Climate Change Assessment for the San Juan Mountain Regions, Southwestern Colorado, USA: A Review of Scientific Research

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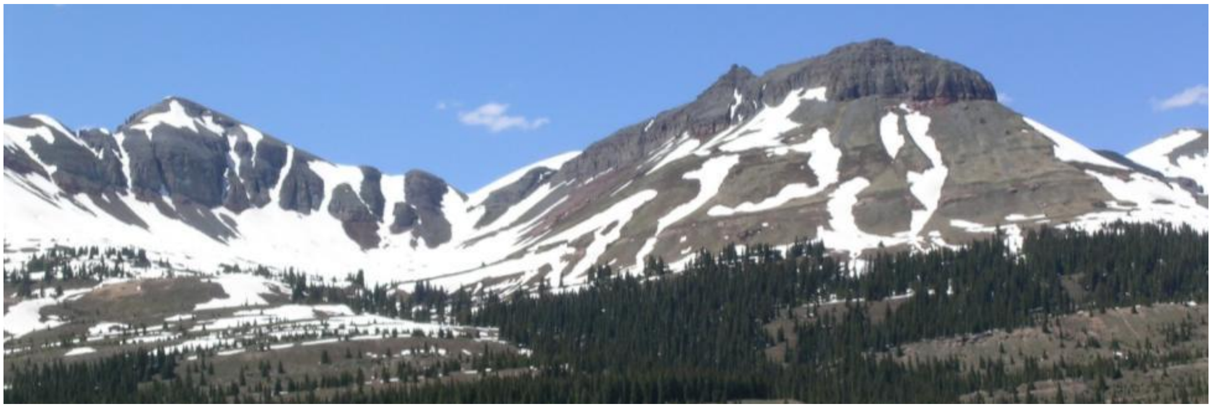

Mountain Studies Institute
SAN JUAN MOUNTAINS, COLORADO

The Mission of the Mountain Studies Institute is to enhance understanding and sustainable use of the San Juan Mountains through research and education.

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Executive Summary

This report is a synthesis of the state of the science regarding the physical and biological aspects of climate change that are important for evaluating impacts to the water resources, ecosystems, and human communities in southwestern Colorado. The scientific findings in this report are presented to identify critical data gaps and to support regional and local efforts towards climate adaptation. The document describes observed conditions and trends, modeling results, and projections of future conditions. It also incorporates a growing body of ecological knowledge that helps us to understand and interpret these historic trends, current conditions, and future projections for managing ecosystems and wildlife in our region. This report is aimed at a fairly technical audience and a shortened outreach version will be released at a later date. Knowledge about climate change and its impacts is evolving. This report is a snapshot of the current state of the science. This is a living document and should be updated as the science progresses.

The following summary provides an overview of observed and projected changes in climate, water, ecosystems, and wildlife resources for southwestern Colorado, based on the work of leading authorities such as the Intergovernmental Panel on Climate Change (IPCC), U.S. Climate Change Science Program, Western Water Assessment, and many individual scientists and institutions. Observations of changes in climate in southwest Colorado are based upon decades of research on past weather trends. This growing body of science demonstrates that southwest Colorado has warmed during the 20th century, and many other changes have been reported.

Each climate model involves representing a multitude of interactions and feedbacks among land, water, oceans, and atmosphere as numerical representations at different scales. Many climate models are available, with variations in terms of detail, levels of complexity, and climate assumptions, and each model has its strengths and weaknesses. Given the complexity of the system being modeled, projected impacts from climate change should not be viewed as explicit predictions, but as a plausible future condition within a range of scenarios. Projects at the North American Regional Climate Change Assessment Program (NARCCAP), National Center for Atmospheric Research (NCAR), University of Colorado at Boulder (UC-Boulder), and more are underway to enhance climate modeling to better represent the mountainous terrain and seasonal weather patterns that affect southwest Colorado's climate.

Temperature

Southwestern Colorado has warmed about 1.1 degree Celsius (˚C) (2.0 degrees Fahrenheit [˚F]) in the last three decades (i.e., 1977 to 2007). This rate of warming is the same as for western Colorado, but greater than the western United States, or any other region of the United States except Alaska. Temperatures are likely to increase by an additional 0.8˚C to 1.9˚C (1.5˚F to 3.5˚F) by 2025 and 1.4˚C to 3.1˚C (2.5˚F to 5.5˚F) by 2050. Summers are projected to warm more than winters. By 2050, typical average monthly temperatures in the summer are projected to be as warm as or warmer than the hottest 10 percent (%) of summers from 1950 to 1999. The climate of the mountains is projected to migrate upward in elevation and the climate of the desert Southwest to progress up into the valleys.

Precipitation

The San Juan Mountain region is difficult to model for precipitation because of complex topography and natural variability in precipitation patterns. Precipitaion projections for the region based on general circulation models (GCMs) are not in consensus on the direction of change in amounts of precipitation, particularly during the monsoon season. Some studies indicate that annual precipitation will decrease slightly, while others project an increase in the winter. Even if the amount of annual precipitation does not change, however, more precipitation may fall as rain, compared to current conditions.

Some models project more variable precipitation patterns with more frequent extreme events.

Evaporation and Evapotranspiration

As a result of increased temperatures, the rate at which water is evaporated from water bodies, soil, and vegetation is very likely to increase. This will make the environment drier even if precipitation stays the same.

Downscaled Future Projections for San Juan Mountain Region

Downscaled climate models that were developed specially for southwestern Colorado provide projections that are based on local data and finer resolution of topography. The models, developed by the National Oceanic and Atmospheric Administration (NOAA) and the University of Colorado—Boulder, project increases in average maximum (daytime) and minimum (nighttime) temperatures in the San Juan region by the middle of the 21st century. Generally the daytime temperatures are predicted to increase an excess of 2.2˚C (4˚F), with potentially higher temperatures in the summer season. Several models show a large nighttime warming (up to 3.3˚C [6˚F]) during winter, particularly at elevations below 8,000 feet. While GCMS vary in their projections, there is a broad consensus among the regional climate models that southwestern Colorado can expect: near normal precipitation or a slight increase during December and January, and near normal precipitation or a decrease from April to June. There is a higher level of uncertainty regarding potential changes in precipitation in July and August, where regional models and GCMS differ in their projections.

Snowpack and Streamflow

Warming temperatures are projected to have significant effects on snowpack, timing of snowmelt, and streamflow even without a decrease in precipitation. From 1978 to 2004, snowmelt shifted about two weeks earlier in western

Colorado. It is likely that in the future, snowpack will continue to melt earlier and peak streamflow will occur earlier in the spring. Decreases are projected in snow water equivalent (SWE), or the amount of water contained in snowpack. The decreases are expected to be greater at elevations below 8,200 feet (i.e., a 20% to 60% reduction of snowpack by the period 2040 to 2069). Above 8,200 feet, the snowpack is anticipated to decrease by 10% to 20%.

Dust on Snow

Dust deposition on the snowpack decreases the albedo (the fraction of light reflected from an object) of snow, resulting in early snowmelt timing and runoff. A dominant source of dust for the San Juan Mountain region is the Colorado Plateau, where warming temperatures, livestock grazing, off-road vehicle use, and military training activities are all contributing to increased aeolian transport of dust. Biological consequences of early snowmelt from dust deposition include delayed phenology that leads to synchronization of plant growth and flowering, altered species interactions, and landscape-scale gene flow via pollination. Changes in timing of snowmelt and runoff affect reservoir operations, water rights, flood risk, wildfire risk, and forest ecology. Also, dust storms can cause nutrient loading where deposition occurs.

Ecosystems

Climate change can influence plant communities directly, by changing temperature and moisture regimes, or indirectly, by changing disturbance regimes like wildfire or insect and disease outbreaks. Increases in temperature and carbon dioxide, and decreases in precipitation, are projected to cause local extinction or shifts in the distribution of plant species. Trends of warming and drying are projected to continue in the West, and it is predicted that by the end of the century, only half of the current climate regimes and their associated plant communities will be maintained.

Current literature does not agree on the direction of change in sagebrush shrublands. Some predict that sagebrush communities will increase in area based on predicted warmer temperatures; others predict a decrease due to land-use activites, expansion of pinyon-juniper and ponderosa pine into sagebrush habitat, and nonnative plant invasion. Gambel oak communities are predicted to increase in range and elevation.

Piñon-juniper woodlands have been known to move up and down along elevational and latitudinal gradients as a result of climate variation. Model predictions of the future range of piñon-juniper woodlands vary depending on whether elevated CO_2 levels are included.

Recent general changes in forests include increases in smaller-diameter trees, reductions in large, old growth trees, alterations in species composition, and range expansions upward in elevation. Changes in forests are related to climate as well as land-use practices such as livestock grazing, forestry practices, and fire suppression over the past century. Predictions for forests based on bioclimatic modeling suggest that ponderosa pine forests will decrease in area, Douglas-fir forests will not change, and Engelmann spruce will be drastically reduced. An important factor in future composition and distribution of forests will be drought tolerance. Drought, along with other factors, has been associated with the decline of aspen forests in lower elevation habitats. Recent outbreaks of insects in Colorado are attributable to a combination of climatic factors, and have occurred at higher elevations and latitudes than previously recorded. Future warmer temperatures are predicted to lead to similar and possibly more intensive outbreaks than those experienced in the last decade.

In alpine ecosystems, the changes have been and will continue to be range expansions and contractions of herbaceous species and overall geographic contraction of the alpine environment. Alpine obligates that are endemic to a small geographic area have few options for movement, and are at risk of regional extinction. Alpine systems may also be impacted by tree encroachment into meadows, which is favored by milder winters, reduced snowpack, and longer growing seasons.

Long-term persistence of fens and wet meadows can be impacted by changes in summer temperatures, length of growing season, annual snowpack snow water equivalent, timing of snowmelt, and alterations in the frequency and strength on monsoon rainstorms. While fens do rely on groundwater, surface water and snowmelt are important elements of the hydrologic regime. Projections of drier conditions in the future indicate that water table depths in fens and wet meadows will be low for longer periods of time during growing season, and may result in drying, decomposition, and vegetation shifts in these ecosystems.

The watershed hydrology and population structures of critical riparian corridors may be affected by earlier spring peak discharge in high elevation streams, and changes in the timing, type, and amount of precipitation. Lowering of water tables along rivers and increased salinization of riverbanks from human activities has aided the spread of invasive tamarisk, and climate models predict that tamarisk will increase in range over the next century.

Exotic plants in general may be better able to tolerate a wide range of climatic conditions and become more successful in the future. Some invasive species have also been shown to respond to increased levels of CO_2 with greater growth than natives. Predicted warming and drying will alter the invasive potential and range of many exotic species of concern, such as cheatgrass, spotted knapweed, yellow star-thistle, red brome, and leafy spurge.

Phenology of Flowering Plants

Across the western United States, snowmelt and plant flowering time for many species have come earlier in the past 40 years. Earlier flowering time can result in fewer flowers per plant, fewer plants flowering, and increased or decreased flowering period. Response to climate change varies by species, and individual response must be understood before predictions about communities can be made.

Early snowmelt affects early flowering species differently than late flowering species. Early flowering species cued to snowmelt timing are more affected in timing and duration of flowering than later flowering species which have different cues. However, early snowmelt and the subsequent reduced soil moisture later in the season can have negative impacts on both early and late flowering species. Early snowmelt can also result in disadvantages to subalpine species from possible reduced pollinator numbers or frost. A reduction in early-season flowers may reduce pollinators and therefore reduce pollination events for late-season flowers that share the same pollinator.

Although a species may have the ability to survive periods of variable climate, depending on time and evolution, decreases in flowering and seed set can have serious repercussions.

Wildfire

Western forests are at great risk of large-scale stand-replacing fires due to warmer temperatures and dry conditions. Climate can influence flammability of living and dead biomass seasonally, through early spring snowmelt and associated mid-summer drought. Drought at the time of, or recently after, bark-beetle outbreaks may also increase wildfire risk. Burn severity is influenced by pre-fire disturbance, species composition, stand structure, topography, and climate. Heterogeneity in a landscape increases its resilience to severe disturbances such as wildfire, so some systems are able to recover from wildfire, whereas others are likely to convert to non-forest systems. With climate change, landscapes may not be able to return to their pre-disturbance state if tree species and genotypes are outside of their climatic niche, pre-disturbance species are replaced with successional or vegetatively reproducing species, or tree seedlings are outcompeted by exotics or native herbaceous and grass species.

Carbon Storage

Currently, forests in the United States act as carbon sinks and take up between 9% and 19% of anthropogenic fossil fuel emissions each year. Although there is interest in retaining or increasing forests as carbon sinks, the implications of various management actions on carbon storage is not fully understood. Atmospheric carbon is taken up and stored in woody and leafy plant material, litter, duff, and soil. Carbon storage varies by vegetation type; forests and peat in fen wetlands generally store the most carbon, and croplands the least. Grasslands and shrublands, which store less carbon than forests, are still important ecosystems for carbon sequestration because of the large geographic area they cover. Forests can sequester carbon for centuries, but may take over 100 years to recover carbon stocks following a fire. Furthermore, forests are dynamic systems, and carbon storage following disturbance can vary over time. Forests that burn according to a natural fire cycle are better carbon sinks than those that have experienced fire suppression. Restoration of fuel and forest structure and a natural fire cycle may be more cost effective than suppressing catastrophic fires. However, prescription burns and thinning treatments are not necessarily useful in all ecosystems.

Wildlife

Ungulate species such as mule deer and elk may be negatively affected by exposure to cold from lack of snow insulation, or by warmer temperatures that may lead to overpopulation. Extirpations and extinctions of species associated with climate change have occurred throughout evolutionary history, when a species cannot change quickly enough, lacks mobility, or is at the limits of its distribution. However, current climate change is expected to exceed rates of global climate change known for mammalian history. Climate change can impact species directly through loss of habitat, heat or drought stress, exposure to cold following early snowmelt, alterations in food availability and quality, and number of competitors or predators. Species can be also be affected indirectly through alteration of environmental cues, increases in disease and pest organisms, alteration in interactions with other individuals or other species, and loss of genetic variation due to decreases in suitable habitat.

Ungulate species such as mule deer and elk may be negatively affected by harsh winters, or by warmer temperatures that may lead to overpopulation. Canada lynx may lose habitat following large-scale stand-replacing wildfires, which are predicted to increase as a result of climate-induced warming and drying. Small-mammal communities will be affected as vegetation communities shift upward because of climate warming and drying. The American pika, a high-elevation species, has limited upward movement from their habitat, and may be especially vulnerable to climate warming. Yellow-bellied marmots have altered the timing of their emergence from hibernation as a result of changes in temperature and snowpack, and those that emerge early may need to re-enter torpor or go without food for weeks. Chipmunks and golden-mantled ground squirrels are showing a similar trend of altering their hibernation phenology.

Impacts of climate change on bird species varies by species and habitat. Sage grouse may be negatively impacted by future drought in their habitat. White-tailed ptarmigan is a high-elevation species that has seen slowed growth and reduced population size from warmer winters. Songbirds and other species may undergo changes in timing of migration and breeding.

Reductions in snowpack and increased temperatures have led to earlier breeding in amphibian species. Drought affects amphibian species by drying of pools prior to the completion of metamorphosis, and by preventing movement between pools.

Colorado’s native fish are adapted to cycles of spring flooding, summer drought, and specific ranges of temperature and water quality. Climate change can affect fish communities directly, through temperature changes that can be fatal or have sublethal effects on growth and reproduction. Indirect effects of climate change on fish include changes in flow regimes, water quality, and susceptibility to disease. Wildfire, floods, and other disturbances that impact fish habitat may have a greater effect on fish populations than temperature changes alone. In some cases, warming water temperatures may increase habitat suitability, where water was previously too cold to be highly productive.

Insects and other pollinators are affected by changes in phenology of flowering plants. In the event of early or late emergence of pollinators, it is important that alternative flowering hosts be available. A mismatch in emergence of pollinators and flowering plants may be more detrimental to pollinators than to plants that have vegetative reproduction or seed banks to rely on.

Table E.S. 1: Summary of Projected Climate Changes for Southwestern Colorado, USA

Climate Variable	General Change Expected	Hypothesis of Change and Reference Period	Size of Projected Change Compared to Recent Changes	Information about Seasonal or Patterns of Change	Confidence	Source(s) and Context
Temperature	Increase	+1.5 to 3.5˚F (2025) +2.5 to 5.5˚F (2050) compared to 1950 –1999 +5.0 to 8.0˚F (2100) compared to 1970–2000	Projected warming much greater than warming observed during the 20th century. Climate of mountains projected to migrate upward in elevation and climate of desert Southwest to progress upward into valleys of the Western Slope.	More warming in summer than winter projected. Average monthly temperatures in the summer projected to be as warm as or warmer than the hottest 10% of summers from 1950 to 1999.	**High confidence** that southwestern Colorado will warm. Good agreement of models exists.	Ray et al. (2008) Projections derived from an ensemble of climate models from IPCC (2007b) simulating climate change with high and low greenhouse gas scenarios.
Annual Precipitation	Not much change, but could decrease slightly	0 to −10% (2050) compared to 1950–1999	Projected change is small compared to observed variability.	Greater decrease in precipitation could occur in the summer, with a slight increase in the winter.	**Low confidence.** Models do not agree on direction of change.	
Snowpack SWE*	Decrease	−10 to −20% at elevations >8,200 feet, but lower elevations could lose up to 70%. Across the Colorado River Basin, SWE on April 1: −13 to −15% (2025) −21 to −25% (2055) −29 to −38% (2085) compared to 1950–1999	Projected decrease is more than 20th century except for a period of rapid decline observed in SJMs (-40%) from 1995 to 2005.	Projected decline due to warming temperatures. Therefore most of the decline is projected for late spring in the higher elevations, although declines could occur earlier in the lower elevations.	**Moderate confidence** in snow pack**.** **High confidence** in reduction in spring SWE.	Eleven climate models and two emissions scenarios projected in Christensen and Lettenmaier (2006).
Timing of Snowmelt Runoff	Earlier	5 to 25 days earlier by 2040–2059 5 to 35 days earlier by 2080–2099 compared to 1951–1980	Shift to two to three weeks earlier snowmelt runoff compared to that observed from 1978 to 2004.		**Moderate to high confidence** in direction of change.	Clow (2010); IPCC (2008); Stewart et al. (2004); Ray et al. (2008)

Climate Variable	General Change Expected	Hypothesis of Change and Reference Period	Size of Projected Change Compared to Recent Changes	Information about Seasonal or Patterns of Change	Confidence	Source(s) and Context
Summer Soil Moisture	Decrease	−10 to −25% for lower elevations (2050) −30 to −50% for higher elevations (2050) compared to 1950–2000	Projected change is greater.	Soil moisture in April is projected to increase in high elevations by 20%–50% by 2050.	**Moderate to high confidence** in direction of change.	Eleven climate models and two emissions scenarios projected in Christensen and Lettenmaier (2006).
Annual Streamflow	Decrease	By 2050, the following is anticipated as compared to 1900–1970: −10 to −25% for Upper Colorado River Basin (including outside of Colorado) −5 to −10% for Rio Grande Basin (including outside of Colorado) −10 to −20% for Colorado statewide	Projected change is greater.	Earlier snowmelt streamflows may increase streamflow in late winter, while streamflows in summer after snowmelt are projected to decline.	**Moderate to high confidence** in direction of change. 75% model agreement in sign of change for Rio Grande Basin, 95% for Upper Colorado River Basin, and >90% for Colorado statewide.	Milly et al. (2005)

* SWE = Snow water equivalent: the amount of water stored in the snowpack.

Table E.S. 2: Hypotheses of Change for Biological Systems

General Change Expected	Specific Change Projected Over the Next Century	Observed Changes	Information about Seasonal or Patterns of Change	Confidence	Source(s) and Context
Insect Outbreaks					
Increase in area affected	Increase in drought, stressed trees, overwintering larvae, and beetle outbreaks	Unknown if current outbreak sizes are unprecedented; current outbreaks occurring at higher latitudes and higher elevations than previousrecords	Overwintering of beetle larvae in warmer winters increases chance of large outbreaks	**High** confidence –all references project this change	Hicke et al. 2006; Janetos et al. 2008; Kliejunas et al. 2008; Ryan et al. 2008
Stand-replacing Fire					
Increase in area affected and fire season	Early spring snowmelt and mid-summer drought, overstocking of small-diameter trees in some forests, and fire-enhancing exotic plant species all can lead to an increased wildfire risk	More and more severe fires in systems that previously did not support severe fire	Extension of fire season to year-round; increase in beetle outbreak and drought simultaneously could increase fire potential	**High** confidence –all references project this change	Aber et al. 2001; Bigler et al. 2005; Kulakowski and Veblen 2006, 2007; Liu et al. 2010; Westerling et al. 2006
Sagebrush					
Increase in area based on climate models; **decrease** due to exotic species and fire	Increase in elevation; loss of populations on edge of distribution; loss to catastrophic fire could be high	Current sagebrush fire return intervals are 20 times shorter than historic intervals and fires are more severe	Historically expanded range during warm climatic periods; historically did not have high-severity fire and is slow to recover	**Low** confidence –climate literature suggests increases in range with warming, however habitat literature suggests causes of present decline are expected in the future	Harte and Shaw 1995; Keane et al. 2008; Perfors et al. 2003
Gambel Oak					
Increase in area based on climate models; **increase** following stand-replacing fire	Increase range; increase elevation by up to 300 meters	In recent catastrophic fires in the southwest, some ponderosa pine forests were replaced with re-sprouting shrub species	Following stand-replacing fire in ponderosa pine forests, gambel oak may re-sprout and shift from forest to shrubland	**Moderately High** confidence –single reference regarding climate plus high potential for post-fire succession	Neilson and Wullstein 1983; Rehfeldt et al. 2006

General Change Expected	Specific Change Projected Over the Next Century	Observed Changes	Information about Seasonal or Patterns of Change	Confidence	Source(s) and Context
Piñon-Juniper Woodlands					
Decrease or **increase in area** based on two models; **decrease** with expected future disturbances	Loss of twoneedle piñon and Utah juniper and reduced co-occurrence of these two species; increases in elevation of 500 and 100 meters, respectively	Landscape scale changes in this community occurred as a result of recent regional drought; more drought-tolerant junipers were also affected	Historically twoneedle piñon expanded in wet climatic periods; historic fires were stand-replacing and infrequent; fires today are within this range of variability	**Moderately High** confidence –climate literature suggests both an increase and a decrease; decrease also expected given future disturbance potential	Anderson and Feiler 2009; Keane et al. 2008; Rehfeldt et al. 2006; Romme et al. 2008
Ponderosa Pine Forests					
Decrease or **little change in area** based on climate models; **decrease** following stand-replacing fire	Elevation increase up to 500 meters	Recent increases in small-diameter trees and loss of large-diameter trees; chance for catastrophic fire is high	Historic fires were low-severity and frequent; restoration to fewer young trees and less accumulated fuel will decrease chance of stand-replacing fire and large, short-term (decades to 100 years) carbon losses	**Moderately High** confidence –climate literature suggests both a decrease and no change; potential for stand-replacing fires in untreated forests is high	Bachelet and Neilson 2000; Fulé 2008; Keane et al. 2008; Rehfeldt et al. 2006
Mixed-Conifer Forests					
No change in area based on climate models; **decrease** in dry mixed-conifer forests from stand-replacing fire	Elevation increase up to 500 meters	Recent increases in small-diameter trees and loss of large-diameter trees; chance for catastrophic fire in dry mixed-conifer is high	Historic fires in dry mixed-conifer were similar to ponderosa pine forests and similar restoration measures are suggested; historic fires in mesic mixed-conifer was variable	**Moderate** confidence –climate literature predicts no change, yet potential for stand-replacing fires in untreated dry mixed-conifer forests is high	Bachelet and Neilson 2000; Rehfeldt et al. 2006
Aspen Forests					
Decrease in area and **shift upslope** in elevation	Elevation increase up to 1,000 meters	Recent loss of 17% of the aspen cover type across Colorado (SAD) in marginal habitat at lower elevations, in open stands, on south- and west-facing slopes, and on droughty, poor soil sites	At higher elevations in spruce-fir forests, aspen is successional following stand-replacing disturbance; loss of aspen at lower elevations expected from drought	**Moderate** confidence – climate references suggest decrease; high potential for post-fire succession into this community type in higher elevation forests	Elliott and Baker 2004; Rehfeldt et al. 2009; Worrall et al. 2008; Worrall et al. in press

General Change Expected	Specific Change Projected Over the Next Century	Observed Changes	Information about Seasonal or Patterns of Change	Confidence	Source(s) and Context
Subalpine Forests					
Decrease in area based on climate models; **decrease** with expected future disturbances	Elevation increase up to 300 meters; changes in species composition (i.e., possible reduction in subalpine fir; increase in aspen	Recent increases in small-diameter trees and loss of large-diameter trees; infrequent catastrophic disturbance is normal in this system	Historic fires were stand-replacing and infrequent; fires today are within this range of variability	**High** confidence – climate literature suggests decrease; decrease also expected given future disturbance potential	Bigler et al. 2005; McKenzie et al. 2004; Noss et al. 2006; Rehfeldt et al. 2006; Westerling et al. 2006; Schoennagel et al. 2004
Alpine					
Decrease in area based on climate models; **decrease** from subalpine invasion	Elevation increase in alpine snow cover up to 400 meters	Shifts in species composition have been documented worldwide in recent decades	Subalpine plant establishment upslope reduces available alpine habitat; no movement upslope for alpine species is possible; alpine plants are gradually replaced by subalpine plants	**High** confidence – climate and habitat literature suggest decrease	Bachelet and Neilson 2000; Burkett and Kusler 2000; Diaz and Eischeid 2007; Dirnböck et al. 2003; Elliott and Baker 2004; Elliott and Kipfmueller 2010
Riparian					
Decrease in area based on climate models and from human disturbance (development, sedimentation, etc.)	Earlier spring runoff; reduced summer flows; more evapotranspiration; drier conditions; reduced obligate riparian species	Reduction in cottonwood, willow, and alder; changes in riparian plant community structure; loss of female cottonwood, willow, and boxelder; earlier spring runoff, lowering of water tables, and increased exotic species competition	Reduction in species health; alteration in plant community composition and structure	**High** confidence – climate literature suggests decrease; decrease also expected given future degradation potential	Cayan et al. 2001; Glenn and Nagler 2005; Hultine et al. 2007; Worrall, Adams, and Tharp in press; Worrall et al. 2008

General Change Expected	Specific Change Projected Over the Next Century	Observed Changes	Information about Seasonal or Patterns of Change	Confidence	Source(s) and Context
Wetlands: Fens and Wet Meadows					
Decrease	Drier conditions may result in long-term shifts from hydrophilic vegetation to mesic upland species (fens become wet meadows, wet meadows become dry meadows)	Currently roughly 300 fens in the San Juan Mountain region are in fair to poor condition; the first study of climate change effects on wetlands of the region began in 2010	San Juan Mountain wetlands are very sensitive to the amount and timing of snowmelt and summer monsoon rains	**Moderate** confidence– the first study is being conducted this summer	Chimner and Cooper 2003a,b; Chimner et al. 2010; Cooper et al. 1998
Exotic Plants					
Increase based on climate models; **increase** with expected future disturbances and globalization	Increase in high elevation and cold/drought–tolerant exotic species; alteration of current ranges for cheatgrass and spotted knapweed; increases in range for yellow star-thistle, red brome, and tamarisk	Recent increases in exotic plant richness and cover; changes in species range	Increase or decrease in current ranges based on climate and disturbance history (i.e., varies by species); increases in native species currently not considered invasive (i.e., ruderal species) or invasion by exotics currently not in the area (i.e., red brome)	**High** confidence – climate literature suggests increase in presence and quantity of invasive species	Bradley 2009; Bradley et al. 2009; Pauchard et al. 2009
Phenology					
Seasonal Changes in Plants and Animals Earlier Events	Earlier flowering; earlier hibernation emergence (i.e., marmots); earlier arrival by migratory species (i.e., robins); earlier breeding (i.e., boreal toad and boreal chorus frogs)	Recent flowering of many plants is earlier (i.e., *androsaceae, delphinium, heterotheca, lathyrus, mertensa*, etc.); many hibernating mammals are emerging earlier (i.e., marmots, chipmunks, golden-mantled ground squirrels); many migratory species are arriving earlier (i.e., robins); many species are shifting reproductive behavior (i.e., American coot, killdeer, American robin, boreal chorus frog)	Earlier flowering can lead to reduction in seed production; earlier emergence and earlier arrival at high elevations can lead to exposure to freezing temperatures and/or starvation; shifts in reproduction can lead to reduced reproductive success	**High** confidence –all references project this change	Blumstein 2009; Bruzgul et al. 2005; Cayan et al. 2001; Ditto and Frey 2007; Forrest and Thomson 2010; Forrest et al. 2010; Inouye et al. 2003; Inouye et al. 2000; Miller-Rushing and Inouye 2009; Muths et al. 2003; Saavedra et al. 2003;

General Change Expected	Specific Change Projected Over the Next Century	Observed Changes	Information about Seasonal or Patterns of Change	Confidence	Source(s) and Context
Carbon Storage					
Decrease / **increase**	Shifts in species composition during recovery from drought, beetle outbreaks, stand-replacing fires, etc. May decrease carbon storage; restoration, reforestation, and aforestation will increase carbon storage	Many forests in the United States are now net sinks for carbon owing to recovery from past heavy harvesting and possibly recent increased nitrogen deposition and CO_2 levels; conversion of forests to shrublands and grasslands reduces carbon storage	After a fire, *if* a forest replaces itself, there is little net loss or gain of carbon over the time scale of stand replacement. Over shorter time scales of 50 to 150 years, however, fire reduces carbon storage. Thinning treatments also reduce carbon storage if wood is not used in long-lived products, though this reduction may be far less than stand-replacing fire	**Low** confidence – literature projects increases and decreases	Dore et al. 2008; Morehouse et al. 2008; Reinhardt and Holsinger 2010; Ryan et al. 2010; Saleska et al. 2002; Scott et al. 2006; Wiedinmyer and Neff 2007
Large Mammals					
Decrease in some species	Changes in habitat will impact prey species and forage; increased threat of fire will have direct and indirect impacts; impacts to photoperiod-dependent species	In years of early spring onset, mobility and forage are increased for ungulates and can lead to overpopulation and starvation; loss of subalpine and alpine meadows to warming-induced tree invasion reduces habitat for large carnivores; loss of late-successional forests to drought-induced wildfire reduces habitat for Canada lynx	Long-lived and larger species dependent on photoperiod will find adaptation more difficult than short-lived, less dependent species	**Low to Moderate** confidence– climate literature predicts decreases in some species; too few species discussed in literature for overall conclusion	Bartmann and Bowden 1984; McKenzie et al. 2004; Moir and Huckaby 1994; Wang et al. 2001

General Change Expected	Specific Change Projected Over the Next Century	Observed Changes	Information about Seasonal or Patterns of Change	Confidence	Source(s) and Context
Small Mammals					
Decrease / increase in some species	Earlier emergence from hibernation; movement to higher elevations; impacts to photoperiod-dependent species; impacts of increased high-severity fire occurrence	Emergence from hibernation is earlier in marmots and other small mammals; upward movement of vegetation communities means upward movement of small mammals associated with these communities (red fox and Wyoming ground squirrel); some species cannot move any higher (i.e., American pika); reduction or fragmentation of habitat leads to loss of genetic variation; loss of late-successional forests to wildfire increases habitat for snowshoe hare; drought-induced reductions in predators can increase deer mice	Species already stressed, in decline, or on the margin of their range will be adversely affected; species unable to move short distances will face greatest challenges; long-lived and larger species dependent on photoperiod will find adaptation more difficult than short-lived, less dependent species	**Low to Moderate** confidence– climate literature predicts decreases and some increases depending on species; too few species discussed in literature for overall conclusion	Beever et al. 2003; Ditto and Frey 2007; Epstein 2001; David Inouye, pers. comm. July 23, 2010; Inouye et al. 2000; Lawler et al. 2010; Millar and Westfall 2010; U.S. Fish and Wildlife Service 2005
Birds					
Decrease in some species	Loss of habitat to fire and exotic plant invasion; loss of seed availability due to reduced reproduction in early-flowering herbaceous species; chance of starvation from earlier movement to high elevation grounds; earlier breeding and reduced nesting success	Earlier breeding observed in American robin, red-winged blackbirds, and other species; earlier arrival of American robins from their wintering grounds; decreased nest success in sage grouse; reduced population size in white-tailed ptarmigan	Species already stressed, in decline, or on the margin of their range will be adversely impacted; species unable to move short distances will face greatest challenges; long-lived and larger species dependent on photoperiod will find adaptation more difficult than short-lived, less dependent species	**Low to Moderate** confidence– climate literature predicts decreases in some species; too few species discussed in literature for overall conclusion	Aldridge et al. 2008; Inouye et al. 2000; Torti and Dunn 2005; Wang et al. 2002; Wang, Hobbs, Giesen, and Galbraith 2002

General Change Expected	Specific Change Projected Over the Next Century	Observed Changes	Information about Seasonal or Patterns of Change	Confidence	Source(s) and Context
Amphibians					
Decrease in some species	Earlier breeding; movement upward in elevation; increased stress and disease	Earlier breeding in many species including boreal toads and boreal chorus frogs; changes in male-female ratio of boreal toads; loss of wetland habitat causes declines in many amphibian species including boreal chorus frogs and tiger salamanders; movement up in elevation by the northern leopard frog; impacts to amphibian predators due to reductions in amphibian prey	Warmer temperatures and a drier environment will affect species directly and indirectly through increased stress levels, susceptibility to disease, reduced fecundity, and loss of food and habitat quality and quantity	**Low to Moderate** confidence- climate literature predicts decreases in some species; too few species discussed in literature for overall conclusion	Blaustein et al. 2001; Bruzgul et al. 2005; Corn 2003, 2005; Corn and Muths 2002; Di Rosa et al. 2006; Lawler et al. 2010; McMenamin et al. 2009; Muths et al. 2003
Fisheries					
Decrease / increase in some species	Reduced habitat (salmonids) / increased habitat (i.e., greenback cutthroat trout) depending on species; loss of shade and increased sedimentation from increased wildfire and flooding; movement of native coldwater fish to higher elevation; reduction in food sources	Reduced streamflow and increased water temperature; reduction or fragmentation of habitat leads to loss of genetic variation; food sources may be reduced in quantity or size (i.e., mayflies that emerge earlier in warmer springs have smaller body size and reduced reproduction success)	Warmer temperatures and a drier environment will impact species directly and indirectly through increased stress levels, susceptibility to disease, reduced fecundity, and loss of food and habitat quality and quantity	**Low to Moderate** confidence- climate literature predicts decreases and some increases depending on species; too few species discussed in literature for overall conclusion	Cooney et al. 2005; Harper and Peckarsky 2006; Keleher and Rahel 1996; Williams et al. 2009

Chapter 1: Introduction

1.1 Purpose of this Report

This report is a synthesis of the current state of the science regarding the physical and biological aspects of climate change that are important for evaluating impacts to the water resources, ecosystems, and human communities in southwestern Colorado. In some cases, scientific results are not available for southwestern Colorado. Therefore, we also rely on findings from studies at larger spatial scales or from other regions that have similarities that make them relevant to southwestern Colorado. Except in a few instances, we use results from studies published in peer-reviewed journals.

The scientific findings in this document are presented to identify critical data gaps and to support regional and local efforts towards climate adaptation. The document describes observed conditions and trends, modeling results, and projections of future conditions. It also incorporates a growing body of ecological knowledge that helps us to understand and interpret these historic trends, current conditions, and future projections.

Knowledge about climate change and its impacts is evolving. This is a living document and should be updated as the science progresses. In addition, some topics, such as climate impacts on human health and the built human environment, are beyond the scope of this report. We recommend that additional work be done to synthesize pertinent datasets and research results in these areas.

The audience intended for this report includes professional environmental managers and scientists, including federal, state, and local government agencies, non-governmental organizations, and colleges/universities. While we hope that it also is useful to the public-at-large, it is a fairly technical document and is not meant for a purely lay audience. A shorter version of this report will be released at a later data for non-scientific audiences.

This research review report is a companion document to the Climate, Water, and Forest Assessment reports that the Mountain Studies Institute is concurrently preparing. The first of these assessments is focused on La Plata County and others will be developed as funding allows. The assessment reports synthesize current water, forest, and economic datasets for a specific county or watershed and use the results to identify areas of vulnerability given the future climate scenario described in this research review. The assessment reports further begin to identify action strategies to address these vulnerabilities.

1.2 Understanding Complexity and Levels of Uncertainty

Projections of future climate involve two major areas of uncertainty:
1) uncertainty about future global emissions of greenhouse gases, and
2) uncertainty in our ability to simulate the Earth's climate system in a computer model. Future emissions of greenhouse gases are uncertain because emissions depend on how humans will act. The Intergovernmental Panel on Climate Change (IPCC) has described several different scenarios of future greenhouse emissions that reflect various amounts of demographic, socio-economic, and technology development (IPCC SRES 2000). These scenarios then serve as inputs to global climate models. Simulating the global climate system is extremely complex and involves a multitude of interactions and feedbacks among land, water, and the atmosphere. A series of such models, developed by research centers around the globe, vary in the way they model the global climate system, including the strength of various climate feedbacks like cloud cover and humidity. This series of models is used to project a range of plausible future climate scenarios (Figure 2-1).

In the mountain terrain of southwestern Colorado, temperature, precipitation, snowpack, streamflow, and other environmental factors are extremely variable over time and space (Rangwala and Miller 2010). Furthermore, elevation data only goes up to 11,000 feet (Ray et al. 2008) and therefore are not adequate in the highest elevations of the San Juan Mountains. At the global scale, however, the combination of data from thousands of climate stations reduces much of the inter-annual variability. For example, a warmer-than-average year in one region of the globe can compensate for a cooler-than-average year somewhere else. Therefore, the effects of human-caused climate change are detected more easily at this large scale (Millar 2007). As the spatial scale of analysis decreases, however, variability over time increases. Since the complexity of terrain (i.e., mountains and valleys) is critical in driving local climate, climate projections at smaller spatial scales are even more difficult to estimate.

The effect of climate on biological systems can be even more complex than climate itself. The direct impacts of a changing climate, such as warming temperatures and drier conditions, interact with other agents of change, such as invasive species and wildfire occurrence. Human actions, such as land-use change and land management practices, also play an important role For example, while our ecological knowledge about how increasing temperatures affect a certain ecosystem may be fairly advanced, the combination of different factors (such as rising temperatures combined with variable grazing practices and the potential spread of invasive plants) may result in much less certainty about the future.

Many people understand that economists cannot be 100% certain about the future of our economy—and that they can’t even be 100% certain about the causes of change once it’s happened. Climate scientists face similar difficulties. Understanding certainty is an important part of climate science. If many models agree on the direction and relative magnitude of environmental change, then the level of certainty and the likelihood of the result increase. Similarly, if other types of studies conducted by different scientists agree on a result and that result passes the scrutiny of the scientific community, the level of its certainty, or likelihood, increases.

Given the complexity of earth and human systems and our inability to perfectly simulate them, projected impacts from climate change should be viewed not as exact predictions, but as plausible future conditions that can help us analyze our vulnerability and risk. Thus, our best estimates of what could happen, with some degree of likelihood, can serve as a useful guide for a range of adaptation measures.

Major climate change assessments like those conducted by the IPCC and the U.S. Climate Change Science Program involve hundreds to thousands of experts (IPCC, various). The major assessments use specific language to communicate level of agreement and uncertainty across these perspectives. IPCC terminology indicates the assessed likelihood, using expert judgment, of an outcome or result. For example, for the IPCC, *very likely* refers to a likelihood of >90%, and *likely* means a likelihood of >66% to 90%. We refer to these likelihood estimates as well as using the terminology of *high*, *moderate*, and *low confidence*. High confidence refers to strong agreement among results. Moderate confidence indicates some agreement. Low confidence describes little agreement.

1.3 Climate Downscaling Approaches

To prepare for future climate change, land and water resource managers are interested in knowing how climate variables—such as surface air temperature and precipitation—will be different in the future relative to the present. Projections of such changes are provided by the general circulation models, or GCMs. However, GCMs produce climate output at spatial scales of 150–600 km (100–400 miles). This spatial resolution is too coarse to address the local issues that often concern resource managers and is too coarse for most models predicting environmental impacts, which may require information at scales of 1–50 km (1–30 miles). Downscaling techniques are therefore used to generate climate output at finer spatial scale from GCM output (Ray et al 20008). Statistical and dynamical downscaling, or a combination of the two, are the current approaches to reducing the scale of climate models to a smaller region.

Downscaling Methods: Strengths and Limitation

Statistical downscaling establishes statistical relationships between GCM output and the observed local climate to correct for the GCM biases in the local climate. The GCM projection is then fed into the statistical model to estimate future changes to the local climate. Until recently, this has been the predominant downscaling method. One of the main advantages of statistical downscaling is that it is computationally inexpensive and it can be done on a personal computer. Limitations include the fact that GCM outputs are typically archived at monthly time steps, which forces the user to develop statistics to transform monthly values into the daily or sub-daily values that are typically needed for impact models. In addition, users are forced to derive other climate variables of interest that may not be archived. Furthermore, there is a general understanding that statistical relationships among monthly averages may be more robust than for daily values, at least when one considers the large spatial scales of the GCMs. Finally, although a statistical relationship may be valid for the current climate regime, it may not remain valid into the future when different atmospheric and oceanic forcing conditions or feedback processes may be at work. Therefore, statistical downscaling procedures may produce artifacts, in certain cases, such that the climate signal in the downscaled data is not commensurate with the original signal from the GCMs.

In contrast to statistical downscaling, dynamical downscaling typically involves nesting a regional climate model (RCM) inside a GCM over the region of interest. The GCM then provides meteorological conditions, typically every six hours, at the boundaries of the RCM. RCMs provide better topographical representations than GCMs, which is especially important for mountainous regions, and they also provide better local- to regional-scale atmospheric dynamics that may, for example, improve the simulation of summertime weather and rain shadows. Dynamically downscaled projections can be produced at a variety of spatial scales, sometimes as small as 1 km. However, generally these efforts are limited to the 25–50 km range because of computational limitations. One of the main advantages of dynamical downscaling is that it is more physically based, and therefore has fewer assumptions about the future climate. In addition, a greater number of output climate variables from these RCMs, relevant to resource managers, are being typically archived at sub-daily time scales. A major disadvantage of dynamical downscaling is that it is computationally expensive and can be a very lengthy process. In addition, if impact modelers require data at a finer spatial resolution than is provided by the RCMs, they would still need to make use of statistical methods to further downscale the data.

When making use of downscaled climate projections, it is preferable to consider a range of projections—in the case of statistical downscaling, a range of GCM projections, and in the case of dynamical downscaling, projections produced by various RCM–GCM combinations. Each GCM and RCM varies in how it represents atmosphere, ocean, and/or land interactions. In addition, there are significant differences among RCMs in how they interact with GCMs. Because each model has its strengths and weaknesses, there is generally no one "best" GCM or RCM–GCM combination.

Models as Tools

Throughout this summary, predictive models are referenced to aid in our understanding of future climate change impacts. It is important to remember that even the most advanced computer models have limitations. Predictions should be considered estimations to gain insight into the range of possibilities, but not facts (Bachelet and Neilson 2000; Millar et al. 2007). Models cannot possibly incorporate all factors impacting ecosystems and commonly do not include estimations for disturbances such as grazing impacts, exotic plant invasion, disease outbreak, or land-use changes (Bachelet and Neilson 2000; Baron et al. 2009). Neither do models incorporate biological details such as plant recruitment, microclimate and microtopography, or forest subtypes (e.g., warm dry vs. cool moist mixed-conifer [e.g., Korb et al. 2007] or persistent piñon-juniper woodlands vs. piñon-juniper savannas vs. wooded shrublands [e.g., Romme et al. 2008]). Models, like long-term monitoring, defining reference conditions, or determining thresholds, are tools to 1) aid in our understanding of past, present, and future vegetation response to climate, and 2) aid managers in making more informed management decisions (Millar et al. 2007). Current and future management must be based on science with frequent reassessment of treatment priorities that emphasize resistance and resilience, as past experiences may not be a valid guide into new and rapidly changing conditions (Baron et al. 2009; Millar et al. 2007).

Chapter 2: Temperature, Precipitation, Moisture, and Streamflow

This chapter provides an overview of observed and projected changes in climate and water resources at various scales: global, western United States, Colorado, and southwestern Colorado.

2.1 The Global Context

"Warming of the climate system is unequivocal, as is now evident from observations of increases in global average air and ocean temperatures, widespread melting of snow and ice and rising global average sea level. Observational evidence from all continents and most oceans shows that many natural systems are being affected by regional climate changes, particularly temperature increases. Most of the observed increase in global average temperatures since the mid-20th century is *very likely* due to the observed increase in anthropogenic GHG (greenhouse gas) concentrations." *—excerpted from the IPCC Fourth Assessment Report (AR4) Synthesis—Summary for Policy Makers, 11/17/2007.*

Global Observations

Global greenhouse gas emissions due to human activities have grown since pre-industrial times, with an increase of 70% between 1970 and 2004. The greatest human-caused contribution to the increased greenhouse effect is the large amount of carbon dioxide produced by burning of carbon-based fossil fuels such as coal, oil and natural gas (IPCC 2007a). Other human-caused sources of greenhouse gases include industry, deforestation, and agriculture. Until 2000, the rate of CO_2 emissions per unit energy produced was declining, but that has reversed since 2000 (IPCC 2007b; Raupach et al. 2007).

Worldwide, temperature is rising, and rates of warming are increasing. Global mean surface temperatures have risen 0.7ºC (~1.3ºF) from 1906 to 2005. The rate of warming over the last 50 years was almost double that over the last 100 years. Nine of the 10 warmest years on record occurred from 1995 to 2004. This same decade is considered the warmest of the past 1,000 to 2,000 years (Backlund et al. 2008).

Of the more than 29,000 observational data series, from 75 studies, that show significant change in many physical and biological systems, more than 89% are consistent with the direction of change expected as a response to warming (IPCC 2007b). Global average sea level has risen about 8 cm since 1961 (at an average rate of 1.8 mm/yr) and almost 4 cm since 1993 (at 3.1 mm/yr). Annual average Arctic sea ice extent has shrunk by 2.7% per decade since 1978, with larger decreases in summer of 7.4% per decade. Mountain glaciers and snow cover on average have declined in both hemispheres. In northern regions of the globe, more precipitation now comes as rainfall, and snowpacks are melting earlier. Globally, the area affected by drought has *likely* increased since the 1970s. It is *very likely* that over the past 50 years: cold days, cold nights, and frosts have become less frequent over most land areas, and hot days and hot nights have become more frequent. It is *likely* that: heat waves have become more frequent over most land areas, and the frequency of heavy precipitation events has increased over most areas. Since the 1950s, the northern hemisphere has had earlier spring onset, and a longer frost-free season (IPCC 2007a; National Academy of Sciences 2007).

Global Projections for the Future

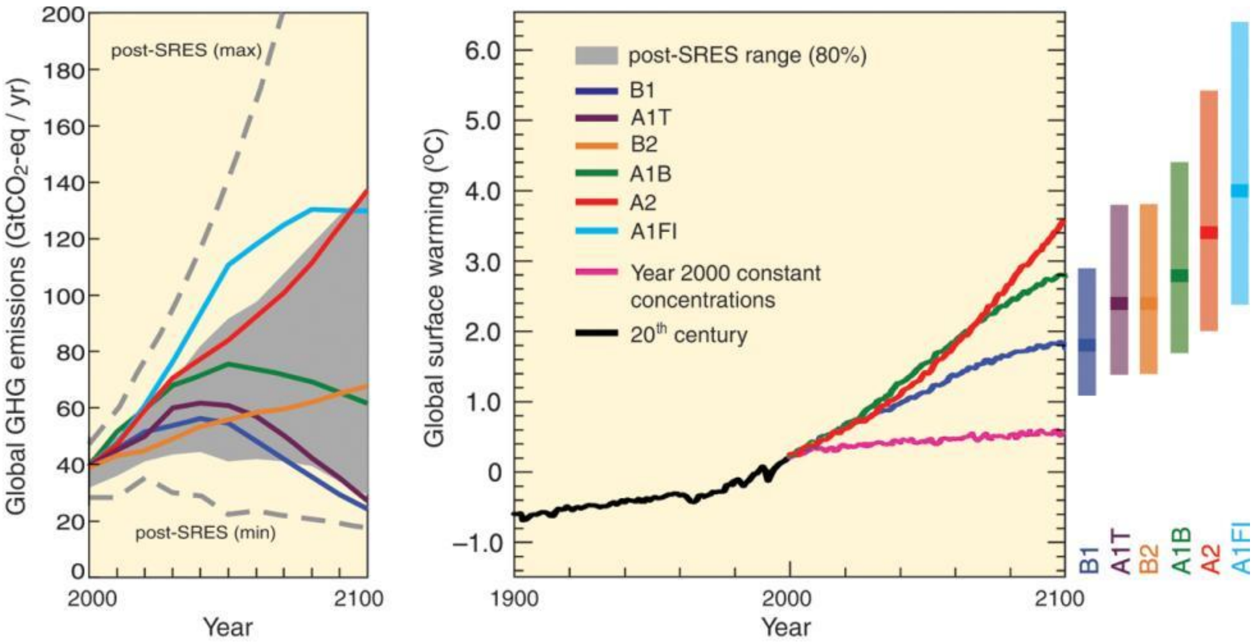


Figure 2-1 Global greenhouse gas emission scenarios

Left panel shows global greenhouse gas (GHG) emission scenarios from 2000 to 2100, and the right panel illustrates 20th-century observed trend in global surface warming with the projected warming trajectories from the GHG emissions scenarios. These projections are the average of many global circulation models. The bars on the far right of the figure indicate the best estimate (solid line within each bar) and the likely range assessed for the six scenarios at 2090–2099. All temperatures are relative to the baseline period 1980–1999. (From IPCC [2007a]: AR4 Synthesis Report, Figure SPM.5)

There is high agreement and much evidence that with current climate change mitigation policies and related sustainable development practices, global greenhouse gas emissions will continue to grow over the next few decades. An increase of global greenhouse gas emissions by 25% to 90% (carbon dioxide equivalents) is projected between 2000 and 2030 (IPCC 2007a). Currently, we are exceeding the A2 emissions scenario (see Figure 2-1), which was thought to be a fairly accelerated rate of change when the IPCC developed this scenario two decades ago.

Sophisticated computer models of the climate system are used to project global patterns of future climatic conditions under various scenarios of greenhouse gas concentrations (see Figure 2-1). Over the next 20 years, a warming of about 0.2˚C (~0.7˚F) is projected for a range of emissions scenarios. Even if the concentrations of all greenhouse gases had been kept constant at year 2000 levels over this 20-year period, a further warming of about 0.1˚C (~0.4˚F) per decade would be expected. Afterwards, temperature projections increasingly depend on specific emissions scenarios. Based on a range of low to high emissions scenarios, projections for 2090–2099, show that global mean temperature will have increased 1.8˚C to 4.0˚C (~3.2˚F to 7.1˚F) relative to 1980–1999. Warming is expected to be greatest over land, especially the high northern latitudes (IPCC 2007a, p.23).

Computer models also are used to estimate changes to physical and biological systems. These projections include: *very likely* continued contraction of snow covered area, increases in thaw depth over most permafrost regions, and decrease in sea ice extent; *very likely* increase in frequency of hot extremes, heat waves and heavy precipitation events; and *very likely* precipitation increases in high latitudes and *likely* decreases in most subtropical land regions. There is *high confidence* that by mid-century, annual river runoff and water availability are projected to increase at high latitudes (and in some tropical wet areas) and decrease in some dry regions in the mid-latitudes and tropics. There is also *high confidence* that many semi-arid areas (including the western United States) will suffer a decrease in water resources due to climate change (IPCC 2007a).

2.2 Western North America

Temperature

The rate of warming is not the same everywhere across the planet (IPCC 2007b). The western part of the United States is warming faster than the global average. Over almost the entire western United States, there have been increases in both cool-season and warm-season temperatures between 1916 and 2003 (Hamlet et al. 2007; Mote et al. 2005). Although the rate of change varies with location and the time period examined, average warming has been about 1˚C (1.8˚F) per century over the 1916 to 2003 time period (Hamlet et al. 2007). The rate of increase from 1947 to 2003 is roughly double that of the longer period from 1916 to 2003, and much of the observed warming has occurred from about 1975 to present.

Annual mean temperature increases in the western United States over the next century are expected to range from 2˚C to 4˚C (~3.6˚F to 5.4˚F) at the low end of the uncertainty range, to 4˚C to 6˚C (~8.9˚F to 10.7˚F) at the upper end of the uncertainty range (IPCC 2007b; Miles et al. 2007). Beyond mid-century, future warming is dependent on greenhouse gas emission levels in the next few decades, which are dependent on human activities.

Precipitation, Snowpack, Moisture, and Streamflow

There have been increases in winter (November–March) precipitation since 1930 over much of the western United States, although patterns are variable in different regions (Figure 2-2; Mote et al. 2005). In many places in the western United States, increased temperatures (Figure 2-3) have led to more precipitation falling as rain rather than snow at lower elevations, earlier snowmelt and snowmelt-driven streamflow (Marks and Seyfried, 2011; Hamlet et al. 2007; Stewart et al. 2005), and reduced spring snowpack (Barnett et al. 2008; Mote 2003; Mote et al. 2005). Projected precipitation changes in the West over the next century are more uncertain than temperature changes, as models do not agree on the direction of change. In addition, projected changes in precipitation patterns differ by region. Summer rains in the Southwest may intensify and shift to the north. Winter rains might decrease in the Southwest but increase in the northern half of the West (Salathé 2006). Inter-annual and inter-decadal variability via El Niño–La Niña cycles may also intensify (Timmermann et al. 1999), producing extreme winter events in both the Southwest and the Northwest.

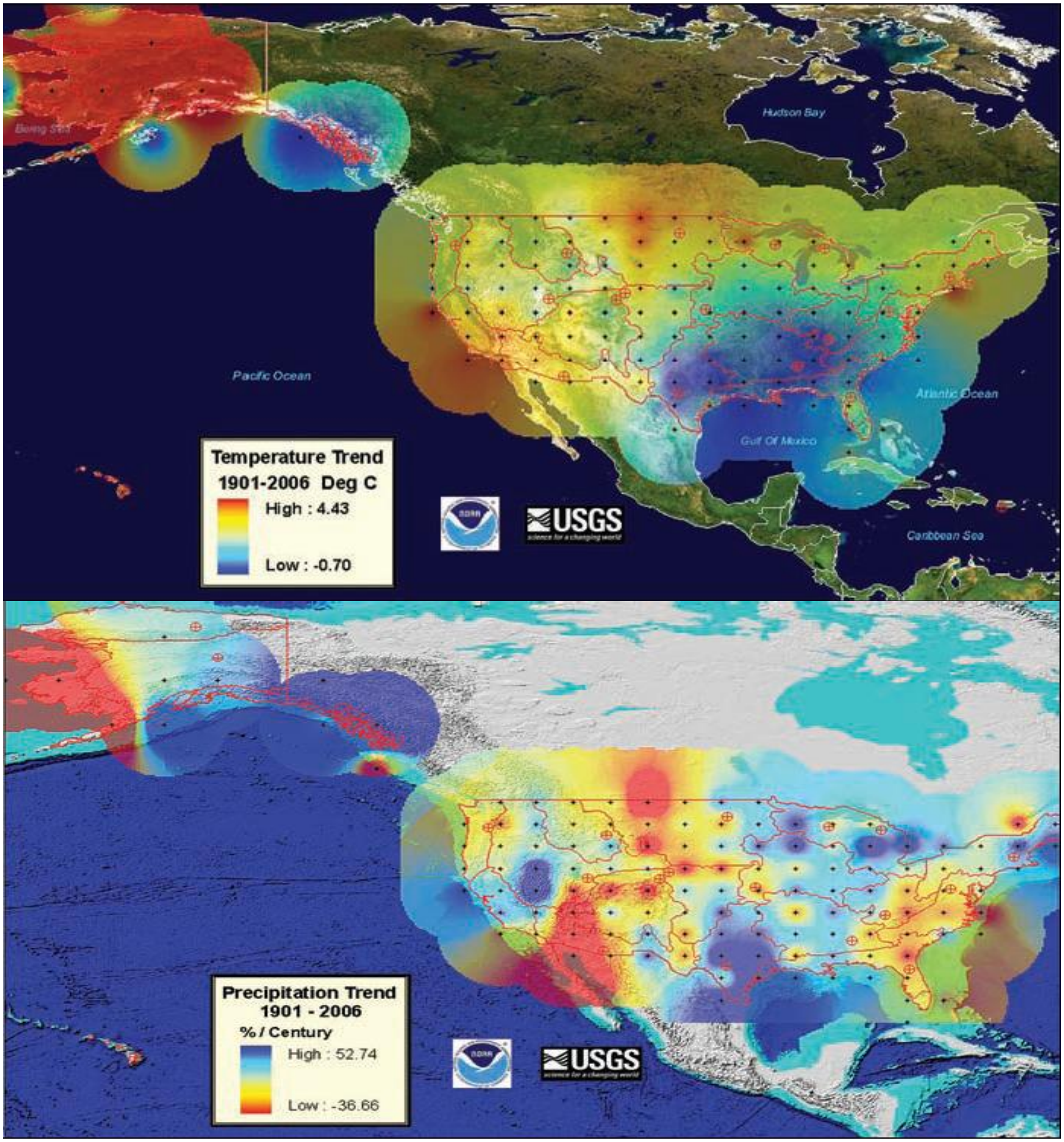
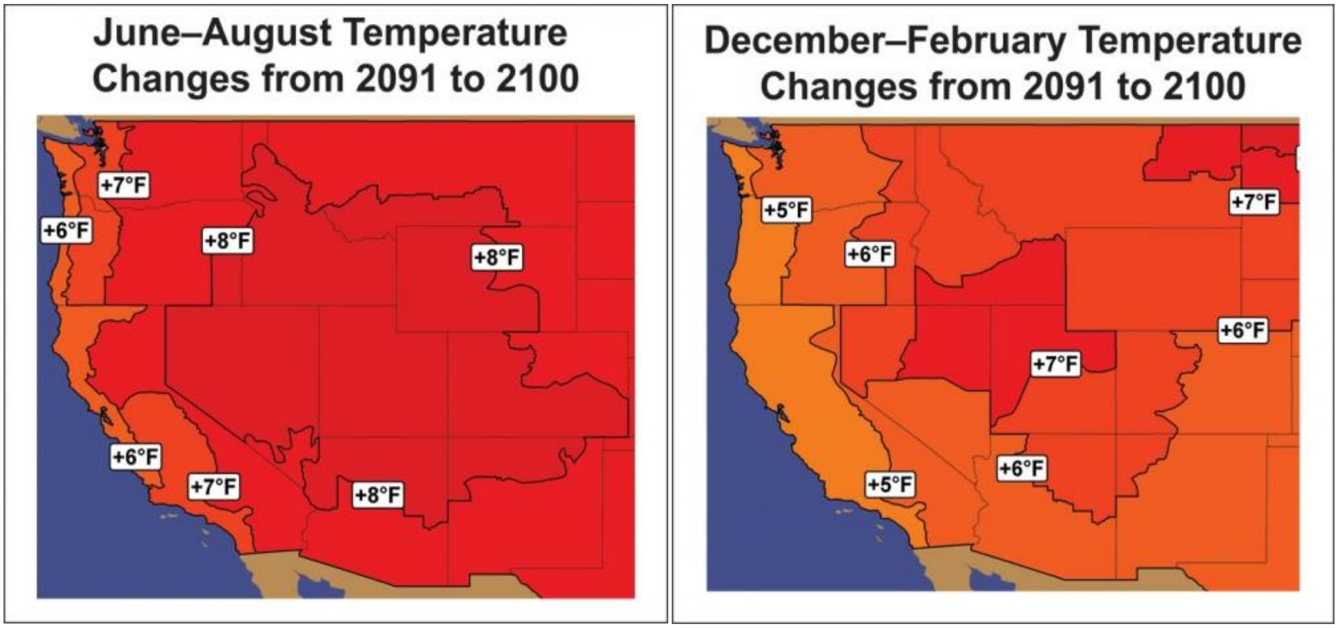


Figure 2-2 Trends in temperature and precipitations over the past century

Upper Panel: Mapped trends in temperature across the lower 48 states and Alaska (1901–2006). These data, which show the regional pattern of warming in the United States, are averaged from weather stations across the country using stations that have as complete, consistent, and high quality records as can be found. Bottom Panel: Precipitation changes over the past century from the same weather stations as for temperature (1901–2006). The changes are shown as percentage changes from the long-term average. Figure is courtesy of NOAA's National Climate Data Center and the U.S. Geological Survey.


June–August Temperature Changes from 2091 to 2100

December–February Temperature Changes from 2091 to 2100

Figure 2-3 Projected summer and winter temperature changes for the last decade of this century

Using an "ensemble" of 18 global climate models and the moderate "A1B" emissions scenario, projected warming in the Intermountain West and Southwest at the end of this century is approximately 7-8˚F for summer (June-August) and 5-7˚F for winter (December-February), relative to average temperatures 1971-2000. Figure from the NOAA Earth System Research Laboratory (ESRL).

Warming and changes in the form, timing, and amount of precipitation will be *very likely* to lead to earlier melting and significant reductions in snowpack in the western mountains by 2050 (IPCC 2008). For the mountainous regions of the western United States, snowmelt provides approximately 70% of annual streamflow (Mote et al. 2005). Projections for mountain snowmelt-dominated watersheds in the West suggest an increase in winter and early spring flows (raising flooding potential) and a substantial decrease in summer streamflows (Mote et al. 2005; Knowles et al, 2006; IPCC 2008; Luce and Holden 2009). This reduction in summer streamflow could have major implications for fisheries, wildlife, water supply, and agriculture, particularly in drier regions. The current and expected future trends in hydrology suggest a coming crisis in water supply for the western United States (Barnett et al. 2008).

Increased temperatures also may result in decreased soil moisture in arid regions of the western United States (Miles et al. 2007). Changes in soil moisture are projected to differ by region, however. In the Pacific Northwest, it is projected that mountainous regions will have 80% or less of historical average soil moisture, while arid regions will have 90% to 95% of historical soil moisture (Miles et al. 2007).

While there exists some uncertainty over precipitation trends, warmer temperatures and higher rates of evapotranspiration with climate change in some areas, such as the southwest United States, *will likely* lead to increased drought frequency and severity. Overall, drought-affected areas are projected to increase in extent (IPCC 2007a). Although increased temperatures *will likely* lead to decreased runoff in some areas, increased frequency of heavy precipitation events *will likely* lead to increased flood risk in many regions (IPCC 2007). Earlier snowmelt and runoff owing to increased temperatures could also lead to increased winter and spring flooding.

2.3 Colorado

Temperature

Colorado's climate since 1930 shows a warm period in the 1930s and 1950s, a cool period through the 1960s and 1970s, and a consistent warming trend since about 1970. Statewide, mean surface temperatures in Colorado have increased an average of about 1˚C (~2˚F) over 30 years from 1977 to 2006. Because of a cool period in the 1960s and 1970s, the 50-year trend from 1957 to 2006 also shows a similar 1˚C (~2˚F) warming. All parts of Colorado except for the southeastern corner warmed since 1977. Minimum temperatures show more warming, on average, than maximum temperatures. Also, spring temperatures tended to warm the most, and winter warming greater than summer or fall (Ray et al. 2008).

Computer climate models project that Colorado will have an annual mean warming of 0.8°C to 2.0°C (~1.5°F to 3.5°F) by 2025, relative to 1950–1999 average temperatures, and 1.4°C to 3.1°C (~2.5°F to 5.5°F) by 2050 (Figure 2-4). Summers are projected to warm more than winters, with typical average monthly temperatures in the summer as warm as or warmer than the hottest 10% of summers from 1950 to 1999. In general, the climate of the mountains is projected to migrate upward in elevation and the climate of the desert Southwest to progress up into the valleys of the Western Slope (Ray et al. 2008).

Precipitation, Moisture, and Streamflow

Owing to high variability in rain and snowfall, no consistent long-term trends in annual precipitation have been detected in Colorado. Elsewhere in the western United States, a widespread and large increase in the proportion of precipitation falling as rain rather than snow, and a reduction in the amount of snow water equivalent (SWE) have been observed (Mote et al 2005; Clow 2010; Luce and Holden 2009). Most of this decline in snowpack has occurred below about 8,200 feet elevation. Most of Colorado's snowpack occurs above 8,200 feet, however, and changes in proportion of rain to snow and SWE have been small and less significant compared to much of the West. Snowmelt-driven increases in streamflow have shifted an average of two weeks earlier in Colorado in the period from 1978 to 2004 (Clow 2010). Streamflows in Colorado are heavily dependent on precipitation stored as snow.

The lowest five-year period of Colorado River natural flow since records began in the late 1800s occurred during the 2000 to 2004 drought period. Tree-ring records suggest that more frequent and severe droughts have occurred throughout the West (including Colorado) in the last 1,000 years, however (Ray et al. 2008; Lukas 2008).

Colorado is located between the Southwest, which is projected to experience a drier climate in the future, and the more northern areas that are projected to

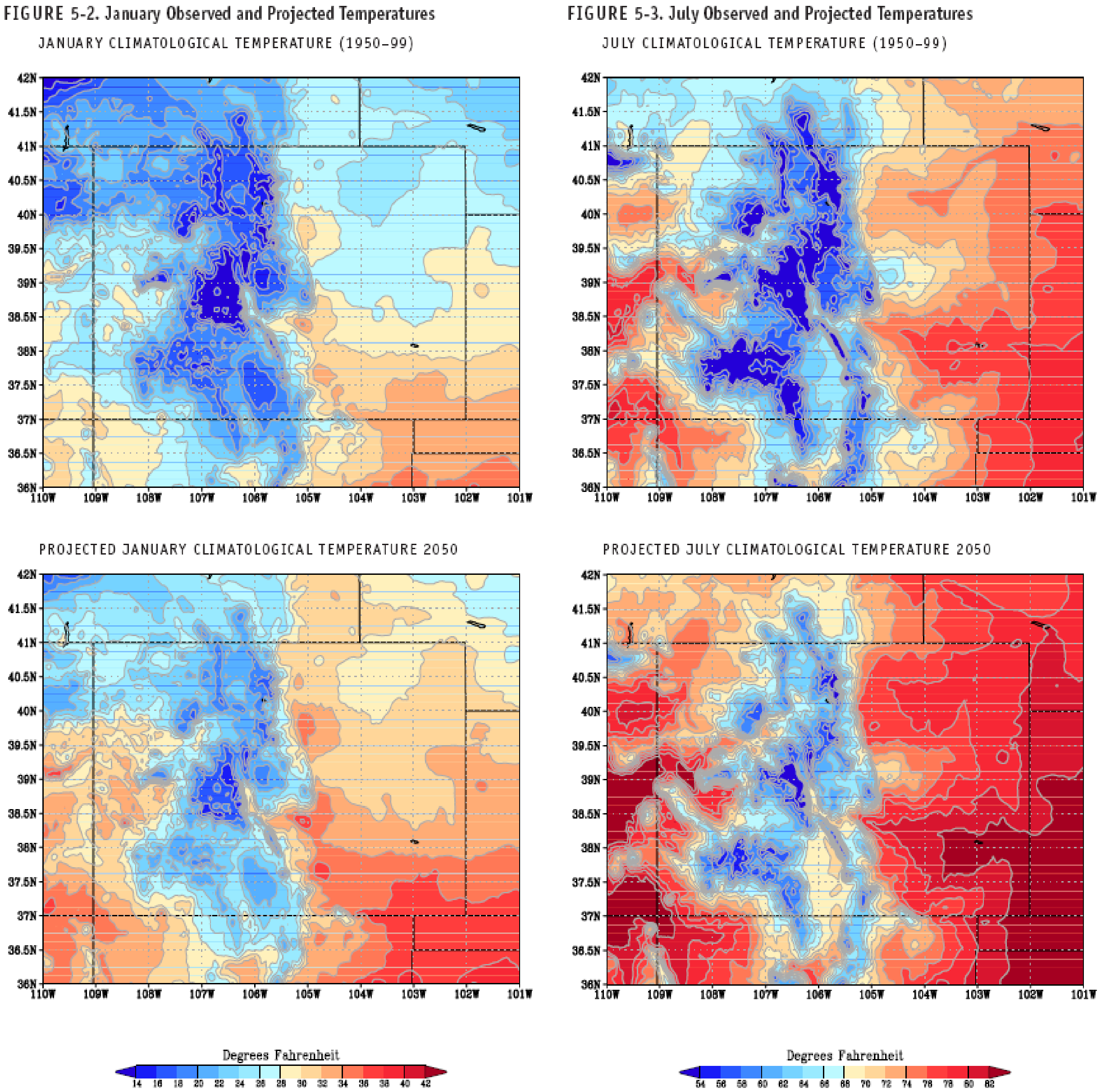


FIGURE 5-2. January Observed and Projected Temperatures

FIGURE 5-3. July Observed and Projected Temperatures

Figure 2-4 Observed and projected future temperatures for Colorado

Left Panel: January observed average daily temperature in Colorado for 1950–1999 (top left) and projections for 2050 (bottom left). By 2050 the January climate of the Eastern Plains has moved northward by a distance greater than half the state. The climate zones of the mountains have migrated upward in elevation, and the climate of the desert Southwest has progressed into the valleys of the Western Slope. Right Panel: July observed average daily temperature in Colorado for 1950–1099 (top right) and projections for 2050 (bottom right). For July, the temperatures on the Eastern Plains have moved westward and upslope, such that the temperature regime near the western Kansas border has reached the Front Range by 2050. Projections were calculated by adding the multi-model average temperature changes to the observed climatology. Observed climatological averages are from PRISM (Di Luzio et al. 2008), and projected changes from the IPCC (CMIP3) 22-model average for the A1B emissions scenario. Figures from Ray et al. (2008).

receive greater precipitation in the future. Climate model projections for precipitation in Colorado do not show substantial agreement, making future changes in the amount and seasonal patterns of precipitation uncertain.

Warming temperatures are projected to have significant effects on snowpack, timing of snowmelt, and streamflow even without a decrease in precipitation, however. Modest declines of 10% to 20% in snowpack SWE are projected for Colorado’s high elevations (>8,200 feet), but lower elevations could lose a much greater percentage of snowpack from snow falling as rain. With decreasing snowpack and increased evapotranspiration from warmer temperatures, drier conditions will exist overall.

At high elevations, winter temperatures are more likely to remain below freezing, even with warming, but temperatures in the spring may rise above freezing earlier. Therefore, continued shifts toward earlier snowmelt and snowmelt-driven streamflow pulses are projected in the future. Projections for the Upper Colorado River Basin also suggest that April soil moisture may increase due to earlier snowmelt, but soil moisture in May through July may decline by 10% to 40% in many locations (Ray et al. 2008). Regional climate models for the monsoon season (July through September) vary in projecting changes to precipitation, with some models showing increases and other decreases (Rangwala and Miller 2010). Like other snowmelt-dominated areas of the West, Colorado could experience more flooding in spring and drier conditions in the summer. Climate model projections give varying results for future estimates of annual river runoff, but a study of multiple models projects a decrease of 5% to 10% for the Arkansas (62% model agreement on the direction of change) and Rio Grande (75% model agreement on the direction of change), 10% to 25% reduction for the Upper Colorado (95% model agreement on the direction of change), and no appreciable change for the Platter/Missouri Basin by about 2050 compared to a 1900–1970 baseline period. These projections are for the entire river basin, however, and include areas outside of

Colorado. The average change in river runoff across Colorado is projected to be a 10% to 20% decrease (>90% model agreement) by 2050 (Ray et al. 2008).

2.4 Southwestern Colorado

Temperature, Precipitation, Moisture, and Streamflow

Few climate studies have focused just on southwestern Colorado. Often the best information we have to apply to climate preparedness at our local scale comes from investigations at medium scales, including Colorado and the western United States. Here we present what additional information does exist at a smaller scale.

An analysis of regional trends in annual average temperature for Colorado shows that from 1977 to 2006 the trend for southwest climate division in Colorado is the same as the average trend for the entire state (i.e., +1˚C (~2˚F) over 30 years). For the past 50- and 75-year periods, however, a significant warming trend was not observed. This lack of a longer-term climate trend at this small-spatial scale could be due to natural local variability masking any underlying global climate change signal. The southwestern Colorado climate division includes monitoring stations from the southwestern corner of the state up to the western and southern San Juan Mountains, but not including the northern San Juan Mountains (Ray et al. 2008).

An independent analysis by Dr. Imtiaz Rangwala and Dr. James Miller of observed surface air temperatures from National Weather Service (NWS) and SNOTEL (SNOwpack TELemetry) stations in the San Juan Mountain region from 1906 to 2005 suggests the following main conclusions:

- Warming of over 1˚C (1.8˚F) between 1910 and 2009, mostly occurring after 1993 (Figure 2-5);
- General pattern of this warming is similar to what has happened globally and in the western United States; however, a more rapid warming occurred in the San Juan Mountains between 1993–2005;

- Long-term gradual warming (1910–1990) also occurred in nighttime temperatures;
- Recent warming is strongly influenced by high daytime temperatures in spring and summer, possibly owing to decreases in snow cover and soil moisture; and
- The warming signal is more robust in the minimum temperature, both in the long and the short term (Rangwala 2008; Rangwala and Miller 2010; Imtiaz Rangwala, personal communication).

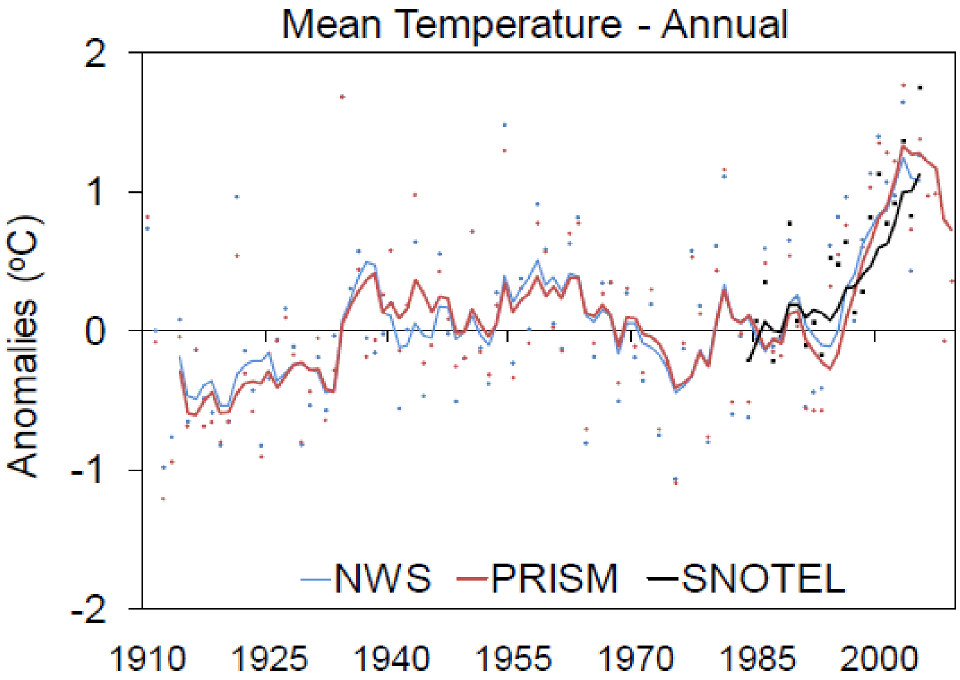


Figure 2-5 Anomalies in the mean annual surface air temperature

Anomalies in the mean annual surface air temperature from 1906–2005 in the San Juan Mountain region compared to the 1960–1990 baseline period from National Weather Service (NWS) (in blue), SNOTEL (in black), and modeled PRISM (in red) datasets. Small circles describe the absolute value of the anomalies in the surface temperature. The trend described by the thick dark line is a 5-year running mean of the data. Prior to 1950, only six or fewer stations existed, while after 1950 there were 26 stations. Figure courtesy of Imtiaz Rangwala, NOAA Earth System Research Laboratory.

Warming during the 1985 to 2005 period was greatest in the winter for the NWS Stations and in the spring and summer for SNOTEL Stations. This difference in seasons could reflect the difference in station elevations. The SNOTEL sites (average elevation = 10,500 feet) are about 2,500 feet higher than the NWS sites (average elevation = 7,000 feet), and the SNOTEL sites experience the bulk of snowmelt later in the year than the NWS sites.

Downscaled Future Projections for the San Juan Mountain Region

Researchers at NOAA and at the University of Colorado—Boulder examined dynamically downscaled temperature and precipitation projections available from the North American Regional Climate Change Assessment Program (NARCCAP) for southwestern Colorado for a historic (1971-1999) and a future (2041-2070) period (Rangwala and Miller 2010). These projections are available at a 50 km resolution. The area of investigation extended from 36.0°-38.5° North latitude and 105.5°-110.0° West longitude, and was centered on the San Juan Mountains. Outputs from five different combinations of RCMs and GCMs are discussed here.

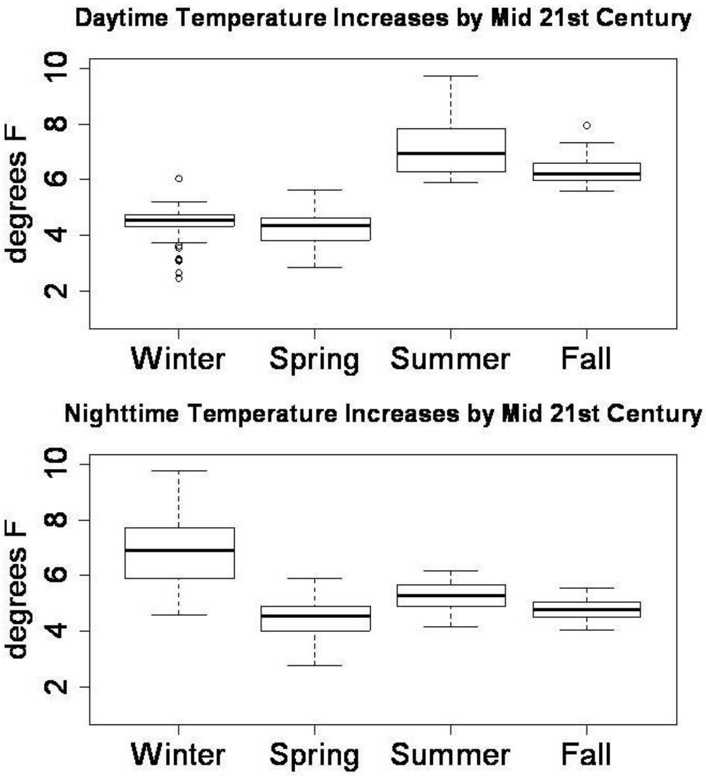


Figure 2-6 Changes in the average maximum (daytime) and the minimum (nighttime) temperatures by mid-21st century in the San Juan Mountain region from one NARCCAP model (CRCM3+CGCM3)

The boxes describe 25th and the 75th percentile values. The dark line is the median value and the error bars are 1.5 times the inter-quartile range. Figure courtesy of Imtiaz Rangwala, NOAA Earth System Research Laboratory. Data subsequently published in Rangwala et al.2012).

Figure 2-6 shows seasonal increases in the average maximum (daytime) and minimum (nighttime) temperatures in the San Juans by the middle of the 21st century relative to the late 20th century for one RCM–GCM combination (Rangwala et al. 2012). These results are very similar for all the NARCCAP model runs analyzed, except for changes in the minimum temperature during winter. RCMs focused on southwest Colorado project warming of maximum (daytime) and minimum (nighttime) temperatures generally in excess of 2.2°C (4°F) in all seasons (Rangwala et al 2012). Summer has the highest daytime warming (>3.3°C [6°F]). By the mid-21st century, the normal daytime summer temperatures are projected to be similar to those observed in 2002, the year with the highest recorded summer temperatures in the San Juans. Preliminary analysis suggests that high summer temperatures are caused by increased drying of the land surface. Several models also show large nightime warming (>3.3°C [6°F]) during winter, particularly at lower elevations (<8,000 feet). The analysis suggests that this warming may be, in part, owing to the projected early snowmelt, increased moistening of the land and lower atmosphere, and reduced snow cover at lower elevations. These interacting, reinforcing factors may positively feedback on the longwave heating of the land surface at night (Rangwala et al 2012).

Figure 2-7 shows the differences between mid-21st century and late-20th century precipitation in southwestern Colorado for five NARCCAP model runs. There is a broad consensus among the GCM-RCM combinations for certain seasonal effects and lack of concensus in others. Specifically, it is agreed that southwestern Colorado can expect: near normal precipitation or a slight increase during December and January, near normal precipitation or a decrease from April to June, and a decrease in precipitation in July and August. In contrast, a comparison of historic modeled and observed precipitation showed that, in general, the five model runs overpredicted precipitation except during the much wetter months such as July and August when three to four of the models underpredicted precipitation. Further, none of the models adequately

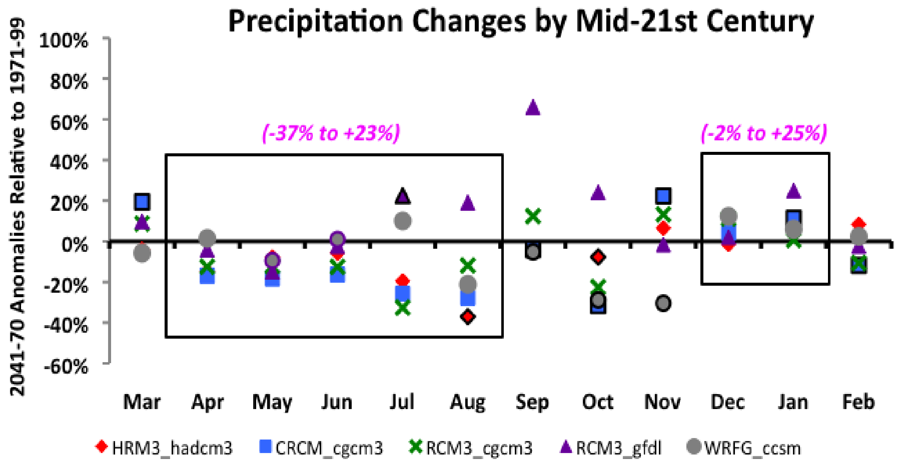


Figure 2-7 Percent change in total monthly precipitation for the San Juan Mountain region by mid-21st century (2041–2070) as compared to the late 20th century (1971–1999) from five NARCCAP model runs

Figure courtesy of Karen Cozzetto, University of Colorado—Boulder.

captured the dry to wet transition associated with the influence of the North American Monsoon in Colorado. The precipitation analysis did not take into account any effects of evapotranspiration, which will also affect moisture availability.

Evapotranspiration will increase as a result of rising air temperatures and will reduce overall water availability.

These RCMs capture observed variability in temperature and precipitation between 1979 and 2005 when forced with observed meteorological boundary conditions from the National Centers for Environmental Prediction Reanalysis. However, caution should be exhibited towards relying on these projections because only four GCMs were used by NARCCAP to provide boundary conditions for downscaling. The temperature and precipitation projections from these GCMs do not span the entire range of projections from the 20+ models participating in the Fourth IPCC Assessment Report (IPCC 2007). Thus, the NOAA/CU analysis may not capture the full range of possible future climatic responses coming out of the international climate modeling effort. In addition, the climatic effects of mountainous topography may not be adequately captured by the RCMs because the highest elevations in the RCM representations of the San Juan Mountains only approach 11,000 feet instead of the actual 14,000+ feet. Nevertheless, it is expected that the continuing research on these NARCCAP model projections, as opposed to statistical downscaling, will allow researchers to piece together a fuller picture of the regional climatic changes associated with process-oriented changes in variables, such as snowpack, soil moisture, and atmospheric humidity. This effort should provide useful insights to land and water resource managers in developing their planning framework for future climate change.

Snowpack and Streamflow

The behavior of annual snowfall in the San Juan Mountain region from 1906 to 1990 does not show any particular trend and varies between ±20% of the mean for the 1960 to 1990 period. However, there is a decreasing trend since 1995. There is a 25% decrease in snowfall at NWS Stations between 1995 and 2005, which is also confirmed by the trends observed at the SNOTEL sites. Similarly, observations of snowpack SWE from both SNOTEL and Snowcourse sites suggest a 40% decrease between 1995 and 2005. The snow depth measurements from Snowcourse sites show a decline of 25% during the same time period. On a monthly basis, April demonstrates the largest decrease in snow depth during the 1996–2005 period, even though, the snowfall during April has not changed significantly over that period (thus reflecting a change in SWE). These observations are not explained by climatic phenomena in the Pacific Ocean (El Niño/La Niña and the Pacific Decadal Oscillation) (Rangwala 2008; Rangwala and Miller 2010; Clow 2010).

Trends in the timing of snowmelt and associated peak of runoff has shifted earlier into the spring (i.e., two weeks over a 29 year study). The trends for southwestern Colorado are consistent with the average for the state of Colorado for the 1978 to 2004 period (Clow 2010). Shifts toward earlier snowmelt and resulting changes in streamflow during this 29-year period have been greater for the San Juan Mountains compared to more northern mountain regions in Colorado, however. Variations in springtime air temperatures and

SWE explain much of the change in snowmelt and streamflow timing (Clow 2010).

Like other regions in the western United States, variability in annual river flows in southwestern Colorado is heavily influenced by annual precipitation. Jeff Lukas (2008) conducted a study of long-term annual flow was conducted for the Animas River using tree-ring records to reconstruct a flow record from 1470 to 2002. The gaged record (1928 to 2002) was used to calibrate the tree-ring reconstruction. Figure 2-8 below shows Animas River at Durango reconstructed annual flows from 1470–2002 (green line), with a five-year running average (black line) to highlight multi-year droughts. The orange line is the average gaged flow (595,000 acre-feet). The reconstruction shows several multi-year droughts more severe than any in the gaged record. The 11-year drought from 1873–1883 (red arrow) was the longest drought in the reconstruction, while the five-year drought from 1622–1626 (orange arrow) had the lowest five-year average flow (332,000 acre-feet, 56% of normal). The reconstructed annual flow for 2002 (brown arrow) was the lowest in the entire 533-year record—indicating what a truly extreme year it was, although three other years (yellow arrows) had reconstructed flow similar to 2002 (Lukas 2008).

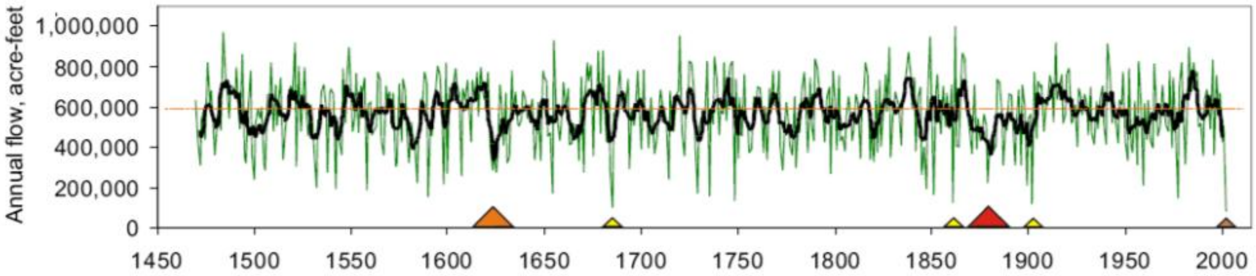


Figure 2-8 Tree-ring reconstructions of streamflow

Tree-ring reconstructions of streamflow for the Animas River at Durango from 1470–2002 (Lukas 2008).

The streamflow reconstruction shows that variability of Animas River flow since 1900 was less than during the four centuries prior to 1900, and flows were higher on average after 1900. In addition to more extreme flows, the record of past flows included several droughts longer than 2000–2004. While the conditions of the past will not be exactly repeated in the future, they can provide a guide to the full range of conditions that might be expected—including droughts longer and more severe than 2000–2004 (Lucas 2008). Any reductions in annual streamflow due to climate change will be superimposed on natural variability and other changes in human patterns (Luce Holden 2009).

Dust on Snow

Dust deposition onto snow cover has been shown to reduce late-spring snowpack depth and accelerate spring snowmelt in the San Juan Mountains (Figure 2-9; Painter et al. 2007). Dust deposition on snow is not a new occurrence; however, dust load levels in the western United States have increased by 500% above historical averages following settlement in the nineteenth century (Neff et al. 2008). In Colorado, the timing of snowmelt and snowmelt runoff has shown a substantial shift towards earlier in the year (Clow 2010). Dust deposition and early snowmelt are a concern because of implications for water resources (Clow 2010; Mote et al. 2005), biological systems (Steltzer et al. 2009), and biogeochemical cycles (Neff

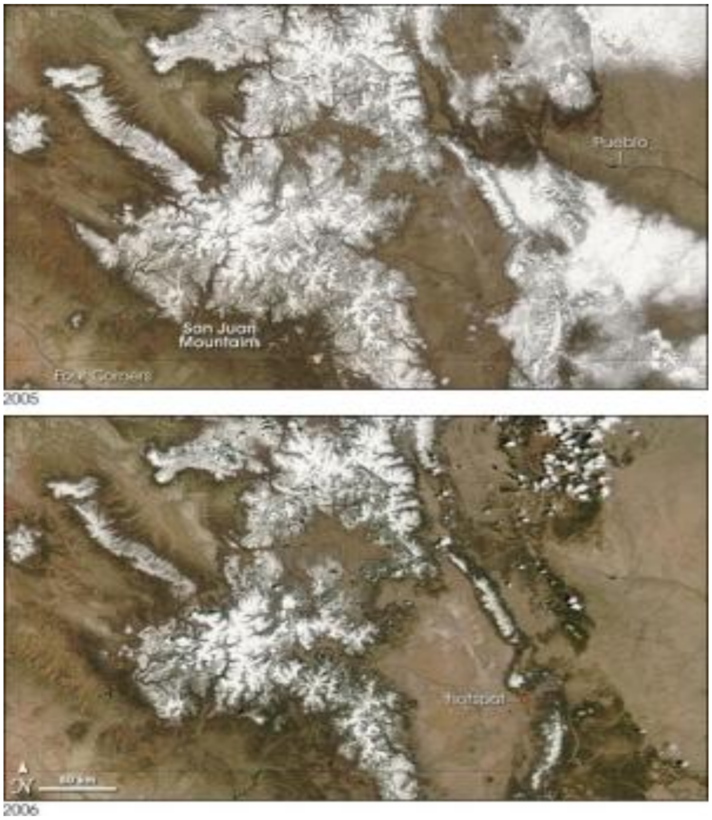


Figure 2-9 Variations in snowpack extent in the San Juan Mountains 2005 & 2006

Shows the extent of snowpack in the San Juan Mountains in 2005 (above) and 2006 (below) on the same date. 2005 had four dust events, while 2006 had eight events and shows a reduced snow extent. Photo credit NASA: http://earthobservatory.nasa.gov/IOTD/view.php?id=7842 et al.2008; Reynolds et al. 2001; Rhoades et al. 2010).

Aeolian, or wind-driven, dust events originate when wind erosion causes soil particles to become suspended, making them available for long-distance transport on a regional, continental, or global scale (Field et al. 2010). In the spring, as dust-laden air masses are pushed up and over the San Juan Mountains, gravitational settling and orographic precipitation deposit particulates (Lawrence et al. 2010). Dust deposited in the San Juan Mountains likely has a regional source area, as indicated by the physical properties of the dust (i.e., silt and clay-sized particles) and isotopic composition of deposited dust, and a February 2006 dust event in the southern Rocky Mountains has been linked to the Colorado Plateau region (Lawrence et al. 2010; Neff et al. 2008; Rhoades et al. 2010). Wind erosion in dryland source areas is increased when the biological and physical soil crusts are disturbed, by activities such as livestock grazing, off-road vehicles, military training, and fire (Field et al. 2010). Reynolds et al. (2001) show much greater soil loss from a grazed grassland surface than from nearby undisturbed surfaces stabilized by biological soil crusts. Climate warming is likely to exacerbate the problem by decreasing soil moisture in arid regions, which results in reduced vegetative cover and slower recovery from disturbance (Field et al. 2010). High elevation ice cores show that dust deposition on mountain snow has occurred throughout much of recent history, and increases in dust deposition correlate with periods of intense land use (Neff et al. 2008; Painter et al. 2007).

Dust deposition onto snow surface reduces albedo (Warren and Wiscombe 1980), therefore increasing solar radiation absorption and accelerating snowmelt (Flanner et al. 2009). Once a dust layer is deposited, the snowpack will only reflect 50%–60% of the incoming visible light (Painter et al. 2007). In 2006, it was found that snow cover duration in the San Juan Mountains was shortened by 18 to 35 days after deposition of disturbed desert dust (Painter et al. 2007). Shortened snow cover duration is of importance because the mountain snowpack is a key component of the hydrologic cycle in the western United States; it acts as a natural reservoir, storing water throughout the winter and releasing it in spring and early summer (Clow 2010; Mote et al. 2005). Early snowmelt has implications for reservoir operations, flood risk, water rights, wildfire, and forest ecology (Clow 2010).

Early snowmelt from dust deposition can also have biological impacts. Through a series of snowmelt manipulations, Steltzer et al. (2009) showed that phenology exhibited a threshold response to the timing of snowmelt (Figure 2-10). Greening and flowering were delayed before seasonal warming, and after warming there was a linear relationship between snowmelt and phenology (Steltzer et al. 2009). A change from staggered to synchronized plant growth across alpine landscapes could result in decreased nutrient retention, altered species interactions, and changes in landscape-scale gene flow via pollination (Steltzer et al. 2009).

Biogeochemical changes occur as a result of dust events because the nutrient-rich soil fine fraction is preferentially eroded by wind, leading to depletion of

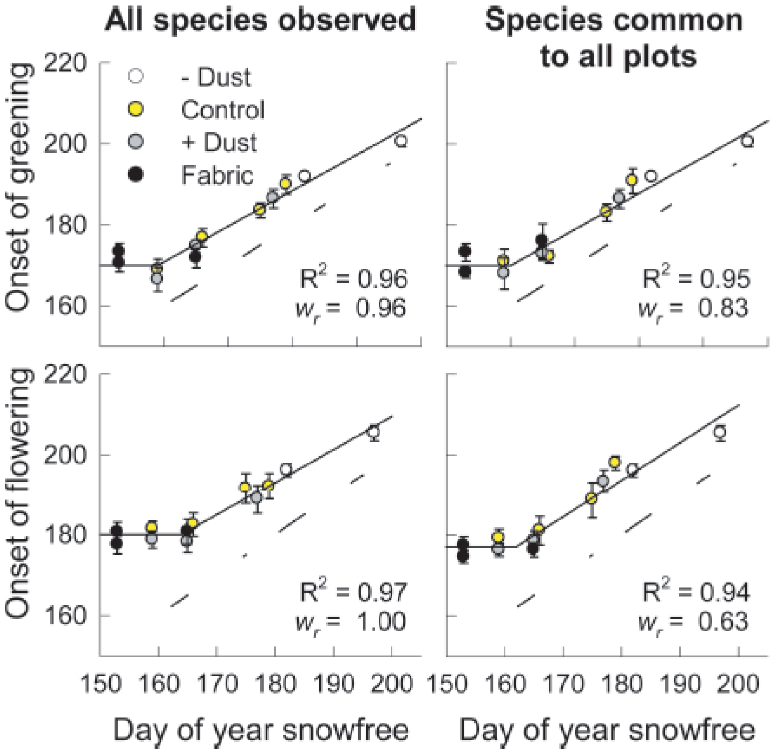


Figure 2-10 Onset of greening and flowering in relation to timing of snowmelt

Dates for the onset of greening and flowering are shown for all species observed within a plot (left) and for the five species common to all plots (right). Statistics report the weight of support for each model (Wr) and the proportion of variation explained (R^2) by the model. The dashed line is the 1:1 line. Data are means ±1 SE (n=12). Overlapping values were offset by up to 1.5 days for clarity. From Steltzer et al. 2009.

nutrients in the source area and nutrient loading where the dust is deposited (Field et al. 2010; Neff et al. 2008). Aeolian dust from the Colorado Plateau, which is within the source region of dust to the San Juan Mountains, generally has high concentrations of base cations, nitrogen, and phosphorus (Neff et al. 2008; Reynolds et al. 2001). In addition, Lawrence et al. (2010) found that snowpack layers in the San Juan Mountains affected by dust deposition events had significantly higher concentrations of base cations than layers that did not have dust. Following a dust deposition event in the Colorado Front Range, the snowpack was found to have higher pH, calcium, and acid neutralizing capacity; additionally, the chemical composition and concentrations of major solutes exceeded any monthly snowpack sample analyzed during the past 17 years for the study area (Rhoades et al. 2010). Base cation loading in high elevation lakes could be a factor in mitigating generalized regional increases in acid deposition (Neff et al. 2008), and is also likely to enhance nutrient availability in soils (Rhoades et al. 2010).

Chapter 3: Ecosystems and Phenology of Plant Flowering

3.1 Ecosystems

Introduction

Over the past 12,000 years, range, diversity, and pattern of vegetation have depended upon many physical factors including topography, seed dispersal distance, temperature, and precipitation (Anderson and Feiler 2009; Coats et al. 2008; Perfors et al. 2003). Increases in temperature and CO_2, as well as periods of decreased precipitation experienced in recent decades across the western United States (Figure 3-1) have greatly influenced native and exotic plant species, causing shifts in their distributions and, in some cases, leading to decreased ecosystem health (e.g., Allen 2009; Allen et al. 2010; Bigler et al. 2007; Breshears et al. 2005; Vankat 2010; Worrall et al. 2008). These trends are projected to continue given forecasts for continued warming and drying in the West, where water and temperature are the main limits to plant productivity (Allen et al. 2010; Harte and Shaw 1995; Loik et al. 2004). Rehfeldt et al. (2006) predict that only about half of the current climate regimes and associated plant distributions in the West will be maintained by the century's end. Models suggest that the preferred ranges of many plant species may shift by hundreds of kilometers over the next century; this will be the greatest shift in vegetation range since the last glaciation (Neilson, Pitelka, et al. 2005).

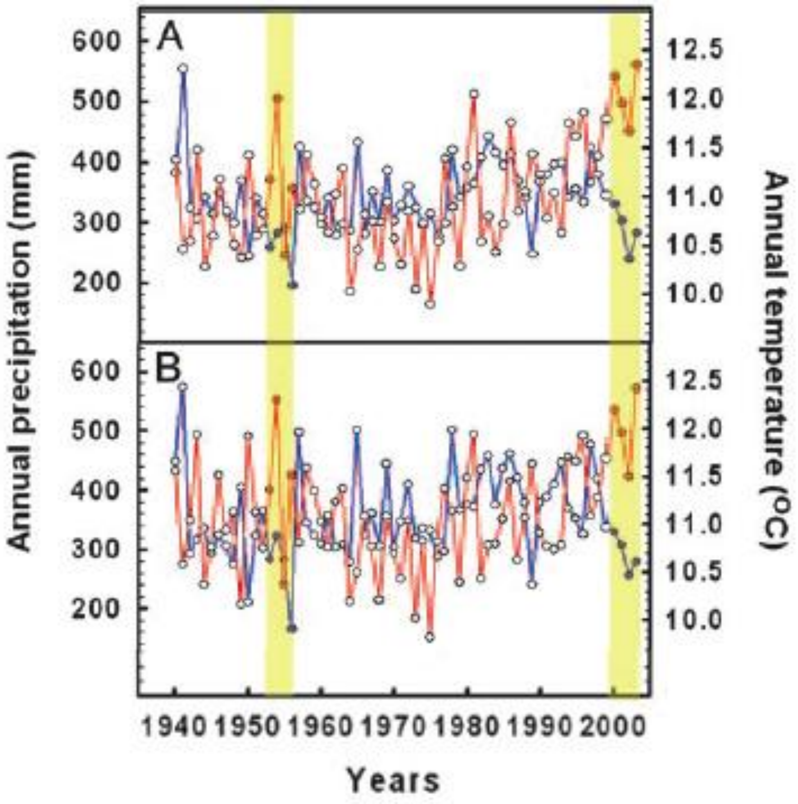


Figure 3-1 Southwestern North American climate

Annual mean precipitation (mm) and average of maximum and minimum temperatures (°C) for (A) all weather stations in the Four Corners Region and (B) only stations in or near piñon-juniper woodlands within that region. Shaded bands indicate drought episodes. From Figure 1 of Breshears et al. 2005.

Plant communities across the West differ in species composition and structure, yet in most systems, disturbance is an integral component of ecosystem health. Although disturbance can rejuvenate some systems, it may cause enduring and sometimes negative changes in others (Ryan et al. 2008). Changes in climate not only affect plant communities directly through modifying temperature and moisture availability, but indirectly through altering disturbance regimes such as the severity and frequency of wildfire and outbreaks of insects and disease. Climate effects on disturbance may impact future ecosystems as much as the effects of climate itself (Ryan et al. 2008). Each plant community is predicted to respond individually to climate change based on physiological characteristics and requirements, and both the current and historic range of the dominant species (Crookston et al. 2010).

Shrublands and Woodlands

Sagebrush (*Artemisia* spp.) shrubland is an extensive vegetation type of the western United States; historically, its range expanded during warm climatic periods (Anderson and Feiler 2009; Harte and Shaw 1995; Perfors et al. 2003). Currently, sagebrush shrubland is being lost to woody plant encroachment, land-use activities, and exotic plant invasion that are in turn driven by a combination of fire suppression, fire exclusion via grazing, and climatic change over the past century (Aldridge et al. 2008; Neilson, Lenihan, et al. 2005). Model forecasts for the future climate of sagebrush regions vary in terms of precipitation, but most models agree that temperature will increase across its range (Neilson, Lenihan, et al. 2005). Artificial warming experiments on sagebrush at the Rocky Mountain Biological Laboratory in Gothic, Colorado suggest seedling establishment, growth rate, and biomass of sagebrush will increase with warmer temperatures (Harte and Shaw 1995; Perfors et al. 2003). These experiments, however, do not account for projected elevated CO_2 that will exacerbate woody undergrowth and encroachment and ultimately the reduction of sagebrush shrublands (Neilson, Lenihan, et al. 2005). It is projected that populations at the southern edge of the distribution may be lost with expansion upward in elevation into areas currently with fragmented shrubland communities (Perfors et al. 2003). In the Great Basin, it is projected that along with warming, if precipitation increases in the future, woodlands will decrease in elevation displacing sagebrush; but if precipitation decreases, woodlands may go up in elevation but frost-sensitive desert shrubs may move up in elevation and displace sagebrush (Neilson, Lenihan, et al. 2005). With either scenario, sagebrush communities are at risk.

Another common shrubland community, Gambel oak (*Quercus gambellii*), is predicted by the end of the century to increase its range and increase its elevation by as much as 300 meters (Rehfeldt et al. 2006). This model however, does not incorporate elevated CO_2, nor does it take into account the asexual reproductive habit of this species, which enables relict populations of the plant to survive when climatic conditions would otherwise prevent the completion of the sexual cycle (Neilson and Wullstein 1983). Ronald P. Neilson of Oregon State University (personal communication August 2, 2010) feels that Gambel oak will likely move north, on the east (and possibly west) side(s) of the Rocky Mountains. This species has been shown to be sensitive to spring freeze (e.g., reduction in seed set due to exposure to spring freeze) and summer drought (reduction in viability of seeds) (Neilson and Wullstein 1983). These two events, which also reduce seedling establishment and maturation, are projected to increase in the future (e.g., see Inouye 2008). No models currently take into account the projected four- to six- week earlier shift in the start of spring; this earlier spring can have great and negative impacts on frost-sensitive deciduous species such as Gambel oak.

Piñon-juniper woodlands are another extensive vegetation community of the western United States and are sensitive to climate variation (Breshears et al. 2005). Over the last 10,000 years, the range of piñon-juniper has moved up and down along elevational and latitudinal gradients (Barger et al. 2009). There have been retractions during dry periods and expansions in wet and cool climatic periods, such as that seen in the 1920s across the Southwest, when both twoneedle piñon (*Pinus edulis*) and ponderosa pine (*P. ponderosa*) had major recruitment episodes (Barger et al. 2009). Water and heat stress in these and other forest communities can very quickly lead to major declines in growth, trigger extensive insect and disease outbreaks, and cause high mortality levels well past the time of drought (Allen 2009; Breshears et al. 2005). In the southwest United States between 2002 and 2003, water stress resulted in infestation of piñon beetles (*Ips confusus*) and the impending death of over a million hectares of twoneedle piñon, equaling up to 80% of the population (Figure 3-2; Allen et al. 2010; Breshears et al. 2005; Romme et al. 2006). This "global change type" drought was different than previously recorded droughts in that it was associated with both low precipitation and higher temperatures;

twoneedle piñon trees of all size and age classes were killed and at a greater magnitude and extent than in the past (Breshears et al. 2005; Romme et al. 2006). Associated species were also impacted; for example, blue gramma (*Boutelua gracilis*) grass cover was reduced by over 50% in plots in New Mexico and oneseed juniper (*Juniperous monosperma*), a species considered more drought-tolerant than twoneedle piñon, lost up to 26% cover in four measured locations within the region. Heavily impacted areas could take many decades to re-establish overstory structure (Breshears et al. 2005).

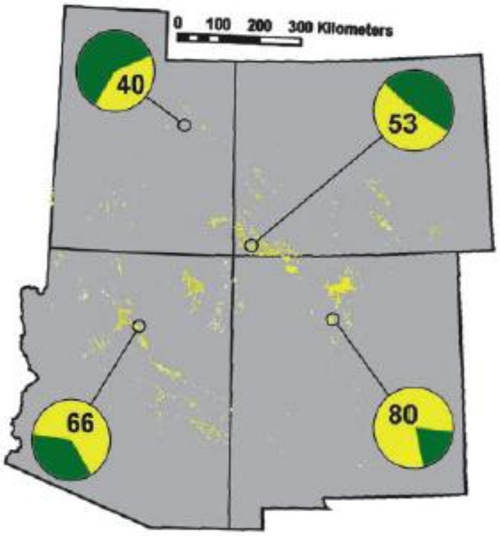


Figure 3-2 Regional drought-induced vegetation changes

Yellow portions of the pie charts show 40% to 80% tree mortality of non-seedling trees for four study areas in the Four Corners Region. Aerial survey data were collected by the U.S. Forest Service. From Figure 3 of Breshears et al. 2005.

Projections for climate in the West include increases in temperature and reduction in precipitation; even areas that currently are not water-stressed may become so in the future (Allen et al. 2010). Although much twoneedle piñon was lost as a result of the recent drought, at least temporarily, in the Southwest, other studies indicate current and projected expansion of its range to the east onto the Great Plains (Anderson and Feiler 2009). Models incorporating projected elevated CO_2 suggest an overall increase in piñon-juniper woodland by the end of this century (Bachelet and Neilson 2000; Neilson, Lenihan, et al. 2005). Although twoneedle piñon require a cool wet climate to successfully reproduce and establish (Barger et al. 2009), inter-decadal variability in climate may allow establishment every few hundred years, allowing for replacement (Ronald P. Neilson, Oregon State University, personal communication July 28, 2010). Alternatively, bioclimatic models that do not incorporate elevated CO_2 suggest a "drastic reduction" in both twoneedle piñon and Utah juniper (*Juniperus osteosperma*) in the future, accompanied by increases in elevation of 500 and 100 meters, respectively (Rehfeldt et al. 2006). The composition of woodlands is also expected to change. The Rehfeldt et al. (2006) model suggests the ratio of summer to total precipitation is the most important variable for the prediction of these two species' occurrence. Currently, twoneedle piñon and Utah juniper co-occur in roughly 35% of woodland areas; by the century's end it is predicted that the two species will share just 6% of their area (Rehfeldt et al. 2006). From the studies of Neilson, Pitelka, et al. (2005) and Rehfeldt et al. (2006), it is clear that variables included in or excluded from bioclimatic models have great affects on model outcomes; models that incorporate the most information may give us the clearest picture of the future.

Forests

Forests of the western United States are primarily dominated by conifers and can be loosely grouped into ponderosa pine forests (*Pinus ponderosa*), mixed-conifer forests (including ponderosa pine, Douglas-fir [*Pseudotsuga menziesii*], white fir [*Abies concolor*], and aspen [*Populus tremuloides*]), and subalpine forests (including Engelmann spruce [(*Picea engelmanii*], subalpine fir [*Abies lasiocarpa*], and lodgepole pine [*Pinus contorta*]). In general, changes in these forests in recent decades include increases in smaller-diameter trees, reduction in large, old growth trees, alterations in species composition, and range expansions upward in elevation (Figure 3-3; e.g., Fulé et al. 2009). Platt and Schoennagel (2009) report significant increases in tree cover at middle and low elevations of Colorado's Front Range, in particular in ponderosa pine forests and shrub communities. Many studies throughout the West report decreases in large-diameter trees and overstocking of smaller-diameter trees (Fulé et al. 2002; Keane et al. 2002; Korb et al. 2007; Mast et al. 1999; Vankat 2010; van Mantgem et al. 2009; Wiedinmyer and Hurteau 2010). The reasons for overstocking of small trees are largely climate-related (e.g., the right moisture and temperature must occur together for establishment), but also relate to land-use changes such as livestock grazing over the past century and fire suppression. The reasons for loss of old growth trees are climate-related (older trees are more drought-intolerant and susceptible to insect attack), but are largely a result of fire suppression and other human-caused land-use changes. For example, in ponderosa forests of the Southwest, frequent low-severity fire was a key disturbance prior to settlement; this type of fire has been largely replaced with high severity "stand-replacing" wildfire, which kills old trees (Mast et al. 1999). However, anecdotal evidence from field observations suggests that old growth ponderosa pine may be more resistant to recent wildfires than previously thought. In 2010, a field study is addressing this question (Laurie Swisher, San Juan National Forest, personal communication, April 29, 2010). In addition, logging of old growth trees and competition from dense young trees in modern forests contribute to the loss of old growth.

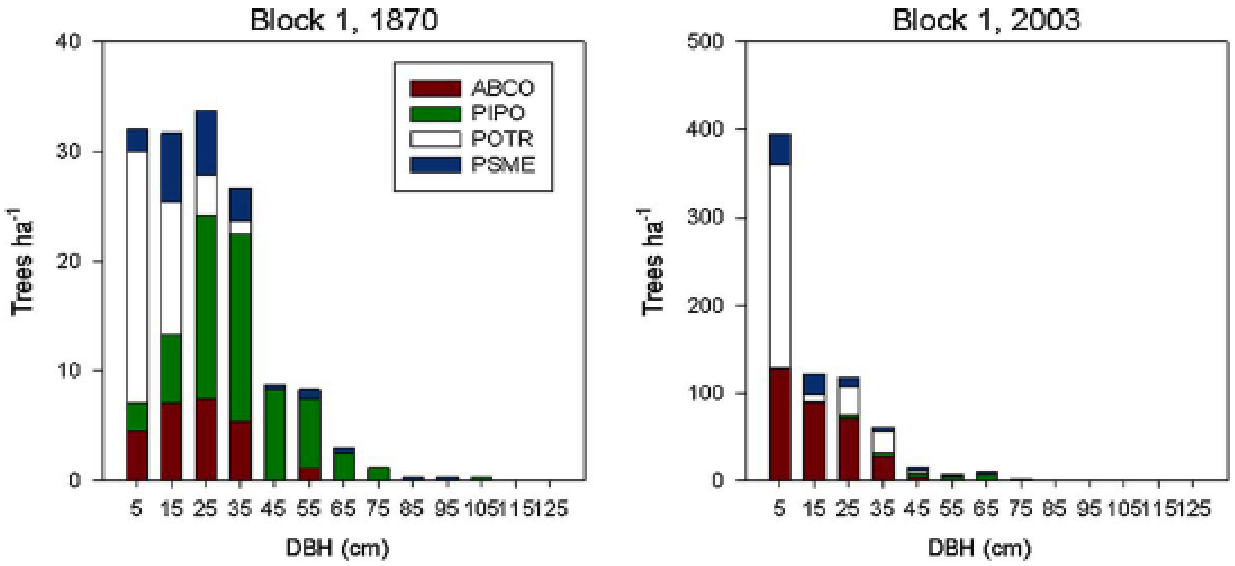


Figure 3-3 Diameter distributions by species in one study block in a warm dry mixed-conifer forest of southwestern Colorado

Reconstructed in 1870 and measured in 2003, showing an increase in forest density by approximately an order of magnitude and a shift toward dominance by mesic conifers (ABCO=white fir; PIPO=ponderosa pine; PSME=Douglas-fir) and aspen (POTR=aspen). From Figure 4 of Fulé et al. 2009.

Recently foresters have included climate variables in their models to understand the potential influences of climate change on forest community structure as dynamic interactions of mortality, regeneration, and species viability (Crookston et al 2010). Based upon bioclimatic factors, models suggest that by the end of the century the land area covered by ponderosa pine forests will decrease, Douglas-fir forests will not change, and Engelmann spruce will be "drastically reduced" (Rehfeldt et al. 2006). Similarly, models by Bachelet and Neilson (2000) suggest little change in the size of western pine or Douglas-fir communities and a decrease in western spruce-fir forests in the future. Elevational range is expected to increase by up to 500 meters for both ponderosa pine and Douglas-fir and up to 300 meters for Engelmann spruce (Rehfeldt et al. 2006). In the model of Rehfeldt et al. (2006), moisture stress was an important predictor for distributions of all three of these species. Indeed, tolerance to drought will greatly impact the composition and distributions of future forests. For example, Bigler et al. (2007) found that drought in Northern Colorado spruce-fir forests had greater impact on subalpine fir than on Engelmann spruce, especially in relation to early season drought, and that lodgepole pine (*Pinus contorta*) was not significantly impacted. This study shows a range of variability of high elevation tree species in the Southern Rockies in response to warming and drying.

Aspen already has begun showing signs of decline in its lower elevation habitats. A 10% reduction in the aspen cover type on the Mancos-Dolores Ranger District of the San Juan National Forest, Colorado, between 2005 and 2006 was directly correlated with regional drought (Rehfeldt et al. 2009; Worrall et al. 2008). Surrounding forests suffered similar, but less severe impacts. This loss of the aspen cover type was determined to be 17% across the state by 2009 and 19% in the Southwest Region including the Mancos-Dolores Ranger District (Worrall et al. in press). Here, elevation and soil type played important roles in the sudden die-off. Sudden aspen decline, as this die-off was called, occurred in primarily open stands of aspen at lower elevations, in drier vegetation communities characterized by the presence of ponderosa pine, snowberry (*Symphoricarpos rotundifolius*), and yarrow (*Achillea millefolium*), on flatter or south- or west-facing slopes, and in drought-prone soils (Figure 3-4; Worrall et al. in press; Worral et al. 2008).

Worrall et al. (in press) list three types of factors that contributed to SAD:
1) inciting factors which are physiological or biological factors that cause stress in the short-term—in this instance, acute drought and high temperatures during the growing season, 2) predisposing factors which are biological or environmental factors that affect plant physiology such as the location and density of stands, and 3) contributing factors that actually kill the debilitated trees; here, these were higher than normal populations of insects and a canker. Bioclimatic models show that the period between 2000 and 2003 was the most severe in a recorded 56-year period for this region in terms of high summer temperature, moderately warm winters, lowest precipitation, and most extreme temperature/precipitation indices (Worrall et al. in press; Rehfeldt et al. 2009). The loss of canopy seems to have followed the loss of roots initially; the loss of roots is presumed to be from the extensive drought described above, though

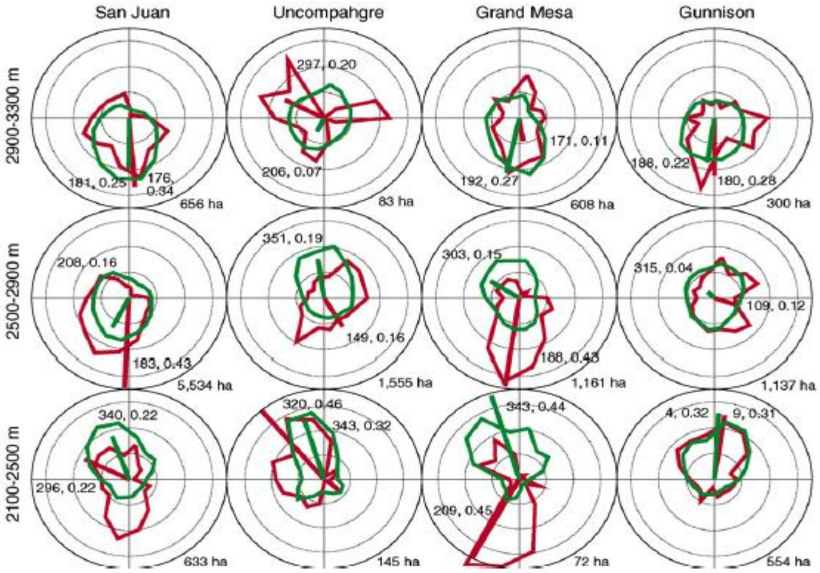


Figure 3-4 Aspect of healthy aspen and mortality

Aspect (slope orientation) of healthy aspen (green) and mortality (red) in three elevational bands on four national forests. Graphs are arranged like a compass dial. Polygon vertices represent relative frequency of aspect in 18 20-degree classes. Radial lines indicate the corresponding slope-weighted angular mean of aspect and its r-value. Concentric, dotted circles represent 0.05 units of relative frequency and 0.125 units of r. The value at the lower right of each graph is the approximate area of mortality represented in the graph. From Figure 5 of Worrall et al. 2008.

freezing has not been ruled out (Worrall et al. in press).

This mortality may extend into the future with poor asexual regeneration from deteriorated roots and a requirement for moist conditions for successful sexual reproduction (Elliott and Baker 2004; Rehfeldt et al. 2009). Poor regeneration has been associated with succession into conifer forest, or type conversion into shrub or grassland given less available water (Elliott and Baker 2004; Worrall et al. in press). It is not known if the slowing of this decline in recent years will be the end of SAD in Colorado, or if future droughts will result in more aspen loss (Worrall et al. in press). Although the damage from SAD has been severe, the majority of the aspen cover type in Colorado is still healthy and, even in impacted areas, roughly half of the canopy is still intact (Worrall et al. in press). Models (which do not account for increased CO_2 or vegetative reproduction) for the future of these forests show that 58% of the sites that experienced SAD in 2005 and 2006 will be outside of the climatic range of aspen by the year 2030, and 76% will be outside the range by 2060 (Rehfeldt et al. 2009). According to these models, by the end of the century the range of aspen will both decline in area and increase in elevation by up to 1,000 meters; the recent die-off largely occurred in areas already determined most likely to lose aspen in the future. The future of aspen lies in resilience of stands to regenerate from existing roots; this effort may need to be aided by land managers in areas of high herbivore pressure and marginal regeneration (Worrall et al. in press).

Aspect has also been shown to play an important role in the distribution and survival of tree species in the West, with southern and southwestern aspects having greater temperatures and more evapotranspiration (Bigler and Veblen 2009; Elliot and Baker 2004; Elliott and Kipfmueller 2010; Vankat 2010; Worrall et al. 2008). Bigler and Veblen (2009), in their study of spruce-fir forests of northern Colorado, found that tree species lived longer on wetter and cooler north-facing aspects. Similarly, Elliott and Baker (2004), in a study of aspen at treeline in southwestern Colorado, found invasion of aspen into alpine meadows occurring on warmer, south-facing slopes (Elliott and Baker 2004).

Recent studies of tree cover along Colorado's Front Range show a 6% increase in tree cover on south-facing slopes and no significant increase on north-facing slopes between 1938 and 1999 (Platt and Schoennagel 2009). Elliott and Kipfmueller (2010), in a study of subalpine conifer expansion into alpine across the Front Range of Colorado, found saplings and young trees established on south-facing aspects and seedlings establishing on north-facing aspects. They reason this may be attributable to very recent alterations in climate, making south-facing aspects that were suitable habitat in recent decades now unsuitable as a result of soil moisture content being too low to allow regeneration success. They predict that with continued warming and reduction in precipitation, future recruitment into south-facing aspects will be limited and north-aspect recruitment will continue upslope.

Warming can lead to tree mortality by increasing water deficits and drought stress and enhancing the growth and reproduction of insects and pathogens. These factors have been the leading cause of tree mortality across western forests in the last 35 years (van Mantgem et al. 2009). Historically, native conifer borers (mountain pine beetles [*Dendroctonus ponderosae*], spruce beetles [*Dendroctonus rufipennis*], and piñon beetles) and defoliators (Western spruce budworm [*Choristoneura occidentalis*]) occurred in low numbers and killed weak trees occasionally. Every so often, numbers of these insects increased to epidemic proportions and large numbers of trees were killed (Romme et al. 2006). Recent outbreaks of insects in Colorado are attributable to a combination of climatic factors: drought-stressed trees, warm summers that accelerate insect growth and development, warm winters which allow insect larvae to survive, and abundant trees in often dense forests (Figure 3-5; Hansen et al. 2010; Romme et al. 2006). Recent outbreaks have occurred at higher latitudes and higher elevations than previously recorded, and it is likely that higher temperatures and drier conditions are at least partly responsible (Romme et al. 2006). It is unclear, however, if the current high elevation insect activity is unprecedented, since data on precise spatial patterns of prehistoric

Figure 3-5 Satellite image of the extensive attack by mountain pine beetle in lodgepole pine forests in Colorado (comparison of 2002 and 2007)

Pre-outbreak image taken October 2002 (left) and post-outbreak image taken August 2007(right). Images courtesy of DigitalGlobe, Inc. (http://digitalglobe.com).

outbreaks are lacking (Romme et al. 2006). Even so, mountain pine beetles are cold-intolerant species, with their development closely correlated to seasonal temperature fluctuations and their death imminent from cold temperature exposure (Régnière and Bentz 2007). In the survival of mountain pine beetles, temperature is said to be more important than parasites, predators, competition, or food resources (Régnière and Bentz 2007); the increased minimum winter temperature in recent decades has played an important role in their over-wintering survival and their increase to epidemic numbers (Figure 3-6).

Because some insects, such as mountain pine beetles, require larger trees with thicker bark for laying eggs, older trees (which may also be less vigorous) are at greater risk of death from insects (Hicke et al. 2006). Following outbreaks, smaller trees and non-susceptible species may grow more rapidly with less competition and can lead to a type conversion (e.g., conifer forests to aspen forests). There are disadvantages to rapid growth, however, such as low wood density, reduced mechanical strength, and greater risk of damage from wind or insects (Bigler and Veblen 2009). Other past considerations following outbreaks have included increased wildfire risk from dead needles that act as ladder fuel and snags which act as lightning rods (Figure 3-7; Bebi et al. 2003; Negrón et al. 2008; Romme et al. 2006).

Recent studies from subalpine forests of Colorado which have long histories of both catastrophic fire and beetle outbreaks have found that there may be no relationship between post-beetle kill forests and either fire severity or extent (Kulakowski and Veblen 2007); the risk of catastrophic fire following beetle kill is greatest in the years during or immediately after the outbreak and in relation to drought (Bebi et al. 2003). In reverse, past wildfire may aid in protecting future forests from beetle outbreaks; as forests recover and trees are smaller they are less susceptible to beetles (Kulakowski and Veblen 2006, 2007). Although bark beetle outbreaks are natural disturbances and beetle-killed forests are a natural stage in forest re-development (Régnière and Bentz 2007; Romme et al. 2006), they may fall outside of management criteria, especially in areas of scenic, recreational, or timber value. Currently however, there are no practical management strategies at large scales for preventing outbreaks (Romme et al. 2006), although partial cut treatments are reported to reduce losses of lodgepole and ponderosa to mountain pine beetles, and spruce to spruce beetles (Hansen et al. 2010).

It is predicted that future warmer temperatures may lead to similar and possibly more intensive outbreaks than were experienced in the last decade (Janetos et al. 2008). For example, mountain pine beetle populations in higher elevations and latitudes of the western United States and Canada are expected to increase on a large scale (Hicke et al. 2006; Kliejunas et al. 2008; Ryan et al. 2008). In addition, spruce beetle outbreaks are expected to rise in the future, and models suggest the recent increased temperatures and decreased moisture which affected the rate of development in piñon ips beetle will continue into the future (Ryan et al. 2008). With the observed and projected warming and drying trends, trees will likely become more stressed, and the interactions of trees with insects and pathogens will "have numerous, mostly adverse, consequences to forest ecosystems" (Kliejunas et al. 2008).

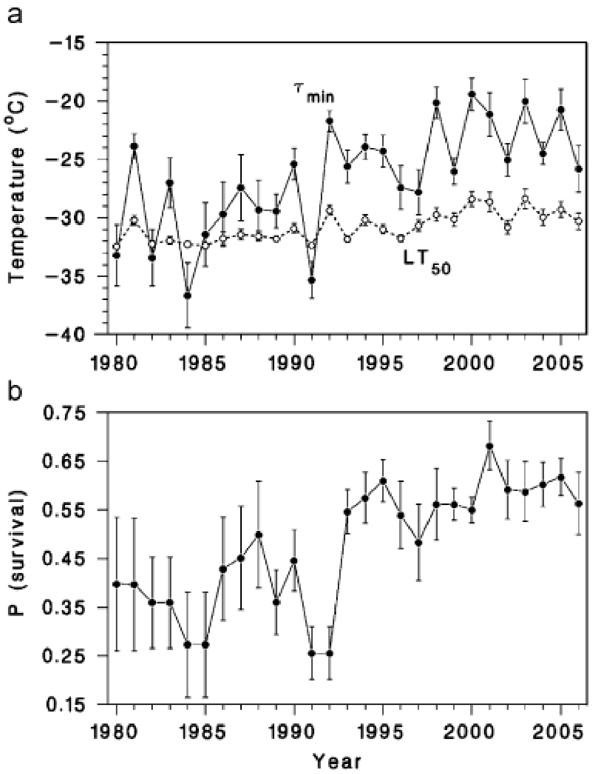


Figure 3-6 Impact of cold temperature on mountain pine beetle

Model results from daily 1980–2006 temperature records at 10 stations in Idaho showing the impact of cold temperature on mountain pine beetle. Annual average (± SEM) (a) extreme minimum phloem temperature, lowest median supercooling point (LT50) when lethal ice crystals form in the insect tissue, and (b) D. ponderosae larval survival from cold exposure. From Figure 6 of Régnière and Bentz 2007.

Figure 3-7 Lodgepole pine and spruce-fir forests

Lodgepole pine and spruce–fir forests typically burn at high intensity even without previous insect activity. It is widely believed that insect outbreaks set the stage for intense forest fires, but there is little scientific evidence for such a connection. From Figure 6 of Romme et al. 2006.

Alpine

Alpine plant communities are treeless and contain many habitats such as wetlands, wet meadows, dry tundra, snow and ice fields, fellfields, talus and scree slopes, and rock faces. Here, the most noted changes have been and will be the range expansions and contractions of herbaceous species and overall geographic contraction of the alpine environment. Both the extent and duration of alpine snow cover is projected to decrease in the future, with snow cover elevated by up to 400 meters upslope in some regions of the United States by the century's end (Burkett and Kusler 2000). Competitive interactions between plants are and will continue to be influenced by changes in temperature, precipitation, and soil moisture. This is especially clear in ecotones such as between subalpine and alpine environments, where faster growing subalpine or exotic plants can rapidly overtake slow-growing alpine plants (Burkett and Kusler 2000). In a recent study of the alpine vegetation of four summits in the San Juan Mountains, Crawford et al. (in review) found 11% of their observed species to be subalpine plants that likely had encroached upward into alpine habitats, as they previously had not been documented as alpine species in herbarium records. Alpine obligates have few options for movement given predicted climate warming, and southernmost populations or populations on small alpine patches may be at risk of regional extinction. This can be detrimental to species that are endemic to a small geographic range, leading to global extinctions. In the San Juan Mountains, Crawford et al. (in review) determined that 24% of the alpine species were at risk of local extinction given future warming and drying scenarios.

The encroachment of forests into the alpine (Dirnböck et al. 2003; Elliott and Baker 2004; Elliott and Kipfmueller 2010) causes competition with herbaceous species and further reduces habitat available to alpine plants and animals (Moir and Huckaby 1994). Tree seedlings may become established upslope in microhabitats often created by large boulders, terrace risers, or krummholz mats. As seedlings grow, they alter microsite conditions through soil development and protection from the wind and sky; this perpetuates additional tree encroachment upslope (Figure 3-8; Resler 2006; Smith et al. 2003). Tree advancement into the alpine can alter cycles of carbon, water, and nutrients (Malanson et al. 2007). Tree invasion into meadows has increased greatly over the last century across the West (Moir and Huckaby 1994). Given predictions of milder winters, reduced snowpack, and longer growing seasons, tree seedling establishment into meadows at treeline will be favored in the future, if enough moisture is available (Moir and Huckaby 1994; Moir et al. 1999). The present climate that supports the treeless alpine environment may decrease drastically in a warmer future with less snowpack duration (Bachelet and Neilson 2000; Burkett and Kusler 2000; Diaz and Eischeid 2007).

Wetlands: Fens & Wet Meadows

Figure 3-8 Patches of krummholz develop into tree islands at their leeward extent and aid in retaining snow

Photo from Libby Flats, Medicine Bow Mountains, WY. From Figure 5 of Malanson et al. 2007.

Wetlands include fens, permanent wetlands and seasonal wetlands. Fens are peat-forming wetlands supported in part by groundwater input. Fens are often saturated with water for most of the year, allowing little oxygen to enter the soil, which greatly reduces decomposition. This allows dead leaves and roots to gradually accumulate to form organic-rich soils called peat. Fens are often areas of high biodiversity and are regionally important refugia for rare plant and animal species that are otherwise limited to colder environments of boreal and arctic regions (Cooper 1996). Species such as whitebristle cottongrass (*Eriophorum altaicum*), buckbean (*Menyanthes trifoliata*), purple marshlocks (*Comarum palustre*), and Baltic sphagnum (*Sphagnum balticum*) occur within the southern Rocky Mountains, disjunct by hundreds or thousands of km from their main ranges in boreal North America (Cooper et al. 2002; Weber 2003). Rocky Mountain peatlands also provide important habitat for elk (*Cervus elaphus*), moose (*Alces alces*), and many species of amphibians and migratory birds (Chimner et al. 2010). Wet meadows are grass- and sedge-dominated wetlands that are seasonally saturated in the spring, but usually dry out by mid-summer.

Understanding changes in summer temperatures, length of growing season, annual snowpack SWE, timing of snowmelt, and alterations in the frequency and strength of monsoon rainstorms are critical to understanding the long-term persistence of fens and wet meadows. Lowered water table levels can result in the overall decomposition of the peat (Chimner and Cooper 2003a, 2003b; Chimner et al. 2002,). If water table levels remain adequately high, however, warming temperatures can increase peat accumulation (Sullivan et al. 2008). Over time, drying can shift plant communities from wetland hydrophilic vegetation (like sedges) to more mesic upland species (Cooper et al. 1998). In the San Juan Mountains, the depth of the water table is very sensitive to the amount and timing of snowmelt and summer monsoon rains. In years with early snowmelt or weak monsoons, water tables drop for extended periods of time (Figure 3-9; David Cooper, Colorado State University, personal communication 6/13/2010). With the projections of drier conditions in the future, it is expected that water table depths in fens and wet meadows will remain low for longer periods of time and may result in long-term drying, decomposition, and vegetation shifts in these ecosystems, especially those in marginal areas or already affected by other disturbances such as drainage ditches and roads. Some fens may become wet meadows and wet meadows may become dry meadows that lack conditions for wetland flora and fauna. Whether or not summer rains will increase, decrease, or

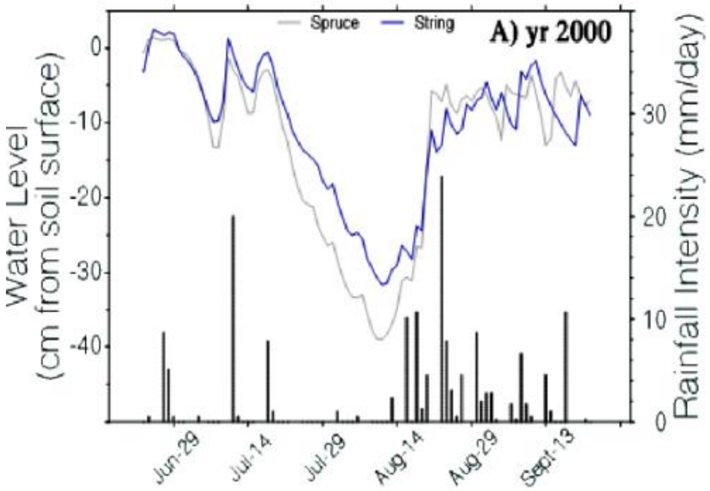


Figure 3-9 Drops in water table levels in fens

Extended periods of no summer rain can lead to steep drops in water table levels in fens. This graph shows water level (gray and blue lines) in Spruce and String Fen (Prospect Basin, San Miguel County, CO) in relationship to rainfall intensity (vertical bars). From David Cooper, Colorado State University.

remain unchanged is uncertain, however. If the monsoons persist, it may lessen the impact of earlier and reduced snowmelt on these ecosystems. In the San Juan Mountains, it is estimated that out of roughly 2,000 fens in the region, 15% (or 300 of them) are in fair to poor condition (Chimner et al. 2010).

The first study to specifically investigate how climate change affects fens in the Southern Rockies began in the summer of 2010. It is a cooperative study with Dr. Kate Dwire of the U.S. Forest Service Rocky Mountain Research Station and Dr. David Cooper of Colorado State University. The study asks how climate affects fen hydrologic regime, carbon storage and loss, and vegetation community composition. The two main study areas are the San Juan Mountains in southwestern Colorado, which are influenced by a summer monsoon season, and the Medicine Bow Mountains in Wyoming, which do not receive monsoon storms. Both high elevation (subalpine) and lower elevation (montane) fens will be included in each region.

Riparian Areas

Riparian areas are highly productive corridors that bisect and connect all aforementioned plant communities. They are critical habitat for many plant and animal species, and are valued for their biodiversity, water quality, and recreational opportunities (Hultine et al. 2007). Throughout the West in recent decades, many high elevation streams have experienced earlier peak spring discharge as a result of earlier snowmelt (Cayan et al. 2001). These shifts, as well as the predicted changes in the timing, type, and amount of precipitation, will impact watershed hydrology and population structure of riparian areas (Hultine et al. 2007). Studies of riparian communities in both Canada and the Sierra Nevada have linked recent hot dry summers with reduction in native cottonwood (*Populus* spp), and both cottonwood and willow (*Salix* spp), (Glenn and Nagler 2005). In container-grown cottonwood (*Populus fremontii* and *Populus deltoides* var. *wislizenii*) and willow (*Salix gooddingii*), Hartwell et al. (2010) found less water was required to maintain healthy cottonwood plants than willow plants. This study may provide insight into the drought tolerance of these important riparian community members.

Additional considerations in a drier future include changes in riparian plant community structure where female trees of the dioecious cottonwood, willow, and boxelder (*Acer negundo*) become less common (Hultine et al. 2007). In general, males of these species are more drought-tolerant and are found on less desirable microsites than the greater resource–requiring females, thereby potentially outliving drought-intolerant females in future severe drought conditions. Because females recycle more precipitation into the atmosphere than males, this imbalance in the sexes could impact hydrological processes in the future (Hultine et al. 2007) as well as sexual reproduction.

Another common riparian species in the western United States is thinleaf alder (*Alnus incana* ssp. *tenuifolia*); this species stabilizes soils, hosts nitrogen-fixing bacteria, and provides habitat for wildlife and fish. Since the late 1980s land managers have documented severe declines in thinleaf alder throughout its range, with 37% of its stems dead and 29% diseased in 2004 surveys (Worrall 2009). It is thought that cytospora canker (caused by *Cytospora umbrina* and *Valsa melanodiscus*) infected this plant historically during warm periods when the plants were stressed, but that the plant recovered in intermittent cool periods. For example, in 2006 canker growth occurred primarily between June 20 and July 20, during a time of highest heat and tree stress (Figure 3-10; Worrall, Adams, and Tharp in press). Since 1976, however, there have been no intermittent cool periods and the species is stressed from high temperatures, low humidity, and wind (but not water, since it grows in riparian areas), causing more susceptibility to the canker and little or no recruitment to replace lost individuals (Worrall Adams, and Tharp in press; Worrall 2009). Future warming could cause the continuation of the epidemic and loss of additional alder populations (Worrall Adams, and Tharp in press), at least in their lower elevational range.

Both climate and land-use changes affect rivers. Most rivers in the arid West are dammed, diverted, and pumped to supply water for human uses, resulting in the lowering of water tables to depths below thresholds required for many native tree species (Hartwell et al. 2010). These changes, along with increased salinization of riverbanks from the above human activities, have aided the spread of drought- and salt-tolerant exotic tree species such as tamarisk, also called saltcedar (*Tamarix* spp.; Glenn and Nagler 2005). Climate models project that tamarisk will increase in range over the next century (Bradley et al. 2009). The tamarisk leaf beetle (*Diorhabda elongate*) has been used successfully to defoliate tamarisk while leaving native species intact. This biocontrol method is controversial, however. While historically it was thought that saltcedar was a great user of precious waters in arid lands, recent studies have demonstrated that the plant has comparable evapotranspiration rates to native phreatophytes (e.g., cottonwood and willow) and, although saltcedar provides a poorer habitat than native riparian species to most wildlife, it may be useful to some species that have adapted to it (e.g., willow flycatcher [*Empidonax traillii*], the endangered Southwestern willow flycatcher [*Empidonax traillii extimus*], and black-chinned hummingbirds [*Archilochus alexandri*]; Glenn and Nagler 2005).

Invasive Species

In general, exotic plants have the capability of altering ecosystem function and can reduce biodiversity on large scales. Projected levels of CO_2 are expected to stimulate the growth of most plants, with some invasive exotics responding with even greater growth than natives (Janetos et al. 2008). Exotics may be better able to tolerate a wide range of climatic conditions and become more successful in the future than natives that are more limited by pollinators and seed dispersers (Janetos et al. 2008) or water/flooding requirements (see for example Glenn and Nagler 2005).

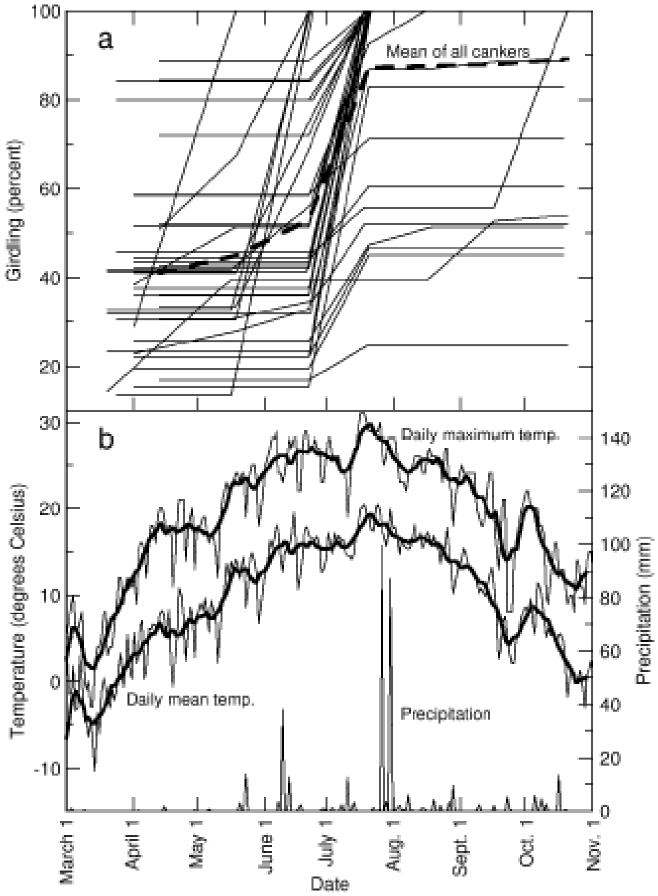


Figure 3-10 Canker growth in thinleaf alder (*Alnus icana* spp. *tenuifolia*)

(a) Development of 39 marked cankers that grew from March–October 2006, expressed as percent girdling. The bold dashed line is the average of all cankers. (b) Weather at the Gunnison 3sw weather station for the corresponding period. Heavy lines are 10-day moving averages of temperatures. From Figure 3 of Worrall, Adams, and Tharp in press.

Until recently, it was thought that higher elevation environments were relatively resistant to invasion by exotics (Pauchard et al. 2009).

Increasingly, exotic plants are reported from mountainous areas, even into the alpine (Julie Crawford, personal observations: toadflax [*Linaria dalmatica*] in Arizona and dandelion [*Taraxacum officinale*] in Colorado). Native cushion plants at altitude have even been documented to aid in establishment of such exotic species (Pauchard et al. 2009). With increased use of mountain areas for recreation, more roads and increased accessibility, and with globalization of the nursery trade, there is a greater probability that more and more cold- and drought-tolerant species from around the world will become invasive in the West, even at high elevations (Figure 3-11).

Predicted warming and drying across the West will alter the invasive potential and range of many of the exotic species of greatest concern. For example, in the arid West, increased drying likely will result in an increased invasion risk of cheatgrass (*Bromus tectorum*) across parts of Montana, Wyoming, Utah, and Colorado (Bradley 2009). This species as well as spotted knapweed (*Centaurea stoebe* ssp. *micranthos*) will likely shift distributions, as some currently occupied sites will become unsuitable with predicted increases in precipitation (Bradley 2009; Bradley et al. 2009). Other species are expected to increase in range (yellow star-thistle [*Centaurea solstitialis*] and red brome [*Bromus rubens*]), whereas still others can be expected to contract their current range (leafy spurge [*Euphorbia esula*]; Bradley et al. 2009).

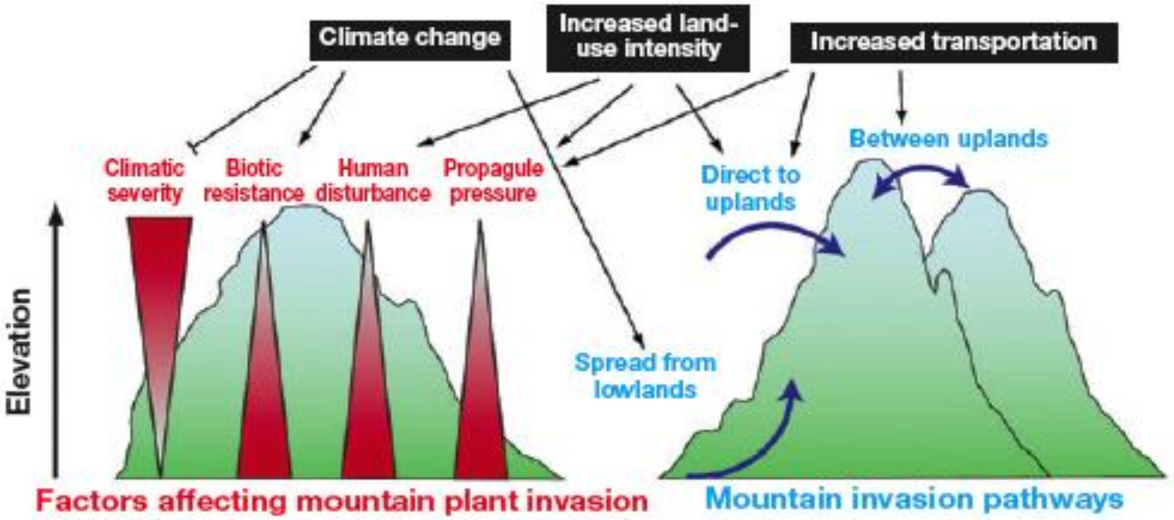


Figure 3-11 Global change and mountain plant invasion

Aspects of global change (black boxes) are likely to directly affect climatic severity, biotic resistance, human disturbance, and propagule pressure (black arrows for increasing effect, bar for decreasing effect), leading to potentially increased invasion of mountains in the near future. Global change will also affect invasion pathways (blue arrows) through increased commerce/agriculture and the introduction of non-native species directly to high-altitude areas. By reducing climatic barriers at higher elevations, climate change could increase the chances of an upward movement of lowland invasive plants. From Figure 3 of Pauchard et al. 2009.

3.2 Phenology of Flowering Plants

The correlative effects of winter snow accumulation, springtime temperatures, and snowmelt timing are associated with growing season soil moisture, plant growth, length of growing season, and the production of both flowers and seeds in many flowering species of the Rocky Mountains (Dunne et al. 2003; Forrest et al. 2010; Inouye et al. 2002; Inouye et al. 2003; Saavedra et al. 2003). Both snowmelt and flowering time of many species have come earlier in the past 40 years across the western United States (Figure 3-12; Cayan et al. 2001; Forrest et al. 2010; Forrest and Thomson 2010; Miller-Rushing and Inouye

2009). Given predicted warming and drying scenarios, flowering time of subalpine meadow species in the Rocky Mountains could advance up to 11 days for every two weeks earlier of snowmelt or 2°C increase in average spring/summer soil temperature (Dunne et al. 2003). Early flowering can result in fewer flowers per plant (Inouye et al. 2002), fewer plants flowering (Saavedra et al. 2003), and either an increased (Dunne et al. 2003) or reduced (Inouye et al. 2002) flowering period. There are no overarching rules about species response to climate change because response varies on a species-by-species and a year-to-year basis depending on global, regional, and local weather patterns (Inouye 2008; Inouye et al. 2002; Lambrecht et al. 2007). The variability in species response must first be understood before predictions of flowering plant community responses can be made (Miller-Rushing and Inouye 2009).

Early snowmelt leads to less available soil moisture later in the season (Inouye et al. 2003; Saavedra et al. 2003). In general, earlier flowering species that are cued to snowmelt timing are more affected by reduced snowpack or earlier snowpack melting (both in timing and duration of flowering) than late flowering species which are more cued to later season temperature, precipitation, and soil moisture (Dunne et al. 2003; Forrest et al. 2010; Lambrecht et al. 2007; Miller-Rushing and Inouye 2009). For example, Miller-Rushing and Inouye (2009), in their study at the Rocky

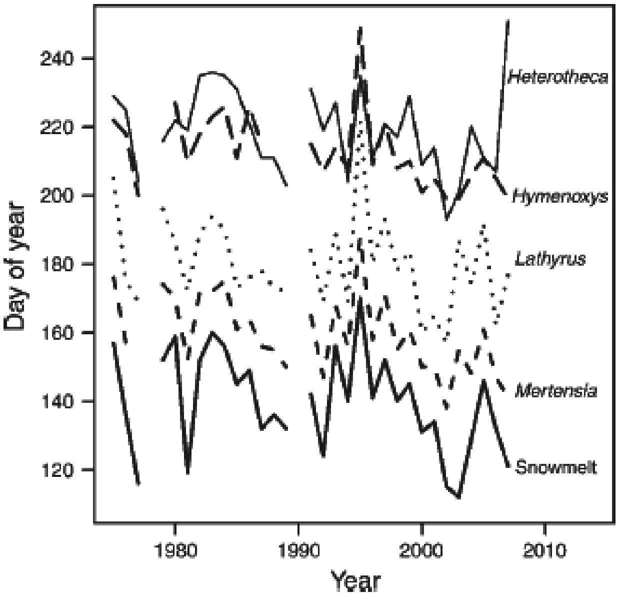


Figure 3-12 Relationship between peak flowering dates and snowmelt

Time series of peak flowering dates of the four species and snowmelt date (bottom line on graph) for the period 1975–2007 (excluding 1978 and 1990) from the Rocky Mountain Biological Laboratory, in Gothic, Colorado. From Figure 1 of Forrest et al. 2010.

Mountain Biological Laboratory (RMBL) in Gothic, Colorado, found that two early flowering species and one late flowering species (injured by frost) had decreased flower abundance in years of early snowmelt. Dry years with little soil moisture have been linked with reduced flower production in the late flowering subalpine larkspur (*Delphinium barbeyi*; Figure 3-13; Inouye 2008; Inouye et al. 2002) and in the early flowering annual rock jasmine (*Androsace septentrionalis*; Inouye et al. 2003). Similarly, Stinson (2005) found in the later flowering varileaf cinquefoil (*Potentilla diversifolia*) at RMBL that low soil moisture and high temperatures, not snowmelt date, were the largest factors leading to death of plants in suboptimal sites.

In addition to observations from long-term monitoring trends from recent decades, as mentioned above, predictions of the impacts of future climate changes on flowering plants can be made through experimental warming scenarios. Artificial warming experiments mimic the following effects that have been observed in recent "warm" years: earlier snowmelt

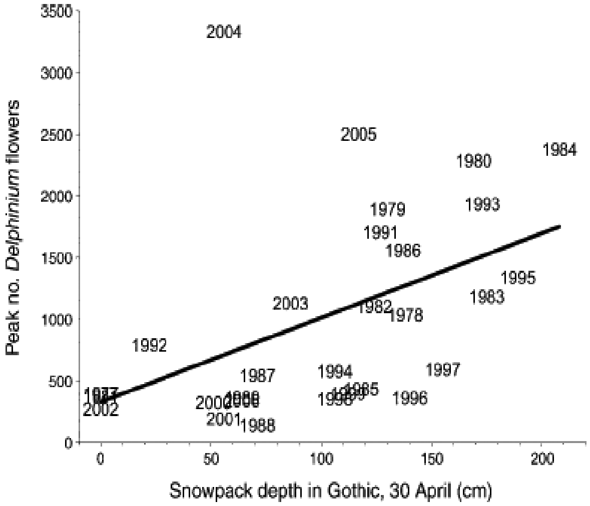


Figure 3-13 Relationship between peak abundance of early flowering *Delphinium barbeyi* flowers and the amount of snow left on the ground on 30 April of that year

From Figure 3 of Inouye 2008.

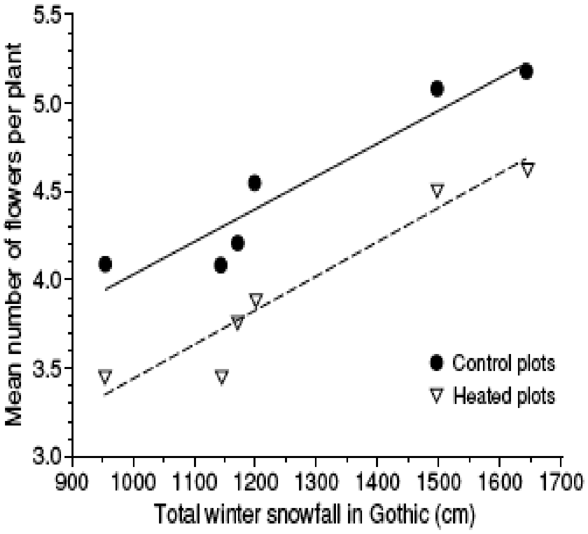


Figure 3-14 Relationship between the mean number of flowers per plant and total snowfall in the previous winter

Data from artificially heated and control plots at RMBL, 1994-1999. From Figure 6 of Saavedra et al. 2003.

in spring, increased soil temperature, lower soil moisture during the growing season, and increased nitrogen mineralization (Lambrecht et al. 2007; Saavedra et al. 2003). In a series of artificial warming experiments at RMBL, scientists have found that response depends on the species and complex physical and phonological interactions. Plants may show effects as increased, decreased, and no change in flowering in warmed plots (Lambrecht et al. 2007). In these studies, warmed plots have shown a decrease in both numbers of flowering plants and number of flowers per plant (Figure 3-14; Saavedra et al. 2003) and extension of the flowering period for early bloomers (Dunne et al. 2003) compared with control plots. Roy et al. (2004) found plants within artificially warmed plots expressed water stress and exacerbated damage from herbivore and pathogen attack; the earlier the plants emerged from snow in the spring, the more damage was incurred. Saavedra et al. (2003) report reduced biomass in studied species from limited water caused by a warming experiment. These studies indicate that, earlier snowmelt timing and reduced soil moisture later in the season can have negative or positive impacts on both early and later flowering subalpine species of the Rocky Mountains. While detailed studies outside of those at RMBL have not yet been completed in the southern Rocky Mountain or Four Corners Region, some studies are underway. Despite this, it is likely that these general results can be translated to other ecosystems in the area (David Inouye, University of Maryland, personal communication, May 25, 2010).

Although there are advantages to early spring flowering, such as high light and moisture levels early in the season, the disadvantages of possible reduced pollinator numbers or frost are significant (Forrest and Thomson 2010). Microsites such as nurse plants, snow accumulation areas, cold air drainages, shaded vs. sunnier aspects, and even small elevational differences play an important role in snowmelt timing, limiting soil moisture loss, and reducing the impacts of frost damage (Inouye 2008; Saavedra et al. 2003; Welden 1985). Earlier snowmelt may cause plants to break dormancy earlier and may leave some species more, and other species less, susceptible to freezing damage (Loik et al. 2004). Many plant species have frost-sensitive buds, ovaries, and leaves, and can have fewer flowers in early snowmelt years due to frost damage (Inouye 2000, 2008).

The continuation of variable early season frosts due to earlier loss of snow insulation in the springtime, could lead to reductions in frost-sensitive plant species in the future (Forrest and Thomson 2010; Inouye 2008; Inouye et al. 2002). This reduction of frost-sensitive species can affect animal species dependant on them for food; this has been documented in voles, squirrels, butterflies, other insects, and may be important for some large mammal species as well (Inouye 2000, 2008; Inouye et al. 2002; Saavedra et al. 2003).

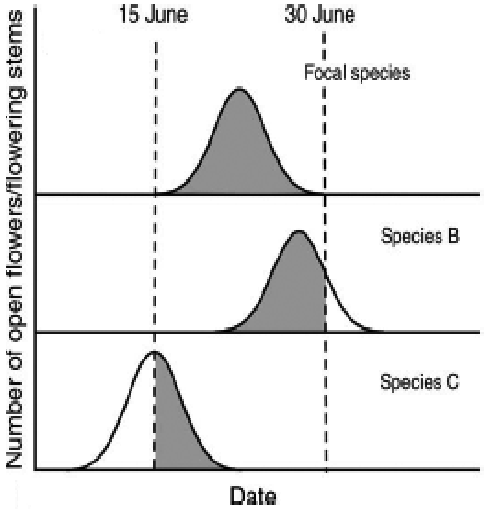


Figure 3-15 Hypothetical flowering curves

Conceptual diagram illustrating hypothetical flowering curves for three species in one year. The period of flowering overlap is the area between the dashed lines. From Figure 2 in Forrest et al. 2010.

Which plant species overlap in time may depend upon snowmelt timing and can greatly impact reproductive success (Figure 3-15; Forrest et al. 2010). Similarly, a reduction in early-season flowers can reduce or eliminate pollinators and can reduce pollination events for these and later flowering species sharing the same pollinators (Forrest et al. 2010; Saavedra et al. 2003). For example, a reduction in reproduction was documented in later flowering scarlet gilia (*Ipomopsis aggregata*) following reduced flowering in twolobe larkspur (*Delphinium nuttallianum*), an early-flowering species utilizing the same pollinator (Saavedra et al. 2003). A reduction in flowering results in reduced plant recruitment through seeds; over time even perennial species could be lost in some areas (Inouye 2008; Miller-Rushing and Inouye 2009;

Saavedra et al. 2003). For example, over the past nine years, the number of fivenerve helianthella (*Helianthella quinquenervis*) plants in study plots at RMBL has decreased significantly (Inouye 2008). Similarly, tall fringed bluebells (*Mertensia ciliata*) have disappeared from study plots at RMBL in recent years, suggesting this species could disappear from the study area completely in the future (Miller-Rushing and Inouye 2009).

There have been no seedbank studies at RMBL, and subalpine meadow seedbank literature is depauperate. Because these meadows and alpine habitats often share many species in common, we report here literature from alpine seedbanks which are thought to be stable (representing the aboveground vegetation), long-term persistent (due to regular disturbance and harsh environment), and variable across the landscape based on climate, reproductive success, seed morphology, soil structure, disturbance, and density of safe sites (Erschbamer et al. 2001; Hülber et al. 2005; Isselin-Nondedeu and Bédécarrats 2007; Klug-Pümpel and Scharfetter-Lehrl 2008; Molau and Larsson 2000; Schwienbacher et al. 2010). It is recognized, however, that alpine seedbanks contain fewer seeds than those of other biomes (Molau and Larsson 2000), that alpine seedbanks are less species-rich than aboveground vegetation (Welling et al. 2004; Molau and Larsson 2000; Körner 1999), and that alpine seedbanks decrease in both numbers of seed and in germinability with depth in soil (Klug-Pümpel and Scharfetter-Lehrl 2008; Körner 1999; Molau and Larsson 2000). In addition, seedlings require periods of moisture or moisture-holding microenvironments for germination and seedling survival is generally low (Körner 1999). These factors do not bode well for future recovery from existing seedbanks if seedrain is reduced for any reason, although Schwienbacher et al. (2010) suggest persistent seedbanks allow alpine plants to cope with unpredictability. Since many herbaceous species in the subalpine are long-lived, these species may return in a good year with ample precipitation and a good seed crop (David Inouye, University of Maryland, personal communication, July 15, 2010) coupled with seed dispersal into area with deteriorated seedbanks (e.g., via wind and ants; see Turnbull et al. 1983). Where local extirpation occurs, opportunistic, drought-tolerant, weedy, or highly mobile species will likely increase in abundance (Saavedra et al. 2003).

Figure 3-16 ***Mertensia ciliata***

Staggered flower timing to increase pollination potential. Photo by Julie Crawford.

Ultimately, whether a species is impacted by climate change or not depends on such things as its ability to self-pollinate, change resource allocation among early and late season flowers, stagger its flowering time, and alter its flowering duration (Forrest and Thomson 2010). For example bluebells (*Mertensia* spp.) have earlier-opening flowers located higher on the plant and later-opening flowers lower on the plant. In this way flowering period is extended, as are the chances of pollination later in the season if earlier flowers failed to set seed (Figure 3-16; Forrest and Thomson 2010). Late season pollination should not be counted on, however, especially given the increased probability of mid-summer drought (Inouye et al. 2003) or a late season frost (Inouye 2000). A species' ability to survive periods of variable climate will change with time and evolution (Forrest and Thomson 2010), but outcrossing and reproduction by seed is essential for speciation and decreases in flowering and seed set can have severe repercussions for the future of many species (Inouye et al. 2003; Inouye et al. 2002). Losses of dominant species and their ecosystem functions must be replaced by either subdominant co-occurring species or newly migrated species (Neilson, Pitelka, et al. 2005).

Chapter 4: Wildfire and Carbon Storage

4.1 Wildfire

Introduction

In the western United States, wildfire is considered one of, if not the, most important natural disturbances of forests, woodlands, and shrublands (Bebi et al. 2003; Kliejunas et al. 2008; McKenzie et al. 2004; Noss et al. 2006). With the intensification of managed landscapes in the early 20th century, fire suppression became a standard management strategy across the West, leading to disruptions in the natural fire cycle and reduced health of many ecosystems (Keane et al. 2002; Noss et al. 2006; Wright and Heinselman 1973). Fire was deemed not in the public interest and even foresters and ecologists were focused on stable systems leading to climax vegetation, with fire seen as an interruption to that goal (Clements 1916; Wright and Heinselman 1973). High levels of logging and grazing of domestic livestock also contributed to the alteration of fire regimes across the West (Keane et al. 2002). Today we understand that fire is an important part of many ecosystems: it aids in nutrient cycling, regulates the density and composition of vegetation, creates habitat for wildlife, structures landscape pattern, and influences water delivery across watersheds (Noss et al. 2006). In systems adapted to severe disturbance, native plant and animal species richness are often greatest during the re-growth period following disturbance (Noss et al. 2006).

Historically, natural fire regimes varied between vegetation types and geographic areas (Table 4-1). In general, sagebrush communities had small, infrequent, low-severity fire (Keane et al. 2008). Lower elevation and drier forests, such as ponderosa pine forests, had frequent low-severity fire and surface fire-adapted species (Fulé 2008; Fulé et al. 2009; Keane et al. 2008;

Table 4-1. Summary of major ecosystem historic fire regimes and return intervals in the western United States.

Ecosystem	Historic Fire Regime	Historic Fire Return Interval
Sagebrush communities	small, low-severity fire	35–80 years in productive sites; 100–200+ years in unproductive sites
Persistent piñon-juniper woodlands	large, high-severity fire	200+ (1,000 years)
Ponderosa pine	large, low-severity fire	3–14 years
Dry mixed-conifer forests	large, low-severity fire	9–30 years
Mesic mixed-conifer forest	mixture of high- and low-severity fire	intermediate between dry mixed-conifer and spruce–fir
Spruce–fir forests	large, high-severity fire	100–400 years

McKenzie et al. 2004). Higher elevation forests, wetter forests, and persistent piñon-juniper woodlands had infrequent high-severity fires (Huffman et al. 2008; Keane et al. 2008; Romme et al. 2008; Westerling et al. 2006. Intermediate forests had a mosaic of ground and crown fires (Figure 4-1; McKenzie et al. 2004; Noss et al. 2006). Those forests and woodlands adapted to infrequent high-severity fires have subsequently been less impacted by fire suppression activities (Bigler et al. 2005; McKenzie et al. 2004; Noss et al. 2006; Schoennagel et al. 2004; Westerling et al. 2006). While stand-replacing fires in higher elevation forests and woodlands today may seem as cause for alarm, they actually are well within the natural range of variability (Keane et al. 2008; Romme et al. 2008). In contrast, stand-replacing fire in sagebrush communities, ponderosa pine, and dry mixed–conifer forests may

Figure 4-1 A mosaic fire pattern in Yellowstone National Park, WY approximately one year after the major 1988 fires

From Figure 2 of Noss et al. 2006.

indeed be catastrophic and are out of the range of natural variability (Covington 2000; Fulé et al. 2009; Keane et al. 2008;).

Sagebrush and Woodlands

Historically, fire-return intervals in sagebrush ecosystems are estimated between 35 and 80 years in productive sites and 100 to 200 or more years in unproductive sites (though some researchers suggest even longer intervals; Keane et al. 2008). Sagebrush recovery following wildfire is slow; reproduction is by seed in all but one *Artemisia* species, seeds do not travel far from the mother plant, and seedbanks are not long-lived. Therefore, recovery must occur from viable seed that survived the fire or from seedrain of adjacent unburned areas. It is estimated that recovery takes a minimum of 30 years in a productive sagebrush site without exotic species competition, and longer in unproductive sites (Keane et al. 2008).

Modern sagebrush communities are commonly impacted by livestock grazing, fire suppression, encroachment by conifers that alter fuel loads and reduce structural diversity, and invasion by exotic species such as cheatgrass (*Bromus tectorum*) that alter fire regimes. It is estimated that because of these impacts, current sagebrush fire-return intervals are 20 times shorter than historic intervals and fires are more severe and damaging (Keane et al. 2008). The trend of larger and more severe wildfire in sagebrush communities is expected to continue into the future. This is largely because of the predicted range shifts of cheatgrass and increased density of the closely related red brome (*B. rubens*) across the region due to a warmer future climate (Bradley 2009; Bradley et al. 2009). Currently in the Southwest, red brome has been found in Arizona, New Mexico, and Utah (USDA Natural Resources Conservation Service 2010); it should be watched for in Colorado. With exotic plant invasion also comes competition with sagebrush seedlings and reduction in their survival. This research suggests that conversion of many sagebrush shrublands into exotic-dominated grasslands with short fire-return intervals will occur in the future.

Persistent piñon-juniper woodlands (where trees are the major component) have a very long fire-return interval of over 200 years, with some estimates as high as 1,000 years (Keane et al. 2008; Romme et al. 2008). Fires in this system are typically high-severity stand-replacing fires and are often associated with severe drought, such as experienced during the late 1500s when extensive loss of woodlands took place across the West (Keane et al. 2008). Exotic species are a threat to woodland communities also, and cheatgrass can alter fire regimes in this system as it does in sagebrush communities. It is expected in the future that with higher temperatures, increased drought, increased foliage from elevated CO_2 levels, and greater area invaded by exotic plants, that recovery from high-severity fire might include movement from woodlands to grass or shrublands. These new ecosystems may burn more frequently and limit recovery of native plant species as a result of exotic plant competition and shortened recovery between disturbances (Bradley et al. 2006; Crawford et al. 2001; Fulé 2008).

Forests

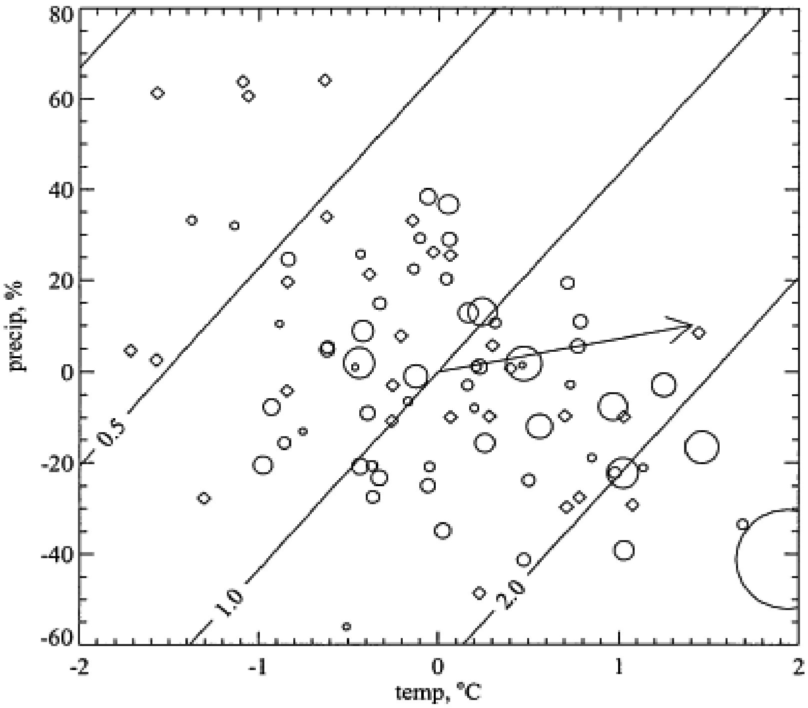
Today, Western forests are generally considered overstocked with small-diameter trees, with large old trees less common than they were historically (Figure 4-2; Fulé et al. 2002; Fulé et al. 2009; Keane et al. 2002; Mast et al. 1999;

Figure 4-2 Changes in vegetation structure and composition due to fire exclusion and logging

1909 (top) and 1979 (bottom) photographs from Bitterroot NF. From Figure 1 of Keane et al. 2002.

Wiedinmyer and Hurteau 2010). Warmer temperatures and dry conditions put these forests at great risk of large-scale stand-replacing wildfires (Fulé 2008; Liu et al. 2010; McKenzie et al. 2004; North et al. 2009; Westerling et al. 2006; Wiedinmyer and Hurteau 2010). The historic fire interval for ponderosa pine forests was 3 to 14 years. Dry mixed-conifer forest had fires every 9 to 30 years, while cool wet mixed-conifer had longer fire intervals. Spruce–fir forests had fires only every 100 to more than 400 years (Fulé et al. 2009; Kulakowski et al. 2006).

Across Western forests in the last 30 years, the numbers of fires, the severity of fires, and the area burned have all increased greatly, especially in hotter, drier years, compared to the 1970 to 1986 period (Figure 4-3; Keane et al. 2008; Keane et al. 2002; McKenzie et al. 2004; Westerling et al. 2006). Drought conditions in the inter-mountain West in 1999-2000 contributed to the worst wildfire season in the previous 50 years (NOAA 2000).



Data for one year are represented by a circle whose position indicates the summer temperature (x-axis) and precipitation (y-axis) anomalies and whose area is proportional to area burned. Largest circles tend to appear in the lower right (warm, dry summers). Contour lines indicate the mean area burned from multiple regression, as a ratio to the value at the origin. The arrow shows the direction of climatic change indicated by computer modeling. From Figure 1 from McKenzie et al. 2004.

Figure 4-3 Area burned in Colorado

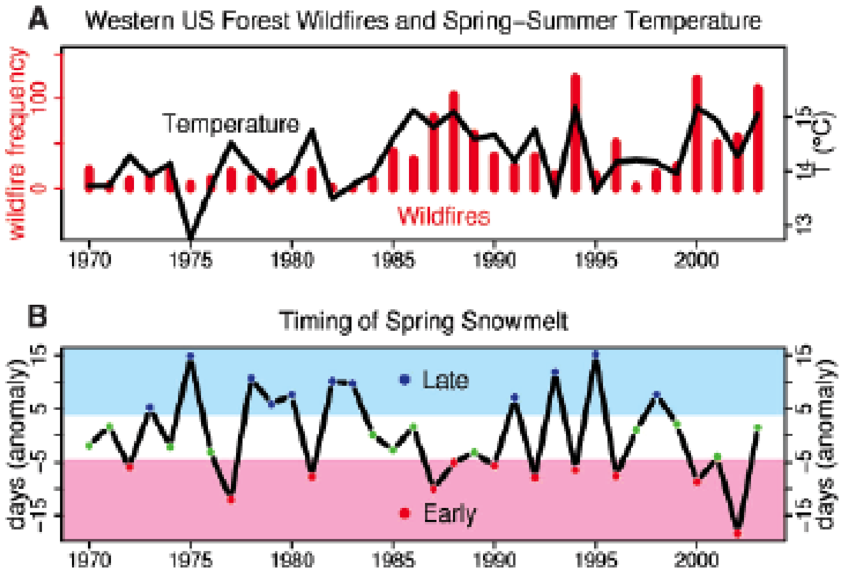
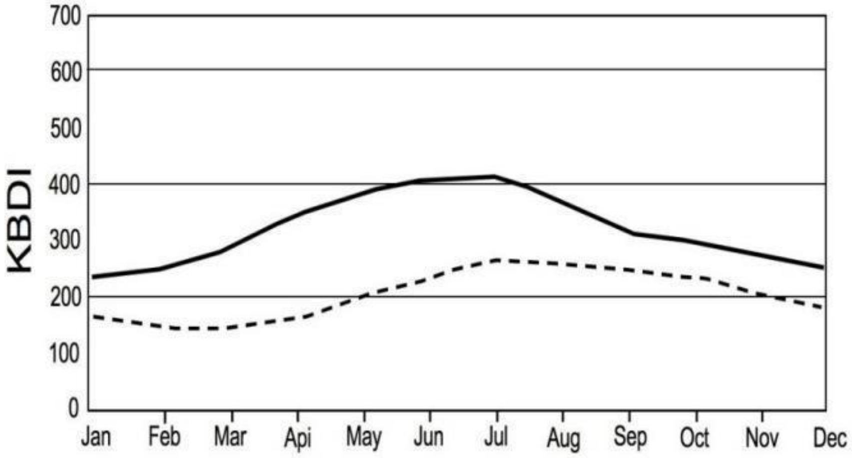


Figure 4-4 Frequency of forest wildfires and timing of spring snowmelt

(A) Annual frequency of large (>400 ha) forest wildfires (bars) and mean March to August temperature for the western United States (line). Spearman's rank correlation between the two series is 0.76 (P < 0.001). Wilcoxon test for change in mean large forest fire frequency after 1987 was significant (W=42; P<0.001). (B) First principle component of center timing of streamflow in snowmelt-dominated streams (line). Low (pink shading), middle (no shading), and high (light blue shading) tercile values indicate early, mid, and late timing of spring snowmelt, respectively. From Figure 1 of Westerling et al. 2006.

Climate may be a more important factor to future wildfire potential than disturbance or land-use/management history, as climate can impact flammability of living and dead forest biomass seasonally (Bigler et al. 2005; Westerling et al. 2006). For example, early spring snowmelt and associated mid-summer drought are highly correlated to wildfire occurrence (Figure 4-4; Westerling et al. 2006). National-scale biogeochemical models suggest that there will be an increase in wildfire throughout the western United States in the future (Aber et al. 2001). Predictions include an extension of the current fire season of May through November to become year-round in the future (Liu et al. 2010). In the Southwest, model-based projections of the Keetch-Byram Drought Index (KBDI) for the end of this century indicate there will be a 50% increase in the current KBDI level, which is already double the national average (Figure 4-5; Liu et al. 2010). KBDI, which incorporates maximum temperature and total precipitation, is essentially an indicator of soil moisture deficit and is significant in determining fire potential. Droughts that have been considered extreme in the past may become more typical in the future (Fulé 2008) and most large fires



Monthly variations of KBDI for present (dashed line) and future (solid line) projected change for the southwest United States. From Figure 4b of Liu et al. 2010.

Figure 4-5 KBDI in the southwest United States

in the contiguous United States are related to moderate to severe drought (Keane et al. 2008).

As a result of recent warming and drying, bark-beetle outbreaks have become more widespread and common in the West (also see Ecosystems section of this report). While it was previously believed that beetle-killed trees posed a great fire risk, recent research from northwestern Colorado suggests that beetle outbreak by itself is not likely to increase the extent or severity of future wildfire, but that drought at the time of, or shortly after beetle kill, may increase wildfire risk (Bigler et al. 2005; Kulakowski and Veblen 2006, 2007). Given the forecast for increased drought and increased beetle outbreaks, it is likely that these events may coincide more frequently in the future.

There are many factors that influence burn severity, such as pre-fire disturbance, species composition, stand structure, topography, and climate (Bigler et al. 2005). In research following the Rodeo-Chediski fire of Arizona, fire behavior was greatly impacted by pre-fire treatments that altered ponderosa pine forest stand structure (Keane et al. 2008). Here, portions of the landscape that had been treated with prescription fire or thinning and burning within the past 11 years displayed mostly surface fire behavior (Keane et al. 2008).

Untreated forests in this same fire incurred high mortality; long-term projections (>100 years) are for these forests to shift to shrublands.

In a series of studies in subalpine forests of Colorado, researchers have shown that severe wildfire can alter species composition and stand structure, thereby influencing future disturbances for decades or even centuries (Bigler et al. 2005; Kulakowski and Veblen 2006, 2007). For example, recovery from stand-replacing fires may include the production of young stands that are less susceptible to many types of disturbances (e.g., flammability increases with the age of a forest; Bigler et al. 2005). Recovery via succession to another cover type may alter species composition from a more flammable forest (e.g., spruce and fir trees retain their lower branches due to a greater tolerance for shade) to a less flammable forest (e.g., most pine species shed their lower branches due to an intolerance of shade). Similarly, succession to aspen forest may act as a deterrent to future fire, as aspen have a high moisture content and mesic (moist) understory (Bigler et al. 2005). An important lesson can be learned, however, from a recent study by Schoennagel et al. (2004) of the 2002 Hayman Fire, which burned nearly 56,000 ha of Colorado subalpine forests. Here, extreme climatic conditions (high wind and low relative humidity) were more important to fire behavior than stand structure or species composition from previous fires or stand treatments. In this fire, a large portion of the area burned experienced high-severity fire leading to high overstory mortality.

A landscape’s resilience, or capability to withstand severe disturbance, lies in its heterogeneity. Heterogeneity in landscapes promotes heterogeneity in species, habitat structure, and ecosystem functions; these in turn increase redundancy, complexity, and resilience to severe disturbances (Schoennagel et al. 2008). Some ecosystems, such as those dominated by serotinous-coned trees, are more resilient to severe wildfire (e.g., Yellowstone National Park lodgepole forests), while others, like ponderosa pine forests of the Southwest, are more likely to convert to non-forest systems without management effort pre- or post-fire (Schoennagel et al. 2008). For example, in recent fires in northern Arizona, areas that burned severely converted from ponderosa pine forest to grass- or shrub-dominated landscapes following the fires (Figure 4-6; Crawford et al. 2001; Griffis et al. 2000; Keane et al. 2008). In these systems, following high-severity fires, and with increased competition from a recovering herbaceous understory, tree species can take decades to begin regrowth and may remain in a changed state for the long term (Figure 4-7; Bradley et al. 2006; Crawford et al. 2001; Dore et al. 2008; Reinhardt and Holsinger 2010). With climate change, post-disturbance systems may be incapable of returning to their previous states if: a) tree species and genotypes are outside of their climatic niche (Baron et al. 2009; Ryan et al. 2010), b) pre-disturbance species are replaced with successional species such as aspen or vegetatively reproducing shrubs such as Gambel oak (Bigler et al. 2005), or c) if exotic or early successional native herbaceous and grass species outcompete tree seedlings (Crawford et al. 2001; Griffis et al. 2000).

Certainly management strategies today can greatly impact these ecosystems in the future and fires that are suppressed today could lead to more severe disturbance in the future (Kulakowski and Vebler 2006). With increased drought we can also expect increased fire risk in the future. It is important to understand the natural fire cycles of the complex ecosystems of the western United States in order to better prepare for this future.

Figure 4-6 Northern Arizona ponderosa pine forest after a high-severity fire

Example from northern Arizona following 1996 high-severity fire: ponderosa pine forest has been converted to a forb-dominated landscape which persists today. Photo by Julie Crawford.

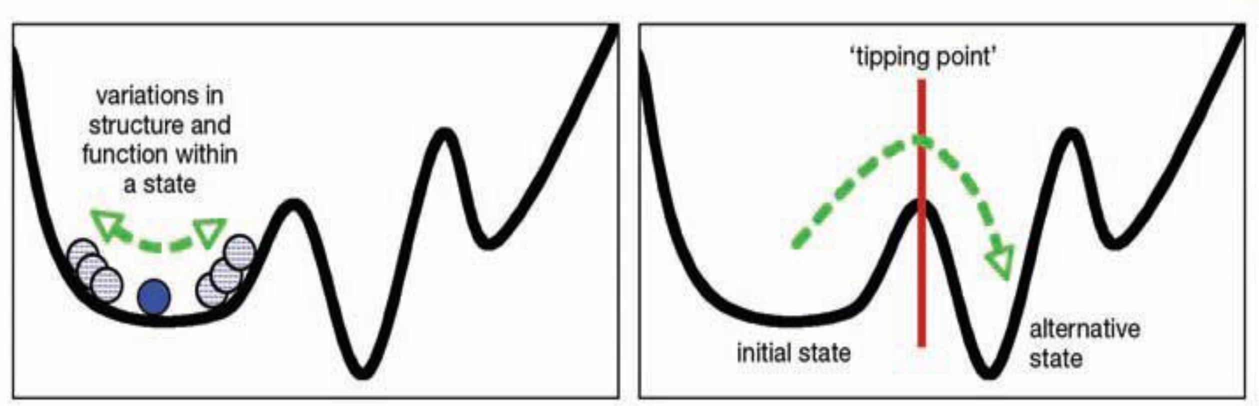


Figure 4-7 Ball and Cup Theory

Change in ecosystem structure and function in response to changes in stress levels or disturbance regimes may be gradual and linear up to a certain point(s), and then change dramatically and profoundly. Once in an alternate state, plant cover, composition and seedbanks, and soil physical properties, nutrient status and water holding capacity, etc., may have been altered to the point that it is difficult for the system to revert to its previous state even if the stresses or disturbance causing the change are relaxed. From Ryan et al. 2008:116.

4.2 Carbon Storage, Fire, and Management

Forests are very important carbon sinks and take up between 9% and 19% of the U.S. anthropogenic fossil fuel emissions each year (Ryan et al. 2010; Wiedinmyer and Hurteau 2010). Current forests are recovering from past land uses such as agriculture, pasture, or timber harvest. Fertilization by increased atmospheric deposition of nitrogen and atmospheric concentrations of carbon dioxide may also be contributing to growth. The resulting current carbon sink is not expected to continue indefinitely, however (Ryan et al. 2010), and interest in retaining or increasing forests as a carbon sink is growing. Despite the importance of fire as a natural disturbance agent, less than 6% of all emissions nationwide come from fire (Figures 4-8, 4-9; Wiedinmyer and Neff 2007). It is estimated that 78% of all carbon emissions from fires in the western United States, however, stem from fire in coniferous forests (Wiedinmyer and Neff 2007).

Various management alternatives are available (such as avoiding deforestation, planting trees, reducing harvesting amount or interval, increasing forest growth, fuels management to reduce fire threat, and using wood biomass to replace fossil fuel energy), but often the full implications of these possible actions on carbon storage and other resource values are not understood. Furthermore, carbon inventories have not been conducted for many forests and may vary significantly even within land cover types, depending on conditions. Similar attention already has been directed at agricultural systems to lower emissions of greenhouse gases via practices such as no-tillage, no or reduced fertilizer application, improved cropping systems, land restoration, land use change, and irrigation and water management. In contrast to forests, U.S. agriculture is a net greenhouse gas source, providing about 7%–8% of the emissions, but just 3%–4% percent of the carbon uptake (EPA 2007).

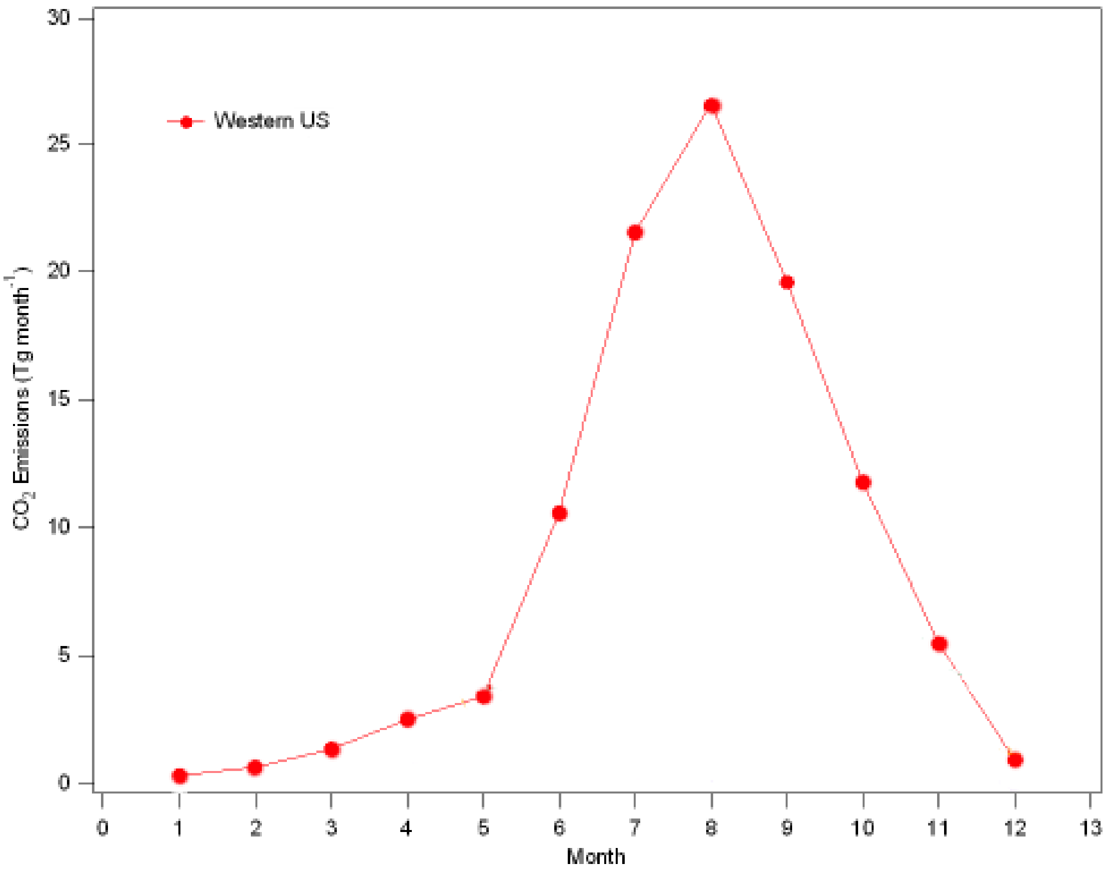


Figure 4-8 CO2 emissions from fires in the western United States

Annually averaged CO2 emissions (2002–2006) from fires the western United States (AZ, CA, CO, ID, MT, NM, NV, OR, UT, WA, WY). Adapted from Figure 2 of Wiedinmyer and Neff 2007.

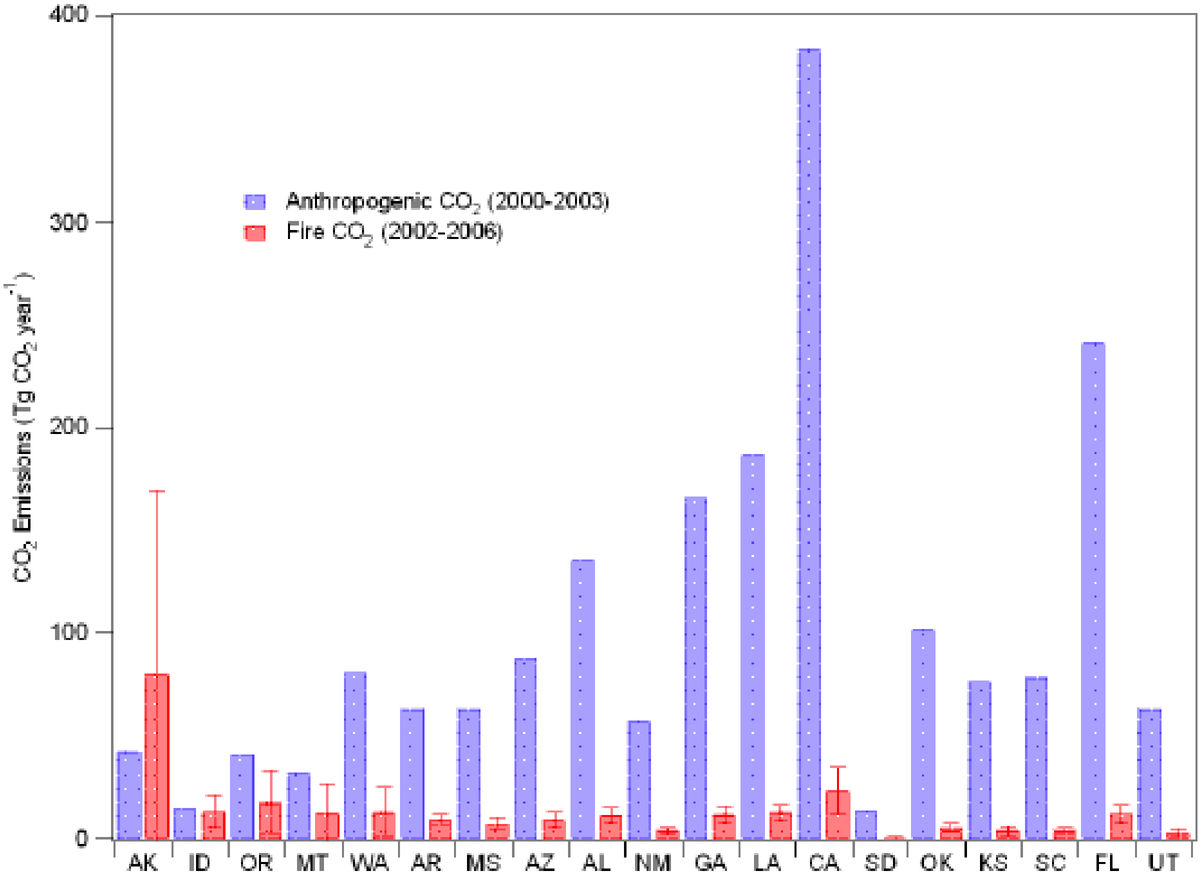


Figure 4-9 Anthropogenic and fire CO2 emissions

Annually-averaged anthropogenic emissions (2000-2003) of CO2 and fire emissions (2002-2006) for states where average fire emission is >5% of the states' anthropogenic emissions. Error bars represent the standard deviation. From Figure 4 of Wiedinmyer and Neff 2007.

Ecosystems take up and store atmospheric carbon in woody and leafy plant material, litter, duff, and soil. Carbon storage is not static in ecosystems. Carbon dioxide uptake fluctuates between vegetation communities throughout the season in relation to precipitation (Dore et al. 2008; Scott et al. 2006). Carbon cycling can vary following shifts in species composition (Saleska et al. 2002). Following disturbance, carbon stocks are transferred from living to dead biomass (Dore et al. 2008; Morehouse et al. 2008; Wiedinmyer and Neff 2007). Climate-related changes resulting in earlier spring onset and summer droughts have been linked to slower uptake of atmospheric CO_2 in high elevation coniferous forests (Hu et al. 2010). Similarly, Kueppers and Harte (2005) found that subalpine forests in Colorado could lose up to half of their current carbon storage as a result of projected warming and drying. In artificial heating experiments at the Rocky Mountain Biological Laboratory in Colorado, Saleska et al. (2002) found that shifts in species composition from highly productive forbs to less productive shrubs resulted in reduced high quality litter, decomposition, and soil carbon storage.

Carbon storage varies by vegetation type; in general, croplands store less carbon than grasslands, grasslands store less than shrublands, shrublands less than woodlands, and woodlands less than forests. Conversion of marginal croplands into grasslands can reduce agricultural emissions and, if managed properly, can result in carbon sinks over time (EPA 2005). In addition, grasslands that employ grazing management strategies such as reduction of stocking rate and addition of nitrogen-fixing legumes increase both the quantity and quality of grass and, therefore, increase carbon sequestration potential (Brown et al. 2010; EPA 2005). In forests that have been permanently converted to shrub or grasslands following catastrophic events, the post-disturbance environment will become a lesser carbon sink (Reinhardt and Holsinger 2010; Ryan et al. 2010) and possibly a future carbon source. However, shrublands recover more quickly than forests and the rate of carbon accumulation in at least one shrub, sagebrush (*Artemisia tridentata*), following fire is reported to be greatest in the first 20 years of recovery (Cleary et al. 2010). Even though grasslands and shrublands sequester less carbon than forests, because of their enormous geographic area in the western United States, they are important ecosystems for carbon sequestration (Brown et al. 2010; Gilmanov et al. 2003). Similarly, piñon-juniper woodlands make up 18% of the Southwest landscape and recent widespread regional dieback in this system resulted in a loss of 4.6 teragram (Tg) total aboveground carbon, roughly 39 times the concurrent carbon loss to wildfire and management treatments within or near to the national forests of the study region (Figure 4-10; Huang et al. 2010). Because of the dry climate in this region,

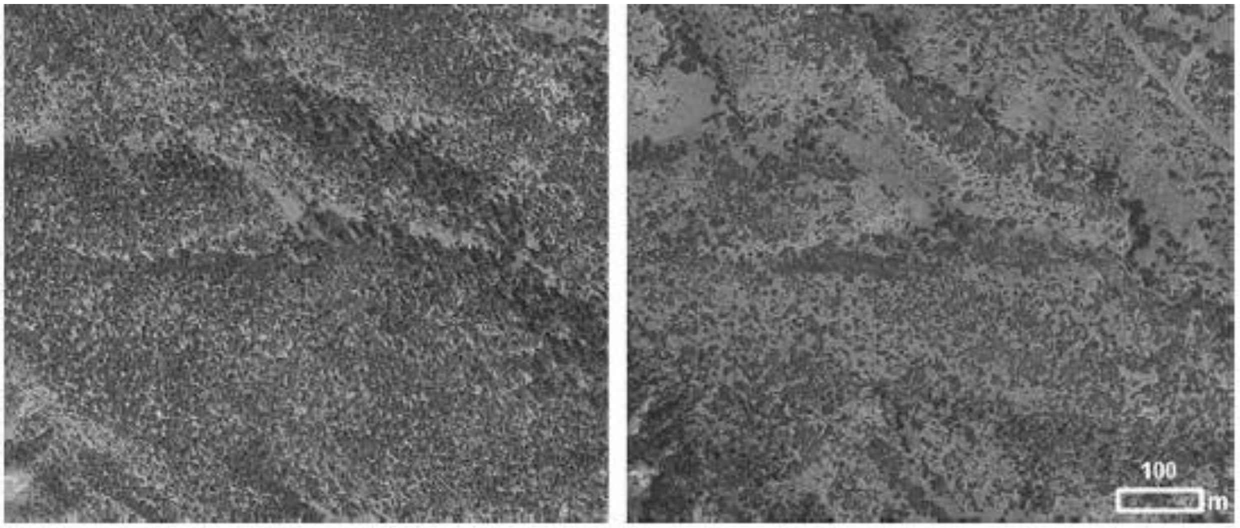


Figure 4-10 Dieback of piñon-juniper vegetation on the Colorado Plateau

Illustrations of tree loss during the dieback period (2001–2005) in the piñon–juniper vegetation of the Colorado Plateau due to dieback. Left: pre-drought; right: post-drought (at 37.15°N, 107.95°W) using 1-m resolution aerial photographs obtained in 1990s and 2005. From Figure 6e-f of Huang et al. 2010.

decomposition of woody debris is slow and total carbon release could take decades; carbon storage will be impacted until the system recovers (Huang et al. 2010).

Forests can sequester carbon for centuries, though at a reduced rate with age. Forests that have burned according to a more natural fire cycle are better carbon sinks than those that have experienced fire suppression (North et al. 2009). In the past, larger trees held large carbon stocks; in today's dense forests, more carbon is stored in smaller-diameter fire-sensitive tree species (North et al. 2009; Ryan et al. 2010; Wiedinmyer and Hurteau 2010). With high aboveground carbon storage—prevalent in overstocked, dense forests—comes greater carbon loss following catastrophic fire (Cleary et al. 2010; Dore et al. 2008). Following fire and other disturbance, ecosystems will shift from being carbon sources back to being carbon sinks through the regrowth of vegetation (Reinhardt and Holsinger 2010; Ryan et al. 2010; Wiedinmyer and Neff 2007), but the amount of time this takes varies with both habitat type and location (Cleary et al. 2010; Dore et al. 2008; Huang et al. 2010). In a study of twoneedle piñon (*Pinus edulis*) and Utah juniper (*Juniperous osteosperma*) loss in southwestern Colorado from disease and insect outbreaks following drought, Huang et al. (2010) suggest recovery of live carbon storage could take more than a decade in young stands and even longer for mature stands. In ponderosa pine (*Pinus ponderosa*) forests of northern Arizona, Dore et al. (2008) found a high-severity burn area was still a moderate carbon source ten years following the fire. Recovery of carbon stocks in a forest after fire may take over 100 years (Figure 4-11; Ryan et al. 2010).

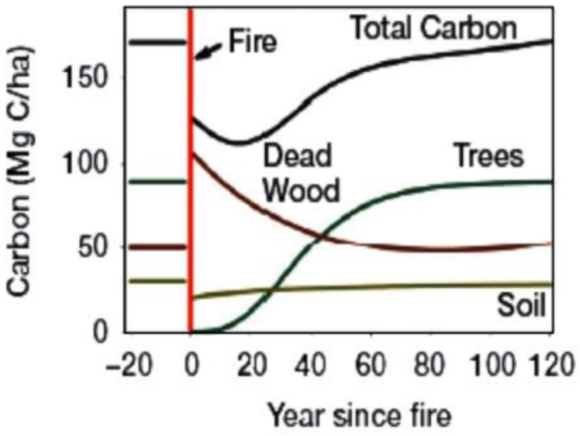


Figure 4-11 Carbon sequestration following a fire

If a forest regenerates after a fire, and the recovery is long enough, the forest will recover the carbon lost in the fire and in the decomposition of trees killed by the fire. Here, carbon stored in forests as live trees, dead wood, and soil change after fire. Carbon sequestration can continue for centuries, though at a reduced rate. From Figure 3 of Ryan et al. 2010.

Recovery depends on both the severity of the disturbance and the short- and long-term climatic conditions. Given predictions of climate warming and drying, there may be higher probability of additional disturbances, such as beetle outbreak or high-severity fire, occurring before the forest has recovered from an initial disturbance, resulting in a net decrease in stored carbon (Ryan et al. 2010). It is important to remember that forests, like other communities, are dynamic biological systems that gain and lose carbon through a balance of processes, and their ability to store carbon varies over the time since disturbance (Figure 4-11). This variability may be balanced spatially as well by the many stages of recovery found across a larger landscape. For example, even extensive disturbance affects only a fraction of the landscape at any one time, and this heterogeneity on a landscape scale plays an important role in buffering against large disturbances (also see the Wildfire section; Kulakowski and Veblen 2007). Many researchers suggest that the restoration of fuel and forest structure and burning patterns similar to historic levels is more cost-effective, both in terms of dollars and carbon, than suppressing catastrophic fire (Finkral and Evans 2008; Fulé 2008; Keane et al. 2008; Reinhardt and Holsinger 2010). Restoration treatments may contribute significantly to attaining a more healthy presettlement condition in some, but not all, forests (Figure 4-12; Mast et al. 1999; Schoennagel et al. 2004). For example, on the largest and most severe wildfire in the Southwest to date, the 2002 Rodeo-Chediski fire of Arizona, areas that were treated with prescription burn and/or thinning prior to the wildfire experienced mostly surface-fires and did not suffer crown-fire, as the rest of the burn area did (Keane et al. 2008). In contrast, during the 2002 Hayman fire in subalpine forests of the Front Range of Colorado that historically burn with high severity, pre-fire treatment did little to alter fire behavior on severe weather days when the fire had high growth and only made a moderate difference during days of small fire growth (Schoennagel et al. 2004). In the Hayman fire example, however, treatment areas were small and that may have contributed to their lack of effectiveness (Schoennagel et al. 2004). Care should be given with regard to prescription burns or thinning

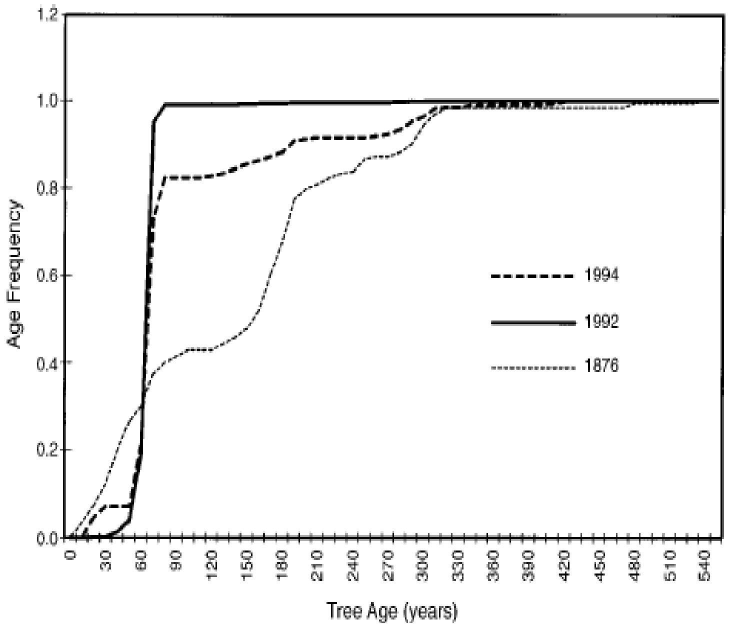


Figure 4-12 Cumulative frequency distributions of ponderosa pine age structure at Gus Pearson Natural Area, AZ

Cumulative frequency distributions of ponderosa pine age structure at Gus Pearson Natural Area, AZ in 1876 (presettlement forest), 1992 (after fire exclusion), and 1994 (after restoration treatment). From Figure 6 of Mast et al. 1999.

treatments as, in general, these treatments are not useful in all ecosystems (Schoennagel et al. 2004), are not enough to reverse wildfire trends (Westerling et al. 2006), and may facilitate exotic plant invasion (Romme et al. 2008).

Wiedinmyer and Hurteau (2010) suggest prescription fire, because it reduces the probability of catastrophic fires, and can reduce CO_2 emissions in the western United States by as much as 25%, or even more in some systems (Figure 4-13). Ryan et al. (2010) conclude more research is needed regarding thinning treatments, as some studies do not show their benefit, though many do. For example, in seven habitat types of the northern Rocky Mountains, crown fire events were reduced following prescription burn, and all but eliminated following thinning (Reinhardt and Holsinger 2010). Although treated stands had lower carbon stocks immediately following wildfire, they had more carbon in living vegetation and less in dead, as a result of reduced tree mortality from treatment (Reinhardt and Holsinger 2010). Here, treated stands had smaller additional post-fire carbon stock declines and took less time to revert back to a carbon sink. Similarly, Finkral and Evans (2008) found a less severe decrease in carbon stocks following thinning and catastrophic burn over no treatment and catastrophically burned ponderosa pine stands. Fulé et al. (2009) and North et al. (2009) suggest that in dry warm mixed-conifer forests, understory thinning to reduce surface fuels, thinning small-diameter trees and ladder fuels, and removal of only fire-sensitive species in larger size classes are good ways to retain carbon sinks and reduce emissions of high-severity fire. Similarly, modeling of eight different fuel treatments on tree-based carbon storage and release over a century from Sierra Nevada mixed-conifer forests showed that in wildfire-prone forests, carbon stocks were best protected by fuel treatments that produced a low-density stand structure dominated by large, fire resistant pines (Hurteau and North 2009). Reinhardt and Holsinger (2010) suggest fuel treatments are important to reduce competition for moisture and light and allow the growth of individual trees into large size classes.

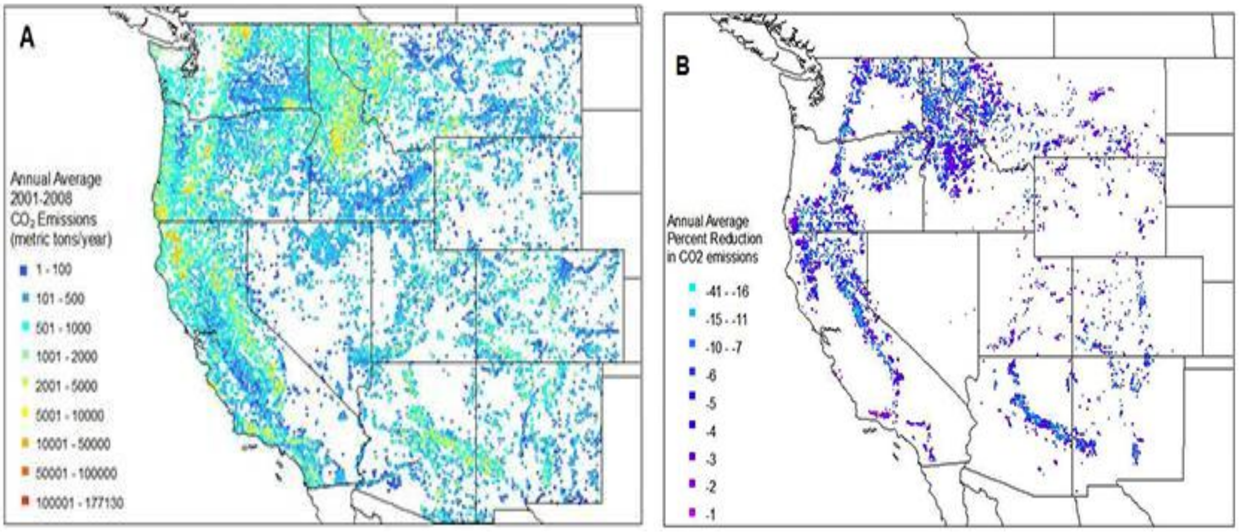


Figure 4-13 Prescription burns can reduce CO2 released by western wildfires

A) Estimated annual CO2 emissions from 2001–2008, B) Extent to which those emissions could have been reduced by a systematic program of prescribed burns. By Christine Wiedinmyer, NCAR; courtesy Environmental Science & Technology (http://www2.ucar.edu/news/prescribed-burns-may-help-reduce-us-carbon-footprint).

One important consideration, however, is that restored landscapes require maintenance of ecological processes; for example, fire should be incorporated into management plans for post-restoration maintenance (Noss et al. 2006). In addition, Noss et al. (2006) recommend post-fire environments should not be salvage-logged or reseeded with exotic species for erosion control. In this way, standing dead wood remains a small carbon sink and there is the greatest possibility for natural recovery.

Future land management must focus on ecological processes (Millar et al. 2007), must be adaptive (Baron et al. 2009), and should include: a) treatment to reduce severe fire potential in appropriate ecosystems, b) management for increased soil conservation, c) restoration of degraded lands, d) preservation of unfragmented lands, e) afforestation, and f) reforestation (Brown et al. 2010; Breshears and Allen 2002). There is controversy, however, as to where to place efforts. Gilmanov et al. (2003) suggest that improving rangeland management in the western United States could increase carbon sequestration more quickly and at lower cost than creating forest through afforestation. Yet, afforestation of forests cleared in the 19th and 20th centuries would sustain carbon storage rates for decades (Ryan et al. 2010). Some suggest that relying on a forest carbon sink in the future is tenuous (Wiedinmyer and Hurteau 2010). Ultimately, reduction of emissions is a more important factor in balancing global carbon emissions (Breshears and Allen 2002). Relying on carbon sequestration is not a solution, but the management of healthy grasslands, shrublands, and forests for carbon sequestration is an important part of the balance.

While little information is currently available for carbon storage in southwestern Colorado, there are a few studies underway. For example, a collaborative effort between University of Colorado—Boulder and the San Juan Public Lands Center (USFS/BLM) will conduct a carbon inventory and management demonstration project with three components: (1) conducting a baseline carbon stock assessment across an elevational gradient for the San Juan Mountains, (2) developing carbon stock distribution maps for the area, and (3) evaluating the potential implications of emerging federal carbon policy approaches on resource management. For updates on this and other studies, see <http://moab.colorado.edu/ClimateImpactsPublicLands/Welcome2.html>.

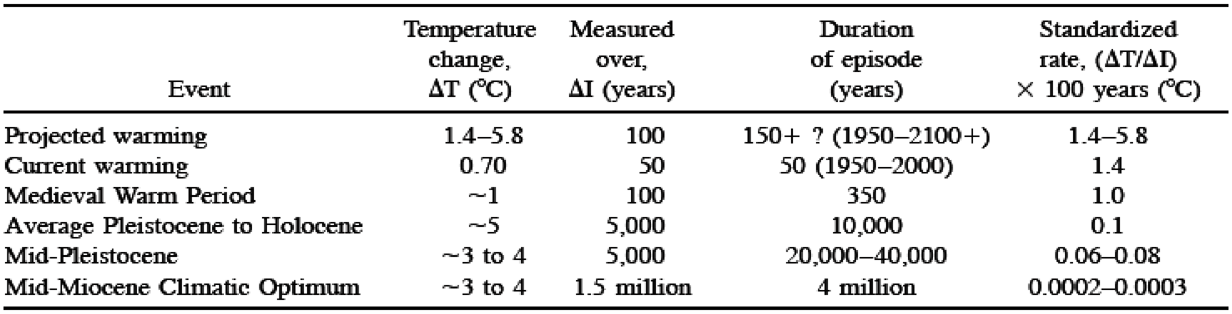
Chapter 5: Wildlife

Introduction

Animal species have many mechanisms for coping with alterations in their environment. Responses include migration and shifts in range, abundance, food sources, and timing of seasonal events (i.e., phenology). Those species/individuals that cannot change quickly enough, lack mobility, or are at the elevational/latitudinal limits of their distribution may be extirpated locally or go extinct globally as a result of projected environmental changes associated with global warming. Extirpations and extinctions of animal species associated with changing climatic conditions have occurred over evolutionary history (Barnosky et al. 2003; Carey 2009). The difference between climate change in the past century and that experienced over previous millennia is the rate of change, with the most rapid warming of the last millennium occurring in the last 30 years of the 20th century (Table 5-1; Barnosky et al. 2003; Carey 2009). Considering both past documented and projected future warming, we will soon exceed rates of global climate change known for all of mammalian history (Barnosky et al. 2003). Because many species today are descendants of species that survived previous climate change events, they may have inherited physiological or behavioral mechanisms that will help them to cope with altered temperature and precipitation regimes (Carey 2009). It is suggested, however, that at least with bird species, response to climate change is more behavioral than evolutionary (Torti and Dunn 2005).

There is concern in the media and among some scientists that studies attempting to statistically correlate observed phenomena with recorded climate events are anecdotal and not proof of causality (Carey 2009). The number of observations across the globe that report climate-related shifts in species phenology and distribution provide weight of evidence, however (Parmesan and Yohe 2003). There is a growing body of scientific support that animal species

Table 5-1 Global warming episodes.



Event	Temperature change, ΔT (°C)	Measured over, ΔI (years)	Duration of episode (years)	Standardized rate, (ΔT/ΔI) × 100 years (°C)
Projected warming	1.4–5.8	100	150+ ? (1950–2100+)	1.4–5.8
Current warming	0.70	50	50 (1950–2000)	1.4
Medieval Warm Period	~1	100	350	1.0
Average Pleistocene to Holocene	~5	5,000	10,000	0.1
Mid-Pleistocene	~3 to 4	5,000	20,000–40,000	0.06–0.08
Mid-Miocene Climatic Optimum	~3 to 4	1.5 million	4 million	0.0002–0.0003

"Projected warming" is from IPCC and "Current warming" reflects measured values from 1950 to 2000. From Table 1 of Barnosky et al. 2003.

worldwide are being impacted by shifts in climate, but the number of such studies from the southern Rocky Mountains is limited. As such, literature focusing on species that occur in this area, but involving studies conducted in other parts of the United States, is drawn upon for this review.

Climate change affects animal species both directly and indirectly and may help or hurt species or populations over time (e.g., Coakley et al. 1999; Wang et al. 2002). Direct impacts include loss of habitat (e.g., the drying of water sources for aquatic animals), heat or drought stress (e.g., smaller body size/loss of reproductive ability), exposure to extreme cold following premature snowmelt, alterations in food availability and quality both in time and space, and the number of competitors or predators. Indirect impacts may be of even greater consequence to some species (Yang and Rudolf 2010) and include alteration of environmental cues (e.g., photoperiod shifts), increases in disease and pest organisms, alteration in the interaction with other individuals of the same species (e.g., mating) or of other species (e.g., pollination), and loss of genetic variation due to decreases in both the size of and movement between suitable habitats.

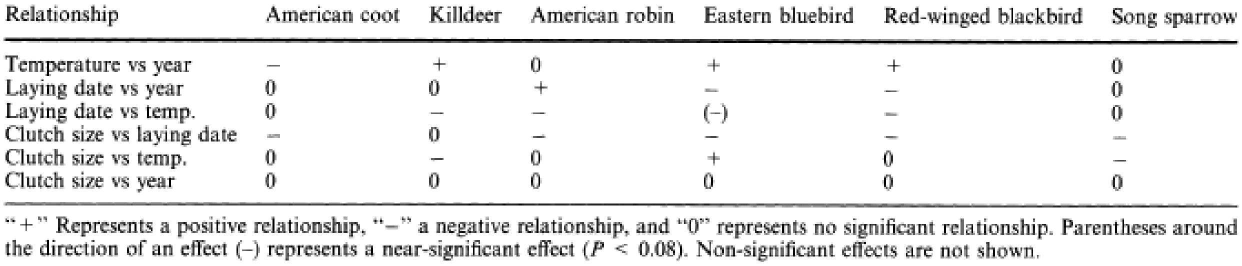
Figure 5-1Yellow-bellied marmot (*Marmota flaviventris*)

Example of species dependent on photoperiod, yellow-bellied marmot, which has been impacted by climate change. Photo credit David Inouye 2010.

Long-lived and larger species dependent upon photoperiod, such as marmots (e.g., Figure 5-1) and bighorn sheep (*Ovis canadensis*) may find it more difficult to adapt to climate change than short-lived, less dependent species (Bronson 2009; Hass 1997). Photoperiod in bighorn sheep is a major factor influencing the timing of breeding and birthing. If lambs are born too early in the season they are vulnerable to severe weather and their mothers may not find adequately nutritious food to support lactation; if born too late in the season, they may not gain enough mass to survive the winter (Hass 1997).

Photoperiod is not a cue for many small mammals of deserts and dry grasslands in the Temperate Zone, as they are more in tune with sporadic rainfall and food availability and may reproduce following these unpredictable events (Bronson 2009). Any species with a more diverse diet is likely to fare better than those with restricted diets (Harper and Peckarsky 2006; Torti and Dunn 2005). Bird species that can breed following unpredictable food availability are likely to cope best with climate change (Carey 2009). Many bird species, however, are sensitive in various aspects of reproduction in response to climate (Table 5-2; Torti and Dunn 2005). Species that hibernate or migrate long distances may be adversely impacted, though mobility over short distance is an important advantage (Carey 2009; Harper and Peckarsky 2006; Inouye et al. 2000). Species or populations that are stressed, are already in decline, or are at the margins of their range will face the biggest challenges and also be more vulnerable to disease (Alford et al. 2007; Coakley et al. 1999; Corn 2005).

Table 5-2 Sensitivity of select bird species to temperature changes.



Relationship	American coot	Killdeer	American robin	Eastern bluebird	Red-winged blackbird	Song sparrow
Temperature vs year	–	+	0	+	+	0
Laying date vs year	0	0	+	–	–	0
Laying date vs temp.	0	–	–	(–)	–	0
Clutch size vs laying date	–	0	–	–	–	–
Clutch size vs temp.	0	–	0	+	0	–
Clutch size vs year	0	0	0	0	0	0

"+" Represents a positive relationship, "–" a negative relationship, and "0" represents no significant relationship. Parentheses around the direction of an effect (–) represents a near-significant effect ($P < 0.08$). Non-significant effects are not shown.

Laying date is negatively related to warmer temperature; clutch size is negatively related to laying date. From Table 5 in Torti and Dunn 2005.

Large Mammals

Ungulate species such as mule deer (*Odocoileus hemionus*) and elk (*Cervus elaphus*) increase in population size in years with early onset of spring when mobility is increased and forage potential is high; harsh winters negatively impact both species, and warmer temperatures may lead to larger populations (Bartmann and Bowden 1984; Wang et al. 2001). Should climate change result in overpopulation of either species, less palatable and lower-quality woody plants may be utilized, meadows could be over-grazed and lose plant diversity, and starvation could ensue. Many alpine meadow species are intolerant of ungulate grazing and overgrazing could result in lower seed production (e.g., see Galen 1990). Selective grazing by ungulate species of flowering forbs over grasses can also lead to losses of forb seed production and changes in plant species composition (Hülber et al. 2005). Concentrations of large grazing animals may encourage tree invasion into high elevation meadows (Moir and Huckaby 1994). Tree invasion or the loss of herbaceous diversity in these meadows greatly reduces habitat quality for large mammals such as black bears (*Ursus americanus*) (Moir and Huckaby 1994).

In addition, increased drought may lead to large-scale stand-replacing wildfire, resulting in the loss of late successional habitat for wildlife species dependent on old forests such as the Canada lynx *(Lynx canadensis*; McKenzie et al. 2004). In addition, Canada lynx, with its long legs and large well-furred feet, is adapted to and dependent upon cold winters with deep snow for extended periods (U.S. Fish and Wildlife Service [USFWS] 2009). Continued warming that threatens these climatic conditions also would have adverse effects on Canada lynx populations (USFWS 2005). The primary prey of Canada lynx is the snowshoe hare (*Lepus americanus*); this species may also be adversely affected by reductions in snow, however it can also benefit from climate change as disturbances such as wildfire and beetle outbreaks may result in greater snowshoe hare habitat and therefore greater foraging habitat for Canada lynx (USFWS 2005). In 2009, a lawsuit was filed against the U.S. Fish and Wildlife Service, claiming that recent expansions to the habitat of the Canada lynx are not sufficient to protect the endangered animal from climate change.

Small Mammals

As vegetation communities shift upward in elevation to adapt to drying and warming throughout the West, so will predator, prey, and competitor species move upward from habitats that become unsuitable (Lawler et al. 2010). Already, observations at the Rocky Mountain Biological Laboratory in Gothic, Colorado (near Crested Butte) have shown upward movement of the Wyoming ground squirrel (*Spermophilus elegans*) and red fox (*Vulpes vulpes*; David Inouye, University of Maryland, personal communication July 23, 2010).

For some species such as American pika (*Ochotona princeps*), however, there is limited upward movement possible from their alpine habitats. The American pika, which lives in western North America's alpine rock slopes, is thought to be especially vulnerable to climate warming. Known for its unique behavior of "haying" vegetation under rocks for winter consumption, this non-hibernating species must maintain an extremely high metabolism to combat cold winter temperatures (Smith and Weston 1990). While this adaptation is beneficial in winter, high summer temperatures can be fatal due to body temperatures that lie just below the threshold for heat stress (Li et al. 2001; Yang et al. 2006). These talus obligates can utilize their habitat for behavioral thermoregulation (Smith and Weston 1990). In hot temperatures, American pikas can retreat into the depths of rock slopes, where temperatures average 4°C degrees cooler than those on the surface (Millar and Westfall 2010). Despite this behavioral adaptation, population dynamics are influenced by temperature and precipitation, as evidenced by studies documenting a strong influence of snowmelt patterns on collared and American pika survival and reproduction (Kreuzer and Huntly 2003; Morrison and Hik 2007; Smith 1974).

Recent evidence suggests that the American pika is responding negatively to climate change through local population extirpation in parts of its range (Beever et al. 2003; Millar and Westfall 2010). In the Great Basin, extirpations have been successfully modeled as a function of chronic heat stress (average summer temperature) and extreme cold stress (number of days below -10°C; Beever et al. 2010). These results have led to a status review of the species by the USFWS for possible listing under the Endangered Species Act (ESA). The USFWS ruled not to include the American pika in the ESA, but an appeal is pending. In the Sierra Nevada, Millar and Westfall (2010) found that wetter, cooler sites with ice/flowing water beneath the talus were more likely to support American pika populations than those lacking these characteristics. Ongoing research in Wyoming, Colorado, and New Mexico is addressing the question of whether populations in the southern Rockies have shifted as a result of modern climate change. In the summer of 2008, 69 historically recorded populations of American pika were revisited to document changes in distribution.

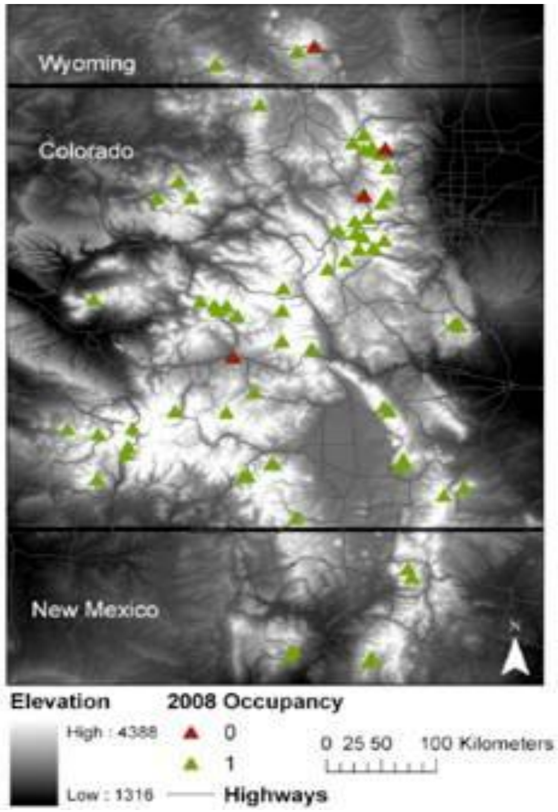


Figure 5-2 Historically occupied American pika sites

Historically occupied American pika sites, resurveyed in 2008. Occupied sites are green; unoccupied sites are red triangles. From L. Peterson Erb, unpublished data.

Results indicate that local population extirpations have been relatively few in the southern Rockies: only four of 69 sites lacked recent sign of their presence (Figure 5-2; Liesl Peterson Erb, University of Colorado—Boulder, personal communication 7/21/2010). In 2009 a three-year investigation began to measure abundance at 25 of these sites, and densities will be compared across sites to identify declining populations among those sampled.

Yellow-bellied marmots (*Marmota flaviventris*), studied at RMBL since 1975, have altered the timing of their emergence from hibernation because of changes in both temperature and snowpack (Figure 5-3; Blumstein 2009; Inouye et al. 2000). Roughly one-half of the population emerges from hibernation when half of the snow cover is gone and food supplies are more available; those that emerge before this may need to re-enter torpor or go without food for weeks. Any lengthier of a mismatch between timing of emergence and food availability could lead to starvation (Inouye et al. 2000). Similarly at RMBL, both chipmunks (*Tamius minimus*) and golden-mantled ground squirrels (*Spermophilus lateralis*) are changing their hibernation phenology in relation to date of first bare ground and date of first *Mertensia* flower; they are following a similar trend as yellow-bellied marmots (David Inouye, University of Maryland, personal communication, July 23, 2010).

In addition to physiological alterations, changes may take place in the spread of disease in small mammals in relation to climate. For example, drought-induced reductions of deer mouse (*Peromyscus maniculatus*) predators in the early 1990s in the southwest United States, followed by a wet winter and ample food supply for the mouse, led to a 10-fold increase in mouse populations and an outbreak of the disease Hantavirus pulmonary syndrome in humans (Epstein 2001).

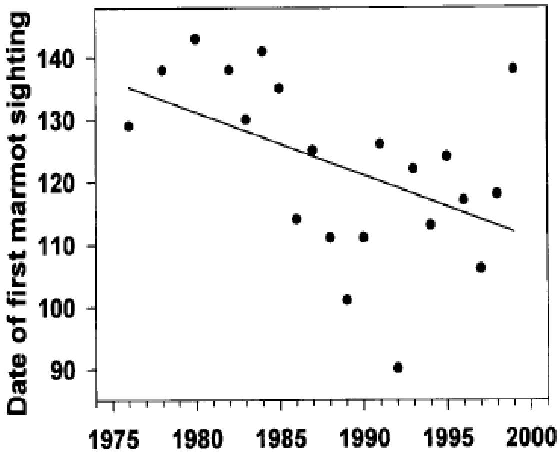


Figure 5-3 Date of first marmot sightings at RMBL

Julian date of the first sighting of a marmot at RMBL each year from 1976 to 1999 (data missing for 3 years), r2=0.23, p=0.03. From Figure 4 of Inouye et al. 2000.

Other concerns include the direct habitat loss due to the upward movement of vegetation communities; this will likely impact chipmunks (*Tamias quadrivittatus* and *T. minimus*), red squirrels (*Tamiasciurus hudsonicus*), woodrats (*Neotoma mexicana*), and other mammals associated with conifer forests in the mountains of the southwest United States (Ditto and Frey 2007). Reduction in habitat quality and quantity through increased fragmentation of these habitats is predicted and could cause loss of genetic variation/gene flow from genetic drift and inbreeding (Ditto and Frey 2007).

Birds

Bird species such as sage grouse (*Centrocercus minimus* and *C. urophasianus*) may be negatively impacted by increased future drought conditions in their native sagebrush habitat. Sage grouse, already surviving in roughly half their historic range, have suffered from decreased nest success and chick survival. These impacts are likely to continue as a result of the reduction in quality

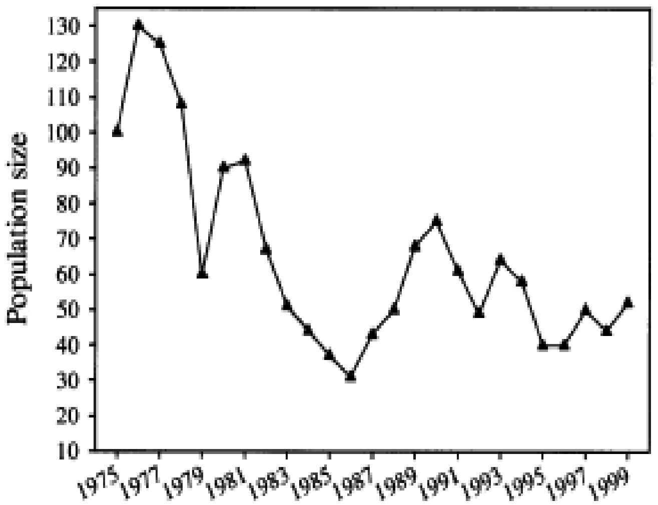


Figure 5-4 White-tailed ptarmigan breeding population at Rocky Mountain NP

Size of white-tailed ptarmigan breeding population at Trail Ridge Road, Rocky Mountain NP, CO, from 1975 to 1999. From Figure 2 in Wang et al. 2002.

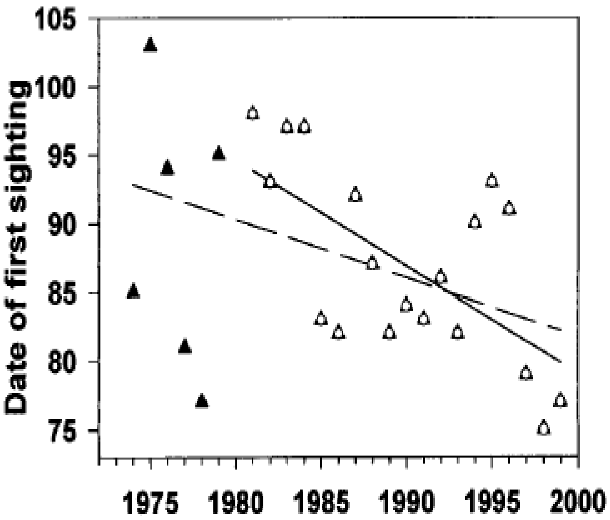


Figure 5-5 Date of first sighting of a robin at RMBL

Julian date of the first sighting of a robin at RMBL each year from 1974 to 1999. The two lines are regressions, including 1974-1980 (▲ and dashed line; P=0.109) and data from 1981-1999 (Δ and solid line; P=0.003). From Figure 3 of Inouye et al. 2000.

habitat, potential for greater livestock and exotic plant impacts, and increased fire severity from drought conditions (Aldridge et al. 2008).

High elevation species such as the white-tailed ptarmigan (*Lagopus leucurus*) have seen slowed growth and reduced population size from recent warmer winters (Figure 5-4; Wang et al. 2002) and these populations may (Wang et al. 2002; Wang, Hobbs, Giesen, and Galbraith 2002) or may not (Sandercock et al. 2005) see reduced reproductive ability from the potential impacts of changes in climate.

Songbird species such as the American robin (*Turdus migratorius*) and red-winged blackbirds (*Agelaius phoeniceus*) have been reported arriving and breeding earlier throughout their ranges (Figure 5-5; Torti and Dunn 2005; Inouye et al. 2000). Other species (e.g., killdeer [*Charadrius vociferous*]), are tending toward this shift in earlier breeding, at least in warm winter years (Torti and Dunn 2005). Over time and with continued warming, this alteration in breeding time may become more permanent (Torti and Dunn 2005). Long-range migratory bird species may be leaving for high elevation summer grounds earlier as a result of increased temperatures in their over-wintering areas (Inouye et al. 2000). Upon arrival at high elevation sites still covered in snow, they must move to lower elevations to find food or face starvation.

Amphibians

There have been many studies showing that reduction in snowpack and/or increases in temperature have led, and will continue with climate warming to lead to, earlier breeding in amphibian species. Amphibian species in the United States (including the boreal toad *Bufo boreas*) studied in Oregon, Michigan, and Canada showed some early breeding in relation to increases in temperature and early snowmelt, though this was not consistent within species or populations (Figure 5-6; Blaustein et al. 2001; Corn 2003). In a study by Corn and Muths (2002), boreal chorus frogs (*Pseudacris triseriata maculata*) in Colorado bred earlier in the spring during dry years between 1986 and 2001 (Figure 5-7); timing was dependent upon maximum snow accumulation, not temperature. Populations of boreal chorus frog and boreal toad, among other species, declined as a result of wetland desiccation in Yellowstone National Park over a 16-year period of warmer annual temperatures and decreased precipitation (McMenamin

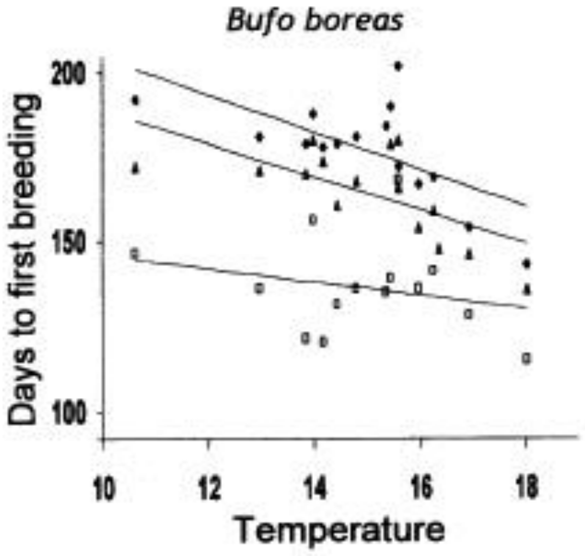


Figure 5-6 Days to first breeding of *Bufo boreas*

Estimated number of days to first breeding from 1 January as related to temperature for Bufo boreas (Todd Lake [◆], Three Creeks Lake [▲], and Lost Lake [□]) in Oregon. From Figure 1 of Blaustein et al. 2001.

et al. 2009), though these results are controversial (Patla et al. 2009).

Similarly, the northern leopard frog (*Rana pipiens*) may alter its range northward and upward in elevation as a result of warming (Lawler et al. 2010). The northern leopard frog requires patches of aquatic habitat connected by drainages for dispersal for functioning meta-populations (Sredl 1997). Drought can impact all amphibian species with the drying of pools prior to completion of metamorphosis (Corn 2005) or by preventing movement between pools (McMenamin et al. 2008). In turn, predators of amphibians—such as herons, coyotes, and birds of prey—will be impacted by declines in amphibians, and species in general will be negatively impacted by loss of wetlands (McMenamin et al. 2009).

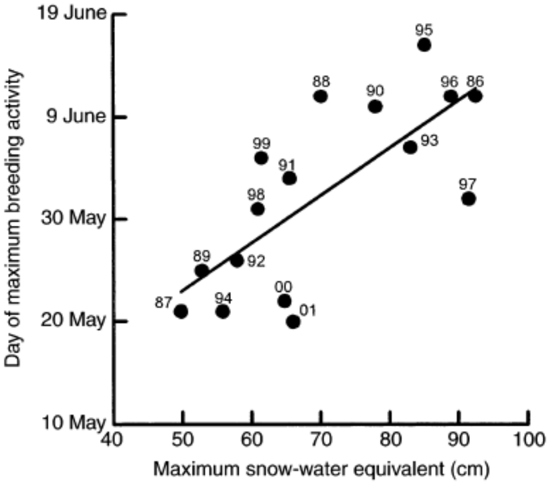


Figure 5-7 Relationship between calling activity by male boreal chorus frogs and maximum snow accumulation

Relationship between calling activity by male boreal chorus frogs and maximum snow accumulation (snow water equivalent, cm) at Lily Pond, CO, USA. Data are labeled by year of observation. From Figure 1 of Corn and Muths 2002.

Studies of boreal toad populations in Rocky Mountain National Park, Colorado over a 10-year period showed a decline in the number of male toads (Figure 5-8) and in the number of egg masses (Muths et al. 2003). Declines may be due to the fungus *Batrachochytrium dendrobatidis* (Bd) found at the site, along with other secondary invaders and may (Alford et al. 2007; Corn et al. 2005; Di Rosa et al. 2006; Muths et al. 2008) or may not (Skerratt et al. 2007) be correlated with increases in temperature. Certainly Bd seems to be restricted from colder environments (Muths et al. 2008), and warming temperatures may shrink these restricted areas. Because the Bd fungus occurs in the environment and may be carried by individuals without symptom (Di Rosa et al. 2006; Skerratt et al.

2007), it is readily available should stress (such as imposed by climate change) raise susceptibility to it or other pathogens (Di Rosa et al. 2006; McMenamin et al. 2009).

Finally, warmer dryer climates may negatively impact tiger salamanders (*Ambystoma tigrinum*) by loss of habitat, decrease in body size, reduction in population numbers, and altering genetic diversity of populations over time (Bruzgul et al. 2005). One positive aspect to early emergence and breeding by amphibians is the reduction of exposure to harmful ultra-violet B (UVB) radiation, which increases later in the season (Corn 2005).

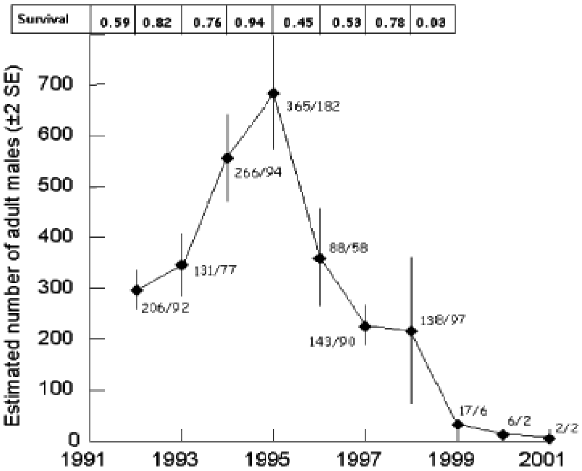


Figure 5-8 Number of male boreal toads in North Fork drainage, Yellowstone NP

Estimated number of male boreal toads in the entire North Fork drainage, Yellowstone NP. Numbers next to means are number captured/number recaptured, and numbers at top between years are the estimated survival rates. Symbols for 1999–2001 represent total numbers caught divided by the mean capture probability (0.51). From Figure 2 of Muths et al. 2003.

Native Fish

Colorado's native fish are adapted to cycles of spring flooding, summer drought, and specific ranges of temperature and water quality. Over the past 150 years, land use has intensified, bringing with it changes in stream and river flows from damming, water diversion, and flood control. Other stresses related to increased human use include predation and competition by stocked non-native fish, pollution from agricultural and urban runoff, destruction of vegetation and stream banks by livestock grazing, siltation, changes in water temperature, and increased susceptibility to disease (Colorado Division of Wildlife 2009; Grahame and Sisk 2002). For example, in a nearby semi-arid system (Gila River, New Mexico), the presence of non-native fish coupled with low flows reduced abundance of individual native species and compromised persistence of native communities; in this case, a natural flow regime alone was unlikely to conserve native fish assemblages (Propst et al. 2008). The Colorado River cutthroat trout (*Oncorhynchus clarkii pleuriticus)*, Rio Grande cutthroat trout (*O. clarkii virginalis)*, Rio Grande sucker (*Catostomus plebeius*), Rio Grande chub (*Gila pandora*), razorback sucker (*Xyrauchen texanus*), bonytail chub (*Gila elegans*), roundtail chub (*Gila robusta robusta*), and Colorado pikeminnow (*Ptychocheilus lucius*) are all native to the Colorado and Rio Grande river basins, but are currently designated either endangered, threatened, or a species of special concern in Colorado (Colorado Division of Wildlife 2009).

Restoration and protection efforts are underway for these species, but future scenarios of a warmer, dryer climate will likely exacerbate these already existing stresses. Changes in climate can affect fish communities directly through temperature effects on fish physiology, and indirectly through effects on flow regimes, water quality, and susceptibility to disease. Even when the increase in temperature is not sufficient to prove fatal, the sublethal impacts on growth and reproduction may cause significant changes in the structure and composition of fish communities (Ficke et al. 2007).

In the Rocky Mountain region, increases of 1°C, 2°C, 3°C, 4°C, or 5°C (1.8°F, 3.6°F, 5.4°F, 7.2°F, or 9.0°F) in mean July air temperature are projected to reduce the geographic area containing suitable salmonid habitat by 16.8%, 35.6%, 49.8%, 62.0%, or 71.8%, respectively (Keleher and Rahel 1996). In addition to warmer water, changes in streamflow and the increasing frequency and intensity of other disturbances, such as wildfire and flooding, are among the factors associated with climate change that are likely to have impacts on native trout populations in the western United States. As much as 65% of the habitat currently occupied by westslope cutthroat trout (*Oncorhynchus clarkia lewisi*), and 29% of that occupied by Colorado River cutthroat trout, will be at high risk from one or more of the three main factors: rising summer temperatures, increased flooding, and increased wildfire (Figure 5-9; Williams et al. 2009). Within the next 50 years, wildfire, floods, and other disturbances may have a greater impact on population persistence than increasing water temperature alone. Risks likely will vary substantially within subspecies.

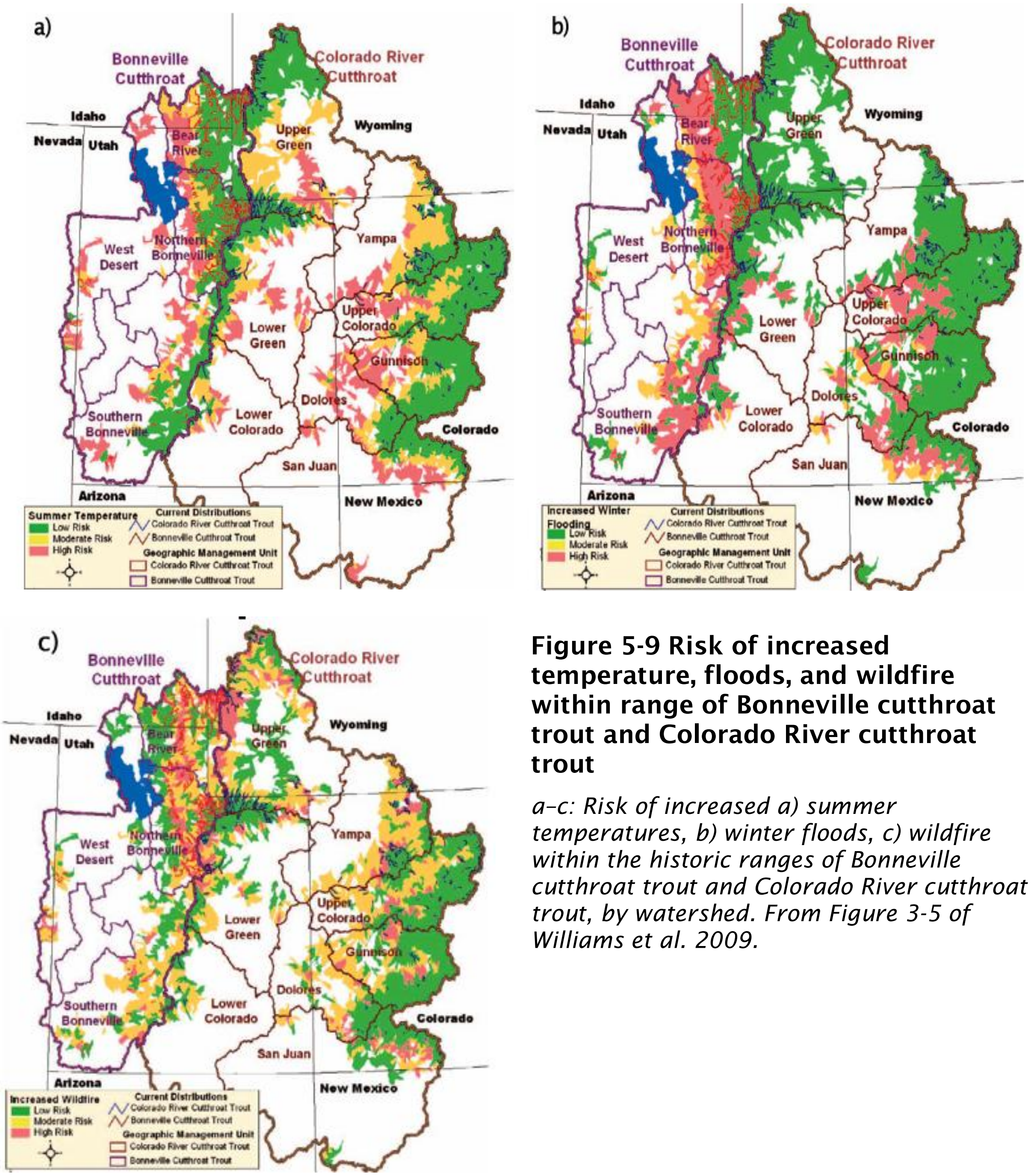


Figure 5-9 Risk of increased temperature, floods, and wildfire within range of Bonneville cutthroat trout and Colorado River cutthroat trout

a–c: Risk of increased a) summer temperatures, b) winter floods, c) wildfire within the historic ranges of Bonneville cutthroat trout and Colorado River cutthroat trout, by watershed. From Figure 3-5 of Williams et al. 2009.

Stress from climate change is likely to compound existing problems associated with habitat degradation and competition from introduced salmonids (Williams et al. 2009). As warming proceeds, salmonid populations would be forced into increasingly higher elevations and would become fragmented as suitable habitat for coldwater fish becomes separated from main river channels and restricted to headwater streams (Keleher and Rahel 1996). In addition, the impact of whirling disease (*Myxobolus cerebralis*) on juvenile rainbow trout (*Oncorhynchus mykiss*; a non-native species) and cutthroat trout in the Rocky Mountains may become more severe as summer water temperatures and periods of low flow increase (Hiner and Moffitt 2001; Schisler et al. 1999). Whirling disease is particularly detrimental to juveniles and can severely limit recruitment in both the wild and in hatcheries. Experimental infection of juvenile trout (rainbow, brook, brown, and four cutthroat subspecies) from the Colorado River resulted in mortality rates of about 89% within a month (Thompson et al. 1999).

Warming water temperatures may also increase habitat suitability for some species in certain circumstances. In a study of greenback cutthroat trout (*Oncorhynchus clarkii stomias*) in Rocky Mountain National Park, modeled warming of stream temperatures increased the likelihood of translocation success (i.e., introduction or re-introduction). This result occurred because greenbacks have been restricted through competition with non-natives to sub-optimal habitats, which are generally too cold to be highly productive (Cooney et al. 2005).

Trout may also be affected by changes in their food sources. For example, the mayfly (*Baetis bicaudatus*) is extremely important in stream environments where they eat algae and are a food source for predatory fish. This species has been shown to emerge earlier with decreased snowpack, resulting in smaller body size, reduction in fitness, and negative impacts on mating, placement of eggs on exposed rocks, and egg development (Harper and Peckarsky 2006).

Climate change impacts are expected to show considerable spatial variation with differences among streams, however, and an in-depth study of southwestern Colorado has not been completed.

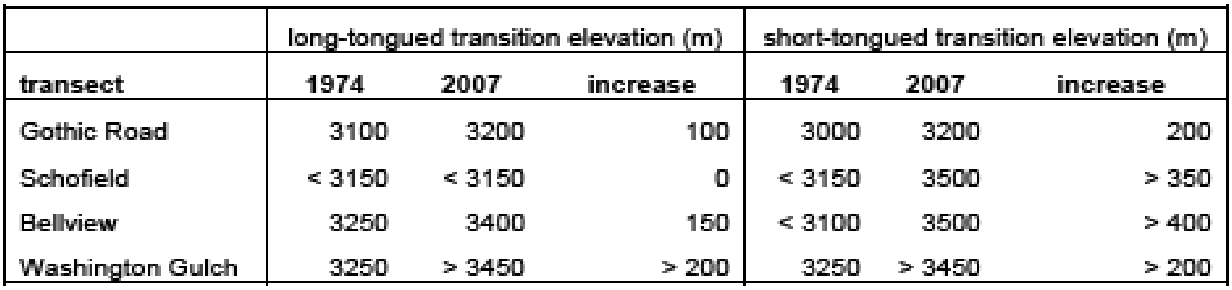
Insects and Other Pollinators

The relationship between vegetation community (habitat) or plant species (food source) to animal species is important to consider in future climate scenarios (also see Ecosystems and Phenology sections of this report). Frost damage to plants or buds from early spring snowmelt or impacts from drought on plants can negatively affect flower and seed production. A diversity of plant species, including a wide range of flowering phenologies, benefit pollinators and birds that eat seeds from plants, or larvae hosted by plants (Carey 2009; Hegland et al. 2009). Although there is preference for certain flower species by pollinators such as bees and butterflies, it is important that alternative early and late flowering hosts be available in the event of early or late emergence of pollinators (Carey 1994). For example, the silvery blue butterfly (*Glaucopsyche lygdamus*) requires diverse hosts to ensure food should drought or late frost impact plants (Carey 1994).

A diversity of flowering plants is also important for pollinators to find high quality nectar sources (Elliott 2009). Should early-season pollinators have limited food supply and not survive, later-season plants will be affected as well (Hegland et al. 2009). This mismatch is more detrimental to pollinators than to plants, which may have vegetative reproduction or seedbanks to rely on, at least in the short term (see Phenology section). Climate warming could lead to continued earlier emergence in some insect species, while flowering time may not change at the same rate, resulting in either the movement of insects to lower elevations or their local extirpation (Miller 2007). Bees may be responding to recent warming, as seen in a re-measure of a historic bee transect in subalpine meadows of Colorado where lower elevation species increased in numbers and higher elevation species decreased over time (Table

5-3; Miller 2007). Another species that greatly impacts seed set (and thus both community composition and pattern) are aphids (Galen 1990). Adler et al. (2007) found aphid numbers had no affect or were reduced following artificial manipulation of temperature and snowmelt, although Roy et al. (2004) speculate this could be attributable to the drying effects of the heaters.

Table 5-3 Summary of elevation changes in the transition of dominance between low and high elevation bees.



	long-tongued transition elevation (m)			short-tongued transition elevation (m)		
transect	**1974**	**2007**	**increase**	**1974**	**2007**	**increase**
Gothic Road	3100	3200	100	3000	3200	200
Schofield	< 3150	< 3150	0	< 3150	3500	> 350
Bellview	3250	3400	150	< 3100	3500	> 400
Washington Gulch	3250	> 3450	> 200	3250	> 3450	> 200

Greater than or less than symbols mean no transition occurred within the transect. From Table 2 of Miller 2007.

Conclusion

Changes in Southwest Colorado's climate are having profound effects on organisms and ecosystems throughout the San Juan Mountain region. Although uncertainties remain, there is published evidence of observed responses of plant and animal communities and natural processes to variability and changes in the regional climate within the last century. Additionally, many models project continued changes and responses during the next decade and beyond. This report provides a synthesis of current information. Because of the great range of topics covered and rapid progress of the state of climate science, this document should be continually updated as climate models are refined and new information is published. However, this document can be used as a guide to what range of effects and changes may affect ecosystems. In this way, we can frame a new way of planning for different evolving scenarios, rather than static environments or a specific state of an ecosystem.

In this review, we found that sagebrush, piñon, and aspen communities are sensitive to drought and likely will be affected by climate change. Additionally, American Pika are particularly sensitive to changes in climate and may prove useful indicators for monitoring changes in the San Juan Mountain alpine. For many other ecological processes, communities or species, non-climate stressors such as pollution, habitat loss or fragmentation, and increased human pressure are anticipated to drive changes over the next century. For instance, ponderosa pine forests are already considered to be outside of their historic range of variability and changes in fire regimes may affect these ecosystems most strongly. Still, other systems and resources may respond to climate change by increasing in population size or range. In other cases, we lack published evidence that link recent trends in a resource to link it to climate change; more developed hypotheses may be generated in the future.

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