

# Climate Change for the New Mexico State Wildlife Action Plan

Produced for the NMDGF by RMRS, Albuquerque

## 1. How have climates and habitat changed in New Mexico?

Over the last century temperatures have increased by approximately 1.2°C within the Western U.S. (Spears et al. 2013). Measures of hot and cold spells show 21<sup>st</sup> century temperatures are well above normal mean monthly maximums and minimums (Wuebbles et al. 2013). Still, recent trends have not yet exceeded heat spells observed during the 1930's though there has been a highly significant decrease in record breaking cold months (Wuebbles et al. 2013). In New Mexico, mean surface air temperature increased by 1°C from 1985 to 2005, although most of this warming occurred in the last decade of that period (Rangwala and Miller 2010). Temperatures within the Rio Grande Basin during the period 1995-2004 were more than 1.1°C higher than 1961-1990 (D'Antonio and Watkins 2006). On average, mean annual temperature in New Mexico has increased by 0.3°C per decade since 1976 (Enquist and Gori 2008). Seasonally, mean temperatures have increased more during winter than spring or summer months. The southwestern, central, and northwestern regions of New Mexico have seen the greatest increase in temperatures, particularly within the Jemez Mountains (Enquist and Gori 2008). Other areas have not shown appreciable changes in temperature over this period including parts of the Gila River headwaters, the Zuni Mountains and the Sangre de Cristo Mountains. However, these observations are rare and somewhat localized as the remaining mountain ranges have recorded increases in temperatures.

Drought and precipitation within New Mexico is driven by interannual and multi-decade variations in ocean-atmosphere interactions (Spears et al. 2013). On average, mean annual precipitation has increased within New Mexico since the 1970s though not all regions have experience similar trends (Enquist and Gori 2008). Mid- to high-elevation forests and woodlands have generally experience warmer and drier conditions over the last 2 decades (Enquist and Gori 2008), whereas many grassland habitats have experience warmer and wetter conditions over the same period. However, the amount of precipitation falling during the monsoon period has decreased over the last 20 years across this region (Arias et al. 2012). In addition, the frequency and intensity of precipitation events has increased over the past 4 decades (Wuebbles et al. 2013). The southwestern United States has experienced extended and severe drought several times over the last century including during the 1930s, 1950s, 1990's and 2002-2004 (Notaro et al. 2012). The 1950's drought was the most severe on record with precipitation lower than any other time in the last 350 years. During this period, many lowland desert species and upland conifer woodlands experienced widespread die off. More recently, drought has been associated with widespread tree mortality (Breshears et al. 2005b).

The effects of climate change are already apparent across the region where warming trends have resulted in a number of observable changes to the hydrological cycles (Hurd and Coonrod 2008). A consequence of warming temperatures and precipitation change has been reduced stream flows (Enquist and Gori 2008, Rood et al. 2008, Lundquist et al. 2009). Average stream flows have decreased by 2% and evapotranspiration has increased over the last century (Fields et al. 2007). Spring and summer flows have declined by as much as 25% over the last 30 years. Declines in spring snowpack and a reduction in the percent of winter precipitation falling as snow are primary drivers of these trends (Spears et al. 2013). Seventy-four percent of western U.S. mountain weather stations have shown an increase in the fraction of precipitation falling as rain rather than snow and snow water equivalents have declined 15–30% since 1950s (Fields et al. 2007). In addition, peak flows from snowmelt are arriving earlier across the West (McCabe and Wolock 2007, Lundquist et al.). Over the last two decades, snowpack has declined over the majority of New Mexico's mountain ranges and peak snowmelt flows occur on average one week earlier than they did 50 years ago (Enquist and Gori 2008). Importantly, these changes appear to be the result of warmer temperatures rather than changes in precipitation (Fields et al. 2007).

Observed changes in climate vary across the diverse range of landscapes within New Mexico. In general, high elevation sites have experience greater degree of warming and drying than lower elevation sites (Enquist and Gori 2008). Specifically, watersheds within the Sierra San Luis/Peloncillo Mountains, the Jemez Mountains and the southern Sangre de Cristo Mountains have experience the greatest amount of warming from 1991–2005. The Bottomless Lakes, Bitter Lake, Blue River/Eagle Creek, Western Plains of San Augustine, Salt Basin/Northern Brokeoff Mountains, Middle Pecos River, Rio Agua Negra, Salado Creek, Grulla National Wildlife Refuge and Pastura Grasslands have experience less change from climate impacts over this same time period. Most areas within the Chihuahuan desert experienced warmer and drier conditions during 2000–2005 but both wetter and dryer warming conditions from 1991–2005 (Enquist and Gori 2008). Within the Chihuahuan desert, the Bottomless Lakes, Lost River, Pecos River, Bitter Lake, Rio Felix, and Lower Hondo have had significant increasing mean maximum and minimum temperatures over 1970–2005 (Enquist and Gori 2008). In their report, Enquist and Gori (2008) also assess changes for key conservation area that were designated by combining The Nature Conservancy's conservation areas and New Mexico's Department of Game and Fish key areas (as identified in NMDGF's Comprehensive Wildlife Strategy (New Mexico Department of Game and Fish 2006)). Conservation areas within Arizona-New Mexico Mountains experienced largely warmer and drier conditions over the 1991–2005 periods particularly over the 2000–2005 time period. The Northern Black Range, Sacramento Mountains, Mogollon divide, Mimbres River and Gila River had significant increases in mean average minimum temperatures. The Sacramento Mountains also had a significant increasing trend in mean annual maximum temperature. On average, the Colorado Plateau was warmer and had both wetter and drier spots for 1991–2005 and was consistently warmer and drier from 2000–2005. For the Southern Rockies, most conservation areas were warmer and wetter over 1991–2005 but warmer and drier when only considering 2000–2005 data. Specifically, the Jemez Mountains, Rio Hondo, Southern Sangre de Cristo Mountains all had significant higher mean annual minimum and maximum temperature trends over 1970–2006. These areas also held the highest number of drought sensitive species

of any region, leaving them particularly vulnerable to continued warming trends. Apache highland areas were consistently warmer and drier. Southern and shortgrass prairie areas tended to fall along warmer-wetter conditions for 1991–2005 but many conservation areas switched to warmer and drier means for 2000–2005.

## **2. How will climate and habitat change in the 21<sup>st</sup> century?**

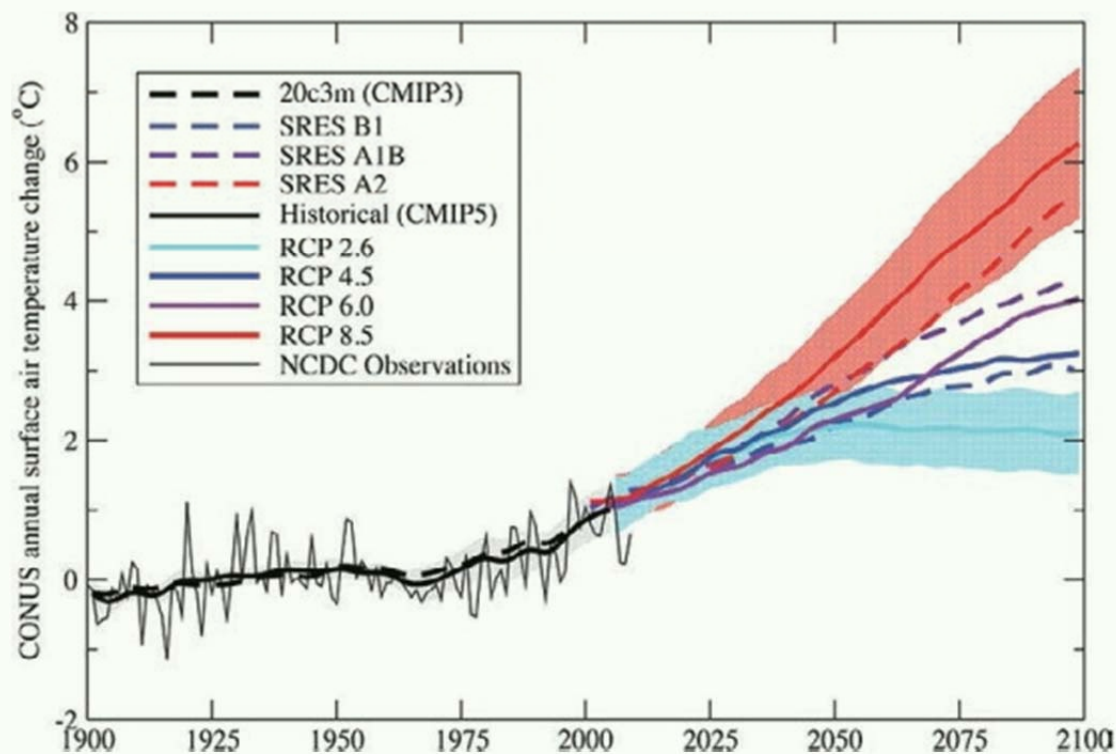
Mean global temperature is expected to increase by 1 to 6°C over the next 100 years (IPCC 2007). This rise in temperature will be accompanied by increases in extreme weather events, higher maximum temperatures, more intense precipitation events, more heat waves, more drought (mid-latitudes), and fewer frost days (Easterling et al. 2000). Climate projections identify the southwestern United States as a hotspot for climate change (Seager et al. 2007). The southwestern United States is expected to experience relatively large temperature increases as compared to global projections with increases in the severity and duration of drought periods, heat waves, and greater variation in precipitation events that will lead to more wildfire, insect outbreaks, increased evapotranspiration and salinization (Easterling et al. 2000, Fields et al. 2007, Garfin and Lenart 2007). The bimodal distribution of precipitation within the Southwest has proven difficult to capture within climate models. Still, a general consensus exists that annual precipitation is likely to decline over the next century, exacerbating many of the effects of increasing temperatures such as reduced snow pack and lower water flow in spring and summer.

### **Climate models and estimating future climate regimes**

Climate studies have long used coupled General Circulation Models (also called Global Climate Models: GCMs) to explore the mechanisms of climate change. GCMs are mathematical representations of large scale atmospheric and ocean processes that determine weather conditions. Climate is affected by a number of factors, referred to as forcings, including radiative output from the sun, volcanic eruptions, and changing greenhouse gas emissions. Using GCMs, scientists simulate future climates under a range of standardized scenarios that represent different emission and radiative forcings. Climate projections discussed in this chapter are based on data from either phase 3 or phase 5 of the Coupled Model Intercomparison Project (CMIP3 and CMIP5, respectively). The CMIP is a standardized protocol for comparing and evaluating the output of coupled atmospheric-ocean general circulation models (AOGCMS-<http://cmip-pcmdi.llnl.gov/>) with participation by nearly the entire international climate modeling community. The IPCC's third and fourth Assessment Reports (TAR and AR4 2012) were based upon CMIP3 simulations that included output from 23 coupled atmosphere-ocean general circulation models (AOGCM). The latest simulations based on CMIP5 models (2014) includes over 50 models and are generally thought to have better resolution and incorporate additional complexities. In addition, anthropogenic forcings change between CMIP3 and CMIP5 generations. Where CMIP3 projections were based on 4 Special Report on Emissions Scenarios (SRES A1, B1, A2, B2), CMIP5 simulations use Representative Concentration Pathways (RCP2.5, RCP4.5, RCP6.5 and RCP8.5). The SRES were based on storylines predicting emissions under different socioeconomic, technological and political conditions that relate to greenhouse gas concentrations and radiative forcings. The SRES range from a low impact B1 future to a future described by increasing population growth and continued accumulations of greenhouse

gas emissions (SRES A2). The RCP are not associated with storylines but rather cover a range of potential radiative forcings that could occur under a diverse range of economic, political or technological conditions. The lowest, RCP2.5, assumes emissions peak 2010-2020 and then decline, in RCP4.5 emissions peak at 2040, in RCP6.5 emissions peak at 2080 and in RCP8.5 emissions continue to rise throughout the 21<sup>st</sup> century.

Studies comparing projections between CMIP3 and CMIP5 generations show similar levels of climate sensitivity (ability to predict trends) and uncertainty (Maloney et al. 2013, Sheffield et al. 2013, Wuebbles et al. 2013). Difference among the SRES or RCP account for the greatest variations in projected temperature and precipitation among the two sets of models (Figure 1). This is important as the majority of data for New Mexico habitats and species are based on CMIP3 based scenarios of future conditions though studies are beginning to incorporate new climate model data. Within this report, we describe current projections for New Mexico climates based upon the most current CMIP5 projections. Discussions of habitat change, species impacts are based upon both sets of models.

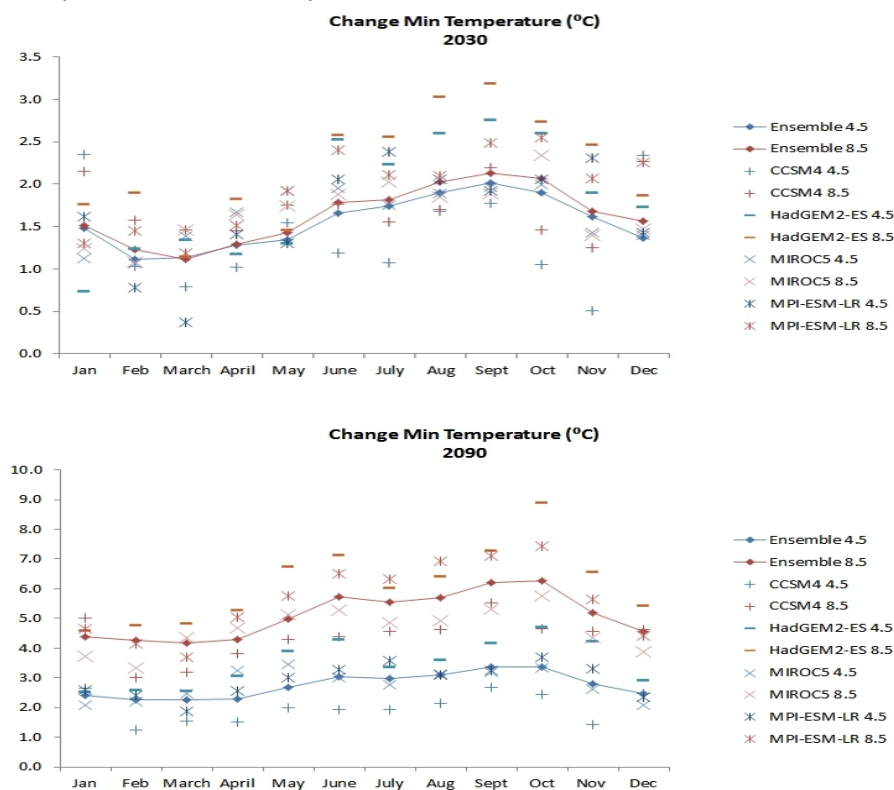


**Figure 1.** Projected CMIP3 and CMIP5 annual temperature changes (°C) for the U.S. Multimodel average (lines) and range (shown for RCP8.5 and RCP2.6 only, for illustrative purposes) relative to the 1901–60 average. Shaded regions for the higher RCP8.5 and lower RCP2.6 scenarios represent one standard deviation across the models. The standard deviation range in intermediate scenarios (RCP4.5 and RCP6.0) is similar but omitted here for clarity. (From Wuebbles et al. 2013: CMIP5 Climate Model Analyses: Climate Extremes in the United States. *Bull. Amer. Meteor. Soc.*, **95**, 571–583. ©American Meteorological Society. Used with permission.)

## Key climatic trends for temperature in New Mexico

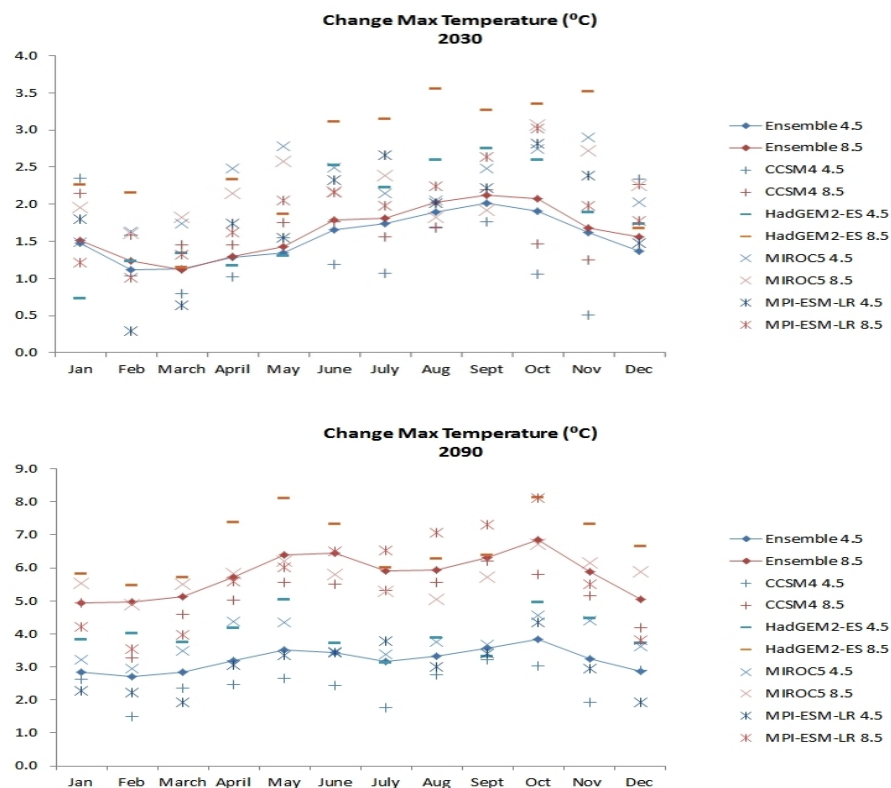
Mean annual temperature is expected to increase by 3.3°C by 2061–2090 from the 1971–2000 average (D'Antonio and Watkins 2006). Monthly temperatures are projected to increase by 1 to 2°C by 2030 and by 2 to 5°C by 2090 (Figures 2.1 and 2.2). Results are shown for the multimodel ensemble mean of all models as well as for three individual models under medium (RCP4.5) and high (RCP8.5) forcing scenarios. Overall, means are better at replicating historic climate than any single model (Maloney et al. 2013), but individual models may provide better estimates for regional phenomenon like monsoons and extreme events.

Monthly minimum and maximum temperatures increase the most during summer months for the time spanning 2020–2040. Maximum temperatures appear to increase slightly more than mean minimum temperatures. Late century predictions (2080–2100) vary dramatically according the RCP. Under low RCP4.5 futures that assume peak emissions levels in 2040, temperatures increase an additional 2.5 to 3° C. Under steady emissions increase (RCP8.5), minimum temperatures increase by 4.5–6.5°C and maximum temperatures increase by 5–7°C.



**Figure 2.1.** Change from current (1970–2005) in °C of average monthly minimum temperatures in New Mexico for 2030 (average of conditions 2020–2040) and 2090 (average of conditions 2080–2100). Values are given for a model average, or ensemble, across 23 GCMs as well as for three models known to perform well with regional climate phenomenon under a low RCP4.5 and high RCP8.5 emission scenario. Climate data downloaded from the National Climate Change Viewer, USGS. [http://www.usgs.gov/climate/landuse/clu\\_rd/apps/nccv\\_viewer.asp](http://www.usgs.gov/climate/landuse/clu_rd/apps/nccv_viewer.asp)





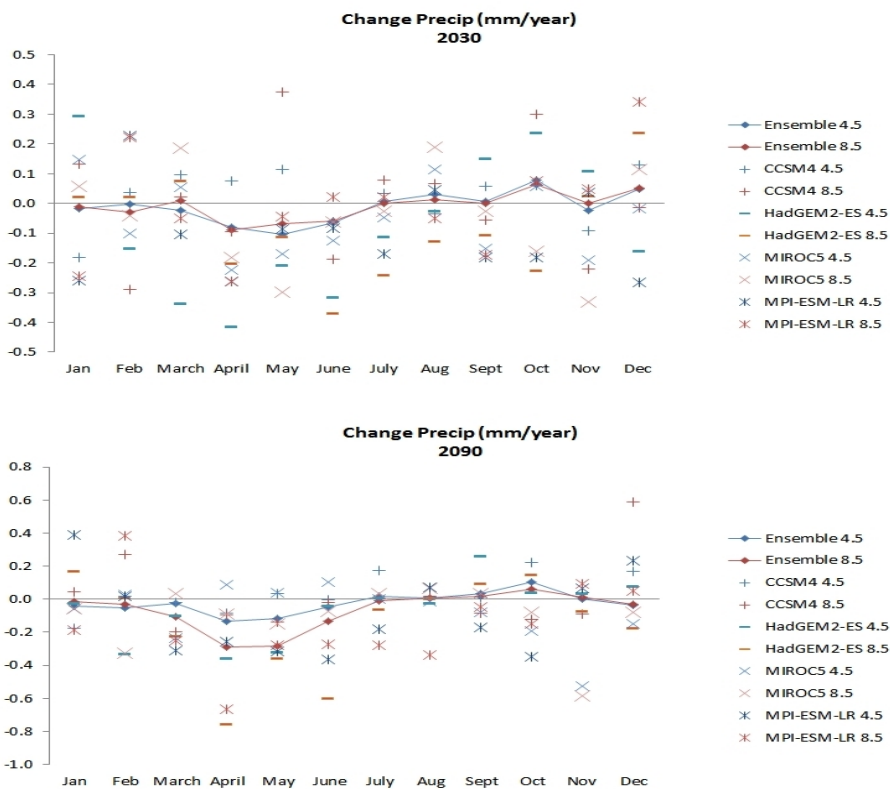
**Figure 2.2.** Change from current (1970–2005) in °C of average monthly maximum temperatures in New Mexico for 2030 (average of conditions 2020–2040) and 2090 (average of conditions 2080–2100). Values are given for a model average, or ensemble, across 23 GCMs as well as for three models known to perform well with regional climate phenomenon under a low RCP4.5 and high RCP8.5 emission scenario. Climate data downloaded from the National Climate Change Viewer, USGS. [http://www.usgs.gov/climate\\_landuse/clu\\_rd/apps/nccv\\_viewer.asp](http://www.usgs.gov/climate_landuse/clu_rd/apps/nccv_viewer.asp)

### Key climatic trends for precipitation in New Mexico

In general, mean annual precipitation is expected to decrease over mid-latitude and semi-arid regions under the high RCP8.5 scenario (Spears et al. 2013). Precipitation trends vary somewhat under lower RCP scenarios. Short duration events are likely to become more intense overall with fewer weak storms. Extreme precipitation events are likely to become more frequent and more intense (Collins et al. 2013) and at the same time, the number of precipitation events overall are likely to decline (Spears et al. 2013). These patterns may already be observed within the U.S. (Min et al. 2011).

The greatest percentage of annual rainfall in New Mexico occurs during the monsoon season (July–September). The North American monsoon is poorly represented by most models (Gutzler et al. 2005). However, among those that appear able to capture trends for the Southwest, models project drier winter and spring seasons and shorter monsoon periods with either more or less precipitation (Swain and Hayhoe 2015). Models generally project less mean annual precipitation though there is considerable disagreement on how much precipitation might

change (Pascale et al. 2015). Overall, CMIP5 models project reduced winter and spring precipitation (-30–50%) and essentially no change in late summer/fall precipitation. However, for the core monsoon area that includes NM, CMIP5 historical comparisons (1979–2005) appear to underestimate precipitation during the March–September period. Most models project a decrease of precipitation for December–July but show less consensus for late summer rainfall, with some models predicting an increase in September–October amounts and others showing a decline (Maloney et al. 2013). In part, conditions change corresponding to the RCP used in the analysis, with the most severe drying expected under the high RCP8.5 (Figure 2.3). There is high model agreement for a decrease in monsoonal precipitation as well as increase in number of annual dry days by nearly a month (Pascale et al. 2015) under the high RCP8.5. These findings are also supported under less severe RCP4.5 scenarios (Lee and Wang 2014). However, model agreement is quite low for onset of monsoonal moisture with some models projecting an advance and some a delay. The frequency of the El Niño–Southern Oscillation (ENSO), the primary driver of North America’s interannual variation in precipitation does not appear to change under either RCP4.5 or RCP8.5 scenarios (Maloney et al. 2013).



**Figure 2.3.** Change from current (1970–2005) average monthly precipitation in New Mexico for 2030 (average of conditions 2020–2040) and 2090 (average of conditions 2080–2100). Values are given for a model average, or ensemble, across 23 GCMs as well as for three models known to perform well with regional climate phenomenon under a low RCP4.5 and high RCP8.5 emission scenario. Climate data downloaded from the National Climate Change Viewer, USGS. [http://www.usgs.gov/climate\\_landuse/clu\\_rd/apps/nccv\\_viewer.asp](http://www.usgs.gov/climate_landuse/clu_rd/apps/nccv_viewer.asp)

Figure 2.3 shows the range of potential changes to monthly precipitation amounts within New Mexico. Ensemble mean projections follow trends noted within regional comparisons, particularly for spring season declines in precipitation. Among individual models, HadGem2-ES performed better than the ensemble mean at simulating North American monsoon events (Sheffield et al. 2013). HadGem2-ES shows much drier condition across New Mexico during winter and spring months and wetter conditions during monsoon periods under RCP4.5 but not RCP8.5 (Figure 2.3).

## **Extreme Weather**

Climate extremes are likely to be accentuated under global warming with increased likelihood of more extreme dry and wet seasons (Wuebbles et al. 2013, Swain and Hayhoe 2015). Many areas are likely to experience novel climate regimes with mean climate conditions projected to be hotter and drier than has been recorded (Notaro et al. 2012). Extreme conditions may be more important for predicting habitat and species response to climate change as these may be more limiting than mean conditions. CMIP5 projections show an increase in extreme high and low monthly temperatures over the next century. Hot spell temperatures increase by at least 7°C in the northern Rockies (Wuebbles et al. 2013). Rare cold events are likely to warm by 1–2°C under the lower RCP2.6 scenario and as much as 5–8°C under RCP8.5 by the end of the century (Wuebbles et al. 2013). Under RCP2.6, annual minimum temperature extremes are likely to occur half as frequently by the end of the century and under RCP8.5 they cease to occur at all (Wuebbles et al. 2013). The maximum temperature of rare heat events are projected to increase by 1°C under RCP2.6 and by 6–7°C under RCP8.5 by the end of the century. Annual maximum temperature extremes are projected to occur 4–10 times more frequently by the end of the century (as compared to 1986–2005) under RCP2.6 and annually by the end of the century under RCP8.5.

Extreme precipitation events are likely to become more frequent and more intense (Collins et al. 2013, Wuebbles et al. 2013). Models show more extreme winter precipitation events (Dominguez et al. 2010), and an increase in the frequency of more extreme precipitation events (Watterson and Dix 2003, Watterson 2005, Sun et al. 2007). Both CMIP3 and CMIP5 model projections show a greater percentage of annual precipitation falling in just the top 1% of events over time, another indication for more intense storms. The fraction of precipitation falling during these intense events is projected to increase by 50% under a RCP 4.5 (mid-low) scenario and by 90% under a high RCP8.5 scenario. As noted previously, these are more likely to be rain than snow events.

Drought and precipitation within New Mexico is driven by interannual and multi-decade variations in ocean-atmosphere interactions. The 1950's drought was the most severe in the U.S. with precipitation lower than any other time in the last 350 years (Notaro et al. 2012). During that time, many lowland desert species and upland conifer woodlands experienced widespread die off. Widespread tree mortality has also been noted for more recent droughts (Breshears et al. 2005b). Climate projections indicate an increase in the frequency and duration of droughts within the southwest. Drought severity is also likely to be worsened in the future by warming that increases evaporation rates (Gutzler and Robbins 2010). Projections of increased



frequency and severity of drought are particularly extreme for the southwestern U.S. (Seager et al. 2013). By 2050, mean forest drought-stress is likely to exceed that of the most severe droughts in the last 1000 years (Williams et al. 2010). Unprecedented forest drought stress conditions are projected to occur over 20% of the 21st century (Williams et al. 2010).

### **3. How will climate change interact with existing stressors?**

The rapid rate of climate change is soon likely to exceed the range of natural climate variability and accelerate the rate at which habitats are degraded and species are lost (Jackson and Overpeck 2000, Walther et al. 2002, Overpeck et al. 2003, Thomas et al. 2004, Hannah et al. 2005). Climate change can interact with other stressors triggering non-linear threshold effects. A prominent example of current climate change effects within western North America is the widespread die-off of conifer species driven by the interaction of drought, insects and fire (Breshears et al. 2005b). Climate change is ubiquitous, influencing many processes directly and many more indirectly and to varying degrees and rates. This leads to a complexity of interactive synergistic and antagonistic effects. As a result few interactions have a clear direction. Increased fire activity is likely to favor fire-adapted species causing shifts in plant communities (McKenzie et al. 2004). Temperature and moisture conditions affect tree host susceptibility to pathogens, pathogen transmission among trees, and the range of both hosts and pathogens that can both increase and decrease infestations. Drier conditions are likely to reduce plant productivity but increased CO<sub>2</sub> concentrations can support increased growth, water efficiency, and resistant to diseases (Sturrock et al. 2011).

#### **3.1 Fire & fire suppression**

Change to disturbance regimes, in particular fire, accounts for a large proportion of observed climate change impacts on wildlife habitat. Increasing spring and summer temperatures, reduced soil and fuel moisture, and drought contribute to increased wildfire activity (Ryan et al. 2008). Wildfires are already observed to be larger, more frequent, and intense under recent conditions characterized by higher temperatures and drought (Westerling et al. 2006, Lettenmaier 2008). Future wildfire potential is expected to increase dramatically in southwestern forests as a result of projected drier and hotter conditions (Brown et al. 2004, Spracklen et al. 2009). Increasing temperatures are likely to increase the number of burn days and acres burned (McKenzie et al. 2004). Though drought conditions tend to lead to increased frequency and extent of wildfires, they may also reduce wildfire risk through reduction of fine fuels (Ford et al. 2012).

Changes in wildfire regimes have many implications for New Mexico habitats. Drought-fire interactions are very likely to disproportionately affect lowland forest communities. Where fires are very large, forests and woodlands may suffer a loss of regeneration potential and leading to significant changes in forest composition and structure (Williams et al. 2010). Increased wildfire is likely to encourage the establishment of exotic grass species in fire sensitive shrubland and desert habitats (Crist et al. 2014). Post fire landscapes are at an increased risk of erosion from

wind and rain, particularly in areas with high slopes (Enquist and Gori 2008). Not all systems are equally impacted by fire, however, and increased wildfire may be beneficial for grassland habitats (Ford et al. 2012).

### 3.2 Invasive non-native/alien species plants & animals

The distribution of both native and exotic invasive species will change in response to climate change. Some plant invasive species, like drought tolerant tamarisk, may be favored under future conditions, while others, like Russian olive, may begin to retreat from hot areas (Perry et al. 2012). Within aquatic systems, warmer waters may facilitate the establishment of aquatic invasive species such as the quagga mussel and simultaneously reduce the effectiveness of biological and chemical control agents (Hellmann et al. 2008). Warming waters may facilitate the spread of cold-limited invasive fish species. On the other hand increased fragmentation of water bodies may act to reduce spread of some exotic species (Hellmann et al. 2008). Drought may increase the susceptibility of higher-elevation ecosystems to invasion by exotic grasses, which in turn increases wildfire risk (Ford et al. 2012). Increases in fire and insects also favor invasive plant species that dominate disturbed habitats. Changes in the timing of precipitation (from summer to winter dominated rainfall) and increasing CO<sub>2</sub> is expected to increase the encroachment of woody species into grasslands (Morgan et al. 2007). These conditions may also exacerbate human related disruptions to grasslands (Hansen et al. 2001, Jetz et al. 2007).

### 3.3 Problematic native plants & animals

Pathogens and pests are strongly influenced by environmental conditions. Increased crowding in shrinking ponds can increase disease transmission as has been noted for avian cholera within waterfowl during drought years (Smith et al. 1990).

Drought stressed forests and woodlands are more susceptible to pests and pathogens (Dale et al. 2001). Within new Mexico, large outbreaks of bark beetle infestations caused extensive dieback in forests during two extreme drought events in the 1950s and 2000s (Allen and Breshears 1998, Breshears et al. 2005b, Ryan et al. 2008). Insects are strongly influenced by weather conditions. Warmer temperatures and longer growing seasons can boost insect populations by increasing overwinter survival, increasing developmental rates, and facilitate range expansions (Logan et al. 2003, Williams et al. 2008). At the same time, increasing temperatures and drought stress plants and increase their susceptibility to infestation. In turn, increased tree mortality due to insect outbreaks may increase fire risk through changes in fuel loads and can lead to further destabilization as erosion and sedimentation increase.

Grasslands are likely to be highly vulnerable to invasive species under climate change (Chambers and Pellant 2008, Morgan et al. 2008). Of particular concern for grasslands is that climate change may increase invasion of woody species (Morgan et al. 2007, Enquist and Gori 2008). Creosote (*Larrea tridentata*) has been spreading into grama grass (*Bouteloua* spp.) dominated grasslands in central New Mexico (Gill and Burke 1999) and honey mesquite (*Prosopis glandulosa*) and creosote have been spreading into black grama (*B. eriopoda*)

grasslands in southern New Mexico (Buffington and Herbel 1965, Gibbens et al. 2005). Increased CO<sub>2</sub> concentrations and greater variability in precipitation favors the establishment and expansion of C3 species (i.e. woody shrubs) over C4 (grasses) plants (Knapp et al. 2002). Of the climate drivers of woody plant encroachment, drought and shifts towards increased winter precipitation seem to be the most important (Brown et al. 1997, Pennington and Collins 2007, Báez et al. 2013, Munson et al. 2013). Warming winter temperatures can also favor shrubs though extremes during summer may actually increase mortality within shrub species (Backlund et al. 2008, Ryan et al. 2008). There are some mechanisms by which grass species may be favored by climate changes. Warmer conditions and increased summer monsoon rainfall could favor grass species like black grama within Chihuahuan desert ecosystems (Peters 2002). Also, elevated CO<sub>2</sub> appears to dampen simulated losses of semiarid grasslands in an analysis of the Southwest (Notaro et al. 2012). Increased temperatures and longer dry periods increases the likelihood of wildfires in grassland habitats (Moritz et al. 2012), which can prevent establishment of woody species (Ford et al. 2012).

### 3.4 Pathogens & microbes

Climate change will also affect many plant diseases, which are strongly influenced by environmental conditions (Sturrock et al. 2011). Several diseases of trees including root pathogens (e.g. *Armillaria* spp.) and canker pathogens are more likely to reach epidemic levels where trees are weakened by heat stress and drought (Sturrock et al. 2011). Warm midsummer temperatures have been linked to explosive growth in *Cytospora* cankers (*Valsa melanodiscus*) and increased mortality in *Alnus incana* subsp. *tenuifolia* in southwestern Colorado. Sudden aspen decline, a disease of *Populus tremuloides*, presents a good example of how climate may impact forest species. This disease does not have a single etiological agent but rather is diagnosed by rapid, synchronous branch dieback and tree mortality at a very large scale. Drought has proven to be an important initial condition leading to sudden aspen decline. Further, sudden aspen decline in Colorado is almost entirely limited to the edge of aspen's climate envelope suggesting that this condition is strongly dependent on environmental conditions (Rehfeldt et al. 2009, Sturrock et al. 2011). Not all diseases will benefit from warming conditions, however. White pine blister rust (*Cronartium ribicola*), present within New Mexico's Sacramento and White mountains, may decline if there are fewer wet periods in early spring and summer when temperatures are suitable cool for the spread of the basidiospore (Sturrock et al. 2011).

### 3.5 Other ecosystem modifications

Land use change can increase fragmentation and reduce connectivity of habitat for wildlife populations, limiting their capacity to adapt to changing conditions. In Southwest riparian systems, drought and intense heat are likely to lead to increased species mortality and the shrinkage and fragmentation of riparian habitat, issues compounded by over-extraction of water and invasive species (Palmer et al. 2009). The Rio Grande is already suffering from excessive water extraction and is considered at risk of more extreme flood events due to the urbanization of its watersheds (Palmer et al. 2009). It has been noted that altered patterns of

land use, water withdrawal, and species invasions will likely dwarf climate change impacts to Southwestern aquatic systems, although effects are intertwined (Meyer et al. 1999).

### 3.5 Climate Change Effects

#### 3.5.1 Fragmentation of habitat

Increasing temperatures will increase evaporation rates and reduce water availability for many species that depend upon ephemeral water bodies for breeding, foraging and stopover sites. Increased crowding at remaining resources could increase incidence of disease, predation and the impact of spreading invasive species. On the other hand increased fragmentation of some water bodies may act to reduce spread of some exotic species. Climate change will also increase human needs for water further reducing aquatic resources for wildlife (Perry et al. 2015). Snag and tree roost sites may become rarer for many birds or disconnected from important foraging sites for bats (Bagne et al. 2011, Friggens et al. 2013).

#### 3.5.2 Ecosystem encroachment

Climate change may increase invasion of woody species into grassland communities (Morgan et al. 2007, Enquist and Gori 2008). Creosote (*Larrea tridentata*) has been spreading into grama grass (*Bouteloua* spp.) dominated grasslands in central New Mexico (Gill and Burke 1999) and honey mesquite (*Prosopis glandulosa*) and creosote have been spreading into black grama (*B. eriopoda*) grasslands in southern New Mexico (Buffington and Herbel 1965, Gibbens et al. 2005). Increased CO<sub>2</sub> concentrations and greater variability in precipitation favors the establishment and expansion of C3 species (i.e. woody shrubs) over C4 (grasses) plants (Knapp et al. 2002). Of the climate drivers of woody plant encroachment, drought and shifts towards increased winter precipitation seem to be the most important (Brown et al. 1997, Pennington and Collins 2007, Báez et al. 2013, Munson et al. 2013). Warming winter temperatures can also favor shrubs though extremes during summer may actually increase mortality within shrub species (Backlund et al. 2008, Ryan et al. 2008). There are some mechanisms by which grass species may be favored by climate changes. Warmer conditions and increased summer monsoon rainfall could favor grass species like black grama within Chihuahuan desert ecosystems (Peters 2002). Also, elevated CO<sub>2</sub> appears to dampen simulated losses of semiarid grasslands in an analysis of the Southwest (Notaro et al. 2012). Increased temperatures and longer dry periods increases the likelihood of wildfires in grassland habitats (Moritz et al. 2012), which can prevent establishment of woody species (Ford et al. 2012).

#### 3.5.3 Changes in temperature regimes

Mean annual temperature is expected to increase by 3.3°C by 2061–2090 from the 1971–2000 average (D’Antonio and Watkins 2006). Monthly temperatures are projected to increase by 1 to 2°C by 2030 and by 2 to 5°C by 2090 (Figures 2.1 and 2.2). Results are shown for the multimodel ensemble mean of all models as well as for three individual models under medium (RCP4.5) and high (RCP8.5) forcing scenarios. Overall, ensemble models are better at replicating historic

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Monthly minimum and maximum temperatures increase the most during summer months for the time spanning 2020–2040. Maximum temperatures appear to increase slightly more than mean minimum temperatures. Late century predictions (2080–2100) vary dramatically according the RCP. Under low RCP4.5 futures that assume peak emissions levels in 2040, temperatures increase an additional 2.5 to 3° C. Under steady emissions increase (RCP8.5), minimum temperatures increase by 4.5–6.5°C and maximum temperatures increase by 5–7°C.

### 3.5.4 Changes in precipitation & broad-scale hydrological regimes

In general, mean annual precipitation is expected to decrease over mid-latitude and semi-arid regions under the high RCP8.5 scenario (Spears et al. 2013). Precipitation trends vary somewhat under lower RCP scenarios. Short duration events are likely to become more intense overall with fewer weak storms. Extreme precipitation events are likely to become more frequent and more intense (Collins et al. 2013) and at the same time, the number of precipitation events overall are likely to decline (Spears et al. 2013). These patterns may already be observed within the U.S. (Min et al. 2011).

The greatest percentage of annual rainfall in New Mexico occurs during the monsoon season (July–September). The North American monsoon is poorly represented by most models (Gutzler et al. 2005). However, among those that appear able to capture trends for the Southwest, models project drier winter and spring seasons and shorter monsoon periods with either more or less precipitation (Swain and Hayhoe 2015). Models generally project less mean annual precipitation though there is considerable disagreement on how much precipitation might change (Pascale et al. 2015). Overall, CMIP5 models project reduced winter and spring precipitation (–30–50%) and essentially no change in late summer/fall precipitation. However, for the core monsoon area that includes New Mexico, CMIP5 historical comparisons (1979–2005) appear to underestimate precipitation during the March–September period. Most models project a decrease of precipitation for December–July but show less consensus for late summer rainfall, with some models predicting an increase in September–October amounts and others showing a decline (Maloney et al. 2013). In part, conditions change corresponding to the RCP used in the analysis, with the most severe drying expected under the high RCP8.5 (Figure 2.3). There is high model agreement for a decrease in monsoonal precipitation as well as increase in number of annual dry days by nearly a month (Pascale et al. 2015) under the high RCP8.5. These findings are also supported under less severe RCP4.5 scenarios (Lee and Wang 2014). However, model agreement is quite low for onset of monsoonal moisture with some models projecting an advance and some a delay. The frequency of the El Niño–Southern Oscillation (ENSO), the primary driver of North America’s interannual variation in precipitation does not appear to change under either RCP4.5 or RCP8.5 scenarios (Maloney et al. 2013).

### 3.5.5 Severe/extreme weather events

Climate extremes are likely to be accentuated under global warming with increased likelihood of more extreme dry and wet seasons (Wuebbles et al. 2013, Swain and Hayhoe 2015). Many areas are likely to experience novel climate regimes with mean climate conditions projected to be hotter and drier than has been recorded (Notaro et al. 2012). These extreme conditions may be more important for predicting habitat and species response to climate change as these may be more limiting than mean conditions. CMIP5 projections show an increase in extreme high and low monthly temperatures over the next century. Hot spell temperatures increase by at least 7°C in the northern Rockies (Wuebbles et al. 2013). Rare cold events are likely to warm by 1–2°C under the lower RCP2.6 scenario and as much as 5–8°C under RCP8.5 by the end of the century (Wuebbles et al. 2013). Under RCP2.6, annual minimum temperature extremes are likely to occur half as frequently by the end of the century and under RCP8.5 they cease to occur at all (Wuebbles et al. 2013). The maximum temperature of rare heat events are projected to increase by 1°C under RCP2.6 and by 6–7°C under RCP8.5 by the end of the century. Annual maximum temperature extremes are projected to occur 4–10 times more frequently by the end of the century (as compared to 1986–2005) under RCP2.6 and annually by the end of the century under RCP8.5. Extreme precipitation events are likely to become more frequent and more intense (Collins et al. 2013, Wuebbles et al. 2013). Models show more extreme winter precipitation events (Dominguez et al. 2010), and an increase in the frequency of more extreme precipitation events (Watterson and Dix 2003, Watterson 2005, Sun et al. 2007). Both CMIP3 and CMIP5 model projections show a greater percentage of annual precipitation falling in just the top 1% of events over time, another indication for more intense storms. The fraction of precipitation falling during these intense events is projected to increase by 50% under a RCP 4.5 (mid-low) scenario and by 90% under a high RCP8.5 scenario.

Drought and precipitation within New Mexico is driven by interannual and multi-decade variations in ocean-atmosphere interactions. The 1950's drought was the most severe in the U.S. with precipitation lower than any other time in the last 350 years (Notaro et al. 2012). During this time, many lowland desert species and upland conifer woodlands experienced widespread die off. Widespread tree mortality has also resulted from more recent drought events (Breshears et al. 2005b). Climate projections indicate an increase in the frequency and duration of droughts within the Southwest. Drought severity is also likely to be worsened in the future by increased evaporation rates with warming (Gutzler and Robbins 2010). Projections of increased frequency and severity of drought are particularly extreme for the southwestern U.S. (Seager et al. 2013). By 2050, mean forest drought-stress is likely to exceed that of the most severe droughts in the last 1000 years (Williams et al. 2010). Unprecedented forest drought stress conditions are projected to occur over 20% of the 21st century (Williams et al. 2010).



## 4. Assessments of ongoing and projected changes to New Mexico wildlife habitats

### 4.1. Terrestrial systems

#### 4.1.1. Statewide:

Temperature and precipitation define the environmental and hydrological conditions that determine vegetation composition at large scales. At smaller scales vegetation responds to topography, competition and physical impacts of animal species. The sensitivity of vegetation to climate change relates to the degree to which smaller scale processes ameliorate or exacerbate climate impacts. Direct impacts of climate change on vegetation results from conditions that limit establishment, growth, productivity and life history events. Indirect effects include impacts from changing disturbance regimes and biogeochemistry. Changes in the timing of critical events like spring floods and increases in climate extremes like winter and early spring flooding, heat waves, and drought will cause shift in vegetation communities by disrupting ecological processes, recruitment and survival (Raymond et al. 2014). Water limitations (e.g. water use deficits) are the primary determinant limiting plant growth within the southwestern U.S. Where water is available (e.g. high elevation forests), temperature becomes the more important limiting factor (e.g. growing season). For most of New Mexico future rising temperatures will increase evapotranspiration and the likelihood of water balance deficits, which will limit plant growth and favor drought tolerant species (Raymond et al. 2014).

Still, many studies show that temperature alone drives or is sufficient to lead to observed or predicted changes (Williams et al. 2010), tree growth; (Currie 2001), biodiversity; (Garfin and Lenart 2007); (Hansen et al. 2001), plant and animal species richness; (Notaro et al. 2012), plant species distributions). This is likely a result of the influence that temperature has not only on evapotranspiration, which can amplify water stress during drought, but on hydrological dynamics driving water availability within the arid Southwest (Williams et al. 2013). In an analysis of 170 tree and shrub species across the Southwest, Notaro et al. (2012) identified three important predictors of future range of North American species: temperature (with precipitation changes modifying patterns driven by warming trends), growing degree days, and annual mean temperature. Two species examined, Engelmann spruce (*Picea engelmannii*) and mesquite (*Prosopis juliflora*), are limited by high and low temperatures, respectively. Engelmann spruce is a cold tolerant, high elevation, evergreen conifer and the most abundant tree species within the Southwest (Notaro et al. 2012), and mesquite, a drought tolerant tree/shrub, is found in hotter regions. Under future climate scenarios, the distribution of conditions suitable for Englemann spruce are expected to decline in the Southwest, whereas those suitable for mesquite are likely to expand and shift northward (Notaro et al. 2012). Similarly, climate changes are likely to lead to an expansion of conditions associated with yucca species and a majority of the oak species, but contractions for other species, including sagebrush (*Artemesia* spp.), ponderosa pine, (*Pinus ponderosa*) and mountain hemlock (Hansen et al. 2001, Notaro et al. 2012).

Importantly, our understanding of climate impacts for New Mexico wildlife habitats is still growing and subject to change. Climate projections can fall outside of known historical ranges preventing a perfect view of future conditions and change (Currie 2001, McKenney et al. 2007, Williams et al. 2013). In addition, changes in atmospheric CO<sub>2</sub>, which not only drives climate changes but also influences plant water use efficiency and growth, may modulate vegetation response to hotter and drier conditions (Notaro et al. 2012). Climate change impacts on habitats are likely to be most dramatic near ecotones (Allen and Breshears 1998, Kupfer et al. 2005, Joyce et al. 2008). Transitions between forests and woodlands and grasslands and shrublands are strongly dependent on precipitation and temperature regimes. Climate related changes to fire and invasive species also influence transitions between biomes.

Overall, high elevation evergreen forests are likely to decline due to increased drought and temperatures (Notaro et al. 2012). Alpine habitats are likely to all but disappear (Hansen et al. 2001). Mid and lower elevation forests and woodlands may expand upslope but will be more susceptible to increased fire and drought conditions at lower elevation limits. Prairie ecosystems of eastern New Mexico may experience an increase biodiversity as climate become more suitable for Chihuahuan desert species (Notaro et al. 2012, Crist et al. 2014). Shrublands and arid woodlands are expected to expand into grasslands in the Interior West and Great Plains as they follow favorable climate regimes (Hansen et al. 2001).

### Grasslands

Grasslands are likely to be highly vulnerable to invasive species under climate change (Chambers and Pellant 2008, Morgan et al. 2008). Of particular concern for grasslands is that climate change may increase invasion of woody species (Morgan et al. 2007, Enquist and Gori 2008). Creosote (*Larrea tridentata*) has been spreading into grama grass (*Bouteloua* spp.) dominated grasslands in central New Mexico (Gill and Burke 1999) and honey mesquite (*Prosopis glandulosa*) and creosote have been spreading into black grama (*B. eriopoda*) grasslands in southern New Mexico (Buffington and Herbel 1965, Gibbens et al. 2005). Increased CO<sub>2</sub> concentrations and greater variability in precipitation favors the establishment and expansion of C3 species (i.e. woody shrubs) over C4 (grasses) plants (Knapp et al. 2002). Of the climate drivers of woody plant encroachment, drought and shifts towards increased winter precipitation seem to be the most important (Brown et al. 1997, Pennington and Collins 2007, Báez et al. 2013, Munson et al. 2013). Warming winter temperatures can also favor shrubs though extremes during summer may actually increase mortality within shrub species (Ryan et al. 2008, Backlund et al. 2008).

There are some mechanisms by which grass species may be favored by climate changes. Warmer conditions and increased summer monsoon rainfall could favor grass species like black grama within Chihuahuan desert ecosystems (Peters 2002). Also, elevated CO<sub>2</sub> may dampen declining trends in semiarid grasslands (Notaro et al. 2012). Increased temperatures and longer periods of dry conditions increases the likelihood of wildfires in grassland habitats (Moritz et al. 2012), which can prevent establishment of woody species (Ford et al. 2012).

Still, studies simulating potential range changes in grasslands under future climate scenarios, agree that grassland habitats are likely to decline overall. Rehfeldt et al. (2006) showed that climates associated with Plains grasslands decline significantly whereas semidesert grasslands increase somewhat within New Mexico. Notaro et al. (2012) projects widespread loss of grass vegetation, particularly across central New Mexico. These losses corresponded to the large spring-summer drying trends in climate models (Notaro et al. 2012). Grass die-off over the next 70 years was most strongly correlated with changes in precipitation, whereas tree cover response was inversely correlated with temperature (Notaro et al. 2012).

Ultimately, grasslands in good condition are not likely to be negatively impacted by temperature increases but will be influenced by changes in precipitation. Increased annual precipitation is likely to benefit grass species at grassland/scrubland interfaces but will favor woody species at woodland/grassland interfaces. Similarly, increased precipitation during monsoon seasons will favor grassland species, whereas increased winter precipitation will favor shrubland and woodland habitats. Predicted declines in spring precipitation appear likely to negatively impact grasslands.

### *Shrublands*

Temperature appears to be the most important climate variable for predicting tree and shrub plant species distributions across the Southwest (Notaro et al. 2012). Shrublands are like to respond positively to increased annual mean temperature and increased minimum and mean winter temperatures (Notaro et al. 2012). Increased winter precipitation is also associated with shrubland expansion. Increased precipitation during warm months can have a positive effect within shrub range but may lead to transition to non-shrub habitat at grass-shrub transition zones (Crist et al. 2014). Increases in maximum temperatures may have a negative impact on shrublands when accompanied by drought. Fire frequency is projected to increase within montane, desert and xeric shrublands, and temperate shrublands (Moritz et al. 2012), and is likely to favor grasslands.

### *Forests*

Response of forests to climate change will be modulated through changes in environmental conditions, insect pests, disease, and fire. Temperature appears to be the most important climate variable for predicting tree and shrub plant species distributions across the Southwest (Notaro et al. 2012). Temperature amplifies water limitations allowing for increased tree stress and mortality, particularly during drought periods (Williams et al. 2013). Winter temperatures appear more important for predicting evergreen species distributions, whereas mean annual and mean summer temperature had a stronger influence on deciduous species (Notaro et al. 2012). Future wildfire potential is expected to increase dramatically in southwestern forests as a result of drier hotter conditions (Brown et al. 2004, Spracklen et al. 2009, Moritz et al. 2012). Increasing temperatures are likely to increase the number of burn days and acres burned (McKenzie et al. 2004). Increased fire activity is likely to favor fire adapted species causing shifts in plant communities (McKenzie et al. 2004).

Most woody species and shrubs are expected to shift northward to track suitable climate conditions. Many higher elevation species experience range contractions as suitable climates disappear. Alpine and subalpine habitats may experience dramatic changes including movement of trees into these areas and an increase in the density of trees. Lower elevation forest species are likely to move upslope. However, the complexity of terrain and individual dispersal abilities means that it is unlikely that species will be able to track the climate envelopes exactly. Further, estimated shifts in conditions for communities (e.g. biomes) are likely to differ from realized responses as these communities are comprised of species which respond uniquely to climate change (e.g. Rehfeldt et al. 2006).

Climate change will affect many tree diseases, which are strongly influenced by environmental conditions (Sturrock et al. 2011). Piñon-juniper and spruce-fir forest ecosystems seem to be especially at increased risk of fire, insect and pathogen impacts under climate change (Beukema et al. 2007). Several diseases of trees including root pathogens (e.g. *Armillaria* spp.) and canker pathogens are more likely to reach epidemic levels where trees are weakened by heat stress and drought (Sturrock et al. 2011). Warm midsummer temperatures have been linked to explosive growth in *Cytospora* cankers (*Valsa melanodiscus*) and increased mortality in *Alnus incana* subsp. *tenuifolia* in southwestern Colorado. Sudden aspen decline, a disease of *Populus tremuloides* is likely to increase with increasing drought (Sturrock et al. 2011). Conversely, white pine blister rust (*Cronartium ribicola*) present within New Mexico's Sacramento and White mountains, is likely to decline with drier spring and summer conditions (Sturrock et al. 2011). Additionally, increased CO<sub>2</sub> concentrations can support increased growth, water efficiency, and disease resistance (Sturrock et al. 2011).

#### *Piñon- juniper woodlands*

Piñon-juniper woodlands (*Pinus edulis* and *Juniperus monosperma*) have recently spread into Ponderosa pine woodlands in north central New Mexico (Allen and Breshears 1998). Juniper species have also expanded into grasslands in southwestern New Mexico (Romme et al. 2009). However, woodland species, especially piñon trees, are highly susceptible to attack by bark beetles (*Ips confusus*) and twig beetle (*Pityophthorus opaculus*). Warmer temperatures increase bark beetle survival and developmental rates leading to more severe outbreaks (Bentz et al. 2010). Temperature amplifies water limitations allowing for increased tree stress and mortality, particularly during drought periods (Williams et al. 2013). Drought conditions and delayed onset of monsoons have increased mortality in infested piñon pine (Gustafson et al. 2015). Though somewhat more drought tolerant, Junipers also experience mortality under persistent droughts (Breshears et al. 2005a, Gaylord et al. 2013). It is likely that these widespread mortality events will become more frequent under climate conditions currently projected to occur.

Most woody species and shrubs are expected to shift northward to track suitable climate conditions. However, the complexity of terrain and individual dispersal abilities means that it is unlikely that species will be able to track their climate envelopes exactly. Further, estimates of shifts in conditions for communities (e.g. biomes) are also likely to differ from realized responses as these communities are comprised of species which respond uniquely to climate

change (e.g. Rehfeldt et al. 2006). Increased annual and seasonal precipitation is likely to benefit many woodland species though increase variability is expected to have a negative influence. Fires are expected to increase in woodland habitats (Moritz et al. 2012) and may lead to a greater shift to grassland or shrubland habitats in ecotones.

### *Riparian Habitats*

Riparian vegetation is not only affected by increased temperature, precipitation variation, and CO<sub>2</sub> but by factors relating to the timing and volume of river flows (Meyer et al. 1999). Prolonged heatwaves and higher maximum temperatures are likely to cause heat stress for many riparian plants. High temperatures are known to reduce photosynthesis in tamarisk, particularly where average growing season temperatures exceed 25°C (Perry et al. 2012). Lowland plants like Fremont cottonwood (*Populus fremontii*), desert willow (*Chilopsis linearis*), netleaf hackberry (*C. laevigata*), mulefat/seep willow (*Baccharis salicifolia*) are likely to shift upstream as temperatures rise. High elevation plants already at the limits of river basins and may begin to disappear. Russian olive may decline in parts of New Mexico as it is limited by warm temperatures. Tamarisk, which is cold limited, may be able to spread northward (though this is mainly an issue for states north of New Mexico).

Climate change is likely to disrupt phenology within riparian plant communities, potentially increasing mortality and decreasing regeneration in native species. Warmer spring temperatures can lead to early seed dispersal in species such as Fremont cottonwood (*Populus fremontii*), Goodding's willow (*Salix gooddingii*) and Sandbar willow (*S. exigua*) causing mismatches between dispersal and the conditions necessary for establishment (e.g. flooding) (Perry et al. 2012). Early spring emergence and warmer falls may increase growing season for many plants and increase productivity, but could also increase frost injuries when late spring frosts occur (Perry et al. 2012). Warmer autumn temperatures could affect seed dispersal of autumn fruiting riparian trees like box elder (*Acer negundo* L.), netleaf hackberry (*Celtis laevigata*) and slow the development of cold-hardiness in some species like cottonwoods (Perry et al. 2012).

Flow dynamics have a strong influence on the composition of riparian plant communities. Climate changes that reduce streamflow are expected to reduce the abundance of native early successional species in favor of herbaceous species and late successional and drought-tolerant woody species (Perry et al. 2012). Warmer and prolonged growing seasons will increase water use through increased evapotranspiration potentially reducing water availability and lowering water tables. Cottonwoods and willows are somewhat drought intolerant and may be vulnerable to lowering groundwater tables. In contrast, introduced tamarisk and Russian olive (*E. angustifolia*) are more drought tolerant (Perry et al. 2012). Reduced base flows during the summer lowers ground water levels and may lead to loss of native species' seedlings and young trees (Perry et al. 2012). Rapid declines in water tables also stress mature trees. Over time, these conditions encourage the establishment of drought tolerant species and may skew tree sex ratios for cottonwood and box elder trees (males survive better) (Perry et al. 2012). In more ephemeral, monsoon driven rivers, lower flows will have even more profound impacts with decreases in drought intolerant cottonwoods and willows likely and increased in drought

tolerant species (Perry et al. 2012). As such, shifts from perennial to intermittent flows in many areas have large consequences for plant community composition. Tamarisk is most likely to benefit in these areas as they are tolerant of intermittent flows and prefer disturbed sites (Perry et al. 2012).

Changes to fluvial dynamics are likely to affect many native riparian plant species, which rely on flood-driven substrates for seed dispersal and germination. Rain-dominated winter precipitation means less snowpack and an overall reduction in flood magnitudes and late summer flows, which can reduce geomorphic variation and lead to system stabilization and homogenization. Without spring floods many riparian forests may become dominated by shade-tolerant late-stage successional species and more perennial herbaceous species over annual grasses (Perry et al. 2012). These conditions are likely to favor the presence of several invasive species including Russian olive, tamarisk, cheatgrass (*Bromus tectorum*), leafy spurge (*Euphorbia esula*- currently very restricted to portions of northern New Mexico) and Canada thistle (*Cirsium arvense*) (Perry et al. 2012). Earlier spring floods resulting from earlier snowmelt may reduce cottonwood seedling recruitment if seed release and peak flow lose synchrony (Perry et al. 2012). Alternatively, increased winter flood magnitudes as might happen with rain on snow events would increase fluvial disturbance and geomorphic diversity and lead to more heterogeneous tree stands and less herbaceous cover (Perry et al. 2012). These events might increase cottonwood and willow recruitment. Increases in late season summer floods are not thought to benefit native species because they occur too late to coincide with willow and cottonwood seed release and may actually scour away seeds (Perry et al. 2012). Tamarisk, which produce seeds throughout the summer, would be able to take advantage of these late season flows (Perry et al. 2012).

Overall, riparian habitats are at high risk of negative impacts from climate related changes in species phenology and hydrological cycles. Drought and intense heat are likely to increase riparian plant mortality and shrink riparian habitat, issues compounded by over-extraction of water and invasive species (Palmer et al. 2009). The Rio Grande is already recognized as suffering from excessive water extraction and at risk of more extreme flood events due to the urbanization of its watersheds (Palmer et al. 2009). The interactive effects of land use, water withdrawal, species invasions, and climate change pose a real threat to the persistence of functional aquatic systems in the southwest (Meyer et al. 1999).

#### 4.1.2. Summaries of specific vegetation responses by Ecoregion

The section highlight studies and vulnerability assessments that identify potential impacts of climate change for specific species and habitats within each of New Mexico's Ecoregions. Appendix 1 lists macrogroups, biomes and ecological response units per Ecoregion and Appendix 2 lists the macrogroup association of species modeled in Notaro et al. 2012.

##### 4.1.2.1. Western Cordillera/Southern Rockies

Climate conditions association with Subalpine conifer forests are projected to decline by 95% by the end of the century (Rehfeldt et al. 2006; Table 4.1). Montane conifer forests decline by 53%



and Great Basin conifer woodlands increase by about 25% by 2090. Conditions suitable for Plains grasslands increase by 75% though ultimately comprise a small amount of the area within this ecoregion. Future climate is also likely to support Great Basin Montane scrub and desert scrub habitats.

**Table 4.1.** Predicted percent area suitable for each biome as simulated by Rehfeldt et al. 2006.

<b>WESTERN CORDILLERA</b>	<b>Current</b>	<b>2030</b>	<b>2090</b>
Great Basin Montane Scrub	--	0.05	0.06
Great Basin Conifer Woodland	0.28	0.39	0.36
Great Basin Desertscrub	--	0.01	0.06
Plains Grassland	0.03	0.01	0.11
Rocky Mountain Montane Conifer Forest	0.51	0.45	0.38
Rocky Mountain Subalpine Conifer Forest	0.18	0.09	0.01

Most type plant species for the Central Rocky Mountain dry forest, intermountain woodland, southern rocky mountain woodland macrogroups are projected to lose suitable habitat under climate change. Simulations show general declines for dominant tree species including *Pinus edulis*, *Picea engelmannii* and *Juniperus osteosperma* under future climate regimes (Rehfeldt et al. 2006). *Pinus flexilis* (limber pine), an alpine species located in northern New Mexico, was projected to have one of the largest range contractions within all evergreens across the Southwest (Notaro et al. 2012). Williams et al. (2010) estimated that future growth of *Pinus edulis*, *P. ponderosa* and *Pseudotsuga menziesii* will decrease across the Southwest, including New Mexico. *Quercus gambelii* was the only species to exhibit a potential increase in suitable area (Rehfeldt et al. 2006).

By the end of the century, *Pseudotsuga menziesii*, *P. ponderosa*, and *P. edulis* may shift upslope by as much as 500 m, *P. engelmannii* and *Q. gambelii* 300 m, and *Juniperus osteosperma* by about 100m (Rehfeldt et al. 2006).

Moisture stress and winter cold are limiting for *P. ponderosa* (Rehfeldt et al. 2006). While cold limitations may ease, precipitation changes are likely to play a dominate role in persistence of these species (Rehfeldt et al. 2006). *Picea engelmannii*, a high elevation evergreen conifer and the most abundant tree species in the southwestern U.S., is projected to decline substantially across the entire Southwest (Notaro et al. 2012). Southwestern ponderosa pine and Douglas-fir are likely to be at greatest risk for drought induced mortality at lower level elevations, whereas piñon pine appears sensitive throughout its range.

Rehfeldt 2006 notes that projections for *P. ponderosa* and *Q. gambelii*, two important species in southern montane forests, do not respond similarly to climate change nor do they mimic projections for the Montane forests biomes (slight increase in area overall). Similarly individual results for *J. osteosperma* and *P. edulis*, indicator species for the Great Basin conifer woodlands (most similar to Intermountain Woodland Macrogroup or Southern Rocky Mountain Woodland

Macrogroup), show that they would co-occur only over 6% (from current 35%) of their distributions. Therefore, it is likely that where conditions support great Basin Woodlands (which increase over time), species composition within those woodlands is likely to be quite different from present.

Within the Southern Rocky Mountain Montane Shrubland Macrogroup, *Artemisia tridentata* (big sagebrush) is expected to lose large areas of suitable habitat under climate change (Notaro et al. 2012).

Triepke et al. (2014) performed a vulnerability analysis for Ecological Response Units (ERU) across New Mexico and Arizona. The ERU classification system represents major ecosystem types of the Southwest, and groups vegetation according to site potential and historic disturbance regime such that each plant community within an ERU shares successional patterns, physiognomy and community dynamics (Triepke et al. 2014). In most cases ERUs are coarser than LANDFIRE Biophysical Settings, but are identical in concept. Vulnerability for each ERU, represented by image segments averaging 10-20 ha, was calculated as the relative probability of type conversion given the breadth of the climate envelope of the ERU, the magnitude of projected climate change at a given location, and the historic climate of the ERU as reflected its climate envelope. Ponderosa Pine Forest, Mixed Conifer Forest with Frequent Fire, Shortgrass Prairie, Spruce-Fir forest, Pinyon-Juniper woodlands, Sagebrush Shrubland, Great Basin grasslands, and Juniper Grass were the predominate (>1% of area) ERU's within the Southern Rockies Ecoregion (Table 4.2). Of these, most (greater than 40%) of the Sagebrush Shrublands and Mixed Conifer Forests were classified as low vulnerability and the majority of the remaining ERUs were considered medium vulnerability. Juniper grass also had a large proportion (27.9%) of area classified as highly vulnerable to climate impacts.

**Table 4.2.** Macrogroup-Ecological Response Units (ERU) crosswalk for ERUs with >1% area representation within the Ecoregion.

<b>ERU</b>	<b>Macrogroup</b>
Colorado Plateau / Great Basin Grassland	M118 - Intermountain Basins Cliff, Scree & Badland Sparse Vegetation
	M169 - Great Basin & Intermountain Tall Sagebrush Shrubland & Steppe
	M170 - Great Basin & Intermountain Dwarf Sagebrush Shrubland & Steppe
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
	M499 - Western North American Cool Semi-Desert Ruderal Scrub & Grassland
Juniper Grass – Cold	M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland
	M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
	M043 - California Chaparral
	M087 - Chihuahuan Semi-Desert Grassland
	M086 - Chihuahuan Desert Scrub
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
	M010 - Madrean Lowland Evergreen Woodland

Mixed Conifer – Frequent Fire	M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland
	M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
	M118 - Intermountain Basins Cliff, Scree & Badland Sparse Vegetation
	M011 - Madrean Montane Forest & Woodland
	M022 - Southern Rocky Mountain Lower Montane Forest
PJ Woodland – Cold	M048 - Central Rocky Mountain Montane-Foothill Grassland & Shrubland
	M168 - Rocky Mountain & Vancouverian Subalpine-High Montane Mesic Meadow
	M501 - Central Rocky Mountain Dry Lower Montane-Foothill Forest
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
	M049 - Southern Rocky Mountain Montane Shrubland
Ponderosa Pine Forest	M010 - Madrean Lowland Evergreen Woodland
	M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland
	M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
	M049 - Southern Rocky Mountain Montane Shrubland
	M048 - Central Rocky Mountain Montane-Foothill Grassland & Shrubland
Sagebrush Shrubland	M168 - Rocky Mountain & Vancouverian Subalpine-High Montane Mesic Meadow
	M118 - Intermountain Basins Cliff, Scree & Badland Sparse Vegetation
	M022 - Southern Rocky Mountain Lower Montane Forest
	M501 - Central Rocky Mountain Dry Lower Montane-Foothill Forest
	M022 - Southern Rocky Mountain Lower Montane Forest
Shortgrass Prairie	M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland
	M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
	M169 - Great Basin & Intermountain Tall Sagebrush Shrubland & Steppe
	M170 - Great Basin & Intermountain Dwarf Sagebrush Shrubland & Steppe
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
Spruce-Fir Forest	M052 - Great Plains Sand Grassland & Shrubland
	M053 - Great Plains Shortgrass Prairie
	M498 - Great Plains Ruderal Grassland & Shrubland
	M512 - North American Warm Desert Ruderal Scrub & Grassland
	M020 - Rocky Mountain Subalpine-High Montane Conifer Forest
	M049 - Southern Rocky Mountain Montane Shrubland
	M168 - Rocky Mountain & Vancouverian Subalpine-High Montane Mesic Meadow
	M022 - Southern Rocky Mountain Lower Montane Forest

#### 4.1.2.2. Western Sierra Madre Piedmont/Madreal Archipelago

Simulations of vegetation for this region indicate drastic transitions in climate and associated habitats (Rehfeldt et al. 2006). Semi-desert grasslands experience the greatest decrease in area of suitable climate (approximately 66% decline) and are largely replaced by Chihuahuan desert scrub type habitat, which increases by more than 400%. Madreal-Tranvolcanic Woodlands and Plains Grasslands Biomes all but disappear.

**Table 4.3.** Predicted percent area suitable for each biome as simulated by Rehfeldt et al. 2006.

Chihuahuan Desertscrub	0.14	0.41	0.61
Madreal-Tranvolcanic Woodland	0.14	0.21	0.01
Plains Grassland	0.05	--	--
Semidesert Grassland	0.66	0.37	0.31
Sonoran Desertscrub	--	--	0.06

The Chihuahuan Desert Scrub Macrogroup [aka to Chihuahuan Creosotebush desert scrub] is stressed by drought, which can reduce vegetation cover though this rarely results significant mortality (Crist et al. 2014). However, drought followed by high intensity precipitation events can increase erosion and drought combined with grazing and reduced wildfire has been associated with the expansion of this habitat type into what were formally grasslands. Chihuahuan Desert Scrub is sensitive to fire, which is increased with the invasion of exotic grass species.

Specific predictions for species within this macrogroup show that *Prosopis juliflora* is expected to expand under climate change (Notaro et al. 2012). In addition, *Quercus toumeyi* (toumey oak), currently occurring in the southernmost part of New Mexico, had one of the largest range expansions within a recent study (Notaro et al. 2012).

From previous studies we know transition between grass and shrub species is highly dependent upon temperature and precipitation. Within Sonoran habitats in Arizona, perennial grasses in mesic mesquite savannas declined with declining precipitation, and cacti increased. *Prosopis velutina* declined in response to increased mean annual temperature (Munson et al. 2012). In xeric upland areas, leguminous *Cercidium microphyllum* declined in hillslopes and *Fouquieria splendens* decreased on south and west facing slopes in response to increasing mean annual temperature (Munson et al., 2012). In xeric shrublands, *Larrea tridentata* and hemiparasite *Krameria grayi* decreased with decreasing cool season precipitation and increasing aridity (Munson et al. 2012). Fire is also a strong determinant of plant community composition. The Montane Chaparral/Warm Interior Chaparral macrogroup [aka Mogollon chaparral ecosystems] needs approximately 20 years to recover from fire and is at risk of decline if future conditions lead to increased wildfire frequency (Crist et al. 2014). The Madreal Montane Forest and Woodland macrogroup [madreal montane conifer-oak forest and woodland system] can change rapidly with changes to fire regime. In general, oaks are favored under more frequent

fire regime. Climate change can affect insect and disease outbreaks within these forests (Crist et al. 2014)

The Chihuahuan Semi-Grassland macrogroup [aka Apacherian-chihuahuan semi-desert grassland and steppe systems] is a fire dependent ecosystem. Though there is expectations for widespread increases in fire frequency (Moritz et al. 2009), this could encourage invasive species like Lehmann and Boer lovegrasses (*Eragrostis lehmanniana* and *Eragrostis curvula*) to the detriment of this habitat. Erosion of topsoil is also an issue for this habitat type especially where drought and high precipitation events alternate (Crist et al. 2014).

Within the Sky Island region of southern Arizona, increased temperatures alone led to increases in desert scrub habitat and upslope movement of most other habitat types (Kupfer and et al. 2005). Madrean Evergreen classes and areas near ecotones were considered most sensitive to negative impacts due to climate change in this region (Kupfer et al. 2005). The Madrean Lowland Evergreen Woodland macrogroup [aka Madrean pinyon-juniper woodland ecological systems] is affected by climate, drought, insects, pathogens, and herbivory (Crist et al. 2014). Climate change is known to have influenced the distribution of this habitat. Increase winter precipitation has been associated with its expansion. Drought often leads to insect outbreaks within stressed trees resulting in widespread mortality. Areas with this type of habitat are highly susceptible to erosion (Crist et al. 2014).

Simulations show substantial declines in suitable habitat for dominant tree species found within the Madrean lowland evergreen woodland, Madrean Montane Forest and Woodland, and Madrean Lowland Evergreen Woodland macrogroups. *Pinus edulis*, *Picea engelmannii* and *Juniperus osteosperma* under future climate regimes (Rehfeldt et al. 2006). Southwest ponderosa pine and Douglas-fir are likely to be at greatest risk for drought induced mortality at lower level elevations, whereas piñon pine appears sensitive throughout its range. Douglas fir, *P. menziesii*, experiences less loss of suitable area though those areas with suitable conditions shift substantial across the landscape (Rehfeldt et al. 2006). By the end of the century, only 54% of current *P. menziesii* habitat overlaps with future distribution of ideal conditions (Rehfeldt et al. 2006). Much of its range within southern New Mexico will become unsuitable by the end of the century. Overall, *Pinus leiophylla* is expected to shift northward by as much as 400 km and as much as 100m up slope by the end of the century (Rehfeldt et al. 2006). *Pseudotsuga menziesii*, *P. ponderosa*, and *P. edulis* may shift upslope by as much as 500 m, *P. engelmannii* and *Q. gambelii* 300 m, and *Juniperus osteosperma* by about 100m (Rehfeldt et al. 2006). In addition, growth rates of *Pinus edulis*, *P. ponderosa* and *Pseudotsuga menziesii* will decrease across the southwest, including New Mexico, as a result of changing climates (Williams et al. 2010).

Within the Semi-Desert Grassland macrogroup simulations show general declines for *Juniperus osteosperma* under future climate regimes (Rehfeldt et al. 2006). *Populus angustifolia* (narrowleaf cottonwood), present in patches in the region, is predicted to have significant range contractions under future conditions (Notaro et al. 2012).

Triepke et al. (2014) performed a vulnerability analysis for Ecological Response Units (ERU) across New Mexico and Arizona. The ERU classification system represents major ecosystem types of the Southwest, and groups vegetation according to site potential and historic disturbance regime such that each plant community within an ERU shares successional patterns, physiognomy and community dynamics (Triepke et al. 2014). In most cases ERUs are coarser than LANDFIRE Biophysical Settings, but are identical in concept. Vulnerability for each ERU, represented by image segments averaging 10-20 ha, was calculated as the relative probability of type conversion given the breadth of the climate envelope of the ERU, the magnitude of projected climate change at a given location, and the historic climate of the ERU as reflected its climate envelope. Semi-Desert Grassland, Chihuahuan Desert Scrub, Madrean Encinal Woodlands, Madrean Pinyon-Oak Woodland, Juniper Grass, Shortgrass Prairie, Interior Chaparral, Pinyon-Juniper Evergreen Shrub and Mixed Conifer with Frequent Fire were the predominate (>1% of area) ERU's within the Madrean Archipelago Ecoregion (Table 4.4). Of these, most (greater than 40%) of the Semi-Desert Grassland, Chihuahuan Desert Scrub, Juniper Grass and Mixed Conifer with Frequent Fire were classified as very highly vulnerable to climate change impacts. However, this analysis may overestimate vulnerability of Chihuahuan Desert Scrub Habitat because Triepke et al. (2014) estimate its climate envelope as it exists within New Mexico. New Mexico represents the northernmost extent of Chihuahuan Desert Scrub habitat and, therefore, its estimated climate envelope is based on a relatively cooler and wetter habitat as compared to its overall range which includes much of northeastern Mexico. Thirty-six percent of Madrean Encinal woodlands were classified as highly vulnerable and 39% as very highly vulnerable. Sixty-four percent of Madrean Pinyon-oak woodlands were classified at medium vulnerability. More than 90% of Interior Chaparral and Pinyon-Juniper Evergreen Shrub were classified at moderate vulnerability.

**Table 4.4.** Macrogroup-Ecological Response Units (ERU) crosswalk for ERUs with >1% area representation within the Ecoregion.

<b>ERU</b>	<b>Macrogroup</b>
<b>Madrean</b>	
Chihuahuan Desert Scrub	M086 - Chihuahuan Desert Scrub M087 - Chihuahuan Semi-Desert Grassland M512 - North American Warm Desert Ruderal Scrub & Grassland
Interior Chaparral	M087 - Chihuahuan Semi-Desert Grassland M088 - Mojave-Sonoran Semi-Desert Scrub M171 - Great Basin & Intermountain Dry Shrubland & Grassland M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
Juniper Grass – High Sun Mild	M043 - California Chaparral M087 - Chihuahuan Semi-Desert Grassland M086 - Chihuahuan Desert Scrub M171 - Great Basin & Intermountain Dry Shrubland & Grassland M010 - Madrean Lowland Evergreen Woodland



	M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland
	M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
	M118 - Intermountain Basins Cliff, Scree & Badland Sparse Vegetation
Madrean Encinal Woodland	M088 - Mojave-Sonoran Semi-Desert Scrub
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
	M091 - Warm Interior Chaparral
	M010 - Madrean Lowland Evergreen Woodland
	M011 - Madrean Montane Forest & Woodland
Madrean Pinyon-Oak Woodland	M010 - Madrean Lowland Evergreen Woodland
	M091 - Warm Interior Chaparral
	M088 - Mojave-Sonoran Semi-Desert Scrub
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
	M011 - Madrean Montane Forest & Woodland
Mixed Conifer – Frequent Fire	M011 - Madrean Montane Forest & Woodland
	M022 - Southern Rocky Mountain Lower Montane Forest
	M048 - Central Rocky Mountain Montane-Foothill Grassland & Shrubland
	M168 - Rocky Mountain & Vancouverian Subalpine-High Montane Mesic Meadow
	M501 - Central Rocky Mountain Dry Lower Montane-Foothill Forest
PJ Evergreen Shrub	M010 - Madrean Lowland Evergreen Woodland
	M011 - Madrean Montane Forest & Woodland
	M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
	M043 - California Chaparral
	M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
Semi-Desert Grassland – High Sun Mild	M086 - Chihuahuan Desert Scrub
	M088 - Mojave-Sonoran Semi-Desert Scrub
	M091 - Warm Interior Chaparral
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
	M087 - Chihuahuan Semi-Desert Grassland
	M088 - Mojave-Sonoran Semi-Desert Scrub
	M512 - North American Warm Desert Ruderal Scrub & Grassland
	M010 - Madrean Lowland Evergreen Woodland

#### 4.1.2.3. South Central, Semi-arid Prairies/High Plains

Great Basin Conifer Woodland habitat is projected to decline within this ecoregion (Rehfeldt et al. 2006). Semi-Desert Grassland type climate expands within this region, whereas Plains grassland type climate decline by more than 50%. Climate suitable for Chihuahuan desert scrub begins to appear by 2030 and accounts for almost 10% of the total area by 2090.

**Table 4.5.** Predicted percent area suitable for each biome as simulated by Rehfeldt et al. 2006.

<b>SOUTH CENTRAL SEMIARID PRAIRIES</b>	<b>Current</b>	<b>2030</b>	<b>2090</b>
Chihuahuan Desertscrub	--	0.02	0.09
Great Basin Conifer Woodland	0.36	0.24	0.12
Plains Grassland	0.45	0.37	0.17
Semidesert Grassland	0.18	0.37	0.62

Within the Semi-Desert Grassland macrogroup simulations show general declines for *Juniperus osteosperma* under future climate regimes (Rehfeldt et al. 2006).

The Chihuahuan Desert Scrub macrogroup [akin to Chihuahuan Creosotebush desert scrub] is stressed by drought which can reduce vegetation cover though it rarely causes significant mortality (Crist et al. 2014). Drought followed by high intensity precipitation events can increase erosion. Drought combined with grazing and reduce wildfire has been associated with the expansion of this this habitat type into what were formally grasslands. This habitat type is sensitive to fire, which is increased with the invasion of exotic grass species.

Within the Chihuahuan Desert Scrub macrogroup, conditions suitable for *Prosopis juliflora* are expected to expand (Notaro et al. 2012) and *Quercus toumeyi* (toumey oak), currently occurring in the southernmost part of New Mexico, experiences a large expansion (Notaro et al. 2012). *Ostrya knowltonii* (Knowlton's hophornbeam) is expected to have the largest range expansion of deciduous trees in the entire SW and *Populus angustifolia* (narrowleaf cottonwood) is predicted to have the largest range contraction (Notaro et al. 2012).

The Chihuahuan Semi-Desert Grassland macrogroup [aka Apacherian-Chihuahuan Semi-Desert Grassland and steppe systems] is fire dependent though increased fire due to invasion by Lehmann and Boer lovegrasses (*Eragrostis lehmanniana* and *Eragrostis curvula*) will be problematic for this area. Erosion of topsoil is an issue for this habitat type especially where drought and high precipitation events alternate (Crist et al. 2014).

Triepke et al. (2014) performed a vulnerability analysis for all ecosystems across New Mexico and Arizona. Vulnerability was considered the relative probability of type conversion given the breadth of the climate envelope of the ecosystem, and the magnitude of projected climate change, and pre-1990 climate at a given location.

Triepke et al. (2014) performed a vulnerability analysis for Ecological Response Units (ERU) across New Mexico and Arizona. The ERU classification system represents major ecosystem types of the Southwest, and groups vegetation according to site potential and historic disturbance regime such that each plant community within an ERU shares successional patterns, physiognomy and community dynamics (Triepke et al. 2014). In most cases ERUs are coarser than LANDFIRE Biophysical Settings, but are identical in concept. Vulnerability for each ERU, represented by image segments averaging 10-20 ha, was calculated as the relative probability of type conversion given the breadth of the climate envelope of the ERU, the magnitude of projected climate change at a given location, and the historic climate of the ERU

as reflected its climate envelope. Short-Grass Prairie, Semi-Desert Grassland, Juniper Grass, Mixed Grass Prairie, Shinnery Oak, Sandsage, Ponderosa Pine Forest, Pinyon-Juniper Woodland and Pinyon-Juniper Grass were the predominate (>1% of area) ERU's within the High Plains Ecoregion (Table 4.6). Of these, most of the Shortgrass prairie (57%), Mixed-Grass Prairie (85%), Shinnery Oak (41%), Sandsage (42%), and Pinyon-Juniper Grass habitats (42%) were classified as highly vulnerable. Sixty-two percent of Semi-desert grassland habitat was highly vulnerable and 70% of Juniper grass was classified as having medium vulnerability. Ponderosa Pine Forest and Pinyon-Juniper Woodlands predominantly fell within the medium vulnerability class.

**Table 4.6.** Macrogroup-Ecological Response Units (ERU) crosswalk for ERUs with >1% area representation within the Ecoregion.

<b>ERU</b>	<b>Macrogroup</b>
Juniper Grass –	M043 - California Chaparral
Cold Juniper Grass	M087 - Chihuahuan Semi-Desert Grassland
– High Sun Mild	M086 - Chihuahuan Desert Scrub
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
	M010 - Madrean Lowland Evergreen Woodland
	M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland
	M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
	M118 - Intermountain Basins Cliff, Scree & Badland Sparse Vegetation
Mixed-Grass Prairie	M051 - Great Plains Mixedgrass & Fescue Prairie
PJ Grass – Cold	M010 - Madrean Lowland Evergreen Woodland
Temp	M011 - Madrean Montane Forest & Woodland
	M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
	M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
PJ Woodland –	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
Cold	M049 - Southern Rocky Mountain Montane Shrubland
	M010 - Madrean Lowland Evergreen Woodland
	M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland
	M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
Ponderosa Pine	M049 - Southern Rocky Mountain Montane Shrubland
Forest	M048 - Central Rocky Mountain Montane-Foothill Grassland & Shrubland
	M168 - Rocky Mountain & Vancouverian Subalpine-High Montane Mesic Meadow
	M118 - Intermountain Basins Cliff, Scree & Badland Sparse Vegetation
	M022 - Southern Rocky Mountain Lower Montane Forest
	M501 - Central Rocky Mountain Dry Lower Montane-Foothill Forest

Sandsage	M052 - Great Plains Sand Grassland & Shrubland
	M053 - Great Plains Shortgrass Prairie
	M086 - Chihuahuan Desert Scrub
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
	M052 - Great Plains Sand Grassland & Shrubland
Semi-Desert Grassland – High Sun	M087 - Chihuahuan Semi-Desert Grassland
	M086 - Chihuahuan Desert Scrub
	M088 - Mojave-Sonoran Semi-Desert Scrub
	M091 - Warm Interior Chaparral
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
	M087 - Chihuahuan Semi-Desert Grassland
	M088 - Mojave-Sonoran Semi-Desert Scrub
	M512 - North American Warm Desert Ruderal Scrub & Grassland
	M010 - Madrean Lowland Evergreen Woodland
	M052 - Great Plains Sand Grassland & Shrubland
Shortgrass Prairie	M053 - Great Plains Shortgrass Prairie
	M498 - Great Plains Ruderal Grassland & Shrubland
	M512 - North American Warm Desert Ruderal Scrub & Grassland

#### 4.1.2.4. Cold Deserts/Colorado Plateaus

The greatest change for this region appears to be an almost complete loss of climate conditions that support Plains Grassland type habitats (from 20% of current area to 3% of area by 2090) and an increase in Great Basin Desert scrub habitats (Rehfeldt; Table 4.7). Great Basin Shrub-grasslands are projected to no longer have any area with suitable habitat within this region. Instead, many of these areas shift to Great-Basin Desert scrub. Great Basin Conifer woodlands type climates shift but do not experience a great deal of change in total area.

**Table 4.7.** Predicted percent area suitable for each biome as simulated by Rehfeldt et al. 2006.

<b>COLD DESERTS</b>	<b>Current</b>	<b>2030</b>	<b>2090</b>
Chihuahuan Desertscrub	--	0.01	0.07
Great Basin Conifer Woodland	0.46	0.68	0.51
Great Basin Desertscrub	0.05	0.17	0.30
Great Basin Shrub-Grassland	0.23	0.06	--
Plains Grassland	0.20	0.02	0.03
Rocky Mountain Montane Conifer Forest	--	0.01	--
Semidesert Grassland	0.06	0.06	0.09

In cold desert shrublands, increases in fire frequency due to invasion of exotic annuals (e.g. *Bromus tectorum*), has led to the simplification of shrubland habitats (Knapp 1996, Ford et al. 2012). *Artemisia tridentata* (big sagebrush) a cold desert species was also among the highest ranked species for range contractions under climate change (Notaro et al. 2012).

The Chihuahuan Semi-Desert Grassland macrogroup [aka Apacherian-chihuahuan semi-desert grassland and steppe systems] is fire dependent though increased fire due to invasion by Lehmann and Boer lovegrasses (*Eragrostis lehmanniana* and *Eragrostis curvula*) will be problematic for this area. Erosion of topsoil is an issue for this habitat type especially where drought and high precipitation events alternate (Crist et al. 2014). Within the Semi-desert grassland macrogroup simulations show general declines for *Juniperus osteosperma* under future climate regimes (Rehfeldt et al. 2006).

The Chihuahuan Desert Scrub macrogroup [akin to Chihuahuan Creosotebush desert scrub] is stressed by drought which can reduce vegetation cover though it rarely causes significant mortality (Crist et al. 2014). Drought followed by high intensity precipitation events can increase erosion. Drought combined with grazing and reduce wildfire has been associated with the expansion of this habitat type into what were formally grasslands. This habitat type is sensitive to fire, which is increased with the invasion of exotic grass species.

Simulations show general declines for dominant tree species including *Pinus edulis*, *Picea engelmannii* and *Juniperus osteosperma* under future climate regimes (Rehfeldt et al. 2006). *Pinus flexilis* (limber pine) an alpine species located in northern New Mexico was projected to have one of the largest range contractions within all evergreens across the Southwest (Notaro et al. 2012). Williams et al. (2010) estimated that future growth of *Pinus edulis*, *P. ponderosa* and *Pseudotsuga menziesii* will decrease across the southwest, including New Mexico. By the end of the century, *Pseudotsuga menziesii*, *P. ponderosa*, and *P. edulis* may shift upslope by as much as 500 m, *P. engelmannii* and *Q. gambelii* 300 m, and *Juniperus osteosperma* by about 100m (Rehfeldt et al. 2006). *Quercus gambelii* was the only species to exhibit a potential increase in suitable area (Rehfeldt et al. 2006).

Moisture stress and winter cold are limiting for *P. ponderosa* (Rehfeldt et al. 2006). While cold limitations may ease, precipitation changes are likely to play a dominate role in persistence of these species (Rehfeldt et al. 2006). *Picea engelmannii* is a high elevation evergreen conifer and the most abundant tree species in the southwestern U.S. is projected to decline substantially across the entire southwest (Notaro et al. 2012). Southwestern ponderosa pine and Douglas-fir are likely to be at greatest risk for drought induced mortality at lower level elevations, whereas piñon pine appears sensitive throughout its range.

Rehfeldt 2006 notes that projections for *P. ponderosa* and *Q. gambelii*, two important species in southern montane forests, do not respond similarly to climate change nor do they mimic projections for the Montane forests biomes (slight increase in area overall). Similarly individual results for *J. osteosperma* and *P. edulis*, indicator species for the Great Basin conifer woodlands (most similar to Intermountain woodland macrogroup or Southern rocky mountain woodland macrogroup), show that they would co-occur only by 6% (from current 35%). Therefore, it is likely that where conditions support great basin woodlands (which increase over time), species composition within those woodlands is likely to be quite different from present.

Triepke et al. (2014) performed a vulnerability analysis for Ecological Response Units (ERU) across New Mexico and Arizona. The ERU classification system represents major ecosystem types of the Southwest, and groups vegetation according to site potential and historic disturbance regime such that each plant community within an ERU shares successional patterns, physiognomy and community dynamics (Triepke et al. 2014). In most cases ERUs are coarser than LANDFIRE Biophysical Settings, but are identical in concept. Vulnerability for each ERU, represented by image segments averaging 10-20 ha, was calculated as the relative probability of type conversion given the breadth of the climate envelope of the ERU, the magnitude of projected climate change at a given location, and the historic climate of the ERU as reflected its climate envelope. Great Basin grassland, Sagebrush shrubland, Pinyon-Juniper grass, woodland and sagebrush, Juniper grass, Ponderosa Pine forest, Short-grass prairie, Intermountain Salt Scrub, Semi-desert grassland, Mixed Conifer with frequent fire, Shinnery Oak, Sandsage, and Ponderosa Pine forest were the predominate (>1% of area) ERU's (Table 4.8). The Great Basin Grassland, Shortgrass Prairie, Pinyon-Juniper Sagebrush, and Sandsage were the most vulnerable and given a Very Highly Vulnerable classification. Location within the landscape had an influence on vulnerability classification. The majority (69%) of Juniper Grass in areas with high insolation was classified as highly vulnerable, whereas only 37% of Juniper Grass in areas with lower insolation was considered highly vulnerable. Most other ERU's, including Sagebrush Shrubland, Semi-desert grasslands, Mixed-conifer with frequent fire, Pinyon-Juniper woodlands were found to have between 30-50% of their area within high or moderate vulnerability classes. Sagebrush Shrubland was the only ERU with a significant proportion (38%) of habitat classified as low vulnerability.

**Table 4.8.** Macrogroup-Ecological Response Units (ERU) crosswalk for ERUs with >1% area representation within the Ecoregion.

<b>ERU</b>	<b>Macrogroup</b>
Colorado Plateau / Great Basin Grassland	M118 - Intermountain Basins Cliff, Scree & Badland Sparse Vegetation
	M169 - Great Basin & Intermountain Tall Sagebrush Shrubland & Steppe
	M170 - Great Basin & Intermountain Dwarf Sagebrush Shrubland & Steppe
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
	M499 - Western North American Cool Semi-Desert Ruderal Scrub & Grassland
Intermountain Salt Scrub	M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland
	M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
	M082 - Warm & Cool Desert Alkali-Saline Wetland
	M093 - Great Basin Saltbush Scrub
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
Juniper Grass – Cold Juniper Grass	M118 - Intermountain Basins Cliff, Scree & Badland Sparse Vegetation
	M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
	M043 - California Chaparral
	M087 - Chihuahuan Semi-Desert Grassland



– High Sun Mild	M086 - Chihuahuan Desert Scrub M171 - Great Basin & Intermountain Dry Shrubland & Grassland M010 - Madrean Lowland Evergreen Woodland M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland M118 - Intermountain Basins Cliff, Scree & Badland Sparse Vegetation
Mixed Conifer – Frequent Fire	M011 - Madrean Montane Forest & Woodland M022 - Southern Rocky Mountain Lower Montane Forest M048 - Central Rocky Mountain Montane-Foothill Grassland & Shrubland M168 - Rocky Mountain & Vancouverian Subalpine-High Montane Mesic Meadow M501 - Central Rocky Mountain Dry Lower Montane-Foothill Forest
PJ Grass – Cold Temp	M010 - Madrean Lowland Evergreen Woodland M011 - Madrean Montane Forest & Woodland M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland M171 - Great Basin & Intermountain Dry Shrubland & Grassland M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
PJ Sagebrush	M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland M169 - Great Basin & Intermountain Tall Sagebrush Shrubland & Steppe M171 - Great Basin & Intermountain Dry Shrubland & Grassland M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
PJ Woodland – Cold	M171 - Great Basin & Intermountain Dry Shrubland & Grassland M049 - Southern Rocky Mountain Montane Shrubland M010 - Madrean Lowland Evergreen Woodland M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
Ponderosa Pine Forest	M049 - Southern Rocky Mountain Montane Shrubland M048 - Central Rocky Mountain Montane-Foothill Grassland & Shrubland M168 - Rocky Mountain & Vancouverian Subalpine-High Montane Mesic Meadow M118 - Intermountain Basins Cliff, Scree & Badland Sparse Vegetation M022 - Southern Rocky Mountain Lower Montane Forest M501 - Central Rocky Mountain Dry Lower Montane-Foothill Forest
Sagebrush Shrubland	M022 - Southern Rocky Mountain Lower Montane Forest M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland

Sandsage	M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
	M169 - Great Basin & Intermountain Tall Sagebrush Shrubland & Steppe
	M170 - Great Basin & Intermountain Dwarf Sagebrush Shrubland & Steppe
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
	M052 - Great Plains Sand Grassland & Shrubland
	M053 - Great Plains Shortgrass Prairie
	M086 - Chihuahuan Desert Scrub
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
	M052 - Great Plains Sand Grassland & Shrubland
	M087 - Chihuahuan Semi-Desert Grassland
Semi-Desert Grassland – High Sun Mild	M086 - Chihuahuan Desert Scrub
	M088 - Mojave-Sonoran Semi-Desert Scrub
	M091 - Warm Interior Chaparral
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
	M087 - Chihuahuan Semi-Desert Grassland
	M088 - Mojave-Sonoran Semi-Desert Scrub
	M512 - North American Warm Desert Ruderal Scrub & Grassland
	M010 - Madrean Lowland Evergreen Woodland
	M052 - Great Plains Sand Grassland & Shrubland
	M053 - Great Plains Shortgrass Prairie
Shortgrass Prairie	M498 - Great Plains Ruderal Grassland & Shrubland
	M512 - North American Warm Desert Ruderal Scrub & Grassland

#### 4.1.2.5. Warm Deserts/ Chihuahuan Desert

This region is likely to experience a sharp increase in climate conditions that support Chihuahuan desert scrub and reduction in areas that support semidesert grasslands (Table 4.5; Rehfeldt et al. 2006). Areas with climates that support woodlands disappear completely by midcentury.

**Table 4.9.** Predicted percent area suitable for each biome as simulated by Rehfeldt et al. 2006.

<b>WARM DESERTS</b>	<b>Current</b>	<b>2030</b>	<b>2090</b>
Chihuahuan Desertscrub	0.42	0.66	0.79
Great Basin Conifer Woodland	0.16	0.04	--
Madrean-Transvolcanic Woodland	0.01	--	--
Plains Grassland	0.01	--	--
Semidesert Grassland	0.40	0.30	0.16
Sonoran Desertscrub	--	--	0.04

The Chihuahuan Desert Scrub macrogroup [akin to Chihuahuan Creosotebush desert scrub] is stressed by drought which can reduce vegetation cover though it rarely causes significant

mortality (Crist et al. 2014). Drought followed by high intensity precipitation events can increase erosion. Drought combined with grazing and reduce wildfire has been associated with the expansion of this this habitat type into what were formally grasslands. This habitat type is sensitive to fire, which is increased with the invasion of exotic grass species. Specific predictions for species within this macrogroup show that *Prosopis juliflora* is expected to expand under climate change (Notaro et al. 2012). In addition, *Quercus toumeyi* (toumey oak), currently occurring in the southernmost part of New Mexico, had one of the largest range expansions within a recent study (Notaro et al. 2012). Interestingly, *Ostrya knowltonii* (Knowlton's hophornbeam) is expected to have the largest range expansion of deciduous trees in the entire Southwest and *Populus angustifolia* (narrowleaf cottonwood) is predicted to have the largest range contraction (Notaro et al. 2012).

Simulations show substantial declines in suitable habitat for dominant tree species found within the Madrean lowland evergreen woodland and Madrean montane forest and woodland macrogroups. *Pinus edulis*, *Picea engelmannii* and *Juniperus osteosperma* under future climate regimes (Rehfeldt et al. 2006). Southwest ponderosa pine and Douglas-fir are likely to be at greatest risk for drought induced mortality at lower level elevations, whereas pinyon pine appears sensitive throughout its range. Douglas fir, *P. menziesii*, experiences less loss of suitable area though those areas with suitable conditions shift substantial across the landscape (Rehfeldt et al. 2006). By the end of the century, only 54% of current *P. menziesii* habitat overlaps with future distribution of ideal conditions (Rehfeldt et al. 2006). Much of its range within southern New Mexico will become unsuitable by the end of the century. Overall, *Pinus leiophylla* is expected to shift northward by as much as 400 km and as much as 100m up slope by the end of the century (Rehfeldt et al. 2006). *Pseudotsuga menziesii*, *P. ponderosa*, and *P. edulis* may shift upslope by as much as 500 m,, *P. engelmannii* and *Q. gambelii* 300 m, and *Juniperus osteosperma* by about 100m (Rehfeldt et al. 2006). In addition, growth rates of *Pinus edulis*, *P. ponderosa* and *Pseudotsuga menziesii* will decrease across the southwest, including New Mexico, as a result of changing climates (Williams et al. 2010).

The Chihuahuan Semi-Desert Grassland macrogroup [aka Apacherian-chihuahuan semi-desert grassland and steppe systems] is fire dependent though increased fire due to invasion by Lehmann and Boer lovegrasses (*Eragrostis lehmanniana* and *Eragrostis curvula*) will be problematic for this area. Erosion of topsoil is an issue for this habitat type especially where drought and high precipitation events alternate (Crist et al. 2014). Within the Semi-desert grassland macrogroup simulations show general declines for *Juniperus osteosperma* under future climate regimes (Rehfeldt et al. 2006).

Triepke et al. (2014) performed a vulnerability analysis for Ecological Response Units (ERU) across New Mexico and Arizona. The ERU classification system represents major ecosystem types of the Southwest, and groups vegetation according to site potential and historic disturbance regime such that each plant community within an ERU shares successional patterns, physiognomy and community dynamics (Triepke et al. 2014). In most cases ERUs are coarser than LANDFIRE Biophysical Settings, but are identical in concept. Vulnerability for each ERU, represented by image segments averaging 10-20 ha, was calculated as the relative

probability of type conversion given the breadth of the climate envelope of the ERU, the magnitude of projected climate change at a given location, and the historic climate of the ERU as reflected its climate envelope. Semi-Desert Grassland, Chihuahuan Desert Scrub, Chihuahuan Salt Desert Scrub, Short-Grass Prairie, Shinnery Oak, Pinyon-Juniper Grass And Evergreen Shrub, Juniper Grass, Sandsage, Mountain Mahogany Mixed Shrubland, Madrean Pinyon-Oak Woodland, and Interior Chaparral were the predominate (>1% of area) ERU's within the Chihuahuan Desert Ecoregion (Table 4.10). Six of these ERU's, Semi-Desert Grassland, Chihuahuan Desert Scrub, Chihuahuan Salt Desert Scrub, Short-Grass Prairie, Shinnery Oak, and Sandsage had 70% or more of their area with vulnerability of very high to climate change impacts. However, this analysis may overestimate vulnerability of Chihuahuan desert scrub and salt desert scrub habitats because Triepke et al. (2014) estimate their climate envelope as they exist within New Mexico. New Mexico represents the northernmost extent of Chihuahuan desert scrub habitat and, therefore, its estimated climate envelope is based on a relatively cooler and wetter habitat as compared to its overall range which includes much of Mexico. Forty-nine percent of Pinyon-Juniper Grass was considered Very Highly Vulnerable. Juniper Grass and Mountain Mahogany mixed shrubland fell more evenly across Medium, High and Very High Vulnerability. Madrean Pinyon-oak woodland (45%) and Interior Chaparral (92%) tended to be within the Medium Vulnerability class.

**Table 4.10.** Macrogroup-Ecological Response Units (ERU) crosswalk for ERUs with >1% area representation within the Ecoregion.

<b>ERU</b>	<b>Macrogroup</b>
Chihuahuan Desert Scrub	M086 - Chihuahuan Desert Scrub M087 - Chihuahuan Semi-Desert Grassland M512 - North American Warm Desert Ruderal Scrub & Grassland
Chihuahuan Salt Desert Scrub	M086 - Chihuahuan Desert Scrub
Interior Chaparral	M087 - Chihuahuan Semi-Desert Grassland M088 - Mojave-Sonoran Semi-Desert Scrub M171 - Great Basin & Intermountain Dry Shrubland & Grassland M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
Juniper Grass – High Sun Mild	M043 - California Chaparral M087 - Chihuahuan Semi-Desert Grassland M086 - Chihuahuan Desert Scrub M171 - Great Basin & Intermountain Dry Shrubland & Grassland M010 - Madrean Lowland Evergreen Woodland M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland M118 - Intermountain Basins Cliff, Scree & Badland Sparse Vegetation

Madrean Pinyon-Oak Woodland	M010 - Madrean Lowland Evergreen Woodland
	M091 - Warm Interior Chaparral
	M088 - Mojave-Sonoran Semi-Desert Scrub
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
	M011 - Madrean Montane Forest & Woodland
Mountain Mahogany Mixed Shrubland	M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland
	M022 - Southern Rocky Mountain Lower Montane Forest
	M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
	M043 - California Chaparral
	M049 - Southern Rocky Mountain Montane Shrubland
	M091 - Warm Interior Chaparral
	M094 - Cool Interior Chaparral
	M168 - Rocky Mountain & Vancouverian Subalpine-High Montane Mesic Meadow
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
	M010 - Madrean Lowland Evergreen Woodland
PJ Evergreen Shrub	M011 - Madrean Montane Forest & Woodland
	M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
	M043 - California Chaparral
	M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
PJ Grass – High Sun Mild	M010 - Madrean Lowland Evergreen Woodland
	M011 - Madrean Montane Forest & Woodland
	M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
	M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
Sandsage	M052 - Great Plains Sand Grassland & Shrubland
	M053 - Great Plains Shortgrass Prairie
	M086 - Chihuahuan Desert Scrub
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
	M052 - Great Plains Sand Grassland & Shrubland
Semi-Desert Grassland – High Sun	M087 - Chihuahuan Semi-Desert Grassland
	M086 - Chihuahuan Desert Scrub
	M088 - Mojave-Sonoran Semi-Desert Scrub
	M091 - Warm Interior Chaparral
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
	M087 - Chihuahuan Semi-Desert Grassland
	M088 - Mojave-Sonoran Semi-Desert Scrub

Shortgrass Prairie	M512 - North American Warm Desert Ruderal Scrub & Grassland
	M010 - Madrean Lowland Evergreen Woodland
	M052 - Great Plains Sand Grassland & Shrubland
	M053 - Great Plains Shortgrass Prairie
	M498 - Great Plains Ruderal Grassland & Shrubland
	M512 - North American Warm Desert Ruderal Scrub & Grassland

#### 4.1.2.6. Upper Gila Mountains/Arizona New Mexico Mountains

This region contains a diverse set of habitats. Under future climate conditions, Chihuahuan desert scrub and semidesert grasslands are likely to experience range expansions (Table 4.5; Rehfeldt et al. 2006). High elevation forests (e.g. Rocky Mountain Montane Conifer Forests) are expected to decline, probably replaced by Great Basin Conifer Woodland type habitats that begin to shift up in elevation.

**Table 4.11.** Predicted percent area suitable for each biome as simulated by Rehfeldt et al. 2006.

<b>UPPER GILA MOUNTAINS</b>	<b>Current</b>	<b>2030</b>	<b>2090</b>
Chihuahuan Desertscrub	0.01	0.05	0.10
Great Basin Conifer Woodland	0.60	0.61	0.50
Great Basin Shrub-Grassland	0.01	--	--
Great Basin Desertscrub	--	--	0.01
Madrean-Transvolcanic Woodland	0.02	0.03	--
Plains Grassland	0.01	--	0.01
Rocky Mountain Montane Conifer Forest	0.32	0.18	0.08
Semidesert Grassland	0.03	0.13	0.29

Both *Prosopis juliflora* and *Quercus toumeyi* (toumey oak), currently within the Chihuahuan Desert Scrub Macrogroup [aka Chihuahuan Creosotebush desert scrub], have among the largest range expansions under a recent simulation analysis (Notaro et al. 2012). In general, Chihuahuan desert scrub is stressed by drought, which can reduce vegetation cover though rarely causes significant mortality (Crist et al. 2014). However, drought combined with grazing and reduced wildfire has been associated with the expansion of this habitat type into grassland habitats. Drought followed by high intensity precipitation events can increase erosion not only for this vegetation group but for Chihuahuan semi-grassland systems (aka Apacherian-chihuahuan semi-desert grassland and steppe systems). And for both ecosystems, climate changes which favor Lehmann and Boer lovegrasses (*Eragrostis lehmanniana* and *Eragrostis curvula*) will be problematic.

Madrean lowland evergreen woodland species (aka pinyon-juniper woodland ecological systems) are affected by climate, drought insects, pathogens, and herbivory (Crist et al. 2014). Drought often leads to insect outbreaks within stressed trees resulting in widespread mortality.

Increased drought and associated increases in fire are likely to cause widespread mortality of trees and change to forest composition. Within Madrean montane forest and woodland macrogroup increased fire is likely to favor oak species. Fire return intervals shorter than 20 years will negatively impact the Montane chaparral and warm interior chaparral macrogroups (aka Mogollon chaparral ecosystems in Crist et al. 2014). Increased winter precipitation has been associated with its expansion. Areas within this type of habitat are highly susceptible to erosion (Crist et al. 2014).

For woodland and forest macrogroups in this region, simulations show general declines in area of suitable habitat for dominant tree species including *Pinus edulis*, *Picea engelmannii*, *Pseudotsuga menziesii*, and *Juniperus osteosperma* under future climate regimes (Rehfeldt et al. 2006; Williams 2010). With these declines, many of these species also shift to the north and upslope. Specifically, ideal conditions for *Pinus leiophylla* are expected to shift northward on average by as much as 400 km and as much as 100m up slope by the end of the century (Rehfeldt et al. 2006). *P. menziesii*, *P. ponderosa*, and *P. edulis* may shift upwards in elevation by 500 m, *P. engelmannii* and *Q. gambelii* 300 m, and *Juniperus osteosperma* by about 100m upslope (Rehfeldt et al. 2006). Higher elevation species like Ponderosa pine and Douglas-fir are likely to be at greatest risk for drought induced mortality at lower level elevations, whereas pinyon pine appears sensitive throughout its range (Williams et al. 2010). In contrast, *Quercus gambelii* is expected to experience an increase of suitable area across the region (Rehfeldt et al. 2006). *Populus angustifolia* (narrowleaf cottonwood) is predicted to have the largest range contraction (Notaro et al. 2012).

Triepke et al. (2014) performed a vulnerability analysis for Ecological Response Units (ERU) across New Mexico and Arizona. The ERU classification system represents major ecosystem types of the Southwest, and groups vegetation according to site potential and historic disturbance regime such that each plant community within an ERU shares successional patterns, physiognomy and community dynamics (Triepke et al. 2014). In most cases ERUs are coarser than LANDFIRE Biophysical Settings, but are identical in concept. Vulnerability for each ERU, represented by image segments averaging 10-20 ha, was calculated as the relative probability of type conversion given the breadth of the climate envelope of the ERU, the magnitude of projected climate change at a given location, and the historic climate of the ERU as reflected its climate envelope. The Arizona New Mexico Mountain Ecoregion contains the greatest number of ERUs (Table 4.12). The 10 most dominate (>3% of total area) include Semi-Desert Grassland, Great Basin Grassland, Ponderosa Pine Forest, Pinyon-Juniper Grass (two types of insolation), Shortgrass Prairie, Pinyon-Juniper Evergreen Shrub, Mixed Conifer with Frequent Fire, and Pinyon-Juniper Woodland. Unlike ERU's from other ecoregions, the majority of types here fell within the Medium vulnerability class. Only Shortgrass Prairie had a high proportion (63%) of land classified as Very High Vulnerability. Pinyon-Juniper Evergreen Shrub was classified as 50% Low vulnerability and 49% Medium Vulnerability.

**Table 4.12.** Macrogroup-Ecological Response Units (ERU) crosswalk for ERUs with >1% area representation within the Ecoregion.

**ERU**

**Macrogroup**

Chihuahuan Desert Scrub	M086 - Chihuahuan Desert Scrub M087 - Chihuahuan Semi-Desert Grassland M512 - North American Warm Desert Ruderal Scrub & Grassland
Colorado Plateau / Great Basin Grassland	M118 - Intermountain Basins Cliff, Scree & Badland Sparse Vegetation M169 - Great Basin & Intermountain Tall Sagebrush Shrubland & Steppe M170 - Great Basin & Intermountain Dwarf Sagebrush Shrubland & Steppe M171 - Great Basin & Intermountain Dry Shrubland & Grassland M499 - Western North American Cool Semi-Desert Ruderal Scrub & Grassland M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
Gambel Oak Shrubland	M048 - Central Rocky Mountain Montane-Foothill Grassland & Shrubland M049 - Southern Rocky Mountain Montane Shrubland M010 - Madrean Lowland Evergreen Woodland M011 - Madrean Montane Forest & Woodland M020 - Rocky Mountain Subalpine-High Montane Conifer Forest M022 - Southern Rocky Mountain Lower Montane Forest M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland M091 - Warm Interior Chaparral M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
Interior Chaparral	M087 - Chihuahuan Semi-Desert Grassland M088 - Mojave-Sonoran Semi-Desert Scrub M171 - Great Basin & Intermountain Dry Shrubland & Grassland M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
Intermountain Salt Scrub	M082 - Warm & Cool Desert Alkali-Saline Wetland M093 - Great Basin Saltbush Scrub M171 - Great Basin & Intermountain Dry Shrubland & Grassland M118 - Intermountain Basins Cliff, Scree & Badland Sparse Vegetation M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
Juniper Grass – Low Sun Mild	M043 - California Chaparral M087 - Chihuahuan Semi-Desert Grassland M086 - Chihuahuan Desert Scrub M171 - Great Basin & Intermountain Dry Shrubland & Grassland M010 - Madrean Lowland Evergreen Woodland M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland



	M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
	M118 - Intermountain Basins Cliff, Scree & Badland Sparse Vegetation
Madrean Encinal Woodland	M088 - Mojave-Sonoran Semi-Desert Scrub
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
	M091 - Warm Interior Chaparral
	M010 - Madrean Lowland Evergreen Woodland
	M011 - Madrean Montane Forest & Woodland
Madrean Pinyon-Oak Woodland	M010 - Madrean Lowland Evergreen Woodland
	M091 - Warm Interior Chaparral
	M088 - Mojave-Sonoran Semi-Desert Scrub
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
	M011 - Madrean Montane Forest & Woodland
Mixed Conifer – Frequent Fire	M011 - Madrean Montane Forest & Woodland
	M022 - Southern Rocky Mountain Lower Montane Forest
	M048 - Central Rocky Mountain Montane-Foothill Grassland & Shrubland
	M168 - Rocky Mountain & Vancouverian Subalpine-High Montane Mesic Meadow
	M501 - Central Rocky Mountain Dry Lower Montane-Foothill Forest
Mixed Conifer w/ Aspen	
	M020 - Rocky Mountain Subalpine-High Montane Conifer Forest
	M168 - Rocky Mountain & Vancouverian Subalpine-High Montane Mesic Meadow
	M022 - Southern Rocky Mountain Lower Montane Forest
Montane / Subalpine Grassland	M048 - Central Rocky Mountain Montane-Foothill Grassland & Shrubland
	M049 - Southern Rocky Mountain Montane Shrubland
	M020 - Rocky Mountain Subalpine-High Montane Conifer Forest
	M168 - Rocky Mountain & Vancouverian Subalpine-High Montane Mesic Meadow
	M022 - Southern Rocky Mountain Lower Montane Forest
Mountain Mahogany Mixed Shrubland	M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland
	M022 - Southern Rocky Mountain Lower Montane Forest
	M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
	M043 - California Chaparral
	M049 - Southern Rocky Mountain Montane Shrubland
	M091 - Warm Interior Chaparral
	M094 - Cool Interior Chaparral
	M168 - Rocky Mountain & Vancouverian Subalpine-High Montane Mesic Meadow
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
PJ Evergreen Shrub	M010 - Madrean Lowland Evergreen Woodland
	M011 - Madrean Montane Forest & Woodland

	M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
	M043 - California Chaparral
	M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
PJ Grass – Cold	M010 - Madrean Lowland Evergreen Woodland
Temp PJ Grass –	M011 - Madrean Montane Forest & Woodland
High Sun Mild	M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
	M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
PJ Sagebrush	M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland
	M169 - Great Basin & Intermountain Tall Sagebrush Shrubland & Steppe
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
	M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
PJ Woodland –	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
Cold PJ Woodland	M049 - Southern Rocky Mountain Montane Shrubland
– Mild	M010 - Madrean Lowland Evergreen Woodland
	M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland
	M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
Ponderosa Pine –	M011 - Madrean Montane Forest & Woodland
Evergreen Oak	M022 - Southern Rocky Mountain Lower Montane Forest
Ponderosa Pine	M049 - Southern Rocky Mountain Montane Shrubland
Forest	M048 - Central Rocky Mountain Montane-Foothill Grassland & Shrubland
	M168 - Rocky Mountain & Vancouverian Subalpine-High Montane Mesic Meadow
	M118 - Intermountain Basins Cliff, Scree & Badland Sparse Vegetation
	M022 - Southern Rocky Mountain Lower Montane Forest
	M501 - Central Rocky Mountain Dry Lower Montane-Foothill Forest
Sagebrush	M022 - Southern Rocky Mountain Lower Montane Forest
Shrubland	M026 - Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland
	M027 - Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland
	M169 - Great Basin & Intermountain Tall Sagebrush Shrubland & Steppe
	M170 - Great Basin & Intermountain Dwarf Sagebrush Shrubland & Steppe
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
Sandsage	M052 - Great Plains Sand Grassland & Shrubland

	M053 - Great Plains Shortgrass Prairie
	M086 - Chihuahuan Desert Scrub
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
	M052 - Great Plains Sand Grassland & Shrubland
	M087 - Chihuahuan Semi-Desert Grassland
Semi-Desert	M086 - Chihuahuan Desert Scrub
Grassland – High	M088 - Mojave-Sonoran Semi-Desert Scrub
Sun	M091 - Warm Interior Chaparral
	M171 - Great Basin & Intermountain Dry Shrubland & Grassland
	M087 - Chihuahuan Semi-Desert Grassland
	M088 - Mojave-Sonoran Semi-Desert Scrub
	M512 - North American Warm Desert Ruderal Scrub & Grassland
	M010 - Madrean Lowland Evergreen Woodland
Shortgrass	M052 - Great Plains Sand Grassland & Shrubland
Prairie	M053 - Great Plains Shortgrass Prairie
	M498 - Great Plains Ruderal Grassland & Shrubland
	M512 - North American Warm Desert Ruderal Scrub & Grassland

## 4.2. Aquatic Systems

### 4.2.1. Statewide:

River flow and reservoir and lake levels are strongly dependent upon rainfall during monsoon season (July-September) and winter snowpack (November- March) (Enquist et al. 2008). Approximately 40% of annual precipitation falls during the monsoonal storms in July and August. Another 20% falls during spring and fall months. Winter precipitation accounts for the remaining 40% and is driven by frontal activity over the Pacific Ocean, which varies from year to year depending on the El Nino Southern Oscillation (ENSO). The Pacific Decadal Oscillation (PDO) and Atlantic multi-year oscillation (AMO) also influence winter precipitation though they fluctuate on larger multi-decadal scales and act to enhance or dampen ENSO driven trends. Currently about 75% of winter precipitation falls as snow in mountainous areas (Enquist and Gori, 2008).

Climate change alters many factors that influence hydrological cycles including precipitation timing, amount, storm intensity and rain-snow ratios (IPCC 2007). These have a number of cascading effects on water volume, quality, and erosion within watersheds in New Mexico. Despite variations among climate models, all support predictions for less snow, earlier snow melt and increase variability in the timing and intensity of storms. Within New Mexico, most flowing streams depend upon winter snow accumulations for spring and summer flows. Years with poor snowpack levels often result in very low flows by the time monsoon storms begin. These impacts have consequences not only for flowing stream bodies but for seeps and springs. Ephemeral water bodies will be warmer and experience increased evaporation rates reducing their value and availability as habitat.

Future hydrological regimes will reflect snowpack and precipitation changes. Increased rainfall versus snow during winter months will reduce snowpack and increase likelihood of atypical winter floods. Reduced snowpack and earlier, more rapid snowmelt will result in earlier peak flows and less water during low summer flow periods. Warm season runoff is projected to decline substantially over the Southwestern U.S. and Southern Rockies (Spears et al. 2013); Table 4.1). Hoerling et al. 2009 (Hoerling et al. 2009) estimate a 2-9% reduction in runoff flows for each degree Celsius increase in temperature in the Upper Colorado region. Hurd and Coonrod (2007) predict future annual reductions in the Rio Grande flow of 2.3%–13.7% in 2030 and 8.3–28.7% in 2080 relative to the baseline period of 1970–2000. D’Antonio (2006) estimates an even more drastic reduction in Rio Grande spring runoff by the end of the century.

**Table 4.13.** Percent change (from 1990) in hydrological metrics for three reaches of the Rio Grande. Adapted from Spears et al. 2013.

Hydroclimate Variable	2020	2070
<b>Northern (Rio Chama)</b>		
Mean Annual temperature (F)	1.9	5.3
Mean Annual Precipitation (%)	-1.1	-2.5
Mean Annual Runoff	-0.2	-11
Mean December-March Runoff	4.8	8.6
Mean April-July Runoff	-1.3	-21.7
<b>Middle (Rio Grande near Otowi)</b>		
Mean Annual temperature (F)	1.9	5.2
Mean Annual Precipitation (%)	-1.5	-2.4
Mean Annual Runoff	-4.4	-19.9
Mean December-March Runoff	-3.1	-12
Mean April-July Runoff	-2.5	-21.8
<b>Southern (Rio Grande at Elephant Butte Dam)</b>		
Mean Annual temperature (F)	1.9	5.1
Mean Annual Precipitation (%)	-0.9	-1.9
Mean Annual Runoff	-4.1	-16.4
Mean December-March Runoff	-3.6	-10.9
Mean April-July Runoff	-1.6	-20

Vulnerability of aquatic systems to climate variation depends upon their size and reliance on snowpack (Meyer et al. 1999). Changes in snowpack, including amount and snowmelt rates, influence not only water availability but the timing of peak flow volumes. Rain-snow transition zones are projected to shift dramatically within New Mexico and nearly all the mountain ranges are considered at-risk snowpacks likely to decline substantially over the next century (Klos et al. 2014). By 2035-2065, mountain ranges within the Southern Rockies Ecoregion and the Arizona/New Mexico Mountains will have a much briefer period of snowfall and a greater amount of winter precipitation falling as rain (Klos et al. 2014). Only the northernmost mountains within the Colorado plateau will continue to receive snow-dominated precipitation though even here most months appear to have a rain-snow mix. At the watershed level, changes to the extent of areas dominated by snowfall, rain-snow mixes and rainfall are

dramatic. The upper Pecos loses all its snow-dominated extent and increases its rain dominated extent by 23%. The Rio Grande–Elephant Butte Watershed (HUC1302) is estimated to increase its rain dominated extent by 51%, the Rio Grande headwaters lose 29% of their snow-dominated extent, and the Gila watershed becomes entirely rain dominated (Klos et al. 2014). Though New Mexico appears to lose much of its snowfall-conducive temperatures, the steep elevational gradients may delay or reduce this loss at local scale (Klos et al. 2014).

Increased temperatures also have a number of impacts for aquatic habitats and species. Warmer waters may facilitate the establishment of aquatic invasive species such as the quagga mussel and simultaneously reduce the effectiveness of biological and chemical control agents (Hellmann et al. 2008). Warmer water also encourages algae growth potentially leading to eutrophic conditions in lakes (Lettenmaier 2008). Decreased precipitation and increased temperatures are expected to decrease riparian vegetation cover and increase erosion leading to increased sedimentation in many stream and river systems (Theobald et al. 2010). Extreme weather events and post fire erosion and debris flows can also reduce water quality and biochemical processes. Feedbacks between runoff volume, water quality, evapotranspiration and erosion commonly lead to degradation in aquatic habitats (Lettenmaier et al. 2008).

### **Assessments of climate change impacts for watersheds within New Mexico**

Two climate change vulnerability assessments have considered either watershed or basins within New Mexico. Enquist and Gori (2008) ranked vulnerability of watersheds in New Mexico according to their magnitude of exposure to climate change and biological diversity. In general, lower elevation watersheds have experienced greater drying than high elevation watersheds though about 93% of watersheds overall showed some decrease in moisture availability over the 1970-2006 study periods. There tended to be more drying at drier (lower elevation) watersheds though some watersheds, primarily in the southeast quadrant of the state, appeared to experience less drying for summer and fall seasons. The Jemez, Cloverdale, and Playas Lake watersheds were identified as the most vulnerable due to the magnitude of observed moisture stress. The Pecos Headwaters, Upper Rio Grande, Upper Gila, and San Francisco watersheds have less moisture stress.

Theobald et al. (2010) reviewed and analyzed threats to riparian ecosystems in the Western United States using a risk assessment approach that considered human modification, climate change and hydrological system. The lower Colorado River and Great Basin regions contain the most modified watersheds. This effect is more pronounced in steeper and more arid part of the west, in particular, within the Southern Rocky mountains. Overall the highest combined threat score was found for western Washington, the Great Basin, southern Idaho and northern Utah, and southern Arizona and New Mexico. Theobald et al. (2010) found decreased flows in the Rio Grande region due to increased discharge but predicted increased flow for Colorado and Great Basin Regions. Southern Arizona and New Mexico received very high riparian threats scores. Flow fragmentation was among the worst for watersheds in Arizona and New Mexico though these same watersheds were not among those with the highest degree of modified riparian area.

## 4.2.2. Summaries by Aquatic Habitat Type

### *Statewide perennial tanks (springs, stream seepage, or precipitation filled)*

Water bodies found in perennial tanks fed by springs, stream seepage or precipitation are likely to experience increased water temperatures and increased rates of evaporation, reducing their size and, in some situations, leading to intermittent presence in the landscape. Increased evaporation may also increase salinization of water in certain areas. Increased drought will reduce the abundance and hydroperiod of catchments. Tanks found in areas prone to wildfire and floods could experience increased sedimentation and infill and lower water quality. Where water tables become too low, many tanks will cease to exist. Many of these systems are isolated increasing the risk of endemic species extinction as a result of their loss or degradation.

### *Statewide ephemeral catchments (playas, pools, tinajas, kettle, tanks)*

Playas are depressional wetlands resulting from accumulations of precipitation and surface runoff (Gage and Cooper 2013). Their exclusive reliance on direct precipitation and runoff means these systems are highly vulnerable to potential changes in precipitation. In eastern New Mexico, Playas may be especially vulnerable to climate impacts under future drier conditions (Matthews 2008). Where they persist, increased variation in precipitation events will reduce the hydroperiod of many catchments. Increased drought will also reduce the abundance and hydroperiod of catchments. Increases in salinity due to increase evaporation and reduced precipitation may exacerbate the rate of species invasions (Meyer et al. 1999). Many of these systems are isolated increasing the risk of endemic species extinction as a result of their loss or degradation.

### *Statewide perennial lakes, cirques, ponds*

Lakes and reservoir response to climate change is influenced by their thermal stratification and depth (Spears et al. 2013). Warmer waters may facilitate the establishment of aquatic invasive species such as the quagga mussel and simultaneously reduce the effectiveness of biological and chemical control agents (Hellmann et al. 2008). Increases in salinity due to increase evaporation and reduced precipitation may exacerbate the rate of species invasions and lead to widespread changes in food webs (Meyer et al. 1999). Warmer water can encourage algae growth leading to eutrophic conditions in lakes (Lettenmaier 2008). Higher water temperatures have multiple effects for temperature-dependent species (Eaton and Scheller 1996, Johnson et al. 2005). Systems that become isolated are at an increased risk of endemic species extinction as a result of climate change.

### *Statewide ephemeral nonplaya habitats (ephemeral marsh/cienega/spring/seep, ephemeral cold/cool water streams, ephemeral warm water streams)*

Marsh/cienega/spring/seeps are at high risk to the synergistic effects of human related habitat disturbance and natural processes including climate change (Friggens et al. 2012). Currently, these habitats are limited due to decreasing water tables as a result of land conversion and channelization. Increased temperatures will increase evaporation and evapotranspiration

leading to greater rates of water loss and decreased hydroperiod. Increased variability in annual precipitation, delayed onset of monsoons and potentially dryer spring conditions will also reduce the availability of these habitats. Water quality is also likely to decrease where post fire flooding and erosion from high intensity events cause increased water turbidity, sedimentation and infill.

#### *Perennial reservoirs, warm water (by ecoregion)*

Overall, reservoirs and other open water habitats may be more buffered from climate impacts because they are relatively stable over time as compared to flowing water and ephemeral systems (Matthews 2008). Water warming will be less severe in larger water bodies as compared to catchments and ponds. Still, lakes and reservoir impacts are influenced by their thermal stratification and depth (Spears et al. 2013) and there is a risk that in these systems, water column turnover periods, important for nutrient cycles within lake systems, would be disrupted by climate related changes to water temperature and volume (Matthews 2008).

Reservoirs can be sensitive to changes in inflow, with substantial drops in reservoir levels from small reductions in runoff (Christensen et al. 2004, Christensen and Lettenmaier 2007). Demand for water is expected to increase under warming conditions (Perry et al. 2012) leading to increased water shortages. Efforts to maintain reservoir storage and delivery under drier hotter climates will decrease flow variability and flow magnitude exacerbating direct effects of climate change on river and riparian systems.

Warmer waters may facilitate the establishment of aquatic invasive species such as the quagga mussel and simultaneously reduce the effectiveness of biological and chemical control agents (Hellmann et al. 2008). Warmer water can also encourage algae growth leading to eutrophic conditions in lakes (Lettenmaier 2008). Higher water temperatures have multiple effects for temperature-dependent species (Eaton and Scheller 1996, Johnson et al. 2005). Increased water temperature could promote productivity and expand habitat for warm water species (Perry et al. 2012). Warmer waters could also lead to the expansion of invasive species in both aquatic and riparian habitats (Rood et al. 2008, Theobald et al. 2010). For cool-adapted aquatic species, warmer temperatures can increase thermal stress, create migration barriers, and reduce reproductive success (Perry et al. 2012, Raymond et al. 2014). Cool water refugia might decrease or disappear within reservoirs. Collectively these impacts can change trophic dynamics within lakes and reservoirs leading to complex impacts.

#### *Perennial reservoirs, cold water*

Cool water reservoirs may be more susceptible to changes in inflow resulting from climate changes. Reservoirs within the Colorado River Basin are likely to be very sensitive to changes in inflow, with substantial drops in reservoir levels from small reductions in runoff (Christensen et al. 2004, Christensen and Lettenmaier 2007). Reservoirs on upper tributaries to the Colorado River are considered more vulnerable to the changes in flow timing and snowmelt than those along lower systems (Spears et al. 2013). Increased water temperature could increase productivity and habitat for warm water species (Perry et al. 2012) but at the expense of cool-

adapted species (Raymond et al. 2014). For cool-adapted species, increased temperatures can increase thermal stress, create migration barriers, and reduce reproductive success (Perry et al. 2012). Cool water refugia might decrease substantially within these reservoirs. Warmer waters may facilitate the establishment of aquatic invasive species such as the quagga mussel and simultaneously reduce the effectiveness of biological and chemical control agents (Hellmann et al. 2008). Warmer water can also encourage algae growth leading to eutrophic conditions in lakes (Lettenmaier 2008).

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#### *Perennial cold/cool water Streams (~1<sup>st</sup> and 2<sup>nd</sup> order, by ecoregion)*

Climate change will decrease the availability of cool water stream habitat suitable for cold-adapted species. Many reaches within lower elevation and southern sites may no longer be suitable for coldwater species.

Precipitation phase can influence spring snowpack, rain-on-snow flood risks and the timing of snowmelt-driven streamflows in mountain catchments (Klos et al. 2014). The loss of snowfall-conductive temperatures within New Mexico has implications for the timing of spring snow melt and soil moisture as well as the persistence of cool water streams. Loss of snowpack is predicted for most of New Mexico's mountain ranges, which will result in reduced spring flood events and summer flows. Lower streamflow amounts are likely to warm more quickly in response to increasing air temperatures (Spears et al. 2013). Warming water will result in a loss of favorable growth periods and a reduction in the extent of available habitat for species dependent upon cold water habitats (Fang et al. 2004a, b). For cool-adapted species, increased temperatures can increase thermal stress, create migration barriers, fragment habitat, and reduce reproductive success (Meyer et al. 1999, Perry et al. 2012). Expansion of invasive species in both aquatic and riparian habitats are also likely under warmer conditions (Theobald et al. 2009; Rood et al. 2008). Changes in sediment load and channel morphology as a result of altered precipitation regimes, erosion, and extreme events will disrupt riparian community composition and lower water quality (Ward et al. 2002). Water quality declines are also likely in areas prone to wildfires and post-fire flood events. Wildfire can also remove vegetation and increase erosion with subsequent negative consequences for water quality.

#### *Perennial warm water (~3<sup>rd</sup> +4<sup>th</sup> +5<sup>th</sup> order, by ecoregion)*

River corridors support a disproportionate amount of biodiversity in the Southwest (Pase and Layser 1977). Climate change is likely to reduce the availability and quality of perennial warm water systems, particularly in the southern part of New Mexico.

In Southwest riparian systems, drought and intense heat are likely to lead to increases in drought mortality and shrinkage and fragmentation of riparian habitat, issues compounded by over-extraction of water and invasive species (Palmer et al. 2009). Milly et al. 2007(Milly et al.



2007) project a substantial decrease in annual runoff in the Southwest under warmer conditions. Several preexisting conditions increase the vulnerability of New Mexico river systems to climate change. First, perennial river systems are snowpack driven, which leaves them less buffered against the drying trends associated with global warming (Palmer et al. 2009). Second, many of these systems are dammed or within clear cut or urbanized watersheds, which reduces their resilience to increasing climate variability. Changes to water temperature and flow regimes are buffered in free-flowing rivers with protected watersheds where vegetation and wetlands slow and absorb the impacts of increased rates of snowmelt and temperatures. Lack of free-flowing rivers and functional wetlands translates to increase erosion, habitat destruction and ultimately, loss of water. In addition, dammed rivers tend to experience more drawdown of water, leaving little water available to sustain environmental flows (Palmer et al. 2009). Higher water temperatures have multiple effects for temperature-dependent species (Johnson et al. 2005; Eaton and Scheller 1996) including the likely expansion of invasive species in both aquatic and riparian habitats (Theobald et al. 2009; Rood et al. 2008). Increased salinity as a result of increases in evaporation rates may also become a problem. In the western Great Plains, increased salinity is a leading factor predicted to lead to loss of endemic fish species, many of which are already near their thermal tolerance limit (Meyer et al. 1999). Water quality declines are likely in areas prone to wildfires and post-fire flood events. Wildfire can also remove vegetation and increase erosion with similar negative consequences for water quality.

#### *Perennial marsh/cienega/spring/seep*

Currently wetland habitats in New Mexico are threatened by drought and land disturbance. Though no models are known that predict future conditions for meadows, it is likely that these systems will continue to struggle to persist and are likely to decline under climate change.

The relative abundance of marsh ecosystems depends upon regional elevational and geographic gradients (Burkett and Kusler 2000). Temperature and precipitation strongly influence marsh formation, persistence and function and, as a result, marshes are very sensitive to climate fluctuations (Perry et al. 2012, Gage and Cooper 2013)). In the semi-arid environment of New Mexico the overall abundance of wetlands tends to be greater at higher elevations though local physiographic characteristics can also play a role. Beavers can also play a role in marsh formation where beaver ponds can lead to the establishment of marsh communities (Gage and Cooper 2013).

Changes in precipitation and evaporation rates due to increased temperatures can change the seasonality, depth and duration of marsh (or wetland or playa) hydroperiod with consequences for marsh function and vegetation dynamics. In particular, hydrological variability is recognized as a predictor of vegetation patterns in marshes. For fens, mean water depth is a significant predictor of vegetation patterns (Gage and Cooper 2013). Marshes may be more resilient than playas to some precipitation changes as they also rely on surface or groundwater inflows (Gage and Cooper 2013). Alpine wetlands are likely to be highly susceptible to negative impacts of climate changes because they are likely to lose relic species that cannot disperse to new sites. Lowering of watertables as a result of hotter dryer conditions will decrease decomposition in

wetland soils and reduce carbon storage potential. Elevated CO<sub>2</sub> may affect wetland plants by increasing growth rates and biomass. Wetlands are often highly fragmented limiting the capacity of wetland dependent species to migrate to new locations as temperature and water levels change and increasing the risk of endemic species extinction. Wetland hydrology may change considerably with changes in the timing of snowmelt, reduced snow pack and winter flows resulting from increased rain versus snowfall. Increases in the frequency of summer drought periods will lead to the transition of many wetlands from permanent to more ephemeral ponds (Poff et al. 2011).

## **5. Vulnerability of SGCN to climate change**

Climate change is already altering ecosystems and presents a substantial threat to the conservation of biodiversity (Hughes 2000; Peñuelas and Filella 2001; Root et al. 2003). Subsequent change in extinction risk will vary by species, taxonomic group, region, and time elapsed leading to questions about where to focus conservation efforts (Peterson et al. 2002, Thomas et al. 2004, MacLean and Wilson 2011). In the Southwest, climate change analyses have primarily focused on rare or special status species, but fish and invertebrate species have rarely been assessed (Friggens et al. 2013a). Grasslands also tend to be underrepresented in recent assessments (Friggens et al. 2013a).

Response of species to climate change is particularly important in the context of SGCN, because ongoing conservation efforts could be overwhelmed by additional impacts or new stressors may be overlooked leading to missed opportunities for intervention. Below we outline how climate change affects species then demonstrate the utility of assessing vulnerability with a brief examination of 21 SGCN. Finally, we outline how information on vulnerability to climate change can be used to improve management actions illustrated by two case studies.

### **5.1. What is climate change vulnerability?**

Although there is some controversy over the precise use and meaning of *vulnerability* in a climate change context, most think of it as the susceptibility to negative impacts (Fussel 2007, Hinkel 2011). Low vulnerability can be taken as low susceptibility or resilience to negative impacts. Some species may experience increasing or expanding populations. Climate change vulnerability is sometimes split into effects due to exposure, sensitivity, and adaptive capacity (Glick et al. 2011). Exposure is climate and climate-related phenomena (e.g., fire, floods) while sensitivity (i.e., how exposure is experienced) and adaptive capacity (i.e., ability to reduce negative impact) are properties of the species that can help predict how they will respond. Difficulty in predicting response of species arises, in part, because projections of exposure tend to be large scale (i.e., several km<sup>2</sup>) while individual plants and animals interact with their environment at much smaller scales.

When species conservation is the goal, vulnerability is measured by change in extinction risk and is generally deduced from projected geographic shifts in suitable range (i.e., climate envelope approaches) or by identifying species traits that predict climate change response (i.e.,

changes in survival or reproduction) or through a combination of these approaches (Bagne et al. 2014, Notaro et al. 2013, Preston et al. 2008).

#### 5.1.1. What makes a species vulnerable to climate change?

Many species are expected to incur negative impacts from climate change (Bagne et al. 2014; Gardali et al. 2012; Foden et al. 2008). Species already at risk of extinction may be particularly vulnerable to these impacts. A review of special status species in the Middle Rio Grande revealed additional vulnerability or population declines for at-risk terrestrial vertebrate species indicating that additional conservation effort will be needed (NABCI 2010, Bagne et al. 2014). Similarly, a review of freshwater fish in California found climate change vulnerability to be positively correlated with current extinction risk (Moyle et al. 2013).

Mountainous regions and associated taxa are particularly vulnerable to change as precipitation and temperature vary rapidly across a relatively small area (Lawler et al. 2009). Importantly, mountains can create isolated islands of habitat particularly where surrounding flatlands have very different environments such as in the Southwest. Wetland, riparian, and aquatic habitats are also particularly vulnerable in the Southwest because their distribution is very localized and they have already been heavily modified and degraded (Patten 1998).

Vulnerability to climate change will vary by populations, species, and taxonomic group because of differences in exposure, sensitivity, and adaptive capacity (Glick et al. 2011). Warmer temperatures, more variable precipitation, and lower net primary productivity are particularly pronounced in the Southwest making New Mexico species exposed to greater change than many other parts of the U.S. (Seager et al. 2007, Reeves et al. 2014). For all taxonomic groups, specialist and sedentary species are thought to be more vulnerable than generalist and highly mobile species (Foden et al. 2008, Gilman et al. 2010). Relevant to many fish and riparian species in New Mexico, we expect warmer water temperatures, earlier peak flows, more variable rainfall, and lower summer base flows (Furniss et al. 2013, more). Although most birds are highly mobile and can readily shift with habitats, migratory species are particularly vulnerable to timing mismatches and disparate habitat change on wintering grounds, breeding sites, and stopover sites (Visser et al. 2004, Visser et al. 2008). Reptiles may be a particularly vulnerable group to negative impacts and tend to not be well represented on species priority lists (Sinervo et al. 2010, Bagne et al. 2014). Mammals, although generally mobile, may be vulnerable to habitat change if geographically isolated (e.g., high elevation, riparian) or migratory. Arthropods are also not well represented in climate change assessments, but like most vertebrates, arthropods are expected to move northward and to higher elevations with specialists more vulnerable to declines than generalists (Brantley and Ford 2012). Because arthropod development is directly related to temperature, populations and range can shift dramatically (Robinet and Roques 2010). Similarly, higher metabolic rates in warmer waters may rapidly expend stored reserves in non-diapausing aquatic insects (Sweeney et al. 1992). Mollusc and crustacean species are vulnerable because they tend to be endemic and freshwater habitats where many New Mexico species reside are one of the most threatened worldwide (Dudgeon et al. 2006).

Species can respond to climate change through a number of different pathways, which can be categorized into habitat, physiology, phenology, and interactions (Bagne et al. 2011). Habitat loss and fragmentation are already major driving forces in declining natural populations. The location and condition of suitable habitats will be further altered by changes in temperature and precipitation (McCarty 2001, Sekercioglu et al. 2008, Ibanez et al. 2008). Shifting habitats can be inaccessible to species with low dispersal ability, may move outside protected boundaries, and are likely not perfectly analogous (Jiguet et al. 2007, ref for protected). Physiological requirements and limitations related to temperature and moisture determine critical components of energetics, survival, and reproduction (Helmuth et al. 2005, Bernardo and Spotila 2006, Sinervo et al. 2010). A species may be intolerant of the range of new ambient conditions, more restricted in activity, or subject to more extreme climate-related events such as fires or storms (Walsberg 2000, Bernardo and Spotila 2006, Sinervo et al. 2010). Higher metabolic costs for ectotherms during warmer winters when food resources are limited could decrease survival (Kaspari et al. 2000, Brantley and Ford 2012). For many species, phenology or timing of activities (e.g., reproduction, migration) is triggered by temperature or moisture cues and, thus, prone to alteration with changing climate. When this timing is altered so that it no longer matches the timing of critical resources or favorable conditions, then survival or reproduction decline (Dunn and Winkler 1999, Both et al. 2006). Finally, response of one species may trigger population change in another such as through predator-prey relationships, disease, pollination, parasitism, or mutualism. These changes in biotic interactions can further alter vulnerability if tied to survival or reproduction (Freed et al. 2005, Memmott et al. 2007, Gilman et al. 2010).

#### 5.1.2. What makes a species resilient to climate change?

Any of the factors noted above; habitats, physiology, phenology, or interactions can shift to more favorable conditions for a species in a given location. As some habitat types contract, others expand. Elevated CO<sub>2</sub> levels and warmer temperatures can enhance plant growth and lengthen growing season providing more forage or longer breeding periods (Morgan et al. 2001). More variable and extreme weather can have positive effects on availability of ephemeral waters, maintenance of some spawning habitats, and prevention of encroachment of woody plants. Species experiencing more favorable conditions may be those that shift their range to New Mexico, although these are unlikely to be current SGCN.

Several species traits are associated with resilience to climate change. Generalist species can switch to different prey or host species and thus are not as sensitive to changes in species with more restricted diets (Chessman 2013, Moyle et al. 2013). Similarly, wide-ranging species typically tolerate a larger array of environmental conditions. Any species can benefit when conditions that limit population growth (i.e., cold winters) are improved. Warm-water fishes, for example, may be more tolerant of warming conditions than cold-water species and may invade newly suitable locations (Moyle et al. 2013). Aestivation, torpor, inactive life stages, and low metabolic rates can improve a species adaptive capacity to cope with fluctuating resources (Humphries et al. 2002, Bronson 2009). For example, although warmer waters increase metabolic demands, aquatic insects that diapause can avoid ensuing higher energy

requirements (Sweeney et al. 1992). Species traits such as irruptive migration and explosive breeding can allow a species to avoid timing mismatches and rapidly respond to resource variability (Visser et al. 2004). Longer, more flexible, and more productive reproductive periods can improve response to increasingly variable and unpredictable conditions, although species with shorter reproductive periods may be favored during drought (Jiguet et al. 2007, Chessman 2013, Moyle et al. 2013). Climate change vulnerability for one species may benefit another if it is an important predator or competitor.

### 5.1.3. Combining climate-related effects with other stressors

As demonstrated by the process for selecting SGCN, there are many stressors on species populations and negative effects of climate change are just one subset to be considered when prioritizing species or actions. Climate change is an important consideration as additional stress on species already prone to extinction could overwhelm conservation efforts. Unfortunately, the very nature of populations of SGCN makes them prone to exacerbating effects of climate change. Many SGCN have very restricted ranges and sometimes are comprised of only a single population. Thus, these species are particularly vulnerable to shifting climate and habitats as small isolated ranges comprise little habitat variability and little opportunity for dispersal. Additionally, local population variability in response to a more varying climate can quickly initiate an Allee effect when small declines can lead to extinction (Opdam and Washer 2004).

By adding to or altering impacts already affecting species, climate change modifies extinction risk and creates a complex challenge for conservation practitioners (McCarty 2001; MacNally 2009). Climate-driven threats, such as exceeding physiological thresholds, have often been overlooked in setting special status species and need to be considered especially in anticipation of future population trends (Bagne et al. 2014). Climate change effects can also exacerbate other stressors such as fire, invasive species, or landscape connectivity that may already be implicated in species decline. For New Mexico, exacerbating effects are particularly applicable to loss of aquatic and riparian habitat. Higher temperatures and more variable rainfall will reduce already limited surface water (Serrat-Capdevila et al. 2007, Theobald et al. 2010). Hotter, drier, and more variable conditions encourage fires that remove vegetation, favor invasives such as tamarisk, and deposit sediments in aquatic habitats (Swetnam and Betancourt 1990, Westerling et al. 2006). Damage from cattle grazing is exacerbated during higher temperatures when cattle preferentially graze near water (DelCurto et al. 2005). Greater water withdrawal from agriculture and residents is expected with warmer temperatures further straining aquatic and riparian habitats (Foti et al. 2012). Changing geographic locations of suitable conditions (i.e. climate envelope) exacerbates issues related to fragmentation and land conversion, which restrict movement in highly mobile species and limit the ability of species to respond.

### 5.2. Species vulnerability assessment example

Vulnerability assessments can be used to examine the relative vulnerability among a group of species and to identify the mechanisms of individual species vulnerability. We conducted a brief vulnerability assessment of 21 SGCN to demonstrate the pathways to climate change response

and how this information could be used in conservation efforts. This should not be interpreted as a full vulnerability assessment for any of these species. Species selection was based on available climate change information and to encompass a wide variety of attributes for SGCN in Tier 1 and 2.

We modified a vulnerability index, SAVS (System for Assessing Vulnerability of Species to climate change), to examine the vulnerability of individual species (Bagne et al. 2011). SAVS is a scoring system that uses a series of questions about species traits or factors to predict response based on habitat, physiology, phenology, and interspecies interactions so that areas of vulnerability or resilience can be identified. Future population trend will depend on the balance of these factors (e.g., more traits predicting lower survival and reproduction than higher), the relative strength among individual effects (e.g., exceeding physiological tolerance overrides vegetation shift), and local conditions that alter to exposure (e.g., slope or recent fire can alter flood risk). We chose 12 of the original 22 SAVS climate change factors to simplify interpretation, but also to cover a diverse array of climate change effects. We interpreted the factors responsible for climate change response broadly so as to include all taxonomic groups rather than just terrestrial vertebrates, which SAVS was originally designed for (Table 5.1). This reinterpretation of SAVS retained the ecological target of individual questions, but expanded scope and terminology to accommodate more taxonomic groups and aquatic habitats. For example, Question H1, “Is associated breeding habitat vegetation expected to change?” was easily expanded to include large-scale aquatic habitat features such as temperature and stream flow to capture whether suitable range is expected to expand, contract or shift (Table 5.3). We considered all life stages for invertebrates (Radchuk et al. 2013). We included a total score to allow the reader to quickly interpret the balance of vulnerable and resilient traits and not as an indication of priority. Life history information for scoring was mostly obtained from BISON-M and Federal Register documents from USFWS Endangered Species Program. Other sources included vulnerability assessments from Fort Huachuca, Arizona (Bagne and Finch 2013); Middle Rio Grande, New Mexico (Friggens et al. 2013b); Sky Islands, Arizona (Coe et al. 2012); and birds and reptiles in the Southwest (van Riper et al. 2014).

### 5.2.1. Which SGCN are most vulnerable to climate change effects?

The brief assessment of 21 species reveals the diversity of vulnerabilities and, in some cases, resilience across the factors (Table 5.2). These are in addition to any threats not related to climate change. For this set of 12 factors, almost half the species were much more vulnerable than resilient including Jemez Mountains salamander, Chiricahua leopard frog, Capulin Mountain arctic butterfly, Southwestern willow flycatcher, spikedace, Gila trout, New Mexico ridgenose rattlesnake, and Mexican gartersnake. Other species like pinyon jay, Socorro isopod, and Texas hornshell had traits indicative of both vulnerability and resilience making it more difficult to project how these various vulnerability factors will collectively affect populations (Table 5.2). Species such as pinyon jay, black-tailed prairie dog, and white-sided jack rabbit had few vulnerabilities for this set of factors, but still face other climate and non-climate threats and lacked information in some key areas. Note that a score of 0 could indicate a neutral response

or an unknown response. Based on these knowledge gaps, important research topics for predicting climate change response were noted in Table 5.3.

Several common areas of vulnerability emerged from our analysis. Changes related to reduced snowpack, water withdrawals, higher evaporation rates, and reduced annual precipitation were cited in vulnerability of 14 species. The change in fire risk, particularly for high severity fires, was important to several terrestrial and aquatic species. Increase in fire interacts with invasion by many exotic plants resulting in further ecosystem changes. Fires, invasive plants, and water issues are not new, but additional vulnerability underscores the critical management needs in this area. Resilience was not as common as vulnerability and tended to be more unique among species. High mobility can allow a species to shift with changing conditions, but human and natural barriers can impede movement and potential for dispersal needs to be considered by managers. For many species, important diseases or parasites were not known or lacked predictions related to climate. Physiological tolerance and the risk of exceeding thresholds were assumed for a number of species, but there is little detailed information available for any species on this critical topic.

### 5.3.How to manage climate change vulnerability

Climate change is an important consideration for the success of species conservation programs whether through exacerbation of current threats or production of new impacts. Exacerbation of current threats may require intensified conservation efforts while threats unique to climate change will require innovative strategies (Bagne et al. 2014). Assessment of climate change vulnerability adds a future outlook on threats that complements traditional conservation approaches, which have focused on threats that are ongoing or were responsible for declines historically. The key to finding effective management actions is to identify factors responsible for vulnerability or resilience for a species.

Management actions designed to cope with climate change effects are commonly termed adaptation. Adaptation strategies can be categorized as resistance (e.g., remove woody encroachment from a grassland, build a fire break around a small isolated population), resilience (apply prescribed fire to a fire-adapted landscape, maintain habitat connections), transition or response (i.e., create corridors, re-vegetate with resistant genotypes, move individuals to new locations), and realignment (i.e., manage for new conditions, focus on provision of ecosystem services)(Millar et al. 2007, Peterson et al. 2011). Resistance strategies are only a short-term solution if climate change induces new or exacerbating negative effects, but often describe the intensive and localized management of rare and isolated species (Heller and Zavaleta 2009). Realignment to future conditions has had little discussion related to species conservation.

The predicted response of species to various climate-related factors can indicate the targets of adaptation actions. Habitats need to be managed under the expectation that they will change and shift over time. Specific components of habitat (i.e., snags, breeding ponds) can also be targeted for management if expected to decline. A species vulnerable because of low dispersal ability may benefit from translocation or creation of corridors. If a species is sensitive to extreme events such as prolonged drought, high severity wildfires, and intense flooding then

action plans can be developed to anticipate and take necessary emergency action (Bagne and Finch 2013). Heat sensitivity may be addressed by providing buffered habitat elements such as shade or deep pools. Management may need to target response of interacting species. Similarly, some traits generating resilience may be enhanced through management such as creation of reserves where habitats are expanding or protection of hibernacula or aestivation sites that provide good temperature buffering. Conservation of genetic diversity may enhance resilience to physiologically limiting conditions (Heller and Zavaleta 2009).

Any landscape or reserve planning needs to take into account how habitats shift over time (Hodgson et al. 2009). Landscape connectivity can facilitate movement when habitats are expected to shift and provide a greater diversity of microclimates. Establishment of corridors, matrix management, and expansion of reserve networks can all improve connectivity (McLachlan 2007, Hodgson et al. 2009). In the absence of connectivity or dispersal potential, assisted migration can move individuals to new environments and is a valid strategy for preventing species extinction, but is also controversial (McLachlan et al. 2007). Research will be needed to make informed decisions before initiating assisted migration to address knowledge gaps such as interspecies relationships, dispersal distance, and detailed habitat requirements (McLachlan et al. 2007). Translocation, or movement of individuals to historically occupied locations is less controversial and may help species cope with short-term habitat change, dispersal barriers, or increasing population fluctuations. Programs to move populations, however, tend to be costly and are often unsuccessful (Fischer and Lindenmayer 2000).

Climate change can make some types of management more difficult. Individual threats may be harder to manipulate under changing climate conditions such water flow, which declines through multiple pathways including warmer temperatures, more variable rainfall, lower snowpack, and greater demand. Fire management will also become more difficult as warmer weather and more frequent drought limit the window for applying prescribed fire and make suppression problematic. Multiple factors indicating vulnerability may make it more difficult to reverse declining trends, but can also offer alternative approaches. For example, it may be more practical to create artificial waters than to regulate water withdrawals. In addition to a single species focus, a list of species and their vulnerabilities can be used to identify management issues common among species making efforts more efficient. An assessment from Fort Huachuca, Arizona found management of fire and fuels, invasive species, natural and artificial waters, and landscape planning were relevant to multiple species (Bagne and Finch 2013).

When faced with uncertainty or few management options to target vulnerability, there are several possible approaches. “No-regrets” adaptation options are actions to increase populations or reduce stressors regardless of climate change (Peterson et al. 2011). Mitigation of other stressors (e.g., invasive species, habitat loss) is often recommended in lieu of addressing climate change effects, but many impacts are interrelated and the increasing vulnerability for many species indicates that conservation efforts will need to be intensified over time. “Win-win” options confer benefits under both future climate projections and current conditions (Peterson et al. 2011). Fire management, invasive species control, and watershed improvement often fall within this category. Habitat quality can be improved through these



types of actions, thus enhancing resilience of species to climate change and disturbance. Uncertainty can also arise because of lack of information and assessing vulnerability can help pinpoint research needs. For many SGCN, a broader set of suitable conditions than those encompassed by currently occupied locations may need to be considered, because historical range and the full breadth of suitability are often poorly known (Heller and Zavaleta 2009). Population monitoring can be a useful tool when effects or management options are uncertain or funds are limited. Furthermore, monitoring is needed to determine the success of any implemented actions.

Opportunities for improved species management can also arise with climate change and should be anticipated. For example, low water levels can create barriers and stress non-native fish and amphibian populations which facilitates their removal to the benefit of native amphibians and fish, which can be more tolerant of drying (Doubledee et al. 2003, Bagne and Finch 2013). Removal or control of non-native plants or animals may be more successful when they are stressed by climate extremes. For example, invasive aquatic species may decline where intolerant of warmer or more saline waters (Higgins & Wilde 2005, Rahel & Olden 2008). Preventative and early intervention programs to control invasive species can be applied where range expansion is predicted and are cheaper and more effective (Davies & Johnson 2011). Climatic variation will also include wet or productive years, which can be timed to correspond with habitat restoration or translocation programs. Exploitation of the vulnerabilities of undesirable species can be summarized as a “kick them when they’re down” strategy and fits well with “no-regrets” and “win-win” strategies of climate change adaptation (Peterson et al. 2011, Bagne and Finch 2013).

The above discussion can be summarized in the following general recommendations for coping with climate change:

1. Use species-specific response (vulnerability and resilience) to enhance populations of SGCN, reduce undesirable species, and develop research projects
2. Anticipate range shifts to adapt local programs and reserve networks to shifting SGCN or to implement early intervention for invasive species
3. Use resistance strategies such as reduction of non-climate stressors for short-term time frames
4. Expect long-term conservation of vulnerable SGCN to require intensified effort, innovative approaches, and flexibility to large-scale ecosystem change
5. Use climate variability to time management actions during periods favorable to goals
6. Implement monitoring to detect population trends and evaluate success of adaptation actions

Taking the vulnerabilities and resilience of the 21 SGCN (Table 5.2 and 5.3), we undertook case studies on two species to demonstrate the relevance of vulnerability to conservation action. For each species, we connected vulnerable or resilient factors to potential targeted actions. These case studies are only a demonstration of how to use climate change response in management applications and should not be taken as actual conservation plans.

**Table 5.1.** Factors and scores indicating vulnerability or resilience to climate change for short version of SAVS (System to Assess the Vulnerability of Species to climate change, Bagne et al. 2011) as modified for this report. All life stages considered. A score of zero is also used when the species response is unknown.

SAVS factor identifier	Factor/trait indicating vulnerability or resilience to climate change
Habitat H1	Habitat area at a large-scale (i.e., range) is expected to increase (-1), decrease (1), or stay the same (0) in New Mexico
Habitat H3	Specific habitat components within suitable range are expected to increase (-1), decrease (1) or stay the same (0)
Habitat H6	This species is able to disperse to new areas (-1), not able to disperse (1), or can only disperse rarely (0)
Physiology PS1	Limits to temperature or moisture tolerance are expected to be exceeded (1), more favorable (-1), or not changed (0)
Physiology PS3	Climate-related disturbance events (i.e., fires, floods) that cause widespread mortality or reproductive failure are expected to increase (1), decrease (-1), or stay the same (0)
Physiology PS5	This species possesses (-1) or does not possess (1) specialized strategies or traits to cope with highly variable resources
Phenology PH1	This species uses (1) or does not use (0) temperature or moisture cues to initiate activities related to survival or reproduction
Phenology PH3	Timing of critical activities are closely tied and can track resource peaks (-1), separated in space or time (1), or this species does not time activities to critical resource peaks (0)
Phenology PH4	This species has a long reproductive period or can reproduce more than once per year (-1) or only reproduces once or less per year with a short reproductive period (1)
Interactions I1	Important food resources are expected to increase (-1), decrease (1), or stay the same/are broad (0)
Interactions I3	This species depends on another species for which populations are expected to increase (-1), decrease (1), or stay the same (0)
Interactions I4	Prevalence of disease or parasites that cause widespread mortality or reproductive failure is expected to increase (1), decrease (-1), or unchanged (0)

**Table 5.2.** Scores by factor from short SAVS version (Table 6.1) for 21 SGCN showing areas of vulnerability and resilience. Zero can indicate neutrality or missing information (see Table 6.3). Total score shows the balance of resilience and vulnerability for quick comparison of this set of 12 factors, but is not a full assessment of climate change vulnerability.

Common name	Scientific name	H1	H3	H6	PS1	PS3	PS5	PH1	PH3	PH4	I1	I3	I4	Total score
Jemez Mountains salamander	<i>Plethodon neomexicanus</i>	1	0	1	1	1	-1	1	-1	1	0	0	-1	3
Chiricahua leopard frog	<i>Lithobates chiricahuensis</i>	1	1	1	0	1	1	1	-1	1	0	0	0	6

Arizona treefrog	<i>Hyla wrightorum</i>	1	1	1	1	0	1	0	0	1	0	0	1	7
Capulin Mountain arctic butterfly	<i>Oeneis alberta capulinensis</i>	1	0	1	1	1	-1	1	0	1	0	0	0	5
Southwestern willow flycatcher	<i>Empidonax traillii eximius</i>	1	1	-1	1	1	1	1	1	-1	1	0	1	7
Mexican spotted owl	<i>Strix occidentalis lucida</i>	1	0	-1	1	0	1	0	0	1	1	0	0	4
Pinyon jay	<i>Gymnorhinus cyanocephalus</i>	1	0	-1	0	1	-1	0	-1	0	1	0	1	1
Socorro isopod	<i>Thermosphaeroma thermophilum</i>	1	0	1	0	1	1	0	0	-1	-1	0	0	2
Noel's amphipod	<i>Gammarus desperatus</i>	1	0	1	0	1	1	0	0	1	-1	0	0	4
Spikedace	<i>Meda fulgida</i>	1	-1	1	0	1	1	1	-1	1	0	0	1	5
Gila trout	<i>Oncorhynchus gilae</i>	1	0	1	0	1	1	1	0	1	0	0	1	7
Colorado pikeminnow	<i>Ptychocheilus lucius</i>	1	0	-1	0	0	1	1	-1	1	0	0	1	3
White-sided jack Rabbit	<i>Lepus callotis</i>	1	0	-1	0	0	1	0	0	-1	1	0	0	1
Meadow jumping Mouse	<i>Zapus hudsonius</i>	1	0	1	0	1	1	1	0	1	0	1	0	7
Black-tailed prairie dog	<i>Cynomys ludovicianus</i>	1	0	-1	0	0	-1	0	0	1	1	0	0	1
Texas hornshell	<i>Popenaias popeii</i>	1	0	1	1	1	-1	0	0	1	0	1	0	5
Koster's springsnail	<i>Juturnia kosteri</i>	1	0	1	0	1	1	0	0	-1	-1	0	0	2
Slevin's bunchgrass lizard	<i>Sceloporus slevini</i>	-1	0	1	0	0	1	1	0	1	0	0	0	3
NM ridgenose rattlesnake	<i>Crotalus willardi obscurus</i>	1	0	1	1	1	1	1	0	1	0	0	0	7
Mexican gartersnake	<i>Thamnophis eques</i>	1	1	0	1	1	1	0	0	1	1	0	0	7

**Table 5.3.** Details related to vulnerability and resilience to climate change across 12 factors for 21 SGCN. Information needs notes where scoring was uncertain or where interacting threats were expected.

Common name	Critical vulnerabilities	Resilience	Information needs
Jemez Mountains salamander	Decreased soil moisture, physiological limitations, low dispersal	Torpor, subterranean habits buffer from surface conditions	Fire effects (severity, season, frequency) on population dynamics, climate effects on chytridiomycosis infection
Chiricahua leopard frog	Declining water flow, faster pond evaporation, dispersal barriers	Timing flexible to water levels	Climate effects on chytridiomycosis infection
Arizona treefrog	High severity fire risk, evaporation and decreased flow of breeding waters, low dispersal, trematode infection		Climate effects on monsoon timing, response to extreme events
Capulin Mountain arctic butterfly	Heat sensitive, two asynchronous populations, warmer temperatures harmful to overwintering larvae, low dispersal	Inactive life phases, more rapid development	Host plant species
Southwestern willow flycatcher	Exacerbating threats to habitats (fire, exotic plants, floods, water withdrawals), brood parasitism (cowbirds resilient), nest failure during drought	Highly mobile, flexibility in habitat use, re-nesting behavior	Thermal tolerances and locations of favorable microclimates, effects of nest timing on success
Mexican spotted owl	Heat sensitive, high severity fire risk	Variability in habitat use and risk (e.g., canyons more resilient), mobile (especially juveniles)	Favorable microclimate distribution
Pinyon jay	High severity fire risk, tree mortality, drought effects on habitat and reproduction	Highly mobile, nomadic, close timing of activities with resources	Use of alternative food sources and habitats, reproductive timing flexibility, climate effects on West Nile virus
Socorro isopod	Water table drop, drought, reduced precipitation, low dispersal	Long reproductive period, algae growth enhanced	Fire effects, CO <sub>2</sub> fertilization effects on populations, effects of water temperature or chemistry change

### 5.3.1. Case study 1: Arizona Treefrog

Given the twelve factors in climate change response, Arizona treefrog is vulnerable from multiple pathways including reduction in terrestrial and breeding habitat, low dispersal, desiccation, high severity wildfire, limited reproductive period, and disease (Table 5.2 and 5.3). None of the factors examined indicated resilience, although we discuss some potential positive effects of reduced stream flow below. Management for Arizona treefrog under changing climate conditions will be complicated by vulnerability of both aquatic and terrestrial habitats as well as currently small and isolated populations. There is considerable uncertainty in predicting response, because there are very few published studies on this species. Although high severity fire is clearly deleterious to habitats and may kill adults, how other variables related to fire affect treefrogs and their habitat are unknown. In addition, future timing of monsoons is important to available breeding habitats, but is not well projected making the associated population response unclear.

Despite uncertainty, vulnerability traits of the Arizona treefrog point towards potential management actions. Fire management could be used to increase resilience of upland woodland habitats to drought and fire mortality. Low severity prescribed fire can reduce the risk of stand-replacing fires detrimental to treefrog populations while preserving a mosaic of unburned patches. Fire risk could also be reduced by creation of fire breaks or application of mechanical treatments. Resilience of adult treefrogs to desiccation might be increased by management actions that promote leaf litter and accumulated debris if it can be assumed that these can buffer adults from extreme conditions. Upland habitats may also shift upwards in elevation, thus corridors to facilitate movement should be maintained or, in the absence of potential dispersal, plausibility of assisted migration should be investigated.

Regulation of water withdrawals as well as wetland and stream restoration could help maintain breeding habitats. Occupied waters that are more resilient to water flow declines (e.g., deeper ponds, greater canopy cover) should be identified and prioritized for conservation. The potential to increase water availability in habitats more vulnerable to drying should be evaluated and rescue measures considered. Artificial waters can in some cases be managed to maintain water levels during critical periods such as during breeding or prolonged droughts. Shift of permanent streams to intermittent cycles may increase potential breeding habitats, but only if they are within dispersal distance of current populations, adjacent upland habitats are suitable, and populations of predators are small. Attention to changing flow will be critical, as management action (e.g., control of predators, habitat linkage) may be needed to make these newly intermittent waters suitable. Reduced and more variable stream flow is also an opportunity for more effective control of non-native fish and amphibians as many are less resilient to decreased water flow and newly isolated populations may more easily be controlled (Doubledee et al. 2003). The relatively higher vulnerability for non-natives could also benefit populations of Gila trout and Chiricahua leopard frog.

Chytridiomycosis has not been identified in wild populations, but snails that host the trematode, *Ribeiroia ondatrae*, have been implicated in observed limb deformities (Johnson and Sutherland 2003). Snail hosts are often present in stock ponds and tanks, which may be

more heavily used under drying conditions (Bagne and Finch 2013). Management targeting reduction of snail hosts and trematode infection may be appropriate. Monitoring should include measures of disease, hosts, or parasites so early intervention can be implemented.

### 5.3.2. Case study 2: Black-tailed Prairie Dog

Given the examined factors involved in climate change response, black-tailed prairie dogs had few vulnerable traits along with some indicators of resilience. In particular, the potential for expansion of grasslands in New Mexico could provide additional habitats. Conversely, prairie dogs are vulnerable to plague, which can kill entire colonies, but complex life cycle and varied climate response make prediction uncertain. Human impacts (e.g., shooting, poisoning) on this species can be high, thus are a source of non-climate stress that can be reduced to increase resilience.

Although expansion of open vegetation types is expected, many areas are likely to see an increase on shrub/scrub habitat, which is less suitable. Conversion of grasslands to scrublands is expected for several macrogroups leading to a reduction in habitat. Where grasslands are likely to replace woodlands (such as areas in eastern New Mexico that are allowed to experience fire), transition to suitable prairie dog habitat will take time, as various elements of the ecosystem are not expected to change at the same rate. Different species assemblages in expansion areas are expected as different species respond in different ways with unknown effects on habitat suitability. In general, prairie dogs can use a variety of grassland habitat types and have been reported from open woodlands. However, black-tailed prairie dogs are only associated with shortgrass prairie and desert grassland habitat types, which appear likely to decrease (see Section 4). Prairie dogs modify their habitat, thus have some capacity to engineer their own resilience (e.g. slow woody species encroachment) to climate change by maintaining preferred habitat features. Management that favors open habitats such as prescribed fire, mechanical removal of woody vegetation, and livestock grazing can be used to enhance prairie dog habitats. Predicted areas of grassland expansion can be evaluated to gauge potential for future suitability and to prioritize locations for conservation.

The interaction of plague with changing climate is likely to be important, but is not well known especially for Southwest climates. Plague is an introduced pathogen that threatens long-term viability of prairie dog populations and more study is needed to predict response in New Mexico. The distribution of plague is projected to expand north and east, thus conditions in New Mexico may become less favorable (Nakazawa et al. 2007, p. 537). Prediction of plague response is complicated by the interaction of the disease causing bacteria (*Yersinia pestis*), the vector (e.g., flea), susceptible prairie dogs and other flea hosts (e.g., *Onychomys*) (Gage and Kosoy 2005, Lorange 2005). High summer temperatures tend to reduce plague outbreaks while periods of higher rainfall, which are expected irregularly, tend to increase flea vectors, transmission, and plague outbreaks, thus prediction of future plague effects is complex and uncertain (Parmenter et al. 1999, Stapp et al. 2004). Flea infestation increases with anthropogenic disturbance, thus management focused on minimizing disturbance or prioritizing conservation in remote areas may increase resilience (Friggens 2010). Roads and streams create barriers to disease transmission and are associated with lower plague occurrence

(Collinge et al. 2005). Lower population densities, such as following periods of low rainfall, wildfire, or drought may also decrease transmission and aid in disease management. Colony isolation during plague outbreaks may be manipulated by creation of temporary barriers particularly during late spring when most intercolony dispersal occurs (Garrett and Franklin 1988). Vaccination and application of insecticides at burrows are potential management tools to increase resilience to plague (Seery et al. 2003, Rocke et al. 2010).

High genetic variation in prairie dogs has been noted with several subspecies proposed, thus variation in response to climate is expected among populations and may incur some resilience to the species as a whole. Maintenance of genetic diversity will require landscape level planning. As opposed to isolation during plague outbreaks, dispersal to new habitats may be encouraged by removal of barriers during late spring. Low reproductive rates mean proactive management is important to avoiding low populations, which recover slowly. Food subsidies, translocation, and predator control are all options for increasing populations (Truett et al. 2001). Management plans should establish triggers and protocols for initiating these types of interventions.

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**Appendix 1.** List of NVCS Macrogroups, Brown Biomes and Ecological Response units for each Ecoregion.

<b>SWAP designations</b>	<b>Macrogroups (&gt;1% area)</b>	<b>Brown Biomes</b> (Rehfeldt et al. 2006: >1%, bold indicates dominate type)	<b>Ecological Response Units (ERU)</b> (Triepke et al. 2014: >1% and in order of abundance)
Southern Rockies	Great Plains Mixedgrass & Fescue Prairie, Great Plains Shortgrass Prairie, Great Basin & Intermountain Dry Shrubland & Grassland, Great Basin & Intermountain Tall Sagebrush Shrubland & Steppe, Southern Rocky Mountain Montane Shrubland, Rocky Mountain & Vancouverian Subalpine-High Montane Mesic Meadow, Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland, Rocky Mountain Subalpine-High Montane Conifer Forest, Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland, Southern Rocky Mountain Lower Montane Forest	Great Basin Montane Scrub, <b>Great Basin Conifer Woodland</b> , Western Alpine Tundra, Great Basin Desertscrub, <b>Rocky Mountain Subalpine Conifer Forest</b> , <b>Rocky Mountain Montane Conifer Forest</b> , Plains Grassland	Ponderosa Pine Forest, Mixed Conifer – Frequent Fire, Shortgrass Prairie, Spruce-Fir Forest, PJ Woodland – Cold, Sagebrush Shrubland, Colorado Plateau / Great Basin Grassland, Juniper Grass – Cold
High Plains	Great Basin & Intermountain Dry Shrubland & Grassland, Great Plains Sand Grassland & Shrubland, Chihuahuan Semi-Desert Grassland, Chihuahuan Desert Scrub, Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland, Great Plains Shortgrass Prairie	<b>Great Basin Conifer Woodland</b> , Chihuahuan Desertscrub, <b>Semidesert Grassland</b> , Rocky Mountain Montane Conifer Forest, <b>Plains Grassland</b>	Shortgrass prairie, Semi-Desert Grassland – High Sun, Juniper Grass – High Sun Mild, Mixed-Grass Prairie, Juniper Grass – Cold, Shinnery Oak, Sandsage, Ponderosa Pine Forest, PJ Woodland – Cold, PJ Grass – Cold Temp

**Appendix 1.** List of NVCS Macrogroups, Brown Biomes and Ecological Response units for each Ecoregion.

Colorado Plateaus	<p>Chihuahuan Semi-Desert Grassland, North American Warm Semi-Desert Cliff, Scree &amp; Rock Vegetation, Warm &amp; Cool Desert Alkali-Saline Wetland, Intermountain Basins Cliff, Scree &amp; Badland Sparse Vegetation, Great Basin &amp; Intermountain Tall Sagebrush Shrubland &amp; Steppe, Great Basin Saltbush Scrub, Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland, Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland Great Basin &amp; Intermountain Dry Shrubland &amp; Grassland Great Plains Shortgrass Prairie, Great Plains Sand Grassland &amp; Shrubland, Madrean Lowland Evergreen Woodland, North American Warm Semi-Desert Cliff, Scree &amp; Rock Vegetation, Chihuahuan Semi-Desert Grassland, Chihuahuan Desert Scrub</p>	<p>Great Basin Montane Scrub, <b>Great Basin Conifer Woodland</b>, Great Basin Desertscrub, Chihuahuan Desertscrub, Semidesert Grassland, Rocky Mountain Subalpine Conifer Forest, <b>Great Basin Shrub-Grassland</b>, Rocky Mountain Montane Conifer Forest, <b>Plains Grassland</b></p> <p><b>Great Basin Conifer Woodland</b>, Madrean Montane Conifer Forest, Madrean-Transvolcanic Woodland, <b>Chihuahuan Desertscrub</b>, <b>Semidesert Grassland</b>, Rocky Mountain Montane Conifer Forest, Plains Grassland</p>	<p>Colorado Plateau / Great Basin Grassland, Sagebrush Shrubland, PJ Grass – Cold Temp, Juniper Grass – Cold, PJ Woodland – Cold, Ponderosa Pine Forest, Shortgrass Prairie, Intermountain Salt Scrub, PJ Sagebrush, Semi-Desert Grassland – High Sun Mild, Juniper Grass – High Sun Mild, Mixed Conifer – Frequent Fire, Sandsage Semi-Desert Grassland – High Sun, Chihuahuan Desert Scrub, Chihuahuan Salt Desert Scrub, Shortgrass Prairie, Shinnery Oak, Juniper Grass – High Sun Mild, Sandsage, PJ Evergreen Shrub, Mountain Mahogany Mixed Shrubland, Madrean Pinyon-Oak Woodland, Interior Chaparral, PJ Grass – High Sun Mild</p>
Chihuahuan Deserts			



**Appendix 1.** List of NVCS Macrogroups, Brown Biomes and Ecological Response units for each Ecoregion.

<p>Arizona/New Mexico Mountains</p>	<p>Chihuahuan Desert Scrub, Rocky Mountain &amp; Vancouverian Subalpine-High Montane Mesic Meadow, Great Plains Shortgrass Prairie, Warm Interior Chaparral, Rocky Mountain Subalpine-High Montane Conifer Forest, Great Basin &amp; Intermountain Dry Shrubland &amp; Grassland, Madrean Montane Forest &amp; Woodland, Southern Rocky Mountain Two-needle Pinyon - One-seed Juniper Woodland, Chihuahuan Semi-Desert Grassland, Madrean Lowland Evergreen Woodland, Intermountain Singleleaf Pinyon - Utah Juniper - Western Juniper Woodland, Southern Rocky Mountain Lower Montane Forest</p>	<p><b>Great Basin Conifer Woodland</b>,Madrean Montane Conifer Forest, Western Alpine Tundra, Madrean-Transvolcanic Woodland, Chihuahuan Desertscrub, Semidesert Grassland, Rocky Mountain Subalpine Conifer Forest, Great Basin Shrub-Grassland, <b>Rocky Mountain Montane Conifer Forest</b>, Plains Grassland</p>	<p>Semi-Desert Grassland – High Sun, Colorado Plateau / Great Basin Grassland, Ponderosa Pine Forest, PJ Grass – Cold Temp, Shortgrass Prairie, Juniper Grass – High Sun Mild, PJ Evergreen Shrub, Mixed Conifer – Frequent Fire,Juniper Grass – Cold, PJ Woodland – Cold, Chihuahuan Desert Scrub, PJ Grass – High Sun Mild, Madrean Pinyon-Oak Woodland, PJ Woodland – Mild, Ponderosa Pine – Evergreen Oak, Mountain Mahogany Mixed Shrubland, Montane / Subalpine Grassland, Mixed Conifer w/ Aspen, Madrean Encinal Woodland, Interior Chaparral, Intermountain Salt Scrub, Juniper Grass – Low Sun Mild ,Gambel Oak Shrubland, Sandsage, Sagebrush Shrubland, PJ Sagebrush</p>
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**Appendix 1.** List of NVCS Macrogroups, Brown Biomes and Ecological Response units for each Ecoregion.

Madrean Archipelago	Warm Interior Chaparral, North American Warm Semi-Desert Cliff, Scree & Rock Vegetation, Madrean Lowland Evergreen Woodland, Chihuahuan Desert Scrub, Chihuahuan Semi-Desert Grassland	Great Basin Conifer Woodland, <b>Madrean-Transvolcanic Woodland, Chihuahuan Desertscrub, Semidesert Grassland</b> , Rocky Mountain Montane Conifer Forest, Plains Grassland	Semi-Desert Grassland – High Sun Mild, Chihuahuan Desert Scrub, Madrean Encinal Woodland, Madrean Pinyon-Oak Woodland, Juniper Grass – High Sun Mild, Interior Chaparral, PJ Evergreen Shrub
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**Appendix 2.** Species modeled in Notaro et al. 2012 and corresponding NVCS Macrogroups. Red indicates projected range contraction (across SW) and green indicate expansion (across SW).

Macrogroup	Species	Area Change (km)	Area Change (%)	Robustness (%)
M011 <i>Pinus leiophylla</i> <i>Pseudotsuga menziesii</i> / <i>Quercus hypoleucoides</i> Madrean Montane Forest & Woodland Macrogroup				
	<i>Pinus leiophylla</i>	-5,704	-50	64
	<i>Pseudotsuga menziesii</i> (MA)*	-69,401	-50	100
	<i>Quercus emoryi</i> (M)	42,781	32	64
M020 <i>Abies lasiocarpa</i> <i>Picea engelmannii</i> <i>Pinus albicaulis</i> Rocky Mountain Forest Macrogroup				
	<i>Abies lasiocarpa</i> (MA)*	-87,147	-67	100
	<i>Picea engelmannii</i> (MA)*	-120,422	-96	100
	<i>Pinus albicaulis</i> *	-57,042	-100	100
M026 <i>Pinus monophylla</i> <i>Juniperus osteosperma</i> <i>Juniperus occidentalis</i> Intermountain Woodland Macrogroup				
	<i>Juniperus osteosperma</i> (MA)*	-6,971	-1	57
	<i>Pinus monophylla</i> *	-37,394	-37	71
M027 <i>Pinus edulis</i> <i>Juniperus monosperma</i> Southern Rocky Mountain Woodland Macrogroup				
	<i>Juniperus monosperma</i> (MA)	-71,302	-18	86
	<i>Pinus edulis</i> (MA)*	35,809	8	71
M034 Rocky Mountain & Great Basin Montane Riparian Forest Macrogroup				
	<i>Picea engelmannii</i> (MA)*	-120,422	-96	100
	<i>Populus angustifolia</i>	-379,329	-88	100
	<i>Populus balsamifera</i>	0	0	X
M036 Southwest Riparian Forest Macrogroup				
	<i>Celtis laevigata</i>	0	0	X
	<i>Platanus wrightii</i>	7,288	6	64
	<i>Populus deltoids</i>	-107,429	-81	100

**Appendix 2.** Species modeled in Notaro et al. 2012 and corresponding NVCS Macrogroups. Red indicates projected range contraction (across SW) and green indicate expansion (across SW).

<i>Populus fremontii</i>	207,252	42	79
<i>Salix laevigata</i>	316	100	50

M049 *Quercus gambelii* *Cercocarpus montanus* *Purshia* spp. Southern Rocky Mountain Montane Shrubland Macrogroup

<i>Amelanchier alnifolia</i>	-44,999	-61	86
<i>Amelanchier utahensis</i>	-199,013	-48	100
<i>Prunus virginiana</i>	-51,337	-45	100

M075 Western North American Montane to Alpine Wet Shrubland & Wet Meadow Macrogroup

<i>Alnus oblongifolia</i>	62,429	61	71
<i>Alnus tenuifolia</i>	-87,781	-68	100
<i>Crataegus douglasii</i>	-47,535	-79	100
<i>Crataegus erythropoda</i>	-251,618	-83	100
<i>Crataegus saligna</i>	-420,526	-96	100
<i>Crataegus tracyi</i>	459,188	200	100
<i>Salix amygdaloides</i>	-196,161	-68	100
<i>Salix bebbiana</i>	-633	-2	50
<i>Salix bonplandiana</i>	0	0	X
<i>Salix exigua</i>	5,387	X	50
<i>Salix geyeriana</i>	-98,872	-98	100
<i>Salix laevigata</i>	316	100	50
<i>Salix lasiandra</i>	-63,380	-68	100
<i>Salix lasiolepis</i>	0	0	X
<i>Salix nigra</i>	35,809	200	86
<i>Salix petiolaris</i>	633	100	57
<i>Salix scouleriana</i>	-105,844	-87	100
<i>Salix taxifolia</i>	0	0	X

M086 *Larrea tridentata* *Flourensia cernua* *Prosopis* spp. Chihuahuan Desert Scrub Macrogroup

<i>Prosopis juliflora</i>	192,992	53	100
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**Appendix 2.** Species modeled in Notaro et al. 2012 and corresponding NVCS Macrogroups. Red indicates projected range contraction (across SW) and green indicate expansion (across SW).

M091 <i>Quercus turbinella</i> <i>Arctostaphylos pungens</i> <i>Ceanothus greggii</i> Warm Interior Chaparral Macrogroup				
	<i>Quercus turbinella</i> (M)	14,894	8	64
M092 North American WarmDesert XericRiparian Scrub Macrogroup				
	<i>Chilopsis linearis</i> (T)	135,950	46	79
M094 <i>Arctostaphylos patula</i> <i>Ceanothus velutinus</i> <i>Quercus vaccinifolia</i> Montane Chaparral Macrogroup				
	<i>Arctostaphylos pringlei</i>	346,371	65	100
M158 <i>Juniperus ashei</i> <i>Juniperus pinchotii</i> <i>Quercus mohriana</i> Scrub, Shrub & Open Vegetation Macrogroup				
	<i>Juniperus pinchotii</i>	-1,901	-100	79
	<i>Quercus mohriana</i>	-21,549	-79	79
M169 <i>Artemisia tridentata</i> <i>Artemisia tripartita</i> ssp. <i>tripartita</i> <i>Purshia tridentata</i> Great Basin & Intermountain Shrubland & Steppe Macrogroup				
	<i>Artemisia tridentata</i> (M)	-259,541	-76	100
M501 <i>Pinus ponderosa</i> var. <i>ponderosa</i> <i>Pseudotsuga menziesii</i> <i>Pinus flexilis</i> Central Rocky Mountain Dry Forest Macrogroup				
	<i>Pinus ponderosa</i> (MA)*	-233,872	-47	100
	<i>Pseudotsuga menziesii</i> (MA)*	-69,401	-50	100
M022 <i>Abies concolor</i> <i>Pseudotsuga menziesii</i> <i>Picea pungens</i> Forest Macrogroup				
	<i>Abies concolor</i> (MA)	55,774	21	64
	<i>Picea pungens</i> (M)*	-187,604	-81	100
	<i>Pseudotsuga menziesii</i> (MA)*	-69,401	-50	100
M028 <i>Populus deltoides</i> <i>Fraxinus pennsylvanica</i> / <i>Salix</i> spp. Great Plains Flooded Forest Macrogroup				
	<i>Fraxinus pennsylvanica</i>	0	0	X
	<i>Populus deltoids</i>	-107,429	-81	100
M10 <i>Juniperus deppeana</i> <i>Pinus cembroides</i> <i>Quercus arizonica</i> Madrean Lowland Evergreen Woodland Macrogroup				
	<i>Juniperus deppeana</i> (MA)	-55,140	-32	86

**Appendix 2.** Species modeled in Notaro et al. 2012 and corresponding NVCS Macrogroups. Red indicates projected range contraction (across SW) and green indicate expansion (across SW).

<i>Pinus cembroides</i>	-10,140	-25	64
<i>Quercus</i>	24,718	13	71
<i>arizonica</i> (M)			