pine.

Capacity to Adapt to Climate Change

Individual RM bristlecone pine trees can attain lifespans of > 2,000 years, making them the oldest trees in Colorado (Brunstein and Yamaguchi 1992). This extreme longevity suggests that trees are able to tolerate considerable variations in climate. However, it is unclear what mechanisms enable trees to be so tolerant of climatic variability.

RM bristlecone pine has a very low genetic diversity, which increases its vulnerability to climate change (Schoettle et al. 2012b). While the overall genetic diversity is low, there is variation among populations that is likely a result of limited gene flow across distant mountaintop populations. This variation is correlated with site temperature and precipitation and suggests that the isolation has led to local adaptation (Schoettle et al. 2012b). The populations along the Front Range harbor the greatest genetic diversity and may serve as an important source of genetic variation to support adaptation in the future (Schoettle et al. 2012b). Mortality caused by white pine blister rust, and the direct and indirect effects of a changing climate may lead to a reduction of the already low overall genetic diversity through the loss of rare alleles or extirpation of small, isolated populations (Kim et al. 2003).

Evolutionary adaptation is most likely to occur under conditions where environmental change is gradual, the initial population size is large, generation time is short, and genetic variation is high (Aitken et al. 2008). Unfortunately, none of these conditions exist for RM bristlecone pine in Colorado. As discussed above, the habitat of RM bristlecone pine is rapidly changing, its populations are small and isolated, the species has delayed maturation and slow population turnover, and its genetic diversity is low. In total, the species is not positioned well for sustained evolutionary potential in a changing climate. RM bristlecone pine has tolerance to many individual stressors, but the novel combinations of the stressors coupled with the species' low genetic diversity and conservative life history strategy limit its capacity for rapid adaptation, making it vulnerable to irreversible impacts (Schoettle et al. 2022a). Proactive management, e.g., plantings and silvicultural treatments done before populations decline, to increase the abundance of younger cohorts with more diverse genetic combinations would improve RM bristlecone pine's adaptive capacity (Schoettle and Coop 2017; Schoettle and Sniezko 2007; Schoettle et al. 2012a,b; Schoettle et al. 2019b).

Subalpine Fir (*Abies lasiocarpa* var. *lasiocarpa*)

Climate change vulnerability rating: moderate to high

Confidence in climate change vulnerability rating: Limited information with moderate agreement

Subalpine fir is highly sensitive to warm, dry conditions, and climate change is expected to directly compromise the survival of regenerating and mature trees. A Very Hot and Dry future climate scenario would be the most detrimental to subalpine fir survival in the Colorado Front Range; however, all scenarios would likely incite some level of mortality. Bioclimatic envelope models on par with the Hot future climate scenario suggest that by 2060, suitable climate space will be lost primarily in low elevations, with only marginal gains at higher elevations. In addition to these direct effects of warming, climate change will compromise subalpine fir indirectly by increasing the prevalence of fire and of an insect and disease disorder, subalpine fir decline. Very little information exists about subalpine fir's ability to respond plastically to changing environmental conditions, to evolve, and to migrate, but what is known suggests that it may have only some capacity to adapt through these mechanisms.

Habitat

Subalpine fir is widely distributed in western North America. Its range extends from the Yukon to Arizona and New Mexico north to south, and from Alaska to Colorado west to east (Alexander et al. 1990; Little 1971; NRCS 2020). Subalpine fir is also widely distributed in the Front Range, dominating or co-dominating about 0.5 million ac (0.2 million ha) of forest on the PSI-AR alone (ARP 2019; Little 1971; NRCS 2020; PSICC 2019).

Subalpine fir occurs in areas in the Front Range where winters are long and cold and summers are short and cool. On the PSI-AR, the subalpine fir zone has long-term (1981 to 2010) mean annual precipitation values between ca. 21 and 39 in (54 and 98 cm), and long-term mean annual temperature values between ca. 32 and 38 °F (0 and 4 °C) (ARP 2019; PRISM 2020; PSICC 2019). Most precipitation falls as snow to form a deep winter snowpack. This zone ranges in elevation from ca. 9,100 to 11,700 ft (2,700 to 3,600 m) (ARP 2019; PSICC 2019; USGS 2020). Subalpine fir grows on a wide range of soil types (Alexander et al. 1990).

In the Front Range, subalpine fir is a key member of subalpine forests (table 1; figs. 1, 4). Subalpine fir rarely grows in pure stands (ARP 2019; PSICC 2019). It most commonly grows in stands with Engelmann spruce; it is also commonly found growing in stands with lodgepole pine and quaking aspen (ARP 2019; PSICC 2019). Understories in stands with a subalpine fir component vary in their composition but commonly include heartleaf arnica (*Arnica cordifolia*), gooseberry currant (*Ribes montigenum*), and blueberry (*Vaccinium* spp.) (Johnston 1987; Peet 1981). Subalpine fir can be either a very long-lived seral species or a climax species (Alexander et al. 1990).

Ecology

Reproductive Ecology

While subalpine fir can reproduce vegetatively by layering, particularly near treeline, its primary mode of reproduction is by seed (Alexander et al. 1990; Holtmeier and Broll 2017). Reproductive maturity in the Front Range can occur around 25 to 30 years of age under open conditions, but can be delayed by many decades under closed canopies (Andrus et al. 2020b). An appreciable amount of viable seed may only be produced a couple times a decade (Noble and Ronco 1978). Seeds are wind-dispersed, and most probably fall within about 264 ft (80 m) of parent trees (Noble and Ronco 1978). Seeds that have been collected and stored indoors can lose viability quickly if not kept under ideal conditions, and so the longevity of seeds in the soil seedbank is probably only a year or so (Edwards 2008).

Ecology of Seedlings, Saplings, and Mature Trees

Consistent with its preference for habitats with cold winters, cool summers, and deep winter snowpacks, subalpine fir is relatively intolerant of drought. Niinemets and Valladares (2006) assigned it a drought tolerance score of 2.0 out of 5.0, a score lower than that of all major Front Range tree species in the subalpine zone except quaking aspen. Seedling establishment events in the Front Range, for example, can be less apt to occur in years with lower spring snowpacks and/or with warmer and drier summer conditions than average (Andrus et al. 2018; Hessl and Baker 1997). The generally slow growth of mature subalpine fir trees also can be slowed further by warmer and drier conditions, particularly at sites that are already on the warm, dry end of the climatic spectrum (Alexander et al. 1990; Villabla et al. 1994). Increases in the mortality of mature subalpine fir trees can likewise be driven by warmer and drier conditions (Andrus et al. 2021; Bigler et al. 2007; Reich et al. 2016).

Subalpine fir is well-adapted to shade. Its shade tolerance score is 4.8 out of 5.0, which is higher than the scores given to its major tree associates in the Front Range (Niinemets and Valladares 2006). Subalpine fir seedlings in particular can germinate and establish in deep forest environments with a high degree of shade and a thick forest floor (Knapp and Smith 1982). Seedlings and saplings can persist in the understory for decades and then quickly release following a low- to moderate-severity overstory disturbance event such as an insect outbreak, as was found at the Fraser Experimental Forest on the AR (Collins et al. 2011). However, more severe overstory disturbance events that dramatically reduce shade can cause declines in seedling and sapling growth (Collins et al. 2011).

The high-elevation forests supporting subalpine fir generally experience significant wildfire events only once every ca. 200 to 500 years or longer (Romme and Knight 1981; Schoennagel et al. 2004; Sibold et al. 2006). Often these events occur in association

with extreme region-wide droughts, as climatic conditions in high-elevation forests are otherwise usually insufficient to support wildfire (Higuera et al. 2014; Sibold and Veblen 2006; Sibold et al. 2006). If wildfire can be supported climatically, fire behavior tends to be extreme due to high fuel loads, and subalpine fir mortality tends to be high (Schoennagel et al. 2004; Sibold et al. 2006). However, even when fire behavior is less severe, subalpine fir mortality can still be high, as it lacks any notable fire-resistant traits (Stevens et al. 2020). Subalpine fir establishment after wildfire tends to be protracted, peaking one to many decades following the establishment of an initial cohort of more shade-intolerant tree species like lodgepole pine, quaking aspen, or Engelmann spruce (Aplet et al. 1988; Veblen 1986; Veblen et al. 1991a). That said, subalpine fir can still make up a notable proportion of the initial postfire cohort, depending on patterns of burn severity and adjacent forest composition (Aplet et al. 1988; Schapira et al. 2021; Veblen 1986).

Subalpine fir decline, a complex but poorly understood disorder brought about by multiple interacting factors, is currently regarded as one of the most widespread causes of subalpine fir mortality in the Colorado Front Range (Harvey et al. 2021; Lalande et al. 2020; Reich et al. 2016). The western balsam bark beetle (*Dryocoetes confusus*), a native insect that preferentially kills older trees of low vigor, is typically a major contributor to subalpine fir decline (Harvey et al. 2021; Lalande et al. 2020). Beyond the western balsam bark beetle, contributors to the decline commonly include high stand density, drought, Armillaria (*Armillaria* spp.) root disease, and other insect and disease agents (Harvey et al. 2021; Lalande et al. 2020; Reich et al. 2016). A current, widespread decline event has been ongoing in the Front Range and across Colorado since the early 1990s; between 2008 and 2017, it killed subalpine fir trees across more than 1.8 million ac (0.7 million ha) (Harvey et al. 2021; Lalande et al. 2020; Reich et al. 2016). Surviving subalpine fir trees in the understory are commonly available to recruit into the overstory following subalpine fir decline, fostering resilience (Redmond and Kelsey 2018).

Potential Direct Impacts of Climate Change

Climate change appears to have already caused increases in the mortality of mature subalpine fir trees in the northern Front Range (Andrus et al. 2021; Smith et al. 2015). Annual mortality rates due to climate-driven stress increased over a 37-year period (1982 to 2019), paralleling increases in summer climatic water deficit and the frequency of extremely warm (90th percentile) summer days. All four future climate scenarios would therefore be likely to cause additional increases in subalpine fir mortality rates, particularly the Very Hot and Dry scenario.

Climate change also has already driven reductions in the frequency of subalpine fir establishment events in the northern Front Range in recent years (Andrus et al. 2018). These reductions, observed over a 35-year period (1975 to 2010), were coincident with declining spring snowpacks and warmer and drier summer conditions. Further reductions in establishment events would therefore likely continue under all four future climate scenarios examined here, but would likely be the least severe under the Warm and Wet scenario and the most severe under the Very Hot and Dry scenario.

A bioclimatic envelope modeling study provides insight into how future climate change, and in particular the Hot future climate scenario, may lead to a net loss of suitable climate space for subalpine fir in the Front Range (Crookston 2009). Approximately 4.5 million ac (1.8 million ha) in the Front Range were estimated to be currently climatically suitable for this species (box 1; table 3; fig. 19). By 2060, only 3.4 million ac (1.4 million ha) were predicted to be climatically suitable, and by 2090, that number dropped to 1.3 million ac (0.5 million ha). Additionally, 23 percent (2060) and 71 percent (2090) of this suitable area was categorized as threatened, indicating that its climatic suitability was predicted to be marginal at best. Areas that were predicted to be climatically suitable for subalpine fir in 2060 and/or 2090 were typically at higher elevations relative to current climatically suitable areas.

Potential Indirect Impacts of Climate Change

Wildfire

Climate change has already increased the frequency of drought in subalpine forests across the northern Front Range and surrounding area, leading to an increase in area burned that is unprecedented in the last 2,000 years (Higuera et al. 2021). Indeed, in just the last 20 years (2000 to 2020), the fire rotation interval for this landscape has shortened from every 204 to 315 years to 117 years (Higuera et al. 2021). The trend of increasing fire activity is likely to continue, especially under the Hot and the Very Hot and Dry climate scenarios (Heidari et al. 2021; Rangwala 2024, this report; Rocca et al. 2014; Temperli et al. 2015; Turner et al. 2022). Because subalpine fir is poorly adapted to survive even low-severity wildfire, continued increases in fire activity could drive major losses of the species (Stevens et al. 2020). Moreover, because subalpine fir can establish poorly in the first few decades after wildfire even under ideal climatic conditions, and because it is expected to have fewer establishment events under a warmer climate, these losses could be long-lasting (Andrus et al. 2018; Aplet et al. 1988; Davis et al. 2023; Veblen 1986; Veblen et al. 1991a).

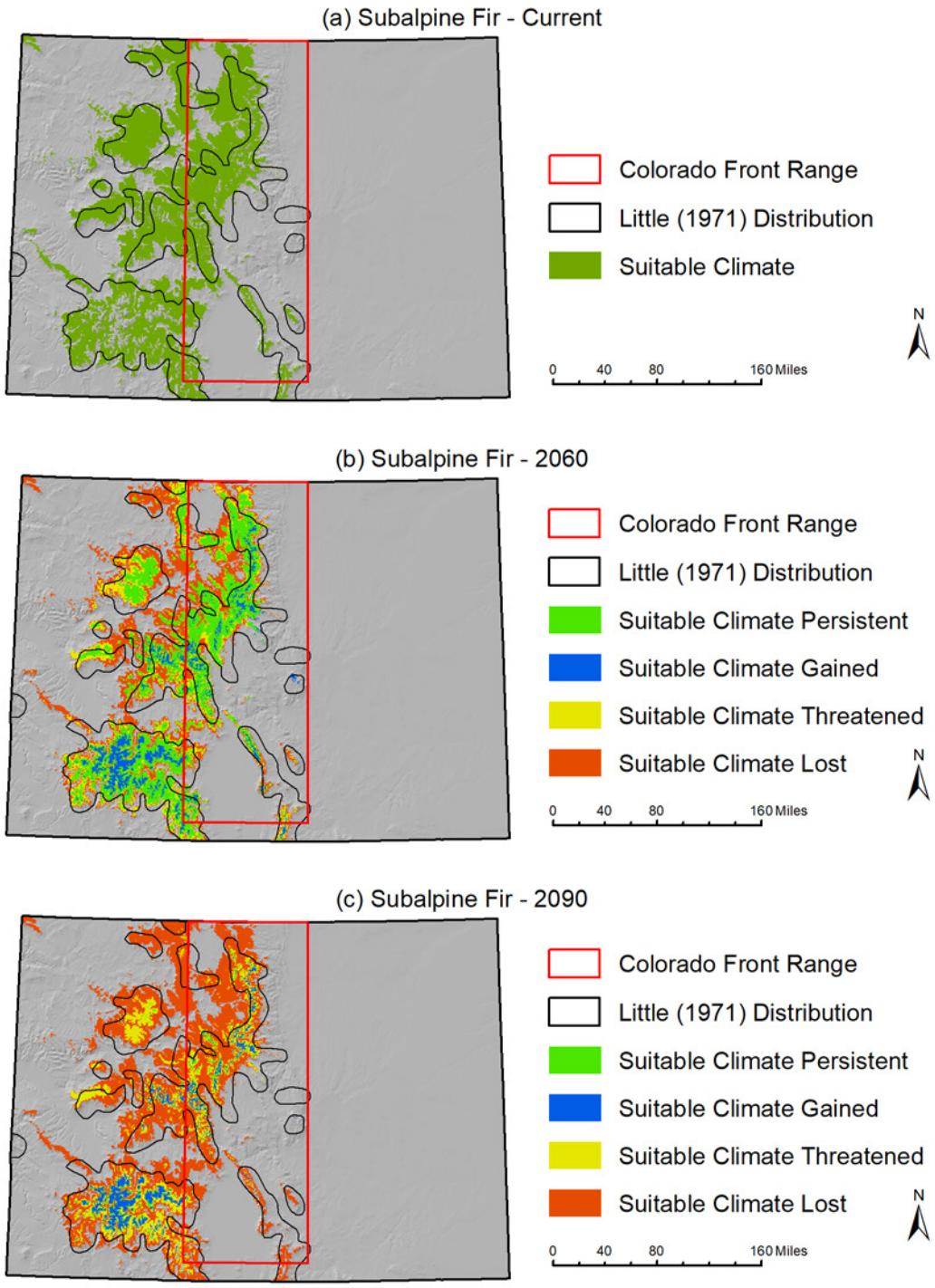


Figure 19a-c—The current and potential future suitable climate space of subalpine fir (Crookston 2009). Panel (a) depicts areas in Colorado where the current climate is suitable for the species. Panels (b) and (c) depict areas in Colorado where suitable climate in the future is predicted to be either lost, threatened, persistent, or gained under the HADCM3-A2 general circulation model (GCM), a GCM that crosswalks relatively well to the one representing our Hot future climate scenario. See box 1 for more information about how current and potential future climate suitability were determined. For all panels, the map of Little (1971) is used to represent the general current distribution of the species.

Insects and Diseases

Severe drought appears to be a major driver of subalpine fir mortality attributed to subalpine fir decline; Harvey et al. (2021) showed that at the scale of the U.S. Rocky Mountains, the extent of subalpine fir decline increased during and following years with high Palmer Drought Severity Indices (i.e., extreme droughts). However, the role of warm, dry sites is less clear. Reich et al. (2016) found that in Colorado, the decline was concentrated in marginal portions of the Engelmann spruce-subalpine fir zone, where site conditions are relatively warm and dry relative to the core of the zone. Meanwhile, Harvey et al. (2021) found that in the Colorado Front Range, the decline was more prevalent in wetter sites, and speculated that this may be because wetter sites had greater subalpine fir abundance and therefore had more exposure to the decline's biotic components (e.g., western balsam bark beetles and *Armillaria* root disease) than their drier counterparts. Despite the uncertainty surrounding the role of site conditions, it is likely that subalpine fir mortality due to subalpine fir decline will become increasingly prevalent across the Front Range under a changing climate, particularly if climatic changes increase the frequency of extreme droughts, as the Hot, Very Hot and Wet, and Very Hot and Dry scenarios do.

Subalpine fir may potentially benefit from climate-driven insect outbreaks that kill co-occurring tree species. Rodman et al. (2022a) compiled field data from 16 studies conducted in beetle-impacted subalpine forests across the southern Rockies and analyzed the data to characterize postoutbreak forest recovery. They found that subalpine fir was dominant in only about 10 percent of the plots prior to the outbreaks, but after the outbreaks it was dominant in about 28 percent of the plots. Moreover, growth simulations done with the Forest Vegetation Simulator predicted that subalpine fir would be dominant in about 32 percent of the plots in 30 years. Subalpine fir dominance may be able to be retained through climatic change in some of these expanded sites, particularly those that are in cooler, wetter environments.

Capacity to Adapt to Climate Change

A rainfall exclusion study of established subalpine fir and Engelmann spruce seedlings near Mount Evans indicated that both species acclimate poorly to summer drought conditions (Goke and Martin 2022). Net photosynthesis declines were twice as severe for spruce compared to fir, yet neither species showed significant stomatal control in response to drought treatments (Goke and Martin 2022). The authors suggest that seedlings of these species prioritize carbon assimilation over drought mitigation, which may leave them vulnerable to future droughts (Goke and Martin 2022). Yet traits like water use efficiency and photosynthetic capacity vary among adult individuals across open and closed canopy forests, suggesting saplings and mature trees exhibit a higher degree of physiological plasticity than seedlings (Broderson et al. 2006; Carter and Smith 1988; Knapp and Smith 1981).

The limited genetics research on subalpine fir in the Front Range suggests that the species has some potential to adapt evolutionarily to climate change. Shea (1990) collected genetic material of both Engelmann spruce and subalpine fir from two adjacent stands at Niwot Ridge, finding that the overall genetic diversity of fir was comparable to that of spruce as well as to that found elsewhere for other conifer associates, such as lodgepole pine and Douglas-fir. Shea (1990) also indicated that fir's genetic diversity was much greater across space than across age classes, likely due to site-related selection pressures and relatively short pollen and seed dispersal distances that limit gene flow.

Subalpine fir's current distribution indicates that there is some opportunity for it to migrate within the Front Range in response to climate change. The southern limit of this species currently extends into New Mexico, which may enable it to withstand any latitudinal range shifts that might occur (Alexander and Shepperd 1990; Little 1971). Additionally, there are opportunities for gains in climatically suitable area above the current upper elevational limit, although these are considerably smaller than projected losses at the species' lower elevational limit (Crookston 2009). However, subalpine fir's relatively large seeds with short dispersal distances, low fecundity, and time to maturity make it unlikely to migrate in pace with climate change.

Engelmann Spruce (*Picea engelmannii* var. *engelmannii*)

Climate change vulnerability rating: moderate to high

Confidence in climate change vulnerability rating: moderate information with moderate to high agreement

We expect Engelmann spruce in the Front Range to be moderately to highly vulnerable to climate change. Increases in temperature are predicted under all four future climate scenarios. Warmer temperatures and concomitant reductions in available moisture will likely directly drive losses in spruce abundance and decreases in spruce growth, except perhaps at the coolest, wettest sites, where increases in temperature may benefit spruce. More significantly, though, warmer temperatures will likely indirectly drive losses in spruce abundance by driving increases in area burned and in spruce beetle activity; indeed, such increases have already been documented, particularly in the northern Front Range. Engelmann spruce's ability to migrate in response to a changing climate will likely be restricted by limited upslope opportunities, relatively short seed dispersal distances, and long generation times. Thus, any potential future gains in spruce habitat will likely be outweighed by losses in warmer, drier sites and in areas experiencing wildfire and/or spruce beetle disturbances.

Habitat

Engelmann spruce is a broadly distributed North American conifer. It ranges from British Columbia and Alberta south to Arizona and New Mexico, and from Washington, Oregon, and California east to Montana, Wyoming, and Colorado (Alexander and Shepperd 1990; Little 1971; NRCS 2020). Engelmann spruce is prominent in the Front Range (ARP 2019; Little 1971; NRCS 2020; PSICC 2019). It is a major component of forests on roughly 1.2 million ac (0.5 million ha) of the PSI-AR, as well as on many thousands of other acres, such as at Rocky Mountain National Park (ARP 2019; PSICC 2019).

Engelmann spruce occupies high-elevation areas with long, cold winters and short, cool summers. Long-term (1981 to 2010) mean annual precipitation in the PSI-AR Engelmann spruce zone, for example, ranges from about 17 to 39 in (44 to 98 cm), while long-term mean annual temperature ranges from about 32 to 40 °F (0 to 5 °C) (ARP 2019; PRISM 2020; PSICC 2019). Most precipitation falls as snow, with a winter snowpack typically persisting from October or November to May or June. Engelmann spruce on the PSI-AR generally occurs from approximately 9,100 to 11,700 ft in elevation (2,700 to 3,600 m) (ARP 2019; PSICC 2019; USGS 2020). It often defines the upper treeline at the highest elevations, particularly on mesic sites and north-facing slopes. Engelmann spruce grows on a variety of soil types, so long as they are moist (Alexander and Shepperd 1990).

Engelmann spruce in the Front Range is a key subalpine forest species (table 1; figs. 1, 4). While it can grow in pure stands, it is most commonly found growing alongside subalpine fir (ARP 2019; PSICC 2019). It is also commonly found growing with lodgepole

pine, quaking aspen, and Rocky Mountain bristlecone pine (ARP 2019; PSICC 2019). Herbaceous and shrubby species in the understory of Engelmann spruce dominated and co-dominated forests are varied, but can include heartleaf arnica (*Arnica cordifolia*), gooseberry currant (*Ribes montigenum*), and blueberry (*Vaccinium* spp.) (Johnston 1987; Peet 1981). Engelmann spruce can be extremely long-lived (> 800 years old) and can therefore function as either a seral or a climax species (Alexander and Shepperd 1990; Brown et al. 1995).

Ecology

Reproductive Ecology

Engelmann spruce regenerates primarily from seeds; layering is not thought to contribute significantly to regeneration except near treeline (Alexander and Shepperd 1990; Holtmeier and Broll 2017). Engelmann spruce seed production is episodic and appears to be at least partially controlled by climate. For example, yearly seed production at the Fraser Experimental Forest was highly variable over a 40-year time period, with intermediate levels of production occurring regularly and high levels occurring only infrequently (Buechling et al. 2016). High seed production during this period was shaped by climatic conditions across multiple years, including low spring snowfall (and putatively increased growing season length) a year before seed dispersal and high summer temperatures in the year of seed dispersal (Buechling et al. 2016). The small, winged seeds are wind-dispersed, and most fall within 300 ft (91 m) of parent trees, although some seeds can fall as far as 600 ft (183 m) from parent trees (Alexander and Edminster 1983). Seeds reaching the soil seedbank are readily lost due to predation and a short viability window (Johnson and Fryer 1996).

Ecology of Seedlings, Saplings, and Mature Trees

Engelmann spruce is only somewhat drought tolerant. Niinemets and Valladares (2006) calculated its drought tolerance score as 2.6 out of 5.0. The roots of Engelmann spruce seedlings develop slowly, making them particularly susceptible to dry conditions (Noble 1973). Alexander (1984) found that adequate soil moisture was a leading determinant of survival for seedlings growing in a seed sowing experiment at the Fraser Experimental Forest. Andrus et al. (2018) found that broad-scale Engelmann spruce establishment events in the northern Front Range tended to occur in years when the spring snowpack water content was greater than average and/or when summers were cooler and wetter than average. Hill et al. (2019) showed that on the Gunnison National Forest, Colorado, later dates of last frost (indicative of cooler, wetter conditions in late spring and early summer) were correlated with greater Engelmann spruce seedling establishment, but that there was a tension between it and longer frost-free periods, which also increased establishment. Drought can also affect mature Engelmann spruce trees, yet the direction of the response is often site-specific. For example, Villalba et al. (1994) found that Engelmann spruce trees on xeric sites in the northern Front Range tended to put on wider annual growth rings in years with wetter growing seasons than with drier growing seasons. Additionally, at xeric sites, they found that the rings of Engelmann

spruce trees tended to be wider when growing season temperatures were cooler than when they were warmer (Villalba et al. 1994). However, at mesic to hydric sites, greater precipitation did not tend to strongly affect growth as water was not limiting, and cooler temperatures actually decreased growth (Villalba et al. 1994). Bigler et al. (2007) showed that Engelmann spruce trees in the northern Front Range have experienced episodes of drought-induced mortality over the last ~100 years, with slower-growing trees more susceptible than faster-growing trees.

Engelmann spruce is tolerant of shade. It has a shade tolerance score of 4.5 out of 5.0 (Niinemets and Valladares 2006), only slightly lower than that of its most common associate, subalpine fir. In fact, researchers working in the northern Front Range have demonstrated that for seedlings, some degree of shading strongly enhances survival and growth, and on southern aspects, it is essential (Knapp and Smith 1982; Noble and Alexander 1977). The benefits of shade appear to diminish as Engelmann spruce ages (Jacobs 2011).

The forests supporting Engelmann spruce are adapted to infrequent, extensive, high-severity wildfire events. Historically, such events typically occurred once every ~200 to 500 years or longer, often in association with extremely dry regional climatic conditions (Romme and Knight 1981; Sibold and Veblen 2006; Sibold et al. 2006). These conditions predisposed forests, which generally contained abundant fuels, to extreme fire behavior over large areas (Schoennagel et al. 2004; Sibold et al. 2006). Mortality from fire was typically high, as Engelmann spruce does not have many fire-adapted traits that promote survival (Stevens et al. 2020). After historical wildfires, Engelmann spruce was commonly a prominent member of the initial cohort of regenerating trees, although this cohort could take one or more decades to establish (Aplet et al. 1988; Veblen 1986; Veblen et al. 1991a).

Spruce beetles (*Dendroctonus rufipennis*) are the most significant insect and disease agent of Engelmann spruce in the Front Range. This native species has long caused widespread and severe mortality when beetle populations reach epidemic levels, particularly for large mature trees (Hart et al. 2014b; Rodman et al. 2022a; Veblen et al. 1991b). Epidemics appear to be triggered, at least in part, by warmer and/or drier climatic conditions (DeRose et al. 2013; Hart et al. 2014a, 2017). One such epidemic has recently occurred across Colorado, affecting an estimated ~1.0 million ac (~400,000 ha) between 2009 and 2018 (Derderian et al. 2016; Hart et al. 2017; Hicke et al. 2020). Chapter 8 contains additional information about spruce beetles and other insects and diseases affecting Engelmann spruce (Negrón 2024, this report).

Potential Direct Impacts of Climate Change

Three field-based observational studies have already documented negative climate change effects on Engelmann spruce in the Front Range. The first of these studies, conducted by Andrus et al. (2018), examined the frequency of broad-scale Engelmann spruce establishment events in the northern Front Range, and how these events were

tied to climate. As mentioned above, they found that establishment events tended to occur in years when the spring snowpack water content was greater than average and/or when summers were cooler and wetter than average. They also found that in the recent half of the study period (1975 to 2010), a downward trend in the number of establishment events coincided with trends of declining snowpacks and increasing summer temperatures and moisture deficits. The other two studies, conducted by Andrus et al. (2021) and Smith et al. (2015), examined how background mortality rates of mature Engelmann spruce trees at Niwot Ridge correlated with climate variability. Their findings indicated that mortality rates increased over a 37-year period (1982 to 2019), particularly in old growth stands, and that increases in mortality rates coincided with warmer and drier climatic conditions.

A field-based experiment by Kueppers et al. (2017) provides insight into how a warmer future climate may limit Engelmann spruce regeneration in the Front Range. Working at forest, treeline, and alpine sites at Niwot Ridge, the authors looked at how warming treatments affected the recruitment of Engelmann spruce seedlings over a 4-year period. They found that warming treatments decreased first-year seedling recruitment at all sites, including the treeline and alpine sites. However, seedlings became less sensitive to warming as they matured over the next 3 years, possibly because many of the most heat-sensitive individuals had already been culled.

The bioclimatic envelope modeling done by Crookston (2009) suggests that the availability of suitable climate space for Engelmann spruce will decline dramatically in the Front Range under a future climate analogous to that of our Hot scenario. Model results indicated that currently, approximately 5.8 million ac (2.3 million ha) are climatically suitable for this species (box 1; table 3; fig. 20). By 2060, only 3.1 million ac (1.2 million ha) were predicted to be climatically suitable; by 2090, that number fell to only 1.7 million ac (687,000 ha). Moreover, 20 percent (2060) and 82 percent (2090) of these suitable acres were categorized as threatened, meaning that their climatic suitability was predicted to be marginal at best.

Gray and Hamann (2013) also used bioclimatic envelope modeling to predict that, across the U.S. Rocky Mountains, future suitable climate space for Engelmann spruce would move up in elevation. They used an ensemble of 18 GCMs, and thus their results may be most comparable to our middle-of-the-road Hot future climate scenario. Specifically, they predicted that relative to the suitable climate space of the 1961 to 1990 reference period, the 2050 climate space of Engelmann spruce will be approximately 548 ft (167 m) higher.

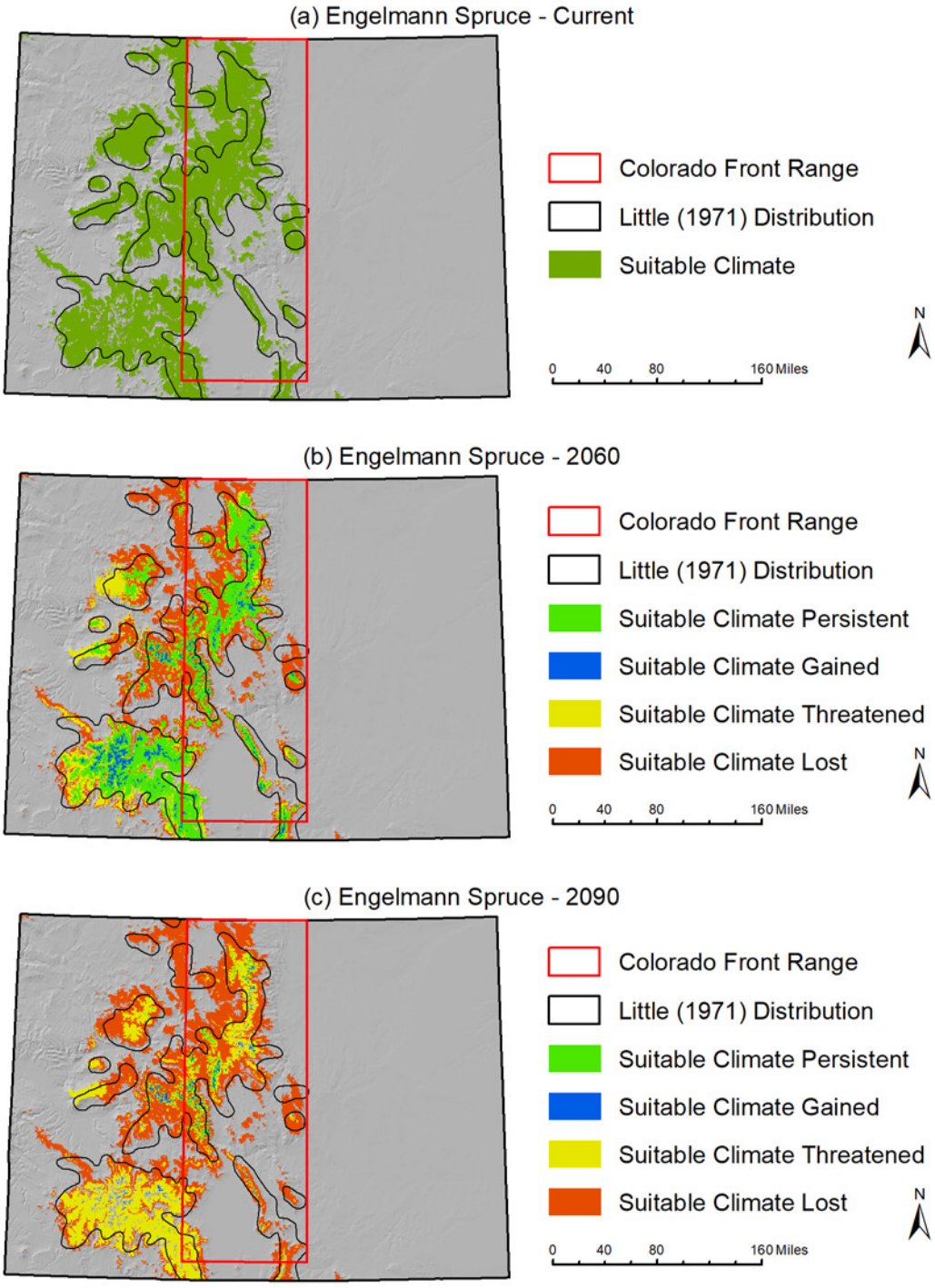


Figure 20a-c—The current and potential future suitable climate space of Engelmann spruce (Crookston 2009). Panel (a) depicts areas in Colorado where the current climate is suitable for the species. Panels (b) and (c) depict areas in Colorado where suitable climate in the future is predicted to be either lost, threatened, persistent, or gained under the HADCM3-A2 general circulation model (GCM), a GCM that crosswalks relatively well to the one representing our Hot future climate scenario. See box 1 for more information about how current and potential future climate suitability were determined. For all panels, the map of Little (1971) is used to represent the general current distribution of the species.

Potential Indirect Impacts of Climate Change

Wildfire

Subalpine forests have naturally high fuel loads, and historically they tended to burn with stand-replacing severity every two or more centuries during extreme drought years (Higuera et al. 2014; Sibold and Veblen 2006; Sibold et al. 2006). Climate change is increasing the frequency of drought in subalpine forests, leading to an increase in annual area burned that is unprecedented in the last 2,000 years (table 2; Higuera et al. 2021). The record-setting 2020 wildfire season exemplified this: the drought-driven Cameron Peak, East Troublesome, and Williams Fork Fires together burned over 412,000 ac (167,000 ha) in the northern Front Range, often as crown fires that resulted in considerable Engelmann spruce mortality (table 2; Rodman et al. 2022b). Increases in area burned are likely to continue in the future, especially under the warmest and driest future climate scenarios (Heidari et al. 2021; Rocca et al. 2014; Temperli et al. 2015; Turner et al. 2022; Yue et al. 2013). Engelmann spruce is poorly equipped to survive even low-severity fire, and thus future increases in fire activity would be expected to drive considerable increases in mortality (Stevens et al. 2020).

Engelmann spruce is adapted to recover from its native stand-replacing wildfire regime through postfire seedling establishment (Aplet et al. 1988; Veblen 1986; Veblen et al. 1991a). While protracted seedling establishment following wildfires is not necessarily uncharacteristic for the species, a changing climate already appears to be slowing establishment rates (Andrus et al. 2018, Aplet et al. 1988; Davis et al. 2023; Veblen 1986; Veblen et al. 1991a). Future climatic changes over the next 10 to 30 years may further restrict seedling establishment, particularly at lower-elevation and south-facing sites (Davis et al. 2023).

Insects and Diseases

Recent spruce beetle outbreaks have caused substantial losses of mature, large-diameter spruce trees across the western United States, including on the PSI-AR (fig. 21; DeRose et al. 2013; Hart et al. 2017; Hicke et al. 2020; Rodman et al. 2022a). Warmer and/or drier conditions appear to have played an important role in fostering recent outbreaks. For example, Hart et al. (2017) found that in the southern Rocky Mountains, the initiation and spread of recent bark beetle outbreaks were positively associated with various measures of drought. Similarly, DeRose et al. (2013) found that across the Interior West, higher maximum summer and minimum winter temperatures were related to spruce beetle outbreak occurrences. Collectively, these authors posit that warmer, drier conditions may have increased beetle survival and reproduction, while simultaneously stressing trees and reducing their capacity to defend against beetle attacks. Moreover, analyses conducted with a range of GCMs suggested that continued warming and drying will exacerbate outbreaks, at least until host trees are depleted (DeRose et al. 2013; Foster et al. 2018; Temperli et al. 2015).



Figure 21—Recent spruce beetle outbreaks have killed a large number of Engelmann spruce trees in the Front Range, including these trees near the Long Draw Road on the Roosevelt National Forest. Warm and/or dry conditions appear to have played an important role in fostering these outbreaks (DeRose et al. 2013; Hart et al. 2017). Continued warming and/or drying will likely exacerbate outbreaks, at least until host trees are depleted (DeRose et al. 2013; Foster et al. 2018; Temperli et al. 2015). USDA Forest Service photo by Paula Fornwalt.

Following the mortality of mature Engelmann spruce trees due to spruce beetle, spruce recovery is most commonly facilitated by subcanopy trees that established prior to the outbreak (Andrus et al. 2020a; Redmond and Kelsey 2018; Rodman et al. 2022a; Veblen et al. 1991b). In contrast to wildfire, it is uncommon for a stand previously dominated or co-dominated by spruce to be completely devoid of preoutbreak spruce trees (Andrus et al. 2020a; Rodman et al. 2022a; Veblen et al. 1991b). The vulnerability of these surviving trees, and of any newly establishing trees, to climate change would likely be similar to that of undisturbed stands (Andrus et al. 2018; Crookston 2009; Gray and Hamann 2013; Kueppers et al. 2017).

Capacity to Adapt to Climate Change

Research on Engelmann spruce plasticity, while limited, suggests that seedlings have little innate ability to tolerate climate change. Goke and Martin (2022) conducted a two-year rainfall exclusion study of established Engelmann spruce seedlings near Mount Evans, and their findings indicated that spruce seedlings acclimated poorly to simulated drought conditions. Relative to ambient conditions, spruce seedlings exposed to simulated drought exhibited net photosynthesis declines of 78 percent, and they showed little ability to close stomates to increase water use efficiency (Goke and Martin 2022). However, larger spruce seedlings showed better stomatal control than smaller seedlings, suggesting that individuals may be able to better respond to drought as they grow (Brodersen et al. 2006; Goke and Martin 2022; Knapp and Smith 1981).

The genetic diversity of Engelmann spruce may afford it some opportunity to evolutionarily adapt in situ to climate change. Shea (1990) found a fair amount of genetic diversity across individuals at Niwot Ridge, for example. That said, Shea (1990) also found that for individuals in very close proximity (around 177 ft [54 m]), genetic diversity was much more limited, and is probably an artifact of local selection pressures and limited gene flow due to short pollen and seed dispersal distances.

Engelmann spruce may also have some ability to withstand climate change in the Front Range through migration. The species is broadly distributed, which could potentially allow for latitudinal and elevational migration within the current Engelmann spruce zone (Rehfeldt 2004). Elevational migration above the current Engelmann spruce zone is also possible, although it will likely be highly constrained by topography (Crookston 2009; Gray and Hamann 2013). Moreover, the migration of Engelmann spruce both latitudinally and elevationally is likely to be hampered by a variety of factors, including seed dispersal distances and the time required to reach reproductive maturity; thus, migration rates are unlikely to keep pace with climate change (Alexander and Edminster 1983; Alexander and Shepperd 1990).

Conclusions

Take-Home Trends

In writing this chapter, we aimed to provide land managers with detailed assessments of the vulnerability of nine major tree species in the Front Range to climate change. A summary of our assessments is presented in table 4.

Table 4—The climate change vulnerability rating assigned in this assessment of nine major tree species in the Colorado Front Range, along with the confidence in that rating.

Species	Vulnerability rating	Confidence in vulnerability rating
Twoneedle pinyon	Moderate	Limited information with moderate agreement
Ponderosa pine	Moderate	Moderate information with high agreement
Douglas-fir	Moderate	Moderate information with high agreement
Lodgepole pine	Moderate to high	Moderate information with moderate agreement
Quaking aspen	Moderate	Moderate information with moderate agreement
Limber pine	High	Moderate information with moderate agreement
Rocky Mountain bristlecone pine	Moderate to high	Moderate information with moderate agreement
Subalpine fir	Moderate to high	Limited information with moderate agreement
Engelmann spruce	Moderate to high	Moderate information with moderate to high agreement

As our confidence ratings suggest, a major trend that emerged is that the vulnerability of all tree species is somewhat uncertain due to unknown patterns and rates of future climate change, as well as due to research gaps and conflicting research findings. Thus, though our vulnerability assessments were grounded in the available relevant literature, we view them more as educated guesses than as definitive evaluations. Some species were particularly understudied. For example, very little research has been done on twoneedle pinyon in the Front Range, one of the few species that may stand to benefit under climate change. However, even for more well-studied species such as ponderosa pine and Engelmann spruce, our confidence in the overall vulnerability rating was never particularly high. Additional research for each of the nine tree species would help boost this confidence.

Differences in vulnerability ratings across tree species indicate that lower-elevation species such as twoneedle pinyon, ponderosa pine, and Douglas-fir may be somewhat less vulnerable to climate change than higher-elevation species like Engelmann spruce, subalpine fir, and Rocky Mountain bristlecone pine. Other vulnerability assessments have arrived at similar conclusions, including those done for the northern Rocky Mountains of Idaho, Montana, and Wyoming, and for the Pacific Northwest of Washington and Oregon (Devine et al. 2012; Hansen and Phillips 2015; Keane et al. 2018). In part, this trend may be because lower-elevation species tend to have more opportunities for upslope migration, and because they tend to be more adapted to frequent disturbance, especially wildfire.

A final trend worth noting is that climate change appears poised to impact tree species more through indirect mechanisms than direct ones. In particular, climate change × wildfire interactions have the potential to cause significant mortality across large areas of each species' range, and for most species, to drive major declines in subsequent tree establishment. Insects and disease are also expected to interact with climate change to cause significant negative impacts for many species.

Adaptation Strategies

It was beyond the scope of this chapter to translate our assessments into adaptation strategies that managers could implement to reduce ongoing and anticipated negative climate change impacts to Front Range forests and woodlands (Peterson et al. 2011). This is clearly a critical next step. Fortunately, many managers are already conducting activities that should help buffer forests and woodlands from climate change effects. For example, tree thinning treatments are occurring across the ponderosa pine and Douglas-fir zones to reduce high-severity wildfire risk stemming from overly dense forest conditions and ongoing and future climate warming (Barrett et al. 2021; Briggs et al. 2017; Cannon et al. 2018). Not only should these treatments mitigate wildfire risk, they should improve the vigor of the remaining trees and decrease their potential for mortality due to drought (Bradford et al. 2022). Similarly, planting treatments are ongoing in the ponderosa pine and Douglas-fir zones in areas where these species are not naturally re-establishing after high-severity wildfire (Guiterman et al. 2022; Marshall et al. 2023). In box 2, we highlight an adaptation strategy that is being implemented to conserve limber pine and RM bristlecone pine in the face of white pine blister rust, climate change, and other stressors. We hope that the content presented in this chapter can aid managers as they develop additional adaptation strategies to help Front Range forests and woodlands weather changing climatic conditions.

Box 2

Adapting limber pine and Rocky Mountain bristlecone pine to promote their persistence in the face of white pine blister rust, climate change, and other stressors

Cronartium ribicola, the nonnative fungal pathogen that causes the lethal disease white pine blister rust, was first discovered in Colorado in 1998 on limber pine on the Roosevelt NF and in 2003 on Rocky Mountain bristlecone pine (along with limber pine) on the San Isabel NF and the Great Sand Dunes National Park and Preserve. Since then, the pathogen has continued to spread. It is estimated that more than half of the limber and Rocky Mountain bristlecone pine habitat in Colorado regularly has climatic conditions suitable for blister rust invasion and the other portions may also become infected as the pathogen becomes more common on the landscape (Howell et al. 2006; Kearns et al. 2014). The life cycle of *C. ribicola* requires two hosts: a five-needle white pine and an alternate host, most commonly *Ribes* spp. (McDonald et al. 2006). Infections are shed annually by the deciduous alternate hosts and are perennial on the pine hosts and accumulate over the life of the tree (Geils et al. 2010). The airborne spores of the pathogen produced on the alternate host enter the tree via needles and the fungus colonizes the branches and grows into the tree bole (fig. 16). The resultant cankers kill branches and eventually the tree. White pine blister rust infects and kills trees of all ages and threatens population sustainability. Branch mortality reduces seed production, reducing regeneration capacity well before tree mortality. Regeneration capacity is further reduced by seedling mortality.

The compounding effects of blister rust, as well as mountain pine beetle, Ips beetle, and dwarf mistletoe, in a warming climate threaten the persistence of limber pine and Rocky Mountain bristlecone pine and their ecosystems. These same stressors have impacted whitebark pine in the northern U.S. Rockies to such an extent that the species was listed as Threatened under the Endangered Species Act in fall 2022 (USFWS 2022). The proportion of dead limber pine lags behind that observed in whitebark pine by approximately a decade, suggesting that limber pine populations may be following a similar trajectory as whitebark pine (Goeking and Windmuller-Campione 2021). Limber pine is already proposed for listing under the Canadian Species at Risk Act (COSEWIC 2014; Government of Canada 2012).

Active management is recommended in the southern U.S. Rockies to accelerate adaptation to white pine blister rust and prevent limber pine and Rocky Mountain bristlecone pine from following the same trajectory as whitebark pine or limber pine to the north (Burns et al. 2008; Schoettle et al. 2019a). Paramount to this effort is increasing the number of trees with genetic resistance to blister rust in forest stands through seedling planting and protecting resistant seed trees from mortality by other stressors (e.g., mountain pine beetle and fire). Although blister rust and these pines have not co-evolved, low levels of genetic resistance to the pathogen naturally occur in both species (Schoettle et al. 2011, 2014, 2022b). However, more work is needed to identify trees and populations with genetic resistance. Including limber pine and bristlecone pine in postdisturbance planting mixes with other species (e.g., Engelmann spruce) would help forest recovery on the harsher portions of the treatment area and provide trees in the area that will spread genetic resistance traits in pollen and seed to nearby pines and forests. Because these pine species mature so slowly, proactively establishing trees with blister rust resistance in healthy stands in anticipation of blister rust invasion is also recommended to accelerate the increase and spread of disease-resistant genes (Schoettle et al. 2019b, 2022a).

References

Abatzoglou, J.T.; Williams, A.P. 2016. Impact of anthropogenic climate change on wildfire across western U.S. forests. Proceedings of the National Academy of Sciences of the United States of America. 113(42): 11770–11775. https://doi.org/10.1073/pnas.1607171113

Adams, H.D.; Kolb, T.E. 2004. Drought responses of conifers in ecotone forests of northern Arizona: Tree ring growth and leaf δ13C. Oecologia. 140(2): 217–225. https://doi.org/10.1007/s00442-004-1585-4

Adams, H.D.; Kolb, T.E. 2005. Tree growth response to drought and temperature in a mountain landscape in northern Arizona, USA. Journal of Biogeography. 32(9): 1629–1640. https://doi.org/10.1111/j.1365-2699.2005.01292.x

Aitken, S.N.; Yeaman, S.; Holliday, J.A.; [et al.]. 2008. Adaptation, migration or extirpation: Climate change outcomes for tree populations. Evolutionary Applications. 1: 95–111. https://doi.org/10.1111/j.1752-4571.2007.00013.x

Alberta Whitebark and Limber Pine Recovery Team [Alberta]. 2014. Alberta limber pine recovery plan 2014–2019. Alberta Species at Risk Recovery Plan No. 35. Edmonton, Alberta: Alberta Environment and Sustainable Resource Development. 61 p. https://open.alberta.ca/dataset/45ceb67a-7b44-4beb-a5a4-2b72a9b2e118/resource/05362a91-7f67-4af3-b8ea-e8e40f91a5dc/download/2014-sar-limberpine-recoveryplan-sep2014.pdf [Accessed 6 June 2024]

Alexander, R.R. 1974. Silviculture of subalpine forests in the central and southern Rocky Mountains: The status of our knowledge. Research Paper RM-121. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station.

Alexander, R.R. 1984. Natural regeneration of Engelmann spruce after clearcutting in the central Rocky Mountains in relation to environmental factors. Research Paper RM-254. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. https://www.fs.usda.gov/research/treesearch/65304

Alexander, R.R.; Edminster, C.B. 1983. Engelmann spruce seed dispersal in the central Rocky Mountains. Research Note RM-424. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. https://www.fs.usda.gov/research/treesearch/65300

Alexander, R.R; Shearer, R.C.; Shepperd, W.D. 1990. *Abies lasiocarpa* (Hook.) Nutt. Subalpine fir. In: Silvics of North America, Volume 1: Conifers. Agriculture Handbook https://www.fs.usda.gov/research/treesearch/1547

Alexander, R.R; Shepperd, W.D. 1990. *Picea engelmannii* Parry ex Engelm. Engelmann spruce. In: Silvics of North America, Volume 1: Conifers. Agriculture Handbook 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 187–203. https://www.fs.usda.gov/research/treesearch/1547

Allen, C.D.; Macalady, A.K.; Chenchouni, H.; [et al.]. 2010. A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. Forest Ecology and Management. 259: 660–684. https://doi.org/10.1016/j.foreco.2009.09.001

Andrus, R.A.; Chai, R.K.; Harvey, B.J.; [et al.]. 2021. Increasing rates of subalpine tree mortality linked to warmer and drier summers. Journal of Ecology. 109(5): 2203–2218. https://doi.org/10.1111/1365-2745.13634

Andrus, R.A.; Hart, S.J.; Veblen, T.T. 2020a. Forest recovery following synchronous outbreaks of spruce and western balsam bark beetle is slowed by ungulate browsing. Ecology. 101(5): e02998. https://doi.org/10.1002/ecy.2998

Andrus, R.A.; Harvey, B.J.; Hoffman, A.; [et al.]. 2020b. Reproductive maturity and cone abundance vary with tree size and stand basal area for two widely distributed conifers. Ecosphere. 11(5): e03092. https://doi.org/10.1002/ecs2.3092

Andrus, R.A.; Harvey, B.J.; Rodman, K.C.; [et al.]. 2018. Moisture availability limits subalpine tree establishment. Ecology. 99: 567–575. https://doi.org/10.1002/ecy.2134

Aoki, C.F.; Romme, W.H.; Rocca, M.E. 2011. Lodgepole pine seed germination following tree death from mountain pine beetle attack in Colorado, USA. The American Midland Naturalist. 165(2): 446–451. https://doi.org/10.1674/0003-0031-165.2.446

Aplet, G.H.; Laven, R.D.; Smith, F.W. 1988. Patterns of community dynamics in Colorado Engelmann spruce-subalpine fir forests. Ecology. 69(2): 312–319. https://doi.org/10.2307/1940429

Arapaho and Roosevelt National Forests and Pawnee National Grassland [ARP]. 2019. Field sampled vegetation spatial database. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Arapaho and Roosevelt National Forests and Pawnee National Grassland.

Araújo, M.B.; Peterson, A.T. 2012. Uses and misuses of bioclimatic envelope modeling. Ecology. 93(7): 1527–1539. https://doi.org/10.1890/11-1930.1

Bai, Y.; Thompson, D.; Broersma, K. 2004. Douglas-fir and ponderosa pine seed dormancy as regulated by grassland seedbed conditions. Rangeland Ecology and Management. 57: 661–667. https://doi.org/10.2111/1551-5028(2004)057[0661:DFAPPS]2.0.CO;2

Baker, W.L. 1992. Structure, disturbance, and change in the bristlecone pine forests of Colorado, USA. Arctic and Alpine Research. 24: 17–26. https://doi.org/10.2307/1551316

Barger, N.N.; Adams, H.D.; Woodhouse, C.; [et al.]. 2009. Influence of livestock grazing and climate on pinyon pine (*Pinus edulis*) dynamics. Rangeland Ecology and Management. 62(6): 531–539. https://doi.org/10.2111/.1/REM-D-09-00029.1

Barger, N.N.; Woodhouse, C. 2015. Piñon pine (*Pinus edulis* Engelm.) growth responses to climate and substrate in southern Utah, USA. Plant Ecology. 216(7): 913–923. https://doi.org/10.1007/s11258-015-0478-4

Barrett, J.W. 1966. A record of ponderosa pine seed flight. Research Note PNW-38. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station.

Barrett, K.J.; Cannon, J.B.; Schuetter, A.M.; [et al.]. 2021. Effects of collaborative monitoring and adaptive management on restoration outcomes in dry conifer forests. Forest Ecology and Management. 488: 119018. https://doi.org/10.1016/j.foreco.2021.119018

Battaglia, M.A.; Gannon, B.; Brown, P.M.; [et al.]. 2018. Changes in forest structure since 1860 in ponderosa pine dominated forests in the Colorado and Wyoming Front Range, USA. Forest Ecology and Management. 422: 147–160. https://doi.org/10.1016/j.foreco.2018.04.010

Battaglia, M.; Smith, F.W.; Shepperd, W.D. 2009. Predicting mortality of ponderosa pine regeneration after prescribed fire in the Black Hills, South Dakota, USA. International Journal of Wildland Fire. 18(2): 176–190. https://doi.org/10.1071/WF07163

Bentz, B.J.; Hansen, E.M.; Vandygriff, J.C.; [et al.]. 2021. Rocky Mountain bristlecone pine (*Pinus aristata*) is a confirmed host to mountain pine beetle (*Dendroctonus ponderosae*). Western North American Naturalist. 81: 19–26. https://doi.org/10.3398/064.081.0102

Bentz, B.J.; Hood, S.A.; Hansen, E.M.; [et al.]. 2016. Defense traits in the long-lived Great Basin bristlecone pine and resistance to the native herbivore mountain pine beetle. New Phytologist. 213(2): 611–624. https://doi.org/10.1111/nph.14191

Bentz, B.J.; Millar, C.; Vandygriff, J.; [et al.]. 2022. Great Basin bristlecone pine mortality: Causal factors and management implications. Forest Ecology and Management. 509: 120099. https://doi.org/10.1016/j.foreco.2022.120099

Bentz, B.J.; Régnière, J.; Fettig, C.J. 2010. Climate change and bark beetles of the western United States and Canada: Direct and indirect effects. BioScience. 60(8): 602–613. https://doi.org/10.1525/bio.2010.60.8.6

Bigler, C.; Gavin, D.G.; Gunning, C.; [et al.]. 2007. Drought induces lagged tree mortality in a subalpine forest in the Rocky Mountains. Oikos. 116(12): 1983–1994. http://dx.doi.org/10.1111/j.2007.0030-1299.16034.x

Blodgett, J.T.; Sullivan, K.F. 2004. First report of white pine blister rust on Rocky Mountain bristlecone pine. Plant Disease. 88(3): 311–311. https://doi.org/10.1094/PDIS.2004.88.3.311A

Borgman, E.M.; Schoettle, A.W.; Angert, A.L. 2014. Using among-year variation to assess maternal effects in *Pinus aristata* and *Pinus flexilis*. Botany. 92: 805–814. https://doi.org/10.1139/cjb-2014-0085

Borgman, E.M.; Schoettle, A.W.; Angert, A.L. 2015. Assessing the potential for maladaptation during active management of limber pine populations: A common garden study detects genetic differentiation in response to soil moisture in the southern Rocky Mountains. Canadian Journal of Forestry Research. 45(4): 496–505. http://dx.doi.org/10.1139/cjfr-2014-0399

Bottero, A.; D'Amato, A.W.; Palik, B.J.; [et al.]. 2017. Density-dependent vulnerability of forest ecosystems to drought. Journal of Applied Ecology. 54(6): 1605–1614. https://doi.org/10.1111/1365-2664.12847

Bower, A.D.; McLane, S.C.; Eckert, A.; [et al.]. 2011. Conservation genetics of high elevation five-needle white pines. In: Keane, R.E.; Tomback, D.F.; Murray, M.P.; [et al.], eds. The future of high-elevation, five-needle white pines in Western North America: Proceedings of the high five symposium. Proceedings RMRS-P-63. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 98–117. https://www.fs.usda.gov/research/treesearch/38207

Bradford, J.B.; Birdsey, R.A.; Joyce, L.A. 2008. Tree age, disturbance history, and carbon stocks and fluxes in subalpine Rocky Mountain forests. Global Change Biology. 14(12): 2882–2897. https://doi.org/10.1111/j.1365-2486.2008.01686.x

Bradford, J.B.; Shriver, R.K.; Robles, M.D.; [et al.]. 2022. Tree mortality response to drought-density interactions suggests opportunities to enhance drought resistance. Journal of Applied Ecology. 59(2): 549–559. https://doi.org/10.1111/1365-2664.14073

Brandt, L.; He, H.; Iverson, L.; [et al.]. 2014. Central hardwoods ecosystem vulnerability assessment and synthesis: A report from the central hardwoods climate change response framework project. Gen. Tech. Rep. NRS-124. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 254 p. https://doi.org/10.2737/NRS-GTR-124

Breshears, D.D.; Carroll, C.J.; Redmond, M.D.; [et al.] 2018. A dirty dozen ways to die: Metrics and modifiers of mortality driven by drought and warming for a tree species. Frontiers in Forests and Global Change. 1: 4. https://doi.org/10.3389/ffgc.2018.00004

Breshears, D.D.; Cobb, N.S.; Rich, P.M.; [et al.]. 2005. Regional vegetation die-off in response to global-change-type drought. Proceedings of the National Academy of Sciences of the United States of America. 102(42): 15144–15148. https://doi.org/10.1073/pnas.0505734102

Bretfeld, M.; Franklin, S.B.; Peet, R.K. 2016. A multiple-scale assessment of long-term aspen persistence and elevational range shifts in the Colorado Front Range. Ecological Monographs. 86(2): 244–260. https://doi.org/10.1890/15-1195.1

Briggs, J.S.; Fornwalt, P.J.; Feinstein, J.A. 2017. Short-term ecological consequences of collaborative restoration treatments in ponderosa pine forests of Colorado. Forest Ecology and Management. 395: 69–80. https://doi.org/10.1016/j.foreco.2017.03.008

Brodersen, C.R.; Germino, M.J.; Smith, W.K. 2006. Photosynthesis during an episodic drought in *Abies lasiocarpa* and *Picea engelmannii* across an alpine treeline. Arctic, Antarctic, and Alpine Research. 38(1): 34–41. https://doi.org/10.1657/1523-0430(2006)038[0034:PDAEDI]2.0.CO;2

Brown, P.M.; Battaglia, M.A.; Fornwalt, P.J.; [et al.]. 2015. Historical (1860) forest structure in ponderosa pine forests of the northern Front Range, Colorado. Canadian Journal of Forest Research. 45(11): 1462–1473. https://doi.org/10.1139/cjfr-2014-0387

Brown, P.M.; Kaufmann, M.R.; Shepperd, W.D. 1999. Long-term, landscape patterns of past fire events in a montane ponderosa pine forest of central Colorado. Landscape Ecology. 14(6): 513–532. https://doi.org/10.1023/A:1008137005355

Brown, P.M.; Schoettle, A.W. 2008. Fire and stand history in two limber pine (*Pinus flexilis*) and Rocky Mountain bristlecone pine (*Pinus aristata*) stands in Colorado. International Journal of Wildland Fire. 17(3): 339–347. https://doi.org/10.1071/wf06159

Brown, P.M.; Shepperd W.D.; Brown C.C.; [et al.]. 1995. Oldest known Engelmann spruce. Research Note RM-RN-534. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 534 p. https://doi.org/10.2737/RM-RN-534

Brunstein, F.C.; Yamaguchi, D.K. 1992. The oldest known Rocky Mountain bristlecone pines (*Pinus aristata* Engelm). Arctic and Alpine Research. 24: 253–256. https://doi.org/10.1080/00040851.1992.12002955

Buechling, A.; Martin, P.H.; Canham, C.D.; [et al.]. 2016. Climate drivers of seed production in *Picea engelmannii* and response to warming temperatures in the southern Rocky Mountains. Journal of Ecology. 104(4): 1051–1062. https://doi.org/10.1111/1365-2745.12572

Burns, D.A. 2004. The effects of atmospheric nitrogen deposition in the Rocky Mountains of Colorado and southern Wyoming, USA—A critical review. Environmental Pollution. 127(2): 257–269. https://doi.org/10.1016/S0269-7491(03)00264-1

Burns, K.S. 2006. White pine blister rust surveys in the Sangre de Cristo and Wet Mountains of southern Colorado. Biological Evaluation R2-06-05. Lakewood, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Region, Renewable Resources. 22 p.

Burns, K.S.; Schoettle, A.W.; Jacobi, W.R.; [et al.]. 2008. Options for the management of white pine blister rust in the Rocky Mountain Region. Gen. Tech. Rep. RMRS-GTR-206. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 26 p. https://doi.org/10.2737/RMRS-GTR-206

Burns, K.S.; Tinkham, W.T.; Leddy, K.A.; [et al.]. 2023. Interactions between white pine blister rust, bark beetles, and climate over time indicate vulnerabilities to limber pine health. Frontiers in Forests and Global Change. 6: 1149456. https://doi.org/10.3389/ffgc.2023.1149456

Burns, R.M.; Honkala, B.H., tech. coords. 1990a. Silvics of North America: 1. Conifers. Agriculture Handbook 654. Washington, DC: U.S. Department of Agriculture, Forest Service. 675 p. https://www.fs.usda.gov/research/treesearch/1547

Burns, R.M.; Honkala, B.H., tech. coords. 1990b. Silvics of North America: 2. Hardwoods. Agriculture Handbook 654. Washington, DC: U.S. Department of Agriculture, Forest Service. 877 p. https://www.fs.usda.gov/research/treesearch/1548

Butler, P.R.; Iverson, L.; Thompson III, F.R.; [et al.]. 2015. Central Appalachians forest ecosystem vulnerability assessment and synthesis: A report from the Central Appalachians climate change response framework project. Gen. Tech. Rep. NRS-146. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 310 p. https://doi.org/10.2737/NRS-GTR-146

Cannon, J.B.; Barrett, K.J.; Gannon, B.M.; [et al.]. 2018. Collaborative restoration effects on forest structure in ponderosa pine-dominated forests of Colorado. Forest Ecology and Management. 424: 191–204. http://dx.doi.org/10.1016/j.foreco.2018.04.026

Carroll, C.J.; Knapp, A.K.; Martin, P.H. 2017. Dominant tree species of the Colorado Rockies have divergent physiological and morphological responses to warming. Forest Ecology and Management. 402: 234–240. http://dx.doi.org/10.1016/j.foreco.2017.07.048

Carroll, C.J.; Martin, P.H.; Knapp, A.K. 2019. Temperature induced shifts in leaf water relations and growth efficiency indicate climate change may limit aspen growth in the Colorado Rockies. Environmental and Experimental Botany. 159: 132–137. http://dx.doi.org/10.1016/j.envexpbot.2018.12.014

Carsey, K.S.; Tomback, D.F. 1994. Growth form distribution and genetic relationships in tree clusters of *Pinus flexilis*, a bird-dispersed pine. Oecologia. 98: 402–411. https://doi.org/10.1007/BF00324230

Carter, G.A.; Smith, W.K. 1988. Microhabitat comparisons of transpiration and photosynthesis in three subalpine conifers. Canadian Journal of Botany. 66(5): 963–969. https://doi.org/10.1139/b88-139

Casper, A.M.A.; Jacobi, W.R.; Schoettle, A.W.; [et al.]. 2016. Restoration planting options for limber pine (*Pinus flexilis* James) in the southern Rocky Mountains. Journal of the Torrey Botanical Society. 143(1): 21–37. https://doi.org/10.3159/TORREY-D-14-00085

Cerezke, H.F. 1995. Egg gallery, brood production, and adult characteristics of mountain pine beetle, *Dendroctonus ponderosae* Hopkins (Coleoptera: Scolytidae), in three pine hosts. The Canadian Entomologist. 127: 955–965. https://doi.org/10.4039/Ent127955-6

Chambers, M.E.; Fornwalt, P.J.; Malone, S.L.; [et al.]. 2016. Patterns of conifer regeneration following high-severity wildfire in ponderosa pine-dominated forests of the Colorado Front Range. Forest Ecology and Management. 378: 57–67. https://doi.org/10.1016/j.foreco.2016.07.001

Chapman, T.B.; Schoennagel, T.; Veblen, T.T.; [et al.]. 2020. Still standing: Recent patterns of post-fire conifer refugia in ponderosa pine-dominated forests of the Colorado Front Range. PLoS One. 15(1): e0226926. https://doi.org/10.1371/journal.pone.0226926

Chapman, T.B.; Veblen, T.T.; Schoennagel, T. 2012. Spatiotemporal patterns of mountain pine beetle activity in the southern Rocky Mountains. Ecology. 93(10): 2175–2185. https://doi.org/10.1890/11-1055.1

Clark, J.S.; Andrus, R.; Aubry-Kientz, M.; [et al.]. 2021. Continent-wide tree fecundity driven by indirect climate effects. Nature Communications. 12: 1242. https://doi.org/10.1038/s41467-020-20836-3

Cleaver, C.M.; Jacobi, W.R.; Burns, K.S.; [et al.]. 2015. Limber pine in the central and southern Rocky Mountains: Stand conditions and interactions with blister rust, mistletoe, and bark beetles. Forest Ecology and Management. 358: 139–153. https://doi.org/10.1016/j.foreco.2015.09.010

Cleaver, C.M.; Jacobi, W.R.; Burns, K.S.; [et al.]. 2016. Limber pine regeneration and white pine blister rust in the central and southern Rocky Mountains. Forest Science. 63(2): 151–164. https://doi.org/10.5849/forsci.16-052

Clement, J.M.; Cheng, A.S. 2011. Using analyses of public value orientations, attitudes and preferences to inform National Forest planning in Colorado and Wyoming. Applied Geography. 31: 393–400. https://doi.org/10.1016/j.apgeog.2010.10.001

Cocke, A.E.; Fule, P.Z.; Crouse, J.E. 2005. Forest change on a steep mountain gradient after extended fire exclusion: San Francisco Peaks, Arizona, USA. Journal of Applied Ecology. 42: 814-823. https://doi.org/10.1111/j.1365-2664.2005.01077.x

Cole, H.M.; Andrus, R.A.; Butkiewicz, C.; [et al.]. 2022. Outbreaks of Douglas-fir beetle follow western spruce budworm defoliation in the southern Rocky Mountains, USA. Forests. 13(3): 371. https://doi.org/10.3390/f13030371

Cole, K.L.; Ironside, K.E.; Arundel, S.T.; [et al.]. 2008. Modeling future plant distributions on the Colorado Plateau: An example using *Pinus edulis*. In: van Riper, C. III; Sogge, M.K., eds. The Colorado Plateau III. Integrating research and resources management for effective conservation. Tucson, AZ: The University of Arizona Press: 319–330. https://www.fs.usda.gov/research/treesearch/31525

Collins, B.J.; Rhoades, C.C.; Hubbard; R.M.; [et al.]. 2011. Tree regeneration and future stand development after bark beetle infestation and harvesting in Colorado lodgepole pine stands. Forest Ecology and Management. 261(11): 2168–2175. https://doi.org/10.1016/j.foreco.2011.03.016

Coop, J.D.; Massatti, R.T.; Schoettle, A.W. 2010. Subalpine vegetation three decades after stand-replacing fire: Effects of landscape context and topography on plant community composition, tree regeneration, and diversity. Journal of Vegetation Science. 21: 472–487. https://doi.org/10.1111/j.1654-1103.2009.01154.x

Coop, J.D.; Parks, S.A.; Stevens-Rumann, C.S; [et al.]. 2022. Extreme fire spread events and area burned under recent and future climate in the western USA. Global Ecology and Biogeography. 31: 1949–1959. https://doi.org/10.1111/geb.13496

Coop, J.D.; Schoettle, A.W. 2009. Regeneration of Rocky Mountain bristlecone pine (*Pinus aristata*) and limber pine (*Pinus flexilis*) three decades after stand-replacing fires. Forest Ecology and Management. 257(3): 893–903. https://doi.org/10.1016/j.foreco.2008.10.034

Coop, J.D.; Schoettle, A.W. 2011. Fire and high-elevation, five-needle pine (*Pinus aristata* & *P. flexilis*) ecosystems in the southern Rocky Mountains: What do we know? In: Keane, R.E.; Tomback, D.F.; Murray, M.P.; [et al.], eds. The future of high-elevation, five-needle white pines in Western North America: Proceedings of the high five symposium. Proceedings RMRS-P-63. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 164–173. https://www.fs.usda.gov/research/treesearch/38217

Committee on the Status of Endangered Wildlife in Canada [COSEWIC]. 2014. COSEWIC assessment and status report on the limber pine *Pinus flexilis* in Canada. Ottawa, ON: Canadian Wildlife Service, Environment Canada. 49 p. https://www.canada.ca/en/environment-climate-change/services/species-risk-public-registry/cosewic-assessments-status-reports/limber-pine-pinus-flexilis-2014.html [Accessed 18 June 2024]

Conlisk, E.; Castanha, C.; Germino, M.J.; [et al.]. 2018. Seed origin and warming constrain lodgepole pine recruitment, slowing the pace of population range shifts. Global Change Biology. 24(1): 197–211. https://doi.org/10.1111/gcb.13840

Corlett, R.T.; Westcott, D.A. 2013. Will plant movements keep up with climate change? Trends in Ecology and Evolution. 28(8): 482–488. https://doi.org/10.1016/j.tree.2013.04.003

Creeden, E.P.; Hicke, J.A.; Buotte, P.C. 2014. Climate, weather, and recent mountain pine beetle outbreaks in the western United States. Forest Ecology and Management. 312: 239–251. https://doi.org/10.1016/j.foreco.2013.09.051

Crookston, N.L. 2009. Plant species and climate profile predictions. Moscow, Idaho: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. https://charcoal2.cnre.vt.edu//climate/species/index.php [Accessed 4 June 2020]

Davis, K.T.; Dobrowski, S.Z.; Higuera, P.E.; [et al.]. 2019. Wildfires and climate change push low-elevation forests across a critical climate threshold for tree regeneration. Proceedings of the National Academy of Sciences of the United States of America. 116(13): 6193–6198. https://doi.org/10.1073/pnas.1815107116

Davis, K.T.; Higuera, P.E.; Dobrowski, S.Z.; [et al.]. 2020. Fire-catalyzed vegetation shifts in ponderosa pine and Douglas-fir forests of the western United States. Environmental Research Letters. 15(10): 1040b8. https://doi.org/10.1088/1748-9326/abb9df

Davis, K.T.; Robles, M.D.; Kemp, K.B; [et al.]. 2023. Reduced fire severity offers near-term buffer to climate-driven declines in conifer resilience across the western United States. Proceedings of the National Academy of Sciences of the United States of America. 120(11): e2208120120. https://doi.org/10.1073/pnas.2208120120

Decker, K.; Fink M. 2015. Ecosystems. In: Decker, K.; Grunau, L.; Handwerk, J.; [et al.], eds. Climate change vulnerability assessment for Colorado Bureau of Land Management. Fort Collins, CO: Colorado Natural Heritage Program, Colorado State University. https://cnhp.colostate.edu/download/documents/2015/CCVA_for_Colorado_BLM_final.pdf [Accessed 18 June 2024]

Derderian, D.P.; Dang, H.; Aplet, G.H. 2016. Bark beetle effects on a seven-century chronosequence of Engelmann spruce and subalpine fir in Colorado, USA. Forest Ecology and Management. 361: 154–162. https://doi.org/10.1016/j.foreco.2015.11.024

DeRose, R.J.; Bentz, B.J.; Long, J.N. 2013. Effect of increasing temperatures on the distribution of spruce beetle in Engelmann spruce forests of the Interior West, USA. Forest Ecology and Management. 308: 198–206. https://doi.org/10.1016/j.foreco.2013.07.061

Devine, W.; Aubry, C.; Bower, A.; [et al.]. 2012. Climate change and forest trees in the Pacific Northwest: A vulnerability assessment and recommended actions for National Forests. Olympia, WA: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region. 102 p.

Dudley, M.M; Burns, K.S.; Jacobi, W.R. 2015. Aspen mortality in the Colorado and southern Wyoming Rocky Mountains: Extent, severity, and causal factors. Forest Ecology and Management. 353: 240–259. https://doi.org/10.1016/j.foreco.2015.06.002

Eager, T.J. 1999. Factors affecting the health of pinyon pine trees (*Pinus edulis*) in the pinyon-juniper woodlands of western Colorado. In: Monsen, S.B.; Stevens, R., eds. Proceedings: Ecology and management of pinyon-juniper communities within the Interior West. Proceedings RMRS-P-9. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 397–399. https://www.fs.usda.gov/research/treesearch/33237

Edwards, D.G.W. 2008. Abies P. Mill. Fir. In: The woody plant seed manual. Agriculture Handbook No 727. Washington, DC: U.S. Department of Agriculture, Forest Service: 149–198. https://www.fs.usda.gov/research/treesearch/32626

Elliott, G.P.; Baker, W.L. 2004. Quaking aspen (*Populus tremuloides* Michx.) at treeline: A century of change in the San Juan Mountains, Colorado, USA. Journal of Biogeography. 31: 733–745. https://www.jstor.org/stable/3554842

Fettig, C.J.; Asaro, C.; Nowak, J.T.; [et al.]. 2022. Trends in bark beetle impacts in North America during a period (2000–2020) of rapid environmental change. Journal of Forestry. 120(6): 693–713. https://doi.org/10.1093/jofore/fvac021

Floyd, M.L.; Clifford, M.; Cobb, N.S. 2009. Relationship of stand characteristics to drought-induced mortality in three Southwestern piñon–juniper woodlands. Ecological Applications. 19(5): 1223–1230. https://doi.org/10.1890/08-1265.1

Floyd, M.L.; Hanna, D.D.; Romme, W.H. 2004. Historical and recent fire regimes in piñon–juniper woodlands on Mesa Verde, Colorado, USA. Forest Ecology and Management. 198(1–3): 269–289. https://doi.org/10.1016/j.foreco.2004.04.006

Floyd, M.L.; Romme, W.H.; Hanna, D.P.; [et al.]. 2017. Historical and modern fire regimes in piñon-juniper woodlands, Dinosaur National Monument, United States. Rangeland Ecology and Management. 70(3): 348–355. https://doi.org/10.1016/j.foreco.2004.04.006

Floyd, M.L.; Romme, W.H.; Hanna, D.D. 2021. Effects of recent wildfires in piñon-juniper woodlands of Mesa Verde National Park, Colorado, USA. Natural Areas Journal. 41(1): 28–38. https://doi.org/10.3375/043.041.0105

Forcella, F. 1981. Ovulate cone production in pinyon: Negative exponential relationship with late summer temperature. Ecology. 62(2): 488–491. https://doi.org/10.2307/1936722

Fornwalt, P.J.; Huckaby, L.S.; Alton, S.K.; [et al.]. 2016. Did the 2002 Hayman Fire, Colorado, USA, burn with uncharacteristic severity? Fire Ecology. 12(3): 117–132. https://doi.org/10.4996/fireecology.1203117

Fornwalt, P.J.; Kaufmann, M.R.; Huckaby, L.S. 2009. Effects of past logging and grazing on understory plant communities in a montane Colorado forest. Plant Ecology. 203(1): 99–109. https://doi.org/10.1007/s11258-008-9513-z

Fornwalt, P.J.; Stevens-Rumann, C.S.; Collins, B.J. 2018. Overstory structure and surface cover dynamics in the decade following the Hayman Fire, Colorado. Forests. 9(3): 152. https://doi.org/10.3390/f9030152

Foster, A.C.; Martin, P.H.; Redmond, M.D. 2020. Soil moisture strongly limits Douglas-fir seedling establishment near its upper elevational limit in the southern Rocky Mountains. Canadian Journal of Forest Research. 50(8): 837–842. https://doi.org/10.1139/cjfr-2019-0296

Foster, A.C.; Shuman, J.K.; Shugart, H.H. 2018. Modeling the interactive effects of spruce beetle infestation and climate on subalpine vegetation. Ecosphere. 9(10): e02437. https://doi.org/10.1002/ecs2.2437

Gaylord, M.L.; Kolb, T.E.; Pockman, W.T.; [et al.]. 2013. Drought predisposes piñon-juniper woodlands to insect attacks and mortality. New Phytologist. 198: 567–578. https://doi.org/10.1111/nph.12174

Geils, B.W.; Hummer, K.E.; Hunt, R.S. 2010. White pines, *Ribes*, and blister rust: A review and synthesis. Forest Pathology. 40: 147–185. https://doi.org/10.1111/j.1439-0329.2010.00654.x

Gergel, D.R.; Nijssen, B.; Abatzoglou, J.T.; [et al.]. 2017. Effects of climate change on snowpack and fire potential in the western USA. Climatic Change. 141: 287–299. https://doi.org/10.1007/s10584-017-1899-y

Gibson, K.; Negrón, J.F. 2009. Fire and bark beetle interactions. In: Hayes, J.L.; Lundquist, J.E., comps. The western bark beetle research group: A unique collaboration with Forest Health Protection—Proceedings of a symposium at the 2007 Society of American Foresters conference. Gen. Tech. Rep. PNW-GTR-784. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 134 p. https://www.fs.usda.gov/research/treesearch/40038

Giunta, A.D.; Jenkins, M.J.; Hebertson, E.G.; [et al.]. 2016. Disturbance agents and their associated effects on the health of interior Douglas-fir forests in the central Rocky Mountains. Forests. 7(4): 80. https://doi.org/10.3390/f7040080

Glatzel, G.; Geils, B. 2009. Mistletoe ecophysiology: Host–parasite interactions. Botany. 87: 10–15. https://doi.org/10.1139/B08-096

Glick, P.; Stein, B.A.; Edelson, N.A. 2011. Scanning the conservation horizon: A guide to climate change vulnerability assessment. Washington, DC: National Wildlife Federation. 168 p. https://www.fs.usda.gov/research/treesearch/37406

Goeking, S.A.; Windmuller-Campione, M.A. 2021. Comparative species assessments of five-needle pines throughout the western United States. Forest Ecology and Management. 496: 119438. https://doi.org/10.1016/j.foreco.2021.119438

Goke, A.; Martin, P.H. 2022. Poor acclimation to experimental field drought in subalpine forest tree seedlings. AoB Plants. 14(1): plab077. https://doi.org/10.1093/aobpla/plab077

Government of Canada. 2012. Order amending Schedule 1 to the Species at Risk Act. P.C. 2012-837. SOR/2012–113. Canada Gazette Part II. 146(14): 1514-1562. https://gazette.gc.ca/rp-pr/p2/2012/2012-07-04/html/sor-dors133-eng.html [Accessed 18 June 2024]

Graham, R.T. 2003. Hayman fire case study. Gen. Tech. Rep. RMRS-GTR-114. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 396 p. https://doi.org/10.2737/RMRS-GTR-114

Gray, L.K.; Hamann, A. 2013. Tracking suitable habitat for tree populations under climate change in western North America. Climatic Change. 117(1): 289–303. https://doi.org/10.1007/s10584-012-0548-8

Gray, C.A.; Runyon J.B.; Jenkins M.J.; [et al.]. 2015. Mountain pine beetles use volatile cues to locate host limber pine and avoid non-host Great Basin bristlecone pine. PloS One. 10: e0135752. https://doi.org/10.1371/journal.pone.0135752

Guiterman, C.H.; Gregg, R.M.; Marshall, L.A.; [et al.]. 2022. Vegetation type conversion in the U.S. Southwest: Frontline observations and management responses. Fire Ecology. 18: 6. https://doi.org/10.1186/s42408-022-00131-w

Guz, J.; Gill, N.S.; Kulakowski, D. 2021. Long-term empirical evidence shows post-disturbance climate controls post-fire regeneration. Journal of Ecology. 109(12): 4007–4024. https://doi.org/10.1111/1365-2745.13771

Hadley, K.S.; Veblen, T.T. 1993. Stand response to western spruce budworm and Douglas-fir bark beetle outbreaks, Colorado Front Range. Canadian Journal of Forest Research. 23(3): 479–491. https://doi.org/10.1139/x93-066

Hansen, A.J.; Phillips, L.B. 2015. Which tree species and biome types are most vulnerable to climate change in the U.S. northern Rocky Mountains? Forest Ecology and Management. 338: 68–83. https://doi.org/10.1016/j.foreco.2014.11.008

Hart, S.J.; Veblen, T.T.; Eisenhart, K.S.; [et al.]. 2014a. Drought induces spruce beetle (*Dendroctonus rufipennis*) outbreaks across northwestern Colorado. Ecology. 95(4): 930–939. https://doi.org/10.1890/13-0230.1

Hart, S.J.; Veblen, T.T.; Kulakowski, D. 2014b. Do tree and stand-level attributes determine susceptibility of spruce-fir forests to spruce beetle outbreaks in the early 21st century? Forest Ecology and Management. 318: 44–53. https://doi.org/10.1016/j.foreco.2013.12.035

Hart, S.J.; Veblen, T.T.; Schneider, D.; [et al.]. 2017. Summer and winter drought drive the initiation and spread of spruce beetle outbreak. Ecology. 98(10): 2698–2707. https://doi.org/10.1002/ecy.1963

Harvey, B.J.; Andrus, R.A.; Battaglia, M.A.; [et al.]. 2021. Droughty times in mesic places: Factors associated with forest mortality vary by scale in a temperate subalpine region. Ecosphere. 12(1): e03318. https://doi.org/10.1002/ecs2.3318

Heidari, H.; Arabi, M.; Warziniack, T. 2021. Shifts in hydroclimatology of U.S. megaregions in response to climate change. Environmental Research Communications. 3(6): 065002. https://doi.org/10.1088/2515-7620/ac0617

Hennon, P.E.; Frankel, S.J.; Woods, A.J.; [et al.]. 2020. A framework to evaluate climate effects on forest tree diseases. Forest Pathology. 50: e12649. https://doi.org/10.1111/efp.12649

Hermann, R.K.; Lavender, D.P. 1990. *Pseudotsuga menziesii* (Mirb.) Franco. Douglas-fir. In: Silvics of North America, Volume 1: Conifers. Agriculture Handbook 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 527–540. https://www.fs.usda.gov/research/treesearch/1547

Hessl, A.E.; Baker, W.L. 1997. Spruce and fir regeneration and climate in the forest-tundra ecotone of Rocky Mountain National Park, Colorado, USA. Arctic and Alpine Research. 29(2): 173–183. https://doi.org/10.2307/1552044

Hicke, J.A.; Logan, J.A.; Powell, J.; [et al.]. 2006. Changing temperatures influence suitability for modeled mountain pine beetle (*Dendroctonus ponderosae*) outbreaks in the western United States. Journal of Geophysical Research. 111: G02019. https://doi.org/10.1029/2005JG000101

Hicke, J.A.; Xu, B.; Meddens, A.J.; [et al.]. 2020. Characterizing recent bark beetle-caused tree mortality in the western United States from aerial surveys. Forest Ecology and Management. 475: 118402. https://doi.org/10.1016/j.foreco.2020.118402

Higuera, P.E.; Briles, C.E.; Whitlock, C. 2014. Fire-regime complacency and sensitivity to centennial- through millennial-scale climate change in Rocky Mountain subalpine forests, Colorado, USA. Journal of Ecology. 102(6): 1429-1441. https://doi.org/10.1111/1365-2745.12296

Higuera, P.E.; Shuman, B.N.; Wolf, K.D. 2021. Rocky Mountain subalpine forests now burning more than any time in recent millennia. Proceedings of the National Academy of Sciences of the United States of America. 118(25): e2103135118. https://doi.org/10.1073/pnas.2103135118

Hill, E.; Ex, S.; Aldridge, C.; [et al.]. 2019. Overstory density, short growing seasons, and moisture limit Engelmann spruce establishment over time in high-elevation managed stands. Canadian Journal of Forest Research. 49(1): 64–75. https://doi.org/10.1139/cjfr-2018-0268

Hinds, T.E.; Wengert, E.M. 1977. Growth and decay losses in Colorado aspen. Research Paper RM-193. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. https://www.fs.usda.gov/research/treesearch/30450

Holtmeier, F.K.; Broll, G. 2017. Layering in the Rocky Mountain treeline ecotone: Clonal conifer groups' distribution, structure, and functional role. Trees. 31(3): 953–965. https://doi.org/10.1007/s00468-017-1520-z

Howell, B.E.; Burns, K.S.; Kearns, H.S.J. 2006. Biological evaluation of a model for predicting presence of white pine blister rust in Colorado based on climatic variables and susceptible white pine species distribution. Biological Evaluation. R2-06-04. Lakewood, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Region, Renewable Resources. 15 p.

Huckaby, L.S.; Moir, W.H. 1998. Forest communities at Fraser Experimental Forest, Colorado. The Southwestern Naturalist. 43(2): 204–218. https://www.jstor.org/stable/30055357

Huckaby, L.S.; Kaufmann, M.R.; Fornwalt, P.J.; [et al.]. 2003. Identification and ecology of old ponderosa pine trees in the Colorado Front Range. Gen. Tech. Rep. RMRS-GTR-110. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 47 p. https://doi.org/10.2737/RMRS-GTR-110

Huckaby, L.S.; Kaufmann, M.R.; Stoker, J.M.; [et al.]. 2001. Landscape patterns of montane forest age structure relative to fire history at Cheesman Lake in the Colorado Front Range. In: Vance, R.K.; Edminster, C.B.; Covington, W.W.; [et al.], comps. Ponderosa pine ecosystems restoration and conservation: Steps toward stewardship. Proceedings RMRS-P-22. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 19–27. https://www.fs.usda.gov/research/treesearch/46667

Huffman, D.W.; Crouse, J.E.; Chancellor, W.W.; [et al.]. 2012. Influence of time since fire on pinyon–juniper woodland structure. Forest Ecology and Management. 274: 29–37. https://doi.org/10.1016/j.foreco.2012.02.014

Huffman, D.W.; Fule, P.Z.; Pearson, K.M.; [et al.]. 2008. Fire history of pinyon–juniper woodlands at upper ecotones with ponderosa pine forests in Arizona and New Mexico. Canadian Journal of Forest Research. 38(8): 2097–2108. https://doi.org/10.1139/X08-053

Intergovernmental Panel on Climate Change [IPCC]. 2001. Climate change 2001: Impacts, adaptation and vulnerability. Contribution of working group II to the third assessment report of the Intergovernmental Panel on Climate Change. Cambridge, UK: Cambridge University Press.

Intergovernmental Panel on Climate Change [IPCC]. 2007. Parry, M.L.; Canziani, O.F.; Palutikof, J.P.; [et al.], eds. Climate change 2007: Impacts, adaptation and vulnerability. Contribution of working group II to the fourth assessment report of the intergovernmental panel on climate change. Cambridge, UK: Cambridge University Press. 976 p.

Jacobs, D.F. 2011. Reforestation of a salvage-logged high-elevation clearcut: Engelmann spruce seedling response to tree shelters after 11 growing seasons. Western Journal of Applied Forestry. 26(2): 53–56. https://doi.org/10.1093/wjaf/26.2.53

Janowiak, M.K.; D'Amato, A.W.; Swanston, C.W.; [et al.]. 2018. New England and northern New York forest ecosystem vulnerability assessment and synthesis: A report from the New England Climate Change Response Framework project. Gen. Tech. Rep. NRS-173. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 234 p. https://doi.org/10.2737/nrs-gtr-173

Jarvis, D.S.; Kulakowski, D. 2015. Long-term history and synchrony of mountain pine beetle outbreaks in lodgepole pine forests. Journal of Biogeography. 42(6): 1029–1039. https://doi.org/10.1111/jbi.12489

Johnson, E.A; Fryer, G.I. 1996. Why Engelmann spruce does not have a persistent seed bank. Canadian Journal of Forest Research. 26(5): 872–878. https://doi.org/10.1139/x26-095

Johnson, D.; Jacobi, W.R. 2000. First report of white pine blister rust in Colorado. Plant Disease. 84(5): 595–595. https://doi.org/10.1094/PDIS.2000.84.5.595D

Johnston, B.C. 1987. Plant associations of region two: Potential plant communities of Wyoming, South Dakota, Nebraska, Colorado, and Kansas. R2-ECOL-87-2. Lakewood, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Region. 429 p.

Jump, A.S.; Marchant, R; Peñuelas, J. 2009. Environmental change and the option value of genetic diversity. Trends in Plant Science. 14(1): 51–58. https://doi.org/10.1016/j.tplants.2008.10.002

Jump, A.S.; Peñuelas, J. 2005. Running to stand still: Adaptation and the response of plants to rapid climate change. Ecology Letters. 8(9): 1010–1020. https://doi.org/10.1111/j.1461-0248.2005.00796.x

Kane, J.M.; Kolb, T.E. 2014. Short- and long-term growth characteristics associated with tree mortality in southwestern mixed-conifer forests. Canadian Journal of Forestry Research. 44: 1227–1235. https://doi.org/10.1139/cjfr-2014-0186

Kane, J.M.; Kolb, T.E.; McMillin, J.D. 2014. Stand-scale tree mortality factors differ by site and species following drought in southwestern mixed conifer forests. Forest Ecology and Management. 330: 171–182. https://doi.org/10.1016/j.foreco.2014.06.042

Kashian, D.M.; Romme, W.H.; Regan, C.M. 2007. Reconciling divergent interpretations of quaking aspen decline on the northern Colorado Front Range. Ecological Applications. 17: 1296–1311. https://doi.org/10.1890/06-1431.1

Kaufmann, M.R.; Eckard, A.N. 1977. Water potential and temperature effects on germination of Engelmann spruce and lodgepole pine seeds. Forest Science. 23(1): 27–33. https://doi.org/10.1093/forestscience/23.1.27

Kaufmann, M.R.; Fornwalt, P.J.; Huckaby, L.S.; [et al.]. 2001. Cheesman Lake: A historical ponderosa pine landscape guiding restoration in the South Platte watershed of the Colorado Front Range. In: Vance, R.K.; Edminster, C.B.; Covington, W.W.; [et al.], comps. Ponderosa pine ecosystems restoration and conservation: Steps toward stewardship. Proceedings RMRS-P-22. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 9–18. https://www.fs.usda.gov/research/treesearch/46666

Kaufmann, M.R.; Regan, C.M.; Brown, P.M. 2000. Heterogeneity in ponderosa pine/Douglas-fir forests: Age and size structure in unlogged and logged landscapes of central Colorado. Canadian Journal of Forest Research. 30(5): 698–711. https://doi.org/10.1139/x99-255

Keane, R.E.; Mahalovich, M.F.; Bollenbacher, B.L.; [et al.]. 2018. Effects of climate change on forest vegetation in the Northern Rockies Region. In: Halofsky, J.E.; Peterson, D.L.; Dante-Wood, S.K.; [et al.], eds. Climate change vulnerability and adaptation in the northern Rocky Mountains. Gen. Tech. Rep. RMRS-GTR-374. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. Chapter 6: 128–173. https://www.fs.usda.gov/research/treesearch/55991

Keane, R.E.; Schoettle, A.W.; Tomback, D.F. 2021. Effective actions for managing resilient high elevation five-needle white pine forests in western North America at multiple scales under changing climates. Forest Ecology and Management. 505: 119939. https://doi.org/10.1016/j.foreco.2021.119939

Kearns, H.S.; Jacobi, W.R. 2007. The distribution and incidence of white pine blister rust in central and southeastern Wyoming and northern Colorado. Canadian Journal of Forest Research. 37: 462–472. https://doi.org/10.1139/x06-231

Kearns, H.S.; Jacobi, W.; Reich, R.; [et al.]. 2014. Risk of white pine blister rust to limber pine in Colorado and Wyoming, USA. Forest Pathology. 44: 21–38. https://doi.org/10.1111/efp.12065

Keith, R.P.; Veblen, T.T.; Schoennagel, T.L.; [et al.]. 2010. Understory vegetation indicates historic fire regimes in ponderosa pine-dominated ecosystems in the Colorado Front Range. Journal of Vegetation Science. 21(3): 488–499. https://doi.org/10.1111/j.1654-1103.2009.01156.x

Kennard, D.K; Moore, A.J. 2013. Fire history, woodland structure, and mortality in a piñon-juniper woodland in the Colorado National Monument. Natural Areas Journal. 33(3): 296–306. https://doi.org/10.3375/043.033.0308

Kim, M.S.; Brunsfeld, S.J.; McDonald, G.I.; [et al.]. 2003. Effect of white pine blister rust (*Cronartium ribicola*) and rust-resistance breeding on genetic variation in western white pine (*Pinus monticola*). Theoretical and Applied Genetics. 106: 1004–1010. https://doi.org/10.1007/s00122-002-1141-8

Klutsch, J.G.; Negrón, J.F.; Costello, S.L.; [et al.]. 2009. Stand characteristics and downed woody debris accumulations associated with a mountain pine beetle (*Dendroctonus ponderosae* Hopkins) outbreak in Colorado. Forest Ecology and Management. 258(5): 641–649. https://doi.org/10.1016/j.foreco.2009.04.034

Knapp, A.K.; Smith, W.K. 1981. Water relations and succession in subalpine conifers in southeastern Wyoming. Botanical Gazette. 142(4): 502–511. https://www.jstor.org/stable/247432

Knapp, A.K.; Smith, W.K. 1982. Factors influencing understory seedling establishment of Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*) in southeast Wyoming. Canadian Journal of Botany. 60(12): 2753–2761. https://doi.org/10.1139/b82-337

Kolb, T.E.; Fettig, C.J.; Ayres, M.P.; [et al.]. 2016. Observed and anticipated impacts of drought on forest insects and diseases in the United States. Forest Ecology and Management. 380: 321–334. https://doi.org/10.1016/j.foreco.2016.04.051

Krannitz, P.G.; Duralia, T.E. 2004. Cone and seed production in *Pinus ponderosa*: A review. Western North American Naturalist. 64: 208–218. https://scholarsarchive.byu.edu/wnan/vol64/iss2/8 [Accessed 21 June 2024]

Kreider, M.R.; Yocom, L.L. 2021. Low-density aspen seedling establishment is widespread following recent wildfires in the western United States. Ecology. 102(10): e03436. https://doi.org/10.1002/ecy.3436

Krist, F.J.; Ellenwood, J.R.; Woods, M.E.; [et al.]. 2014. 2013–2027 National insect and disease forest risk assessment. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Forest Health Technology Enterprise Team. 199 p. https://www.fs.usda.gov/foresthealth/technology/pdfs/2012_RiskMap_Report_web.pdf [Accessed 21 June 2024]

Kueppers, L.M.; Conlisk, E.; Castanha, C.; [et al.]. 2017. Warming and provenance limit tree recruitment across and beyond the elevation range of subalpine forest. Global Change Biology. 23(6): 2383–2395. https://doi.org/10.1111/gcb.13561

Kulakowski, D.; Jarvis, D. 2011. The influence of mountain pine beetle outbreaks and drought on severe wildfires in northwestern Colorado and southern Wyoming: A look at the past century. Forest Ecology and Management. 262(9): 1686–1696. https://doi.org/10.1016/j.foreco.2011.07.016

Kulakowski, D.; Jarvis, D. 2013. Low-severity fires increase susceptibility of lodgepole pine to mountain pine beetle outbreaks in Colorado. Forest Ecology and Management. 289: 544–550. https://doi.org/10.1016/j.foreco.2012.10.020

Lalande, B.M.; Hughes, K.; Jacobi, W.R.; [et al.]. 2020. Subalpine fir mortality in Colorado is associated with stand density, warming climates and interactions among fungal diseases and the western balsam bark beetle. Forest Ecology and Management. 466: 118133. https://doi.org/10.1016/j.foreco.2020.118133

Landhäusser, S.M.; Pinno, B.D.; Mock, K.E. 2019. Tamm review: Seedling-based ecology, management, and restoration in aspen (*Populus tremuloides*). Forest Ecology and Management. 432: 231–245. https://doi.org/10.1016/j.foreco.2018.09.024

Latif, Q.S.; Truex, R.L.; Sparks, R.A.; [et al.]. 2020. Dry conifer forest restoration benefits Colorado Front Range avian communities. Ecological Applications. 30(6): e02142. https://doi.org/10.1002/eap.2142

Lawson, T.; Blatt, M.R. 2014. Stomatal size, speed, and responsiveness impact on photosynthesis and water use efficiency. Plant Physiology. 164(4): 1556–1570. https://doi.org/10.1104/pp.114.237107

League, K.; Veblen, T. 2006. Climatic variability and episodic *Pinus ponderosa* establishment along the forest-grassland ecotones of Colorado. Forest Ecology and Management. 228(1–3): 98–107. https://doi.org/10.1016/j.foreco.2006.02.030

Li, P.; Adams, W.T. 1989. Range-wide patterns of allozyme variation in Douglas-fir (*Pseudotsuga menziesii*). Canadian Journal of Forest Research. 19(2): 149–161. https://doi.org/10.1139/x89-022

Little, E.L. 1938. The earliest stages of pinyon cones. Research Note 46. Tucson, AZ: U.S. Department of Agriculture, Forest Service, Southwestern Forest and Range Experiment Station.

Little Jr., E. 1971. Atlas of United States trees, Volume 1: Conifers and important hardwoods. Miscellaneous Publication 1146. Washington, DC: U.S. Department of Agriculture, Forest Service. 313 p. https://www.biodiversitylibrary.org/item/22546

Logan, J.A.; Macfarlane, W.W.; Willcox, L. 2010. Whitebark pine vulnerability to climate-driven mountain pine beetle disturbance in the greater Yellowstone ecosystem. Ecological Applications. 20: 895–902. https://doi.org/10.1890/09-0655.1

Long, J.N.; Mock, K. 2012. Changing perspectives on regeneration ecology and genetic diversity in western quaking aspen: Implications for silviculture. Canadian Journal of Forest Research. 42(12): 2011–2021. https://doi.org/10.1139/x2012-143

Lotan, J.E.; Critchfield, W.B. 1990. *Pinus contorta* Dougl. ex. Loud. Lodgepole pine. In: Silvics of North America, Volume 1: Conifers. Agriculture Handbook 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 302–315. https://www.fs.usda.gov/research/treesearch/1547

Macalady, A.K.; Bugmann, H. 2014. Growth-mortality relationships in piñon pine (*Pinus edulis*) during severe droughts of the past century: Shifting processes in space and time. PLoS One. 9(5): e92770. https://doi.org/10.1371/journal.pone.0092770

Maini, J.S.; Cayford, J.H., eds. 1968. Growth and utilization of poplars in Canada. Ottawa, ON: Department of Forestry and Rural Development, Forestry Branch, Departmental Publication. 1205: 46–69.

Malone, S.L.; Fornwalt, P.J.; Battaglia, M.A. 2018a. Mixed-severity fire fosters heterogeneous spatial patterns of conifer regeneration in a dry conifer forest. Forests. 9(1): 45. https://doi.org/10.3390/f9010045

Malone, S.L.; Schoettle, A.W.; Coop, J.D. 2018b. The future of subalpine forests in the southern Rocky Mountains: Trajectories for *Pinus aristata* genetic lineages. PLoS one. 13: e0193481. https://doi.org/10.1371/journal.pone.0193481

Marchetti, S.B.; Worrall, J.J.; Thomas, E. 2011. Secondary insects and diseases contribute to sudden aspen decline in southwestern Colorado, USA. Canadian Journal of Forest Research. 41(12): 2315–2325. https://doi.org/10.1139/x11-106

Marshall, L.A.E.; Fornwalt, P.J.; Stevens-Rumann, C.S.; [et al.]. 2023. North-facing aspects, shade objects, and microtopographic depressions promote the survival and growth of tree seedlings planted after wildfire. Fire Ecology. 19: 26. https://doi.org/10.1186/s42408-023-00181-8

Mastrandrea, M.D.; Field, C.B.; Stocker, T.F.; [et al.]. 2010. Guidance note for lead authors of the IPCC Fifth Assessment Report on consistent treatment of uncertainties. Intergovernmental Panel on Climate Change (IPCC). https://www.ipcc.ch/site/assets/uploads/2018/05/uncertainty-guidance-note.pdf [Accessed 20 June 2024]

McCambridge, W.F.; Hawksworth, F.G.; Edminster, C.B.; [et al.]. 1982. Ponderosa pine mortality resulting from a mountain pine beetle outbreak. Research Paper RM-235. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. https://doi.org/10.2737/RM-RP-235

McCullough, I.M.; Davis, F.W.; Williams, A.P. 2017. A range of possibilities: Assessing geographic variation in climate sensitivity of ponderosa pine using tree rings. Forest Ecology and Management. 402: 223–233. https://doi.org/10.1016/j.foreco.2017.07.025

McDonald, P.M. 1980. Seed dissemination in small clearcuttings in north-central California. Research Paper PSW-RP-150. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station. https://doi.org/10.2737/PSW-RP-150

McDonald, G.I.; Richardson, B.A.; Zambino, P.J.; [et al.]. 2006. *Pedicularis* and *Castilleja* are natural hosts of *Cronartium ribicola* in North America: A first report. Forest Pathology. 36: 73–82. https://doi.org/10.1111/j.1439-0329.2006.00432.x

McDowell, N.G.; Grossiord, C.; Adams, H.D.; [et al.]. 2019. Mechanisms of a coniferous woodland persistence under drought and heat. Environmental Research Letters. 14(4): 045014. https://doi.org/10.1088/1748-9326/ab0921

Meddens, A.J.; Hicke, J.A. 2014. Spatial and temporal patterns of Landsat-based detection of tree mortality caused by a mountain pine beetle outbreak in Colorado, USA. Forest Ecology and Management. 322: 78–88. https://doi.org/10.1016/j.foreco.2014.02.037

Meeuwig, R.O.; Bassett, R.L. 1983. Pinyon-juniper. In: Burns, R.M., tech. comp. Silvicultural systems for the major forest types of the United States. Agricultural Handbook 455. Washington, DC: U.S. Department of Agriculture, Forest Service: 84–86. https://www.fs.usda.gov/research/treesearch/32974

Menon, M.; Bagley, J.C.; Friedline, C.J.; [et al.]. 2018. The role of hybridization during ecological divergence of southwestern white pine (*Pinus strobiformis*) and limber pine (*P. flexilis*). Molecular Ecology. 27: 1245–1260. https://doi.org/10.1111/mec.14505

Millar, C.I.; Westfall, R.D.; Delany, D.L.; [et al.]. 2015. Recruitment patterns and growth of high-elevation pines in response to climatic variability (1883–2013), in the western Great Basin, USA. Canadian Journal of Forestry Research. 45: 1299–1312. https://doi.org/10.1139/cjfr-2015-0025

Minott, J.A.; Kolb, T.E. 2020. Regeneration patterns reveal contraction of ponderosa forests and little upward migration of pinyon-juniper woodlands. Forest Ecology and Management. 458: 117640. https://doi.org/10.1016/j.foreco.2019.117640

Mirov, N.T. 1967. The genus *Pinus*. New York: Ronald Press. 602 p.

Mitton, J.B.; Duran, K.L. 2004. Genetic variation in piñon pine, *Pinus edulis*, associated with summer precipitation. Molecular Ecology. 13(5): 1259–1264. https://doi.org/10.1111/j.1365-294X.2004.02122.x

Mitton, J.B.; Ferrenberg, S.M. 2012. Mountain pine beetle develops an unprecedented summer generation in response to climate warming. The American Naturalist. 179(5): E163–E171. https://doi.org/10.1086/665007

Mitton, J.B.; Grant, M.C. 1996. Genetic variation and the natural history of quaking aspen. BioScience. 46(1): 25–31. https://doi.org/10.2307/1312652

Mitton, J.B.; Grant, M.C.; Yoshino, A.M. 1998. Variation in allozymes and stomatal size in pinyon (*Pinus edulis Pinaceae*), associated with soil moisture. American Journal of Botany. 85(9): 1262–1265. https://doi.org/10.2307/2446636

Mock, K.E.; Rowe, C.A.; Hooten, M.B.; [et al.]. 2008. Clonal dynamics in western North American aspen (*Populus tremuloides*). Molecular Ecology. 17: 4827–4844. https://doi.org/10.1111/j.1365-294X.2008.03963.x

Moir, W.H.; Ludwig, J.A. 1979. A classification of spruce-fir and mixed conifer habitat types of Arizona and New Mexico. Research Paper RM-207. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 47 p.

Monahan, W.B.; Cook, T.; Melton, F.; [et al.]. 2013. Forecasting distributional responses of limber pine to climate change at management-relevant scales in Rocky Mountain National Park. PloS One. 8(12): e83163. https://doi.org/10.1371/journal.pone.0083163

Monitoring Trends in Burn Severity [MTBS]. 2022. Burned areas boundaries dataset (1984–2020). https://www.mtbs.gov [Accessed 9 June 2022]

Mooney, K.A.; Linhart, Y.B.; Snyder, M.A. 2011. Masting in ponderosa pine: Comparisons of pollen and seed over space and time. Oecologia. 165: 651–661. https://doi.org/10.1007/s00442-010-1742-x

Moyes, A.B.; Castanha, C.; Germino, M.J.; [et al.]. 2013. Warming and the dependence of limber pine (*Pinus flexilis*) establishment on summer soil moisture within and above its current elevation range. Oecologia. 171: 271–282. https://doi.org/10.1007/s00442-012-2410-0

Moyes, A.B.; Germino, M.J.; Kueppers, L.M. 2015. Moisture rivals temperature in limiting photosynthesis by trees establishing beyond their cold-edge range limit under ambient and warmed conditions. New Phytologist. 207: 1005–1014. https://doi.org/10.1111/nph.13422

Natural Resources Conservation Service [NRCS]. 2020. The PLANTS Database. Greensboro, NC: U.S. Department of Agriculture, Natural Resources Conservation Service, National Plant Data Team. https://plants.usda.gov [Accessed 23 August 2020]

Negrón, J.F. 1998. Probability of infestation and extent of mortality associated with the Douglas-fir beetle in the Colorado Front Range. Forest Ecology and Management. 107(1–3): 71–85. https://doi.org/10.1016/S0378-1127(97)00319-8

Negrón, J.F. 2024. Chapter 8: Insects and diseases and climate change in Colorado Front Range forests. In: Hanberry, B.B.; Foster, A., eds. Colorado Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-GTR-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 293-322. https://doi.org/10.2737/RMRS-GTR-438

Negrón, J.F.; Cain, B. 2019. Mountain pine beetle in Colorado: A story of changing forests. Journal of Forestry. 117(2): 144–151. https://doi.org/10.1093/jofore/fvy032

Negrón, J.F.; Huckaby, L. 2020. Reconstructing historical outbreaks of mountain pine beetle in lodgepole pine forests in the Colorado Front Range. Forest Ecology and Management. 473: 118270. https://doi.org/10.1016/j.foreco.2020.118270

Negrón, J.F.; Popp, J.B. 2004. Probability of ponderosa pine infestation by mountain pine beetle in the Colorado Front Range. Forest Ecology and Management. 191: 17–27. https://doi.org/10.1016/j.foreco.2003.10.026

Negrón, J.F.; Popp, J.B. 2009. The flight periodicity, attack patterns, and life history of *Dryocoetes confusus* Swaine (Coleoptera: Curculionidae: Scolytinae), the western balsam bark beetle, in north central Colorado. Western North American Naturalist. 69(4): 447–458. https://doi.org/10.3398/064.069.0404

Negrón, J.F.; Wilson, J.L. 2003. Attributes associated with probability of infestation by the piñon ips, *Ips confusus* (Coleoptera: Scolytidae), in piñon pine, *Pinus edulis*. Western North American Naturalist. 63(4): 440–451. https://scholarsarchive.byu.edu/wnan/vol63/iss4/4

Negrón, J.F.; Lynch, A.M.; Schaupp Jr., W.C.; [et al.]. 2014. Douglas-fir tussock moth and Douglas-fir beetle-caused mortality in a ponderosa pine/Douglas-fir forest in the Colorado Front Range, USA. Forests. 5(12): 3131–3146. https://doi.org/10.3390/f5123131

Negrón, J.F.; Schaupp Jr., W.C.; Pederson, L. 2011. Flight periodicity of the Douglas-fir beetle, *Dendroctonus pseudotsugae* Hopkins (Coleoptera: Curculionidae: Scolytinae) in Colorado, USA. The Coleopterists Bulletin. 65(2): 182–184. https://doi.org/10.1649/072.065.0219

Nicotra, A.B.; Atkin, O.K.; Bonser, S.P.; [et al.]. 2010. Plant phenotypic plasticity in a changing climate. Trends in Plant Science. 15: 684–692. https://doi.org/10.1016/j.tplants.2010.09.008

Nigro, K.M.; Rocca, M.E.; Battaglia, M.A.; [et al.]. 2022. Wildfire catalyzes upward range expansion of trembling aspen in southern Rocky Mountain beetle-killed forests. Journal of Biogeography. 49(1): 201–214. https://doi.org/10.1111/jbi.14302

Niinemets, Ü.; Valladares, F. 2006. Tolerance to shade, drought, and waterlogging of temperate northern hemisphere trees and shrubs. Ecological Monographs. 76(4): 521–547. https://doi.org/10.1890/0012-9615(2006)076[0521:TTSDAW]2.0.CO;2

Noble, D.L. 1973. Engelmann spruce seedling roots reach depth of 3 to 4 inches their first season. Research Note RM-241. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. https://www.fs.usda.gov/research/treesearch/65255

Noble, D.L.; Alexander, R.R. 1977. Environmental factors affecting natural regeneration Engelmann spruce in the central Rocky Mountains. Forest Science. 23(4): 420–429. https://doi.org/10.1093/forestscience/23.4.420

Noble, D.L.; Ronco Jr, F. 1978. Seedfall and establishment of Engelmann spruce and subalpine fir in clearcut openings in Colorado. Research Paper RM-RP-53. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 8 p.

Oliver, W.W.; Ryker, R.A. 1990. *Pinus ponderosa* Dougl. ex Laws. Ponderosa pine. In: Silvics of North America, Volume 1: Conifers. Agriculture Handbook 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 413–424. https://www.fs.usda.gov/research/treesearch/1547

Owens, J.N.; Blake, M.D. 1985. Forest tree seed production. A review of the literature and recommendations for future research. Canadian Forest Service. No. PI-X-53.

Parks, S.A.; Holsinger, L.M.; Panunto, M.H.; [et al.]. 2018. High-severity fire: Evaluating its key drivers and mapping its probability across western U.S. forests. Environmental Research Letters. 13(4): 044037. https://doi.org/10.1088/1748-9326/aab791

Parmenter, R.R.; Zlotin, R.I.; Moore, D.I.; [et al.]. 2018. Environmental and endogenous drivers of tree mast production and synchrony in piñon–juniper–oak woodlands of New Mexico. Ecosphere. 9(8): e02360. https://doi.org/10.1002/ecs2.2360

Pataki, D.E.; Oren, R.; Smith, W.K. 2000. Sap flux of co-occurring species in a western subalpine forest during seasonal soil drought. Ecology. 81: 2557–2566. https://doi.org/10.1890/0012-9658(2000)081[2557:SFOCOS]2.0.CO;2

Peet, R.K. 1981. Forest vegetation of the Colorado Front Range: Composition and dynamics. Vegetatio. 45: 3–75. https://www.jstor.org/stable/20037022

Peltier, D.M.; Ogle, K. 2019. Legacies of more frequent drought in ponderosa pine across the western United States. Global Change Biology. 25(11): 3803–3816. https://doi.org/10.1111/gcb.14720

Pelz, K.A.; Rhoades, C.C.; Hubbard, R.M.; [et al.]. 2018. Severity of overstory mortality influences conifer recruitment and growth in mountain pine beetle-affected forests. Forests. 9: 536. https://doi.org/10.3390/f9090536

Perala, D.A. 1990. *Populus tremuloides* Michx. quaking aspen. In: Silvics of North America, Volume 2: Hardwoods. Agriculture Handbook 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 555–569. https://www.fs.usda.gov/research/treesearch/1548

Peterson, D.L.; Millar, C.I.; Joyce, L.A.; [et al.]. 2011. Responding to climate change in national forests: A guidebook for developing adaptation options. Gen. Tech. Rep. PNW-GTR-855. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 109 p. https://doi.org/10.2737/PNW-GTR-855

Petrie, M.D.; Wildeman, A.M.; Bradford, J.B.; [et al.]. 2016. A review of precipitation and temperature control on seedling emergence and establishment for ponderosa and lodgepole pine forest regeneration. Forest Ecology and Management. 361: 328–338. https://doi.org/10.1016/j.foreco.2015.11.028

Picotte, J.J.; Peterson, B.; Meier, G.; [et al.]. 2016. 1984–2010 trends in fire burn severity and area for the conterminous U.S. International Journal of Wildland Fire. 25: 413–420. https://doi.org/10.1071/WF15039

Pike and San Isabel National Forests and Comanche and Cimarron National Grasslands [PSICC]. 2019. Field sampled vegetation spatial database. U.S. Department of Agriculture, Forest Service, Pike and San Isabel National Forests and Comanche and Cimarron National Grasslands.

Potter, K.M.; Hipkins, V.D.; Mahalovich, M.F.; [et al.]. 2015. Nuclear genetic variation across the range of ponderosa pine (*Pinus ponderosa*): Phylogeographic, taxonomic and conservation implications. Tree Genetics and Genomes. 11(3): 1–23. https://doi.org/10.1007/s11295-015-0865-y

PRISM Climate Group [PRISM]. 2020. 30-Year normals (1981–2010), Version M2. Corvallis, OR: Oregon State University. http://prism.oregonstate.edu [Accessed 20 August 2020]

Rangwala, I. 2024. Chapter 3: Future climate scenarios for Colorado's Front Range forests. In: Hanberry, B.B.; Foster, A., eds. Colorado Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-GTR-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 39-56. https://doi.org/10.2737/RMRS-GTR-438

Redmond, M.D.; Barger, N.N. 2013. Tree regeneration following drought-and insect-induced mortality in piñon–juniper woodlands. New Phytologist. 200(2): 402–412. https://doi.org/10.1111/nph.12366

Redmond, M.D.; Kelsey, K.C. 2018. Topography and overstory mortality interact to control tree regeneration in spruce-fir forests of the southern Rocky Mountains. Forest Ecology and Management. 427: 106–113. https://doi.org/10.1016/j.foreco.2018.05.057

Redmond, M.D.; Cobb, N.S.; Clifford, M.J.; [et al.]. 2015. Woodland recovery following drought-induced tree mortality across an environmental stress gradient. Global Change Biology. 21(10): 3685–3695. https://doi.org/10.1111/gcb.12976

Redmond, M.D.; Cobb, N.S.; Miller, M.E.; [et al.]. 2013. Long-term effects of chaining treatments on vegetation structure in piñon–juniper woodlands of the Colorado Plateau. Forest Ecology and Management. 305: 120–128. https://doi.org/10.1016/j.foreco.2013.05.020

Redmond, M.D.; Forcella, F.; Barger, N.N. 2012. Declines in pinyon pine cone production associated with regional warming. Ecosphere. 3(12): 1–14. https://doi.org/10.1890/ES12-00306.1

Redmond, M.D.; Kelsey, K.C.; Urza, A.K.; [et al.]. 2017. Interacting effects of climate and landscape physiography on piñon pine growth using an individual-based approach. Ecosphere. 8(3): e01681. https://doi.org/10.1002/ecs2.1681

Redmond, M.D.; Weisberg, P.J.; Cobb, N.S.; [et al.]. 2018. Woodland resilience to regional drought: Dominant controls on tree regeneration following overstorey mortality. Journal of Ecology. 106(2): 625–639. https://doi.org/10.1111/1365-2745.12880

Régnière, J.; Nealis, V.G. 2019. Influence of temperature on historic and future population fitness of the western spruce budworm, *Choristoneura occidentalis*. International Journal of Pest Management. 65(3): 228–243. https://doi.org/10.1080/09670874.2018.1541113

Rehfeldt, G.E. 2004. Interspecific and intraspecific variation in *Picea engelmannii* and its congeneric cohorts: Biosystematics, genecology, and climate change. Gen. Tech. Rep. RMRS-GTR-134. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 18 p. https://doi.org/10.2737/RMRS-GTR-134

Rehfeldt, G.E.; Ferguson, D.E.; Crookston, N.L. 2009. Aspen, climate, and sudden decline in western USA. Forest Ecology and Management. 258: 2353–2364. https://doi.org/10.1016/j.foreco.2009.06.005

Rehfeldt, G.E.; Jaquish, B.C.; López-Upton, J. [et al.]. 2014. Comparative genetic responses to climate for the varieties of *Pinus ponderosa* and *Pseudotsuga menziesii*: Realized climate niches. Forest Ecology and Management. 324: 126–137. https://doi.org/10.1016/j.foreco.2014.02.035

Rehfeldt, G.E.; Worrall, J.J.; Marchetti, S.B.; [et al.]. 2015. Adapting forest management to climate change using bioclimate models with topographic drivers. Forestry. 88(5): 528–539. https://doi.org/10.1093/forestry/cpv019

Rehfeldt, G.E.; Ying, C.C.; Spittlehouse, D.L.; [et al.]. 1999. Genetic responses to climate in *Pinus contorta*: Niche breadth, climate change, and reforestation. Ecological Monographs. 69(3): 375–407. https://doi.org/10.1890/0012-9615(1999)069[0375:GRTCIP]2.0.CO;2

Reich, R.M.; Lundquist, J.E.; Hughes, K. 2016. Host-environment mismatches associated with subalpine fir decline in Colorado. Journal of Forestry Research. 27(5): 1177–1189. https://doi.org/10.1007/s11676-016-0234-1

Reinhardt, K.; Castanha, C.; Germino M.J.; [et al.]. 2011. Ecophysiological variation in two provenances of *Pinus flexilis* seedlings across an elevation gradient from forest to alpine. Tree Physiology. 31: 615–625. https://doi.org/10.1093/treephys/tpr055

Reinhardt, K.; Germino, M.J.; Kueppers, L.M.; [et al.]. 2015. Linking carbon and water relations to drought-induced mortality in Pinus flexilis seedlings. Tree Physiology. 35: 771–782. https://doi.org/10.1093/treephys/tpv045

Renwick, K.M.; Rocca, M.E.; Stohlgren, T.J. 2016. Biotic disturbance facilitates range shift at the trailing but not the leading edge of lodgepole pine's altitudinal distribution. Journal of Vegetation Science. 27(4): 780–788. https://doi.org/10.1111/jvs.12410

Restaino, C.M.; Peterson, D.L.; Littell, J. 2016. Increased water deficit decreases Douglas-fir growth throughout western U.S. forests. Proceedings of the National Academy of Sciences of the United States of America. 113: 9557–9562. https://doi.org/10.1073/pnas.1602384113

Rhoades, C.C.; Fegel, T.S.; Hubbard, R.M.; [et al.]. 2022. Limited seed viability in long-dead serotinous lodgepole pine trees in the Southern Rockies, USA. Forest Ecology and Management. 526: 120565. https://doi.org/10.1016/j.foreco.2022.120565

Rhoades, C.C.; Hubbard, R.M.; Elder, K.; [et al.]. 2020a. Tree regeneration and soil responses to management alternatives in beetle-infested lodgepole pine forests. Forest Ecology and Management. 468: 118182. https://doi.org/10.1016/j.foreco.2020.118182

Rhoades, C.C.; Hubbard, R.M.; Hood, P.R; [et al.]. 2020b. Snagfall the first decade after severe bark beetle infestation of high-elevation forests in Colorado, USA. Ecological Applications. 30(3): e02059. https://doi.org/10.1002/eap.2059

Rhoades, C.C.; Pelz, K.A.; Fornwalt, P.J. [et al.]. 2018. Overlapping bark beetle outbreaks, salvage logging and wildfire restructure a lodgepole pine ecosystem. Forests. 9(3): 101. https://doi.org/10.3390/f9030101

Roberts, L.; McCulley, R.; Burke, I.; [et al.]. 2004. Indications of deep soil water usage by limber pine (*Pinus flexilis*) and skunkbush sumac (*Rhus aromatica*) in northeastern Colorado: An oxygen isotope study. The American Midland Naturalist. 152: 178–182. https://doi.org/10.1674/0003-0031(2004)152[0178:IODSWU]2.0.CO;2

Robinson, D.C.; Geils, B.W. 2006. Modelling dwarf mistletoe at three scales: Life history, ballistics and contagion. Ecological Modelling. 199: 23–38. https://doi.org/10.1016/j.ecolmodel.2006.06.007

Rocca, M.E.; Brown, P.M.; MacDonald, L.H.; [et al.]. 2014. Climate change impacts on fire regimes and key ecosystem services in Rocky Mountain forests. Forest Ecology and Management. 327: 290–305. https://doi.org/10.1016/j.foreco.2014.04.005

Rodman, K.C.; Andrus, R.A.; Carlson, A.R.; [et al.]. 2022a. Rocky Mountain forests are poised to recover following bark beetle outbreaks but with altered composition. Journal of Ecology. 110: 2929–2949. https://doi.org/10.1111/1365-2745.13999

Rodman, K.C.; Davis, K.T.; Chambers, M.E.; [et al.]. 2022b. The historic 2020 fire year in northern Colorado and southern Wyoming: A landscape assessment to inform post-fire forest management. Southwest Ecological Restoration Institutes. 29 p. https://www.fs.usda.gov/research/treesearch/65991

Rodman, K.C.; Veblen, T.T.; Andrus, R.A.; [et al.]. 2021. A trait-based approach to assessing resistance and resilience to wildfire in two iconic North American conifers. Journal of Ecology. 109(1): 313–326. https://doi.org/10.1111/1365-2745.13480

Rodman, K.C.; Veblen, T.T.; Battaglia, M.A.; [et al.]. 2020a. A changing climate is snuffing out post-fire recovery in montane forests. Global Ecology and Biogeography. 29: 2039–2051. https://doi.org/10.1111/geb.13174

Rodman, K.C.; Veblen, T.T.; Chapman, T.B.; [et al.]. 2020b. Limitations to recovery following wildfire in dry forests of southern Colorado and northern New Mexico, USA. Ecological Applications. 30(1): e02001. https://doi.org/10.1002/eap.2001

Romme, W.H.; Allen, C.D.; Bailey, J.D.; [et al.]. 2009. Historical and modern disturbance regimes, stand structures, and landscape dynamics in pinon–juniper vegetation of the western United States. Rangeland Ecology and Management. 62(3): 203–222. https://doi.org/10.2111/08-188R1.1

Romme, W.H.; Knight, D.H. 1981. Fire frequency and subalpine forest succession along a topographic gradient in Wyoming. Ecology. 62(2): 319–326. https://doi.org/10.2307/1936706

Ronco, F. 1990. *Pinus edulis* Engelm. Pinyon. In: Silvics of North America, Volume 1: Conifers. Agriculture Handbook 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 327–337. https://www.fs.usda.gov/research/treesearch/1547

Rother, M.T.; Veblen, T.T. 2016. Limited conifer regeneration following wildfires in dry ponderosa pine forests of the Colorado Front Range. Ecosphere. 7(12): e01594. https://doi.org/10.1002/ecs2.1594

Rother, M.T.; Veblen, T.T. 2017. Climate drives episodic conifer establishment after fire in dry ponderosa pine forests of the Colorado Front Range, USA. Forests. 8(5): 159. https://doi.org/10.3390/f8050159

Rother, M.T.; Veblen, T.T.; Furman, L.G. 2015. A field experiment informs expected patterns of conifer regeneration after disturbance under changing climate conditions. Canadian Journal of Forest Research. 45(11): 1607–1616. https://doi.org/10.1139/cjfr-2015-0033

Ryker, R.A.; Potter, D.R. 1970. Shade increases first-year survival of Douglas-fir seedlings. Research Note INT-119. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 4 p.

Schapira, Z.H.; Stevens-Rumann, C.S.; Shorrock, D. 2021. Subalpine tree seedlings: Assessing aging methodology and drivers of establishment. Forest Ecology and Management. 497: 119516. https://doi.org/10.1016/j.foreco.2021.119516

Schoennagel, T.; Sherriff, R.L.; Veblen, T.T. 2011. Fire history and tree recruitment in the Colorado Front Range upper montane zone: Implications for forest restoration. Ecological Applications. 21(6): 2210–2222. https://www.jstor.org/stable/41416649

Schoennagel, T.; Veblen, T.T.; Romme, W.H. 2004. The interaction of fire, fuels, and climate across Rocky Mountain forests. BioScience. 54(7): 661–676. https://doi.org/10.1641/0006-3568(2004)054[0661:TIOFFA]2.0.CO;2

Schoettle, A.W. 2004. Ecological roles of five-needle pine in Colorado: Potential consequences of their loss. In: Sniezko, R.A.; Samman, S.; Schlarbaum, S.E.; [et al.], eds. 2004. Breeding and genetic resources of five-needle pines: Growth, adaptability and pest resistance. IUFRO Working Party 2.02.15. Proceedings RMRS-P-32. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 124–135. https://www.fs.usda.gov/research/treesearch/52826

Schoettle, A.W.; Coop, J.D. 2017. Range-wide conservation of *Pinus aristata*: A genetic collection with ecological context for proactive management today and resources for tomorrow. New Forests. 48: 181–199. https://doi.org/10.1007/s11056-017-9570-z

Schoettle, A.W.; Negrón, J.F. 2001. First report of two cone and seed insects on *Pinus flexilis*. Western North American Naturalist. 61(2): 252–254. https://scholarsarchive.byu.edu/wnan/vol61/iss2/15

Schoettle, A.W.; Rochelle, S.G. 2000. Morphological variation of *Pinus flexilis* (Pinaceae), a bird-dispersed pine, across a range of elevations. American Journal of Botany. 87(12): 1797–1806. https://doi.org/10.2307/2656832

Schoettle, A.W.; Sniezko, R.A. 2007. Proactive intervention to sustain high-elevation pine ecosystems threatened by white pine blister rust. Journal of Forest Research. 12(5): 327–336. https://doi.org/10.1007/s10310-007-0024-x

Schoettle, A.W.; Burns, K.S.; Cleaver, C.M.; [et al.]. 2019a. Proactive limber pine conservation strategy for the greater Rocky Mountain National Park area. Gen. Tech. Rep. RMRS-GTR-379. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 81 p. https://www.fs.usda.gov/research/treesearch/57621

Schoettle, A.W.; Burns, K.S.; McKinney, S.T.; [et al.]. 2022a. Integrating forest health conditions and species adaptive capacities to infer future trajectories of the high elevation five-needle white pines. Forest Ecology and Management. 521: 120389. https://doi.org/10.1016/j.foreco.2022.120389

Schoettle, A.W.; Goodrich, B.A.; Hipkins, V.; [et al.]. 2012b. Geographic patterns of genetic variation and population structure in *Pinus aristata*, Rocky Mountain bristlecone pine. Canadian Journal of Forest Research. 42: 23–37. https://doi.org/10.1139/x11-152

Schoettle, A.W.; Jacobi, W.R.; Waring, K.M.; [et al.]. 2019b. Regeneration for resilience framework to support regeneration decisions for species with populations at risk of extirpation by white pine blister rust. New Forests. 50: 89–114. https://doi.org/10.1007/s11056-018-9679-8

Schoettle, A.W.; Kegley, A.; Sniezko, R.A.; [et al.]. 2022b. Preparing for invasion: Rust resistance in limber, Great Basin bristlecone, and Rocky Mountain bristlecone pines. In: Murray, M.P.; Smith, C.M.; McKinney, S.T.; [et al.], eds. 2021. Research and management of high-elevation five-needle pines in Western North America: Proceedings of the second high-five conference. The Press at Cal Poly Humboldt: 77–80. https://www.fs.usda.gov/research/treesearch/64863

Schoettle, A.W.; Klutsch, J.; Sniezko, R. 2012a. Integrating regeneration, genetic resistance, and timing of intervention for the long-term sustainability of ecosystems challenged by non-native pests: A novel proactive approach. In: Proceedings of the fourth international workshop on the genetics of host-parasite interactions in forestry: Disease and insect resistance in forest trees. Gen. Tech. Rep. PSW-GTR-240. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 112–123. https://www.fs.usda.gov/research/treesearch/42527

Schoettle, A.W.; Sniezko, R.A.; Kegley, A.; [et al.]. 2011. Preliminary overview of the first extensive rust resistance screening tests of *Pinus flexilis* and *Pinus aristata*. In: Keane, R.E.; Tomback, D.F.; Murray, M.P.; [et al.], eds. The future of high-elevation, five-needle white pines in Western North America: Proceedings of the high five symposium. Proceedings RMRS-P-63. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 265–269. https://www.fs.usda.gov/research/treesearch/38235

Schoettle, A.W.; Sniezko, R.; Kegley, A.; [et al.]. 2014. White pine blister rust resistance in limber pine: Evidence for a major gene. Phytopathology. 104: 163–173. https://doi.org/10.1094/PHYTO-04-13-0092-R

Schopmeyer, C.S. 1974. Seeds of woody plants in the United States. Agricultural Handbook 450. Washington, DC: U.S. Department of Agriculture: 645–655.

Schubert, G.H. 1971. Growth response of even-aged ponderosa pine related to stand density levels in Arizona. Journal of Forestry. 69(12): 857–860. https://doi.org/10.1093/jof/69.12.857

Schuster, W.S.; Alles, D.L.; Mitton, J.B. 1989. Gene flow in limber pine: Evidence from pollination phenology and genetic differentiation along an elevational transect. American Journal of Botany. 76(9): 1395–1403. https://doi.org/10.1002/j.1537-2197.1989.tb15118.x

Shankman, D.; Daly, C. 1988. Forest regeneration above tree limit depressed by fire in the Colorado Front Range. Bulletin of the Torrey Botanical Club. 115: 272–279. https://doi.org/10.2307/2996159

Shaw, J.D.; Steed, B.E.; DeBlander, L.T. 2005. Forest inventory and analysis (FIA) annual inventory answers the question: What is happening to pinyon-juniper woodlands? Journal of Forestry. 103(6): 280–285. https://doi.org/10.1093/jof/103.6.280

Shea, K.L. 1990. Genetic variation between and within populations of Engelmann spruce and subalpine fir. Genome. 33: 1–8. https://doi.org/10.1139/g90-001

Shepperd, W.D.; Edminster, S.B.; Mata, S.A. 2006. Long-term seedfall, establishment, survival, and growth of natural and planted ponderosa pine in the Colorado Front Range. Western Journal of Applied Forestry. 21: 19–26. https://doi.org/10.1093/wjaf/21.1.19

Shepperd, W.D.; Noble, D.L. 1976. Germination, survival, and growth of lodgepole pine under simulated precipitation regimes: A greenhouse study. Research Note RM-328. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 4 p.

Sherriff, R.L.; Platt, R.V.; Veblen, T.T.; [et al.]. 2014. Historical, observed, and modeled wildfire severity in montane forests of the Colorado Front Range. PloS One. 9(9): e106971. https://doi.org/10.1371/journal.pone.0106971

Sherriff, R.L.; Veblen, T.T. 2007. A spatially-explicit reconstruction of historical fire occurrence in the ponderosa pine zone of the Colorado Front Range. Ecosystems. 10(2): 311–323. https://doi.org/10.1007/s10021-007-9022-2

Shinneman, D.J.; Baker, W.L. 2009. Historical fire and multidecadal drought as context for pinon–juniper woodland restoration in western Colorado. Ecological Applications. 19(5): 1231–1245. https://doi.org/10.1890/08-0846.1

Shinneman, D.J.; Baker, W.L.; Rogers, P.C.; [et al.]. 2013. Fire regimes of quaking aspen in the Mountain West. Forest Ecology and Management. 299: 22–34. https://doi.org/10.1016/j.foreco.2012.11.032

Sibold, J.S.; Veblen, T.T. 2006. Relationships of subalpine forest fires in the Colorado Front Range with interannual and multidecadal-scale climatic variation. Journal of Biogeography. 33(5): 833–842. https://doi.org/10.1111/j.1365-2699.2006.01456.x

Sibold, J.S.; Veblen, T.T.; González, M.E. 2006. Spatial and temporal variation in historic fire regimes in subalpine forests across the Colorado Front Range in Rocky Mountain National Park, Colorado, USA. Journal of Biogeography. 33(4): 631–647. https://doi.org/10.1111/j.1365-2699.2005.01404.x

Sidder, A.M.; Kumar, S.; Laituri, M.; [et al.]. 2016. Using spatiotemporal correlative niche models for evaluating the effects of climate change on mountain pine beetle. Ecosphere. 7(7): e01396. https://doi.org/10.1002/ecs2.1396

Sieg, C.H.; McMillin, J.D.; Fowler, J.F.; [et al.]. 2006. Best predictors for postfire mortality of ponderosa pine trees in the Intermountain West. Forest Science. 52(6): 718–728. https://doi.org/10.1093/forestscience/52.6.718

Simonson, S.E.; Opler, P.A.; Stohlgren, T.J.; [et al.]. 2001. Rapid assessment of butterfly diversity in a montane landscape. Biodiversity and Conservation. 10(8): 1369–1386. https://doi.org/10.1023/A:1016663931882

Sindewald, L.A.; Tomback, D.F.; Neumeyer, E.R. 2020. Community structure and functional role of limber pine (*Pinus flexilis*) in treeline communities in Rocky Mountain National Park. Forests. 11(8): 838. https://doi.org/10.3390/f11080838

Smith, J.M.; Paritsis, J.; Veblen, T.T.; [et al.]. 2015. Permanent forest plots show accelerating tree mortality in subalpine fir forests of the Colorado Front Range from 1982 to 2013. Forest Ecology and Management. 341: 8–17. https://doi.org/10.1016/j.foreco.2014.12.031

Smithers, B.V.; North, M.P.; Millar, C.I.; [et al.]. 2018. Leap frog in slow motion: Divergent responses of tree species and life stages to climatic warming in Great Basin subalpine forests. Global Change Biology. 24: e442–e457. https://doi.org/10.1111/gcb.13881

Steele, R. 1990. *Pinus flexilis* James. Limber pine. In: Silvics of North America, Volume 1: Conifers. Agriculture Handbook 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 348–354. https://www.fs.usda.gov/research/treesearch/1547

Steinberg, P.D. 2002. *Pseudotsuga menziesii* var. *glauca*. In: Fire effects information system. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. https://www.fs.usda.gov/database/feis/plants/tree/psemeng/all.html [Accessed 30 September 2022]

Steinhoff, R.J.; Andresen, J.W. 1971. Geographic variation in *Pinus flexilis* and *Pinus strobiformis* and its bearing on their taxonomic status. Silvae Genetica. 20(5–6): 159–67.

Stevens, J.T.; Kling, M.M.; Schwilk, D.W.; [et al.]. 2020. Biogeography of fire regimes in western U.S. conifer forests: A trait-based approach. Global Ecology and Biogeography. 29(5): 944–955. https://doi.org/10.1111/geb.13079

Sthultz, C.M.; Gehring, C.A.; Whitham, T.G. 2007. Shifts from competition to facilitation between a foundation tree and a pioneer shrub across spatial and temporal scales in a semiarid woodland. New Phytologist. 173: 135–145. https://doi.org/10.1111/j.1469-8137.2006.01915.x

Swanston, C.W.; Janowiak, M.K.; Brandt, L.A.; [et al.]. 2016. Forest adaptation resources: Climate change tools and approaches for land managers, 2nd ed. Gen. Tech. Rep. NRS-GTR-87-2. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 161 p. https://doi.org/10.2737/NRS-GTR-87-2

Taylor, J.E.; Mathiasen, R.L. 2002. Limber pine dwarf mistletoe. Forest Insect and Disease Leaflet. U.S. Department of Agriculture, Forest Service. 171 p.

Temperli, C.; Veblen, T.T.; Hart, S.J.; [et al.]. 2015. Interactions among spruce beetle disturbance, climate change and forest dynamics captured by a forest landscape model. Ecosphere. 6(11): 1–20. https://doi.org/10.1890/ES15-00394.1

Tintor, W.L.; Woodhouse, C.A. 2021. The variable climate response of Rocky Mountain bristlecone pine (*Pinus aristata* Engelm.). Dendrochronologia. 68: 125846. https://doi.org/10.1016/j.dendro.2021.125846

Tomback, D.F.; Achuff, P. 2010. Blister rust and western forest biodiversity: Ecology, values and outlook for white pines. Forest Pathology. 40(3-4): 186–225. https://doi.org/10.1111/j.1439-0329.2010.00655.x

Tomback, D.F.; Achuff, P.; Schoettle, A.W.; [et al.]. 2011. The magnificent high-elevation five-needle white pines: Ecological roles and future outlook. In: Keane, R.E.; Tomback, D.F.; Murray, M.P.; [et al.], eds. The future of high-elevation, five-needle white pines in Western North America: Proceedings of the high five symposium. Proceedings RMRS-P-63. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 2–28. https://www.fs.usda.gov/research/treesearch/38188

Tomback, D.F.; Schoettle, A.W.; Chevalier, K.E.; [et al.]. 2005. Life on the edge for limber pine: Seed dispersal within a peripheral population. Ecoscience. 12: 519–529. https://doi.org/10.2980/i1195-6860-12-4-519.1

Troendle, C.A.; King, R.M. 1985. The effect of timber harvest on the Fool Creek watershed, 30 years later. Water Resources Research. 21(12): 1915–1922. https://doi.org/10.1029/WR021i012p01915

Turner, M.G.; Braziunas, K.H.; Hansen, W.D.; [et al.]. 2022. The magnitude, direction, and tempo of forest change in greater Yellowstone in a warmer world with more fire. Ecological Monographs. 92(1): e01485. https://doi.org/10.1002/ecm.1485

Urza, A.K.; Weisberg, P.J.; Chambers, J.C.; [et al.]. 2019. Shrub facilitation of tree establishment varies with ontogenetic stage across environmental gradients. New Phytologist. 223(4): 1795–1808. https://doi.org/10.1111/nph.15957

U.S. Fish and Wildlife Service [USFWS]. 2022. Endangered and threatened wildlife and plants; Threatened species status with section 4(d) rule for whitebark pine (*Pinus albicaulis*). Federal Register. 87(240): 76,882–76,917. https://www.govinfo.gov/content/pkg/FR-2022-12-15/pdf/2022-27087.pdf#page=1 [Accessed 20 June 2024]

U.S. Geological Survey [USGS]. 2020. National elevation dataset. Reston, VA: U.S. Department of Interior, U.S. Geological Survey, 3D Elevation Program. https://nationalmap.gov/elevation.html. [Accessed 20 August 2020].

U.S. National Vegetation Classification [USNVC]. 2020. United States national vegetation classification database, V2.03. Washington, DC: Federal Geographic Data Committee, Vegetation Subcommittee. https://usnvc.org. [Accessed 20 January 2020].

Vander Wall, S.B.; Balda, R.P. 1977. Coadaptations of the Clark's Nutcracker and the pinon pine for efficient seed harvest and dispersal. Ecological Monographs 47(1): 89–111. https://doi.org/10.2307/1942225

Veblen, T.T. 1986. Age and size structure of subalpine forests in the Colorado Front Range. Bulletin of the Torrey Botanical Club. 113(3): 225–240. https://doi.org/10.2307/2996361

Veblen, T.T.; Donnegan, J.A. 2005. Historical range of variability for forest vegetation of the National Forests of the Colorado Front Range. Golden, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Region. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5386430.pdf [Accessed 20 June 2024]

Veblen, T.T.; Hadley, K.S.; Reid, M.S. 1991a. Disturbance and stand development of a Colorado subalpine forest. Journal of Biogeography. 18(6): 707–716. https://doi.org/10.2307/2845552

Veblen. T.T.; Hadley, K.S.; Reid, M.S.; [et al.]. 1991b. The response of subalpine forests to spruce beetle outbreak in Colorado. Ecology. 72(1): 213–231. https://doi.org/10.2307/1938916

Villalba, R.; Veblen, T.T.; Ogden, J. 1994. Climatic influences on the growth of subalpine trees in the Colorado Front Range. Ecology. 75(5): 1450–1462. https://doi.org/10.2307/1937468

Webster, K.L.; Johnson, E.A. 2000. The importance of regional dynamics in local populations of limber pine (*Pinus flexilis*). Ecoscience. 7: 175–182. https://doi.org/10.1080/11956860.2000.11682586

West, D.R.; Briggs, J.S.; Jacobi, W.R. 2014. Mountain pine beetle-caused mortality over eight years in two pine hosts in mixed-conifer stands of the southern Rocky Mountains. Forest Ecology and Management. 334: 321–330. https://doi.org/10.1016/j.foreco.2014.09.012

Williams, A.P.; Cook, B.I.; Smerdon, J.E. 2022. Rapid intensification of the emerging southwestern North American megadrought in 2020–2021. Nature Climate Change. 12: 232–234. https://doi.org/10.1038/s41558-022-01290-z

Williams, T.J.; Tomback, D.F.; Grevstad, N.; [et al.]. 2020. Temporal and energetic drivers of seed resource use by Clark's nutcracker, keystone seed disperser of coniferous forests. Ecosphere. 11: e03085. https://doi.org/10.1002/ecs2.3085

Windmuller-Campione, M.A.; Long, J.N. 2016. Limber pine (*Pinus flexilis* James), a flexible generalist of forest communities in the Intermountain West. PLoS One. 11: e0160324. https://doi.org/10.1371/journal.pone.0160324

Wion, A.P.; Weisberg, P.J.; Pearse, I.S.; [et al.]. 2020. Aridity drives spatiotemporal patterns of masting across the latitudinal range of a dryland conifer. Ecography. 43(4): 569–580. https://doi.org/10.1111/ecog.04856

Wooten, J.T.; Stevens-Rumann, C.S.; Schapira, Z.H.; [et al.]. 2022. Microenvironment characteristics and early regeneration after the 2018 Spring Creek Wildfire and post-fire logging in Colorado, USA. Fire Ecology. 18: 10. https://doi.org/10.1186/s42408-022-00133-8

Worrall, J.J.; Egeland, L.; Eager, T.; [et al.]. 2008. Rapid mortality of *Populus tremuloides* in southwestern Colorado, USA. Forest Ecology and Management. 255: 686–696. https://doi.org/10.1016/j.foreco.2007.09.071

Wright, J.L.; Roy, S.S. 2022. Geospatial characteristics of Colorado wildfire occurrences from 2001 to 2020. Remote Sensing Applications: Society and Environment. 28: 100872. https://doi.org/10.1016/j.rsase.2022.100872

Yang, R.C.; Yeh, F.C. 1993. Multilocus structure in *Pinus contorta* Dougl. Theoretical and Applied Genetics. 87(5): 568–576. https://doi.org/10.1007/BF00221880

Yang, R.C.; Yeh, F.C. 1995. Patterns of gene flow and geographic structure in *Pinus contorta* Dougl. Forest Genetics. 2: 65–75.

Yue, X.; Mickley, L.J.; Logan, J.A.; [et al.]. 2013. Ensemble projections of wildfire activity and carbonaceous aerosol concentrations over the western United States in the mid-21st century. Atmospheric Environment. 77: 767–780. https://doi.org/10.1016/j.atmosenv.2013.06.003

Zier, J.L; Baker, W.L. 2006. A century of vegetation change in the San Juan Mountains, Colorado: An analysis using repeat photography. Forest Ecology and Management. 228(1–3): 251–262. https://doi.org/10.1016/j.foreco.2006.02.049

Zweifel, R.; Bangerter, S.; Rigling, A. 2012. Pine and mistletoes: How to live with a leak in the water flow and storage system? Journal of Experimental Botany. 63: 2565–2578. https://doi.org/10.1093/jxb/err432

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Future-Oriented Wildlife Management in the Pike-San Isabel and Arapaho-Roosevelt National Forests Under a Warming Climate: Anticipating Change and Facing Uncertainties With a Focus on Avian Species

Angela M. White

Introduction

Climate change has impacted terrestrial, freshwater, and ocean ecosystems across the globe changing disease dynamics, disturbance regimes, and nutrient cycling, resulting in the degradation of key ecosystem services (Cooley et al. 2022; Parmesan et al. 2022). Birds provide a diverse array of ecosystem services, including pest control, pollination, seed dispersal, and nutrient cycling (Sekercioğlu et al. 2004, 2006), and are of conservation concern due to human impacts on avian diversity and abundance (Gaston et al. 2003; Jetz et al. 2007; Rosenburg et al. 2019). Relative to other taxa, birds are predicted to be less vulnerable to climate change due to their vagility and ability to shift their range and phenology in response to environmental changes (Pacifici et al. 2017; Siegel et al. 2014; Tingley et al. 2009); however, a species' ability to adapt can vary significantly with cascading impacts on communities. For example, avian range shifts can result in new species interactions and vulnerabilities that may have consequences for ecosystem function and stability (Weiskopf et al. 2020).

Planning and responding to climate concerns with the appropriate environmental and conservation actions often rely on an understanding of individual species' sensitivity to climate-induced changes, the rate and magnitude of climate-induced change, and the species' ability to adapt to these changes (Fischlin et al. 2007). The application of adaptive management approaches is not only complicated by the uncertainties in species' responses to climate change, but also differences in climate projections that predict different rates and magnitude of change. Therefore, predicting climate change impacts on biodiversity remains a major challenge (Bellard et al. 2012; Pacifici et al. 2015; Pereira et al. 2010), yet managers are frequently tasked with the identification, prioritization, and implementation of adaptive management options to lessen the impacts of climate change on the resources they manage.

Although we cannot predict how most species will respond to climate change, several assessments have identified species or habitats in the Rocky Mountain Region that are particularly vulnerable to climate change (Halofsky et al. 2018a,b; Rice et al. 2018). For avian species, the Forest Service Manual for the Rocky Mountain Region (USDA Forest Service Manual, Chapter 2670) provides a list of 21 species that are considered sensitive due to their geographic distribution, dispersal ability, population dynamics, life history strategies, and habitat requirements (table 1). Risk-reduction management actions to protect these vulnerable species are likely to mitigate impacts, but traditional conservation strategies focused on resisting change may not be possible or economically prudent under a changing climate (Aplet and Cole 2012). This chapter adds to the existing assessments applicable to the Pike-San Isabel and Arapaho-Roosevelt National

Forests (PSI-AR) while applying future-oriented conservation planning strategies (van Kerkhoff et al. 2019; Wyborn et al. 2016; fig. 1) that are designed around anticipating changes, recognizing societal values, shared governance, and ongoing learning.

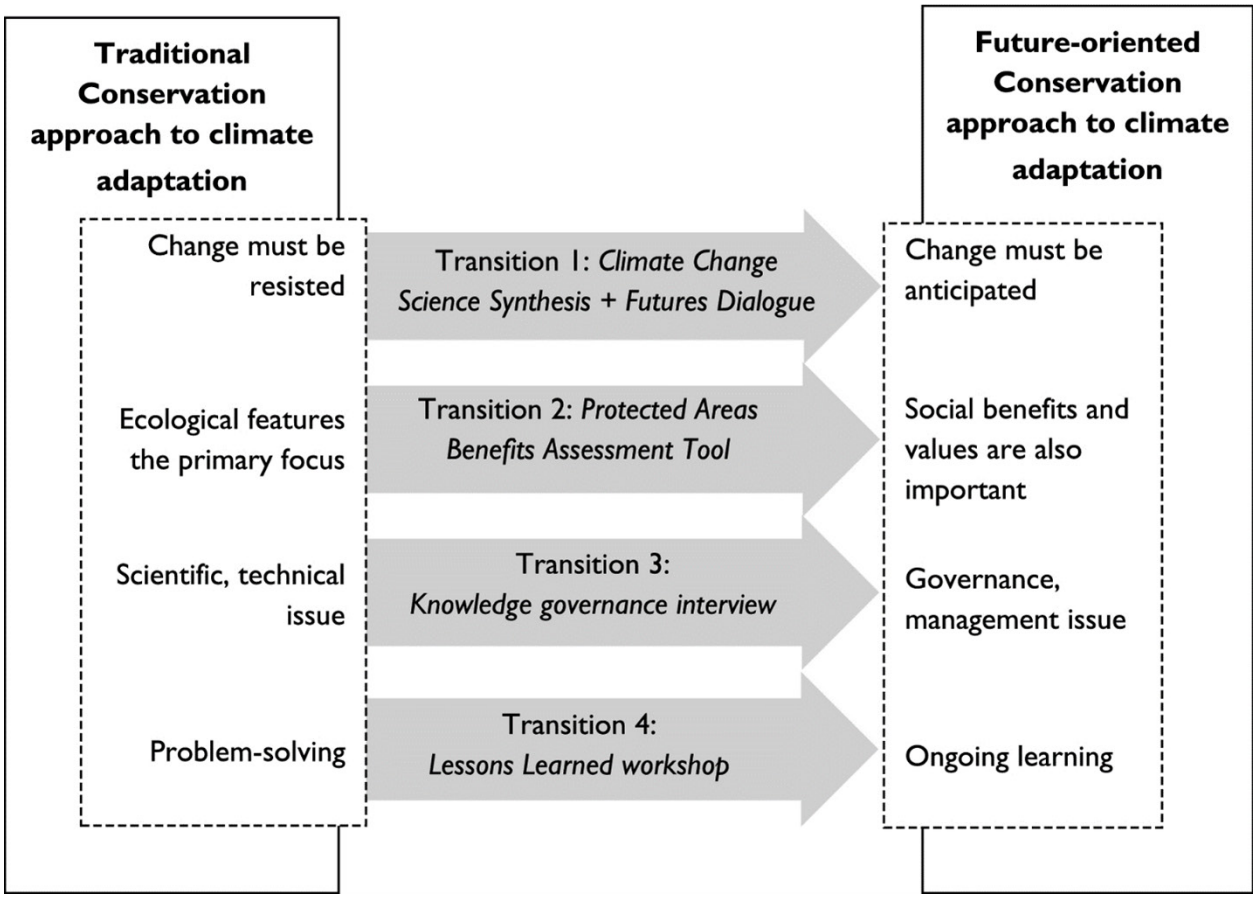


Figure 1—Four conceptual transitions and the tools that support them, from traditional conservation thinking and practices towards future-oriented conservation. Figure from van Kerkhoff et al. 2019.

Table 1—List of sensitive avian species identified in the Forest Service Manual for Rocky Mountain Region (USDA Forest Service Manual, Chapter 2670) that are found within the Pike (P), San Isabel (SI), Arapaho (A), and Roosevelt (R) National Forests. Residency status (i.e., breeding, non-breeding, migrant, or annual resident) and rationale for inclusion are given for each species.

Species	Status	Rationale
Northern goshawk (*Accipiter gentilis)*	Resident (PS-IAR)	Widely distributed across coniferous forests, but locally rare. This species requires mature coniferous and aspen forests, and there have been significant declines in habitat quantity and quality due to harvest and/or development activity in previously suitable nesting habitat. Most local range on Forest Service lands.
Boreal owl (*Aegolius funereus)*	Resident (PSI-AR)	Evidence of long-term population decline, patchily distributed in the late-seral spruce forest habitat. Species is a secondary cavity nester dependent on woodpeckers and/or natural processes for nest site construction.
Short-eared owl (*Asio flammeus)*	Resident (AR), Non-breeding (PSI)	Long-term habitat declines of high-structured grasslands for nesting. Declining population trend.
Burrowing owl (*Athene cunicularia)*	Breeding (PSI-AR)	Primarily nests in prairie dog burrows in open grasslands; declining black-tailed prairie dog (candidate for ESA listing) colonies provide principal habitat. Declining population trends across range.
American bittern (*Botaurus lentiginosus)*	Breeding (PSI-AR)	Declining population trend, likely due to habitat decline. Loss and degradation of wetlands with robust emergent vegetation, due to drainage, filling, conversion to agriculture or recreational use, siltation, and pollution; needed for nesting and wintering habitat.
Ferruginous hawk (*Buteo regalis)*	Resident (PSI-AR)	High concern for declining habitat, localized downward population trends, and sensitivity to disturbance at nest sites. Habitat of open pinyon-juniper woodlands has declined with fire suppression and industrial and recreational activities impact reproductive success.
Chestnut-collared longspur (*Calcarius ornatus)*	Migrant (PSI-AR)	Habitat is declining range wide due to conversation of grasslands to croplands. Declining population trends across range.
Greater sage-grouse (*Centrocercus urophasianus)*	Resident (PSI-AR)	Significant population declines range wide, with several local extirpations linked to loss, fragmentation, and degradation of sagebrush habitat by conversion to croplands, mining and energy development, urbanization, invasive species, increase wildfire size, and overgrazing.
Mountain plover (*Charadrius montanus)*	Breeding (PSI-AR), Migrant (AR)	Significant population decline at key breeding areas. This species is associated with shortgrass and shrub-steppe habitats throughout its breeding and wintering range. Disturbance by prairie dogs, bison, heavy grazing, fire, drought, and cultivation create preferred large, flat grassland expanses with sparse, short vegetation, and bare ground.

Table 2 continued.

Species	Status	Rationale
Black tern (*Chlidonias niger*)	Breeding (PSI-AR)	Declining population trends with small and localized breeding populations leading to vulnerability of local extirpation. Primary threat is habitat loss and conversion of freshwater marsh habitat for nesting sites.
Northern harrier (*Circus hudsonius*)	Resident (PSI-AR)	This species is widely distributed, but declining population trends throughout western United States. Livestock grazing affects habitat quality of wetlands or grasslands with tall dense vegetation and high residual cover.
Olive-sided flycatcher (*Contopus cooperi*)	Breeding (PSI-AR)	Continent-wide declining population trend likely due to habitat alteration or loss on wintering grounds in Central and South America. Declines in breeding habitat quality and vulnerability to habitat modification by fire suppression, salvage logging, and timber management practices.
White-tailed ptarmigan (*Lagopus leucura*)	Resident (PSI-AR)	Populations are regionally isolated and population sizes are small, with poor ability to colonize new areas, thus susceptible to local extirpation. Many populations may not be self-sustaining. Cadmium from mining may be affecting lifespans of this species. Most range in Forest Service lands.
Loggerhead shrike (*Lanius ludovicianus*)	Breeding (PSI-AR)	Substantial population decline from historic levels, reasons uncertain. This species prefers open habitat of grasses and forbs of low stature, interspersed with bare ground and shrubs or tress, and often is associated with cropland and pastures.
Lewis's woodpecker (*Melanerpes lewis*)	Resident (PSI-AR)	Both habitat and population declines. Depends on large-diameter standing snags and diseased trees in late-successional forests, and mature lowland riparian cottonwood forests, both habitat types being lost or degraded.
Long-billed curlew (*Numenius americanus*)	Breeding (PSI)	Declining population trend. Habitat of grassland is being converted to cropland and is unable to adjust to grazed lands or cropland.
Cassin's sparrow (*Peucaea cassinii*)	Breeding (PSI)	While numbers fluctuate markedly from year to year, significant decrease in population across North America. Species needs structured grasslands that are often lost with intensive grazing and as grasslands turn to shrublands.
Purple martin (*Progne subis*)	Migrant (PSI-AR), Breeding (PSI)	Since areas of necessary habitat components are relatively sparse and widely separated, breeding colonies are small and vulnerable to local extirpation. This species is associated with large stands of pure mature aspen that have healthy populations of cavity excavators and large diameter trees with heart rot near large open areas with summer water.

Table 2 continued.

Species	Status	Rationale
Flammulated owl (*Psiloscops flammeolus*)	Breeding (PSI-AR)	Habitat vulnerability of late successional ponderosa pine habitat with historical alteration. Declining population trends and low reproductive rates makes this species vulnerable to local or regional extirpation.
McCown's longspur (*Rhynchophanes mccownii*)	Migrant (PSI-AR) Breeding (AR)	Extremely restricted national range with weak declining population trend data range wide. Productivity linked to rangeland health.
Brewer's sparrow (*Spizella breweri*)	Breeding (PSI-AR)	Significant and historic declines in high risk, vulnerable habitat due to sagebrush conversion and invasion of cheatgrass. Strong population declines across species range, including its heart—the Great Basin.

Pike-San Isabel and Arapaho-Roosevelt National Forests (PSI-AR)

The PSI-AR lies within the Southern Rocky Mountains Steppe–Open Woodland–Coniferous Forest–Alpine Meadow Province of the United States Temperate Steppe Ecoregion (McNab et al. 2007; fig. 2). The climate of this ecoregion is characterized by short, warm summers and long, cold winters. Average current temperatures (1985–2015) in the ecoregion vary substantially throughout the year with average daily temperatures ranging typically from 57 °F in the summer (June, July, August) to 20 °F in the winter (December, January, February). Precipitation is moderate, increasing with altitude where it generally occurs as snow. Topography varies from high mountains to gently rolling valleys in the north to steeply sloped, precipitous, flat-topped mountains in the south. Vegetation is varied and includes predominately lodgepole pine, ponderosa pine, alpine tundra, and sagebrush cover types in the northern sections and ponderosa pine, pinyon-juniper, fir-spruce, chaparral-mountain shrub, and alpine tundra in the south.

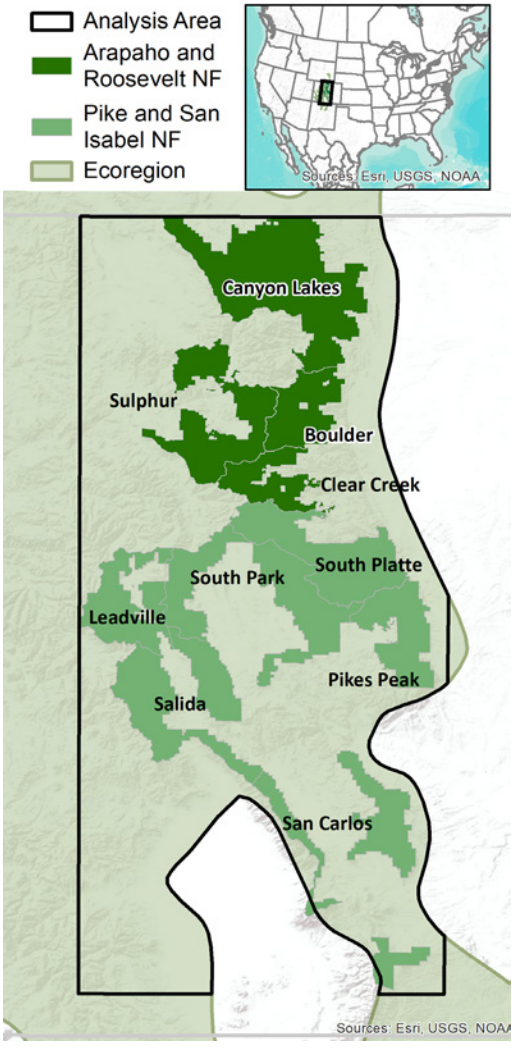


Figure 2—The location of the Pike, San Isabel, Arapaho, and Roosevelt (PSI-AR) National Forests (including district boundaries) within the Southern Rocky Mountains Steppe–Open Woodland–Coniferous Forest–Alpine Meadow Province of the United States Temperate Steppe Ecoregion focal area.

Climate Change Projections for the PSI-AR

Future climate projections can be used by decision-makers to proactively address the impacts of climate change on the resources they manage. However, a range of assumptions about the magnitude and pace of future emissions are used by scientists to develop different emission scenarios, upon which climate model projections are based. Different general circulation models (GCMs) provide alternative representations of the Earth's natural climate variability and response to emissions that influence predicted change in global surface temperatures. These global models are too coarse (100 km resolution) to resolve many important regional climatic patterns, and statistical downscaling techniques are used to translate large-scale data into smaller spatial scales appropriate for managers. Therefore, climate projections are dependent upon choice of GCM, emission scenario, and downscaling method. Differences in the outputs of these projections complicate our ability to manage adaptively for a set of potential "known" future conditions.

Rangwala et al. 2024 (this report) provided four downscaled climate models that forecast climatic conditions for the period 2035–2065. Relative to current conditions (1985–2015), these models resulted in future climates that were Warm and Wet (WW), Hot (H), Very Hot and Wet (VHW), and Very Hot and Dry (VHD). All four climate projections predicted an increase in daily temperatures (fig. 3), although increases in temperature varied seasonally and by climate model. The predicted increase in temperature was lowest in the WW scenario (ESM2M.rcp8.5). The H scenario (CNRM-CM5.rcp8.5) predicted similar temperature increases as the WW scenario in spring (defined here as the months of March, April, May) and fall (September, October, November), but relatively higher summer (June, July, August) and winter (December, January, February) temperatures. Temperatures in the VHW (CanESM2.rcp8.5) and VHD (IPSL-CM5A-MR.rcp8.5) scenarios predicted the largest temperature increases, with average summer temperatures increasing by > 5 °F. Increases in the VHD model tended to be higher than the VHW scenario, except in the spring months when VHW temperatures exceeded that of VHD. In the spring, average daily temperatures in the VHW scenario shifted relative to current conditions by ~7 °F, such that this scenario resulted in a range of mean daily spring temperatures that had little overlap with current spring temperatures (fig. 3).

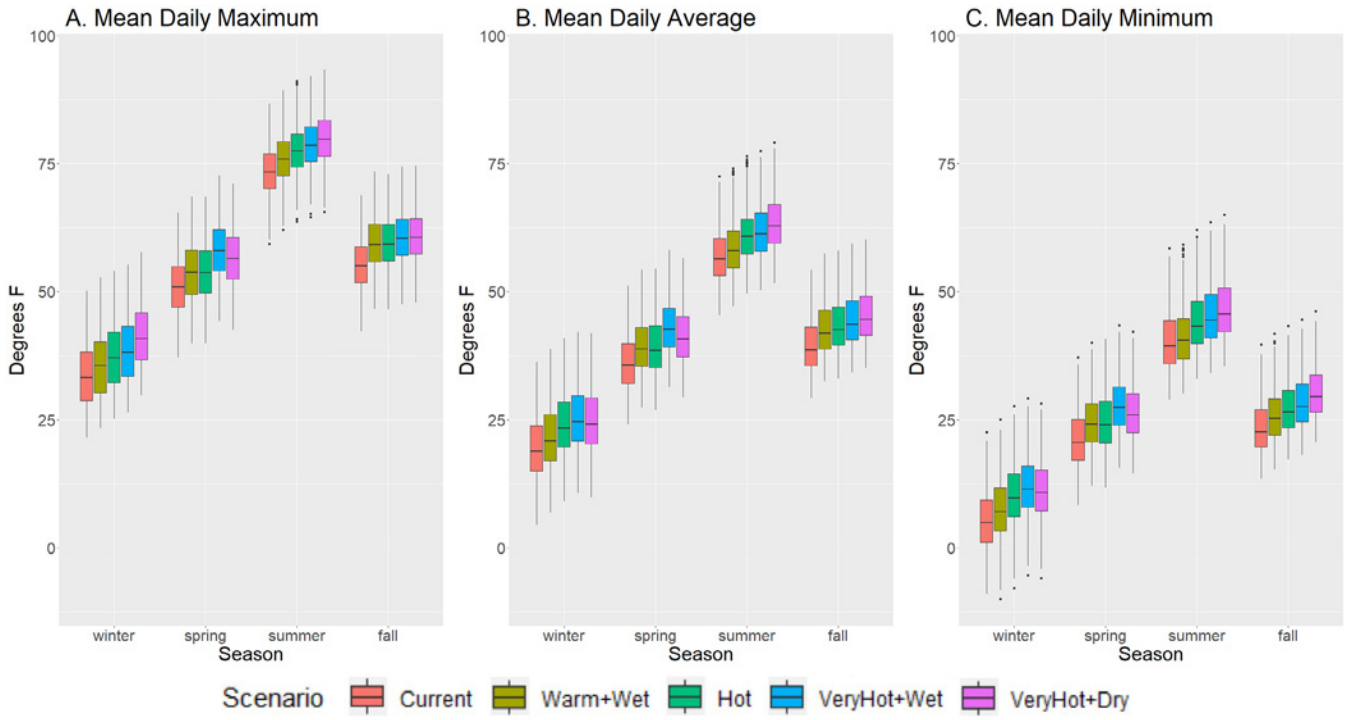


Figure 3a-c—Boxplots displaying mean daily maximum, daily average, and daily minimum temperatures predicted under four different climate change scenarios relative to current temperatures.

Increases in temperature not only varied by scenario and season, but also varied across the PSI-AR, with more extreme increases in spring temperature in the southwestern ranger districts (fig. 4). By contrast, in the summer months, temperature increases in these ranger districts were predicted to be less severe, although the degree of difference varied by climate model (fig. 4). In the fall and winter, temperatures tended to be highest in the Canyon Lakes Ranger District.

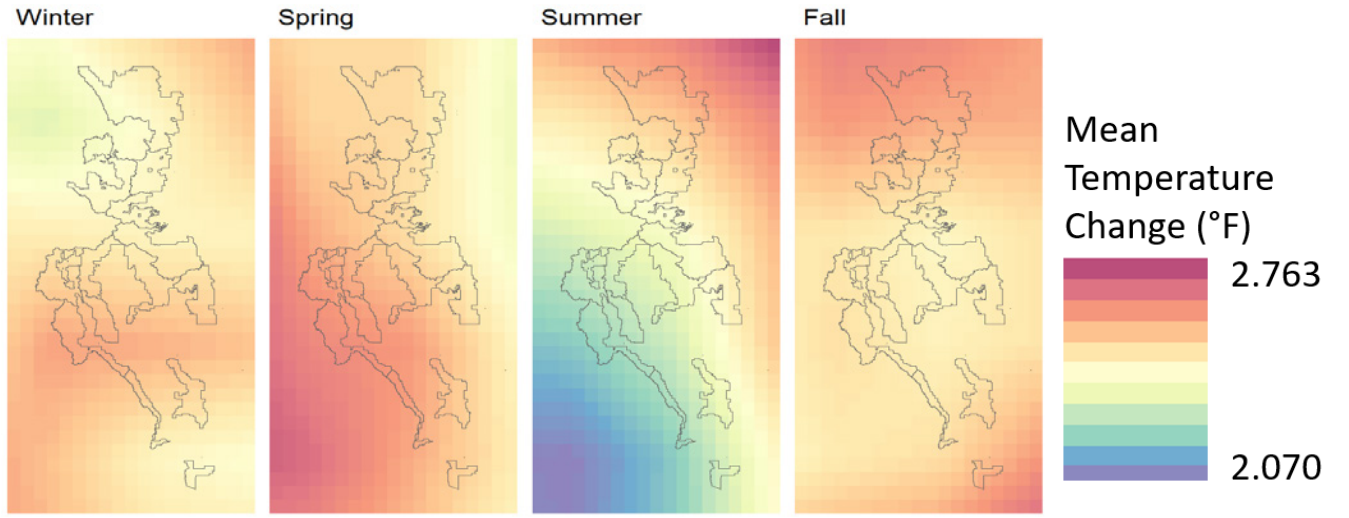


Figure 4a—Average mean temperature change(degrees F) predicted across the PSI-AR by season under four climate scenarios.



Figure 4b—Standard deviations (degrees F) predicted across the PSI-AR by season under four climate scenarios.

Climate Change Impacts on Avian Species Within the PSI-AR

Frequently the climatic condition over a species' current distribution is used as a proxy for a species' physiological tolerances (i.e., temperature and moisture requirements necessary for a species to survive and reproduce). Based on a species' physiological tolerance, climate change models can be used to predict where a species' distribution is likely to contract (reduce in size), expand (increase in size), and/or shift (occur in a different location). Existing datasets on bird species distributions indicate that 269 bird species have current distributions that include parts of the PSI-AR (BirdLife

International 2018). Birds use habitat within the PSI-AR for breeding (132 species), wintering (69 species), for migration (36), or to support their populations year-round (101 species, Appendix A). For most of these species the focal area is only a small portion of their entire distribution (see fig. 5); for these species local management decisions are less likely to influence a species conservation status as these species range across wide environmental gradients. Species with larger portions of their range (> 10 percent) within the focal area include brown-capped rosy finch (*Leucosticte australis*) and Gunnison sage-grouse (*Centrocercus minimus*), and the migratory range of the lark bunting (*Calamospiza melanocorys*). In addition, two species have genetically distinct breeding populations that occur within the focal area: the least flycatcher (*Empidonax minimus*) and western bluebird (*Sialia mexicana*). In mountainous regions, observed changes in climate have not only resulted in changes to the geographic range of avian species, but also the timing of their seasonal life cycles (phenology).

Anticipating Climate-Induced Changes in Species Distributions

How vulnerable an individual species is to changes in climate will be influenced by its exposure to climatic changes and its sensitivity to these changes. To better understand the probability of changes in avian communities under each of the four climate projections, I first calculated the change in mean and maximum temperatures across the distributions of each of the 132 species that breed within the PSI-AR (BirdLife International 2018) at a 30-m pixel resolution. For each pixel, I classified the predicted change in temperature as < 3.9 °F, 3.9–4.4 °F, 4.5–5.0 °F, 5.1–5.5 °F, 5.6–6.0 °F, and > 6 °F based on NatureServe's Climate Change Vulnerability Index (Young et al. 2011, fig. 6). The Climate Change Vulnerability Index can be used to interpret the likely effect of changes in temperature on a species' vulnerability to climate change whereby a species vulnerability increases with increasing changes in temperature.

As the predicted change in temperature was not always uniform across the species distribution (e.g., fig 6, bottom row), an overall species vulnerability index was assigned to each species under each scenario based on the temperature changes experienced across the majority of their range. Based on these calculations, all species would be experiencing an increase in temperature across their distributions, with the VHD and VHW scenarios predicting temperature increases > 5.6 °F (fig. 7). As changes in species' distributions have already occurred in several mountainous bird species in the northern hemisphere in response to current climate change (reviewed in Scridel et al. 2018), we would predict changes in species distributions even under the more moderate WW and H climate scenarios (< 3.9 °F). In addition to range contractions, we need to anticipate a redistribution of species, including the addition of species whose ranges will likely shift to include parts of the PSI-AR where they currently do not occur.

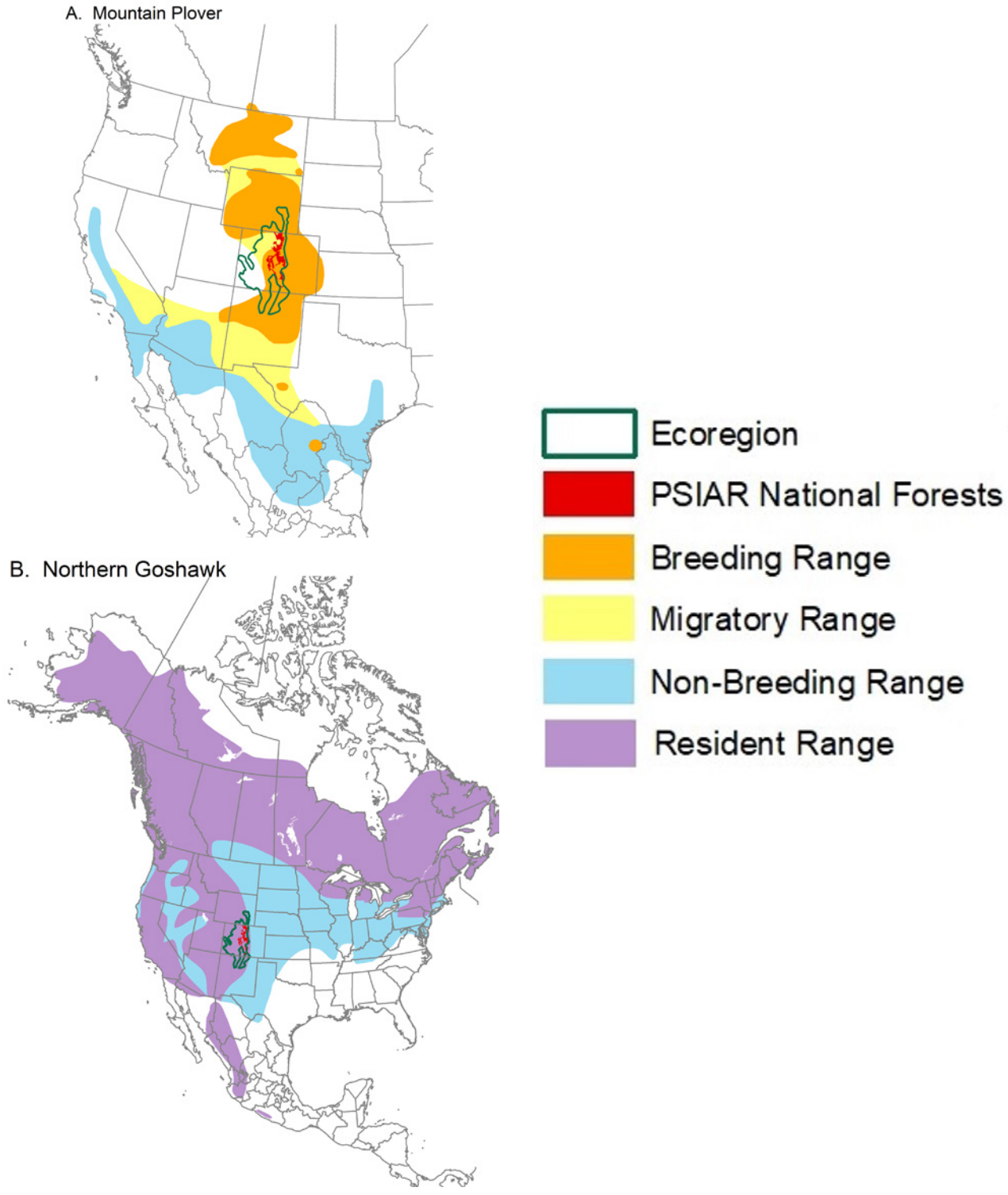


Figure 5a-b—Distribution of two bird species found within the Pike, San Isabel, Arapaho, and Roosevelt (PSI-AR) National Forests that are included in table 1. A green outline delineates the Southern Rocky Mountains Steppe–Open Woodland–Coniferous Forest–Alpine Meadow Province of the United States that was used as the ecological boundary for the PSI-AR (outlined in red).

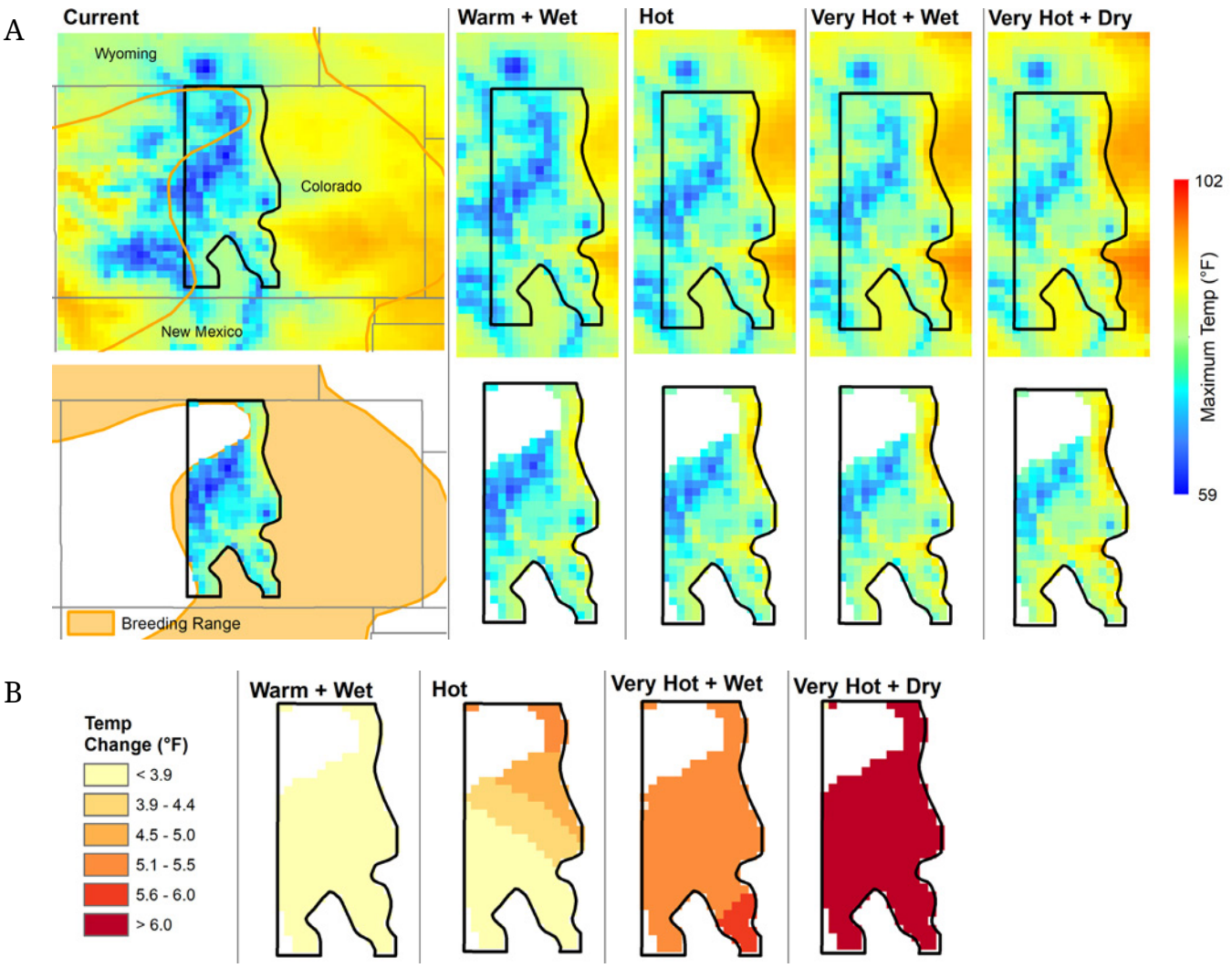


Figure 6a-b—Example of workflow used to assign species vulnerability index. (a) Mean maximum summer temperatures (June, July, August) experienced currently (1985–2015) compared to future temperatures (2035–2065) predicted under four different climate change scenarios (top). Second row provides an example of a single species current distribution (orange) within the larger ecoregion assessment area (black outline) that was used in individual species calculations. (b) shows how the scenario impacted projections of temperature across the range of the species.

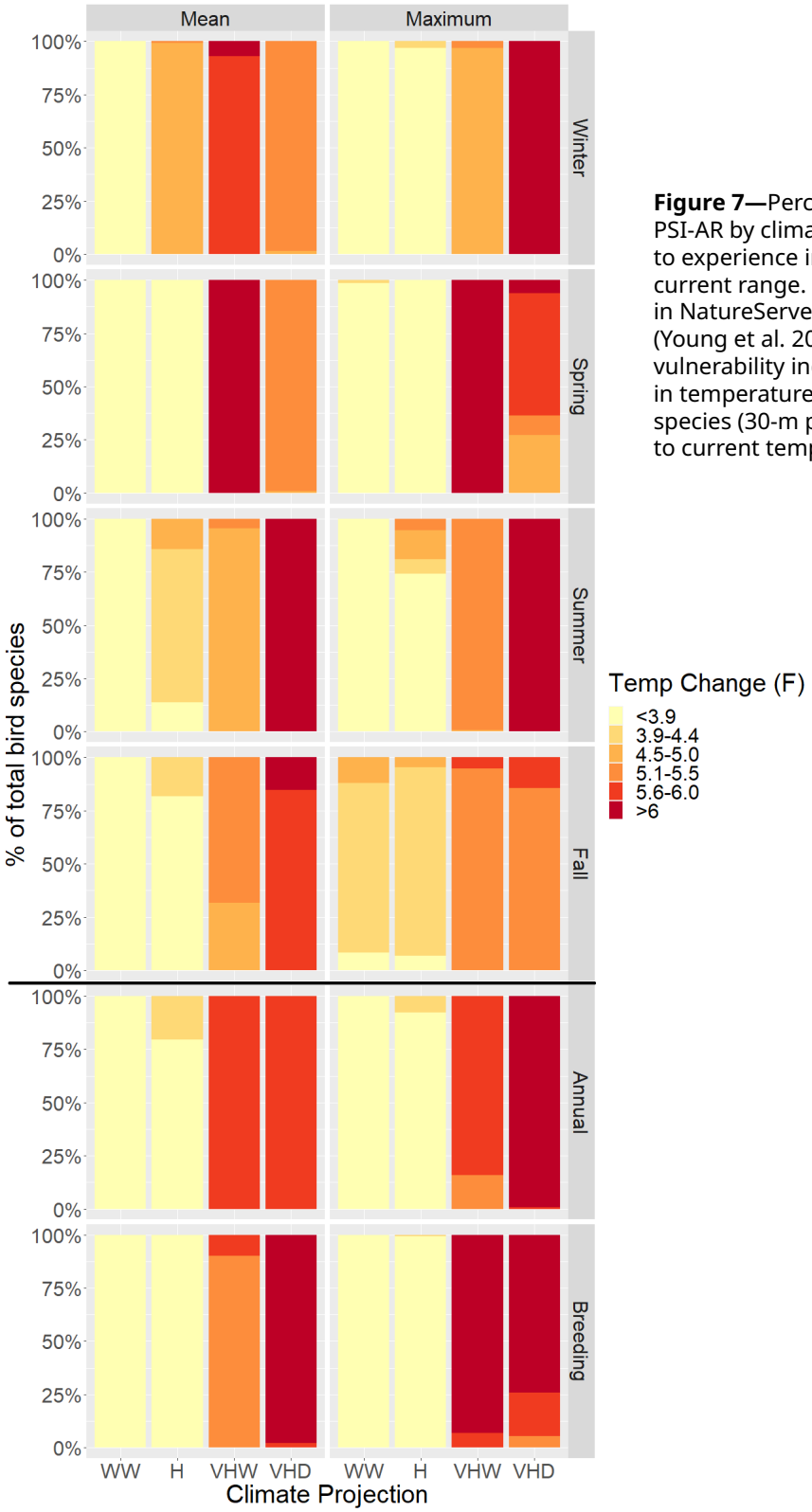


Figure 7—Percentage of 132 species breeding within the PSI-AR by climate scenario and season that are predicted to experience increases in temperature across their current range. Temperature bins are based on guidelines in NatureServe's Climate Change Vulnerability Index (Young et al. 2011). Counts of species into a species vulnerability index is based on spatially-explicit changes in temperature across the current distribution of each species (30-m pixel) under each climate scenario relative to current temperatures.

Anticipating Climate-Induced Changes in Phenology

Bird species that are year-around residents of the PSI-AR (101 of the 269 species) are exposed to temperatures that vary dramatically throughout the year from mean daily temperatures of 15 °F in the winter to 60°F in the summer (fig. 3). These species would have experienced evolutionary pressure to thrive within the range, average, and extreme conditions experienced throughout the year and are adapted to a wide range of climatic conditions. As year-round residents, these species are better positioned to respond to local changes in climate and timing of these changes. Conversely, migratory species, and spring migrants in particular (i.e., those species that use the PSI-AR for breeding grounds), require special consideration, as they must time their departure from wintering grounds with conditions and resource availability at their breeding grounds. Recent work has shown that population declines are most widely reported in migratory species that are not advancing their migration dates in response to climate change (Møller et al. 2008).

Within the PSI-AR, 132 species have populations that generally begin breeding in late spring to begin breeding (Birdlife International 2018). Species that are unable to shift the timing of spring migration will not only have a shorter duration of time in which to breed, but will face increased breeding season temperatures (Halupka and Halupka 2017), particularly under the VHW and VHD scenarios (fig. 7). Changes in the timing of spring migration are the most commonly reported phenological responses to climate change (reviewed in Gill et al. 2014) and can lead to phenological mismatches with prey resources (e.g., resources may peak in response to earlier spring conditions) before birds arrive/fledge, which can lead to lower reproductive success (Ross et al. 2017). Earlier spring conditions are predicted under all climate scenarios (2-3 weeks earlier; Rangwala et al. 2024 (this report).

Anticipating Climate-Induced Changes in Habitat

Forest structure and composition provide the environmental conditions needed to support essential food, resting, and breeding requirements for wildlife and changes in forest structure will impact suitable habitat for any given species (Fontaine and Kennedy 2012; Stephens et al. 2014; White et al. 2013). Therefore, forest structure and composition, often targeted by management actions such as fuel reduction (i.e., forest thinning) and mediated by temperature and precipitation, will also determine whether species can persist on the landscape.

The amount of annual precipitation within the PSI-AR ecoregion is also highly variable and seasonal, with most precipitation occurring in the spring and summer. All climate change scenarios, with the exception of the VHD scenario, predict an increase in summer and spring precipitation relative to current conditions. In contrast, the VHD scenario predicts a decrease in precipitation throughout the year (fig. 8). Under all

models, seasonal changes in precipitation and temperature are predicted to lead to a shift in snowline elevation from its current levels (8,400 ft) to 9,360 ft in the WW scenario, to as much as 10,400 ft in the VHD scenario. Decreases in snow will have negative impacts on some wildlife species that are dependent on snow for hunting and or thermoregulatory requirements (Penczykowski et al. 2017), and the volume and timing of snowmelt will impact fish, amphibians, and other riparian-associated species (Smith et al. 2024, this report).

Although changes in precipitation are often used as a proxy for water availability, the availability of water for plants also depends on ambient temperature, soil type, vegetation conditions, and whether precipitation falls as rain or snow. A more informative proxy for water availability for plants is climatic water deficit (CWD). CWD is when the evaporative demand of vegetation exceeds annual water availability, and mean annual CWD across all scenarios is expected to increase slightly relative to the current deficit (fig. 8). Therefore, despite predictions that precipitation may increase, all models predict that vegetation will likely be more water stressed in the future—a deficit which is expected to be most severe in the VHD model. Water-stressed vegetation is likely to influence the availability and abundance of resources for birds including fruits/seeds and insects (Barton and Ives 2014; McCluney et al 2012). Changes in temperature and CWD can also change habitat structure and composition by altering the probability of insect outbreaks and severity of wildfires—changes already being observed within the PSI-AR (Chapman et al. 2020). The pace of these changes is likely to be most rapid under the VHD model compared to the other climate scenarios as changes in temperature and moisture availability will also influence seedling establishment and survival (Davis et al. 2019).

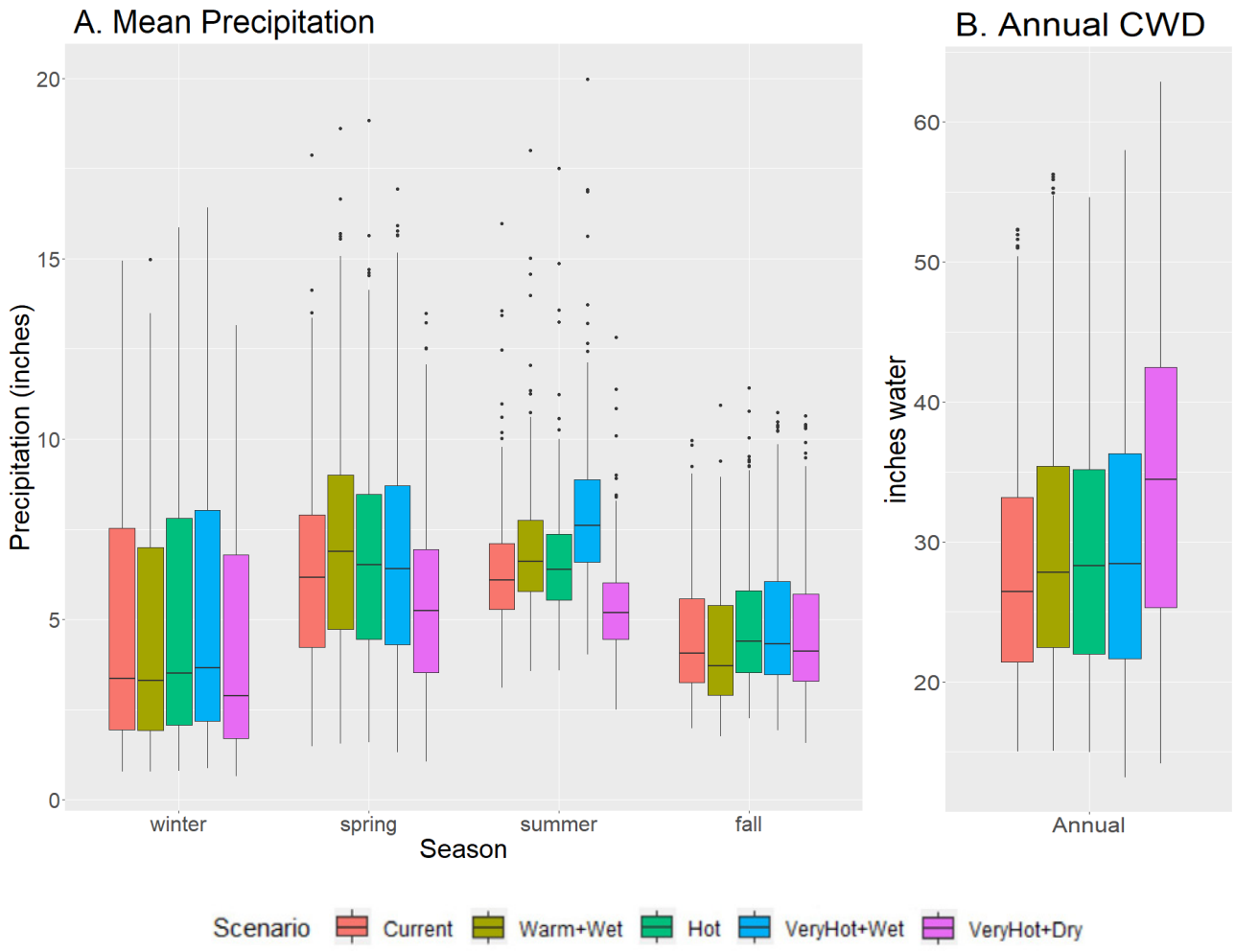


Figure 8a-b—Boxplots displaying seasonal precipitation under current and future climate scenarios. Patterns of precipitation throughout the year influence annual climatic water deficit.

Embracing the Uncertainties and Moving Forward

Adaptive capacity describes the ability of individuals within a species to change in response to stimuli or inputs from the environment, thus the overall vulnerability of a species will ultimately be influenced by its adaptive capacity. Although we have some ability to predict how populations of some species are likely to respond to changing environmental conditions, differences in climate change models can lead to different future conditions with corresponding management challenges. In addition, information about how biodiversity and ecosystems will respond is lacking, although some evidence suggests that species are not adapting quickly enough (Radchuk et al. 2019). Important questions arise of which we rarely have immediate answers: *How is a species' reproductive success impacted as their environmental thresholds are reached? Which species can shift their current distributions in response to a warming climate? What happens to ecosystems when there is species loss or turnover?* These uncertainties can be paralyzing, and the four climate change scenarios modeled here predict different futures that will impact species differently based on their life history. These uncertainties can make it challenging to invest in specific management interventions. However, we do have a high level of confidence in certain climate-induced changes.

The climate is warming with greater warming occurring at higher latitudes. In mountainous areas, different topoclimates within narrow elevational gradients result in high levels of species biodiversity, and mountains support one-quarter of terrestrial biodiversity and contain nearly half of the world's biodiversity hotspots. A warming climate has been linked to increased drought and wildfire and there is a large body of evidence that animals and plants are responding to climate change through shifts in their distribution, phenology (e.g., timing and duration of breeding, migration), and changes in population size.

Most species do not exist within the confines of social-political boundaries in which management actions are implemented (e.g. fig. 2). Therefore, when attempting to address the impacts of climate change on the social-ecological sustainability of resources, it will be prudent to consider both adaptation strategies to maintain current populations and resources, and strategies that anticipate change in the distribution of species both within and outside of National Forest boundaries (i.e., shared governance, Transition 3, fig. 1). Under the WW scenario, a focus on more conventional conservation strategies (fig. 1) may help support the conservation of existing species and habitats within their current boundaries over the near-term. However, under the VHD and VHW scenarios, changes in climate are likely to result in intense pressure on species to shift their current distributions. Therefore, in addition to assessing climatic vulnerabilities, we should anticipate climate-induced changes in species distributions, work across social-political boundaries, and invest in shared learning as we navigate the many uncertainties that stall management action (fig. 1).

Future-Oriented Conservation Approach and Guidelines

Climate change adaptation refers to the process of adjusting strategies to reduce risks to natural and human systems from changes in climate. Many tools have been developed to help managers articulate and evaluate different climate adaptation approaches (Abrahms et al. 2017). These tools are not typically designed to help establish conservation and management goals, which are driven as much by societal values, economic constraints, and political feasibility (Transition 2, fig. 1), as they are by scientific knowledge (Camacho et al. 2010). Instead, they are designed to help assess climate risk based on scientific knowledge. In many cases, our scientific knowledge is lacking due to uncertainty in future conditions, incomplete knowledge of species adaptive capacity, novel species assemblages, and cascading effects on ecosystem function and stability.

Below are some general conservation guidelines to address scientific uncertainties and need for management action under a future-oriented conservation approach to climate adaptation:

1. Change must be anticipated
    a. Expect changes in species distribution, abundance, and phenology.
    b. Minimize barriers to movement (within and across jurisdictions).
    c. Protect trailing-edge populations where appropriate (i.e., resist change), while facilitating population transition to other areas.
2. Recognize societal benefits and values
    a. Identify societal values, economic constraints, and political feasibility as part of a conservation strategy.
    b. Emphasize maintaining ecosystem function and ecosystem services.
3. Governance management
    a. Work across political boundaries to protect ecosystems and the species which rely on them.
    b. Rethink and redefine measures of success (i.e., maintaining species within an ecoregion, or sustaining functional diversity).
4. Invest in and apply ongoing learning
    a. Embrace heterogeneity in thought and practice—protect against loss by supporting more than one possible result.
    b. Refrain from thinking of change as “failure.”
    c. Focus learning and research on reducing uncertainties that hinder management action.

References

Abrahms, B.; DiPietro, D.; Graffis, A.; [et al.]. 2017. Managing biodiversity under climate change: Challenges, frameworks, and tools for adaptation. Biodiversity and Conservation. 26(10): 2277–2293. https://doi.org/10.1007/s10531-017-1362-4

Aplet, G.H.; Cole, D.N. 2012. The trouble with naturalness: Rethinking park and wilderness goals. In: Cole, D.N.; Yung, L., eds. Beyond naturalness: Rethinking park and wilderness stewardship in an era of rapid change. Washington, DC: Island Press: 12–29.

Barton, B.T.; Ives, A.R. 2014. Species interactions and a chain of indirect effects driven by reduced precipitation. Ecology. 95(2): 486–494. https://doi.org/10.1890/13-0044.1

Bellard, C.; Bertelsmeier, C.; Leadley, P.; [et al.]. 2012. Impacts of climate change on the future of biodiversity. Ecology Letters. 15(4): 365–377. https://doi.org/10.1111/j.1461-0248.2011.01736.x

BirdLife International and Handbook of the Birds of the World [Birdlife International]. 2018. Bird species distribution maps of the world. Version 2018.1. Available at https://datazone.birdlife.org/species/requestdis

Camacho, A.E.; Doremus, H.; McLachlan, J.S.; [et al.]. 2010. Reassessing conservation goals in a changing climate. Issues in Science and Technology. 26(4): 21–26. https://issues.org/p_camacho/

Chapman, T.B.; Schoennagel, T.; Veblen, T.T.; [et al.]. 2020. Still standing: Recent patterns of post-fire conifer refugia in ponderosa pine-dominated forests of the Colorado Front Range. PLoS One. 15(1): e0226926. https://doi.org/10.1371/journal.pone.0226926

Cooley, S.; Schoeman, D.; Bopp, L.; [et al.]. 2022. Oceans and coastal ecosystems and their services. In: Pörtner, H.-O.; Roberts, D.C.; Tignor, M.; [et al.], eds. Climate change 2022: Impacts, adaptation and vulnerability. Contribution of working group II to the sixth assessment report of the Intergovernmental Panel on Climate Change. Cambridge, UK, and New York, NY, USA: Cambridge University Press: 379–550. https://www.ipcc.ch/report/ar6/wg2/downloads/report/IPCC_AR6_WGII_Chapter03.pdf [Accessed 24 June 2024]

Davis, K.T.; Dobrowski, S.Z.; Higuera, P.E.; [et al.]. 2019. Wildfires and climate change push low-elevation forests across a critical climate threshold for tree regeneration. Proceedings of the National Academy of Sciences of the United States of America. 116(13): 6193–6198. https://doi.org/10.1073/pnas.1815107116

Fischlin, A.; Midgley, G.F.; Price, J.T.; [et al.]. 2007. Ecosystems, their properties, goods, and services. In: Parry, M.L.; Canziani, O.F.; Palutikof, J.P.; [et al.], eds. Climate change 2007: Impacts, adaptation and vulnerability. Contribution of working group II to the fourth assessment report of the Intergovernmental Panel on Climate Change. Cambridge, UK: Cambridge University Press: 211–272. https://www.ipcc.ch/site/assets/uploads/2018/02/ar4-wg2-chapter4-1.pdf [Accessed 24 June 2024]

Fontaine, J.B.; Kennedy, P.L. 2012. Meta-analysis of avian and small-mammal response to fire severity and fire surrogate treatments in U.S. fire-prone forests. Ecological Applications. 22(5): 1547–1561. https://doi.org/10.1890/12-0009.1

Gaston, K.J.; Blackburn, T.M.; Goldewijk, K.K. 2003. Habitat conversion and global avian biodiversity loss. Proceedings of the Royal Society of London. Series B: Biological Sciences. 270(1521): 1293–1300. https://doi.org/10.1098/rspb.2002.2303

Gill, J.A.; Alves, J.A.; Sutherland, W.J.; [et al.]. 2014. Why is timing of bird migration advancing when individuals are not? Proceedings of the Royal Society B: Biological Sciences. 281(1774): 20132161. https://doi.org/10.1098/rspb.2013.2161

Halofsky, J.S.; Conklin, D.R.; Donato, D.C.; [et al.]. 2018a. Climate change, wildfire, and vegetation shifts in a high-inertia forest landscape: Western Washington, USA. PLoS One. 13(12): e0209490.

Halofsky, J.S.; Donato, D.C.; Franklin, J.F.; [et al.]. 2018b. The nature of the beast: Examining climate adaptation options in forests with stand-replacing fire regimes. Ecosphere. 9(3): e02140. https://doi.org/10.1002/ecs2.2140

Halupka, L.; Halupka, K. 2017. The effect of climate change on the duration of avian breeding seasons: A meta-analysis. Proceedings of the Royal Society B: Biological Sciences. 284(1867): 20171710. https://doi.org/10.1098/rspb.2017.1710

Jetz, W.; Wilcove, D.S.; Dobson, A.P. 2007. Projected impacts of climate and land-use change on the global diversity of birds. PLOS Biology. 5(6): e157. https://doi.org/10.1371/journal.pbio.0050157

McCluney, K.E.; Belnap, J.; Collins, S.L.; [et al.]. 2012. Shifting species interactions in terrestrial dryland ecosystems under altered water availability and climate change. Biological Reviews. 87(3): 563–582. https://doi.org/10.1111/j.1469-185X.2011.00209.x

McNab, W.H.; Cleland, D.T.; Freeouf, J.A.; [et al.] 2007. Description of ecological subregions: sections of the conterminous United States. Gen. Tech. Rep. WO-76B, 76: 1–82. https://doi.org/10.2737/WO-GTR-76B

Møller, A.P.; Rubolini, D.; Lehikoinen, E. 2008. Populations of migratory bird species that did not show a phenological response to climate change are declining. Proceedings of the National Academy of Sciences of the United States of America. 105(42): 16195–16200. https://doi.org/10.1073/pnas.0803825105

Pacifici, M.; Foden, W.B.; Visconti, P.; [et al.]. 2015. Assessing species vulnerability to climate change. Nature Climate Change. 5(3): 215–224.

Pacifici, M.; Visconti, P.; Butchart, S.H.; [et al.] 2017. Species' traits influenced their response to recent climate change. Nature Climate Change. 7(3): 205–208. https://doi.org/10.1038/nclimate3223

Parmesan, C.; Morecroft, M.; Trisurat, Y.; [et al.]. 2022. Terrestrial and freshwater ecosystems and their services. In: Pörtner, H.-O.; Roberts, D.C.; Tignor, M.; [et al.], eds. Climate change 2022: Impacts, adaptation and vulnerability. Contribution of working group II to the sixth assessment report of the Intergovernmental Panel on Climate Change. Cambridge, UK, and New York, NY, USA: Cambridge University Press: 197–377. https://www.ipcc.ch/report/ar6/wg2/downloads/report/IPCC_AR6_WGII_Chapter02.pdf [Accessed 24 June 2024]

Penczykowski, R.M.; Connolly, B.M.; Barton, B.T. 2017. Winter is changing: Trophic interactions under altered snow regimes. Food Webs. 13: 80–91. https://doi.org/10.1016/j.fooweb.2017.02.006

Pereira, H.M.; Leadley, P.W.; Proença, V.; [et al.]. 2010. Scenarios for global biodiversity in the 21st century. Science. 330(6010): 1496–1501. https://doi.org/10.1126/science.1196624

Radchuk, V.; Reed, T.; Teplitsky, C.; [et al.]. 2019. Adaptive responses of animals to climate change are most likely insufficient. Nature Communications. 10(1): 3109. https://doi.org/10.1038/s41467-019-10924-4

Rangwala, I. 2024. Chapter 3: Future climate scenarios for Colorado's Front Range forests. In: Hanberry, B.B.; Foster, A., eds. Colorado Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-GTR-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: **XX-XX** https://doi.org/10.2737/RMRS-GTR-438

Rice, J.R.; Joyce, L.A.; Regan, C.; [et al.]. 2018. Climate change vulnerability assessment of aquatic and terrestrial ecosystems in the U.S. Forest Service Rocky Mountain Region. Gen. Tech. Rep. RMRS-GTR-376. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. https://doi.org/10.2737/RMRS-GTR-376

Rosenberg, K.V.; Dokter, A.M.; Blancher, P.J.; [et al.]. 2019. Decline of the North American avifauna. Science. 366(6461): 120–124. https://doi.org/10.1126/science.aaw1313

Ross, M.V.; Alisauskas, R.T.; Douglas, D.C.; [et al.]. 2017. Decadal declines in avian herbivore reproduction: Density-dependent nutrition and phenological mismatch in the Arctic. Ecology. 98(7): 1869–1883. https://doi.org/10.1002/ecy.1856

Scridel, D.; Brambilla, M.; Martin, K.; [et al.]. 2018. A review and meta-analysis of the effects of climate change on Holarctic mountain and upland bird populations. Ibis. 160(3): 489–515. https://doi.org/10.1111/ibi.12585

Şekercioğlu, Ç.H. 2006. Increasing awareness of avian ecological function. Trends in Ecology & Evolution. 21(8): 464–471. https://doi.org/10.1016/j.tree.2006.05.007

Şekercioğlu, Ç.H.; Daily, G.C.; Ehrlich, P.R. 2004. Ecosystem consequences of bird declines. Proceedings of the National Academy of Sciences of the United States of America. 101(52): 18042–18047.

Siegel, R.; Pyle, P.; Thorne, J.; [et al.]. 2014. Vulnerability of birds to climate change in California's Sierra Nevada. Avian Conservation and Ecology. 9(1). https://doi.org/10.5751/ACE-00658-090107

Smith, M. 2024. Chapter 4: Water resources of the Front Range National Forests. In: Hanberry, B.B.; Foster, A., eds. Colorado Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-GTR-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 57-122. https://doi.org/10.2737/RMRS-GTR-438

Stephens, S.L.; Bigelow, S.W.; Burnett, R.D.; [et al.]. 2014. California spotted owl, songbird, and small mammal responses to landscape fuel treatments. BioScience. 64(10): 893–906. https://doi.org/10.1093/biosci/biu137

Tingley, M.W.; Monahan, W.B.; Beissinger, S.R.; [et al.]. 2009. Birds track their Grinnellian niche through a century of climate change. Proceedings of the National Academy of Sciences of the United States of America. 106(Supplement 2): 19637–19643. https://doi.org/10.1073/pnas.0901562106

USDA Forest Service Manual [FSM] and Forest Service Handbooks [FSH]. 2011. Office of the Chief of the Forest Service. https://www.fs.usda.gov/about-agency/regulations-policies/national-directives [Accessed 24 June 2024]

van Kerkhoff, L.; Munera, C.; Dudley, N.; [et al.]. 2019. Towards future-oriented conservation: Managing protected areas in an era of climate change. Ambio. 48(7): 699–713. https://doi.org/10.1007/s13280-018-1121-0

White, A.M.; Zipkin, E.F.; Manley, P.N.; [et al.]. 2013. Conservation of avian diversity in the Sierra Nevada: Moving beyond a single-species management focus. PLOS ONE. 8(5): e63088. https://doi.org/10.1371/journal.pone.0063088

Weiskopf, S.R.; Rubenstein, M.A.; Crozier, L.G.; [et al.]. 2020. Climate change effects on biodiversity, ecosystems, ecosystem services, and natural resource management in the United States. Science of the Total Environment. 733: 137782. https://doi.org/10.1016/j.scitotenv.2020.137782

Wyborn, C.; van Kerkhoff, L.; Dunlop, M.; [et al.]. 2016. Future oriented conservation: Knowledge governance, uncertainty and learning. Biodiversity and Conservation. 25: 1401–1408. https://doi.org/10.1007/s10531-016-1130-x

Young, B.; Byers, E.; Gravuer, K; [et al.]. 2011. Guidelines for using the NatureServe climate change vulnerability index. Arlington, VA: NatureServe. https://www.natureserve.org/sites/default/files/guidelines_natureserveclimatechangevulnerabilityindex_r2.1_apr2011.pdf [Accessed 24 June 2024]

Appendix A: List of avian species whose distribution overlaps portions of the Pike San Isabel and Arapaho-Roosevelt National Forests

List of avian species whose distribution overlaps portions of the Pike-San Isabel and Arapaho-Roosevelt National Forests (BirdLife International 2018). Species are organized by family and listed alphabetically by scientific name.

Capital letters in columns indicate that the species occurs throughout a National Forest (PSI = Pike-San Isabel, AR = Arapaho-Roosevelt), whereas lowercase letters refer to Ranger Districts (bo = Boulder, cc = Clear Creek, cl = Canyon Lakes, le = Leadville, pa = South Park, pl = South Platte, pp = Pike's Peak, sa = Salida, sc = San Carlos, su = Sulphur).

Ranger Districts that are italicized and in parentheses indicate a species occurs throughout the National Forest except for the Ranger District(s) indicated. Species with smaller ranges or in taxonomic families with fewer species may warrant special consideration.

Family	Common name *(Scientific name)*	Year-round	Breeding	Migratory route	Nonbreeding
Accipitridae	Northern goshawk *(Accipiter gentilis)*	PSI, AR			
	Golden eagle *(Aquila chrysaetos)*	PSI, AR			
	Rough-legged buzzard *(Buteo lagopus)*				PSI, AR
	Ferruginous hawk *(Buteo regalis)*	PSI, AR			
	Northern harrier *(Circus hudsonius)*	PSI, AR			
	Bald eagle *(Haliaeetus leucocephalus)*	PSI, AR			
	Cooper's hawk *(Accipiter cooperii)*	PSI, AR			
	Sharp-shinned hawk *(Accipiter striatus)*	PSI, AR			
	Red-tailed hawk *(Buteo jamaicensis)*	PSI, AR			
	Swainson's hawk *(Buteo swainsoni)*		PSI, AR		
Aegithalidae	Bushtit *(Psaltriparus minimus)*	PSI *(le)*			
Alaudidae	Horned lark *(Eremophila alpestris)*		PSI, AR		
Alcedinidae	Belted kingfisher *(Megaceryle alcyon)*	PSI, AR			
Anatidae	Wood duck *(Aix sponsa)*		AR, le, pa, pl		
	Northern pintail *(Anas acuta)*		cl		pa, pl, pp
	Green-winged teal *(Anas crecca)*	PSI, AR			
	Mallard *(Anas platyrhynchos)*	PSI, AR			
	Snow goose *(Anser caerulescens)*			le, pl, sa, su	
	Ross's goose *(Anser rossii)*			PSI *(pp)*, cc, su	sc
	Lesser scaup *(Aythya affinis)*		cl, sc	PSI, AR	
	Redhead *(Aythya americana)*		PSI, AR		
	Ring-necked duck *(Aythya collaris)*	cl, su		bo, cl, su	PSI *(cl)*, AR
	Canvasback *(Aythya valisineria)*	PSI, cc	AR		
	Canada goose *(Branta canadensis)*	PSI, AR			
	Cackling goose *(Branta hutchinsii)*			PSI-AR *(sc)*	pl, pp, sa, sc
	Bufflehead *(Bucephala albeola)*	su			PSI, AR

Family	Common name *(Scientific name)*	Year-round	Breeding	Migratory route	Nonbreeding
	Common goldeneye *(Bucephala clangula)*				PSI, AR
	Barrow's goldeneye *(Bucephala islandica)*				PSI *(pp)*, AR
	Tundra swan *(Cygnus columbianus)*				
	Hooded merganser *(Lophodytes cucullatus)*				PSI *(sa)*, AR
	American wigeon *(Mareca americana)*	PSI, AR			
	Gadwall *(Mareca strepera)*		PSI, cl, su		
	Common merganser *(Mergus merganser)*	PSI *(pp)*, AR			pa, pl, pp, sc
	Red-breasted merganser *(Mergus serrator)*			PSI, AR	
	Ruddy duck *(Oxyura jamaicensis)*		PSI, AR		
	Northern shoveler *(Spatula clypeata)*		PSI, AR		
	Cinnamon teal *(Spatula cyanoptera)*		PSI, AR		
	Blue-winged teal *(Spatula discors)*		PSI, AR		
Anhingidae	American white pelican *(Pelecanus erythrorhynchos)*		cl, pl, pp, sc	PSI, AR	
Apodidae	White-throated swift *(Aeronautes saxatalis)*		PSI, AR		
	Chimney swift *(Chaetura pelagica)*		AR	sc	
Ardeidae	Great blue heron *(Ardea herodias)*	PSI, AR			
	American bittern *(Botaurus lentiginosus)*		PSI, AR	sc	
	Cattle egret *(Bubulcus ibis)*		PSI, AR		
	Snowy egret *(Egretta thula)*		PSI *(le)*, AR		
	Black-crowned night-heron *(Nycticorax nycticorax)*		PSI, AR		
Bombycillidae	Cedar waxwing *(Bombycilla cedrorum)*	PSI, AR			pa, pp, sc
	Bohemian waxwing *(Bombycilla garrulus)*				PSI, AR
Calcariidae	Lapland longspur *(Calcarius lapponicus)*				PSI, AR
	Chestnut-collared longspur *(Calcarius ornatus)*			PSI, AR	
	Snow bunting *(Plectrophenax nivalis)*				PSI, AR
	McCown's longspur *(Rhynchophanes mccownii)*		bo, cl, su	PSI-AR *(cl)*	

Family	Common name *(Scientific name)*	Year-round	Breeding	Migratory route	Nonbreeding
Caprimulgidae	Common nighthawk *(Chordeiles minor)*		PSI, AR		
	Common poorwill *(Phalaenoptilus nuttallii)*		PSI, AR		
Cardinalidae	Lazuli bunting *(Passerina amoena)*		PSI *(pa, pp)*, AR		
	Blue grosbeak *(Passerina caerulea)*		pa, pp, sc		
	Indigo bunting *(Passerina cyanea)*		PSI		
	Rose-breasted grosbeak *(Pheucticus ludovicianus)*			PSI-AR *(le, sa)*	
	Black-headed grosbeak *(Pheucticus melanocephalus)*		PSI, AR		
	Hepatic tanager *(Piranga hepatica)*		pl, pp, sa, sc		
	Western tanager *(Piranga ludoviciana)*		PSI, AR	pp, sc	
	Dickcissel *(Spiza americana)*		bo, cl, pa, pp		
Cathartidae	Turkey vulture *(Cathartes aura)*		PSI, AR		
	Brown creeper *(Certhia americana)*	PSI, AR			
Charadriidae	Mountain plover *(Charadrius montanus)*		PSI, AR	cl, su	
	Snowy plover *(Charadrius nivosus)*		sa, sc		
	Semipalmated plover *(Charadrius semipalmatus)*			cl	
	Killdeer *(Charadrius vociferus)*	PSI, AR			
Cinclidae	American dipper *(Cinclus mexicanus)*	PSI, AR			
Columbidae	Band-tailed pigeon *(Patagioenas fasciata)*		le, pl, sa, sc		
	Mourning dove *(Zenaida macroura)*	PSI, AR			
Corvidae	California scrub-jay *(Aphelocoma californica)*	PSI *(cl)*, AR			
	American crow *(Corvus brachyrhynchos)*	PSI, AR			
	Common raven *(Corvus corax)*	PSI, AR			
	Blue jay *(Cyanocitta cristata)*	AR, pa, pl, pp			
	Steller's jay *(Cyanocitta stelleri)*				
	Pinyon jay *(Gymnorhinus cyanocephalus)*	bo, cl, le, pl, sa, sc			
	Clark's Nutcracker *(Nucifraga columbiana)*	PSI, AR			

Family	Common name *(Scientific name)*	Year-round	Breeding	Migratory route	Nonbreeding
	Canada jay *(Perisoreus canadensis)*	PSI, AR			
	Black-billed magpie *(Pica hudsonia)*	PSI, AR			
Cuculidae	Yellow-billed cuckoo *(Coccyzus americanus)*		sc		
	Black-billed cuckoo *(Coccyzus erythropthalmus)*		AR, pa, pl, pp	sc	
	Greater roadrunner *(Geococcyx californianus)*	sc			
Emberizidae	Black-throated sparrow *(Amphispiza bilineata)*		pp		
	Lark bunting *(Calamospiza melanocorys)*		cl	PSI, AR	
	Lark sparrow *(Chondestes grammacus)*		PSI, AR		
	Dark-eyed junco *(Junco hyemalis)*	PSI, AR			
	Lincoln's sparrow *(Melospiza lincolnii)*		PSI, AR	pa, pl, pp, sa	
	Song sparrow *(Melospiza melodia)*	PSI, AR			
	Canyon towhee *(Melozone fusca)*	le, pl, sa, sc			
	Savannah sparrow *(Passerculus sandwichensis)*		cl, su	PSI, AR	
	American tree sparrow *(Passerella arborea)*				PSI, AR
	Slate-colored fox sparrow *(Passerella schistacea)*		PSI *(pp, sc), AR (cl)*	PSI *(le, sa), AR (cl)*	
	Cassin's sparrow *(Peucaea cassinii)*		pp, pp, sc		
	Green-tailed towhee *(Pipilo chlorurus)*		PSI, AR		
	Spotted towhee *(Pipilo maculatus)*	sc	6		sc
	Vesper sparrow *(Pooecetes gramineus)*		PSI, AR		
	Brewer's sparrow *(Spizella breweri)*		PSI, AR	sc	
	Clay-colored sparrow *(Spizella pallida)*			PSI, AR	
	Chipping sparrow *(Spizella passerina)*		PSI *(pl)*, AR	cc, pa, pp	
	White-throated sparrow *(Zonotrichia albicollis)*			PSI, AR	bo, cl, sa, sc, su
	White-crowned sparrow *(Zonotrichia leucophrys)*	cc, le, pl, sa, su			PSI *(le)*, AR
	Harris's sparrow *(Zonotrichia querula)*			AR *(cc)*	

Family	Common name *(Scientific name)*	Year-round	Breeding	Migratory route	Nonbreeding
Falconidae	Merlin *(Falco columbarius)*	AR, le, pl, sa			PSI
	Prairie falcon *(Falco mexicanus)*	PSI, AR			
	Peregrine falcon *(Falco peregrinus)*		PSI, AR	cl	
	American kestrel *(Falco sparverius)*	PSI, AR			
Fringillidae	Common redpoll *(Acanthis flammea)*				AR *(cc)*
	Cassin's finch *(Haemorhous cassinii)*	PSI *(pp)*, AR			
	House finch *(Haemorhous mexicanus)*	PSI, AR			
	Evening grosbeak *(Hesperiphona vespertina)*	PSI, AR			
	Black rosy-finch *(Leucosticte atrata)*				PSI *(pp)*, AR
	Brown-capped rosy-finch *(Leucosticte australis)*		PSI *(pp)*, AR		PSI, AR
	Gray-crowned rosy-finch *(Leucosticte tephrocotis)*				PSI *(pp)*, AR
	Red crossbill *(Loxia curvirostra)*	PSI, AR			
	White-winged crossbill *(Loxia leucoptera)*	le			PSI, AR
	Pine grosbeak *(Pinicola enucleator)*	PSI, AR			
	Pine siskin *(Spinus pinus)*	PSI, AR			
	Lesser goldfinch *(Spinus psaltria)*		PSI, AR		
	American goldfinch *(Spinus tristis)*	AR *(cc)*			PSI-AR *(cl)*
Gruidae	Sandhill crane *(Antigone canadensis)*			PSI, AR	
Hirundinidae	Barn swallow *(Hirundo rustica)*		PSI, AR		
	Cliff swallow *(Petrochelidon pyrrhonota)*		PSI, AR		
	Purple martin *(Progne subis)*		le	PSI, AR	
	Bank swallow *(Riparia riparia)*		PSI, AR		
	Northern rough-winged swallow *(Stelgidopteryx serripennis)*		PSI, AR		
	Tree swallow *(Tachycineta bicolor)*		PSI, AR		
	Violet-green swallow *(Tachycineta thalassina)*		PSI, AR		

Family	Common name *(Scientific name)*	Year-round	Breeding	Migratory route	Nonbreeding
Icteridae	Red-winged blackbird *(Agelaius phoeniceus)*	PSI, AR			
	Bobolink *(Dolichonyx oryzivorus)*		PSI, AR	pp, sc	
	Brewer's blackbird *(Euphagus cyanocephalus)*	PSI, AR *(le, sa)*	le, pl, sa, sc, su		
	Bullock's oriole *(Icterus bullockiorum)*		PSI, AR		
	Orchard oriole *(Icterus spurius)*		cl		
	Brown-headed cowbird *(Molothrus ater)*		PSI, AR		
	Great-tailed grackle *(Quiscalus mexicanus)*				
	Common grackle *(Quiscalus quiscula)*		PSI, AR		
	Western meadowlark *(Sturnella neglecta)*	PSI, AR	cl, su		
	Yellow-headed blackbird *(Xanthocephalus xanthocephalus)*		PSI, AR		
Laniidae	Northern shrike *(Lanius borealis)*				PSI, AR
	Loggerhead shrike *(Lanius ludovicianus)*		PSI, AR		
Laridae	Black tern *(Chlidonias niger)*		PSI, AR	sc	
	California gull *(Larus californicus)*			PSI, AR	
	Ring-billed gull *(Larus delawarensis)*		cl	AR *(cc)*	PSI-AR *(cl)*
	Bonaparte's gull *(Larus philadelphia)*			PSI, AR	
	Franklin's gull *(Larus pipixcan)*			PSI, AR	
	American herring gull *(Larus smithsonianus)*			PSI *(pp), AR (bo)*	PSI *(le, sa)*, AR
	Forster's tern *(Sterna forsteri)*		PSI, AR	le, pl, sa, sc	
Mimidae	Gray catbird *(Dumetella carolinensis)*		PSI, AR		
	Northern mockingbird *(Mimus polyglottos)*	PSI, AR			
	Sage thrasher *(Oreoscoptes montanus)*		PSI, AR		
	Brown thrasher *(Toxostoma rufum)*		PSI *(le, sa)*, AR		
Motacillidae	American pipit *(Anthus rubescens)*		PSI, AR	cl, pa, pp, sc	
Odontophoridae	Gambel's quail *(Callipepla gambelii)*	sc			
	Scaled quail *(Callipepla squamata)*	pp, sc			

Family	Common name *(Scientific name)*	Year-round	Breeding	Migratory route	Nonbreeding
	Northern bobwhite *(Colinus virginianus)*	cl			
Pandionidae	Osprey *(Pandion haliaetus)*		PSI *(pp, sc)*, AR	PSI	
Paridae	Juniper titmouse *(Baeolophus ridgwayi)*	PSI *(pa)*			
	Black-capped chickadee *(Poecile atricapillus)*	PSI, AR			
	Mountain chickadee *(Poecile gambeli)*	PSI, AR			
Parulidae	Wilson's warbler *(Cardellina pusilla)*		PSI *(pp)*, AR	PSI *(sa), AR (bo)*	
	MacGillivray's warbler *(Geothlypis tolmiei)*		PSI, AR	pa, pp, sc	
	Common yellowthroat *(Geothlypis trichas)*		PSI, AR		
	Yellow-breasted chat *(Icteria virens)*		PSI, AR		
	Orange-crowned warbler *(Leiothlypis celata)*		cl, cc, le, pl, su	PSI, AR	
	Tennessee warbler *(Leiothlypis peregrina)*			PSI, AR	
	Nashville warbler *(Leiothlypis ruficapilla)*			PSI, cc	
	Virginia's warbler *(Leiothlypis virginiae)*		PSI *(le, sa)*		
	Black-and-white warbler *(Mniotilta varia)*			PSI, AR	
	Northern waterthrush *(Parkesia noveboracensis)*			PSI, AR	
	Ovenbird *(Seiurus aurocapilla)*		pp		
	Yellow-rumped warbler *(Setophaga coronata)*		PSI, AR	cl	
	Grace's warbler *(Setophaga graciae)*		sc		
	Black-throated gray warbler *(Setophaga nigrescens)*		PSI *(pa, pp)*		
	Yellow warbler *(Setophaga petechia)*		PSI, AR		
	American redstart *(Setophaga ruticilla)*		bo, cl, pa, pp	PSI, AR	
	Townsend's warbler *(Setophaga townsendi)*			PSI, AR	
Passeridae	House sparrow *(Passer domesticus)*	PSI, AR			
Phalacrocoracidae	Double-crested comorant *(Nannopterum auritus)*		AR, pa, pl, pp	PSI *(pa)*	
Phasianidae	Gunnison sage-grouse *(Centrocercus minimus)*	sa			
	Greater sage-grouse *(Centrocercus urophasianus)*	PSI *(pp)*, AR			

Family	Common name *(Scientific name)*	Year-round	Breeding	Migratory route	Nonbreeding
	Dusky grouse *(Dendragapus obscurus)*	PSI *(pp, sc)*, AR			
	White-tailed ptarmigan *(Lagopus leucura)*	PSI *(sc)*, AR			
	Wild turkey *(Meleagris gallopavo)*	PSI, AR			
	Ring-necked pheasant *(Phasianus colchicus)*	PSI, AR			
	Sharp-tailed grouse *(Tympanuchus phasianellus)*				
Picidae	Northern flicker *(Colaptes cafer)*	PSI, AR			
	Downy woodpecker *(Dryobates pubescens)*	PSI, AR			
	Hairy woodpecker *(Leuconotopicus villosus)*	PSI, AR			
	Red-headed woodpecker *(Melanerpes erythrocephalus)*		PSI, AR		
	Lewis's woodpecker *(Melanerpes lewis)*	PSI, AR			
	Three-toed woodpecker *(Picoides tridactylus)*	PSI, AR			
	Red-naped sapsucker *(Sphyrapicus nuchalis)*		PSI, AR		
	Williamson's sapsucker *(Sphyrapicus thyroideus)*		PSI, AR		
Podicipedidae	Clark's grebe *(Aechmophorus clarkii)*		PSI, AR		
	Western grebe *(Aechmophorus occidentalis)*		PSI, AR		
	Horned grebe *(Podiceps auritus)*			cl	pl
	Eared grebe *(Podiceps nigricollis)*	pa, pl, pp	PSI *(pp)*, AR		
	Pied-billed grebe *(Podilymbus podiceps)*	PSI, AR			
	Blue-gray gnatcatcher *(Polioptila caerulea)*		sc		
Psittacidae	Carolina parakeet *(Conuropsis carolinensis)*	PSI, AR			
Rallidae	American coot *(Fulica americana)*	cl	PSI, AR		
	Sora *(Porzana carolina)*		PSI, AR		
	Virginia rail *(Rallus limicola)*		PSI, AR		
Recurvirostridae	Black-necked stilt *(Himantopus himantopus mexicanus)*		PSI *(le, sa)*		
	American avocet *(Recurvirostra americana)*		PSI, AR		

Family	Common name *(Scientific name)*	Year-round	Breeding	Migratory route	Nonbreeding
Regulidae	Ruby-crowned kinglet *(Regulus calendula)*		cl, le, sa		
	Golden-crowned kinglet *(Regulus satrapa)*	PSI, AR			
Scolopacidae	Spotted sandpiper *(Actitis macularius)*		PSI, AR		
	Upland sandpiper *(Bartramia longicauda)*			PSI, AR	
	Pectoral sandpiper *(Calidris melanotos)*			PSI, AR	
	Least sandpiper *(Calidris minutilla)*			PSI, AR	
	Semipalmated sandpiper *(Calidris pusilla)*			PSI, AR	
	Wilson's snipe *(Gallinago delicata)*	PSI, AR			
	Long-billed dowitcher *(Limnodromus scolopaceus)*			PSI, AR	
	Long-billed curlew *(Numenius americanus)*		pa, pp, sc		
	Wilson's phalarope *(Steganopus tricolor)*		cl	PSI, AR	
	Lesser yellowlegs *(Tringa flavipes)*			PSI, AR	
	Greater yellowlegs *(Tringa melanoleuca)*			PSI, AR	
	Solitary sandpiper *(Tringa solitaria)*				
Sittidae	Red-breasted nuthatch *(Sitta canadensis)*	PSI, AR			
	White-breasted nuthatch *(Sitta carolinensis)*	PSI, AR			
	Pygmy nuthatch *(Sitta pygmaea)*	PSI, AR			
Strigidae	Northern saw-whet owl *(Aegolius acadicus)*	PSI *(cl)*, AR			bo, cl, pa, pp, sc
	Boreal owl *(Aegolius funereus)*	PSI, AR			
	Short-eared owl *(Asio flammeus)*	AR			PSI, AR *(bo, cl)*
	Long-eared owl *(Asio otus)*	PSI *(pp, sa)*, AR			PSI
	Burrowing owl *(Athene cunicularia)*		PSI, AR		
	Great Horned owl *(Bubo virginianus)*	PSI, AR			
	Northern Pygmy-owl *(Glaucidium californicum)*	PSI, AR			
	Eastern Screech-owl *(Megascops asio)*	cl			
	Western Screech-owl *(Megascops kennicottii)*	le, sa, sc			

Family	Common name *(Scientific name)*	Year-round	Breeding	Migratory route	Nonbreeding
	Flammulated owl *(Psiloscops flammeolus)*		PSI, AR *(su)*		
	Spotted owl *(Strix occidentalis)*				
Sturnidae	European starling *(Sturnus vulgaris)*	PSI, AR			
Threskiornithidae	White-faced ibis *(Plegadis chihi)*		PSI, AR	cl	
Trochilidae	Broad-tailed hummingbird *(Selasphorus platycercus)*		cl, sc, su		
Troglodytidae	Canyon wren *(Catherpes mexicanus)*	PSI, AR			
	Marsh wren *(Cistothorus palustris)*	PSI, AR	cl, su		
	Rock wren *(Salpinctes obsoletus)*	sc	PSI, AR		
	Bewick's wren *(Thryomanes bewickii)*		cl		
	House wren *(Troglodytes aedon)*		PSI, AR		
	Winter wren *(Troglodytes hiemalis)*				PSI *(le, sa)*
Turdidae	Veery *(Catharus fuscescens)*		PSI, AR	sc	
	Hermit thrush *(Catharus guttatus)*		PSI, AR		
	Swainson's thrush *(Catharus swainsoni)*		PSI *(pp)*, AR	PSI, AR	
	Townsend's solitaire *(Myadestes townsendi)*	PSI, AR			
	Mountain bluebird *(Sialia currucoides)*	PSI, AR	cl		
	Western bluebird *(Sialia mexicana)*	sc	PSI, AR *(bo, cl)*		
	American robin *(Turdus migratorius)*	PSI, AR			
Tyrannidae	Olive-sided flycatcher *(Contopus cooperi)*		PSI, AR		
	Western wood-pewee *(Contopus sordidulus)*		PSI, AR		
	Hammond's flycatcher *(Empidonax hammondii)*		PSI, AR	sc	
	Least flycatcher *(Empidonax minimus)*		AR	cl, pa, pp	
	Dusky flycatcher *(Empidonax oberholseri)*		PSI, AR		
	Cordilleran flycatcher *(Empidonax occidentalis)*		PSI, AR		
	Willow flycatcher *(Empidonax traillii)*		PSI, AR		
	Gray flycatcher *(Empidonax wrightii)*		PSI *(pp)*		

Family	Common name *(Scientific name)*	Year-round	Breeding	Migratory route	Nonbreeding
	Ash-throated flycatcher *(Myiarchus cinerascens)*				
	Say's phoebe *(Sayornis saya)*		PSI, AR		
	Eastern kingbird *(Tyrannus tyrannus)*		AR, pa, pl, pp		
	Western kingbird *(Tyrannus verticalis)*		PSI, AR		
Tytonidae	Barn owl *(Tyto alba)*	PSI, AR			
Vireonidae	Cassin's vireo *(Vireo cassinii)*			PSI, AR	
	Warbling vireo *(Vireo gilvus)*		PSI, AR		
	Red-eyed vireo *(Vireo olivaceus)*		AR, pl, pp	pl, sc	
	Plumbeous vireo *(Vireo plumbeus)*		PSI, AR		

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Community Involvement to Increase Monitoring Capacity for Climate-Sensitive Resources

Matthew Fairchild and Brice B. Hanberry

Climate-driven changes in environmental conditions and disturbance patterns have informed predictive models with expected distributional shifts of forest habitat types (Fornwalt et al. 2024) and myriad changes for vulnerable biophysical resources across ecosystems of the Front Range of Colorado (e.g., Colorado Parks and Wildlife 2015; Smith 2024; White 2024). Strategies for managing these forests adaptively in a changing climate are more likely to realize desired outcomes when decisions are based on relevant and current data (see Moir and Block 2001). Although natural resource monitoring programs generating data often require substantial investments of time and labor (Lovett et al. 2007), community science and crowd-sourcing data collection can expand monitoring capacity for climate-sensitive biophysical resources (McKinley et al. 2017). Limited agency staff and funding for routine monitoring and field data collection justifies engagement of interested public groups who can make valuable contributions to resource management (i.e., community science). Community science projects can permit monitoring of critical resources that otherwise would not be possible or extend the spatial and temporal scale of monitoring (McKinley et al. 2017). Managers may find employing community science particularly useful for examining sensitive biophysical resources that may change under altered climate conditions (e.g., Chandler et al. 2017). In addition to increasing capacity to collect data, by recruiting volunteers, managers can better deliver information to the public, connect greater numbers of the public with the mission of the U.S. Department of Agriculture, Forest Service, and increase the public value of national forests, which will likely be crucial to managing forests adaptively in the face of climate change.

Community science has been in existence for decades and successful programs have capitalized on the interests of nature enthusiasts. Beginning in the early 1900s, the Audubon Society began the Christmas Bird Count as a means to understand the diversity and trends of birds within the United States (Butcher et al. 1990; Stewart 1954). Other efforts to engage the public in the science and management of our natural resources have been developed and promoted in more recent times to examine, for example, nesting birds, invasive plant species, pollinators (see Bonney et al. 2009). Some of those efforts are more broadly accessible through the internet (e.g., Gallo and Waitt 2011) and others through more local engagement with specific conservation (e.g., Firehock and West 1995) or community-based forestry groups (e.g., Ballard et al. 2008). Regardless of the platform, interested community members are likely accessible for those managers and scientists willing to invest resources in managing such programs and able to pique the interests of the public.

Monitoring encompasses a broad spectrum of environmental measures used for informing land management decisions and is also used to evaluate management actions (Lovett et al. 2007). The burgeoning number and diversity of community science programs in recent years (MacPhail and Colla 2020) provide ample opportunity

for natural resource managers seeking to build or expand monitoring capacity (see table 1). Community science can aid both managers and scientists to collect far more observations than possible with professional staff, which in turn can be used to improve knowledge of the natural world and develop decision support tools (McKinley et al. 2017). For example, when crowd-sourced observations are coupled with precipitation gauges, the accuracy of models characterizing peak flows on ungauged watersheds was improved across space and time (Starkey et al. 2017). Other hydrology-related programs focus on water quality monitoring at regional or national scales, yet with the advent of relatively low-cost dataloggers there is more opportunity for using volunteer scientists for hydrologic monitoring (Buytaert et al. 2014; Njue et al. 2019). Capitalizing on the enthusiasm of birders, Cornell Lab of Ornithology's eBird program has surpassed over 1 billion bird observations globally (eBird 2021), leading to improved distribution models for numerous migratory bird species (Sullivan et al. 2014), while their backyard Nestwatch program has improved understanding of reproductive biology of birds (e.g., Bailey et al. 2020) and how climate change is altering reproductive success of conservation species (e.g., Callery et al. 2022). Other community science programs have improved early detection of forest insect and disease occurrence, allowing for rapid and targeted management responses (e.g., de Groot et al. 2023; Mair et al. 2016). Several community-based forestry programs employ aspects of crowd-sourcing data collection to adaptively create management prescriptions and monitor outcomes (Ballard et al. 2008), which can link to the monitoring emphasis in the 2012 Forest Service Planning Rule to examine the effects on species diversity and hydrologic function (USDA FS 2012). Through the Collaborative Forest Landscape Restoration Program (CFLRP) of the Forest Service, project teams often rely on community-based monitoring of restoration outcomes (Schultz et al. 2012), one of which on the Uncompahgre National Forest uses community scientists to help collect data used to evaluate wildfire risk reduction (McKinley et al. 2017).

Within the Front Range mountains of Colorado, one ongoing community science project occurring on the Arapaho-Roosevelt National Forests is part of a global monitoring effort on ungauged streams. Trained volunteer scientists use the Stream Tracker application (Stream Tracker 2021) on smart phone devices to collect presence/absence data on streamflow to better understand hydrology of seasonally, flowing nonperennial streams (fig. 1). Volunteers report on streams encountered along popular hiking trails as well as those streams infrequently examined along forest road networks. Since 2017, community scientists have reported stream flow data across 222 sites in numerous intermittent streams within the Arapaho-Roosevelt National Forests. Since the project's inception, more than 100 volunteers have recorded well over 5,000 streamflow observations globally (Stream Tracker 2021). Such observations have improved insight into the seasonality of streamflow under current climate conditions, supported the development of streamflow duration models for ungauged streams, and helped volunteers to learn how their monitoring contributes to broader scientific understanding and aids management decisions.

Table 1—Examples of community science monitoring programs that operate at a broad national scale and regionally within the central Rocky Mountains.

National monitoring programs	Fields of study	Program links
eBird	Bird distribution Phenology of birds	https://ebird.org/home
iNaturalist	Plants, invertebrates, birds, reptiles and amphibians, fish, mammals	https://www.inaturalist.org/
The USA National Phenology Network	Phenology of plants and animals	https://www.usanpn.org/
Monarch Larva Monitoring Project (MLMP)	Occurrence in Monarch butterflies	https://mlmp.org/
Nature's Notebook	Track seasonal changes in plants and animals	https://www.usanpn.org/natures_notebook
NABat	Occurrence of bats in space and time	https://sciencebase.usgs.gov/nabat/
Project Budburst	Seasonal changes in plant life cycles	https://budburst.org/

Regional monitoring programs	Fields of study	Program Links
Stream Tracker	Hydrology of ungauged streams	https://www.streamtracker.org/
Riverwatch of Colorado	Water quality	https://coloradoriverwatch.org/
Mountain Studies Institute	Noxious weed mapping Bighorn sheep monitoring PikaNet	https://www.mountainstudies.org
University of Wyoming Biodiversity Institute	Mullen Wildfire Community Science Initiative Rocky Mountain Amphibian Project Wyoming BioBlitz	https://www.wyomingbiodiversity.org/index.php/community-science
Colorado Pika Project	Records observations of Pika	https://pikapartners.org/
CoCoRahs (Community Collaborative Rain, Hail, and Snow Network)	Observations of precipitation in rural and remote areas of Colorado	https://www.cocorahs.org/



Figure 1a-b—Photographs of community scientists performing field work to aid Forest Service resource managers: (A) Trout Unlimited volunteer locates and retrieves stream temperature datalogger (photograph by Tom Lindsey, Trout Unlimited); (B) Stream Tracker volunteers make repeat observations (dry versus wet) of streamflow presence in ungauged stream (courtesy photos by Kira Puntenney-Desmond, Colorado State University).

Other community science efforts on national forests in Colorado include assisting in evaluation of land-use impacts, animal abundance, and physical habitat conditions. In southwestern Colorado, range condition assessments performed by local ranchers help to add more data points and bolster information from the limited number of surveys that agency staff are able to complete (USDA Forest Service 2021). Using the U.S. Geologic Survey North American Bat Monitoring (NABat) program, volunteer scientists are monitoring roost sites on the Pike and Routt National Forests, which has helped document declines in bat populations in other regions of the western United States (Rodhouse et al. 2019) and helps multiple agencies acquire bat monitoring data needed for better managing conservation challenges facing imperiled bat species across North America (Reichert and Soileau 2020). In return, volunteers learn about the ecological and economic importance of bats and contribute to improving management. Within alpine ecosystems, interested public help the White River National Forest and agency partners monitor American pika abundance and occupancy due to the species' importance in the ecosystem and vulnerability to climate change outcomes (Front Range Pika Project 2021). Trout Unlimited volunteers have helped staff on the Arapaho-Roosevelt National Forests to deploy and retrieve stream temperature dataloggers (see fig. 1), resulting in a substantial increase in the number of streams with year-round temperature monitoring, which will ultimately help managers evaluate where stream temperatures are warming in response to climate change and develop sound conservation approaches for climate-sensitive aquatic species (e.g., cutthroat trout *Oncorhynchus clarki* subspecies, Susan's purse-making caddisfly *Ochrotrichia susanae*).

Community science programs can provide information over broader geographic and longer time scales than most land management monitoring programs (McKinley et al. 2017). The advent of mobile computing and utility-based applications available on smartphones has given rise to several platforms capable of streamlining data workflows and, thus, supporting community science initiatives (Newman et al. 2012), many of which have assembled data sufficient for broad-scale assessments that show spatial and temporal shifts in species distributions at regional scales (e.g., Parmesan et al 1999; Sullivan et al. 2014). Whereas some community science projects have developed specific applications (e.g., eBird 2021; StreamTracker 2021), other data collection platforms, such as Epicollect (2021) and Survey123 (2021), are highly customizable to address a range of community science topics and facilitate easy deployment of trained volunteers. Once useful community science projects are developed and adapted, smartphone applications such as these are helpful in managing data and allowing participants to see their observations and track progress.

Even lack of time, funds, and expertise for taxonomic identification of diverse or rare species are challenges that may be particularly amenable to community science projects in conjunction with species identification applications (e.g., Ožana et al. 2019; Sullivan et al. 2014). The iNaturalist application (2021) applies an algorithm to suggest species identifications of photographs to assist with identification, which is followed by

confirmation from trained scientists to obtain research grade data. Public engagement by collecting photo observations in National Forests followed by uploading to iNaturalist may allow for development of species lists and distributions. One such effort used iNaturalist on Forest Service lands in Arizona to examine observational trends in 34 plant species in space and time, identifying five plant species that might need improved conservation on Federal lands (Souther et al. 2021). Similar efforts may empower community members to become familiar with the diversity species on National Forests, developing strong interests in certain taxonomic groups and contributing to increasing knowledge about rare species.

Developing and managing community science projects requires a commitment of resources and time in the form of training volunteers, developing platforms and organization structures, and real costs to agencies and organizations; however, many successful community projects report positive returns on invested agency resources. The European-developed Freshwater Watch (2021), which aims to compare aquatic ecosystem dynamics (e.g., water quality, biodiversity) across a range of land-use impacts, returns on average 6 hours of data collected by volunteer scientists per hour of instruction and engagement by lead agency scientists and managers (Thornhill et al. 2016). In the case of stream temperature monitoring on the Arapaho-Roosevelt National Forests, a modest amount of training and direction (e.g., 7 days per year) provided from Forest Service biologists to volunteer scientists who steward datalogger networks yields more than a doubling in the density and extent of monitoring sites (AR-NFs, 1996-2024, unpublished data) and a cumulative equivalent of more than 1,000 hours of labor. For web-enabled programs (e.g., Stream Tracker), where simplified streamflow presence or absence data can be collected on a smartphone application without specialized equipment and self-guided training for volunteers, the return on investment in training volunteer scientists can yield high rates of return as volunteers contribute data across time and space once they become familiar with recording observations in the application. However, such programs often work best and generate the most accessible data repositories when sufficient paid staff are available to manage incoming data (MacPhail and Colla 2020) and monitoring programs are built in collaboration with community groups (Conrad and Hilchey 2011).

Well-managed community science programs can provide large volumes of data for monitoring changing environmental conditions on national forests and where those changes are occurring rapidly or are muted, which will likely enable managers to prioritize actions for their most critical of resource concerns in a more informed manner. Although initiating and overseeing a community science project does take time, the engagement can be mutually beneficial. Volunteer scientists can provide much needed data, particularly for climate-sensitive biophysical resources, while experiencing new activities that are important for managing natural resources. Engaging community members in management can increase public awareness of resource management issues and inspire members of the public to take action in

meaningful and beneficial ways, helping to create a culture of shared stewardship, and connecting people to the land (MacPhail and Colla 2020). Community science not only can help forest managers improve data collection across spatiotemporal scales and make more informed conservation decisions, but also serves to engage sectors of the public in those decisions and foster public engagement (Aceves-Bueno et al. 2015; McKinley et al. 2017). Further, ample reviews of effective community science programs and existing datasets (e.g., Buytaert et al. 2014; Conrad and Hilchey 2011; de Groot et al. 2023; MacPhail and Colla 2020; Njue et al. 2019; Silvertown 2009) are available to help national forest managers develop community science programs and acquire crowd-sourced information instructive for managing climate-vulnerable biophysical resources. More resources about using public participation to support management needs are available at the USDA Forest Service Citizen Science website (USDA FS 2021).

References

Aceves-Bueno, E.; Adeleye, A.S.; Bradley D.; [et al.]. 2015. Citizen science as an approach for overcoming insufficient monitoring and inadequate stakeholder buy-in in adaptive management: Criteria and evidence. Ecosystems. 18: 493–506. https://doi.org/10.1007/s10021-015-9842-4

AR-NFs. 1996-2024. Stream temperature data. Unpublished data on file at Fort Collins, Colorado: AR-NFs, Supervisor's Office, Biological and Physical Resources.

Ballard, H. L.; Fernandez-Gimenez, M.E.; Sturtevant, V.E. 2008. Integration of local ecological knowledge and conventional science: A study of seven community-based forestry organizations in the USA. Ecology and Society. 13(2): 37. http://www.ecologyandsociety.org/vol13/iss2/art37/

Bonney, R.; Ballard, H.; Jordan, R.; [et al.]. 2009. Public participation in scientific research: Defining the field and assessing its potential for informal science education. A CAISE Inquiry Group Report. Washington, DC: Center for Advancement of Informal Science Education (CAISE): 58. https://files.eric.ed.gov/fulltext/ED519688.pdf [Accessed 24 June 2024]

Butcher, G.S.; Fuller, M.R.; McAllister, L.S.; [et al.]. 1990. An evaluation of the Christmas Bird Count for monitoring population trends of selected species. Wildlife Society Bulletin. 18(2): 129–134. https://www.jstor.org/stable/3782126

Buytaert, W.; Baez, S.; Bustamante, M.; [et al.]. 2014. Web-based environmental simulation: Bridging the gap between scientific modeling and decision-making. Environmental Science and Technology. 46: 1971–1976. https://doi.org/10.1021/es2031278

Callery, K.R.; Schulwitz, S.A.; Hunt, A.R.; [et al.]. 2022. Phenology effects on productivity and hatching-asynchrony of American kestrels (*Falco sparverius*) across a continent. Global Ecology and Conservation. 36: e02124. https://doi.org/10.1016/j.gecco.2022.e02124

Chandler, M.; See, L.; Copas, K.; [et al.]. 2017. Contribution of citizen science towards international biodiversity monitoring. Biological Conservation. 213: 280–294. https://doi.org/10.1016/j.biocon.2016.09.004

Colorado Parks and Wildlife. 2015. State wildlife action plan: A strategy for conserving wildlife in Colorado. Denver, CO: State of Colorado Department of Natural Resources. https://cpw.widen.net/s/gnjwdl87fn/co_swap_fullversion [Accessed 4 March 2022]

Conrad, C.C.; Hilchey, K.G. 2011. A review of citizen science and community-based environmental monitoring: Issues and opportunities. Environmental Monitoring and Assessment. 176: 273–291. https://doi.org/10.1007/s10661-010-1582-5

de Groot, M.; Pocock, M.J.O.; Bonte, J.; [et al.]. 2023. Citizen science and monitoring forest pests: A beneficial alliance? Current Forestry Reports. 9:15–32. https://doi.org/10.1007/s40725-022-00176-9

eBird. 2021. eBird. https://ebird.org/home [Accessed 1 March 2021]

Epicollect. 2021. Free and easy-to-use mobile data-gathering platform. https://five.epicollect.net/ [Accessed 14 January 2021]

Firehock, K.; West, J. 1995. A brief history of volunteer biological water monitoring using macroinvertebrates. Journal of the North American Benthological Society. 14 (1): 197–202. https://doi.org/10.2307/1467734

Fornwalt, P.J.; Wion, A.P.; Schoettle, A.W.; [et al.]. 2024. Chapter 5: Vulnerability of major Colorado Front Range tree species to climate change. In: Hanberry, B.B., ed. Colorado Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-GTR-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. p. 123-243. https://doi.org/10.2737/RMRS-GTR-438

Freshwater Watch. 2021. Freshwater Watch, Earthwatch Europe. https://earthwatch.org.uk/freshwater/ [Accessed 9 March 2021]

Front Range Pika Project. 2021. Front Range pika project. https://pikapartners.org/ [Accessed 14 January 2021]

Gallo, T.; Waitt, D. 2011. Creating a successful citizen science model to detect and report invasive species. BioScience. 61(6): 459–465. https://doi.org/10.1525/bio.2011.61.6.8

iNaturalist. 2021. How it works. Accessed at: https://www.inaturalist.org/ [Accessed 14 January 2021]

Lovett, G.M.; Burns, D.A.; Driscoll, C.T.; [et al.]. 2007. Who needs environmental monitoring? Frontiers in Ecology and Environment. 5(5): 253–260.

MacPhail, V.J.; Colla, S.R. 2020. Power of the people: A review of citizen science programs for conservation. Biological Conservation. 249: 108739.

Mair, L.; Harrison, P.J.; Jönsson, M.; [et al.]. 2016. Evaluating citizen science data for forecasting species responses to National Forest management. Ecology and Evolution. 7: 368–378. https://doi.org/10.1002/ece3.2601

McKinley, D.C.; Miller-Rushing, A.J.; Ballard, H.L.; [et al.]. 2017. Citizen science can improve conservation science, natural resource management, and environmental protection. Biological Conservation. 208: 15–28. https://doi.org/10.1016/j.biocon.2016.05.015

Moir, W.H.; Block, W.M. 2001. Adaptive management on public lands in the United States: Commitment or rhetoric? Environmental Management. 28(2): 141–148. https://doi.org/10.1007/s002670010213

Njue, N.; Stenfert Kroese, J.; Gräf, J.; [et al.]. 2019. Citizen science in hydrological monitoring and ecosystem services management: State of the art and future prospects. Science of the Total Environment. 693: 133531. https://doi.org/10.1016/j.scitotenv.2019.07.337

Newman, G.; Wiggins, A.; Crall, A.; [et al.]. 2012. The future of citizen science: Emerging technologies and shifting paradigms. Frontiers in Ecology and Environment. 10(6): 298–304. https://doi.org/10.1890/110294

Ožana, S.; Burda, M.; Hykel, M.; [et al.]. 2019. Dragonfly Hunter CZ: Mobile application for biological species recognition in citizen science. PLoS One. 14(1): e0210370. https://doi.org/10.1371/journal.pone.0210370.

Parmesan, C.; Ryrholm. N.; Stefanescu, C.; [et al.]. 1999. Poleward shifts in geographical ranges of butterfly species associated with regional warming. Nature. 399: 579–583. https://doi.org/10.1038/21181

Reichert, B.E.; Soileau, S.C. 2020. U.S. Geological Survey science in support of the North American Bat Monitoring Program (NABat): U.S. Geological Survey Fact Sheet 2020–3008. https://doi.org/10.3133/fs20203008

Rodhouse, T.J.; Rodríguez, R.M.; Banner, K.M.; [et al.]. 2019 Evidence of region-wide bat population decline from long-term monitoring and Bayesian occupancy models with empirically informed priors. Ecology and Evolution. 9: 11078–11088. https://doi.org/10.1002/ece3.5612

Schultz, C.A.; Jedd, T.; Beam, R.D. 2012. The collaborative forest landscape restoration program: A history and overview of the first projects. Journal of Forestry. 110(7): 381–391. https://doi.org/10.5849/jof.11-082

Silvertown, J. 2009. A new dawn for citizen science. Trends in Ecology and Evolution. 24:467–71. https://doi.org/10.1016/j.tree.2009.03.017

Smith, M. 2024. Chapter 4: Water resources of the Front Range National Forests. In: Hanberry, B.B., ed. Colorado Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-GTR-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. p. 57-122. https://doi.org/10.2737/RMRS-GTR-438

Souther, S.; Randall, V.; Lyndon, N. 2021. The use of citizen science to achieve multivariate management goals on public lands. Diversity. 3: 293. https://doi.org/10.3390/d13070293

Starkey, E.; Parkin, G.; Birkinshaw, S.; [et al.]. 2017 Demonstrating the value of community-based ("citizen science") observations for catchment modelling and characterisation. Journal of Hydrology. 548: 801–817. https://doi.org/10.1016/j.jhydrol.2017.03.019

Stewart, P.A. 1954. The value of the Christmas Bird Counts. The Wilson Bulletin. 66(3): 184–195. https://www.jstor.org/stable/4158325

Stream Tracker. 2021. Stream Tracker. https://www.streamtracker.org/ [Accessed 14 January 2021]

Sullivan, B.L.; Aycrigg, J.L.; Barry, J.H.; [et al.]. 2014. The eBird enterprise: An integrated approach to development and application of citizen science. Biological Conservation. 169: 31–40. https://doi.org/10.1016/j.biocon.2013.11.003

Survey123. 2021. ArcGIS Survey123. https://survey123.arcgis.com [Accessed 14 January 2021]

Thornhill, I.; Loiselle, S.; Lind, K.; [et al.]. 2016. The citizen science opportunity for researchers and agencies. BioScience. 66(9): 720–721. https://doi.org/10.1093/biosci/biw089

United States Department of Agriculture Forest Service [USDA FS]. 2012. National Forest System land management planning, under 36 CFR Part 219. Federal Register 77: 21260-21276 [April 9, 2012]. https://www.federalregister.gov/documents/2012/04/09/2012-7502/national-forest-system-land-management-planning [Accessed 24 June 2024]

United States Department of Agriculture Forest Service [USDA FS]. 2021. Citizen science. https://www.fs.usda.gov/working-with-us/citizen-science [Accessed 14 January 2021]

White, A. 2024. Chapter 6. Future-oriented wildlife management in the Pike-San Isabel and Arapaho-Roosevelt National Forests under a warming climate: Anticipating change and facing uncertainties. In: Hanberry, B.B., ed. Colorado Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-GTR-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. p. 245-279. https://doi.org/10.2737/RMRS-GTR-438

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Insects and Diseases and Climate Change in Colorado Front Range Forests

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Introduction

Insects and diseases are major agents of change in forest ecosystems (Costello et al. 1995; Raffa et al. 2008). Their activities can result in scattered to extensive tree mortality across landscapes (Lundquist and Negrón 2000; Raffa et al. 2008). As natural components of forests, native insects, parasites, and pathogens contribute to shape forest structure and composition, contribute to nutrient cycling, and create habitat for wildlife species among other services (Negrón and Cain 2019). Innumerable species of insects, fungi, and other forms of life are part of the biological diversity of forests. There is a general consensus that abiotic disturbances such as fires and windstorms and biotic disturbances such as insects and diseases will increase in frequency and intensity as climate change manifests. Yet, there are still unresolved knowledge gaps, particularly on the effects of insect-associated organisms and interactions with their hosts (Jactel et al. 2019; Seidl et al. 2017). In this chapter, an overview of what is known about the general effects of a warming climate on pathogens and insects are presented followed by a general description of the most relevant insects and pathogens in the Colorado Front Range. Lastly, comments on how a warming climate may affect these organisms are offered.

General Effects of Climate Change on Forest Insects and Pathogens

Insects are poikilothermic organisms, meaning that they are not able to regulate body temperature and are, therefore, directly influenced by temperatures in their environment. Insects develop faster with increasing temperatures up to an optimal temperature with higher temperatures negatively impacting development. Insects exhibit differential life cycles and strategies to utilize their hosts and as a result effects of climate change on insects may vary in different ecosystems, habitats, seasons, and bioclimatic regions, among other factors.

In general, the literature supports the idea of increased frequency and intensity of forest insect activity under climate change. The predicted changes to insect presence and abundance under climate change may have the following effects: (1) potentially a higher number of generations and higher survival of overwintering stages with warmer winter temperatures; (2) weakened tree resistance mechanisms resulting from increased stress such as droughts; (3) increased breeding habitat such as blowdowns from increased storm activity; and (4) changes in food quality associated with elevated atmospheric carbon dioxide, particularly for defoliators (Jactel et al. 2019). However, a closer look reveals the complexity of these processes and that many other aspects of the interaction between hosts and insects under warming temperatures need to be considered. For example, increased temperatures can also increase mortality of some insects, and result in smaller insects with lower dispersal ability. A broader temperature variability can result in higher mortality. For example, extreme cold spells that happen

before an insect's supercooling period can kill overwintering bark beetle brood (Dooley et al. 2015). The effects of climate change on natural insect enemies such as parasitoids and predators remain largely unknown. In sum, a differential response by different insect species and guilds to climate change, novel interactions between insects and their hosts as they exhibit their own response, and changes in the frequency and severity of abiotic agents such as fires and windstorms make predictions about their individual responses challenging (see Jactel et al. 2019; Pureswaran et al. 2018 and references therein).

Climate change impacts on bark beetles and defoliators will affect range expansion or contractions (changes in geographical distribution), insect abundance, and outbreak patterns (Pureswaran et al. 2018). For example, range expansion of mountain pine beetle (*Dendroctonus ponderosae*) farther north in British Columbia and east into Alberta is well-documented (Cullingham et al. 2011; Safranyik et al. 2010). However, the occurrence of range expansion is not as well documented with defoliators. The processionary moth (*Thaumetopoea pityocampa*, not in Colorado) has shown both positive and negative behaviors in terms of range expansion (Battisti et al. 2006; Robinet et al. 2014). In terms of insect abundance, increased numbers could result in more intense outbreaks. For instance, the spruce beetle (*Dendroctonus rufipennis*) can transition from a 2-year life cycle to a 1-year life cycle under warm conditions providing their populations with the ability to rapidly increase in numbers (Hansen et al. 2001; Werner and Holsten 1985; Werner et al. 2006). In terms of outbreak patterns, the western spruce budworm (*Choristoneura occidentalis*) outbreaks are becoming more intense and frequent (Swetnam and Lynch 1989), and bark beetle outbreaks are expected to behave similarly (Seidl et al. 2017).

Plant pathogens, their hosts, and their interactions are also strongly influenced by environmental conditions; therefore, they will also be affected by climate change. Some effects will include changes in pathogen growth, rates of reproduction and infection, as well as variations in plant-host susceptibility. As a result of warming temperatures, changes in host tree distributions are expected, primarily contracting; however, migrations to areas with more suitable conditions are expected. Pathogens affecting drought-stressed hosts may increase where precipitation levels decline and temperatures increase. In addition, their life cycles may be affected; for example, fungal spore release and the life cycle of insect vectors may be affected (see Sturrock et al. 2011 and citations therein).

Ecology of Prevalent Forest Insect and Diseases on the Colorado Front Range

Important insects that interact with forests in the Colorado Front Range include various species of bark beetles such as the mountain pine beetle, spruce beetle, Douglas-fir beetle (*Dendroctonus pseudotsugae*), and the western balsam bark beetle (*Dryocoetes confusus*) (table 1). Notable defoliators include the western spruce budworm and the Douglas-fir tussock moth (*Orgyia pseudotsugata*). Although forest pathogens do occur in the Colorado Front Range, in general their impacts are less. The most important, perhaps, is dwarf mistletoes that affect several species of pines. White pine blister rust, an introduced disease, has become locally important causing tree mortality, reducing regeneration, and increasing susceptibility to mountain pine beetle (Cleaver et al. 2015).

Table 1—Forest insects and diseases acting as important disturbance agents in the Colorado Front Range. Modified from the Field Guide to Insects & Insects of the Rocky Mountain Region, RMRS-GTR-241 (Rocky Mountain Region, Forest Health Protection 2010).

Disturbance	Name	Primary host(s)
Bark Beetles	Mountain pine beetle	Ponderosa pine, Lodgepole pine, Limber pine
	Spruce beetle	Engelmann spruce, Blue spruce
	Douglas-fir beetle	Douglas-fir
	Western balsam bark beetle	Subalpine fir
	Pinyon ips	Pinyon pine
Defoliators	Western spruce budworm	Douglas-fir, Engelmann spruce, Blue spruce, White fir, Subalpine fir
	Douglas-fir tussock moth	Douglas-fir
Diseases	Mistletoes (parasitic plants)	Lodgepole pine, Limber pine, Pinyon pine, Douglas-fir, Bristlecone pine, Ponderosa pine
	Armillaria root disease	All common tree species and in all major forest types in the Rocky Mountain Region
	White pine blister rust	Limber pine, Bristlecone pine
	Sudden aspen decline / Cytospora	Aspen

Bark Beetles

Mountain Pine Beetle (Dendroctonus ponderosae)

This bark beetle is widely distributed across the western U.S. conifer forests. It attacks and kills about 15 species of pines but most of its activity is in ponderosa (*Pinus ponderosa*) and lodgepole pine (*Pinus contorta*) forests as these are the most abundant hosts in the Colorado Front Range. In the Colorado Front Range, the insect has a 1-year life cycle with emergence and dispersal occurring from late July through mid-August (McCambridge 1964; Negrón 2019; Tishmack et al. 2005; West et al. 2016). Mountain pine beetle prefers dense stands of larger-diameter trees. Data from the Colorado Front Range indicate that in ponderosa pine stands, the insect prefers stands with a ponderosa pine basal area > 74 ft^2/ac and trees with a d.b.h. > 7 inches (Negrón and Popp 2004). Data from the Sulphur Ranger District (RD) of the Arapaho-Roosevelt National Forests (NFs) indicate that in lodgepole pine forests, stands with a basal area > 59 ft^2/ac and mean d.b.h. of 7 inches are more likely to be infested. Increased mortality levels occur when lodgepole pine basal area is > 106 ft^2/ac (Negrón and Klutsch 2017). These levels are likely applicable throughout the Pike-San Isabel and Arapaho-Roosevelt National Forests (PSI-AR).

An extensive outbreak affecting ponderosa pine forests in the late 1970s affected about 2 million acres from the Colorado-Wyoming State line south to the San Carlos RD of the Pike-San Isabel NFs (McCambridge et al. 1982). A smaller outbreak occurred in lodgepole pine in the Sulphur RD of the Arapaho-Roosevelt NFs in the 1980s (Peltz and Smith 2012). A recent extensive mountain pine beetle outbreak occurred from the late 1990s to about the mid-2010s and extended from the southern Rocky Mountains north to British Columbia in Canada. In Colorado, 3.4 million acres were affected with mortality occurring primarily in lodgepole pine forests. Mortality was also observed to a lesser extent in ponderosa pine, limber pine (*Pinus flexilis*), and bristlecone pine (*Pinus aristata*). Klutsch et al. (2009), working in the Sulphur RD of the Arapaho-Roosevelt NFs, indicated mortality levels ranging from little to complete mortality in stands. Across the landscape, tree density and basal area were reduced by 62 percent and 71 percent, respectively, in infested stands and mean tree diameter was reduced by 53 percent.

Spruce Beetle (Dendroctonus rufipennis)

The geographic range of the spruce beetle is transcontinental from Alaska east to New Foundland, parts of the northeastern United States and the southern Rockies south to Arizona and New Mexico. All species of spruce are utilized as hosts, but in Colorado, Engelmann spruce (*Picea engelmannii*) is its primary host with some attacks also occurring in Colorado blue spruce (*Picea pungens*). It has also attacked and caused mortality to lodgepole pine (Mercado 2020 and references therein). The insect has primarily a 2-year life cycle but under warm conditions can complete its life cycle in one year. This has been the case during the extensive outbreaks in Alaska (Werner and Holsten 1985; Werner et al. 2006). Outbreaks of the insect are often associated with blown-down trees where populations can increase rapidly. A major outbreak in

Colorado occurred in the 1940s in the White River and Grand Mesa National Forests after a blowdown in 1939 (Massey and Wygant 1954). Another extensive outbreak occurred in the early 2000s after a large blowdown in 1997 in the Hanhs' Peak-Bear's Ears Ranger District of the Routt National Forest. Using tree-ring analysis, Veblen et al. (1991) suggested that another major outbreak occurred in western Colorado in the 1850s. According to Schmid and Frye (1977), in Colorado the most susceptible stands to spruce beetles are those in drainage bottoms with mean spruce d.b.h. > 16 inches, a live spruce component > 60 percent, and with basal areas exceeding 150 ft^2/ac. Extensive mortality occurred in the spruce-fir forests of the Canyon Lakes Ranger District since about the 2000s, killing most large-diameter trees.

Douglas-fir Beetle (Dendroctonus pseudotsugae)

This species of bark beetle uses Douglas-fir (*Pseudotsuga menziesii*) as its only host in Colorado. Its primary dispersal flight occurs towards the end of June (Negrón et al. 2011) and overwinters in the larval stage. Outbreak populations of Douglas-fir beetle can increase to epidemic levels when a precipitating factor occurs such as the availability of downed trees and trees with nonlethal fire injury. In Colorado, the primary precipitating factor has been defoliation events from western spruce budworm or the Douglas-fir tussock moth (Cole et al. 2022; Lessard and Schmid 1990; Negrón 1998; Negrón et al. 2014). Defoliation results in reduced carbohydrates compromising tree defenses against bark beetles facilitating successful beetle colonization (Webb 1981; Webb and Karchesy 1977; Wright et al. 1979). The Douglas-fir beetle also prefers large-diameter trees. The Douglas-fir beetle prefers stands with a high percentage of the basal area represented by Douglas-fir, high tree densities, and poor growth during the last 5 years prior to attack. Mortality levels are influenced by the initial host basal area (Negrón 1998).

Western Balsam Bark Beetle (Dryocoetes confusus)

Although not adequately studied, over the past three decades, the western balsam bark beetle has caused high tree mortality of its primary host in Colorado, which is subalpine fir (*Abies lasiocarpa*). The only study on the biology of the insect in Colorado examined its life cycle, flight periodicity, and attack patterns in downed trees. The insect completes its life cycle in 1 year, overwintering in the larval stage, and emerges and disperses in mid-July. Preference for attacks is on the underside of downed trees, which is where populations usually increase (Negrón and Popp 2009).

Pinyon Ips (Ips confusus)

Pinyon ips is the primary bark beetle associated with pinyon pine (*Pinus edulis*) in Colorado and the southwest. Outbreaks occur during drought events. Although small outbreaks occur as a result of downed trees from logging operations, two major outbreaks across the Southwest, including Colorado, have coincided with major extensive severe droughts (Breshears et al. 2005; Shaw et al. 2005). Little is known about the basic biology of this insect. It has multiple generations per year, depending on geographic locations. The insect attacks all sizes of trees, particularly those with

dwarf mistletoe infections and in denser stands (Negrón and Wilson 2003). In general, pinyon ips beetles are favored by dry and warm climatic conditions. In addition, combined with an increased likelihood of pinyon ips infestation associated with higher pinyon dwarf mistletoe (*Arceuthobium divaricatum*) infections may result in pinyon ips populations increasing in relevance under climate change.

Defoliating Lepidoptera

Western Spruce Budworm (Choristoneura occidentalis)

Western spruce budworm is a major defoliator in western conifer forests in North America that utilizes various tree species as hosts. In Colorado, the preferred species are Douglas-fir and white fir (*Abies concolor*) and, to a lesser extent, subalpine fir and Engelmann spruce. Defoliation from the insect can result in reduced growth and height, top-kill, and mortality among other effects (Carlson et al. 1983; MacLean 1985). The insect has one generation per year with adults active in July–August. Egg laying takes place primarily on the underside of needles. The insect overwinters as a first instar larvae and the following spring begin feeding in new buds and young needles. Stand conditions preferred by the insect are multi-storied stands with high host tree densities. These conditions provide ample food and facilitate dispersal (Beckwith and Burnell 1982; Fellin et al. 1983; Hadley and Veblen 1993; Johnson and Denton 1975; Maclauchlan and Brooks 2009). Western spruce budworm outbreaks can trigger Douglas-fir beetle eruptions when larger trees are defoliated (Lessard and Schmid 1990; Negrón 1998).

Swetnam and Lynch (1989), using tree-ring analysis, estimated that about nine outbreaks of western spruce budworm have occurred between 1700 and 1983 in the Colorado Front Range. Reduced tree growth rates resulting from defoliation lasted for an average of 13 years while the return interval between outbreaks averaged 35 years. Outbreaks are hypothesized to be increasing in synchronicity because of changes in forest structure and species composition resulting from harvesting and fire suppression. These events have fostered the development multi-storied stands with high host tree densities (Swetnam and Lynch 1989)—conditions where western spruce budworm thrive as indicated above.

Douglas-fir Tussock Moth (Orgyia pseudotsugata)

Douglas-fir tussock moth is another major defoliator in western conifer forests. However, in Colorado, it has been historically more frequently found in urban environments with little and only localized activity in forested ecosystems. Since the early 1990s, outbreaks are occurring in forest lands in the Pike National Forest and surrounding areas (Negrón et al. 2014). The Douglas-fir tussock moth has one generation per year, overwintering in the egg stage in host tree foliage. Adults are active during summer and into the fall. Females are flightless so primary dispersion is by the first instar larvae which suspend on silken threads from the foliage, facilitating wind transport to other host trees, a behavior called “ballooning.” Outbreaks tend to be brief, most commonly up to 3 years, but can persist as long as 7 years. Outbreaks collapse

primarily by the pressure exerted by parasites, predators, and a nuclear polyhedrosis entomopathogenic virus. Defoliation by the insect can result in top-kill or tree mortality. Defoliation also results in tree growth reduction, although long-term growth rates may increase in surviving trees because of tree mortality of surrounding trees or increased nutrient availability, or both (Alfaro and Shepherd 1991; Wickman 1980). A study in the Pike National Forest indicated significant reductions in basal area caused by the Douglas-fir tussock moth in heavily defoliated stands compared to lightly defoliated stands. Douglas-fir beetle populations increased because of the outbreak but there were no differences in tree mortality between heavily and lightly defoliated stands. Douglas-fir tussock moth killed more trees < 6 inches in d.b.h. while Douglas-fir-beetle killed primarily larger-diameter trees (Negrón et al. 2014).

Balsam Woolly Adelgid (Adelges picea)

The balsam woolly adelgid is an invasive species of true firs that spread to the Interior West in the 1980s and into Utah in the 2010s (Davis et al. 2020). The insect is not present in Colorado but will likely spread into the Colorado Front Range in the future where subalpine fir occurs. Infestations can cause swelling at branch nodes (gouting) where feeding has occurred and can also occur in the main stem. Heavy infestations by the insect can kill trees of any age within 2 years by the sap-sucking activity of the insect to the stem, while lesser infestations cause a slow decline lasting several years by the gouting and stunted tree growth (Hrinkevich et al. 2016).

Diseases

Dwarf Mistletoes (Arceuthobium spp.)

Dwarf mistletoes are parasitic plants that infect conifers in the Colorado Front Range including ponderosa pine, lodgepole pine, limber pine, pinyon pine, and Douglas-fir. Mistletoes use nutrients and water from their host trees. Infections can result in reduced growth, reduced seed and cone production, and mortality when infections are severe (Geils and Hawksworth 2002). In the Central Rockies, about 51 percent of surveyed forests exhibited infections with healthy stands comprising 1.5 higher volumes of mistletoe than infected stands (Tainter and Baker 1996). Infections can also make trees more vulnerable to bark beetle attacks and facilitate mortality of drought-stressed trees (Kliejunas 2011). Trees can be deformed and “witches’ brooms” are produced. These deformities are important for many wildlife species and other organisms, particularly insects, fostering biological diversity. Dwarf mistletoe seeds are produced in female plants and are released explosively. Once seeds fall in a new tree and germinate, a root is produced that penetrates the phloem and xylem of the tree, which is how mistletoe obtains resources from the host. Many species of dwarf mistletoes occur in the Colorado Front Range using a variety of host trees (table 2).

Table 2—Dwarf mistletoes and primary hosts in the Colorado Front Range. Modified from the Field Guide to Insects & Insects of the Rocky Mountain Region, RMRS-GTR-241 (Rocky Mountain Region, Forest Health Protection 2010).

Dwarf mistletoe (DM) species	Primary host
Lodgepole pine DM, *Arceuthobium americanum*	Lodgepole pine
Limber pine DM, *A. cyanocarpum*	Limber and bristlecone pine
Pinyon DM, *A. divaricatum*	Pinyon pine
Douglas-fir DM, *A. douglasii*	Douglas-fir
Southwestern DM, *A. vaginatum subsp. cryptopodium*	Ponderosa pine

Armillaria Root Disease (Armillaria spp.)

Armillaria root disease is the most widely distributed root disease in Colorado (Johnson 1984). In the Colorado Front Range, it occurs primarily in spruce-fir and mixed-conifer forests with higher mortality occurring in subalpine fir trees (Worrall et al. 2004). It is caused by various species of fungi, although *Armillaria solidipes* appears to be the most common. Armillaria root disease infects numerous hosts including all conifers in the Colorado Front Range. Infected trees often exhibit a flow of resin at the base of the tree and thinning of the crown. As the infections move through root contact and rhizomorphs, the disease can create disease centers where adjacent trees are killed, although this is not always the case. Species of *Armillaria* can be secondary agents of tree disease infecting stressed hosts, decomposing dead roots, or aggressive primary pathogens killing healthy trees (Kliejunas 2011; Wargo and Shaw 1985). Root diseases cause growth reductions, mechanical failure that can result in windthrow, and direct tree mortality (Shaw and Kile 1991). Root disease can also make trees more susceptible to bark beetle attacks (Geils and Hawksworth 2002; Tkacz and Schmitz 1986) suggesting that increases in root disease incidence could also exacerbate bark beetle infestations.

White Pine Blister Rust (Cronartium ribicola)

The white pine blister rust (*Cronartium ribicola*) pathogen was introduced from Asia into North America in the early 1900s (Bega 1978) and was first detected in Colorado in 1999 in limber pine in northern Larimer County (Johnson and Jacobi 2000). By 2003, infestations were detected in the Sangre de Cristo and Wet Mountains in limber and bristlecone pine (Blodgett and Sullivan 2004; Burns 2006; USDA FS 2013). The pathogen continues to move in limber to the south and east of the Rockies in Colorado and Wyoming (Jacobi et al. 2018). In combination with other stressors such as drought, and particularly the mountain pine beetle, high mortality levels have been reported for various species of white pines in locations across western North America (Kliejunas 2011). The disease is observed more frequently in limber pine than bristlecone pine although overall incidence is low (Burns 2006). Infections result in branch dieback and tree and sapling mortality (Kliejunas 2011). The rust requires two hosts for completion of its life cycle utilizing primarily *Ribes* spp. and a five-needle pine.

Sudden Aspen Decline and Cytospora Canker of Aspen

Sudden aspen (*Populus tremuloides*) decline was first observed in southwestern Colorado in the mid-2000s and by 2006 almost 150,000 acres were affected. It presents as a rapid synchronous branch dieback, thinning of crowns, and tree mortality at a landscape scale (Dudley et al. 2015; Kliejunas et al. 2009; Worrall et al. 2008). Aspen decline is related to drought conditions (Worall et al. 2010, 2013). Secondary factors such as western tent caterpillar defoliation and bark beetles may have exacerbated mortality levels (Dudley et al. 2015; Worrall et al. 2008).

Cytospora canker of aspen is a common bark disease caused by the fungus *Valsa sordida*. Infections occur primarily when the fungus enters the host through wounds on the bark surface and kills the cambium. It is also a stress pathogen as it occurs more commonly in stressed trees (McIntyre et al. 1996). Cytospora canker can cause dead branches but most importantly it can cause tree mortality of stressed trees, particularly by drought. Healthy trees can often limit colonization.

Effects of Climate Change on Colorado Front Range Forest Insects and Diseases

Climate change will have significant effects on insects as they are particularly sensitive to temperatures. Their short generation times and high biotic potential may lead to faster adaptations to new climatic conditions of higher temperatures compared to their hosts (Jactel et al. 2019). Climate change is anticipated to result in more intense droughts that can weaken trees and compromise their defenses making them more susceptible to insect infestation. More intense storms may facilitate the occurrence of blowdowns that can foster outbreaks of some bark beetle species. Increased atmospheric carbon dioxide will influence host physiology and interactions with associated insects; however, these interactions are not well understood. Finally, climate change will also influence the ecology of natural enemies of insects such as parasites, predators, and a suite of organisms associated with bark beetles such as fungi, mites, and nematodes. Changes in temperature and moisture will influence the life cycle, development, and infectivity of diseases. Effects on host tree physiology and migration will also influence the response of pathogenic organisms.

Bark Beetles

The most important effects of climate change on bark beetles will be manifested through increased temperatures (Bentz et al. 2010). Warmer temperatures will impact overwintering survival and developmental time. Other important effects will include how host trees respond to climate possibly affecting the relationship between beetle species and its host. Climate change will also impact a suite of organisms associated with bark beetles such as mites, fungi, nematodes, and other microorganisms that strongly influence insect development and population dynamics (Jactel 2019).

Cold winter temperatures are a major mortality agent for overwintering bark beetle populations (Bentz et al. 2010; Cole 1981). To survive cold winter temperatures, mountain pine beetle larvae begin the process of "cold-hardening," producing alcohols that protect the larvae from freezing. This begins as temperatures decline towards the winter (Bentz and Mullins 1999). While this adaptation fosters survival of the larvae, cold temperature can still result in mortality. An increase in winter temperatures will facilitate increased winter survival that can lead to higher populations and increased tree mortality. Mountain pine beetles begin attacking new trees in the early to mid-summer and "mass attack," that is, large numbers of beetles attack the tree to be able to overcome the defenses of the tree. For this to occur, mountain pine beetles need to emerge in synchrony. Mountain pine beetle exhibits differential developmental thresholds during its larval instars that fosters synchronous emergence. Warmer temperatures could increase this synchrony and enhance population success (Powell and Bentz 2009). However, warmer temperatures could also increase developmental rates causing the insects to enter winter as an adult, a cold-susceptible stage.

Warmer temperatures can also influence habitat suitability. For example, spruce beetle typically exhibits a 2-year lifecycle, but under warmer conditions they can complete development in 1 year. This change in developmental timing can result in faster buildup of populations and increased tree mortality (Werner and Holsten 1985; Werner et al. 2006). Models have indicated an increase in the probability of spruce beetle transitioning from a 2-year life cycle to 1-year life cycle (Bentz et al. 2010; Hansen et al. 2001).

Warming temperatures will also impact host trees. Changes in water availability to trees will result in physiological stresses that compromise trees' ability to maintain defenses such as resin production. However, elevated atmospheric carbon dioxide could also result in increased resin availability for defense and increased carbohydrates improving the nutritional quality of the host (see Jactel 2019).

In addition to the major bark beetles, pinyon ips may be an insect worthy of increased attention. In general, ips beetles are favored by warmer and drier conditions as they prefer to colonize trees under stress. In addition, the likelihood of infestations by pinyon ips increases with dwarf mistletoe infections (Negrón and Wilson 2003), which may also be exacerbated under climate change and are well distributed in the Pike-San Isabel National Forests (table 4). The combination of these two factors may elevate the importance of this insect in the Pike-San Isabel National Forests.

Mountain Pine Beetle Degree-Day Scenario

A mountain pine beetle scenario is presented as an example of the potential effect of climate change on the habitat distribution of mountain pine beetle in the Arapaho-Roosevelt National Forests. Pheromone traps catch data from several years in lodgepole and ponderosa pine stands indicate that peak flight occurs around the end of July to

early August (fig. 1). Data from several years of successive pheromone trap catches indicate that 898 is the mean number of degree days from peak flight in one year to peak flight the following year using a base temperature of 5.5 °C (Negrón, unpublished data).

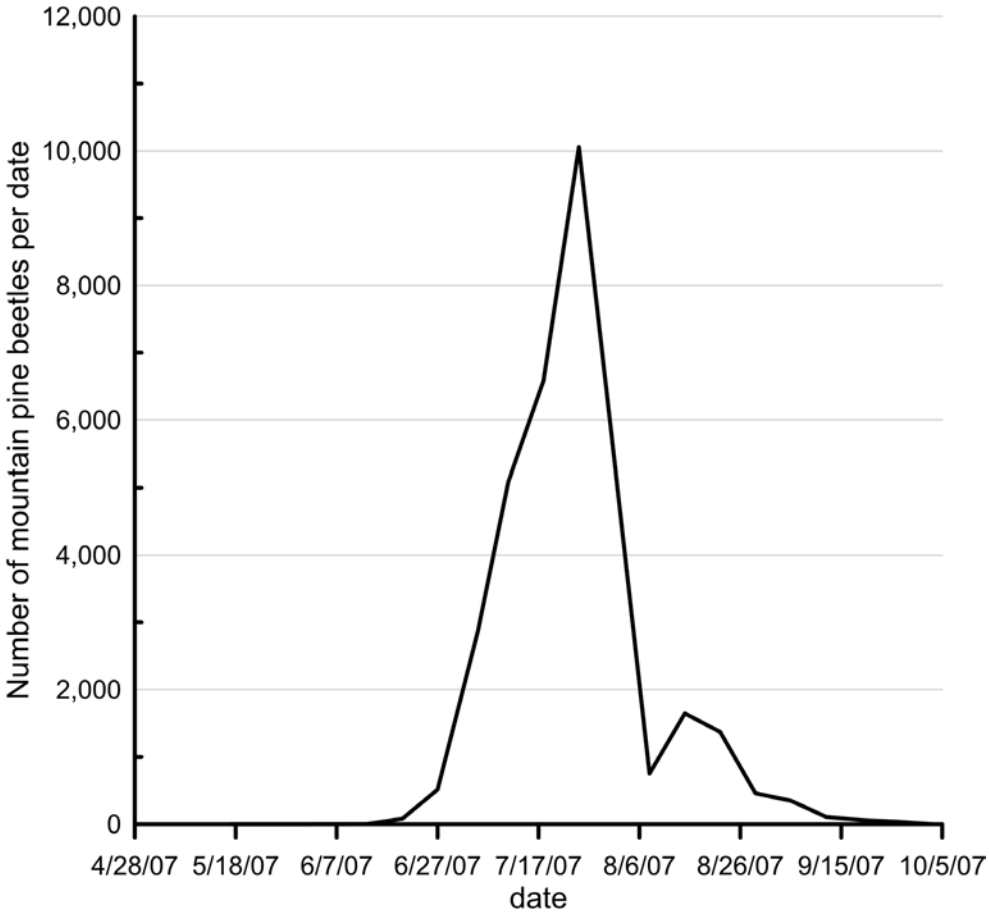


Figure 1—Example of typical mountain pine beetle flight periodicity at the Arapaho-Roosevelt National Forests using data from 2007. Note peak flight occurring at the end of July.

Matthews et al. (2018, 2019) examined signals of climate change across the United States including growing degree days (base 5 °C) and prepared maps comparing recent conditions to future projections under two scenarios of potential change (RCPs 4.5 and 8.5), and four different time periods. Data from the RPC 4.5 maps for the area comprised by the Arapaho-Roosvelt National Forests and plotting an isoline at 898-degree days indicate how the areas of biological suitability where mountain pine beetle can exhibit peak flight between successive years change (fig. 2). In brief, it can be seen how the suitability areas shrink over time, generally transitioning to higher elevations. It should be noted that this is a scenario that does not consider other effects of climate change of mountain pine beetle populations or host responses.

Balsam Wooly Adelgid

Balsam wooly adelgid is expected to continue to spread throughout the range of subalpine fir causing decline and mortality. Extreme winter and summer temperatures can compromise survival and spread. Predictive models of susceptibility to the insect for the Colorado Front Range were like those of forests in northern Utah where the species is now present and causing tree mortality (Hrinkevitch et al. 2016). Conditions associated with a warming climate may allow increased survival and spread of the insect. As the climate becomes hotter, it could further improve favorable conditions for the insect establishment, such as warmer September temperatures needed for the winter survival of the first instar nymphs. That, combined with climate that brings wet and cool spring conditions, especially in May, could incite mortality events (Hrinkevich 2016).

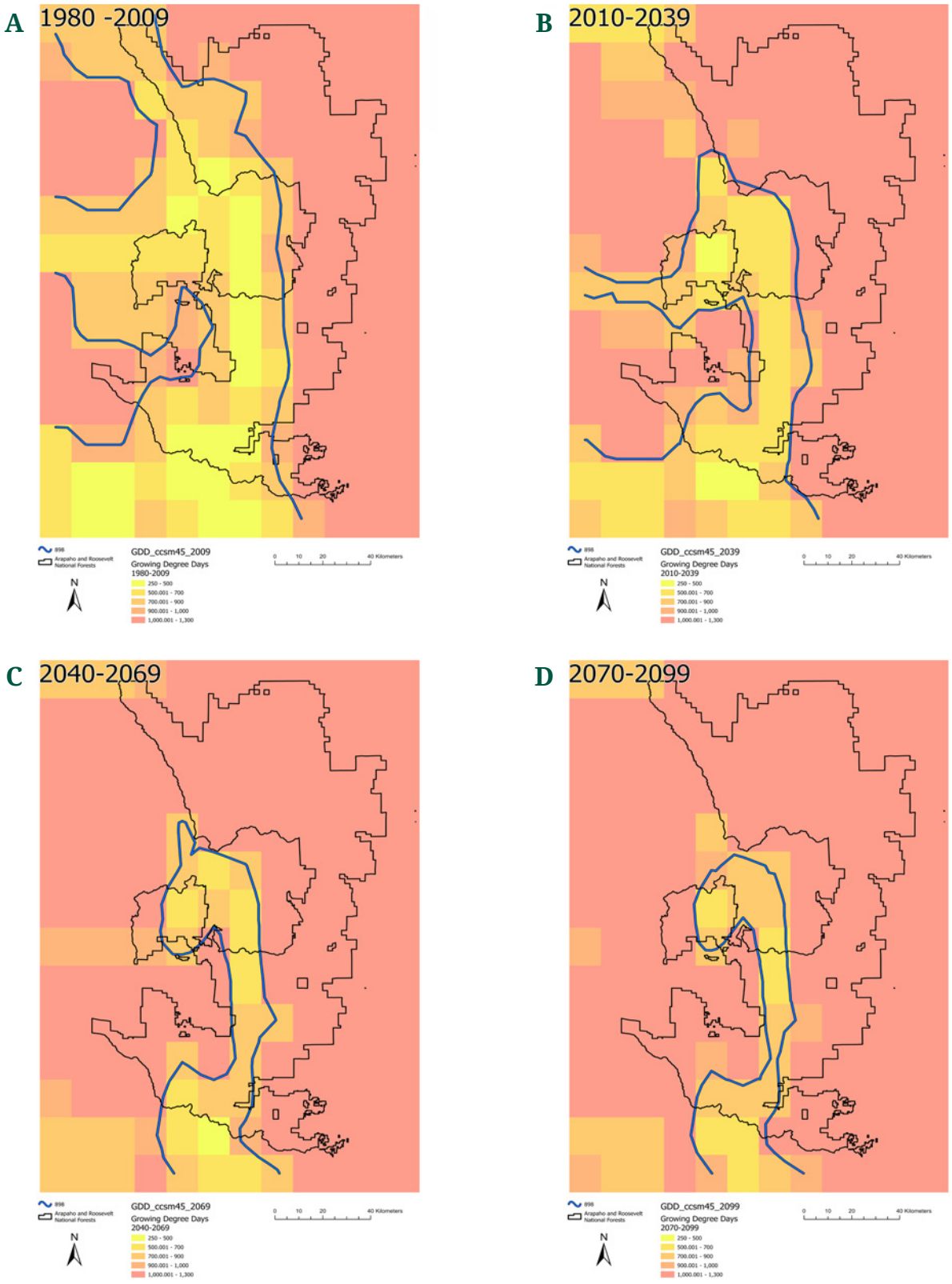


Figure 2—Recent and projected degree-day accumulations at the Arapaho-Roosevelt National Forests. Data come from projections by Matthews et al. 2018, 2019. Blue lines enclose areas of suitability for mountain pine beetle peak flight between successive years: (a) 1980–2009, (b) 2010–2039, (c) 2040–2069, and (d) 2070–2099.

Defoliators

Weather conditions govern many biological processes of defoliators and their interactions with their hosts. Although there is a paucity of information on the potential effects of climate change on defoliators in the PSI-AR, available knowledge on the ecology of these organisms can help answer some questions.

Climate change will influence western spruce budworm in diverse ways so it is challenging to portray what the future may look like. Western spruce budworm outbreaks have increased in duration, intensity, and extent. Historical land use and warming climate are considered as drivers (Gray 2008; Swetnam and Lynch 1993; Williams and Liebhold 1995) but causal mechanisms are poorly understood (Peltonen et al. 2002). Western spruce budworm outbreaks can cover large geographical areas and last decades (Swetnam and Lynch 1989, 1993). Temperature patterns and fluctuations and rainfall can affect development, dispersal, feeding, fecundity, and survival. Early spring temperatures help synchronize larval emergence with bud break. Warm springs can result in earlier emergence while cold temperatures can delay emergence decoupling synchronization; either way, it is maladaptive to the larvae by increasing exposure to natural enemies and limiting food quality. Warm temperatures in the fall and winter can result in increased metabolic expenditures by the larvae; colder temperatures can increase the risk of frost after emerging from their winter habitat; and dispersal limited by increased rainfall. Cool springs and early summer can reduce developmental speed making the larvae more susceptible to predation. (See Beckwith and Burnell 1982; Carlson et al. 1983; Chen et al. 2003; Johnson and Denton 1975; O'Connor et al. 2015; Régnière et al. 2012; Thomson 1979; Thomson and Benton, 2007).

Western spruce outbreaks in the southwestern United States have been associated with increased spring and summer precipitation, which may foster abundant and better-quality food resources (Flower et al. 2014; Ryerson et al. 2003; Swetnam and Betancourt 1998; Swetnam and Lynch 1993). On the other hand, several studies have related outbreak initiation with reduced moisture prior to the outbreak (Flower et al. 2014; Senf et al. 2017; Xu et al. 2019). The outbreak association with increased moisture has been documented in the San Juan Mountains of southwestern Colorado and in northern New Mexico while the association with reduced moisture prior to outbreaks is more typical of the Northwest and British Columbia (Flower et al. 2014; Senf et al. 2017; Xu et al. 2019)

It will be important to monitor western spruce budworm populations as climate change manifests. Tree-ring reconstructions in the southern Rockies, including the Colorado Front Range, suggest average outbreak return intervals every 35 years with a range of 14–58 years (Swetnam and Lynch 1989). Although an outbreak occurred in the Arapaho-Roosevelt NFs in the 1970s, populations are currently scarce (table 3). Western spruce budworm is currently well distributed in the Pike-San Isabel National Forests (table 4).

Table 3—Number of FIA plots by forest type in the Arapaho-Roosevelt National Forests with presence of dwarf mistletoe, root disease, or western spruce budworm. Note that dwarf mistletoe occurrence was widespread, with only small detections of root disease occurring mostly from lodgepole pine up to subalpine forests. Western spruce budworm was not detected. Data from FIA measurements from 2009–2018.

Forest type	FIA plots (number)	Dwarf mistletoe	Root disease	Western spruce budworm
Douglas-fir	14	50.0	0.0	0.0
Ponderosa pine	24	25.0	0.0	0.0
Engelmann spruce	17	5.9	5.9	0.0
Engelmann spruce/subalpine fir	47	17.0	2.1	0.0
Subalpine fir	16	18.8	0.0	0.0
Lodgepole pine	67	46.3	1.5	0.0
Bristlecone pine	1	0.0	0.0	0.0
Limber pine	5	0.0	0.0	0.0
Aspen	19	10.5	0.0	0.0

Douglas-fir tussock moth outbreaks are short-lived, and their duration is strongly influenced by the epidemiology of a nuclear polyhedrosis virus that regulates populations. It is unknown how the virus will respond to climate change; if negatively impacted, Douglas-fir tussock moth outbreaks may increase in duration. Warmer temperatures may increase the likelihood of outbreaks in drier sites.

Table 4—Number of FIA plots by forest type in the Pike-San Isabel National Forests with presence of dwarf mistletoe, root disease, or western spruce budworm. Note that dwarf mistletoe and western spruce budworm occurrence was widespread, with only small detections of root disease mostly occurring in subalpine forests. Data from FIA measurements from 2009–2018.

Forest type	FIA no. of plots	Dwarf mistletoe	Root disease	Western spruce budworm
Pinyon-juniper woodland	13	15.4	0.0	0.0
Douglas-fir	64	14.1	1.6	26.6
Ponderosa pine	44	15.9	0.0	20.5
White fir	7	0.0	0.0	42.9
Engelmann spruce	59	0.0	3.4	11.9
Engelmann spruce/subalpine fir	14	7.1	7.1	14.3
Subalpine fir	2	0.0	0.0	0.0
Blue spruce	1	0.0	0.0	0.0
Lodgepole pine	31	32.3	0.0	3.2
Foxtail/bristlecone pine	5	20.0	0.0	0.0
Limber pine	12	0.0	0.0	8.3
Aspen	39	2.6	0.0	22.9

Pathogens

Dwarf Mistletoe

Dwarf mistletoe can cause tree mortality particularly when trees become stressed due to drought conditions (Kliejunas et al. 2009). Tree mortality caused by dwarf mistletoe is the result of declining growth rates in the long term, particularly when infections become severe towards the top of the tree (Hawksworth and Wiens 1996). For the most part, the geographic distribution of mistletoe species follows that of its hosts, but their range is limited by cold temperatures (Hawksworth and Weins 1996). This suggests that range expansion is possible under increased temperatures with mistletoes following host migration.

Stanton (2007), working with mature ponderosa pine, indicated that the growth rate of infected trees is more sensitive than those of uninfected trees and that moisture stress can exacerbate mistletoe infections. Increasing temperatures and drought frequency anticipated under climate change may have beneficial effects to mistletoe infections.

Dwarf mistletoe is currently widespread in both the Arapaho-Roosevelt and the Pike-San Isabel National Forests (tables 3 and 4), particularly in the ponderosa pine, lodgepole pine, and Douglas-fir forest types, indicating that it is one of the most important agents to monitor as climate change continues to manifest. Mistletoe infections in pinyon will make them susceptible to pinyon ips.

Armillaria Root Disease (Armillaria spp.)

Tree stress is a primary risk factor for Armillaria infection; therefore, increased stress of host trees associated with warmer temperatures and drought may result in increased levels of *Armillaria* (Kliejunas et al. 2009; Shaw and Kile 1991; Sturrock et al. 2011). In addition, increased levels of atmospheric carbon dioxide may foster increased root growth, which may increase the likelihood of an infection to move from one tree to an adjacent one through root contact (O'Neill 1994). Host distribution will also be key to the future distribution of *Armillaria*. While an increase in activity is expected, the area of climatic suitability for Douglas-fir, a primary host, may decrease (Klopfenstein et al. 2009).

Root disease is not widely distributed in either the Arapaho-Roosevelt or the Pike-San Isabel National Forests. However, it is present in spruce-fir stands of both Forests (tables 3 and 4). If the disease increases with tree stress, it may become a factor in these Forests.

White Pine Blister Rust (Cronartium ribicola)

White pine blister rust is considered a cool weather disease. Development, dispersal, and germination of rust spores are directly affected by environmental conditions (Kliejunas et al. 2009), and conditions for successful infection have been examined (Spaulding 1922; Sturrock et al. 2011; Van Arsdel 1956). The fungus requires high

humidity and temperatures below 68 °F. Reduced infection levels are expected under the high temperatures and dryer conditions anticipated under climate change (Kinloch 2003; Kliejunas 2011; Sturrock et al. 2011).

Sudden Aspen Decline and Cytospora Canker of Aspen

Both conditions are associated with stress conditions and drought and will likely increase under warming temperatures and more frequent droughts (Kliejunas et al. 2009; Worrall et al. 2013); however, interactions of these conditions with climate change have not been examined.

Concluding Remarks

Warming temperatures and changes in precipitation patterns associated with climate change will influence the biology and ecology of forest insects and diseases. In general, the literature suggests an increase in frequency and intensity of outbreaks, range expansions, and increases in population numbers will be affected by a warming climate. In the long term, effects will also be governed by how host trees respond to climate change as their distributions will change and species are expected to migrate. Changes in host distribution may have positive or negative effects on insects and diseases. Increased stress in hosts associated with drought may make trees more susceptible to some insects and diseases synergizing and influencing tree mortality in the most affected areas. The effects of climate change on associated organisms of insects and diseases, such as natural enemies and vectors, remain largely unknown. In sum, some insect and diseases will exhibit positive responses such as faster growth, larger insects, range expansion, and increased capability to successfully infest their hosts while others may exhibit negative responses such as decoupling host and insect phenology and fostering maladaptive life cycles.

While our understanding of these interactions necessitates additional research, a few general statements can be made. Outbreaks of mountain pine beetle may increase in frequency and intensity because of increased drought-induced stress in their hosts and increased winter temperatures that foster increased survival in insect overwintering populations. Spruce beetle outbreaks may also become more common as warmer temperatures foster 1-year life cycles, which support population growth along with increased stress in hosts caused by drought. Pinyon ips could increase as the insect prefers warmer and drier conditions and is exacerbated by dwarf mistletoe infections, which are also expected to increase due to host stress. Western spruce budworm populations will likely increase during periods of increased moisture. Armillaria root disease will likely increase in abundance in higher-elevation spruce-fir stands because of increased tree stress. At low elevations, mistletoe parasitism would be more prevalent and white pine blister rust may decrease as it is more successful under cooler wetter conditions. Continued monitoring of insect and disease populations should be increased by managers and practitioners as climate change continues to manifest and increased research efforts are needed to knowledge gaps.

References

Alfaro, R.I.; Shepherd, R.F. 1991. Tree-ring growth of interior Douglas-fir after one year's defoliation by Douglas-fir tussock moth. Forest Science. 37: 959–964. https://doi.org/10.1093/forestscience/37.3.959

Atkins, D.H.; Davis, T.S.; Stewart, J.E. 2021. Probability of occurrence and phenology of pine wilt disease transmission by insect vectors in the Rocky Mountains. Ecological Solutions and Evidence. 2(1): e12044. https://doi.org/10.1002/2688-8319.12044

Battisti, A.; Stastny, M.; Buffo, E.; [et al.]. 2006. A rapid altitudinal range expansion in the pine processionary moth produced by the 2003 climatic anomaly. Global Change Biology. 12: 662–71. https://doi.org/10.1111/j.1365-2486.2006.01124.x

Beckwith, R.C.; Burnell, D.G. 1982. Spring larval dispersal of the western spruce budworm (Lepidoptera: Tortricidae) in north-central Washington. Environmental Entomology. 11: 828–832. https://doi.org/10.1093/ee/11.4.828

Bega, R.V. 1978. Rusts. In: Bega, R.V., tech. coord. Diseases of Pacific Coast conifers. Agricultural Handbook 521. Washington, DC: U.S. Department of Agriculture, Forest Service.

Bentz, B.J.; Mullings, D.E. 1999. Ecology of mountain pine beetle (Coleoptera: Scolytidae) cold hardening in the Intermountain West. Environmental Entomology. 28: 577–587. https://doi.org/10.1093/ee/28.4.577

Bentz, B.J.; Régnière, J.; Fettig, C.J.; [et al.]. 2010. Climate change and bark beetles of the western United States and Canada: Direct and indirect effects. Bioscience. 60: 602–613. https://doi.org/10.1525/bio.2010.60.8.6

Blodgett, J.T.; Sullivan, K.F. 2004. First report of white pine blister rust on Rocky Mountain bristlecone pine. Plant Disease. 88: 311. https://doi.org/10.1094/PDIS.2004.88.3.311A

Breshears, D.D.; Cobb, N.S.; Rich, P.M.; [et al.]. 2005. Regional vegetation die-off in response to global-change-type drought. Proceedings of the National Academy of Sciences of the United States of America. 102: 15144–15148. https://doi.org/10.1073/pnas.0505734102

Burns, K.S. 2006. White pine blister rust surveys in the Sangre de Cristo and Wet Mountains of Southern Colorado. Biological Evaluation R2-06-05. Denver, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Region, Renewable Resources.

Carlson, C.E.; Fellin, D.G.; Schmidt, W.C. 1983. The western spruce budworm in northern Rocky Mountain forests: A review of ecology, insecticidal treatments, and silvicultural practices. In: Management of second growth forests, the state of knowledge and research needs. O'Laughlin, J.; Pfister, R.D., eds. Missoula, MT: University of Montana, Montana Forest and Conservation Experiment Station.

Chen, Z.; Clancy, K.M.; Kolb, T.E. 2003. Variation in budburst phenology of Douglas-fir related to western spruce budworm (Lepidoptera: Tortricidae) fitness. Journal of Economic Entomology. 96: 77–387. https://doi.org/10.1093/jee/96.2.377

Cleaver, C.M.; Jacobi, W.R.; Burns, K.S.; [et al.]. 2015. Limber pine in the central and southern Rocky Mountains: Stand conditions and interactions with blister rust, mistletoe, and bark beetles. Forest Ecology and Management. 358: 139–153. https://doi.org/10.1016/j.foreco.2015.09.010

Cole, W.E. 1981. Some risks and causes of mortality in mountain pine beetle populations: A long-term analysis. Research in Population Ecology. 23: 116–144. https://doi.org/10.1007/BF02514096

Cole, H. M.; Andrus, R. A.; Butkiewicz, C.; [et al.]. 2022. Outbreaks of Douglas-fir beetle follow western spruce budworm defoliation in the Southern Rocky Mountains, USA. Forests. 13(3): 371. https://doi.org/10.3390/f13030371

Costello, J.D.; Leopold, D.J.; Smallidge, P.J. 1995. Pathogens, patterns, and processes in forest ecosystems. Bioscience. 45: 16–24. https://doi.org/10.2307/1312531

Cullingham, C.I.; Cooke, J.E.K.; Dang, S.; [et al.]. 2011. Mountain pine beetle host-range expansion threatens the boreal forest. Molecular Ecology. 20: 2157–2171. https://doi.org/10.1111/j.1365-294X.2011.05086.x

Davis, G.; Lowrey, L.; Eckberg, T.; [et al.]. 2020. Notes on balsam woolly adelgid, *Adelges piceae* (Ratzeburg, 1844) (Hemiptera: Adelgidae), range expansion in Idaho, Montana, and Utah. The Pan-Pacific Entomologist. 96: 129–133. https://doi.org/10.3956/2020-96.3.129

Dooley, E.M.; Six, D.L.; Powell, J.A. 2015. A comparison of mountain pine beetle (Coleoptera: Curculionidae, Scolytinae) productivity and survival in lodgepole and whitebark pine after a region-wide cold weather event. Forest Science. 61: 235–246. https://doi.org/10.5849/forsci.14-014

Dudley, M.M.; Burns, K.S.; Jacobi, W.R. 2015. Aspen mortality in the Colorado and southern Wyoming Rocky Mountains: Extent, severity, and causal factors. Forest Ecology and Management. 353: 240–259. https://doi.org/10.1016/j.foreco.2015.06.002

Fellin, D.G.; Shearer, R.C.; Carlson, C.E. 1983. Western spruce budworm in the Northern Rocky Mountains: Biology, ecology, and impacts. Western Wildlands. 9: 2–7.

Flower, A.; Gavin, D.G.; Heyerdahl, E.K.; [et al.]. 2014. Drought-triggered western spruce budworm outbreaks in the interior Pacific Northwest: A multi-century dendrochronological record. Forest Ecology and Management. 324: 16–27. https://doi.org/10.1016/j.foreco.2014.03.042

Geils, B.W.; Hawksworth, F.G. 2002. Chapter 5: Damage, effects, and importance of dwarf mistletoes. In: Geils, B.W.; Cibrian Tovar, J.; Moody, B., tech. coords. Mistletoes of North American Conifers. Gen. Tech. Rep. RMRS-GTR-98. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 57–65. https://www.fs.usda.gov/research/treesearch/27738

Gray, D.R. 2008. The relationship between climate and outbreak characteristics of the spruce budworm in eastern Canada. Climatic Change. 87: 361–383. https://doi.org/10.1007/s10584-008-9478-x

Hadley, K.S.; Veblen, T.T. 1993. Stand response to western spruce budworm and Douglas-fir bark beetle outbreaks, Colorado Front Range. Canadian Journal of Forest Research. 23: 479–491. https://doi.org/10.1139/x93-066

Hansen, E. M.; Bentz, B.J.; Turner, D.L. 2001. Temperature-based model for predicting univoltine brood proportions in spruce beetle (Coleoptera: Scolytidae). The Canadian Entomologist. 133: 827–841. https://doi.org/10.4039/Ent133827-6

Hawksworth, F.G.; Wiens, Dt. 1996. Dwarf mistletoes: Biology, pathology, and systematics. Agricultural Handbook 709. Washington, DC: U.S. Dept. of Agriculture, Forest Service. 410 p. https://www.fs.usda.gov/research/treesearch/4699

Hrinkevich, K.H.; Progar, R.A.; Shaw, D.C. 2016. Climate risk modelling of balsam woolly adelgid damage severity in subalpine fir stands of Western North America. PLoS ONE. 11:e0165094. https://doi.org/10.1371/journal.pone.0165094

Jacobi, W.R.; Kearns, H.S.J.; Cleaver, C.M.; [et al.]. 2018. Epidemiology of white pine blister rust on limber pine in Colorado and Wyoming. Forest Pathology. 48:e12465. https://doi.org/10.1111/efp.12465

Jactel, H.; Koricheva, J.; Castagneyrol, B. 2019. Response of forest insect pests to climate change: Not so simple. Current Opinion in Insect Science. 35: 103–108. https://doi.org/10.1016/j.cois.2019.07.010

Johnson, D.W. 1984. An assessment of root diseases in the Rocky Mountain Region. Technical Report R2-29. Lakewood, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Region.

Johnson, P.C.; Denton, R.E. 1975. Outbreaks of the western spruce budworm in the American northern Rocky Mountain area from 1922 through 1971. Gen. Tech. Rep. INT-20. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station.

Johnson, D.W.; Jacobi, W.R. 2000. First report of white pine blister rust in Colorado. Plant Disease. 84: 595. https://doi.org/10.1094/PDIS.2000.84.5.595D

Kinloch Jr., B.B. 2003. White pine blister rust in North America: Past and prognosis. Phytopathology. 93: 1044–1047. https://doi.org/10.1094/PHYTO.2003.93.8.1044

Kliejunas, J. T. 2011. A risk assessment of climate change and the impact of forest diseases on forest ecosystems in the western United States and Canada. Gen. Tech. Rep. PSW-GTR-236. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 70 p. https://doi.org/10.2737/PSW-GTR-236

Kliejunas, J.T.; Geils, B.W.; Glaeser, J.M.; [et al.]. 2009. Review of literature on climate change and forest diseases of western North America. Gen. Tech. Rep. PSW-GTR-225. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 54 p. https://doi.org/10.2737/PSW-GTR-225

Klopfenstein, N.B.; Kim, M.; Hanna, J.W.; [et al.]. 2009. Approaches to predicting potential impacts of climate change on forest disease: An example with *Armillaria* root disease. Res. Pap. RMRS-RP-76. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 10 p. https://doi.org/10.2737/RMRS-RP-76

Klutsch, J.G.; Negrón, J.F.; Costello, S.L.; [et al.]. 2009. Stand characteristics and downed woody debris accumulations associated with a mountain pine beetle (*Dendroctonus ponderosae* Hopkins) outbreak in Colorado. Forest Ecology and Management. 258: 641–649. https://doi.org/10.1016/j.foreco.2009.04.034

Lessard, E.D.; Schmid, J.M. 1990. Emergence, attack densities, and host relationships for the Douglas-fir beetle, *Dendroctonus pseudotsugae* Hopkins in northern Colorado. Great Basin Naturalist. 50: 333–338. https://scholarsarchive.byu.edu/gbn/vol50/iss4/7

Lundquist, J.E.; Negrón, J.F. 2000. Endemic forest disturbances and stand structure of ponderosa pine (*Pinus ponderosa*) in the Upper Pine Creek Research Natural Area, Black Hills National Forest, South Dakota, USA. Natural Areas Journal. 20: 126–132. https://www.jstor.org/stable/43911897

MacLauchlan, L.; Brooks, J.E. 2009. Influence of past forestry practices on western spruce budworm defoliation and associated impacts in southern British Columbia. British Columbia Journal of Ecosystems and Management. 10: 37–49. https://doi.org/10.22230/jem.2009v10n2a418

MacLean, D.A. 1985. Effects of spruce budworm outbreaks on forest growth and yield. In: Recent advances in spruce budworm research, Proc. CANUSA spruce budworm res. symp., Sanders, C.J.; [et al.], eds. Ottawa, Ontario: Canadian Forest Service. https://ostrnrcan-dostrncan.canada.ca/handle/1845/239356

Massey, C.L.; Wygant, N.D. 1954. Biology and control of the Engelmann spruce beetle in Colorado. Circular No. 944. Washington, DC: U.S. Department of Agriculture, Forest Service. 35 p.

Matthews, S.N.; Iverson, L.R.; Peters, M.P.; [et al.]. 2018. Assessing potential climate change pressures across the conterminous United States: Mapping plant hardiness zones, heat zones, growing degree days, and cumulative drought severity throughout this century. Res. Map NRS-9. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 31 p. https://doi.org/10.2737/NRS-RMAP-9

Matthews, S.N.; Iverson, L.R.; Peters, M.P.; [et al.]. 2019. Climate change pressures for the conterminous United States: Plant hardiness zones, heat zones, growing degree days, and cumulative drought severity. Fort Collins, CO: Forest Service Research Data Archive. https://doi.org/10.2737/RDS-2019-0001.

McCambridge, W.F. 1964. Emergence period of Black Hills beetles from ponderosa pine in the central Rocky Mountains. Research Note RM-32. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 4 p.

McCambridge, W.F.; Hawksworth, F.G.; Edminster, C.B.; [et al.]. 1982. Ponderosa pine mortality resulting from a mountain pine beetle outbreak. Research Paper RM-235. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 7 p. https://doi.org/10.2737/RM-RP-235

McIntyre, G.A.; Jacobi, W.R.; Ramaley, A.W. 1996. Factors affecting Cytospora canker occurrence on aspen. Journal of Arboriculture. 22: 229–233. https://doi.org/10.48044/jauf.1996.035

Mercado, J.E. 2020. Improved identification and new records of *Dendroctonus* bark beetles attacking *Pinus contorta* in the subalpine forest of the southern Rocky Mountains. Forests. 11(6): 656. https://doi.org/10.3390/f11060656

Negrón, J.F. 1998. Probability of infestation and extent of mortality associated with the Douglas-fir beetle in the Colorado Front Range. Forest Ecology and Management. 107: 71−85. https://doi.org/10.1016/S0378-1127(97)00319-8

Negrón, J. 2019. Biological aspects of mountain pine beetle in lodgepole pine stands of different densities in Colorado, USA. Forests. 10, 18. https://doi.org/10.3390/f10010018

Negrón, J.F.; Cain, B. 2019. Mountain pine beetle in Colorado: A story of changing forests. Journal of Forestry. 117: 144–151. https://doi.org/10.1093/jofore/fvy032

Negrón, J.F.; Klutsch, J.G. 2017. Probability of infestation and extent of mortality models for mountain pine beetle in lodgepole pine forests in Colorado. Res. Note RMRS-RN-77. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 13 p. https://doi.org/10.2737/RMRS-RN-77

Negrón, J.F.; Lynch, A.M.; Schaupp Jr., W.C.; [et al.]. 2014. Douglas-fir tussock moth- and Douglas-fir beetle-caused mortality in a ponderosa pine/Douglas-fir forest in the Colorado Front Range, USA. Forests. 5: 3131–3146. https://doi.org/10.3390/f5123131

Negrón J.F.; Popp, J.B. 2004. Probability of ponderosa pine infestation by mountain pine beetle in the Colorado Front Range. Forest Ecology and Management. 191: 17−27. https://doi.org/10.1016/j.foreco.2003.10.026

Negrón J.F.; Popp, J.B. 2009. The flight periodicity, attack patterns, and life history of *Dryocoetes confusus* Swaine (Coleoptera: Curculionidae: Scolytinae), the western balsam bark beetle, in north-central Colorado. Western North American Naturalist. 69: 447–458. https://doi.org/10.3398/064.069.0404

Negrón, J.F.; Schaupp Jr., W.C.; Pederson, L. 2011. Flight periodicity of the Douglas-fir beetle, *Dendroctonus pseudotsugae* Hopkins (Coleoptera: Curculionidae: Scolytinae) in Colorado, USA. Coleopterits Bulletin. 65: 182–184. https://doi.org/10.1649/072.065.0219

Negrón, J.F.; Wilson, J.L. 2003. Attributes associated with probability of infestation by the pinon ips, *Ips confusus* (Coleoptera: Scolytidae) in pinon pine, *Pinus edulis*. Western North American Naturalist. 63: 440–451. https://scholarsarchive.byu.edu/wnan/vol63/iss4/4

O'Connor, C.D.; Lynch, A.M.; Falk, D.A. [et al.]. 2015. Post-fire forest dynamics and climate variability affect spatial and temporal properties of spruce beetle outbreaks on a Sky Island mountain range. Forest Ecology and Management. 336: 148–162. https://doi.org/10.1016/j.foreco.2014.10.021

O'Neill, E.G. 1994. Responses of soil biota to elevated atmospheric carbon dioxide. Plant and Soil. 165: 55–65. https://doi.org/10.1007/BF00009962

Peltonen, M.; Liebhold, A.M; Bjørnstad, O.N; [et al.]. 2002. Spatial synchrony in forest insect outbreaks: roles of regional stochasticity and dispersal. Ecology. 83: 3120–3129. https://doi.org/10.1890/0012-9658(2002)083[3120:SSIFIO]2.0.CO;2

Peltz, K.A.; Smith, F.W. 2012. Thirty-year change in lodgepole and lodgepole/mixed conifer forest structure following 1980s mountain pine beetle outbreak in western Colorado. Forest Ecology and Management. 336: 148–162. https://doi.org/10.1016/j.foreco.2012.05.032

Powell, J.A.; Bentz, B.J. 2009. Connecting phenological predictions with population growth rates for mountain pine beetle, an outbreak insect. Landscape Ecology. 24: 657–672. https://doi.org/10.1007/s10980-009-9340-1

Pureswaran, D.S.; Roques, A.; Battisti, A. 2018. Forest insects and climate change. Current Forestry Reports. 4: 35–50. https://doi.org/10.1007/s40725-018-0075-6

Raffa, K.F.; Aukema, B.H.; Bentz, B.J.; [et al.]. 2008. Cross-scale drivers of natural disturbances prone to anthropogenic amplification: Dynamics of biome-wide bark beetle eruptions. Bioscience. 58: 501–518. https://doi.org/10.1641/B580607

Régnière, J.; St-Amant, R.; Duval, P. 2012. Predicting insect distributions under climate change from physiological responses: Spruce budworm as an example. Biological Invasions. 14: 1571–1586. https://doi.org/10.1007/s10530-010-9918-1

Robinet, C.; Roques, A. 2010. Direct impacts of recent climate warming on insect populations. Integrative Zoology. 5: 132−142. https://doi.org/10.1111/j.1749-4877.2010.00196.x

Robinet, C.; Rousselet, J.; Roques, A. 2014. Potential spread of the pine processionary moth in France: Preliminary results from a simulation model and future challenges. Annals of Forest Science. 71: 149–60. https://doi.org/10.1007/s13595-013-0287-7

Rocky Mountain Region, Forest Health Protection. 2010. Field guide to diseases & insects of the Rocky Mountain Region. Gen. Tech. Rep. RMRS-GTR-241. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 336 p. https://doi.org/10.2737/RMRS-GTR-241

Ryerson, D.E.; Swetnam, T.W.; Lynch, A.M. 2003. A tree-ring reconstruction of western spruce budworm outbreaks in the San Juan Mountains, Colorado, USA. Canadian Journal of Forest Research. 33: 1010–1028. https://doi.org/10.1139/x03-026

Safranyik, L.; Carroll, A.L.; Régnière, J.; [et al.]. 2010. Potential for range expansion of mountain pine beetle into the boreal forest of North America. The Canadian Entomologist. 142: 415–442. https://doi.org/10.4039/n08-CPA01

Schmid, J.M.; Frye, R.H. 1977. Stand ratings for spruce beetles. Research Note RM-309. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 4 p.

Seidl, R.; Thom, D.; Kautz, M.; [et al.]. 2017. Forest disturbances under climate change. Nature Climate Change. 76: 395–402. https://doi.org/10.1038/nclimate3303

Senf, C.; Campbell, E.M.; Pflugmacher, D.; [et al.]. 2017. A multi-scale analysis of western spruce budworm outbreak dynamics. Landscape Ecology. 32: 501–514. https://doi.org/10.1007/s10980-016-0460-0

Shaw III, C.G.; Kile, G.A. 1991. Armillaria root disease. Agricultural Handbook 691. Washington, DC: U.S. Department of Agriculture, Forest Service. 233 p. https://doi.org/10.1093/jof/103.6.280

Shaw, J.D.; Steed, B.E.; DeBlander, L.T. 2005. Forest inventory and analysis (FIA) annual inventory answers the question: What is happening to pinyon-juniper woodlands? Journal of Forestry. 103: 280−285.

Spaulding, P. 1922. Investigations of the white-pine blister rust. Bulletin 957. Washington, DC: U.S. Department of Agriculture, Forest Service. 100 p.

Stanton, S. 2007. Effects of dwarf mistletoe on climate response of mature ponderosa pine trees. Tree-ring Research. 63: 69–80. https://doi.org/10.3959/1536-1098-63.2.69

Sturrock, R.N.; Frankel, S.J.; Brown, A.V.; [et al.]. 2011. Climate change and forest diseases. Plant Pathology. 60: 133−149. https://doi.org/10.1111/j.1365-3059.2010.02406.x

Swetnam, T.W.; Betancourt, J.L. 1998. Mesoscale disturbance and ecological response to decadal climatic variability in the American Southwest. Journal of Climate. 11: 3128–3147. https://doi.org/10.1175/1520-0442(1998)011<3128:MDAERT>2.0.CO;2

Swetnam, T.W.; Lynch, A.M. 1989. A tree-ring reconstruction of western spruce budworm history in the Southern Rocky Mountains. Forest Science. 35: 962–986. https://doi.org/10.1093/forestscience/35.4.962

Swetnam, T.W.; Lynch, A.M. 1993. Multicentury, regional-scale patterns of western spruce budworm outbreaks. Ecological Monographs. 63: 399–424. https://doi.org/10.2307/2937153

Tainter F.H.; Baker, F.A. 1996. Principles of forest pathology. New York, NY: John Wiley & Sons, Inc. 805 p.

Thomson, A.J. 1979. Evaluation of key biological relationships of western spruce budworm and its host trees. BC-X-186. Victoria, BC: Environment Canada, Forestry Service, Pacific Forest Research Centre. 19 p.

Thomson, A.J.; Benton, R. 2007. A 90-year sea warming trend explains outbreak patterns of western sprue budworm on Vancouver Island. Forestry Chronicle. 83: 867–869.

Tishmack, J; Mata, S.A.; Schmid, J.M. 2005. Mountain pine beetle emergence from lodgepole pine at different elevations near Fraser, CO. Res. Note RMRS-RN-27. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 5 p. https://doi.org/10.2737/RMRS-RN-27

Tkacz, B.M.; Schmitz, R.F. 1986. Association of an endemic mountain pine beetle population with lodgepole pine infected by Armillaria root disease in Utah. Research Note INT-353. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 8 p.

USDA Forest Service [USDA FS]. 2013. Forest insect & disease conditions in the Rocky Mountain Region. Report R2-14-RO-32. Harris, J.L., comp. Lakewood, CO: USDA Forest Service, Rocky Mountain Region, State, Private Forests, Tribal Relations, & Forest Health Protection. 77 p.

Van Arsdel, E.P.; Riker, A.J.; Patton, R.F. 1956. The effects of temperature and moisture on the spread of white pine blister rust. Phytopathology. 46: 307–318.

Veblen, T.T.; Hadley, K.S.; Reid, M.S.; [et al.]. 1991. Methods of detecting past spruce beetle outbreaks in Rocky Mountain subalpine forests. Canadian Journal of Forest Research. 21: 242−254. https://doi.org/10.1139/x91-030

Wargo, P.M.; Shaw III, C.G. 1985. Armillaria root rot: The puzzle is being solved. Plant Disease. 69: 826–832. https://doi.org/10.1094/PD-69-826

Webb, W.L. 1981. Relation of starch content to conifer mortality and growth loss after defoliation by the Douglas-fir tussock moth. Forest Science. 27: 224–332. https://doi.org/10.1093/forestscience/27.2.224

Webb, W.L.; Karchesy, J.J. 1977. Starch content of Douglas-fir defoliated by the tussock moth. Canadian Journal of Forest Research. 7: 186–188. https://doi.org/10.1139/x77-026

Werner, R.A.; Holsten, E.H. 1985. Factors influencing generation times of spruce beetles in Alaska. Canadian Journal of Forest Research. 15: 438–443. https://doi.org/10.1139/x85-070

Werner, R.A.; Holsten, E.H.; Matsuoka, S.M.; [et al.]. 2006. Spruce beetles and forest ecosystems in south-central Alaska: A review of 30 years of research. Forest Ecology and Management. 227: 195–206. https://doi.org/10.1016/j.foreco.2006.02.050

West, D.R.; Briggs, J.S.; Jacobi, W.R.; [et al.]. 2016. Mountain pine beetle host selection between lodgepole and ponderosa pines in the southern Rocky Mountains. Environmental Entomology. 45: 127−141. https://doi.org/10.1093/ee/nvv167

Wickman, B.E. 1980. Increased growth of white fir after a Douglas-fir tussock moth outbreak. Journal of Forestry. 78: 31–33. https://doi.org/10.1093/jof/78.1.31

Williams, D.W.; Liebhold, A.M. 1995. Forest defoliators and climatic change: Potential changes in spatial distribution of outbreaks of western spruce budworm (Lepidoptera: Tortricidae) and gypsy moth (Lepidoptera: Lymantriidae). Environmental Entomology. 24:1–9. https://doi.org/10.1093/ee/24.1.1

Worrall, J.J.; Egeland, L.; Eager, T.; [et al.]. 2008. Rapid mortality of *Populus tremuloides* in southwestern Colorado, USA. Forest Ecology and Management. 255: 686–696. https://doi.org/10.1016/j.foreco.2007.09.071

Worrall, J.J.; Marchetti, S.B.; Egeland, L.; [et al.]. 2010. Effects and etiology of sudden aspen decline in southwestern Colorado, USA. Forest Ecology and Management. 260: 638–648. https://doi.org/10.1016/j.foreco.2010.05.020

Worrall, J.J.; Rehfeldt, G.E.; Hamann, A.; [et al.]. 2013. Recent declines of *Populus tremuloides* in North America linked to climate. Forest Ecology and Management. 299: 35–51. https://doi.org/10.1016/j.foreco.2012.12.033

Worrall, J.J.; Sullivan, K.F.; Harrington, T.C.; [et al.]. 2004. Incidence, host relations and population structure of *Armillaria ostoyae* in Colorado campgrounds. Forest Ecology and Management. 192: 191–206. https://doi.org/10.1016/j.foreco.2004.01.009

Wright, L.C.; Berryman, A.A.; Gurusiddajah, S. 1979. Host resistance to the fir engraver beetle, *Scolytus ventralis* (Coloeoptera: Scolytidae): Effect of defoliation on wound terpene and inner bark carbohydrate concentrations. Canadian Entomologist. 111: 1255–1262. https://doi.org/10.4039/Ent1111255-11

Xu, B.; Hicke, J.A.; Abatzoglou, J.T. 2019. Drought and moisture availability and recent western spruce budworm outbreaks in the Western United States. Forests. 10(4): 354. https://doi.org/10.3390/f10040354

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Fire Risk and Climate Change in the Colorado Front Range

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Introduction

Wildfires are natural disturbances that are fundamental in many ecosystems of the United States. However, when they burn intensely in places where human infrastructure exists, wildfires can cause catastrophic damage. The Colorado Front Range exemplifies this tension between fire-adapted ecosystems and human development. The native vegetation throughout most of the Front Range evolved over millennia with fire as an important process driving vegetation composition and structure (Romme et al. 2003). But the Front Range has also been a highly desirable place to live since European-American settlers first arrived in the mid-1800s (Veblen and Lorenz 1986), and the area has experienced rapid expansion of urban, suburban, and exurban development in the last century (Liu et al. 2015; Theobald and Romme 2007; Veblen and Lorenz 1986). Since 1989, the Front Range has experienced several wildfires that have become urban fire disasters (table 1).

The gravity of wildfire risk on the Colorado Front Range is underscored by the 2020 and 2021 fire seasons. The 2020 Cameron Peak Fire, now the largest wildfire in Colorado's recorded history, burned over 200,000 acres and destroyed 444 structures near the Canyon Lakes Ranger District on the Arapaho-Roosevelt (AR) National Forests. Also in 2020, the Calwood and Lefthand Fires together burned over 13,000 acres, with the Calwood destroying 26 homes near the Boulder Ranger District, and the East Troublesome Fire destroyed 580 structures mostly in the Grand Lake community, while burning almost 200,000 acres within the AR and surrounding lands before moving into Rocky Mountain National Park and jumping the divide. The weather that contributed to the 2020 fires was unusually hot, dry, and windy, and extended much further into the fall than is typical historically. Then, in late December of 2021, dry and very windy (some gusts strong enough to rate as hurricane speed) conditions drove the Marshall Fire in the span of hours, from grass wildlands into subdivisions where homes quickly became the fuel that carried the fire, resulting in two fatalities and the loss of 1,058 structures. Such conditions may become more common in the future. As climate change, insects and disease, human ignitions, and wildland-urban interface (WUI) expansion coalesce to exacerbate wildfire risk on the Colorado Front Range, agencies, organizations, communities, and individuals must work collaboratively and share the investment responsibility for preventive mitigation and planning.

Table 1—Fires that have caused substantial property loss or damage in the Colorado Front Range in recent decades.

Year	Fire name	Location	Final size (acres)	Number of structures lost	Approximate suppression cost (millions)	Approximate insured losses (millions, except as noted)
1989	Black Tiger	Near Boulder	2,100[a]	44[a]	$10[a]	$1[a]
2002	Hayman	Between Denver and Colorado Springs	138,114[b]	600[b]	$42[b]	$39[b]
2010	Fourmile Canyon	Near Boulder	6,181[c]	172[c]	$10[c]	$217[d]
2012	High Park	Larimer County	87,284[c]	371[c]	$38[c]	$114[d]
2012	Waldo Canyon	Colorado Springs	18,947[c]	347[c]	$2[c]	$454[d]
2013	Black Forest	Near Colorado Springs	14,280[c]	533[c]	$10[c]	$421[d]
2020	Cameron Peak	East of Walden	208,913[c]	444[c]	$133[c]	$71[d]
2020	East Troublesome	Near Granby	193,812[c]	580[c]	$16[c]	$543[d]
2021	Marshall	Near Superior	6,080[c]	1,058[c]	$2[c]	$2 billion[d]

[a]Black Tiger Fire Case Study
[b]Hayman Fire Case Study
[c]ICS-209 reports
[d]Rocky Mountain Insurance (http://www.rmiia.org/catastrophes_and_statistics/Wildfire.asp, accessed 10 November 2022)

As indicated in Front Range Forest Service land and resource management plans, fire managers recognize the potential benefit of allowing wildfire to play a natural role in ecosystem dynamics when and where fire does not threaten human life and property. Managers also recognize the importance of managing fuel conditions by removing fuel mechanically or by applying prescribed fire within and around the WUI, near homes and infrastructure. They take initiative to propose and secure funding to prepare and implement fuel treatments with diverse partners across land ownership boundaries (see, for example: https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fseprd680611.pdf, a report that describes a collaborative prescribed fire implementation effort designed to foster resilient landscapes and reduce wildfire risk to communities in the northern Front Range). To satisfy desired conditions of reduced fire hazard, lands adjacent to private property and infrastructure would have dispersed patterns of fuel types and arrangements (redundant natural and human-made barriers) designed to moderate

wildfire behavior, reduce the rate of fire spread in and around communities, and provide safe locations for firefighter suppression efforts should a wildfire move into the WUI. Under these conditions, wildfire, should it occur, would be constrained to surface fuels, resulting in short flame lengths, and reducing the potential for fire behavior that lofts fire brands onto nearby structures and limits suppression effectiveness. Fire and natural resource managers, however, face many complex challenges in realizing these conditions in the WUI. For example, although wildfire plays an important role in ecosystem dynamics and reduces fuel loads, a summary of fire occurrence from Short (2022) reveals that from 1992 to 2020, 99 percent of fires in the Front Range were suppressed before growing to 300 acres. Additionally, varying levels of wildfire risk awareness and diverse private landowner privacy and aesthetic preferences complicate the challenge of implementing fuels mitigation work. And while wildfire hazard can be reduced through well-planned and targeted fuels treatments, the vulnerability of structures is highly dependent upon the condition of the area directly surrounding them, making homeowner responsibility another critical factor in efforts to reduce overall WUI wildfire risk.

In this chapter, we address wildfire risk on the Colorado Front Range and investigate ways in which risk is developing due to changes in vegetation, fuel, climate, and the expansion of the WUI. We begin with a definition of wildfire risk, followed by an examination of the factors contributing to each component of risk, and an evaluation of how each component may change in the future. We then summarize current wildfire exposure and risk using several available spatial data resources. Finally, we discuss mitigation strategies that target different components of wildfire risk.

Quantifying and mapping wildfire risk in the WUI is an important step that can help managers, local leaders, and the public understand potential wildfire scenarios and take actions to avoid detrimental outcomes. We structure our discussion around the quantitative wildfire risk assessment framework following concepts first outlined by Finney (2005). This framework couples probabilistic fire behavior outputs, calculated using stochastic fire behavior modeling systems, with a characterization of potential fire impacts on specific resources and assets to produce a robust wildfire risk metric. The national cohesive strategy science report (Calkin et al. 2011) identified this framework as the best available science-based decision tool for fire and fuel management across spatial scales. We follow terminology aligned with the established quantitative wildfire risk assessment framework as articulated by Scott et al. (2013) and Thompson et al. (2016) to explore how wildfires potentially impact life, property, and valued resources in the Colorado Front Range.

Wildfire Risk Defined

Thompson et al. (2016) define risk as: “A measure of the probability and consequence of uncertain future events.” In the context of wildfire, the uncertain future events are fires. Adapting terms from Scott et al. (2013), we define wildfire risk as the combination of wildfire hazard and vulnerability (fig. 1).

Wildfire risk = Wildfire Hazard * Vulnerability

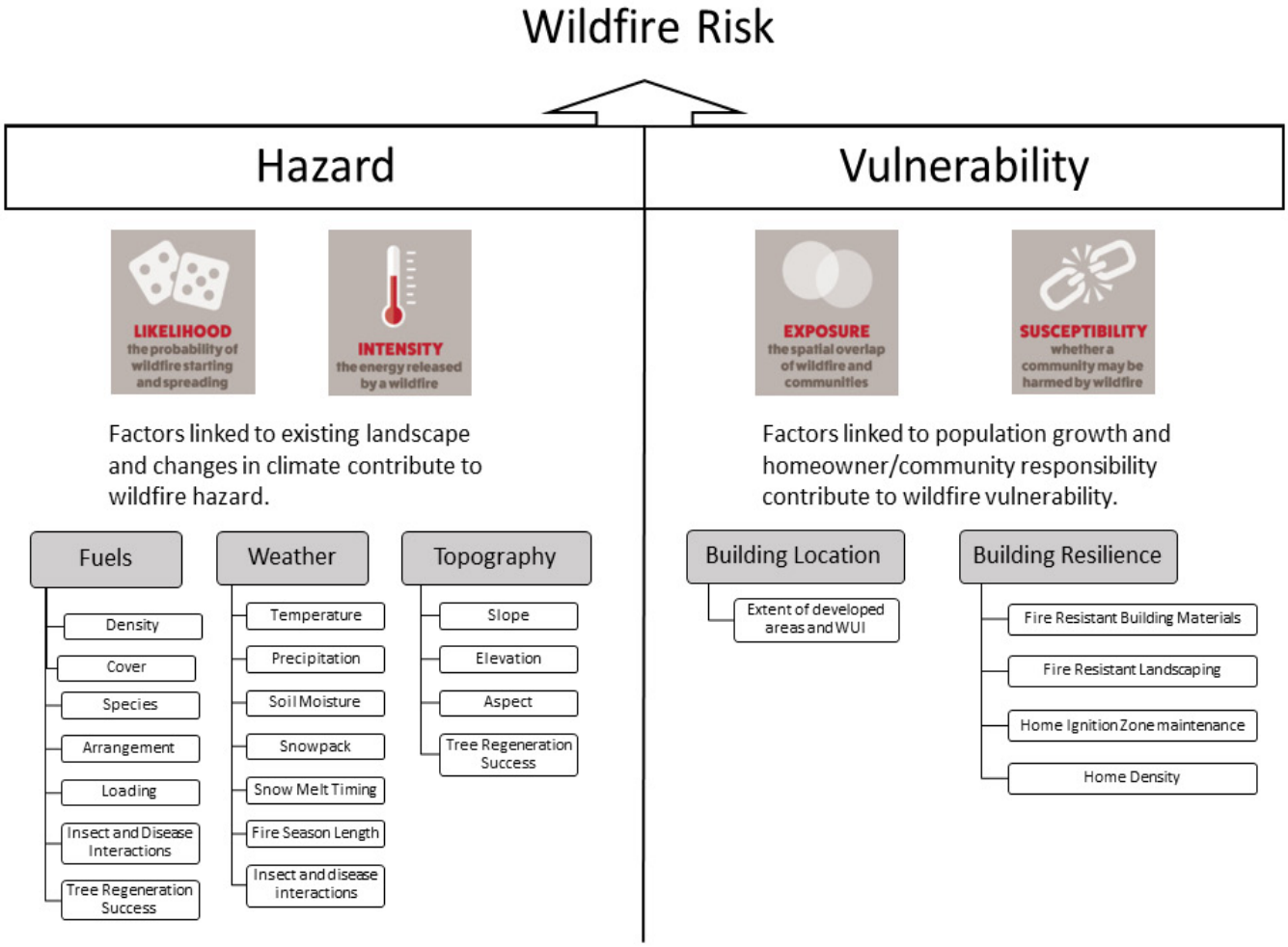


Figure 1—Components of wildfire risk.

Wildfire hazard is the potential for wildfires to cause damage, loss or harm to people, infrastructure, equipment, natural resources, or property. It incorporates both the likelihood of wildfire exposure and the energy released by the fire (or the fire intensity). Vulnerability is a measure of how an asset or resource might be impacted by a fire, based on its exposure to the hazard and how susceptible it is to damage at different

intensities. For example, consider a home located in a forest that has a high likelihood of burning in a high-intensity fire, but the homeowner has worked to make improvements to the structure and surrounding property (i.e., building with fire-resistant roofing and siding, installing screening on home vents, removing burnable material directly surrounding structures) that make the home less susceptible to the impact of direct flame impingement or lofted embers. This home is exposed to wildfire because of its location, and the wildfire hazard is high. The home's wildfire risk, however, may be only moderate because its susceptibility to damage is low.

Note that in this chapter we focus on risk to specific resources and assets on the landscape that are important to communities and land management agencies. Operational risk, which addresses the health and safety of responders, is also very important, but is not explicitly addressed here.

Wildfire Hazard in the Front Range and How It May Change

Wildfire hazard is determined by topography, fuels, and weather. Topography describes the shape of landscape features (including elevation, aspect, and slope of the land surface) and is a relatively static factor. Fuels are the components of vegetation that can burn, from the ground to the highest treetops. They include both dead material, such as litter, duff, and branches, and living vegetation, such as grasses, shrubs, and trees. Fuels are described by type, amount, and configuration. Finally, weather is important in the dynamics of how fires ignite and spread; temperature, precipitation, cloudiness, and humidity affect fuel moisture, while atmospheric stability and wind impact rate and direction of fire spread. The interaction between topography, fuels, and weather drives ignition and spread patterns of a wildfire, ultimately determining wildfire likelihood and intensity. Over long time periods, there are feedbacks between fuels and climate; these have shaped Front Range forests historically (Addington et al. 2018; Battaglia et al. 2018; Brown et al. 2015; Sherriff et al. 2014). How these components each change in the future, separately and in conjunction, will dictate future wildfire hazard.

Decades of land use and vegetation management have altered the configuration of fuels across Front Range landscapes. The practice of active wildfire exclusion via suppression by land management agencies throughout the 20th century has changed fuel loads and vegetation characteristics of forests (Battaglia et al. 2018; Hessburg et al. 2019), affecting some forest types more than others (Schoennagel et al. 2004). Especially in lower-elevation ponderosa pine and dry mixed-conifer forests in the Front Range, fire exclusion has caused ecological changes, with ramifications for wildfire hazard: increased forest density, fewer forest openings, reduced stand heterogeneity (structure diversity and patchiness), and an increase in shade-tolerant species creates more uniform fuel conditions conducive for fire to spread horizontally (across the landscape) and vertically (from surface into tree crowns) (Addington et al. 2018).

The changes in vegetation and fuel conditions have led to consequential shifts in fire regimes. The term "severity" is typically used when describing effects of fire retrospectively or conceptually defining fire regimes (Morgan et al. 2014). Severity is an ecological concept closely linked to fire intensity, and we can make inferences about shifts in intensity, and therefore hazard, by considering shifts in the severity component of fire regimes. Historically in Front Range forests, low- and moderate-severity fires occurred more frequently than today (Addington et al. 2018). Less frequent, mixed- or high-severity fires occurred in high-elevation subalpine forests (Buechling and Baker 2004; Schoennagel et al. 2004), and a mixed-severity fire regime was dominant in the northern Colorado Front Range (McKinney 2019; Sherriff et al. 2014). Predicted fire behavior continues to indicate a mixed-severity fire regime in some places, but with climate-driven changes that could have important ecological consequences. In some respects, recent fires in the Front Range are not fundamentally different from historical events under extreme weather conditions (Sherriff et al. 2014), but there is evidence that Front Range fires in recent decades have had an increasing proportion of high severity (Dillon et al. 2011) and that high-severity patch sizes are larger (Davis et al. 2019; Fornwalt et al. 2016; Stevens-Rumann and Morgan 2019). In lower-elevation ponderosa pine and mixed-conifer forests that have gone a century or more without fire, this can mean that fires burning under extreme weather conditions today result in large areas of tree mortality that do not regenerate into new forests after fire (Chambers et al. 2016; Davis et al. 2019; Rodman et al. 2020; Stevens-Rumann and Morgan 2019; Walker et al. 2018).

The changes to wildfire hazard with changing climate and land management practices will likely not be unidirectional. Where shifts toward higher-severity fire regimes occur, we can expect the potential for high-intensity wildfire to increase for a period, leading to increased hazard. Where high-severity fire leads to complete shifts in vegetation type, however, expected fire intensity in subsequent fires will depend on characteristics of the newly established vegetation. In some cases, this may mean decreased potential for high-intensity fire and decreased hazard. We explore some specific considerations around wildfire intensity and probability related to climate change, insects and disease, and human influences in the sections that follow.

Climate Change and Fire

Across the western United States, climate change in recent decades has altered the weather patterns that influence how wildfires burn and the type of vegetation capable of growing back after a fire. Increased fire size and fire severity may largely be explained by extreme weather conditions (98–99th percentile wind speed and relative humidity), when suppression efforts are often unsuccessful (Finney 2020; Sherriff et al. 2014). Increased temperatures, decreased precipitation, and earlier snowmelt have all contributed to drier fuel conditions conducive to longer fire seasons and larger fires (Abatzoglou and Williams 2016; Flannigan et al. 2013; Jolly et al. 2015; Litschert et al. 2012; Littell et al. 2009; Spracklen et al. 2009; Vose et al. 2018). Wildfire activity is

expected to increase in frequency (Westerling et al. 2006) and severity (Flannigan et al. 2013) due to climate warming. By the middle of this century, the annual area burned in the western United States could increase two to six times (Vose et al. 2018). From the perspective of wildfire hazard, this means that many places may experience an increase in the likelihood of wildfire and more days annually above critical fire weather thresholds that can produce higher intensity fire behavior (Riley and Loehman 2016).

Changes in climate will very likely influence wildfire activity and wildfire hazard in the southern Rocky Mountains, including the Colorado Front Range. Similar to other areas in the western United States, increasing temperatures and changing precipitation patterns are expected to cause drier fuel conditions (Addington et al. 2018; Liu et al. 2013; Lukas et al. 2014; Rocca et al. 2014), likely leading to longer fire seasons and increases in fire frequency and burned area (Litschert et al. 2012; Littell et al. 2009; Liu et al. 2015; Stavros et al. 2014). Fire activity may vary by vegetation type (Littell et al. 2009), with a greater impact expected in ponderosa pine and dry mixed-conifer forests (Rocca et al. 2014). However, Schoennagel et al. (2004) suggest that moisture conditions are an important influence on fire frequency and severity in spruce-fir forests, implying that drier conditions due to a changing climate may increase fire activity in those areas as well. Climate change may generate more frequent or widespread extreme fire weather events, increasing high fire danger days by several days or weeks (see Chapter 3 in this report).

Wildfire hazard in a changing climate will likely also be impacted by landscape-scale changes in vegetation. In most forests across the western United States, the climate under which mature trees established was different from the current conditions (Rodman et al. 2020). Disturbance events like wildfires become catalysts for change, and the vegetation that grows after a fire is often different from what existed prefire (Davis et al. 2020; Parks et al. 2019). Particularly in forests like lower-elevation ponderosa pine and mixed-conifer forests in the Front Range that are at the edge of their environmental tolerance, evidence already exists for decreased tree regeneration following fires in recent decades (Davis et al. 2019; Stevens-Rumann and Morgan 2019; Stevens-Rumann et al. 2018). This is due to climate conditions such as vapor pressure deficit, soil moisture, and soil surface temperature that influence the success of tree seedling establishment (Davis et al. 2019). According to climate change projections for the mid-21st century, 18–61 percent of these “trailing-edge forests” in the intermountain western United States, including the Front Range, are at risk of conversion to nonforest, and 6–30 percent of all forests in this region are at risk of conversion (Parks et al. 2019). As vegetation composition on the landscape changes, so too will the components of wildfire hazard.

Insects and Disease

Insects and disease, which are sometimes tied to climate change (Kolb et al. 2016), will play a role in changing wildfire hazard in Colorado. Insects with the most notable impact on Front Range forests are mountain pine beetle (*Dendroctonus ponderosae*) and spruce beetle (*Dendroctonus rufipennis*). Since 1996, extensive mountain pine beetle outbreaks have affected lodgepole pine and ponderosa pine forests in Colorado (Negrón and Cain 2019). A later outbreak of spruce beetle is continuing to progress across the State; 2021 was the tenth consecutive year that spruce beetle caused the most widespread destruction among insect agents (CSFS 2021). Early after an outbreak of either beetle, needle flammability (and thus crown fire potential) is expected to increase (Jolly et al. 2012; Page and Jenkins 2007; Page et al. 2014; Schoennagel et al. 2012). Years after initial outbreak, fuel accumulation (fallen needles, branches, and trees) and increased wind speeds may increase surface fire intensity due to changes in stand structure and canopy fuels (Klutsch et al. 2009; Page and Jenkins 2007), but specific effects on crown fire potential are uncertain (see Negrón et al. 2018).

Another important ramification of beetle outbreaks is that standing and downed dead trees can jeopardize firefighter safety and complicate wildfire suppression efforts. Firefighters report unexpected intense fire behavior in forests in all phases of mountain pine beetle attacks during Colorado wildfire events, prompting changes in fire suppression tactics (Moriarty et al. 2019). Weak, dead trees threaten to fall on firefighters, and downed trees impede travel, increasing firefighter exposure, and slowing fire operations (Page et al. 2013).

With climate change anticipated to increase mountain pine beetle and spruce beetle activity (Bentz et al. 2010; Raffa et al. 2013), the potential for increased tree mortality has implications for future wildfire behavior and hazard. Though uncertainty exists around the specifics of how climate change may interact with insect and disease outbreaks to affect wildfire, land managers can investigate potential local climate effects to estimate a range of possible futures and to guide mitigation strategies. A variety of tools are available for such efforts, including downscaled climate scenarios (Lukas et al. 2014), bioclimate envelope models (Worrall et al. 2016), ecosystem process models (Loehman et al. 2020), and state and transition models (Daniel and Frid 2012).

Fire Occurrence and Human Ignitions

Ignitions and ignition sources critically influence wildfire likelihood and therefore represent another important component of wildfire hazard. Across the United States, human-caused ignitions have expanded the fire season to triple the length of the lightning-caused fire season, and account for 84 percent of all wildfires and 44 percent of total area burned (Balch et al. 2017).

Historically, both human and lightning-caused ignitions have shaped fire regimes of the Front Range. Native Americans are known to have intentionally used fire for land

management purposes (Simmons 2000), but the arrival of Euro-American settlers in the late 1800s brought increased wildfire activity due to ignitions associated with settlement activity (Rodman et al. 2019). Successful fire prevention and suppression, combined with less favorable climate conditions for ignition, led to a drop in fire activity through much of the 20th century, but the number of ignitions and area burned have been on the rise again since the mid-1980s (Rodman et al. 2019). Human-caused ignitions, along with expanding human population and climate change, have been major factors in this rise. Summary of fire occurrence data from Short (2022) reveals that within the 16 counties encompassing the Front Range urban corridor, humans caused 73 percent of fires that occurred from 1992 to 2020, accounting for 71 percent of the area burned. Also, ignitions are more likely in areas with more human activity, and fires that start closer to where people live have a higher likelihood to impact communities. Humans caused 97 percent of wildfires in the WUI between 1992 and 2015 (Mietkiewicz et al. 2020), and simulated fires starting from ignitions on private land were more likely to result in fires that impact people than ignitions on Federal land (Haas et al. 2015). Human-caused fire plays a significant role in fire hazard in the Front Range, and regional analyses of spatial and temporal human ignition patterns could inform fire-safe public education strategies.

Wildfire Vulnerability in the Front Range and How It May Change

Assessing the vulnerability of resources and assets to wildfire is just as critical to any evaluation of wildfire risk as the characterization of hazard. From the perspective of a land management agency, the resources or assets of interest for a quantitative wildfire risk assessment may include resource areas identified in land and resource management plans (i.e., timber stands, threatened and endangered species, historically significant structures, or populated areas). However, a State, county or municipal entity may focus primarily on human life and property. For example, Boulder County Fire Management's mission is to "provide efficient, effective public safety services to the residents and visitors of Boulder County." Just as hazard is made up of the two subcomponents of wildfire likelihood and intensity, vulnerability too, has two subcomponents: exposure and susceptibility. Unlike hazard, however, where driving factors are biophysically based and linked to climate conditions, the drivers of vulnerability are primarily social. In the case of wildfire risk in the Front Range WUI, exposure has to do with where homes and other assets associated with human life and property overlap with areas of wildfire hazard. Susceptibility, then, refers to how easily those homes and other assets will ignite and incur damage.

Where Homes Are Exposed to Wildfire: The Wildland-Urban Interface

Homes, property, and other susceptible assets in the built environment are exposed to wildfire in the wildland-urban interface (WUI). Ultimately, human lives are at risk when the places where people live are exposed to wildfire. The Federal Register defines the WUI as "the area where humans and their development meet or intermix with wildland fuel" (USDA and USDOI 2001). The WUI contains 1 housing unit per 0.3

to 40 acres; about 10 percent of the land area and about one-third of all housing units in the conterminous United States are located in the WUI (Radeloff et al. 2018). The WUI includes both places where homes are directly embedded in a matrix of wildland vegetation, known as "intermix," and where homes may not be immediately adjacent to wildland fuels but are within a distance where embers from wildfire can still be a threat, known as "interface" (Martinuzzi et al. 2015). Houses located anywhere in the WUI will be exposed to wildfire, given sufficient vegetation to carry fire, resulting in a tension between periodic fire that occurs in ecosystems and the need to minimize negative impacts to people and their homes from wildfire. Total WUI area in the United States increased by 33 percent from 1990 to 2010 and 50 percent between 1970 and 2000 (Radeloff et al. 2018; Theobald and Romme 2007). Projections of the future WUI require information about migration rates, which will be affected by climate change, and different socioeconomic and demographic factors, resulting in uncertainty (EPA 2010); however if land-use regulations do not change, development in the Front Range is expected to occur in areas of high wildfire hazard (Platt et al. 2011).

Colorado is one of the States with greatest recent population expansion and expected future expansion into the WUI. The number of WUI homes increased by 70 percent or more in Colorado during 1990 to 2010 (Radeloff et al. 2018). As of 2017, 2.9 million people, or half of the State's total population, live in Colorado's WUI, compared to 2 million people during 2013 (some increase may be due to changed source data; CSFS 2018). A 20 percent increase in Colorado's population is expected between 2000 and 2030; Theobald and Romme (2007).

Susceptibility of Structures to Wildfire

If a wildfire occurs in the WUI, a structure will have a certain level of exposure, based on its proximity to varying levels of fire intensity, but the degree of damage it incurs will also greatly depend on its susceptibility to fire. Contrary to the common expectation that most home destruction results from direct flame impingement from high-intensity crown fires, wildfire damage to most homes and structures is actually due either to low-intensity surface fires creeping through vegetation in contact with the structure, or to wind-borne embers igniting after landing on a roof or wooden deck or blowing in through vents (Cohen 2000). Investigations of several Front Range fires (Fourmile Canyon, Graham et al. 2012; Hayman, Graham 2003; Waldo Canyon, Quarles et al. 2013) indicate that home destruction is commonly driven by local flammability conditions, including the type, structure, and configuration of vegetation surrounding the home, along with the building materials used in home construction.

Given these lessons learned from previous WUI "disasters," homeowner mitigation activities recommended to reduce home susceptibility to wildfire include reducing flammable vegetation and other wildfire hazards in the zone directly surrounding individual structures ("home ignition zone"; HIZ [Cohen 2010; Cohen and Stratton 2008), and using fire-resistant building materials and techniques. Recommended actions include routine home and yard maintenance: clearing trees and brush around

homes; cleaning roofs and gutters; stacking firewood away from structures; and moving propane tanks at least 30 feet from away from structure, and "home-hardening" improvements: installing double paned, tempered glass windows; screening entry holes (e.g., roof vents); and using nonflammable building materials for decks, roofs, and siding.

Wildfire Risk in Front Range Landscapes: Risk Analysis Reveals Areas of Concern

Wildfire risk assessment and associated risk analytics are recognized as tools to aid land managers with their responsibility of addressing wildfire hazard within their jurisdictions, and to enable collaboration among multiple agencies to reduce risk across ownership boundaries. The U.S. Government Accountability Office advised that Federal agencies use risk analysis for making "fully informed" decisions about wildfire risk-reduction projects (US GAO 2004). A risk-based approach forms the spatial framework for the National Cohesive Wildland Fire Management Strategy (Calkin et al. 2011). The risk assessment framework is scalable, meaning the same methodology can be applied to analysis areas ranging from small to large (e.g., from ranger districts to regions, or even national extents), with the finest resolution input data available for the area of interest producing the most reliable results (Calkin et al. 2011; Scott et al. 2013). Because wildfire risk maps depict areas that are more likely to incur damage from wildfire, managers can use them to identify and prioritize hazardous fuel reduction activities (Addington et al. 2020; Scott et al. 2016; Thompson et al. 2013; Thompson et al. 2016), measure the effects of fuel treatment on WUI and other resources (Ager et al. 2010), or compare management strategies in fire prone landscapes (Ager et al. 2012).

Given current levels of wildfire hazard and vulnerability, there is potential for fire to cause substantial damage to people and communities in the Front Range, where 85 percent of Colorado's population lives. Researchers and fire management professionals at both the Federal and State level have leveraged advancements in analytical tools to evaluate wildfire exposure and risk to Front Range communities in recent years. Haas et al. (2015) used fire behavior modeling to simulate wildfire ignition and spread under severe, but not uncommon, weather events in the 11 Front Range counties that contain large areas of USDA Forest Service lands. Their goal was to assess the potential for exposure to human populations from wildfires igniting in different locations within these 11 counties. The resultant product depicts the absolute population (aPOP) directly exposed to wildfire. Their analysis highlighted that even though fire sizes are often smaller near places with higher population density, those fires can still result in considerable exposure to populated areas. Also, their findings suggest that communities in El Paso, Douglas, and Jefferson Counties have relatively more wildfire risk than elsewhere in the Front Range (Haas et al. 2015).

Ager et al. (2019) evaluated wildfire exposure and transmission across ownership boundaries for the entire western United States. Their methods built upon fire behavior modeling conducted in a consistent manner across the United States that simulates fire ignition and spread across a full range of statistically possible weather scenarios (not

just severe weather) (Finney et al. 2011; Short et al. 2020). Similar to Haas et al. (2015), they found that in Colorado most wildfire exposure to human populations is from fires igniting on private lands or local community jurisdictions, and only a minority of exposure comes from fires starting on Federal lands. Relative to other western states like California and Arizona with very high levels of community wildfire exposure, relatively few (only two) Colorado communities were ranked by Ager et al. (2019) in the top 100 communities exposed to wildfire in the western United States; both were in the Front Range: Evergreen (rank = 90) and Colorado Springs (rank = 99).

Spatial datasets depicting wildfire risk in Colorado have also been produced for the U.S. Forest Service Rocky Mountain Region (Vogler et al. 2022) and the Colorado State Forest Service (CSFS 2018). These datasets integrate components of wildfire hazard and vulnerability in a spatial quantitative wildfire risk assessment framework. The CSFS products are available to the public online through the Colorado Wildfire Risk Assessment Portal (COWRAP; https://csfs.colostate.edu/wildfire-mitigation/colorado-forest-atlas/).

Another suite of online wildfire risk products is part of a national Forest Service project called Wildfire Risk to Communities (WRC; http://wildfirerisk.org/, downloadable datasets also available [Scott et al. 2020]). One of the WRC products, Risk to Homes, is a map that combines wildfire likelihood and intensity from the same fire behavior modeling (Short et al. 2020) used by Ager et al. (2019), with home vulnerability described by a generalized estimation of fire impact to homes. The Risk to Homes map characterizes the relative risk to a home, if one existed at every 30-meter pixel on the landscape, and generally depicts where WUI fire risk concentrates within the Front Range (fig. 2).

Each wildfire hazard and risk product available in Colorado was developed for a specific purpose and could thus explain a different part of the overall wildfire risk picture. The COWRAP Wildfire Risk Map is a composite metric, combining wildfire hazard with a suite of highly valued resources and assets including: WUI, drinking water importance areas, riparian areas, and forest assets (fig. 3). This map could be useful for land management agencies who must consider the relative contribution of different assets (beyond homes and structures) to a comprehensive risk profile. COWRAP also provides a WUI Wildfire Risk Index map (fig. 4) intended to specifically portray risk to homes and people as a single value, akin to the aPOP (mapped in fig. 5) and Risk to Homes metrics, and in contrast to the Wildfire Risk Map which integrates multiple values. Some difference in percent area of risk and exposure as measured by the different metrics is expected; each metric was calculated using different source data, objectives, and underlying assumptions. The COWRAP Wildfire Risk covers much more of the total forest (and buffered) areas because it is mapped across all areas (populated and uninhabited), whereas the WUI Wildfire Risk Index and aPOP metrics are constrained to populated areas. The Risk to Homes product is mapped across all areas but only considers risk to one type of asset (homes) at every pixel.

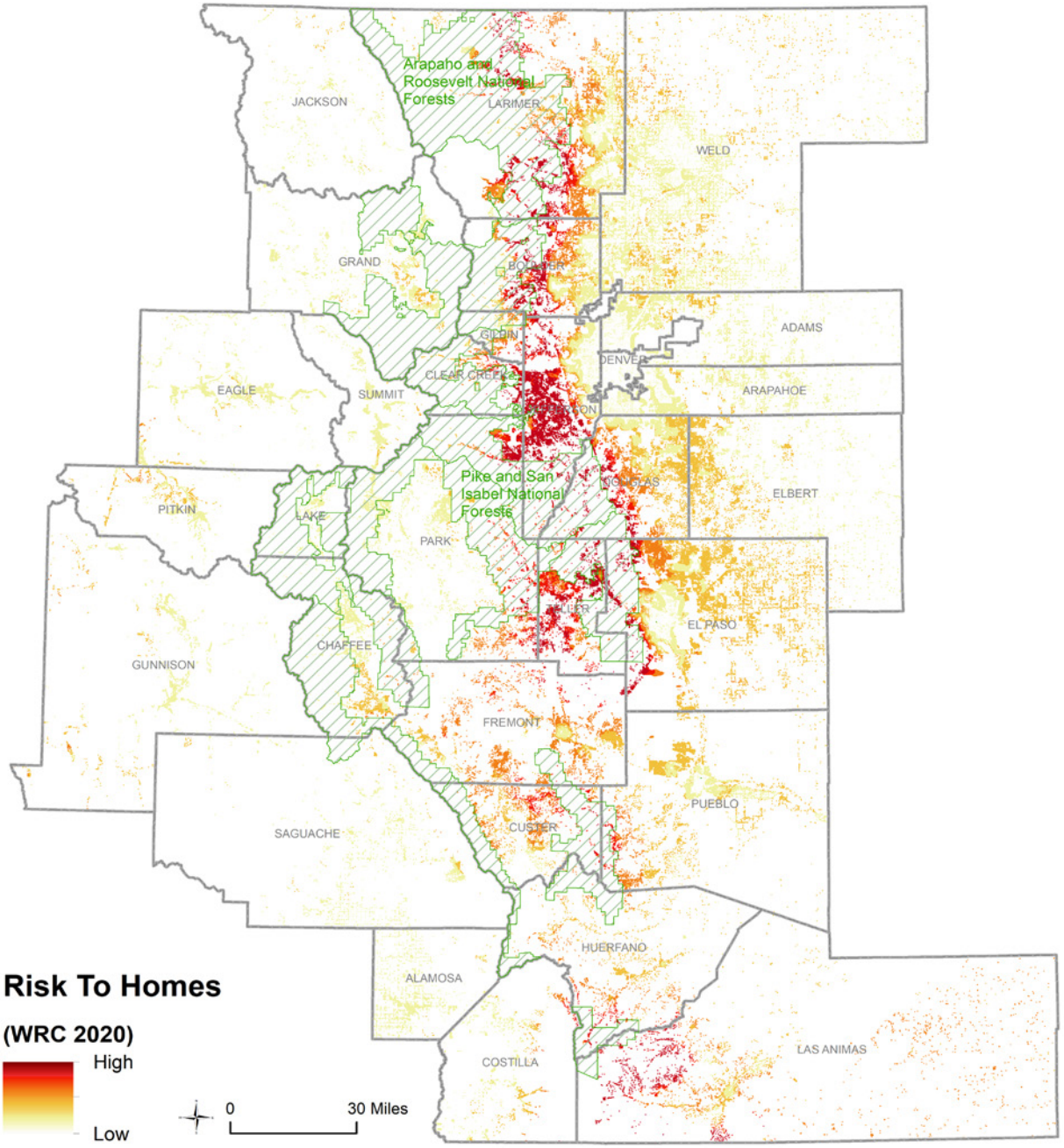


Figure 2—Wildfire Risk to Communities Risk to Homes (Scott et al. 2020) mapped where housing unit density is greater than zero in the Colorado Front Range.

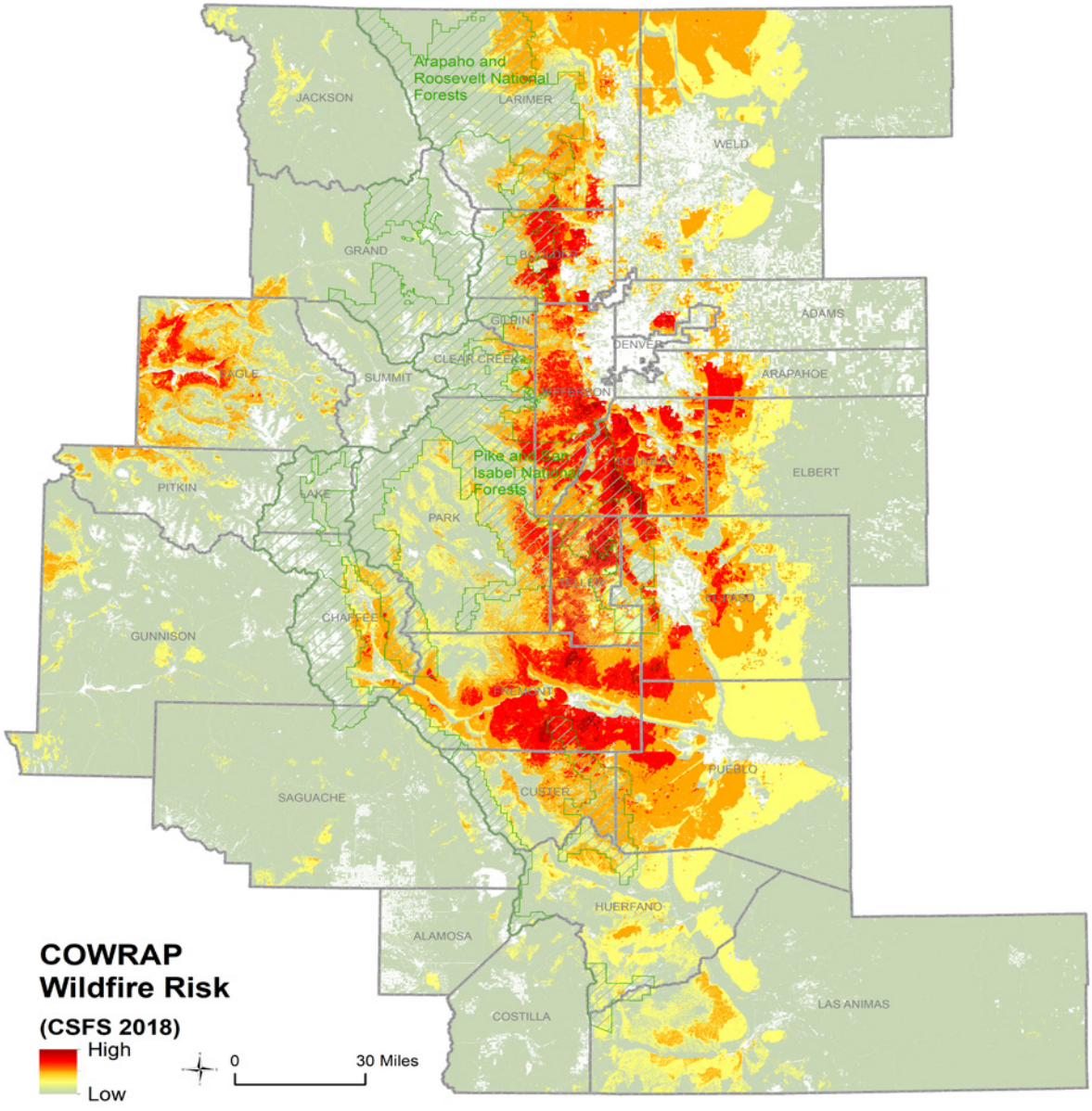


Figure 3—Colorado Risk Assessment Wildfire Risk (CSFS 2018; 2024) mapped in the Colorado Front Range.

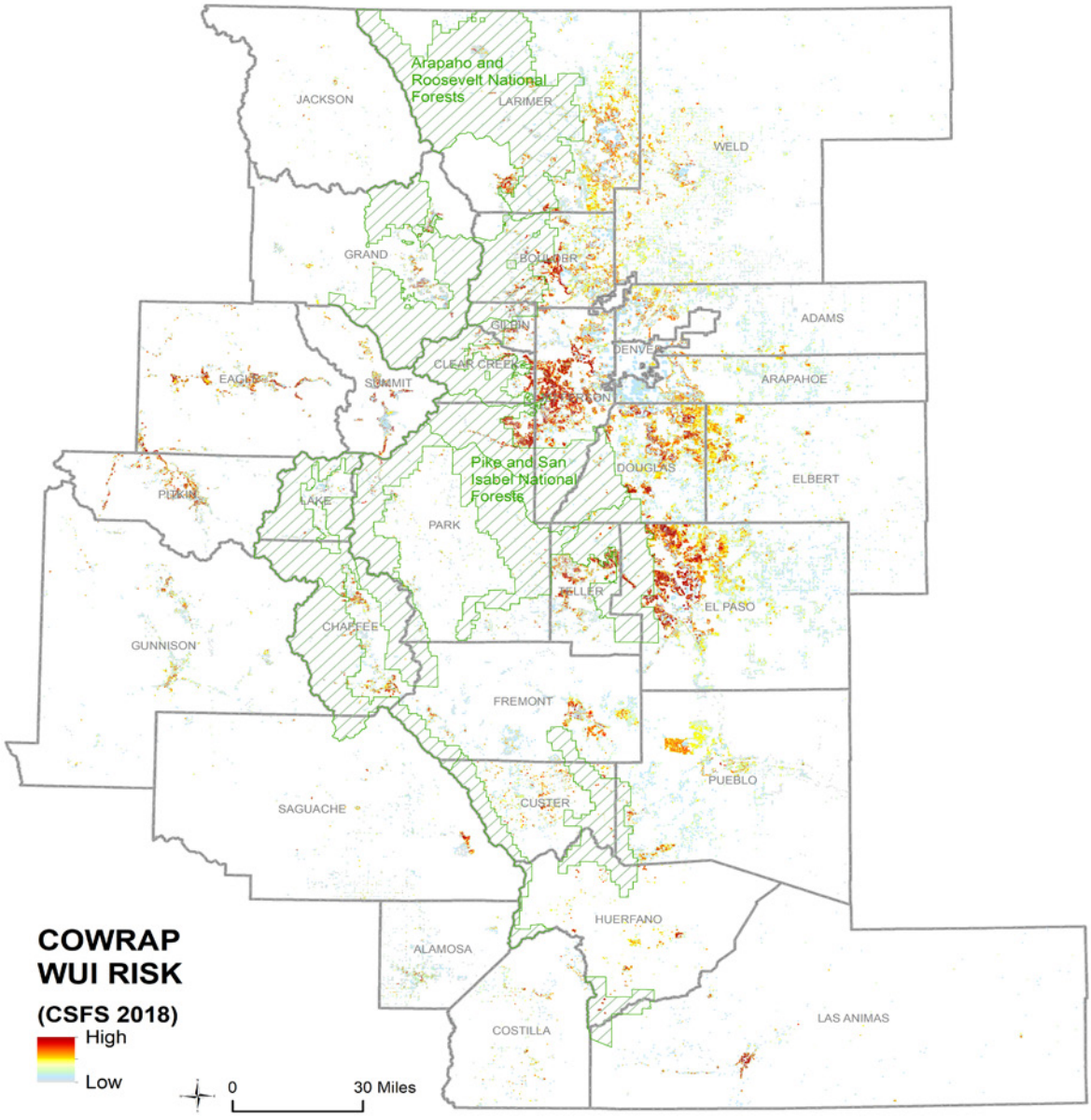


Figure 4—Colorado Risk Assessment WUI Wildfire Risk (CSFS 2018; 2024) mapped in the Colorado Front Range.

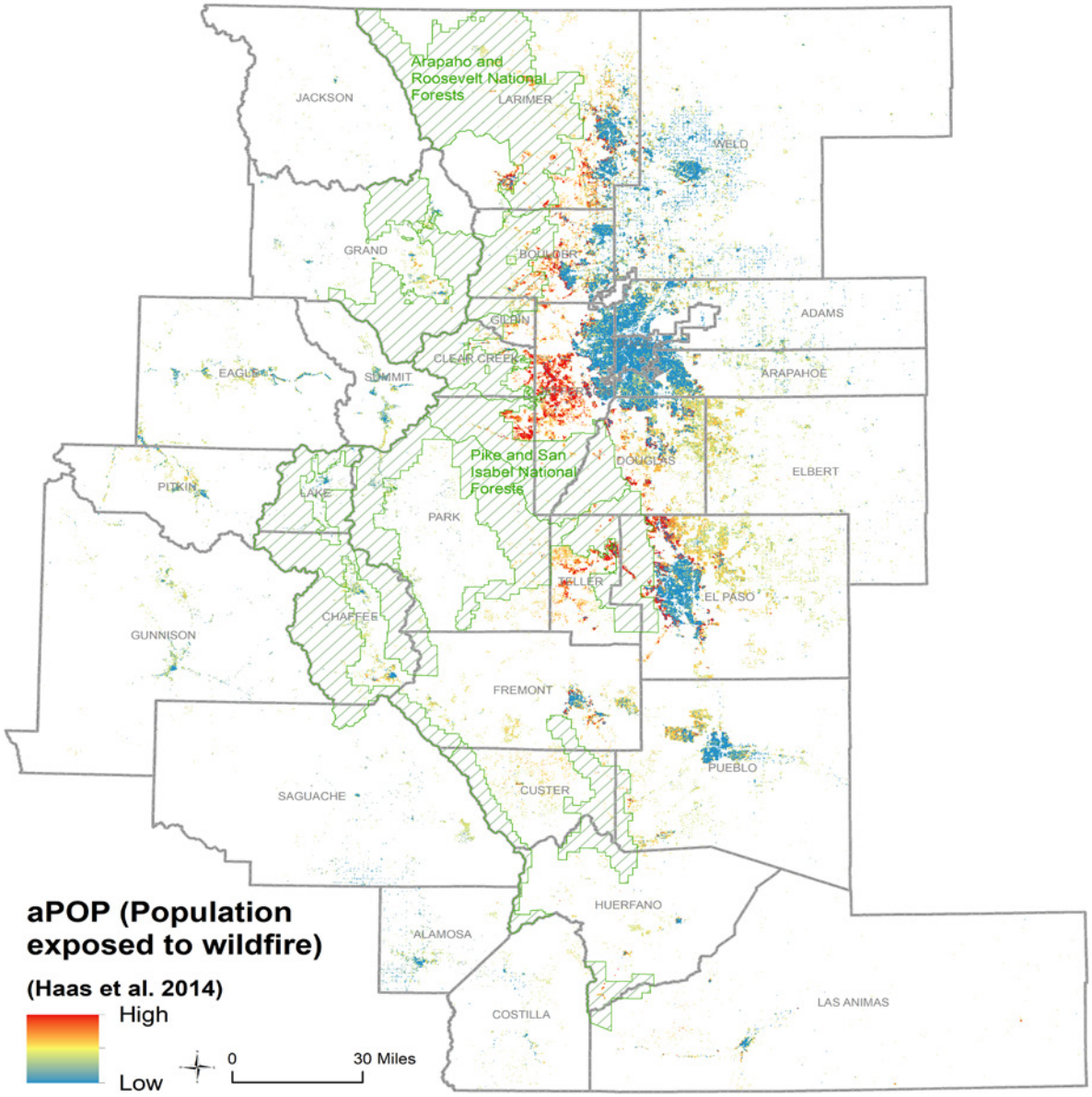


Figure 5—Absolute population directly exposed to wildfire (aPOP; Haas et al. 2015) mapped in the Colorado Front Range.

Concern about hazardous fuels conditions across Front Range landscapes can prompt managers to target specific areas for hazard reduction treatments. The COWRAP Fire Intensity Scale (FIS) map (fig. 6) is a spatial data product that maps potential wildfire intensity to indicate where fuel conditions could contribute to destructive fire behavior. Natively, the COWRAP FIS includes five intensity classes, from class 1 representing very low intensity where fires are easy to suppress, to class 5 representing very high intensity where fires are very difficult/dangerous to suppress and likely to cause damage. For consistency with other metrics in our tabular summary, we binned the

original five COWRAP FIS values into three classes: “low” (flames less than 2 feet; includes COWRAP FIS classes 1 and 2); “moderate” (flames up to 8 feet with short-range spotting possible; includes COWRAP FIS class 3); and “high” (flames up to 150 feet with short-range spotting common or profuse, and possibly frequent long-range spotting; includes COWRAP FIS classes 4 and 5). According to the COWRAP FIS data as classified (table 2), much of the Front Range exceeds the desired condition of low-intensity surface fire behavior as indicated in land and resource management plans, as 40 to 50 percent of area in and around the AR and PSI NFs could experience high wildfire intensity, with flame lengths between 8 and 150 feet and medium to long range spotting.

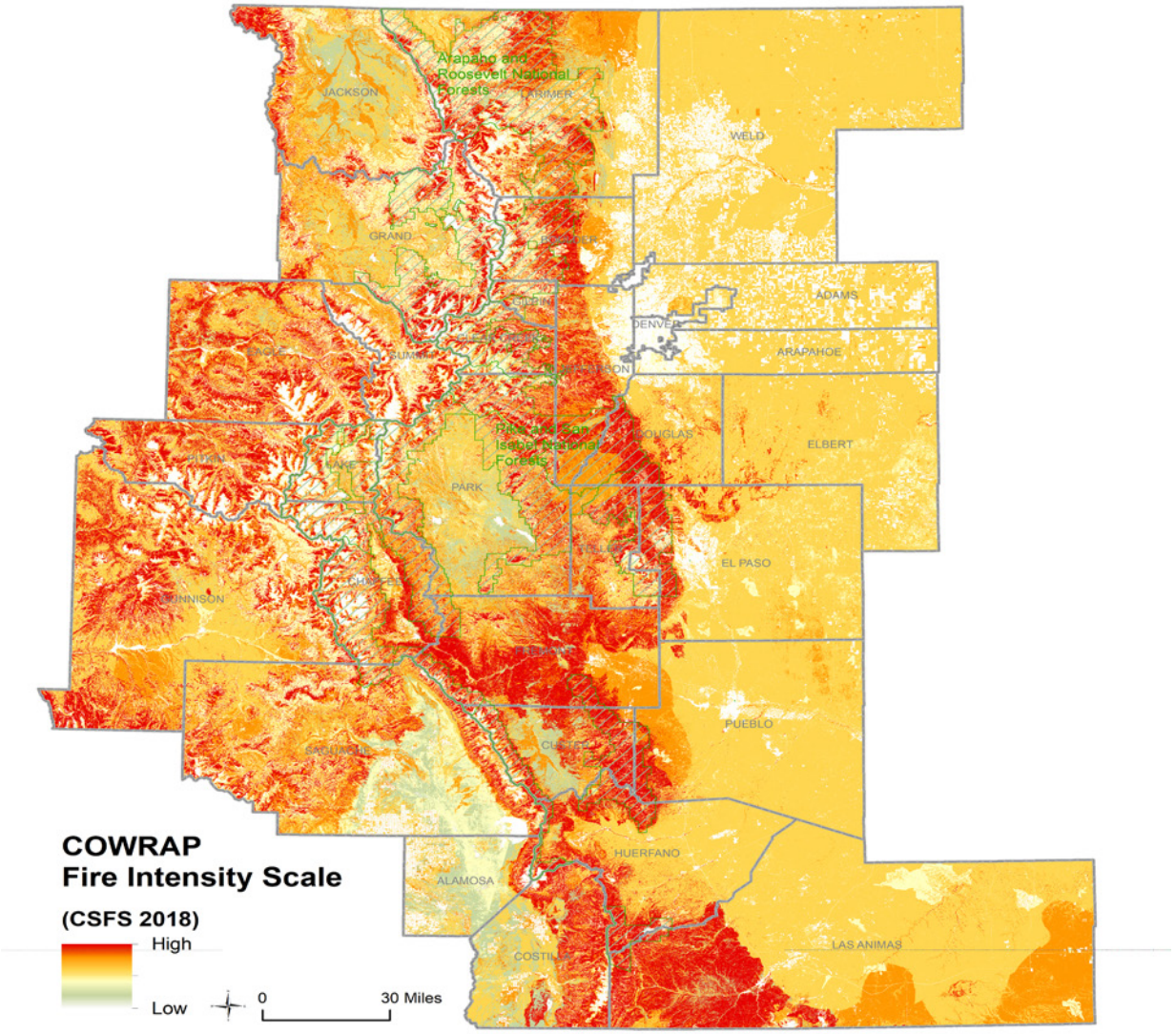


Figure 6—Colorado Risk Assessment Fire Intensity Scale (CSFS 2018; 2024) mapped in the Colorado Front Range.

Table 2—Percent area of each wildfire exposure and risk metric in low, moderate, and high categories for PSI-AR National Forests and the area within 0.5 and 1 miles of national forest boundaries. Data layer sources: Wildfire risk, Fire Intensity Scale, and Wildfire WUI Risk from Colorado State Forest Service (2018), Risk to Homes from Wildfire Risk to Communities (WRC 2020), and aPOP from Haas et al. (2014).

Exposure and Risk Metric	Location	Low (%)		Moderate (%)		High (%)	
		AR	PSI	AR	PSI	AR	PSI
Wildfire risk	Inside forest boundary	74.5	60.7	11.6	19.2	3.2	11.1
	Within 0.5 mile	69.5	60.3	14.4	17.3	5.7	9.8
	Within 1 mile	69.9	62.2	14.1	17.6	5.9	9.4
Fire intensity scale	Inside forest boundary	38.8	22.2	7.7	12.1	42.8	56.7
	Within 0.5 mile	33.7	20.6	16.5	16.2	39.4	50.6
	Within 1 mile	33.3	20.6	18.4	16.9	38.3	51.6
WUI wildfire risk	Inside forest boundary	3.8	1.7	1.3	0.7	0.5	0.4
	Within 0.5 mile	9.9	6.3	3.5	3.4	3.2	2.4
	Within 1 mile	10.1	6.7	3.6	3.5	3.3	2.5
Risk to homes	Inside forest boundary	0.8	1.2	2.6	0.8	2.9	2.2
	Within 0.5 mile	4.2	4.5	5.7	4.6	7.1	6.4
	Within 1 mile	4.4	4.4	5.8	5.3	7.4	6.1
aPOP	Inside forest boundary	0.1	0.2	1.9	0.9	0.6	0.3
	Within 0.5 mile	0.6	0.6	5.3	3.5	3.1	2.5
	Within 1 mile	0.6	0.6	5.2	3.7	3.4	2.4

Analysis of fire and risk metrics such as those described above help Federal, State, or local managers compare and contrast wildfire risk within and across jurisdictions. As an example, we consider an analysis of wildfire risk or exposure within and around (within 1 mile) all of the Ranger Districts in the Arapaho-Roosevelt (AR) and Pike-San Isabel (PSI) National Forests (tables 2 and 3). If a Forest Service regional fire manager wanted to prioritize resources among Forests, they may consider that 27 percent of the 1-mile area surrounding PSI Forest boundaries is in moderate and high COWRAP Wildfire Risk categories and note that the same figure calculated for the AR is lower (20 percent). This result is likely due to a combination of factors: modeled flame lengths are greater in the PSI National Forests, and vulnerable resources (e.g., critical water supply for Denver and Aurora) cover more area. However, the percentage of area in and around the AR mapped as low, moderate, or high WUI Risk (classes were aggregated from the original nine classes) is double that of the same categories around the PSI, indicating that the COWRAP WUI Risk Index is sensitive to the greater extent of developed area (WUI) near the AR. As expected for risk metrics tuned to populated areas, the percentage of area in moderate to high categories (class thresholds determined by 40th and 90th percentile values) of aPOP and WRC Risk to Homes metrics

is also higher around the AR. Depending on objectives, managers could consider information from both single value risk metrics (i.e. COWRAP WUI Risk Index, WRC Risk to Homes, and aPOP), along with those that incorporate multiple values at risk (COWRAP Wildfire Risk Index) to make nuanced prioritization and allocation decisions.

Similar comparisons are possible among ranger districts within national forests. For example, within the AR, the percentage of the Boulder Ranger District in low, moderate, or high classes of all three single value metrics (COWRAP WUI Risk, WRC Risk to Homes, and aPOP) is more than three times that of any other district on the AR (table 3). This is a result of the relatively high population and number of structures within the district boundary. For the multiple value COWRAP Wildfire Risk Index and the Fire Intensity Scale, a higher percentage of the Sulphur and Canyon Lakes Ranger Districts are in low and moderate classes, while the Boulder Ranger District has the highest percent area in the high risk/high-intensity classes; 18 percent of the Boulder Ranger District is at high risk and 50 percent consists of fuels that could result in high-intensity fires. These results indicate that the Boulder Ranger District has more WUI exposed to wildfire, and more area with high-intensity fuels than other ranger districts within the AR. When new fire behavior is simulated with fuels input layers that have been updated to reflect landscape conditions following more recent fire seasons, risk metrics will likely decrease within and around the footprints of the large fires that burned on the AR National Forests reflecting reduced hazard due to fuel reductions in the burned areas.

Table 3—Percent area of each wildfire exposure and risk metric in low, moderate, and high categories within PSI-AR Ranger Districts. Data layer sources: Wildfire risk, Fire Intensity Scale, and Wildfire WUI Risk from Colorado State Forest Service (2018), Risk to Homes from Wildfire Risk to Communities (WRC 2020), and aPOP from Haas et al. (2014). L = Low, M = Moderate, H = High.

National Forest	Ranger District	Wildfire risk			Fire intensity scale			WUI wildfire risk			Risk to homes			aPOP		
		L	M	H	L	M	H	L	M	H	L	M	H	L	M	H
AR	Sulphur	83.8	0.0	0.0	43.6	6.5	33.7	0.6	0.1	0.1	1.0	0.2	0.0	0.1	0.3	0.0
	Canyon Lakes	74.3	18.4	1.1	38.8	9.7	45.3	3.8	1.2	0.3	0.3	2.8	3.8	0.0	1.8	0.6
	Clear Creek	79.2	4.6	0.2	37.0	4.3	42.7	2.9	1.3	0.6	0.4	3.3	1.2	0.0	1.7	0.3
	Boulder	54.8	14.9	18.4	32.0	5.9	50.2	10.4	3.5	1.8	2.6	5.5	6.7	0.3	5.3	2.1
PSI	San Carlos	59.5	28.2	7.4	13.4	9.0	72.8	0.6	0.4	0.2	0.0	0.6	1.5	0.0	0.4	0.1
	South Park	71.7	14.3	7.9	28.9	17.9	47.1	2.0	0.5	0.2	2.6	1.4	1.9	0.5	1.2	0.1
	South Platte	42.1	27.4	27.1	20.1	7.7	68.7	1.9	0.8	0.4	0.1	0.9	3.6	0.0	0.9	0.5
	Leadville	73.9	0.7	0.0	30.4	8.4	35.7	1.6	0.4	0.2	1.9	0.0	0.0	0.3	0.5	0.0
	Salida	74.5	12.8	0.5	21.8	14.9	50.9	1.5	0.6	0.2	1.7	1.1	0.0	0.1	0.9	0.0
	Pikes Peak	32.5	33.6	28.5	18.0	11.9	64.8	2.9	1.8	1.6	0.0	0.6	7.4	0.0	1.2	1.9

Risk assessments conducted for smaller spatial extents (i.e., county, watershed, or smaller) offer finer spatial resolution map products targeted to specific objectives. Front Range communities with community wildfire protection plans sometimes generate wildfire risk maps for those plans (e.g., Boulder County Community Wildfire Protection Plan 2011) and some communities involved in the Community Planning Assistance for Wildfire program (https://cpaw.headwaterseconomics.org/) receive hazard maps based on locally customized fire behavior modeling, and a suite of maps specifically relevant to community land-use planning (Karau and Noonan-Wright 2018).

At the time of this publication, some of the wildfire hazard and risk metrics mentioned will be outdated, and new products will be available. Our discussion of the products that were available at the time of this writing is intended to illustrate the types of maps and analytics that may exist and provide examples of how they may be used. We advise managers to ensure that wildfire risk products match the timeframes and spatial scales appropriate for management objectives within the extent of the landscape under their jurisdiction.

Risk Mitigation Opportunities and Efforts

Front Range communities already face significant wildfire risk today. As the effects of climate change and WUI expansion combine to escalate that risk into the future (Liu et al. 2015), a robust mitigation strategy is critical. Schoennagel et al. (2017) propose policy and management measures that promote ecosystem and community resilience through a set of approaches, including: suppressing fewer fires, prescribing more fire, focusing fuel treatments where they will be most effective at reducing hazard near structures, fostering fire-adapted communities through incentives that decrease risk (homeowner mitigation work in the area directly around structures), and encouraging adoption of fire-safe land-use planning codes and regulations. Further, Calkin et al. (2014) suggest that we frame the WUI fire risk problem primarily as a home ignition problem; in the situation of a fire that overwhelms suppression capacity, home survival will depend upon the vulnerability of the structure, so mitigation responsibility should lie with the homeowner, rather than a land management agency. Ultimately, the most successful strategies for mitigating wildfire risk will be multi-pronged, with property owners, community leaders, and public agencies working together.

Because wildfire risk integrates the two components of hazard and vulnerability, we can consider mitigation challenges and opportunities for each component individually. Reducing wildfire hazard typically requires making changes to vegetation such that fuel characteristics are altered enough to reduce expected intensity of subsequent wildfires. This can be accomplished through landscape-scale fuels treatments, prescribed fire, and managed wildfire. Programs that reduce human ignitions can also address hazard by decreasing wildfire likelihood. Reducing wildfire vulnerability requires a focus on the particular resources or assets of concern in the risk equation. Where natural resources are the concern, fuels treatments that aim to reduce fire intensity may address vulnerability of the resource in addition to reducing hazard. Where assets

and infrastructure in the built environment are the concern, mitigation opportunities (and challenges) will be more focused on social behavior and land-use policies. The risk mitigation opportunities we outline here should be considered a baseline for the adaptable solutions agencies and communities will need to develop in order to become and remain fire resilient in the face of a changing climate.

Hazardous Fuels Management Efforts by Land Management Agencies

The Forest Service, Colorado State Forest Service, and several collaboratives (e.g., Front Range Fuels Treatment Partnership Roundtable https://frontrangeroundtable.org/) already collaborate to improve forest resilience and prioritize fuel treatment and prescribed burning efforts across public and private boundaries. These partnerships follow the ethos embodied in the National Cohesive Wildland Fire Management Strategy (USDA 2011) that robust communication and collaboration across levels of government (Federal, State, local) and among organizations and communities boosts fire management success. The cohesive strategy lays out three fundamental goals that should underpin any successful cross-boundary fire management strategy. These are: (1) creating and maintaining fire-resilient landscapes; (2) creating and maintaining fire-adapted communities; and (3) providing for safe and effective wildfire response (USDA 2011). Hazardous fuels management can help to address all these goals, and it will be most effective at reducing wildfire hazard when undertaken in a collaborative and strategic manner.

Successful risk-based decision-making requires a well-crafted problem statement and clear objectives (Calkin et al. 2014). Without clear objective statements or problem framing, well-intended mitigation actions will fail to reduce risk. For example, fuel treatments within the Fourmile Canyon wildfire perimeter failed to moderate fire intensity or slow fire spread during the incident, likely because the treatments were not specifically targeted to mitigate conditions of a “problem fire” scenario as could occur under very windy and dry conditions (Graham et al. 2012). As the climate changes, today’s “problem fire” conditions may become much more common, requiring managers to not only carefully consider wildfire risk metrics relevant to the spatial extent of their jurisdiction and their management goals, but also to the likelihood that treatment configurations that successfully mitigated wildfire impacts in the past may no longer be effective.

When appropriately parameterized, wildfire risk assessments can highlight landscape conditions that if mitigated, will reduce expected fire behavior, and therefore likely reduce negative effects to resources and assets. The variety of currently available risk maps and metrics affords managers choices, but they are also faced with the challenge of sorting through the options and selecting, or developing, a product that appropriately addresses their concerns. A risk assessment developed for a natural resource land management agency to prioritize fuels treatments may not be directly applicable for wildfire-aware municipal land-use planning.

One tool that can help with strategic placement of fuel treatments to protect specific resources and assets is risk transmission analysis. Transmission analysis employs standard risk assessment components, along with simulated wildfire ignitions and perimeters derived from fire behavior modeling, to highlight areas that are more likely to be the source of wildfires that reach communities or assets. This type of analysis reveals how different land tenures (or ownership types and management categories) contribute to wildfire exposure on other land tenures (Ager et al. 2018; Ager et al. 2019; Haas et al. 2015; Palaiologou et al. 2019). As such, it allows managers to locate and prioritize fuels treatments and mitigation efforts within their jurisdiction aimed at reducing exposure on other jurisdictions (Ager et al. 2014; Scott et al. 2016).

During wildfire incidents, fire managers strive to make informed decisions about how best to engage fire or use fire as a vegetation management tool. Ideally, a risk-based decision framework is in place before fire season (Taber et al. 2013), so agency officials and incident management teams can avoid contributing to the "wildfire paradox" wherein excessive fire suppression perpetuates hazardous fuel accumulation (Calkin et al. 2014). Some managers have begun to use Potential Operational Delineations (PODs); (Dunn et al. 2017; O'Connor et al. 2016; Thompson et al. 2016; Wei et al. 2019) to inform their prefire planning and incident management frameworks. PODs are areas whose boundaries are physical features that may successfully contain wildfire under the right conditions (roads, ridges, streams, fuel breaks/treatments, natural barriers, etc.). Wildfire risk metrics can be summarized within each POD, and PODs can then be classified into strategic response categories. During the prefire season, fire managers, land managers, resource specialists, and planners can review the relative wildfire risk within each POD and transmitted from each POD to nearby areas. Incident response strategies can be assigned that maximize safe and effective response, protection of community assets, and alignment with land and resource management plan objectives (Greiner et al. 2020). After PODs are initially created and classified, the framework can inform strategic decision-making during wildfire incidents (O'Connor and Calkin 2019) and in planning and prioritizing future hazardous fuels treatments that will most effectively reduce wildfire risk (Thompson et al. 2016). As climate and fuel conditions change, PODs and their associated strategies can be continually reassessed.

Homeowner Responsibility and Risk Perception

While strategic fuels mitigation work can help to reduce WUI wildfire hazard, it is important to note that fuels reduction on Federal lands alone will not prevent WUI fire disasters (Calkin et al. 2014). Communities may be best protected by placing fuel treatments close to homes and modifying ignitability of structures (Cohen 2000).

Addressing the vulnerability component in the wildfire risk equation is a critical aspect of making communities truly fire adapted. Homeowner awareness and motivation are paramount in WUI wildfire mitigation success, and cognizance and acceptance of the need to prepare for wildland fire has grown along with WUI expansion. Nonetheless,

homeowners exhibit varying levels of willingness or ability to take action to reduce their wildfire risk. This is due to differing levels of motivation, familiarity with the issues and solutions, availability of resources, education, and existence and acceptance of community outreach efforts (Kent et al. 2003; Stein et al. 2013). For example, although most survey respondents (96 percent) within the Fourmile Canyon Fire evacuation zone had performed some mitigation within the 30-foot home perimeter, installing screens over roof vents was a measure with one of the lowest reported frequencies (29 percent; Brenkert-Smith and Champ 2011), despite it being one of the most effective ways to prevent home ignition from embers. Perceptions of risk vary, along with misinformation about risk and belief in the ability to control fire damage. Threats anticipated in the distant future may not outweigh immediate concerns, such as desired landscaping and siding, or motivate outlays of resources to prepare for future risk, particularly when residents question the magnitude of risk and doubt the effectiveness of preparatory measures (McCaffrey 2009). A survey of WUI residents in Colorado indicated that some people were attached to the current landscape and did not intend to alter it until threatened by wildfire (Brenkert-Smith et al. 2006). Homeowners may also perceive that the cost of fire-resistant building materials is significantly greater than standard materials; however, Quarles and Pohl (2018) found that though a homeowner who retrofits certain elements of existing structures may incur significant expense, a new home built with wildfire resistant materials would cost roughly the same as a typical home, and those materials may be more durable. Barriers to mitigation action often include limited time, money, assistance, and technical knowledge, and perception of the required costs compared to the benefits. Additionally, the number of houses built within perimeters of past fires (occurring between 1990 and 2015) increased at a greater rate than houses built outside of past fire boundaries, indicating that many homeowners are either unaware of local wildfire risk, or they are willing to accept the risk (Alexandre et al. 2015).

Proximity to a national forest can also reduce perceptions of risk. Property owners adjacent to large areas of public land often are reliant on the government to protect them from natural disasters by covering losses and providing wildfire suppression services. This mindset removes the incentive for landowners in the WUI to take responsibility for reducing their own wildfire risk through their decisions about construction materials and landscaping (OIG 2006). Busby and Albers (2010) theorized that certain fire management policies foster a sense of complacency among private WUI landowners who benefit from fuels treatments on nearby public lands, but fail to conduct treatments on their own property, while also encouraging further settlement into undeveloped areas due to a perceived security that adjacent fuels are already mitigated.

Community Responsibility

While individual homeowner actions are essential to reduce the potential for wildfire damage to property, entire communities ideally would be adapted to fire such that fire could pass through a community without causing extensive damage. Fire-adapted communities are "knowledgeable, engaged communities where actions of residents and agencies in relation to infrastructure, buildings, landscaping and the surrounding ecosystem lessen the need for extensive protection actions and enable the communities to safely accept fire as part of the surrounding landscape" (Fire Adapted Communities Learning Network; https://fireadapted.org/). Thus, communities do not rely solely on suppression for protection. The entire community, including government agencies and emergency responders, works together to remove fuels, reduce human-caused ignitions, harden structures, prepare for emergency response, provide prevention education, and form partnerships to address barriers that deter people from participating in fire risk-reduction activities (Stein et al. 2013).

Front Range communities have access to information and organizations that assist with wildfire mitigation. Since 2002, the Firewise program of the National Fire Protection Association (https://www.nfpa.org/, https://www.nfpa.org/Education-and-Research/Wildfire/Firewise-USA) has enlisted communities to develop community-wide wildfire hazard mitigation plans and encourage homeowners to create defensible space on individual properties. In addition to educational tools, communities benefit through access to organizational help and expert guidance, community-building, and funding and assistance. Both the USDA Forest Service and the Colorado State Forest Service facilitate various grant programs that support individual and community mitigation efforts, often where a focused investment at all levels produces projects that have a cumulative effect on mitigating the risk across landscapes. Colorado also has an active Community Wildfire Protection Plan (CWPP; Healthy Forests Restoration Act of 2003; https://csfs.colostate.edu/wildfire-mitigation/community-wildfire-protection-plans/) program, with more than 250 locally supported plans that aim to reduce each community's exposure to wildfire damage by identifying and prioritizing fuels treatment opportunities and providing guidance to reduce the ignitability of structures. The Ready, Set, Go! Program offered by fire departments encourages improved coordination and communication between emergency response agencies and communities: "ready" is the defensible space message promoted by Firewise; "set" encourages situational awareness during a fire; and "go" prepares people for a safe and speedy evacuation (https://www.wildlandfirersg.org/). Additionally, Boulder County's Wildfire Partners (https://www.wildfirepartners.org/) assists county homeowners to identify, fund, and complete necessary mitigation work on their properties.

Incentive and education programs could be useful in changing homeowner risk perception and motivation for mitigation. Disclosure of wildfire risk, similarly to flood risk, may reduce willingness of people to move into fire-prone areas in the first place, affecting spread of future housing development. In Colorado Springs, home price and

wildfire risk were positively correlated (that is, most desirable housing locations had greatest home prices and greatest risk) until the Colorado Springs Fire Department posted the wildfire risk of housing parcels within the WUI (Donovan et al. 2007). To encourage mitigation to existing homes, financial assistance for homeowners is often available through grants, low-interest loans, and tax rebates, of up to $2,500 per taxpayer per year (Colorado Department of Revenue; https://www.colorado.gov/pacific/sites/default/files/Income65.pdf; Boulder County Community Wildfire Protection Plan 2011). This type of incentive program promotes homeowner mitigation action, while the specter of insurance premium increases or termination of coverage along with education about the problem of underinsurance are effective disincentives to inaction (see Boulder County Community Wildfire Protection Plan 2011) for an example highlighting the Fourmile Canyon Fire).

Although voluntary educational and community programs often are preferred over mandatory regulations, efforts that integrate wildfire mitigation into the land-use planning and building codes will decrease community vulnerability. Zoning and planning authority for WUI development rests with State, county, and municipal governments (OIG 2006). Community Planning Assistance for Wildfire (https://cpaw.headwaterseconomics.org/) is a program that surveys a community's existing codes and regulations, then helps planners address wildfire risk to incorporate improved land-use planning strategies. If home ignition zone guidelines are fair and part of a larger community effort of fire-safe building codes and planning to discourage development in high-risk areas, then homeowners are likely to be compliant (Stein et al. 2013). Conversely, conflicts with homeowner association restrictions or best management practices for soil and water protection or energy conservation may reduce compliance. Codes and standards are available from the National Fire Protection Association (http://www.nfpa.org/catalog/).

As communities expand into previously undeveloped areas, the safest strategy is to avoid building in areas with potential for wildfire hazard. Ridge tops, saddles, canyons, and tops of steep slopes are high risk locations; steeper slopes will result in faster moving fire, with longer flame lengths, and thus, may be a better location for other land uses than housing. For example, parks and golf courses provide both defensible space and safety zones for firefighters to be successful with suppression efforts. However, without changes to land-use planning policies and regulations, Front Range development is more likely to occur in areas of more frequent and intense fire behavior (Platt et al. 2011). Planning guidelines that reduce WUI sprawl and design transportation routes for easy ingress and egress will result in safer, more effective wildfire defense and efficient evacuation of people from Front Range WUI communities where topography is often steep and dissected.

Summary

In this chapter, we described the current and potential future dynamics of wildfire hazard and vulnerability within Colorado Front Range forests and communities. Wildfire hazard, determined by topography, fuels and weather, has been shaped by past land-use management practices and will continue to evolve due to interactions among a changing climate, insect and disease impacts, and human ignition patterns. The level of wildfire vulnerability within the Front Range depends upon whether resources or assets are located where wildfire is likely and the degree of damage that would result should a wildfire occur. Wildfire risk mitigation will be most successful if both components of risk are addressed. For example, forest managers could reduce hazard through strategic placement of hazardous fuels reduction efforts while communities and homeowners adopt initiatives that promote fire-resistant construction materials and vegetation management in the home ignition zone.

Wildfire risk assessment tools and analytics may support wildfire mitigation planning decisions. The value of any assessment will depend upon the clarity of project objectives and the suitability of tools and metrics selected to support the effort. We presented a summary of a suite of wildfire risk metrics mapped in the Front Range primarily to demonstrate a few ways that managers might critically evaluate and select appropriate tools to address different wildfire risk management questions. Though the metrics that we analyzed may benefit from an update (where fuel conditions represent those prior to more recent wildfires), and new products will be available at the time this chapter is published, results to date reveal that:

- Much of the Front Range is modeled to have the potential for surface fire behavior with flame lengths greater than 8 feet.
- WUI wildfire risk is higher in and around AR National Forests as compared to PSI National Forests, whereas moderate and high integrated wildfire risk (where multiple values and assets are considered) is more prevalent on the PSI National Forests.
- WUI wildfire risk is highest in the Boulder Ranger District of the AR National Forests and the Pikes Peak Ranger District on the PSI National Forests.

In the context of the risk assessment framework, these results indicate that wildfire hazard is elevated in the PSI National Forests, while structure exposure and vulnerability are elevated in and around the AR National Forests. This information could help customize mitigation efforts to address specific local risk conditions. For example, in areas with elevated wildfire hazard, fuels reduction activities may be most appropriate, whereas in areas with relatively high structure vulnerability, the priority might be to foster fire-adapted communities.

Our summary of risk metrics also indicates that moderate and high wildfire risk is not constrained within the AR and PSI National Forests boundaries. Collaboration and communication among all Federal, State, and local stakeholders will be essential to effectively manage wildfire risk in the Front Range WUI. We encourage land management and fire management officials at all levels of government, along with community collaboratives and private citizens, to leverage risk analytics and mapping tools relevant to specific project objectives, and thereby make informed strategic decisions that reduce hazard and risk.

Wildfire risk is in the Colorado Front Range is likely to increase with climate change due to increases in warmer, drier, and potentially windier weather conditions, regardless of fuel conditions. Concurrently, an expansion of the WUI will compound risk by increasing the area of the landscape that includes human resources and assets, while also increasing the likelihood of human-caused fire. Land management agencies and local collaboratives can provide fuel and vegetation management, fire preparedness and fire suppression, particularly by sharing limited resources among Federal and State agencies, counties, and fire protection districts. Nonetheless, Federal agencies do not have the power to prevent all fires, control the location of development on non-Federal lands, or create defensible space for private residences. Due to increasing likelihood of intense wildfires that will be difficult to suppress, fire mitigation efforts focused on structures and their immediate surroundings will be most effective at allowing communities to co-exist with wild areas and natural processes, thus freeing land management agencies to realign actions with their missions to sustain the health, diversity, and productivity of public lands.

References

Abatzoglou, J.T.; Williams, A.P. 2016. Impact of anthropogenic climate change on wildfire across western U.S. forests. Proceedings of the National Academy of Sciences of the United States of America. 113(42): 11770-11775. https://doi.org/10.1073/pnas.1607171113

Addington, R.N.; Aplet, G.H.; Battaglia, M.A.; [et al.]. 2018. Principles and practices for the restoration of ponderosa pine and dry mixed-conifer forests of the Colorado Front Range. Gen. Tech. Rep. RMRS-GTR-373. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 121 p. https://doi.org/10.2737/RMRS-GTR-373

Addington, R.N.; Tavernia, B.G.; Caggiano, M.D.; [et al.]. 2020. Identifying opportunities for the use of broadcast prescribed fire on Colorado's Front Range. Forest Ecology and Management. 458. https://doi.org/10.1016/j.foreco.2019.117655

Ager, A.A.; Day, M.A.; McHugh, C.W.; [et al.]. 2014. Wildfire exposure and fuel management on western U.S. national forests. Journal of Environmental Management. 145: 54–70. https://doi.org/10.1016/j.jenvman.2014.05.035

Ager, A.A.; Palaiologou, P.; Evers, C.R.; [et al.]. 2018. Assessing transboundary wildfire exposure in the southwestern United States. Risk Analysis. 38(10): 2105–2127. https://doi.org/10.1111/risa.12999

Ager, A.A.; Palaiologou, P.; Evers, C.R.; [et al.]. 2019. Wildfire exposure to the wildland urban interface in the western U.S. Applied Geography. 111: 102059. https://doi.org/10.1016/j.apgeog.2019.102059

Ager, A.A.; Vaillant, N.M.; Finney, M.A. 2010. A comparison of landscape fuel treatment strategies to mitigate wildland fire risk in the urban interface and preserve old forest structure. Forest Ecology and Management. 259(8): 1556–1570. https://doi.org/10.1016/j.foreco.2010.01.032

Ager, A.A.; Vaillant, N.M.; Finney, M.A.; [et al.]. 2012. Analyzing wildfire exposure and source–sink relationships on a fire prone forest landscape. Forest Ecology and Management. 267: 271–283. https://doi.org/10.1016/j.foreco.2011.11.021

Alexandre, P.M.; Mockrin, M.H.; Stewart, S.I.; [et al.]. 2015. Rebuilding and new housing development after wildfire. International Journal of Wildland Fire. 24(1): 138–149. https://doi.org/10.1071/WF13197

Balch, J.K.; Bradley, B.A.; Abatzoglou, J.T.; [et al.]. 2017. Human-started wildfires expand the fire niche across the United States. Proc Natl Acad Sci U S A. 114(11): 2946–2951. https://doi.org/10.1073/pnas.1617394114

Battaglia, M.A.; Gannon, B.; Brown, P.M.; [et al.]. 2018. Changes in forest structure since 1860 in ponderosa pine dominated forests in the Colorado and Wyoming Front Range, USA. Forest Ecology and Management. 422: 147–160. https://doi.org/10.1016/j.foreco.2018.04.010

Bentz, B.J.; Régnière, J.; Fettig, C.J.; [et al.]. 2010. Climate change and bark beetles of the Western United States and Canada: Direct and indirect effects. BioScience. 60(8): 602–613. https://doi.org/10.1525/bio.2010.60.8.6

Boulder County Community Wildfire Protection Plan. 2011. https://www.bouldercounty.org/disasters/wildfires/mitigation/community-wildfire-protection-plan/ [Accessed 28 May 2024]

Brenkert-Smith, H.; Champ, P.A. 2011. Fourmile Canyon: Living with wildfire. Fire Management Today. 71(2): 31–37

Brenkert-Smith, H.; Champ, P.A.; Flores, N. 2006. Insights into wildfire mitigation decisions among wildland–urban interface residents. Society & Natural Resources. 19(8): 759–768. https://doi.org/10.1080/08941920600801207

Brown, P.M.; Battaglia, M.A.; Fornwalt, P.J.; [et al.]. 2015. Historical (1860) forest structure in ponderosa pine forests of the northern Front Range, Colorado. Canadian Journal of Forest Research. 45(11): 1462–1473. DOI: 10.1139/cjfr-2014-0387. https://doi.org/10.1139/cjfr-2014-0387

Buechling, A.; Baker, W.L. 2004. A fire history from tree rings in a high-elevation forest of Rocky Mountain National Park. Canadian Journal of Forest Research. 34(6): 1259–1273. DOI: 10.1139/x04-012. https://doi.org/10.1139/x04-012

Busby, G.; Albers, H.J. 2010. Wildfire risk management on a landscape with public and private ownership: Who pays for protection? Environmental Management. 45(2): 296–310. https://doi.org/10.1007/s00267-009-9381-x

Calkin, D.E.; Ager, A.A.; Thompson, M.P.; [et al.]. 2011. A comparative risk assessment framework for wildland fire management: The 2010 cohesive strategy science report. Gen. Tech. Rep. RMRS-GTR-262. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 63 p. https://doi.org/10.2737/RMRS-GTR-262

Calkin, D.E.; Cohen, J.D.; Finney, M.A.; [et al.]. 2014. How risk management can prevent future wildfire disasters in the wildland-urban interface. Proceedings of the National Academy of Sciences of the United States of America. 111(2): 746–751. https://doi.org/10.1073/pnas.1315088111

Chambers, M.E.; Fornwalt, P.J.; Malone, S.L.; [et al.]. 2016. Patterns of conifer regeneration following high severity wildfire in ponderosa pine-dominated forests of the Colorado Front Range. Forest Ecology and Management. 378: 57–67. https://doi.org/10.1016/j.foreco.2016.07.001

Cohen, J.D. 2000. Preventing disaster: Home ignitability in the wildland-urban interface. Journal of Forestry. 98(3): 15–21. https://doi.org/10.1093/jof/98.3.15

Cohen, J.D. 2010. The wildland-urban interface fire problem. Fremontia. 38(2)–38(3): 16–22. https://www.cnps.org/wp-content/uploads/2018/03/Fremontia_Vol38-No2-3.pdf

Cohen, J.D.; Stratton, R.D. 2008. Home destruction examination: Grass Valley Fire, Lake Arrowhead, California. https://www.fs.usda.gov/treesearch/pubs/31544 [Accessed 28 May 2024]

Colorado State Forest Service (CSFS). 2018. The 2018 report on the health of Colorado's forests: Colorado State Forest Service. https://csfs.colostate.edu/media/sites/22/2019/03/FINAL-307714_ForestRpt-2018-www.pdf [Accessed 28 May 2024]

Colorado State Forest Service (CSFS). 2021. The 2021 report on the health of Colorado's forests: Colorado State Forest Service. https://csfs.colostate.edu/wp-content/uploads/2022/03/2021_Forest_Health_Report.pdf [Accessed 28 May 2024]

Colorado State Forest Service (CSFS). 2024. Colorado forest atlas wildfire risk reduction planner [internet application software]. Colorado State Forest Service: Fort Collins, CO. Available from: https://coloradoforestatlas.org/

Daniel, C.J.; Frid, L. 2012. Predicting landscape vegetation dynamics using state-and-transition simulation models. In: Kerns, B.K.; Shlisky, A.J.; Daniel, C.J., eds. First landscape state-and-transition simulation modeling conference. Gen. Tech. Rep. PNW-GTR-869. Portland, OR; U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 5–22. https://www.fs.usda.gov/research/treesearch/42566

Davis, K.T.; Dobrowski, S.Z.; Higuera, P.E.; [et al.]. 2019. Wildfires and climate change push low-elevation forests across a critical climate threshold for tree regeneration. Proceedings of the National Academy of Sciences. 116(13): 6193–6198. https://doi.org/10.1073/pnas.1815107116

Davis, K.T.; Higuera, P.E.; Dobrowski, S.Z.; [et al.]. 2020. Fire-catalyzed vegetation shifts in ponderosa pine and Douglas-fir forests of the western United States. Environmental Research Letters. 15(10): 1040b8. https://doi.org/10.1088/1748-9326/abb9df

Dillon, G.K.; Morgan, P.; Holden, Z.A. 2011. Mapping the potential for high severity wildfire in the western United States. Fire Management Today. 71(2): 25–28.: 25–28.

Donovan, G.H.; Champ, P.A.; Butry, D.T. 2007 Wildfire risk and housing prices: A case study from Colorado Springs. Land Economics. 83(2): 217–233. https://www.jstor.org/stable/27647762

Dunn, C.J.; Thompson, M.P.; Calkin, D.E. 2017. A framework for developing safe and effective large-fire response in a new fire management paradigm. Forest Ecology and Management. 404: 184–196. https://doi.org/10.1016/j.foreco.2017.08.03

EPA. 2010. National Center for Environmental Assessment (NCEA)—Global Change Research Program (GCRP) ICLUS v1.3. Housing density for the conterminous USA. https://edg.epa.gov/metadata/catalog/main/home.page

Finney, M.A. 2005. The challenge of quantitative risk analysis for wildland fire. Forest Ecology and Management. 211(1): 97–108. https://doi.org/10.1016/j.foreco.2005.02.010

Finney, M.A. 2020. The wildland fire system and challenges for engineering. Fire Safety Journal. 103085. https://doi.org/10.1016/j.firesaf.2020.103085

Finney, M.A.; McHugh, C.W.; Grenfell, I.C.; [et al.]. 2011. A simulation of probabilistic wildfire risk components for the continental United States. Stochastic Environmental Research and Risk Assessment. 25(7): 973–1000. https://doi.org/10.1007/s00477-011-0462-z

Flannigan, M.; Cantin, A.S.; de Groot, W.J.; [et al.]. 2013. Global wildland fire season severity in the 21st century. Forest Ecology and Management. 294: 54–61. DOI: https://doi.org/10.1016/j.foreco.2012.10.022

Fornwalt, P.J.; Huckaby, L.S.; Alton, S.K.; [et al.]. 2016. Did the 2002 Hayman Fire, Colorado, USA, burn with uncharacteristic severity? Fire Ecology. 12(3): 117–132. DOI: 10.4996/fireecology.1203117. https://doi.org/10.4996/fireecology.1203117

Graham, R.T. 2003. Hayman Fire case study. In: Graham, R.T., ed. 2003. Hayman Fire case study. Gen. Tech. Rep. RMRS-GTR-114. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 1–32. https://www.fs.usda.gov/research/treesearch/28713

Graham, R.T.; Finney, M.A.; McHugh, C.W.; [et al.]. 2012. Fourmile Canyon Fire findings. Gen. Tech. Rep. RMRS-GTR-289. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 110 p. https://doi.org/10.2737/RMRS-GTR-289

Greiner, S.M.; Schultz, C.A.; Kooistra, C. 2020. Pre-season fire management planning: the use of Potential Operational Delineations to prepare for wildland fire events. International Journal of Wildland Fire. https://doi.org/10.1071/WF20124

Haas, J.R.; Calkin, D.E.; Thompson, M.P. 2015. Wildfire risk transmission in the Colorado Front Range, USA. Risk Analysis. 35(2): 226–40. https://doi.org/10.1111/risa.12270

Hessburg, P.F.; Miller, C.L.; Parks, S.A.; [et al.]. 2019. Climate, environment, and disturbance history govern resilience of western North American forests. Frontiers in Ecology and Evolution. 7: 239. https://doi.org/10.3389/fevo.2019.00239

Jolly, W.M.; Cochrane, M.A.; Freeborn, P.H.; [et al.]. 2015. Climate-induced variations in global wildfire danger from 1979 to 2013. Nature Communications. 6: 7537. https://doi.org/10.1016/j.foreco.2011.12.022

Jolly, W.M.; Parsons, R.; Hadlow, A.M.; [et al.]. 2012. Relationships between moisture, chemistry, and ignition of *Pinus contorta* needles during the early stages of mountain pine beetle attack. Forest Ecology and Management. 269: 52–59. DOI: 10.1016/j.foreco.2011.12.022. https://doi.org/10.1016/j.foreco.2011.12.022

Karau, E.C.; Noonan-Wright, E. 2018. Wildfire hazard assessment for community land-use planning: A case study in Chelan County, WA. In: Hood, S.; Drury, S.; Steelman, T.; [et al.], eds. Fire continuum conference; Missoula, MT. Proceedings RMRS-P-78. Fort Collins, CO; U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 289–293. https://www.fs.usda.gov/research/treesearch/63232

Kent, B.; Gebert, K.M.; McCaffrey, S.; [et al.]. 2003. Social and economic issues of the Hayman fire. In: Graham, R.T., ed. 2003. Hayman Fire case study. Gen. Tech. Rep. RMRS-GTR-114. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 315–388. https://www.fs.usda.gov/research/treesearch/14904

Klutsch, J.G.; Negrón, J.F.; Costello, S.L.; [et al.]. 2009. Stand characteristics and downed woody debris accumulations associated with a mountain pine beetle (*Dendroctonus ponderosae* Hopkins) outbreak in Colorado. Forest Ecology and Management. 258(5): 641–649. DOI: 10.1016/j.foreco.2009.04.034. https://doi.org/10.1016/j.foreco.2009.04.034

Kolb, T.E.; Fettig, C.; Ayres, M.P.; [et al.]. 2016. Observed and anticipated impacts of drought on forest insects and diseases in the United States. https://doi.org/10.1016/j.foreco.2016.04.05

Litschert, S.E.; Brown, T.C.; Theobald, D.M. 2012. Historic and future extent of wildfires in the Southern Rockies Ecoregion, USA. Forest Ecology and Management. 269: 124–133. https://doi.org/10.1016/j.foreco.2011.12.024

Littell, J.S.; McKenzie, D.; Peterson, D.L.; [et al.]. 2009. Climate and wildfire area burned in western U.S. ecoprovinces, 1916–2003. Ecological Applications. 19(4): 1003–1021. https://doi.org/10.1890/07-1183.1

Liu, Y.; Goodrick, S.L.; Stanturf, J.A. 2013. Future U.S. wildfire potential trends projected using a dynamically downscaled climate change scenario. Forest Ecology and Management. 294: 120–135. https://doi.org/10.1016/j.foreco.2012.06.049

Liu, Z.; Wimberly, M.C.; Lamsal, A.; [et al.]. 2015. Climate change and wildfire risk in an expanding wildland–urban interface: A case study from the Colorado Front Range Corridor. Landscape Ecology. 30(10): 1943–1957. https://doi.org/10.1007/s10980-015-0222-4

Loehman, R.A.; Keane, R.E.; Holsinger, L.M. 2020. Simulation modeling of complex climate, wildfire, and vegetation dynamics to address wicked problems in land management. Frontiers in Forests and Global Change. 3(3). https://doi.org/10.3389/ffgc.2020.00003

Lukas, J.; Barsugli, J.; Doesken, N.; [et al.]. 2014. Climate change in Colorado: A synthesis to support water resources management and adaptation. Boulder, CO: University of Colorado.

Martinuzzi, S.; Stewart, S.I.; Helmers, D.P.; [et al.]. 2015. The 2010 wildland-urban interface of the conterminous United States. Research Map NRS-8. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 124 p. DOI: https://doi.org/10.2737/NRS-RMAP-8

McCaffrey, S. 2009. Crucial factors influencing public acceptance of fuels treatments. Fire Management Today. 69(1): 9–12. https://www.frames.gov/documents/usfs/fmt/fmt_69-1.pdf

McKinney, S.T. 2019. Systematic review and meta-analysis of fire regime research in ponderosa pine (*Pinus ponderosa*) ecosystems, Colorado, USA. Fire Ecology. 15(1): 38. https://doi.org/10.1186/s42408-019-0056-6

Mietkiewicz, N.; Balch, J.K.; Schoennagel, T.; [et al.]. 2020. In the line of fire: Consequences of human-ignited wildfires to homes in the U.S. (1992–2015). Fire. 3(3): 50. https://doi.org/10.3390/fire3030050

Morgan, P.; Keane, R.E.; Dillon, G.K.; [et al.]. 2014. Challenges of assessing fire and burn severity using field measures, remote sensing and modelling. International Journal of Wildland Fire. 23(8): 1045–1060. https://doi.org/10.1071/WF13058

Moriarty, K.; Cheng, A.S.; Hoffman, C.M.; [et al.]. 2019. Firefighter observations of “surprising” fire behavior in mountain pine beetle-attacked lodgepole pine forests. Fire. 2(2): 34. https://doi.org/10.3390/fire2020034

Negrón, J.F.; Cain, B. 2019. Mountain pine beetle in Colorado: A story of changing forests. Journal of Forestry. 117(2): 144–151. https://doi.org/10.1093/jofore/fvy032

O'Connor, C.D.; Calkin, D.E. 2019. Engaging the fire before it starts: A case study from the 2017 Pinal Fire (Arizona). Wildfire. 28(1): 14–18.

O'Connor, C.D.; Thompson, M.P.; Rodríguez y Silva, F. 2016. Getting ahead of the wildfire problem: Quantifying and mapping management challenges and opportunities. Geosciences. 6(3): 35. https://doi.org/10.3390/geosciences6030035

OIG. 2006. Audit report: Forest Service large fire suppression costs. Report Number: 08601-44-SF. https://www.fs.usda.gov/sites/default/files/media_wysiwyg/08601-44-sf_recommendations_11-20-06.doc

Page, W.G.; Alexander, M.E.; Jenkins, M.J. 2013. Wildfire's resistance to control in mountain pine beetle-attacked lodgepole pine forests. The Forestry Chronicle. 89(06): 783–794. DOI: 10.5558/tfc2013-141. https://doi.org/10.5558/tfc2013-141

Page, W.G.; Jenkins, M.J. 2007. Predicted fire behavior in selected mountain pine beetle-infected lodgepole pine. Forest Science. 53: 662–674. https://doi.org/10.1093/forestscience/53.6.662

Page, W.G.; Jenkins, M.J.; Runyon, J.B. 2014. Spruce beetle-induced changes to Engelmann spruce foliage flammability. Forest Science. 60(4): 691–702. DOI: 10.5849/forsci.13-050. https://doi.org/10.5849/forsci.13-050

Palaiologou, P.; Ager, A.A.; Evers, C.R.; [et al.]. 2019. Fine-scale assessment of cross-boundary wildfire events in the western United States. https://doi.org/10.5194/nhess-19-1755-2019

Parks, S.A.; Dobrowski, S.Z.; Shaw, J.D.; [et al.]. 2019. Living on the edge: Trailing edge forests at risk of fire-facilitated conversion to non-forest. Ecosphere. 10(3): e02651. https://doi.org/10.1002/ecs2.2651

Platt, R.V.; Schoennagel, T.; Veblen, T.T.; [et al.]. 2011. Modeling wildfire potential in residential parcels: A case study of the north-central Colorado Front Range. Landscape and Urban Planning. 102(2): 117–126. https://doi.org/10.1016/j.landurbplan.2011.03.015

Quarles, S.; Leschak, P.; Cowger, R.; [et al.]. 2013. Lessons learned from Waldo Canyon: FAC mitigation assessment team report: Insurance Institute for Business & Home Safety. https://www.nwfirescience.org/sites/default/files/publications/Waldo-Canyon-Rpt-FINAL.pdf

Quarles, S.; Pohl, K. 2018. Building a wildfire-resistant home: Codes and costs. Headwaters Economics. November. https://headwaterseconomics.org/wp-content/uploads/building-costs-codes-report.pdf

Radeloff, V.C.; Helmers, D.P.; Kramer, H.A.; [et al.]. 2018. Rapid growth of the U.S. wildland-urban interface raises wildfire risk. Proceedings of the National Academy of Sciences of the United States of America. 115(13): 3314–3319. https://doi.org/10.1073/pnas.1718850115

Raffa, K.F.; Powell, E.N.; Townsend, P.A. 2013. Temperature-driven range expansion of an irruptive insect heightened by weakly coevolved plant defenses. Proceedings of the National Academy of Sciences of the United States of America. 110(6): 2193–2198. https://doi.org/10.1073/pnas.1216666110

Riley, K.L.; Loehman, R.A. 2016. Mid-21st-century climate changes increase predicted fire occurrence and fire season length, Northern Rocky Mountains, United States. Ecosphere. 7(11): e01543. DOI: 10.1002/ecs2.1543. https://doi.org/10.1002/ecs2.1543

Rocca, M.E.; Brown, P.M.; MacDonald, L.H.; [et al.]. 2014. Climate change impacts on fire regimes and key ecosystem services in Rocky Mountain forests. Forest Ecology and Management. 327: 290–305. https://doi.org/10.1016/j.foreco.2014.04.005

Rodman, K.C.; Veblen, T.T.; Battaglia, M.A.; [et al.]. 2020. A changing climate is snuffing out post-fire recovery in montane forests. Global Ecology and Biogeography. 29(11): 2039–2051. https://doi.org/10.1111/geb.13174

Rodman, K.C.; Veblen, T.T.; Saraceni, S.; [et al.]. 2019. Wildfire activity and land use drove 20th-century changes in forest cover in the Colorado Front Range. Ecosphere. 10(2): e02594. https://doi.org/10.1002/ecs2.2594

Romme, W.H.; Veblen, T.T.; Kaufmann, M.R.; [et al.]. 2003. Ecological effects of the Hayman Fire–Part 1: Historical (pre-1860) and current (1860–2002) fire regimes. In: Graham, R.T., ed. Hayman Fire case study. Gen. Tech. Report. RMRS-GTR-114. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 181–195. https://doi.org/10.2737/RMRS-GTR-114

Schoennagel, T.; Balch, J.K.; Brenkert-Smith, H.; [et al.]. 2017. Adapt to more wildfire in western North American forests as climate changes. Proceedings of the National Academy of Sciences of the United States of America. 114(18): 4582–4590. https://doi.org/10.1073/pnas.1617464114

Schoennagel, T.; Veblen, T.T.; Negrón, J.F.; [et al.]. 2012. Effects of mountain pine beetle on fuels and expected fire behavior in lodgepole pine forests, Colorado, USA. PLoS One. 7(1): e30002. https://doi.org/10.1371/journal.pone.0030002

Schoennagel, T.; Veblen, T.T.; Romme, W.H. 2004. The interaction of fire, fuels, and climate across Rocky Mountain forests. BioScience. 54(7): 661–676. https://doi.org/10.1641/0006-3568(2004)054[0661:tioffa]2.0.co;2

Scott, J.H.; Gilbertson-Day, J.W.; Moran, C.; [et al.]. 2020. Wildfire risk to communities: Spatial datasets of landscape-wide wildfire risk components for the United States. Fort Collins, CO: Forest Service Research Data Archive. https://doi.org/10.2737/RDS-2020-0016

Scott, J.H.; Thompson, M.P.; Calkin, D.E. 2013. A wildfire risk assessment framework for land and resource management. Gen. Tech. Rep. RMRS-GTR-315. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 83 p. https://doi.org/10.2737/rmrs-gtr-315

Scott, J.H.; Thompson, M.P.; Gilbertson-Day, J.W. 2016. Examining alternative fuel management strategies and the relative contribution of National Forest System land to wildfire risk to adjacent homes–A pilot assessment on the Sierra National Forest, California, USA. Forest Ecology and Management. 362: 29–37. https://doi.org/10.1016/j.foreco.2015.11.038

Sherriff, R.L.; Platt, R.V.; Veblen, T.T.; [et al.]. 2014. Historical, observed, and modeled wildfire severity in montane forests of the Colorado Front Range. PLoS One. 9(9): e106971. https://doi.org/10.1371/journal.pone.0106971

Short, K.C. 2022. Spatial wildfire occurrence data for the United States, 1992–2020. [FPA_FOD_20221014]. 6th edition. Fort Collins, CO: Forest Service Research Data Archive. https://doi.org/10.2737/RDS-2013-0009.6

Short, K.C.; Finney, M.A.; Vogler, K.C.; [et al.]. 2020. Spatial datasets of probabilistic wildfire risk components for the United States (270m). 2nd edition. Fort Collins, CO: Forest Service Research Data Archive. https://doi.org/10.2737/RDS-2016-0034-2

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

Spracklen, D.V.; Mickley, L.J.; Logan, J.A.; [et al.]. 2009. Impacts of climate change from 2000 to 2050 on wildfire activity and carbonaceous aerosol concentrations in the western United States. Journal of Geophysical Research: Atmospheres. 114(D20). https://doi.org/10.1029/2008JD010966

Stavros, E.N.; Abatzoglou, J.T.; McKenzie, D.; [et al.]. 2014. Regional projections of the likelihood of very large wildland fires under a changing climate in the contiguous Western United States. Climatic Change. 126(3–4): 455–468. https://doi.org/10.1007/s10584-014-1229-6

Stein, S.M.; Menakis, J.; Carr, M.A.; [et al.]. 2013. Wildfire, wildlands, and people: Understanding and preparing for wildfire in the wildland-urban interface—A Forests on the Edge report. Gen. Tech. Rep. RMRS-GTR-299. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 36 p. https://doi.org/10.2737/RMRS-GTR-299

Stevens-Rumann, C.S.; Kemp, K.B.; Higuera, P.E.; [et al.]. 2018. Evidence for declining forest resilience to wildfires under climate change. Ecology Letters. 21(2): 243–252. https://doi.org/10.1111/ele.12889

Stevens-Rumann, C.S.; Morgan, P. 2019. Tree regeneration following wildfires in the western U.S.: A review. Fire Ecology. 15(1): 15. https://doi.org/10.1186/s42408-019-0032-1

Taber, M.A.; Elenz, L.M.; Langowski, P.G. 2013. Decision making for wildfires: A guide for applying a risk management process at the incident level. Gen. Tech. Rep. RMRS-GTR-298WWW. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 59 p. https://doi.org/10.2737/RMRS-GTR-298

Theobald, D.M.; Romme, W.H. 2007. Expansion of the U.S. wildland–urban interface. Landscape and Urban Planning. 83(4): 340–354. DOI: 10.1016/j.landurbplan.2007.06.002. https://doi.org/10.1016/j.landurbplan.2007.06.002

Thompson, M.P.; Bowden, P.; Brough, A.; [et al.]. 2016. Application of wildfire risk assessment results to wildfire response planning in the southern Sierra Nevada, California, USA. Forests. 7(3): 64. https://doi.org/10.3390/f7030064

Thompson, M.P.; Gilbertson-Day, J.W.; Scott, J.H. 2016. Integrating pixel- and polygon-based approaches to wildfire risk assessment: Application to a high-value watershed on the Pike and San Isabel National Forests, Colorado, USA. Environmental Modeling & Assessment. 21(1): 1–15. https://doi.org/10.1007/s10666-015-9469-z

Thompson, M.P.; Scott, J.H.; Langowski, P.G.; [et al.]. 2013. Assessing watershed-wildfire risks on National Forest System Lands in the Rocky Mountain Region of the United States. Water. 5(3): 945–971. https://doi.org/10.3390/w5030945

Thompson, M.P.; Zimmerman, T.; Mindar, D.; [et al.]. 2016. Risk terminology primer: Basic principles and a glossary for the wildland fire management community. Gen. Tech. Rep. RMRS-GTR-349. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 13 p. https://doi.org/10.2737/RMRS-GTR-349

USDA. 2011. A national cohesive wildland fire management strategy. Washington, DC: Wildland Fire Leadership Council. 43 p.

USDA and USDOI. 2001. Urban wildland interface communities within the vicinity of Federal lands that are at high risk from wildfire. https://www.gao.gov/products/gao-04-705

U.S. Government Accountability Office [US GAO]. 2004. Wildland fires: Forest Service and BLM need better information and a systematic approach for assessing the risks of environmental effects. GAO-04-705. https://www.gao.gov/assets/gao-04-705.pdf

Veblen, T.T.; Lorenz, D.C. 1986. Anthropogenic disturbance and recovery patterns in montane forests, Colorado Front Range. Physical Geography. 7(1): 1–24. https://doi.org/10.1080/02723646.1986.10642278

Vogler, K.C.; Gilbertson-Day, J.W.; Napoli, J.; [et al.]. 2022. Contemporary wildfire hazard across Colorado [report for Rocky Mountain Region United States Forest Service]. Missoula, MT: Pyrologix LLC. http://pyrologix.com/reports/COAL_HazardReport.pdf. [Accessed 31 October 2019].

Vose, J.M.; Peterson, D.L.; Domke, G.M.; [et al.]. 2018. Forests. In: Reidmiller, D.R.; Avery, C.W.; Easterling, D.R.; [et al.], eds. Impacts, risks, and adaptation in the United States: Fourth national climate assessment, volume II. Washington, DC: U.S. Global Change Research Program: 232–267. https://doi.org/10.7930/NCA4.2018.CH6

Walker, R.B.; Coop, J.D.; Parks, S.A.; [et al.]. 2018. Fire regimes approaching historic norms reduce wildfire-facilitated conversion from forest to non-forest. Ecosphere. 9(4): e02182. https://doi.org/10.1002/ecs2.2182

Wei, Y.; Thompson, M.P.; Scott, J.H.; [et al.]. 2019. Designing operationally relevant daily large fire containment strategies using risk assessment results. Forests. 10(4): 311. https://doi.org/10.3390/f10040311

Westerling, A.L.; Hidalgo, H.G.; Cayan, D.R.; [et al.]. 2006. Warming and earlier spring increase western U.S. forest wildfire activity. Science. 313(5789): 940–943. DOI: 10.1126/science.1128834. https://doi.org/10.1126/science.1128834

Worrall, J.J.; Marchetti, S.B.; Rehfeldt, G.E. 2016 Bioclimate models and change projections to inform forest adaptation in southwestern Colorado: Interim report. Technical Report R2-68. Golden, CO: United States Department of Agriculture, Forest Service, Rocky Mountain Region, Forest Health Protection, State and Private Forestry and Tribal Relations. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fseprd500723.pdf [Accessed 31 October 2019]

Infrastructure, Flood Risk, and Sustainable Operations in Front Range National Forests

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Infrastructure, Extreme Events, Climate Change, and Management Concerns

Infrastructure refers to a collection of facilities and services, typically encompassing energy, water and wastewater, waste management, transportation, public health, food and agriculture, telecommunications, and banking (National Research Council 1993). Systems of public buildings, known as public works, contribute important governmental functions and may be incorporated as infrastructure. The U.S. Department of Agriculture, Forest Service considers infrastructure to comprise structures and facilities for administrative, transportation, recreation, or special uses, such as dams and camping concessions (OIG 2017a). The complete Forest Service infrastructure portfolio contains 40,000 buildings, 27,000 recreational sites such as campgrounds, 370,000 miles of roads, 13,000 road and trail bridges, and 3,000 dam structures (OIG 2017a). Public utilities and corridors and privately owned structures, such as oil and gas wells, also occur on National Forest System lands. Infrastructure typically consists of socioeconomic development, but natural areas and hydrologic features also may be considered infrastructure with functions in supplying clean air and water (National Research Council 1993).

Infrastructure is vulnerable to extreme weather events and stresses from changing climate through flooding, drought, wildfire, landslides and debris flows, extreme heat, ice, strong winds, freeze-thaw cycles, and snowmelt (Mora et al. 2018). Multiple outcomes result from extreme events, including interactions (Hanberry 2020). Drought and warming may stress water supplies and indoor cooling systems, create tree mortality and hazard trees, and increase road dust (Hanberry 2020). Drought and warming may increase probability of wildfire, which threatens infrastructure and natural resources, exacerbates flooding, and generates smoky and low visibility conditions (Karau et al. 2024; Rangwala 2024). After vegetation is removed by fire, the likelihood of flooding and debris flow may increase following rain events (Keller et al. 2021). Flooding and associated soil erosion may generate infrastructure failure of administrative buildings, roads, bridges, and recreation sites near streams, steep slopes, and unstable soils (Furniss and Howe 2016; Furniss et al. 2018). Extreme events may cause road closures, emergency evacuation, disruption of livelihoods, injury and mortality, and breakdown of infrastructure networks, such as electrical and water supplies and emergency services (Hanberry 2020; Mora et al. 2018). Climate change likely will increase frequency of extreme events (Karau et al. 2024; Rangwala 2024).

Anticipation of potential infrastructure vulnerabilities due to exposure to natural hazards as a consequence of extreme weather and climate change can help inform management priorities and guide plans to increase infrastructure resilience (Keller et al. 2021). Infrastructure without ongoing maintenance or near its lifecycle end is more

likely to fail due to structural weakness, under-design below minimum standards, or more demanding conditions than anticipated (Furniss et al. 2018). Many stream crossings with older culverts were designed to accommodate 25-year peak streamflows, whereas current standards typically require sizing for 100-year flows, resulting in older culverts that are failing due to inadequate culvert capacity and increased plugging hazards (Furniss et al. 2018). Current and future stream flows also may exceed the capacity of bridges and dams (Furniss et al. 2018). Extreme snowfalls can limit transportation and access, while warmer and less snowy winters reduce maintenance and snow removal costs and allow early access to facilities (Keller et al. 2021). Drought and warming above 90 °F may reduce bridge integrity and cause heaving, buckling, and cracking in roads, increasing maintenance costs (Klein et al. 2015). However, if roads are drier and warmer in the future, with reduced erosion and freeze-thaw episodes, weather may cause less damage than under current conditions (Neumann et al. 2015). Variable weather or seasonal changes create problems for scheduling of contracts and maintenance work (Keller et al. 2021). Consequences may be unknown and underestimated, even though infrastructure is vulnerable to extreme weather events.

The Front Range of central Colorado (fig. 1) already may be experiencing climate change and risks to infrastructure caused by design for historical climate conditions instead of current extreme events and future climate change (Lall et al. 2018). Managers and planners for the Rocky Mountain Region and the Arapaho-Roosevelt National Forests and Pike-San Isabel National Forests requested (1) delineation of 100-year-flood boundaries to identify infrastructure within 100-year-flood boundaries for future avoidance of these floodplains; (2) management guidelines for infrastructure that is resilient to climate change and extreme weather events; and (3) information about incorporation of sustainable operations. Based on these management concerns, we specifically assessed flood risk to infrastructure of recreational sites and buildings by developing a GIS layer of 100-year-flood inundation using modeled streamflow rates during 1994 to 2018 (Zheng and Hanberry 2020). However, more work with resource managers, hydrologists, and engineers is needed to assess both this model and risks to infrastructure. Roads, trails, and campgrounds near streams, which are a strong indicator of potential flood hazard, in national forests and grasslands of the Rocky Mountain Region were identified by Furniss and Howe (2016) and manager concerns about fire risk to national forests and the wildlands urban interface are addressed in this report (Karau et al. 2024). We also suggested management practices and sustainable operations for infrastructure and resources for detailed information. In particular, Keller et al. (2021) provided extensive treatments for adapting transportation infrastructure to the effects of climate change and additional national core best management practices are available for roads, trails, and recreation on National Forest System lands (USDA FS 2012).

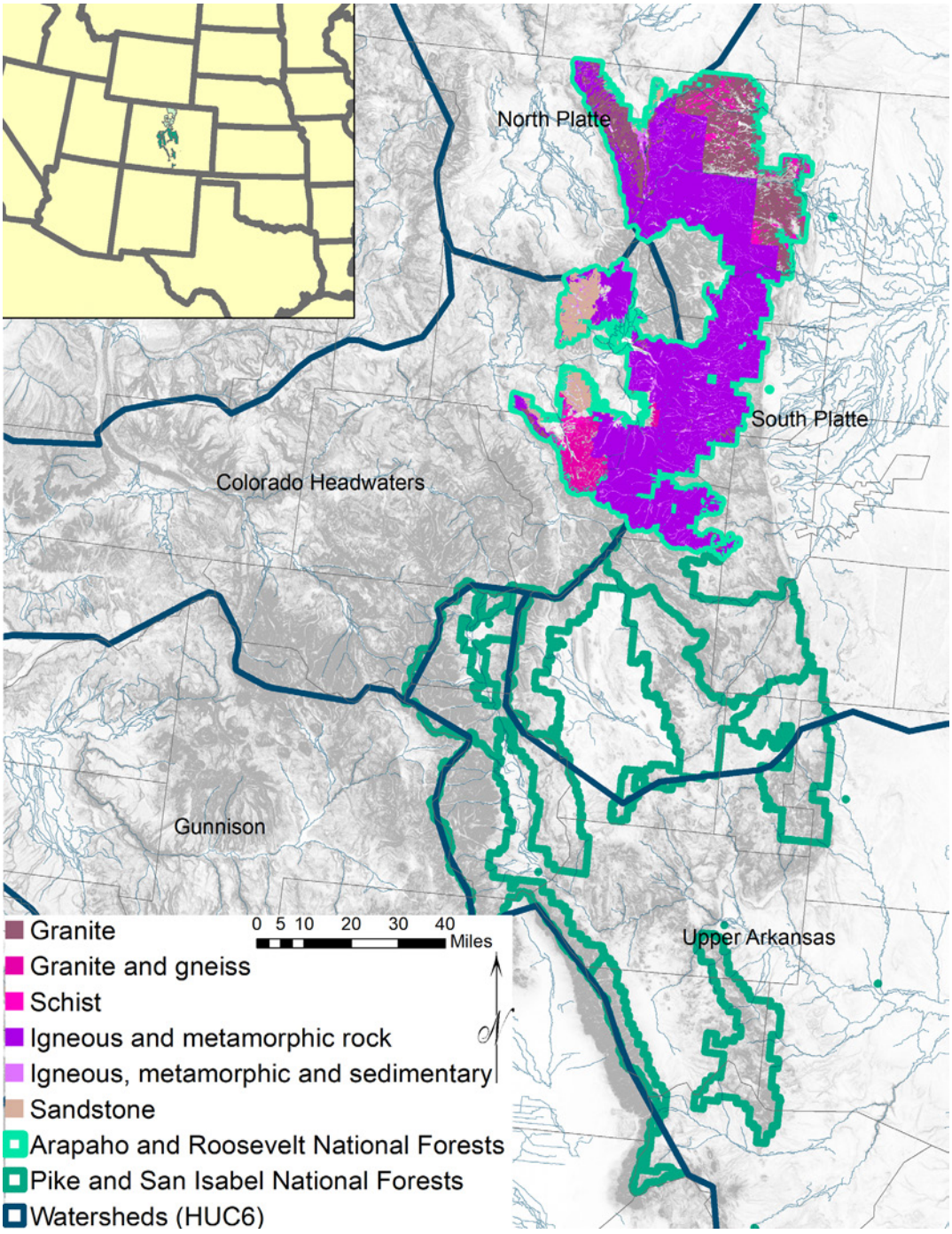


Figure 1—The setting for the Front Range region in central Colorado, including the Arapaho and Roosevelt National Forests and Pike and San Isabel National Forests, watersheds, and topographic roughness (darker gray has greater topographic difference). The Rocky Mountains are comprised primarily of igneous and metamorphic rock, as demonstrated in the Arapaho and Roosevelt National Forests.

Infrastructure and Flooding

Flooding is damaging through direct rainfall, overbank flow, channel scour, soil erosion, treefall, and landslide (Keller et al. 2021). Flooding stresses stream crossings of culverts and bridges, road and drainage systems, and unstable slopes through transportation of large volumes of water and debris, resulting in the potential for failure during high runoff periods (Keller et al. 2021). Consequences of failure include reduced water quality and loss of designated uses due to a range of limited, short-term impacts to severe, long-term impacts (Furniss and Howe 2016). Landslides, or movement of rock and soil due to saturated soil on steep slopes, often worsen flooding and roads commonly increase landslide risks (Furniss et al. 2018). Heavy rains after fire or drought, which reduce vegetation, may result in debris flows (Keller et al. 2021).

Flooding effects on infrastructure involve damage to structures and roads and failure of culverts, dams, and bridges, causing road and trail closures and millions of dollars in basic repairs (ARP 2013; Wobus et al. 2014). Costs consist of transportation disruption, infrastructure relocation or abandonment, and loss of property and even lives (Pralle 2019). In the early hours of June 11, 2010, a flash flood in the Albert Pike Recreation Area campground, Ouachita National Forest, Arkansas, claimed the lives of 20 people (USDA 2010). The area was flood-prone and in a 100-year floodplain. However, the event was judged to be beyond a 500-year return period magnitude, so most safety measures may have been ineffective except for avoidance of locating campgrounds in floodplains (USDA 2010). The 2013 Colorado Front Range Flood produced an estimated $16.8 million in damages to structures in the Arapaho-Roosevelt National Forests, and 10 fatalities, and $2.9 billion in damages for 18 counties in the Front Range (ARP 2013). Other historical peak discharge events in Colorado are recorded (USGS 2022a). Currently, flood events in the Upper Colorado River region average a cost of $27 million per year (Wobus et al. 2014). Flooding costs in the future are expected to average about $52 million per year (Wobus et al. 2014). Damage to infrastructure also can affect water quality and aquatic ecosystems (Furniss and Howe 2016). For example, spills from mine tailings dams on or near National Forest System lands may result in health risks from pollutants, such as at the Gold King Mine wastewater spill near Silverton, Colorado, during 2015 (OIG 2017a).

Current Flood Risk

Currently, data about flooding and their recurrence intervals are not publicly available in the United States (Hydrologic Engineering Center 2023). The Federal Emergency Management Agency (FEMA) ratings are the standard in flood risk (National Research Council 2009; OIG 2017b; Pralle 2019). Flood maps typically are based on a variety of methodologies, including field-intensive data, historical flooding information, and negotiation with 22,000 communities (Pralle 2019). Less populated areas often are excluded from mapping (Pralle 2019). Maps are updated when new flooding events occur (Pralle 2019; Wing et al. 2017). That is, these flood maps are not able to provide

accurate assessments of current flood risk for the United States due to varying quality, incompleteness, and lack of funds (estimated at billions of dollars) to revise maps with new information (National Research Council 2009; OIG 2017b; Pralle 2019). For example, after September 2013 flooding in Colorado, Colorado's Legislature funded the Colorado Hazard Mapping Program to determine flood hazard area (Colorado Water Conservation Board 2019).

Flood hazard mapping and modeling in the United States is in a developmental stage (Hydrologic Engineering Center 2023). Global flood models that are openly accessible do not provide detailed information for smaller streams (Bernhofen et al. 2018). A variety of proprietary flood risk models have proliferated in recent years (Risk & Investment 2017). However, due to lack of open access and transparency, the completeness and accuracy of these maps generally is unknown but likely to provide flood risk to houses in urban areas for insurance purposes, with a focus on urban storm water drainage based on impervious surface area and localized storm water drainage infrastructure (Risk & Investment 2017). A national 30-m flood hazard model is available, at a cost, even though flow of urban storm water may not be modeled as accurately as other proprietary flood risk maps (Wing et al. 2017).

Given lack of available model information, Furniss and Howe (2016) evaluated vulnerability of roads, trails, and campgrounds in national forests and grasslands of the Rocky Mountain Region by using road, trail, and campground proximity of 100 feet and 300 feet to streams, thus indicating potential flood hazard. The Arapaho-Roosevelt National Forests and Pawnee National Grasslands contained 812 road miles and 21 recreational sites within 100 feet of streams and 1,219 road miles and 32 recreational sites within 300 feet of streams, out of 5,106 road miles and 79 recreational sites, with 7,786 road and motor trail stream crossings (Furniss and Howe 2016). The Pike and San Isabel National Forests and Cimarron and Comanche National Grasslands contain 1,111 road miles and 35 recreational sites within 100 feet of streams and 1,617 road miles and 66 recreational sites within 300 feet of streams, out of 5,880 road miles and 180 recreational sites, with 8,962 road and motor trail stream crossings (Furniss and Howe 2016).

To provide information about 100-year flood discharge, or a 1 percent chance of being equaled or exceeded during any year under current stationary conditions, we used the following method (documented in Zheng et al. 2018). The National Water Model (NWM version 2.0) is a new tool that simulates hourly water flow for 2.7 million stream locations (National Hydrography Dataset; NHDPlusV2) based on more than 8,000 U.S. Geological Survey gauges in the contiguous United States (Office of Water Prediction 2019). We used the retrospective (1994 to 2018) simulation to extract the maximum flow rate for each stream reach. We then applied the USGS bulletin 17B flood frequency analysis method to determine discharge rates, from the log-Pearson Type III frequency distribution (IACWD 1982). We supplied discharge rates at multiple intervals, including the 100-year recurrence interval, for about 5,000 stream reaches. To determine area

and height of inundation, we used the height above nearest drainage (HAND) method (Rennó et al. 2008; Rodda 2005) and HAND data (Liu et al. 2018; Zheng et al. 2018), based on hydrologic terrain, to estimate inundation rather than more computationally intensive modelling. We applied synthetic rating curves from the HAND raster to convert the 100-year flow rates to corresponding water levels (Zheng et al. 2018), using the Manning's equation with a predefined channel roughness value of 0.05.

An approximate flood mapping approach with simplified hydraulic assumptions is not likely to generate products as accurate as locally detailed engineering models, but this modeling approach is realizable at a continental scale and provides information about floodplains where other models are not available. Caveats include multiple sources of error that may affect these models, although after identification of problems, models will improve. Bias may occur in the National Water Model simulations, NHD stream locations, and digital elevation models. For example, obstructions such as bridges, roads, and river intersections may interfere with determination of hydrologic terrain. Validation shows that the HAND models may be comparable with observed floods and computationally intensive hydrological models (Afshari et al. 2018; Johnson et al. 2019; Nobre et al. 2016). A challenge for flood modeling is depicting stormwater drainage in urban areas when heavy rainfall overwhelms local drainage capabilities, which this model will not represent well (Zhang et al. 2018), albeit urban modeling may not be relevant to forested lands.

Facilities Within 100-Year Flood Zones

Of the approximately 200 open recreational areas (all GIS layers from the USDA Forest Service Enterprise Data Warehouse) that may be high risk due to overnight stays (that is, campgrounds, camping areas, horse camps, group campgrounds, hotels, lodges, lookout/cabins, resorts, and recreation residences), 62 were within 300 feet of 100-year flood zones (table 1; resource managers and engineers will need to evaluate accuracy and implications of the results). Seventeen of these recreational areas were in the Arapaho-Roosevelt National Forests and 45 were in the Pike-San Isabel National Forests (fig. 2). The GIS points do not represent dimensions and also may not be centered on the recreational area sites, and thus we applied a large 300-foot buffer around the points signifying recreational sites. Some recreational areas may extend well into the flood zones. For example, the Dutch George Campground of Arapaho-Roosevelt National Forests was inundated by the 100-year flood zone (fig. 3).

Table 1—Overnight stay recreational areas and associated wastewater sites likely at greatest risk of inundation during a 100-year flood based on 100 (wastewater) to 300 (recreational areas) foot proximity to GIS point locations to a modeled flood zone. The model is not an engineered design and point locations are imprecise.

Arapaho and Roosevelt National Forests		
Site name	**Type**	**Capacity**
Ansel Watrous	Campground	90
Big Bend	Campground	45
Big South	Campground	20
Browns Park	Campground	120
Camp Dick	Campground	255
Chambers Lake	Campground	305
Dutch George	Campground	120
Kelly Flats	Campground	150
Lower Narrows	Campground	40
Mountain Park	Campground	320
North Fork Poudre	Campground	45
Peaceful Valley	Campground	90
Sleeping Elephant	Campground	75
Stove Prairie Landing	Campground	45
Tom Bennett	Campground	60
Tunnel	Campground	245
Upper Narrows	Campground	40
Peaceful Valley Host Site	Holding Tank	0
Camp Dick Host Site #2	Holding Tank	0
Camp Dick Host Site #1	Holding Tank	0

Pike and San Isabel National Forests		
Site name	**Type**	**Capacity**
Blue Lake	Campground	75
Buffalo	Campground	205
Cascade	Campground	105
Chalk Lake	Campground	105
Coaldale	Campground	40
Cottonwood Lake	Campground	125

Table 1 continued.

Pike and San Isabel National Forests		
Site name	**Type**	**Capacity**
Cove	Campground	20
Davenport	Campground	60
Deer Creek	Campground	65
Dexter	Campground	75
Fourmile	Campground	70
Geneva Park	Campground	125
Goose Creek	Campground	50
Green Mountain	Campground	30
Halfmoon East	Campground	30
Halfmoon West	Campground	80
Happy Meadows	Campground	40
Iron City	Campground	75
Jefferson Creek	Campground	90
Kite Lake	Campground	78
Lodgepole	Campground	175
May Queen	Campground	135
Monarch Park	Campground	264
Mt. Princeton	Campground	95
Ophir Creek	Campground	155
Osprey	Campground	65
Ouzel	Campground	65
Parry Peak	Campground	130
Platte River	Campground	50
Purgatoire	Campground	115
Riverside	Campground	95
Spillway	Campground	120
Springer Gulch	Campground	75
St. Charles	Campground	75
Weston Pass	Campground	70
Winfield	Camping Area	35
Ponderosa	Group Campground	120
Spruce	Group Campground	60

Table 1 continued.

Pike and San Isabel National Forests		
Site name	**Type**	**Capacity**
Crescent Mining Camp	Lookout/Cabin	6
Clyde	Recreation Residence	0
Fox Lake	Recreation Residence	0
Rock Creek	Recreation Residence	0
Stage Road	Recreation Residence	0
Vicksburg	Recreation Residence	0
Weston Pass	Recreation Residence	70
Blue Mtn	Toilet-Vault	0
Spruce Grove Cement	Toilet-Vault	0
Whiteside	Toilet-Vault	0
Spruce Grove Stick Built	Toilet-Vault	0
Spillway	Toilet-Vault	0
Elevenmile	Toilet-Vault	0

Of the approximately 800 active buildings, six were within 50 feet of flood zones, which were in the Pike and San Isabel National Forests (table 1). These buildings were vaulted toilets, and thus represent potential water pollution sources, and also may indicate campgrounds that extend into flood zones. Three wastewater holding tanks in the Arapaho-Roosevelt National Forests were within 100 feet of flood zones. No oil or gas wells were within 50 feet of flood zones. There were 363 road intersections within the flood zone, of which 191 were in the Arapaho-Roosevelt National Forests and 172 were in the Pike-San Isabel National Forests.

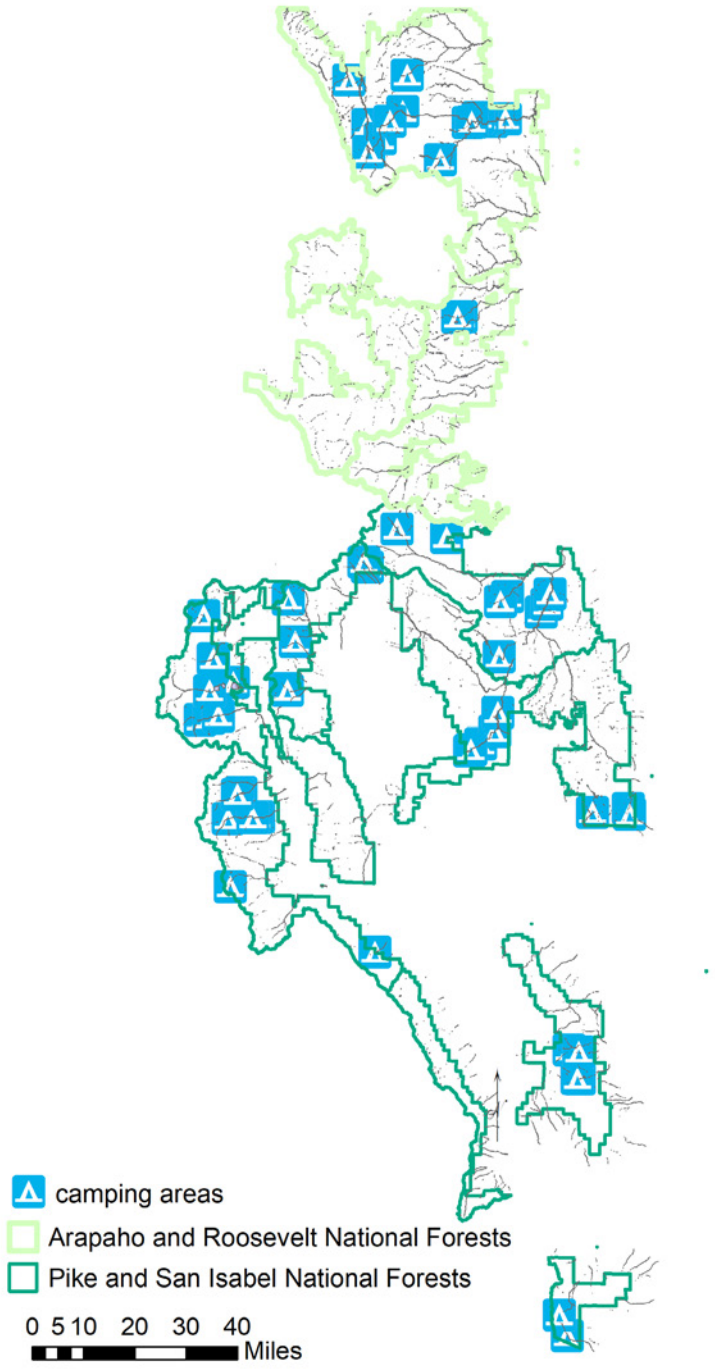


Figure 2—Location of 17 camping areas within 300 feet of 100-year flood zones in the Arapaho and Roosevelt National Forests and 45 camping areas within 300 feet of 100-year flood zones in the Pike and San Isabel National Forests. These areas may be at greatest risk of inundation based on proximity of GIS point locations indicating overnight stay recreational areas to a model. However, the model is not an engineered plan and point locations are dimensionless and may be imprecise.



Figure 3—Example of a campground area in comparison to a GIS point location for the Dutch George campground of Arapaho and Roosevelt National Forests and inundation of the campground during 100-year flood (green zone).

Future Flood Risk and Infrastructure

Current flood risk is not well-established and future flood risk due to climate warming is even more uncertain. Flood risk based on current stream flows may not remain stable under future stream flows and stream channels (Rangwala 2024; Wenger et al. 2010). Therefore, “100-year” flood risk is a misnomer and instead represents a 1 percent chance of flooding under current conditions. Stream geometry also may change over time along with other hydroclimate changes.

Greater frequency of intense precipitation events likely will increase risks of flooding, landslides, and infrastructure damage to roads, dams, bridges, and culverts (Furniss et al. 2018). Winter flooding events and peak flows may increase as precipitation shifts from snow to rain and greater winter soil moisture may increase landslide risk (Furniss et al. 2018). One coarse method of identifying areas at greater risk in the future is to locate areas that have increases in realized precipitation during the past 30 years and areas that are expected to increase in precipitation, runoff, or extreme rainfall events in the future, which overall indicate potentially increasing flooding (Rangwala 2024).

Percent change in 1.5-year flood (F1.5, percent), 25-year flood (F25, percent), and absolute change in center of flow mass timing (CFM, days) between historical data (1977–2006) and end of century (2071–2099) projections were produced from daily runoff and baseflow from the Variable Infiltration Capacity (VIC) macroscale hydrologic model (Liang et al. 1994). The F1.5, F25, and CFM stream metrics are based on the

ensemble mean of 10 general circulation models associated with a scenario that assumes balanced use of fossil fuels and renewable energy sources (Wenger et al. 2010). Data are available for each stream segment in the 1:100,000 scale National Hydrography Dataset (NHDPlus version 2) of the western United States (https://data.fs.usda.gov/geodata/edw/datasets.php?xmlKeyword=Hydro+Flow+Metrics+West). F1.5 and F25 represent flood levels that are expected to occur every 1.5 and 25 years, respectively, and CFM represents the day of the water year when half of the total annual stream flow has been exceeded.

Over two-thirds of streams within the Arapaho-Roosevelt and Pike-San Isabel National Forests are projected to experience decreases in 1.5-year floods (bankfull flows) by the end of the century (fig. 4). The remaining streams are projected to remain constant or experience moderate increases (10 to 20 percent) in bankfull flows. Stream flows that can lead to larger floods (F25) may increase in over one-third of Arapaho-Roosevelt and Pike-San Isabel National Forests streams (fig. 5). The largest of these increases (> 25 percent change in 25-year floods) is projected to occur within tributaries of large Front Range canyons and the Upper Arkansas River. The timing of peak flows (CFM) may occur 2 to 6 weeks earlier by the end of the century due to snow melting earlier in the year (fig. 6). These changes in the frequency and timing of floods may intensify damage to roads, bridges, and other infrastructure near perennial streams.

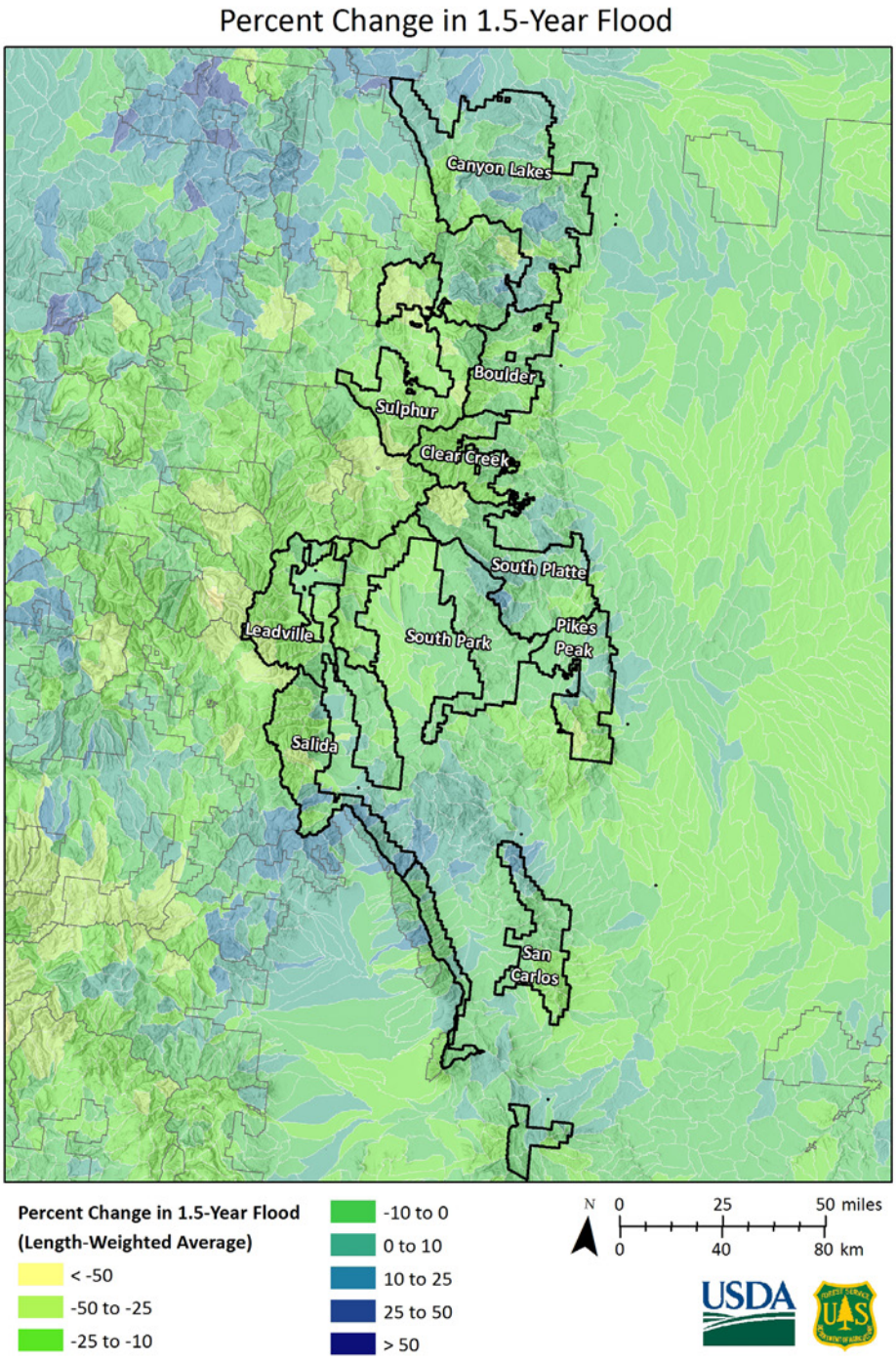


Figure 4—Percent change in 1.5-year flood between historical data (1977–2006) and future (2071–2099) projections.

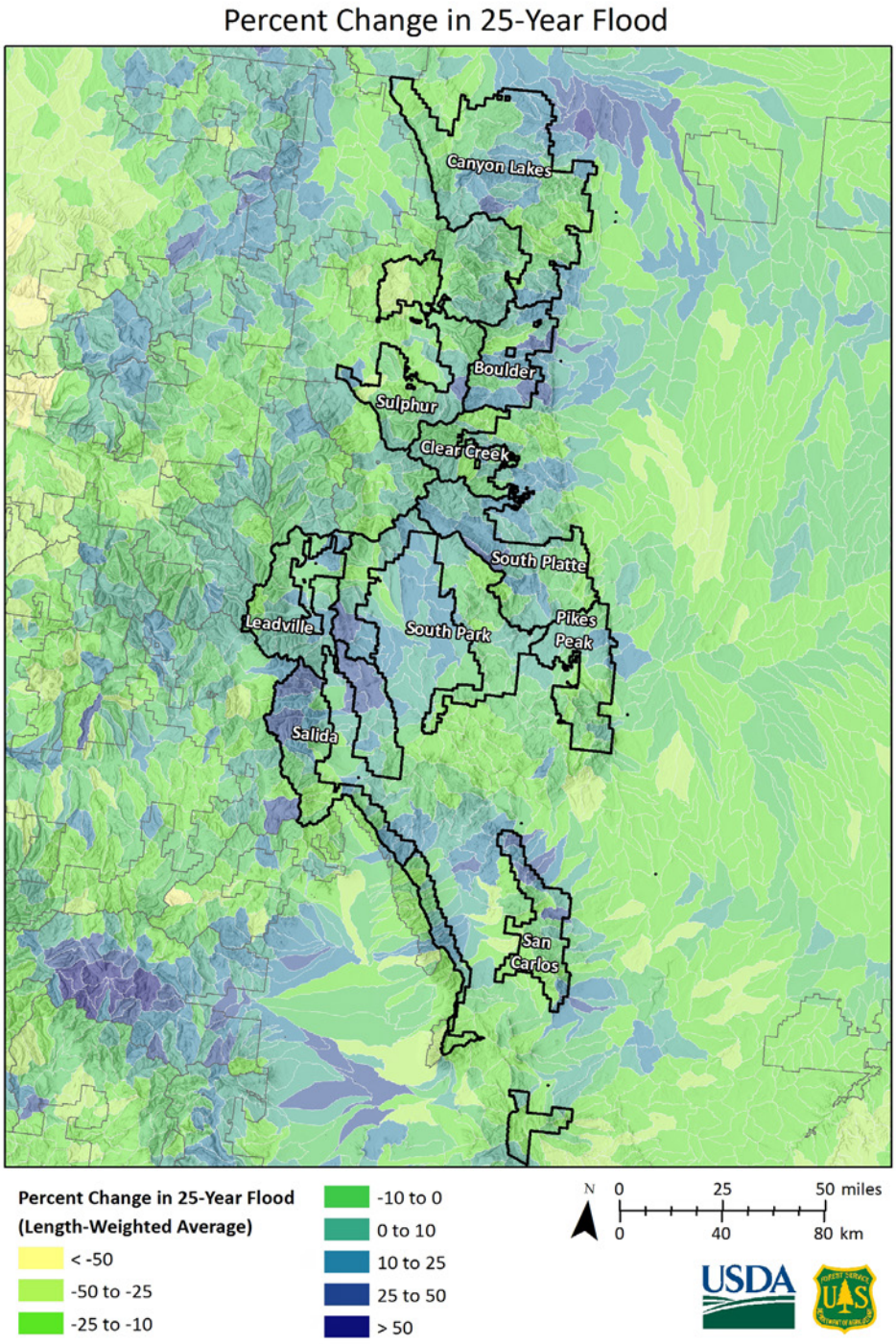


Figure 5—Percent change in 25-year flood between historical data (1977–2006) and future (2071–2099) projections.

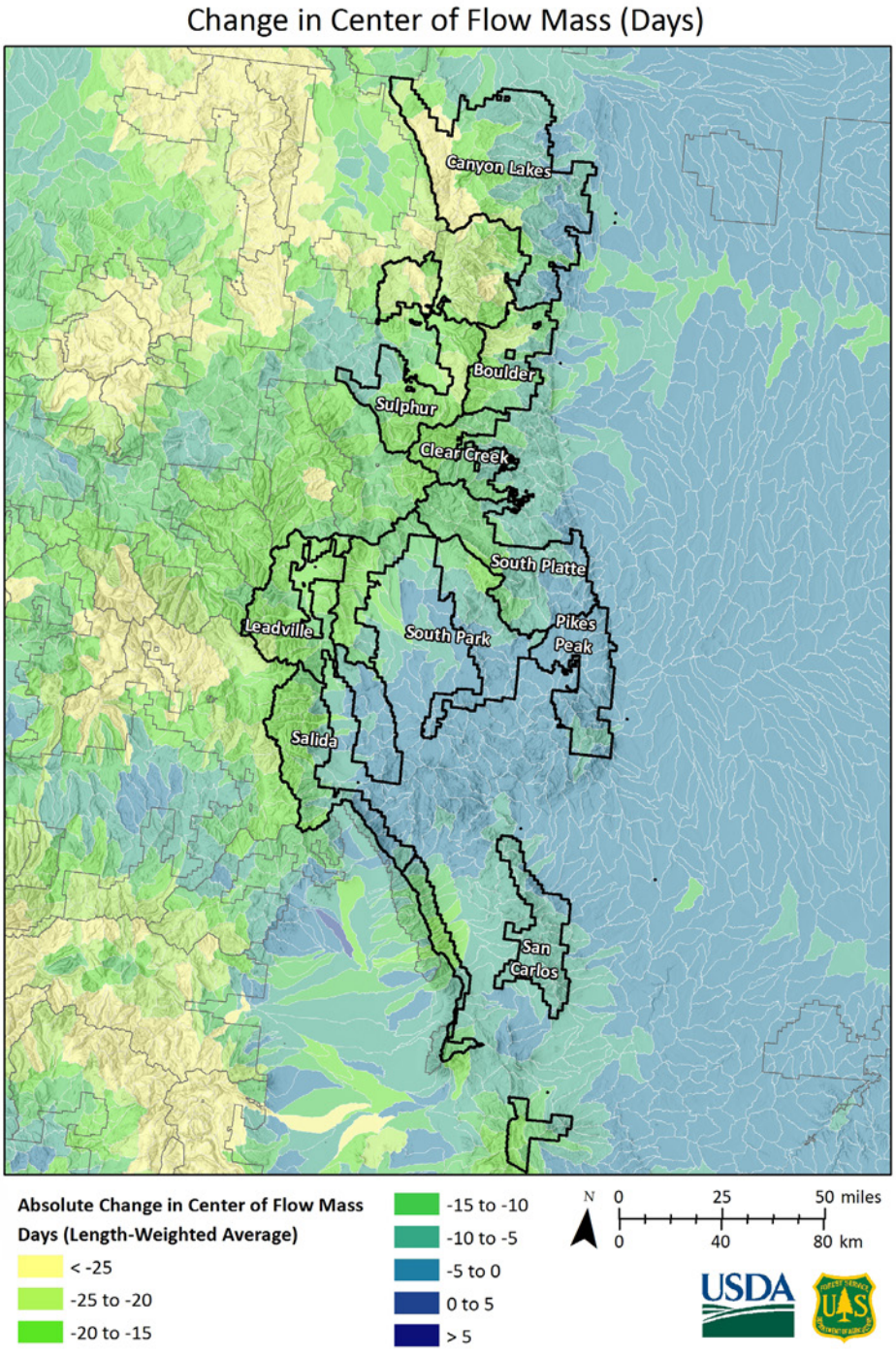


Figure 6—Absolute change in center of flow mass days between historical data (1977–2006) and future (2071–2099) projections.

Infrastructure Management For Extreme Weather Events and Climate Change

More resilient infrastructure designs and implementing best management practices are central to adapting to climate change. First, risks to the infrastructure can be assessed and consequences of failure evaluated. Then, specific structural mitigations require an engineered plan. Keller et al. (2021) presented a thorough range of infrastructure adaptation treatments for road maintenance, management, and closure; road-stream encroachment; road surface drainage improvements; culvert and stream-crossing structure protection and improvements; fords and low-water crossings; bridge protection and improvements; erosion prevention and erosion control; slope stabilization measures; roadway freeze-thaw effects; and fire detours (table 2). Simple strategies involve regularly cleaning out culverts, maintaining roads and ensuring surface drainage measures are functional, modifying bridge and culvert designs to accommodate greater flows, preventing culvert plugging, avoiding potential stream diversion at road-stream crossings, and correcting scour problems at bridges (Keller et al. 2021). Stabilization treatments, planned by engineers and specialists, are intended to minimize susceptibility of structures, streambanks, and steep slopes to failure and erosion. Measures can be taken to reduce flow velocity and concentration of flow and redirect water flow as needed. Erosion control, streambank protection, and scour protection encompass the use of materials such as vegetation, soil bioengineering with plants and woody debris, or conventional engineering elements of rock riprap, gabions, concrete, and turf-reinforcing mats (Keller et al. 2021). Closing and relocating roads or other infrastructure from flood zones is sometimes the best solution in terms of risk and cost-effectiveness (Keller et al. 2021). Keller and Sherar (2003) summarized many of the key best management practices for engineering design and function of roads and Furniss and Howe (2016) listed numerous ways to make road systems more resilient.

Table 2—Vulnerabilities and adaptation actions (modified from Keller et al. 2021).

Infrastructure Type	Vulnerability	Adaptation Action
Facilities	Damage from fires, floods, and wind	• Move vulnerable buildings and facilities • Develop evacuation plans • Use fire-resistant materials (e.g., metal roofs) and cover holes to protect from embers (e.g., gutters) • Thin for defensible space around structures
	Depleted water systems	Drill or deepen water wells
	Lack of available water during droughts	Increase water efficiency, decrease water use, plan for droughts
	Slope failures	Construct stable cut and fill slopes
	Rockfall	Add drainage and deep-rooted vegetation
	Debris flows	Add slope stabilization measures as needed
	Erosion	• Maintain adequate ground cover • Diffuse concentrated water • Armor eroding sites
Roads	Damage from fires, floods, and wind	• Inspect, maintain, repair • Decommission unneeded roads • Designate detour routes • Remove hazard trees
	Road-stream encroachment	Move vulnerable sections of roads out of channel migration zone
	Landslide prone areas	• Redirect streamflow away from stream bank • Stabilize unstable slopes
	Erosion/sedimentation	• Outslope roads • Maintain adequate ground cover • Improve drainage • Armor weak streambanks
	Water quality degradation	Use frequent rolling dips, cross-drains, and leadoffs
	Gully formation	• Armor pipe and dip outlets • Armor eroding ditches
	Pavement freeze-thaw damage	Use springtime road use restrictions

Table 2 continued.

Infrastructure Type	Vulnerability	Adaptation Action
Culverts	Culvert failures	Keep channel clear
	Washout from poor installation	Repair damaged culvert sections
	Lack of capacity	Add stream-diversion protection dips
	Plugging with debris, logs, and sediment	• Add trash racks to trap debris • Use stream simulation designs
Bridges	Bridge failure	Keep channel clear of debris and vegetation
	Foundation scour	Armor foundations and areas of scour potential
	Loss of clearance or capacity	Redirect channel flow away from abutments
	Timber bridges burned by wildfire	• Remove vegetation near bridges • Remove aggradation deposits • Increase clearance

National core best management practices are available for roads, trails, and recreation on National Forest System lands; these practices particularly are effective due to the requirement that road storm damage surveys be conducted shortly after storms (USDA FS 2012). Surveying roads after storms is critical to detect and prevent problems from insufficient drainage, unstable slopes, or vulnerable infrastructure. Understanding causes of failure and designs and prescriptions that reduce likelihood of future failures are discussed by Furniss and Howe (2016). Additionally, it is important to assess damage or potential for damage to water bodies to establish causes of failures and determine mitigation at damaged sites and preventive actions at similar sites.

The *Transportation Resiliency Guidebook* (USDA FS 2018) supplies a conceptual framework for practitioners at a local level to consider climate change and prepare for effects on the transportation system. New Forest Service bridges and bridge replacements, as well as culverts, are designed in accordance with the agency's aquatic organism passage stream simulation guide, making bridges and culverts substantially more resilient to climate change (Furniss et al. 2018). The Forest Service Transportation Analysis Process is intended to maintain a sustainable road network in accordance with best management practices, integrating a map of the recommended minimum road system and lists of unneeded roads and key issues with a road risk assessment to prioritize road infrastructure (Furniss and Howe 2016; USDA FS 2012). The Program for Emergency Relief for Federally Owned Roads was established to assist Federal agencies with repairing Federal roads that have incurred serious damage from a natural disaster (USDOT FHWA 2014). The Federal Highway Administration (USDOT FHWA 2012 and 2017) and the Transportation Resiliency Guidebook (USDA FS 2018) both provide a climate change and extreme weather vulnerability assessment framework for assessing vulnerability to climate change and extreme weather events.

Interactive with the Transportation Analysis Process, the Watershed Condition Framework model rated various factors that influence watershed condition (Keller et al. 2021). Every national forest in the United States completed a Watershed Condition Assessment at the sub-watershed scale (HUC-6; 10,000–40,000 acres) during 2010 (Keller et al. 2021). The model is based on 12 watershed condition indicators, for which road density, condition, and proximity to streams contributed to the ratings.

Sustainable Operations

Recommendations about sustainable operations incorporate reducing water and energy consumption and increasing water and energy efficiency, purchasing sustainable goods and services, reducing waste generation, and reusing and recycling materials for sustainable resource use and reducing greenhouse gas emissions (USDA FS 2014). The USDA has a Sustainable Buildings Implementation Plan with lists of resources available at other agencies (USDA 2011). The Leadership in Energy and Environmental Design (LEED) Green Building Rating System was developed by the U.S. Green Building Council (USGBC) for sustainable building and development practices (USDA 2015). The LEED rating systems are available for new construction and existing building operations and maintenance (USDA 2015).

Sustainable operations involve a shift toward renewable energy for climate change mitigation. Embarking on atmospheric carbon drawdown and net-zero carbon emissions is possible through existing technologies of rapid electrification based on renewable energy, such as rooftop solar, electric vehicles, and heat pumps (Griffith et al. 2020). Solar systems on buildings are one option to consider. Although buildings differ in size, grid-tied inverter systems cost about $2 to $3 per watt (S. Rivers, Black Hills Solar, Inc., personal communication). Average capacity installed for a residence typically is 4 kW to 6 kW and larger office buildings may use at minimum 10 kW (that is, $20,000 to $30,000; EnergySage 2019). Battery systems will increase the price compared to a grid-tied only solar system. In addition, depending on location, electrician costs may range from $500 to $2000 for connecting the inverter to the solar and electric panels (S. Rivers, Black Hills Solar, Inc., personal communication).

Conclusions and Future Assessments

Management concerns included identifying infrastructure within 100-year-flood boundaries. Before preparing for future flooding risk, current flood risk first needs to be established, albeit flood risk may change over time because stream flows are not stationary (Wenger et al. 2010). We modeled 100-year-flood floodplain boundaries to identify infrastructure potentially vulnerable to current flooding, based on 1994 to 2018 water flows. We used a method to approximate current flood risk that, while comprehensive, is not likely to identify all 100-year floodplains and vulnerable camping areas. Other models and confirming evidence such as a known history of floods and

hydric soils will help detect 100-year floodplains and vulnerable infrastructure. Work conducted by resource managers, hydrologists, and engineers is needed to evaluate this model and assess flood risk at recreational areas.

Based on the identified 100-year floodplains and vulnerable camping areas, more intensive local assessments can be made, incorporating modeling, field assessment, and engineered plans. Avoiding identified areas in 100-year floodplains for future development is prudent, particularly for campgrounds. For already built recreational areas, river camping is enjoyable to many visitors, but it may be necessary to balance demand for recreation against risks to human lives due to catastrophic flash floods, which are unpredictable in time and space, and sometimes offer little warning. To communicate flood risks of infrastructure to the public, posted signs should provide a visible warning of probable flood heights near infrastructure in inventoried 100-year floodplains (USDA FS 2010). Early warning systems of sirens, visitor safety communication, and evacuation plans and training are other effective safety practices (USDA FS 2010). Closing access during some seasons or removing infrastructure is an option where risks to public safety are great, such as where flooding events and 100-year floodplains are documented and recreation use occurs.

Flooding risks are likely to change with a warming climate because both precipitation and stream channels are not stationary in time (Rangwala 2024; Wenger et al. 2010). A simple strategy to identify future risk may be to identify infrastructure near streams that will receive greater precipitation in the form of extreme rainfall events or runoff in the future (Furniss and Howe 2016). Stream flows that can lead to larger floods may increase in over one-third of Arapaho-Roosevelt and Pike-San Isabel National Forests streams, with greatest increases (> 25 percent change in 25-year floods) projected to occur within tributaries of large Front Range canyons and the Upper Arkansas River. However, any areas may be vulnerable to extreme events, which are difficult to predict (Hydrologic Engineering Center 2023; Karau et al. 2024; Mora et al. 2018).

Future work ideally will continue to develop a comprehensive risk assessment of exposure and vulnerability of infrastructure, integrating a complete portfolio of infrastructure at risk, such as roads, bridges, and water supplies, to climate-related events such as floods, landslides, drought, and other natural hazards at national forest scales. Assessments are key to evaluating risk and setting priorities for work that can or needs to be done for resilience. This process involves forming an interdisciplinary team of resource managers and engineers to examine site data and history, compile available models, and rank infrastructure risk using a risk matrix to prioritize needed work or adaptation measures (Keller et al. 2021). Detailed information about environmental and administrative context, infrastructure condition and alternative options, biological resources, climate projections, and values likely are needed for accurate determinations of risk. Although models may not exist for current or future natural hazard exposure, a comprehensive assessment, along with development of a database, will help identify risks that can be known now and information needs for risk management.

Similarly, detailed information typically is not available at larger extents, but site scale inventories can be developed and combined, for example, to map potential for landslides and high debris flow hazards after fires (USGS 2022b). Preferably, a map of all infrastructure in relation to potential hazards will be produced, with an associated risk rating, although such a map has not yet been produced for any National Forest.

References

Afshari, S.; Tavakoly, A.A.; Rajib, M.A.; [et al.]. 2018. Comparison of new generation low-complexity flood inundation mapping tools with a hydrodynamic model. Journal of Hydrology. 556: 539–556. https://doi.org/10.1016/j.jhydrol.2017.11.036

Arapaho and Roosevelt National Forests and Pawnee National Grassland [ARP]. 2013. Flood impact assessment report. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5440234.pdf [Accessed 31 October 2019]

Bernhofen, M.V.; Whyman, C.; Trigg, M.A.; [et al.]. 2018. A first collective validation of global fluvial flood models for major floods in Nigeria and Mozambique. Environmental Research Letters. 13: 104007. https://doi.org/10.1088/1748-9326/aae014

Colorado Water Conservation Board. 2019. Colorado hazard mapping & risk MAP portal. https://coloradohazardmapping.com/hazardMapping/floodplainMapping/ [Accessed 31 October 2019]

EnergySage. 2019. Commercial and industrial solar panel installations. https://www.energysage.com/solar/101/types-solar-installations/commercial-and-industrial-solar/ [Accessed 31 October 2019]

Furniss, M.J.; Howe, C. 2016. Regional-scale climate change vulnerability assessment for infrastructure in the National Forests and Grasslands of the Rocky Mountain Region: Including roads, trails, and recreational facilities. USDA Forest Service, Rocky Mountain Region. 117 p. https://www.fs.usda.gov/research/treesearch/61145

Furniss, M.J.; Little, N.J.; Peterson, D.L. 2018. Effects of climate change on infrastructure. In: Halofsky, J.E.; Peterson, D.L.; Ho, J.J.; [et al.]., eds. Climate change vulnerability and adaptation in the Intermountain Region [Part 2]. Gen. Tech. Rep. RMRS-GTR-375. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 339–362. Chapter 11. https://www.fs.usda.gov/research/treesearch/56121

Griffith, S.; Calisch, S.; Fraser, L. 2020. Rewiring America: A handbook for winning the climate fight. https://www.rewiringamerica.org/handbook [Accessed 31 July 2021]

Hanberry, B.B. 2020. Compounded heat and fire risk for future U.S. populations. Sustainability. 12(8): 3277. https://doi.org/10.3390/su12083277

Hydrologic Engineering Center. 2023. FEMA's future of flood risk data initiative. https://www.hec.usace.army.mil/confluence/hecnews/spring-2023/fema-s-future-of-flood-risk-data-initiative?focusedCommentId=139736276 [Accessed 9 July 2023]

Interagency Committee on Water Data [IACWD]. 1982. Guidelines for determining flood flow frequency. Bulletin 17B, Interagency Committee on Water Data, Hydrology Subcommittee. https://water.usgs.gov/osw/bulletin17b/dl_flow.pdf [Accessed 22 January 2020]

Johnson, J.M.; Munasinghe, D.; Eyelade, D.; [et al.]. 2019. An integrated evaluation of the National Water Model (NWM)–Height Above Nearest Drainage (HAND) flood mapping methodology. Natural Hazards Earth Systems Science. 19: 2405–2420. https://doi.org/10.5194/nhess-19-2405-2019

Karau, E.C.; Hanberry, B.B.; Dillon, G.K. 2024. Chapter 9: Fire risk and climate change in the Colorado Front Range. In: Hanberry, B.B.; Foster, A., eds. Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. **XX-XX**. https://doi.org/10.2737/RMRS-GTR-438

Keller, G; Sherar, J. 2003. Low-volume roads engineering: Best management practices field guide. Washington, DC: U.S. Department of Agriculture, Forest Service, Office of International Programs and U.S. Agency for International Development. 158 p. https://roadsforwater.org/wp-content/uploads/2020/11/LOW-RESOLUTION-ROADS-BMP-GUIDE.pdf

Keller, G.R.; Boak, L.J.; Furniss, M.J. 2021. Climate change and infrastructure in the Sierra Nevada. In: Halofsky, J.E.; Peterson, D.L.; Buluç, L.Y; [et al.], eds. Climate change vulnerability and adaptation for infrastructure and recreation in the Sierra Nevada. Gen. Tech. Rep. PSW-GTR-272. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 49–179. Chapter 4. https://www.fs.usda.gov/research/treesearch/63218

Klein, R., Chinowsky, P., Bohren, L. 2015. Chapter 8—Transportation sector. In: Gordon, E.; Ojima, D., eds. Colorado climate change vulnerability study. Boulder and Fort Collins, CO: University of Colorado and Colorado State University. pp. 109-126. https://climate.colorado.gov/colorado-climate-change-vulnerability-study

Lall, U.; Johnson, T.; Colohan, P.; [et al.]. 2018: Water. In: Reidmiller, D.R.; Avery, C.W.; Easterling, D.R. [et al.]., eds. Impacts, risks, and adaptation in the United States: Fourth national climate assessment, volume II. Washington, DC: U.S. Global Change Research Program: 145–173. https://doi.org/10.7930/NCA4.2018

Liang, X.; Lattenmaier, D.P. 1994. A simple hydrologically based model of land surface water and energy fluxes for general circulation models. Journal of Geophysical Research. 99(D7): 14415–14428. https://doi.org/10.1029/94JD00483

Liu, Y.Y.; Maidment, D.R.; Tarboton, D.G.; [et al.]. 2018. A CyberGIS integration and computation framework for high-resolution continental-scale flood inundation mapping. Journal of the American Water Resources Association. 54: 770–784. https://doi.org/10.1111/1752-1688.12660

Mora, C.; Spirandelli, D.; Franklin, E.C.; [et al.]. 2018. Broad threat to humanity from cumulative climate hazards intensified by greenhouse gas emissions. Nature Climate Change. 8:1062–1071. https://doi.org/10.1038/s41558-018-0315-6

National Research Council. 1993. In our own backyard: Principles for effective improvement of the Nation's infrastructure. Washington, DC: The National Academies Press. 128 p. https://doi.org/10.17226/2205

National Research Council. 2009. Mapping the zone: Improving flood map accuracy. Washington, DC: The National Academies Press. 136 p. https://doi.org/10.17226/12573

Neumann, J.E., Price, J., Chinowsky, P. [et al.]. 2015. Climate change risks to U.S. infrastructure: Impacts on roads, bridges, coastal development, and urban drainage. Climatic Change. 131:97–109. https://doi.org/10.1007/s10584-013-1037-4

Nobre, A.D.; Cuartas, L.A.; Momo, M.R.; [et al.]. 2016. HAND contour: A new proxy predictor of inundation extent. Hydrological Processes. 30: 320–333. https://doi.org/10.1002/hyp.10581

Office of Inspector General [OIG]. 2017a. Forest Service deferred maintenance. https://www.usda.gov/oig/webdocs/08601-0004-31.pdf [Accessed 22 January 2020]

Office of Inspector General [OIG]. 2017b. FEMA needs to improve management of its flood mapping programs. https://www.documentcloud.org/documents/4066233-OIG-17-110-Sep17.html [Accessed 22 January 2020]

Office of Water Prediction. 2019. The National Water Model. https://registry.opendata.aws/nwm-archive/ [Accessed 22 January 2020]

Pralle, S. 2019. Drawing lines: FEMA and the politics of mapping flood zones. Climatic Change. 152: 227–237. https://doi.org/10.1007/s10584-018-2287-y

Rangwala, I. 2024. Chapter 3: Future climate scenarios for Colorado's Front Range forests. In: Hanberry, B.B.; Foster, A., eds. Front Range climate change vulnerability assessment for National Forests. Gen. Tech. Rep. RMRS-438. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: p. 39-56. https://doi.org/10.2737/RMRS-GTR-438

Rennó, C.D.; Nobre, A.D.; Cuartas, L.A.; [et al.]. 2008. HAND, a new terrain descriptor using SRTM-DEM: Mapping terra-firme rainforest environments in Amazonia. Remote Sensing of Environment. 112: 3469–3481. https://doi.org/10.1016/j.rse.2008.03.018

Risk & Investment. 2017. Comparing flood maps. https://riskandinsurance.com/comparing-flood-maps/ [Accessed 22 January 2020]

Rodda, H.J. 2005. The development and application of a flood risk model for the Czech Republic. Natural Hazards. 36: 207–20. https://doi.org/10.1007/s11069-004-4549-4

U.S. Department of Agriculture [USDA]. 2010. Albert Pike review team report. https://media.arkansasonline.com/news/documents/2010/10/22/albertpikereviewteamreport0924.pdf [Accessed 31 October 2019]

U.S. Department of Agriculture [USDA]. 2011. Sustainable buildings implementation plan. https://www.dm.usda.gov/emd/docs/SustainableBuildingsImpPlan.pdf [Accessed 31 October 2019]

U.S. Department of Agriculture, Forest Service [USDA FS]. 2010. Recreation visitor safety report. https://media.arkansasonline.com/news/documents/2010/10/22/recreationsafetyreport1018.pdf [Accessed 31 October 2019]

U.S. Department of Agriculture, Forest Service [USDA FS]. 2012. National best management practices for water quality management on National Forest System lands. Volume 1: National core BMP technical guide. FS-990a. Washington, DC: U.S. Department of Agriculture, Forest Service. https://www.fs.usda.gov/naturalresources/watershed/pubs/FS_National_Core_BMPs_April2012.pdf

U.S. Department of Agriculture, Forest Service [USDA FS]. 2014. Sustainable operations. https://www.fs.fed.us/sustainableoperations/index.shtml [Accessed 21 February 2022]

U.S. Department of Agriculture, Forest Service [USDA FS]. 2018. U.S. Forest Service transportation resiliency guidebook: Addressing climate change impacts on U.S. Forest Service transportation assets. Washington, DC: U.S. Department of Agriculture, Forest Service.

U.S. Department of Transportation, Federal Highway Administration [USDOT FHWA]. 2012. Climate change and extreme weather vulnerability assessment framework. http://www.fhwa.dot.gov/environment/climate_change/adaptation/resources_and_publications/vulnerability_assessment_framework/fhwahep13005.pdf [Accessed 31 October 2019]

U.S. Department of Transportation, Federal Highway Administration [USDOT FHWA]. 2014. Emergency relief for federally owned roads: Disaster assistance manual. FHWA-FLH-15-001. Washington, DC. 115 p. https://highways.dot.gov/sites/fhwa.dot.gov/files/disaster-assistance-manual.pdf

U.S. Department of Transportation, Federal Highway Administration [USDOT FHWA]. 2017. Vulnerability assessment and adaptation framework. 3rd edition. FHWA-HEP-18-020. Washington, DC. 86 p. https://www.fhwa.dot.gov/environment/sustainability/resilience/adaptation_framework/climate_adaptation.pdf

U.S. Geological Survey [USGS]. 2022a. Colorado flood database. https://co.water.usgs.gov/projects/COFloodDB/html/COFloodMap.html [Accessed 21 February 2022]

U.S. Geological Survey [USGS]. 2022b. Emergency assessment of post-fire debris-flow hazards. https://landslides.usgs.gov/hazards/postfire_debrisflow/ [Accessed 21 February 2022]

Wenger, S.J.; Luce, C.H.; Hamlet, A.F.; [et al.]. 2010. Macroscale hydrologic modeling of ecologically relevant flow metrics. Water Resources Research. 46: W09513. https://doi.org/10.1029/2009WR008839

Wing, O.E.; Bates, P.D.; Sampson, C.C.; [et al.]. 2017. Validation of a 30 m resolution flood hazard model of the conterminous United States. Water Resources Research. 53: 7968–7986. https://doi.org/10.1002/2017WR020917

Wobus, C.; Lawson, M.; Jones, R.; [et al.]. 2014. Estimating monetary damages from flooding in the United States under a changing climate. Journal of Flood Risk Management. 7: 217–229. https://doi.org/10.1111/jfr3.12043

Zhang, J.; Huang, Y.F.; Munasinghe, D.; [et al.]. 2018. Comparative analysis of inundation mapping approaches for the 2016 flood in the Brazos River, Texas. Journal of the American Water Resources Association. 54: 820–833. https://doi.org/10.1111/1752-1688.12623

Zheng, X.; Hanberry, B. B. 2020. Modeled 100-year flood zones for Front Range National Forests. Fort Collins, CO: Forest Service Research Data Archive. https://www.fs.usda.gov/rds/archive/catalog/RDS-2020-0073 [Accessed 21 February 2022]

Zheng, X.; Tarboton, D.G.; Maidment, D.R.; [et al.]. 2018. River channel geometry and rating curve estimation using height above the nearest drainage. Journal of the American Water Resources Association. 54: 785–806. https://doi.org/10.1111/1752-1688.12661

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Vulnerability of Ecosystem Services Along the Colorado Front Range

Travis Warziniack, Elizabeth Cox, Pam Froemke, Delilah Jaworski, Josh Sidon

Introduction

Ecosystem services, broadly defined, are the benefits humans get from nature. When first introduced, the phrase "ecosystem services" was meant to capture many of the benefits resource managers (and humans, in general) had largely ignored in economic analyses, such as purification of water and pollination of plants (Daily 1997). Managing for ecosystem services meant improving the condition of the natural system (i.e., natural capital) in a way that made it more likely that those services would be provided (Brown et al. 2007). Ecosystem services were initially held distinct from "ecosystem goods," which include tangible environmental products like timber, fish, and minerals. Ecosystem goods, because they form the basis of many economies, have long been studied and valued. Today, ecosystem services and ecosystem goods are usually lumped together and referred to generically as ecosystem services.

The Multiple-Use Sustained-Yield Act of 1960 directs national forests to be managed for outdoor recreation, range, timber, watershed, wildlife, and fish. The 2012 Forest Service Planning Rule further directed national forests to manage for key ecosystem services beyond the five traditional multiple uses, where key ecosystem services are defined as ecosystem services that are: (1) important to people in the broader landscape or beyond the plan area itself, and (2) likely to be affected by the management plan under revision. In practice, there is little difference between multiple-use and ecosystem services as used by Forest Service staff, and the terms multiple-use and key ecosystem services are often used interchangeably (Jaworski et al. 2018).

One of the primary purposes of an ecosystem services approach to management is to align policy outcomes with benefits to the people and communities that are connected to the forests. In many cases, focusing on the human benefits derived from national forests also expands the geographic scope of management beyond national forest boundaries. As recognized in the Shared Stewardship mission of the Forest Service, national forests are part of a landscape that provides natural capital to local economies, defines regional cultures, and regulates global climate. Social and economic forces, therefore, play an important role in understanding ecosystem services. They help us identify the users and stakeholders of the forest resources, the types of natural resources they value, and the impacts people are likely to have on the natural resources. Understanding how ecosystem services are vulnerable can help with long-term forest planning, adaptation to climate change, and engagement with local communities.

The first part of this chapter describes some of the potential hazards to ecosystem services from climate change and the social forces that are defining the Colorado Front Range (Front Range) that are likely to impact the condition and demand for ecosystem services. Climate impacts include increased frequency of fire, flood, and drought, insects and disease, and decreased snowpack; social forces include population growth

and changing demographics. Other sections focus on specific ecosystem services, including water and watershed health, mining, forage for wildlife and livestock, and carbon sequestration (an example of ecosystem services on two national forests is given below in box 1). These ecosystem services are likely to be either affected by climate change, affected by socioeconomic forces in the region (like large changes in the region's population), or play an important role in climate change policy and mitigation. (Note that some ecosystem services (e.g., recreation) are addressed in resource-specific chapters elsewhere in this report and not addressed here.) The chapter concludes with potential uses of this assessment for management.

Box 1—Within U.S. Forest Service Region 2, the Rio Grande and Grand Mesa and Gunnison (GMUG) National Forests have completed or are in the process of completing their forest plan revisions according to the 2012 Planning Rule. Key ecosystem services identified and addressed in those plans are shown below.

Rio Grande	GMUG
Fish and wildlife	Forage for grazing
Forage	Timber
Mineral deposits	Cultural, historic, sacred sites
Nontimber forest products	Outdoor recreation and human enjoyment of fish and wildlife species
Timber	Solitude, spirituality, sense of escape
Cultural, historic, sacred sites	Water resources
Landscapes and features providing recreation and scenery	--
Solitude, spirituality, sense of escape	--
Water quantity, quality, storage, and flood control	--
Pollination	--
Support of habitat, species diversity, abundance, and distribution	--

Drivers of Change

Climate change is already affecting ecosystem services and communities throughout the Front Range, with changes expected to become more severe in the future. Since the late 19th century, temperatures in the Pike-San Isabel and Arapahoe-Roosevelt National Forests (PSI and AR, respectively) have increased by 2 °F, with an expected increase by mid-century of 2.5 to 5.5 degrees. These temperature increases have significant impacts on water-related ecosystem services by increasing the frequency of drought and extreme runoff events, shifting the timing of annual runoff, increasing fire frequency, and increasing water demand for human use.

Social changes currently underway along the Front Range compound impacts due to climate change. The dominant social force likely to affect ecosystem services on the Front Range is population growth, expected to affect both the supply of and demand for ecosystem services. The conversion of open space to urban development, for example, can cause habitat fragmentation and adversely affect water quality while it can also increase demand for recreation opportunities and water for municipalities.

Between 1980 and 2010, Colorado's population grew by 74 percent, with the highest growth rates seen along the Front Range and I-70 corridor. The Front Range Urban Corridor is now home to an estimated 4.5 million people. According to the Colorado State Demography Office, population is expected to increase 61 percent statewide between 2010 and 2050, with the Front Range and I-70 corridor continuing to see higher than average rates of growth. Figure 1 shows population and per capita personal income projections for the Denver metropolitan statistical area[1] from Wear and Prestemon (2019) for five Shared Socioeconomic Pathways (SSPs). The SSPs are trajectories of population, demographics, and income scenarios over the next century. The SSP projections are bounded by SSP5 (high growth) and SSP3 (low growth). They are usually paired with climate change scenarios to examine interactions between social forces, climate change, and other natural resources. Figure 2 shows Colorado-wide population projections by county for the low growth (SSP3) and high growth (SSP5) SSPs. Front Range counties Douglas, El Paso, and Arapaho Counties are projected to have the highest growth rates in the State; projections for personal income per capita show the highest growth in Douglas, Clear Creek, and Gilpin Counties.

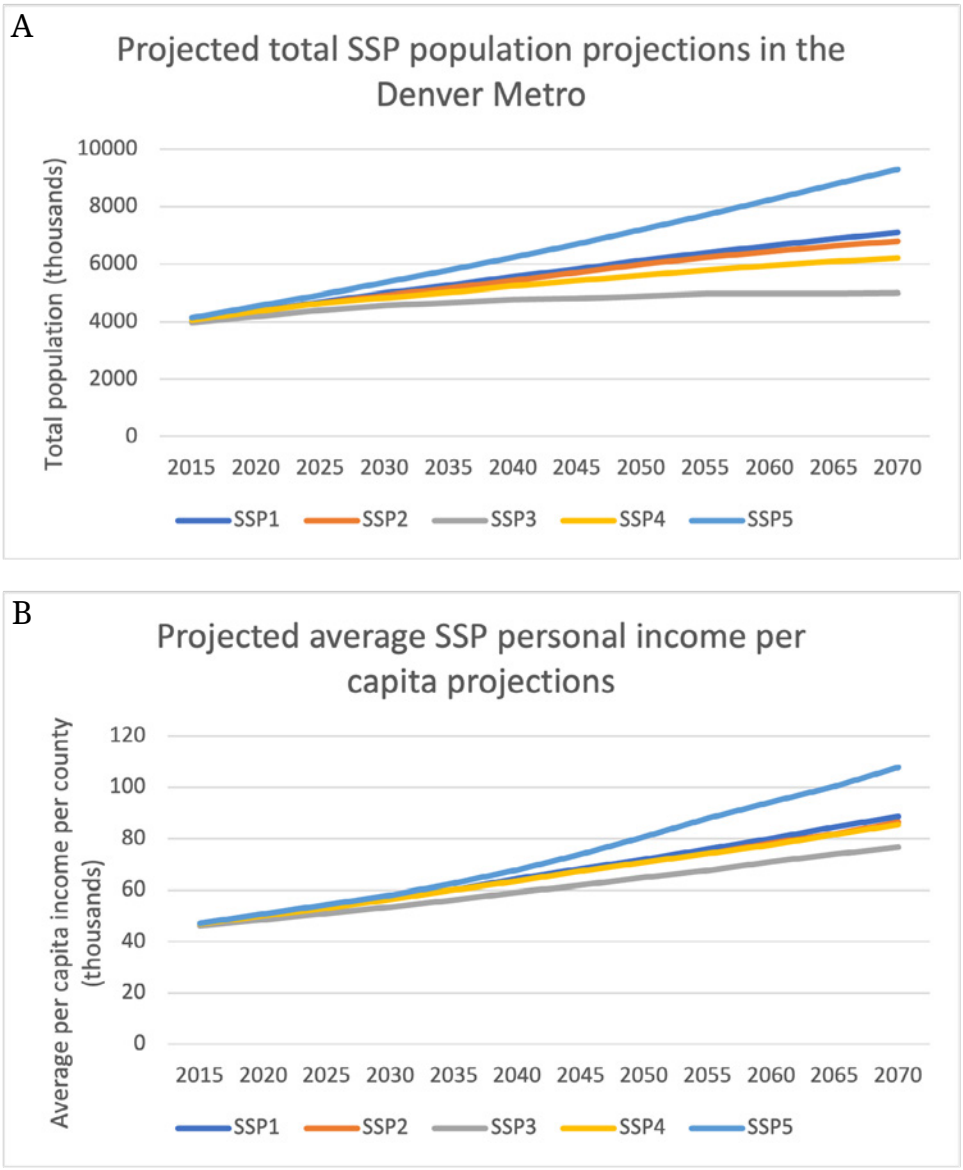


Figure 1a-b—Population and personal income projections for the Denver metro (Wear and Prestemon 2019).

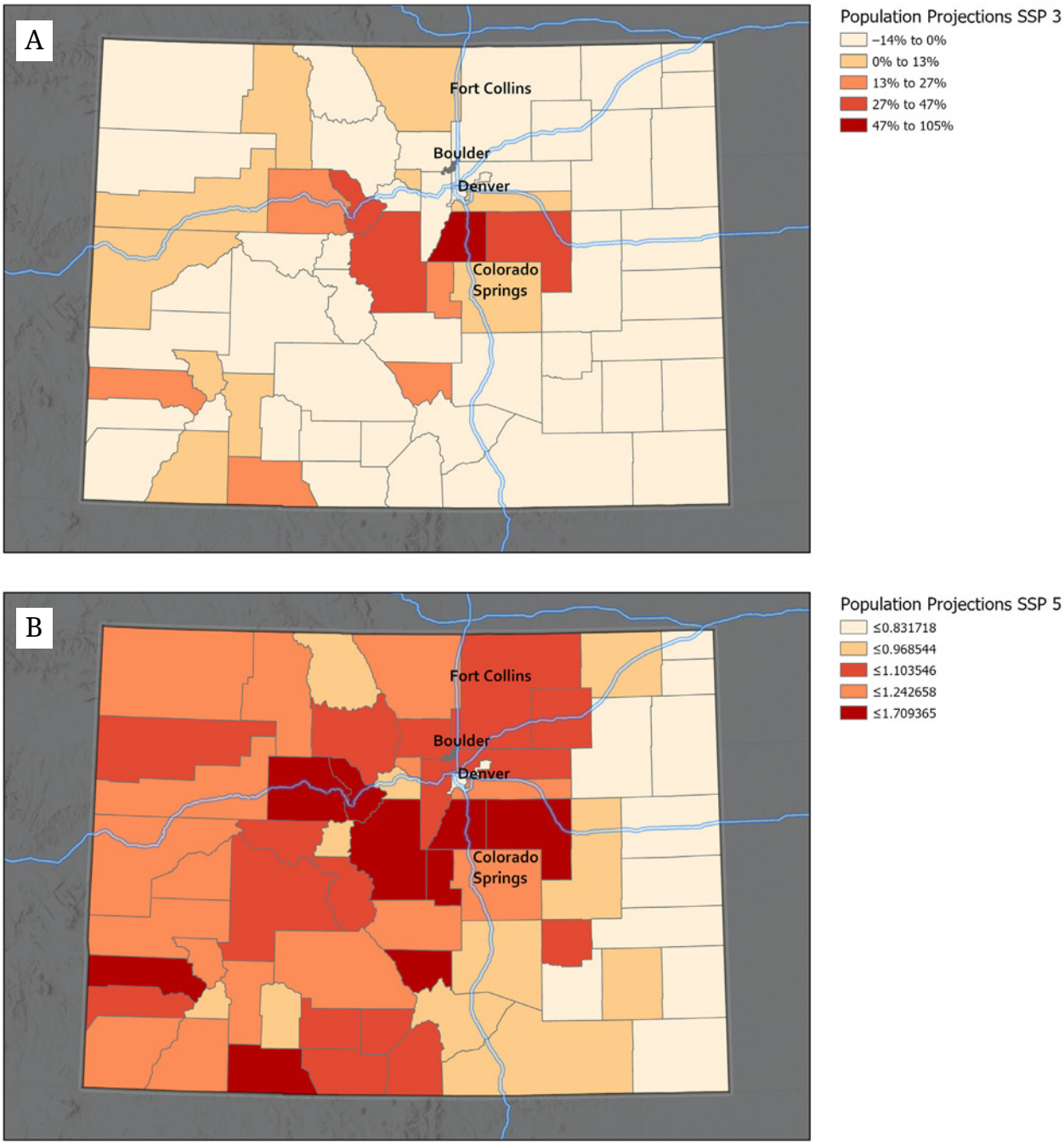


Figure 2a-b—County-level population growth projections for Colorado (Wear and Prestemon 2019).

Much of this growth will spill into the wildland-urban interface (WUI), an area of particular concern for the U.S. Forest Service. Figure 3 shows the projected growth in Colorado's WUI. Knight et al. (1995) find that, because much of the higher-elevation land in the Front Range is federally owned, private developable land in the WUI is often confined to the lower elevations, which are more likely to contain sensitive meadow, grassland, and riparian zones and are more fire prone. Threats to the region, particularly watersheds, due to development can be seen on the interactive Forest Service maps for the Forests to Faucets 2.0 Story Map.[2] Increases in fencing, buildings, and roads that accompany WUI development increase noise, human presence, and habitat fragmentation, and have been linked to reduction of native species and the introduction of exotic species. Domestic cats that range outdoors have the potential to significantly decrease songbird populations and depress prey populations for small and medium-sized predators (Loss et al. 2013). While these impacts are not directly related to climate, they are likely to increase sensitivity to climate change.

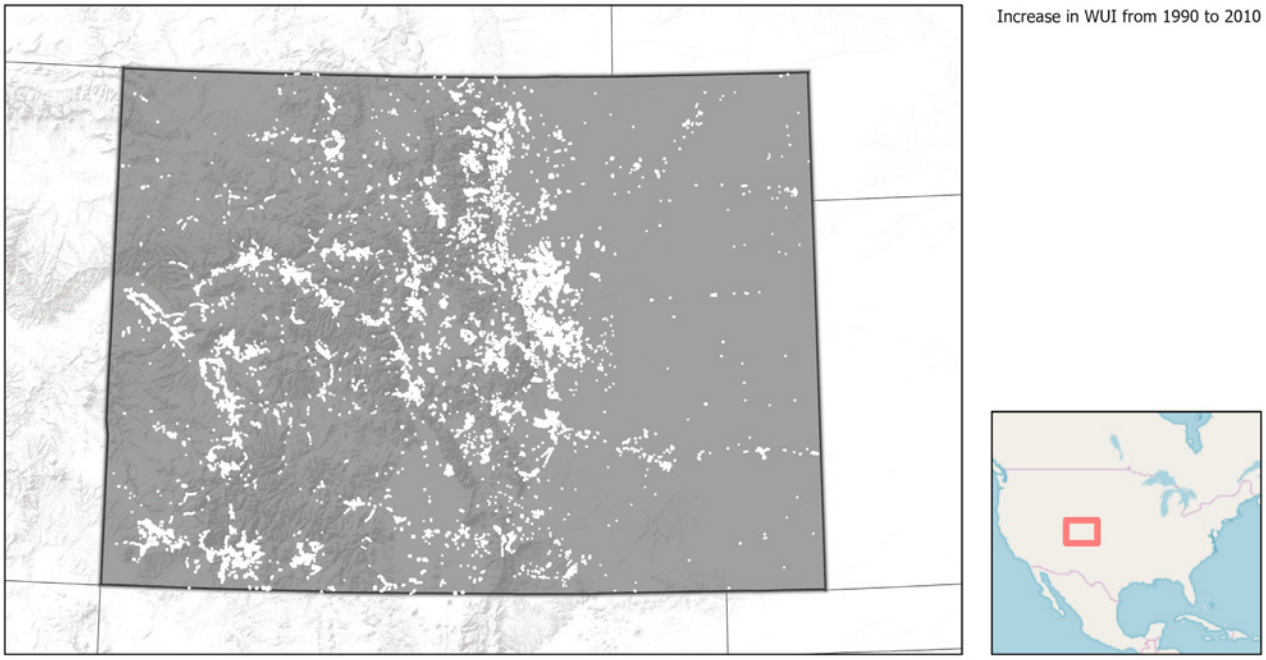


Figure 3—Increase in wildland-urban interface (WUI) throughout Colorado (1990–2010). White areas show the expansion of WUI from 1990 to 2010.

Social Vulnerability Index

An oft-cited measure of vulnerability to climate change and other environmental stressors is the "social vulnerability index." Such indices attempt to measure a community's vulnerability and resilience to climate change, particularly to sudden extreme events like fires and floods that are projected to be more common. Herein, we examine variables included in the Center for Disease Control's (CDC) social vulnerability index (SVI), which uses 15 variables from the U.S. Census at the tract level to help officials assess a community's vulnerability to hazards. The CDC's SVI index shows the relative vulnerability of census tracts throughout the United States, in aggregate and disaggregated according to four themes: socioeconomic status, household composition and disability, minority status and language, and housing and transportation (figs. 4 and 5). Low values indicate low vulnerability and high values indicate high vulnerability. The socioeconomic status theme includes a number of income variables; household composition includes variables related to age and disabilities; minority status and language includes ethnic, racial, and language information; and the housing and transportation theme includes variables related to housing density and occupancy rates, number of people in homes, and whether households have vehicles.

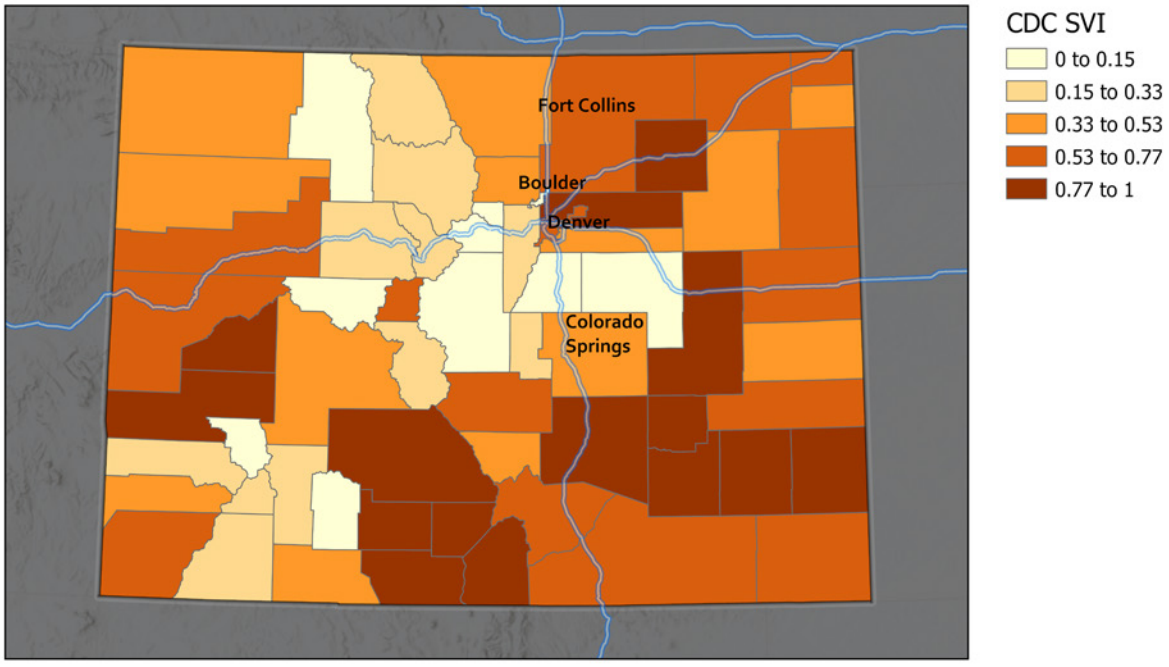


Figure 4—Aggregate Social Vulnerability Index (SoVI) for Colorado (0 = low vulnerability, 1 = high vulnerability)

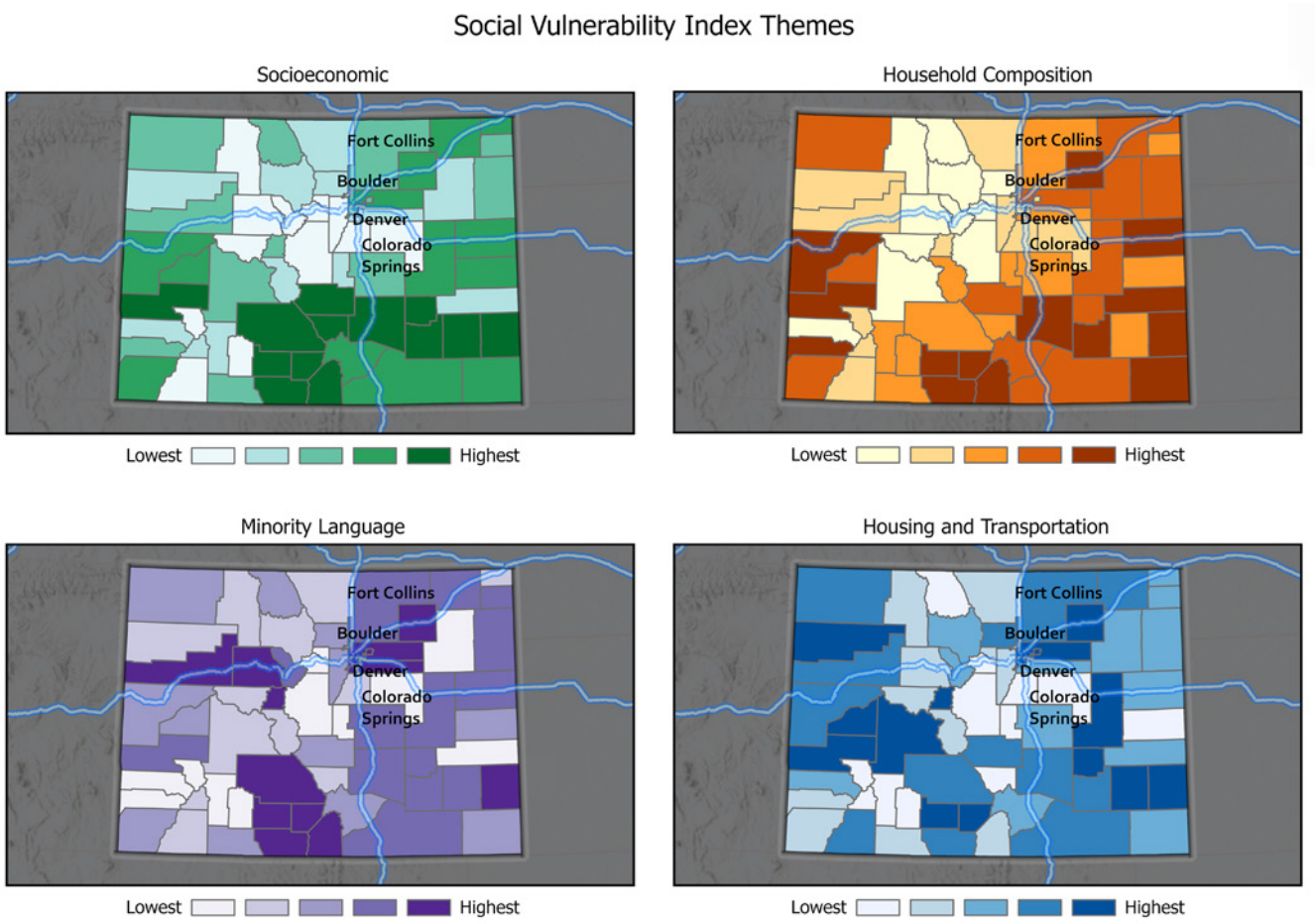


Figure 5—Social Vulnerability Index (SoVI) by theme (0 = low vulnerability, 1 = high vulnerability).

Table 1 compares SVI scores for census tracts in Colorado that overlap with Front Range national forests boundaries and for census tracts that do not. On average, tracts near the national forests are less vulnerable across all categories except Housing and Transportation. Strong economic growth in many Front Range communities have increased regional incomes (Douglas County, Colorado, has the sixth highest median household income among all counties in the United States), and the average unemployment rate for census tracts near Front Range national forests is about 13 percent lower than that for the rest of the State. The high vulnerability score for Housing and Transportation relative to other scores on the Front Range reflects a high percentage of multi-unit housing, which is not surprising given the role of tourism and second homes in many of the mountain communities. At the time of this writing, however, many of the mountain communities are heavily impacted by COVID-19, and the relatively close living quarters and mixing of people in tourist communities has been highlighted as a potential cause for higher-than-average rates of spread and an exceptional challenge for these communities.

Table 1—Relative SVI (social vulnerability index) rankings for Front Range and non-Front Range (OTHER) Census tracts (0 = low vulnerability, 1 = high vulnerability).

Census tract type	Total SVI	Socio-economic	Household composition	Minority status	Housing and transportation
Front Range	0.3816	0.3946	0.3876	0.2566	0.5109
Other	0.5072	0.5062	0.5055	0.5149	0.5004

Notably, even though the region might not be highly vulnerable according to indices like the SVI, individual households and community groups might be highly vulnerable. Regional averages can mask many issues households face, particularly when an area is experiencing rapid growth. Higher property values mean higher rents and the potential to displace poorer residents. Such displacement has been shown to adversely affect social-emotional outcomes for early and middle school children. And while there may be job growth in rural Front Range communities, many of those jobs are low paying jobs in the service and tourism sectors. In 2016, the tourism sector was the second largest provider of jobs in Colorado.[3] During recessionary periods, the demand for tourism-related goods and services generally decreases more than for other goods. Mountain communities in the Front Range experienced this phenomenon following the collapse of the real estate market in 2008. The hospitality and tourism industry suffered as a result of the economic crisis and subsequent recession. During the recovery period from the recession, occupancy rates at ski lodges in the Rocky Mountains dropped by approximately 12.4 percent (Berwyn 2009). Similar impacts seem to currently be playing out due to the COVID-19 pandemic, resulting in declines in the service sector, and closing of company-provided housing for ski resort and other hospitality workers. While a good long-term investment for regional growth, such dependency on an industry with cyclical variability gives the region lower resiliency from economic turmoil. Social changes and the unique features of rural and WUI communities near national forests suggest the need for a new vulnerability index, instead of the one used by the CDC, that captures local employment related to the Forest Service.

Minority Populations and Engagement With Front Range National Forests

By 2044, the United States is projected to be a majority-minority nation (Colby and Ortman 2015). While not directly tied to climate change, national forests that fail to recognize and manage for this trend will fail to meet the Forest Service's goal of delivering benefits to the public (the entire public) and risk becoming irrelevant (Chavez 2000; Dawson et al. 2016; Flores et al. 2018). Adapting to climate change should account for the diversity of stakeholders. Flores et al. (2018) investigated differences between the demographics of communities surrounding national forests and visitors on the forests. They measure what they call an "inequality gap," that measures the disparity in national forest utilization between racial minorities and their white counterparts. The residents of counties around the Arapaho-Roosevelt and Pike-San Isabel National Forests are 39 percent and 42 percent minorities. Comparatively,

minorities make up 8 percent and 4 percent of total visitation on the Arapaho-Roosevelt and Pike-San Isabel National Forests. These percentages imply an inequality gap for the Arapaho-Roosevelt of 31, and an inequality gap for the Pike-San Isabel of 38. The national average inequality gap is 23, making the Front Range national forests among the least diverse national forests in the country, which is true, in general, for national forests throughout Region 2. Chavez (2000) suggests the best way to include under-represented groups is to invite, include, and involve minorities in management decisions and programs. "Inviting" could be as simple as making sure outreach materials are multilingual. "Including" means including diverse groups in decision processes and soliciting diverse opinions on planned projects. "Involving" means getting diverse groups involved in implementing, a key part of which is including minorities in the workforce.

Understanding social networks among minorities and people that describe themselves as having the ability to speak English "less than well" is important to a community's ability to cope with climate change, especially with regard to providing emergency assistance during disasters. During the 2013 Front Range floods, more than 11,000 people were evacuated, requiring community-wide coordination with emergency services, local volunteers, and the National Guard. After being displaced from their residences, non-English speakers and immigrant households often have fewer options to stay with friends and fewer informal short-term housing options. Minority groups are more vulnerable during emergency situations due to (1) conflict between minority communities and local agencies and governments and (2) diminished social and resource capital (Fothergill 1999). Often, local agencies in minority communities have less resources than their Caucasian counterparts. Those who have access to adequate resources may be less inclined to use emergency services due to a lack of trust in government officials, particularly among undocumented immigrants. Less wealthy minority communities also tend to have a higher dependency on alternate or public forms of transportation, which may prevent individuals from having the ability to flee from a dangerous situation (Fothergill 1999).

Water Resources

About 42 percent of Colorado's water originates on forested land (Brown et al. 2008). Most of this begins in higher elevations as snowfall and becomes available for consumptive use with spring runoff. Tables 2 and 3 show the watersheds that get at least 10 percent of their water from the Arapaho-Roosevelt and Pike-San Isabel National Forests (Luce 2015) with surface water intakes for household consumption. Surface water intakes from these national forests serve over 2.6 million people and 2.3 million people on the Arapaho-Roosevelt and Pike-San Isabel, respectively. Note, some of the watersheds are shared between these national forests and are counted toward both forests' totals. The Upper South Platte, for example, serves about 1.6 million people and has contributions from both Forests.

While water yield is provided at a fairly fine spatial scale, historical water demand data is much coarser. This section makes use of Brown et al. (2013), who project water use for 4-digit HUCs for several climate change scenarios. The 4-digit Hydrologic Unit Codes (HUCs) used in this section are the North Platte (HUC 1018), South Platte (HUC 1019), Upper Arkansas (HUC 1102), Upper Cimarron (HUC 1104), and Colorado Headwaters (HUC 1401).

Table 2—Contributions to 8-digit HUCs (hydrologic unit codes) from the Arapaho-Roosevelt National Forests.

HUC name	8-digit HUC	Water contribution (%)	Forest land occupied (%)	Population served
Colorado Headwaters	14010001	21	23	26,829
Upper Laramie	10180010	66	12	198
Upper South Platte	10190002	15	3	1,574,518
Clear	10190004	80	46	228,954
St. Vrain	10190005	60	39	373,320
Big Thompson	10190006	13	37	210,577
Cache La Poudre	10190007	75	39	199,252
Crow	10190009	12	24	19,032

Table 3—Contributions to 8-digit HUCs (hydrologic unit codes) from the Pike-San Isabel National Forests.

HUC name	8-digit HUC	Water contribution (%)	Forest land occupied (%)	Population served
South Platte Headwaters	10190001	82	48	420,625
Upper South Platte	10190002	64	50	1,574,518
Arkansas Headwaters	11020001	82	46	16,943
Upper Arkansas	11020002	38	13	210,907
Fountain	11020003	58	23	78,378
Huerfano	11020006	45	13	7,113
Purgatoire	11020010	12	7	10,902

The largest user of water on the Front Range is irrigated agriculture, accounting for 89 percent of total consumptive water use in 2010.[4] Significant advances have been made in irrigation efficiency, leading to an overall decrease in water withdrawals for agriculture in the Front Range. Whether or not that trend will continue in a warmer climate is uncertain. Most projections show water use increasing in the agricultural sector to compensate for rising temperatures, more frequent drought, and increased needs by plants (Warziniack et al. 2022). All else equal, this would imply an increase in water demand for agriculture on the Front Range. However, there is increasing pressure to divert both land and water out of agriculture to support residential growth, and increased frequency of drought and extreme rainfall events could adversely affect the agricultural sector. Figure 6 shows projections by Brown (2013) in consumptive use of water by economic sector. These projections account for trends in regional acreage devoted to agriculture, improvements in irrigation efficiency, and impacts of rising temperatures on crops' needs, but do not account for competition for water among uses, namely between agriculture and household use. Brown's (2013) projections reflect desired water use assuming water is not more constraining than it is today, and therefore, likely overestimate actual withdrawals in the face of increasing scarcity.

The highest value and fastest growing categories of water use on the Front Range are domestic and public water use (water used by households and in public spaces) and commercial and industrial water use (water used by firms and manufacturers). Usage per capita among households has been decreasing due to improvements in appliances and more reliable water metering. These gains in efficiency, however, do not offset large population and economic growth experienced on the Front Range. Even allowing for gains in water use efficiency, consumptive water use for households is expected to double by the end of the century compared to 2010 levels. Water use in the commercial and industrial sector is expected to grow even faster, seeing a projected 280 percent increase over 2010 levels by the end of the century. These increases in demands combined with a decrease in water yield makes the Front Range one of the most vulnerable regions in the country with respect to water shortage (Foti et al. 2012).

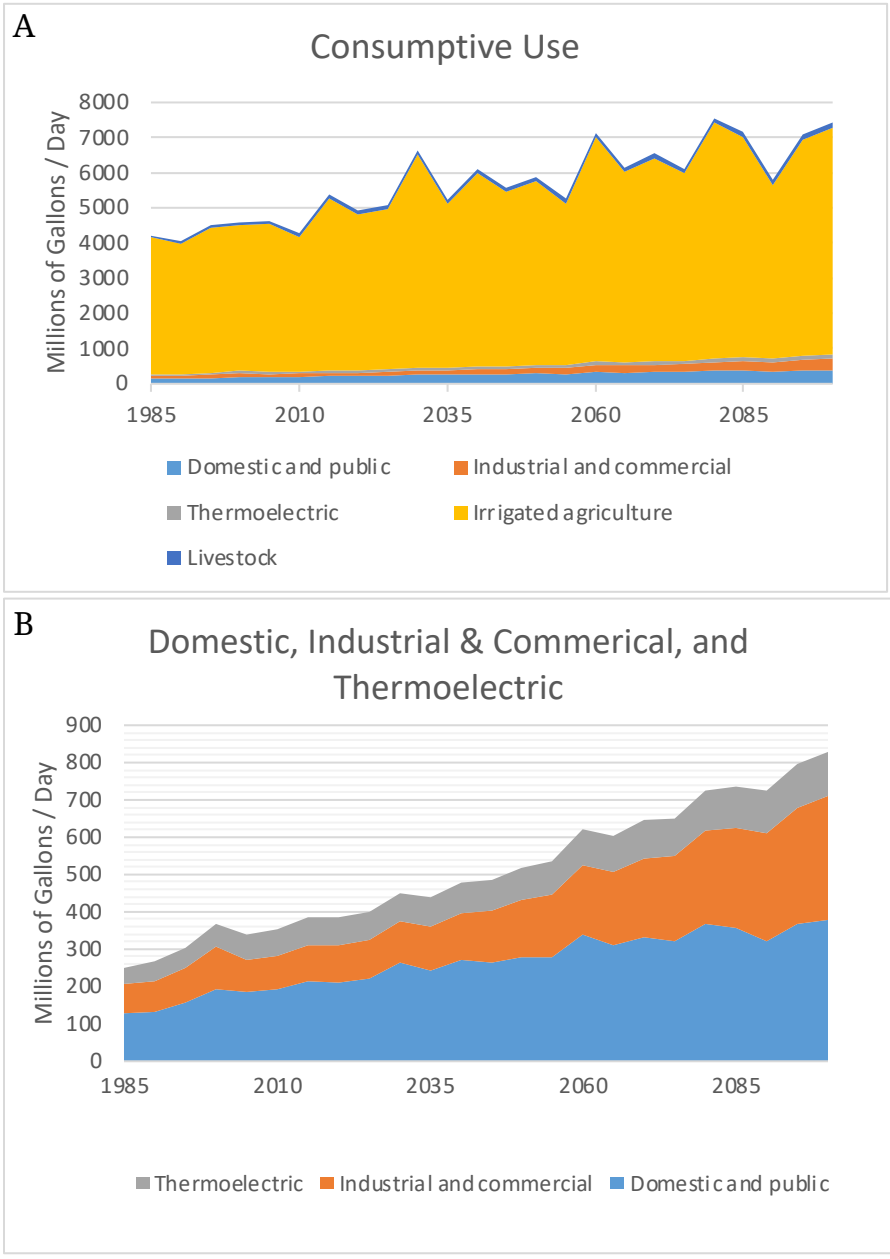


Figure 6a-b—Consumptive water use by economic sector on the Front Range.

Timing of Water Availability, Floods, and Drought

Climate change scenarios predict more frequent droughts, earlier and less reliable runoff from snowpack, and lower summer flows (Ehsani et al. 2017; Fassnacht et al. 2018). Climate models for Front Range forests show increases in spring runoff of 20 to 80 percent and decreases of 10 to 50 percent in Summer and Fall. Such conditions might necessitate greater water storage in reservoirs, particularly as warmer temperatures and drought increase the need for irrigation for crops. With lower water yields and higher demands, many of the area's existing reservoirs could occasionally be drawn down by the end of the century (fig. 7). Warziniack and Brown (2019) find that between 5 and 25 percent of the water currently going to agriculture would be needed to cover future household demands as the region gets both drier and more populous. Several regional storage projects have begun in order to respond to aging water infrastructure and a growing population. The Arapaho-Roosevelt National Forests have three proposals under review to increase storage and diversion capacity for existing projects. These proposals would likely affect the Seaman Reservoir expansion, Windy Gap Firming Project, and Gross Reservoir expansion.

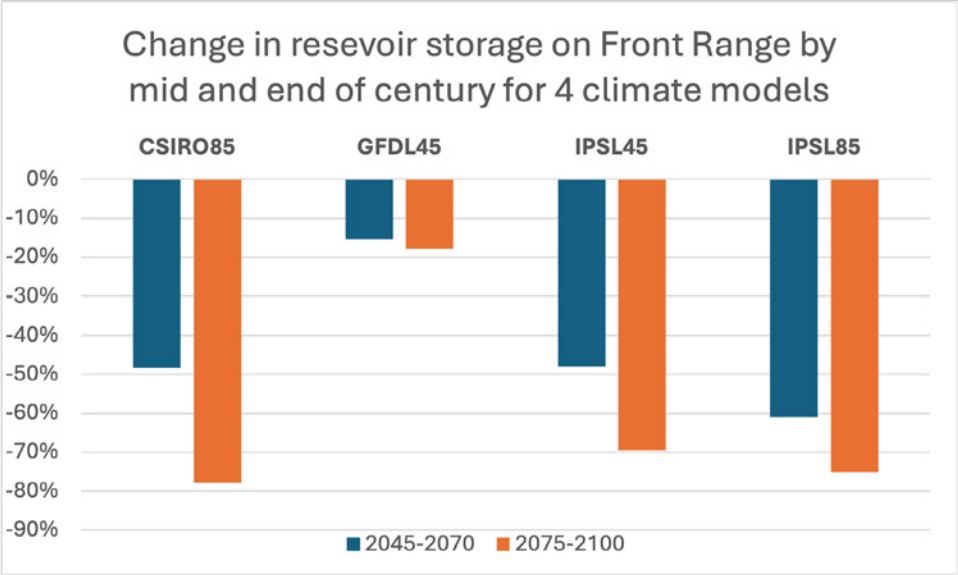


Figure 7—Reservoir storage for CSIRO, GFDL, and IPSL global climate models (GCMs) and Relative Concentration Pathways (RCPs) 4.5 and 8.5. These projections do not account for increased capacity due to infrastructure and storage investments. (Brown et al. 2008).

Climate change is also expected to lead to a higher frequency of floods in some areas of the Front Range (Human 2012). Increased runoff is predicted to be caused by a shift from snow to rain in the shoulder seasons, and from hail to rainfall in summer months. These shifts will cause more flash floods throughout the region. The 2013 flood in the Front Range caused approximately $16 million in damages to roads, bridges, trails, and other facilities. There was additional damage to resources including, but not limited to, archaeological sites, fisheries, and vegetation (ARP 2013).

In addition to damage to roads, homes, businesses, and ecosystems, floods pose a variety of other health-related risks to the community. Following the 2013 flood, tests conducted by the Colorado Health Department showed that the Big Thompson River and Boulder Creek watersheds contained high levels of *E. coli*. Both high temperatures that encourage the growth of *E. coli* bacteria and increased frequency of floods create substantial water quality vulnerabilities along the Front Range.

Water Quality and Watershed Health

Brown and Froemke (2012) calculate risk indices for sediment, nutrient, and toxics for watersheds throughout the country (figure 8). They find less risk and lower variability in conditions for Front Range watersheds compared to an average U.S. watershed. Many of the key watersheds are on protected lands, highlighting the role of national forests in providing much of the region's water supply. Major risks are due to WUI development and wildfire as shown on the Forests to Faucets 2.0 project maps (https://storymaps.arcgis.com/collections/4e450a6c7ed24f0cbae4abc1c07843b7?item=1). In 2020, the two largest wildfires in Colorado's history burned in key Front Range municipal watersheds. Harmful algal blooms are also becoming more prevalent, including in municipal drinking water supply reservoirs and in sources for irrigation water. Nutrients (nitrogen) running off burned forests are contributing to algal blooms in some reservoirs, but as population increases, public water supplies are increasingly at risk. Problems are likely to get worse in the face of growing urban areas and warmer climates.

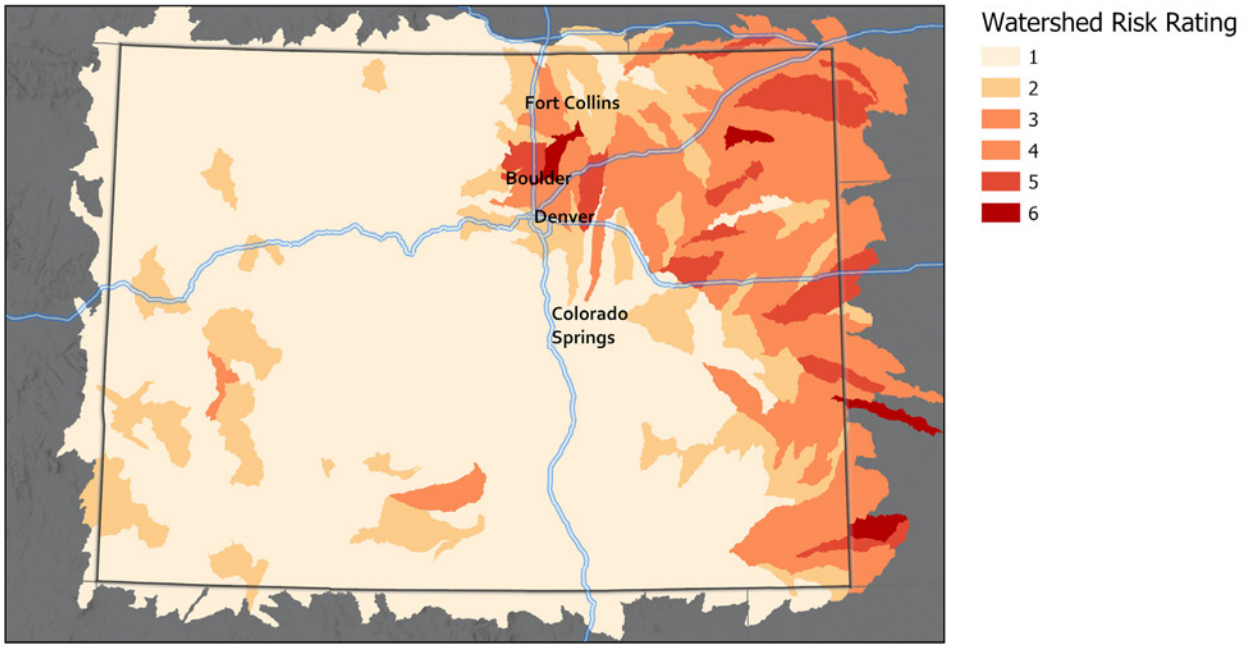


Figure 8—Risk rating for Colorado watersheds (Brown and Froemke 2012).

Risks to Water Quality Related to Mining Operations

There are over 800 abandoned mines or adits (entrances to underground mines) on the Pike-San Isabel, seven of which are considered dangerous or extremely dangerous for safety and human health. Approximately another 70 wells are considered potentially dangerous. In the Arapaho-Roosevelt National Forests, only two abandoned mines are considered dangerous, and 13 are considered potentially dangerous. Figure 9 shows the mines throughout Colorado and their safety classification.

Most of the literature related to mining activity and climate change focuses on acid rock drainage and the increased movement of pollutants such as metal sulfides due to extreme rainfall events and flooding. Watersheds containing active or abandoned mines are affected by complex interactions of surface and subsurface flows that introduce acidity and metals to the receiving stream. These influences may also emanate from natural sources in the underlying bedrock. Some ecosystems on the Front Range have not completely recovered from mining activities that took place in the 19th century, which continue to impact stream channels, sedimentation, and release of toxic chemicals (Schmidt et al. 2012; Wohl 2006, 2011). The Central City/Clear Creek Superfund site, for example, is a product of one of the most active mining areas in the state during the gold and silver rushes of the 1800s. Mining activity left behind tailings, impoundments, and draining mine adits that significantly impact groundwater and heavy metal contamination throughout the watershed. This work is ongoing, and to date, none of the performance measures have been met for remediation.[5]

Risks associated with mine drainage and leaching depend on precipitation and frequency of floods, fires, and other large-scale disturbances (Ossa-Moreno et al. 2018) as well as the level of human and wildlife exposure, all of which are expected to become more common in the future. In wetter years, higher precipitation can introduce more groundwater into the mine works, increasing the amount of exposed rock to water and the amount of water that percolates through tailings and waste rock. Most abandoned mines have exposed metal sulfides, which are harmful when in contact with water supplies and surrounding aquatic ecosystems (Colorado Geological Survey 2016; Parsons 1977; Vasilatos et al. 2015). Effects of acid mine drainage include turning surface waters red, coating creek beds with white aluminum, coating creek beds with metal deposits, and killing aquatic life. When left alone, groundwater that interacts with dangerous mine substances becomes contaminated and enters surface waters.

However, increased flow also dilutes pollutants, and surface waters tend to have lower concentrations of pollutants during wet periods. The low-flow periods, which will become more common with climate change, represent a critical time period in terms of aquatic toxicity; containment sources are undiluted by snowmelt and/or rainfall, and concentrations of mining-related constituents are often at their highest levels (Runkel et al. 2016).

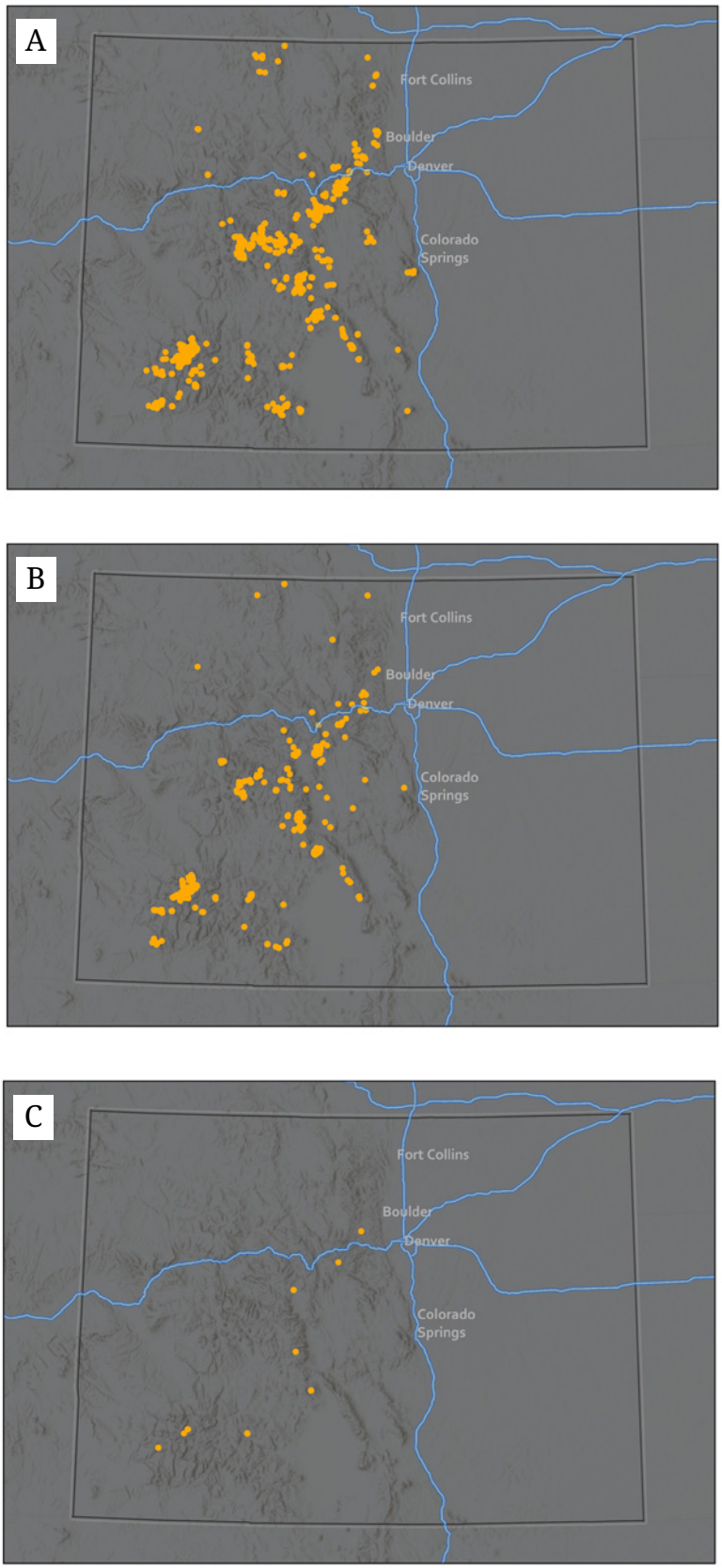


Figure 9a-c—Mines throughout Colorado at (a) slight, (b) significant, and (c) extreme risk to safety and human health.

Flooding and high levels of precipitation can cause collapses in mine roofs, create sinkholes, and cause further contamination (Polak et al. 2016; Whyatt and Varley [n.d.]). Intense rains may also flood mine shafts, damage mine works, and cause other safety concerns. In 2015, an EPA contractor attempting to repair a collapsed section of the adit at the bulkhead of the Gold King mine punctured the bulkhead, causing the release of 3 million gallons of mine water that had built up behind the bulkhead. Following the accident, the river was laced with heavy metals, causing the Animas River to face discoloration throughout Colorado, New Mexico, and Utah.

Rangeland Forage

As fragmentation of the region's rangeland on private lands increases, Federal lands will become more important for species that rely on rangeland forage. According to the Forest Service's Benefits at a Glance reports (https://www.fs.usda.gov/emc/economics/at-a-glance/benefits-to-people.shtml) the Arapaho-Roosevelt National Forests and Pawnee National Grassland support nearly 65,000 animal unit months (AUMs, defined as forage required for one cow-calf pair for one month) of livestock, approximately 3,720 local jobs, and $157,000,000 in local labor income. The Pike-San Isabel National Forests and the Cimarron and Comanche National Grasslands combine to support 152,000 AUMs, 5,260 local jobs, and $212,000,000 in local labor income.

Reeves and Mitchell (2012) find that 29 percent of Colorado's rangeland has been modified by human use. An additional 256,000 acres statewide are at risk from residential development by 2030. Reeves and Mitchell (2012) found that forage demand was below forage supply for most of Colorado, and that in the absence of drought and other disturbances, the ability of the region's rangeland to support livestock is adequate, though conditions vary at more localized levels.

Rangeland health will also be affected by livestock's dependence on reliable water supplies. Currently, the driest month of the year in the Front Range is June, which shifts in some climate scenarios. Seasonal water availability will affect plant competition and possibly cause changes in grass species. High temperatures will also degrade digestibility of forage for livestock and other grazing species (through increases in lignin in plants). Because of the seasonal shift, forage quality will likely increase more among C3 plants than C4 plants and have consequences for nutritional value (Reeves and Mitchell 2012; Trlica 2013). Frequent flash floods may also affect the formation of roots for plants, thus degrading forage health due to seasonal shifts and more extreme precipitation patterns.

Higher temperatures also increase the probability of morbidity and death among livestock and change microbial communities that spread vector-borne diseases, food-borne diseases, and host resistance (Rojas-Downing et al. 2017). In 2012 the agricultural industry across the Front Range was affected by outbreaks of anthrax, equine herpesvirus-1, vesicular stomatitis, and West Nile virus. Anthrax development is typically associated with periods of heavy rainfall, flooding, drought, and other

ecological changes, all of which are projected to become more frequent. Wittmann et al. (2001) showed warmer temperatures led to tick infestations (which thrive in high temperatures), causing livestock to lose about 18 percent of their bodyweight.

Carbon Sequestration

The Forest Service Planning Rule requires national forests to develop baseline carbon stocks and consider impacts of management actions on carbon stocks. Much of this work is being done by the agency's Office of Sustainability and Climate (USDA FS 2015) and Forest Service Research and Development (Heath et al. 2011). The following is taken directly or summarized from USDA (2015), "Baseline estimates of carbon stocks in forests and harvested wood products for National Forest System units, Rocky Mountain Region":

Forests sequester carbon in seven primary stocks. They are:

- Live trees—All live woody vegetation at least 1 inch in diameter at breast height (d.b.h.). Some estimates also separate above and below ground stocks in live trees, in which case, below ground includes coarse living roots more than 2 mm in diameter.
- Understory—All live herbaceous and woody vegetation up to 1-inch d.b.h.
- Standing dead trees—Nonliving, but otherwise follow the same definition as live trees and include estimates for below ground roots more than 2 mm in diameter.
- Down dead wood—All nonliving woody biomass with a diameter of at least 7 cm at transect lying on the ground.
- Forest floor—Litter, fumic, and humic layers and all nonliving biomass with a diameter less than 7.5 cm at transect intersection lying on the ground above the mineral soil.
- Soil organic carbon—All organic material in soil to a depth of 1 meter but excluding the coarse roots of the pools mentioned earlier.
- Harvested wood products.

In general, national forests have greater carbon density than private land (28 percent more per forested hectare). Thus, management and impacts of these forests can substantially affect the region's stock of carbon sequestered by forests.

Over the last couple of decades, many of the forests in the Rocky Mountain Region have switched from being a carbon sink to a carbon source, due to both disturbances and aging effects. Stands that were established in the late 1800s and early 1900s reached their maximum rates of productivity around the 1970s. Rates of accumulation declined naturally as these forests matured. This slowing, paired with disturbances over recent decades, has caused the forests to become carbon sources. The latest round of disturbances has led to regrowth in some areas, and these areas are likely to become carbon sinks as rates of growth increase.

The largest pool of carbon in Region 2 is aboveground carbon, followed by soil organic carbon, and carbon stored in matter on the forest floor (fig. 10). These three categories have seen an increase in carbon sequestered between 1990 and 2013. Compared to other national forests in Region 2, the Arapaho-Roosevelt and Pike-San Isabel National Forests fall near the middle among all forests in R2 for sequestered carbon and carbon density (figure 11). Total carbon stored has increased on the Arapaho-Roosevelt but decreased on the Pike-San Isabel, largely in aboveground storage (figure 12).

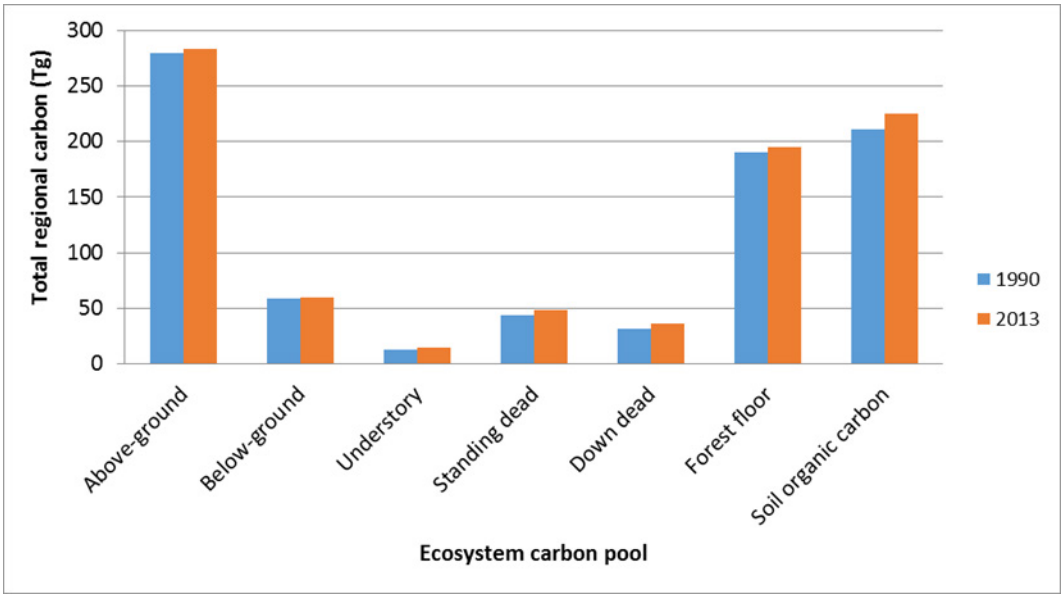


Figure 10—Ecosystem carbon pools for Region 2.

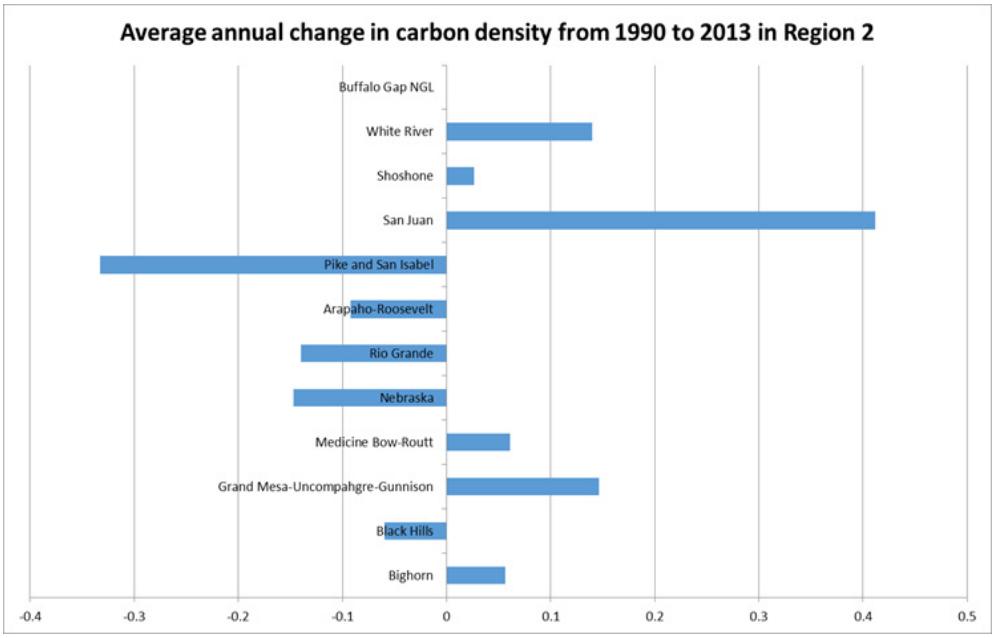
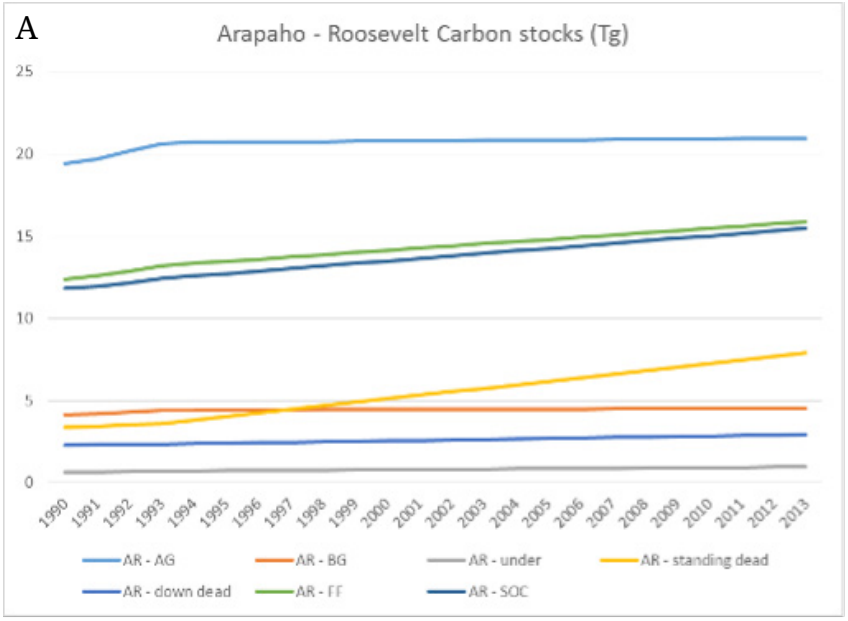


Figure 11—Average annual change in carbon density.



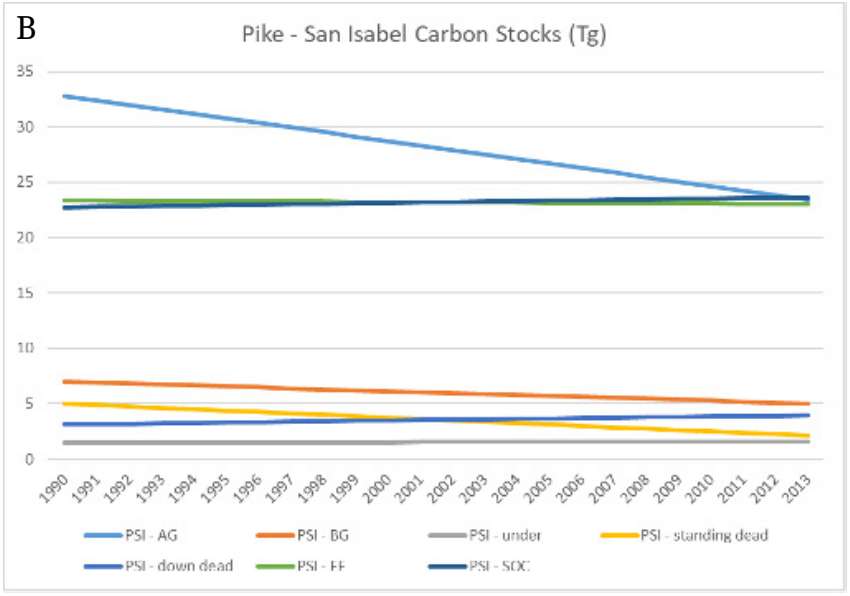


Figure 12a-b—Carbon stocks in the (a) Arapaho-Roosevelt and (b) Pike-San Isabel National Forests.

Key Points and Management Implications

Climate change impacts are likely to be amplified by social forces taking place, especially rapid population growth and shifts in demographic characteristics. Increased population living on the Front Range and in WUI communities means more people will be exposed to fires, floods, and drought, and the resulting impacts on mines and water resources. Increased fragmentation of non-Federal lands will make Front Range national forests even more important for providing ecosystem services.

In general, the Front Range population is wealthier and more educated than the national average, which suggests the region is not particularly vulnerable to climate and environmental change. These regional averages mask small pockets of highly vulnerable populations. Vulnerable people will become more vulnerable as rising home prices and lower wages strain households. Low-income families in the region are more likely to live in close living quarters, which as seen by the combined effects of the COVID-19 pandemic and 2020 wildfires, can amplify the impacts of disease outbreaks and natural disasters. The most vulnerable often lack the resources and support networks to absorb shocks as easily as the more affluent, pushing poor and vulnerable people further to the margins.

A key takeaway from research on social vulnerability is the Forest Service's role in supporting rural economies, in direct employment by the Forest Service and contractors, but also the role of Federal lands in supporting activities like grazing, recreation, and clean water. The Forest Service, with a commitment to serving the American people and increasing diversity of our workforce, is in a unique position to connect with vulnerable populations. Adapting to climate change should account for the diversity of stakeholders by inviting, including, and involving minorities in management decisions and programs. Management should seek to understand the social networks of vulnerable populations to facilitate their climate adaptation and to provide emergency assistance during disasters. Furthermore, given the unique features of WUI and forest-dependent communities, there is a need for more tailored and nuanced vulnerability measures that capture the role the Forest Service plays in local communities.

Notes

[1]Adams, Arapahoe, Broomfield, Clear Creek, City of Denver, Douglas, Elbert, El Paso, Gilpin, Jefferson, Larimer, and Park counties.

[2]https://www.fs.usda.gov/ecosystemservices/FS_Efforts/forests2faucets.shtml

[3]https://choosecolorado.com/key-industries/tourism/ accessed 27 March 2020.

[4]Consumptive-use is water that is evaporated, transpired, incorporated into products or crops, consumed by humans or livestock, or otherwise removed from an immediate water environment (water body, surface- or ground-water source, basin) (Shaffer et al.).

[5]https://cumulis.epa.gov/supercpad/SiteProfiles/index.cfm?fuseaction=second.Healthenv&id=0800257 accessed May 25, 2020.

References

Arapaho and Roosevelt National Forests and Pawnee National Grassland [ARP]. 2013. Flood impact assessment report. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5440234.pdf [Accessed 31 October 2019]

Berwyn, B. 2009. U.S., Colorado tourism hit hard by recession. https://www.summitdaily.com/news/u-s-colorado-tourism-hit-hard-by-recession/

Brown, T.C.; Bergstrom, J.C.; Loomis, J.B. 2007. Defining, valuing and providing ecosystem goods and services. Natural Resources Journal. 47(2): 329–376. https://www.jstor.org/stable/24889176

Brown, T.C.; Foti, R.; Ramirez, J.A. 2013. Projected freshwater withdrawals in the United States under a changing climate. Water Resources Research. 49(3): 1259–1276. https://doi.org/10.1002/wrcr.20076

Brown, T.C.; Froemke, P. 2012. Nationwide assessment of nonpoint source threats to water quality. BioScience. 62(2): 136–146. https://doi.org/10.1525/bio.2012.62.2.7

Brown, T.C.; Hobbins, M.T.; Ramirez, J.A. 2008. Spatial distribution of water supply in the coterminous United States. Journal of the American Water Resources Association. 44(6): 1474–1487. https://doi.org/10.1111/j.1752-1688.2008.00252.x

Chavez, D.J. 2000. Invite, include, and involve! Racial groups, ethnic groups, and leisure. In: Allison, M.T.; Schneider, I.E., eds. Diversity and the recreation profession: Organizational perspectives. State College, PA: Venture Publishing: 179–191.

Colby, S.L.; Ortman, J.M. 2015. Projections of the size and composition of the U.S. population: 2014 to 2060, population estimates and projections. Current Population Reports. P25-1143. U.S. Census Bureau.

Colorado Geological Survey. 2016. Abandoned mine lands. https://coloradogeologicalsurvey.org/hazards/aml/

Daily, G.C. (ed.). 1997. Nature's services. Washington, DC: Island Press. 392 p.

Dawson, C.P.; Cordell, K.; Watson, A.E.; [et al.]. 2016. The U.S. wilderness managers' survey: Charting a path for the future. Journal of Forestry. 114(3): 298–304. https://doi.org/10.5849/jof.15-007

Ehsani, N.; Vörösmarty, C.J.; Fekete, B.M.; [et al.]. 2017. Reservoir operations under climate change: Storage capacity options to mitigate risk. Journal of Hydrology. 555: 435–446. https://doi.org/10.1016/j.jhydrol.2017.09.008

Fassnacht, S.; Venable, N.; McGrath, D.; [et al.]. 2018. Sub-seasonal snowpack trends in the Rocky Mountain National Park Area, Colorado, USA. Water. 10(5): 562. https://doi.org/10.3390/w10050562

Flores, D.; Falco, G.; Roberts, N.S.; [et al.]. 2018. Recreation equity: Is the Forest Service serving its diverse publics? Journal of Forestry. 116(3): 266–272. https://doi.org/10.1093/jofore/fvx016

Fothergill, A.; Maestas, E.G.; Darlington, J.D. 1999. Race, ethnicity, and disasters in the United States: A review of the literature. Disasters. 23(2): 156–173. https://doi.org/10.1111/1467-7717.00111

Foti, R.; Ramirez, J.A.; Brown, T.C. 2012. Vulnerability of U.S. water supply to shortage: A technical document supporting the Forest Service 2010 RPA Assessment. Gen. Tech. Rep. RMRS-GTR-295. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. https://doi.org/10.2737/RMRS-GTR-295

Halofsky, J.E.; Peterson, D.L.; Ho, J.J.; [et al.], eds. 2018. Climate change vulnerability and adaptation in the Intermountain Region [Parts 1 and 2]. Gen. Tech. Rep. RMRS-GTR-375. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 513 p.
Part 1: https://doi.org/10.2737/RMRS-GTR-375PART1
Part 2: https://doi.org/10.2737/RMRS-GTR-375PART2

Heath, L.S.; Smith, J.E.; Woodall, C.W. [et al.]. 2011. Carbon stocks on forestland of the United States, with emphasis on USDA Forest Service ownership. Ecosphere. 2(1): 1–21. https://doi.org/10.1890/ES10-00126.1

Human, K. 2012. A future without hail? Boulder, CO: NOAA Physical Sciences Laboratory. https://www.esrl.noaa.gov/psd/news/2012/010912.html

Jaworski, Delilah; Kline, Jeffrey D.; Miller, Chris; [et al.]. 2018. Evaluating ecosystem services as management outcomes in national forest and grassland planning assessments. Gen. Tech. Rep. PNW-GTR-968. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 38 p. https://doi.org/10.2737/PNW-GTR-968

Knight, R.L.; Wallace, G.N.; Riebsame, W.E. 1995. Ranching the view: Subdivisions versus agriculture. Conservation Biology. 9(2): 459–461. https://www.jstor.org/stable/2386791

Loss, S.R.; Will, T.; Marr, P.P. 2013. The impact of free-ranging domestic cats on wildlife of the United States. Nature Communications. 4(1): 1–8. https://doi.org/10.1038/ncomms2380

Luce, C. 1970. National forest contributions to streamflow: Rocky Mountain Research Station. https://www.fs.usda.gov/research/rmrs/projects/national-forest-contributions-streamflow

Ossa-Moreno, J.; Mcintyre, N.; Ali, S.; [et al.]. 2018. The hydro-economics of mining. Ecological Economics. 145: 368-379. https://doi.org/10.1016/j.ecolecon.2017.11.010

Parsons, J. 1977. Effects of acid mine wastes on aquatic ecosystems. Water, Air, and Soil Pollution. 7(3). https://doi.org/10.1007/bf00284129

Polak, K.; Różkowski, K.; Czaja, P. 2015. Causes and effects of uncontrolled water inrush into a decommissioned mine shaft. Mine Water and the Environment. 35(2): 128–135. https://doi.org/10.1007/s10230-015-0360-6

Reeves, M.C.; Mitchell, J.E. 2012. A synoptic review of U.S. rangelands: A technical document supporting the Forest Service 2010 RPA Assessment. Gen. Tech. Rep. RMRS-GTR-288. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 128 p. https://doi.org/10.2737/RMRS-GTR-288

Rojas-Downing, M.; Nejadhashemi, P.; Harrigan, T.; [et al.]. 2017. Climate change and livestock: Impacts, adaptation, and mitigation. Climate Risk Management. 16: 145–163. DOI: https://doi.org/10.1016/j.crm.2017.02.001

Runkel, R.L.; Kimball, B.A.; Nimick, D. A.; [et al.]. 2016. Effects of flow regime on metal concentrations and the attainment of water quality standards in a remediated stream reach, Butte, Montana. Environmental Science & Technology. 50(23): 12641–12649. https://doi.org/10.1021/acs.est.6b03190

Schmidt, T.S.; Clements, W.H.; Wanty, R.B.; [et al.]. 2012. Geologic processes influence the effects of mining on aquatic ecosystems. Ecological Applications. 22: 870–879. https://doi.org/10.1890/11-0806.1

Shaffer, K.; Runkle, D.L. 2007. Consumptive water: Use coefficients for the Great Lakes Basin and climatically similar areas. Reston, VA: U.S. Geological Survey. https://doi.org/10.3133/sir20075197

Trlica, M.J. 2013. Grass growth and response to grazing. Fact Sheet No. 6.108. Fort Collins, CO: Colorado State University Extension. https://extension.colostate.edu/topic-areas/natural-resources/grass-growth-and-response-to%20grazing-6-108/

Turner, S. 2018. Denver water quality warning in effect because of *E. coli* concerns. https://kdvr.com/news/local/denver-water-quality-warning-in-effect-due-to-e-coli/

U.S. Department of Agriculture, Forest Service [USDA FS]. 2015. Baseline estimates of carbon stocks in forests and harvested wood products for National Forest System units; Rocky Mountain Region. 41 p. Whitepaper. https://www.fs.usda.gov/sites/default/files/rocky-mountain-region-carbon-assessment.pdf

Vasilatos, C.; Koukouzas, N.; Alexopoulos, D. 2015. Geochemical control of acid mine drainage in abandoned mines: The case of Ermioni Mine, Greece. Procedia Earth and Planetary Science. 15: 945–950. https://doi.org/10.1016/j.proeps.2015.08.151

Warziniack, T.; Arabi, M.; Brown, T.C.; [et al.]. 2022. Projections of freshwater use in the United States under climate change. Earth's Future. 10(2): e2021EF002222. https://doi.org/10.1029/2021EF002222

Warziniack, T.; Brown, T.C. 2019. The importance of municipal and agricultural demands in future water shortages in the United States. Environmental Research Letters. 14(8): 084036. https://doi.org/10.1088/1748-9326/ab2b76

Wear D.N.; Prestemon, J.P. 2019. Spatiotemporal downscaling of global population and income scenarios for the United States. PLoS One. 14(7): e0219242. https://doi.org/10.1371/journal.pone.0219242

Whyatt, J.; Varley, F. 2008. July. Catastrophic failures of underground evaporite mines. In: Proceedings of the 27th international conference on ground control in mining. Morgantown, WV: College of Engineering and Mineral Resources, West Virginia University. https://www.cdc.gov/niosh/mining/UserFiles/works/pdfs/cfoue.pdf

Wittmann, E.J.; Mellor, P.S.; Baylis, M. 2001. Using climate data to map the potential distribution of *Culicoides imicola* (Diptera: Ceratopogonidae) in Europe. Revue Scientifique et Technique. 20: 731–740. http://dx.doi.org/10.20506/rst.20.3.1306

Wohl, E. 2006. Human impacts to mountain streams. In: James, L.A.; Marcus, W.A., eds. The human role in changing fluvial systems. Proceedings of the 37th International Binghamton geomorphology symposium. Geomorphology. 79: 217–248. https://doi.org/10.1016/j.geomorph.2006.06.020

Effects of Climate Change on Outdoor Recreation Benefits in the Pike-San Isabel and Arapaho-Roosevelt National Forests

José J. Sánchez, Christopher Huber, John Loomis

Introduction

Millions of Americans enjoy recreating in the outdoors. In 2017, 49 percent of the United States (U.S.) population participated in at least one outdoor activity (Outdoor Foundation 2018). The U.S. Department of Agriculture, Forest Service (2017) estimates there were 148 million visits to national forests during fiscal year 2016. In 2018, there were more than 318.2 million visits to national parks (Ziesler 2019). Across all national park units in Colorado, recreation visits totaled more than 7.6 million in 2018, the largest proportion of which were to Rocky Mountain National Park in northern Colorado— which shares a border with the Arapaho-Roosevelt National Forests—with nearly 4.6 million visitors (Ziesler 2019).

Outdoor recreation provides significant economic contributions to the overall U.S. economy. The latest report from the U.S. Department of Commerce, Bureau of Economic Analysis (BEA) estimates the outdoor recreation economy accounted for 2.2 percent of gross domestic product (GDP) in 2016 (BEA 2019). The BEA (2019) also reports that the outdoor recreation economy grew slightly faster (1.7 percent) than the overall U.S. economy (1.6 percent). Boating and fishing were the outdoor recreation activities with the largest overall economic impact, accounting for $36.9 billion of real gross output, in other words, the industry's sales. The Outdoor Industry Association (2019) reports that the national outdoor recreation economy generates annually $887 billion in consumer spending and supports 7.6 million American jobs. Additionally, it provides $65.3 billion in Federal tax revenue and $59.2 billion in state and local tax revenue.

Colorado's outdoor recreation also provides a substantial contribution to the State economy. The Outdoor Industry Association (2019) reports that 71 percent of Colorado residents participate annually in outdoor recreation. Additionally, they report that outdoor recreation generates $28 billion in consumer spending and provides $2 billion in State and local tax revenue. Outdoor recreation also supports 229,000 jobs in Colorado. The Statewide Comprehensive Outdoor Recreation Plan (2018) uses a broader definition of outdoor recreation to estimate economic output; for example, their definition includes activities such as team sports, golf, and playground activities. The implications of including a wider range of activities will result in larger estimated economic contributions from consumer spending. This report estimated that outdoor recreation contributes $62.5 billion in total economic output (value of all sales of goods and services) and $35 billion in value added (in other words, gross domestic product, or GDP). Furthermore, the Statewide Comprehensive Outdoor Recreation Plan (2018) report finds that outdoor recreation supports over 511,000 jobs in Colorado, when considering a broader definition of recreation activities.

According to the U.S. Census Bureau, Colorado has one of the fastest growing populations among U.S. States, increasing 14.8 percent in population size between 2010 and 2020 (U.S. Census Bureau 2021). In 2020, Colorado's population was more than 5.7 million residents (U.S. Census Bureau 2020); the Colorado State Demography Office predicts the State's population to increase to more than 7.4 million people by 2050 (Colorado State Demography Office 2022; Colorado State Demography Office, n.d.). Most of the State's population (more than 83 percent in 2020) lives along the Front Range, a relatively concentrated urban area situated near the foothills where the Great Plains meet the Rocky Mountains. This urban corridor takes its name from the nearby mountain range called the "Front Range" and spans roughly from Fort Collins (Larimer County) and Greeley (Weld County) in northern Colorado to Pueblo (Pueblo County) in the south. Most of the State's projected population growth is expected to be concentrated along the Front Range urban corridor, with an estimated addition of more than 1.4 million people between 2010 to 2050 (Colorado State Demography Office 2022). With expected population growth, recreation visitation to national forests and other nearby public lands is expected to increase.

Considering outdoor recreation's role of being economically and culturally important to Colorado, the State faces a series of challenges to forest management in the coming decades. Among these challenges are an expected increase in state population, effects of crowding, and the effects from a changing climate on availability of and access to outdoor recreation opportunities. Specifically, climate change is expected to have an impact to the region as temperatures are projected to increase in the future by 2.5 to 5.5 ^{o}F as well as a snowpack decrease in the range of 10 to 40 percent (see Chapter 3 in this report). As a result, it can be expected that outdoor recreation may be affected by climate change via two drivers: first, through a direct effect on recreation demand, and second, through an indirect effect from the quantity and quality of environmental goods and services (Loomis and Crespi 1999; Mendelsohn and Markowski 1999; Shaw and Loomis 2008; White et al. 2016).

The overarching objective of this report is to investigate, from an economic perspective, current outdoor recreation use on the Arapaho-Roosevelt and Pike-San Isabel National Forests, and to consider what future outdoor recreation use may look like under the conditions of changing climate and a growing state population. In the next section of this report, supply of outdoor recreation on the AR and PSI National Forests is discussed. This is subsequently followed by an investigation into current outdoor recreation visitor statistics for these four Forests. Next, we discuss two different concepts of the economic significance of outdoor recreation: first, how communities benefit economically from tourism and visitor expenditures and second, the economic benefits to individual recreators. We pay special attention to how visitors demand the various outdoor recreation uses on these forests and estimate a total economic benefit of current recreation use. Next, we investigate future recreation use into the year 2060 under the scenario of a changing climate. Lastly, findings are summarized, and management implications are documented.

Outdoor Recreation Supply

Outdoor recreation is a prominent use of the Arapaho-Roosevelt and Pike-San Isabel National Forests in Colorado. The Pike and San Isabel National Forests (PSI-NF) are in central Colorado, while the Arapaho-Roosevelt National Forests (AR-NF) are situated in northcentral Colorado (fig. 1). Proximity to these Forests is the expanding metropolitan Front Range, and the variety of types of recreation uses makes these Forests among the most popular outdoor recreation destinations in Colorado. Outdoor recreation on the PSI-NFs and the AR-NFs encompasses a spectrum of uses, including hunting, fishing, mountain biking, sight-seeing, hiking, backpacking, and skiing and snowboarding, among many others.

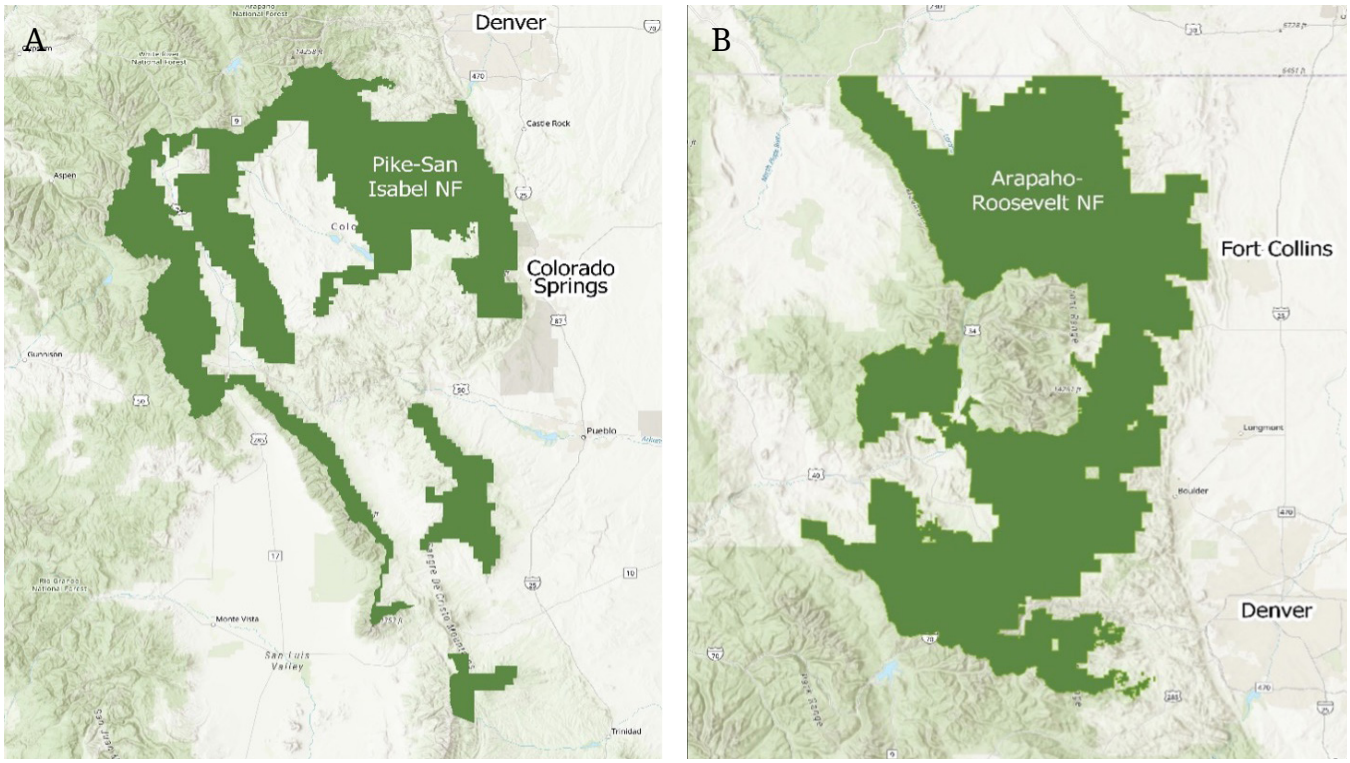


Figure 1a-b—Boundaries for Pike-San Isabel and Arapaho-Roosevelt National Forests.

Eldora (680 acres)[1], Loveland Ski Area (1,800 acres)[2], and Winter Park Resort (3,081)[3] are ski areas situated in the AR-NFs and are among the closest ski areas to Denver and the Front Range urban corridor. (Echo Mountain Resort is another ski area close to Denver within the AR-NFs management boundary but is located on private land.) Ski Cooper (470 acres)[4], near Leadville, Colorado, is in the PSI-NFs and is one of the oldest ski areas in the U.S. and began as Camp Hale, a mountaineering training ground for the U.S. Army's 10th Mountain Division during World War II. In addition to in-bounds skiing, Ski Cooper also operates snowcat tours on 2,600 acres of backcountry skiing in the Pike-San Isabel and White River National Forests. Monarch Mountain (800 acres)[5] near Salida,

Colorado, maintains both in-bounds and guided backcountry skiing via snowcat tours in the PSI-NFs. The AR-NFs and PSI-NFs also provide opportunities for backcountry skiing and snowboarding, cross-country skiing, and snowshoeing in accessible winter locations such as Cameron Pass west of Fort Collins, Berthoud Pass near Winter Park, and Indian Peaks Wilderness west of Boulder.

Both the AR-NFs and the PSI-NFs have unique places for camping, hiking, scenic drives, and trails (USDA FS, n.d.-a). In total, the PSI-NFs maintains 94 developed campgrounds: eight are managed by the Pikes Peak Ranger District; 11 are managed by the San Carlos Ranger District; 16 are managed by the Salida Ranger District; 17 are managed by the Leadville Ranger District; 21 are managed by South Park Ranger District; and 21 are managed by the South Platte Ranger District (USDA FS, n.d.-b). The AR-NFs maintain 54 developed campgrounds: seven of which are managed by the Boulder Ranger District west of Boulder; 24 are managed by the Canyon Lakes Ranger District west of Fort Collins and Loveland; eight are managed by the Clear Creek Ranger District near Idaho Springs; and 15 are managed by the Sulphur Ranger District near Winter Park, Grand Lake, and Granby (USDA FS, n.d.-c).

In terms of the total number of trails available in these Forests, the PSI-NFs and AR-NFs have approximately 411 miles and 478 miles of trails, respectively, for hiking and horseback riding (Furniss and Howe 2016). Well-known trails include the Colorado Trail and the Continental Divide Trail. The Colorado Trail includes a total of 485 trail-miles, crossing six wilderness areas and six national forests from Denver to Durango (TCTF, n.d.). The trail is maintained and administered by both the Forest Service and the Colorado Trail Foundation. This continuous trail is known as one of the premier long-distance trails in the country for hiking, horseback riding, and mountain biking. Leadville Ranger District hosts several sections of the Colorado trail, providing views of beautiful mountains and gorgeous forest landscape (USDA FS n.d.-d). The Continental Divide Trail totals 3,100 miles that cross five U.S. States and is advertised as one of the world's greatest long-distance trails (CDTC n.d.). Nearly a quarter of the trail's total length (740 miles) is in the State of Colorado and crosses both the AR-NFs and PSI-NFs. Recreationists use the trail for fishing access, hiking, horseback riding, hunting, mountain biking, and skiing. Another unique hiking trail is the Devil's Head Lookout Tower in Douglas County, which is the last operational lookout tower on the Front Range and sits on the highest point of the Rampart Range between Colorado Springs and Denver.

In addition to hiking in general, climbing Colorado's "Fourteeners" (mountain peaks over 14,000 feet) is a popular activity in this region. The AR-NFs contains three Fourteeners partially or fully within the forest boundary. PSI-NFs contains an additional 23 of the total 54 Fourteeners in the State of Colorado (USDA FS n.d.-a). Hiking Fourteeners can be challenging but provides unique climbing and viewing experiences. The Colorado Fourteeners Initiative estimates that the closest Fourteeners to Denver collectively see hundreds of thousands of hiker use days per year (CFI n.d.). Among all

Colorado Fourteeners, the Fourteener with the highest amount of hiking use is Mount Bierstadt at 35,000 to 40,000 hiker-days per year; it is located in the PSI-NFs but can be accessed at the Guanella Pass Trailhead in the AR-NFs.

There are other recreation opportunities in the area that include nonmotorized water recreation, such as rafting, kayaking, and recreational fishing. In terms of stream reach lengths, the AR-NFs contains 3,186 miles (or 5,128 km) of streams and the PSI-NFs contains 4,577 miles (or 7,366 km) of streams throughout the respective forests (USGS 2019). Although not all streams can be boated, many can be fished by anglers for recreational fishing purposes. Those that can be boated can represent exciting whitewater experiences. For example, the Arkansas River runs through Browns Canyon National Monument (jointly managed by the Forest Service and Bureau of Land Management) and provides unique boating opportunities including scenic views from the river. Each year more than 175,000 recreationists visit the upper section of Arkansas River for nonmotorized water recreation (one of the most heavily used rivers for whitewater rafting in the U.S.) ranging from scenic float trips to class V rafting (Arkansas River Outfitters Association 2019).

Viewing scenery is also an important recreation use year-round especially during the fall foliage season from September through October. Some well-known places during the autumn include Kenosha Pass near Grant and Jefferson, Colorado (USDA FS, n.d.-e), and the Peak-to-Peak Scenic Byway (55 miles) situated between Estes Park and Black Hawk, Colorado (CDOT, n.d.). Located near Browns Canyon National Monument between Granite and Poncha Springs and Salida is the Collegiate Peaks Scenic Byway (57 miles), which parallels the Sawatch Range containing the highest concentration of Fourteeners in the U.S. People can also drive to the top of Pikes Peak, a Fourteener managed by the PSI-NFs near Colorado Springs. Other notable scenic driving routes in the AR-NFs and PSI-NFs include the Highway of Legends Scenic Byway (110 miles) and the Colorado River Headwaters (80 miles), Lariat Loop (40 miles), and Top of the Rockies (115 miles) National Scenic Byways. Frontier Pathways National Scenic Byway, located on the southern end of the PSI-NFs near the Spanish Peaks and Sangre de Cristo mountain ranges, is also a notable scenic drive (103 miles). The Cache La Poudre (101 miles in length) and Guanella Pass (22 miles) Scenic Byways are other notable scenic driving routes near the Front Range (CDOT, n.d.).

The PSI-NFs fully or partially contains nine wilderness areas. These wilderness areas range in size from the Spanish Peaks Wilderness at 1,937 acres to the Sangre de Cristo Wilderness at 179,304 acres (table 1) (Landres and Meyer 2000; Wilderness Connect 2019). For the PSI-NFs, the combined size of the nine designated wilderness areas totals 746,738 acres. The AR-NFs fully or partially maintains 10 areas designated as wilderness and that range in size from the Byers Peak Wilderness at 8,801 acres to the Indian Peaks Wilderness at 74,973 acres (Landres and Meyer 2000; Wilderness Connect 2019). The total combined size for wilderness areas in the AR-NFs is 370,031 acres. These wilderness areas provide important ecosystem services to society, such as maintaining

critical habitat for wildlife, supplying clean water for downstream users (Meldrum and Huber 2019), and providing unique outdoor recreation opportunities, including hiking, backpacking, and solitude-oriented experiences (Loomis and Richardson 2006; Walsh et al. 1984).

Table 1—Number of acres by Wilderness Area for the Pike-San Isabel and the Arapaho-Roosevelt National Forests.

Pike-San Isabel NF	Acres[a]
Buffalo Peak Wilderness	41,281
Collegiate Peaks Wilderness	165,864
Greenhorn Mountain Wilderness	23,116
Holy Cross Wilderness	122,446
Lost Creek Wilderness	114,864
Mount Evans Wilderness	73,459
Mount Massive Wilderness	24,467
Sangre de Cristo Wilderness	179,304
Spanish Peaks Wilderness	1,937
Total acres:	**746,738**

Arapaho-Roosevelt NF	Acres[a]
Byers Peak Wilderness	8,801
Cache la Poudre Wilderness	9,482
Comanche Peak Wilderness	68,025
Indian Peaks Wilderness	74,973
James Peak Wilderness	17,109
Mount Evans Wilderness	73,459
Neota Wilderness	9,927
Never Summer Wilderness	20,847
Rawah Wilderness	74,408
Vasquez Peak Wilderness	13,000
Total acres:	**370,031**

[a]Landres and Meyer 2000; Wilderness Connect 2019.

Outdoor Recreation Use Statistics - Current and Projected

Outdoor Recreation on the Arapaho-Roosevelt and Pike-San Isabel National Forests

People participate in a wide range of outdoor recreation activities on the AR-NFs and the PSI-NFs. To better understand people's satisfaction and use of these national forests, the Forest Service conducts the National Visitor Use Monitoring (NVUM) survey, which rotates sampling on all forests and grasslands throughout the National Forest System every 5 years.[6] For this analysis, we use data collected in 2015 and 2016 for the AR-NFs and PSI-NFs, respectively. A two-step process is used to arrive at total number of days of participation by recreation activity per year. First, total annual visitation is multiplied by percent main activity participation, both of which are drawn from the NVUM survey, to arrive at annual visits by main activity per year (tables A-1 and A-2 in the Appendix). Main activity participation is defined by NVUM as the primary activity for this recreation visit to the national forest. The second step is to account for length of stay per visit by main activity. Here, conversion factors for Region 2 of the Forest Service (Rocky Mountain Region) in the Rosenberger et al. (2017) outdoor recreation use values report are used to convert "visits" into "days." The conversion factor is defined as "the average number of calendar days per national forest visit" and were imputed by Rosenberger et al. (2017) using NVUM survey results. For each activity, "visitation" is multiplied by the "conversion factor" to arrive at "days" (table A-3 in the Appendix), which are the relevant units for the economic valuation. Days by main activity are summarized for the PSI-NFs and AR-NFs into five climate-sensitive categories of activities (categorized the same as Winter et al. 2021): warm-weather activities, winter activities, wildlife activities, gathering forest products, water-based activities (not including fishing), and other activities (table 2). Each of the five climate-sensitive categories combine recreation activities that are expected to be similarly affected by environmental conditions (Hand and Lawson 2018; Hand et al. 2018).

These calculations show more than 5,377,000 days per year across all outdoor recreation activities for the PSI-NFs. Warm-weather activities collectively have the most visitor days (2,713,000), with viewing natural features having the highest number of days per year (960,000) when looking at the individual activity level (shown graphically in fig. 2). This is followed by hiking/walking (926,000) and developed camping (351,000). Additional warm-weather activities contributing to this category include primitive camping, bicycling, backpacking, other nonmotorized use, picnicking, and horseback riding. The bundled other activities grouping had the next highest amount of total days as a category (1,647,000). Within this climate-sensitive category, driving for pleasure (704,000) had the highest number of activity days, followed by motorized trail activity (403,000) and relaxing (221,000). Wildlife activities collectively have the third most visitor days (661,000), with fishing (552,000) being the most popular activity in this category followed by viewing wildlife (69,000). Downhill skiing (224,000 days) was the most popular activity in the winter activity category.

Table 2—Activity days per year (1,000s) by main activity for the Arapaho-Roosevelt and Pike-San Isabel National Forests.

Activity	Pike-San Isabel NF Days (1,000s)	Arapaho-Roosevelt NF Days (1,000s)
Warm-weather activities		
Hiking/walking	926.9	817.9
Viewing natural features	960.2	818.9
Other nonmotorized	39.1	34.1
Bicycling	154.6	84.0
Developed camping	351.4	719.4
Picnicking	29.4	9.7
Backpacking	53.5	38.6
Horseback riding	9.6	0.0
Primitive camping	189.2	74.0
Total	**2,713.8**	**2,596.5**
Winter activities		
Downhill skiing	224.9	1191.0
Cross-country skiing/snowshoeing	17.1	87.9
Snowmobiling	32.2	42.5
Total	**274.3**	**1,321.4**
Wildlife activities		
Fishing	552.6	228.7
Hunting	38.7	480.9
Viewing wildlife	69.7	29.7
Total	**661.1**	**739.3**
Gathering forest products		
Gathering forest products	27.2	4.9
Total	**27.2**	**4.9**
Water-based activities, not including fishing		
Motorized water activities	0.0	5.2
Non-motorized water	54.3	15.2
Total	**54.3**	**20.4**
Other activities		
Relaxing	221.2	110.9
Driving for pleasure	704.7	398.9
Some other activity	120.1	64.0

Table 2 continued.

Activity	Pike-San Isabel NF Days (1,000s)	Arapaho-Roosevelt NF Days (1,000s)
Off-highway vehicle use	170.0	92.1
Nature center activities	7.1	0.0
Motorized trail activity	403.1	48.9
Nature study	9.0	0.0
Resort use	0.2	0.0
Visiting historic sites	11.6	3.1
Other motorized activity	0.0	0.0
Total	**1,647.0**	**717.8**
Total days (1,000's)	**5377.6**	**5400.4**

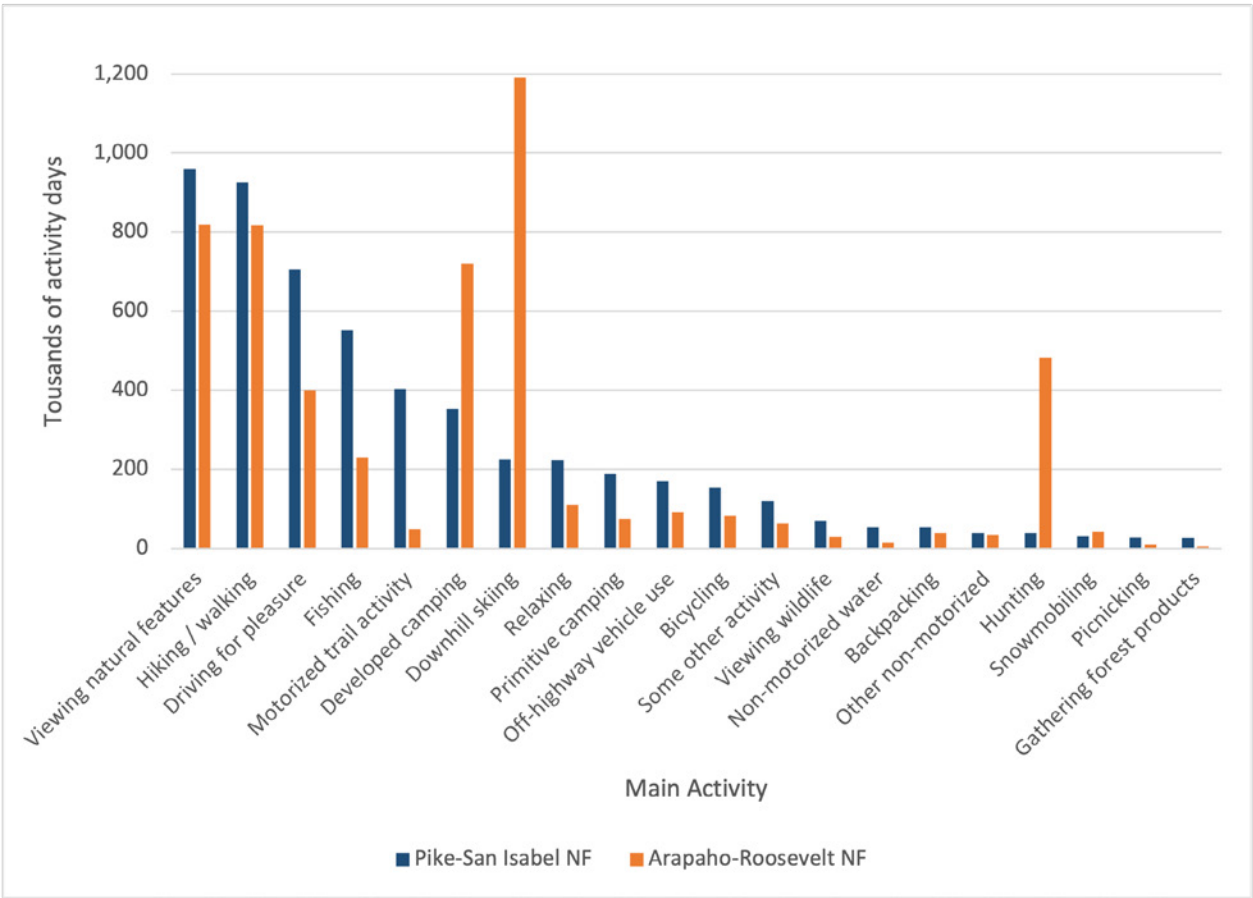


Figure 2—Top 20 activity days per year (1,000s) by main activity for the Arapaho-Roosevelt and Pike-San Isabel National Forests.

The AR-NFs have a similar number of total days per year at 5,400,000 across all outdoor recreation activities (table 2). However, there are noticeable differences in numbers of activity days by main activity when comparing the AR-NFs to the PSI-NFs. The most striking difference is the annual number of days downhill skiing (1,191,000), which has the highest number of activity days across all activities for the AR-NFs (fig. 2). The warm-weather activities grouping has the most activity days (2,596,000), with viewing natural features (818,000) having the largest number of activity days per year, followed by hiking/walking (817,000) and developed camping (719,000). Hunting (480,000) has the highest number of activity days in the wildlife activities category, while driving for pleasure (398,000) is the most popular activity in the other activities category.

As an aside, we can use NVUM reported data on the number of annual visitors to wilderness areas to investigate a rough approximation of intensity of wilderness use. (These data are implicitly captured in table 2 and are imbedded in the total activity counts at the Forest level.) Based on the 2016 NVUM data, there are approximately 297,338 visits to wilderness areas in the PSI-NFs each year, yielding a result of 2.5 wilderness visits per acre per year. For the AR-NFs, there are 271,867 Wilderness visits, based on 2015 NVUM data, yielding a result of 1.4 Wilderness visits per acre per year. Compared to other Wilderness Areas, the PSI-NFs and AR-NFs wilderness areas are highly visited when evaluating use on a per acre basis. For example, in California, which is the most populous State and has the second most wilderness acres in the U.S., the Angeles National Forest has approximately 122,557 acres with 291,649 visits across its five designated wilderness areas. The San Bernardino National Forest has 151,341 acres with 40,334 visits across its eight designated wilderness areas. The wilderness visits per acres are 0.42 and 3.75 for the Angeles and San Bernardino National Forests, respectively. These estimates show the intensity of use to otherwise “pristine” wilderness environments.

The estimated number of activity days by main outdoor recreation activity on the PSI-NFs and the AR-NFs shown in table 2 were then used to investigate the economics of outdoor recreation use and projected changed in activity days in response to a changing climate.

Economics of Outdoor Recreation

There are two common ways to think about the economic significance of outdoor recreation to these national forests. The first—a regional economic analysis—measures how jobs and business activities are supported by expenditures made by national forest visitors to the regional economy (Loomis and Walsh 1997; White 2017). The second—economic valuation—measures the value or benefits to individual recreators (Segerson 2017). The two concepts and approaches—regional economic analyses and economic valuation—measure the economic significance of outdoor recreation from different perspectives. The following subsections discuss these concepts and describe regional

economic effects from visitor spending on jobs and business activities and economic valuation of outdoor recreation to individual recreators.

Outdoor Recreation Expenditures and an Overview of Regional Economic Effects Analyses

There are two subtly different ways to consider how regional economies, employment, and business activities are affected by outdoor recreation and tourism dollars. The first is an economic contributions analysis, which measures the gross economic activity of the regional economy from total visitor expenditures, including both local and nonlocal visitor expenditures (Cullinane Thomas et al. 2019). In a contributions analysis, results can be interpreted as the relative importance of all forest visitor spending to the regional economy. This is contrasted with the second type of regional economic analysis called an economic impact analysis, which focuses on net changes to the regional economy from "new" money brought in exclusively from nonlocal visitor expenditures (Cullinane Thomas et al. 2019). For this report, we investigate the expenditures made by both types of forest visitors but do not estimate the economic effects to the broader regional economy. In particular, expenditures are separated by both local and nonlocal visitors to gain a foundational understanding of how local economies may be affected by local and nonlocal tourism dollars.

To begin to understand how tourism affects local economic activities in and near these Forests, the local economic region must be defined, and the amount of money spent by local and nonlocal forest visitors within the defined regional economy must be quantified. It is important to distinguish between local and nonlocal forest visitors because their trip behaviors, characteristics, and expenditures can be different. For this report, the local region is defined as a 50-mile boundary around each forest (PSI-NFs and AR-NFs, respectively). This 50-mile boundary is a standard approach used by the Federal agencies to define which economies and their associated activities can be counted as part of the local economic region near the forest (White et al. 2016). Using this definition, people who live inside the 50-mile boundary around the forest are considered locals and their associated spending to make an outdoor recreation trip to the forest would be classified as local spending. In contrast, people who live outside the 50-mile buffer are considered nonlocals to the forest. Their associated spending is bringing new money into the defined local region and is generating new economic activity (i.e., jobs and business activities). Total expenditures within 50 miles of each forest are estimated for local and nonlocal visitation using the procedures outlined in White (2017). For the PSI-NFs, total annual spending is considerably higher among those traveling from outside a 50-mile radius, than for locally originating visitation (table 3). Lodging expenditures account for 25.2 percent of expenditures among nonlocal visitation, followed by gas and oil at 20.2 percent, and then by restaurants at 18.8 percent. Gas and oil is the primary spending category for local visitors, followed by groceries at 24.7 percent. The average total spending per party per trip is $255 for nonlocals and $131 for locals.

Table 3—Total annual expenditures within 50 miles of the survey site by nonlocal and local visitation to the Pike-San Isabel National Forest, by spending category.

Spending category	Nonlocal spending[a,b]		Local spending	
	Total annual expenditures Thousands of dollars ($2018)[c]	Spending for each category Percent	Total annual expenditures Thousands of dollars ($2018)[c]	Spending for each category Percent
Lodging	49,227	25.2	2,712	4.3
Camping fees	5,596	2.9	3,698	5.8
Restaurant	36,703	18.8	7,757	12.2
Groceries	27,577	14.1	15,639	24.7
Gas and oil	39,603	20.2	21,707	34.3
Local transportation	1,379	0.7	203	0.3
Entry fees	9,103	4.7	3,726	5.9
Recreation and entertainment	12,153	6.2	1,647	2.6
Sporting goods	7,026	3.6	5,136	8.1
Souvenirs and other expenses	7,292	3.7	1,141	1.8
Total	**195,659**	**100**	**63,366**	**100**

Spending totals were calculated using the procedures outlined in White (2017) and parameters and spending averages used by the USDA Forest Service office of Ecosystem Management Coordination.
[a]Nonlocal refers to trips that required traveling more than 50 miles to the survey site.
[b]National Visitor Use Monitoring (NVUM) data source is the fourth round of the NVUM surveys.
[c]NVUM data, 2016.

For the AR-NFs, the total annual expenditure is similar for those traveling from outside a 50-mile radius and for locally originating visitation (table 4). Gas and oil expenditures account for 23.8 percent of expenditures among nonlocal visitation, followed by restaurant at 17.7 percent, and then by lodging expenses at 15.5 percent. For the PSI-NFs, gas and oil is also the primary spending category for local visitation, followed by groceries at 23.4 percent. The total spending (per party/trip) is $285 for nonlocals and $119 for locals.

Table 4—Total annual expenditures within 50 miles of the survey site by nonlocal and local visitation to the Arapaho-Roosevelt National Forests, by spending category.

Spending category	Nonlocal spending[a,b]		Local spending	
	Total annual expenditures Thousands of dollars ($2018)[c]	Spending for each category Percent	Total annual expenditures Thousands of dollars ($2018)[c]	Spending for each category Percent
Lodging	13,717	15.5	1,795	2.1
Camping fees	2,597	2.9	5,111	6.0
Restaurant	15,672	17.7	9,468	11.1
Groceries	12,464	14.1	19,855	23.4
Gas and oil	21,063	23.8	26,769	31.5
Local transportation	188	0.2	91	0.1
Entry fees	9,989	11.3	9,801	11.5
Recreation and entertainment	7,080	8.0	4,494	5.3
Sporting goods	3,732	4.2	6,621	7.8
Souvenirs and other expenses	1,950	2.2	915	1.1
Total	**88,452**	**100**	**84,920**	**100**

Spending totals were calculated using the procedures outlined in White (2017) and parameters and spending averages used by the USDA Forest Service office of Ecosystem Management Coordination.
[a]Nonlocal refers to trips that required traveling more than 50 miles to the survey site.
[b]National Visitor Use Monitoring (NVUM) data source is the fourth round of the NVUM surveys.
[c]NVUM data, 2015.

Economic Valuation of Outdoor Recreation

Understanding the economic benefits of outdoor recreation, as opposed to the regional economic effects from visitor spending, provides a useful framework for evaluating tradeoffs because: (1) maximizing net economic benefits can be an important objective when facing limited resources and competing uses; (2) it provides a way to weigh the outcome competing projects or objectives in commensurate units (dollars); and (3) it provides a measure of the net benefits across all recreators (Young and Loomis 2014). Since there are no market-driven entrance fees required to participate in outdoor recreation on the PSI-NFs and the AR-NFs, meaning there are little to no entrance fees to access the National Forest, nonmarket valuation techniques must instead be relied upon to quantify the economic value of outdoor recreation activities (Loomis and Walsh 1997). Nonmarket valuation methods can be generally grouped into two broad categories: revealed preference methods and stated preference methods (Champ et al. 2017), both of which have been widely used as standard approaches to valuing outdoor recreation (Loomis and Walsh 1997). The basic approach to revealed preference methods includes estimating value from observed behaviors in related markets. These

techniques infer value from observed data, such as number of trips or site visits, and travel related expenditures incurred during the visit, which are then used in estimating a travel cost model. The number of trips a visitor takes over a period of time and their travel related expenditures allow estimation of the overall demand function to visit a given site. From this demand function the consumer surplus or the amount the visitor would pay in excess of their travel costs (i.e., their net willingness to pay—WTP) is obtained. In contrast, the basic approach to the second broad grouping of nonmarket valuation techniques, stated preference methods, is to ask people via a survey or interview to state their value based on a hypothetical change to a good or service (Johnston et al. 2017). Stated preference techniques include the contingent valuation method (Boyle 2017) and choice models (conjoint or choice experiments; see Holmes et al. [2017] for background on these types of approaches), both of which are flexible enough to accommodate use values (e.g., outdoor recreation) as well as nonuse (or passive use) values (e.g., existence value).

When circumstances preclude conducting an original nonmarket valuation study, the benefit transfer method uses existing nonmarket valuation estimates from previously conducted revealed and stated preference studies and applies them new context (Boyle and Bergstrom 1992; Johnston et al. 2015; Richardson et al. 2015; Rosenberger and Loomis 2017). For example, a benefit for hiking/walking at one site might be used to estimate benefits at another site. There are four different types of benefit transfer: (1) a unit value transfer using a single estimate from the recreation site most similar to the site we are trying to value (the value estimated at Site A applied to Site B); (2) an average value transfer of several existing estimates drawn from the literature (the average value estimated at all sites in a region applied to Site B; (3) a function transfer that applies an estimated site demand or willingness to pay function from a past study to the new unstudied site or context; and (4) using a meta-regression demand or valuation equation estimated from a pool of past valuation studies at several sites (see Johnston et al. [2015] and Rosenberger and Loomis [2017] for comprehensive discussions of benefit transfer methods). The ideal situation for benefit transfer would be to use an existing value estimate drawn from a study that was conducted in the same location (e.g., AR-NFs) for the same type of good or service (e.g., deer hunting) being examined in this new policy context. Rosenberger et al. (2017) offer that the error when conducting a benefit transfer is likely to be smallest when the original study site and new policy site are similar. One measure of reliability between different benefit transfer approaches is the absolute percent transfer error (PTE), which compares the degree of difference between the transferred value estimate with a known policy value estimate (Rosenberger 2015). Rosenberger (2015) and Rosenberger and Loomis (2017) cataloged estimates of absolute PTE from value (unit or average) transfers and function transfers. They found that value transfers had mean absolute PTE of 140 percent (median = 45 percent), while benefit function transfers (including meta-analyses) performed better

with a mean absolute PTE of 65 percent (median = 36 percent) (Rosenberger, 2015), suggesting that function transfers offer an improvement over unit or average value transfers.

For this analysis, we conduct a benefit transfer by relying on the per person, per day average value estimates for Region 2 of the National Forest System generated by Rosenberger et al. (2017). Their predicted average values are based on a meta-analysis of pooled economic valuation studies on outdoor recreation across the United States and Canada over the period 1958 to 2015 (see the Recreation Use Values Database [RUVD 2016] for a complete list of studies used in their meta-analysis). Several of the Rosenberger et al. (2017) primary activity groupings are aggregated into two or more NVUM activities. For example, NVUM includes a "primitive camping" category and a "backpacking" category, both of which are aggregated into the "backpacking" primary activity in the Rosenberger et al. (2017) meta-analysis (see table 1 on page 5 of their report). We follow their categorization scheme provided to apply a value to these aggregated activity types. Table 5 summarizes the estimated per person, per day economic benefits (or WTP) by activity type.

Table 5—Estimates per person, per day and total economic benefits by activity type for the Pike -San Isabel and Arapaho-Roosevelt National Forests (adapted from Rosenberger et al. 2017; and based on visitation data collected in 2016 for PSI-NF and 2015 for AR-NF).

Activity	Average value estimate (2016, $ per person, per day)	Total annual value estimate ($1,000s; 2016)	
		Pike-San Isabel NFs	Arapaho-Roosevelt NFs
Warm-weather activities			
Hiking/walking	84.12	77,972	68,803
Viewing natural features	59.79	57,408	48,964
Other nonmotorized	64.67	2,528	2,203
Bicycling	86.40	13,357	7,254
Developed camping	35.28	12,396	25,380
Picnicking	48.84	1,435	475
Backpacking	32.81	1,755	1,267
Horseback riding	64.67	620	—
Primitive camping	32.81	6,208	2,427
Total	—	**173,679**	**156,772**
Winter activities			
Downhill skiing	81.89	18,417	97,532
Cross-country skiing/Snowshoeing	56.18	962	4,941

Table 5 continued.

Activity	Average value estimate (2016, $ per person, per day)	Total annual value estimate ($1,000s; 2016)	
		Pike-San Isabel NFs	Arapaho-Roosevelt NFs
Snowmobiling	50.11	1,616	2,128
Total	—	**20,995**	**104,601**
Wildlife activities			
Fishing	71.18	39,336	16,280
Hunting	77.08	2,985	37,070
Viewing wildlife	59.79	4,169	1,774
Total	—	**46,490**	**55,124**
Gathering forest products			
Gathering forest products	64.67	1,759	316
Total	—	**1,759**	**316**
Water-based activities, not including fishing			
Motorized water activities	58.04	—	304
Non-motorized water	108.59	5,894	1,651
Total	—	**5,894**	**1,955**
Other activities			
Relaxing	64.67	14,305	7,170
Driving for pleasure	64.67	45,576	25,798
Some other activity	64.67	7,768	4,136
Off-highway vehicle use	50.11	8,520	4,613
Nature center activities	59.79	422	—
Motorized trail activity	50.11	20,200	2,448
Nature study	59.79	535	—
Resort use	64.67	10	—
Visiting historic sites	59.79	696	186
Other motorized activity	50.11	—	—
Total	—	**98,033**	**44,352**
Total for all activities	—	**346,850**	**363,121**

— indicates no data

The per person, per day economic WTP estimates are then multiplied by the number of days (table 2) to arrive at the total annual value by activity for the PSI-NFs and the AR-NFs (table 5). Results for the PSI-NFs show the total estimated economic benefit across all recreation activities to be more than $346 million per year. The results for the AR-NFs show a similar magnitude in total estimated benefits at more than $363 million per year. When comparing among the climate-sensitive groupings (warm-weather activities, winter activities, wildlife activities, gathering forest products, water-based activities [not including fishing], and other activities), it becomes clear that warm-weather activities (e.g., bicycling, hiking/walking, picnicking) are the most-highly valued type of recreation use per year for both Forests ($173 million for the PSI-NFs and $156 million for the AR-NFs). Winter activities (e.g., skiing, snowmobiling) are the next most valued recreation use for the AR-NFs at $104 million per year and are primarily driven by the estimated value of downhill skiing ($97 million per year). For the PSI-NFs, the next most valuable grouping is other activities ($98 million per year), which includes activities such as off-highway vehicle use, motorized trail activity, and driving for pleasure.

The main limitation of our results arises from fact that the underlying meta-analysis does not provide confidence intervals or other measures of uncertainty for the per person, per activity day values. This means our aggregated total economic benefits shown in table 5 suggest a certain level of precision not acknowledged in the results. Furthermore, some of the studies in the meta-analysis do not consider substitution effects, which can lead to overestimation of values.

Climate Change Vulnerability Assessment: Relationships Between Climate Change and Outdoor Recreation

Climate change is expected to influence the region's temperature as climate models forecast an increase of 2.5 to 5.5 °F as well as a decrease in snowpack of 10 to 40 percent (see Chapter 3 in this report). Outdoor recreation may be affected by climate change via two drivers: first, through a direct effect on recreation demand, and second, through an indirect effect from the quantity and quality of environmental goods and services (Loomis and Crespi 1999; Mendelsohn and Markowski 1999; Shaw and Loomis 2008). In a study focusing on national parks, Fisichelli et al. (2015) examined the relationship between climate and future park visitation by evaluating historical monthly mean air temperatures and historical National Park Service visitation from 340 parks since 1979. They found visitation increased with increasing average monthly temperature but decreased with temperature greater than 25 degrees Celsius. As a result of higher temperatures under alternative climate scenarios, they predict future visitation to generally increase across the park system. Their findings indicate increased temperatures could increase visitation during the nonpeak shoulder seasons, as well as other public lands that are in either higher-latitude and/or high-altitude environments, such as the AR and PSI National Forests. Similarly, Loomis and Richardson (2006) investigated changes in Rocky Mountain National Park visitation behavior from climate

change. The authors conclude that climate change is expected to benefit traditional recreationists, such as hikers and campers, due to warmer temperature and a longer season, while winter recreation uses would decline in visitation. However, there have been some climactic changes in the study area since 2006, such as the Cameron Peak and East Troublesome Fires, that have resulted in extended trail closures (e.g., Roaring Creek Trail; USDA FS, n.d.-g). Furthermore, for the same study area, Richardson and Loomis (2004) found that there is a decrease in visitation under the extreme heat climate conditions. Askew and Bowker (2018) found that recreation will respond differently to climate change, depending on location and activity. For example, in the Rocky Mountain Region, climate change is expected to negatively impact fishing, hunting, and motorized water activities, while having a positive impact on swimming. Scott et al. (2008) investigated climate change implications for snowmobiling in the U.S. northeast region and found that it was more vulnerable than skiing due to greater dependence on natural snowfall. Similarly, White et al. (2016) reports that on Federal lands, snowmobiling and undeveloped skiing are the outdoor recreation activities likely to have the greatest decline in participation under climate change.

We are interested in the potential impacts to outdoor recreation use on the AR-NFs and the PSI-NFs from climate change, and to quantify these projected changes, we rely on forecasts published in Chapter 12 of the "Update to the Forest Service 2010 Resources Planning Act Assessment" (USDA FS 2016). This referenced report provides modeled regional recreation participation projections at the four Resources Planning Act (RPA) regions, including the Rocky Mountain RPA that contains the AR-NFs and PSI-NFs. The report provides percent change in number of recreation days by activity projected to the year 2060 under two scenarios: with and without climate change (table 6). The analysis uses external projections of demographics, economic conditions, land use, and climate factors to estimate future per capita participation and average annual days per participant on national forest units under climate change. The participation rate and average days per participant are multiplied by the RPA population projections to calculate total participants and total annual days for the Rocky Mountain RPA region. To incorporate uncertainty about future recreation use, the RPA assessment developed multiple scenarios based on future economic, social, economic, and environmental conditions (see Bowker et al. 2012 and USDA FS 2016 for additional details). There are several limitations to these forecasts, which leads to population growth being a driving factor in forecasted results changes (Bowker et al. 2012). Emerging activities brought about by new technologies and changing tastes and preferences may not be accurately reflected in the models and, to the extent that data were collected during periods of participation growth, may overestimate future use. Bowker et al. (2012) state that another drawback to their analysis is the challenge in accounting for future congestion, limited supply, and relative price changes on growth in participation.

Table 6—Percent change in recreation days by recreation activity for the Rocky Mountain Region projected to 2060 (adapted from table 12-20 in USDA FS 2016).

Activity	Outdoor recreation activity group (USDA FS 2016)	2060 average change in total participants (%)	
		Without climate change	With climate change
Warm-weather activities			
Hiking/walking	Day hiking	98	110
Viewing natural features	Nature viewing	79	78
Other nonmotorized	Nature viewing	79	78
Bicycling	Challenge activities	97	101
Developed camping	Visiting developed sites	92	86
Picnicking	Visiting developed sites	92	86
Backpacking	Primitive area use	60	48
Horseback riding	Horseback riding	69	54
Primitive camping	Primitive area use	60	48
Winter activities			
Downhill skiing	Developed skiing	162	161
Cross-country skiing	Undeveloped skiing	80	53
Snowmobiling	Motorized snow activities	11	3
Wildlife activities			
Fishing	Fishing	62	35
Hunting	Hunting	22	5
Viewing wildlife	Nature viewing	79	78
Gathering forest products			
Gathering forest products	Nature viewing	79	78
Water-based activities, not including fishing			
Motorized water activities	Motorized water activities	109	86
Nonmotorized water	Floating	57	54
Other activities			
Relaxing	Visiting developed sites	92	86
Driving for pleasure	Nature viewing	79	78
Some other activity	Nature viewing	79	78
Off-highway vehicle use	Motorized off road	68	65

Table 6 continued.

Activity	Outdoor recreation activity group (USDA FS 2016)	2060 average change in total participants (%)	
		Without climate change	With climate change
Nature center activities	Visiting interpretative sites	116	113
Motorized trail activity	Motorized off road	68	65
Nature study	Nature viewing	79	78
Resort use	Visiting developed sites	92	86
Visiting historic sites	Visiting interpretative sites	116	113
Other motorized activity	Motorized off road	68	65

For our analysis, we multiply the average percent change in number of days across all RPA scenarios for the Rocky Mountain RPA (table 12-20, USDA FS 2016) with activity days per year (table 2); this yields average annual projected number of days with and without climate change in 2060 for both the PSI-NFs (table 7) and AR-NFs (table 8). Taking the difference between the two climate change scenarios provides a net change in the number of days by activity resulting from climate change for each national forest. The net change in number of days is summed by climate-sensitive groupings (warm-weather activities, winter activities, wildlife activities, gathering forest products, water-based activities [not including fishing], and other activities) to investigate the general effects from climate change impacts on similar outdoor recreation activity types. In the next subsections, we discuss the potential climate impacts to each climate-sensitive grouping. The other activities category (e.g., off-highway vehicle use, motorized trail activity, driving for pleasure) is omitted from further discussion because, although climate change may impact these uses implicitly, other important factors such as infrastructure and interpretive materials are likely to be more important determinates in their projected recreation uses.

Table 7—Projected and net changes in recreation days at the Pike-San Isabel National Forests with and without climate change in 2060.

Activity	Projected days in 2060 (1,000s)		Net effects of climate change
	without climate change	with climate change	
Warm-weather activities			
Hiking/walking	1,835.3	1,946.5	111.2
Viewing natural features	1,718.7	1,709.1	-9.6
Other nonmotorized	70.0	69.6	-0.4
Bicycling	304.5	310.7	6.2
Developed camping	674.6	653.5	-21.1

Table 7 continued.

Activity	Projected days in 2060 (1,000s)		Net effects of climate change
	without climate change	with climate change	
Picnicking	56.4	54.7	-1.8
Backpacking	85.6	79.2	-6.4
Horseback riding	16.2	14.8	-1.4
Primitive camping	302.7	280.0	-22.7
Total	**5,064.1**	**5,118.1**	**54.0**
Winter activities			
Downhill skiing	589.2	587.0	-2.2
Cross-country skiing/ snowshoeing	30.8	26.2	-4.6
Snowmobiling	35.8	33.2	-2.6
Total	**655.9**	**646.4**	**-9.5**
Wildlife activities			
Fishing	895.3	746.0	-149.2
Hunting	47.2	40.7	-6.6
Viewing wildlife	124.8	124.1	-0.7
Total	**1067.3**	**910.8**	**-156.5**
Gathering forest products			
Gathering forest products	48.7	48.4	-0.3
Total	**48.7**	**48.4**	**-0.3**
Water-based activities, not including fishing			
Motorized water activities	—	—	—
Nonmotorized water activities	85.2	83.6	-1.6
Total	**85.2**	**83.6**	**-1.6**
Other activities			
Relaxing	424.7	411.4	-13.3
Driving for pleasure	1,261.5	1254.4	-7.0
Some other activity	215.0	213.8	-1.2
Off-highway vehicle use	285.7	280.6	-5.1
Nature center activities	15.2	15.0	-0.2
Motorized trail activity	677.2	665.1	-12.1
Nature study	16.0	15.9	-0.1

Table 7 continued.

Activity	Projected days in 2060 (1,000s)		Net effects of climate change
	without climate change	with climate change	
Resort use	0.3	0.3	0.0
Visiting historic sites	25.2	24.8	-0.3
Other motorized activity	—	—	—
Total	**2,920.8**	**2,881.4**	**-39.4**
Total days (1,000s)	**9,842.0**	**9,688.8**	**-153.2**

— Indicates no data

Table 8—Projected and net changes in recreation days at the Arapaho-Roosevelt National Forests with and without climate change in 2060.

Activity	Projected days in 2060 (1,000s)		Net effects of climate change
	without climate change	with climate change	
Warm-weather activities			
Hiking/walking	1,619.5	1,717.6	98.1
Viewing natural features	1,465.9	1,457.7	-8.2
Other nonmotorized	61.0	60.6	-0.3
Bicycling	165.4	168.8	3.4
Developed camping	1,381.2	1,338.0	-43.2
Picnicking	18.7	18.1	-0.6
Backpacking	61.8	57.1	-4.6
Horseback riding	—	—	—
Primitive camping	118.4	109.5	-8.9
Total	**4,891.7**	**4,927.5**	**35.7**
Winter activities			
Downhill skiing	3,120.5	3,108.5	-11.9
Cross-country skiing/snowshoeing	158.3	134.6	-23.7
Snowmobiling	47.1	43.7	-3.4
Total	**3,325.9**	**3,286.9**	**-39.1**

Table 8 continued.

Activity	Projected days in 2060 (1,000s)		Net effects of climate change
	without climate change	with climate change	
Wildlife activities			
Fishing	370.5	308.8	-61.8
Hunting	586.7	505.0	-81.8
Viewing wildlife	53.1	52.8	-0.3
Total	**1,010.4**	**866.6**	**-143.8**
Gathering forest products			
Gathering forest products	8.8	8.7	-0.05
Total	**8.8**	**8.7**	**-0.05**
Water-based activities, not including fishing			
Motorized water activities	10.9	9.7	-1.2
Nonmotorized water	23.9	23.4	-0.5
Total	**34.8**	**33.2**	**-1.7**
Other activities			
Relaxing	212.9	206.2	-6.7
Driving for pleasure	714.1	710.1	-4.0
Some other activity	114.5	113.8	-0.6
Off-highway vehicle use	154.6	151.9	-2.8
Nature center activities	—	—	—
Motorized trail activity	82.1	80.6	-1.5
Nature study	—	—	—
Resort use	—	—	—
Visiting historic sites	6.7	6.6	-0.1
Other motorized activity	—	—	—
Total	**1,284.9**	**1,269.3**	**-15.6**
Total days (1,000s)	**10,556.5**	**10,392.0**	**-164.4**

— Indicates no data

Warm Weather Activities

Warm-weather activities are currently the most common recreation use in PSI-NFs (50 percent of total days) and AR-NFs (48 percent of total days). Within the warm-weather activities group for the PSI-NFs, viewing natural features and hiking/walking are the most popular recreation activities, 17.9 percent and 17.2 percent of total days, respectively. For the AR-NFs, viewing natural features and hiking receive the most days per year at 15.1 percent and 15.2 percent, respectively, followed by developed camping at 13.3 percent. Using the projected changes from the U.S. Forest Service 2010 RPA Assessment (USDA FS 2016; table 6), total recreation days are expected to increase by a combined net effect of 54,000 days for the PSI-NFs and 35,700 for the AR-NFs across all warm-weather recreation activities by 2060 as a result of climate change (tables 7–8).

When considering the specific recreation activities in the warm-weather group, it becomes clear that the impacts from climate change are expected to disproportionally affect specific recreation activities. Specifically, one of the most popular recreation activities, hiking, will see net increases in the number of visitation days under climate change in both the PSI and AR NFs (tables 7-8). Under the no climate change scenario, the number of hiking days are also expected to increase and are predicted to nearly double (an increase of 98 percent) by 2060 (table 6). Comparatively, hiking days are expected to more than double (an increase of 110 percent) in the climate change scenario by 2060, yielding a net increase of 111,000 hiking days for the PSI-NFs and a net increase of 98,000 hiking days for the AR-NFs attributable to climate change. A net increase is also expected for days bicycling attributable to climate change for the PSI-NFs (6,200 days) and the AR-NFs (3,400 days). In contrast, for the remaining warm-weather recreation activities, the models predicted a slight net decrease in warm-weather activity days on both national forests as a result of climate change. Among warm-weather activities, the largest net decrease in warm-weather activity days resulting from climate change are predicted to be primitive camping (22,700 days for the PSI-NFs and 8,900 days for the AR-NFs) and developed camping (21,100 days for the PSI-NFs and 43,200 for the AR-NFs).

Keep in mind that these are net effects between the with and without climate change scenarios, and the gross number of days for all warm-weather activities are expected to increase by at least 48 percent (primitive camping forecasts from table 6) relative to the current recreation use estimates from 2015 and 2016 levels. An explanation for these rates of growth under each climate scenario is largely due to population growth (USDA FS 2016). Although higher minimum temperatures required for warm-weather activities are expected to increase use with climate change, higher maximum levels may dampen or shift demand to different locations or activities due to extreme heat events (Bowker et al. 2012; Richardson and Loomis 2004).

Winter Activities

Winter activities account for 24.5 percent of total activity days per year for the AR-NFs. Downhill skiing accounts 22.1 percent of total activity days, cross-country skiing comprise of 1.6 percent of total days, and snowmobiling comprises 0.8 percent of total days. For the PSI-NFs, the proportion of days in the winter activities grouping is comparatively lower at 5.1 percent of total forest days and is largely concentrated in downhill skiing (4.2 percent of total days). Generally, climate change is expected to have a negative effect on winter activities as warmer seasonal temperatures will cause relatively shorter winter seasons, with more precipitation as rain rather than snow, and a higher snow line in the mountains. Specifically, past research predicts that downhill skiing, snowboarding, cross-country skiing, and snowmobiling are each expected to decrease in number of days available for these winter activities (Wobus et al. 2017).

Using projected changes from the U.S. Forest Service 2010 RPA Assessment (USDA FS 2016) (table 6), winter activity recreation days are expected to decrease by 9,500 days on the PSI-NFs and 39,100 days on the AR-NFs in the climate change scenario (tables 7–8). The difference between each forest's predicted net change to winter activities is because the AR-NFs' current total recreation days is disproportionally higher for this climate-sensitive category (24.5 percent) compared to the PSI-NFs' (5.1 percent). Cross-country skiing (which includes snowshoeing) will have the largest net decrease in days (4,600 fewer days for the PSI-NFs and 23,700 fewer days for the AR-NFs) between the two climate change scenarios. Downhill skiing for each forest is expected to experience a net decrease in visitation days of 2,200 for the PSI-NFs and 11,900 for the AR-NFs as result of climate change. Wobus et al. (2017) argue that downhill skiing and snowboarding may be more resilient to climate change because resorts could rely on artificial snowmaking whereas cross-country skiing, snowshoeing, and snowmobiling must rely on natural snowfall. Despite the expected net decrease in number of days for winter activities, the climate change scenario predicts positive gross increases in the recreation days in 2060 relative to current levels. Specifically, downhill skilling is expected to increase by 161 percent, cross-country skiing by 53 percent, and snowmobiling by 3 percent for each Forest relative to the current number of days (table 6). However, because of shorter seasons, the number of days within a single year when winter activities are possible will decrease and winter activities may see an increase in congestion and crowding, thus reducing benefits to recreationists. For example, Fonner and Berrens (2014) found that U.S. downhill ski areas participants prefer some level of crowding, but higher levels of crowding become a negative experience for skiers. To reduce crowding for downhill skiing, Walsh et al. (1983) found that Colorado skiers would be willing to pay a premium for lift tickets with shorter lift line waiting times. A more recent study from Norway shows similar results; skiers are willing to pay more than the standard lift ticket for reduce wait times (Lien et al. 2022).

Wildlife Activities

Wildlife activities are distinct from other types of recreation as these activities depend on the distribution, abundance, and health of desired target species (Hand et al. 2018). Wildlife activities considered in this analysis include consumptive (hunting and fishing) and nonconsumptive (wildlife viewing) activities. Both the PSI-NFs and AR-NFs have the similar percentage of total days (12.3 and 13.7 percent) in the wildlife activities category. The most popular activities in this category are fishing for PSI-NFs (10.3 percent of total days) and hunting for AR-NFs (8.9 percent of total days). Using projected changes from the U.S. Forest Service 2010 RPA Assessment (USDA FS 2016) (table 6), wildlife activity recreation participation between the with and without climate change scenarios is expected to decline by a net 156,500 days for the PSI-NFs and 143,800 days for the AR-NFs (tables 7-8). For the PSI-NFs, most of the decline in wildlife activity days is concentrated in fishing which is predicted to have a net decline of 149,200 days by 2060. For the AR-NFs, hunting is expected to decrease by a net 81,800 days and fishing decrease by a net of 61,800 days. Factors for the expected decline in fishing are higher summer and annual temperatures and changes in water availability for fish habitat. For hunting, this decline is due to increases in human population density and a reduction in per capita access. Warmer and drier conditions affecting habitat are contributing factors to the net decrease in participation in hunting from climate change (USDA FS 2016). Viewing wildlife is also expected to experience a decrease in the number of days for each Forest but not as dramatically as fishing and hunting.

Forest Product Gathering

Forest product gathering consists of Christmas tree harvesting, mushroom gathering, wood collection for personal use, as well as other nontimber products. This broad category is among the least popular activities as <1 percent of PSI-NFs days fall into this activity category and <1 percent for AR-NFs. The U.S. Forest Service 2010 RPA Assessment (USDA FS 2016) indicates there will be a slight decrease in visits for forest product gathering with climate change (tables 7–8).

Water-Based Activities (Not Including Fishing)

For the water-based category, the recreation activities include nonmotorized water activities and motorized water activities. Nonmotorized water activities are 1.0 percent of total PSI-NFs visits and <1 percent of total AR-NFs visits. Motorized water activities account for <1 percent of total recreation days for the PSI-NFs and <1 percent in AR-NFs. These relatively low activity days might be due to high cost of rafting and motorized boating. Furthermore, streams, rivers, lakes, and reservoirs provide boating opportunities, but they may be commonly paired with fishing (Hand et al. 2018) or could be considered a relatively scarce resource compared to the high availability of other outdoor recreation opportunities on the Forest such as the number of trails for hiking or backpacking. The U.S. Forest Service 2010 RPA Assessment (USDA FS 2016) shows climate change will have a negative effect on activity days for water-based activities, which are predicted to experience a net decrease of 1,600 days for the PSI-

NFs and 1,700 for the AR-NFs by 2060 (tables 7–8). The timing of spring runoff and overall availability of water in rivers, lakes, and reservoirs may be key drivers for these changes.

Adapting Recreation to the Effects of Climate Change

Based on these modeled results, climate change is expected to affect future recreation participation on the PSI-NFs and AR-NFs. However, these forecasts do not account for emerging outdoor recreation activities brought about by new technologies and changing tastes and preferences (Bowker et al. 2012). Future congestion and limited supply place constraints on the amount of recreation use that Forest Service sites can accommodate with current infrastructure. Adaptation strategies may be needed by recreation participants and public land managers to ensure recreation activities can be accessible in the future. Dante-Wood and Hoang (2018) provide a general framework of adaptation strategies for landscape-scale land management for the Northern Rockies. Jardine and Long (2014) have outlined several approaches to promote climate change resilience in the Sierra Nevada and southern Cascade Range that can be adapted to the study area. They suggest focusing on longer time horizons for planning, setting adaptable objectives with the ability to revisit and modify as needed, and use of valuation tools, monitoring, and research. In addition, the Interagency Visitor Use Management Council has guidebooks that may be useful towards planning and monitoring of recreation use levels and selection of indicators (IVUMC 2019).

Adaptation by Recreation Participants

Increasing temperatures and shorter winter and longer summer seasons will impact the demand and supply of outdoor recreation. Recreationists may need to adjust the timing, location, and type of activity with a changing climate and environmental conditions. The timing of seasonal visitation patterns may shift to shoulder seasons, such as recreating earlier or later in the summer or winter. Where people choose to recreate may change as the result of hotter temperatures where they may decide to hike at higher elevations such as high-alpine areas, where temperatures are cooler. Recreationists may benefit from developing alternative visitation plans in case preferred recreation sites are no longer available. It's possible that alternative sites or activities might be available; however, the alternative location may be less appealing to individuals or family connection to a site or activity might be lost, or the lost activity and location may be viewed as irreplaceable due to its unique nature (Winter et al. 2014). In addition, recreationists have already been forced to adapt due to other indirect climate change effects. For example, recent wildfires have caused several trails to close for an extended period of time. Therefore, recreationists are force to find alternative recreation sites to engage in the same activity, switch activity, or forgo the activity by staying at home.

Adaptation by Public Land Managers

Public land managers may need to adapt to changing environmental conditions and future demand for recreation opportunities. Regardless of the effects from a changing climate, as the absolute growth in visitor days (largely due to population growth) and demand for certain high-quality recreation opportunities increases, sufficient staffing and resources may be required to continue to manage these recreation sites. Limited supply constrains recreation use that Forest Service sites can accommodate with current infrastructure, and it is uncertain whether existing facilities will be able to accommodate increased demand in future recreation activities (White et al. 2016). With increased demand and limited supply, one implication is to consider planning for substitution effects between recreation activities. For example, people who would otherwise be camping may shift to day hiking or walking if campsites are full and they cannot stay overnight. One way this might be accomplished is for the Forest Service to consider maintaining and increasing partnerships and volunteer programs. Community involvement and partnerships have been instrumental in providing recreation opportunities and restoring recreation sites. Some examples of successful partnerships with Forest Service are the Colorado Trail Foundation (TCTF, n.d.) and the Continental Divide Trail Coalition (CDTC, n.d.). Through these partnerships, the Forest Service can acquire the additional labor needed to open developed campsites and national forest roads earlier especially if the season is expanded as is expected under climate change.

Further, the agency can build more campsites and trails at higher elevations that can be used during the hotter mid-summer temperatures that might make lower-elevation trails and campsites undesirable. The increasing technological capabilities of smartphones, apps, and other devices can potentially help public land managers communicate with recreationists for possible trail closures, types of recreation activities available, and safety information. For example, the U.S. Forest Service Interactive Visitor Map (USDA FS, n.d.-f) provides information on the types of recreation opportunities available on each national forest. National Forest apps can potentially help managers provide recreationists with trail information, possible trail closure, and substitute sites. Social media can also be useful for a subset of interested individuals seeking forest updates.

Conclusion and Implications

The biggest challenge for Forest Service managers on both the PSI-NFs and the AR-NFs is the substantial increase in visitor use expected due to the large population increases that are expected along the Front Range. Specifically, recreation days on both national forests are expected to double by 2060. While climate change will result in some recreation activities seeing larger increases, the biggest challenge will be designing plan alternatives to deal with general increases in visitor use. This might entail different alternatives such as: (1) building additional recreation infrastructure (e.g., campsites and trails, especially at higher elevations) along with the means to finance and maintain this expanded infrastructure (e.g., utilizing fee structures permissible under the Federal Lands Recreation Enhancement Act); or (2) imposing rationing mechanisms to balance the demand for recreation and limited supply of recreation opportunities. In any climate future, managers will need to manage congestion and its impacts to the recreational experience.

Notes

[1]https://www.eldora.com/the-mountain/mountain-stats-policies-faqs/mountain-stats
[2]https://skiloveland.com/the-mountain/trail-map-mountain-stats/
[3]https://www.winterparkresort.com/the-mountain/mountain-information/statistics
[4]https://www.skicooper.com/mountain-stats/
[5]https://www.skimonarch.com/hours-stats/
[6]National Visitor Use Monitoring Program (NVUM) web-application: https://www.fs.usda.gov/about-agency/nvum

References

Arkansas River Outfitters Association. 2019. The Arkansas River. http://arkansasriveroutfitters.org/arkansas-river/ [Accessed 13 July 2023]

Askew, A.E.; Bowker, J.M. 2018. Impacts of climate change on outdoor recreation participation: Outlook to 2060. Journal of Park and Recreation Administration. 36: 97–120. https://doi.org/10.18666/JPRA-2018-V36-I2-8316

Bowker, J.M.; Askew, A.E.; Cordell, H.K. 2012. Outdoor recreation participation in the United States—Projections to 2060: A technical document supporting the Forest Service 2010 RPA Assessment. Gen. Tech. Rep. SRS-160. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 34 p. https://doi.org/10.2737/SRS-GTR-160

Boyle, K.J. 2017. Contingent valuation in practice. In: Champ, P.A.; Boyle, K.J.; Brown, T.C., eds. A primer on nonmarket valuation. 2nd edition. Dordrecht, The Netherlands: Springer: 83–132. https://doi.org/10.1007/978-94-007-7104-8_4

Boyle, K.J.; Bergstrom, J.C. 1992. Benefit transfer studies: Myths, pragmatism, and idealism. Water Resources Research. 28(3): 675–683. https://doi.org/10.1029/91WR02591

Bureau of Economic Analysis [BEA]. 2019. Outdoor recreation satellite account: Updated statistics for 2012–2016. BEA 18–48. https://www.bea.gov/system/files/2018-09/orsa0918_0.pdf [Accessed 23 July 2019]

Champ, P.A.; Boyle, K.J.; Brown, T.C., eds. 2017. A primer on nonmarket valuation. 2nd edition. Dordrecht, The Netherlands: Springer, 504 p. https://doi.org/10.1007/978-94-007-7104-8

Colorado Fourteeners Initiative [CFI]. [n.d.]. Colorado 14ers statewide report card. https://www.14ers.org/stay-informed/colorado-14ers-statewide-report-card/ [Accessed 27 March 2020]

Colorado State Demography Office. 2022. Preliminary population forecasts for Colorado regions, 2000–2050 [Data set]. Department of Local Affairs. https://demography.dola.colorado.gov/assets/html/population.html [Accessed 7 July 2023]

Colorado State Demography Office. [n.d.]. Forecasts Methodology. https://drive.google.com/uc?export=download&id=1MvvnPBkmcS-QpY7roZNri7NAVDbl2PXb [Accessed 6 July 2023]

Continental Divide Trail Coalition [CDTC]. [n.d.]. Explore the trail. https://continentaldividetrail.org/explore-the-trail/ [Accessed 17 August 2019]

Cullinane Thomas, C.; Koontz, L.; Cornachione, E. 2019. 2018 National park visitor spending effects: Economic contributions to local communities, states, and the nation. Natural Resource Report NPS/NRSS/EQD/NRR—2019/1922. Fort Collins, CO: National Park Service. https://irma.nps.gov/DataStore/DownloadFile/621641 [Accessed 13 July 2023]

Dante-Wood, S.K; Hoang, L. 2018. Conclusions. In: Halofsky, J.E.; Peterson, D.L.; Dante-Wood, S.K.; [et al.], eds. Climate change vulnerability and adaptation in the Northern Rocky Mountains. Gen. Tech. Rep. RMRS-GTR-374. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. Part 2: 469–475 https://www.fs.usda.gov/research/treesearch/55998

Fisichelli, N.A.; Schuurman, G.W.; Monahan, W.B.; [et al.]. 2015. Protected area tourism in a changing climate: Will visitation at U.S. National Parks warm or overheat? PLoS One. 10(6): e0128226. https://doi.org/10.1371/journal.pone.0128226

Fonner, R.C.; Berrens, R.P. 2014. A hedonic pricing model of lift tickets for U.S. alpine ski areas: Examining the influence of crowding. Tourism Economics. 20(6): 1215–1233. https://doi.org/10.5367/te.2013.0338

Furniss, M.J.; Howe, C. 2016. Regional-scale climate change vulnerability assessment for infrastructure in the national forests and grasslands of the Rocky Mountain Region: Including roads, trails, and recreational facilities. Lakewood, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Region; Rocky Mountain Research Station. 117 p. https://www.fs.usda.gov/rm/pubs_journals/2016/rmrs_2016_furniss_m001.pdf

Hand, M.S.; Lawson, M., 2018. Effects of climate change on recreation in the northern Rockies Region. In: Halofsky, J.E.; Peterson, D.L.; Dante-Wood, S.K.; [et al.]., eds. Climate change vulnerability and adaptation in the Northern Rocky Mountains. Gen. Tech. Rep. RMRS-GTR-374. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. Part 2: 398–433. https://www.fs.usda.gov/research/treesearch/55995

Hand, M.S.; Smith, J.W.; Peterson, D.L.; [et al.]. 2018. Effects of climate change on outdoor recreation. In: Halofsky, J.E.; Peterson, D.L.; Ho, J.J.; [et al.]., eds. 2018. Climate change vulnerability and adaptation in the Intermountain Region. Gen. Tech. Rep. RMRS-GTR-375. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. Part 2: 316–338. https://www.fs.usda.gov/research/treesearch/56120

Holmes, T.P.; Adamowicz, W.L.; Carlsson, F. 2017. Choice experiments. In: Champ, P.A.; Boyle, K.J.; Brown, T.C., eds. A primer on nonmarket valuation. 2nd edition. Dordrecht, The Netherlands: Springer: 133–186. https://doi.org/10.1007/978-94-007-7104-8_5

Interagency Visitor Use Management Council [IVUMC]. 2019. Monitoring guidebook: Evaluating effectiveness of visitor use management. 1st edition. Denver, CO. https://visitorusemanagement.nps.gov/Content/documents/highres_final_Monitoring_Guidebook_Edition_One_IVUMC.pdf [Accessed 13 July 2023]

Jardine, A.; Long, J.W. 2014. Synopsis of climate change. In: Long, J.W.; Quinn-Davidson, L.; Skinner, C.N., eds. Science synthesis to support socioecological resilience in the Sierra Nevada and southern Cascade Range. Gen. Tech. Rep. PSW-GTR-247. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 71–81. https://www.fs.usda.gov/research/treesearch/46955

Johnston, R.J.; Boyle, K.J.; Adamowicz, W.; [et al.]. 2017. Contemporary guidance for stated preference studies. Journal of the Association of Environmental and Resource Economists. 4(2): 319–405. https://doi.org/10.1086/691697

Johnston, R.J.; Rolfe, J.; Rosenberger, R.S.; [et al.], eds. 2015. Benefit transfer of environmental and resource values: A guide for researchers and practitioners. Dordrecht, The Netherlands: Springer. 582 p. https://doi.org/10.1007/978-94-017-9930-0

Landres, P.; Meyer, S. 2000. National Wilderness Preservation System database: Key attributes and trends, 1964 through 1999. Gen. Tech. Rep. RMRS-GTR-18. Ogden, UT: U.S. Department of Agriculture, Rocky Mountain Research Station. 98 p. https://doi.org/10.2737/RMRS-GTR-18

Lien, G.; Haugom, E.; Alnes, P.K.; [et al.]. 2022. Willingness to pay for fast-track tickets in the alpine skiing industry. SSRN. http://dx.doi.org/10.2139/ssrn.4234081

Loomis, J.; Crespi, J. 1999. Estimated effects of climate change on selected outdoor recreation activities in the United States. In: Mendelsohn R.; Neumann, J.E., eds. The impact of climate change on the United States economy. Cambridge, The United Kingdom: Cambridge University Press: 289–314. https://doi.org/10.1017/CBO9780511573149.012

Loomis, J.B.; Richardson, R.B. 2006. An external validity test of intended behavior: Comparing revealed preference and intended visitation in response to climate change. Journal of Environmental Planning and Management. 49(4): 621–630. https://doi.org/10.1080/09640560600747562

Loomis, J.B.; Walsh, R.G. 1997. Recreation economic decisions: Comparing benefits and costs. 2nd edition. State College, PA: Venture Publishing, Inc. 440 p.

Meldrum, J.; Huber, C. 2019. An economic perspective on the relationship between wilderness and water resources. Natural Areas Journal. 39(1): 33–45. https://doi.org/10.3375/043.039.0103

Mendelsohn, R.; Markowski, M. 1999. The impact of climate change on outdoor recreation. In: Mendelsohn R.; Neumann, J.E., eds. The impact of climate change on the United States economy. Cambridge, The United Kingdom: Cambridge University Press: 267–288. https://doi.org/10.1017/CBO9780511573149.011

Outdoor Foundation. 2018. 2018 Outdoor participation report. https://outdoorindustry.org/participation/ [Accessed 21 August 2019]

Outdoor Industry Association. 2019. The outdoor recreation economy. https://outdoorindustry.org/ [Accessed 21 August 2019]

Outdoor Industry Association. 2019. The outdoor recreation economy for Colorado. https://outdoorindustry.org/state/colorado [Accessed 21 August 2019]

Recreation Use Values Database [RUVD]. 2016. Corvallis, OR: Oregon State University, College of Forestry. http://recvaluation.forestry.oregonstate.edu/ [Accessed 13 July 2023]

Richardson, L.; Loomis, J.; Kroeger, T.; [et al.]. 2015. The role of benefit transfer in ecosystem service valuation. Ecological Economics. 115: 51–58. https://doi.org/10.1016/j.ecolecon.2014.02.018

Richardson, R.B.; Loomis, J.B. 2004. Adaptive recreation planning and climate change: A contingent visitation approach. Ecological Economics. 50(1–2): 83–99. https://doi.org/10.1016/j.ecolecon.2004.02.010

Rosenberger, R. 2015. Benefit transfer validity and reliability. In: Johnston, R.; Rolfe, J.; Rosenberger, R.; [et al.], eds. Benefit transfer of environmental and resource values: A handbook for researchers and practitioners. Dordrecht, The Netherlands: Springer: 307–326. https://doi.org/10.1007/978-94-017-9930-0_14

Rosenberger, R.S.; Loomis, J.B. 2017. Benefit transfer. In: Champ, P.A.; Boyle, K.J.; Brown, T.C., eds. A primer on nonmarket valuation. 2nd edition. Dordrecht, The Netherlands: Springer: 431–462. https://doi.org/10.1007/978-94-007-7104-8_11

Rosenberger, R.S.; White, E.M.; Kline, J.D., [et al.]. 2017. Recreation economic values for estimating outdoor recreation economic benefits from the National Forest System. Gen. Tech. Rep. PNW-GTR-957. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 33 p. https://doi.org/10.2737/PNW-GTR-957

Scott, D.; Dawson, J.; Jones, B. 2008. Climate change vulnerability of the U.S. northeast winter recreation–tourism sector. Mitigation and Adaptation Strategies for Global Change. 13: 577–596.

Segerson, K. 2017. Valuing environmental goods and services: An economic perspective. In: Champ, P.A.; Boyle, K.J.; Brown, T.C., eds. A primer on nonmarket valuation. 2nd edition. Dordrecht, The Netherlands: Springer: 1–25. https://doi.org/10.1007/978-94-007-7104-8_1

Shaw, W.D.; Loomis, J.B. 2008. Frameworks for analyzing the economic effects of climate change on outdoor recreation. Climate Research. 36: 259–269. https://doi.org/10.3354/cr00748

Statewide Comprehensive Outdoor Recreation Plan [SCORP]. 2018. The 2017 economic contributions of outdoor recreation in Colorado: A regional and county-level analysis. https://cpw.state.co.us/Documents/Trails/SCORP/Final-Plan/SCORP-AppendixF-EconomicContributions.pdf [Accessed 13 July 2023]

The Colorado Department of Transportation [CDOT]. [n.d.]. Colorado: The official guide to the scenic and historic byways. http://edition.pagesuite-professional.co.uk/html5/reader/production/default.aspx?pubname=&pubid=edfe2883-9dfe-4cb6-ac27-261c2bbba921 [Accessed 13 July 2023]

The Colorado Trail Foundation [TCTF]. [n.d.]. The Colorado Trail. https://coloradotrail.org/ [Accessed 13 July 2023]

U.S. Census Bureau. 2020. 2020 census redistricting data (Public Law 94-171)— Colorado, Table P1 [Data set]. https://data.census.gov/table?g=040XX00US08&tid=DECENNIALPL2020.P1 [Accessed 5 July 2023]

U.S. Census Bureau. 2021. Historical population change data (1910–2020) [data set]. https://www.census.gov/data/tables/time-series/dec/popchange-data-text.html [Accessed 5 July 2023]

USDA Forest Service [USDA FS]. 2016. Future of America's forests and rangelands: Update to the Forest Service 2010 Resources Planning Act Assessment. Gen. Tech. Rep. WO-94. Washington, DC: U.S. Department of Agriculture, Forest Service. 250 p. https://doi.org/10.2737/WO-GTR-WO-GTR-94

USDA Forest Service [USDA FS]. 2017. U.S. Forest Service national visitor use monitoring survey results national summary report—Data collected FY 2012 through FY 2016. https://www.fs.usda.gov/sites/default/files/2016-National-Summary-Report.pdf [Accessed 12 July 2023]

USDA Forest Service [USDA FS]. n.d.-a. Special places. Pike-San Isabel National Forests & Cimarron and Comanche National Grasslands. https://www.fs.usda.gov/attmain/psicc/specialplaces [Accessed 27 March 2020]

USDA Forest Service [USDA FS]. n.d.-b. Campground camping. Pike-San Isabel National Forests & Cimarron and Comanche National Grasslands. https://www.fs.usda.gov/activity/psicc/recreation/camping-cabins/?recid=12403&actid=29 [Accessed 13 July 2023]

USDA Forest Service [USDA FS]. n.d.-c. Campgrounds at-a-glance. Arapaho & Roosevelt National Forests and Pawnee National Grassland. https://www.fs.usda.gov/detailfull/arp/recreation/?cid=fseprd527741&width=full [Accessed 13 July 2023]

USDA Forest Service [USDA FS]. n.d.-d. Leadville Ranger District Colorado Trail sections. Pike-San Isabel National Forests & Cimarron and Comanche National Grasslands. https://www.fs.usda.gov/recarea/psicc/recreation/hiking/recarea/?recid=81971&actid=51 [Accessed 13 July 2023]

USDA Forest Service [USDA FS]. n.d.-e. Kenosha Pass area. Pike-San Isabel National Forests & Cimarron and Comanche National Grasslands. https://www.fs.usda.gov/recarea/psicc/recreation/natureviewing/recarea/?recid=82584&actid=64 [Accessed 13 July 2023]

USDA Forest Service [USDA FS]. n.d.-f. U.S. Forest Service interactive visitor map. https://www.fs.usda.gov/ivm/ [Accessed 13 July 2023]

USDA Forest Service [USDA FS]. n.d.-g. Cameron Peak fire closure. Arapaho & Roosevelt National Forests and Pawnee National Grassland. https://www.fs.usda.gov/alerts/arp/alerts-notices/?aid=60887 [Accessed 13 July 2023]

U.S. Geological Survey [USGS]. 2019. National hydrography dataset. Ver. USGS National Hydrography Dataset Best Resolution (NHD) for Hydrologic Unit (HU) 4 – 2001. https://www.usgs.gov/core-science-systems/ngp/national-hydrography/access-national-hydrography-products [Accessed 13 July 2023]

Walsh, R.G.; Miller, N.P.; Gilliam, L.O. 1983. Congestion and willingness to pay for expansion of skiing capacity. Land Economics. 59(2): 195–210 https://doi.org/10.2307/3146048

Walsh, R.G.; Radulaski O.; Lee, L. 1984. Value of hiking and cross-country skiing in roaded and nonroaded areas of a national forest. In: Kaiser, F.; Schweitzer, D.; Brown, P., eds. Proceedings for economic value analysis of multiple-use forestry: 176–187

White, E. 2017. Spending patterns of outdoor recreation visitors to national forests. Gen. Tech. Rep. PNW-GTR-961. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 70 p. https://doi.org/10.2737/PNW-GTR-961

White, E.M.; Bowker, J.M.; Askew, A.E.; [et al.]. 2016. Federal outdoor recreation trends: Effects on economic opportunities. Gen. Tech. Rep. PNW-GTR-945. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Station. 46 p. https://doi.org/10.2737/PNW-GTR-945

Wilderness Connect. 2019. National Wilderness Preservation System database. W.A. Franke College of Forestry and Conservation's Wilderness Institute, The University of Montana. https://wilderness.net/practitioners/wilderness-areas/search.php?STATE=6&AGENCY=FS#resultsSection [Accessed 13 July 2023]

Winter, P.L.; Long, J.W.; Lake, F.K. 2014. Sociocultural perspectives on threats, risks, and health. In: Long, J.W.; Quinn-Davidson, L.; Skinner, C.N., eds. Science synthesis to support socioecological resilience in the Sierra Nevada and southern Cascade Range. Gen. Tech. Rep. PSW-GTR-247. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 569–598. https://www.fs.usda.gov/research/treesearch/46972

Winter, P.L.; Sánchez, J.J.; Olson, D.D. 2021. Effects of climate change on outdoor recreation in the Sierra Nevada bioregion. In: Halofsky, J.E.; Peterson, D.L; Buluc, L.; [et al.], eds. Climate change vulnerability and adaptation for infrastructure and recreation in the Sierra Nevada. Gen. Tech. Rep. PSW-GTR-272. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 181–244. Chapter 5. https://www.fs.usda.gov/research/treesearch/63219

Wobus, C.; Small, E.E.; Hosterman, H.; [et al.]. 2017. Projected climate change impacts on skiing and snowmobiling: A case study of the United States. Global Environmental Change. 45: 1–14. https://doi.org/10.1016/j.gloenvcha.2017.04.006

Young, R.A.; Loomis, J.B. 2014. Determining the economic value of water: Concepts and methods. 2nd edition. New York, NY: Resources for the future. New York: Routledge. 358 p. https://doi.org/10.4324/9780203784112

Ziesler, P.S. 2019. Statistical abstract: 2018. Natural resource data series. NPS/NRSS/EQD/NRDS—2019/1219. Fort Collins, CO: National Park Service. https://irma.nps.gov/DataStore/DownloadFile/620857 [Accessed 14 July 2023].

Appendix

Table A1—Participation frequency (%) and visits (1000s) by recreation activity at the Pike and San Isabel National Forests (National Visitor Use Monitoring sampling in 2016).

Activity	Main activity (%)	Visitation (1,000s)
Warm-weather activities		
Hiking/walking	19.0	842.6
Viewing natural features	19.7	872.9
Other nonmotorized	0.7	32.6
Bicycling	3.2	140.5
Developed camping	2.9	130.1
Picnicking	0.6	26.7
Backpacking	0.5	21.4
Horseback riding	0.2	7.4
Primitive camping	1.8	78.8
Total	**48.6**	**2,153.1**
Winter activities		
Downhill skiing	5.1	224.9
Cross-country skiing	0.4	17.1
Snowmobiling	0.6	26.9
Total	6.1	268.9
Wildlife activities		
Fishing	10.4	460.5
Hunting	0.7	29.8
Viewing wildlife	1.4	63.4
Total	**12.5**	**553.7**
Gathering forest products		
Gathering forest products	0.6	24.7
Total	**0.6**	**24.7**
Water-based activities, not including fishing		
Motorized water activities	0.0	0.0
Nonmotorized water	1.1	49.3
Total	**1.1**	**49.3**

Table A1 continued.

Activity	Main activity (%)	Visitation (1,000s)
Other activities		
Relaxing	3.3	147.5
Driving for pleasure	14.4	640.7
Some other activity	2.3	100.1
Off-highway vehicle use	3.2	141.7
Nature center activities	0.2	7.1
Motorized trail activity	7.0	310.1
Nature study	0.2	9.0
Resort use	0.0	0.1
Visiting historic sites	0.2	10.6
Other motorized activity	0.0	0.0
Total	**30.8**	**1,366.7**
Total Visits	—	**4,434.0**

— Indicates no data.
*Visits of each category does not sum to Total Visits due to rounding and excluding “No Activity Reported”.

Table A2—Participation frequency (%) and visits (1000s) by recreation activity at the Arapaho and Roosevelt National Forests (National Visitor Use Monitoring sampling in 2015).

Activity	Main activity (%)	Visitation (1,000s)
Warm-weather activities		
Hiking/walking	16.7	743.6
Viewing natural features	16.7	744.5
Other nonmotorized	0.6	28.4
Bicycling	1.7	76.3
Developed camping	6.0	266.4
Picnicking	0.2	8.8
Backpacking	0.3	15.4
Horseback riding	0.0	0.0
Primitive camping	0.7	30.8

Table A2 continued.

Activity	Main activity (%)	Visitation (1,000s)
Total	**42.9**	**1914.3**
Winter activities		
Downhill skiing	26.7	1191.0
Cross-country skiing	2.0	87.9
Snowmobiling	0.8	35.4
Total	**29.5**	**1314.4**
Wildlife activities		
Fishing	4.3	190.6
Hunting	8.3	369.9
Viewing wildlife	0.6	27.0
Total	**13.2**	**587.5**
Gathering forest products		
Gathering forest products	0.1	4.4
Total	**0.1**	**4.4**
Water-based activities, not including fishing		
Motorized water activities	0.1	4.8
Nonmotorized water	0.3	13.8
Total	**0.4**	**18.6**
Other activities		
Relaxing	1.7	73.9
Driving for pleasure	8.1	362.7
Some other activity	1.2	53.3
Off-highway vehicle use	1.7	76.7
Nature center activities	0.0	0.0
Motorized trail activity	0.8	37.6
Nature study	0.0	0.0
Resort use	0.0	0.0
Visiting historic sites	0.1	2.8
Other motorized activity	0.0	0.0
Total	**13.6**	**607.0**
Total Visits	—	**4,458.8**

— Indicates no data.
*Visits of each category does not sum to Total Visits due to rounding and excluding “No Activity Reported”.

Table A3—Days (1000s) and conversion factor by recreation activity per sampling year at the Pike and San Isabel National Forests and the Arapaho and Roosevelt National Forests.

Activity	Conversion factor (R2)	Days (1,000s)	
		Pike-San Isabel NF (2016)	Arapaho-Roosevelt NF (2015)
Warm-weather activities			
Hiking/walking	1.1	926.9	817.9
Viewing natural features	1.1	960.2	818.9
Other nonmotorized	1.2	39.1	34.1
Bicycling	1.1	154.6	84.0
Developed camping	2.7	351.4	719.4
Picnicking	1.1	29.4	9.7
Backpacking	2.5	53.5	38.6
Horseback riding	1.3	9.6	0.0
Primitive camping	2.4	189.2	74.0
Total	—	**2,713.8**	**2,596.5**
Winter activities			
Downhill skiing	1.0	224.9	1,191.0
Cross-country skiing	1.0	17.1	87.9
Snowmobiling	1.2	32.2	42.5
Total	—	**274.3**	**1,321.4**
Wildlife activities			
Fishing	1.2	552.6	228.7
Hunting	1.3	38.7	480.9
Viewing wildlife	1.1	69.7	29.7
Total	—	**661.1**	**739.3**
Gathering forest products			
Gathering forest products	1.1	27.2	4.9
Total	—	**27.2**	**4.9**
Water-based activities, not including fishing			
Motorized water activities	1.1	0.0	5.2
Nonmotorized water	1.1	54.3	15.2
Total	—	**54.3**	**20.4**

Table A3 continued.

Activity	Conversion factor (R2)	Days (1,000s)	
		Pike-San Isabel NF (2016)	Arapaho-Roosevelt NF (2015)
Other activities			
Relaxing	1.5	221.2	110.9
Driving for pleasure	1.1	704.7	398.9
Some other activity	1.2	120.1	64.0
Off-highway vehicle use	1.2	170.0	92.1
Nature center activities	1.0	7.1	0.0
Motorized trail activity	1.3	403.1	48.9
Nature study	1.0	9.0	0.0
Resort use	2.1	0.2	0.0
Visiting historic sites	1.1	11.6	3.1
Other motorized activity	1.2	0.0	0.0
Total	—	**1,647.0**	**717.8**
Total days (1,000s)	—	**5,377.6**	**5,400.4**

— Indicates no data.

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Conclusions

Thomas Timberlake, Donna Shorrock, Delilah Jaworski

Introduction

This report provides a foundation for climate change response on the Arapaho-Roosevelt and Pike-San Isabel National Forests located along the Front Range and mountainous regions of Colorado. The preceding chapters outline the best available scientific information on the impacts of climate change on the Arapaho-Roosevelt and Pike-San Isabel National Forests. This vulnerability assessment supports national forest managers in meeting expectations to consider climate change in their work, including land management planning guided by the 2012 Planning Rule for implementing the National Forest Management Act (36 CFR 219) and environmental analysis and decision making under the National Environmental Policy Act (42 USC § 4321). The importance of considering climate change in national forest management is highlighted in a series of recent Federal government-wide, department, and agency-specific policies, including Executive Order on Tackling the Climate Crisis at Home and Abroad (Executive Order No. 14,008, 2021), updated National Environmental Policy Act guidance from the Council on Environmental Quality, the USDA Secretary's Memorandum on Climate Resilience and Carbon Stewardship of America's National Forests and Grasslands (USDA 2022), and the Forest Service's Climate Adaptation Plan (USDA FS 2022).

Responding to climate change represents a key contemporary challenge for managers. It will require managers to modify existing management activities, explore new practices, and grapple with considerable levels of uncertainty. Climate change exacerbates existing management challenges, including intensifying recreation pressures in a region with a growing population, restoring fire-adapted forests, and restoring landscapes impacted by recent wildfires. This vulnerability assessment offers a starting point for managers and scientists to work together to take on these challenges. These collaborative efforts have begun, including through a science-management workshop that occurred as part of this vulnerability assessment and that is described below; however, tackling climate change will also require additional work building on this vulnerability assessment. This conclusions chapter provides some initial ideas to support future adaptation based on this assessment.

Vulnerability Assessment Workshop and Opportunities for Future Engagements

Vulnerability assessment processes offer opportunities for scientists and managers to learn from one another about impacts of climate change and potential management responses (Peterson et al. 2011; Swanston et al. 2016; Timberlake and Schultz 2019). To address this need, scientists and managers involved in this assessment convened for a 2-day workshop in November 2019 in Fort Collins, Colorado. The 35 attendees included staff from both national forest units, the Rocky Mountain Regional Office, and scientists

affiliated with the USDA Forest Service Rocky Mountain and Pacific Southwest Research Stations, the Northern Institute of Applied Climate Science, Colorado State University, and the United States Geological Survey Climate Adaptation Science Centers. During the workshop, scientists presented on climate change impacts to different resources, and workshop participants worked through a structured scenario planning approach to explore how to achieve management goals under different future climate scenarios (Murphy et al. 2016; Rowland et al. 2014).

For this scenario planning exercise, participants applied a modified version of the Climate Change Response Framework, which provides a step-by-step approach for developing adaptation approaches and tactics based on the strategies of resistance, resilience, and transition. In this context, the framework was modified to apply specifically to scenario planning (Millar et al. 2007; Swanston et al. 2016). Scenarios provided a picture of future conditions, and the Climate Change Response Framework walked participants through a structured process to consider implications for management and adaptation. Topics discussed by breakout groups included managing forests to reduce risks to insects and wildfire, climate-smart watershed restoration, developing climate-adapted winter recreation infrastructure, and managing changing social expectations in light of changing landscapes. These workshop discussions provided an initial opportunity for staff on these national forests to consider adaptation actions to take in response to climate change; however, they did not result in specific lists of adaptation actions. Moving forward, additional organized workshops and informal discussions may be necessary in order to apply the information in this vulnerability assessment to identify specific adaptation actions relevant to specific projects and activities occurring on these national forests.

Key Concepts in Forest Adaptation

A commonly used framework for guiding forest adaptation involves developing a "toolbox" of adaptation strategies based on the concepts of resistance, resilience, and transition (Millar et al. 2007; Peterson et al. 2011; Swanston et al. 2016). During the workshops described above, participants used these concepts to think through adaptation strategies, and this section briefly summarizes the concepts as they may be relevant for future applications of this assessment. Resistance describes the goal of forestalling or resisting change with the intent of maintaining current conditions or features on the landscape. Resilience aims to allow some level of change or disturbance but ensures that the system is able to recover and retain the same general composition, structure, pattern, and ecological processes following disturbance. Transition (or response) describes the goal of facilitating ecosystem transition and moving the system to a new state.

In designing projects and applying this vulnerability assessment, it may be useful to consider how treatment prescriptions may vary if guided by resistance, resilience, or transition, and how they may be affected under different climate scenarios. For

example, restoration of dry mixed-conifer forests under a resistance strategy may involve uniform thinning to reduce stand density coupled with fuel breaks to keep fire out of treated stands, whereas a resilience strategy may involve more substantial density reductions and thinning to create clumps of trees and gaps, leaving a residual structure able to accommodate low-severity surface fire under most conditions. A transition approach might involve treatments that promote a shift in species composition to favor ponderosa pine (*Pinus ponderosa*) with full removal of Douglas-fir (*Pseudotsuga menziesii*) and lodgepole pine (*Pinus contorta*), and that result in much lower residual densities. Alternatively, a transition treatment may involve allowing a burned area to transition from forest to grassland following a high-severity fire. By comparing potential treatment options across these concepts, managers may develop a better sense of the full range of possibilities. Ultimately, selecting the appropriate treatment will require considering site-specific conditions, management goals, and constraints.

Adapting to Changes

Climate change adaptation will require practitioners to translate information on climate vulnerabilities, including what is included in this assessment, and adaptation concepts like those described above, into feasible management actions. Many of these actions will build on things that managers are already doing, but they may need to modify these activities to address changes in timing, location, and intensity associated with climate change. This section provides some theoretical examples of these types of changes building on information in previous chapters of the vulnerability assessment, though it does not provide a comprehensive summary of the rest of the report.

As the preceding chapters demonstrate, climate change will lead to shifts in seasons and more variable conditions. Notably, the climate change projections considered in this report (Chapter 3) indicate that the study area will undergo changes in seasonal temperatures, duration of fire seasons, and timing of stream runoff, as compared to current conditions. In response, managers may need to shift the timing of when they implement certain management activities, allowed uses, and seasonal restrictions. For example, longer fire seasons may require looking earlier in the spring and later in the fall for appropriate burn windows for prescribed fire. Similarly, winters with lower snowpack may limit opportunities for pile burning and managers may need to concentrate these activities in the years that offer sufficient snowpack. Other examples of changes in timing include the potential for shorter seasons for winter recreation and longer seasons for warm-weather uses, which has implications for opening dates for recreation sites and roads. Preparing for these yearly variations, as well as changes in seasonality within years, will require enhanced flexibility in business practices, staffing, and out-year planning.

In addition to considering timing, managers will also need to consider how climate change will play out in space. Changes in temperature and precipitation patterns will affect the suitability of locations for supporting species, public uses, and management activities. In this landscape in particular, effects of climate change will interact with topography, and features like slope, aspect, and elevation will be important considerations in identifying adaptation actions. For example, in implementing postfire reforestation projects, silviculturists will need to make site-specific determinations of which species and genotypes are likely to survive given projected changes in climate, while considering factors like north-facing slopes that remain comparatively cooler. These types of careful considerations of changes in space are also relevant for managing watersheds, recreation, and other resource areas. Wildfires, flooding, and other disturbances may make the current locations of developed recreational sites less desirable or unsafe for recreational activities. Maps and other information in this vulnerability assessment help inform these considerations.

Climate change will intensify existing stressors to the landscape, and managers will need to modify the intensity of management activities accordingly. For example, climate change projections indicate an increase in days per year with high fire danger. As the chapter on forests indicates, more intensive drought conditions will impact seedling survival and tree mortality. Intensified drought conditions will also increase susceptibility to insect outbreaks. Thinning is already a common activity to build resilience to these stressors, as it reduces stress on individual trees, removes ladder fuels, and helps forests better accommodate surface fires while reducing the risk of crown fires. However, preparing for climate change may necessitate thinning over larger areas or increasing the intensity of the activity by thinning to lower residual densities.

Climate change may also contribute to intensified uses of public lands in the area. For example, disturbance events like fires and flooding may make some recreation areas unavailable or less desirable for recreation leading to intensified use at unaffected sites. More broadly, increased temperatures in communities along the Front Range may drive increased recreational visits as people look for opportunities to escape the heat and travel to higher elevations. Approaches for responding to these effects include permitting and reservation systems to manage crowding, barriers and signage to prevent resource damage from intensified use, and coordinated messaging and public education campaigns to provide recreationists with up-to-date information. Intensified use may also require more frequent and thorough maintenance activities, including cleaning of facilities and trail work.

Managing Under Uncertainty

Climate-informed management requires accepting the inherent uncertainty associated with climate change. While it is certain that changes will occur, there is still uncertainty in anticipating what the future will hold. For example, as Chapter 3 highlights, temperatures are projected to increase and changes in precipitation patterns are projected to occur; however, the specific magnitude of these changes depends on the models used and assumptions driving these models. Similarly, while previous chapters describe probable trends associated with climate change, it is still difficult to identify exactly when and where effects of climate change will show up on the landscape. The scenario planning process used at the vulnerability assessment workshop and described above offered one approach for considering uncertainty. A need for the future will be to develop management approaches that enable effective and intentional adaptation actions while also accepting the high levels of uncertainty associated with climate change.

Monitoring is essential in managing under uncertainty. Monitoring both indicators related to climate, such as snowpack, and those related to ecological, hydrological, and social responses to climate change can help improve understanding of ongoing trends. For example, ongoing monitoring of postfire tree regeneration trends along with climate indicators provides insights on how forests are recovering from fire under increasing temperatures. Monitoring approaches are also a key element of adaptive management, which generally describes structured efforts to learn while managing and adjust in response to this learning. Adaptive management approaches vary in their complexity but may involve asking questions and making hypotheses, designing projects as experiments, and using control sites. Across these efforts, robust monitoring is crucial for answering questions and facilitating learning that can further refine future management activities (Larson et al. 2013).

Applications in Decisionmaking

Vulnerability assessments offer peer-reviewed syntheses of scientific information that can help managers integrate climate change considerations into existing decisionmaking processes (Timberlake and Schultz 2019). Climate change considerations are broadly relevant to many facets of national forest management, including forest plan revision under the 2012 Planning Rule, project-level environmental analysis under the National Environmental Policy Act (NEPA), and other forms of strategic planning and prioritization. Recent policies emphasize the need to intentionally and explicitly consider climate change across these settings (USDA FS 2022).

Forest planning is a particularly appropriate situation for applying vulnerability assessments, given the longer timeframes and landscape scales associated with land management planning. Further, robust considerations of climate change in land

management plans facilitate the development of projects that address climate change (Timberlake and Schultz 2019). The 2012 Planning Rule, which guides land management planning under the National Forest Management Act, requires considerations of climate change throughout the planning process. The rule requires that assessment reports developed to inform plans consider climate change as a system driver and stressor (36 CFR 219.6). Land management plans developed under the rule are expected to consider climate change in providing for ecosystem integrity and multiple uses (36 CFR 219.8 and 219.10). Additionally, the monitoring program must address whether there are measurable changes to the plan area resulting from climate change (36 CFR 219.12). Collectively, the planning framework and its three phases are intended to enable adaptation to changing conditions.

Beyond land management planning, vulnerability assessments are also directly relevant to project-level planning, including NEPA analyses (Timberlake and Schultz 2019). In 2023, the Council on Environmental Quality released updated interim NEPA Guidance on Consideration of Greenhouse Gas Emissions and Climate Change, which guides Federal agencies' consideration of climate change in NEPA. The guidance indicates that NEPA analyses should consider the effects of climate change on proposed actions, the affected environment, and environmental effects, and highlights the value of using available assessments for meeting this expectation (CEQ 2023). The vulnerability assessment provides general context on observed and projected changes in climate, as well as detailed descriptions of climate effects across different resource areas, which can be efficiently incorporated into NEPA analyses. In addition, the vulnerability assessment can support other forms of decisionmaking, including programmatic multi-year planning, project prioritization, and discipline-specific processes like watershed restoration action plans.

Looking Forward

In 2022, the Forest Service released its Climate Adaptation Plan that highlights the importance of adapting to climate change to the agency's ability to achieve its mission and aligns agency efforts with expectations in executive orders and other high-level policies (USDA FS 2022). This national plan highlights high level challenges brought on by climate change, including changing wildfire regimes and intensifying extreme events, and identifies adaptation actions that managers can implement to address these challenges. The Forest Service Climate Adaptation Plan serves as a call to action for all agency staff and their partners to understand and address climate change in the landscapes where they work. In 2023, Region 2 will draft a regional climate action plan tiering to the six climate change vulnerabilities described in the national adaptation plan. The regional action plan will provide specific adaptation actions that tier to this and other vulnerability assessments. This vulnerability assessment provides a useful resource to help managers on the Arapaho-Roosevelt and Pike-San Isabel National Forests take on this national priority and manage their lands and resources under climate change.

References

Council on Environmental Quality (CEQ). 2023. National Environmental Policy Act guidance on consideration of greenhouse gas emissions and climate change. Federal Register. 88(5): 1196–1212. https://www.federalregister.gov/documents/2023/01/09/2023-00158/national-environmental-policy-act-guidance-on-consideration-of-greenhouse-gas-emissions-and-climate

Executive Order No. 14,008. 86 FR 7619. (Jan. 27, 2021). https://www.federalregister.gov/documents/2021/02/01/2021-02177/tackling-the-climate-crisis-at-home-and-abroad

Larson, A.J.; Belote, R.T.; Williamson, M.A.; [et al.]. 2013. Making monitoring count: Project design for active adaptive management. Journal of Forestry. 111(5): 348–356. https://doi.org/10.5849/jof.13-021

Millar, C.I.; Stephenson, N.L.; Stephens, S.L. 2007. Climate change and forests of the future: Managing in the face of uncertainty. Ecological Applications. 17(8): 2145–2151. https://doi.org/10.1890/06-1715.1

Murphy, D.; Wyborn, C.; Yung, L. 2016. Engaging communities and climate change futures with multi-scale, iterative scenario-building. Human Organization. 75(1): 33–46. https://doi.org/10.17730/0018-7259-75.1.33

Peterson, D.L.; Millar, C.I.; Joyce, L.A.; [et al.]. 2011. Responding to climate change in national forests: A guidebook for developing adaptation options. Gen. Tech. Rep. PNW-GTR-855. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 1–109. https://doi.org/10.2737/PNW-GTR-855

Rowland, E.R.; Cross, M.S.; Hartmann, H. 2014. Considering multiple futures: Scenario planning to address uncertainty in natural resource conservation. Washington, DC: U.S. Fish and Wildlife Service. https://doi.org/10.1016/j.crm.2017.07.002

Swanston, C.W.; Janowiak, M.K.; Brandt, L.A.; [et al.]. 2016. Forest adaptation resources: Climate change tools and approaches for land managers. 2nd edition. Gen. Tech. Rep. NRS-GTR-87-2. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 161 p. http://dx.doi.org/10.2737/NRS-GTR-87-2

Timberlake, T.J.; Schultz, C.A. 2019. Climate change vulnerability assessment for forest management: The case of the U.S. Forest Service. Forests. 10: 1–22. https://doi.org/10.3390/f10111030

U.S. Department of Agriculture [USDA]. 2022. Secretary's Memorandum 1077-004: Climate resilience and carbon stewardship of America's national forests and grasslands. https://www.usda.gov/directives/sm-1077-004 [Accessed 5 February 2023]

USDA Forest Service [USDA FS]. 2022. Climate adaptation plan. FS-1196. Washington, DC: U.S. Department of Agriculture, Forest Service. https://www.usda.gov/sites/default/files/documents/4_NRE_FS_ClimateAdaptationPlan_2022.pdf