ECOREGIONAL ASSESSMENT REPORT

CLIMATE CHANGE SUPPLEMENT

NORTHERN GREAT BASIN

RAPID ECOREGIONAL ASSESSMENT

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Acronyms and Abbreviations

%	percent
°F	degree Fahrenheit
°C	degree Celsius
BCSD	Bias Correction and Spatial Downscaling
BLM	Bureau of Land Management
CA	Change Agents
CE	Conservation Element
CEG	Conservation Element Group
CIG	Climate Impact Group
CMIP	Coupled Model Inter-Comparison Project
DGVM	Dynamic Global Vegetation Model
FGDC	Federal Geographic Data Committee
GCM	Global Climate Models
IPCC	Intergovernmental Panel on Climate Change
km	kilometer
m	meter
MACA	Multivariate Adaptive Constructed Analogs
MC	MAPSS - CENTURY dynamic global vegetation model
NGB	Northern Great Basin
PRISM	Parameter-Elevation Regressions on Independent Slopes Model
RCP	Representative Concentration Pathway
REA	Rapid Ecoregional Assessment
RegCM3	Regional Climate Model 3
sq. km	square kilometers
SWE	Snow Water Equivalent
VDDT	Vegetation Dynamics Development Tool
VIC	Variable Infiltration Capacity
W/m2	Watts per meter squared

1 Introduction

1.1 Overview of the Rapid Ecoregional Assessment Process

The Bureau of Land Management (BLM) is currently evaluating a wide variety of environmental challenges to western ecosystems while maintaining their mission of resource management, and authorizing multiple uses of the public lands under the jurisdiction of each field office. These challenges transcend land ownership and administrative jurisdictions, and necessitate a landscape-scale approach to evaluate of potential changes and threats to these ecosystems. A comprehensive Rapid Ecoregional Assessment (REA) was completed for the Northern Great Basin Ecoregion in 2013 (SAIC 2013). An REA is the BLM's first step toward a broader initiative to systematically develop and incorporate landscape-scale information into the evaluation and eventual application to management of public land resources and is one of a suite of tools available to resource managers and field personnel for assessing natural resource values. This Climate Change Supplement to the Northern Great Basin REA summarizes the existing knowledge of the potential effects of climate change to the Northern Great Basin ecoregion. This report provides technical summaries detailing observed and projected changes for the ecoregion's climate, change agents (CA), and conservation elements (CE), and applies that latest generation of global climate modeling from the Integrated Scenarios dataset which includes regionally downscaling results, dynamic vegetation modeling, and land surface hydrology modeling to evaluate potential effects to the CEs.

In general terms, the purpose of the REA process is to document important regional resource values and patterns of environmental change that may not be evident when managing smaller, local land areas separately. The REA process maintains a focus at the scale of the ecoregion to understand more fully the ecological conditions and trends; encompass the extent of natural and human influences; and identify opportunities for resource conservation, restoration, and sustainable development. REAs define the core ecological elements of the ecoregion, CEs, define the relevant parameters, and describe and map areas of high ecological value. REAs then gauge the potential of these values to be affected by environmental CAs. This REA synthesizes existing information, rather than conduct research or collect new data. As part of this synthesis, a better understanding of critical data gaps also emerges.

This supplemental effort specifically applies recently available climate data for the ecoregion. The reader is directed to the Northern Great Basin REA (SAIC 2013) for a comprehensive review of ecological characteristics of the ecoregion, the REA process, and the potential future effects from non-climate related CAs.

1.2 Overview of the Northern Great Basin Ecoregion

The study area for this REA is comprised of two ecoregions, the Northern Basin and Range and the Snake River Plains, identified going forward as the Northern Great Basin (Figure 1-1). The Northern Great Basin encompasses southeastern Oregon, portions of southern Idaho, northern Nevada, and a small extension into northeastern California and northwestern Utah. It is the northern extent of the larger Basin and Range physiographic province and also includes the important upper Snake River drainage system. Most of the ecoregion is dominated by sagebrush steppe ecosystems on the desert floor, but distinct vegetation zones related to relief and elevation also exist including juniper, mountain mahogany, aspen and riparian habitats. In the upper elevations Douglas-fir and aspen stands occur up the sub-alpine zone, which supports primarily low-growing shrubs, grasses, and forbs. Wildlife species of importance to the region include bighorn sheep (*Ovis canadensis*), mule deer (*Odocoileus hemionus*), elk (*Cervus elaphus*), pronghorn (*Antilocapra*

americana), pygmy rabbit (*Brachylagus idahoensis*), golden eagle (*Aquila chrysaetos*), and greater sage-grouse (*Centrocercus urophasianus*). Important habitats in the ecoregion include migration corridors and areas for overwintering pronghorn, as well as key habitat for greater sage-grouse. Federal agencies manage the majority of land in this ecoregion but large areas of tribal and private agricultural lands are present as well. Historical and current land use includes mining, livestock grazing, agriculture, and recreation. Current management priorities include energy development, wild horse and burro management, and invasive plant species (particularly cheatgrass).

Human populations in the ecoregion are concentrated along the Snake River corridor and land development remains an important CA. Much of the Snake River Plains ecoregion is used as cropland and federally managed rangeland, in which the distribution and extent of native vegetation communities have been significantly altered. Land use issues focus on the impacts of farming and livestock grazing, residential and commercial development, invasive annual grasses, dispersed recreation, surface water and groundwater withdrawal for irrigation, and soil erosion.
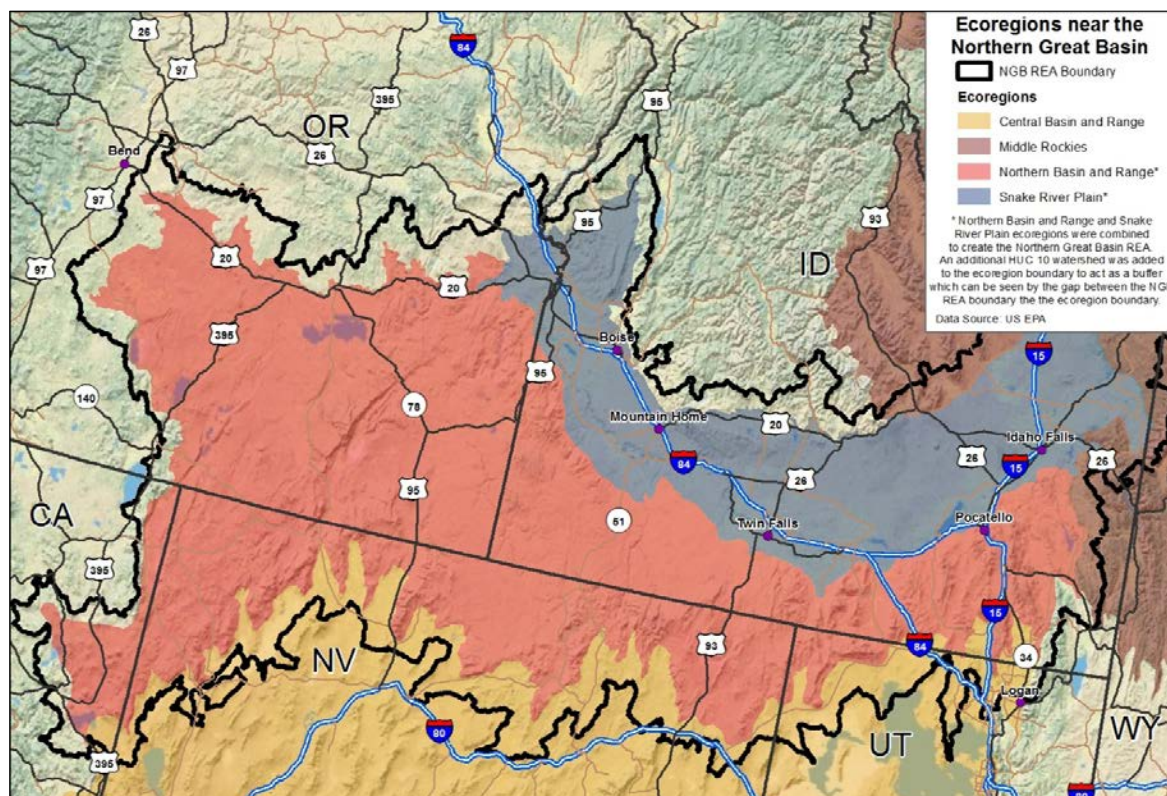
Figure 1-1. Northern Great Basin Ecoregion with Other Adjoining REA Boundaries

1.3 Organization of the Document

The following sections in Chapter 1 briefly summarize possible (i.e. modeled) future climate scenarios as well as the possible effects of changing climate scenarios on selected CEs and CAs within the ecoregion. Chapter 2 comprehensively presents the approach to climate change modeling, and data and assumptions employed, and Chapter 3 offers the results of modeled projections for temperature, precipitation, and other climate variables at mid-century and end of the century time slices. Chapter 4 provides climate change assessments for selected CAs, CEs, and conservation element groups (CEGs) that were analyzed in the 2013 REA (SAIC 2013). Chapter 5 contemplates lessons learned, included data gaps and limitations.

1.4 Brief Review of Modeled Climate Change and Associated Effects

Climate Change

Atmospheric carbon dioxide concentration has varied over earth's history. Based on measurements in Antarctic ice cores, carbon dioxide concentrations have fluctuated between 170 and 300 parts per million (ppm) over the past 800,000 years. As a result of human activities since the Industrial Revolution, carbon dioxide levels have increased and currently exceed 400 ppm. Based on historical temperature measurement records, there has been an increase in average global temperature of at least 1°C from 1960 to present, compared to the 1901-1960 average. Multiple additional indicators of a changing global climate have been identified, including increases in sea temperatures, higher sea level, and decreased Arctic sea ice. If anthropogenic sources of greenhouse gases continue to increase, as expected, global average temperatures will likely continue to rise.

Climate scientists from around the world collaborate regularly to produce Global Climate Model (GCM) simulations. The products from GCM simulations are at a coarse spatial scale, typically 150 to 300 kilometers (km). Downscaling GCM results bridges the gap between the coarser resolution provided by the GCM modelers and the finer resolution data needed by decision-makers and impact assessors. This report presents GCM data downscaled using the Multivariate Adaptive Constructed Analogs (MACA) method. The modeling results are presented using an ensemble mean of the 10 highest-ranked models for the Pacific Northwest.

This supplement utilized the data outputs from the Integrated Scenarios of the Future Northwest Environment project which is based on the latest generation of global climate models (CMIP5) to portray as accurately as possible the Northwest's future climate. The Integrated Scenarios data is composed of outputs from downscaled climate data, the hydrologic models, and the vegetation models (Figure 1-2).
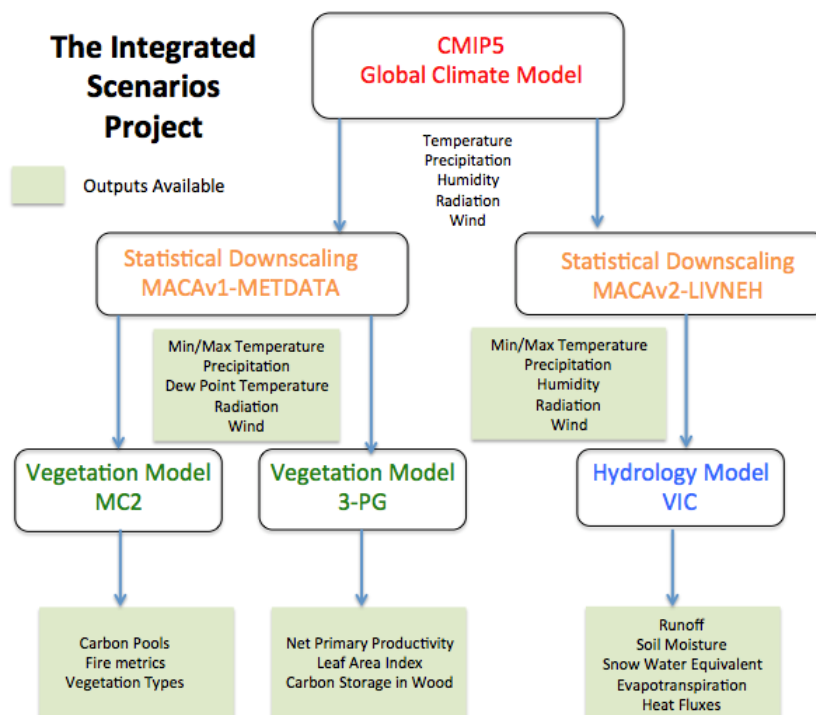
Figure 1-2. Integrated Scenarios Modeling Outputs

Source: Integrated Scenarios Project - http://climate.nkn.uidaho.edu/IntegratedScenarios/model_products.php

Based on these models, temperatures are projected to increase in the future throughout the ecoregion. The current annual average maximum daily temperature is 14.6 °C (58 °F). In a low emissions scenario (RCP 4.5) annual average maximum daily temperatures are projected to increase by 2.8 °C (5.0 °F) at mid-century (2040-2069) and 3.5 °C (6.3 °F) at the end of the century (2070-2099). In the high emissions scenario (RCP 8.5) temperatures could increase by 3.6 °C (6.5 °F) at mid-century and 5.8 °C (10.4 °F) at the end of the century. The largest increases in temperature will occur in the summer months.

Precipitation is also projected to increase in the future throughout the ecoregion. In the low emissions scenario (RCP 4.5), the annual mean precipitation is projected to increase by 7 percent at mid-century and 9 percent at the end of the century. In the high emissions scenario (RCP 8.5) annual mean precipitation is projected to increase by 9 percent at mid-century and 18 percent at the end of the century. In general, there is more uncertainty in the precipitation projections than the temperature projections in the global climate models. The modeling predicts that the ecoregion as a whole will experience very minor decreases in future wind speeds for all scenarios and seasons. In the ecoregion the downward shortwave solar radiation is projected to decrease slightly, indicating slightly higher cloud cover.

1.5 Summary of the Possible Effects of Climate Change on Change Agents and Conservation Elements

This section summarizes the results of this Climate Change Supplement analysis. Citations have not been included in this summary. Full citations are included for each respective change agent or conservation element in Chapter 4.

Invasive Annual Grasses

Across the Northern Great Basin ecoregion invasive annual grasses cover millions of acres and are a threat to various communities from salt desert shrub and sagebrush steppe to ponderosa pine forests. Of main concern is cheatgrass (*Bromus tectorum*), medusahead (*Taeniatherum caput-medusae*) and ventenata (*Ventenata dubia*). For the ecoregion, both the spring and the summer are expected to have slight increases in precipitation with a 6 to 9 percent increase predicted in spring months and a 3 to 9 percent increase predicted in summer months. An increase in spring precipitation is advantageous to cheatgrass, as it provides additional water when cheatgrass is in its growing season. An increase in summer precipitation would generally support sagebrush and native bunchgrasses.

The risk of wildfire is also likely to increase. The dynamic vegetation modeling (MC2) predicts the fraction of area burned in the ecoregion is expected to increase by approximately 200 percent in the future scenarios compared to the historical conditions (1970-2000). Previous modeling by Bradley (2009) found that most the Northern Great Basin ecoregion is suitable for cheatgrass. Overall, the evidence from the MACA-downscaled global climate modeling and MC2 fire modeling indicates that invasive plants may be favored by many of the elements associated with climate change. However, it is a challenge to predict the specific effects of climate change on current and potential invasive plants like cheatgrass due to the uncertainty of climate projections, the heterogeneous landscapes of the Intermountain West, and complex interactions of multiple elements which can produce inconsistent. Given the uncertainty, management should embrace multiple pathways and rely on adaptive management with respect to invasive annual grasses and climate change in the ecoregion.

Shrubs (Sagebrush & Salt Desert Shrub)

The shrubs CE group included sagebrush and salt desert scrub. Sagebrush ecosystems are widespread across the western United States (U.S.) and serve as important habitat for a variety of species. Sagebrush habitats are considered some of the most endangered ecosystems and their widespread degradation and

vulnerability has led to broad-based ecosystem conservation efforts. Salt desert shrubs ecosystems tend to occur in the basins, on sites where soils may be salt-affected and heat and aridity are locally the greatest.

The broad ecological amplitude of the various species of sagebrush present in the ecoregion and the uncertainties involved in predicting climate change make predictions a complex undertaking. Generally, all the range-wide modeling efforts predict a decline of sagebrush in the southern periphery of its range with expansion of sagebrush northward and at higher elevations. Species distribution modeling studies suggest that portions of sagebrush habitat in the ecoregion that appear to be in the stable portion of suitable sagebrush climate niche may present important restoration and conservation opportunities for the preservation of sagebrush and sagebrush-obligate species. However, dynamic vegetation modeling coupled with state and transition models show that the prevalence of cheatgrass, the expansion of junipers, and the apparent recent trend toward larger and more frequent wildfires are interrelated factors that need to be taken into account when considering the effects of climate change on sagebrush communities.

The MC2 modeling predicts a significant loss and redistribution of the temperate shrublands within the ecoregion. The temperate shrublands are modeled to be largely replaced by temperate evergreen needleleaf woodlands. The MC2 vegetation model simulates this extensive woody expansion throughout the U.S., which is caused in part from increase CO_2 concentrations resulting in increased water efficiency of woody plants. Given the complex interactions between CO_2 concentrations, climate and wildfire, the results from the MC2 model for the ecoregion should be carefully interpreted. Alternatively, the effects of increased CO_2 concentrations is generally not included in species distribution models, and based on the MC2 results, increased CO_2 levels may have a significant effect on increasing woody vegetation types which has not been accounted for in previous sagebrush modeling. In addition, as the ecoregion becomes warmer and slightly moister, the soil temperature/moisture regime of sagebrush communities will shift. These shifts were analyzed with respect to greater sage-grouse habitat (discussed below).

Conifer Trees (Juniper & Other Conifer)

Conifers are an integral component of forest communities at higher elevations in the Northern Great Basin. Common dominant species are Douglas-fir (*Pseudotsuga menziesii*), lodgepole pine (*Pinus contorta*), ponderosa pine (*Pinus ponderosa*) and Engelmann spruce (*Picea engelmannii*). Western juniper (*Juniperus occidentalis*) and Utah juniper (*J. osteosperma*) were also selected because of aesthetic and wildlife habitat values. Juniper are also of interest due to their potential to expand into and alter sagebrush steppe communities.

The dynamic vegetation modeling results predict a substantial expansion of temperate evergreen woodlands (e.g. juniper). The woodland is modeled to expand and replace much of the shrubland in the ecoregion. Similarly, the results for the forest vegetation types (other conifer) predicts a significant forest expansion due to climate change. This expansion is based on the increased CO_2 concentrations increasing the net primary productivity in the ecoregion resulting in more carbon in the vegetation and more success of woody species. The extensive woodland expansion is not consistent with the species distribution modeling presented in the previous REA for western and Utah juniper. Overall, the species distribution model may underestimate the range for tree species because it does not include increased productivity and water efficiency from elevated CO_2 concentrations. However, the MC2 dynamic vegetation modeling likely overestimates the increase in range of tree species by not including a nitrogen-based limitation to the increase in productivity expected from the elevated CO_2 concentrations.

Aspen

Aspen was selected as a CE due to concerns over recent declines of aspen stands and the need for increased aspen management and restoration efforts. Two vegetation types, mixed conifer aspen and

Rocky Mountain aspen, were combined to make up one class for aspen. Based on the USFS Moscow Forestry Sciences Laboratory climate model results for aspen, aspen is projected to significantly decline in the ecoregion in 2030 and 2060. Based on the VIC hydrologic modeling, the summer soil moisture (July 1st) generally increases with the projected increase in precipitation in the low-lying areas. However, reduced snowpack in the higher elevation areas results in a reduction in soil moisture from melting snow when compared to historical conditions. The predicted reductions of summer soil moisture on July 1st appear to overlap with the reductions in aspen. Overall, climate change would be expected to reduce the aspen viability, mainly through increased soil moisture stress in the summer in mountainous areas in the ecoregion.

Hydrology (Groundwater, Springs & Seeps, Streams & Rivers, Open Water, Riparian & Wetlands)

Across the landscape surface water, groundwater, and geological characteristics control the presence of perhaps the most important element in shaping the West – water. Together with climate, water availability is a key variable in the development and sustenance of ecological communities.

Based on modeling predictions, the ecoregion will shift towards more rain dominant behavior as the region's temperatures warm. Currently, based on historical data, 85 percent of the basins would be classified as rainfall dominant. The rainfall dominant basins increase to 98 percent of the ecoregion by end of the 21st century. Modeling generally predicts a slight increase in overall annual flow (10 to 20 percent) by the end of the century for most sites. The increase in flow is projected to occur in the winter months. Peak flows generally shift one month earlier due to the reduction in snowpack. Spring and summer flows are projected to be decrease with less available snowpack. The modeling does not include changes in the streamflow response due to a reduced groundwater recharge in slow draining systems. Vegetation and agriculture rely heavily on soil moisture. Summer soil moisture is expected to increase slightly in low lying areas due to increased spring and summer precipitation. However, soil moisture is expected to decline slightly in the mountains due to reduced snowpack. While summer soil moisture would slightly increase, the periodic droughts that occur in the ecoregion would likely be more severe with high evapotranspiration rates due to increased temperatures.

Vulnerable Soils

Generally, the areas most vulnerable to water erosion are silt-textured and on steeper slopes. Based on the modeling projections, annual precipitation is expected to increase by 7 to 9 percent at mid-century (2040-2069) and 11 to 18 percent at the end of the century (2070-2099). This annual increase in precipitation would be expected to increase the rainfall erosivity in the region by 8 to 11 percent from 2040-2069 and 13 to 21 percent from 2070-2099.

Soils vulnerable to wind erosion are also fine textured but generally occur in flatter areas. The greatest threat to soils vulnerable to wind erosion is increasing wildfire frequency and severity in the ecoregion. The exposure of soils following recent wildfires have resulted in wind erosion events in the ecoregion that are as great (or greater) in magnitude than many previously studied environments in Africa, Australia, and the U.S. Overall, the average summer wind speed (June, July, and August) is projected to slightly decrease 0.09 to 0.12 m/s at the mid-century (2040-2069) and decrease 0.11 to 0.16 m/s at the end of the century (2070-2099). These changes range from 1 to 3 percent of the historical average wind speeds. Therefore, predicted changes in wind speed related to climate change are not likely to have a significant effect on wind erosion rates in the ecoregion. However, the fraction of area burned is expected to increase with climate change in the ecoregion by 200 percent. This will likely result in more catastrophic fires which are often followed by large wind erosion.

Mule Deer

Mule deer are one of the most widely distributed and economically and socially important animals in western North America. Over the past century, mule deer populations have fluctuated throughout their range; however, trends in the latter part of the 20th century indicated that mule deer populations were declining.

Mule deer in the Northern Great Basin ecoregion inhabit areas primarily classified as sagebrush and other shrub-steppe habitats. The potential changes in vegetation described above will likely impact mule deer habitat in the ecoregion. In addition, winter snowpack and summer precipitation are also important to mule deer survival. Overall, the expected reduction in winter snowpack, especially in March and the slight projected increase in summer precipitation due to climate change should have a favorable impact on the mule deer populations in the ecoregion. However, increasingly severe droughts associated with warming temperatures will likely have a periodic negative impact on mule deer populations.

Pronghorn

Climate change is expected to result in a reduction of snowpack in March which should slightly increase available pronghorn habitat in the higher elevation mountainous regions. Higher annual precipitation (especially in spring) is expected to increase forage resources and could benefit the pronghorn populations. However, the MC2 modeling predicts a substantial increase in total vegetation carbon. The increased growth of woodland species, like juniper, would likely reduce the available pronghorn habitat as pronghorn strongly avoid forested habitats due to the cover provided to its predators. The total vegetation carbon increase is due in part to the estimated carbon dioxide fertilization effect assumed by the model. The MC2 model may overestimate the woody expansion due to the carbon dioxide fertilization effect by not accounting for limits in other nutrients (especially nitrogen).

Greater Sage-grouse

The greater sage-grouse is considered an umbrella species for sagebrush-associated vertebrates. As described above, the prevalence of cheatgrass, the expansion of junipers, and the recent trend toward larger and more frequent wildfires are interrelated factors that are effects of climate change on sagebrush communities which could impact greater sage-grouse. In addition, soil temperature and moisture regimes of sagebrush communities are a strong indicator of resilience to disturbance and resistance to invasive annual plants. Highly resilient and resistant (cool/moist) sagebrush communities are important sage-grouse habitat in the ecoregion.

In general, climate change would be expected to result in a shift from cool/dry and cool/moist regimes to warm/dry and warm/moist regimes which are less resilient and resistant to annual grass invasion. Cool/moist sagebrush currently makes up approximately 43 percent of priority sagebrush habitat in the ecoregion. At mid-century (2040-2069), cool/moist sagebrush would be reduced to 2 to 6 percent of the priority habitat with most areas transitioning to a warm/moist regime. In RCP 8.5, late period 2070-2100, there would be almost no cool/moist sagebrush and some sagebrush areas would shift to hot/dry and hot/moist regimes. Consistent with the invasive grass analysis in Section 4.1, modeled climate change would result in a shift in temperature/moisture regimes for sagebrush communities which would likely make the sagebrush communities and corresponding sage-grouse habitat less resilient to disturbance and less resistant to invasion from invasive annual grasses. Based on the MC2 modeling, increasing CO_2 levels and climate change could result in an increase in temperature evergreen woodlands in the ecoregion in the place of sagebrush. This may result in increased juniper encroachment into sagebrush communities which can negatively affect sage-grouse populations. Overall, climate change would be expected to

exacerbate the dual stresses of juniper encroachment and invasive annual grass expansion in existing sage-grouse habitat.

Pygmy Rabbit

The pygmy rabbit (*Brachylagus idahoensis*) is the smallest rabbit species in North America and occupies sagebrush-steppe communities within the Great Basin and adjacent Intermountain West. Sagebrush communities are threatened by invasive annual grasses and juniper expansion. Overall, climate change would be expected to exacerbate the dual stresses of juniper encroachment and invasive annual grass expansion in existing pygmy rabbit habitat. The temperature/moisture regimes will likely shift from cool/moist sagebrush communities to warm/moist communities, which will make them less resilient and resistant and could alter the species composition of the communities. Based on the MC2 modeling, climate change could result in an increase in temperature in evergreen woodlands in the ecoregion in the place of sagebrush. Warming temperatures will also reduce the snowpack within pygmy rabbit habitat. With less snow cover, predatory protection in the winter may decrease. The amount of pygmy rabbit habitat burned by wildfire is also likely to increase with the warming temperatures. With more wildfires expected, less snowpack, and continued expansion of juniper into sagebrush communities, pygmy rabbit habitat will likely continue to contract due to the effects from climate change.

Coldwater Fish

The coldwater fish assemblage for the Northern Great Basin ecoregion includes bull trout, redband trout, mountain whitefish, Lahontan cutthroat trout, and Yellowstone cutthroat trout. These species were selected to represent the assemblage due to their sensitivity to changes in hydrology and habitat quality, in addition to hybridization, competition, and predation pressures associated with introduced species. The NorthWest Stream Temperature project (NorWeST) developed a comprehensive interagency stream temperature database and high-resolution climate scenarios for the Northwest U.S. This database of stream temperatures along with climate modeling predictions (CMIP3) has been used to evaluate climate refugia in the northwest U.S. for bull trout and cutthroat trout (Isaak *et al.* 2015). Habitat for coldwater fish is expected to decline by more than 50 percent in the ecoregion due to rising temperatures. In the ecoregion, the Climate Shield modeling identifies 632 km of bull trout habitat with a probability of occupation of 10 percent or more. This is modeled to be reduced to 210 km by 2080, mainly due to forecasted temperature increases. The Climate Shield modeling also identifies 2,610 km of cutthroat habitat with a probability of occupation of 10 percent or more under present conditions. This is modeled to be reduced to 1,206 km by 2080 due to climate change.

In addition to the increased August temperatures, the August streamflow will likely decrease. Based on the Integrated Scenarios VIC modeling results of selected streamflow sites in the ecoregion, the August streamflow would decrease by approximately 20% (ranging from 2 to 41 percent) in the RCP 4.5 scenario at mid-century and by up to 40 percent (ranging from 10 to 59%) in RCP 8.5 scenario by end the century. This reduction in streamflow would decrease available habitat and connectivity for coldwater fish, adding additional stress to cold water fish species.

Columbia Spotted Frog

The Columbia spotted frog (*Rana luteiventris*) was selected as a CE due to losses of historically known occupied sites, reduced numbers of individuals within local populations, and declines in the reproduction of those individuals. Within the ecoregion, this species' distribution associates with low population areas such as the Owyhee Mountain region and Boise National Forest. A recently completed study on climate suitability (historic, current, and future) (Pilliod *et al.* 2015) shows that the suitable climate conditions in the Great Basin have been reduced 50 percent over the last century. Future climate suitability is expected to

continue to decline significantly (77–97%) due to seasonal shifts in temperature, precipitation, and stream flow patterns associated with climate change. Reduced snowpack and warming at higher elevations has the potential to increase population viability for montane frog populations. However, Pilliod *et al.* (2015) did not predict an increase in future climate suitability at higher elevations. Less snowpack could negatively impact wetland hydrology in the summer months and the Great Basin mountain ranges have steep topography which may limit the formation of wetland habitat. If climate suitability projections are realized, the existing, isolated Columbia spotted frog populations in the ecoregion could disappear. Human intervention and cooperation of private, state, and federal landowners may be needed to facilitate the long-term persistence of the Columbia spotted frog in the ecoregion in the future.

2 Climate Change Modeling Background

The surface of the earth and lower atmosphere is livable due to the greenhouse effect. Radiation from the sun warms the earth and the planet loses energy through emission of infrared radiation. Greenhouse gases (primarily water vapor, carbon dioxide, methane, ozone, and nitrous oxide) trap a portion of the infrared radiation and radiate it back to the surface, thereby warming the surface (the greenhouse effect). Without the greenhouse gases, the surface of the earth would be 60 degrees Fahrenheit (°F) (33 degrees Celsius [°C]) cooler (Walsh *et al.* 2014). At that temperature, all water on earth would freeze, and life as we know it would not exist.

One of the primary greenhouse gases is carbon dioxide. Atmospheric carbon dioxide concentration has varied over earth's history. Based on measurements in Antarctic ice cores, carbon dioxide concentrations have fluctuated between 170 and 300 parts per million (ppm) over the past 800,000 years. As a result of human activities since the Industrial Revolution, carbon dioxide levels have increased and currently exceed 400 ppm (Walsh *et al.* 2014). Based on historical temperature measurement records, there has been an increase in average global temperature of at least 1°C from 1960 to present, compared to the 1901-1960 average (Walsh *et al.* 2014). Multiple additional indicators of a changing global climate, have been identified including increases in sea temperatures, higher sea level, and decreased Arctic sea ice. If anthropogenic sources of greenhouse gases continue to increase, as expected, global average temperatures will likely continue to rise (Intergovernmental Panel on Climate Change [IPCC] 2013).

Climate scientists from around the world collaborate regularly to produce GCM simulations. Since the completion of the 2013 Ecoregional Assessment Report (SAIC 2013) a new generation of GCMs from the Coupled Model Inter-Comparison Project 5 (CMIP5) has become available, as widely used in the most recent IPCC Assessment Report (Fifth) (IPCC 2014). The CMIP5 utilizes a different approach to estimating atmospheric concentrations of greenhouse gases and their effects on climate. The previous GCM results used in the 2013 Northern Great Basin rapid ecoregional assessment (i.e., Regional Climate Model 3) are based on different rates of forecasted emissions. The next generation CMIP5 GCM results establish a range of changes in radiative forcing which could be caused by several processes including the change in the emission or change in land use. Radiative forcing is the result of the balance of incoming and outgoing energy in the Earth-atmosphere system, measured in Watts per square meter (W/m^2). Two scenarios modeled by CMIP5 models include: (1) Representative Concentration Pathway (RCP) 8.5 which represents an increase of the radiative forcing to 8.5 W/m^2 in 2100 and (2) RCP 4.5 which is a stabilization scenario where radiative forcing is stabilized at 4.5 W/m^2 in 2100. Previous greenhouse gas scenarios for CMIP3 have close analogues to the CMIP5 scenarios (Table 2-1). The project total CO_2 emissions in each CMIP3 and CMIP5 scenarios are shown in Figure 2-1. In this report, the results from both the low scenario (RCP 4.5) and high scenario (RCP 8.5) are presented.

Table 2-1. Comparison of CMIP5 and CMIP3 Scenarios (Snover *et al.* 2013)

New CMIP5 Scenarios	Characteristics	Comparison to Old CMIP3 Scenarios
RCP 2.6	An extremely low scenario that reflects aggressive greenhouse gas reduction and sequestration efforts.	No analogue in previous scenarios.
RCP 4.5	A low scenario in which greenhouse gas emissions stabilize by mid-century and fall sharply thereafter.	Very close to B1 by 2100, but higher emissions at mid-century.
RCP 6.0	A medium scenario in which greenhouse gas emissions increase until stabilizing in the final decades of the 21st century.	Similar to A1B by 2100, but closer to B1 at mid-century.
RCP 8.5	A high scenario that assumes continued increases in greenhouse gas emissions until the end of the 21st century	Nearly identical to A1F1.
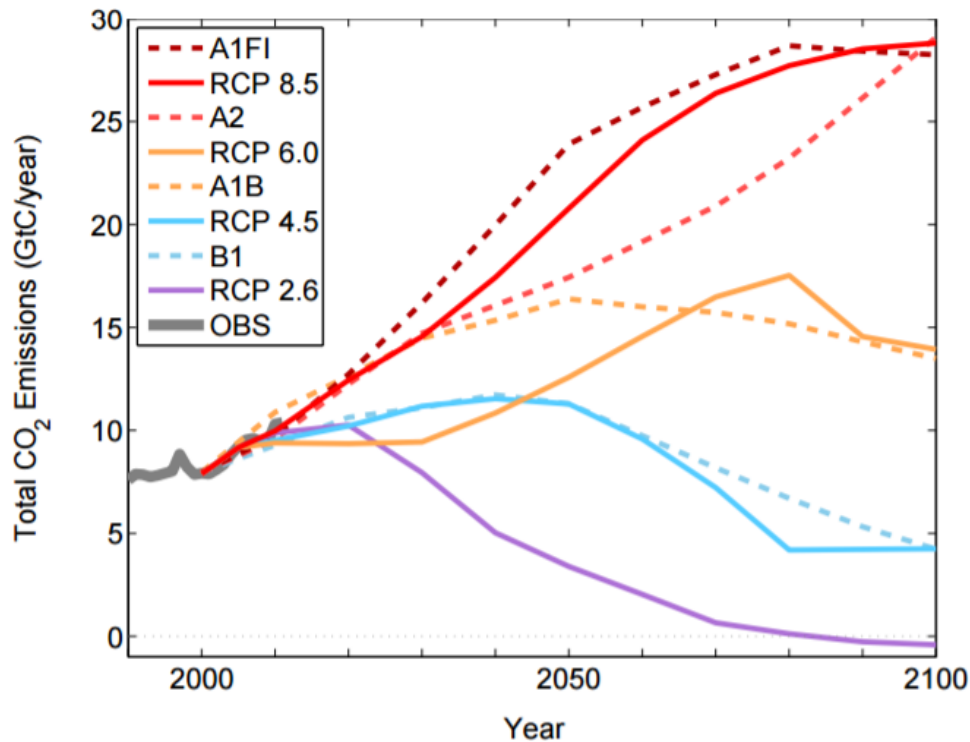
Figure 2-1. Future Greenhouse Gas Scenarios (Snover *et al.* 2013)

2.1 Downscaling Global Climate Model Simulations

The products from GCM simulations are at a coarse spatial scale, typically 150 to 300 kilometers (km). Downscaling GCM results bridges the gap between the coarser resolution provided by the GCM modelers and the finer resolution data needed by decision-makers and impact assessors (Rana and Moradkhani 2015). The most common downscaling methods are dynamic and statistical downscaling. Dynamic downscaling typically uses regional climate models nested within the GCMs, which ensures consistency between climatological variables. However, dynamic downscaling requires much more computational time than statistical downscaling. Statistical downscaling applies the output from GCMs to the region by using a series of equations to relate variations in global climate to variations in local climate based on the observed historical climate record (Figure 2-2). Bias correction and spatial downscaling (BCSD) and Multivariate Adaptive Constructed Analogs (MACA) are two statistical downscaling methods that have been used for impact studies in the study region (Rana and Moradkhani 2015). While both methods outperformed results obtained from direct interpolation, MACA displayed advantages in predicting in temperature, humidity, wind, and precipitation due to its ability to jointly downscale temperature and dew point temperature, and its use of analog patterns rather than interpolation (Abatzoglou and Brown 2012).

The previous climate change analysis in the Northern Great Basin rapid ecoregional assessment presented results on a 15-km grid which is the native coarse resolution of the third generation of the Regional Climate Model 3 (RegCM3). The MACA downscaling method allows the representation of the modeling results on a 4-km grid that can match the native resolution of the Parameter-elevation Regressions on Independent Slopes Model (PRISM) dataset. PRISM data is the official climatological data for the U.S. Department of Agriculture and is recognized as one of the highest quality spatial climate data sets currently available. The University of Idaho MACA dataset includes the downscaled GCM results for 20

models for RCP 4.5 and RCP 8.5. Of the 40 models used in CMIP5, the 20 that the University of Idaho selected have daily outputs for precipitation, temperature, wind, and humidity.

Variables that are downscaled using the MACA method include 2-meter(m) maximum/minimum temperature, 2-m maximum/minimum relative humidity, 10-m zonal and meridional wind, downward shortwave radiation at the surface, 2-m specific humidity, and precipitation accumulation all at the daily timestep represented on a 4-km grid. The MACA data is divided into three separate datasets, MACAv1 METDATA, MACAv2 LIVNEH, and MACAv2 METDATA. The main differences between these datasets are spatial resolution of the datasets, spatial domain (Western U.S. vs Contiguous U.S.), and different downscaling methods and training datasets (http://maca.northwestknowledge.net/). This supplement report utilizes the MACAv2 METDATA. Climate forcings in the MACAv2-METDATA were drawn from a statistical downscaling of GCM data from the CMIP5 (Taylor *et al.* 2010) utilizing a modification of the MACA (Abatzoglou and Brown 2012) method with the METDATA (Abatzoglou 2011) observational dataset as training data. The MACAv2 METDATA has been post processed more than the MACAv1 datasets and includes seasonal monthly bins, time slices for two future scenarios with differences from the historical dataset already calculated and a multi model mean from 20 global climate models.
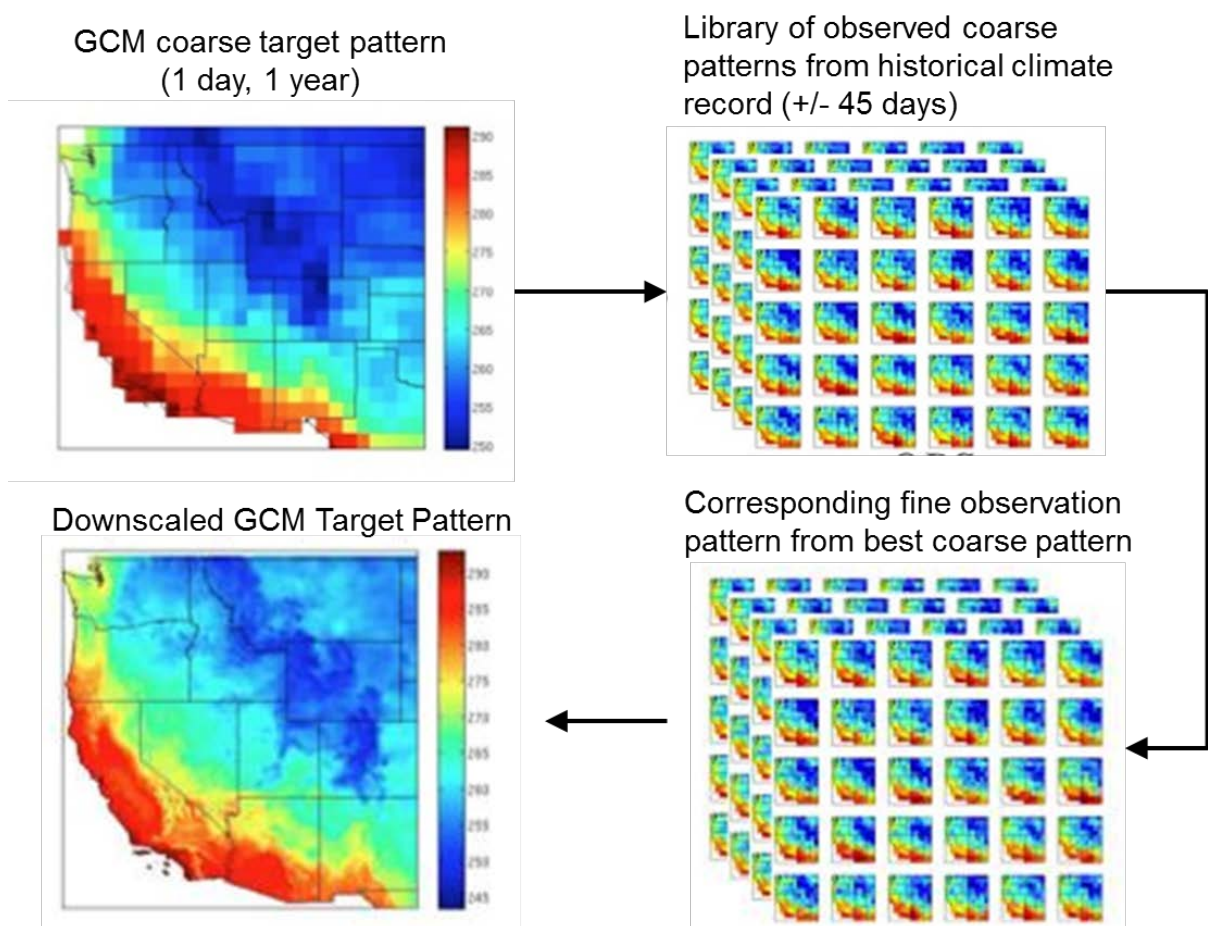
Figure 2-2. MACA Downscaling Process (adapted from University of Idaho 2014)

2.2 Ensemble Mean Selection

This study utilizes the results of several GCMs to define future meteorological conditions in the project region. This multi-model ensemble approach has been shown to improve predictions compared to an approach that only uses a single model. The development of mean values from a set of model predictions helps to offset extreme and extraneous predictions of individual models (Mote *et al.* 2011).

To develop the multi-model ensemble, ten GCMs were selected from the University of Idaho MACA model set based on how accurately they predict historical climate conditions in the project region. Inclusion of at least ten GCMs to develop a multi-model ensemble is consistent with the recommendations of the Mote *et al.* 2011 study, which recommends using as many GCM projections as possible for this purpose.

Selection of the most accurate GCMs was based on the results of a study conducted by Rupp *et al.* 2013. This comprehensive study evaluated the ability of 41 CMIP5 GCMs to simulate historical climate within the U.S. Pacific Northwest, which encompasses most of the REA project region. This study focused on the evaluation of GCM monthly temperature and precipitation predictions in both temporal and spatial scales. Figure 2-3 shows the results of the GCM evaluation and rankings conducted by Rupp *et al.* (2013).
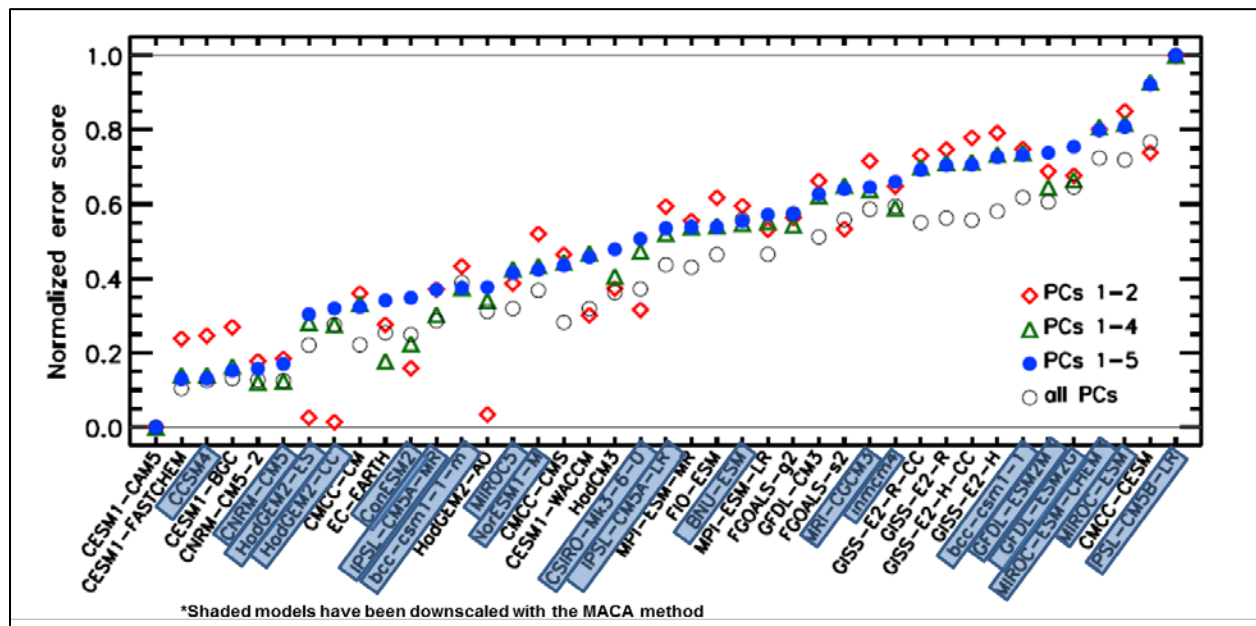
Figure 2-3. CMIP5 models ranked by 18 performance metrics (Rupp *et al.* 2013 [Figure 3])

The GCMs in the MACA model set that coincide with the highest 10 rankings identified by Rupp *et al.* 2013 were then identified for use in this study and include the following (with Rupp *et al.* 2013 total model ranking):

1 - CCSM4 (3)	6 - IPSL-CM5A-MR (12)
2 - CNRM-CM5 (6)	7 - bcc-csm1-1-m (13)
3 - HadGEM2-ES 365 (7)	8 - MIROC5 (15)
4 - HadGEM2-CC 365 (8)	9 - NorESM1-M (16)
5 - CanESM2 (11)	10 - CSIRO-Mk3-6-0 (20)

The down-scaled climate modeling outputs for these 10 GCMs from the MACA dataset were acquired to develop a multi-model ensemble mean for key variables (temperature, precipitation, humidity, and wind) that can impact potential CEs and CAs. The University of Idaho also provides a multi-model mean of all 20 MACA downscaled models. This 20-model mean was acquired as well for key variables for quality assurance to assess the difference between an ensemble mean of the best performing 10 models and a 20-model mean of all available models.

2.2.1 Global Climate Model and Downscaling Uncertainty

There is very high confidence that GCMs reproduce the general features of the global-scale annual mean surface temperature increase over the historical period. However, although the simulation of large-scale patterns of precipitation has improved, models continue to perform less well for precipitation than for surface temperature. At regional scales, precipitation is not simulated as well, and the assessment remains difficult owing to observational uncertainties (Flato *et al.* 2013).

Some GCM errors can be traced to uncertainty in the representation of processes. For the atmosphere, cloud processes, including convection and its interaction with boundary layer and larger-scale circulation, remain major sources of uncertainty. The uncertainty in cloud processes in turn create uncertainty in radiation which propagate through the coupled climate system. Other sources of uncertainty is the distribution of aerosols, the vertical and horizontal mixing and convection in the ocean, simulation of sea ice, and parameterizations of nitrogen limitation and forest fires impacts on biogeochemical cycles (Flato *et al.* 2013). Uncertainty can be examined by looking at outputs of different GCMs with a common model and experiment. The multi-model ensemble can be used to estimate the range and standard deviation for key variables to estimate the uncertainty of the projections. However, it is possible that future changes in climate could be outside of the range projected by climate models (Mauger *et al.* 2015; Flato *et al.* 2013). To address uncertainty in this analysis we have included a multi-model ensemble of GCMs and present the multi-model mean and the standard deviation from the mean.

The MACA method assumes that the climate distribution does not change much over time and only the mean changes, that is, it is stationary in the variance and skew of the distribution. The MACA method is also sensitive to the geographic extent, with the influence of dependence differing for various variables, and ability of GCMs to accurately simulate climate in the region. The accuracy of the downscaling results also typically degrades near the tails of the distribution, resulting in extremes to generally be underestimated (Rana and Moradkhani 2015).

The MACA data utilized is based on climate analogs developed from the METDATA archive. The METDATA is based on daily data from the North American Land Data Assimilation System using monthly temperature, precipitation, and humidity from PRISM (Daly *et al.* 2008). The PRISM dataset was compared to observed station data in Nevada (Jeton *et al.* 2006). The analysis concluded that the precipitation-elevation relations developed to distribute precipitation may not be appropriate for all mountainous areas. Some areas in northern Nevada had large differences between the PRISM results and recorded data. Due to the coarseness of the PRISM grid cells, the sparseness of the long-term precipitation data, and the broad range of differences between the PRISM estimates and the recorded data, the optimum use of the PRISM data set is for large-scale studies where long-term averages are required (Jeton *et al.* 2006).

2.3 Modeling Climate Influences on Vegetation Distribution

Climate change is expected to alter the distribution of plant species, the composition of plant communities, the structure of vegetation, and the processing of nutrients, water, and carbon within the ecosystem. The influences of climate on vegetation are complex and the long-term effects of climate on

vegetation makes it difficult to predict potential vegetation responses based solely on theory or field experiments. Computer simulation models often are used to integrate theory and experimental results to project vegetation responses under future climate conditions (Kerns and Peterson 2014). These computer vegetation models are generally statistically based, process based, or a combination of the two. These models include species distribution modeling, the state and transition model, and the dynamic global vegetation model (DGVM). They vary in their approach and are often distinguished by the following criteria (Daniel and Frid 2011):

A. Degree to which ecosystem processes, such as succession and disturbance, are simulated mechanistically;
B. If the models are deterministic (predict a single future) or stochastic (i.e. predict a distribution of possible futures);
C. Scale at which ecosystem processes are represented (e.g. gap (m2), stand (ha), region (km 2);
D. Extent to which the spatial dynamics of ecosystem processes are represented explicitly (e.g. disturbance spread over time); and,
E. Range of ecosystems that the models can be applied.

Statistical species distribution modeling, also known as climate envelope-modeling, habitat modeling, and niche-modeling, estimates the similarity of the conditions at any site to the conditions at the locations of known occurrence with the aim of predicting species ranges with climate data as predictors. Species distribution models are popular due to their ability to generate complex predictions without high computational requirements. However, most statistical approaches use very basic relationships between species or vegetation distributions to infer plausible future distributions. The statistical models do not factor in disturbance (e.g. wildfire or insect outbreaks), complex ecological interactions (e.g. dispersal of seed by wildlife), or local conditions (e.g. slope, aspect, soils, etc.), or interspecific interactions that also determines species' actual distributions (Littell *et al.* 2013; Miller *et al.* 2015). These uncertainties limit the use of the results for inferring the actual change that may occur. The REA completed for the ecoregion presented the results of the USFS Moscow Forestry Sciences Laboratory species distribution modeling for various tree species (Douglas fir, Engelmann spruce, lodgepole pine, and ponderosa pine). The models produce a viability that ranges from 0.0 to 1.0, where species with a viability below 0.5 have little chance of a persisting in future climate scenarios.

The state and transition model is a tool for identifying potential state changes in which a parcel of land can find itself in at any point in time through a series of transitions, both natural and anthropogenic, which land between these states (Westoby *et al.* 1989; Daniel and Frid 2011). State changes are the loss or addition of functional groups, changes is soils or hydrographs outside the historical range, the presence of highly invasive species, and long-term or large-scale human-caused disturbances on the landscape (e.g., energy development). The state and transition model has been further refined by assigning probabilities to each of the transitions pathways, which provides a landscape approach and provides the ability to look at communities as well as ecosystems (Daniel and Frid 2011). Conceptual state and transition models are also developed using historical information, local and professional knowledge, inventory monitoring, and experimental data. Background information and existing data previously gathered can be linked with local and professional knowledge from workgroups to develop initial diagrams and narratives for each component of a "state and transition" for an ecological site (Caudle *et al.* 2013). State and transition models have been useful in exploring "what if" scenarios for management options and are used to evaluate the sensitivity of the system to specific parameterizations. State and transition models can incorporate the effects of multiple disturbances, biotic interactions, and several management scenarios (Miller *et al.* 2015). However, state and transition models lack statistically robust techniques for relating climate data to species distributions.

Process based models simulate the underlying physiological, ecosystem, and disturbance processes that drive changes in vegetation. Examples of process models include gap models, biogeochemical models, and dynamic global vegetation models. Gap models are used to examine vegetation changes and interactions between on a daily time step over a fine spatial scale. Biogeochemical models are process-based models that simulate carbon, water, and mineral cycles in terrestrial ecosystems. For climate change they are primarily used to estimate carbon storage and fluxes (Kerns and Peterson 2014). Dynamic global vegetation models project changes in vegetation properties at broad spatial scales (thousands of square kilometers) over annual to decadal time steps.

A type of dynamic global vegetation model is the MAPSS-CENTURY version two (MC2). M stands for MAPSS (a biogeography model) and C stands for CENTURY (a biogeochemistry model) that are linked together to generate the dynamic global vegetation model. MC2 uses climate projections (current or future scenarios) to drive vegetation dynamics and simulates changes in distribution of plant functional types, carbon, and wildfire at broad scales (Bachelet 2013; DGVM 2015; Peterman *et al.* 2014). This climate change supplement utilized the MC2 model results developed during the Integrated Scenarios of Climate, Hydrology and Vegetation project, which was a collaboration between the Northwest Climate Science Center, the University of Idaho, Conservation Biology Institute, and the University of Washington. The goal was to model future changes in climate, hydrology, and vegetation over the western U.S. from the coast to the Great Plains. The vegetation modeling efforts using the downscaled CMIP5 projections in the Pacific Northwest were produced through the use of the process-based MC2 vegetation model with a large number of the most recent global climate projections. The MC2 model is made up of 3 modules, 1) a biogeography module (MAPS model) that simulates change in plant functional types and life forms over time, 2) a biogeochemistry module (CENTURY model) that accounts for pools and fluxes of carbon and nutrients; and 3) a fire module which simulates fire effects in the landscape (Figure 2-4). A detailed description of the modeling methodology is provided by Sheehan *et al.* (2015).
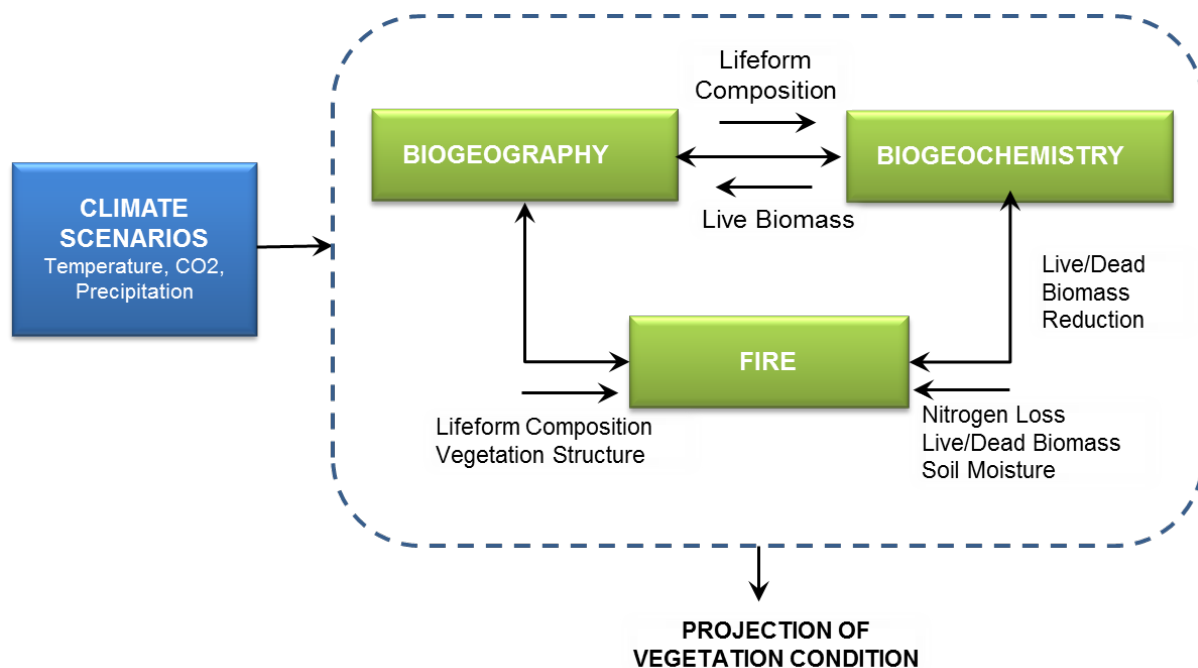
Figure 2-4. Graphic representation of MC2 DGVM (Source: Peterman *et al.* 2014)

2.3.1 MC2 Vegetation Model Limitations

The MC model is advantageous because it uses climate scenarios to predict vegetation structure and produces a spatial output with a visual picture; however, there are limitations, such that there are multiple factors and complex interactions being simulated. MC2 simulates potential vegetation most adapted to the climate drivers. However, in reality vegetation is long lived and endures under suboptimal conditions remaining in a metastable state until disturbance or natural mortality removes this legacy vegetation. Modeling fire also presents challenges as fire occurrence, spread, and intensity depend on inherently unpredictable factors including seasonal weather extremes, immediate weather conditions, ignition occurrence, and other factors that may affect fuel load and conditions. The effects of CO_2 concentration on water use efficiency and plant productivity are still not completely understood. While the CO_2 fertilization effect in MC2 is moderate, it does result in greater woody plant production and fuel accumulation. The addition of insect attacks, disease, and invasive species to the model would allow the model to better reflect the effects of these influences on vegetation dynamics (Sheehan *et al.* 2015). The model also does not incorporate human impacts (e.g. that are thought to be more important than climate). The model incorporates different scales such that broad patterns of climate drive vegetation but microclimate, geomorphology, soils, animals, and humans all operate at small scales and the model design does not include fine scale processes (Bachelet 2013).

There are advantages and disadvantages to every model developed and to overcome some of the model limitations scientists have incorporated multiple models. For example, Miller *et al.* (2015) combined species distribution models with state and transition models to examine processes of vegetation growth and disturbance relevant to whitebark pine (*Pinus albicaulis*). State and transition models and species distribution models were combined to account for the anticipated impacts of climate change, biotic interactions, and disturbances, while also allowing for the exploration of management options (Miller *et al.* 2015). MC2 is the second version of the MC1 dynamic global vegetation model that was created to assess potential impacts of global climate change on ecosystem structure and function.

2.4 Modeling Climate Influences on Hydrology

Water resources are sensitive to changes in climate. Climate change poses challenges to resource managers seeking the most effective strategies to adapt, maintain, and restore rivers, watersheds, and aquatic ecosystems in the ecoregion. Managers benefit from accurate analyses of historical streamflows and predictions of future hydrologic behavior (Safeeq *et al.* 2014). Both empirical and numerical models have been routinely used for predicting future streamflows and improving the understanding of hydrological functioning at varying spatial and temporal scales. In large watershed- and regional-scale studies, land surface models, such as the large-scale Variable Infiltration Capacity (VIC) model, have been widely employed in the Pacific Northwest. Studies include regional-scale changes in snowpack (Hamlet *et al.* 2005), water resources (Hamlet *et*

Variable Infiltration Capacity - Three Layer (VIC-3L) Macroscale Hydrologic Model
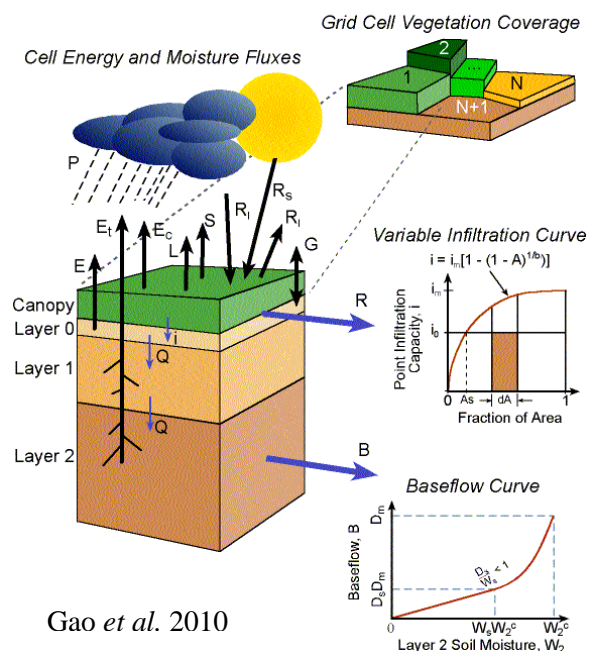
Gao *et al.* 2010

al. 2007), droughts (Shukla and Wood 2008), and energy (Hamlet *et al.* 2010).

The VIC model was designed to integrate with the general circulation model. The outputs from the circulation models (temperature, precipitation, and wind speeds) are inputs to the VIC model. The VIC model balances the water and surface energy budgets within each grid cell accounting for evaporation (E), transpiration (E_t), soil moisture (S), and runoff (R). Outputs of VIC include snow water equivalent (SWE), soil moisture, runoff, streamflow, evapotranspiration, and potential evapotranspiration, which are useful in describing the habitat of many species ranging from fish to trees (Mote 2014).

There are two principal datasets that are often used to evaluate hydrologic projections in the greater Pacific Northwest:

i. **Integrated Scenarios for the Future Northwest Environment**. The current set of projections, developed by Mote *et al.* in 2015, which stem from the newer 2013 IPCC report.
ii. **The Pacific Northwest Hydroclimate Scenarios Project**. A previous set of projections, developed by Hamlet *et al.* in 2010, which are based on the climate projections used in the IPCC's 2007 report.

This climate change supplement currently uses the Integrated Scenarios dataset. The newer Integrated Scenarios hydrologic projections appear to contain biases in the mountain regions and assume that winter temperatures are unrealistically cold at high elevations, which impacts the simulations of snow water equivalent and its effects on streamflow (Mauger *et al.* 2015). The Integrated Scenarios data is being refined through calibration that may eliminate the issues associated with the temperature bias. VIC Model Limitations, an analysis of VIC model performance, which compared the observed and simulated streamflow, found both strength and weakness in the VIC model that is important to understand for the successful application of the model (Safeeq *et al.* 2014). The VIC model performs reasonably well in capturing year-to-year variability in observed streamflow which provides confidence in using the model to estimate hydrologic trends for climate change assessments (Hamlet *et al.* 2007; Silva-Hidalgo *et al.* 2009). However, performance was poorer for predicting the magnitude and year-to-year variability in observed low flows and total summer streamflow. This poorer performance could be problematic, given the importance of summer flows for aquatic organisms and municipal water supply.

2.5 Data Evaluation

The Statement of Work requires that relevant climate data be evaluated for data quality as part of this task to ensure that the data used in the modeling process is appropriate to derive a suitable outcome in the analysis stage. A comparison of the Integrated Scenarios Project data sets is provided in Table 2-2. The goal of the evaluation process is to determine the overall quality of the climate datasets as measured by the eleven criteria identified in Appendix 5 within the Bureau of Land Management (BLM) Data Quality Management Guide. The Federal Geographic Data Committee (FGDC) compliant data must contain metadata as part of the data source information. Metadata was either acquired as part of the climate dataset, or as additional files paired with the data. The information contained within the metadata file is often relevant to the data quality itself. Therefore, each climate dataset will be examined to determine the quality of the associated metadata. Each data quality criteria will be given a score from 0 to 4 (0 = unknown, 1 = low, 2 = moderate, 3 = high, 4 = very high) for a total possible score of 44. The summing of the eleven data quality criteria allows for a quantitative comparison of all the criteria.

Table 2-2. Comparison of the Integrated Scenarios Climate Modeling Data

Parameter	Statistical Downscaling MACA v2	Vegetation Model MC2	Hydrology Model VIC
CMIP	5	5	5
Time slice Historic	1971 – 2005	1971 – 2000	1971 – 2000
Time slice Future	2040 – 2069 2070 – 2099	2036 – 2065 2071 – 2100	2040 – 2069 2070 – 2099
Statistical Downscaling	MACAv2-METDATA	MACAv1-METDATA	MACAv2-LIVNEH
GCMs	20 (+ 20 multi model mean)	12	10
Emissions	RCP 4.5 & 8.5	RCP 4.5 & 8.5	RCP 4.5 & 8.5
Variables	Temp (Max/Min) Precipitation Solar Radiation Wind	Carbon Pools Fire Metrics Vegetation Types	Runoff Soil Moisture Snow Water Equivalent Evapotranspiration Heat Fluxes
Spatial Domain	Contiguous U.S.	Western U.S.	Western U.S.
Spatial Resolution	4km	4km	6km
Seasonal Bins	Yes	No	NA

2.5.1 Challenges Encountered Using Data Quality Evaluation with Climate Data

No Attributes

The MACA, Conservation Biology Institute MC2 Vegetation and Climate Impact Group (CIG) hydro-climate datasets are all stored as floating point rasters. There are no attributes to review for validity, non-duplication, completeness and relationship validity data quality evaluators.

No Metadata

Most of the climate datasets that are being reviewed are stored as netcdf or ASCII rasters and aren't accompanied by FDGC metadata. Netcdf files contain some 'attribute' information that could be considered metadata but it is limited to what the user stores when creating the netcdf. Some netcdf attributes contain very detailed information while others contain brief descriptions of the data and contact information. The MC2 vegetation climate data was the only dataset with traditional metadata but it was fairly limited to mostly an abstract, use constraints and contact info.

Spatial Accuracy

Since most climate data is fairly coarse (large pixel/cell size), the spatial accuracy data quality evaluator that would normally be used doesn't really apply the same way it would for smaller cell size raster data (aerial imagery) or vector data (roads, boundaries, etc.).

2.5.2 Data Quality Evaluation Results

Overall, the climate datasets scored fairly high in overall quality due to the fact that they are relatively recent and released as a whole rather than in small pieces (timeliness and consistency). The raster output layers are mostly based on netcdf files exported directly from the climate modelers so there is little chance of anyone altering the data before use in the REA (derivation integrity and thematic accuracy). The CIG hydroclimatic dataset currently available is based on the older CMIP3 so it was scored slightly lower since its data is older and based on the previous CMIP (timeliness and thematic accuracy).

2.6 Climate Change Modeling References

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3 Patterns of Future Climate Change in the Ecoregion

3.1 Existing Climate Patterns

The Northern Great Basin ecoregion has low to moderate topographic relief interspersed with complex terrain, which affects temperature and precipitation at a wide variety of spatial and temporal scales (see Williams Jr. 1972; Mock 1996; Daly 2006). One of the main influences on the climate of the ecoregion is the prevailing westerly flow and associated Polar storm systems, which transports moisture from the Pacific Ocean into the ecoregion from fall through spring. This westerly flow regime provides the majority of annual precipitation to the region and moderates temperatures compared to colder and drier Arctic air masses that occasionally enter the ecoregion from the north. From roughly July through September, weak westerly flow is often replaced by southerly flow that circulates around the subtropical high pressure system that builds into the Rocky Mountain region and produces the North American monsoon. This regime results in drier conditions across the ecoregion, but it can still produce substantial rain showers, especially in higher terrain. This activity affects the eastern portion of the ecoregion more than the west.

During general westerly flow, the Cascade Mountains, Coastal Ranges, and northern extension of Sierra Nevada mountain range produce a rain shadow effect in the ecoregion. Mountain ranges within the ecoregion that also contribute to localized climate effects include the Blue Mountains, Hart Mountain, Poker Jim Ridge, and Steen's Mountain in the west, the northward extensions of the parallel ranges of the Great Basin to the south, the Middle Rockies to the east, and the Boise and Sawtooth ranges to the north. The projection of the Owyhee Mountains northward along the western border of Idaho almost bisects the Northern Great Basin ecoregion. The Snake River Plain and the Owyhee Uplands are large areas of low to moderate topographic relief while relatively narrow basins are present within most of the ranges along the southern boundary of the ecoregion.

The climate of the Northern Great Basin is also affected by a number of external drivers. A correlation analysis of winter precipitation from 1926 to 2007 with the Southern Oscillation Index (SOI) indicates that the area of the ecoregion lies in a transition zone between the Southwest, which is negatively correlated with the SOI, and the Northwest, which is positively correlated with the SOI. A positive SOI value is associated with La Niña conditions and a more northerly Polar storm track whereas a negative SOI value is associated with El Niño conditions with a more southerly storm track. One analysis showed that the Owyhee Uplands experiences similar effects to those that occur in the Northwest while the portion of the ecoregion within Utah experiences similar effects to those that occur in the Southwest (Wise 2010).

3.2 Temperature Projections

The pattern of average annual temperatures for the ecoregion indicates the effects both of altitude and latitude. Highest annual values are found at lower elevations, including the Snake River Valley and open valleys of the Boise, Payette, and Weiser Rivers. The lowest annual values occur at higher elevations in the mountains (WRCC 2014). The average daily maximum temperature from 1971 to 2000 is estimated to be 14.6 °C for the ecoregion, with localized values ranging from 3 to 19.5 °C, as shown in Figure 3-1.

Temperatures are projected to increase in the future throughout the ecoregion. Figure 3-2 displays predictions of average maximum daily temperatures across the ecoregion based on the multi-model mean results. These data show that annual mean maximum daily temperatures for the ecoregion are predicted to increase by the following:

- For RCP 4.5, 2.8 ± 0.5 °C at mid-century (2040-2069) and 3.5 ± 0.6 °C at the end of the century (2070-2099).
- For RCP 8.5, 3.6 ± 0.5 °C at mid-century (2040-2069) and 5.8± 0.8 °C at the end of the century (2070-2099).

The modeling results determined that the range in average maximum daily temperature increases across the ecoregion do not exceed 0.9 °C for any of the four scenarios presented in Figure 3-2. These multi-model mean data also show a distinct trend of increasing temperatures from west to east across the ecoregion. This pattern could in part be a function of proximity to the Pacific Ocean and its moderating effect on temperatures, as the lowest predicted values occur in the western end of the ecoregion. The east-west gradient of temperature change also hints at the possibility of effects from the parallel Rocky Mountains to the east.

Figure 3-3 provides seasonal and annual predictions of average maximum daily temperatures for the ecoregion, based on the multi-model mean results. Table 3-1 also summarizes these temperature predictions in tabular form. Figure 3-4 provides seasonal and annual predictions of average minimum daily temperatures for the ecoregion, based on the multi-model mean results. Table 3-2 also summarizes these temperature predictions in tabular form. The annual average maximum and minimum daily temperatures for the ecoregion are nearly identical. There are some seasonal differences. Summer (June through August) is predicted to have the greatest temperature increases for both the minimums and the maximums. However, for the daily maximum temperature Winter (December through February) has the lowest increase and for the daily minimum Spring (March through May) has the lowest increase.

Table 3-1. Projected Change in Average Maximum Daily Temperature (°C) for the Northern Great Basin Ecoregion

Period	RCP 4.5						RCP 8.5					
	2040-2069			2040-2069			2040-2069			2040-2069		
Annual	2.8	±	0.4	3.5	±	0.6	3.6	±	0.5	5.8	±	0.8
Spring	2.7	±	0.8	3.5	±	0.9	3.3	±	0.9	5.3	±	1.0
Summer	3.3	±	0.6	3.9	±	0.7	4.3	±	0.7	6.7	±	1.0
Fall	2.7	±	0.5	3.5	±	0.7	3.7	±	0.6	5.8	±	1.1
Winter	2.5	±	0.5	3.2	±	0.5	3.2	±	0.6	5.3	±	0.7

Table 3-2. Projected Change in Average Minimum Daily Temperature (°C) for the Northern Great Basin Ecoregion

Period	RCP 4.5						RCP 8.5					
	2040-2069			2070-2099			2040-2069			2070-2099		
Annual	2.7	±	0.4	3.4	±	0.5	3.5	±	0.4	5.7	±	0.8
Spring	2.3	±	0.7	3.0	±	0.8	3.0	±	0.8	4.7	±	0.9
Summer	2.9	±	0.7	3.6	±	0.8	4.0	±	0.7	6.5	±	1.2
Fall	2.4	±	0.5	3.1	±	0.7	3.4	±	0.7	5.6	±	1.2
Winter	2.9	±	0.5	3.7	±	0.6	3.8	±	0.6	6.0	±	0.8

3.2.1 Spring (March, April, May)

Based on the modeling projections, the spring (March, April, May) average maximum daily temperature would increase by 2.7 ± 0.8°C at mid-century (2040-2069) and 3.5 ± 0.9 °C at the end of the century (2070-2099) for RCP 4.5 and by 3.3 ± 0.9 °C at mid-century (2040-2069) and 5.3± 1.0 °C at the end of

the century (2070-2099) for RCP 8.5 (See Figure 3-3 and Table 3-1). In addition, the spring average minimum daily temperature would increase by 2.3 ± 0.7°C at mid-century (2040-2069) and 3.0 ± 0.8 °C at the end of the century (2070-2099) for RCP 4.5 and by 3.0 ± 0.8 °C at mid-century (2040-2069) and 4.7± 0.9 °C at the end of the century (2070-2099) for RCP 8.5 (See Figure 3-4 and Table 3-2). Overall, the eastern portion of the ecoregion would experience the greatest temperature increase and the western portion the lowest temperature increase in Spring (See Appendix A: Figure A-1). A study conducted by Nayak *et al.* (2010) from 1962 to 2006 observed an increase in temperature in spring which resulted in a decreasing proportion of snow to rain at all elevations. As a result, streamflow seasonally shifted to increased winter and early spring flows and reduced late spring and summer flows (Nayak *et al.* 2010).

3.2.2 Summer (June, July, August)

The summer (June, July, August) is especially important because of the potential effect of temperature on increasing potential evapotranspiration. Increased temperatures in the summer could result in reduced soil moisture, higher plant stress, and an increase in wildfire risk. Based on the modeling projections, the summer average maximum daily temperature would increase by 3.3 ± 0.6°C at mid-century (2040-2069) and 3.9 ± 0.7 °C at the end of the century (2070-2099) for RCP 4.5 and by 4.3 ± 0.7°C at mid-century (2040-2069) and 6.7 ± 1.0 °C at the end of the century (2070-2099) for RCP 8.5 (Figure 3-3 and Table 3-1). Additionally, the summer average minimum daily temperature would increase by 2.9 ± 0.7°C at mid-century (2040-2069) and 3.6 ± 0.8 °C at the end of the century (2070-2099) for RCP 4.5 and by 4.0 ± 0.7°C at mid-century (2040-2069) and 6.5 ± 1.2 °C at the end of the century (2070-2099) for RCP 8.5 (See Figure 3-4 and Table 3-2). Overall, the northern portion of the ecoregion is projected to have the greatest temperature increase and the southern portion the lowest temperature increase in summer (See Appendix A: Figure A-2). Summer temperatures are also projected to increase proportionally more than during any other season.

3.2.3 Fall (September, October, November)

Based on the modeling projections, the fall (September, October, November) average maximum daily temperature is expected to increase by 2.7 ± 0.5 °C at mid-century (2040-2069) and 3.5 ± 0.7 °C at the end of the century (2070-2099) for RCP 4.5 and by 3.7 ± 0.6 °C at mid-century (2040-2069) and 5.8 ± 1.1 °C at the end of the century (2070-2099) for RCP 8.5. The fall average minimum daily temperature also is expected to increase by 2.4 ± 0.5 °C at mid-century (2040-2069) and 3.1 ± 0.7 °C at the end of the century (2070-2099) for RCP 4.5 and by 3.4 ± 0.7 °C at mid-century (2040-2069) and 5.6 ± 1.2 °C at the end of the century (2070-2099) for RCP 8.5 (See Figure 3-4 and Table 3-2).The southeast portion of the ecoregion is projected to have the greatest temperature increase and the northwest portion the lowest temperature increase in fall (See Appendix A: Figure A-4).

3.2.4 Winter (December, January, February)

Based on the modeling projections, the winter maximum daily temperature is expected to increase by 2.5 ± 0.5°C at mid-century (2040-2069) and 3.2 ± 0.5 °C at the end of the century (2070-2099) for RCP 4.5 and by 3.2 ± 0.6°C at mid-century (2040-2069) and 5.3 ± 0.7 °C at the end of the century (2070-2099) for RCP 8.5 (Figure 3-1). In addition, the winter minimum daily temperature is expected to increase by 2.9 ± 0.5°C at mid-century (2040-2069) and 3.7 ± 0.6 °C at the end of the century (2070-2099) for RCP 4.5 and by 3.8 ± 0.6°C at mid-century (2040-2069) and 6.0 ± 0.8 °C at the end of the century (2070-2099) for RCP 8.5 (See Figure 3-4 and Table 3-2). The central portion of the ecoregion (Snake River Plain) is projected to have the greatest temperature increase and the western and northeast portions the lowest temperature increases in winter (See Appendix A: Figure A-4).
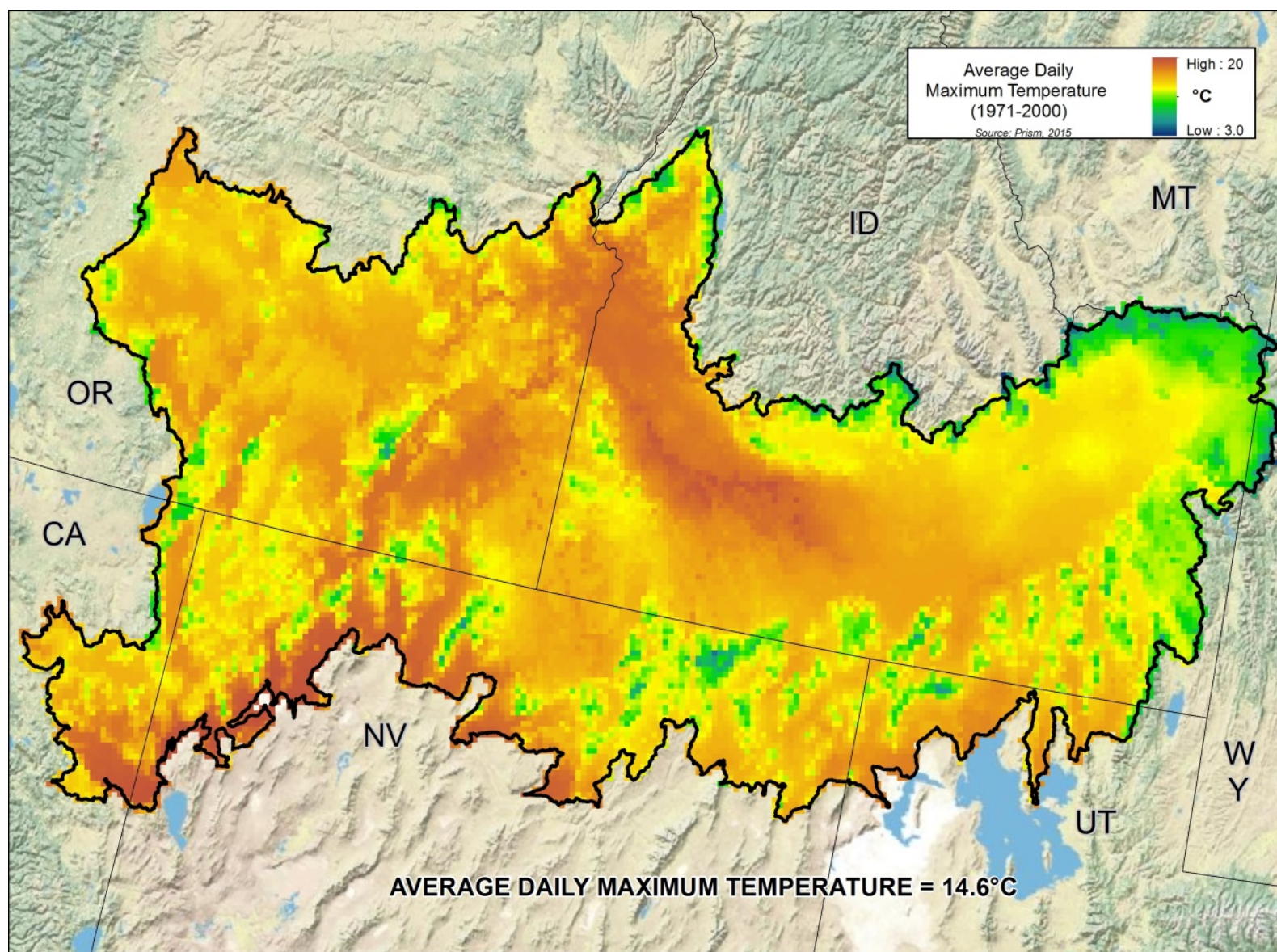
Figure 3-1. Annual Average Maximum Daily Temperature
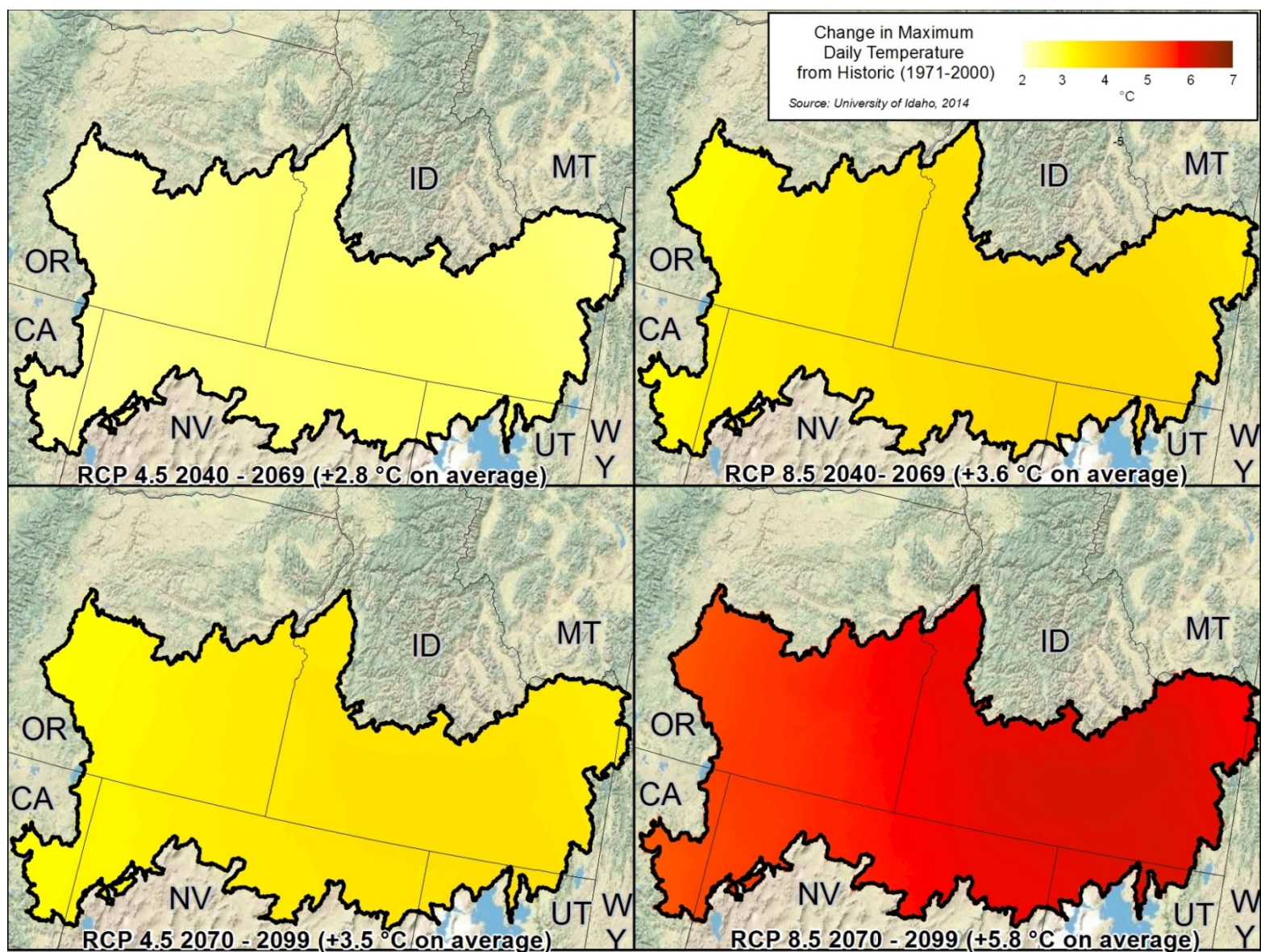
Figure 3-2. Mean Annual Change in Maximum Daily Temperature
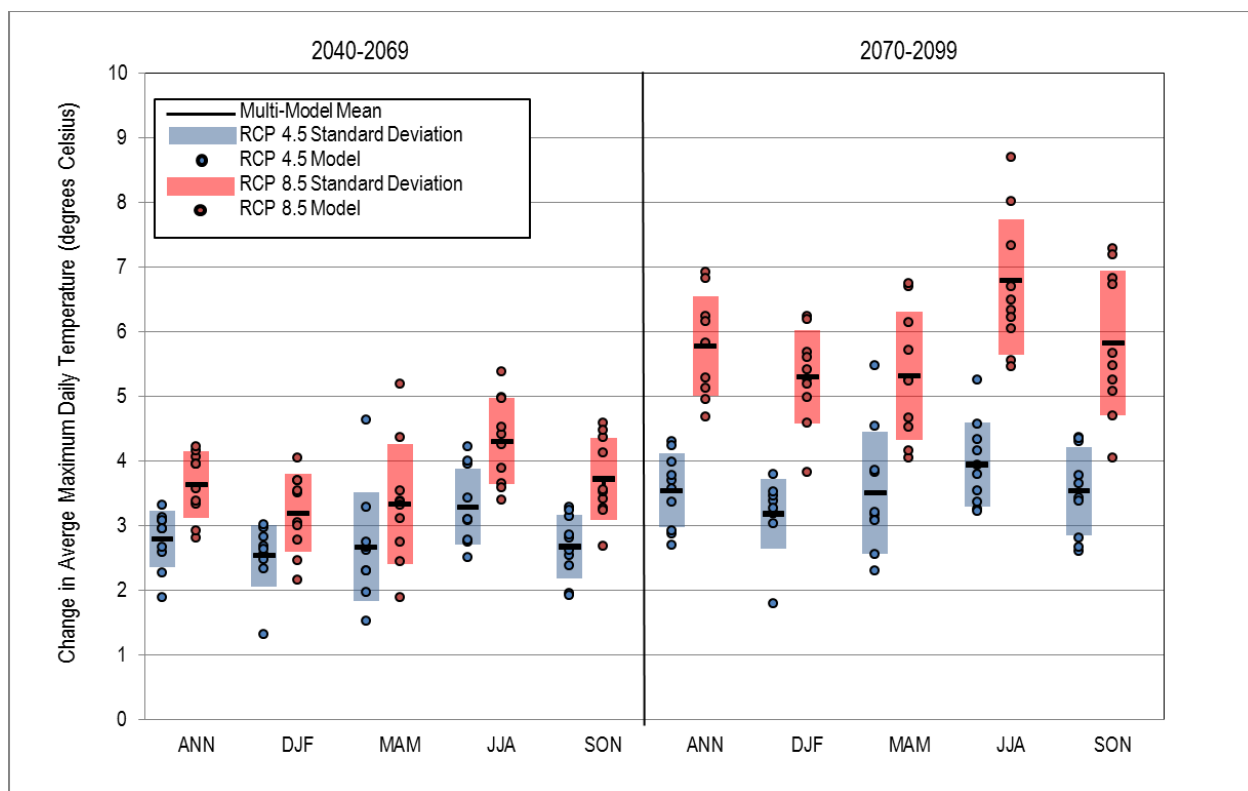
Figure 3-3. Change in Average Maximum Daily Temperature for the Northern Great Basin Ecoregion
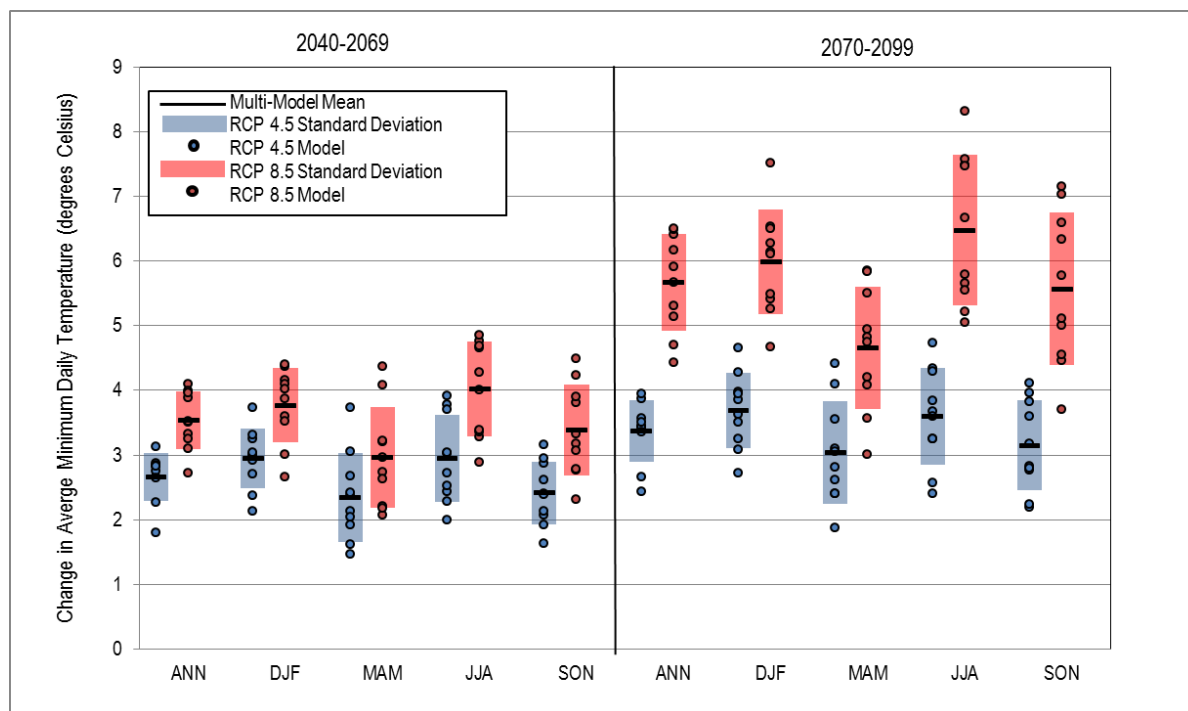
Figure 3-4. Change in Average Minimum Daily Temperature for the Northern Great Basin Ecoregion

3.3 Precipitation Projections

The main source of moisture for precipitation in the ecoregion is the Pacific Ocean. In summer and early fall, atmospheric circulation from the south at higher levels also can produce rain shower and thunderstorm activity. This moisture largely originates from the Gulf of Mexico and Gulf of California and occurs as part of the North American monsoon. The annual average precipitation map for the ecoregion shows the influence of physiography, with the higher elevations having much higher annual precipitation totals compared to lower elevations. Large areas of the ecoregion, including much of Snake River Plains and the lower elevations of the southwestern valleys, receive less than 10 inches of annual precipitation. The annual precipitation averaged over the entire ecoregion from 1971 to 2000 is estimated to be 15 inches (38 cm), with localized values ranging from 6 in (15 cm) to 85 in (216 cm), as shown in Figure 3-5.

The peak season for precipitation at individual locations within the ecoregion is either winter or spring, as determined from a review of historical precipitation records for 18 locations across the ecoregion (Western Regional Climate Center 2016). Generally, the winter precipitation peak occurs in the western part of the ecoregion and the spring precipitation peak occurs to the east. Precipitation during these seasons occurs from Polar storm systems. However, localized convection from surface heating and moisture circulated into the region from the south can contribute to early fall and late spring precipitation. While almost all stations have minimum monthly precipitation in the summer and early fall, these months contribute to larger percentages of the total annual precipitation at locations in the eastern part of the ecoregion compared to the west. This is the case because the eastern part of the ecoregion is more longitudinally aligned with the southerly flow from the monsoon compared to locations further west.

Precipitation is projected to increase in the future throughout the ecoregion. Figure 3-6 displays predictions of changes in annual average precipitation across the ecoregion based on the multi-model mean results. These data show that annual mean precipitation for the ecoregion is predicted to increase by the following:

- For RCP 4.5, 7 percent at mid-century (2040-2069) and 9 percent at the end of the century (2070-2099).
- For RCP 8.5, 9 percent at mid-century (2040-2069) and 18 percent at the end of the century (2070-2099).

The data in Figure 3-5 show that the smallest increases in precipitation would occur in the west/northwest and extreme eastern portions of the ecoregion. The highest increases in precipitation would occur in the central and southcentral parts of the ecoregion.

Figure 3-7 provides seasonal and annual predictions of average precipitation for the ecoregion, based on the multi-model mean results. Table 3-3 also summarizes these precipitation predictions in tabular form. Winter is predicted to have the largest seasonal increase and spring the lowest increase. The following discusses seasonal precipitation changes in more detail.

3.3.1 Spring

Based on the modeling projections, spring precipitation is expected to increase by 5.5 to 9.0 percent at mid-century (2040-2069) and 6.5 to 9.3 percent at the end of the century (2070-2099). Overall, the northeastern portion of the ecoregion would experience the greatest precipitation increase and the western and southeast portions the lowest precipitation increases in spring (See Appendix A: Figure A-5).

3.3.2 Summer

Based on the modeling projections, summer precipitation is expected to increase by 4.0 to 5.4 percent at mid-century (2040-2069) and 11.4 to 15.3 percent at the end of the century (2070-2099). Generally, the southern and southeastern portions of the ecoregion would experience the greatest precipitation increases and the northern portion the lowest precipitation increases (including minor decreases) in summer (See Appendix A: Figure A-6).This precipitation pattern may show the influence of southerly monsoonal flow, which would produce higher precipitation in the elevated terrain in the south and a resulting rain shadow effect to locations further north.

3.3.3 Fall

Based on the modeling projections, fall precipitation is expected to increase by 3.4 to 9.6 percent at mid-century (2040-2069) and 7.5 to 15.5 percent at the end of the century (2070-2099). In general, the southcentral and central portions of the ecoregion would experience the greatest precipitation increases and the western and eastern portions the lowest precipitation increases (including minor decreases in the west) in fall (See Appendix A: Figure A-7).

3.3.4 Winter

Based on the modeling projections, the winter precipitation is expected to increase by 12 to 13 percent at mid-century (2040-2069) and 14 to 30 percent at the end of the century (2070-2099). In general, the southern and central portions of the ecoregion would experience the greatest precipitation increases and the northwestern, northcentral, and eastern portions the lowest precipitation increases in fall (See Appendix A: Figure A-8).

Table 3-3. Projected Change in Precipitation in Percent for the Northern Great Basin Ecoregion

Period	RCP 4.5						RCP 8.5					
	2040-2069			2040-2069			2040-2069			2040-2069		
Annual	6.9	±	5.6	9.5	±	7.2	9.4	±	7.2	17.7	±	9.9
Spring	5.5	±	6.9	6.5	±	9.1	9.0	±	10.8	9.3	±	8.2
Summer	5.4	±	10.4	11.4	±	17.4	4.0	±	11.8	15.3	±	32.3
Fall	3.4	±	8.9	7.5	±	8.6	9.6	±	9.4	15.5	±	12.5
Winter	11.8	±	9.3	13.9	±	12.8	12.8	±	9.9	30.2	±	13.7
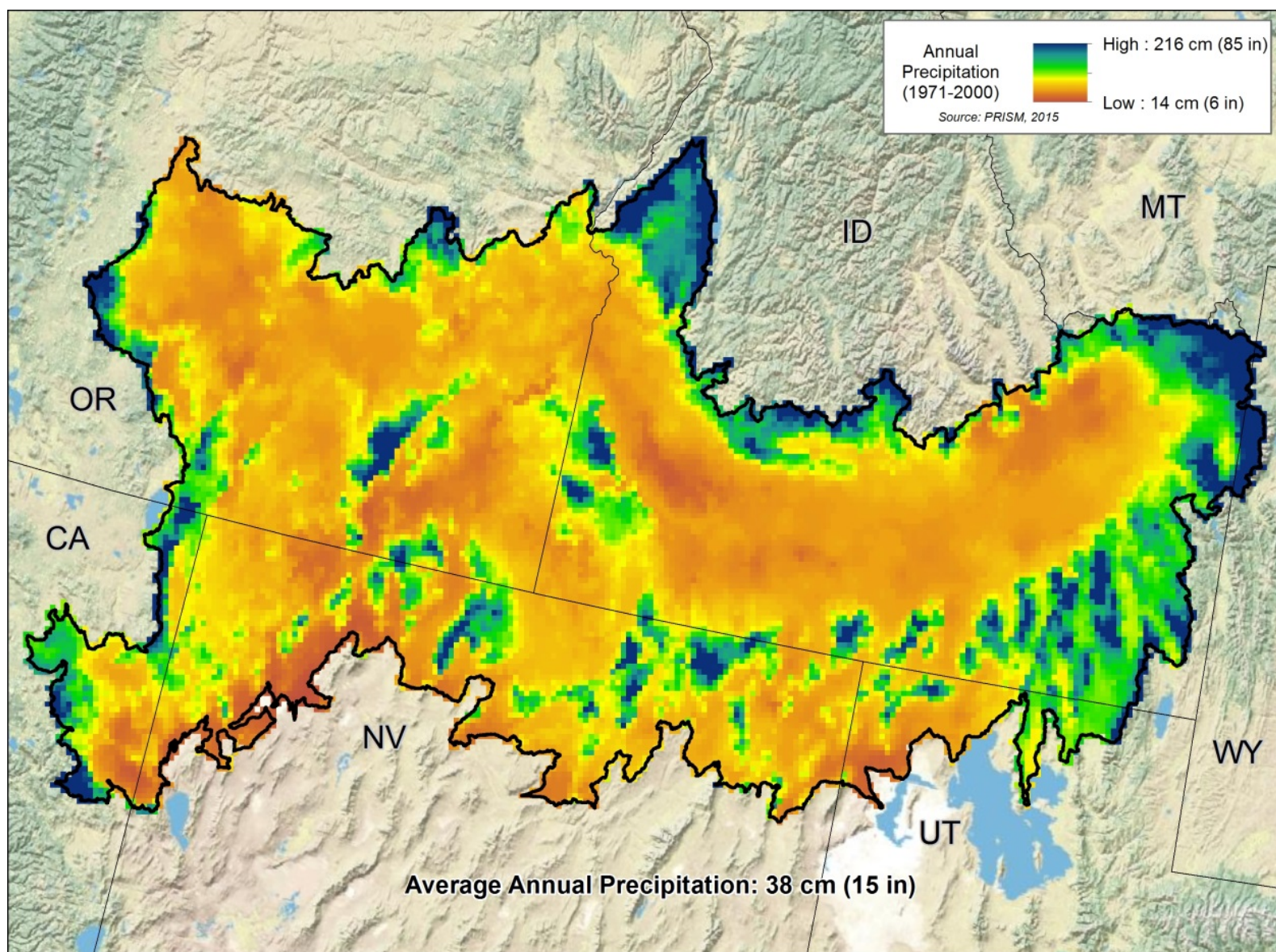
Figure 3-5. Average Annual Precipitation
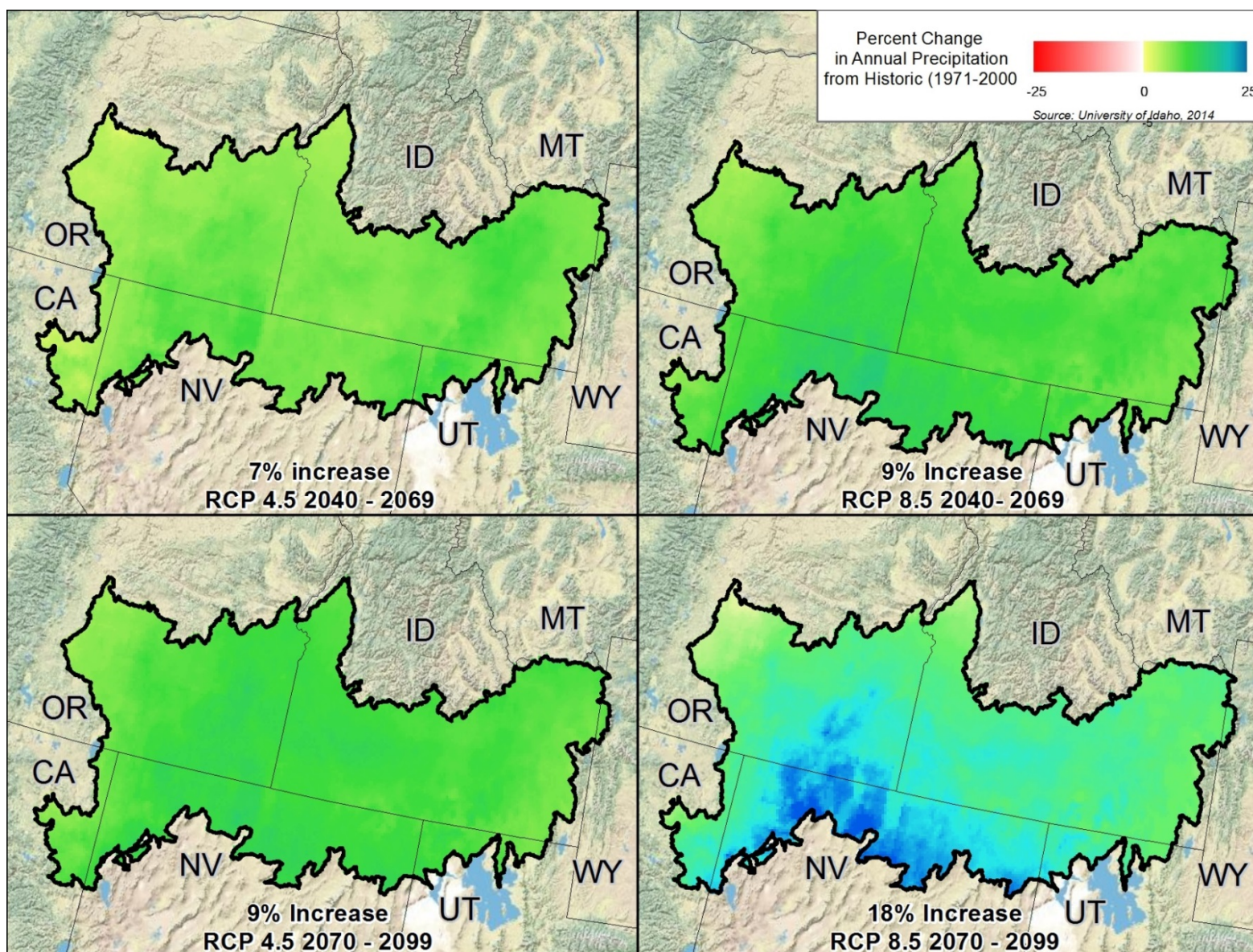
Figure 3-6. Change in Annual Precipitation
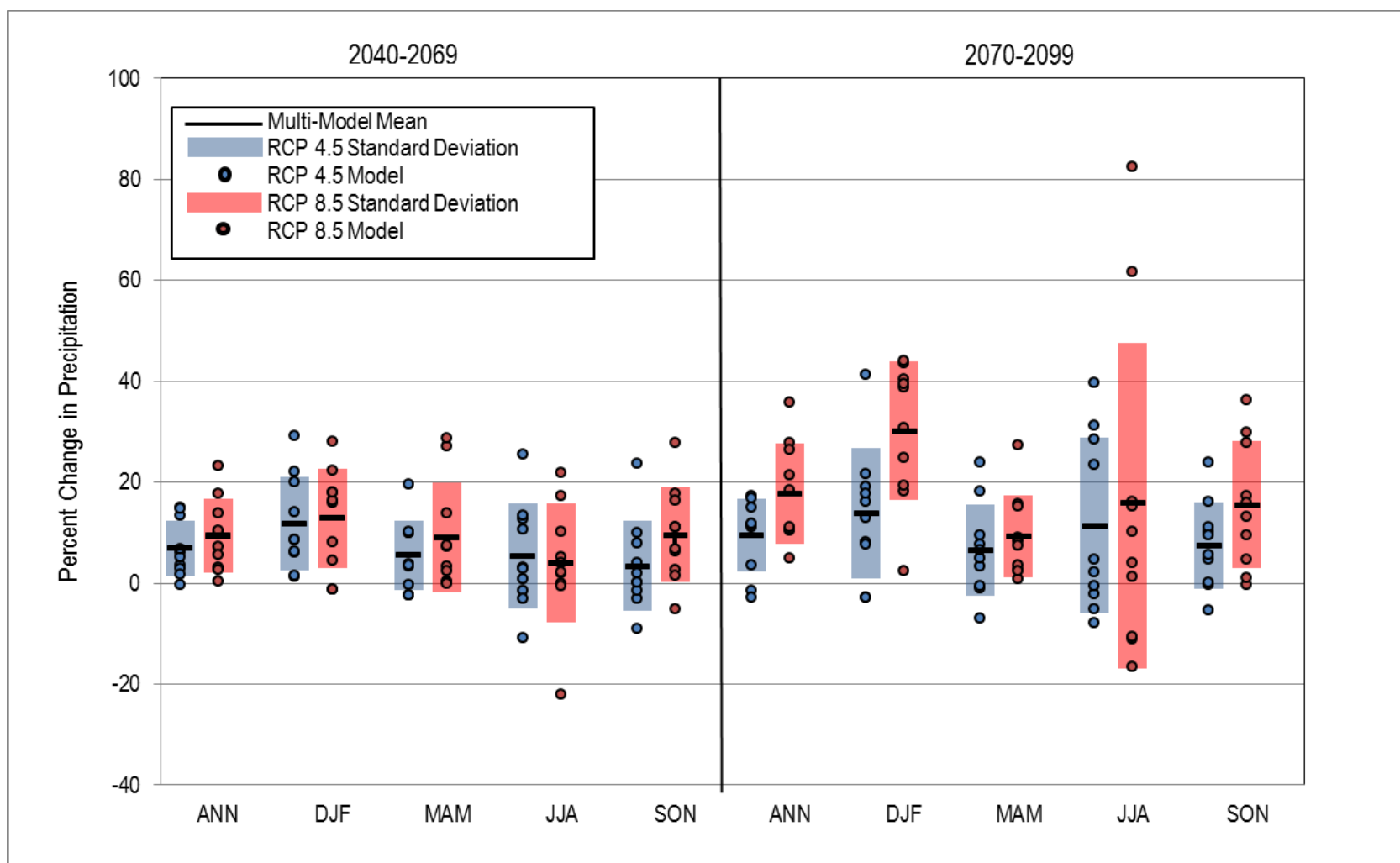
Figure 3-7. Percent Change in Precipitation for the Northern Great Basin Ecoregion

3.4 Projections for Other Variables

3.4.1 Wind Speed

Wind speeds in the ecoregion are the strongest during the spring and lightest during the fall (WRCC 2016). Table 3-4 summarizes these wind speed predictions in tabular form. The modeling predicts that the ecoregion as a whole will experience very minor decreases in future wind speeds for all scenarios and seasons, except that winter would experience very little change in wind speeds. Fall would experience the largest seasonal decrease in wind speeds of any season.

Table 3-4. Projected Change in Average Wind Speed for the Northern Great Basin Ecoregion (m/s)

Period	RCP 4.5						RCP 8.5					
	2040-2069			2040-2069			2040-2069			2040-2069		
Annual	-0.1	±	0.1	-0.1	±	0.1	-0.1	±	0.1	-0.1	±	0.1
Spring	-0.1	±	0.2	-0.1	±	0.1	-0.1	±	0.1	-0.1	±	0.2
Summer	-0.1	±	0.1	-0.1	±	0.1	-0.1	±	0.1	-0.2	±	0.1
Fall	-0.1	±	0.1	-0.1	±	0.1	-0.2	±	0.1	-0.3	±	0.1
Winter	0.0	±	0.1	0.0	±	0.1	0.0	±	0.1	0.0	±	0.1

3.4.2 Downward Shortwave Radiation at the Surface

The available solar energy at a given site is quantified by the downward solar (shortwave) radiation at the surface. This quantity is affected by mostly by cloud cover, but also can be affected by water vapor, trace gases, and aerosols. In the ecoregion the downward shortwave solar radiation is projected to decrease slightly (Table 3-5).

Table 3-5. Projected Change in Downward Solar Radiation in the Northern Great Basin Ecoregion (W/m^2)

Period	RCP 4.5						RCP 8.5					
	2040-2069			2040-2069			2040-2069			2040-2069		
Annual	-0.6	±	2.2	-0.8	±	2.4	-1.7	±	2.9	-3.6	±	3.8
Spring	-0.6	±	4.7	0.1	±	5.6	-1.9	±	6.7	-1.5	±	7.5
Summer	1.0	±	2.5	0.7	±	2.8	-0.4	±	3.1	-3.7	±	5.1
Fall	-0.1	±	0.1	-0.1	±	0.1	-0.2	±	0.1	-0.3	±	0.1
Winter	-3.3	±	2.2	-4.6	±	3.0	-4.9	±	3.2	-8.0	±	3.4

3.4.3 Relative Humidity

Relative humidity is available for some but not all models. The results for relative humidity are not presented due to the lack of the data across all the GCMs.

3.5 Patterns of Future Climate Change References

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4   Climate Change Assessments

This section provides climate change assessments for selected CAs, CEs, and CEGs that were analyzed in the previous REA. Table 4-1 provides a summary of the CAs, CEs, and CEGs that were analyzed and the specific climate modeling outputs (described in Sections 2 and 3) which may positively or negatively affect each agent or element. For each CA, CE, or CEG the assessment includes a summary of the agent or element, a conceptual model, a scientific literature review of the climate influences on the agent or elements, a recommended geoprocessing approach, climate change output modeling results relevant to each CA, CE, or CEG, and management recommendations. Invasive grasses are the only CA addressed, therefore the approach, especially with respect to conceptual models, is different than for the other CEs.

Table 4-1. Climate Change Assessment Categories and Relevant Datasets

Change Agent/Conservation Element Group/ Conservation Element (CA/CEG/CE)	Climate (MACA)	Vegetation (MC2)	Hydrology CIG
Invasives (Annual Grasses)	●Spring Precipitation ●Summer Precipitation	●Fraction of Area Burned	
Shrubs (Sagebrush and Salt Desert Shrub)		●Shrubland Distribution ●Vegetation Carbon	
Tree (Juniper and Other Conifer)		●Woodland Distribution ●Forest Distribution	
Aspen		●Aspen Distribution Overlain on Modeling Results	●Summer Soil Moisture
Vulnerable Soils	●Summer Wind Speed ●Precipitation	●Fraction of Area Burned	
Hydrology (Groundwater, Springs & Seeps, Streams & Rivers, Open Water, Riparian & Wetlands)	●Precipitation		●Ratio of April 1 SWE to Winter Precipitation ●Projected Streamflow
Mule Deer	●Spring Precipitation ●Summer Precipitation		●March SWE
Pronghorn	●Spring Precipitation ●Annual Precipitation	●Woodland Distribution/Vegetation Carbon	
Greater Sage-grouse	●Precipitation ●Temperature	●Woodland Distribution/Vegetation Carbon	
Pygmy Rabbit		●Woodland Distribution/Vegetation Carbon ●Fraction of Area Burned	●SWE
Coldwater Fish			●Projected Streamflow
Spotted Frog			

SWE = Snow Water Equivalent

4.1 Invasives (Annual Grasses)

Across the Northern Great Basin ecoregion invasive annual grasses cover millions of acres and are a threat to various communities from salt desert shrub and sagebrush steppe to ponderosa pine forests. Of main concern is cheatgrass, medusahead (*Taeniatherum caput-medusae*), and ventenata (*Ventenata dubia*). Cheatgrass is a winter annual that colonizes open areas, especially sagebrush steppe communities, after wildfire and ground disturbance, and can be dispersed by livestock, wildlife and vehicles (Young *et al.* 1969; Beckstead *et al.* 2010). The combination of its high seed production (seed bank densities ranging from 10,000 to 30,000 seeds per square meter) and ability to complete its life cycle prior to native perennial bunchgrasses facilitates the colonization and spread of cheatgrass. Furthermore, cheatgrass can germinate in most habitats in the fall or spring, which provides additional advantages (Rivera *et al.* 2011). After repeated disturbances, cheatgrass replaces native perennial grasses. Standing dead cheatgrass forms a dense layer of dry fine fuels that can substantially decrease the return interval of fire (Creutzberg *et al.* 2015). Cheatgrass is thought to have influenced the change in wildfire frequency from return intervals of 32 to 70 years to every 5 years (Pellant 1996). A region wide comparison of land cover maps and historical fire data found that cheatgrass has increased fire activity across the Great Basin. Fires were more likely to start in cheatgrass areas than in other vegetation types and the cheatgrass dominated habitat was associated with increased fire frequency, size, and duration (Balch *et al.* 2013). Studies have also shown the effects of wildfire on cheatgrass are not consistent across the sagebrush steppe. Many studies have found cheatgrass increases in abundance, biomass, or seed production following fire. However, long-term studies have found that cheatgrass response to fire is temporary, with cheatgrass dominating the first 2 years following fire, declining and fluctuating subsequent years, to being negligible after a 10-year period without additional fire (Hosten and West 1994). In addition, some other studies have found that cheatgrass has no response to fire (Menke and Muir 2004). Cheatgrass distribution may be affected by a multitude of factors including livestock grazing (Diamond *et al.* 2010; Reisner *et al.* 2013) and soil borne pathogens (Meyer *et al.* 2014). Improper livestock grazing reduces cheatgrass invasion resistance by decreasing bunchgrass abundance, shifting bunchgrass composition, and thereby increasing connectivity of gaps between perennial plants while trampling further reduces resistance by reducing biological soil crusts (Reisner *et al.* 2013). Once cheatgrass has been established, targeted cattle grazing in spring can reduce the flame length and rate of spread of fires and reduce the potential for catastrophic fires in late summer (Diamond *et al.* 2010).

Livestock grazing can exacerbate cheatgrass dominance when it results in the alteration of native bunchgrass and biological soil crusts (Reisner *et al.* 2013). In cheatgrass-dominated rangelands, targeted grazing can be used to reduce the potential for catastrophic fires during peak fire season in the Northern Great Basin (Diamond *et al.* 2010). Cheat grass die-off events have been observed in the ecoregion where the seed from the previous year fails to emerge. This is occurs in entire stands but the causes are still unknown. The presence of fungus genera including *Fusarium*, *Alternaria*, *Embellisia*, *Aspergillus*, and *Phoma* have been identified in the seed of cheat grass and has been suggested to play a role in cheatgrass die-off (Major 2013; Meyer *et al.* 2014; Meyer *et al.* 2016). Microenvironmental factors that affect levels of water stress are also thought to be important for the relative abundance of soil-borne pathogens and presence of disease on cheatgrass in the field (Meyer *et al.* 2014).

Ventenata and medusahead are also winter annuals that fall in the same functional group as cheatgrasss (Bansal *et al.* 2014). Both annuals are aggressively spreading into sagebrush communities and areas dominated by cheatgrass (Bansal *et al.* 2014). Ventenata and medusahead usually germinate a few weeks later than cheatgrass in the fall. Neither species is palatable to livestock due to their high silica content. Areas invaded by ventenata have yield reductions of 50 percent or more within a few growing seasons (Prather and Steele 2009) Medusahead infestations, although less widespread than cheatgrass in this ecoregion, form monocultures and similarly outcompete native plant species, increasing the risk of large, severe wildfires.

4.1.1 Invasives Conceptual Model

A conceptual model specific to invasive species was not developed for the previous REA; however, invasives are identified as CAs in the conceptual models for most CEs. Figure 4-1 is a state and transition conceptual model developed by McIver *et al.* (2011), which shows the factors and pathways for the state and transitions of sagebrush and invasive annual grassland. Drier systems (10-12 inches annual precipitation areas) as depicted in Figure 4-1, are much more prone to full conversion to annuals, whereas the higher precipitation systems are more resistant (McIver *et al.* 2011). The sagebrush ecosystem reference state consists of dynamic community phases, which change from shrub-dominated to grass-dominated when fire removes sagebrush. Recovery of sagebrush in burned locations requires the establishment of seedlings by seeds surviving in the soil or being dispersed from sagebrush plants that escaped the fire. These sagebrush seedlings grow slowly, increasing in size and dominance over time, and eventually lead to late successional communities represented by a combination of sagebrush and perennial grasses. In sagebrush-dominated areas, fires converts the communities to perennial (or annual depending on ecotype and condition when burned and fire severity) grass-dominated areas, and the cycle continues (Pyke 2011). Where annual grasses invade into the sagebrush community, fire occurs more frequently and the perennial plants (sagebrush and grasses) are eventually lost. The new invasive annual grassland is a stable state due to the competitiveness of invasive annual grasses and frequent fire preventing a transition back to perennial grasses, forbs, and sagebrush. The effects of climate change, wildfire, livestock grazing, and insect, disease, and fungi on invasive annual grasses are provided in Table 4-2.

Table 4-2. Change Agents and their Principal Effects on Invasive Annual Grass Ecosystems

Change Agent	Principal Effects on Invasive Annual Grasses
Climate change	Climate change may influence the spread of cheatgrass, ventenata, and/or medusahead by changing the habitat suitability for invasive annual grasses (Bradley 2009) and affecting the frequency wildfires (Jolly *et al.* 2015). Changing climate conditions could have different effects on the various invasive species. Wetter and warmer winters/springs and hotter summer conditions would provide suitable habitat for invasive annuals, especially at the lower elevations where Wyoming big sagebrush dominates. These areas are easily invaded by cheatgrass, which promotes more frequent fire return intervals.
Wildfire	Cheatgrass tends to have a positive response to fire, with increased biomass or seed production; however, this is not consistent across the sagebrush steppe. The dominance of cheatgrass increases fire frequency in invaded areas, which in turn, favors the establishment of this species by reducing competition with perennials Without fire or other disturbances, eventually (~10 years) perennials can re-establish in cheatgrass-dominated areas. However, if fire occurs frequently (every 10 years or less), cheatgrass can dominate for decades. (Mata-Gonzalez et al. 2007).
Livestock grazing	Livestock may reduce perennial grasses and forbs by grazing and impact cryptogamic crust by trampling, creating areas suitable for annual grass invasion. Cheatgrass could increase as a result of improper grazing of native perennial grasses. Livestock are unlikely to reduce the biomass of ventenata or medusahead due to their poor palatability, and the seeds are likely to be dispersed by the animals.
Insects, disease and fungi	Insects and disease appear to have a minor role in this ecosystem compared to the CAs mentioned above. Fungi have been identified in the seed of cheatgrass and are thought to contribute to die-offs; however, data is inconclusive and requires further investigation.

REFERENCE STATE
Sagebrush Steppe (10-12" Precipitation)

Phase Ib:
Sagebrush/Native
Forbs/Grasses

Fire *

No Fire

Phase Ia:
Native Perennial
Forbs/Grasses

Moderate
Grazing
Impact

Phase II
Sagebrush/Native
Forbs/Grasses

Fire

Heavy
Grazing
Impact

Phase III
Sagebrush/Native
Forbs/Grasses

Fire
???

Biotic
Threshold

Fire
???

DEGRADED STATE

Phase IV:
Cheatgrass-
Dominated

Frequent
Fire*

Invasive
Perennial
Seeds

Fire

Phase V:
Cheatgrass+
Invasive Perennials

Source: McIver *et al.* 2010

State-and-transition model for the sage/cheat system (10- to 12-inch precipitation zone)
focusing on vegetation only. Font size indicates relative dominance of vegetation life form
within each phase. *Fire is assumed to be severe enough to kill most of the woody vegetation.

**Figure 4-1. Conceptual Model for Transition from Sagebrush
to Invasive Annual Grassland (McIver *et al.* 2010)**

4.1.2 Climate Influences

Cheatgrass, ventenata, and medusahead are widespread across the western U.S. and can withstand a range of climates. Relatively little is known about ventenata and medusahead's response to the climatic environment compared to cheatgrass. Cheatgrass can germinate over a wide range of constant and fluctuating temperatures. Seeds can germinate at temperatures just above freezing but germination is inhibited at temperatures above 86 °F (Evans and Young 1972). Cheatgrass is capable of out-competing native species for water and nutrients in spring since cheatgrass is actively growing while many native species are still initiating growth (Pellant 1996). Medusahead grows where extended periods of great cold are lacking (Aubin 2011). It tends to occur in areas that have relatively mild to cold temperatures in winter and hot temperatures in summer. Areas that receive fall, winter, and spring moisture and dry summers provide suitable conditions. It occurs in areas with annual precipitation of 10 to 40 inches, with an upper limit of approximately 50 inches. Cheatgrass is found primarily growing at elevations less than 7,000 feet (Leger *et al.* 2009), but has been documented at elevations of 13,100 feet and higher (Zouhar 2003).

The influence of climate change on future invasions of annual grasses depends on climate suitability that defines a potential species range and the mechanisms that facilitate invasions and contractions. Cheatgrass germination, establishment, and growth depend on adequate precipitation from fall rains and/or from spring rains or spring snowmelt. High precipitation events during the winter/spring increase the cheatgrass biomass (Chambers *et al.* 2007). Cheatgrass growth following wet winters can be so dramatic as to be detected by satellite (Bradley and Mustard 2005). Cheatgrass can fail to establish during drier years and growth is limited during long-term droughts (Chambers *et al.* 2007). The increase predicted in winter precipitation in the ecoregion and associated increase in more extreme wet winters (Abatzoglou and Kolden 2011) could result in the expansion of cheatgrass into drier areas that are marginal habitats (Bradley *et al.* 2016). The potential range of cheatgrass depends also on competitive interactions with the invaded ecosystem. Increases in summer precipitation results in stronger competition from native perennial grass species with more productivity from the native community (Bradley 2009; Chambers *et al.* 2014). However, increases in summer temperatures may reduce overall summer moisture availability, even with a slight increase in summer precipitation, which could negatively affect perennial grasses.

Winter and spring temperatures affect the rate and timing of spring germination for cheatgrass. Colder temperatures generally decrease germination rates, plant establishment, and growth and reproduction of cheatgrass (Chambers *et al.* 2007). Spring soil temperature was the best predictor of the timing and rate of cheatgrass germination across elevation gradients in sagebrush ecosystems of Nevada and Utah (Roundy *et al.* 2007). With increased temperatures cheatgrass will likely spread upward in elevation (Bradley *et al.* 2016). Temperature increases due to climate change may also increase the length of the fire season as well which results in an invasive annual grasses/wildfire positive feedback loop (Abatzoglou and Kolden 2011).

Bansal *et al.* (2014) examined cheatgrass, ventenata, and medusahead's response to soil and watering treatments and found that all species had similar responses. All species had greater shoot growth with increasing clay content in the soil. Medusahead had intrinsically more allocation of biomass to roots and greater root growth in response to environmental manipulations compared to cheatgrass or ventenata. Greater root growth combined with high responsiveness to precipitation is an adaptive strategy for resource uptake in dry climates with variable precipitation (Bansal *et al.* 2014). Consequently, medusahead may have a competitive advantage over other invasive grasses and thus spread faster than others across the western U.S. in response to climate change (Bansal *et al.* 2014). Additionally, climate change is expected to increase temperature and spring precipitation, which could increase invasive annual grass germination rates (Aubin 2011).

In desert shrublands like the Northern Great Basin, fire is influenced by the production of fine fuels that occur in years with wet winters and springs (Bradley *et al.* 2016). Balch *et al.* (2013) found correlations between the frequency and size of fires in areas dominated by cheatgrass and precipitation during the preceding winter. Wetter winters at lower elevations could increase cheatgrass biomass production and promote fires, which result in the mortality of native shrub species and further enable the expansion of cheatgrass (Chambers *et al.* 2014). Cheatgrass-driven fires are also likely if wet winters are accompanied by warmer summers (Bradley *et al.* 2016).

4.1.2.1 Modeling Future Range Shifts with Climate Change

Modeling the impact of climate change on invasive annual grass is generally incorporated into models developed for sagebrush or sage-grouse. Experimental methods have also been used to examine invasive species distribution and predict future shifts in species. Most models and experimental studies available have focused on cheatgrass and there were no studies found that examined the impacts of climate change on ventenata and medusahead.

To better understand climate influences on the distribution of cheatgrass Bradley (2009) used bioclimatic envelope models. The climatic variables that most constrained cheatgrass distribution included summer precipitation, average annual precipitation, spring precipitation, winter maximum temperature, and winter minimum temperature. Sensitivity analysis was also performed on potential cheatgrass distributions using the predictions of ten atmosphere-ocean general circulation models (GCMs) for the year 2100. The model predicted that invasion risk in the western U.S. will decrease in southern Nevada and Utah and expand in parts of Wyoming and Montana. Cheatgrass had the greatest increase in areas with large reductions in summer precipitation. These results could likely be due to the negative effect a decrease in summer precipitation would have on native competing species. Also, the frequency and extent of fire could increase with reduced summer precipitation. Bradley's (2009) analysis highlighted the uncertainty in future climatic conditions, which may result in expansion, contraction, or range shifts of cheatgrass. Overall, Bradley (2009) suggests that throughout the range of all future scenarios, the Northern Great Basin and Snake River Plain region would likely remain suitable for cheatgrass and at risk for invasion or expansion of cheatgrass (Figure 4-2).

It is important to consider that bioclimate models based solely on climate suitability do not account for key factors such as biotic interactions, substrate, the genetic and phenotypic composition of species, and limits to species dispersal. Therefore, the ability of bioclimatic models to forecast the effects of climate change or the spread of invasive species has rarely been adequately tested (Jeschke and Strayer 2008; Pearson and Dawson 2003). Experimentation has been found to be important in understanding the mechanisms that control species range limits (Pfeifer-Meister *et al.* 2013). For example, Bradley's bioclimate envelope model predicts some range contraction of cheatgrass with warming (Bradley 2009). However, Zelikova's *et al.* (2013) experimental study showed that warming could result in increased biomass and annual reproductive output of cheatgrass growth when soil moisture was adequate. During dry years when soil moisture was inadequate, cheatgrass growth diminished.

Additional experimental studies were conducted by Compagnoni and Adler (2014), who measured cheatgrasses response to warming across an elevation gradient in Utah. This study showed that warming could increase cheatgrass density in years with normal to high precipitation. Furthermore, warming caused the greatest increase in cheatgrass performance in high elevation sites where it currently was not problematic. These results suggest that warming due to climate change may increase the susceptibility of higher elevation areas to cheatgrass invasion.

Creutzburg *et al.* (2015) modeled the interrelated effects of climate change, disturbances, and management activities on sagebrush-steppe landscapes with a focus on impacts to the greater sage-grouse habitat. This model integrated information from a dynamic global vegetation model, sage-grouse habitat climate envelope model, and state-and-transition simulation model to project broad scale vegetation

dynamics on 23.5 million acres in southeastern Oregon. Four climate scenarios were evaluated, including continuing current climate and three climate scenarios of global climate change (HadGEM, NorESM, and MRI) from the CMIP5 under RCP 8.5. Management scenarios were also incorporated into the analyses. All climate scenarios projected an expansion of moist shrub-steppe and contraction of dry shrub-steppe, with exotic grasses increasing substantially in the first several decades with increased fire frequency. However, increasing precipitation late in the century resulted in predicted decreases of exotic grasses after initial increases. The Creutzberg *et al.* (2015) model further highlights the sensitivity of cheatgrass invasion to precipitation and wildfire frequency.

In addition to temperature and precipitation, the increased CO_2 concentrations associated with climate change could increase the indigestible portion of above ground plant materials in cheatgrass. These CO_2-induced qualitative changes could, in turn, result in potential decreases in herbivory and decomposition with subsequent effects on the aboveground retention of cheatgrass biomass. Overall, increasing atmospheric CO_2 concentrations above preambient levels may have contributed significantly to cheatgrass productivity and fuel load with subsequent effects on fire frequency and intensity (Ziska *et al.* 2005). The conversion from shrubland to annual grassland also results in a transformation from slow to rapid carbon cycling. The shallow root system of cheatgrass creates less carbon deposition in the deeper soil layers. In addition, wildfire results in the rapid transfer of above ground biomass carbon to atmospheric carbon in both the initial fire of standing shrub biomass and with subsequent increased fire frequency associated with the colonization of non-native annual grasslands (Meyer 2012).

4.1.3 Geoprocessing Modeling and Analysis

Species distribution models for cheatgrass viability have found that the summer precipitation, average annual precipitation, and spring precipitation were the best climate variables for cheatgrass in developing a model of future cheatgrass suitability (Bradley 2009). The projected changes in spring and summer precipitation are presented in Figure 4-3 and 4-4.

The MC2 model does not simulate invasives and likely underestimates the role of fire in changing the trajectory of grasslands, for example, in the intermountain West (Bachelet et al. 2015). Therefore, MC2 grassland vegetation type modeling results are not presented for invasive annual grasses. Increased wildfire size and frequency may increase the spread of invasive annual grasslands in the ecoregion. The estimated increase in fraction area burned is provided for the MC2 results.

4.1.4 Results

An increase in spring precipitation is advantageous to cheatgrass, as it provides additional water when cheatgrass is in its growing season. An increase in summer precipitation would generally support sagebrush and native bunchgrasses. For the ecoregion, both the spring and the summer are expected to have slight increases in precipitation with a 6 to 9 percent increase predicted in spring months (Figure 4-3) and a 3 to 9 percent increase predicted in summer months (Figure 4-4).

The fraction of area burned each year is presented for historical and climate conditions in Figure 4-5. The historical conditions are not based on the actual fires which occurred but the modeling of fire by the MC2 model using historical temperature and precipitation from 1970 to 2000. The fraction of area burned in the ecoregion is expected to increase by 200 percent compared to the historical conditions.

Previous modeling by Bradley (2009) found that most of the Northern Great Basin ecoregion is suitable for cheatgrass. Overall, the evidence from the MACA-downscaled global climate modeling and MC2 fire modeling indicates that invasive plants may be favored by many of the elements associated with climate change, which could enhance the risk of invasive grass expansion. However, it is a challenge to predict the specific effects of climate change on current and potential invasive plants like cheatgrass due to the uncertainty of climate projections, the heterogeneous landscapes of the Intermountain West, and complex

interactions of multiple elements which can produce inconsistent patterns (Bradley 2010; Runyon *et al.* 2012; Bradley *et al.* 2016). Given the uncertainty, management should embrace multiple pathways and rely on adaptive management with respect to invasive annual grasses and climate change in the ecoregion (Bradley *et al.* 2016).
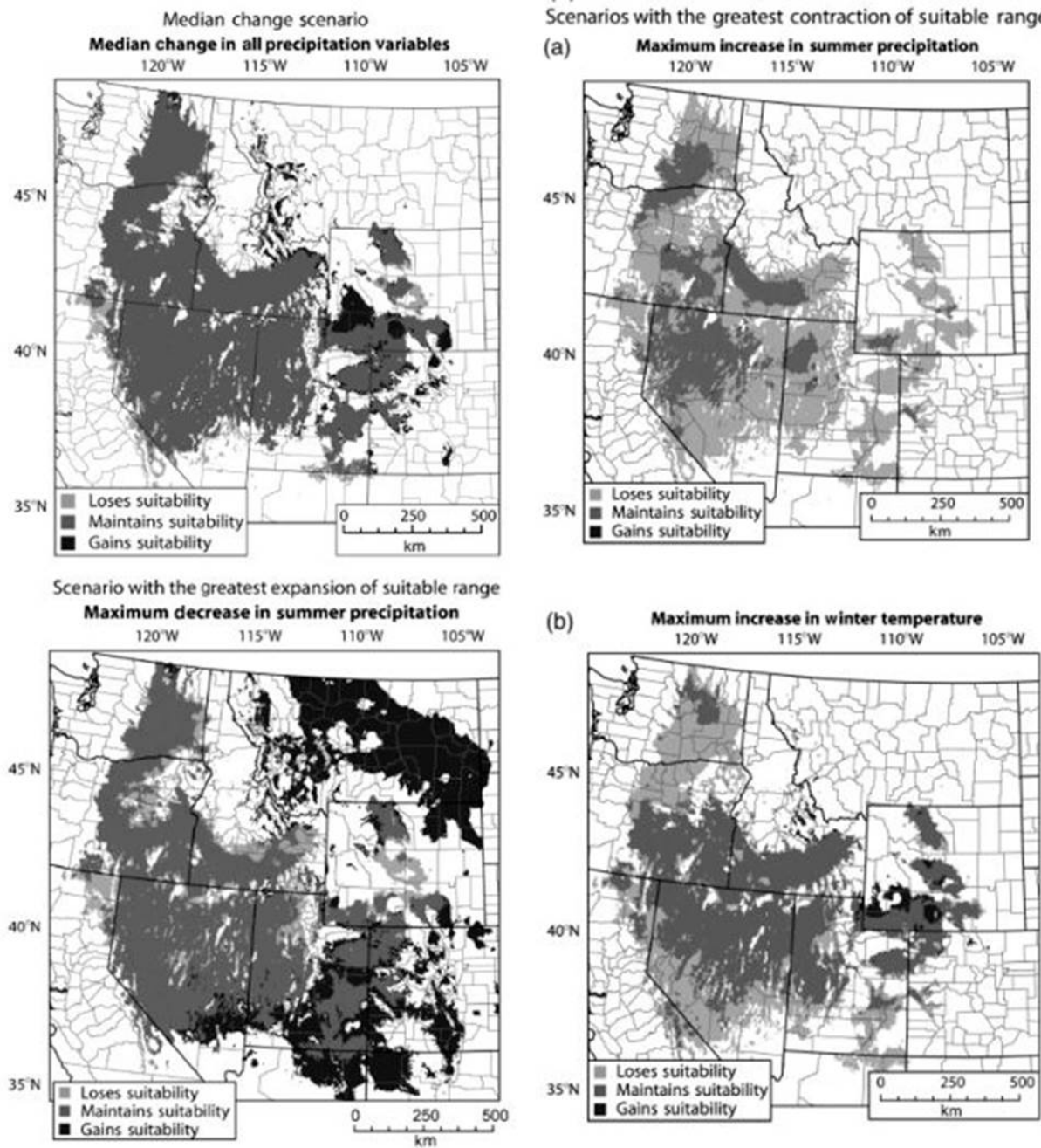
Figure 4-2. Predicted climatic habitat for cheatgrass under various climate model outputs (Bradley 2009)
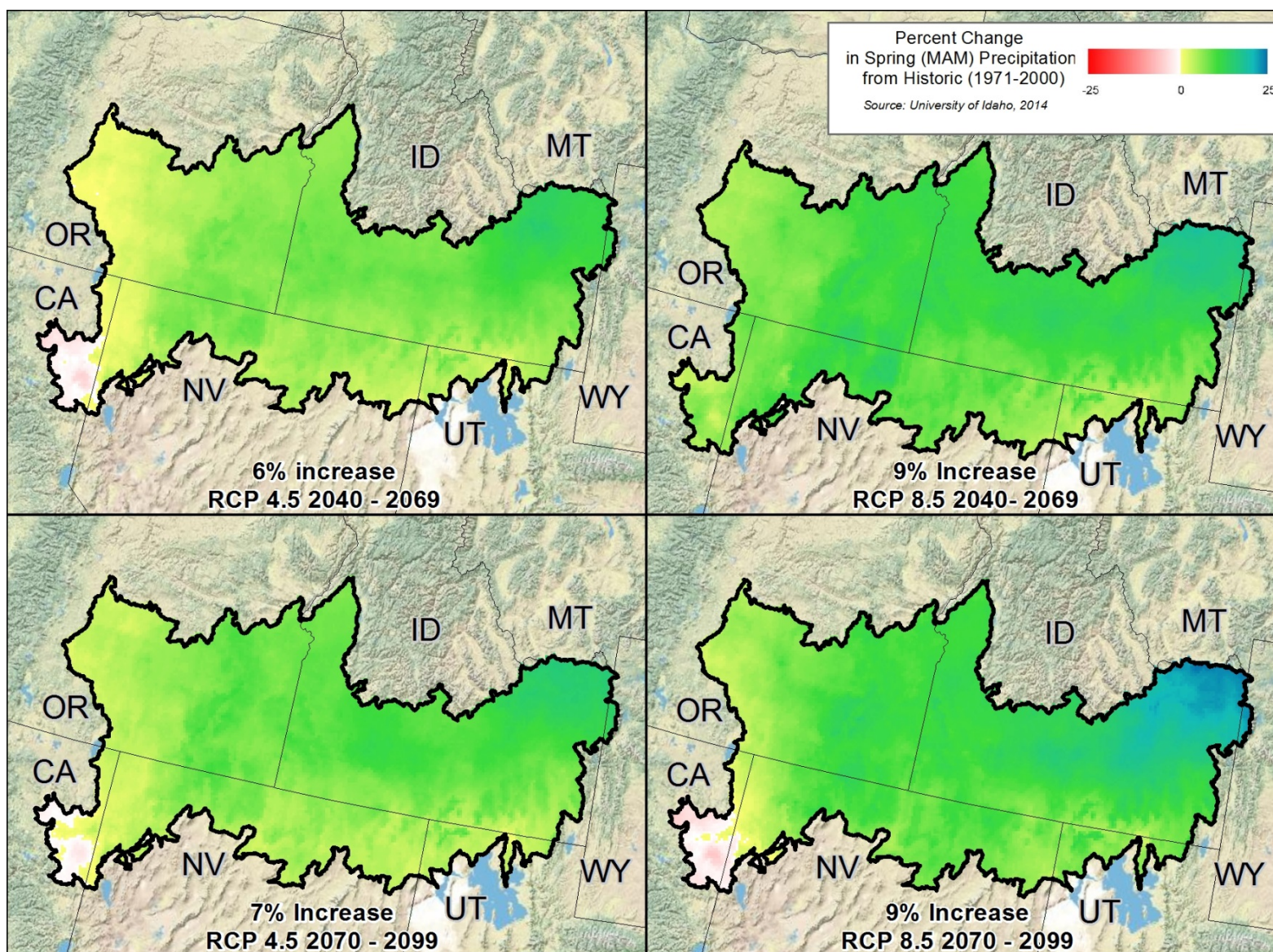
Figure 4-3. Percent Change Predicted in Spring Precipitation
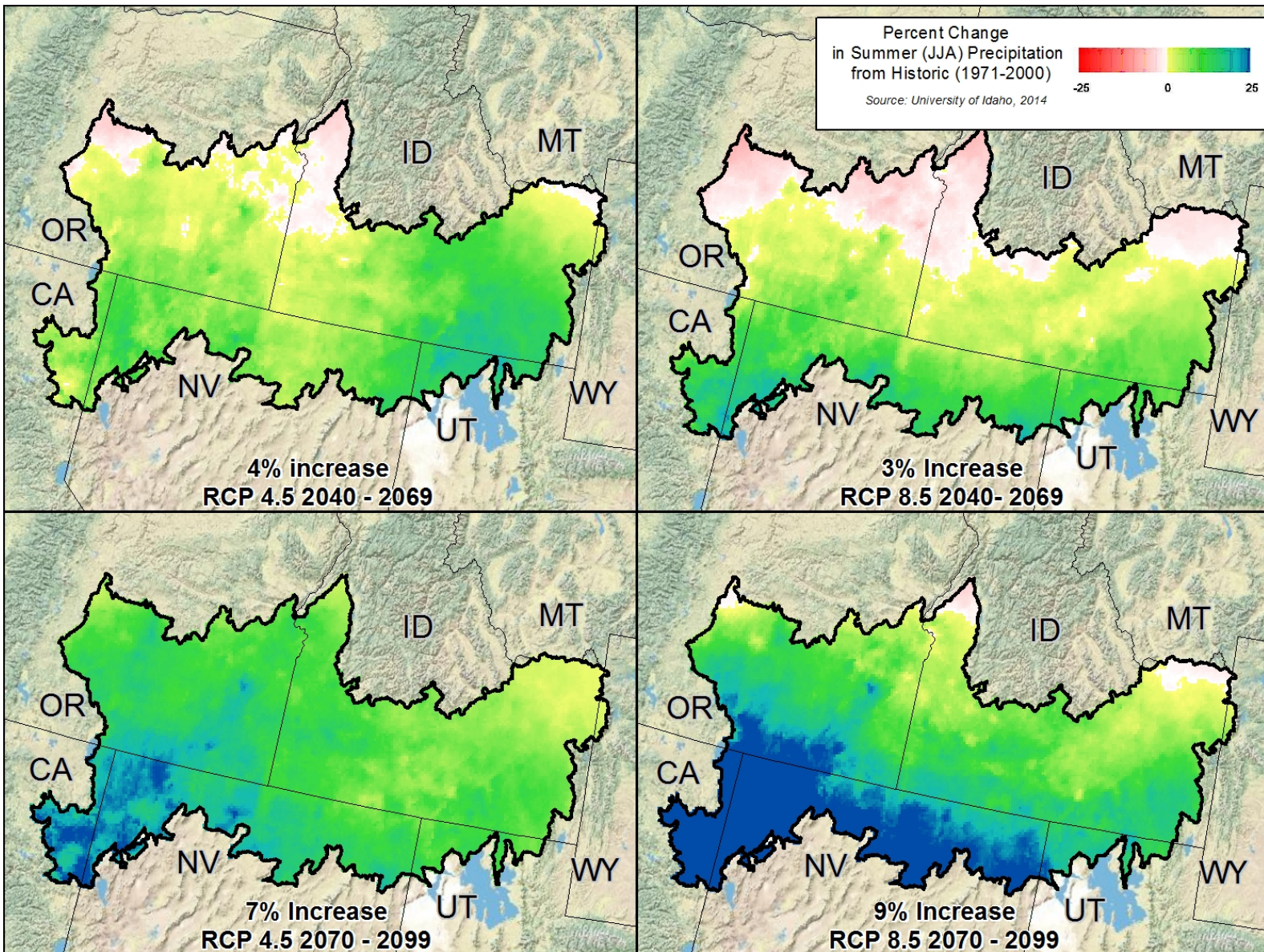
Figure 4-4. Percent Change Predicted in Summer Precipitation
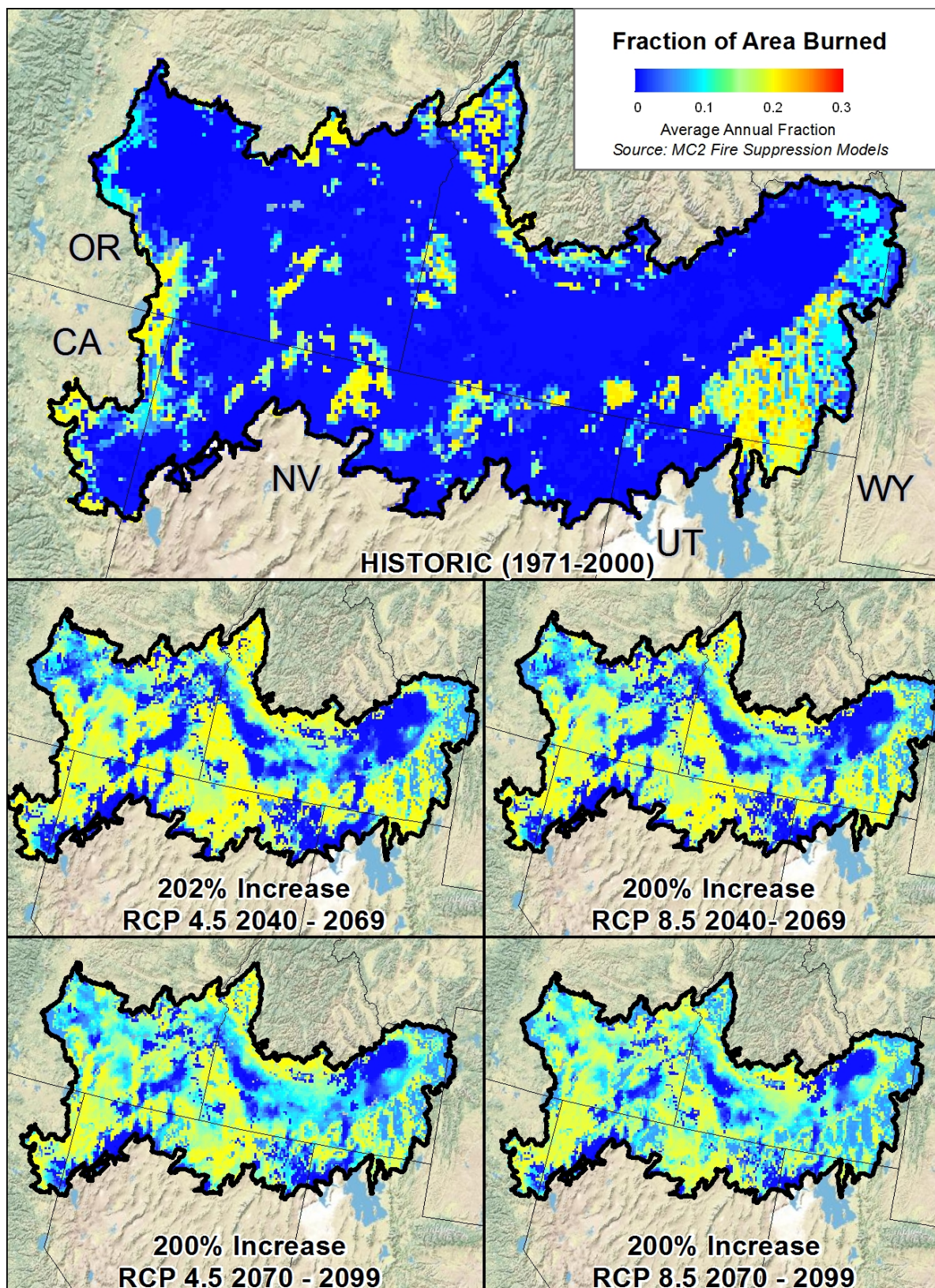
Figure 4-5. Fraction of Area Burned based on MC2 Vegetation Modeling

4.1.5 Invasives References

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4.2 Shrubs (Sagebrush and Salt Desert Shrub)

Sagebrush ecosystems are widespread across the western U.S. and serve as important habitat for a variety of species. Sagebrush habitats are considered some of the most endangered ecosystems and their widespread degradation and vulnerability has led to broad-based ecosystem conservation efforts (e.g., Davies *et al.* 2011; Knick and Connelly 2011; Great Basin Restoration Initiative 2012; Great Basin Consortium 2012). Sagebrush is often separated into two natural vegetation types: 1) sagebrush steppe, where perennial bunchgrasses are frequently codominant and 2) sagebrush, where sagebrush is often the dominant plant (Miller *et al.* 2011). In the Northern Great Basin ecoregion there are numerous species of sagebrush (*Artemisia* spp.) that dominate different sites, generally assorting along soil temperature and moisture gradients. The three most common big sagebrush (*Artemisa tridentata*) subspecies include Wyoming big sagebrush (ssp. *wyomingensis*), basin big sagebrush (ssp. *tridentata*), and mountain big sagebrush (ssp. *vaseyana*). Common perennial grasses include Idaho fescue (*Festuca idahoensis*), bluebunch wheatgrass (*Pseudoroegneria spicata*), Thurber's needlegrass (*Acnatherum thurberianum*), needle and thread (*Hesperostipa comata*), Columbia needlegrass (*Acnatherum nelsonii*), western needlegrass (*A. occidentalis*), California brome (*Bromus carinatus*), squirreltail (*Elymus elymoides*), and sandberg bluegrass (*Poa secunda*). Shorter-statured species including black sagebrush (*Artemisia nova*), little sagebrush (*Artemisia arbuscula*), and silver sagebrush (*Artemisia cana*) are especially important to greater sage-grouse (*Centrocercus urophasianus*), pygmy rabbit (*Brachylagus idahoensis*), and many other obligate species within the ecoregion (Miller *et al.* 2011). These sagebrush species differ significantly in their ability to recover after fire or other disturbance. Sagebrush ecosystems are also commonly used for grazing by domesticated livestock, especially cattle. With decreasing elevation and increasing soil salinity, sagebrush dominated systems give way to salt desert shrub systems.

Salt desert shrubs ecosystems tend to occur in the basins, on sites where soils may be salt-affected, and heat and aridity are locally the greatest. In basins, soils become progressively finer toward the bottom of the basin and precipitation generally declines with decreases in elevation (Blaisdell and Holmgren 1984; Brooks and Chambers 2011; Haubensak *et al.* 2009; Dragt and Provencher 2005). The dominant shrubs may vary considerably from site to site with many areas strongly dominated by a single shrub species. In general, topographic gradients are gentle in areas occupied by salt desert shrub. The saltbush or goosefoot family (Chenopodiaceae) is extremely well represented in salt desert shrub habitats with numerous species of saltbush (e.g, *Atriplex confertifolia*), greasewood (*Sarcobatus* spp.), winterfat (*Krascheninnikovia lanata*), gray molly (*Kochia americana*), and hopsage (*Grayia spinosa*). Black sagebrush (*Artemisia nova*), budsage (*Picrothamnus* [*Artemisia*] *desertorum*), basin big sagebrush (*Artemisia tridentata* subsp. *tridentata*), and rabbitbrush (*Chrysothamnus viscidiflorus*) may be co-dominants or locally dominant. There are a variety of associated perennial grasses such as sandberg bluegrass (*Poa secuda*), bluebunch wheatgrass, Idaho fescue, and Thurber's needlegrass on ranges in good condition. Salt desert shrub ecosystems are commonly used for livestock grazing.

4.2.1 Shrubs Conceptual Model

A conceptual model of sagebrush ecosystems in the ecoregion is presented in Figure 4-6 and a conceptual model of salt desert shrub is shown in Figure 4-7. These models show the general relationships among CAs including climate change, wildfire, livestock grazing, invasive species, land treatments, and insects and disease. In addition, it shows the relationship of these CAs with the sagebrush-steppe plant association. Tables 4-3 and 4-4 summarize the key aspects of CAs relating to sagebrush-steppe and salt desert shrub, respectively.

A key factor not shown in the model is the type or subspecies of sagebrush and the characteristics of the ecological sites. Wyoming big sagebrush, which occurs at lower elevations on drier, less productive sites, is especially vulnerable to conversion to cheatgrass monocultures after fire. In contrast, mountain big

sagebrush, which occurs at higher elevations with higher precipitation, cooler conditions, and more productive sites, is less vulnerable to cheatgrass invasion but is susceptible to juniper invasion under conditions of infrequent wildfire (Miller *et al.* 2011; McIver *et al.* 2010). Sagebrush ecosystems that have been invaded by cheatgrass or juniper are also susceptible to increased wind and water erosion, especially on sloping ground and after fire. This loss of soil can ultimately lead to site degradation and difficulty to return the site to its original shrub or shrub-steppe condition.

Table 4-3. Change Agents and their Principal Effects on Sagebrush Ecosystems

Change Agent	Principal Effects on Sagebrush
Climate change	Climate change may influence sagebrush through its complex influences on the establishment and spread of invasive species and wildland fire. Changing climate conditions could have different effects on the various sagebrush species. Hotter-drier conditions would tend to adversely affect sagebrush especially at the lower elevations where Wyoming big sagebrush dominates. Sites dominated by Wyoming big sagebrush are easily invaded by cheatgrass, which promotes more frequent fire return intervals. Wyoming big sagebrush is killed outright by fire and lacks a persistent seedbank, and is therefore easily lost from local systems as a result of repeated fires.
Wildfire	Frequency, intensity, and areal extent of wildfires are of the greatest importance to sagebrush dominated ecosystems and are in turn affected by characteristics of the vegetation (fuel characteristics) and livestock grazing (which affects vegetation and soils). The dominant big sagebrush subspecies (Wyoming big sagebrush, basin big sagebrush, and mountain big sagebrush) lack the ability to resprout after fire and tend to have short-lived seeds. Because of this, dispersal from surviving (unburned) individuals becomes very important in regeneration, making the areal extent of the fire and the completeness vs. patchiness of the burn critical factors in regeneration. Differences in site productivity as well as variation in seed longevity in the soil may play a role in the rates at which systems dominated by mountain big sagebrush recover from fire compared to Wyoming big sagebrush (e.g., see Wijayratne and Pyke 2009). Seeds of both species are very short-lived unless covered with soil. At suitable sites, very infrequent fire coupled with diminished shrub and perennial grass cover set the stage for invasion by fire-intolerant juniper, which further reduces cover of sagebrush and other shrubs and perennial grasses.
Livestock grazing	Effects of livestock grazing on sagebrush are associated with inappropriate grazing practices or techniques. Livestock may reduce fine fuels as well as shrub and cryptogamic crust by grazing as well as trampling. Reduced fine fuels and shrub cover may lower both the frequency and intensity of wildfires. However, fire frequency may increase if cheatgrass increases as a result of improper grazing of native perennial grasses. In addition, wild horses and burros exceeding appropriate management levels can have similar effects.

Change Agent	Principal Effects on Sagebrush
Invasive species	Cheatgrass and many invasive annual grasses in the ecoregion are short-lived annuals that generally grow in response to winter precipitation and invade open areas where they deplete soil moisture early in the growing season because of their early root growth. They set seed and die early in the growing season and the remaining fine fuels are important in the ignition and carrying of rangeland fires during the dry summer months. They regenerate more rapidly than shrubs after fire promoting a short-return interval fire cycle favoring cheatgrass and selecting against most shrub species including non-sprouting sagebrush species. Other threats to sagebrush ecosystems include the expansion of native vegetation, including western juniper (*Juniperus occidentalis*) and Utah juniper (*J. osteosperma)*, beyond their historic range. Western and Utah juniper have expanded rapidly over the past 140 years into cool, moist sagebrush steppe often converting these areas to woodlands (Soule and Knapp 1999). Utah juniper and western juniper have expanded their distributions into sagebrush steppe since the mid-1800s and especially in the early 1900s. Three phases of increasing juniper expansion into sagebrush steppe (Phase I-III) are recognized (Miller *et al.* 2005). They are distinguished by characteristics including juniper land cover type on the site and the degree of annual leader growth on individuals (which declines as junipers age).
Land treatments	Historical land treatments (e.g. chaining and herbicide application), which involved soil disturbance and removal of native shrub land cover, increased the potential for cheatgrass invasion and wildfire as depicted in the model (Morris and Rowe 2014). More recently, land treatments have involved attempts to restore the native shrub and perennial grass cover (especially after wildfire) and reduce cheatgrass cover. These restorative treatments are not specifically addressed in the conceptual model.
Insects and disease	Insects and disease appear to have a minor role in this ecosystem compared to the CAs mentioned above. However, there has been an increase in outbreaks of the sagebrush defoliator moth (*Aroga websteri*). The sagebrush defoliator moth can cause widespread damage, especially since *Artemesia* spp. are the exclusive larval host. In large numbers larvae can kill hostplants and reduce the production of foliage and flowering by surviving plants for years. Climate change could influence the insect population timing and survival (Bentz *et al.* 2008). In addition, dead sagebrush patches caused by insects equates to dry woody fuels, which could increase fire risk.

Sagebrush-Steppe

Figure 4-6. Sagebrush Conceptual Model

Table 4-4. Change Agents and their Principal Effects on Salt Desert Shrub Ecosystems

Change Agent	Principal Effects on Salt Desert Shrub
Climate change	Climate change may influence salt desert shrub principally through its complex influences on the establishment and spread of invasive species and wildfire. Salt desert shrub communities occupy the lowest elevations and harshest sites in the ecoregion. High temperatures, aridity, and accumulated mineral salts characterize these sites. Upward migration of salt desert shrub in response to climate change is likely to be constrained by soil requirements.
Wildfire	In the ecoregion, wildfire ties directly to growth and persistence of invasive annuals in salt desert shrub. Because of the low productivity, low fuel availability, and low vegetation continuity in salt desert shrub communities, wildfire is believed to have been very infrequent under pre-settlement conditions (>500 years), but has become prevalent in recent years. The increase in wildfire in the salt desert shrub ecosystem is associated with the spread and increasing dominance of invasive annuals, particularly cheatgrass, which have altered vegetation composition and fuel continuity. Wildfire frequency and extent are increased by cheatgrass, while fire intensity is reduced as the shrub component is reduced by fires and is replaced by cheatgrass. A few shrubs such as four-wing saltbush (*Atriplex canescens*), sickle saltbush (*A. falcate*), and black greasewood (*Sarcobatus vermiculatus*), are capable of post-fire resprouting; however, most of the native shrubs typical of salt desert shrub lack specialized adaptations for post-fire regeneration. In contrast, the invasive annual grasses increase in dominance after fire, capitalizing on nutrient release and greater availability of soil moisture due to lack of other vegetation. Additionally, biological soil crusts regenerate slowly after fire, especially when dense stands of annual grasses emerge.
Livestock grazing	Effects of livestock grazing on salt desert shrub ecosystems are associated with inappropriate grazing practices or techniques. The relationship between livestock grazing and salt desert shrub ecosystems is complex. Livestock may reduce fine fuels as well as shrub cover as well as biological soil crusts (cryptogamic crusts) by grazing as well as trampling. Reduced fine fuels and shrub cover may lower both the frequency and intensity of wildfires. Biological soil crusts are important in soil stabilization and are likely adversely affected by trampling associated with heavy grazing. Regeneration of damaged soil crusts is challenging because of low and erratic precipitation and salt-affected soils.
Invasive species	Invasive species most prevalent in the salt desert shrub ecosystem include cheatgrass, halogeton (*Halogeton glomeratus*), Russian-thistle (*Salsola tragus*), and various mustards (Brassicaceae). Cheatgrass and other invasive annual grasses and forbs are short-lived plants that generally grow in response to winter precipitation and invade open areas where they deplete soil moisture early in the growing season because of their early root growth. They set seed and die early in the growing season and the remaining fine fuels are important in the ignition and carrying of rangeland fires during the dry summer months. They regenerate more rapidly than shrubs after fire, promoting a short-return interval fire cycle favoring cheatgrass and selecting against most shrub species.
Land treatments	Historical land treatments (e.g., chaining and herbicide application), which involved soil disturbance and removal of native shrub land cover, increased the potential for cheatgrass invasion and wildfire as depicted in the model (Morris and Rowe 2014). More recently, land treatments have involved attempts to restore the native shrub and perennial grass cover (especially after wildfire) and reduce cheatgrass cover. These restorative treatments are not specifically addressed in the conceptual model. Salt desert shrub, including associated biological soil crusts, is slow to recover following disturbance given the harshness and aridity of the environment and the great year-to-year variation in precipitation.
Insects and disease	Insects and disease appear to have a minor role in salt desert shrub compared to the CAs mentioned above.

Figure 4-7. Salt Desert Shrub Conceptual Model

4.2.2 Climate Influences

Sagebrush ecosystems in the western U.S. typically occur across a strong latitudinal gradient, which suggests adaptation to a range of temperature conditions (Bradley 2010). Annual precipitation in the sagebrush range of the Northern Great Basin ecoregion is extremely variable from year to year. The wettest years on record were 1938 and 1993 with about 530 mm of precipitation, whereas the driest year was 1994 with 140 mm (Bates *et al.* 2006). Sagebrush habitats transition to salt desert shrub in areas receiving <200 mm of annual precipitation, and to pinyon-juniper at higher elevations and precipitation levels. The distribution of sagebrush habitat is likely to shift if precipitation patterns are altered by climate change. Particular sagebrush communities have demonstrated resilience to short term (less than four years) climate perturbations. For example, increased winter precipitation combined with summer drought appears unlikely to cause major changes to productivity of *A. tridentata* communities in the Northern Great Basin (Bates *et al.* 2006). A shift in climate patterns will influence sagebrush ecosystems and to better understand species distribution patterns, climatic influences will likely need to be examined at the subspecies level.

Salt desert shrub communities occupy the harshest sites in the ecoregion with temperature extremes, aridity, and accumulated fine mineral salt affected soils. Elevated levels of saline groundwater may be present in areas occupied by greasewood and certain other halophytes, which have elevated water requirements coupled with tolerance to the elevated salt and mineral content. This community is usually located in the valley bottom or basins, which contain playas that may be periodically flooded. Because salt desert shrub communities are so widely distributed and more related to edaphic than climate factors, the climate is harder to characterize than that of other ecosystem types (West 1983). Total average precipitation for salt desert shrub ecosystems is typically <200mm. Diurnal and seasonal variations in temperature are among the highest in the region. Salt desert shrub communities have very low average relative humidity, low average temperatures, and colder winter extremes than in the nearby sagebrush ecosystems. Summer temperatures are often greater than in the nearby sagebrush ecosystems as well. The uncertainties involved in predicting climate change coupled with the complexity and heterogeneity of conditions and species in salt desert shrub make climate predictions on salt desert shrub complex and difficult.

4.2.2.1 Modeling Climate Influences on Sagebrush Distribution

Climate change poses a substantial long-term risk to sagebrush ecosystems (Bradley 2010; Schlaepfer *et al.* 2012; Homer *et al.* 2015; Creutzburg *et al.* 2015; Still and Richardson 2015). Impacts of climate change on plants could include altering flowering phenologies, an increase in mortality, and an alteration in seed size and quality. Climate conditions may cause currently occupied locations to be unsuitable for sagebrush and could create potential shifts in ecosystem distributions. Modeling the impact of climate change on sagebrush ecosystems is underway; however, inconsistencies in climate projections (e.g. atmosphere-ocean general circulation model [AOGCM]) and differences among model methodologies (e.g. bioclimatic envelope models, linear models, dynamic global vegetation model, and state-and-transition model) have resulted in variations in the predicted distribution of sagebrush. Various climate change modeling efforts have been used to examine sagebrush distribution and predict future shifts in species.

Bradley (2010) analyzed the impacts of climate change on sagebrush in Nevada and California. Ten AOGCM projections were applied to two bioclimatic envelope models (Maxent and Mahalanobis distance). Spatial models were also used to develop state-wide landscape scale risk assessment in association with land use and invasion of cheatgrass and pinyon juniper woodland. Climatic variables of monthly average temperatures and precipitation, and maximum and minimum temperatures from 1971 to 2000 were incorporated into the models. Results showed that 95 percent of sagebrush occurred in areas with annual precipitation greater than 19 cm but less than 52 cm and average minimum temperatures that range from 5.5 to 3.9 °C and average maximum temperatures that range from 10.8 to 18.5 °C . Summer precipitation and

temperature were the best predictors for sagebrush regional distribution, which suggests that summer conditions may have the most impact on longer-term sagebrush viability. Climate change is expected to negatively impact sagebrush at the edge of its current range, particularly in southern Utah, southern Nevada, and eastern Washington. Climate change alone is not thought to be the primary variable affecting the future distribution of sagebrush. Land use changes, juniper woodland expansion, and cheatgrass invasion are also expected to pose major challenges for the conservation of sagebrush and sagebrush obligate species.

Bioclimatic models were also used by Still and Richardson (2015) to examine the distribution of Wyoming big sagebrush (*Artemisia tridentata* spp. *wyomingensis*). Since *Artemisia* covers a wide range and has multiple subspecies it is practical to assess climate profiles at the subspecies level. Wyoming big sagebrush is the most abundant and widespread subspecies of big sagebrush and occupies the warmest and driest areas of the species range. The climate envelope of Wyoming big sagebrush was modeled using Random Forests multiple-regression tree for contemporary and future climates (decades 2050). Still and Richardson's (2015) modeling technique followed Rehfeldt *et al.* (2006) (as explained in Rehfeldt *et al.* 2009 and Crookson *et al.* 2010), which creates a climate profile based from the Random Forest algorithm. The bioclimate model was derived from Wyoming big sagebrush presence and absence points and seven climate variables, of which six were related to temperature and one was related to precipitation (mean annual precipitation). Results indicate that under current conditions, nearly 267 million acres are suitable for Wyoming big sagebrush. In 2050, there will be a 39 percent reduction in suitable habitat (only 66 million acres will be suitable) and 32 percent of the current climate niche for Wyoming big sagebrush is predicted to be lost and 28 percent will be gained. Similar to findings by Bradley (2010) and Schlaepfer *et al.* (2012), the most vulnerable sagebrush areas lie at the southern periphery of the subspecies range including the lower elevations of the Great Basin. Areas that will be more suitable for Wyoming sagebrush to persist include the higher elevations of the Great Basin. However, it is important to consider other factors, such that a subspecies may not be able to colonize areas even where the climatic niche has expanded and suitable habitat is present. This study demonstrates how bioclimatic models can be used to determine which areas are likely to sustain conditions suitable for Wyoming big sagebrush in the upcoming decades, as well as areas that could be used for restoration and conservation (Still and Richardson 2015).

Schlaepfer *et al.* (2012) compared climate-based species distribution models against ecohydrological species distribution models for big sagebrush. The ecohydrology model included a process-based soil-water balance model (SOILWAT) which utilizes a daily time step and a multiple layer, mechanistic model of the soil. Under current conditions both approaches produced comparable sagebrush distributions, though the results from the ecohydrology approach were slightly less accurate. Results suggest that there is a strong decrease in habitat suitability for big sagebrush from 2070 to 2099 in the southern part of the current sagebrush range, with increases in suitable habitat for big sagebrush predicted in the northern parts and at higher elevations. These findings are consistent with Bradley (2010) and Still and Richardson (2015) as discussed above. In 2015, Schlaepfer *et al.* (2015) estimated current and future regeneration under 2070 to 2099 CMIP5 climate conditions at trailing and leading edges that were previously identified. The results of this study predicted an increased probability of regeneration of sagebrush at the leading edge and decreased probability at the trailing edge compared to current levels. This suggests that it will be difficult to restore sagebrush at the trailing edge and that there will be potential conflicts for land managers in maintaining existing grasslands at the northern edge from sagebrush expansion.

In another study in southwestern Wyoming, Homer *et al.* (2015), used remote sensing data (LANDSAT) to determine five sagebrush ecosystem components (bare ground, herbaceous, litter, sagebrush, and shrub) and then compared them to the daily precipitation records from 1984 to 2011. These results were used to create a linear model that examined the relationship between sagebrush abundance and precipitation and then applied to future (2050) precipitation patterns (using Intergovernmental Panel on Climate Change [IPCC] A1B and A2 scenarios). Bare ground was the only component that increased

under both scenarios whereas litter, herbaceous, shrub, and sagebrush showed a decreasing trend in abundance. These future 2050 values were then applied to a contemporary (circa 2006) greater sage-grouse habitat model. Overall, under the two 2050 IPCC scenarios, predicted climate change impacts to sagebrush habitat showed a 12 percent decrease of current sage-grouse nesting habitat, a 4 percent reduction of summer habitat, and less than one percent of new potential habitat gained from 2006 to 2050. Similar to other studies (e.g. Bradley 2010; Still and Richardson 2015; Schlaepfer *et al.* 2012; Schlaepfer *et al.* 2012) peripheral sagebrush habitats will be less resilient to change.

To better understand sagebrush steppe landscapes, Creutzburg *et al.* (2015) integrated information from a dynamic global vegetation model, a sage-grouse habitat climate envelope model, and a state-and-transition simulation model. This model was used to project broad-scale vegetation dynamics and potential sage-grouse habitat on 23.5 million acres in southeastern Oregon. Four climate scenarios were evaluated, including continuing current climate and three climate scenarios of global climate change (HadGEM, NorESM, and MRI) from the CMIP5 under RCP 8.5. Management scenarios were also incorporated into the analyses. All climate scenarios projected an expansion of moist shrub-steppe and contraction of dry shrub-steppe, but varied in the extent of xeric shrub-steppe. Additionally, the Creutzberg *et al.* (2015) model showed a 26 percent increase in wildfire under the current climate and an increase of two- to four-fold under all climate scenarios with exotic grasses also expanding rapidly. If there is no management, juniper is also predicted to expand in southeastern Oregon. However, as climate change increased, the suitable juniper habitat is also reduced because of more frequent wildfire. Overall, this study showed that the rangeland condition will likely decline in the future due to the prevalence of exotic grasses and juniper on the landscape. Sagebrush steppe (sage-grouse habitat) is projected to decline in the first several decades but increase in area under the three climate change scenarios later in the century. Current levels of management treatments were not able to counter the threats of exotic grass and juniper encroachment. However a restoration scenario with higher levels of treated juniper was effective in maintaining woodland encroachment near current levels in priority treatment areas. In general, climate change impacts were more influential than management impacts in the eastern Oregon rangelands, with climate change effects having the potential to be both positive and negative for sagebrush steppe ecosystems.

Soil temperature and moisture regimes of sagebrush communities are a strong indicator of resilience to disturbance and resistance to invasive annual plants (Table 4-5) (Chambers *et al.* 2014). The available data for the soil temperature and moisture regimes were recently compiled from the Natural Resources Conservation Service (NRCS) Soil Survey Geographic Database (SSURGO) with gaps filled with the NRCS State Soil Geographic Database (STATSGO) (Chambers *et al.* 2014). In general, resilience of sagebrush communities (i.e. the capacity of an ecosystem to regain its fundamental structure after disturbance) increases along the elevation gradient due to increasing abundance and cover of fire-tolerant native perennials (Chambers *et al.* 2014). Resistance to invasion also increases at higher elevations due to lower climate suitability for cheatgrass and greater competition from the native community (Chambers *et al.* 2014).

The broad ecological amplitude of the various species of sagebrush present in the ecoregion and the uncertainties involved in predicting climate change make predictions a complex undertaking. Generally, all the range-wide modeling efforts predict a decline of sagebrush in the southern periphery of its range with expansion of sagebrush northward and at higher elevations. Studies suggest that portions of sagebrush habitat in the ecoregion that appear to be in the stable portion of suitable sagebrush climate niche may present important restoration and conservation opportunities for the preservation of sagebrush and sagebrush-obligate species. However, dynamic vegetation modeling coupled with state and transition models show that the prevalence of cheatgrass, the expansion of junipers, and the apparent recent trend toward larger and more frequent wildfires are interrelated factors that need to be taken into account when considering the effects of climate change on sagebrush communities.

Table 4-5. Resistance and Resilience of Ecological Types (Chambers *et al.* 2014)

Ecological Type	Typical shrubs:	Resilience based on Productivity	Resistance to Invasive Grasses
Cold and Moist (Cryic/Xeric)	Mountain big sagebrush, snowfield sagebrush, snowberry, serviceberry, silver sagebrush, and/or low sagebrushes	Moderately High	High
Cool and Moist (Frigid/Xeric)	Mountain big sagebrush, antelope bitterbrush, snowberry, and/or low sagebrushes Piñon pine and juniper potential in some areas	Moderate High	Moderate
Warm and Moist (Mesic/Xeric)	Wyoming big sagebrush, mountain big sagebrush, Bonneville big sagebrush, and/or low sagebrushes Piñon pine and juniper potential in some areas	Moderate	Moderately Low
Cool and Dry (Frigid/Aridic)	Wyoming big sagebrush, black sagebrush, and/or low sagebrushes	Low	Moderate
Warm and Dry (Mesic/Aridic)	Wyoming big sagebrush, black sagebrush and/or low sagebrushes	Low	Low

4.2.2.2 Modeling Climate Influences on Salt Desert Shrub

There were no studies found that analyzed the impacts of climate change on salt desert shrub. Modeling in southeastern Oregon by Creutzburg *et al.* (2015) excluded the lowland salt desert shrub communities from their analysis because they are restricted to topographic features (such as playas) that are not adequately modeled using their methods.

4.2.3 Geoprocessing Modeling and Analysis

Predicted Changes in Types of Sagebrush

At a landscape scale, greater sage-grouse require large, interconnected expanses of sagebrush ecosystems, with varying densities and heights of sagebrush cover across different soil moisture regimes (Doherty *et al.* 2008). Potential changes to sagebrush ecosystems are discussed in Section 4.2. To evaluate potential changes in sagebrush-steppe types which could impact sage-grouse, the potential changes in ecological types of sagebrush based on temperature (cold, cool, warm) and moisture (moist/dry) were evaluated using the thresholds provided in Table 4-6 using PRISM data and MACA climate outputs. The moisture regimes are divided into dry and moist categories by an annual precipitation of 32 cm (Creutzburg *et al.* 2015) which is consistent with the annual precipitation of greater than 12 inches described in Chambers *et al.* (2014) for the moist regimes. Thresholds for temperature regimes (cool, warm, hot) are based on standard NRCS temperature regimes: Frigid (Cool) <8 °C, Mesic (Warm) 8 to 15 °C, and Thermic (Hot) 15 to 22 °C. To estimate the annual average increase in temperature, the change in annual average maximum and minimum temperatures provided by the MACA downscaled datasets were averaged.

Table 4-6. Ecological Types Based on Soil Moisture and Temperature Regime

Ecological Type		Soil Temperature Regime		
		Cool (Frigid) Annual Temp <8 °C	Warm (Mesic) 8 to 15 °C	Hot (Thermic) 15 to 22 °C
Moisture	Dry (Aridic)	Cool and Dry	Warm and Dry	Hot and Dry

Regime	Annual Precip <32 cm	(Frigid/Aridic)	(Mesic/Aridic)	(Mesic/Aridic)
	Moist (Xeric) Annual Precip >32 cm	Cool and Moist (Frigid/Xeric)	Warm and Moist (Mesic/Xeric)	Hot and Moist (Mesic/Xeric)

MC2 Modeling for Temperate Shrubland

In addition, the MC2 dynamic global vegetation model results showing the historic 1971–2000 vegetation types were compared to the modeled vegetation under the RCP 4.5 and 8.5 scenarios for the 2036–2065 and 2071–2100 time slices. These vegetation models are based on the CMIP5 MACA downscaled datasets. Areas that are predicted to be sagebrush are the temperate shrubland vegetation class. Areas that are predicted to be salt desert shrub are in the xeromorphic shrubland vegetation class (Table 4-7).

Table 4-7. Conservation Elements and MC2 Vegetation Classes

Conservation Element	MC2 Vegetation Class	Common Species
Sagebrush	Temperate shrubland	Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) and antelope bitterbrush (*Purshia tridentata*)
Juniper	Temperate Evergreen Needleleaf Woodland	Western juniper (*Juniperus occidentalis*), mountain big sagebrush (*Artemisia tridentata* spp. *vaseyana*)
Salt Desert Shrub	Xeromorphic shrubland	Greasewood (*Sarcobatus vermiculatus*), saltbush (*Atriplex canescens*), and Wyoming big sagebrush

Source: Adapted from Halofsky *et al.* 2013

4.2.4 Results

The results of the species distribution models that show the future viability of sagebrush or salt bush were not presented in the previous REA. Since the REA process was completed, modeling efforts that have examined sagebrush species distribution within the ecoregion have been performed. As discussed above, the Schlaepfer *et al.* (2012) model for distribution of big sagebrush in the Western U.S., which used both climatic and ecohydrology, shows that most of the big sagebrush in the ecoregion is stable; however, there is some decrease in sagebrush projected in the northern Utah portion of the ecoregion (Figure 4-8). Bradley's (2010) bioclimatic envelope models also show similar trends of sagebrush in the ecoregion where climate change is expected to negatively impact sagebrush at the edge of its current range (Figure 4-9). The Still and Richardson (2015) bioclimatic envelope models for Wyoming big sagebrush suggests that by 2050 there will be a contraction of Wyoming big sagebrush habitat in the western portion of the Snake River Plain and in northern Utah (Figure 4-10).

The current sagebrush distribution developed for the REA is presented in Figure 4-11. Wyoming/Big Basin sagebrush communities are the dominant sagebrush type in the ecoregion. The soil temperature/moisture regime based on soil mapping by the NRCS is presented for the current distribution of sagebrush in the ecoregion in Figure 4-12. Using the thresholds provided in Table 4-6, the temperature/moisture regimes for sagebrush based on the historic PRISM data is provided in Figure 4-13. The temperature/moisture regimes has slightly more moist regimes than the NRCS mapping. The predicted change in sagebrush types based on the MACA-downscaled climate outputs are graphically displayed in Figure 4-14. In general, climate change would be expected to result in a shift from cool/dry and cool/moist regimes to warm/dry and warm/moist regimes which are less resilient and resistant to invasive grass invasion. In RCP 8.5, late period 2070-2100, some sagebrush areas would shift to hot/dry and hot/moist regimes. As no sagebrush in the ecoregion currently occur in hot/dry and hot/moist regimes, this could shift to a new vegetation type. Overall, consistent with the invasive grass analysis in Section 4.1, climate change would result in a shift in temperature/moisture regimes for sagebrush

communities which would likely make the sagebrush communities less resilient to disturbance and less resistant to invasion from invasive annual grasses.

The MC2 DGVM results are presented for the historic 1971–2000 temperate shrubland vegetation type and compared to the predicted temperate shrubland vegetation under the RCP 4.5 and 8.5 scenarios for the 2036–2065 and 2071–2100 time slices (Figure 4-15). The MC2 modeling predicts a significant loss and redistribution of the temperate shrublands within the ecoregion. Temperate shrublands generally correspond to Wyoming big sagebrush and antelope bitterbrush in the ecoregion (Table 4-7). The temperate shrublands are modeled to be largely replaced by temperate evergreen needleleaf woodlands. The MC2 vegetation model simulates this extensive woody expansion throughout the U.S., which is caused in part from an increase CO_2 concentrations resulting in increased water efficiency of woody plants (Bachelet *et al.* 2015). However, modelers note that the expansion into grass-dominated areas would be overestimated where increased invasive grasses increase fine fuel sources, which would result in more frequent fires that could prevent tree and shrub establishment (Bachelet *et al.* 2015). Given the complex interactions between CO_2 concentrations, climate, and wildfire, the results from the MC2 model for the ecoregion should be carefully interpreted. Alternatively, the effects of increased CO_2 concentrations is generally not included in species distribution models, and based on the MC2 results, increased CO_2 may have a significant effect on increasing woody vegetation types.

The differentiation of vegetation types between shrubland, woodland, and forest in the MC2 model is based on the modeled vegetation carbon. Different base calibrations for the MC2 model will have different potential vegetation types. The default base calibration for MC2 results is the CONUS (Continental U.S.). However in 2015, Creutzberg *et al.* (2015) modified the base calibration to improve the modeling of rangeland ecosystems in southeast Oregon (SEO). This involved an adjustment of parameters for vegetation carbon thresholds, including combining grassland and shrubland into a shrub steppe category which contains significant components of shrub and grassland forms. They also raised the thresholds for woodland and forest vegetation types. A similar study in the Blue Mountains (BLUE_MTNS) of northeast Oregon also raised the threshold for shrubland vegetation types. The total vegetation carbon is provided in Figure 4-16, classified according to the thresholds breaks in Table 4-8. Overall, total vegetation carbon is expected to increase 43 to 73 percent with climate change. However, the reduction in shrubland presented based on the MC2 results may be overestimated as studies within or adjacent to the ecoregion applied higher thresholds for shrubland vegetation carbon in determining vegetation types. In addition, as discussed the previous section, the MC2 model does not factor in the effects of invasive plants like cheatgrass which may increase the fire frequency, thus reducing the expansion of woody plants and vegetation carbon in the ecoregion.

Table 4-8. Vegetation Type Carbon Thresholds (g/m2)

Vegetation Type	CONUS Base Calibration	BLUE_MTNS Base Calibration	SEO Base Calibration (Creutzberg *et al.* 2015)
Grassland	0 – 385	0 – 385	0 – 3000
Shrubland	385 – 1150	385 – 1822	Shrub steppe
Woodland	1150 – 3000	1822 – 2852	3000 – 3800
Forest	3000 +	2852 +	3800 +

Source: Cruetzberg *et al.* 2015, MC2 Model Documentation
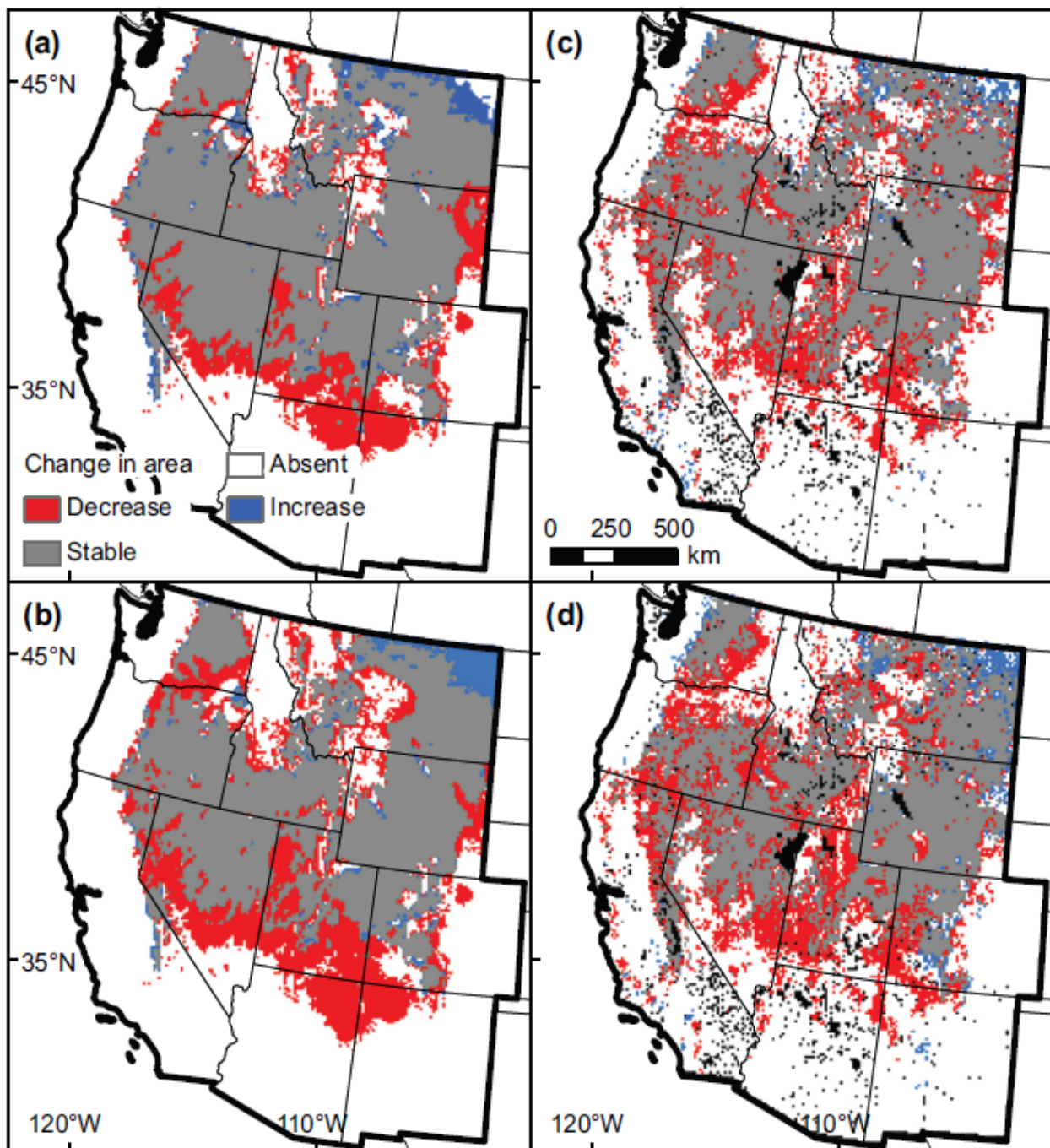
Figure 4-8. Maps of change in predicted sagebrush ecosystem area (Schlaepfer *et al.* 2012 [(Figure 3])

Sagebrush ecoystem area based on ensemble SDM using the climate dataset (a) – 9b) and the eco-hydrolocal dataset (c)-(d) bewteen the current 1970-1999 climate and future B1 (a), (c) and A2 (b), (d) 2070 -2099 climate scenarios of western U.S. Black cells indicate data not available.

(A) Current sagebrush distribution

Sagebrush present

(B) BEMs of current suitable climate

Current suitable climate
MD
MAXENT
MD + MAXENT

(C) Projected climatic suitability in 2100

\# Models (out of 20) predicting suitability
>10 8 6 4 2 0
Low — High
Risk from climate change
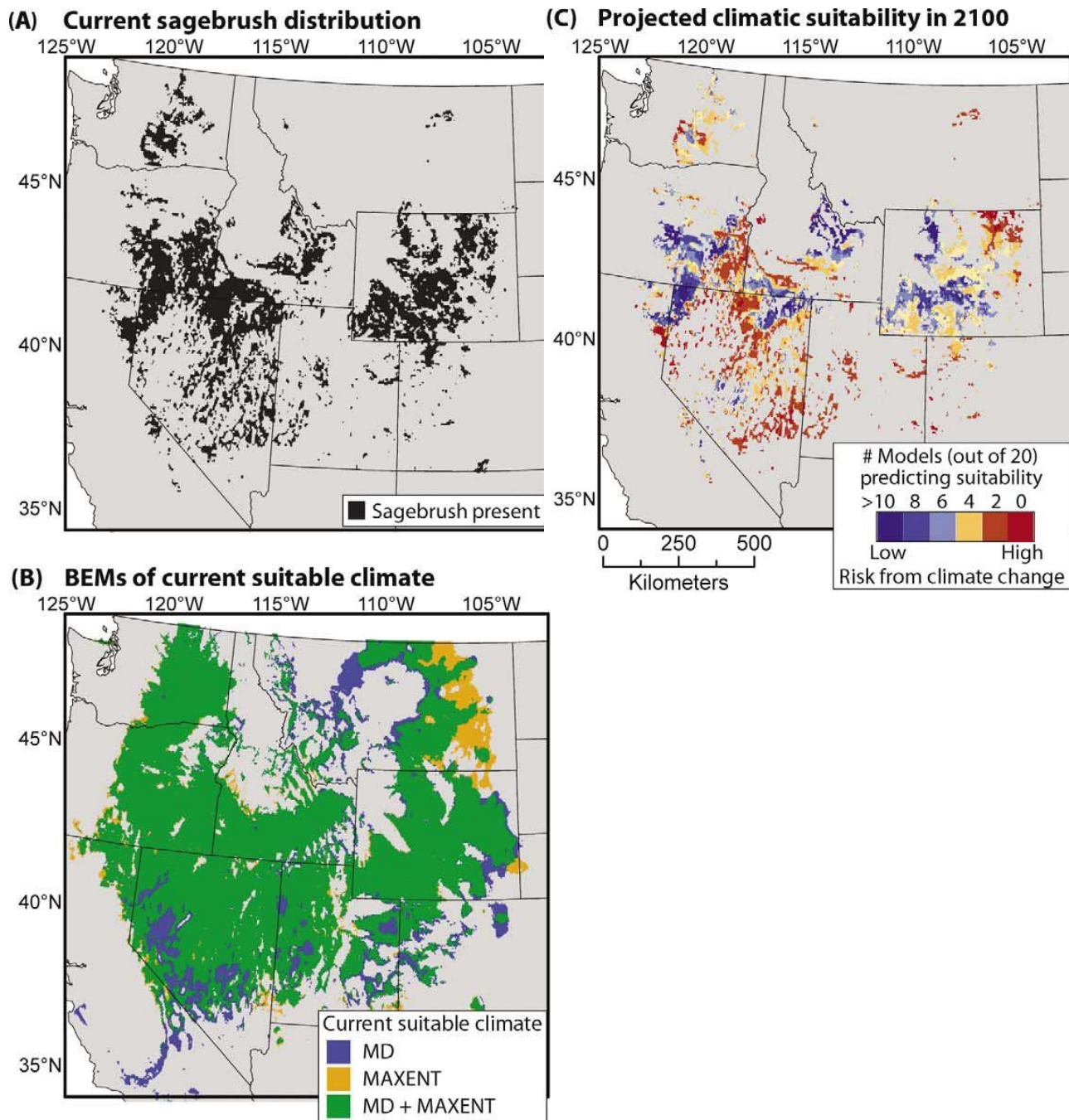
Figure 4-9. Current climatic suitability for sagebrush (*Artemisia* spp.) in the western U. S. and future climate change risk to sagebrush (Bradley 2010 [Figure 8])

(A) Current sagebrush distribution based on sagebrush (Comer et al. 2002) scaled to 4 km resolution is shown in black.(B) Suitable climate for sagebrush from bioclimatic envelope models based on Mahalanobis distance (MD) and maximum entropy (MAXENT). (C) Within the current sagebrush distribution, combined number of envelope models and AOGCMs (out of 20) that project maintained climatic suitability by 2100. Warmer colors are projected to be climatically suitable under fewer climate scenarios and are at greater risk from climate change. Cooler colors remain climatically suitable in multiple climate scenarios and are at lesser risk from climate change.
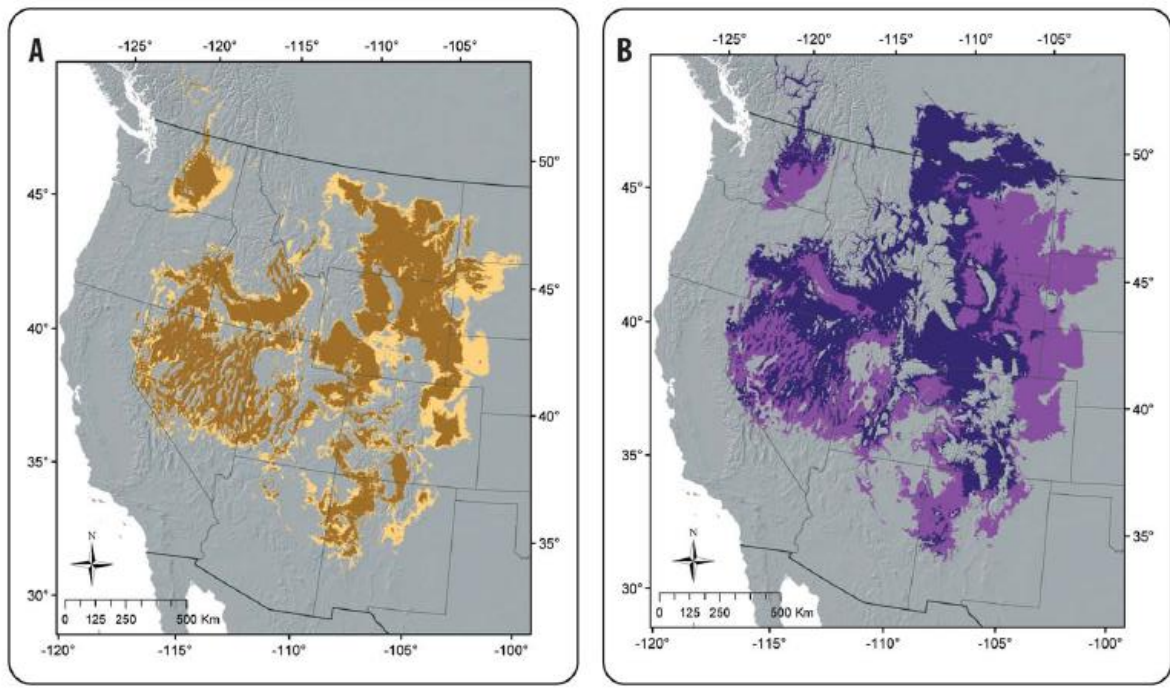
Figure 4-10. Mapped projection of contemporary and 2050 climate niche of Wyoming Big Sagebrush (Still and Richardson 2015 [Figure 1])

A) Mapped projection of the contemporary climate niche of Artemesia tridentata subsp. wyomingensis. Dark brown represents higher probability of occurance (>0.75), whereas light brown represents lower probability (0.5 to 0.75). B) Mapped projection of the change in climate niche between contemporary and decade 2050. Dark purple represents areas that are predicted to have suitable climate for this subspecies in decade 2050 (i.e. stable or expanded), whereas light purple represents areas that are predicted to have unsuitable climates (i.e. contracted).
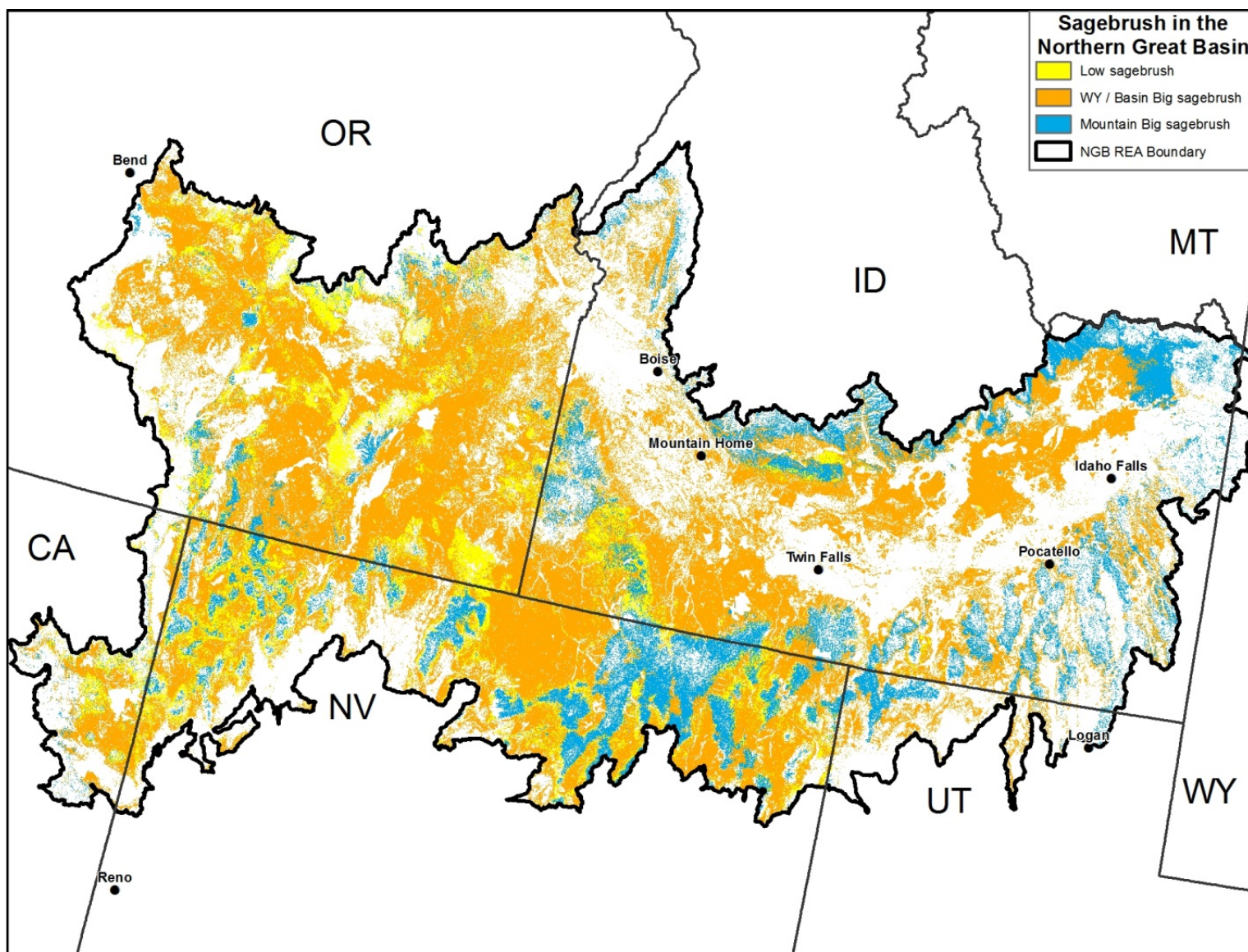
Figure 4-11. Sagebrush Distribution in the Northern Great Basin (from REA)
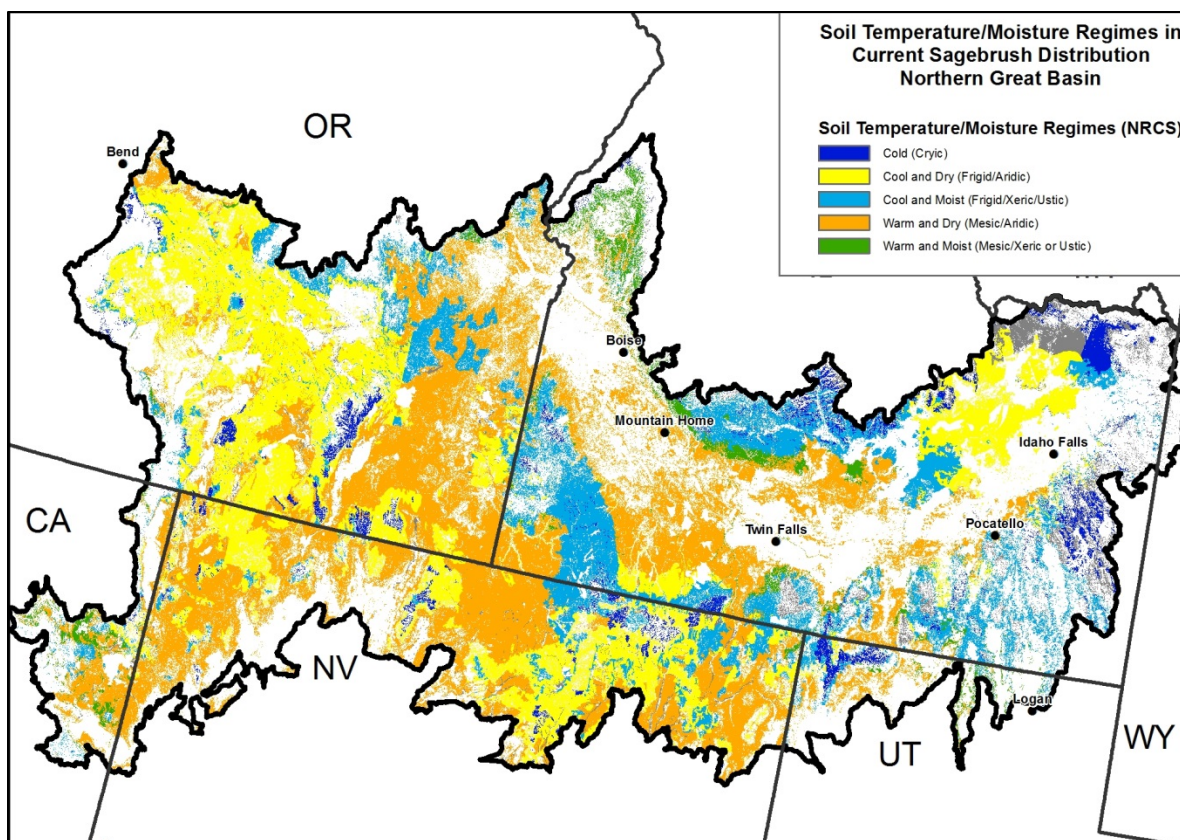
Figure 4-12. NRCS Temperature/Moisture Regimes in Current Sagebrush Distribution
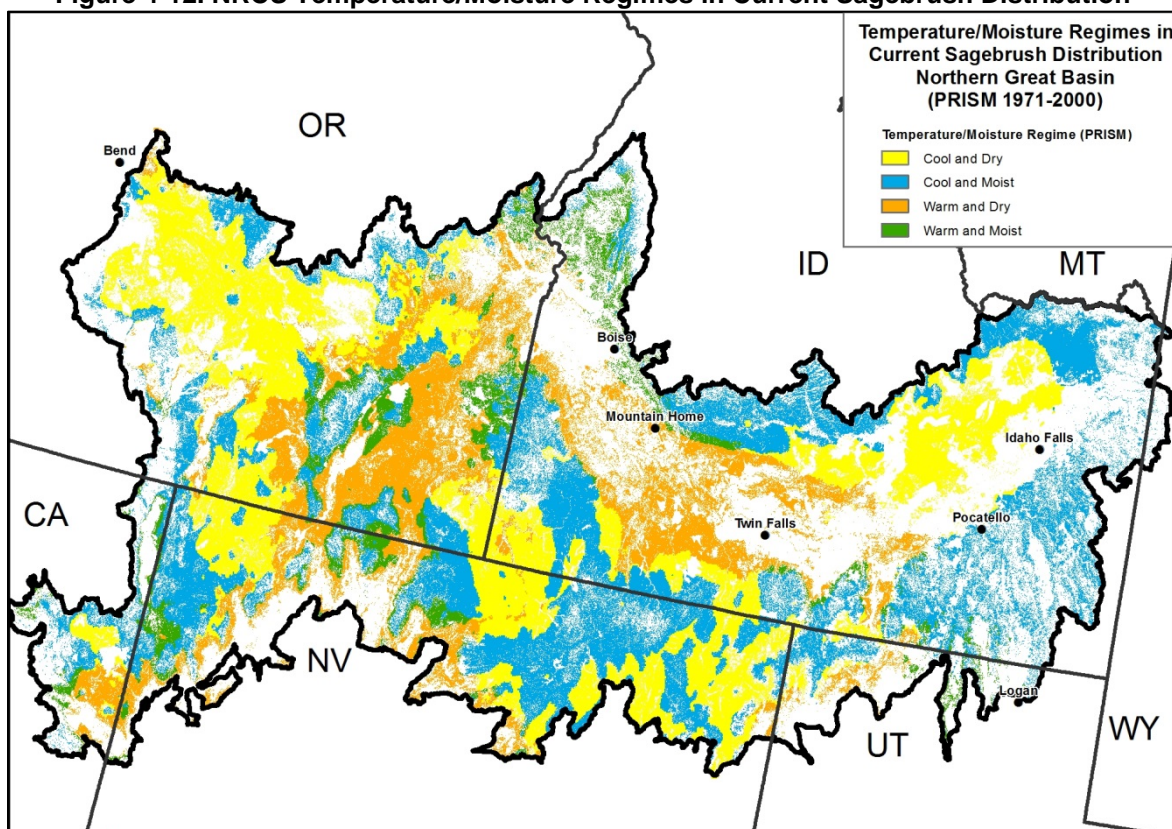
Figure 4-13. PRISM Temperature/Moisture Regimes in Current Sagebrush Distribution
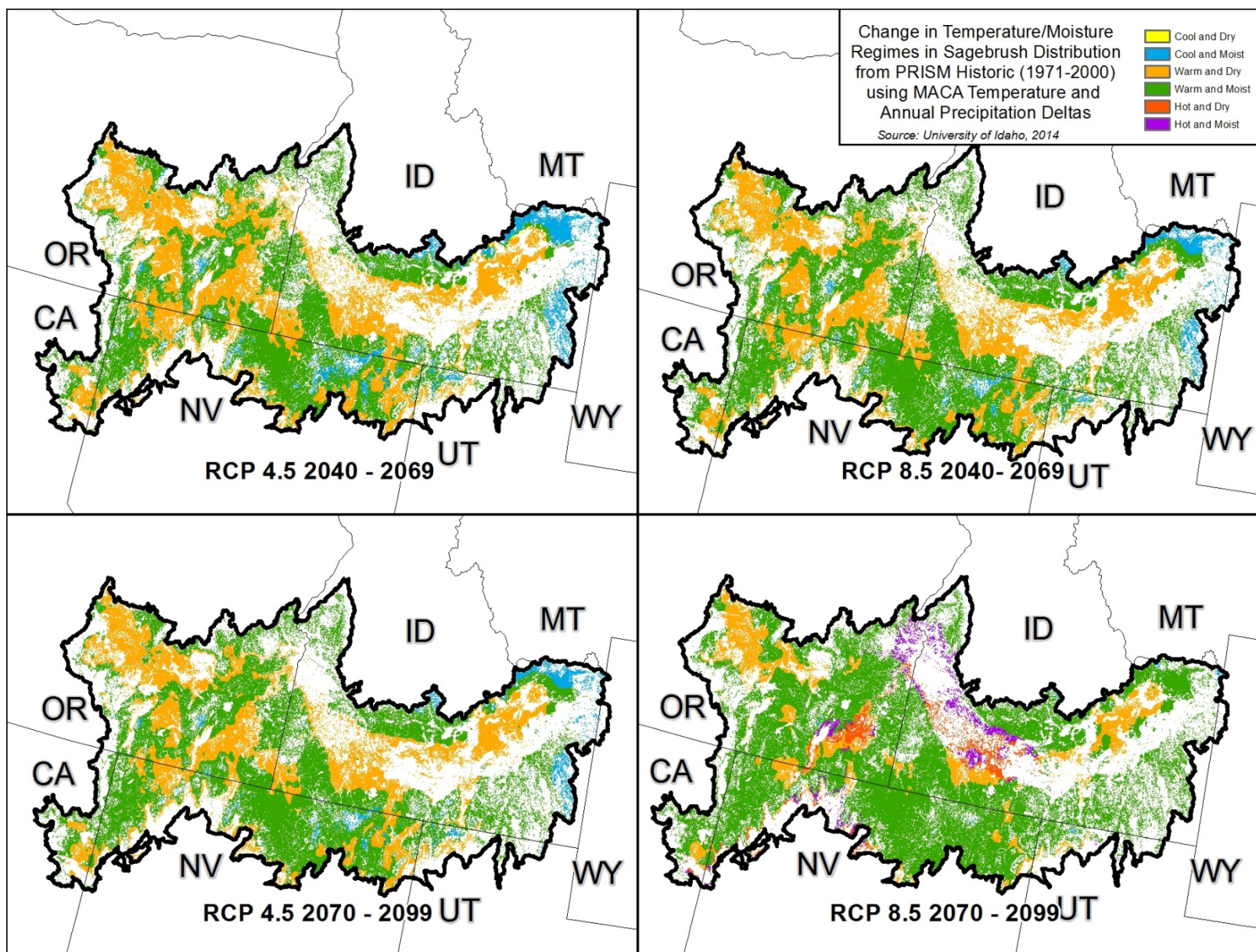
Figure 4-14. Projected Temperature/Moisture Regimes in Greater Sage-grouse Habitat
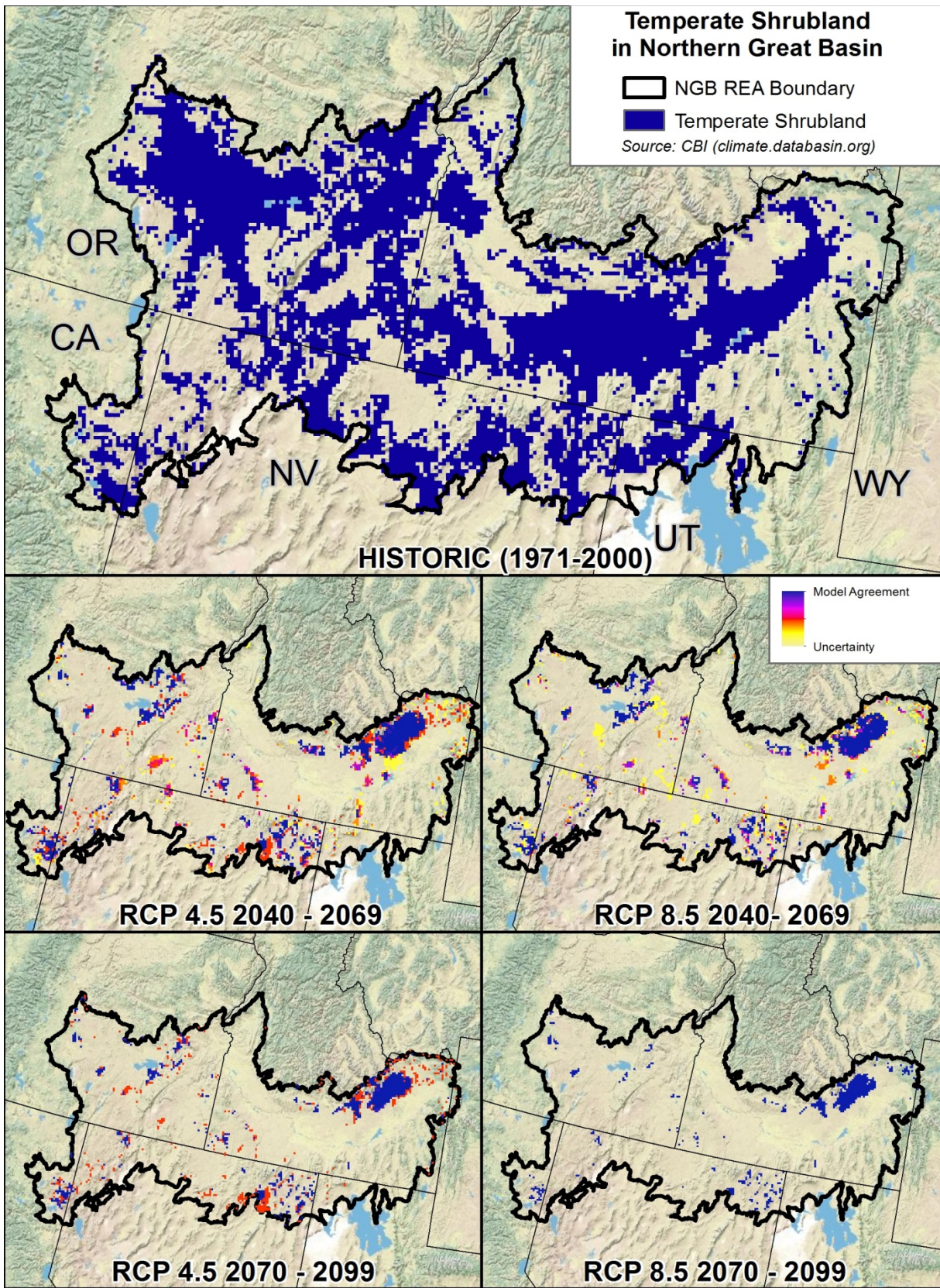
Figure 4-15. MC2 Temperate Shrubland Vegetation Distribution
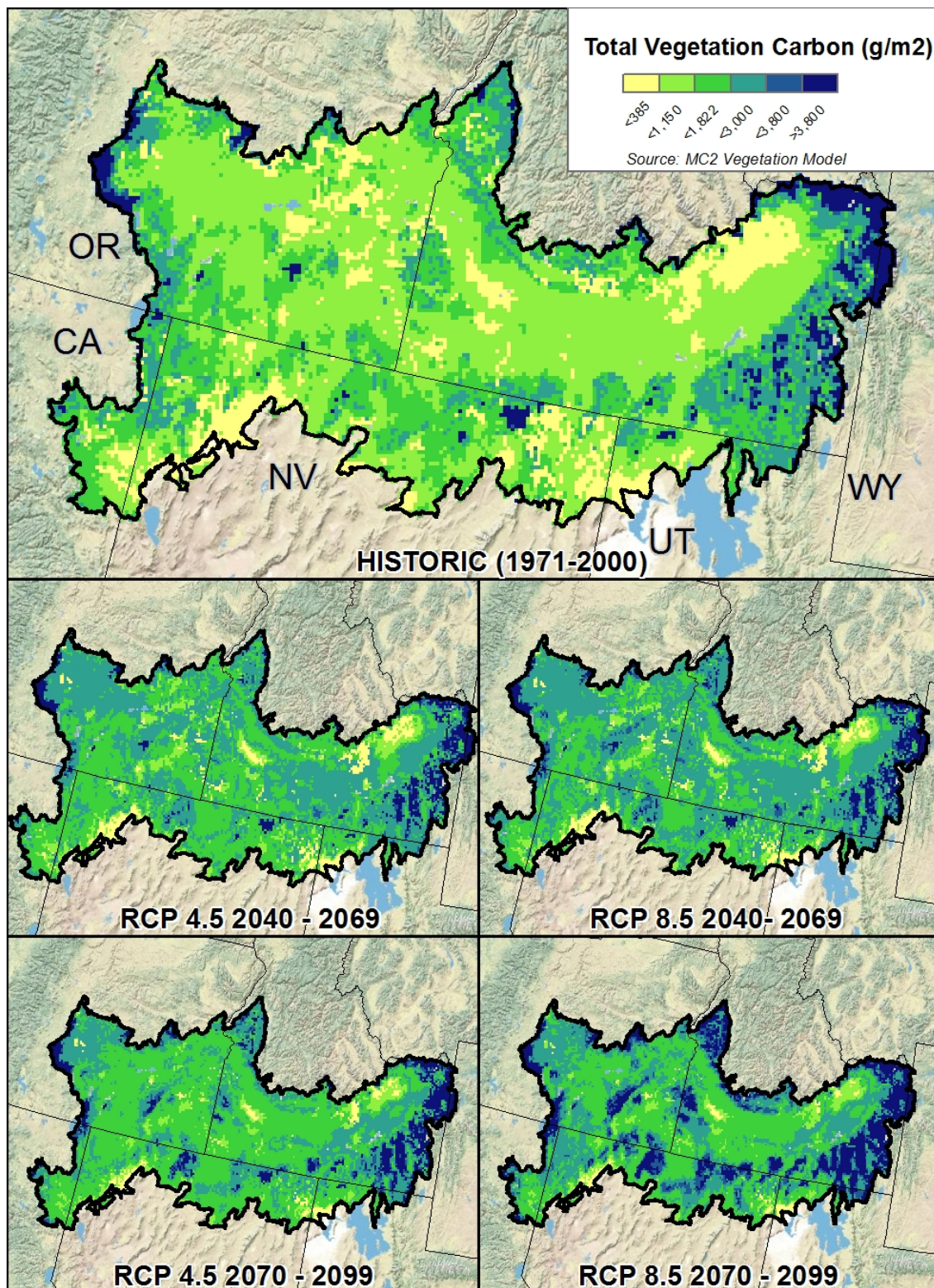
Figure 4-16. MC2 Total Vegetation Carbon Results

Higher total vegetation carbon would indicate increasing woody vegetation. The Creutzberg et al. (2015) study in eastern Oregon portion of the REA classified vegetation as woodland at greater than 3,000 g/m2. The shrubland distribution in Figure 4-15 is based on a 1,150 g/m2 threshold for woodland (base calibration for the MC2 modeling) which may be too low for the ecoregion.

4.2.5 Shrubs References

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4.3 Trees (Juniper and Other Conifer)

Conifers are an integral component of forest communities at higher elevations in the Northern Great Basin. Common dominant species are Douglas-fir (*Pseudotsuga menziesii*), lodgepole pine (*Pinus contorta*), ponderosa pine (*Pinus ponderosa*), Engelmann spruce (*Picea engelmannii*), western juniper (*Juniperus occidentalis*), and Utah juniper (*J. osteosperma*). These communities provide important wildlife habitat, forest products, and a myriad of other ecological services. Conifers serve as an important food source and habitat for various fauna, many of which are specially adapted for the high-elevation conditions. Conifer forests are the primary species used for timber harvest. Douglas-fir and other conifers have been the focus of additional attention because of their ability to rapidly colonize and establish in sagebrush and aspen communities. A combination of overgrazing, changes in microenvironment and climatic patterns, and fire suppression may contribute to conifer expansion into these communities. Conifer establishment is increasingly common and leads to new management decisions on existing stands and conifer expansion into sagebrush and aspen communities.

Western juniper and Utah juniper dominate large areas across the Intermountain Region, including the Northern Basin and Range Ecoregion. In the ecoregion, western juniper is prevalent in Oregon, northeastern California, northwestern Nevada, and southwestern Idaho. It is geographically replaced by Utah juniper to the south and east. Utah juniper has an extensive distribution in Nevada and Utah and is present in southeastern Idaho. Along the California-Nevada border, western juniper is represented by two subspecies: typical western juniper (subspecies *occidentalis*), which occurs as woodlands in sagebrush-steppe, and Sierra juniper (subspecies *australis*), which differs from subspecies *occidentalis* in being a large tree occurring in montane forested habitats at higher elevations. Sierra juniper occurs mostly south of the ecoregion. The ecological relationships of the Utah juniper and the typical western juniper are very similar.

Except for large individuals in relatively fire safe sites, both Utah and western junipers are killed outright by fire and do not resprout. For these species to regenerate, seeds must survive a fire or disperse back into a burned area. Establishment typically takes place under a shrub (Miller *et al.* 2005). All of this means that a certain amount of time must elapse to allow post-fire reestablishment of shrubs and seed dispersal of junipers to take place before western or Utah junipers can begin to reoccupy a burned site.

4.3.1 Trees Conceptual Models

4.3.1.1 Juniper Conceptual Model

A conceptual model of juniper ecosystems in the ecoregion is presented in Figure 4-17. It shows the general relationships among CAs including climate change, wildfire, livestock grazing, invasive species, land treatments, insects and disease, and restoration. Table 4-9 provides a summary of the CAs and their principal effects on other juniper ecosystems.

Table 4-9. Change Agents and their Principal Effects on Juniper Ecosystems

Change Agent	Principal Effects on Junipers
Climate change	A decrease in temperature and increase in precipitation favor juniper ecosystems. Wet, mild conditions promote vigorous growth in junipers and these climate patterns are thought to be responsible for the increase and spread of juniper establishment during the late 1800s and early 1900s (Miller *et al.* 2008; Miller and Rose 1995).
Wildfire	Fire is considered to be the most important factor in limiting juniper occurrence as well as encroachment into shrub-grassland communities. Except for large individuals in relatively fire

Change Agent	Principal Effects on Junipers
	safe sites, both Utah and western junipers are killed outright by fire and do not resprout. For these species to regenerate, seeds must survive a fire or disperse back into a burned area. Establishment typically takes place under a shrub (Miller *et al.* 2005). Post-fire recovery of native vegetation in higher elevation stands is more rapid and these stands are less susceptible to cheatgrass invasion and type conversion.
Livestock grazing	By reducing the fine fuels of perennial grasses, livestock grazing indirectly made wildfire less frequent, allowing juniper expansion (e.g., see Miller and Rose 1999; Miller *et al.* 2008). Juniper expansion has been documented in relict ungrazed areas as well as in grazed areas (Soule and Knapp 1999).
Invasive species	Low elevation sites into which junipers have expanded are vulnerable to cheatgrass invasion which can ultimately lead to a type conversion and soil loss.
Land treatments	Land treatment, including tree thinning and seeding with shrub and/or grasses, could influence the distribution of junipers. Junipers are often controlled in areas to reduce their expansion into shrub-grassland communities.
Insects and disease	Juniper trees are long-lived and resistant to most insects and diseases (Knapp and Soule 1999), although some species use junipers as a food source (Miller *et al.* 2005).
Restoration	Restoration of western and Utah juniper stands is addressed in detail in Miller *et al.* (2005), Miller *et al.* (2007), and Tausch *et al.* (2009), with special attention to analyzing the site and its potential. Because of their aesthetic value and value as wildlife habitat, old growth stands of junipers are considered valuable and receive management attention. Old-growth woodlands that provide valuable habitat for cavity-nesting birds should be maintained. Restoration of stands in which juniper density has increased to undesirable levels is possible, and entails thinning of trees (often using chain saws) and possible reseeding of suitable native grass and shrub species, if needed. Combinations of cutting and fire may also be effective. Fire alone can be effective in Phase I and early Phase II stands if cover of native grasses is sufficient to support regeneration of grass cover. However, use of fire to thin post-settlement junipers that have infilled within or adjacent to an old-growth stand would have to be implemented with caution. This is because the increased tree density within or adjacent to old growth stands creates the potential to fuel stand-replacing wildfire, killing the old growth trees.

4.3.1.2 Other Conifer Conceptual Model

A conceptual model of other conifer ecosystems in the ecoregion is presented in Figure 4-18. It shows the general relationships among CAs including climate change, wildfire, livestock grazing, invasive species, land treatments, and insects and disease. Table 4-10 provides a summary of the CAs and their principal effects on other conifer ecosystems.

Table 4-10. Change Agents and their Principal Effects on Other Conifer Ecosystems

Change Agent	Principal Effects on Other Conifers
Climate change	Climate change, in particular toward hotter and drier conditions, may alter the current distribution of coniferous forests and is thought to weaken the trees making them more vulnerable to insect attack.
Wildfire	Douglas-fir can survive low intensity surface fires but are killed by moderate to high intensity fires and must regenerate from seed. Douglas-fir is somewhat shade tolerant and can encroach into the understory of forested habitats (e.g., aspen or other conifers). In contrast, lodgepole

Change Agent	Principal Effects on Other Conifers
	pine and Engelmann spruce require open habitat for regeneration and regenerate from seed after a wildfire.
Livestock grazing	Livestock grazing can trample vegetation and indirectly make wildfire less frequent by reducing the fine fuels of perennial grasses, which could allow conifer expansion.
Invasive species	Fuel buildup in the understory is related to the cover of invasives, particularly knapweed (*Centaurea virgata*) and smooth brome (*Bromus inermis)* which can invade conifer communities when openings exist. Fire frequency and intensity influences the cover of invasives. For example, smooth brome has been controlled with repeated prescription burn treatments (Wilson and Stubbendieck 1997).
Land treatments	Timber harvest and windthrow events are also important CAs. Older trees are generally more vulnerable to windthrow events (Steil *et al.* 2009). Timber harvest may initially reduce conifer canopy cover; however, if harvest is conducted using sustainable methods tree canopy cover could increase. Wildlife and livestock browsing on resprouts, buds, needles, and cambium may retard growth and regeneration of conifers.
Insects and disease	Insects that infest conifers include Tussock moths (*Euproctis similis)*, western spruce budworms (*Choristoneura occidentalis*), and mountain pine beetles (*Dendroctonus ponderosae*). Diseases that infect trees are wood-rot basidiomycete fungus and white pine blister rust fungus. Bark beetle population sizes have rapidly increased in recent years. An increase in temperatures at high elevations is thought to be the variable that allows these insects to survive and reproduce at higher elevations than previously known to occur (Bentz 2008).
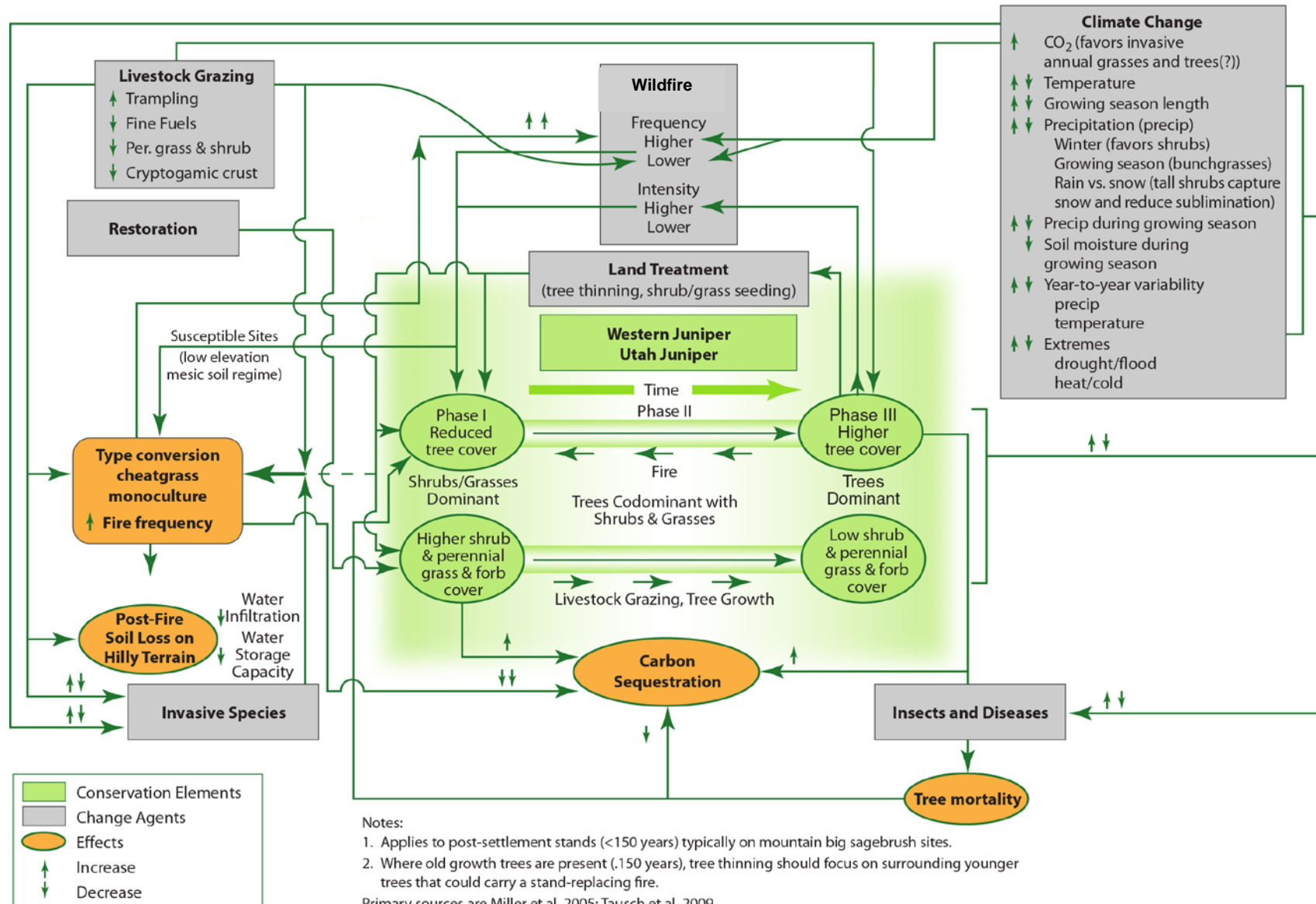
Figure 4-17. Western Juniper and Utah Juniper Conceptual Model

Other Conifers
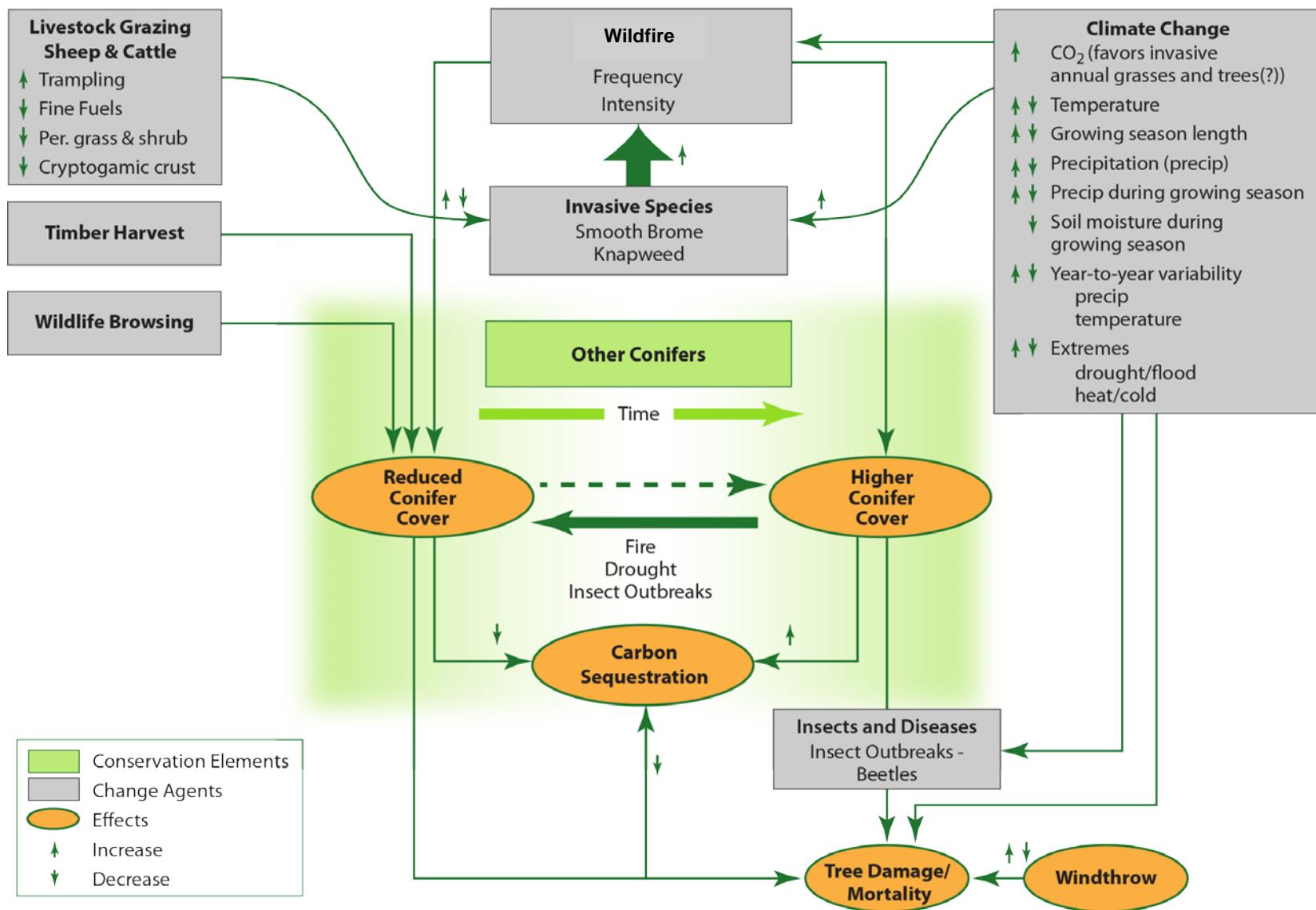
Figure 4-18. Other Conifer Conceptual Model

4.3.2 Climate Influences

4.3.2.1 Juniper Climate Influences

Juniper grows in a wide range of environments. Precipitation across most of the western juniper zone varies between 10 and 15 inches (Gedney *et al.* 1999), most of which falls during the winter and spring (October through June). Utah juniper generally is found in areas of 12 to 18 inches of precipitation (Zlatnik 1999). However, juniper can grow in areas receiving less than 10 inches or highs of 20 inches of precipitation annually (Zlatnik 1999). Western juniper occupies elevations ranging from 1,000 to 8,000 feet (ft), though, most western juniper woodlands and savannas are found between 2,000 and 6,000 ft (Gedney *et al.* 1999). Western juniper is usually not found above 7,000 feet because its foliage is damaged by extreme winter temperatures (Miller and Rose 1995). Similarly, Utah juniper occurs at 3,000 to 8,000 feet elevation (Zlatnik 1999).

Since the last major glacial period, the distribution and abundance of the western juniper has changed dynamically with shifting climate conditions (Miller *et al.* 2005). During the past 130 years Utah and western junipers have been expanding at unprecedented rates (Miller *et al.* 2008). The ability of junipers to expand into neighboring communities is thought to be caused by a variety of factors as well as their interactions, which include: climate influences during the past two centuries, decreased fire return intervals, livestock grazing, and increases in carbon dioxide concentrations in the atmosphere (Miller *et al.* 2005; Miller *et al.* 2008). Winters in 1850 to 1916 were milder and wetter than the current long-term average precipitation of the Great Basin region (Miller *et al.* 2005). Wet, mild conditions promote vigorous growth in western juniper (Miller *et al.* 2005). The wet and mild recruitment conditions coupled with the introduction of livestock reduced fuel loads and the fire return interval allowing western junipers to expand. Fire is the most important factor which limits conifer encroachment into shrub or grassland communities (Miller *et al.* 2005). The probability that western juniper will establish and successfully mature increases when the fire return interval is 70 years or more. Typical fire return intervals for mountain big sagebrush and Idaho fescue range from 6 to 17 years (Miller *et al.* 2005).

Specifically in the ecoregion, in the Reynolds Creek Experimental watershed in southwestern Idaho, Sankey *et al.* (2010) demonstrated that the aerial extent of western juniper has increased 85 percent within the area examined since 1965, indicating an encroachment rate of approximately 2 percent per year. An examination of seven study locations of junipers (both Utah and western) in the Intermountain West found that over 90 percent of the junipers established post-Eurasian settlement after the 1860s. Under current climatic conditions, conifers are likely to continue expanding into shrub-steppe plant communities (Miller *et al.* 2008).

Increased CO_2 concentrations in the atmosphere increases the productivity of forests (Norby *et al.* 2005). A study of four intact forests sites which increased free atmosphere CO_2 found a median increase of net primary productivity of 23 percent under 550 ppm (Norby *et al.* 2005). This CO_2 fertilization effect has also been observed in western juniper. An analysis of juniper growth rings in the early 20th century and the later part of the 20th century found that overall growth increased by 23 percent in the later part of the century where CO_2 concentrations were higher (Knapp et al. 2001). Higher atmospheric CO_2 concentrations are hypothesized to favor trees over grasslands and contribute to the "woody thickening" observed in savannas worldwide (Bragg *et al.* 2013). In arid regions, rising CO_2 concentration is likely to be greater because stomata close more often to conserve water so an increase in CO_2 has a larger effect (Bragg *et al.* 2013). However, follow-up studies have found that the increase in net primary productivity eventually declines as tree stands become limited by available nitrogen (Norby *et al.* 2010). The CO_2 fertilization effects also affect plants species differently with C3 plants (most trees) benefiting more than C4 plants (Bragg *et al.* 2013).

The shift to juniper dominance reduces the biomass of vegetation in the understory which provides soil cover. This increased bare ground can result in increased soil erosion (Kuhn *et al.* 2007). Pierson *et al.* (2007) found that juniper-dominated hillslopes produced significantly more soil surface runoff and erosion than hillslopes with juniper removed in southeast Oregon. One possible consequence of increased juniper dominance is that a greater portion of precipitation falling in these arid subbasins is used by juniper trees, resulting in reduced stream flows and groundwater recharge (Kuhn *et al.* 2007). However, in arid west regions, any soil water made available by woody plant removal is often stored in the soil and then directly evaporated.

4.3.2.2   Other Conifer Climate Influences

Climate change is projected to result in rising temperatures and changes in the precipitation cycle, including less snow and more rain, less snowpack, earlier spring snowmelt, lower summer streamflows, and longer summer dry periods (Scott *et al.* 2013). Water is the major limiting factor on forests in the region as most of the precipitation occurs outside of the growing season. With most forest currently experiencing seasonal water limitations, increases in the summer water deficit could result in decreased seedling regeneration and tree growth and increased mortality due to insect outbreaks and wildfires (Littell *et al.* 2013). The actual effects on montane forests will be driven by how these broad-scale climate changes affect the local microclimate conditions. Montane forest structure and composition depend on the water requirements of individual species, water holding capacity of the local soil, the aspect and slope which controls solar energy inputs as well as complicating factors caused by increased intensity of disturbances resulting from more arid conditions (e.g. wildfire and insect outbreaks) (Scott *et al.* 2013).

Ponderosa pine occurs on drier and more nutrient deficient sites when compared to other montane conifers (Scott *et al.* 2013). Ponderosa pine tolerates fairly warm temperatures as long as there is enough moisture during the growing season. Summer drought, in combination with future projected higher temperatures, is likely to reduce ponderosa pine regeneration in the drier, lower elevation areas.

Douglas-fir is one of the most common tree species and is of high commercial importance in North America. Compared to other tree species, Douglas-fir can thrive in a variety of habitats and climate conditions (Weiskittel *et al.* 2012). Climate factors controlling Douglas-fir include total and growing season precipitation, growing days (days above 5 °C), and the mean temperature of the coldest month (Rehfeldt *et al.* 2006).

Lodgepole pine is adapted to heterogeneous forest landscapes in the mid to high elevations. It is relatively resistant to frost injury and can often survive in frost pockets where other species cannot. It is generally found on moist soils and thus grows well on slopes with northern and eastern aspects. It has a high ability to regenerate due to cone serotiny, high seed viability, early rapid growth, and ability to survive in high soil moisture conditions. Engelmann spruce occupies the highest and coldest environments in the western U.S. It has a low tolerance for high temperatures and drought, especially during the first five years of establishment.

4.3.2.3   Modeling Climate Influences on Juniper and Other Conifer Ecosystems

Observed relationships between climate and plant response have been used to develop models to project future species and ecosystem responses to climate change (e.g., McKenney *et al.* 2007; Rehfedt *et al.* 2006). In general, model simulations indicate large potential changes in the climatic suitability for some plant species and habitats in the Northwest. Statistical models of tree species-climate relationships show that due to unique climate tolerances of individual species, the species range shifts are expected to be species specific rather than collections of currently associated species shifting as a community (Rehfeldt *et al.* 2006; McKenney *et al.* 2007). Some species have limited dispersal ability or require a narrow

biophysical niche. These sensitive species may be lost from existing communities if changes to the climate occur faster than a species ability to disperse within its suitable climate (Littell *et al.* 2010; Scott *et al.* 2013). Within the ecoregion, four key tree species (ponderosa pine, Douglas-fir, lodgepole pine, and Engelmann spruce) are discussed in further detail below.

Rehfeldt *et al.* (2006) were able to predict the distribution of ponderosa pine through the use of summer, winter precipitation, summer temperatures and growing degree days greater than 5 °C. With increasing temperatures, ponderosa pine may expand its range into areas currently dominated by Douglas-fir (Scott *et al.* 2013). The distribution of ponderosa pine at lower elevation, drier areas is related to the available soil moisture which is controlled by the local soil texture and depth. Widespread recruitment of ponderosa pine occurs in conditions where there is sufficient moisture and fewer fires. Such conditions may become less likely under future climate scenarios (Scott *et al.* 2013).

Modeling by Rehfeldt *et al.* (2006) projected that the area occupied by Douglas-fir across North America would change little (-2 to 11 percent) but it would become displaced in space. In the ecoregion, at higher elevations and with warmer temperatures, it may expand on to sites that are dominated by cold, hardy lodgepole pine, but it will be limited by growing season frosts (Scott *et al.* 2013). At lower elevation and drier sites it may be replaced by ponderosa pine (Scott *et al.* 2013).

Lodgepole pine will retract from the dry sites where Douglas-fir may be favored with increased average temperatures and where cold does not limit Douglas-fir establishment (Scott *et al.* 2013). With increased temperatures, spruce may expand into areas that are currently limited by cold; however, they may be replaced at lower elevations by trees that are better adapted to drier conditions (e.g. lodgepole pines or Douglas-fir) (Scott *et al.* 2013). Subalpine forests are projected to substantially decrease in areas as temperatures increase in most of the Northwest (Littell *et al.* 2013). Extreme events like droughts, heat waves, frosts, and windfalls could also affect forest systems. Subapline forests are especially susceptible to droughts. Increased drought can result in extensive tree mortality and shifts in tree species. Fir are most susceptible to drought, followed by spruce, and then pines (Bigler *et al.* 2007).

Climate change is likely to substantially affect the distribution, growth, and functioning of the forests of the ecoregion. The spatial distribution of suitable climate for many important tree species and vegetation types may change considerably by the end of the 21st century, and some vegetation types, such as subalpine forests, will become extremely limited compared to their current distribution (Littell *et al.* 2013).

To better understand changes associated with specific species (e.g. sage-grouse), species distribution models have been incorporated into MC2 and state and transition models, or the "mega model" (Creutzburg et al. 2015; Miller *et al.* 2015; Peterman 2014). The distribution and expansion of junipers was examined during this process. Results from this "mega model" predicted that juniper invasion is much less variable among the different climate scenarios and that the type of management has more impacts on juniper occurrence (Creutzburg *et al.* 2015). In addition, the current landscape composition presents a high risk for expansion of juniper and climate change is projected to reduce juniper relative to current climate.

4.3.3 Geoprocessing Modeling and Analysis

As discussed in Section 4.3.2, the previous REA presented the climate envelope or species distribution modeling for key conifer tree species in the ecoregion. For the climate change supplement, the MC2 DGVM results showing the historic 1971–2000 vegetation types are compared to the modeled vegetation under the RCP 4.5 and 8.5 scenarios for the 2036–2065 and 2071–2100 time slices. These vegetation models are based on the CMIP5 MACA downscaled datasets. The MC2 model simulates the mixture of

tree lifeform (evergreen needleleaf, evergreen broadleaf, and deciduous broadleaf trees), as well as grass life form (C3 and C4) and shrublands likely to occur based on the results of the climate model. The MC2 vegetation model and its vegetation types are fairly generalized. As discussed in Section 4.2.3, the categorization of forest, woodland, and shrublands is based on modeled vegetation carbon.

The other tree and juniper CEs generally translate to the MC2 vegetation classes shown in Table 4-11. The MC2 vegetation class is shown in a composite map of all 12 available modeling scenarios. Each pixel is given a value based on the number of models that predict the occurrence of that particular vegetation class in that pixel.

Table 4-11. Conservation Elements and MC2 Vegetation Classes

Conservation Element	MC2 Vegetation Class	Common Species
Other Conifer	Subalpine Forest	Engelmann spruce (*Picea engelmannii*), subalpine fir (*Abies lasiocarpa*), and lodgepole pine (*Pinus contorta*)
	Cool Needleleaf Forest	Douglas-fir (*Pseudotsuga menziesii*), ponderosa pine (*Pinus ponderosa*)
	Temperate Evergreen Needleleaf Forest	Ponderosa pine, lodgepole pine, Douglas-fir
Juniper	Temperate Evergreen Needleleaf Woodland	Western juniper (*Juniperus occidentalis*), mountain big sagebrush (*Artemisia tridentata* spp. *vaseyana*)

Source: Adapted from Halofsky *et al.* 2013

4.3.4 Results

The USFS Moscow Forestry Sciences Laboratory developed climate models for various tree species modeling the effects of climate change in various future scenarios. These results were presented in the REA. Figure 4-19 shows the current and 2060 viability for Utah and western juniper within the Northern Great Basin based on the modeling done by the USFS (Crookson *et al.* 2010).

The MC2 DGVM results are presented for the historic 1971–2000 temperate evergreen needleleaf woodland type and compared to the predicted temperate evergreen needleleaf woodland type under the RCP 4.5 and 8.5 scenarios for the 2036–2065 and 2071–2100 time slices (Figure 4-20). Similarly, the results for the forest vegetation types (other conifer) are presented in Figure 4-21. The MC2 modeling predicts a significant woodland expansion due to climate change. The woodland is modeled to expand and replace much of the shrubland in the ecoregion. This expansion is based on the increased CO_2 concentrations which would increase the net primary productivity in the ecoregion resulting in more carbon in the vegetation and more success of woody species. The extensive woodland expansion is not consistent with the species distribution modeling presented in the previous REA for western and Utah juniper. The species distribution modeling does not factor in the effects of CO_2 fertilization. Generally, these results should be interpreted carefully. Experiments have found that the increase in net primary productivity caused by CO_2 fertilization eventually declines as tree stands become limited by available nitrogen (Norby *et al.* 2010). However, the MC2 model assumes an increase in productivity CO_2 fertilization at a constant rate through time. Overall, the species distribution model may underestimate the range for tree species because it does not include increased productivity and water efficiency from elevated CO_2 concentrations. The MC2 dynamic vegetation modeling likely overestimates the increase in range of tree species by not including a nitrogen-based limitation to the increase in productivity expected from the elevated CO_2 concentrations.
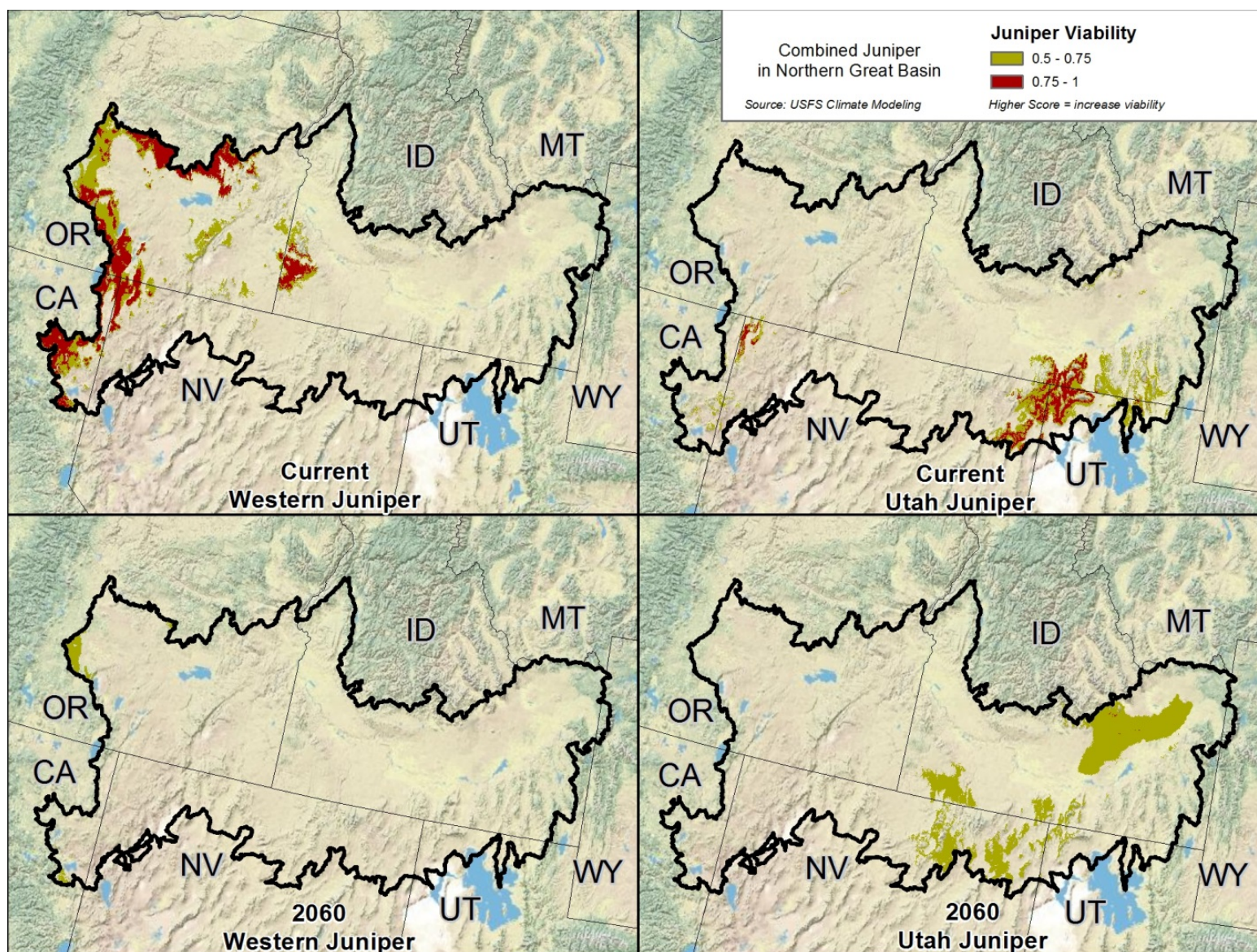
Figure 4-19. Projected Juniper Viability based on USFS Modeling
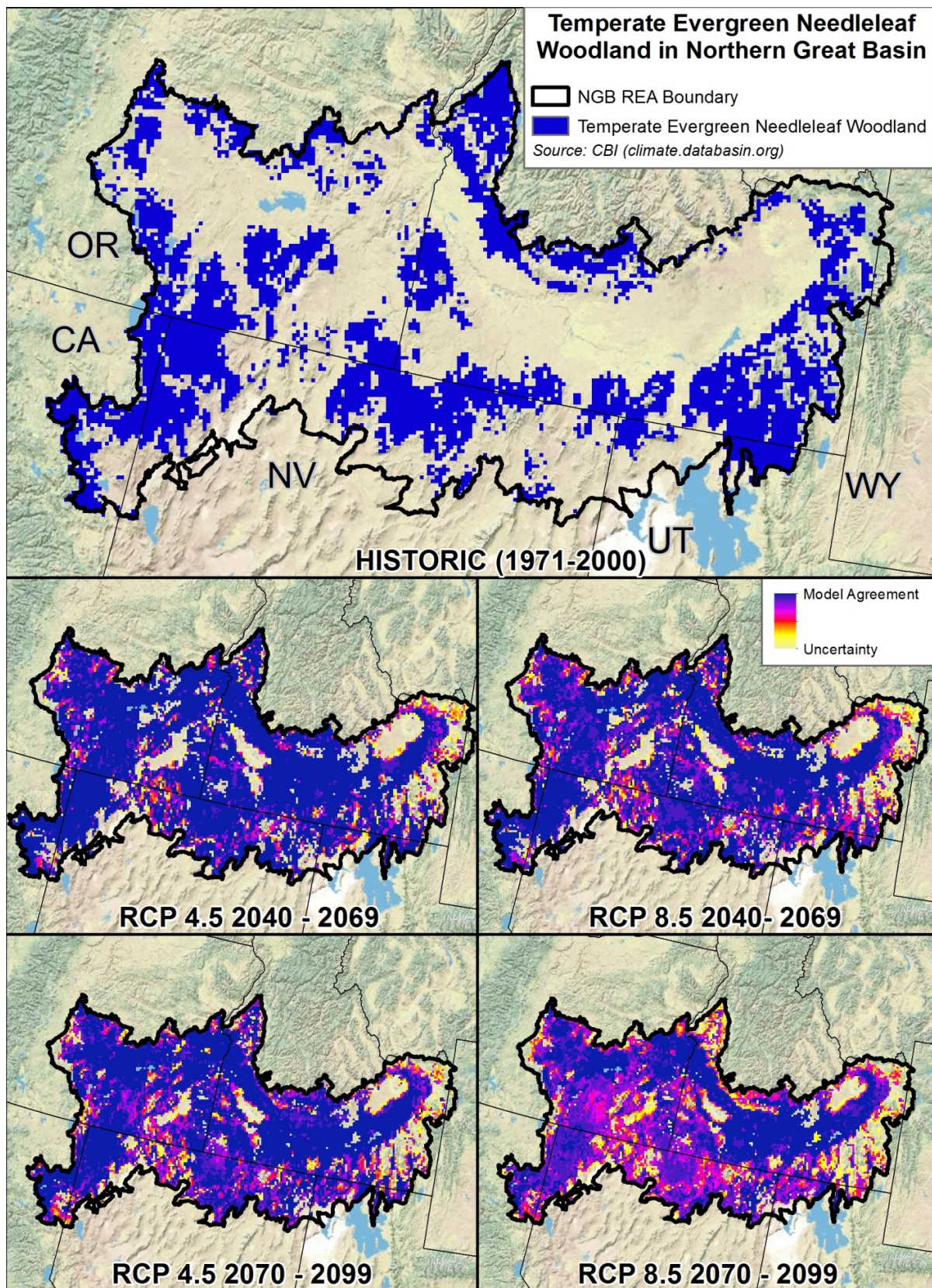
Figure 4-20. MC2 Temperate Evergreen Needleleaf Woodland Vegetation Distribution
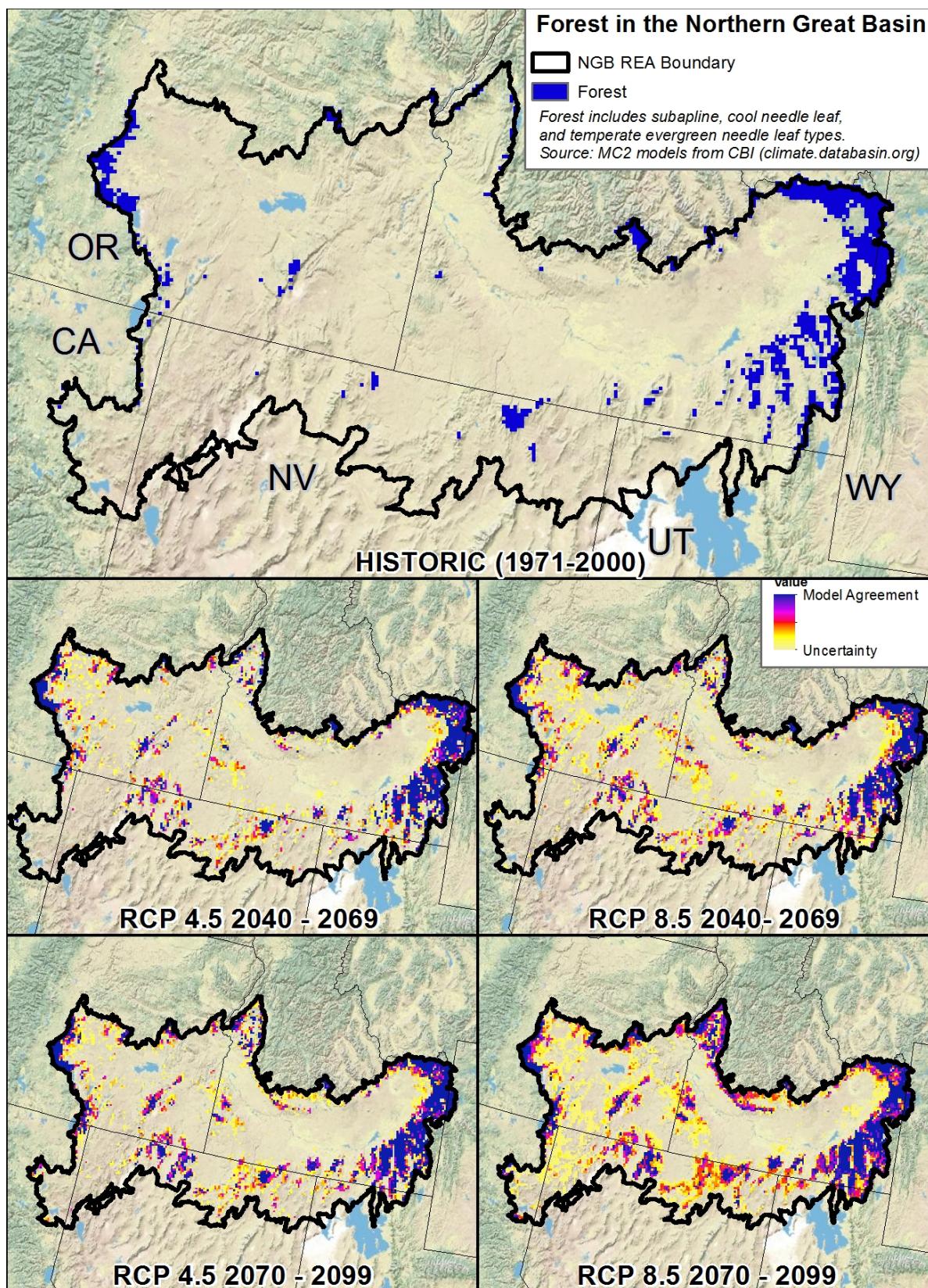
Figure 4-21. MC2 Forest Distribution

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4.4 Aspen

Quaking aspen (*Populus tremuloides*; hereafter referred to as aspen) is one of the most widely distributed tree species in the U.S., found from Canada to central Mexico. Across the intermountain region of western North America, aspen forests provide unique, biologically diverse communities (Strand *et al.* 2009). Aspens grow in distinct clones, and this system includes upland forests and woodlands dominated by aspens that is sometimes mixed with conifers (<25 percent relative tree cover). Aspen usually occurs within a mosaic of many plant associations and may be surrounded by a diverse array of other communities, including coniferous forests, grasslands, wetlands, and meadows. The understory structure ranges from complex, with multiple shrub, forb, and herbaceous layers to simple, dominated by grasses (Faber-Langendoen et al.2011). In the western U.S., aspens commonly occur as a disturbance-dependent species, seral to conifer species (Strand *et al.* 2009). In mixed aspen/conifer stands, periodic fires are sometimes necessary to prevent conifer dominance and possible loss of the aspen stand (DeByle *et al.* 1987).

Aspen stands provide habitat for a diversity of species and are one of the few broad-leaved trees that can grow at high elevations. Elevations generally range from 1,493 to 2,743 meters (4,900 to 9,000 feet), but occurrences can be found at lower elevations in some regions on cooler, north aspects and mesic sites. Aspen stands can occur on gentle to moderate slopes, in swales, or on level sites. Soils are usually deep and well developed with rock often absent and textures that range from sandy loam to clay loam. Aspens prefer a temperate climate with a relatively long growing season, typically cold winters, and deep snow. Aspens grow in large clonal colonies that form from a single seedling and spread by root suckers. The occurrence of aspen is primarily limited by soil moisture (adequate soil moisture is required to meet high evapotranspirative demand), length of growing season, and temperatures. Mean annual precipitation where aspens occur is generally greater than 38 centimeters (15 inches) and typically greater than 51 centimeters (20 inches), except in semi-arid environments where occurrences are restricted to mesic microsites such as seeps or areas below large snow drifts.

An increasing decline of aspen populations was first noticed in the late 1990's in Utah (Huang and Anderegg 2012). In 2004, widespread branch dieback and mortality of whole portions of aspen stands occurred across Utah, Wyoming, Arizona, and Colorado landscapes (Nijhuis 2008). This scale of aspen mortality had never been seen before and the phenomenon was named "sudden aspen decline," or SAD (Frey *et al.* 2004). Increased declines of aspen stands have resulted in the need for aspen management and restoration efforts.

4.4.1 Aspen Conceptual Model

A conceptual model of aspen ecosystems in the ecoregion is presented in Figure 4-22. CAs include climate change, wildfire, livestock grazing and wildlife browsing, invasive species, and insects and disease. These CAs are intertwined in complex feedback loops that are often correlated and have an effect on each other. CAs of greatest importance to this CE are climate change, insects and disease, and wildfire. Table 4-12 provides a summary of the CAs and their principal effects on aspen ecosystems. Change agent effects can vary geographically as well depending upon the stability of aspen stands. Occurrences of the aspen system originate with and are maintained by stand-replacing disturbances, such as avalanches, crown fire, insect outbreak, disease and windthrow, or clearcutting by man or beaver, within the matrix of conifer forests, with which they compete for space and resources (Faber-Langendoen et al. 2011; Strand *et al.* 2009a, 2009b).

Table 4-12. Change Agents and their Principal Effects on Aspen Ecosystems

Change Agent	Principal Effects on Junipers
Climate change	Climate change, in particular hot and dry conditions, is thought to weaken the trees, making them more vulnerable to insect attack and disease and at risk for Sudden Aspen Decline (see insects and disease below).
Wildfire	Aspen has been widely regarded as a fire-adapted species and typically regenerates asexually and prolifically after fire (e.g. Yang *et al.* 2015; Shinneman *et al.* 2013). However, new studies have found more diverse responses of aspen to fire, depending on various aspen functional types (e.g., seral aspen and stable aspen) and topoclimatic conditions (Shinneman *et al.* 2013). Fire can promote increased aspen coverage through the mortality of competing species while stimulating aspen re-sprouting or opening space for aspen colonization.
Livestock grazing and wildlife browsing	Aspens are a highly preferred forage species for domestic cattle, deer, and elk. Feeding on bark is done by a variety of small and large mammals. Big game will often damage saplings and larger trees by rubbing their antlers against the stems. Wounding the stem bark predisposes trees to attachment or entry by damaging diseases and insects. According to the Idaho Department of Lands, aspen decline in southern Idaho is likely caused by a combination of factors including increased conifer encroachment due to fire suppression, insects and disease, and heavy ungulate grazing on regenerating aspen trees.
Invasive species	There is little information available on invasive species that pose a direct threat to aspens. However, nonnative species such as Kentucky bluegrass (*Poa pratensis*), quackgrass (*Agropyron repens*), bull thistle (*Circium vulgare*), and houndstongue (*Cynoglossum officinale*) have been recorded in aspen communities. Aspen stands are usually seral to conifers and often intermixed within coniferous forests, with some stands being faced with conifer encroachment. In Nevada aspen stands are relatively stable and generally are not faced with conifer encroachment. In contrast in Utah, aspen are seral and undergo succession toward a conifer canopy (Guyon and Hoffman 2011).
Insects and disease	In recent years, many aspen stands have exhibited declines from the effects of several CAs and mortality from Sudden Aspen Decline and biotic vectors. Bronze poplar borer larva are known to weaken the trees and make the trees more susceptible to fungal infections, and bark beetles cut off the tree's nutrient supply (Nijhuis 2008). Defoliating insects that cause damage include leaf-feeding larvae of Lepidoptera. Pathogens primarily infect aspen clones that are already stressed by drought, insects, wind damage, heavy livestock, and wildlife use. Natural disturbances such as wildfires or disease usually prompt aspen clones to send up numerous fresh sprouts, but new growth is rare in Sudden Aspen Decline-affected stands. Studies suggest that several interacting factors are contributing to causes of Sudden Aspen Decline. Site-related effects (low elevations, south and southwest aspects, open stands), higher temperatures, drought stress (Hogg *et al.* 2008; Rehfeldt *et al.* 2008; Worrall *et al.* 2008; Fairweather *et al.* 2008; Worrall *et al.* 2010), and climate change influence the occurrence and health of aspen stands.

Aspen

Figure 4-22. Aspen Conceptual Model
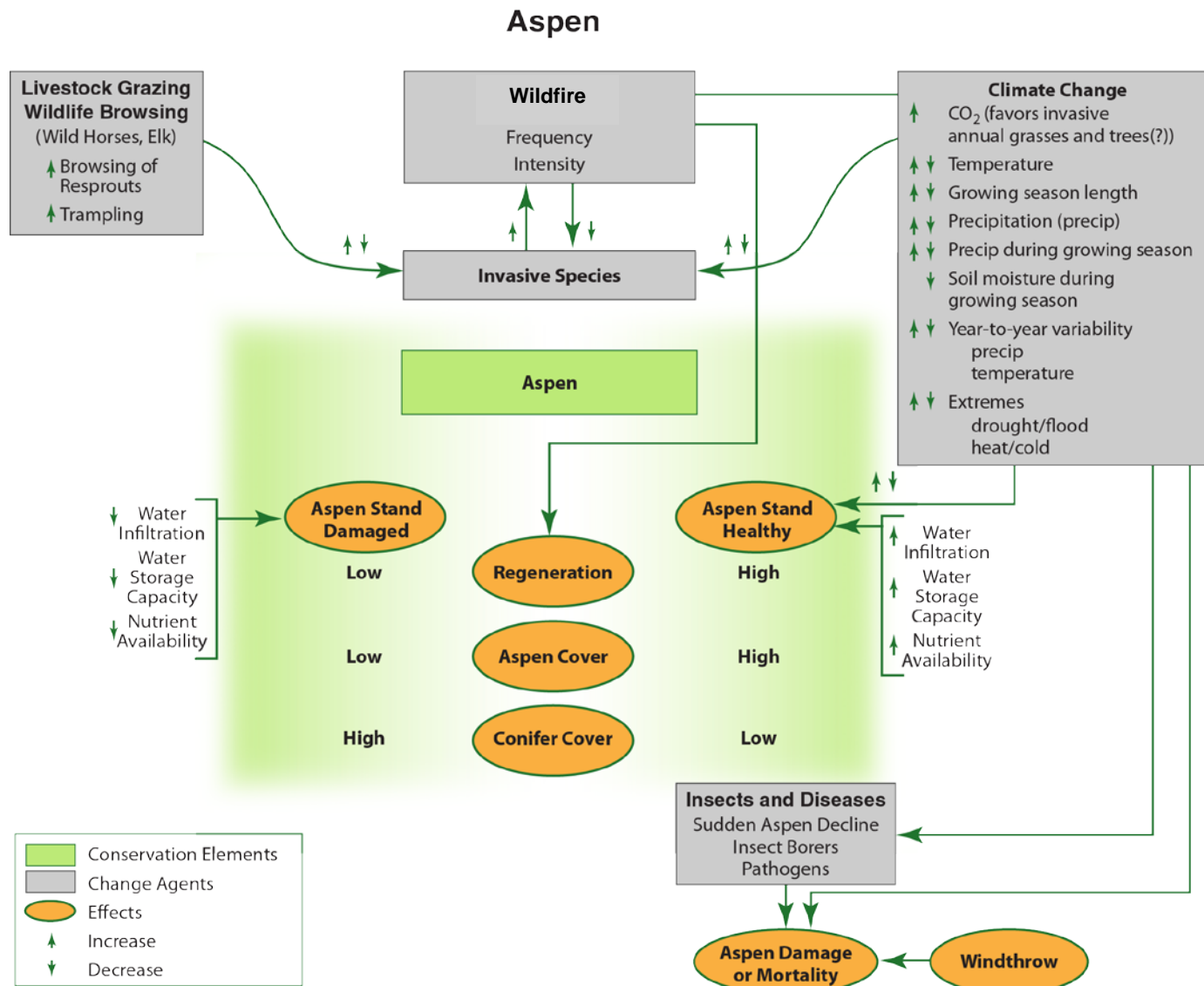
Aspen health: Healthy = Full aspen crowns with little to no die-off (<25% overstory mortality, <25% conifer cover);
Damaged = Dead or dying aspen stands with considerable to full overstory die-off and/or foliage loss (25-100% overstory mortality, <25% conifer cover).
(*Source:* Oukrop et al. 2011).

4.4.2 Climate Influences

Aspen trees in western North America typically occur where there is a water surplus (annual precipitation exceeds potential evapotranspiration). Aspen tolerate extremely cold air temperatures. However, cold soil temperature that is less than 6 °C appears to stress aspen trees, inhibiting root growth and water intake (Morelli and Carr 2011). Aspens also function poorly in hot and dry conditions, especially if temperatures are greater than 25 °C.

Climate change could lead to higher temperatures and an increase in moisture stress which could advance aspen mortality and decrease regeneration in western North America. Since aspen is limited by the amount of water available, severe droughts have been observed to cause the death and decline of aspen (Rehfeldt *et al.* 2009). In the northern Rocky Mountains, aspen dieback was noticeable two years following the extremely dry year in 2002 and one year following the dry period in 2000 to 2003 (Rehfeldt *et al.* 2009). Recent episodes of decline, from 2000 to 2010, suggest local shifts of the aspen climate niche could be responsible. In the west, declines generally occurred at the margins of the aspen geographic distribution at the edge of the climate niche (Worral *et al.* 2013).

The Intermountain Region (USDA Forest Service R4), which generally includes the Northern Great Basin, experienced a significant drought from 1999 to 2004, immediately prior to an episode of aspen dieback (Guyon and Hoffman 2011). Following this, surveys reported different patterns of aspen mortality caused by a variety of damage agents (e.g., animals foraging on resprouts or seedlings, insects, diseases) and varying susceptibility of different stem sizes (Guyon and Hoffman 2011). Some stands experiencing dieback were still capable of regenerating although recruitment was below the threshold suggested for successful aspen sustainment (O'Brien et al. 2010). Drought has been known to cause the loss of seral aspen stands and contribute to a decline in aspen regeneration and may continue to be an important factor in future mortality (Steed and Kearns 2010). The physiological mechanisms of how drought induces Sudden Aspen Decline are currently being investigated (Huang and Anderegg 2012). With warmer temperatures, droughts are expected to be more severe due to climate change.

4.4.2.1 Modeling Climate Influences on Aspen Distribution

The paleo-distribution of aspen, and its change with climate, could help anticipate future changes. Unfortunately, aspen pollen is poorly preserved, so past distributions are poorly understood (Worrall *et al.* 2013). A variety of climate models have been used to examine aspen distribution and predict future shifts of the species. The USFS Moscow Forestry Sciences Laboratory developed climate models to examine the distribution of aspen and effects of climate change for various future scenarios. Although there is substantial uncertainty in individual projections, modeled climate change shows the potential for dramatic reduction in aspen viability by 2030 (see Figure 4-23) and further reduction throughout the ecoregion by 2060 (Figure 4-24). The USFS Moscow Forestry Sciences Laboratory found that the annual dryness index (a ratio of growing-degree-days to annual precipitation) was the best predictor for aspen distribution. Aspen range appears to be limited mostly by moisture stress (Rehfeldt *et al.* 2009).

Worral *et al.* (2013) developed a bioclimate model of aspen across all of North America that projected losses of aspen habitat in the south (U.S. and Mexico) and gains in habitat in the north (Canada). Overall, through the century (2000-2100) in North America, the suitable climate for aspen is projected to remain constant, with the gains balancing the losses. However, many of the existing aspen stands in the ecoregion are at the edge of their climatic niche. Losses that have occurred in northeastern Nevada, southeastern Idaho, and northeastern Utah are expected to continue in the ecoregion while the suitable climates for aspen shift north and higher in elevation (Worrall *et al.* 2013). Bioclimatic models determined that the most important variables in predicting aspen occurrence was mean maximum temperature in the warmest month and summer precipitation from April to September (Worrall *et al.* 2013).

Collectively, these bioclimate studies suggest that the critical factor for aspen persistence is adequate summer soil moisture. Factors that could reduce summer moisture in the ecoregion's predominantly winter precipitation regime include decreasing overall precipitation, early melting of snowpack (e.g., due to spring rains at elevations where aspens occur at higher elevations), and increased evapotranspiration during summer (e.g., as a result of higher temperatures). Any factor that contributes to water stress during aspens growing season, especially when stress is prolonged over numerous years, would lead to increased susceptibility to insects and disease and could result in aspen declines. Bioclimate models have not incorporated key biological and disturbance processes that could affect aspen and its realized niche on the landscape-scale (e.g. Crookson *et al.* 2010; Worral *et al.* 2013). Dynamic forest landscape models have been developed to address succession-disturbance interactions.

In southwestern Idaho, Stand *et al.* (2009b) developed a conceptual state-and-transition model for upland aspen/conifer stands occurring across a range of topographic positions. The model was parameterized using field data in the vegetation dynamics computer simulation model Vegetation Dynamics Development Tool (VDDT), which examined the current and future aspen distribution under varying fire regimes. Model results indicate that average fire return intervals of 50 to 70 years are desirable for maintenance of aspen in upland areas where conifers are present. Under the current fire regime in the area many upland aspen/conifer stands will likely be lost within 80 to 200 years. The proportion of conifers and browsing pressure were suggested to be the two most important variables affecting aspen regeneration (Strand *et al.* 2009a).

In another study, Stand *et al.* (2009a) developed a spatially explicit landscape simulation model to assess the effects of current and historic wildfire regimes and prescribed burning programs on landscape vegetation composition across two mountain ranges in the Owyhee Plateau, Idaho. Under current fire regimes and in the absence of management activities, loss of seral aspen stands will continue to occur over the next two centuries. However, a return to historic fire regimes (burning 12 to 14 percent of the modeled landscape per decade) would maintain the majority of aspen stands in early and mid-seral woodland stages and minimizes the loss of aspen. A fire rotation of 70 to 80 years was estimated for the historic fire regime while the current fire regime resulted in a fire rotation of 340 to 450 years, emphasizing the fact that fire is currently lacking in the system. Implementation of prescribed burning programs and treating aspen and young conifer woodlands according to historic fire occurrence probabilities, are predicted to prevent conifer dominance and loss of aspen stands. The results presented for the Owyhee Plateau are likely applicable to semi-arid aspen woodlands across the American West where succession to conifers is one major cause of aspen decline.

Within the Northern Great Basin ecoregion, a forest landscape model was developed for the Jarbidge and Mountain City Ranger Districts of the Humboldt-Toiyabe National Forest (Yang *et al.* 2015). This model included the coupling of the LANDIS-II (land disturbance model) with a fine-scaled climate water deficit model which factored in the topography (aspect/elevation) on soil moisture availability. The focus of the model was how the interactions of different fire-climate scenarios at a landscape scale could affect aspen distribution and associated conifer and shrub species over the next 150 years. The Yang *et al.* (2015) model simulations suggest that many aspen stands could persist without fire for centuries under current climate conditions; however, aspen coverage was greater with fires. An increase in fire activity may favor aspen, with an increase in aspen at its upper elevation limits adjacent to coniferous forest, but may also favor a reduction of aspen at lower elevation limits adjacent to xeric shrubland. The frequent fire scenario (30 to 60 year return interval) resulted in the favoring of aspen over subalpine fir at higher elevations and shrub and herbaceous species over aspen at lower elevations. Climate change (simulated 2 to 5 °C increase in temperature) produced only a modest increase in aspen distribution at upper elevations, a more substantial upward shift in the lower elevation range, and an overall contraction to the most mesic environments, leading to an overall reduction of aspen range at the landscape level. Overall, Yang *et al.* (2015) demonstrated that while projected distribution of aspen may greatly reduce in more arid portions

of their existing range, there remains certain environmental settings at the landscape scale where aspen would persist due to the topography, resource availability, and species interactions.

Landscape level models, such as MC2, a dynamic global vegetation model (DGVM), uses climate projections (current or future scenarios) to drive vegetation dynamics and simulates changes in distribution of plant functional types, carbon, and wildfire at broad scales (Bachelet 2013; DGVM 2015; Peterman *et al.* 2014). MC2 has limitations because it only examines vegetation communities, thus incorporating aspen stands into multiple vegetation communities (e.g. temperate evergreen needleleaf forest, temperate evergreen needleleaf woodland, and subalpine forest), which provides an overestimate or underestimate of distribution patterns depending on which vegetation type is examined (Figure 4-25). To overcome some of the limitations, other models such as state and transition and/or species distribution models are integrated into MC2; however, to date there is no published literature available that has combined multiple models to examine broad scale changes in aspen distribution within the Northern Great Basin Ecoregion.

Ongoing Studies

The Northwestern Climate Center is currently funding a similar study to Yang *et al.* (2015), which examines the Jarbidge Mountains (led by Douglas Shinneman and Timothy Link), coupling similar hydrological and landscape-disturbance models to evaluate the climate change effects on aspen distribution and productivity in the Central and Northern Rockies. This on-going study (2014-2016) should provide finer-scale predictions of climate change impacts on aspen distribution in other portions of the ecoregion.

4.4.3 Geoprocessing Modeling and Analysis

The key climate dependencies for aspen have been developed by the USFS Moscow Forestry Sciences Lab to examine the distribution of aspen and effects of climate change for various future scenarios in the ecoregion. The results of the future viability of aspen were presented in the previous REA and are presented in this Climate Change Supplement (Figure 4-23 and 4-24).

Aspen stands have been incorporated into multiple vegetation communities in the MC2 modeling results (e.g. temperate evergreen needleleaf forest, temperate evergreen needleleaf woodland, and subalpine forest). The estimated existing aspen distribution has been overlain with MC2 model results to demonstrate how selecting for a broad-scale vegetation type to determine impacts to aspen can overestimate or underestimate the expected changes to aspen (Figure 4-25).

Aspen distribution is heavily dependent on the soil moisture. CIG Western Hydroclimate data provides the estimated percentile change in soil moisture for all months of the year in gridded format for the Western U.S. As an example, the July 1st soil moisture as a percentile change from the historical mean is presented in Figure 4-26 for areas of the ecoregion that include aspen.

4.4.4 Aspen Results

Based on the USFS Moscow Forestry Sciences Laboratory climate model results for aspen, aspen is projected to significantly decline in the ecoregion in 2030 (Figure 4-23) and 2060 (Figure 4-24). Based on the VIC hydrologic modeling, the summer soil moisture (July 1st) generally increases with the projected increase in precipitation in the low-lying areas. However, reduced snowpack in the higher elevation areas results in a reduction in soil moisture from melting snow when compared to historical conditions. The predicted reductions of summer soil moisture on July 1st (Figure 4-25) appear to overlap with the reductions in aspen viability shown in Figures 4-23 and 4-24. Overall, climate change would be expected to reduce the aspen viability, mainly through increased soil moisture stress in the summer in mountainous areas in the ecoregion.
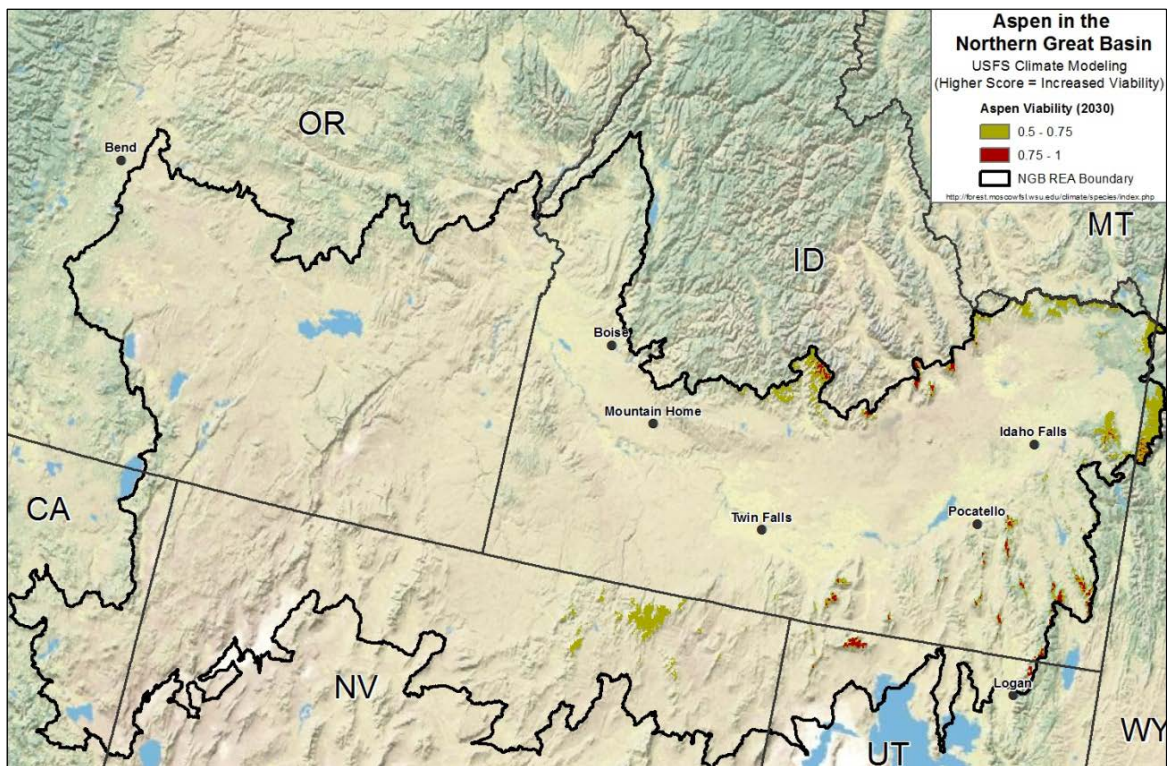
Figure 4-23. USFS Moscow Forestry Sciences Laboratory climate model results for aspen viability across the Northen Great Basin in 2030
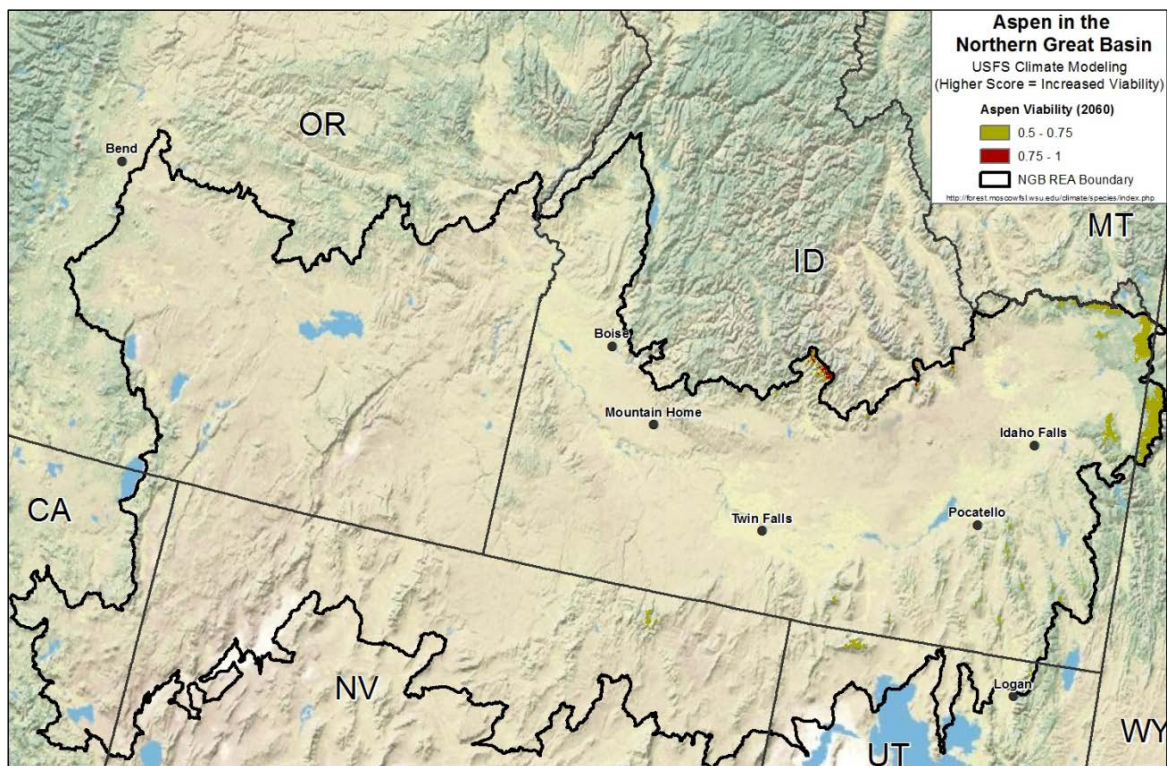
Figure 4-24. USFS Moscow Forestry Sciences Laboratory climate model results for aspen viability across the Northern Great Basin in 2060
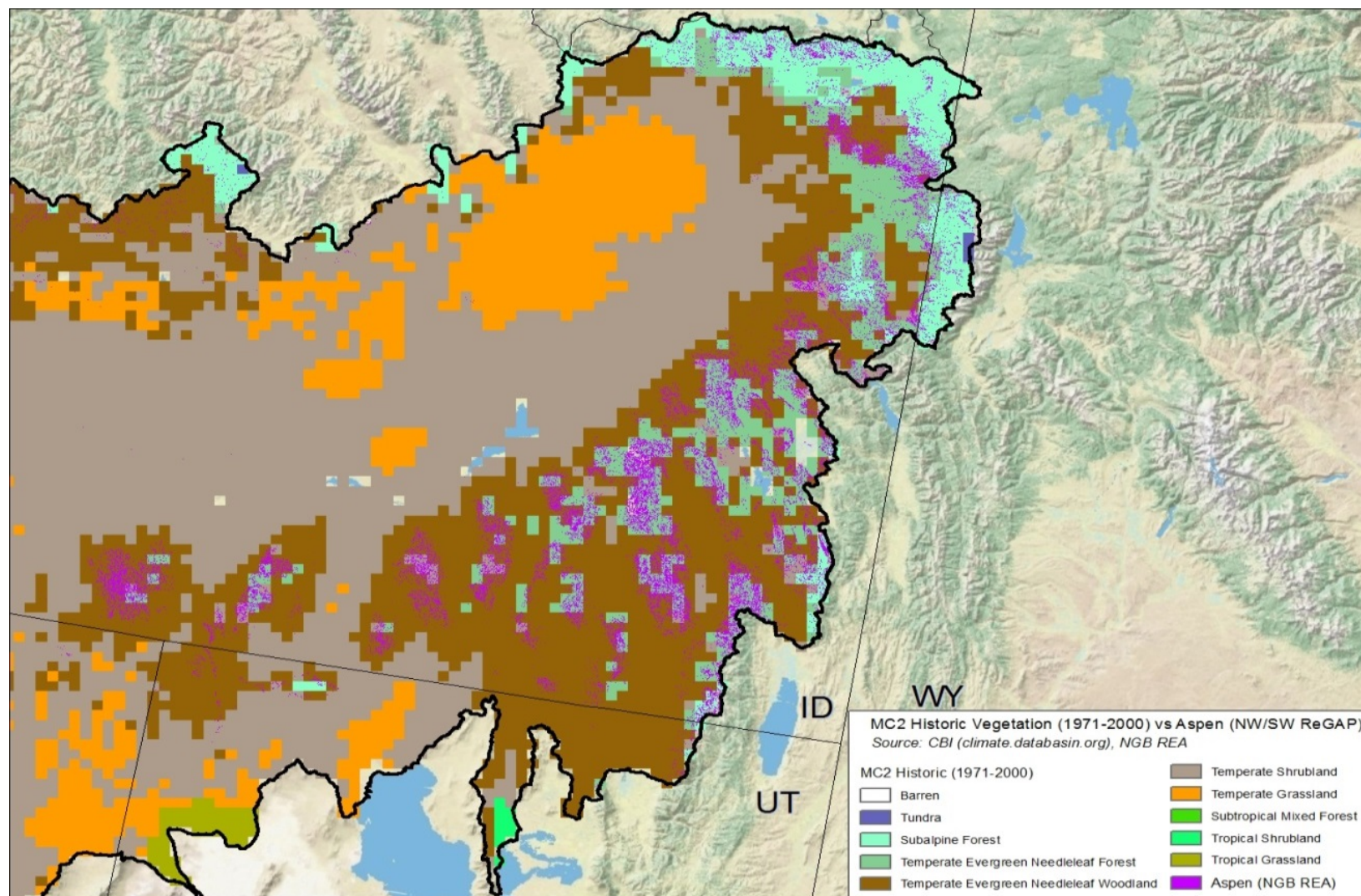
Figure 4-25. MC2 results and Aspen distribution for the eastern portion of the Northern Great Basin. Aspen occurs within multiple different vegetation types
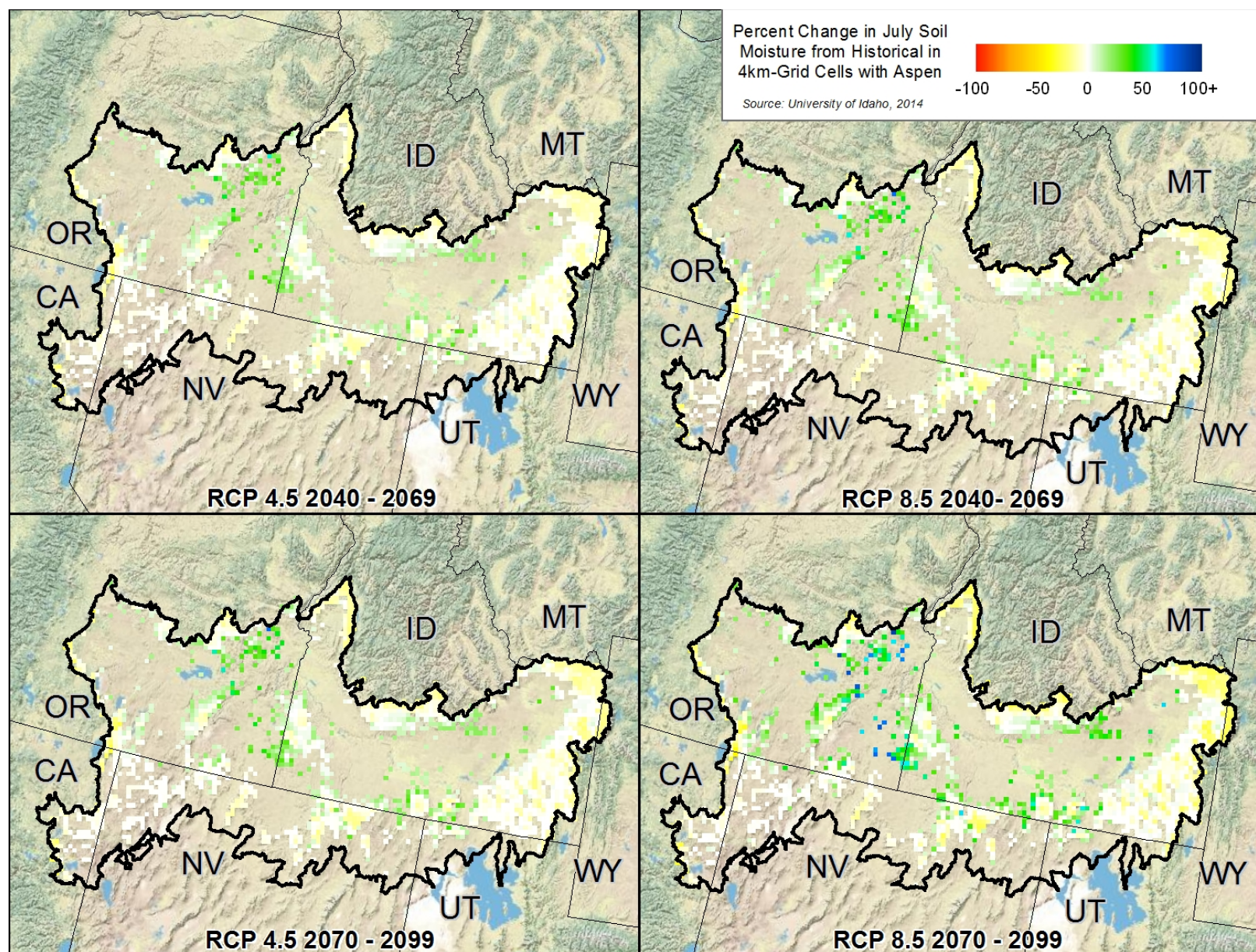
Figure 4-26. Percent Change in July Soil Moisture in 4-km Grid Cells with Aspen

4.4.5 Aspen References

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4.5 Vulnerable Soils

Vulnerable soils are defined as soils susceptible to wind or water erosion. Soil erosion caused by water and wind is a natural process; however, human activities have accelerated the natural erosion process in some areas, which can cause widespread soil loss or degradation with ecosystem-level impacts. Vulnerable soils typically have fine texture (e.g., loess) and may be on sloping terrain or exposed to a long fetch in the direction of prevailing high winds. The fetch is the length over which a given wind has blown. Lack of protective cover by vegetation, biological soil crust (cryptogams), rock, or gravel contributes to the vulnerability of soil to wind or water erosion (Ravi *et al.* 2011).

The northern portion of the ecoregion (Snake River Plain) is located in the Northwest Wheat and Range Region, which includes about 10 million acres of cropland in parts of eastern Washington, north central Oregon, northern Idaho, southeastern Idaho, southwestern Montana, western Wyoming, northwestern Utah, northern California, and other western U.S. regions (USDA 2008). This area suffers high erosion throughout the winter season due to the combination of winter precipitation, intermittent freezing and thawing of soils, steep land slopes, and improper management practices (Greer *et al.* 2006). Midwinter erosion events involve rainfall and/or snowmelt on thawing soils (McCool 1999). The frost layer near the surface limits infiltration and creates a super-saturated moisture condition where almost all rainfall and snowmelt becomes runoff (USDA 2008). These midwinter erosion events occur most intensively where the soil has been finely tilled. Less erosion occurs as a result of cropping management systems such as no-till and pasture, where little mechanical disturbance has occurred (USDA 2008).

4.5.1 Vulnerable Soils Conceptual Model

A conceptual model of vulnerable soils in the ecoregion is presented in Figure 4-27. Change agents of greatest importance to this CE are activities that remove vegetation cover. Conventionally tilled agricultural lands, wildfire, and poorly managed livestock grazing are the CAs with the greatest potential to expose vulnerable soils to wind and water erosion. Other potential CAs include large scale vegetation removal, mining, off-highway vehicles, and climate change. Multiple CAs have the ability to accelerate soil erosion or lead to high soil loss.

4.5.2 Climate Influences

Future climate change is expected to impact the extent, frequency, and magnitude of soil erosion (Mullan 2013). The most direct of these impacts is a projected increase in the erosive power of rainfall. The erosive power of rainfall increases with increasing rainfall amounts and increasing rainfall intensity (Wischmeier and Smith 1978). Increasing temperatures lead to an increase in the moisture holding capacity of the atmosphere (about 7 percent per 1 °C) (Nearing *et al.* 2005). The increased water vapor results in more intense precipitation events (Nearing *et al.* 2005). Very heavy precipitation events have increased throughout the U.S. and are projected to increase in all regions (Melillo *et al.* 2014).

In addition, during the winter months in the ecoregion, snowpack provides protective cover of the soil surface. As temperature warms, there would be a decrease in snowpack on the ground and an increase in rain and snowmelt in winter months. This reduction in snowpack and increase in runoff would facilitate more midwinter erosion events. Based on the Water Erosion Prediction Project modeling of sites near Moscow, Idaho, an increase in temperature of 4 °F resulted in a modeled increase in soil erosion of 192 percent of conventionally tilled agricultural lands. Most of the erosion losses occurred during the winter from October to January. With the warming, soil losses in February to May are tempered by the earlier increase in plant growth (Farrell 2015).

Based on remote sensing data from 1988 to 2009 for the Snake River Plain, the number of days that the ground is frozen ranges from 20 to 40 days per year with the ground covered in snow 60 to 100 days per year (Mastin and Josberger 2014). Climate change simulations of the Reynolds Creek watershed found that over time as temperatures increase and snowpack decreases with climate change, the days when the ground is frozen would likely substantially decrease by nearly 90 percent in 2080 when compared to the baseline (Mastin and Josberger 2014). These changes in snow cover and frost/thaw during the winter could potentially result in more mid-winter runoff and more erosion in the winter months in the ecoregion.

Other more indirect impacts include changes in plant biomass which affects the canopy cover and ground cover (Nearing *et al.* 2005). As vegetation cover is reduced, soil erosion exponentially increases (USACE 2004). In the ecoregion, large, severe wildfires have recently exposed large areas of bare soil to wind erosion. Erosion rates measured on these areas have been as great (or greater) in magnitude as many previously studied environments in Africa, Australia, and the United States (Sankey *et al.* 2009). With warmer temperatures and earlier snowmelt in the ecoregion, wildfires will start earlier in the spring, last later into the fall, and burn more acreage (Melillo *et al.* 2014). As the climate warms, lightning ignition is expected to increase along with fire severity and intensity (Stavros *et al.* 2014). Increased wind erosion could result in increased dust deposition on mountain snow cover. Dust in snow enhances absorbed solar radiation and increases melt rates, threatening snowmelt-fed water resources in the ecoregion (Painter *et al.* 2007).

In portions of the ecoregion, where plant cover is sparse, large amounts of the ground surface are covered by specialized organisms that form biological soil crusts (biocrusts). Biocrusts fix carbon and nitrogen, stabilize soils, and influence hydrology (Ferrenberg *et al.* 2015). Livestock/human trampling and off-road vehicles are known to destroy biocrusts and alter ecosystem function (Belnap and Eldridge 2003). Recent experiments on the Colorado Plateau manipulated the local climate above biocrusts by watering and warming based on predicted changes in climate. The simulated increase in water and warming resulted in a shift in the biocrust to degraded, early successional states (Ferrenberg *et al.* 2015). Experimental climate change was found to have similar impacts on biocrust communities as would physical disturbance. Early successional biocrusts are more prone to erosion, dust production, and reduced water infiltration (Belnap and Eldridge 2003). The changes in the biocrusts could have complex, long-lasting effects on local ecosystem processes (Ferrenberg *et al.* 2015).

Soil and water are essential resources for agricultural production and livestock grazing. Changes in production practices can have more effect than climate change on soil erosion. For example, there is a large disparity in soil erosion losses in agricultural fields which are under conventional tillage practices and no tillage practices (Farrell 2015). However, changes in climate will exacerbate the effects of management practices that do not protect the soil surface from the forces of rainfall (Mellilo *et al.* 2014; Farrell 2015). Erosion can be managed on agricultural lands and grazing lands through maintenance of cover on the soil surface which reduces the effect of rainfall intensity. Many studies on climate change effects on soil erosion have focused on the direct impacts of climate change, whereas factoring in future land use and management that may change as a result of climate change is crucial in understanding the future potential for increased soil erosion as a result of climate change (Mullan *et al.* 2012).
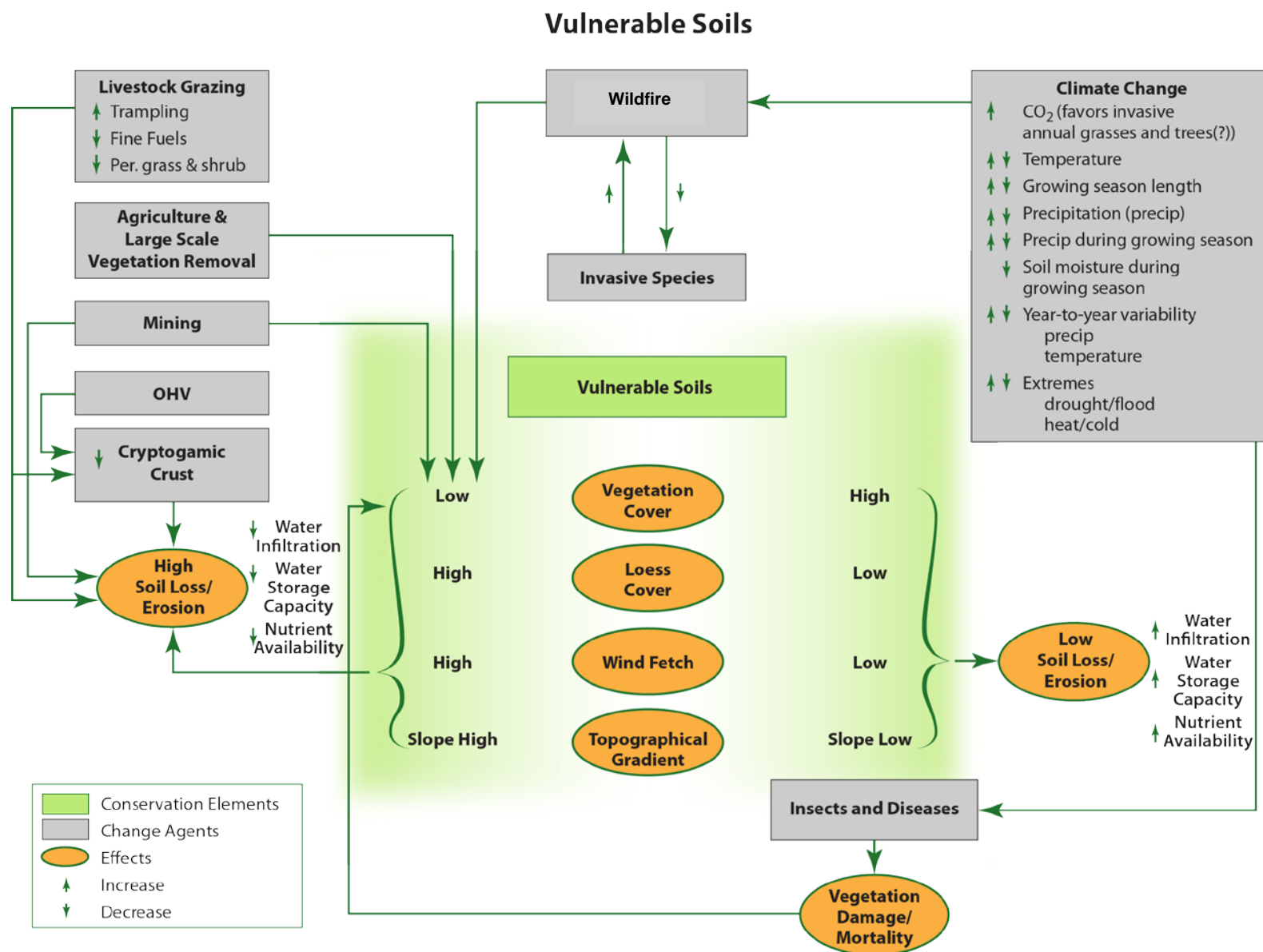
Figure 4-27. Vulnerable Soils Conceptual Model

Cold deserts in the Interior West store much of their carbon below ground in deeper soil layers. In deserts the carbon is stored 95 percent as soil organic carbon and 5 percent as standing biomass (Meyer 2012). However, the potential increase in fire frequency due to climate change and invasive annual grasses may reduce the ability of cold desert soils to retain soil organic carbon. The displacement of desert shrubs by invasive annual grasses through increased frequency of fire results in a transformation from slow to rapid carbon cycling, the cessation of carbon deposition in deeper soil layers, and the rapid conversion of aboveground biomass carbon to atmospheric carbon (Meyer 2012). With the combination of high belowground allocation of carbon and relatively high biomass production, cold deserts, such as those that occur in the ecoregion, are good candidates for carbon sequestration. The restoration of degraded cold desert shrublands would reduce the potential for excessive wind erosion and could result in significant sequestration of carbon in the soil (Meyer 2012).

4.5.3 Geoprocessing Modeling and Analysis

There is a wide range of climate products available for the ecoregion. The key climate variables that can be used to predict potential changes in water and wind erosion rates due to climate change are:

- A change in precipitation as precipitation is a key component used to estimate the rainfall erosivity of the climate in the ecoregion.
- A change in the average wind speed as average wind speed is a key variable in the calculation of the climate factor for the wind erosion potential.
- A change the future occurrence of wildfire with respect to intensity, frequency, and magnitude which could expose vulnerable soils to erosion.

4.5.3.1 Increases in the Erosive Power of Rainfall

Rainfall erosivity is a measure used to estimate how the impact of raindrops and rate of associated runoff affect sheet and rill erosion rates. Global climate models do not provide the details required to directly calculate the rainfall erosivity (R) factor (Segura *et al.* 2014; Biasutti and Seager 2015). Long-term continuous data sets of both the amounts and the maximum 30-minute rainfall intensity are required to calculate the R factor. However, methods to approximately estimate the rainfall erosivity using monthly or annual precipitation records have been developed.

Two recent studies (Segura *et al.* 2014; Biasutti and Seager 2015) used the monthly average rainfall to estimate the projected change in rainfall erosivity for the continental U.S. The erosivity was estimated using the modified Fourier Index, which formulates that a month of intense rainfall creates more erosion than the same rainfall spread over a longer time. Segura et al. (2014) projected changes based on three models and three emissions scenarios from the CMIP3 generation of GCMs. Biasutti and Seager (2015) projected changes using 21 models from the CMIP5 generation for the RCP 8.5 emissions scenario. The data utilized was bias-corrected and downscaled. Both Segura et al. (2014) and Biasutti and Seager (2015) predicted increases in rainfall erosivity in the ecoregion (northwestern U.S.) due to climate change. Biasutti and Seager (2015) also concluded that uncertainty was large for predictions of erosivity in the interior and southern portions of the continental U.S. Biasutti and Seager (2015) attempted to calculate the rainfall erosivity based on daily rainfall estimates as well, but were hindered by model deficiencies and lack of complete erosivity records for historical observations.

The calculation of projected change in rainfall erosivity for the ecoregion based on the predicted monthly precipitation for the MACA-downscaled models is computationally complex. Two recent studies have undertaken this calculation for the continental U.S. For the purposes of the climate change supplement the predicted change in erositivity will be estimated based on a statistical relationship for the annual precipitation. Renard and Freimund in 1994 (as cited in Nearing 2001) developed statistical relationships

between the R factor and total annual precipitation (P) based on data from 155 locations. The relationships are as follows (Nearing 2001):

$$P \ (mm) < 850 \ m \qquad R \ factor = 0.04830P^{1.610}$$

$$P \ (mm) > 850 \ m \qquad R \ factor = 5.87.8 - .1219 \ P + 0.004105P^2$$

These equations will be applied to the change in precipitation estimated in the MACA datasets to calculate the projected increase in rainfall erosivity expected in the ecoregion. The results will be compared to the results from Segura *et al.* (2014) and Biasutti and Seager (2015).

Changes in Average Wind Speed

Wind erosion is also a function of climate. Changes in temperature, precipitation, and wind speed all could influence the wind erosion potential in the region. An overall increase in wind speed would be expected to increase the wind erosion potential.

Changes in Future Wildfire Dynamics

The soils are generally most vulnerable to erosion following large wildfires in the ecoregion. Wildfires are most likely to occur in the summer months. The analysis in the REA compared existing the FSIM modeled wildfire risk with the areas of high wind erosion potential. Based on the review of the available data, there was not a similar dataset of FSIM wildfire probability which also factors in the effects of climate change. To evaluate the change in wildfire risk, the Fraction of Area Burned estimated by the MC2 model was compared to areas that are susceptible to wind ersion.

4.5.4 Results

Based on the modeling projections, annual precipitation is expected to increase by 7 to 9 percent at mid-century (2040-2069) and 11 to 18 percent at the end of the century (2070-2099). This annual increase in precipitation would be expected to increase the rainfall erosivity in the region by 8 to 11 percent from 2040-2069 and 13 to 21 percent from 2070-2099.

Overall, the average summer wind speed (June, July, and August) is projected to slightly decrease by 0.09 to 0.12 m/s at the mid-century (2040-2069) and decrease by 0.11 to 0.16 m/s at the end of the century (2070-2099) (Figure 4-28). These changes range from 1 to 3 percent of the historical average wind speeds. Therefore, predicted changes in wind speed related to climate change are not likely to have a significant effect on wind erosion rates in the ecoregion. However, the fraction of area burned is expected to increase with climate change in the ecoregion by 200 percent. Much of this increase will occur where soils are vulnerable to wind erosion (Figure 4-29). This will likely result in more catastrophic fires which are often followed by large wind erosion.
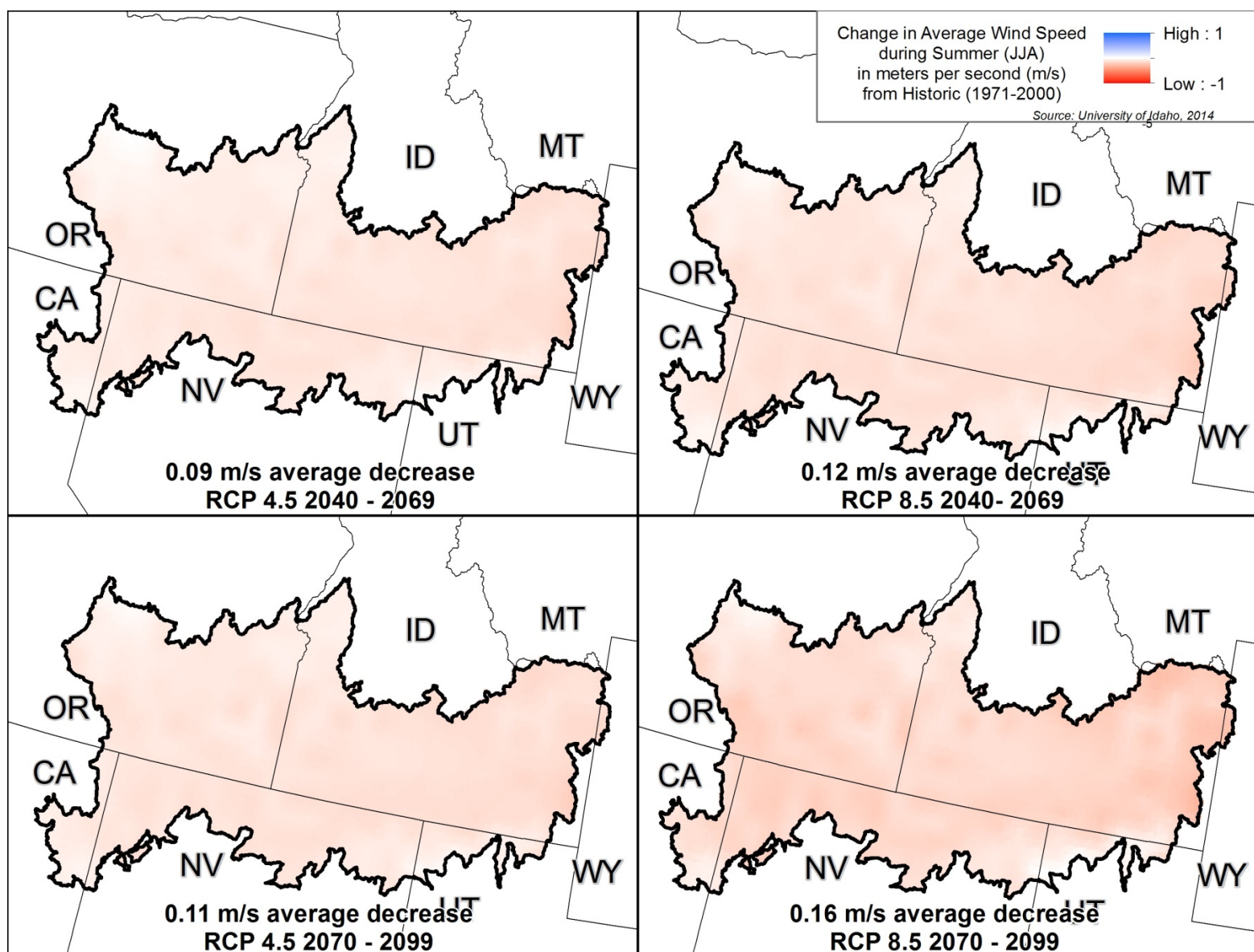
Figure 4-28. Predicted Change in Average Wind Speed during the Summer (meters/second)
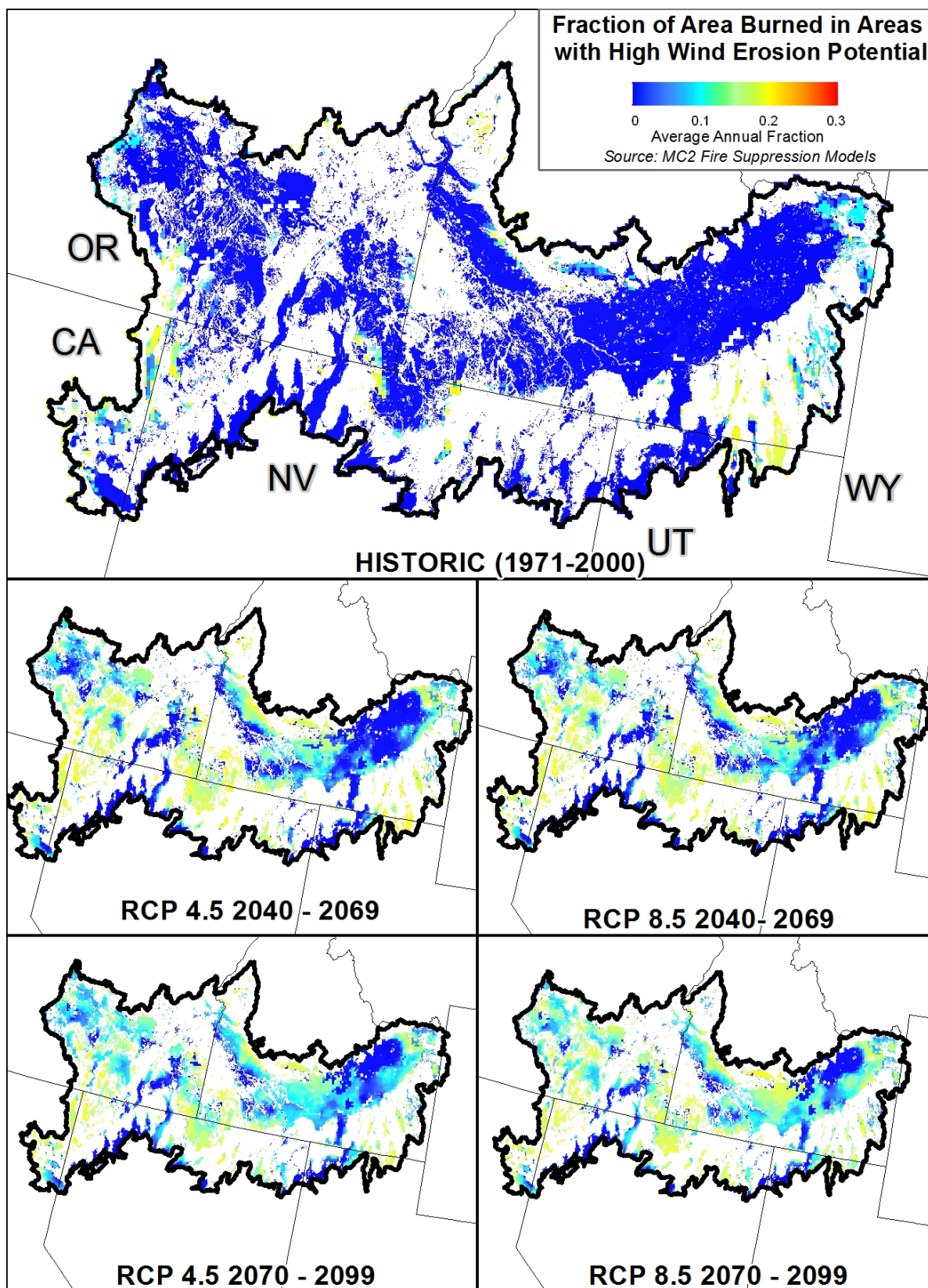
Figure 4-29. Fraction of Area Burned in Areas with High Wind Erosion Potential

4.5.5 Vulnerable Soils References

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4.6 Hydrology (Groundwater, Springs & Seeps, Streams & Rivers, Open Water, Riparian & Wetlands)

Across the landscape surface water, groundwater, and geological characteristics control the presence of perhaps the most important element in shaping the West – water. Together with climate, water availability is a key variable in the development and sustenance of ecological communities. Springs and seeps are known as biological hotspots, associated with unique aquatic ecosystems. Perennial streams and rivers are the lifeblood of the Northern Great Basin ecoregion and reflect effects occurring on a landscape scale throughout the watershed. Flow regulation due to dam operations can alter the natural hydrology of perennial streams and rivers. Dams are operated for multiple purposes in the ecoregion: flood control, water storage, and hydroelectricity. Reservoirs created by dams make up the majority of open water in the Northern Great Basin.

Most of the prime agricultural land in the ecoregion is under cultivation and agriculture which accounts for 97 percent of the overall water use. The water demand in the ecoregion is met by approximately 75 percent surface water diversion and 25 percent groundwater withdrawals. Groundwater withdrawals increased in the ecoregion by over 50 percent, from 1995 to 2005, as agricultural lands have shifted from surface water to groundwater irrigation (Kenny *et al.* 2009).

4.6.1 Hydrology Conceptual Model

A conceptual model for hydrology in the ecoregion is presented in Figure 4-30. The most significant CA is development, which includes dams, diversions, groundwater pumping, agricultural, and other water uses. Climate change can also influence the hydrology directly or indirectly by altering the timing, duration, and amount of available water in the form of snowmelt and rainfall. A portion of the available water is subject to evapotranspiration, which is influenced by solar radiation, topographic shading, vegetation density, and cloudiness, all of which are influenced by climate. The remaining available water, fed by mountain streams, springs, and subsurface outflow, consists of surface runoff and groundwater discharge. Other factors that may influence hydrology to a much lesser extent are: wildfire that may lead to the removal of vegetation in the watershed; livestock grazing that may trample and consume vegetation, which in time may alter the groundwater levels; and mining that could contaminate groundwater and also alter groundwater levels. Vegetation cover is important in capturing and slowing precipitation runoff velocity, both from snow and rain, to allow time for water to infiltrate into the substrate and recharge into the groundwater reservoir. Without vegetative cover, precipitation runs off quickly and leaves the watershed.

4.6.2 Climate Influences

Climate projections indicate that the ecoregion will experience temperature increases in both cool and warm seasons, and a slight increase in annual precipitation (7 to 18 percent), with the highest increases expected to occur in the winter (11 to 30 percent) (Section 3). Warmer temperatures in the future will result in more winter precipitation falling as rain rather than snow throughout much of the ecoregion, particularly in mid-elevation basins where average winter temperatures are near freezing (Hamlet *et al.* 2013). The shift from less snow to more rain will result in:

- less accumulation of winter snow pack;
- higher winter streamflows in response to storm events;
- earlier spring snowmelt;
- earlier peak spring streamflow; and
- lower summer streamflows.
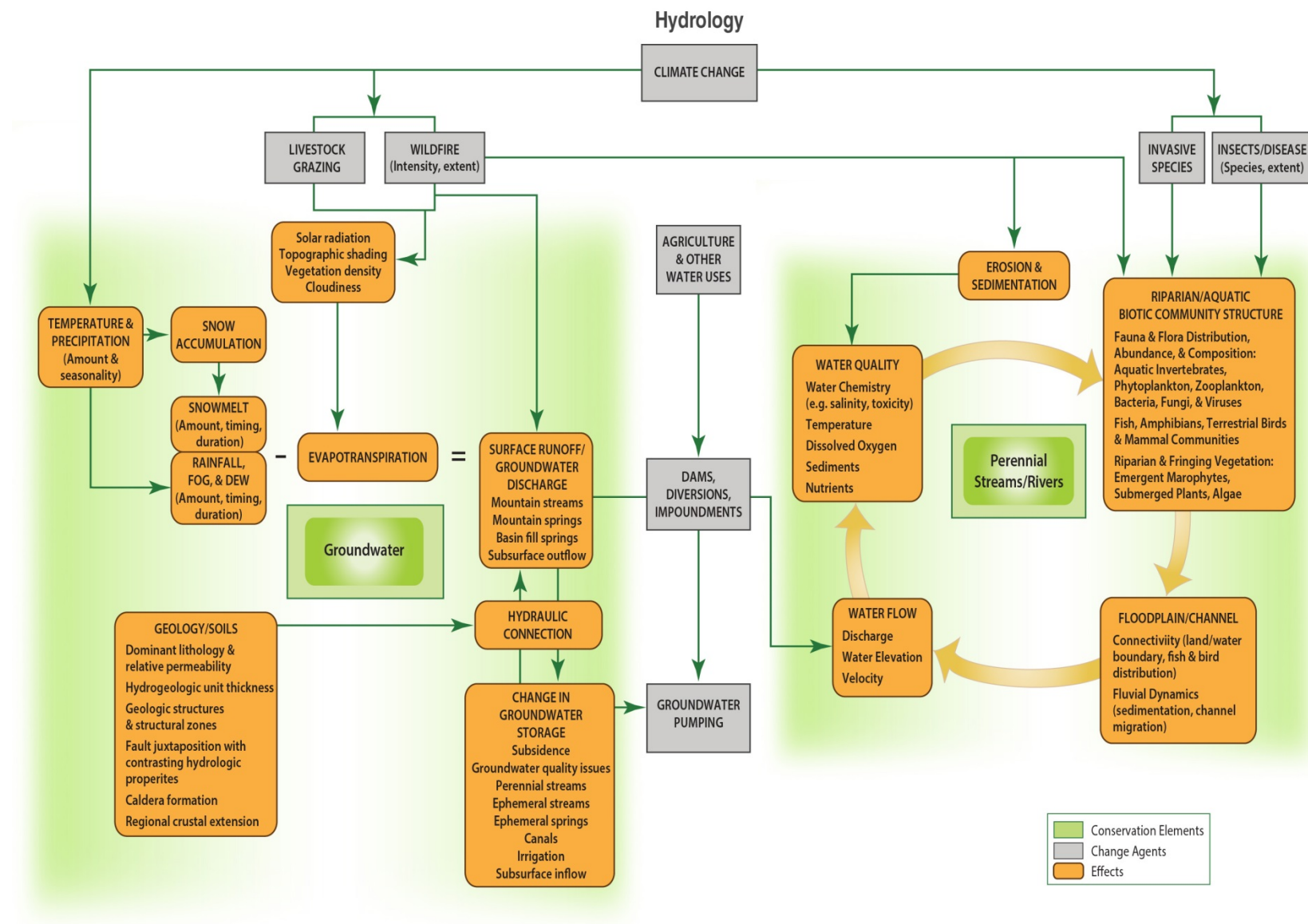
Hydrology

Figure 4-30. Hydrology Conceptual Model

Observed climate trends in the Owyhee Uplands on Reynolds Creek Experimental Watershed have already measured seasonal shifts in streamflow due to increased temperatures, with larger streamflows in winter and early spring and reduced streamflow in summer (Goodrich *et al.* 2011). Springtime irrigation diversions also have increased between 1970 and 2007 in the Snake River basin due to the earlier snowmelt, reducing the spring soil moisture and increasing the springtime need for irrigation (Hoekema and Sridhar 2011).

For the reservoir systems in the ecoregion with flood control objectives, warming could result in increased winter runoff to manage during flood control operations. This may require higher winter releases from reservoirs to be able to manage flood control risks. Higher winter releases may affect the summer season supply, with spring refill beginning with less winter carryover storage. Substantial reductions in summer streamflow, which will emerge in coming decades, will adversely affect many water users, including farmers who rely on irrigation, resident and anadromous fish, and summertime hydropower production (BOR 2013). Based on recent climate projections (CMIP3 and CMIP5), it appears plausible that precipitation increases throughout the Northern Great Basin could occur in association with increased temperatures. Such increased precipitation could offset some portion of summer runoff decreases associated with warming alone (BOR 2013). Projected reductions in late spring and summer snowmelt runoff are largely balanced by increases in winter and early spring runoff, as more precipitation is projected to fall as rain rather than snow.

The predominant water demand (97 percent) in the ecoregion is for agricultural irrigation (Kenny *et al.* 2009). As temperature increases, so does plant water consumption and surface water evaporation rates. With reduced surface water availability in the summer, groundwater use would likely increase to meet the deficit in water demand. Groundwater withdrawals can eventually result in a reduction in aquifer discharge, reducing the flow of springs, streams, and the extent of groundwater dependent wetlands and riparian systems (Bredehoft and Durbin 2009).

Mountains are generally wetter and cooler than adjacent basins, and groundwater in the West is derived mainly from mountain precipitation. Snowmelt provides more recharge in western mountains than does rain (Earman *et al.* 2006). Large, sustained infiltrations of water are needed to break through the thick unsaturated zones and snowpack store and then slowly release precipitation from several storms at once. Isotopic studies in western settings have suggested that 50 to 90 percent of recharge is from snowmelt (Earman *et al.* 2006). As the temperatures warm in the 21st century, mountain-block recharge would also be expected to decline, as recharge areas shrink and snow available for snowmelt infiltration dwindles. Groundwater dynamics can have a significant role in determining streamflow responses to warming (Tague *et al.* 2008). Local variations in bedrock geology and its variations in volume and seasonal fluxes of groundwater will likely result in significant variability in how streamflow responds to climate change. For example, hydrologic modeling of the porous, volcanic formations of the High Cascades in Oregon (which are similar to those in the Snake River plain) indicated that the High Cascade volcanic formations had greater absolute reductions in summer streamflow with predicted temperature increases than shallower subsurface systems in the Western Cascades (Tague *et al.* 2008). With warming temperatures, snowpack will be reduced and will melt earlier in the season, resulting in less groundwater recharge in the mountains. In slow-draining systems like the High Cascades, the groundwater recharge supports summer streamflows through spring discharge. With less recharge from snowmelt, summer streamflows will be reduced (Tague and Grant 2009). Coupling the groundwater dynamics with the snowpack data, Safeeq *et al.* (2013) analyzed historical streamflow trends and also concluded that summer streamflows in watersheds that drain slowly from deep groundwater and receive precipitation from snow are most sensitive to climate warming. The rivers and streams in the northern and eastern portion of the ecoregion (e.g. Boise River) are both snow-melt dominant and slow-draining (Safeeq *et al.* 2013) and summer streamflows are expected to be especially sensitive to climate warming.

Observed warming trends have changed the flood risks of watersheds in the 20th century (Hamlet and Lettenmaier 2007). Snowmelt dominant systems have reduced flood risks due to the reduction in snowpack from warming. Transitional systems (both rain and snowmelt driven) generally have increased flood risks. Rain dominant systems show little to no systematic change (Hamlet and Lettenmaier 2007). Warming in the late 20th century has also coincided with increased interannual precipitation variability (both extreme wet conditions and extreme dry conditions). It is not clear whether the change in precipitation variability over the region in the late 20th century is systematic in nature (like temperature increases), or whether it is due to natural variability that has been coincident with large-scale warming (Hamlet and Lettenmaier 2007).

While trends toward increasing precipitation in the ecoregion have been forecasted, it is also likely that the region will continue to experience substantial interannual to interdecadal variability in precipitation (BOR 2013). Natural resource managers should continue to seek operations that are robust to a wide range of hydrologic conditions (wet or dry, extreme high or low flows, etc.) while preparing for a future with less springtime snowpack and snow-fed runoff and reduced summer streamflow for most ecoregion watersheds (Hamlet *et al.* 2013).

4.6.2.1 Hydrology-Related Climate Change Studies in the Ecoregion

The Climate Impact Group has developed hydrologic projections for the Western U.S. derived from the CMIP3 and CMIP5 global model archive. The projected changes in hydrology were obtained by using a Variable Infiltration Capacity (VIC) macroscale hydrologic model to translate the downscaled changes in temperature and precipitation to changes in hydrology. Simulated changes in hydrology are available for 23 different variables, including snow water equivalent, soil moisture, potential evapotranspiration, actual evapotranspiration, and runoff. The Climate Impact Group is currently updating the Hydroclimate modeling results, based on the temperature and precipitation outputs from the CMIP5 model, which was downscaled using the MACA method.

4.6.3 Geoprocessing Modeling and Analysis

There are a wide range of hydrologic products available for the ecoregion from the CIG Hydroclimate Modeling (Hamlet *et al.* 2013). However, the key hydrologic impacts will be demonstrated by the following variables:

- A change from snowmelt dominant to rainfall dominant watersheds estimated by the ratio of the April 1 snow water equivalent to the total winter precipitation.
- Shifts in runoff patterns with more winter runoff and less spring snowmelt represented by monthly hydrographs for major streams in the ecoregion.
- Changes in soil moisture as a percentile of the historical mean for July 1st. Soil moisture has important implications for agriculture as well as impacting the risk of wildfire.

4.6.4 Results

Rainfall dominant basins are categorized based on the ratio of the April 1 snow water equivalent to the total winter precipitation (October to March). If less than ten percent of the winter precipitation is stored as snowpack on April 1, the system is classified as a rainfall dominant system (Littell *et al.* 2014). The Climate Impact Group provides the ratio of the April 1 snow water equivalent to the total winter precipitation ratio by the Hydrologic Unit Code 10 – basin for the Western U.S. The ratios for the historical condition and the A1B Scenario averaged for 2060-2099 are provided in Figure 4-31 and 4-33. Based on modeling predictions, the ecoregion will shift towards more rain dominant behavior as the region's temperatures warm. Currently, based on historical data, 85 percent of the basins would be

classified as rainfall dominant (Figure 4-31). The rainfall dominant basins increase to 98 percent of the ecoregion by end of the 21st century (Figure 4-32). The classification of a rainfall vs. snowmelt dominated system from the CMIP5 generation VIC modeling was not available. Therefore the results from CMIP3 are presented in Figures 4-31 and 4-32.

Simulated average monthly hydrology is available for 24 stream gaging stations in the region. These monthly hydrographs were created based on the Western Hydroclimate modeling by the Climate Impact Group. For example, the results of six sites are presented in Figures 4-33 and 4-34. Modeling generally predicts a slight increase in overall annual flow (10 to 20 percent) by the end of the century for most sites. The increase in flow is projected to occur in the winter months. Peak flows generally shift one month earlier due to the reduction in snowpack. Spring and summer flows are projected to decrease with less available snowpack. The modeling does not include changes in the streamflow response due to a reduced groundwater recharge in slow draining systems discussed above (Safeeq *et al.* 2013).

Vegetation and agriculture rely heavily on soil moisture. CIG Integrated Scenarios data provides the estimated change in total soil moisture for all months of the year in gridded format for the Western U.S. Increases in summer soil moisture would generally reduce water stress on native vegetation and a decrease would increase plant stress and the water use requirements for agriculture in the ecoregion. The change in total soil moisture for July 1st is provided in Figure 4-35. Lower elevation areas are expected to have a slight increase in soil moisture due to increased precipitation. However, in mountainous areas summer soil is likely to decrease the reduction in snowpack.
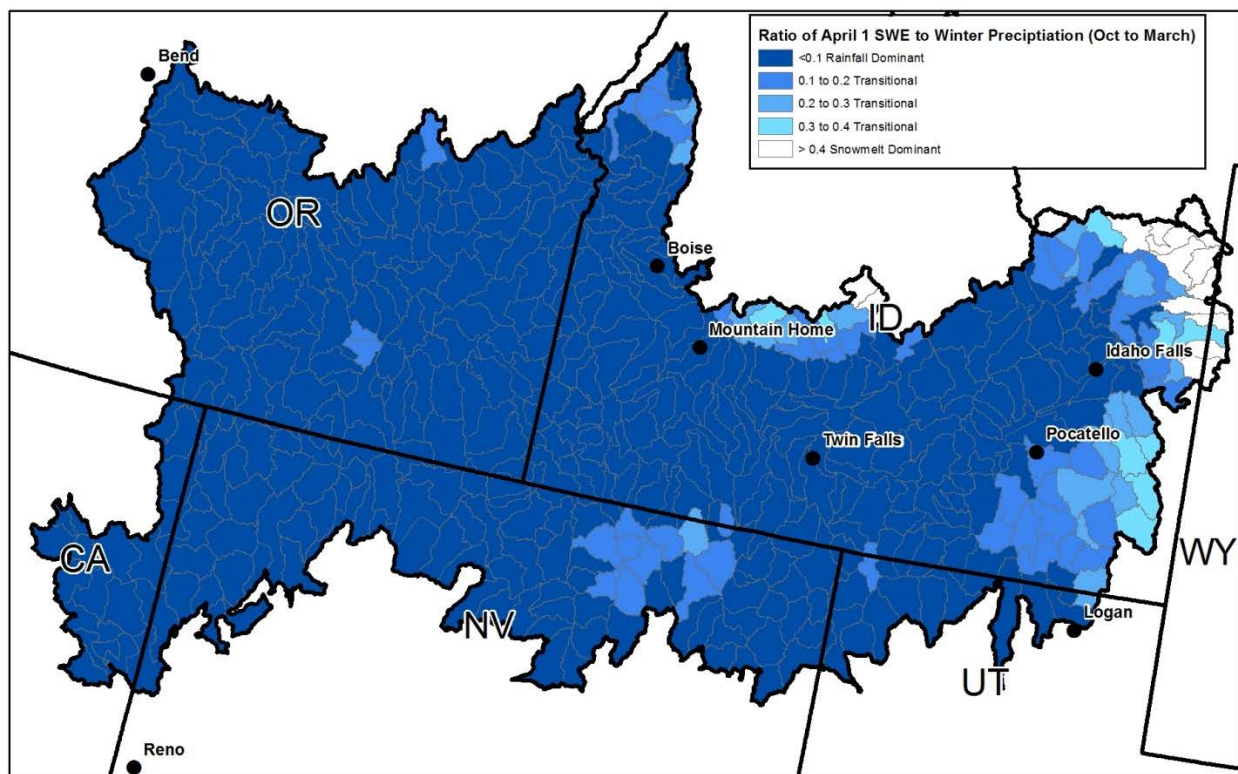
Figure 4-31. Watershed Classification – Historical
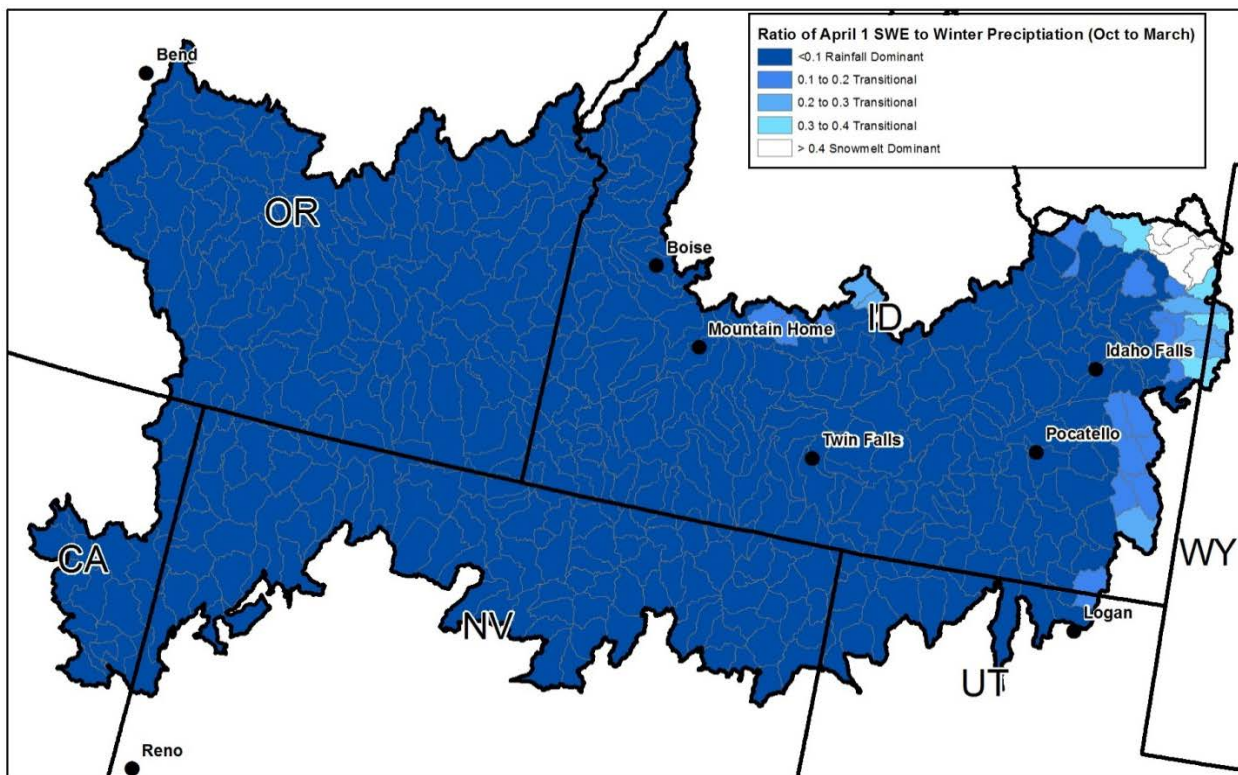
Figure 4-32. Watershed Classification – A1B (2060-2099)
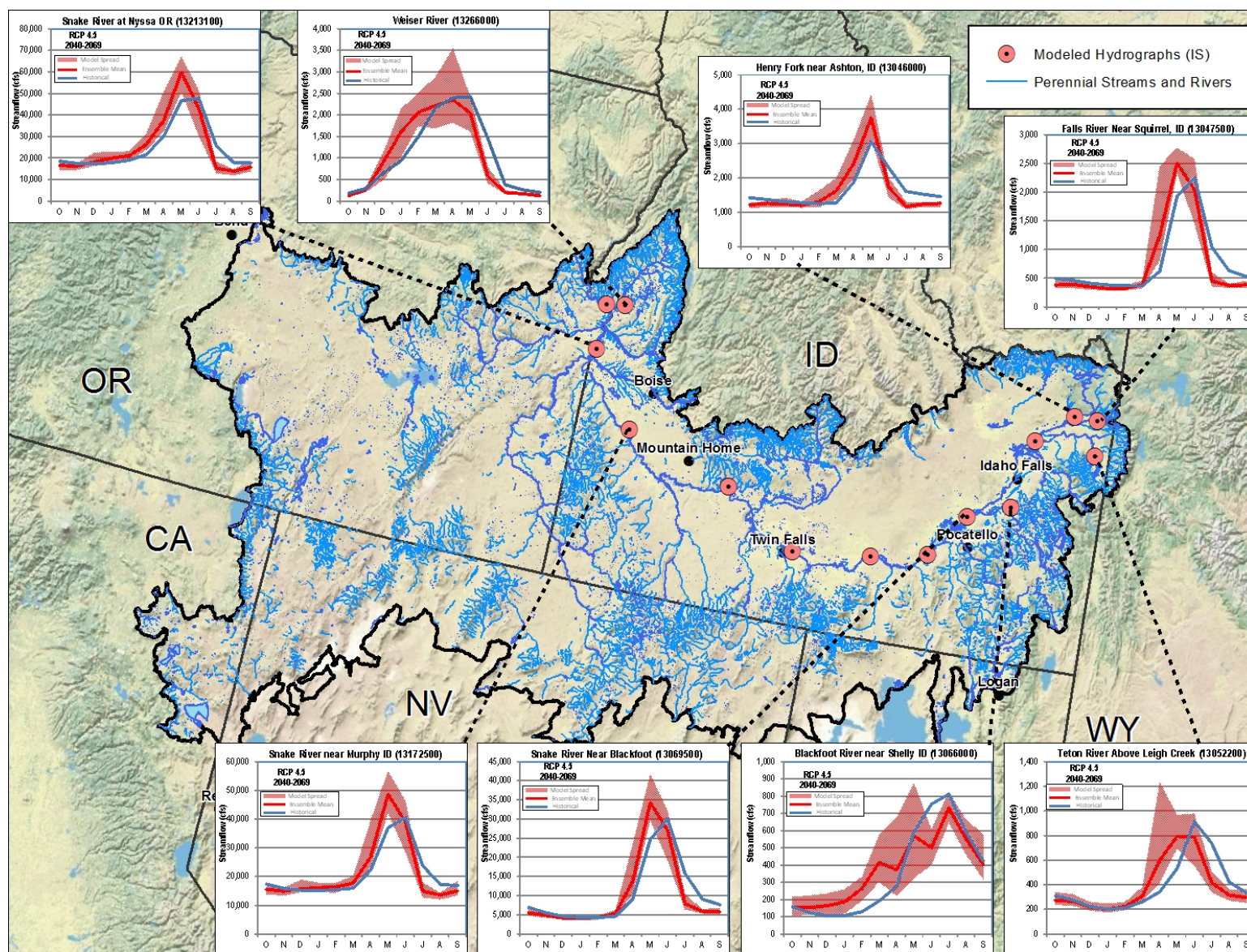
Figure 4-33. Selected Projected Hydrographs (CMIP5)
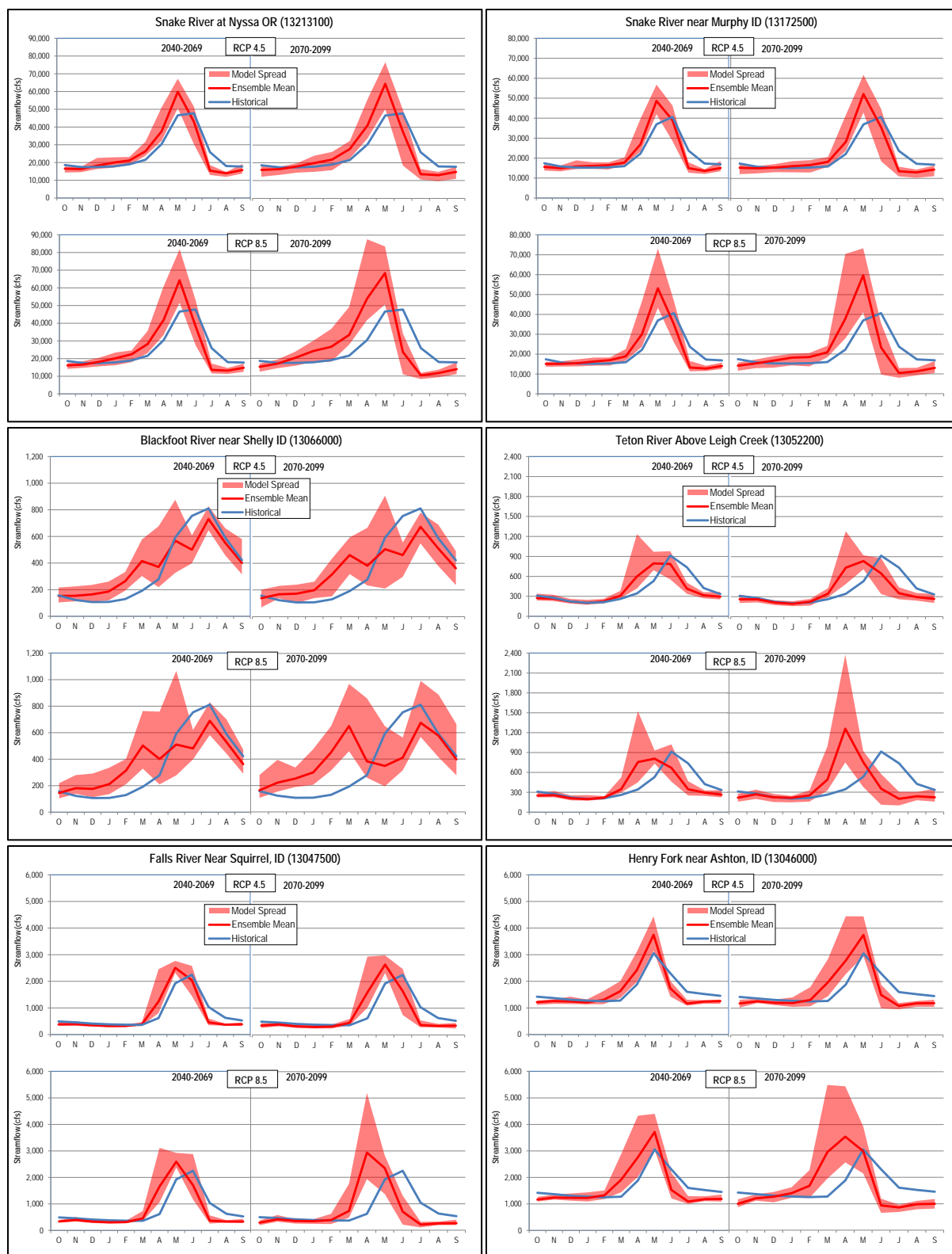
Figure 4-34. Projected Hydrographs under various Climate Change Scenarios
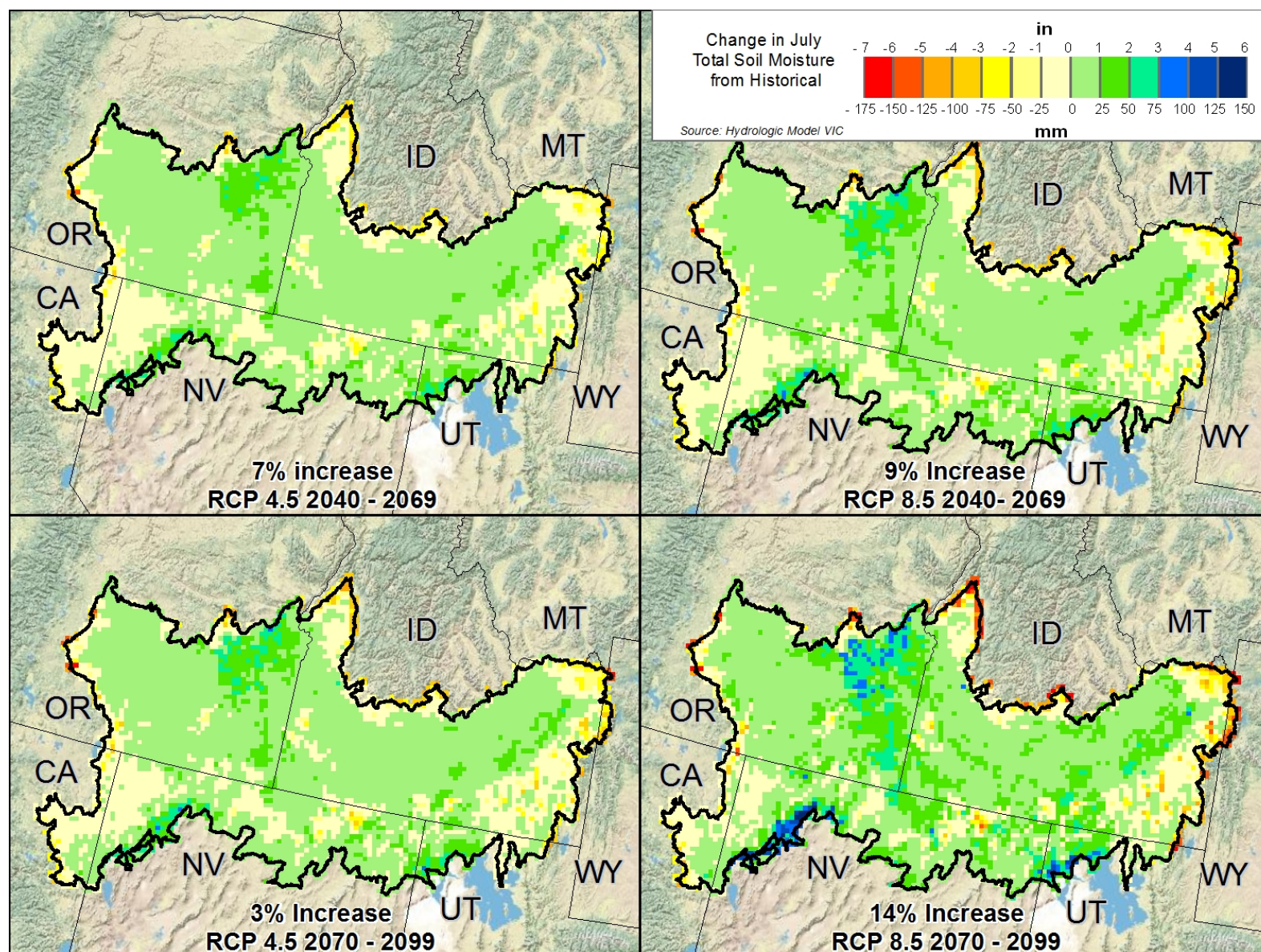
Figure 4-35. Projected Total Change in Soil Moisture in July

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4.7 Mule Deer

The mule deer is one of the most common and recognizable large mammals and is popular for hunting and wildlife viewing. Mule deer are one of the most widely distributed and economically and socially important animals in western North America (Cox *et al.* 2009). Over the past century, mule deer populations have fluctuated throughout their range; however, trends in the latter part of the 20th century indicated that mule populations were declining. Much of this decline can be attributed to direct habitat loss (mainly winter range), a loss of browse species and deteriorating forage base, and weather extremes including large-scale droughts and fire suppression (Heffelfinger and Messmer 2003). This led to the creation of a Mule Deer Working Group in 1998. The 2016 population report by the Mule Deer Working Group (MDWG 2016) found that in most Western states and provinces most mule deer populations are stable or recently recovering from previous declines. For example, the estimated Idaho mule deer population has increased from 2011 to 2014. However, the Oregon mule deer population is stable but has not recovered from previous declines (MDWG 2016). Besides habitat loss issues, mule deer were chosen as a CE because of their status as an important game species in all of the Northern Great Basin states.

Mule deer in the Northern Great Basin ecoregion inhabit areas primarily classified as sagebrush (*Artemisia* spp.) and other shrub-steppe habitats. Throughout the ecoregion riparian and woodland areas are often interspersed within the sageshrub steppe, providing a mosaic of habitat types across the landscape. An important aspect of good mule deer habitat is the juxtaposition of browse and security (Leckenby *et al.* 1982). The diverse environmental and climatic conditions across the species' range result in dynamic relationships between mule deer and their habitats (Cox *et al.* 2009). Mule deer are generalists, but are typically highly selective foragers (browsers) that rely on specific components of palatable shrubs and forbs. Vegetation disturbance and subsequent renewal is a key element to maintaining high quality deer habitat; however, many natural disturbance regimes have been altered over the last several decades.

4.7.1 Mule Deer Conceptual Model

Conceptual system models were developed as part of the pre-assessment phase to assist with the determination of key factors that are important to mule deer in the Northern Great Basin ecoregion. The model was developed to provide an ecological framework and justification for the choice of indicators that were used in assessing CA threats for this CE. This model incorporates the identified CAs, as well as potential effects from the actions of the CAs on both the landscape and local habitat/terrain levels (Figure 4-36). This model begins with depicting the important habitat components (or functions and values) for mule deer required throughout the year. It then incorporates the identified CAs, as well as potential effects from the actions of the CAs on a landscape level (Figure 4-36). Unlike the other fine-filter CE ungulates that require large home ranges (bighorn sheep and pronghorn), the conceptual models for mule deer did not utilize a local habitat/terrain scale analysis. Mule deer are more of a habitat generalist than the other two species and remain near areas with high vegetative cover rather than in the high visibility areas preferred by bighorn sheep and pronghorn.

The CAs considered for this analysis include development, climate change, wildfire and invasive species, and disease, depicted in brown boxes across the top of the model (Figure 4-36). The mule deer model shows the pathway of CAs that affect mule deer landscape variables and thereby, the mule deer functions and values, depicted in the lower box. The predicted results of CA effects are presented in blue boxes in Figure 4-36.

Table 4-13. Change Agents and their Principal Effects on Mule Deer

Change Agent	Principal Effects on Mule Deer
Development	Fences, road networks, and increased human disturbance associated with energy and housing developments can influence the effectiveness of mule deer migration routes (Sawyer *et al.* 2005). Migration bottlenecks are created in areas along historic migration routes where topography, vegetation, or other landscape features may restrict animal movements to narrow or limited regions, which can be exacerbated by the addition of human influences such as development. Roads are widely recognized as having a range of direct, indirect, and cumulative effects on wildlife and their habitats (Trombulak and Frissell 2000; Gaines *et al.* 2003; Wisdom *et al.* 2004). Road construction often accompanies other development types listed in the model (Figure 4-63) including urban/exurban, agriculture, oil/gas, and alternative energy developments. Mule deer comprised the largest percentage of wildlife species killed as a result of vehicle collisions (62 percent) (Craighead *et al.* 2001).
Climate change	Climate change may increase environmental extremes and influence habitat changes for the mule deer in the Northern Great Basin through the effects of summer and winter precipitation and temperature regime changes on forage resources, winter migration, invasive species, and disease. Global warming patterns are also projected to lead to loss of sagebrush winter ranges and increasing coniferous communities, which will ultimately reduce habitat quality for mule deer during winter (Lutz *et al.* 2003). Climate change may also affect other CAs including wildfire, invasive species, and disease. The climate change impacts on mule deer are discussed in more detail below in Section 4.7.2.
Wildfire and invasive species	Fire generally has a beneficial impact on mule deer habitat by stimulating earlier greenup the following spring, increasing availability and nutritional quality of forage and more herbaceous plants. However, fire can also facilitate the spread of invasive annual grasses including cheatgrass, medusahead, and ventenata, which have low value to foraging mule deer and may reduce shrub cover and browse availability (d'Antonio and Vitousek 1992; d'Antonio 2000; Brooks *et al.* 2004). Increasing fire intervals that are supported by the abundance of fine fuels (e.g., cheatgrass and invasive forbs) tend to reduce and ultimately eliminate browse species that deer heavily rely on (Link *et al.* 2006). Cox *et al.* (2009) estimate that the historic 30- to 100-year fire cycle has been reduced to a 5- to 10-year cycle in portions of the region due to the abundance of invasive cheatgrass. Conversely, in some areas the absence of fire for 50 years or more can facilitate conifer encroachment, canopy closure, and deterioration of herbaceous and shrub understories, also resulting in deterioration of mule deer open and varied habitats (Cox *et al.* 2009). Increased fire size can also dislocate and concentrate herds into smaller habitat areas.
Disease	There are several diseases (e.g., chronic wasting disease [CWD], bluetongue disease, epizootic hemorrhagic disease [EHD]) that affect mule deer which may increase in prevalence and result in negative effects on populations in the Northern Great Basin, especially those undergoing stresses from other factors. Currently, CWD, which attacks the brains of infected deer, elk, and moose and is always fatal, is primarily concentrated in wild deer in the mid-western states, the northern Rockies states, and in July 2012 was detected in Texas (Chronic Wasting Disease Alliance 2012). Farmed and captive deer in a wider geographic area have tested positive for CWD (Chronic Wasting Disease Alliance 2012). Within the Northern Great Basin states only Utah is known to have recorded occurrences of CDW. Hemorrhagic disease is caused by either of two closely related viruses, epizootic hemorrhagic disease (EHD) virus or bluetongue virus (SCWDS Group 2012). Because symptoms produced by these viruses are indistinguishable, a general term, hemorrhagic disease is often used when the specific virus responsible is unknown. These viruses are seasonal, transmitted by biting flies, and fatal within 24 hours after a 7- to 10-day incubation period. Wild ruminants have been infected including mule deer, pronghorn, and bighorn sheep (SCWDS Group 2012).

DEVELOPMENT
- Urban/Exurban
- Agriculture
- Alternative Energy
- Oil & Gas
- Rangeland Treatments
- Recreation/Hunting
- Roads
- Grazing

CLIMATE CHANGE
- Temperature
- Snowpack
- Drought
- Precipitation

INVASIVE SPECIES
- Conifer Encroachment
- Cheatgrass
- Invasive Forbs

WILDFIRE
- Frequency
- Severity
- Burn Intervals

DISEASE
- Chronic Wasting
- EHID and Bluetongue (Hemorrhagic Disease)

MORTALITY & DISTURBANCE
- Recreation/ Disturbance
- Hunting
- Roadkill
- Reduced Access/ Corridors

LANDSCAPE EFFECTS
- Vegetation shifts (latitudinal & elevational)
- Habitat Fragmentation (winter range and migration route)
- Succession/Vegetation Transactions (juniper woodlands, browse species)
- Habitat Loss (winter range)
- Movement Corridors

Functions & Values

Mule Deer
- Survival
- Reproduction/Recruitment
- Distribution
- Density

LEGEND
Change Agents
Effects

Figure 4-36. Mule Deer Conceptual Model

4.7.2 Climate Influences

In the ecoregion, there is a large annual variation in mule deer survival which is in part dependent on the climate (Hurley et al. 2011). Spring precipitation and soil moisture are important for the production of high quality mule deer forage and summer precipitation is critical in maintaining forage quality. Increased rainfall in summer promotes the growth of forbs, an important forage for mule deer, and can increase new growth in autumn germinating annual grasses (e.g. cheatgrass) as well as delay plant senescence, prolonging access to higher quality forage (Hurly *et al.* 2014). An analysis of Nevada's mule deer populations and summer precipitation data from 1978 to 2002 found over 80 percent of the variation in mule deer population could be explained by a 6-year average of monthly precipitation received in July, August, and September (Wasley 2004). A recent study of mule deer fawn winter survival in Idaho found that autumn plant growth (measured by remote sensing) had twice the effect on mule deer fawn body mass as spring plant growth (Hurley *et al.* 2014). In semiarid climates available forage in autumn may be more important than spring for survival over the winter. Warming temperatures associated with climate change will exacerbate the effects of drought by increasing rates of evapotranspiration when water availability is already limited. The impacts of drought on mule deer populations will likely be more severe in the future.

Severe winters with lower than normal temperatures and increased snow pack are also known to reduce the mule deer densities (Hurley *et al.* 2011). Below average precipitation in the summer, which reduces forage quality, followed by above-average winter precipitation restricting winter migration, can result in low over-winter mule deer survival and reduced population growth. Mule deer are less affected by severe cold weather than by high levels of snow cover, which restrict access to forage. Gilbert *et al.* (1970) stated that snow depth over 18 inches precluded use of winter range by deer, but energy costs of locomotion for mule deer increase significantly at 10 inches (25 centimeters), regardless of the density of snow (Parker *et al.* 1984). Conversely, high summer precipitation coupled with low winter precipitation could reduce winter fawn mortality (Hurley *et al.* 2011). With the predicted increases in temperature and slight increases in summer precipitation, juvenile mule deer growth may result in better body conditions going into the winter and less mortality. Climate change models project lower snowfall amounts in much of western North America, which may increase access to winter ranges for mule deer; however, global warming patterns are projected to lead to loss of sagebrush winter ranges and increasing coniferous communities, which will ultimately reduce the range of habitat quality over winter (Lutz *et al.* 2003).

Autumn migration is highly variable among individual mule deer and associated with patterns of winter weather such as temperature and snowpack, as well as the individual life history characteristics (Monteith *et al.* 2011). On the other hand, spring migration is much more synchronous among individuals and seems to coincide with decreasing snow depth and advances in plant phenology (Monteith *et al.* 2011). A recent study of migrating ungulates and satellite imagery confirmed the green wave hypothesis, that mule deer track or "surf" the leading edge of spring green-up to maximize high quality forage intake (Merkle *et al.* 2016). The likely association between seasonal migration and weather conditions provides evidence that those migratory patterns may be altered by global climate change (Monteith *et al.* 2011). However, the apparent adaptive behavioral strategy of mule deer to time migration in response to environmental conditions may reduce the potential for migratory patterns to be mismatched with food availability, especially because climate change alters seasonal patterns of plant growth (Monteith *et al.* 2011).

Climate change may also indirectly affect mule deer through impacts on the spread of invasive species and diseases. Climate-induced changes could expose native plant communities to invasive plant species or exacerbate current invasive plant problems, which may alter range forage quality and fire regimes (see Section 4.1, *Invasive Annuals*) Generally, ecoregional differences in the impact to mule deer populations are expected to occur as climate change progresses (DeVos and McKinney 2007). Within the Northern Great Basin, expanded distribution of woody species, reduced availability of high quality winter forage, increased frequency of stand-converting wildfires, and spread of invasive plants and insects have

increased in the past 150 years, resulting in different biotic communities and interactions between species (Cox *et al.* 2009). Recent research on factors that influence distributions of biting midges, which serve as vectors of bluetongue and epizootic hemorrhagic disease, suggests that climate change could alter the distribution and incidence of mule deer diseases (Schmidtmann *et al.* 2011). Increased CO_2 concentrations could also potentially affect the nutritional quality and quantity of mule deer forage (DeVos and McKinney 2007). As global climate change progresses, the extent of these changes and altered biological interactions are expected to increase.

4.7.3   Geoprocessing Modeling and Analysis

4.7.3.1   Predicted Changes in Snow Water Equivalent

Changes in snow depth could affect the available mule deer winter habitat and alter the timing of spring-time tracking of the green-up. The previous REA evaluated snow levels in March to estimate the effects of snow levels on Mule Deer based on the following rationale.

- Mule deer densities on winter range are highest from January through March;

- Mule deer body reserve depletion (starvation, fat loss) is nearing its maximum at the end of winter (March);

- Mule deer begin migrating off winter range in late March and most have moved to transitional ranges by late May when fawns are born;

- Mule deer are most predation prone in March when they are in high density, poor body condition, and in their last trimester of gestation; and

- Mule deer move off winter range where snow melt exposes fresh green areas with deeper snow that will green up later and most likely not be used much as all the deer will follow the receding snowline uphill.

The changes in March snow water equivalent were evaluated for climate change impacts on mule deer winter and summer ranges. In addition, to analyze how changes in snow water equivalent may affect winter mule deer habitat, the estimated change in January SWE equivalent was also compared to the (WAFWA) winter habitat (Winter Range, Winter Concentrations, Year-round Habiat).

4.7.3.2   Predicted Changes in Precipitation

Precipitation amounts and timing affects the growth and quality of forage for mule deer as well as the distribution and movement of deer. Higher summer precipitation is expected to increase forage resources and could benefit the growth of mule deer fawns. Summer precipitation has been shown to be a limiting factor for Nevada mule deer population, with the summer precipitation totals explaining 80 percent of the variation in the mule deer population (Wasley 2004). Summer precipitation improves available autumn forage which is important for overwinter survival (Hurley *et al.* 2014). With the sensitivity of mule deer to summer precipitation, the changes in summer precipitation was compared to mule deer summer habitat in the ecoregion.

4.7.4   Results

The March SWE is expected to decline across the ecoregion due to climate change (Figure 4-37 and 4-38). The reduction in snowpack in March should improve conditions for mule deer in allowing mule deer to move off winter ranges sooner where snow melt exposes fresh green growth. Midwinter SWE (January) is also expected to decline slightly within or adjacent to mule deer winter ranges. The reduction of snowpack in the winter could expand the available habitat for mule deer winter ranges.

Based on the modeling projections, summer precipitation is expected to increase by 4 to 5 percent at mid-century (2040-2069) and 11 to 15 percent at the end of the century (2070-2099). Generally, the southern and southeastern portions of the ecoregion would experience the greatest precipitation increases and the northern portion the lowest precipitation increases (including minor decreases) in summer (See Appendix A: Figure A-6). Increased summer precipitation results in increased forage availability in autumn, which increases the body mass of mule deer fawns, thus improving the probability of winter survival (Hurley *et al.* 2014). However, increasingly severe droughts associated with warming temperatures will likely have a periodic negative impact on mule deer populations. Overall, the expected reduction in winter snowpack, especially in March, and the slight projected increase in summer precipitation due to climate change should have a favorable impact on the mule deer populations in the ecoregion.
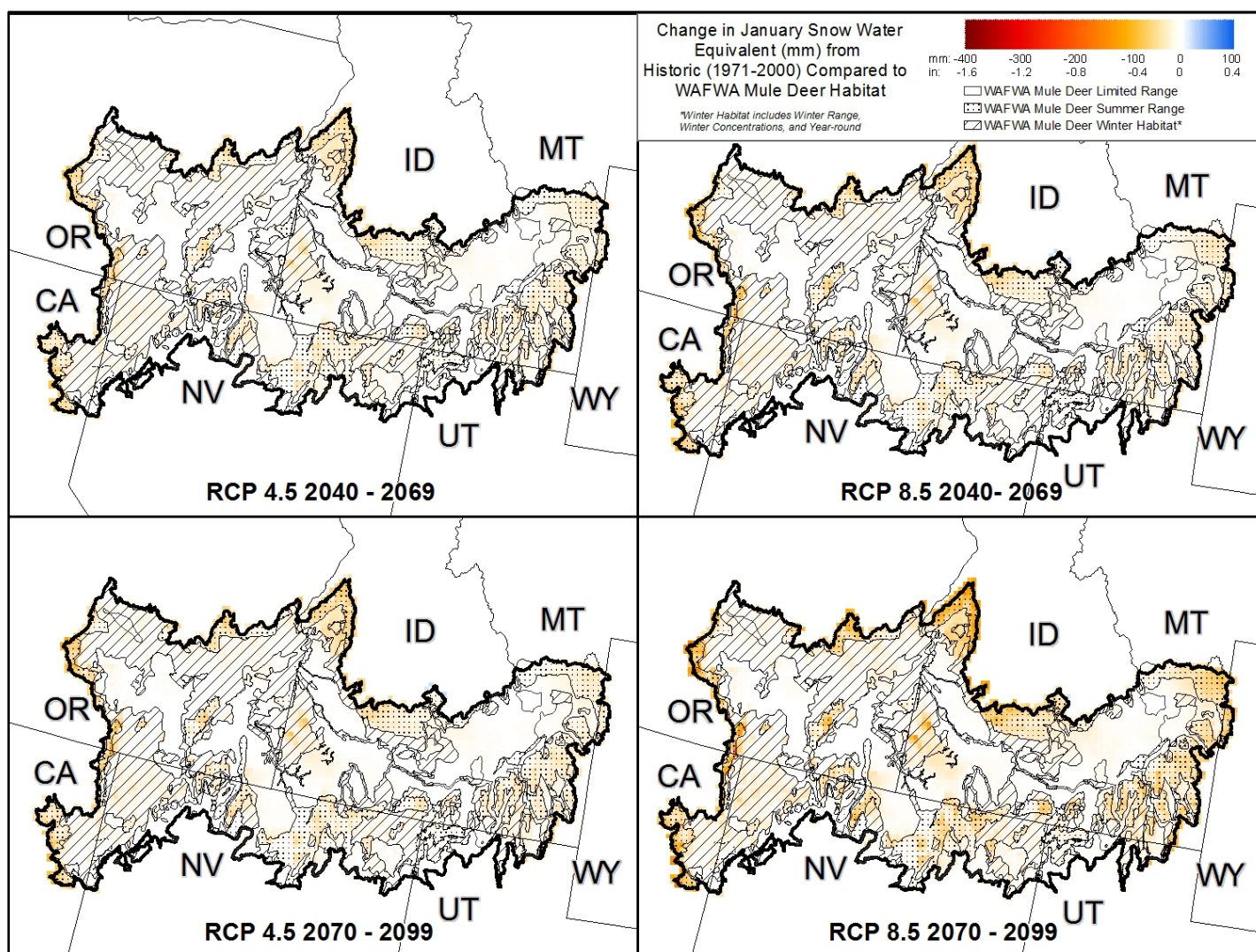
Figure 4-37. Projected Changes in January Snow Water Equivalent Compared to Winter Mule Deer Habitat
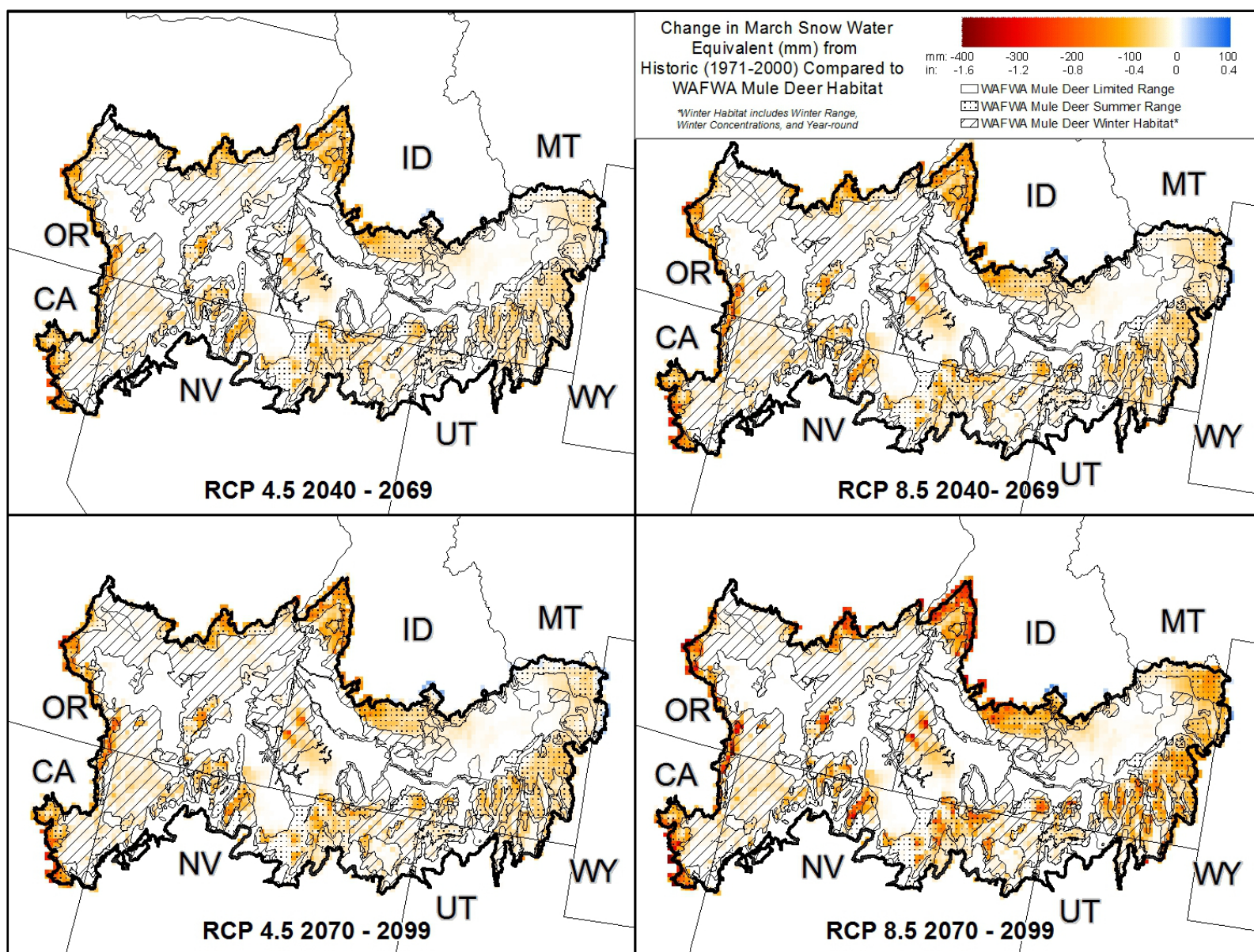
Figure 4-38. Projected Changes in March Snow Water Equivalent compared to Mule Deer Habitat

4.7.5 Mule Deer References

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4.8 Pronghorn

Pronghorn are adapted to a grassland and shrub-steppe ecosystem. Characteristics of good pronghorn habitat include large areas of open rangeland and relatively flat or undulating terrain with high visibility (NDOW 2003). Pronghorn are highly selective browsers varying food sources seasonally depending on the availability of vegetation. Forbs are the most preferred food source when available in spring and summer (Howard 1995). Feeding preferences for shrubs are intermediate and generally dominate winter diets (Bayless 1969). Big sagebrush (*Artemesia tridentata*), rabbitbrush (*Chrysothamnus spp.*), and bitterbrush (*Purshia tridentata*) are particularly important pronghorn forage in the Northern Great Basin ecoregion.

Typically, habitat for pronghorn consists of relatively flat, open native sagebrush and grassland habitats free of encroaching trees, fragmenting infrastructure (roads, fences, and oil/gas development), and other anthropogenic disturbances. Pronghorn strongly avoid forested habitats and they cannot jump over fences. Slope is also an important indicator of pronghorn habitat. Studies suggest that pronghorn avoid slopes of greater than 20 percent and prefer areas where the slopes are less than 10 percent (Yoakum 2004a; Longshore and Lowrey 2007). Snow depth above 15 inches (38 centimeters) appears to limit pronghorn use of winter range in some areas.

While pronghorn can migrate up to 170 miles in the Northern Great Basin, some pronghorn populations migrate shorter distances while some populations don't migrate at all (Sawyer *et al.* 2005; Feeney *et al.* 2004; Sawyer and Lindzey 2000). Non-migratory individuals have also been documented to stay on the winter ranges throughout the year (White *et al.* 2007). Adult pronghorn in the vicinity of the Idaho National Energy Laboratory in the eastern Snake River Plain (SRP) overwinter at the lower end of valleys on the edge of the SRP, moving variable distances (up to 40 miles [64 kilometers]) up the valleys during spring migration (Hoskinson and Tester 1980). Pronghorn in northeastern California may migrate up to 93 miles (150 kilometers) between summer and winter range (CDFG 2000). Pronghorn summer and winter ranges have been identified in other portions in the Northern Great Basin ecoregion (e.g., reported in BLM 2004, 2007), and state wildlife agencies have provided additional sources of distribution data for this CE.

4.8.1 Pronghorn Conceptual Model

Conceptual system models were developed as part of the pre-assessment phase to assist with the determination of the key factors that are important to pronghorn in the Northern Great Basin ecoregion. The model was developed to provide an ecological framework and justification for the choice of indicators that were used in assessing CA threats for this CE. This model incorporates the identified CAs, as well as potential effects from the actions of the CAs on both the landscape and local habitat/terrain levels (Figure 4-39).

The CAs considered for this CE analysis include development, climate change, wildfire and invasive species, and disease, which are depicted in brown boxes across the top of the model in Figure 4-39 and discussed in Table 4-14. The pronghorn conceptual model shows the pathway of CAs that affect the pronghorn landscape variables and thereby, the pronghorn functions and values, depicted in the lower box in Figure 4-39. The predicted results of CA effects are presented in blue boxes in Figure 4-39.

Because pronghorn require large home ranges, the conceptual model depicts effects on two spatial scales, the landscape level and the local habitat/terrain scale. This allows the model to focus site-level CA effects on localized CE habitat needs that may differ from landscape level effects. Examples of local terrain and habitat types recognized in the conceptual model include winter range, water sources, and areas with high visibility. Although the habitat requirements depicted in the Local Habitat/Terrain blue box in Figure 4-39

may be unmappable in some instances, they are included because they affect habitat choices for pronghorn. These include "High Visibility" areas preferred by pronghorn to be able to detect approaching predators and, conversely, areas that have visual obstructions such as trees or tall shrubs that are generally avoided by pronghorn.

Table 4-14. Change Agents and their Principal Effects on Pronghorn Antelope

Change Agent	Principal Effects on Pronghorn Antelope
Development	Most of the pronghorn range throughout the Northern Great Basin ecoregion is fragmented and affected by roads, agriculture, and development. The type of agriculture may have different effects on pronghorn. For example, irrigated hayfields are favored by pronghorn but other types, such as row crops, can adversely affect this species. Pronghorn are a shrubsteppe obligate species, and as sagebrush communities are converted, degraded, or altered, pronghorn populations are expected to decline. Pronghorn evolved in open landscapes without vertical barriers to movement. Fences often severely impede pronghorn movements (Spillet *et al.* 1967; Oakley and Riddle 1974; Mitchell 1980; Barrett 1982; Pyrah 1978; Hailey 1979). Unlike deer species, pronghorn can't jump over fences. There is strong evidence that, if prevented from seasonal migration by obstacles, pronghorn may experience massive die-offs (Ryder *et al.* 1984). Roads may also impair pronghorn access and use of winter range and seasonal movements. In southwestern Wyoming (Sheldon 2005) and in Arizona (Van Riper *et al.* 2001) unfenced roads appeared not to be a barrier to pronghorn movement, but the combination of heavy traffic volume (Buechner 1950) and fences along roads can become considerable barriers to pronghorn movement (Ockenfels *et al.* 2007). Divided, interstate, and other high-volume (i.e., >2,000 average annual daily traffic) highways are usually fenced to restrict. pronghorn movements to designated crossing structures. However, Yoakum (2004b) speculated that pronghorn behavior may prevent the use of under- and overpasses of high-volume highways. The recent expansion of energy development in the West also has the potential to have impacts on pronghorn and their long distance migration corridors (Hebblewhite 2008). Berger *et al.* (2006) found that some pronghorn continued to use areas that were heavily developed, whereas other individuals showed strong avoidance of such areas. Energy development resulted in avoidance of heavily developed areas by pronghorn and the total abandonment of the Jonah Field in Wyoming, which had previously been an important winter transition range (Beckmann et al. 2012). Sawyer *et al.* (2002) suggested that energy development could sever migration corridors for pronghorn and could influence the distribution of pronghorn on winter ranges. In the Northern Great Basin, the near-term future includes wind and solar energy development, not oil and gas. Wind energy creates roads and rotating turbine blades which could cause visual disturbance for pronghorn. However, a recent study in south-central Wyoming did not detect that wind energy development influenced pronghorn movement behavior, resource selection, or mortality risk (Taylor 2014). Some solar facilities remove large areas of land that could lead to habitat fragmentation.
Climate change	Climate change effects on big game species are primarily related to changes in vegetation communities, fire regimes, plant productivity, water availability, and the amount and persistence of snow pack affecting winter range. The climate change effects on pronghorn are discussed in more detail below in Section 4.8.2.
Wildfire and invasive species	Wildfire is considered one of the key factors affecting pronghorn migration and winter habitat. Moderate fire return intervals and mixed severity fires are necessary to maintain the mixed composition of sagebrush communities that provide the forage and open migration habitat pronghorn require. Heterogeneous, or mosaic, burn patterns can create better habitat than homogeneous burn patterns. Large homogenous burn patterns can result in type conversion from sagebrush to annual grassland. Aggressive fire suppression can cause densification of sagebrush conditions with loss of the understory grasses and forbs and increasing dead crowns,

Change Agent	Principal Effects on Pronghorn Antelope
	tree encroachment, and loss of the open character of the landscape. Pronghorn generally avoid trees and woodland habitats within 100 meters (Ockenfels *et al.* 1994; Yoakum 2004a). Wildfire is a major factor in the nutritional ecology and habitat dynamics for pronghorn. Pronghorn are adapted to a mosaic of age classes of sagebrush and other shrubs maintained by natural fire regimes, which tend to produce a reliable source of high quality browse and sufficient cover for fawns from predators. Due to cheatgrass invasion, improper grazing, homogenization of sagebrush structure, and increasing frequency of droughts, wildfires are now larger and occur more frequently, reducing habitat quality and quantity of sagebrush communities. With an increasing number of fires exceeding 100,000 acres during the last decade, fire is currently a major contributing factor to the transition of many shrubsteppe ecological states to grass dominated conditions (especially when coupled with cheatgrass invasions). For example, increasing cheatgrass cover is predicted for northwestern Nevada and other portions of the ecoregion with significant risk of invasion into some of the best and least-fragmented pronghorn habitat that remains in the ecoregion (See *Wildfire CA Package* from previous REA).
Disease	Hemorrhagic disease is caused by either of two closely related viruses, epizootic hemorrhagic disease (EHD) virus or bluetongue virus (SCWDS Group 2012). Because disease features produced by these viruses are indistinguishable, a general term, hemorrhagic disease, often is used when the specific virus responsible is unknown. These viruses are seasonal, transmitted by biting flies, and fatal within 24 hours after a 7- to 10-day incubation period. The wild ruminants have been infected include pronghorn, deer, and bighorn sheep. EHD was reported as severe in white-tailed deer populations in the central and eastern portions of Montana in 2011 (Pierce 2011). It is not certain at this time if invasive species and/or disease will play an important role in the analyses for pronghorn. These CAs can be explored in more detail and refined as the REA process moves forward.
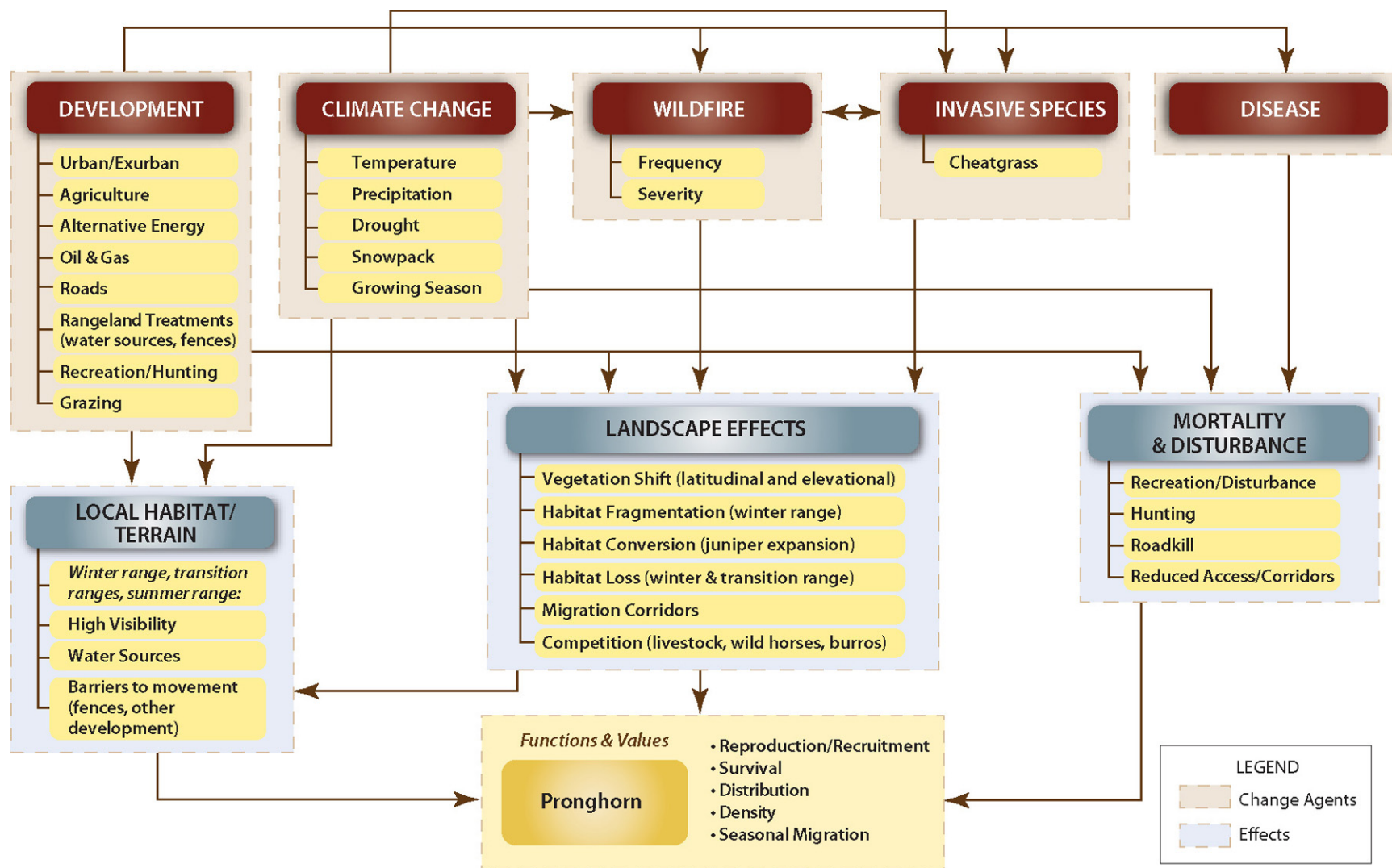
Figure 4-39. Pronghorn Conceptual Model

4.8.2 Climate Influences

Climate change effects on big game species are primarily related to plant productivity, water availability, the amount and persistence of snow pack, and changes in vegetation communities and fire regimes. The predicted changes associated with climate change for the Northern Great Basin pronghorn ranges include potential increases in temperature and changes in winter, spring, and summer precipitation, which would play a role in pronghorn fitness and reproductive success. Numerous studies have determined that fawn survival is one of the most significant concerns for the species and is directly correlated with timing, duration, and distribution of rainfall during winter and summer months (Byers and Hogg 1995; Fairbanks 1994; Gregg *et al.* 2001; Bright and Hervert 2005). Pronghorns are highly dependent on nutritious forage and undernourishment in females leads to an increase in births of underweight fawns, fawn mortality, and in extreme cases, could increase the likelihood of abortion during gestation (Hoffman et al. 2010). In addition, if spring or annual precipitation decreases and temperature increases, resulting in excessively dry conditions, the foraging quality of succulent forbs and grasses in spring and summer ranges would be detrimental to the health of pronghorn populations (Utah Division of Wildlife Resources 2009). Conversely, if spring precipitation increases it would lead to a more nutritious range of foraging options and increased access to water, resulting in decreases in both fawn and adult mortality (Bright and Hervert 2005).

Where this precipitation falls as snow, it may restrict pronghorn from moving between habitats and thus seasonal migrations. The greater the winter severity, the farther individuals and herds must travel to winter ranges with snow depths that can be tolerated by pronghorn (Creek 1967; Yoakum 1978; Guenzel 1986; Raper *et al.* 1989; Sawyer and Lindzey 2000). Generally, changes in climate and vegetative conditions trigger the onset and length of seasonal movements for pronghorn (Hoskinson 1980). If global climate change has an effect on timing of seasonal temperature changes, which is a trigger for pronghorn migration, this may affect initiation of migration. High accumulation of snow and below normal temperatures can result in pronghorn population losses. Deep snow restricts pronghorn access to winter forage causing individuals to perish from starvation (Smyser et al. 2006; Smyser 2005).

 Climate changes may also prompt significant shifts in vegetation, which will affect pronghorn populations due to their sensitivity to changes in foraging conditions (Prato 2009; Brown *et al.* 2006). Conifer encroachment could reduce the habitat visibility, which could over time affect the survival rates of fawns and adults (Richardson 2006). Increased early season precipitation in February and March and increased spring temperatures may also favor the spread of cheatgrass in pronghorn habitats, which may displace native bunchgrasses in shrubsteppe communities and exacerbate fire frequency and extent by providing more abundant continuous fuel sources during the dry summer months.

4.8.3 Geoprocessing Modeling and Analysis

Predicted Changes in Snow Water Equivalent

Snow depth data for the month of March was selected for analysis based on the importance of snowmelt and spring greenup to provide nutrition to late gestational does, and in restricting the amount of habitat and migration corridors that are available to migratory pronghorn. Most of the modeled pronghorn habitat in the REA had snow depths in the preferred range (less than 15 inches) with the exceptions of higher elevation mountainous regions. A reduction in spring snow depth may increase available pronghorn habitat in the higher elevation mountainous regions.

Predicted Changes in Precipitation

Precipitation amounts and timing affects the growth and quality of forage for pronghorn. Higher annual precipitation (especially in spring) is expected to increase forage resources and could benefit the growth of pronghorn (Bright and Hervert 2005; Utah Division of Wildlife Resources 2009).

Juniper Expansion

Pinyon-juniper expansion is a threat to the pronghorn. Woody brush encroachment could reduce the habitat visibility, which could over time affect the survival rates of fawns and adults (Richardson 2006). The portion of existing Northern Great Basin pronghorn habitat that is vulnerable to pinyon-juniper encroachment was estimated based on the MC2 modeling results (Figure 4-40).

4.8.4 Results

Based on the previous modeling of snow depth, most of the ecoregion had snow depths less than 15 inches in March. Climate change is expected to result in a reduction of snowpack in March which should slightly increase available pronghorn habitat in the higher elevation mountainous regions. The estimated total vegetation carbon by the MC2 modeling was compared to modeled pronghorn habitat. Higher annual precipitation (especially in spring) is expected to increase forage resources and could benefit the growth of pronghorn. Based on the modeling projections, spring precipitation is expected to increase by 6 to 9 percent at mid-century (2040-2069) and 7 to 9 percent at the end of the century (2070-2099). In addition, MC2 modeling predicts a substantial increase in total vegetation carbon. The total vegetation carbon increase is due in part to the estimated carbon dioxide fertilization effect assumed by the model. The MC2 model may overestimate the woody expansion due to the carbon dioxide fertilization effect by not accounting for limits in other nutrients (especially nitrogen). The increase growth of woodland species like juniper, would likely reduce the available pronghorn habitat as the species strongly avoids forested habitats due to the cover provided to its predators.
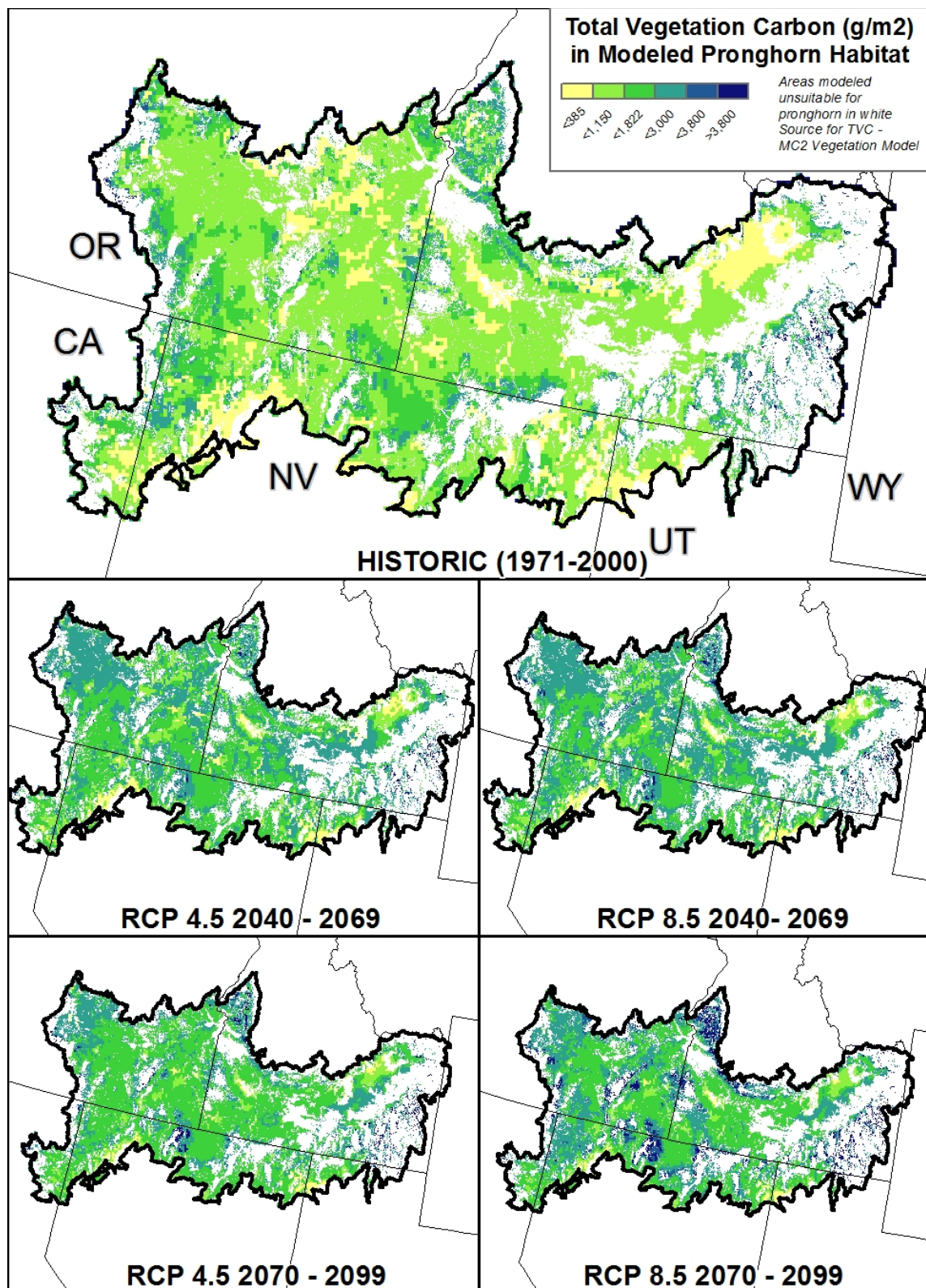
Figure 4-40. Total Vegetation Carbon in Modeled Pronghorn Habitat
Values of total vegetation carbon above 3,000 g/m2 have been classified as woodland in a previous southeastern Oregon study (Creutzberg et al. 2015). Higher total vegetation carbon values (especially above 3,000 g/m2) would indicate increasing woody vegetation which could negatively affect pronghorn.

4.8.5 Pronghorn References

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4.9 Greater Sage-grouse

The greater sage-grouse is a widespread sagebrush-obligate species that has become an icon and symbol for conserving sagebrush across the western U.S. Long-term population declines have resulted in greater sage-grouse being absent from much of their pre-Euro American settlement range (Schroeder *et al.* 2004). At the landscape scale, greater sage-grouse require large, interconnected expanses of sagebrush ecosystems, with varying densities, heights, and ages of sagebrush (Doherty *et al.* 2008). They also occur across a wide range of soil moisture and temperature regimes. Not all sagebrush habitats have been found to be suitable for greater sage-grouse. They generally don't appear to use very open sites with a high proportion of bare ground or exposed rock. Creutzberg *et al.* (2015) model simulations suggest that projected changes in climate may affect vegetation potential by increasing the amount of moist shrub steppe and causing periodic increases in xeric shrub steppe, where conditions are climatically unsuitable for sage-grouse. Greater sage-grouse are most closely linked to systems dominated by three subspecies of big sagebrush: little sagebrush, black sagebrush, and silver sagebrush (Miller *et al.* 2011), though in the Northern Great Basin silver sagebrush is less common.

Sagebrush steppe vegetation types vary in resilience to disturbance depending on the species or subspecies and site characteristics. Sagebrush systems as a whole are generally not considered resilient to frequent and large-scale disturbance (Davies *et al.* 2009). Silver sagebrush and threetip sagebrush (*Artemisia tripartite*) may resprout after fire but the other species that are important to greater sage-grouse are killed by fire and must regenerate from seed. Sagebrush species occurring on wetter, more productive sites (e.g., mountain big sagebrush) have greater ability to recover from disturbance than species or subspecies growing on drier, less productive sites (e.g., Wyoming big sagebrush) (Miller *et al.* 2014). Disturbance events result in alternate stable states (vegetation conditions) in many semiarid systems, as described in greater detail in the coarse-filter vegetation model for sagebrush (Section 4.2). Altering a native disturbance regime (e.g., fire frequency, drought and Aroga moth [Lepidoptera:Gelechiidae] outbreaks) may drive a sagebrush community across a threshold to another stable state (e.g., grassland, woodland) that is not suitable for greater sage-grouse.

There is considerable variation in migration distances among greater sage-grouse populations. Some migratory populations move large distances (often >20 kilometers [km]) between different seasonal habitats, and occupy large home ranges (>600 square kilometers [km2]). Life cycle components related to habitat (Connelly *et al.* 2011) include the following:

1 Lek sites are typically located in sparse to short grassland or human-made openings within sagebrush communities. Sagebrush immediately surrounding lek sites is used for feeding, resting, cover from weather, and security from predators.

2 Nesting habitat, usually 1.3 to 5.1 km from the lek, requires a sagebrush canopy that provides cover from predation and a healthy grass understory.

3 Early brood-rearing habitat is characterized by the chicks' requirements for escape cover (sagebrush canopy), and access to water and food resources (primarily arthropods and forbs).

4 Summer into fall during late brood-rearing, is when greater sage-grouse may shift to areas that support green vegetation, such as riparian habitats, springs and seeps, agricultural croplands, irrigated hayfields, sagebrush uplands, and high elevation meadows.

5 Winter habitat, occurs primarily where sagebrush is exposed above the snow. Exposed sagebrush is used for feed and cover; greater sage-grouse feed almost exclusively on sagebrush in the winter.

4.9.1 Greater Sage-grouse Conceptual Model

Conceptual system models were developed as part of the pre-assessment phase to assist with the determination of the key factors that are important to sage-grouse in the Northern Great Basin ecoregion. The model was developed to provide an ecological framework and justification for the choice of indicators that were used in assessing CA threats for this CE. This model incorporates the identified CAs, as well as potential effects from the actions of the CAs on both the landscape and local habitat/terrain levels (Figure 4-41).

The CAs considered for this CE analysis include development, climate change, invasive species, wildfire, and insects and disease, depicted in brown boxes across the top of the model (Table 4-15). As mentioned above, suitable greater sage-grouse habitat depends upon the stability of healthy sagebrush ecosystems. Because the details of transitions between sagebrush vegetation states are presented in the course filter sagebrush (Section 4.2), they are not repeated in the greater sage-grouse model. However, the greater sage-grouse system model does indicate the relationships between the CAs that act upon the greater sage-grouse habitat needs and thereby, on the greater sage-grouse functions and values, depicted in the lower box. The predicted results of CA effects are presented in blue boxes in Figure 4-41.

Table 4-15. Change Agents and their Principal Effects on Sage-grouse

Change Agent	Principal Effects on Sage-grouse
Development	In the last few decades, development, including infrastructure expansion (roads, pipelines, and transmission lines), mining, and establishment of wind energy facilities in proximity to greater sage-grouse leks and in winter habitat, have directly reduced the amount of suitable habitat available for greater sage-grouse and have introduced noise and human presence that may also have adverse effects (Hollaran 2005; Kaiser 2006; Aldridge and Boyce 2007; Doherty *et al.* 2008; Naugle *et al.* 2009; Harju *et al.* 2010). Historic conversion of sagebrush to pasture, cropland, or irrigated hayfields has been widely recognized as a dominant factor in the early declines of greater sage-grouse populations. Rangeland vegetation treatment practices are analyzed as a type of land development primarily because land conversions conducted to improve forage quality for livestock and in turn remove sagebrush adversely affect greater sage-grouse habitat quality. Current rangeland and grazing practices attempt to maintain adequate shrub cover while rejuvenating the understory component. Conflicts between land use changes and greater sage-grouse-occupied habitat remain high across the species' range and population expansion may only be possible on large protected areas such as public lands unaffected by development and private land conservation easements of sufficient size. Other development types included in the model include transmission lines or towers, which can both present strike hazards to flying greater sage-grouse and provide perches from which avian predators can hunt (Ellis 1987; Hall and Haney 1997; Braun 1998; Gilmer and Weihe 1977; Steenhof *et al.* 1993; Beck *et al.* 2006). Greater sage-grouse collisions with rangeland fences have also been documented (Christiansen 2009; Gruver 2009). Hydrological diversions and impoundments change the local hydrology and may affect the brood-rearing and/or summer/fall habitat. Human urban/exurban development can directly increase mortality, introduce pet predators into the environment, and indirectly affect greater sage-grouse by increasing noise and disturbance, which is often not tolerated by these noise sensitive species.
Climate change	Climate change that alters vegetation growing conditions, especially those plants important for greater sage-grouse, has the potential to directly change habitat availability and quality. Locations of habitats suitable for greater sage-grouse may change under future climate change scenarios. Predictions seem to be clearer for habitats at the extremes, such as those at highest elevations and northern latitudes. Drought conditions can also affect greater sage-grouse. Drought can occur when winter conditions have low precipitation, a combination of low

| Change Agent	Principal Effects on Sage-grouse
	precipitation and high temperatures, low snowpack, or when there are dry summer conditions (Bumbaco and Mote 2010). Climate effects are discussed in more detail below in Section 4.9.2.
Invasive species	Invasive species occurrences and fire history are often linked, as shown in the coarse filter sagebrush model (Section 4.2), and may contribute to an increase in pinyon-juniper woodlands, which corresponds to a decrease in greater sage-grouse habitat in the Intermountain West (Miller and Tausch 2001). Tree establishment within sagebrush communities generally decreases forb availability due to moisture depletion (Bates *et al.* 2000). The greatest threat in Wyoming big sage communities is invasion by annual grasses (e.g., cheatgrass, medusahead), because these fuels can increase the fire frequency from greater than 100 years to less than 10 years (Whisenant 1990). In addition, annual grasslands can persist for many decades with little recovery back towards native vegetation even in the absence of fire.
Wildfire	Under natural conditions, moderate fire return intervals and mosaic or heterogeneous burn patterns promote the mixed composition of sagebrush communities required by greater sage-grouse for leks, nesting, and brood rearing. However, the ecological role of fire has changed significantly. In conjunction with climate change and the expansion of invasive annual species, wildfire now covers larger areas more frequently, reducing habitat quality and quantity for greater sage-grouse (Connelly and Braun 1997; Connelly *et al.* 2000; Nelle *et al.* 2000; Fischer *et al.* 1996). At lower elevations on drier sites, more frequent wildfires cover large extents that have contributed to vegetation type conversion from sagebrush to invasive grass monocultures. Elsewhere, fire suppression has promoted expansion of juniper woodland into sagebrush sites. The predominant impacts of wildfire are expected to occur at the vegetation community level as sagebrush sites shift from one state to another with changes in disturbance regimes.
Insects and disease	Since 2002, the West Nile virus has been a source of mortality in greater sage-grouse, as it can simultaneously reduce juvenile, yearling, and adult survival (Walker and Naugle 2011). The West Nile virus has the greatest potential for population-level effects among all parasites and infectious diseases identified in greater sage-grouse (Christiansen and Tate 2011). It has been detected in 10 states and may result in persistent low-level mortality and severe outbreaks leading to local extinctions and/or regional population declines (Walker and Naugle 2011). Small, isolated populations on the fringe of the greater sage-grouse range would be most affected (eastern California), as an outbreak could reduce populations below a recoverable size. Larger low to mid elevation populations of greater sage-grouse, which are annually inflicted with West Nile virus in northern Nevada and southern Idaho, may absorb the impacts if population growth is still supported by quality habitat (Walker and Naugle 2011). West Nile virus incidence is thought to be related to a combination of biotic and abiotic factors, including the increase in available surface water (breeding sites for the West Nile virus mosquito vector) associated with irrigated agriculture (usually not on BLM lands) and livestock tanks and ponds. The risk of West Nile virus is expected to increase as temperatures increase with predicted climate change. Insects that affect sagebrush could also affect sage-grouse. For example, the Aroga moth (Lepidoptera:Gelechiidae), also known as the sagebrush defoliator, can cause disturbance to sagebrush communities which could result in decreased suitability for sage-grouse (Bolshakova and Evans 2014). In northern Utah, suitable habitats for Aroga moths include intermediate elevations (1,800 to 2,000 m), and abundance and feeding damage was highest on north-facing stands of sagebrush, characterized by lower values of incident solar radiation.
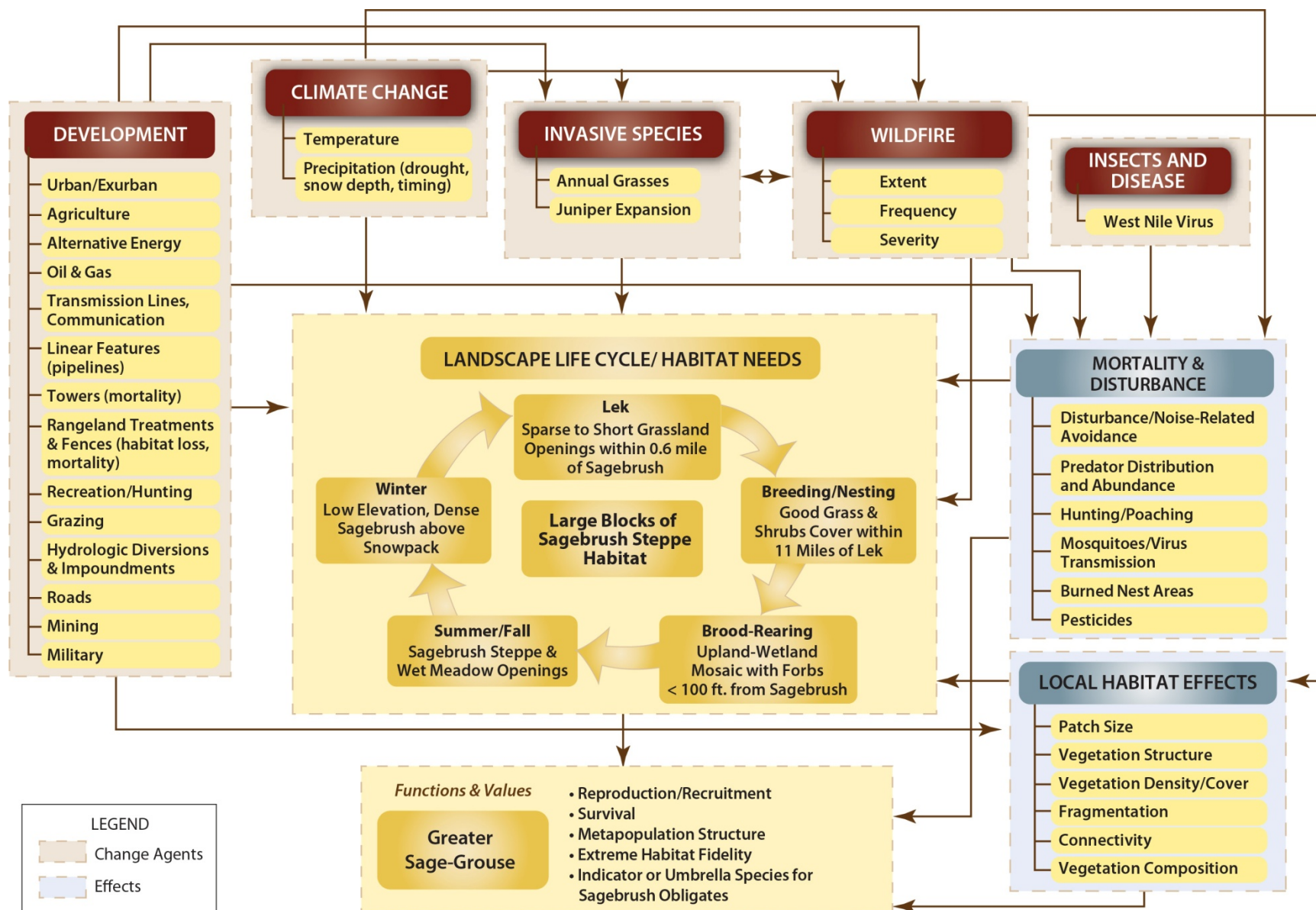
Figure 4-41. Greater Sage-grouse Conceptual Model

4.9.2 Climate Influences

Climate change poses a substantial long-term risk to sage-grouse, particularly through their direct link to sagebrush ecosystems (Bradley 2010; Schlaepfer *et al.* 2012; Homer *et al.* 2015; Creutzburg *et al.* 2015; Still and Richardson 2015). Generally, all range-wide modeling efforts predict a decline of sagebrush in the southern periphery of its range with expansion of sagebrush northward and at higher elevations (see Section 4.2). However, inconsistencies in climate projections and differences among model methodologies (e.g. bioclimatic envelope models, linear models, dynamic global vegetation models, and state-and-transition models) have resulted in variations in the predicted distribution of sagebrush.

For southeast Oregon, Creutzburg *et al.* (2015) integrated information from a dynamic global vegetation model (MC2), a sage-grouse habitat climate envelope model, and a state-and-transition simulation model. The study found that the rangeland condition will likely decline in the future due to the prevalence of exotic grasses and juniper on the landscape. Sage-grouse habitat (sagebrush steppe) is projected to decline in the first several decades but increase in area later in the century. In general, climate change impacts were more influential than management impacts in the eastern Oregon rangelands, with climate change effects having the potential to be both positive and negative for sagebrush steppe ecosystems. The dynamic vegetation modeling coupled with state-and transition models show that the prevalence of cheatgrass, the expansion of junipers, and the apparent recent trend toward larger and more frequent wildfires are interrelated factors that need to be taken into account when considering the effects of climate change on sagebrush communities (Creutzburg *et al.* 2015).

In another study in southwestern Wyoming, Homer *et al.* (2015), used remote sensing data (LANDSAT) to determine five sagebrush ecosystem components (bare ground, herbaceous, litter, sagebrush, and shrub) and then compared them to the daily precipitation records from 1984 to 2011. These results were used to create a linear model that examined the relationship between sagebrush abundance and precipitation which was then applied to future (2050) precipitation patterns. These future 2050 values were compared to a contemporary (circa 2006) greater sage-grouse habitat model. Overall, the two 2050 IPCC scenarios (A1B and A2 scenarios) predicted a 12 percent decrease of current sage-grouse nesting habitat, a 4 percent reduction of summer habitat, and less than one percent of new potential habitat gained from 2006 to 2050. It is important to note that this study is located east of the continental divide and these areas have a much higher proportion of the annual summer precipitation than does the Northern Great Basin region, thus plant responses will differ as a result. Nevertheless, similar to other studies (e.g. Bradley 2010; Still and Richardson 2015; Schlaepfer *et al.* 2012), peripheral sagebrush habitats will be less resilient to change.

Increasing temperatures associated with climate change could also result in the spread of the West Nile virus to higher elevations and an earlier onset of transmission (Naugle *et al.* 2005). Schrag *et al.* (2011) developed a West Nile virus transmission risk model based on predicted temperatures for Montana and Wyoming. The model predicted an overall trend toward increasing transmission risk during peak transmission season (June through August) (Schrag *et al.* 2011).

Over the last several decades there have been changes in types of extreme weather events. Heat waves have become more frequent and intense, especially in the West (Melillo *et al.* 2014). Drought and severe winters have been found to decrease survival of greater sage-grouse broods and adults (Holloran 2005; Moynahan *et al.* 2007; Aldridge and Boyce 2008). A mark and recapture study just south of the ecoregion in Eureka County found that climatic processes had a strong relationship with adult survival and recruitment. Blomberg *et al.* (2012) examined the characteristics of climate and landscape disturbance on greater sage-grouse populations. The variability in annual precipitation or snow depth explained as much as 75 percent of the variance in greater sage-grouse population size. Because of its lifespan, the sage-grouse is able to capitalize on favorable reproductive conditions during wetter than average rainfall (Blomberg *et al.* 2012). Overall, a changing climate with more frequent droughts would negatively impact sage-grouse recruitment and survival (Blomberg *et al.* 2012).

Climate change has the additional potential to exacerbate the spread of annual invasive plants, as well as woody plants such as juniper, displacing native sagebrush communities (Neilson *et al.* 2005). Greater sage-grouse recruitment was consistently low in areas with a substantial exotic grassland footprint even following years of favorable rainfall (Blomberg *et al.* 2012). Thus elevated temperatures due to climate change may increase the competitive ability of cheatgrass at higher elevations, expanding its range into sites where it currently is not widespread.

Climate change may also increase the spread of juniper at higher elevations due to increased precipitation in winter and spring and overall warmer temperatures (Neilson *et al.* 2005). Additionally, greater sage-grouse distribution patterns are influenced by conifers. Sage-grouse numbers were found to be reduced within 100 meters of pinyon-juniper due to increased predation from raptors perching on the trees/shrubs (Commons *et al.* 1999). Baruch-Mordo *et al.* (2013) reported that a 4 percent conifer cover would preclude leks within 1 km and Doherty *et al.* (2008) found that sage-grouse avoid conifers in habitats (i.e. on a 650 m^2 scale).

4.9.3 Geoprocessing Modeling and Analysis

Predicted Changes in Types of Sagebrush

At a landscape scale, greater sage-grouse require large, interconnected expanses of sagebrush ecosystems, with varying densities and heights of sagebrush cover across different soil moisture regimes (Doherty et al. 2008). Potential changes to sagebrush ecosystems are discussed in Section 4.2. To evaluate potential changes in sagebrush-steppe types which could impact sage-grouse, the potential changes in ecological types of sagebrush based on temperature (cold, cool, warm) and moisture (moist/dry) were evaluated using the thresholds provided Section 4.2.3 regarding sagebrush.

Predicted Changes in Juniper Expansion

Woody brush encroachment has been found to be detrimental to sage-grouse populations. The potential expansion of temperate evergreen woodlands throughout the ecoregion is discussed in Section 4.3. The portions of existing sage-grouse habitat that are vulnerable to pinyon juniper encroachment were estimated based on the MC2 modeling results and compared to sage-grouse habitat.

4.9.4 Results

The predicated change in sagebrush types based on the MACA-downscaled climate output estimates changes in precipitation and temperature which are provided in Table 4-16 and graphically displayed in Figure 4-43 and 4-44. In general, climate change would be expected to result in a shift from cool/dry and cool/moist regimes to warm/dry and warm/moist regimes which are less resilient and resistant to invasive grass invasion. In RCP 8.5, late period 2070-2100, some sagebrush areas would shift to hot/dry and hot/moist regimes. As no sagebrush in the ecoregion currently occur in hot/dry and hot/moist regimes, this could shift to a new vegetation type. Overall, consistent with the invasive grass analysis in Section 4.1, climate change would result in a shift in temperature/moisture regimes in greater sage-grouse habitat which would likely make the sagebrush communities and corresponding sage-grouse habitat less resilient to disturbance and less resistant to invasion from invasive annual grasses.

Based on the MC2 modeling, which factors in CO_2 fertilization effects, increasing CO_2 levels and climate change could result in an increase in temperate evergreen woodlands in the ecoregion in the place of sagebrush (Figure 4-45). This may result in increased juniper encroachment into sagebrush communities which can negatively affect sage-grouse populations. Overall, climate change would be expected to exacerbate the dual stresses of juniper encroachment and invasive annual grass expansion in existing sage-grouse habitat.

Table 4-16. Projected Temperature/Moisture Regimes in Greater sage-grouse Habitat

GSG Habitat	Time Slice	Temperature/Moisture Regime					
		Cool and Dry	Cool and Moist	Warm and Dry	Warm and Moist	Hot and Dry	Hot and Moist
All Habitat	Current	33.7%	35.8%	21.4%	9.1%		
Priority	Current	36.1%	43.4%	13.1%	7.4%		
General	Current	31.1%	27.1%	30.9%	11.0%		
All Habitat	Mid RCP 4.5	0.1%	6.3%	40.6%	53.0%		
Priority	Mid RCP 4.5	0.2%	6.7%	31.5%	61.6%		
General	Mid RCP 4.5	0.1%	5.8%	50.9%	43.3%		
All Habitat	Mid RCP 8.5	0.0%	2.5%	34.7%	62.7%		
Priority	Mid RCP 8.5	0.0%	2.3%	25.1%	72.6%		
General	Mid RCP 8.5		2.7%	45.7%	51.6%		
All Habitat	Long RCP 4.5	0.0%	3.0%	34.5%	62.5%		
Priority	Long RCP 4.5	0.0%	2.8%	25.0%	72.2%		
General	Long RCP 4.5		3.3%	45.2%	51.5%		
All Habitat	Long RCP 8.5		0.1%	15.2%	78.7%	3.1%	2.9%
Priority	Long RCP 8.5		0.1%	9.4%	88.7%	0.9%	1.0%
General	Long RCP 8.5		0.1%	21.7%	67.4%	5.7%	5.1%
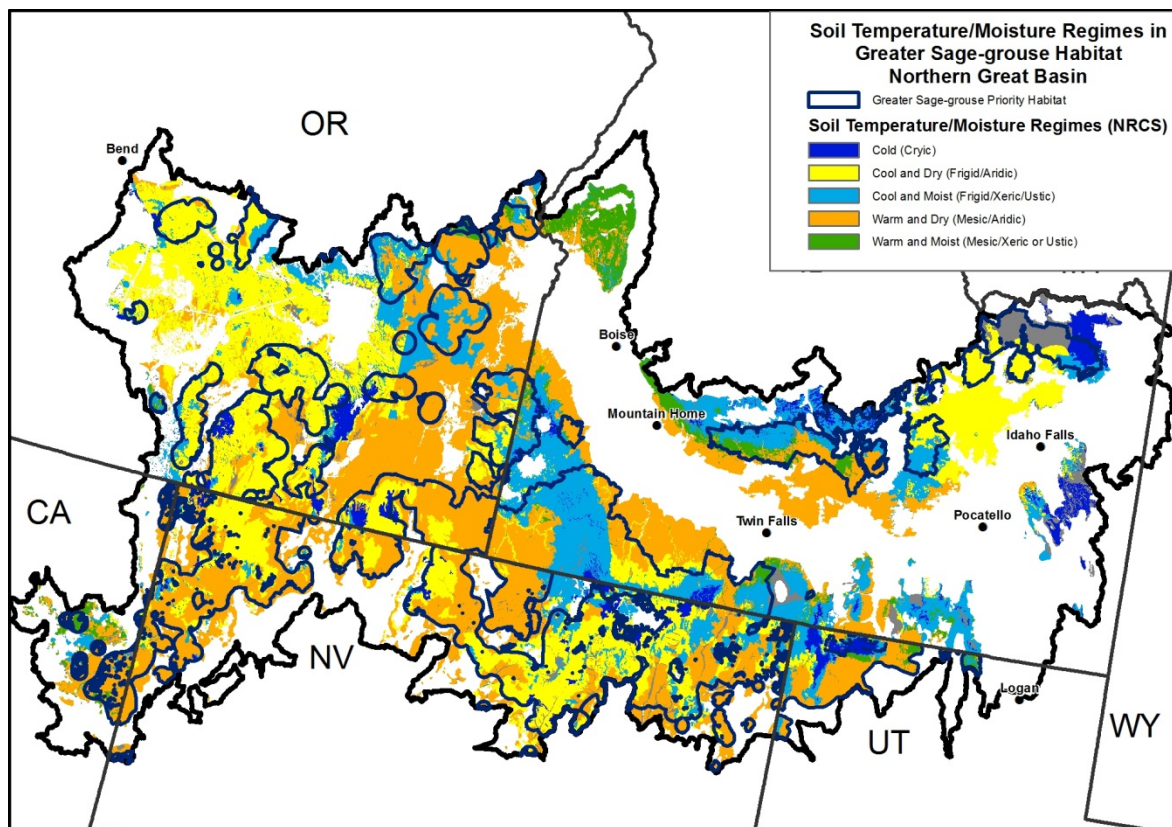
Figure 4-42. NRCS Soil Temperature/Moisture Regimes in Greater Sage-grouse Habitat
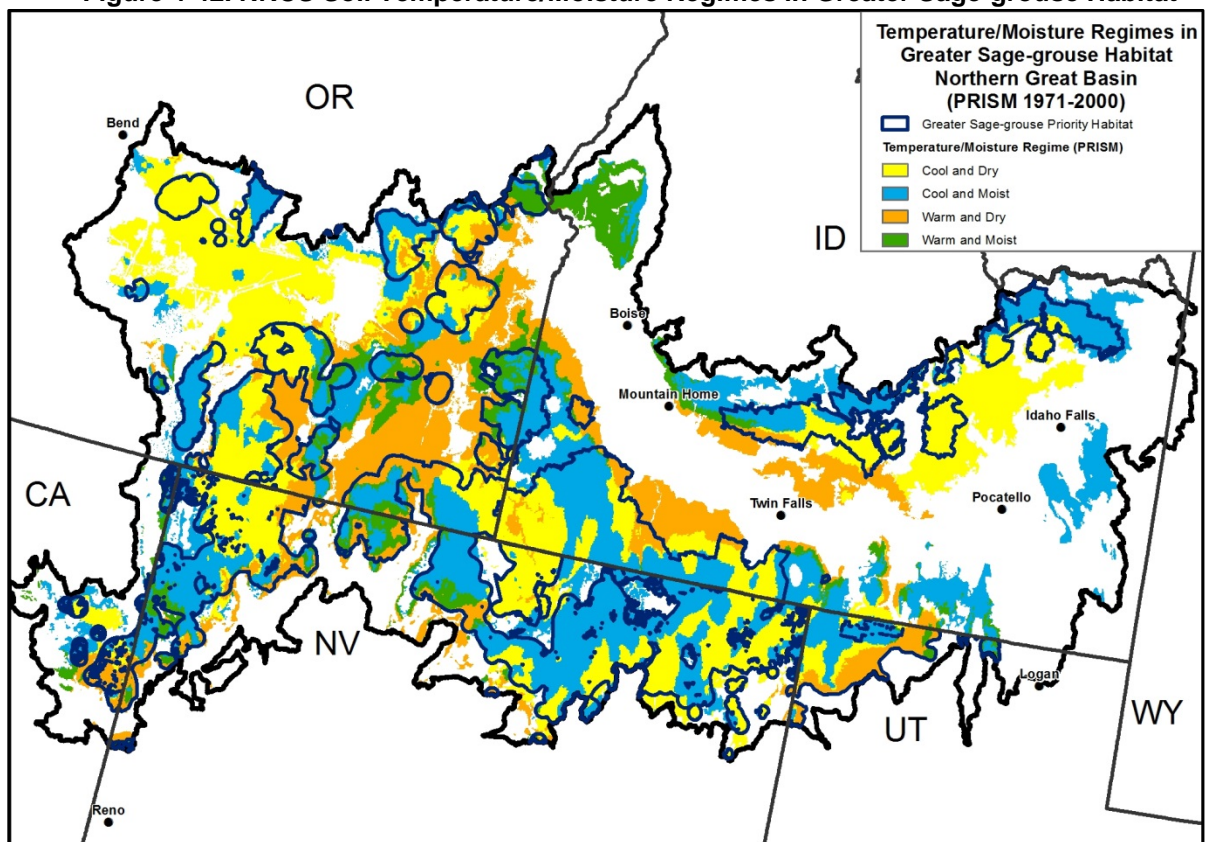
Figure 4-43. PRISM-based Temperature/Moisture Regimes in Greater Sage-grouse Habitat
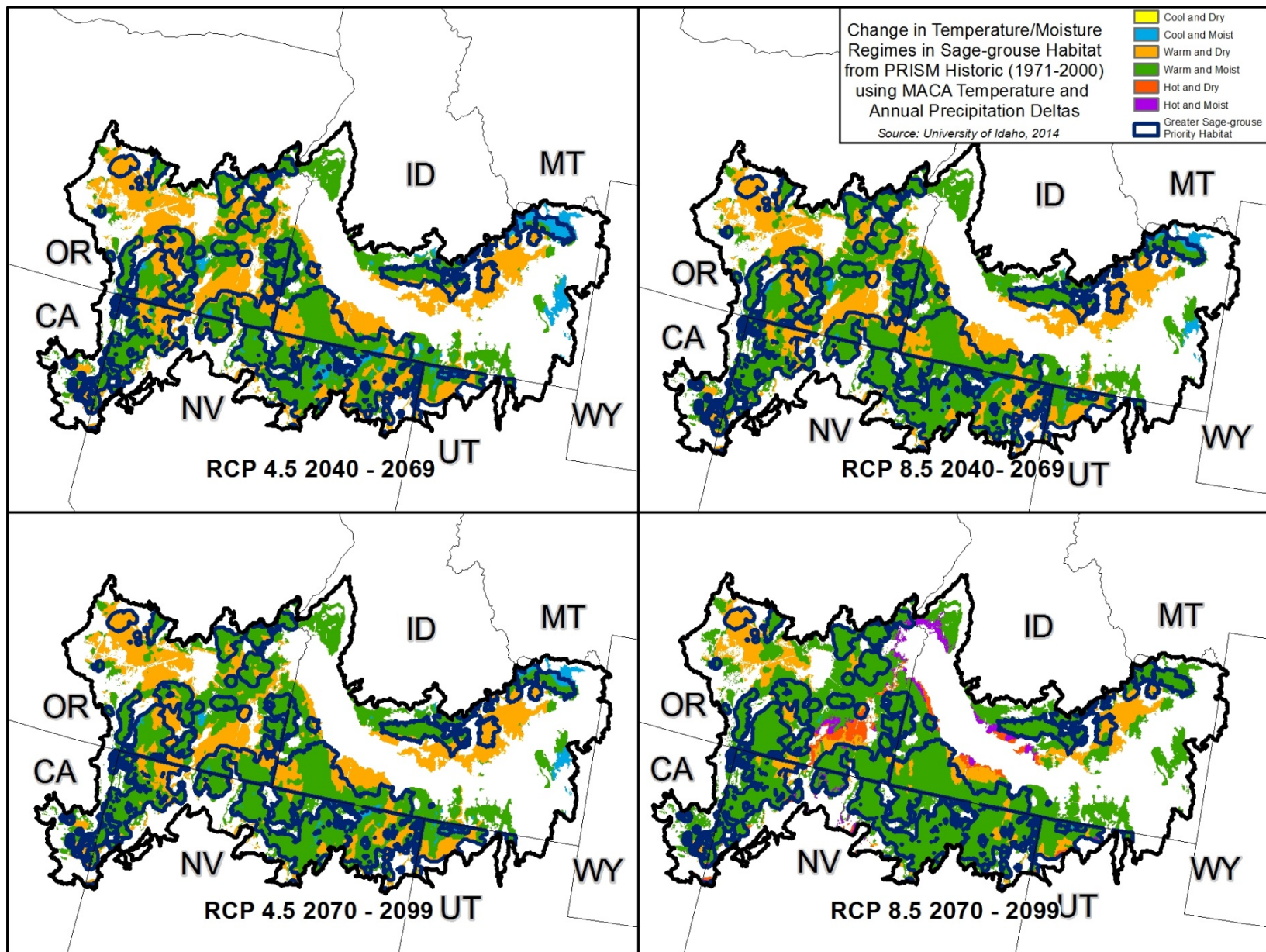
Figure 4-44. Projected Temperature/Moisture Regimes in Greater Sage-grouse Habitat
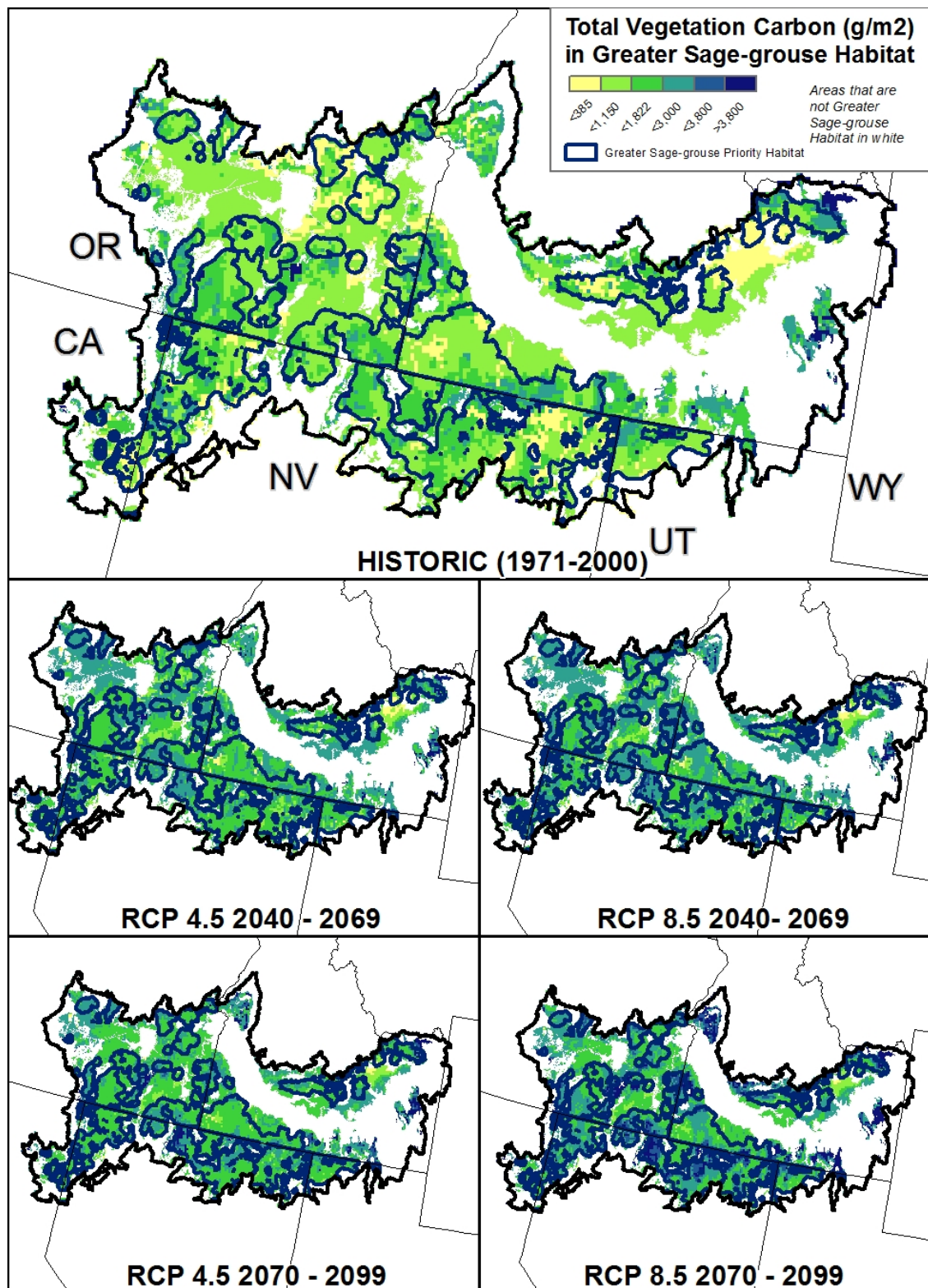
Figure 4-45. Projected Total Vegetation Carbon in Greater Sage-grouse Habitat

Values of total vegetation carbon above 3,000 g/m2 have been classified as woodland in a previous southeastern Oregon study (Creutzberg et al. 2015). Higher values (especially above 3,000 g/m2) would indicate increasing woody vegetation which could negatively affect sage-grouse.

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4.10 Pygmy Rabbit

The pygmy rabbit (*Brachylagus idahoensis*) is the smallest rabbit species in North America and occupies sagebrush-steppe communities within the Great Basin and adjacent Intermountain West. Pygmy rabbits are found at elevations ranging from about 1,500 to 2,600 meters and are active year round. The distribution of the species is widespread but populations are disjunct within a large geographic range, which roughly stretches from southeastern Oregon, through central Nevada, to western Utah, into southern Idaho and is also found in Washington, California, Montana, and Wyoming. A Pygmy rabbit's home range is extremely variable according to season, sex, and location. In the winter, pygmy rabbits tend to stay close to their burrows (within 98 feet) and during summer males have been documented to range across 69.7 acres, though more commonly between approximately 4 and 25 acres (USFWS 2010). The pre-Euro-American range of the pygmy rabbit occurred over a larger area than today. Multiple factors, including invasive species, frequent fires, improper grazing, conversion of land to agriculture, and energy development amongst others, have contributed to recent declines in both quantity and quality of sagebrush habitats required by pygmy rabbits (Edgel *et al.* 2014).

The pygmy rabbit is a sagebrush obligate and relies year-round on big sagebrush (*Artemisia tridentata*) for food (51 to 99 percent of the diet) and cover from thermal extremes and predators (Crawford 2008; Gabler *et al.* 2001). Several studies have identified the presence of taller, denser stands of big sagebrush, relative to surrounding unused areas, as an essential feature of pygmy rabbit habitat (Crawford 2008; Larrucea and Brussard 2008a; Rachlow and Svancara 2006; Hagar and Lienkaemper 2007; Gabler *et al.* 2001). Habitat selection appears to be based on a complex of vegetation and soil characteristics (Gabler *et al.* 2001). Important vegetative characteristics include composition (i.e., sagebrush), horizontal and vertical shrub cover, shrub height (greater than 65 cm), and deep (greater than 60 cm) friable soils (Edgel 2013; Larrucea & Brussard 2008a; Hagar and Lienkaemper 2007; Weiss & Verts 1984; Roberts 2001). For example, in Utah, sites occupied by pygmy rabbit had greater amounts of horizontal obscurity, were located at higher elevations, had a greater percentage of sagebrush and shrub understory, and had less decadent sagebrush than unoccupied sites (Edgel 2013). Because pygmy rabbits excavate burrow systems, soil characteristics including depth and texture also may influence their selection of habitat (Weiss and Verts 1984). Pygmy rabbits prefer soils that consist of mostly sand with some silt and clay, which provides ease of burrow excavation and minimizes burrow collapse (Schmalz *et al.* 2014; Hagar and Lienkaemper 2007).

4.10.1 Pygmy Rabbit Conceptual Model

A conceptual system model was developed to determine the key factors that are important to the life cycles of the pygmy rabbit in the Northern Great Basin ecoregion. The model was developed to provide an ecological framework and justification for the choice of indicators that were used in assessing CA threats for this CE. The conceptual model depicts the important habitat components required throughout the year for the pygmy rabbit and incorporates the identified CAs, as well as potential effects from the actions of the CAs, on both the landscape and local habitat levels (Figure 4-46).

The CAs considered for this CE analysis include development, climate change, invasive species, and wildfire, depicted in brown boxes across the top of the model (Table 4-17; Figure 4-46). Additional CAs may act on pygmy rabbit but have not been included in the model due to lack of sufficient information. These include disease, such as the plague, and a genetic bottleneck due to the low genetic diversity in some areas. As mentioned in Section 4.10, pygmy rabbit habitat depends upon the stability of healthy big sagebrush ecosystems. Section 4.2 (sagebrush) details the transitions between sagebrush vegetation states, which are not repeated in in this section. However, the pygmy rabbit model does indicate the relationships between the CAs that act upon pygmy rabbit habitat and thereby the functions and values supporting

pygmy rabbit, which are depicted in the lower box. The predicted results of CA effects are presented in blue boxes in Figure 4-46.

Table 4-17. Change Agents and their Principal Effects on Pygmy Rabbit

Change Agent	Principal Effects on Pygmy Rabbit
Development	The various types of human development identified in the model affect important pygmy rabbit habitat requirements as indicated in the Local Habitat Effects box (Figure 4-46), including changes in patch size, vegetation density and cover, vegetation height or composition, soil structure, and increased fragmentation of suitable habitat patches. Large, land-intensive developments especially affect the natural patchy distribution of suitable sagebrush communities resulting in changes to pygmy rabbit behavior, movements, genetic exchange among populations, and feeding habits (Crawford 2008). As indicated by the Mortality and Disturbance box (Figure 4-46), competition with other species has been identified as an issue for pygmy rabbits; for example, suitable habitat occupancy decreased with presence of cottontails (*Syviligus* spp.) (Larrucea and Brussard 2008b). These studies also found that an increase in cheatgrass (invasive grass) reduced pygmy rabbit occupancy. Maintaining connectivity between patches of adequate size was found to be of great importance for pygmy rabbit populations studied in southwest Idaho (Burak 2006). Any of the CAs that fragment habitat patches and limit successful rabbit dispersal among patches would have adverse effects, potentially leading to local extirpations. Pierce *et al.* (2011) found that fragmentation (edge habitat) resulted in increased competitors (cottontail and jackrabbit [*Lepus californicus*]). Additionally, predation, particularly by mammals (e.g. coyotes, red foxes, badgers, bobcats) is a major source of mortality for pygmy rabbits and could increase as a result of fragmentation (Pierce *et al.* 2011; Crawford 2008). However, little information is currently available, especially at the landscape level, to assess effects of fragmentation on pygmy rabbit habitat use and population dynamics, including genetic analysis of metapopulations (Hagar and Lienkaemper 2007). For this reason, the model includes several local habitat effects factors that can be evaluated to determine effects to pygmy rabbits (Figure 4-46). Large-scale habitat fragmentation through agricultural development over the last 200 years in southern Idaho has reduced by at least 20 percent into what now can be considered three separate sub-populations (Roberts 2003). These islands of habitat support continuous big sagebrush and connectivity is still rated as good. A broad belt of dry-land and irrigated farms now form a travel barrier for rabbits along the Snake River from Ashton to Mountain Home, which separates populations that exist to the north and south of the river (Roberts 2003). Future development may affect the suitable habitat quality and availability for the pygmy rabbit. These include urban/exurban expansion, agriculture (especially when land conversion occurs), alternative and traditional energy exploration and development, and linear features (especially pipelines that disrupt vegetation and soil structure).
Climate change	Climate change effects on small mammal species are primarily related to changes in vegetation communities, fire regimes, and the amount and persistence of snow pack affecting snow cover. Climate effects are discussed in more detail below in Section 4.10.2.
Invasive species	The presence of pygmy rabbits has been found to be negatively associated with increased occurrence of cheatgrass (Larrucea and Brussard 2008a). Additionally, invasive annual grasses indirectly influence pygmy rabbit habitat because an increasing dominance of invasive annuals produces fuel for wildfire and facilitates shorter fire return intervals. Wildfire can alter the habitat and create soil conditions vulnerable to invasive species, particularly by cheatgrass and medusahead, which will continue to alter the fire regime and reduce persistent shrubby vegetative cover required by pygmy rabbits. The potential expansion of conifers beyond their historic range could indirectly affect pygmy rabbit. For example, Utah and western juniper have expanded rapidly over the past 140 years into sagebrush steppe, often converting these areas

| Change Agent	Principal Effects on Pygmy Rabbit
	to woodlands, thus reducing sagebrush habitat suitable for pygmy rabbits (Soule and Knapp 1999).
Wildfire	Wildfire covers large areas and occurs frequently, reducing habitat quality and quantity of mature sagebrush communities used by pygmy rabbits. Increased fire frequency reduces the time available between burns for sagebrush to fully mature, which modifies the habitat and availability between burn events. The increase in fire frequency in the West within the last century poses serious threats to pygmy rabbit persistence (Gabler *et al.* 2001; Roberts 2003). Burn pattern is also important for pygmy rabbit habitat such that a mosaic burn pattern could retain suitable pygmy rabbit habitat. One of the three current sub-populations in Idaho (on lands south of the Snake River) may possibly be considered isolated and fragmented with an uncertain future. The occupied habitat for another sub-population (in Owyhee County) has been severely burned at lower elevations but higher elevations appear to be relatively intact (Roberts 2003).
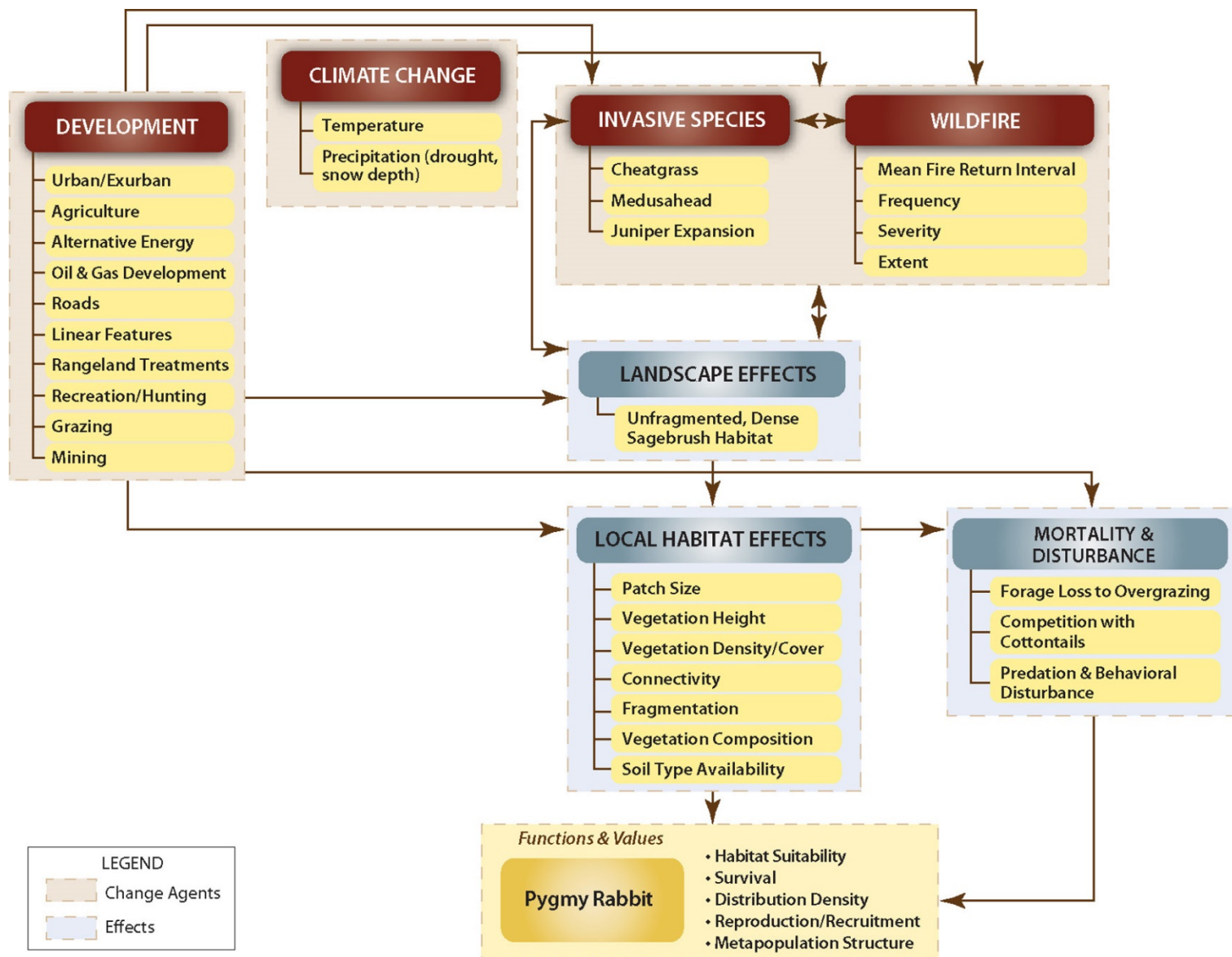
Figure 4-46. Pygmy Rabbit Conceptual Model

4.10.2 Climate Influences

Climate change effects on small mammal species are primarily related to changes in vegetation communities, fire regimes, and the amount and persistence of snow pack affecting snow cover. Increased temperatures and decreased precipitation over the last several decades has led to the decline in both quantity and quality of sagebrush-steppe habitats required by pygmy rabbits (Grayson 2006). Based on historical records from the Great Basin, warmer and drier conditions could cause a decline in the range of the pygmy rabbit (USFWS 2010). The pygmy rabbit is highly sensitive to decreases in density and increases in fragmentation of their sagebrush habitat (Himes and Drohan 2007). Recent climate change trends, along with changes in fire return intervals especially fire suppression policies, has resulted in pinyon pine and juniper invasion into sagebrush habitat at higher elevations. This vegetation shift is undesirable to the pygmy rabbit regardless of the extent of the invasion. Larrucea and Brussard (2008b) found that the presence of even a few trees provided a perch or sufficient cover for avian and mammal predators, and thus resulted in the absence of pygmy rabbits all together.

The range of pygmy rabbit has contracted due to multiple factors at different scales at both higher and lower elevations. At higher elevations there has been a decrease in the availability of habitat, particularly due to pinyon and juniper invasion into sagebrush communities. This encroachment is believed to have the potential to force a shift in pygmy rabbit populations from the foothills into lower elevations, which previously contained meadow-like vegetation but overtime has included more sagebrush species. However at lower elevations, warming temperatures have attributed to a decrease in snowpack (Larrucea and Brussard 2008b). Pygmy rabbits prefer to inhabit areas with a longer duration of snowpack that provides cover from predation during the winter, and increasing temperatures have caused less snow for the rabbits to use for building tunnels to access sagebrush forage. Pygmy rabbits have been extirpated from numerous sites at lower elevations, and are now inhabiting elevations averaging 157 meters higher than before, where they can retain the desired snowpack cover (Larrucea and Brussard 2008b). Moreover, the combined processes of pinyon-juniper encroachment at higher elevations and increasing temperatures with reduced snowpack at lower elevations may result in a heavy loss of pygmy rabbit habitat (Larrucea and Brussard 2008b).

Besides these range shifts, the greatest threat from climate change to the pygmy rabbit is the increased severity of fires with the potential of periodically destroying suitable habitat. Historically, fires burned more frequently in smaller areas and at much lower intensities. Recently, due to past fire exclusion policies and the increase in invasive species such as cheatgrass, which provide an effective fuel source, fires are much larger and more intense (Menakins *et al.* 2002). Fires have severely burned large areas of land, making it difficult for sagebrush communities to recolonize and have since consumed 16 percent of historical pygmy rabbit habitat (Larrucea and Brussard 2008b).

Invasive species occurrences and fire history are often linked, as shown in the coarse filter sagebrush model (Section 4.2). The presence of pygmy rabbits has been found to be negatively associated with increased occurrence of cheatgrass (Larrucea and Brussard 2008a). Cheatgrass roots can form dense mats that may make burrowing difficult for pygmy rabbits (Larrucea and Brussard 2008a). Cheatgrass is also unpalatable and indigestible except for a short period in spring (Larrucea and Brussard 2008a). Cheatgrass-dominated areas also can create a barrier to dispersal as the physical structure providing protection from sagebrush is lost (Larrucea and Brussard 2008b). Additionally, the rapid spread of other invasive grasses such as medusahead and ventenata also pose a threat to pygmy rabbit habitats.

4.10.3 Geoprocessing Modeling and Analysis

4.10.3.1 Predicted Changes in Sagebrush

Pygmy rabbit require expanses of tall, dense sagebrush. Potential changes to sagebrush ecosystems are discussed in Section 4.2 and 4.8. To evaluate potential changes in sagebrush-steppe types which could impact pygmy rabbit, the potential changes predicted for the sagebrush steppe will be compared to existing pygmy rabbit habitat for the ecoregion.

4.10.3.2 Predicted Changes in Temperature and Changes in Snowpack

Warmer temperatures reduce snowpack, which provides predatory protection for the pygmy rabbit in the winter. Without snowpack there is a higher predatory pressure on the pygmy rabbit in the winter. Changes in temperature and snowpack due to climate change will be evaluated using the predicted change in winter temperatures and estimated changes in snow water equivalent for the ecoregion.

4.10.3.3 Predicted Changes in Vegetation

Pinyon-juniper expansion is a threat to the pygmy rabbit. Trees provide perches for avian predators and shade and cover for mammalian predators (Larrucea and Brussard 2008b). The portions of existing pygmy habitat that is vulnerable to pinyon juniper encroachment will be estimated based on the MC2 modeling results.

4.10.3.4 Predicted Increase in Fraction of Area Burned

More homogeneous burn patterns can also have an effect on pygmy rabbit habitat by eliminating sagebrush communities over large areas.

4.10.4 Results

Pygmy rabbit habitat is generally also habitat for the greater sage-grouse. As discussed in Section 4.8, sagebrush communities are threatened by invasive annual grasses and juniper expansion. Overall, climate change would be expected to exacerbate the dual stresses of juniper encroachment and invasive annual grass expansion in existing pygmy rabbit habitat. The temperature/moisture regimes will likely shift from cool/moist sagebrush communities to warm/moist communities, which will make them less resilient and resistant and could alter the species composition of the communities. Based on the MC2 modeling, climate change could result in an increase in temperate evergreen woodlands in the ecoregion in the place of sagebrush (See Section 4.2 and 4.3). The potential increase in junipers provides roost for predators in pygmy rabbit habitat. The areas of pygmy rabbit habitat where expansion may have the highest potential are shown in Figure 4-47. Warming temperatures will also reduce the snowpack in the ecoregion. The reduction in March SWE in pygmy rabbit habitat is shown in Figure 4-48. With less snow cover, predatory protection in the winter may decrease. The areas burned by wildfire are also likely to increase with the warming temperatures. The projected fraction of area burned in pygmy rabbit habitat is shown in Figure 4-49. With more wildfires expected, less snowpack, and continued expansion of juniper into sagebrush communities, pygmy rabbit habitat will likely continue to contract due to the effects from climate change.
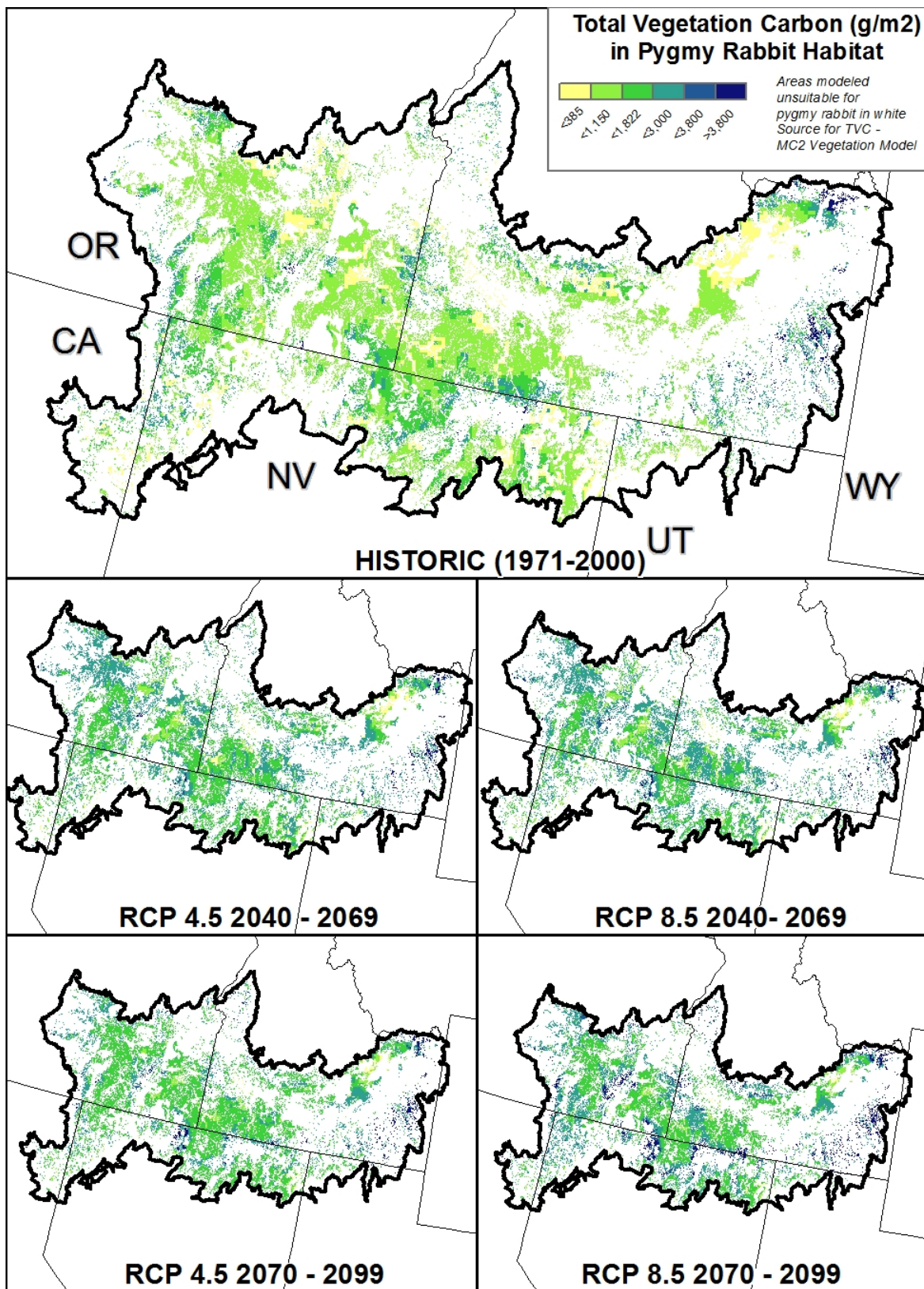
Figure 4-47. Total Vegetation Carbon in Pygmy Rabbit Habitat

Values of total vegetation carbon above 3,000 g/m2 have been classified as woodland in a previous southeastern Oregon study (Creutzberg et al. 2015). Higher values (especially above 3,000 g/m2) would indicate increasing woody vegetation which could negatively affect sage-pygmy rabbit.

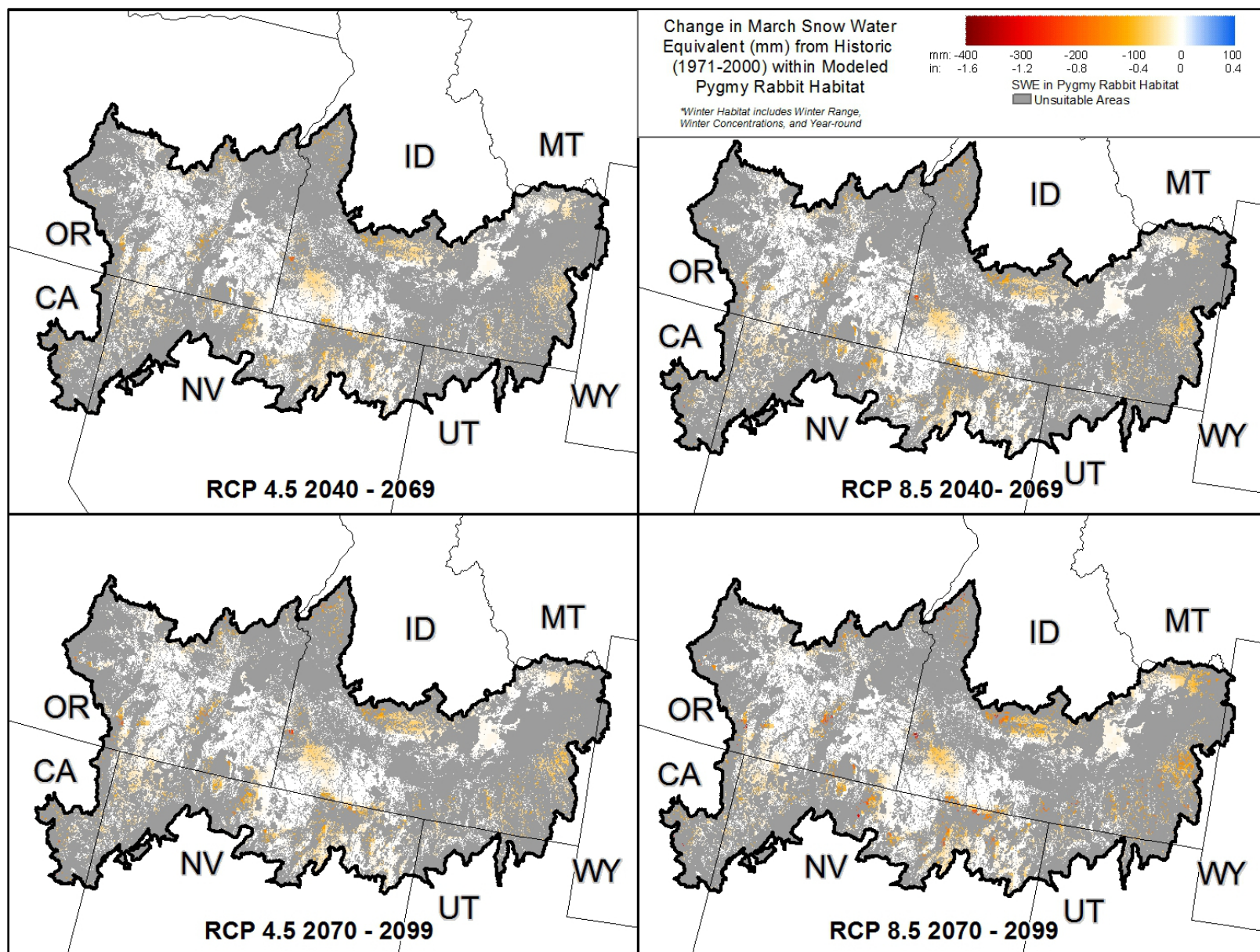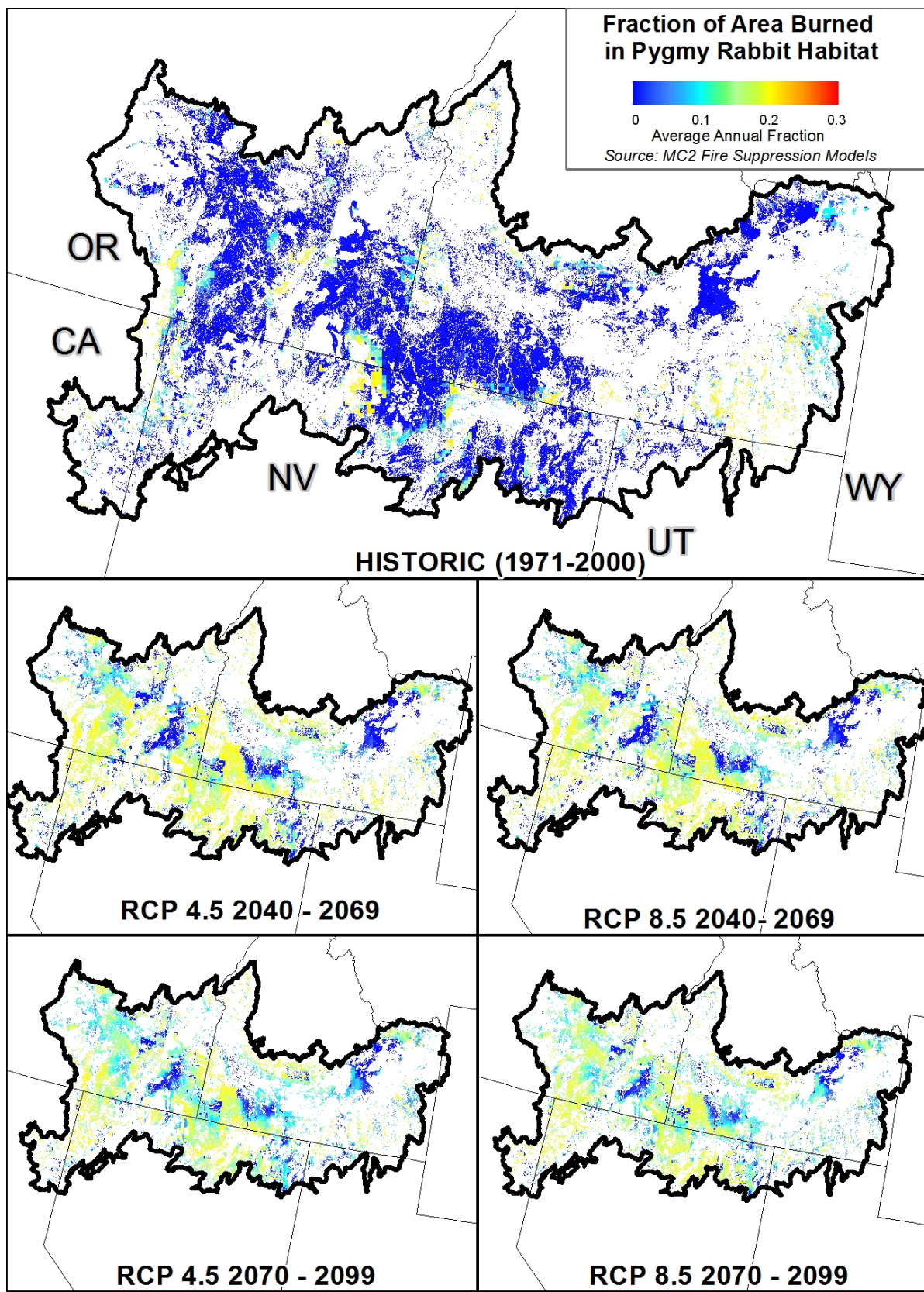**Figure 4-48. Change in March Snow Water Equivalent in Pygmy Rabbit Habitat**

Figure 4-49. Fraction of Area Burned in Pygmy Rabbit Habitat

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4.11 Coldwater Fish

The coldwater fish assemblage for the Northern Great Basin ecoregion includes bull trout, redband trout, mountain whitefish, Lahontan cutthroat trout, and Yellowstone cutthroat trout. These species were selected to represent the assemblage due to their sensitivity to changes in hydrology and habitat quality, in addition to hybridization, competition, and predation pressures associated with introduced species. Summary descriptions of the coldwater fish species are provided in Table 4-18 based on the *Coldwater Fish and Bull Trout CE Packages* provided in Appendix B of the 2011 REA.

Table 4-18. Coldwater Fish Species

Species	Description
Bull trout	Bull trout (*Salvelinus confluentus*), an Endangered Species Act threatened species, currently occurs in less than half of its historic range. Of all salmonids, due to its sensitivity to environmental conditions, bull trout are considered excellent indicators of water quality. Bull trout have the most specific habitat requirements of salmonids, including the "Four Cs": Cold, Clean, Complex, and Connected habitat. Bull trout require colder water temperature than most salmonids, very clean stream substrates for spawning and rearing, and complex and connected habitats, including streams with riffles, deep pools, undercut banks, and lots of large logs for rearing, annual spawning, and feeding migrations.
Redband trout	Non-anadromous rainbow trout (*Oncorhynchus mykiss*) that are found primarily east of the Cascade Mountains in the U.S. are often called redband trout (*O. m. gairdneri*). Redband trout like cool temperatures in clean and clear waters. As the range over which this subspecies occurs includes reaches that become frozen over, these fish require deeper pools in which to overwinter. Mature redband trout, typically age 3 or older, prefer to spawn in riffle or end of pool tail-out habitats that provide gravels free of fine-grained sediment to ensure proper oxygenation of their eggs (Behnke 2002).
Mountain whitefish	Mountain whitefish (*Prosopium williamsoni*) are a broadly distributed, long-lived (up to 11 years) salmonid native to western North America rivers, streams, and lakes (Wydoski and Whitney 2003). In general their mouths are smaller than most salmonids, with prey resources primarily consisting of adult and larval aquatic insects. However, larger mountain whitefish also prey on crayfish, leeches, and occasionally small fish (Wydoski and Whitney 2003). Though slightly more tolerant of warmer water temperatures and higher turbidity than cutthroat trout, mountain whitefish prefer large, deep, clear, cold rivers (Behnke 2002). The general in-stream temperature in habitats where mountain whitefish occur generally ranges from 48 to 52 °F (8.9 to 11.1 °C).
Lahontan cutthroat trout	Historically, Lahontan cutthroat trout (*Oncorhynchus clarkii henshawi*) were found in a wide variety of cold-water habitats including large, terminal, alkaline lakes, slow, meandering rivers, mountain rivers, and small headwater tributary streams. Generally, Lahontan cutthroat trout occur in cool flowing water with available cover of well-vegetated and stable stream banks in healthy riparian zones. Lahontan cutthroat trout prefer areas where there are stream velocity breaks, and in relatively silt-free, rocky riffle-run areas (USFWS 1995).
Yellowstone cutthroat trout	The Yellowstone cutthroat trout (*Oncorhynchus clarki bouvieri*) was historically native to the Yellowstone river drainage in south-central Montana and Wyoming, and in the Snake River drainage in Wyoming, Idaho, Utah, and Nevada (Gresswell 1995). At present, 91 percent of the current range lies within the boundary of Yellowstone National Park.

4.11.1 Coldwater Fish Conceptual Model

Conceptual system models were developed to provide an ecological framework and justification for the choice of indicators that were used in assessing CA threats for this CE. This model incorporates the identified CAs (Table 4-19), as well as potential effects from the actions of the CAs on both the landscape and local habitat/terrain levels (Figure 4-50).

The primary CAs for the coldwater fish species are identified across the top of the figure in red and their effects on habitat functions important to this species are identified in gray boxes below (Figure 4-50). The CAs that are key to the distribution and status of these species include development, climate change, wildfire, invasive species, and disease. The functions and values of aquatic systems for coldwater fish include habitat suitability, connectivity (i.e., fish passage), and maintenance of the genetic integrity of coldwater fish populations. The CAs may significantly affect these functions and values in the Northern Great Basin ecoregion for coldwater fish by affecting habitat suitability (i.e., water quality and quantity, channel configuration, aquatic and riparian habitat features, hydrologic regimes, presence of non-native fish species and diseases, and physical barriers to movement). For example, human development and resource use may degrade water quality, create barriers to fish movement among habitats, reduce availability of water, and introduce non-native fish that hybridize with, compete with, or prey upon native coldwater fish.

Table 4-19. Change Agents and their Principal Effects on Coldwater Fish

Change Agent	Principal Effects on Coldwater Fish
Development	Dams, improperly placed or sized culverts, irrigation diversions, and other migration barriers have negatively affected individuals and habitat and interfered with metapopulation dynamics. Large dams influence the hydrograph and water temperature, support non-native fish, and completely fragment native fish populations. Coldwater releases from large dams may provide refugia for coldwater fish. Culverts and smaller diversions may be seasonal barriers, fragmenting habitat and reducing availability of water. As a result, populations have become increasingly fragmented, with reduced resiliency in the face of environmental changes. Surface and groundwater extraction for urban and exurban populations, agricultural irrigation, and industrial development adversely affect native coldwater fish populations by reducing base flows with resulting increases in stream temperature and fragmented habitat. Poorly screened intakes can result in stranding small or young fish in ditches and fields.
Climate change	Reduced snowpack, water temperature changes, precipitation changes (timing and amount), and greater fluctuations in stream hydrographs will likely be significant stressors on native coldwater fish species for spawning, rearing, and other life cycle needs. Climate effects are discussed in more detail below in Section 4.11.2.
Wildfire	Wildfire affects aquatic habitats and biota through water quality changes including sedimentation and debris flows. Significant changes in the size and intensity of wildfires in Northern Great Basin communities due to suppression practices or climate change are a concern for aquatic species. Climate change will increase the likelihood of wildfires in the presence of fuels and ignition sources in relation to the timing of snowmelt (Haak *et al.* 2010), depending on the vegetation community and fire history of the area. Dunham *et al.* (2003a) indicated that an increase in wildfire prevalence would likely result in a corresponding decrease of riparian habitat function and benefit to the associated stream or river system, and has been considered a disturbance that may provide an ecological advantage for non-native fish species. Large, severe wildfire can contribute to increases in stream temperature and sedimentation attributable to riparian vegetation loss. In sufficient quantities, the additional sediment can degrade water quality conditions and smother spawning habitats and eggs. Historically, fish populations have

| Change Agent	Principal Effects on Coldwater Fish
	been able to recolonize after a fire in most aquatic systems (Gresswell 1999), but dams, diversions, and culverts have fragmented many populations, making recolonization of recovered aquatic systems difficult and in some cases impossible. Additionally, with predicted stream temperature increases in the future, thermal barriers may also hamper recolonization.
Invasive species	The introduction of non-native fishes in the waters occupied by native salmonid populations has led to hybridization of native redband trout and Yellowstone cutthroat trout with non-native rainbow trout. Additionally, competition with non-native fish species and predation by non-native fish species are among the primary concerns for persistence of native trout species. The impact of hybridization of redband trout with hatchery rainbow trout has become so problematic that it has necessitated monitoring of genetic purity within systems in which pure redband trout still occur (May *et al.* 2012). Introduced fish species such as channel catfish, smallmouth bass, and walleye likely influence population dynamics and distribution of the coldwater fish assemblage through competition, habitat degradation, and predation. Distribution and abundance of these introduced species is likely to increase with warmer water temperatures.
Disease	All species of trout and salmon may become infected with the parasite responsible for whirling disease (*Myxobolus cerebralis*), an introduced disease agent that was first identified in the U.S. in 1956 and is now present in Idaho (Idaho Invasive Species Council Technical Committee 2007) and other states in the Northern Great Basin (Elwell *et al.* 2010). The presence of the parasite does not always cause dramatic population losses, but can be a serious problem in hatcheries and has had severe impacts on some wild trout populations (Whirling Disease Initiative 2011). Infectious hematopoietic necrosis virus (IHN) and other pathogens affect salmonids and other hosts, and require continued monitoring within hatchery systems. The significance of disease as a CA for native coldwater fishes is unknown at present but is included in the conceptual model due to the potential for spread of pathogens from hatchery facilities into habitats of wild salmonid populations.
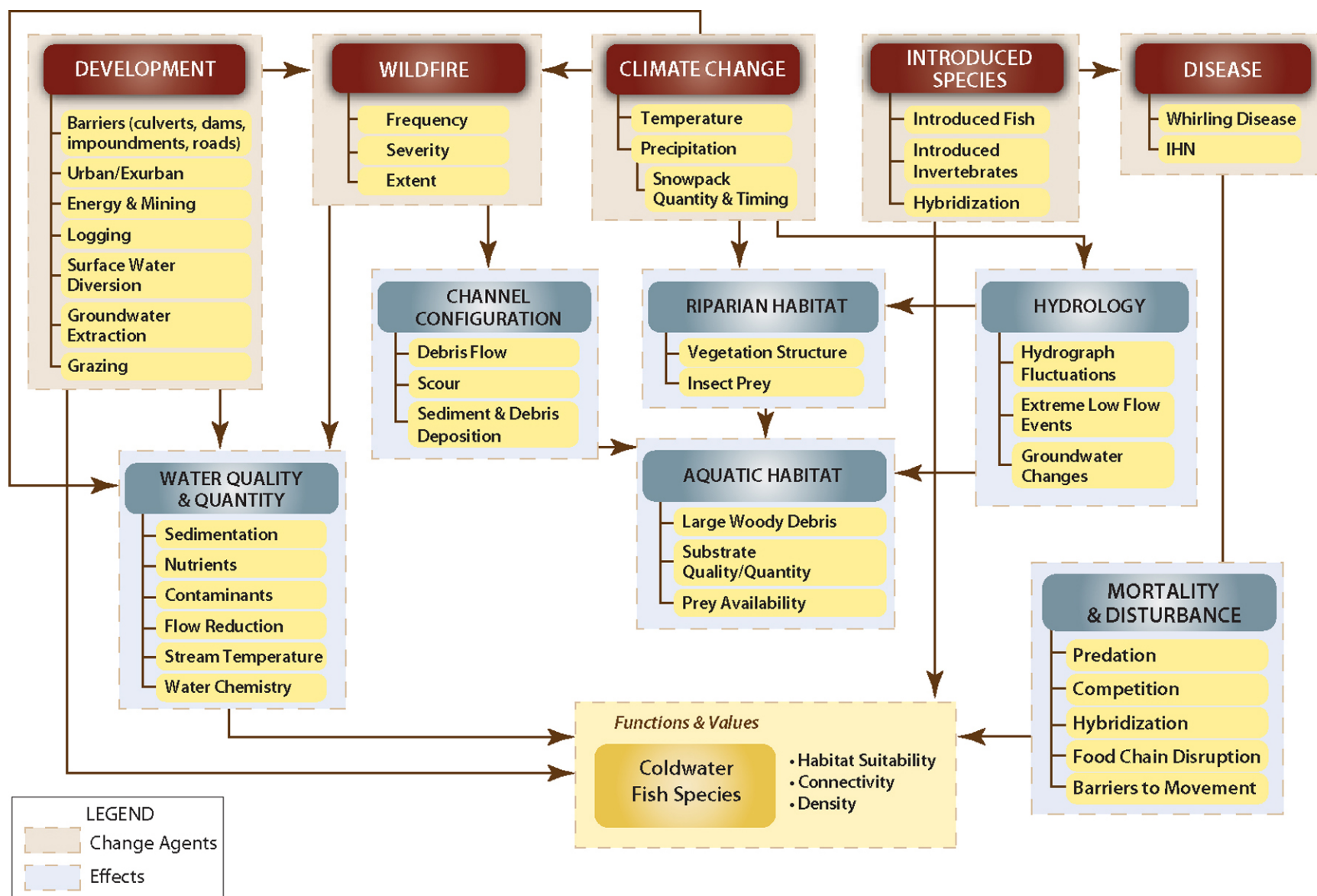
Figure 4-50. Coldwater Fish Conceptual Model

4.11.2 Climate Influences

The U.S. Bureau of Reclamation (USBR 2010) summarized literature on the past and projected effects of climate change on hydrology and water resources in western states with implications for resource objectives featured in the Bureau of Reclamation planning processes. The report suggests that future climate conditions featuring water temperature changes, precipitation increasingly occurring as rain rather than snow, reduced snowpack and earlier snowmelt runoff, and greater fluctuations in stream hydrographs combined with increased colonization of non-natives will likely be significant stressors on native coldwater fish species.

4.11.2.1 Increased water temperature

Reduced snowpack is expected to increase stream temperature variability. Colder groundwater and spring discharges entering streams provide habitat for coldwater fish species in many locations in the Northern Great Basin; for example, redband trout use pools where temperature is moderated by groundwater inflow (Zoellick 1999), but these effects are site-specific and cannot be adequately modeled with available information.

The distribution of coldwater fish is influenced by temperature, both directly (through avoidance behavior or mortality at extremes of thermal tolerances) and indirectly (by affecting physiological processes such as growth) (Brinkman *et al.* 2013). Each freshwater fish species is likely to respond to climate-driven changes differently, depending to a great extent on the species' thermal tolerances during different life stages and their ability to move from thermally unsuitable habitat to suitable habitat. In general, younger life stages have a narrower range of thermal tolerance than older life stages, although that narrower range is usually within the adult's range. Thus, different life stages may use different stream segments to maximize growth rates, emphasizing the importance of availability of the range of suitable habitats and connectivity between them. Overall, climate change is expected to reduce the extent of water courses habitable by native coldwater fish species (Isaak *et al.* 2015) and shift their distributions northward and to higher elevations (Ficke *et al.* 2007; Rahel *et al.* 2008).

Concerns for coldwater fish species are illustrated in the recent literature. Lahontan cutthroat trout, bull trout, and mountain whitefish all have a similar narrow thermal tolerance range, particularly during early fall spawning periods (Dunham *et al.* 2003b; Brinkman 2013), and climate change will probably shrink thermally suitable habitat for these species (Rieman *et al.* 2007; Wenger *et al.* 2011). Mountain whitefish may face proportionately greater habitat loss from climate change because they occupy larger, lower elevation streams where water temperature is expected to rise above thermal tolerances, and they have difficulty expanding into colder, smaller, high-gradient headwater streams, where cutthroat trout and bull trout populations find refuge (Brinkman *et al.* 2013). Warmer stream temperatures may facilitate invasion by nonnative predators and competitors that flourish under warmer temperature regimes (Hughes and Herlihy 2012; Rahel *et al.* 2008). Due to the bull trout's greater dependence on cold water than the other coldwater fish species in the ecoregion, a reduction in the availability of colder water habitats represents a particularly significant risk of reducing suitable habitat conditions for this species. High certainty projections showed that the majority of bull trout habitat is going to become unsuitable by the year 2080 (Wenger *et al.* 2013). Numerous studies found that bull trout occupation is declining, specifically at lower elevation sites with warm temperatures (Rieman *et al.* 2007; Eby *et al.* 2014), and confirm that bull trout are abandoning sites within their historical range and retreating to higher elevation sites as water temperature increases. As coldwater fish species migrate to higher elevations, however, they may find less suitable habitat because precipitation changes may result in decreased headwater stream flow (Luce *et al.* 2013).

The importance of thermal refugia is illustrated by physiological studies of desert and montane populations of redband trout. Redband trout demonstrated a thermal tolerance when exposed to high

temperatures for short intervals, but only as long as there was sufficient time to recover in cooler temperatures (Wehrly *et al.* 2007; Cassinelli and Moffitt 2010). The presence of cool water refugia created by subsurface flows or stratification of low flowing pools is important for physiological recovery when periods of low stream flow and excessive heating occur during the summer months.

4.11.2.2 Reduced streamflow

The USBR report (2010) also predicted decreased summer streamflows of up to 26 percent relative to the historic average. This reduction would amplify the required mitigation for USACE mandated flows and increase the competition between water users. In addition, this decrease in streamflow can become detrimental to the coldwater fish populations that are abandoning sites within their historical range and retreating to higher elevation sites as water temperature increases, because precipitation studies have found that the fish species attempting to migrate to higher elevations may find less suitable habitat at higher site locations due to the decrease in headwater streamflow (Luce *et al.* 2013).

Water quantity issues associated with climate change include effects of persistent severe drought and impacts on recruitment due to sudden runoff events during hatching and emergence of larvae (USBR 2010). Extreme low flows during severe droughts decrease survival of adults due to decreased volume of water, associated increases in water temperatures, increased susceptibility to predation, and reduced ability to move to more favorable habitats. Truncated stream habitats isolate fish near cold-water headwaters and may reduce the genetic diversity of populations through the loss of a migratory population or, because the population is so reduced, through the lack of immigration/emigration within the population. Loss of genetic thermal tolerance in small populations is a significant concern. For example, numerous studies have been conducted to evaluate resilience and responses of redband trout to typical summertime temperature cycles. Narum *et al.* (2013) found that redband trout populations with high genetic diversity have evolved adaptive heat shock responses to be able to withstand increasing summer temperatures. However, the study emphasized their concern for small, isolated populations that may not contain enough genetic diversity to adapt to changing climates. These small populations may be at risk of extinction due to small size, although these remaining cold-water streams could also act as climate-safe and invasion-resistant refuge habitats for native salmonids (Isaak and Young 2014).

The whitefish has been observed preferring large streams (>15 meters) with connectivity to large river habitats to complete their lifecycle (Meyer *et al.* 2009; Benjamin *et al.* 2014). Due to predicted changes in temperature and discharge caused by climate change, the migratory patterns of the mountain whitefish may also become altered when streamflow decreases (Benjamin *et al.* 2014).

4.11.2.3 Other Effects

Reduced snowpack increases stream temperature variability and under climate change, the snowpack will be replaced by periodic rain events as the climate gets warmer. These events would contribute to an increase in fine sediment deposition in small tributaries, warmer instream temperatures, and declines in streamflow. The input of finer-grained sediments can bury eggs and fry, and reduce the oxygenation of bull trout eggs, effectively decreasing bull trout egg survivability. As populations are already isolated by migrational barriers, maintaining suitable spawning habitat is essential to avoid additional declines and local coldwater fish extinctions (Rieman *et al.* 1997).

The continued colonization of native salmonid habitat by non-native species is expected to continue, and possibly be exacerbated by warming stream temperatures with the result of increased competition and hybridization with non-natives. In areas where native coldwater fishes may have a narrow range of thermal tolerance, many introduced salmonids are successful across a wide range of thermalscapes. For example, although the Yellowstone cutthroat trout has displayed some adaptive potential to changing

climatic conditions, they may have a narrower range of thermal tolerance than do introduced species such as the brook trout and rainbow trout. This means any increases to coldwater fish growth potential (with future warmer stream temperatures) could be offset by temperature-mediated competition with non-natives, as well as the continued colonization of these non-natives within the native fishes range (Al-Chokhachey *et al*. 2013).

4.11.3 Geoprocessing Modeling and Analysis

4.11.3.1 Increased water temperature

Stream temperature regulates the distribution and abundance of coldwater fish in the ecoregion. The NorthWest Stream Temperature project (NorWeST) developed a comprehensive interagency stream temperature database and high-resolution climate scenarios for the Northwest U.S (Isaak *et al*. 2011). The NorWeST database consists of data contributed by >60 state, federal, tribal, and private resource agencies and >50,000,000 hourly temperature recordings at >15,000 unique sites (Isaak *et al*. 2011).Those data are used with spatial statistical network models to accurately predict summer temperatures for historical and future climate scenarios in all reaches in the Northwest (Figure 4-51). The database provides the historical August stream temperatures in the ecoregion for current and future climate scenarios. This database of stream temperatures along with climate modeling predictions (CMIP3) has been used to evaluate climate refugia in the northwest U.S. for bull trout and cutthroat trout (Isaak *et al*. 2015).

4.11.3.2 Reduced Streamflow

Simulated average monthly hydrology is available for selected stream gaging stations in the region. These monthly hydrographs were created based on the Integrated Scenarios modeling by the Climate Impact Group (See Section 2.4 and Section 4.6). Modeling generally predicts a slight increase in overall annual flow (10 to 20 percent) by the end of the century for most sites. The increase in flow is projected to occur in the winter months. Peak flows generally shift one month earlier due to the reduction in snowpack. Spring and summer flows are projected to decrease due to reduced snowpack. In the ecoregion, late summer is a key period for coldwater fish because water temperatures are highest and streamflow is generally at its lowest levels. To estimate the potential impacts on coldwater fish, the estimated change in August streamflow for the six (6) sites shown in Figure 4-35 were evaluated.

4.11.3.3 Modeling Limitations

The NorWeST streamflow temperature modeling does not factor in changes in streamflow due to climate change in the temperature analysis. The modeling also does not include changes in the streamflow response due to a reduced groundwater recharge in slow draining systems (Safeeq *et al*. 2013).

4.11.4 Results

In the ecoregion, the Climate Shield modeling (Isaak *et al*. 2015) identifies 632 km of bull trout habitat with a probability of occupation of 10 percent or more (Figure 4-52). This is modeled to be reduced to 210 km by 2080, mainly due to forecasted temperature increases. The Climate Shield modeling also identifies 2,610 km of cutthroat habitat with a probability of occupation of 10 percent or more under present conditions. This is modeled to be reduced to 1,206 km by 2080 due to climate change.

In addition to the increased temperatures in August, the August streamflow will likely be reduced. An example of the reduction in streamflow is show for Teton Creek (Figure 4-53). Modeling of select streamflow in the ecoregion estimated that August streamflow would decrease by approximately 20% (ranging from 2 to 41 percent) in the RCP 4.5 at mid-century and by up to 40 percent (ranging from 10 to 59%) in RCP 8.5 by end the century. The reduction also contribute to higher stream temperatures and reduce available habitat and connectivity adding additional stress to cold water fish species.
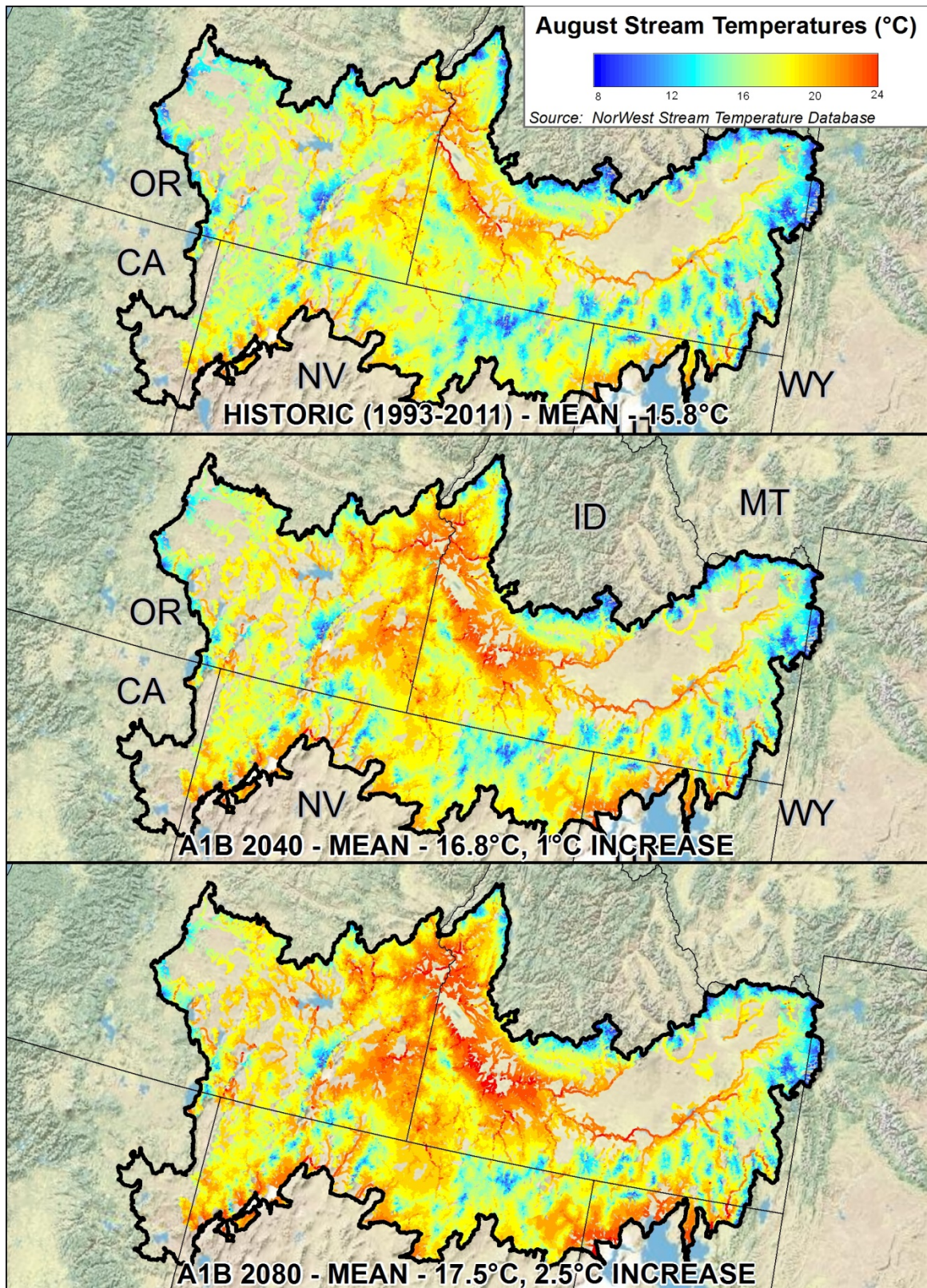
Figure 4-51. Historic and Projected August Stream Temperature
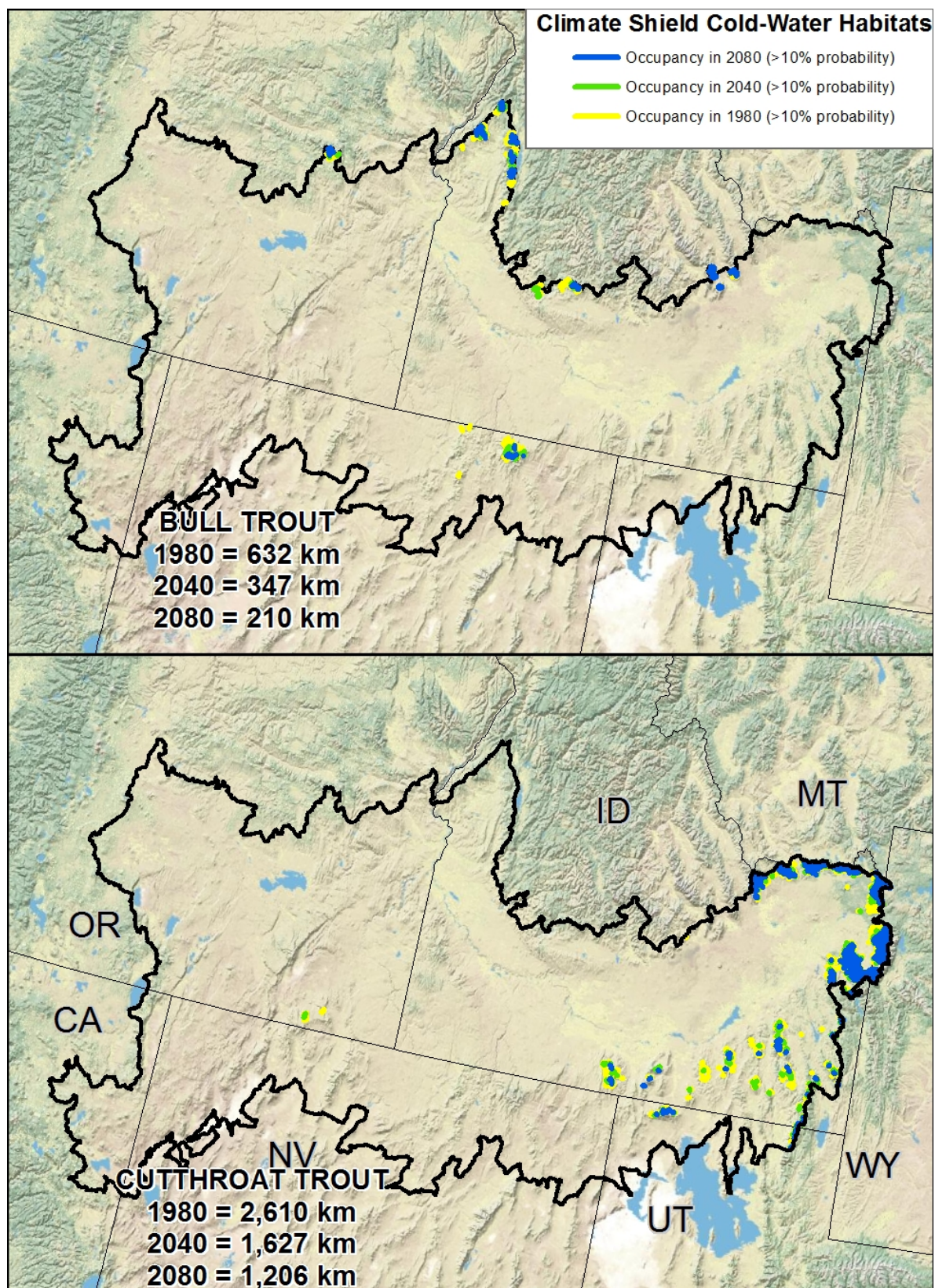
Figure 4-52. Climate Shield - Coldwater Fish Habitat

Climate Shield Cold-Water Habitats
- Occupancy in 2080 (>10% probability)
- Occupancy in 2040 (>10% probability)
- Occupancy in 1980 (>10% probability)

BULL TROUT
1980 = 632 km
2040 = 347 km
2080 = 210 km

CUTTHROAT TROUT
1980 = 2,610 km
2040 = 1,627 km
2080 = 1,206 km

OR, CA, NV, ID, MT, UT, WY
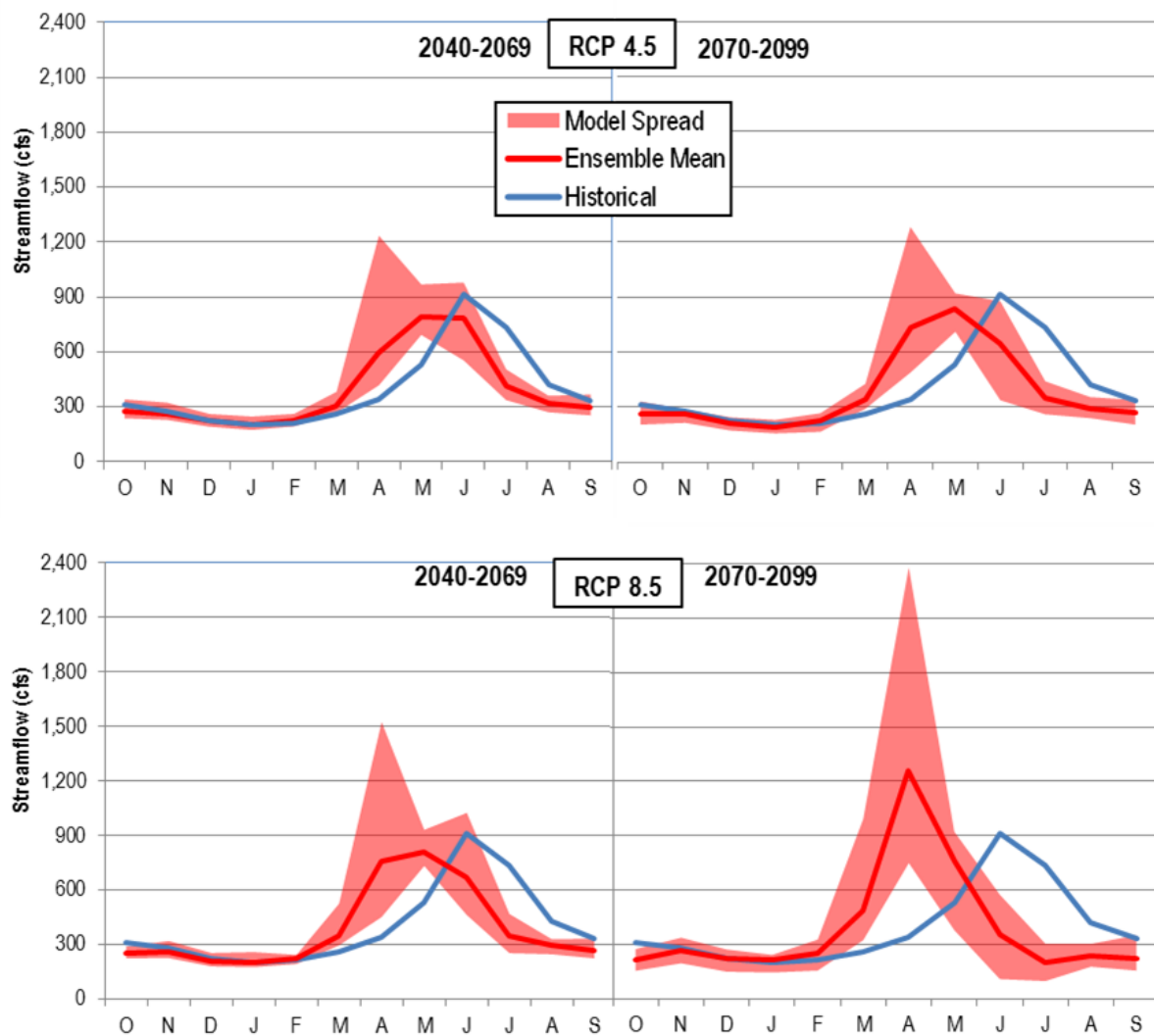
Figure 4-53. Projected Streamflows of Teton River above Leigh Creek

4.11.5 Coldwater Fish References

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4.12 Spotted Frog

The Columbia spotted frog is strongly associated with clear, slow-moving or ponded surface waters with little shade and relatively constant water temperatures (Munger *et al.* 1996; Reaser and Pilliod 2005; Bull 2005; Wilson 2006). Selected breeding/larval rearing sites provide a variety of herbaceous emergent, floating, and submergent vegetation (Bull 2005; Pearl *et al.* 2007). Although they are known to use temporary bodies of water for breeding in more mesic parts of their range, in more arid portions of the ecoregion breeding sites are predominantly associated with permanent water sources including springs, shallow ponds, and shorelines of streams and lakes (Pearl *et al.* 2007; Wilson 2006), although egg laying may occur in shallow seasonally flooded areas associated with permanent sources. A variety of CAs, discussed below, affect the availability and condition of spotted frog habitats.

Anuran habitats must provide the major resources for the annual cycle: reproduction, foraging, and hibernation/estivation (Sinisch 1990). These life cycle stages are central in the system model (Figure 4-54). In some circumstances, these resources may be located in the same habitat patch (e.g., a breeding pond with adequate summer and winter habitat), but for many Columbia spotted frogs, some or all of these resources are spatially separated, requiring seasonal migrations among different, sometimes distant, water bodies (Pilliod *et al.* 2002). For example, up to 50 percent of adult female Columbia spotted frogs migrated between aquatic breeding and summer habitats separated by 1,640 feet (500 meters) or more of high elevation dry coniferous forests (Pilliod *et al.* 2002). Bull and Hayes (2001) found that pond size, proximity to other permanent water, and water temperature were associated with frog movements. Presence of predators and food supply are important elements of breeding and summer habitats (Bull and Hayes 2001; Pilliod *et al.* 2010). Post-breeding travel frequently follows streams and riparian corridors (Turner 1960), but spotted frogs will cross dry, grazed grasslands and sagebrush uplands (Reaser and Pilliod 2005; Bull and Hayes 2001). Spotted frog migrations often appear to follow shortest-distance travel routes through dry open forest even when stream corridors were available nearby (Pilliod *et al.* 2002). Columbia spotted frog overwintering habitats are different from breeding and summer habitats, usually requiring a silt or muck layer for hibernation and sufficient oxygen levels beneath frozen pond surfaces (Bull and Hayes 2002; Pilliod *et al.* 2002; Bull 2005; Reaser and Pilliod 2005). Thus, conservation concerns include protecting not only suitable aquatic habitats for breeding/larval rearing, summer, and winter life cycle stages but also the stream, riparian, and overland corridors that connect these habitats. Pilliod *et al.* (2002) recommended protecting diverse water bodies and surrounding uplands within 1 kilometer of breeding ponds in high elevation sites. Bull and Hayes (2001) recommended protecting permanent pond, river, and stream habitat within at least 1,640 feet (500 meters) of breeding ponds in northeastern Oregon.

4.12.1 Spotted Frog Conceptual Model

Conceptual system models were developed to provide an ecological framework and justification for the choice of indicators that were used in assessing CA threats for this CE. This model depicts the important habitat components for the Columbia spotted frog (or functions and values) required throughout the year and incorporates the identified CAs, as well as potential effects from the actions of the CAs, on a landscape and local level (Figure 4-54).

The CAs considered for this CE analysis include development, climate change, and invasive species and disease, depicted in brown boxes across the top of the model (Figure 4-54). Suitable Columbia spotted frog habitat depends upon presence of healthy shallow aquatic sites for egg deposition, wetland riparian zones for adult foraging and overland movement, and suitable overwintering sites that remain above freezing temperatures. The Columbia spotted frog model shows the pathways of CAs that affect spotted frog landscape variables and local habitat requirements and thereby, the species' functions and values, depicted in the lower box. Blue boxes contain the predicted results of CA effects.

Analyses of long-term data from Columbia spotted frog breeding sites in the Great Basin show a significant variation in growth that was strongly associated with patch characteristics, drought frequency, and restoration actions (Hossack *et al.* 2013).

Table 4-20. Change Agents and their Principal Effects on Spotted Frog

Change Agent	Principal Effects on Spotted Frog
Development	Many anthropogenic and natural CA effects determine the availability and condition of suitable spotted frog habitats and the distribution and persistence of spotted frog populations. Habitat loss, degradation, and fragmentation is a combined result of past and current human development influences related to agriculture, livestock grazing, hydrologic diversions, mining, and climate change (USFWS 2015). Most of the development CAs depicted in the system model (Figure 4-54) affect hydrology and water quality in spotted frog habitats. Development tends to increase impervious surface, putting increased demands on existing wetlands and streams to carry runoff. Stream dredging and straightening lead to floodwaters rising and falling at an increased rate. Spotted frog breeding habitat at the margins of shallow wetlands and ponds can be affected by more pronounced and rapid water level fluctuations. Eggs laid during or immediately following late winter rains are often left exposed to freezing and desiccation by rapidly dropping water levels (Richter and Azous 1995). Water diversions and impoundments for agriculture, groundwater extraction, and rangeland management are included in the system model to indicate many other direct and indirect effects on spotted frog habitat, including lost or reduced surface and groundwater flow, flooding of desirable shallow-water habitat, changes in water temperatures, and increased habitat for predatory fish. In semi-arid areas, springs represent a stable permanent source of water for breeding, feeding, and overwintering frogs (Patla and Peterson 1996; Munger 2003). Population growth in water bodies with permanent hydroperiod was approximately 9 percent greater than in sites that dried regularly (Hossack *et al.* 2013). Diversion of springs for livestock watering can lead to loss of associated riparian habitats and wetlands. Livestock grazing affects riparian and stream ecosystems throughout the range of the Columbia spotted frog (Minshall *et al.* 1989; Munger *et al.* 1996; Reaser 1997; Engle 2002), but the magnitude of this threat in terms of reproductive success and survival is uncertain in the literature (reviews in USFWS 2005; Patla and Keinath 2005). Management of beaver populations is an important element affecting the availability of suitable spotted frog habitat (Reaser 1997; Nevada Department of Wildlife 2006; Oregon Department and Fish and Wildlife 2006). Past beaver and beaver dam removal practices have negatively affected spotted frog habitat (USFWS 2005). The effects of mining on water quality and quantity in general, and amphibians in particular, include addition of toxic substances into streams (such as methylmercury and other trace metals), altered stream morphology, and effects on groundwater and aquifers (Nelson *et al.* 1991; USFWS 2008; Nevada Department of Wildlife 2010). Aquatic habitat loss and degradation have had both local and landscape-level effects on Columbia spotted frogs, as shown in the system model. Loss of connectivity between aquatic habitats and fragmentation of habitat patches have contributed to the isolation of remaining spotted frog populations with implications for the long-term persistence of populations.
Climate change	Spotted frogs are highly vulnerable to natural drought events which sometimes cause local extirpations of populations (Turner 1962; Munger *et al.* 2002; Wilson 2006). In the western states, documented warming trends will produce large hydrological changes due to reduced snowpack and earlier melting. Direct effects on spotted frogs may include evaporative loss of some ponds due to higher summer temperatures with reduced survival during overland migration, earlier reproduction and more rapid larval development, and shorter hibernation periods (Corn 2003; Corn 2005; Patla and Keinath 2005). Climate effects are discussed in more detail below in Section 4.12.2.

| Change Agent	Principal Effects on Spotted Frog
Invasive species and disease	At the level of effects on individual survival and reproductive success, shown as mortality and disturbance in the system model, the primary CAs are invasive predatory fish species (salmonids and bass), bullfrogs (Monello and Wright 1999; Pilliod and Peterson 2001; Munger *et al.* 1996), and disease (chrytridiomycosis [chytrid] and ranavirus). Chytrid has not been associated with any large die-off of Columbia spotted frogs (Rollins-Smith *et al.* 2005; Adams *et al.* 2010) but monitoring of its occurrence and a better understanding of how it affects this species is needed (Russell *et al.* 2010). Malformations of frogs, which generally lead to higher mortality rates, are a common problem in Columbia spotted frog populations outside the Great Basin Distinct Population Segment (Johnson *et al.* 2002). These malformations are associated with the presence and abundance of trematodes (Ribeiroia) and parasitic snails (Planorbella) in anthropogenic wetlands and stock ponds. The level of malformations in the Great Basin Distinct Population Segment of Columbia spotted frogs is currently not significant but the range extension of Planorbella into the ecoregion may result in a greater threat.
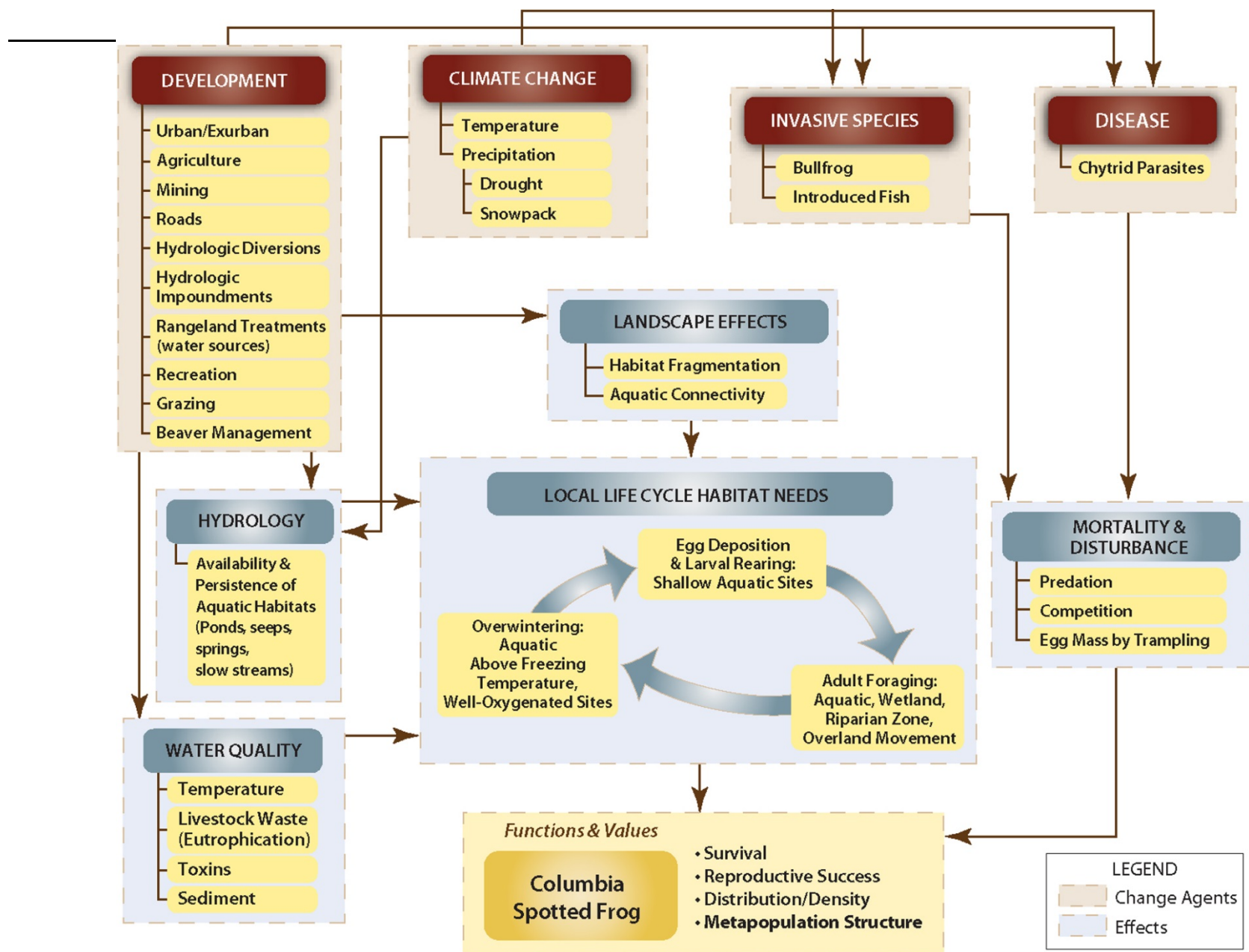
Figure 4-51. Columbia Spotted Frog Conceptual Model

4.12.2 Climate Influences

The Columbia spotted frog is a widely distributed species that once occupied large portions of the Intermountain West, from central Nevada to Alaska. The spotted frog is able to breed in a variety of freshwater habitats and climates, ranging in elevation from 1,640 to 9,960 feet (500 to 3,036 meters) (Pilliod *et al.* 2015). However, in the Great Basin the spotted frog exists in small isolated populations with little gene flow due to the unsuitability of intervening habitat (Pilliod *et al.* 2015).

Pilliod *et al.* (2015) developed a model of current climate suitability and used it to predict the historic and future distribution of suitable climates for the Columbia spotted frog (Figure 4-55). The suitable climate conditions in the Great Basin have been reduced by 50 percent over the last century. Future climates at the currently occupied locations are predicted to become warmer throughout the year and drier during the frog's activity period (May through September). Fall and winter precipitation may increase, but as rain instead of snow. Earlier runoff and lower summer base flows may reduce the already limited connectivity between isolated populations in the Great Basin. However, milder winters, longer growing seasons, and wetter falls could improve survival and dispersal. On the whole, seasonal shifts in temperature, precipitation, and stream flow patterns associated with climate change could reduce habitat suitability and connectivity for frogs and other aquatic species in the Great Basin region.

Effects of wildfire depend on the main vegetation types present in the vicinity of spotted frog habitat (sagebrush shrublands, desert shrublands, and pinyon-juniper woodlands), and the size and condition of riparian vegetation adjacent to occupied water bodies (USFWS 2015). Riparian vegetation often recovers relatively rapidly following fire; however, recovery rates depend on the condition of the riparian area prior to the fire, fire severity, post-fire flooding, and post-fire management. Most negative effects to aquatic species after wildfire are due to the immediate loss or alteration of habitat, including loss of riparian cover, and indirect effects such as post-fire hydrologic events. Loss of riparian vegetative cover leading to increased water temperature, for example, could affect the viability of spotted frog populations in post-fire habitats. Changing climate has affected summer temperatures and the timing of snowmelt, which have contributed to increasing the length of the wildfire season, wildfire frequency, and the size of wildfires (McKenzie *et al.* 2004; Westerling *et al.* 2006). Although Columbia spotted frogs evolved in a fire-prone environment, increases in wildfire frequency and severity due to increased fuel loads, exotic species, and effects of climate change have increased the threat to this species (USFWS 2015). The threat is exacerbated by the current fragmented and isolated state of occupied habitat, which makes recolonization of extirpated areas more challenging.

Climate change could also have a positive impact on higher elevation portions of the Columbia spotted frog range. A 9-year demographic study of Columbia spotted frogs in the Bitterroot Mountains of Montana documented an increase in survival and breeding as winter severity declined. Thus, a warming climate with less severe winters could increase population viability for montane frog populations. The study demonstrated that amphibians and other alpine or boreal ectotherms that occur at or near their thermal ecological limits could benefit from the milder winters provided by a warming climate (McCaffery *et al.* 2010; McCaffery *et al.* 2012).

Hossack *et al.* (2013) evaluated the long-term population trends of spotted frogs across their range. Spotted frog populations have been observed to rapidly grow in restored wetlands in areas of historical declines. Hossack *et al.* (2013) recommends maintaining large areas of habitat, as small habitat areas have greater vulnerability to extreme events like droughts and wildfires. Pond construction and other wetland restoration efforts can effectively increase growth rates of the spotted frog and are becoming increasingly important to mitigating the effects of more severe droughts associated with climate change (Hossack *et al.* 2013).
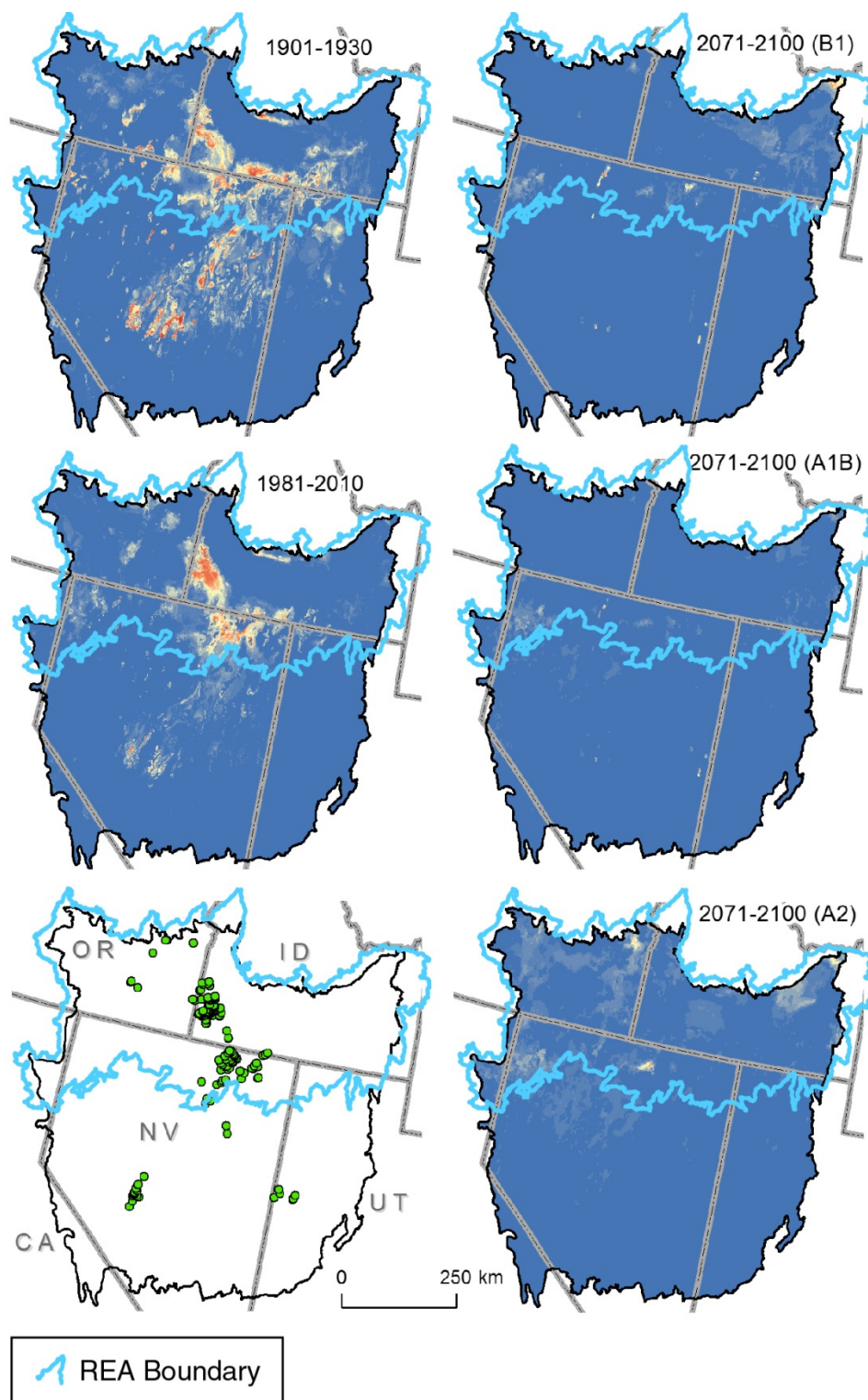
Figure 4-55. Geographic projections of past, recent, and future climate suitability for Columbia spotted frogs in the Great Basin (black line) (Pilliod *et al.* 2015 [Figure 1])

Each panel uses an equivalent color ramp, with cooler colors indicating lower probability of suitability and warmer colors indicating greater suitability (range = 0–0.97). Points shown in green are current breeding locations (points in Utah were not used in model development).

4.12.3 Geoprocessing Modeling and Analysis

There was no geoprocessing modeling and analysis completed for Columbia spotted frog. A recent study (Pilliod *et al.* 2015) provided a comprehensive climate change impact analysis for Columbia spotted frog in the ecoregion.

4.12.4 Results

A recently completed study on climate suitability (historic, current, and future) (Pilliod *et al.* 2015) shows that the suitable climate conditions for Columbia Spotted Frog in the Great Basin have been reduced 50 percent over the last century. Future climate suitability is expected to continue to decline significantly (77–97%) due to seasonal shifts in temperature, precipitation, and stream flow patterns associated with climate change. Reduced snowpack and warming at higher elevations has the potential to increase population viability for montane frog populations (McCaffrey *et al.* 2012). However, Pilliod *et al.* (2015) did not observe an increase in future climate suitability at higher elevations. Less snowpack could negatively impact wetland hydrology in the summer months and the Great Basin mountain ranges have steep topography which may limit the formation of wetland habitat (Pilliod *et al.* 2015). If climate suitability projections are realized, the existing, isolated Columbia spotted frog populations in the ecoregion could disappear. Human intervention and cooperation of private, state, and federal landowners may be needed to facilitate the long-term persistence of the Columbia spotted frog in the ecoregion in the future (Pilliod *et al.* 2015).

4.12.5 Spotted Frog References

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5 Lessons Learned

This assessment benefited greatly from the previously completed REA effort and the Integrated Scenarios Dataset collection. The Integrated Scenarios of the Future Northwest Environment project utilizes the global climate models from CMIP5 to portray as accurately as possible the Northwest's future climate. The Integrated Scenarios data is composed of outputs from downscaled climate data (MACA), the hydrologic models (VIC), and the vegetation models (MC2). All these models utilized the GCM output from CMIP5 which supported the Fifth Assessment Report for the IPCC and represent the next-generation climate change framework for resource managers. When work began at the start of this climate change supplement, many of the datasets were not yet available. Overtime, the vegetation and hydrological model outputs were processed and added to the large collection of climate data hosted by the Applied Science Lab at the University of Idaho. Climate change science and climate modeling efforts are ever evolving and refining. Every month new data products become available along with the completion of research projects. For projects like these, it can be a challenge to balance when to wait for new products to be finished versus presenting data that will be superseded in a couple years. The amount of studies available and number of models available can be overwhelming. Resource managers and decision makers need useful, up-to-date climate information that can be applied to improve policy outcomes. In many cases, however, the implications of climate change for a particular species or resource is not fully understood. Results of models and studies may conflict and the current answer maybe that more research is needed. This chapter provides an overall summary of the analyses that were completed and provides lessons learned on how to make the process better in the future. As the available information on climate change is ever-expanding and refining, this section highlights the current significant data gaps, important on-going research for the ecoregion, and recommendations for future analyses.

5.1 Summary of Analysis and Reporting

The scope of this climate change supplement to the REA and the evaluation of CEs (coarse and fine filters) relative to their interactions with the climate change required the identification and evaluation of hundreds of additional climate datasets and a massive effort to develop maps of not only where these resources are located within a multi-state area but also what is happening as a result of climate change to these resources in each of those states. Substantial resources were dedicated to the development and creation of the geospatial output products. Where data were available, the geospatial output of all the fine-filter, coarse-filter, and CA analyses were compared to the applicable Integrated Scenarios data products. Although the REA products will be useful to resource managers in the future, it is important to understand the limitations associated with this type of analysis.

5.2 Data Limitations

Because this analysis substantially relied on large scale multi-state datasets, it is subject to all the limitations in accuracy and precision associated with the original data. These limitations were previously discussed in the REA and repeated here. It is important to note that the results of the bioclimatic analysis are heavily biased/influenced by the resolution of the predictor data (bioclimatic factors) as well as the values assigned as thresholds from the literature. The inherent bias in this type of approach starts with the 30×30 m Landsat pixel that likely includes (reflects) native vegetation, invasive vegetation, bare ground, litter, etc. There is high variability within the cell, even though a single value (attribute) is assigned to that cell. In other words, just because a pixel returns a positive result for whatever the attribute is that it supposedly reflects doesn't mean that every square meter within that pixel contains that attribute.

In addition, attempting to apply quantitative values for elevation, temperature and precipitation across a particular species distribution in an area with a semi-arid climate might not be completely accurate.

Sometimes, physiological details of species abilities are known and can be related to environmental data and therefore reasonably modeled. Upon review of all of the figures in this REA, it must be recognized that there is a mixture of data quality, generalization and similar specifications on the target species. There are clear limitations with this approach and the results that are based on these biases must be used with all of this in mind. Although the best available data were used at the time of this assessment, there are several limitations to the data and the methods used to complete the REA. Most of these were beyond the control of the study team. Some of these included:

o   Lack of ecoregion-wide datasets. Some states in the ecoregion actively collect and store geospatial resource data and other states did not.
o   Some states provided very fine scale data that were not appropriate for use at the landscape scale or would not match data from other states.
o   Although some ecoregion-wide datasets were obtained (e.g., WAFWA), the way the states collected or categorized the information varied from state to state which is evident when state data is combined by groups like WAFWA.
o   Point occurrence records are initially biased due to the fact that researchers are actively seeking out the species.
o   Point occurrence data may be historic in nature and represent areas where the species no longer occurs.
o   Records typically only indicated species that were present in an area and not absences data. Absence of the species from other areas may only indicate that those areas were not surveyed.
o   Development of some of the species assemblages was not conducive to an assemblage type analysis because of the different habitat requirements of the species. For example, the various fish species could not be modeled as an assemblage because of the different habitat requirements of each of the species.
o   Rolling the analysis up to the watershed level or 4km grid also dilutes the original data.

5.3 Significant Data Gaps, On-going Research, and Recommendations for Future Analyses

The following section provides a summary of the data gaps, on-going research and recommendations for analysis for some of the agents/elements analyzed in this analysis.

Invasives (Annual Grasses)

Most models and experimental studies available have focused on cheatgrass and there were no studies found that examined the impacts of climate change on ventenata and medusahead. In addition, climate change could result in a substantial expansion of red brome (*Bromus rubens*) northward into the southern Great Basin Desert and Colorado Plateau. If cheatgrass retreats due to the climate no longer being suitable, other invasive species, like red brome may occupy those vacated lands (Bradley *et al.* 2016). The effects of climate change on other invasive annual grasses (ventenata, medusahead, red brome) and forbs would benefit from research.

Shrubs

There is very limited information available on climate change impacts to salt desert scrub communities. Species in these communities, like winter fat, are very important forage sources for mule deer and pronghorn during winter. More research is needed regarding climate change impacts to salt desert scrub communities. There is an on-going study to update the regeneration and suitability analysis for sagebrush based on ecohydrology (Schlaepfer *et al.* 2012, Schlaepfer *et al.* 2014) using the CMIP5 GCM results.

The modeling also plans to quantify soil moisture regimes, plant communities. Initial results from these studies were published in October 2016 (Palmquist *et al.* 2016).

Trees

The USFS Moscow Forestry Sciences Laboratory developed climate models for various tree species modeling the effects of climate change in various future scenarios used the CMIP3 scenarios and climate outputs. These results were presented in the REA. The results for the Species-Climate profiles could be updated using the climate variables from the CMIP5 modeling results. The MC2 modeling predicts significant increases in woodland and forest vegetation in the ecoregion, mainly due to the fertilization effect of CO_2. The MC2 modeling assumptions regarding the fertilization effect of CO_2 may need to be updated to limit the CO_2 fertilization effect as other nutrients (e.g. nitrogen) becoming limiting to increased growth.

Aspen

Regional-scale models of aspen distribution based on bioclimatic envelopes may underestimate the range and occurrence of aspen. At the landscape-scale, the landscape characteristics (e.g. slope, aspect, soils, contributing watershed) are important determinants where aspen stands could occur (Yang *et al.* 2015). The Northwestern Climate Center is currently funding a similar study (led by Douglas Shinneman and Timothy Link) to the Yang *et al.* 2015 studying the Jarbidge Mountains which couples similar hydrological and landscape-disturbance models to evaluate the climate change effects on aspen distribution and productivity in the Central and Northern Rockies. This on-going study (2014-2016) should provide finer-scale predictions of climate change impacts on aspen distribution in other portions of the ecoregion.

Hydrology

The rain-dominant versus snow-melt dominated watershed classification were based on the previous CMIP3-based Western Hydroclimate VIC modeling. The rain-dominant vs. snow-melt dominant could be updated to the Integrated Scenarios CMIP5-based modeling results. The Integrated Scenarios hydrology modeling (VIC) does not include impacts from reduced groundwater recharge for slow draining systems. With warming temperatures, snowpack will be reduced and will melt earlier in the season, resulting in less groundwater recharge in the mountains. In slow-draining systems the groundwater recharge supports summer streamflows through spring discharge. With less recharge from snowmelt, summer streamflows will be reduced (Tague and Grant 2009). Recent soil moisture modeling of sagebrush habitat (low-lying areas) found that groundwater recharge would likely increase with increased winter precipitation (Palmquist *et al.* 2016). Groundwater recharge is important in supporting springs and summer streamflows in the ecoregion. More study regarding climate change impacts to groundwater recharge is needed.

Pronghorn

WAFWA data are not available for all states in the ecoregion (data for Oregon is not available) and are mapped at different resolutions. Better mapping of pronghorn habitat, similar to mule deer, may allow for a better understanding of potential climate change impacts to pronghorn.

Coldwater Fish

The Climate Shield modeling for coldwater fish species are based on the CMIP3 global climate model results. Additional Climate Shield modeling utilizing downscaled CMIP5 data would be useful to understand the climate change impacts with the most recent results.

Columbia Spotted Frog

The climate suitability modeling for the Columbia spotted frog are on the VIC modeling based on the CMIP3 GCM results. Additional climate suitability modeling for the Columbia spotted frog could be completed based on the outputs of the Integrated Scenarios VIC Hydrology model which are based on the CMIP5 GCM outputs.

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Appendix A – Additional Figures

This section will be included in the Final Report.