Vulnerability of riparian obligate species to the interactive effect of fire, climate and hydrological change

Final Report

Agreement #R12PG80468
(FS Agreement #13-IA-11221632-006)

12/30/2014
USDA Forest Service
Rocky Mountain Research Station

BLANK

OVERVIEW

This project had two primary goals: 1) To develop a process for integrating data from multiple sources to improve predictions of climate impacts for wildlife species; and 2) To provide data on climate and related hydrological change, fire behavior under future climates, and species' distributions for use by researchers and resource managers.

We present within this report the process used to integrate species niche models, fire simulations, and vulnerability assessment methods and provide species' reports that summarize the results of this work. Species niche model analysis provides information on species' distributions under three climate scenarios and time periods. Niche model analysis allows us to estimate the degree to which species are likely to be affected by climate impacts to habitat. Fire simulation models were generated under a single climate scenario for the same three time periods and are used, in conjunction with species response profiles, to generate risk maps that represent the potential for negative fire impacts. For a given time period, these risk maps represent the cumulative impact from climate (change area of predicted habitat) and fire (positive, neutral or negative impact on predicted habitat). The output from these modeling efforts can identify the relative impacts among species and habitats, potential refugia, and guide management priorities. However, climate change response is also a function of the innate sensitivity and adaptive capacity of a species' that contributes to their capacity to deal not only with habitat change but changes to resource availability, phenology and biotic interactions. To consider these factors with estimates for changes in habitat, we generated species vulnerability scores using a modified version of a climate change vulnerability assessment system. Through these methods, we provide data on the relative impact of climate change on species via change to habitat availability and the likelihood of climate change resulting in negative impacts via species' vulnerability scores. These measures are then integrated using a risk analysis approach to identify critical issues over time and among species.

We are providing geospatial data layers of climate, fire, biome and predicted species distributions for download at our project website. Links to presentations, data descriptions and zip files containing data layers can be found here. Over the next few months, we will continue to upload webinars and new training tutorials that demonstrate the application of these datasets to new questions and species. Climate and environmental data can readily be used to generate new models for additional species or other applications to describe habitats and future conditions within New Mexico. Initial fire model output is available as raster images and tabulated values that can be used in analyses of wildfire risk or hazardous fuels prioritization or to create additional value-specific wildland fire risk maps when paired with spatial data depicting highly valued resources. Geospatial fire risk layer (merged intensity and fire probability) for species assessments conducted within this study are also available.

Our analysis shows challenges ahead for wildlife species inhabiting the Rio Grande. Climate predictions show a much hotter future with more sporadic precipitation events. Species inhabiting riparian habitats within the Southwest are commonly dependent on the relatively moist and densely vegetation sites. Most species experience declines or substantial shifts in the distribution of suitable habitat over time. Both climate and fire pose large risks to integrity of riparian habitats. The fragmented nature of the remaining habitat along the Rio Grande increases the risk caused by disturbances and highlight the importance of preserving refugia.

Vulnerability of riparian obligate species to the interactive effect of fire, climate and hydrological change

Final report prepared for the Bureau of Reclamation's WaterSMART Program under Applied Science Grants for the Desert Landscape Conservation Cooperative and the Southern Rockies Landscape Conservation Cooperative

Report Authors:

Megan M. Friggens, Ph.D.
Research Ecologist
USFS Rocky Mountain Research Station
meganfriggens@fs.fed.us

Rachel Loehman, Ph.D.
Research Landscape Ecologist
USGS Alaska Science Center
rloehman@usgs.gov

Lisa Holsinger, M.S.
Ecologist
USFS Rocky Mountain Research Station
lmholsinger@fs.fed.us

Deborah Finch, Ph.D.
Program Manager and Supervisory Biologist
USFS Rocky Mountain Research Station
dfinch@fs.fed.us

Recommended Citation:

Friggens, M., Loehman, R., Holsinger, L., and D. Finch. 2014. Vulnerability of riparian obligate species to the interactive effect of fire, climate and hydrological change. Final Report for Interagency Agreement #13-IA-11221632-006. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 213p.

For Further Information Contact:

Megan M. Friggens, Ph.D.
USDA Forest Service
Rocky Mountain Research Station
333 Broadway SE, Suite 115
Albuquerque, NM 87102
meganfriggens@fs.fed.us

CONTENTS

1. INTRODUCTION

Climate change is expected to have multiple direct and indirect impacts on ecosystems in the interior western U.S. (Christensen et al., 2007; IPCC 2013). Global climate predictions for the Southwest include higher temperatures, more variable rainfall, and more drought periods, which will likely exacerbate the ongoing issues relating to wildfire and water allocation in the region (Christensen et al., 2007). Of particular concern to managers are the effects of climate and related changes on riparian habitats, which support a disproportionate amount of the biodiversity in the region. The Rio Grande Basin contains important water sources and habitats for municipalities, agriculture, recreation and wildlife in Colorado, New Mexico and Texas (Figure 1). The Rio Grande Basin also contains critical habitat for a number of riparian dependent species including the endangered southwestern willow flycatcher and the Rio Grande silvery minnow (BOR 2011). Climate change is expected to alter river flows through modified precipitation regimes and higher temperatures that increase evapotranspiration rates (Hansen 1991). Mean annual runoff is projected to decrease from 7.3 to 14.4% by 2050 (BOR 2011). Increased fire potential, though not well defined for riparian habitats, is also an issue where it may favor the establishment of exotic species like Tamarisk and accelerate the degradation of riparian forests (Ellis 2001). In addition, human populations in the region are expected to grow considerably, putting more pressure on natural systems competing for resources. Because competing land and water use leaves the riparian habitats of rivers like the Rio Grande are highly vulnerable to degradation, resource managers need information and tools to identify future conditions under various climate and fire scenarios. By assessing and understanding the impacts of climate and related disturbance change on these important habitats, managers will be able to better focus limited resources on the most critical needs as well as identify opportunities for promoting natural regeneration of riparian woodland and wetland habitats.

This project uses a coupled modeling approach that combines species distribution modeling (bioclimate envelope models, Iverson et al., 2011), fire behavior models (Finney et al., 2012) and vulnerability assessment methods (Bagne et al., 2011) to generate spatially explicit estimates of species vulnerability to the interactive effects of climate change and fire. Bioclimate envelope models are a powerful tool for estimating climate change response of species and recent efforts have extended the applicability of these tools towards generating estimates of species vulnerability (e.g. Iverson et al., 2011). However, these models do not capture the influence of disturbances like fire and invasive species on future habitat suitability potentially leading to overly optimistic projections of species ranges (Iverson et al., 2011). To generate more accurate information on future impacts to species, we have integrated estimates of species habitat requirements, individual components of species vulnerability, and the future fire and climate effects, to create species distribution models and data that represent not only climatic limitations but species’ adaptive capacity. Our goal is to help managers assess alternatives for preventing species’ declines under climate change by providing spatially explicit estimates of habitat conditions under various future climates as well as specific information on the vulnerabilities important for predicting species’ response to climate change.

Study approach

The first step of this project involved summarizing current climate and vegetation projections for Rio Grande habitats within New Mexico (Section 2). We downloaded and processed climate and hydrological data (BOR, 2011 and 2013) based upon the World Climate Research Programme's

(WCRP's) Coupled Model Intercomparison Project, phase 3 (CMIP3), multi-model dataset to generate maps of future climate and hydrological conditions at three future time periods 2030, 2060 and 2090. We also describe expected changes to vegetation using analyses by Rehfedt et al. (2006, 2012). The second phase of this project involved the Large Fire Simulation (FSim) system (Finney et al., 2011) to estimate future fire behavior for the study region (Section 3). Output from these models provide data on burn probability and fire intensity that are used to determine the fire risk of species' habitats under changing climate. Third, we use maximum entropy methods (Maxent 3.3.3, Phillips et al., 2006) to model species' distributions based on climate, hydrological, biophysical, and biome characteristics. We populated these models with species' records from museum and study data, climate layers generated from BOR datasets (2011, 2013), as well as habitat and vegetation data from Rehfeldt et al., 2006. Model output provides information not only on distributions of suitable habitat but the relevance of each environmental variable for predicting species' presence. Fourth, we modified a System for Assessing Vulnerability of Species (SAVS) to climate change (Bagne et al., 2011) to quantify nonmodeled predictors of species' vulnerability to population loss under climate change. These predictors include characteristics such as dispersal capacity, drought sensitivity and potential changes to biotic interactions.

Finally, we integrate these methods and provide estimates of the interactive effect of climate and fire through a series of data products including:

1. Species distribution maps that showing future habitat suitability and fire risk;
2. Species vulnerability scores identifying sources of potential sensitivity or adaptive capacity;
3. Risk matrices that merge measures of exposure (habitat change) and adaptive capacity (vulnerability), which can be used to prioritize species and or management actions (Iverson et al., 2011).

Site Description

The study site contains 11 subbasins within three basins selected based on their proximity (within 50km) to the Rio Grande (Figure 1.1). Three primary streams drain into the Rio Grande within New Mexico: 1) the Rio Chama, the most significant tributary, 2) the Jemez River, and 3) the San Jose/Rio Puerco Drainage. Further south additional, smaller watersheds drain mountains in southern New Mexico though these are often more ephemeral. The habitat contained within these subbasins varies dramatically from the north to south. The headwaters of the Rio Grande constitute the upper Rio Grande watershed and begin in Colorado where the Rio Grande flows as a narrow fast river through a forest mountain landscape (Parcher et al., 2010). Habitat in this alrea is described as constituting big sagebrush shrubland (20%), Pinyon Juniper woodland (16%), Ponderosa Pine woodland (12%), and various mixed conifer woodland types (13%) (NRCS 2008). The Rio Grande-Santa Fe Watershed (HUC8 13020201), begins at Otowi and ends at Chochiti Lake and consists of Pinyon Juniper woodlands (35%), Juniper woodlands and savanna (16%), Semi-desert grasslands (16%), and other mixed conifer woodlands, riparian woodlands and shrublands and grassland habitats (remaining 12%). At Albuquerque, the river broadens and slows as it enters a landscape of sloping flood plains and grassland desert habitats (US Geological Survey 1996). The Rio Grande flows another 515 km to the international boundary between Mexico, New Mexico and Texas. Most of this section is contained within the El Paso-Las Cruces watershed (HUC8 13030102), which flows from Caballo Reservoir dam to El Paso (NRCS 2008) and is characterized as a Chihuahuan desert habitat (33%), Chihuahuan

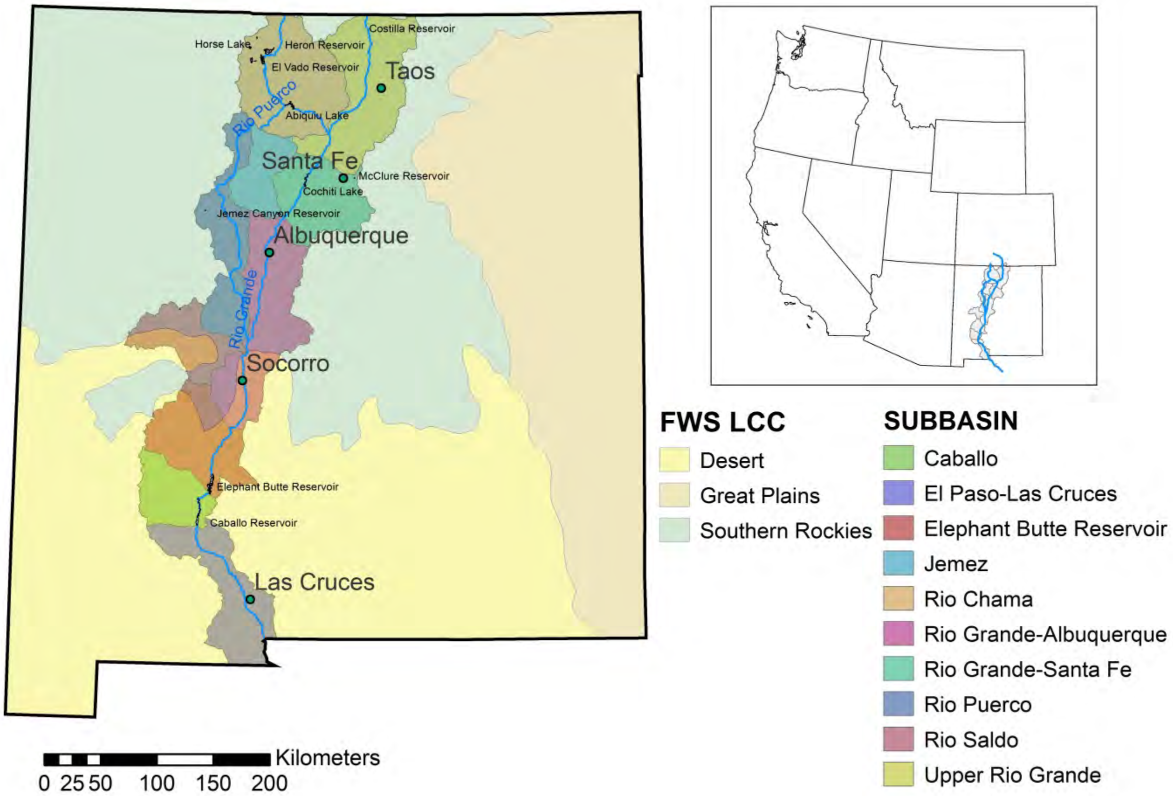


Figure 1.1. Study area consists of 9 subbasins associated with the Rio Grande in New Mexico. Hydrographic data from the USDA, Service Center Agencies, National Hydrography Dataset (NHD)-24 k. Hydrological unit subbasins data from USDA/NRCS National Geospatial Management Center's "Watershed Boundary Data- Hydrological Unit".

semidesert grasslands (18%), dune and sand scrub (15%), and mesquite upland scrub (10%) (Parcher et al., 2010).

The historic flow regime has been greatly impacted by irrigation diversions and agricultural reservoirs in the lower part of the system. Most notably, irrigation activities have increased the relative magnitude and duration of summer peak flows while reducing flow levels associated with snowmelt. In addition, several modifications, including the Rio Grande Reservoir in the headwaters, Cochiti Reservoir (built in 1973) located about 80 miles north of Albuquerque, Elephant Butte reservoir (1916), and Caballo Reservoir (1938) to the south affect natural flow regimes. The southern half of the Rio Grande is severely over allocated. Beginning at Albuquerque and continuing southward, annual potential evaporation can exceed 1000% of annual precipitation (Levings et al., 1998 in Parcher et al., 2010). Appropriated surface water rights along the Rio Grande-Rio Bravo in CO and NM usually exceed mean annual flow.

Land Use: The headwaters typically lie within National Forest Land (Carson, Santa Fe, Cibola, and Gila), whereas the main stem of the Rio Grande flows through large tracts of Bureau of Land Management holdings, as well as the Middle Rio Grande Conservancy District and Elephant

Butte Irrigation District (Figure 1.2). Cultivated cropland or orchards occupy about 7% of the basin. Several reaches of the Rio Grande are surrounded by agriculture lands, particularly in the Española Valley, Middle Rio Grande Valley, and the Mesilla Valley. Other reaches are used extensively for livestock grazing.

Climate: Annual precipitation along the Rio Grande ranges from 1120 mm in the northern reach to less than 200 mm in the southern portion. Approximately 70% of precipitation that drives river flow derives from snowpack in the northernmost part of the Rio Grande basin. The Rio Grande flow is characterized by a spring peak, generally between early April and mid-May, corresponding to spring snow melt followed by a lesser peak in late summer as monsoon storms provides additional precipitation. Fall and winter flows are generally lower (Bullard and Wells 1992).

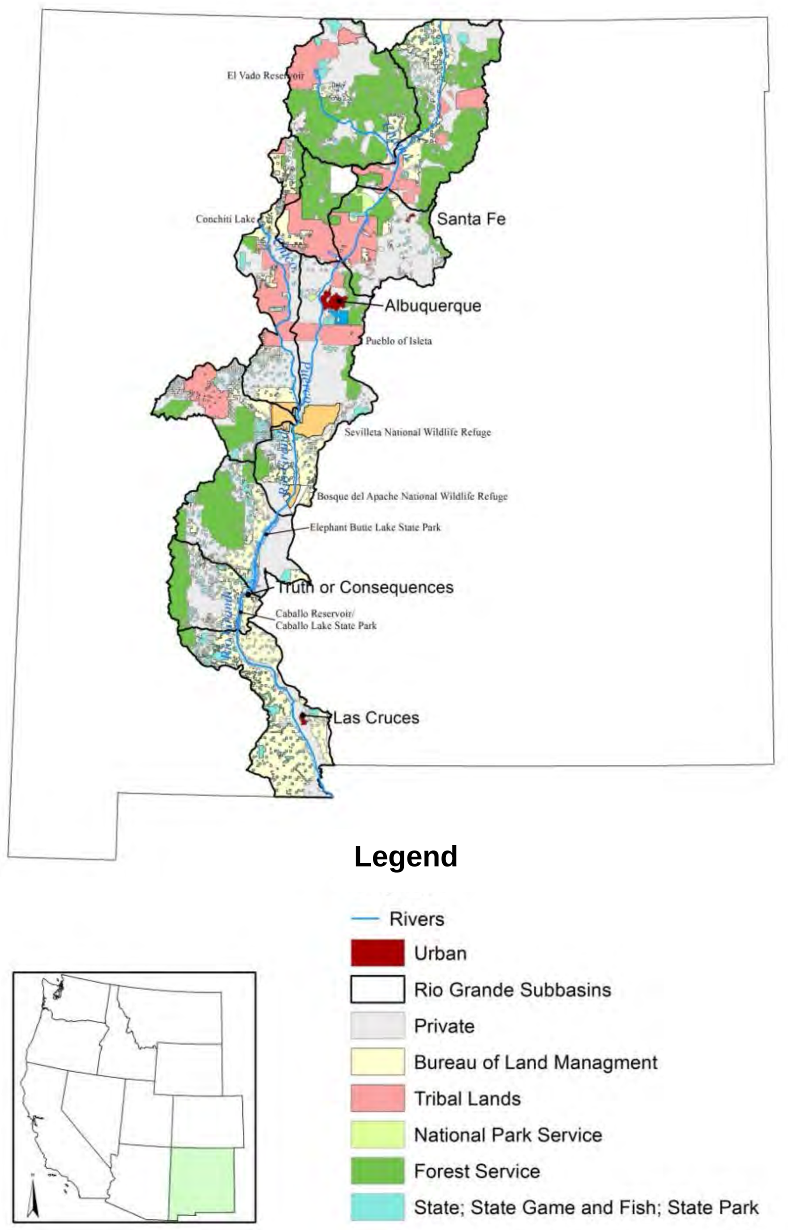


Figure 1.2. Distribution of private, state and federal land within the study area. Forest Service lands predominate in the northern reaches and Bureau of Land Management in the south.

Species Selection

We selected twelve species for this study based on the following criteria:

1. Species relies on the riparian corridor or riparian habitat.
2. Species is of conservation interest (Federal or state status, in decline or invasive).
3. Data is available regarding habitat needs of species.
4. Sufficient data records (minimum 20 localities) are available for niche model analysis.

We also tried to select species whose distribution is tied to a habitat or feature characteristic important to multiple species. We generated a master list of species which was sent out to LCC coordinators, species experts, Region 3 (New Mexico and Arizona) U.S. Forest Service biologists, and cooperators for comment. Ultimately, we focused on 13 species (Table 1.1).

Table 1.1. Species selected for analysis. "I" indicates an introduced species.

Lucy's warbler	*Oreothlypis luciae*
Southwestern willow flycatcher	*Empidonax traillii extimus*
Western yellow-billed cuckoo	*Coccyzus a. occidentalis*
New Mexican meadow jumping mouse	*Zapus hudsonius luteus*
Hispid cotton rat	*Sigmodon hispidus*
Long-legged bat	*Myotis volans*
Occult bat	*Myotis occultus*
Yuma bat	*Myotis yumanensis*
Black-necked gartersnake	*Thamnophis cyrtopsis*
Western painted turtle	*Chrysemys picta belii*
American bullfrog (I)	*Lithobates catesbeiana*
Northern leopard frog	*Lithobates pipiens*

The *Myotis* species do not necessary conform to the criteria used in our initial list but were the only bats to have sufficient location records for analysis. Further, the three species assessed here represent a range of habitat associations that were considered good representatives for other bat species.

Data Delivery

Data generated as part of this project is available from the RMRS website: http://www.fs.fed.us/rm/grassland-shrubland-desert/research/projects/vulnerable-obligate-species/ or by contacting the report author.

Portions of this final report, including species reports, as well as the original data files can be downloaded from the website. Data layers include climate (Section 2), fire behavior (Section 3), and species data layers (Section 4.2). Additional data, presentations and tutorials not presented within this final report are also available on the website. All data and metadata conforms to the guidelines outlined by the Federal Geographic Data committee.

References

1. 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. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO.
2. Bureau of Reclamation, 2013. Downscaled CMIP3 and CMIP5 Climate and Hydrology Projections: Release of Downscaled CMIP5 Climate Projections, Comparison with preceding Information, and Summary of User Needs, prepared by the U.S. Department of the Interior, Bureau of Reclamation, Technical Services Center, Denver, Colorado. 47pp.
3. Bureau of Reclamation. 2011. West-wide climate risk assessments: bias-corrected and spatially downscaled surface water projections, prepared by the U.S. Department of the Interior, Bureau of Reclamation, Technical Services Center, Denver Colorado, March 2011, 138 pp.
4. Christensen, J. R., et al., 2007: Regional climate Projections. In: Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovermental Panel on Climate Change [Solomon, S., D. Quin, M. Manning, et al. (eds.)]. Cambridge University Press, Cambridge, New York, NY, USA.
5. Ellis, L.M. 2001. Short-term response of woody plants to fire in a Rio Grande riparian forest, central New Mexico, U.S.A. Biological Conservation 97: 159-170.
6. Finney, M.A., C.W. McHugh, I.C. Grenfell, K.L. Riley, and K.C. Short. 2011. A Simulation of Probabilistic Wildfire Risk Components for the Continental United States. Stochastic Environmental Research and Risk Assessment 25:973-1000.
7. Foden, W.B. et al. 2013. Identifying the world's most climate change vulnerable species: a systematic trait-based assessment of all birds, amphibians, and corals. PLoS ONE. 8, doi:10.1371/journal.pone.0065427.
8. Friggens, M., and M. Mathews. 2013. Risk-based framework and risk case studies. *In* Vose, Peterson, Patel-Weyn National Climate Assessment – Forest Sector Technical Report.
9. Friggens, M., Bagne, K., Finch, D., Falk, D., Triepke, J. and A. Lynch. 2013. Review and Recommendations for Climate Change Vulnerability Assessment Approaches with Examples from the Southwest. Gen. Tech. Rep. RMRS-GTR-309. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO.
10. Friggens, M.M., Finch, D. M, et al. 2013. Vulnerability of Species to climate change in the southwest: terrestrial species of the middle Rio Grande. Gen. Tech. Rep. RMRS-GTR-306. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO.
11. Gardali, T., N.E. Seavy, R.T. DiGaudio, and L.A. Comrack. 2012. A climate change vulnerability assessment of California's at-risk birds. PLos ONE 7 doi:10.1371/journal.pone.0029507
12. Hanson, R.L., 1991, Evapotranspiration and Droughts, *In* Paulson, R.W., Chase, E.B., Roberts, R.S., and Moody, D.W., Compilers, National Water Summary 1988-89--Hydrologic Events and Floods and Droughts: U.S. Geological Survey Water-Supply Paper 2375, p. 99-104.
13. IPCC, 2013. Intergovernmental Panel on Climate Change (IPCC) Fifth Assessment Report (AR5). P. Romero-Lankao, J. B. Smith, et al. 2014. Chapter 26. North America. http://ipcc-wg2.gov/AR5/images/uploads/WGIIAR5-Chap26_FGDall.pdf
14. Iverson et al. 2011. Lessons learned while integrating habitat, dispersal, disturbance, and life-history traits into species habitat models under climate change. Ecosystems 14: 1005-1020.

15. Natural Resources Conservation Service. 2008. Upper Rio Grande Water Shed: Hydrologic Unit Code 13020101 U.S. Department of Agriculture Rapid Assessment. Available at: http://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/nrcs144p2_060580.pdf
16. Notaro, M., et al. 2012. Projected vegetation changes for the American SW: Combined dynamical modeling and bioclimate envelope approach. Ecological Applications.
17. Parcher, J.W. Woodward, D.G. and R.A. Durall. 2010. A Descriptive Overview Of The Rio Grande-Rio Bravo Watershed. Journal of Transboundary Water 1: 159-178. Available at: Resourceshttp://wrri.nmsu.edu/publish/journal_of_transboundary/Parcher.pdf
18. Phillips, S. J., Anderson, R. P, and R. E Schapire. 2006. Maximum entropy modeling of species geographic distributions. Ecological Modelling 190: 231-259.
19. Rehfeldt, G.E., Crookston, N.L., Warwell, M.V., and J.S. Evans. 2006. Empirical Analyses of Plant-Climate Relationships for the Western United States. International Journal of Plant Sciences 167:1123-1150.
20. Young, B., E. Byers, K. Gravuer, K. Hall, G. Hammerson, and A. Redder. 2011. Guidelines for using the NatureServe climate change vulnerability index. NatureServe, Arlington, Virginia.

2. DATA LAYERS

See Species Analysis (Section 4) for details on presence records.

2.1 Future Climate Conditions

Climate data used for this analysis is based upon the World Climate Research Programme's (WCRP's) Coupled Model Intercomparison Project phase 3 (CMIP3) multi-model dataset. We downloaded climate data based on three general circulation models (GCMs) under the A2 emission scenario from the downscaled CMIP3 Climate and Hydrology Projections archive at http://gdo-dcp.ucllnl.org/downscaled_cmip_projections/ (Reclamation 2011). Data was downloaded for historic (1950-2013) and future (2020-2100) time periods. The models were: 1) Canadian Center for Climate Modeling and analysis (CCC) using the CGCM3.1 model, 2) Met office, Hadley Centre (HAD), using the HadCM3 model, and (3) Geophysical Fluid Dynamics laboratory (GFDL) using the CM2.1 model (Fig. 2). Data is provided at the resolution of 1/8 degree. We consider these GCMs to represent the range of scientific approaches used by climate modeling communities (Towes and Allen 2009). The SRESA2 emission scenario is considered the high emission pathway where technological change and economic growth more fragmented, slower, higher population growth (Emissions pathway descriptions and illustrations are available online at the IPCC Data Distribution Centre). At the time of this analysis, observed trends in CO_2 and other emissions had exceeded estimates under the SRES A2 higher emission scenarios (Raupach et al., 2007; Ganguly et al., 2009).

We selected GCMs to represent the range of predicted conditions that might occur within NM under a scenario of increasing emissions. Huntley et al., 2006 in a similar analysis selected one model from each model group ECHAM4, HadCM3, and GFDL to represent global dry, medium and wet projections, respectively (e.g. Cubasch et al., 2001). In general, assessments of individual climate model performance as determined by comparing observed versus modeled historical trends shows considerable variability according to the parameter of interest. Importantly those models that most accurately predicted 20th century precipitation, HadCM3, HadGEM, MRI-CGCM3.2 and MIROC3.2 predict different changes for future conditions in the southwestern U.S. HadCM3 was found to slightly overestimate winter precipitation and GFDL2.1 led to the greatest over estimation (McAfee et al., 2011). Overall the Hadley models, including HadCM3 had the smallest Precipitation error for the western U.S. though more local estimations had considerable positive bias. For precipitation during the monsoon season (July through September), CGCM2.3 project small decreases whereas HadCM3 a slight increase in monsoon Precipitation (Garfin et al., 2010 and McAfee et al., 2011). In an analysis of model capacity to accurately describe southwestern precipitation, Garfin et al., (2010) the models selected for this analysis were ranked as intermediate. Arritt et al., (2000) found the second generation HadCM2 accurately predicted onset and characteristics of Precipitation. When comparing predictions of mean summer temperature for GFDL and CGCM3 models, HadCM3 tended to predict greater increases in Temperature (hotter than observed) and GFDL predicted lower temperatures than observed (Salazar et al., 2007). Others report HadCM3 and ECHAM5 (not considered in our analysis) perform best with Southwest temperature, precipitation and anomalies (Notaro et al., 2012; Dominguez et al., 2010). Notaro reports the top five performing models as ECHAM3, HadCM3, CCSM3.0, HadGem1, and CNRM-CM3. For this analysis, relative predictions among the models show HadCM3 generated intermediate increases in mean temperature (+4.6 by 2090) and slight increasing annual precipitation by 2090, GFDL predicts the greatest increase in mean temperature (+5.16) and greatest decline in precipitation, and

CGCM3.1 generates intermediate temperature increases (+4.82) and moderate declines in annual precipitation (Figure 2.1).

We downloaded monthly values for two time periods, 1950-2013 and 2020-2100, for the following variables: Total Precipitation, mean maximum daily Temperature, mean minimum daily Temperature, soil moisture content, state 1st day of month , snow water equivalent in snow pack, state 1st day of month, stream flow (surface runoff + base flow), actual ET , natural vegetation PET, and open water surface PET. From temperature and precipitation variables, we calculated 19 bioclimate variables (Table 2.1, O'Donnell and Ignizio 2012; also see workslim.org/bioclim) commonly used for species distribution models for four time periods (1970-2013 "current", 2020-2040 "2030", 2050-2070 "2060", and 2080-2100 "2090")(SAS 9.3).

Table 2.1. Climate data layers produced for the current study. For each variable listed, images of averaged values for current (1970-2013), 2020-2040, 2050-2070, and 2080-2100 time periods are available from the project website.

Climate data layers (20-year averages)

BIO1 = Annual Mean Temp
BIO2 = Mean Diurnal Range (Mean of monthly (max temp - min temp))
BIO3 = Isothermality (BIO2/BIO7) (* 100)
BIO4 = Temp Seasonality (standard deviation *100)
BIO5 = Max Temp of Warmest Month (°C)
BIO6 = Min Temp of Coldest Month (°C)
BIO7 = Temp Annual Range (BIO5-BIO6)
BIO8 = Mean Temp of Wettest Quarter (°C)
BIO9 = Mean Temp of Driest Quarter (°C)
BIO10 = Mean Temp of Warmest Quarter (°C)
BIO11 = Mean Temp of Coldest Quarter (°C)
BIO12 = Annual Precip (mm)
BIO13 = Precip of Wettest Month (mm)
BIO14 = Precip of Driest Month (mm)
BIO15 = Precip Seasonality (Coefficient of Variation)
BIO16 = Precip of Wettest Quarter (mm)
BIO17 = Precip of Driest Quarter (mm)
BIO18 = Precip of Warmest Quarter (mm)
BIO19 = Precip of Coldest Quarter (mm)
Aridity Index
Annual and seasonal Evapotranspiration (mm)
Annual and seasonal Potential Evap Natural Vegetation (mm)
Annual and seasonal Potential Evap Water (mm)
Annual and seasonal Runoff (mm)
Annual and seasonal Soil Moisture Content
Annual and seasonal Snow Water Equivalent (only relevant to northern areas)

In addition we calculated and aridity index (P/PET) where an AI<0.2 indicates arid/hyperarid and AI>0.5 indicating dry conditions (http://www.cgiar-csi.org/wp-content/uploads/2012/11/Global-Aridity-and-Global-PET-Methodology.pdf). We then created a series of raster layers for each time period and each variable in ArcMap 10.1.

Projected changes for the Rio Grande Basin show increasing mean and max temperatures for all months from 1990's levels (Reclamation 2011; Fig. 2.1). Overall, precipitation is expected to increase through the 2020's and 2050's but decline during the later decades of the century (Reclamation 2011). Mean annual precipitation is expected to decrease thought the extent and direction of change varies throughout the year (Fig. 2.1). The models used in this analysis show different patterns for Precipitation: HadCM3.1 predictions by 2090 show increasing summer Precipitation but the greater declines for winter months, whereas GFDL and CGCM 3.1 show more consistent declines across the entire year. The BOR hydrological projections rely on a VIC hydrological model that estimates water balance at each model grid cell. The VIC model applied in the BOR water projections gives Precipitation one of two fates: runoff or evapotranspiration. Runoff is also expected to show some increase in the first half of the century but declines starting in the 2050's (Reclamation 2011). Winter season runoff shows increasing trends whereas spring summer levels generally decrease. The latter corresponds with expected declines in snow water equivalent from 1990's levels (Reclamation 2011).

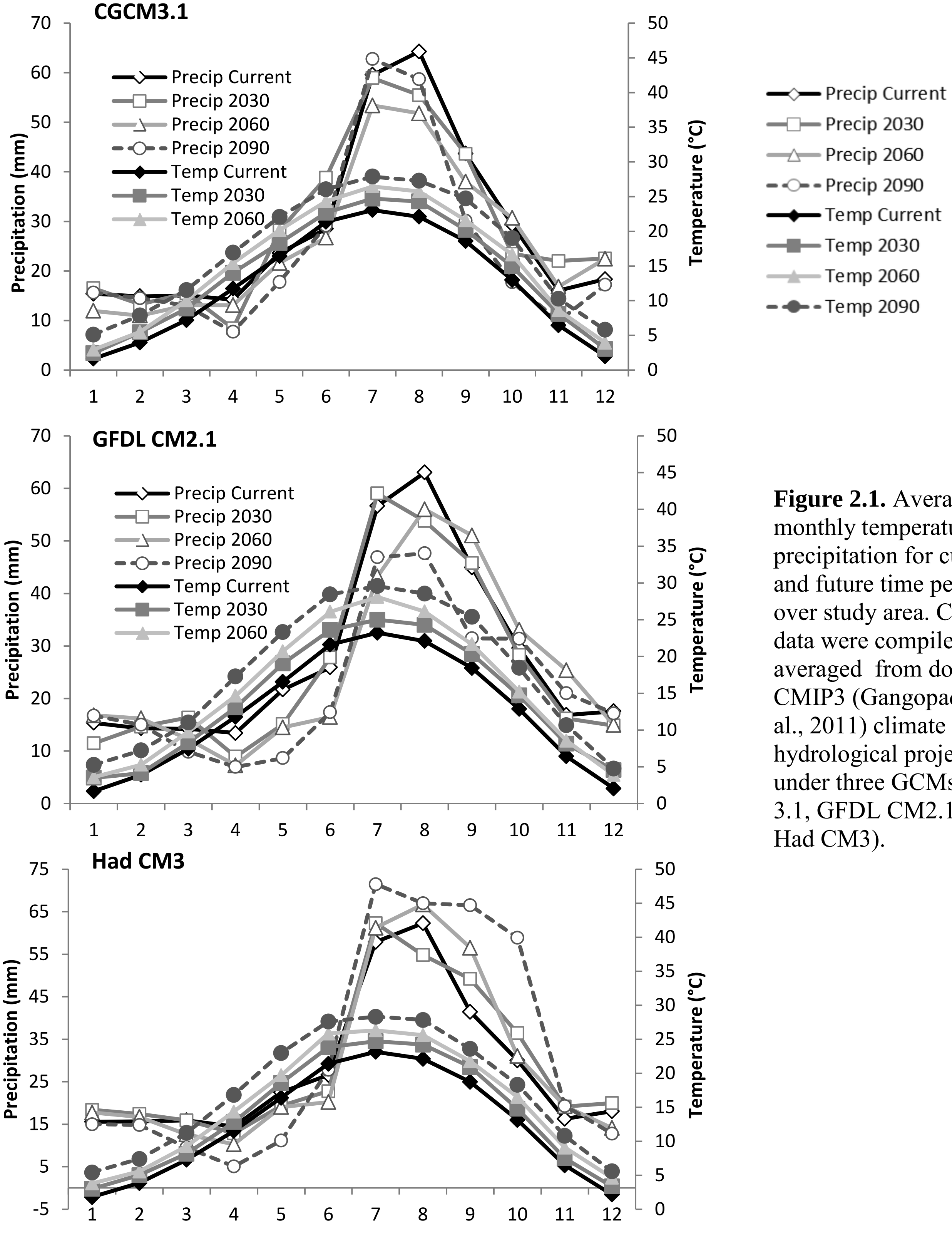


Figure 2.1. Average monthly temperature and precipitation for current and future time periods over study area. Climate data were compiled and averaged from downscaled CMIP3 (Gangopadhyay et al., 2011) climate and hydrological projections under three GCMs (CGCM 3.1, GFDL CM2.1, and Had CM3).

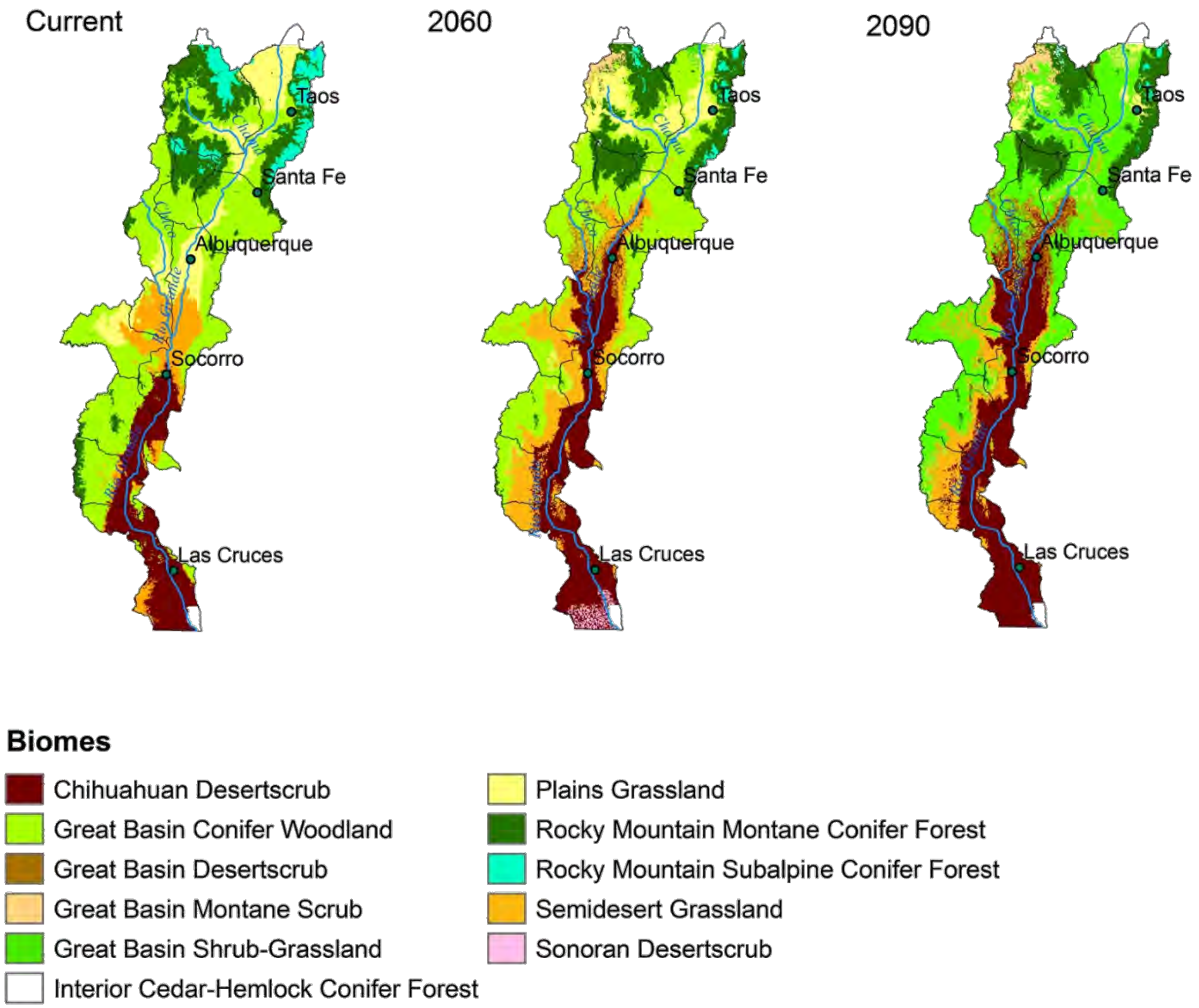
2.2 Biophysical variables

Elevation layers were derived from 1 degree digital elevation model for Colorado, New Mexico, Utah and Arizona, from Data Basin (http://app.databasin.org) originally produced by the United States Geological Survey. We used the slope tool in ArcMap 10.1 to generate a slope data layer from the elevation data.

Creeks and river features were extracted from the National Hydrography Dataset -24k (gway_1933069_03_NHD24k). We created a vector file of all named and perennial creek and river features (derived from nhd24kst) and merged it with a layer representing reservoirs and ponds (derived from nhd23kwb) greater than 0.2 km^2 in area. We calculated Euclidean distance (ArcMap 10.1) in kilometers from the water features and used this to create a layer representing distance to water.

2.3 Habitat variables

Raster layers representing 25 biomes at four distinct time periods (current, 2030, 2060, and 2090) were downloaded from http://forest.moscowfsl.wsu.edu/climate/publications.php (Rehfeldt et al., 2006, 2012). Biome classifications and change over time are shown in Figure 2.2. Rehfeldt used random forests to predict distribution of biomes for contemporary and future (2030, 2060, 2090) time periods under three GCM and two emission scenarios. Projections were generated at a 1km resolution. Final results are presented as consensus rasters where pixels were classified as a biome when at least 4 model runs supported that outcome.


Current
2060
2090
Taos
Santa Fe
Albuquerque
Socorro
Las Cruces
Chama
Rio Grande
Biomes
Chihuahuan Desertscrub
Great Basin Conifer Woodland
Great Basin Desertscrub
Great Basin Montane Scrub
Great Basin Shrub-Grassland
Interior Cedar-Hemlock Conifer Forest
Plains Grassland
Rocky Mountain Montane Conifer Forest
Rocky Mountain Subalpine Conifer Forest
Semidesert Grassland
Sonoran Desertscrub

References

1. Arritt, R. W., Goering, D.C., and C. J. Anderson. 2000. Geophysical research letters 27: 565-568.
2. Bureau of Reclamation. 2013. Downscaled CMIP3 and CMIP5 Climate and Hydrology Projections: Release of Downscaled CMIP5 Climate Projections, Comparison with preceding Information, and Summary of User Needs, prepared by the U.S. Department of the Interior, Bureau of Reclamation, Technical Services Center, Denver, Colorado. 47pp.
3. Bureau of Reclamation. 2011. West-wide climate risk assessments: bias-corrected and spatially downscaled surface water projections, prepared by the U.S. Department of the Interior, Bureau of Reclamation, Technical Services Center, Denver Colorado, March 2011, 138 pp.
4. Cubasch, U. Meehl, G.A., Boer, G.J., Stouffer, R.J., Dix, M., Noda, A., Senio, C.A., Raper, S. and K.S. Yap. 2001. Projections of future climate change. In Climate Change 2001: The scientific basisi (eds J.T. Houghton, Y. Ding, D.J. Griggs, M. Noguer, P. J. van der Linden, X. Dai, K. Maskell and C. A. Johnson), pp 525-5882. Cambridge: Cambridge University Press.
5. Dominguez, F., J. Cañon, and J. Valdes. 2010. IPCC-AR4 climate simulations for the Southwestern US: the importance of future ENSO projections. Climatic Change 99: 499- 514.
6. Gangopadhyay S. and T. Pruitt. 2011. West-Wide Climate Risk Assessments: Bias-Corrected and Spatially Downscaled Surface Water Projections. Water and Environmental Resources Division and Water Resources Planning and Operations Support Group, Technical Memorandum No. 86-68210–2011-01
7. Ganguly, A.R., K. Steinhauser, D.J. Erickson III, M. Branstetter, E.S. Parish, N. Singh, J.B. Drake, and L. Buja. 2009. Higher trends but larger uncertainty and geographic variability in 21st century Temperature and heat waves. PNAS 106: 15555-15559.
8. Garfin, G., J. Eischeid, M. Lenart, K. Cole, K. Ironside, and N. Cobb. 2010. Downscaling Climate Projections in Topographically Diverse Landscapes of the Colorado Plateau in the Arid Southwestern U.S. The Colorado Plateau IV: Shaping Conservation Through Science and Management. Charles Van Riper III, Brian F. Wakeling, and Thomas D. Sisk (eds.), University of Arizona Press, pp. 21-44.
9. Hanson, R.L. 1991. Evapotranspiration and Droughts *In* Paulson, R.W., Chase, E.B., Roberts, R.S., and Moody, D.W., Compilers, National Water Summary 1988-89--Hydrologic Events and Floods and Droughts: U.S. Geological Survey Water-Supply Paper 2375: 99-104.
10. McAfee, S. A., Russell, J.L., and P.J. Goodman. 2011. Evaluating IPCC AR4 cool-season Precip simulations and projections for impacts assessment over North America. Climate Dynamics 37: 2271-2287.
11. Raupach, M.R., G. Marland, P. Ciais, C.L. Quéré, J.G. Canadell, G.t Klepper, and C.B. Field. 2007. Global and regional drivers of accelerating CO_2 emissions. PNAS 104:10288-1029.
12. Rehfeldt, G.E., Crookston, N.L., Saenz-Romero, C., and E.M. Campbell. 2012. North American vegetation model for land-use planning in a changing climate: A solution to large classification problems. Ecological Applications 22: 119-141.
13. Rehfeldt, G.E., Crookston, N.L., Warwell, M.V., and J.S. Evans. 2006. Empirical Analyses of Plant-Climate Relationships for the Western United States. International Journal of Plant Sciences 167:1123-1150.

14. Salazar, B.S., Finley, A.O., HAmmerling, D., Steinsland, I. Wang, X. and P. Delamater. 2011. Comparing and blending regional climate model predictions for the American Southwest. Journal of Agricultural, Biological and Environmental Statistics 16: 586-605.
15. Stoleson, S.H., Chook, R. S., and D. M. Finch. 2000. Breeding biology of Lucy's warbler in southwester New Mexico. Western Birds 31: 235-242.
16. Toes, M.W. and D. M. Allen. 2009. Evaluating different GCMs for predicting spatial recharge in an irrigated arid region. Journal of Hydrology 374: 265-281.
17. van Riper, C., III., Hatten, J.R., Giermoakowski, J.T. Mattson, D., Holmes, J.A., Johnson, M.J. et al. 2014. Projecting climate effects on birds and reptiles of the Southwestern United States: U.S. Geologica Survey Open-File Report 2014-1050, 100p.
18. Webb, R.H., Leacke, S.A., and R.M. Turner. 2007. The ribbon of green: change in riparian vegetation in the southwestern United States. The University of Arizona Press, Tucson. 462 pp.

3. FIRE ANALYSIS: FIRE SIMULATION AND FIRE TYPES

Wildfire Risk Assessments provide a framework for estimating fire impacts to habitats and species (Calkin et al., 2010; Scott et al., 2013). These frameworks are based on three components: coverage for highly valued resources, estimate burn probability, and fire response functions. For this study, highly valued resources are represented as predicted habitat for our study species at three future time periods. Burn probabilities are estimated using fire simulation programs (Section 3.1) and were used to define fire types (Section 3.2). Fire response functions translate fire effects into net value change to described resource. In this case, species response to fire types was determined from a review of the literature (Section 3.3).

3.1 Large Scale Fire Simulations (FSim)

Wildland Fire Simulation Modeling. Our objective was to quantify the likelihood for wildfire and its potential intensity to evaluate possible changes to the fire disturbance regime under future climate scenarios for the Rio Grande study area. Of the fire modeling systems that simulate natural variability and fire probability (Scott et al., 2013), we chose the geospatial Fire Simulation (FSim) system (Finney et al., 2011) because it: simulates large fires (i.e. which typically account for 95% of area burned) on an annual basis; integrates variability in fuel moisture, wind speed and direction throughout the fire season; incorporates the effects of fire suppression; and because model inputs and calibration for the study area were developed and available from the 2014 national Fire Program Analysis (FPA) effort. FSim uses current land-cover data, historical wildfire data and weather data for defined geographical areas (Fire Planning Units – FPUs, Figure 3.1) and simulates fires in those FPUs for thousands of years to capture the full range of variation across the complex topology of fuel patterns, weather sequences, and fire spread. Output from model simulations includes an overall burn probability (Figure 3.2), relative burn probabilities at six flame lengths (Figure 3.3), and the mean fireline intensity (Figure 3.4).

To simulate FSIM for the Rio Grande study area, we used the model inputs developed from the FPA project for three distinct geographic areas or Fire Planning Units (FPU) including the Northern, Central and Southern New Mexico FPUs. Geospatial land-cover inputs consisted of surface fuel characteristics (i.e. fire behavior fuel model), canopy fuel characteristics (canopy base height, canopy bull density), vegetation (canopy cover and height), and topography (slope, aspect, elevation) – which all originated from the Landfire Refresh project and were resampled from 30-m to a 270-m resolution. Historical wildfire occurrence data needed for model simulations consisted of fire records from the years 1992-2010 (Short 2014); these data in conjunction with weather records were used in the model to determine the probability of large fire starts on any given data.

Weather inputs required by FSim include hourly wind-speed during the burning period, daily Precipitation amount and duration, maximum and minimum Temp, maximum and minimum relative humidity, and 1300-hour observations for relative humidity and Temp.

Figure 3.1. Fire Planning Units for the Western U.S. The current study estimated fire behavior for three future time periods for 001, 005 and 003 within New Mexico.

With the exception of windspeed, these eight weather variables are integrated to calculate the Energy Release Component (ERC) for fuel model G (Cohen and Deeming 1985) - which is a metric that essentially reflects short and long term changes in fuel moisture content caused by Precipitation and changes in Temperature and humidity and can represent the influence of fuel moisture on fire behavior (Finney et al., 2011). More specifically, FSim uses ERC to determine, using logistic regression, the probability that a large fire will ignite on a given day, and then ERC and windspeed are both used to determine the behavior of those fires that do ignite (i.e. fire spread and size). To model future weather, we obtained climate predictions developed using the Multivariate Adaptive Constructed Analogs (MACA) method (Abatzoglou and Brown 2011) which downscales model output from various global climate models (GCMs) of the Coupled Model Inter-Comparison Project 5 (CMIP5). Due to time constraints, we only simulated future conditions based on one GCM and chose the GFDL-ESM-2m because of its demonstrated success in historical climate simulation (Dunne et al., 2012, Scheffield et al., 2013). We also chose to model future conditions predicted by the CMIP5 8.5 Representative Concentration Pathway (RCP) – a scenario with the highest estimated Temperature increases (4.0 to 6.1°C by 2100; Rogelj et al., 2012) but the one that appears increasingly most likely to occur (Peters et al., 2013). We modeled three projected time periods to capture potential climate conditions over the near-term (2020-2040), mid-century (2050-2070) and end-of-century (2080-2100).

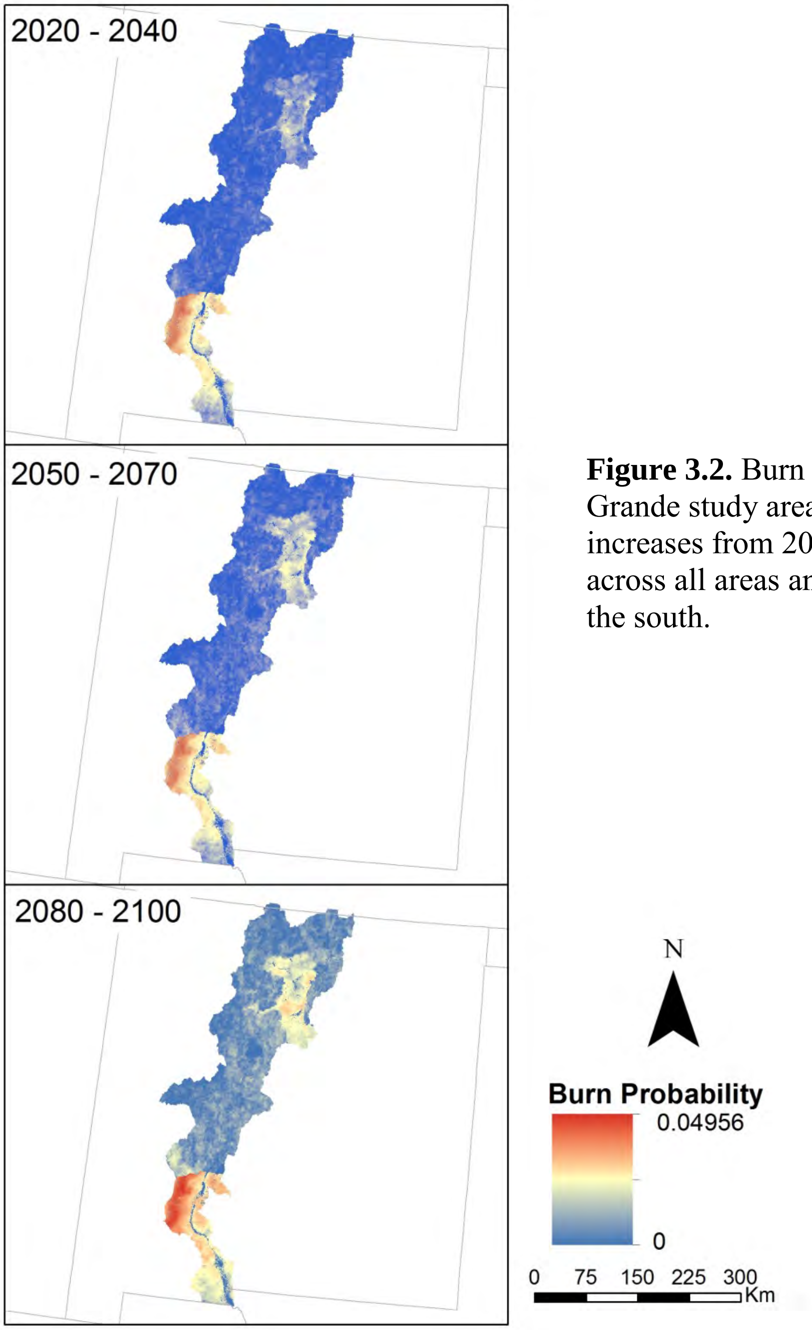


Figure 3.2. Burn probability over the Rio Grande study area shows significant increases from 2020-2040 to 2080-2100 across all areas and especially high values in the south.

Downscaled weather variables available for the RCP85 scenario from GFDL-ESM-2m GCM included daily minimum and maximum temperature, precipitation amount, and relative humidity; however data for wind-speed and Precipitation duration were not readily available. To appropriately model weather in FSim, we needed to maintain the correlations among Precipitation (both amount and duration), Temperature and relative humidity. We decided to preserve these relationships by starting with recent weather observations (1992-2010) from RAWS weather stations for each FPU (Jemez for Northern New Mexico; Grants for Central New Mexico; Dripping Springs for Southern New Mexico) and calculated monthly differences between observed records and future predictions for each weather variable and then assumed these proportional changes on a daily basis to develop future weather data streams. To estimate

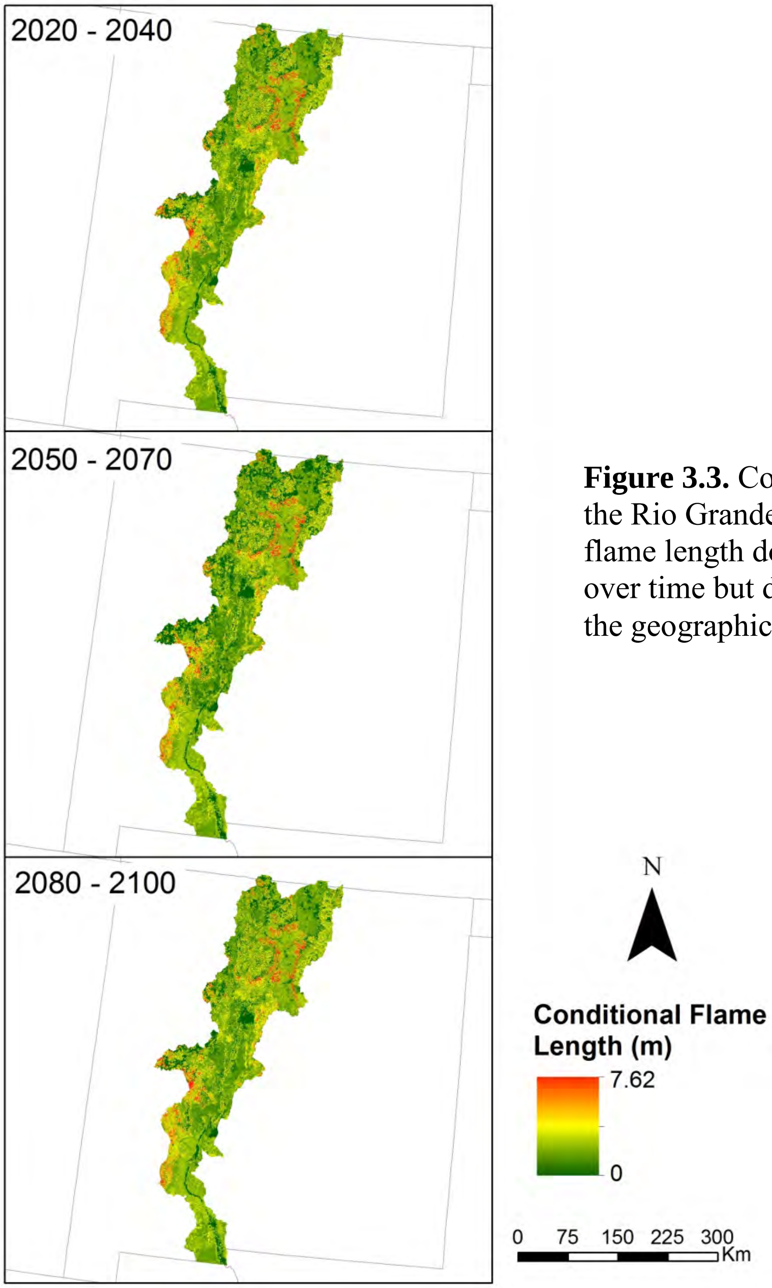


Figure 3.3. Conditional flame length across the Rio Grande study area. Conditional flame length does not change considerably over time but does vary substantially over the geographic area.

future Precipitation duration (currently unavailable from downscaled climate projections), we simply applied the same relative changes calculated from Precipitation amount to determine the Precipitation duration variable. For wind-speed, a variable also absent from downscaled projections, we could only use historic wind data and therefore model results assume that winds do not change in the future. We used the program Fire Family Plus 4.1 to calculate daily ERCs from these variable estimates for each FPU.

To obtain stable estimates of fire behavior across each landscape, fire ignition and growth was

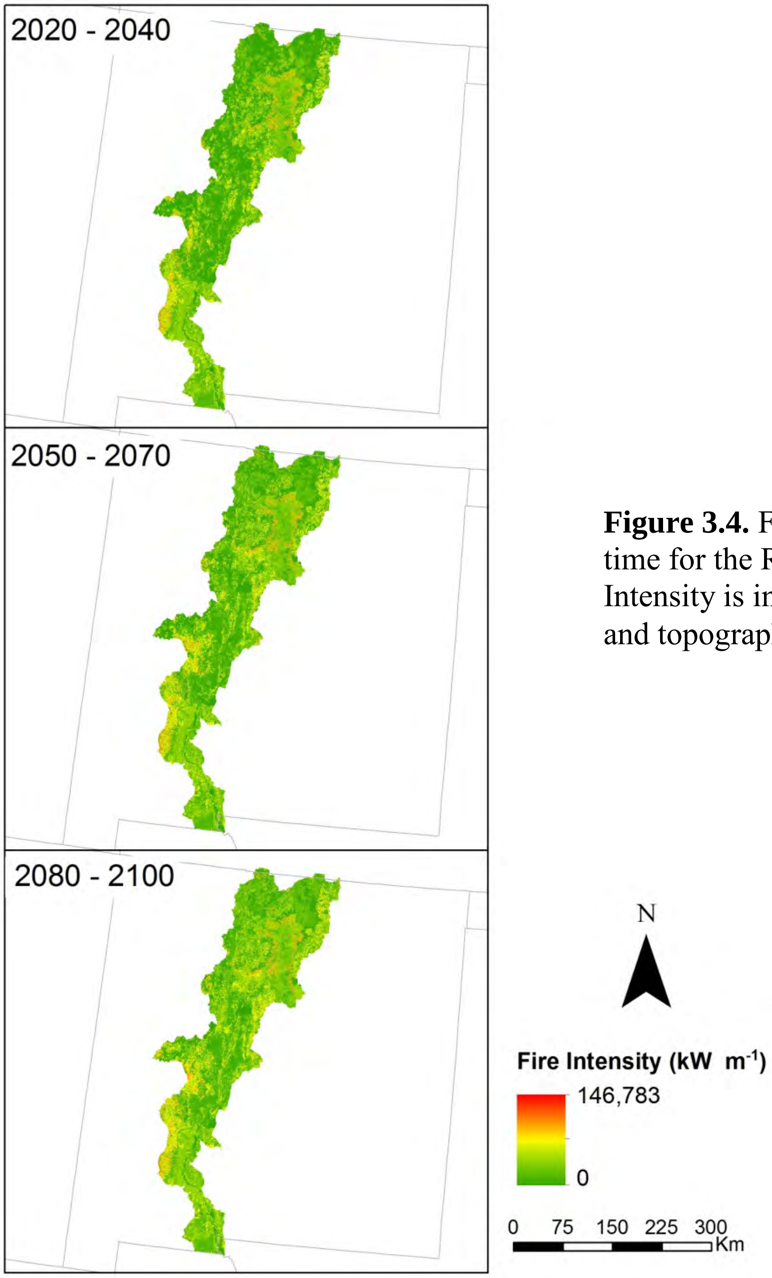


Figure 3.4. Fire intensity increases over time for the Rio Grande study area. Intensity is influenced by fuels, weather and topography.

simulated by FSim for 10,000 potential annual weather scenarios (i.e.'years') for each FPU.

We ran simulations for this extended length because FSim needs sizeable samples of potential scenarios to estimate the probability for the rare events of large fires. That is, FSim was not projecting 10,000 years into the future but generating a large sample of possible fire seasons based on projected future climate conditions. We assumed that current levels of fire suppression would continue into the future, which for the FSim model was determined in the FPA effort to be equivalent to a suppression factor of 2 for Northern New Mexico, 5 for Central New Mexico, and 3 for Southern Mexico. To simulate large fires, FSim requires estimates of live fuel moisture values for the upper percentiles (80th, 90th and 97th) of ERC, and we used the same values used in the FPA project based on estimates by Finney et al., (2011). In addition, FSim uses spatial estimates of the distribution of ignitions based on fire size from the 1992-2010 period, and we used fire ignition layers from the FPA project which for the Northern and Central New Mexico FPUs, was based on ignition distributions for fires larger than 50 acres, and for Southern New Mexico, for fires larger than 300 acres.

3.2 Fire Impacts to Species' Habitat

For the purpose of estimating impacts to species' habitat under climate and fire, we categorized FSim output into 5 fire type classifications:

1: shrub with torching
2: shrub without torching
3: forest with torching
4: forest without torching
5: grass or non-veg

To create these data files, we developed vegetation type layer from Landfire biophysical settings (BpS) layer (http://www.landfire.gov/NationalProductDescriptions20.php) that described vegetation type as forest, shrub, grass and non-vegetation. Conditional Flame Length (CFL) layers for each time period were classified into four categories (taking after Calkin et al., 2010):

1. Low = 0-0.61 -> 0.62;
2. Mod=0.61-1.83 -> 1.83;
3. High=1.83-3.66 -> 3.66;
4. Very High=3.66-7.62 -> 7.62

Using the Canopy Base Height layer from Landfire, we estimated potential torching behavior by defining torching as where Canopy Base Height $\leq$ classified CFL. For instance if CBH was 0.54 and CFL reclassified is 1.83 (i.e. flame length exceeds vegetation height), torching was considered possible. Fire Type by Vegetation Type layers were created by combining vegetation layers with torching potential (Figure 3.5).

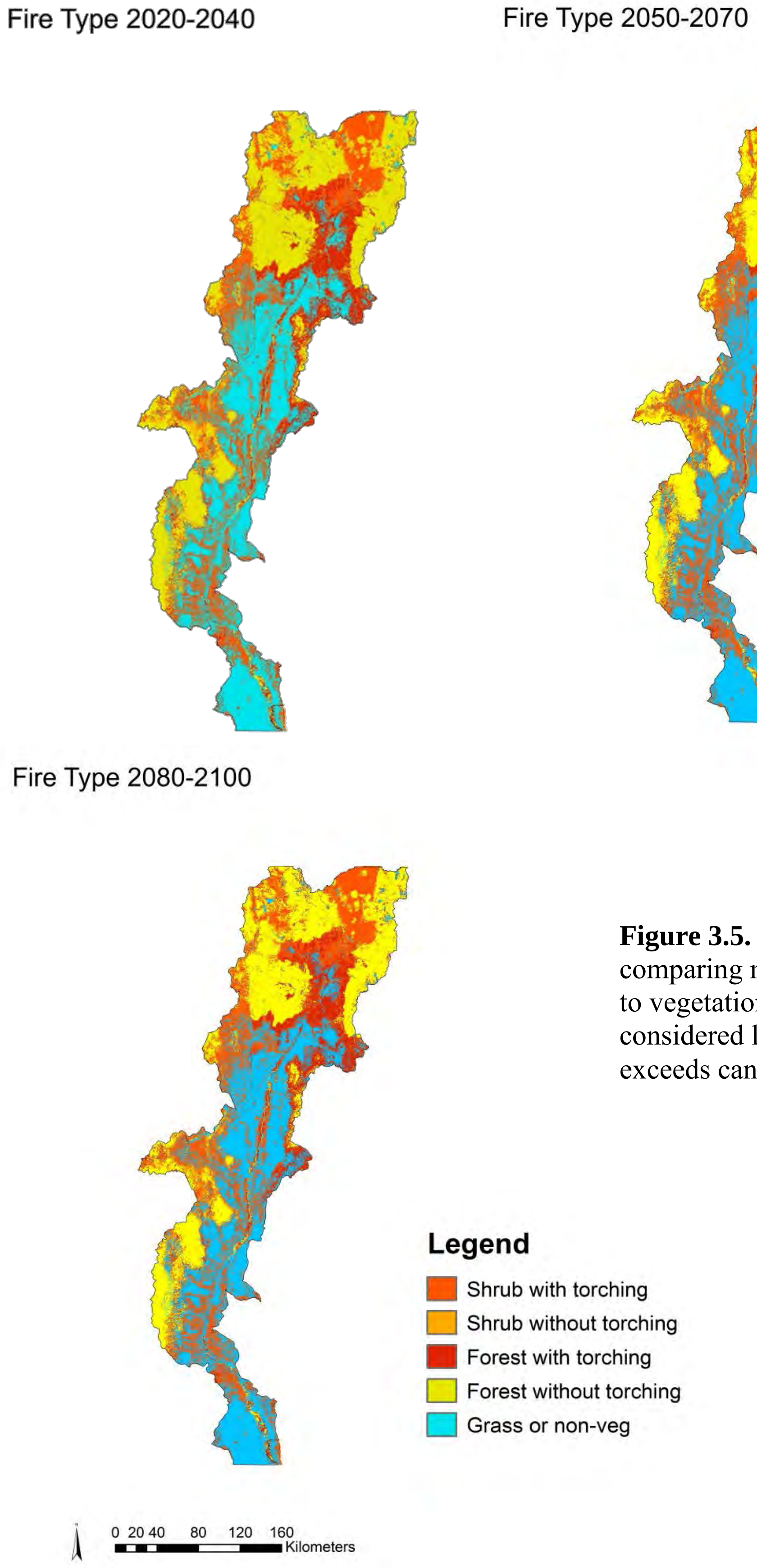


Figure 3.5. Fire types generated by comparing mean conditional flame length to vegetation height. Torching is considered likely where flame length exceeds canopy base height.

3.3 Species Response Functions

We developed a simple method for assessing impacts to species from wildfire based on the response functions described for the Wildfire risk framework (Calkin et al., 2010 and Scott et al., 2013). Response functions relate fire effects to realized impacts for a resource of interest by quantifying a net value change in the resource under different fire classes. For the purpose of this study we classified fire into one of 5 categories based on the likelihood of certain fire behaviors. Our classifications were: 1) shrub with torching; 2) shrub without torching; 3) forest with torching; 4) forest without torching, and: 5) grass or non-veg (see Fire Methods section for details on how these were calculated). To estimate risk to species, we scored species' short term response to fire as highly positive (+2), positive (+1), neutral (0), negative (-1), or highly negative (-2) to each fire type (Table 3.1). Scores were determined through literature review and according to previous assessments and classifications (e.g. Friggens et al., 2013).

Fire can benefit wildlife by increasing water availability, insect abundance, encouraging new plant growth and increasing the structural heterogeneity of habitats (Buchalski 2006; Pilliod et al., 2006; Smith 2008). Low-severity burns within riparian habitats can be beneficial by removing ground cover, releasing nutrients to remaining vegetation, and encouraging new growth. Riparian vegetation may be more resilient to wildfire, given that water is present and willows are known to be positively affected by low-severity burns that do not damage root crowns (Coleman 2011). However, current conditions resulting from decades of fire suppression support more destructive high severity wildfires that can increase the homogenization of landscape and reduce habitat availability for wildlife (Pilliod et al., 2006). Fuel accumulation and restricted flows combined with predicted reductions in snowpack, earlier snowmelt, and increased drought suggest wildfires will be larger, more frequent, and more severe in the southwestern riparian habitats (Stromberg and Chew 2002; Fulé 2010, Westerling et al., 2006). The effects of these high-intensity wildfires include the removal of vegetation, the degradation of subbasin condition, altered stream behavior, and increased sedimentation of streams (Westerling et al., 2006; FWS, 2014; Fulé 2010).

According to Pilliod et al, 2006, wildlife species that tend to benefit immediately post fire are associated with open canopies and forest floors and rely on an insect prey base. Characteristics of species with negative response include those associated with downed wood, snags, dwarf mistletoe, dense forests and closed canopies as well as small mammals that prefer shrub cover to avoid predators. Direct mortality from a fire is generally considered rare because it is assumed species are able to seek cover (Pilliod et al., 2006). Exceptions are when fire occurs during the breeding season and results in increased mortalities for shrub or ground nesting species. Mobility is not necessarily a guarantee of survivorship as losses have been noted for birds and other fast movers (e.g. lizards, Coleman 2011).

Within riparian ecosystems, wildfires can reduce vegetative cover and result in greater soil erosion, subsequently resulting in increased sediment flows in streams (Fulé 2010). The severity of a burn influences the post-fire landscape including the spatial distribution of litter, down wood, snag and vegetation, which in turn, influence wildlife resource availability and interactions. Over longer periods of time, high severity fires may lead to post-fire ash flows and flooding that can result in significant declines of resident fish and amphibian communities and impact predator populations (e.g. snakes)(Coleman 2011; FWS 2014). Increased sedimentation also reduces visibility within the water column, hampering the hunting ability of aquatic predators and suffocating amphibian eggs. Fire can also favor the establishment of exotic species (e.g. Smith et al., 2009). A wildfire in the Bosque forest of the Middle Rio Grande caused significant decreases in in litter depth, canopy cover, large and medium cottonwoods and an increase in small cottonwoods, cottonwood snags, forb cover and saltcedar following wildfire (Smith et al., 2005). Composition and structural changes resulting from fire have severe implications for species that rely on native habitat, particularly birds and small mammals. Habitat loss due to wildfire is among the threats recognized for the southwestern willow flycatcher, the yellow billed cuckoo, and the Chiricahua leopard frog (USFWS 2007; USFWS 2014).

The scores determined for this study represent our understanding of the current conditions and likelihood for severe fires within the study area. These scores are not a final determination of fire responses and potential risks for the studied species. Future researchers may wish to modify these scores based upon additional evidence, perceived differences, or to apply to additional study sites. For the purpose of this exercise, we focus on the impact of fire on species assuming that the study area represents the total of available habitat. In reality, species with large regional distributions may not experience large impacts from local fire effects. We do not consider the seasonality of fire though future efforts would benefit from its inclusion since spring and fall fires have unique effects on plant response. Smith (2008) notes that summer burns are likely to be more severe than winter burns and thus have more impacts for effects on vegetation and wildlife. For wildlife, early-summer fire in grassland can increase native forbs and thus benefit many bird species, especially raptors (Smith 2008). However, for other species early fires are likely to result in increased mortality for species with non-mobile young (e.g. Bats, Lacki et al., 2009) and destruction of potential nesting habitats for species that rely on shrubs and dense vegetation (Southwestern willow flycatcher). Given predictions for increased number, size and severity of fire, we assumed during scoring that all species could be exposed to early season fires.

Background data for scores

Amphibians

Amphibians are thought to not respond well to prescribed fire (Pilliod et al., 2006). However, at least one toad, *Bufo boreas,* appears to be attracted to burn areas (Pilliod et al., 2006). Hossak and Corn (2007) identify positive and neutral responses to fire for three amphibians in the Northern U.S. Some species appear to be drawn to burn sites, perhaps due to the low quality of habitats with fire suppression. Amphibian response to fire is likely to be species and location specific (Hossack and Pilliod, 2011). Fire may benefit herpetofauna by maintaining wetlands where fire suppression is associated with increased trees, reduced water levels, and drying of bogs (Russell et al., 1999). Though largely beneficial, several species have been extirpated after fires led to landslides and large debris fields (Hosack and Pilliod, 2011). Among the negative impacts observed, fire can increase sedimentation and sediment load which inhibits gas exchange by eggs. Fire may also have negative impacts for species that require leaf litter or other surface cover that is burned (Pilliod et al., 2006). Upland species, like Salamanders, in particular often rely on moist conditions, litter and wood.

The federal register (FWS, 2006) for the listing of the Mexican gartersnake notes that:

> "Sedimentation can adversely affect fish populations used as prey by northern Mexican or narrow-headed gartersnakes by: (1) Interfering with respiration; (2) reducing the effectiveness of fish's visually based hunting behaviors; and (3) filling in interstitial spaces (spaces between cobbles, etc., on the stream floor) of the substrate, which reduces reproduction and foraging success of fish (Wheeler *et al.,* 2005, p. 145). Excessive sediment also fills in intermittent pools required for amphibian prey reproduction and foraging."

The American Bullfrog, *Rana catesbeania,* may be indifferent to fire impacts in the short term and likely to benefit in the long term. However, its reliance on permanent ponds and association with dense vegetation was considered indicative of sensitivity to high intensity fires, which could result in increased sedimentation, inhibited reproductive success. Importantly, fire severity and resulting erosion and debris flows are not directly modeled in the current exercise. Negative effects are assumed for crowning fires within both shrub and forest habitats. These effects are largely manifest through increased likelihood of sedimentation and debris flows during egg and tadpole development. Though crowning fires are not necessarily going to lead to these negative impacts, they are an indication that fire could be problematic within areas labeled high impact. Less intense (non-crowning) fires are assumed to not have a negative impact on this species.

The reliance of the Northern Leopard frog, *Rana pipiens,* on permanent ponds and association with dense vegetation was considered indicative of sensitivity to high intensity fires, which could result in increased sedimentation and loss of thermal cover. Again, negative effects are assumed for crowning fires within both shrub and forest habitats. These effects are largely manifest through increased likelihood of sedimentation and debris flows during egg and tadpole development.

Table 3.1. Species response scores used for fire risk maps. Bat scores were divided to reflect differences in the direction of impact for foraging and roosting sites.

Species	**shrub with torching**	**shrub without torching**	**forest with torching**	**forest without torching**	**grass or non-veg**
Empidonax t. extimus	-2	-1	-2	-1	0
Vermivora luciae	-2	-2	-2	-1	0
Coccyzus a. occidentalis	-2	-1	-2	-1	0
Myotis yumanensis - roosting	0	0	-2	0	0
Myotis yumanensis -foraging	+1	+1	+1	+1	0
Myotis occultus- roosting	0	0	-2	0	none
Myotis occultus--foraging	+1	+1	+1	+1	0
Myotis volans- roosting	0	0	-2	0	none
Myotis volans--foraging	+1	+1	+1	+1	0
Sigmodon hispidus	-2	-1	-2	-1	-1
Zapus luteus	-2	-1	-2	-1	-2
Chrysemys picta belii	-2	-1	-2	-1	-1
Thamnophis cyroptsis	-2	0	-2	0	0
Rana pipiens	-1	0	-1	0	0
Rana catesbeania	-1	0	-1	0	0

Birds

Fire can alter bird habitat by changing vegetation composition and structure which influences nest site availability, foraging substrates, and nest predation rates. Prescribed or wildfire during breeding season is likely to reduce habitat for shrub and ground dwelling species (Finch et al., 1997). Loss of snags due to fire will harm cavity nesters (Finch et al., 1997 but see Smith et al., 2005) though such losses can increase beetle food in the short term. There is some concern that fire can encourage the spread the exotic Tamarisk species especially during drought years (Smith et al., 2005). Cicadas emerged earlier on burned versus unburned plots along the Rio Grande increasing potential mismatches between this important food source and breeding events (Smith et al., 2006). The biggest risk from wildfire for bird species along the Rio Grande is the further degradation and isolation of limited habitat (FWS, 2014). Fire is considered a major threat to remaining habitat for the southwestern willow flycatcher (FWS Species Report http://ecos.fws.gov/speciesProfile/profile/speciesProfile.action?spcode=B094) and is listed as a concern for determination of threatened status for the Western Yellow-billed cuckoo.

Bird response to fire has been studied in relation to nest and foraging behaviors. Riparian associated birds in Oregon that preferred burned habitats (at least in the short term) tended to forage insects from air, ground or bark, have closed nests, and nest either on the ground or in the canopy (Klamath Bird Observatory 2009). Birds that glean insects from foliage, had open-cup nest, or that nest in shrubs did not prefer burned habitats. In their study of burn severity and bird populations in Arizona forests, Kotliar et al., (2007) found that species that are less likely in burned areas included sub-aerial insectivores, ground foragers and foliage gleaners though some ground foraging species were also found in greater abundance on burned sites. Kotliar et al., (2007) note that species associated with the shrub layer, which resprout vigorously following fire, were more common on severely burned landscapes. Smith (2008) found that bird response to wildfire in arid habitats in Arizona related to increased seed production by both native and nonnative grasses. Seed eating birds that did not rely on grass or litter cover increased substantially after burns, whereas species that responded negatively to fire tend to prefer shrub cover (Smith 2008). In New Mexico, secondary cavity nesting birds show both positive and negative associations with severely burned sites (Smith et al., 2005).

Spatial patterns of burn severity and time of ecosystem recovery are likely to be important determinants of bird response (Kotliar et al., 2007). Most species occurred across all burn severities and they found species richness was similar across all patches due to species replacement. Kotliar et al., (2007) observed strong negative responses were largely within two years post-fire whereas positive responses tended to increase with time. Seventy one percent of the species in their gradient analysis exhibited either positive or neutral density responses to fire effects (Kotliar et al., 2007). Their findings suggest that the majority of species may tolerate or benefit from many of the ecological changes that occur across the severity gradients.

For this study, we considered fire a risk factor for increased habitat loss for all three bird species (Table 3.1). Southwestern willow flycatchers and Western yellow-billed cuckoos already suffer habitat loss and fire is likely to destroy important nesting habitats within remaining suitable sites. Warblers were considered at high risk of negative impacts for all fires occurring in shrub habitat and fires in forests with torching and at moderate risk for forest fires without torching.

Small mammals

We distinguished between foraging and roosting effects for bats and consider grassland fire for *Sigmodon* and *Zapus* species (Table 3.1). Small mammal mortality from fire can be directly caused by burns, heat stress, asphyxiation, physiological stress, trampling by other animals, or predation while fleeing fire (Sullivan 1995). Indirect causes of mortality can include changes in quality and quantity of food, nest site availability, predation, parasitism, disease, increased competition, and social interactions (Sullivan, 1995). Shrews, voles, read-backed voles, rabbits, snowshoe hares, red squirrels and northern flying squirrels were found to be negatively impacted by short term consequences from prescribed fire (Pilliod et al., 2006). Deer mice, chipmunks, pocket gophers, and ground squirrels which tend to favor disturbances were found to be less affected by prescribed fires (Pilliod et al., 2006). Wildfires are likely to have similar if not more profound impacts on small mammal communities.

Wildfire and especially post wildlife flooding are identified as especially problematic for the endangered Meadow mouse, *Zapus luteus* (FWS, 1995). From the Federal Registrar:

> “Following fires, we found that, depending on fire intensity and the subsequent ash and debris flow within stream reaches, New Mexico meadow jumping mouse populations can be significantly affected and likely extirpated, even when 15 km (9 mi) of continuous suitable habitat existed prior to the fire (Sugarite Canyon; Frey 2006d, pp. 18-21; 2012b, p. 16).”

We considered the New Mexico Meadow mouse as very vulnerable to high intensity fires within the riparian habitats, fires within grass dominated habitats, and moderately vulnerable to less intense fires (Table 3.1).

In general, hispid cotton rats, *Sigmodon hispidus*, are thought to respond negatively to fire in the short term but recover rapidly once vegetation cover has been reestablished. Hispid cotton rats are known to experience mortality from fires though the great majority are able to escape injury by retreating into burrows or unburned areas. Most differences in abundances are attributed to differential movement and recruitment between study plots after burns. Kauffman et al., (1990) considered hispid cotton rats as fire-negative, since they are associated with plant debris and are foliage eaters. However, they are often reported as increasing after fire with the highest populations found in the first few growing seasons after fire (Sullivan 1995 and references therein). Hispid cotton rats were more numerous (1.2 individuals/burned versus 0.16/individuals unburned) on burned treatments in Konza prairie, whereas Meadow jumping mouse was about the same (Knoof and Samson1999). Fire appears to have a positive effect on cottonrats in southwestern forests when they occur every 3-4 years. More and less frequent fires are associated with smaller cotton rat populations (Sullivan1995 and references therein). There are no studies of the hispid cotton rat in New Mexico or southwestern forests. Given their strong preference for thick vegetation and the relative scarcity of such habitat outside of the riparian forests in which they exist, we rated this species as sensitive to all types of fire (Table 3.1).

Myotis yumanensis, M. occultus and M. volans are likely to benefit from most fire activity except where roosts are impacted. Bats are able to escape fire mortality and are drawn to the insects within newly burned habitats (Malison and Baxeter 2010; Lacki et al., 2009). In California, bats benefited from the clearing of vegetation, which improved foraging success, and increased insect abundance due to early successional habitat (Bhucalski et al., 2013). Fire can also increase roosting habitat for species that rely on snags or tree cavities (Lacki et al., 2009). The long term

(>5 years) consequences of fire are probably positive. Over large periods of time, fire contributes to habitat heterogeneity associated with improved conditions for these species (Bhucalski et al., 2013). Fire suppression is considered one of causes leading to reduced habitat for bat species in the SW (Chung-MacCoubrey 2005). Buhcalski et al., (2013) note that *M. yumanensis* in particular may prefer emergent insects and had increased activity levels in riparian habitat burned at moderate to high severity.

Bats may be negatively impacted by fire when burns happen in the early to mid-spring seasons especially around roosting maternity sites (Lacki et al., 2009). At best, fire would represent a short term disturbance to colonies and at worst fire could lead to the loss of roost site and mortality of non-volant young. In addition, large fires that reduce habitat heterogeneity will lead to a decline in habitat quality. Pilliod et al., (2006) considered *Myotis volans*, silver haired bats, and other species at risk of negative fire impacts if important roost sites under bark of tall, large diameter tree of in cavities of large snags are destroyed. Bat response to fire appears to relate to clearing of vegetation (better for foraging), increasing insect abundance (due to increasing early successional plants and emergent aquatic insects), and fire is assumed to increase the quantity and quality of roosting habitat by creating dead and dying trees. In Idaho, greater bat diversity was associated with a high severity fire in mixed conifer riparian habitat (Malison and Baxeter 2010). However, emphasis is made that bats are likely dependent on a mosaic of burn habitats with foraging best in early successional landscapes and roosting best in late successional landscapes.

To account for the potential negative impacts of fire on roost sites, we classified areas expected to experience crown fires as high risk. Though the presence of torching does not necessarily predict habitat loss, the indication for torching is used to qualify the potential risk for bat species. Within this study, areas with a prediction for torching fires are considered more likely to result in negative impacts for species than areas with non-torching fires.

Reptiles

Most reptiles appear to do well with prescribed fire (Pilliod et al., 2006). Direct mortality is probably rare and is most often associated with individuals undergoing ecdysis (Russel et al., 1999). Low-severity burns within riparian habitats can be beneficial by removing ground cover, releasing nutrients to remaining vegetation, and encouraging new growth. Riparian vegetation may be more resilient to wildfire, given that water is present and willows are known to be positively affected by low-severity burns that do not damage root crowns (Coleman 2011). However, high-severity burns are likely to reduce basking sites and a loss of cover could increase the risk of predation. Over longer time periods high severity fires may lead to post-fire ash flows, flooding that can result in significant declines of resident fish and amphibian communities a primary source of prey for gartersnake populations (Coleman 2011; FWS 2014).

From the federal register for the Mexican gartersnake (FWS 2014) cites negative impacts from prey loss and reduced predation efficiency due to sedimentation:

> “Effects to northern Mexican and narrow-headed gartersnake habitat from wildfire should be considered in light of effects to the structural habitat and effects to the prey base... Sedimentation can adversely affect fish populations used as prey by northern Mexican or narrow-headed gartersnakes by: (1) Interfering with respiration; (2) reducing the effectiveness of fish's visually based hunting behaviors; and (3) filling in interstitial spaces (spaces between cobbles, etc., on the stream floor) of the substrate, which reduces reproduction and foraging success of fish

(Wheeler *et al.,* 2005, p. 145). Excessive sediment also fills in intermittent pools required for amphibian prey reproduction and foraging."

Further, "The presence of adequate interstitial spaces along stream floors may be particularly important for narrow-headed gartersnakes. Hibbitts *et al.,* (2009, p. 464) reported the precipitous decline of narrow-headed gartersnakes in a formerly robust population in the San Francisco River at San Francisco Hot Springs from 1996 to 2004. The exact cause for this decline is uncertain, but the investigators suspected that a reduction in interstitial spaces along the stream floor from an apparent conglomerate, cementation process may have affected the narrow-headed gartersnake's ability to successfully anchor themselves to the stream bottom when seeking refuge or foraging for fish (Hibbitts *et al.*2009, p. 464). These circumstances would likely result in low predation success and eventually starvation. Other areas where sedimentation has affected either northern Mexican or narrow-headed gartersnake habitat are Cibecue Creek in Arizona, and the San Francisco River and South Fork Negrito Creek in New Mexico…The San Francisco River in Arizona."

We ranked the black-headed gartersnake as at risk of mortality or negative habitat impact under conditions where forest or shrub habitats are expected to have torching fires. Non torching fires were not considered as problematic (Table 3.1).

Research regarding fire impacts for the western painted turtle is limited. Research for the more terrestrial eastern box turtle shows significant mortality from fire in tallgrass prairies and other habitats (Leunsmann 2006). However, it is likely box turtles with access to burrows are able to survive fires. Hatchling and juvenile eastern box turtles appear to hide under litter, which exposes them to fire, and leads to increased mortality. Eastern box turtles are often not able to escape the active line of fire and many suffer burn scares as a result (Leunsmann 2006 and references therein). Frequent fires may result in more mortality, increase fragmentation of habitat and reduced turtle populations.

Leunsmann 2006 concludes high-severity fires that kill trees and scorch canopies are likely to be detrimental to turtles that favor forests. Additionally, loss of the litter layer by fire could also have negative impacts because litter is used extensively for cover throughout the year. The Governor's Drought Task Force identified *C. p. belii* as a species susceptible to increased mortality due to drought and wildfire.

We considered areas with predicted torching type fires as at high risk of negative impact for the western painted turtle due either directly to exposure to fire or associated changes in habitat. Areas with non-torching fires were classified as having a moderate risk of negative fire impact for this species (Table 3.1).

References

1. Abatzoglou , J.T. and Brown TJ. 2012. A comparison of statistical downscaling methods suited for wildfire applications. International Journal of Climatology 32(5):772-780.
2. Batista, C.G. 2002. *Rana catesbeiana* (Bullfrog). Effects on native anuran community. Herpetol. Rev. 33:131.
3. Buhcalski, M.R., Fontaine, J. B., Heady III, P. A., Hayes, J.P. and W. F. Frick. 2013. Bat response to differing fire severity in mixed-conifer Forest California, USA. PLOS One 8: e57884
4. Calkin, D.E., Ager, A.A., Gilbertson-Day, J., eds. 2010. Wildfire risk and hazard: procedures for the first approximation. Gen. Tech. Rep. RMRS-GTR-235. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 62 p.
5. Coleman, S. 2011. *Thamnophis rufipunctatus* and *Thamnophis eques* information post-Wallow Fire. White paper drafted by Stephanie Colemen, Apache-Sitgreaves National Forest. 4 pp.
6. Dunne, J.P., John, J.G., Adcroft, A.J., Griffies, S.M., Hallberg, R.W., Shevliakova E., Stouffer R.J., et al. 2012. GFDL's ESM2 Global Coupled Climate-Carbon Earth Systemm Models. Part I: physical formulation and baseline simulation characteristics. Journal of Climate 25:6646-6665.
7. Finch, D., and Sutherland, E.K. 2007. Achieving fuel reduction goals and restoring riparian zones to healthy conditions: Riparian ecosystem dynamics in relation to fire in the Rocky Mountains. In: Baldwin, V. C. Accomplishing and applying National Fireplan Research and Development from 2001-2005. Gen. Tech. Rep. GTR-RMRS-187. Fort Collins, CO.: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. p. 32-33.
8. Finch, D.M., J.L. Ganey, W. Yong, R.T. Kimball, and R. Sallabanks. 1997. Effects and interactions of fire, logging and grazing. Pages l03-106, In W.M. Block and D.M. Finch (technical editors). Songbird Ecology in Southwestern Ponderosa Pine Forest, A Literature Review. USDA Forest Service General Technical Report GTR-292. Rocky Mountain Forest and Range Experiment Station, Fort Collins. Colorado.
9. Finney, M.A., McHugh, C.W., Grenfell, I.C., Riley, K.L., and K.C. Short. 2011. A simulation of probabilistic wildfire risk components for the continental United States. Stoch Environ Res Risk Assess 25:973-1000.
10. Fritz, L.K. and F.B. Samson (eds). 1999. Ecology and conservation of Great Plains vertebrates. Springer-Verlag, New York, 327p
11. Fulé, P. 2010. Interactions of climate change, fire regimes, and hydrologic regimes. Presented at the Arizona Climate Change Adaptation Workshop for Natural Resource Managers of the Four Forest Restoration Initiative Area. April 7-8, 2010. Flagstaff, Arizona.
12. Hossack B.R., and Corn P.S. 2007. Responses of pond-breeding amphibians to wildfire: short-term patterns in occupancy and colonization. Ecological Applications 17: 1403-1410
13. Hossack, B.R., and Pilliod, D.S., 2011, Amphibian responses to wildfire in the western United States- Emerging patterns from short-term studies. Fire Ecology 7: 129-144
14. Hossack, B.R., W. H. Lowe and P.S. Corn. 2012. Rapid Increases and Time-Lagged Declines in Amphibian Occupancy after Wildfire. Conservation Biology 27: 219-228. doi: 10.1111/j.1523-1739.2012.01921.x
15. Kaufman, D. W., Finck, El. J., and Kaufman, G.A. 1990. Small mammals and grassland fires. In: Collins, Scott L., Wallace, Linda L., eds. Fire in North American tallgrass prairies. Norman, OK: University of Oklahoma Press: 46-80.

16. Klamath Bird Observatory and Bureau of Land Management. 2009. Short term negative response noted for southwestern Oregon birds after prescribed fire. Version 1.2. Riparian fuel treatments in intermittent and perennial stream riparian areas: Effectiveness and ecological effects. Rep. No. KBO-2009-0008. Klamath Bird Observatory, Ashland, OR. http://www.klamathbird. org/images/stories/kbo/pdfs_dsts/riparian_fuel_ treatments_v1.2.pdf
17. Kotliar, N.B., P. L. Kennedy, and K. Ferree. 2007. Avifaunal responses to fire in southwestern montane forests along a burn severity gradient. Ecological Applications 17: 491–507.
18. Lacki, M.J., Cox, D.R., Dodd, L.E., and M.B. Dickinson. 2009. Response of northern bats (*Myotis septentrionalis*) to prescribed fires in eastern Kentucky forests. Journal of Mammalogy 90: 1165-1175.
19. Luensmann, P. 2006. *Terrapene carolina*. 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/ [2014, December 9].
20. Malison, R.L., and C.V. Baxeter. 2010. The fire pulse: wildfire stimulates flux of aquatic prey to terrestrial habitats driving increases in riparian consumers. Canadian Journal of Fisheries and Aquatic Sciences 67: 570-579
21. Peters, G.P., Andrew, R.M., Boden, T., Canadell, J.G., Ciais, P., Le Quéré, Marland G., Raupach M.R., and C. Wilson. 2013. The challenge to keep global warming below 2°C. Nature Climate Change 3: 4-6.
22. Pilliod, D.S., Bull, E., Hayes, J.L., Wales, B.C. 2006. Wildfire and invertebrate response to fuel reduction treatments in dry coniferous forests of the Western United States: a synthesis. Gen. Tech. Rep. RMRS-GTR-173. Fort Collins, CO.: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 34p.
23. Rogelj J., Meinshausen M., and R. Knutti 2012. Global warming under old and new scenarios using IPCC climate sensitivity range estimates. Nature Climate Change 2: 248-253.
24. Russell, K.R., D.H. Van Lear, and D.C. Guynn, Jr. 1999. Prescribed fire effects on herpetofauna: Review and Management implications. Wildlife Society Bulletin 27: 374-384.
25. Scott J.H., Thompson M.P., and Calkin D.E. 2013. A wildfire risk assessment framework for land and resource management. Gen Tech. Rep. RMRS-GTR-315. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 83 p.
26. Scott J.H., Thompson M.P., and Calkin D.E. 2013. A wildfire risk assessment framework for land and resource management. Gen Tech. Rep. RMRS-GTR-315. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 83 p.
27. Sheffield J., A.P. Barrett, B. Colle, D.N. Fernando, R. Fu, K.L. Geil, Q. Hu, J. Kinter, S. Kumar, B. Langenbrunner, K. Lombardo, et al. , 2013: North American climate in cmip5 experiments. part i: evaluation of historical simulations of continental and regional climatology. Journal of Climate 26: 9209–9245.
28. Short, K.C. 2014. Spatial wildfire occurrence data for the United States, 1992-2012 [FPA_FOD_20140428]. 2nd Edition. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. http://dx.doi.org/10.2737/RDS-2013-0009.2
29. Smith, B.E. and D.A. Keinath. 2007. Northern Leopard Frog (*Rana pipiens*): a technical conservation assessment. [Online]. USDA Forest Service, Rocky Mountain Region.

Available: http://www.fs.fed.us/r2/ projects/scp/assessments/northernleopardfrog.pdf [071414].

30. Smith, D. M., D. M. Finch, C. Gunning, R. Jemison and J.F. Kelly. 2009. Post-wildfire Recovery of Riparian Vegetation during a Period of Water Scarcity in the Southwestern USA. Fire Ecology Special Issue 5, Page 38
31. Smith, D. M., Kelly, J.F., and D.M. Finch. 2005. Avian nest box selection and nest success in burned and unburned southwestern riparian forest. Journal of Wildlife Management 71: 411-421.
32. Smith, D.M., J.F. Kelly, and D. M. Finch. 2006. Cicada emergence in southwestern riparian forest: Influences of wildfire and vegetation composition. Ecological Applications 16: 1608–1618
33. Smith, J. K., compiler. 2008. Research Project Summary: Effects of prescribed fires in semi-desert plant communities in southeastern Arizona. 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/ [2014, December 9].
34. Stromberg, J.C. and M. K. Chew. 2003. Fire and riparian ecosystems. Southwest Hydrology 2: 23.
35. Sullivan, J. 1995a. *Bubo virginianus*. 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/ [2014, December 17].
36. Sullivan, J. 1995b. *Sigmodon hispidus*. 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/ [2014, December 9].
37. U.S. Fish and Wildlife Service. 2014. Rules and Regulations Endangered and Threatened Wildlife and Plants, Threatened Status for the Northern Mexican Gartersnake and Narrow-Headed Gartersnake, Final Rule Federal Register Vol. 79, No. 130.
38. U.S. Fish and Wildlife Service. 2014. Endangered and Threatened Wildlife and Plants, Determination of Threatened Status for the Western Distinct Population Segment of the Yellow-billed Cuckoo (*Coccyzus americanus*), Final Rule Federal Register Vol. 79, No. 192.
39. Westerling, A.L., H.G. Hidalgo, D.R. Cayan, and T.W. Swetnam. 2006. Warming and earlier spring increase western U.S. forest wildfire activity. Science 313: 940-943.

4. SPECIES ANALYSIS

4.1 Niche Model Analysis

We use Maximum Entropy (MaxEnt 3.3.3) methods to estimate change in suitable habitat due climate change in order to determine how and to what extent species will be exposed to unsuitable conditions (Phillips et al., 2008). MaxEnt has become a popular tool for these types of estimations as it can work with presence only (e.g. museum) data, is user friendly, and has been shown to be able to accurately represent suitable habitat with as little as 10 samples (Wisz et al., 2008).We created models of suitable habitat for each species based on collection and survey samples from New Mexico and climate, biophysical, and vegetation characteristics of habitats along the Rio Grande. We then project future suitability by applying these models to future conditions as predicted by three climate models at three times periods, 2030, 2060 and 2090. We then compare current and future distributions to explore to what degree habitats are likely to change along the Rio Grande and the consequences of those changes for thirteen species (Section5).

4.1.1 Observational data

Presence data for each species was collected from museum records and research studies. We search data network portals including Arctos, VertNet (includes MaNIS, HerpNET, ORNIS) and the Global Biodiversity Information Facility (GBIF). We downloaded occurrence records for observations after 1970, to correspond with the timeframe of historic climate data. We checked downloaded data for errors and converted species locations NAD1983. For most species, we removed observations with location uncertainty greater than 1km. Records without an assigned geodetic datum were plotted on the datum WGS84, though the collections note that not all specimens have defined datum. In the event that an observation datum is misidentified (does not conform to our assumption that it is WGS84), a reported location may be misplaced by up to 500 meters. This margin of error was deemed acceptable for this study; we are not concerned with site specific characteristics but rather shifts in the overall presence of suitable habitat, the modeling of which concerns relatively low resolution data (+ or – 1 km).

Amphibians

Amphibian data was downloaded from Herpnet, Arctos and the GBIF. We compiled data from searches on *Rana pipiens* (Northern leopard frog), *Lithobates catesbieanus* (American Bullfrog) as well as included *Rana catesbieanus*. Museum data without define datum were assigned designations of NAD1927 for occurrences before 1984 and WGS84 for occurrences after 1984. Most data had accuracy to two decimal places.

Birds

Most museum records were accessed through the ORNIS 2 (ornis2.ornisnet.org), MANIS, ARTOS. In addition, we used nest and transect survey data conducted by The Rocky Mountain Research station in the Middle Rio Grande Valley during the 1990's and 2000's (Smith et al., 2006; Finch and Sutherland 2007). Only specimens reported or collected during breeding season (June-September) were used in the analysis. We converted UTM from study data to WGS84 using the spreadsheet of Dr. Steve Dutch (http://www.uwgb.edu/dutchs/SITEMAP.HTM).

Mammals

Most data was accessed from Arctos based on searches for *Sigmodon hispidus*. *Sigmodon hispidus berlandie*ri and specimens without a subspecies designation were recorded as S. hispidus. For the New Mexico Meadow jumping mouse, we records identified as *Zapus hudsonius luteus*. We also crosschecked our museum records with those of Malaney et al., (2012) and incorporated new observations from Malaney et al., (2012) into our data set.

Reptiles

Data was accessed from Herpnet based on searches for *Chrysemys picta* for the Western painted turtle and *Thamnophis cyrtopsis* for the black-necked gartersnake.

4.1.2. MaxEnt modeling

Maximum Entropy modeling was performed to provide and index of breeding habitat suitability (Royle et al., 2012). We used the Maxent program (version 3.3.1 – see http://www.cs.princeton.edu/~schapire/maxent, see Phillips et al., 2006, 2009). Maxent uses a maximum likelihood method to model species' distributions by generating a probability distribution over the pixels in a grid of the study area, where observed species presence are the sample points within the study area. The probability distribution is estimated to maximize entropy (i.e., that is the closest to uniform) under a set of constraints or variables assumed to represent suitable habitat values at species occurrence locations. To achieve maximum entropy, the expected value of each environmental variable distribution must match its empirical mean (the mean over the sample points). MaxEnt measures how well the predicted distribution fits the sample points as compared to a uniform distribution, which is used to represent the probability distribution of the model (e.g. represents a measure of the likelihood of the samples). In this way, MaxEnt determines what characteristics of used areas are more suitable than those of a background sample (Royle et al., 2012).

We ran MaxEnt algorithms for all species using a subset of continuous environmental and biophysical variables and one categorical predictor, biome (Section 2, see Phillips et al., 2006 regarding categorical variables). Based on the literature and other modeling work (e.g. Mathews et al., 2011; Huntley et al., 2006; O'Donnell and Ignizio 2012), we selected variables we believed would be important for determining (or limiting) species distribution (Merow et al., 2013 discusses importance of using biological meaningful variables). For some models (e.g. *Myotis occultus*, Northern leopard frog) we eliminated variables that reduced model gain during initial runs. The data analysis for this paper was generated using SAS/STAT software, Version 9.3 of the SAS System for Windows. We used Pearson's r to assess correlation among the variables (Proc Corr) and minimize redundant variables (significant Pearson's correlations greater than 0.80) for each model by keeping those that were most likely to be biologically relevant. Pearson's correlation coefficients for model variables are available upon request.

Potential evapotranspiration represents water loss through plants would expire given unlimited precipitation and was included to represent characteristics of the habitat not represented by other measures (e.g. productivity and plant functional group). Potential evapotranspiration was included to represent potential productivity changes under warmer weather conditions. Potential evapotranspiration has been related to species richness of insects, an important food base for birds (Entling et al., 2012).

MaxEnt requires all variables to be the same cell size and extent. We used bilinear interpolation to resample climate, biome and distance to water layers (see Section 2) to a pixel size of 0.0083 ° by 0.0083 ° (corresponding to elevation and slope layers). The processed climatic variables (at the original resolution), all resampled variables, and the occurrence localities are available upon request. For all models, we used the default regularization value of 1 because we did not detect signs of overfitting linear, quadratic and hinge functions were included in the models.

We used 15-fold cross-validation to assess model fit. During each model run, 15% of the data was used to train models and the remaining points were used for testing. Subsets were randomly selected for each of 10 model iterations. We used threshold-independent Area Under the Curve (AUC) of the Receiving Operator Characteristic (ROC) curve as a metric of model accuracy. The AUC provides information on model performance without requiring output to be translated in to binary outputs indicating presence or no presence. The AUC value ranges between 0.5 and 1.0, where values of 0.5 indicate no difference between scores of specificity and sensitivity and scores of 1.0 indicate no overlap between the distributions of the scores. As noted by Yackulic et al. (2012), if detection covaries with covariates used in the model, the AUC values represent how well the model classifies detections versus background (which may or may not represent a true absence). For this reason, AUC values may not be a good measure of whether a particular model is good or not. Indeed, when using presence only data, caution is advised (Jimènez-Valerde 2011; Phillips et al., 2006). For the purposes of the current analysis, we consider the MaxEnt generated AUC values to represent how well our predictor variables can explain presence of species and assume presence is correlated with suitable habitat. Thus, high AUCs indicate working models.

We explored individual variable importance using a jackknife analysis within the Maxent model. Maxent runs series of models that sequentially excludes each variable. Variable importance is determined by the increase in gain of the model with and without variables. Gain is a measure of model fit that where a likelihood statistic is used to maximize the probability of presences in relation to background data. MaxEnt assigns a percentage to the environmental variable based on the increase in gain associated with the variable. Through this process, Maxent assesses the contribution of each variable to the model gain and its importance for predicting presence of species.

Bias Files

We used a bias correction file to account for potential sampling bias that can occur with presence only data, particularly as found in museum data (Phillips et al., 2009; Yackulic et al., 2013) for bird and bat species. Specifically, we created a sampling layer for MaxEnt by delineating areas within 10 km of an observed presence for each species. These buffer zones represented the typical areas and habitats surveyed for birds and bats along the Rio Grande. The bias correction directed MaxEnt to draw background data from the buffer areas, which were then used to validate the model and then applied the model to the entire study area. We included bias files during initial runs for small mammals (2km), amphibians (1km), and reptiles (1km) but dropped these files from final analysis to improve model performance.

4.1.3. Change to suitable habitat

To translate MaxEnt output to maps representing suitable versus nonsuitable habitat, we relied on a threshold calculated by equalizing sensitivity and specificity for training samples. This method is assumed to equalize the risk of over and under specifying actual suitability (van Riper et al., 2013). We choose this measure to maximize the likelihood that this analysis would accurately identify areas suitable for each species but also capture potential sites of interest (e.g. Liu et al., 2005). We created new layers representing each modeled output to create consensus maps of predicted change where consensus was determined by the prediction of presence by at least 2 models (see Species' Reports, Section 5). We also assessed results from individual models based upon each GCM to estimate the range of possible outcomes. We compared the distribution of suitable habitat over different land groups using BLM New Mexico Surface Ownership layers (vector file from http://rgis.unm.edu).

4.2 Calculating Adaptive Capacity

We built upon previous climate change vulnerability assessments conducted for the Middle Rio Grande (Friggens et al., 2013). This assessment used a prototype of the System for Assessing Vulnerability of Species (SAVS) to climate change (Bagne et al., 2011). For this study, we adapted the SAVS scoring system to summarize potential vulnerabilities or areas of adaptive capacity for a species that can used in conjunction with species niche model analysis to estimate risk (Friggens and Mathews 2013). Vulnerability, according to the IPCC definition, is an integrated measure of the expected magnitude of adverse effects to a system caused by a given level of certain external stressor. The SAVS system was designed to be a stand-alone tool for identify species' vulnerability through a tally of species' traits and expected impacts. Traits and criteria were selected based on an exhaustive review of the literature on potential climate impacts for amphibians, birds, mammals and reptiles and for all three elements of vulnerability: exposure, sensitivity, and adaptive capacity (Bagne et al., 2011). The SAVS has been employed for a number of studies in the Southwest (Bagne and Finch 2012; Bagne and Finch 2013; Coe et al., 2013; Friggens et al., 2013; Bagne et al., 2014). Adaptive capacity is one element contributing to the overall expected vulnerability of a species to climate impact (Figure 4.1). For species, adaptive capacity is often realized as the ability of an organism to cope with increased resource variations. For many traits adaptive capacity and sensitivity are analogues on one another. For instance, a species with good ability to disperse to new habitat is said to have increased adaptive capacity for dealing with climate related habitat shifts (Bagne et al., 2011). The lack of dispersal capacity for a species is often considered characteristic of species that is likely to be sensitive to potential climate impacts.

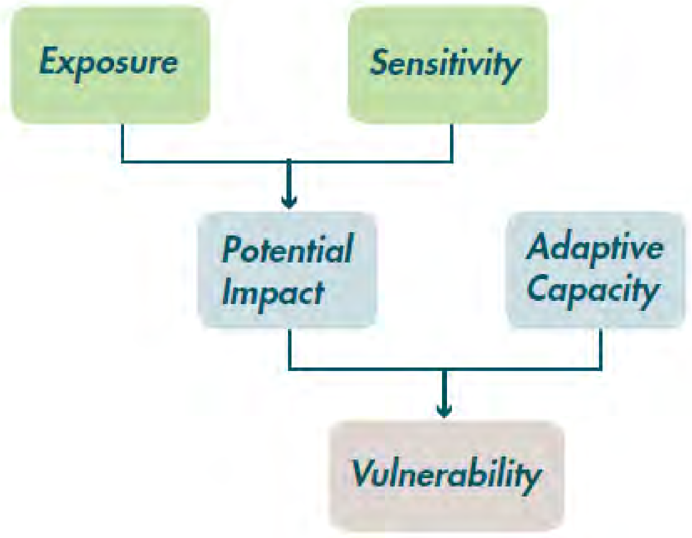


Figure 4.1. Vulnerability is measured as the collective impact of the exposure to a disturbance plus the sensitivity of the system to that disturbance, and moderated by adaptive capacity. From Glick et al., 2011.

To modify SAVS for use here, we streamlined the original questionnaire to include only those traits that dealt with non-modeled predictors of response (Table 4.2). Specifically, we eliminated most questions regarding habitat changes. Exceptions include questions dealing with habitat quality and habitat features (e.g. snags, ponds), neither of which are modeled adequately in the niche model analysis. A major consideration for this modification scheme is the treatment of migration. SAVS considers climate impacts for both breeding and nonbreeding habitat (Bagne et al., 2011). In addition, potential risks associated with migration are considered including species' requirement for additional habitats (or stopover sites) and potential for timing mismatches. In the current study, the collective output of models and vulnerability assessments represent impacts to summer habitat along the Rio Grande; winter habitat no longer contributes to vulnerability scores. However, the potential for species that migrate to be at a greater risk of habitat disruption (reliance on multiple sites), and timing mismatches still contribute to the scores generated for this study.

After responding to each question, scores are tallied based on the number of questions that fell within vulnerable, resilient or neutral categories. Final scores are scaled to a range of -6 (less vulnerability) to +6 (increased vulnerability). Importantly, these scores represent our current knowledge and assumptions regarding species' ecology. New research may lead to data and revisions in these scores.

Of the 12 species assessed here, the southwestern willow flycatcher had the greatest potential sensitivity to climate affects (Figure 4.2). The northern leopard frog also exhibited a high score indicating increased sensitivity to climate impacts (Figure 4.2). Further details are available for each species within their species' report (Section 5).

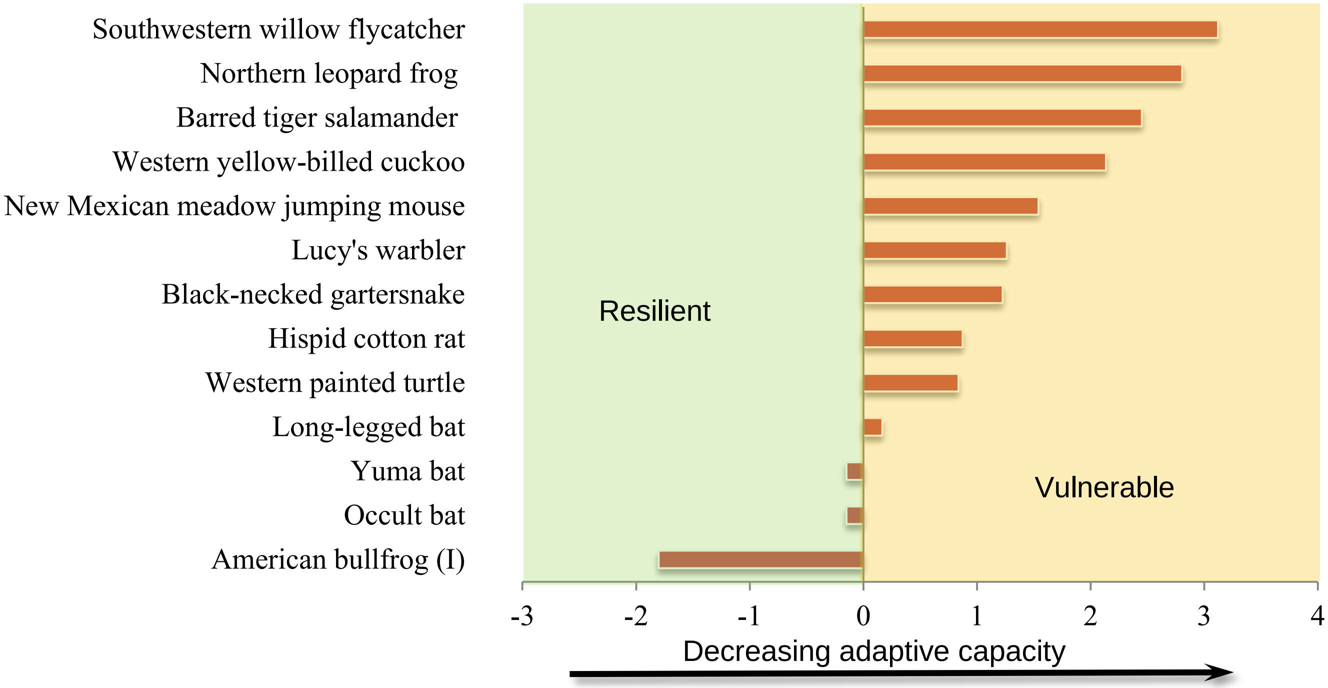


Figure 4.2. Vulnerability scores for 13 species residing in the Rio Grande Basin in New Mexico. Score are based on a modified version of SAVS (Bagne et al., 2011), where increasingly positive values represent increasing vulnerability. "I" indicates introduced species.

Table 4.2. Questions and short explanation used to score species vulnerability (or potential adaptive capacity) to climate change impacts. See Bagne et al., (2011) for discussion on how these traits were selected and further discussion on how species are assigned a score of "v" vulnerable, "r" resilient, or "n" neutral.

Non-modeled habitat features

1. Changes to non-modeled habitat components. Does this species rely on snags, vegetation structure (e.g. dense grasses, closed canopy) or other features not included in niche model analysis?
2. Change in habitat quality. Considers features relating to pond Temp, water quality, habitat productivity, etc.
3. Dispersal ability. Score based on limitations either due to limited dispersal ability or high site fidelity.
4. Reliance on migratory or transitional habitats. This variable accounts for differential habitat uses, particularly by migratory species, not considered within the models.

Physiological adaptations

5. Increase or decrease in physiological range limitation. Are there indications of heat related loss of offspring, water limitations, or reduced tolerance for extreme tempertures?
6. Sex ratios determined by temperature or food changes. Temperature dependent sex determination may put species at increased risk of skewed sex ratios (Bagne et al., 2011)**.** Recent evidence also shows nutritional state might skew sex ratios for some small mammal species (Cameron et al., 2008).
7. Response to predicted extreme weather events/disturbances. Accounts for known sensitivities to drought (desiccation), heat waves, or extreme conditions.
8. Changes to daily activity period. Will future climates limit or extend daily forage or reproductive activities?
9. Variable life history traits or coping strategies. Here we consider physiological traits (e.g. torpor) or behaviors (food caching) that allow species to cope with resource variations.
10. Ability to outlive limiting conditions. Reproduction can be limited under unfavorable conditions like drought. Populations may decline where individuals are not able to outlive limiting conditions.

Phenology

11. Migrates/hibernates in response to weather cues. Species are at an increased risk of timing mismatches between the initiation of important life history events and resources (e.g. nesting sites, leaf bud, insect emergence, river flow) when they rely on a weather cue.
12. Reliance on weather mediated resource (e.g. insect emergence). Species are at an increased risk of timing mismatches between the initiation of important life history events and resources when important resources rely on a weather cue.

Table 4.2. Questions and short explanation used to score species vulnerability (or potential adaptive capacity) to climate change impacts. See Bagne et al., (2011) for discussion on how these traits were selected and further discussion on how species are assigned a score of "v" vulnerable, "r" resilient, or "n" neutral.

---

13. Spatial or temporal separation between critical resources and life history stages. Risk of mismatch increases as distance between initiation of event and required resource increases. This could mean geographic distances as for distinct breeding or wintering grounds or refer to extended time as for species that have long gestation periods.

14. Can adjust timing of critical activities. Species may be able to breed throughout the year, a characteristic associated with increase resilience to timing mismatches. Species that have a narrow reproductive window (single event per year or short breeding season) are likely to be more vulnerable to seasonal changes in weather events and resource pulses.

Biotic interactions

15. Likelihood for decreased food resource. Species may rely on food resources that are known to change. Often, species with a diverse diet are given a neutral score.

16. Likelihood of increase predation. In cases where a population of a species is known to be limited by predation, we consider whether predation is likely to increase or decrease in the future.

17. Loss of important symbiotic species. Considers whether species have important symbiosis with other species and the likelihood that these interactions will become more or less frequent.

18. Increase in high mortality/morbidity disease. Considers whether species are or will be more or less susceptible to disease.

19. Increased competitive pressures. Considers whether climate changes might amplify or create new competitive interactions.

References

1. 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. 28 p. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO.
2. Bagne, K.E., and D.M. Finch. 2012. Vulnerability of species to climate change in the Southwest: threatened, endangered, and at-risk species at the Barry M. Goldwater Range, Arizona. Gen. Tech. Rep. RMRS-GTR-284. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO.
3. Bagne, K.E., and D.M. Finch. 2013. Vulnerability of species to climate change in the Southwest: threatened, endangered, and at-risk species at Fort Huachuca, Arizona. Gen. Tech. Rep. RMRS-GTR-302. U.S. Department of Agriculture, Forest Service Rocky Mountain Research Station. Fort Collins, CO.
4. Cameron, E.Z., Lemons, P.R., Bateman, P.W., and N.C. Bennett. 2008. Experimental alteration of litter sex ratios in a mammal. Proceedings of the Royal Society: Biological Studies 275: 323-327.
5. Coe, S., Finch, D., and M. Friggens. 2012. Applying a decision-support tool for assessing vulnerability of wildlife to climate change: a case study on the Coronado National Forest, AZ. USFS-GTR-273. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. Fort Collins, CO.
6. Entling MH, Schweiger O, Bacher S, Espadaler X, Hickler T, et al., (2012) Species Richness-Environment Relationships of European Arthropods at Two Spatial Grains: Habitats and Countries. PLoS ONE 7(9): e45875. doi:10.1371/journal.pone.0045875
7. Friggens, M., and M. Mathews. 2013. Risk-based framework and risk case studies. *In* VVose, J. M., Peterson, D. L., Patel-Weynand, T., eds. 2012. Effects of climatic variability and change on forest ecosystems: a comprehensive science synthesis for the U.S. Forest Sector. Gen. Tech. Rep. PNW-GTR-870. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 265 p.
8. Friggens, M., Bagne, K., Finch, D., Falk, D., Triepke, J. and A. Lynch. 2013. Review and Recommendations for Climate Change Vulnerability Assessment Approaches with Examples from the Southwest. *USFS-RMRS-GTR*-309. U.S. Department of Agriculture, Forest Service Rocky Mountain Research Station. Fort Collins, CO.
9. Friggens, M., Finch, D., Bagne, K., Coe, S., and D. Hawksworth. 2013. Vulnerability of Species to Climate Change in the Southwest: Terrestrial Species of the Middle Rio Grande. USFS- RMRS-GTR-306. U.S. Department of Agriculture, Forest Service Rocky Mountain Research Station. Fort Collins, CO.
10. Glick, P. Stein, B.A., and N. A. Edelson. 2011. Scanning the Conservation Horizon: A guide to climate change vulnerability assessment. National Wildlife Federation, Washington DC. 168p.
11. Huntley, B., Collingham, Y.C., Green, R. E., Hilton, G. M, Rahbek, C. and S. G. Willis. 2006. Potential impacts of climate change upon geographical distribution of birds. Ibis 148: 8-28.
12. Iverson et al. 2011. Lessons learned while integrating habitat, dispersal, disturbance, and life-history traits into species habitat models under climate change. Ecosystems 14: 1005-1020.
13. Liu, C., Berry, P. M., Dawson, T. P. and R. G. Pearson. 2005. Selecting thresholds of occurrence in the prediction of species distributions. Ecography 28: 385-393.

14. McRae, B. H., Schumaker, N. H., McKane, R. B., Busing, R. T., Solomon, A.M., and C. A. Burdick. 2008. A multi-model framework for simulating wildlife population response to land-use and climate change. Ecological Modeling 219: 77-91.
15. Merrow, C., Smith, M.J. and J.A. Silander. 2013 A practical guide to MaxEnt for modeling species' distributions: what it does, and why inputs and settings matter. Ecography 36: 001–012. (doi:10.1111/j.1600-0587.2013.07872.x)
16. O'Donnell, B. S., and D.A. Ignizio. 2012. Bioclimatic predictors for supporting ecological application in the conterminous United States: U.S. Geological Survey Data Series 69. 10p.
17. Phillips, S. J., Anderson, R. P, and Schapire R. E. 2006. Maximum entropy modeling of species geographic distributions. Ecological Modelling 190: 231-259.
18. Rehfeldt, et al., 2006. Empirical analyses of plant-climate relationships for the western United States. U.S. Department of Agriculture, Forest Service, RMRS. 167: 1123-1150.
19. Royle, J. et al., 2012. Likelihood analysis of species occurrence probability from presence-only data for modelling species distributions. – Methods Ecol. Evol. 3: 545–554.
20. Wisz, M.S., Hijmans, R. J., Li, J., Peterson, A. T., Graham, C. H., Guisan, A., and NCEASE Predicting Species Distributions Working Group. 2008. Effects of sample size on performance of species distribution models. Diversity and Distributions 14: 763-773.
21. Yackulic, C. B., Chandler, R. Zipkin, E. F., Royle, J.A., Nicols, J.D., Grant, E.H. C., and S. Veran. 2013. Presence-only modelling using MAXENT: when can we trust the inferences? Methods in Ecology and Evolution 4: 236-243.

SPECIES REPORTS

The following species reports contain summaries and discussion of niche model analysis, adjusted vulnerability scores, and fire risk predictions. Finalized version of these species reports, data layers and maps are available for download at the RMRS Project Website at: http://www.fs.fed.us/rm/grassland-shrubland-desert/research/projects/vulnerable-obligate-species/

Acronyms

WCRP	World Climate Research Programme
CMIP3	Coupled Model Intercomparison Project phase 3 multi-model dataset.
GCMs	General circulation models
CGCM3.1	Canadian Center for Climate Modeling and analysis (CCC)
Had_CM3	Met office, Hadley Centre (HAD)
GFDL_CM2.1	Geophysical Fluid Dynamics laboratory using the CM2.1 model

List of Species

Common Name	***Genus species***
Amphibians	
1. American bullfrog (I)	*Lithobates catesbeiana*
2. Northern leopard frog	*Lithobates pipiens*
Birds	
3. Lucy's warbler	*Oreothlypis luciae*
4. Southwestern willow flycatcher	*Empidonax traillii extimus*
5. Western yellow-billed cuckoo	*Coccyzus a. occidentalis*
Mammals	
6. Hispid cotton rat	*Sigmodon hispidus*
7. New Mexican meadow jumping mouse	*Zapus hudsonius luteus*
8. Occult bat	*Myotis occultus*
9. Long-legged bat	*Myotis volans*
10. Yuma bat	*Myotis yumanensis*
Reptiles	
11. Black-necked gartersnake	*Thamnophis cyrtopsis*
12. Western painted turtle	*Chrysemys p. belii*

American Bullfrog (*Rana catesbeiana*)

Overview

Climate Impacts	
Habitat Change	
2030	29-34% loss
2060	25-44% loss
2090	25-81% loss
Adaptive capacity	Moderate
Fire Response	Negative

Status:
The American bullfrog is considered of "least concern" by the IUCN Red List. This species is introduced to New Mexico where it is consider an invasive species able to outcompete native amphibian fauna.

Range and Habitat:
Widespread across North America (Figure 1), although introduced in the western part of the continent including the Middle Rio Grande in New Mexico. It occurs in mostly permanent waters and is often associated with dense vegetation. It also uses temporary pools near larger permanent bodies of water. In New Mexico, the bullfrog generally occurs below 2100 m (Degenhardt et al. 1996).

American bullfrogs breed in permanent water including ponds, lakes, rivers, and irrigation ponds. Some note ephemeral ponds are not suitable for breeding because of the duration of metamorphosis (Degenhardt et al. 1996), but others note variation in metamorphosis duration and some use of temporary ponds (Bury and Whelan 1984).

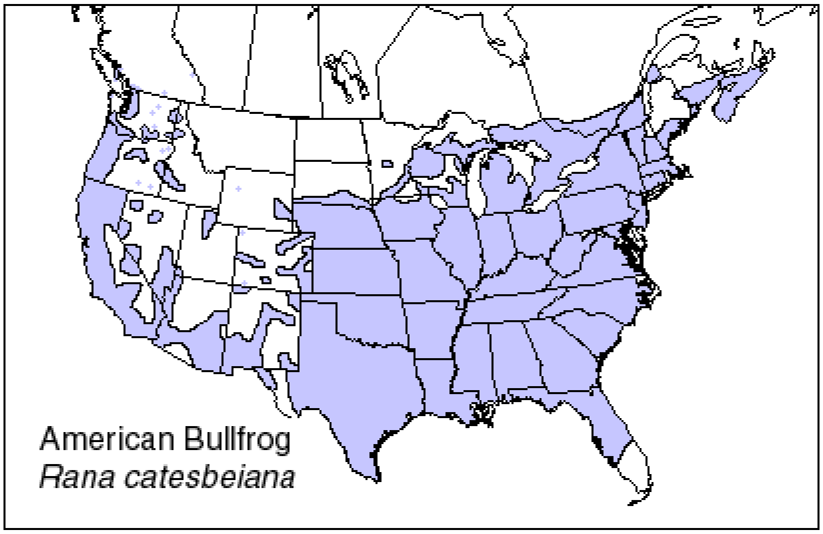


Figure 1. Distribution of *R. catesbeiana* in the U.S.

American Bullfrog (*Rana catesbeiana*)

Climate Change Impacts and Adaptive Capacity

Adaptive capacity score = - 1.4 (moderate)

The greatest issue facing bullfrogs under changing climate is their reliance on permanent or long lasting water bodies. Ephemeral ponds are typically not considered suitable because of the duration of metamorphosis, which takes 1-2 years (Degenhardt et al. 1996). However, others note variation in metamorphosis duration and some use of temporary ponds (Bury and Whelan 1984). Warmer water may benefit American bullfrog tadpolesas they show a preference for warm waters which allows them to develop more quickly and reduces their exposure to predators. The American bullfrog probably requires deep water for hibernation, especially where winter temperatures are low (Degenhardt et al. 1996). Frogs have been found in shallow water and terrestrial environments (Friet and Pinder 1990).

This species is not dispersal limited. The American bullfrog can move up to 159 m in one night and has been found at isolated temporary ponds (Degenhardt et al. 1996). Smith and Green 2005 reviewed dispersal distances of many amphibians and found many toads (*Bufo* spp.) and the American bullfrog are able to move up to and well over one kilometer. In the Middle Rio Grande, seasonal movements of the American bullfrog are probably limited and transitional habitats are not required. Overland movements are limited to periods with warm rains and drought may limit dispersal to new areas. However, rain is likely to be sufficient to allow dispersal in at least in some years.

Physiology:
The American bullfrog exhibits a number of potential positive attributes that may help it survive warming climates (Table 1). The Bullfrog has historically occupied habitats with a broad range of temperatures (Bury and Whelan 1984) and recent introductions and expansions into new habitat indicate a lack of limiting thresholds (Figure 1). American bullfrogs are considered warm adapted species that prefer temperatures above 26°C. (Bachmann 1969). However, it is important to note that egg development is impaired in water above 31°C (Degenhardt et al. 1996), a lower threshold than observed for native amphibian species. High water temperatures could reduce the reproductive window for bullfrogs during the summer months.

The American bullfrog does not appear to be overly affected by flooding or drought and none have been associated with direct mortality. Droughts, heat waves and more variable precipitation patterns could limit movements in dry conditions, but is unlikely to increase mortality. The American bullfrog is primarily nocturnal and thereby able to avoid extreme temperatures during the hottest periods of the day. The estimated lifespan of the American bullfrog is 8-10 years, allowing it to outlive average expected drought periods (5-7 years) when breeding opportunities may be limited.

Bullfrogs do not exhibit adaptations for resource limitations. The American bullfrog does hibernate in the winter and is generally not tolerant of freezing conditions (Degenhardt et al. 1996), but it is not clear that this is an advantage under warming conditions. Yearling American bullfrog males use alternative mating strategies and do not maintain territories when male-male competition is intense and instead attempt to intercept females attracted to larger males (Howard 1984). However, it is not clear that such a strategy would improve species response to climate impacts.

Phenology:

The American bullfrog appears quite resilient to event timing changes. Overland movements of the American bullfrog are associated with rainfall, but these movements are probably not critical to survival or fecundity in Middle Rio Grande habitat. This species does not rely on temperature or precipitation cues and there is little separation between events related to survival or fecundity for the American bullfrog. The American bullfrog has a prolonged reproductive period with a large variation in the dates of sexual receptivity for individual females and males remaining reproductively active throughout (Degenhardt et al. 1996). Females can produce two clutches per year.

Biotic Interactions:

The American bullfrog also appears to be quite resilient to changes in food resources and disease. The bullfrog has a diverse diet that includes a wide variety of invertebrates, snakes, rodents, frogs, and salamanders (Degenhardt et. al. 1996). The American bullfrog has various predators though it is relatively unpalatable to fish. Tadpoles are eaten by salamanders, other frogs, as well as adult bullfrogs. Chytridiomycosis is a major disease associated with amphibian declines. Bullfrogs are resistant to chytridiomycosis, but an efficient carrier of the pathogen (Daszak et. al. 2004). Though the bullfrog is susceptible to pollution and environmental toxins, tadpole malformation caused by toxins is not expected to change with climate change. American bullfrogs tend to have a competitive advantage over native frog species and bullfrog tadpoles generally outcompete native tadpoles (Kupferberg 1997).

Table 1. Scoring sheet for American Bullfrog (*Rana (Lithobates) catesbeiana).* “v” indicates a vulnerability or sensitivity, “r” indicates a resiliency or increased adaptive capacity, “n” indicates neutral response.

Question	Score	Notes
1. Changes to non-modeled habitat components	v	Pond Scarcity Possible
2. Change in habitat quality	n	Sensitivities Not Identified
3. Dispersal ability (Site fidelity or other limitations)	r	Good
4. Reliance on migratory or transitional habitats	n	No
5. Increase or decrease in physiological range limitation	n	No
6. Sex Ratios determined by temperature or food changes	n	No
7. Response to predicted extreme weather events/disturbances	n	No

Question	Score	Notes
8. Changes to daily activity period	n	No
9. Variable life history traits or coping strategies	n	No
10. Ability to outlive limiting conditions	r	Yes
11. Migrates/hibernates in response to weather cues	n	No
12. Reliance on weather mediated resource (e.g. insect emergence)	r	No
13. Spatial or temporal separation between critical resources and life history stages	r	No
14. Can adjust timing of critical activities	r	Yes
15. Likelihood for decreased food resource	n	Not Known
16. Likelihood of increase predation	n	Not Known
17. Loss of important symbiotic species	n	No
18. Increase in high mortality/morbidity disease	n	Not Known
19. Increased competitive pressures	n	Not Known

American Bullfrog (*Rana catesbeiana*)

Niche Model Analysis

Model AUC values were 0.95 for all three climate scenarios. Habitat is expected to decline under all three models though there is considerable variation among climate scenarios (Table 2, Figures 2 and 3).

Table 2. Percent of original predicted habitat under three climate models.

Time	CGCM3.1	GFDL CM2.1	Had_CM3	Average
2030	0.69	0.71	0.66	0.69
2060	0.72	0.56	0.55	0.61
2090	0.19	0.25	0.71	0.39

Lower elevation sites closer to water were the most important habitat for *R. catesbeiana* (Table 2). Overall, habitats with higher JJA precipitation and lower AI values were more suitable for *R. catesbeiana.* American bullfrogs were associated with areas with intermediate mean temperature in JJA. Generally habitats with higher winter precipitation (DJF) were more likely to be suitable in full model, but results based on models using just bio17 show the greatest likelihood of presence in areas with low values. Suitability was greatest for low elevation sites close to water across a range of slope values.

Table 3. Variables in order of importance for species habitat models. Relationship between variable and predicted suitability is listed as positive (+) negative (-) for linear and exponential curves or quadratic for parabola-like curves.

Variables	CGCM3.1	GFDL CM2.1	Had_CM3	Relationship
Distance to water	44.5	43.9	45.1	-
Elevation	43.3	42.9	43.6	-
Mean Temp JJA (bio11)	4.9	4.3	4.8	Quadratic
Biome	2.6	2.4	2.6	-ChiDeSCr, +PlnsGsslnd
AI (aridity index)	1.7	3.8	1.2	-
Slope	1.3	1.3	1.3	+
Precip JJA (bio16)	1.0	0.9	0.9	+
Precip DJF (bio17)	0.7	0.4	0.4	+

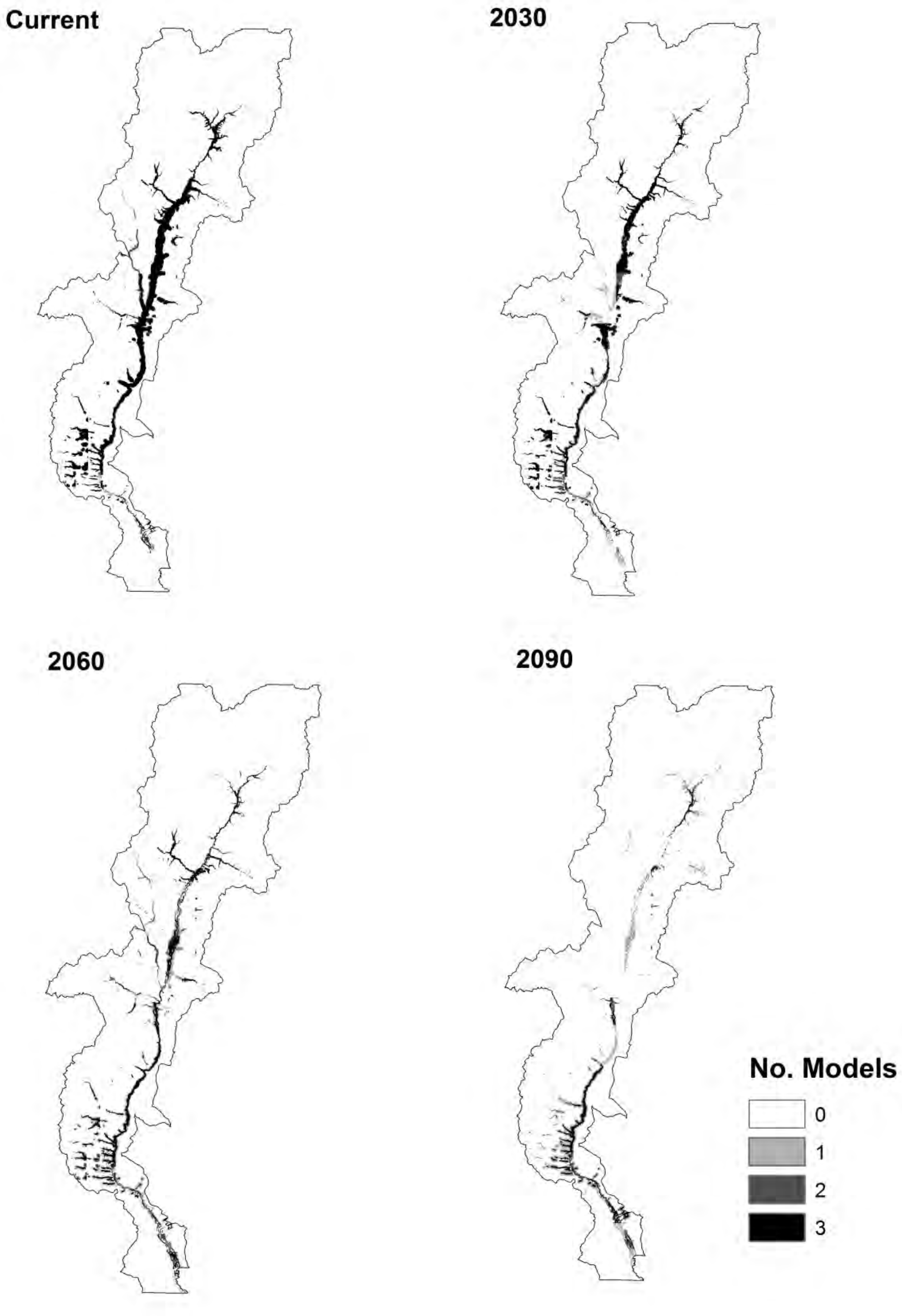


Figure 2. Distribution of suitable habitat for *Rana catesbeiana* under three climate futures. Habitat was modeled under three climate models: Had_CM3, CGCM3.1, and GFDL_CM2.1. Shading indicates the number of models that predict suitable habitat for a given area.

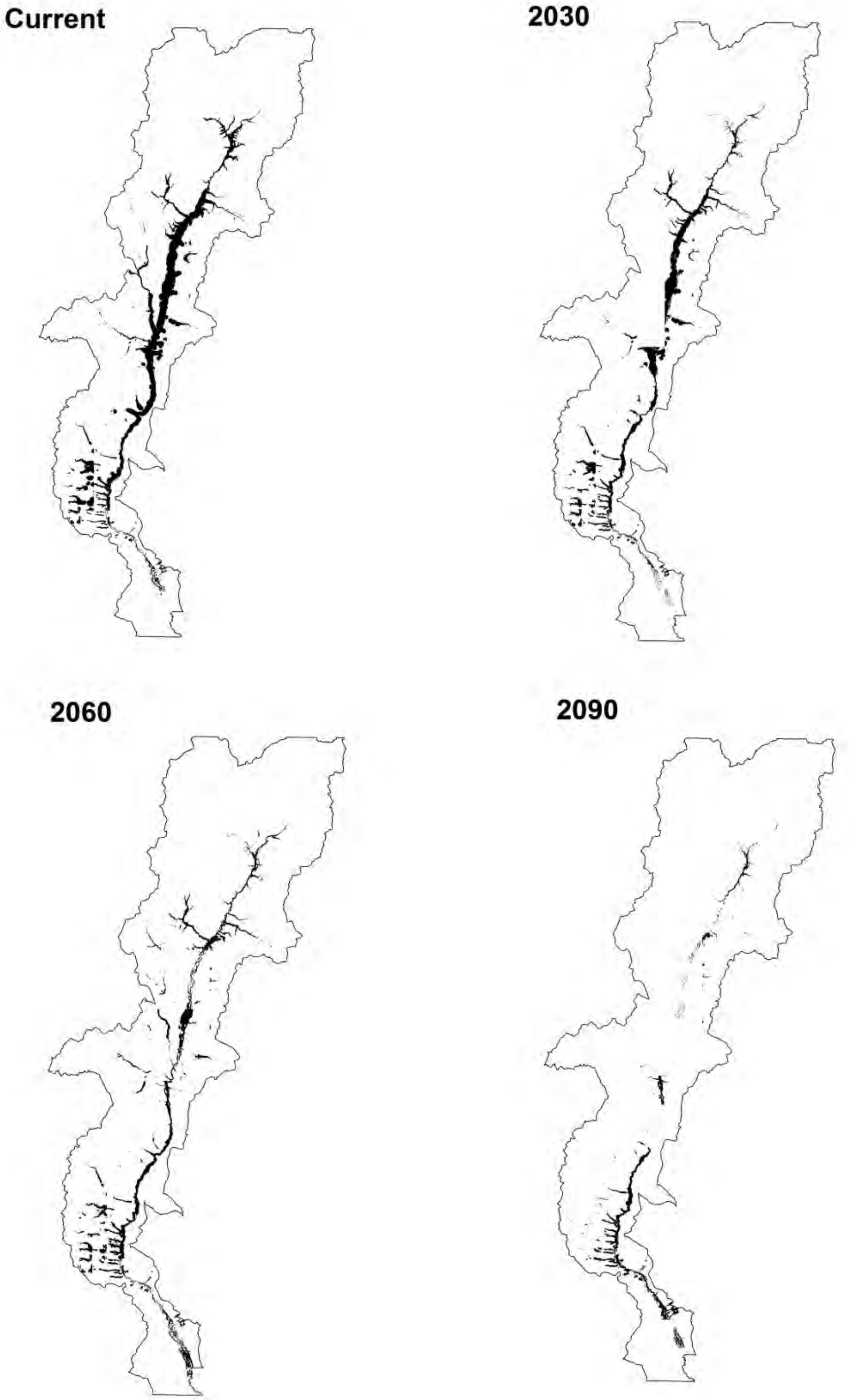


Figure 3. Two thirds consensus models of suitable habitat for *Rana catesbeiana* under three climate futures. Habitat was modeled under three climate models: Had_CM3, CGCM3.1, and GFDL_CM2.1. Black areas indicate predicted suitable habitat estimated by at least two models.

American Bullfrog (*Rana catesbeiana*)

Fire Impacts

The American Bullfrog may be indifferent to fire impacts in the short term and likely to benefit in the long term. However, its reliance on permanent ponds and association with dense vegetation was considered indicative of sensitivity to high intensity fires, which could result in increased sedimentation and inhibited reproductive success. Importantly, fire severity and resulting erosion and debris flows are not directly modeled in the current exercise. Negative effects are assumed for crowning fires within both shrub and forest habitats. These effects are largely manifest through increased likelihood of sedimentation and debris flows during egg and tadpole development. Though crowning fires are not necessarily going to lead to these negative impacts, they are an indication that fire could be problematic within areas labeled high impact. Less intense (non-crowning) fires are assumed to not have a negative impact on this species.

Long term effects were not deemed a significant source of risk or benefit and are not displayed.

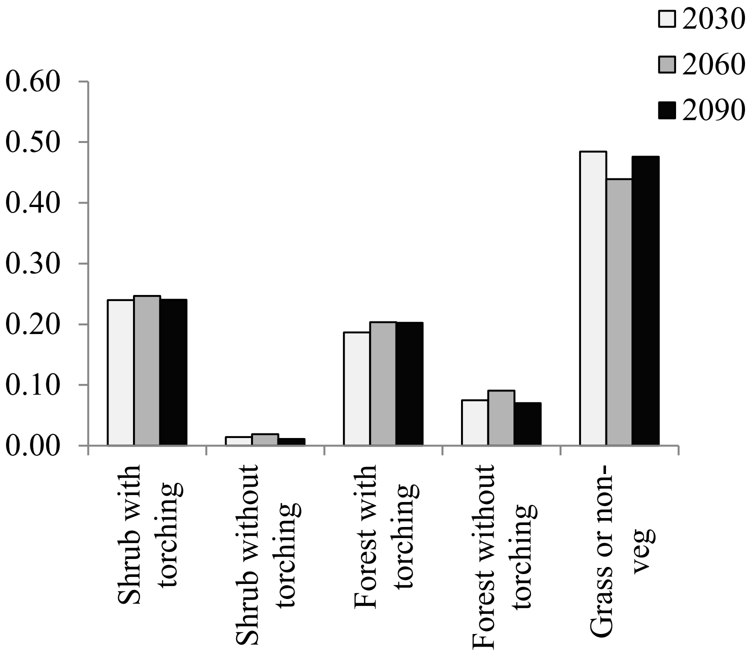
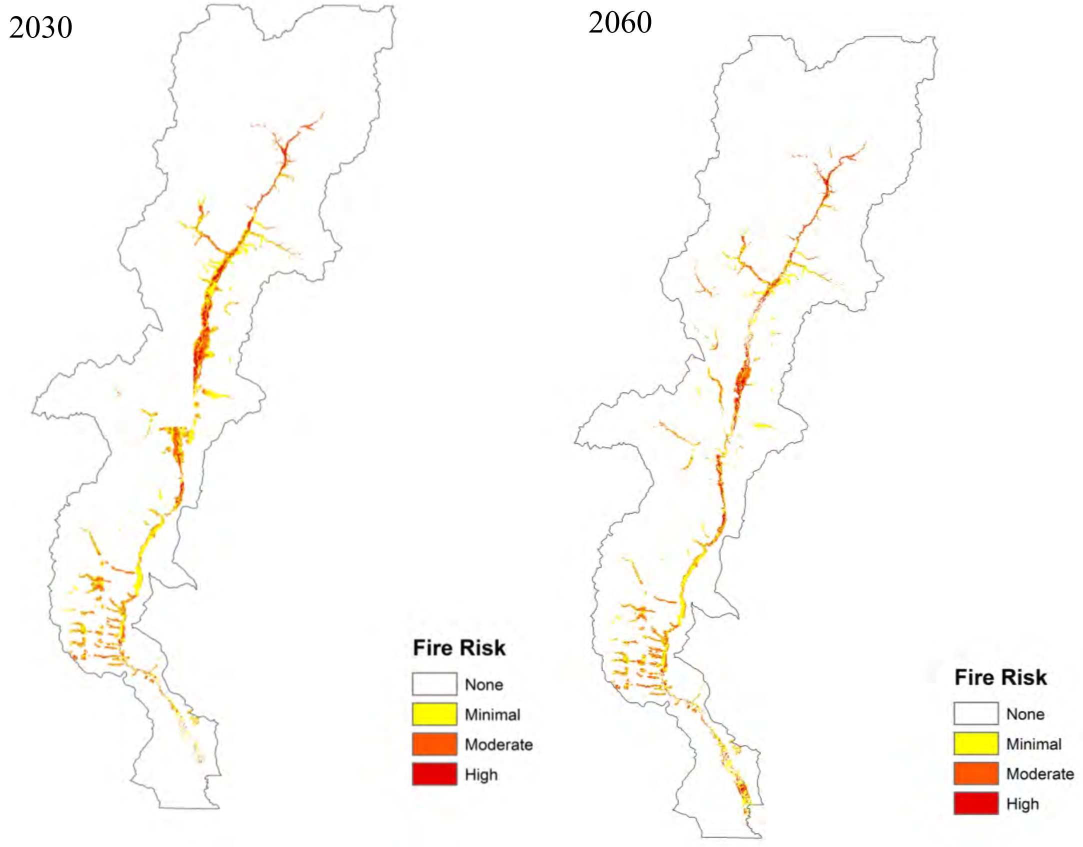


Figure 4. Percent of habitat falling within each fire type category.



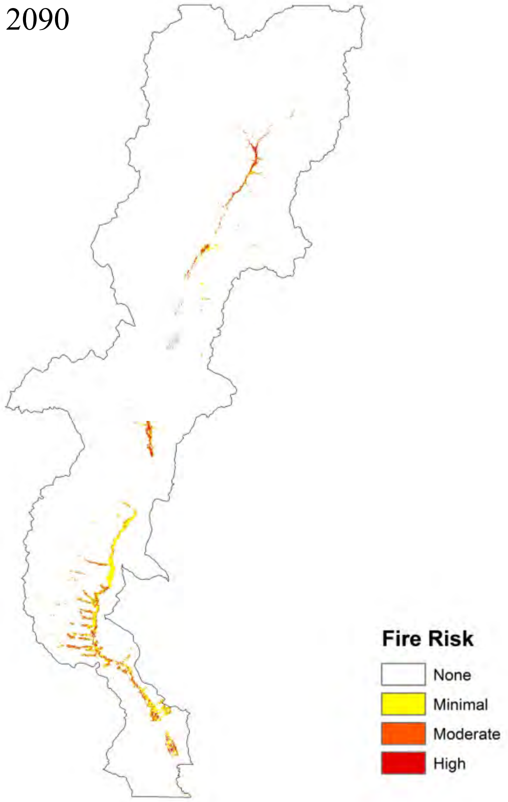


Figure 5. Fire impacts for *Rana catesbeiana* habitat. Fire intensity was calculated using FSim (Finney et. al. 2011) for each time period based on the GFDL-ESM-2m GCM under the RCP85 scenario. Fire risk maps were created by overlaying estimated species' habitat, predicted fire characteristic (shrub with or without torching, forest with or without torching or grass/nonveg) and species' response (negative, neutral, positive).

References

1. Bachmann, K. 1969. Temperature adaptations of amphibian embryos. American Naturalist 103: 115 – 130.
2. 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.
3. Degenhardt, G., C. Painter, and A. Price. 1996. Amphibians and Reptiles of New Mexico. UNM Press, Albuquerque, NM. 431 pp
4. Finney, M.A., C.W. McHugh, I.C. Grenfell, K.L. Riley, and K.C. Short. 2011. A Simulation of Probabilistic Wildfire Risk Components for the Continental United States. Stochastic Environmental Research and Risk Assessment 25: 973-1000.
5. Friet, S.C. and A.W. Pinder. 1990. Hypoxia during natural aquatic hibernation of the bullfrog. American Zoologist. 30:69A.
6. Friggens, M.M., D. M. Finch, K.E Bagne, S.J. Coe, and D.L. Hawksworth. 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.
7. Howard, R. 1983. Sexual selection and variation in reproduction success in a long-lived organism. American Naturalist 122: 301-325.
8. Howard, R. 1984. Alternative mating behaviors of young male bullfrogs. American Zoologist 24: 397-406
9. Kupferberg, S. 1997. Bullfrog (*Rana catesbeiana*) invasion of a California river: the role of larval competition. Ecology 78:1736-1751.
10. 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.

Northern Leopard Frog (*Rana pipiens*)

Overview

Impacts	
Habitat Change	
2030	-22% to +20%
2060	-10% to +50%
2090	-40% to +20%
Adaptive capacity	Very Low
Fire Response	Negative

Status:
The Northern leopard frog (*Rana pipiens*) is listed as "Least Concern" by the IUCN Red List (http://www.iucnredlist.org). However, this species is considered in decline, especially within the southern range of its habitat, due to habitat loss, introduced predators, disease and pollution (AmphibiaWeb).

Range and Habitat:
The Northern leopard frog occurs from Canada to Kentucky and is somewhat at the southern end of its distribution in New Mexico (Figure 1). This species has been introduced in California. Within New Mexico, it occurs at elevations from 1120 to 3050m (Degenhardt et. al. 1996).

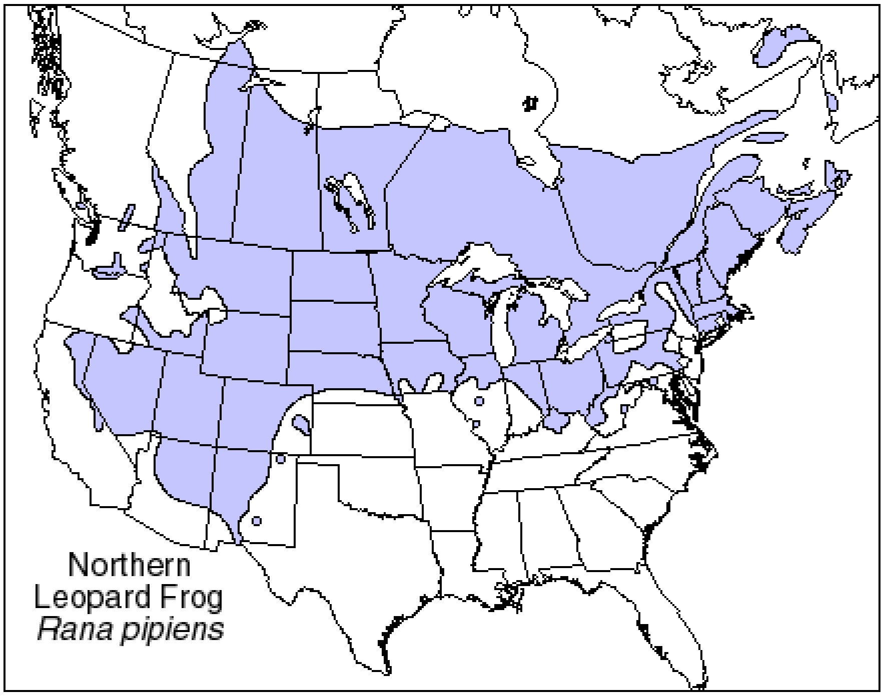


Figure 1. Distribution of *R. pipiens* in North America.

Northern Leopard Frog (*Rana pipiens*)

Climate Change Impacts and Adaptive Capacity

Adaptive Capacity score = 2.8 (very low)

Northern leopard frogs appear to have a limited capacity to cope with climate related changes (Table 1). Leopard frogs rely on limited resources within the habitats they occupy and are somewhat restricted in movements. Northern leopard frogs are mostly aquatic, occurring in permanent ponds and wetlands and are most commonly associated with water sources with dense aquatic vegetation (BISON-M 2009; Degenhardt et. al. 1996). They will use irrigation ditches, and, in summer months, may forage in wet meadows and unmowed fields (BISON-M 2009; Degenhardt et. al. 1996). Northern leopard frogs lay their eggs in still, shallow water, but also require permanent deep ponds, lakes, or streams for hibernation (BISON-M 2009). Permanent water bodies and ponds are likely to experience more rapid evaporation and loss under climate change, thus becoming rarer. Declines are expected in wetland vegetation where foraging occurs and deep water sources for hibernation (Friggens et. al. 2013). Though Northern leopard frogs can migrate 2-3 kilometers (BISON-M 2009), movements are largely restricted to riparian corridors. This species also exhibits high site fidelity; 98% of returned to their home pond after a one-kilometer displacement (Dole 1968).

Northern leopard frogs may also experience reduced habitat quality due to changes in the quantity of aquatic vegetation and introduced predators such as bullfrogs, fish, and crayfish (BISON-M 2009). Water fluctuations during breeding (mid-April-July and September-October in New Mexico (Degenhardt et. al. 1996) reduce reproductive success (Gilbert et. al. 1994), and these may be more likely with earlier and more rapid snowmelt and increased fire activity along the Rio Grande. The Northern leopard frog is resilient in that it does not require transitional or migratory habitats.

Physiology:

The Northern leopard frog does not exhibit many characteristics associated with increased adaptive capacity (Table 1). It is not prone to temperature dependent sex ratios and is not likely to experience activity limitations in response to increasing temperatures. However, this species has experienced reproductive failure following pond drying during drought (BISON-M 2009) With a lifespan of 6 years in captivity, it is assumed that most Northern leopard frogs (BISON-M 2009) would not outlast droughts of more than 5 years.

Phenology:

The Northern leopard frog is not dependent upon discretely timed resources and responds to its immediate environment, which is considered more adaptable under changing climate regimes. However, this species initiates breeding in response to temperature cues and produces a single clutch of eggs per year (BISON-M 2009), which increases its vulnerability to climate related timing mismatches and clutch failures.

Biotic Interactions:

The Northern leopard frog is a generalist and eats a wide variety of invertebrates and some vertebrates lending it some adaptive capacity for dealing with shifts in resource availability. However, Northern leopard frogs are vulnerable to predation and competition by the American bullfrog and introduced fish. These exotic species are blamed in part for population declines in the Northern Leopard frog in New Mexico. Introduced crayfish eat tadpoles are also considered a major impact (BISON-M 2009). Bullfrogs are expected to be tolerant of changing conditions and a likely source of increased sensitivity for the Northern leopard frog (Friggens et. al. 2013). Northern leopard frogs are also prone to increased disease. Declines of populations in Arizona and Colorado may have been in part due to Chytridiomycosis (*Batrachochytrium dendrobatidis*) (Bradley et. al. 2002; Milius 1998). Chytridiomycosis may increase with crowding in permanent water sources. Populations will also be at an increase of extirpation due to pollution as climate limits alternative habitats for this species. Bullfrogs also compete for food and crayfish remove vegetation associated with foraging causing significant issues for the Northern Leopard frog (BISON-M 2009). Crayfish are likely tolerant of climate change as they can forage from multiple trophic levels.

Table 1. Scoring sheet for Northern Leopard Frog (*Rana pipiens*). "v" indicates a vulnerability or sensitivity, "r" indicates a resiliency or increased adaptive capacity, "n" indicates neutral response.

Question	Score	Notes
1. Changes to non modeled habitat components	v	Loss of ponds
2. Change in habitat quality	v	Increases in exotic predators
3. Dispersal ability (Site fidelity or other limitations)	v	Limited to moist environ
4. Reliance on migratory or transitional habitats	n	No
5. Increase or decrease in physiological range limitation	n	No
6. Sex Ratios determined by temperature or food changes	n	No
7. Response to predicted extreme weather events/disturbances	v	
8. Changes to daily activity period	n	No
9. Variable life history traits or coping strategies	n	No
10. Ability to outlive limiting conditions	v	Shortlived
11. Migrates/hibernates in response to weather cues	v	Breeding cues to temp
12. Reliance on weather mediated resource (e.g. insect emergence)	n	No specific need
13. Spatial or temporal separation between critical resources and life history stages	r	Responds to immediate environment
14. Can adjust timing of critical activities	v	No, single clutch per year

Question	Score	Notes
15. Likelihood for decreased food resource	n	Not predicted at this time
16. Likelihood of increase predation	v	American Bullfrogs/ crayfish
17. Loss of important symbiotic species	n	None known
18. Increase in high mortality/morbidity disease	v	Chytrid presence at higher elevations
19. Increased competitive pressures	v	American Bullfrogs

Northern Leopard Frog (*Rana pipiens*)

Niche Model Analysis

The models performed well with AUC values of 0.88, 0.91, and 0.95 for GFDL_CM2.1, HadCM3 and CGCM3.1, respectively. Model results were mixed however, with CGCM3.1 predicting an increase of 25% by the end of the century, no measurable change under GFDL_CM2, and a 40% reduction under Had_CM3 (Table 2, Figures 2 and 3).

Table 2. Percent of original predicted habitat under three climate models.

Time	CGCM3.1	GFDL CM2.1	Had_CM3	Average
current	1	1	1	1
2030	1.19	0.89	0.78	0.95
2060	1.53	1.48	0.90	1.30
2090	1.25	1.06	0.60	0.97

Rana pipiens habitat tended to be associated with riparian habitats within more arid areas, with moderate values for Isothermality and potential evaporation of natural vegetation and peaking at higher winter minimum temperatures (Table 3). Suitability increased for habitats with either very low values or very high runoff values.

Table 3. Variables in order of importance for species habitat models. Relationship between variable and predicted suitability is listed as positive (+) negative (-) for linear and exponential curves or quadratic for parabola-like curves.

Variable	CGCM3.1	GFDL CM2.1	Had_CM3	Relationship
Distance to water	57.2	56.1	56.2	-
Annual runoff	18.0	19.2	19.1	Peaks at low and high values
Biome	13.3	13.0	13.2	+ChiDsScrb, PlnsGrss, GrtBaCnWd, SemidGrss, MMconFor
Min Temp Jan (bio6)	5.2	4.73	5.4	Quadratic
Isothermality (bio3)	2.5	4.4	3.23	Peaks at low values
Pot Evap Nat Veg	2.0	1.7	1.6	Peaks at low values
AI	1.7	0.9	1.4	+

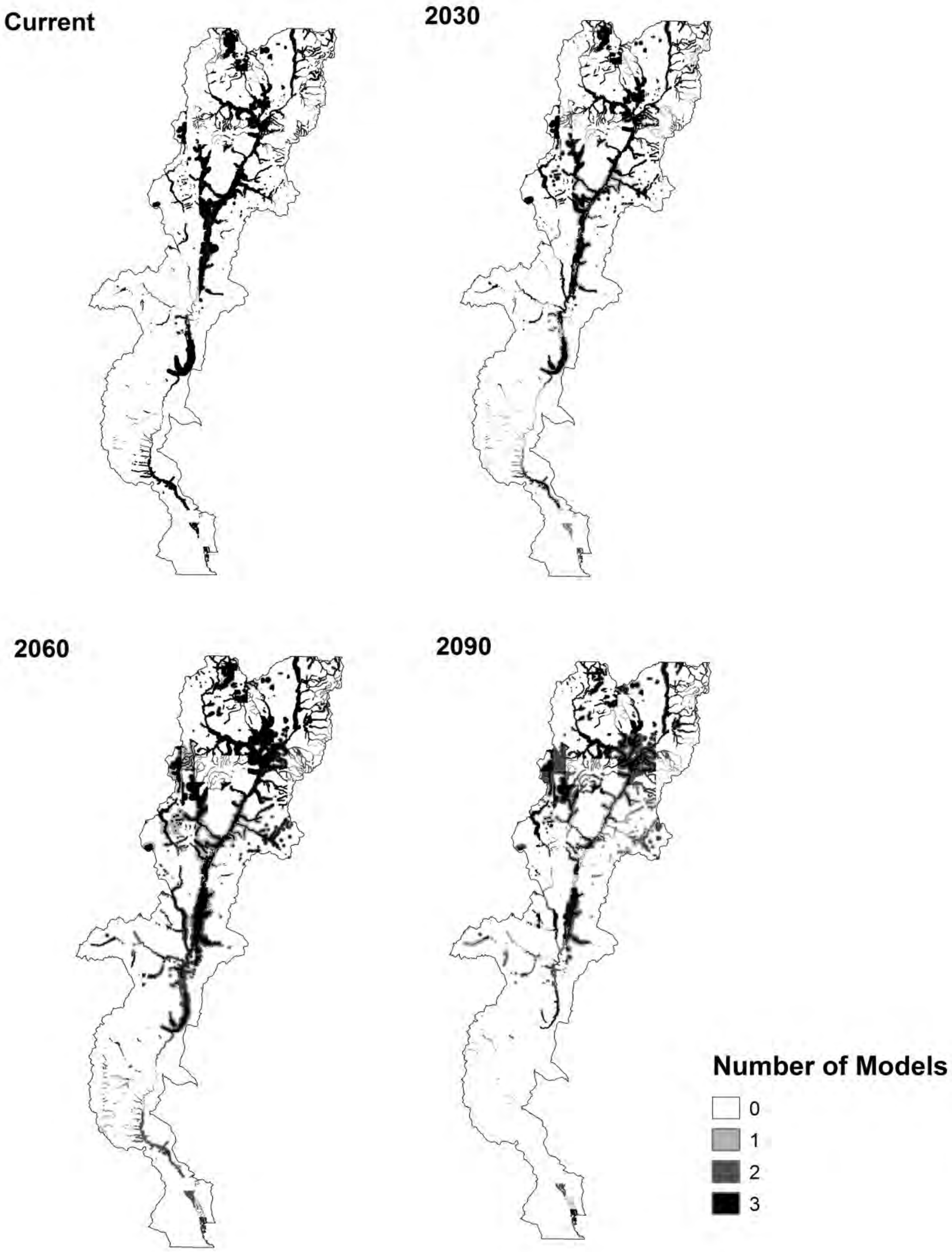


Figure 2. Distribution of suitable habitat for *Rana pipiens* under three climate futures. Habitat was modeled under three climate models: Had_CM3, CGCM3.1, and GFDL_CM2.1. Shading indicates the number of models that predict suitable habitat for a given area.

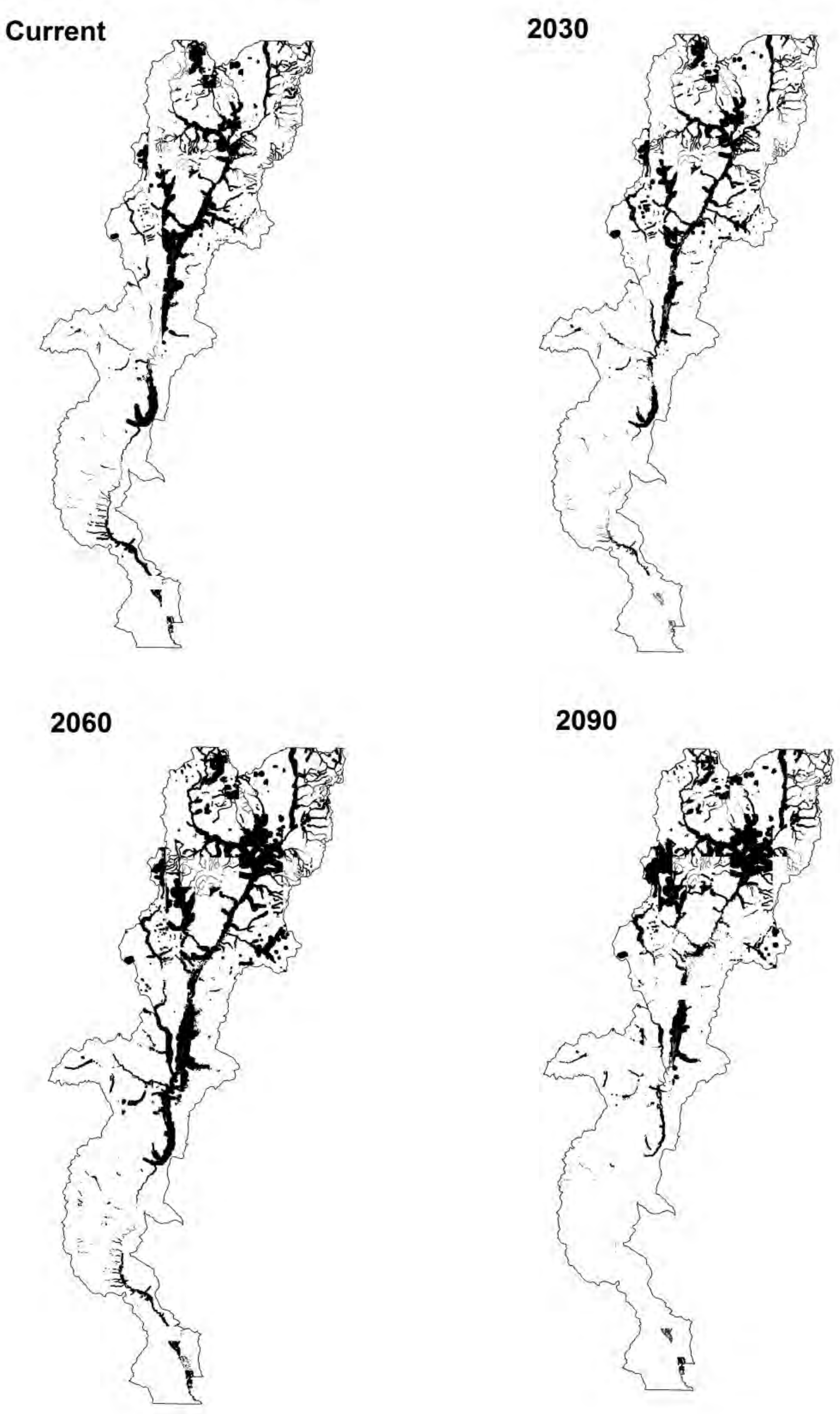


Figure 3. Two thirds consensus models of suitable habitat for *Rana pipiens* under three climate futures. Habitat was modeled under three climate models: Had_CM3, CGCM3.1, and GFDL_CM2.1. Black areas indicate predicted suitable habitat estimated by at least two models.

Northern Leopard Frog (*Rana pipiens*)

Fire Impacts

The reliance of the Northern leopard frogs on permanent ponds and association with dense vegetation was considered indicative of sensitivity to high intensity fires, which could cause increased sedimentation and loss of thermal cover. Importantly, sedimentation and fire severity are not directly modeled in the current exercise. Negative effects are assumed for crowning fires within both shrub and forest habitats. These effects are largely manifest through increased likelihood of sedimentation and debris flows during egg and tadpole development. Though crowning fires are not necessarily going to lead to these negative impacts, they are an indication that fire could be problematic within areas labeled high impact. Less intense (non-crowning) fires are assumed to not have a negative impact on this species.

Long term effects were not deemed a significant source of risk or benefit and are not displayed.

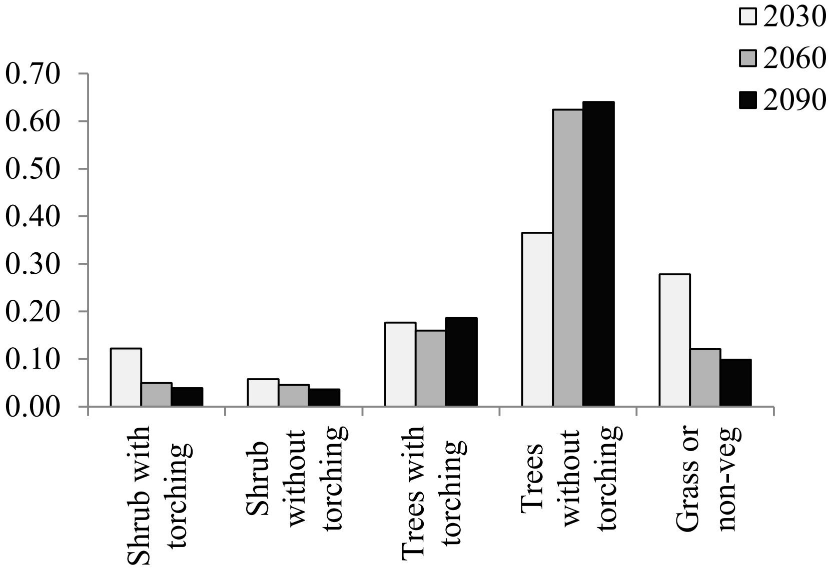
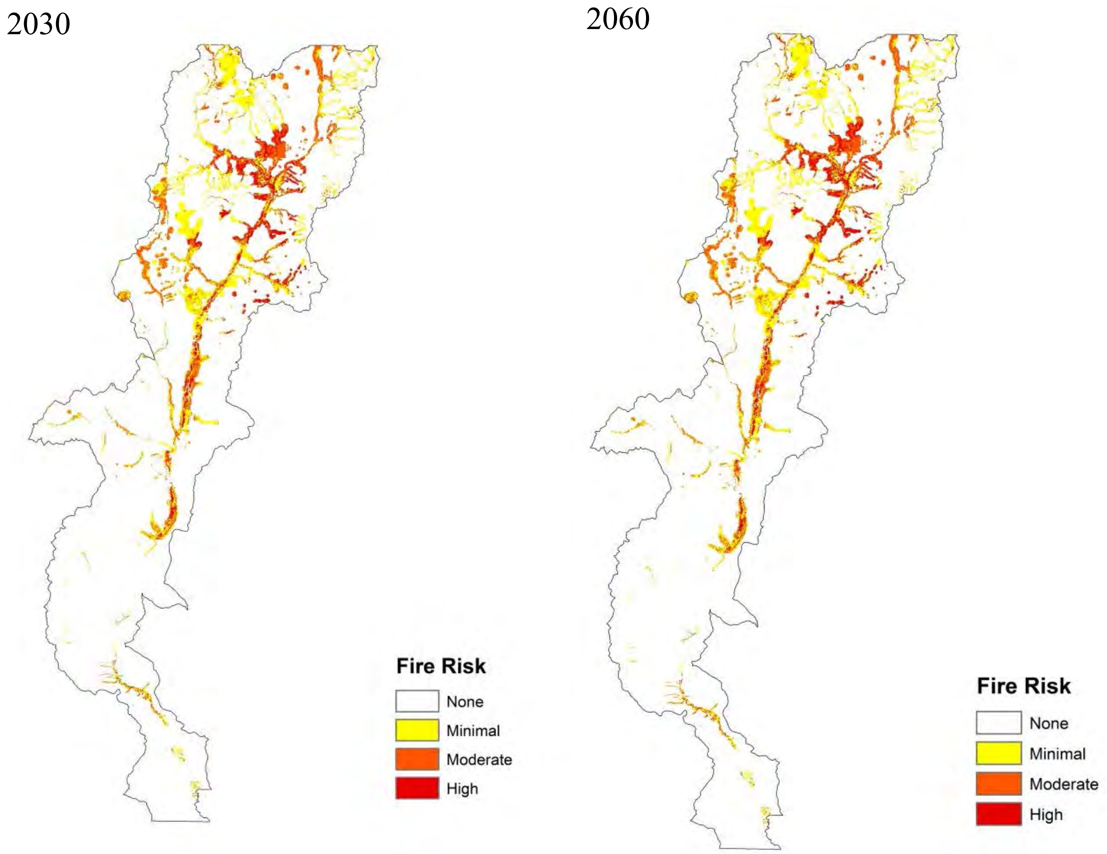


Figure 4. Percent of habitat falling within each fire type category.



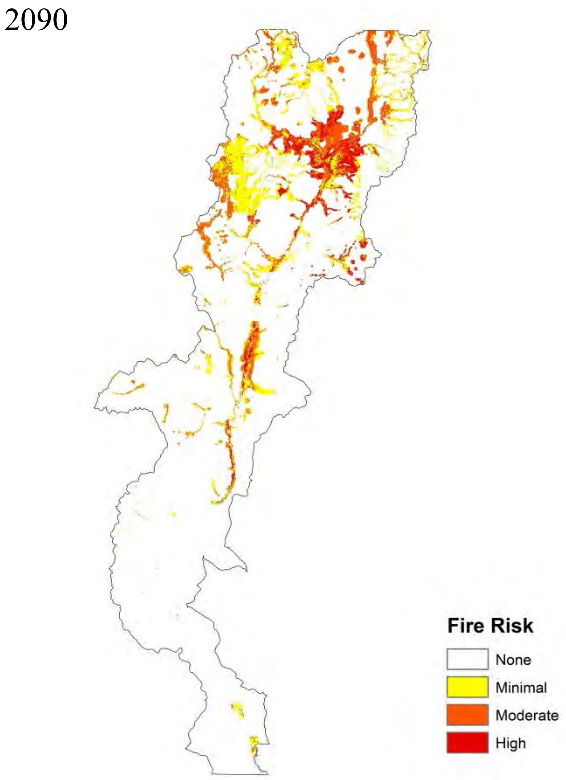


Figure 5. Fire impacts for *Rana pipiens* predicted habitat. Fire intensity was calculated using FSim (Finney et. al. 2011) for each time period based on the GFDL-ESM-2m GCM under the RCP85 scenario. Fire risk maps were created by overlaying estimated species' habitat, predicted fire characteristic (shrub with or without torching, forest with or without torching or grass/nonveg) and species' response (negative, neutral, positive).

References:

1. BISON-M. 2009. Biotic Information System of New Mexico. New Mexico Game and Fish Department. <http://www.bison-m.org>
2. 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
3. Carey, C., N. Cohen, and L. Rollins-Smith. 1999. Amphibian declines: an immunological perspective. Developmental & Comparative Immunology 23: 459-472.
4. Coleman, S. 2011. Thamnophis rufipunctatus and Thamnophis eques information post-Wallow Fire. White paper drafted by Stephanie Colemen, Apache-Sitgreaves National Forest. 4 pp.
5. Degenhardt, G., C. Painter, and A. Price. 1996. Amphibians and Reptiles of New Mexico. UNM Press, Albuquerque, NM. 431 pp.
6. Dole, J.W. 1968. Homing in leopard frogs, *Rana pipiens*. Ecology 49:386-399.
7. Emery, A. R., A. H. Berst, and K. Kodaira. 1972. Under-Ice Observations of Wintering Sites of Leopard Frogs. Copeia 1: 123-126.
8. Finney, M.A., C.W. McHugh, I.C. Grenfell, K.L. Riley, and K.C. Short. 2011. A Simulation of Probabilistic Wildfire Risk Components for the Continental United States. Stochastic Environmental Research and Risk Assessment 25: 973-1000.
9. Friggens, M.M., D. M. Finch, K.E Bagne, S.J. Coe, and D.L. Hawksworth. 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.
10. Gilbert, M., R. Leclair, Jr., and 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.
11. 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.
12. Hossack, B. R., Pilliod, D. S., 2011, Amphibian responses to wildfire in the western United States- Emerging patterns from short-term studies. Fire Ecology, v. 7, no. 2, p. 129-144
13. Milius, S. 1998. Fatal skin fungus found in U.S. frogs. Science News 154: 7.
14. Russell, K.R., D.H. Van Lear, and D.C. Guynn, Jr. 1999. Prescribed fire effects on herpetofauna: Review and Management implications. Wildlife Society Bulletin 27: 374-384.
15. Smith, B.E. and D.A. Keinath. (2007, January 16). Northern Leopard Frog (Rana pipiens): a technical conservation assessment. [Online]. USDA Forest Service, Rocky Mountain Region. Available at: http://www.fs.fed.us/r2/ projects/scp/assessments/northernleopardfrog.pdf.
16. Werner, E. and K. Glennemeier. 1999. Influence of Forest Canopy Cover on the Breeding Pond Distributions of Several Amphibian Species. Copeia 1999: 1-12.

Lucy’s Warbler (*Oreothlypis luciae*)

Overview

Predicted Impacts	
Habitat Change	
2030	62-68% Loss
2060	77-84% Loss
2090	72-82% Loss
Adaptive capacity	Low
Fire Response	Negative

Status:
Lucy’s warblers are classified a “least concern” by the IUCN red list. Within New Mexico it is considered a species of Greatest Conservation Need (NMDGF, 2006) and Natural Heritage New Mexico ranks it as SB3 (Breeding population is vulnerable). Lucy’s warblers are listed as threatened in Mexico.

Range and Habitat: Lucy’s Warbler (*Oreothlypis luciae*)
The Middle Rio Grande is the far northwestern portion of the Lucy’s warbler’s range (Figure 1). Lucy’s are found in desert and riparian areas of the southwestern United States and northwestern Mexico (Johnson et al., 1997). This species occurs in driest habitat of breeding Southwest warblers (Johnson et al., 1997). In most of its range, Lucy’s warblers are most often found in mesquite woodlands. They also breed in cottonwood-willow riparian woodlands, sycamore-oak woods and salt cedar stands (Johnson et al., 1997). In the Middle Rio Grande, Lucy’s warblers breed mostly in cottonwood Bosque rather than mesquite.

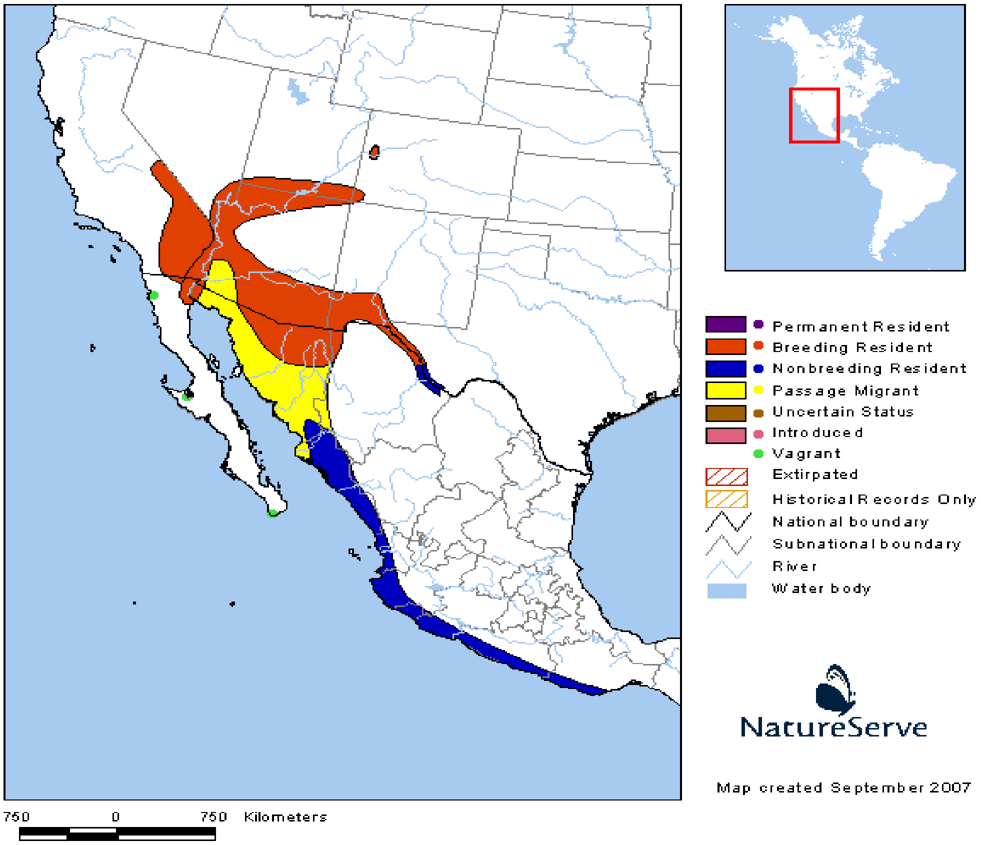


Figure 1. Range of *Oreothlypis luciae.*

Lucy's Warbler (*Oreothlypis luciae*)

Climate Change Impacts and Adaptive Capacity

Adaptive Capacity Score = 1.3 (Low)

Lucy's warblers exhibit more traits associated with increased vulnerability to climate change than traits associated with increase coping potential (Table 1). Lucy's warblers require cavities or "pseudo-cavities" for nesting. They will use natural cavities, tree cracks, openings behind pieces of loose bark, abandoned woodpecker holes, abandoned verdin nests, and rarely locations such as bank burrows, abandoned cliff swallow nests, exposed root tangles, etc. (Johnson et al., 1997). Large cottonwoods and willows were important nesting sites in similar habitats in southern New Mexico and were associated with habitat use (Stoleson et al., 2000). These species are considered less tolerant of water table declines and were expected to decline due to climate impacts within the Middle Rio Grande (Friggens et al., 2013).

The highest densities of Lucy's warblers are reached in closed canopy riparian mesquite habitats, but also dense populations in salt cedar thickets and cottonwood (Johnson et al., 1997). It is not known if this association is related to thermal or predator protection but there is a strong likelihood for reductions in habitat features along the Rio Grande. Lucy's Warblers are long distance migrants that winter along the Pacific Coast of Mexico (Johnson et al., 1997). This species may also rely on stopover sites as transients have been found in various locations in northwest Mexico. Reliance on multiple, spatially distinct sites is associated with an increased risk of negative impacts as migrating animals are exposed to a greater range of conditions and likelihood of habitat change. Conversely, migrant species are able to disperse to new habitats, an important trait for coping with potential climate changes.

Physiology:

Though Lucy's warblers appear to be quite resilient to hot climates there are some indications that this species may be near its physiological tolerances: It is the smallest of the wood warblers, thus more prone to physiological limitations; Lucy's warblers migrate early, which is thought to be a mechanism for avoiding seasonal extreme temperatures (Stoleson et al., 2000). However, among the habitats in which it currently resides, the Rio Grande is not the hottest or driest conditions known within its range. We identified Lucy's warblers as adapted for hot climate (unlikely to reach physiological limit) but prone to mortality from extreme events that are expected to increase including heat, drought, and intense rainfall due to their small size (Table1). We also consider this species as sensitive to resource variations (no torpor or other mechanisms for reducing metabolic requirements), and unlikely to outlive potentially limiting conditions (drought). Banding data shows Lucy's Warblers can live at least 5 years (Johnson et al., 1997). Recent droughts, which may limit reproduction, had duration of 5 years and this species was not considered likely to outlive potential limiting conditions.

Phenology:

Cues to initiate migration or breeding in Lucy's warblers are not known. Lucy's warblers leave breeding grounds earlier than most migrants and are generally gone by beginning of September (Johnson et al., 1997). In southern New Mexico in similar habitats, Lucy's warblers were mostly gone by the beginning of July (Stoleson et al., 2000). This species also migrates early in Arizona, arriving in early March. Males arrive before females, but overall individual arrival is synchronous occurring within a short period (Johnson et al., 1997). Advantages of this strategy are unknown, but avoidance of summer heat has been suggested. Warming means high temperatures will likely extend into current breeding season and thus may not be avoided without concurrent timing changes in breeding. This species is also at risk due to a moderate level of temporal and geographic separation between wintering grounds and initiation of breeding. Lucy's warblers are believed to raise two broods per year (Johnson et al., 1997), a trait that confers greater adaptation to changing conditions.

Biotic Interactions:

There were no additional sensitivities or potential adaptations that might influence Lucy's warbler interactions under climate change (Table 1). Lucy's warblers feed exclusively on insects and small arthropods year round (Johnson et al., 1997). Important prey items include leafhoppers, caterpillars, beetles, true bugs, and spiders (Yard 1996). Though they specialize in gleaning food from leaves and branches of trees and shrubs, we cannot at this time predict an overall trend for these food resources. Overall, prey should remain available though taxa may vary. Eggs and nestlings have been reported as prey for snakes, wood rats, lizards, and Gila Woodpecker but these are not known to drive population changes within the warbler. Lucy's warblers use verdin and woodpecker nests, but not wholly dependent on these species and the warbler will nest in a wide variety of natural cavities and crevices. It is believed Lucy's Warblers may be more susceptible to body parasites than other warblers since they are cavity nesters and frequently reuse nest sites, but no information on associated mortality (Johnson et al., 1997). Lucy's are reported as a common cowbird host (Friedmann and Kiff 1985), but no brood parasitism observed in RMRS data for Middle Rio Grande (n = 10). There are also no estimates of mortality related to this parasitism.

Table 1. Score sheet for *Oreothlypis luciae.* "v" indicates a vulnerability or sensitivity, "r" indicates a resiliency or increased adaptive capacity, "n" indicates neutral response.

Question	Score	Notes
1. Changes to nonmodeled habitat components	v	Closed canopy species
2. Change in habitat quality	n	None known
3. Dispersal ability (Site fidelity or other limitations)	r	Good
4. Reliance on migratory or transitional habitats	v	Yes
5. Increase or decrease in physiological range limitation	n	Not Predicted
6. Response to predicted extreme weather events/disturbances	v	Drought Sensitive
7. Changes to daily activity period	n	Not likely
8. Variable life history traits or coping strategies	v	No torpor
9. Ability to outlive limiting conditions	v	No
10. Sex Ratios determined by temperature or food changes	n	No
11. Migrates/hibernates in response to weather cues	n	No
12. Reliance on weather mediated resource (e.g. insect emergence)	v	Reproduction timed to insects
13. Spatial or temporal separation between critical resources and life history stages	v	Yes, long distance migrant
14. Can adjust timing of critical activities	r	Yes, multiple clutches
15. Likelihood for decreased food resource	n	Not Predicted
16. Likelihood of increase predation	n	Not Predicted
17. Loss of important symbiotic species	n	Not Predicted
18. Increase in high mortality/morbidity disease	n	Not Predicted
19. Increased competitive pressures	n	Not Predicted

Lucy’s Warbler (*Oreothlypis luciae*)

Niche Model Analysis

Model AUC values were 0.96, 0.97, and 0.96 for CGCM3, GDFLCM2 and HadCM3, respectively. Lucy’s warbler habitat declines under all three models (Table 1, Figs. 2, 3) with much of the loss occurring in the near future and little change after 2060.

Table 2. Percent of original predicted habitat under three climate models.

Time	CGCM3.1	GFDL CM2.1	Had_CM3	Average
current	1	1	1	1
2030	0.38	0.33	0.32	0.34
2060	0.21	0.16	0.23	0.20
2090	0.27	0.18	0.25	0.23

Elevation, distance to water and diurnal variation in temperature were the most important variables for predicting suitable habitat for Lucy’s warblers (Table 3). In general, suitable habitat was associated with areas found nearer water and at lower elevations. Suitability of habitat was greater for areas that experienced greater range of diurnal temperatures and intermediate values of July maximum temperatures. Suitability was negatively related to low potential evapotranspiration of natural vegetation but increased with increasing values above a certain threshold.

Table 3. Variables in order of importance for species habitat models. Relationship between variable and predicted suitability is listed as positive (+) negative (-) for linear and exponential curves or quadratic for parabola-like curves.

Variables	CGCM3.1	GFDL CM2.1	Had_CM3	Relationship
Distance to Water	56.06	56.80	55.63	-
Mean Diurnal Temp Range (bio2)	15.08	13.10	14.50	+
Elevation	13.41	15.30	14.68	Peaks at low values then -
Max Temp JJA (bio5)	5.28	5.70	5.47	+ Peaks at high values
Pot Evap Natural Veg	4.98	4.60	4.20	+ Peaks at intermediate values
Biome	2.65	2.30	2.95	+ ,-
Slope	1.75	1.20	1.57	-
Isothermality (bio3)	0.59	0.50	0.65	Peaks at low and high values
Precip JJA(bio18)	0.10	0.20	0.11	-
Annual Precip	0.05	0.30	0.19	Peaks at low values
Aridity Index	0.03	0.00	0.05	Peaks at low values then -
Precip August (bio13)	0.01	0.00	0.01	Peaks at moderately low values

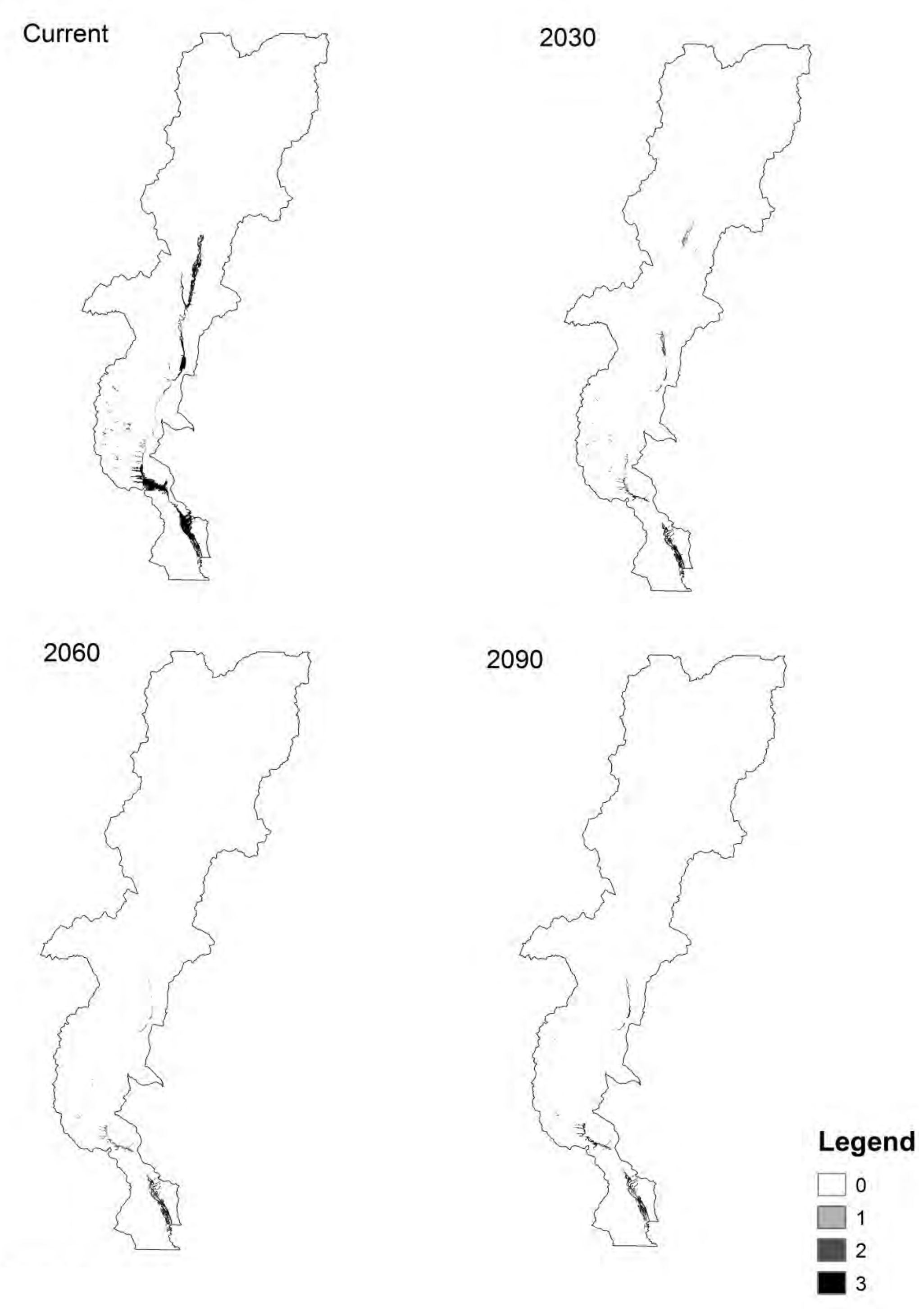


Figure 2. Distribution of suitable habitat for *Oreothlypis luciae* for three future time periods. Future habitat predictions are based on three climate models: Had_CM3, CGCM3.1, and GFDL_CM2.1. Shading indicates the number of models that predict suitable habitat for a given area.

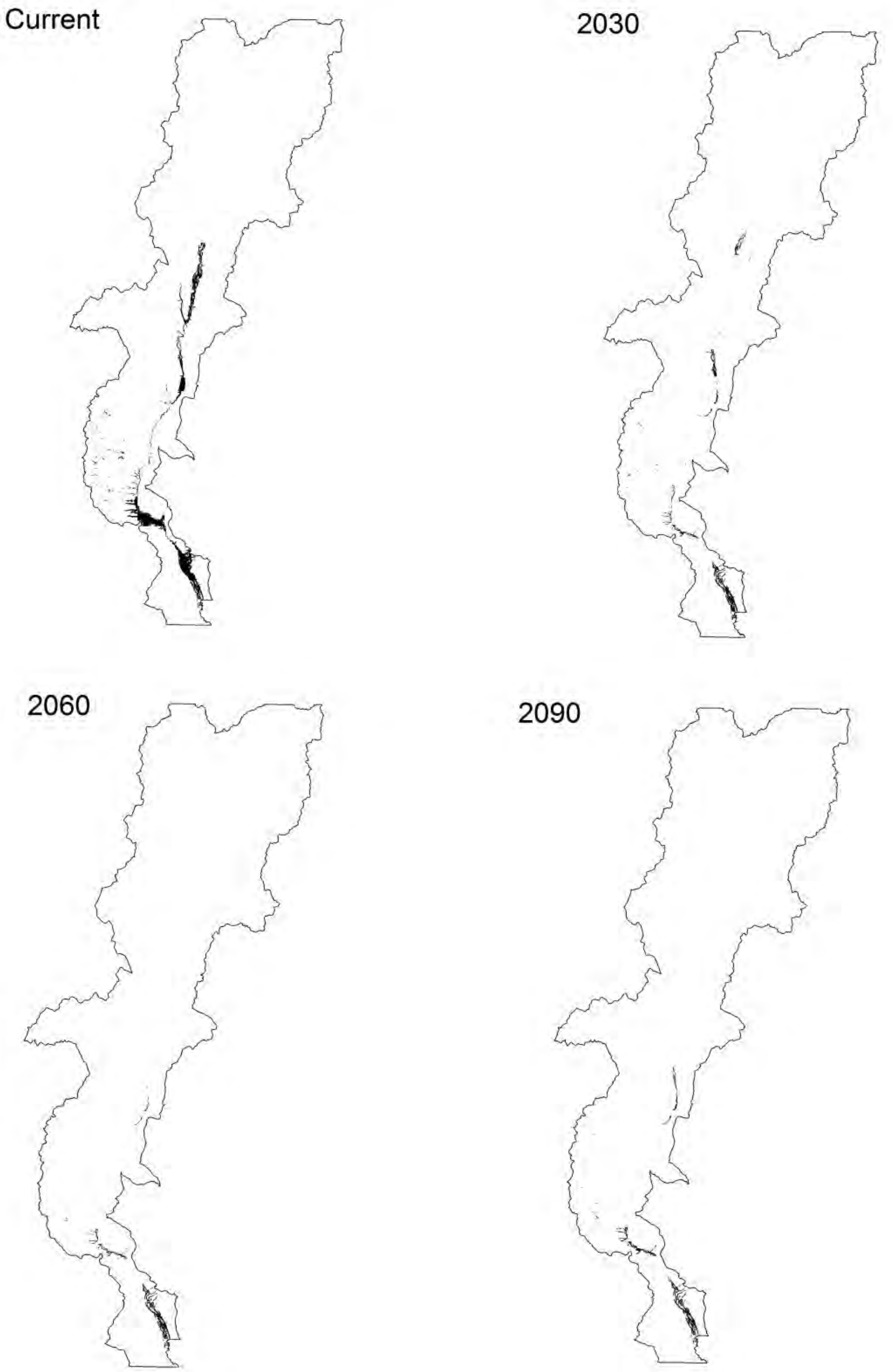


Figure 3. Two-thirds consensus models for suitable habitat for *Oreothlypis luciae* for three future time periods. Future habitat is predicted where at least two of three climate models, Had_CM3, CGCM3.1, and GFDL_CM2.1, identify suitable habitat.

Lucy's Warbler (*Oreothlypis luciae*)

Fire Analysis

Fire can alter bird habitat by changing vegetation composition and structure which influences nest site availability, foraging substrates, and nest predation rates. Prescribed or wildfire during breeding season is likely to reduce habitat for shrub and ground dwelling species (Finch et al., 1997). Loss of snags due to fire will harm cavity nesters (Finch et al., 1997 but see Smith et al., 2005), though such losses can increase beetle food in the short term. There is some concern that fire can encourage the spread the exotic *Tamarisk* (salt cedar) species especially during drought years (Smith et al., 2005). Cicadas emerged earlier on burned versus unburned plots along the Rio Grande increasing potential mismatches between this important food source and breeding events (Smith et al., 2006). The biggest risk from wildfire for bird species along the Rio Grande is the further degradation and isolation of limited habitat (FWS 2014). Fire is considered a major threat to remaining habitat for the southwestern willow flycatcher (FWS Species Report http://ecos.fws.gov/speciesProfile/profile/speciesProfile.action?spcode=B094) and is listed as a concern for determination of threatened status for the Western yellow-billed cuckoo.

We considered the Lucy's warbler at risk of habitat declines due to wildfire. Warblers were considered at high risk of negative impacts for all fires occurring in shrub habitat and fires in forests with torching and at moderate risk for forest fires without torching (Figures 4 and 5).

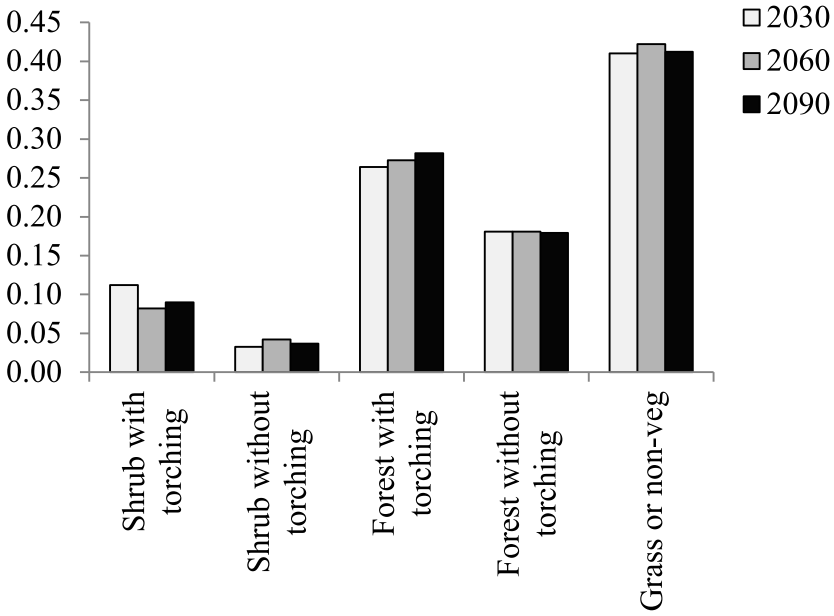
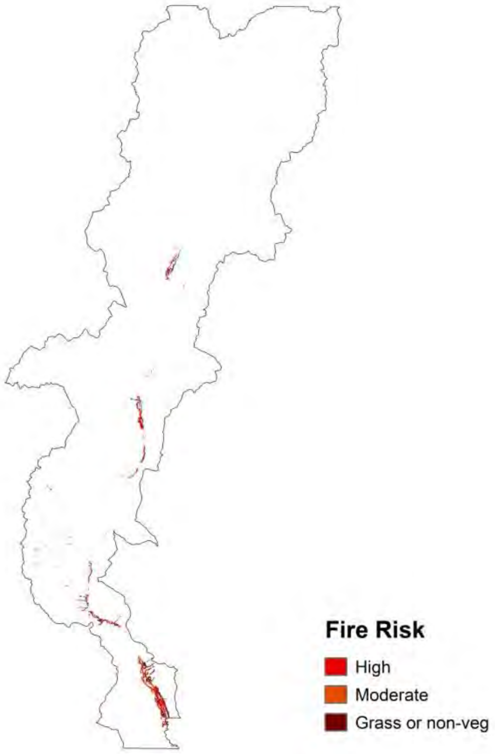
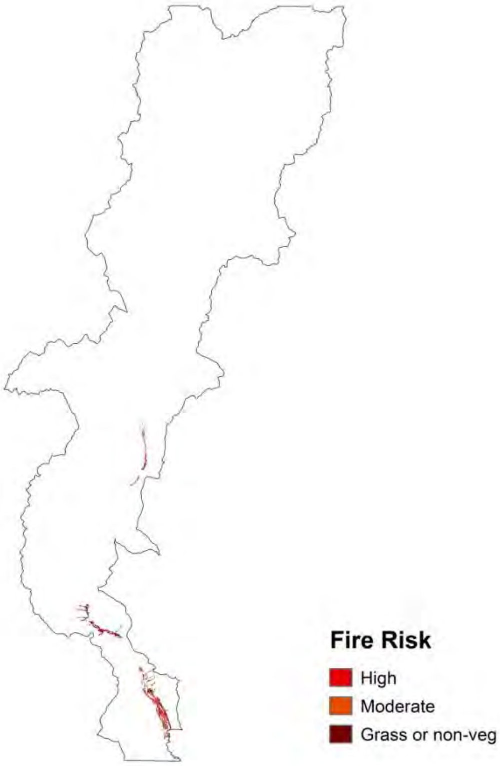


Figure 4. Percent of habitat falling within each fire type category.

2030



2060



2090

Figure 5. Fire impacts for *Oreothlypis luciae* habitat under three climate futures. Fire intensity was calculated using FSim (Finney et al., 2011) for each time period based on the GFDL-ESM-2m GCM under the RCP85 scenario. Fire risk maps were created by overlaying estimated species' habitat, predicted fire characteristic (shrub with or without torching, forest with or without torching or grass/nonveg) and species' response (negative, neutral, positive).

References:

1. 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.
2. Finch, D.M., J.L. Ganey, W. Yong, R.T. Kimball, and R. Sallabanks. 1997. Effects and interactions of fire, logging and grazing. Pages l03-106, In W.M. Block and D.M. Finch (technical editors). Songbird Ecology in Southwestern Ponderosa Pine Forest; A Literature Review. USDA Forest Service General Technical Report GTR-292. Rocky Mountain Forest and Range Experiment Station, Fort Collins. Colorado.
3. Fish and Wildlife Service. 2014. Endangered and Threatened Wildlife and Plants; Determination of Threatened Status for the Western Distinct Population Segment of the Yellow-billed Cuckoo (*Coccyzus americanus*). Final Rule 50 CFR Part 17.
4. Friedmann, H. and L. F. Kiff. 1985. The parasitic cowbirds and their hosts. West. Found. Vertebr. Zool. 2:226-304.
5. Friggens, M.M., Finch, D.M, et al. 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
6. Johnson, R., Roy, Yard, H. K. and B.T. Brown. 1997. Lucy's Warbler (*Oreothlypis 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.
7. New Mexico Department of Game and Fish. 2006. Comprehensive Wildlife Conservation Strategy for New Mexico. New Mexico Department of Game and Fish. Santa Fe, New Mexico. 526 pp + appendices. http://www.wildlife.state.nm.us/conservation/comp_wildlife_cons_strategy/
8. Smith, D. M., Kelly, J.F., and D.M. Finch. 2005. Avian nest box selection and nest success in burned and unburned southwestern riparian forest. Journal of wildlife management 71: 411-421.
9. 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.
10. 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.
11. 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

Southwestern Willow Flycatcher (*Empidonax traillii extimus*)

Overview

Predicted Impacts	
Habitat Change	
2030	48-50% Loss
2060	54-62% Loss
2090	62-71% Loss
Adaptive capacity	Very Low
Fire Response	Negative

Status:
The Southwestern willow flycatcher has been on the federal endangered species list since 1995.

Range and Habitat:
The Southwestern Willow flycatcher inhabits riparian areas in the southwestern U.S. (Figure 1). It winters in southern Mexico, Central America and northern South America (Sedgwick 2000). In the Middle Rio Grande, the Southwestern willow flycatcher migrates through willow, cottonwood and saltcedar stands (Hunter 1988; Cartron et al., 2008). It is common in New Mexico during migration in the spring and fall, but also breeds in a few areas along the Middle Rio Grande. This species is associated with dense shrubby and wet habitats and typically nests in flooded areas with willow dominated habitat (Sedgwick 2000). Generally, the Southwestern willow flycatcher does not occupy areas dominated by exotics (Skoggs and Marshall 2000), but can successfully nest in saltcedar-dominated habitats (Skoggs et al., 2006).

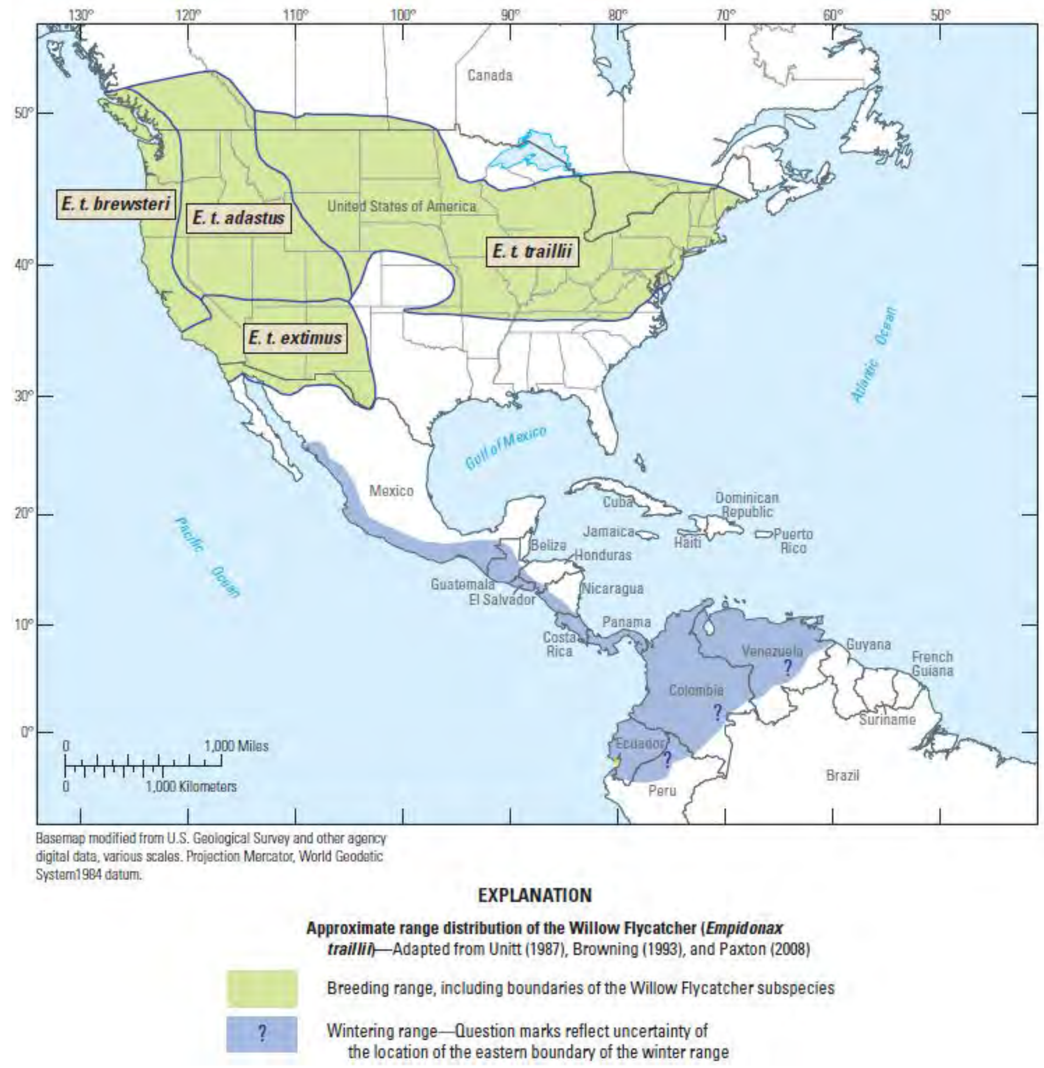


Figure 1. Distribution of *Empidonax traillii* subspecies. From Sogg et al., 2010, USGS.

Southwestern Willow Flycatcher (*Empidonax traillii extimus*)

Climate Impacts and Adaptive Capacity

Adaptive capacity score = 2.5 (very low)

There are a number of indications for potential negative impacts for the flycatcher under changing climate (Table 1). The Southwestern willow flycatcher uses shrubs and small trees for nesting substrates. Increased shrub cover is associated with reproductive success of the Southwestern willow flycatcher (Bombay et al., 2003). Additionally, willow flycatchers will not nest if water is not flowing (Johnson et al., 1999). Flooding may be associated with reduced predation by mammalian predators (Cain et al., 2003). Flooding due to precipitation changes may increase with warmer winter temperatures, but lower overall water output and advancement of flood pulse may shift pulse too early to benefit nesting.

Low fecundity and starvation in Southwestern willow flycatcher nestlings has been associated with low snowpack and drying of marshes in Oregon (Sedgwick 2000). Saltcedar, while used for nesting, may be limiting for nesting in hotter climates, because it does not provide needed shade (Hunter 1988; Cartron et al., 2008). Decreased streamflow will likely drop water tables and favor saltcedar over willow which might increase habitat quality for the flycatcher. This may decrease flycatcher reproductive success because of microclimate disadvantage in saltcedar, which will be more critical as temperatures increase.

Reliance on multiple, spatially distinct sites is associated with an increased risk of negative impacts as migrating animals are exposed to a greater range of conditions and likelihood of habitat change. Conversely, migrant species are able to disperse to new habitats, an important trait for coping with potential climate changes. The Southwestern willow flycatcher is highly mobile and has a good capacity to shift with changes in habitats. This species exhibits fairly high site fidelity to breeding grounds (Sedgwick 2000), but is known to recolonize habitats.

Physiology:

There are several indications that the southwestern willow flycatcher is sensitive to increased temperatures (Table 1). Decline of willow flycatcher populations are associated with spread of tamarisk, which may not have the thermal protection of broadleaf shrubs (Hunter 1988; Cartron et al., 2008), though nests successfully in tamarisk in many areas (Skoggs et al., 2006). Its association with moist and shady microclimates may relate to its relatively late season nesting and accompanying hot temperatures (Skoggs and Marshall 2000). The Southwestern Willow flycatcher is not expected to be exposed to greater storms because it uses interior migration (Bagne et al., 2011). No mortalities have been noted due to extreme weather with the exception of nestlings. However, these flycatchers are drought sensitive due to effects on food resources and drying of habitat. This species does not possess an ability to reduce metabolic energy or water requirements and does not have alternative life history strategies to cope with variable resources or climate conditions. The Southwestern Willow flycatcher can live up to 11 years, but mean life span around one year (Sedgwick and Iko 1997).

Phenology:

Photoperiod likely important for timing migration and no change in cue expected. However, the Southwestern willow flycatcher has a short nesting season that is thought to be limited by resource availability (Sedgwick 2000). Insects may emerge earlier or become more variable with more variable rainfall. There is also a potential difference in migrant arrival and insect emergence because this species is a long distant migrant. One advantage for this species is that it is a late season breeder that can raise multiple broods minimizing reproductive losses due to spring storms or other sources of nest loss.

Biotic Interactions:

Willow flycatchers are primarily insectivorous (Sedgwick 2000). Willow flycatchers are dependent upon localized food sources during nesting and tend to specialize on insects that are associated with water. Years of low rainfall are known to reduced food supplies and lower reproductive success in the flycatcher. Future dry periods with increase rainfall variability may exacerbate losses due to resource changes. Flycatchers are preyed upon by various species though overall changes in predation rates are not expected (Sedgwick 2000). Southwestern willow flycatcher nests are parasitized by brown-headed cowbirds, which are considered a threat under climate change. Currently, willow flycatcher reproduction occurs relatively late, which does not allow them to escape cowbird brood parasitism (Robinson et al., 1995). The shorter migration distance in cowbirds will likely allow cowbirds to keep pace with any advancement in breeding by willow flycatchers. In addition, cowbirds possess a number of resilience traits to climate change (Friggens et al., 2013).

Table 1. Vulnerability scores for *Empidonax traillii extimus.* "v" indicates a vulnerability or sensitivity, "r" indicates a resiliency or increased adaptive capacity, "n" indicates neutral response.

Question	Score	Notes
1. Changes to non-modeled habitat components	v	Flooded habitats
2. Change in habitat quality	v	Declines under dryer conditions
3. Dispersal ability (Site fidelity or other limitations)	r	Good
4. Reliance on migratory or transitional habitats	v	Yes
5. Increase or decrease in physiological range limitation	v	May be sensitive to extreme heat
6. Response to predicted extreme weather events/disturbances	n	Drought sensitive
7. Changes to daily activity period	n	Not expected
8. Variable life history traits or coping strategies	v	No
9. Ability to outlive limiting conditions	v	No
10. Sex Ratios determined by temperature or food changes	n	No
11. Migrates/hibernates in response to weather cues	n	No, daylight likely cue

Question	Score	Notes
12. Reliance on weather mediated resource (e.g. insect emergence)	v	Yes, breeding timed to insects
13. Spatial or temporal separation between critical resources and life history stages	v	Yes, long distance migrant
14. Can adjust timing of critical activities	r	Yes, 2 or more clutches per year
15. Likelihood for decreased food resource	v	Specialist on aquatic insects
16. Likelihood of increase predation	n	Not expected
17. Loss of important symbiotic species	n	None known
18. Increase in high mortality/morbidity disease	v	Cow bird parasitism may increase
19. Increased competitive pressures	n	Not expected

Southwestern Willow Flycatcher (*Empidonax traillii extimus*)

Niche Model Analysis

Model AUC values were 0.83, 0.85 and 0.86 for CGCM3, GFDLCM2, and HadCM3, respectively. Models predictions were similar under all three climate scenarios with declines in suitable habitat by approximately 2/3^{rds} (Table 2, Figures 2 and 3).

Table 2. Percent of original predicted habitat under three climate models.

Time	CGCM3.1	GFDL CM2.1	Had_CM3	Average
current	1	1	1	1
2030	0.58	0.50	0.55	0.54
2060	0.49	0.38	0.47	0.44
2090	0.38	0.29	0.34	0.33

Distance to water, slope, diurnal range of temperature, and biome were the most important variables for predicting suitable habitat for the Southwestern willow flycatcher (Table 3). Habitat suitability decreased with increasing distance from water. Slope contributed to the model both as a linear and hinge function with higher quality habitat associated more strongly with areas with low slope. Suitability was greatest for areas with intermediate values of potential evapotranspiration of natural vegetation. Mean diurnal range of temperatures that fell within values of 17 and 19 °C were most strongly associated with suitable habitat whereas areas with ranges less than 17 °C were less suitable. This species is positively associated with riparian habitats in Semidesert Grassland biomes and negatively associated with riparian habitats in Great Basin Conifer Woodland and Plains grassland biomes.

Table 3. Variables in order of importance for species habitat models. Relationship between variable and predicted suitability is listed as positive (+) negative (-) for linear and exponential curves or quadratic for parabola-like curves.

Variables	CGCM3.1	GFDL CM2.1	Had_CM3	Relationship
Distance to Water	41.59	43.50	41.82	-
Slope	14.77	13.43	15.14	-
Mean Diurnal Temp Range (bio2)	10.03	10.45	9.64	+ at low values
Biome	9.78	9.69	11.14	+SemDesGssd, -GB ConWdld, PlnsGssd
Pot Evap Natural Veg	5.61	5.09	5.52	Quadratic
Precip August (bio13)	4.59	4.23	4.58	-
Precip JJA (bio18)	4.23	3.22	4.04	-
Max Temp JJA (bio5)	3.96	3.58	3.14	+

Variables	CGCM3.1	GFDL CM2.1	Had_CM3	Relationship
Annual Precip	3.03	4.75	2.85	-
Elevation	2.22	1.99	2.01	Peaks at low values
Aridity Index	0.21	0.07	0.10	Peaks at low values
Isothermality (bio3)	0.00	0.01	0.01	Peaks at low and high values

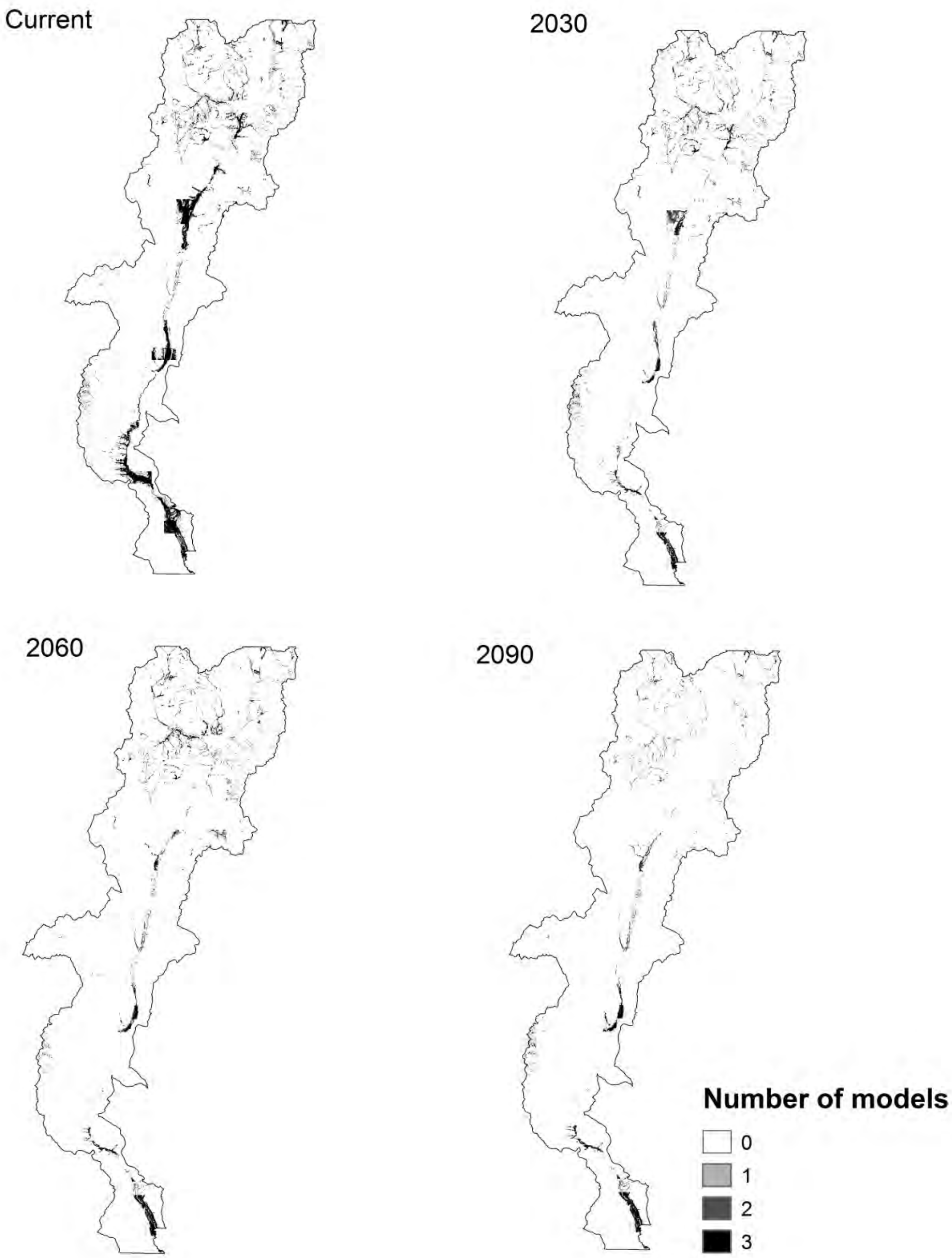


Figure 2. Distribution of suitable habitat for *Empidonax t. extimus* for three future time periods. Future habitat predictions are based on three climate models: Had_CM3, CGCM3.1, and GFDL_CM2.1. Shading indicates the number of models that predict suitable habitat for a given area.

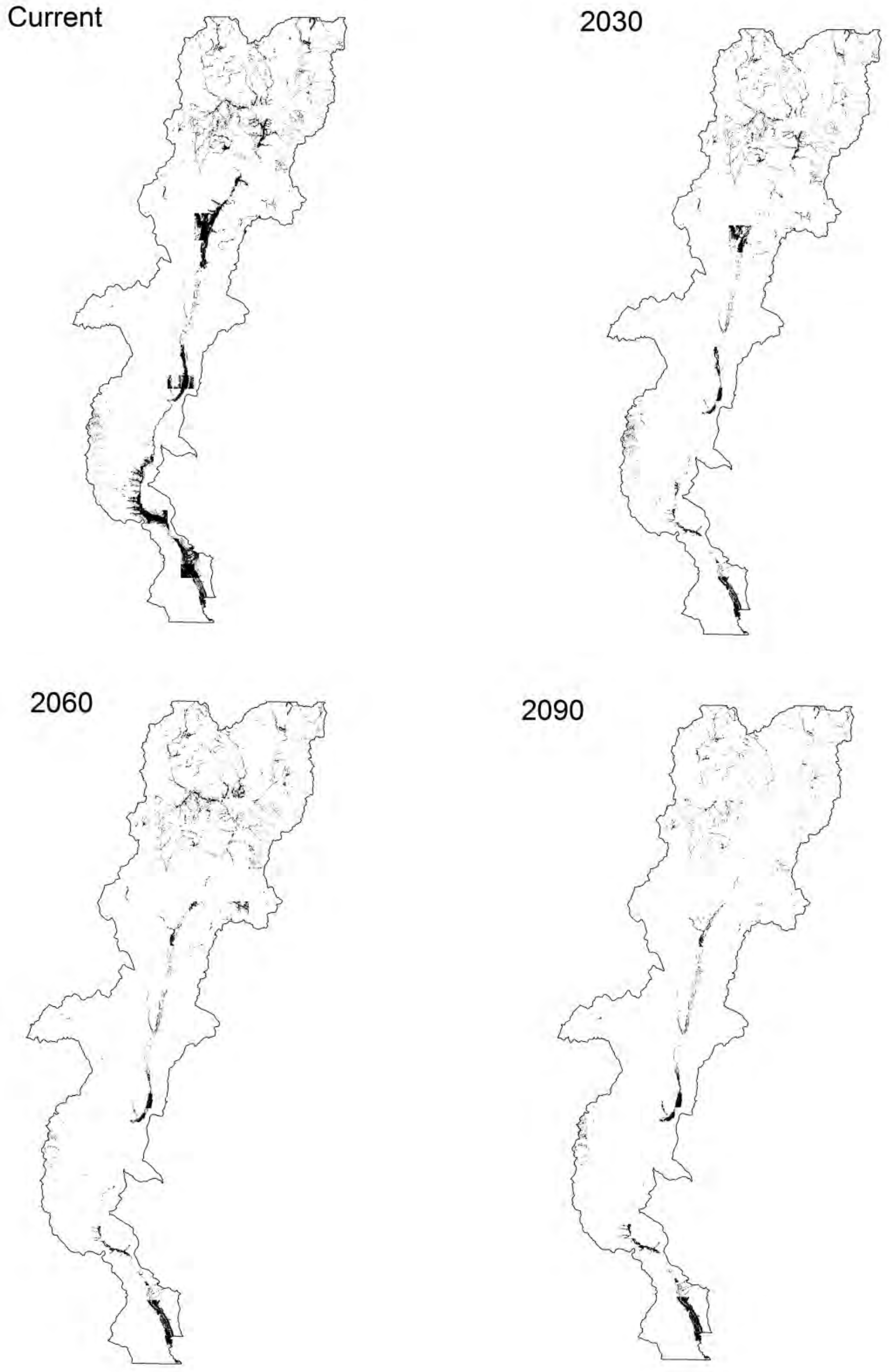


Figure 3. Two-thirds consensus models for suitable habitat for *Empidonax t. extimus* for three future time periods. Future habitat is predicted where at least two of three climate models, Had_CM3, CGCM3.1, and GFDL_CM2.1, identify suitable habitat.

Southwestern Willow Flycatcher (*Empidonax traillii extimus*)

Fire Analysis

Fire can alter bird habitat by changing vegetation composition and structure which influences nest site availability, foraging substrates, and nest predation rates. Prescribed or wildfire during breeding season is likely to reduce habitat for shrub and ground dwelling species (Finch et al., 1997). Loss of snags due to fire will harm cavity nesters (Finch et al., 1997 but see Smith et al., 2005) though such losses can increase beetle food in the short term. There is some concern that fire can encourage the spread the exotic *Tamarix* species especially during drought years (Smith et al., 2005). Cicadas emerged earlier on burned versus unburned plots along the Rio Grande increasing potential mismatches between this important food source and breeding events (Smith et al., 2006). The biggest risk from wildfire for bird species along the Rio Grande is the further degradation and isolation of limited habitat (FWS 2014). Fire is considered a major threat to remaining habitat for the southwestern willow flycatcher (FWS Species Report http://ecos.fws.gov/speciesProfile/profile/speciesProfile.action?spcode=B094) and is listed as a concern for determination of threatened status for the Western Yellow-billed cuckoo. Fire may increase shrubby habitats, though Paxton et al., (1996) noted that fire destroyed habitat.

We considered the southwestern willow flycatcher at risk of habitat declines due to wildfire. Flycatchers were considered at high risk of negative impacts for torching fires in shrub and forest habitats and at moderate risk for habitat loss with non-torching fire (Figures 4 and 5).

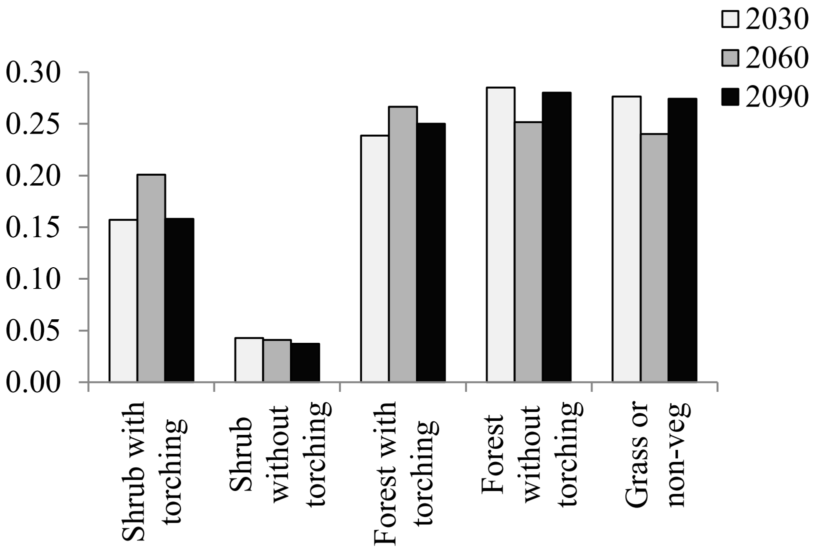


Figure 4. Percent of habitat falling within each fire type category.

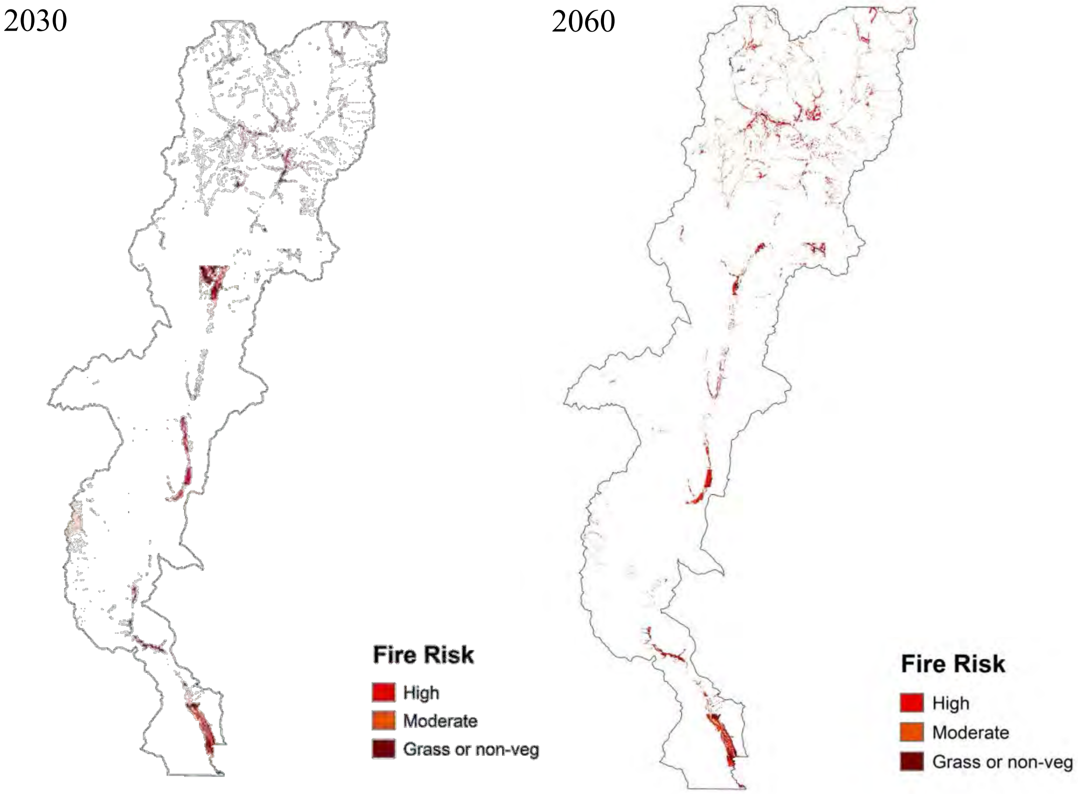


Figure 5. Fire impacts for *Empidonax traillii extimus* habitat under three climate futures. Fire intensity was calculated using FSim (Finney et al., 2011) for each time period based on the GFDL-ESM-2m GCM under the RCP85 scenario. Fire risk maps were created by overlaying estimated species' habitat, predicted fire characteristic (shrub with or without torching, forest with or without torching or grass/nonveg) and species' response (negative, neutral, positive).

References:

1. 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. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO.
2. 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.
3. 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.
4. 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.
5. Fish and Wildlife Service. 2014. Endangered and Threatened Wildlife and Plants; Determination of Threatened Status for the Western Distinct Population Segment of the Yellow-billed Cuckoo (*Coccyzus americanus*). Final Rule 50 CFR Part 17.
6. 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.
7. Friggens, M.M., Finch, D. M, et al. 2013. Vulnerability of Species to climate change in the southwest: terrestrial species of the middle Rio Grande. Gen. Tech. Rep. RMRS-GTR-306. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO.
8. 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.
9. 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.
10. 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.
11. 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.
12. 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.
13. Sedgwick, J. A. 2000. Willow Flycatcher (Empidonax traillii). *In* The Birds of North America, No. 533 (A. Poole and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA.
14. Sedgwick, J. A. and W. M Iko. 1997. Costs of Brown-headed Cowbird parasitism to Willow Flycatchers. Studies in Avian Biology 18: 167-181.
15. 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.

16. 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.

Western Yellow-billed Cuckoo (*Coccyzus americanus occidentalis*)

Overview

Predicted Impacts	
Habitat Change	
2030	59-79% Loss
2060	57-67 % Loss
2090	44-91% Loss
Adaptive capacity	Very Low
Fire Response	Negative

Status:
Cuckoo populations in New Mexico have declined precipitously since the 1960's as humans have altered riparian habitats (Howe 1986). Population size of The Western yellow-billed cuckoo may vary greatly from year-to-year even in undisturbed habitats, presumably due to changes in food supply. The Western yellow-billed cuckoo is considered species of high concern at risk from drought and wildfire in New Mexico by the Wildlife and Wildfire Work Group of the Governor's Task Force on Drought. Currently, the Western yellow-billed cuckoo is designated as threatened under the Endangered Species Act (FWS, 2014).

Range and Habitat:
The breeding range of the Western yellow-billed cuckoo formerly included most of eastern North America and parts of Northern Mexico (FWS 2014). The Western yellow-billed cuckoo is thought to dwell west of the continental divide at northern latitudes and west of the Pecos River at southern latitudes (Figure 1). This cuckoo species is associated with riparian habitats with willow and cottonwood (Hughes 1999). The Western yellow-billed cuckoo is restricted to riparian areas, which provide cooler and more humid environments, in hot regions.

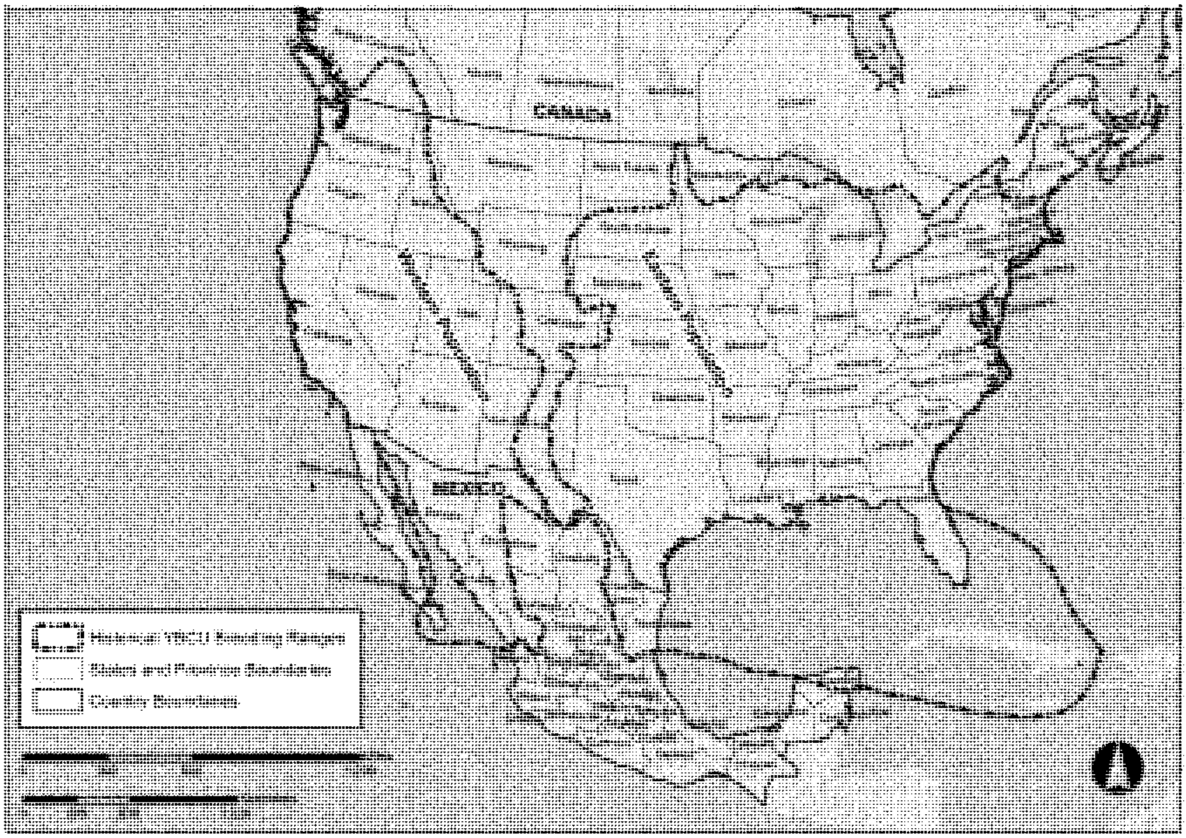


Figure 1. Historical Breeding range of the Eastern and Western yellow-billed cuckoos based on the American Ornithological Union's checklist (from FWS 2014)

Western Yellow-billed Cuckoo (*Coccyzus americanus occidentalis*)

Climate Change Impact and Adaptive Capacity

Adaptive Capacity Score= -2.5 (Very low)

Western yellow-billed cuckoos exhibit a number of vulnerabilities to potential climate impacts (Table 1). Its habitat requirements includes fairly large trees and dense vegetation (Hughes 1999), which were considered likely to decline in previous climate change assessments (Friggens et al., 2013). The Western yellow-billed cuckoo is a long distance migrant that relies on sites throughout Mexico and Central America. In addition, for 2-3 weeks before breeding, they may occupy upland vegetation including pinyon, oak, juniper, and manzanita (Hughes 1999). Reliance on multiple, spatially distinct sites is associated with an increased risk of negative impacts as migrating animals are exposed to a greater range of conditions and likelihood of habitat change. Conversely, migrant species are able to disperse to new habitats, an important trait for coping with potential climate changes.

Physiology:

The Western yellow-billed cuckoo does not appear to have physiological adaptations for dealing with additional temperature increases, increased storms, and drought (Table 1). In particular, the cuckoo appears be sensitive to hot dry conditions. Incubating adults and nestlings have been observed panting on hot days (Hughes 1999). Western yellow-billed cuckoo eggs may be prone to drying, which reduces hatchability (Laymon and Halterman 1987). Drought is considered a limiting condition in terms of food availability for the Western yellow-billed cuckoo (Hughes 1999). Cuckoos may also experience increased mortality with increased exposure to hurricanes, but exposure is probably not high for western populations (Hughes 1999).

This species does not torpor nor does it exhibit other traits that are employed to deal with more variable resource availability. The Western yellow-billed cuckoo does engage in both inter- and intra-specific brood parasitism. This behavior allows females to lay larger clutches by putting extra eggs in nests of other individuals. This behavior is thought to increase breeding opportunities and populations during high resource years, but no effect of this behavior on populations has been documented during low resource years. Rates of brood parasitism are also not well known, but currently 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 are unknown. This species is not thought to live more than 5 years and therefore is unlikely to outlive limiting conditions (Friggens et al., 2013).

Phenology:

Western yellow-billed cuckoos exhibit mostly vulnerabilities to potential climate impacts (Table 1). Moisture may be related to breeding timing, at least in some populations (see below), but not known as cue. The Western yellow-billed cuckoo is thought to initiate breeding to time with abundance of local food or periods of greatest precipitation. This species is a long distance migrant and therefore at risk of mismatch in conditions between breeding and wintering grounds.

Populations of the cuckoo in the western U.S. are believed to raise only one brood per season during a relatively short breeding season (Hughes 1999). Recent information found evidence that yellow-billed cuckoos may breed a second time in western Mexico after migrating from the north (Rohwer et al., 2009), which may allow this species take advantage of seasonal resources in multiple locations. However, it is unknown cuckoos within habitats along the Rio Grande exhibit this behavior.

Biotic Interactions:

Western yellow-billed cuckoos feed primarily on large insects such as caterpillars, grasshoppers, crickets, katydids, cicadas. They also feed on 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). Cicadas are resilient to high temperatures (Heath and Wilkin 1970) and populations of at least one species (Apache cicadas) increased due to habitat changes associated with declining water tables (Glinski and Ohmart 1984) and increased Tamarix. Thus, cuckoo food sources could increase in the future. The risk that these food resources will appear outside of the cuckoos nesting season is considered within the phenology scores.

Raptors may be an important predator during migration and upon arrival on wintering grounds. Snakes, mammals, and birds are known nest predators (Hughes 1999). A wide variety of predators and climate influences led to no predictions in predation rate.

Yellow-billed cuckoos may use other bird’s nests for eggs, but this is a facultative trait that may only rarely occur in the Western yellow-billed cuckoo (Hughes 1999). Numerous diseases and parasites have been documented for The Western yellow-billed cuckoo, but no information indicating significant negative effects on populations. Seldom subject to brood parasitism by brown-headed cowbird (*Molothrus ater*) as nesting duration is short (Hughes 1999). There is some evidence of competition with black-billed cuckoos in the eastern U.S (Hughes 1999), but black-billed cuckoos are not found in its range in New Mexico.

Table 1. Score sheet for *Coccyzus a. occidentalis*. "v" indicates a vulnerability or sensitivity, "r" indicates a resiliency or increased adaptive capacity, "n" indicates neutral response.

Question	Score	Notes
1. Changes to nonmodeled habitat components	v	Large trees, snags
2. Change in habitat quality	n	Not Predicted
3. Dispersal ability (Site fidelity or other limitations)	r	Good
4. Reliance on migratory or transitional habitats	v	Yes
5. Increase or decrease in physiological range limitation	v	Potential temperature threshold
6. Response to predicted extreme weather events/disturbances	v	Potential mortalities from storms
7. Changes to daily activity period	n	Not known
8. Variable life history traits or coping strategies	v	No
9. Ability to outlive limiting conditions	v	No
10. Sex Ratios determined by temperature or food changes	n	No
11. Migrates/hibernates in response to weather cues	n	Uses daylight as cue
12. Reliance on weather mediated resource (e.g. insect emergence)	v	Yes
13. Spatial or temporal separation between critical resources and life history stages	v	Yes
14. Can adjust timing of critical activities	v	Single clutch per year
15. Likelihood for decreased food resource	r	Increases possible for cicadas
16. Likelihood of increase predation	n	Not predicted
17. Loss of important symbiotic species	n	None determined
18. Increase in high mortality/morbidity disease	n	Not predicted
19. Increased competitive pressures	n	Not predicted

Western Yellow-billed Cuckoo (*Coccyzus americanus occidentalis*)

Niche Model Analysis

Model AUC values were 0.95, 0.94, and 0.94 for CGCM3.1, GFDL_CM2.1, and Had_CM3, respectively. Habitat declines under all three models though the degree of habitat loss varies under the different scenarios (Table 2, Figures 2 and 3).

Table 2. Percent of original predicted habitat under three climate models.

Time	**CGCM3.1**	**GFDL CM2.1**	**Had_CM3**	**Average**
current	1	1	1	1
2030	0.37	0.21	0.41	0.33
2060	0.33	0.43	0.38	0.38
2090	0.09	0.36	0.11	0.19

For the yellow-billed cuckoo, distance to water and elevation were very important and potential annual evapotranspiration, slope and diurnal temperature range somewhat important predictors of suitable habitat (Table 3). Overall, cuckoo habitat suitability declined with increasing distance from water and elevation. Under GFDL_CM2.1, and in contrast to the other two models, August precipitation was more important than slope and max temperature during JJA. Habitat suitability was strongly and negatively associated with maximum summer (JJA) temperature under the GFDL_CM2.1, slightly negative under Had_CM3 and negatively related in only 1 of the 15 runs under CGCM3.1. A number of interactions were found for the cuckoo: a negative association with ai*bio3, bio18*bio5 and bio18*petnatveg, bio2*slope and bio 5*distance to water. Generally, suitability declined with increasing values of potential evapotranspiration of natural vegetation at low values but increases with increasing values after a minimum threshold is reached. The yellow-billed cuckoo was positively associated with riparian habitats within the Chihuahuan desert scrub Biome and negatively associated with riparian habitats within the Semidesert Grassland Biomes.

Table 3. Variables in order of importance for species habitat models. Relationship between variable and predicted suitability is listed as positive (+) negative (-) for linear and exponential curves or quadratic for parabola-like curves.

Variables	CGCM3.1	GFDL CM2.1	Had_CM3	Relationship
Distance to Water	56.90	59.57	56.74	-
Elevation	20.18	18.88	19.71	Associated with lower elevations
Mean Diurnal Temp Range (bio2)	5.27	4.74	4.33	+
Pot Evap Natural Veg	4.57	4.56	5.77	Quadratic
Slope	3.43	2.77	3.76	-
Max Temp JJA (bio5)	2.32	2.64	2.23	+
Precip August (bio13)	2.05	2.89	0.49	Peaks at lower values
Precip JJA (bio18)	1.57	1.47	2.21	Peaks at lower values
Biome	1.56	1.84	0.92	
Aridity Index	1.50	0.29	3.30	- after 13
Isothermality (bio3)	0.44	0.33	0.22	-
Annual Precip	0.22	0.01	0.31	Peaks at lower values

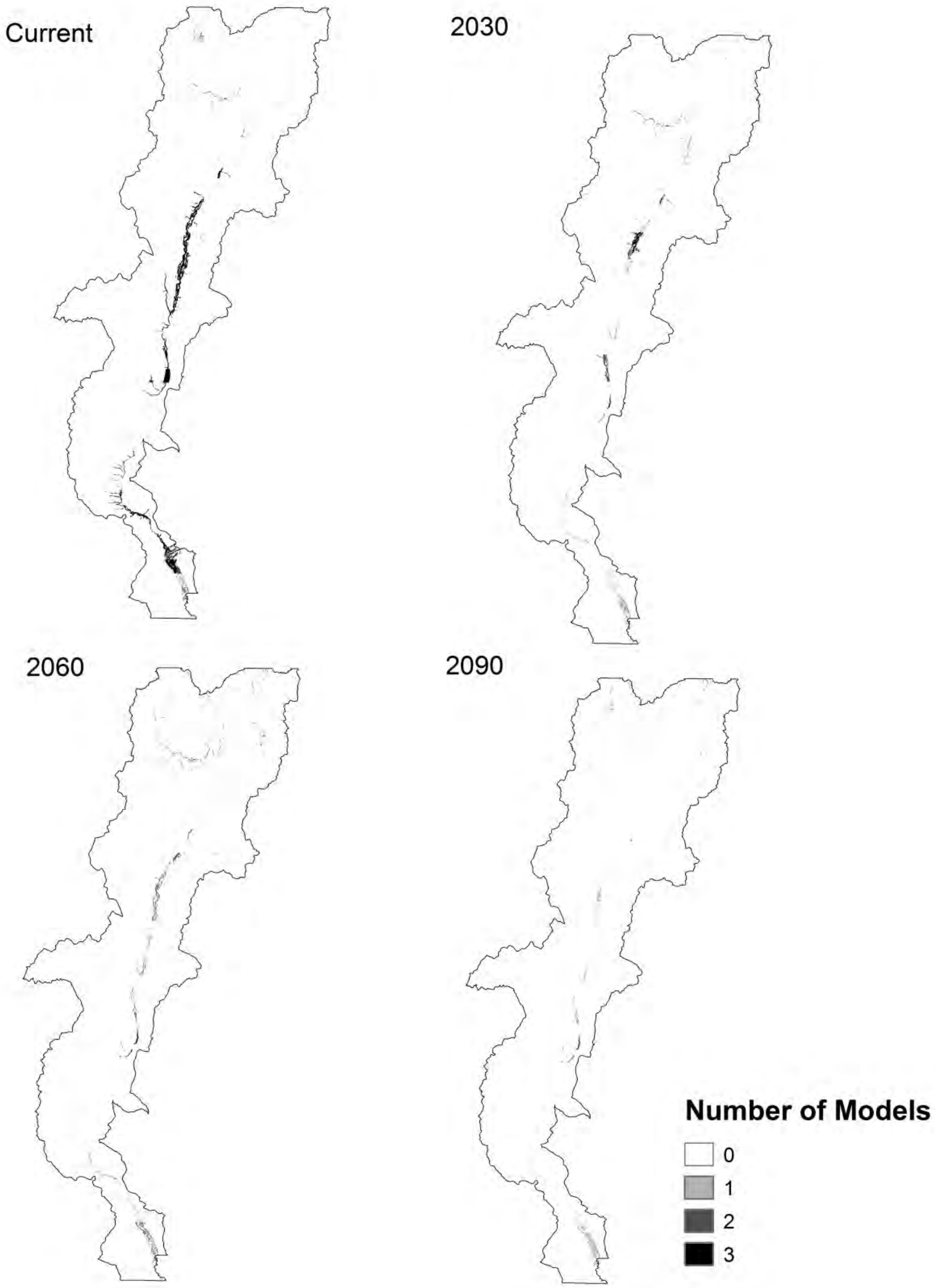


Figure 2. Distribution of suitable habitat for *Coccyzus a. occidentalis* for three future time periods. Future habitat predictions are based on three climate models: Had_CM3, CGCM3.1, and GFDL_CM2.1. Shading indicates the number of models that predict suitable habitat for a given area.

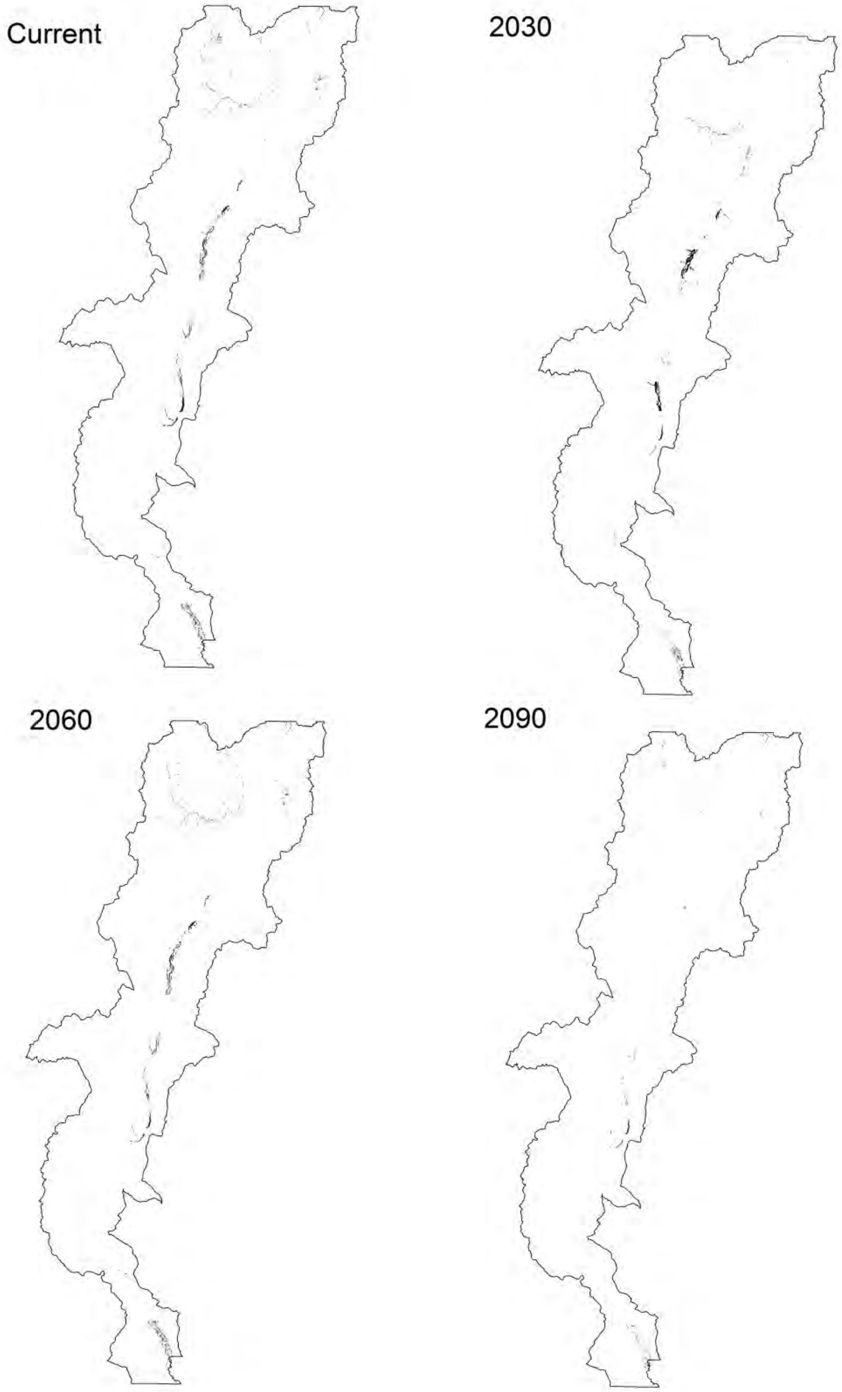


Figure 3. Two-thirds consensus models for suitable habitat for *Coccyzus a. occidentalis* for three future time periods. Future habitat is predicted where at least two of three climate models, Had_CM3, CGCM3.1, and GFDL_CM2.1, identify suitable habitat.

Western Yellow-billed Cuckoo (*Coccyzus americanus occidentalis*)

Fire Analysis

Fire can alter bird habitat by changing vegetation composition and structure which influences nest site availability, foraging substrates, and nest predation rates. Prescribed or wildfire during breeding season is likely to reduce habitat for shrub and ground dwelling species (Finch et al., 1997). Loss of snags due to fire will harm cavity nesters (Finch et al., 1997 but see Smith et al., 2005) though such losses can increase beetle food in the short term. There is some concern that fire can encourage the spread the exotic Tamarisk species especially during drought years (Smith et al., 2005). Cicadas emerged earlier on burned versus unburned plots along the Rio Grande increasing potential mismatches between this important food source and breeding events (Smith et al., 2006). The biggest risk from wildfire for bird species along the Rio Grande is the further degradation and isolation of limited habitat (FWS, 2014). Fire is considered a major threat to remaining habitat for the southwestern willow flycatcher (FWS Species Report http://ecos.fws.gov/speciesProfile/profile/speciesProfile.action?spcode=B094) and is listed as a concern for determination of threatened status for the Western yellow-billed cuckoo. The Western yellow-billed cuckoo is considered species of high concern at risk from drought and wildfire in New Mexico by the Wildlife and Wildfire Work Group of the Governor's Task Force on Drought

We considered the Western yellow-billed cuckoo at risk of habitat declines due to wildfire. Cuckoos were considered at high risk of negative impacts for all fires occurring in shrub habitat and fires in forests with torching and at moderate risk for forest fires without torching.

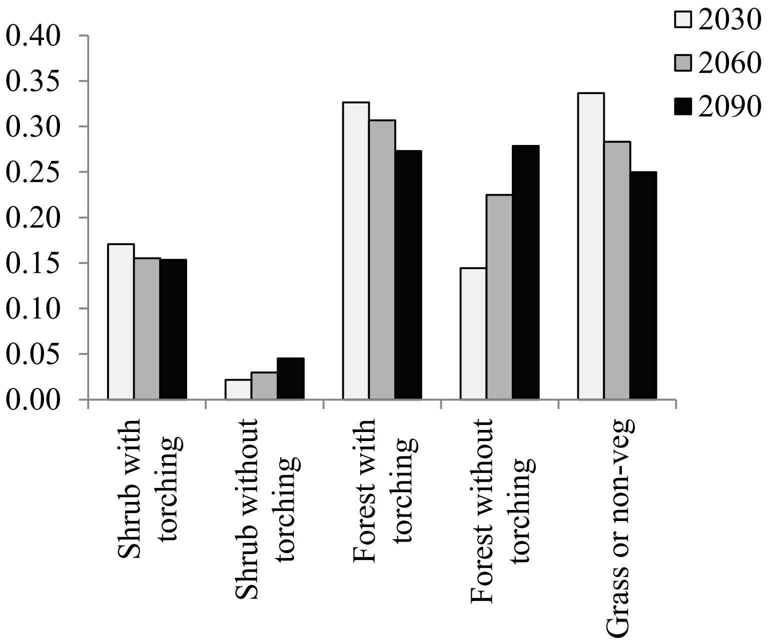
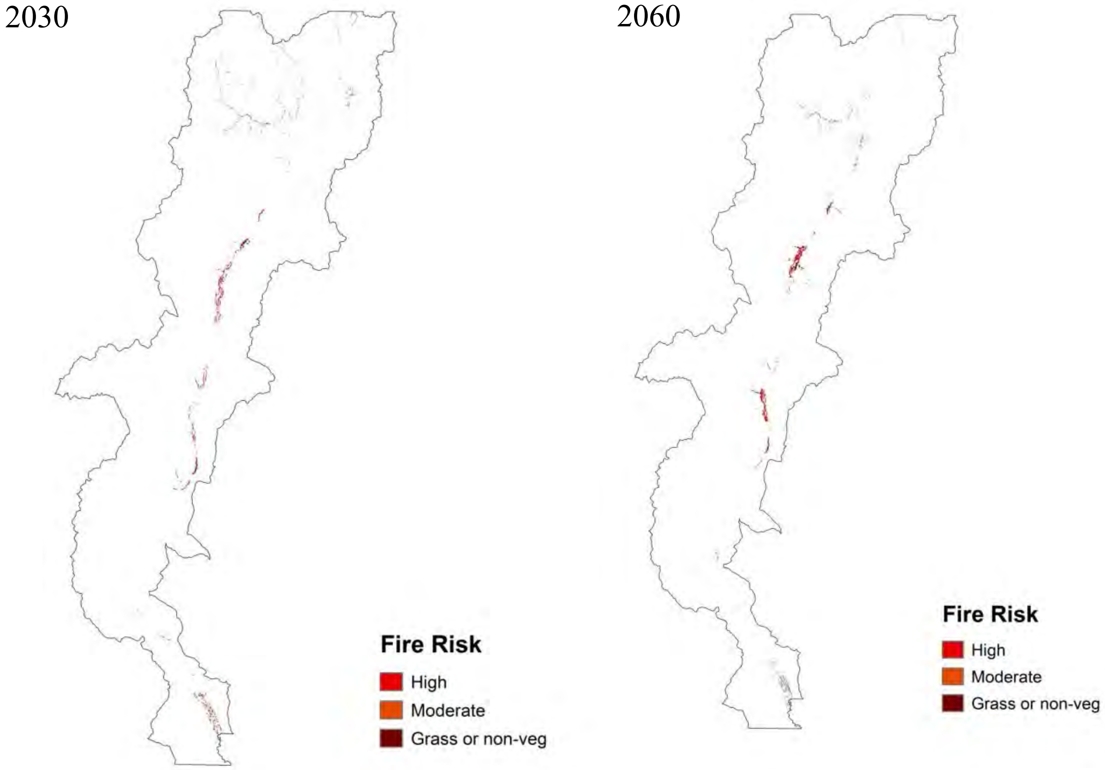


Figure 4. Percent of habitat falling within each fire type category.


2030

2060


2090

Figure 5. Fire impacts for *Coccyzus a. occidentalis* habitat under three climate futures. Fire intensity was calculated using FSim (Finney et al., 2011) for each time period based on the GFDL-ESM-2m GCM under the RCP85 scenario. Fire risk maps were created by overlaying estimated species' habitat, predicted fire characteristic (shrub with or without torching, forest with or without torching or grass/nonveg) and species' response (negative, neutral, positive).

References:

1. 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. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO.
2. Fish and Wildlife Service. 2014. Endangered and Threatened Wildlife and Plants; Determination of Threatened Status for the Western Distinct Population Segment of the Yellow-billed Cuckoo (*Coccyzus americanus*). Final Rule 50 CFR Part 17.
3. 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.
4. Friggens, M.M., Finch, D. M, et al. 2013. Vulnerability of Species to climate change in the southwest: terrestrial species of the middle Rio Grande. Gen. Tech. Rep. RMRS-GTR-306. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO.
5. 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.
6. Heath, J.E. and P.J. Wilkin. 1970. Temperature Responses of the Desert Cicada, *Diceroprocta apache* (Homoptera, Cicadidae). Physiological Zoology 43:145-154.
7. 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.
8. 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
9. 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.
10. 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.
11. 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.
12. 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.

Occult Bat (*Myotis occultus*)

Overview

Predicted Impacts	
Habitat Change	
2030	43-44% loss
2060	52-70% loss
2090	80-84% loss
Adaptive capacity	Intermediate
Fire Response	Positive Overall
Short term	neutral
Long term	Positive

Status:

Currently 56% of the bat species present in North America are listed or being considered for listing under the Endangered Species Act. The primary reasons for bat population declines include habitat loss, water contamination, pesticide poisoning and, most importantly, disturbance and destruction of day roosts (Evelyn et al., 2004). Taxonomic status of this species has been debated for some time but recent evidence has validated the designation of *M. occultus* (NatureServe 2009; Piaggio et al., 2002). This species is considered vulnerable on a global status because of its small range and limited distribution. It has an S4 status within New Mexico (NatureServe 2009). BISON-M (2009) lists this species as rare or uncommon in New Mexico a BLM and USFS sensitive species as well as a Species of Greatest Conservation Need in New Mexico.

Range and Habitat:

The range of *Myotis occultus* includes southeastern Califorina, central and eastern Arizona, New Mexico, and potentially Colorado and Texas (Figure 1). *Myotis occultus* are often seen feeding over bodies of water and are typically associated with riparian areas (AZ Game and Fish Heritage Report; BISON-M 2009). One colony was known to occur on the Bosque del Apache National Wildlife Refuge, but its status is unknown since their use of a building as a roost site was prevented. *Myotis occultus* is considered a resident of areas around large bodies of water and transient in areas that do not contain water. Vegetation zone is not thought to be important determinant of the presence of *M. occultus* (Heritage Data-Arizona Game and Fish Department). BISON-M (2009) reports this species to utilizes desert-scrub, ponderosa pine, spruce-fir, deciduous riparian and coniferous riparian habitat types. Interestingly, this bat is typically found at higher elevations (>1830 m) though it has been found in low elevation reaches of the Colorado and Verde rivers in AZ.

Though they have been captured in late fall and very early spring, this species is not thought to be active during winter months (Geluso 2007). They probably hibernate in the vicinity of their summer range and are considered nonmigratory for this study. No hibernacula are known for this species though they have been found in a mine in the northern Sonora in December (Heritage Data-Arizona Game and Fish Department).

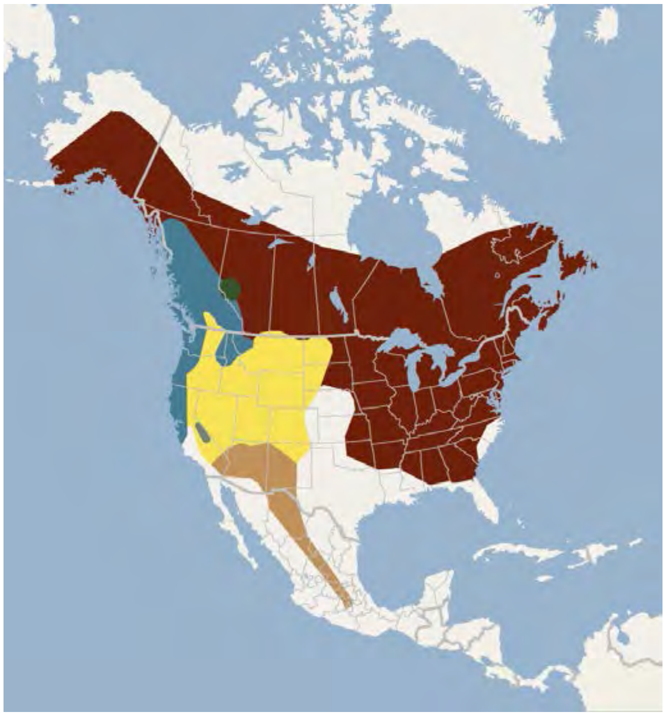


Figure 1. Distribution of *Myotis lucifugus* according to Fenton and Barclay, 1980. *M. l. occultus* is represented in tan.

Occult Bat (*Myotis occultus*)

Climate Change Impacts and Adaptive Capacity

Adaptive Capacity Score = -0.147 (Intermediate)

Like many bats, *M. occultus* probably favor riparian areas because of the concentration of insects and open water, which aids ecolocation. Fast running water has been shown to negatively affect bat ecolocation (Grindal et al., 1999). The occult bat is a water surface forager and so the presence of water appears quite important to the presence of this species. They are also reported to use hawking and to forage in shrubbery. Roost sites appear to be varied and have often been reported in bridge structures. In Arizona, nursery colonies have been found in snags and under exfoliating bark of Ponderosa pine trees indicating this species appears to prefer trees. Natureserve lists standing snags/hollow trees as important habitat factors. This species may use snags in winter. The potential reliance of *Myotis occultus* on snags and large trees was seen as a potential issue under climate change (Table 1).

Physiology Limits:
Myotis occultus appear to be resilient to physiological stresses due to warming air temperature (Table 1). Aerial foraging is limited by insect activity which in turn is limited by temperature (Anthony et al., 1981), but the bats themselves are not considered especially sensitive to air temperatures. BISON-M (2009) cites a Colorado dataset which claims that this species selects summer roosts with temperatures between 5 and 55°C. Higher temperatures are favorable for gestation and lactation phases of female reproductive cycles (McCain 2007; Grindal et al., 1999). Conversely, males and juveniles seek out cooler habitats that allow them to utilize torpor. Little brown bats (*M. lucifugus*) survive hibernation with about half the fat reserves of other heterotherms by limiting the number of times they arouse. The energetically intensive nature of arousal is the primary reason disturbance to hibernacula can be so costly to this and other bat species (Fenton and Barclay 1980).

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). Evidence supports the notion that bats rely on a water source in such environments. Lactating females have an increased need for water (Adams and Hayes 2008). Duration of lactation period is important since species with longer nursing times will have a greater water needs. *Myotis occultus* is reported to be able to increase urine concentration more than the other subspecies of *M. lucifugus* (AnimalDiversity 2009).

Bats exhibit several flexible strategies for dealing with resource variation. Torpor is used by bats to reduce evaporative loss (Carpenter, 1969) though females lose ability to enter torpor in later stages of pregnancy (AnimalDiversity 2009). Female bats are known to reabsorb/abort embryos (Anthony et al., 1981) and will abandon neonates if stressed. Gestation may also be delayed under cool conditions in order that female bats may continue to enter torpor (seen in Pipistrelles, Racey 1973). Females may forgo reproduction in harsh years and will not ovulate if their fat reserves are too depleted when they arise from hibernation (Chung-MacCoubrey 2005).

Though one report showed mass mortality due to cave flooding (Fenton and Barclay 1980), this impact is location specific and there were no other indications that this species will experience increased mortality due to climate impacts. The lifespan of *Myotis occultus* is reported as 6-7 years though one 31 year old male individual was captured (Animaldiversity 2009).

Phenology:

Myotis occultus may be negatively impacted by shifts in the timing of seasonal changes in temperature and insect emergence. Timing and metabolic characteristics of bat hibernation is influenced by temperature (Twente et al., 1985). Additionally, reproductive cycles are generally tied to insect activity, which may shift under changing climate regimes (Anthony et al., 1981). Temperature probably relates directly to insect emergence, although there is some risk of unnaturally warm spring days (that occur before frost season is over) leading to premature emergence. Though females have some capacity to modify gestation and ovulation, the result of non-favorable conditions is the loss of pregnancy. Occult bats ovulate once per year and give birth in the spring to a single offspring (BISON-M 2009).

Biotic Interactions:

The occult bat forages on flying insects including mosquitoes and midges (Heritage Data-Arizona Game and Fish Department). There may be some preference for beetles in populations that inhabit the Middle Rio Grande Bosque (E. Valdez, pers. comm). Bats commonly fall prey to a variety of raptors and animals when young and have been found with high numbers of ectoparasites (E. Valdez, pers. comm). However, there is no record that *M. occultus* populations are regulated by predators, disease or competitive interactions.

Table 1. Scoring sheet for the occult bat (*Myotis occultus).* "v" indicates a vulnerability or sensitivity, "r" indicates a resiliency or increased adaptive capacity, "n" indicates neutral response.

Question	Score	Notes
1. Changes to non-modeled habitat components	v	Large trees may be important roost sites
2. Change in habitat quality	n	No direct link
3. Dispersal ability (Site fidelity or other limitations)	r	Good
4. Reliance on migratory or transitional habitats	n	Non migratory
5. Increase or decrease in physiological range limitation	n	Not limited
6. Sex ratios determined by temperature or food changes	n	No
7. Response to predicted extreme weather events/disturbances	n	Not known
8. Changes to daily activity period	n	No
9. Variable life history traits or coping strategies	r	Torpor. Delayed implantation
10. Ability to outlive limiting conditions	r	Long lived
11. Migrates/hibernates in response to weather cues	v	Yes
12. Reliance on weather mediated resource (e.g. insect emergence)	v	Yes
13. Spatial or temporal separation between critical resources and life history stages	r	Probably not
14. Can adjust timing of critical activities	v	No. Single reproductive event
15. Likelihood for decreased food resource	n	Not predictable at this time
16. Likelihood of increase predation	n	No major predation impact
17. Loss of important symbiotic species	n	None known
18. Increase in high mortality/morbidity disease	n	Not predictable at this time
19. Increased competitive pressures	n	Not predictable at this time

Occult Bat (*Myotis occultus*)

Niche Model Analysis

The model AUC values were 0.82 for all three climate scenarios. Models showed very similar declines in suitable habitat (Table 2, Figures 2 and 3).

Table 2. Percent of original predicted habitat under three climate models.

Time	CGCM3.1	GFDL CM2.1	Had_CM3	Average
current	1	1	1	1
2030	0.67	0.64	0.67	0.66
2060	0.48	0.30	0.48	0.42
2090	0.19	0.16	0.20	0.18

Elevation and slope were the most important predictors of suitability for *M. occultus* (Table 3). *M. occultu*s was associated with sites below 2000m but showed an increasing probability of occurrence in areas as slope increased (Table 3). Suitability also increased with increasing isothermality and potential evapotranspiration. Habitat suitability declined in areas where mean winter temperatures exceeded 3.15°C.

Table 3. Variables in order of importance for species habitat models. Relationship between variable and predicted suitability is listed as positive (+) negative (-) for linear and exponential curves or quadratic for parabola-like curves.

Variables	CGCM3.1	GFDL CM2.1	Had_CM3	Relationship
Elevation	32.3	31.0	31.5	-
Slope	21.5	20.7	21.2	+
Annual Pot Evap Nat Veg	19.3	19.4	20.2	+
Isothermality (bio3)	6.4	7.7	7.0	+
Mean Temp DJF (bio11)	9.2	8.9	8.9	- (After 3°C)

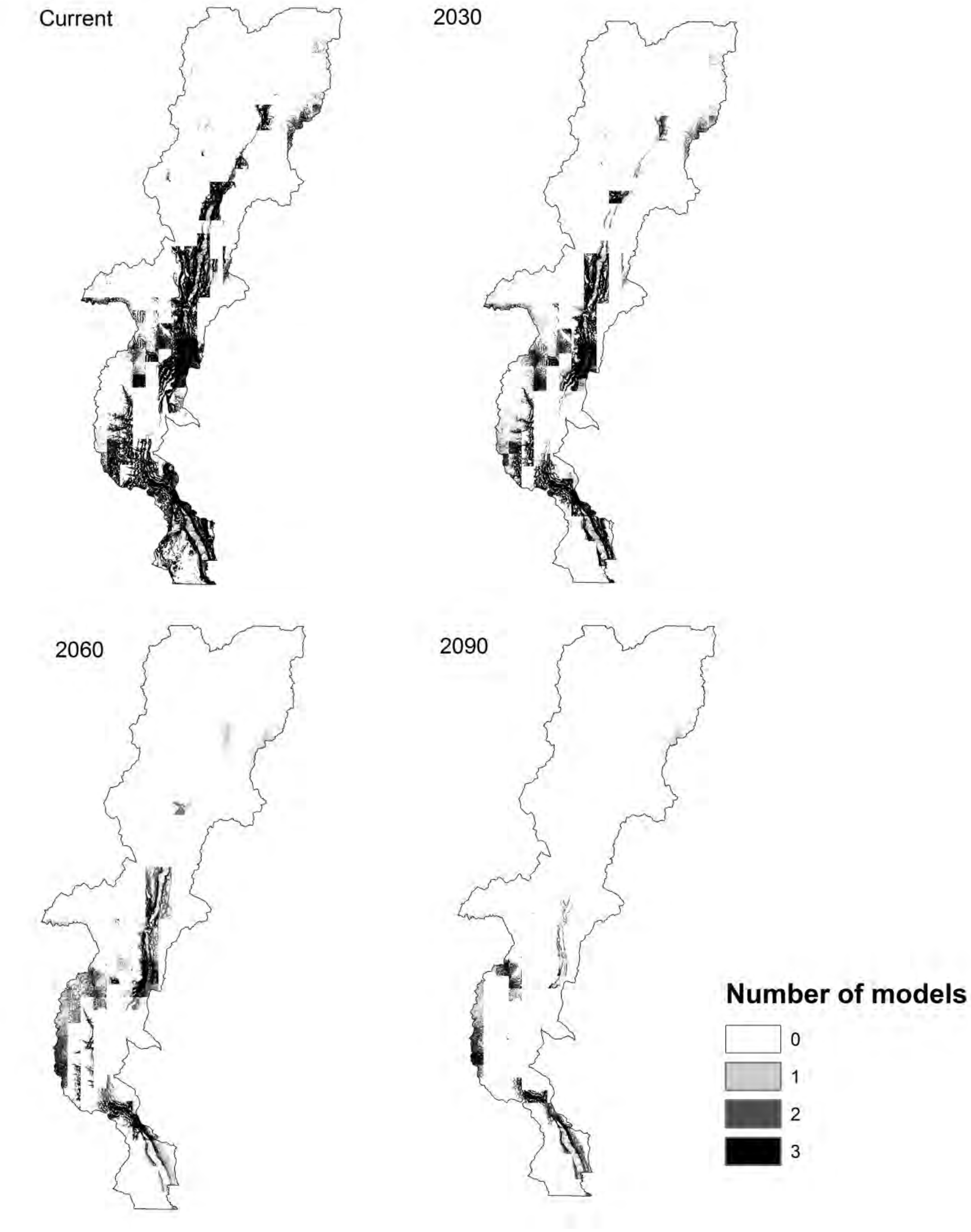


Figure 2. Distribution of suitable habitat for *Myotis occultus* under three climate futures. Habitat was modeled under three climate models: Had_CM3, CGCM3.1, and GFDL_CM2.1. Shading indicates the number of models that predict suitable habitat for a given area.

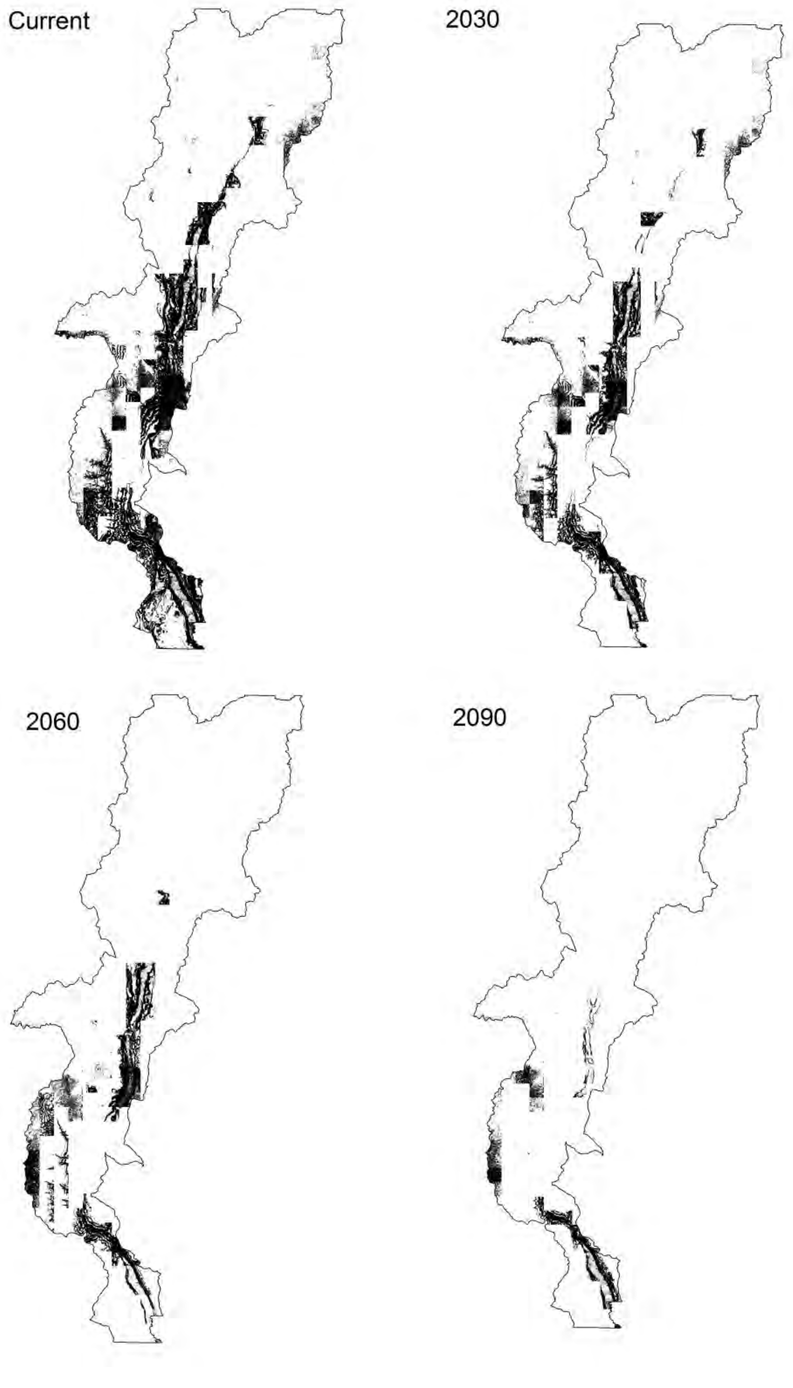


Figure 3. Two thirds consensus models of suitable habitat for *Myotis occultus* under three climate futures. Habitat was modeled under three climate models: Had_CM3, CGCM3.1, and GFDL_CM2.1. Black areas indicate predicted suitable habitat estimated by at least two models.

Occult Bat (*Myotis occultus*)

Fire Impacts

Myotis occultus is likely to benefit from most fire activity except where roosts are impacted. Bats are able to escape fire mortality and are drawn to the insects within newly burned habitats (Malison and Baxter 2010; Lacki et al., 2009). In California, bats benefited from the clearing of vegetation, which improved foraging success and increased insect abundance in early successional habitat (Bhucalski et al., 2013). Fire can also increase roosting habitat for species that rely on snags or tree cavities (Lacki et al., 2009). The long term (>5 years) consequences of fire are probably positive. Over large periods of time, fire contributes to habitat heterogeneity associated with improved conditions for these species (Bhucalski et al., 2013). Fire suppression is considered one of causes leading to reduced habitat for bat species in the SW (Chung-MacCoubrey 2005).

Negative impacts from fire can occur when burns happen in the early to mid-spring, especially around roosting maternity sites (Lacki et al., 2009). At best, fire would represent a short term disturbance to colonies and at worst fire could lead to the loss of roost site and mortality of non-volant young. In addition, large fires that reduce habitat heterogeneity will lead to a decline in habitat quality. To account for the potential negative impacts of fire on roost sites, we classified areas expected to experience crown fires as high risk. Though the presence of torching does not necessarily predict habitat loss, the indication for torching is used to qualify the potential risk for bat species. Here, torching fires are associated with a greater risk than non-torching fires resulting in higher risk to roost sites and less benefit for foraging sites (Figures 4-6).

.

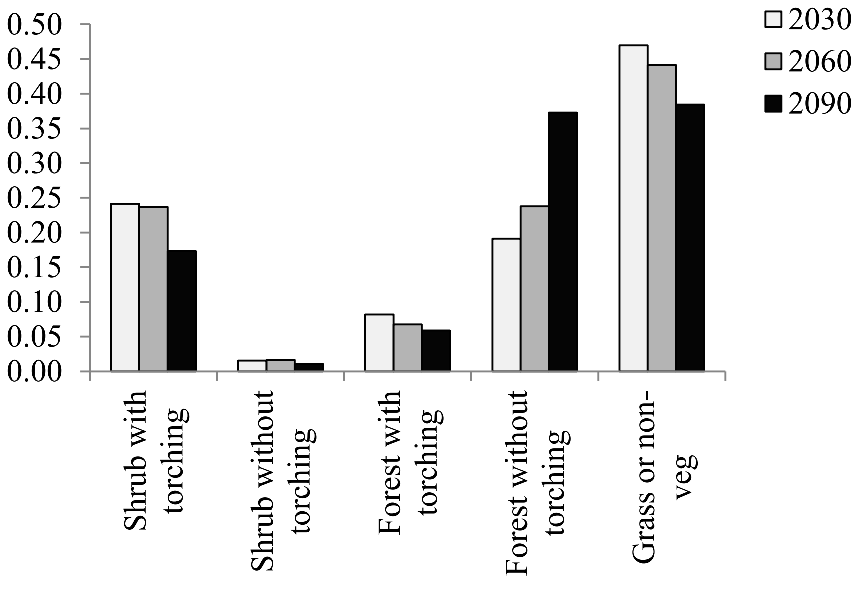
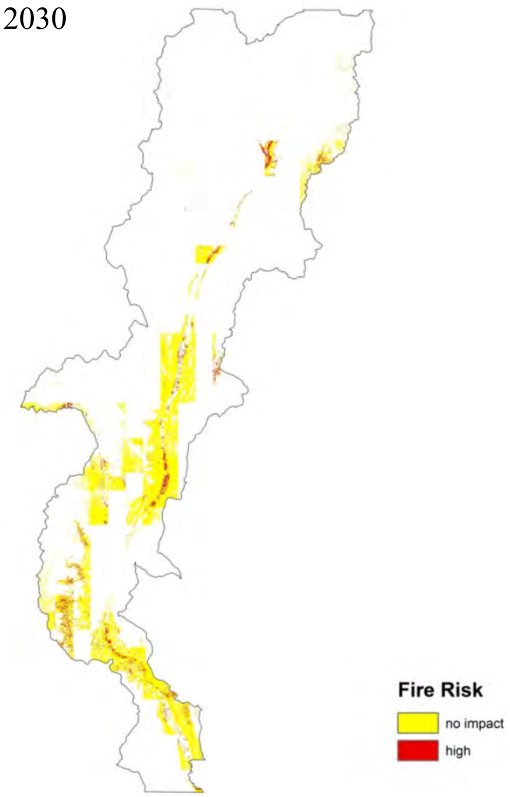
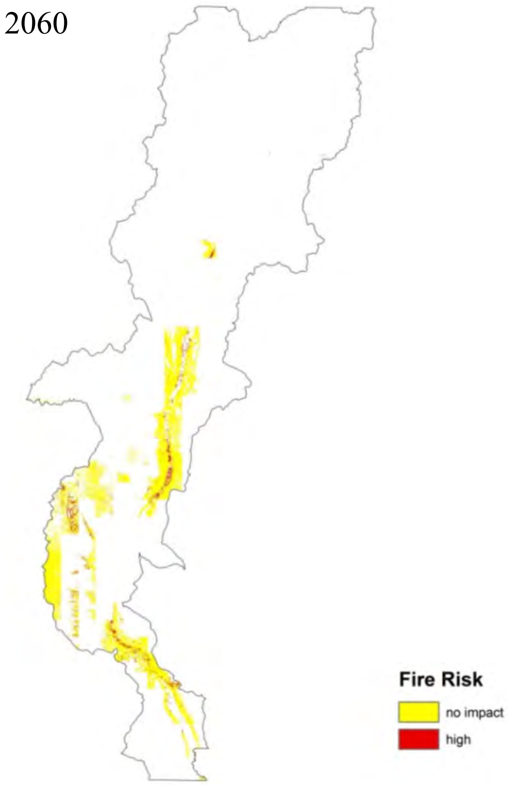


Figure 4. Percent of habitat falling within each fire type category.





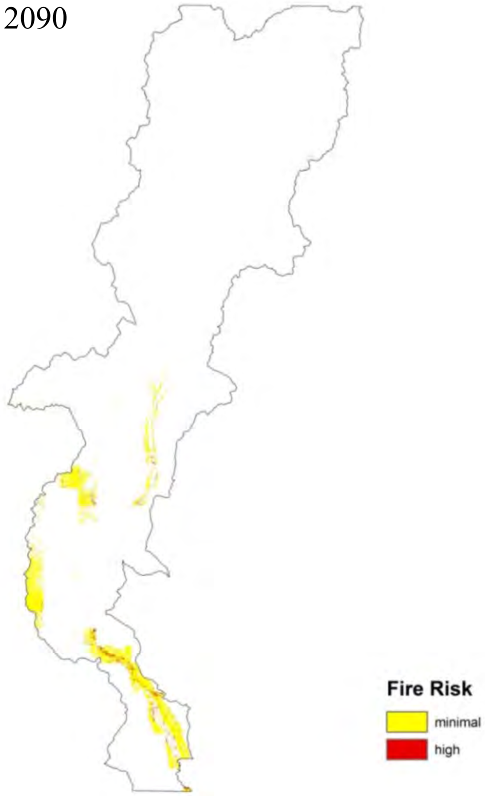
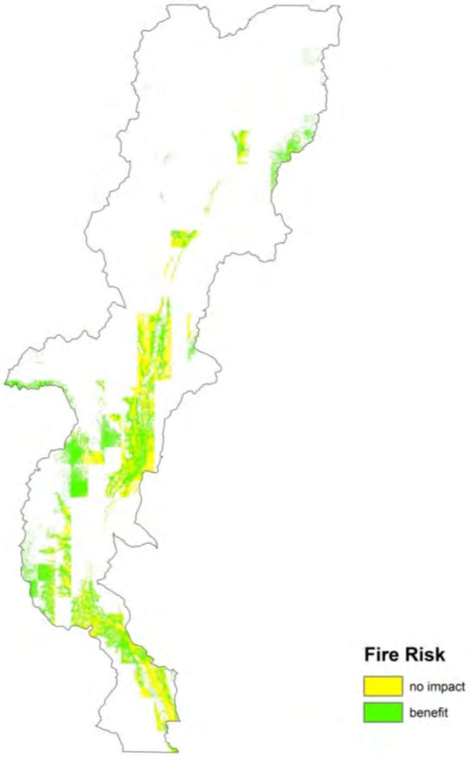
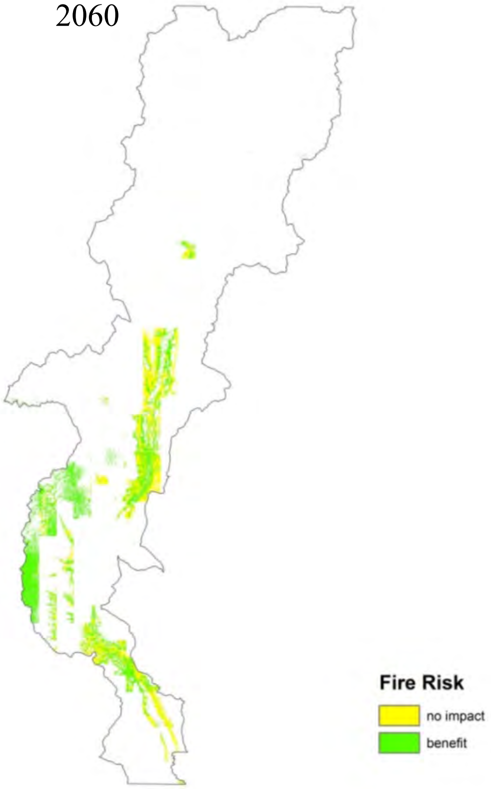


Figure 5. Fire impacts for *Myotis occultus* roosting habitat under three climate futures. Fire intensity was calculated using FSim (Finney et al., 2011) for each time period based on the GFDL-ESM-2m GCM under the RCP85 scenario. Fire risk maps were created by overlaying estimated species' habitat, predicted fire characteristic (shrub with or without torching, forest with or without torching or grass/nonveg) and species' response (negative, neutral, positive).

2030



2060



2090

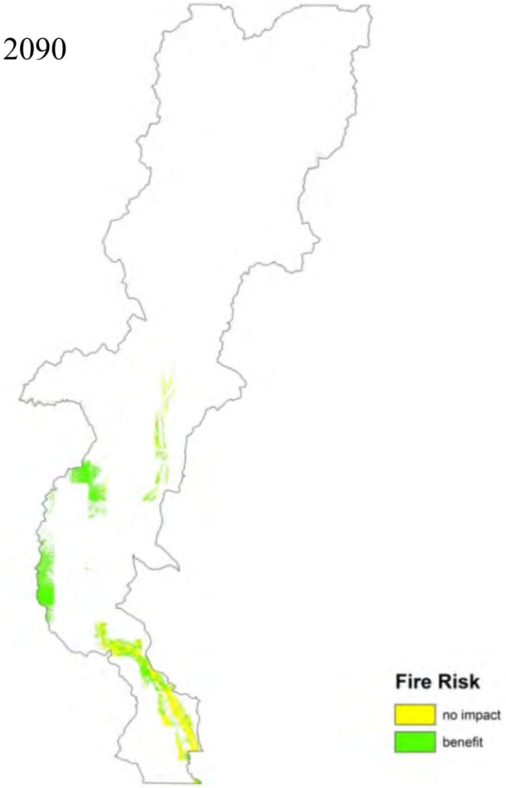


Figure 6. Fire impacts for *Myotis occultus* foraging habitat under three climate futures. Fire intensity was calculated using FSim (Finney et al., 2011) for each time period based on the GFDL-ESM-2m GCM under the RCP85 scenario. Fire risk maps were created by overlaying estimated species' habitat, predicted fire characteristic (shrub with or without torching, forest with or without torching or grass/nonveg) and species' response (negative, neutral, positive).

References:

1. 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.
2. 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.
3. 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. http://www.bison-m.org
4. Carpenter, R. E. 1969. Structure and function of the kidney and the water balance of desert bats. Physiological Zoology 42: 288-301.
5. Chung-MacCoubrey, A. L. 2005. Use of pinyon-juniper woodlands by bats in New Mexico. Forest Ecology and Management 204: 209-220.
6. Fenton, M. B., and R. M. R. Barclay. 1980. Myotis lucifugus. Mammal. Species No. 142.
7. 8pp.
8. Finney, M.A., C.W. McHugh, I.C. Grenfell, K.L. Riley, and K.C. Short. 2011. A Simulation of Probabilistic Wildfire Risk Components for the Continental United States. Stochastic Environmental Research and Risk Assessment 25: 973-1000.
9. Geluso, K. 2007. Winter activity of bats over water and along flyways in New Mexico. The Southwestern Naturalist 52: 482-492.
10. Grindal, S. D., J.L. Morissette, and R.M. Brigham. 1999. Concentration of bat activity in riparian habitats over an elevational gradient. Can. J. Zool. 77: 972-977.
11. Lacki, M. J., J. S. Johnson, E.D. Dodd, and M. D. Baker. 2007. Prey consumption of insectivorous bats in coniferous forest of North-Central Idaho. Northwest Science 81: 199-205.
12. 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.
13. Malison, R.L., and C.V. Baxter, 2010. The fire pulse: wildfire stimulates flux of aquatic prey to terrestrial habitats driving increases in riparian consumers. Canadian Journal of Fisheries and Aquatic Sciences 67: 570-579.
14. 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-
15. 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).
16. Piaggio, A. J., Valdez, E. W., Bogan, M. A. and Spicer, G. S. 2002. Systematics of *Myotis occultus* (Chiroptera: Vespertilionidae) inferred from sequences of two mitochondrial genes. *Journal of Mammalogy* 83: 386-395.
17. Racey, P. A. 1973. The time of onset of hibernation in Pipistrelle bats, *Pipistrellus pipistrellus*. Journal of Zoology 171: 465-467.
18. Scheel, D., T.L.S. Vincent, and G.N. Cameron. 1996. Global warming and the species richness of bats in Texas. Conservation Biology 10: 452-464.

19. Twente, J. W., J. Twente and V. Brack. 1985. The duration of the period of hibernation of three species of vespertilionid bats. II. Laboratory studies. Canadian Journal of Zoology 63: 2955-2961.
20. Warner, R. M. 1985. Interspecific and temporal dietary variation in an Arizona bat community. Journal of Mammalogy 66: 45-61.
21. Warner, R. M., and N. J. Czaplewski. 1984. *Myotis volans*. Mammalian Species 224: 1-4.

Long-legged Bat (*Myotis volans*)

Overview

Climate Impacts	
Habitat Change	
2030	-40% to none
2060	-94% to +7%
2090	-95% to +57%
Adaptive capacity	Intermediate
Fire Response	Positive Overall
Roosts	Neutral
Foraging habitat	Positive

Status:

BISON-M reports concerns with the status of this species due to limited information. *Myotis volans* is considered demonstrably secure in New Mexico by the Natural Heritage Global rank. This species is included on the BLM list of sensitive species and within New Mexico is considered a state sensitive species. In 1996, *M. volans* was listed as a federal species of concern, but this status was removed in 2003. Currently it is listed as “Least Concern” on the IUCN Red List.

Range and Habitat:

Myotis volans inhabits a large range covering western North America (Figure 1), including Alaska and central Mexico. *Myotis v. interior* (type locality in Taos, New Mexico) are typically found in montane habitats and less frequently in arid lowland areas, whereas *M. v. volans* (type locality in Mexico) are common in the desert regions of Baja California. *Myotis volans* are reported in New Mexico from May through September which indicates they use areas in New Mexico for maternity roosting sites. *Myotis volans* is considered a seasonal migrant in New Mexico (BISON-M) and is present within the state from at least May through September. Populations in Texas, Oklahoma, and Kansas appear to be resident whereas Arizona and California populations appear to migrate.*Myotis volans* is primarily associated with forests, though it is also found in riparian and desert habitats (Warner and Czaplewski 1984). Chung-MacCoubrey (2005) found *M. volans* commonly associated with pinyon-juniper habitat in New Mexico. *Myotis volans* is considered a montane species and relatively rare in arid lowland areas. *Myotis volans* may be more common at lower elevations during migration (Frey 1996).

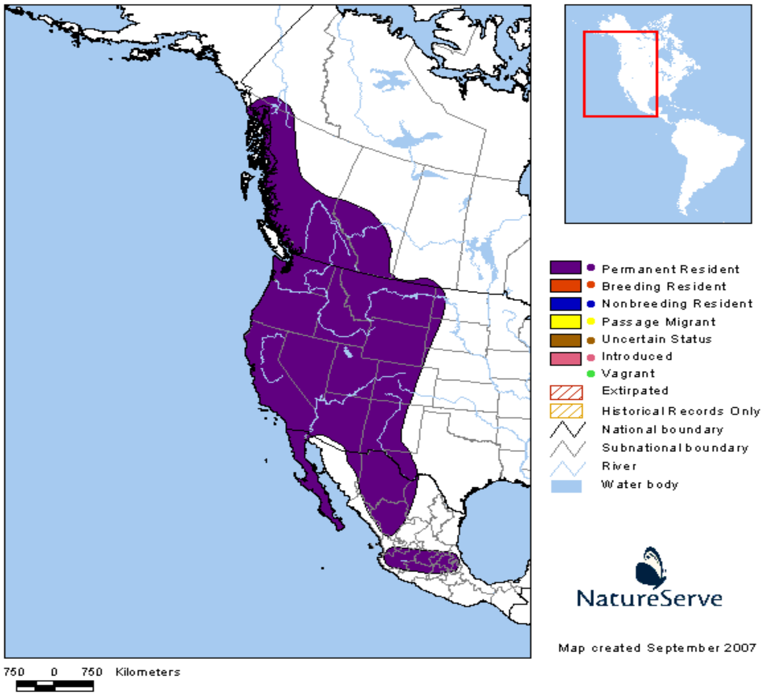


Figure 1. Distribution of *Myotis volans*. From NatureServe 2009.

Long-legged Bat (*Myotis volans*)

Climate Change Impacts and Adaptive Capacity

Adaptive Capacity Score= 0.4830 (intermediate)

The long-legged bat is an aerial forager that may be dependent on canopy structure, a potential source of vulnerability if forest structure changes (Table 1). *Myotis volans* form large colonies and are reported to use a variety of roosts including abandoned buildings, cracks and crevices, and under bark (BISON-M 2009; Arroyo-Barales and Alvarez-Castaneda 2008). Day and night roosts may vary. Arroyo-Barales and Alvarez-Castaneda (2008) noted that maternity roosts are most common in large trees. Winter hibernacula have been reported from caves and mines and are thought to be locally distributed (BISON-M).

Myotis volans is an aerial forager that may be dependent on canopy structure, which may be a source of vulnerability if forest structure changes (Table 1). Scheel et al., (1996) categorize *M. volans* as a crevice roosting species (and not a generalist) and project that such species will experience distribution decreases in response to global warming in TX as roost sites and habitat become spatially separated. Roost disturbance has a huge impact on bat populations, especially during hibernation (frivolous use of energy) or when roosts contain young (fall to their death). Displacement during hibernation can result in death by starvation if bats are forced to deplete their fat reserves (BISON-M 2009)

Physiology:
Bats require a water source for survival, especially in hot climates. However, there is little evidence that these bats will be directly limited by temperature. *Myotis volans* are most active at temperatures between 12 and 18 °C (Warner and Czaplewski 1984) a range somewhat narrower than the other *Myotis* species assessed within this study. Peak activity is 3-4 hours after sunset. Higher temperatures are favorable for gestation and lactation phases of female reproductive cycles (McCain 2007), but lactating females have an increased need for water (Adams and Hayes 2008). 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). Lack of water will limit the capacity of this species to persist within New Mexico under warmer climates.

Myotis volans have some capacity to deal with resource variation. Most bat species inhabiting this region undergo hibernation and use torpor to mitigate food shortages and evaporative water loss. There is some risk that high temperature roosts may prevent hibernation and torpor. Female bats are known to reabsorb/abort embryos (Anthony et al., 1981) and will abandon neonates if stressed. Females may forgo reproduction in harsh years and will not ovulate if their fat reserves breed in their first year although males are thought to breed by their 2nd year (Warner and Czaplewski 1984). Individuals may live up to 21 years in the wild, which improves the chances that they will be able to outlive droughts and other limiting conditions.

Phenology:

Myotis volans may be more sensitive than other bat species to phenology as they are thought to follow seasonal shifts in habitat (Table 1). Timing and metabolic characteristics of bat hibernation is influenced by temperature (Twente et al., 1985). Additionally, reproductive cycles are generally tied to insect activity, which may shift under changing climate regimes (Anthony et. al. 1981). Temperature probably relates directly to insect emergence, although there is some risk of unnaturally warm spring days (that occur before frost season is over) leading to premature emergence. Though females have some capacity to modify gestation and ovulation, the result of non-favorable conditions is the loss of pregnancy. *Myotis volans* give birth to a single young per year (Warner and Czaplewski 1984) leaving them at an increased risk of reproductive failure due to timing mismatches.

Biotic Interactions:

We did not find evidence for a strong interaction effect due to climate change (Table 1). *Myotis volans* is an aerial forager that prefers moths, though they do feed on a variety of other insects (Warner and Czaplewski 1984; Warner 1985; Bison-M 2009). In a comparison of several species, *M. volans* showed the least diversity in its diet (Warner 1985). *Myotis volans* was considered to have a moderate level of diet specialization on moths in another study (Lacki et al., 2007). It pursues prey in the air rather than skimming water surfaces (Warner and Czaplewski 1984). This bat species has had several parasite reports including mites, chiggers, fleas, bat bugs and nycteribiid flies. It has also been found to harbor rabies (Warner and Czaplewski 1984). However, there is no known relationship between expected climate changes and incidence or mortality from these parasites and diseases.

Table 1. Scoring sheet for Long-legged bat (*Myotis volans).*"v" indicates a vulnerability or sensitivity, "r" indicates a resiliency or increased adaptive capacity, "n" indicates neutral response.

Question	Score	Notes
1. Changes to non-modeled habitat components	v	Distance between roost and foraging sites may increase. Reduced canopy.
2. Change in habitat quality	n	not known
3. Dispersal ability (Site fidelity or other limitations)	r	Good dispersal
4. Reliance on migratory or transitional habitats	v	Migrates
5. Increase or decrease in physiological range limitation	v	Water limitation
6. Sex Ratios determined by temperature or food changes	n	no
7. Response to predicted extreme weather events/disturbances	n	Not known
8. Changes to daily activity period	n	No
9. Variable life history traits or coping strategies	r	Torpor and delayed gestation

Question	Score	Notes
10. Ability to outlive limiting conditions	r	Long-lived
11. Migrates/hibernates in response to weather cues	v	Yes
12. Reliance on weather mediated resource (e.g. insect emergence)	v	Yes
13. Spatial or temporal separation between critical resources and life history stages	r	Not known. Likely local migrant
14. Can adjust timing of critical activities	v	Limited reproductive window
15. Likelihood for decreased food resource	n	Moth specialist. Not predictable at this time
16. Likelihood of increase predation	n	No major predation impact
17. Loss of important symbiotic species	n	None known
18. Increase in high mortality/morbidity disease	n	Not predictable at this time
19. Increased competitive pressures	n	Not predictable at this time

Long-legged Bat (*Myotis volans*)

Niche Model Analysis

Model AUC values were 0.86, 0.85, and 0.84 for CGCM3, GDFLCM2 and HadCM3, respectively. Models predicted either an substantial decline (>90%) in suitable habitat or a 50% increase (Had_CM3) (Table 2, Figures 2 and 3).

Table 2. Percent of original predicted habitat under three climate models.

Time	CGCM3.1	GFDL CM2.1	Had_CM3	Average
current	1	1	1	1
2030	0.60	0.89	1.03	0.84
2060	0.06	0.24	1.07	0.45
2090	0.13	0.05	1.57	0.59

Across all three climate futures average summer precipitation (JJA), slope, and biome accounted for 70% of variable contributions (Table 3). Suitability increased rapidly with increasing monsoon precipitation and declined at higher values (highest values for areas with a mean of 31 mm). Similarly, suitability increased with increasing potential evaporation of natural vegetation up to a point after which this variable became less important for determining change in suitability (suitability includes sites with values greater than 29mm). *Myotis volans* was positively associated with habitats within Plains Grassland and Great Basin Conifer Woodland biomes and had a slight negative association with habitats within the Rocky Mountain Montane Conifer Forest biome. This species was also associated with areas with lower mean winter precipitation. Divergence in habitat trends relates to differences among the climate scenarios: Had_CM3 showed an increase in average summer precipitation, whereas the other models showed declines.

Table 3. Variables in order of importance for species habitat models. Relationship between variable and predicted suitability is listed as positive (+) negative (-) for linear and exponential curves or quadratic for parabola-like curves.

Variable	CGCM3.1	GFDL CM2.1	Had_CM3	Relationship
Precip JJA (bio16)	46.9	44.1	42.8	+
Slope	15	13.9	15.5	+
Biome	9.7	11.4	10.1	+Plns Gssld, GB ConWdld -RMMonConFor
Precip DJF (bio17)	8.6	5.9	6.9	Peaks at low values
Annual Pot Evap Nat Veg	6.4	7	6.9	Quadratic
Distance to water	6.1	7.6	5.8	-
Isothermality (bio3)	3.9	5.4	8.6	- (after 49)
Temp Annual Range (bio7)	3.4	4.7	3.4	-

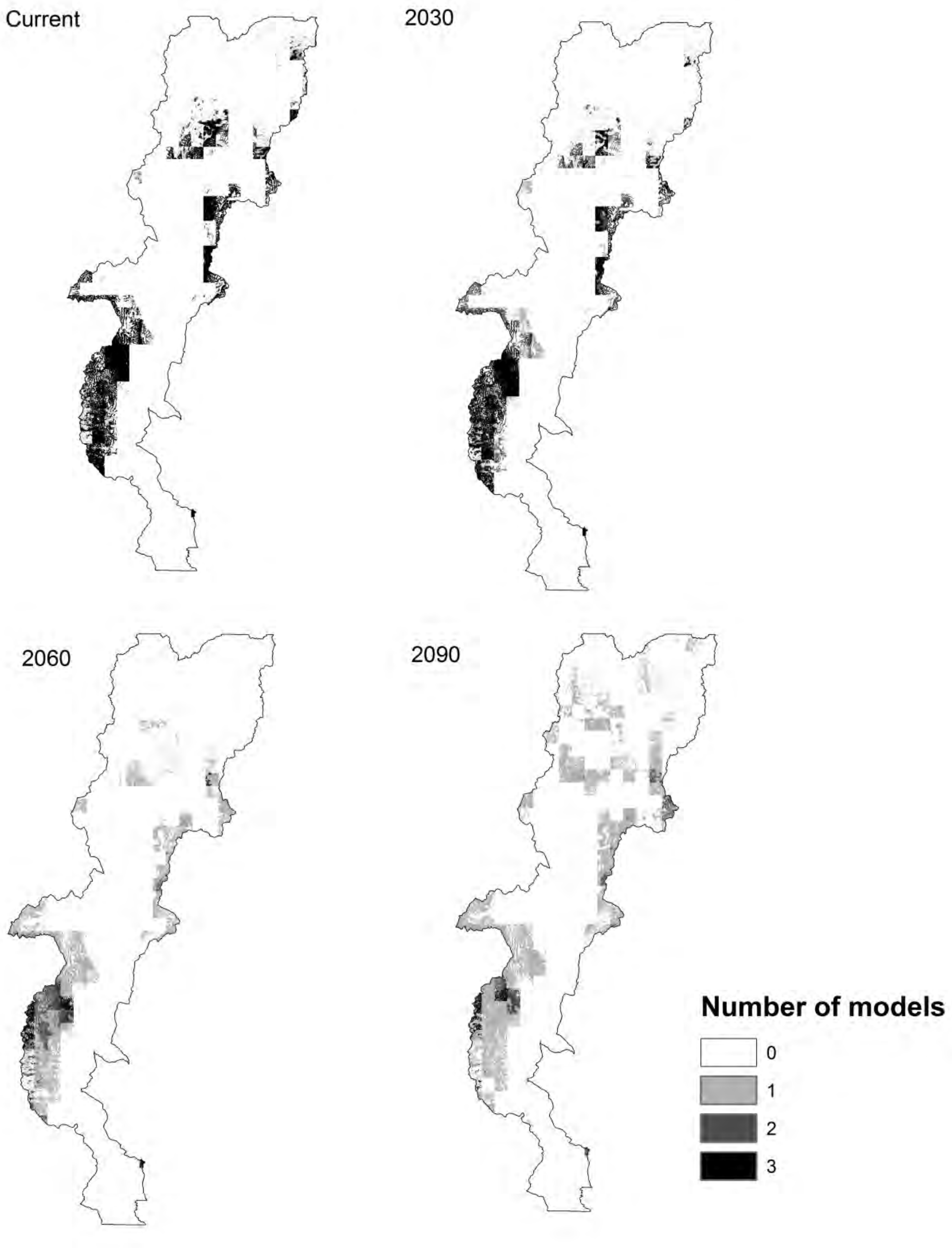


Figure 2. Distribution of suitable habitat for *Myotis volans* under three climate futures. Habitat was modeled under three climate models: Had_CM3, CGCM3.1, and GFDL_CM2.1. Shading indicates the number of models that predict suitable habitat for a given area.

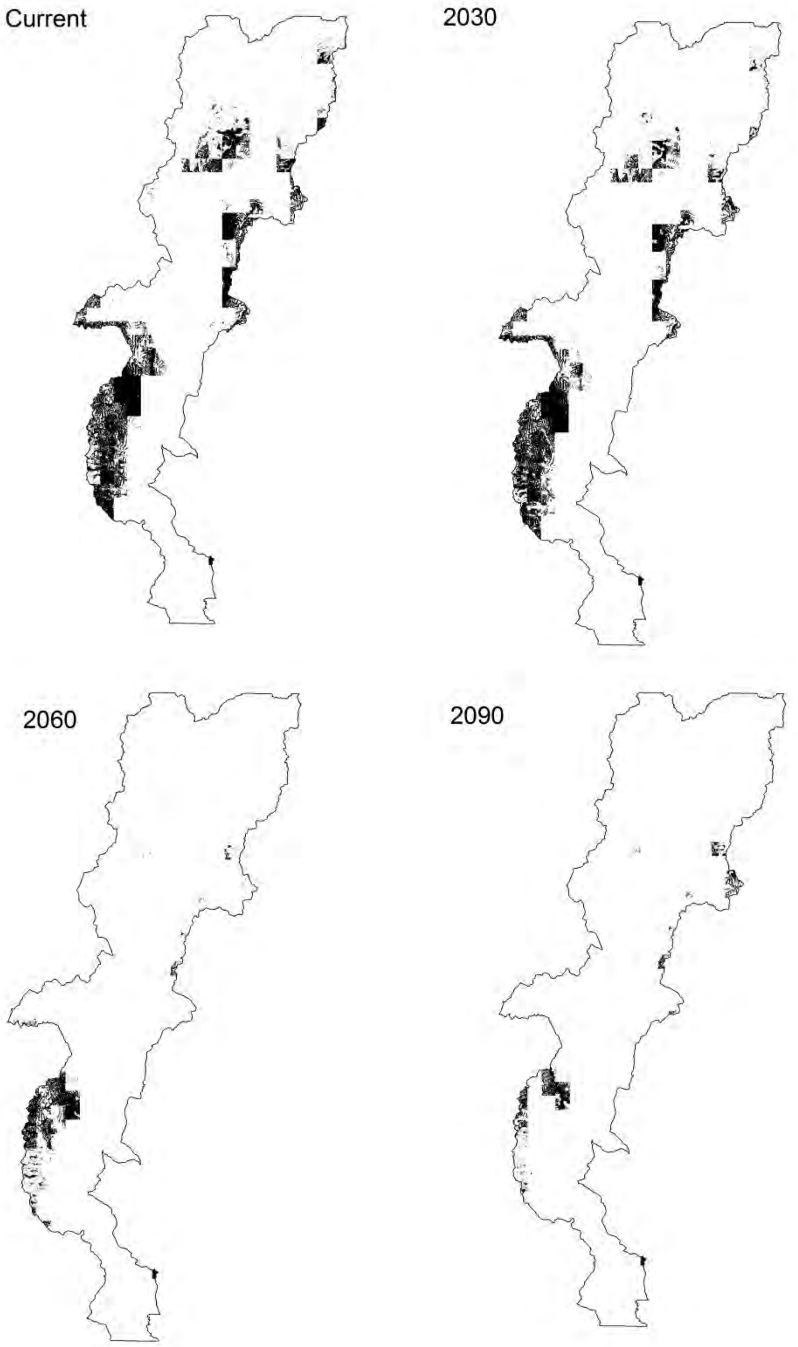


Figure 3. Two thirds consensus models of suitable habitat for *Myotis volans* under three climate futures. Habitat was modeled under three climate models: Had_CM3, CGCM3.1, and GFDL_CM2.1. Black areas indicate predicted suitable habitat estimated by at least two models.

Long-legged Bat (*Myotis volans*)

Fire Impacts

Myotis volans is likely to benefit from most fire activity except where roosts are impacted. Bats are able to escape fire mortality and are drawn to the insects within newly burned habitats (Malison and Baxter 2010; Lacki et al., 2009). In California, bats benefited from the clearing of vegetation, which improved foraging success, and increased insect abundance in early successional habitat (Bhucalski et al., 2013). Fire can also increase roosting habitat for species that rely on snags or tree cavities (Lacki et al., 2009). The long term (>5 years) consequences of fire are probably positive. Over large periods of time, fire contributes to habitat heterogeneity associated with improved conditions for these species (Bhucalski et al., 2013). Fire suppression is considered one of the causes leading to reduced habitat for bat species in the SW (Chung-MacCoubrey 2005).

Negative impacts from fire can occur when burns happen in the early to mid-spring especially around roosting maternity sites (Lacki et al., 2009). At best, fire would represent a short term disturbance to colonies and at worst fire could lead to the loss of roost site and mortality of non-volant young. In addition, large fires that reduce habitat heterogeneity will lead to a decline in habitat quality. To account for the potential negative impacts of fire on roost sites, we classified areas expected to experience crown fires as high risk. Though the presence of torching does not necessarily predict habitat loss, the indication for torching is used to qualify the potential risk for bat species. Here, torching fires are associated with a greater risk than non-torching fires resulting in higher risk to roost sites and less benefit for foraging sites (Figures 4-6).

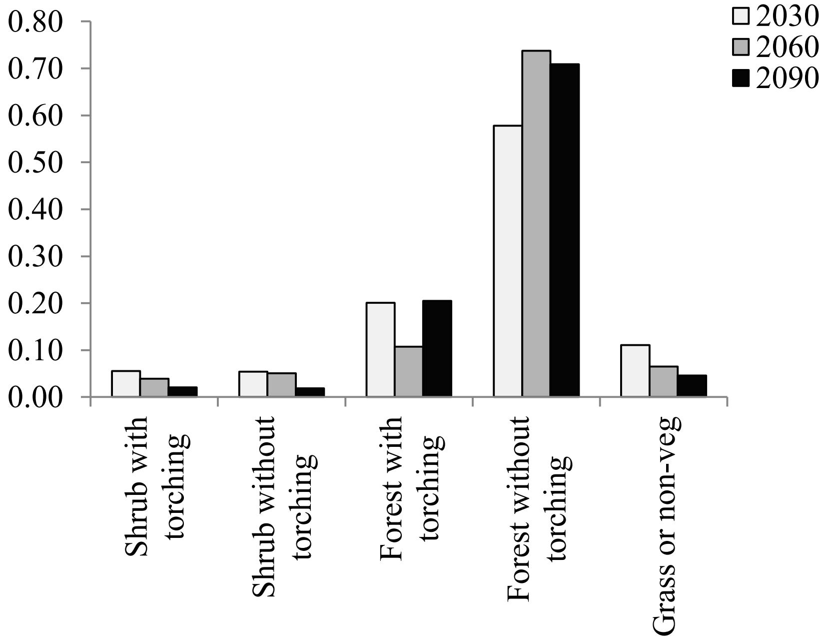
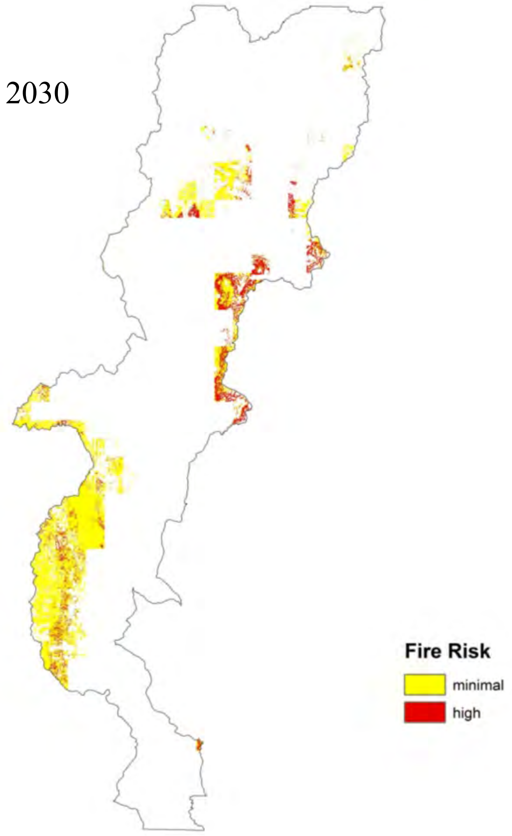


Figure 4. Percent of habitat falling within each fire type category.



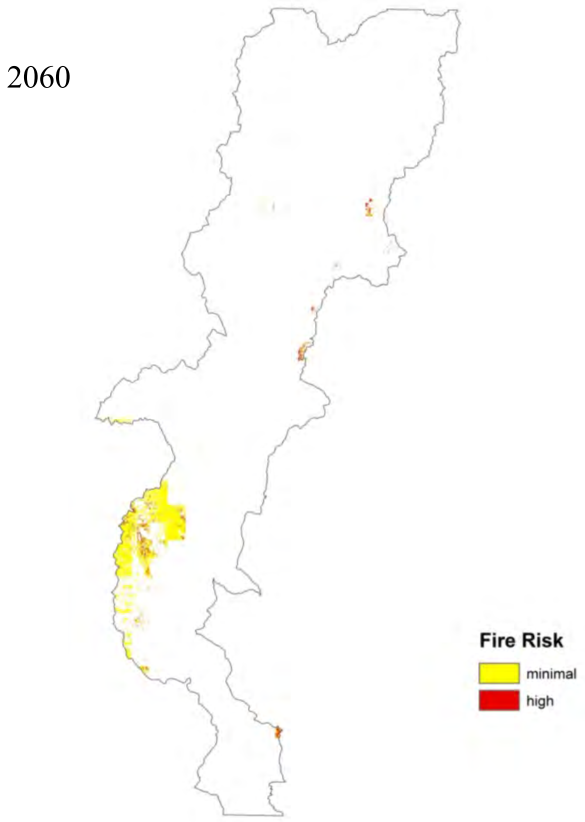
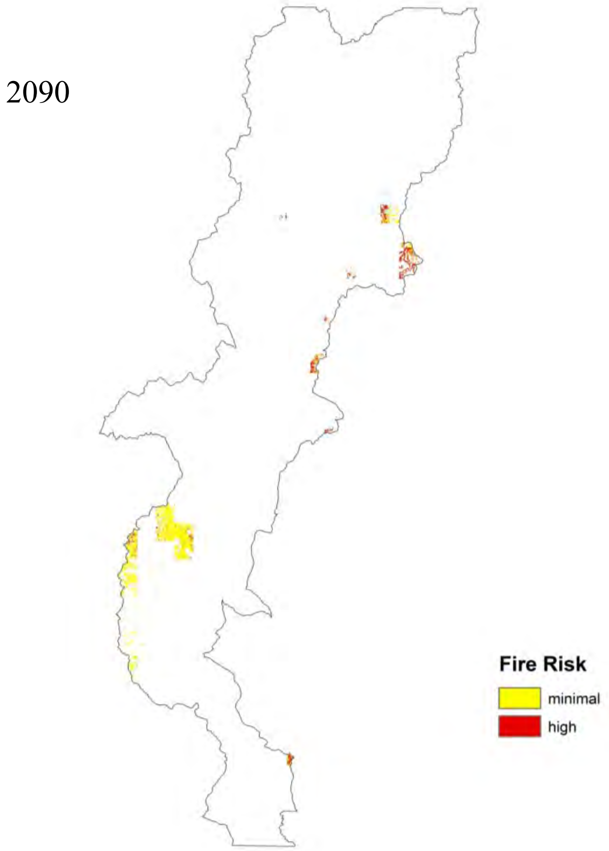
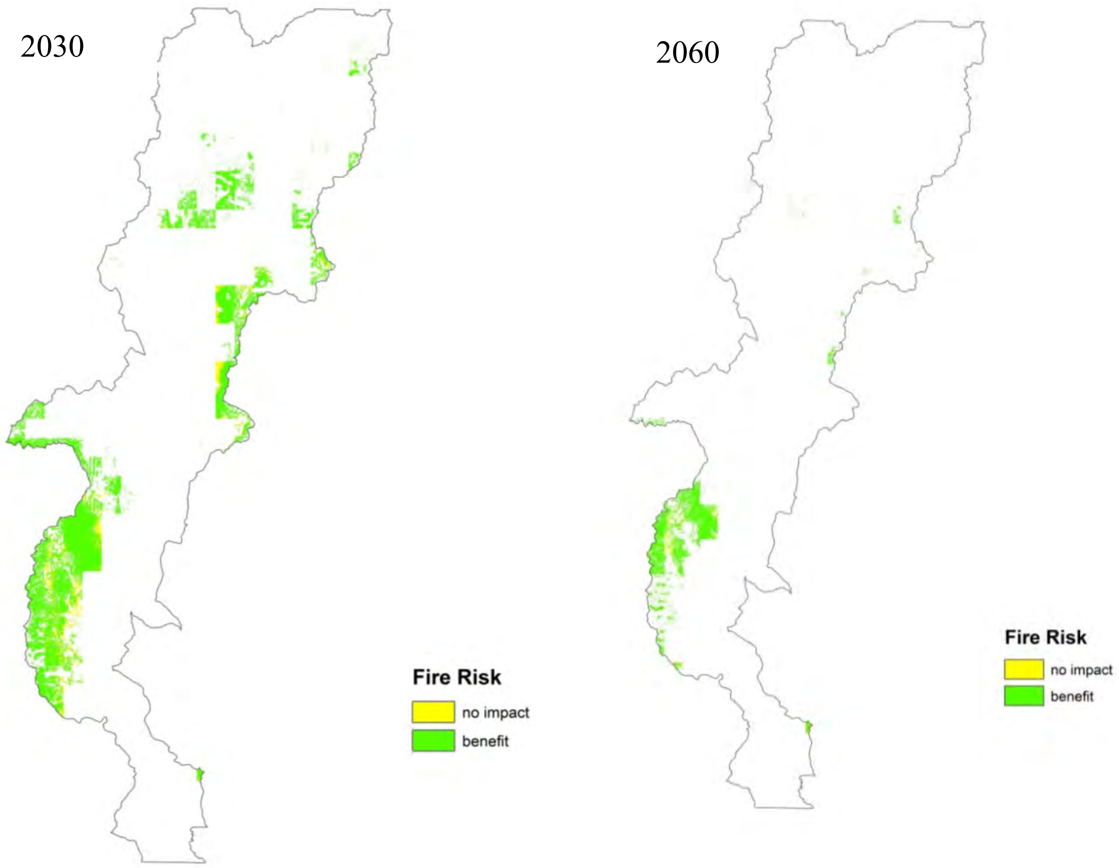


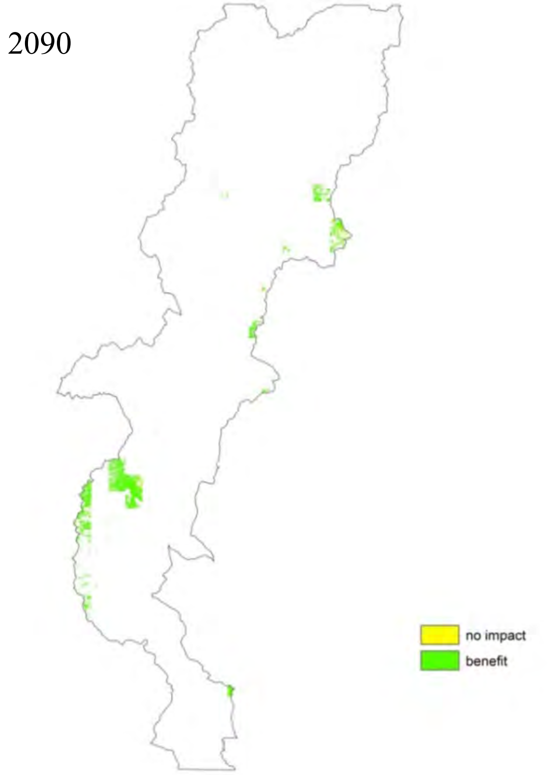
Figure 5. Fire impacts for *Myotis volans'* roosting habitat under three climate futures. Fire intensity was calculated using FSim (Finney et al., 2011) for each time period based on the GFDL-ESM-2m GCM under the RCP85 scenario. Fire risk maps were created by overlaying estimated species' habitat, predicted fire characteristic (shrub with or without torching, forest with or without torching or grass/nonveg) and species' response (negative, neutral, positive).





2030

2060



2090

Figure 6. Fire impacts for *Myotis volans’* foraging habitat under three climate futures. Fire intensity was calculated using FSim (Finney et al., 2011) for each time period based on the GFDL-ESM-2m GCM under the RCP85 scenario. Fire risk maps were created by overlaying estimated species’ habitat, predicted fire characteristic (shrub with or without torching, forest with or without torching or grass/nonveg) and species’ response (negative, neutral, positive).

References:

1. 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.
2. 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.
3. Arroyo-Cabrales, J. and S.T. Álvarez-Castañeda. 2008. *Myotis volans*. The IUCN Red List of Threatened Species. Version 2014.3. <www.iucnredlist.org>. Downloaded on 08 December 2014.
4. 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
5. Buhcalski, M.R., J.B. Fontaine, P.A. Heady III, J.P. Hayes and W.F. Frick. 2013. Bat response to differing fire severity in mixed-conifer Forest California, USA. PLOS One 8: e57884
6. Carpenter, R E. 1969. Structure and function of the kidney and the water balance of desert bats. Physiological Zoology 42: 288-301.
7. Chung-MacCoubrey, A. L. 2005. Use of pinyon-juniper woodlands by bats in New Mexico. Forest Ecology and Management 204: 209-220.
8. Finney, M.A., C.W. McHugh, I.C. Grenfell, K.L. Riley, and K.C. Short. 2011. A Simulation of Probabilistic Wildfire Risk Components for the Continental United States. Stochastic Environmental Research and Risk Assessment 25: 973-1000.
9. Frey, J.K., and T.L. Yates. 1996. Mammalian Diversity in New Mexico. New Mexico Journal of Science 36: 4-37.
10. Geluso, K. 2007. Winter activity of bats over water and along flyways in New Mexico. The Southwestern Naturalist 52: 482-492.
11. Grindal, S.D., J.L. Morissette and R. M. Brigham. 1999. Concentration of bat activity in riparian habitats over an elevational gradient. Can. J. Zool. 77: 972-977.
12. Lacki, M.J., D.R. Cox, L.E. Dodd and M.B. Dickinson. 2009. Response of northern bats (*Myotis septentrionalis*) to prescribed fires in eastern Kentucky forests. Journal of Mammalogy 90: 1165-1175.
13. Lacki, M. J., J.S. Johnson, E.D. Dodd and M D. Baker. 2007. Prey consumption of insectivorous bats in coniferous forest of North-Central Idaho. Northwest Science 81: 199-205.
14. 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.
15. Malison, R.L., and C.V. Baxter, 2010. The fire pulse: wildfire stimulates flux of aquatic prey to terrestrial habitats driving increases in riparian consumers. Canadian Journal of Fisheries and Aquatic Sciences 67: 570-579.
16. 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.
17. 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).

18. Scheel, D., T.L.S. Vincent, and G.N. Cameron. 1996. Global warming and the species richness of bats in TX. Conservation Biology 10: 452-464.
19. Twente, J.W., J. Twente and V. Brack. 1985. The duration of the period of hibernation of three species of vespertilionid bats. II. Laboratory studies. Canadian Journal of Zoology 63: 2955-2961.
20. Warner, R.M. 1985. Interspecific and temporal dietary variation in an Arizona bat community. Journal of Mammalogy 66: 45-61.
21. Warner, R.M. and N. J. Czaplewski. 1984. *Myotis volans*. Mammalian Species 224: 1-4.

Yuma Bat (*Myotis yumanensis*)

Overview

Predicted Impacts	
Habitat Change	
2030	21-71% Loss
2060	40-83% Loss
2090	80-92% Loss
Adaptive capacity	Intermediate
Fire Response	Positive Overall
Roost	Neutral
Forage	Positive

Status:

Currently 56% of the bat species present in North America are listed or being considered for listing under the Endangered Species Act. The primary reasons for bat population declines include habitat loss, water contamination, pesticide poisoning and, most importantly, disturbance and destruction of day roosts (Evelyn et al., 2004). There are five subspecies of *Myotis. yumanensi*s, all of which inhabit SW desert areas (Sims 2000). The status of *M. yumanensis* is stable according to the Natural Heritage ranking but is considered sensitive by the New Mexico Bureau of Land Management. In addition, BISON-M (2009) cites several concerns including 1) a lack of information, 2) high risk of disturbance (easily impacted), and 3) habitat degradation.

Range and Habitat:

Myotis yumanensis is found in a variety of habitats ranging from Canada to Central Mexico (Figure 1). This bat is strongly associated with water, more so than most other bat species (Sims 2000). Chung-MacCoubrey 2005 notes that *M. yumanensis* are commonly captured at lower elevations. *Myotis yumanensis* is a year round resident in New Mexico (NatureServe 2009) though other sources list this as a migratory species (BISON-M 2009). It appears that these bats are year-round residents in riparian, agricultural, urban, and scrub habitats in New Mexico (BISON-M 2009). They have also been reported from mines (BISON-M 2009).

Bat species diversity is greatest at mid-elevation habitats in arid regions where temperature and precipitation are optimal for bat reproduction (McCain 2007). Reproductive females are thought to choose low elevation sites because higher temperatures typically lead to higher insect concentrations and aid embryo development (Grindal et al., 1999). Conversely, males and juveniles seek out cooler habitats that allow them to utilize torpor. *Myotis yumanensis* is highly commensal with man (BISON-M 2009).

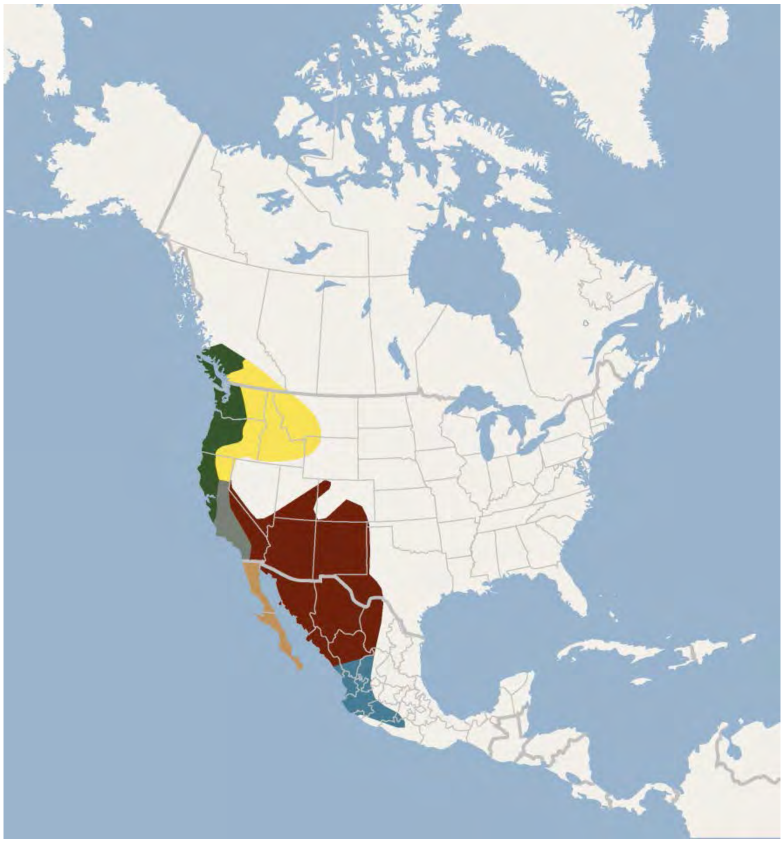


Figure 1. Distribution of *Myotis yumanensis* according to Kays and Wilson 2009.

Yuma Bat (*Myotis yumanensis*)

Climate Change Impacts and Adaptive Capacity

Adaptive Capacity Score = -0.15 (intermediate)

In general, *M. yumanensis* has some capacity to cope with environmental change (Table 1). The greatest risk for bats outside of habitat loss is the potential for changes in the abundance and timing of insect food resources. Little is known about how insect populations will be influenced by climate change. For this report, insect abundance is presumed to be adequate provided foraging habitat is available. Uncertainty exists around where bats roost and their eating habits.

The Yuma bat is known to use a diversity of roost sites. Scheel et al., (1996) categorize *M. yumanensis* as a crevice dwelling species. However, evidence seems to support the notion that these bats roost in trees as well. This species is commonly cited as using human structures for day roost sites, but may in fact be utilizing trees near human structures (Evelyn et al., 2004). Evelyn et al., (2004) found that bats had fairly strong preference for living large diameter trees. Yuma bats are usually found foraging only near bodies of water or streams, (BISON-M 2009) perhaps due in part to an increased need for water by lactating females (Adams and Hayes 2008). They exhibit some site fidelity behavior with average roost fidelity of 4.8 days and associated travel length of 2km in one study (Evelyn et al., 2004).

Physiology Limits:

Myotis yumanensis appear to be resilient to physiological stresses due to climate change (Table 1). Aerial foraging is limited by insect activity which in turn is limited by temperature (Anthony et al., 1981), but the bats themselves are not considered especially sensitive to air temperatures. Yuma bats have been found to forage in temperatures ranging from 18-28° C (BISON-M 2009). These bats appear to like very warm roosts for nurseries and will seek to keep roosts at 40°C (BISON-M 2009).

Though bats show little ability to concentrate urine or other physical traits for water conservation, 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). Bats exhibit several flexible strategies for dealing with resource variation. Torpor is used to reduce evaporative loss (Carpenter 1969). Female bats are known to reabsorb/abort embryos (Anthony et al., 1981) and will abandon neonates if stressed. Gestation may also be delayed under cool conditions in order that female bats may continue to enter torpor (seen in Pipistrelles, Racey 1973). Females may forgo reproduction in harsh years and will not ovulate if their fat reserves are overly depleted when they arise from hibernation (Chung-MacCoubrey 2005). Reproductive females are thought to choose low elevation sites because higher temperatures typically lead to higher insect concentrations and aid embryo development (Grindal et al., 1999). Conversely, males and juveniles seek out cooler habitats that allow them to utilize torpor.

Phenology:

Myotis yumanensis may be negatively impacted by shifts in the timing of seasonal changes in temperature and insect emergence. Timing and metabolic characteristics of bat hibernation is influenced by temperature (Twente et al., 1985). Additionally, reproductive cycles are generally tied to insect activity, which may shift under changing climate regimes. Though females have some capacity to modify gestation and ovulation, the result of non-favorable conditions is the loss of pregnancy. Bats ovulate once per year and give birth in the spring to a single offspring.

Biotic Interactions:

The Yuma bat eats a variety of insects that are usually captured using water-surface foraging techniques. Though *M. yumansis* has been found infected by rabies there are no indications that climate is going to increase incidence. There is no evidence that predation or competitive interactions influence bat demographics to a significant degree.

Table 1. Scoring sheet for Yuma bat (*Myotis yumanensis).* "v" indicates a vulnerability or sensitivity, "r" indicates a resiliency or increased adaptive capacity, "n" indicates neutral response.

Question	Score	Note
1. Changes to non-modeled habitat components	v	Large trees may be important roost sites. Forages over open water bodies.
2. Change in habitat quality	n	Commensal with humans, diversity of roost sites
3. Dispersal ability (Site fidelity or other limitations)	r	Good dispersal ability. Some site fidelity observed
4. Reliance on migratory or transitional habitats	n	Not likely in New Mexico
5. Increase or decrease in physiological range limitation	n	Seems to tolerate high temperatures well
6. Sex Ratios determined by temperature or food changes	n	No
7. Response to predicted extreme weather events/disturbances	n	None known
8. Changes to daily activity period	n	Minimal- nocturnal species
9. Variable life history traits or coping strategies	r	Females can delay or abort pregnancies under low resources. Use torpor to minimize water loss.
10. Ability to outlive limiting conditions	r	Bats are generally long lived (>20 years)
11. Migrates/hibernates in response to weather cues	v	Yes
12. Reliance on weather mediated resource (e.g. insect emergence)	v	Yes
13. Spatial or temporal separation between critical resources and life history stages	r	No

Question	Score	Note
14. Can adjust timing of critical activities	v	Single reproductive event per year. Single offspring.
15. Likelihood for decreased food resource	n	Not predictable at this time
16. Likelihood of increase predation	n	No major predation impact
17. Loss of important symbiotic species	n	None known
18. Increase in high mortality/morbidity disease	n	Not predictable at this time
19. Increased competitive pressures	n	Not predictable at this time

Yuma Bat (*Myotis yumanensis*)

Niche Model Analysis

Model AUC values were 0.87, 0.87 and 0.85 for CGCM3, GDFLCM2 and HadCM3, respectively. Suitable habitat declines rapidly under all three models (Table 2, Figures 2 and 3).

Table 2. Percent of original predicted habitat under three climate models.

Time	CGCM3.1	GFDL CM2.1	Had_CM3	Average
current	1	1	1	1
2030	0.39	0.68	0.79	0.62
2060	0.16	0.14	0.60	0.30
2090	0.08	0.19	0.20	0.16

Variables describing temperature conditions, and in particular the range of temperature variations, contributed the most to model fit (Table 3). Models predicted areas with intermediate minimum winter temperature values, relatively high temperature variations (Diurnal temperature range and isothermality) and moderate annual temperature ranges as having the highest probability of occurrence. Predicted probability declined rapidly with increasing distance from water and was greatest at lower elevations (below 1600-1700m).

Table 3. Variables in order of importance for species habitat models. Relationship between variable and predicted suitability is listed as positive (+) negative (-) for linear and exponential curves or quadratic for parabola-like curves.

Variable	CGCM3.1	GFDL CM2.1	Had_CM3	Relationship
Isothermality (bio3)	25.6	31.2	28	Peaks at high values
Annual Pot Evap Nat Veg	17.7	19.2	21.6	+
Distance to water	15.3	12.8	14.8	-
Annual Range Temp (bio7)	12.4	13.5	12.5	Peaks at lower values
Min Temp January (bio6)	11.2	6.3	11.1	+ (above -0.20°C)
Mean Temp DJF (bio11)	8.7	9.3	6.8	Quadratic
Elevation	4.8	4.1	3.8	-
Diurnal Temp Range (bio2)	4.2	3.4	1.3	+ (when greater than 19°C)

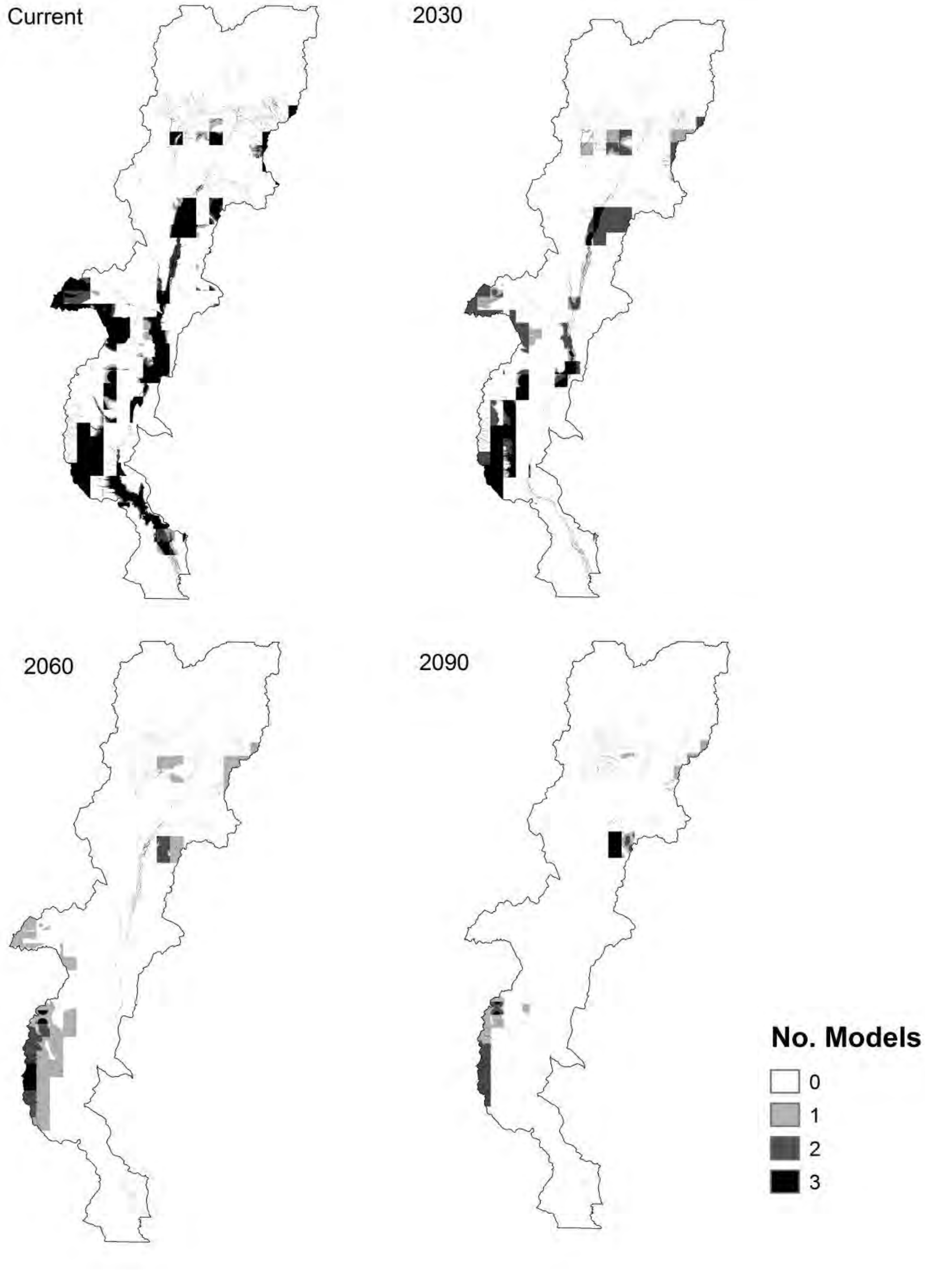


Figure 2. Distribution of suitable habitat for *Myotis yumanensis* for three future time periods. Future habitat predictions are based on three climate models: Had_CM3, CGCM3.1, and GFDL_CM2.1. Shading indicates the number of models that predict suitable habitat for a given area.

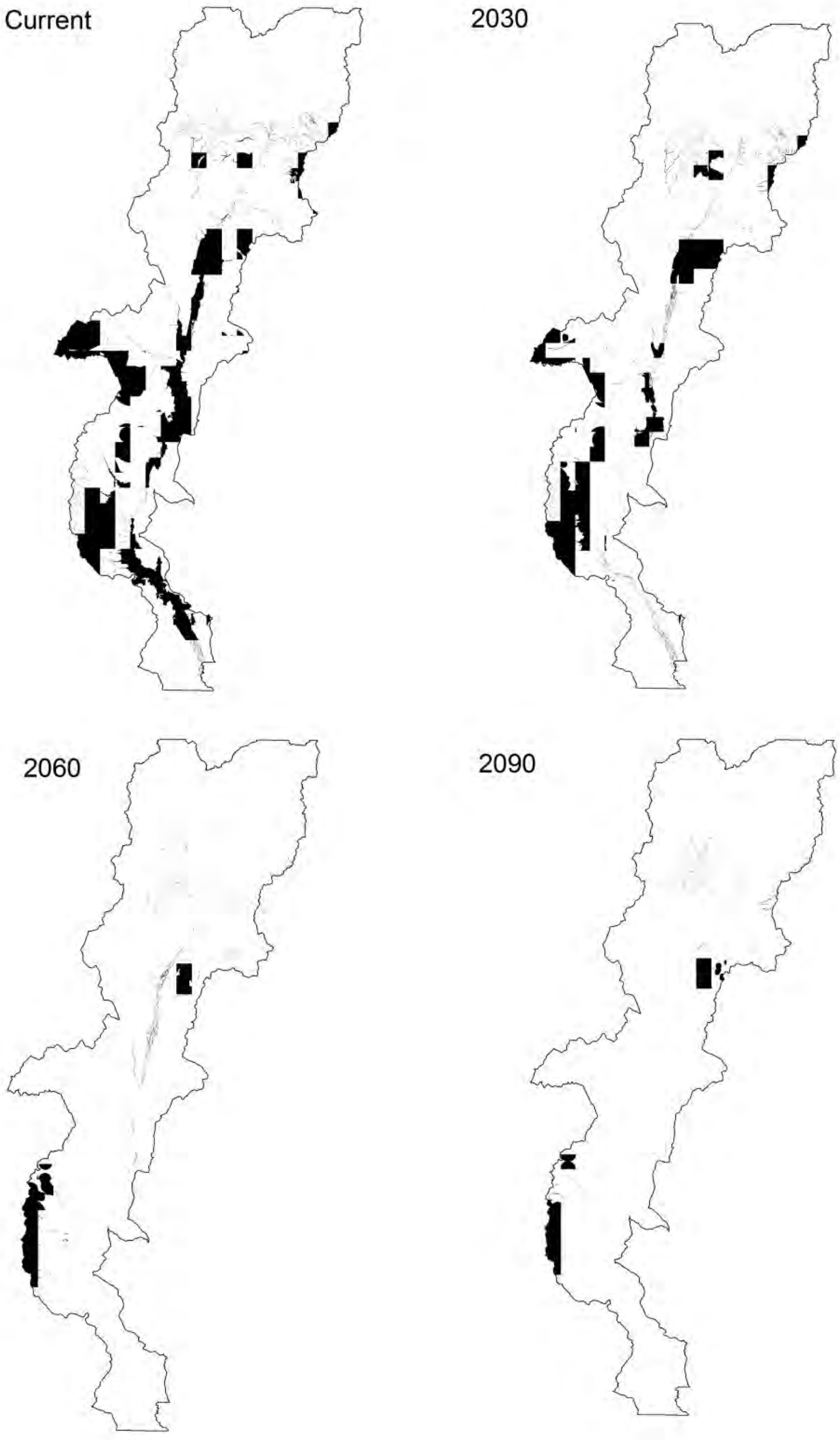


Figure 3. Two thirds consensus models of suitable habitat for *Myotis yumanensis* for three future time periods. Habitat was modeled under three climate models: Had_CM3, CGCM3.1, and GFDL_CM2.1. Black areas indicate predicted suitable habitat estimated by at least two models.

Yuma Bat (*Myotis yumanensis*)

Fire Impacts

Myotis yumanensis is likely to benefit from most fire activity except where roosts are impacted. Bats are able to escape fire mortality and are drawn to the insects within newly burned habitats (Malison and Baxter 2010; Lacki et al., 2009). In California, bats benefited from the clearing of vegetation, which improved foraging success, and increased insect abundance in early successional habitat (Bhucalski et al., 2013). Fire can also increase roosting habitat for species that rely on snags or tree cavities (Lacki et al., 2009). The long term (>5 years) consequences of fire are probably positive. Over large periods of time, fire contributes to habitat heterogeneity associated with improved conditions for these species (Bhucalski et al., 2013). Fire suppression is considered one of the causes leading to reduced habitat for bat species in the SW (Chung-MacCoubrey 2005). Buhcalski et al., (2013) note that *M. yumanensis* in particular may prefer early successional insects and had increased activity levels in riparian habitats in California.

Negative impacts from fire can occur when burns happen in the early to mid-spring especially around roosting maternity sites (Lacki et al., 2009). At best, fire would represent a short term disturbance to colonies and at worst fire could lead to the loss of roost site and mortality of non-volant young. In addition, large fires that reduce habitat heterogeneity will lead to a decline in habitat quality. To account for the potential negative impacts of fire on roost sites, we classified areas expected to experience crown fires as high risk. Though the presence of torching does not necessarily predict habitat loss, the indication for torching is used to qualify the potential risk for bat species. Here, torching fires are associated with a greater risk than non-torching fires resulting in higher risk to roost sites and less benefit for foraging sites (Figures 4-6).

.

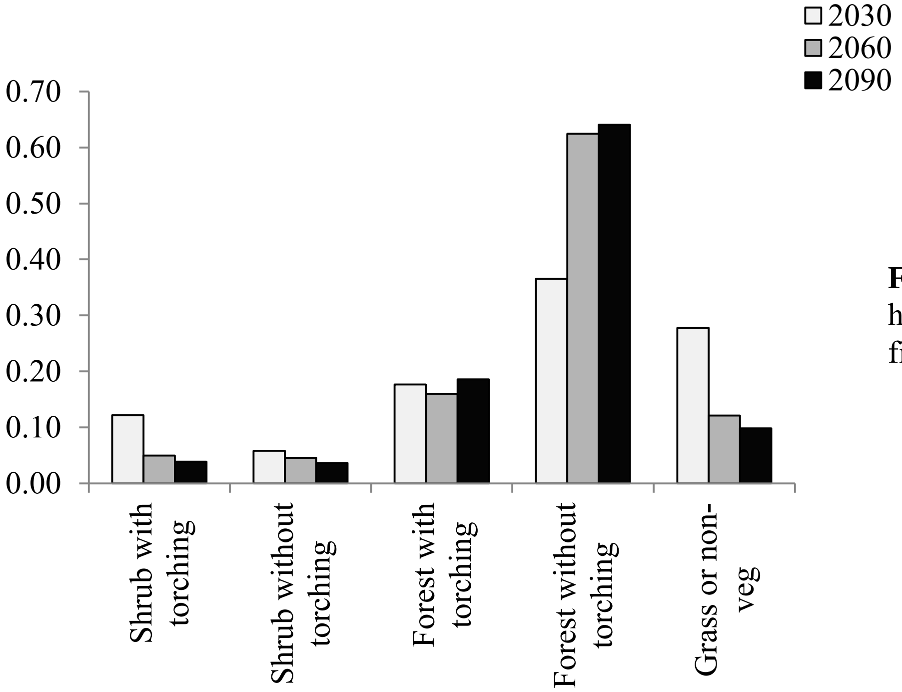
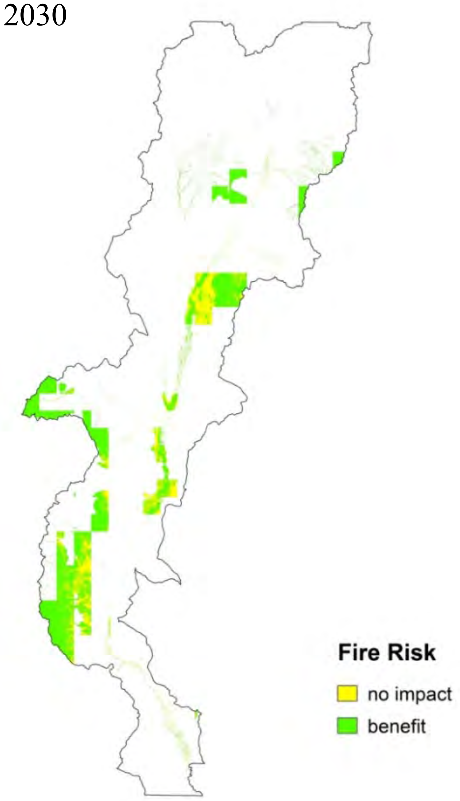
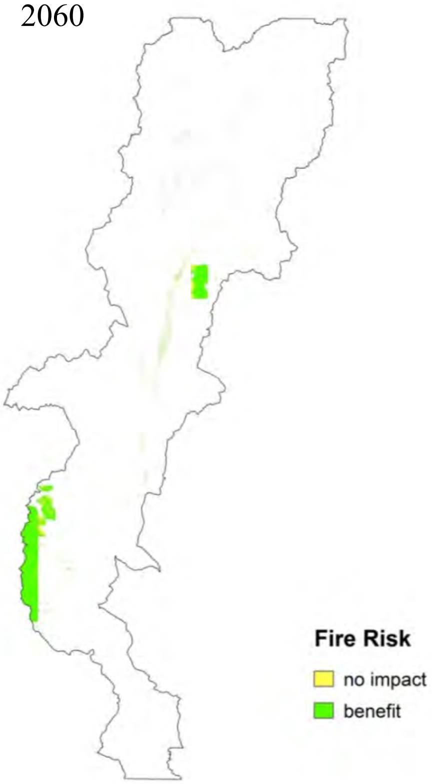


Figure 4. Percent of habitat falling within each fire type category.

2030

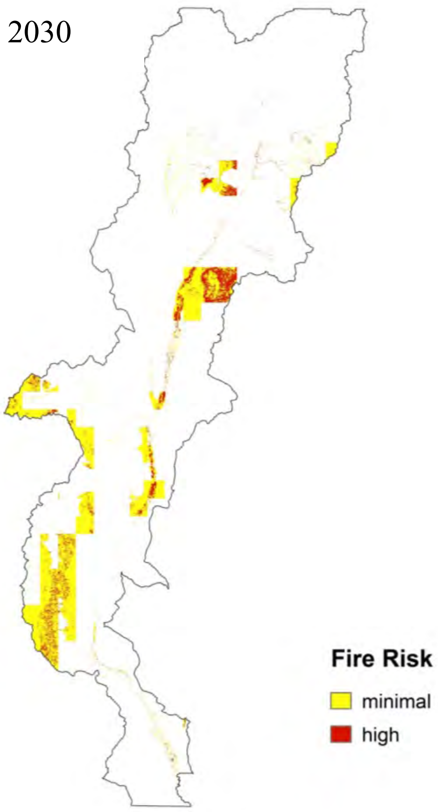


2060



2090

Figure 5. Fire impacts for *Myotis yumanensis* foraging habitat under three climate futures. Fire intensity was calculated using FSim (Finney et al., 2011) for each time period based on the GFDL-ESM-2m GCM under the RCP85 scenario. Fire risk maps were created by overlaying estimated species' habitat, predicted fire characteristic (shrub with or without torching, forest with or without torching or grass/nonveg) and species' response (negative, neutral, positive).


2030


2060

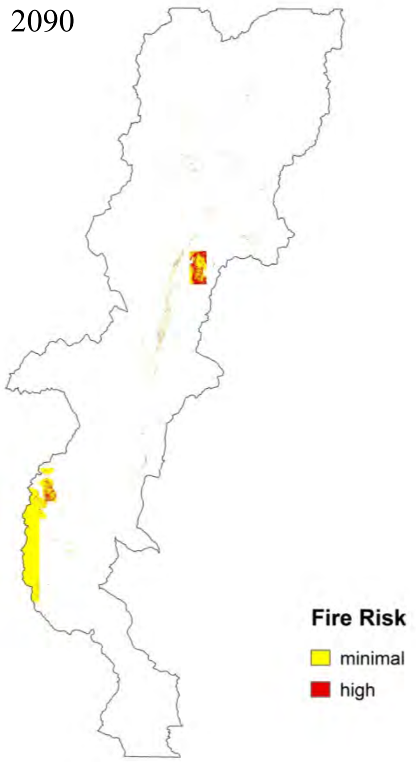

2090

Figure 6. Fire impacts for *Myotis yumanensis* roost habitat under three climate futures. Fire intensity was calculated using FSim (Finney et al., 2011) for each time period based on the GFDL-ESM-2m GCM under the RCP85 scenario. Fire risk maps were created by overlaying estimated species' habitat, predicted fire characteristic (shrub with or without torching, forest with or without torching or grass/nonveg) and species' response (negative, neutral, positive).

References:

1. 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.
2. Bhucalski, M.R., Fontaine, J.B., Heady III, P.A., Hayes, J. P. and W.F. Frick. 2013. Bat response to differing fire severity in mixed-conifer forest California, USA. PLOS One 8: e57884.doi:10.1371/journal.pone.0057884
3. 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. http://www.bison-m.org
4. Carpenter, R.E. 1969. Structure and function of the kidney and the water balance of desert bats. Physiological Zoology 42: 288-301.
5. Chung-MacCoubrey, A.L. 2005. Use of pinyon-juniper woodlands by bats in New Mexico. Forest Ecology and Management 204: 209-220.
6. Evelyn, M.J., D.A. Stiles, 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.
7. Finney, M.A., C.W. McHugh, I.C. Grenfell, K.L. Riley, and K.C. Short. 2011. A Simulation of Probabilistic Wildfire Risk Components for the Continental United States. Stochastic Environmental Research and Risk Assessment 25: 973-1000.
8. Geluso, K. 2007. Winter activity of bats over water and along flyways in New Mexico. The Southwestern Naturalist 52: 482-492.
9. Grindal, S.D., J.L. Morissette, and R.M. Brigham. 1999. Concentration of bat activity in riparian habitats over an elevational gradient. Canadian Journal of Zoology 77: 972-977.
10. Kays, R.W., and D.E. Wilson. 2009. Princeton Field Guides: Mammals of North America, Second edition. Princeton
11. University Press, Princeton, NJ.Lacki, M. J., J.S. Johnson, E.D. Dodd, and M. D. Baker. 2007. Prey consumption of insectivorous bats in coniferous forest of North-Central Idaho. Northwest Science 81: 199-205.
12. 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.
13. Malison, R.L. and C.V. Baxter. 2010. Effects of wildfire of varying severity on benthic stream insect assemblages and emergence. Journal of North American Benthological Society 29: 647-656.
14. 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.
15. NatureServe. 2009. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia. Available: http://www.natureserve.org/explorer. [2009, February 18].
16. Scheel, D., T.L.S. Vincent, and G.N. Cameron. 1996. Global warming and the species richness of bats in TX. Conservation Biology 10: 452-464.
17. Sims, K. 2000. "*Myotis yumanensis*" (On-line), Animal Diversity Web. Accessed: http://animaldiversity.ummz.umich.edu/site/accounts/information/Myotis_yumanensis.html. [2009, February 18]

18. Twente, J. W., J. Twente, and V. Brack. 1985. The duration of the period of hibernation of three species of vespertilionid bats. II. Laboratory studies. Canadian Journal of Zoology 63: 2955-2961.

New Mexico Meadow Jumping Mouse (*Zapus hudsonius luteus*)

Overview

Predicted Impacts	
Habitat Change	
2030	12-15% Loss
2060	8-41% Loss
2090	10-42% Loss
Adaptive capacity	Low
Fire Response	Negative

Status:

Meadow jumping mice (*Zapus hudsonius)* contain a number of subspecies of which, *Z. h. luteus* inhabits New Mexico and the Jemez Mountains. Recent molecular evidence supports the current taxonomic status of subspecies and suggests that *Z. h. luteus* may be among the most distinct of the western subspecies (King et al., 2006). This species has been considered endangered in New Mexico since 1983 and is a candidate for Federal T&E listing. This species has experienced 82% reduction in populations due to habitat loss. *Zapus h. luteus* is currently considered endangered by New Mexico and threatened by Arizona State Heritage Groups and is listed as Globally Rare or Critically Endangered within each of the states it exists in by the IUCN Red List of Threatened Species (http://http://www.iucnredlist.org).

Range and Habitat:

This species is thought to have been more widespread in the past during more mesic climate conditions. The current distribution of *Z. h. luteus* represents relic populations that were able to persist in suitable habitats in northern latitudes and higher elevations (Morrison 1992). Common hibernation sites include logs or underground areas with nests made from leaves or grass (Whitaker 1972).

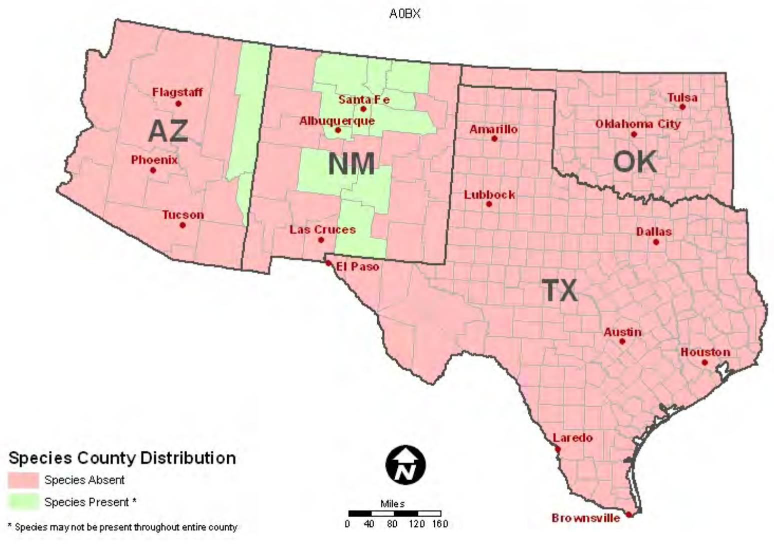


Figure 1. Range of *Zapus hudsonius luteus* in the Southwest. Produced by FWS.

New Mexico Meadow Jumping Mouse (*Zapus hudsonius luteus*)

Climate Change Impacts and Adaptive Capacity

Adaptive capacity = 1.55 (Low)

The New Mexico Meadow Mouse does not exhibit adaptive characteristics for dealing with habitat change (Table 1). This species has a fairly tight requirement for moist habitats and has little dispersal ability. *Zapus h. luteus* is associated with permanent waterways that contain dense vegetation. This species appears to be particularly dependent on riparian habitat, perhaps as a result of living in relatively more arid habitats than other *Zapus* (Frey and Malaney 2009). Frey and Malaney (2009) found *Z. h. luteus* most commonly in ungrazed, emergent, herbacious riparian habitats dominated by tall, dense stands of sedges on saturated soils. Others have noted that *Zapu*s make their nests on dry soil. Vegetation is probably more important than soil moisture to this species. *Zapus h. luteus* has also been reported from human-made mesic habitats such as irrigated waterways in the absence of the natural riparian habitat. Still, a recent survey found mice restricted to small areas of well-developed riparian habitat along isolated stream reaches (Frey and Malaney 2009). In addition, these authors noted that *Z. luteus* was further restricted to grazing exclosures at one site in the Sacramento Mountains (Frey and Malaney 2009).

Physiology Limits:
There is little information for the meadow jumping mouse regarding potential physiological adaptations for dealing with changing conditions. Exposure to lethal temperatures or dry conditions is determined by habitat availability, which is already considered in the niche model analysis. However, there are two indications for increased meadow jumping mouse mortality: 1. Increased winter flooding could kill hibernating mouse populations (Frey 2005) and, 2. Extended drought could result in greater fragmentation of suitable habitat as certain riparian zones become too dry for the mouse. Because future conditions likely include increased drought and habitat fragmentation appears to be a very important driver of *Z. h. luteus* population declines, we consider this species vulnerable to climate conditions. 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).

The meadow jumping mouse does not present any long term mechanisms for dealing with resource variation and is assumed to be short-lived as is typical for a mammal of its size. While these mice are described as profound hibernators and remain in hibernation longer than many other mammals, underweight or small animals often do not survive hibernation (Whitaker 1972). The great majority of fat accumulation occurs in the weeks just prior to hibernation. We do not consider hibernation an advantage under warmer climates (See Friggens et al., 2013).

Phenology:
The New Mexico meadow jumping mouse appears to be quite vulnerable to potential timing mismatches resulting from climate change. Hibernation is cued by day length (Muchlinksi 1980) and emergence is related to soil temperature (Muchlinski, 1988). The emergence of jumping mice and subsequent reproductive activities are tied to general requirement for forage.
Day length cues create issues when temperature change results in different seasonal cycles of plant growth and reproduction. Temperature is also a problematic cue for emergence as there is

risk of early emergence and increased death due to frosts. Though little information is available regarding reproduction in *Z. h. luteus* it is thought that females have at least two litters per year with each litter averaging 5 offspring. Gestation takes about 20 days and young have full coats and open eyes at 4 weeks (Whitaker, 1972).

Biotic Interactions:

New Mexico meadow jumping mice are primarily herbivores and show a preference for grass seeds (Quimby 1951). They also eat fruits of plants and will consume insects, especially upon emergence from hibernation (Quimby 1951). *Zapus hudsonius* in general (Whitaker 1972) have been reported as prey items of hawks and owls, weasels, rattlesnakes and even frogs. It is likely they are an important food source for many species. The New Mexico meadow jumping mouse is positively associated with the presence of beavers and is harmed from beaver removal (Frey 2006). Beavers are predicted to be slightly vulnerable to future climate changes, particularly as they pertain to water flows (Friggens et al., 2013). *Zapus h. luteus* has been reported as a host for a variety of parasites including fleas, ticks, botflies and intestinal helminthes though none appear to be particularly important to the health of these animals.

Table 1. Scoring sheet for New Mexico meadow jumping mouse (*Zapus hudsonius luteus).* "v" indicates a vulnerability or sensitivity, "r" indicates a resiliency or increased adaptive capacity, "n" indicates neutral response.

Question	Score	Notes
1. Changes to habitat component	n	None predicted
2. Changes to habitat quality	n	Not predicted
3. Dispersal ability (Site fidelity or other limitations)	v	Limited
4. Reliance on migratory or transitional habitats	n	No
5. Increase or decrease in physiological range limitation	n	No
6. Sex Ratios determined by temperature or food changes	n	No
7. Response to predicted extreme weather events/disturbances	v	Drought
8. Changes to daily activity period	n	No
9. Variable life history traits or coping strategies	n	Some
10. Ability to outlive limiting conditions	v	No
11. Migrates/ hibernates in response to weather cues	v	Yes
12. Reliance on weather mediated resource (e.g. insect emergence)	n	No
13. Spatial or temporal separation between critical resources and life history stages	v	Yes
14. Can adjust timing of critical activities	r	Yes

Question	Score	Notes
15. Likelihood for decreased food resource	n	Unknown, not predicted
16. Likelihood of increase predation	n	Not predicted
17. Loss of important symbiotic species	v	Yes
18. Increase in high mortality/morbidity disease	n	No
19. Increased competitive pressures	n	No

New Mexico Meadow Jumping Mouse (*Zapus hudsonius luteus*)

Niche Model Analysis

The model AUC values were 0.96 for all three models (Table 2). Total area of habitat does not declines, with the most severe (42% decline) seen under the dryer GFDL_CM2.1 scenarios. Shifts in habitat appear likely to increase fragmentation and isolation of remaining mouse populations.

Table 2. Percent of original predicted habitat under three climate models.

Time	CGCM3.1	GFDL_CM2.1	Had_CM3	Average
current	1	1	1	1
2030	0.85	0.86	0.87	0.86
2060	0.92	0.69	0.84	0.81
2090	0.80	0.68	0.90	0.79

Physiographic variables were important predictors for *Z. h. luteus* habitat (Table 3). Distance to water was the single most important variable for all three models. Mice were associated with riparian areas within the Rocky Mountain Montane Conifer Forest, Chihuahuan Desert and Plains Grassland Biomes. Slope was less important under the Had CM3 model. Mice were negatively associated with Annual Potential Evaporation of Natural Vegetation for areas with values less than 52. Suitability increased with increasing precipitation JJA at high values and decreased with increasing precipitation at very low values. Suitability increased with increasing winter precipitation (DJF) at low values but was not related in areas with higher winter precipitation. These patterns agree with the tendency to find this species in humid habitats within a large array of habitat types including low elevation sites (Frey and Malaney 2009)

Table 3. Variables in order of importance for species habitat models. Relationship between variable and predicted suitability is listed as positive (+) negative (-) for linear and exponential curves or quadratic for parabola-like curves.

Variable	CGCM3.1	GFDL_CM2.1	Had_CM3	Relationship
Distance to water	61.2	59.8	61.4	-
Biome	17.5	15.6	17.0	+ RMM ConFor, ChiDes, PlainsGssd,
Annual Pot Evap Nat Veg	11.0	12.1	12.5	Quadratic
Slope	5.6	7.0	3.6	Peaks at high values
Elevation	3.0	3.3	3.9	Peaks at low values
Precipitation JJA (bio16)	1.5	1.3	1.4	- at low values
Precip DJF (bio17)	0.2	1.0	0.1	-

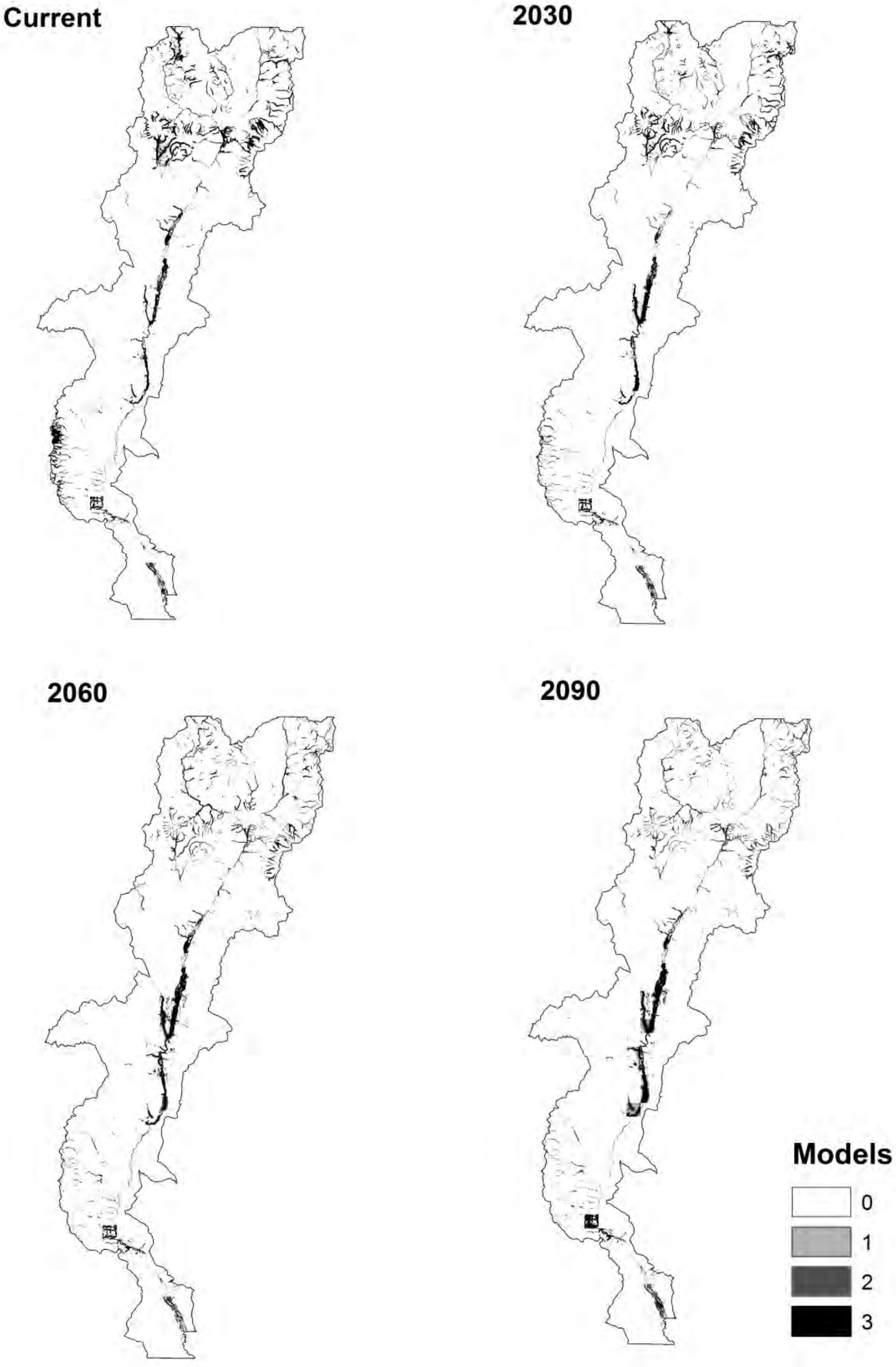


Figure 2. Distribution of suitable habitat for Zapus *hudsonius luteus* for three future time periods. Future habitat predictions are based on three climate models: Had_CM3, CGCM3.1, and GFDL_CM2.1. Shading indicates the number of models that predict suitable habitat for a given area.

New Mexico Meadow Jumping Mouse (*Zapus hudsonius luteus*)

Fire Analysis

Small mammal mortality from fire can be directly caused by burns, heat stress, asphyxiation, physiological stress, trampling by other animals, or predation while fleeing fire (Sullivan 1995). Indirect causes of mortality can include changes in quality and quantity of food, nest site availability, predation, parasitism, disease, increased competition, and social interactions (Sullivan 1995). Direct mortality from a fire is generally considered rare because it is assumed species are able to seek cover (Pilliod et al., 2006). However, fire during breeding season could lead to increased mortalities for shrub or ground nesting species. According to Pilliod et al., 2006, characteristics of species that might benefit immediately post fire include those associated with open canopies and open forest floors and those that eat insects (more long term response e.g. small mammals).Characteristics of species with negative impact include species associated with downed wood, snags, dwarf mistletoe, dense forests and closed canopies and small mammals that prefer shrub cover to avoid predators (short term).

Wildfire and especially post wildlife flooding are identified as especially problematic for the endangered meadow jumping mouse, Zapus luteus (FWS 2006). From the federal register:
"Following fires, we found that, depending on fire intensity and the subsequent ash and debris flow within stream reaches, New Mexico meadow jumping mouse populations can be significantly affected and likely extirpated, even when 15 km (9 mi) of continuous suitable habitat existed prior to the fire (Sugarite Canyon; Frey 2006; Malaney et al., 2012). Therefore, we estimate that stream lengths should be at least two to three times of those characterized by Frey (2011, p. 29) in order to have adequate population sizes necessary to persist through these types of stochastic and catastrophic events."

Given these conclusions and the potentially large negative impacts for fires within the riparian habitats upon which this species relies, we rated this species as sensitive to all shrub and forest fires.

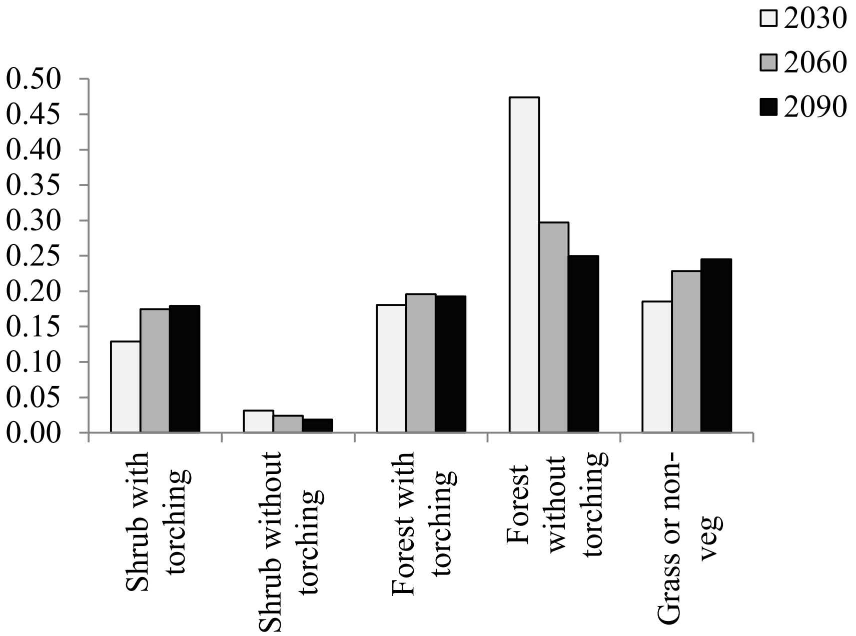
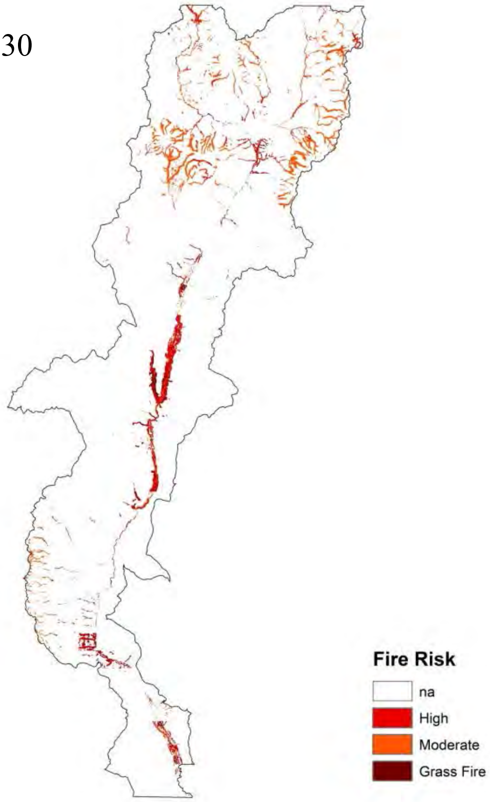
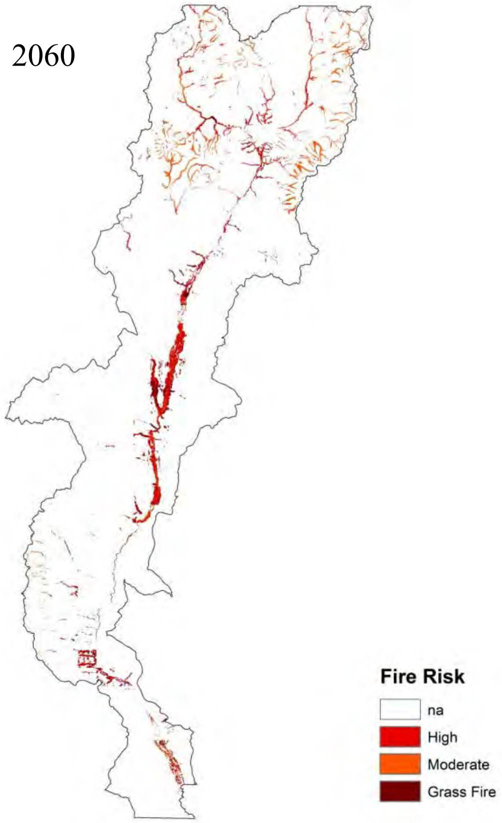


Figure 3. Percent of habitat falling within each fire type category.

2030



2060



2090

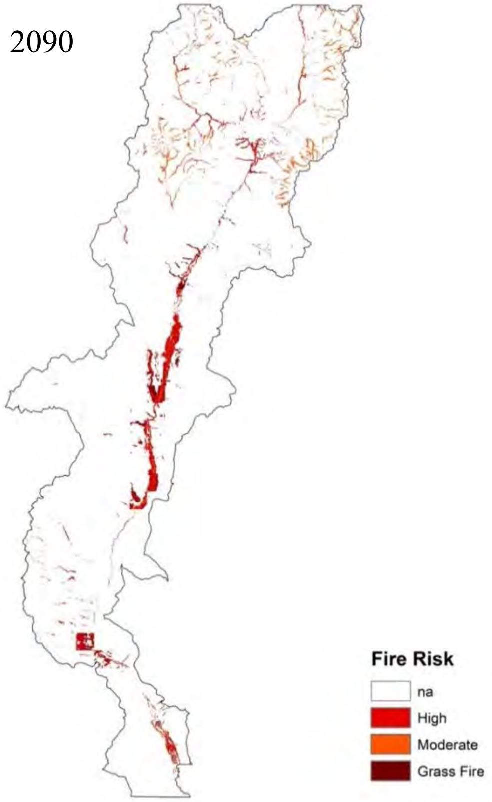


Figure 4. Fire impacts for *Zapus luteus* habitat under three climate futures. Fire intensity was calculated using FSim (Finney et al., 2011) for each time period based on the GFDL-ESM-2m GCM under the RCP85 scenario. Fire risk maps were created by overlaying estimated species' habitat, predicted fire characteristic (shrub with or without torching, forest with or without torching or grass/nonveg) and species' response (negative, neutral, positive).

References

1. Finney, M.A., C.W. McHugh, I.C. Grenfell, K.L. Riley, and K.C. Short. 2011. A Simulation of Probabilistic Wildfire Risk Components for the Continental United States. Stochastic Environmental Research and Risk Assessment 25: 973-1000.
2. Frey, J. K. 2005. Status assessment of montane populations of the New Mexico meadow jumping mouse (*Zapus hudsonius luteus*) in New Mexico. Report to New Mexico Department of Game and Fish, Santa Fe, New Mexico. Contract # 05-516.57. 74 pp. + appendices.
3. 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. Final report to New Mexico Dept. Game & Fish, Conservation Services Division, Santa Fe, contract 06-516.0000.0049. 78 pp. + CD.
4. Frey, J. K. and J. L. Malaney. 2009. Decline of the meadow jumping mouse (*Zapus hudsonius luteus*) in two mountain ranges in New Mexico. Southwest Naturalist 54: 31-44.
5. Friggens, M.M., Finch, D. M, et al. 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
6. 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.
7. Malaney, J. L., Frey, J. K., and J.A. Cook. 2012. The biogeographic legacy of an imperiled taxon provides a foundation for assessing lineage diversification, demography and conservation genetics. Diversity and Distributions 18: 689-703.
8. Morrison, J.L. 1992. Persistence of the Meadow Jumping Mouse, *Zapus hudsonius luteus,* in New Mexico. The Southwestern Naturalist 37: 308-311.
9. Muchlinski, A.E. 1980. The effects of day length and temperature on the hibernating rhythm of the meadow jumping mouse (*Zapus hudsonius*). Physiological Zoology 53: 410-418.
10. Muchlinski, A.E. 1988. Population Attributes Related to the Life-History Strategy of Hibernating *Zapus hudsonius* Journal of Mammalogy. 69: 860-865.
    Whitaker, J. O., Jr. 1972. *Zapus hudsonius*. Mammalian Species 11: 1-7.

Hispid Cotton Rat (*Sigmodon hispidus*)

Overview

Predicted Impacts	
Habitat Change	
2030	-7 to +84%
2060	68 to 487% Increase
2090	229 to 474% Increase
Adaptive capacity	Intermediate to low
Fire Response	Negative

Status:
Sigmodon hispidus is classified as "a species of least concern" by the IUCN (http://www.iucnredlist.org).

Range and Habitat:
Sigmodon hispidus inhabits dense grass habitats along the south and southeastern U.S. (Figure 1). It has been recently reported as expanding northward (including northern New Mexico) and westward where it might exclude pygmy mice and prairie voles. The hispid cotton rat is associated with croplands, hedgerows, desert, grasslands, herbaceous fields, and savanna (Natureserve 2008). It favors dense vegetation, particularly grasses, and is also found in cattails along streams and in desert scrub habitats (Natureserve 2008). Hispid cotton rats are strongly associated with grassy patches that have some shrub overstory and they have little or no affinity for dicot-dominated patches (Sullivan 1995).

Habitat selectivity differs between sexes, according to breeding status and season. Male hispid cotton rats are less choosey than females. In the southern part of their ranges, males selected habitats (grassy, shrubby, and mixed) approximately in proportion to availability, whereas female hispid cotton rats chose mixed habitats more often than expected based on availability (Sullivan 1995). Also in the southern extent of their range, rats were found to select grasslands with few shrubs during spring, and areas with more shrubby cover in fall (Sullivan 1995). Cotton rats use burrows or fallen logs/debris for nests (Natureserve 2008).

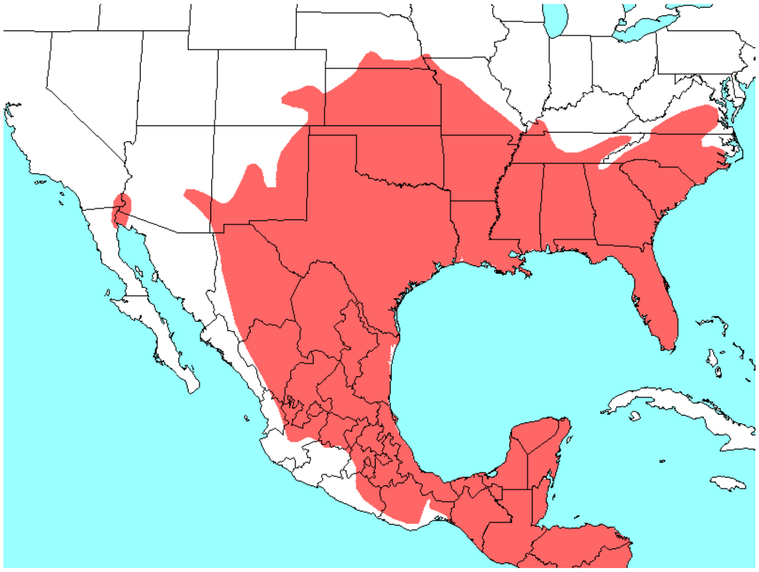


Figure 1. Distribution of *Sigmodon hispidus* (from NatureServe)

Hispid Cotton Rat (*Sigmodon hispidus*)

Climate Change Impacts and Adaptive Capacity

Adaptive Capacity = 0.87 (Intermediate to low)

The hispid cotton rat does not appear to well adapted for dealing with habitat change (Table 1). Reproduction in the hispid cotton rat is tied to habitat quality, which in turn is tied to precipitation, particularly in marginal habitat (Davis and Schmidley 1997). Dense vegetation appears to be an important habitat characteristic for this species, probably as cover for protection from predators. This species was considered dispersal limited in previous assessments (Friggens et al., 2013). Hispid cotton rat populations are largely sedentary and population density is primarily driven by reproductive effort as opposed to immigration (Sullivan 1995). In many parts of its range, local extinctions are frequent (Sullivan 1995). Maximum dispersal distance was found to be 100m and greatest distance of homing capability was 1,500m.

Physiology Limits:

Sigmodon hispidus does not exhibit physiological adaptations that might improve it resiliency to physiological stresses (Table 1). In New Mexico, hispid cotton rats are not found in areas with a mean annual temperature lower than 55° F (13° C) and a growing season shorter than 180 days (Sullivan 1995). Severe reductions in hispid cotton rat populations have been associated with sharp declines in vegetative biomass and cover and following severe weather such as drought and, in the northernmost parts of its range, extremely cold winters (Sullivan 1995). Reduced populations have also been noted during drought periods in Texas (Davis and Schmidley 1997).

Sigmodon do not cache food and are active year round (Natureserve 2008; Davis and Schmidley 1997) and are fairly short-lived. One study in a Kansas remnant prairie recaptured very few hispid cotton rats after an 8 month period (Sullivan 1995). Collectively, information on the cotton rat indicates they are moderately vulnerable to population declines resulting from prolonged limiting conditions. Populations are prone to extreme population fluctuations (Davis and Schmidley 1997).

Phenology:

The Hispid cotton rat does exhibit flexibility in the timing of life history events that represents increased adaptive capacity (Table 1). Reproductive effort in the cotton rat is tied to resource availability such that years with greater resources lead to a large number of litters. Therefore, though an opportunistic breeder, there is a relationship between fitness and discrete resource pulses. However, it is not clear how resource pulses might change given the diverse diet of this species, nor is there information regarding how this might affect fitness. It appears that this species is able to respond immediately to changes in resources, which enables it to maximize reproductive output under good conditions. Additionally, cotton rats breed throughout the year and do not differentiate special breeding habitat (Natureserve 2008). They are prolific breeders that can produce several litters of 2-10 altricial young per year (Davis and Schmidley 1997).

Biotic Interactions: Hispid cotton rats have generalized interactions with food and other species. Cottons rats are omnivorous, feeding primarily on green vegetation though they also consume insects and other small animals (Sullivan 1995). Hispid cotton rats are preyed on by many birds and reptiles, and by other mammals (Sullivan 1995) but none appear to have an overt influence on populations (Friggens et al., 2013).

Hispid cotton rats are a reservoir for some human diseases including rabies, Chagas disease, and Venezuelan equine encephalitis (Sullivan 1995), though it is unclear whether these diseases negatively affect the rats themselves. Marsh rice rats (*Oryzomys palustris*) replaced cotton rats in one area where water additions caused the water table to rise (Sullivan 1995). Studies comparing competition between *S. hispidus* and *S. fulviventer* do not concur and it is unclear which species is competitively dominant (see discussion in Friggens et al., 2013).

Table 1. Scoring sheet for the hispid cotton rat (*Sigmodon hispidus).*

Question	Score	Notes
1. Changes to habitat component	n	None predicted
2. Changes to habitat quality	v	Less productive, less dense vegetation
3. Dispersal ability (Site fidelity or other limitations)	v	Not great, extirpations noted
4. Reliance on migratory or transitional habitats	n	No
5. Increase or decrease in physiological range limitation	n	No
6. Sex ratios determined by temperature or food changes	n	No
7. Change in limiting threshold	v	Tied to temp, extirpations due to drought
8. Changes to daily activity period	n	No
9. Variable life history traits or coping strategies	v	No
10. Ability to outlive limiting conditions	v	No
11. Migrates/ hibernates in response to weather cues	n	No
12. Reliance on weather mediated resource (e.g. insect emergence)	n	No
13. Spatial or temporal separation between critical resources and life history stages	r	Responds to immediate environment
14. Can adjust timing of critical activities	r	Yes, reproduces quickly
15. Likelihood for decreased food resource	n	Not predicted
16. Likelihood of increase predation	n	Not predicted
17. Loss of important symbiotic species	n	Not predicted
18. Increase in high mortality/morbidity disease	n	Not predicted

Question	Score	Notes
19. Increased competitive pressures	n	Not predicted

Hispid Cotton Rat (*Sigmodon hispidus*)

Niche Model Analysis

Model AUC values were 0.91, 0.89, and 0.92 for HadCM3, GDFLCM2, and CGCM3, respectively. All models predict and increase in the area of suitable habitat (Table 2, Figures 2, 3). In 2030, increases in suitable habitat represent an expansion of available habitat. However, by years 2060 and 2090, there is little overlap in current and future areas of suitable habitat. This lack of overlap between current and future habitat predictions is a source of concern and may effectively represent a total loss of habitat if cotton rats are unable to shift their distributions due to dispersal limitations or non-modeled habitat issues.

Table 2. Percent of original predicted habitat under three climate models.

Time	CGCM3.1	GFDL CM2.1	Had_CM3	Average
current	1	1	1	1
2030	4.19	1.84	0.93	2.32
2060	4.67	3.15	1.68	3.17
2090	4.74	2.29	3.78	3.61

The cotton rat was strongly associated with wetlands present in Rocky Mountain Alpine Conifer Forest Biome and had a slightly positive association with habitats in the Great Basin Conifer Woodland and SemiDesert Grassland Biomes (Table 3). The cotton rat was negatively associated with habitats in the Chihuahuan Desert Biome. Highly suitable sites were characterized with warmer summer temperatures, greater summer precipitation, and lower winter precipitation. Cotton rat habitat was found in areas with intermediate annual temperature values. Cotton rat habitat is more likely to exist below 2000m, and declines quickly at higher elevations. Suitability increased with Isothermality where values where below 36 or above 38.

Table 3. Variables in order of importance for species habitat models. . Relationship between variable and predicted suitability is listed as positive (+) negative (-) for linear and exponential curves or quadratic for parabola-like curves.

Variable	CGCM3.1	GFDL CM2.1	Had_CM3	Relationship
Biome	39.0	37.0	40.0	+ RMAConFor, GBConWd, SemiDesGssd, -ChiDes
Max Temp JJA (bio5)	20.3	21.0	13.4	+
Mean Diurnal Temp Range (bio2)	13.9	14.4	13.8	+
Elevation	10.2	11.0	15.3	- (after 2000m)
Mean Temp (bio11)	9.8	11.1	11.8	Quadratic
Precip DJF (bio14)	2.8	1.6	2.3	-

Variable	CGCM3.1	GFDL CM2.1	Had_CM3	Relationship
Precip JJA (bio18)	2.1	2.1	2.1	+
Temp Annual Range (bio7)	1.5	0.6	0.6	Peaks at low and high values
Isothermality (bio3)	0.5	1.1	0.8	Peaks at intermediate values

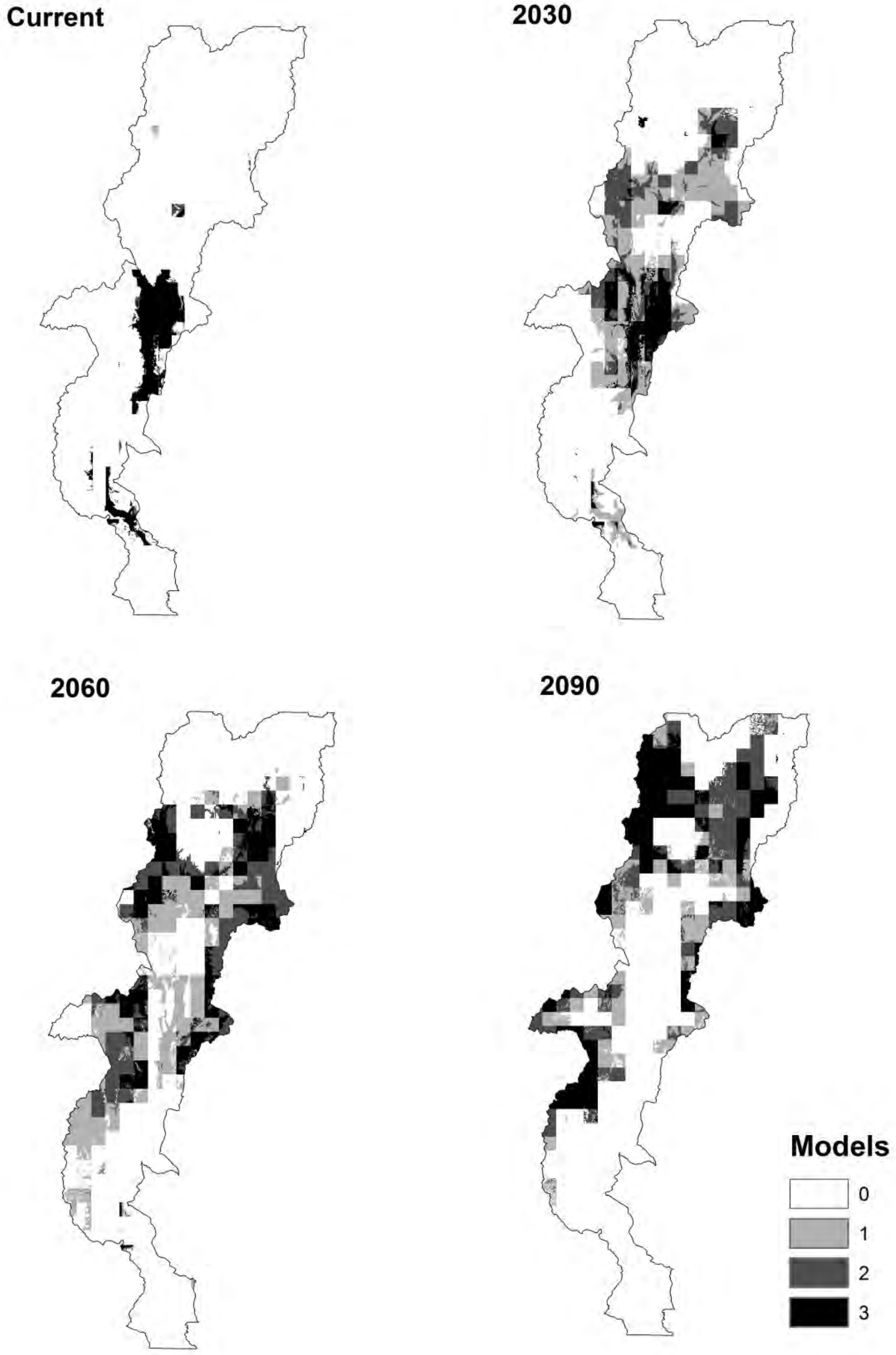


Figure 2. Distribution of suitable habitat for *Sigmodon hispidus* for three future time periods. Future habitat predictions are based on three climate models: Had_CM3, CGCM3.1, and GFDL_CM2.1. Shading indicates the number of models that predict suitable habitat for a given area.

Hispid Cotton Rat (*Sigmodon hispidus*)

Fire Impacts

Small mammal mortality from fire can be directly caused by burns, heat stress, asphyxiation, physiological stress, trampling by other animals, or predation while fleeing fire (Sullivan 1995). Indirect causes of mortality can include changes in quality and quantity of food, nest site availability, predation, parasitism, disease, increased competition, and social interactions (Sullivan 1995). Direct mortality from a fire is generally considered rare because it is assumed species are able to seek cover (Pilliod et al., 2006). However, fire during breeding season could lead to increased mortalities for shrub or ground nesting species. According to Pilliod et al., (2006), characteristics of species that might benefit immediately post fire include those associated with open canopies and open forest floors and those that eat insects (more long term response e.g. small mammals).Characteristics of species with negative impact include species associated with downed wood, snags, dwarf mistletoe, dense forests and closed canopies and small mammals that prefer shrub cover to avoid predators (short term).

Hispid cotton rats are known to experience mortality from fires though the great majority are able to escape injury by retreating into burrows or unburned areas. In general, hispid cotton rats are thought to respond negatively to fire in the short term, but recover rapidly once vegetation cover has been reestablished. Kauffman et al., (1990) rated hispid cotton rats as fire-negative, since they are associated with plant debris and are foliage eaters. However, they are often reported as increasing after fire with the highest populations found in the first few growing seasons after fire (Sullivan 1995 and references therein). Fire appears to have a positive effect in southwestern forests when they occur every 3-4 years. More and less frequent fires are associated with smaller cotton rat populations (Sullivan 1995 and references therein). There are no studies of the hispid cotton rat in New Mexico or southwestern forests. Given their strong preference for thick vegetation and the relative scarcity of such habitat outside of the riparian forests in which they exist, we rate this species as sensitive to all shrub and forest fires.

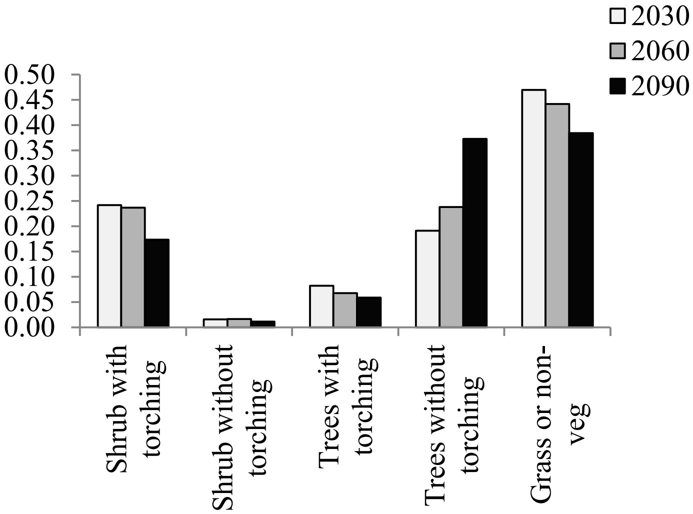
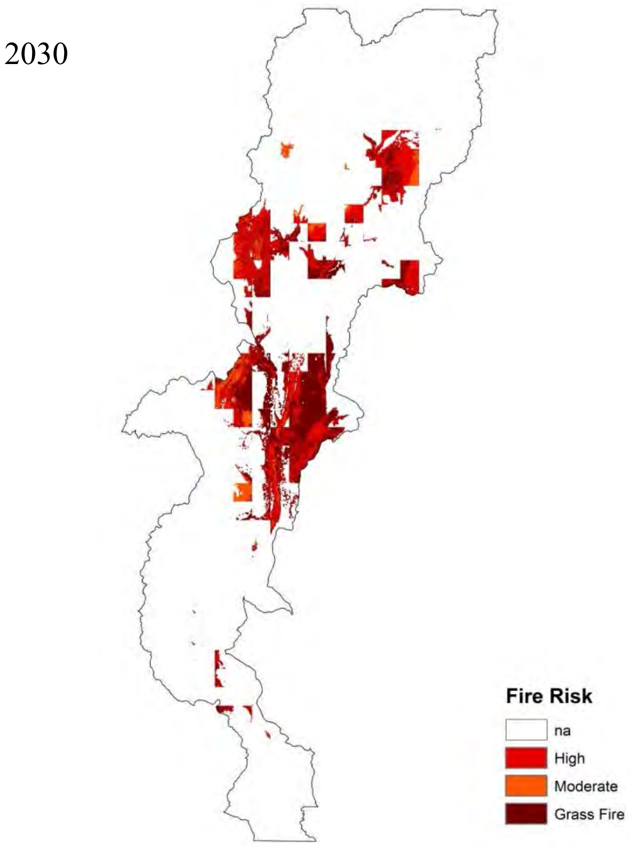
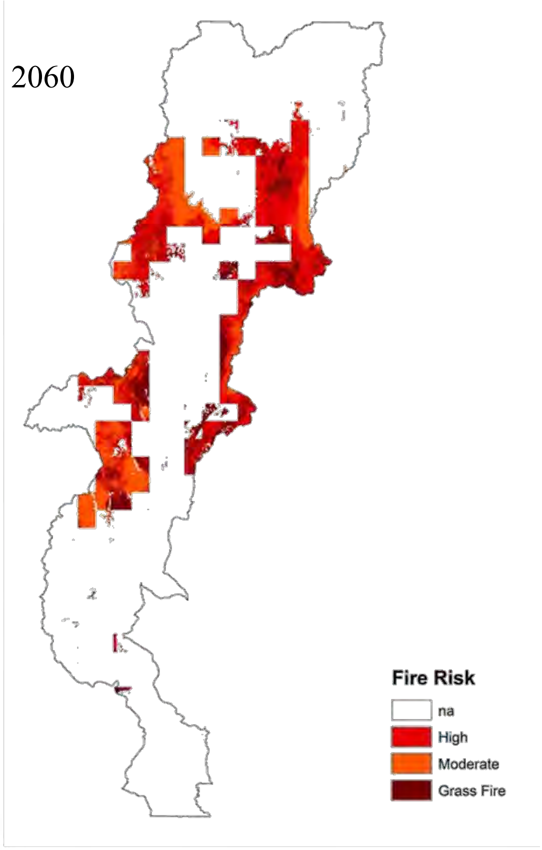


Figure 3. Percent of habitat falling within each fire type category.





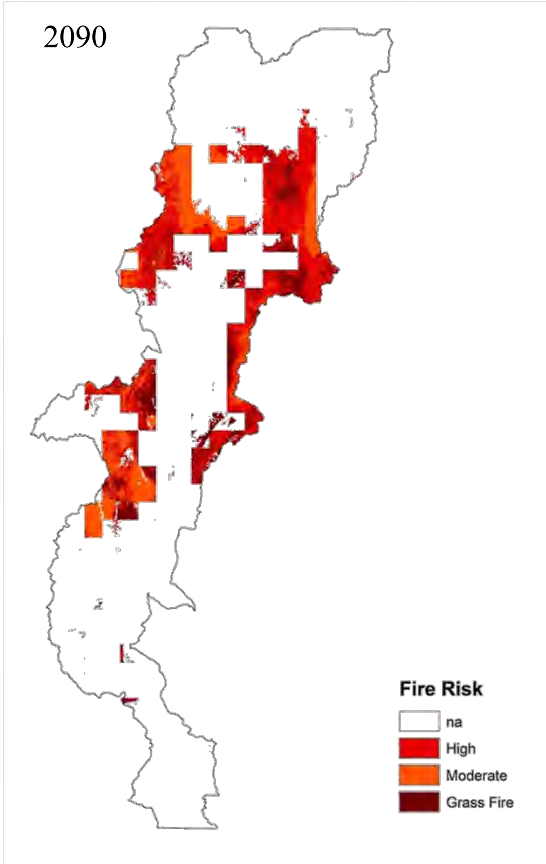


Figure 4. Fire impacts for *Sigmodon hispidus* habitat under three climate futures. Fire intensity was calculated using FSim (Finney et al., 2011) for each time period based on the GFDL-ESM-2m GCM under the RCP85 scenario. Fire risk maps were created by overlaying estimated species' habitat, predicted fire characteristic (shrub with or without torching, forest with or without torching or grass/nonveg) and species' response (negative, neutral, positive).

References:

1. 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.
2. Cameron, G. N. and S.R. Spence. 1981. *Sigmodon hispidus*. Mammalian Species 158: 1-9.
3. Davis, W.B., and D. J. Schmidley. 1997. *The Mammals of Texas Online Edition*. Texas Tech University. Available: http://www.nsrl.ttu.edu/tmot1/Default.htm
4. Finney, M.A., C.W. McHugh, I.C. Grenfell, K.L. Riley, and K.C. Short. 2011. A Simulation of Probabilistic Wildfire Risk Components for the Continental United States. Stochastic Environmental Research and Risk Assessment 25: 973-1000.
5. Friggens, M.M., D. M. Finch, K.E Bagne, S.J. Coe, and D.L. Hawksworth. 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.
6. Kaufman, D.W., E. J. Finck, and G. A. Kaufman. 1990. Small mammals and grassland fires. In: Collins, Scott L.; Wallace, Linda L., eds. *Fire in North American tallgrass prairies*. Norman, OK: University of Oklahoma Press: 46-80. [14195]
7. Natureserve. 2008. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.0 NatureServe, Arlington, Virginia. Available at: http://www.natureserve.org/explorer.
8. Pilliod, D.S., E. Bull, J.L. Hayes, and B.C. Wales. 2006. Wildfire and invertebrate response to fuel reduction treatments in dry coniferous forests of the Western United States: a synthesis. Gen. Tech. Rep. RMRS-GTR-173. 34p.
9. Sullivan, J. 1995. *Sigmodon hispidus*. 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/ [2008, December 30].

Range Distribution Credit: Data provided by NatureServe in collaboration with Bruce Patterson, Wes Sechrest, Marcelo Tognelli, Gerardo Ceballos, The Nature Conservancy — Migratory Bird Program, Conservation International — CABS, World Wildlife Fund — US, and Environment Canada — WILDSPACE.

Other References: Other References: Animal Diversity Website (http://animaldiversity.org.) and Mammals of Texas online, both referenced 12-26-08 to 12-29-08.

Black-necked Gartersnake (*Thamnophis cyrtopsis*)

Overview

Predicted Impacts	
Habitat Change	
2030	23 to 50% Loss
2060	66 to 81% Loss
2090	71 to 88% Loss
Adaptive capacity	Low
Fire Response	Mixed

Status:
The black-necked gartersnake (*Thamnophis cyrtopsis*, Kennicott, 1860) is considered a species of least concern by the IUCN Red List (http://www.iucnredlist.org) and as S5 (Demonstrably secure) by New Mexico Natural Heritage in 1991. Within Mexico, *T. cyrtopsis* was listed as Threatened in 1995. This species was known as *T. dorsalis* in the 1960's. There are also a variety of other older names (e.g. animaldiversity.org notes *T. eques* as an important one).

Range and Habitat:
Thamnophis cyrtopsis is found in forest, rangeland and wetland (including streams and channels) habitat types (BISON-M 2009). They are usually found in association with water and cottonwood, scrub oak, and pinyon-pine woodlands in New Mexico. AnimalDiversity (http://animaldiversity.org/) lists desert, grassland/herbaceous, shrubland/chaparral, and woodlands among its habitat associations. It also states that this species is commonly found in the vicinity of streams or similar water sites and is largely restricted to these areas in the Southwestern United States (BISON-M 2009). This gartersnake is reported as uncommon in wetlands of the Bosque del Apache Wildlife Refuge (BISON-M 2009) though they may have been common along the Middle Rio Grande 20 years ago (Hink and Ohmart 1984).

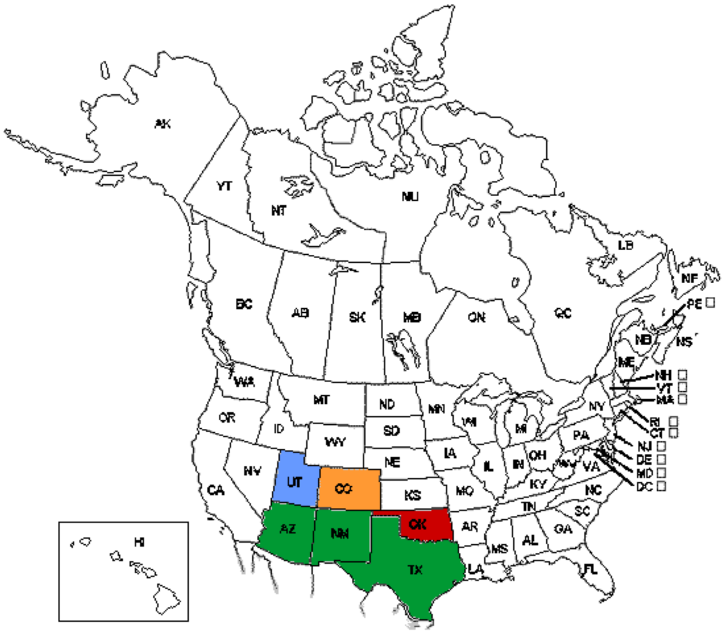


Figure 1. Range of *T. cyrtopsis* in the southwestern U.S. From NatureServe.

Black-necked Gartersnake (*Thamnophis cyrtopsis*)

Climate Change Impacts and Adaptive Capacity

Adaptive Capacity Score = 1.22 (low)

Overall, the black-necked gartersnake does not appear to have traits indicative of increased or decreased adaptive capacity to habitat changes (Table 1). Little information is available about the black-necked gartersnake and we relied on information on closely related species where relevant for the following assessment. *Thamnophis cyrtopsis* tends to be found in areas with water and benefits from riparian restoration and presence of water tanks, which points to its reliance on water. Potential loss of riparian habitat due to increase evapotranspiration, declining snowpack and increase variation in precipitation events are likely to negatively impact this species.

We did not include a prediction for increased or decreased resilience due to habitat quality changes (Table 1). Females of other *Thamnophis* spp. tend to have less variability in clutch size and timing 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). This difference could reflect an adaptive response or could reflect the influence of variable food sources on clutches (i.e. spring clutch characteristics are determined by previous years (Seigel and Ford 2001). *Thamnophis cyrtopsis* may be dependent upon riparian areas when in the more xeric Southwestern U.S. Aquatic elements of habitat are important especially as they relate to increased fish diversity. The black-necked gartersnake depends on fish and leopard frogs. Terrestrial habitat components include vegetation and rocks used primarily for cover. *Thamnophis cyrtopsis* needs to be able to burrow (or retreat) far enough below ground that it escapes freezing in the winter. It utilizes underground burrows/dens which are unlikely to be affected by climate change and does not rely on specific habitat components. We did consider this species to have good dispersal ability. Some species of gartersnake are reported to travel several km to hibernaculum. AnimalDiversity (NatureServe) notes that colubrids can move between areas of up to a few kilometers apart, and medium sized colubrids (such as gartersnakes) are easily able to move 1-2km. In addition, the black-necked gartersnake has an advantage in that it does not require additional migration sites.

Physiology:

There are a number of indications that this species will be negatively affected by higher temperatures and reduced precipitation (Table 1). In New Mexico, *T. cyrtopsis* maintains its body temperatures at 22-32°C when air temperatures are 16-35°C (BISON-M 2009 and references therein). Experiments on other species of gartersnakes (*Thamnophis radix, Natrix sipedon, Elaphe o. obsoleta* and others) indicate that very few snakes can survive temperatures over 40°C (104°F) (Lueth 1941). Snake metabolism increases with increasing temperature such that the length of life of starved snakes is highly dependent on temperature (Leuth 1941). Other effects of increasing temperature include increasing metabolic demands, shortened hibernation periods and accelerated onset of reproduction (Seigel et al., 2000 and see Friggens et al., 2013). This species' tendency to inhabit sites with water probably buffers it from some of the negative effects of high temperature. Still, reproduction in *T. marcianus* is reported as being extremely reduced following two drought years, presumably due the effects of drought on prey base (Ford and Karges 1987) and this species was considered at risk under prolonged drought conditions. From a vulnerability assessment of *T. marcianus* (Friggens et al., 2013): Average lifespan is reported as 2 years though sexual maturity and max size are not reached until 3 or 4 years (i.e. most die before breeding) (Animal diversity website).

However, the black-necked gartersnake is not prone to temperature dependent sex ratios and is unlikely to experience a change in activity. *Thamnophis cyrtopsis* is diurnal though sometimes active at night as well. Seasonally, it is active from April through October. The seasonal active period may increase with warmer temperatures. More variable precipitation patterns could negatively impact seasonal activity and higher summer temperatures are likely to reduce daily activity given physiological limitation (see Huey and Tewskbury 2009). However, *T. cyrtopsis* appears to be able to hunt during both day and night and it is assumed that this behavior is driven by needs related to hunting or foraging rather than temperature (see discussions in Bagne et al., 2011; Friggens et al., 2013). As such, higher temperatures may limit this species to more night activity, but will not necessary represent a loss of hunting time. Further, seasonal active period may increase with warmer temperatures offsetting somewhat negative impacts from very high temperatures. Finally, though this species hibernates, hibernation is not an optional life history event. Hibernation is initiated as cold weather limits activity. The capacity to hibernate is not considered to add to a species' adaptive capacity under warming conditions.

T. marcianus are able to undergo atresia and reabsorption of vitelline follicles, which ultimately would help snakes adjust clutch size just prior to ovulation (Ford and Karges 1987). These allow the animal to match reproductive effort with resource availability. However, it is not clear that such a strategy leads to a net increase in reproductive success and it is not considered a beneficial adaptation. There are no other known beneficial behaviors.

Phenology:

T. marcianus may be prone to negative impacts from changes in phenology. *T. marcianus* hibernation is initiated by cold weather (BISON-M 2009; Jacob and Painter, 1980). In temperate zones, offspring must be born early enough to be able to grow enough to survive hibernation (onset of cold weather), but females need to ensure that they do not emerge from hibernation too early when risk of lethal cold temperatures may still be possible (Seigel et al., 2000). Timing of precipitation events may also impact this species. From a vulnerability assessment of *T. marcianus* (Friggens et al., 2013): "Vitellogenesis appears to be tied to rainy periods (in arid environments) and/or when prey base is good. Both movement of snakes and prey availability (amphibians) are strongly tied to precipitation (Seigel and Ford 2001). Snakes have been found relatively far from water sources during wet periods (Seigel et al., 2000) which may indicate an important mechanism for dispersal."

Birth is presumably timed to abundance peaks in resources though survival of young is not necessarily limited to peak prey abundance (i.e. some will survive even if born on either side of the peak). Emergence from hibernation cues to temperature, which may or may not reflect rise in prey abundance. *Thamnophis cyrtopsis* are viviparous and produce 6-25 young. This species is thought to mate in the fall and young have been reported born in mid-August. They reproduce once/year.

Biotic Interactions:
The black-necked gartersnake (*Thamnophis cyrtopsis*) probably eats mainly adult and larval amphibians. There are several indications that gartersnake food resources are likely to decline (FWS 2013; Friggens et al.2013). Common gartersnakes (from Animal diversity website) are often predated by large fish, bull frogs, other snakes, squirrels, shrews, crows, hawks, raccoons, shrews and foxes (Seigel and Ford 2001). No specific information was available regarding the black-necked gartersnake. There were no reported symbiotic relationships for this species. One article notes that snakes maintained at higher temperatures were prone to eye and skin infections though it does not specify the agent (Leuth 1941).

Competition from the American Bullfrog may be an issue for the gartersnake. From a vulnerability assessment of *T. marcianus* (Friggens et al., 2013): "Introduced bullfrogs and sunfish are named as important competitors, which not only eat small snakes but the food that the snakes eat (interestingly, bullfrog tadpoles are often a primary prey item for the snakes-Rosen, 1991) (from the Sonoran Desert Museum website)". Bullfrogs are expected to benefit somewhat under future climate scenarios. However, for this assessment we consider *T. marcianus* less prone to bullfrog issues as it appears to have broader habitat associations than that seen for the common gartersnake.

Table 1. Score sheet for *Thamnophis cyrtopsis*. "v" indicates a vulnerability or sensitivity, "r" indicates a resiliency or increased adaptive capacity, "n" indicates neutral response.

Question	Score	Notes
1. Changes to non-modeled habitat components	**n**	None known
2. Change in habitat quality	**n**	None predicted
3. Dispersal ability (Site fidelity or other limitations)	**r**	Good
4. Reliance on migratory or transitional habitats	**n**	No
5. Increase or decrease in physiological range limitation	**n**	Not likely
6. Sex Ratios determined by temperature or food changes	**n**	No
7. Response to predicted extreme weather events/disturbances	**v**	Drought sensitive
8. Changes to daily activity period	**n**	Probably not
9. Variable life history traits or coping strategies	**n**	None identified
10. Ability to outlive limiting conditions	**v**	Short lived
11. Migrates/hibernates in response to weather cues	**v**	Yes

Question	Score	Notes
12. Reliance on weather mediated resource (e.g. insect emergence)	**n**	No
13. Spatial or temporal separation between critical resources and life history stages	**n**	No
14. Can adjust timing of critical activities	**v**	No
15. Likelihood for decreased food resource	**v**	Yes
16. Likelihood of increase predation	**n**	Variable, not predictable
17. Loss of important symbiotic species	**n**	None
18. Increase in high mortality/morbidity disease	**n**	No
19. Increased competitive pressures	**n**	Perhaps

Black-necked Gartersnake (*Thamnophis cyrtopsis*)

Niche Model Analysis

Model AUC values were 0.85 across all three models. All models agreed that suitable habitat declines for *T. cyrtopsis* within the study area but models differ on the extent and rate of decline (Table 2).

Table 2. Percent of original predicted habitat under three climate models.

	CGCM3.1	GFDL_CM2	Had_CM3	Average
current	1	1	1	1
2030	0.50	0.74	0.87	0.70
2060	0.19	0.20	0.44	0.28
2090	0.12	0.19	0.29	0.20

Areas with higher minimum January temperatures were most suitable for the gartersnake (Table 3). The black-necked gartersnake also appears to avoid habitats with the highest maximum July temperatures. Suitability declined with distance from water and increased with increasing mean annual runoff. Suitability was associated with low elevation sites. The most suitable sites had low annual soil moisture values, with a sharply negative relationship between annual soil moisture content and suitability at higher values. While site suitability increased in areas with either low or high Isothermality, there was a generally negative association with increasing annual range of temperatures.

Table 3. Variables in order of importance for species habitat models. Relationship between variable and predicted suitability is listed as positive (+) negative (-) for linear and exponential curves or quadratic for parabola-like curves.

	CGCM3.1	GFDL_CM2	Had_CM3	Relationship
Min Temp January (bio6)	41.8	43.1	41.8	Peaks at high values
Distance to Water	25.5	24.8	25.1	-
Annual Runoff (mm)	9.2	8.8	8.8	+
Isothermality (bio3)	5.0	8.0	6.6	Peaks at low and high values
Max Temp July (bio5)	7.1	5.5	6.2	- (after 32.6°C)
Elevation	6.5	5.8	6.2	-
Ann Temp Range (bio7)	3.6	2.6	4.2	-
Annual Mean Soil Moisture Content	1.5	1.7	1.35	Peaks at low values

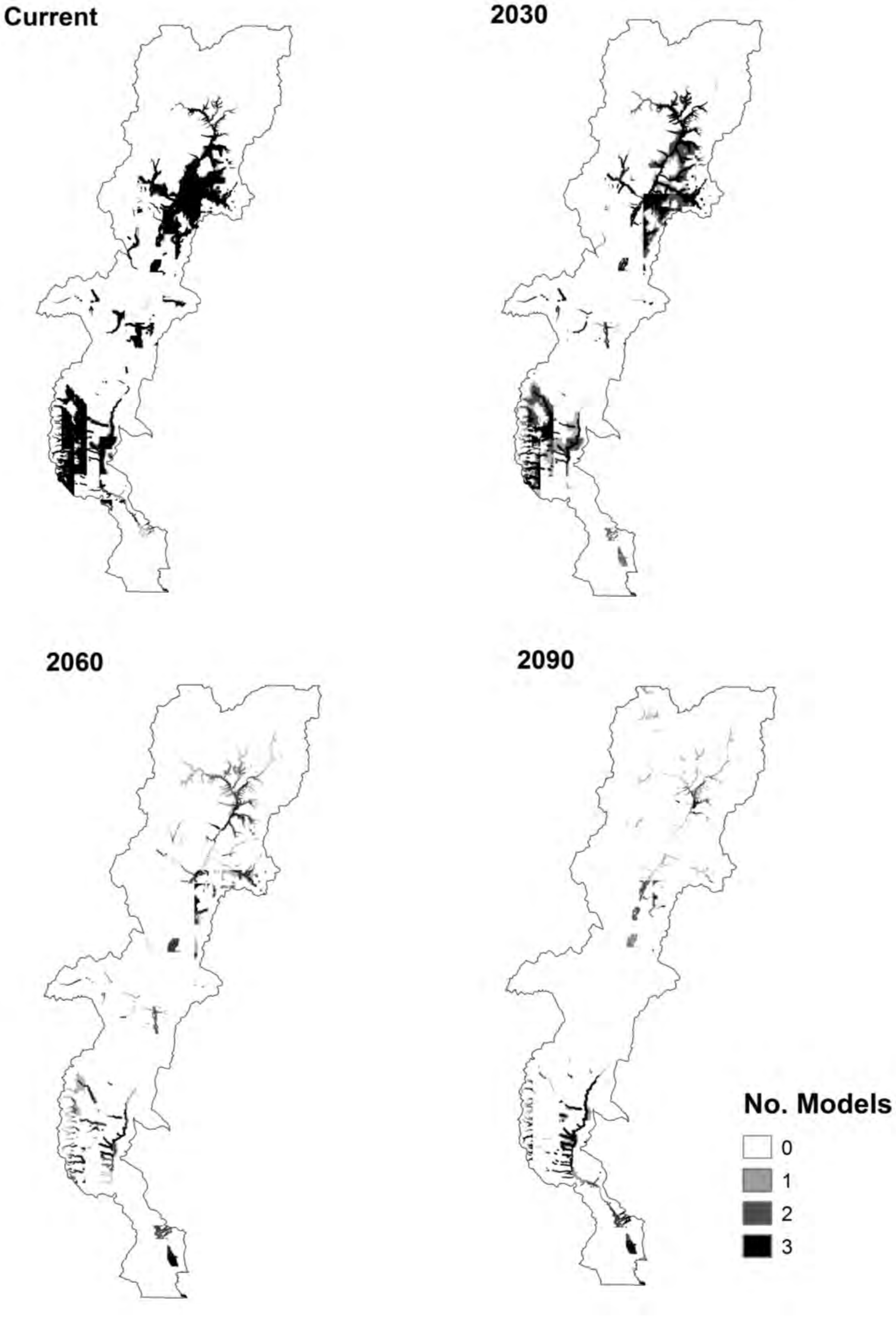


Figure 2. Distribution of suitable habitat for *Thamnophis cyrtopsis* for three future time periods. Future habitat predictions are based on three climate models: Had_CM3, CGCM3.1, and GFDL_CM2.1. Shading indicates the number of models that predict suitable habitat for a given area.

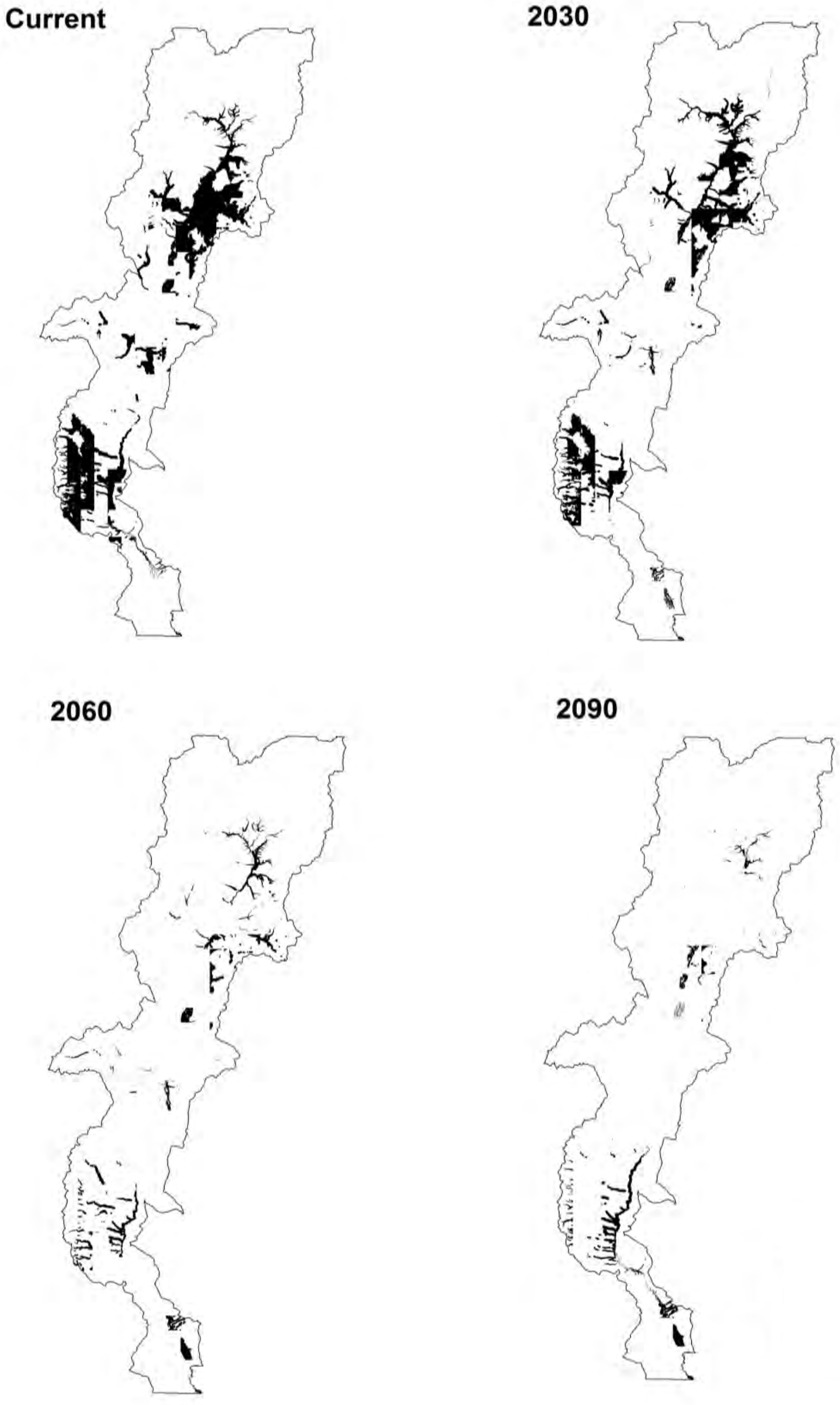


Figure 3. Two-thirds consensus models for suitable habitat for *Thamnophis cyrtopsis* for three future time periods. Future habitat is predicted where at least two of three climate models, Had_CM3, CGCM3.1, and GFDL_CM2.1, identify suitable habitat.

Black-necked Gartersnake (*Thamnophis cyrtopsis*)

Fire Analysis

According to Pilliod et al., (2006), characteristics of species that might benefit immediately post fire include those associated with open canopies and open forest floors and those that eat insects. Characteristics of species with negative impact include species associated with downed wood, snags, dwarf mistletoe, dense forests and closed canopies and small mammals that prefer shrub cover to avoid predators.

Low-severity burns within riparian habitats can be beneficial by removing ground cover, releasing nutrients to remaining vegetation, and encouraging new growth. Riparian vegetation may be more resilient to wildfire, given that water is present and willows are known to be positively affected by low-severity burns that do not damage root crowns (Coleman 2011). However, high-severity burns are likely to reduce basking sites and a loss of cover could increase the risk of predation. Over longer time periods high severity fires may lead to post-fire ash flows, flooding that can result in significant declines of resident fish and amphibians, a primary source of prey for gartersnake populations (Coleman 2011; FWS 2013). While information on the black-necked gartersnake is limited, data is available for similar species, in particular the Mexican gartersnake.

From the federal register (FWS 2013) for the Mexican gartersnake:

> “Effects to northern Mexican and narrow-headed gartersnake habitat from wildfire should be considered in light of effects to the structural habitat and effects to the prey base. Post-fire effects vary with burn severity, percent of area burned within each severity category, and the intensity and duration of precipitation events that follow (Coleman 2011, p. 4).”

Further, excess sedimentation and turbidity may reduce hunting success:

> “The presence of adequate interstitial spaces along stream floors may be particularly important for narrow-headed gartersnakes. Hibbitts *et al.*(2009, p. 464) reported the precipitous decline of narrow-headed gartersnakes in a formerly robust population in the San Francisco River at San Francisco Hot Springs from 1996 to 2004. The exact cause for this decline is uncertain, but the investigators suspected that a reduction in interstitial spaces along the stream floor from an apparent conglomerate, cementation process may have affected the narrow-headed gartersnake’ s ability to successfully anchor themselves to the stream bottom when seeking refuge or foraging for fish (Hibbitts *et al.*2009, p. 464). These circumstances would likely result in low predation success and eventually starvation. Other areas where sedimentation has affected either northern Mexican or narrow-headed gartersnake habitat are Cibecue Creek in Arizona, and the San Francisco River and South Fork Negrito Creek in New Mexico…The San Francisco River in Arizona.”

We ranked the black-headed gartersnake as at risk of mortality or negative habitat impact under conditions where forest or shrub habitats are expected to have torching fires. Non torching fires were not considered as problematic.

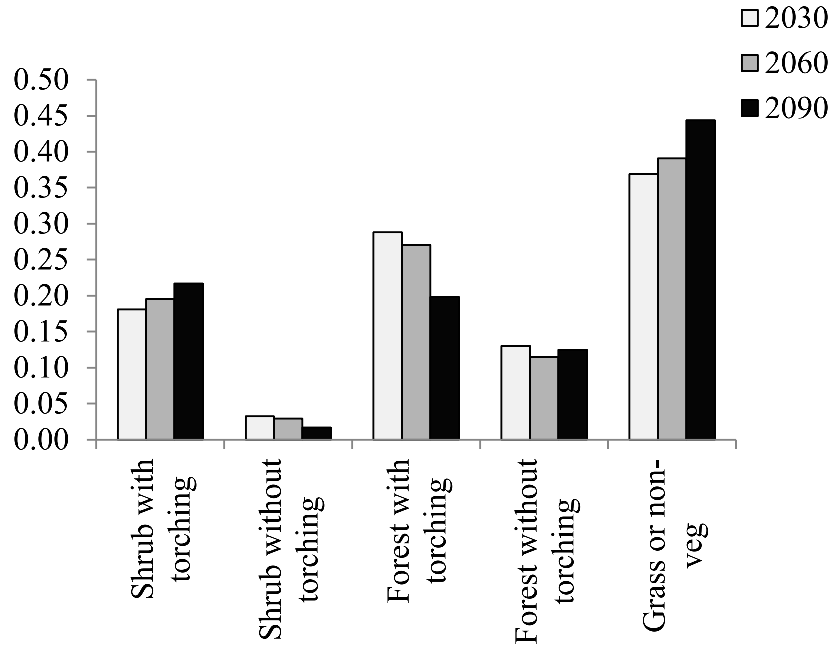
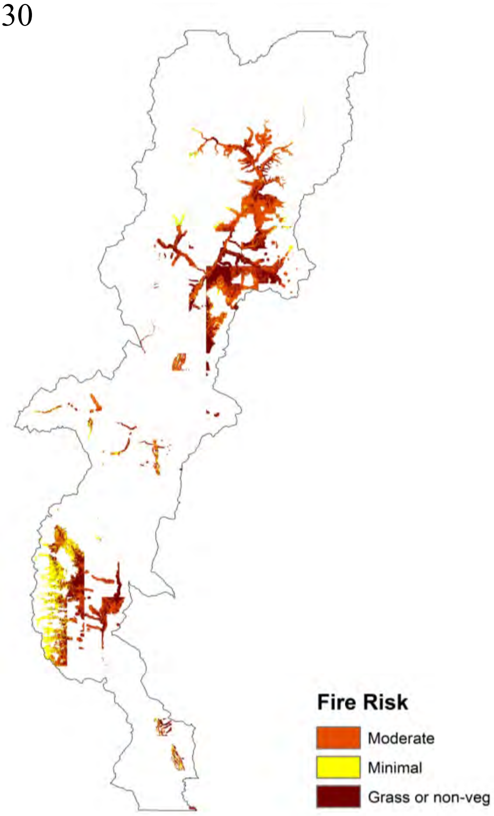
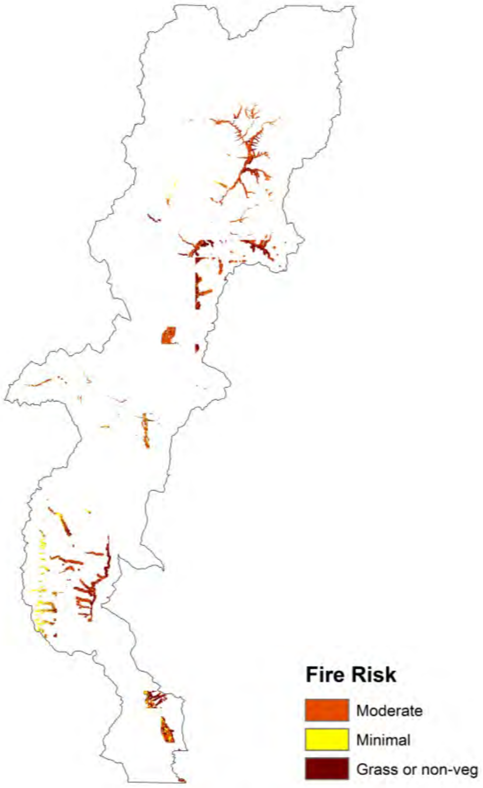


Figure 4. Percent of habitat falling within each fire type category.

2030



2060



2090

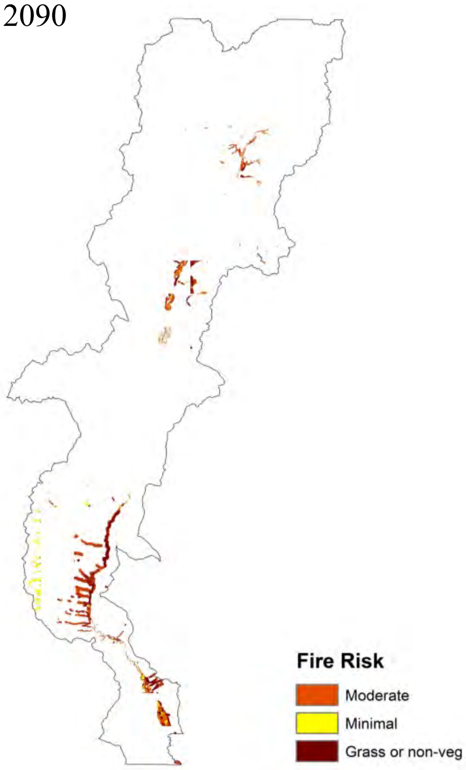


Figure 5. Fire impacts for *T. cyrtopsis* habitat under three climate futures. Fire intensity was calculated using FSim (Finney et al.2011) for each time period based on the GFDL-ESM-2m GCM under the RCP85 scenario. Fire risk maps were created by overlaying estimated species' habitat, predicted fire characteristic (shrub with or without torching, forest with or without torching or grass/nonveg) and species' response (negative, neutral, positive).

References:

1. Animal Diversity Web (online). Accessed at http://animaldiversity.org.
2. 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. 28 p. USDA, Forest Service, Rocky Mountain Research Station, Fort Collins, CO.
3. 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. http://www.bison-m.org
4. Coleman, S. 2011. *Thamnophis rufipunctatus* and *Thamnophis eques* information post-Wallow Fire. White paper drafted by Stephanie Colemen, Apache-Sitgreaves National Forest. 4 pp.
5. Finney, M.A., C.W. McHugh, I.C. Grenfell, K.L. Riley, and K.C. Short. 2011. A Simulation of Probabilistic Wildfire Risk Components for the Continental United States. Stochastic Environmental Research and Risk Assessment 25: 973-1000.
6. Ford, N.B., and J. P. Karges. 1987. Reproduction in the checkered gartersnake, *Thamnophis marcianu*s, from southern Texas and northeastern Mexico: seasonality and evidence for multiple clutches. The Southwestern Naturalist 32: 93-101.
7. Friggens, M.M., Finch, D. M, et al. 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
8. Hink, V.C. and R.D. Ohmart.1984. Middle Rio Grande Biological Survey. Submitted to the U.S. Army Corps of Engineers, Albuquerque District. Contract # DACW47-81-C-0015.
9. 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.
10. Huey, R. B. and J. J. Tewksbury. 2009. Can behavior douse the fire of climate warming? PNAS 106: 3647-3648.
11. Jacob, J.S. and C.W. Painter. 1980. Overwinter thermal ecology of *Crotalus viridis* in the north-central plains of New Mexico. Copeia 1980: 799–805.
12. Leuth, F. X. 1941 Effects of temperature on snakes. Copeia 3: 125-132.
13. 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).
14. Pilliod, D.S., Bull, E., Hayes, J.L., Wales, B.C. 2006. Wildfire and invertebrate response to fuel reduction treatments in dry coniferous forests of the Western United States: a synthesis. Gen. Tech. Rep. RMRS-GTR-173. Fort Collins, CO.: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 34p.
15. Rosen, P.C. 1991. Comparative field study of thermal preferenda in gartersnakes (*Thamnophis*). Journal of Herpetology 25: 301-312.
16. 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.
17. 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.

18. U.S. Fish and Wildlife. 2013. Federal determination for critical habitat for Mexican and narrow headed gartersnakes. Federal register 78 no 132, 2013.

Western Painted Turtle (*Chrysemys picta belli*)

Overview

Predicted Impacts	
Habitat Change	
2030	26 to 68 % Loss
2060	48 to 82 % Loss
2090	96 to 100 % Loss
Adaptive capacity	Moderately Low
Fire Response	Mixed

Status:
Chrysemys picta has four subspecies, of which only *C. p. belli* is found within New Mexico. Collectively, *C. picta* are considered "Demonstrably Secure" in the Natural Heritage global rank but more locally, *C. p. belli* is considered a species of greatest conservation need in New Mexico and critically imperiled in Arizona by State Heritage Programs (NMDGF 2006).

Range and Habitat:
The western painted turtle is found throughout the western U.S. though its range is limited to riparian systems in warmer areas like the Southwest (Figure 1). *Chrysemys picta belli* is an aquatic species that is found in slow-moving or still water bodies. This species has been found on the margins of lakes and river pools, streams, ditches and cattle tanks. They can be present in ponds as much as a mile from a river and are typically not found in the river itself, but in water bodies associated with rivers. Nests are dug in a variety of soils (with a depth of at least 9 cm) in a sunny area.

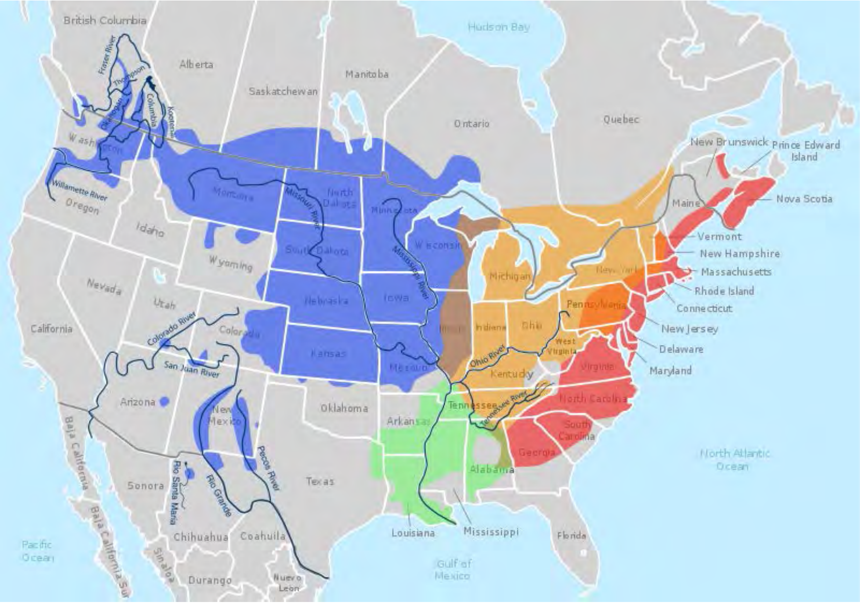


Figure 1. Range of *Crysemys picta belli* (in blue)

Western Painted Turtle (*Chrysemys picta belli*)

Climate Impacts and Adaptive Capacity

Adaptive Capacity Score = 0.84 (Moderately low)

Though *C. p. belli* is likely to be sensitive to a number of future changes, it scored neutral for many traits (Table 1). The western painted turtle is closely associated with wet habitats in the southwest, with few specific requirements outside of the presence of water. There were no indications that this species will be overtly negatively impacted by perceived changes in habitat quality and *C. belli* does not rely on additional transitional habitats. *Chrysemys p. belli* prefers slow moving waters that are shallow and have muddy bottoms with underwater vegetation and logs upon which to bask. Hibernation sites may be a limiting factor as this species is sensitive to anoxic conditions. Reductions in the availability of water bodies are recognized as a potential risk for this species. Dispersal appears good for this species: they have the ability to home 1-2 miles when displaced and some individuals have traveled as much as 26 kilometers. Others note that this species is able to traverse several kilometers overland to move between water bodies. However, the potential for dispersal to new suitable sites is limited by a lack of alternative sites to the study area, which reduces the functional benefit of this capacity. In addition, females tend to stay within 150 meters of water for nest sites, indicating some dispersal limitations (COSEWIC 2006).

Physiology:
Chrysemys picta belli have a number of traits within the physiological category that might indicate increased sensitivity to climate change (Table 1). Critical maximum temperature reported for this species varies geographically (BISON-M 2009), but environmental moisture is critical to the successful development of eggs. Though higher temperatures can increase juvenile development, hatchling success has been reduced at higher temperatures when on dry substrates. Optimal nest temperatures are 28-30°C (COSEWIC 2006), which could be exceeded in many locations within New Mexico. Additionally, it has been suggested that turtles in New Mexico nest under canopy and near water because they are already near the upper threshold for ideal incubation temperatures (COSEWIC 2006). This species also exhibits temperature dependent sex ratios. Females are produced when eggs are exposed to very low or higher temperatures (>29°C) and males result when eggs are exposed to temperatures < 26 °C. The Governor's Drought Task Force identified *C. p. belli* as a species susceptible to increased mortality due to drought and wildfire. This species is also known to be susceptible to hurricane related mortalities (COSEWIC 2006). *Chrysemys picta* is probably not at risk of reduced activity periods because it can submerge in water to mitigate extreme temperatures. The loss of water is considered as a potentially limiting habitat component rather than as a physiological issue. It does not possess specialized traits for dealing with variable resources. In New Mexico, male *C. picta* are reported to be sexually mature at 3 years and females at 5 years and are assumed to be long lived as are most turtle species. Thus, it is likely this species will be able to survive extended drought periods.

Phenology:
Though *Chrysemys picta* most likely uses a temperature cue for hibernation (BISON-M 2009), it is probably only slightly affected by timing changes. This species does not rely on distinctly

timed or remotely located resources that might increase the risk of mismatch between life history events and resource availability (Friggens et al., 2013). The western painted turtle appears to have a great deal of flexibility during the breeding season that would likely give it an advantage under changing climate. Breeding season begins with mating in Spring, followed by egg laying in June or July and emergence from nest in August. However, many hatchlings will overwinter in the nest and emerge the next spring. Mating may also occur in the fall in some populations and females can produce 1-3 clutches.

Biotic Interactions:

Chrysemys picta belli has a generalized diet that includes invertebrates, small vertebrates and plants (BISON-M 2009). Adults tend to consume more plant material. Nest predation can account for up to 21% of net loss. Young turtles may also fall prey to mink, muskrats, crows, racers, bullfrogs, fish and other turtles. We did not find evidence for an overall increase or decrease in food resources or predators or other potential interactions.

Table 1. Scoring sheet for the western painted turtle (*Chrysemys picta).*"v" indicates a vulnerability or sensitivity, "r" indicates a resiliency or increased adaptive capacity, "n" indicates neutral response.

Question	Score	Notes
1. Changes to non-modeled habitat components	v	Reduced water
2. Change in habitat quality	n	No
3. Dispersal ability (Site fidelity or other limitations)	v	Limited to water
4. Reliance on migratory or transitional habitats	n	No
5. Increase or decrease in physiological range limitation	v	Temperature thresholds
6. Sex Ratios determined by temperature or food changes	v	Yes
7. Response to predicted extreme weather events/disturbances	v	Drought sensitive
8. Changes to daily activity period	n	No
9. Variable life history traits or coping strategies	n	No
10. Ability to outlive limiting conditions	r	Yes
11. Migrates/hibernates in response to weather cues	v	Yes
12. Reliance on weather mediated resource (e.g. insect emergence)	r	No
13. Spatial or temporal separation between critical resources and life history stages	n	Unknown
14. Can adjust timing of critical activities	r	Multiple clutches, flexible breeding season

Question	Score	Notes
15. Likelihood for decreased food resource	n	Not predicted
16. Likelihood of increase predation	n	Not predicted
17. Loss of important symbiotic species	n	None known
18. Increase in high mortality/morbidity disease	n	Not predicted
19. Increased competitive pressures	n	Not predicted

Western Painted Turtle (*Chrysemys picta belli*)

Niche Model Analysis

Model AUC values were 0.97 for all three models. Habitat declines under all climate scenarios with substantial loss of suitable habitat by 2090 under all scenarios (Table 2, Figures 2 and 3).

Table 2. Percent of original predicted habitat under three climate models.

	CGCM3.1	**GFDL_CM2**	**Had_CM3**	**Average**
current	1	1	1	1
2030	0.74	0.32	0.54	0.54
2060	0.52	0.32	0.18	0.34
2090	0.04	0.00	0.01	0.02

Biophysical variables were most important for distinguishing suitable habitat (Table 3). Suitability declined with increasing distance from water and slope. The most suitable sites occurred at lower elevations. Suitability increased with increasing minimum winter temperature. Western painted turtles were associated with riparian habitats in Plains Grassland Biome and less likely to be present in riparian habitats within the SemiDesert Grassland Biome. Hinge functions were evident for a number of variables. Suitability was negatively related to annual potential evaporation of natural vegetation above values of 45mm. Mean Annual Runoff was positively associated with suitability at lower values. There was a negative relationship with Aridity index, but only for the low range of values. The lack of riparian habitats (areas with close proximity to water, low elevation) within the Plains Grassland Biome appears to drive patterns of expected loss in these models.

Table 3. Variables in order of importance for species habitat models. Relationship between variable and predicted suitability is listed as positive (+) negative (-) for linear and exponential curves or quadratic for parabola-like curves.

Variable	**CGCM3.1**	**GFDL_CM2**	**Had_CM3**	**Relationship**
Distance to water	27.8	28.6	27.7	-
Elevation	19.8	9.6	18.9	-
Slope	11.2	10.8	11.4	-
Biome	9.7	9.8	9.6	-SemDesGssd, + Plns Grssd
Mean Temp DJF (bio11)	8.9	9.4	9.2	Quadratic
Pot Evap Natural Vegetation	8.2	6.5	7.7	Peaks below 45mm
Mean Annual Runoff	6.4	4.5	7.0	Peaks at high values
Min Temp January (bio6)	3.4	3.3	4.2	+
Max Temp July (bio5)	2.9	14.8	2.2	+
Mean Temp JJA (bio10)	1.3	0.1	1.2	-
Aridity Index (AI)	0.2	2.4	0.8	-
Isothermality (bio3)	0.1	0.2	0.0	+,-

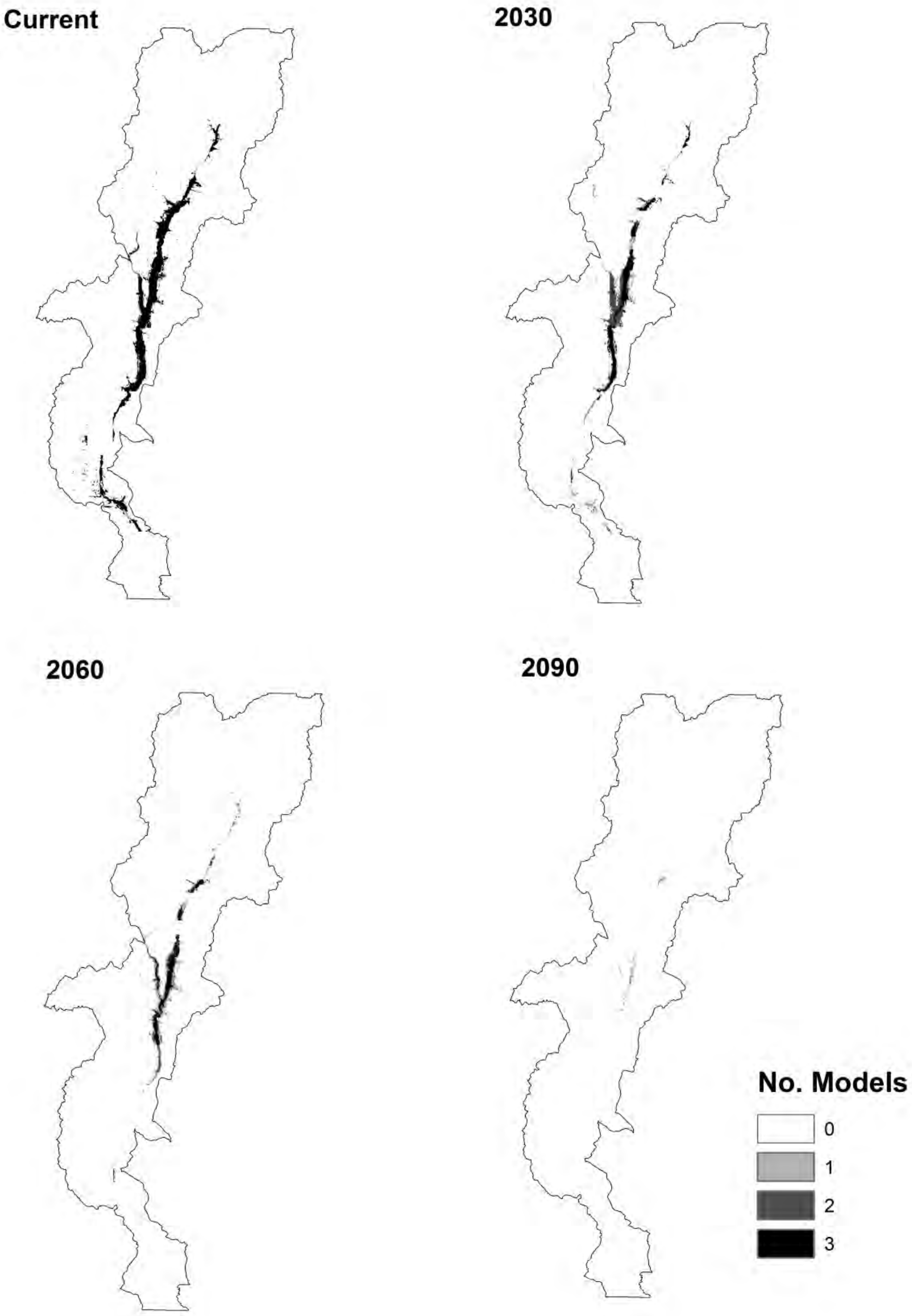


Figure 2. Distribution of suitable habitat for *Chrysemys p. belli* for three future time periods. Future habitat predictions are based on three climate models: Had_CM3, CGCM3.1, and GFDL_CM2.1. Shading indicates the number of models that predict suitable habitat for a given area.

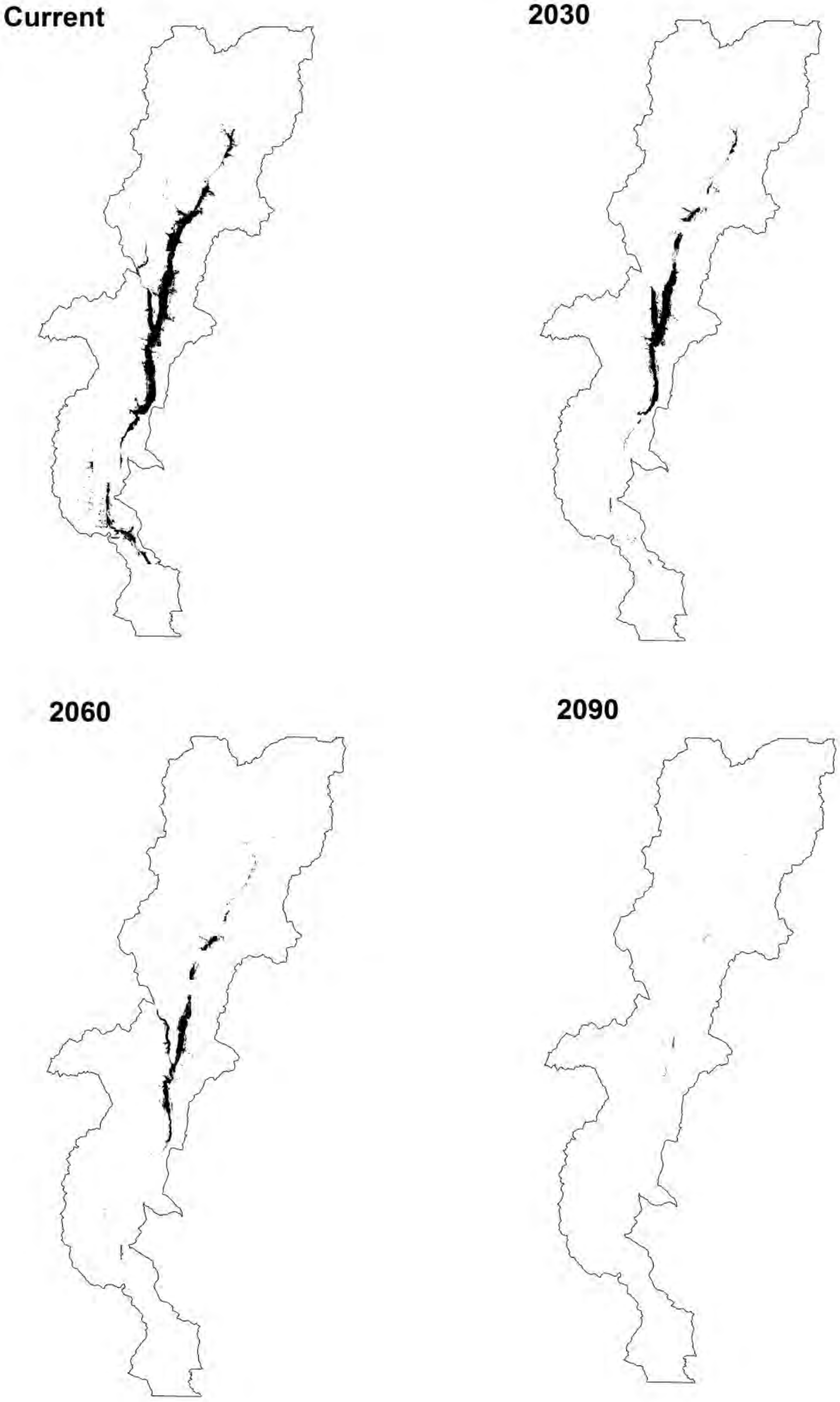


Figure 3. Two-thirds consensus models for suitable habitat for *Chrysemys p. belli* for three future time periods. Future habitat is predicted where at least two of three climate models, Had_CM3, CGCM3.1, and GFDL_CM2.1, identify suitable habitat.

Western Painted Turtle (*Chrysemys picta belli*)

Fire Analysis

Most reptiles appear to be ok with prescribed fire (Pilliod et al., 2006). According to Pilliod et al., (2006), characteristics of species that might benefit immediately post fire include those associated with open canopies and open forest floors and those that eat insects (long term response e.g. small mammals). Characteristics of species with negative impact include species associated with downed wood, snags, dwarf mistletoe, dense forests and closed canopies and small mammals that prefer shrub cover to avoid predators (short-term response).

Low-severity burns within riparian habitats can be beneficial by removing ground cover, releasing nutrients to remaining vegetation, and encouraging new growth. Riparian vegetation may be more resilient to wildfire, given that water is present and willows are known to be positively affected by low-severity burns that do not damage root crowns (Coleman 2011).

Research is limited regarding fire impacts for the western painted turtle. Research for the more terrestrial eastern box turtle shows significant mortality from fire in tallgrass prairies and other habitats (Luensmann 2006). However, it is likely box turtles with access to burrows are able to survive fires. Hatchling and juvenile eastern box turtles appear to hide under litter, which exposes them to fire, and leads to increased mortality. Eastern box turtles are often not able to escape the active line of fire and many suffer burn scars as a result (Luensmann 2006 and references therein). Frequent fires may result in more mortality, increase fragmentation of habitat and reduced turtle populations.

Luensmann (2006) concludes high-severity fires that kill trees and scorch canopies are likely to be detrimental to turtles that favor forests. Additionally, loss of the litter layer by fire could also have negative impacts because litter is used extensively for cover throughout the year. Though largely aquatic, similar effects could be envisioned for the western painted turtle where fires remove shading canopies and cause nestling mortalities. The Governor’s Drought Task Force identified *C. p. belli* as a species susceptible to increased mortality due to drought and wildfire.

We considered areas with predicted torching type fires as at high risk of negative impact due either directly to exposure to fire or associated changes in habitat. Areas with non-torching fires were classified as having a moderate risk of negative fire impact for this species.

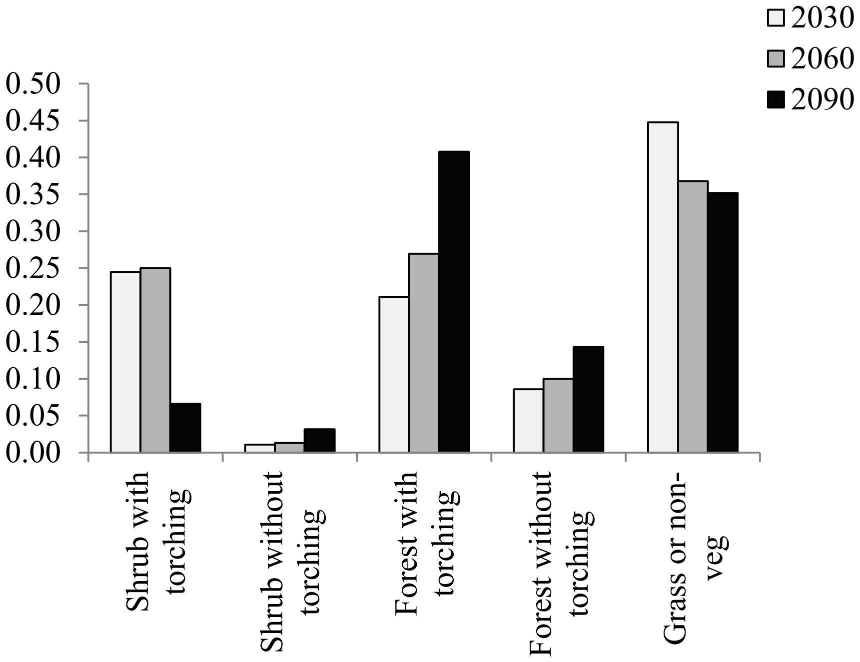
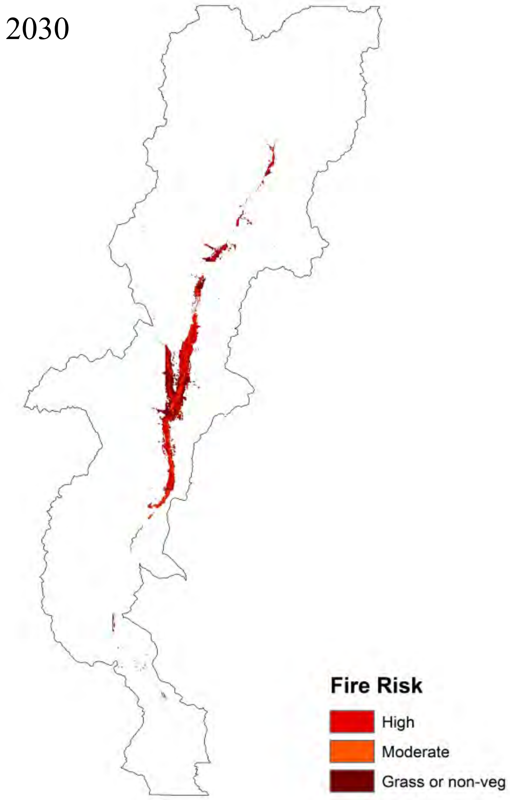


Figure 4. Percent of habitat falling within each fire type category.



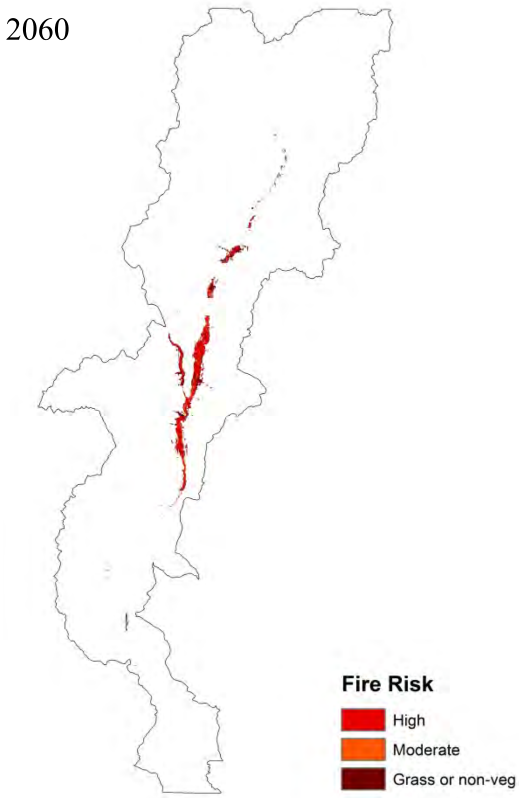
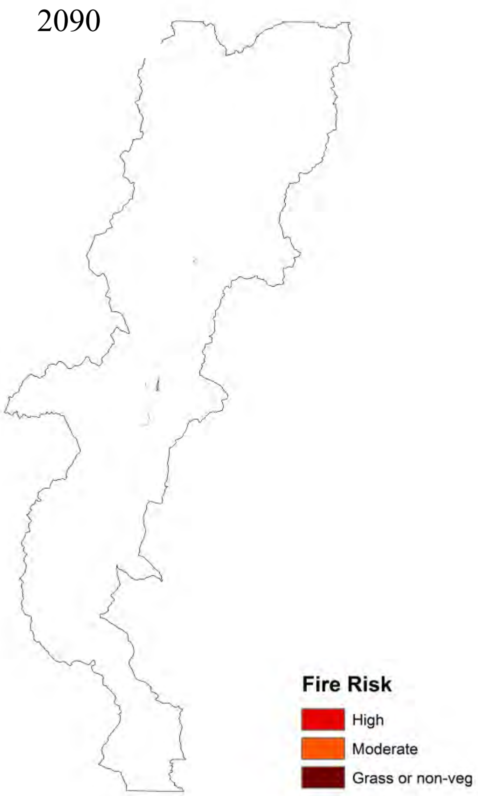


Figure 5. Fire impacts for *Chrysemys picta belli* habitat under three climate futures. Fire intensity was calculated using FSim (Finney et al., 2011) for each time period based on the GFDL-ESM-2m GCM under the RCP85 scenario. Fire risk maps were created by overlaying estimated species' habitat, predicted fire characteristic (shrub with or without torching, forest with or without torching or grass/nonveg) and species' response (negative, neutral, positive).



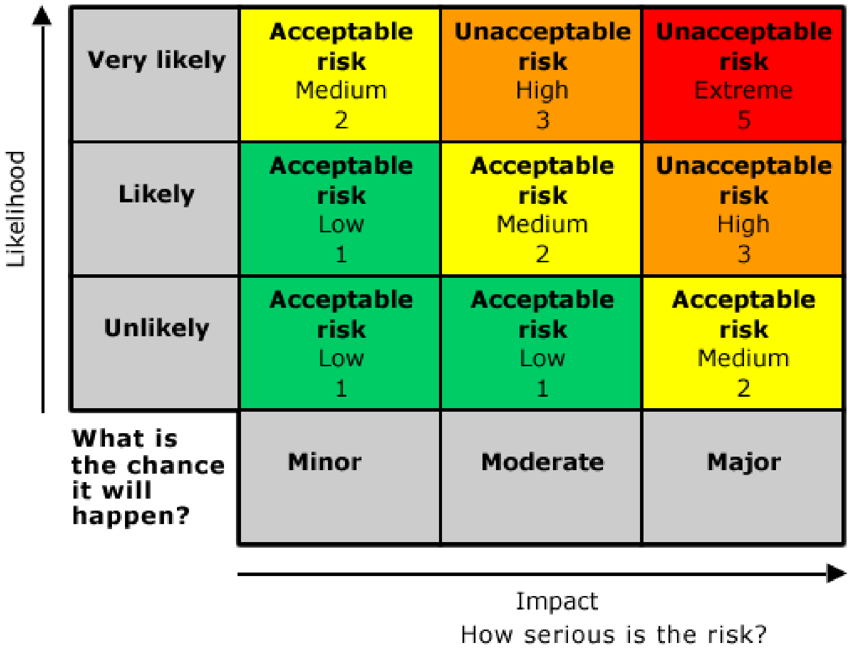
References:

1. 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
2. Coleman, S. 2011. *Thamnophis rufipunctatus* and *Thamnophis eques* information post-Wallow Fire. White paper drafted by Stephanie Colemen, Apache-Sitgreaves National Forest. 4 pp.
3. 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.
4. Finney, M.A., C.W. McHugh, I.C. Grenfell, K.L. Riley, and K.C. Short. 2011. A Simulation of Probabilistic Wildfire Risk Components for the Continental United States. Stochastic Environmental Research and Risk Assessment 25: 973-1000.
5. Fish and Wildlife Service. 2013. Federal determination for critical habitat for Mexican and narrow headed gartersnakes. Federal register 78: 132, 2013.
6. Friggens, M.M., D. M. Finch, K.E Bagne, S.J. Coe, and D.L. Hawksworth. 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.
7. Luensmann, P. S. 2006. *Terrapene carolina*. 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/ [2014, December 9].
8. New Mexico Department of Game and Fish. 2006. Comprehensive Wildlife Conservation Strategy for New Mexico. New Mexico Department of Game and Fish. Santa Fe, New Mexico. 526 pp +
9. Pilliod, D.S., Bull, E., Hayes, J.L., Wales, B.C. 2006. Wildfire and invertebrate response to fuel reduction treatments in dry coniferous forests of the Western United States: a synthesis. Gen. Tech. Rep. RMRS-GTR-173. Fort Collins, CO.: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 34p.

6. RISK MATRIX

Risk analyses have long been used in security and social science industries and more recently have come to the attention of natural resource managers. Within natural resource management, risk analysis and assessment provides a way to minimize subjectivity in ranking risks and allows activities to be prioritized based on a common understanding of the ranking process (Ojima et al., 2014). Importantly, risk management approaches also allow decision makers to incorporate uncertainty and resource values into their decision making processes. Risk-based frameworks have been recognized as an effective approach for dealing with impacts and uncertainty in resource planning under climate change (IPCC, 2007; Ojima et al., 2014). Recently, risk-based frameworks were applied to species, watersheds, and forest ecosystems to better understand the implications of climate change for management and planning within the U.S. (Vose et al., 2012; Ojima et al., 2014).

Risk is defined as the likelihood of an impact occurring and the magnitude of that impact on a resource of interest (Yohe 2010). At the core of the risk based-framework is the probability-impact matrix (Figure 6.1), which considers both the likelihood and magnitude of a disturbance or impact. Within the context of climate change, minor impacts occur when climate change is unlikely to have a measurable effect on structure, function, or processes within a specified timeframe (e.g., 2030s, 2050s, 2090s). Moderate impacts include situations where climate change will cause at least one measurable effect on structure, function, or processes within a specified timeframe. Major impacts occur in situations where climate change will cause multiple or irreversible effects on structure, function, or processes within a specified timeframe (Yohe, 2010). Likelihood can be considered low when climate change impacts are unlikely to be measurable within the specified timeframe; medium where climate change impacts are likely to be measurable within the specified timeframe; and high where climate change impacts are very likely (or have already been observed) within or before the specified timeframe. Using this



Very likely	**Acceptable risk** Medium 2	**Unacceptable risk** High 3	**Unacceptable risk** Extreme 5
Likely	**Acceptable risk** Low 1	**Acceptable risk** Medium 2	**Unacceptable risk** High 3
Unlikely	**Acceptable risk** Low 1	**Acceptable risk** Low 1	**Acceptable risk** Medium 2
What is the chance it will happen?	**Minor**	**Moderate**	**Major**

Figure 6.1. A conceptual risk matrix used to help identify risks associated with disturbances. This approach can be applied to climate change issues to prioritize management decisions.

scheme, a resource of interest can be categorized along a spectrum of risk, which corresponds to an action need. For example, low risk situations might be considered acceptable and require no action, whereas high risk situations indicate a need for immediate action.

We use a risk-based framework for wildlife species inhabiting the Rio Grande Bosque to highlight management needs under climate change. To do this, we follow the example laid out by Ojima et al. (2014) and Friggens and Mathews (2012) and apply a modified probability-impact matrix to Rio Grande species. The modified matrix borrows from the related concept of climate change vulnerability, where species' response is a function of exposure, species sensitivity, and potential adaptive capacity (IPCC 2007; Glick et al., 2011). The sensitivity of a species to a given climate disturbance or condition (exposure) represents the impact of that condition on the species. Adaptive capacity refers to the ability of the organism to ameliorate this impact. Vulnerability assessments based upon this concept are a proven tool for identifying issues for species and habitat under climate change (Glick et al. 2012; Friggens et al., 2013). In a risk matrix modified to consider climate change impacts for species, likelihood is represented by a measure of adaptive capacity. Though not an explicit temporal element, adaptive capacity represents the likelihood that a given impact will result in a negative response. Vulnerability assessments describe the why and what of species' response to climate change and rank species according to the potential for negative consequences. We use the risk-based framework to take the next step in the assessment process and begin to identify the when and how to prevent those consequences.

How to Interpret a Risk Matrix

The following risk matrices represent the capacity of species to deal with the indirect and direct impacts of climate change. These risk matrices do not consider socioeconomic or socio-ecological issues. To better represent uncertainty in future conditions, we estimate risk under three climate scenarios (see Section 2). For this study, magnitude of impact is estimated as the total change (+ or -) in area of suitable habitat under climate change as predicted through species niche models (Section 4.1). Change is represented as a departure from the starting value of 1 where 1.5 represents a 50% increase in available habitat and 0.5 represents a 50% loss. Our target time periods are 2030, 2060 and 2090).

Likelihood of impact is represented by the expected vulnerability of species to climate impacts as estimated using a modified version of the SAVS tool (Bagne et al., 2011, see also Section 4.2). Under the vulnerability framework for assessing species' impacts (Friggens et al., 2013), species are more or less susceptible to climate impacts based on their inherent sensitivities or adaptive capacity. Species with high adaptive capacity are likely to be less vulnerable to the same exposure to change than species with low adaptive capacity. Thus, the likelihood of impact can be viewed as a reciprocal of vulnerability: highly vulnerable species are more likely to be impacted by climate change. Higher scores indicate increased vulnerability or less capacity to deal with climate impacts. Negative scores indicate highly resilient species.

Four possible outcomes are possible within the risk matrix (Figure 6.2): Very high, Moderate and Low Risk, and Potential Benefit. Corresponding to each risk outcome are management need categories. We chose management categories that could be generalized across a broad audience.

Under this scheme, management plans and actions are most urgent for species that fall within the Very High area of the chart. For these species, immediate actions to preserve habitat or breeding populations may be needed to reduce the negative impacts from climate change. Assessments are indicated for species with Moderate risk. Assessments of species status, other threats, and the need for new, modified or additional management plans are starting points. Assessments and research might also be used to decipher the relative importance of climate effects on species as a way to identify management intervention points. Species within the Moderate Risk category are likely to already be experiencing negative impacts as a result of changing climate and might benefit from mitigation strategies that reduce additive stressors. Species which fall within Low Risk areas should be considered susceptible to negative impacts but are at less risk of near term population declines. Monitoring and assessment are appropriate actions to determine if species are likely to suffer declines. Long term planning strategies to reduce stressors and habitat loss are likely to be successful for species within the Low Risk categories. Assessment is also indicated for species falling within the Potential Benefit category. Many of the characteristics associated with increased resilience or decreased vulnerability to climate change are found within invasive species. Though it is feasible that some native animals will benefit under climate change, there is a strong likelihood that increased variation in weather and resources will favor exotic species. Assessments of species falling within the Potential Benefit category should consider whether the species will experience range expansion or population increases to the detriment of native species and ecosystems.

Results and Discussion

Results averaged across all three climate scenarios show the majority of species fall within a Very High Risk category at some point during the study period (Figure 6.2). All bird and reptile speices, the Yuma bat, and the occult bat fell within the highest risk category by 2060 (Figure 6.2). The New Mexico meadow jumping mouse and the Northern leopard frog began within a moderate risk category but shift to Very High risk by the end of the century due to habitat loss. The American bullfrog and hispid cotton rat remained somewhat stable throughout the study time period. Though the hispid cotton rat was predicted to experience substantial increases in habitat under future climate conditions, there is almost no overlap between current and future predicted ranges, which may in effect represent habitat loss rather than gain for this small mammal species (see species account, Section 5).

Graphing data for multiple time periods and climate scenarios (Figs 6.3-6.6), shows variation in the response of species to climate impacts over the next century. A few species, like the birds (Figure 6.4), show rapid declines in habitat, whereas other species, like the reptiles (Figure 6.6) and the long-legged bat (Figure 6.5a) showed a steadier decline over time. For many species within this study, outcomes did not change across climate scenarios (Figs. 6.4 (Birds), 6.5b (Rodents), and to some extent 6.6 (Reptiles)). For others, climate scenario had a large impact (Figs. 6.2 (Amphibians), 6.4a (Bats)). Notable is the trend for the long-legged bat where habitat predictions increase under HadCM3.1 but decline under the other two scenarios. Predictions also differed for the Northern leopard frog although here declines of suitable habitat were expected under the HadCM3.1 scenario but increase under the other models. Under HadCM3.1, average winter precipitation decreases but monsoon precipitation increases. Both the long-legged bat and the Northern leopard frog predictions appear to track, in part, these precipitation trends with the long-legged bat associated with areas with higher winter precipitation (which decline over time) and the Leopard frog favoring areas with greater monsoon precipitation (which increase) under the HadCM3.1 scenario (see Species Reports, Section 5). Under the current analysis, we see unique trends in habitat availability for these species across different climate models, which points to a potential high level sensitivity to changing weather conditions, especially as it regards seasonal precipitation events. The remaining species showed less sensitivity to specific climate variables though overall, conditions appear to become less suitable.

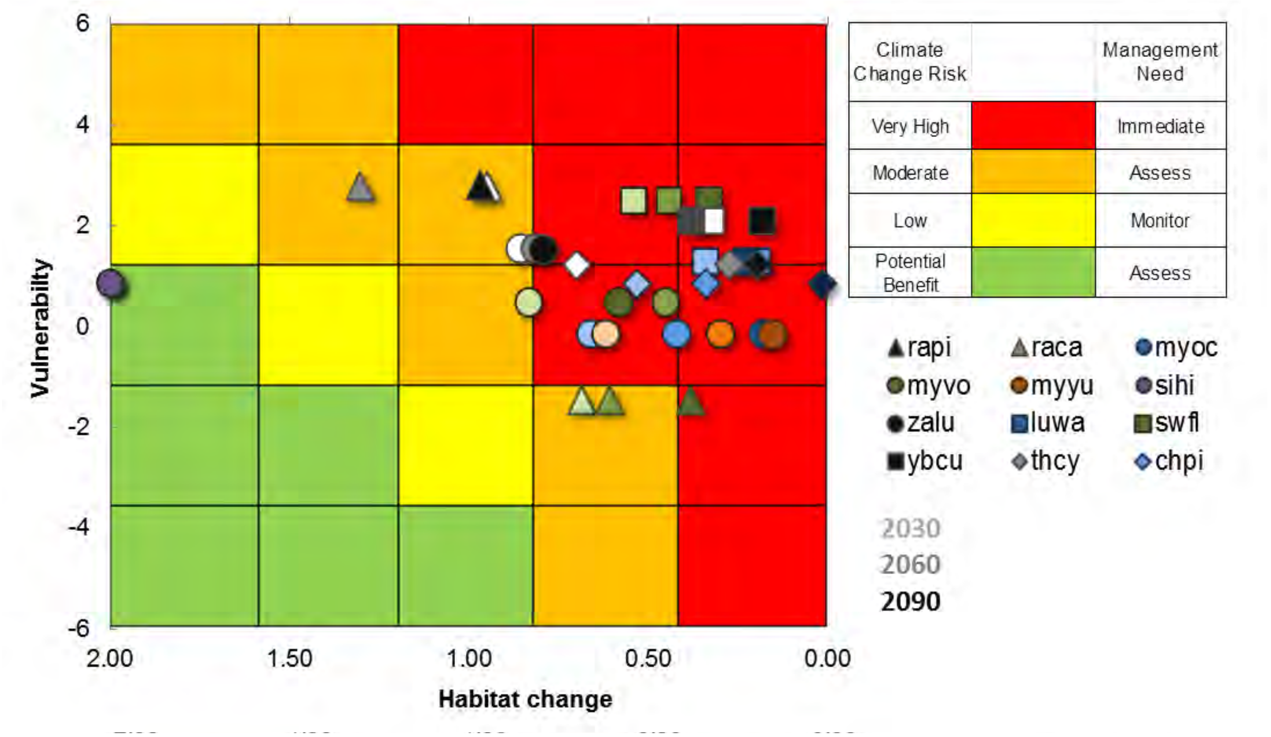
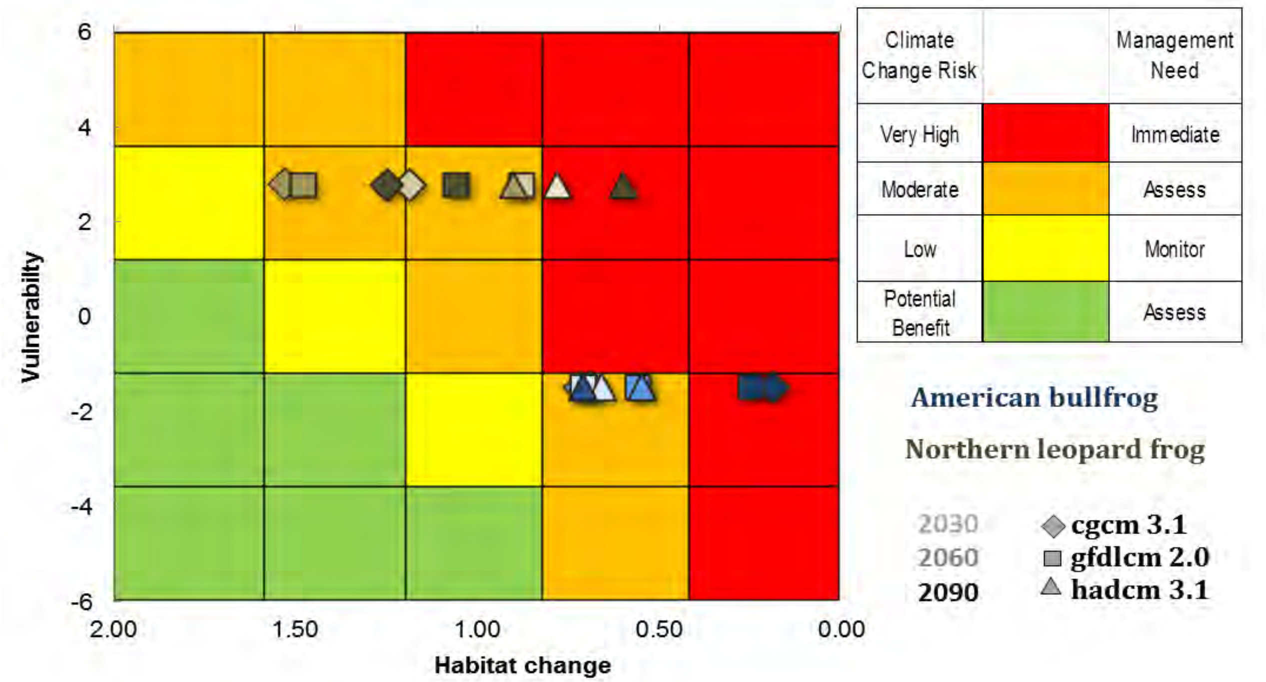


Figure 6.2. Assessment of risk for 12 species residing along the Rio Grande in New Mexico for three future time periods. Risk is a function of the expected magnitude of change in habitat availability and the likelihood of negative impacts from non-modeled predictors of climate change vulnerability (e.g. species' adaptive capacity and sensitivity). Each species is represented by a unique symbol and color scheme (b&w, blue, green, orange) falling along a spectrum (light to dark) representing time period. Species codes: rapi= *Rana pipiens* (Northern Leopard frog), raca= *R. catesbeiana* (American bullfrog), myoc=*Myotis occultus* (Occult bat), myvo= *M. volans* (long-legged bat), myyu= *M. yumanensis* (Yuma bat), sihi=*Sigmodon hispidus* (Hispid cotton rat), zalu=*Zapus luteus* (New Mexico Meadow jumping mouse), luwa= Lucy's warbler (*Oreothlypis luciae*), swfl=Southwestern willow flycatcher (*Empidonax traillii extimus*), ybcu= Yellow-billed cuckoo (*Coccyzus a. occidentalis*), thcy= *Thamnophis cyrtopsis* (black-necked gartersnake), and chpi= *Chrysemys picta belli* (Western painted turtle).

Amphibians (Figure 6.3)

The Northern leopard frog appears to be at greater risk from climate impacts than the American bullfrog. Northern leopard frogs had a much higher vulnerability score but more variable trends for habitat over time, whereas the American bullfrog was more resilience but also more likely to lose habitat under warming conditions. In general the leopard frog was associated with riparian habitat in hotter areas (see species report, Section 5). The predicted loss of bullfrog habitat concurs with the knowledge that bullfrogs require persistent water bodies for metamorphosis and are somewhat sensitive to high water temperatures (see Friggens et al., 2013). In an assessment of climate change vulnerability that did not include modeled output of habitat availability, the leopard frog was considered vulnerable due to the potential for increased interaction with American bullfrogs (Friggens et al., 2013). The greater resilience of the bullfrog to climate impacts and its capacity to reduce leopard frog populations through predation and competition contributed to its high ranking. However, warmer conditions may have the potential to reduce negative pressures by American bullfrogs if they do reduce habitat suitability as modeled in here. On the other hand, climate change is likely to reduce the availability of breeding ponds for both species and increase crowding, which will negatively impact leopard frogs, especially where both species coexist. The results of this analysis indicate the need for assessments and the development of management plans to mitigate habitat loss.



Climate Change Risk		Management Need
Very High	🟥	Immediate
Moderate	🟧	Assess
Low	🟨	Monitor
Potential Benefit	🟩	Assess

Figure 6.3. Assessment of risk for two amphibians residing along the Rio Grande in New Mexico for three time periods and under three climate models. Risk is a function of the expected magnitude of change in habitat availability and the likelihood of negative impacts from non-modeled predictors of climate change vulnerability (e.g. species' adaptive capacity and sensitivity). Each species is represented by a unique color scheme (blue, tan) with shading indicating time period and shape indicating climate model.

Birds (Figure 6.4)

All three bird species ranked as under Very High risk due to climate change. Lucy's warblers are likely to be most resilient to climate changes but are predicted to experience substantial declines in suitable breeding habitat. The Southwestern willow flycatcher is the most vulnerable to climate impacts but is expected to experience slightly less breeding habitat loss. The Western yellow-billed cuckoo experienced the greatest loss of breeding habitat. These results emphasize the need for strong actions towards preserving areas known to support nesting and foraging activities. Lucy's warblers, not currently considered a species of conservation concern, may warrant additional attention including the development of management plans to mitigate potential habitat loss.

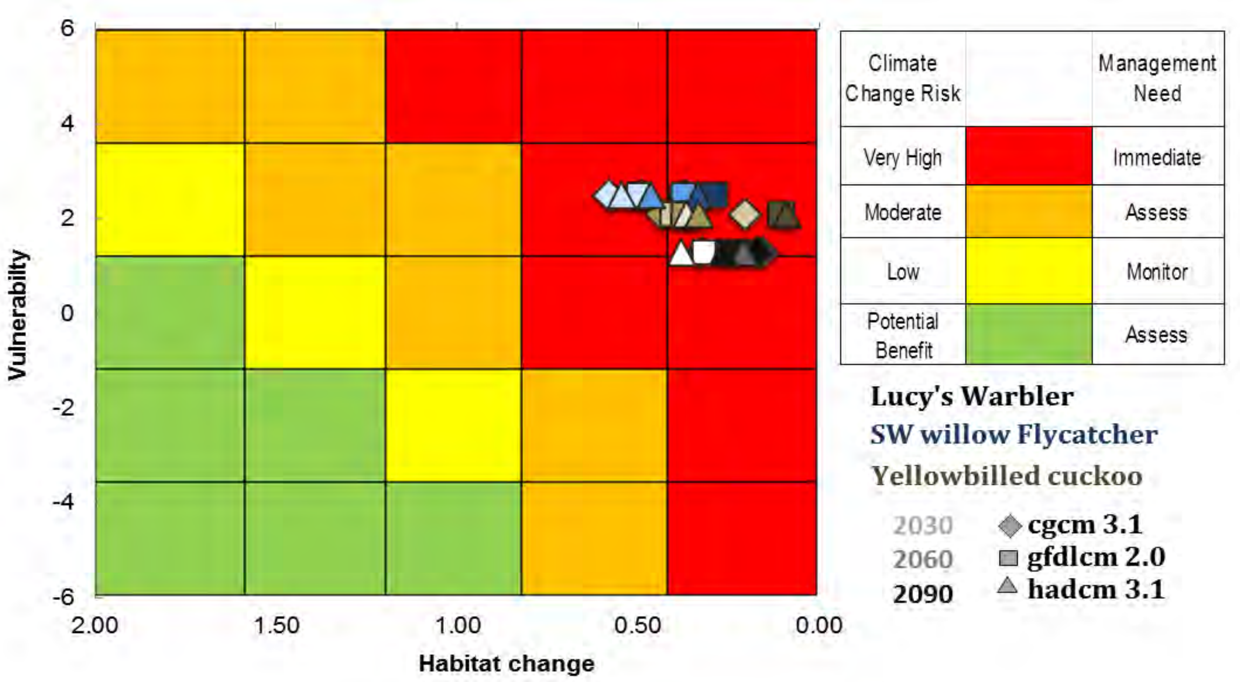


Figure 6.4. Assessment of risk for three birds residing along the Rio Grande in New Mexico for three time periods and under three climate models. Risk is a function of the expected magnitude of change in habitat availability and the likelihood of negative impacts from non-modeled predictors of climate change vulnerability (e.g. species' adaptive capacity and sensitivity). Each species is represented by a unique color scheme (black, blue, tan) with shading indicating time period and shape indicating climate model.

Small mammals (Figure 6.5)

All three bat species have the potential to experience substantial declines in suitable habitat. The Yuma bat and occult bat appear to be at particularly high risk due to climate change impacts. For these species immediate management action may be needed to prevent population declines. The long-legged bat showed range of possible responses under different climate scenarios. Though the long-legged bat is predicted to experience substantial declines in suitable habitat under dryer scenarios, habitat was predicted to increase under slightly wetter conditions. The bat species assessed here have distinct associations with habitats within New Mexico (see species reports). From our analysis, we cannot determine whether long-legged bat trends are driven by habitat associations which tend to fall within a range of climate conditions or whether they are a result of physiological thresholds that determine range limits. The risk matrix indicates a need for assessment of the long-legged bat in the short term and a potential to transition to active management over time. Among the assessment needs, studies of the relationship between bat roost and foraging requirements under variable weather conditions would improve our capacity to determine whether it is the physical features (trees, snags, water) or conditions (temperature range) or both that are important for limiting this species' distribution and whether this species might be at risk of population declines in the future.

The meadow mouse was predicted to lose habitat, especially under the hottest climate model. The meadow mouse is indicated for assessment in the near future and management actions by mid-century. The hispid cotton rat was predicted to experience an increase in habitat within the study area and categorized as likely to benefit. Primarily, habitat increases were due to warming conditions in Northern New Mexico, which appear similar to conditions in the current cotton rat range. However, there are two important caveats to this outcome: 1) new predicted suitable habitat exists far to the north of this species' current distribution (see species report); and 2) our models did not contain measures of vegetation structure (dense grasses) associated with hispid cotton rat presence. Though climate conditions may be favorable to cotton rats in northern reaches, it is not known whether these areas will have vegetation characteristics typically associated with this species and these results should be interpreted cautiously. Further assessment is indicated for the hispid cotton rat.

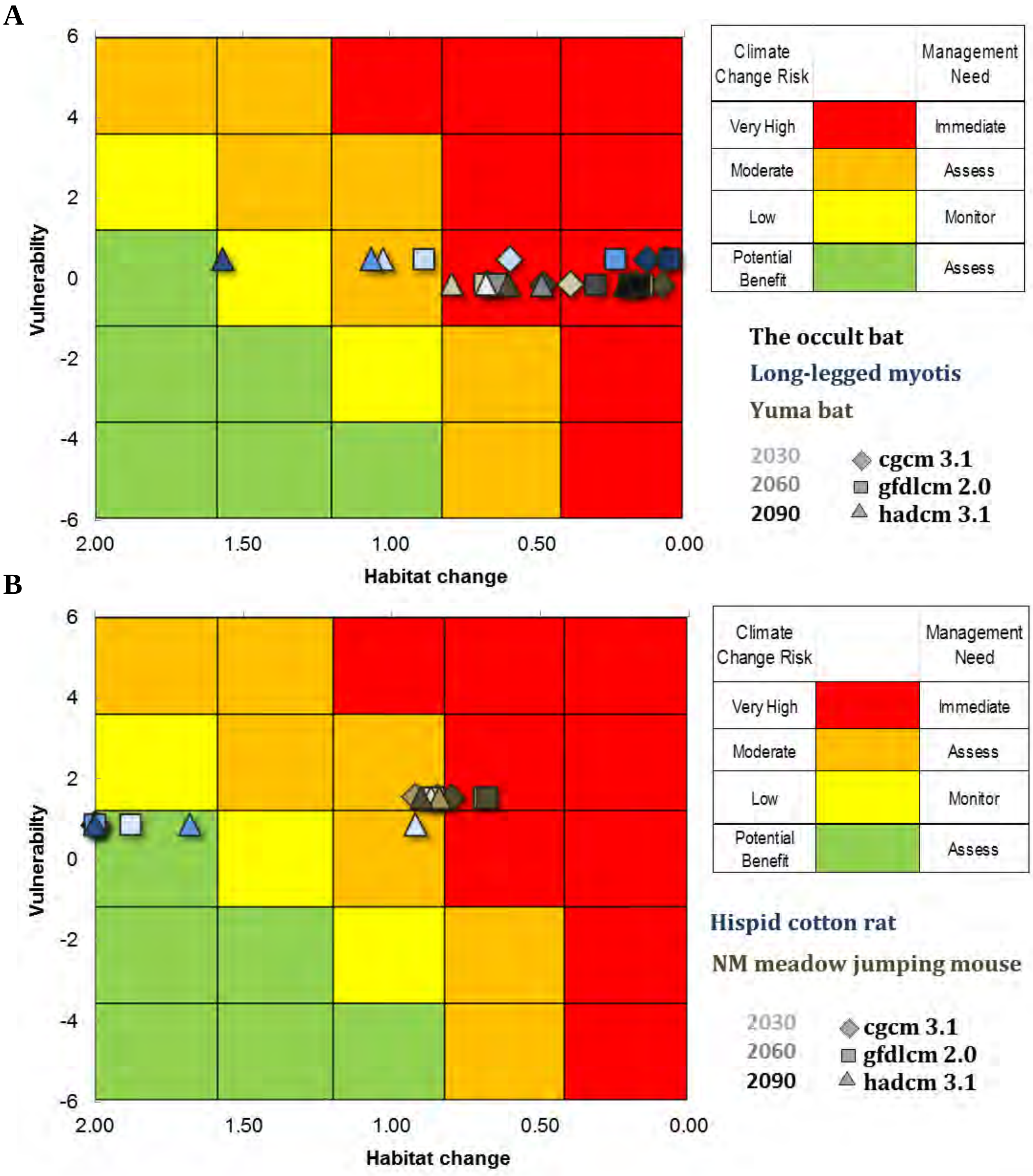


Figure 6.5. Assessment of risk for some small mammals (A: bats, B: rodents) residing along the Rio Grande in New Mexico for three time periods and under three climate models. Risk is a function of the expected magnitude of change in habitat availability and the likelihood of negative impacts from non-modeled predictors of climate change vulnerability (e.g. species' adaptive capacity and sensitivity). Each species is represented by a unique color scheme (black, blue, tan) with shading indicating time period and shape indicating climate model.

*The Hispid cotton rat was estimated to experience up to a 4-fold increase in suitable habitat. We changed all values to 2 to maintain consistency among figures

Reptiles (Figure 6.6)

Both reptile species ranked as at Very High risk by year 2030. Though models show variability among climate scenarios, habitat suitability declines steadily until the end of the century. The Western painted turtle experiences the most severe declines in habitat. This species is considered a species of greatest conservation need in New Mexico and critically imperiled in Arizona by State Heritage Programs. The black-necked gartersnake is considered secure within New Mexico but these results would suggest that this species should be reassessed, closely monitored, and considered within local management plans.

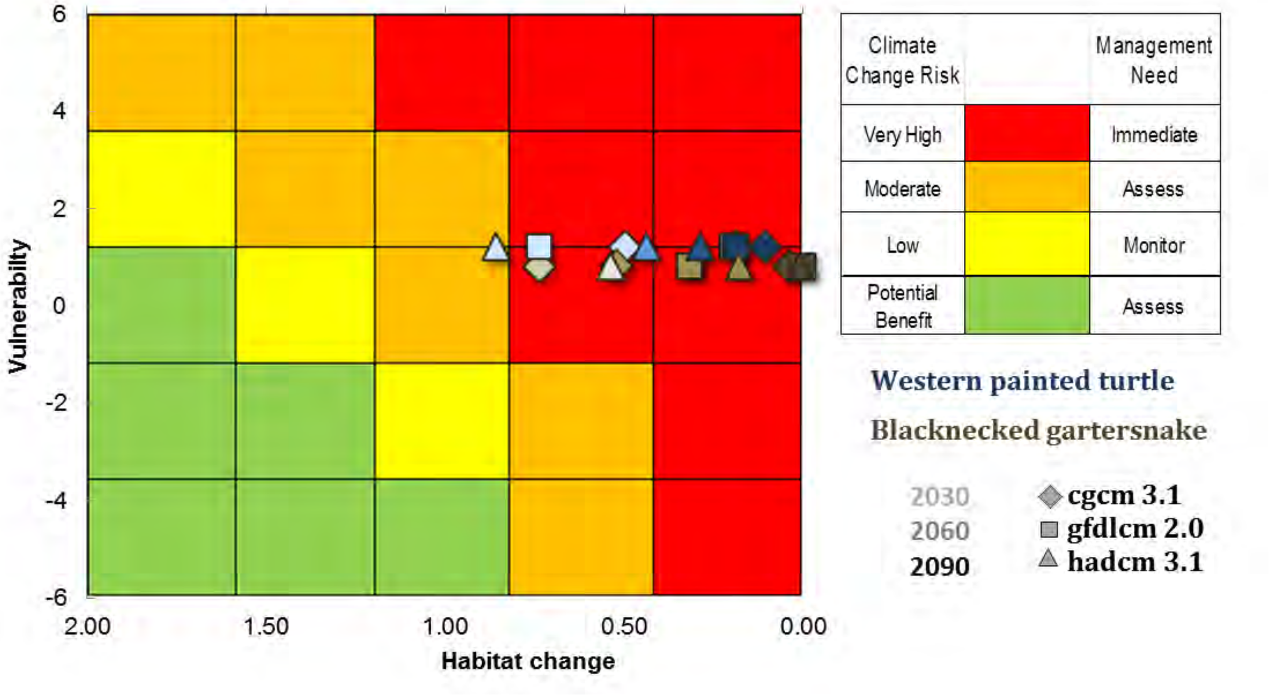


Figure 6.6. Assessment of risk for two reptiles residing along the Rio Grande in New Mexico for three time periods and under three climate models. Risk is a function of the expected magnitude of change in habitat availability and the likelihood of negative impacts from non-modeled predictors of climate change vulnerability (e.g. species' adaptive capacity and sensitivity). Each species is represented by a unique color scheme (blue, tan) with shading indicating time period and shape indicating climate model.

Conclusion

The application of risk-based frameworks to natural resource management issues is a promising method for prioritizing management under climate change. A risk matrix allows users to quickly identify the effect of both the magnitude and likelihood of climate change impacts. The risk matrix used here was specifically developed to integrate data from two important analysis methods: mechanistic species niche models and quantitative assessments of species vulnerability. This method can be applied to other species and geographic areas using available species' distribution data and vulnerability assessment tools are available (e.g. SAVS, NatureServe's CCVI, Young et al., 2011; and others see Friggens et al., 2013 for review). It is also possible to customize the current application to better fit individual organizations or agency management options. The risk categories and management implications used here were chosen to demonstrate the applicability of this system for climate change analysis but are easily modified to incorporate different risk levels and associated management implications. Other measures of magnitude and impact can also be used to identify management priorities for a variety of targets including watersheds, forests, and landscapes (e.g. Yohe et al., 2012; Ojima et al., 2014).

In general, species inhabiting the Rio Grande Bosque and associated upland areas appear to be at high risk of population declines as a result of climate impacts. Species that fell within the highest risk category either were expected to experience greater than 70% habitat loss, habitat loss with an above neutral vulnerability score, or had very high vulnerability scores (Figure 6.2). Importantly, neither our niche model nor our vulnerability assessment analyses account for human related changes in water availability and our results may underrepresent impacts for species that require open and/or flowing water. Of the species studied here, the Southwestern willow flycatcher, the New Mexico meadow mouse and the Western yellow-billed cuckoo are listed as endangered or threatened by the Endangered Species Act (ESA). For these species, our risk analyses are likely to substantiate expected population trends and may or may not influence current management plans. Management plans are more variable for the remaining species given at Very High risk ranking, however. Given the trends predicted within this analysis, there is a need to consider increased assessment and conservation planning activities for these species.

References

1. Bagne, K. E., M. M. Friggens, and D. M. Finch. 2011. A System for Assessing Vulnerability of Species (SAVS) to Climate Change. *USFS-GTR*-257. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. Fort Collins, CO.
2. Friggens, M., and M. Mathews. 2012. Risk-based framework and risk case studies. *In* Vose, J. M., Peterson, D. L., Patel-Weynand, T., eds. 2012. Effects of climatic variability and change on forest ecosystems: a comprehensive science synthesis for the U.S. Forest Sector. Gen. Tech. Rep. PNW-GTR-870. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 265 p.
3. Friggens, M., Bagne, K., Finch, D., Falk, D., Triepke, J. and A. Lynch. 2013. Review and Recommendations for Climate Change Vulnerability Assessment Approaches with Examples from the Southwest. *USFS-RMRS-GTR*-309. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. Fort Collins, CO.
4. Friggens, M., Finch, D., Bagne, K., Coe, S., and D. Hawksworth. 2013. Vulnerability of Species to Climate Change in the Southwest: Terrestrial Species of the Middle Rio Grande. USFS- RMRS-GTR-306. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. Fort Collins, CO.
5. Glick, P. Stein, B.A., and N. A. Edelson. 2011. Scanning the Conservation Horizon: A guide to climate change vulnerability assessment. National Wildlife Federation, Washington DC. 168p.
6. Intergovernmental Panel on Climate Change (IPCC) (2007), Climate Change 2007: The Scientific Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, edited by S. Solomon et al., Cambridge Univ. Press, New York.
7. Ojima, D., Iverson, L., Sohngen, B., Vose, J., Woodall, C., Domke, G., Peterson, D., Littell, J., Matthews, S., Prasad, A., Peters, M., Yohe, G., and M. Friggens. 2014. Risk Assessment. *In* Peterson, D.L., Vose, J.M., Patel-Weynand, T., eds. Climate change and United States forests. The Netherlands: Springer: 223-244.
8. Vose, J. M., Peterson, D. L., Patel-Weynand, T., eds. 2012. Effects of climatic variability and change on forest ecosystems: a comprehensive science synthesis for the U.S. Forest Sector. Gen. Tech. Rep. PNW-GTR-870. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 265 p.
9. Yohe, G., and R. Leichenko. 2010. Adopting a risk-based approach. Annals of the New York Academy of Sciences. 1196: 29–40.
10. Young, B., E. Byers, K. Gravuer, K. Hall, G. Hammerson, and A. Redder. 2011. Guidelines for using the NatureServe climate change vulnerability index. NatureServe, Arlington, Virginia.