United States
Department
of Agriculture

Forest Service

Rocky Mountain
Research Station

General Technical Report
RMRS-GTR-306

July 2013

Vulnerability of Species to Climate Change in the Southwest: Terrestrial Species of the Middle Rio Grande

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Friggens, Megan M.; Finch, Deborah M.; Bagne, Karen E.; Coe, Sharon J.; Hawksworth, David L. 2013. **Vulnerability of species to climate change in the Southwest: terrestrial species of the Middle Rio Grande.** Gen. Tech. Rep. RMRS-GTR-306. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 191 p.

ABSTRACT

We used a vulnerability scoring system to assess the vulnerability of 117 vertebrate species that occur in the Middle Rio Grande Bosque (MRGB) to expected climate change. The purpose of this project was to guide wildlife managers on options and considerations for climate change adaptation. The 117 species occur regularly in the MRGB during the breeding season, winter, or year-round. In general, future climate scenarios predict warmer temperatures with an altered precipitation regime that will likely lead to reduced water levels in the MRGB. This assessment points to several key issues relating to future habitat changes and individual species physiology that are expected to affect species survival under climate change. Decreased availability of mesic sites is expected to directly impact many amphibian and reptile populations and is expected to have indirect affects for birds and mammals primarily through changes in habitat availability. We predicted that phenological changes will negatively impact many species within all taxonomic groups through altered timing of weather events and river flow. Riparian-dependent species received some of the highest vulnerability scores. Species already at the southern limit of their distributional range were also predicted to be more likely to be vulnerable to climate change. The assessment also identified important data gaps. Management for species conservation under future climate conditions will require increased research and monitoring, greater integration of landscape-scale approaches, consideration of future land-use scenarios, and increased understanding of the consequences of species' interactions. We review the specific implications of climate change for wildlife in the MRGB in order to identify intervention points and approaches that may achieve management goals.

Keywords: climate change, southwestern riparian habitats, vulnerability assessment, Bosque, wildlife management, SAVS

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ACKNOWLEDGMENTS

This work was supported by a Research and Development grant through the USDA Forest Service, Washington Office, Climate Change Initiative and a U.S. Fish and Wildlife Service (USFWS) Bosque Initiative Grant (Final product for Agreement No. 201819H705). We thank the Nature Conservancy and the University of Arizona for input and contributions during the creation and implementation of the species vulnerability tool used in this assessment. We also thank Cyndie Abeyta (USFWS) and Alan Soloman (USFS, WO) for encouragement and inspiration. We acknowledge the Program for Climate Model Diagnosis and Intercomparison (PCMDI) and the World Climate Research Programme's (WCRP) Working Group on Coupled Modeling (WGCM) for their roles in making available the WCRP CMIP3 multi-model dataset. Support of this dataset is provided by the Office of Science, U.S. Department of Energy. Finally, we thank Erica Rowland and Molly Cross (Wildlife Conservation Society) and Katherine Dwire (Air, Water, and Aquatic Environments program, RMRS)—their comments improved this manuscript greatly.

CONTENTS

LIST OF FIGURES

LIST OF TABLES

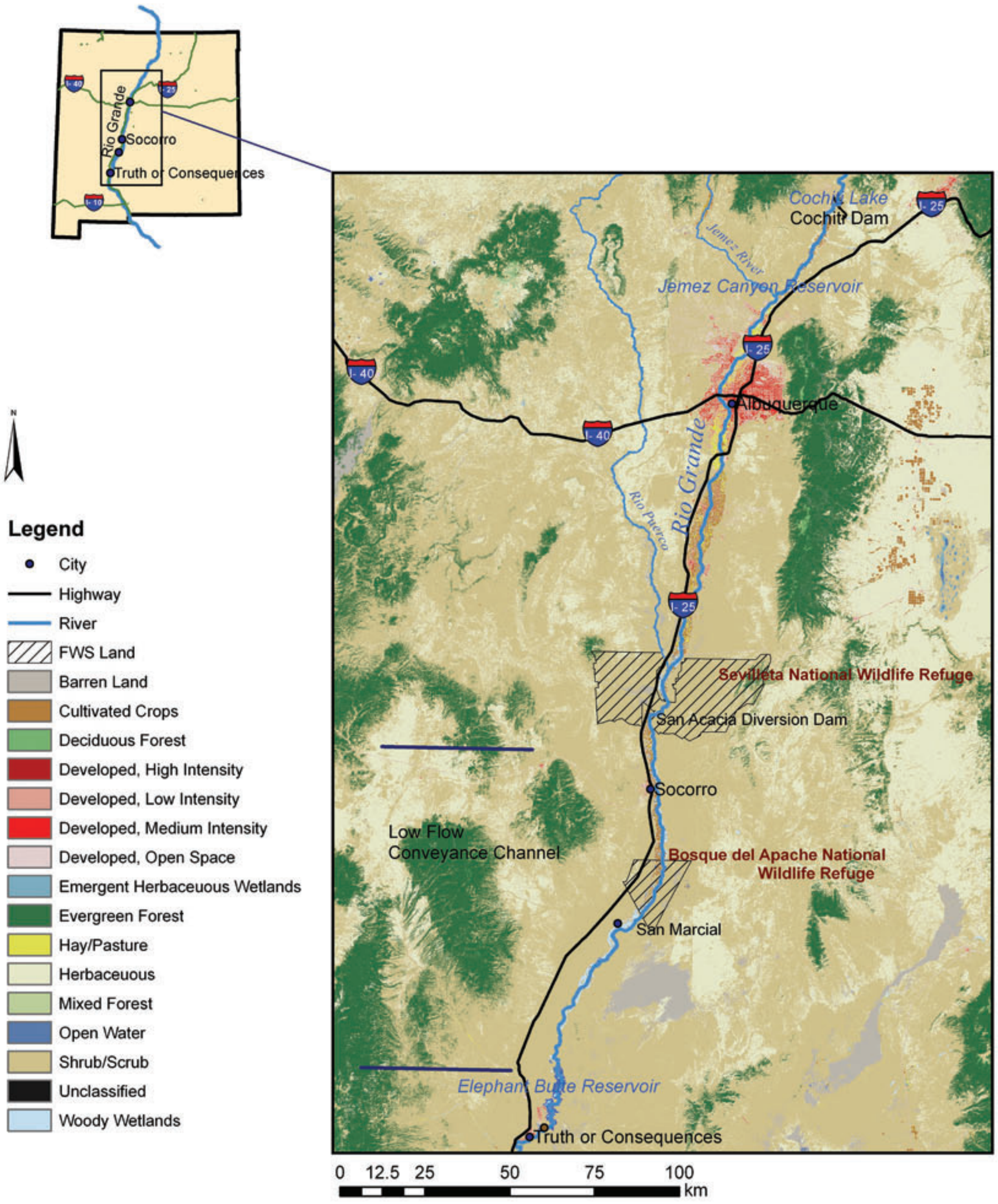


Figure 1.1. The MRG Valley in New Mexico. Our assessment specifically considered species that depend upon the Bosque forest and river stretching approximately 270 km from just south of Cochiti Dam to just north of Elephant Butte Reservoir. The majority of the study area is comprised of shrub/scrub habitat with a narrow forested riparian corridor.

Chapter 1. Introduction

1.1 Implications of Climate Change

Global climate change has the potential to affect habitats and species worldwide within a relatively short period of time and appears to already be altering ecosystems (reviewed in Hughes 2000; McCarty 2001; Peñuelas 2001; Root and others 2003). Natural selection provides the mechanism whereby species can adapt to changes in their environments (Skelly and others 2007), but the speed with which the climate is currently changing may preclude adaptation in many species (Janzen 1994; Visser 2008). In addition to current conservation challenges such as habitat loss, pollutants, and exploitation that have long been part of species management programs, climate change is a relatively new and growing challenge facing conservation and natural resource managers.

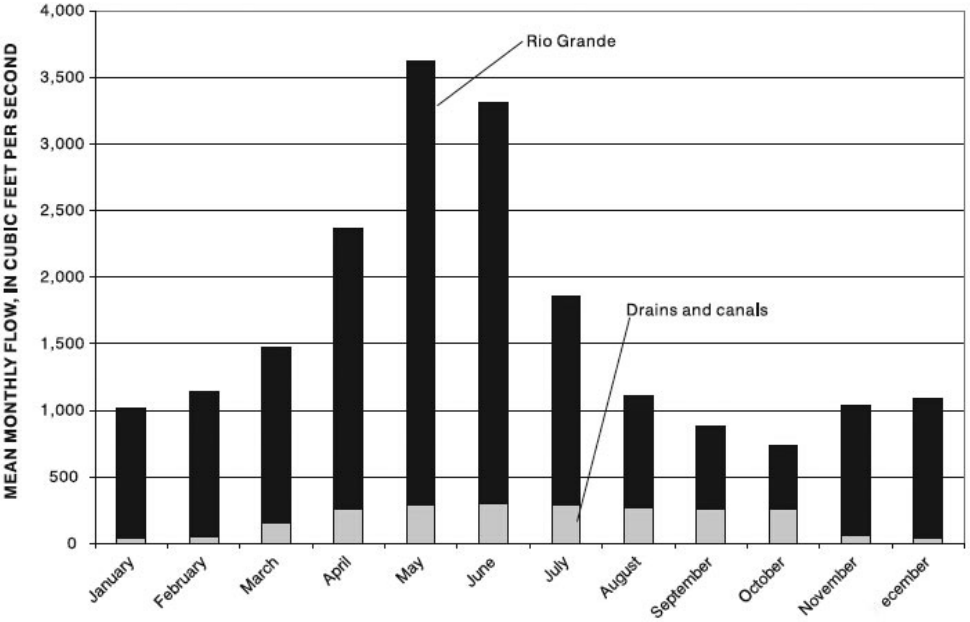
Climate change poses unique challenges to wildlife management, in part, because rates of temperature and precipitation change will vary depending on geographic location and elevation, and different rates of change are expected within and across geographic landscapes. Overlaid on this backdrop, physiological response of individual species to changing climates will vary. Such individual responses may ultimately lead to new species assemblages and interactions over time. There are many uncertainties regarding the nature and direction of impacts on species as a result of global warming. The first step toward gaining information that will allow us to evaluate future conditions and communities is to identify the relative vulnerability of species to climate change.

To prepare for climate change and assist the development of realistic management objectives, managers need place-based approaches relevant to specific locations and habitats (Palmer and others 2009). We present the results of a climate change vulnerability assessment for an important riparian system within the southwestern United States, the Middle Rio Grande (MRG) (Fig. 1.1). This chapter presents background information regarding the MRG, including historical climate trends. Place-based approaches as used in this assessment must begin with a locally relevant scenario(s) of future conditions, which are presented in Chapter 2. Results of the species climate change assessment are presented in Chapter 3. Management strategies are reviewed and discussed in Chapter 4 within the context of both anticipated impacts to the MRG and species-specific vulnerabilities to climate change.

1.2 The Middle Rio Grande Bosque (MRGB)

The Rio Grande is the fifth longest river in North America and supports over 407 vertebrate species (USACE 2011). The Rio Grande Watershed comprises nearly 1.9 million acres in New Mexico (CWCS 2006). The Middle Rio Grande (MRG) is a 180-mile segment of the river and associated riparian habitat that stretches from Cochiti Dam to the headwater of the Elephant Butte reservoir in New Mexico (Roberts 2005; Fig. 1.1). The MRG experiences a semi-arid/arid climate with an average maximum temperature of 70 ˚F (21.1 ˚C) and a minimum of 44 ˚F (6.6 ˚C) (USACE 2011). Average annual precipitation is 7.88 inches, most of which falls during the monsoon season from July through September. The flow of the Rio Grande is affected by both snowmelt and rainfall, with a typical annual flow characterized as low in the winter, peaking during spring coincident with

Figure 1.2. Hydrograph of a monthly flow volumes, averaged over the period 1942-1998, showing spring peak characteristic of the MRG. Adapted from the USGS Water Investigations report 02-4131 (2002).



snow melt, and low by June (Fig. 1.2). Smaller peak flows can also occur with monsoon activity in late summer (CWCS 2006). The lands adjacent to MRG are managed by several organizations, including the Middle Rio Grande Conservancy District (MRGCD), Bureau of Reclamation, Army Corps of Engineers, New Mexico Department of Game and Fish, U.S. Fish and Wildlife Service (USFWS), New Mexico State Parks, City of Albuquerque Parks and Recreation Division, and private landowners (Crawford and others 1993; CWCS 2006) (Fig. 1.4). The MRGCD was formed in 1943 to address diverse stakeholder needs and owns approximately 30,000 acres of the Bosque (Roberts 2005). The MRG also contains about 60,000 acres of high density of agriculture adjacent to its banks (Fullerton and Batts 2003).

Although considerably diminished from its historical extent, this region supports an extensive native cottonwood (*Populus deltoides* ssp. *wislizenii*) gallery forest or bosque (Howe and Knopf 1991; Fig. 1.3) along with important wetlands, including the Bosque del Apache National Wildlife Refuge. Native understory species include New Mexico olive (*Foresteria neomexicana*), coyote willow (*Salix exigua*), Gooding willow (*Salix gooddingii*), and seep willow (*Baccharis* spp.). Russian olive (*Eleganus angustifolia*) and saltcedar (*Tamarix* spp.) are dominant understory exotics (Fig. 1.3). Surrounding uplands vary from natural vegetation types dominated by desert scrub and grasslands to agricultural fields to a primarily urban environment as the river passes through Albuquerque, the largest city in New Mexico (Fig. 1.1).

Over 250 bird species, 54 native mammal species, 39 reptiles, and 11 amphibians are known to use the river and associated habitats (Fullerton and Batts 2003; USACE 2011). Species diversity overall is enhanced by the natural mosaic of structures and vegetation types that are historically known to occur in the MRG (Roberts 2005). The MRG is a major stopover site for many migrating bird and bat species. Bird densities tend to be highest in densely vegetated and edge habitat with consistently high densities observed for habitats with cottonwood overstory and Russian olive understory (Hink and Ohmart 1984). Emergent marsh and wetland habitats also support high densities and a wide diversity of bird species. Reptiles tend to utilize areas with sandy soils and that lack dense forest canopy.

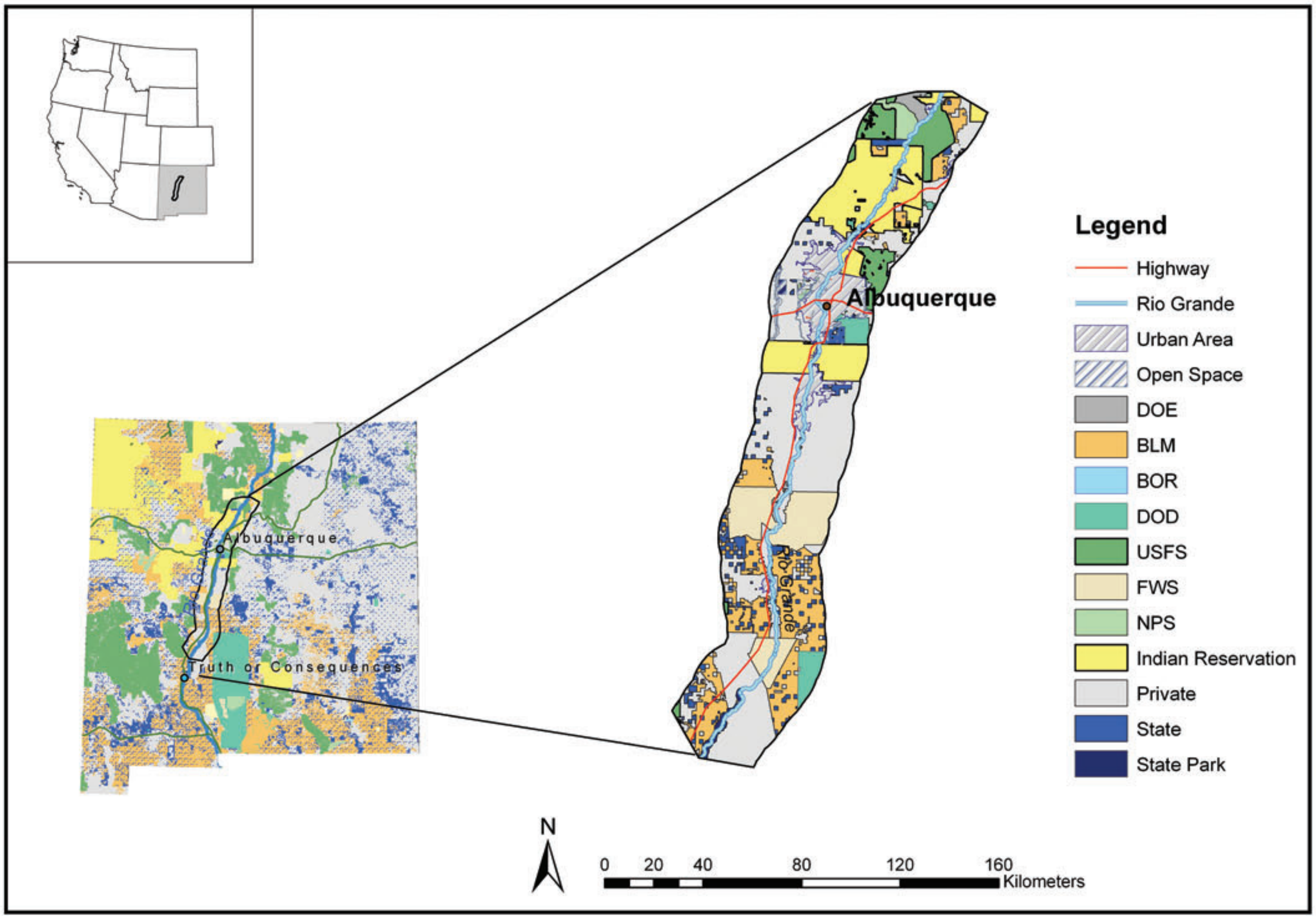


Figure 1.4. Land ownership in a 20-km buffer along the study site that corresponds with the MRG.

Amphibians are more common in mesic or aquatic environments. Mammal species prefer moist sites with dense vegetation (Hink and Ohmart 1984). The historic mosaic of habitats was maintained, in part, through the geomorphic structure of the MRG, which was characterized as a wide, shallow braided channel with high sediment loads and frequent flooding (USACE 2011). The habitats present today are much more homogenous than those of the past, and recent declines of habitat diversity within the MRG due to land conversion, dams, channelization, and other causes have led to species loss. Human-induced changes alone are known to have caused or contributed to the extirpation of over 20 species, and an additional 30 species warrant some degree of management intervention.

Over the last century, 90% of the riparian area in New Mexico and Arizona is estimated to have been converted to other land uses (CWCS 2006; Krzysik 1990; NMDGF 1988). Development has resulted in large-scale changes in the hydrology and biology of the Rio Grande and the overall degradation of many habitats, ecosystem functions, and processes (Fullerton and Batts 2003). Flow of the water in the MRG has been modified by dams, agriculture, and extraction for domestic use resulting in dramatic changes: upstream reaches of the Rio Grande are estimated to receive 50% of the pre-development flow volume (Fig. 1.5) and downstream reaches receive less than 10% of historic flow volumes (Fullerton and Batts 2003). Natural flood regimes have been reduced or eliminated altogether (Fullerton and Batts 2003). Current modifications and absence of floods combined with erosion

Figure 1.3.Typical habitat within the MRG. Top left—River bank in Albuquerque. This shows a small patch of willow with Russian olive in the far background. Top right—View of opposite bank showing the prominence of Russian olive in the riverbank vegetation. Bottom left—Open cottonwood forest in an area within the Albuquerque city limits that no longer receives regular floods. Bottom right—Cottonwood forest with emerging tamarisk (black arrows) and Russian olive (white arrows) understory.

control practices and salt build-up in substrates inhibit the establishment of new cottonwood individuals and stands (Fullerton and Batts 2003). Previous to drain and ditch construction, wetlands were the most extensive habitat types of the MRG valley. Wetlands have decreased by as much as 93% since 1918, experiencing the greatest historical decline of any floodplain community (Roberts 2005; USACE 2011). Recent climate changes, which show increased temperatures but little change in precipitation are likely to exacerbate issues relating to water loss and reduced flows (Fig. 1.6; Daly and others 2004; Gibson and others 2002). In addition, intensive grazing and logging in northern watersheds has increased sediment input and accelerated channel aggradation, or elevation, to the detriment of many native habitats (Roberts 2005; USACE 2011). Invasive species are also an important management issue within the Middle Rio Grande Bosque (MRGB). Saltcedar (*Tamarix* spp.) and Russian olive have become dominant in many reaches and are prevalent throughout the MRGB (Fullerton and Batts 2003) (Fig. 1.3). Russian olive tends to dominate in northern reaches, whereas Tamarisk is common in southern areas where it often forms monotypic stands (Crawford and others 1993). The presence of these species not only impacts wildlife through changes in the

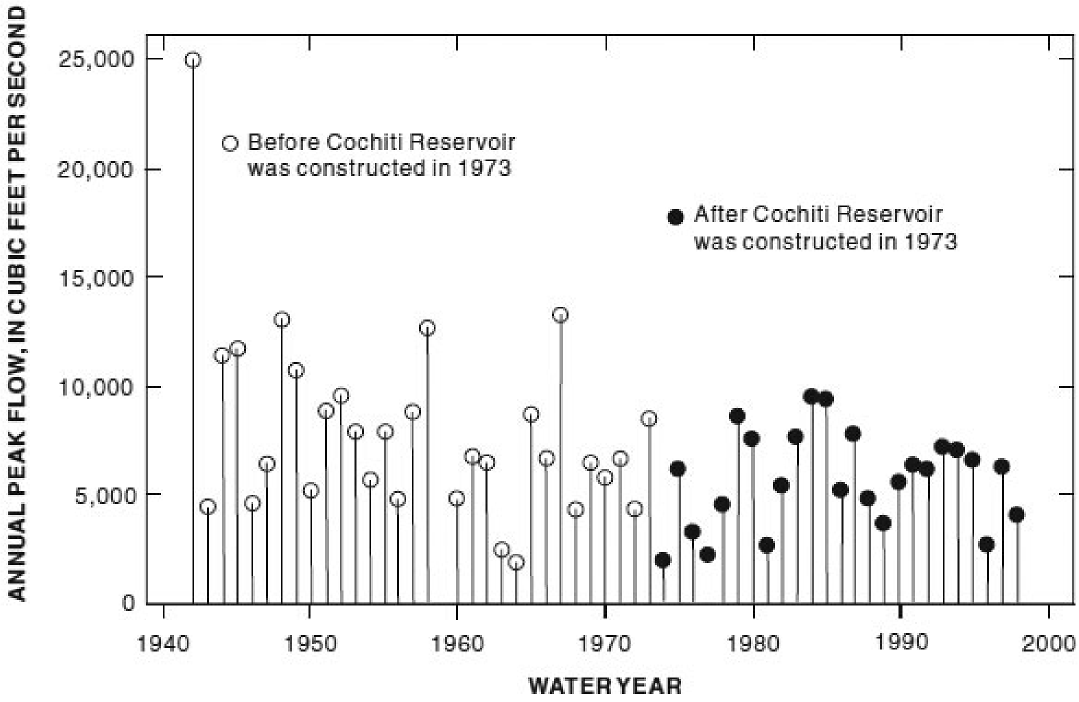


Figure 1.5. Annual peak flow along at the Albuquerque gage, pre and post Cochiti Dam (Reservoir), which went online in 1973. Adapted from the USGS Water Investigations report 02-4131 (2002).

structure of native riparian forests, but through chemical and other physiological changes (water draw and shade) as well as reduced regeneration of native species.

1.3 Climate Change in the Southwestern United States and MRGB

Mean global temperature is expected to increase by 1 to 6 °C (1.8-10 °F) over the next 100 years (IPCC 2007). The southwestern United States is expected to experience relatively large temperature increases (Seager and others 2007). Specific predictions for the Southwest are: increased severity and duration of droughts, more heat waves, greater variation in precipitation, increased wildfires, increased insect outbreaks by species like the bark beetle and gypsy moth, and increased evapotranspiration and salinization (Easterling and others 2000; Field and others 2007; Garfin and Lenart 2007). Recent warming trends have already impacted rivers and streams within the region (Figs. 1.5, 1.6, 2.2). Stream flows have decreased by 2% in the Rocky Mountain region over the last century (Field and others 2007), and evapotranspiration has increased by 55 mm in the Colorado and Columbia River basins since 1950 (Field and others 2007). Since the 1950s, 74% 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 to 30% (Field and others 2007; USACE 2011; Fig. 2.2). Changes to winter snow-rain ratios have important implications for western and southwestern riparian systems because they relate directly to spring and summer flow volume. Trends are generally weak or inconclusive regarding overall snowpack levels but several sources indicate earlier snowmelts, increased winter flows and early onset of springtime peak flows as compared to the first half of the century (McCabe and Wolock 1999; USACE 2011). Peak stream flows across the western United States occurred from one to four weeks earlier in 2002 as compared to 1948 (Field and others 2007). Importantly, these changes were attributed to warmer temperatures and not to changes in precipitation (Field and others 2007), which have been more

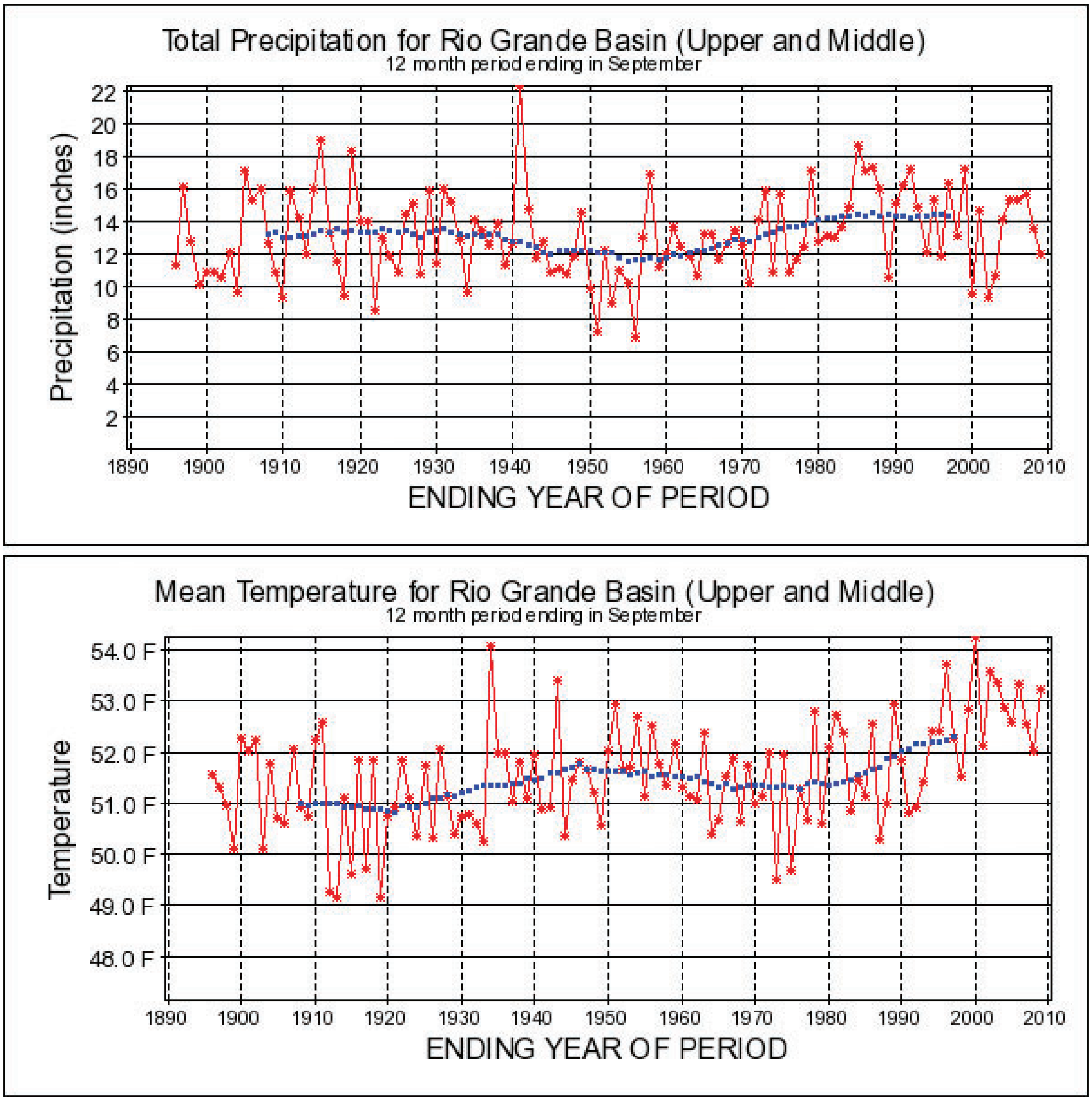


Figure 1.6. Observed temperature (top) and precipitation (bottom) changes for a 100-year period. From Western Climate Mapping Initiative (WestMap) available at: http://www.cefa.dri.edu/Westmap/. Red line indicates annual time series for the given geographic region. Blue line indicates 25-year moving annual mean values, where each value is plotted on the center year of its respective 25-year period. WestMap data are derived from the PRISM climate mapping system (Daly and others 2004; Gibson and others 2002).

difficult to project, particularly for the summer monsoon season (Mitchell and others 2002).

Within the Rio Grande Basin, temperatures have increased over the last century, with most of the warming in the last two decades (Fig. 1.6). This recent warming has been greatest (+1 °C) in the northern reaches of the Rio Grande Basin where the Rio Grande originates. The consequences of this temperature change are already evident in the Middle and Lower Rio Grande Valley (Figs. 1.6, 2.2 "SRM"). Increased winter and spring temperatures have led to a greater proportion of precipitation falling as rain rather than snow and an overall reduction in snowpack, both of which have direct consequences for spring and summer flow levels (Enquist and Gori 2008; Rood and others 2006). The hydrological consequences of increased temperatures include changes to flow volume, duration, and seasonality (Hurd and Coonrod 2008), which have impacted water availability, water quality, habitats, and wildlife in the Rio Grande. Observed trends of decreased stream flow, greater proportion of precipitation falling as rain rather than snow, and earlier snowmelt are expected to continue (Fig. 2.2; see Chapter 2 of this report; USACE 2011).

1.4 Climate and Management Challenges

Climate change will exacerbate ongoing management issues in the MRGB. Riparian systems in the southwest have been severely altered by human land use and recent droughts (Roberts 2005). Desert riparian systems are considered among the most endangered in the United States (Noss and others 1995). In 1993, the Rio Grande was considered the most endangered river in America by the national conservation group American Rivers (Roberts 2005) and, more recently, was designated one of the top 10 rivers at risk in the world by the World Wildlife Federation (Wong and others 2007). In Southwest riparian systems, drought and intense heat are likely to lead to increases in drought-related species mortality and shrinkage and fragmentation of riparian habitat—issues compounded by over-extraction of water and spread of invasive species (Palmer and others 2009). The Rio Grande is already recognized as suffering from excessive water extraction (Palmer and others 2009; Wong and others 2007). Changes to water temperature and flow regimes are buffered in free-flowing rivers with protected watersheds because vegetation and wetlands slow and absorb the impacts of increased rates of snowmelt and temperatures. Conversely, rivers such as the Rio Grande that include urbanized and logged watersheds and dams are less resilient to warming trends because they experience more evaporation, have less capacity to buffer changes in flow (and are associated with greater risk of extreme flood events), and contribute less to the recharge of ground water than unmodified rivers (sensu Palmer and others 2009). Lack of free-flowing rivers and functional wetlands can result in increased erosion, habitat destruction, and, ultimately, loss of water, leaving little water available to sustain environmental flows (Palmer and others 2009). In addition, rivers such as the MRG that rely on snowpack rather than groundwater are considered less buffered against the drying trends associated with global warming (Palmer and others 2009) because they are prone to extreme fluctuations in hydrological cycles with variations in climate.

Despite the relative scarcity of riparian habitat and the issues facing these habitats, river corridors continue to support a disproportionate amount of the diversity in Southwest flora as well as resident and migratory faunal communities (Pase and Layser 1977). Already, a number of species have been identified as at risk to changes in climate in New Mexico. With increasing droughts projected and potential loss of important riparian habitats, species sensitive to drought conditions, such as the southwestern willow flycatcher (*Empidonax trailii extimus*), are likely to experience further decline (Enquist and Gori 2008). For other species such as western painted turtles (*Chrysemys picta*), leopard frogs (*Lithobates pipiens*), and New Mexico garter snakes (*Thamnophis sirtalis dorsalis*), which rely heavily on perennial marsh/cienega habitats, increasing fragmentation and loss of riparian habitat will have profoundly negative impacts. Here, we examine the impact of climate change on a major riparian system in the Southwest, the MRGB, and its associated vertebrate taxa.

Chapter 2. Methods for Assessing the Vulnerability of Species in the MRGB to Climate Change

We assessed the vulnerability of vertebrate species within the MRGB to identify the relative risk of species to the possible effects of climate change. We used a pilot version of a decision-support tool, System for Assessing Vulnerability of Species (SAVS), developed by the U.S. Forest Service, Rocky Mountain Research Station (RMRS) for assessing the vulnerability of terrestrial vertebrates to climate change (Bagne and others 2011). The scoring system uses simple criteria for predicting the response of terrestrial vertebrate species to climate change and was specifically designed to be applied by managers.

2.1 Species Vulnerability Assessment Tool

The assessment tool is a questionnaire (with multiple-choice responses selected by the user) based on criteria that contribute to the potential vulnerability of vertebrate species to climate change. Questions are grouped into four categories by theme: habitat, physiology, phenology, and biotic interactions (Table 2.1; Appendix 1). Each question addresses one or more of the following aspects of vulnerability: exposure, sensitivity, and/or adaptive capacity (Bagne and others 2011). Each question focuses on a trait that influences individual species' response to the direct and indirect effects of changes in temperature, precipitation, and extreme events, which are important in driving population dynamics and natural selection (Boag and Grant 1984; Easterling and others 2008; Parmesan and others 2000), as well as related climate phenomena such as frost, snowpack, and sea level. Before scoring, estimates of potential exposure to limiting conditions are generated through the creation of a future climate scenario that outlines expected changes in weather, disturbance regimes, and important biological elements within and including the species' habitat. Sensitivity and adaptive capacity relate to species' traits, which are considered to influence the likelihood that a species will experience a negative impact from the change or, alternatively, whether it can adapt to the change in such a way to minimize impact to the population.

The traits represented by questions in the habitat category are exposure to habitat loss (both breeding and non-breeding), decline in habitat components and habitat quality, ability to disperse to new habitat, and requirements for stopover sites (i.e., migratory habitat) (Table 2.1). By examining these aspects of habitat change, we are able to determine the influence of species habitat use and evaluate species' capacities to deal with future change or exposure to increased change in suitable habitat. Physiological traits include traits that might influence a species' survival through effects on its energetics or physiological limitations. Recorded physiological limits, sensitivities to extreme events, temperature-determined sex ratios, torpor and hibernation, and adaptations to deal with or reproduce under variable resources are included in this category. Phenology sensitivities include reliance on climate cues that may change in the future, reliance on timed events that are influenced by climate, temporal or spatial separation between life history stages, and initiation of activity and capacity to successfully time reproduction to resources that vary. Finally, we consider species sensitivity and exposure to changes in five primary biotic interactions: food, predators, symbionts, disease, and competitors.

Table 2.1. Questions, standard responses for taxon groups, and range of points possible used to score species vulnerability to climate change. Scores which were consistent across all species within a taxon are highlighted. Scores of 1 indicate vulnerability, 0 indicate no effect or change or information, and -1 indicate resilience. Score ranges that fall outside the standard 1, 0, -1 range are also highlighted under the Score Range column.

	Amphibians	Birds	Mammals	Reptiles	**Score Range**
HABITAT					
1. Is the area or location of the general associated vegetation type used for breeding activities by this species expected to change?					2 /-2
2. Is the area or location of the general associated vegetation type used for non-breeding activities by this species expected to change?					2 /-2
3. Are specific habitat components required for breeding expected to change within associated vegetation type?					1/ -1
4. Are specific habitat components required during non- breeding expected to change within associated vegetation type?					1/ -1
5. Are specific habitat components required for survival expected to change within associated vegetation type?					1/ -1
6. Within habitats occupied, are features of the habitat associated with better reproductive success or survival expected to change?					1/ -1
7. What is the potential for this species to disperse?		-1			1/ -1
8. Does this species require additional habitats during migration?	0			**0**	1, 0
PHYSIOLOGY					
1. Are limiting physiological conditions expected to change?					1/ -1
2. Is sex ratio determined by temperature?	0	0	0		1,0
3. Are disturbance events that affect survival or reproduction expected to change?					1/ -1
4. Is activity period expected to change?					1/ -1
5. Does this species possess an ability to reduce metabolic energy or water requirements?	0			0	1/ -1
6. Does this species have lower energy requirements or possess the capacity to store energy or water in the longterm?	-1	1		-1	1, -1, none
7. Does this species have employ adaptive strategies?					0, -1
8. Can this species outlive periods where reproduction is limited?					1/ -1
PHENOLOGY					
1. Does this species use temperature or moisture cues to initiate activities?					1, 0
2. Are activities tied to discrete resource peaks that are expected to change?					1/ -1
3. What is the separation between cues/activities and discrete events/ resources?	-1			0	1/ -1
4. Does this species have more than one reproduction event per year?					1, -1
INTERACTIONS					
1. Are important food resources for this species expected to change?					1/ -1
2. Are important predator populations expected to change?					1/ -1
3. Are populations of symbiotic species expected to change?					1/ -1
4. Is prevalence of disease in this species expected to change?					1/ -1
5. Are populations of important competing species expected to change?					1/ -1

Vulnerability scores result from a tally of points; a response that reflects anticipated vulnerability to climate change is given a positive point value, one that reflects anticipated resilience to climate change is given a negative point value, and one that reflects a predicted absence of an effect or neither a benefit nor a cost relative to climate change is given a point value of zero. For each species, an overall score is calculated considering all criteria together. In addition, questions are grouped into four categories—habitat, physiology, phenology, and biotic interaction—and a score is calculated for each category. The pilot version used in this assessment included 25 questions, resulting in overall scores ranging from -20 to +20 and categorical scores ranging from -5 to +5 (Table 2.1; Box 1).

Box 1. Interpreting species vulnerability

Output scores are based on a continuous linear scale representing the balance between vulnerable and resilient traits. To aid interpretation, we grouped scores into three classes representing vulnerable, neutral, and resilient species. It is important to note that species scores near zero represent either an equal mix of traits contributing toward vulnerability and resilience or the predominance of truly neutral (score = 0, no effect) scores for the vulnerability factors. We categorized scores to simplify interpretation and help us identify species or groups of species that may be at greater risk or increase benefit to climate changes (Fig. 3.1).

Species assessed in this study tended to fall out among those that scored near +5 and those that scored near +1 range, with the latter group less likely to be vulnerable to climate change (Figs. 3.1, 3.3, 3.5, 3.7, and 3.9). Therefore, we made an arbitrary cutoff point to indicate vulnerability when overall scores exceed +2 (on a potential scale from -20 to +20) or exceed +1 for categorical scores (scale -5 to +5). Similarly, species were categorized as resilient for overall scores less than -2 overall or less than -1 within a category. Scores that fall within the range of -2 to +2 or -1 to +1 for overall or categorical scores, respectively, were generally considered neutral.

2.2 Methods: Background

Species Selection

In SAVS, species are assessed within a geographical extent defined by the user; in this case, this included the MRGB and a 20-km buffer of adjacent upland habitats (Fig. 2.1). In addition, areas important to migrating species (e.g., wintering grounds) were included in the evaluation of potential change.

Vertebrate species for the region were initially identified using *A Field Guide to the Plants and Animals of the Middle Rio Grande Bosque* (Catron and others 2008). RMRS site data on birds, amphibians, reptiles, and bats were also used to identify and/or verify candidate species for scoring. We included species that reside within the MRGB for at least part of the year but excluded waterfowl and species that occur primarily in upland habitats (Table 3.1; Appendix 1). Both breeding and wintering populations were included in the assessment list. Species that use the

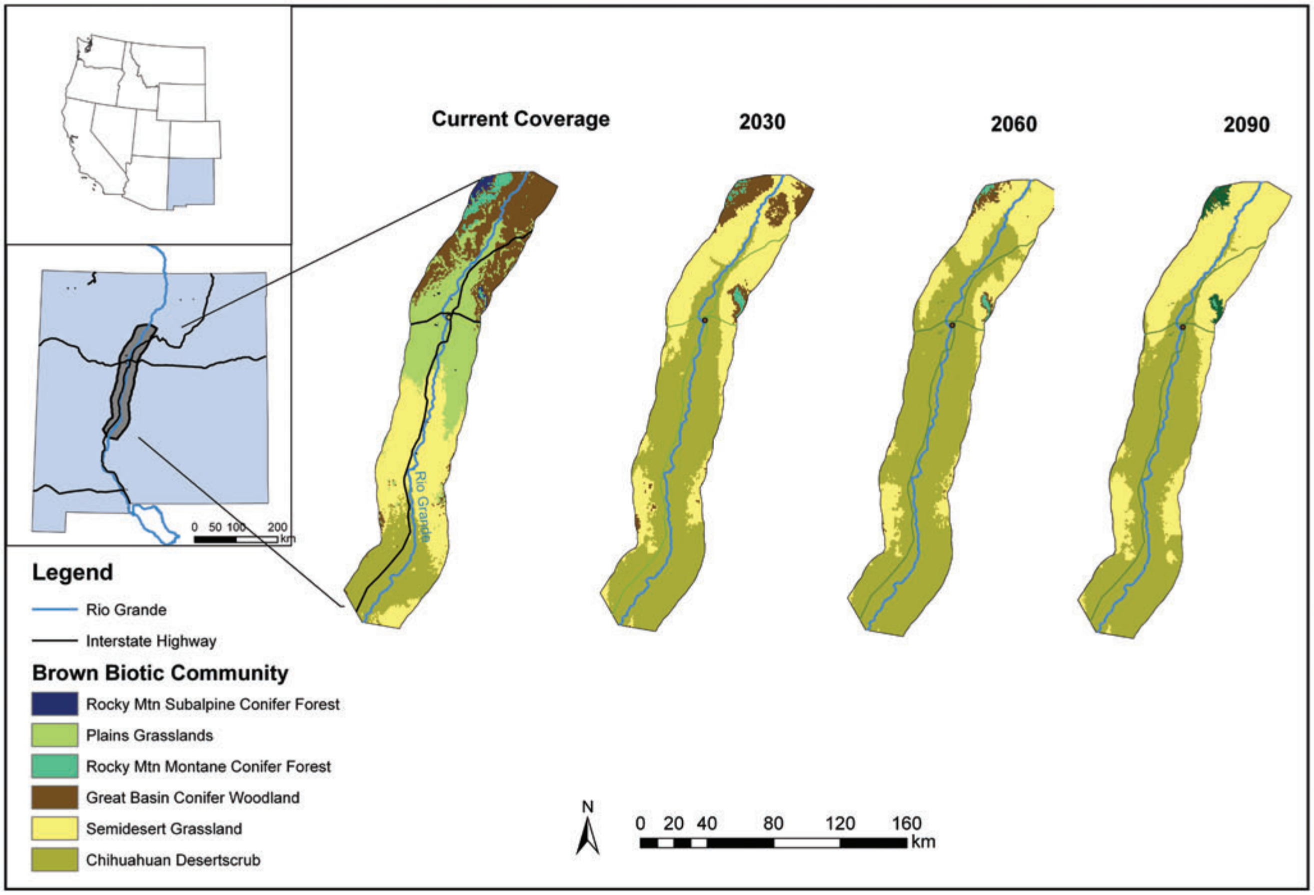


Figure 2.1. Land cover for a 20-km buffer zone around the Rio Grande in New Mexico. Here we show vegetation projections within this buffer zone for 2005 and three future time periods as produced by Rehfeldt and others (2006). Chihuahuan desert scrub becomes predominate under future climate regimes.

riparian corridor solely for migration, or otherwise had an intermittent or transient presence within the MRGB, were not included in the assessment.

Future Climate and Vegetation Scenario for the MRG

To initiate the assessment process, we first developed a scenario of climate change and vegetation response to estimate exposure of species to climate change. This scenario was used to provide context for selecting the appropriate response and to score species. To make predictions for the MRGB, we gathered information on both observed historical trends and projections for changes in climate and related phenomena, disturbances, and vegetation. Recent climate trends and impacts were presented in the background material of Chapter 1. For future conditions, we used climate modeling tools available from Climate Wizard and Rehfeldt and others (2006) and primary literature sources to create a predicted climate change scenario for the MRGB (Tables 2.2, 2.3; Figs. 2.1, 2.2). We relied on projections and information covering periods 20 to 60 years in the future.

ClimateWizard, developed by The Nature Conservancy, University of Washington, and University of Southern Mississippi, provides estimated changes in precipitation and temperatures using projections from the World Climate Research Programmer’s Coupled Model Intercomparison Project phase 3 and can incorporate a number of emission scenarios (IPCC 2000) and general circulation

Table 2.2. Data collected from Climate Wizard for changes in precipitation and temperature. Precipitation projections were retrieved in 2009 for a period representing change in average precipitation 2010-2060 and are given for the upper (U) and lower (L) reaches of the MRG. Temperature projections represent departure from averages that are expected to occur in 2050 and 2080.

	Precipitation (inches)		**Departure in Average Temperature (˚ F)**	
	U	**L**	**2050**	**2080**
Winter (Dec-Feb)	0 to -1.0	0 to -0.5	+3.0-4.0	+5.5
Spring (Mar-May)			+5.5-6.0	+6.0
Summer (June-Aug)			+5.5	+6.0
Fall (Sep-Nov)			+5.5	+6.0

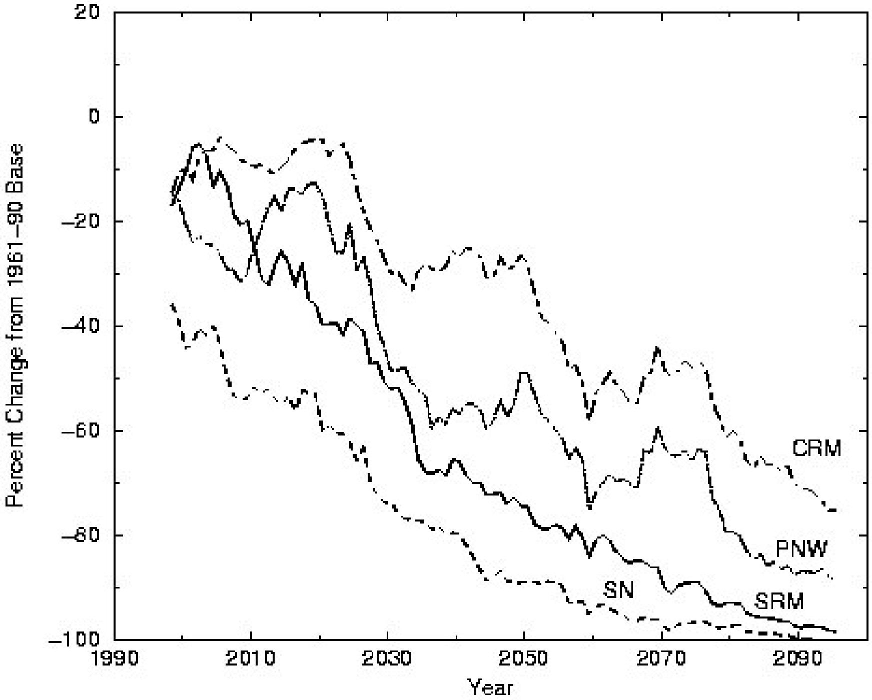
Table 2.3. Summary of climate projections and predictions used as the basis for scoring species assessments inhabiting the Middle Rio Grande.

Impact	**Source(s)**
Less open water, reduced flow	Hurd and Coonrod 2008; Johnson and others 1995; Milly and other 2005
Spring floods occur earlier	Annear and others 2004; Rood and others 2003
Reduced riparian habitat available due to decreased stream flows and longer droughts	Saunders and others 2008; Shafroth and others 2002
Cottonwood gallery forests likely to decline due to lower water flows and increasing wildfires, which reduce recruitment and increase mortality	Howe and Knopf 1991
Loss of mature trees due to fire and disease	Obedzinski and others 2001
Loss/reduction of native vegetation and replacement by invasive species due to increased fire and lower water tables.	Obedzinski and others 2001
Increase in *Tamarix* spp. and, potentially, Russian olive trees	Annear and others 2004; Obedzinski and others 2001; Shafroth and others 2002; Stromberg, 1998
Replacement of grassland and woodland habitats with scrubland	Merritt and Cooper 2000; Rehfeldt and others 2006
Reduced vegetation cover due to increased aridity	Saunders and others 2008
Potential increase in exotic grass species (which might increase vegetation cover)	Obedzinski and others 2001
Increased fire intensity and frequency and prolonged fire season	Swetnam and Betancourt 1990
Increased duration of drought that last five years or more	Seager 2007
More intense droughts (drier conditions)	Seager 2007

models (GCMs). We used data generated from an ensemble GCM under an A2 emission scenario for the 30 year average covering 2010 to 2060 (Table 2.2).

Vegetation projections created by Rehfeldt and others (2006) provide current and future community compositions based on the IPCC IS92a scenario (1% increase in greenhouse gases per year after 1990) and two GCMs—the Hadley Center (HadCM3GGA1) and the Canadian Center for Climate Modeling and Analysis (CGCM2_ghga).We downloaded raster images produced by Rehfeldt (available from http://forest.moscowfsl.wsu.edu/climate/customData/) of Brown

Figure 2.2. Results of six ensembles of hydroclimate projections for the Rio Grande basin above the Elephant Butte Dam: annual total precipitation (top left), annual mean temperature (top right), April 1st snow water equivalent (middle left), annual runoff (middle right), December-March runoff season (bottom left), and April-July runoff season (bottom right). The heavy black line is the annual time series of 50 percentile values (i.e., ensemble-median). The shaded area is the annual time series of 5th to 95th percentile. Adapted from BOR Secure Water Report (http://www.usbr.gov/climate/SECURE/docs/SECUREWaterReport.pdf_). Original figure from McCabe and Wolock 1999.



biotic communities projected for years 2005 and 2090 and shapefiles for the middle Rio Grande Basin into ArcMap 9.3 to create current (Year 2005) and projected (Year 2050, 2090) vegetation maps for the MRGB (Fig. 2.1). We estimated percent change by calculating the difference in percentage of each Brown biotic community (Brown and others 1998) between successive time periods. Using this method, we analyzed projections for the MRGB as well as upland areas surrounding the MRGB.

Temperature and Precipitation

Climate projections are presented in Table 2.2. The largest monthly change of +5 °F (2.75 °C) is projected for July, August, and October (Climate Wizard, downloaded May 2009). Climate Wizard projections for the MRG north of Albuquerque predict an average of 4 °F (2.2 °C) increase. Precipitation change is predicted to either remain the same or decrease. Annual precipitation may decrease as much as a 0.5 inches for lower reaches of the MRG and 1 inch for upper reaches (Table 2.2).

Hydrology

Warmer temperatures are predicted to have many consequences for the hydrology of the MRG (Table 2.3). Earlier spring peak stream flows are projected (Rood and others 2003; Annear and others 2004) as well as a large (up to 30%) reduction in the overall stream flow for the Rio Grande (Hurd and Coonrod 2008; Johnson and others 1995). These changes, along with the impact of human population growth on future water demand (McAda and Barroll 2002), lead to a strong likelihood that the water table will drop substantially and the Rio Grande may become intermittent in some areas, particularly the southern reaches (Fig. 2.1) (Hurd and Coonrod 2008). Milly and others (2005) used 12 different models to assess future changes to runoff in rivers of the United States and project a substantial decrease in annual runoff in the Southwest under future conditions. McCabe and Wolock (1999) also estimated a drastic reduction in snowpack with warming climate (Fig. 2.2). As a result of hydrological changes, we predict that there will be less open water, shorter duration for ephemeral ponds, and a decline in wetland habitats.

Vegetation Communities

Projections for 2030 showed decreases in Great Basin Conifer Woodlands (from 10 to 1%), Semi-Desert Grasslands (38 to 25%), and a complete loss of Plains Grasslands (estimated to comprise 52% of the current habitat) (Fig. 2.1). Chihuahuan Desert Scrub was predicted to increase from 0 to 74%. In addition, hydrological changes are predicted to impact the MRGB vegetation in a number of ways (Table 2.3). First, there is likely to be a general loss of riparian vegetation and a narrowing of the riparian corridor. In particular, these reductions will be apparent in ephemeral or intermittent streams where reduced flow is likely to reduce areal extent of riparian vegetation (Shafroth and others 2002). This, in turn, will affect many areas currently linked to tributaries (e.g., Rio Puerco) as well as the lower reaches of the Rio Grande where the stream flows already exist only below the surface in many years. There may also be a shift to early non-forested seral stages of vegetation along some reaches of the river due to increased fire and the establishment of emergent plant types in channels with little water (Johnson and others 1995). Drought tolerant grasses and forbs, as well as exotic species such as Tamarisk, are likely to be favored and will further reduce natural woody vegetation in the riparian zone (Obedzinski and others 2001).

Gallery cottonwood forests (*Populus* spp.) along the Rio Grande are in decline (Howe and Knopf 1991), and increased future stress due to water limitations and increased fire could favor the establishment of Tamarisk and other exotics over cottonwood (Obedzinski and others 2001). We predict that Tamarisk stands are likely to increase in area within the MRGB to the detriment of cottonwoods. Changes in the timing of peak flows, sustained high peak flows, and winter floods are all thought to favor the recruitment of tamarisk, which has a prolonged establishment period (Annear and others 2004; Shafroth and others 2002; Stromberg 1998). Once established, shading caused by exotic species reduces cottonwood recruitment (Obedzinski and others 2001). In addition, tamarisk is relatively resistant to fire and drought (Shafroth and others 2002). These changes have been noted in other southwestern river systems. In recent years, as temperatures have increased and precipitation decreased, researchers have observed declines in cottonwood and willow species along the San Pedro River in Arizona and dramatic increases in mesquite, though no change in tamarisk has occurred (Stromberg 2006). In Colorado, hydrologic changes similar to those projected (i.e., reduced flows and shallower water) has led to loss of cottonwood in favor of desert scrub habitats (Merritt and Cooper 2000).

Overall, we expect a loss of canopy tree species in the MRGB. Vigorous resprouting of shrubs and trees after fires and related disturbances are expected to lead to at least a temporary increase in shrub habitats. However, for longer projection periods (beyond 20 years), we expect an overall decrease in primary plant cover due to the synergistic effects of increased disturbance and decreased water availability. Some diseases may become more problematic for native vegetation in the future and exacerbate the loss of riparian forest habitat. Mistletoe infections can reduce tree vigor and lead to mortalities, especially under conditions of water stress (Obedzinski and others 2001). Increases in the willow borer that affect willow species may also be more problematic in the future as plants become more stressed (Obedzinski and others 2001).

Extreme Events

Timing and volume of peak flow, fire, storms, and heat waves have the potential to result in significant change within the MRGB. Changes to disturbance regimes include an increased risk of damaging fire, earlier and more intense floods, longer and more severe droughts, and an increased risk of storms. We considered the potential for all of these factors in our assessment. Although extreme weather conditions and disturbances are more difficult to project than changes in temperature, we considered their potential effect to be more critical to wildlife populations than changes in average conditions (Parmesan and others 2000). Most climate models predict that droughts will increase in the Southwest and are likely to increase in severity (Seager 2007). Droughts can reduce the competitive ability of native plant species against exotic species that are better adapted to dry conditions with less flooding (Obedzinski and others 2001). The drought period from 2000 to 2004 was one of the longest recorded in the West for the past century (Saunders and others 2008). We used this recent five-year drought as a conservative expectation for drought duration to assess potential impact on vertebrate species. Water stress will leave many species more susceptible to disturbances, such as disease, fire, and insect outbreaks, and will contribute to the overall decline in riparian vegetation within the MRGB (Saunders and others 2008).

We also expect an increase in fire frequency and severity within the MRG, fewer frost days, more intense precipitation events, and higher maximum temperatures (Easterling and others 2000). High temperatures and more variable rainfall will increase conditions favorable to wildfires (Swetnam and Betancourt 1990).Though tamarisk is known to be relatively resistant to fire and drought (Shafroth and others 2002), both it and native cottonwood are susceptible to high mortality from high severity fire (Ellis 2001). Within the MRG, tamarisk and cottonwoods are known to resprout after fires, but recent high intensity burns appear to favor the establishment of the invasive species over native cottonwoods (Ellis 2001). Though fires are quite common in the Southwest in general, we know little about the historic fire regime within the MRGB. It is likely that fires did occur in the Bosque forests though with less severity than commonly seen today. Decreased flooding and fire suppression efforts within the riparian forests have led to high fuel loads and a number of high intensity fires. Insect outbreaks, which result in significant defoliation as seen with the forest tent caterpillar (*Malacosoma* spp.), may leave plants and, in particular, cottonwoods more vulnerable to mortality due to stresses (Obedzinski and others 2001), ultimately contributing to build-up in woody debris and likelihood of high severity fires.

Though water flows in general are expected to decrease, we also expect an increase in severe flood events due to rapid melting of snow under warming conditions. High volume, high velocity flood events can scour riparian habitats and have been responsible for declines in native woody species in the Verde River in Arizona. Within this same system, peak storm flows have increase eight-fold over a 30-year period indicating that damaging floods may occur more often as a result of increasing storm intensity as well as more rapid snow melt (Obedzinski and others 2001). We generally assumed that floods may become more damaging through changes in the timing and volume of flow but, at the same time, less likely to coincide with important establishment periods of native vegetation.

Migratory Habitats

For migrating species, climate change will impact multiple habitats unpredictably. These species are generally more vulnerable to potential negative effects because changes across multiple habitats are more likely to occur at different rates or result in different habitat characteristics across the range of the species (Wormworth and Mallon 2006). Predictions of future range shifts in bird species that overwinter in Mexico and South America are expected to lead to a 20% decline in bird species in the eastern Midwest; we assume that similar changes are likely to affect bird species migrating throughout the western United States. A recent study of Mexican bird fauna (1179 species) predicted more than 40% of the local communities will experience some degree of turnover as a result of changes to habitat from fluctuations in temperature and precipitation (Peterson and others 2002). The greatest number of extinctions has occurred in the open Chihuahuan desert and northwest coastal plain. Pounds and others (1999) identified increases in lowland and premontane bird species at higher elevation sites within Costa Rica's highland forests as a result of warming temperatures.

Birds use a variety of habitats in Central and South America many of which are likely to experience profound change. Specific Latin American projections are: (1) increased temperatures in many areas (e.g., glacier retreat in Columbia, Peru, and Ecuador); (2) climate is expected to become more suitable for tropical vegetation toward the south, whereas northern South America is projected to be drier; (3) increased sea level rise which threatens mangroves; (4) tropical forests are expected to be replaced by savannahs accompanied by an increased risk of fire (Magrin and others 2007). Latin America has already experienced increases in extreme weather, such as flooding, windstorms, and heat waves, and the trend is expected to continue (Magrin and others 2007). Areas of Mexico are also subject to widespread buffelgrass introduction and invasion, which reduces shrub habitats in favor of grasslands (Burquez-Montijo and others 2002).

Rainfall in Central America has increased in some areas and decreased in others over the last 50 years (Magrin and others 2007), while average and maximum temperatures have increased and precipitation has slightly decreased in southern Mexico between 1960 and 2004 (Peralta-Hernandez and others 2009). Future changes in temperature for Mexico range from 1.6 to 2.5 °C (Peterson and others 2006). Analysis by the Intergovernmental Panel on Climate Change Earth simulator shows significant temperature increases and a net reduction in rainfall over Mexico (Pérez and others 2008). Intense anomalies are more prevalent in the eastern Pacific, with specific consequences for northern Mexico. The maximum number of dry days is likely to increase in central and southern Mexico but may decrease in more northern areas.

The effects of climate change in South America are most notable with respect to the loss of glaciers at high altitudes. Future extreme precipitation and temperature events may have significant impacts for the Amazon and northern coasts of Peru (Pérez and others 2008). Within the Caribbean basin, hurricanes have become more intense, and widespread die off in coral are evidence of recent warming trends. The rate of temperature increase is likely to be the greatest in the Andes, resulting in rapid tropical glacier retreat and changes to Andes montane system (Pérez and others 2008). The Amazon basin is also expected to experience drastic temperature increases and disrupted precipitation cycles with consequences on carbon storage, soil temperatures, and, ultimately, the climate of the entire South American region (Pérez and others 2008).This area may eventually transition to a savannah type of habitat. As it does, the important role of the Amazon forest to sequester carbon and contribute moisture to the atmosphere will be reduced and could lead to desertification of many other habitats within South America. Important coastal habitats are likely to be affected by rising sea level within the Caribbean basin and coast of Ecuador appears to be most vulnerable to coastal flooding issues (Pérez and others 2008).

Biological Information Sources

For species-specific biological information, we used published literature sources (i.e., primary scientific literature and books), including searching various web databases such as the Web of Science® and Google Scholar. Search criteria included species' scientific and common names and specific terms relating to relevant limitations or requirements where needed (e.g., bat kidney function and hibernation + species name).We also used species information available through Amphibiaweb (http://amphibiaweb.org/), Birds of North America (http://bna.birds.cornell.edu/bna/), BISON-M (http://www.bison-m.org/), NatureServe's Explorer (http://www.natureserve.org/explorer/), and online publications such as the U.S. Geological Survey Disease publication (http://www.nwhc.usgs.gov/publications/field_manual/), Mammals of Texas (http://www.nsrl.ttu.edu), Western Governors' Drought Task Force report (http://www.westgov.org/climate), and Journal of Mammalogy mammal accounts (http://www.science.smith.edu/departments/Biology/VHAYSSEN/msi/msiaccounts.html). We also consulted with species experts and RMRS unpublished data when published data were unavailable.

Chapter 3. Vulnerability of Species in the Middle Rio Grande Bosque to Climate Change

We conducted scoring according the process described in Chapter 2 and as outlined in Bagne and others (2011). We scored the 117 species in this assessment based on their possession of or response to 25 characteristics that indicate potential vulnerability or resilience to climate changes (see Appendix 1, Table 2.1). Scores refer only to response to climate change and related phenomena, such as disturbance, and do not assess vulnerability to all factors that affect populations. In this chapter, we present the results of the assessment scoring process and discuss general trends among taxa with respect to these scoring variables (Box 1). We also calculated uncertainty scores (Appendix 2) based upon the availability and quality of data regarding species traits. Though not used in the following discussion, uncertainty scores provide a good indication of how confident we were in the assigned score for a particular species and category. Appendix 3 contains specific details regarding data and decision making processes for each of the species scored in this assessment.

3.1 Summary of Results

Our results show a skew in the distribution of vulnerable to resilient species. We found fewer species were predicted to be without impact or at benefit from climate change and most fell within the vulnerable category. In addition, species that fell within the resilient group typically did not achieve a score that mirrored the magnitude or scale of the vulnerable scores; the most vulnerable species received a score of 11.2, whereas the most resilient received a score of -3.3.

Sixty-two percent of all species, 69% of birds, 62% of reptiles, 44% of amphibians, and 57% of mammals had scores indicating vulnerability (overall score >2). Overall vulnerability scores for the 117 species ranged from a low (indicating resilience) of -4.4 to a high (indicating vulnerability) of 11.5 (Table 3.1). Bird species had the highest average and median scores of any taxonomic group and the largest range of variation between minimum and maximum scores (Table 3.1). Of the four categories—habitat, physiology, phenology, and biotic interactions—birds had the highest (most vulnerable) scores for physiological indicators of vulnerability, amphibians had the highest vulnerability for habitat and biotic interactions, and reptiles had the highest average phenology score. The distribution of overall scores within each taxonomic grouping was mostly unimodal and skewed toward higher vulnerability scores, except for amphibians, where the distribution was bimodal (Fig. 3.1). The following discussion highlights vulnerability trends within habitat, physiology, phenology, and biotic interaction categories for each taxon.

Table 3.1. Summary of categorical and overall vulnerability scores for 117 terrestrial vertebrates residing within the Middle Rio Grande Bosque, NM. Categorical scores range from -5 to +5 and Overall scores are scaled to -20 to +20. Bold values are taxon averages. (I) indicates introduced species.

Common Name	Species	Habitat	Physiology	Phenology	Interactions	Overall
Amphibians		**1.17**	**0.40**	**0.37**	**0.67**	**3.18**
Plains spadefoot toad	*Spea bombifrons*	-1.22	0.00	-0.83	0.00	-2.05
American bullfrog (I)	*Lithobates [Rana] catesbeiana*	1.39	-1.43	-5.00	0.00	-1.99
Couch's spadefoot toad	*Scaphiopus couchii*	-0.72	0.00	2.08	0.00	0.32
New Mexico spadefoot toad	*Spea multiplicata*	-0.72	0.00	2.08	0.00	0.32
Great Plains toad	*Anaxyrus cognatus*	-0.72	1.43	2.08	0.00	1.92
Woodhouse's toad	*Anaxyrus woodhousii*	3.00	0.71	-2.08	1.00	5.19
Barred tiger salamander	*Ambystoma mavortium mavortium*	2.50	0.71	2.08	1.00	6.73
Western chorus frog	*Pseudacris triseriata*	3.00	1.43	2.08	1.00	8.33
Northern leopard frog	*Lithobates pipiens*	4.00	0.71	0.83	3.00	9.87
Reptiles		**1.11**	**0.07**	**0.75**	**0.14**	**2.40**
Desert grassland whiptail	*Aspidoscelis uniparens*	-1.70	0.00	-0.42	0.00	-2.70
Desert or common kingsnake	*Lampropeltis getula*	-0.06	-1.43	2.50	0.00	-0.19
Western diamondback rattlesnake	*Crotalus atrox*	-0.56	-0.71	2.50	0.00	-0.19
Western/prairie rattlesnake	*Crotalus viridis*	-0.56	-0.71	2.50	0.00	-0.19
Ornate box turtle	*Terrapene ornata*	1.00	0.00	-2.08	0.00	0.51
Western hognosed snake	*Heterodon nasicus*	-0.06	-1.43	2.50	1.00	0.58
Gophersnake	*Pitouphis catenifer*	-0.56	0.00	2.50	0.00	0.58
Common gartersnake	*Thamnophis sirtalis*	-0.06	-0.71	2.50	0.00	0.58
Racer	*Coluber constrictor*	-0.06	0.00	2.50	0.00	1.35
Prairie lizard or sw fence lizard	*Sceloporus cowlesi*	0.50	0.71	-0.42	0.00	1.41
Western whiptail	*Aspidoscelis tigris*	0.50	0.71	-0.42	0.00	1.41
Glossy snake	*Arizona elegans*	-0.06	0.71	2.50	0.00	2.18
Western terrestrial gartersnake	*Thamnophis elegans*	-0.06	0.71	2.50	0.00	2.18
New Mexico whiptail	*Aspidoscelis neomexicana*	0.50	0.00	2.50	0.00	2.24
Plains black-headed snake	*Tantilla nigriceps*	0.50	0.00	2.50	0.00	2.24
Red-eared slider (I)	*Trachemys scripta*	2.50	0.00	-2.08	0.00	2.82
Coachwhip	*Masticphis flagellum*	1.94	-1.43	2.50	0.00	2.88
Side-blotched lizard	*Uta stansburiana*	0.50	2.14	-0.42	0.00	2.95
Snapping turtle (I)	*Chelydra serpentina*	3.00	0.00	-2.08	0.00	3.59
Chihuahuan spotted whiptail	*Aspidoscelis exsanguis*	2.50	0.00	-0.42	0.00	3.72
Checkered whiptail	*Aspidoscelis tesselata*	2.50	0.00	-0.42	0.00	3.72
Many-lined skink	*Eumeces multivirgatus*	1.50	0.00	2.50	0.00	3.78
Western painted turtle	*Chrysemys picta*	3.00	0.71	-2.08	0.00	4.36
Big Bend slider	*Trachemys geigeae*	3.50	0.00	-2.08	0.00	4.36
Spiny softshell turtle	*Apalone spinifera*	3.00	0.71	-2.08	0.00	4.36
Checkered gartersnake	*Thamnophis marcianus*	1.44	0.71	-0.42	2.00	4.42

Table 3.1. *Continued.*

Common Name	Species	Habitat	Physiology	Phenology	Interactions	Overall
Desert spiny lizard	*Sceloporus magister*	2.50	0.71	-0.42	0.00	4.49
Black-necked gartersnake	*Thamnophis cyrtopsis*	1.44	0.71	2.50	1.00	5.26
Great Plains skink	*Eumeces obsoletus*	3.50	0.00	2.50	0.00	6.86
Birds		0.40	2.26	0.03	0.46	3.69
Greater roadrunner	*Geococcyx californianus*	-3.33	1.43	-2.08	0.00	-4.42
Brown-headed cowbird	*Molothrus ater*	-2.78	1.43	-3.33	0.00	-4.29
House finch	*Carpodacus mexicanus*	-2.78	2.14	-3.33	0.00	-3.53
Mourning dove	*Zenaida macroura*	-2.78	2.86	-3.33	1.00	-1.99
European starling (I)	*Sturnus vulgaris*	-2.78	2.14	-0.83	0.00	-1.99
American kestrel	*Falco sparverius*	-2.28	2.14	-2.08	1.00	-1.22
Blue grosbeak	*Passerina caerulea*	-2.28	2.14	0.83	1.00	0.38
Indigo bunting	*Passerina cyanea*	-1.78	2.86	-0.42	0.00	0.38
Spotted towhee	*Pipilo maculatus*	-1.67	2.14	0.83	0.00	0.51
Northern rough-winged swallow	*Stelgidopteryx serripennis*	-0.67	2.86	-2.08	0.00	1.22
Eastern bluebird	*Sialia sialis*	-0.06	2.14	-3.33	1.00	1.35
Lesser goldfinch	*Carduelis psaltria*	-0.56	1.43	0.83	0.00	1.35
Black phoebe	*Sayornis nigricans*	1.44	0.71	-2.08	0.00	1.99
Ash-throated flycatcher	*Myiarchus cinerascens*	-0.67	2.14	0.83	0.00	2.05
Western kingbird	*Tyrannus verticalis*	-1.17	2.86	0.83	0.00	2.05
Bewick's wren	*Thryomanes bewickii*	0.44	2.14	-2.08	0.00	2.12
Phainopepla	*Phainopepla nitens*	0.44	2.14	-2.08	1.00	2.82
Great horned owl	*Bubo virginianus*	0.44	0.71	0.83	1.00	2.88
Ladder-backed woodpecker	*Picoides scalaris*	-0.06	2.14	0.83	0.00	2.95
Bushtit	*Psaltriparus minimus*	1.44	2.14	-2.08	0.00	3.65
Black-chinned hummingbird	*Archilochus alexandri*	0.44	2.14	0.83	1.00	4.42
Black-capped chickadee	*Poecile atricapillus*	2.44	0.71	0.83	0.00	5.19
Hairy woodpecker	*Picoides villosus*	2.44	0.71	0.83	0.00	5.19
Barn swallow	*Hirundo rustica*	-0.17	2.86	2.08	2.00	5.90
Red-winged blackbird	*Agelaius phoeniceus*	2.44	2.14	-2.08	1.00	5.90
Yellow-breasted chat	*Icteria virens*	-0.17	4.30	0.83	1.00	5.90
Western meadowlark	*Sturnella neglecta*	1.94	2.14	-0.83	1.00	5.96
Bank swallow	*Riparia riparia*	0.94	2.86	0.83	1.00	6.03
Black-headed grosbeak	*Pheucticus melanocephalus*	1.94	2.14	0.83	0.00	6.03
Lucy's warbler	*Vermivora luciae*	1.44	2.86	0.83	0.00	6.03
Bullock's oriole	*Icterus bullockii*	1.94	2.86	0.83	0.00	6.79
Cooper's hawk	*Accipiter cooperii*	1.94	2.14	0.83	1.00	6.79
Northern flicker	*Calaptes auratus*	1.94	2.14	0.83	1.00	6.79
Western screech-owl	*Megascops kennicottii*	1.94	2.86	0.83	0.00	6.79
White-breasted nuthatch	*Sitta carolinensis*	2.44	2.14	-0.42	1.00	6.79

Table 3.1. *Continued.*

Common Name	**Species**	**Habitat**	**Physiology**	**Phenology**	**Interactions**	**Overall**
Downy woodpecker	*Picoides pubescens*	2.94	2.14	0.83	0.00	7.56
Summer tanager	*Piranga rubra*	1.44	4.29	0.83	0.00	7.56
Western wood-pewee	*Contopus sordidulus*	2.44	2.86	0.83	0.00	7.56
Western yellow-billed cuckoo	*Coccyzus americanus occidentalis*	1.44	3.57	3.75	-1.00	7.56
Cliff swallow	*Petrochelidon pyrrhonota*	0.89	2.86	3.75	2.00	8.33
Common yellowthroat	*Geothylypis trichas*	1.44	2.86	3.75	1.00	8.40
Southwestern willow flycatcher	*Empidonax traillii extimus*	2.94	2.86	3.75	2.00	11.47
Mammals		0.84	0.63	-0.18	0.22	2.14
Crawford's or desert shrew	*Notiosorex crawfordi*	-1.72	1.67	-3.33	0.00	-2.89
Jackrabbit	*Lepus californicus*	-2.78	1.67	-1.67	1.00	-2.82
Big brown bat	*Eptesicus fuscus*	-0.56	-0.83	2.08	0.00	-0.28
Rock squirrel	*Spermophilus variegatus*	0.50	0.83	-3.33	0.00	-0.21
Deer mouse	*Peromyscus maniculatus*	0.00	0.00	0.00	0.00	0.00
Fringed bat	*Myotis thysanodes*	-0.06	-0.83	2.08	0.00	0.52
Raccoon	*Procyon lotor*	0.44	-0.83	0.83	0.00	0.52
Ord's kangaroo rat	*Dipodomys ordii*	0.50	0.00	-0.83	0.00	0.52
Silky pocket mouse	*Perognathus flavus*	0.50	0.83	-2.08	0.00	0.59
Botta's pocket gopher	*Thomomys bottae*	0.50	0.83	-2.08	0.00	0.59
Bobcat	*Lynx (Felis) rufus*	-0.56	1.67	-1.67	1.00	0.59
Coyote	*Canis latrans*	-0.56	0.83	1.25	0.00	0.66
Occult bat	*Myotis occultus*	0.44	-0.83	2.08	0.00	1.32
Stripped skunk	*Mephitis mephitis*	1.44	-0.83	-1.67	1.00	1.32
Pallid bat	*Antrozous pallidus*	0.44	-0.83	2.50	0.00	1.39
White-footed mouse	*Peromyscus leucopus*	1.50	0.83	-3.33	0.00	1.39
Long-legged bat	*Myotis volans*	0.94	-0.83	2.08	0.00	2.12
Desert cottontail	*Sylvilagus audubonii*	0.44	3.33	-3.33	0.00	2.19
Elk	*Cervus canadensis*	0.44	0.83	0.83	0.00	2.19
Porcupine	*Erethizon dorsatum*	0.44	0.83	1.25	0.00	2.26
Long-tailed weasel	*Mustela frenata*	-0.06	1.67	1.25	0.00	2.26
Yuma bat	*Myotis yumanensis*	1.94	-1.67	2.08	0.00	2.92
Mountain lion	*Puma concolor*	1.44	0.83	-1.67	1.00	2.99
Western harvest mouse	*Reithrodontomys megalotis*	2.50	0.83	-3.33	0.00	2.99
Tawny-bellied cotton rat	*Sigmodon fulviventer*	1.44	2.50	-3.33	0.00	2.99
Muskrat	*Ondatra zibethicus*	1.44	1.67	-1.67	0.00	2.99
Hispid cotton rat	*Sigmodon hispidus*	1.50	2.50	-3.33	0.00	3.06
Mexican free-tailed bat	*Tadarida brasiliensis*	0.94	-0.83	5.00	0.00	3.79
Piñon mouse	*Peromyscus truei*	2.50	1.67	-3.33	0.00	3.79
White-throated woodrat	*Neotoma albigula*	2.00	0.83	-1.67	1.00	3.86

Table 3.1. *Continued.*

Common Name	Species	Habitat	Physiology	Phenology	Interactions	Overall
Mule deer	*Odocoileus hemionus*	0.44	2.50	1.25	0.00	3.86
Grey fox	*Urocyon cinereoargenteus*	1.44	0.83	1.25	0.00	3.86
Mexican woodrat	*Neotoma mexicana*	2.50	1.67	-1.67	0.00	4.66
Beaver	*Castor canadensis*	2.94	-0.83	1.25	1.00	5.39
Hoary bat	*Lasiurus cinereus*	1.44	0.00	5.00	0.00	5.39
Black bear	*Ursus americanus*	1.94	0.00	3.75	1.00	6.3
New Mexico meadow jumping mouse	*Zapus hudsonius luteus*	2.50	1.67	0.83	1.00	7.13

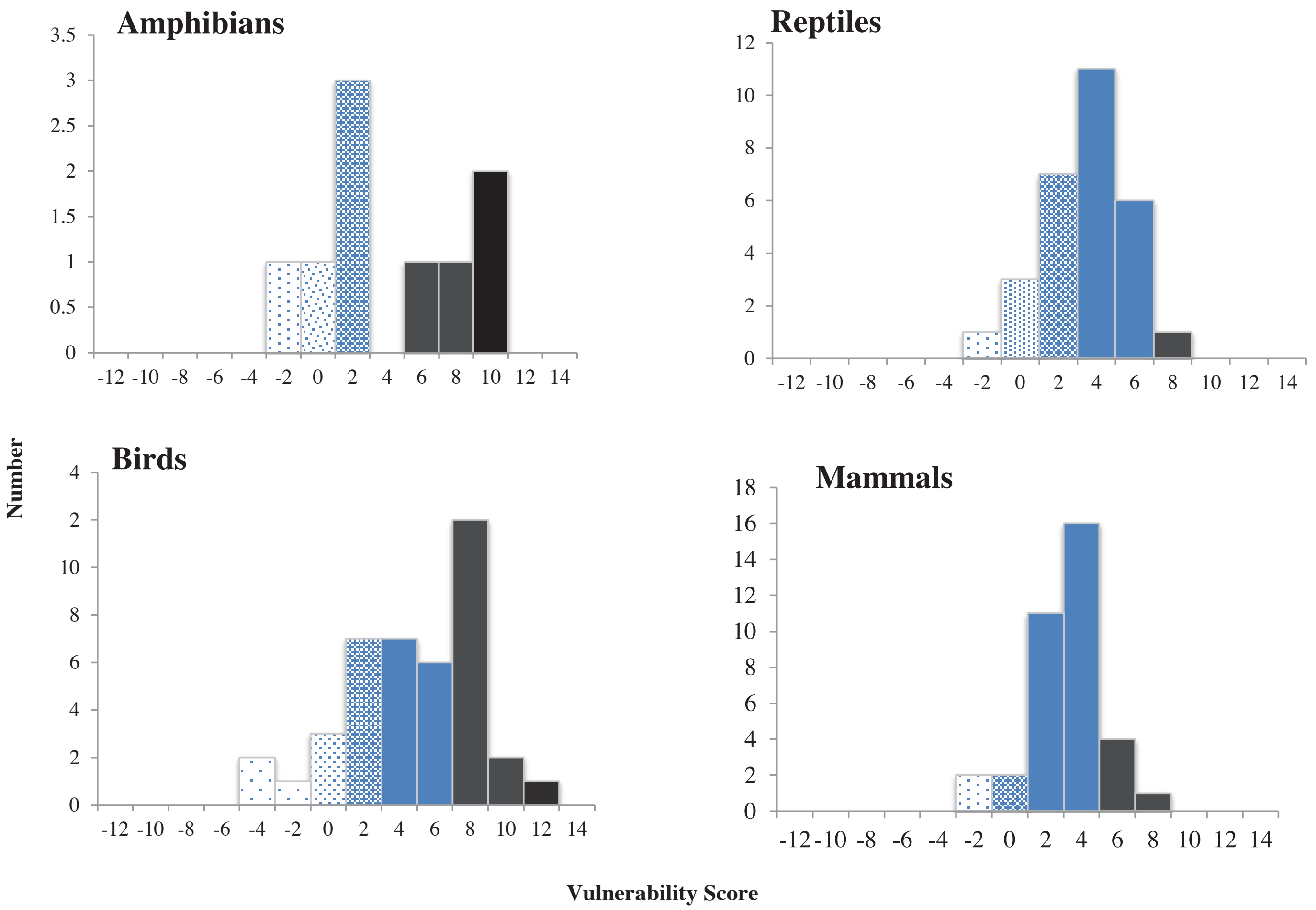


Figure 3.1. Histograms showing trends in scoring (overall vulnerability to climate change) across taxonomic groups. Total potential maximum and minimum scores are -20 and +20, respectively, though realized maximum and minimum for this assessment were -4.2 and +11.47. For the purpose of interpretation, species were divided into groups that represented high vulnerability or resilience (<-2 or >+2, respectively, black), moderate vulnerability or resilience (<-1 or >1, respectively, grey), and neutral response (>-1 but <1, white). Note: Y-axis differs among individual histograms.

Amphibian Overview

We assessed two frogs, northern leopard frog (*Lithobates* [*Rana*] *pipiens*) and western chorus frog (*Pseudacris triseriata*); three spadefoot toads, Couch's spadefoot (*Scaphiopus couchii*), New Mexico spadefoot (*Spea multiplicata*), and plains spadefoot (*S. bonifrons*); two toads, Great Plains toad (*Anaxyrus cognatus*) and Woodhouse's toad (*A. woodhousii*); on salamander, the barred tiger salamander (*Ambystoma mavortium*); and one introduced species, the American bullfrog (*Lithobates* [*Rana*] *catesbeiana*) (Table 3.1; Figs. 3.1, 3.2; Appendix 3). Of the nine species of amphibians assessed, we considered four (44%) vulnerable to climate change. The amphibian with the highest overall score was the northern leopard frog (9.9). Four species had neutral scores and only one, the American bullfrog (-2.0), appeared to be somewhat resilient to climate change effects. We predicted species with water-dependent larval or adult life stages to be the most vulnerable to future climate projections.

We used several specific predictions and assumptions about future conditions under climate change for assessing amphibian species (Box 2). We found variables related to habitat loss to be a major issue for many amphibian species. Each of the four most vulnerable species (western chorus frog, barred tiger salamander, northern leopard frog, and Woodhouse toad) had high habitat scores (Fig. 3.2). The three spadefoot toad species had slightly negative scores for habitat components as did the Great Plains toad. For many species, water availability, including the persistence time of ephemeral ponds and water quality, is projected to decrease (Box 2). The relatively low (or neutral) scores obtained for the three species of spadefoot toad reflects our expectation for increasing scrub and grassland habitats in the MRGB. Though we expect the habitat of the Great Plains toad to also increase, its stronger dependence on riparian habitat components led to its expected neutral response. Our assessment projected that American bullfrog would be the most tolerant of future trends in climate. Although the habitat for this species is expected to

Box 2. Summary of climate projections and predictions used as the basis for scoring species assessments inhabiting the Middle Rio Grande. See Chapter 2 for references.

- Less open water, reduced flow.
- More early springtime floods.
- Reduced riparian habitat due to decreased stream flows and longer droughts.
- Cottonwood gallery forests likely to decline due to lower water flows and increasing wildfires, which reduce recruitment and increase mortality.
- Loss of mature trees due to fire and disease.
- Loss/reduction of native vegetation and replacement by invasive species due to increased fire and lower water tables.
- Increase in *Tamarix* spp. and, potentially, Russian olive trees.
- Replacement of grassland and woodland habitats with scrubland.
- Reduced vegetation cover due to increased aridity.
- Potential increase in exotic grass species (might increase vegetation cover).
- Increased fire intensity and frequency. Prolonged fire season.
- Increased duration of drought that last five years or more.
- More intense droughts (drier conditions).

decrease, the American bullfrog is quite tolerant of warm water (in contrast to the native frog species) and is able to survive within a variety of habitats and consume a variety of food resources. Thus, though we found the bullfrog vulnerable to some future changes, it possessed enough resilient traits to drive the overall score toward resilience. None of the nine amphibians migrate, and dispersal was considered a limitation for most species.

Physiological sensitivity to desiccation at multiple life stages and temperature-related activity limitations appeared to be important in the majority of the amphibians, and six of nine species had scores indicating vulnerability in the physiology category (Table 3.1; Fig. 3.2). We considered all amphibians to be resilient from having an ectothermic metabolism, which may allow them, through lower energy requirements, to survive low-resource conditions better than endotherms (Bagne and others 2011). We also considered the ability of spadefoot toads and barred tiger salamanders to aestivate to avoid desiccation a resiliency characteristic.

We found the greatest variation among amphibian scores in the phenology category. Some species, such as the bullfrog, showed a high degree of resilience to timing changes, and others, such as the spadefoots, appeared quite vulnerable to expected changes in the phenology of water sources. Phenology was the most important vulnerability indicator for spadefoot toads whose breeding is tightly tied to rainfall patterns and the availability of ponds, the latter of which is positively correlated to mortality of spadefoot tadpoles (Degenhardt and others 1996).

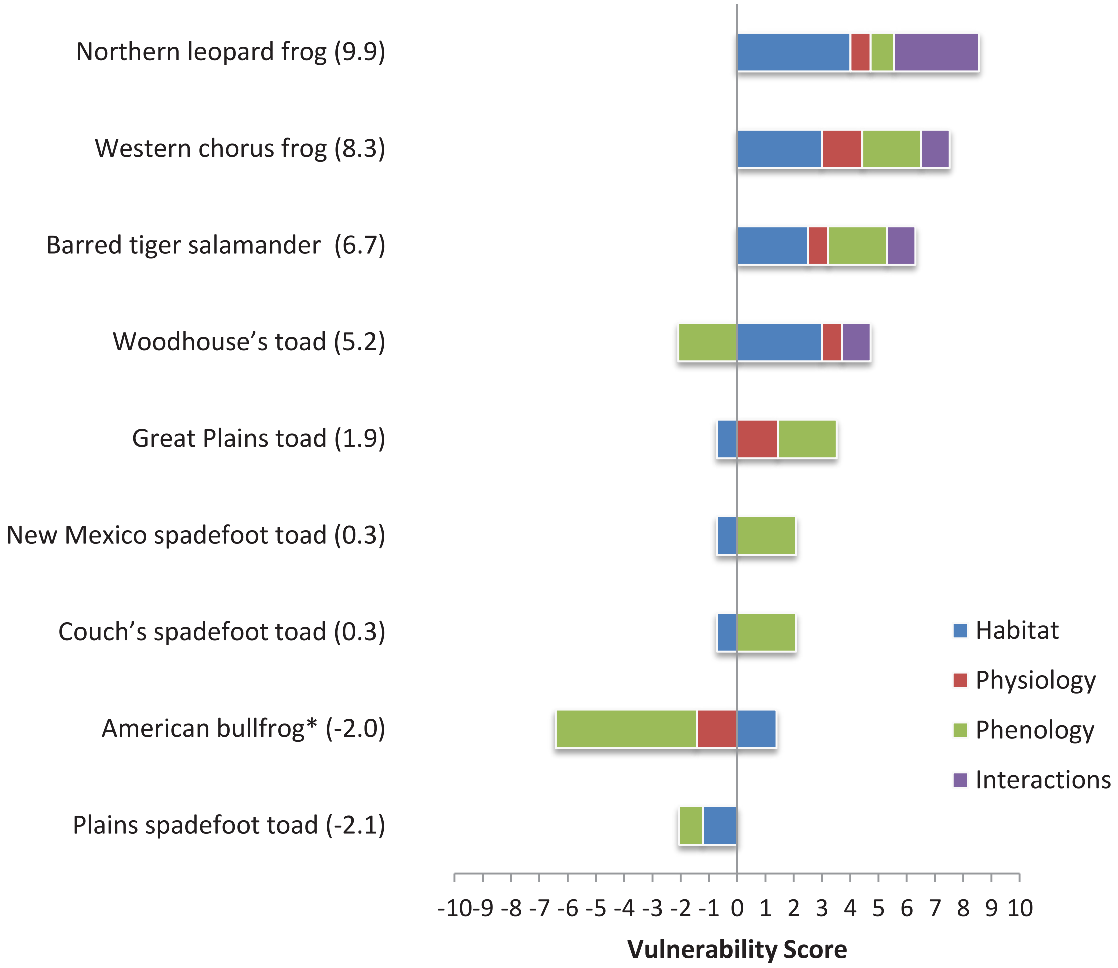


Figure 3.2. Climate change vulnerability scores for amphibian species inhabiting the MRG, New Mexico. Overall vulnerability scores are in parentheses. Asterisk indicates introduced species. See text for details.

Spadefoot toads require breeding ponds for two to four months and often rely on ponds formed by heavy warm rains of the monsoon season (BISON-M 2009; Degenhardt and others 1996). These types of ephemeral aquatic habitats are probably at greatest risk of change due to changes in evaporation rates and storm timing and intensity. With the exception of the American bullfrog, we considered all amphibians to be at an increased risk of timing mismatches because they reproduce only once per year.

The northern leopard frog had the largest score in the biotic interactions category. Three other species had lower scores though still were scored as being negatively impacted (Fig. 3.2). Competition and predation by the American bullfrog, especially pertinent given potential resilience of bullfrogs to climate change, and decline in amphibian prey were the most common reasons for high vulnerability scores for all species. The American bullfrog is also resistant to the chytrid fungus, which we expected to negatively impact the other anuran species, such as the northern leopard frog and barred tiger salamander.

Reptile Overview

We assessed 29 reptiles—4 are mostly aquatic (turtles), 3 are semi-aquatic (primarily garter snakes), and 22 are mostly, or strictly, terrestrial species (Table 3.1; Figs. 3.3; Appendix 3). Species that were strongly associated with the riparian area or water bodies of the MRGB, as opposed to xeric-adapted species, tended to have the highest scores (indicating vulnerability). We found 18 of 29 (62%) reptiles to be vulnerable to expected climate change in the MRG Valley. Scores ranged from 6.9 for the most vulnerable species, the Great Plains skink (*Eumeces obsoletus*), to -2.7 for the least vulnerable species, the desert grassland whiptail (*Aspidoscelis uniparens*) (Table 3.1). Ten reptile species received neutral scores (-2 to 2), and two species received a score greater than 2, indicating vulnerability. Habitat and phenology categories were the most consistent determinants of vulnerability (Table 3.1, Fig. 3.3). Within these categories, species' scores indicated high vulnerability or were slightly negative, indicating neutral or resilient responses. All of the most vulnerable species had high scores in the habitat category.

Specific areas of concern for future reptile vulnerability include risks associated with changing fire regimes and loss of habitat. We did not expect suitable habitat area to increase for any of the reptiles considered. We generally expected reptiles that depend on mesic environments in the MRGB to be more likely to suffer population declines, whereas adjacent terrestrial species are less likely to be negatively affected by projected climate conditions. We found the aquatic turtles and the skink, which require very moist environments and generally do not disperse great distances away from moist sites, most vulnerable to habitat issues (Fig. 3.3). As with the amphibians, loss of perennial marsh and cienega habitats is likely to have negative consequences for reptiles and, in particular, for populations of western painted turtles and New Mexico gartersnakes. We considered the availability of other habitat components, such as sand bars used for nesting, dense vegetation, and still water bodies, to be important determinants of vulnerability for many reptiles. We also considered soil moisture requirements for nesting or hibernating to be critical habitat components susceptible to change. For instance, the snapping turtle may lose potential hibernation sites as river beds dry. Also, hibernation sites may be a limiting factor for the western painted turtle, which is sensitive to anoxic conditions (BISON-M 2009).

Vulnerability to habitat loss is not solely related to the direct loss of mesic conditions. The ornate box turtle (*Terrapene ornate*) is strongly associated with the

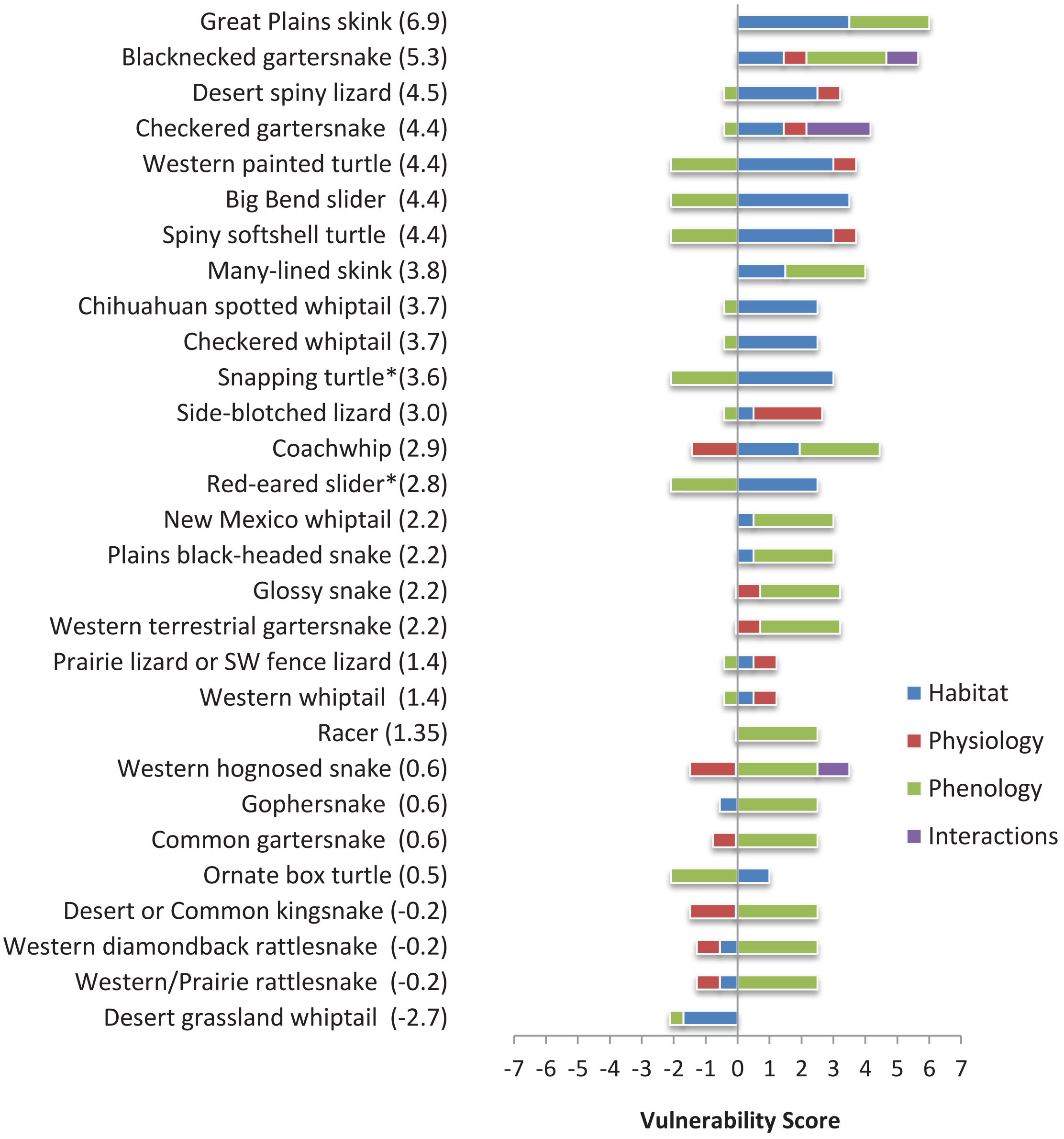


Figure 3.3. Climate change vulnerability scores for reptile species inhabiting the MRG, New Mexico. Overall vulnerability scores are in parentheses. Asterisk indicates introduced species. See text for details.

open grasslands, a habitat threatened by invasive species, and changes to fire and precipitation regimes (CRACM 2006). Similarly, increased invasion by tamarisk has been directly associated with habitat degradation for the Big Bend slider (*Trachemys geigeae*) and contributed to its vulnerability score (Stuart and Ward 2009).

Physiology scores tended to be lower for reptiles than for other taxon, and we predicted most reptiles will have an overall neutral response to physiological climate effects. Ectotherms are generally considered less sensitive to resource variation than other taxa because of reduced energy requirements, but observed interaction between temperature and metabolic effects are more complicated. Ectotherms could be vulnerable to negative consequences of higher temperatures on metabolism. Snake metabolism increases with increasing temperature such that the ability of a snake to survive starvation is highly dependent on temperature (Leuth 1941). For instance, snakes kept at low temperatures (15 °C) lived an average of 104 days without food, whereas snakes kept at 36 °C lived an average of

10.6 days. In general, we scored snakes as more vulnerable than lizards to physiological issues, most commonly high temperatures. Exceptions included snakes that are xeric adapted (e.g., rattlesnakes). Warmer conditions may extend breeding seasons for some reptile species though the ultimate impact on fecundity depends on changes in precipitation events (see Appendix 1 or Bagne and others [2011] for further discussion). Die-offs of several lizard species in Costa Rica have also been attributed to increased temperatures (Sinervo and others 2010). We considered semi-aquatic species to be somewhat buffered from temperature extremes due to their use of water. Similarly, nocturnal or crepuscular species were not expected to be overly influenced by increasing temperatures. Though reptiles are able to hibernate, this was not considered to contribute to the resilience of any species under warming conditions. Typically, hibernation sites were underground, and we did not consider them well buffered from the environment (see Bagne and others 2011). In addition, we did not consider obligatory hibernation or aestivation as a source of resilience with respect to traits that may confer metabolic advantage under warming conditions, particularly if these events are initiated without adequate fat reserves. For many hibernating species and, in particular, snakes, reproductive success is strongly tied to available fat reserves following hibernation. Instead, we considered aestivation a resilient trait for surviving extreme weather conditions and considered hibernation in response to cold a neutral trait. We considered all of the freshwater turtles assessed to be at increased risk to population declines because they exhibit temperature-dependent sex ratios (Gibbons and others 2000). Though long generation times in some species, such as rattlesnakes and turtles, makes them more vulnerable to extinction from habitat loss, hunting, and other factors that affect populations (Gibbons and others 2000), we considered species with lifespans greater than five years to be resilient to potentially negative impacts from drought.

Phenology scores were highly variable among reptiles, with some species showing a high potential for negative impacts and others predicted to show little negative effect. All of the reptile species rely on temperature cues to initiate hibernation. We found a number of potential timing mismatches between critical activities, such as breeding or hibernation, and favorable conditions. We used several proxies to indicate species reliance on resource pulses: high degree of variation in clutch size, reduced number of reproductive events in a year due to changing conditions, and narrow reproductive windows (e.g., a few weeks of the active season). Both movement of gartersnakes and availability of prey species, in particular amphibians, are strongly tied to precipitation (Seigel and Ford 2001) and indicate potential areas of vulnerability. Among species possessing resilient traits, turtles, such as the ornate box turtle, have a flexible capacity to control the timing of reproductive events through sperm storage and controlled fertilizations (see species accounts presented in BISON-M and the Animal Diversity web site). Similarly, many lizard species, including the desert spiney lizard, prairie lizard, side-blotched lizard (Pianka and Parker 1975; in southern climates), checkered whiptail, New Mexico whiptail, and western whiptail (Pianka 1970), produce multiple clutches per year, a trait that minimizes the risk of reproductive failure within a single year under increasingly variable conditions. Chihuahuan spotted whiptails were also assumed to have multiple clutches given data on other species within this genus. Conversely, many snakes only reproduce biennially, which increases the likelihood that limiting conditions or time events will result in reproductive failure and lead to population declines over time.

We did not find identifiable interaction effects for many reptile species. We found the strongest effects within the biotic interactions category for reptile species that preyed upon amphibians, specifically anurans. We expect three species will experience food declines as a result of climate change: western hog-nosed snake, black-necked garter snake, and checkered garter snake (Seigel and Ford 2001). Reliance on invertebrate populations also will undoubtedly affect many reptile species in the future, but little specific information is available to identify trends within populations of predators and prey.

Bird Overview

Of the 42 species of birds assessed, 29 (69%) species had scores reflecting an overall vulnerability to climate change (Table 3.1; Fig. 3.4; Appendix 3). Scores ranged from 11.5 for the southwestern willow flycatcher (*Empidonax trailii extimu*) to -4.42 for the greater roadrunner (*Geococcyx californianus*). The three most resilient species—greater roadrunner, house finch (*Carpodacus mexicanus*) and brown-headed cowbird (*Molothrus ater*)—are habitat generalists with a good

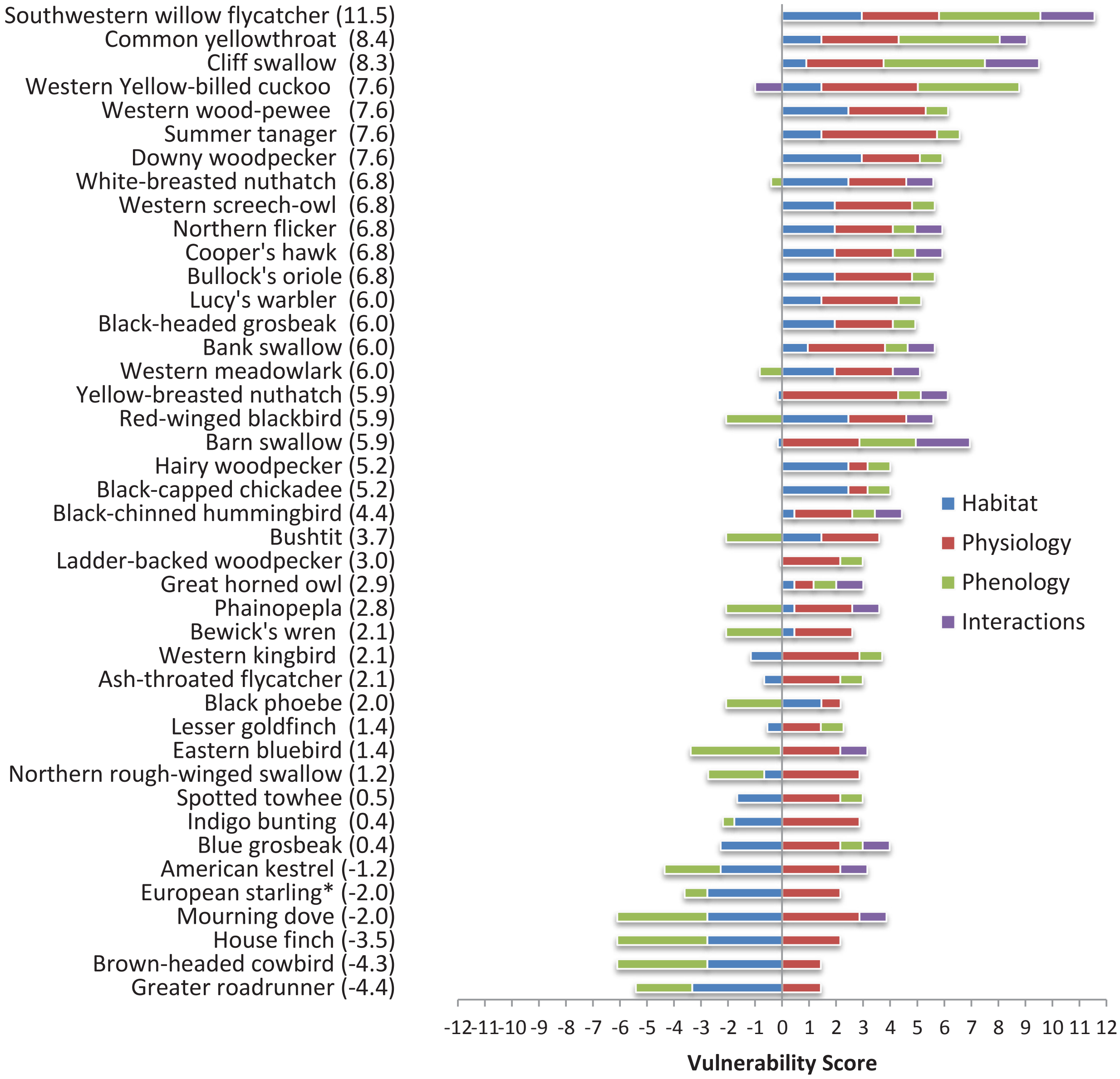


Figure 3.4. Climate change vulnerability scores for bird species inhabiting the MRG, New Mexico. Asterisk indicates introduced species. Overall vulnerability scores are in parentheses. See text for details.

capacity to respond to resource variation. The five species with the highest vulnerability—southwestern willow flycatcher, common yellowthroat, cliff swallow, western yellow-billed cuckoo, and western wood-pewee—were vulnerable in all four categories (Fig. 3.4). The four species of woodpeckers had overall scores indicating vulnerability. Of the four raptor species, western screech owl (*Megascops kennicottii*) was the most vulnerable followed by Cooper's hawk (*Accipiter cooperii*) and great horned owl (*Bubo virginianus*). By contrast, the American kestrel (*Falco sparverius*) had a negative score.

We found about half of the bird species (23 species) vulnerable to changes in their habitat. The southwestern willow flycatcher (Federally listed as endangered), the western yellow-billed cuckoo (a candidate for Federal listing), and the common yellowthroat depend on riparian habitat and were among the most vulnerable to potential population declines under climate projections. We also considered species that rely on cavities, particularly those in trees, vulnerable partly due to climate-related changes in cavity availability. These species included ash-throated flycatcher (Brush and others 1990; Cardiff and Dittman 2000), chickadees (Smith 1993), Bewick's wren (Kennedy and White 1997), nuthatches (Grubb and Pravosudov 2008), and Lucy's warbler (Johnson and others 1999). Long-term survival of large trees, particularly cottonwoods in the MRGB, may be threatened by reductions in water as well as increases in more drought-tolerant invasive tree species that are not suitable for cavity excavation (Kranjcec and others 1998). We scored three woodpeckers—the downy woodpecker, hairy woodpecker, and northern flicker—vulnerable to habitat loss because they are dependent to some degree on mature forest habitat (Jackson and others 2002; Jackson and Ouellet 2002; Wiebe and Moore 2008). The ladder-backed woodpecker was the only woodpecker species we thought would be likely to experience little or no change in habitat area, primarily because it successfully utilizes shrubby habitats (Lowther 2001; Winternitz 1998). All birds received resilience scores for their high capacity to disperse and ability to colonize new areas. Many migrants showed higher vulnerability scores than non-migrants (Table 3.2). However, four long-distant migrant species showed low vulnerability scores—blue grosbeak, western kingbird, indigo bunting, and eastern bluebird.

Scores in the physiology category were very similar for all birds. All but one species, the black phoebe, were vulnerable to future warming and drying trends. We considered the majority of birds vulnerable to limiting physiological conditions, particularly drought. Specifically, we considered the southwestern willow flycatcher, the western yellow-billed cuckoo, and the common yellowthroat sensitive to high temperatures. We also considered nestlings of many species that utilize non-forested sites susceptible to heat and extended heat waves that could cause certain nest materials to dry out and crumble or become too hot inside for eggs or nestlings to survive (e.g., sparrow species, Baltosser and Russell 2000; Brown and Brown 1999; Cabe 1995). We considered a reduction in daily activity periods due to warming temperatures likely to occur for only a few species. However, we did expect extreme events, such as storms and late frosts, to contribute to species vulnerability, particularly those migrating along the coast or with demonstrated mortalities. Swallows are especially vulnerable to periods of bad weather because it limits their foraging time, causing them to starve to death or become weakened to the point where they are easily caught by predators (Brown and Brown 1999b; Garrison 1999). Sudden cold spells in the spring have killed hundreds of thousands of barn swallow adults (Baltosser and Russell 2000; Brown and Brown 1999). Very cold, wet, or hot weather may kill European starling nestlings (Cabe 1995).

Table 3.2. Passerines of the Middle Rio Grande assessed for vulnerability to climate change. Species which migrate tended to be more vulnerable. Overall score is presented in parentheses. Shading indicates migrant. Introduced species indicated with "I".

Common Name	Scientific Name	Migrates?	Habitat score	Phenology score
Brown-headed cowbird (-4.3)	*Molothrus ater*	No	-2.78	-3.33
House finch (-3.5)	*Carpodacus mexicanus*	No	-2.78	-3.33
European starling, I (-2.0)	*Sturnus vulgaris*	No	-2.78	-0.83
Mourning dove (-2.0)	*Zenaida macroura*	No	-2.78	-3.33
Blue grosbeak (0.4)	*Passerina caerulea*	Yes	-2.28	0.83
Indigo bunting (0.4)	*Passerina cyanea*	Yes	-1.78	-0.42
Spotted towhee (0.5)	*Pipilo maculatus*	No	-1.67	0.83
Northern rough-winged swallow (1.2)	*Stelgidopteryx serripennis*	Yes	-0.67	-2.08
Eastern bluebird (1.4)	*Sialia sialis*	No	-0.06	-3.33
Lesser goldfinch (1.4)	*Carduelis psaltria*	No	-0.56	0.83
Black phoebe (2.0)	*Sayornis nigricans*	No	1.44	-2.08
Ash-throated Flycatcher (2.1)	*Myiarchus cinerascens*	Yes	-0.67	0.83
Bewick's wren (2.1)	*Thryomanes bewickii*	No	0.44	-2.08
Western kingbird (2.1)	*Tyrannus verticalis*	Yes	-1.17	0.83
Phainopepla (2.8)	*Phainopepla nitens*	No/Yes	0.44	-2.08
Bushtit (3.7)	*Psaltriparus minimus*	No	1.44	-2.08
Black-capped chickadee (5.2)	*Poecile atricapillus*	No	2.44	0.83
Western meadowlark (5.6)	*Sturnella neglecta*	No	1.94	-0.83
Barn swallow (5.9)	*Hirundo rustica*	Yes	-0.17	2.08
Red-winged blackbird (5.9)	*Agelaius phoeniceus*	No/sometimes	2.44	-2.08
Yellow-breasted chat (5.9)	*Icteria virens*	Yes	-0.17	4.33
Bank swallow (6.0)	*Riparia riparia*	Yes	0.94	0.83
Black-headed grosbeak (6.0)	*Pheucticus melanocephalus*	Yes	1.94	0.83
Lucy's warbler (6.0)	*Vermivora luciae*	Yes	1.44	0.83
Bullock's oriole (6.8)	*Icterus bullockii*	Yes	1.94	0.83
White-breasted nuthatch (6.8)	*Sitta carolinensis*	No/Irruptive?	2.44	-0.42
Summer tanager (7.6)	*Piranga rubra*	Yes	1.44	0.83
Western wood-pewee (7.6)	*Contopus sordidulus*	Yes	2.44	0.83
Cliff swallow (8.3)	*Petrochelidon pyrrhonota*	Yes	0.89	3.75
Common yellowthroat (8.4)	*Geothylypis trichas*	Yes	1.44	3.75
Southwestern willow flycatcher (11.5)	*Empidonax traillii extimus*	Yes	2.94	3.75

Red-winged blackbirds may be at risk of mortality due to flooding because they nest over water.

We found 19 species resilient to potential changes in the timing of resource availability. Resilient species were typically food generalists and able to produce multiple clutches per year. The southwestern willow flycatcher, western yellow-billed cuckoo, and common yellowthroat rely on climate driven cues and/or resource pulses, which are likely to change under future scenarios. We considered all migrants at risk of a timing mismatch between initiation of migration and availability of critical resources at the destination site. These species include: ash-throated flycatcher, bank swallow, barn swallow, Bewick's wrens, black-headed grosbeak, blue grosbeak, Bullock's oriole, bushtits, cliff swallows, common yellowthroat, summer tanager, western kingbird, western wood-Pewee's, southwestern willow flycatcher, and yellow-bellied chat (Bemis and Rising 1999; Brown and Brown 1996; Hill 1995; Garrison 1999; Guzy and Ritchison 1999; Rising and Williams 1999; Robinson 1996; Rosenberg and others 1982; Sedgwick 2000; Svensson 1986; Thompson and Nolan 1973; Turner and Rose 1989; Wingfield 2008; Table 3.2). We considered resident birds generally unaffected by timing mismatch issues. However, we found the effect of climate on the availability of important food resources an important factor in determining vulnerability for many bird species. We considered most insectivores, whether migrating or not, vulnerable to distinctly timed resources. Food abundance in spring is an important factor driving fecundity in bird species breeding in temperate regions (review by Martin 1987). Typically, increases in insect abundance (a critical resource for these species) occur in spring, and changes in the timing of precipitation and increases in temperature have the potential to alter the timing of insect availability and abundance and, hence, fecundity. We found numerous examples of potential effects among our assessed species. For the blue grosbeak, timing of breeding is associated with monsoons (Rosenberg and others 1982), which is thought to relate to important insect populations for feeding young. Earlier nests in California populations of Bullock's orioles did better than later nests (Rising and Williams 1999), and breeding season was lengthened during wetter summers with more insects (Rising and Williams 1999). For the western kingbird, clutch sizes were larger in a year when insect abundances were higher in Kansas (Murphy 1988 in Gamble and Bergin 1996), consistent with another study in Arizona (Blancher and Robertson 1987 in Gamble and Bergin 1996). The southwestern willow flycatcher has a short nesting season that is thought to be limited by resource availability (Sedgwick 2000). Emergence of periodical cicadas can increase reproductive success in red-winged blackbirds (Strehl and White 1986) and cicada emergence is related to soil temperatures (Smith and others 2006; Williams and Simon 1995), leading to the likelihood of future timing issues. We also considered the black-capped chickadee, a seed eating species that also relies on insects during breeding season (Smith 1993), vulnerable to changes in the timing of food resources. In contrast, we did not score another seed eating species, the lesser goldfinch, as vulnerable because data were sparse. The white-breasted nuthatch, which is reported to consume both seeds and insects in spring, received a neutral score because, although there are potential changes in the timing of insect availability and abundance, consumption of seeds from the previous season potentially makes this species less dependent on insect flush in spring. In California, egg laying of phainopeplas coincided with fruiting for summer populations (Walsberg 1977) and, although this species will also eat insects, we considered it likely to be negatively impacted by changes to the timing of fruiting periods.

We expected changes in biotic interactions to negatively impact 15 species but benefit 7 species. Most of the aerial foraging bird species obtained positive overall scores: cliff swallow, barn swallow, ash-throated flycatcher, northern rough-winged swallow, and bank swallow. An exception is the eastern bluebird. We considered some insectivores, especially those that specialize on aquatic insects and have a demonstrated tendency to suffer from weather-associated starvation, at risk of food declines. For instance, bank swallows are aerial predators that feed almost exclusively (>99% of diet) on flying or jumping insects, most of which rely on aquatic developmental stages. Studies of European populations have found them to be vulnerable to starvation on their wintering grounds when low rainfall reduces their food supply (Szep 1995). Presumably, MRGB populations may also be at risk as precipitation is predicted to be reduced and periods of drought are expected to increase. We considered phainopeplas, which feed primarily on mistletoe berries, susceptible to food declines. Wintering populations of phainopeplas vary dramatically in size and location depending on weather and the availability of fruiting plants (Chu and Walsberg 1999). However, drought was found to make mistletoe-infested plants more susceptible to mortality in the Mojave but not in the Sonoran desert (Spurrier and Smith 2007), which is more similar to New Mexico habitats. Therefore, we did not expect the occurrence of mistletoe-infested plants to change in desert regions. However, phainopeplas feed extensively on New Mexico privet berries (RMRS data) during the summer in the MRG, which are vulnerable to severe freezes and possibly drought. Given predictions for increased droughts, mistimed frosts and observed impacts on phainopepla populations, we considered this species at risk of food declines due to climate change. Willow flycatchers are primarily insectivorous (Sedgwick 2000), with a fairly diverse food base. However, willow flycatchers are dependent upon localized food sources during nesting and, in particular, insects that rely on water for some phase of their lifecycle, and so were considered at risk of decline.

Mammal Overview

We assessed 37 mammals for their potential vulnerability to climate change in the MRG Valley (Table 3.1; Fig. 3.5; Appendix 3). Mammal scores ranged from a high vulnerability score of 7.1 for the New Mexican jumping mouse (*Zapus hudsonius luteus*) to a low score of -2.9 for the desert shrew (*Notiosorex crawfordi*). We found 51% (21 species) vulnerable to population declines associated with climate change, and 5% (2 species) had scores indicating resilience. In general, species that appeared most at risk were the following types: (1) those with a high reliance on riparian areas (e.g., New Mexican jumping mouse and beaver, *Castor canadensis*); (2) those that require dense vegetation or specific vegetation features (e.g., woodrats, *Neotoma* spp.); and (3) those that were at risk of a timing mismatch between critical resources and breeding (e.g., hoary bat, *Lasiurus cinereus*; black bear, *Ursus americanus*). Species that we scored as having a neutral response tended to be opportunistic breeders (e.g., jackrabbit, *Lepus californicus*; desert shrew) with a wide diversity of habitat associations, including habitats that are expected to increase in the future (Table 3.1).

We expect climate-induced habitat changes to have widely varying effects for mammal species. Of all the taxonomic groups, mammal species exhibited the greatest variation in body size, metabolism, range size, and life history traits. Bats are likely to be particularly hard hit by changing vegetation and water dynamics. Climate-related shifts in riparian habitat and potential loss of important foraging grounds (trees, shrubs, and open water) will have high energetic costs for maternal colonies of bat because nursing animals will have to travel greater distances between roosts and foraging grounds. For example, we expect shifts in foraging habitat to negatively impact the Mexican free-tailed bat, *Tadarida brasiliensis,* which, although widespread, is considered near threatened by the International

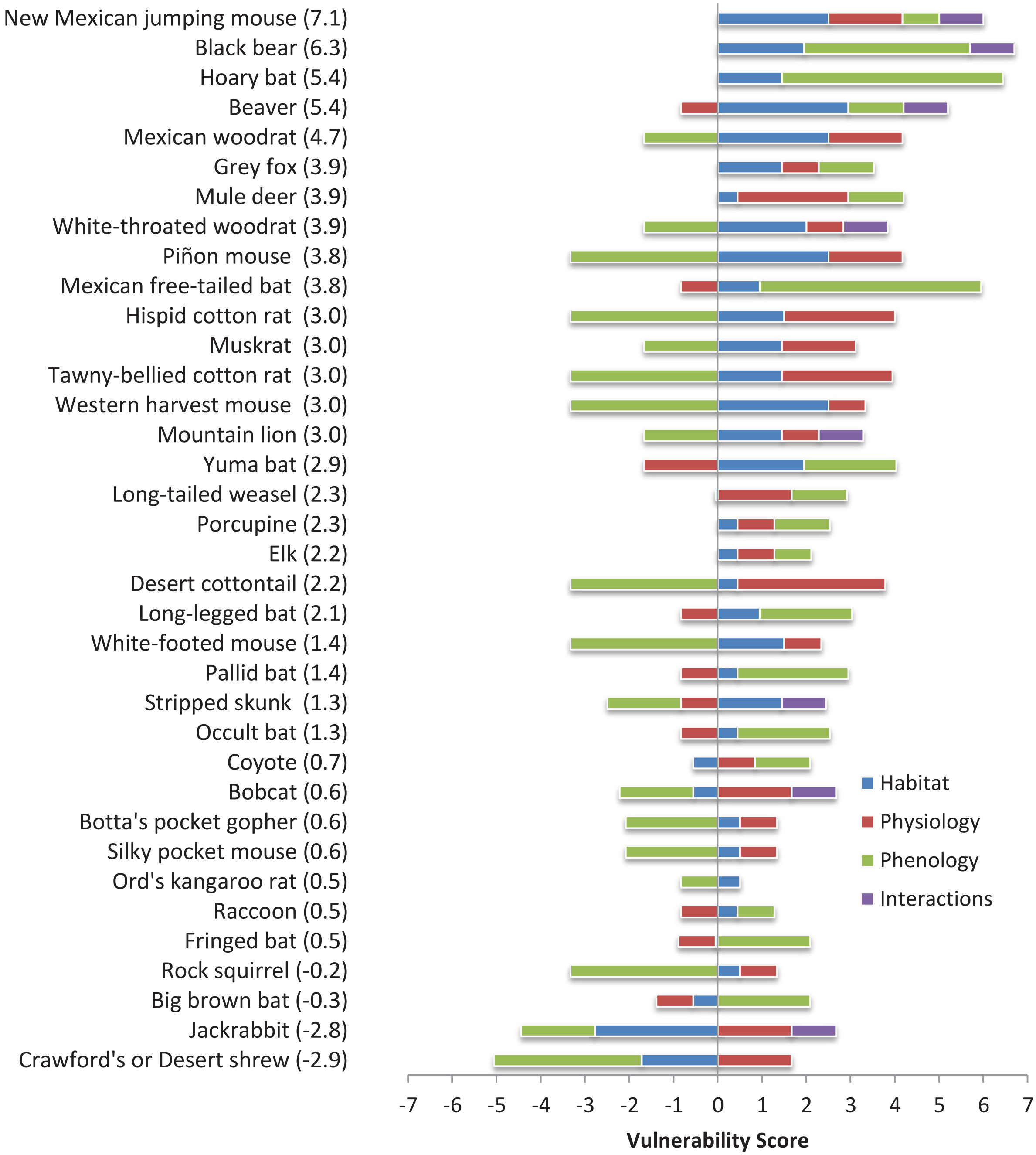


Figure 3.5. Climate change vulnerability scores for mammal species inhabiting the MRG, New Mexico. Asterisk indicates introduced species. See text for details.

Union for Conservation of Nature (Red List Category) and has designation of S3 (vulnerable) in New Mexico. Free-tailed bats form large maternity colonies in caves, a finite resource unlikely to shift in geographic space with important forage resources. Therefore, free-tailed bats are likely to experience some loss of linkages between forage and roosting sites, which will increase energy expenditures (Scheel and others 1996). We also considered wintering habitat at risk of decline for migrating bat species, specifically the Mexican free-tailed bat because it is known to be threatened by disturbance, pesticides, and habitat destruction in its tropical wintering grounds (NatureServe 2009). Climate change is likely to increase the rate of these losses.

Species that rely on the presence of surface water (e.g., beaver; Yuma bat, *Myotis yumanensis*) or vegetation features, such as dense sedges or grasses, received the highest vulnerability scores in the habitat category. Species associated with scrub habitat and open areas (e.g., black-tailed jackrabbit and desert shrew) received negative scores indicating resilience for habitat. Data on habitat quality were scarce for most mammals or, if available, dealt with aspects of habitat related to questions we addressed within other categories. For instance, quality often regarded food availability, which was addressed in the first question of the biotic interaction scoring. We only expect three species—beaver and two cotton rats (*Sigmodon* spp.)—to be negatively impacted by either changes to water quality or changes to vegetation characteristics associated with habitat quality. Beavers require somewhat consistent water sources and may not do well under dryer regimes, or under conditions that include damaging floods. Precipitation appears to be an important correlate of habitat quality for hispid cotton rats, particularly in marginal habitat (Texas Mammals online). Habitat use and preference of tawny-bellied cotton rats appears to be positively correlated with the density of monocot grass species, probably due to its role as a predator avoidance mechanism (Sullivan 1995). Among carnivores, we only found evidence for habitat declines for the gray fox (*Urocyon cinereoargenteus*). Though (like other carnivores) the gray fox has generalized habitat needs, we also determined that it specifically depends upon riparian habitat in its New Mexico range (Fritzell and Haroldson 1982). Gray fox, however, is essentially absent from grassland or open habitats that do not have rocky outcrops or some juniper encroachment (BISON-M 2009). Only three species—hoary bat, big brown bat, and long-legged bat—migrate, and we considered these species vulnerable to changes across multiple disparate habitats.

We found short life spans, susceptibility to extreme weather events, and sensitivity to changes in precipitation to be the most common traits exhibited by species with high scores in the physiology category (Appendix 1; Appendix 3). We did not find evidence that high temperatures alone were a limiting condition for most mammals, but we did find that water availability was a potential limiter for many populations. Several bat species have a strong reliance on open water for metabolic needs, lactation in females, and as a foraging habitat and were more sensitive to physiological effects of climate change than many other species (Adams and Hayes 2008; Carpenter 1969; Wilkins 1989). We were most likely to score bats, particularly those that are foliage roosting, as vulnerable to future storm and extreme weather events. Storms were listed as a significant source of mortality for big brown bats (Kurta and Baker 1990) and are likely to negatively impact foliage roosting species such as the Hoary bat (Willis and Brigham 2005). Conversely, we also found several traits among bat species that contribute to resiliency: bats use torpor to reduce evaporative loss (Carpenter 1969), bats are long lived (Hermanson and O'Shea 1983; Mulheisen and Berry 2000; Rasheed and others 1995; Wilkins 1989), and females can reabsorb or abort embryos when reproduction is likely to fail (Anthony and others 1981; Chung-MacCoubrey 2005).

The highest scores in the phenology category were for three species of bats due to their reliance on cues for both hibernation and migration. Hibernation was the most common activity associated with temperature cues for mammals. As with birds, long-distance migration led to greater vulnerability scores due to the risks associated with timing asynchrony between environmental cues and critical resources (see Bagne and others 2011), though this only applied to the hoary bat and Mexican free-tailed bat. We generally assumed species relied on timed resource peaks when they utilized few sources, for example, bears rely on masting events, and bat species specialize on insect prey (see Chapter 4, Table 4.2; Appendix 1).

Though shifts in temperature regimes will result in a change in the timing of emergence and reproduction in insect prey bases, we recognize that this shift does not necessarily create a net loss of prey but rather a change in the type of species available during a certain period. Still, hoary bat migration through New Mexico is hypothesized to be timed to cutworm emergence, a factor that we felt indicated potentially increased vulnerability (Valdez and Cryan 2009). In general, we considered species that were likely to adjust migration timing in response to changes in resources to be more vulnerable to climate effects because such shifts carry greater risk of mistimed arrival, exposure to inclement weather, and asynchrony between changes in winter and summer habitats. For bats, it is believed that they migrate later in the season if warmer temperatures lead to an increase in night-flying insects during colder months (Geluso 2007). We considered species with no reliance on temperature or moisture cues and/or short generation times to be resilient to potential phenological issues. For example, the *Sigmodon* spp. do not rely on temperature or moisture cues, do not hibernate, and are able to produce multiple litters per year.

Biotic interactions generally did not have a large impact on scores for mammal species. One exception was coyotes (*Canis latrans*), black-tailed jackrabbits, and bobcats (*Lynx rufus*). It is generally recognized that the abundance and reproductive success of many predators, including bobcats, grey fox, and coyotes, is influence by jackrabbit populations (Best 1996; BISON-M 2009; Flinders and Chapman 2003). We expect the major prey species of coyotes, including the black-tailed jackrabbit, to experience steady or increased predation pressure because coyotes are likely to be resilient to climate change (Table 3.1). We also expect bobcats, which are more specialized predators and competitively inferior to coyotes (Anderson and Lovallo 2003; Larivière and Walton 1997), to experience greater competitive pressure and therefore be more vulnerable to climate impacts on the biotic community.

3.2 Discussion

The results of this assessment highlight important expectations and potential issues for species within the MRGB. The information summarized here is the first step toward integrating climate change into management plans and identifying future research needs. In addition, the assessment process highlights the inherent challenges of trying to forecast the ultimate outcomes of climate changes on future animal populations. Within the current assessment, we found three areas that presented challenges to our assessment process: (1) the complexity of interactions between disturbance processes and future expected changes; (2) defining clear outcomes based upon species traits; and (3) information gaps in species biology. Following, we consider each of these issues and discuss how we addressed them within the context of this assessment. Through this discussion, we provide context for climate change assessment processes that we hope not only informs the interpretation of our findings but also the interpretation and implementation of other climate change vulnerability assessments.

Disturbances such as fire and drought can rapidly alter ecosystems and affect a large number of species within a short time period. However, these effects are difficult to quantify over longer time periods. For instance, fire is associated with short-term pulses in resources, such as food or snags, or beneficially opens up habitat for some MRGB species (see Chapter 4). However, wildfires may also result in habitat decline in the long-term, especially in the case of severe fires, which eliminate snags and favor invasive species. To best reconcile potential alternatives in disturbance outcomes, we based our scoring decisions upon our understanding of the relative sensitivity of species to direct and indirect disturbance effects. For instance, we weighed the relative impact of increased fire frequencies on mortality versus the impact of fire-related habitat changes. We also defined a target period of 30 years. For example, in the case of snag availability, though we expected short-term increases due to fire activity and senescence of cottonwoods, we felt continued fires would result in a net decline in snags at our 30 year target date.

Like fire, drought can have multiple direct and indirect consequences for animal populations, which could introduce redundancy into the scoring process. Measures of relative vulnerability to climate change become less meaningful if they are based upon redundant scoring processes (Bagne and other 2011). For some species, a shift to more arid environments as a result of prolonged drought will lead to loss of associated vegetation (i.e., habitat loss), which will increase mortality through loss of protective cover (increased predation) and decreased food resources. If food availability was used as a proxy of habitat quality, we predicted increased vulnerability due to likely habitat changes and to either decreased habitat quality or decline in food availability, but not both. In this way, we accounted for the effect of drought on food resources in a single area of the assessment. However, we found several instances where drought has multiple additive effects for species. For instance, for many amphibians, drought-induced changes to MRG hydrology are likely to reduce habitat availability (riparian vegetation), habitat component availability (ponds), habitat quality (duration/temperature of ponds), and food sources for aquatic species. It is important that the assessment process, as well as our interpretation of assessment results, be flexible and amendable to examining these complex issues in context of multiple potential outcomes.

We noted some interesting complexities regarding two characteristics used by this assessment: type of habitat use and hibernation. Specialization, whether for certain habitat features or food resources, is often associated with increased risk or sensitivity to stochastic events. This assessment did not categorize vulnerability according to species specialization per se but instead relied on a series of questions regarding habitat requirements at multiple scales and our expectation for changes in those required elements. We found that specialists tended to receive vulnerable scores, but this effect may be driven as much by the capacity to judge habitat change for specialists as a realized vulnerability. For the woodpeckers, our capacity to predict habitat changes for specific vegetation types allowed us to predict increased vulnerability for three species that specialized on riparian areas versus the ladder-backed woodpecker, which is known to utilize more shrubby habitats, despite all four species being sensitive to loss of nesting cavities. However, in another example, the American bullfrog received a neutral Habitat score despite a clear reliance on mesic environments and an extensive requirement for long lasting ponds (Table 3.1). In part, this neutrality was due to the bullfrog's broad habitat associations, which resulted in a mix of positive and negative habitat-specific predictions that were used to identify potential response. Ultimately, it was more difficult to make definitive decisions for species that used a diversity of habitats, especially in absence of evidence that points to the relative importance of various habitat associations.

Another trait clearly related to climate change effects but less clearly related to a negative or positive response was hibernation. Hibernation is a cost saving mechanism for species that deal with resource shortages and was considered to confer resilience in this assessment. However, for some species, hibernation could actually prove less adaptive in warmer climates. Higher temperatures may cause animals to burn through reserves more quickly and increase starvation risk, a major source of mortality for hibernating bats. For the black bear, we considered its capacity to hibernate a neutral characteristic, conferring neither resilience nor vulnerability under climate change (see Table 3.1). Bears are likely to be at a disadvantage under warmer climates where temperatures rise above minimums conducive to metabolic savings. During hibernation, metabolic rates are reduced by ~50% with a corresponding drop in body temperature of 7 to 8 °C (BISON-M 2009). This temperature drop is limited by the size of the black bear as it is unable to reduce its body temperature to those seen in smaller animal species (Larivière 2001). Thus, bears are easily aroused during their dormant state and some populations in the south remain active during winter months (BISON-M 2009; Larivière 2001). Increased temperature could also lead to a reduced number of good denning sites, which are defined as those buffered from changes in ambient temperature. In addition to temperature, food resources play an important role in hibernation as energy during hibernation is derived from body fat (BISON-M 2009). If fat reserves are depleted, bears may be at risk of succumbing to severe dehydration (Larivière 2001). Thus, inadequate fat reserves and warmer temperatures will reduce benefit and could, in some instances, result in increased mortality where hibernating behavior is still triggered by cold temperatures. Bats, such as the pallid bat, also rely on hibernation during cold months and undergo nocturnal cycles of torpor as a means to deal with cool temperatures and low food or fat reserves and to reduce water loss (Hermanson and O'Shea 1983; Rambaldini and Brigham 2008). However, the benefits of seasonal torpor are not always clear, again because the energetic benefit of this state corresponds to ambient temperature (Hermanson and O'Shea 1983).

Perceptions of risk or sensitivity were influenced by the quality and volume of data, which varies greatly across the species studied. Data availability also influenced trends seen for individual traits or questions within the assessment. For instance, most species received neutral vulnerability scores and high uncertainty values (Appendix 2) in the biotic interaction category. There is a paucity of information that identifies important predator-prey-competitor relationships for most species. Important variables related to food resources, such as regional differences in diet and projected changes for many food sources, were often unknown. There was generally more published information about food habitat of birds compared to the other taxonomic categories, but there were still information gaps or studies that lacked sufficient detail to make localized predictions for our study area. Disease was a threat for many species, and we used a number of environmental correlates of disease risk for scoring because there are few studies that examine the relationship between recent climate changes and incidence of disease (Tables 3.3, 3.4). For instance, increased crowding due to reductions in pond area and pond availability increases the overall risk of disease for amphibians and other aquatic obligates. However, as with food resources, we often could not predict future changes in disease incidence with reasonable confidence. Lack of information, however, does not preclude these relationships from having strong effects on species' responses to climate change, and our lack of strong predictions regarding future biotic effects should not be taken as an indication that these effects are unimportant.

Neutral scores can result from a prediction of no effect (no benefit or cost associated with exposure to change), an equal balance of vulnerable and resilient trait, or a lack of capacity to determine an effect (see Bagne and others 2011 for discussion of uncertainty). Therefore, neutral scores (e.g., score near 0) do not necessarily translate to a lack of realized impact for a species due to climate change, particularly if that score is accompanied by a high uncertainty value (Appendix 2).

Uncertainty scores identified areas where information was insufficient and varied among specific taxa. Uncertainty tended to be highest for birds and reptiles and lower for mammals. A number of species, particularly within the reptile group, had little to no information regarding multiple questions. Of all the taxonomic groups, the available information was most limited for reptiles. Among the *Aspidoscelis* lizards, ecological characteristics were determined largely from data presented in field guides (Stebbins 1985) because the primary literature was largely focused on the genetic characteristics of this genus. For species in this genus, we assumed that all five shared characteristics where biology of an individual species was unknown. We also found a lack of information regarding thermal maxima to determine reptile species vulnerability to rising temperatures though the implications of extreme temperatures are clear in laboratory experiments. There is a need for research that examines the influence of extreme temperatures and sustained high temperatures on survival of both amphibian and reptile species. Additional data, as they become available, may change scores and assessment results. Examination of category and scores for individual questions along with uncertainty can help identify the source of neutrality. To best inform management prioritization, it is important consider the biological relevance of individual score criteria as well as the source of uncertainty.

Table 3.3. Some common diseases known to infect and cause mortality in rodent species that inhabit the MRGB. These diseases are not necessarily known from populations residing within the MRGB.

Species	Disease	Source
Mexican wood rat	plague	Cornely and Baker 1986
Silky pocket mouse	Fort Morgan virus and plague	Best and Skupski 1994
Raccoons	canine distemper, respiratory disease, and pneumonia (chronic pleurisy)	Lotze and Anderson 1979
Hispid cotton rats	rabies, Chagas' disease, and Venezuelan equine encephalitis	Sullivan 1995
Rock squirrels	plague, tularemia, brucellosis, Q-fever, and Rock Mountain spotted fever	Oaks and others 1987

Table 3.4. Diseases known to infect and cause mortality in carnivore species found in the MRGB. Not all diseases are necessarily present within the study area.

Species	Disease	Source
Coyotes	rabies virus, and mange	BISON-M 2009
Bobcat	rabies, cat-scratch fever, gastric enteritis, feline pan leukopenia, plague, tularemia, brucellosis, salmonellosis, leptopsriosis and respiratory infections	Larivière and Walton 1997; Anderson and Lovallo 2003; Currier 1983
Mountain lion	puma lentivirus, bartonellosis, and feline panleukopenia virus	Pierce and Bleich 2003
Skunk	pneumonia, distemper, letptospirosis, canine distemper, leptospirosis, infectious canine hepatitis, rabies, and Powassan virus	Rosatte and Larivière 2003; Wade-Smith and Verts 1982
Weasel	Tularemia, canine distemper, rabies and other diseases	BISON-M 2009; Fritzell and Haroldson 1982; Svendsen 2003
Black bear	Trichinosis	BISON-M 2009

Chapter 4. Management Under Climate Change

4.1 Managing Species for Climate Change

Species may adapt to changes in climate by changing their location through dispersal, behaviorally changing their physiology or phenology, or by evolving over time (The Heinz Center 2008). However, not all species will be able to adequately respond to expected changes, particularly given the rapid rate of global change and the existence of numerous non-climate stressors. For these species, human intervention will be required to prevent or mitigate adverse effects of climate change. Where intervention is not possible to maintain a species population, alternate strategies focused on maintaining ecosystem function or integrity may be more appropriate. Adaptive management, where wildlife managers continually adapt activities according to information learned from current management efforts and monitoring, is a popular and necessary strategy under climate change (The Heinz Center 2008, 2011). Recent efforts have synthesized strategies for managing species and biodiversity from a variety of sources (The Heinz Center 2008; also see Table 4.1, Box 3). Management strategies to cope with climate change, or adaptation strategies, include efforts to mitigate non-climate stressors, conserve existing biodiversity and function, restore habitats or processes, and manage for ecological function. Ongoing plans for management of the MRGB include actions for each of these elements, although actions were not always designed with climate impacts in mind. The Comprehensive Wildlife Conservation Strategy (CWCS 2006) for New Mexico identifies the following management strategies specific to the conservation of Southwest riparian systems:

- Work with Federal/state/tribal/private entities to protect, restore, conserve, and create aquatic habitats.
- Collaborate to increase public awareness and understanding of aquatic function, services, and value; to emphasize risk of non-natives and seek acceptance of "in stream flow" water rights for wildlife; and to promote water conservation.
- Collaborate to mitigate and prevent invasive species in aquatic areas.
- Adopt standardized monitoring and survey methods; improve current data management systems.
- Collaborate to implement existing plans.
- Collaborate to identify and promote grazing methods that promote long-term sustainability of natural ecosystems and grazing productivity.
- Conduct risk assessments for disease.
- Determine ways to prevent the lowering of groundwater levels.
- Collaborate with Federal and state agencies to reduce water extraction and road building; reduce or mitigate impacts of other human land and water use practices.
- Work with the U.S. Forest Service to reduce wildfire effects on water quality and reduce or prevent negative effects from fire management practices.
- Increase compliance with regulations.

Without knowledge of species biology, ecology, interactions, climate conditions, and human dimensions, we cannot identify future effects nor make decisions that will result in positive outcomes. Monitoring is a core element to adaptive management and assists managers in determining species response to implemented actions

Table 4.1. Strategies proposed by various reviews for managing habitats and wildlife for climate change. Similar strategies are listed together with more detailed examples to the right. Sources are listed for reference and as an indication of the frequency that the strategy is suggested in the literature.

Strategy	**Examples**	**Source**
Reduce impacts of other threats (e.g., habitat fragmentation/conversion, pollution, desertification, and exotic species)	• Reduce anthropogenic stress • Reduce non-climate stressors • Prevent and control invasive species • Measures to reduce risk of catastrophic wildfires • Reduce likelihood of catastrophic events affecting populations • Increase agricultural productivity to reduce pressure on natural resources • Implement seasonal bans, protected areas, and payments for environmental services	Fischlin and others 2007; Inkley et al 2004; Intersecretarial Commission on Climate Change 2007; Hansen, Biringer and Hoffman 2003; Julius and West 2007; Mitchell and others 2007; Mukheibir and Ziervogal 2006; The Heinz Center 2008; The Wildlife Management Institute, 2008; The United Nation Environment Program
Protect key ecosystem features (corridors, keystone species, etc.); promote conditions for ecosystem function	• Ensure ecosystem processes • Preserve and strengthen natural buffering functions within watersheds • Captive breeding • Relocation	Fischlin and others 2007; Julius and West 2007; Mitchell 2007; Inkley and others 2004; Intersecretarial commission on Climate Change 2007; Hansen, Biringer and Hoffman 2003; Palmer and others 2009; The Heinz Center 2008; The United Nations Environment Program
Design reserves able to withstand shifts and disturbances	• Create refugia in areas not expected to experience drastic changes. • Protect multiple replicates of habitat • Include representative habitats and species in refuges • Manage for diverse conditions • Maintain genetically diverse and connected communities • Include lands adjacent to rivers or headwaters	Fischlin and others 2007; Inkley et al 2004; Julius and West 2007; Palmer and others 2009; The Heinz Center 2008; The United Nations Environment Program (include areas outside of refuges)
Monitor indicator species to understand impacts and improve understanding of future impacts		Inkley and others 2004; Mukheibir and Ziervogel 2006
Recognize global climate change as factor in conservation and when projecting conditions; incorporate climate change into plans; optimize sectoral responses to climate change		Inkley 2004; Mitchell 2007; Mukheibir and Ziervogal 2006
Focus on landscape-level management plans	• Incorporate medium and long-range planning • Coordinate species migration management over broad regions • Establish and review biological corridors to ensure the adaptive capacity of ecosystems and species • Promote species dispersal	Intersectretarial commission on Climate Change 2007; Mukheibir and Ziervogal 2006 (sensu); Mitchell and others 2007; Palmer and others 2009
Provide information to managers in a timely manner; provide technical assistance to local managers.		Palmer and others 2009; The United Nations Environment Program
Restore and preserve threatened habitats	• Enhance or replace lost ecosystem services (pollination and seed dispersal) • Restore wetlands for flood control • Create artificial wetlands (increase level of dykes) • Restore entire watersheds, not just streams	Fischlin and others 2007; McNeely and Schutyser 2003 (increase protected habitat); Mitchell and others 2007; Palmer and others 2009; The Heinz Center 2008

in order to inform future decisions. Monitoring efforts also provide data and information regarding the degree of ecosystem change as a result of human modified flows (Crawford and others 1996; Roberts 2005). Environmental monitoring provides information on current and changing conditions and is already emphasized in the MRGB Biological Plan (Crawford and others 1993). Ecosystem monitoring, also emphasized in the Biological Plan, provides information on important variables such as leaf litter and arthropod communities (Crawford and others 1996). Population monitoring of plant and animal species is critical to efforts to identify or minimize climate change impacts on sensitive species, reduce uncertainties, and identify critical actions (Table 4.1). As an example, many of the species assessed within this case study rely on insect prey, but little information exists on invertebrate population dynamics under different climate conditions, which limits our predictions. The uncertainty scores generated as part of this assessment process point to other gaps in our knowledge for species inhabiting the MRGB (Appendix 2). We need information not just on species but also on the interactive effects of climate change with grazing, fire, and invasive species that will ultimately determine future conditions. As research and information expands in many critical areas, monitoring allows us to update our priorities and assessments. However, future monitoring protocols must include measures of climate change impacts in order that future assessments are able to accurately gauge condition and plan restoration activities.

Improvements for wildlife monitoring programs have been suggested in light of potential climate change effects (The Heinz Center 2011):

1. Adjust data collection protocols to include elements or population parameters that are sensitive to climate changes.
2. Incorporate additional data sources related to climate parameters in monitoring protocols.
3. Determine and monitor indicators of climate change effects (demographics of species, stream flow temperatures, snowpack, and flood regimes).
4. Monitor indicators of both long- and short-term changes in wildlife populations (fecundity versus direct mortality).
5. Develop effective methods for assessing and reporting results so they can be incorporated into adaptive management plans.

In addition, research will continue to be an important route to provide basic and applied biological information on riparian species, particularly with respect to climate change. Modeling efforts to project future species distributions and dispersal are also useful tools, although they can include large levels of uncertainty (The Heinz Center 2011).

Current management limitations to species conservation are many: cost, lack of knowledge regarding species response, uncertainty regarding the degree of change, limited jurisdictions for management units, laws, and physical locations of current reserves. Although likely to improve conditions for many species, attempts to restore the MRGB to historic conditions are probably not likely or possible because restoration assumes some return toward historical conditions (Crawford and others 1996). The MRGB has experienced substantial changes to its geomorphic and biotic makeup over the last 50 years (see Chapter 1) and the continued presence of dams and growing human population limits future potential restoration (Crawford and others 1996). Changes to hydrological regimes and, in particular, the potential for significant decline in water availability will remain an issue even without climate change (Fig. 4.1). However, careful management to reduce

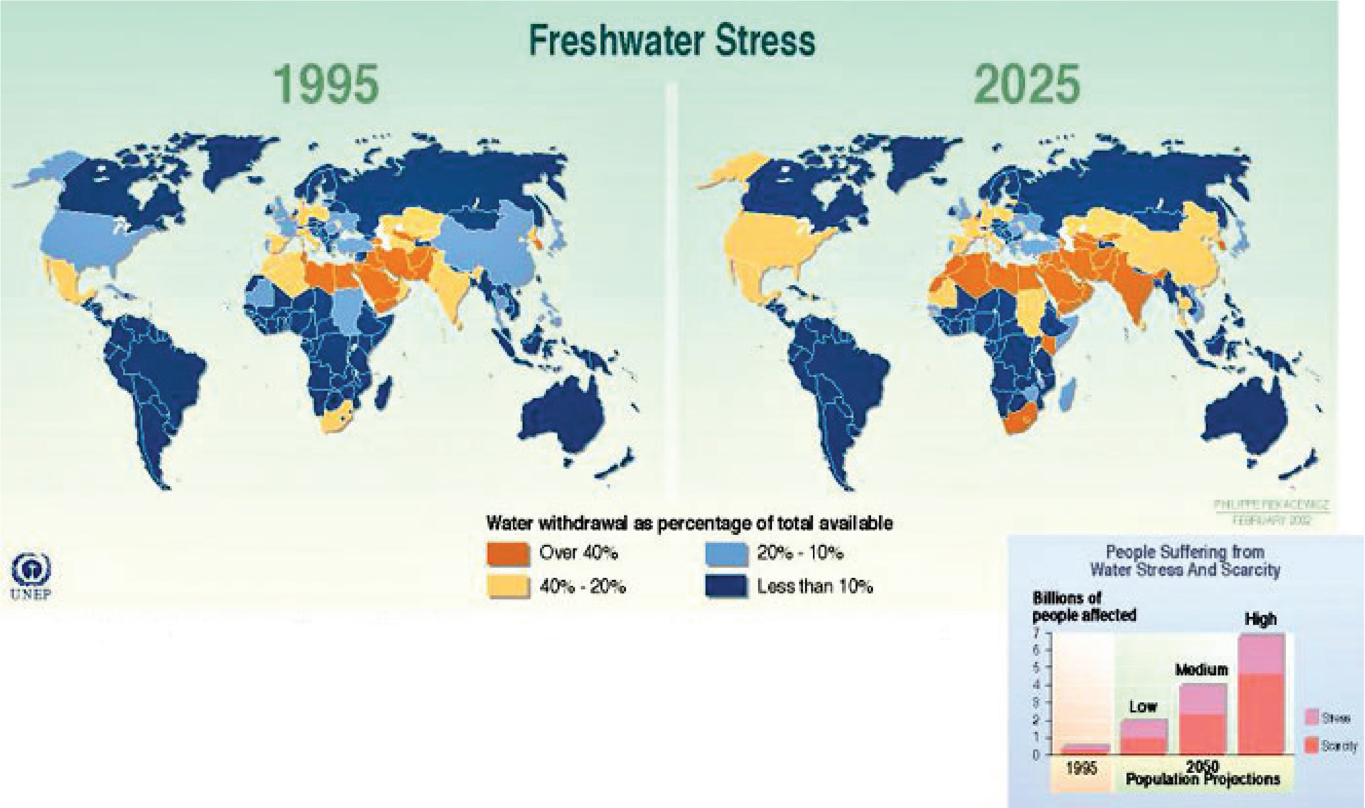


Figure 4.1. Freshwater stress index for 2025 shows that water withdrawal will become a greater issue in this region.

or reverse ongoing habitat degradation due to human activities (e.g., fire and dams) may provide a mechanism for increasing riparian forests and, thereby, species resilience to climate changes (see Box 3). In recognition of this, the Biology Interagency Team (see Roberts 2005 for history) developed the Middle Rio Grande Ecosystem Bosque Biological Management Plan, which includes 21 recommendations, to serve as a planning guide for stakeholders, scientists, and conservationists interested in preserving the MRG and its associated habitats (Crawford and others 1993). By building upon ongoing efforts to reduce risks of wildfire and preserve important habitat, future management may be able to preserve natural function of the riparian communities of the MRG. Actions are being implemented to improve river function and habitat sustainability, thereby helping to mitigate future stress to animal populations arising from increasing temperatures (e.g., USACE 2011; Roberts 2005). However, we must continue to link current management activities and recommendations to information on likely climate effects to increase the potential effectiveness of such strategies for future populations of wildlife (Table 4.1). This vulnerability assessment provides a starting point for developing new management strategies, evaluating ongoing activities, and identifying priorities for the future management of the MRGB

4.2 Lessons from the Assessment Process

Droughts, temperature increases, water availability, and interactions of climate with ongoing disturbance processes were important drivers of predicted negative impacts for the MRGB species. Table 4.2 identifies species with known sensitivities to these processes and disturbances. The following discussion covers a small

Box 3. Specific suggestions outlined for management of riparian habitats under climate change. Most of these suggestions correspond with Crawford and other's (1993) 21 Biological Management Recommendations.

1. Maintain or enhance early successional vegetation and ensure connectivity between summer and winter ranges for big game species (Wildlife Management Institute 2008).
2. Plant riparian vegetation, increase use of swales, use stormwater detention ponds or natural stream corridors (Developed for salmon by Natural Resources Canada Group 2003).
3. Create artificial wetlands (various).
4. Protect/restore key winter and or summer range for mule deer and elk (Wildlife management institute 2008).
5. Ensure adequate environmental flow through using conjunctive groundwater/surface water management (Palmer and others 2009).
6. Eliminate/minimize grazing impact (Obedzinski and others 2001).
7. Reduce stand density to improve moisture and sunlight conditions for plant establishment (Obedzinski and others 2001).
8. Increase plant species diversity using artificial seedbeds, planting and selective thinning (Obedzinski and others 2001).
9. Used timed interval flooding to maintain riparian function (Obedzinski and others 2001).

percentage of the species assessed and reflects, in part, a limited state of knowledge regarding basic biology of many of the species. It is likely that the themes discussed here have far reaching implications to the biological community at large.

Drought

Drought duration, frequency, and severity are expected to increase due to climate change (Seager and others 2007). Droughts stress vegetation and affect survival and establishment of plant species, thus affecting the entire biotic community. Drought may exacerbate ongoing issues with invasive species, many of which are able to tolerate extreme conditions. For instance, abiotic conditions such as soil moisture are a primary determinant of whether Tamarisk or cottonwoods predominated in post-fire vegetation communities (Crawford and others 1996; Ellis 2000). Droughts will also be a central issue to water management because they heavily influence the ability to maintain ecological flows. Although it is likely that drought will negatively impact much of the MRGB wildlife, documented incidents of drought-induced population declines are somewhat rare. We found many terrestrial vertebrate species of the MRGB most negatively affected by the impacts of drought on in habitat availability. Among the documented cases of species decline, we note the following species of concern:

- Drought is associated with large population declines of amphibian species such as the barred tiger salamander (Rustigian and others 2003), Couch's spadefoot toads (Tinsley and Tocque 1996), and Great Plains spadefoot (Bateman and others 2008).

Table 4.2. Species with demonstrated mortality due to extreme weather events. Specialist behaviors that may be important under changing climates are also noted. Introduced species indicated with an (I).

Common name	Scientific name	Drought	Increased fire	Heat waves	Flood	Specialist
Amphibians						
American bullfrog (I)	*Lithobates catesbeiana*				x	use permanent ponds
Barred tiger salamander	*Ambystoma mavortium mavortium*	x				
Couch's spadefoot toad	*Scaphiopus couchii*	x				
Great plains toad	*Bufo cognatus*	x				
New Mexico spadefoot toad	*Spea multiplicata*	x				
Northern leopard frog	*Lithobates pipiens*	x				use permanent ponds
Plains spadefoot toad	*Spea bombifrons*	x				
Western chorus frog	*Pseudacris triseriata*	x				
Woodhouse's toad	*Bufo woodhousii*	x				
Reptiles						
Big bend slider	*Trachemys geigeae*	x				
Black-necked gartersnake	*Thamnophis cyrtopsis*	x				x
Checkered gartersnake	*Thamnophis marcianus*	x				x
Common gartersnake	*Thamnophis sirtalis*	x				
Painted turtle	*Chrysemys picta*	x				
Racer	*Coluber constrictor*		x			
Red-eared slider (I)	*Trachemys scripta*	x			x	
Side-blotched lizard	*Uta stansburiana*		x			
Spiny softshell turtle	*Apalone spinifera*				x	
Western diamondback rattlesnake	*Crotalus atrox*		x			
Western hognose snake	*Heterodon nasicus*					x
Western terrestrial gartersnake	*Thamnophis elegans*	x				
Western whiptail	*Aspidoscelis tigris*	x				
Western/prairie rattlesnake	*Crotalus viridis*		x			
Birds						
Bank swallow	*Riparia riparia*	x			x	aquatic insects
Barn swallow	*Hirundo rustica*			x		
Black-chinned hummingbird	*Archilochus alexandri*	x		x		nectar drinking species-Flowering is sensitive to droughts, etc.
Cliff swallow	*Petrochelidon pyrrhonota*	x		x		
Eastern bluebird	*Sialia sialis*					
European starling (I)	*Sturnus vulgaris*			x		

Table 4.2. *Continued.*

Common name	Scientific name	Drought	Increased fire	Heat waves	Flood	Specialist
Phainopepla	*Phainopepla nitens*	x				x
Red-winged blackbird	*Agelaius phoeniceus*	x			x	
Southwestern willow flycatcher	*Empidonax traillii extimus*					aquatic insects
Summer tanager	*Piranga rubra*			x		x
Mammals						
Beaver	*Castor canadensis*				x	
Bobcat	*Lynx rufus*	x				
Botta's pocket gopher	*Thomomys bottae*					
Coyote	*Canis latrans*	x				
Hoary bat	*Lasiurus cinereus*		x*			cutworms
Long-legged bat	*Myotis volans*					moths
Mexican free-tailed bat	*Tadarida brasiliensis*					moths
Mountain lion	*Puma concolor*	x				
Occult bat	*Myotis occultus*					may specialize on beetles in the MRGB (Valdez, pers. comm.)
Pallid bat	*Antrozous pallidus*			x*		

*inferred from indirect evidence for sensitivity to such an event

- Reptile species, including western whiptails and gartersnakes, may also be prone to population declines during drought years. Western whiptail populations crashed during an extreme drought in California (Pianka 1970). Drought-related reductions in vegetation density, which reduced survivorship among sensitive whiptail hatchlings, were probably a primary mechanism for this effect. Reproduction in the gartersnake, *Thamnophis marcianus,* was greatly reduced following two drought years, presumably due to effects of drought on prey base (Ford and Karges 1987). In most snakes, reductions in prey base not only affect hatchling survival but also prevent egg development in females. Therefore, despite a paucity of information regarding most gartersnakes overall, the increased risk of population declines due to increasing drought conditions was well supported by scientific evidence.
- Among the birds assessed, drought had a demonstrated effect on screech owls and red-wing black birds. Screech owl populations in southwestern Arizona declined 70% in three years during a drought (Hardy and others 1999b). Mortality in this species was often attributed to starvation, particularly of nestlings, which is frequently associated with weather. Severe drought has been reported to cause large-scale reproductive failure in red-winged blackbirds (Brenner 1966).
- Drought is likely to be a problem for many carnivore species, which are sensitive to prey declines. Severe drought is known to affect prey availability and reduces pregnancy rates in bobcats (Larivière and Walton 1997). Black bear are also sensitive to the effects of drought on food resources, specifically those associated with masting events; coyote populations have been observed to decline; and long-tailed weasels have reduced reproductive success (Chapter 3).

Although carnivores are considered habitat and prey generalists, a characteristic associated with resilience, declines of productivity associated with drought are likely to have a significant negative impact.

- Several rodent species, including the New Mexican meadow jumping mouse, muskrat, and cotton rat, are known to be negatively affected by drought. For the jumping mouse, the effect is indirect through habitat changes: drought fragments suitable habitat as certain riparian zones become too dry, which limits dispersal, isolates populations, and increases stochastically driven extinction (Frey 2005). Muskrats have suffered severe population declines in Idaho through increases in predation resulting from an increase in exposed burrow entrances during drought (Errington 1939). *Sigmodon* spp. have shown severe declines in response to consecutive drought years, largely due to its effect on food resources (Sullivan 1995; Texas Mammals Online).
- Adult mule deer have a high survival rate, and most doe deaths were attributed to natural mortality influenced by weather (Smith 1991). However, drought has had clear and negative consequences on mule deer reproduction, leading to population declines (Leopold and Krausman 1991).

Increased Temperatures

Temperature increases will have a number of direct and indirect effects for species inhabiting the MRGB. Issues relating to phenology, physiological limitations, and limitations to daily activity periods that could impact reproductive success were found for multiple taxa. Of these three mechanisms, we found the most common issue for species facing increasing temperatures was the potential for changes to phenology. In particular, we identified the following vulnerabilities relating to changes in temperature or precipitation cues:

- We found most amphibians and many reptiles are likely to experience restrictions to overland movements (e.g., frogs, salamander, and toads) and difficulties with migrations (e.g., gartersnakes and turtles) that rely on rainfall or extended moist periods.
- Earlier temperature rises will have an impact on hibernating species such as black bears, bats, and some rodents. Hibernation, migration, and parturition in bats vary with climate conditions (Hermanson and O'Shea 1983), and we generally found bats vulnerable to risks associated with reliance on weather-related cues. Additionally, hibernation and migration of many bat species is tied to insect emergence (Popa-Lisseanu and Voigt 2009), which, in turn, is tied to temperature (Valdez and Cryan 2009).
- We considered black-chinned hummingbirds, which time migration and breeding to plant flowering (Baltosser and Russell 2000), vulnerable to a mismatch with resources because of the great spatial distance between resources and cues (Baltosser and Russel 2000). Different rates of climate change across the migratory route of the hummingbird will lead to a greater risk of mistimed activity (Bagne and others 2011).
- Increasing temperatures are expected to lead changes to the hydrology of the MRG and, in particular, result in earlier spring flows and an overall reduction to summer flows. This will affect many species, including beavers, aquatic turtles, and amphibians, through changes in habitat as discussed in the Water section (this Chapter).

Among the species likely to experience negative impacts due to a physiological limitation, we found:

- Reptiles will experience direct negative impacts of high temperatures on egg and juvenile survival, particularly when paired with reduced water availability. Higher temperatures are likely to be problematic for species with soft eggs such as the snapping turtle, ornate box turtle, western painted turtle, and Big Bend slider, which are vulnerable to temperature effects on egg survival (Finkler 1999, 2001; Redder and others 2006). In addition, hatchlings were often cited as sensitive to high temperatures (e.g., western painted turtle and snapping turtle; see Brooks and others 1991; Cagle and others 1993; COSEWIC 2006; Finkler 1999, 2001), drought-related decreases in vegetation cover (e.g., ornate box turtle; see Nieuwolt 1996; Redder and others 2006) or both (e.g., Big Bend slider and red-eared slider; see Delmas and others 2008; Stuart and Ward 2009; Western Governors' Drought Task Force). In addition, we considered all turtle species at risk of population decline due to increasingly skewed sex ratios.
- The black-chinned hummingbird was one of few bird species we considered vulnerable due to physiological limitations. The MRGB represents the southern portion of the hummingbird's range, and although it is most abundant in southern portions, its small size and lower nest success with reduced shading are thought to indicate intolerance to increasing temperatures (Baltosser and Russell 2000). The black-chinned hummingbird's range has expanded northward in the Pacific Northwest over the last century, probably in accordance with rising temperatures (Baltosser and Russell 2000).
- Very hot weather is also known to kill European starling nestlings (Cabe 1993).
- We considered the pallid bat vulnerable to increasing temperatures because it is known to be sensitive to higher temperatures, which affect its capacity to forage and can limit roost suitability to those with good insulation (BISON-M 2009; Hermanson and O'Shea 1983). Although other bat species are known to tolerate higher temperatures, several authors noted that tolerance for high temperatures is not a key factor for bat survival, but, rather, high temperatures can prevent bats from entering torpor, a critical behavior that enables bats to survive food and water shortages (Carpenter 1969, 1988). The relationship between temperature and bat survival, however, is not clear and appears to vary among bat species. For instance, Wilkins (1989) reported that Mexican free-tailed bats live only one-quarter to one-third as long as hibernating species under cooler temperatures but appear to fare much better than other species under warmer conditions.
- Though widespread, skunks are not usually found in the very arid Southwest and are near the southern limit of their range in New Mexico (Wade-Smith and Verts 1982). This and the local dependence of this species on more mesic riparian habitats point toward a physiologically mediated limitation on the range of striped skunk. Long-tailed weasels become quiescent during hot weather and are less alert (Sheffield and Thomas 1997), which could affect survival under warming trends.
- *Neotoma* spp., in particular the white-throated woodrat, may already exist close to their upper thermal maximum (i.e., lethal limit) in desert areas (Macèdo and Mares 1988). The white-throated woodrat moderates temperatures by reducing pelage, utilizing efficient vascular systems, and staying within burrow (Macèdo and Mares 1988). *Neotoma albigula* are nocturnal and fossorial and select succulent foods, features typical of a species sensitive to high temperatures (Macèdo and Mares 1988). Though this woodrat species is predominantly

southern ranging, it inhabits montane and other "cool" environments, which may indicate some sensitivity to high temperatures. BISON-M (2009) reported that when found in warmer sites, woodrats are often associated with boulders, which are thought to provide the preferred microclimate. Population declines may, thus, be associated with a number of high temperature factors including direct mortality, reduced activity time, and limitations in burrow sites with the appropriate microclimate.

We also predicted that many species in the MRG will experience reductions in activity periods due to increasing temperatures:

- For most reptiles, we recognized that increased daily maximum temperature will potentially limit daytime foraging opportunities to the extent that reptiles are unable to fulfill metabolic needs, which has been blamed for local extirpations of some lizard species (Huey and Tewksbury 2009). However, increases in seasonal activity periods could counter potential loss to daily activity to some extent. Still, many species are also sensitive to precipitation changes, which, combined with potential effects of daily temperature fluctuations, is compelling evidence for a potential negative effect on lizard energy budgets (Huey and Tewksbury 2009; Sinervo and others 2009). We considered the side-blotched lizard, southwestern fence lizard, racer, and gopher snake to be at risk of reduced activity due to higher daytime temperatures (Brennan 2008). Huey and Tewksbury (2009) indicated that the spotted Chihuahuan whiptail may have difficulties with adequate foraging time under warming regimes (Winne and Keck 2004). Gopher snakes were considered to be at risk of exposure to temperatures in excess of their thermal maxima and which can reduce daily foraging time (Diller and Wallace 1996). We also considered the ornate box turtle, which is most active in mornings or immediately after rain (Niedzielski 2002; Redder and others 2006) to be sensitive to temperature losses in excess of benefit to increased growing season.
- For mammals, we only considered species with documented temperature sensitivities vulnerable to temperature-related changes in foraging and reproductive periods. Small desert mammal species exposed to higher soil temperatures (30 °C minimum) may experience a reduction in energy savings associated with seasonal dormancy by either increasing resting metabolic rate or reducing the time that a species can undergo torpor, which is likely to lead to reductions in activity times in particularly hot environments (Walsberg 2000). We applied this theory to nocturnal species, which are already limited to three to five hour foraging periods where temperatures are ~30 °C. For instance, *Dipodomys ordii* are only known to initiate torpor in Canada where temperatures are low enough to make this strategy feasible (Bender and others 2007). Bears, the only carnivores in this group, were also considered sensitive to climate-related reductions in activity periods. Bears are described as primarily crepuscular and diurnal though they can shift to more nocturnal activity patterns during fall months or in the vicinity of human populations (BISON-M 2009). Diurnal activity is thought to be driven by the need to increase food intake, and activity is reported to decline due to rain, snow, or extreme temperatures (Larivière 2001). Rock squirrels are also likely to experience increased limitations during the hottest part of the year. Elk are reported to be regulated by hunting, disease, and snow (BISON-M 2009). Typically, large herbivore populations are limited by the effects of winter weather on adult survival and yearling recruitment (Wang and others 2002). Therefore, warming, especially that occurs in winter months, may lead to increases in abundance. However, recent modeling efforts have found that spring

precipitation increased elk population growth, whereas mean monthly summer temperatures were negatively associated with elk survival (Wang and others 2002). Ultimately, warmer, drier conditions will likely not favor elk because although mild winters may increase fawn survival, extreme events (ice storms, heat waves, etc.) negatively impact elk survival.

- We noted six species of birds that were likely to experience some limitation due to physiological sensitivity to increasing temperatures. The western yellow-billed cuckoo is considered at risk of reaching its physiological limit under warmer conditions. The yellow-billed cuckoo is restricted to riparian areas in hot regions, which are generally cooler and more humid than the surroundings. Further, incubating adults and nestlings have been observed panting on hot days (Hughes 1999), and eggs may be prone to drying, which reduces hatchability (Laymon and Halterman 1987). Four other passerine species had evidence supporting intolerance of hot temperatures and were considered at risk of reaching a potentially limiting physiological threshold: the summer tanager, yellow-breasted chat, southwestern willow flycatcher, and Indigo bunting (Eckerle and Thomson 2001; Robinson 1996; Walsberg 1993).

Water

Water availability drove many of the vulnerabilities of the 117 species assessed for the MRGB. Water is a critical resource that is in high demand and in ever increasingly short supply in the Southwest. The many competing economic, social, and cultural interests that have long dominated management priorities complicate water management for biodiversity. Riparian habitats have already suffered severe reductions and alterations from human development adding to changes in temperature and precipitation that is exacerbating regional water issues. Dams are recognized as a major issue for fish, amphibians, and water dispersed plants and have already been associated with local extirpations of fish in the Southwest (Fagan and others 2002; Palmer and others 2009). In the recent assessment by the Comprehensive Wildlife Conservation Strategy, ephemeral natural catchments, perennial marsh/cienga/spring/seeps, and riparian habitats were found to be at the greatest risk to the synergistic effects of human-related habitat disturbance and natural processes such as climate change. These habitats are limited primarily to the bosque found in Albuquerque, New Mexico, and the Low Flow Conveyance Channel in Sierra and Socorro counties (Fig. 1.1). These habitats, including oxbow lakes, marshes and backwater features, are limited in area because water tables have declined due to land conversion and channelization (Fig. 4.2). As highlighted by the scoring results, species dependent on surface water for any life stage are vulnerable to reductions in water availability and changes to the timing and conditions of water availability.

The following are some examples of species with vulnerabilities related to water:

- Loss of perennial marshes and cienegas are likely to have negative consequences for populations of leopard frogs, and plans to reconstruct the Low Flow Conveyance Channel (see Fig. 1.1), which will reduce flooding and cause further decline in wetland habitats, are expected to increase negative impacts for amphibians (see Chapter 1).
- Beavers were considered prone to habitat quality declines related to climate change effects on river flow. Beavers require a permanent flowing water source and a minimal amount of water in order to use their lodges (Jenkins and Busher

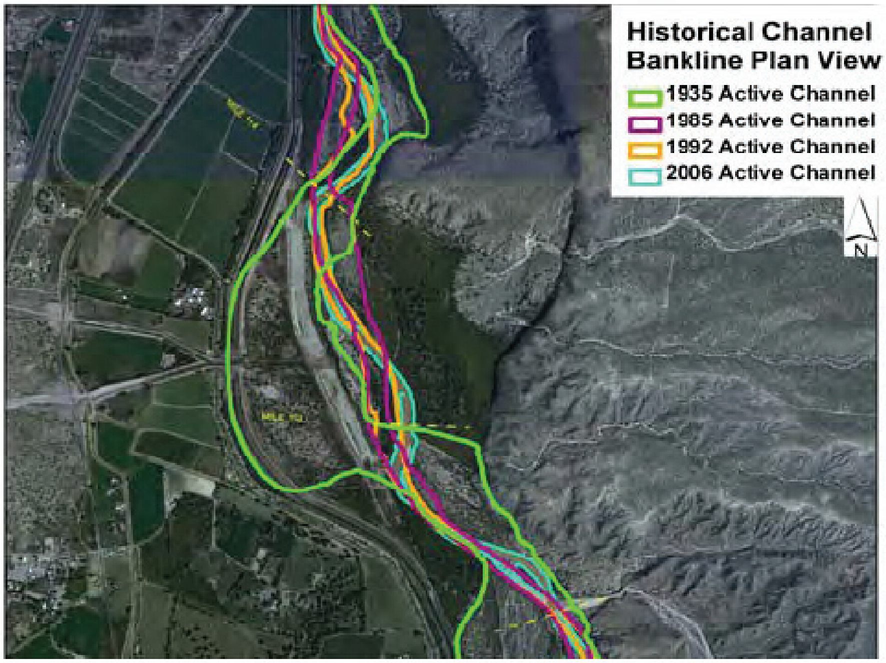


Figure 4.2. Channelization has caused considerable loss of riparian forest habitat. This figure shows the extent of such changes over the last 80 years. The construction of Cochiti Dam in 1973 led to considerable change in the geophysical nature of the MRG (upper panel), and introduction of the low conveyance channel in the southern MRG had considerable impact (lower panels). Adapted from Padilla and Baird (2005).

1979). Typically, beavers are forced to find more suitable habitat when siltation has reduced the depth of a beaver pond (Struebel 2000). Availability of food is considered an important determinant of habitat suitability.

- Similarly, areas prone to flooding are unsuitable for muskrats because sloughs and marches will silt up and muskrats cannot inhabit fast moving waterways (BISON-M 2009; Willner and others 1980).
- Several bat species show a strong reliance on free water sources, particularly during lactation when females have an increased need for water (Adams and Hayes 2008). Evidence supports the notion that bats rely on a water source and, thus, are probably more sensitive to the loss of open water sources than other mammal species. The big brown bat and Mexican free-tail bat cannot exist without a water source (Carpenter 1969; Wilkins 1989). Bats show little ability to concentrate urine or other physical traits for water conservation, and it is likely that behavior, specifically roost selection, plays a large role in thermoregulation in bats and their capacity to persist in arid environments (Carpenter 1969).

Wildfire

Fire was an important consideration during the assessment, particularly for any habitat-related effects, as it can rapidly initiate successional habitat progression or convert habitats to new types. Fire is under-studied in Southwest riparian areas, and species, including invasive species, have variable responses to fire disturbances. Fire is a function of fuel loads and weather conditions with the ultimate outcome, in terms of severity and intensity, determined by site level composition of topography, soils, and vegetation (Dwire and Kauffman 2003). Little is known about historic fire regimes in southwestern riparian habitats though many of the characteristics of these habitats (e.g., moist fuels, lower temperatures, and presence of water) mean that fires were probably less intense and frequent in riparian zones than in surrounding upland areas (Dwire and Kauffman 2003). More recently, human activity has led to an increase in ignition events and more frequent fires, while fuel accumulation due to changes in flood regimes and habitat modifications has led to more high intensity fires (Busch and Smith 1993; Ellis 2000).

Native and invasive species' response to fire varies by species involved, intensity of fire, seasonality of fire, and post fire conditions (Dwire and Kauffman 2003; Ellis 2001). For instance, Fremont cottonwoods, *Populus fremontii*, in Arizona are highly susceptible to fire-related mortalities (Busch and Smith 1993). Cottonwoods native to the MRG, *P. deltoids*, have recovered well after at least one high severity fire (Ellis 2000). Tamarisk is generally considered better able to establish in post-fire sites than cottonwoods (e.g., Busch 1995; Busch and Smith 1993) and, once present, prevents establishment of other species. At least one study has shown that tamarisk is better able to withstand post fire increases in salinity (Busch and Smith 1993). In addition, fire-related stress initiates increased flowering in tamarisk species and contributes to their ability to colonize new sites post fire. However, many of these observations regard short-term responses, and long-term (>three years) studies have failed to find significant post fire successional trends (Busch 1995). In part, this may be due to variations in tamarisk survival related to the seasonality of fire (Ellis 2001; Dwire and Kauffman 2003) and the potential of some post fire conditions (e.g., flooding) to favor cottonwood establishment (see Ellis 2001). In addition, fire may benefit other invasive species such as the tree of heaven (*Ailanthus altissima*), cheatgrass (*Bromus tectorum*), and Russian mulberry (*Morus alba*) (the latter not currently established in the MRGB) (USACE 2011). The ultimate outcome of fire for vegetation and the consequential impact on wildlife species depends upon many factors that provide the setting that can favor the establishment of very distinct post-fire communities. For this assessment, we relied on documented cases to account for potential fire due to synergistic climate interactions.

As with fire-habitat effects, we felt care must be taken to distinguish between short- and long-term costs and benefits of future fire regimes on native fauna. For instance, we determined both potential negative and positive effects for carnivores inhabiting the MRG. Carnivores are likely to experience changes in food availability if prey decline due to fire-related mortalities or are attracted to newly burned sites. Hunting success for predators could improve where cover is reduced or reduced cover may hinder a predator's ability to ambush prey. Fire initiated succession of and/or habitat conversion of forest systems can lead to large changes in habitat suitability over time for some species; and, ultimately, more frequent and intense fires are likely to have a deleterious effect on mature riparian forest habitat with negative downstream consequences for most species, including carnivores.

Among the species potentially able to benefit from increased fire, we found the following:

- Prescribed fire could improve habitat for the Great Plains skink by opening understory vegetation (Fitch 1955) in areas where fossorial habits protect it from fire-related mortality (Western Governors' Drought Task Force).
- Fire in desert and semidesert communities reduces shrub cover and creates more open areas that are favored by Ord's kangaroo rats (BISON-M 2009).
- Northern flickers, which are known to occur in clear cut or burned areas and require open habitat for foraging (Weibe and Moore 2008) may benefit from increased fire frequency.
- Many mammals, including bobcats, mountain lions, rodents, and deer, benefit from the structural and qualitative changes created in a fire landscape that includes a mosaic of burned and unburned areas (from the Prescribed Fire and Fire Effects Research Work Unit 1996; BISON-M 2009).
- Elk and mule deer are drawn to the increased nutrient content of forage on recently burned sites (BISON-M 2009).

Other species, especially amphibians and dispersal-limited animals, may suffer mortalities due to the direct and indirect (habitat based) effects of fire.

- Fire is known to result in temporary declines in meadowlark populations (Pylypec 1991).
- Reduction, though not extirpation, in summer tanager numbers following wildfires was found for the MRG (Smith and others 2006).
- Human impacts relating to fragmentation and changes in fire regimes may exacerbate expected losses of habitat for the hairy woodpecker (Jackson and others 2002; Vierling and others 2008).
- Fire has been known to destroy the dens of one rattlesnake species (*Croatalis viridis*) and is likely to cause losses in similarly denning western diamond rattlesnakes.
- Western chorus frogs may be at risk if fire reduces the logs and debris used for thermal protection during hibernation (BISON-M 2009).
- Fire may reduce survival of the side-blotched lizard (BISON-M 2009) through vegetation removal though varying effects have been noted depending on location (Barbault and Maury 1981, and sources from BISON-M 2009).
- Removal of ground cover by fires can increase sedimentation and ash deposition in water, which negatively affects habitats for a number of species with aquatic life stages (Amphibiaweb 2008; Degenhardt and others 1996).

4.3 Identifying Management Strategies

Palmer and others (2008, 2009) organized management strategies for river ecosystems into two categories: (1) proactive measures such as restoration and land purchases that increase river resilience, and (2) reactive measures that arise in response to damage or to mitigate ongoing issues. Proactive measures are more desirable as they may be more ecologically and fiscally effective and are the focus of this assessment. However, many efforts are already underway in response to issues in the MRG (Crawford and others 1996; USACE 2011). These efforts combine both proactive and reactive elements and are likely to increase the resilience of the MRG to climate effects. However, climate change must have greater emphasis within these plans in the future to enable better management practices that incorporate realistic methods to successfully conserve biodiversity and discontinue efforts that are likely to be ineffective under global warming trends (Table 4.1; Box 3).

This assessment summarizes background and vulnerability information that can be used to evaluate how and which species may be impacted under a changing climate. Relevant management actions may then be designed in the context of their potential effectiveness for conserving species under future conditions. The list of potential actions for MRGB conservation is long and mirrors the size of the problem. Solutions and plans for adapting to climate change (Table 4.1; Box 3) are extensive and, ideally, could be fully implemented. However, limited resources and the rapid advancement of climate change require prioritization of some activities. A number of management strategies have relevance across a spectrum of the species assessed in this study. We briefly outline the actions related to these strategies.

Prioritization

Protection of species that are most likely to go extinct is a common approach in conservation management and follows a long tradition in conservation biology. Rare species may be at greater risk of extinction under global change due to their sensitivity to stochastic events and lack of resilience (The Heinz Center 2011). Indeed, Hoyle and James (2006) and Fischlin and others (2007) noted that climate thresholds exist above which extinction probabilities for some species increase dramatically. Sixty-two percent (72) of species in this assessment were identified as vulnerable to population declines associated with climate change. This large number points to the already difficult decisions managers need to make about where to focus limited resources. It is clear from our assessment that threatened and rare species are among the most vulnerable to climate change (Table 3.1). In the next 30 years within the MRGB, the 10 species most vulnerable to population declines associated with climate change are the southwestern willow flycatcher, northern leopard frog, common yellowthroat, western chorus frog, cliff swallow, yellow-billed cuckoo, western wood-pewee, summer tanager, downy woodpecker, and New Mexico meadow jumping mouse. The additional vulnerability of the southwestern willow flycatcher, northern leopard frog, and New Mexico jumping mouse to climate change indicates high risk for extinction and highlights need the for intervention strategies that incorporate climate effects. Through the assessment process, managers are better able to judge the potential effectiveness of strategies whether it be greater protection for current conservation areas or more drastic intervention. Assisted translocation or assisted migration of species may be necessary where conservation efforts are limited, although it can be difficult to predict the future suitability of translocation sites and biological consequences of such moves. Captive breeding programs are often viewed as a last resort, and though they have been successful in extreme cases, such as for the black-footed ferret and California condor, they are often not viable in the long term. Rare species are already likely to be detrimentally impacted by disturbances, and negative climate effects add urgency for management plans to consider climate change.

Though we cannot gauge whether vulnerable species that are widespread will be equally vulnerable to climate change in other occupied areas, these species often play a disproportionately large role in their community and their loss or decline, even if restricted to the MRG, may have many ramifications. Management implications are also found by exploring patterns found within resilient species. Populations resilient to the effects of future climate are less likely to need modified management plans. Conversely, problematic pest or invasive species that are likely to benefit from future conditions may need additional intervention to prevent establishment or colonization under climate change.

Prioritization need not focus just on species ranks but can also consider common aspects of vulnerability that might benefit a large number of species (Bagne and others 2011). For instance, habitat issues drove vulnerability scores for all taxonomic groups, except birds where physiological effects were more important overall. However, measures to improve or restore habitat (reviewed in later sections), such as carefully managing the timing of water releases, have far reaching implications that will benefit multiple species. For instance, well-timed water releases will improve the establishment of cottonwoods to the benefit of birds and bats, reduce severe fire risk and improve habitat quality for bosque dependent species, ensure periods of insect prey availability and benefit of species like the southwestern willow flycatcher, and provide essential microhabitat for turtles and amphibians.

Landscape-Scale Planning

Single species management is often limited in both scope and long-term effectiveness, particularly under scenarios where habitat is expected to change drastically. Landscape- or ecosystem-level plans often are better for long-term objectives and are more viable approaches for increasing the adaptation of species by anticipating and encouraging range shifts. Connectivity of habitats is critical for many species. At a large scales, landscapes sustain populations, and at smaller scales, connectivity allows for dispersal to new sites. Thus, actions that preserve large and continuous swaths of land benefit species conservation (Palmer and others 2009). Still, depending on the focus, landscape-scale plans may be difficult to implement due to political boundaries, societal perceptions, and opposing management objectives.

Increasing the area and connectivity of habitat will be important for mammal species, including bats, large carnivores, and ungulates (Lacki and Baker 2007). Bats, such as the hoary bat, that rely on tree roosts are negatively affected by fragmentation. The future of this bat in the MRGB will probably be strongly affected by how well conservation and restoration activities preserve continuous tracks of suitable habitat. In particular, management actions need to focus on areas surrounding existing bat roosts. We also predict that landscape-level planning that incorporates needs of carnivore species will become increasingly important. Although carnivores as a group are able to disperse to new areas, habitat disturbance is considered a threat to all species. Carnivores, like ungulates, often require large home ranges and are susceptible to fragmentation and land conversion. For instance, mountain lions avoid habitat with roads and human modified vegetation (Van Dyke and others 1983). Similarly, ungulates have large home range needs, and further reductions in habitat within the MRGB are likely to cause decreases in elk populations (Catron and others 2008). Finally, the MRGB does not exist in isolation and is impacted by conditions outside of the boundaries of this study. An issue for many of the case studies of riparian habitat restoration presented in Chaney and others (1993) (e.g., Big Creek, Utah) showed that upstream erosion and sediment deposition can affect downstream recovery. Efforts to improve upstream and upland habitat, as might happen through thinning practices, may be necessary to fully achieve recovery after disturbances such as grazing or fire and to maximize species persistence through time.

Management of Invasive Species

Invasive animal species can have a profound effect on native species primarily through competitive exclusion. Bullfrogs and introduced fish are major predators for most amphibian species assessed in this report and have been blamed, in part, for population declines in the northern leopard frog (Chapter 3). Introduced crayfish, for example (not assessed here), eat tadpoles and are considered a major impact on frog populations (BISON-M 2009). Bullfrogs were expected to be tolerant of changing conditions and, thus, the leopard frog was considered vulnerable to increased predation under warming conditions. Successful management for invasive species under climate change will need to consider individual effects on species and identify potential intervention points that may favor native populations. Although overall the bullfrog was resilient compared to other amphibian species, it is a riparian obligate and will similarly suffer from reductions in aquatic habitats. Importantly, we found that bullfrog eggs have the lowest water temperature threshold of any of the amphibians assessed, and bullfrogs have the longest metamorphosis period and require particularly long lasting water bodies, two characteristics that may be exploited in future scenarios.

Another pervasive issue regards the influence and role of invasive plant species on the ecology of Southwest riparian systems. Vegetation management for riparian habitat restoration has focused on reduction of invasive species (Fullerton and Batts 2003; Shafroth and others 2007). Tamarisk (*Tamarix* spp.) has been a major focus of management and restoration attempts within the MRGB. To a lesser degree, Russian olive (*Elaeagnus angustifolia*) and non-native grass species are also problematic. Tamarisk has a reputation for creating monotypic cultures that crowd out natural communities and are associated with many negative consequences with respect to fire risk, water draw down, and floodplain loss (Taylor and others 2006; USACE 2011). Recent studies and reviews, however, are starting to point toward a new strategy for tamarisk management (Shafroth and others 2007; Stromberg and others 2009). The presumed capacity of tamarisk to actively crowd out native species is based on correlative evidence (Stromberg and others 2009). More recent information indicates that this species becomes dominant within river systems not due to an aggressive colonization so much as having the capacity to establish in sites that are less suitable for native flora due to anthropogenic disturbances such as grazing and alteration of flows (Stromberg and others 2009). In fact, cottonwoods and willow appear to have a competitive advantage where appropriate soil and moisture conditions are present. Further, although cottonwood-dominated habitats appear to support the greatest biodiversity (Ellis 1995; Shafroth and others 2007), tamarisk also plays a role in maintaining habitats for native species. Tamarisk is ubiquitous in some Southwest riparian habitats, and there are numerous examples of species using these habitats for food and shelter (Ellis 1995; Shafroth and others 2007; Stromberg and others 2009). For species that rely on mid-story layers of habitat, tamarisk may provide nesting and foraging sites (Finch and Hawksworth 2006), and insectivore birds appear readily able to use tamarisk-dominated habitat (Ellis 1995). Cuckoo populations, which fluctuate greatly with food availability, benefit during cicada cycles (Heath and Wilkin 1970), and populations of at least one species (*Apache cicadas*) increased when host tamarisk trees increased (Glinski and Ohmart 1984). Lucy's warblers, yellow-breasted chats, and blue grosbeaks are known to do well in habitats dominated by exotic plant species (Eckerle and Thomson 2001; Ellis 1995), including tamarisk- and mesquite-dominated habitats. Removal of the exotic shrub understory and downed dead wood was associated with declines in Bewick's wren and spotted towee populations in the MRG (RMRS Fuels Study data). Still, these examples may represent species with flexible habitat requirements rather than those that have a specific association with tamarisk. Many species are more strongly associated or do better with native riparian communities and, in particular, Lucy's warbler, timber gleaning species (e.g., White-breasted nuthatch), and timber drillers (e.g., woodpecker species) remain tied to native cottonwood-dominated habitats (Eckerle and Thomson 2001; Ellis 1995; Johnson and others 1997; Rosenberg and others 1991). For other species, such as the southwestern willow flycatcher that utilizes saltcedar, it is not clear that these habitats provide a quality substitute for native habitat.

In areas where tamarisk is a primary component of the system (Stromberg and others 2009), efforts to reduce tamarisk may neither be feasible nor desirable. In addition, information is lacking regarding the effects of chemical and physical tamarisk removal on aquatic and biological communities, although effects are likely minimal (Shafroth and others 2007). A more recent approach to tamarisk control has been release of the saltcedar leaf beetle (*Diorhabda elongata*) as a biocontrol agent. As with other removal efforts, there may be unintended consequences that need to be considered carefully (Hultine and other 2010). The balance of costs and benefits of tamarisk habitats and their removal are unclear. An alternate approach may focus on restoration of native species and prioritize in areas that have not transitioned to tamarisk stands (Stromberg and others 2009). Success of site recovery, as measured by native species growth and cover after tamarisk removal, is strongly dependent upon continued restoration efforts, including well timed floods (Shafroth and others 2007). Where post removal restoration efforts are expected to be minimal, removal of tamarisk is not likely to result in restoration of healthy native ecosystems (Crawford and others 1996; Shafroth and others 2007). By shifting management toward maximizing conditions conducive to native species growth, restoration efforts can promote important habitat components and reduce the spread of tamarisk and other invasive species (Stromberg and Chew 2002; Stromberg and others 2009).

Restoration Under Climate Change

Restoration activities that improve habitat and increase resilience are critical to preserve biodiversity in the face of climate change (Fig. 4.3). In addition, greater biodiversity has been linked to greater ecosystem resilience. Plans to create a mosaic of wetlands, willow stands, grasslands, and cottonwood gallery forests may increase biodiversity and foster self-sustainability of the habitats (Fullerton and Batts 2003). Degraded riparian areas lack vegetation communities that stabilize and protect streambanks and provide shade (Chaney and others 1993). Water tables and saturation zones are typically lowered in poor systems and degraded systems have less subsurface water storage. Reduced or absent summer flows and warm waters lead to generally poor habitat for fish and wildlife. Restored riparian areas are characterized as having diverse vegetation communities that shade stream reaches and stabilize streambanks. Water tables are elevated and subsurface water storage increases. Water is cooler in summer and flows are more constant and contain a greater volume throughout the season.

Although dams and levees have significantly altered hydrologic regimes and the flood plain, controlled water releases can be used for ecological benefit in some cases. Maintaining favorable flood regimes is one step toward encouraging native plant establishment and may favor some wildlife taxa, although there are numerous implementation issues because the water that flows in the MRG is already fully

Figure 4.3. Habitat improvement projects such as that developed through a partnership between the non-profit Rio Grande Community Farm and the City of Albuquerque can provide food and shelter for animals by planting wildlife friendly crops on publicly owned open space. Photo Credit: U.S. Department of Agriculture.

allocated. In addition, water in the Rio Grande basin has some of the highest marginal values (i.e., the cost to develop and deliver additional water sources) in the United States; thus, water has a very high economic value and costs for transfers are high (USFWS 2002). Various treaties and flood control acts mandate specific water allocations that must be met for private, Tribal, interstate and national entities (Whitney 1996). Water is sometimes leased or purchased by Government agencies and private organizations to protect in-stream flow (Landry 1998). However, this may not be possible for the MRG. Albuquerque municipal rights are entirely obligated leaving a relatively small (i.e., privately owned) portion available for conservation purchases. Water rights for ecological roles have some legal authority for species under protection of the Endangered Species Act (ESA) if removal of water would result in mortality or "take" or threatens critical habitat. In 2003, the Tenth Circuit Court of Appeals ruled that the Bureau of Reclamation had the discretion to reduce contract deliveries and restrict diversion to meet ESA duties for the endangered Rio Grande silvery minnow (Rio Grande Silvery Minnow, et al. v. Keys, 333 F.3d 1109, 1114-1115 [10th Cir. 2003]). Within the MRG, preservation or restoration of water flow is a major step toward improving and conserving natural habitats. Proactive measures to reduce water loss, preserve functioning wetlands, and prevent further degradation and conversion of ecosystems are needed to limit the degree to which the MRGB is affected by climate change (CWSS; Palmer and others 2009). Summer releases from dams, with particular attention to flow timing, duration, and frequency, could help to maintain ecosystems and prevent river drying (CWSS; Palmer and others 2008). For instance, well-timed flood events with slow drawdown improve the survival and establishment of native cottonwoods (Bhattacherjee and others 2006; Ellis 2001; Taylor and others 2006). In addition, water quality issues related to higher temperatures and sediment inputs can be offset by maintaining river function and vegetation (Palmer and others 2008). A number of geomorphic manipulations, such as increasing floodplain connectivity, reducing and preventing bank erosion, recreating remnant channels, and altering floodplain elevation, can encourage overbank flooding and improve river functionality to the benefit of many wildlife species (Fullerton and Batts 2003). Though water is a finite source within the MRG, it is widely recognized that these actions will improve its preservation. Indeed, reconnecting the floodplain to the river is recognized as a major goal in recent restoration plans (USACE 2011).

However, activities associated with restoration can cause a number of disruptions that may adversely affect wildlife. In addition, managing for ecosystem function could lead to shifts away from natural communities or lower biodiversity (The Heinz Center 2011). Many options are available to help mitigate temporary disruptions, and action is warranted where future shifts to new ecosystems in the MRGB are unavoidable. The recent Environmental Assessment prepared by the U.S. Army Corps of Engineers in anticipation of large-scale restoration activities in the Albuquerque reach of the Rio Grande has proposed mitigation measures (USAC 2011). From the results of this assessment, we have additional considerations that incorporate the potential implication of climate change:

- The timing of restoration activities should consider climate-related changes in species phenology. For example, fire treatments pose risks to many species where thinning activities disturb nest sites. Currently, seasonal limitations are in place to avoid bird nesting, and major disruptions are not permitted beginning August 15 through April 15 (USAC 2011). However, if species phenology changes significantly in response to climate change, these periods may need to be adjusted. Phenology of all affected species needs to be considered, especially those sensitive to disturbances such as bats.

- For many species, reducing non climate-related threats during restoration is important. For example, herbicides pose high risks to amphibians (USACE 2001). Grazing may exacerbate disturbance related to restoration treatments. Warming conditions and increased variability to river flow will reduce the capacity of the riparian habitats and individual species to recover from disturbances. Decisions on land use and conversion should consider the overall effect of human activities plus potential consequences of climate change for habitat loss as discussed in this and previous chapters.
- Restoration goals aimed at preserving a mosaic of habitat structure types are most likely to reduce biodiversity loss. The best defense against the myriad potential outcomes is to preserve a diversity of habitats, which may include non-native species. Preserving a wide diversity of habitats under climate change scenarios is recommended where uncertainty is prevalent. In this way, we hedge against future change by maximizing the ability of the river to shift to a new steady state that maintains ecosystem services.
- Captive breeding and translocation activities might be considered at least temporarily during restoration in anticipation of asynchrony among species needs, climate change effects, and new habitat created by restoration. Some species, such as box turtles, tend to not recover or re-establish in sites from which they have been extirpated (Doroff and Keith 1990). Preventing disturbance at critical sites is the best solution for these turtles, but temporary relocations may be essential to maintaining populations for the future.
- Implement selective thinning and removal treatments for invasive species such as tamarisk and focus treatments on areas with the greatest potential to maintain native communities (see discussion earlier in this chapter regarding invasive species).
- Restore with future conditions in mind: there is little benefit in creating willow habitat in locations where future conditions preclude regular flows. Similarly, strategies for invasive species management and preservation of native wildlife species will need to consider the potential capacity of the site to support species given their unique requirements and the progressive nature of global warming.
- This assessment also highlighted the importance of nesting sites or microclimate conditions for species persistence. For instance, preservation of nest sites is likely to be an important strategy for reptile species such as rattlesnakes and box turtles, which are known to exhibit high site fidelity. Microclimate is important for some birds, particularly for protection from extreme heat, and hibernation sites may be a limiting factor for western painted turtles, which require oxygenated soils (BISON-M 2009).
- Connectivity of habitats is critical for many species. At a large scale, landscapes sustain populations, and at smaller scales, connectivity allows for dispersal to new sites. Thus, actions that preserve large and continuous swaths of land benefits species conservation (Palmer and others 2009).

Wildland-Urban Interface

The interface between urban, suburban, and wild areas can exacerbate a number of climate change issues, including fire, disease, and invasive species. Fire ignition events increase in areas of high human activity (Busch and Smith 1993; Ellis 2000). The wildland-urban interface can increase disease exchange between wildlife and domestic animals. Exposure to pollution and human activities can reduce immune defenses in wildlife. Insect outbreaks are worsened by drought, late spring frosts, wind throw, and air pollution (Haack and Byler 1993). Changes in land use due to or in spite of the downstream hydrological consequences of warming temperatures will put additional pressure on species already experiencing habitat degradation and population declines. Land use change can threaten the persistence of species (e.g., agricultural encroachment and *Myotis thysanodes*; Rasheed and others 1995). The suggestions outlined in Tables 4.1 and 4.2 for managing species under climate change include actions that will often be in direct conflict with human land use.

The proximity of human populations to wildlands also increases public interest in management activities. Research and strategies that identify areas and species likely to be most negatively impacted by climate change can be used to conduct outreach activities and minimize conflicts between stakeholders with varying values of biodiversity. Stakeholder buy-in for strategies that aim to conserve or restore bosque habitat is critical to the success of riparian management, particularly when resources are limited. Local and regional buy-in can lead to greater participation and a sense of ownership, which increases success. In addition, stakeholder-driven strategies may increase effectiveness of management practices by supporting immediate action over verification processes (Chaney and others 1993). Within the Duck River of Idaho, fishermen, ranchers, and local residents formed a partnership that identified common goals and implemented actions that led to volunteer labor and fundraising efforts. The formation of a focused objective allowed this group to successfully restore riparian habitat based upon the desires of the group for improved habitat function in advance of results of studies and reports that may or may not have influenced management strategies. Success of restoration or management activities is also dependent on compliance with newly implemented regulations, which is more likely to happen with stakeholder participation in the planning processes. Chaney and others (1993) demonstrated that non-compliance by grazing permittees thwarted efforts to restore the Big Creek in Utah. A fundamental requirement for management of a multiuse, multijurisdictional resource as the MRGB is to develop a set of clearly designed objectives or desired future conditions that incorporates the needs of wildlife, ecosystems, and the values of stakeholders.

Grazing

Extensive and irreversible degradation of western riparian zones occurred in the late 19th and early 20th century due to severe overgrazing. Arguably, affected riparian zones in the Southwest have never recovered from this intense period of use. Grazing practices have changed since this time and many rangelands report improved conditions. However, the impact of grazing remains a significant issue for riparian areas and is likely to exacerbate the negative consequences of climate change. The negative consequences of grazing are many. Overgrazing of natural grasses reduces infiltration of rain water into the soil, thereby reducing ground water recharge and water availability for plants. Soil compaction and loss of vegetation also contribute to erosion and increased sediment loads in streams. Increased sediment loads along with reduced shade decreases water quality (Chaney and others 1993). Though data are scarce with respect to southwestern riparian systems, available information suggests that grazing has a large potential to negatively impact native vegetation and physical environment (Belsky and others 1999). Grazing by wildlife is also understudied but generally considered less detrimental to the system, probably because wildlife populations are not large enough to impact the system to the degree seen with concentrated livestock populations. Within most riparian systems, livestock has contributed to the widespread decline of native habitat and contributed to substantial erosion (Belsky and others 1999). Most studies of grazing in riparian areas show severe and widespread damage to native vegetation communities as well as increases in invasive species and erosion with a corresponding decrease in ecosystem resilience and recovery from fire and floods (Belsky and others 1999). Grazing has been associated with a shift away from habitat dominated by Fremont cottonwood toward salt-cedar (*Tamarix chinensis*) in the San Pedro River in southern Arizona (Stromberg 1997).

In case studies presented by Chaney and others (1993), limiting the access of livestock to riparian zones led to streambank stabilization and reduced erosion and generally improved riparian condition. With bank stabilization and reduced channelization, streamflow and water depth increased. Typically, as a result, vegetation returns, further increasing water depth, lowering water temperature, and improving fish habitat. Modified grazing practices may include exclusion of livestock from riparian stream banks by fencing, piping water to troughs, parceling upland areas, and implementing grazing rotations. These methods improved forage production, stock weight gain, and riparian habitat in Idaho (Chaney and others 1993). However, for other arid streams such as Mahogany Creek in Nevada, only full cessation of grazing activity allowed vegetation to recover. Within the Three Bar wildlife area in Arizona, cessation of grazing led to the development of a densely vegetation mix stand riparian habitat that improved conditions for both black bear and turkey.

However, grazing can have positive effects by increasing vegetation diversity, improving forage for early and mid-successional wildlife species, contributing to patchy habitat dynamics, opening up areas of dense vegetation, reducing invasive species, and stimulating plant growth (Holecheck and others 2004). In particular, livestock grazing has been associated with enhanced forage for elk by increasing the availability and quality of preferred grass species (Holechek and others 2004). Grazing in the northern Great Plains had a positive effect for native woodlands (Obedzinski and others 2001). In the Pacific Northwest, grazing contributed to a more open canopy, a desirable effect given efforts to reduce density to prevent insect outbreaks. Cattle grazing has also been associated with increased wildlife diversity in studies of New Mexico and Arizona (Chihuahuan desert, Nelson and others 1999; Desert grasslands, Bock and others 1984). Still, no studies indicate what level of grazing is appropriate for southwestern riparian systems, and overwhelmingly, evidence points toward a negative impact. We do know that a number of species reviewed in this assessment are negatively impacted by grazing. In particular, livestock grazing in riparian areas adversely affects bats (BISON-M) and has been known to reduce population of Botta's gophers as well as New Mexico meadow jumping mice (Frey and Malaney 2009). For some species and components of the riparian habitat, grazing may not be compatible, particularly where compaction of soils, increased bank erosions, and loss of vegetation cover destroys important habitat components.

Managers need to carefully determine locations and time periods where gazing is appropriate as well as sustainable stocking rates and density and annual condition and productivity of forage (Chaney and others 1993). These factors will change depending on range condition, soil type, and shifts in temperature and precipitation. The negative impacts of grazing, particularly by livestock, associated with habitat degradation are very likely to be exacerbated under climate warming. Sensitive habitats include those already degraded, areas with reduced water flow, areas with invasion problems, and areas important for emergent vegetation. Managers will have to carefully weigh potential benefits against the cost of this type of disturbance on the various habitats within the MRGB in light of climate change pressures. It is likely that implementing practices that minimize the negative consequences of climate change, such as preventing browsing on young native plants and preventing grazing altogether in ecologically sensitive areas, especially after restoration efforts, will become increasingly important in the future.

Legislative Actions

Without strong legal and administrative policies, efforts to restore or promote species conservation are limited. Further, restoration efforts often depend on or include measures to partner with local and regional stakeholders to conserve water and reduce impacts of human activities. Such collaborations require integrated strategies and common objectives. A review of legislative protection and mandates is beyond the scope of this assessment. However, a number of practices that will contribute to species conservation while building community and public involvement have been identified by Fullerton and Batts (2003):

- Purchase of water rights.
- Transfer existing development rights to riparian corridor to less sensitive areas outside of the riparian corridor or to locations with more resilient characteristics due to low disturbance and human activities.
- Conservation easements.
- Grazing management.
- Floodplain zoning ordinances.
- Incentive programs to reduce use of water (e.g., switch to low water use crops, retrofit irrigation systems to improve efficiency, and create mandates for municipalities).

Utilizing Opportunities

Not all species are going to be negatively impacted by climate change, and for some, such as those currently limited by minimum temperature thresholds, warming trends may favor establishment in new habitats. For instance, black phoebe populations are thought to be cold limited with New Mexico representing the northern extent of their current range (Wolf 1997), and could benefit from warmer winters. Middleton (1984) found that American goldfinches from warmer climates lived longer than those from colder climates, suggesting winter temperatures or storms are a factor in survival. This may also be true of the lesser goldfinch considered in this assessment. In addition, as noted in the Prioritization section, managers can take advantage of negative impacts expected for problematic species such as introduced predators or invasive plants.

Although the transition from one state and community assemblage to another has some ecological risk, there is some chance that we will see increased diversity in some regions and that the Southwest might be the new refuge for currently at-risk species. Our challenge in managing for climate change is to identify the unpermitable risks and areas where we can most effectively support our values while recognizing new opportunities for desirable outcomes. We must also evaluate the future value of current conservation sites across a broad range of species. It may be that species-rich bird communities are going to shift outside of conservation areas, but other species may arrive as they experience improved conditions.

Through assessment, managers can also identify novel opportunities in a changing climate to achieve management goals. For instance, invasive species exhibit a wide range of tolerance to fire (Grace and others 2000). Fires that are planned to coincide with sensitive periods in a noxious weed's lifecycle will improve the likelihood of a desired outcome. Different burn responses between invasive and native species may be exploited to reduce survival in undesirable species. For instance, fire may be an effective management tool for tamarisk if used for several consecutive years within the same area or if focused during periods when tamarisk is sensitive to fire impacts (Grace and others 2000). Fires, both prescribed and wild, may also improve conditions for native reptiles. Prescribed fire could improve habitat for the Great Plains skink (Fitch 1955) in areas where fossorial habits protect it from fire-related mortality (Western Governors' Drought Task Force). Increased fires may attract mule deer, which appear to prefer to forage in burned over unburned areas, although preferences vary seasonally (BISON-M 2009). Timing management efforts to maximize the effectiveness of invasive plant and animal control, identifying important roost sites, and constructing habitat elements (bat roost structures) in anticipation of potential new arrivals are possible strategies for maximizing the benefits of changing climates.

Conclusion

Changes in habitat resulting from climate change and the interaction of climate with other disturbances such as fire, hydrological changes associated with human activities, and continued development will determine the ultimate effect of climate change on species in the MRGB. Management actions must consider both small- and large-scale processes and couple place-based climate projections with estimates of population growth and development (Palmer and others 2009).

The goal of climate change vulnerability assessments is to predict how ecosystems and species will be affected by future conditions and identify management actions to mitigate negative impacts. Due to the large numbers of species identified as vulnerable and the expected exacerbation of current stressors, ongoing efforts to manage or restore riparian habitat and function will continue to be important. This MRGB assessment points to several focal issues, many already known, for species residing in and dependent upon the MRGB habitat. Climate-induced changes in habitat cover, particularly relating to increased fragmentation, direct effects of increased temperatures, and species sensitivity to disturbances such as fire and storms are likely to be very important drivers of species loss or persistence. In addition, we found a few examples of how climate change will influence species through changes in biotic interactions. This assessment allowed us to compare species vulnerability to climate change within the MRGB and identify common themes of impact and points for management intervention. More detailed knowledge of species life history and ecology are needed to improve predictions of species response, but even incomplete knowledge can inspire actions that are robust under uncertainty or that are unlikely to have negative consequences.

Once systems or species are determined to be at risk of decline under future warming scenarios, managers and planners must decide how to apply this information to their respective needs. Management actions may focus on the most vulnerable species, identify vulnerabilities of rare or keystone species, or determine common themes for a variety of species and create broad reaching actions. Results of this assessment point to the importance of considering both single species and a landscape-scale approach for the successful management of wildlife species within the MRGB. Managers should strive to maintain structurally diverse forested ecosystems and restore or create self-sustaining diverse riparian plant communities (see Crawford and others 1996). Proactive measures to reduce water loss, preserve functioning wetlands, and prevent further degradation and conversion of ecosystems are needed to limit the degree to which the MRGB is affected (Crawford and others 1996; Palmer and others 2009; CWSS 2006). Management strategies that aim to increase sustainable riparian systems will need to incorporate restoration, minimization of negative effects (e.g., grazing and water draw), and adaptive management strategies that consider multiple future scenarios.

Successful management under climate change will require us to recognize opportunities for improved management, continue our efforts to engage a diverse set of stakeholders, and continue to promote the value of biodiversity. To maximize effectiveness, future management plans will need to address not just average conditions but also years of high impact, such as those with very low flows or large fires. We must also remain flexible to changing conditions and new knowledge about species and alternative management strategies. Priorities in the Southwest may need to shift toward maximizing the establishment of species that will survive in a more xeric landscape or that allow for transitions in biotic communities. However, strategies that shift management focus to new natural states, as might occur where landscapes and habitat become unsuitable for species, require knowledge of how species respond to new conditions. This assessment is the first step towards providing information relevant to resource managers and other stakeholders and informing management decisions under climate change.

Literature Cited

Adams, R. A. and M. A. Hayes. 2008. Water availability and successful lactation by bats as related to climate change in arid regions of western North America. Journal of Animal Ecology 77: 1115-1121.

Allan J. F. and A. S. Flecker. 1993. Biodiversity conservation in running waters. BioScience 43: 32-43.

Allison, L. J., P. E. Brunkow, and J. P. Collins. 1994. Opportunistic breeding after summer rains by Arizona tiger salamanders. Great Basin Naturalist 54: 376-379.

Amphibiaweb: Information on amphibian biology and conservation. 2009. Berkeley, California. http://www.amphibiaweb.org.

Anderson, E. M. and M. J. Lovallo. 2003. Bobcat and Lynx. In: Wild Mammals of North America: biology, management, and conservation. 2nd ed. Feldhamer, G. A., Thompson, B. C., and J. A. Chapman (Eds). The John Hopkins University Press, MD, USA.

Annear, T., Chisholm, I., Beecher, H., [and others]. 2004. Instream flows for riverine resources, revised edition, Cheyenne, WY. 268 p.

Anthony, E. L. P., Stack M. H. and T. H. Kunz. 1981. Night roosting and the nocturnal time budget of the Little Brown Bat, *Myotis lucifugus*: Effects of reproductive status, prey density, and environmental conditions. Oecologia 51: 151-156.

Archer, S. 1994. Woody plant encroachment into southwestern grasslands and savannas: rates, patterns and proximate causes. Pages 13-68 in M. Vavra, W. A. Laycock, and R. D. Pieper (eds.), Ecological implications of livestock.

Armour, C., D. Duff, and W. Elmore. 1994. The effects of livestock grazing on western riparian and stream ecosystem. Fisheries 19: 9-12.

Atchley, M. C., A. G. de Soyza, and W. G. Whitford. 1999. Arroyo water storage and soil nutrients and their effects on gas-exchange of shrub species in the northern Chihuahuan Desert. Journal of Arid Environments 43: 21-33.

Bachmann, K. 1969. Temperature adaptations of amphibian embryos. American Naturalist 103: 115-130.

Bagne, K. E., M. M. Friggens, and D. M. Finch. 2011. A System for Assessing Vulnerability of Species (SAVS) to Climate Change. Gen. Tech. Rep. RMRS-GTR-257. Fort Collins, CO: U. S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 28 p.

Bahre, C. J. and M. L. Shelton. 1993. Historic vegetation change, mesquite increases, and climate in southeastern Arizona. Journal of Biogeography 20: 489-504.

Bailey, R. G. 1988. Ecogeographic analysis: a guide to the ecological division of land for resource management. Miscellaneous Publication No. 1465. US Forest Service, Washington, D.C. 16 p.

Bailey, R. G. 1995. Description of the ecoregions of the United States. 2nd edition revised and expanded (1st edition 1980). Miscellaneous Publication No. 1391. US Forest Service, Washington, D.C. 108 p., with separate map at 1:7,500,000.

Bailey, R. G. 1998. Ecoregions map of North America: explanatory note. Miscellaneous.

Baker, Jr., M. B., P. F. Ffolliott, L. F. DeBano, and D. G. Neary, 2004. Riparian Areas of the Southwestern United Sates. Lewis Publishers, Boca Raton, Florida, 408 p.

Baker, R. H. and K. A. Shump, Jr. 1978. *Sigmodon fulviventer*. Mammalian Species 94: 1-4.

Baker, R. J. and D. L. Spencer. 1965. Late Fall reproduction in the desert shrew. Journal of Mammalogy 46: 330.

Baltosser, W. H. and S. M. Russell. 2000. Black-chinned Hummingbird (*Archilochus alexandri*). In The Birds of North America, No. 495 (A. Poole and F. Gill, Eds.) The Birds of North America, Inc. Philadelphia, PA.

Barbault, R. and M-E. Maury. 1981. Ecological organization of a Chihuahuan desert lizard community. Oecologia 51: 335-342.

Barnett, T. P., D. W. Pierce, H. G. Hidalgo, C. Bonfils, B. D. Santer, T. Das, G. Bala, A. W. Wood, T. Nozawa, A. A. Mirin, D. R. Cayan, and M. D. Dettinger. 2008. Human-Induced Changes in the Hydrology of the western United States. Science 2319: 1080-1083.

Bateman, H., M. Harner, and A. Chung-MacCoubrey. 2008. Abundance and reproduction of toads (*Bufo*) along a regulated river in the southwestern United States: Importance of flooding in riparian ecosystems. Journal of Arid Environments 72: 1613-1619.

Beier, P. 1993. Determining minimum habitat areas and habitat corridors for cougars. Conservation Biology 7: 94-108.

Beissinger, Steven R., J. Michael Reed, Joseph M. Wunderle, Jr., Scott K. Robinson, and Deborah M. Finch. 2000. Report of the AOU Conservation Committee on the Partners in Flight species prioritization plan. Auk 117: 549-561.

Belsky, A. J., A. Matzke, and S. Uselman. 1999. Survey of Livestock Influences on Stream and Riparian Ecosystems in the Western United States. Journal of Soil and Water Conservation 54: 419-431.

Bemis, C. and J. D. Rising. 1999. Western Wood-Pewee (*Contopus sordidulus*). In The Birds of North America, No. 451 (A. Poole and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA.

Bender, D. J., D. L. Gummer, S. Robertson, A. Tuecher, P. Knaga, E. Baird, and E. Jochum. 2005. Conservation management of Ord's kangaroo rats and sandy habitats of the Middle Sand Hills of Alberta. Report for the Canadian Forces Base Suffield, Medicine Hat, Alberta. 33 p.

Best, T. L. 1996. *Lepus californicus*. Mammalian Species 530: 1-10.

Bhattacharjee, J., J. P. Taylor, Jr., and L. M. Smith. 2006. Controlled flooding and staged drawdown for restoration of native cottonwoods in the Middle Rio Grande Valley, New Mexico, USA. Wetlands 26: 691-702.

Blancher, P. J. and R. J. Robertson. 1985. Predation in relation to spacing of kingbird nests. Auk 102: 654-658.

Boag, P., and P. Grant. 1984. The classical case of character release—Darwin's finches (*Geospiza*) on Isla Daphne Major. Galapagos Biological Journal of the Linnean Society 22: 243-287.

Bock, C. E., J. H. Bock, W. R. Kenny, and V. M. Hawthorne. 1984. Responses of birds, rodents, and vegetation to livestock exclosure in a semidesert grassland site. Journal of Range Management 37: 239-242.

Bock, C. E., V. A. Saab, T. D. Rich, and D. S. Dobkin. 1993. Effects of livestock grazing on neotropical migratory land birds in western North America. In D. M. Finch, P. W. Stangel (eds.), Status and management of neotropical migratory birds. U.S. Forest Service, Gen. Tech. Rep. RM-229.

Bombay, H. L., M. L. Morrison, L. S. Hall. 2003. Scale perspectives in habitat selection and animal performance for willow flycatchers (*Empidonax traillii*) in the central Sierra Nevada, California.Studies in Avian Biology 26: 60-72.

Bonter, D. N. and W. M. Hochalka. 2003. Declines of chickadees and corvids: Possible impacts of West Nile Virus. American Birds: 2002-2203 pp. 22-25.

Both, Christiaan, and Luc te Marvelde. 2007. Climate change and timing of avian breeding and migration throughout Europe. Climate Research 35: 93-105.

Bradley, G. A., P. C. Rosen, M. J. Sredl, T. R. Jones, and J. E. Longcore. 2002. Chytridiomycosis in Native Arizona Frogs. Journal of Wildlife Diseases 38:206–212.

Brooks R. J., M. L. Bobyn, D. A. Galbraith, J. A. Layfield, and E. G. Nancekivell. 1991. Maternal and environmental influences on growth and survival of embryonic and hatching snapping turtles (*Chelydra serpentina*). Canadian Journal of Zoology 69: 2667-2676.

Brown, C. R. and M. B. Brown. 1996. Coloniality in the Cliff Swallow: the effect of group size on social behavior. University of Chicago Press, Chicago.

Brown, Charles R. and Mary B. Brown. 1995. Cliff Swallow (*Petrochelidon pyrrhonota*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/149.

Brown, Charles R. and Mary Bomberger Brown. 1999. Barn Swallow (*Hirundo rustica*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/452

Brown, D. E., F. Reichenbacher, S. E. Franson. 1998. A classification of North American biotic communities. University of Utah Press, Salt Lake City. 141 p.

Brush, T., B. W. Anderson and R. D. Ohmart. 1990. Habitat use by cavity-nesting birds of desert riparian woodland. Pp. 191-198 in Managing wildlife in the Southwest symposium (P. R. Krausman and N. S. Smith, eds.). Arizona Chapter of the Wildlife Society, Phoenix.

Burquez-Montijo, A., M. Miller, and A. Martinez-Yrizar. 2002. Mexican grasslands, thornscrub, and the transformation of the Sonoran desert by invasive exotic buffelgrass. Pages 126-146 In B. Tellman, ed., Arizona-Sonoran Desert Museum Studies in Natural History, Tucson, AZ.

Butler, C. J. 2003. The disproportionate effect of global warming on the arrival dates of short-distance migratory birds in North America. Ibis 145: 484-495.

Cabe, Paul R. 1993. European Starling (*Sturnus vulgaris*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu.bnaproxy.birds.cornell.edu/bna/species/048.

Cagle K. D., G. C. Packard, K. Miller, and M. J. Packard. 1993. Effects of the microclimate in natural nests on development of embryonic painted turtles, *Chrysemys picta*. Functional Ecology 7: 653-660.

Cagle K. D., G. C. Packard, K. Miller, and M. J. Packard. 1993.Effects of the microclimate in natural nests on development of embryonic painted turtles, *Chrysemys picta*. Functional Ecology 7: 653-660.

Capinera, J. L., and D. R. Horton. 1989. Geographic Variation in Effects of Weather on Grasshopper Infestation. Environmental Entomology 18: 8-14.

Cardiff, S. W., and D. L. Dittmann. 2002. Ash-throated Flycatcher (*Myiarchus cinerascens*). In The Birds of North America, No. 664 (A. Poole and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA.

Carey, C., N. Cohen, and L. Rollins-Smith. 1999. Amphibian declines: an immunological perspective. Developmental & Comparative Immunology 23: 459-472.

Carpenter, R. E. 1969. Structure and function of the kidney and the water balance of desert bats. Physiological Zoology 42: 288-301.

Casey, D. and D. Hein. 1983. Effects of heavy browsing on a bird community in deciduous forest. Journal of Wildlife Management 47: 829-836.

Catron, J. E., Lightfoot, D. C., Mygatt, J. E., Brantley, S. L. and T. K. Lowrey. 2008. A field guide to the plants and animals of the Middle Rio Grande Bosque. University of New Mexico Press, Albuquerque, NM. 375 p.

Cayan, D. R., Dettinger, M. D., Diaz, H. F., and Graham, N. E. 1998. Decadal variability of precipitation over Western North America. Journal of Climate 11: 3148-3166.

Cayan, D. R., Karrerdiener, S. A., Dettinger, M. D., Caprio, J. M., Peterson, D. H. 2001. Changes in the onset of spring in the western United States. Bulletin of the American Meteorological Society 82: 399-415.

Chaney, E., Elmore, W., and W. S. Platts. 1990. Livestock grazing on western riparian areas. Report produced for the U.S. Environmental Protection Agency by the Northwest Resource Information Center, Inc., Eagle, ID. Denver, CO: U.S. Environmental Protection Agency. 45 p.

Chaney, Ed; Elmore, Wayne; Platts, William S. 1993. Managing change: livestock grazing on western riparian areas. Report produced for the U.S. Environmental Protection Agency by the Northwest Resource Information Center, Inc. Eagle, ID. Denver, CO: U.S. Environmental Protection Agency. 31 p.

Chu, M. and G. Walsberg. 1999. Phainopepla (*Phainopepla nitens*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/415.

Chung-MacCoubrey, A. L. 2005. Use of pinyon-juniper woodlands by bats in New Mexico. Forest Ecology and Management 204: 209-220.

Comprehensive Wildlife Conservation Strategy for New Mexico [CWCS]. 2006. New Mexico Department of Game and Fish. Santa Fe, New Mexico. 526 pp. + appendices. Available: http://fws-mcfwru.nmsu.edu/cwcs/New_Mexico_CWCS.htm.

Conine, K. H., B. W. Anderson, R. D. Ohmart, and J. F. Drake. 1979. Responses of riparian species to agricultural habitat conversions. Pp. 248-262 in Strategies for protection and management of floodplain wetlands and other riparian ecosystems (R. R. Johnson and J. F. McCormick, tech. coords.). U.S. Department of Agriculture, Forest Service. Gen. Tech. Rep. WO-12.

Committee on the Status of Endangered Wildlife in Canada [COSEWIC]. 2006. COSEWIC assessment and status report on the Western Painted Turtle *Chrysemys picta bellii* (Pacific Coast population, Intermountain-Rocky Mountain population and Prairie/Western Boreal–Canadian Shield population) in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa. vii + 40 pp. (www.sararegistry.gc.ca/status/status_e.cfm).

Crawford, C. S., Ellis, L. M., Shaw, D. and N. E. Umbreit. 1999. Restoration and monitoring in the middle Rio Grande Bosque: Current status of flood pulse related efforts. USDA Forest Service Proceedings RMRS-P-7: 158-163.

Crawford, C. S., Cully, A. C., Leutheuser, R., Sifuentes, M. S., White, L. H., and J. P. Wilber. 1993. Middle Rio Grande ecosystem: bosque biological management plan. Report submitted to U.S. Fish and Wildlife Service, Albuquerque, New Mexico.

Crimmins, M. 2006. Synoptic climatology of extreme fire-weather conditions across the Southwest United States. International Journal of Climatology 26: 1001-1016.

Degenhardt, Painter, and Price. 1996. Amphibians and Reptiles of New Mexico. University of New Mexico Press, Albuquerque, NM. 431 p.

Delmas, V., Bonnet, X., Girondot, M., Prèvot-Julliard, A-C. 2008. Varying hydric conditions during incubation influence eggs water exchange in hatchling phenotype in the red-eared slider turtle. Physiological and Biochemical Zoology 81: 345-355.

Diller, L. V. and R. L. Wallace. 1996. Comparative ecology of two snake species (*Crotalus viridis* and *Pituophis melanoleucus*) in Southwestern Idaho. Herpetologica 52: 434-360.

Dixon, M. D., J. C. Stromberg, J. Price, H. Galbraith, A. Fremier, and E. Larsen. In review. Climate change and riparian vegetation response. Chapter 8, In: Stromberg, J. and B. Tellman (eds.). Ecology and Conservation of Desert Riparian Ecosystems: The San Pedro River Example. University of Arizona Press, Tucson, AZ.

Dixon, M. D., J. C. Stromberg, J. T. Price, H. Galbraith, A. K. Fremier, and E. W. Larsen. 2009. Potential Effects of Climate Change on the Upper San Pedro Riparian Ecosystem. In: Ecology and Conservation of the San Pedro River, J. C. Stromberg and B. J. Tellman (Editors). University of Arizona Press, Tucson, Arizona, pp. 57-72.

Dwire, K. A. and J. B. Kauffman. 2003. Fire and Riparian Ecosystems in Landscapes of the Western USA. Forest Ecology and Management 178: 61-74.

Easterling, D., G. Meehl, C. Parmesan, S. Changnon, T. Karl, and L. Mearns. 2000. Climate extremes: Observations, modeling, and impacts. Science 289: 2068-2074.

Eckerle, K. P. and C. F. Thompson. 2001. Yellow-breasted Chat (*Icteria virens*). In The Birds of North America, No. 575 (A. Poole and F. Gill., Eds.). The Birds of North America, Inc., Philadelphia, PA.

Ellis, L. M. 1995. Bird use of saltcedar and cottonwood vegetation in the Middle Rio Grande Valley of New Mexico, U.S.A. Journal of Arid Environments 30: 339-349.

Emanuel, K. 2005. Increasing destructiveness of tropical cyclones over the past 30 years. Nature 436: 686-688.

Enquist, C., and D. Gori. 2008. Implications of recent climate change on conservation priorities in New Mexico. Climate Change Ecology and Adaptation Program, The Nature Conservancy, New Mexico.

Errington, P. L. 1939. Reaction of Muskrat Populations to Drought. Ecology 20: 168-186.

Field, C. B., L. D. Mortsch, M. Brklacich, D. L. Forbes, P. Kovacs, J. A. Patz, S. W. Running and M. J. Scott. 2007. Chapter 14 : North America. Pages 617-652 in M. Parry, O. Costanza, J. Palutikoff, P. J. van der Linden and C. E. Hanson (eds.), Climate Change 2007: Impacts, Adaptation and Vulnerability, Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, Cambridge University Press, Cambridge, UK.

Findley, J. S. 1987. The Natural History of New Mexican Mammals. University of New Mexico Press, Albuquerque, pp. 96-97.

Finkler, M. S. 1999. Influence of water availability during incubation on hatchling size, body composition, desiccation tolerance, and terrestrial locomotor performance in the snapping turtle *Chelydra serpentina*. Physiological and Biochemical Zoology 72: 714-722.

Fire Effects Information System [Online] (1996, September). Prescribed Fire and Fire Effects Research Work Unit, Rocky Mountain Research Station (producer). Available: www.fs.fed.us/database/feis/.

Fischlin, A., Midgely, G. F., Price, J. T., Leemans, R. and others 2007. Chapter 4: Ecosystems, their properties, goods and services In Climate Change 2007: Impacts, adaptation, vulnerability. Contribution of working group II to the fourth assessment report of the Intergovernmental Panel on Climate Change (IPCC). Cambridge University Press, Cambridge.

Fitch H. S. 1955. Habits and Adaptations of the Great Plains Skink (*Eumeces obsoletus*). Ecological Monographs 25: 59-8.

Flinders, J. T., and J. A. Chapman. 2003. *Lepus californicus*. In Wild Mammals of North America: Biology, management, and Conservation. Feldhamer, G. A., Thompson, B. C. and J. A. Chapman (Eds). Johns Hopkins University Press, Baltimore, MD. 1197 p.

Foden, W., G. Mace, J.-C. Vie, A. Angulo, S. Butchart, L. DeVantier, H. Dublin, A. Gutsche, S. Stuart, and E. Turak. 2008. Species susceptibility to climate change impacts. In: J.-C. Vie, C. Hilton-Taylor, and S. Stuart (eds.). The 2008 Review of the IUCN Red List of Threatened Species. IUCN Gland, Switzerland.

Ford, N. B., and J. P. Karges. 1987. Reproduction in the checkered garter snake, *Thamnophis maricanus*, from southern Texas and northeastern Mexico: seasonality and evidence for multiple clutches. The Southwestern Naturalist 32: 93-101.

Frey, J. K. 2005. Status assessment of montane populations of the New Mexico meadow jumping mouse (*Zapus hudsonius luteus*) in New Mexico. Santa Fe, Final report submitted to New Mexico Department of Game and Fish: 74 + appendices on CD.

Frey, J. K. and J. L. Malaney. 2009. Decline of the meadow jumping mouse (*Zapus hudsonius luteus*) in two mountain ranges in New Mexico. The Southwestern Naturalist 54: 31-44.

Friend, M. and J. C. Franson, tech. eds. 1999. Field Manual of Wildlife Diseases: General Field Procedures and Diseases in Birds. U.S. Geological Survey, Washington, D.C.

Fritzell, E. K., and K. J. Haroldson. 1982. *Urocyon cinereaoargenteus*. Mammalian Species 189: 1-8.

Fullerton, W., and D. Batts. 2003. A Framework for a restoration vision for the Rio Grande: Hope for a living river. The Alliance for Rio Grande Heritage, World Wildlife Fund. 142 p.

Gamble, L. R. and T. M. Bergin. 1996. Western Kingbird (*Tyrannus verticalis*). In The Birds of North America, No. 227 (A. Poole and F. Gill, eds.). The Academy of Natural Sciences, Philadelphia, PA and The American Ornithologists' Union, Washington, D.C.

Garfin, G. and M. Lenart. 2007. Climate change effects on Southwest water resources. Southwestern Hydrology 6(1):16-17, 34.

Garrison, Barrett A. 1999. Bank Swallow (*Riparia riparia*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/414.

Geluso, K. 2007. Winter activity of bats over water and along flyways in New Mexico. The Southwestern Naturalist 52: 482-492.

George, T. Luke, Ada C. Fowler, Richard L. Knight and Lowell C. McEwen. 1992. Impacts of a Severe Drought on Grassland Birds in Western North Dakota. Ecological Applications 2: 275-284.

Gibbons, J. W., Scott, D. E., Ryan. T. J., Buhlmann, K. A., Tuberville, T. D., Metts, B. S., Greene, J. L., Mills, T., Leiden, Y., Poppy, S. and C. T. Winne. 2000. The global decline of reptiles, déjà vu amphibians. BioScience 50: 653-666.

Gilbert, M., R. Leclair, Jr., R. Fortin. 1994. Reproduction of the Northern Leopard Frog (*Rana pipiens*) in Floodplain Habitat in the Richelieu River, P. Quebec, Canada. Journal of Herpetology 28: 465-470.

Gitay, H., Suárez, A., and Watson, R. T. 2002. Climate change and Biodiversity. IPCC technical paper, Intergovernmental Panel on Climate Change, Geneve 77 p.

Glinski, R. L. and R. D. Ohmart. 1984. Factors of Reproduction and Population Densities in the Apache Cicada (*Diceroprocta apache*). The Southwestern Naturalist 29: 73-79.

Goguen, C. B., and N. E. Mathews. 2001. Brown-headed cowbird behavior and movements in relation to livestock grazing. Ecological Applications 11: 1533-1544.

Grace, J. B., Smith, M. D., Grace, S. L., Collins, S. L. and T. J. Stohlgren. 2002. Interaction between fire and invasive plants in the temperate grassland of North America. Pages 40-65. In K. E. M. Galley and T. P. Wilson (eds.). Proceedings of the Invasive Species Workshop: the Role of Fire in the Control and Spread of Invasive Species. Fire Conference 2000: the First National Congress on Fire Ecology, Prevention, and Management. Miscellaneous Publication No.11, Tall Timbers Research Station, Tallahassee, FL.

Grubb, Jr., T. C. and V. V. Pravosudov. 2008. White-breasted Nuthatch (*Sitta carolinensis*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; http://bna.birds.cornell.edu/bna/species/054.

Guzy, M. J., and G. Ritchison. 1999. Common Yellowthroat (*Geothylypis trichas*). In The Birds of North America, No. 448 (A. Poole and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA.

Haack, R. A. and J. W. Byler. 1993. Insects and pathogens: regulators of forested ecosystems. Journal of Forestry 91: 32-37.

Hardy, P. C., M. L. Morrison and R. X. Barry. 1999. Abundance and habitat associations of Elf Owls and Western Screech-Owls in the Sonoran Desert. Southwestern Naturalist 44: 311-323.

Hart, J. S. 1971. Rodents In Comparative Physiology of thermoregulation. Vol II Mammals Whittow (ed.) Academic Press, Inc., New York. Pp. 1-147.

Heath, James E. and Peter J. Wilkin. 1970. Temperature Responses of the Desert Cicada, *Diceroprocta apache* (Homoptera, Cicadidae). Physiological Zoology 43: 145-154.

Heller, N. E., Sanders, N. J., Shors, J. W. and D. M. Gordon. 2007. Rainfall facilitates the spread, and time alters the impact of the invasive Argentine ant. Oecologia 155: 385-95.

Hermanson, J. W. and T. J. O'Shea. 1983. *Antrozous pallidus*. Mammalian Species 213: 1-8.

Hill, G. E. 1995. Black-headed Grosbeak (*Pheucticus melanocephalus*). In The Birds of North America, No. 143 (A. Poole and F. Gill, eds.). The Academy of Natural Sciences, Philadelphia and The American Ornithologists' Union, Washington, D.C.

Hink, V.C., and R. D. Ohmart. 1984. Middle Rio Grande Biological Survey. U.S. Army Corps of Engineers, Albuquerque District, New Mexico. Contract No. DACW47-81-C-0015, Arizona State University.

Hogg, I. D., and D. D. Williams. 1996. Response of stream invertebrates to a global warming thermal regime: an ecosystem-level manipulation. Ecology 77: 395-407.

Holecheck, J. L., T. T. Baker, and J. C. Boren. 2004. Impacts of controlled grazing versus grazing exclusion on rangeland ecosystems: what we have learned. New Mexico State University Cooperative Extension Service, Range Improvement Task Force Report 57. Las Cruces, New Mexico. 42 p.

Holechek, J. L., R. D. Pieper, and C. H. Herbel. 1998. Range management: principles and practices, third edition. Prentice Hall, New Jersey.

Howe, W. H. 1986. Status of the Yellow-billed Cuckoo (*Coccyzus americanus*) in New Mexico. Rep. 516.6-75-09. New Mexico Department of Game and Fish, Santa Fe.

Howe, William H. and Fritz L. Knopf. 1991. On the Imminent Decline of Rio Grande Cottonwoods in Central New Mexico. The Southwestern Naturalist 36: 218-224.

Hoylman, A. M., A. Peterson, J. V. H. Constable, J. B. Picone, J. T. Ball. 1998. Climate Change and Wetland Processes in the Southwest United States: USBR Cooperative Agreement 2-FC-30-O0200 Final Report Response of Riparian Communities to Rising CO_2 Levels. Available from: http://www.osti.gov/bridge/servlets/purl/762877-CyAGPg/webviewable/762877.pdf.

Huey, R. B. and J. J. Tewksbury. 2009. Can behavior douse the fire of climate warming? Proceedings of the National Academy of Sciences 106: 3647-3648.

Hughes, J. M. 1999. Yellow-billed Cuckoo (*Coccyzus americanus*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/418.

Hultine, K. R., J. Belnap, C. van Riper, J. R. Ehleringer, P. E. Dennison, M. E. Lee, P. L. Nagler, K. A. Snyder, S. M. Uselman, and J. B. West. 2010. Tamarisk biocontrol in the western United States: ecological and societal implications. Frontiers in Ecology and the Environment 8: 467-474.

Hunter, W. C., R. D. Ohmart and B. W. Anderson. 1988. Use of Exotic Saltcedar (*Tamarix chinensis*) by Birds in Arid Riparian Systems. The Condor 90: 113-123.

Hurd, Brian H. and Julie Coonrod. 2008. Climate change risks New Mexico's waterways: its byways and its flyways. Water Resources IMPACT 10(4).

Ingold, J. 1993. Blue Grosbeak (*Guiraca caerulea*). In The Birds of North America, No. 79 (A. Poole and F. Gill, Eds.). The Academy of Natural Sciences and the American Ornithologists' Union, Philadelphia, PA.

Inkley, D. B., Anderson, M. G., Blaustein, A. R., Burkett, V. R., Felzer, B. and others. 2004. Global climate change and wildlife in North America. Wildlife Society Technical Review 04-2. The Wildlife Society, Bethesda, Maryland, USA 26 p.

Intersecretarial Commission on Climate Change. 2007. National Strategy on Climate Change: Mexico. Available at: http://www.un.org/ga/president/61/follow-up/climatechange/Nal_Strategy_MEX_eng.pdf.

Intergovernmental Panel on Climate Change [IPCC]. 2000. Special Report on Emissions Scenarios: A special report of Working Group III of the Intergovernmental Panel on Climate Change. Nakićenović, N., and Swart, R. and others. Cambridge University Press, Cambridge, UK.

Intergovernmental Panel on Climate Change [IPCC]. 2007. Climate change 2007: the physical science basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Soloman, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K. B. Averyt, M. Tignor, and H. L. Miller (eds.). Cambridge University Press, Cambridge, UK. 996 p.

Irwin, K. 1985. Foraging ecology and reproduction of the Black Phoebe in Humboldt County. Thesis. California State University, Humboldt.

Jackson, J. A. and H. R. Ouellet. 2002. Downy Woodpecker (*Picoides pubescens*). In The Birds of North America, No. 613 (A. Poole and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA.

Jackson, J. A., H. R. Ouellet and B. J. Jackson. 2002. Hairy Woodpecker (*Picoides villosus*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu.bnaproxy.birds.cornell.edu/bna/species/702.

Janzen, F. J. 1994. Climate change and temperature-dependent sex determination in reptiles. Proceedings of the National Academy of Sciences 91: 7487-7490.

Johnson, K., M. Patricia, C. Black and K. Score. 1999. Reproductive failure of endangered southwestern willow flycatchers on the Rio Grande, New Mexico. The Southwestern Naturalist 44: 226-231.

Johnson, W. C., M. D. Dixon, R. Simons, S. Jenson, and K. Larson. 1995. Mapping the response of riparian vegetation to possible flow reductions in the Snake River, Idaho. Geomorphology 13: 159-173.

Johnston, D. W. and T. P. Haines 1957. Analysis of Mass Bird Mortality in October, 1954. The Auk 74: 447-458.

Jones, C. A. and C. N. Baxter. 2004. *Thomomys bottae*. Mammalian Species 742: 1-14.

Jones, H. L. and J. M. Diamond. 1976. Short-time-base studies of turnover in breeding bird populations on the California Channel Islands. Condor 78: 526-549.

Julius, S. H., and J. M. West. 2007. Preliminary review of adaptation options for climate sensitive ecosystems and resources. U.S. Climate Change Science Program. Synthesis and Assessment Product 4.4.

Justus, J., Sandomir, M., Urquhart, T., and B. Ewan. 1977. Developmental rates of two species of toads for the desert southwest. Copeia 3: 592-594.

Kennedy, E. D. and D. H. White. 1997. Bewick's Wren (*Thryomanes bewickii*). In The Birds of North America, No. 315 (A. Poole and F. Gill, eds.). The Academy of Natural Sciences, Philadelphia, PA and The American Ornithologists Union, Washington, D.C.

Kennedy, E. D. and D. W. White. 1996. Interference Competition from House Wrens as a Factor in the Decline of Bewick's Wrens. Conservation Biology, Vol. 10, No. 1: 281-284.

Kilham, L. 1971. Roosting Habits of White-Breasted Nuthatches. Condor 73: 113-114.

King, O. 1960. Observations on Oklahoma toads. Southwestern Naturalist 5: 102-103.

King, T. L., Switzer, J. F., Morrison, C. L., Eackles, M. S., Young, C. C., Lubinski, B. A., and P. Cryan. 2006. Comprehensive genetic analyses reveal evolutionary distinction of a mouse (*Zapus hudsonius preblei*) proposed for delisting from the US endangered species Act. Molecular Ecology 15: 4331-4359.

Klimkiewicz, M. and A. Futcher. 1987. Longevity records of North American birds: Coerebinae through Estrildidae. Journal of Field Ornithology 58: 318-333.

Knapton, R. W. 1988. Nesting Success is Higher for Polygamously Mated Females Than for Monogamously Mated Females in the Eastern Meadowlark. The Auk 105: 325-329.

Kocan, R. M. and S. R. Amend. 1972. Immunologic status of Mourning Doves following an epizootic of trichomoniasis. Journal of Wildlife Diseases 8: 176-180.

Kranjcec, J., J. M. Mahoney, and S. B. Rood. 1998. The responses of three riparian cottonwood species to water table decline. Forest Ecology and Management 110: 77-87.

Kroodsma, D. E. 1974. Song learning, dialects, and dispersal in the Bewick's Wren. Z. Tierpsychologie 35: 352-380.

Krupa, J. J. 1994. Breeding biology of the Great Plains toad in Oklahoma. Journal of Herpetology 28: 217-224.

Krzysik, A. J. 1990. Biodiversity in riparian communities and watershed management. In R. E. Riggins, E. B. Jones, R. Singh, and P. A. Rechard (eds.), Watershed planning and analysis in action. Proceedings of the symposium held at Durango, Colorado, July 9-11, 1990. American Society of Civil Engineers, New York.

Kupferberg, S. 1997. Bullfrog (*Rana catesbeiana*) invasion of a California river: the role of larval competition. Ecology 78: 1736-1751.

Kurta, A. 1995. Mammals of the great lakes region: revised edition. The University of Michigan Press, U.S.A.

Kurta, T. H., and R. H. Baker. 1990. *Eptesicus fuscus*. Mammalian Species No. 356: 1-10. American Society of Mammalogists 1-7.

Laakkonen, J., Fisher, R. N., and T. J. Case. 2003. Factors influencing the variation in capture rates of shrews in southern California, USA. Acta Theriologica 48: 157-166.

Laakkonen, J., Fisher, R. N., and T. J. Case. 2001. Effect of land cover, habitat fragmentation and ant colonies on the distribution and abundance of shrews in southern California. Journal of Animal Ecology 70: 776-788.

Lackey, J. A., Huckaby, D. G., and B. G. Ormiston. 1985. *Peromyscus leucopus*. Mammalian Species 247: 1-10.

Lacki, M. J., Johnson, J. S., Dodd, E. D. and M. D. Baker. 2007. Prey consumption of insectivorous bats in coniferous forest of North-Central Idaho. Northwest Science 81: 199-205.

Ladeau, S. L., P. P. Marra, A. M. Kilpatrick and C. A. Calder. 2008. West Nile Virus Revisited: Consequences for North American Ecology. Bioscience: 937-946.

Landry, C. 1998. Market transfers of water for environmental protection in the western United States. Water Policy 1: 457-469.

Lanyon, W. E. 1963. Experiments on species discrimination in *Myiarchus* flycatchers. American Museum Novitates 2126.

Larivière, S. 2001. *Ursus americanus*. Mammalian Species 647: 1-11.

Larivière, S. and Walton, L. R. 1997. *Lynx rufus*. Mammalian Species 563: 1-8.

Lasiewski, R. C. 1964. Body temperatures, heart and breathing rate, and evaporative water loss in hummingbirds. Physiological Zoology 37: 212-223.

Laskey, A. R. 1946. Some Bewick Wren nesting data. Migrant 17: 39-43.

Laymon, S. A. and M. D. Halterman. 1987. Can the western subspecies of the yellow-billed cuckoo be saved from extinction? Western Birds 18: 19-25.

Leopold, B. D. and P. R. Krausman. 1991. Factors influencing desert mule deer distribution and productivity in Southwestern Texas. Southwestern Naturalist 36: 67-74.

Leuth, F. X. 1941. Effects of temperature on snakes. Copeia 3: 125-132.

Lewis, S. E. 1993. Effect of climatic variation on reproduction by pallid bats (*Antrozous pallidus*). Canadian Journal of Zoology 71: 1429-1433.

Lewis, S. E. 1996. Low roost-site fidelity in pallid bats: associated factors and effect on group stability. Behavioral Ecology and Sociobiology 39: 335-344.

Li, P., and T. Martin. 1991. Nest-site selection and nesting success of cavity-nesting birds in high elevation forest drainages. Auk 108: 405-418.

Ligon, J. D. 1969. Some aspects of temperature relations in small owls. Auk 86: 458-472.

Liknes, E. T. and D. L. Swanson. 1996. Seasonal variation in cold tolerance, basal metabolic rate, and maximal capacity for thermogenesis in White-breasted Nuthatches *Sitta carolinensis* and Downy Woodpeckers *Picoides pubescens*, two unrelated arboreal temperate residents. Journal of Avian Biology 27(4): 279-288.

Lowther, P. E. 1993. Brown-headed Cowbird (*Molothus ater*). In The Birds of North America, No. 47 (A. Poole and F. Gill, Eds.). Philadelphia: The Academy of Natural Sciences; Washington, D.C.: The American Ornithologists Union.

Macèdo, R. H. and M. A. Mares. 1988. *Neotoma albigula*. Mammalian Species 310: 1-7.

Magrin, G., C. Gay García, D. Cruz Choque, J. C. Giménez, A. R. Moreno, G. J. Nagy, C. Nobre and A. Villamizar. 2007. Latin America. Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, M. L. Parry, O. F. Canziani, J. P. Palutikof, P. J. van der Linden and C. E. Hanson, Eds., Cambridge University Press, Cambridge, UK 581-615.

Martin, T. E. 1987. Food as a limit on breeding birds: a life-history perspective. Annual Review of Ecology and Systematics: 18: 453-487.

Mathews, W. J. and E. G. Zimmerman. 1990. Potential effects of global warming on native fishes on the southern Great Plains and Southwest. Fisheries 15: 26-32.

Maurer, E. P., L. Brekke, T. Pruitt, and P. B. Duffy. 2007. Fine-resolution climate projections enhance regional climate change impact studies. Eos Transactions American Geophysical Union 88: 504.

Mawdsley, Jonathan R., Robin O'Malley, and Dennis S. Ojima. 2009. A review of climate-change adaptation Strategies for Wildlife Management and Biodiversity Conservation. Conservation Biology 5: 1080-1089.

McAda, D. P. and Barroll, P. 2002. Simulation of ground-water flow in the Middle Rio Grande Basin between Cochiti and San Acacia, New Mexico. U.S. Geological Survey Water Resources Investigations Report 02-4200.

McCabe, G. J. and D. M. Wolock. 1999. General-circulation-model simulations of future snowpack in the western United States. Journal of the American Water Resources Association 35(6): 1473-1484.

McCain, C. M. 2007. Could temperature and water availability drive elevational species richness patterns? A global case study for bats. Global Ecology and Biogeography 16: 1-13.

McCarty, J. 2001. Ecological consequences of recent climate change. Conservation Biology 15: 320-331.

Memmott, J., P. G. Craze, N. M. Waser, and M. V. Price. 2007. Global warming and the disruption of plant–pollinator interactions. Ecology Letters 10: 710-717.

Merritt, D. M. and D. J. Cooper. 2000. Riparian vegetation and channel change in response to river regulation: a comparative study of regulated and unregulated streams in the Green River Basin, USA. Regulated Rivers: Research and Management 16: 543-564.

Milly, P. C. D., K. A. Dunne, and A. V. Vecchia. 2005. Global patterns of trends in streamflow and water availability in a changing climate. Nature 438: 347-350.

Mitchell, D. L., D. Ivanova, R. Rabin, K. Redmond, and T. J. Brown. 2002. Gulf of California sea surface temperatures and the North American monsoon: Mechanistic implications from observations. Journal of Climate 15: 2261-2281.

Mitchell, R. J., Morecroft, M. D., Acreman, M., and others. 2007. England biodiversity strategy—towards adaptation to climate change. Available at: http://www.defra.gov.uk/wildlife-countryside/resprog/findings/ebs-climate-changes.pdf.

Mitrovich, M. J., J. E. Diffendorfer, and N. Robert. 2009. Behavioral response of the Coachwhip (*Masticophis flagellum*) to habitat fragment size and isolation in an urban landscape. Journal of Herpetology 43(4): 646-656.

Mukheibir, P., and G. Ziervogel. 2006. Framework for adaptation to climate change in the city of Cape Town. Available at: http://www.erc.uct.ac.za/publication/Framework%20for%20adaptation%20to%20CC%20in%20the%20city%20of%20Cape%20Town%20-%20FAC4t.pdf.

Mulheisen, M. and K. Berry. 2000. "*Eptesicus fuscus*." Available at Animal Diversity Web (5/05): http://animaldiversity.ummz.umich.edu/site/accounts/information/Eptesicus_fuscus.html.

National Research Council. 1994. Restoring and protecting marine habitat: the role of engineering and technology. National Academy Press, Washington, D.C.

NatureServe. 2009. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia. Available http://www.natureserve.org/explorer. (Accessed Dec 30, 2008).

Nelson, T., J. L. Holechek, R.Valdez, and M. Cardenas. 1997. Wildlife numbers on late and mid-seral Chihuahuan Desert rangelands. Journal of Range Management. 50: 593-599.

Nelson, J., J. L. Holecheck, and R.Valdez. 1999. Wildlife plant community preference in the Chihuahuan Desert. Rangelands 21: 9-12.

New Mexico Department of Game and Fish [NMDGF]. 1988. Handbook of species endangered in New Mexico 6-253: 1-2. Santa Fe, New Mexico. 150 p.

Niedzielski, S. 2002. "*Terrapene Carolina*" (On-line), Animal Diversity Web. Accessed January 19, 2010 at http://animaldiversity.ummz.umich.edu/site/accounts/information/Terrapene_carolina.html.

Noss, R. F., E. T. LaRoe, III, and J. M. Scott. 1995. Endangered ecosystems of the United States: a preliminary assessment of Loss and Degradation. Biological Report 28, National Biological Survey, Washington, D.C.

Obedzinski, R. A., C. G. Shaw, and D. G. Neary, 2001. Declining Woody Vegetation in Riparian Ecosystems in the Western United States. Western Journal of Applied Forestry 16: 169-181.

Omhart, R. D. and B. W. Anderson, 1982. North American Desert Riparian Ecosystems. In: References Handbook on the Deserts of North America, G. L. Bender (Editor). Greenwood Press, Westport, Connecticut, pp. 433-479.

Palmer, M. A., D. P. Lettenmaier, N. L. Poff, S. L. Postel, R. Richter, and R. Warner. 2009. Climate Change and river ecosystems: Protection and adaptation options. Environmental Management 44: 1053-1068.

Palmer, M. A., Reidy, C. A., C. Nilsson, M. Flörke, J. Alcamo, P. S. Lake, and N. Bond. 2008. Climate change and the world's river basins: anticipating management option. Frontiers in Ecology and the Environment 6: 81-89.

Panjabi, A. O., E. H. Dunn, P. J. Blancher, [and others]. 2005. The Partners in Flight handbook on species assessment. Version 2005. Partners in Flight Technical Series No. 3.

Parker, W. S. and E. R. Pianka. 1975. Comparative ecology of populations of the lizard *Uta stansburinana*. Copeia 4: 615-632.

Parmesan, C., T. Root, and M. Willig. 2000. Impacts of extreme weather and climate on terrestrial biota. Bulletin of the American Meteorological Society 81: 443-450.

Pase, C. P. and E. F. Layser. 1977. Classification of riparian habitat in the southwest. Pages 5-9 in R. R. Johnson and D. A. Jones (tech. coords.), Importance, preservation and management of riparian habitat: a symposium. U.S. Forest Service, General Technical Report RM-43. Tucson, Arizona.

Paxton, E. H., J. Owen and M. K. Sogge. 1996. Southwestern Willow Flycatcher response to catastrophic habitat loss. Report by the USGS Colorado Plateau Research Station, Flagstaff, AZ.

Peñuelas, J. and I. Filella. 2001. Responses to a warming world. Science 294: 793-795.

Peralta-Hernandez, A., R. C. Balling Jr., and L. R. Barba-Martinez. 2009. Analysis of near-surface diurnal temperature variations and trends in southern Mexico. International Journal of Climatology 29: 205-209.

Pérez, E. P., J. M. Méndez Pérez and V. M. Rueda. 2008. High Spatial Resolution Climate Change Scenarios for Mexico Based on Experiments Conducted with the Earth Simulator. In Visualizing Future Climate in Latin America: Results from the application of the Earth Simulator.

Peterson, A. T., Ortega-Huerta, M. A., Bartley, J., Sanchez-Cordero, V., Soberon, J., Buddemeier, R. H. and Stockwell, D. R. B. 2002. Future projections for Mexican faunas under global climate change scenarios. Nature 416: 626.

Pianka, E. R. 1970. Comparative autecology of the lizard *Cnemidophouus tigris* in different parts of its geographic range. Ecology 51: 703-720.

Popa-Lisseanu, A. G. and C. C. Voigt. 2009. Bats on the Move. Journal of Mammalogy 90: 1283-1289.

Pounds, J. A., Fogden, M. P. L. and J. H. Campbell. 1999. Biological response to climate change on a tropical mountain. Nature 398: 611:615.

Pylypec, B. 1991. Impacts of fire on bird populations in a fescue prairie. Canadian Field-Naturalist 105: 346-349.

Rasheed, S. A., P. F. J. Garcia, and S. L. Holyoyd.1995. Status of the fringed myotis in British Columbia. Ministry of Environment, Lands and Parks, Wildlife Branch, Victoria, British Columbia, Canada, Wildlife Working Report WR-73.

Redder, A. J., C. K. Dodd, Jr., and D. Keinath. 2006. Ornate Box Turtle (*Terrapene ornata ornata*): a technical conservation assessment. [Online]. USDA Forest Service, Rocky Mountain Region. Available: http://www.fs.fed.us/r2/projects/scp/assessments/ornateboxturtle.pdf.

Rehfeldt, Gerald E., Nicholas L. Crookston, Marcus V. Warwell, and Jeffrey S. Evans. 2006. Empirical analyses of plant-climate relationships for the western United States. International Journal of Plant Sciences 167: 1123-1150.

Rising, J. D. and P. L. Williams. 1999. Bullock's Oriole (*Icterus bullockii*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; http://bna.birds.cornell.edu/bna/species/416.

Roberts, L. 2005. Middle Rio Grande Ecosystem Bosque Biological Management Plan. The First Decade: a Review and Update. Aurora Publishing LCC, Albuquerque, New Mexico. 153 p.

Robinson, W. D. 1996. Summer Tanager (*Piranga rubra*). In The birds of North America, No. 248 (A. Poole and F. Gill, eds.). The Academy of Natural Sciences, Philadelphia, PA, and the American Ornithologists' Union, Washington, D.C.

Rood, S. B., Pan, J., Gill, K. M., Franks, C. G., Sameulson, G. M., and A. Shepard. 2006. Declining summer flow so the Rocky Mountain rivers: Changing seasonal hydrology and probable impacts of floodplain forests. Journal of Hydrology 349: 397-410.

Root, T. 1988. Atlas of wintering North American birds: an atlas of Christmas Bird Count data, 312 p. University of Chicago Press, Chicago, Illinois.

Root, Terry, Jeff T. Price, Kimberly R. Hall, Stephen H. Schneider, Cynthia Rosenzweig and J. Alan Pounds. 2003. Fingerprints of global warming on wild animals and plants. Nature 421: 57-60.

Rosatte, R. and S. Larivière. 2003. In Wild Mammals of North America: Biology, management, and Conservation. Feldhamer, G. A., Thompson, B. C. and J. A. Chapman (Eds). Johns Hopkins University Press, Baltimore, MD. 1197 p.

Rosenberg, K. V., R. D. Ohmart, and B. W. Anderson. 1982. Community organization of riparian breeding birds: response to an annual resource peak. Auk 99: 260-274.

Rosenberg, K. V., R. D. Ohmart, W. C. Hunter, and B. W. Anderson. 1981. Birds of the Lower Colorado River valley. University of Arizona Press, Tucson.

Russell, S. M. and G. Monson. 1998. The birds of Sonora. University of Arizona Press, Tucson.

Rustigian, H. L., M. V. Santelmann, and N. H. Schumaker. 2003. Assessing the potential impacts of alternative landscape designs on amphibian population dynamics. Landscape Ecology 18: 65-81.

Safi, K. and G. Kerth. 2004. A comparative analysis of specialization and extinction risk in temperate-zone bats. Conservation Biology 18: 1293-1303.

Sanz, Juan José. 2002. Climate change and birds: have their ecological consequences already been detected in the Mediterranean region? Ardeola 49: 109-120.

Saunders, Stephen, Charles Montgomery, Tom Easley, and Theo Spencer. 2008. Hotter drier West: the West's changed climate. Rocky Mountain Climate Organization and Natural Resources Defense Council, New York. Available: www.nrdc.org/policy.

Scheel, D., Vincent, T. L. S. and Guy N. Cameron. 1996. Global warming and the species richness of bats in TX. Conservation Biology 10: 452-464.

Seager, Richard, Mingfang Ting, Isaac Held, [and others]. 2007. Model projections of an imminent transition to a more arid climate in southwestern North America. Science 316: 1181-1184.

Sedgwick, J. A. 2000. Willow Flycatcher (*Empidonax trailii*). In The Birds of North America, No. 533 (A. Poole and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA.

Seigel, R. A. and Ford, N. B. 2001. Phenotypic plasticity in reproductive traits: geographical variation in plasticity in a viviparous snake. Functional Ecology 15: 36-42.

Shafroth, P. B., J. R. Cleverly, T. L. Dudley, J. P. Taylor, C. Van Riper, III, E. P. Weeks, and J. N. Stuart. Profile. Control of Tamarix in the Western United States: implication for water salvage, wildlife use and riparian restoration. Environmental Management 35: 231-246.

Shafroth, Patrick B., Juliet C. Stromberg, and Duncan T. Patten. 2002. Riparian vegetation response to altered disturbance and stress regimes. Ecological Applications 12: 107-123.

Sinervo, B., [and others]. 2010. Erosion of lizard diversity by climate change and altered thermal niches. Science 328: 895-899.

Skelly, D., L. Joseph, H. Possingham, L. Freidenburg, T. Farrugia, M. Kinnison, and A. Hendry. 2007. Evolutionary responses to climate change. Conservation Biology 21: 1353-1355.

Skogge, M. and R. Marshall. 2000. A survey of current breeding habitats. Pages 43-56 in Finch, D. and Stoleson, eds. Status, ecology, and conservation of the Southwestern willow flycatcher. USDA Forest Service, Rocky Mountain Research Station, RMRS-GTR-60, Ogden, Utah.

Smith, F. A., Browning, H. and U. L. Shepard. 1998. The influence of climate change on the body mass of woodrats *Neotoma* in an arid region of New Mexico, USA. Ecography 21: 140-148.

Smith, M., J. Kelly, and D. Finch. 2006. Cicada emergence in Southwestern riparian forest: influence of wildfire and vegetation composition. Ecological Applications 16: 1608-1616.

Smith, S. M. 1991. The Black-capped Chickadee: behavioral ecology and natural history. Cornell University Press, Ithaca, NY.

Smith, S. M. 1993. Black-capped Chickadee. In The Birds of North America, No. 39 (A. Poole, P. Stettenheim, and F. Gill, Eds.). Philadelphia: The Academy of Natural Sciences; Washington, D.C.: The American Ornithologists' Union.

Smith, W. P. 1991. *Odocoileus virginianus*. Mammalian Species 388: 1-13.

Sogge, M. K.; Paxton, E. H.; Tudor, April A. 2006. Saltcedar and Southwestern Willow Flycatchers: Lessons From Long-term Studies in Central Arizona. Pages 238-241 in Aguirre-Bravo, C.; Pellicane, P. J.; Burns, Denver P.; and Draggan, S., Eds. 2006. Monitoring Science and Technology Symposium: Unifying Knowledge for Sustainability in the Western Hemisphere Proceedings. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, RMRS-P-42CD. Fort Collins, CO.

Spurrier, S. and K. G. Smith. 2007. Desert mistletoe (*Phoradendron californicum*) infestation correlates with blue palo verde (*Cercidium floridum*) mortality during a severe drought in the Mojave desert. Journal of Arid Environments 69: 189-197.

Stebbins, R. C. 1985. A Field Guide to Western Reptiles and Amphibians. Second edition. Houghton Mifflin Company, Boston, Massachusetts.

Stoleson, S. H., R. S. Shook, and D. M. Finch. 2000. Breeding biology of Lucy's warbler in southwestern New Mexico. Western Birds 31: 235-242.

Strehl, C. E. and J. White. 1986. Effects of superabundant food on breeding success and behavior of the Red-winged Blackbird. Oecologia 70: 178-186.

Streubel, D. 2000. American Beaver from http://imnh.isu.edu/digitalatlas/bio/mammal/Rod/Beaver/beaver.htm and taken from Leege, T. A. 1968. Natural movement of beavers in southeastern Idaho. Journal of Wildlife Management 32: 973-976.

Stromberg, J. 1998. Dynamics of Fremont cottonwood (*Populus fremontii*) and saltcedar (*Tamarix chinensis*) populations along the San Pedro River, Arizona. Journal of Arid Environments 40: 133-155.

Stromberg, J. C. 1997. Growth and survivorship of Fremont cottonwood, Goodding willow and saltcedar seedlings after large floods in central Arizona. Great Basin Naturalist 57: 198-208.

Stromberg, J. C., Chew, M. K., Nagler, P. L., and E. P. Glenn. 2009. Changing perception of change: the role of scientists in Tamarix and river Management. Restoration Ecology 17: 177-186.

Stromberg, J. C., S. J. Lite, T. J. Rychener, L. Levick, M. D. Dixon, and J. W. Watts. 2006. Status of the riparian ecosystem in the upper San Pedro River, Arizona: Application of an assessment model. Environmental Monitoring and Assessment 115: 145-173.

Stromberg, J. C., V. B. Beauchamp, M. D. Dixon, S. J. Lite, and C. Paradzick. 2007. Importance of low-flow and high-flow characteristics to restoration of riparian vegetation along rivers in arid southwestern United States. Freshwater Biology 52: 651-679.

Stromberg, Juliet C., Scott D. Wilkins and James A. Tress. 1993. Vegetation-Hydrology Models: Implications for Management of *Prosopis velutina* (Velvet Mesquite) Riparian Ecosystems. Ecological Applications 3: 307-314.

Stuart, J. N. and J. P. Ward. 2009. *Trachemys gaigeae* (Hartweg 1939) Big Bend Slider, Mexican Plateau Slider, Jicotea de la Mesita Mexicana. IUCN/SSC Tortoise and Freshwater Turtle Specialist Group.

Sullivan, J. 1995. *Dipodomys ordii*. In: Fire Effects Information System, [Online]. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer). Available: http://www.fs.fed.us/database/feis/.

Svensson, S. 1986. Number of pairs, timing of egg-laying and clutch size in a subalpine Sand Martin *Riparia riparia* colony, 1968-1985. Ornis Scandinavica 17: 221-229.

Swetnam, T. W. and J. L. Betancourt. 1990: Fire–Southern oscillation relations in the southwestern United States. Science 24: 1017-1020.

Taylor, J. P., L. M. Smith, and D. A. Haukos. 2006. Evaluation of woody plant restoration in the Middle Rio Grande: ten years after. Wetlands 26: 1151-1160.

The Heinz Center. 2008. Strategies for managing the effects of climate change on wildlife and ecosystems. Washington, DC. 43 p.

Thibault, K. M., and J. H. Brown. 2008. Impacts of an extreme climatic event on community assembly. Proceedings of the National Academy of Sciences 105: 3410-3415.

Thompson, C. W. and V. Nolan, Jr. 1973. Population Biology of the Yellow-breasted Chat (*Icteria virens*) in southern Indiana. Ecological Monographs 43: 145-171.

Tinsley, R. C., and K. Tocque. 1996. The population dynamics of a desert anuran, *Scaphiopus couchii*. Austral Ecology 20: 376-384.

Turner, A. K. and C. Rose. 1989. Swallows and martins an identification guide and handbook. Houghton Mifflin Co., Boston, MA.

U.S. Army Corps of Engineers, Albuquerque District. 2011. Environmental Assessment for the Middle Rio Grande Bosque Restoration Project. Available: http://www.spa.usace.army.mil/Portals/16/docs/environmental/fonsi/MRG%20Bosque%20Final%20Environmental%20Assessment.pdf.

U.S. Department of the Interior Bureau of Reclamation Technical Service Center. 2011. West-Wide Climate Risk Assessments: Bias-Corrected and Spatially Downscaled Surface Water Projections. Technical Memorandum No. 86: 68210–2011-01–SECURE Water Act Section 9503(c,)Reclamation Climate Change and Water, Report to Congress, 2011. Authors, Contributors, and Editors: Alexander, Patty; Brekke, Levi; Davis, Gary; and others.

Valdez, E. W. and P. M. Cryan. 2009. Food habits of the hoary bat (*Lasiurus cinereus*) during spring migration through New Mexico. The Southwestern Naturalist 54: 195-200.

Vergara, W., Kondo, H. Pèrez E. and others. Latin America and Caribbean Region Sustainable Development Working Paper 30. Regional Climate Change Scenarios for South America: The CREAS project Jose A. Marengo CPTEC/INPE from the Conference on Climate Change and Official Statistics, Oslo, Norway April 14-16, 2008.

Visser, M. E. 2008. Keeping up with a warming world; assessing the rate of adaptation to climate change. Proceedings of the Royal Society, Biological Sciences 275: 649-659.

Walsberg, G. E. 1977. Ecology and energetics of contrasting social systems in the phainopepla (Aves: Ptilogonatidae). University of California Publications in Zoology 108: 1-63.

Walsberg, G. E. 2000. Small mammals in hot deserts: Some generalizations revisited. BioScience 50: 109-120.

Walther, G., S. Beissner, and C. Burga. 2005. Trends in the upward shift of alpine plants. Journal of Vegetation Science 16: 541-48.

Wang, G., Hobbs, N. T., Singer F. J., Ojima, D. S. and B. C. Lubow. 2002. Impacts of climate changes on elk population dynamics in rocky mountain national park, Colorado, U.S.A. Global Change 54: 205-223.

Werner, E. and K. Glennemeier. 1999. Influence of Forest Canopy Cover on the Breeding Pond Distributions of Several Amphibian Species. Copeia 1999: 1-12.

Whitney, J. C. 1996. The Middle Rio Grande: Its ecology and management. In: Shaw, D. W., Finch, D. M., tech coord. Desired future conditions for Southwestern riparian ecosystems: bringing interests and concerns together. 1995 Sept. 18-22. USDA Forest Service, RMRS, Gen. Tech. Rep. RM-GTR-272.

Wiebe, K. L. and W. S. Moore. 2008. Northern Flicker (*Colaptes auratus*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/166a.

Wilkins, K. T. 1989. *Tadarida brasiliensis*. Mammalian Species No. 331. The American Society of Mammalogists 1-10.

Williams, B. K., R. C. Szaro, and D. D. Shapiro. 2007. Adaptive Management: The U.S. Department of Interior technical guide. Adaptive management working group, U.S. Department of Interior, Washington, D.C.

Williams, K. S., and C. Simon. 1995. The ecology, behavior, and evolution of periodical cicadas. Annual Review of Entomology 40: 269-295.

Williams, P. H. 1986. Environmental change and the distributions of British bumble bees (*Bombus*, Latr.). Bee World 67: 50-61.

Williams, P. L. 1988. Spacing behavior and related features of social organization in Northern Orioles of central coastal California. Doctorate. University of California, Berkeley.

Willis, C. K. R., and R. M. Brigham. 2005. Physiological and Ecological aspects of roost selection by reproductive female hoary bats (*Lasiurus cinereus*). Journal of Mammalogy 86: 85-94.

Willner, G. R., Feldhamer, G. A., Zucker, E. E., and J. A. Chapman. 1980. *Ondatra zibethicus*. Mammalian Species 141: 1-8.

Wingfield, J. C. 2008. Comparative endocrinology, environment, and global change. General and Comparative Endocrinology 157: 207-216.

Winne, C. T. and M. B. Keck. 2004. Daily activity patterns of whiptail lizards (Squamata: Teiidae: *Aspidoscelis*): a proximate response to environmental conditions or an endogenous rhythm? Functional Ecology 18: 314-321.

Winternitz, B. L. 1998. Hairy woodpecker. In Kingery, H. E., ed. Colorado Breeding Bird Atlas. Colorado Breeding Bird Atlas Partnership and Colorado Division of Wildlife.

Wong, C. M., Williams, C. E., Pittock, J., Collier, U. and P. Schelle. 2007. World's top 10 rivers at risk. World Wildlife Federation International. Gland, Switzerland.

Wormworth, J. and Karl Mallon. 2006. Bird species and climate change: the global status report version 1.1. World Wildlife Federation for Nature, Australia.

Young, B., E. Byers, K. Gravuer, K. Hall, G. Hammerson, and A. Redder. 2009. NatureServe Climate Change Vulnerability. NatureServe, Arlington, VA.

Appendix 1.

Scored criteria for each trait considered in the assessment of the vulnerability of 117 terrestrial species to climate change in the Middle Rio Grande Bosque, New Mexico. For each criterion, a species may be vulnerable (v), resilient (r), or regarded as not affected or no known/predictable effect (n). Introduced species are indicated with an (I).

	Question	Changes to breeding habitat	Changes to non-breeding habitat	Changes to breeding component	Changes to non-breeding component	Changes to breeding habitat quality	Changes to non-breeding habitat quality	Dispersal ability	Changes to stop-over sites	Physiological limit	Temperature-determined sex ratios	Exposure to extreme weather	Activity limitations	Capacity for metabolic inhibition	Exposure to resource limitations	Variable life history traits	Ability to outlive limiting conditions	Reliance on temp or precipitation cue	Reliance on weather mediated resource	Spatial or temporal separation	Timing mismatches during breeding	Food resources	Predation	Symbiotic relationship	Disease	Competitors
	No.	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**	**21**	**22**	**23**	**24**	**25**
Common name	***Scientific name***																									
Amphibians																										
American bullfrog (I)	*Lithobates catesbeiana*	v	v	v	n	r	n	r	n	n	n	n	n	n	r	n	r	n	r	r	r	n	n	n	n	n
Barred tiger salamander	*Ambystoma m. mavortium*	v	n	v	n	v	n	v	n	n	n	v	v	n	r	r	v	v	v	r	v	n	n	n	v	n
Couch's spadefoot	*Scaphiopus couchii*	r	r	v	n	v	n	v	n	n	n	v	v	n	r	r	n	v	v	r	v	n	n	n	n	n
Great Plains toad	*Bufo cognatus*	r	r	v	n	v	n	v	n	n	n	v	v	n	r	n	v	v	v	r	v	n	n	n	n	n
New Mexico spadefoot toad	*Spea multiplicata*	r	r	v	n	v	n	v	n	n	n	v	v	n	r	r	n	v	v	r	v	n	n	n	n	n
Northern leopard frog	*Lithobates pipiens*	v	v	v	v	v	n	v	n	n	n	v	n	n	r	n	v	v	n	r	v	n	v	n	v	v
Plains spadefoot toad	*Spea bombifrons*	r	r	v	n	v	n	n	n	n	n	v	v	n	r	r	n	v	v	r	r	n	n	n	n	n
Western chorus frog	*Pseudacris triseriata*	v	v	v	n	n	n	v	n	n	n	v	n	v	r	n	v	v	v	r	v	n	n	n	v	n
Woodhouse's toad	*Bufo woodhousii*	v	v	v	n	n	n	v	n	n	n	n	v	n	r	n	v	v	n	r	r	n	n	n	v	n

	Question	Changes to breeding habitat	Changes to non-breeding habitat	Changes to breeding component	Changes to non-breeding component	Changes to breeding habitat quality	Changes to non-breeding habitat quality	Dispersal ability	Changes to stop-over sites	Physiological limit	Temperature-determined sex ratios	Exposure to extreme weather	Activity limitations	Capacity for metabolic inhibition	Exposure to resource limitations	Variable life history traits	Ability to outlive limiting conditions	Reliance on temp or precipitation cue	Reliance on weather mediated resource	Spatial or temporal separation	Timing mismatches during breeding	Food resources	Predation	Symbiotic relationship	Disease	Competitors
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Common name	***Scientific name***																									
Reptiles																										
Big Bend slider	*Trachemys geigeae*	v	v	n	v	n	v	v	n	n	v	v	n	n	r	n	r	v	r	n	r	n	n	n	n	n
Black-necked gartersnake	*Thamnophis cyrtopsis*	v	v	n	n	n	n	r	n	n	n	v	n	n	r	n	v	v	n	n	v	v	n	n	n	n
Checkered gartersnake	*Thamnophis marcianus*	v	v	n	n	n	n	r	n	n	n	v	n	n	r	n	v	v	n	n	r	v	n	n	n	v
Checkered whiptail	*Aspidoscelis tesselata*	v	v	n	n	n	n	v	n	n	n	n	n	n	r	n	v	v	n	n	r	n	n	n	n	n
Chihuahuan spotted whiptail	*Aspidoscelis exsanguis*	v	v	n	n	n	n	v	n	n	n	n	n	n	r	n	v	v	n	n	r	n	n	n	n	n
Coachwhip	*Masticphis flagellum*	v	v	v	n	n	n	r	n	n	n	n	n	n	r	n	r	v	n	n	v	n	n	n	n	n
Common gartersnake	*Thamnophis sirtalis*	n	n	n	v	n	n	r	n	n	n	v	n	n	r	n	r	v	n	n	v	n	n	n	n	n
Desert grassland whiptail	*Aspiodoscelis uniparens*	r	r	n	n	n	n	v	n	n	n	n	n	n	r	n	v	v	n	n	r	n	n	n	n	n
Desert or common kingsnake	*Lampropeltis getula*	n	n	v	n	n	n	r	n	n	n	n	n	n	r	n	r	v	n	n	v	n	n	n	n	n
Desert spiny lizard	*Sceloporus magister*	v	v	n	n	n	n	v	n	n	n	n	v	n	r	n	v	v	n	n	r	n	n	n	n	n
Glossy snake	*Arizona elegans*	n	n	v	n	n	n	r	n	n	n	n	n	n	r	n	v	v	n	n	v	n	n	n	n	n
Gophersnake	*Pitouphis catenifer*	n	n	n	n	n	n	r	n	v	n	n	v	n	r	n	r	v	n	n	v	n	n	n	n	n
Great Plains skink	*Eumeces obsoletus*	v	v	v	v	n	n	v	n	n	n	n	n	n	r	n	v	v	n	n	v	n	n	n	n	n
Many-lined skink	*Eumeces multivirgatus*	n	n	v	v	n	n	v	n	n	n	n	n	n	r	n	v	v	n	n	v	n	n	n	n	n

New Mexico whiptail	*Aspidoscelis neomexicana*	n	n	n	n	n	n	v	n	n	n	n	n	n	r	n	v	v	n	n	v	n	n	n	n	n
Ornate box turtle	*Terrapene ornata*	n	n	n	n	n	v	v	n	n	v	n	v	n	r	n	r	v	r	n	r	n	n	n	n	n
Plains black-headed snake	*Tantilla nigriceps*	n	n	n	n	n	n	v	n	n	n	n	n	n	r	n	v	v	n	n	v	n	n	n	n	n
Prairie lizard or southwestern fence lizard	*Sceloporus cowlesi*	n	n	n	n	n	n	v	n	n	n	n	v	n	r	n	v	v	n	n	r	n	n	n	n	n
Racer	*Coluber constrictor*	n	n	v	n	n	n	r	n	n	n	v	v	n	r	n	r	v	n	n	v	n	n	n	n	n
Red-eared slider (I)	*Trachemys scripta*	v	v	n	n	n	n	v	n	n	v	v	n	n	r	n	r	v	r	n	r	n	n	n	n	n
Side-blotched lizard	*Uta stansburiana*	n	n	n	n	n	n	v	n	v	n	v	v	n	r	n	v	v	n	n	r	n	n	n	n	n
Snapping turtle (I)	*Chelydra serpentina*	v	v	n	v	n	n	v	n	v	v	n	n	n	r	n	r	v	r	n	r	n	n	n	n	n
Spiny softshell turtle	*Apalone spinifera*	v	v	v	n	n	n	v	n	n	v	v	v	n	r	n	r	v	r	n	r	n	n	n	n	n
Western diamondback rattlesnake	*Crotalus atrox*	n	n	n	n	n	n	r	n	n	n	v	n	n	r	n	r	v	n	n	v	n	n	n	n	n
Western hognosed snake	*Heterodon nasicus*	n	n	v	n	n	n	r	n	n	n	n	n	n	r	n	r	v	n	n	v	v	n	n	n	n
Western painted turtle	*Chrysemys picta*	v	v	n	v	n	n	v	n	v	v	v	n	n	r	n	r	v	r	n	r	n	n	n	n	n
Western terrestrial gartersnake	*Thamnophis elegans*	n	n	n	v	n	n	r	n	n	n	v	n	n	r	n	v	v	n	n	v	n	n	n	n	n
Western whiptail	*Aspioscelis tigris*	n	n	n	n	n	n	v	n	n	n	v	n	n	r	n	v	v	n	n	r	n	n	n	n	n
Western/prairie rattlesnake	*Crotalus viridis*	n	n	n	n	n	n	r	n	n	n	v	n	n	r	n	r	v	n	n	v	n	n	n	n	n
Birds																										
American kestrel	*Falco sparverius*	r	r	v	n	n	n	r	n	n	n	n	n	v	v	n	v	n	v	r	r	n	n	v	n	n
Ash-throated flycatcher	*Myiarchus cinerascens*	n	r	v	n	n	n	r	v	n	n	n	n	v	v	n	v	n	v	v	r	n	n	n	n	n
Bank swallow	*Riparia riparia*	v	n	n	n	n	n	r	v	n	n	v	n	v	v	n	v	n	v	v	r	v	n	n	n	n
Barn swallow	*Hirundo rustica*	r	v	n	n	n	n	r	v	n	n	v	n	v	v	n	v	v	v	v	r	n	n	n	v	v
Bewick's wren	*Thryomanes bewickii*	n	n	v	v	n	n	r	n	n	n	n	n	v	v	n	v	n	v	r	r	n	n	n	n	n
Black phoebe	*Sayornis nigricans*	v	v	n	n	n	n	r	n	r	n	n	v	v	v	n	r	v	n	r	r	n	n	n	n	n
Black-capped chickadee	*Poecile atricapillus*	v	v	v	n	v	n	r	n	n	n	n	n	r	v	n	v	n	v	r	v	n	n	n	n	n
Black-chinned hummingbird	*Archilochus alexandri*	n	n	n	n	v	n	r	v	v	n	v	n	r	v	n	v	n	v	v	r	n	n	n	v	n

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Common name	***Scientific name***																									
Black-headed grosbeak	*Pheucticus melanocephalus*	v	v	n	n	n	n	r	v	n	n	n	n	v	v	n	v	n	v	v	r	n	n	n	n	n
Blue grosbeak	*Passerina caerulea*	r	r	n	n	n	n	r	v	n	n	n	n	v	v	n	v	n	v	v	r	n	n	n	v	n
Brown-headed cowbird	*Molothrus ater*	r	r	n	n	n	n	r	n	n	n	n	n	v	v	n	n	n	n	r	r	n	n	n	n	n
Bushtit	*Psaltriparus minimus*	v	v	n	n	n	n	r	n	n	n	n	n	v	v	n	v	n	v	r	r	n	n	n	n	n
Bullock's oriole	*Icterus bullockii*	v	v	n	n	n	n	r	v	n	n	v	n	v	v	n	v	n	v	v	r	n	n	n	n	n
Cliff swallow	*Petrochelidon pyrrhonota*	v	r	n	n	n	v	r	v	n	n	v	n	v	v	n	v	n	v	v	v	n	n	n	v	v
Common yellowthroat	*Geothlypis trichas*	v	n	v	n	n	n	r	v	n	n	v	n	v	v	n	v	n	v	v	v	n	n	n	n	v
Cooper's hawk	*Accipiter cooperii*	v	v	v	n	n	n	r	n	n	n	n	n	v	v	n	v	n	v	r	v	n	n	n	v	n
Downy woodpecker	*Picoides pubescens*	v	v	v	v	n	v	r	n	n	n	n	n	v	v	n	v	n	v	r	v	n	n	n	n	n
Eastern bluebird	*Sialia sialis*	n	n	n	v	n	n	r	n	n	n	n	n	v	v	n	v	n	n	r	r	n	n	n	v	n
European starling (I)	*Sturnus vulgaris*	r	r	n	n	n	n	r	n	n	n	v	n	v	v	n	n	v	v	r	r	n	n	n	n	n
Great horned owl	*Bubo virginianus*	n	n	v	v	n	n	r	n	n	n	n	n	v	v	n	r	n	v	r	v	n	n	v	n	n
Greater roadrunner	*Geococcyx californianus*	r	r	r	n	n	n	r	n	n	n	n	v	r	v	n	v	n	v	r	r	n	n	n	n	n
Hairy woodpecker	*Picoides villosus*	v	v	v	v	n	n	r	n	n	n	n	n	v	v	n	r	n	v	r	v	n	n	n	n	n
House finch	*Carpodacus mexicanus*	r	r	n	n	n	n	r	n	n	n	n	n	v	v	n	v	n	n	r	r	n	n	n	n	n
Indigo bunting	*Passerina cyanea*	r	r	n	n	v	n	r	v	v	n	n	n	v	v	n	v	n	n	v	r	n	n	n	n	n

Ladder-backed woodpecker	*Picoides scalaris*	n	n	v	n	n	n	r	n	n	n	n	n	v	v	n	v	n	v	r	v	n	n	n	n	n
Lesser goldfinch	*Carduelis psaltria*	n	n	n	n	n	n	r	n	n	n	r	n	v	v	n	v	v	n	r	v	n	n	n	n	n
Lucy's warbler	*Vermivora luciae*	n	v	v	n	n	n	r	v	n	n	v	n	v	v	n	v	n	v	v	r	n	n	n	n	n
Mourning dove	*Zenaida macroura*	n	r	r	r	n	n	r	n	n	n	v	n	v	v	n	v	n	n	r	r	n	n	n	v	n
Northern flicker	*Calaptes auratus*	v	v	v	n	n	n	r	n	n	n	n	n	v	v	n	v	n	v	r	v	n	n	n	n	v
Northern rough-winged swallow	*Stelgidopteryx serripennis*	n	r	n	n	n	v	r	v	n	n	v	n	v	v	n	v	n	v	r	r	n	n	n	n	n
Phainopepla	*Phainopepla nitens*	v	n	n	n	n	n	r	n	n	n	n	v	v	v	r	v	n	v	r	r	v	n	n	n	n
Red-winged blackbird	*Agelaius phoeniceus*	v	v	n	v	v	n	r	n	n	n	v	n	v	v	r	v	n	v	r	r	n	n	n	v	n
Spotted towhee	*Pipilo maculatus*	r	n	n	n	n	n	r	n	n	n	n	n	v	v	n	v	n	v	r	v	n	n	n	n	n
Southwestern willow flycatcher	*Empidonax traillii extimus*	v	v	v	n	v	n	r	v	v	n	n	n	v	v	n	v	n	v	v	v	v	n	v	n	n
Summer tanager	*Piranga rubra*	v	n	v	n	n	n	r	v	v	n	v	v	v	v	n	v	n	v	v	r	n	n	n	n	n
Western meadowlark	*Sturnella neglecta*	v	v	n	n	v	n	r	n	n	n	n	n	v	v	n	v	v	v	r	r	n	n	n	v	n
Western kingbird	*Tyrannus verticalis*	r	n	n	n	n	n	r	v	n	n	v	n	v	v	n	v	n	v	v	r	n	n	n	n	n
Western screech-owl	*Megascops kennicottii*	v	v	v	n	n	n	r	n	n	n	v	n	v	v	n	v	n	v	r	v	n	n	n	n	n
Western wood-pewee	*Contopus sordidulus*	v	v	v	n	n	n	r	v	n	n	v	n	v	v	n	v	n	v	v	r	n	n	n	n	n
Western yellow-billed cuckoo	*Coccyzus a. occidentalis*	v	n	v	n	n	n	r	v	v	n	v	n	v	v	n	v	n	v	v	v	r	n	n	n	n
White-breasted nuthatch	*Sitta carolinensis*	v	v	v	n	n	v	r	n	n	n	n	n	v	v	n	v	n	n	r	v	n	n	v	n	n
Yellow-breasted chat	*Icteria virens*	n	v	r	r	v	n	r	v	v	n	v	v	v	v	n	v	n	v	v	r	n	n	n	v	n
Mammals																										
Beaver	*Castor canadensis*	v	v	v	v	v	n	r	n	n	n	v	n	r	r	n	n	n	n	n	v	v	n	n	n	n
Big brown bat	*Eptesicus fuscus*	n	n	n	n	n	n	r	n	n	n	v	n	r		n	r	v	v	r	v	n	n	n	n	n
Black bear	*Ursus americanus*	v	v	n	n	v	n	r	n	n	n	n	v		r	n	r	v	v	n	v	v	n	n	n	n

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Common name	***Scientific name***																									
Bobcat	*Lynx rufus*	n	n	n	n	n	n	r	n	n	n	v	n	v	v	n	r	n	n	n	r	n	n	n	n	v
Botta's pocket gopher	*Thomomys bottae*	n	n	n	n	n	n	v	n	n	n	n	n	v	r	n	v	n	v	r	r	n	n	n	n	n
Coyote	*Canis latrans*	n	n	n	n	n	n	r	n	n	n	n	n	v	v	n	r	n	n	n	v	n	n	n	n	n
Crawford's or desert shrew	*Notiosorex crawfordi*	r	r	n	n	n	n	v	n	v	n	n	n	r	v	n	v	n	n	r	r	n	n	n	n	n
Desert cottontail	*Sylvilagus audubonii*	n	n	v	v	n	n	r	n	n	n	n	v	v	v	n	v	n	n	r	r	n	n	n	n	n
Elk	*Cervus elaphus*	v	n	n	n	n	n	r	n	n	n	n	n	v	v	n	r	n	v	r	v	n	n	n	n	n
Fringed bat	*Myotis thysanodes*	n	n	v	n	n	n	r	n	v	n	n	n	r		n	r	v	v	r	v	n	n	n	n	n
Grey fox	*Urocyon cinereoargenteus*	v	v	n	n	n	n	r	n	n	n	n	n	v	v	n	r	n	n	n	v	n	n	n	n	n
Hispid cotton rat	*Sigmodon hispidus*	n	n	n	n	v	v	v	n	n	n	v	n	v		n	v	n	n	r	r	n	n	n	n	n
Hoary bat	*Lasiurus cinereus*	v	n	v	n	n	n	r	v	n	n	v	n	r	v	n	r	v	v	v	v	n	n	n	n	n
Jackrabbit	*Lepus californicus*	r	r	n	n	n	n	r	n	n	n	n	n	v	v	n	n	n	n	n	r	n	v	n	n	n
Long-legged bat	*Myotis volans*	v	n	n	n	n	n	r	v	v	n	n	n	r		n	r	v	v	r	v	n	n	n	n	n
Long-tailed weasel	*Mustela frenata*	n	n	n	v	n	n	r	n	v	n	n	n	v	r	n	v	n	n	n	v	n	n	n	n	n
Mexican free-tailed bat	*Tadarida brasiliensis*	v	n	n	n	n	n	r	v	n	n	n	n	r	v	n	r	v	v	v	v	n	n	n	n	n

Mexican woodrat	*Neotoma mexicana*	v	v	n	n	n	n	v	n	v	n	n	n	v	r	n	v	n	n	n	r	n	n	n	n	n
Mountain lion	*Puma concolor*	v	v	n	n	n	n	r	n	n	n	n	n	v	v	n	r	n	n	n	r	v	n	n	n	n
Mule deer	*Odocoileus hemionus*	n	n	v	n	v	n	r	n	v	n	v	n	v	v	n	r	n	n	n	v	n	n	n	n	n
Muskrat	*Ondatra zibethicus*	v	v	n	n	n	n	r	n	n	n	v	n	v	r	n	v	n	n	n	r	n	n	n	n	n
New Mexican meadow jumping mouse	*Zapus hudsonius luteus*	v	v	n	n	n	n	v	n	n	n	v	n	n		n	v	v	n	v	r	n	n	v	n	n
Occult bat	*Myotis occultus*	v	n	n	n	n	n	r	n	v	n	n	n	r		n	r	v	v	r	v	n	n	n	n	n
Ord's kangaroo rat	*Dipodomys ordii*	n	n	n	n	n	n	v	n	n	n	v	n	v	r	n	r	v	v	r	r	n	n	n	n	n
Pallid bat	*Antrozous pallidus*	v	n	n	n	n	n	r	n	v	n	n	n	r		n	r	v	n	n	v	n	n	n	n	n
Piñon mouse	*Peromyscus truei*	v	v	n	n	n	n	v	n	n	n	v	n	r	v	n	v	n	n	r	r	n	n	n	n	n
Porcupine	*Erethizon dorsatum*	v	v	n	n	n	n	r	n	n	n	n	n	v	v	n	r	n	n	n	v	n	n	n	n	n
Raccoon	*Procyon lotor*	v	v	n	n	n	n	r	n	n	n	n	n	v	r	n	r	v	n	v	r	n	n	n	n	n
Rock squirrel	*Spermophilus variegatus*	n	n	n	n	n	n	v	n	n	n	n	v	r		n	v	n	n	r	r	n	n	n	n	n
Silky pocket mouse	*Perognathus flavus*	n	n	n	n	n	n	v	n	n	n	n	n	r	v	n	v	v	n	r	r	n	n	n	n	n
Striped skunk	*Mephitis mephitis*	v	v	n	n	n	n	r	n	v	n	n	n	r		n	r	n	n	n	r	n	v	n	n	n
Tawny-bellied cotton rat	*Sigmodon fulviventer*	v	v	n	n	n	n	r	n	n	n	v	n	v		n	v	n	n	r	r	n	n	n	n	n
Western harvest mouse	*Reithrodontomys megalotis*	v	v	n	n	n	n	v	n	n	n	n	n	r	v	n	v	n	n	r	r	n	n	n	n	n
White-footed mouse	*Peromyscus leucopus*	v	v	n	n	n	n	v	n	n	n	n	n	r	v	n	v	n	n	r	r	n	n	n	n	n
White-throated woodrat	*Neotoma albigula*	v	n	n	n	v	n	v	n	v	n	n	n	v	r	n	n	n	n	n	r	n	v	n	n	n
Yuma bat	*Myotis yumanensis*	v	v	v	n	n	n	r	n	n	n	n	n	r		n	r	v	v	r	v	n	n	n	n	n

Appendix 2. Uncertainty Scores for MRG Species

Potential scores were 0 = sufficient data, 1 = minimal data, and 2 = no data or conflicting data. Uncertainty was determined by individual users and based upon their interpretation of the scoring system and feelings regarding the availability of data and information for use to assign score. Due to the subjective nature of the uncertainty scores, they are best interpreted as a general estimation of data gaps. They are not based upon a quantitative analysis of available peer-reviewed literature. Introduced species are indicated with an (I).

	Name	Habitat	Physiology	Phenology	Biotic	Sum score
Amphibians						
	American bullfrog (I)	0	1	2	1	4
	Barred tiger salamander	0	0	0	1	1
	Couch's spadefoot toad	1	1	1	2	5
	Great Plains toad	0	1	0	2	3
	New Mexico spadefoot toad	0	1	0	2	3
	Northern leopard frog	1	1	1	1	4
	Plains spadefoot toad	0	1	0	1	2
	Western chorus frog	1	1	2	1	5
	Woodhouse's toad	1	1	1	1	4
Reptiles						
	Big Bend slider	0	1	1	1	3
	Black-necked gartersnake	1	0	0	1	2
	Checkered gartersnake	0	0	1	0	1
	Checkered whiptail	0	0	0	1	1
	Chihuahuan spotted whiptail	2	2	0	1	5
	Coachwhip	1	0	0	0	1
	Common gartersnake	0	1	1	1	3
	Desert grassland whiptail	2	2	1	1	6
	Desert spiny lizard	1	1	1	0	3
	Desert/common kingsnake	0	1	0	0	1
	Glossy snake	0	0	1	0	1
	Gophersnake	0	0	0	1	1
	Great Plains skink	0	1	1	1	3
	Many-lined skink	2	2	1	1	6
	New Mexico whiptail	0	1	1	1	3
	Ornate box turtle	0	1	1	1	3
	Plains black-headed snake	2	1	1	1	5
	Prairie lizard or southwestern fence lizard	2	1	1	0	4
	Racer	1	0	0	0	1
	Red-eared slider (I)	0	0	1	0	1
	Side-blotched lizard	0	0	0	0	0
	Snapping turtle (I)	1	0	1	0	2

	Name	Habitat	Physiology	Phenology	Biotic	Sum score
	Spiny softshell	1	0	1	0	2
	Western diamondback Rattlesnake	1	1	0	0	2
	Western hognosed snake	0	0	0	0	0
	Western painted turtle	0	0	1	1	2
	Western terrestrial Gartersnake	0	0	1	0	1
	Western whiptail	1	0	0	0	1
	Western/prairie rattlesnake	0	0	1	1	2
Birds						
	American kestrel	0	1	1	1	3
	Ash-throated flycatcher	1	1	1	1	4
	Bank swallow	1	1	2	1	5
	Barn swallow	2	1	1	0	4
	Bewick's wren	1	1	1	0	3
	Black phoebe	1	1	2	2	6
	Black-capped chickadee	0	1	1	1	3
	Black-chinned hummingbird	1	1	1	0	3
	Black-headed grosbeak	1	1	1	0	3
	Blue grosbeak	1	2	2	1	6
	Brown-headed cowbird	0	1	1	1	3
	Bushtit	1	1	1	1	4
	Bullock's oriole	1	2	1	2	6
	Cliff swallow	2	1	0	0	3
	Common yellowthroat	1	0	1	1	3
	Cooper's hawk	0	1	2	1	4
	Downy woodpecker	0	0	1	0	1
	Eastern bluebird	1	1	2	1	5
	European starling (I)	1	1	0	0	2
	Great horned owl	0	0	0	0	0
	Greater roadrunner	0	0	0	0	0
	Hairy woodpecker	0	2	1	1	4
	House finch	0	0	0	1	1
	Indigo bunting	0	1	0	1	2
	Ladder-backed woodpecker	0	1	2	1	4
	Lesser goldfinch	1	1	1	1	4
	Lucy's warbler	1	2	2	1	6
	Mourning dove	0	0	0	0	0
	Northern flicker	0	0	1	0	1
	Northern rough-winged swallow	1	2	1	1	5
	Phainopepla	1	0	1	0	2
	Red-winged blackbird	0	1	1	0	2

	Name	**Habitat**	**Physiology**	**Phenology**	**Biotic**	**Sum score**
	Spotted towhee	0	0	1	1	2
	Southwestern willow flycatcher	0	1	1	0	2
	Summer tanager	1	1	1	1	4
	Western meadowlark	1	1	2	1	5
	Western kingbird	1	1	1	1	4
	Western screech-owl	1	1	1	1	4
	Western wood-pewee	1	1	1	0	3
	Western yellow-billed cuckoo	2	2	1	1	6
	White-breasted nuthatch	0	1	1	0	2
	Yellow breasted chat	0	2	2	1	5
Mammals						
	Beaver	0	1	1	0	2
	Big brown bat	0	0	0	0	0
	Black bear	0	1	1	1	3
	Bobcat	0	1	1	0	2
	Botta's pocket gopher	1	0	0	1	2
	Coyote	0	0	1	1	2
	Crawford's or desert shrew	1	0	0	0	1
	Desert cottontail	0	1	1	1	3
	Elk	0	0	1	0	1
	Fringed bat	1	0	1	0	2
	Grey fox	1	2	1	1	5
	Hispid cotton rat	0	1	1	0	2
	Hoary bat	1	0	0	0	1
	Jackrabbit	0	0	0	0	0
	Long-legged bat	0	0	1	0	1
	Long-tailed weasel	0	0	0	1	1
	Mexican free-tailed bat	1	0	0	1	2
	Mexican woodrat	1	1	0	0	2
	Mountain lion	0	1	0	0	1
	Mule deer	0	0	0	0	0
	Muskrat	0	1	1	0	2
	New Mexican meadow jumping mouse	0	1	1	1	3
	Occult bat	0	0	1	0	1
	Ord's kangaroo rat	0	0	0	0	0
	Pallid bat	0	0	0	0	0
	Pinyon mouse	1	0	0	0	1
	Porcupine	0	0	0	0	0
	Raccoon	0	1	1	0	2

	Name	Habitat	Physiology	Phenology	Biotic	Sum score
	Rock squirrel	0	1	1	1	3
	Silky pocket mouse	0	0	1	0	1
	Striped skunk	0	0	1	0	1
	Tawny bellied cotton rat	0	1	1	0	2
	Western harvest mouse	0	0	0	0	0
	White-footed mouse	0	0	0	0	0
	White-throated woodrat	0	0	0	0	0
	Yuma bat	0	0	0	0	0

Appendix 3. Detailed species assessment results from the report: "Vulnerability of Species to Climate Change in the Southwest: Terrestrial Species of the Middle Rio Grande" by Megan M. Friggens, Karen E. Bagne, Sharon J. Coe, Deborah M. Finch, and David L. Hawksworth.

Contents

Amphibian Results

Habitat Scores

Vegetation Association: Habitat scores considered trends (decreases/increases) for associated vegetation and required habitat components, changes in habitat quality, requirements for additional habitat and capacity of species to move with transitions in critical habitat features. Most amphibians were expected to experience some decline in their associated vegetation (Table A1). Species such as the northern leopard frog, western chorus frog, and American bullfrog, which are mostly aquatic and rely on moist conditions, received vulnerability scores for a number of questions (Appendix 1). Amphibians can occur in a variety of habitats but commonly rely on permanent ponds and wetlands (BISON-M 2009; Degenhardt and others 1996). We felt it was likely that these species will experience significant declines in breeding and nonbreeding habitats as the climate warms. Most amphibians require moist environments and lack characteristics, such as the ability to aestivate, that other species use to withstand dry conditions. An exception are the spadefoot toad and barred tiger salamander.

We predicted that the Great Plains toad, Woodhouse's toad, and barred tiger salamander will experience increased loss of breeding habitat but no change in

Table 1A. Habitat characteristics for amphibians assessed for climate change vulnerability.

Species	Breeding habitat vegetation association	Habitat quality indicator
Great Plains toad (Amphibiaweb 2008; Degenhardt and others 1996; Stebbins 1985)	ponds, wetlands	Pond longevity
Woodhouse's toad (Amphibiaweb 2008; Degenhardt and others 1996; Stebbins 1985)	ponds, wetlands	
Barred tiger salamander (Amphibiaweb 2008; Sarell 2004)	Riparian forest areas and wetlands	Emergent vegetation and detritus for egg deposition and zooplankton as larval food sources Pond longevity
Couch's spadefoot toad	Chihuahuan desert scrubland, mesquite and creosote flats, mesquite Bosque, shortgrass prairie, and irrigated agricultural lands	Pond longevity
New Mexican spadefoot toads (BISON-M 2009; Degenhardt and others 1996)	grasslands, shrublands, occasionally, open forest and irrigated agricultural lands	Pond longevity
Plains spadefoot toad (Degenhardt and others 1996)	Grasslands, avoids river bottoms	Pond longevity
American bullfrog (Degenhardt and others 1996)	Often associated with dense vegetation and generally areas below 2100 m	Tadpoles prefer warm water, which allows them to develop more quickly and avoid predators.
Northern leopard frog (Degenhardt and others 1996)	Permanent ponds and wetlands. Associated with dense vegetation.	Quantify of aquatic vegetation and lack of predators.
Western chorus frog (BISON-M 2009; Degenhardt and others 1996)	Wide range of habitats and elevations in meadows, lakes and flooded fields	Unknown

vegetation associations used during non-breeding seasons (Table 1A; Degenhardt and others 1996; Amphibiaweb 2008; Stebbins 1985). In contrast, we predicted increases in both breeding and non-breeding terrestrial habitats for the spadefoot toads. In general, terrestrial habitat associations of the spadefoot may increase with the expected decrease in canopy forest (Degenhardt and others 1996). However, vegetation association may be a poor predictor of future spadefoot habitat as these toads occur in a variety of habitats and more strongly associate with soil conditions than vegetation. Therefore, effective increases in associated habitat will not necessarily benefit spadefoot toads as was predicted in this assessment.

Habitat Components: We assumed that ponds in general will become less abundant under future conditions and considered ephemeral water sources especially (as compared to permanent water bodies) at greater risk of becoming rarer under future conditions (Chapter 3, Box 2). Two species—the American bullfrog and northern leopard frog—use permanent water sources almost exclusively. We gave them both a vulnerability score for potential losses in breeding habitat component due to the likelihood of decreases in all water sources (BISON-M 2009). The bullfrog has an especially long requirement of 1-2 years to complete metamorphosis and so is more reliant on long lasting water sources, whereas leopard frogs are known to have more versatile site selection behaviors (Bury and Whelan 1984; Degenhardt and others 1996). The remaining species fall out between those that use a variety of permanent, semi-permanent, and temporary water bodies (the Great Plains and Woodhouse's toads, Amphibiaweb 2008; BISON-M 2009; Degenhardt and others 1996), and those that exclusively use temporary breeding ponds (the spadefoot toads, BISON-M 2009; Degenhardt and others 1996; Tiger salamanders, Amphibiaweb 2008; Sarell 2004). Mortality of spadefoot tadpoles is high where the availability of ponds is reduced (Degenhardt and others 1996). We considered most non-breeding habitat components unlikely to be affected by climate change. The Great Plains toad, New Mexico spadefoot toad, Plains spadefoot, Couch's spadefoot toads, and Woodhouse's toad need loose soil or animal burrows to aestivate (Amphibiaweb 2008; Degenhardt and others 1996; Ruibal and others 1969; Stebbins 1985). These soil properties are not expected to be affected by climate. However, we did find that requirements for moist soils could result in a vulnerable score (Appendix 1). For instance, deep, friable soils are needed for the adult barred tiger salamanders to aestivate (Amphibiaweb 2008). Similarly, we considered the reliance of the western chorus frogs on protective features that could be prone to declines in the future as a predictor of potential vulnerability. Northern leopard frogs use permanent deep ponds, lakes, or streams for hibernation (BISON-M 2009), leaving them vulnerable to non-breeding habitat component loss as well.

Habitat quality: Habitat quality scores were based on our expectations of the effect of changing pond longevity (or persistence) and of fire and temperature pollution on amphibian survival and reproductive success. Pond longevity is positively associated with higher survival of Couch's spadefoot toad larvae, Great Plains toad larvae, and New Mexico spadefoot toad larvae (BISON-M 2009; Degenhardt and others 1996). In addition, Great Plains toad tadpole development is slower in crowded conditions (Krupa 1994), a situation likely to increase as ponds become less available. We predicted that most larval amphibian forms will suffer from exposure to silt, increased ultraviolet (UV) associated with vegetation loss, and overly warm water. Higher water temperatures tend to increase the rate of development in ectotherms, including amphibians, which could reduce exposure to predators and may counter some of the negative consequences of increasing water temperatures for egg mortality. For example, barred tiger salamanders require precipitation that is adequate and sustained enough to provide ephemeral water bodies lasting for at least 4 months during the warm season (Sarell 2004). Survival of salamander tadpoles is positively associated with precipitation and negatively affected by drought (Church and others 2007).While increased temperatures and longer droughts would be disadvantageous to salamanders, earlier metamorphosis due to warmer waters may assist individuals in avoiding predation and surviving low water conditions (Amphibiaweb 2008). In the case of the salamander, we considered potential drying of critical habitat and risk of increased tadpole mortality to be of greater importance than the benefit of accelerated rate of metamorphosis. We also considered sedimentation and ash deposition resulting from fire an issue contributing to the vulnerability of the salamander and frog species. We also believe decreased aquatic vegetation due to drier conditions would decrease habitat quality for the northern leopard frog and barred tiger salamander because vegetation density was associated with increased tadpole survival. Vegetation surrounding ponds buffers water from extreme temperature change, provides food, and may reduce UV exposure, all of which negatively impact frog species (Amphibiaweb 2008). In addition, water fluctuations during breeding season (mid-April-July and September-October) is known to reduce reproductive success for the northern leopard frog (Gilbert and others 1994). We did not consider decreased vegetation to be an issue for the Great Plains toad or American bullfrog, which tend to associate with greater vegetation cover and height or more dense vegetation, because effects on reproduction or survival are unknown (Anderson and others 1999). The American bullfrog was the only species considered resilient to quality changes under climate change. Specifically, this species might benefit from warm water, which is known to improve metamorphosis of tadpoles. However, bullfrog eggs are sensitive to very high temperatures, and reduced vegetation and pond availability as well as other conditions noted above may have negative impacts to this species, as is predicted for the other amphibian species we assessed.

We scored all amphibians neutral on elements of non-breeding habitat quality (e.g., depth of soil). Deeper soils are associated with better survival of some species (Couch's spadefoot toad, New Mexico spadefoot; Plains spadefoot, Ruibal and others 1969), but soil depth is unlikely to be affected by climate change.

We considered amphibians that rely on mesic environments for movement dispersal to be limited because the xeric habitats surrounding the river act as effective barriers to dispersal. We also relied on information regarding species' behaviors, such as site fidelity, that may limit dispersal potential when identifying dispersal limited species. Smith and Green (2005) reviewed dispersal distances of many amphibians and found that toads (*Bufo* spp.) were able to move up to and well over 1 km. In contrast, Semlitsch (2000) considered aquatic-breeding amphibians only able to disperse within 1 km of their developmental site and migrations limited to within 200 m of aquatic breeding habitat. We considered the American bullfrog neither resilient nor vulnerable due to its dispersal capacity; it can move up to 159 m in one night and has been found in isolated temporary ponds (Degenhardt and others 1996), suggesting that this species can move overland to new ponds. We also gave the Great Plains toad a neutral score because we found no information regarding site fidelity, and the species has been reported to migrate 1.6 km to breeding sites. Conversely, we considered northern leopard frogs vulnerable due to dispersal limitations—though they can migrate 2-3 km (BISON-M 2009), leopard frog movements are restricted to riparian corridors and it is unlikely that they can circumvent dams and other barriers to move into other stream reaches. Northern leopard frogs also exhibit high site fidelity: 98% of breeding adults returned to their home pond after a 1-km displacement (Degenhardt and others 1996). Although spadefoot toads are also known to disperse 1-2 km, there is little information on long-range movements (James 1998). However, we considered spadefoot toads' dispersal to be limited because adults exhibit high site fidelity, which limits their potential movement activity over time (Greenberg and Tanner 2005), and long-distance (2 km) movements are associated only with juvenile dispersal from breeding wetlands after metamorphosis (Klassen 1998; Landreth and Christensen 1971). The remaining amphibians—western chorus frog, Woodhouse's toad, and tiger salamanders—were similarly considered vulnerable to dispersal limitations (Amphibiaweb 2008; Sarell 2004).

We did not consider migration an issue for amphibians inhabiting the Middle Rio Grande (MRG). Seasonal movements of species, as seen with the American bullfrog, are probably limited and transitional habitats are not required.

Physiology Scores

We scored species based on information on thermal maxima and sensitivity to temperature extremes for larval, juvenile, and adult stages.

Physiological limitations: We did not consider evidence strong enough to predict either vulnerability or resilience of amphibians to rising temperatures or changing precipitation. All amphibians exhibited a broad range of tolerances although a number of unique characteristics among these species are worth noting. We based our scores on information regarding current range, which we used as a proxy for climate suitability (Bagne and others 2011) and known temperature thresholds. The American bullfrog is generally considered a warm adapted species that prefers temperatures above 26 °C (Bachmann 1969). However, egg development is impaired in water above 31 °C (Degenhardt and others 1996), which represents the lowest threshold of the amphibians in this assessment. The spadefoot toads appear most tolerant of both extreme temperatures and limited rains (Dimmett and Ruibal 1980b; Moore 1937; Wasserman 1970). Couch's spadefoot and the New Mexico spadefoot toad exist at the northern edge of their range (Degenhardt and others 1996), which may indicate these species are limited by minimum temperatures. Thus, warming temperatures could increase suitable habitat in more northern latitudes. However, these toads do not develop normally in water above 32.5 °C indicating an important maximum temperature threshold that could limit the benefit of warm weather. Conversely, the Plains spadefoot toad and the northern leopard frog are near the southern extents of their distributions (Degenhardt and others 1996) and have lower thermal maxima, which could indicate they are limited by high temperatures.(Zweifel 1968). The critical thermal maximum of the barred tiger salamander is reported as 36.5 ˚C for aquatic and 37.5 ˚C for terrestrial habitats (Smith 2003) though dehydration lowers these thresholds (Claussen 1977). Western chorus frogs occur throughout North America though only in the northern part of New Mexico. Critical thermal maximum of Western chorus frogs is around 37 °C. Though Woodhouse's toads range from Montana to Mexico and across North America and typically occur at lower elevations (900 to 2400 m) in New Mexico (Degenhardt and others 1996), juvenile toads avoid high temperatures (above 32 °C) and show greater preference for lower temperatures when food is limited or under dry conditions (O'Conner and Tracy 1993). We considered species such as the barred tiger salamanders and woodhouse toad to be able to reduce their exposure to very high temperatures through behavioral mechanisms (e.g., burrowing). In addition, the nocturnal amphibians, including the Couch's spadefoot, may be buffered to some extent from temperature extremes.

Extreme Events: We considered most amphibians sensitive to drought and likely to experience increased mortality. In the Habitat section, we expected drought to lead to reduced pond availability and duration, which are considered elements of habitat availability and quality, respectively. Here, we associate drought with massive population declines of several amphibian species, including the barred tiger salamanders (Rustigian and others 2003), Couch's spadefoot toads (Tinsley and Tocque 1996), Great Plains spadefoot toads (Bateman and others 2008), and New Mexico spadefoot toads (Cottonwood 1986; Degenhardt and others 1996). As a result, we predict that the increase in droughts and, in particular, the more intense and longer droughts projected for the Southwest will lead to increased amphibian mortality. The American bullfrog and the Woodhouse's toad were not considered vulnerable to drought-related population declines because their reliance on permanent water bodies was assumed to afford them greater protection from drought related mortality.

We did not find strong evidence supporting negative impacts from floods, heat waves, fire, or storms. Flooding is expected to be more intense and earlier in the season but not necessarily more frequent. Floods were considered a potential issue for the American bullfrog, but flooding is not known to be associated with direct mortality and so we did not incorporate this into a vulnerability score. Flooding may benefit Woodhouse toads, at least in the short term, as this species is tolerant of habitat degradation, but this effect was not expected to be a strong benefit.

Activity Periods: We predicted that some species will be buffered from extreme temperatures by water or through nocturnal habits (e.g., American bullfrog and western chorus frog) and would not experience a reduction in favorable conditions for foraging or reproduction. Conversely, we expect most terrestrial species, such as spadefoots or adult barred tiger salamanders, to experience reductions in potential active periods due to reduced precipitation. For these species, wet or high humidity is required in order to engage in above-ground activity. For example, the barred tiger salamander and Woodhouse's toad are known to reduce active periods as a result of reductions in moisture (Amphibiaweb 2008; Degenhardt and others 1996). The number of days needed to forage and build sufficient reserves for aestivation ranges from 3 days for Couch's spadefoot (Degnehardt and others 1996), 8 days for the New Mexico spadefoot, to 22 days for Plains spadefoot (Dimmet and Ruibal 1980). The Plains spadefoot toad appears to be the most flexible regarding activity need (Degenhardt and others 1996) and will emerge during droughts when humid conditions exist (Amphibiaweb 2008; Bragg 1965; Ruibal 1969).

Hibernation/aestivation/torpor: Animals that hibernate or aestivate in buffered hibernacula, or hibernation locations, are considered more resilient to climate change than those that do not hibernate or that use hibernacula exposed to environmental variations. We considered the western chorus frog to be vulnerable to weather variation during hibernation because its hibernacula are exposed to environmental variations. We scored the remaining amphibians as neither at benefit nor increased vulnerability because their hibernacula exist in either water bodies (bullfrogs and northern leopard frogs, see Degenhardt 1996; Emery and others 1972) or burrows (spadefoot toads and barred tiger salamanders, see Degenhardt and others 1996; Ruibal and others 1969), which we considered buffered from changes in ambient temperature. Importantly, the Great Plains toad and barred tiger salamander use active metabolic inhibition, which could increase the cost of hibernation, i.e., burn up resources more quickly, under warming temperatures (see Chapter 3).

Metabolic advantage: Species that utilize long-term food and water reserves are better able to deal with increased variability in resources as a result of climate changes. In addition, we consider ectotherms better adapted to resource variations than endotherms due to their inherently lower metabolic rates. Thus, we gave all amphibians a score indicating resilience with respect to metabolic requirements. Additionally, we found that the Great Plains toad and the spadefoot toads are able to store energy and water over multiple months, thus allowing them to survive long periods of drought (Dimmett and Ruibal 1980b; Ruibal 1962).

Alternative life forms: Some species have phenotypic plasticity which may allow them to adjust to in their environment. Among amphibians we assessed, spadefoot toads are able to express two types of morphs in response to environmental conditions (BISON-M 2009; Degenhardt and others 1996), yearling American bullfrog males can use alternative mating strategies in response to crowding (Howard 1984), and larval forms of tiger salamanders have the capacity to become sexually active in long standing ponds (Amphibiaweb 2008). However, we only considered the morphological variations present in larval spadefoot toads as potentially valuable adaptations for future climate conditions; we gave all other species a neutral score. Spadefoot toads will develop into either a fast growing and carnivorous larval form or a slow growing and omnivorous larval form to cope with differences in pond longevity and food availability. Ephemeral pond conditions favor carnivorous larvae (Phennig 1992). Interestingly, in Couch's spadefoot, larvae that grow quickly, as during drier periods, have less longevity than the slow-growing forms (Tinsley and Tocque 1996).

Capacity to outlive limiting conditions: Drought was the primary limiting condition for amphibians, and we use drought duration of 5 years to define the period of limitation that might be expected under climate change. We only considered the American bullfrog, which is estimated to live 8-10 years, likely to survive drought and successfully reproduce. We considered spadefoot toads unlikely to benefit or experience increasing vulnerability due to future drought conditions. Spadefoot toads are long-lived and probably not limited by duration of drought. However, only 5% of the Couch's spadefoot toad population lives to 10 years and beyond (Tinsley and Tocque 1996). Furthermore, adult survivorship and larvae development rates of Couch's spadefoot appear to be negatively affected by drought (Tinsley and Tocque 1996). However, this species can assimilate food to survive hibernation very quickly (Dimmett and Ruibal 1980b) and carnivorous larvae develop quickly. Ultimately, we predict these species are unlikely to see a change in limiting conditions because, though negatively impacted by drought, these toads are adapted to the desert environment in which they live.

We considered western chorus frogs, barred tiger salamanders, northern leopard frogs, and Great Plains toads vulnerable to limiting condition that could arise in population decline. Though the Great Plains toad often lives 10 years and may live 20 years in Canada (James 1998), information from the Sonoran desert, which has similarly dry conditions of the MRG, indicates maximum lifespan of 6 years (Sullivan and Fernandez 1999). One captive northern leopard frog lived 6 years (BISON-M 2009).Tiger salamanders typically live 2-5 years (Sarell 2004). Western chorus frogs have very high juvenile and tadpole mortality rate (BISON-M 2009). Longevity of western chorus frog and woodhouse toads is unknown though it is probably only a few years (Amphibiaweb 2008).

Phenology Scores

We considered amphibians vulnerable to phenological changes if they: (1) relied on climate cues to initiate activities (breeding, hibernation); (2) relied on resources that occurred during limited times in a year; or (3) dealt with spatial or temporal separation between initiation of activities and availability of resources. We found the American bullfrog to be most resilient to phenological changes. Overland movements were not considered critical to survival or fecundity in MRG habitat, there was no temporal variation in events related to survival or fecundity, and the bullfrog does not appear to rely on cues or have critical timing issues between activities. Typically, species that required rain events for overland movement (e.g., barred tiger salamander) or breeding ponds (e.g., spadefoot toads) were most vulnerable to potential phenology effects. Activities such as breeding and emergence are likely closely tied with rainfall and temperature for the western chorus, spadefoot toads, Great Plains toad, and tiger salamander (Amphibiaweb 2008; Degenhardt and others 1996; Dimmett and Ruibal 1980a; Ruibal and others 1969). Reproduction of woodhouse toads and northern leopard frogs is cued to temperature (BISON-M 2009; Bragg 1940). Emergence from hibernation and aestivation is also triggered by environmental conditions or signals for some species (e.g., in spadefoots, frequency vibrations associated with heavy rain and thunder; Dimmett and Ruibal 1980a). Salamanders use post frost period to cue emergence and rainfall cues for migration of both juveniles and adults to and from breeding ponds.

We did not find evidence for amphibian reliance on timed food resources but there were several examples demonstrating the importance of precipitation and temperature on activity. Temporary pond formation is important to Western chorus frogs, spadefoot toads and timing of frost and precipitation is important for overland movements and activities of the spadefoot toads, Great Plains toad and tiger salamander (Amphibiaweb 2008; Degenhardt and others 1996; Ruibal and others 1969).

We did not find evidence for negative impacts of distantly placed resources and cues among the amphibians (see Bagne and others 2011 for discussion). For spadefoot toads, cues such as rainfall and vibrations from rain drops are directly related to availability of resources such as ponds and termites. In addition, spadefoot toads are explosive breeders with high population increases during favorable times and are not likely to have large temporal variation in breeding initiation (Degenhardt and others 1996).

We considered the couch's and New Mexico spadefoots, Great Plains toad, western chorus frogs, barred tiger salamander, and northern leopard frogs, which have one breeding event per year, vulnerable to timing mismatch during breeding (BISON-M 2009; Sarell 2004). Though the barred tiger salamander might be able to have two reproductive events if habitat is available (Allison and others 1994), Church and others (2007) suggested the latter is prohibitively costly for females and this was not considered a common trait in our assessment. We considered the Plains spadefoot, Woodhouse's toads, and American bullfrog resilient to breeding timing mismatches because they can all breed more than once a year when conditions are favorable (Bragg 1944; Degenhardt and others 1996; Klassen 1998) or have extended breeding seasons (woodhouse toad, Woodward 1982).

Biotic Interaction Scores

Food: We did not find clear evidence to support definitive predictions of increasing or decreasing interactions for amphibians and gave many questions a neutral score. For many amphibians, we could not predict trends because most species had a diverse prey base (BISON-M 2009; Bragg 1964; Degenhardt and others 1996). One study from Canada indicated that stream invertebrates may be reduced by increased temperatures (Hogg and Williams 1996) and higher water temperatures may inhibit invertebrate food populations, but other prey items are likely to remain available.

Interestingly, some of the strongest (and most negative) expectations for future interactive effects regarded the American bullfrog, which is an important predator and competitor for many species. The American bullfrog preys on wide variety of invertebrates, snakes, rodents, frogs, and salamanders and often eats beetles and snails in New Mexico (Degenhardt 1996). The American bullfrog has various predators and, though adults are relatively unpalatable to fish, tadpoles are eaten by salamanders, other frogs, and adult bullfrogs.

Predators: We did not find clear indications for changed impact by predators because predator diversity is high for most amphibians and we have little information regarding predator impacts on populations (Amphibiaweb 2009; Degenhardt and others 1996). One exception is the northern leopard frog, which we expect to be vulnerable to potential increases in bullfrog predation. Bullfrogs and introduced fish are major predators of leopard frogs and are blamed in part for population declines in the northern leopard frog (BISON-M 2009). Predators also pose a significant threat and considerable impact on salamanders. However, warmer temperatures are conducive to nocturnal migration movement of salamanders, which is thought to be a means of predator evasion (Amphibiaweb 2008). Ultimately, we did not feel that we could attribute a change in predation pressure for the salamander given the current available information.

Disease: We found disease a threat for many species under climate change. Increased crowding due to reductions in pond area and availability increases the overall risk of disease for amphibians. The risk of local extirpation for population will also increase due to pollution as climate limits alternative habitats for this species. The threat of Chytridiomycosis, associated with major amphibian declines, may increase with crowding in permanent water sources. Bullfrogs are resistant to Chytridiomycosis but an efficient carrier of the pathogen (Daszak and others 2004). We considered leopard frogs, western chorus frogs, and Woodhouse's toads (BISON-M 2009; Bradley and others 2002; Milius 1998; Voyles and others 2007) vulnerable to increased risk of chytrid fungus infections. Though chytrid fungus is known to impact Great Plains toad (Muths and others 2003), we did not consider it vulnerable to increased disease. We considered salamander populations vulnerable to increased bacterial diseases and fungus (*Batrachochytrium*) as well as Ranavirus (Sarell 2004) under climate change. No known diseases are known to result in widespread mortality for Couch's spadefoot toads, the New Mexico spadefoot toad, and the Great Plains toad.

Competitive: We did not find evidence for changing competitive interactions for any species except the northern leopard frog (BISON-M 2009). We considered the northern leopard frog at risk of negative impacts from the American bullfrog, especially in permanent water bodies. American bullfrog tadpoles generally outcompete native tadpoles (Kupferberg 1997) and though bullfrogs are prone to predation and exclusion from fish (Sarell 2004), the extent of this competitive relationship is not clear and fish population response will vary by species. The Couch's spadefoot and Plains spadefoot toads may compete with various other temporary pond breeders (mostly spadefoots), but community structure is generally determined by competitive interactions that primarily exist within species (Dayton and Fitzgerald 2001).

Symbionts: We found no evidence of significant symbiotic relationships within this group of animals. Woodhouse's toads hybridize with some other species of toads including Great Plains toads (Amphibiaweb 2008). Rodents, such as *Perognathus parvus* (Great Basin pocket mouse) and *Thomomys talpoides* (pocket gophers), have been identified as important because salamanders use their burrows (Sarell 2004). However, burrowing rodents as a group are somewhat more resilient to climate change and we did not expect their populations to change much in the future.

Reptile Results

Habitat Scores

Habitat scores considered decreases and increases for associated vegetation, required habitat components, changes in habitat quality, requirements for additional habitat and capacity of species to move with spatial and temporal transitions in critical habitat features.

Vegetation Association: We expect riparian associates including checkered gartersnake, blacknecked gartersnake, desert spiny lizard (Menke 2003; Vitt and others 1981), Chihuahuan spotted lizards (Catron and others 2008), and checkered whiptail to be negatively impacted by declines of associated habitat. Coachwhips, though they depend on open grasslands and thick clumps of vegetation, also are reported to be sensitive to fragmentation (Mitrovich and others 2009), and we considered this species likely to experience disruption to their preferred habitat with potential negative effects on population size.

We did not expect species with diverse habitat associations to be negatively affected by declines in associated vegetation types. Specically, we did not consider the southwestern fence lizard, New Mexico whiptail, western whiptail, glossy snake, racer snake, gopher snake, and many-lined skink at risk of loss of primary habitat associations (BISON-M 2009; Brennen 2008; Catron and others 2008). We also could not determine a clear negative outcome of vegetation changes on a few species with more selective habitat needs. Though the common gartersnake is considered a semiaquatic species, Zimmerman (2002) noted that this species is adapted for extreme conditions (e.g., very hot, dry) and inhabits a variety of vegetation types. Therefore, we did not consider common gartersnake exclusively dependent on riparian habitat, so loss of riparian habitat was not considered a risk to populations. Similarly, we did not consider the terrestrial gartersnake likely to experience negative impacts due to loss of riparian woodlands or conversion of plant communities in the MRGB. Information was very limited for the Plains black-headed snake though its uncommon appearance in the MRGB may indicate that riparian habitat is not critical for the presence of this species. We considered it neither vulnerable to nor likely to benefit from future changes (Catron 2008). We also expected relatively little change to habitats important to the western diamond-backed rattlesnake, the most common rattlesnake species in the MRGB (Catron and others 2008), and the prairie rattlesnake. The desert grassland lizard was the only species whose association with the MRGB appears to be related to the presence of current xeric sites (sandy bottoms, desert grasslands), and we expect its preferred xeric habitat to increase.

Habitat components: Important habitat components primarily included moist microhabitats for egg laying and incubation. We considered the many-lined skink, Great Plains skink, glossy snake, the racer, western hognosed snake, common or desert kingsnake, coachwhip, and all gartersnakes at risk of loss of important moist nesting habitats (see BISON-M 2009; Catron and others 2008; Fitch 1955; Forest Encyclopedia Network). The remaining reptiles, which did not have a clearly identified reliance on moist nest sites, were given neutral scores.

Habitat Quality: In some instances, we could not find a clear association of species with riparian vegetation or specific components like moist nest sites but did determine species that relied on microhabitats provided by the river. In such cases, we considered species vulnerable to increasing aridity, which results in a decline in the quality of habitat. For instance, the ornate box turtle is expected to experience declining habitat quality through decreased vegetation cover, which has been associated with increased death of juveniles. In another example, the Great Plains skink is known to prefer open canopy thick grasslands of the MRGB (Catron and others 2008; Dawson 1960). Though tree decline will probably favor the presence of the skink, we ultimately expect drier conditions and reductions in the amount of thick grass habitat to leave the skink more vulnerable to population declines.

Dispersal: We considered both potential dispersal distance as well as capacity to travel overland in our judgment of species vulnerability to habitat shifts. In general, we considered species that were unlikely to traverse the xeric habitat surrounding the MRGB to be dispersal limited. Most reptiles, including lizards and skinks, fell into this category. For these species, we expect movements to be largely restricted to northern movements along the riparian corridor but to be additionally limited by barriers including dams, roads, and other structures that will effectively limit potential northward expansions. For instance, turtles are known to travel distances of 10 km, but we generally considered them dispersal limited because their movements are limited to the riparian corridor and they are unlikely to surmount barriers such as dams (e.g., western painted turtle, COSEWIC 2006; Big Bend slider, Stuart and Ward 2009). We also considered the red-eared slider, an invasive species, dispersal limited because it only disperses to leave its nest or hibernate and spends most of its time in water (Dewey and Kuhrt 2002). We considered most snakes, including both rattle snakes species, to disperse distances sufficient enough to allow them to relocate to new habitat. The Animal Diversity Website (NatureServe) noted that colubrids are known to move between areas of up to a few kilometers apart, and medium-sized colubrids, such as gartersnakes, are able to move 1-2 km. The only snake considered potentially dispersal limited was the Plains black-headed snake (*Tantilla nigriceps*) because of its small size and secretive fossorial nature.

Physiology Scores

Physiological limitations: Limiting conditions for reptiles include potential negative impacts of high temperatures on egg and juvenile survival and lack of adequate water. We found evidence that higher temperatures could result in greater egg and hatchling mortality, especially in the face of increasingly arid conditions for turtles. Specifically, increased temperatures are likely to negatively impact species with soft eggs, such as the snapping turtle, ornate box turtle, western painted turtle, and Big Bend slider, which are vulnerable to temperature effects on egg survival (see Finkler 1999, 2001; Redder and others 2006). In addition, hatchlings were often cited as sensitive to high temperatures (e.g., western painted turtle, snapping turtle, see Brooks and others 1991; Cagle and others 1993; COSEWIC 2006; Finkler 1999, 2001), drought-related decreases in vegetation cover (e.g., ornate box turtle, see Nieuwolt 1996; Redder and others 2006) or both (e.g., Big Bend slider and red-eared slider, see Demas and others 2008; Governor's Drought Task Force; Stuart and Ward 2009). Water availability affects incubation time, hatchling size, and mobilization of yolk reserves in the red-eared slider such that drier conditions will result in reductions in post hatchling locomotor performance, growth, and survival (Dema and others 2008). We considered the gopher snake at risk of experiencing temperatures in excess of its thermal maxima and at risk for reduce forage time due to increasing daytime temperatures (Diller and Wallace 1996).

For other reptiles, we did not find enough information regarding thermal maxima to determine adult reptile species vulnerability. Commonly, reptile species had thermal maxima ranging from 39.9 to 43 ˚C, which we did not consider a limitation under future conditions. We also did not consider aquatic diurnal species at risk of reduced foraging time since presence of water allows them to buffer themselves from extreme temperatures. Though experiments on gartersnake species (*Thamnophis radix, Natrix sipedon, Elaphe o. obsoleta,* and others) indicate that very few snakes can survive temperatures over 40 °C (104 °F) (Leuth 1941), we did not consider them vulnerable to limiting conditions related to temperature extremes. We did find a number of negative consequences of note for reptiles regarding warming temperatures. First, there may be potential metabolic consequences for snakes because metabolism increases with increasing temperature such that the length of life of starved snakes is highly dependent on temperature (Leuth 1941). Second, we found one article noting that gartersnakes maintained at higher temperatures were prone to eye and skin infections though the study did not specify the agent (Leuth 1941). Third, more energy is spent when ambient temperatures are high, so warmer climates will likely shorten hibernation period. Fourth, warmer weather may accelerate onset of reproduction in gartersnake species (Seigel and others 2000). Gibbons and others (2000) noted that empirical evidence from studies of freshwater turtles shows that increased temperatures increase juvenile growth rates and onset of maturity and results in skewed sex ratio. The collective impact of these effects may cause substantial changes in the reptile fauna within the MRGB and across the Southwest in the future.

We did not consider most lizard species, which generally show a high tolerance to high temperatures and exhibit behavioral avoidance mechanisms, vulnerable to future extreme conditions (e.g., desert spiny lizard, Vitt and others 1981; the whiptails, *Aspidoscelis inornata* and *A. gularis*, Winne and Keck 2004). However, we did find evidence for population change in side-blotched lizards (*Uta stansburiana*) in response to changes in precipitation (Parker and Pianka 1975) and considered this species at risk of facing physiologically limiting conditions under future scenarios.

Sex Ratios: All turtle species were considered at risk of population decline due to skewed sex ratios. This was the only taxonomic group to be scored as vulnerable to this potential effect.

Extreme Weather: Many species showed sensitivity to extreme weather events (Appendix 1; Table 4.2). Spiney soft-shell turtles may be susceptible to flooding and subsequent changes in sandbar availability (BISON-M 2009). Drought negatively impacted western whiptails in California, which experienced a population crash in response to the negative effects of reduced vegetation as a result of extreme drought on whiptail hatchlings (Pianka 1970). Reproduction in *Thamnophis marcianus* was similarly reduced following two drought years, though for this species it was thought to be due to effects of drought on prey base (Ford and Karges 1987). We considered all gartersnake species likely to be negatively impacted by increasing drought.

Fire is known to negatively impact reptile species, though we were not always able to predict a clear negative effect on our assessed species. The side-blotched lizard may not be well adapted to fire but vegetation removal (via grazing) had varying effects on populations depending on location (Barbault and Maury 1981; sources from BISON-M 2009), so we gave this species a neutral score for impacts from extreme events. In contrast, fire has been known to destroy the dens of the prairie rattlesnake and we considered future increases in fires a threat for this species as well as the western diamondback rattlesnake, which has similar denning habits.

Hibernation/Fat Storage: All reptiles in this study hibernate, which is a characteristic associated with increased resilience because it is often used to survive periods of resource limitation (Bagne and others 2011). For warm-blooded mammals that remain active through a range of temperature, food limitation is a plausible mechanism for the evolutionary development of hibernation. However, reptile hibernation is not clearly derived from resource limitation because ectotherms experience reduced metabolic functions as a result of dropping temperatures. Therefore, reptiles must hibernate during cold months, whether or not food resources are available. Reptiles must gain sufficient fat during warm months to sustain themselves through cold periods when activity is not possible, and these fat reserves must be adequate enough to ensure egg production at the onset of warmer conditions. Meeting these obligations may be more difficult under more variable weather conditions that can affect food availability, and warmer temperatures during hibernation may increase the rate at which reptiles burn through fat reserves (see first question of this section). Future warming trends may extend the seasonal activity period for many ectothermic species and may eliminate periods of hibernation altogether. However, the risk of frost and storms may also increase for species active at earlier and later periods of the year with potential increases in mortality. For this assessment, we did not consider the capacity of species to hibernate during cold periods a benefit or source of vulnerability.

Activity Periods: Reductions in activity due to higher temperatures may be a source of vulnerability for many species that balance foraging and reproductive activities during limited time periods. For most reptiles, increased daily maximum temperature is recognized as potentially limiting daytime foraging opportunities and has been blamed for local extirpations of some lizard species (Huey and Tewksbury 2009). However, increases in seasonal activity periods could counter potential loss to daily activity. We gave many species a neutral score for this question despite compelling evidence for a potential negative effect on lizard energy budgets (Huey and Tewksbury 2009; Sinervo and others 2009) because of conflicts in predicted response that created large uncertainties. Still, we considered several species that exhibit clear avoidance mechanisms such as seasonal changes in forage behavior(e.g., side-blotched lizard, southwestern fence lizard, racer, and gopher snake) at risk of reduced activity due to higher daytime temperatures (Brennan 2008). Similarly, we considered the ornate box turtle, which is most active in mornings or immediately after rain (Niedzielski 2002; Redder and others 2006), sensitive to temperature induced reductions in activity in excess of benefit via increased growing season. Huey and Tewksbury (2009) indicated that the spotted Chihuahuan whiptails may have difficulties under warming regimes (Winne and Keck 2004), which influenced our score of this species under this category. Species are also sensitive to precipitation changes as well as daily temperature fluctuations. Both movement of snakes and prey availability (amphibians) are strongly tied to precipitation (Seigel and Ford 2001).

Outlive Limiting Conditions: Species that lived less than 5 years were considered vulnerable to prolonged limiting conditions associated with drought. Drought was considered a limiting condition for all reptiles in this study. All the turtles assessed in the study were long lived whereas many of the lizards were short lived (<5 years). Therefore, most turtles were considered to have some level of resilience to extended conditions such as drought that may result in reproductive loss. Conversely, lizards are generally not expected to outlive periods that could lead to reproductive failure and we considered these species at risk of population declines under increasingly severe droughts.

Phenology Scores

All reptiles hibernate in response to cold weather and we considered all species prone to risks associated with potential changes in cues used to initiate and emerge from hibernation. Temperature is probably the sole driver for onset of hibernation for most reptiles (e.g., snapping turtle, Ultsch 2008; box turtles, Niedzielski 2002; common gartersnake, Jacob and Painter 1980). As the timing of temperature cues change, there is an increased risk of mismatch with important food resources that may or may not be equally influenced by changing climate conditions. Further, reptile emergence needs to occur late enough to avoid lethal frosts (Seigel and others 2000) and premature warming before frost seasons have ended could have drastic and negative effects for many reptile species. We also noted that vitellogenesis (production of ova) in the checkered gartersnake appears to be tied to rainy periods in arid environments and/or when prey base is good, which is also a mechanism for increased mistiming issue (Seigel and others 2000).

Almost all reptile species exhibit behaviors that are indicative of reliance on timed food resources, including having a limited breeding season. However, this association was often not quantifiable and we gave many reptiles a neutral score for this trait (Appendix 1). Distinctions were made between species that fed on animals versus those that grazed water plants. Turtles often show a greater capacity to adjust the timing of egg and hatchling stages and we considered this group more resilient to phenological risks than other reptiles. We gave turtles a neutral score for effects of resource timing. Overland movement of the checkered gartersnakes is associated with wet periods (Seigel and others 2000), and we gave this species a positive score because this association indicates some reliance on appropriately timed rainfall for necessary dispersal periods.

We did not consider any reptile species at an increased risk of asynchrony due to spatially or temporally distinct timed events because none of the reptiles migrate. In addition, reptiles exhibit a number of traits that may help them deal with temporal separations in resource availability. For instance, snakes are able to undergo atresia and reabsorption of vitelline follicles, which helps females adjust clutch size prior to ovulation, thus reducing potential energy losses (Ford and Karges 1987). Such flexibility is important for species like the reptiles, which can have long incubation or parturition times that may leave them at an increased risk for mismatch between initiation of breeding and conditions for hatchlings. As an example, in the checkered gartersnake, ovulation occurs approximately 2 months after initiation of egg production and birth occurs about 3 months after ovulation (Ford and Karges 1987; Seigel and others 2000). Though initiation of egg production is tied to weather conditions, ovulation and birth are not and conditions may change drastically over this time period.

Finally, we consider species that are able to produce multiple clutches or reproduce over a large period of time in a given year (extended breeding season) to be more resilient to potential risks associated with mismatch of timed resources and reproductions. Reptiles were nearly evenly divided between those species that can vary timing of annual reproduction and those that time reproductive events with critical resources (Appendix 1). The Great Plains skink had one of the most limited reproductive periods of just a few weeks (Fitch 1955). Less was known about the black-headed snake, but we assumed it produces a single clutch per year and has a short life span given its small size.

Biotic Interactions Scores

We found the strongest evidence for climate effects on biotic interactions for reptile species that prey upon amphibians and, specifically, anurans. Three species are expected to experience food declines as a result of climate change: western hog-nosed snake, black-necked gartersnake, and checkered garter snake (Seigel and Ford 2001). Females of other *Thamnophis* spp. tend to have less variability in clutch size and timing of reproduction in environments with more consistent food resources (frog tadpoles in manmade ponds). They are also receptive to males within a day of emerging for hibernation. In contrast, females in environments with periodic variation in prey base show greater variation in clutch size and number of clutches per year and are not receptive for at least 28 days after emerging for hibernation (Seigel and Ford 2001). Thus, reproductive strategies, though they could reflect an adaptive response, could also reflect the influence of variable food sources on clutch size (i.e., spring clutch characteristics are determined by previous years [Seigel and Ford 2001]).

Bird Results

Columbiformes (doves and pigeons)

There was a single species in this order, the mourning dove (*Zenaida macroura*). The mourning dove was among the most resilient species scored, in large part because it was considered resilient to habitat and phenology changes relative to other species.

Habitat Scores

Habitat scores reflect expected trends (decreases/increases) for associated vegetation and required habitat components, changes in habitat quality, requirements for additional habitat, and capacity of species to move with transitions in critical habitat features.

Mourning doves breed in a wide variety of habitats (Otis and others 2008), and there was little indication that they will experience significant change in habitat under climate change. Mourning doves generally avoid breeding in dense forests, preferring to nest on the ground, on ledges, and in shrubs and trees. Although frequency of use of various substrates varies regionally, we expect this species to experience an increase in suitable nesting sites (Otis and others 2008).

Though there are few studies on winter habitat use, mourning doves are typically found in grain fields, parks, fields, and other open areas and tend to move between roost sites (woodlot areas) at night to feeding areas (e.g., grain fields) during the day (Tomlinson and others 1994). We expect them to see an increase in suitable wintering habitat. Mourning doves feed almost exclusively on the ground and require open habitats for foraging, generally avoiding to feed in areas with thick ground vegetation (Otis and others 2008). Therefore, we expect this species to benefit from warming trends, which will result in less vegetation in the MRG.

Mourning dove populations in the northern portion of the species' range are migratory, whereas those in mid-latitudes are partial migrants (Otis and others 2008). Thus, we did not consider this species prone to negative effects associated with reliance on migratory habitats.

Physiology Scores

We based physiology scores on expected responses to extreme climate, extreme weather events, potential limitations to daily activity, metabolic characteristics, adaptive capacity traits, and ability to outlive limiting conditions.

The wide range and habitat use that includes open and arid areas (e.g., deserts, urban and suburban areas, and farms and agricultural areas; see Otis and others 2008) of mourning doves suggests this species is not overtly impacted by high temperatures. With the exception of cases of extreme heat rises, we assumed that doves could tolerate increases of several degrees Celsius. The MRG is in the southwestern portion of mourning dove range in the United States, but doves are known to breed from coast to coast and from southern Canada to Panama. In addition, doves have many physiological coping mechanisms for high heat, including augmented cutaneous evaporation, panting, and gular flutter during high temperatures (Otis and others 2008). However, the dove requires surface water for drinking at least every day or so, which did not contribute to the scores of this assessment but is recognized as a potential impact on species presence (Slade 1969 in Otis and others 2008).

Nests are frequently destroyed by windstorms or rainstorms (Otis and others 2008); 33% of nestling and fledgling mortality was attributed to weather in Alabama (Grand and others 1984 in Otis and others 2008). Ten percent (20 of 195) of nest failures with a known cause in RMRS data were attributed to weather; nests were usually blown down during storms. However, we did not have clear evidence that climate change in New Mexico will involve greater frequency or severity of storms during breeding season in the MRG. Therefore, we did not consider the dove vulnerable to increased storms. However, we would like to note that Otis and others (2000) cited weather and predation as primary influences on nesting success.

Despite clear indications that this species is able "to reduce metabolic rate, body temperature, and heart rates when subjected to decreasing ambient temperatures below critical temperature of 30°C, i.e., becomes torpid when subjected to decreasing ambient temperatures and food restrictions (Otis and others 2008), we did not consider the dove among species able to benefit from metabolic variation because it was not clear to what extent this occurs in natural environment versus laboratory environments.

We considered doves to be vulnerable to prolonged drought and other limiting periods as individuals are expected to survive for only a single breeding season (Otis and others 2008).

Phenology Scores

We considered it likely that individuals nesting in New Mexico use photoperiod as an important cue for initiation of breeding activities. Given their long breeding season, they are likely tied less to "local predictive information" (Wingfield 2008) than other bird species. Doves primarily consume seeds, and we did not consider them reliant on pulsed resources to the extent insectivorous species rely on insect abundance. In part, we presumed that this species consumes seeds from the previous spring, and potentially fall during the non-growing season, although there are no records of caching behavior for this species. According to Otis and others (2000), weather and predation appear to be the primary influences on nesting success; thus, we considered changes in the timing of precipitation and increases in temperature as able to alter the fecundity for this species. We considered mourning doves resilient to timing mismatches during the breeding season because they can produce 3-6 broods per year (references in Westmoreland and others 1986).

Biotic Interaction Scores

We did not find any measurable or predicted vulnerability for doves in biotic interactions. Doves feed almost exclusively (99% of diet) on seeds (both cultivated and wild plant species; Mirarchi 1993a in Otis and others 2008). When available, cultivated crops are their main food (e.g., corn, wheat, sorghum, and sunflowers; Ibid). The diverse diet of doves precluded predictions for future food availability under climate change. Adults and young are preyed upon by raptors, mammals, and reptiles; falcons and accipiters are possibly the most common predators on adults; raccoons and domestic cats and dogs may be the most common nest predators but other predators have been documented (Otis and others 2008); and there are no direct effects expected from climate change. Trichomoniasis outbreaks have caused tens of thousands of deaths in the past and small outbreaks still occur (see references in Otis and others 2008). Outbreaks have often occurred during periods of hot, humid weather (Ibid). Though this disease is believed to be the leading cause of natural mortality in mourning doves, we did not feel that we could tie future outbreaks to climate trends at the time of this assessment. Doves are also prone to contract avian pox, which can be fatal (Ibid), but again we did not find a clear link between future conditions in the MRGB and known outbreaks of avian pox. Doves are not known to have strong competitive interactions (Poling and Hayslette 2006 in Otis and others 2008).

Strigiformes (owls)

The MRG has two owl species: the western screech-owl (*Megascops kennicottii*) and the great horned owl (*Bubo virginianus*). We found the screech-owl was more vulnerable to climate changes largely due to greater sensitivities in habitat and physiology categories.

Habitat Scores

Habitat scores reflect expected trends (decreases/increases) for associated vegetation and required habitat components, changes in habitat quality, requirements for additional habitat, and capacity of species to move with transitions in critical habitat features.

We considered western screech-owls at risk of breeding habitat declines but predicted no effect for great horned owls. Screech owls breed in a variety of desert and low-elevation woodland habitats, including residential and suburban areas (Cannings and Angell 2001), with the highest density of birds found in deciduous riparian woods. While existing human habitats will remain, new housing developments are not likely to be suitable (Cannings and Angell 2001) and declines in cottonwood-dominated Bosque will cause a decrease in preferred habitat. Great horned owls breed in a variety of habitats, including deciduous, mixed, or conifer forests, and show a preference for open and secondary-growth woodlands, swamps, orchards, and agricultural areas (Houston and others 1998). We do not expect these to change much for this owl.

We expect both owl species to see decreases in nest sites. Great horned owls require open areas and perches for hunting; nest sites are most often built by other bird species (e.g., corvids, raptors), but can include snags with broken off tops, holes in cliffs, and abandoned buildings and platforms (Houston and others 1998). Screech-owls have a general requirement for cavities for nesting and roosting and will use both natural cavities, including old woodpecker holes, and nest boxes (Cannings and Angell 2001). We considered both owls to be more dependent on larger trees than smaller trees. Older larger trees are likely more susceptible to mortality, and many in the Bosque are senescing with little regeneration (Howe and Knopf 1991). In addition, reduced water flows and increased fires will negatively affect the abundance of large trees capable of supporting the type of nest site observed for these owls. Therefore, we expect great horned owls to experience declines in winter roosting sites that include trees, snags, thick brush, cavities, ledges, and human structures (Houston and others 1998). Our expectations for a loss of trees and snags and an overall reduction in vegetation cover also contributed to this owl's score. In constrast, we did not expect screech-owl winter roosts, which include areas under bark and in caves and crevices (Cannings and Angell 2001) to change.

Reproductive success of great horned owls is tied to food supply and, particularly in arid or semiarid regions, to prey cycles and precipitation (Smith and Murphy 1979 in Houston and others 1998). Similarly, adequate food supply is critical for successful western screech-owl nesting (Cannings and Angell 2001). However, there is no information to indicate how survival varies by habitat with respect to factors tied to climate change. We therefore did not factor in a vulnerability score related to habitat quality.

We also did not expect migration or dispersal to be problematic in the future. Neither the great horned owl nor the western screech-owl migrates, and they are not prone to issues related to additional exposure to climate-mediated habitat changes (Cannings and Angell 2001; Houston and others 1998). Great horned owls in northern portions of their range have been known to undergo movements in response to population crashes of snowshoe hare indicating some capacity for irruptive movements (Houston and others 1998).

Physiology Scores

We based physiology scores on expected responses to extreme climate, extreme weather events, potential limitations to daily activity, metabolic characteristics, adaptive capacity traits, and ability to outlive limiting conditions.

Neither owl was considered at risk of reaching physiological thresholds in New Mexico. Both species have wide ranges within North America and inhabit a variety of climates and vegetation communities, indicating they are able to tolerate a wide range of precipitation and temperature conditions (Cannings and Angell 2001; Houston and others 1998). In addition, great horned owls are known to engage active and passive avoidance methods for dealing with high temperatures, including increasing core body temperature slightly (Chaplin and others 1984 in Houston and others 1998), panting, and perching in cooler microsites. Western screech-owls are able to use a larger portion of their upper respiratory track for cooling than some other small owls, leading to larger capacity for evaporative cooling. Still, owls are typically inactive during hot periods and will compress contour feathers and open wings to reduce body temperatures (Ligon 1969).

There were no instances of reported mortality due to extreme weather events for either owl species**.** However, screech-owl populations in southwestern Arizona declined 70% in 3 years during a drought (Hardy and others 1999). Mortality due to starvation is frequently associated with weather, either directly or indirectly. Because drought periods are expected to increase and nestlings are prone to starvation, we did consider the western screech-owl vulnerable to future changes in extreme weather (drought).

We did not expect either species to experience reductions to activity periods due to their nocturnal hunting habits (Packard 1954 in Houston and others 1998). It is worth noting, however, that there is some indication that owls are inactive during hot periods (Ligon 1969).

We did not consider either species in possession of adaptations for dealing with food or water shortages. Though there are reports of great horned owls in Yukon (Houston and others 1998) and western screech-owls in Idaho caching prey (Cannings and Angell 2001), we did not feel the caches would be accessible for more than a few days.

Only the western screech-owl was considered at risk of sustained limiting conditions. As with many other taxa, owls are expected to experience reproductive limitations related to drought. The great horned owl, possibly the longest lived of owls in North America (Houston and others 1998) (record for a banded individual was at least 28 years, 7 months [Nero 1992 in Houston and others 1998]), was considered likely to outlive limiting conditions represented by a 5-year drought. The oldest known western screech-owls in the wild lived more than 13 years (Clapp and others 1983), but the average life-span in one studied population was less than 2 years (Cannings and Angell 2001). For this reason, we considered the screech-owl susceptible to limiting conditions that could outlast individual reproductive opportunities.

Phenology Scores

We considered great horned owls and western screech-owls vulnerable to changes in timing of food resources and at increased risk of nest failure due to changes in phenology but resilient to changes in climate cues and issues related to distantly placed resources. Food supply influences reproduction and survival of great horned owls (Houston and others 1998) and western screech-owls (Cannings and Angell 2001). In addition, clutch size is positively tied to precipitation (sensu wet seasons versus dry season), presumably due to increases in food supply (Dixon 1914 in Houston and others 1998). Clutch size has also been positively correlated with jackrabbit and rodent abundance, respectively (references in Houston and others 1998). Changes in the timing of precipitation and increases in temperature have the potential to alter the timing of prey availability and abundance and hence fecundity. Each owl raises a single brood per year and though pairs may re-nest if their nest is destroyed or predated early in the breeding cycle (Cannings and Angell 2001; Marti 1969), we did not consider this capacity to represent a significant extension of the breeding period (hence a resilience). Great horned owls are a "predictable breeder," where it is thought that photoperiod is the predominant proximate factor leading to hormonal changes that drive reproductive events (Dawson and others 2001). Similarly, though juvenile western screech-owl dispersal is related to hormone levels (Belthoff and Dufty 1998), breeding cues are unknown and we assumed it was related to photoperiod rather than climate. Since both great horned owls and screech-owls are year-round residents in the MRG, there is little temporal separation in prey abundance (a critical resource) and nesting behaviors. The western screech-owl has longer incubation and developmental stages than some other birds, which could leave them more vulnerable to mismatches than shorter generation species (Cannings and Angell 2001). However, this time period is not unusual for other taxa (e.g., mammals) and we did not consider them vulnerable to this risk.

Biotic Interaction Scores

We found one area where biotic interactions were influenced by climate change. Great horned owls can use sites for nesting that were not produced by other birds (e.g., a snag with a broken-off top), but frequently use abandoned stick nests of diurnal birds of prey, mostly red-tailed hawks (Houston and others 1998). Given this owl's propensity for using sites produced by other species and the likelihood of other raptors being vulnerable to climate change, we considered this species vulnerable to declines in a symbiont. We did not consider this to be an issue for the western screech-owl. Perhaps due to its smaller size, the western screech-owl has less specific habitat requirements. For the remaining questions, either we found no evidence of effect (predation) or no clear trends could be estimated (e.g., food resources). Both great horned owls and screech-owls feed on a wide variety of prey including mammals (~90% of the great horned owl diet), birds (~10%), amphibians, reptiles, insects, and other invertebrates (Houston and others 1998). Owls are opportunistic hunters and their main prey varies greatly depending on the season and location. Drought has an effect on these animals via food resources, but this impact is not easily quantified given the diverse prey based utilized by each species.

Adult great horned owls are considered safe from most predators and nest predation is generally very low (Houston and others 1998). Larger owls are the primary predators on adult western screech-owls (Cannings and Angell 2001). Juvenile owls experience the most risk from predation as they are frequently in a weakened condition from starvation. Great horned owlets and eggs may be eaten by ravens, raccoons, opossums, etc., when adults are away from nest; fledglings are at the most risk and may fall prey to coyotes, foxes, bobcats, etc., before they can fly well (Ibid). However, it is not clear that this has a significant effect on owl populations.

Western screech-owls compete with other cavity nesters for cavities (Cannings and Angell 2001), but the extent of competition and effects on screech-owls are unknown.

Piciformes (woodpeckers)

We assessed four species of woodpeckers inhabiting the MRG: the ladder-backed woodpecker (*Picoides scalaris*), downy woodpecker (*Picoides pubescens*), hairy woodpecker (*Picoides villosus*), and northern flicker (*Colaptes auratus*).

Habitat Scores

Habitat scores reflect expected trends (decreases/increases) for associated vegetation and required habitat components, changes in habitat quality, requirements for additional habitat, and capacity of species to move with transitions in critical habitat features.

We found three woodpeckers—the downy woodpecker, hairy woodpecker, and northern flicker—at risk of breeding habitat declines due to expected loss of large trees and riparian woodlands. Each of these species has diverse habitat associations but is dependent to some degree on mature forest habitat (Jackson and others 2002; Jackson and Ouellet 2002; Wiebe and Moore 2008). The downy woodpecker is likely to be most at risk since it is found primarily in riparian areas in the Southwest. We found the ladder-backed woodpecker to be the only species likely to experience little or no change in associated vegetation primarily because it successfully utilizes more shrubby habitats (Lowther 2001; Winternitz 1998 in Lowther 2001). In the MRG, this species is reported to occupy primarily cottonwood woodland but also "willow vegetation along ditches" (Cartron 2008), habitats that are likely to decline under future conditions. However, this species also breeds in arid regions of the Southwest in areas dominated by desert shrubs, and we felt it is likely that this species will continue to utilize the MRG as vegetation transitions to more shrub-dominated habitat.

The primary habitat components important to these woodpecker species are large trees and snags that are able to support cavities (Jackson and Ouellet 2002; Lowther 2001; Wiebe and Moore 2008). Flicker population density appears to be closely tied to the availability of suitable nest-cavity substrates (Wiebe and Moore 2008). The long-term survival of cottonwoods in the Bosque may be threatened by reductions in water as well as increases in more drought-tolerant invasive tree species, which are not suitable for cavity excavation (Kranjcec and others 1998). Though the ladder-backed woodpecker is known to use a variety of alternate trees, including Joshua tree, willow, walnut (*Juglans* sp.), oak, hackberry, pine, mesquite, agave, and *Yucca* spp. (Lowther 2001), many of these alternate species are not present in the MRG. We assumed this species also relies on native cottonwood and willow vegetation and is thus vulnerable to loss of breeding components (i.e., nest sites) due to climate change

The downy woodpecker and hairy woodpecker were considered at risk due to declines in non-breeding roost sites. Northern flickers, which do not appear to depend on cavities for roosting during the non-breeding season, were not considered to be at a higher risk (Wiebe and Moore 2008). Downy woodpecker nests are not reused (Jackson and Ouellet 2002), so birds must excavate a new cavity each year and also excavate new roost cavities in the fall (Kilham 1962 in Jackson and Ouellet 2002), which may lead to increased vulnerability.

We found that the downy woodpecker appears to be sensitive to fragmentation (Doherty and Grubb 2002), which we considered a vulnerability due to the likely effect of future changes on the Bosque habitat that will lead to an effective decrease in habitat quality for this species.

None of these woodpecker species migrate; thus, habitat scores were similar for non-breeding associations.

Physiology Scores

We based physiology scores on expected responses to extreme climate, extreme weather events, potential limitations to daily activity, metabolic characteristics, adaptive capacity traits, and ability to outlive limiting conditions.

We did not find evidence that the woodpeckers were overly sensitive to high temperatures, and many express coping mechanisms for dealing with heat. Except for downy woodpeckers, these species all have ranges south of the MRG, which suggests that these species are probably not regularly exceeding temperature thresholds (Lowther 2001). The ladder-backed woodpecker has been observed to drink water from crevices on trees or other plants (Short 1982 in Lowther 2001).

None of the woodpecker species was considered prone to widespread mortality due to extreme events. One study reported a large number of dispersing downy woodpeckers killed by a storm (Segal 1960), and reports of weather related mortality include nests or roosts blown down in storms and nest trees hit by lightning (Jackson and Ouellet 2002); however, these are expected to affect only a few individuals in any given year. Additionally, all species are resident in the MRG and therefore are not exposed to potential changes in storms along a migratory route. There were some indications that some species appear sensitive to cold weather. Cold, wet weather is believed to be one of main causes of nestling mortality in downy woodpeckers. However, there are no known nest failures due to weather in RMRS data (n = 93). We also did not consider cold weather as a condition currently limiting woodpecker range and, therefore, did not count warming as a possible benefit (i.e., reduction in limiting condition, sensu Bagne and others 2012).

Daily activity patterns of the woodpeckers assessed appear to be limited more by cold than warm temperatures and we did not expect climate change to reduce foraging time for these birds. Based on a radio tracking study of 77 adults, cold or rainy weather was associated with northern flickers spending less time foraging (Wiebe and Elchuk 2003 in Wiebe and Moore 2008). No information was found in Wiebe and Moore (2008) indicating limits on daily foraging regime relative to high temperatures.

We did not consider woodpeckers to be able to take advantage of behaviors or metabolic adaptation to allow them to deal with resource variation. Although Burchsted (1987 in Jackson and Ouellet 2002) reported an observation of one downy woodpecker that appeared to be caching poison-ivy berries in a bark crevice, overall, it appears that this species does not have long-term food storage. Hairy woodpeckers are not known to use torpor and though some food caching is recorded, the quantity seems limited (Jackson and others 2002). We found no information to indicate that northern flickers (Lowther 2001) or ladder-backed woodpeckers use long-term food storage (Wiebe and Moore 2008).

We found differences among the woodpeckers that influenced our scores regarding species' ability to outlive extended (>5 year) limiting conditions, here drought.

The hairy woodpecker, which is reported to live up to 15 years (Jackson and others 2002), likely has a lifespan long enough to survive potentially limiting conditions and we did not consider this species sensitive to this element. We considered the other species vulnerable to population declines under expected conditions because their lifespans may not be sufficient to survive extended droughts. No information was available on mean longevity for downy woodpeckers (Jackson and Ouellet 2002), though record longevity is 11 years (Clapp and others 1983). Based on band recoveries through 1998, longevity record of the ladder-backed woodpecker is 4 years 6 months (see Lowther 2001), although as for many species, very few band recoveries exist. Annual survivorship of northern flicker adults >1 year old was 43%, which appears to be lower, on average, than many other woodpecker species (Wiebe 2006b in Wiebe and Moore 2008).

Phenology Scores

Woodpecker species were scored similarly for phenology. We did not consider any of the species at increased risk due to a reliance on weather driven cues. Woodpecker breeding season occurs at the same time each year (i.e., spring). In predictable breeders such as the woodpecker, photoperiod is the predominant proximate factor controlling leading to hormonal changes that gonadal changes required for reproduction (Dawson and others 2001). We did find evidence to score all species as vulnerable to mismatch with timed food resources because food abundance in spring is an important factor driving fecundity in bird species breeding in temperate regions (review by Martin 1987). The timing of insect emergence and peak abundance in spring may change due to changes in precipitation and temperature. In particular, hairy woodpecker breeding is often associated with insect emergence, which is sensitive to timing changes from warmer temperatures and food may be limited in winter (Askins 1981). We did not consider any of the woodpeckers at an increased risk related to distantly placed resources or cues since all species assessed here are year-round residents in the MRG. We considered all woodpeckers vulnerable to timing mismatches during breeding because each species typically raises a single brood (Dawson 1923 in Lowther 2001). Downy woodpecker, hairy woodpecker, and northern flicker pairs are thought to attempt re-nesting (Jackson and others 2002; Jackson and Ouellet 2002; Wiebe and Moore 2008) but only if nest failure happens early in the season, so we did not considered this behavior a true source of resilience.

Biotic Interaction Scores

With the exception of competitive interactions, we did not find evidence that supported vulnerable or resilient scores for future biotic interactions. Woodpeckers are generalists, though primarily insectivores. Downy woodpeckers primarily consume insects and other arthropods as well as fruits, seeds, some cambium tissue, and sap (Jackson and Ouellet 2002). The ladder-backed woodpecker feeds on insects and arthropods from cactus, willows, cottonwoods, and mesquite (Lowther 2001). Flickers feed mostly on insects, primarily on ants, but also on predaceous ground beetles and a variety of fruits and seed during cold months (Wiebe and Moore 2008). Predator interactions are also general and we did not consider predation important under climate change. We did not consider woodpeckers reliant on other species but they were vulnerable to potential losses in symbionts.

Though we found literature on competitive interactions for all woodpecker species, only the northern flicker was considered vulnerable to future changes. European starlings have been implicated in some declines for northern flicker populations (Jackson and others 2002; Wiebe and Moore 2008), and flickers are known to compete with European starlings and American kestrels for nest cavities. We expect both European starlings and American kestrels to be minimally impacted by climate change. Downy woodpeckers compete with other birds, including chickadees, nuthatches, titmice, house wrens, house sparrows, European starlings, and other woodpecker species, for cavities for nesting and roosting (references in Jackson and Ouellet 2002). We expect many of these competing species to be vulnerable to climate change, but we could not attribute a population decline of downy woodpeckers to any species and so considered this a neutral effect. Little information exists regarding interactions of the ladder-backed woodpecker with other species, but they were assumed to experience issue similar to that recorded for other species (Lowther 2001). European starlings have also been implicated in declines for hairy woodpecker populations (Jackson and others 2002; Wiebe and Moore 2008). Though generalist species such as starlings may be more resilient to changes in climate, they are not common where hairy woodpeckers nest. We did not consider brood parasitism an issue for any of the MRG woodpecker species as it is not common in cavity nesting species (Lowther 2001).

Apodiformes (swifts and hummingbirds)

A single Apodiformes species was assessed for the MRGB: the black-chinned hummingbird, *Archilochus alexandri*.

Habitat Scores

Black-chinned hummingbirds were considered vulnerable to habitat loss. Though they breed in a variety of habitats, including desert scrub, shrublands, suburban areas, and orchards, they are most frequently found in riparian habitats, and, in the MRG, highest densities are found in cottonwood gallery forest with a thick shrub understory (especially Russian olive) (D. Hawksworth, pers. comm.). Irrigation and ornamental plantings have allowed the species to expand into previously unused areas in the Southwest (Baltosser and Russell 2000). Black-chinned hummingbirds require shaded, secure nest sites for successful nesting, and nest success in southeastern Arizona was much lower when a severe freeze greatly reduced leaf cover (Baltosser 1989). Therefore, this species was considered vulnerable to climate-induced reductions in habitat quality associated with potential reductions in cover possible with increased drought periods and lower water availability.

Black-chinned hummingbirds are long-distance migrants that rely mostly on riparian areas during migration (Baltosser and Russell 2000). This assessment does not examine differences in risk associated with different habitat associations, but it may be more of an issue for species that use especially rare habitats such as the hummingbird.

Physiology Scores

The black-chinned hummingbird was one of few species considered vulnerable because of physiological limitations. The MRG is in the southern portion of range and though it is most abundant in southern portions, its small size and lower nest success with reduced shading was thought to indicate intolerance to increasing temperatures (Baltosser and Russell 2000). The black-chinned hummingbird's range has expanded northward in the Pacific Northwest over the last century (Baltosser and Russell 2000).

Black-chinned hummingbirds can go into torpor at night and during periods of cold weather to conserve energy (Lasiewski 1964). Birds with adequate energy reserves do not use torpor (Hainsworth and others 1977), indicating this is an energy saving strategy. This was one of three birds assessed (the others were the black-capped chickadee and the greater roadrunner) that appear able to use this method of metabolic savings.

Phenology Scores

Black-chinned hummingbirds, which time migration and breeding to plant flowering (Baltosser and Russell 2000), were considered vulnerable to changes in timed resources as well as distantly placed resources and cues. Timing of flowering plants will change in the future and the hummingbirds are not able to predict changes in flowering prior to migration. Black-chinned hummingbirds (Baltosser and Russell 2000) are known to raise multiple broods per season, one source of resilience for this species.

Biotic Interaction Scores

Flower nectar and small insects and spiders are the hummingbird's main food source. Sugar water from feeders is now also an important food source (Baltosser and Russell 2000). Feeder sources will likely remain relatively constant but reduced rainfall may reduce flowering plant availability. Black-chinned hummingbirds are susceptible to Candidiasis, a yeast infection that can destroy the tongue and bill tip (Baltosser and Russell 2000). It can be spread through feeders, which may become increasingly important, particularly during drought years. No other significant effects were observed for this species.

Cuculiformes (cuckoos)

The MRG contains two species of cuckoos: the western yellow-billed cuckoo, *Coccyzus americanus occultidus*, and the greater roadrunner, *Geococcyx californianus*.

Habitat Scores

The western yellow-billed cuckoo is expected to experience habitat declines due to its association with riparian habitats containing willow and cottonwood (Hughes 1999), which are expected to decrease with reduced stream flows, lower water tables, higher temperatures, and changing flood regimes. Cuckoo populations in New Mexico have declined precipitously since the 1960s as humans have altered riparian habitats (Howe 1986). In contrast, greater roadrunners occur in a variety of arid and semi-arid regions, including open grasslands, pinyon-juniper woodlands, scrub blackbrush, creosote scrub, tamarisk thickets, and riparian areas (Hughes 1996). Greater roadrunners will avoid heavily forested areas and open habitats that lack suitable nesting substrates (Hughes 1996). There was no predicted change in winter habitat area for either species. Greater roadrunners do not migrate (Hughes 1996) and are found year-round in the same habitats (see above). While winter habits are not well known, the western yellow-billed cuckoo seems to be associated with a number of scrub and woodland habitats in South America (Hughes 1999) that were not expected to result in a net decline.

Western yellow-billed cuckoos are expected to lose nesting substrate (habitat component) because they rely on trees large enough to have large horizontal branches to support their nests (Hughes 1999). Conversely, greater roadrunners require isolated thickets of small trees and/or bushes (for nesting sites) close to open or short-grass areas for foraging (Folse 1974 in Hughes 1996). Roadrunner nests are usually in a thorny bush, small tree, or cactus above ground, often in a well-concealed location (Hughes 1996). Increased aridity is expected to lead to more open areas and an increase in arid-adapted vegetation (e.g., cactus) resulting in an increased nesting habitat for roadrunners.

As with other birds, both cuckoo species were assumed to have adequate dispersal capacities to keep up with habitat changes if needed. Though roadrunners show site tenacity to breeding location (Hughes 1996) and are non-migratory, the range of this species has expanded northward and eastward in the twentieth century (Hughes 1996) and it is considered capable of movement in response to changing habitat.

Greater roadrunners do not migrate (Hughes 1996) and, therefore, do not have additional habitat requirements. The western yellow-billed cuckoo is a long-distance migrant through Mexico and Central America and was considered vulnerable to the additive risk of relying on multiple changing habitats. In addition, for 2-3 weeks before breeding, it may occupy upland vegetation, including pinyon, oak, juniper, and manzanita (Hughes 1999).

Physiology Scores

Greater roadrunners are not expected to experience physiological limitations. They occur in arid and semi-arid regions and show physiological adaptations to heat and aridity (e.g., mechanisms to keep brain cool and reabsorb water). Greater roadrunners also undergo behavioral responses such as gular fluttering, panting, and extending wings to expose sides of body (Hughes 1996). Conversely, the western yellow-billed cuckoo is considered at risk of reaching its physiological limit under warmer conditions. The yellow-billed cuckoo is restricted to riparian areas in hot regions, which are generally cooler and more humid than surroundings. Further, incubating adults and nestlings have been observed panting on hot days (Hughes 1999), and eggs may be prone to drying, which reduces hatchability (Laymon and Halterman 1987).

Neither cuckoo was considered at risk due to extreme events. Though there may be indications for increased mortality in roadrunners due to storms, they were not considered prone to large-scale effects from extreme weather events. Norris and Elder (1982 in Hughes 2011) reported that adult roadrunners are at greater risk of mortality if winter snow cover persists; however, climate projections predict less snow and earlier snowmelt. Western yellow-billed cuckoos may experience increased mortality with increased exposure to hurricanes, but exposure is probably not high for western populations (Hughes 1999). More intense rain storms are also predicted for South America (Magrin and others 2007), which may increase mortality for some species, but the effect on cuckoos is largely unknown.

Roadrunners, but not cuckoos, were considered at risk of limitations to active periods under warmer climates. Most roadrunners are active in morning and late afternoon (1530–2000 h) and ~ 50% less active during midday hours when adults frequently rest in the shade (Calder 1967b in Hughes 1996). Daily active periods may be directly restricted to the point of significantly reducing survival.

Both species had adaptations that may help them deal with resource variations though only the roadrunner was believed to be able to express it enough to confer some level of resilience**.** Non-incubating roadrunner adults may conserve energy at night by lowering core body temperature from about 40 °C to 34 °C (Vehrencamp 1982). Adults can also warm themselves in cold weather by sunbathing using an exposable black skin patch on their back (Hughes 1996). The western yellow-billed cuckoo has both inter- and intra-specific brood parasitism. This may allow the species to lay larger clutches and lay extra eggs in nests of other individuals. This behavior has the potential to increase breeding success during years of high resource levels. Increased breeding opportunities through brood parasitism may be expected to increase populations during high resource years, but no effect of this behavior on populations has yet been documented. Rates of brood parasitism are also not well known, but there are few records of occurrence, which may indicate this behavior is too rare to affect populations. Some pairs nesting in California have one or more helpers that assist in raising nestlings (Nolan and Thompson 1975), which may also be a strategy for increasing reproduction, but, again, effect on populations and frequency of this behavior is unknown.

Neither species is not thought to be able to outlive periods that may result in extended reproductive failure. The greater roadrunner was considered vulnerable to reduced survival and populations due to extended droughts. Drought is considered a limiting condition in terms of food availability for the western yellow-billed cuckoo as well (Hughes 1999).

Phenology Scores

Neither cuckoo was thought to rely on climate mediated cues. Both species were considered sensitive to timing of food resources, which may change under climate warming. Changes in precipitation and temperature may alter timing of insect emergence, which is an important food source for both species. Breeding initiation in the western yellow-billed cuckoo may be correlated with abundance of local food or periods of greatest precipitation.

The greater roadrunner is a year-round resident in the MRG. Typically, increases in food abundance (a critical resource) occur in spring so there is little temporal separation. The greater roadrunner is reported to have a bimodal reproductive cycle (Hughes 1996; Ohmart 1973), whereas the yellow-billed cuckoo is believed to raise only one brood per season during a relatively short breeding season (Hughes 1999). Recent evidence indicates that yellow-billed cuckoos may breed a second time in western Mexico after migrating from the north (Rohwer and others 2009), which would allow this species to take advantage of seasonal resources in multiple locations, but it is unknown if MRG cuckoos exhibit this behavior.

Biotic Interaction Scores

Roadrunners are opportunistic omnivores with about 90% of their diet comprising animal material (Bryant 1916 in Hughes 1996). No predictions were made regarding future food supplies for roadrunners, but yellow-billed cuckoos were considered potentially likely to experience increased food resources. Yellow-billed cuckoos also have a varied diet and feed primarily on large insects such as caterpillars, grasshoppers, crickets, katydids, and cicadas but will also eat small lizards, frogs, eggs, fruits, seeds, and nestlings. Cuckoo populations fluctuate greatly with food availability and increase dramatically in years of highest insect abundance such as tent caterpillar infestations and cicada cycles (Heath and Wilkin 1970). Though tent caterpillar infestations are likely to decline with their host trees, cicadas are resilient to high temperatures (Heath and Wilkin 1970). Populations of at least one species (Apache cicadas) increased due to habitat changes associated with declining water tables due to increased tamarisk trees (Glinski and Ohmart 1984).

Predation effects were not considered significant for either species. Adult roadrunners are infrequently caught by raptors such as red-tailed hawks and Cooper's hawks; eggs and nestlings are taken by a variety of predators, including coyotes, raccoons, skunks, snakes, crows, and ravens (Hughes 1996). Vulnerability to disease is also not well known. These birds are seldom subject to brood parasitism by brown-headed cowbirds (*Molothrus ater*) as nesting duration is short in the yellow-billed cuckoo (Hughes 1999) and roadrunner nesting behavior is not conducive to parasitism. There is some evidence of competition with black-billed cuckoos in the eastern United States (Hughes 1999), but black-billed cuckoos are not found in the MRG. No significant competitive effects are known for roadrunners.

Falconiformes (hawks and falcons)

Two species in the MRG are reviewed here: the American kestrel, *Falco sparverius*, and Cooper's hawks, *Accipiter cooperii*.

Habitat Scores

Loss of canopy is likely to benefit the American kestrel, which breeds in a wide variety of open to semi-open habitats, including meadows, grasslands, deserts, wooded streams, burnt forest, early old field successional communities, open parkland, agricultural fields, and both urban and suburban areas (Bird and Palmer 1988). In contrast, Cooper's hawks, which have a large range but breed exclusively in forests (deciduous, mixed, evergreen, and deciduous riparian), are expected to experience a loss of habitat. Cooper's hawk habitat increasingly includes suburban and urban areas for breeding (references in Curtis and others 2006). Conversion to urban and suburban areas is not directly linked to climate change but it is expected to reduce suitable riparian breeding habitat. Winter habitats of American kestrels are generally similar in vegetative structure to those occupied during the breeding season, except often with more woody vegetation (i.e., open patches are smaller), and suitable nesting trees may or may not be present (Smallwood 1987, 1988). Again, loss of forest habitat is expected to benefit this species. Similarly, Cooper's hawks also presumably use the same habitats in winter and thus are expected to be exposed to the same losses to riparian habitat.

For the American kestrel, cavities were considered less important for non-breeding roosting sites since dense vegetation may also provide suitable sites. However, limiting factors for the kestrel include suitable prey, perches, and low, open vegetation for hunting (Balgooyen 1976 in Smallwood and Bird 2002). Conversely, though not a cavity nester, Cooper's hawks require large trees for nest substrates and were considered at additional risk due to loss of nonbreeding habitat elements as a result of climate change.

Habitat quality was difficult to quantify for the majority of animal species. For many, such as the American kestrel, there was evidence for differential nesting success rates in cross-site comparisons, but these differences could not be attributed to a specific cause and could not be qualified here.

Kestrels and Cooper's hawks (Hoffman and others 2002 in Curtis and others 2006) are present year-round in the MRG and populations include both resident birds and migrant/wintering birds from farther north.

Physiology Scores

There were no indications that either species would reach a physiological threshold under future conditions. American kestrels occupy a wide range of semi-open habitats (see Habitat Question 1) suggesting they experience warm temperatures during breeding. New Mexico is in the center of this species' breeding range. Cooper's hawks are also widespread.

Neither species was considered at risk of increased mortality (or widespread mortality) due to extreme climate conditions. There is no information of an effect on American kestrel. Nest failures in Cooper's hawks have been attributed to windstorms (Curtis and others 2006) but it is not clear that windstorms will increase in MRG in the future.

With the exception of cases of extreme heat rises, it seems unlikely that seasonal or daily activity periods will be restricted to the point of significantly reducing survival for either species. In a study of American kestrels during winter in Florida, where prey densities were high, kestrels seemed to satisfy their daily energy demands relatively early in day; the daily temporal pattern of foraging was also related to the activity patterns of prey (Smallwood and Bird 2002).

American kestrels cannot use torpor and though this species will cache uneaten remains of prey and surplus kills (Collopy 1977; Toland 1984), storage is for only a few hours to a few days (Toland 1984). During the breeding season, Cooper's also hawks cache prey (Curtis and others 2006), but this was not considered a long term food source (Bagne and others 2011).

Precipitation is predicted to decline in the Southwest over time, and periods of extreme reductions in rainfall that reduce food resources to the point that birds have insufficient food for breeding may last >5 years. American kestrels have a mean life span of 5 years 2 months (Bird and Palmer 1988; Roest 1957); many individuals are expected to live less than the mean.

Phenology Scores

Kestrels and Cooper's hawks were not considered to rely on temperature or precipitation cues. Both species were considered vulnerable to changes in the timing of food resources. Based on information for other raptors, changes in the timing of precipitation and increases in temperature have the potential to alter the timing of prey availability and abundance and, hence, fecundity. Food supply influences reproduction and survival in the American kestrel (Smallwood and Bird 2002). Productivity varies with habitat, presumably in response to prey availability (Smallwood and Bird 2002). Low productivity has been associated with rain during nestling period (Smallwood and Natale 1998 in Smallwood and Bird 2002). For the American kestrels and Cooper's hawks, which overwinter in the MRG, there is little risk due to separation of resources and key life history events. In addition, the American kestrel can re-nest after nest failure, with some pairs reported to rear a second brood after successful first nest (Smallwood and Bird 2002).

However, Cooper's hawks raise only one brood, and despite being able to re-nest if clutch is lost early in season, the Cooper's hawk was considered vulnerable to phenological mismatch.

Biotic Interaction Scores

Neither species was considered at risk nor likely to benefit from climate-related changes in prey or predators. Kestrels feed primarily on insects and small rodents but also frogs, reptiles, and small birds with diet varying with prey availability and location (Smallwood and Bird 2002). Cooper's hawks feed primarily on a wide variety of small-to-medium-sized birds and small mammals and occasionally reptiles and amphibians (Curtis and others 2006). No natural predators appear to pose a significant impact to these species' populations.

An interesting source of vulnerability relates to a symbiotic relationship for the cavity nesting American kestrel. Since this species is an obligate secondary cavity nester, meaning that they do not excavate cavities in trees themselves, they rely on natural cavities or those excavated by woodpeckers (Smallwood and Bird 2002). Given the size of this species (large enough that natural cavities may be harder to find than for smaller secondary cavity nesting species) and the likelihood of woodpeckers being vulnerable to climate change, this species was scored as vulnerable due to the loss of important symbiotic species. Interestingly, many populations of cavity nesters are expected to decline, which should reduce competition for species like the American kestrel. However, it isn't clear that this will occur in advance of reduced cavities or not.

Disease may be an issue for Cooper's hawks but not American kestrels. Though kestrel adults and young are subject to a variety diseases and parasites (see review in Smallwood and Bird 2002), the authors did not report any diseases resulting in mass mortality. Conversely, Cooper's hawks may be at an increased risk of trichomoniasis, the primary cause of death among the half of all nestling mortality in Arizona (Boal and Mannan 1999). Nestlings were likely infected after feeding upon infected doves. Given the large population of doves in the MRG, which are considered resilient to climate change, future exposure is likely to increase for Cooper's hawks leading to increased disease incidence. Numerous other diseases and parasites (blood parasites and trematodes) have been recorded (summarized in Curtis and others 2006), but there are no reports of widespread deaths.

Passeriformes

Most of the birds assessed for this report fell within the order Passeriformes. Table 3.2 lists both the common and scientific names of the species reviewed in this section. A few species had traits that may be important under climate change but that were not included in our assessment tool. In particular, chickadees may undergo irruptive movements over long distances when food is scarce or due to habitat alterations (Smith 1993). Phainopeplas are considered nomadic and populations fluctuate regionally with changing resources (Walsberg 1977). These behaviors may mitigate negative impacts of climate change on food resources. Though potentially important, this was not considered within this assessment.

Habitat Scores

Vegetation Association: We did not consider species at risk or likely to benefit from climate-induced habitat changes if they prefer or tolerate well open woodlands or scrub habitat. These species include ash-throated flycatcher (Cardiff and Dittman 2002), eastern bluebirds (Gowaty and others 1998), lesser goldfinch (Watt and Willoughby 1999), and bushtit (Sloane 2001). Eastern bluebirds are a recent arrival to the MRG and are found almost exclusively in riparian woodland that has been cleared of understory by fuels reduction efforts or post-fire rehabilitation (Catron and others 2007). We gave neutral scores for Bewick's wrens, Lucy's warblers, and yellow-breasted chats. Bewick's wrens, though resident in the riparian forest, are associated with human alterations of the habitat and are able to use a variety of vegetation classes. Interestingly, though we did not consider them at risk in this assessment, Bewick's wren populations in the MRG declined with the removal of the exotic shrub understory and downed dead wood (RMRS Fuels Study data), which may indicate some sensitivity to future changes. Lucy's warblers and yellow-breasted chats are known to do well in habitats dominated by exotic plant species (Eckerle and Thomson 2001) and though Lucy's warblers prefer to use cottonwood-willow riparian habitats (Johnson and others 1997; Rosenberg and others 1991), we felt these species were unlikely to be overtly affected by vegetation changes.

We expected most species would experience declines in breeding habitat and associated vegetation. We scored species as vulnerable when they demonstrated positive association with cottonwood or willow habitats or exhibit negative associations with exotic species expected (e.g., Tamarisk). These included black phoebe, which breeds in a wide variety of riparian habitats (Wolf 1997), but seems to avoid salt cedar (Conine and others 1979); phainopeplas, which specialize on cottonwoods heavily infested with mistletoe (RMRS data); and the southwestern willow flycatcher (Sedgwick 2000). Generally, the southwestern willow flycatcher does not occupy areas dominated by exotics (Sogge and Marshall 2000) but can successfully nest in salt cedar-dominated habitats (Sogge and others 2006). We also considered species such as black-capped chickadee (Smith 1993), white-breasted nuthatch (Grubb and Pravosudov 2008), and summer tanager (Carton and others 2008), which rely on mature forests or closed canopy habitats as vulnerable to climate related loss in associated vegetation. The summer tanager may be able to utilize salt cedar habitats in New Mexico (Hunter and others 1988; Robinson 1996) but avoids fragmented forests in the East (Rosenberg and others 1999). We also considered species that exclusively associate with riparian areas or wetlands, such as the common yellowthroat (Szaro and Jakle 1985 in Guzy and Ritchison 1999) and the red-winged blackbird as vulnerable to breeding habitat loss. We scored two species that prefer edge habitat—Bullock's orioles (Rising and Williams 1999) and western wood-pewees (Bemis and Rising 1999)—as vulnerable to associated vegetation loss because edge habitat is likely to decline in the long term even though it is likely to increase in the short term.

We predicted increases in scrub habitat to negatively affect habitat suitability for two swallow species and western meadowlarks. Bank swallows and cliff swallows breed in a wide variety of open or semi-open habitats near water wherever suitable nesting sites exist and are not tied to a specific vegetation type (Garrison 1999). However, increases in desert scrub habitat at the expense of grasslands and loss of open water were considered to represent a loss of suitable habitat for these species (Brown and Brown 1995). Western meadowlarks are most common in native or restored grasslands and in the southwestern United States and are frequently found in farmlands and grasslands in river valleys (Davis and Lanyon 2008). Future transitions away from grasslands in the MRGB will negatively impact this species.

We expect that a number of species that prefer open, desert, or shrubby habitat, including blue grosbeaks, barn swallows (Ingold 1993), brown-headed cowbirds (Lowther 1993), European starlings (Cabe 1993), spotted towees (Greenlaw 1996), western kingbirds (Gamble and Bergin 1996), indigo buntings (Payne 2006), and house finches (Hill 1993), will experience increases in breeding habitat. Unlike the black-headed grosbeak, blue grosbeaks will probably experience habitat increases as the riparian area shifts to more shrubby species (Ingold 1993). Spotted towhees breed in a variety of habitats that contain a dense shrubby layer with or without an overstory of taller trees (Greenlaw 1996). In RMRS data, towhee numbers declined after the removal of the exotic understory, and spotted towhees have declined where shrubby areas become more open (e.g., due to grazing or other factors; Greenlaw 1996). The indigo bunting inhabits shrubby habitats, open woodlands, and fields (Payne 2006). Blue grosbeaks may also benefit from abandonment of agricultural fields and spread of salt cedar (Ingold 1993). Brown-headed cowbirds (Johnson and van Riper 2004), European starlings (Cabe 1993), western kingbirds (Gamble and Bergin 1996), and house finches (Hill 1993) are known to respond positively or associate with human disturbed habitats, which contributed to their resiliency scores.

Wintering Habitats: We expected many migrants to experience declines in wintering habitats located in South and Central America. We considered species that rely on marshes and similar habitats located in central or South America to be vulnerable to habitat loss (e.g., barn swallows, Brown and Brown 1999) or habitat shifts (cliff swallow, Brown and Brown 1995). Similarly, we considered species that rely on riparian habitats and wetlands in southern regions during winter, such as cliff swallow (Brown and Brown 1995), Lucy's warbler (information lacking but see Johnson and others 1997), black-headed grosbeak (Hill 1995), and southwestern willow flycatcher (Lynn and others), as vulnerable to winter habitat loss. In addition, we considered tropical deforestation a threat to wintering habitats of western wood-pewees (Bemis and Rising 1999) and Bullock's orioles (Rising and Williams 1999). We could not identify specific trends for winter habitat for bank swallows and considered winter habitat as unlikely to decline for wide ranging species, such as common yellowthroat, summer tanager, western kingbird, and yellow-breasted chat, which occupy a wide variety of habitats in winter (Eckerle and Thomson 2001; Gamble and Bergin 1996; Guzy and Ritchison 1999; Robinson 1996).

We expected potential expansions in wintering habitat for the ash-throated flycatcher, which associate with open and semi-open areas (Cardiff and Dittman 2002), and the blue grosbeak and the indigo bunting, both of which winter in grasslands and thickets in Mexico and Central America.

For most resident species, such as the black phoebe, white-breasted nuthatch, red-winged blackbird, and house finch, we gave similar scores for breeding and non-breeding habitats. We considered white-breasted nuthatches, black phoebes, black-headed grosbeaks, and red-winged blackbirds as vulnerable to winter habitat declines, whereas we expected house finches to experience an increase in habitat over time. In one exception, we expected spotted towhees, which use additional brushy habitats, including areas such as marshes and mesquite thickets (Greenlaw 1996), as unlikely to experience any significant change in winter habitat.

Habitat Component Change: Species that rely on tree cavities were considered vulnerable to climate-related changes in cavity availability. These species included ash-throated flycatchers (Brush and others 1990; Cardiff and Dittman 2002), black-capped chickadees (Smith 1993), Bewick's wrens (Kennedy and White 1997), white-breasted nuthatches (Grubb and Pravosudov 2008), and Lucy's warblers (Johnson and others 1999; also associated with cottonwood trees, Stoleson and others 2000). While white-breasted nuthatches do occupy cavities in the non-breeding season, it is not clear whether they require cavities for roosting (no information in Grubb and Pravosudov 2008 or Kilham 1971 indicating that they are required) and we did not score these species as vulnerable to future component loss.

We also scored species that rely on dense foliage or tree canopies as vulnerable to habitat component loss. Among these species are red-winged blackbirds (require dense cover during migration and winter, Yasukawa and Searcy 1995), summer tanagers (nest in trees, particularly cottonwoods, Carton and others 2008), western wood-pewees (nest in trees, Bemis and Rising 1999, particularly cottonwoods, Bemis 1996), southwestern willow flycatchers (nest in shrubs and small trees near flowing water, Johnson and others 1999), and common yellowthroats (nest in reeds, grasses, or sedges, Stewart 1953 in Guzy and Ritchison 1999).

We gave most passerines a neutral score for habitat component change. Banks swallows, barn swallows, and cliff swallows use natural features such as arroyos, cliffs, and banks as well as human structures, which are not expected to be overly influenced by climate change (Brown and Brown 1999; Turner and Rose 1989). Similarly, though considered a major limiting factor, we did not expect suitable nest sites (rock faces, dirt banks, and human structures) for black phoebe populations to change. Other bird species such as black-headed grosbeak (Hill 1995), Bullock's oriole (Rising and Williams 1999), indigo bunting (Payne 2006), house finch (Hill 1993), lesser goldfinch (Watt and Willoughby 1999), phainopepla (Chu and Walsberg 1999), and western kingbird (Gamble and Bergin 1996) have generalized substrate needs. Bluebirds require cavities for nest sites, but these can be naturally occurring or excavated by woodpeckers (Gowaty and others 1998). Naturally occurring nest sites in structures or rocks are not expected to be altered and flexibility in nest site selection may ameliorate the negative impact of loss of tree cavities on nest site availability. Similarly, European starlings are cavity nesters but are able to use a wide variety of substrates from woodpecker holes to buildings to cliffs (Cabe 1993). Brown-headed cowbirds require nests of other birds. Though we recognized that declines in the numbers of host nests (but not clutch size) effectively reduce the availability of this important habitat component, we felt that hosts vulnerable to climate changes will be replaced by more resilient species, thus resulting in no net effect on availability for the cowbird.

We considered future changes to increase the availability of habitat components for yellow-breasted chats, which use thick shrubby habitats for breeding (Eckerle and Thomson 2001) that may increase with mature cottonwood mortality and increased fires.

Habitat Quality: Habitat quality was often hard to discern and many species received a neutral score. Exceptions included Lucy's warblers, which we scored as resilient because they reach their highest densities in closed canopy riparian mesquite habitats but also show dense populations in salt cedar thickets and cottonwood Bosques (Johnson and others 1997). We found habitat changes resulting in increased vulnerability due to declines in quality for species that prefer particular vegetation structure. Black-capped chickadees appear to do better in undisturbed mature forest (van Oort and Otter 2005), and we expect them to be negatively impacted by the effects of increased drought and fire. We considered water loss an issue for red-winged blackbirds because studies show nests built over deep water are more successful than nests built over shallow water or dry land (Searcy and

Yasukawa 1995). Decreases in water availability are also likely to reduce habitat quality for southwestern willow flycatchers (Bombay and others 2003; Sedgwick 2000). Western meadowlarks are sensitive to density and height of grasses, with a preference for an intermediate density (Davis and Lanyon 2008), and we considered future conditions, including reduced grass density and increased invasive species, as problematic for this species. Eastern bluebirds require habitats with little or no understory for foraging areas (Gowaty and Plissner 1998). Fragmentation may be a problem for white-breasted nuthatches (Doherty and Grubb 2002), yellow-breasted chats (Burhans and Thompson 1999), and indigo buntings (Weldon and Haddad 2005).

Stopover Sites: Most species assessed were long distance migrants (see Table 2.3 and Appendix 1) and as such, were prone to issues related to their use of multiple habitats that may be exposed differentially to climate change. This assessment does not evaluate differences in risk associated with different habitat associations, but we considered it more of an issue for species that use especially rare or highly endangered (e.g., riparian) habitats. Wintering grounds of Lucy's warblers are along the Pacific Coast of Mexico, and we felt the short duration between its presence on wintering and breeding grounds to indicate no stopover (Johnson and others 1997).

Some species have both resident and migratory populations. For instance, southernmost populations of phainopeplas are sedentary, but northernmost birds may migrate over 1000 km to breeding grounds. A few phainopeplas winter in the lower MRG, but this species is mostly nomadic and may move further south or perhaps into Arizona. Red-winged blackbirds are resident in much of their range, including New Mexico, but northern and high-altitude populations are short-distance migrants (Yasukawa and Searcy 1995). White-breasted nuthatches are generally a year-round resident (Grubb and Pravosudov 2008) but some populations in northern and western regions undergo irruptive movements in some years and may leave their home range to exploit rich food sources (e.g., a nearby bird feeder). However, we did not consider these species at an increased risk of climate effects on habitat change because it is unclear how far individuals from MRG migrate and we did not have information on whether they require transitional habitats.

Physiology Scores

Physiological Limitations: We considered two species as likely to benefit from future warming trends: the black phoebe and the lesser goldfinch. Pheobes are thought to be cold limited in New Mexico (Wolf 1997) and could potentially benefit from future warmer winters. American goldfinches from warmer climate apparently lived longer than those from colder climates suggesting winter temperatures or storms are a factor in survival (Middleton 1984). Thus, we assumed warmer winter temperatures would increase the likelihood of survival of lesser goldfinches in a similar manner.

We did not consider most birds at risk of experiencing an increase in limiting conditions such as extended periods of very high daily maximum temperatures. Many, such as ash-throated flycatchers (Cardiff and Dittman 2002) and western wood-pewees (Bemis and Rising 1999), exhibit behavioral coping strategies for dealing with extreme heat such as becoming less active at midday, perching in shade, and/or compressing body plumage and holding folded wings away from body. Though some losses of Bewick's wrens nests were attributed to extreme heat (Kennedy and White 1997), we considered this more a reflection of poor artificial nesting substrate than actual heat intolerance.

We did not consider species such as Lucy's warblers and phainopeplas, which are at the northern edge of their breeding range along the MRG, to be at risk of reaching a physiological limit (Johnson and others 1997). We also did not consider species existing at the southwestern boundary of their ranges, including bank swallows, barn swallows, red-winged blackbirds, western kingbirds, and western meadowlarks (Brown and Brown 1999; Davis and Lanyon 2008; Howell 1975; Yasukawa and Searcy 1995; Root 1988), as heat intolerant since there was little evidence of negative heat effects for these species. House finches have expanded their range in North America in the last century; it currently extends well into Mexico (Hill 1993). Though the house finch does not occur in deserts far from water, the persistence of riparian habitats will allow the persistence of this species, and it was not considered threatened by additional extreme temperatures.

We considered four species vulnerable to potentially limiting physiological threshold: summer tanager, yellow-breasted chat, southwestern willow flycatcher, and indigo bunting. The MRG is north and west of the center of the summer tanager's range. It has been proposed that summer tanagers have not invaded salt cedar habitats in the lower Colorado River because summers are too hot (Hunter and others 1988), which we took to suggest a physiological threshold for high summer temperatures in this species. Similarly, the MRG represents the southern portion of the yellow-breasted chat's breeding range, which extends from southern Canada to northern Mexico. Chats are restricted to riparian areas in the West, whereas in the East, they occupy various shrubby habitats (Eckerle and Thomson 2001), which may indicate some intolerance for dry or hot conditions. Declines of willow flycatchers are associated with the spread of tamarisk, which may not have the thermal protection of broadleaf shrubs (Cartron and others 2008; Hunter 1988), although this species nests successfully in tamarisk in many areas (Sogge and others 2006). The indigo bunting only became established in the Southwest in the 1940s and generally occurs only in a few locations (Payne 2006); we considered it generally dependent upon more mesic environments and not likely to tolerate increased temperatures and drier conditions in the Southwest.

Extreme Events: Extreme drought has the potential to cause widespread bird declines. However, unless widespread mortalities were noted for species that were directly attributed to drought, we did not consider species vulnerable beyond effects already accounted for in habitat and biotic interaction sections. Drought could negatively impact brown-headed cowbird populations through its effect on host species. However, this relationship was difficult to quantify and is largely unknown. Severe drought can result in large-scale reproductive failure in red-winged blackbirds (Brenner 1966). In addition, this species may be at risk due to flooding as they nest over water.

We found sensitivity to storms during migration a primary indicator of potential vulnerability of passerines to future extreme events in the MRGB. This pertains both to species on their migration grounds and species within the MRGB that have observed instances of storm or heat wave related mortality. Migrating species which were considered at risk of increased exposure to hurricanes or other serious storms included: Bullock's oriole (Magrin and others 2007), ash-throated flycatcher (Cardiff and Dittman 2002), western kingbird (Gamble and Bergin 1996), and yellow-breasted chat (Magrin and others 2007). In 1951, hundreds of common yellowthroats (and other birds) died due to exposure to storms in the Gulf of Mexico and adjacent areas (Padre Island, Texas in 1951; James 1956 in Guzy and Ritchison 1999; Wiedenfeld and Wiedenfeld 1995 in Guzy and Ritchison 1999).

We did not consider migrants such as black-headed grosbeak (Hill 1995), blue grosbeaks (Ingold 1993), southwestern willow flycatchers (Sedgewick 2000), and indigo buntings (Payne 2006), which use interior migration routes, at an increased risk of hurricane related mortality. Similarly, white-breasted nuthatches are typically a year-round resident in the MRG and therefore are not exposed to potential changes in storms along a migratory route.

Cold spells may also have a detrimental effect on bird populations, especially if they occur during nesting periods. We considered increased cold a threat to future populations of Bullock's orioles, summer tanagers, western kingbirds, western wood-pewees, bank and barn swallows, European starlings, and Lucy's warblers. Late spring/early summer cold spells are known to rapidly reduce Bullock's oriole populations. Summer tanagers have been recorded in mass mortality events related to cold fronts during migration (Johnston and Haines 1957), which led the authors to consider them to be sensitive to extreme weather events. High winds and rain are a major cause of mortality in western kingbird eggs and nestlings (Gamble and Bergin 1996). Evidence exists that western wood-pewees experience nest failures due to weather in the MRG (RMRS data, six of 17 [35.0%] failed due to weather [most or all were nests blown down in storms]). Barn and bank swallows are especially vulnerable to periods of bad weather since it limits their foraging time and ability, thereby leading to starvation or weakness to the point where they are easily caught by predators (Brown and Brown 1999b; Garrison 1999). Swallow nests are also frequently destroyed in storms. Severe heat waves and droughts have also caused heavy nestling mortality in both species (Baltosser and Russell 2000; Brown and Brown 1999). Very cold, wet, or hot weather may kill European starling nestlings (Cabe 1993). We found no mortality information for Lucy's warblers, but we considered them at risk of extreme weather-related mortalities because their small size may make them more prone to mortality. Unusual cold spells that freeze mistletoe berries have dramatically reduced populations and resulted in reproductive failure in phainopepla (Anderson and Ohmart 1978; Rosenberg and others 1991); but we did not have enough information on potential changes in occurrence of freeze events for New Mexico and scored this species as neutral.

Most other birds were sensitive to extreme weather conditions that were not clearly defined for the MRG and we gave these species neutral scores. For instance, we did not consider resident species at either an increased or decreased risk of mortality in the future when they experienced mortality due to cold winters (versus cold spells during nesting). Severe winters in the 1970s were correlated with sharp declines in two species of wrens (winter and Carolina wrens), but severe winters have not been correlated with a long-term decline observed in Bewick's wren (Wilcove 1990 in Kennedy and White 1997). Black phoebes experienced increases in nest failures in California during extended periods of rainy weather (Irwin 1985) and heavy rains are known to reduce phoebe foraging time. Similarly, sudden declines in bushtit populations may be related rainy winters (Sloane 2001). However, we felt it unlikely that the Southwest will experience extended periods of rainy weather, and while predictions regarding precipitation are variable, most predictions agree that precipitation events are likely to be more intense and more irregular and to expect changes to timing. Therefore, we could not predict that the risk of nestling mortality will decrease or that foraging time will increase. Eastern bluebirds are known to communally roost in cavities to stay warm (Gowaty and others 1998), but there is little indication that cold temperatures cause deaths and even less indication that cold temperatures are limiting the MRG region. Deaths of western meadowlarks have been reported from ice storms and deep snow, which are uncommon in New Mexico.

Activity Periods: We did not expect daily activity patterns are to be overly influenced for most species. Exceptions were: phainopepla, which exhibits shade seeking behavior in hotter climates (Walsberg 1993); summer tanager, which is usually inactive in the hottest parts of the day (Robinson 1996) and has demonstrated behaviors indicative of heat intolerance; and yellow-breasted chat, which is usually inactive during the hottest part of the day (Eckerle and Thomson 2001). We considered these species vulnerable to reductions in activity periods due to climate changes.

Metabolic Advantage and Adaptations: We were only able to identify black-capped chickadees as able to reduce metabolic rates (Chaplin 1976). Though we are not sure what advantage this holds for the chickadee under warming regimes because it is primarily a mechanism to save energy during cold periods, the capability is generally considered an advantage and this species was given a resiliency score. We found a few instances of food storage behavior in birds, including the black-capped chickadee and white-breasted nuthatch, both of which cache food (mostly seeds, Smith 1993; Grubb and Pravosudov 2008). However, we did not consider these species to possess an advantage due to uncertainty with respect to the future availability of plant species that might be used in caching and a potentially short (28 days for the chickadee) memory for these caches (Hitchcock and Sherry 1990).

Variable Life History: Red-winged blackbirds were one of the few species we considered to have a variable life history component that may infer advantages under climate change. Males are polygynous with each male having up to 15 females in their territory. Higher polygynous rates are found in territories with the best habitat (Yasukawa and Searcy 1995) and also where valuable resources are concentrated (Ewald and Rohwer 1982). This type of mating system is related to resources and, although not demonstrated, flexibility is potentially advantageous under future conditions. We did not consider other polygynous birds, such western meadowlark whose males are usually polygynous with two females, to have an advantage under climate change. In eastern meadowlarks, polygyny may increase reproductive success through better parental care (Knapton 1988); however, we did not consider this a relevant trait with respect to climate variables as species specific data were not available.

Capacity to Outlive Limiting Conditions: Precipitation is predicted to decline in the Southwest over time, and periods of extreme reductions in rainfall that reduce food resources to the point that birds have insufficient food for breeding may last >5 years. We considered most birds vulnerable to this limiting condition since most adult birds will not outlive such periods. Though most passerines live longer than 5 years, they generally exhibit low survival rates. We did find several exceptions, however. Though weather (especially heavy rain) has a significant effect on populations at the northern edge of the black phoebe's range in California and Oregon (Irwin 1985), limiting factors in the rest of the phoebe's range are not well known. In addition, annual survival rate of black phoebes in California was estimated at over 70% (Schroeder 1985), indicating this species is not likely to experience limiting conditions over its entire lifespan. Though showing a much lower survival rate, we also did not consider eastern bluebirds to be affected by limiting periods. Numerous 6- and 7-year-old eastern bluebirds have been documented in long-term studies, but estimated annual survival rates for adults range from 38 to 47% (Pinkowski 1971a; Plissner and Gowaty 1996). European starlings can live up to 21 years, though annual mortality is estimated at 33-77% (Cabe 1993). However, this species is not known to have climate-related limiting conditions. Brown-headed cowbirds can live up to 10 years, but annual adult survival is approximately 45% (Lowther 1993). Though this was often considered an indication of vulnerability for most other bird species, evidence exists that cowbirds may not be as limited by drought as their host species, thus there is no indication that limiting conditions (no others identified) exist for the cowbird or will change either way in the future.

Phenology Scores

For most bird species, we considered photoperiod the predominant proximate factor controlling hormonal changes that initiate reproduction (Dawson and others 2001). Therefore, we gave these species neutral scores for the first phenology questions that relate increased vulnerability to reliance on a weather driven cue. Species that were considered influenced by temperature or precipitation cues include barn swallows, black phoebes, European starlings, lesser goldfinches, and western meadowlarks. Male barn swallows are arriving earlier on the breeding grounds as the result of changing climate in a Danish population (Møller 2004), and barn swallows have been arriving both earlier and later in different regions of Europe (Sanz 2002), which may be related to climate on wintering ground or migration corridors (Gordo and others 2005). Populations of cliff swallows in southeast Arizona time their breeding to coincide with summer rains (Brown and Brown 1996) probably to increase the likelihood that hatching time coincides with insect emergence. However, it is unlikely the cue is related to rainfall; we gave the cliff swallow a neutral score. European starlings have many behaviors under hormonal control that are altered seasonally by photoperiod (Cabe 1993). They are known to adjust breeding time to spring temperatures by as much as 14 days because of the changes in gonadal growth with temperature (Meijer and others 1999). They are already predicted to be laying eggs earlier in Europe (Both and te Marvelde 2007), which may improve nestling survival (Moore 1984). Breeding of the lesser goldfinch may be timed to precipitation but is not well studied, especially in the Interior West (Watt and Willoughby 1999). Though, for this assessment, we did assume that breeding initiation in New Mexico is tied to precipitation. The western meadowlark has also been observed to breed earlier in southern parts of its range, which we took to indicate this species relies on temperature cues. Though we did not consider the Lucy's warbler to be reliant on weather cues, many believe that migration of Lucy's warblers is timed to avoid hot summer temperatures (Johnson and others 1997; Stoleson and others 2000).

We considered all migrants at risk of mismatch between initiation of migration and critical resources at the destination site. These species include: ash-throated flycatcher, bank swallow, barn swallow, Bewick's wren, black-headed grosbeak, blue grosbeak, Bullock's oriole, bushtit, cliff swallow, common yellowthroat, summer tanager, western kingbird, western wood-pewee, southwestern willow flycatcher, and yellow-bellied chat (Svensson 1986; Garrison 1999; Turner and Rose 1989; Hill 1995; Rosenber and others 1982; Rising and Williams 1999; Wingfield 2008; Brown and Brown 1996; Guzy and Ritchison 1999; Robinson 1996; Bemis and Rising 1999; Sedgwick 2000; Thompson and Nolan 1973, respectively). We considered all resident birds resilient to mismatches related to distantly place cues and resources.

Food abundance in spring is an important factor driving fecundity in bird species breeding in temperate regions (review by Martin 1987). Typically, increases in insect abundance (a critical resource for these species) occur in spring. Changes in the timing of precipitation and increases in temperature have the potential to alter the timing of insect availability and abundance, and hence, fecundity. Therefore, we scored most insectivores as vulnerable to distinctly timed resources. For many of these species, this relationship was inferred through knowledge of the species' primary eating habits, though we were able to identify specific examples. As mentioned earlier, timing of blue grosbeak breeding is associated with monsoons (Rosenberg and others 1982), which is thought to relate to important insect populations for feeding young. Earlier nests in California populations of Bullock's orioles did better than later nests (Rising and Williams 1999) and breeding season is also lengthened during wetter summers (Rising and Williams 1999), probably due to the positive effect of precipitation on insect abundance. For the western kingbird, clutch sizes were larger in a year when insect abundance was higher in Kansas (Murphy 1988 in Gamble and Bergin 1996) and Arizona (Blancher and Robertson 1987 in Gamble and Bergin 1996). The southwestern willow flycatcher has a short nesting season that is thought to be limited by resource availability (Sedgwick 2000). Emergence of periodical cicadas is related to soil temperatures (Smith and others 2006; Williams and Simon 1995) and can increase reproductive success in red-winged blackbirds (Strehl and White 1986). We considered only one seed-eating species, the black-capped chickadee, vulnerable to changes in the timing of food resources, even though it is known to rely on insects during breeding season (Smoth 1993). Seed eaters such as the lesser goldfinch, which we considered neither at risk nor expected to benefit, were given a neutral score largely because data were sparse. White-breasted nuthatches, which consume both seeds and insects in spring, were also given a neutral score. In the case of the nuthatch, though changes in the timing of precipitation and increases in temperature have the potential to alter the timing of insect availability and abundance, consumption of seeds from the previous season may make this species less dependent on insect flush in spring. We did consider phainopeplas vulnerable to changes in time resources since egg laying in certain populations has been associated with fruiting, which changes with changing climate conditions (Walsberg 1977). No information was available on the potential influence of timed resources on black phoebes, eastern bluebirds, lesser goldfinches, and indigo buntings, and we did not consider these species vulnerable or resilient to potential risks associated with timed events.

We considered species that had an extended breeding season or multiple broods more resilient to potential mistiming issues during the breeding season. Brown-headed cowbirds have an extended breeding season which is influenced by the breeding seasons of its host species and appears to be able to adjust to changes in the timing of host species (Butler 2003). Eastern bluebirds also have a prolonged breeding season and can raise multiple broods. Phainopeplas will re-nest if their nest is destroyed early in the breeding cycle, though it is uncertain whether or not phainopeplas raise multiple broods. There are no records of second broods in the same area, but pairs may be creating second nests in different habitats (Chu and Walsberg 1999). Barn swallows (Brown and Brown 1999), black phoebes (Wolf 1997), black-chinned hummingbirds (Baltosser and Russel 2000), blue grosbeaks (Ingold 1993), bushtits (Sloane 2001), western meadowlarks (Bemis and Rising 1999), indigo buntings (Payne 2006), house finches, Bewick's wrens (in Kansas and others 1997), and eastern bluebirds are known to raise multiple broods per season.

Species that were considered vulnerable to phenology changes because they only produced a single brood within a relatively short period (<2 months) included black-capped chickadees (Smith 1993), black-headed grosbeaks (Hill 1995), cliff swallows (Brown and Brown 1995), white-breasted nuthatches (Grubb and Pravosudov 2008), lesser goldfinches (Watt and Willoughby 1999), common yellowthroats (Guzy and Ritchison 1999), and spotted towhees (Greenlaw 1996).

We considered European starlings (Cabe 1993), Lucy's warblers (Johnson and others 1997), ash-throated flycatchers (Cardiff and Dittman (2002), summer tanagers (Colorado River Valley, Robinson 1996), and western kingbirds (references in Gamble and Bergin 1996) capable of double brooding. In addition, we considered species that are able to re-nest late in the season resilient to timing mismatch during breeding. Bank swallow pairs will re-nest if the nest fails up to halfway through the breeding cycle, and their breeding season may last several months (Garrison 1999). Red-winged blackbirds will initiate multiple breeding attempts per season due to nest failures, but it is extremely rare for them to raise more than one brood per season. The southwestern willow flycatcher typically only raises one brood per year; however, some pairs will raise a second brood or re-nest after a nest failure, particularly in the northern part of their distribution (Sedgwick 2000). Yellow-breasted chats readily re-nest if nests are destroyed (Eckerle and Thomson 2001).

Biotic Interaction Scores

Food Resources: Most insectivores were considered neither vulnerable nor resilient to climate related changes in food. For the most part, these species were considered generalists (many opportunistically eat vegetation) and there were not clear expectations with respect to the overall abundance in insects. Ash-throated flycatchers (Cardiff and Dittman 2002), Bewick's wrens (Kennedy and White 1997), black phoebes (Verbeek 1975), black-capped chickadees (Smith 1993), blue grosbeaks (Ingold 1993; Rosenberg, Ohmart, and Anderson 1982), Bullock's orioles (Guzy and Ritchison 1999; Sloane 2001), eastern bluebirds (Gowaty and others 1998), European starlings (Cabe 1993), red-winged blackbirds (also eat grain, Yasukawa and Searcy 1995), spotted towhees (eat plant matter in the non-breeding season Davis 1957b in Greenlaw 1996), and western kingbirds (Gamble and Bergin 1996) received neutral scores.

Seed-eating species were also given neutral scores when their diets were diverse. The lesser goldfinch diet consists primarily of seeds, but also some fruits and flowers (Watt and Willoughby 1999). The indigo bunting eats mostly seeds, especially grass seeds, but also various insects (particularly in the breeding season) and berries (Payne 2006).The house finch eats buds, seeds, and fruits of many species (Hill 1993). Though drought is likely to reduce these food sources, the reliance of this species on human landscapes will probably mitigate negative climate impacts by providing alternative food resources.

Some insectivores were clearly specialists, but were also able to opportunistically prey upon or use other food resources. Lucy's warblers are among the most specialized insectivores, feeding exclusively on insects and small arthropods year round (Johnson and others 1997; Yard 1996). Summer tanagers are specialists on bees and wasps, though they also eat a wide variety of other insects and fruit during migration and on the wintering grounds. Western wood-pewees primarily feed on flying insects, especially bees, flies, ants, wasps, beetles, moths, and bugs (Bemis and Rising 1999). None of these species were considered vulnerable to future declines in food resources because they eat a variety of food. Western meadowlarks feed on a wide variety of grains, seeds, and insects (Davis and Lanyon 2008). White-breasted nuthatches consume a variety of insects as well as nuts and seeds (reference in Grubb and Pravosudov 2008). Yellow-breasted chats feed on small invertebrates, mainly insects and spiders, as well as fruits and berries when available (Eckerle and Thomson 2001).

Some insectivores, especially those that specialize on aquatic insects and have a demonstrated tendency to suffer from weather-mediated starvation were considered at risk of food declines. For these species, we felt that their demonstrated sensitivity to food losses should be reflected in biotic interactions even if the mechanism (drought) was considered elsewhere. Bank swallows are aerial predators that feed almost exclusively (>99% of diet) on flying or jumping insects, most of which rely on aquatic developmental stages. Studies of European populations have found them vulnerable to starvation on their wintering grounds when rains fail and their food supply is reduced (Szep 1995). Presumably, MRG populations may also be at risk as precipitation is predicted to be reduced and periods of drought are expected to increase. Though drought effects were often considered in physiology (particularly for the last question) and phenology sections, this species was scored as vulnerable to food declines given its demonstrated sensitivity to food loss and its reliance on riparian associated insects. Phainopeplas, which feed primarily on mistletoe berries, were also considered susceptible to food declines. Wintering populations of phainopeplas vary dramatically in size and location depending on weather and the availability of fruiting plants (Chu and Walsberg 1999). Drought was found to make mistletoe infested plants more susceptible to mortality in the Mojave Desert, but not in the Sonoran Desert (Spurrier and Smith 2007), which is more similar to New Mexico habitats. Therefore, occurrence of mistletoe-infested plants is not expected to change in during summer in the MRG. In the MRG, phainopeplas feed extensively on New Mexico privet berries (RMRS data), which are vulnerable to severe freezes and possibly drought. Given predictions for increased droughts, mistimed frosts, and observed impacts on phainopepla populations, this species was considered at risk of food declines due to climate change. Willow flycatchers are primarily insectivorous (Sedgwick 2000), with a fairly diverse food base. However, willow flycatchers are dependent upon localized food sources during nesting and, in particular, insects that rely on water for some phase of their life cycle or are otherwise drawn to water, which were considered at risk of decline.

Predators: Most species did not have a demonstrated or predicted change in predators. Many were preyed upon by a diverse range of predators or were not known to be overly influenced by predation. American kestrels may be important predators of bank swallows (Garrison 1999). Kestrels seem to be resilient to future climate effects, but the impact of this on bank swallow populations has not yet been quantified.

Symbionts: Symbiotic relationships were only important to secondary cavity nesters. Lucy's warblers use verdin and woodpecker nests, but are not wholly dependent on these species as they can also nest in a wide variety of natural cavities and crevices. White-breasted nuthatches, on the other hand, are not known to excavate their own cavities and generally use natural cavities or holes constructed by woodpeckers, and were considered sensitive to the decline of these species.

Disease: Bank swallows were among those species that had demonstrable disease interactions. Infestations by various parasites, especially mites and/or blowfly larvae, are one of the main causes of nestling mortality. Infestations increase later in the breeding season and are more prevalent in larger colonies. Nestling survivorship may be reduced by ectoparasites (Brown and Brown 1986). Warmer climates could have two impacts that would cause increased ectoparasite loads and greater mortality. First, warmer temperatures speed development and increase overwinter survival of larvae. Second, loss of riparian habitat may reduce foraging grounds and lead to greater concentrations of birds, which increase transmission. Numerous parasites and diseases have been documented in cliff swallows. Infestations of various ectoparasites, especially the swallow bug (*Oeciacus vicarius*), are the major cause of death among cliff swallow nestlings (Chapman and George 1991). In addition to causing direct mortality, these parasites frequently reduce growth rates and body size in nestlings making them more susceptible to starvation and other types of mortality. Larger colonies and later season nests are at a greater risk from ectoparasites. Severe infestations may cause entire colonies to desert their nests simultaneously. Infestations of blowflies results in nestling mortality in eastern bluebirds and may increase with warmer weather (Mason 1944; Merino and Potti 1996).

We considered the threat of West Nile Virus and other mosquito mediated diseases as neutral in this assessment. It is suspected that chickadees may be especially vulnerable to West Nile Virus (Bonter and Hochalka 2003). While precipitation and ambient temperature likely play a role in pathogen transmission, forecasting changes in disease intensity is currently difficult (Ladeau and others 2008). It remains to be seen whether this disease will become more prevalent in the MRG. If so, many species will be at risk. The lesser goldfinch is social and may be vulnerable to crowding, though it is not yet known to suffer widespread mortalities due to disease or parasitism.

Competitors: Competitive interactions were the most commonly recorded effects within biotic interactions. However, the ultimate impact of climate changes on these interactions was difficult to assess in many instances. This species competes with other species for nest cavities and has been observed evicting other species from cavities (e.g., woodpeckers, titmice, wrens, bluebirds, and flickers; see references in Cardiff and Dittmann 2002; Lanyon 1963). Furthermore, wintering individuals aggressively defend fruiting trees in Sonora, Mexico (Russell and Monson 1998). However, it was not possible to attribute this behavior to a realized increase in competitive interactions for other MRGB species since the flycatcher is not expected to be particularly resilient to climate change.

European starlings and house finches may compete with bank swallows, though the effect of this on bank swallow populations is not known (Garrison 1999). Conversely, house sparrows are considered serious competitors for barn swallows and are believed to have caused local extinctions of barn swallows in some areas (Weisheit and Creighton 1989). House sparrows were not directly assessed in this assessment but are highly adaptable, benefit from human development, and have a prolonged breeding season, and thus may be expected to do well under climate change scenarios leading to increased competition for nest sites. House wrens frequently destroy nests and evict Bewick's wrens from cavities (Kennedy and White 1996). Competition with house wrens is correlated with the decline of Bewick's wrens throughout the eastern United States, and Bewick's wren nesting success is lower in areas where house wrens co-occur (house wrens remove eggs and destroys nests; Kennedy and White 1997). Though not currently sympatric in the MRG, house wren expansion resulting from climate change would have a large impact on this species. Similarly, most nest losses in black-headed grosbeak populations in the Sandia Mountains of New Mexico were due to predation by western scrub-jays and Stellar's jays (Hill 1988b in Hill 1995). These species do not overlap to a significant degree in the MRG, but future changes may affect distributions in such a way as to increase competitive interactions. Bullock's orioles will defend nectar resources from tanagers and wrens (Cruden and Hermann-Parker 1977). Cliff swallows are frequent nest parasites of conspecifics within their colony. House sparrows can be a major competitor for swallows as they will evict swallows from their nests during the summer or establish themselves in nest gourds over the winter. In the northeastern United States, severe competition with house sparrows is believed to limit the breeding range of cliff swallows (Silver 1993). House sparrow control programs have greatly increased the size of cliff swallow colonies in many instances. Competition with house sparrows is believed to be much lower in the western United States, but may be locally important where house sparrows are common. House sparrows are common in the MRG region and with prolonged breeding and broad adaptability may be expected to increase.

Cowbird parasitism was noted for several species, but is not currently considered a threat for most due to a lack of apparent effect on bird populations. However, for each of these species, increases in the populations of brown-headed cowbirds, which are considered resilient to climate affects, may lead to increased parasitism. Brood parasitism is low with most hole-nesting species (Smith 1993). Brood parasitism is not common in black-headed grosbeaks based on Hill (1995), but this species is frequently parasitized by brown-headed cowbirds in the MRG (RMRS data). Blue grosbeak nests are frequently parasitized by brown-headed cowbirds, and parasitized nests often fledge no genetic young, though nests late in the season can escape parasitism (D. Hawksworth, pers. comm.). Bushtit nests, which are gourd-shaped pendulous nests, are very rarely parasitized by brown-headed cowbirds (Smith 1979). Brown-headed cowbirds parasitize the nests of common yellowthroats with many studies reporting parasitism rates of 20-50% (Guzy and Ritchison 1999). Furthermore, studies in Minnesota and Michigan suggest that brood parasitism there is a major regulator of the population (Hofslund 1957 in Guzy and Ritchison 1999). Lucy's warblers are reported as a common cowbird host (Friedmann and Kiff 1985), but no brood parasitism was observed in RMRS data for MRG (n = 10). Red-winged blackbirds are very frequently parasitized by cowbirds; parasitism rates of up to 74% have been recorded, although rates vary widely (Blankespoor and others 1982; Searcy and Yasukawa 1995). Summer tanagers are affected by brood parasitism by brown-headed cowbirds, but effects on success are not well known (Robinson 1996) and they are not commonly parasitized in MRG Bosque (D. Hawksworth, pers. comm.). Reports of brood parasitism by brown-headed cowbirds are rare in western kingbirds (Friedmann 1963 and Smith 1972 in Gamble and Bergin 1996), possibly a result of western kingbirds rejecting cowbird eggs in the wild as they do in captive experiments (Rothstein 1975 in Gamble and Bergin 1996).

Western meadowlarks are a very frequent cowbird host (Davis and Lanyon 2008), and numerous published papers report over 50% of meadowlark nests parasitized by cowbirds. Early season nests are less vulnerable to parasitism because cowbirds are fewer or absent at the beginning of meadowlark breeding. Southwestern willow flycatcher nests are parasitized by brown-headed cowbirds, with failure of the genetic young. Currently willow flycatcher reproduction occurs relatively late, which does not allow for breeding attempts too early from cowbird brood parasitism (Robinson and others 1995). Yellow-breasted chats are very frequent cowbird hosts in New Mexico and elsewhere (Eckerle and Thomson 2001). Fifteen of 41 (37%) chat nests in RMRS dataset were parasitized by cowbirds. Brown-headed cowbirds commonly lay eggs in indigo bunting nests and infested nests fledge fewer genetic young and have higher predation rates (Payne 2006).

Mammals Results

Chiroptera (Bats)

We assessed eight bat species: big brown bat (*Eptesicus fuscus*), fringed bat (*Myotis thysanodes*), occult bat (*Myotis occultus*), pallid bat (*Antrozous pallidus*), long-legged bat (*Myotis volans*), yuma bat (*Myotis yumanensis*), Mexican free-tailed bat (*Tadarida brasiliensis*), and hoary bat (*Lasiurus cinereus*).

Habitat

Habitat scores considered trends (decreases/increases) for associated vegetation and required habitat components, changes in habitat quality, requirements for additional habitat, and capacity of species to move with transitions in critical habitat features. Bats generally breed in the fall, delay implantation and give birth the following spring. Females form maternity colonies during the summer while nursing young. For most bat species, we considered the habitat requirements of these maternity colonies for questions regarding breeding habitat and winter habitat or hibernacula was considered non-breeding habitat.

Vegetation Association: The Yuma bat is the only year-round resident of MRG habitat considered in this assessment (Chung-MacCoubrey 2005). BISON-M (2009) indicates the Rio Grande Valley as a stop-way for many bats (e.g. big brown bats) en route to hibernacula. In general, vegetation type or zone is not commonly thought to be an important determinant of the presence of bat species (e.g. Occult bat Heritage Data-Arizona Game and Fish Department). Rather, bats need sites with appropriate roost sites and foraging grounds. Riparian areas are often favored habitat by bats because of the concentration of insects and open water, which aids echolocation. The healthy riparian gallery of the MRGB provides both and most bats species were considered sensitive to the loss of riparian forest during summer residence. Among those bats that appear to depend on riparian habitat, we considered the pallid bat, the occult bat, the hoary bat, the long-legged bat, and the Yuma bat (Frey 1999; Geluso 2007; Hermanson and O'Shea 1983) vulnerable to habitat loss. Other species, such as the big brown bat, do not appear to rely on riparian habitat though they are commonly reported from the area (Valdez and Cryan 2009). We also felt that fringed bats were slightly resilient to habitat changes in the MRG because they are largely transient in the summer and do not appear to rely on riparian features.

We did not consider any bats vulnerable to changes in non-breeding (winter) vegetation associations because most species migrate or hibernate during winter months. We did note that the Yuma bat is more strongly associated with water than most other bat species which could represent an increased sensitivity to habitat disturbance (Sims 2000).

Habitat Components: Bats commonly use special habitat components (Bagne and others 2011) that we considered likely to decline under future conditions. Specifically, we expect declines in snags, tree roosts, and open, slow moving waters to negatively impact the fringed bat, Yuma bat, and hoary bat (Shump and Shump 1982). The Yuma bat is commonly cited as using human structures for day roost sites (Scheel and others 2007), but may in fact be utilizing trees near human structures (Evelyn and others 2004). Though surveys in other states show a preference for rock crevices (e.g., Schell and others 1996 in Texas), fringed myotis in New Mexico exclusively use snags and live trees (Chung-MacCoubrey and others 1996). The hoary bat is the only species in this region to raise its young solitarily in the open foliage of trees (Willis and Brigham 2005) and appears to have a fairly strong preference for large diameter trees that are still alive (Evelyn and others 2004). The big brown bat, occult bat (though they may prefer trees), long-legged bat, pallid bat, Yuma bat, and Mexican free-tailed bat use a wide variety of roosting structures, including those that are made by humans (Cartron and others 2008; Hermanson and O'Shea 1983; Kurta and Baker 1990; Scheel and others 1996; Wilkins 1989), and we did not consider these species vulnerable to loss of habitat components.

Habitat Quality: Riparian areas are often favored by bats because these habitats harbor a concentration of insects, and the presence of open water aids ecolocation. Slow waters might also be an important feature of habitat because fast running water can negatively impact the effectiveness of bat ecolocation (Grindal and others 1999).Yuma bats, in particular, are usually found only near bodies of water or streams, above which they forage (BISON-M), and we considered them vulnerable to habitat quality declines due to decreasing availability of ideal forage grounds. We also considered hoary bats, which soar and glide along waterways and above tree tops while foraging, and the occult bat, a water surface forager reliant on slow moving water, vulnerable to declines in habitat quality. The occult bat and long-legged bats were not considered prone to this vulnerability. The occult bat uses hawking and forages in shrubbery; their presence in areas without water is considered transient (BISON-M). Long-legged bats are aerial foragers and are not thought to rely on open bodies of water (Frey 1999). However, long-legged bats probably rely on canopy structure, which we expect to decline.

Dispersal and Migration: We considered all bat species good dispersers and not vulnerable to shifts in habitat. We considered the hoary bat and Mexican free-tailed bat vulnerable to climate change impacts on migratory habitats. Though many other species, such as the fringed bat, long-legged bat, and big brown bat, migrate, it is likely on a much smaller scale (Rasheed and others 1995), and we did not consider these species vulnerable to climate effects on additional habitats.

Physiology

We found similar physiological sensitivities and resilience capacities for the bat species in response to expected climate changes. The primary *limiting conditions* for bats within the MRGB related to increasing temperature. We considered pallid bats vulnerable to increasing temperatures because high temperatures affect their capacity to forage and can limit roost selection (BISON-M 2009; Hermanson and O'Shea 1983). Other bat species appear much more resilient to high temperatures. In particular, the big brown bat has a good capacity to deal with extreme conditions (Carpenter 1969). Similarly, Carpenter (1969) reported that Mexican free-tailed bats occupy roosts at temperatures reaching 40 °C. Further, Mexican free-tailed bats appear to fair much better than other species under warmer conditions though they may live only one-fourth to one-third as long as hibernating species under cooler temperatures (Wilkins 1989). However, several authors noted that bats may actually be less resilient to higher temperature because it could prevent bats from entering torpor, an important mechanism enabling them to survive food and water shortages (Carpenter 1988).

Water availability may also represent a physiological limiting condition if future conditions become more arid. Bats show little ability to concentrate urine or other physical traits for water conservation and it is likely that behavior, specifically roost selection, plays a large role in thermoregulation in bats and their capacity to persist in arid environments (Carpenter 1969). Several species show a strong reliance on free water sources, particularly during lactation when female bats have an increased need for water (Adams and Hayes 2008). In one study, the effect of increased warming and coinciding reductions in water availability is predicted to reduced lactating capacity in females to the extent that there will be declines in bat populations (Adams and Hayes 2008). We considered the big brown bat and Mexican free-tailed bat vulnerable to reductions in metabolic water sources (Carpenter 1969; Wilkins 1989). The pallid bat has adaptations for dry conditions and reduces food intake when water isn't available (Hermanson and O'Shea 1983), but the long-term benefits of this capacity are probably somewhat limited. We did not consider hoary bats sensitive to reductions in free water because they have a relatively good capacity to conserve water perhaps due to their thick pelage, which protects against heat and loss (Shump and Shump 1982).

Extreme Events: We considered increasing frequency of storms a likely issue and source of mortality for big brown bats (Kurta and Baker 1990) and foliage roosting species such as the hoary bat and pallid bat, which may experience greater exposure during summer months (Willis and Brigham 2005).

Activity Limitations: We did not consider any of the bats at risk of climate related limitations to activity periods (Appendix 1; see Bagne and others 2011). Though sensitive to high temperatures, the pallid bat is nocturnal and thus the impact of higher temperatures on active periods was presumed minimal. In addition, increased temperatures may benefit bats on a seasonal scale. Aerial foraging is limited by insect activity, which, in turn, is limited by temperature (Anthony and others 1981). Higher temperatures are favorable for gestation and lactation phases of female reproductive cycles (McCain 2007).

Metabolic Inhibition: Torpor is used opportunistically in many mammal species and we considered the capacity to torpor a source of resiliency. However, the benefits of seasonal torpor are not always clear, particularly when the energetic benefit corresponds to ambient temperature, which means less energy savings under warmer conditions. Metabolic rates for bats during torpor vary positively with ambient temperatures (Hermanson and O'Shea 1983). Thus, higher temperatures may cause animals to burn through reserves more quickly and increase its risk of starvation, a major source of mortality for hibernating bats.

We considered daily torpor capacity, which helps bats deal with resource or water shortages, a beneficial trait, but seasonal torpor patterns, particularly those driven solely by temperature, a neutral trait under changing climate. To reduce overlap with other sections that consider timing issues and weather cues, species were considered increasingly vulnerable (or less resilient) to climate change as they moved from having the capacity to undergo daily torpor without relying on seasonal torpor, to those that underwent both daily and seasonal torpor but the latter was less efficient under warmer conditions (hibernacula exposed), to those that had no capacity to torpor but could store water or fat over the long term, to those that could not reduce metabolic energy requirements or store energy in the long term. Bats fell within one of the first two categories (with the exception of the Mexican Free-tailed bat) (Table 3.1, Appendix 1). For instance, we considered pallid bats to have a metabolic advantage because they hibernate (Hermanson and O'Shea 1983) during cold months and undergo nocturnal cycles of torpor as a means to deal with cool temperatures, low food, or fat reserves and to reduce water loss (Hermanson and O'Shea 1983; Rambaldini and Brigham 2008).

Phenology

Climate Cues: Hibernation, migration and parturition in bats varies with climate conditions (Hermanson and O'Shea 1983), and we generally consider bats vulnerable to risks associated with reliance on weather related cues. Hibernation and migration of many bat species is tied to insect emergence (Popa-Lisseanu and Voigt 2009), which, in turn, is tied to temperature (Valdez and Cryan 2009). Local migration of Mexican free-tailed bats from maternity roosts is largely determined by local conditions: specifically movements are correlated with weather patterns, the stress of overheating in caves and gas (ammonia) buildup (Tuttle 1994). Migration of the hoary bat is also thought to be initiated by temperature (or barometric pressure) (Geluso 2007) and temperature may also cue initiation of torpor (Twente and others 1985).

Asynchrony with Discretely Timed Resources: Insect emergence, especially when it consisted primarily of winged insects (e.g., big brown bat, hoary bats, long-legged bat, and Mexican free-tailed bat), was considered a qualifying discrete event and indicative of increased risk of timing mismatch. Long-distance migrants such as the hoary bat and Mexican free-tailed were considered vulnerable to potential mismatch between onset of migration and presence of event (insects).

Mismatch Due to Distantly Placed Resources and Events: Like birds, we considered long-distance migrants (hoary bat and Mexican free-tailed bat) at risk of increased mismatch due to distantly placed resources. For the hoary bat, migration through New Mexico is hypothesized to be timed to insect, in particular the cutworm, emergence (Valdez and Cryan 2009). Migrations may also change if warmer temperatures lead to an increase in night-flying insects during colder months (longer residence in New Mexico) (Geluso 2007). For many bats, temperature also determines arousal from hibernation and insect presence, and we considered these events to be closely tied. Therefore, we considered hibernating bat species as resilient to potential mismatches related to great distance between climate and resource, unless they also hibernated.

Asynchrony During Reproduction: All bat species have a single, tightly timed reproductive cycle per year (Kurta and Baker 1990; O'Farrell and Studier 1980), and we considered them to be resilient to changes in the timing of resources. Temperature has several effects for bat parturition, including delayed gestation under cold climates, but that was not considered relevant to this assessment (Racey 1973). Some bats, such as the pallid bats, in which reproductive synchrony is important as young are left in nursery crèches while mothers forage, may benefit if warmer temperatures lead to greater degree of synchrony among mothers (cooler weather often decreases such) (Lewis 1993). Still, the impact of temperature on so many aspects of bat ecology is likely to have more risk associated with them than benefit.

Biotic interactions

Prey: We did not identify a clear positive or negative trend in food resources under climate change. All bats assessed were insectivores and predictions are limited with respect to climate effects on insect taxa. However, insect abundance and response to climate change will have resounding impacts on the biotic communities for many species, especially bats, which have high energetic costs while nursing young—females must eat their weight in insects each night while lactating (Kurta and Baker 1990). Though a specialist (Shump and Shump 1982), the hoary bat was not considered vulnerable to climate related losses in prey as a relationship among prey species, climate, and bat survival has not been established. Similarly, long-legged bats are aerial foragers that prefer moth insects and may be considered specialists and, therefore, vulnerable, but they also feed on a variety of other insects (Warner 1985, Warner and Czaplewski 1984).

Disease: We believe increased transmission of parasites and disease is likely with loss of roosts, reduced roost choices, and overcrowding (Hermanson and O'Shea 1983). Rabies poses a real threat to bat populations (e.g., big brown bats as described in Kurta and Baker 1990), is directly transmitted, and we expect crowding will increase the incidence of this disease. The colonial lifestyle of bats, especially those that form large colonies such as the Mexican free-tailed, also leaves them susceptible to epidemic disease (McCracken 1986).

Shrew

The MRG harbors one shrew species, the Crawford's or desert shrew (*Notiosorex crawfordi*). We found the shrew to be the most resilient of mammal species assessed in this report, though it has a number of physiological traits that might limit its populations under climate change.

Habitat Scores

We expect the shrew to experience increases in its preferred habitat along the MRG, though it is less clear how changes in habitat components and characteristics associated with quality will affect this species. The desert shrew (also called the gray shrew) has a broad distribution in southwestern and south-central United States and northern Mexico and is commonly associated with semi-desert scrub habitats with or without permanent water source (Armstrong and Jones 1972). The MRG Field Guide (Catron and others 2008) states that this may be one of the most common, if least observed, mammals in the Bosque. The desert shrew is unique from other shrews because it inhabits relatively xeric habitats and is relatively social (no antagonism when caged together). Laakonen and others (2002) reported positive association with scrub habitat and negative associations for other flora in California habitats and, thus, it appears the shrew may be favored by future expected conditions in the MRG. Shrews also appear to be somewhat resilient to fragmentation due to small range (Laakkonen and others 2001). However, the shrew has a generalized need for cover, and reduced cover negatively affects shrew capacity to hide from predators (Armstrong and Jones 1972). It is unclear whether shift to scrub will result in less cover since this is the habitat most commonly associated with the species, so we considered the potential loss of habitat components as neither potentially benefiting nor harming future shrew populations. However, the shrew is quite vulnerable to habitat changes that do not favor its presence as it has very limited dispersal ability (Armstrong and Jones 1972), and we did consider this vulnerability under the dispersal question of this section.

Physiology Scores

The desert shrew has several “adaptations” distinct to its taxon, which are thought to relate to its capacity to survive in arid environments: it is nocturnal, has highly efficient kidneys and a relatively low resting metabolic rate, and is often observed to torpor when conditions are dry (Simons and others 1990). However, it may rely on mesic environments under hot conditions and at lower elevations this animal tends to inhabit more moist environments (Allen 2000). We felt that there were enough indications of a physiological sensitivity to high temperatures due to the reliance of the shrew on riparian areas in lower (hotter) elevations and suggestions that it may prefer moist conditions (Simons and others 1990) to consider this species vulnerable to future temperature extremes.

We do not, however, predict a change in activity levels due to climate change for the shrew. There are contrasting views of how activity levels are influenced by climate with one study finding no change in capture rates with varying precipitation (Laakkonen and others 2001, 2003), whereas another point to increases in activity with precipitation (Simons and others 1990). Precipitation patterns are not well known for the future and the lack of a clearly defined trend in shrew activity levels inhibits future predictions of potential changes in shrew activity.

Starvation induced torpor common in desert shrews (Allen 2000) and we considered this species resilient to resource variations in the short term due to this capacity. Lindstedt (1980), who induced shallow torpor with food limitation (to maintain body weight), observed that shrews only enter this state at ambient temperatures falling between 20-25 °C and suggested that there are two “preferred” states for the species: one near 38 °C and the other, reserved for emergency situations, near 28 °C. This may indicate some cost of higher temperatures on the effective metabolic savings for shrews under warmer conditions. We considered the shrew vulnerable to climate-induced changes in resources over longer periods because it does not have the capacity to store or inhibit energy use over the long term and has an extremely high metabolic rate. We also considered desert shrews, which live 1-2 years (Allen 2000), sensitive to drought related declines in populations.

Phenology Scores

We considered desert shrews resilient to most phenology issues. Shrews are active year-round and do not rely on resources that may be limited in space or time. Shrews also have a very short generation time and populations can respond to changes in their immediate environment: females become productive at 2 months (Allen 2000) and have 2-3 litters of 3-5 (Armstrong and Jones 1972) pups (1-6 according to Punzo and Lopez 2003) per year. Breeding season is typically during early and late summer, but shrews may also breed in the fall (Baker and Spencer 1965) during mild weather, and there is at least one report of year-round reproduction (Simons and others1990).

Biotic Interaction Scores

We did not consider the desert shrew at risk of significant change in any of its biotic interactions. Shrews have general diets that include insects and small vertebrates

(Armstrong and Jones 1972). Though shrews have a very high metabolism—they must eat 75% of their body weight per day and are extremely susceptible to loss of prey items (Allen 2000)—there are no clear trends in invertebrate abundance under climate change. Though not considered an issue in this assessment, future studies on invertebrate populations will influence our interpretation of this potential effect for the shrew. Laakonen and others (2001) found a strong negative association between introduced Argentine ants and shrew capture success, potentially indicating a negative competitive interaction. Rainfall events facilitate the spread of the Argentine ant (Heller and others 2007). However, it is not clear from climate predictions that future scenarios will favor or inhibit the inevitable arrival of the ant to the MRGB.

Rodents

Rodents comprised the most diverse group of mammals and the rodents of the MRG are expected to exhibit a wide range of responses to climate change. We assessed 15 species of rodents (Fig. A1).

Habitat Scores

Vegetation Association: Rodents are year-round residents within the MRG and breeding and non-breeding habitats were given similar scores for all species. We did not expect any rodents to see increases in associated habitat. We gave neutral scores to species with general habitat associations or that were more strongly associated with abiotic correlates than vegetation. Porcupines, hispid cotton rats (*Sigmodon hispidus*), and pocket mice (*Perognathus flavus*) were considered habitat generalists. Porcupines utilize a broad range of habitats, are widespread, and

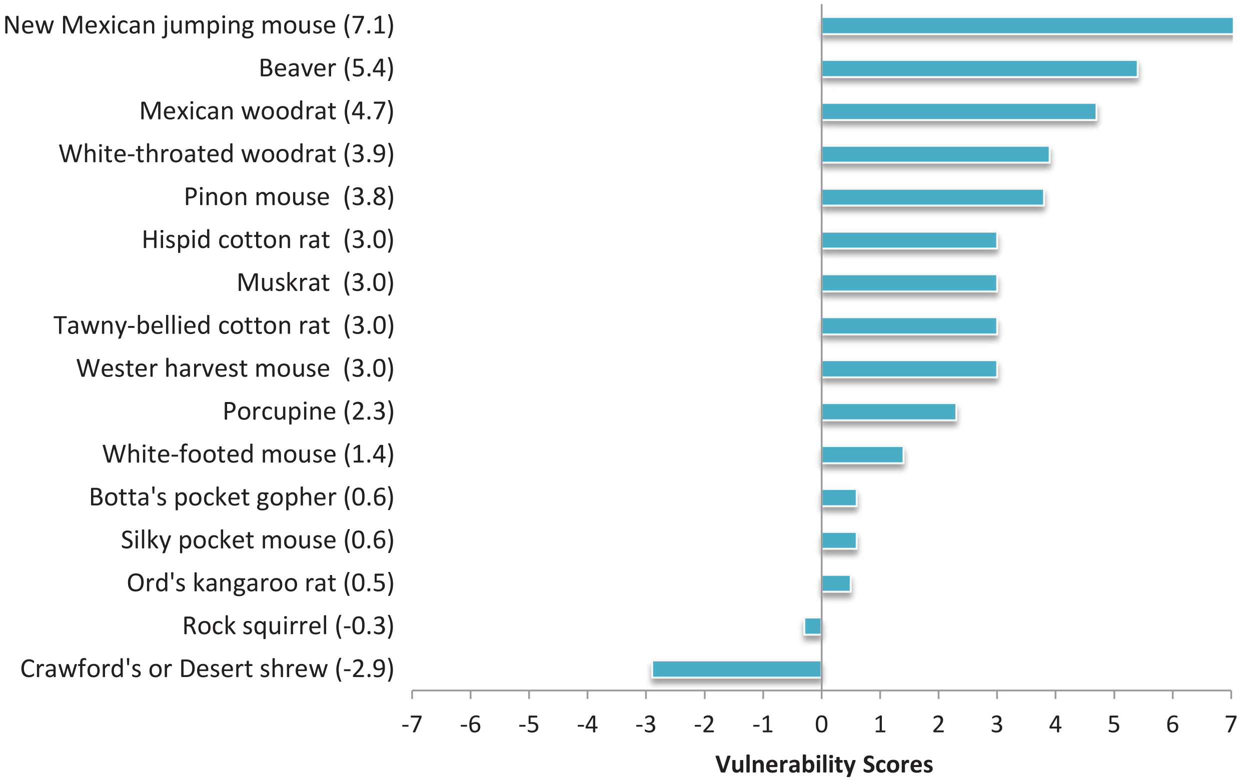

Figure A1. Overall scores and ranking of 15 rodent species.

are especially common in the western United States and in Canada (Woods 1973). The hispid cotton rat is strongly associated with grassy patches that have some shrub overstory and favors dense vegetation, particularly grasses, though it is also found in cattails along streams and in desert scrub habitats (NatureServe 2008; Sullivan 1995). Pocket mice (*P. flavus*) have somewhat general habitat preferences (Davis and Schmidly 1994). Many rodents rely on physical features not expected to be influenced by climate affects: Ord's kangaroo rat (association with sand type, Fitzgerald and others 1994; Moulton and others 1981; Ribble and Samson 1987), rock squirrels (rocky habitats, Oaks and others 1987), and pocket gopher (soil characteristics, Jones and Baxter 2004).

We considered rodents likely to experience habitat loss when they: (1) associate with riparian habitat or features and thus will experience a reduction in needed elements, or (2) rely on the mesic environment to deal with high temperatures. We considered the meadow jumping mouse vulnerable to future loss of riparian habitats and associated vegetation structure (Frey and Malaney 2009). The Mexican woodrat was considered at risk of loss because its association with riparian habitats is thought to derive from a requirement for cooler sites (Cornely and Baker 1986). Similarly, white-throated woodrats are dependent upon the riparian corridor in reaches of the Rio Grande that are surrounded by xeric habitat (Frey 1995; Hoffmeister 1986). Western harvest mice appear to depend upon the dense vegetation provided by riparian areas (BISON-M 2009; Webster and Jones 1982). Similarly, the tawny-bellied cotton rat typically inhabits well-developed grasslands (Baker and Shump 1978) and is presently limited to southwestern New Mexico and the MRG Valley from Socorro to Sandoval counties (Mohlenrich 1961). Though a widespread species, the white-footed mouse tends to inhabit riparian areas and ravines in prairie or semi-desert habitats (Lackey and others 1985) and was considered at risk due to changes associated with habitat shifts, though not necessarily habitat loss given its flexibility in habitat associations. The pinyon mouse is closely associated with rocky habitat and rocky slopes with a rather exclusive association with pinyon-juniper habitat (Hoffmeister 1981), which is limited in the MRGB. The silky pocket mouse is common to desert grasslands but also has clear associations with riparian habitats and has been found in sycamore, cottonwood, and rabbit brush riparian habitats (BISON-M 2009).

Habitat Component Change: For many species, habitat components were often not unique features from habitat type already considered under vegetation associations (e.g., thick grasses of riparian habitats for western harvest mice and the tawny bellied cotton rat), and many species received neutral scores indicating no expected effect. In one exception, we considered the New Mexico meadow jumping mouse vulnerable to both habitat and component loss because it appears to be particularly dependent on riparian habitat and is associated with permanent waterways that contain dense vegetation (Frey and Malaney 2009).We also made exceptions for species that had generalized habitat needs and, as a result, were not expected to experience a decline (or increase) in preferred habitat type, but did appear to rely on specific nesting or denning substrates (e.g., dense vegetation). We considered beavers and muskrats, which rely on the presence of water, vulnerable to reduced flows and hydrological changes resulting from climate change. Specially, beavers were considered to be at risk due to habitat component loss where water level, duration of flow, and vegetation characteristics where considered important elements. Most other rodents use a diversity of nesting sites or lacked a clear requirement or utilized substrates that are not likely to be influenced by climate variation and were not considered vulnerable to component loss.

Habitat Quality: For most species, we were not able to determine a change in habitat quality as a result of climate change. We did find evidence that beavers, muskrats, and hispid cotton rats may experience declines in habitat quality due to climate change effects on river flow. Beavers require permanent flowing water sources in order to use their lodges (Jenkins and Busher 1979) and must move once siltation has reduced the depth of a beaver pond (Struebel 2000). Similarly, muskrats need slow moving water of a minimum depth and silting due to flood activity, or they need fast moving waterways—both have negative impacts on habitat suitability for the muskrat (BISON-M 2009; Willner and others 1980). Though not directly related to water volume or depth, reproduction in the hispid is tied to precipitation, particularly in marginal habitat (Texas mammals), and habitat use by the cotton rat appears to positively correlate with the density of monocot or other grass species (Sullivan 1995). We expect greater variation and potentially declines in both these habitat features with negative impacts for the hispid cotton rat.

Dispersal: We considered most rodent species with small home ranges, including the New Mexico meadow jumping mice, silky pocket mice, Mexican woodrats, white throated woodrats, and *Peromyscus* mice (Best and Skupski 1994; BISON-M 2009; Lackey and others 1985; Hoffmeister 1981), vulnerable to climate effects that shift habitats because they are dispersal limited. We also considered rodents that established elaborate nesting systems such as the kangaroo rats (Sullivan 1995 and references therein) or that relied on certain habitat features such as the rock squirrel (cover, rocky habitat) for movements, dispersal limited and vulnerable. For instance, the average home range of a western harvest mouse is 200-300 ft with a homing ability up to 1000 ft (Webster and Jones 1982). Though this represents a relatively generous distance as compared to other rodents, the western harvest mouse is dependent upon thick vegetation for movement, and we expect that it may have more limited dispersal opportunities under scenarios of increasingly sparse habitat. We considered the dispersal ability of rock squirrels to be small because, though this species might be able to move significant distances, they will only move to sites that have appropriate rocky habitat (Oaks and others 1987). Presumably, rock squirrels already inhabit suitable habitat within the MRGB and surrounding area and movement to other suitable areas would represent movements to other canyons or mountain ranges, which we consider unlikely. Interestingly, we expect the tawny bellied cotton rat, which can disperse several kilometers (NatureServe 2008), able to disperse adequately to shift its range with shifting habitat and climate conditions, but we found the hispid cotton rat to be dispersal limited. Unlike their congener, hispid cotton rats are less mobile and are not considered good dispersers due to both physiological and behavioral mechanisms (NatureServe 2008; Sullivan 1995).

In contrast, we found beavers and porcupines resilient to shifts in habitat and climate conditions because they have good dispersal ability. Beavers, though limited to the riparian corridor, may be less inhibited by barriers such as dams, which will allow them to move up into tributaries (NatureServe 2008; Payne 1989). Porcupines also have good dispersal ability with an average annual home range of 70 ha (BISON-M 2009). Porcupines exhibit site fidelity with respect to dens (Woods 1973; http://sibr.com/mammals/M145.html), but we did not consider this a strong enough behavior to discount their potential to move with shifting habitats.

Stopover Sites: We did not assess any migrating rodent species and so did not find any rodents vulnerable to additional climate changes at stopover sites.

Physiology Scores

Physiological Limits: We did not find evidence that higher temperatures would negatively affect many nocturnal fossorial species, including *Peromyscus* species, western harvest mice, pocket mice, and gophers (Garrison and Best 1990; Jones and Baxter 2004; Lackey and others 1985; Webster and Jones 1982;). Habitat and physiological adaptations are likely to protect these species from extreme temperatures. For instance, the white-footed mouse and pinyon mouse conserve water by urine concentration (Hoffmeister 1981; Lackey and others 1985; Webster and Jones 1982) and western harvest mice have an exceptionally high evaporative water loss (Webster and Jones 1982), which is probably quite effective for dealing with high temperatures provided an adequate source of water is available. Similarly, we predict beaver lodges will provide a buffer from temperature extremes. We also did not consider *Sigmodon* spp., porcupines, and the New Mexico meadow jumping mouse vulnerable to rising temperatures. *Sigmodon* are more limited by cold than high temperatures (Sullivan 1995) and there is some potential that this species may be able to expand its range in northern parts of is range. The meadow jumping mouse's reliance on riparian habitats was assumed to directly relate to intolerance of heat and desiccation. However, since exposure to lethal temperatures or dry conditions is determined by habitat availability, it is unlikely this species will experience more mortality under future climate regimes outside of that already considered in the Habitat section.

In contrast, activity limitations suggest a temperature threshold for rock squirrels, though individuals can survive with no physiological consequences for up to 100 days without water (Oaks and others 1987). However, because this rodent is able to mitigate high temperatures through behavioral adaptations and is adapted to arid conditions, we did not consider the rock squirrel resilient nor at increased risk due to future climate conditions.

Neotoma albigula has features typical of a species sensitive to high temperatures: it is nocturnal and fossorial and selects succulent foods (Macèdo and Mares 1988). However, unlike other fossorial species, which were not considered vulnerable to direct impacts from high temperatures, we found a number of indications that led us to consider *Neotoma* spp., particularly the white-throated rat, vulnerable to temperature increases. The white-throated woodrat is able to moderate temperatures by reducing pelage, having efficient vascular systems, and staying within burrow (Macèdo and Mares 1988). However, *Neotoma* spp. may already exist close to their upper threshold (lethal limit) of temperature in desert areas (Macèdo and Mares 1988). In addition, though the woodrat species is predominantly southern ranging, it inhabits montane and other "cool" environments, which may indicate some sensitivity to high temperatures. BISON-M (2009) reports that, when found in warmer sites, woodrats are often associated with boulders which are thought to provide the preferred microclimate. There is also evidence for temperature mediated reduction in body size with climate warming (Burgman's rule, see Smith and others 1998). Finally, juveniles are known to be susceptible to mortalities related to den sites with inappropriate microclimates (BISON-M 2009).

Extreme Conditions: We did not find a demonstrated link between extreme events and mortality for most rodent species. Exceptions include muskrats, *Sigmodon* spp., beavers, and meadow jumping mouse, which we found at risk of increased mortality due to extreme conditions. Muskrats have experienced mortality due to extreme conditions, including droughts (Errington 1939), and climate effects relating to the North Atlantic Oscillation may drive population cycles in eastern

Canada (Haydon and others 2001). *Sigmodon* spp. suffer severe declines in response to consecutive drought years, largely due to drought effects on food resources (Sullivan 1995; Texas Mammals).

Population fluctuations of the silky pocket mouse are tied to precipitation events that influence vegetation. However, these fluctuations include both strong positive responses to beneficial conditions as well and negative responses to less than ideal conditions and, given a lack of clear predictions regarding precipitation trends, we could not predict the specific impact of future climate extremes for this species. Floods may be an issue for beavers and New Mexico meadow jumping mice. In general, areas with greatly fluctuating flow or water levels are considered poor habitat for beavers (NatureServe). Two indications exist for increased mortality for meadow jumping mice: (1) increased winter flooding could kill hibernating mouse populations (Frey 2005); (2) drought can result in a greater fragmentation of suitable habitat as certain riparian zones become too dry for the mouse. The capacity of the meadow jumping mouse to disperse is limited to riparian corridors and, thus, drought is predicted to isolate populations, leaving them prone to stochastically driven extirpation (Frey 2005). Significant mortality and population declines have been observed in *Dipodomy*s spp. (including *D. ordii*) following a flood in southern Arizona (Portal), whereas *Cheatodipus* spp. responded positively (Thibault and Brown 2008). However, it was not clear how populations in the MRGB would be affected given that their distributions are often associated with soil characteristics rather than proximity to the river (i.e., many animals may reside outside the flood zone).

Reductions to Activity Periods: Walsberg (2000) stated that desert small mammal species, including nocturnal species, are likely to experience reductions in activity times in particularly hot environments. However, we did not consider most rodents at risk of climate-related reductions in activity periods, primarily due to a lack of information and because it was assumed nocturnal would be less affected by temperature increases and that the MRG might provide a more favorable habitat for species under warmer climates. For instance, Dipodomys merriami have demonstrated activity limits in the central Sonoran Desert. However, we felt that the MRGB may be more protected than habitats within the Sonoran Desert and D. ordii populations in the MRG were considered not at risk due to high temperatures. Additional studies that support Walsberg (2000) will require a reassessment of this impact for our assessed species. The only species we considered clearly at risk of activity reductions was the rock squirrel (Oaks and others 1987). Rock squirrels are diurnal and activity is usually restricted to temperatures between 15 ˚C and 27-30 ˚C (Oaks and others 1987).

Metabolic Advantage: Rodents showed a wide range of behavioral and physiological mechanisms for dealing with resource shortages. Many rodents are able to undergo torpor and many species are able to store food and were considered resilient to short term resource shortages.

For *D. ordii*, only populations in Canada are thought to be able to torpor (Bender and others 2007). This would indicate a mechanism caused by cold weather and, as temperatures are not expected to decrease, it probably will not be available for *D. ordii* in the MRGB. However, *Dipodomys* spp. are thought to cache food, a potential source of resiliency for this species. Beavers were among the most resilient with respect to metabolic advantages. They can torpor during cold periods and are known to cache food. They also store a substantial amount of fat in their tails. Mexican woodrats are not known to torpor but do exhibit caching habits (Cornely and Baker 1986). Silky pocket mice also torpor, but do not hibernate and were not thought to cache more than 5 days' worth of food. White-footed mice experience daily torpor in response to the combined stimuli of cold weather and reduced food sources (Lackey and others 1985). However, not all individuals undergo voluntary torpor and there may be a genetic component as to whether an animal does or does not torpor (Lackey and others 1985). Pinyon mice also use torpor to deal with water stress, but this may not be a good long-term strategy as it results in less concentrated urine and wetter feces, which may increase issues related to water limitations (Hoffmeister 1981). Hibernation occurs in some populations (Nevada) of western harvest mice and torpor is used in response to osmotic stress or water deprivation (Webster and Jones 1982). These mice are not able to store food or undergo seasonal torpor, however. Though apparently unable to torpor, pocket gophers do create rooms for food storage (Jones and Baxter 2004). These effects canceled out under physiology. Muskrats do not hibernate and there was no mention of torpor. However, it does appear that muskrats can store food for winter, which may lend to them greater adaptability under warming conditions (Errington 1939).

We expect neither positive (resilience) nor negative impacts relating to species' metabolic requirements for the *Sigmodon* spp., rock squirrels, and New Mexico meadow jumping mice in response to resource variation. *Sigmodon* spp. do not exhibit food storage behaviors and do not have the capacity to reduce metabolic rates (NatureServe 2008; Texas Mammals). Rock squirrels are considered facultative hibernators (Oaks and others 1987). For the purpose of this assessment, facultative (versus obligatory) hibernation was considered a potential resilience factor because it allows an animal to save energy when conditions are conducive to hibernation. However, facultative hibernation can only be considered to confer resilience under warming climates when species' metabolic savings are not related to environmental temperatures. For most species, metabolic savings relates directly to the temperature at which hibernation occurs and a lack of cool winters may limit the realized benefit of this adaptation for the squirrel and other species (Walsberg 2000). Though some squirrels have been observed to use food caches, food caching is not established within rock squirrel populations to the extent that it can be generally considered a definite capability (Oaks and others 1987). Meadow jumping mice have been described as profound hibernators and remain in hibernation longer than many other mammals (Whitaker 1972). However, underweight or small animals often do not survive hibernation (Whitaker 1972), and, as with the rock squirrel, we considered increases in winter temperatures a significant factor that will reduce potential metabolic savings generated through this behavior.

Ability to Outlive Limiting Conditions: We considered most species unlikely to outlive and, therefore, reproduce under extended droughts lasting 5 years. Exceptions include wild raccoons, which have been known to live 12 years, and rock squirrels, which have been reported to live up to 41 months in one mark recapture study (Oaks and others 1987).

Phenology Scores

We found a clear association between reproductive activity and rainfall and/or temperature conditions for the *Dipodomys* spp., the silky pocket mouse, and beavers, and we considered all three species vulnerable to future changes in climate related cues. *Dipodomys ordii* and *D. merriami* initiate reproductive cycles in response to climate (temperature and precipitation and precipitation only, respectively; Garrison and Best 1990; Lima and others 2008; Sullivan 1995). Further, experimental evidence suggests that an intake of 6-methoxybenzoxazolinone, a compound found in young tissue of some grasses, could induce or increase reproductive output female Ord's kangaroo rats (Rowsemitt and O'Connor 1989). For the silky pocket mice, reproductive activity is tied to food production, which is tied to precipitation (BISON-M 2009). A study of *Perognathu parvus* in Washington showed a link between *Perognathus* populations and winter precipitation (Dunigan and others 1980). Timing and duration of breeding in beavers may vary according to latitude and habitat, which could indicate an effect of local weather (Jenkins and Busher 1979).

We also considered the New Mexico meadow jumping mouse, raccoon, and rock squirrel, which use both endogenous and exogenous cues, vulnerable to changes in weather cues. The entry of meadow jumping mice into hibernation is cued by day length but emergence is related to soil temperature (Muchlinski 1988). Onset of raccoon breeding period may be triggered by increased photoperiod or increased temperature (see Bissonetete and Csech 1938, 1939 or Sanderson and Nalbandov 1973). The rock squirrel's reproduction is known to cue to melatonin (day length), though cold will induce torpor state (Oaks and others 1987). Reproduction appears to be endogenously mediated or related to day length for most other rodent species and they were not considered vulnerable to the effect.

We did not consider many rodents vulnerable to changes in resource pulses because it is not known how resource availability and breeding season or birth are related for most rodents. For small mammals in arid environments, there is probably a substantial effect of precipitation driven changes in primary productivity. However, it is not clear how resource pulses might change given the diverse diet of many species nor is there information regarding how this might affect fitness. For example, the emergence of jumping mice from hibernation and subsequent reproductive activities are tied to general requirement for forage and, while not a discreet event, indicate some reliance on resource events. However, we could not quantify the nature of this relationship in such a way to predict a clear change in populations as a result of changes in the timing of forage material. As a result, we gave most species a neutral score for the second phenology question relating to reliance on resource pulses. We also gave *Sigmodon* and *Reithrodontomys* neutral scores though they are known to cue directly to vegetation (Webster and Jones 1982) because their populations typically fluctuate widely in response to changing conditions and commonly experience extirpations during shortages. We were not able to definitively predict a change in such events (i.e. increase) due specifically to climate change. One exception is the white footed mouse, whose reproductive cycle is clearly tied to resource pulses and who was given a score indicating increased vulnerability under future conditions.

We generally considered mice and rats resilient to issues of distantly placed cues and resources because, though the relationship between these events was largely unknown (e.g., Mexican woodrat), rodents do not migrate and have short gestation periods. Exceptions include the raccoon, which we considered sensitive to potential changes in cues, and, since neither cue (temperature or day length) correspond directly with conditions several months in the future when young are born, it was also considered sensitive to potential changes in resources because their breeding season is limited and shows a definitive springtime peak.

We considered most rodent species including Ord's kangaroo rat (BISON-M 2009; Garrison and Best 1990), *Peromyscus* spp., *Sigmodon* spp., muskrats (BISON-M 2009; NatureServe 2008; Willner and others 1980), meadow jumping mouse

(Whitaker 1972), Mexican woodrats (Cornely and Baker 1986), and rock squirrels resilient to timing issues during reproduction because they are able to produce multiple litters. We considered two species, the pocket mouse and the beaver, vulnerable to timing mismatches during breeding because they have just one litter per year (BISON-M 2009).

Pocket gophers, which persist almost exclusively in underground burrows, presented a unique challenge for assessment of phenology. There is mixed evidence of whether pocket gophers cue to precipitation changes or presence of particular plants, though ultimately this effect does not appear to be overtly important to the timing of species reproduction (Bandoli 1981). Pocket gophers are able to respond quickly to changes in the immediate environment and produce multiple litters; however, there are clear indications that burrowing activity changes with changes in environmental conditions, though the mechanism is not clear (Bandoli 1981; Jones and Baxter 2004). Thus, we considered a change in timing of food resources a potential issue for this species, though in general this species does not possess characteristics associated with critical phenological sensitivities.

Biotic Interaction Scores

Many rodent species had generalized relationships with other community members, and we found very few indications that would lead us to consider species vulnerable due to future change in biotic interactions.

Food Resources: Most rodents have a diverse diet that can be characterized as opportunistic. We considered only a few species at risk of food declines and found no evidence for food increases for any species. For some species, declines in food resources were considered indirectly in habitat quality and exposure to extreme events and were not counted within this section. We did not consider granivores, such as Ord's kangaroo rat and silky pocket mice (Best and Skupski 1994) vulnerable to climate change, largely due to the lack of information and knowledge about plant productivity under warmer scenarios. We did consider the beaver, which prefers newly emergent woody species and is prone to out-eating its surroundings (Jenkins and Busher 1979), at risk of food shortage due to climate changes. Though porcupines also eat trees, we felt this species had greater capacity to survive on alternative food resources because it has a more varied diet than beavers (BISON-M 2009; Woods 1973). The most important food source for white-throated woodrats is cactus species (BISON-M 2009), but it is unknown how limited this species is with respect to its diet and we were not able to determine whether food may increase or decrease for the rat under future conditions.

Predators: We did not quantify predator effects for most rodent species because population dynamics are more commonly driven by resource changes than predator influences. Humans and their related activities were commonly the most influential sources of mortality, which we did not consider an influence of climate change. Two exceptions are white-throated woodrats (BISON-M 2009; Macèdo and Mares 1988) and *Peromyscus* spp., which are preyed upon by coyotes. We expect coyote predation may increase in the future, resulting in negative impacts for these species.

Symbionts: Symbiotic relationships, where they exist, are not expected to be influenced by climate change. Western harvest mice benefit from the presence of meadow voles, which create runways in vegetation, but this relationship was considered not critical (BISON-M 2009). The New Mexico meadow jumping mouse is positively associated with the presence of beavers and is harmed from beaver removal (Frey 2006b). Beavers are predicted to be slightly vulnerable to climate changes, particularly pertaining to water flows.

Disease: Interestingly, data on *Sigmodon hispidus* cite disease as the primary cause of population crashes, which essentially offsets the high reproductive rate of these species. (NOTE: This assertion from The mammals of Texas-online, could not be verified in literature sources.)

Competition: We found better data for competitive interactions between rodent species as compared to other biotic interactions. We found several indications that competitive influences are important to *Neotoma* spp. and such interaction might become more important in the future. Although their ranges overlap, the Mexican woodrat appears to replace the white-throated woodrat where they co-occur. Interestingly, the white-throated woodrat may be replacing *N. micropus* (not assessed in this study) in southwestern plains and the southern Rio Grande Valley as these regions continue to desertify (BISON-M 2009). There appears to be some competitive exclusion between the cotton rat species (hispid cotton rat and tawny-bellied cotton rat) assessed in this study (Petersen 1979). A hiatus between the northern and southern segments of the range of tawny-bellied cotton rat may be caused by the recent arrival of the hispid cotton rat into much of the Rio Grande Valley (BISON-M 2009; Mohlhenrich 1961). However, exclusion studies have found that the hispid cotton rat is displaced and suffers reduced productivity in the presence of tawny-bellied cotton rat in a mesquite habitat. The effect of climate change on the interaction of these species is dependent upon the type of change in habitats along the MRG. Western harvest mice may compete with house mice in some areas (BISON-M 2009), but we did not consider this a significant influence in the current assessment. Botta's gophers appear to be displaced from favorable habitat when *Pappogeomys* and *Geomys* spp. are present (Jones and Baxter 2004), and, in Colorado, gopher burrows were found at a significantly different depth than burrows of *Pappogeomys catanop*s where they were sympatric. We were unable, however, to make a prediction of potential effects of pocket gopher competitors in the MRG because these species are not known from the area.

Lagomorphs

We assessed two lagomorphs within the MRG: black-tailed jackrabbits (*Lepus californicus*) and desert cottontails (*Sylvilagus audubonii*).

Habitat Scores

We expect the black-tailed jackrabbit will experience an increase in suitable habitat. It is widespread, occurring from the upper low tropics through the lower and upper Sonoran Desert and into the transition zone (Best 1996), and prefers heavily grazed areas or areas with sparse grass (Best 1996; Flinders and Chapman 2003). We considered habitat change neutral for desert cottontails, which typically inhabit arid habitats mostly in the Lower Sonoran Life-Zone. Desert cottontails like to hide in shrubby habitat or under manmade and natural structures and cover type appears to be an important factor in determining feeding sites (Chapman and Willner 1978) with a preference for brushy cover.

We considered several potential measures of habitat quality for the black-tailed jackrabbit including: forage, cover, water, predators, competitors, and climatic variables (Flinders and Chapman 2003). We found several indications that habitat quality features will improve for the jackrabbit. Declining range condition, sparser vegetation, overgrazing, and drought have been associated with an increase of favorable habitat (open weedy areas) (Flinders and Chapman 2003). We did not consider the jackrabbit dependent upon specific habitat components. Conversely, desert cottontails use dense cover stands for nesting (Chapman and Willner 1978), and cover composition and relative densities of live trees, shrubs and brush piles are the most important factors influencing habitat use by rabbits (Rintamaki 1986). We considered the potential impact of climate on vegetation change for the cottontail in terms of its impact for important habitat components and habitat quality. Specifically, we expect decreases in vegetation due to increasing aridity to result in a reduction of important thermal and anti-predator features (shrubs, cover) of the habitat, which will result in greater vulnerability. Cover type also appears to be an important factor in determining feeding sites for desert cottontail (Chapman and Willner 1978) with a preference for brushy cover, so we also considered the loss of cover type an issue for survival of individuals.

Both black-tailed jack rabbits and desert cottontails have good dispersal ability (BISON-M 2009; Chapman and Willner 1978), and we did not consider them vulnerable to dispersal limitation.

Physiology Scores

We did not consider either lagomorph at risk of experiencing increased limitation by future expected weather conditions. The black-tailed jack rabbit has several mechanisms for dealing with extreme temperature: it dissipates heat by seeking shade or retreating to burrows; its pelage has insulative and reflective properties, which help reduce heat load; it can concentrate urine to reduce water use; and it avoids extreme water requirements by allowing body temperature to rise to 41 °C (Best 1996; Flinders and Chapman 2003). Still, jackrabbits must consume 120 ml/kg of water through plant resources to meet daily requirement (Best 1996), and the size of jackrabbits litters correlated with precipitation (Best 1996) an indirect influence of precipitation on forage species. Though the cottontail has a high evaporative cooling capacity and high thermal maxima (44.8 ˚C) (Chapman and Willner 1978), free water is critical for survival in dry years if high moisture content plants are not available. In addition, thermal maxima were estimated from an experiment on sedentary animals and may not reflect true limitations under conditions that require movement for forage or predator avoidance. For instance, the desert cottontail is inactive at temperatures greater than 80 ˚F (Chapman and Willner 1978). Though the jack rabbit and desert cottontail are well adapted to the arid conditions in which they exist, we noted during our assessment that these species may exist at their threshold of tolerance, in which case their adaptations are unlikely to help them under increasingly hot and arid conditions.

We did not find strong evidence for extreme weather impacts for either lagomorph species. Black-tailed jackrabbit populations in California experience high mortality due to fires (Best 1996), and fires can lead to higher predation for both species by reducing cover. However, patchy fires in sagebrush and other shrub types may also benefit lagomorphs by improving food value (BISON-M 2009). Therefore, increased fire frequency could result in both positive and negative effects.

We considered the desert cottontail but not the black-tailed jackrabbit to be vulnerable to restrictions in activity periods due to increasing temperatures and storms. Wind has the greatest impact on activity levels for both species (Best 1996; Chapman and Willner 1978). But, while *Lepus californicus* are nocturnal with most activity concentrated at dawn and dusk (Best 1996; BISON-M 2009), desert cottontail activity is concentrated around early morning and evening hours (Chapman and Willner 1978). Additionally, cottontail activity is related to other factors, including habitat (cover available) and season of the year (Chapman and Willner 1978). Given the tendency for this species to restrict its activities to the cool part of the day and become inactive to higher temperatures (>80 °F), it seems quite likely that it may experience a reduction in potential foraging time.

Though they experience high mortality rates, jackrabbits are able to live up to 7 years and are drought tolerant, and we did not consider them especially vulnerable to populations declines resulting from extended limiting periods (e.g., drought) (BISON-M 2009). Conversely, desert cottontails live 1-2 years (Chapman and Willner 1978), and we considered them susceptible to population declines related to prolonged conditions limiting or reducing reproduction in individuals.

Phenology Scores

We did not find strong evidence that jackrabbits rely on timed resources, and, given their somewhat extended and flexible breeding season, we found jackrabbits resilient over all to phenological changes. Litter size varies according to habitat and time of year but 2-5 young are commonly produced (Best 1996) in about 4 litters/year. Reabsorption and pseudo-pregnancies are common in jackrabbits (Flinders and Chapman 2003), indicating that this species is able to respond to its immediate environment. However, as with many other mammals, this response is angled toward reducing output under poor conditions rather than improving output under good conditions (e.g., it does not increase survival or population rates but reduces cost to parent). Desert cottontails received a slightly resilient score overall, reflecting their somewhat more limited breeding seasons, which indicates some reliance on time resources. Still, the desert cottontail exhibits a capacity to respond directly to it surrounding environment and is able to produce multiple litters throughout the year. We expect it will not be negatively affected by changes in the timing of resource pulses (Chapman and Willner 1978).

Biotic Interaction Scores

We predicted no significant change in food resources that would increase or decrease lagomorph survival. In general, black-tailed jackrabbits are considered strict herbivores (with occasional forays into fungi, sand, and gravel) and appear to prefer succulents, perhaps due to their metabolic water requirements (Best 1996). For instance, cacti are often preferred food sources as droughts conditions increase (Best 1996). However, in two instances, black-tailed jackrabbits were reported to feed on a horse carcass. *Sylvilagus audubonii* are strict herbivores that forage on a variety of plants. Rintamaki (1986) noted that species is not as important as moisture content and growth stage with cottontails preferring new growth (shrubs in particular) when available. This could prove to be a point of vulnerability under changing climates that include drier seasons. However, we do not have an accurate gauge to what degree changes in vegetation composition would occur in the MRG.

Predation, hunting, roadkill, fire, drought, hail, and extremely cold weather have all been associated with jackrabbit mortality (BISON-M 2009). For this assessment, we considered predation a source of increasing vulnerability under climate change due to the strong influence of coyotes on jackrabbit populations (Best 1996; Flinders and Chapman 2003). Coyotes are not expected to be negatively impacted by climate change and many predicted habitat changes are associated with increased predator success.

Jackrabbits are infected by a number of pathogens, including *Borreia burgadorferi, Toxoplasma gondii, Coxiella burnettis, Pasteurella tularensis*, *Yersinia pestis, Y. pseudotuberculosis*, and a variety of endo and ecto parasites (Best 1996; Flinders and Chapman 2003). Coccidiodosis was thought to be behind 20% of the winter deaths in California. Tularemia is also a problem (BISON-M 2009). Though these diseases might have significant effects on mammal populations, there is no evidence for or against their increase under climate change scenarios, and we did not consider disease a source of increase vulnerability in the future.

We also did not consider competitive interactions important under warming conditions. The desert cottontail may be negatively impacted by cattle. However, the impact of cattle has not been quantified and it is not possible to identify a climate related change at this time. It is likely that biotic interactions are not as important as food resources to survival of lagomorph species.

Carnivores

We assessed eight species included in the carnivore category: coyotes (*Canis latrans*), mountain lions (*Puma concolor),* long-tailed weasels (*Mustela frenata*), grey fox (*Urocyon cinereoargenteus*), stripped skunk (*Mephites mephitis*), bobcats (*Felis* [*Lynx*] *rufus*), raccoon (*Procyon lotor*), and the black bear (*Ursus americanus*).

Habitat Scores

Vegetation Association: We did not consider species with generalist habits such as coyotes (generalized need for a water source, Bekoff 1977), bobcats (BISON-M 2009), and the long-tailed weasel (BISON-M 2009) vulnerable to habitat loss.

For other species, habitat may become rarer under climate change, particularly where animals rely on habitat specific to riparian areas. Raccoons, which rely on water sources, are likely to be negatively impacted and were considered vulnerable to shifts of both breeding and non-breeding habitat. Mountain lions are found in a variety of habitats though they tend to be more abundant in forested areas and associated with the more heavily vegetated areas within those with forest habitat (BISON-M 2009). In addition, mountain lions require large patches of contiguous forest or native habitat to maintain viable populations (Maehr and Cox 1995), which we believe will decline under climate change. We also considered skunks vulnerable to habitat loss. Though widespread, skunks are not usually found in the very arid Southwest and are near the southern limit of their range in New Mexico, where they appear to rely on riparian areas (Rosatte and Larivière 2003; Wade-Smith and Verts 1982). Black bears are associated with either riparian or montane habitats in New Mexico (BISON-M 2009). In addition, bears require a diversity of habitats to provide their seasonal food needs (Larivière 2001). The gray fox depends upon riparian habitat, though it, like other carnivores, has generalized habitat needs (Fritzell and Haroldson 1982). However, gray fox are essentially absent from grassland or open habitats that do not have rocky outcrops or some juniper encroachment (BISON-M 2009), and we considered this species vulnerable to habitat loss.

Habitat Components: Most carnivores use dens, which we considered unaffected by climate change. Only one carnivore, the raccoon, was considered dependent on a habitat component that might be affected by future conditions. Specifically, raccoons may be vulnerable to reduced flows and other hydrological changes resulting from climate change. Though not quantified in this assessment, proximity of appropriate den sites to forage sites may become an issue for some species. For instance, mountain lion dens are typically set up in rocky areas or areas with thick vegetation (Pierce and Breich 2003) and thermal characteristics of dens are thought to be of primary importance for protecting kittens from thermal maxima (Bleich and others 1996). Dense vegetation may also be an important determinant of success in hunting for mountain lions (Pierce and Bleich 2003), though it is difficult to quantify its influence. Black bears require secluded areas for denning and hibernation (Larivière 2001). Appropriate hibernation sites are needed for successful reproduction and typically include caves, depressions, snags, or other forest shelter (BISON-M 2009). Denning sites are likely to become rarer, especially those that are associated with current forest habitat, but again, it is not possible to gauge potential effect to bear populations.

Habitat Quality: Carnivore populations are sensitive to changes in prey availability and conditions which change hunting success, which was the most common measure of habitat quality used for carnivore species. We expect black bears and long-tailed weasels to experience declines in habitat quality. Bears can survive in any habitat provided there is some sort of cover (required for cover, breeding, and thermoregulation; Jonkel and Cowa 1971) and sufficient food (BISON-M 2009). We felt that increasing aridity and declines in food resources would increase bear vulnerability during breeding but not non-breeding periods. We considered the presence of water a critical element required for the survival of the long-tailed weasel and one likely to decline in the future, thus contributing to weasel vulnerability. Though coyotes and bobcat reproduction is tied to prey availability, their primary prey—rodents—have a wide variation in expected response to climate change, and we did not consider these species vulnerable to change in habitat quality (Bekoff 1977; Larivière and Walton 1997).

Dispersal: Carnivores have good dispersal ability and we gave all species a resilient score.

Physiology Scores

Physiological Limitations: We do not expect most carnivores, including coyotes and mountain lions, to be limited by physiological conditions. The impact of extreme temperatures and heat waves on carnivores is unknown or minor or exposure is moderated by behavior. However, we considered skunks and weasels vulnerable to increased temperatures. Though widespread, skunks are not usually found in the very arid Southwest and are near the southern limit of their range in New Mexico (Wade-Smith and Verts 1982). This, and the general and opportunistic nature of this species, was considered as evidence for a physiologically mediated limitation on the range of stripped skunk. Their dependence on riparian habitats in the arid Southwest is also an indication that this species is not well adapted to xeric landscapes. Long-tailed weasels become quiescent during hot weather and are less alert (Sheffield and Thomas 1997) and were also considered potentially vulnerable to future warming trends.

Extreme Events: In general, carnivores are very sensitive to changes in prey availability, which was often tied to climate and, in particular, drought. However, prey availability is addressed elsewhere, and we did not consider any of the carnivore species additionally affected by drought or other extreme events.

Activity Periods: Bears were the only carnivore considered sensitive to climate-related reductions in activity periods. Bears are described as primarily crepuscular and diurnal though they can shift to more nocturnal activity patterns during fall months or in the vicinity of human populations (BISON-M 2009). Diurnal activity is thought to be typical of bears because it increases forage efficiency. Activity is reported to decline due to rain, snow, or extreme temperatures (Larivière 2001). Changes to climate will disrupt this tendency and probably reduce overall time used for foraging. During fall seasons, bears become more active (and forage into the night hours) in order to put on fat. We believe higher temperatures will limit the maximum number of hours a bears is able to forage in a day during this period. The remaining carnivore species are largely nocturnal or crepuscular, and we do not expect them to experience changes in their activity budgets as a result of warming.

Metabolic Advantages: We found variable evidence among the carnivores regarding potential metabolic advantages for dealing with resource variations. Many carnivores (e.g., grey fox) do not hibernate or cache food and we considered these species vulnerable to increased resource variations. Mountain lions bury kills and will return over a period of a few days (BISON-M 2009; Pierce and Bleich 2003), but we did not consider this a long enough time span to represent a significant resiliency in the face of increasingly variable food resources and considered this species vulnerable.

We considered skunks, long-tailed weasels, and raccoons to be able to engage in advantageous activities. Skunks may undergo mild hypothermia in cold weather (Wade-Smith and Verts 1982). Others report this species as undergoing a deep sleep but not actually hibernating (Rosatte and Larivière 2003). Still, others consider skunks to undergo a torpor like state (like some bears), which is described as a state where animals do not lower body temperature and can be easily aroused. For the purpose of this assessment and to include the beneficial properties of this type of energy conserving mechanism, we considered skunks as able to torpor (though literature does not support the presence of an accompanying reduction in metabolic function). Long-tailed weasels do not hibernate or torpor (Svendsen 2003) but do put on fat reserves for the purpose of winter denning and are known to cache food (Rosatte and Larivière 2003; Sheffield and Thomas 1997; Svendsen 2003). Similarly, raccoons do not hibernate but rely on fat reserves accumulated during the previous summer and fall (Lotze and Anderson 1979).

We determined an overall neutral response for the black bear (scored as "none;" see Appendix 1) from future changes. Though hibernation is a mechanism for surviving resource shortages, it could prove less adaptive in warmer climates. Hibernation may cease in some populations where temperatures fail to cool enough; hibernation sites may be warmer, causing bears to burn through fat reserves more quickly; and conditions may be too poor for bears to get adequate fat reserves to survive and support offspring during winter months. Bears are likely to emerge from hibernation if temperature increases and some populations in the south are active during winter months (BISON-M 2009; Larivière 2001). Increased temperature could also lead to a reduced number of good denning sites because not all bear dens are protected from changes in ambient temperatures. During hibernation, metabolic rates are reduced by ~50% with a corresponding drop in body temperature of 7-8 °C (BISON-M 2009). Warmer temperatures could lead to less metabolic savings during hibernation and increase bear mortality. Energy during hibernation is derived from body fat (BISON-M 2009). If fat reserves are depleted too quickly, bears may be at risk of succumbing to severe dehydration (Larivière 2001). We feel bears are likely to be at a disadvantage under warmer climates where temperatures continue to fall to levels conducive to hibernation.

Surviving Limiting Conditions: Carnivores are generally long lived (BISON-M 2009; Fritzell and Haroldson 1982; Larivière and Walton 1997; Wade-Smith and Verts 1982) and we considered all long-lived predators, including coyotes, bobcats, skunks, long-tailed weasels, and gray fox, not vulnerable to reproductively limiting conditions that would outlast the lifespan of an animal. An exception is the black bear, which though long lived, is sensitive to the effects of drought on food resources specifically associated with masting events. Since the mechanism is well defined, we were able to identify a climate-related effect of drought on the food resource rather than a general sensitivity to the weather event itself.

Phenology Scores

Bears were among the most sensitive carnivores to potential phenological changes. Though reproductive cycles are more related to photoperiod for males and coitus for females, duration and entry into hibernation is cued to temperature as well as food availability, weather, and snow accumulation (Larivière 2001). We did not find evidence for weather cues for other carnivores.

We considered bears, bobcats, and mountain lions vulnerable to changes timed resources. Birth of bear cubs occurs in winter while mothers hibernate. The abundance of fruit and nuts in fall that allows females to support cubs via fat accumulation will change with changing climate scenarios. We considered prey populations a critical resource for bobcats and mountain lions because both populations exhibit periodic (seasonal and annual) fluctuations. Additionally, bobcats require vitamin A for successful implantation and this is only obtained through diet (Anderson and Lovallo 2003).

There are indications that carnivores might be negatively impacted by temporal separation between reproductions and hibernation and resource availability. For instance, risk for mismatch between day length cue and resource pulses (vegetation reproduction) increases as daily temperature changes, though it was not always clear to what degree this might affect fitness. For black bears, weasels, and coyotes, mating occurs in the winter before spring conditions are known, and survival is closely tied to prey availability (BISON-M 2009). Therefore, we considered these species vulnerable to risks associated with distantly time resources. In contrast, mountain lions have a 96-day gestation and were given a neutral score.

Many carnivores have low reproductive rates, and we considered the bear, gray fox, weasels, and skunks, which produce one litter (or biannually in the case of the bear) per year, vulnerable to timing mismatches during breeding (BISON-M 2009; Rosatte and Larivière 2003; Wade-Smith and Verts 1982;). In contrast, bobcats, which are seasonally polyestrous, and mountain lions, which can breed at any time during the year, were considered resilient to issues of mismatch during critical reproductive periods (Anderson and Lovallo 2003; BISON-M 2009).

Biotic Interaction Scores

Prey: We considered mountain lions and black bears vulnerable to climate-related declines in prey. Mountain lions take a diversity of prey ranging in size from rodents, to collared peccaries, and even domestic livestock species (BISON-M 2009). Mountain lions are probably not limited by a lack of any one prey species.

However, there is some concern that large mammals, including the primary prey of this species, will become less abundant in the MRGB, and we considered this an issue for the mountain lion. For black bears, food availability, especially mast (nuts) events during autumn months, affects age at first reproduction, productivity, and cub survival (Larivière 2001). Litter size is related to mother's condition, which, in turn, relates to food availability. Black bears are omnivores and vegetation is reported to have a strong influence on bear population dynamics (BISON-M 2009). Though generalists, bears are highly dependent upon good crops of berries and nuts (masting species such as acorns), which are likely to decline under future scenarios (Larivière, 2001). The remaining carnivores were considered generalists, and we were unable to accurately predict future food availability.

Predators: Predation rates were largely unknown and we assume they would not have an overt impact on carnivore species. Mountain lions are reported to prey upon bears (BISON-M 2009), though overall, there does not appear to be a strong predator influence on bear populations.

Disease: We were unable to identify whether carnivore diseases will increase or decrease under climate change. We did not find evidence for widespread die-offs attributed to disease. Carnivores tend to have high parasite loads and are exposed to a variety of diseases. Table 3.4 presents the diseases known to occur in carnivores of the MRGB. New and continuing studies of some of these diseases, in particular, plague and Tularemia, may change the considerations made in this assessment. In addition, rabies is an important influence on many species and its presence may be strongly influence by changes in population dynamics. However, there is no clear climate link to rabies outbreaks at this time. Leptospirosis, which is spread in water, may also be of concern under future scenarios. Feline panleukopenia is known to affect mortality and morbidity of both bobcats and mountain lions and is highly infectious, but it is unclear how incidence of this disease might change under warming trends.

Competition: We considered bobcats vulnerable to increasing pressure from competitors. Potential competitors for bobcats include coyotes and red fox (in particular, the former is dominant) (Larivière and Walton 1997). The authors suggested that coyotes are competitively dominant because bobcats are more specialized. Anderson and Lovallo (2003) felt that there is considerable evidence for the competitive exclusion of bobcats by coyotes. Coyotes are not expected to be overly impacted by future climate projections, so we considered bobcats vulnerable to increased competition for dwindling resources. No other carnivores were considered vulnerable. Coyotes, black bears, and other small carnivores may compete with mountain lions for smaller prey items. However, it is not unclear whether any of these species will have an important influence on mountain lion populations under future climate scenarios, nor is there clear evidence that any of these species will increase in the future. Marten and mink are competitors where they coexist (not in New Mexico) (BISON-M 2009). There is some evidence that competition with other fox species (red, swift, or kit) influences gray fox distribution, with gray fox selecting habitat intermediate to that preferred by the other species (BISON-M 2009). Fox populations were inversely related to weasel populations in one study (Fritzell and Haroldson 1982). However, competitive influences do not appear to be strong factors in fox population trends in the MRGB.

Ungulates

Two ungulate species are considered residents in the MRG: elk (*Cervus elaphus*) and mule deer (*Odocoileus hemionus*).

Habitat Scores

Vegetation Association: We considered elk but not mule deer vulnerable to reductions in required breeding habitat. In New Mexico, elk are most common in meadows and mountain coniferous forests (BISON-M 2009). Their presence in the MRGB is somewhat incidental. However, populations that do exist are probably highly dependent upon the riparian habitat, especially during calving season. Within the MRGB, elk are reported to forage in open areas, wetlands, and adjacent alfalfa fields and use shrubby vegetation for cover (Catron and others 2008). Since the MRGB is often far from other suitable habitat, we considered resident elk to be dependent on riparian habitat and therefore vulnerable to its loss. Mule deer favor more open terrain, commonly occupying arid, barren, desert ranges, and have been known to avoid forest habitats in Mexico (BISON-M 2009). Though it may be that the mule deer will find the loss of riparian forest habitat beneficial, we interpreted its broad habitat association as neither benefit nor source of vulnerability under future conditions.

Habitat Components and Quality: We did not consider elk vulnerable to loss of important habitat components or reduced habitat quality, but did expect mule deer at risk of decline due to changes in vegetation structure. Forest edge is critical for calving habitat (BISON-M 2009) and elk need a free source of water. However, these are probably the primary reasons elk associate with the riparian habitat and, therefore, no additional vulnerabilities due to specific features or quality of habitat were considered for this species. McCorquodale (1991) found semi-desert shrub habitat actually resulted in higher calving/survivorship rates than forests, presumably due to the overall greater forage availability (quality forage is clumpy in forests and more homogenous in open areas). However, elk success in such habitats is hypothesized to require greater access to water and may be offset by greater predation and exposure risk. We considered vegetation cover a habitat component for mule deer because it is well established that this species requires adequate cover to avoid and escape from predators (Haussamen 1995). In New Mexico Department of Game and Fish (1988) studies of mule deer, fawn survival varied from 8% to 59% and averaged 31%. Because fawn survival rates influenced long-term population trends much more than did adult survival rates, their reliance on vegetation structure was considered an important indicator of future susceptibility to climate effects. Weather patterns and summer cattle grazing intensity were closely associated with fawn survival (Haussamen 1995). Fawns survived better with more precipitation and less grazing. This association was directly attributed to the cover and protection provided by vegetation. Further, population growth and individual reproduction is thought to be highly dependent on the availability of good forage (Anderson and Willmo 1984). In addition, mule deer abundance and distribution is regulated by habitat conditions, particularly in climatic transition zones (e.g., much of New Mexico), which we interpreted as habitat quality effects (BISON-M 2009). Free water may also be an important habitat attribute, though short supply rarely appears to limit distribution (Mackie and others 2003).

Dispersal: We considered both species resilient to shifts in habitat.

Migration: We considered both species to be unaffected (neutral score) by climate-related changes in other habitats. Mule deer engage in seasonal migration movement to higher altitudes during the hot weather, returning to the foothills and valleys in the winter (Anderson and Willmo 1984; BISON-M 2009). However, this migration does not require a dependence upon new habitat because these animals are traveling across a transition zone.

Physiology Scores

Physiological Limitations: Elk and deer depend upon free water sources during conditions of high heat. However, if water is available, both species are able to survive extreme decisions. Therefore, we did not consider either species vulnerable to extreme temperatures. Shifts to nocturnal feeding regimes may indicate elk have a sensitivity to high temperatures that obligates behavioral thermoregulation (Merrill 1991). However, Johnston and Smitchz (1997) estimated elk climate-space from observations that show a high tolerance for solar radiation and temperature extremes. Elk also have a relatively good capacity to unload thermal energy through sweating though this method likely requires a greater water intake (McCorquodale 1991; Merrill 1991). The upper critical thermal range of mule deer is thought to be >25 ˚C, though they are known to survive in areas where extreme temperatures may approach 50 °C (Mackie and others 2003). Above this temperature, animals use behavioral thermoregulation technique to avoid overheating (Peek and others 2002). Deer regulate their temperature through a variety of physiological (vasoconstriction, panting) and behavioral means.

Extreme Events: We considered both species vulnerable to increased mortality due to extreme events. Elk are reported to be regulated by hunting, disease, and snow (BISON-M 2009). Typically, large herbivore populations are limited by the effects of winter weather on adult survival and yearling recruitment (Wang and others 2002). Therefore, warming, especially that occurs in winter months, may lead to increases in abundance. However, recent modeling efforts have found that while spring precipitation increased elk population growth, mean monthly summer temperatures were negatively associated with elk survival (Wang and others 2002). Ultimately, though mild winters favor fawn survival, we do not expect warmer, drier conditions to will not favor elk overall. In addition, there is an increased likelihood of extreme events (ice storms, etc.) that could negatively impact faun survival. Adult mule deer have a good survival rate and most doe deaths were attributed to natural mortality influenced by weather (Smith 1991). However, drought has had clear and negative consequences on mule deer reproduction (Leopold and Krausman 1991) and this was a factor for mule deer under our future scenario. The relationships of fire are less clear: elk and deer are drawn to post-fire sites, but deer are known to be killed by fast moving fires (BISON-M 2009; Prescribed Fire and Fire Effects Research Work Unit 1996).

Activity Patterns: We did not consider elk or deer vulnerable to declines in activity periods under future warming trends. Elk appear to be able to adjust foraging times and exhibited nocturnal activity patterns, which may help them avoid hot periods (McCorquodale and others 1986; Merrill 1991). Deer tend to be most active around sunrise or sunset. Mackie and others (2003) noted that daily and seasonal activity patterns are likely strongly influenced by air temperature regimes, but this refers primarily to effects of cold temperature and snow.

Hibernation/Fat Storage/Metabolic Adaptations: Neither elk nor mule deer are able to store fat or water reserves nor do they exhibit special adaptations for dealing with resource variability. We considered both species vulnerable to potential increases in resource variability and neutral (neither likely to benefit or be harmed) by short-term variations. In addition, we gave both neutral scores for metabolic adaptability because they are endotherms with moderate metabolisms.

Phenology Scores

We did not consider either elk or mule deer vulnerable to changes in cues, resources, or distantly timed/located events. Elk migrate in response to climate (typically snow) variables and time birth to coincide with plant productivity. However, they also appear to have flexibility regarding migration timing and were considered quick responders to habitat change. In particular, elk will follow migration paths determined by the successive budding of forage plants. Therefore, though a long-distant migrant, elk were not perceived to be prone to issues of mismatch due to distantly located cues and resource pulses. We considered both elk and mule deer vulnerable to timing mismatch during their annual breeding cycle deer because they have one birth per year.

Biotic Interaction Scores

There was little evidence of significant biotic interactions for either species. Most issues relate to human domestics and activities.

Food Resources: We did not identify a specific vulnerability for elk or deer. It was not possible to quantify a general loss or increase in forage availability for elk under warming trends because they have a diverse diet. Elk diet changes with phenotypic shifts in plants. Mule deer can out eat their environment (BISON-M 2009) and have been observed to be regulated by spring precipitation events, presumably due to the direct influence of precipitation on forage availability (Leopold and Krausman 1991). However, the mule deer diet was too diverse to delineate a particular trend for this report.

Predators: We did not consider predator influences a significant impact for elk. Conversely, mule deer have a number of predators and, when deer are present below ecosystem carrying capacity, predation can be a significant limiting factor for this species (Ballard and others 2001). In New Mexico, mountain lion predation was considered an additive effect to other sources of mortality, whereas in California and Nevada mountain lion predation was considered compensatory (did not increase overall mortality rates) (Ballard and others 2001). Mountain lions were considered an important limiting factor on mule deer populations in the Kaibab plateau of Arizona (Pierce and Bleich 2003). However, we did not consider mule deer vulnerable to increased predation under climate change because mountain lions were not considered resilient (and thus likely to increase) to climate change in their assessment (see Carnivore section). We did not have enough information to account for potential changes in predation efficiency that might result from changes to habitat cover.

Disease: Elk are reported as regulated by hunting, disease, and snow (BISON-M 2009). Among the diseases that affect elk are chronic wasting disease, *Brucella* spp., lungworms, and *Elaeophorosis* nematode (BISON-M 2009). Though disease has the potential to reduce population, there are no clear mechanisms that predict how climate change will affect exposure and spread of disease in the MRGB. Parasites and diseases contribute largely to mortality rates in mule deer and can make them more susceptible to predators or adverse climatic conditions

(BISON-M 2009). Prion disease, which may be an issue for populations in New Mexico, appears to be influenced more by high population densities than climate (various publications, see Farnsworth and others [2005] for discussion of anthropogenic habitat effects).

Literature Cited

Adams, R. A. and M. A. Hayes. 2008. Water availability and successful lactation by bats as related to climate change in arid regions of western North America. Journal of Animal Ecology 77: 1115-1121.

Allan, J. F. and A. S. Flecker. 1993. Biodiversity conservation in running waters. BioScience 43: 32-43.

Allen, D. 2000. "*Notiosorex crawfordi*" (On-line), Animal Diversity Web. Accessed December 08, 2008 at http://animaldiversity.ummz.umich.edu/site/accounts/information/Notiosorex_crawfordi.html.

Allen, P. E. 1950. Road-runner in eastern Oklahoma. Condor 52: 43.

Allison, L. J., P. E. Brunkow, and J. P. Collins. 1994. Opportunistic breeding after summer rains by Arizona tiger salamanders. Great Basin Naturalist 54: 376-379.

Amphibiaweb: Information on amphibian biology and conservation. 2009. Berkeley, California. Available: http://www.amphibiaweb.org.

Anderson, A. E. and O. C. Wallmo. 1984. *Odocoileus hemionus*. Mammalian Species 219: 1-9.

Anderson, A. M., D. A. Haukos, and J. T. Anderson. 1999. Habitat use by anurans emerging and breeding in playa wetlands. Wildlife Society Bulletin 27: 759-769.

Anderson, B. W. and R. D. Ohmart. 1978. Phainopepla utilization of honey mesquite forests in the Colorado River Valley. Condor 80:334-338.

Anderson, E. M. and M. J. Lovallo. 2003. Bobcat and Lynx. In: Wild Mammals of North America: biology, management, and conservation/ 2nd ed. Feldhamer, G. A., Thompson, B. C., and J. A. Chapman (Eds). The John Hopkins University Press, MD, USA.

Annear, T., Chisholm, I., Beecher, H., [and others]. 2004. Instream flows for riverine resources, revised edition, Cheyenne, WY. 268 p.

Anonymous. 1991. Help for Migratory Bats: BCI and Mexican biologists join in a mutual concern: protection of the bats that cross our borders. BAT 9: 3-4.

Archer, S. 1994. Woody plant encroachment into southwestern grasslands and savannas: rates, patterns and proximate causes. Pages 13-68 in M. Vavra, W. A. Laycock, and R. D. Pieper (eds.), Ecological implications of livestock.

Armour, C., D. Duff, and W. Elmore. 1994. The effects of livestock grazing on western riparian and stream ecosystem. Fisheries 19: 9-12.

Armstrong, D. M. and K. J. Jones, Jr. 1972. *Notiosorex crawfordi*. Mammalian Species 17: 1-5.

Askins, R. 1981. Survival in winter: the importance of roost holes to resident birds. Loon 53: 179-183.

Atchley, M. C., A. G. de Soyza, and W. G. Whitford. 1999. Arroyo water storage and soil nutrients and their effects on gas-exchange of shrub species in the northern Chihuahuan Desert. Journal of Arid Environments 43: 21-33.

Bachmann, K. 1969. Temperature adaptations of amphibian embryos. American Naturalist 103:

Bagne, K. E., M. M. Friggens, and D. M. Finch. 2011. A System for Assessing Vulnerability of Species (SAVS) to Climate Change. Gen. Tech. Rep. RMRS-GTR-257. Fort Collins, CO. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 28 p.

Bahre, C. J. and M. L. Shelton. 1993. Historic vegetation change, mesquite increases, and climate in southeastern Arizona. Journal of Biogeography 20: 489-504.

Bailey, R. G. 1988. Ecogeographic analysis: a guide to the ecological division of land for resource management. Miscellaneous Publication No. 1465. US Forest Service, Washington, D.C. 16 pp.

Bailey, R. G. 1995. Description of the ecoregions of the United States. 2nd edition revised and expanded (1st edition 1980). Miscellaneous Publication No. 1391. US Forest Service, Washington, D.C. 108 pp., with separate map at 1:7,500,000.

Bailey, R. G. 1998. Ecoregions map of North America: explanatory note. Miscellaneous

Baker, R. H. and K. A. Shump, Jr. 1978. *Sigmodon fuviventer*. Mammalian Species 94: 1-4.

Baker, R. J. and D. L. Spencer. 1965. Late Fall reproduction in the desert shrew. Journal of Mammalogy 46: 330.

Baltosser, W. H. 1989. Nectar availability and habitat selection by hummingbirds in Guadalupe Canyon. Wilson Bull. 101:559-578.

Baltosser, W. H. and S. M. Russell. 2000. Black-chinned Hummingbird (*Archilochus alexandri*). In The Birds of North America, No. 495 (A. Poole and F. Gill, Eds.) The Birds of North America, Inc. Philadelphia, PA.

Bandoli, J. H. 1981. Factors influencing seasonal burrowing activity in the pocket gopher, *Thomomys bottae*.

Barbault, R. and M-E. Maury. 1981. Ecological organization of a Chihuahuan desert lizard community. Oecologia 51: 335-342.

Bateman, H., M. Harner, and A. Chung-MacCoubrey. 2008. Abundance and reproduction of toads (*Bufo*) along a regulated river in the southwestern United States: Importance of flooding in riparian ecosystems. Journal of Arid Environments 72: 1613-1619.

Beier, P. 1993. Determining minimum habitat areas and habitat corridors for cougars. Conservation Biology 7: 94-108.

Beissinger, Steven R., J. Michael Reed, Joseph M. Wunderle, Jr., Scott K. Robinson, and Deborah M. Finch. 2000. Report of the AOU Conservation Committee on the Partners In Flight species prioritization plan. Auk 117: 549-561.

Bekoff, M. 1977. *Canis latrans*. Mammalian Species 79: 1-9.

Belthoff, J. R. and A. M. Dufty, Jr. 1998. Corticosterone, body condition and locomotor activity-a model for dispersal in screech-owls. Animal Behavior 55: 405-415.

Bemis, C. and J. D. Rising. 1999. Western Wood-Pewee (*Contopus sordidulus*). In The Birds of North America, No. 451 (A. Poole and F. Gill, eds.), The Birds of North America, Inc., Philadelphia, PA.

Bender, D. J. Bummer, D. L. and R. E. Dzenkiw. 2007. Monitoring protocol for the Ord's Kangaroo Rat. Alberta Species at Risk Report 113.

Bent, A. C. 1948. Life Histories of North American Nuthatches, Wrens, Thrashers and their Allies. U. S. Natural History Bulletin No. 195.

Bergin, T. M. 1993. The influence of habitat structure, reproductive success and nest predation on nest site selection within a canopy nesting guild of avian species. Ph.D. Thesis. Bowling Green State Univ. Bowling Green, OH.

Best, T. L. 1996. *Lepus californicus*. Mammalian Species 530: 1-10.

Best, T. L. and M. P. Skupski. 1994. *Perognathus flavus*. Mammalian Species 471: 1-10.

Bird, D. M., and R. S. Palmer. 1988. American Kestrel. Pp 253-290 in Handbook of North American birds. Vol. 5: diurnal raptors. Pt. 2 (R.S. Palmer, ed.) Yale Univ. Press, New Haven, CT.

BISON-M. 2009. This reference information came from the BISON-M (Biota Information System of NM) database. The information was derived directly from data in this species account. See other references in this account for data verification. Available: http://www.bison-m.org.

Blancher, P. J. and R. J. Robertson. 1985. Predation in relation to spacing of kingbird nests. Auk 102: 654-658.

Blankespoor, G. W., J. Oolman, and C. Uthe. 1982. Eggshell strength and cowbird parasitism of Red-winged Blackbirds. Auk 99: 363-365.

Bleich, V. C., Pierce, B. M., Davis, J. L., and V. L. Davis. 1996. Thermal Characteristics of mountain lion dens. Great Basin Naturalist 56: 276-278.

Boag, P., and P. Grant. 1984. The classical case of character release- Darwin's finches (*Geospiza*) on Isla Daphne Major. Galapagos Biological Journal of the Linnean Society 22: 243-287.

Boal, C. W. and R. W. Mannan. 1999. Comparative breeding ecology of Cooper's Hawks in urban and exurban areas of southeastern Arizona. Journal of Wildlife Management 63: 77-84.

Bock, C. E., J. H. Bock, W. R. Kenny, and V. M. Hawthorne. 1984. Responses of birds, rodents, and vegetation to livestock exclosure in a semidesert grassland site. Journal of Range Management 37: 239-242.

Bock, C. E., V. A. Saab, T. D. Rich, and D. S. Dobkin. 1993. Effects of livestock grazing on neotropical migratory land birds in western North America. In D.M. Finch, P.W. Stangel (eds.), Status and management of neotropical migratory birds. U.S. Forest Service Gen.Tech. Rep. RM-229.

Bombay, H. L., M. L. Morrison, L. S. Hall. 2003. Scale perspectives in habitat selection and animal performance for willow flycatchers (*Empidonax traillii*) in the central Sierra Nevada, California.

Bonter, D. N. and W. M. Hochalka. 2003. Declines of chickadees and corvids: Possible impacts of West Nile Virus. American Birds: 2002-2203: 22-25.

Both, C., and L. Marvelde. 2007. Climate change and timing of avian breeding and migration throughout Europe. Climate Research 35: 93-105.

Bradley, G. A., P. C. Rosen, M. J. Sredl, T. R. Jones, and J. E. Longcore. 2002. Chytridiomycosis in Native Arizona Frogs. Journal of Wildlife Diseases, 38:206–212.

Bragg, A. N. 1964. Further study of predation and cannibalism in spadefoot tadpoles. Herpetologica 20:17-24.

Bragg, A. N. 1940. Observations on the ecology and natural history of Anura. II. Habits, habitat, and breeding of *Bufo woodhousii woodhousii* in Oklahoma. American Midland Naturalist 24: 306-321.

Brennan, T. C. 2008. *Sceloporus cowlesi*. Reptiles of Arizona. http://www.reptilesofaz.org/Lizards-Subpages/h-s-cowlesi.html.

Brenner, F. J. 1966. The influence of drought on reproduction in a breeding population of Red-winged Blackbirds. American Midland Naturalist 76: 201-210.

Brittingham, M. C. and S. A. Temple. 1988. Impacts of supplemental feeding on survival rates of black-capped chickadees. Ecology 69: 581-589.

Brooks, R. J., M. L. Bobyn, D. A. Galbraith, J. A. Layfield, and E. G.Nancekivell. 1991. Maternal and environmental influences on growth and survival of embryonic and hatching snapping turtles (*Chelydra serpentina*). Canadian Journal of Zoology 69: 2667-2676.

Brown, C. R. and M. B. Brown. 1986. Ectoparasitism as a Cost of Coloniality in Cliff Swallows (*Hirundo pyrrhonota*). Ecology 67: 1206-1218.

Brown, C. R. and M. B. Brown. 1996. Coloniality in the Cliff Swallow: the effect of group size on social behavior. Univ. of Chicago Press, Chicago.

Brown, C. R. and M. B. Brown. 1999b. Natural selection on tail and bill morphology in Barn Swallows (*Hirundo rustica*) during severe weather. Ibis 141: 652-659.

Brown, Charles R. and Mary B. Brown. 1995. Cliff Swallow (*Petrochelidon pyrrhonota*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/149.

Brown, C. R. and M. B. Brown. 1999a. Barn Swallow (*Hirundo rustica*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/452.

Brown, D. E., F. Reichenbacher, S. E. Franson. 1998. A classification of North American biotic communities. University of Utah Press, Salt Lake City. 141 pp.

Brown, J. H. and A. Kodric-Brown. 1979. Convergence, competition, and mimicry in a temperate community of hummingbird-pollinated flowers. Ecology 60: 1022-1035.

Brown, W. S. and W. S. Parker. 1982. Niche Dimensions and resource partitioning in a Great Basin desert snake community. In Herpetological communities. U.S. Fish and Wildlife Service. Wildlife Research Report 13: 59-81.

Browning, M. R. 1995. Do downy woodpeckers migrate? Journal of Field Ornithology 66: 12-21.

Brush, T. 1999. Couch's Kingbird (*Tyrannus couchii*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/437.

Brush, T., B. W. Anderson and R. D. Ohmart. 1990. Habitat use by cavity-nesting birds of desert riparian woodland. Pp. 191-198 in Managing wildlife in the Southwest symposium (P. R. Krausman and N. S. Smith, eds.). Arizona Chapter of the Wildlife Society, Phoenix.

Burchsted, A. E. 1987. Downy Woodpecker caches food. Wilson Bulletin 99: 136-137.

Burhans, D. E. and F. R. Thompson III. 1999. Habitat patch size and nesting success of Yellow-breasted Chats. Wilson Bull. 111: 210-215.

Burquez-Montijo, A., M. Miller, and A. Martinez-Yrizar. 2002. Mexican grasslands, thornscrub, and the transformation of the Sonoran desert by invasive exotic buffelgrass. Pages 126-146 In B. Tellman, ed., Arizona-Sonoran Desert Museum Studies in Natural History, Tucson, AZ.

Bury, R. B. and J. A. Whelan. 1984. Ecology and management of the bullfrog. U.S. Fish and Wildlife Service, Resource Publication 155, Washington, D.C.

Butler, C. J. 2003. The disproportionate effect of global warming on the arrival dates of short-distance migratory birds in North America. Ibis 145: 484-495.

Cabe, Paul R. 1993. European Starling (*Sturnus vulgaris*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu.bnaproxy.birds.cornell.edu/bna/species/048.

Cagle, K. D., G. C. Packard, K. Miller, and M. J. Packard. 1993. Effects of the microclimate in natural nests on development of embryonic painted turtles, Chrysemys picta. Functional Ecology 7: 653-660.

Cain, III, J. W., Morrison, M. L., and H. L. Bombay. 2003. Predator activity and nest success of willow flycatchers and yellow warblers. Journal of Wildlife Management 67: 600-610.

Cameron, G. N., and Spencer, S. R. 1981. *Sigmodon hispidus*. Mammalian Species. 158: 1-9.

Campbell, R. W., N. K. Dawe, I. Mctaggert-Cowan, J. M. Cooper, G. W. Kaiser, and M. C. E. McNall. 1997. The birds of British Columbia. Vol. 3. Univ. of British Columbia Press, Vancouver.

Cannings, R. J. and T. Angell. 2001. Western Screech-Owl (*Megascops kennicottii*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/597.

Capinera, J. L., and D. R. Horton. 1989. Geographic Variation in Effects of Weather on Grasshopper Infestation. Environmental Entomology 18:8-14.

Cardiff, S.W., and D. L. Dittmann. 2002. Ash-throated Flycatcher (*Myiarchus cinerascens*). In The Birds of North America, No. 664 (A. Poole and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA.

Carey, C., N. Cohen, and L. Rollins-Smith. 1999. Amphibian declines: an immunological perspective. Developmental & Comparative Immunology 23: 459-472.

Carpenter, R. E. 1969. Structure and function of the kidney and the water balance of desert bats. Physiological Zoology 42: 288-301.

Cartron, J., D. Lightfoot, J. Mygatt, S. Brantley, and T. Lowrey. 2008. A Field Guide to the Plants and Animals of the Middle Rio Grande Bosque. University of New Mexico Press, Albuquerque, NM.

Casey, D. and D. Hein. 1983. Effects of heavy browsing on a bird community in deciduous forest. Journal of Wildlife Management 47: 829-836.

Catron, J. E. Lightfoot, D. C., Mygatt, J. E. Brantley, S. L. and T. K. Lowrey. 2008. A field guide to the plants and animals of the Middle Rio Grande Bosque. University of New Mexico Press, Albuquerque, NM. 375 p.

Cayan, D. R., Dettinger, M. D., Diaz, H. F., and Graham, N. E. 1998. Decadal variability of precipitation over Western North America. Journal of Climate 11: 3148-3166.

Cayan, D. R., Karrerdiener, S. A., Dettinger, M. D., Caprio, J. M., Peterson, D. H. 2001. Changes in the onset of spring in the western United States. Bulletin of the American Meteorological Society 82: 399-415.

Center for Reptile and Amphibian Conservation and Management. Species Account: *Terrapene ornate ornate*. Available: http://herpcenter.ipfw.edu/index.htm?http://herpcenter.ipfw.edu/outreach/accounts/reptiles/turtles/Ornate_Box_Turtle/index.htm&2.

Chaplin, S. B. 1976. The physiology of hypothermia in the Black-capped Chickadee *Parus atricapillus*. Journal of Comparative Physiology B 112:335-344.

Chaplin, S. B. 1982. The energetic significance of huddling behavior in Common Bushtits (*Psaltriparus minimus*). Auk 99: 424–430.

Chapman, B. R. and J. E. George. 1991. The effects of ectoparasites on Cliff Swallow growth and survival. Pages 69-92 in Bird-parasite interactions: ecology, evolution and behaviour. (Loye, J. E. and M. Zuk, Eds.) Oxford Univ. Press, Oxford.

Chapman, J. A. and G. R. Willner. 1978. *Sylvilagus audubonii*. Mammalian Species 106: 1-4.

Chapman, J. A., and R. P. Morgan II. 1974. Onset of the breeding season and size of first litters in two species of cottontails from southwestern TX. The Southwestern Naturalist 19: 277-280.

Christian, K. A. 1982. Changes in the food niche during postmetamorphic ontogeny of the frog *Pseudacris triseriata*. Copeia 1982: 73-80.

Chu, M. and G. Walsberg. 1999. Phainopepla (*Phainopepla nitens*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/415.

Chung-MacCoubrey, A. L. 2005. Use of pinyon-juniper woodlands by bats in New Mexico. Forest Ecology and Management 204: 209-220.

Church, D. R. Bailey, L. L., Wilbur, H. M., Kendall, W. L., and J. E. Hines. 2007. Iteroparity in the variable environment of the salamander *Amystoma tigrinum*. Ecology 88: 891-903.

Clapp, R. B., M. K. Klimkiewicz and A. G. Futcher. 1983. Longevity records of North American birds: Columbidae through Paridae. Journal of Field Ornithology 54: 123-137.

Claussen, D. L. 1977. Thermal acclimation in Amystomatid salamanders. Comparative Biochemistry and Physiology 58A: 333-240.

Collopy, M. W. 1977. Food caching by female American Kestrels in winter. Condor 79:63-68.

Comprehensive Wildlife Conservation Strategy for New Mexico. New Mexico Department of Game and Fish. 2006. New Mexico Department of Game and Fish. Santa Fe, New Mexico. 526 pp + appendices. Available http://fws- mcfwru.nmsu.edu/cwcs/New_Mexico_CWCS.htm

Conine, K. H., B. W. Anderson, R. D. Ohmart, and J. F. Drake. 1979. Responses of riparian species to agricultural habitat conversions. Pp. 248-262 in Strategies for protection and management of floodplain wetlands and other riparian ecosystems (R. R. Johnson and J. F. McCormick, tech. coords.). U.S. Dep. Agric., For. Serv. Gen. Tech. Rep. WO-12.

Cornely, J. E. and R. J. Baker. 1986. *Neotoma mexicana*. Mammalian Species 262: 1-7.

COSEWIC 2006. COSEWIC assessment and status report on the Western Painted Turtle *Chrysemys picta bellii* (Pacific Coast population, Intermountain-Rocky Mountain population and Prairie/Western Boreal-Canadian Shield population) in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa. vii + 40 pp. Available: www.sararegistry.gc.ca/status/status_e.cfm.

Cruden, R. W. and S. M. Hermann-Parker. 1977. Defense of feeding sites by orioles and Hepatic Tanagers in Mexico. Auk 94: 594-596.

Cuesta, L. R. 1974. Comparative breeding ecology of the Western Kingbird (*Tyrannus verticalis*) in three habitat types. Master's Thesis. New Mexico State Univ. Las Cruces.

Currier, M. J. P. 1983. *Felis concolor*. Mammalian Species 200: 1-7.

Curtis, O. E., R. N. Rosenfield and J. Bielefeldt. 2006. Cooper's Hawk (*Accipiter cooperii*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/075.

Davis, S. K. 2005. Nest-site selection patterns and the influence of vegetation on nest survival of mixed-grass prairie passerines. Condor 107(3): 605-616.

Davis, S. K. and W. E. Lanyon. 2008. Western Meadowlark (*Sturnella neglecta*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology: http://bna.birds.cornell.edu/bna/species/104.

Dawson, A., V. M. King, G. E. Bentley, G. F. Ball. 2001. Photoperiodic control of seasonality in birds. Journal of Biological Rhythms 16: 365-380.

Dawson, W. R. 1960. Physiological Responses to Temperature in the Lizard *Eumeces obsoletus*. Physiological Zoology 33: 87-103.

Dayton, G. H., and L. A. Fitzgerald. 2001. Competition, predation, and the distribution of four desert anurans. Oecologia 129: 430-435.

Degenhardt, Painter, and Price. 1996. Amphibians and Reptiles of New Mexico. University of New Mexico Press, Albuquerque, NM. 431 pp.

Delmas, V., Bonnet, X., Girondot, M., Prèvot-Julliard, A-C. 2008. Varying hydric conditions during incubation influence egg water exchange in hatchling phenotype in the red-eared slider turtle. Physiological and biochemical Zoology 81: 345-355.

Dement'Ev, G. P. and N. A. Gladkov. 1968. Birds of the Soviet Union. Vol. 4. (Israel Program for Scientific Translations). Smithsonian Institution and National Science Foundation, Washington, D.C.

DeSante, D. F., D. R. O'Grady, K. M. Burton, P. Velez, D. Froehlich, E. E. Feuss, H. Smith, and E. D. Ruhlen. 1997. The Monitoring Avian Productivity and Survivorship (MAPS) Program sixth and seventh annual report (1995 and 1996) Bird Populations 4(0): 69-122.

Dewey, T. and T. Kuhrt. 2002. "*Trachemys scripta*" (On-line), Animal Diversity Web. Accessed January 19, 2010 at http://animaldiversity.ummz.umich.edu/site/accounts/information/Trachemys_scripta.html.

Dickson, B. and P. Beier. 2002. Home-range and habitat selection by adult cougars in southern California. Journal of Wildlife Management 66: 1235-1245.

Dickson, B. G. and J. S. Jenness. 2005. Influence of vegetation, topography, and roads on cougar movement in Southern California. Journal of Wildlife Management 69: 264-276.

Diller, L. V. and R. L. Wallace. 1996. Comparative ecology of two snake species (*Crotalus viridis* and *Pituophis melanoleucus*) in Southwestern Idaho. Herpetologica 52: 434-360.

Dimmett, M. A. and R. Ruibal. 1980a. Environmental correlates of emergence in spadefoot toads (*Scaphiopus*). Journal of Herpetology 14: 21-29.

Dimmett, M. A. and R. Ruibal. 1980b. Exploitation of food resources by spadefoot toads (*Scaphiopus*). Copeia 1980(4): 854-862.

Dixon, M. D., J. C. Stromberg, J. Price, H. Galbraith, A. Fremier, and E. Larsen. In review. Climate change and riparian vegetation response. Chapter 8, In: Stromberg, J. and B. Tellman (eds.), Ecology and Conservation of Desert Riparian Ecosystems: The San Pedro River Example. University of Arizona Press, Tucson, AZ.

Doherty, P. F. and T. C. Grubb. 2002. Survivorship of permanent-resident birds in a fragmented forested landscape. Ecology 83(3): 844-857.

Doroff, A. M. and L. B. Keith. 1990. Demography and Ecology of an Ornate Box Turtle (*Terrapene ornata*) Population in South-Central Wisconsin. Copeia 2: 38

Dunigan, Jr., P. F. X., Lei, W., and W. H. Rickard. 1980. Pocket mouse population response to winter precipitation and drought. Northwest Science 54: 289-295.

Easterling, D., G. Meehl, C. Parmesan, S. Changnon, T. Karl, and L. Mearns. 2000. Climate extremes: Observations, modeling, and impacts. Science 289: 2068-2074.

Eckerle, K. P. and C. F. Thompson. 2001. Yellow-breasted Chat (*Icteria virens*). In The Birds of North America, No. 575 (A. Poole and F. Gill., Eds.). The Birds of North America, Inc., Philadelphia, PA.

Emanuel, K. 2005. Increasing destructiveness of tropical cyclones over the past 30 years. Nature 436: 686-688.

Emery, A. R., A. H. Berst, and K. Kodaira. 1972. Under-Ice Observations of Wintering Sites of Leopard Frogs. Copeia 1: 123-126.

Enquist, C., and D. Gori. 2008. Implications of recent climate change on conservation priorities in New Mexico. Climate Change Ecology and Adaptation Program, The Nature Conservancy, New Mexico.

Erickson, W., D. Johnson, M. D. Strickland, D. Young, Jr., K. Sernka, and R. Good. 2001. Avian collisions with wind turbines: a summary of existing studies comparisons to other sources of avian collision mortality in the United States. National Wind Coordinating Committee (NWCC), Washington, D.C.

Errington, P. L. 1939. Reaction of Muskrat Populations to Drought. Ecology 20: 168-186.

Evelyn, M. J., Stiles, D. A., and R. A. Young. 2004. Conservation of bats in suburban landscapes: roost selection by Myotis yumanensis in a residential area in California. Biological Conservation 115: 463-473.

Ewald, Paul W. and Sievert Rohwer. 1982. Effects of Supplemental Feeding on Timing of Breeding, Clutch-Size and Polygyny in Red-Winged Blackbirds *Agelaius phoeniceus*. Journal of Animal Ecology 51: 429-450.

Fenton, M. B. and R. M. R. Barclay. 1980. *Myotis lucifugus*. Mammalian Species 142: 1-8.

Field, C. B., L. D. Mortsch, M. Brklacich, D. L. Forbes, P. Kovacs, J. A. Patz, S. W. Running and M. J. Scott. 2007. Chapter 14 : North America. Pages 617-652 in M. Parry, O. Costanza, J. Palutikoff, P. J. van der Linden and C. E. Hanson (eds.), Climate Change 2007: Impacts, Adaptation and Vulnerability, Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, Cambridge University Press, Cambridge, UK.

Findley, J. S. 1987. The Natural History of New Mexican Mammals. University of New Mexico Press, Albuquerque. p 96-97.

Finkler, M. S. 1999. Influence of water availability during incubation on hatchling size, body composition, desiccation tolerance, and terrestrial locomotor performance in the snapping turtle *Chelydra serpentina*. Physiological and Biochemical Zoology 72: 714-722.

Fire Effects Information System [Online] (1996, September). Prescribed Fire and Fire Effects Research Work Unit, Rocky Mountain Research Station (producer). Available: www.fs.fed.us/database/feis/ [1998,March 12].

Fischlin, A., Midgely, G. F., Price, J. T., Leemans, R. and others 2007. Chapter 4: Ecosystems, their properties, goods and services In Climate Change 2007: Impacts, adaptation, vulnerability. Contribution of working group II to the fourth assessment report of the Intergovernmental Panel on Climate Change (IPCC). Cambridge University Press, Cambridge.

Fitch H. S. 1955. Habits and Adaptations of the Great Plains Skink (*Eumeces obsoletus*) Ecological Monographs 25: 59-8.

Fleischer, R. C., M. T. Murphy, and L. E. Hunt. 1985. Clutch size Increase and Intraspecific Brood Parasitism in the Yellow-billed Cuckoo. Wilson Bulletin 97: 127-128.

Flinders, J. T., and J. A. Chapman. 2003. *Lepus californicus*. In Wild Mammals of North America: Biology, management, and Conservation. Feldhamer, G. A., Thompson, B. C. and J. A. Chapman (Eds). Johns Hopkins University Press, Baltimore MD. 1197 p.

Foden, W., G. Mace, J.-C. Vie, A. Angulo, S. Butchart, L. DeVantier, H. Dublin, A. Gutsche, S. Stuart, and E. Turak. 2008. Species susceptibility to climate change impacts. In: J.-C. Vie, C. Hilton-Taylor, and S. Stuart (eds.). The 2008 Review of the IUCN Red List of Threatened Species. IUCN Gland, Switzerland.

Folse, L. J., Jr. 1974. Population ecology of Roadrunners (*Geococcyx californianus*) in south Texas. M.Sc. thesis, Texas A&M Univ., College Station.

Ford, N. B., and J. P. Karges. 1987. Reproduction in the checkered garter snake, *Thamnophis maricanus*, from southern Texas and northeastern Mexico: seasonality and evidence for multiple clutches. The Southwestern Naturalist 32: 93-101.

Ford, N. B., and J. P. Karges. 2001. Rates of water loss and estimates of survival time under varying humidity in juvenile snapping turtles (*Chelydra serpentina*). Copeia 2001: 521-525.

Frey, J. K. 2005. Status assessment of montane populations of the New Mexico meadow jumping mouse (*Zapus hudsonius luteus*) in New Mexico. Santa Fe, Final report submitted to New Mexico Department of Game and Fish: 74 + appendices on CD.

Frey, J. K. 2006. Field surveys for the New Mexico meadow jumping mouse (*Zapus hudsonius luteus*) at historical localities in the Sangre de Cristo Mountains, New Mexico.

Frey, J. K. and J. L. Malaney. 2009. Decline of the meadow jumping mouse (*Zapus hudsonius luteus*) in two mountain ranges in New Mexico.

Frey, Jennifer K. September 29, 1995. Mammals of the Negrito Creek Watershed, Gila National Forest, Reserve Ranger District, Catron County, New Mexico. A Final Report Submitted to: Gila National Forest, Reserve District, USDA Forest Service. 38 pp.

Friedmann, H. and L. F. Kiff. 1985. The parasitic cowbirds and their hosts. Proceedings of the Western Foundation of Vertebrate Zoology 2: 225-304.

Friend, M. and J. C. Franson, tech. eds. 1999. Field Manual of Wildlife Diseases: General Field Procedures and Diseases in Birds. USGS, Washington, D.C.

Friet, S. C. and A. W. Pinder. 1990. Hypoxia during natural aquatic hibernation of the bullfrog. American Zoologist. 30: 69A.

Fritzell, E. K., and K. J. Haroldson. 1982. *Urocyon cinereaoargenteus*. Mammalian Species 189: 1-8.

Fullerton, W., and D. Batts. 2003. A Framework for a restoration vision for the Rio Grande: Hope for a living river. The Alliance for Rio Grande Heritage, World Wildlife Fund. 142 p.

Gamble, L. R. and T. M. Bergin. 1996. Western Kingbird (*Tyrannus verticalis*). In The Birds of North America, No. 227 (A. Poole and F. Gill, eds.). The Academy of Natural Sciences, Philadelphia, PA, and The American Ornithologists' Union, Washington, D.C.

Garfin, G. and M. Lenart. 2007. Climate change effects on Southwest water resources. Southwestern Hydrology 6(1): 16-17, 34.

Garrison, Barrett A. 1999. Bank Swallow (*Riparia riparia*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/414

Garrison, T. E. and T. L. Best. 1990. *Dipodomys ordii*. Mammalian Species 353: 1-10.

Geluso, K. 2007. Winter activity of bats over water and along flyways in New Mexico. The Southwestern Naturalist 52: 482-492.

George, T. Luke, Ada C. Fowler, Richard L. Knight and Lowell C. McEwen. 1992. Impacts of a Severe Drought on Grassland Birds in Western North Dakota. Ecological Applications 2: 275-284.

Gibbons, J. W., Scott, D. E., Ryan, T. J., Buhlmann, K. A., Tuberville, T. D., Metts, B. S., Greene, J. L., Mills, T., Leiden, Y., Poppy, S. and C. T. Winne. 2000. The global decline of reptiles, déjà vu amphibians. BioScience 50: 653-666.

Gilbert, M., R. Leclair, Jr., R. Fortin. 1994. Reproduction of the Northern Leopard Frog (*Rana pipiens*) in Floodplain Habitat in the Richelieu River, P. Quebec, Canada. Journal of Herpetology, Vol. 28: 465-470.

Gitay, H. Suárez, A., and Watson, R.T. 2002. Climate change and Biodiversity. IPCC technical paper, Intergovernmental Panel on Climate Change, Geneve 77 pp.

Glinski, Richard L. and Robert D. Ohmart. 1984. Factors of Reproduction and Population Densities in the Apache Cicada (*Diceroprocta apache*). The Southwestern Naturalist 29: 73-79.

Goguen, Christopher B. and Nancy E. Mathews. 2001. Brown-headed cowbird behavior and movements in relation to livestock grazing. Ecological Applications 11: 1533-1544.

Goldberg, N. H. 1979. Behavioral flexibility and foraging strategies in Cassin's and Western kingbirds (*Tyrannus vociferans* and *Tyrannus verticalis*) breeding sympatrically in riparian habitats in central Arizona. Phd Thesis. Univ. of Illinois, Urbana-Champaign.

Gordo, O., L. Luis Bronton, X. Faerrer, and P. Comas. 2005. Do changes in climate patterns in wintering areas affect the timing of the spring arrival of trans-Saharan migrant birds? Global Change Biology 11:12-21.

Gowaty, Patricia Adair and Jonathan H. Plissner. 1998. Eastern Bluebird (*Sialia sialis*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/381.

Greenberg and Tanner. 2005. Spatial and temporal ecology of eastern spadefoot toads on a Florida landscape. Herpetologica 61: 20-28.

Greenlaw, Jon S. 1996. Spotted Towhee (*Pipilo maculatus*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/263.

Grindal, S. D., Morissette, J. L., and R. M. Brigham. 1999. Concentration of bat activity in riparian habitats over an elevational gradient. Canadian Journal of Zoology 77: 972-977. (Note species: *M. evotis, M. lucifugus, M. thysanodes, M. volans, M. yumanensis, E. fuscus, L. noctivagans* and *L. cinereus*).

Grubb, Jr., T. C. and V. V. Pravosudov. 2008. White-breasted Nuthatch (*Sitta carolinensis*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; http://bna.birds.cornell.edu/bna/species/054.

Grubb, T. C., Jr. 1975. Weather-dependent foraging behavior of some birds wintering in a deciduous woodland. Condor 77: 175-182.

Guzy, M. J., and G. Ritchison. 1999. Common Yellowthroat (*Geothylypis trichas*). In The Birds of North America, No. 448 (A. Poole and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA.

Haack, R. A. and J. W. Byler. 1993. Insects and pathogens: regulators of forested ecosystems. Journal of Forestry 91: 32-37.

Hainsworth, F. R., B. B. Collins and L. L. Wolf. 1977. The function of torpor in hummingbirds. Physiological Zoology 50: 215-222.

Hardy, P. C., M. L. Morrison and R. X. Barry. 1999. Abundance and habitat associations of Elf Owls and Western Screech-Owls in the Sonoran Desert. Southwestern Naturalist 44: 311-323.

Hart, J. S. 1971. Rodents in Comparative Physiology of Thermoregulation. Vol II Mammals Whittow (ed) Academic press, Inc, New York, New York. 1-147.

Haydon, D. T., Stenseth, N. C., Boyce M. S., and P. E. Greenwood. 2001. Phase coupling and synchrony in the spatiotemporal dynamics of muskrat and mink populations across Canada. Proceedings of the National Academy of Science 98: 13149-13154.

Heath, J. E. and P. J. Wilkin. 1970. Temperature Responses of the Desert Cicada, *Diceroprocta apache* (Homoptera, Cicadidae). Physiological Zoology 43: 145-154.

Heller, N. E., Sanders, N. J., Shors, J. W. and D. M. Gordon. 2007. Rainfall facilitates the spread, and time alters the impact of the invasive Argentine ant. Oecologia 155: 385-95.

Hermanson, J. W. and T. J. O'Shea. 1983. *Antrozous pallidus*. Mammalian Species 213: 1-8.

Hill, G. E. 1995. Black-headed Grosbeak (*Pheucticus melanocephalus*). In The Birds of North America, No. 143 (A. Poole and F. Gill, eds.). The Academy of Natural Sciences, Philadelphia and The American Ornithologists' Union, Washington, D.C.

Hill, G. E. 1988. Age, plumage brightness, territory quality, and reproductive success in the black-headed grosbeak. Condor 90: 379-388.

Hill, Geoffrey E. 1993. House Finch (*Carpodacus mexicanus*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu.bnaproxy.birds.cornell.edu/bna/species/046 doi:10.2173/bna.46.

Hink, V. C., and R. D. Ohmart. 1984. Middle Rio Grande Biological survey. U.S. Army Corps of ENginneers, Albuquerque District, New Mexico. Contract No. DACW47-81-C-0015, Arizona State University.

Hitchcock, C. L., and D. F. Sherry. 1990. Long-term memory for cache sites in the Black-capped Chickadee. Animal Behaviour 40: 701-712.

Hoffmeister, D. F. 1981. *Peromsycus truei*. Mammalian Species 161: 1-5.

Hogg, I. D., and D. D. Williams. 1996. Response of stream invertebrates to a global warming thermal regime: an ecosystem-level manipulation. Ecology 77: 395-407.

Holecheck, J. L., T. T. Baker, and J. C. Boren. 2004. Impacts of controlled grazing versus grazing exclusion on rangeland ecosystems: what we have learned. New Mexico State University Cooperative Extension Service, Range Improvement Task Force Report 57. Las Cruces, New Mexico. 42 pp.

Holechek, J. L., R. D. Pieper, and C. H. Herbel. 1998. Range management: principles and practices, third edition. Prentice Hall, New Jersey.

Hood, G. A. and S. E. Bayley. 2008. Beaver (*Castor canadensis*) mitigate the effects of climate on the area of open water in boreal wetlands in western Canada. Biological Conservation 141: 556-567.

Howard, R. 1983. Sexual selection and variation in reproduction success in a long-lived organism. American Naturalist 122: 301-325.

Howard, R. 1984. Alternative mating behaviors of young male bullfrogs. American Zoologist 24: 397-406.

Howe, W. H. 1986. Status of the Yellow-billed Cuckoo (*Coccyzus americanus*) in New Mexico. Rep. 516.6-75-09. N.M. Dep. Game Fish, Santa Fe.

Howe, W. H. and F. L. Knopf. 1991. On the imminent decline of Rio Grande cottonwoods in central New Mexico. The Southwestern Naturalist 36: 218-224.

Howell, T. R. 1975. Bank swallow (*Riparia riparia*), bobolink (*Dolichonyx oryzivorous*), and other birds at a desert reservoir in Chile. Condor 77: 105.

Hoylman, A. M., A. Peterson, J. V. H. Constable, J. B. Picone, J. T. Ball. 1998. Climate Change and Wetland Processes in the Southwest United States: USBR COOPERATIVE AGREEMENT 2-FC-30-O0200 Final Report Response of Riparian Communities to Rising C02 Levels.

Huey, R. B. and J. J. Tewksbury. 2009. Can behavior douse the fire of climate warming? PNAS 106: 3647-3648.

Hughes, J. M. 1996. Greater Roadrunner (*Geococcyx californianus*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/244.

Hughes, Janice M. 1999. Yellow-billed Cuckoo (*Coccyzus americanus*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/418.

Hunter, W. C. 1988. Dynamics of bird species assemblages along a climatic gradient: a Grinnellian niche approach. MS Thesis. Arizona State University, Tempe, AZ. 103 p.

Hunter, William C., Robert D. Ohmart and Bertin W. Anderson. 1988. Use of Exotic Saltcedar (*Tamarix chinensis*) by Birds in Arid Riparian Systems. The Condor 90: 113-123.

Hurd, Brian H. and Julie Coonrod. 2008. Climate change risks New Mexico's waterways: its byways and its flyways. Water Resources IMPACT 10(4).

Ingold, J. 1993. Blue Grosbeak (*Guiraca caerulea*), In The Birds of North America, No. 79 (A. Poole and F. Gill, Eds.). The Academy of Natural Sciences and the American Ornithologists' Union, Philadelphia, PA.

Inkley, D. B. Anderson, M. G. Blaustein, A. R., Burkett, V. R., Felzer, B. and others. 2004. Global climate change and wildlife in North America. Wildlife Society Technical Review 04-2. The Wildlife Society, Bethesda, Maryland, USA 26 pp.

Intersecretarial Commission on Climate Change. 2007. National Strategy on Climate Change: Mexico. Available at: http://www.un.org/ga/president/61/follow-up/climatechange/Nal_Strategy_MEX_eng.pdf.

IPCC. 2007. Climate change 2007: the physical science basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Soloman, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K. B. Averyt, M. Tignor, and H. L. Miller (eds.). Cambridge University Press, Cambridge, UK and New York, NY, USA. 996 pp.

Irwin, K. 1985. Foraging ecology and reproduction of the Black Phoebe in Humboldt County. Master's Thesis. California State Univ., Humboldt.

Jackson, J. A. and H. R. Ouellet. 2002. Downy Woodpecker (*Picoides pubescens*). In The Birds of North America, No. 613 (A. Poole and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA.

Jackson, J. A., H. R. Ouellet and B. J. Jackson. 2002. Hairy Woodpecker (*Picoides villosus*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu.bnaproxy.birds.cornell.edu/bna/species/702 doi:10.2173/bna.702.

James, J.D. 1998. Status of the Great Plains toad (*Bufo cognatus*) in Alberta. Alberta Environmental Protection, Wildlife Management Division, and Alberta Conservation Association, Wildlife Status Report, Number 14, Edmonton, Alberta.

Janzen, F. J. 1994. Climate change and temperature-dependent sex determination in reptiles. Proceedings of the National Academy of Sciences 91: 7487-7490.

Jenkins, S. H. and P. E. Busher. 1979. *Castor canadensis*. Mammalian Species 120: 1-8.

Johnson, D. M., G. L. Stewart, M. Corley, R. Ghrist, J. Hagner, A. Ketterer, B. McDonnell. Newsom, E. Owen, and P. Samuels. 1980. Brown-headed Cowbird (*Molothrus ater*) mortality in an urban winter roost. Auk 97: 299-320.

Johnson, Kristine, Patricia Mehlhop, Charles Black and Kim Score. 1999. Reproductive failure of endangered southwestern willow flycatchers on the Rio Grande, New Mexico. The Southwestern Naturalist 44: 226-231.

Johnson, M. and C. van Riper, III. 2004. Brown-headed Cowbird parasitism of the Black-throated Sparrow in central Arizona. Journal of Field Ornithology 75: 303-311.

Johnson, R. Roy, Helen K. Yard and Bryan T. Brown. 1997. Lucy's Warbler (*Vermivora luciae*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/318.

Johnson, W. C., M. D. Dixon, R. Simons, S. Jenson, and K. Larson. 1995. Mapping the response of riparian vegetation to possible flow reductions in the Snake River, Idaho. Geomorphology 13: 159-173.

Johnston, David W. and T. P. Haines 1957. Analysis of Mass Bird Mortality in October, 1954. The Auk 74: 447-458.

Johnston, K. M. and Schmitz, O. J. 1997. Wildlife and climate change: Assessing the sensitivity of selected species to simulated doubling of atmospheric CO2. Global Change Biology 3: 531-544.

Jones, C. A. and C. N. Baxter. 2004. *Thomomys bottae*. Mammalian Species 742: 1-14.

Jones, H. L. and J. M. Diamond. 1976. Short-time-base studies of turnover in breeding bird populations on the California Channel Islands. Condor 78: 526-549.

Jones, K. B. 1990. Habitat use and predatory behavior of *Thamnophis cyrtopsis* (Serpentes: colubridae) in a seasonally variable aquatic environment. The Southwestern Naturalist 35: 115-122.

Julius, and West. 2007. Preliminary review of adaptation options for climate sensitive ecosystems and resources. U.S. Climate change Science Program. Synthesis and Assessment product 4.4.

Justus, J., M. Sandomir, T. Urquhart, and B. Ewan. 1977. Developmental rates of two species of toads for the desert southwest. Copeia 3: 592-594.

Kennedy, E. D. and D. H. White. 1997. Bewick's Wren (*Thryomanes bewickii*). In The Birds of North America, No. 315 (A. Poole and F. Gill, eds.). The Academy of Natural Sciences, Philadelphia, PA and The American Ornithologists Union, Washington, D.C.

Kennedy, E. D. and D. W. White. 1996. Interference Competition from House Wrens as a Factor in the Decline of Bewick's Wrens. Conservation Biology, Vol. 10, No. 1: 281-284.

Kilham, L. 1971. Roosting Habits of White-Breasted Nuthatches. Condor 73: 113-114.

King, O. 1960. Observations on Oklahoma toads. Southwestern Naturalist 5: 102-103.

King, T. L., Switzer, J. F., Morrison, C. L., Eackles, M. S., Young, C. C., Lubinski, B. A., and P. Cryan. 2006. Comprehensive genetic analyses reveal evolutionary distinction of a mouse (*Zapus hudsonius preblei*) proposed for delisting from the US endangered species Act. Molecular Ecology 15: 4331-4359.

Klimkiewicz, M. and A. Futcher. 1987. Longevity records of North American birds: Coerebinae through Estrildidae. Journal of Field Ornithology 58: 318-333.

Knapton, Richard W. 1988. Nesting success is higher for polygynously mated females than for monogamously mated females in the Eastern meadowlark. The Auk 105: 325-329.

Kocan, R. M. and S. R. Amend. 1972. Immunologic status of Mourning Doves following an epizootic of trichomoniasis. Journal of Wildlife Diseases 8: 176-180.

Kranjcec, J., J. M. Mahoney, and S. B. Rood. 1998. The responses of three riparian cottonwood species to water table decline. Forest Ecology and Management 110: 77-87.

Kroodsma, D. E. 1974. Song learning, dialects, and dispersal in the Bewick's Wren. Z. Tierpsychol. 35: 352-380.

Krupa, J. J. 1994. Breeding biology of the Great Plains toad in Oklahoma. Journal of Herpetology 28: 217-224.

Krzysik, A. J. 1990. Biodiversity in riparian communities and watershed management. In R. E. Riggins, E. B. Jones, R. Singh, and P. A. Rechard (eds.), Watershed planning and analysis in action. Proceedings of the symposium held at Durango, Colorado, July 9-11, 1990. American Society of Civil Engineers, New York, New York.

Kupferberg, S. 1997. Bullfrog (*Rana catesbeiana*) invasion of a California river: the role of larval competition. Ecology 78: 1736-1751.

Kurta, A. 1995. Mammals of the great lakes region: revised edition. The University of Michigan Press, U.S.A.

Laakkonen, J. Fisher, R. N. and T. J. Case. 2001. Effect of land cover, habitat fragmentation and ant colonies on the distribution and abundance of shrews in southern California. Journal of Animal Ecology 70: 776-788.

Laakkonen, J., Fisher, R. N. and T. J. Case. 2003. Factors influencing the variation in capture rates of shrews in southern California, USA. Acta Theriologica 48: 157-166.

Lackey, J. A., Huckaby, D. G., and B. G. Ormiston. 1985. *Peromyscus leucopus*. Mammalian Species 247: 1-10.

Lacki, M. J., Johnson, J. S., Dodd, E. D. and M. D. Baker. 2007. Prey consumption of insectivorous bats in coniferous forest of North-Central Idaho. Northwest Science 81: 199-205.

Ladeau, S. L., P. P., Marra P. P., A. M. Kilpatrick and C. A. Calder. 2008. West Nile Virus Revisited: Consequences for North American Ecology. Bioscience: 937-946.

Lanyon, W. E. 1963. Experiments on species discrimination in Myiarchus flycatchers. American Museum Novitates 2126.

Larivière, S. 2001. *Ursus americanus*. Mammalian Species 647: 1-11.

Larivière, S. and Walton, L. R. 1997. *Lynx rufus*. Mammalian Species 563: 1-8.

Lasiewski, R. C. 1964. Body temperatures, heart and breathing rate, and evaporative water loss in hummingbirds. Physiological Zoology 37: 212-223.

Laskey, A. R. 1946. Some Bewick Wren nesting data. Migrant 17: 39-43.

Laymon, S. A. and M. D. Halterman. 1987. Can the western subspecies of the yellow-billed cuckoo be saved from extinction? Western Birds 18: 19-25.

Leopold, B. D. and P. R. Krausman. 1991. Factors influencing desert mule deer distribution and productivity in Southwestern Texas. The Southwestern Naturalist 36: 67-74.

Leuth, F. X. 1941 Effects of temperature on snakes. Copeia 3: 125-132.

Lewis, S. E. 1993. Effect of climatic variation on reproduction by pallid bats (*Antrozous pallidus*). Canadian Journal of Zoology 71: 1429-1433.

Lewis, S. E. 1996. Low roost-site fidelity in pallid bats: associated factors and effect on group stability. Behavioral Ecology and Sociobiology 39: 335-344.

Li, P., and T. Martin. 1991. Nest-site selection and nesting success of cavity-nesting birds in high elevation forest drainages. Auk 108: 405-418.

Ligon, J. D. 1969. Some aspects of temperature relations in small owls. Auk 86: 458-472.

Liknes, E. T. and D. L. Swanson. 1996. Seasonal variation in cold tolerance, basal metabolic rate, and maximal capacity for thermogenesis in White-breasted Nuthatches *Sitta carolinensis* and Downy Woodpeckers *Picoides pubescens*, two unrelated arboreal temperate residents. Journal of Avian Biology 27: 279-288.

Lima, Mauricio S. K. Morgan Ernest, James H. Brown, Andrea Belgrano, Nils Chr. Stenseth. 2008. Chihuahuan desert Kangaroo Rats: Nonlinear effects of population dynamics, competition, and rainfall. Ecology 89: 2594-2603.

Lindstedt, S. L. 1980. Regulated hypothermia in the desert shrew. Journal of Comparative Physiology 137: 173-176.

Linsdale, J. M. 1949. Survival in birds banded at the Hastings Reservation. Condor 51: 88-96.

Long, S. 2003. "*Thomomys talpoides*" (On-line), Animal Diversity Web. Accessed January 08, 2010 at http://animaldiversity.ummz.umich.edu/site/accounts/information/Thomomys_talpoides.html.

Lotze, J-H, and S. Anderson. 1979. *Procyon lotor*. Mammalian Species 119: 1-8.

Lowther, P. E. 1993. Brown-headed Cowbird (*Molothus ater*). In The Birds of North America, No. 47 (A. Poole and F. Gill, Eds.). Philadelphia: The Academy of Natural Sciences; Washington, D.C.: The American Ornithologists Union.

Lynn, J. C., T. J. Koronkiewicz, M. J. Whitfield, and M. K. Sogge. Willow flycatcher winter habitat in El Salvador, Costa Rica, and Panama: characteristics and threats. Studies in Avian Biology 26: 41-51.

Macèdo, R. H. and M. A. Mares. 1988. *Neotoma albigula*. Mammalian Species 310: 1-7.

Mackie, R. J., Kei, J. G., Pac, D. F., and K. L. Hamlin. 2003. Mule Deer In Wild mammals of North America: Biology, management, and conservation 2nd Ed. by Feldhamer, Thompson, and Chapman. John Hopkins University Press, Baltimore, Maryland 1216 pp.

Maehr, D. S. and J. A. Cox. 1995. Landscape features and panthers in Florida. Conservation Biology 9: 1008-1019.

Magrin, G., C. Gay García, D. Cruz Choque, J. C. Giménez, A. R. Moreno, G. J. Nagy, C. Nobre and A. Villamizar. 2007. Latin America. Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, M. L. Parry, O. F. Canziani, J. P. Palutikof, P. J. van der Linden and C. E. Hanson, Eds., Cambridge University Press, Cambridge, UK. 581-615.

Martin, T. E. 1987. Food as a limit on breeding birds: a life-history perspective. Annual Review of Ecology and Systematics: 18: 453-487.

Mason, E. A. 1944. Parasitism by *Protocalliphora* and management of cavity-nesting birds. Journal of Wildlife Management 8: 232-237.

Mathews, W. J. and E. G. Zimmerman. 1990. Potential effects of global warming on native fishes on the southern Great Plains and Southwest. Fisheries 15: 26-32.

Maurer, E. P., L. Brekke, T. Pruitt, and P. B. Duffy. 2007. Fine-resolution climate projections enhance regional climate change impact studies. Eos Transactions AGU 88(47): 504.

Mawdsley, Jonathan R., Robin O'Malley, and Dennis S. Ojima. 2009. A review of climate-change adaptation Strategies for Wildlife Management and Biodiversity Conservation. Conservation Biology 5: 1080-1089.

McAda, D. P. and Barroll, P. 2002. Simulation of ground-water flow in the Middle Rio Grande Basin between Cochiti and San Acacia, New Mexico. USGS Water Resources Investigations Report 02-4200.

McCain, C. M. 2007. Could temperature and water availability drive elevational species richness patterns? A global case study for bats. Global Ecology and Biogeography 16: 1-13.

McCarty, J. 2001. Ecological consequences of recent climate change. Conservation Biology 15: 320-331.

McCorquodale, S. M. 1991. Energetic considerations and habitat quality for elk in arid grasslands and coniferous forests. Journal of Wildlife Management 55: 237-242.

McCorquodale, Scott M., Kenneth J. Raedeke and Richard D. Taber. 1986. Elk Habitat Use Patterns in the Shrub-Steppe of Washington . The Journal of Wildlife Management 50: 664-669.

McCracken, G. F. 1986. Why Are We Losing Our Mexican Free-tailed Bats? Bat Conservation International 3: 1-2.

Meijer, Theo U. Nienaber, U. Langer and F. Trillmich. 1999. Temperature and Timing of Egg-Laying of European Starlings. The Condor 101: 124-132.

Memmott, J., P. G. Craze, N. M. Waser, and M. V. Price. 2007. Global warming and the disruption of plant–pollinator interactions. Ecology Letters (2007) 10: 710-717.

Menke, S. B. 2003. Lizard community structure across a grassland-creosote bush ecotone in the Chihuahuan Desert. Canadian Journal of Zoology 81: 1829-1838.

Merino, Santiago and Jaime Potti. 1996. Weather Dependent Effects of Nest Ectoparasites on Their Bird Hosts. Ecography 19: 107-113.

Merrill, E. H. 1991. Thermal constraints in the use of cover types and activity time of elk. Applied Behavioral Science 29: 251-267.

Merrit, D. M. and D. J. Cooper. 2000. Riparian vegetation and channel change in response to river regulation: a comparative study of regulated and unregulated streams in the Green River Basin, USA. Regulated Rivers: Research and Management 16: 543-564.

Middleton, A. 1979. Influence of age and habitat on reproduction by the American goldfinch. Ecology 60: 418-432.

Middleton, A. 1984. Longevity of the American goldfinch. Journal of Field Ornithology 55: 383-386.

Milius, S. 1998. Fatal skin fungus found in U.S. frogs. Science News 154: 7.

Mitchell, D. L., D. Ivanova, R. Rabin, K. Redmond, and T. J. Brown. 2002. Gulf of California sea surface temperatures and the North American monsoon: Mechanistic implications from observations. Journal of Climate 15: 2261-2281.

Mitchell, R. J., Morecroft, M. D., Acreman, M., and others. 2007. England biodiversity strategy- towards adaptation to climate change. Available at: http://www.defra.gov.uk/wildlife-countryside/resprog/findings/ebs-climate-changes.pdf.

Mitrovich, Milan J., Jay E. Diffendorfer, and Robert N. 2009. Behavioral Response of the Coachwhip (*Masticophis flagellum*) to Habitat Fragment Size and Isolation in an Urban Landscape. Journal of Herpetology 43: 646-656.

Møller, Anders Pape. 2004. Protandry, sexual selection and climate change. Global Change Biology 10: 2028-2035.

Moore, G. A. 1937. The spadefoot toad under drought conditions. Copeia 4: 225-226.

Moore, J. 1984. European starling reproduction in central New Mexico. Journal of Field Ornithology 55: 254-257.

Moore, W. S. 1965. Northern Flicker (*Colaptes auratus*). In The Birds of North America, No. 166 (A. Poole and F. Gill, eds.). The Academy of Natural Sciences, Philadelphia and The American Ornithologists Union, Washington, D.C.

Morrison, J. L. 1992. Persistence of the Meadow Jumping Mouse, *Zapus hudsonius leuteus*, in New Mexico. The Southwestern Naturalist 37: 308-311.

Muchlinski, A. E. 1980. The effects of daylength and temperature on the hibernating rhythm of the meadow jumping mouse (*Zapus hudsonius*). Physiological Zoology 53: 410-418.

Muchlinski, A. E. 1988. Population attributes related to the life-history strategy of hibernating *Zapus hudsonius*. Journal of Mammalogy. 69: 860-865.

Mukheibir, P., and G. Ziervogel. 2006. Framework for adaptation to climate change in the city of Cape Town. Available at: http://www.erc.uct.ac.za/publication/Framework%20for%20adaptation%20to%20CC%20in%20the%20city%20of%20Cape%20Town%20-%20FAC4t.pdf.

Muths, Erin, P. S. Corn, A. P. Pessier and D. E. Green. 2003. Evidence for disease-related amphibian decline in Colorado. Biological Conservation 110: 357-365.

Mysterud, A. and Eivind, O. 1999. Cover as a Habitat Element for Temperate Ungulates: Effects on Habitat Selection and Demography. Wildlife Society Bulletin 27: 385-394.

National Research Council. 1994. Restoring and protecting marine habitat: the role of engineering and technology. National Academy Press, Washington, D.C.

NatureServe. 2009. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia. Available http://www.natureserve.org/explorer (Accessed: February 18, 2009).

Nelson, T., J. L. Holechek, R. Valdez, and M. Cardenas. 1997. Wildlife numbers on late and mid-seral Chihuahuan Desert rangelands. Journal of Range Management 50: 593-599.

Nelson, J., J. L. Holecheck, and R. Valdez. 1999. Wildlife plant community preference in the Chihuahuan Desert. Rangelands 21: 9-12.

Niedzielski, S. 2002. "*Terrapene carolina*" (On-line), Animal Diversity Web. Accessed January 19, 2010 at http://animaldiversity.ummz.umich.edu/site/accounts/information/Terrapene_carolina.html.

Nieuwolt, P. 1996. Movement, activity and microhabitat selection in the western box turtle *Terrapene ornate luteola*, in New Mexico. Herpetologica 52: 487-495.

New Mexico Department of Game and Fish [NMDGF]. 1988. Handbook of species endangered in New Mexico 6-253: 1-2. Santa Fe, New Mexico. 150 pp.

Nolan, V. and C. F. Thompson. 1975. The Occurrence and Significance of Anomalous Reproductive Activities in Two North American Non-parasitic Cuckoos *Coccyzus* sp. Ibis 117: 496-503.

Noss, R. F., E. T. LaRoe III, and J. M. Scott. 1995. Endangered ecosystems of the United States: a preliminary assessment of Loss and Degradation. Biological Report 28, National Biological Survey, Washington, D.C.

O'Connor, M. and C. Tracy. 1992. Thermoregulation of juvenile toads of *Bufo woodhousii* in the field and in the laboratory. Copeia 1992(3): 865-876.

O'Farrell. 1974. Seasonal activity patterns of rodents in a sagebrush community. Journal of Mammalogy 55: 809-823.

Oaks, E. C., Young, P. J., Kirkland, Jr., G. L., and D. F. Schmidt. 1987. *Spermophilus variegatus*. Mammalian Species 272: 1-8.

Oberlander, G. 1939. The history of a family of Black Phoebes. Condor 41:133-151.

Ohlendorf, H. M. 1971. Competitive relationships among selected species of flycatchers (Tyrannidae) in Trans-Pecos Texas. Ph.D Dissertation. Texas A&M Univ., College Station.

Ohmart, R. D. 1973. Observations on the breeding adaptations of the Roadrunner. Condor 75: 140-149.

Orians, G. H. and M. F. Willson. 1964. Interspecific territories of birds. Ecology 45: 736-745.

Ostrand, W. D., J. A. Bissonette, and M. R. Conover. 1995. Trichomoniasis as a factor in Mourning Dove population decline in Fillmore, Utah. Journal of Wildlife Diseases 31: 87-89.

Otis, D. L., J. H. Schulz, D. Miller, R. E. Mirarchi and T. S. Baskett. 2008. Mourning Dove (*Zenaida macroura*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/117.

Palmer, M. A., D. P. Lettenmaier, N. L. Poff, S. L. Postel, R. Richter, and R. Warner. 2009. Climate Change and river ecosystems: Protection and adaptation options. Environmental Management 44: 1053-1068.

Palmer, M. A., Reidy, C. A., C. Nilsson, M. Flörke, J. Alcamo, P. S. Lake, and N. Bond. 2008. Climate change and the world's river basins: anticipating management option. Frontiers in Ecology and the Environment 6: 81-89.

Panjabi, A. O., E. H. Dunn, P. J. Blancher, [and others]. 2005. The Partners in Flight handbook on species assessment. Version 2005. Partners in Flight Technical Series No. 3.

Parker, W. S. and E. R. Pianka. 1975. Comparative ecology of populations of the lizard *Uta stansburinana*. Copeia 4: 615-632.

Parmesan, C., T. Root, and M. Willig. 2000. Impacts of extreme weather and climate on terrestrial biota. Bulletin of the American Meteorological Society 81:443-450.

Pase, C. P. and E. F. Layser. 1977. Classification of riparian habitat in the southwest. Pages 5-9 in R. R. Johnson and D. A. Jones (tech. coords.), Importance, preservation and management of riparian habitat: a symposium. U.S. Forest Service, General Technical Report RM-43. Tucson, Arizona.

Paxton, E. H., J. Owen and M. K. Sogge. 1996. Southwestern Willow Flycatcher response to catastrophic habitat loss. Report by the USGS Colorado Plateau Research Station, Flagstaff, AZ.

Payne, Robert B. 2006. Indigo Bunting (*Passerina cyanea*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu.bnaproxy.birds.cornell.edu/bna/species/004 doi:10.2173/bna.4.

Peek, J. M., Dennis, B. and T. Hershey. 2002. Predicting population trends of mule deer. Journal of Wildlife management 66: 729-736.

Peet, R. K. 1988. Forests of the Rocky Mountains. In M. G. Barbour and W. D. Billings (eds.), North American vegetation. Cambridge University Press, New York, New York.

Peñuelas, J. and I. Filella. 2001. Responses to a warming world. Science 294: 793-795.

Peralta-Hernandez, A., R. C. Balling Jr., and L. R. Barba-Martinez. 2009. Analysis of near-surface diurnal temperature variations and trends in southern Mexico. International Journal of Climatology 29: 205-209.

Petersen, M. K. 1979. A laboratory test of species exclusion in *Sigmodon fulviventer* and *S. hispidus*. Iowa Stat Journal of Research 53: 301-304.

Phennig, D. W. 1992. Polyphenism in spadefoot toad tadpoles as a logically adjusted evolutionarily stable strategy. Evolution 46: 1408-1420.

Phillips, A. R. 1986. The known birds of North and Middle America. Pt. I. Allan R. Phillips, Denver, CO.

Phillips, A., J. Marshall, and G. Monson. 1964. The birds of Arizona. Univ. of Arizona Press, Tucson.

Pianka, E. R. 1970. Comparative autecology of the lizard *Cnemidophouus tigris* in different parts of its geographic range. Ecology 51: 703-720.

Pierce, B. M. and V. C. Bleich. 2003. Mountain Lion (pages 744-758) In Wild mammals of North America: Biology, management, and conservation 2nd Ed. by Feldhamer, Thompson, and Chapman. John Hopkins University Press, Baltimore, Maryland 1216 pp.

Pinkowski, B. C. 1971a. An analysis of banding recovery data on eastern bluebirds in Michigan and three neighboring states. Jack-Pine Warbler 49: 33-50.

Plissner, J. H. and P. A. Gowaty. 1996. Patterns of natal dispersal, turnover, and dispersal costs in eastern bluebirds. Animal Behavior 51: 1307-1322.

Poling, T. D. and S. E. Hayslette. 2006. Dietary overlap and foraging competition between mourning doves and Eurasian collared-doves. Journal of Wildlife Management 70: 998-1004.

Popa-Lisseanu, A. G. and C. C. Voigt. 2009. Bats on the Move. Journal of Mammalogy 90: 1283-1289.

Potvin, C. L., and J. Bovet. 1975. Annual cycle of patterns of activity rhythms in beaver colonies (*Castor canadensis*). Journal of Comparative Physiology 98: 243-256.

Punzo, F. and R. Lopez. 2003. Observation son nest site selection and litter size in the gray shrew from Presidio County, Texas. Texas Journal of Science.

Pylypec, B. 1991. Impacts of fire on bird populations in a fescue prairie. Canadian Field-Naturalist 105: 346-349.

Redder, A. J., C. K. Dodd, Jr., and D. Keinath. 2006. Ornate Box Turtle (*Terrapene ornata ornata*): a technical conservation assessment. [Online]. USDA Forest Service, Rocky Mountain Region. Available: http://www.fs.fed.us/r2/projects/scp/assessments/ornateboxturtle.pdf.

Rehfeldt, Gerald E., Nicholas L. Crookston, Marcus V. Warwell, and Jeffrey S. Evans. 2006. Empirical analyses of plant-climate relationships for the western United States. International Journal of Plant Sciences 167(6): 1123-1150.

Rintamaki, R. 1986. Desert Cottontail. US Department of Agriculture, Wyoming Soil Conservation Service Technical Notes 104. 2p. Note: Text says data taken from: Ecoregion M3113 Handbook and Habitat Suitability Index Models, Wildlife Species Narratives (literature searches), U.S. Fish and Wildlife Service.

Rising, J. D. and P. L. Williams. 1999. Bullock's Oriole (*Icterus bullockii*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Available: http://bna.birds.cornell.edu/bna/species/416.

Roberson, D. and C. Tenney. 1993. Atlas of the breeding birds of Monterey County, California. Monterey Peninsula Audubon Soc. Monterey, CA.

Robinson, S. K., S. I. Rothstein, M. C. Brittingham, L. J. Petit, and J. A. Grzybowski. 1995. Ecology and behavior of cowbirds and their impact on host populations. In Ecology and Management of Neotropical Migratory Birds (T. E. Martin and D. M. Finch, eds.), Oxford University Press, New York.

Robinson, W. D. 1996. Summer Tanager (*Piranga rubra*). In The birds of North America, No. 248 (A. Poole and F. Gill, eds.). The Academy of Natural Sciences, Philadelphia, PA and the American Ornithologists' Union, Washington, D.C.

Roest, A. I. 1957. Notes on the American Sparrow Hawk. Auk 74: 1-19.

Rohwer, S., K. Hobson, and V. Rohwer. 2009. Migratory double breeding in neotropical migratory birds. Proceedings of the National Academy of Sciences 106: 19050-19055.

Rood, S. B., Pan, J., Gill, K. M., Franks, C. G., Sameulson, G. M., and A. Shepard. 2006. Declining summer flow so the Rocky Mountain rivers: Changing seasonal hydrology and probable impacts of floodplain forests. Journal of Hydrology 349: 397-410.

Root, T. 1988. Atlas of wintering North American birds: an atlas of Christmas Bird Count data. University of Chicago Press, Chicago, Illinois. 312 pp.

Root, Terry, Jeff T. Price, Kimberly R. Hall, Stephen H. Schneider, Cynthia Rosenzweig and J. Alan Pounds. 2003. Fingerprints of global warming on wild animals and plants. Nature 421: 57-60.

Rosatte, R. and S. Larivière. 2003. In Wild Mammals of North America: Biology, management, and Conservation. Feldhamer, G. A., Thompson, B. C. and J. A. Chapman (Eds). Johns Hopkins University Press, Baltimore, MD. 1197 p.

Rosen, P. C. 1991. Comparative field study of thermal preferenda in garter snakes (*Thamnophis*). Journal of Herpetology 25: 301-312.

Rosenberg, K. V., J. D. Lowe, and A. A. Dhondt. 1999. Effects of Forest Fragmentation on Breeding Tanagers: A Continental Perspective. Conservation Biology 13: 568-583.

Rosenberg, K. V., R. D. Ohmart, and B. W. Anderson. 1982. Community organization of riparian breeding birds: response to an annual resource peak. Auk 99: 260-274.

Rosenberg, K. V., R. D. Ohmart, W. C. Hunter, and B. W. Anderson. 1981. Birds of the Lower Colorado River valley. Univ. of Arizona Press, Tucson.

Rowsemitt, C. N. and A. J. O'Connor. 1989. Reproductive function in *Dipodomys ordii* stimulated by 6-MethoxyBensoxazolinone. Journal of Mammalogy 70: 805-809.

Rudd, W. J.; A. L. Ward and L. L. Irwin. 1983. Do Split Hunting Seasons Influence Elk Migrations from Yellowstone National Park? Wildlife Society Bulletin 11: 328-331.

Ruibal, R. 1962. The adaptive value of bladder water in the toad, *Bufo cognatus*. Physiological Zoology 35: 218-223.

Russell, S. M. and G. Monson. 1998. The birds of Sonora. Univ. of Arizona Press, Tucson.

Rustigian, H. L., M. V. Santelmann, and N. H. Schumaker. 2003. Assessing the potential impacts of alternative landscape designs on amphibian population dynamics. Landscape Ecology 18: 65-81.

Safi, K. and G. Kerth. 2004. A comparative analysis of specialization and extinction risk in temperate-zone bats. Conservation Biology 18: 1293-1303.

Saino, N., A. M. Bolzern, and A. P. Møller. 1997. Immunocompetence, ornamentation, and viability of male barn swallows (*Hirundo rustica*). Proceedings of the National Academy of Sciences 94: 549-552.

Sanz, Juan José. 2002. Climate change and birds: have their ecological consequences already been detected in the Mediterranean region? Ardeola 49: 109-120.

Sarell, Mike. 2004. Tiger Salamander from Accounts and Measure for Managing Identified Wildlife- Account V.2004, Canada. Available: http://www.env.gov.bc.ca/wld/frpa/iwms/documents/Amphibians/a_tigersalamander.pdf.

Saunders, Stephen, Charles Montgomery, Tom Easley, and Theo Spencer. 2008. Hotter Drier West: the West's changed climate. Rocky Mountain Climate Organization and Natural Resources Defense Council, New York. Available: www.nrdc.org/policy.

Scheel, D., Vincent, T. L. S. and G. N. Cameron. 1996. Global warming and the species richness of bats in TX. Conservation Biology 10: 452-464.

Schroeder, D. A. 1985. Breeding biology and population limitation in the Black Phoebe (*Sayornis nigricans*) in southern coastal California. Ph.D. Dissertation. Univ. of California, Santa Barbara.

Seager, R., M. Ting, I. Held, [and others]. 2007. Model projections of an imminent transition to a more arid climate in southwestern North America. Science 316: 1181-1184.

Searcy, W. A. and K. Yasukawa. 1995. Polygyny and sexual selection in red-winged blackbirds. Princeton Univ. Press, Princeton, NJ.

Sedgwick, J. A. 2000. Willow Flycatcher (*Empidonax trailii*). In The Birds of North America, No. 533 (A. Poole and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA.

Sedgwick, J. A. and W. M. Iko. 1997. Costs of Brown-headed Cowbird parasitism to Willow Flycatchers. Studies in Avian Biology 18: 167-181.

Segal, S. 1960. Bird tragedy at the Dunes. Indiana Audubon Quarterly. 38: 23-25.

Seigel, R. A. and N. B. Ford. 2001. Phenotypic plasticity in reproductive traits: geographical variation in plasticity in a viviparous snake. Functional Ecology 15: 36-42.

Seigel, R. A., N. B. Ford, and L. A. Mahrt. 2000. Ecology of an aquatic snake (*Thamnophis marcianus*) in a desert environment: Implications of early timing of birth and geographic variation in reproduction. American Midland Naturalist 143: 453-462.

Shafroth, P. B., J. C. Stromberg, and D. T. Patten. 2002. Riparian vegetation response to altered disturbance and stress regimes. Ecological Applications 12: 107-123.

Sheffield, S. R. and H. H. Thomas. 1997. *Mustela frenata*. Mammalian Species 570: 1-9.

Short, L. 1974. Nesting of southern Sonoran birds during the summer rainy season. Condor 76: 21-32.

Shump, K. A., Jr. and A. U. Shump. 1982. *Lasiurus cinereus*. Mammalian Species 185: 1-5.

Sieber, O. 1980. Causal and functional aspects of brood distribution in Sand Martins (*Riparia riparia L.*). Zeitschrift fur Tierpsychologie 52: 19-56.

Silver, M. 1993. Second-year management of a cliff swallow colony in Massachusetts. Bird Observer 21: 150-155.

Simons, L. H., Szaro, R. C. and S. C. Belfit. 1990. Distribution of *Notiosorex crawfordi* and *Sorex arizonae* along an elevational gradient. Journal of Mammalogy 781: 634-640.

Sims, K. 2000. "*Myotis yumanensis*" (On-line), Animal Diversity Web. Accessed February 18, 2009 at http://animaldiversity.ummz.umich.edu/site/accounts/information/Myotis_yumanensis.html.

Skelly, D., L. Joseph, H. Possingham, L. Freidenburg, T. Farrugia, M. Kinnison, and A. Hendry. 2007. Evolutionary responses to climate change. Conservation Biology 21: 1353-1355.

Skogge, M. and R. Marshall. 2000. A survey of current breeding habitats. Pages 43-56 in Finch, D. and Stoleson, eds. Status, ecology, and conservation of the southwestern willow flycatcher. USDA Forest Service, Rocky Mountain Research Station, RMRS-GTR-060, Ogden Utah.

Sloane, S. A. 2001. Bushtit (*Psaltriparus minimus*). The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/598.

Smallwood, J. A. 1987. Sexual segregation by habitat in American Kestrels (*Falco sparverius*) wintering in south-central Florida: vegetative structure and responses to differential prey availability. Condor 89: 842-849.

Smallwood, J. A. 1988. A mechanism of sexual segregation by habitat in American Kestrels (*Falco sparverius*) wintering in south-central Florida. Auk. 105: 36-46.

Smallwood, J. A. and D. M. Bird. 2002. American Kestrel (*Falco sparverius*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/602.

Smith, B. R. 2003. Conservation Assessment of the Tiger Salamander in the Black Hills National Forest, South Dakota and Wyoming. USDA, Forest Service Technical Report. pp 61.

Smith, D. C. and J. Van Buskirk. 1988. Winter territoriality and flock cohesion in the Black-capped Chickadee, *Parus atricapillus*. Animal Behavior. 36: 466-476.

Smith, F. A., Browning, H. and U. L. Shepard. 1998. The influence of climate change on the body mass of woodrats Neotoma in an arid region of New Mexico, USA. Ecography 21: 140-148.

Smith, J. P. 1979. Cowbird Parasitism on Common Bushtit Nest. The Wilson Bulletin, Vol. 91, No.1 (March 1979), pp. 122-123.

Smith, M. A. and D. M. Green. 2005. Dispersal and metapopulation paradigm in amphibian ecology and conservation: are all amphibian populations metapopulations? Ecography 28: 110-128.115-130.

Smith, M., J. Kelly, and D. Finch. 2006. Cicada emergence in Southwestern riparian forest: influence of wildfire and vegetation composition. Ecological Applications 16: 1608-1616.

Smith, S. M. 1991. The Black-capped Chickadee: behavioral ecology and natural history. Cornell Univ. Press, Ithaca, NY.

Smith, S. M. 1993. Black-capped Chickadee. In The Birds of North America, No. 39 (A. Poole, P. Stettenheim, and F. Gill, Eds.). Philadelphia: The Academy of Natural Sciences; Washington, D.C.: The American Ornithologists' Union.

Smith, W. P. 1991. *Odocoileus virginianus*. Mammalian Species 388: 1-13.

Sogge, M. K.; Paxton, E. H.; Tudor, April A. 2006. Saltcedar and Southwestern Willow Flycatchers: Lessons From Long-term Studies in Central Arizona. Pages 238-241 in Aguirre-Bravo, C.; Pellicane, P. J.; Burns, Denver P.; and Draggan, S., Eds. 2006. Monitoring Science and Technology Symposium: Unifying Knowledge for Sustainability in the Western Hemisphere Proceedings. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, RMRS-P-42CD. Fort Collins, CO.

Spurrier, S. and K. G. Smith. 2007. Desert mistletoe (*Phoradendron californicum*) infestation correlates with blue palo verde (*Cercidium floridum*) mortality during a severe drought in the Mojave desert. Journal of Arid Environments 69: 189-197.

Stanback, Mark T. and Anne A. Dervan. 2001. Within-Season Nest-Site Fidelity in Eastern Bluebirds: Disentangling Effects of Nest Success and Parasite Avoidance. The Auk, Vol. 118: 743-745.

Stebbins, R. C. 1985. A Field Guide to Western Reptiles and Amphibians. Second edition. Houghton Mifflin Company, Boston, Massachusetts.

Stoleson, S. H., R. S. Shook, and D. M. Finch. 2000. Breeding biology of Lucy's warbler in southwestern New Mexico. Western Birds 31: 235-242.

Strehl, C. E. and J. White. 1986. Effects of superabundant food on breeding success and behavior of the Red-winged Blackbird. Oecologia 70: 178-186.

Streubel, D. 2000. American Beaver, from Leege, T. A. 1968. Natural movement of beavers in southeastern Idaho. Journal of Wildlife Management 32: 973-976. Available:http://imnh.isu.edu/digitalatlas/bio/mammal/Rod/Beaver/beaver.htm.

Stromberg, J. 1998. Dynamics of Fremont cottonwood (*Populus fremontii*) and saltcedar (*Tamarix chinensis*) populations along the San Pedro River, Arizona. Journal of Arid Environments 40: 133-155. References and further reading may be available for this article. To view references and further reading you must purchase this article.

Stromberg, J. C., S. J. Lite, T. J. Rychener, L. Levick, M. D. Dixon, and J. W. Watts. 2006. Status of the riparian ecosystem in the upper San Pedro River, Arizona: Application of an assessment model. Environmental Monitoring and Assessment 115: 145-173.

Stromberg, Juliet C., Scott D. Wilkins and James A. Tress. 1993. Vegetation-Hydrology Models: Implications for Management of *Prosopis velutina* (Velvet Mesquite) Riparian Ecosystems. Ecological Applications 3: 307-314.

Stuart, J. N. and J. P. Ward. 2009. *Trachemy geigeae* (Hartweg 1939)- Big Bend Slider, Mexican Plateau Slider, Jicotea de la Mesita Mexicana. In: Rodin, A. G. J., Fritchard, P. C. H., van Dijk, P. P. and others (Eds.) Conservation Biology of Freshwater Turtles and Tortoises: A compilation Project o the IUCN/SSC Tortoise and Freshwater Turtle Specialist Group. Chelonian Research Monographs 5: 032.1-032.12.

Suarez, A. V., K. S. Pfennig, and Scott K. Robinson. Nesting success of a disturbance dependent songbird on different kinds of edges. Conservation Biology 11: 928-935.

Sullivan, B. K. 1993. Distribution of the southwestern toad (*Bufo microscaphus*) in Arizona. Great Basin Naturalist 53: 402-406.

Sullivan, B. K. and P. J. Fernandez. 1999. Breeding activity, estimated age-structure, and growth in Sonoran Desert anurans. Herpetologica 55: 334-343.

Sullivan, J. 1995. *Dipodomys ordii*. In: Fire Effects Information System, [Online]. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer). Available: http://www.fs.fed.us/database/feis/.

Svendsen, G. E. 2003. Weasels and Black-footed ferret. In: Wild Mammals of North America: biology, management, and conservation/ 2nd ed. In Feldhamer, G. A., Thompson, B. C., and J. A. Chapman (Eds). The John Hopkins University Press, MD, USA.

Svensson, S. 1986. Number of pairs, timing of egg-laying and clutch size in a subalpine Sand Martin *Riparia riparia* colony, 19681985. Ornis Scandinavica 17: 221-229.

Swanson, D., B. Graves, and K. Koster. 1996. Freezing tolerance/intolerance and cryoprotectant synthesis in terrestrially overwintering anurans in the Great Plains, USA. Journal of Comparative Physiology 166: 110-119.

Sweitzer, R. A. S. H. Jenkins and J. Berger. Near-Extinction of Porcupines by Mountain Lions and Consequences of Ecosystem Change in the Great Basin Desert. Conservation Biology, Vol. 11, No. 6 (Dec., 1997), pp. 1407-1417.

Szep, T. 1995. Relationship between West African rainfall and the survival of central European Sand Martins *Riparia riparia*. Ibis 137: 162-168.

Terborgh, J. 1989. Where have all the birds gone? Princeton University Press, Princeton, N.J.

Tesky, J. L. 1993. *Castor canadensis*. In: Fire Effects Information System, [Online]. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer). Available: http://www.fs.fed.us/database/feis/.

The Mammals of Texas-online. 1997. Davis, W. B. and D. J. Schmidly. The Mammals of Texas. Lisa Bradley (ed). Available from: http://www.nsrl.ttu.edu/tmot1/Default.htm.

The Heinz Center. 2008. Strategies for managing the effects of climate change on wildlife and ecosystems. Washington, DC. 43 pp.

Thibault, K. M., and J. H. Brown. 2008. Impacts of an extreme climatic event on community assembly. PNAS 105: 3410-3415.

Thompson, C. W. and V. Nolan, Jr. 1973. Population Biology of the Yellow-breasted Chat (*Icteria virens*) in Southern Indiana. Ecological Monographs 43: 145-171.

Thompson, C. W., N. Hillgarth, M. Leu, and H. E. McClure. 1997. High parasite load in house finches (*Carpodacus mexicanus*) is correlated with reduced expression of a sexually selected trait. American Naturalist 149: 270-294.

Tinsley, R. C., and K. Tocque. 1996. The population dynamics of a desert anuran, *Scaphiopus couchii*. Austral Ecology 20: 376-384.

Toland, B. 1984. Unusual predatory and caching behavior of American Kestrels in central Missouri. Journal of Raptor Research 18: 107-110.

Tomlinson, R. E., D. D. Dolton, R. R. George, and Mirarchi R. E. 1994. Mourning Dove. in Migratory shore and upland game bird management in North America. International Association of Fish and Wildlife Agencies, Washington, D.C.

Torchin, Mark E., Kevin D. Lafferty, Andrew P. Dobson, Valerie J. McKenzie and Armand M. Kuris. 2003. Introduced species and their missing parasites. Nature 421: 628-630.

Turner, A. K. and C. Rose. 1989. Swallows and martins an identification guide and handbook. Houghton Mifflin Co., Boston, MA.

Tuttle, M. D. The lives of Mexican Free-tailed bats. BATS Magazine 12: 1.

Ultsch, G. R. 2008. Ecology and physiology of hibernation and overwintering among freshwater fishes, turtles, and snakes. Biological Reviews 64: 435-515.

USACE. 2011. Environmental Assessment for the Middle Rio Grande Bosque Restoration Project. U.S. Army Corps of Engineers, Albuquerque District.

V. Johnson, and J. Harris. Common porcupine. California Department of Fish and Game. 1999. R. Duke, S. Granholm (eds) California's Wildlife, Sacramento, CA. Available: http://sibr.com/mammals/M145.html.

Valdez, E. W. and P. M. Cryan. 2009. Food habits of the hoary bat (*Lasiurus cinereus*) during spring migration through New Mexico. The Southwestern Naturalist 54: 195-200.

Valdez, E. W. Ecology of the Occult Myotis (*Myotis occultus*): Investigating Diet, Ectoparasites, Distribution, and morphological variation. Research Task 83279SW.13.0.

Van Dyke, F. G. Rainer, H. Brocke, H. G. Shaw, B. B. Ackerman, T. P. Hemker, and F. G. Lindsey. 1986. Reactions of mountain lions to logging and human activity. Journal of Wildlife Management 50: 95-102.

Van Oort, H. and K. A. Otter. 2005. Natal nutrition and the habitat distributions of male and female Black-capped Chickadees. Canadian Journal of Zoology 83: 1495-1501.

Van Oort, H., K. A. Otter, K. T. Fort, and Z. McDonnell. 2007. Habitat dominance, and the phenotypic quality of male Black-capped Chickadees. Condor 109: 88-96.

Varland, D. E., E. E. Klaas, and T. M. Loughlin. 1993. Use of habitat and perches, causes of mortality, and time until dispersal in post-fledging American Kestrels. Journal of Field Ornithology 64: 169-178.

Vehrencamp, S. L. Body temperatures of incubating versus non-incubating Roadrunners. Condor 84: 203-207.

Verbeek, Nicolaas A. M. 1975. Northern Wintering of Flycatchers and Residency of Black Phoebes in California. Auk. 92: 737-749.

Vierling, K., L. Lentile, and N. Nielson-Pincus. 2008. Preburn characteristics and woodpecker use of burned coniferous forests. Journal of Wildlife Management 72: 422-427.

Visser, M. E. 2008. Keeping up with a warming world; assessing the rate of adaptation to climate change. Proceedings of the Royal Society, Biological Sciences 275: 649-659.

Vitt, L. J. R. C. van loben Sels, and R. D. Ohmart. 1981. Ecological relationships among arboreal desert lizards. Ecology 62: 398-410.

Voyles, J., L. Berger, S. Young, R. Speare, R. Webb, J. Warner, D. Rudd, R. Campbell, L. Skerratt. 2007. Electrolyte depletion and osmotic imbalance in amphibians with chytridiomycosis. Diseases of aquatic organisms 77: 113-118.

Wade-Smith, J. and B. J. Verts. 1982. *Mephitis mephitis*. Mammalian Species 173: 1-7.

Walsberg, G. E. 1977. Ecology and energetics of contrasting social systems in the phainopepla (Aves: Ptilogonatidae). University of California Publications in Zoology 108: 1-63.

Walsberg, G. E. 1993. Thermal consequences of diurnal microhabitat selection in a small bird. Ornis Scandinavica 24: 174-182.

Walsberg, G. E. 2000. Small mammals in hot deserts: Some generalizations revisited. BioScience 50: 109-120.

Walther, G., S. Beissner, and C. Burga. 2005. Trends in the upward shift of alpine plants. Journal of Vegetation Science 16: 541-48.

Wang, G., Hobbs, N. T., Singer, F. J., Ojima, D. S. and B. C. Lubow. 2002. Impacts of climate changes on elk population dynamics in rocky mountain national park, Colorado, U.S.A. Global Change 54: 205-223.

Warner, R. M. 1985. Interspecific and temporal dietary variation in an Arizona Bat community. Journal of Mammalogy 66: 45-61.

Warner, R. M. and N. J. Czaplewski. 1984. *Myotis volans*. Mammalian Species 224: 1-4.

Wasserman, A. O. 1970. *Scaphiopus couchii* Baird. Catalogue of American Amphibians and Reptiles 85.1-85.4.

Watt, D. J. and E. J. Willoughby. 1999. Lesser Goldfinch (*Carduelis psaltria*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu.bnaproxy.birds.cornell.edu/bna/species/392 doi:10.2173/bna.392.

Weathers, W. W. 1983. Birds of Southern California's Deep Canyon. Univ. of Calif. Press, Berkeley and Los Angeles.

Webster, W. D. and J. K. Jones, Jr. 1982. *Reithrodontomys megalotis*. Mammalian Species 167: 1-5.

Weisheit, A. S. and P. D. Creighton. 1989. Interference by House Sparrows in nesting activities of Barn Swallows. Journal of Field Ornithology 60: 323-328.

Welch, D. K., R. J. Robel and L. B. Fox. 1993. Age Structure and Reproductive Activity in a Kansas Beaver Population Transactions of the Kansas Academy of Science (1903-), Vol. 96, No. 3/4 (Oct., 1993), pp. 186-195.

Weldon, A. J. and N. M. Haddad. 2005. The effects of patch shape on indigo buntings: evidence for an ecological trap. Ecology 86: 1422-1431.

Werner, E. and K. Glennemeier. 1999. Influence of Forest Canopy Cover on the Breeding Pond Distributions of Several Amphibian Species. Copeia 1999: 1-12.

Westmoreland, D., L. B. Best and D. E. Blockstein. 1986. Multiple Brooding as a Reproductive Strategy: Time-Conserving Adaptations in Mourning Doves. Auk 103: 196-203.

Whitaker, J. O., Jr. 1972. *Zapus hudsonius*. Mammalian Species 11: 1-7.

White, D. H., B. Chapman, J. Brunjes, IV, R. Raftovish, Jr., and J. Seginak. 1999. Abundance and reproduction of songbirds in burned and unburned pine forests of the Georgia piedmont. Journal of Field Ornithology 70: 414-424.

Wiebe, K. L. and W. S. Moore. 2008. Northern Flicker (*Colaptes auratus*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/166a.

Williams, B. K., R. C. Szaro, and D. D. Shapiro. 2007. Adaptive Management: The U.S. Department of Interior technical guide. Adaptive management working group, U.S. Department of Interior, Washington, D.C.

Williams, K. S., and C. Simon. 1995. The ecology, behavior, and evolution of periodical cicadas. Annual Review of Entomology 40: 269-295.

Williams, P. H. 1986. Environmental change and the distributions of British bumble bees (Bombus, Latr.). Bee World 67: 50-61.

Williams, P. L. 1988. Spacing behavior and related features of social organization in Northern Orioles of central coastal California. Ph.D. thesis. Univ. of California, Berkeley.

Willis, C. K. R., and R. M. Brigham. 2005. Physiological and Ecological aspects of roost selection by reproductive female hoary bats (*Lasiurus cinereus*). Journal of Mammalogy 86: 85-94.

Willner, G. R., Feldhamer, G. A., Zucker, E. E., and J. A. Chapman. 1980. *Ondatra zibethicus*. Mammalian Species 141: 1-8.

Wilson, Scott D. and Joyce W. Belcher. 1989. Plant and Bird Communities of Native Prairie and Introduced Eurasian Vegetation in Manitoba, Canada. Conservation Biology 3: 39-44.

Wingfield, J. C. 2008. Comparative endocrinology, environment, and global change. General and Comparative Endocrinology 157: 207-216.

Winne, C. T. and M. B. Keck. 2004. Daily activity patterns of whiptail lizards (Squamata: Teiidae: Aspidoscelis): a proximate response to environmental conditions or an endogenous rhythm? Functional Ecology 18: 314-321.

Wolf, B. O. 1991. The reproductive biology and natural history of the Black Phoebe (*Sayornis nigricans* Swainson) in central California. Master's Thesis. California State Univ., San Jose.

Wolf, B. O. 1997. Black Phoebe (*Sayornis nigricans*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology: http://bna.birds.cornell.edu/bna/species/268.

Woodrey, M. S. 1990. Economics of caching versus immediate consumption by White-breasted Nuthatches: the effect of handling time. Condor 92: 621-624.

Woods, C. A. 1973. *Erethizon dorsatum*. Mammalian Species 29: 1-6.

Woodward, B. 1982. Persistence and male mating success in *Bufo woodhousii*. Ecology 63: 583-585.

Woodward, B. 1984. Operational sex ratios and sex biased mortality in *Scaphiopus* (Pelobatidae). Southwestern Naturalist 29: 232-233.

Yard, H. K. 1996. Quantitative diet analysis of selected breeding birds along the Colorado River in Grand Canyon National Park. Unpubl. Rep. U.S.G.S. Colorado Plateau Research Station., Flagstaff, AZ.

Yasukawa, K. and W. A. Searcy. 1995. Red-winged Blackbird (*Agelaius phoeniceus*) In The Birds of North America, No. 184 (A. Poole and F. Gill, Eds.) The Academy of Natural Sciences, Philadelphia and The American Ornithologists' Union.

Young, B., E. Byers, K. Gravuer, G. Hammerson, A. Redder and K. Hall. 2010. Guidelines for Using the NatureServe Climate Change Vulnerability Index. Available: http://www.natureserve.org/prodServices/climatechange/pdfs/Guidelines_NatureServeClimateChangeVulnerabilityIndex_r2.0_Apr10.pdf

Zimmerman, R. 2002. "*Thamnophis sirtalis*" (On-line), Animal Diversity Web. Accessed January 26, 2010 at http://animaldiversity.ummz.umich.edu/site/accounts/information/Thamnophis_sirtalis.html.

Zweifel, R. G. 1968. Reproductive biology of anurans of the arid southwest, with emphasis on adaptation of embryos to temperatures. Bulletin of the American Museum of Natural History 140: 1-64.

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