# Appendix F Species and Species Assemblages: Conceptual Models and Ecological Status

Version December 8, 2014

# Table of Contents


**OVERVIEW OF APPENDIX F** ..... **7**

**Overview of the Conceptual Models** ..... **8**

**Overview of the Status Assessment Results** ..... **11**

**DISTRIBUTION MAPPING METHODS** ..... **13**

**STATUS ASSESSMENT METHODS** ..... **14**

**Linking CE Conceptual Models to CE Status Assessments** ..... **14**

**KEAs, Indicators, and Scenarios** ..... **14**

**CE Response Models** ..... **16**

**Overall Ecological Status Scoring** ..... **22**

**Considerations and Limitations for All Species** ..... **22**

**SPECIES AND SPECIES ASSEMBLAGES: CONCEPTUAL MODELS AND ECOLOGICAL STATUS** ..... **24**

**MAMMALS** ..... **25**

F-1 Pronghorn (Antilocapra americana) ..... 25

F-1.1 Conceptual Model ..... 25

F-1.2 Considerations and Limitations ..... 38

F-1.3 Ecological Status Assessment Results and Interpretation ..... 39

F-1.4 References for the CE ..... 44

F-2 Coues White-tailed Deer (Odocoileus virginianus couesi) ..... 48

F-2.1 Conceptual Model ..... 48

F-2.2 Considerations and Limitations ..... 60

F-2.3 Ecological Status Assessment Results and Interpretation ..... 60

F-2.4 References for the CE ..... 66

F-3 Desert Bighorn Sheep (Ovis canadensis) ..... 70

F-3.1 Conceptual Model ..... 70

F-3.2 Considerations and Limitations ..... 81

F-3.3 Ecological Status Assessment Results and Interpretation ..... 82

F-3.4 References for the CE ..... 87

F-4 Black-tailed Prairie Dog (Cynomys ludovicianus) ..... 93

F-4.1 Conceptual Model ..... 93

F-4.2 Ecological Status Assessment ..... 106

F-4.3 References for the CE ..... 107

**REPTILES** ..... **111**

F-5 Desert Box Turtle (Terrapene ornata subspecies luteola) ..... 111

F-5.1 Conceptual Model ..... 111

F-5.2 Considerations and Limitations ..... 118
F-5.3 Ecological Status Assessment Results and Interpretation ..... 119
F-5.4 References for the CE ..... 124

**AMPHIBIANS ..... 127**
F-6 Chiricahua Leopard Frog (Lithobates chiricahuensis) ..... 127
F-6.1 Conceptual Model ..... 127
F-6.2 Considerations and Limitations ..... 143
F-6.3 Ecological Status Assessment Results and Interpretation ..... 144
F-6.4 References for the CE ..... 150

**ASSEMBLAGES ..... 157**
F-7 Grassland Birds ..... 157
F-7.1 Conceptual Model ..... 157
F-7.2 Considerations and Limitations ..... 173
F-7.3 Ecological Status Assessment Results and Interpretation ..... 174
F-7.4 References for the CE ..... 179
F-8 Nectivorous Bats ..... 185
F-8.1 Conceptual Model ..... 185
F-8.2 Considerations and Limitations ..... 196
F-8.3 Ecological Status Assessment Results and Interpretation ..... 197
F-8.4 References for the CE ..... 202

**REFERENCES FOR APPENDIX F ..... 206**

# Tables

Table F-1. Species conservation elements (species CEs) selected for the Madrean Archipelago REA. 8
Table F-2. List of key ecological attributes (KEAs) identified for CEs in the conceptual models with their corresponding indicators and KEA indicator scenarios that were assessed for each of the species CEs. ..... 15
Table F-3. List of key ecological attributes identified for the Chiricahua leopard frog in the conceptual model with the indicators and associated KEA indicator scenarios that were assessed for this CE. ..... 15
Table F-4. CE response model values for each species CE. ..... 19
Table F-5. Key ecological attributes (KEA) of North American Pronghorn (*Antilocapra americana mexicana* and *A. a. americana)* ..... 35
Table F-6. Key ecological attributes (KEA) of Coues white-tailed deer (*Odocoileus virginianus couesi)* ..... 58
Table F-7. Key ecological attributes (KEA) of desert bighorn sheep (Ovis canadensis) ..... 78
Table F-8. Key ecological attributes (KEA) of Black-tailed Prairie Dog (Cynomys ludovicianus) ..... 105
Table F-9. Key ecological attributes (KEA) of the Desert Box Turtle (*Terrapene ornata luteola*) ..... 117
Table F-10. Key ecological attributes (KEA of the Chiricahua Leopard Frog (CLF) (*Lithobates chiricahuensis*) ..... 139
Table F-11. Examples of habitat characteristics for grassland bird species. ..... 164
Table F-12. Key Ecological Attributes (KEA) of the grassland bird assemblage ..... 171
Table F-13. Key ecological attributes (KEA) of nectar feeding bats: Lesser long-nosed bat (*Leptonycteris yerbabuenae*), Mexican long-nosed bat (*Leptonycteris nivalis*), and Mexican long-tongued bat (*Choeronycteris mexicana*) ..... 195

# Figures

Figure F-1. American pronghorn (*A. a. americana*), Animas Valley, Arizona. ..... 26
Figure F-2. American pronghorn (*A. a. americana*), Las Cienegas National Conservation Area, Arizona. ..... 26
Figure F-3. Distribution of pronghorn in the MAR ecoregion. ..... 27
Figure F-4. Conceptual model diagram for the North American pronghorn, showing key ecological attributes ..... 38
Figure F-5. Scores for three indicators for pronghorn: development indicator (1st map), fire regime departure indicator (2nd map), and invasive species indicator (3rd map) for each 30m pixel. 40
Figure F-6. Overall ecological status scores for pronghorn for all three indicators combined (development, fire regime and invasives) for each 30m pixel (top) and 4km grid cells (bottom). ..... 43
Figure F-7. Frequency distribution of the 4km ecological status scores for pronghorn, with cumulative percent. ..... 44
Figure F-8. Coues white-tail deer (*Odocoileus virginianus couesi*). ..... 50
Figure F-9. Distribution of Coues white-tailed deer in the MAR ecoregion. ..... 51
Figure F-10. Conceptual model diagram for the North American Coues white-tailed deer, showing key ecological attributes ..... 60
Figure F-11. Scores for three indicators for Coues white-tailed deer: development indicator (1st map), fire regime departure indicator (2nd map), and invasive species indicator (3rd map) for each 30m pixel. ..... 62

Figure F-12. Overall ecological status scores for Coues white-tailed deer for all three indicators combined (development, fire regime and invasives) for each 30m pixel (top) and 4km grid cells (bottom). 65
Figure F-13. Frequency distribution of the 4km ecological status scores for Coues white-tailed deer, with cumulative percent. 66
Figure F-14. Desert bighorn sheep (*Ovis canadensis*) in southwestern New Mexico. 71
Figure F-15. Distribution of desert bighorn sheep in the MAR ecoregion. 72
Figure F-16. Conceptual model diagram for the desert bighorn sheep, showing key ecological attributes 81
Figure F-17. Scores for three indicators for desert bighorn sheep: development indicator (1st map), fire regime departure indicator (2nd map), and invasive species indicator (3rd map) for each 30m pixel. 83
Figure F-18. Overall ecological status scores for desert bighorn sheep for all three indicators combined (development, fire regime and invasives) for each 30m pixel (top) and 4km grid cells (bottom). 86
Figure F-19. Frequency distribution of the 4km ecological status scores for desert bighorn sheep, with cumulative percent. 87
Figure F-20. Black-tailed prairie dog (*Cynomys ludovidianus*). 93
Figure F-21. Distribution of black-tailed prairie dog in the MAR ecoregion. 94
Figure F-22. Conceptual model diagram for the black-tailed prairie dog, showing key ecological attributes 106
Figure F- 23. Desert box turtle (*Terrapene ornata luteola*), near Patagonia, Arizona. 111
Figure F-24. Distribution of desert box turtle in the MAR ecoregion. 113
Figure F- 25. Conceptual model diagram for the desert box turtle, showing key ecological attributes 118
Figure F-26. Scores for three indicators for desert box turtle: development indicator (1st map), fire regime departure indicator (2nd map), and invasive species indicator (3rd map) for each 30m pixel. 120
Figure F-27. Overall ecological status scores for desert box turtle for all three indicators combined (development, fire regime and invasives) for each 30m pixel (top) and 4km grid cells (bottom). 123
Figure F-28. Frequency distribution of the 4km ecological status scores for desert box turtle, with cumulative percent. 124
Figure F-29. *Chiricahua leopard frog (*Lithobates chiricahuensis*) - Sycamore and Scotia Canyons, Arizona.* 128
Figure F-30. Potential habitat distribution of Chiricahua leopard frog in the MAR ecoregion. 129
Figure F-31. Conceptual model diagram for the Chiricahua leopard frog, showing key ecological attributes 143
Figure F-32. Scores for indicators for Chiricahua leopard frog: development (1st map), aquatic invasive species (2nd map), recent severe burns (3rd map), and water use (4th map) for each 30m pixel. 145
Figure F-33. Overall ecological status scores for Chiricahua leopard frog for all four indicators combined (development, invasives, recent severe burns, and water use) for each 30m pixel (top) and 6th level watersheds (bottom). 149
Figure F-34. Frequency distribution of the 6th-level watershed ecological status scores for Chiricahua leopard frog, with cumulative percent. 150
Figure F-35. Distribution of grassland bird assemblage in the MAR ecoregion. 163

Figure F-36. Conceptual model diagram for the grassland bird assemblage, showing key ecological attributes ..... 173
Figure F-37. Scores for three indicators for grassland bird assemblage: development indicator (1st map), fire regime departure indicator (2nd map), and invasive species indicator (3rd map) for each 30m pixel. ..... 175
Figure F-38. Overall ecological status scores for grassland bird assemblage for all three indicators combined (development, fire regime and invasives) for each 30m pixel (top) and 4km grid cells (bottom). ..... 178
Figure F-39. Frequency distribution of the 4km ecological status scores for the grassland bird assemblage, with cumulative percent. ..... 179
Figure F-40. Distribution of nectivorous bat assemblage in the MAR ecoregion. ..... 192
Figure F-41. Conceptual model diagram for nectar feeding bats, showing key ecological attributes ..... 196
Figure F-42. Scores for three indicators for nectivorous bats: development indicator (1st map), fire regime departure indicator (2nd map), and invasive species indicator (3rd map) for each 30m pixel. ..... 198
Figure F-43. Overall ecological status scores for nectivorous bat assemblage for all three indicators combined (development, fire regime and invasives) for each 30m pixel (top) and 4km grid cells (bottom). ..... 201
Figure F-44. Frequency distribution of the 4km ecological status scores for the nectivorous bat assemblage, with cumulative percent. ..... 202

# Overview of Appendix F

This appendix contains the conceptual models and ecological status assessment results for the species selected as conservation elements (CEs) for the Madrean Archipelago REA. Appendix A describes the methods for selection of the CEs and the change agents (CAs), as well as the collection and organization of management questions (MQs) of interest to many partners active in this ecoregion. Appendices B and C contain the assessment methods used to assess status for all of the CEs: B contains the methodological approaches to the geospatial assessments, while C contains the technical GIS documentation. Other appendices contain the conceptual models and ecological status assessment results for the terrestrial CEs (Appendix D) and aquatic CEs (Appendix E). Three additional appendix volumes contain the ecoregional conceptual model and methods / results for the ecological integrity assessment (Appendix G); the conceptual models, methods and results for assessment of Mesquite Expansion: Restoration Opportunities (Appendix H); and the climate changes methods and results (Appendix I).

The content of this appendix is organized into the following major sections:

1. The Overview of Appendix F explains the content of the appendix to help the reader navigate the content, including a summary of how the CE conceptual models are organized, what material is provided in each one, and how the results of the assessment are organized for each CE.
2. The second section, Distribution Mapping Methods, provides a brief summary of methods used to map the distributions of the species and species assemblage CEs; detailed technical documentation of these methods is provided in Appendix C.
3. The third section, Status Assessment Methods, provides a brief summary of the status assessment methods that are specific to the CEs in this appendix; readers should reference Appendix B for complete details on the scientific rationale and technical approach to the status assessments.
4. The fourth section, Species and Species Assemblages: Conceptual Models and Ecological Status, contains the conceptual models and assessment results for each CE and is the primary focus of this appendix.
5. References for this appendix as a whole are at the very end of the document. (References for each individual CE are at the end of each of the CE sections under Species and Species Assemblages: Conceptual Models and Ecological Status.)

To help visually organize the content for readers, headings are **not** numbered for the sections containing the background or supporting or overview information. In addition, headings for the broader categorizations of the CEs (e.g., Mammals, Reptiles, etc.), are similarly **not** numbered. Sections containing the individual CE assessment content – conceptual models, status assessment results, and other CE-specific information – have outline-numbered headings (e.g., F-1, F-1.1, F-1.2, etc.).

The individual CE content follows the below structure**:**

1. Species X
    - 1.1. Conceptual Model
        - 1.1.1. Classification
        - 1.1.2. Distribution
        - 1.1.3. Population
        - 1.1.4. Reproduction

1.1.5. Habitat and Ecology
1.1.5.1. Habitat
1.1.5.2. Connectivity
1.1.5.3. Food
1.1.5.4. Phenology
1.1.5.5. Predators
1.1.5.6. Demographics
1.1.5.7. Disease and Parasites
1.1.6. Change Agent Effects on the CE
1.1.6.1. List of Primary Change Agents
1.1.6.2. Altered Dynamics and Stressors
1.1.7. Ecological Status: Key Ecological Attributes and Indicators
1.1.7.1. Key Ecological Attributes
1.1.8. Relationship of KEAs to Fundamentals of Rangeland Health
1.1.9. Conceptual Model Diagrams
1.2. CE-Specific Assessment Methods
1.3. Considerations and Limitations
1.4. Ecological Status Assessment Results and Interpretation
1.4.1. Current Ecological Status: Development, Fire Regime, Invasives
1.4.2. Current Ecological Status: All Change Agents
1.5. References for the CE

# Overview of the Conceptual Models

The conceptual models combine text, concept diagrams, and tabular summaries in order to state assumptions about the ecological composition, structure, dynamic processes, and interactions with major CAs within the ecoregion. These conceptual models provided the foundation for developing spatial models to enable gauging the relative ecological status of each Conservation Element (CE). The content included for each species CE conceptual model is described below. The MAR is a highly biologically diverse region, but only a select subset of eight species conservation elements (Table F-1) were chosen for the REA; methods for selection are described in Harkness et al. (2013). The descriptive material builds upon existing species descriptions compiled by NatureServe (see http://www.natureserve.org/explorer/index.htm to search and download existing descriptions). The information developed is generally intended to cover the full range of distribution of the CE, which can extend beyond the ecoregion, but includes characteristics or dynamics as they occur within this ecoregion to the extent that MAR-specific information was available for the species. The list of species CEs for the MAR is provided in Table F-1.

**Table F-1. Species conservation elements (species CEs) selected for the Madrean Archipelago REA.**

| **Mammals** |
|---|
| Pronghorn (*Antilocapra americana*)<br>Coues White-tail Deer (*Odocoileus virginianus couesi*)<br>Desert Bighorn Sheep (*Ovis canadensis*)<br>Black-tailed Prairie Dog (*Cynomys ludovicianus*) |
| **Reptiles** |
| Desert Box Turtle (*Terrapene ornata* subspecies *luteola)* |

| **Amphibians** |
| --- |
| Chiricahua Leopard Frog (*Lithobates chiricahuensis*) |
| **Assemblages** |
| Grassland Birds<br>Botteri's sparrow (*Peucaea botterii*)<br>Grasshopper sparrow (*Ammodramus savannarum*)<br>Scaled quail (*Callipepla squamata*)<br>Baird's sparrow (*Ammodramus bairdii*)<br>Nectivorous Bats<br>Lesser long-nosed bat (*Leptonycteris yerbabuenae*)<br>Mexican long-nosed bat (*Leptonycteris nivalis*)<br>Mexican long-tongued bat (*Choeronycteris mexicana*) |

## Conservation Element Characterization

This section of the conceptual model includes a narrative of the CE classification, distribution, protection status, biology and distribution status, and habitat and ecology with supporting literature cited.

***Classification –*** clarification of element taxonomy and clarification of any anomalies or changes concerning taxonomic distinctness of the species.
***Distribution –*** Total geographic range-wide extent of the species including breeding/non-breeding or seasonal ranges if specified and with a discussion of any MAR specific range issues.
***Population –*** Estimate of the total number of locations where the species is known to occur across its range including information on how the estimate was derived. When available, MAR-specific population information is provided.
***Reproduction –*** discussion of reproduction of the species across its range including information on clutch/litter size and frequency, gestation/incubation period, seasonal timing of reproductive activities, nature and period of any parental care, age of sexual maturity, and size and general nature of breeding aggregations.
***Habitat and Ecology –*** A description of habitats and microhabitats commonly used by the species throughout its range; including any daily seasonal and geographic variation within habitat use and considerations of mobility and connectivity between populations. A summary of the ecology of the species across its range, including information on food, phenology, predators, competitors, demographics, disease and parasites and any other significant ecological factors is also included.

***Change Agent Effects on the CE***

In this section the primary change agents and current knowledge of their effects on the CE are characterized. Some CAs have specific effects on each CE such as the alteration or destruction of habitat or disruption of connectivity between populations. This section lists the known change agents and then moves into describing the altered ecological dynamics of the CE, with a narrative on the effects of CAs on the individual CE.

***Diagrams for the Model***

Each species is represented by one diagram that includes key ecological attributes and stressors and change agents that are currently acting upon the key attributes. Diagrams are a visual representation of

how change agents interact with key ecological attributes for each species. They are designed to highlight key drivers for species ecological health.

### Ecological Status: Key Ecological Attributes and Indicators

NatureServe's ecological integrity assessment framework sets up practical criteria for assessing the ecological status of each CE within an ecoregion (Faber-Langendoen et al. 2006, Unnasch et al. 2009). This section of the conceptual model addresses Key Ecological Attributes and their potential indicators. The ecological status is a way of describing current status via criteria, functionality, or levels of attributes and asks if they are within the normal range of variation. Is it within its "proper functioning condition"? Attributes are direct and indirect measures of a species habitat status or function. Key Ecological Attributes (or their indicators) should be measured to take the "pulse" of a species habitat. High scores indicate high ecological integrity and high ecological functionality.

## Key Ecological Attributes

The key ecological attributes for the CE within the Madrean Archipelago ecoregion are identified in this section. A **key ecological attribute** of a focal ecological resource is a characteristic of the resource's biology, ecology, or physical environment that is critical to the resource's persistence in the face of both natural and human-caused disturbance, e.g., resistance or resilience (Holling 1973, De Leo and Levin 1997, Parrish et al. 2003, Unnasch et al. 2009). Alteration of such a characteristic beyond some critical range of variation will lead to the degradation or loss of the resource within decades or less.

For each CE, a table provides identified key ecological attributes, with a brief definition, a rationale for why it is important for the CE, and a listing of stressors or change agents that might be affecting the key attribute.

Key ecological attributes of a resource include critical or dominant characteristics of the resource, such as specific characteristics of:

a) demographic or taxonomic composition;
b) functional composition;
c) spatial structure;
d) range or extent.

They also include critical biological and ecological processes and characteristics of the environment that:

a) limit the regional or local spatial distribution of the resource;
b) exert pivotal causal influence on other characteristics;
c) drive temporal variation in the resource's structure, composition, and distribution;
d) contribute significantly to the ability of the resource to resist change in the face of environmental disturbances or to recover following a disturbance; or
e) determine the sensitivity of the resource to human impacts.

Conservation of key ecological attributes contributes to current ecological integrity and to the resilience of species in the face of large-scale or long-term stressors (Parrish et al. 2003). The ecological integrity assessment framework (Unnasch et al. 2009) identifies four classes or categories of key ecological attributes: landscape context; resource size or extent; biotic condition; and abiotic condition. These four may overlap, and provide a guide for considering and identifying key ecological attributes. They also provide a basis for integrating information on key ecological attributes.

- "Landscape context" refers both to the spatial structure (spatial patterning and connectivity) of the landscape within which the focal resource occurs; and to critical processes and

environmental features that affect the focal ecological resource from beyond its immediate geographic scope.

- "Size" refers to the numerical size and/or geographic extent of a focal resource.
- "Biotic condition" refers to biological composition, reproduction and health, and succession; and critical ecological processes affecting biological structure, functional organization (e.g., food-web guild structure), and interactions.
- "Abiotic condition" refers to physical environmental features and dynamics within the geographic scope of the focal resource that significantly shape biotic conditions, such as fire, weather, and hydrologic regimes; and soil and geological conditions and dynamics.

Taken together these attributes tell the story of the current status of a species habitat.

### Indicators of Key Ecological Attributes

Assessing the status of key ecological attributes requires explicit identification of indicators (also called metrics) – specific means for measuring their status. These are the detailed metrics that measure the amount or status of each key attribute. There are many potential indicators, and the choice is largely dependent on the purpose of the assessment and available data. An indicator may be a specific, measurable characteristic of the key ecological attribute; or a collection of such characteristics combined into a multi-metric index. Such indicators directly evaluate the condition of the KEAs and their responses to stressors (change agents).

Alternatively, indicators may evaluate the severity and extent of the stressors themselves. Such "stressor-based" indicators may consist of a single measurement type, or a collection of such measurements combined into a multi-metric stressor index. Indicators of stressors are often used as indirect indicators of a key ecological attribute, because data on stressor condition is often far more readily available than data on direct indicators. Examples of stressor-based indicators include measures of overall landscape development such as the Landscape Condition Model methodology (Comer and Hak 2009, Comer and Faber-Langendoen 2013); measurements of invasive non-native annual grass distributions that affect fire regimes; measurements of fragmentation due to development; or measurements of the stress water use places on aquatic natural resources.

During the data evaluation, analysis and assessment of status portions of the REA, the indicators used for each KEA for each CE were identified and are explained in Appendix B. The results of the status assessment are contained in this appendix.

### References for the CE

Literature is listed that is relevant to the classification, distribution, floristic composition, ecological processes, threats, stressors, or management of the CE, in some cases from portions of its range outside of the ecoregion. These are not exhaustive literature surveys, rather are an accumulation of known references. Some documents may be listed that are not cited in the narrative text.

## Overview of the Status Assessment Results

Each CE summary has a section titled **Ecological Status Assessment Results and Interpretation**. This section of the individual CE material presents the results of the CE status assessments, and includes both maps and accompanying interpretive text. Readers are referenced to Appendix B for the overall methodological approach for assessing status, and descriptions of scenarios that were used, including data inputs, process model diagrams, data outputs, and limitations. Appendix C provides the detailed

documentation of the GIS procedures followed to create the CE and CA distributions, and evaluate the scenarios for status.

Maps are provided for each CE showing the status or condition scores for each individual indicator at the resolution of the analysis unit (30m pixels), as well as the CE's overall ecological status scores, which is a combination of all indicators, at both a 30m resolution and rolled up into the 4km grid cell reporting unit. The following series of status results maps and charts are provided for each CE:

**Maps of individual indicator scores**

- Development, 30 meter resolution
- Fire Regime, 30 meter resolution
- Invasives, 30 meter resolution

Note: Chiricahua leopard frog has a different set of indicators, which is explained in the Status Assessment Methods section below, and also in the Chiricahua leopard frog section.

**Maps and charts of comprehensive ecological status assessment results**

- Ecological status, 30 meter resolution
- Ecological status, averaged across 4 km reporting units
- Chart showing frequency distribution of ecological status scores within 4km reporting units

The individual indicator results maps are grouped together for each CE, followed by text explanation and interpretation. The overall ecological status maps and accompanying charts are presented in a second grouping, followed by interpretive text. The interpretive text for the results does include material that is repeated for each CE, so that the reader will not need to return to the methods sections repeatedly.

# Distribution Mapping Methods

The datasets and methods used to map the distribution of species CEs in the Madrean Archipelago assessment area are briefly described here; Appendix C provides the technical details of the datasets and GIS processing steps used to map distributions. In general, existing datasets from the state wildlife agencies served as the foundation for the distribution maps. In a few cases, limited additional spatial modeling was conducted to generate distributions.

For the three ungulate species, pronghorn, Coues white-tailed deer, and desert bighorn sheep, distribution maps of their occupied habitat were created by combining the relevant distribution datasets provided by the AZDGF and the NMGF.

The Natural Heritage New Mexico program, under the auspices of the New Mexico Department of Game and Fish, modeled the potential distribution of desert box turtle across the MAR using New Mexico Natural Heritage data and extending the methods used for modeling distribution in the Arizona portion of the MAR (AZGFD HabiMap) into the New Mexico portion. The resulting data set was used to map current distribution for this CE.

The spatial distribution of the Chiricahua leopard frog (*Lithobates chiricahuensis*) CE was represented by selecting all 6$^{th}$-level watersheds (USDA NRCS Watershed Boundaries (HUC12)) that overlapped with leopard frog recovery units as identified in the USFWS Chiricahua Leopard Frog (*Lithobates chiricahuensis*) Recovery Plan (USFWS 2007). The resulting distribution is wide in scope to allow the ecoregional assessment to address landscape-scale management questions presented by the BLM technical team.

The data used to represent the distribution of the grassland bird assemblage was the distribution map for the MAR Apacherian-Chihuahuan Semi-Desert Grassland and Steppe CE; this dataset was derived from the TNC Arizona grasslands dataset, as described in the Distribution Mapping Methods section of Appendix D (as well as in Appendix C). This single data set was used because there are too many species with distinct habitat needs in the assemblage to feasibly utilize all of the individual distributions for the analysis.

A new distribution map for nectar-feeding bats within the MAR ecoregion was modeled by using roost sites as focal points that were buffered based on proximity to habitat (vegetation types) and elevation. All nectivorous bats roosts in Arizona and New Mexico were buffered using a radius between 25 and 50 miles. (The buffer radius was selected to take in account nightly foraging distances as summarized in USFWS 2013, but is provided as a range to mask the location of roost sites being buffered). These areas were intersected with distributions of five ecosystems[1] expected to support bat habitat and limestone soil distributions (indicating agave habitat) to identify areas with suitable habitat (see Appendix C for a detailed description of the distribution modeling for this assemblage). Because of the sensitivity of bat roost sites to human disturbance, this distribution data is sensitive and was not delivered to the BLM.

[1] The five ecosystems are 1) MAR Apacherian-Chihuahuan Semi-Desert Grassland and Steppe, 2) MAR Madrean Encinal, 3) Sonoron Palo-Verde Mixed Cacti, 4) MAR North America Warm Desert Riparian Woodland and Shrubland Mesquite Bosque and Stream, and 5) MAR North America Warm Desert Lower Montane Riparian Woodland and Shrubland and Stream

# Status Assessment Methods

Appendix B describes the conceptual scientific approach and rationale for the ecological status assessment (**Appendix B: Rationale for Ecological Status Assessment Approach**) and the detailed technical approach for conducting the assessment (**Appendix B: Ecological Status Assessment Technical Approach**). As described there, a raster-based spatial modeling tool, the Landscape Condition Model (LCM), was used to assess ecological status of CEs. Two categories of inputs are needed to assess ecological status using the LCM: 1) the CE response models, and 2) the spatial KEA indicator scenarios. The CE response model is a series of numeric values that characterize how each CA is expected to reduce status or condition of the CE onsite (site intensity values) and, in some cases, offsite (distance values); the response model values were assigned by ecologists on the contractor team using the information on the CE's ecology and dynamics as summarized in the CE's conceptual model. The site intensity values indicate the degree to which the impact of the specified CA features degrades the ecological status of the CE where the CA feature is present. The KEA indicator scenarios are aggregations of spatial raster datasets representing the CA features that were identified to assess each of the indicators for the CE. The starting point of the model is a theoretically perfect status or condition score of 1.0 for each pixel of a CE's distribution; zero is the lowest status score. The LCM tool applies the CE response model values for each of the CA features to the KEA indicator scenarios to calculate overall ecological status scores for the CE across its distribution. Where multiple CA features overlap, the associated response model values were multiplied to approximate a cumulative CA effect. The overall ecological status scores indicate the degree to which the combined CAs present in the CE's distribution degrade the ecological status of the CE, accounting for distance effects as appropriate. Readers should refer to Appendix B for more detail and background on how the status assessment was conducted.

# Linking CE Conceptual Models to CE Status Assessments

It is important that the ecological status assessment of CEs be grounded in what is known about each of the CEs – their ecology, dynamic processes, and stressors. The conceptual models developed for the species and species assemblage CEs of the MAR provided the scientific context and current knowledge base from which to identify the key ecological attributes (KEAs) and their indicators to be assessed to characterize ecological status, and to characterize CE responses to CAs via the CE response models (see **Appendix B: Rationale for Ecological Status Assessment Approach)**.

## KEAs, Indicators, and Scenarios

The status assessment of species CEs in the MAR focused on three primary KEA/indicator pairs for which spatial data were available: 1) Landscape Condition/Development, 2) Fire Regime/Fire Regime Departure, and 3) Biotic Condition/Invasive Species, with the exception of Chiricahua leopard frog and black-tailed prairie dog. For the KEAs specifically identified for species CEs in this REA, data to measure direct indicators of status were not available; therefore, all of the indicators used to assess ecological status are indirect, stressor-based indicators. The KEAs and indicators assessed are listed in Table F-2. For each of the indicators selected for the species status assessments, a KEA indicator scenario representing the spatial extent of the change agents identified to assess the indicator was generated (see **Appendix B: Scenario Generation: Current and Future**); the KEA indicator scenarios associated with each indicator are also listed in Table F-2. The types of spatial data used to represent the relevant change agents are listed in the Indicator Datasets column of Table 3-3 in the main body of the report. In addition, not all KEAs identified for each CE could be assessed due to data gaps; there may be more KEAs identified for an individual species CE than were evaluated to determine ecological status.

**Table F-2. List of key ecological attributes (KEAs) identified for CEs in the conceptual models with their corresponding indicators and KEA indicator scenarios that were assessed for each of the species CEs.** (except for Chiricahua leopard frog and black-tailed prairie dog).

| ***KEA Class:*** **KEA Name** | **Indicator Name** | ***KEA Indicator Scenario*** (as named in Appendix B) | **Type and Description of Indicator Used** |
|---|---|---|---|
| ***Landscape Context:*** **Habitat Condition** | **Development** | ***Current Scenario, Landscape Condition*** | Stressor-based: Modifications to land surface for human use that affects CE habitat directly or indirectly |
| ***Landscape Context:*** **Habitat Availability** | **Development** | ***Current Scenario, Landscape Condition*** | Stressor-based: Modifications to land surface for human use that modifies or destroys the CE habitat |
| ***Biotic Condition:*** **Forage Quality/ Vegetation Composition** | **Invasive Plant Species** | ***Current Scenario, Vegetation Composition*** | Stressor-based: Abundance of invasive species (mesquite and exotic grasses & forbs) |
| ***Abiotic Condition:*** **Fire Regime** | **Fire Regime Departure** | ***Current Scenario, Fire Regime*** | Stressor-based: Altered fire regimes as reflected in successional classes & their proportions |

The Chiricahua leopard frog was assessed using a different suite of four KEAs/indicators (for which spatial data were available), as listed in Table F-3 (and also listed in Table 3-3 in the main report). These indicators address the unique CAs of non-native aquatic species that prey on Chiricahua leopard frogs and the species' life cycle requirement for water. For each of the indicators selected for the leopard frog assessment, a KEA indicator scenario representing the spatial extent of the change agents identified to assess the indicator was generated (see **Appendix B: Scenario Generation: Current and Future**); the KEA indicator scenarios associated with each indicator are also listed in Table F-3. The types of spatial data used to represent the relevant change agents are listed in the Indicator Datasets column of Table 3-3 in the main body of the report.

**Table F-3. List of key ecological attributes identified for the Chiricahua leopard frog in the conceptual model with the indicators and associated KEA indicator scenarios that were assessed for this CE.**

| ***KEA Class*** | **KEAName** | **Indicator** | **KEA Indicator Scenario** | **Indicator Notes** |
|---|---|---|---|---|
| ***Landscape Context*** | **Habitat Condition** | **Development** | ***Current Scenario, Landscape Condition*** | Stressor: Modifications to land surface for human use that affects CE habitat directly or indirectly |
| ***Landscape Context*** | **Habitat Availability** | **Development** | ***Current Scenario, Landscape Condition*** | Stressor: Modifications to land surface for human use that modifies or destroys the CE habitat |
| ***Biotic Condition*** | **Non-native Aquatic Species** | **Invasive Species** | ***Current Scenario, Aquatic Invasives*** | Stressor: Abundance of invasive species (bullfrog and other high-impact species) |
| ***Abiotic Condition*** | **Condition of Water Sources** | **Recent Severe Burns** | ***Current Scenario, Fire Severity*** | Stressor: Recent severe burns that increase sedimentation into CLF Habitat |
| ***Abiotic Condition:*** | **Condition of Water Sources** | **Water Use** | ***Current Scenario, Total Water Use*** | Stressor: Water resource removed from CLF habitat |

Ecological status was not assessed for the black-tailed prairie dog beyond providing a potential habitat model because management questions tied to reintroduction are highly locally driven and therefore not readily addressed by the standard status assessments or data collected for this REA.

# CE Response Models

As described in Appendix B (**Ecological Status Assessment Technical Approach)** and above, the KEA scenarios were input into the LCM in conjunction with a response model for each CE; the LCM first intersected the CE distribution map with the KEA indicator scenario, and then the response model was applied to those intersecting pixels to derive a raster map of the calculated status or condition score for each pixel in the CE's distribution. The response model was constructed using information from the CE conceptual models to characterize how a CE is expected to respond in the presence of the CAs (and in some cases, a distance out from the CA) for a particular indicator; Table F-4 below provides a complete listing of all the CAs used in the KEA indicator scenarios and the associated response values (site intensity and distance) that were assigned for each species. Within each of the following categories of CA features, site intensity and distance decay were assigned the same values for all species: Transportation, Mining and Landfills, Energy, Recreation, and Urbanization. Values are also the same for the specific CA inputs of communications towers, below-ground corridors and above-ground corridors.

## Response Model Values by Indicator

This section provides additional information is provided about each indicator and how the associated site intensity and distance values were assigned in the CE response models for the status assessment. In addition, see **Appendix B: Species Current Scenario Generation Process Model** where more details are provided about the inputs and limitations for the corresponding KEA indicator scenarios for each of the indicators.

### Development Indicator

The development indicator is a stressor-based indicator of the spatial extent and intensity of human modifications to the land surface that alter the habitat of species CEs in the MAR ecoregion. The indicator takes into account the extent and density of urban development; infrastructure such as above- and below-ground distribution corridors, communication towers, and border barriers; a wide range of transportation features; mines and landfills; recreational development; agriculture; and energy development. The site intensity values assigned to the various development features ranged from 0 to 1, with the highest value of 1.0 indicating no ecologically relevant effects, and the lowest value of 0.0 indicating modifications that essentially eliminate all natural cover and ecological functions.

It is important to note that most development features were assigned much lower values than the non-development change agents (i.e., fire regime departure, and invasives); for example, site intensity values for urbanization range from 0.05 to 0.6 for high to low density development, respectively, while the lowest site intensity values for invasives start at 0.65 and range as high as 0.9. This is because many types of development (e.g., high-intensity urban development, roads) have a more severe on-site impact than the other indicators. However, except for urban development, most features associated with the development indicator are highly discrete and localized and usually not readily visible at the scale of the ecological status maps; although not visible at this scale, they are nonetheless pervasive throughout the ecoregion. In particular, effects of many of the very small, local areas of development, or small linear features (e.g. dirt roads) are not readily visible at the scale of the development indicator maps.

**Fire Regime Departure Indicator**
The fire regime departure indicator is an indirect measure of fire regime across the CE's estimated distribution. It is based on the Vegetation Condition Class (VCC) dataset produced by Landfire, which was developed to compare historical reference conditions with current conditions for individual ecological system types. Landfire VCC is calculated based on changes to species composition, structural stage, and canopy closure, and derived by comparing expected (historic) proportions of structural stages with current proportions for the individual ecological system. This then results in a ranking of departure from expected historic range of variability, which can be interpreted as "how has the disturbance regime (for the REA purposes and relevant to this ecoregion: fire regime) changed from its historical variability for this individual CE." Two departure categories, *Severe Vegetation Departure* and *Moderate Vegetation Departure*, were used in the status assessments for this REA and are displayed in the fire regime maps. The two departure categories were assigned varying site intensity values between 0 and 1, depending on how fire regime is understood to affect the species CE. *Severe Vegetation Departure* was usually assigned a site intensity value of 0.65 for species CEs, and *Moderate Vegetation Departure* was usually assigned a higher value of 0.75, reflecting the expected lesser degree of impact.

In addition, recent severe burns are an important indicator of habitat condition for several species. Burn severity data (reflecting burns documented in the 15-year period from 1997 to 2011) were compiled from the Monitoring Trends in Burn Severity website (MTBS), and two classes of severity were used: severe, and moderately severe. Burn severity was used for the following species (Table F-2, Table F-3):

- Chiricahua leopard frog because fire reduces upland cover in habitats and their habitat can be silted in and lost due to post-fire erosion events (USFWS 2011)
- desert box turtle because fire reduces cover in upland habitats and protective cover and food in riparian habitats
- grassland bird assemblage because fire can temporarily destroy habitat for certain grassland bird species and can alter habitat composition

Site intensity values assigned ranged from 0.5 to 0.8 for moderate burn severity fire and 0.4 to 0.7 for high severity fire (Table F-4).

**Invasives Species Indicator**
The invasive species indicator serves as an indirect (stressor-based) measure of vegetation composition, by measuring the cover of invasive species. It is based on a combination of two Integrated Landscape Assessment Project (ILAP, http://westernlandscapesexplorer.info/IntegratedLandscapeAssessmentProject) models of percent cover of 1) non-native grasses and forbs and 2) native woody increasers (mesquite). For each of these, the ILAP data included canopy cover on a continuous scale from 0% to over 90%; for the response models, these continuous variables were broken into three classes of cover (see Table F-4). Each of the three classes was assigned site intensity values between 0 and 1 as shown in Table F-4. Higher values correspond to limited ecological impact, and lower values correspond to greater impact to the CE.

## Other Species-Specific Variations
Due to differences in KEAs for individual species, there are differences in site intensity and distance decay values assigned in the response models for the various CEs. Because pronghorn are particularly sensitive to density of mesquite due to sight-line requirements (Byers 1997, O'Gara & Yoakum 2004) and due to their effects on forage quality, the site intensity values assigned for terrestrial invasives and mesquite cover are lower than those for other species, ranging from 0.55 to 0.75 and 0.3 to 0.7, respectively. Coues white-tailed deer were assigned a higher (better) site intensity value for agriculture

than other species due to their lower sensitivity to agricultural development and ability to utilize a variety of forage.

Desert bighorn avoid low visibility areas with dense vegetation (Hall 1981, Hansen 1980, Ostermann et al. 2001, Risenhoover and Bailey 1985, Shackleton 1997, USFWS 2000). They are known to be highly affected by altered fire regimes that have resulted in the loss of escape terrain due to the increased woody vegetation in areas and reduced sight lines. Consequently, the site intensity value assigned for Fire Regime Departure was lower than for other species: 0.4 for severe departure and 0.6 for moderate departure. The site intensity for agriculture was 0.2 for bighorn because of their very specific forage and habitat needs; if agriculture is present, appropriate forage and habitat would likely be largely obliterated.

Because box turtles have limited mobility (ability to climb over larger barriers or travel long distances around them) the site intensity value assigned for infrastructure (border barrier – pedestrian) was 0.2 as compared to 0.4 for other, more mobile species.

For Chiricahua leopard frog, because of the high mobility of high-impact aquatic invasive species (e.g. American bullfrog, (*Lithobates catesbeianus*), aquatic invasives were assigned a distance decay of 1200 m (0.75 miles). The recent burn severity indicator includes a distance decay of 1600 meters (1 mile) to capture the potential loss of habitat from post-fire erosion events in watersheds where habitat is present. Water use is an indicator for the magnitude of water consumption by people, agriculture, and industry per unit area; and tracks the consumption of surface and ground water together. Both diversions of surface water and pumping of ground water from an aquatic or wetland CE are detrimental to Chiricahua leopard frog habitat. Data available was total water volume used per year, summarized by groundwater basin in AZ and by county in NM. Consequently, the present assessment standardized the water use data by converting the annual rate of consumption in each spatial unit (groundwater basin or county) to volume per year per unit of surface area (see Figures E-2 and E-3 in Appendix E). All 30 m pixels that fall within each groundwater basin or county were assigned the same water use impact score, representing the total water use for that area. The assessment of water use also does not track consumption of water imported from other ecoregions, such as from the Colorado River, which does not take water away from aquatic CE occurrences within the MAR ecoregion. See **Appendix E Aquatic Ecological Systems: Conceptual Models and Ecological Status** for further details about the water use data and its application.

**Table F-4. CE response model values for each species CE.** For each pair of CE columns, site intensity values (abbreviated as "Site Int.") are shown on the left and distance values on the right. Site intensity is a unitless value; distance is listed in meters. Site intensity values range from 0.0 - 1.0 and are relative to each other. Site intensity values reflect how much an activity (as reflected in the indicator) removes ecological status of the CE. A value of 0.05 removes 95% of the status, 0.5 removes 50%, 0.7 30% and so on. Where two or more activities occur within the same pixel, the intensity values were multiplied together. Note that the Vista software requires a minimum distance of 10 meters as an input for distance decay, but because the actual pixel size for analysis is 30 meters, no distance effect is calculated in the model. Where site intensity and distance are blank, that land use or other feature was not assessed for that species.

| | Pronghorn | | Coues White-tailed Deer | | Desert Bighorn Sheep | | Desert Box Turtle | | Chiricahua Leopard Frog | | Grassland Bird Assemblage | | Bat Assemblage | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Indicator** | **Site Int.** | **Distance** | **Site Int.** | **Distance** | **Site Int.** | **Distance** | **Site Int.** | **Distance** | **Site Int.** | **Distance** | **Site Int.** | **Distance** | **Site Int.** | **Distance** |
| **Fire Regime Departure** | | | | | | | | | | | | | | |
| Moderate Departure | 0.75 | 10 | 0.75 | 10 | 0.6 | 10 | 0.75 | 10 | | | 0.75 | 10 | 0.75 | 10 |
| Severe Departure | 0.65 | 10 | 0.65 | 10 | 0.4 | 10 | 0.65 | 10 | | | 0.65 | 10 | 0.65 | 10 |
| **Recent Burn Severity** | | | | | | | | | | | | | | |
| Moderate Severity | | | | | | | 0.8 | 10 | 0.5 | 1600 | 0.8 | 10 | | |
| High Severity | | | | | | | 0.7 | 10 | 0.4 | 1600 | 0.7 | 10 | | |
| **Development** | | | | | | | | | | | | | | |
| **- Infrastructure** | | | | | | | | | | | | | | |
| Border Barrier - Pedestrian | 0.4 | 10 | 0.4 | 10 | 0.4 | 10 | 0.2 | 10 | 0.1 | 10 | 0.5 | 10 | 0.5 | 10 |
| Border Barrier - Vehicle | 0.7 | 10 | 0.7 | 10 | 0.7 | 10 | 0.7 | 10 | 0.6 | 10 | 0.6 | 10 | 0.6 | 10 |
| Communication Towers | 0.3 | 10 | 0.3 | 10 | 0.3 | 10 | 0.3 | 10 | 0.3 | 10 | 0.3 | 10 | 0.3 | 10 |
| Below-Ground Corridors | 0.7 | 10 | 0.7 | 10 | 0.7 | 10 | 0.7 | 10 | 0.7 | 10 | 0.7 | 10 | 0.7 | 10 |
| Above-Ground Corridors | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 |
| **- Transportation** | | | | | | | | | | | | | | |
| Dirt & 4-wheel Drive Roads | 0.7 | 10 | 0.7 | 10 | 0.7 | 10 | 0.7 | 10 | 0.7 | 10 | 0.7 | 10 | 0.7 | 10 |
| Local/Rural/ Private Roads | 0.2 | 10 | 0.2 | 10 | 0.2 | 10 | 0.2 | 10 | 0.2 | 10 | 0.2 | 10 | 0.2 | 10 |

| | Pronghorn | | Coues White-tailed Deer | | Desert Bighorn Sheep | | Desert Box Turtle | | Chiricahua Leopard Frog | | Grassland Bird Assemblage | | Bat Assemblage | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Indicator** | **Site Int.** | **Distance** | **Site Int.** | **Distance** | **Site Int.** | **Distance** | **Site Int.** | **Distance** | **Site Int.** | **Distance** | **Site Int.** | **Distance** | **Site Int.** | **Distance** |
| Primary Highways w/ Limited Access | 0.05 | 10 | 0.05 | 10 | 0.05 | 10 | 0.05 | 10 | 0.05 | 10 | 0.05 | 10 | 0.05 | 10 |
| Primary Highways w/o Limited Access | 0.05 | 10 | 0.05 | 10 | 0.05 | 10 | 0.05 | 10 | 0.05 | 10 | 0.05 | 10 | 0.05 | 10 |
| Airstrips | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 |
| Railroads | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 |
| **-Mining & Landfills** | | | | | | | | | | | | | | |
| High Impact Mines/Landfills | 0.05 | 10 | 0.05 | 10 | 0.05 | 10 | 0.05 | 10 | 0.05 | 10 | 0.05 | 10 | 0.05 | 10 |
| Medium Impact Mines/Landfills | 0.6 | 10 | 0.6 | 10 | 0.6 | 10 | 0.6 | 10 | 0.6 | 10 | 0.6 | 10 | 0.6 | 10 |
| Low Impact Mines/Landfills | 0.9 | 10 | 0.9 | 10 | 0.9 | 10 | 0.9 | 10 | 0.9 | 10 | 0.9 | 10 | 0.9 | 10 |
| **-Energy** | | | | | | | | | | | | | | |
| Geothermal Energy | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 |
| Wind Energy | 0.8 | 10 | 0.8 | 10 | 0.8 | 10 | 0.8 | 10 | 0.8 | 10 | 0.8 | 10 | 0.8 | 10 |
| Solar Energy | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 |
| Oil & Gas Wells | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 |
| **- Recreation** | | | | | | | | | | | | | | |
| Trails - Hiking/ Biking/Horse | 0.9 | 10 | 0.9 | 10 | 0.9 | 10 | 0.9 | 10 | 0.9 | 10 | 0.9 | 10 | 0.9 | 10 |
| **- Agriculture** | | | | | | | | | | | | | | |
| Agriculture | 0.3 | 10 | 0.6 | 10 | 0.2 | 10 | 0.3 | 10 | 0.3 | 10 | 0.3 | 10 | 0.1 | 10 |
| **- Urbanization** | | | | | | | | | | | | | | |
| Low Density Development | 0.6 | 10 | 0.6 | 10 | 0.6 | 10 | 0.6 | 10 | 0.6 | 10 | 0.6 | 10 | 0.6 | 10 |
| Medium Density Development | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 | 0.5 | 10 |
| High Density Development | 0.05 | 10 | 0.05 | 10 | 0.05 | 10 | 0.05 | 10 | 0.05 | 10 | 0.05 | 10 | 0.05 | 10 |
| **Invasives** | | | | | | | | | | | | | | |
| Aquatic Invasives - High Impact Species | | | | | | | | | 0.7 | 1200 | | | | |

| | Pronghorn | | Coues White-tailed Deer | | Desert Bighorn Sheep | | Desert Box Turtle | | Chiricahua Leopard Frog | | Grassland Bird Assemblage | | Bat Assemblage | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Indicator** | **Site Int.** | **Distance** | **Site Int.** | **Distance** | **Site Int.** | **Distance** | **Site Int.** | **Distance** | **Site Int.** | **Distance** | **Site Int.** | **Distance** | **Site Int.** | **Distance** |
| Aquatic Invasives - Low Impact Species | | | | | | | | | 0.7 | 10 | | | | |
| Terrestrial Invasives - Low Cover | 0.75 | 10 | 0.85 | 10 | 0.85 | 10 | 0.85 | 10 | | | 0.85 | 10 | 0.85 | 10 |
| Terrestrial Invasives - Medium Cover | 0.65 | 10 | 0.75 | 10 | 0.75 | 10 | 0.75 | 10 | | | 0.75 | 10 | 0.75 | 10 |
| Terrestrial Invasives - High Cover | 0.55 | 10 | 0.65 | 10 | 0.65 | 10 | 0.65 | 10 | | | 0.65 | 10 | 0.65 | 10 |
| **Mesquite Cover** | | | | | | | | | | | | | | |
| Mesquite - Low Cover | 0.7 | 10 | 0.85 | 10 | 0.85 | 10 | 0.85 | 10 | | | 0.85 | 10 | 0.85 | 10 |
| Mesquite - Medium Cover | 0.5 | 10 | 0.75 | 10 | 0.75 | 10 | 0.7 | 10 | | | 0.7 | 10 | 0.7 | 10 |
| Mesquite - High Cover | 0.3 | 10 | 0.65 | 10 | 0.65 | 10 | 0.6 | 10 | | | 0.6 | 10 | 0.6 | 10 |
| **Water Use** | | | | | | | | | | | | | | |
| Total Water Use - Low | | | | | | | | | 0.95 | 10 | | | | |
| Total Water Use - Medium | | | | | | | | | 0.8 | 10 | | | | |
| Total Water Use - Medium-High | | | | | | | | | 0.7 | 10 | | | | |
| Total Water Use - High | | | | | | | | | 0.5 | 10 | | | | |

## Overall Ecological Status Scoring

An overall "full" scenario (all KEA indicator scenarios combined into one) and associated overall ecological status map were also generated for each CE to provide overall CE status; however, such products typically beg the question of which indicators are driving the status at different locations. Therefore, as described above, the individual KEA indicator scenarios that represent relevant indicators (i.e., Development, Fire Regime Departure, and Invasive Species) were also assessed individually to illuminate their effects and inform understanding and potential management action.

# Considerations and Limitations for All Species

As described in Appendix B (**Ecological Status Assessment Technical Approach** section), geospatial modeling always introduces assumptions and abstractions of actual ecosystem processes and CA effects. The many factors that can be observed and measured in the field cannot be fully captured with existing data and geospatial modeling. While the geospatial results can be field tested to some degree and calibrated to field observations, there will not be a one-to-one comparability between the KEAs and indicators identified in the CE conceptual models and what can be assessed with existing data. This methodology also does not model interactions between CAs, such as an increase in the distribution or intensity of one CA resulting from the presence or effects of another CA. However, in some cases the inputs used for the MAR REA (e.g., fire condition) are based on more complex models that do incorporate such interactions. In addition, it is important to understand that some CAs are indicative of a current potential for impacts on CEs, rather than the actual current extent of the CA; for example, the invasives data from ILAP is a model predicting the likelihood of the presence of invasive species, rather than an actual mapped distribution of them.

Although ILAP had modeled data for percent cover of exotic invasive herbs, the ILAP team notes that it is a model with moderate uncertainty due to the lack of field-based input data for known locations (and cover) of invasive plants. The ILAP model for mesquite density/cover is a better model than that for invasive exotic herbs, because 1) there are more field-based locations for known occurrences of mesquite; and 2) the input data were vegetation sampling plots, which include percent cover estimates and not simply presence/absence. However, both of these datasets are modeled predicted distributions of the two categories of invasives, not actual mapped distributions that have been field-verified. Outside of the ILAP data, there is a lack of comprehensive (MAR-wide) current distribution or risk of occurrence data for exotic invasive plants. This is an important data gap; there are some efforts by local groups (e.g. Southern Arizona Buffelgrass Coordination Center, http://www.buffelgrass.org/SABCC) to develop spatial data for invasives but these are somewhat local in scale, and there do not appear to be any ecoregion-wide comprehensive databases compiled for the MAR.

The Landfire Vegetation Condition Class (VCC) dataset is not a direct measure of fire risk or fire regime departure from expected historical range of variability. It was developed to compare historical reference conditions with current conditions for an individual ecological system type (Rollins et al. 2007). It provides a categorized measure of the difference between current vegetation type and structure, and estimated vegetation type (Biophysical Settings, BpS) and structure from the time just prior to European settlement. It is calculated based on changes to species composition, structural stage, and canopy closure, and derived by comparing expected (historical) proportions of structural stages with current proportions (Rollins et al. 2007) within sufficiently large summary landscape units to adequately represent the historical conditions versus current conditions. Landfire VCC calculations are done within

variable size watersheds (4th, 5th or 6th level watersheds), depending upon the fire regime group to which each vegetation type (BpS) is assigned.

Therefore, the results from the fire regime indicator should not be over-interpreted relative to current fire regime conditions in species CE habitats; rather it provides a useful overview of where disturbance regimes in general are different from the expected historical regimes. Those differences can be due to a number of factors, such as impacts of drought and warmer temperatures over the past 20 to 30 years, increases in invasive grasses that introduce a regime of frequent fires to desert scrub ecosystems, the invasion of mesquite into upland grasslands due to the effects of many decades of land use practices, or effects of grazing or other activities that might alter the structural and compositional characteristics of the ecosystem.

Landscape connectivity is an important consideration for the wildlife species assessed here. At the local scale, permeability is critical for species to avoid predators and access food and water. Bighorn sheep, Coues white-tail deer, pronghorn and Chiricahua leopard frog rely on available surface water for survival or completion of their life cycle; therefore, the ability to move between water resources and forage resources is critical. For terrestrial mammals such as bighorn sheep, Coues white-tailed deer, and pronghorn, seasonal and even daily movements between areas of foraging and breeding are also important (NatureServe 2013a, USFWS 2008).

At the larger landscape scale, connectivity is critical for species population health through gene flow, dispersal, and seasonal migration. Daily and seasonal migratory movement between habitat patches is essential for adaptability to changing forage and water availability, predator avoidance, and gene flow (AZGFD 2011, NatureServe 2013a, USFWS 2008). Ability to move across the landscape in response to fire, drought, and shifts in resource availability is important for climate change adaptation.

Landscape connectivity for species between habitats and populations was not included in the spatial assessment for species in the MAR. Although this is not included in the spatial assessments, it is an important management consideration for species at the landscape level. The efforts of state fish and game agencies, working with the Western Governors Association, to develop connectivity models, wildlife corridors and assessments of crucial habitats for individual species are important (State Wildlife Agencies of the Western U.S. 2014); see for example http://westgovchat.org/about, http://westgovchat.org/map, http://habimap.org/habimap/, and http://nmchat.org/).

Grassland birds are of particular interest and concern in this ecoregion. Treating the diverse suite of grassland birds as a single CE assemblage has limitations. Each of the species found in this assemblage has very different characteristics and needs, and an assessment that is pertinent to them all within the constraints of available data and a rapid assessment, necessarily results in generalized findings that will not shed light on the status of any individual bird species. In addition to different habitat needs (e.g., grass height, structure, grass species composition, amount of bare ground, etc.) some of these species are resident species, while others are migrants here only in the winter, and others migrants here only in the breeding season. Even at the general levels for this assessment, the species in the assemblage do not respond the same way to development or other human disturbances, fire, or non-native or invasive grasses/forbs or shrubs.

# Species and Species Assemblages: Conceptual Models and Ecological Status

# Mammals

## *F-1* Pronghorn *(Antilocapra americana)*

### F-1.1 Conceptual Model

#### F-1.1.1 Classification Comments

Endemic to North America, pronghorn (*Antilocapra americana*) are the last surviving genus in the family Antilocapridae. Historically debated to be part of the Bovidae family, recent genetic studies have confirmed that they belong in Antilocapridae (Janis and Scott 1987; Jones et al. 1992; Wilson and Reeder 1993, 2005), and are more closely related to Giraffidae than to Cervidae or Bovidae despite their antelope-like appearance (Hernandez et al 2005; Murray 2006; Janis 2000). Pronghorn are unique in having horns made of a keratinous sheath over a bony core, similar to bovids, that they shed and re-grow each year, similar to antlers of cervids (AZGFD 2011).

Five subspecies of the North American pronghorn are currently recognized: the American pronghorn (*A. a. americana*, Ord 1815); the Mexican or Chihuahuan pronghorn (*A. a. mexicana*, Merriam 1901); the Baja California or Peninsular pronghorn (*A. a. peninsularis*, Nelson 1912); the Oregon pronghorn (*A. a. oregona*, Bailey 1932); and the Sonoran pronghorn (*A. a. sonoriensis*, Goldman 1945) (ITIS, accessed 18 January 2013; USFWS 2013; O'Gara and Yoakum 2004). The validity of these subspecies and their separate ranges are not fully understood, and are largely defined by population variations in color, size, and location (O'Gara and Yoakum 2004). Genetic studies of North American pronghorn populations led Lee et al. (1994) and Cockrum (1981) to question subspecies status based on these minimal physical and geographical differences; particularly between the American pronghorn and Chihuahuan pronghorn occurring in the Madrean Archipelago ecoregion and western Texas, and between the American pronghorn and Oregon pronghorn populations in Oregon, California, Idaho, and Nevada (Lee et al. 1994; Lee 1992). These studies and other more recent mitochondrial DNA analyses lend support to the identification of clines, rather than distinct subspecies, for pronghorn populations displaying minor phenotypic or genetic differences that reflect adaption to the diverse habitats and environments in which pronghorn range (O'Gara and Yoakum 2004). The genetic integrity of these taxonomic subspecies are further complicated by translocation and reintroduction efforts that began in the early twentieth century and continue today, in an attempt to bring back declining or extirpated populations in the region (USFWS 2013). Currently, subspecies distinctions remain standard for pronghorn populations throughout North America.

Two subspecies are addressed in the Madrean REA, the American pronghorn (*A. a. americana*, Figure F-1 and Figure F-2) and the Mexican or Chihuahuan pronghorn (*A. a. mexicana*), unless otherwise noted. References to the Sonoran pronghorn (*A. a. sonoriensis*) are occasionally used to understand unique adaptation responses relevant to this arid borderlands ecoregion.

**Figure F-1. American pronghorn (*A. a. americana*), Animas Valley, Arizona.** Photo © 2012 Sky Island Alliance/ Melanie Emerson.

**Figure F-2. American pronghorn (*A. a. americana*), Las Cienegas National Conservation Area, Arizona.** Photo © 2012 Sky Island Alliance/ Tim Cook.

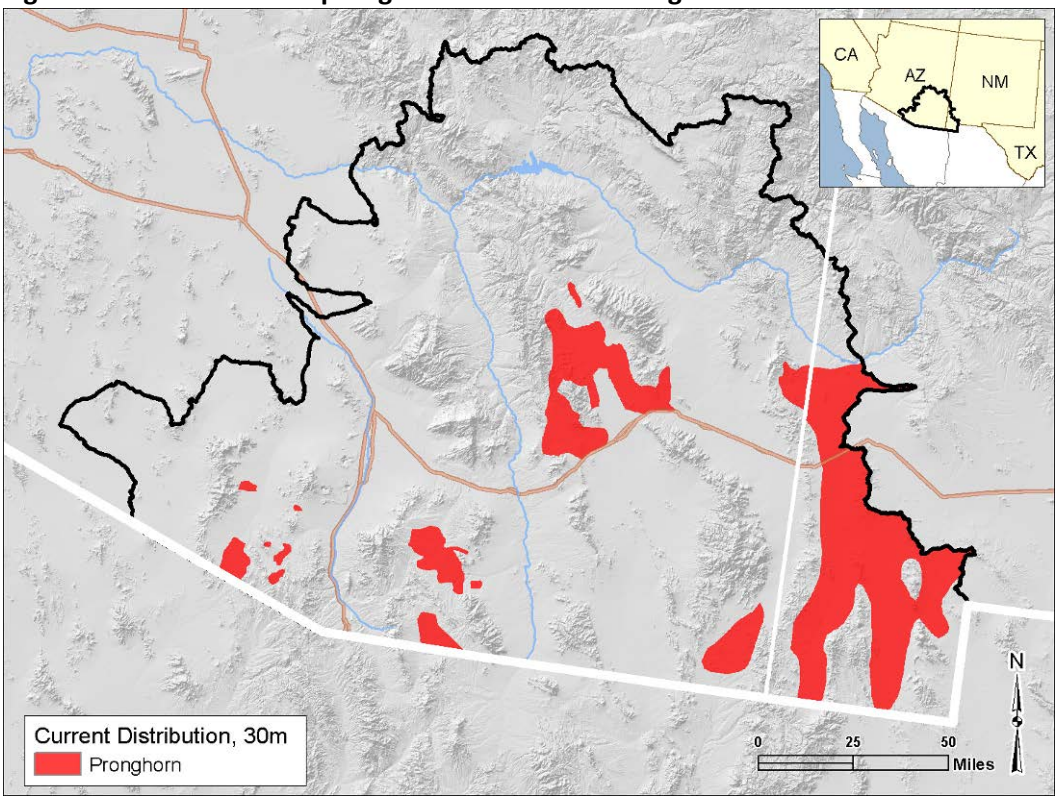
### F-1.1.2 Biology and Distribution Status

#### *F.1.1.2.1 Distribution*

*North America*: Pronghorn once ranged from southern Canada to northern Mexico; and from the Mississippi River and Gulf Coast of Texas to California and the Pacific Ocean. Today pronghorn populations are primarily distributed along the "Spine of the Continent" (Hannibal 2012) in southeastern Oregon, southern Idaho, southern Alberta and Saskatchewan, Montana, and western North Dakota to Arizona, western Texas, and northern Mexico (Byers 1997; O'Gara and Yoakum 2004), and were unsuccessfully introduced to Lanai, Hawaii in 1958 (NatureServe 2007).

*Madrean Archipelago ecoregion*: Mixed populations of the American and Chihuahuan pronghorn subspecies occur within the Madrean Archipelago ecoregion today (AZGFD 2011; Howard 1995; Lee at al 1994; Yoakum 1980). Historically, Sonoran pronghorn was distributed from southern Arizona to the desert plains of central and western Sonora, Mexico (CONANP 2009; Leopold 1959). The Chihuahuan pronghorn's distribution encompasses southeastern Arizona, southwestern New Mexico, and portions of west Texas, extending south in Mexico to Chihuahua, Durango, Coahuila, portions of Nuevo Leon and Tamaulipas, and as far south as the state of Hidalgo (CONANP, 2009). These populations consist of small herds scattered throughout their range. Within the Madrean Archipelago ecoregion of Arizona and New Mexico, seven population localities are currently managed and monitored (AZGFD 2013). In the New Mexico portion of the Madrean Archipelago ecoregion the Chihuahuan pronghorn occurs in the grassland areas of New Mexico Game Management Units 23, 24, 25, 26, and 27. In 2009, the range of the Chihuahuan pronghorn in northern Mexico principally included localities of Chihuahua, Mexico, including La Perla, La Gregoria, San Luis, Terraceño, El Sueco-Moctezuma, Janos-Ascención and Coyame, as well as Valle de Colombia and Rancho El Novillo near Coahuila, Mexico (CONANP 2009; Pallares 1999). The distribution of pronghorn populations in southeastern Arizona appear to be strongly associated with identified Wildlife Linkage zones in that portion of the region (Arizona Wildlife Linkage Workgroup 2006).

**Figure F-3. Distribution of pronghorn in the MAR ecoregion.**




### *F.1.1.2.2 Population*

Chihuahuan pronghorn subspecies populations occurring in Sonora and Chihuahua have CITES I status and are listed as endangered by the Mexican government; the North American pronghorn is listed by IUCN as Least Concern, Ver. 3.1 (Hoffman et al 2008). Recent aerial pronghorn surveys conducted by New Mexico indicate a declining population within the ecoregion due to low recruitment rates. Arizona's statewide pronghorn population status is based on the sum of population estimates gathered from

game management units; these are produced with computer simulation models using data gathered from aerial surveys and annual hunter-reported harvest data. Fixed-wing aircraft surveys are conducted, prior to hunting season, to collect age and sex ratios and population estimates using a simultaneous double count method (AZGFD 2011). An adult pronghorn population of 8,000 was estimated for Arizona (not including Indian reservations), occupying approximately 21,000 square miles of habitat; this population is projected to increase steadily 2 to 4 percent per year over the next ten years (AZGFD 2011). The population trend in Mexico and the southwest United States is in decline, despite the projected upward trend for Arizona, largely due the recent severity of droughts and winters (Hoffman et al 2008; O'Gara 1999). Southern Arizona populations are periodically augmented, and in some cases re-established, with translocated animals from northern Arizona and Texas (AZGFD 2011).

In a series of aerial surveys conducted in 2011, a total of 165pronghorn were counted from the seven herds managed in the Madrean Archipelago ecoregion (AZGFD 2011); as of 2013 this number is between 286 – 407, despite significant decline due to drought in three herds since 2007 (Caroline Patrick-Birdwell, environmental consultant, pers. communication). Seven pronghorn herd populations are currently managed by the Arizona Game and Fish Department. The most recent 2013 population status for these herds is as follows (Caroline Patrick-Birdwell, environmental consultant, pers. communication):

*Altar Valley Population:* 11-20 animals (no management goal in the near future; this population has experienced close to zero fawn recruitment in recent years), located in the Altar Valley and Buenos Aires National Wildlife Refuge, near Sasebe and Arivaca, Arizona (within Arizona Hunt Units 36A, 36B and 36C). Population trend is declining.

*San Rafael Population:*9 animals (management goal is 100), located in the San Rafael Valley and Fort Huachuca Military Reservation near Lochiel, Arizona and the U.S.-Mexico border (Arizona Hunt Units 35A and 35B). Population trend has been declining since 2007.

*Las Ciénegas Population:* 60-80 animals (management goal is 50-100), located at Empire Ranch and the Las Ciénegas National Conservation Area northeast of Sonoita, Arizona (Arizona Hunt Unit 34B).

*Elgin-Rosetree Population*: 19 animals (management goal is 100), located south of the Las Ciénegas Population and bounded on the south by the Huachuca and Patagonia Mountains and Fort Huachuca Military Reservation. Population trend has been declining since 2007.

*North Sulpher Springs Valley-Bonita Population:* 50-80 animals (management goal is 100-200), located in the Sulpher Springs Valley and Allen Flat area north of Interstate-10 and Dragoon, Arizona and west of Highway 191 (Arizona Hunt Units 31 and 32). Population trend has been declining since 2007.

*San Bernardino Valley Population:* 125-175 animals (management goal reached), located in the San Bernardino Valley northeast of Douglas, Arizona (Arizona Hunt Unit 30A).

*Lordsburg Plains Population:* 12-24 animals within Arizona(no management goal) representing an estimated 10% of the total pronghorn population (120-240 animals) on the Lordsburg plains, located in the Day Ranch area of the Animas Valley, spanning across the border of Arizona and New Mexico, east of the Peloncillo Mountains and north of Interstate-10 (Arizona Hunt Unit 28), and also bounded by the Gila River and Burro Mountains (New Mexico Hunt Units 27 and 23). This population moves fluidly between Arizona and New Mexico.

#### *F.1.1.2.3 Reproduction*

*Breeding Systems:* Male pronghorn will exhibit different breeding systems within the same population and this behavior is a direct result of the availability and location of resources, density of the population, and the ratio of males to females in a group (Maher 2000). Territorial breeding behavior (male defense and competition of a defined location containing water, succulent vegetation and the females within this area) occurs when quality resources are isolated or scarce. Dominant breeding behavior occurs when resources uniformly available on the landscape, resulting in the formation of herds of females and multiple males, with the most dominant males in the hierarchy doing most of the breeding. Harem breeding behavior occurs when resources are widely dispersed, population levels are low, or the sex ratio is skewed (1 male to 10 or more females); in this case, a single breeding male will defend a harem of females without regard to a defined territory.

*Gestation:* The gestation period for pronghorn is between 245 and 255 days. Ova are able to develop for up to a month after fertilization before implantation occurs (San Diego Zoo Global Library 2009; O'Gara and Yoakum 2004; Byers 1997; O'Gara 1978). Females may reabsorb embryos under conditions of environmental stress.

*Litter size:* One to two fawns, per breeding female per year. Twins are the norm; occasionally less dominate females will give birth to a single fawn.

*Breeding Season:* Pronghorn in southern habitats breed in the fall, between the months of September and October. Breeding behavior may begin in late summer, as early as July (San Diego Zoo 2009; Byers 1997; O'Gara 1978; O'Gara & Yoakum 2004). Northern pronghorn populations begin their breeding season earlier.
*Birthing Season:* Depending on habitat and environmental conditions, pronghorn generally give birth in the Madrean Archipelago ecoregion between June and July; this differs from northern populations that typically give birth in May and June. Pronghorn exhibit synchronous birthing, with gravid females in a herd giving birth within the same 10 day period (San Diego Zoo 2009; Byers 1997; O'Gara 1978; O'Gara & Yoakum 2004).

*Life history:* Pronghorn weigh 7-9 pounds at birth, which is approximately 18% of the adult female's weight. Young are weaned by 4 to 5 months of age but do not reach their full adult weight for about 4.5 years. Males reach sexual maturity in one year but may not have the opportunity to breed until they are dominant enough to compete with other males. Females are sexually mature at 16 months, but there are cases of females conceiving as young as 5 months (San Diego Zoo 2009; Byers 1997; O'Gara 1978; O'Gara & Yoakum 2004). In the wild, the lifespan of a pronghorn is less than 10 years.

### F-1.1.3 Habitat and Ecology

#### *F.1.1.3.1 Habitat*

Pronghorn have several specific habitat requirements needed for survival. Yoakum et al. (1996) and Jaeger and Fahrig (2004) defined the optimal habitat parameters for the North American pronghorn as elevation, terrain, landscape connectivity, distance from water, and vegetation. In 1995, the Arizona Game and Fish Department conducted a statewide analysis that quantified and ranked pronghorn habitat based on five variables: topographic ruggedness, vegetative structure and species richness, water availability, human disturbance, and fence density and structure (Ockenfels et al. 1996; Ockenfels et al. 2000).

*Elevation:* In Arizona and New Mexico pronghorn are found between 3,000-7,000 feet (914-2,134 meters) elevation, although northern Arizona herds have been documented as high as 10,000 feet (3,048 meters) in summer (AZGFD 2011). New Mexico has also documented several pronghorn populations at high elevations above 8,000 feet (2,438 meters) as high as 10,000 feet (3,048 meters).

*Terrain:* Yoakum et al. (1996) cites that pronghorn prefer open habitat, with flat or with rolling hills, in order to detect and avoid or escape approaching predators; pronghorn are associated with topographic ruggedness of less than 20% slope according to studies in Arizona (AZGFD 2011; Ockenfels et al. 1996; Ockenfels et al. 2000).

*Distance from accessible water:* Pronghorns require regular access to surface water and are usually found within a 4 mile (6.5 km) radius of an open water source (Yoakum et al, 1996). These include but are not limited to natural springs and seeps, human-made wildlife waters and cattle tanks, ponds, lakes, and perennial riparian corridors and streams.

*Vegetation:* Largely associated with grasslands, short-grass prairie, and shrub-steppe, preferably with vegetation ranging 5 to 30 in (13 to 76 cm) in height (Byers 1997, O'Gara & Yoakum 2004). Pronghorn are occasionally found in temperate desert habitat; deserts support less than 1% of the North American pronghorn population, and this consists largely of the Sonoran subspecies (San Diego Zoo 2009, Byers 1997, O'Gara & Yoakum 2004). Based on population distribution data from the Arizona Game and Fish Department (AZGFD 2013) and according to the biotic communities described in Brown and Lowe (1982), pronghorn herds in the Madrean Archipelago ecoregion occur primarily in semi-desert grasslands with the Sulphur Springs Valley population also overlapping the edges and open areas of Madrean oak woodland habitat.

#### *F.1.1.3.2 Landscape Connectivity*

Pronghorn have become the poster child for landscape connectivity and large-scale wildlife corridor conservation (Hannibal 2012), as they require expansive areas and connected habitats to thrive. One population in the Wyoming Greater Yellowstone region is known for having the longest seasonal migratory pathway of any other terrestrial species in the continental U.S., and using the same 100 mile (160 km) route for more than 6,000 years (Hannibal 2012, San Diego Zoo 2009, Byers 1997, O'Gara & Yoakum 2004). In the Madrean Archipelago region these seasonal migrations are not as impressive and are limited within separate herd home ranges, but are essential to maintaining gene flow and finding quality forage and water sources in summer and winter. Barriers to movement result from fragmentation and habitat loss caused by human development, in the form of fencing, mining, urban sprawl, and roads, railroads and highways, among others (AZGFD 2011). In the Madrean Archipelago ecoregion, major barriers include the U.S. – Mexico border and Interstate-10. New Mexico Department of Game and Fish has partnered with the US Forest Service to identify and implement habitat projects designed to enhance historical pronghorn corridors between summer and winter range that has pinyon/juniper encroachment.

Pronghorn cross fences by passing underneath, rather than jumping over; barbed wire, woven wire or fences with the bottom wire lower than 20 inches (< 50cm) off the ground are barriers for pronghorn (Yoakum et al. 1996, Jaeger and Fahrig 2004). Fences become even greater barriers to pronghorn movement (and gene flow due to direct mortality) when they are placed too close to high-traffic roads or highways without a buffer zone that allows individuals to navigate one obstacle at a time (AZGFD 2011; Scott Sprague, AZGFD pers. communication). Pronghorn-friendly fencing must have a smooth

bottom wire at least 20 inches above ground level, or a bottom wire equipped with plastic pipe “goat bars” (AZGFD 2011).

#### *F.1.1.3.3 Food*

Pronghorn require a diversity of vegetation for forage and thrive with higher plant species richness. They are opportunistic herbivores; high-protein, succulent forbs make up the majority of their diet, along with shrubs (i.e. rabbit bush, sagebrush), grasses (i.e. bunchgrass, squirrel-tail) and cacti (*Opuntia* sp.*).* Grasses total approximately 12% of their diet (AZGFD 2011; Yoakum et al. 1996; O'Gara 1978). Availability of both grasses and shrubs is shown to be essential for fawn survival (AZGFD 2011). Pronghorn frequently change their forage preferences season to season, and will seek the most nutritious and succulent forage species available (San Diego Zoo 2009, Byers 1997, Hansen & Clark 1977, O'Gara 1978, O’Gara & Yoakum 2004). In harsh winters, northern Arizona populations had a higher survival rate with more shrub forage available than herds who fed largely on grass (AZGFD 2011).

#### *F.1.1.3.4 Phenology*

Adult and juvenile pronghorn activity patterns are circadian, crepuscular and diurnal (NatureServe 2007, Einarsen 1948, Kitchen 1974). Pronghorn have crepuscular feeding habits, with most foraging activity occurring at dawn and dusk; however, the majority of their time throughout the day is spent foraging, ruminating and sleeping or resting. Approximately 40 to 60 percent of their day is spent feeding, and they sleep without regularity and for frequent, short periods (San Diego Zoo 2009, Byers 1997, Hansen & Clark 1977, O'Gara 1978, O’Gara & Yoakum 2004). Activity patterns for pronghorn are variable, and influenced by region, weather, season, and also by herd dynamics. Some pronghorn populations migrate long distances between summer and winter feeding grounds, although this does not appear to be the case with southern herds that occur in warmer habitats where quality forage is available year-round. Northern herds have shown daily foraging movements that vary greatly, from 0.1 to 0.8 km in the spring and summer to 3.2 to 9.7 km in the fall and winter. In the Madrean ecoregion, home ranges are anywhere between 20 to 40 square miles (AZGFD 2011), and herds travel seasonally between fawning grounds and winter and summer feeding grounds within each home range (Caroline Patrick-Birdwell, environmental consultant, pers. communication).

#### *F.1.1.3.5 Predators*

Coyotes (*Canis latrans*), bobcats (*Lynx rufus*), mountain lions (*Puma concolor*), golden eagles (*Aquila chrysaetos*), gray wolves (*Canis lupus*), and humans are all predators of the pronghorn; the only other predator it evolved with is the jaguar (*Panthera onca*). Fawn predation is of most concern; due to their synchronous birthing behavior, small populations may lose the majority of fawns to heavy predation occurring in a short time period (AZGFD 2011).

#### *F.1.1.3.6 Competitors*

Pronghorn compete with other browsers and grazers on the landscape. O’Gara (1978) calculates that 38 pronghorn can survive on the same amount of forage that is needed for a single cow. Research on the Sonoran pronghorn suggests that the effects of livestock grazing on condition of forage habitat can be a significant stressor on pronghorn populations in combination with drought conditions (USFWS 2002).

#### *F.1.1.3.7 Demographics*

Sparse or small populations can result in random variations in sex ratios, age distributions, and birth and death rates among individuals, which can cause fluctuations in population size and even lead to extinction (USFWS 2002). In very small isolated populations, males may have trouble finding females, leading to decreased recruitment (USFWS 2002). The composition of a herd can change hourly or day to day, forming loosely dispersed groups of 3 to 25 that keep in visual contact when food is readily

available, or forming large herds of 1,000 individuals that are made up of many groups that band together, most often in fall and winter in northern areas where predation is higher (Byers 1997, O'Gara 1990, O'Gara & Yoakum 2004). Dominance hierarchies are maintained by female bands, bachelor bands, and territorial male bands that contain more than one male (San Diego Zoo 2009).

#### *F.1.1.3.8 Disease and Parasites*

Pronghorn can suffer from hemorrhagic disease caused by epizootic hemorrhagic disease viruses and the Bluetongue virus, also known as catarrhal fever, which is a virus transmitted by *Culicoides* sp. during warm summer months. In Wyoming in 1976, Bluetongue virus killed nearly 3,200 pronghorn; and another 300 mortalities occurred in 1984 (Thorne et al. 1988). *Haemonchus contorutus* a stomach worm of sheep and cattle has impacted pronghorn in Texas (McGhee 1981, Wasel 2003). White-tail deer (*Odocoileus virginianus),* mule deer (*Odocoileus hemionus*), and bighorn sheep (*Ovis canadensis*) act as primary, unaffected, hosts that pass it on to other species through their feces; pronghorn ingest vegetation with gastropods, the intermediate host, that have become infected with the larvae of the nematode (Simmons et al. 2002). Hemorrhagic disease has been identified occasionally in free-ranging white-tail deer, mule deer and bighorn sheep in Arizona (Dubay et al. 2006).

### F-1.1.4 Change Agent Characterization

This section of the conceptual model presents a narrative description of the primary change agents and current knowledge of their effects on pronghorn populations. The main change agents identified are forms of altered dynamics: Livestock management, development, and climate change.

#### *F.1.1.4.1 Livestock Management*

Competition for food with grazing cattle is one stressor on pronghorn, most significantly during poor forage conditions, and heavy livestock use coupled with frequent drought periods reduces forbs during the growing season (AZGFD 2011, USFWS 2002). Livestock management acts as a change agent for pronghorn through increased competition for forage and water sources, and increased fencing, cattle guards, and other barriers on the landscape. Livestock management can provide additional water resources available for pronghorn, but only if it is not at cost to water availability (Yoakum et al. 1995) and wildlife friendly fencing may be used. In Mexico, where invasive forage such as buffelgrass (*Pennisetum ciliare*) has been subsidized and planted for livestock grazing, decreased plant diversity and changing fire regimes have detrimental effects. Loss of habitat and movement corridors from fences used to control livestock movements within pastures, between ranches and allotments, and along highways, and habitat degradation from long-term vegetative community changes due to livestock overuse and fire suppression greatly impact pronghorn populations (AZGFD 2011; USFWS 2002; Ockenfels et al. 1994). Cooperative livestock management practices can provide benefits and reduce stressors on pronghorn. New Mexico BLM has utilized various tools to initiate the restoration process within many of the desert grasslands across the southern portions of the state. This restoration process has included the reduction of species that have invaded these grasslands, including creosote and mesquite. This statewide initiative, known as Restore New Mexico, will culminate in the restoration of many desert grassland habitats which are home to pronghorn.

**Stresses:** increased competition, increased habitat fragmentation, change in vegetation structure, decreased plant diversity, decreased forage abundance, decreased water quality, increased soil compaction, habitat degradation, increased behavioral disruption, increased fire frequency and intensity.

**Responses:** decreased population size, reduced home range size, reduced migratory distance, reduced recruitment, increased mortality rate.

### *F.1.1.4.2 Development*

Human development influences pronghorn populations by directly affecting habitat and resources, and may fall into the categories of 1) military and border-defense related activity; 2) mining; 3) rural and urban development including canals; 4) transportation in the form of roads, railroads and highways; and 5) energy development including new transmission lines, and converting open space for solar and wind farming. The U.S. Fish and Wildlife Service (2011) cites that the difference between rates of decline north and south of the border for Sonoran pronghorn populations might be due to high levels of human disturbance on the U.S. side, from illegal immigration, smuggling, and law enforcement response to these activities (USFWS 2011); these stressors are also present in the Madrean Archipelago ecoregion where pronghorn occur. Border infrastructure, including fencing, lighting, roadways, and frequent vehicle traffic, pose a significant barrier to wildlife movement north and south of the border, and large scale mining can have a tremendous impact on ground water resources, water and air quality, and habitat availability. Rural and urban development has a direct impact on ground water levels and habitat resources, and transportation and energy development results in habitat fragmentation and direct mortality from vehicle collisions. Increased human development also infers increased human presence: increased access to more remote areas generated by increased road development may increase direct take from legal and illegal harvest, incidental road kill, and other disturbance. Brown et al (2012) determined that elk and pronghorn did not respond to noise levels associated with motorized human activity, which they suggested was due to the animal's inability to maintain constant responsiveness to these activities. Brown et al also suggested that although less responsiveness may be an investment in fitness, it could potentially decrease predator detection and increase human conflicts.

**Stresses:** reduced habitat/ open space, decreased water availability, increased habitat fragmentation, decreased water quality, habitat degradation, increased behavioral disruption.

**Responses:** decreased population size, reduced recruitment, decreased home range, reduced migratory distance, decreased gene flow/ decreased genetic health, increased mortality rate.

A note on the potential side effects of pronghorn management including sound disturbance: translocation of animals from other herds to augment populations with the goal of stabilizing populations and increasing hunting recreation opportunities (AZGFD 2011), leads to increasing hybridization between subspecies and occasional capture myopathy (mortality due to capture). In addition, pronghorn are routinely pursued by helicopter or fixed-wing aircraft for capture efforts and annual population counts, which may increase stress in the form of behavioral disruption, predator response, herd dispersal and occasional physical injury.

### *F.1.1.4.3 Climate Change*

Pronghorn rely greatly on specific habitat requirements to survive and thrive, which are affected by precipitation, temperature, and other climatic conditions and change through the seasons. Additional studies of these influences in consideration of a changing climate—including increasing temperature extremes, increased fire intensity and frequency, and drought—are needed, but some research outside the Madrean Archipelago ecoregion may provide insight to the responses that may be expected. Annual reproductive success in Sonoran Pronghorn has shown to depend heavily on winter rainfall (Heinz Center 2011), presumably because winter precipitation increases high nutrient forage and surface water resources in spring, a factor that also occurs in the similarly arid Madrean ecoregion. Depending on conditions, pronghorn are able to obtain some of the water that they need from their diet, but many pronghorn populations use water sources, particularly during dry periods. In the absence or reduction of succulent forage pronghorn have been recorded using water developments (O'Gara and Yoakum 1992;

Yoakum 1994). In desert environments, Sonoran Pronghorn, previously thought not to use free water have been detected drinking at water sites (Morgart et al. 2005). Free water does influence the distribution of pronghorn. In Arizona Sonoran pronghorns were typically found to be within 2-6 km of an open water source (Hughes and Smith 1990; Devos and Miller 2005) and fawns were found to bed within .4-.8 km of water (Ticer and Miller 1994). In other parts of the country, pronghorns have been shown to stop using water sources when forage moisture content exceeds 75 percent (Beale and Smith 1970). Decreased water resources and increased presence of predators at available water sites can expose pronghorn to greater risk of mortality; in addition, warmer temperatures and lower precipitation may lead to increased water drawdowns by human populations (Heinz Center 2011, SWCCN 2008). Some studies (Wilson and Krausman 2008) have speculated that fawns and juveniles may be susceptible to hyperthermia during the hottest period of the year (for temperatures reaching 109 $^{o}$F (43 $^{o}$C) or greater), leading to death. Extreme cold temperatures in combination with low forage quality and diversity can also cause direct mortality to pronghorn (AZGFD 2011). The National Drought Mitigation Center (2013) reports that the Madrean Archipelago ecoregion has been experiencing Severe to Extreme drought in the last year, and forecasts this condition will persist or intensify.

**Stresses:** increased change in vegetation structure, change in biotic community, decreased forage availability, decreased water availability, increased distance to water, increased predation, increased fire frequency and intensity, hyperthermia, hypothermia.

**Responses:** decreased population size, reduced recruitment, increased reliance on artificial water, increased daily foraging movements, increased migratory distance, and increased mortality rate.

### F-1.1.5 Ecological Status: Key Ecological Attributes and Indicators

This section of the conceptual model addresses Key Ecological Attributes and their potential indicators. The ecological status is a way of describing current status via criteria, functionality, or levels of attributes and asks if they are within the normal range of variation.

#### *F.1.1.5.1 Key Ecological Attributes*

Table F-5 identifies the key ecological attributes for this CE within the Madrean Archipelago ecoregion. A **key ecological attribute** of a focal ecological resource is a characteristic of the resource's biology, ecology, or physical environment that is critical to the resource's persistence in the face of both natural and human-caused disturbance. Alteration of such a characteristic beyond some critical range of variation will lead to the degradation or loss of the resource within decades or less. The KEAs table lists the identified key ecological attributes, with a brief definition, a rationale for why it is important for the CE, and a listing of stressors or change agents that might be affecting the key attribute.

**Table F-5. Key ecological attributes (KEA) of North American Pronghorn (*Antilocapra americana mexicana* and *A. a. americana)* in the Madrean Archipelago ecoregion.**Indicators for these KEAs can be used to determine the ecological status for this CE; see **Table F-2** for a list of the indicators assessed in this REA.

| ***KEA Class:*** **KEA Name** | **Definition** | **Rationale** | **Stressors** |
|---|---|---|---|
| ***Landscape Context:*** **Landscape and Habitat Condition** | The extent of suitable grasslands, short-grass prairie and shrub-steppe that is intact and unfragmented. | Pronghorn have specific habitat requirements including grasslands with low topographic ruggedness between 3,000—7,000 feet elevation and regular access to surface water. Pronghorn habitat and historical herd home ranges and dispersal movements extend north and south of the U.S.-Mexico border; pronghorn populations occurring in the Madrean ecoregion are directly affected by the physical barriers to gene flow, behavior disturbance, and habitat loss and destruction, among other stressors. | Habitat fragmentation is caused by livestock management (fences, cattle guards, and roads), mining, rural and urban development, transportation features including roads and railroads, energy transmission lines, canals and border infrastructure. Pronghorn habitat and historical herd home ranges and dispersal movements extend north and south of the U.S.-Mexico border; pronghorn populations occurring in the Madrean ecoregion are directly affected by the physical barriers to gene flow, behavior disturbance, and habitat loss and destruction, among other stressors. This can decrease pronghorn home range, recruitment, disrupt migration, decrease gene flow and genetic health and increase mortality. |
| ***Landscape Context:*** **Habitat and Corridor Permeability** | The permeability of the landscape to pronghorn movement based on the absence of barriers such as fences, roads and other developments. | Landscape permeability is a measure of connected habitat that can be modeled at various scales: at a fine/ local scale, permeability is critical to pronghorn ability to avoid predators and access food and water; at a larger landscape scale, it is critical for pronghorn population health through breeding between herds, dispersal, and seasonal migration. Daily and seasonal migratory movement between habitat patches is essential for pronghorn adaptability to changing forage and water availability, predator avoidance, and gene flow (AZGFD 2011). | Barbed wire, woven wire or fences with the bottom wire lower than 20 inches (< 50cm) off the ground are significant barriers for pronghorn movement (Yoakum et al. 1996, Jaeger and Fahrig 2004). Fences also cause direct mortality when placed too close to high-traffic roads or highways without a buffer zone that allow individuals to navigate one obstacle at a time (AZGFD 2011; Scott Sprague, AZGFD pers. communication). Pronghorn fencing must have a smooth bottom wire at least 20 inches above ground level, or a bottom wire equipped with plastic pipe "goat bars" (AZGFD 2011). Fencing that is not permeable for pronghorn restricts available habitat for populations to find forage, water, mates and to avoid predators, and influences gene flow and migratory behavior. Low or absent landscape permeability can also indicate locations of corridor bottlenecks and/or genetic isolation between populations that can lead to decreased genetic health, inbreeding and local extinction. This indicator can include a number of landscape or habitat features that decrease or alter movement patterns of the CE- fences, rough topography, roads, or other barriers to movement. |

| *KEA Class:* KEA Name | Definition | Rationale | Stressors |
|---|---|---|---|
| ***Size/Extent:* Population Condition** | The size and age and sex distribution of pronghorn populations. | Population size and demographic structure affects the viability and health of the population; demographics can affect breeding systems affecting genetic diversity. | Human development may impact population size and demography by reducing suitable habitat, direct disturbance and migration disruption. Other stressors include reduced habitat quality, direct disturbance, mortality due to indirect effects of climate change and impacts from livestock grazing management. |
| ***Biotic Condition:* Quality and Availability of Forage** | The amount and quality of forage available to pronghorn within suitable habitat based on native species composition, vegetation height and amount of invasive ground cover. | Composition of grasses, forbs, succulents and shrubs indicate forage quality and rangeland health. Pronghorn require a diversity of vegetation for forage and thrive with higher plant species richness. They are opportunistic herbivores; high-protein, succulent forbs make up the majority of their diet, with grasses equaling 12% of their diet (AZGFD 2011; Yoakum et al. 1996; O'Gara 1978). Pronghorn are associated with vegetation ranging 5 to 30 in (13 to 76 cm) in height (Byers 1997, O'Gara & Yoakum 2004). Availability of both native grasses and shrubs is shown to be essential for fawn survival (AZGFD 2011). O'Gara (1978) calculates that 38 pronghorn can survive on the same amount of forage that is needed for a single cow; this makes livestock grazing a significant stressor on pronghorn populations, particularly in poor forage conditions (USFWS 2002). | Historically, the main stressor for pronghorn has been the competition for food with grazing cattle (USFWS 2002). Livestock management acts as a change agent for pronghorn through increased competition for forage and water sources. Active grazing leases can indicate habitat with direct competition for forage and water from cattle, and possible reduced forage quality. Precipitation can also affect quality and availability of forage. Changes in timing and amount of precipitation due to climate changes may affect forage. Annual reproductive success in Sonoran Pronghorn has shown to depend heavily on winter rainfall (Heinz Center 2011), presumably because winter precipitation increases high nutrient forage and surface water resources in spring. Decreases in high nutrient forage will stress pronghorn populations forcing them to move more to forage and reach open water and potentially increasing mortality. |
| ***Abiotic Condition:* Availability of Surface Water** | The availability of surface water for drinking and to support forage as measured by annual and seasonal precipitation and the density of surface water resources. | Precipitation affects surface water. Pronghorns require regular access to surface water and are usually found within a 4 mile (6.5 km) radius of an accessible, open water source (Yoakum et al, 1996). | Changes in timing and amount of precipitation due to climate changes may affect available surface water. This may also lead to increased human use of water sources. Decreased water resources and increased presence of predators at available water sites can expose pronghorn to greater risk of mortality. Providing livestock water in habitat that is already suitable to pronghorn in terms of water distribution (3-4 miles distant) may compound impacts to pronghorn by reducing quality forage (Yoakum et al. 1995). Decreases in surface water resources will stress pronghorn populations forcing them to move more to reach open water and potentially increasing mortality. |

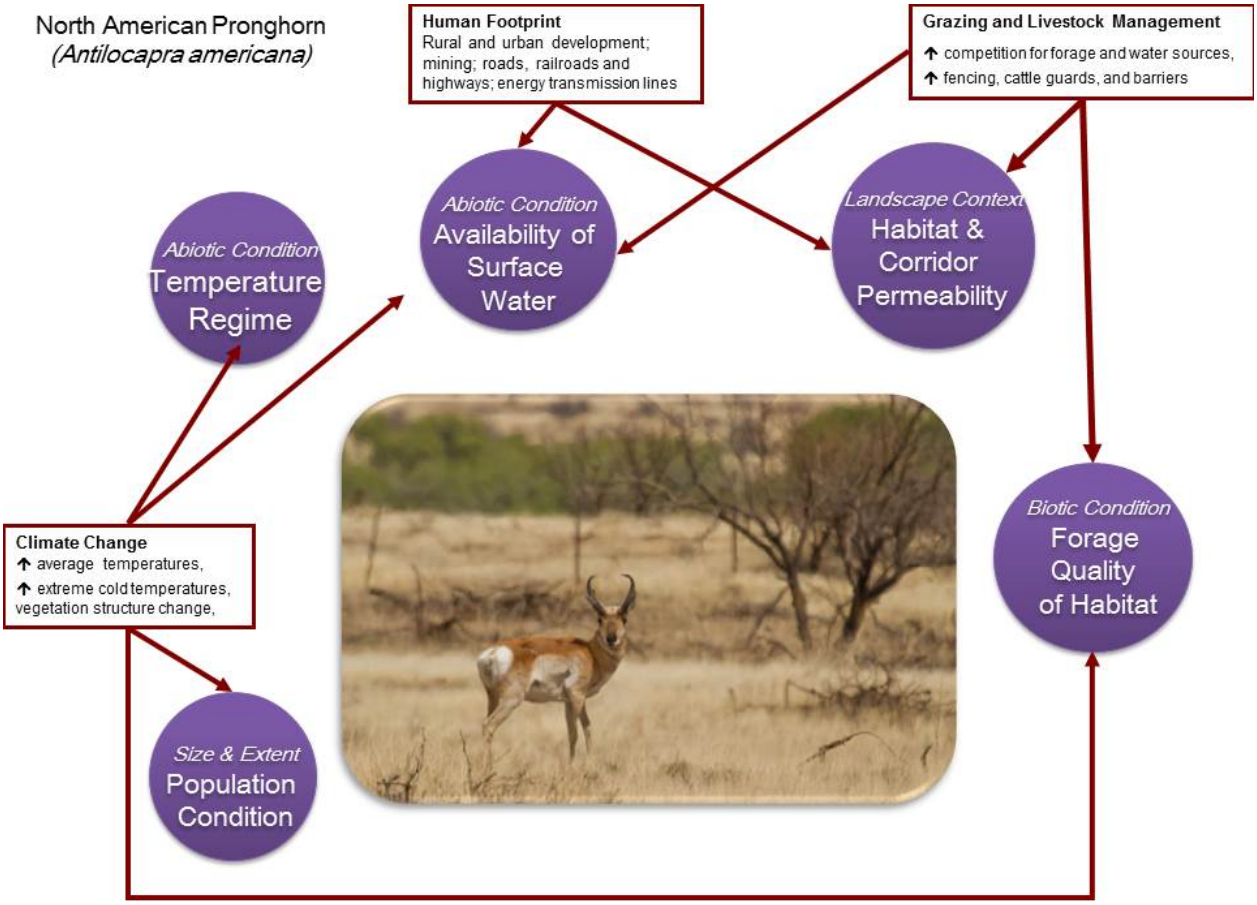
| *KEA Class:* KEA Name | Definition | Rationale | Stressors |
|---|---|---|---|
| ***Abiotic Condition:* Temperature Regime** | The pattern of average temperature fluctuations; the duration, intensity and seasonal timing of temperature extremes. | High temperatures in conjunction with low precipitation can result in greater fire risk (equating to forage and/or habitat loss); in addition, extreme temperatures can affect forage quantity and quality, surface water availability, and may directly result in fawn and juvenile mortalities due to hyperthermia (Wilson and Krausman 2008) during the hottest period of the year if temperatures reach 109 $^{o}$F (43 $^{o}$C) or greater (AZGFD 2011). | Temperature regime, particularly the occurrence extreme hot and cold temperatures, is changing due to climate change climate as is the timing and amount of precipitation. Fawns and juveniles may be susceptible to hyperthermia during the hottest period of the year which can cause direct mortality. |

### F-1.1.6 Conceptual Model Diagram

A conceptual model diagram for the CE provides a visual summary representation of the Key Ecological Attributes and species' life cycle (Figure F-4). These diagrams are intended to show how various stressors interact with categories of Key Ecological Attributes to highlight important drivers for species ecological integrity. The arrows indicate relationships between stressors and but do not indicate the nature of the influence (i.e. positive or negative).

**Figure F-4. Conceptual model diagram for the North American pronghorn, showing key ecological attributes** (by class) for this species and relationship of stressors to KEAs.

Inter-relationship between stressors and key ecological attributes




## F-1.2 Considerations and Limitations

Although fencing is an important change agent for pronghorn (Yoakum et al. 1996, Jaeger and Fahrig 2004), lack of adequate spatial data on fence type and location within pronghorn habitat prevented fencing from being addressed in the status assessment. The U.S. Fish and Wildlife Service (2002) identifies the main historical stressor for pronghorn as competition for food with grazing cattle and the availability of both native grasses and shrubs essential for fawn survival (AZGFD 2011). The status

assessment does not include an analysis of potential forage composition due to lack of adequate data to support such an analysis.

## F-1.3 Ecological Status Assessment Results and Interpretation

This section of the CE summary presents and discusses the results of the ecological status assessment for pronghorn. It addresses each indicator separately, and then addresses the overall assessment, which integrates the results of all individual indicators assessed for the CE. The results are presented using a common framework, in which the status of an indicator – or the combination of all indicators – is scored on a scale from 0.0 to 1.0, where 0.0 indicates a condition of complete replacement of reference ecological conditions due to the impacts of stressors, and 1.0 indicates a condition of no alteration of reference ecological conditions. The same yellow-to-blue color ramp is used for all results, with yellow representing low scores, green moderate scores, and dark blues high scores.

### F-1.3.1 Current Ecological Status: Development, Fire Regime, Invasives

The results of the assessments for the three individual indicators for the KEAs for pronghorn are shown in the three maps in Figure F-5 below.

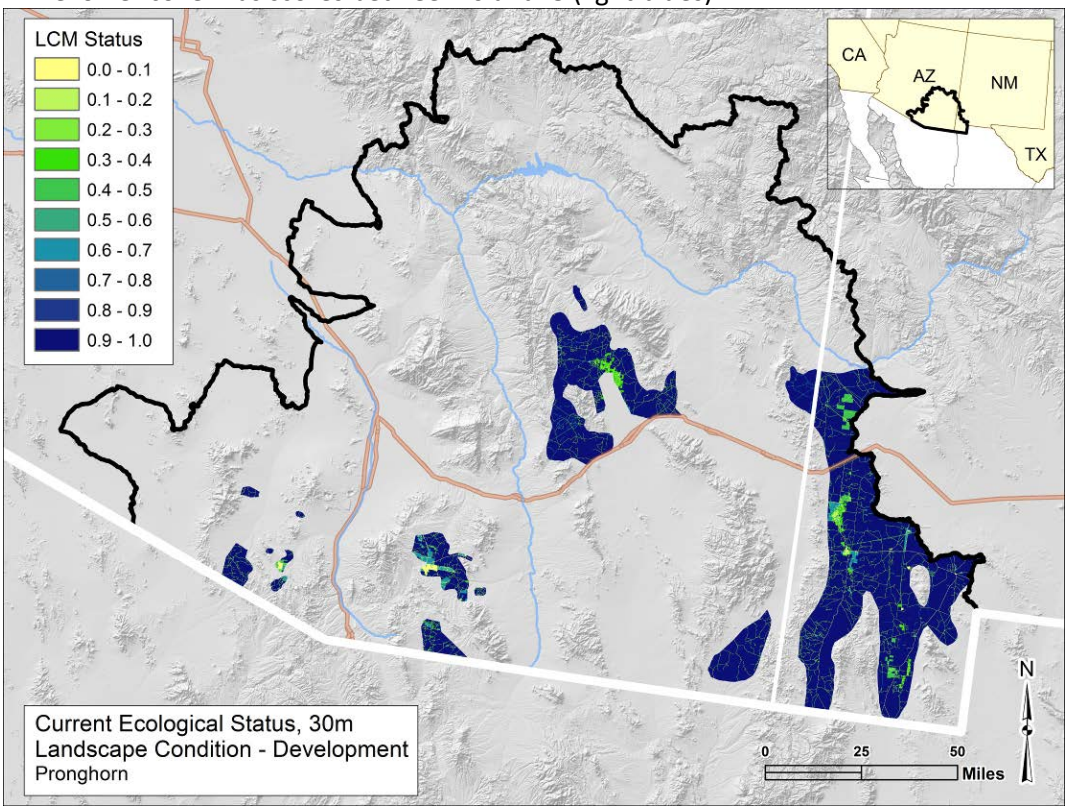
The development indicator results shown in the first map of Figure F-5 reflect a number of large but discrete areas and corridors of intense development (yellows and greens on the map) across the pronghorn's current distribution, representing areas of agricultural and other development. These areas of lower development scores in part reflect the presence of agriculture in Sulphur Springs Valley northwest of Willcox, AZ and in parts of the Animas and Playas valleys of New Mexico, development around the town of Sonoita, as well as major road corridors and other development features. However, most of the pronghorn's distribution in the MAR ecoregion is experiencing little or no impact (dark blue) from the development features reflected in the landscape condition indicator. Where development is present, the primary impacts to pronghorn include direct loss of grassland habitat and potential barriers or impediments to habitat connectivity that can restrict seasonal or daily movement patterns. In rural areas, development features tend to include fencing that poses a barrier to pronghorn movement; however, as noted above, fencing data were not available to be directly spatially assessed in this REA.

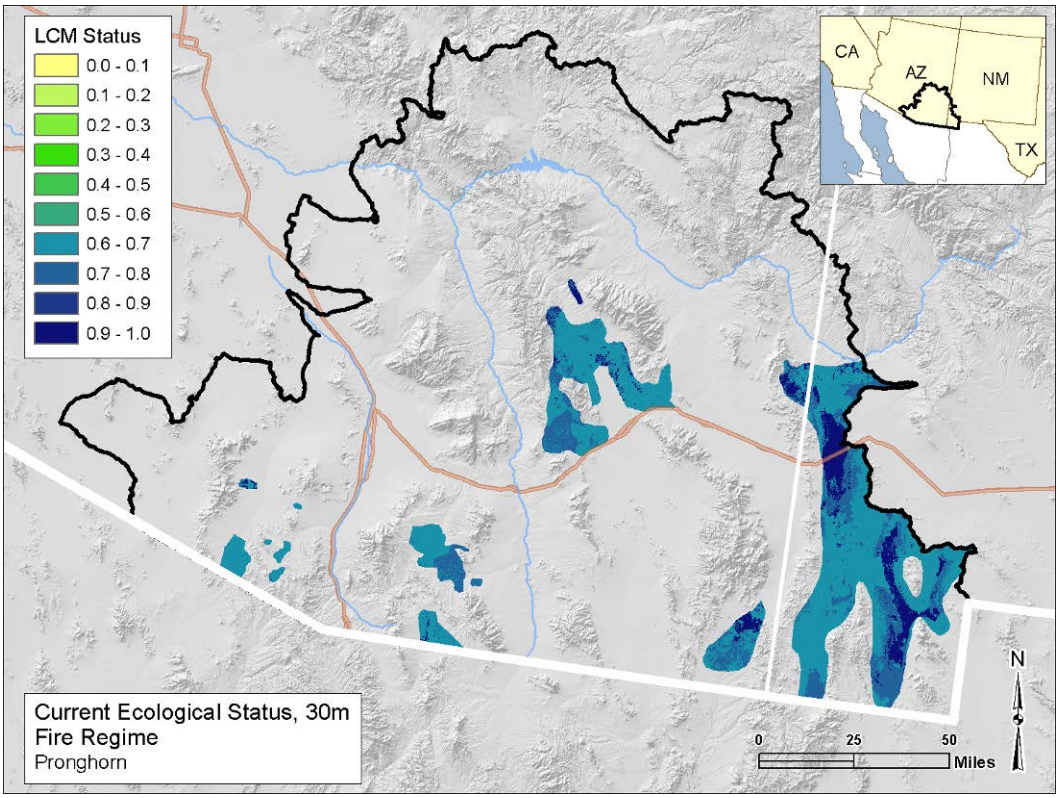
The second map in Figure F-5, the fire regime indicator results, shows that the ecological systems present in the pronghorn's distribution are primarily in the severe ecological departure category (scores of 0.65, shown in turquoise blue), indicating a highly departed fire regime. Because pronghorn are dependent on quality grasslands for forage and breeding, the degree to which fire suppression has affected grassland quality may be impacting pronghorn. In grassland habitat, departure from a characteristic fire regime may indicate more mesquite and other shrub cover that may reduce utility of habitat for pronghorn.

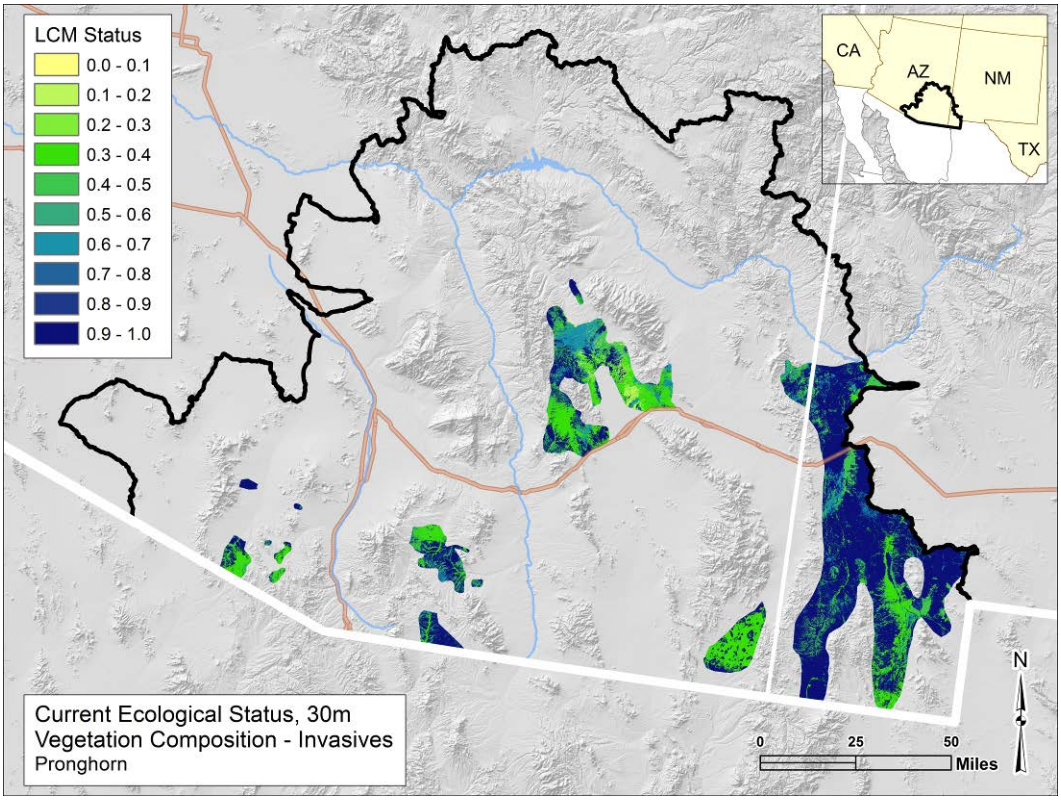
The invasives indicator map (third map in Figure F-5) shows variation in invasives cover within the pronghorn's distribution. In the central part of the ecoregion around the Sulphur Springs Valley and surrounding area, much of the landscape has high cover of mesquite, or high cover of both mesquite and non-native invasives. The same is true around the San Bernadino Valley, in extreme southeastern Arizona. In the New Mexico portion of its distribution, as well as the smaller areas in southwestern Arizona, invasives are variable – not present in the areas shown in dark blue, but high cover of one or both groups of invasives is present in the bright green areas. To the degree that non-native invasives or native woody increasers are negatively affecting forage quality (lower plant diversity), they may be

negatively impacting habitat for this species in the areas where they are modeled to be present at higher percent cover.

**Figure F-5. Scores for three indicators for pronghorn: development indicator (1st map), fire regime departure indicator (2nd map), and invasive species indicator (3rd map) for each 30m pixel.** Yellow (equivalent to 0) indicates high impacts from the CA, dark blue (equivalent to 1) indicates little to no impact from the CA. At the ecoregion scale, many development features are not readily visible (e.g., secondary roads or highways, railroads, small agricultural fields). Only 3 classes of fire regime condition are scored: no to little departure (dark blue), moderate departure and severe departure (lighter blues). For invasives, higher cover of mesquite or invasive exotics have scores between .4 and .6 (light greens), while lower cover has scores between .6 and .8 (light blues).




LCM Status
0.0 - 0.1
0.1 - 0.2
0.2 - 0.3
0.3 - 0.4
0.4 - 0.5
0.5 - 0.6
0.6 - 0.7
0.7 - 0.8
0.8 - 0.9
0.9 - 1.0
CA
AZ
NM
TX
Current Ecological Status, 30m
Fire Regime
Pronghorn
N
0
25
50
Miles


LCM Status
0.0 - 0.1
0.1 - 0.2
0.2 - 0.3
0.3 - 0.4
0.4 - 0.5
0.5 - 0.6
0.6 - 0.7
0.7 - 0.8
0.8 - 0.9
0.9 - 1.0
CA
AZ
NM
TX
Current Ecological Status, 30m
Vegetation Composition - Invasives
Pronghorn
N
0
25
50
Miles

### F-1.3.2 Current Ecological Status: Full Scenario

The results of the three individual status indicators were combined to get an overall ecological status score, per pixel of the CE's distribution, as shown in the first map of Figure F-6 below. The overall, per-pixel status scores for the CE were then averaged across each of the 4 km grid cells, as shown in the second map of Figure F-6 below.

When the three sets of indicator scores are aggregated to obtain the overall ecological status scores (as shown in the first map of Figure F-6), much of the habitat for this species is in somewhat lower to moderate condition, with status scores between 0.4 and 0.7 over roughly 65% of its distribution (Figure F-7). In visually comparing the maps of scores for the three individual indicators (Figure F-5) to the map of the overall status scores (first map of Figure F-6), the cumulative impacts of the change agents reflected by the indicators are clear. The altered fire regime and invasives cover are both driving down scores in much of the ecoregion, and scores drop even lower where agriculture or other development features are also present. These same patterns are reflected when the overall status scores are averaged across each 4 km reporting unit, as shown in the second map of Figure F-6 above.

Based on the indicators assessed in this REA, impediments to connectivity from development features and high densities of mesquite are the primary factors of concern affecting pronghorn habitat across its distribution. Altered fire regimes associated with fire suppression in combination with mesquite encroachment is also a concern due to the long-term negative impacts on quality of forage. Direct habitat loss (from existing development) is much more localized, but where it occurs, it may result in pronghorn being unable to occupy those areas (e.g., urban development, agriculture, and fencing).

**Figure F-6. Overall ecological status scores for pronghorn for all three indicators combined (development, fire regime and invasives) for each 30m pixel (top) and 4km grid cells (bottom).** Yellow scores (equivalent to 0) indicate high impacts from the CAs, dark blue (equivalent to 1) indicate little to no impact from the CAs. In the second map, the score for each 4km cell is an average of the overall ecological status scores of the 30m pixels within the 4km cell that were scored for the CE.

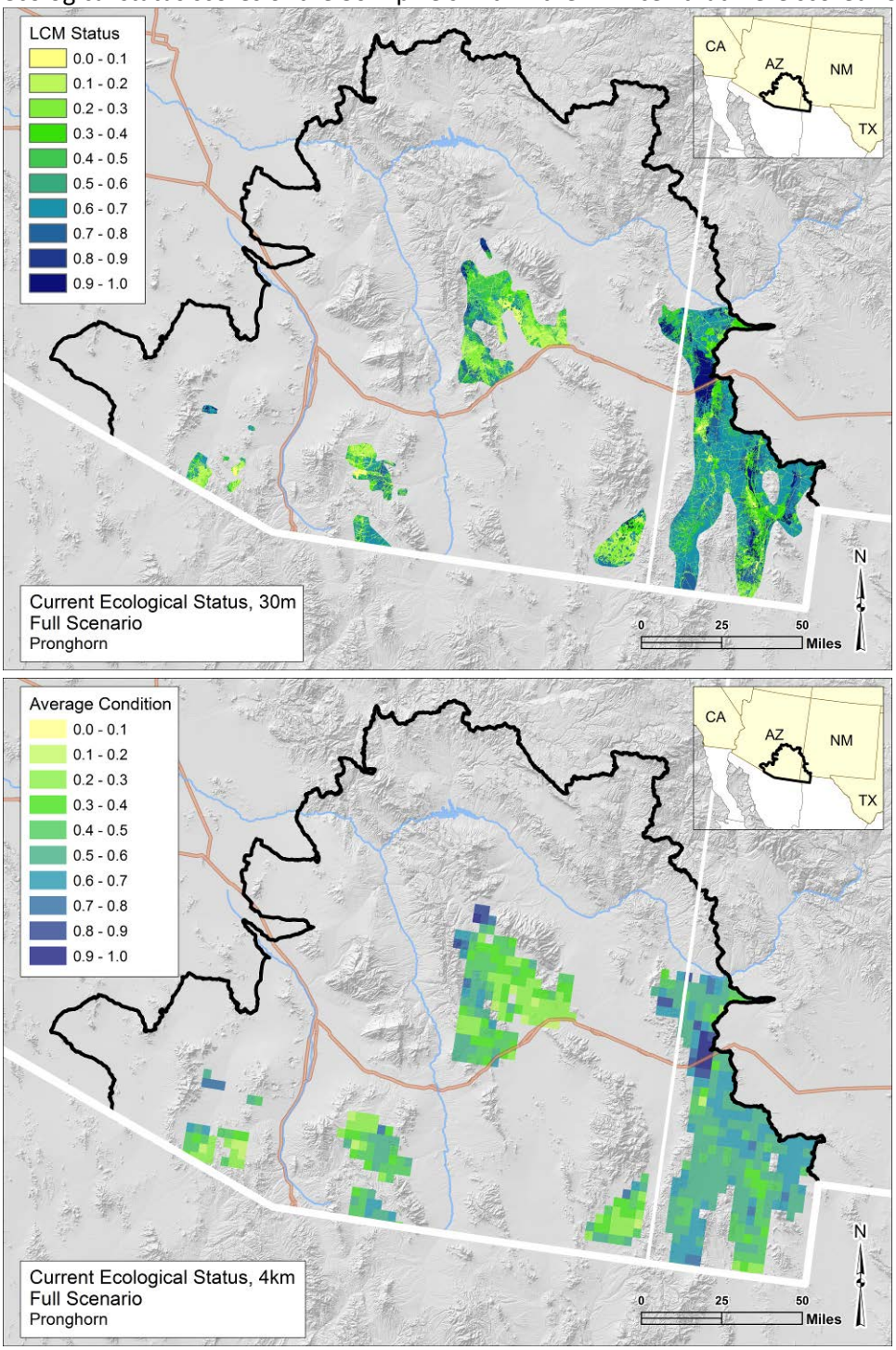
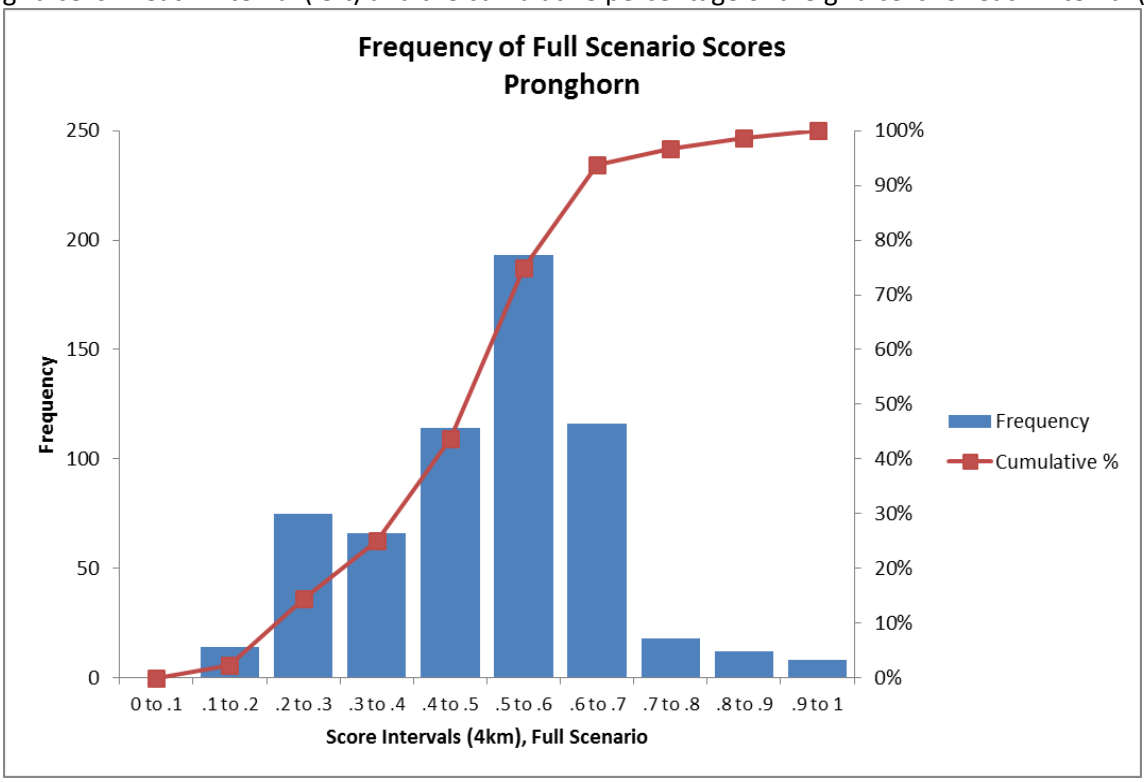


**Figure F-7. Frequency distribution of the 4km ecological status scores for pronghorn, with cumulative percent.** The x-axis represents the 0.1 increment scoring intervals, while the y-axis shows the number of grid cells in each interval (left) and the cumulative percentage of the grid cells for each interval (right).




## F-1.4 References for the CE


Arizona Game and Fish Department (AZGFD). 2011. Arizona Statewide Pronghorn Management Plan. Available:http://www.azgfd.gov/h_f/documents/2011%20Pronghorn%20Management%20Plan%20-%20December%203%202011.pdf

Arizona Game and Fish Department (AZGFD). 2013. HabiMap™ Arizona. Mammal distributions, last revision 2011.Available: http://www.habimap.org. Accessed 18 January 2013.

Arizona Wildlife Linkages Workgroup. 2006. Arizona Wildlife Linkages Assessment. Arizona Department of Transportation, Phoenix, AZ. Available: http://www.azdot.gov/inside_adot/OES/AZ_WildLife_Linkages/assessment.asp

Bright, J. L. and Hervert, J. J. 2005.Adult and fawn mortality of Sonoran pronghorn. Wildlife Society Bulletin, 33:43–50.

Brown C.L., A.R. Hardy, J.R. Barber, K.M. Fristrup, K.R. Crooks, and L.M. Angeloni. 2012. The Effect of Human Activities and Their Associated Noise on Ungulate Behavior. PLoS ONE 7(7): e40505. Available: http://www.plosone.org/article/info%3Adoi%2F10.1371%2Fjournal.pone.0040505

Brown, D.E. and C.H. Lowe. 1982. Biotic Communities of the Southwest [map]. Scale 1:1,000,000. General Technical Report RM-78. U. S. Forest Service, Fort Collins. Reprinted (and revised) 1994 by University of Utah Press, Salt Lake City.

Bureau of Land Management (BLM). March 2011. Colorado Plateau Rapid Ecoregional Assessment: Final Memorandum I-3-c. Pp 138-142. Available: http://www.blm.gov/pgdata/etc/medialib/blm/wo/Communications_Directorate/public_affairs/landscape_approach/documents1.Par.2170.File.dat/FinalCPMemo1-3-c3-21-11R.pdf

Byers, J.A. 1997. American Pronghorn: Social Adaptations and the Ghosts of Predators Past. Univ. Chicago Press, Chicago, Illinois. Pp 300.

Casey L. Brown, Amanda R. Hardy, Jesse R. Barber, Kurt M. Fristrup, Kevin R. Crooks, and Lisa M. Angeloni. 2012. "The Effect of Human Activities and Their Associated Noise on Ungulate Behavior "*PLoS One*7.7 (2012): e40505. Available: http://works.bepress.com/jesse_barber/10

Cockrum, E.L. 1981. Taxonomy of the Sonoran pronghorn. Pages 2-10 in J.S. Phelps, ed., The Sonoran Pronghorn. Spec. Rept. 10. Arizona Game and Fish Department, Phoenix. Pp. 55.

Comer, P., and J. Hak. 2012. Landscape condition model of the western United States. NatureServe, Boulder, CO. In preparation.

DeVos, J. C. and Miller, W. H. (2005), Habitat use and survival of Sonoran pronghorn in years with above-average rainfall. Wildlife Society Bulletin, 33: 35–42.

Dubay, Shelli A., Ted H. Noon, James C. deVos Jr. and Richard A. Ockenfels. 2006. Serologic Survey for Pathogens Potentially Affecting Pronghorn (*Antilocapra Americana*) Fawn Recruitment in Arizona, USA. Journal of Wildlife Diseases 42(4):844-848.

Einarsen, A.S. 1948. The Pronghorn Antelope and Its Management. Monumental Printing Co., Baltimore, Maryland. Pp 238.

Ellis, J. 1970. A computer analysis of fawn survival in the pronghorn antelope. Dissertation, University California, Davis, California, USA.

Hannibal, M.E. 2012. The Spine of the Continent: The Most Ambitious Wildlife Conservation Project Ever Undertaken. Globe Pequot Press, Guilford, CT, USA.

Hansen, R.M. and R.C. Clark. 1977. Foods of elk and other ungulates at low elevations in northwestern Colorado. J. Wildl. Manage. 41:76-80.

Heinz Center for Science Economics and the Environment. 2011. Managing and monitoring Arizona's wildlife in an era of climate change: Strategies and Tools for Success. Report and Workshop Summary, prepared for the U.S. Department of Interior Bureau of Land Management and the Arizona Game and Fish Department. Available: http://www.heinzctrinfo.net/wildlife/wordpress/wp-content/uploads/2011/07/Low-res-Heinz-Arizona-Report_Final_7.11.11.pdf

Hernández Fernández, M. and E. Vbra 2005. A complete estimate of the phylogenetic relationships in Ruminantia: a date species-level supertree of extant ruminants. Biological Reviews 80:269-302.

Hoffmann, M., Byers, J. & Beckmann, J. 2008.*Antilocapraamericana*. In: IUCN 2012. IUCN Red List of Threatened Species. Version 2012.2.<www.iucnredlist.org>. Accessed 18January 2013.

Howard, Janet L. 1995. *Antilocapra americana*. 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/animals/mammal/anam/all.html. Accessed 19 January 2013.

Hughes, K.S. and Smith. 1990. Sonoran pronghorn use of habitat in southwest Arizona. Arizona Coop. Fish and Wildl. Res. Unit Proj. 14-16-009-1564 RWO 6, Final Report, Tucson. 58 pp.

Jaeger, J. A. G., and L. Fahrig.2004. Under what conditions do fences reduce the effects of roads on population persistence? *Conservation Biology*18:1651–1657.

Janis, C. M. 2000.Ruminants of the Neogene of North America. In: Vrba, E. and G. Schaller (eds.) Antelopes, Deer, and Relatives: Fossil Record, Behavioral Ecology, Systematics, and Conservation. Yale University Press, New Haven and London. Pp 26-37.

Janis, C.M, Kathleen M. Scott and Louise L. Jacobs, ed. 1998. Evolution of Tertiary Mammals of North America (Vol. 1): Terrestrial Carnivores, Ungulates, and Ungulate-like Mammals. Cambridge University Press, Cambridge, United Kingdom. Pp. 491-507.

Janis, C.M. and K.M. Scott. 1987. The interrelationships of higher ruminant families with special emphasis on the members of the Cervoidea. Am. Mus. Novit. 2,983: 1-85.

Jones, J. K., Jr., R. S. Hoffman, D. W. Rice, C. Jones, R. J. Baker, and M. D. Engstrom. 1992. Revised checklist of North American mammals north of Mexico, 1991. Occasional Papers, The Museum, Texas Tech University, 146:1-23.

Kitchen, D.W. 1974. Social behavior and ecology of the pronghorn. Wildlife Monograms. 38. Pp 96.

Lee, T.E., Jr. 1992. Mitochondrial DNA and allozyme analysis of pronghorn populations in North America. Ph.D. thesis, Texas A&M Univ., College Station. Pp. 53.

Lee, Thomas E., Jr.; Bickham, John W.; Scott, M. Douglas. 1994. Mitochondrial DNA and allozyme analysis of North American pronghorn populations. Journal of Wildlife Management 58(2): 307-318.

Leopold, A.S. 1959. Fauna  Silvestre de Mexico. Instituto Mexicano de Recursos Naturales Renovables. México. D. F.

Maher, C.R. 2000. Quantitative variation in ecological and hormonal variables correlates with spatial organization of pronghorn (*Antilocapra americana*) males. Behav. Ecol. and Sociobiol. 47 (5): 327-338.

McGhee M. B. et al. 1981. Journal of Wildlife Disease. 17:3 page 353.

Morgart, J. R., J. J. Hervert, P. R. Krausman, J. L. Bright, and R. S. henry. 2005. Sonoran pronghorn use of anthropogenic and natural water sources. Wildlife Society Bulletin 33:51-60.

Murray, L. 2006. The smaller Artiodactyls: peccaries, oxen, deer, and pronghorns. Jefferson, G. and L. Lindsay (eds) In: Fossil Treasures of the Anza-Borrego Desert: the Last Seven Million Years. Sunbelt Publications, San Diego, California. pp. 273-287.

National Drought Mitigation Center. 2013. U.S. Drought Monitor – West [web application]. Lincoln, NE. Available: http://droughtmonitor.unl.edu/DM_west.htm

NatureServe. 2007. InfoNatura: Animals and Ecosystems of Latin America [web application]. Version 5.0. Arlington, Virginia, USA. Available: http://www.natureserve.org/infonatura. Accessed 18 January 2013.

O'Gara, B.W. 1968. A study of reproduction in the female pronghorn (*Antilocapra americana* Ord.) Ph.D. thesis, Univ. Montana, Missoula. Pp. 161.

O'Gara, B.W. 1978.*Antilocapraamericana.*Mammalian Species, American Society of Mammalogists. 90:1-7

O'Gara, B.W., and J.D. Yoakum, eds. 1992. Pronghorn management guides. Proceedings of the Pronghorn Antelope Workshop 15 (supplement). N.p.

O'Gara, B.W. and J.D. Yoakum. 2004. Pronghorn Ecology and Management. University Press of Colorado, Boulder, Colorado.Pp 903.

Ockenfels, R. A. 1994. Factors affecting adult pronghorn mortality rates in central Arizona. Arizona Game and Fish Department, Digest 16, Phoenix. Pp. 11.

Ockenfels, R. A., C. L. Ticer, A. Alexander, J. A. Wennerlund. 1996a. A landscape level pronghorn habitat evaluation model for Arizona. Arizona Game and Fish Department Technical Report 19, Phoenix, Arizona, USA. Pp. 50.

Ockenfels, R. A., C. L. Ticer, A. Alexander, J. A. Wennerlund. 1996b. Statewide evaluation of pronghorn habitat in Arizona. Arizona Game and Fish Department Federal Aid in Wildlife Restoration Project Report W-78-R.Pp 296, Phoenix, Arizona, USA.

Ockenfels, R. A., C. L. Ticer. 2000. A validation of Arizona's landscape-level pronghorn habitat model. *In:* Proceedings of the 19th Biennial Pronghorn Workshop 19:63-70, Sur, Mexico. Available: http://www.cibnor.org/servicios/biblioteca/libros/Validation.pdf

O'Gara, B.W. 1999.*Antilocapraamericana*. In: D. E. Wilson and S. Ruff (eds).*The Smithsonian Book of North American Mammals*, UBC Press, Canada. Pp. 339-341.

Pallares, E. 1999. El retornodel Berrendo: Rescate de unaespecie en peligro de extinción. National Geographic 4 (3): Conservación en América Latina.

Patrick-Birdwell, C., S. Avila-Villegas, J. Neeley, L. Misztal, and N. Deyo. May 2012. Mapping and Assessing the Environmental Impacts of Border Tactical Infrastructure in the Sky Island Region. IN: Proceedings of *Connecting mountain islands and desert seas: biodiversity and management of the Madrean Archipelago III*, Tucson, AZ. In Press.

San Diego Zoo Global Library. 2009. Factsheet: Pronghorn (*Antilocapraamericana).* Available: http://library.sandiegozoo.org/factsheets/pronghorn/pronghorn.htm.Accessed 18 January 2013.

Secretaría de MedioAmbiente y RecursosNaturales, y ComisiónNacional de ÁreasNaturalesProtegidas (CONANP). 2009. PROGRAMA DE ACCIÓN PARA LA CONSERVACIÓN DE LA ESPECIE: Berrendo (Antilocapra americana). Available: http://www.biodiversidad.gob.mx/especies/especies_priori/fichas/pdf/pace_berrendo.pdf. Accessed 18 January 2013.

Simmons, H.A., Steffen, D.J., Armstrong, D.L., Rogers, D.G. 2002.*Parelaphostrongylustenuis*in captive pronghorn antelope (*Antilocapraamericana*) in Nebraska. J. Wildl. Dis. 38(4): 822-25.

Southwest Climate Change Network (SWCCN). 2008. Drought and the Environment. Available: http://www.southwestclimatechange.org/impacts/land/drought in October 2010.

The Integrated Taxonomic Information System (ITIS). 2013. Available: www.itis.gov. Accessed 18 January 2013.

Thorne, E.T., Williams, E.S., Spraker, T.R., Helms, W., Segerstrom, T. 1988. Bluetongue in free-ranging pronghorn antelope (*Antilocapra americana*) in Wyoming: 1976 and 1984. J. Wildl. Dis., 24(1): 113-119.

Ticer, C.L.D. and W.H. Miller. 1994. Pronghorn fawn bed site selection in a semidesert grassland community of central Arizona. Proc. Pronghorn Antelope Workshop 6:86–103.

U.S. Fish and Wildlife Service (USFWS). 2002. Recovery Criteria and Estimates of Time for Recovery Actions for the Sonoran Pronghorn: A Supplement and Amendment to the Final Revised Sonoran Pronghorn Recovery Plan. Available: http://ecos.fws.gov/docs/recovery_plan/031126.pdf.

U.S. Fish and Wildlife Service (USFWS). 2011. Establishment of a Nonessential Experimental Population of Sonoran Pronghorn in Southwestern Arizona. Federal Register, Vol. 76, No. 87, Rules and Regulations 76 FR 25593 25611. Available: http://www.gpo.gov/fdsys/pkg/FR-2011-05-05/pdf/2011-10467.pdf. Accessed 18 January 2013.

U.S. Fish and Wildlife Service (USFWS). 2013. Environmental Conservation Online System Species Profile: Sonoran Pronghorn. Available: http://ecos.fws.gov/speciesProfile/profile/speciesProfile.action?spcode=A009. Accessed 18 January 2013.

Wasel S. M. et al. 2003 Journal of Wildlife Disease. 39(2) 338-346

Whittington, J., M. Musiani, T. Clevenger and C. White. 2008. Effects of Human Activity and Restoration Actions on Animal Movements. Ed. Darren Bender. Proceedings of the 23rd Biennial Pronghorn Workshop, Canmore, Alberta. Available: http://homepages.ucalgary.ca/~berrendo/Draft2008Proceedings.pdf

Wilson, D. E., and D. M. Reeder (ed). 1993. Mammal Species of the World: a Taxonomic and Geographic Reference. Second Edition.Smithsonian Institution Press, Washington, DC. xviii + 1206 pp. Available: http://www.nmnh.si.edu/msw/.

Wilson, D. E., and D. M. Reeder (ed). 2005. Mammal species of the world: a taxonomic and geographic reference. Third edition. The Johns Hopkins University Press, Baltimore. Two volumes. 2,142 pp. Available: http://www.bucknell.edu/msw3/.

Wilson, Ryan R. and Paul R. Krausman. 2008. Possibility of Heat-Related Mortality in Desert Ungulates. Journal of the Arizona-Nevada Academy of Science 40(1):12-15.

Yoakum, Jim. 1980. Habitat management guides for the American pronghorn antelope. Tech. Note 347. Denver, CO: U.S. Department of the Interior, Bureau of Land Management, Denver Service Center. Pp 77.

Yoakum, J.D. 1994. Water requirements for pronghorn. Proc. Pronghorn Antelope Workshop 16:143–15

Yoakum, J.D., B. W. O'Gara and V.W. Howard, Jr. 1995. Pronghorn on Western Rangelands. Pages 211-224 in Krausman, ed. Rangeland Wildlife. The Society for Range Management, Denver, CO. 1996.

# *F-2* Coues White-tailed Deer *(Odocoileus virginianus couesi)*

## F-2.1 Conceptual Model

### F-2.1.1 Classification Comments

Endemic to North America, Coues white-tailed deer (*Odocoileus virginianus couesi*) or Venado Cola Blanca in Mexico, one of 38 subspecies of white-tailed deer found in the new world, is found primarily in central and southeastern Arizona, southwestern New Mexico and north central Mexico. The white-tailed deer was first described in Virginia by Thomas Hariot in 1584 (Heffelfinger 2006). The Coues white-tailed deer was first classified as *Odocoileus virginianus couesi* in 1874 by U.S. Army surgeon Dr. Joseph Rothrock, who collected two specimens in 1874 and suggested they be called *Odocoileus virginianus*

*couesi* for the naturalist Dr. Elliot Coues. Dr. Coues never actually collected specimens (Carmony 1985). The Coues white-tailed deer is most closely related to *O. v. Texanus*, with which it may exhibit some interchange with populations in Durango, Mexico (Findley 1975). There is some controversy regarding the designation of subspecies due to the fact that classification is currently based primarily on morphological characteristics (e.g. size, pelage color, size and shape of male antlers) (Hoffmeister 1986).

White-tailed deer range from 60 degrees north latitude in Central Canada to 15 degrees south latitude in South America: and from the east coast of the United State and Canada to the West Coast. In the United States and Canada there are 16 subspecies. White-tailed deer have not been described in California, Nevada, and Utah (Hesselton and Hesselton 1982). There are 14 subspecies found in Mexico with only Baja and parts of Sonora and Chihuahua void of the species. With 38 identified subspecies, 14 found in Mexico and 16 in Canada and the United States, and a north/south range between 60 degrees north in Canada and 15 degrees south in Peru; the white-tailed deer is the most widely distributed and studied cervid in the American continent, where it inhabits wide variety of habitat types throughout its range (Mandujano 2010, Geist 1998).

The Madrean REA focused exclusively on one subspecies, the Coues white-tailed deer (*Odocoileus virginianus couesi*).

References to the *Texanus* and other subspecies are occasionally used to understand unique adaptation responses relevant to this arid borderlands ecoregions. Within this ecoregion, populations of Coues white-tailed deer in Arizona are currently managed and monitored in at least 17 different Game Management Units (AZGFD 2013). In the New Mexico portion of the Madrean Archipelago Ecoregion the Coues white-tail deer occurs in areas of New Mexico Game Management Units (GMUs) 15, 16, 17, 21, 22, 23, 24, 26, and 27; aerial surveys have been flown in some of the GMUs each year to develop tend data (NMGF 2013). Populations of Coues white-tailed deer in Northern Sonora are managed through established seasons and bag limits. There is no evidence that they are regularly monitored or surveyed.

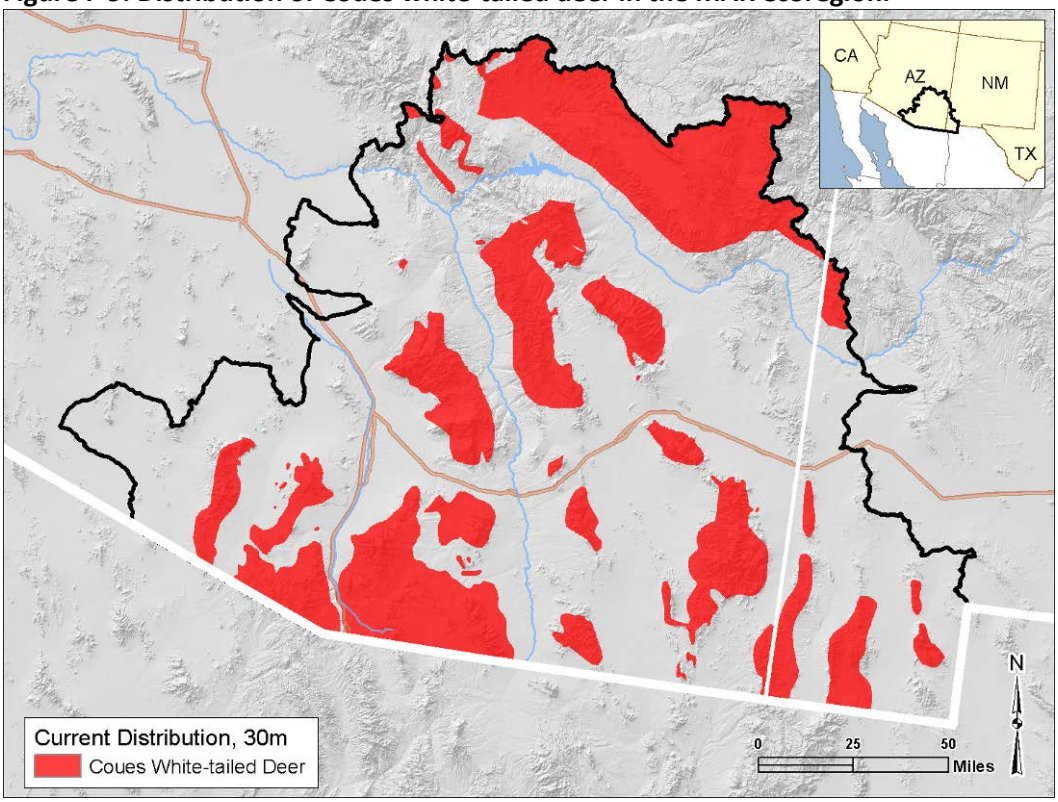
**Figure F-8. Coues white-tail deer (*Odocoileus virginianus couesi*).**

## F-2.1.2 Biology and Distribution Status

### *F.2.1.2.1 Distribution*

Mixed populations of mule deer (*Odocoileus hermionus*) and Coues white-tailed deer (*Odocoileus virginianus couesi*) occur within the Madrean Archipelago ecoregion today. The Coues white-tailed deer is distributed throughout the Madrean Archipelago including both north and south of the border (Hoffmeister 1986). Coues white-tailed deer occur from near the Mogollon Rim in Central Arizona, to south of the Mexican/U.S. border in Durango, Mexico where they tend to blend with other subspecies (Galindo-Leal 1993, Knipe 1977). Isolated populations are found as far east as the Black Range, Gila, Animas, Alamo Hueco, San Luis, Magdalena and Peloncillo Mountains of southwestern New Mexico and as far west as the Ajo Mountains in Organ Pipe Cactus National Monument in western Arizona (Lopez 2006). A single subspecies, Coues white-tailed deer, occurs in Arizona (Hoffmeister 1986). While the species is not usually thought of as a desert animal, the Coues white-tailed deer inhabits virtually every mountain range in Sonora east of the 113$^{th}$ meridian, including the arid Sierras de Alamo, Jojoba, Picu', and Los Mochos (Carmony and Brown 1991). A prominent food source, the deer is highly regarded as source of protein and features prominently in the Vigida (Vikita) ceremony of the Hia C'ed and Tohono O'odham. Their east/west range extends from the foothills of the Sierra Madre in Western Durango, to the far west with a population in the Sierra Cubabi 8km south of Sonoita on the northeast side of Mexico Highway 2. (Felger & Broyles, *Dry Borders: Great Natural Reserves of the Sonoran Desert*).

**Figure F-9. Distribution of Coues white-tailed deer in the MAR ecoregion.**




Home range is vital to Coues white-tailed deer. It must be large enough to provide essential components of food, shelter, and water, but small enough to be totally familiar to deer for survival advantages (Ockenfels 1991). There have been limited studies designed to define the home range of Coues white-tailed deer. The best estimate of Coues white-tailed deer home range comes from research done by

Ockenfels et. al. (1993). The overall area they used during that time and more intensive "core area" estimated does had a home range of averaging 2 $mi^2$ (range = .02 – 7.0 $mi^2$) with a core area of more intensive use of 0.7 $mi^2$. As is typical, bucks had larger home ranges, averaging 4.1 $mi^2$ (range=1.7-6.5 $mi^2$) with a core area of 1.7 $mi^2$ (Heffelfinger 2006). Coues white-tailed deer do not need to migrate, but do have seasonal movements related to cover, food and water availability (Ockenfels 1991).

#### *F.2.1.2.2 Population*

Arizona's statewide Coues white-tailed deer population is based on the sum of population estimates gathered from hunt management units; these are produced with computer simulation models using data gathered from aerial surveys and annual hunter-reported harvest data. Fixed-wing aircraft, helicopter and ground surveys are conducted, prior to hunting season. The current population of Coues white-tailed deer in Arizona is estimated at 82,000 post hunt adults (AZGFD 2013). New Mexico has a smaller population in relation to Arizona, however, Coues populations have been expanding and increasing the past few years based on biologists' observations. Mexico has abundant populations but there are limited surveys and no known estimate of the current size of the Mexican population.

#### *F.2.1.2.3 Reproduction*

*Breeding Systems:* Local fluctuation in reproductive success are related to seasonal and annual reproductive phenology; antler development, rutting behavior, and antler cast for males, while rutting season, fawn drop, and lactations are important time periods for females. Rutting season is partially related to photoperiod, the ratio of diminishing daylight to darkness (Verme and Ullery 1984). Rut takes place earlier in northern white-tailed deer populations beginning in September and continuing for several months terminating in January. Variation in rut timing is related to latitude with southern populations occurring up to 2 months later than northern populations. The rut for Coues White-tailed deer is normally form mid-December into March (McCabe and Leopold 1959, Welles 1959). The Tohono O'odham honor this important source of food by indicating February as Uhwalig Machath, "moon of the deer-mating odor" (The Antler cast for most populations starts in late December and peaks in January. Antler cast generally occurs after breading season and is based on deceasing levels of testosterone in males. Female white-tailed deer are receptive to breeding only 2-3 days during each estrus cycle (Knox et al. 1988) For Coues white-tailed deer, peak of estrus should coincide with the rut (Ockenfels 1994).

***Gestation**:* The gestation period for Coues white-tailed deer is between 200 and 207 days with fawning from mid-July through mid-September, peaking in late July or August (Knipe 1977). However timing can vary from year to year based on timing of conception and nutrition of the doe. Very poor nutrition can delay breeding or lengthen the gestation period (Verme 1965).

***Litter size**:* One to two fawns, per breeding female per year. Twins are the norm after the first year with single fawns the norm in the first breeding cycle.

***Breeding Season**:* Coues white-tailed deer breed between the months of November and March with the peak rut taking place in mid-January. Many of the extremes are likely young, inexperienced buck. The average date of conception is early to mid January (Ockenfels 1991)

***Birthing Season:*** Depending on habitat and environmental conditions, Coues white-tailed deer generally give birth in the Madrean Archipelago ecoregion between mid-July and mid-September. Fawning usually peaks between late July and August. (Heffelfinger 2006)

***Life history**:* Coues white-tailed deer weigh 4 to 6 pounds at birth, which is approximately 10 to15% of the adult female's weight. Young are weaned by 2 to 3 months but already feed on vegetation during the first month. Rapidly growing fawns lose their spots after about two month. Males reach sexual maturity in 1.5 years but may not have the opportunity to breed until they are dominant enough to compete with older males. Females are sexually mature at 18 months. Females often fail to conceive in their first breeding cycle. The Life span of the Coues white-tailed deer is approximately 10 years but they can live to be 20 (Hoffmeister 1986, USFS 2013).

## F-2.1.3 Habitat and Ecology

### *F.2.1.3.1 Habitat*

Knowledge of Coues white-tailed deer habitat requirements are mostly observational in nature. Coues white-tailed deer have several specific habitat requirements needed for survival.

Food: Shrubs, Forbs, Grasses, Mast, Succulents
Cover: Escape cover, Bedding and Fawning cover, Thermal cover
Space: Activity center, Home range
Water: Surface water, Preformed water (Fulbright & Ortega-S 2006)

***Elevation:*** In the Madrean Archipelago, Coues white-tailed deer are found between elevations ranging from 3,000 to 10,000 feet, with the greatest densities of deer concentrated within the 3,500 to 5,500 feet elevation.

***Terrain:*** Coues white-tailed deer occupy rolling hills, bajadas and canyons and mountainous terrain with adequate water sources, food and over story. They seek terrain that decreases the risk of predation and dehydration under extreme conditions, based in part on the degree of cover. At lower elevations the Mesquite habitat provides the greatest amount of hiding and thermal cover and edible food. In order to detect and avoid or escape approaching predators Coues white-tailed deer tend to prefer habitat and bedding areas that allow for adequate visibility (Bello et al. 2001). Both bucks and does prefer slopes of 20 –29 degrees for resting and feeding, usually on the upper third of major and minor ridges. Habitats offering adequate hiding and thermal cover during extremes in summer temperatures are critical to Coues white-tailed deer survival (Ockenfels 1994).

***Distance from accessible water**:* Coues white-tailed deer require regular access to surface water and regularly travel one and a half miles to water during the dry season in May and June (Knipe 1977). Coues white-tailed deer tend to move to permanent water during the dry season where they will stay until the rains return later in the year. The distance to water sources varies between years and between seasons. Coues white-tailed deer select habitat within one half mile of a permanent water source, but show a distinct affinity for areas within one quarter of a mile, with very few animals located more that three-quarters of a mile from water (Ockenfels 1991).

***Vegetation:*** Coues white-tailed deer prefer mountainous habitat, associated with Madrean Evergreen Woodlands (Brown & Lowe 1982). They are found primarily in woodland communities consisting of evergreen oaks or in mixed oak-pinion-juniper. They are also found in ponderosa pine forests, in desert scrub, in deciduous forests and occasionally in spruce-fir (Hoffmeister 1986).

### *F.2.1.3.2 Landscape Connectivity*

In the Madrean Archipelago ecoregion Coues white-tailed deer do not migrate in the traditional sense. But they do show seasonal shifts in habitat induced by the availability of food, water, cover, and the

effects of weather hunting, breeding, livestock grazing and other factors. These seasonal migrations are limited within separate herd home ranges and are not as impressive as long-range migrations found in many other ungulates. However, these seasonal movements are essential to maintaining gene flow and finding quality forage and water sources in summer and winter (Heffeleinger 2006). Coues white-tailed deer will move higher or lower in their home range depending on temperature and snow cover. These deer are generally more abundant on southern and eastern exposures at 6,000 to 7,000 feet in winter reflecting the seasonal movements associated with temperature and snow depth. Coues white-tailed deer are similar to mule deer in that the rut tends to drastically change activity patterns with large-scale movement taking them out of their home range. (Welch 1960).

#### *F.2.1.3.3 Food*

Coues white-tailed deer require a diversity of vegetation for forage and thrive with higher plant species richness. They are opportunistic herbivores; shrubs and trees makes up the highest proportion of their diet. Coues white-tailed deer change their forage preferences season to season and with habitat type and elevational gradient. They will seek the most nutritious and succulent forage species available. In the lower elevation western deserts they consume jojoba, fairy duster, mallow and agave, while in the central highlands of Arizona and at higher elevations they forage on desert ceanothus, ratany, sugar sumac, white sage and hollyleaf buckthorn. Coues white-tailed deer populations found in southeastern Arizona are known to eat a wide range of shrubs and succulents, including velvet-pod mimosa, ocotillo, fairy-duster, barrel cactus, buckwheat, Fendler's ceanothus, oaks, alligator juniper berries and leaves, kidneywood, silk tassel and prickly poppy (Hefflefinger 2006). In Arizona, forbs, shrubs, and trees make up the majority of food, with cacti, grasses and other plants accounting for minor dietary intake throughout the year (Ockenfels 1991).

#### *F.2.1.3.4 Phenology*

Coues white-tailed deer activity patterns are circadian, crepuscular and diurnal. Movement patterns show animals active between 8 am and 10 am and again between 4 pm and 6 pm during the winter months (Krauseman and Ables 1981). During the summer 70% of the deer observed by Knipe (1977) between 9 am and 5 pm were bedded. (Heffelfinger 2006). Activity patterns for Coues white-tailed deer are variable, and influenced by region, weather, season, and also by herd dynamics. High intensity hunting pressure has been shown to induce seasonal movements in Coues white-tailed deer (Welch 1960).

#### *F.2.1.3.5 Predators*

The native prey of the Mexican gray wolf was almost entirely the Coues white-tailed deer. The Arizona wolf would have fed on the white-tailed deer and mule deer. (Brown 1983) With limited populations of Mexican wolves in Coues white-tailed deer habitat, the two most important predators in the Southwest are mountain lions (*Felis concolor*) and coyote (*Canis latrens*). Mountain lions are the most important predator on adult deer, whereas coyote can kill a large number of young fawns (Ballard et al. 2001). Other predators include black bears (*Ursus americanus*), bobcats (*Lynx rufus*), golden eagles (*Aquila chrysaetos*), and possibly an occasional transient jaguar (*Panthera onca*) (Heffelfinger 2006).

#### *F.2.1.3.6 Competitors*

Coues white-tailed deer compete with other browsers and grazers on the landscape. Presence of cattle in whitetail habitat may induce temporary avoidance movement; however, competition with livestock is minimized by the fact that the higher elevations are not grazed as heavily as the lower elevations that would likely be used more intensively by deer. Welch (1959) reported that white-tailed deer actively avoid cattle and in some parts of the southwest overgrazing in dry years is still a problem for whitetails and their habitat (Galindo-Leal et al. 1997).

#### *F.2.1.3.7 Demographics*

Sparse or small populations can result in variations in sex ratios, age distributions, and birth and death rates, which can cause fluctuations in population size. Males tend to stay separate from females during most of the year only coming together to breed. In very small isolated populations, males may have trouble finding females leading to decreased recruitment. The composition of a herd can change hourly or day to day, forming loosely dispersed groups that keep in visual contact when food is readily available. During times of stress, access to adequate food is the main factor in lower recruitment and mortality (Hefflefinger 2006).

#### *F.2.1.3.8 Disease and Parasites*

Hemorrhagic disease has been identified occasionally in free-ranging whitetail deer, mule deer and bighorn sheep in Arizona (Noon et al. 2002) Coues white-tailed deer are susceptible to any number of diseases. The most common are caused by infection with either bluetongue or epizootic hemorrhagic disease. Adenovirus caused hemorrhagic disease, papilloms or fibropapillomas, foot and mouth disease, vesicular stomatitis, bovine viral diarrhea, and other miscellaneous viruses infect whitetails. Bacterial infections can cause sickness in Coues white-tailed deer including; dermatitis, bovine tuberculosis, lumpy jaw, leptospirosis, brain abscess and bovine brucellosis. Chronic wasting disease can infect white-tailed deer and has been documented in south-central New Mexico in close proximity to Coues white-tailed deer populations but not in Arizona.

Parasites are common in deer populations but may become only significant when deer are under stress. Nasal bots, mites, ticks louse flies, flea flies and screwworm are known external parasites. Internal parasites include elaeophorosis, commonly called carotid worm; foot/worm is very common, gastrointestinal nematodes of stomach worm, abdominal worm lungworm and tapeworm (Heffelfinger 2006)

### F-2.1.4 Change Agent Characteristics

This section of the conceptual model presents a narrative description of the primary change agents and current knowledge of their effects on Coues white-tailed deer populations. The main change agents identified are forms of altered dynamics: Livestock management, development, and climate change.

#### *F.2.1.4.1 Livestock Management*

Although cattle and deer do not typically utilize the same forage resources, competition with cattle for forage has the potential to affect fawn survival under conditions of drought or livestock overgrazing, where deer habitat overlaps with grazed areas. Ockenfels (1997) found that under moderate grazing regimes and “normal precipitation, there is little evidence of forage competition between deer and cattle. Heavy livestock use coupled with frequent drought periods reduces available browse (Knipe 1977) and can reduce perennial grass cover needed for fawn survival (D. Brown 1984, M. Brown 1984). Competition between deer and cattle can also occur for non-forage resources. The main food sources for this species are shrubs, and to a lesser extent, trees (Knipe 1977); Gallina et al. (1981) found that shrubs were the highest percentage (51%) of Coues' year-round diet, and together with trees accounted for 83% of deer diet in Mexico. Grasses constitute only a minor part of deer diet but could be important to deer in spring and year-round in certain areas.

**Stresses:** increased competition during drought periods.

**Responses:** decreased reproductive success

### *F.2.1.4.2 Development*

Human development influences Coues white-tailed deer by directly affecting home range, habitat and resources, and may fall into the categories of 1) military and border-defense related activity; 2) mining; 3) rural and urban development; 4) transportation in the form of roads, railroads and highways; and 5) energy development including new transmission lines, canals, and converting open space for solar and wind farming. These stressors are present in the Madrean Archipelago ecoregion where Coues white-tailed deer occur. Border infrastructure, including lighting, roadways, and frequent vehicle traffic, pose a significant barrier to wildlife movement north and south of the border and wherever they may transect home ranges. Large-scale mining can have a tremendous impact on ground water resources, water and air quality, and habitat availability. Rural and urban develop has a direct impact on ground water levels and habitat resources, and transportation and energy development results in habitat fragmentation and direct mortality from vehicle collisions. Increased human development also infers increased human presence: access to more remote areas for recreation and hunting may increase direct take from hunting, road kill, and other disturbance.

**Stresses:** reduced habitat/ open space, decreased water availability, increased habitat fragmentation, decreased water quality, habitat degradation, increased behavioral disruption, increased migratory barriers.

**Responses:** decreased population size, reduced recruitment, decreased home range, reduced migratory distance, decreased gene flow/ decreased genetic health, increased direct mortality rate, increased mortality rate of the population.

### *F.2.1.4.3 Climate Change*

Coues white-tailed deer rely greatly on specific habitat requirements to survive and thrive. These habitat requirements are affected by precipitation, temperature, seasonality, and other climatic conditions. Additional studies of these influences in consideration of a changing climate including increasing temperature extremes, increased fire intensity and frequency, and drought are needed. Although Coues white-tailed deer are able to obtain much of the water that they need from their diet, the absence or reduction of succulent forage requires deer to seek out additional surface water. Decreased water resources and increased presence of predators at available water sites may expose deer to greater risk of mortality, although O'Brien et al. (2006) found that contrary to common perception, rates of predation at human-made water catchments is relatively low so this may not be an issue. In addition, warmer temperatures and lower precipitation may lead to increased aquifer drawdowns by human populations due to both decreased infiltration and increased demand for water (Heinz Center 2011, SWCCN 2008). The National Drought Mitigation Center (2013) reports that the Madrean Archipelago ecoregion has been experiencing "Severe to Extreme" drought in the last year, and forecasts this condition will persist or intensify.

**Stresses:** increased change in vegetation structure, change in biotic community, decreased forage availability, decreased water availability, increased distance to water, increased predation, increased fire frequency and intensity, greater potential for disease transmission, increased average temperatures, hyperthermia.

**Responses:** decreased population size, reduced recruitment, increased daily foraging movements, increased seasonal movement distance, increased mortality rate.

### F-2.1.5 Ecological Status: Key Ecological Attributes and Indicators

This section of the conceptual model addresses Key Ecological Attributes and their potential indicators. The ecological status is a way of describing current status via criteria, functionality, or levels of attributes and asks if they are within the normal range of variation.

#### *F.2.1.5.1 Key Ecological Attributes*

Table F-6 identifies the key ecological attributes for this CE within the Madrean Archipelago ecoregion. A **key ecological attribute** of a focal ecological resource is a characteristic of the resource's biology, ecology, or physical environment that is critical to the resource's persistence in the face of both natural and human-caused disturbance. Alteration of such a characteristic beyond some critical range of variation will lead to the degradation or loss of the resource within decades or less. The KEAs table lists the identified key ecological attributes, with a brief definition, a rationale for why it is important for the CE, and a listing of stressors or change agents that might be affecting the key attribute.

**Table F-6. Key ecological attributes (KEA) of Coues white-tailed deer (*Odocoileus virginianus couesi)* in the Madrean Archipelago ecoregion.** Indicators for these KEAs can be used to determine the ecological status for this CE; see **Table F-2** for a list of the indicators assessed in this REA.

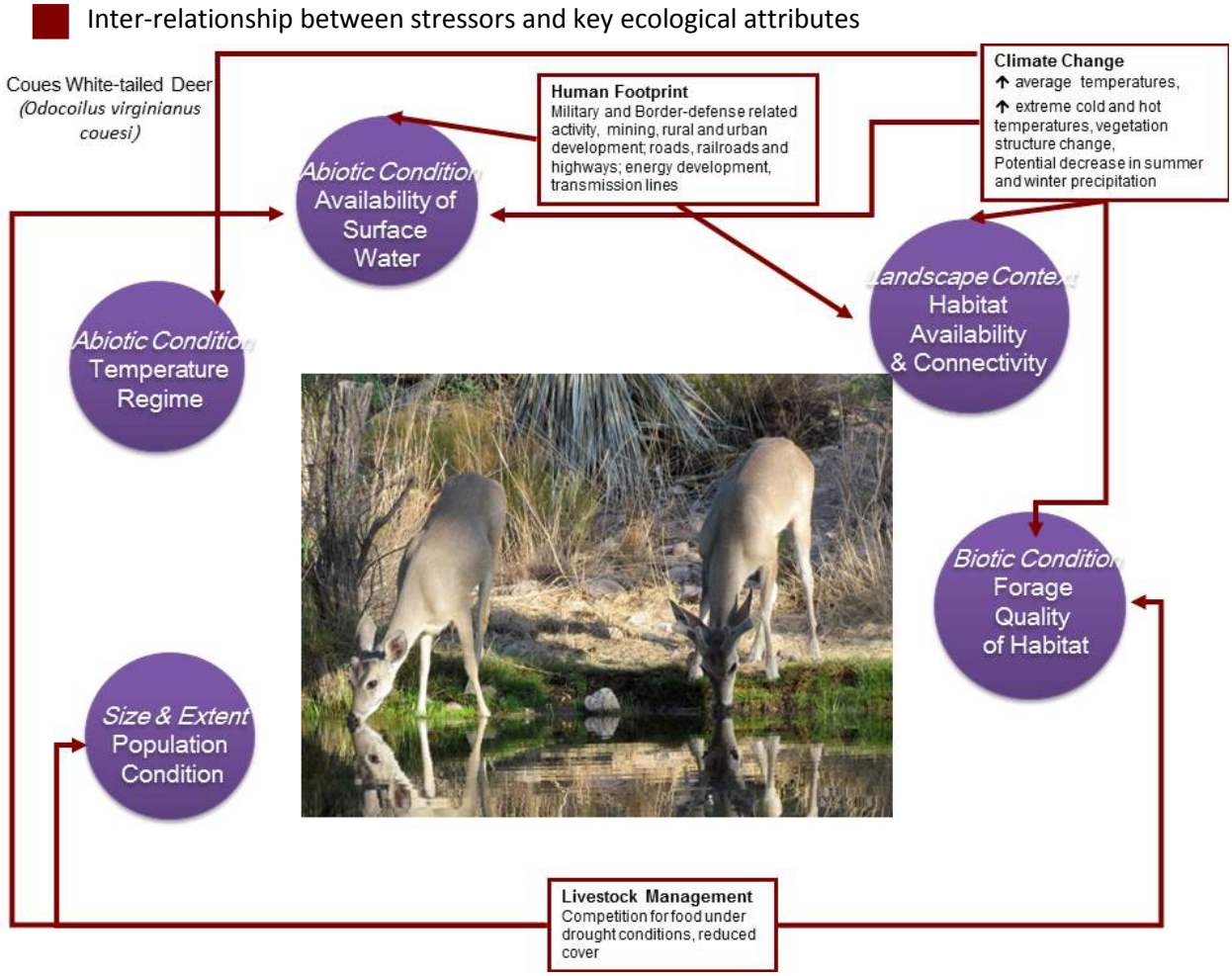
| ***KEA Class:* KEA Name** | **Definition** | **Rationale** | **Stressors** |
|---|---|---|---|
| ***Landscape Context:* Habitat Availability and Connectivity** | The extent and connectedness of mountainous Madrean evergreen woodland and mixed oak-pinyon juniper habitat that is intact. | Daily and seasonal movement between habitat patches within home range is essential for Coues white-tailed deer adaptability to changing forage and water availability, predator avoidance, and gene flow. Coues white-tailed deer prefer habitats with topographic ruggedness of between 20 and 29 degrees of slope (Ockenfels 1994). This habitat preference may restrict movement and home range size as water and forage conditions essential to Coues white-tailed deer survival shift or change on an elevational gradient. Coues white-tailed deer prefer mountainous habitat, associated with Madrean Evergreen Woodlands (Brown 1984). They are found primarily in woodland communities consisting of evergreen oaks or in mixed oak-pinion-juniper. They are also found in Ponderosa pine forests, in desert scrub, in deciduous forests and occasionally in spruce-fir (Hoffmeister 1986). | Stressors include destruction and fragmentation of habitat by human development. Coues white-tailed deer habitat and historical herd home ranges extend north and south of the U.S.-Mexico border. White-tail populations occurring in the Madrean ecoregion are directly affected by the physical barriers to gene flow, behavior disturbance, and habitat loss and destruction, among other stressors, caused by border infrastructure and other border-related activities. Increased road density, direct loss of foraging habitat and increased human interaction as well as disturbance of diurnal bedding sites can have a negative impact on Coues white-tailed deer populations (Ockenfels 1994). |
| ***Size/Extent: Population Condition*** | Population size and demographics of Coues white-tailed deer herds, including sex and age ratios. This may also include evidence of population health (presence of disease or signs of chronic physical stress) which over time affect population size and demographics. | Population size and demographic structure (buck to doe ratios) directly indicates the viability and health of the population; demographics can affect breeding systems affecting genetic diversity. Densities greater than 15/mi$^2$ can indicate population stress due to over-population (Heffelfinger 2006). | Stressors affecting population condition include quality and availability of forage, presence of infectious disease and decreased water availability. |

| *KEA Class:* KEA Name | Definition | Rationale | Stressors |
|---|---|---|---|
| ***Biotic Condition:*** **Forage Quality** | Relative composition, abundance and species richness of native forage plants, average vegetation density and height within habitat areas. | Composition of grasses, forbs, succulents, shrubs and trees indicate forage quality. Coues white-tailed deer require a diversity of vegetation for forage and thrive with higher plant species richness. They are opportunistic herbivores; high-protein, succulent forbs, shrubs and trees make up the majority of their diet, with grasses equaling a small portion of their diet. Availability of both native shrubs and trees is shown to be essential for fawn survival (Ockenfels 1991).Coues white-tailed deer require a diversity of vegetation for forage and thrive with higher plant species richness. They are opportunistic herbivores; high-protein, shrubs make up the majority of their diet, with grasses and other plants making up a minor portion of their diet. Availability of both native shrubs and trees is shown to be essential for fawn survival (Ockenfels 1991). Coues white-tailed deer are associated with vegetation within reach. Access to vegetation ranging from 2.5 inches to approximately 60 inches is necessary. Having access to lower shrubs, forbs and trees is critical for fawns. | Historically, the main stressors for white-tailed deer have been the competition for food with grazing cattle, Hunting mismanagement and loss of habitat (Heffelfinger 2006). During times of drought, livestock management acts as a change agent for white-tailed deer through increased competition for forage and water sources. Grazing allotment boundaries may indicate locations of habitat with direct competition for forage and water from cattle, and possible reduced forage quality. Locations of stock waters are indicators of heavy use of vegetation. |
| ***Abiotic Condition:*** **Availability of Surface Water** | Annual and seasonal precipitation and the resulting density, distribution and distance to surface water accessible by Coues white-tailed deer. | Annual reproductive success in Coues white-tailed deer has shown to depend heavily on winter rainfall. Winter precipitation increases high nutrient forage and surface water resources in spring and early summer. Coues white-tailed deer require regular access to surface water and are usually found within a 3/4-mile radius of an accessible, open water source (Ockenfels 1994). | Changes in precipitation regime and average temperatures may change the availability of surface water. Human use of water, and competition from livestock management will exacerbate this. |
| ***Abiotic Condition:*** **Temperature Regime** | The pattern of average temperature fluctuations; the duration, intensity and seasonal timing of temperature extremes. | Coupes white-tailed deer rely greatly on specific habitat requirements to survive and thrive. These habitat requirements are affected by precipitation, temperature, seasonality, and other climatic conditions. Although Coues white-tailed deer are able to obtain much of the water that they need from their diet, the absence or reduction of succulent forage requires deer to seek out additional surface water. | High temperatures in conjunction with low precipitation can affect forage quantity and quality, surface water availability, and may directly result in fawn and juvenile mortalities due to hyperthermia. Water source congestion during the hottest period of the year can cause over browsing. If conditions persist survival may be threatened (Truett 1971). Climate change also has the potential to cause additional change in the hydrologic regime, through its effects on precipitation form (snow vs. rain), spatial distribution, magnitude, and timing; and through its effects of evapotranspiration rates both within the riparian zone and across the surrounding watershed. Climate change may also cause changes in human water use. |

### F-2.1.6 Conceptual Model Diagram

A conceptual model diagram for each CE provides a visual summary representation of the Key Ecological Attributes and species' life cycle (Figure F-10). These diagrams are intended to show how various stressors interact with categories of Key Ecological Attributes to highlight important drivers for species ecological integrity. The arrows indicate relationships between stressors and but do not indicate the nature of the influence (i.e. positive or negative).

**Figure F-10. Conceptual model diagram for the North American Coues white-tailed deer, showing key ecological attributes** (by class) for the Coues white-tailed deer.




## F-2.2 Considerations and Limitations

Forage quality, including the diversity of plant species and the availability of high-protein trees and shrubs, is important for Coues white-tailed deer fawn survival (Ockenfels 1991). This was an aspect of habitat that was not assessed due to a lack of adequate data.

## F-2.3 Ecological Status Assessment Results and Interpretation

This section of the CE summary presents and discusses the results of the ecological status assessment for Coues white-tailed deer. It addresses each indicator separately, and then addresses the overall assessment, which integrates the results of all individual indicators assessed for the CE. The results are

presented using a common framework, in which the status of an indicator – or the combination of all indicators – is scored on a scale from 0.0 to 1.0, where 0.0 indicates a condition of complete replacement of reference ecological conditions due to the impacts of stressors, and 1.0 indicates a condition of no alteration of reference ecological conditions. The same yellow-to-blue color ramp is used for all results, with yellow representing low scores, green moderate scores, and dark blues high scores.

### F-2.3.1 Current Ecological Status: Development, Fire Regime, Invasives

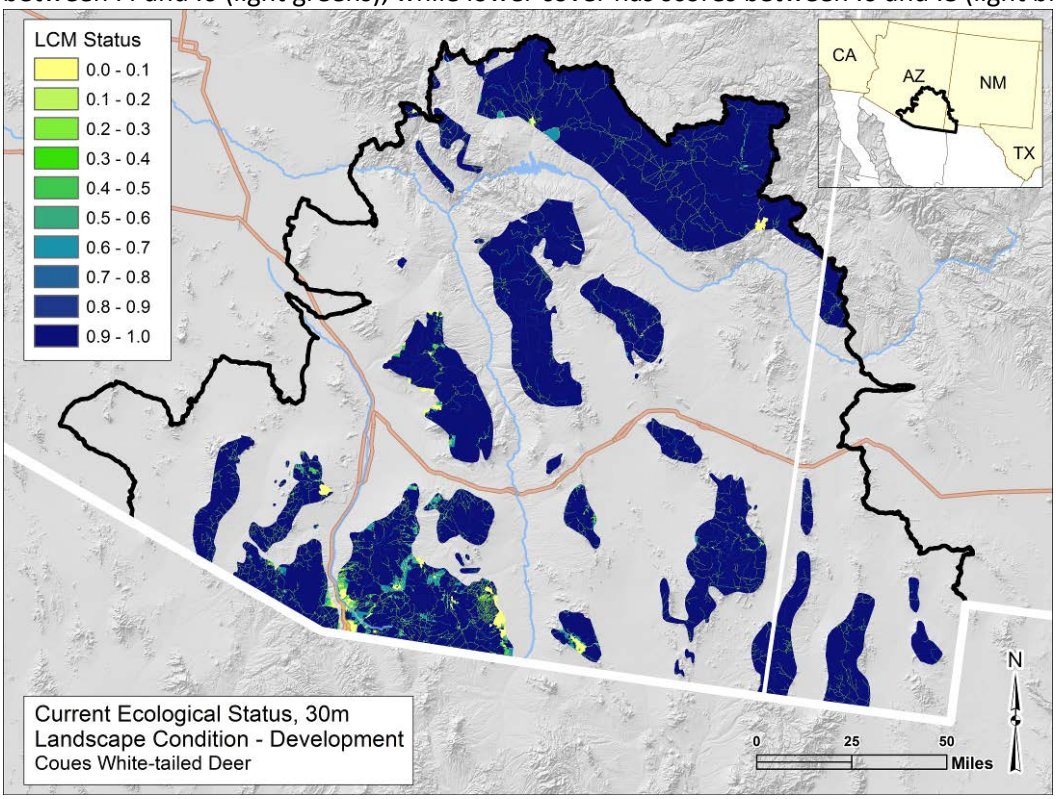
The results of the assessments for the three individual indicators for the KEAs for Coues white-tailed deer are shown in the three maps in Figure F-11 below.

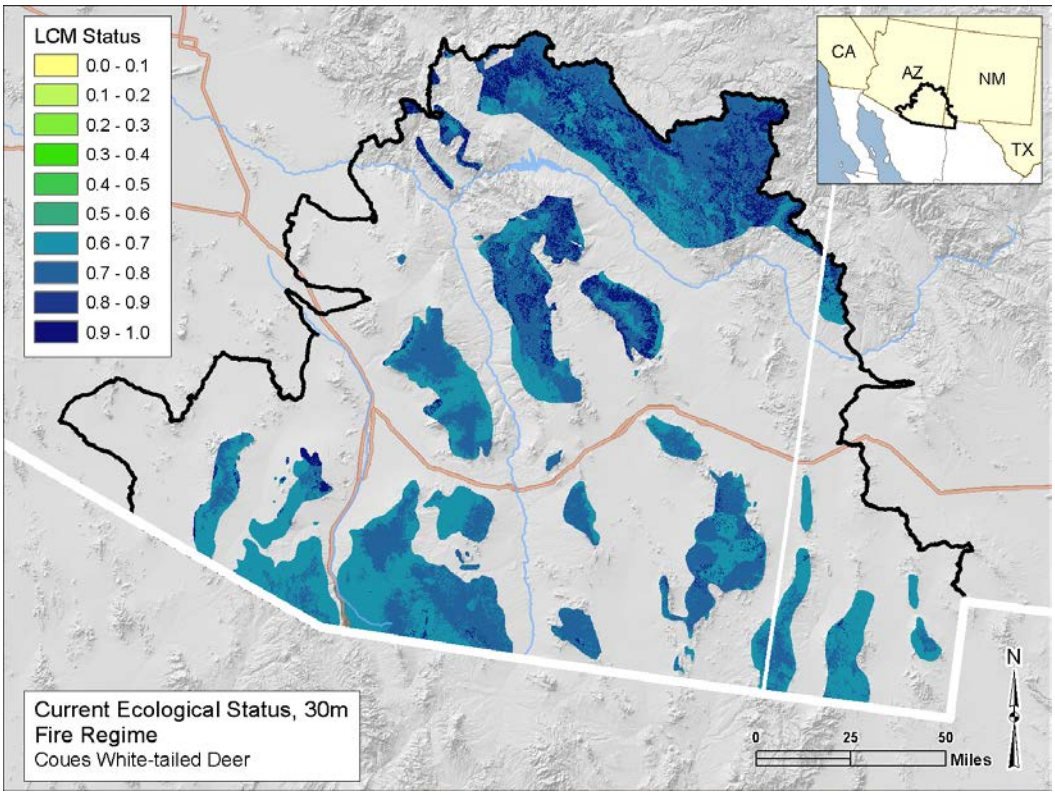
Because Coues white-tailed deer prefer the Madrean woodlands in the lower and middle elevations of many of the sky islands of the ecoregion, their habitat is less impacted by human development features, which tend to be in the valleys and lower elevations of the ecoregion. The development indicator results shown in the first map of Figure F-11 show the majority of its distribution having little impact from development. There are a few discrete areas of extremely poor condition, such as the areas around the cities of Nogales and Sierra Vista in the extreme southern part of the ecoregion, and around mining operations west of Interstate 19 in the southwest part of the MAR and around Morenci in the northern part of the MAR. In addition, major road corridors crisscross the deer's distribution. Where development is present, the primary impact to the Coues deer is direct loss of habitat.

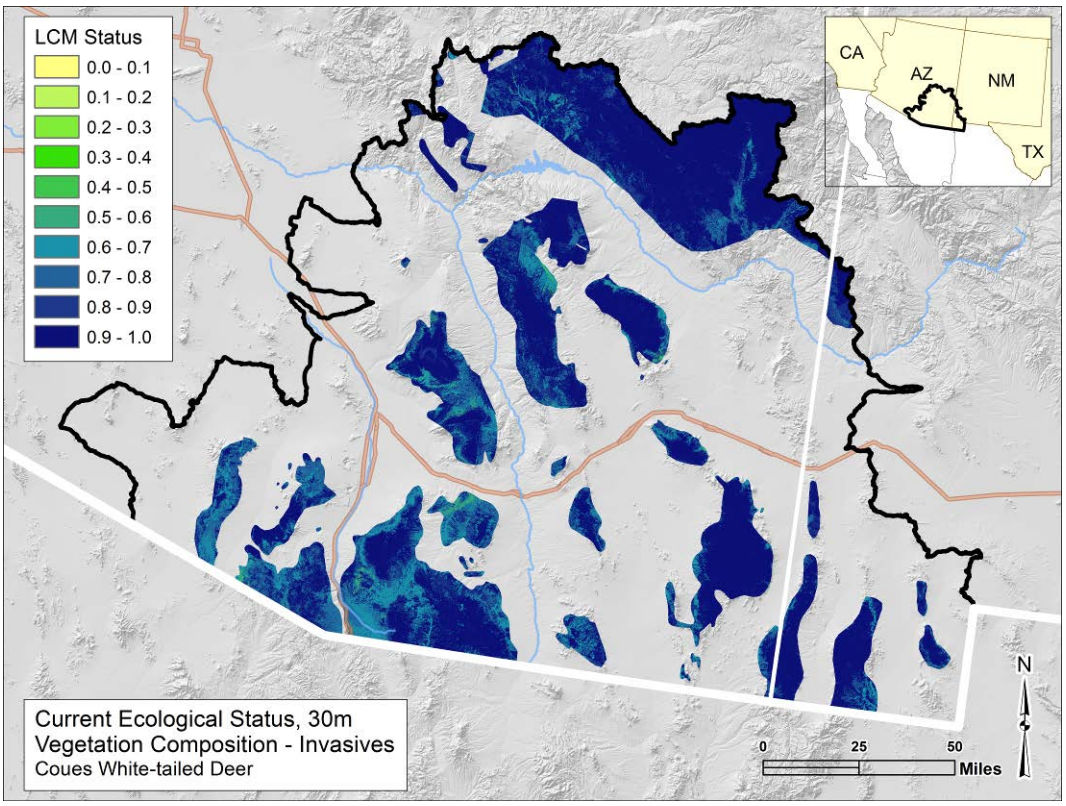
Fire regime results, shown in the second map in Figure F-11, indicate that fire regimes in the ecological systems present in the Coues white-tailed deer distribution are largely split between severe ecological departure (scores of 0.65, shown in turquoise blue) and moderate ecological departure (scores of 0.75); very little of its distribution shows no departure in fire regimes. Although the woodland habitat preferred by Coues white-tailed deer is undergoing changes resulting from altered fire regimes, the type and diversity of food plants it can consume and variety of habitat it utilizes suggest that altered fire regimes may be tolerated by this species.

The invasives results (third map in Figure F-11) show that a fair amount of the deer's distribution is little impacted either by mesquite or by non-native grasses and forbs (darkest blue color). However, there are still significant areas, especially in the southwestern portion of its distribution, that have high cover of mesquite. Because this species utilizes mesquite for cover and forage, mesquite encroachment may have a relatively low impact unless the habitat is converted to a largely mesquite-dominated shrubland.

**Figure F-11. Scores for three indicators for Coues white-tailed deer: development indicator (1st map), fire regime departure indicator (2nd map), and invasive species indicator (3rd map) for each 30m pixel.** Yellow (equivalent to 0) indicates high impacts from the CA, dark blue (equivalent to 1) indicates little to no impact from the CA. At the ecoregion scale, many development features are not readily visible (e.g., secondary roads or highways, railroads, small agricultural fields). Only 3 classes of fire regime condition are scored: no to little departure (dark blue), moderate departure and severe departure (lighter blues). For invasives, higher cover of mesquite or invasive exotics have scores between .4 and .6 (light greens), while lower cover has scores between .6 and .8 (light blues).




LCM Status
0.0 - 0.1
0.1 - 0.2
0.2 - 0.3
0.3 - 0.4
0.4 - 0.5
0.5 - 0.6
0.6 - 0.7
0.7 - 0.8
0.8 - 0.9
0.9 - 1.0
CA
AZ
NM
TX
Current Ecological Status, 30m
Fire Regime
Coues White-tailed Deer
N
0
25
50
Miles


LCM Status
0.0 - 0.1
0.1 - 0.2
0.2 - 0.3
0.3 - 0.4
0.4 - 0.5
0.5 - 0.6
0.6 - 0.7
0.7 - 0.8
0.8 - 0.9
0.9 - 1.0
CA
AZ
NM
TX
Current Ecological Status, 30m
Vegetation Composition - Invasives
Coues White-tailed Deer
N
0
25
50
Miles

### F-2.3.2 Current Ecological Status: Full Scenario

The results of the three individual status indicators were combined to get an overall ecological status score, per pixel of the CE's distribution, as shown in the first map of Figure F-12 below. The overall, per-pixel status scores for the CE were then averaged across each of the 4 km grid cells, as shown in the second map of Figure F-12 below.

When the three sets of indicator scores are aggregated to obtain the overall ecological status scores (as shown in the first map of Figure F-12), much of the habitat for this species is in moderate to higher condition, with scores around 0.5 to 0.8 in many areas (Figure F-13). In visually comparing the maps of scores for the three individual indicators (Figure F-11) to the map of the overall status scores (first map of Figure F-12), the cumulative impacts of the change agents reflected by the indicators are clear. The altered fire regime and invasives cover are both driving down scores in much of the ecoregion, and scores drop even lower where cities, major road corridors, or other development features are also present. These same patterns are reflected when the overall status scores are averaged across each 4 km reporting unit, as shown in the second map of Figure F-12 above.

Based on the indicators assessed in this REA, changes in habitat quality and vegetative structure from altered fire regime is having the most widespread effect on habitat within this species' geographic distribution. While mesquite cover is significant in some portions of its distribution, this species can utilize mesquite for forage and cover. It is notable that higher elevations areas of larger mountain ranges tend to have higher ecological status scores. Direct habitat loss or degradation (from existing development) is much more localized and tends to be in terrain that is less frequently occupied by Coues white-tailed deer.

**Figure F-12. Overall ecological status scores for Coues white-tailed deer for all three indicators combined (development, fire regime and invasives) for each 30m pixel (top) and 4km grid cells (bottom).** Yellow scores (equivalent to 0) indicate high impacts from the CAs, dark blue (equivalent to 1) indicate little to no impact from the CAs. In the second map, the score for each 4km cell is an average of the overall ecological status scores of the 30m pixels within the 4km cell that were scored for the CE.

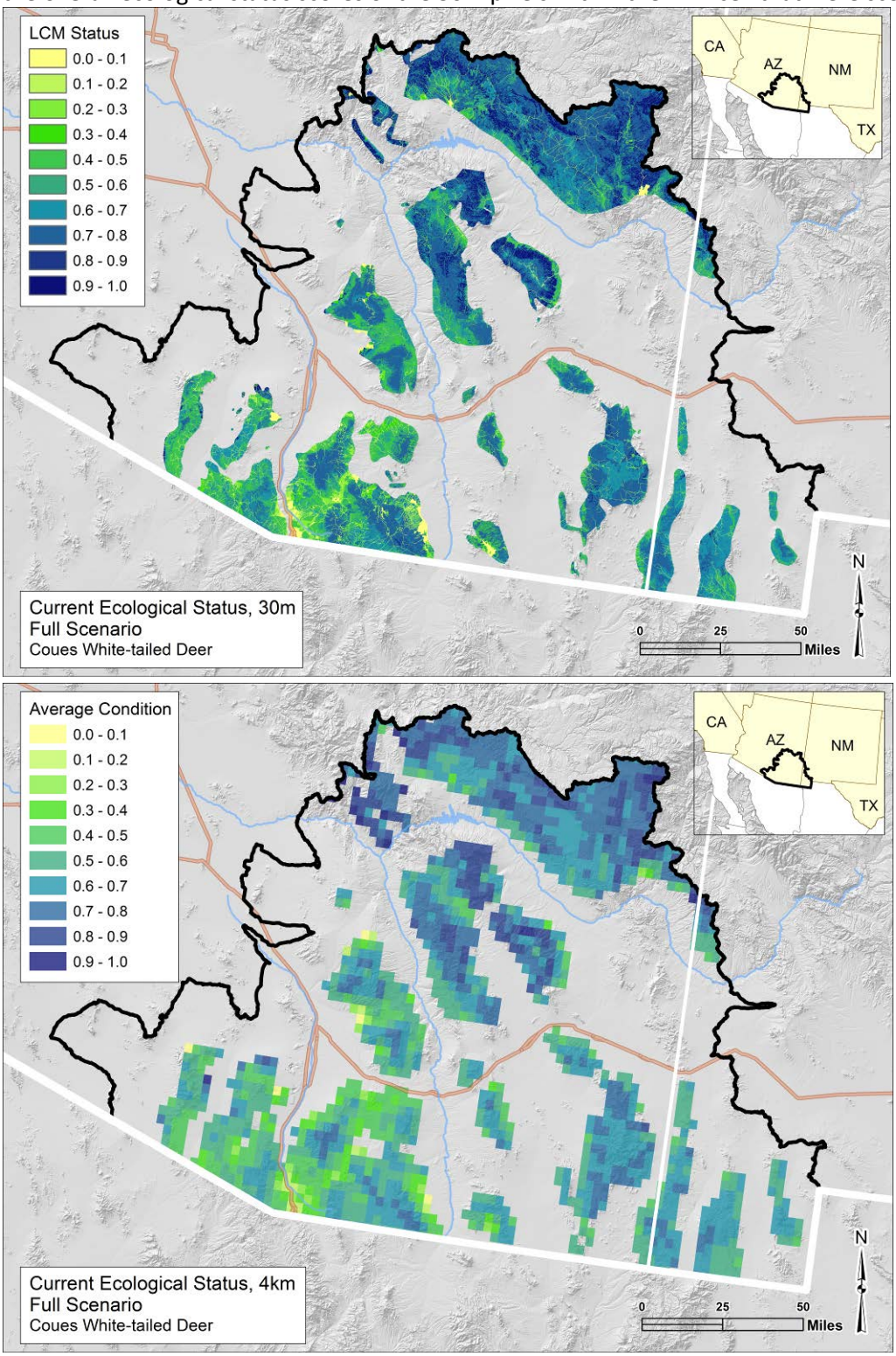
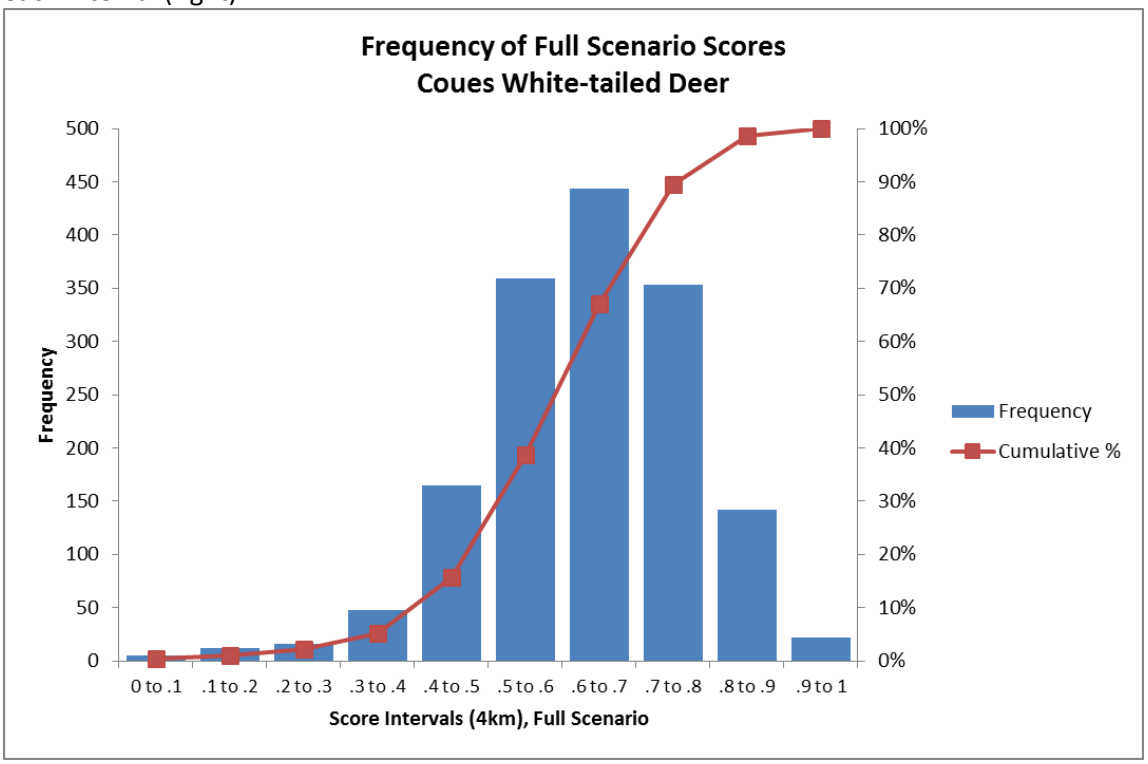


**Figure F-13. Frequency distribution of the 4km ecological status scores for Coues white-tailed deer, with cumulative percent.** The x-axis represents the 0.1 increment scoring intervals, while the y-axis shows the number of grid cells in each interval (left) and the cumulative percentage of the grid cells for each interval (right).




## F-2.4 References for the CE


Arizona Game and Fish Department (AZGFD). 2013. HabiMap™ Arizona. Mammal distributions, last revision 2011.Available: http://www.habimap.org. Accessed 18 January 2013.

Arizona Wildlife Linkages Workgroup. 2006. Arizona Wildlife Linkages Assessment. Arizona Department of Transportation, Phoenix, AZ. Available: http://www.azdot.gov/inside_adot/OES/AZ_WildLife_Linkages/assessment.asp.

Ballard, Warren B. 2013. DEER-PREDATOR RELATIONSHIPS. at: http://texnat.tamu.edu/files/2010/09/008.pdf. Department of Range, Wildlife, and Fisheries Management, Texas Tech University, Box 42125, Lubbock TX 79409, USA.

Bello, Joaquin, Sonia Gallina, and Miguel Equihua, 2001 Characterization and habitat preferences by white-tailed deer in Mexico. J. Range Manage. 54: 537–545.

Brown, D.E. and C.H. Lowe. 1982. Biotic Communities of the Southwest [map]. Scale 1:1,000,000. General Technical Report RM-78. U. S. Forest Service, Fort Collins. Reprinted (and revised) 1994 by University of Utah Press, Salt Lake City.

Brown, D.E. 1983. The Wolf in the Southwest: The Making of an Endangered Species. The University of Arizona Press, Tucson 195pp.

Brown, D. E. 1984. The effects of drought on white-tailed deer recruitment in the arid southwest. Pages 7-12 in P. R. Krausman and N. S. Smith, eds. Deer in the Southwest: a workshop. University of Arizona, Tucson. 131 pp.

Brown, M. T. 1984. Habitat selection by Coues white-tailed deer in relation to grazing intensity. Pages 1-6 in P. R. Krausman and N. S. Smith, eds. Deer in the Southwest: a workshop. University of Arizona, Tucson. 13l pp.

Carmony, Neil and David Brown, 1991. MEXICAN GAME TRAILS. Americans Afield in Old Mexico, 1866-1940. Univ. of Oklahoma Press, Norman. 270 pp.

Melanie Bucci, Natasha Kline and Don E. Swann. 2008. Ungulate Surveys at Saguaro National Park: Past, Present, and Future

Carmony N. B. 1985. Elliot Coues and the White-tailed deer of the southwest. Arizona Wildlife. Special edition of Arizona Wildlife Views. Arizona Game and Fish Department, Phoenix.

Comer, P., and J. Hak. 2012. Landscape condition model of the western United States. NatureServe, Boulder, CO. In preparation.

Ffolliott , Peter F., Hui Chen, Gerald J. Gottfried and Cody L. Stropki, 2012. Coues White-Tailed Deer and Desert Cottontail in the Southwestern Oak Savannas: Their Presence Before and After Burning Events. Journal of the Arizona-Nevada Academy of Science, 44(1):1-5.

Findley, J.S., A.H. Harris, D.E. Wilson, and C. Jones. 1975. Mammals of New Mexico. University of New Mexico Press, Albuquerque, 360pp.

Felger, Richard Stephen (Editor), Bill Broyles (Editor) 2007. Dry Borders: Great Natural Reserves of the Sonoran Desert. University of Utah Press, Salt Lake City 799 pp.

Fulbright , Timothy Edward & Alfanso Orteg-S 2006. White-tailed Deer habitat: Ecology and Management on Rangelands. Texas A&M University Press

Galindo-Leal, Carlos, A. Morales and M. Weber., 1993. Distribution and Abundance of Coues deer and cattle in Michilia Bioshere Reserve, Mexico. The Southwestern Naturalist, 38(2): 127-135.

Geist, V. 1998. Deer of the world: Their evolution, behaviour and ecology. Stackpole Books, Mechanicsburg, Pennsylvania. 421pp.

Hansen, R.M. and R.C. Clark. 1977. Foods of elk and other ungulates at low elevations in northwestern Colorado. J. Wildl. Manage. 41:76-80.

Hanson, Roseann Beggy, Jonathan Hanson 2003. Southern Arizona Nature Almanac. University of Arizona Press, Tucson. 324 pp.

Heffelfinger, Jim 2006. Deer of the Southwest: A Complete Guide to the Natural History, Biology, and Management of Southwestern Mule Deer and White-Tailed Deer. Texas A&M University Press, College Station 282 pp.

Heinz Center for Science Economics and the Environment. 2011. Managing and monitoring Arizona's wildlife in an era of climate change: Strategies and Tools for Success. Report and Workshop Summary, prepared for the U.S. Department of Interior Bureau of Land Management and the Arizona Game and Fish Department. Available: http://www.heinzctrinfo.net/wildlife/wordpress/wp-content/uploads/2011/07/Low-res-Heinz-Arizona-Report_Final_7.11.11.pdf.

Hernández Fernández, M. and E. Vbra 2005. A complete estimate of the phylogenetic relationships in Ruminantia: a date species-level supertree of extant ruminants. Biological Reviews 80:269-302.

Hesselton, W.T., and P.M. Hesselton. 1982. White-tailed deer. Pages 878-901 in J.A. Chapman and G.A. Feldhamer, eds. Wild Mammals of North America. John Hopkins Univ. Press, Baltimore. 1147ppp.

Hoffmeister, D.F. 1986 Mammals of Arizona. University of Arizona Press, Tucson. 602pp.

The Integrated Taxonomic Information System (ITIS). 2013. Available: www.itis.gov. Accessed 18 January 2013.

Knipe, T. 1977. The Arizona whitetail deer. Spec. Rep. 6 Arizona Game and Fish Dept., Phoenix 108pp.

Krausman, P.R. and E.D. Ables 1981.Ecology of the Carmen Mountains white-tailed deer. Scientific monograph Series No. 15. U.S. Department of the Interior, National Park Service, Washington, D.C. 114pp.

Leopold, A.S. 1959. Wildlife of Mexico. University of California Press, Berkeley 568pp. "Fauna Silvestre de Mexico. Instituto Mexicano de Recursos Naturales Renovables. México. D. F."

Leopold, R.A. 1966. A Sand County Almanac. Oxford University Press and Random House, New York. 295pp.

Lopez, Roy G. 2006. Genetic Structuring of Coues white-tailed deer in the Southwestern United States: A Thesis Submitted in Partial Fulfillment of the Requirements for the Degree of Master of Science in Forestry. Northern Arizona University

Mandujano, Salvador, Christian A. Delfín-Alfonso and Sonia Gallina. 2010. Comparison of geographic distribution models of white-tailed deer Odocoileus virginianus (Zimmermann, 1780) subspecies in Mexico: biological and management implications, THERYA. Vol. 1(1): 41-68

Manghini, M. T. and N. S. Smith. 1990. Water use and diurnal seasonal ranges of Coues white-tailed deer. Pages 21-34 in P. R. Krausman and N. S. Smith, eds. Managing wildlife in the southwest. Arizona Chapter of the Wildlife Society, Phoenix. 262 pp.

McCabe, R.A. and A. S. Leopold, 1951. Breeding Season of the Sonora white-tailed deer. Journal of Wildlife Management 15:433-34.

National Drought Mitigation Center. 2013. U.S. Drought Monitor – West [web application]. Lincoln, NE. Available: http://droughtmonitor.unl.edu/DM_west.htm.

NatureServe. 2007. InfoNatura: Animals and Ecosystems of Latin America [web application]. Version 5.0. Arlington, Virginia, USA. Available: http://www.natureserve.org/infonatura. Accessed 18 January 2013.

New Mexico Game and Fish. 2013. at: http://www.wildlife.state.nm.us/.

Noon, T.H., S.L.Wesche, J.R. Heffelfinger, A.F. Fuller, G.A. Bradley, C. Reggiardo, Hemoragic disease in Arizona deer. Journal of Wildlife Diseases 38:177-181.

O'Gara, B.W. 1999.Antilocapraamericana. In: D. E. Wilson and S. Ruff (eds).The Smithsonian Book of North American Mammals, UBC Press, Canada. Pp. 339-341.

O'Brien, C. S., R. B. Waddell, S. S. Rosenstock, and M. J. Rabe. 2006. Wildlife use of water catchments in southwestern Arizona. Wildlife Society Bulletin 34:582-591

Ockenfels, R. A. Coues White-tailed Deer (Odocoileus virginianus couesi): An annotated Bibliography. Arizona Game and Fish Department Special Report Number 18, Phoenix. 106 pp.

Ockenfels, R. A., D. E. Brooks, and C. H. Lewis. 1991. General ecology of Coues white-tailed deer in the Santa Rita Mountains. Ariz. Game and Fish Dept. Technical Report 6, Phoenix. 73 pp.

Ockenfels, R. A., D. E. Brooks. 1994. Summer diurnal bed sites of Coues white-tailed deer. Journal of Wildlife Management 58:70-75.

Ockenfels, R. A., D. E. Brooks. 1997. Coues White-tailed Deer Dietary Overlap with Cattle in Southern Arizona. Pp.89-96 in J.C. deVos ed., Proceedings of the Deer/ Elk Workshop, Rio rico, Arizona 224pp.

Patrick-Birdwell, C., S. Avila-Villegas, J. Neeley, L. Misztal, and N. Deyo. May 2012. Mapping and Assessing the Environmental Impacts of Border Tactical Infrastructure in the Sky Island Region. IN: Proceedings of *Connecting mountain islands and desert seas: biodiversity and management of the Madrean Archipelago III*, Tucson, AZ. In Press.

Robinette, W.L., N.V. Hancock, and D.A. Jones. 1977. The Oak Creek mule deer herd in Utah. Publication No.77-15. Utah Division of Wildlife Resources, Salt Lake City. 148pp.

Smith, N. S. 1984. Reproduction in Coues white-tailed deer relative to drought and cattle stocking rates. Page 13-30 in P. R. Krausman and N. S. Smith, eds. Deer in the southwest: a workshop. University of Arizona, Tucson. 131 pp.

Southwest Climate Change Network (SWCCN). 2008. Drought and the Environment. Available: http://www.southwestclimatechange.org/impacts/land/drought in October 2010.

Tafoya ,Jeffery J., V.W. Howard, Jr., Jon C. Boren. 2000? Diets of Elk, Mule Deer& Coues White-tailed deer on Fort Bayard in Southwestern New Mexico. Range Improvement Task Force, Agricultural Experiment Station, Cooperative Extension Service. New Mexico State University, Las Cruces Report 52. 54pp.

Truett, J.C. 1971. Ecology of the desert mule deer, Odocoileus hemionus crooki Mearns, in southeastern Arizona. Ph.D. Dissertation, University of Arizona, Tucson. 64pp.

United States Forest Service (USFS), 2013. Index of Species Information: Wildlife Species: Odocoileus virginianus. At: www.fs.fed/database/fels/animals/mammal/odvi/all.html.

Verme, L.J. 1965. Reproduction studies on penned white-tailed deer. Journal of wildlife Management 29:74-79.

Welch, J.M. 1960. A study of seasonal movements of white-tailed deer (Odocolileus virginianus couesi) in Cave Creek basin of the Chiricahua Mountains. Master's Thesis, University of Arizona, Tucson. 79pp.

Welles, P. 1959. The Coues deer at Coronado national Memorial. Unpublished manuscript. Coronado National Memorial, National Park Service, 29pp.

White, Robert W. 1961. Some foods of the White-Tailed Deer in Southern Arizona. The Journal of Wildlife Management, Vol. 25. No 4, Oct. 1961.

# *F-3* Desert Bighorn Sheep *(Ovis canadensis)*

## F-3.1 Conceptual Model

### F-3.1.1 Classification Comments

Ancestors of the Desert Bighorn Sheep, also referred to as Mountain Sheep or *Borrego cimarrón*, in the Bovidae family, originated in the mountain and desert regions of Eurasia and crossed the Bering land bridge in the late Pleistocene (Boyce et al 1999, San Diego Zoo Global Library 2009). This Old World sheep evolved into many different and distinct wild sheep (Figure F-14), each adapted to their native habitats.

The Integrated Taxonomic Information System (www.itis.gov; accessed 23 March 2013) recognizes seven subspecies of bighorn sheep: an extinct subspecies of American bighorn (*Ovis canadensis auduboni,* Merriam, 1901)*;* California bighorn sheep *(O. c. californiana,* Douglas, 1829); Rocky Mountain bighorn sheep *(O. c. canadensis,* Shaw, 1804); Peninsular desert bighorn sheep *(O. c. cremnobates ,*Elliot 1904); Desert bighorn sheep*(O. c. Mexicana,* Merriam, 1901); Nelson's desert bighorn sheep *(O. c. nelson;* Merriam, 1897); and the Weems desert bighorn sheep (*O. c. weemsi,* Goldman, 1937)(ITIS, accessed 23 March 2013; Boyce et al 1999).

There has been some disagreement regarding the number of subspecies and the validity of some population distinctions. Wehausen and Ramey (2000) concluded that the extinct American bighorn and populations of California bighorn occurring in British Columbia and Washington are genetically grouped with Rocky Mountain bighorn; extirpated native populations of California bighorn in northeastern California, Oregon, northern Nevada, and southwestern Idaho are actually Nelson's bighorn; and the California bighorn is restricted to just the central and southern Sierra Nevada in California. Wehausen and Ramey (2000) also suggested that the California and Rocky Mountain bighorn Sierra Nevada population in Yakima County, Washington have a unique mtDNA haplotype (Wehausen et al. 2005; USFWS 2008) and should be recognized as a new distinct subspecies (*Ovis canadensis sierrae*).

Ramey (1995) examined mtDNA variation in Rocky Mountain bighorn populations in the southwestern United States and adjacent northern Baja California and found a lack of similar traits between mtDNA haplotype distributions to justify many of the current subspecies designations. Additionally, Ramey (1993) and Wehausen and Ramey (1994) found no morphological support for the separation of desert sheep into four subspecies. Ramey (1995) suggested that Nelson's, Mexican, Peninsular, and Weems desert sheep populations probably should be recognized as a single polytypic subspecies (Nelson's desert bighorn sheep) on the basis of morphological and mtDNA data. As such, the subspecies would include bighorn sheep populations in the Peninsular Ranges of southern California and Baja California. A study of nuclear DNA markers is needed to test this taxonomic arrangement (Ramey 1995). The genetic integrity of these taxonomic subspecies could potentially be complicated by translocation and reintroduction efforts to bring back declining or extirpated populations in the region; bighorn sheep reintroduction efforts began in Arizona in 1979 (McKinney et al 2006).

**Figure F-14. Desert bighorn sheep (*Ovis canadensis*) in southwestern New Mexico.** Courtesy of New Mexico Game and Fish Department website (http://www.wildlife.state.nm.us/conservation/bighorn)

The focus of the Madrean REA is on desert bighorn sheep *(O. c. mexicana*, Figure F-14), unless otherwise noted.

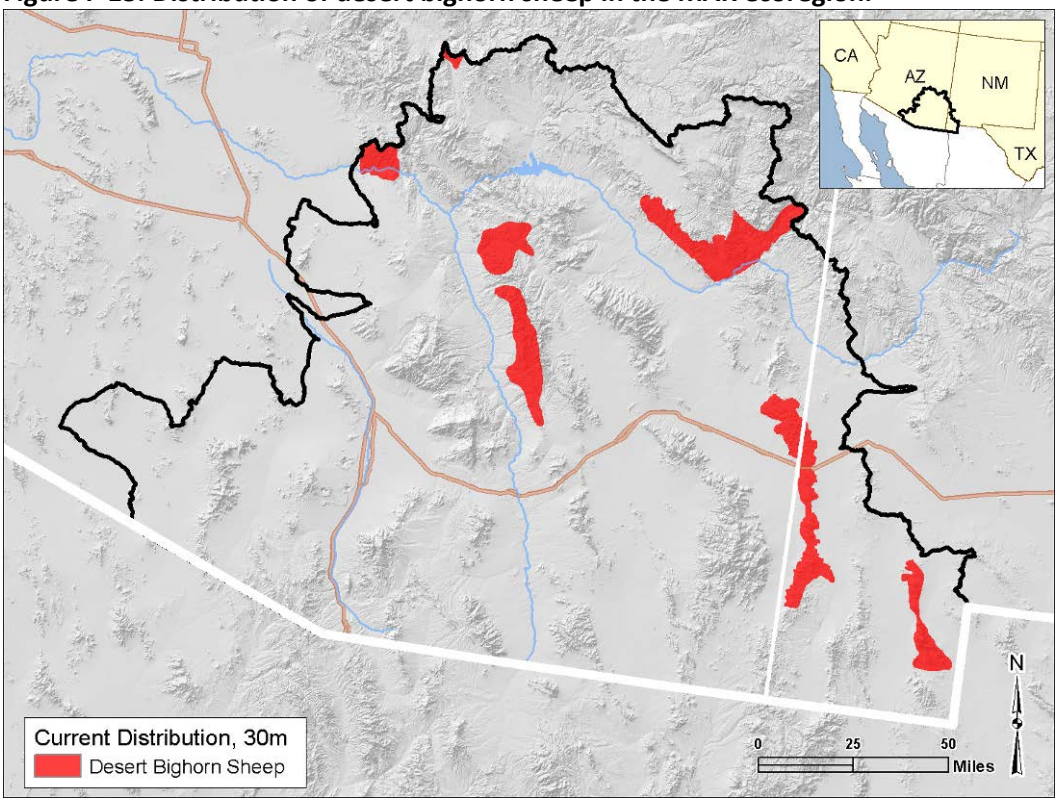
## F-3.1.2 Biology and Distribution Status

### *F.3.1.2.1 Distribution*

*North America*: The bighorn sheep historically ranged across the majority of the mountainous western United States, extending from British Columbia to southern Baja California and northwestern Sonora and Chihuahua (Mexico), and as far east as portions of North Dakota, South Dakota and Nebraska and parts of western Texas.

*Madrean Archipelago ecoregion*: The Mexican desert bighorn sheep (*O. c. mexicana*) is the only subspecies historically distributed throughout the majority of Arizona, New Mexico and northern Mexico (Cowan 1940, Hall 1981).Today bighorn are reduced to fragments of their previous range, with remaining populations restricted to small isolated mountain and canyon areas. Within the Madrean Archipelago ecoregion, occupied and unoccupied bighorn habitat includes the Peloncillo Mountains north of Interstate 10 at San Simon, AZ and extending into New Mexico; the Galiuro Mountains and Galiuro Wilderness encompassing Redfield Canyon Wilderness and Kelsey Canyon; the Galiuro Mountains encompassing Table Mountain and Arivaipa Canyon northeast of Mammoth, AZ; the Superstition Mountains and Superstition Wilderness near the Salt River north of Superior, AZ; Pusch Ridge Wilderness in the Santa Catalina Mountains; the Silverbell Mountains and Ironwood Forest National Monument west of Tucson, AZ; the Hatchet Mountains of New Mexico; and New Mexico's Red Rock area of the Gila River (AZGFD 2013, NMDGF 2013).

**Figure F-15. Distribution of desert bighorn sheep in the MAR ecoregion.**




### *F.3.1.2.2 Population*

In 1991, the total population of desert bighorn in North America was estimated to be 23,055 (Valdez and Krausman 1999). In 1992, the Endangered Species Technical Bulletin 17(3-8):6 reported that Peninsular Ranges desert bighorn populations in the U.S. declined to 1,171 by 1979 and to fewer than 400 in the early 1990s. Helicopter-based surveys of the principle mountain ranges in Sonora in 1992 produced an estimate of 2,000 wild sheep for Sonora (ICUN). Desert bighorn populations occurring in Mexico have CITES Appendix II status (CITES 2010; www.itis.gov, accessed 23 March 2013). California bighorn occurring in Sierra Nevada, CA and the Rocky Mountain bighorn in peninsular ranges of California are listed as Endangered in the U.S. The IUCN lists Weems desert bighorn as Critically Endangered, Peninsular desert bighorn as Endangered, and the remaining subspecies as Lower Risk. New Mexico successfully delisted the desert bighorn from the state's Threatened and Endangered species list in 2011 (NMDGF 2013).

### *F.3.1.2.3 Reproduction*

*Breeding Systems:* During mating season mature males compete for access to females through vigorous head butting behavior. During the rest of the year they live amiably in small bachelor herds apart from female herds.

*Gestation:* Gestation lasts about 175 days (Geist 1971, Shackleton et al. 1999).

*Litter size:* Litter size is 1, twins occurring rarely (Geist 1971, Turner and Hansen 1980).

*Breeding Season:* The timing of the mating season varies throughout their range. Bighorns in southwestern Arizona deserts have an extended season encompassing several months (Krausman et al. 1999), but the season is relatively later and shorter elsewhere, generally November in the northern part

of the range (Shackleton et al. 1999), November-December in some southern California mountains (DeForge 1980).

*Birthing Season:* Lambing generally peaks in March in desert populations (Rubin et al. 2000) with most lambs born between January and March (NMDFD 2003), and May (occasionally April or June) in the remainder of range (Krausman et al. 1999, Shackleton et al. 1999).

*Life history:* Young are weaned in 4-6 months. Females first breed in their second year in southern populations; they may breed in their third year in north populations and occasionally breed in their first year in some areas (Krausman et al. 1999, Shackleton et al. 1999). The ability for females to reproduce begins to decline slightly after eight years of age (Caughley 1977). One of the most important habitat requirements for Bighorn is open, mountainous habitat in close proximity to escape terrain. Escape terrain is particularly important for Ewes during parturition and while raising young (NMDFG 2003). Ewes spend more time than bachelor rams in steep rocky escape terrain to maximize protection for themselves and lambs even though these areas generally have less abundant forage (NMDFG 2003)

### F-3.1.3 Habitat and Ecology

#### *F.3.1.3.1 Habitat*

*Terrain:* Bighorn sheep habitat is typically made up of rough, rocky topography in mountains, foothills or river canyons (Shackleton et al. 1999, Krausman et al. 1999). Suitable escape terrain - steep slopes (i.e. cliffs, talus slopes) with a minimum of 60% slope that provide for predator avoidance is a crucial habitat requirement of desert bighorn sheep (NMDFG 2003). High quality and diversity of available forage plants are important to desert bighorn sheep, but it is noted that the amount of habitat available to sheep is limited by the amount of escape terrain close to open landscapes (McCarty and Bailey 1994). Terrain that allows for seasonal elevational movements between areas of foraging and breeding are also important (NatureServe 2013; USFWS 2008). Escape terrain is described as rock outcroppings and slopes of 27 – 85 % (Shannon et al. 2008). Rocky Mountain Bighorn in the north have been documented to spend as much as 86% of their time within 100 meters of escape terrain in winter (Oldemayer et al. 1971, Erickson 1972), and usually stay within 800 meters of escape terrain throughout the year (Pallister 1974). Desert bighorn use the micro-climates of cliff overhangs, granite outcroppings and slope aspect to rest from direct sun in hot, dry environments (NatureServe 2013; San Diego Zoo Global Library 2009); solar heat on south aspects also reduces cold stress on bighorn sheep in northern habitats (Shackleton et al. 1999).

*Elevation:* Elevation varies considerably, both geographically and seasonally, from as low as 450 meters to over 3,300 meters (Shackleton et al. 1999).

*Distance from accessible water:* Northern populations of bighorn sheep do not usually require free-standing water, instead getting water from succulent vegetation in the summer and snow or ice in the winter (Van Dyke 1978). However, in the Madrean Archipelago region, bighorn have been shown to use water year-round (Gunn 2000) and in New Mexico, most bighorn are found within 1 mile of water (NMDFG 2003). Particularly during summer drought, bighorn may require accessible surface water to augment what moisture they are able to get from forage (Turner 1979, Turner and Weaver 1980, Seegmiller and Ohmart 1981). In the desert, bighorn can survive 10 or more days without drinking, and may meet some water needs by eating barrel cacti (Warrick and Krausman 1989; NatureServe 2013). Bighorn have been observed using free water in all New Mexico populations (NMDFG 2003). The presence of dependable water sources is strongly correlated with population persistence (Epps et al. 2004) and the creation of artificial water sources has been a component of sheep management and recovery for many years.

*Vegetation:* Bighorn sheep occur in mesic to xeric, alpine to desert grasslands or shrub-steppe (Shackleton et al. 1999, Krausman et al. 1999) and avoid low visibility areas with dense vegetation (Hall 1981, Hansen 1980, Risenhoover and Bailey 1985, Shackleton 1997, Ostermann et al 2001, USFWS 2000, San Diego Zoo). In Chihuahuan Desert habitat shrubs dominate their diet but their diet shifts with seasonal availability of newly emergent grasses and forbs during the summer-fall rainy season (Sandoval 1997, NMDFG 2003).

#### *F.3.1.3.2 Landscape Connectivity*

In the Madrean Ecoregion, long-term persistence of desert bighorn may depend on movement corridors between populations (Bleich et al. 1990, Bailey 1992). They also rely on intermountain corridors for movement and to allow colonization of new areas or areas from which they were previously extirpated. Rams may also move long distances between mountain ranges in search of ewes during the rut (Geist 1971). In western Arizona, January-June home range of adult females was 19-27 square km (Seegmiller and Ohmart 1981). Male annual home range can be up to 37 square km in Nevada (Leslie and Douglas 1979).

#### *F.3.1.3.3 Food*

Desert bighorn sheep are opportunistic herbivores and ruminants. Turner (1973) documented 43 species of plants in diet. They are primarily grazers of grass and forbs, but their diet can also include significant amounts of shrubs and occasionally cacti, including *Acacia spp.*, *Encelia spp.*, sweetbush (*Bebbia juncea*), jojoba (*Simmodsia chinensis*) and *Krameria spp.* (Miller and Gaud 1989, Krausman et al. 1999, Shackleton et al. 1999). Diet can vary seasonally.

#### *F.3.1.3.4 Phenology*

Diurnal behavior. Feeding activity peaks at early morning and dusk, and grazing during the day is punctuated with periods of rest and rumination. Daily activity can vary by season; in winter, feeding periods are longer (NatureServe 2013).

#### *F.3.1.3.5 Predators*

Natural predators of the bighorn include coyotes (*Canis latrans*), bobcats (*Lynx rufus*), mountain lions (*puma concolor*), golden eagles (*Aquila chrysaetos*), gray wolves (*Canis lupus*). Coyotes have been shown to primarily prey young bighorn within a herd. In one case up to 80% of the year's lambs were lost to coyote predation (Hebert and Harrison 1988, Harper 1984, Hass 1989). Mountain lions, which are adapted to hunt in rugged terrain, are important predators in the Madrean Archipelago Ecoregion with mountain lions being documented as the cause of mortality in 85% of know-cause mortality in New Mexico desert bighorn (Harrison and Hebert 1988, Krausman et al. 1999, NMDFG 2003). Mountain lions have been cited as the primary factor preventing populations from recovering in New Mexico, not limitations of habitat quality or forage availability (NMDFG 2003).

Bighorn are known to alarm and startle easily at sudden animal movements and other disturbances (San Diego Global Library 2009). It is possible that fire suppression resulting in unnaturally dense vegetation cover may increase predation success (Etchberger et al. 1989).

#### *F.3.1.3.6 Competitors*

Carrying capacity for bighorn can be reduced through grazing by other ungulates, particularly domestic stock including cattle, sheep and burros. Mule deer (***Odocoileus hemionus***) are their primarily natural competitors (Shackleton 1985, USFWS 2000, San Diego Zoo Global Library 2009). Feral honey bees may

interfere with sheep attempting to drink at artificial water sources (guzzlers) in desert environments (Boyce et al. 2003).

#### *F.3.1.3.7 Demographics*

Herds are matrilineal and based on female associations; Gregarious, but for most of the year adult males live apart from females and young (Shackleton et al. 1999, Krausman et al. 1999). Larger males less than 10 years old generally dominate younger males during the breeding season; males older than 10 years decline rapidly in condition. Bighorn tend to be faithful to their natal home range. Young learn from older sheep their home range, escape terrain, water sources and lambing habitat. Rams tend to range, moving between female ewe groups, and rams with larger horns and body size dominate both males and females; linear dominance orders among ewes (Shackleton 1985, USFWS 2000, Valdezand Krausman 1999).

#### *F.3.1.3.8 Disease and Parasites*

Respiratory disease is the most serious disease affecting bighorn sheep. Disease was an important factor contributing to extinction of bighorn in much of their range and continues to limit populations today (Buchener 1960). As an example, contact with stray domestic sheep is believed to have resulted in the death (through bacterial pneumonia) of entire reintroduced herd of 65 in Warner Mountains, California and devastated bighorn populations in the mid and late 1900's (California Department of Fish and Game 1990). Respiratory disease can have large impacts on population dynamics of desert bighorn including all-age mortality events, high rates of mortality in lambs and chronic low-level infection in adults leading to sporadic mortality. Once introduced, respiratory pathogens can spread through interconnected populations for years. This is of particular concern for fragmented or isolated populations that may be pushed to extinction by stochastic events unrelated to disease (IDFG 2010).
Chronic sinusitis, pneumonia, and psoroptic scapies are the most common problems in bighorn sheep, and pathogens including contagious ecthyma virus, bluetongue, Pasteurella, contagious echthyma, infection keratoconjunctivitis, mycoplasma ovipneumoniae, and parainfluenza-3 virus have been found in declining populations. Decreased fitness occasionally occurs due to improperly fitted radio collars that cause physical injury to osseous and dermal tissues (Allen 1980, USFWS2000, Valdez and Krausman 1999, Clark at al. 1985).

### F-3.1.4 Change Agent (CA) Characterization

The main change agents identified are forms of altered dynamics: Livestock management, development, and climate change.

Initial large declines were primarily the result of overhunting, habitat loss and competition with, and diseases and parasites introduced by, domestic stock including cattle, sheep and burros (Cowan 1940, Buechner 1960, Sugden 1961, Stelfox 1971, Goodson 1982, Boyce et al. 1990, Valdez and Krausman 1999). Urban encroachment, human disturbance and habitat changes resulting from removal of fire due to consumption of fine fuels by livestock and from fire suppression were likely factors in the disappearance of bighorn in southern Arizona's Catalina Mountain range in the 1990s, although disease or increased predation may have played a part (Etchberger et al. 1989).

#### *F.3.1.4.1 Livestock Management*

In Idaho, introduced bighorns avoided cattle and were more sensitive to cattle presence than were established populations, based on previous studies (Bissonette and Steinkamp 1996). Decline of desert bighorn populations has been attributed to destruction of habitat and competition with domestic livestock and other native and domestic species (Seegmiller and Ohmart 1981).

**Indicators:** Landscape condition, active grazing leases, pasture management.

**Stresses:** increased competition, change in vegetation structure, decreased plant diversity, decreased forage abundance, increased behavioral disruption.

**Responses:** decreased physical fitness, decreased population size, reduced/ altered home range size.

#### *F.3.1.4.2 Disease and Parasites*

Desert bighorn are more sensitive to diseases and parasites than other native big game species. Desert bighorn populations have been heavily impacted by diseases introduced by livestock as people settled in the Madrean Archipelago Ecoregion. (NMDFG 2003). Psoroptic scabies from domestic sheep have been implicated in the decline of desert bighorn populations in the west (NMDFG 2003). Domestic sheep spread pneumonia, usually fatal for desert bighorn, throughout central and southern New Mexico (NMDFG 2003). In the Madrean Archipelago Ecoregion, even penned flocks of domestic sheep are of concern. Domestic goats were implicated in an outbreak of pneumonia in the Hell's Canyon Rocky Mountain bighorn (Cassirer et al. 1996). In the Arizona portion of the Madrean Archipelago Ecoregion, there was an incident of bighorn in the Silverbell Mountains dying from ecthyma and keratoconjunctivitis following contact with domestic goats. Cattle have not been conclusively shown to transmit disease to bighorn, but cattle do carry disease that are a potential source of bighorn disease (Bailey 1994), including bluetongue virus that is transmitted between ruminant species by a gnat (Luedke et al. 1967). Outbreaks of bluetongue have been documented in bighorn populations in New Mexico. Bunch et al. (1999) provides a general account of diseases and parasites affecting bighorn sheep which include chronic wasting disease, bacterial chronic sinusitis, contagious ecthyma, and parasitic elaeophorosis.

**Indicators:** Domestic stock with disease or parasite infestation, decreased recruitment, increased predation.

**Stresses:** Decreased fitness.

**Responses:** Decreased population size, reduced recruitment, decreased genetic health, increased mortality rate.

#### *F.3.1.4.3 Development*

Loss and alteration of habitat, especially key winter forage sites, is a key threat (Valdez and Krausman 1999, Shackleton et al. 1999, Krausman et al. 1999). Habitat degradation can occur through overgrazing by domestic stock, competition with exotic ungulates (e.g., Aoudad or Barbary, *Ammotragus lervia*), excessive off-road vehicle use, roads and trails, urban development, mining, spread of rangeland weeds, and the loss of water sources (Simpson 1980, Valdez and Krausman 1999, Krausman et al. 1999). Fire suppression and resulting vegetation succession leading to dense cover with the encroachment of tall dense shrubland and forest have been a major cause of habitat loss in Colorado and British Columbia (Davidson 1991, Cannings et al. 1999, Wakelyn 1987). Dense cover resulting from fire suppression is also a factor in herd declines in the Santa Catalina Mountains of Arizona (Etchberger et al. 1989). Fragmentation of habitat reduces or eliminates genetic interchange among populations (Ramey 2000) and reduces the probability of recolonization following local extirpation; both these effects are especially concerning in small populations (fewer than 100 individuals) vulnerable to extirpation (Berger 1990).

**Indicators:** Landscape condition, landscape permeability, habitat loss, groundwater level/ hydrology condition, distance from water, water quality, air quality, light pollution levels, noise index, human-wildlife conflict, distance from roads, wildlife-vehicle collisions, and direct take (hunting).

**Stresses:** Reduced habitat, decreased water availability, increased habitat fragmentation, loss of view shed/ increase in tall dense vegetation, increased behavioral disruption.

**Responses:** decreased population size, reduced recruitment, decreased home range, reduced migratory distance, decreased gene flow/ decreased genetic health, increased mortality rate.

#### *F.3.1.4.4 Climate Change*

Bighorn rely greatly on specific habitat requirements to survive and thrive, which are affected by precipitation, temperature, seasonality, and other climatic conditions. Additional studies of these influences in consideration of a changing climate - including increasing temperature extremes, increased fire intensity and frequency, and drought - are needed, but some research outside the Madrean Archipelago ecoregion may provide insight to the responses that may be expected. Although bighorn are able to obtain much of the water that they need from their diet, the absence or reduction of succulent forage requires bighorn to seek out additional surface water. Decreased water resources and increased presence of predators at available water sites can expose bighorn to greater risk of mortality; in addition, warmer temperatures and lower precipitation may lead to increased water draw-downs by human populations (Heinz Center 2011, SWCCN 2008). The National Drought Mitigation Center (2013) reports that the Madrean Archipelago ecoregion has been experiencing Severe to Extreme drought in the last year, and forecasts this condition will persist or intensify.

**Indicators:** Increased drought, decreased water table/ groundwater level, decreased surface water, increased temperature extremes and duration, increased wildfire.

**Stresses:** increased change in vegetation structure, change in biotic community, decreased forage availability, decreased water availability, increased distance to water, increased predation, increased fire frequency and intensity, hyperthermia.

**Responses:** decreased population size, reduced recruitment, increased daily foraging movements, increased migratory distance, increased mortality rate.

### F-3.1.5 Ecological Status: Key Ecological Attributes and Indicators

This section of the conceptual model addresses Key Ecological Attributes and their potential indicators. The ecological status is a way of describing current status via criteria, functionality, or levels of attributes and asks if they are within the normal range of variation.

#### *F.3.1.5.1 Key Ecological Attributes*

Table F-7 identifies the key ecological attributes for this CE within the Madrean Archipelago ecoregion. A **key ecological attribute** of a focal ecological resource is a characteristic of the resource's biology, ecology, or physical environment that is critical to the resource's persistence in the face of both natural and human-caused disturbance. Alteration of such a characteristic beyond some critical range of variation will lead to the degradation or loss of the resource within decades or less. The KEAs table lists the identified key ecological attributes, with a brief definition, a rationale for why it is important for the CE, and a listing of stressors or change agents that might be affecting the key attribute.

**Table F-7. Key ecological attributes (KEA) of desert bighorn sheep (Ovis canadensis) in the Madrean Archipelago ecoregion.** Indicators for these KEAs can be used to determine the ecological status for this CE; see **Table F-2** for a list of the indicators assessed in this REA.

| ***KEA Class:*** **KEA Name** | **Definition** | **Rationale** | **Stressors** |
|---|---|---|---|
| ***Landscape Context:*** **Functional Landscape Connectivity & Landscape Permeability** | The degree to which the landscape facilitates or impedes bighorn movement within and among habitat patches including connectivity of foraging habitat to escape terrain; and the relative potential for bighorn movement between populations at a regional scale. This may be indicated by the Landscape permeability index and Landscape Condition Model Index and/or assessing potential loss of key habitat association (escape terrain and foraging terrain). | At a local scale, permeability is critical for bighorn sheep to avoid predators and access food and water; at a larger landscape scale, it is critical for breeding between herds, dispersal, and migration between seasonal ranges. In the Madrean Ecoregion, long-term persistence of desert bighorn may depend on movement corridors between populations (Bleich et al. 1990, Bailey 1992). They also rely on intermountain corridors for movement and to allow colonization of new areas or areas from which they were previously extirpated. Rams may also move long distances between mountain ranges in search of ewes during the rut (Geist 1971). In western Arizona, January-June home range of adult females was 19-27 square km (Seegmiller and Ohmart 1981). Male annual home range up to 37 square km in Nevada (Leslie and Douglas 1979). | Stressors include fragmentation and habitat loss, caused by human development in the form of fencing, mining, urban development, and distance to roads, railroads and highways, among others. Fragmentation of habitat reduces or eliminates genetic interchange among populations (Ramey 2000) and reduces the probability of recolonization following local extirpation (Berger 1990). Low or absent landscape permeability can indicate locations of corridor bottlenecks and genetic isolation between populations that can lead to decreased genetic health, inbreeding and local extinction. Because bighorn have a strong flight and stress response to animal movement or disturbance (San Diego Global Library 2009), hiking trails and other areas frequented by humans or domestic dogs may also affect habitat permeability for bighorn sheep. |
| ***Size/Extent:*** **Population Condition** | Population size and demographics of bighorn herds, including sex and age ratios. This may also include evidence of population health (presence of disease or signs of chronic physical stress) which over time affect population size and demographics. | Population size and demographic structure directly indicates the viability and health of the population. Demographics can affect breeding systems, resulting is changes to genetic diversity; evidence of disease or signs of chronic stress may be a precursor to a decrease in population size. Initial large declines in bighorn sheep populations were primarily the result of overhunting, habitat loss and competition with, and diseases and parasites introduced by, domestic stock including cattle, sheep and burros (Cowan 1940, Buechner 1960, Sugden 1961, Stelfox 1971, Goodson 1982, Boyce et al. 1990, Valdez and Krausman 1999). | Stressors include causes of direct mortality such as predation, hunting, poaching, and disease; decreased fitness as the result of poor forage conditions, livestock use of forage, low water availability or flight/stress response to disturbance; and habitat fragmentation resulting in small habitat patches unable to support large herd populations. Urban encroachment, human disturbance and habitat changes resulting from fire suppression were likely factors to the disappearance of bighorn in southern Arizona's Catalina Mountain range in the 1990's, although disease or increased predation may have played a role (Etchberger et al. 1989). |

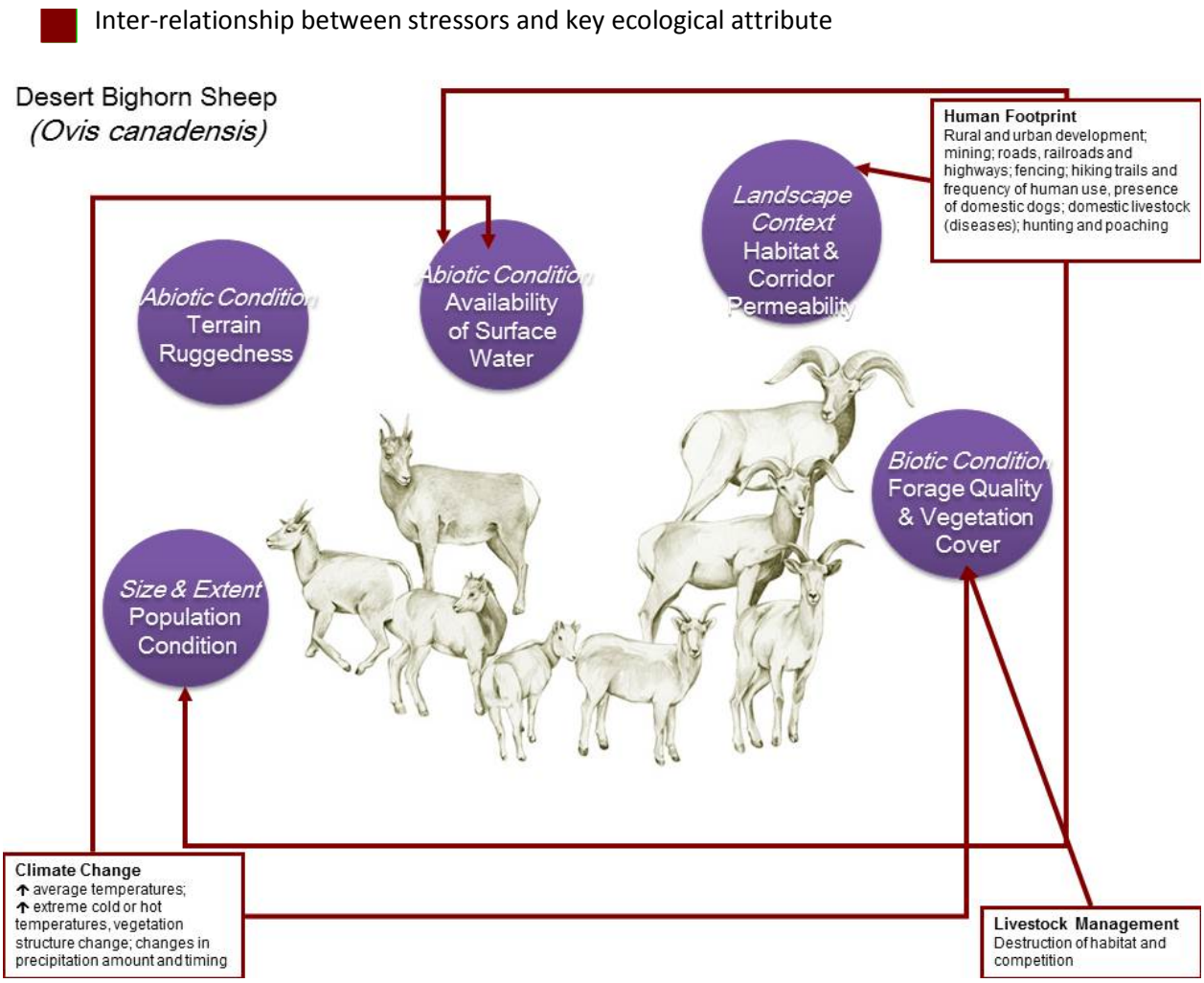
| *KEA Class:* KEA Name | Definition | Rationale | Stressors |
|---|---|---|---|
| ***Biotic Condition:* Forage Quality & Vegetation Cover** | Relative composition, abundance and species richness of native forage plants, average vegetation density and height within bighorn habitat areas. | Bighorn avoid low visibility areas with dense vegetation (Hall 1981, Hansen 1980, Risenhoover and Bailey 1985, Shackleton 1997, Ostermann et al 2001, USFWS 2000, San Diego Zoo). They are primarily grazers of grass and forbs, but their diet can also include significant amounts of shrubs and occasionally cacti, including *Acacia spp.*, *Encelia spp.*, sweetbush (*Bebbia juncea*), jojoba (*Simmodsia chinensis*) and *Krameria spp.* (Miller and Gaud 1989, Krausman et al. 1999, Shackleton et al. 1999). It is possible that unnaturally dense vegetation cover may increase predation success (Etchberger et al. 1989). In the Madrean Archipelago region, particularly during summer drought, or when surface water is less available, bighorn need to augment what moisture they are able to get from forage (Turner 1979, Turner and Weaver 1980, Seegmiller and Ohmart 1981). With high quality forage, bighorn are able to avoid frequent risk of predation at surface water sites. | Seasonal and monsoon precipitation has a large effect on forage quality; drought, invasive plants, livestock impacts to forage condition and fire suppression are also significant stressors on forage condition and vegetation density. |
| ***Abiotic Condition:* Availability of Surface Water** | Annual and seasonal precipitation and the resulting density, distribution and distance to surface water accessible by bighorn. | Madrean Archipelago region, bighorn have been shown to use water year-round (Gunn 2000) and in New Mexico, most bighorn are found within 1 mile of water (NMDFG 2003). Particularly during summer drought, bighorn may require accessible surface water to augment what moisture they are able to get from forage (Turner 1979, Turner and Weaver 1980, Seegmiller and Ohmart 1981). In the desert, bighorn can survive 10 or more days without drinking, and may meet some water needs by eating barrel cacti (Warrick and Krausman 1989; NatureServe 2013). Precipitation affects surface water availability and growth of high-nutrient forage. | Surface water sources include active seeps, springs, water tanks, wildlife waters, riparian and perennial streams, lakes and other sources of water in relation to bighorn habitat areas. Stressors affecting these open water sources include watershed development that alters runoff, infiltration (recharge), and evapotranspiration rates; surface water diversions, transfers, and use; groundwater withdrawals from basin-fill and alluvial aquifers; dams and dam operations; and riparian corridor development. Climate change also has the potential to cause increased drought and changes in human water use. Decreased water resources and increased presence of predators at available water sites can expose bighorn to greater risk of mortality. |

| *KEA Class:* KEA Name | Definition | Rationale | Stressors |
|---|---|---|---|
| ***Abiotic Condition:*** **Terrain Ruggedness** | The topographic roughness (ruggedness) of the landscape or habitat area. A roughness index may be the standard deviation of slope, standard deviation of elevation, slope convexity, or degree of topographic contour. | Bighorn sheep habitat is typically made up of rough, rocky topography in mountains, foothills or river canyons (Shackleton et al. 1999, Krausman et al. 1999). Suitable escape terrain - steep slopes (i.e. cliffs, talus slopes) with a minimum of 60% slope that provide for predator avoidance is a crucial habitat requirement of desert bighorn sheep (NMDFG 2003). High quality and diversity of available forage plants are important to desert bighorn sheep, but it is noted that the amount of habitat available to sheep is limited by the amount of escape terrain close to open landscapes (McCarty and Bailey 1994). Terrain that allows for seasonal elevational movements between areas of foraging and breeding are also important (NatureServe 2013; USFWS 2008). | The availability of rugged terrain to bighorn is directly affected by human activity, including habitat loss through human encroachment and development. Climate change that alters high quality forage or water availability occurring in areas with ideal terrain may be a concern. |

### F-3.1.6 Conceptual Model Diagram

A conceptual model diagram for each CE provides a visual summary representation of the Key Ecological Attributes and species' life cycle (Figure F-16). These diagrams are intended to show how various stressors interact with categories of Key Ecological Attributes to highlight important drivers for species ecological integrity. The arrows indicate relationships between stressors and but do not indicate the nature of the influence (i.e. positive or negative).

**Figure F-16. Conceptual model diagram for the desert bighorn sheep, showing key ecological attributes** (by class) for this species, and indicating relationships between stressors and KEAs. Original artwork by Jane Kim.




## F-3.2 Considerations and Limitations

Desert bighorn have a strong flight and stress response to animal movement or disturbance (San Diego Global Library 2009); therefore, hiking trails and other areas frequented by humans or domestic dogs may affect bighorn sheep habitat and movement. This may mean that smaller or lower intensity development features on the landscape, such as trails, may be having a large effect on sheep if they are

frequented by humans and dogs, while larger features (such as a dirt road) may have less effect if rarely used. Although the flight response is qualitatively understood, the quantitative effects on bighorn are not sufficiently understood and therefore site intensity scores for various development features were not adjusted accordingly. Although bighorn are sensitive to certain kinds of human disturbance, within the MAR they can be found inhabiting reclaimed mine tailings piles where there is still human activity. This may be due to their preference for suitable escape terrain and line-of-sight to avoid predators.

Although the Santa Catalina Mountains in the MAR are not included in the current distribution map for desert bighorn (Figure F-15), these mountains are the site of a reintroduction effort that was initiated in the fall of 2013. This may be important to inform project-level management decisions.

## F-3.3 Ecological Status Assessment Results and Interpretation

This section of the CE summary presents and discusses the results of the ecological status assessment for desert bighorn sheep. It addresses each indicator separately, and then addresses the overall assessment, which integrates the results of all individual indicators assessed for the CE. The results are presented using a common framework, in which the status of an indicator – or the combination of all indicators – is scored on a scale from 0.0 to 1.0, where 0.0 indicates a condition of complete replacement of reference ecological conditions due to the impacts of stressors, and 1.0 indicates a condition of no alteration of reference ecological conditions. The same yellow-to-blue color ramp is used for all results, with yellow representing low scores, green moderate scores, and dark blues high scores.

### F-3.3.1 Current Ecological Status: Development, Fire Regime, Invasives

The results of the assessments for the three individual indicators for the KEAs for desert bighorn sheep are shown in the three maps in Figure F-17 below.

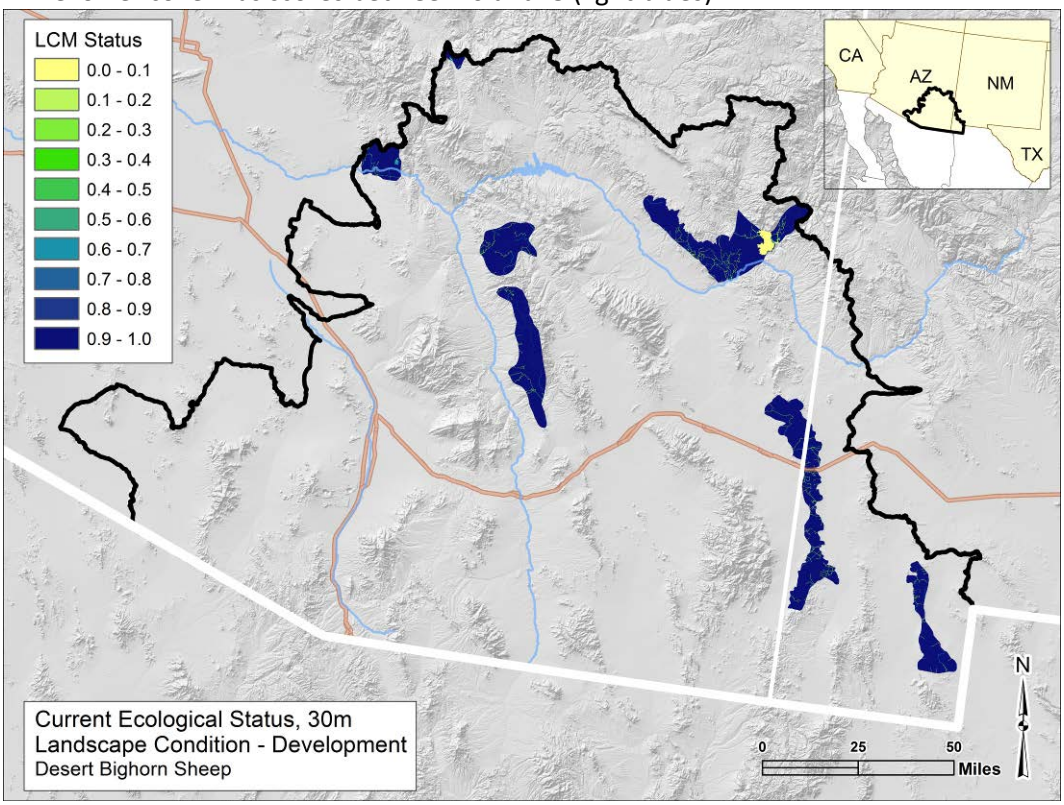
A concern with development is the direct loss of habitat for bighorn. With a localized exception around Morenci, most of the bighorn sheep's habitat has experienced very limited impacts from development, as indicated by the high scores in the first map of Figure F-17. Although there are likely roads throughout its habitat, the terrain it typically occupies and current understanding of road effects on desert bighorn suggest roads are not a major issue for this species.

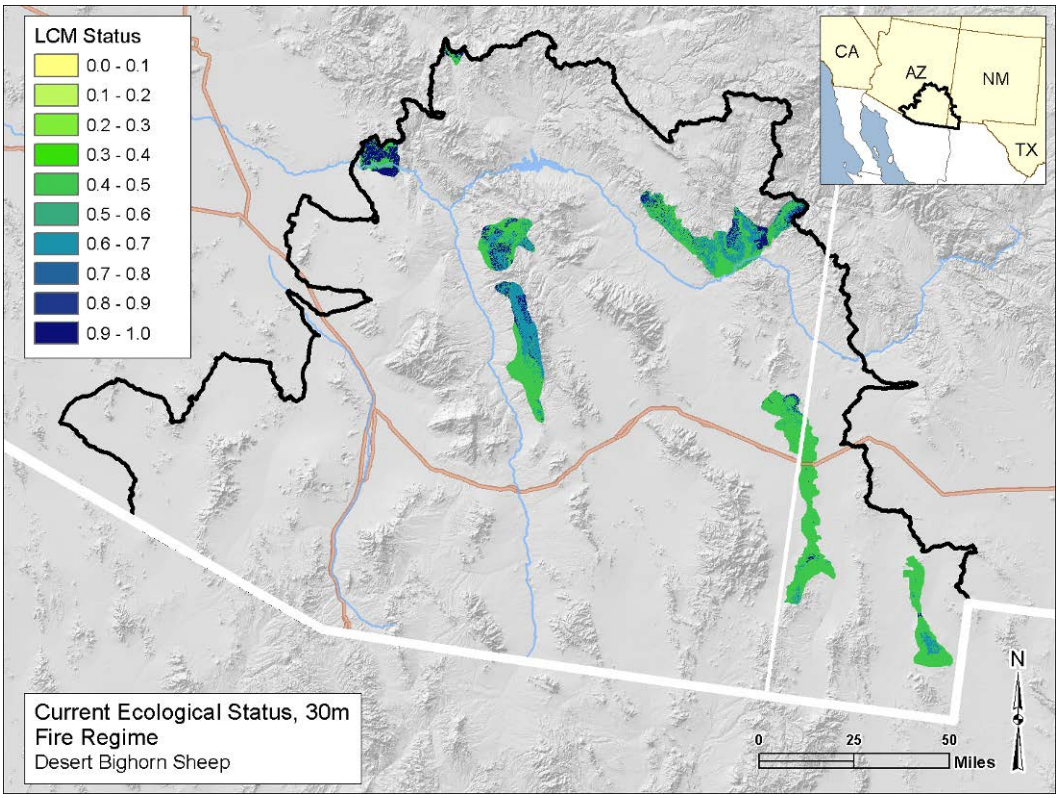
As described in the conceptual model above, other studies indicate fire suppression has altered density and composition of vegetation in desert bighorn habitat in other parts of this species' range, resulting in loss of suitable habitat or herd declines (Cannings et al. 1999, Davidson 1991, Etchberger et al. 1989, Wakelyn 1987). Based on the indicators that were assessed in this REA, the biggest issue affecting the ecological status of desert bighorn habitat is the alteration in the fire regimes in the ecosystems utilized by this species. The second map in Figure F-17 shows that most of the ecological systems in its habitat are in the severe ecological departure category (scores of 0.4), indicating a highly departed fire regime. Given the findings from other parts of desert bighorn range, the highly altered fire regimes may contribute to habitat loss and herd decline.

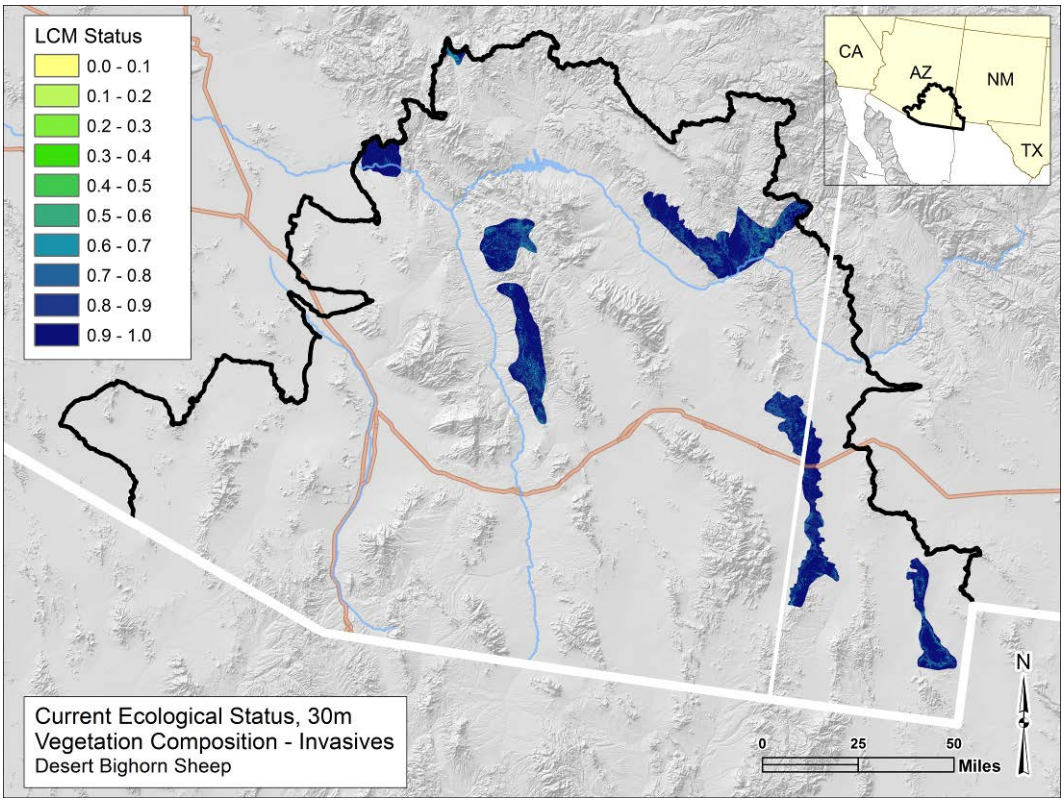
The invasives indicator map (third map in Figure F-17) shows some areas with high cover (>25%) in bighorn habitat; these areas have invasives scores around 0.65 and appear in the turquoise blue pixels interspersed among the dark blue pixels (which indicate no invasives present according to ILAP models of current invasives cover). The areas with "high cover" reflect either non-native grasses or forbs, or mesquite encroachment. As described in the conceptual model above, desert bighorn avoid low visibility areas with dense vegetation (Hall 1981, Hansen 1980, Ostermann et al 2001, Risenhoover and Bailey

1985, Shackleton 1997, USFWS 2000); high densities of mesquite could create unsuitable forage habitat and poor visibility. Based on the ILAP source data, generally there are not large and highly concentrated areas of either mesquite or non-native forbs and grasses in desert bighorn habitat, but instead more dispersed and less dense patterns of these two categories of invasive cover. To the degree that non-native invasives or native woody increasers are affecting vegetation density and fire regimes in bighorn forage habitat, they may also be negatively impacting habitat for this species in the areas where they are modeled to be present at higher percent cover.

**Figure F-17. Scores for three indicators for desert bighorn sheep: development indicator (1st map), fire regime departure indicator (2nd map), and invasive species indicator (3rd map) for each 30m pixel.** Yellow (equivalent to 0) indicates high impacts from the CA, dark blue (equivalent to 1) indicates little to no impact from the CA. At the ecoregion scale, many development features are not readily visible (e.g., secondary roads or highways, railroads, small agricultural fields). Only 3 classes of fire regime condition are scored: no to little departure (dark blue), moderate departure and severe departure (lighter blues). For invasives, higher cover of mesquite or invasive exotics have scores between .4 and .6 (light greens), while lower cover has scores between .6 and .8 (light blues).




LCM Status
0.0 - 0.1
0.1 - 0.2
0.2 - 0.3
0.3 - 0.4
0.4 - 0.5
0.5 - 0.6
0.6 - 0.7
0.7 - 0.8
0.8 - 0.9
0.9 - 1.0
CA
AZ
NM
TX
N
0
25
50
Miles
Current Ecological Status, 30m
Fire Regime
Desert Bighorn Sheep


LCM Status
0.0 - 0.1
0.1 - 0.2
0.2 - 0.3
0.3 - 0.4
0.4 - 0.5
0.5 - 0.6
0.6 - 0.7
0.7 - 0.8
0.8 - 0.9
0.9 - 1.0
CA
AZ
NM
TX
N
0
25
50
Miles
Current Ecological Status, 30m
Vegetation Composition - Invasives
Desert Bighorn Sheep

### F-3.3.2 Current Ecological Status: Full Scenario

The results of the three individual status indicators were combined to get an overall ecological status score, per pixel of the CE's distribution, as shown in the first map of Figure F-18 below. The overall, per-pixel status scores for the CE were then averaged across each of the 4 km grid cells, as shown in the second map of Figure F-18 below.

When the three sets of indicator scores are aggregated to obtain the overall ecological status scores (as shown in the first map of Figure F-18), much of the habitat for this species is in somewhat lower condition, with scores around 0.3, 0.4, and 0.5 (Figure F-19). In comparing the fire regime indicator map (second map in Figure F-17) to the overall status scores, it is clear that fire regime is the main factor driving down the scores, although certainly the interspersion of pixels containing high cover of invasives and the development around Morenci are also contributing. These patterns hold when the overall status scores are averaged across each 4 km reporting unit, as shown in the second map of Figure F-18 above. Based on the indicators assessed in this REA, changes in bighorn forage habitat, including decreased suitability of escape terrain, caused by the altered fire regime is having the most widespread effects on habitat quality for this species.

**Figure F-18. Overall ecological status scores for desert bighorn sheep for all three indicators combined (development, fire regime and invasives) for each 30m pixel (top) and 4km grid cells (bottom).** Yellow scores (equivalent to 0) indicate high impacts from the CAs, dark blue (equivalent to 1) indicate little to no impact from the CAs. In the second map, the score for each 4km cell is an average of the overall ecological status scores of the 30m pixels within the 4km cell that were scored for the CE.

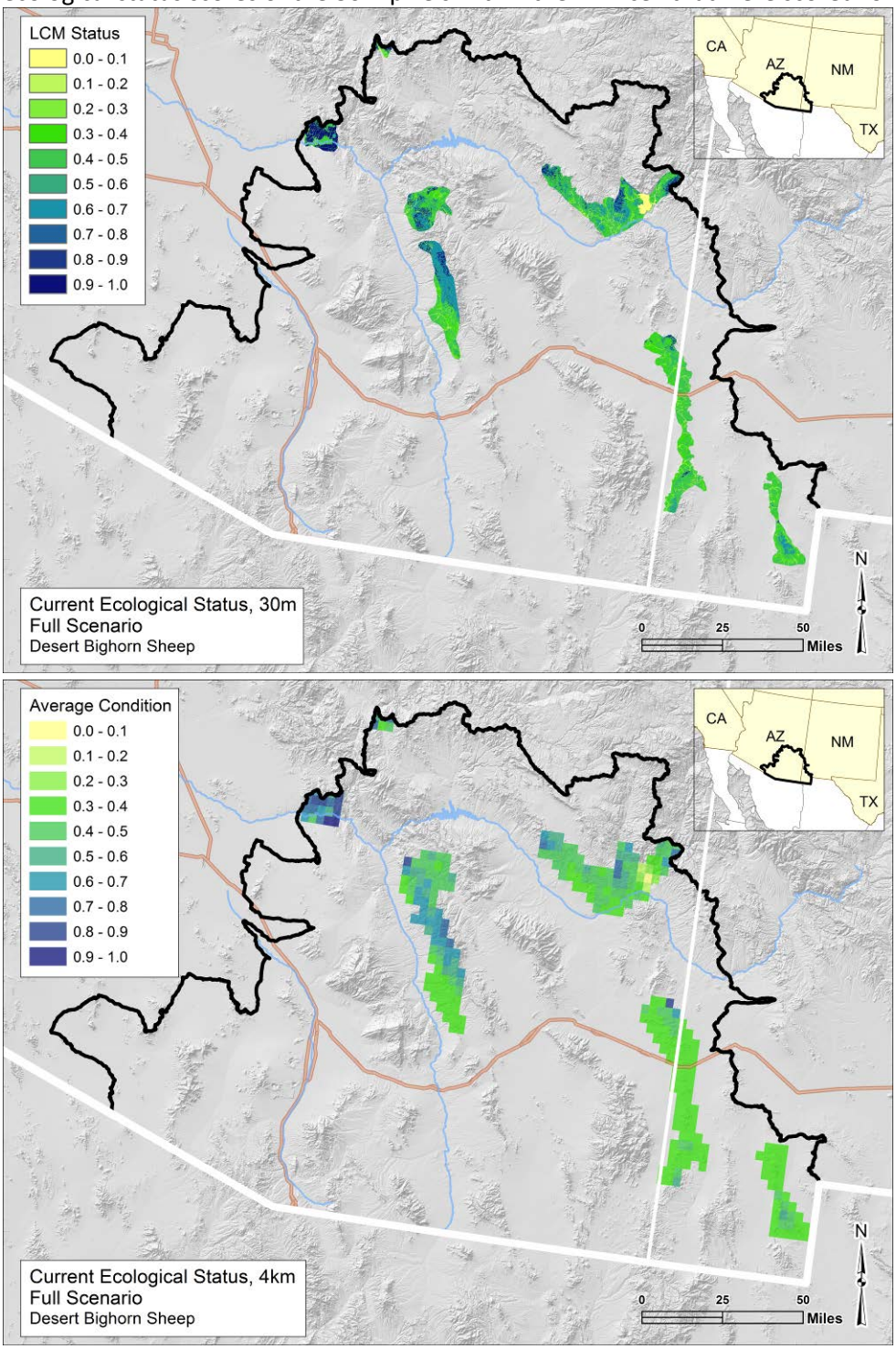
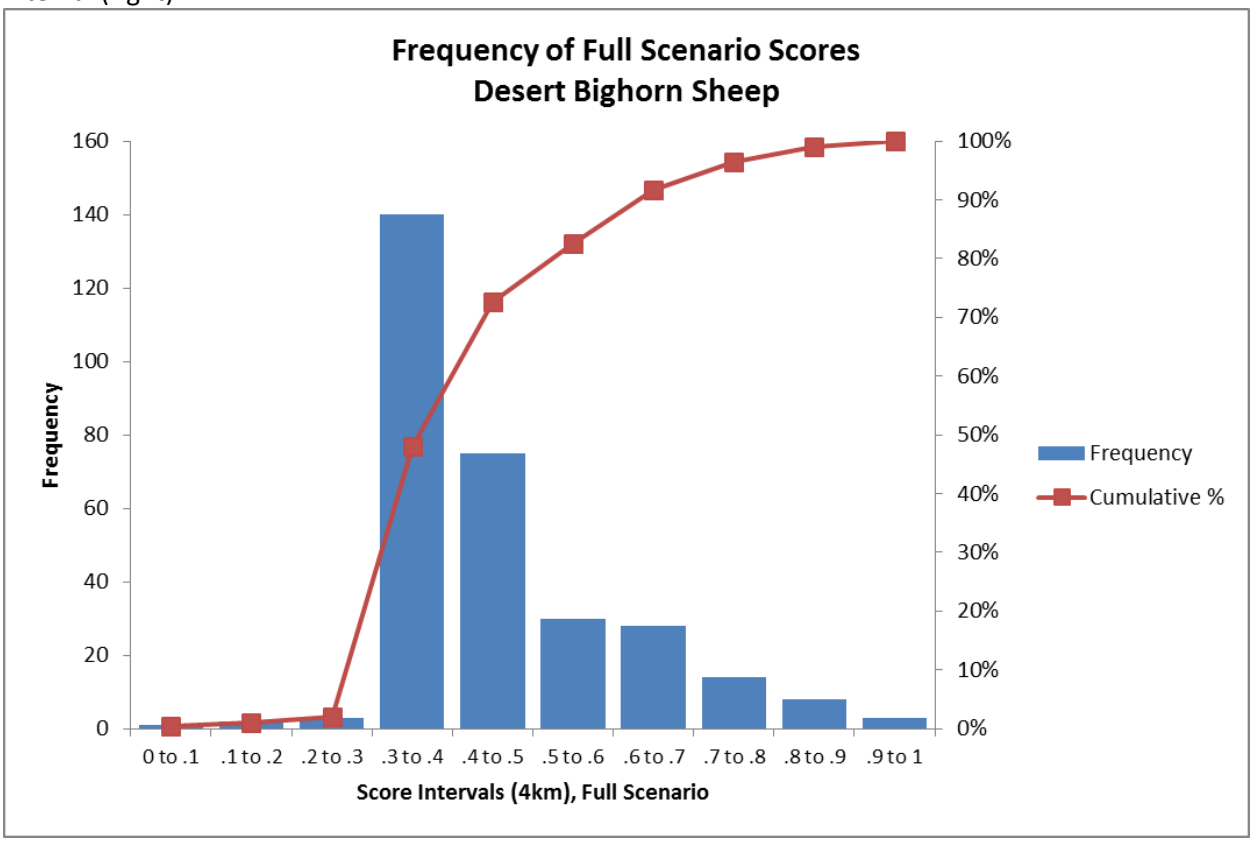


**Figure F-19. Frequency distribution of the 4km ecological status scores for desert bighorn sheep, with cumulative percent.** The x-axis represents the 0.1 increment scoring intervals, while the y-axis shows the number of grid cells in each interval (left) and the cumulative percentage of the grid cells for each interval (right).




## F-3.4 References for the CE


Allen, R. W. 1980. Natural mortality and debility. In: G. Monson and L. Sumner, eds. The Desert Bighorn: Its Life History, Ecology, and Management, pp. 172 - 185. Tucson: The University of Arizona Press. 370 pp.

Aoude, A. 2008. Status of Utah's bighorn sheep and mountain goats in 2008. Biennial Symposium of the Northern Wild Sheep and Goat Council 16:1-2.

Arizona Game and Fish Department (AZGFD). 2013. HabiMap™ Arizona. Mammal distributions, last revision 2011. Accessed 24 March 2013.

Bailey, J. A. 1992. Managing bighorn habitat from a landscape perspective. Biennial Symposium of the North American Wild Sheep and Goat Council 8:49 -57

Berger, J. 1990. Persistence of different-sized populations: an empirical assessment of rapid extinctions in bighorn sheep. Conservation Biology 4:91-98.

Bissonette, J. A., and M. J. Steinkamp. 1996. Bighorn sheep response to ephemeral habitat fragmentation by cattle. Great Basin Naturalist 56:319-325.

Bleich, V. C., J. D. Wehausen, and S. A. Holl. 1990. Desert-dwelling mountain sheep: conservation implications of a naturally fragmented distribution. Conservation Biology 4:383-390.

Boody, G., and B. Devore. 2006. Redesigning agriculture. BioScience 56(10):839-845. Online: http://dx.doi.org/10.1641/0006-3568(2006)56[839:RA]2.0.CO;2.

Boyce, M. B., R. K. Clark and D. J. Jessop. 1990. Recent advances in the diagnosis and treatment of psoroptic scabies in bighorn sheep. Biennial Symposium of the Northern Wild Sheep and Goat Council 7:125-128.

Boyce, W. M., C. S. O'Brien, and E. S. Rubin. 2003. Response of bighorn sheep (*Ovis canadensis*) to feral honey bees (*Apis mellifera*) at water. Southwestern Naturalist 48:81-84.

Boyce, W.M., R.R. Ramey, T.C. Rodwell, E.S. Rubin, and R.S.Singer. 1999. Population subdivision among desert bighorn sheep (*Ovis canadensis*) ewes revealed by mitochondrial DNA analysis. Molecular Ecology, 8(1):33-106.

Bristow, K. and M. Crabb. August 2008. Evaluation of distribution and trans-highway movement of desert bighorn sheep: Arizona Highway 68. Final Report to Arizona Department of Transportation (ADOT) FHWA-AZ-08-588.

Brussard, P. F., D. A. Charlet, D. S. Dobkin, and numerous contributing authors. 2000. Status and trends of the nation's biological resources: Great Basin-Mojave Desert Region. U.S. Department of the Interior, U.S. Geological Survey, Biological Resources Division.

Buechner, H. K. 1960. The bighorn sheep in the United States-its past, present and future. Wildlife Monograph No. 4. 174pp.

Bunch, T. D., W. M. Boyce, C. P. Hibler, W. R. Lance, T. R. Spraker, and E. S. Williams. 1999. Pages 209-237 in R. Valdez and P. R. Krausman (editors). Mountain sheep of North America. University of Arizona Press, Tucson, Arizona. 353 pp.

California Department of Fish and Game (CDFG). 1990. 1989 annual report on the status of California's state listed threatened and endangered plants and animals. 188 pp.

Cannings, S. G., L. R. Ramsay, D. F. Fraser, and M. A. Fraker. 1999. Rare amphibians, reptiles, and mammals of British Columbia. Wildlife Branch and Resources Inventory Branch, B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. 198 pp.

Cassirer, E. F., L. E. Oldenburg, V. L. Coggins, P. Fowler, K. Rudolph, D. L. Hunter, W. J. Foreyt. 1996. Overview and preliminary analysis of a bighorn sheep dieoff, Hell's Canyon 1995-96. Biennial Symposium of the North American Wild Sheep and Goat Council 10:78-86.

Caughley, G. 1977. Analysis of vertebrate populations. John Wiley and Sons, New York, NY.

Chambers, J. C., B. A. Roundy, R. R. Blank, S. E. Meyer, and A. Whittaker. 2007. What makes Great Basin sagebrush ecosystems invasible by *Bromus tectorum*? Ecological Monographs 77:117-145.

Clark, R.K., D.A. Jessup, M.D. Kock, and R.A. Weaver. 1985. Survey of desert bighorn sheep in California for exposure to selected infectious diseases. Journal of the American Veterinary Medical Association, 187:1175-1179.

Comer, P. J., and J. Hak. 2009. NatureServe landscape condition model. Internal documentation for NatureServe Vista decision support software engineering, prepared by NatureServe, Boulder, CO.

Condon, L., A. P. J. Weisberg, and J. C. Chambers. 2011. Abiotic and biotic influences on *Bromus tectorum* invasion and *Artemisia tridentata* recovery after fire. International Journal of Wildland Fire 20:597-604.

Convention on International Trade in Endangered Species (CITES). March 2010. Decisions of the Conference of the Parties to CITES in effect after its 15th meeting. Doha, Qatar. Online: http://www.cites.org/eng/dec/valid15/E15-Dec.pdf.

Cowan, I. M. 1940. Distribution and variation in the native sheep of North America. American Midland Naturalist 24:505-580.

Davidson, P. W. 1991. East Kootenay bighorn sheep enhancement project: completion report. British Columbia Ministry of Environment. Unpublished Wildlife Branch Report. Cranbrook, BC. 183pp.

Debinski, D. M., and R. D. Holt. 2000. A survey and overview of habitat fragmentation experiments. Conservation Biology 14(2):342-355.

DeForge, J. R. 1980. Population biology of desert bighorn sheep in the San Gabriel Mountains of California. Desert Bighorn Council Transactions 24:29-32.

Demarchi, R. A., C. L. Hartwig, and D. A. Demarchi. 1999a. Status of the Rocky Mountain Bighorn Sheep in British Columbia. Unpublished report submitted to Wildlife Branch, British Columbia Ministry of Environment, Lands and Parks, Victoria, BC. 134pp.

Douglas, C. L., and D. M. Leslie, Jr. 1999. Management of bighorn sheep. Pp. 238-262 In: R. Valdez and P. R. Krausman (editors). Mountain sheep of North America. University of Arizona Press, Tucson, Arizona. 353 pp.

Dunbar, M. R. 1992. Theoretical concepts of disease versus nutrition as primary factors in population regulation of wild sheep. Proceedings of the Biennial Symposium of the Northern Wild Sheep and Goat Council 8:174-192.

Erickson, G. L. 1972. The ecology of Rocky Mountain bighorn sheep in the Sun River area of Montana with special reference to summer food habits and range movements. Montana Fish and Game Department, Federal Aid and Wildlife Restoration Project W-120-R-2 and R-3.

Etchberger, R. C., P. R. Krausman, and R. Mazaika. 1989. Mountain sheep habitat characteristics in the Pusch Ridge Wilderness, Arizona. J. Wildl. Manage. 53:902-907.

Festa-Bianchet, M. 1988. Seasonal range selection in bighorn sheep conflicts between forage quality, forage quantity, and predator avoidance. Oecologia 75:580-586.

Geist, V. 1971. Mountain sheep: a study of behavior and evolution. Univ. Chicago Press. Chicago, IL. 384 pp.

Goodson, N. J. 1982. Effects of domestic sheep grazing on bighorn sheep populations: review. Proceedings of the Biennial Symposium of the Northern Wild Sheep and Goat Council 3:287-313.

Gunn, J. 2000. Justification for the continued use of wildlife water developments for the management of bighorn sheep populations in the southwest United States. Report to the Arizona Desert Bighorn Sheep Society. 29pp

Hall, E. R. 1946. Mammals of Nevada. The University of California Press, Berkeley, California.

Hall, E. R. 1981. The Mammals of North America, Vols. I & II. John Wiley & Sons, New York, New York. 1181 pp.

Heinz Center for Science Economics and the Environment. 2011. Managing and monitoring Arizona's wildlife in an era of climate change: Strategies and Tools for Success. Report and Workshop Summary, prepared for the U.S. Department of Interior Bureau of Land Management and the Arizona Game and Fish Department. Available:

http://www.heinzctrinfo.net/wildlife/wordpress/wp-content/uploads/2011/07/Low-res-Heinz-Arizona-Report_Final_7.11.11.pdf

Hoffmeister, D. F. 1986. Mammals of Arizona. University of Arizona Press and Arizona Game and Fish Department. 602 pp.

Idaho Department of Fish and Game (IDFG). 2010. Bighorn sheep management plan 2010. Idaho Department of Fish and Game, Boise, USA

Keeler-Wolf, T. 2007. Mojave Desert scrub vegetation. Pages 609-656 in: M. G. Barbour, T. Keeler-Wolf, and A. A. Schoenherr, editors. 2007. Terrestrial vegetation of California. Third edition. University of California Press, Berkeley.

Krausman, P. R., A. V. Sandoval, and R. C. Etchberger. 1999. Natural history of desert bighorn sheep. Pages 139-191 in R. Valdez and P. R. Krausman, editors. Mountain Sheep of North America. The University of Arizona Press, Tucson, AZ. 353 pp.

Leslie, D. M., Jr., and C. L. Douglas. 1979. Desert bighorn sheep of the River Mountains, Nevada. Wildl. Monogr. No. 66:1-56.

Mazaika, R., P. R. Krausman, and R. C. Etchberger. 1992. Forage availability for mountain sheep in Pusch Ridge Wilderness, Arizona. Southwest. Nat. 37:372-378.

McCarty, G. W. and J. A. Bailey. 1994. Habitat requirements of desert bighorn sheep. Colorado Div. of Wildlife Spec. Rep. No. 69, Denver. 27pp

McKinney, T., J. C. Devos Jr., W. B. Ballard, and S. R. Boe. 2006. Mountain Lion Predation of Translocated Desert Bighorn Sheep in Arizona. Wildlife Society Bulletin. Vol. 34, No. 5 (December 2006), pp. 1255-1263. Online: http://www.jstor.org/stable/4134258. Accessed 23 March 2013.

Miller, G. D., and W. S. Gaud. 1989. Composition and variability of desert bighorn sheep diets. Journal of Wildlife Management 53(3):597-606.

Monson, G. and L. Sumner (editors). 1980. The desert bighorn - its life history, ecology, and management. 370 pp.

NatureServe. 2013. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia. Online:http://www.natureserve.org/explorer. Accessed: 23 March 2013.

New Mexico Department of Game and Fish (NMDGF). 2003. Plan for the Recovery of Desert Bighorn Sheep in New Mexico 2003-2013.

New Mexico Department of Game and Fish (NMDGF). New Mexico's Bighorn Sheep. Online: http://www.wildlife.state.nm.us/conservation/bighorn/. Accessed 23 March 2013.

Oldemeyer, J. L., W. J. Barmore, and D. L. Gilbert. 1971. Winter ecology of bighorn sheep in Yellowstone National Park. Journal of Wildlife Management 35:257-269.

Pallister, G. L. 1974. The seasonal distribution and range use of bighorn sheep in the Beartooth Mountains, with special reference to the West Rosebud and Stillwater herds. Montana Fish and Game Department, Federal Aid Wildlife Restoration Project W-120-R5.

Quinn, M. A. 2004. Influence of habitat fragmentation and crop system on Colombia Basin strubsteppe communities. Ecological Applications 14:1634-1655.

Ramey, R. R., II. 1995. Mitochondrial DNA variation, population structure, and evolution of mountain sheep in the south-western United States and Mexico. Molecular Ecology 4:429-439.

Ramey, R. R., II. 2000. New perspectives on the evolutionary origins, historic phylogeography, and population structure of North American mountain sheep. Pages 9-16 in A. E. Thomas and H. L. Thomas, editors. Transactions of the 2nd North American Wild Sheep Conference. April 6-9, 1999, Reno, NV. 470 pp.

Rosentreter, R., and D. J. Eldridge. 2002. Monitoring biodiversity and ecosystem function: Grasslands, deserts, and steppe. Pages 199-233 in: P. L. Nimis, C. Scheidegger, and P. A. Wolseley, editors. Monitoring with lichens--monitoring lichens. Kluwer Academic Publishers, Dordrecht, The Netherlands.

Rubin, E. S., W. M. Boyce, and V. C. Bleich. 2000. Reproductive strategies of desert bighorn sheep. Journal of Mammalogy 81:769-786.

Ryder, T. J., E. S. Williams, K. W. Mills, K. H. Bowles, and E. T. Thorne. 1992. Effect of pneumonia on population size and lamb recruitment in Whiskey Mountain bighorn sheep. Proceedings of the Biennial Symposium of the Northern Wild Sheep and Goat Council 8:136-146.

Sanchez, O. *Borrego cimarrón*. In: Ceballos, G and G. Oliva, coords. 2005. Los mamíferos silvestres de México. Comisión Nacional para el Conocimiento y Uso de la Biodiversidad. Fondo de Cultura Económica. México. 986pp.

Schmidly, D. J. 2004. The mammals of Texas. Revised edition. University of Texas Press, Austin. xviii + 501 pp.

Schwantje, H. M. 1988. Causes of bighorn sheep mortality and dieoff--literature review. Working Report No. WR-35, Wildlife Branch, British Columbia Ministry of Environment, Victoria, BC.

Seegmiller, R. F., and R. D. Ohmart. 1981. Ecological relationships of feral burros and desert bighorn sheep. Wildlife Monographs No. 78:1-58.

Shackleton, D. M. 1985. Ovis canadensis. Mammalian Species 230:1-9.

Shackleton, D. M., C. C. Shank, and B. M. Wikeem. 1999. Natural history of Rocky Mountain and California bighorn sheep. In: R. Valdez and P. R. Krausman (editors). Mountain sheep of North America. University of Arizona Press, Tucson, Arizona. 353 pp.

Shannon, J. M., D. D. Olson, S. L. Peterson, J. C. Whiting, and J. T. Flinders. 2008. Can GIS Models Predict Habitat Use of Reintroduced Bighorn Sheep? In: Proceedings of the 28th Annual ESRI International User Conference, August 4–8, 2008.

Southwest Climate Change Network (SWCCN). 2008. Drought and the Environment. Available: http://www.southwestclimatechange.org/impacts/land/drought in October 2010.

Simpson, C. D., ed. 1980. Proceedings of the symposium on ecology and management of Barbary sheep. Texas Tech University Press, Lubbock.

Stelfox, J. G. 1971. Bighorn sheep in the Canadian Rockies: a history, 1800-1970. Canadian Field-Naturalist 85:101-122.

Stelfox, J. G. 1975. Range ecology of Rocky Mountain bighorn sheep in Canadian National Parks. Ph.D. thesis, University of Montana, Missoula.

Sugden, L. G. 1961. The California bighorn in British Columbia with particular reference to the Churn Creek herd. B.C. Department of Recreation and Conservation, Victoria.

Turner, J. C. 1979. Osmotic fragility of desert bighorn sheep red blood cells. Comparative Biochemistry and Physiology A 64:167-175.

Turner, J. C., and C. G. Hansen. 1980. Reproduction. Pp. 145-151 IN G. Monson and L. Sumner, eds. The desert bighorn: its life history, ecology, and management. University of Arizona Press, Tucson, AZ.

Turner, J. C., and R. A. Weaver. 1980. Water. Pp. 100-112 IN G. Monson and L. Sumner, eds. The desert bighorn: its life history, ecology, and management. University of Arizona Press, Tucson, AZ.

Turner, J.C. 1973. Water energy and electrolytic balance in the desert bighorn sheep (*Ovis canadensis cremnobates* Elliot). PhD dissertation. University of California, Riverside. 150pp.

U.S. Fish and Wildlife Service (USFWS). 5 August 2008. Designation of critical habitat for the Sierra Nevada bighorn sheep (*Ovis canadensis sierrae*) and taxonomic revision. Federal Register 73(151):45534-45604.

United States Fish and Wildlife Service (USFWS). 2000. Recovery Plan for Bighorn Sheep of the Peninsular Ranges, California. U.S. Fish and Wildlife Service, Portland. O.R. Xv +251pp.

Valdez, R., and P. R. Krausman. 1999. Description, distribution, and abundance of mountain sheep in North America. Pages 3-22 In: R. Valdez and P. R. Krausman, editors. Mountain sheep of North America. University of Arizona Press, Tucson. 353 pp.

Van Dyke, W. A. 1978. Population characteristics and habitat utilization of bighorn sheep, Steens Mountain, Oregon. M.S. thesis, Oregon State University, Corvallis.

Vander Haegen, W. M, S. M. McCorquodale, C. R. Pearson, G. A. Green, and E. Yensen. 2001. Wildlife of eastside shrubland and grassland habitats. Chapter 11, pages 317-341 In: D. H. Johnson and T. A. O'Neil. Wildlife-habitat relationships in Oregon and Washington. Oregon State University Press, Corvallis, OR.

Vander Haegen, W. M., F. C. Dobler, and D. J. Pierce. 2000. Shrub-steppe bird response to habitat and landscape variables in eastern Washington, USA. Conservation Biology 14:1145-1160.

Verts, B. J., and L. N. Carraway. 1998. Land mammals of Oregon. University of California Press, Berkeley. xvi + 668 pp.

Vranckx, G., H. Jacquemyn, B. Muys, and O. Honnay. 2011. Meta-analysis of susceptibility of woody plants to loss of genetic diversity through habitat fragmentation. Conservation Biology, November 2011.

Wakelyn, L. A. 1987. Changing habitat conditions on bighorn sheep ranges in Colorado. Journal of Wildlife Management 51:904-912.

Warrick, G. D., and P. R. Krausman. 1989. Barrel cacti consumption by desert bighorn sheep. Southwest. Nat. 34:483-486.

Wehausen, J. D., and R. R. Ramey, II. 2000. Cranial morphometric and evolutionary relationships in the northern range of Ovis canadensis. Journal of Mammalogy 81:145-161.

Wehausen, J. D., and R. R. Ramey. 1993. A morphometric reevaluation of the peninsular bighorn subspecies. Desert Bighorn Council Transactions 37:1-10.

# *F-4 Black-tailed Prairie Dog (Cynomys ludovicianus)*

## F-4.1 Conceptual Model

### F-4.1.1 Classification Comments

Prairie dogs are rodents within the squirrel family (*Sciuridae*); they occur only in North America. There are five species of prairie dogs in North America: the black-tailed prairie dog (*Cynomys ludovicianus*), the white-tailed prairie dog (*C. leucurus*), the Gunnison's prairie dog (*C. gunnisoni*), the Utah prairie dog (*C. parvidens*), and the Mexican prairie dog (*C. mexicanus*) (Pizzimenti 1975). The Utah and Mexican prairie dogs are currently listed as threatened (49 FR 22339) and endangered (35 FR 8495), respectively. Generally, the black-tailed prairie dog occurs east and north of the other four species in more mesic habitat.

The black-tailed prairie dog was first collected by members of the Lewis and Clark expedition of 1804-1806, the species was first described by Ord in 1815 from a specimen local to the Upper Missouri River (Hall and Kelson 1959); and they were extirpated from Arizona by 1961. The species is found in eastern Montana, eastern Wyoming, eastern Colorado, eastern New Mexico, southwestern North Dakota, western and central South Dakota, western and central Nebraska, western and central Kansas, western and central Oklahoma, northwestern Texas, and in a small area of south-central Canada, and was reintroduced to Arizona in 2008.

**Figure F-20. Black-tailed prairie dog (*Cynomys ludovidianus*).**Photo courtesy Arizona Game and Fish Department website, **www.azgfd.gov**.

### F-4.1.2 Biology and Distribution Status

#### *F.4.1.2.1 Distribution*

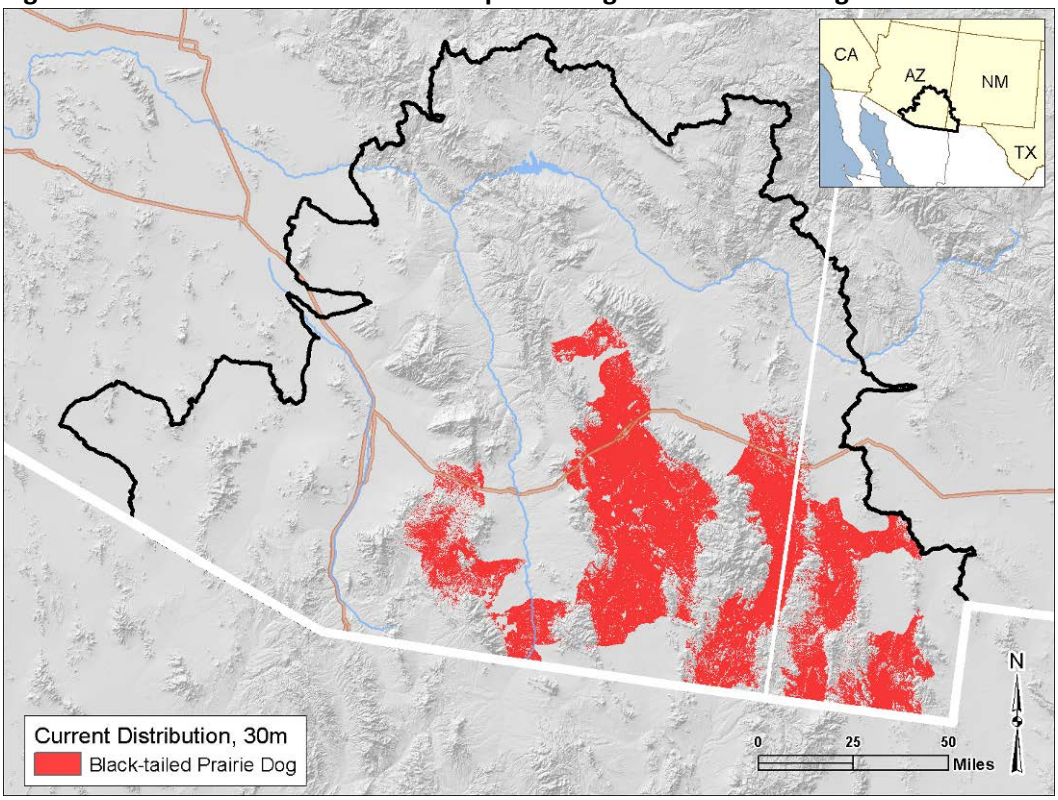
*North America*: The historic range of the black-tailed prairie dog included the Great Plains from southern Saskatchewan south to the Texas panhandle, and southern New Mexico and Arizona into extreme

northern Mexico (Prairie dog coalition, 2013). The US Fish and Wildlife Service (2000) indicated that "Today it occurs from extreme south-central Canada to northeastern Mexico and from approximately the 98th meridian west to the Rocky Mountains. Black-tailed prairie dogs are absent from most of the range which they occupied historically, although remnant populations are widely scattered across much of this area. Notably, many land tracts within the historic range have no black-tailed prairie dogs, some have a few black-tailed prairie dogs, and very few have significant numbers of black-tailed prairie dogs."

In relation to distribution of the black-tailed prairie dog the US Fish and Wildlife Service (2000) states "The extirpation of most historic populations (and burrow systems), vegetative community changes (e.g., brush invasion of grasslands), landscape changes (e.g., cropland conversion, urbanization), and the establishment of sylvatic plague in North America may limit any extensive reoccupation of its former range by the species. Additionally, most historic black-tailed prairie dog occupied habitat no longer exists and source populations for recolonization are often isolated."

*Madrean Archipelago ecoregion*:
Black-tailed prairie dog-occupied habitat existed in southeastern Arizona and New Mexico prior to rodent control efforts (Hall and Kelson 1959).

**Figure F-21. Distribution of black-tailed prairie dog in the MAR ecoregion.**




Information provided by Van Pelt (1999) noted the species in Cochise and Graham Counties. Information from Fagerstone and Ramey (1996) as delineated by the Black-footed Ferret Recovery Foundation documented the species in Cochise and Santa Cruz Counties as well as portions of Pima, Graham, Pinal, and Greenlee Counties. As of August 2013 there are four small colonies within the Las Cienegas National Conservation Area- the result of a reintroduction program launched by the Arizona Game and Fish Department in 2008, (Treadwell, personal communication, 2013).

By 2011, three sites within Las Cienegas NCA had been stocked, but due to extreme drought, each of these colonies had experienced population declines from high predation and low forage availability. To maintain stability and genetic integrity, eighty animals from two sources (New Mexico and Sonora, Mexico) were released at these three sites. It was also determined by a University of Arizona survivorship study conducted in 2011, that the drought conditions were forcing the prairie dogs to travel well outside their colonies and into the tall grass to search for food. This was resulting in high predation, causing the small populations to decline rapidly. It was also determined that annual offspring numbers were low. From 2009-2011, a total of 34 pups emerged, but only 10 emerged in 2011. To combat these issues, the project decided to provide supplemental feed. Initially, the additional food succeeded in reducing predation mortalities. In 2012, the feeding was started earlier in March with the hope of increasing the production of offspring. The result was a virtual population explosion when 132 pups emerged. This technique is now standard operating procedure for new colony establishment. This extra production also allowed the program to source animals for the fourth colony in 2012 from the first three (Treadwell, personal communication, 2013).

In New Mexico, black-tailed prairie dogs historically occupied the desert grasslands of the state. Specific reports by county vary as to the occurrence of the species. The range of the subspecies *C.l. arizonensis* overlaps with that of *C.l. ludovicianus* in Torrance, Chaves, and Lincoln counties (Schmitt 1995).

In 1996, surveys on black-tailed prairie dogs were conducted in New Mexico, including Colfax, Union, Harding, and Mora Counties, covering about 11,500 square miles (Van Pelt 1999). These surveys revealed the presence of a total of 1,191 black-tailed prairie dog individuals in over 41 colonies in Union County, 11 colonies in Colfax County, two colonies in Mora County, and 10 colonies in Harding County. Similar surveys were conducted in 1997 by Paternoster, revealing the presence of 33 colonies of black-tailed prairie dogs in Roosevelt County and 40 colonies in Curry County together totaling about 3,174 occupied acres. Although these two surveys revealed a total of 137 colonies of black-tailed prairie dogs in six counties, current systematic surveys such as these are lacking in the remaining counties of New Mexico (Van Pelt 1999).

Given some gaps in current information concerning their distribution, numbers, viability of isolated populations, impacts of habitat fragmentation, amount and extent of efforts to control (or eradicate) them, population impacts from plague, and other biological factors, the overall status of black-tailed prairie dogs in New Mexico needs further verification. However, there are no data available that would suggest populations in New Mexico are stable and/or improving (Van Pelt 1999).

#### *F.4.1.2.2 Population*

The U.S. Fish and Wildlife Service notes the unavailability of any recent, comprehensive, original, single-source, range wide estimate of occupied habitat for the black-tailed prairie dog and recognizes that such an evaluation would be very difficult to conduct (USFWS, 2000). Several authors have attempted to combine reports of occupied habitat from various sources to provide an overall estimate across the species' range (Fagerstone and Ramey 1996, Knowles 1995, Knowles 1998, Mulhern and Knowles 1995). However, these efforts are not comprehensive because remote portions of habitat have precluded equally accurate and comparable information across the vast species range. Populations are located irregularly, at varying densities, and periodically expand and contract due to various combinations of factors.

Most estimates of prairie dog population trends are based on estimates of the amount of occupied habitat rather than on numbers of individual animals because of the assoicated effort and cost. However, studies have demonstrated that densities of black-tailed praireid dog colonies range from 2 to 18 individuals per acre (5 to 45 per hectare) (Fagerstone and Ramey 1996, Hoogland 1995, King 1955, Koford 1958, Miller et al. 1996). Most prairie dog surveys do not estimate density because of the associated effort and cost. The US Fish and Wildlife Service believes that a review of various estimates of black-tailed prairie dog occupied habitat provides the best available and most reasonable means of determining population trends and the status of the species (USFWS, 2000).

The black-tailed prairie dog may be found intermittently in remnant populations throughout much of its historic range. However, within that range, the amount of occupied black-tailed prairie dog habitat has declined by about 94 to 99 percent since 1900 (Barko 1997, Fagerstone and Ramey 1996, Knowles 1998, Mulhern and Knowles 1995, Wuerthner 1997 and USFWS, 2000). Black-tailed prairie dogs had the most extensive range of all the species of prairie dogs; they probably occupied more area than all other species combined (Hoogland 1995).

At present, the black-tailed prairie dog may be found scattered in remnant populations throughout much of the range that it once occupied. A significant portion of existing black-tailed prairie dog occupied habitat rangewide occurs in a few large complexes. Approximately 36 percent of the remaining occupied habitat for the species in North America occurs in seven complexes, each larger than 10,000 acres (4,000 hectares). In 2000, the USFWS estimated that approximately 768,000 acres (311,000 hectares) of black-tailed prairie dog occupied habitat existed range wide (USFWS, 2000).

The Arizona Game and Fish Department first released 74 black-tailed prairie dogs in 2008 from the Ladder Ranch in New Mexico at the Las Cienegas National Conservation Area. At least four offspring were identified the following spring. An additional 107 prairie dogs were released in 2009 at the previous release site and a new release site which is also in the Las Cienegas National Conservation Area. Due to the sensitive nature of the species during re-establishment, the hunting season for black-tailed prairie dogs is closed. Risks to the species need to be minimized while the population grows and expands in remanant populations throughout much its historic range (AZGFD 2013).

In New Mexico, black-tailed prairie dog was identified as a species of greatest conservation need in the Comprehensive Wildlife Conservation Strategy for New Mexico (NMDGF, 2006). Formerly they were widespread and abundant east of the Rio Grande and in the grasslands of southwestern New Mexico. Colonies were often reported in habitat such as open woodland, and in the southwestern part of the state they occupied semidesert conditions (Findley et al. 1975).

In Northern Mexico, the population size of black tailed prairie dogs is unknown, however estimates conducted in 1990 in the Pancho Villa - Monteverde - Loma de los Ratones y Buenos Aires, Chihuahua, found between 258,000 and 380,000 individuals. In 1993, it was estimated there were over one million individuals distributed in a complex of populations dispersed in approximately 55,000 hectare (136,000 acres); making it, at that time, the largest population in North America (CONABIO, 2011). According to the Arizona Game and Fish Department, there are still two extant colonies in Sonora, Mexico, just south of Las Cienegas NCA in Arizona. In 2011, AGFD and CEDES personnel completed density mapping at these colonies. The La Mesa colonies had 177 acres, and the population estimate was 1,351 individuals with a 95% confidence interval of 931 to 1,771 animals. The Las Palmitas colony had 146 acres, with a population estimate of 1,905 individuals and 1,440 to 2,371 animals at the 95% confidence interval (Treadwell, personal communication, 2013).

### F.4.1.2.3 Reproduction

*Breeding Systems:* Black-tailed prairie dogs are not prolific in comparison to many other rodents. Several biological factors determine the reproductive potential of the species. Females usually do not breed until their second year and typically have 2 to 3 reproductively active years (Hoogland 1995, King 1955, Knowles and Knowles 1994). Survival of young prairie dogs can be high in some circumstances, especially in low density populations where habitat resources are plentiful and repressive factors such as control or disease are not operative (Garrett et al. 1982); however, much lower rates of annual increase or even reductions in colony size can occur where vegetation hinders expansion (Osborn and Allan 1949), or due to high predation rates.

Mating is polygynous, typically with a single male mating with multiple females within his coterie. In cases where there is more than one resident male, usually siblings, females will mate with both. Reproduction occurs once per year in spring, although the timing varies with latitude. Females are typically sexually receptive for only one day of the year (Hoogland 1995) though if conception fails they can undergo a second estrous.

Young are born blind, naked and mostly helpless (Hoogland 1995), and do not emerge from the burrow until around six weeks of age, and are weaned shortly after that (Davis and Schmidly 1994). Interestingly, after emerging from the burrow, but prior to the end of lactation, pups may nurse from females other than their own mother, in an example of 'cooperative breeding' (Hoogland 1995; Texas Parks and Wildlife 2006). Females remain in their natal coterie for life, while males disperse before their first breeding season (Hoogland 1995). Likewise, adult males rarely remain within the same coterie for more than two breeding seasons, probably to reduce the possibility that they will mate with their own offspring. Females can live up to eight years of age, whereas males tend not to live longer than five years in the wild (Hoogland 1995).

*Gestation:* 33-38 days. Pups are born in April or May.

*Litter size:* Female black-tailed prairie dogs produce a single litter, usually 4-5 pups, annually (Hoogland 1995, Knowles and Knowles 1994). Other authors mention an average litter size of 3 to 4 pups with the number of pups ranging between 1 and 8 (USFWS, 2000). One female may produce from 0 to 20 young in her lifetime.

*Breeding Season:* February through March.

*Birthing Season:* April or May.

### F.4.1.2.4 Life History

Prairie dogs are small, stout ground squirrels. The total length of an adult black-tailed prairie dog is approximately 14-17 inches; the weight of an individual ranges from 1 to 3 pounds, males being larger than females. Individual appearances within the species vary in mixed colors of brown, black, gray, and white. Their black-tipped tail and long claws in the front legs for digging are characteristic (CONABIO 2011, Hoogland 1995, USFWS 2000).

Black-tailed prairie dogs are diurnal, burrowing animals. Individuals spend most of the day above ground. They do not hibernate as do white-tailed, Gunnison's, and Utah prairie dogs but may remain underground during bad weather, with a month or more being documented during a severe winter period (Hoogland 1995, Tileston and Lechleitner 1966). The species is very social, living in population

aggregations called colonies, towns, or villages (Hoogland 1995, IUCN 2013). Historically, they generally occurred in large colonies that contained thousands of individuals, covered hundreds of thousands of acres, and extended for miles (Bailey 1905). Most existing colonies are much smaller. When unsuitable habitat such as a hill, tall vegetation, or a stream divides a prairie dog colony, the resulting sub-colonies are called wards (King 1955). Within colonies, prairie dogs live in territorial, harem-polygamous family groups called coteries (Hoogland 1995).

Black-tailed prairie dogs exhibit a high degree of social organization, living in enormous colonies known as 'towns' containing from hundreds to millions of individuals (Hoogland 1995, IUCN 2013). Each colony shares an elaborate network of burrows for shelter and protection against predators, often covering areas of 100 hectares or more (IUCN 2013), with the largest ever recorded colony covering 65,000 square kilometers and containing an estimated 400 million animals (Hoogland 1995, IUCN 2013). Colonies are subdivided into 'wards', and then into smaller family units called 'coteries', populated by a group of closely related females, one or two territorial males, and any offspring under two years of age (Hoogland 1995, Texas Parks and Wildlife 2006). Members of a coterie share food supplies outside of the breeding season and cooperate to aggressively defend their territory from neighbors (Hoogland 1995, Smithsonian 2006). However, while males respond strongly to intrusion by other males they seem oblivious to invading females; females, by contrast, show the most hostility toward invading females. During the breeding season, females aggressively defend their natal burrow against other females and, given the opportunity, will even raid the burrows of other females and kill their pups (Hoogland 1995).

The colonial nature of prairie dogs, especially the black-tailed prairie dog, is a significant characteristic of the species. Hoogland (1995) described the sociality, demography, and population dynamics of the black-tailed prairie dog. Coloniality offers an effective defense mechanism by aiding in the detection of predators and by deterring predators through mobbing behavior. It increases reproductive success through cooperative rearing of juveniles and it aids parasite removal via shared grooming. However, it has been noted that coloniality promotes the transmission of disease, which can significantly suppress populations (Olsen 1981, Hoogland 1995). Accordingly, disease may play a major factor in the population dynamics of the species.

### F-4.1.3 Habitat and Ecology

Many authors have recognized the biological importance of the black-tailed prairie dog as a keystone species (Agnew et al. 1986, Ceballos and Pacheco 1997, Clark et al. 1982, Kotliar et al. 1999, Miller et al. 1994, Reading et al. 1989). Keystone species influence ecosystem functions through their activities in unique and significant ways. The ecological effect caused by a keystone species is disproportionate to its numerical abundance and its removal or decline initiates changes in ecosystem structure and a decline in overall species diversity (Kotliar et al.1999, Mills et al. 1993, Paine 1980, Power et al. 1996, Terborgh 1988). However, Stapp (1998) questioned whether the black-tailed prairie dog is truly a keystone species. He recognized various ecological values of the species, but challenged other authors' view of the overall role of the species. Kotliar et al. (1999) concluded that prairie dogs provide some unique functions compared to other herbivores in the system and that continued decline of the species may lead to a substantial erosion of biological diversity; and, therefore, keystone status is appropriate. The extent to which these interrelationships directly affect the black-tailed prairie dog itself is largely unknown.

Prairie dogs act in several roles inasmuch as they are prey, provide shelter, modify vegetation, and influence ecological processes in a manner not entirely duplicated by other prairie herbivores (Ceballos and Pacheco 1997, Kotliar et al. 1999, List et al. 1997, Miller et al. 1994, Wuerthner 1997). While the

black-tailed prairie dog creates habitat for itself and other species, it also is affected by other species. For example, prairie dogs can create preferential grazing opportunities for herbivores that in turn create opportunities via grazing for the expansion of prairie dog colonies at their perimeters. However, the degree to which the black-tailed prairie dog itself is influenced by these and other prairie species, particularly ungulates, is not well understood. For example, the removal of large numbers of bison (*Bison bison)* and other native ungulates from the North American prairie may have had effects on the ecology of the black-tailed prairie dog that can no longer be fully evaluated. Similarly, the periodic effects of fire no longer influence much of the remaining fragmented prairie environment.

#### *F.4.1.3.1 Habitat*

The black-tailed prairie dog lives in arid and semi arid deserts and grasslands of North America (CONABIO, 2011). Family groups live in burrows in dry prairies with short grass (<30 cms is preferred). Their burrows have an entrance that is surrounded by a pile of dirt, looking like a 'volcano' protecting it from floods and providing good visibility to defend against predators.

Augustine and Baker (2013) note that "our findings and those from previous studies show that areas in the North American Great Plains with prairie dog colonies support higher densities of at least 9 vertebrate species than sites without colonies. Prairie dogs affect habitat for these species through multiple pathways, including creation of belowground refugia, supply of prey for specialized predators, modification of vegetation structure within colonies, and increased landscape heterogeneity."

*Terrain:* Prairie dogs prefer flat lands, with little to no slope, well-drained soils and few rocks (CONABIO, 2011). Roe and Roe (2003) describe the ideal soil as <30% clay, ~50% sand and >70% silt. In Mexico, black-tailed prairie dogs are found between 1400 and 1600 meters (~4600 – 5250 feet) of elevation (Pacheco and Ceballos 2005). Rangewide, they occur between 700 and 2200 meters (~2300 – 7220 feet) elevation.

*Distance from accessible water:* The prairie dog gets most of the water it needs from the plants it eats.

*Vegetation:* Black tailed prairie dogs forages selectively from the plants available in its habitat. Diet also varies seasonally. In the summer black tailed prairie dogs prefer to feed upon wheatgrass (*g. Agopyron*), buffalo grass (*g. Bromus*), grama (*g. Bouteloua*), rabbitbush (*g. Chrusothamnus*), and globemallow (*g. Sphaeralcea*). In the winter they eat prickly pear cactus (*g. Opuntia*), thistles (*g. Cirsium*), and various roots (Hoogland 1995).

#### *F.4.1.3.2 Landscape Connectivity*

Prairie dog dispersal is usually limited to approximately 3 miles (5 kilometers) or less, and individuals dispersing from home colonies generally move into an established colony rather than attempting to initiate a new colony (Garrett and Franklin 1988, Hoogland 1995). Black-tailed prairie dogs avoid certain landforms and soils, and preferentially select some landforms and some soils. These limitations could restrict recruitment of animals into small and declining isolated populations and favor reestablishment of individuals in nearby, recently abandoned colonies over the establishment of new colonies. Prairie dogs use human-made land alterations, like roads, to assist in landscape-level emigration and discovery of potential new habitat for colonization. King (1955) observed two types of emigration among black-tailed prairie dogs. The first type occurred when yearling males move in the spring following the appearance of young in the colony. The second type of emigration consisted of older adults leaving a colony in the spring or summer, possibly to avoid excessive attention and interference by the young-of-the-year.

#### F.4.1.3.3 Food

Black-tailed prairie dogs mainly consume grasses, sedges, forbs (flowering plants), roots and seeds, though they are also known to eat grasshoppers, beetles and other insects (USFWS 2000). In northern Mexico, prairie dogs consume grasses such as *Agopyron* spp., *Bromus* spp., and *Bouteloua* during summer and in winter they consume cactus *Opuntia* spp., herbs (*Cirsium* spp.), shrubs (*Chrysothamnus* spp.) and roots (CONABIO 2011). The animals usually eat all the plants right around its burrow.

#### F.4.1.3.4 Phenology

Black-tailed prairie dogs are diurnal, burrowing animals. Sunrise to 0900 and 1500 to sunset are the peaks of activity on a colony during mid summer (Clark et al. 1971 in Keinath 2004). This pattern is largely driven by the temperature at ground level during these months (June-August), which can be too hot for extended activity above ground (Grant 1995 in Keinath 2004). However, in early season (February-Apri) and late season (September – November), the peak of activity is unimodal and occurs around 1300 (Clark et al. 1971 in Keinath 2004). Activity is most common between the temperatures of 9 to 21 degrees Celsius (15 to 70 degrees F), although short bouts of activity can occur between the temperatures of 24 to 27 degrees Celsius (75 to 80 degrees F). Activity bouts can be affected by high wind speeds (exceeding 11 m/s) (Grant 1995 in Keinath 2004).

Daily behaviors of white-tailed prairie dog observed by Orabona-Cerovski (1991) indicate that they spend about one-third of their time above ground feeding (x = 36%) and one-third sitting erect (x = 33%) possibly scanning for predators. Sitting horizontally accounts for 16% of the daily activity. The remainder of the behaviors (running, vocalizing, fighting, playing, kissing, digging, grooming) made up the remaining 15% of time spent above ground (Orabona-Cerovski 1991 in Keinath 2004).

Mating season runs from February through March. A month after mating, the female will have three to four pups. The pups are born naked and with their eyes closed. They stay in the burrow for about six weeks. They are weaned when they are about seven weeks old but will stay near their mother for another two weeks. Colony activity is at its highest around May and will not begin to decline until late July, when adult males begin to disappear below ground. Adult females will then begin to descend into burrows two to three weeks later (Clark 1973).

In the fall, prairie dogs put on a layer of fat to help them survive in the winter. Black tails do not hibernate as do white-tailed, Gunnison, and Utah prairie dogs (Hoogland 1995, Tileston and Lechleitner 1966), but when the winter weather is extremely cold or snowy, they may go into a light hibernation-like sleep and stay in their burrows for a few days.

#### F.4.1.3.5 Predators

Although reports vary as to those species that require prairie dogs for their survival, at least 9 species depend directly on prairie dogs or their activities to some extent, and another 137 species are associated opportunistically (Kotliar et al.1999). The most obligatory species of this group is the black-footed ferret (*Mustela nigripes*). Probably no other species has a more clearly documented dependence on another species than does the black-footed ferret on the prairie dog (Anderson et al. 1986, Biggins et al. 1986, Clark 1989, Forrest et al. 1988, Henderson et al. 1974, Hillman 1968, Miller et al. 1996). The black-footed ferret is a federally listed endangered species that depends upon prairie dogs as a source of food and uses its burrows for shelter. Any actions that kill prairie dogs or alter their habitat could prove detrimental to black-footed ferrets occupying the affected prairie dog towns.

Other predators include the coyote (*Canis latrans*), kit fox (*Vulpes macrotis*), swift fox (*Vulpes velox*), gray fox (*Urocyon cinereoargenteus*), badger (*Taxidea taxus*), bobcat (*Lynx rufus*), mountain lion (*Puma concolor*), long-tailed weasel (*Mustela frenata*), ferruginous hawk (*Buteo regalis*), burrowing owl (*Athene cunicularia*), Golden eagle (*Aquila chrysaetos*), Bald eagle (*Haliaeetus leucocephalus*), and reptiles such as the Gopher snake (*Pituophis melanoleucus*) and rattlesnakes (*Crotalus* spp.) (CONABIO 2011; USFWS 2000). Clearing away the plants around the burrow helps the prairie dog spot predators.

#### *F.4.1.3.6 Disease and Parasites*

Sylvatic plague—an exotic disease that entered North America in 1900—threatens the survival of prairie dogs. The influence of sylvatic plague on black-tailed prairie dog populations has had significant depressant effects on remnant populations since the late 1900s (USFWS 2000). However, given that about 10% of the historic range is both plague-free and available, limited immune response has been observed in some individuals, and some sites have demonstrated the ability to recover to pre-plague levels, the USFWS has concluded that plague no longer appears to be as significant a threat as previously thought, and is not likely to cause the black-tailed prairie dog to become an endangered species in the foreseeable future (USFWS 2004). In Arizona, sylvatic plague may not occur at elevations below 4,500 feet (1,372 meters), where most of the species occurred historically (Van Pelt, 1999, USFWS, 2000). Bubonic plague from Europe transmits very easily to colonies and wipes them out fast (Prairie Dog Coalition 2013).

Common Black tailed prairie dog parasites include fleas (*Opisocrostis hirsutus*, *O. tuberculatus*, *O. labis*, *Pulex simulans*, *P. irritans* and *Leptopsylla segnis*), mites (*Ixodes kingi* and *Atricholaelaps glasgowi*) and lice (CONABIO 2000, IUCN 2013).

#### *F.4.1.3.7 Threats*

Four major impacts have had a substantial influence on black-tailed prairie dog populations. The first major impact on the species was the initial conversion of prairie grasslands to cropland across its whole range from approximately the 1880s through the 1920s. The second major impact on the species was large-scale control efforts conducted from approximately 1918 to approximately 1972 in efforts to reduce competition between prairie dogs and domestic livestock and to minimize destruction of agricultural crops by prairie dogs. The third major impact on the species was the inadvertent introduction of an exotic disease, sylvatic plague, from the Old World into North American ecosystems around 1900, with the first recorded impacts on the black-tailed prairie dog in 1946 (USFWS 2000). The fourth major impact on the species is habitat fragmentation, which is a serious threat at the local level due to inbreeding. Additionally the remaining colony could be heavily impacted by natural or human-related catastrophic events such as plague or illegal shooting.

Humans pose the greatest threat to prairie dogs, frequently poisoning and shooting the animals and often plowing or bulldozing entire colonies for cropland or development. Many ranchers dislike the animals because they eat grass that ranchers would rather have for their livestock (CONABIO 2011). As a result, the former range and numbers of the black-tailed prairie dog have been dramatically reduced, and the considerable reduction in population numbers has also seriously threatened, amongst others, the black-footed ferret, for which they were virtually sole prey.

The main reason black-tailed prairie dogs were extirpated from Arizona is extensive extermination efforts that were initiated in the early 1900s. Prairie dogs were considered pests and ranchers viewed them as competition with their cattle for grass. While massive poisoning efforts are no longer occurring in Arizona, extermination still occurs in many places within its' range. Poisoning not only kills prairie

dogs but, depending on the type of poison, often kills some animals that eat the poisoned prairie dog. Continued use of poisons could not only reduce prairie dog numbers, but could also cause a decline in the populations of the animals that prey upon them (AZGFD 2013).

*Conservation status* – The Black tailed prairie dog is classified as 'Least Concern' (LC) on the IUCN Red List; 'Rare' in the United States and 'Threatened' in Mexico.

The Arizona Game and Fish Department (AZGFD) classifies all prairie dogs native to the State, black-tailed and Gunnison's, as nongame mammals. In 1999, the hunting season for black-tailed prairie dogs was closed (Van Pelt 1999). Arizona does not require the eradication of prairie dogs for agricultural purposes or promote recreational shooting of prairie dogs (Van Pelt 1999). The black-tailed prairie dog is listed as Species of Greater Conservation Need. This list is being modified into a State list of "Wildlife of Special Concern" and AZGFD has proposed the black-tailed prairie dog for inclusion on the new list (Van Pelt, Arizona Game and Fish Department, pers. comm. 1998; USFWS 2000).

In New Mexico the black-tailed prairie dog was identified as a species of greatest conservation need in the Comprehensive Wildlife Conservation Strategy for New Mexico (NMDGF, 2006).

### F-4.1.4 Change Agent Characterization

This section of the conceptual model presents a narrative description of the primary change agents and current knowledge of their effects on Black tailed prairie dog populations. The main change agents identified are the destruction and modification of habitat, disease and pet trade and shooting.

#### *F.4.1.4.1 Destruction and Modification of Habitat*

Black tailed prairie dog habitat destruction and modification to croplands is the biggest and most widespread change agent for the species across its distribution range. Significant destruction, modification, and curtailment of black-tailed prairie dog habitat and range have occurred for many years. By the end of the 1990s, approximately 33 percent of the historic range of the black-tailed prairie dog and 37 percent of the suitable habitat within its range had been converted to cropland uses in the United States. This fundamental land use change resulted in significant destruction of black-tailed prairie dog habitat, mostly in the eastern portions of the species' range where adequate precipitation favored farming (USFWS 2000). The present threat of large-scale destruction of black-tailed prairie dog habitat through cropland conversion is much less than in the early days of agricultural development due to the fact that land with the highest potential for traditional farming uses was converted many years ago. However, conversion of rangelands and prairie habitat to cropland still occurs in some areas due to continuing improvements in intensive agricultural techniques (USFWS 2000).

Although efforts to control prairie dogs continue in New Mexico, the New Mexico Department of Game and Fish (NMDGF) believes these control activities are now more localized in application and on a smaller scale than in the recent past. Control of prairie dogs is still done for the purpose of protecting grazing lands, agricultural crops, and farming developments. Other control activities, usually in close proximity to human developments, include eradication or control of prairie dogs for their presumed role in transmission of sylvatic plague to humans. Management of black-tailed prairie dogs has not been limited to control. Black-tailed prairie dogs have recently been reintroduced in two areas in Sierra County, and two areas in Hidalgo County are being discussed for possible reintroduction in areas of private ownership on the Gray Ranch (Van Pelt 1999).

In Mexico, List et al. (1997) reported that occupied black-tailed prairie dog habitat in Mexico declined by 34 percent between 1988 and 1996, in part due to rangeland conversion due to farming. In Arizona, the AZGFD noted a 35 percent reduction in grassland habitat along the San Pedro River to mesquite woodland invasion that could be due to the cumulative effects of fire suppression, grazing practices and perhaps the elimination of the black-tailed prairie dog (USFWS 2000).

Within the MAR, black-tailed prairie dog habitat and regional historical ranges extend north and south of the U.S.-Mexico border. Prairie dog populations occurring in the Madrean ecoregion can be affected by physical barriers to gene flow, recruitment, behavior, and habitat loss and destruction, caused by border infrastructure and other border-related activities.

#### *F.4.1.4.2 Disease*

The Fish and Wildlife Service believes that sylvatic plague is likely the most important factor in recent reductions of many black-tailed prairie dog populations throughout a significant portion of the range of the species. Approximately 66 percent of the species' range has been affected by plague (USFWS 2000). Sylvatic plague is a disease caused by the bacterium *Yersinia pestis*, which fleas acquire from biting infected rodents and other species and then transmit via a bite. The disease also can be transmitted directly between animals. The term "sylvatic" refers to the occurrence of the disease in the wild. It also may be referred to in its bubonic, pneumonic, or septicemic forms, depending on the affected portion of the organism in which it is observed. Given the communicability and lethality of plague, an epizootic may affect an entire colony in a similar manner as a pathogen may affect an individual animal. An entire black-tailed prairie dog colony may disappear just as an individual black-tailed prairie dog would die from a plague infection. Plague, once established in an area, becomes persistent and periodically erupts, with the potential to extirpate local black-tailed prairie dog populations (USFWS 2000).

#### *F.4.1.4.3 Collection of Wild Animals and Shooting*

Herron (Texas Parks and Wildlife Department, pers. comm. 1999; USFWS 2000) and others have reported that black-tailed prairie dogs are removed from the wild for sale as pets. Herron was aware of three commercial operators who collectively removed approximately 5,000 individuals from the Texas panhandle and other States annually in the 1990s, although these efforts may have declined. One animal export company in Texas noted that over the past 4 years their company has bought and sold approximately 20,000 black-tailed prairie dogs, largely from the same locations in western and northwestern Texas (Shaw et al. 1993). Miscellaneous reports indicate that this practice occurs elsewhere in the species' range, but the extent of removal of individuals from the wild for use as pets is unknown (USFWS 2000).

One factor impacting black-tailed prairie dog populations in some local areas is recreational (sport or varmint) shooting. Extensive shooting, especially of pregnant females or females nursing young, could significantly reduce annual recruitment and change the ultimate population dynamics of a colony. Gross estimates of the number of modern shooters of prairie dogs and their potential take, based on reports from the field, suggest that hundreds of thousands of black-tailed prairie dogs are probably shot across their range annually. Small local populations already depressed by disease and other adverse influences may suffer shooting impacts as additive losses. Shooting impacts also may contribute to population fragmentation and preclude or delay recovery of colonies reduced by other factors such as sylvatic plague.

### F-4.1.5 Ecological Status: Key Ecological Attributes and Indicators

This section of the conceptual model addresses Key Ecological Attributes and their potential indicators. The ecological status is a way of describing current status via criteria, functionality, or levels of attributes and asks if they are within the normal range of variation.

#### *F.4.1.5.1 Key Ecological Attributes*

Table F-8 identifies the key ecological attributes for this CE within the Madrean Archipelago ecoregion. A **key ecological attribute** of a focal ecological resource is a characteristic of the resource's biology, ecology, or physical environment that is critical to the resource's persistence in the face of both natural and human-caused disturbance. Alteration of such a characteristic beyond some critical range of variation will lead to the degradation or loss of the resource within decades or less. The KEAs table lists the identified key ecological attributes, with a brief definition, a rationale for why it is important for the CE, and a listing of stressors or change agents that might be affecting the key attribute.

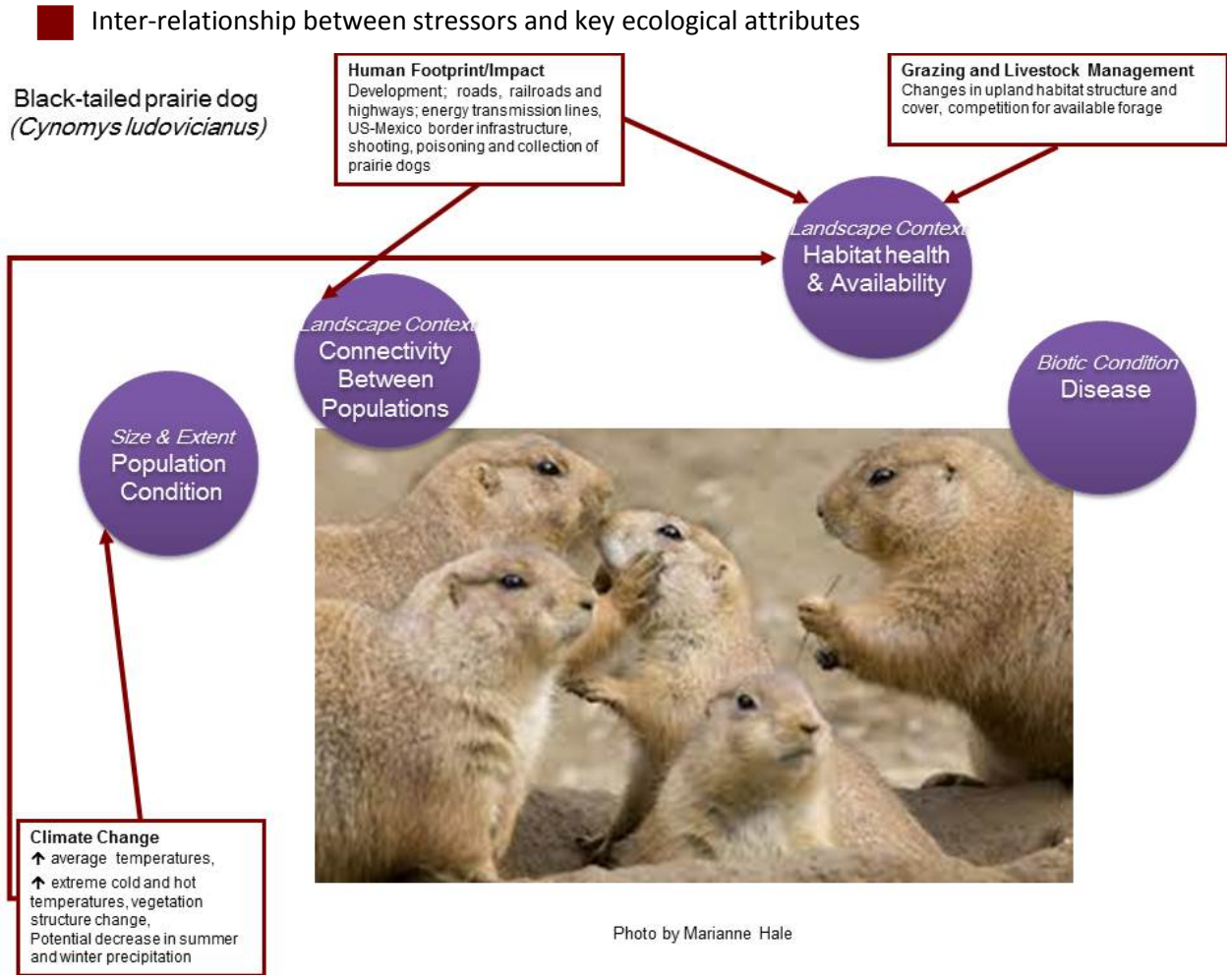
**Table F-8. Key ecological attributes (KEA) of Black-tailed Prairie Dog (Cynomys ludovicianus)** in the Madrean Archipelao ecoregion. *Please note that no spatial assessment was done for this species.*

| ***KEA Class:*** **KEA Name** | **Definition** | **Rationale** | **Stressors** |
|---|---|---|---|
| ***Size/Extent:*** **Population Condition** | The size and age and sex distribution of prairie dog populations. | Prairie dog population size and demographic structure indicates the viability and health of a population, metapopulation or prairie dog colony. Females do not breed until their second year and reproduction occurs once per year during spring. | Shooting of females, pregnant females or females nursing young can significantly reduce annual recruitment. Increased mortality by any cause – poisoning. Extermination efforts by humans had previously extirpated prairie dogs from Arizona. |
| ***Landscape Context:*** **Habitat health and availability** | The extent of intact arid and semi arid deserts and short-grass grasslands suitable for prairie dogs. | Prairie dogs prefer flat lands, with little to no slope, well-drained soils and few rocks (CONABIO, 2011). Roe and Roe (2003) describe the ideal soil as <30% clay, ~50% sand and >70% silt. In Mexico, Black tailed prairie dogs are found between 1400 and 1600 meters of elevation (Pacheco and Ceballos 2005). | Significant destruction, modification, and curtailment of black-tailed prairie dog habitat and range have occurred for many years. Black tailed prairie dog habitat destruction and modification to croplands is the biggest and most widespread change agent for the species across its distribution range.Historically, an important stressor for established prairie dog populations has been diminishing or destroying their habitat, mainly due to conversion to agriculture and cattle grazing (USFWS 2000). Livestock management acts as a change agent for prairie dogs through habitat destruction, change of vegetative communities, and direct mortality due to eradication control efforts. |
| ***Landscape Context:*** **Connectivity between populations** | The connectivity between prairie dog populations along US-Mexico borderlands | Black-tailed prairie dog habitat and regional historical ranges extend north and south of the U.S.-Mexico border. Prairie dog populations occurring in the Madrean ecoregion can be affected by physical barriers to gene flow, recruitment, behavior, and habitat loss and destruction, caused by border infrastructure and other border-related activities. | US-Mexico Border infrastructure & border related activity physical reduce or prevent movement between prairie dog populations north and south of the border that were historically connected. |
| ***Biotic Condition:*** **Prairie dog health and disease** | Disease | The Fish and Wildlife Service believes that sylvatic plague is likely the most important factor in recent reductions of many black-tailed prairie dog populations throughout a significant portion of the range of the species. Approximately 66 percent of the species' range has been affected by plague (USFWS 2000). | Sylvatic plague is caused by the bacterium *Yersinia pestis*, acquired and transmitted by fleas. |

### F-4.1.6 Conceptual Model Diagram

A conceptual model diagram for the CE provides a visual summary representation of the Key Ecological Attributes and species' life cycle (Figure F-22). These diagrams are intended to show how various stressors interact with categories of Key Ecological Attributes to highlight important drivers for species ecological integrity. The arrows indicate relationships between stressors and but do not indicate the nature of the influence (i.e. positive or negative).

**Figure F-22. Conceptual model diagram for the black-tailed prairie dog, showing key ecological attributes** (by class) for this species, and indicating relationships between stressors and KEAs.




## F-4.2 Ecological Status Assessment

No ecological status assessment was conducted for the black-tailed prairie dog due to the unique circumstances of the species in the MAR and inadequate available data. The species was extirpated in the MAR, but has recently been re-introduced. There are currently only a few reintroduced colonies in the region, and they are heavily managed. While development, fire regime, and invasive species may affect black-tailed prairie dog habitat under ordinary circumstances, such effects may not be present or

are otherwise mitigated under an intensive reintroduction and management program. In addition, spatial data for KEAs such as disease, human take of individuals, and livestock management was not sufficient to support an ecological status assessment based on those factors. Management questions tied to reintroduction are highly locally driven and therefore not readily addressed by the standard status assessments or data collected for this REA. The current distribution of the species was modeled using data and parameters from the Arizona Game and Fish Department's HabiMap with the modeling extended into the New Mexico portion of the MAR. The modeled potential distribution for black-tailed prairie dog (Figure F-21) provides landscape-level information that can be used to inform management decisions about where and whether to manage for or reintroduce prairie dogs.

## F-4.3 References for the CE


Agnew, W., D.W. Uresk, and R.M. Hansen. 1986. Flora and fauna associated with prairie dog colonies and adjacent ungrazed mixed-grass prairie in western South Dakota. Journal of Range Management 39(2):135-139.

Anderson, E., S.C. Forrest, T.W. Clark, and L. Richardson. 1986. Paleobiology, biogeography, and systematics of the black-footed ferret, *Mustela nigripes* (Audubon and Bachman), 1851. Great Basin Naturalist Memoirs 8:11-62.

Augustine, D. J. and Baker, B. W. (2013). Associations of Grassland Bird Communities with Black-Tailed Prairie Dogs in the North American Great Plains. Conservation Biology, 27: 324–334.

AZGFD Black-tailed Prairie Dog Reintroduction (Accessed on March 26, 2013). http://www.azgfd.gov/w_c/nongameandendangeredwildlifeprogram/Black-tailedPrairieDog.shtml

Bailey, V. 1905. Biological survey of Texas. U.S. Department of Agriculture, Bureau of Biological Survey. North American Fauna No. 25. Washington, D.C. Pages 89-92.

Barko, V.A. 1997. History of policies concerning the black-tailed prairie dog: a review. In Proceedings of Oklahoma Academy of Science 77:27-33.

Biggins, D.E., M.H. Schroeder, S.C. Forrest, and L. Richardson. 1986. Activity of radio-tagged black-footed ferrets. Great Basin Naturalist Memoirs 8:135-140.

Ceballos, G., and J. Pacheco. 1997. The relationship of prairie dogs and regional biodiversity in Mexico. In Seventh International Theriological Congress, Acapulco, Mexico, September 6-11, 1997. Page 52.

Clark, T.W. 1989. Conservation biology of the black-footed ferret, *Mustela nigripes*. Wildlife Preservation Trust Special Scientific Report No. 3. Wildlife Preservation Trust International, Philadelphia, Pennsylvania. 175 pp.

Clark, T.W., T.M. Campbell III, D.G. Socha, and D.E. Casey. 1982. Prairie dog colony attributes and associated vertebrate species. Great Basin Naturalist 42(4):572-582.

CONABIO (2011). Fichas de especies prioritarias. Perrito de la Pradera Cola Negra (*Cynomys ludovicianus*) Comisión Nacional de Áreas Naturales Protegidas y Comisión Nacional para el Conocimiento y Uso de la Biodiversidad, México D.F.

Davis, W.B. and Schmidly, D.J. (1994) ***The Mammals of Texas***. Texas Parks and Wildlife Department, Texas. Available at: http://www.nsrl.ttu.edu/tmot1/

Fagerstone, K.H., and C.A. Ramey. 1996. Rodents and lagomorphs. In Rangeland Wildlife. P.R. Krausman, ed. The Society for Range Management. Denver, Colorado. Pages 83-132.

Findley, J.S., A.H. Harris, D.E. Wilson, and C. Jones. 1975. Mammals of New Mexico. University of New Mexico Press, Albuquerque, New Mexico. xxii + 360 pp.

Forrest, S.C., D.E. Biggins, L. Richardson, T.W. Clark, T.M. Campbell, K.A. Fagerstone, and E.T. Thorne. 1988. Population attributes for black-footed ferret (*Mustela nigripes*) at Meeteetse, Wyoming, 1981-1985. Journal of Mammalogy 69:261-273.

Garret, M.G., J.L. Hoogland, and W.L. Franklin. 1982. Demographic differences between an old and a new colony of black-tailed prairie dogs (*Cynomys ludovicianus*). American Midland Naturalist.108(1):51-59.

Garrett, M.G., and W.L. Franklin. 1988. Behavioral ecology of dispersal in the black-tailed prairie dog. Journal of Mammalogy 29(2):236-250.

Hall, E.R., and K.R. Kelson. 1959. The Mammals of North America. Volume I. Ronald Press. New York, New York. Pages 364-366.

Henderson, F.R., P.F. Springer, and R. Adrian. 1974. The black footed ferret in South Dakota. South Dakota Game, Fish, and Parks Department Technical Bulletin 4. 37 pp.

Hillman, C.N. 1968. Field observations of black-footed ferrets in South Dakota. Transactions of the North American Wildlife and Natural Resources Conference 33:433-443.

Hoogland, J. (1995) *The Black-tailed Prairie Dog: Social Life of a Burrowing Mammal*. University of Chicago Press, Chicago.

IUCN Red List (Accessed on March, 2013) http://www.iucnredlist.org

Keinath, D.A. 2004. Species assessment for white-tailed prairie dog (*Cynomys leucurus*) in Wyoming. Prepared for U.S. Department of Interior, Bureau of Land Management, Wyoming State Office, Cheyenne, Wyoming.

King, J.A. 1955. Social behavior, social organization, and population dynamics in a black-tailed prairie dog town in the Black Hills of South Dakota. Contrib. Lab. Vert. Biol. University of Michigan, Ann Arbor, Michigan. 123 pp.

Knowles, C.J. 1995. A summary of black-tailed prairie dog abundance and distribution on the central and northern Great Plains. Report to Defenders of Wildlife. Missoula, Montana. 65 pp.

Knowles, C.J. 1998. Availability of black-tailed prairie dog habitat for black-footed ferret recovery. U.S. Fish and Wildlife Service. Unpublished report. 12 pp.

Knowles, C.J., and P.R. Knowles. 1994. A review of black-tailed prairie dog literature in relation to rangelands administered by the Custer National Forest. 78 pp.

Koford, C.B. 1958. Prairie dogs, whitefaces, and blue grama. Wildlife Monograph No. 3. 78 pp.

Kotliar, N.B., B.W. Baker, A.D. Whicker, and G. Plumb. 1999. A critical review of assumptions about the prairie dog as a keystone species. Environmental Management Vol. 24(2):177-192.

List, R., J. Pacheco, and G. Ceballos. 1997. Chapter 7, Fragmentation, populations extinction, and threats to the largest prairie dog (*Cynomys ludovicianus*) complex in North America. In Ecology of Kit Fox (*Vulpes macrotis*) and Coyote (*Canis latrans*) and the Conservation of the Prairie Dog Ecosystem in Mexico. Ph.D. Dissertation. University of Oxford, United Kingdom. 26 pp.

Miller, B., G. Ceballos, and R.P. Reading. 1994. The prairie dog and biotic diversity. Conservation Biology 8:677-681.

Miller, B., R.P. Reading, and S. Forrest. 1996. Prairie Night: Black-footed Ferrets and the Recovery of Endangered Species. Smithsonian Institution Press, Washington and London. 254 pp.

Miller, B., R.P. Reading, and S. Forrest. 1996. Prairie Night: Black-footed Ferrets and the Recovery of Endangered Species. Smithsonian Institution Press, Washington and London. 254 pp.

Mills, L.S., M.E. Soule, and D.F. Doak. 1993. The history and current status of the keystone species concept. BioScience 43:219-224.

Molina-Freaner, F. E. and T. R. Van Devender. Editors. (2010). Diversidad Biológica de Sonora. Universidad Nacional Autónoma de México. Comisión Nacional para el Conocimiento y Uso de la Biodiversidad. Mexico.

Mulhern, D., and C. Knowles. 1995. Black-tailed prairie dog status and future conservation planning. U.S. Forest Service, Rocky Mountain Forest and Range Experiment Station, General Technical Report RM-GTR-298. Fort Collins, Colorado. Pages 19-29.

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/

Olsen, P.F. 1981. Sylvatic plague. In Infectious Diseases of Wild Mammals. J. Davis, L. Karstadt, and D. Trainer, eds. Iowa State University, Ames, Iowa. Pages 232-243.

Osborn, B., and P.F. Allan. 1949. Vegetation of an abandoned prairie-dog town in tall grass prairie. Ecology 30(3):322-332.

Pacheco, J. and G. Ceballos. Perro llanero. In: Ceballos, G and G. Oliva, coords. (2005) Los mamíferos silvestres de México. Comisión Nacional para el Conocimiento y Uso de la Biodiversidad. Fondo de Cultura Económica. México. 986pp.

Paine, R.T. 1980. Food webs: linkage, interaction strength and community infrastructure. Journal of Animal Ecology 49:667-685.

Pizzimenti, J.J. 1975. Evolution of the prairie dog genus *Cynomys*. Occasional Papers of the Museum of Natural History, University of Kansas. 39:1-73.

Power, M.E., D. Tilman, J.A. Estes, B.A. Menge, W.J. Bond, L.S. Mills, G.Daily, J.C. Castilla, J. Lubchenco, and R.T. Paine. 1996. Challenges in the quest for keystones. BioScience 466:9-20.

Prairie Dog Coalition (Accessed on March, 2013) http://www.prairiedogcoalition.org/

Reading, R.P., S.R. Beissinger, J.J. Grensten, and T.W. Clark. 1989. Attributes of black-tailed prairie dog colonies in northcentral Montana, with management recommendations for the conservation of biodiversity. In Montana Bureau of Land Management Wildlife Technical Bulletin No. 2:13-23.

Roe, K. A., and C. M. Roe. 2003. Habitat selection guidelines for black-tailed prairie dog relocations. Wildlife Society Bulletin 31:1246-1253.

Schmitt, Greg. February 1995. Endangered Species Biologist. New Mexico Department of Game and Fish. Santa Fe, NM.

Shaw, J.H., W. McAbee, T.S. Carter, and D.M. Leslie, Jr. 1993. Assessment of black-tailed prairie dog colonies for reintroduction of black-footed ferrets in western Oklahoma. Proceedings of Oklahoma Academy of Science 73:47-52.

Smithsonian National Museum of Natural History (November, 2006) http://www.mnh.si.edu/mna/image_info.cfm?species_id=54

Stapp, P. 1998. A re-evaluation of the role of prairie dogs in Great Plains grasslands. Conservation Biology 12(6):1253-1259.

Terborgh, J. 1988. The big things that run the world - a sequel to E.O. Wilson. Conservation Biology 2:402-403.

Texas Parks and Wildlife (November, 2006) http://www.tpwd.state.tx.us/huntwild/wild/species/prairie/

Tileston, J.V., and R.R. Lechleitner. 1966. Some comparisons of the black-tailed and white-tailed prairie dogs in north-central Colorado. The American Midland Naturalist 75(2):292-315.

U.S. Fish and Wildlife Service. 2000. 12-Month Administrative Finding, Black-Tailed Prairie Dog http://www.fws.gov/mountain-prairie/species/mammals/btprairiedog/12month2000/chapter2.htm (Accessed on March 25, 2013)

U.S. Fish and Wildlife Service. 2004. Endangered and Threatened Wildlife and Plants; Finding for the Resubmitted Petition to List the Black-Tailed Prairie Dog as Threatened. Federal Register 69(159):51217-51226.

U.S. Fish and Wildlife Service: Mountain-Prairie Region – Endangered Species Programme (November, 2006) http://mountain-prairie.fws.gov/species/mammals/btprairiedog/

Van Pelt, W.E. 1999. The black-tailed prairie dog conservation assessment and strategy. Nongame and Endangered Wildlife Program Technical Report 159. Arizona Game and Fish Department, Phoenix, Arizona.

Wuerthner, G. 1997. Viewpoint: the black-tailed prairie dog - headed for extinction? Journal of Range Management 50:459-466.

# Reptiles

## *F-5* Desert Box Turtle *(Terrapene ornata* subspecies *luteola)*

### F-5.1 Conceptual Model

#### F-5.1.1 Classification Comments

The box turtle (*Terrapene ornata luteola,* Figure F- 23 ) is a genus of five species and 10 taxa native to North America. Box turtles are terrestrial members of the American pond turtle family (Emydidae). The eastern box turtle (*T. carolina*) with four extant subspecies (*T. c. carolina, T. c. mexicana, T. c. trianguis,* and *T. yucatana*) and a larger extinct Pleistocene subspecies (*T. c. putnami*) mostly lives in wooded habitats in the eastern United States and eastern Mexico (Crother et al., 2012). The Coahuilan box turtle (*T. coahuila*) is an endangered species endemic to the Cuatro Ciénegas Basin in Coahuila, northeastern Mexico. This species has readapted to living in water with a life style similar to mud turtles (*Kinosternon*). The spotted box turtle (*T. nelsoni*) with two subspecies (*T. n. klauberi* and *T. n. nelsoni*) is a tropical species in the Sierra Madre Occidental and tropical lowlands of western Mexico. The ornate box turtle (*T. ornata*) is a grassland animal in the central and southwestern United States and adjacent states in Mexico. The nominate subspecies (*T. o. ornata*) lives in the central United States from western Indiana, to eastern Texas and into Louisiana. The desert box turtle (*T. o. luteola*) inhabits the driest habitats of all box turtles in western Texas, New Mexico, Arizona, and Chihuahua and Sonora, Mexico. Recent genetic work indicates that the *Terrapene ornata ornata* and *Terrapene ornate luteola* lack distinction phylogenetically (Martin et al., 2013). Martin et al. (2013) recommended that they no longer be considered subspecies. However, for the purposes of this analysis we have focused on the desert box turtle subspecies as it was classified at the time this analysis was conducted.

**Figure F- 23. Desert box turtle (*Terrapene ornata luteola*), near Patagonia, Arizona.** Photos by James C. Rorabaugh.

**Species description** - The ornate box turtle is a small terrestrial turtle; males are usually smaller than females (Redder et al. 2006). The carapace varies in color from black to dark gray to reddish brown, with

yellow lines radiating from the center of every pleural scute. Central scutes have yellow dashes and form a discontinuous mid-dorsal line. In the desert box turtle, the colors of the mature turtles are lighter and more muted than the colors of the young. The yellow radiating stripes on the shell of males disappear with age, but are generally retained in females. The carapace has an oval outline and a domed and dorsally flattened shape resembling a box, with little or no mid-dorsal keel. The plastron is hinged between the hyoplastral and hypoplastral bones (at the joint between the pectoral and abdominal scutes; Dodd 2001, Stebbins 2003), and it can be completely closed against the carapace, allowing box turtles to completely withdraw their head and feet. All plastral scutes have yellow streaks. The skin is brown with yellow spots.

The desert box turtle has 10-16 yellow streaks on pleural scutes as opposed to 5-9 in ornate box turtle (Ward 1978). The carapace ground color is lighter in desert box turtles, and fades to a uniformly straw- or horn-colored in older animals. The male has a bluish head and red or orange eyes, while females have brown or yellow irises. The shell is up to 5.8 inches (146 mm) long (Brennan and Holycross 2006). There are usually four toes on each hind foot. Males have an enlarged, recurved inner claw on the hind feet that is used during copulation. Reproductive-age males have a concave plastron (females are flat or convex), a red iris (females are typically yellow), a more posterior cloacal opening compared to females, a longer and thicker tail, and red coloration on the legs and sometimes on the jaw. Young desert box turtles have the vertebral stripe more pronounced than the yellow scute streaks. The carapace after hatching is nearly flat and circular (Legler 1960). Rings on the scute margins appear to be added with each growing season, allowing for reasonable age estimation up to about age 14 (St. Clair 1998).

### F-5.1.2 Protection Status

Box turtle populations throughout the United States are thought to be in decline due to human-influenced landscape changes. Because of this concern, the Arizona Game and Fish Department's Turtles Project is initiating monitoring efforts in desert box turtle habitat to determine baseline population levels and to develop management plans for the species. In 2009 the Turtles Project launched the citizen scientist based Ornate Box Turtle Watch program where members of the public are encouraged to report box turtle observations in Arizona (www.azgfd.gov/boxtrutlewatch). The desert box turtle has rarely been seen in Sonora in the last 20 years; a study of its status is urgently needed. *Terrapene ornata* is included in CITES Appendix II, which are species that are not necessarily now threatened with extinction but that may become so unless international trade in them is closely controlled. This turtle is protected under a variety of US laws and regulations (review by Dodd 2001). Both *T. ornata* and *T. nelsoni* have *Protegida* status under the Mexican endangered species law, Norma Oficial Mexicana, NOM-059-SEMARNAT-2010 (Diario Oficial de la Federación 2010). In January 2005, the Arizona Game and Fish Department closed the collecting season on desert box turtles in Arizona (Arizona Commission Order 43). Possession of ornate box turtles, *Terrapene ornata*, is prohibited, except for those legally held prior to season closure. The desert box turtle is designated as a Priority Species for Pima County, Arizona in the Sonoran Desert Conservation Plan (Pima County 2013).

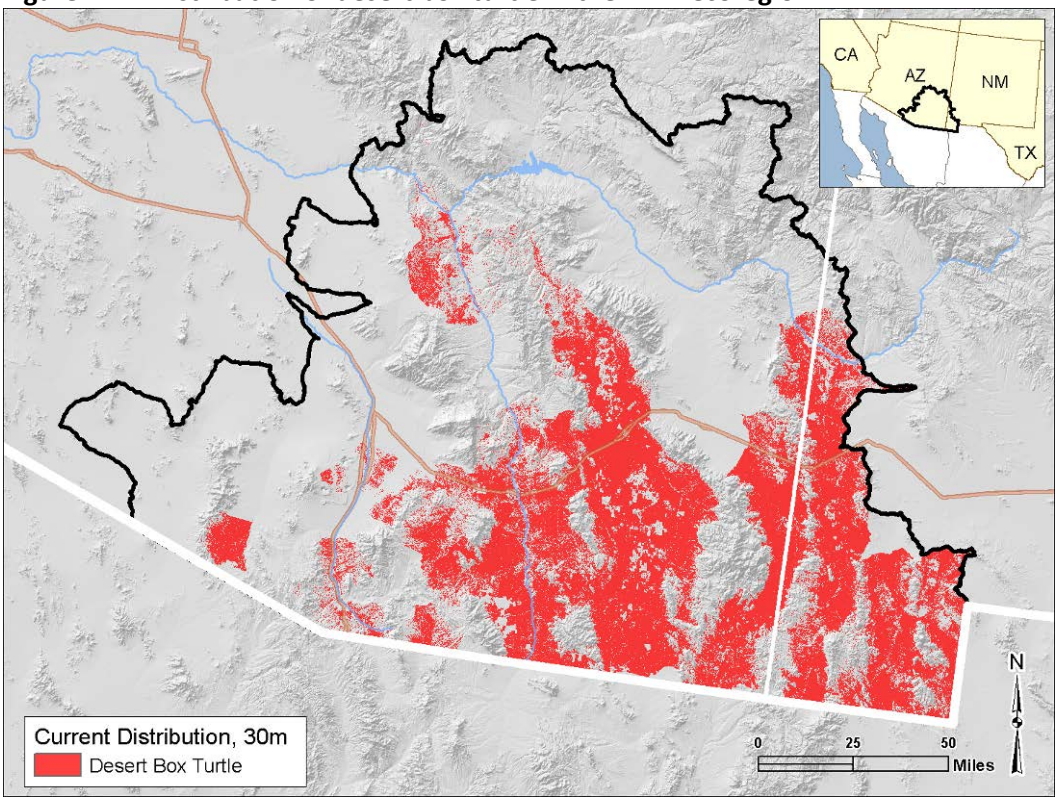
### F-5.1.3 Biology and Distribution Status

#### *F.5.1.3.1 Distribution*

*Terrapene ornata luteola* ranges throughout southeastern Arizona, much of New Mexico, southwestern Texas, and south into northeastern Sonora and much of Chihuahua, Mexico (Stebbins 2003, van Dijk and Hammerson 2011). In Arizona, this turtle is found in the southeastern counties of Cochise, Graham, Pima, and Santa Cruz counties (Brennan 2013, Brennan and Holycross 2006) and Greenlee and Gila (AZGFD, HDMS, 2013). The westernmost desert box turtle populations are in Las Ciénegas National Conservation Area, in the Santa Cruz River near Sahuarita, and Altar Valley in Pima County (Pima County

2013). The southernmost records for desert box turtle in Sonora are near Arizpe in the Río Sonora Valley (30°20'N) and at Nacozari de García (30°19'N), 110 to 115 km south of the Arizona border.

**Figure F-24. Distribution of desert box turtle in the MAR ecoregion.**




### *F.5.1.3.2 Reproduction*

Male desert box turtles can reproduce after they reach 7-8 years of age, and females at 8-9 years (Dawson 1998, Brennan 2013, Brennan and Holycross 2006). Box turtle breeding peaks in the spring and fall but may mate anytime during their active season (AZGFD 2008). Female box turtles can retain the sperm after mating for several years. After mating, females lay an average clutch of 2-3 eggs (up to 8 maximum) during the monsoon season, and incubation lasts about seventy days. Hatchlings are dark in color with a yellow stripe down the center of the shell. They can live 25 years or more.

## F-5.1.4 Habitat and Ecology

### *F.5.1.4.1 Habitat*

In Arizona, desert box turtles mainly inhabit desert grassland, but can also be found in Chihuahuan desertscrub, Madrean evergreen woodland, and Sonoran desertscrub up to 7100 feet (2,165m) in elevation. They require loose soil for burrowing (Hall and Steidel 2003). The southernmost populations near Arizpe are in foothills thornscrub or riparian habitats along the Río Sonora or in oak woodland/grassland at Nacozari de García.

***Elevation:*** In Arizona, the desert box turtle mostly occurs at 3000-6500 ft (915-2165 m) elevation, but has been found at 2000 ft (610 m) and 7100 ft (2165 m; Brennan 2013, Brennan and Holycross 2006). In Sonora, it has been found at 2526-5110 ft (868-1558 m) elevation.

#### *F.5.1.4.2 Food*

Brennan (2013) reports that desert box turtle "feeds on beetles, crickets, grasshoppers, other insects, worms, crayfish, reptiles, eggs, carrion, cactus fruit, cactus pads, berries, and other plant material." They have been seen in shallow temporary ponds in the summer rainy season feeding on tadpoles and aquatic insects. The box turtle often searches under cow dung for beetles and other insects (Legler 1960, Ernst et al. 1994, Stebbins 2003).

#### *F.5.1.4.3 Phenology*

In southeastern Arizona, the desert box turtle takes refuge in subterranean mammal burrows, primarily those of the Banner-tailed kangaroo rat (*Dipodomys spectabilis*), from November through June (Plummer 2004). In New Mexico, the box turtle is associated with black-tailed prairie dog (*Cynomys ludovicianus*) towns (New Mexico Game and Fish Department 1997). Plummer (2004) found that in the fall, entrance into hibernation varied among individual turtles and between years from 18 October to 6 December. Mean daily body temperatures at the beginning of hibernation were about 8°C. From December through February, body temperatures averaged about 9°C and were slightly less than and highly correlated with the 30-cm soil temperature. By April, body temperatures of turtles approached levels exhibited by surface-active turtles (minimum approximately 18°C for moving turtles) but turtles did not emerge from burrows until the summer monsoons began about three months later. Thus, the five-month hibernation period of *T. o. luteola* was contiguous with a three-month estivation period for a total annual period of subterranean refuge of about eight months.

### F-5.1.5 Change Agent Characteristics

#### *F.5.1.5.1 Landscape Modification and Livestock Management*

Landscape level changes in desert grassland, such as large-scale urbanization and conversion of grassland to agricultural fields, have major impacts on desert box turtles populations. Threats may also include habitat loss due to alterations to plant community (AZGFD, 2008). Grazing does not have a significant impact on the desert box turtle, heavier grazing resulting in more open areas likely favors kangaroo rats and more preferred hibernation sites.

Historically, the desert box turtle was reasonably well distributed in desert grassland in southeastern Arizona. Desert box turtle's are often seen on roads, especially during the monsoon season. Hall and Steidl (2003) found that box turtle abundance was lower in high traffic areas, and that mortality was higher on paved roads. In the Madrean Archipelago Ecoregion there has been an increase in vehicle traffic related to border enforcement activities. The amount and speed of vehicle traffic has increased. Roads have been upgraded, experience more frequent vehicular traffic and in some cases are drug daily to clear the road surface to record fresh tracks.

**Stresses:** habitat loss due to urbanization or agriculture; animals killed on roads by vehicles.

**Responses:** loss of populations; population health and integrity of gene pool damaged by removal of ads or removed as pets.

#### *F.5.1.5.2 Collection of Wild Animals*

Substantial numbers of box turtles have been collected in the past for the domestic and international pet trade, with potentially significant population impacts (van Dijk and Hammerson 2011). Box turtles of several species and varieties are commonly kept as pets in the United States and offered for sale online (The Turtle Shack 2013, The Turtle Source 2013).

Although collection of desert box turtles is prohibited by the Arizona Game and Fish Department, individuals encountered crossing roads in suburban and rural areas in southeastern Arizona may be taken home as pets. In Willcox, Arizona there is an annual box turtle race for which people likely collect wild turtles from local roads. Although the eastern ornate box turtle is commonly sold as pets for as much as $250 apiece (the Turtle Shack 2013), the desert box turtle is not offered for sale online.

**Stresses:** increased human presence, animals removed from populations for pets.

**Responses:** loss of populations; population health and integrity of gene pool damaged by removal of adults killed on roads or removed as pets.

#### *F.5.1.5.3 Climate Change*

Climate change is an ongoing process in the Southwest with potential effects on all species and biotic associations (Thompson et al. 1997). Mean annual temperatures rose by 2.0-3.1°F in the American Southwest in the 20$^{th}$ century, and are predicted to rise between 8.1 and 11.0°F in the 21st century (Southwest Regional Assessment Group 2000). Changes in precipitation are more speculative, with contradicting predictions in different models (Southwest Regional Assessment Group 2000, Patterson 1997, Betancourt 2004).

Contradictory predictions about future precipitation regimes make evaluating threats to the desert box turtle due to precipitation difficult. Because desert box turtles' sex is determined by the temperature at which it was it was incubated in the egg, temperature increases may have an affect on populations of this species. Eggs incubated at cooler temperatures produce more males and at warmer temperatures produce more females (Degenhardt et al., 1996). Addtionally, desert box turtles metabolism response to warming temperatures is of concern because as a terrestrial ectotherm, thier metabolism will be affected by ambient temperature (Zug et al. 2001.) Studies of other box turtle subspecies (*Terrapene Carolina triunguis)* indicated that climate change may influence body growth and fecundity causing fewer turtle hatchlings to show positive growth during their first year (McCallum et al., 2009).

Changes in average annual temperatures may cause turtles to enter into hibernation later in the fall, shortening the hibernation-estivation inactivity period, or to come out of hibernation later in the spring. Depending on how this coincides with precipitation changes this may have an effect on turtle survival.

The grassland communities that desert box turtles inhabit may be affected by changes in winter rainfall and fire which could in turn affect the distribution of these communities. In desert grassland, shrub dominance has increased as perennial grass and herbaceous perennial dominance declined twice in the last 4,000 years during warm periods before European arrival (Van Devender 1995). Desert box turtle would likely not be impacted significantly by shrub increases in desert grassland. Moodie and Van Devender (1978) reported fossil box turtles from Cochise County, Arizona, including a Pliocene *Terrapene* cf. *ornata* in Pliocene (ca. 4.0 mya [million years ago]) sediments and *T. ornata* in late Pleistocene (ca. 0.5-1.0 mya) deposits. Considering that desert box turtle has survived in the Madrean Archipelago region during frequent climate and vegetation changes for the last two million years, it would likely not be impacted significantly by shrub increases in desert grassland. The foothills thornscrub-desert grassland ecotone may shift northward in Sonora, but the desert box turtle already lives in this tropical vegetation type near Arizpe.

**Stresses:** Increasing annual temperatures; decreased summer precipitation; prime grass-dominated desert grassland becomes less open as shrubs increase.

**Responses:** Changes in sex ratios within populations, changes in hibernation-estivation activity; increase or decrease of summer precipitation will affect reproduction, food resources, survivorship, and dispersal/gene flow between metapopulations.

### F-5.1.6 Ecological Status: Key Ecological Attributes and Indicators

This section of the conceptual model addresses Key Ecological Attributes and their potential indicators. The ecological status is a way of describing current status via criteria, functionality, or levels of attributes and asks if they are within the normal range of variation.

#### *F.5.1.6.1 Key Ecological Attributes*

Table F-9 identifies the key ecological attributes for this CE within the Madrean Archipelago ecoregion. A **key ecological attribute** of a focal ecological resource is a characteristic of the resource's biology, ecology, or physical environment that is critical to the resource's persistence in the face of both natural and human-caused disturbance. Alteration of such a characteristic beyond some critical range of variation will lead to the degradation or loss of the resource within decades or less. The KEAs table lists the identified key ecological attributes, with a brief definition, a rationale for why it is important for the CE, and a listing of stressors or change agents that might be affecting the key attribute.

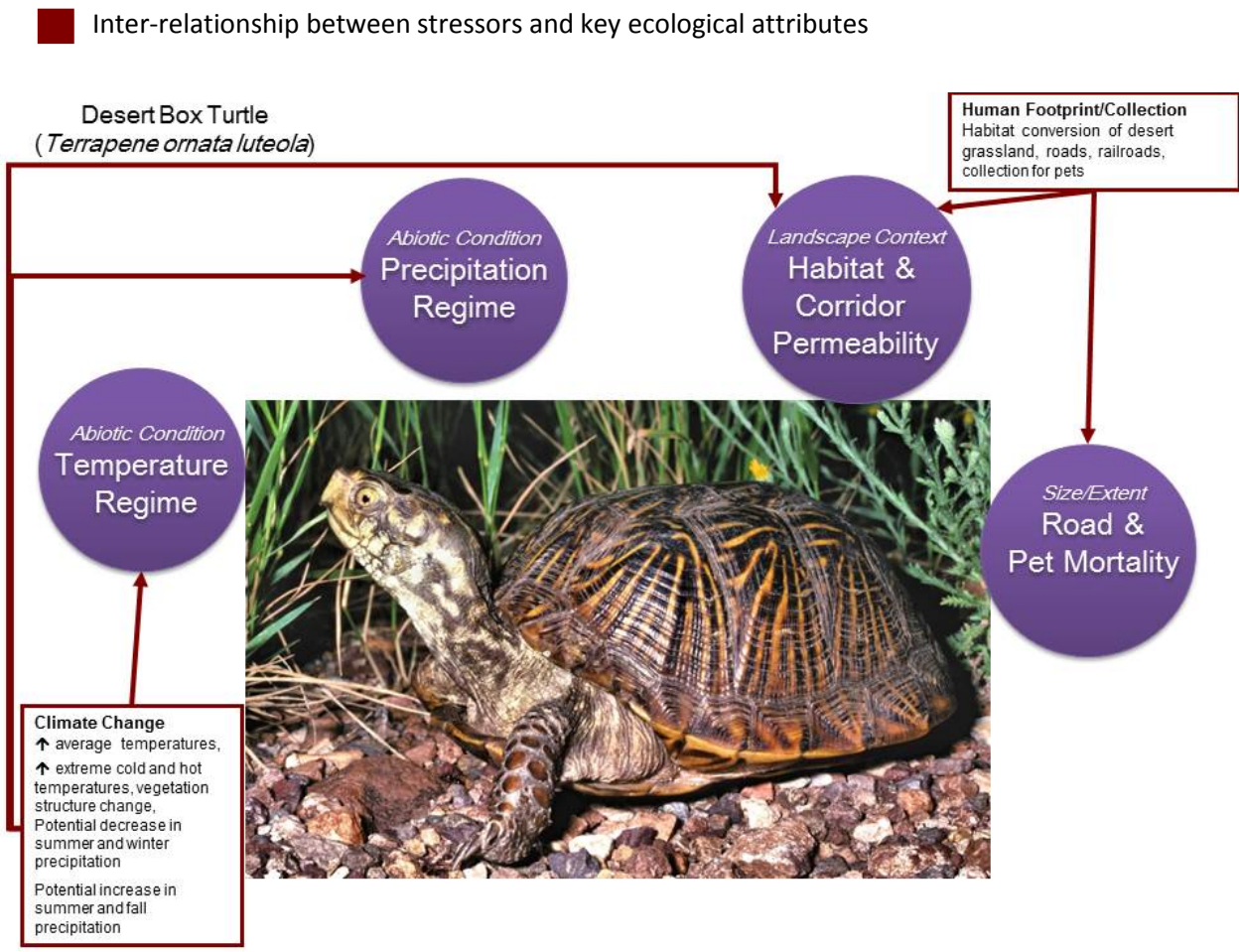
**Table F-9. Key ecological attributes (KEA) of the Desert Box Turtle (*Terrapene ornata luteola*) in the Madrean Archipelago ecoregion.** Indicators for these KEAs can be used to determine the ecological status for this CE; see **Table F-2** for a list of the indicators assessed in this REA.

| ***KEA Class:*** **KEA Name** | **Definition** | **Rationale** | **Stressors** |
|---|---|---|---|
| ***Landscape Context:*** **Habitat Availability and Quality** | The extent of desert grassland habitat that is unfragmented by development and/or roads. | Ecological conditions and landscape dynamics are affected by land use. Land use impacts vary in their intensity where they occur, as well as their ecological effects with distance. | Factors that affect landscape condition in desert box turtle habitat can be abiotic (drought, fire) or related to human activities. Landscape level changes in desert grassland, such as large-scale urbanization and conversion of grassland to agricultural fields, have major impacts on desert box turtles populations. Turtles may suffer direct mortality crossing roads. |
| ***Population Condition:*** **Population Size & Demographics** | The size and age classes of the turtle population. | Mortality and survivorship are opposing processes in population dynamics. Human related mortality stresses natural populations. | Animals killed on roads by vehicles; animals removed for pets. Development of roads fragmented desert box turtle habitats and gene pools. Changes in frequency and speed of vehicular traffic will also affect turtle mortality. Turtles are often seen on roads, especially during the monsoon season (Brennan and Holycross 2006). |
| ***Abotic Condition:*** **Precipitation regime** | The mean annual and seasonal precipitation and fluctuations in them. | Turtles come out of burrows when the summer monsoons begin and turtle mating typically takes place during the monsoon season. Precipitation changes would also effect the availability of the turtle's food sources. | Although there are contradictory predictions about future precipitation regimes, there is agreement on a trend of increased aritdy. Changes in timing of precipitation, amount and frequency of precipitation events. |
| ***Abiotic Condition:*** **Temperature regime** | This mean annual and seasonal temperature and periodic extreme events. | Seasonal temperatures and periodic extreme events control the distributions of most animals and plants. Seasonal temperature changes dictate when turtles enter into hibernation in the fall and ambient temperature during egg incubation affects sex determination. | With increased annual temperatures, desert box turtle would likely enter into hibernation later in the fall, shortening the hamibernation-estivation inactivity period. |

### F-5.1.7 Conceptual Model Diagram

A conceptual model diagram for the CE provides a visual summary representation of the Key Ecological Attributes and species' life cycle (Figure F- 25). These diagrams are intended to show how various stressors interact with categories of Key Ecological Attributes to highlight important drivers for species ecological integrity. The arrows indicate relationships between stressors and but do not indicate the nature of the influence (i.e. positive or negative).

**Figure F- 25. Conceptual model diagram for the desert box turtle, showing key ecological attributes** (by class) for this species, and indicating relationships between stressors and KEAs.




## F-5.2 Considerations and Limitations

To date there has been little research on the status of desert box turtles in the MAR but they are thought to be in decline due to human-influenced landscape changes. Death due to collisions with cars on roads is noted above as an important influence on the species. All road types were assigned the same site intensity scores across all species, but this may be a limitation of the box turtle ecological status

results as road infrastructure may be having more of an influence on this species survival and gene flow. There was not adequate information to incorporate this into the assessment in a more species-specific way but this remains an important consideration for management.

## F-5.3 Ecological Status Assessment Results and Interpretation

This section of the CE summary presents and discusses the results of the ecological status assessment for desert box turtle. It addresses each indicator separately, and then addresses the overall assessment, which integrates the results of all individual indicators assessed for the CE. The results are presented using a common framework, in which the status of an indicator – or the combination of all indicators – is scored on a scale from 0.0 to 1.0, where 0.0 indicates a condition of complete replacement of reference ecological conditions due to the impacts of stressors, and 1.0 indicates a condition of no alteration of reference ecological conditions. The same yellow-to-blue color ramp is used for all results, with yellow representing low scores, green moderate scores, and dark blues high scores.

### F-5.3.1 Current Ecological Status: Development, Fire Regime, Invasives

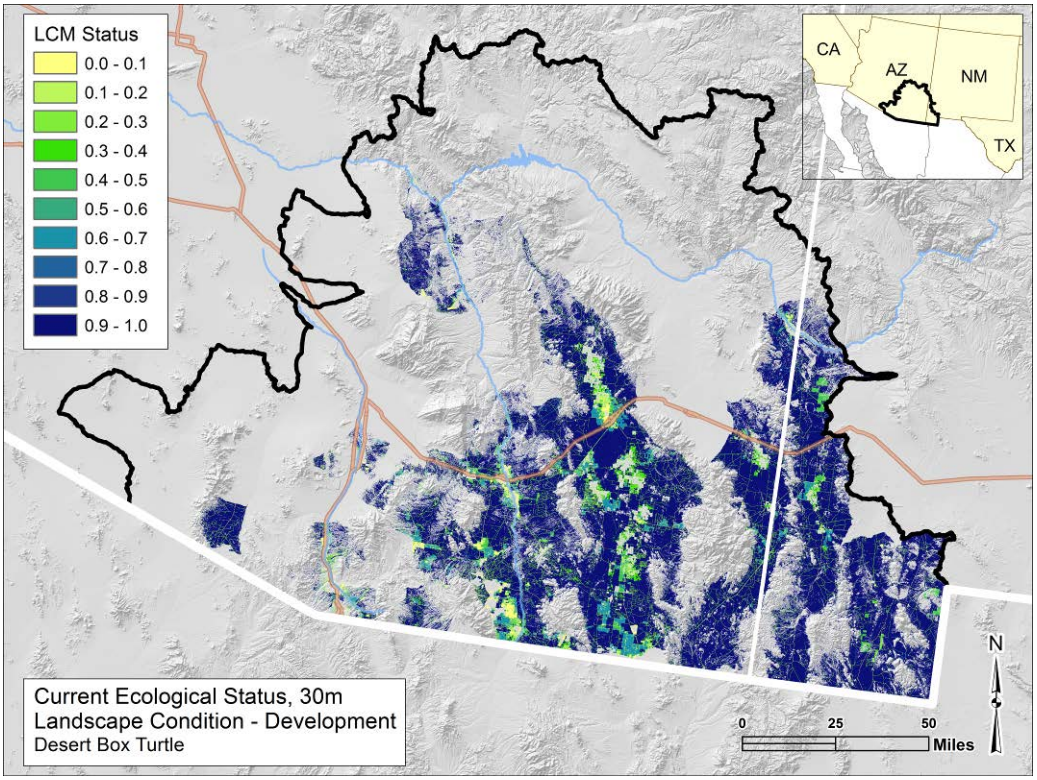
The results of the assessments for the three individual indicators for the KEAs for desert box turtle are shown in the three maps in Figure F-26 below.

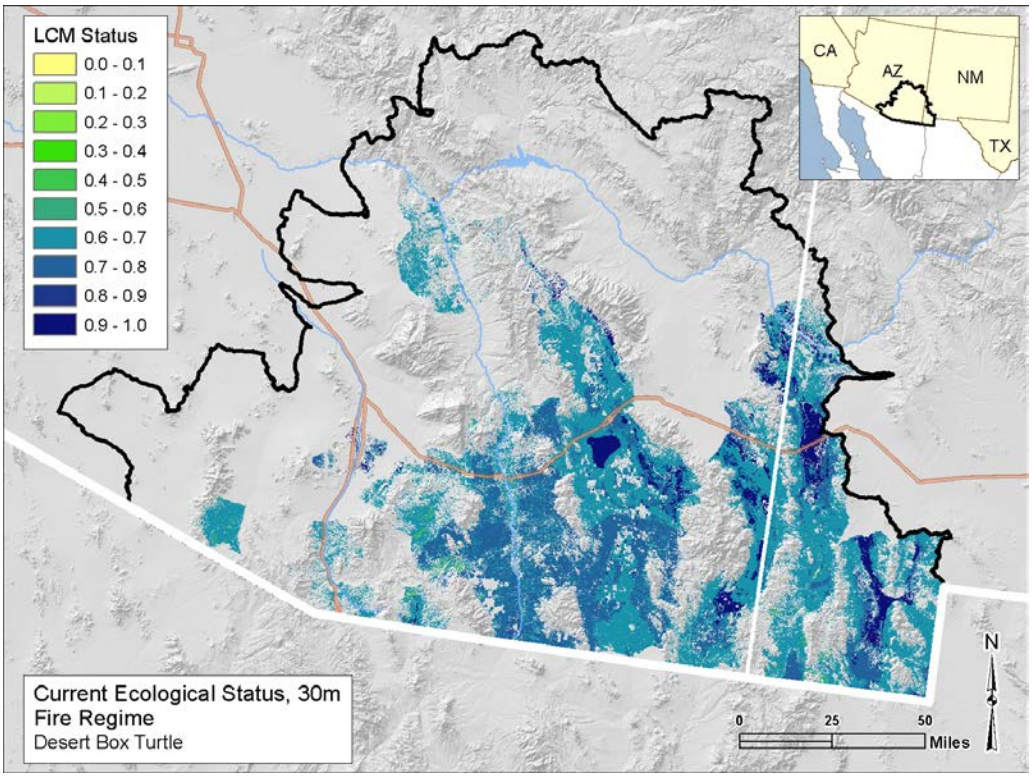
The development indicator results shown in the first map of Figure F-26 reflect a number of areas of development (yellows and greens on the map) across the turtle's current distribution. These areas of lower development scores reflect a range of development features, including municipalities, agriculture, transportation and other features. However, a fair amount of the turtle's distribution in the MAR ecoregion is experiencing little or no impact (dark blue) from the development features reflected in the landscape condition indicator. Where development is present, the primary impacts to turtles include direct loss of grassland habitat and increased disruption of activities due to increased transportation infrastructure and increased traffic. It is worth noting that turtles are likely to be more sensitive to smaller roads than other larger species and that although they may be impacting a relatively small area of the turtle's distribution, if they are occurring in areas critical to movement and gene flow for turtle populations, they may be of higher concern.

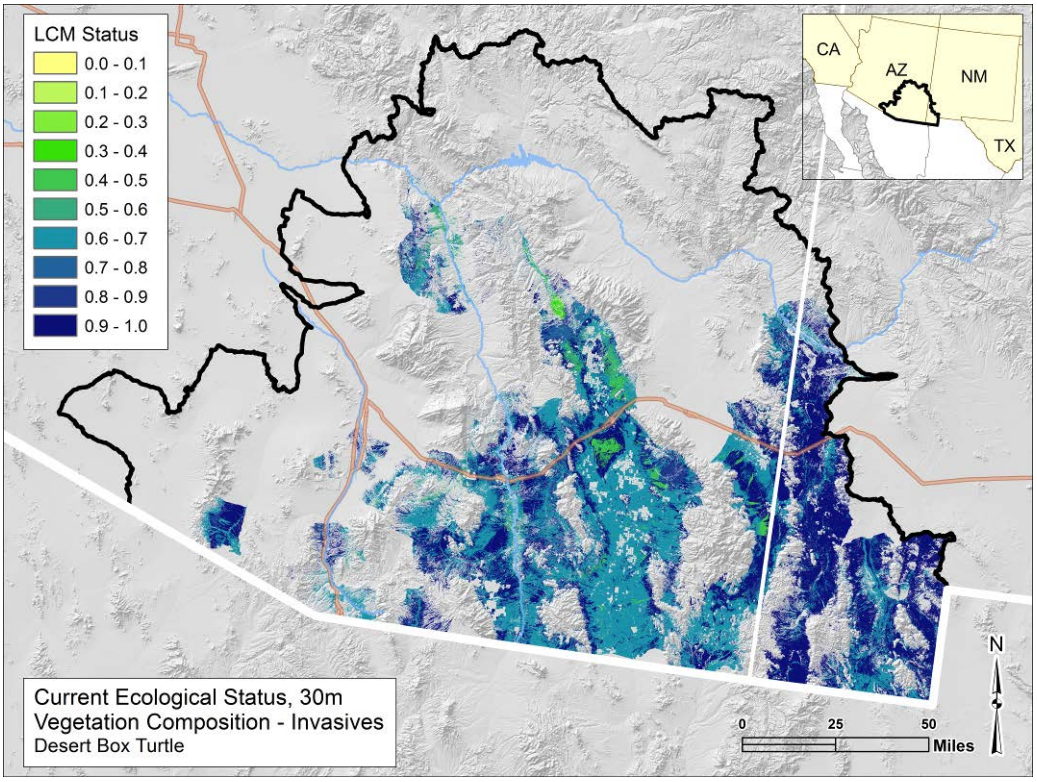
Fire regime results, shown in the second map in Figure F-26, indicate that fire regimes in the ecological systems present in the turtle's distribution are largely split between severe ecological departure (scores of 0.65, shown in turquoise blue) and moderate ecological departure (scores of 0.75); very little of its distribution shows no departure in fire regimes. To the degree that fire is maintaining habitat supporting this species (e.g., semi-desert grasslands, Madrean woodlands), the altered fire regime is of potential concern for this assemblage. However, there is not enough information to conclusively state whether the effects of altered fire have had any effects on this species.

The invasives results (third map in Figure F-26) show that a fair amount of the turtle's distribution is impacted either by mesquite or by non-native grasses and forbs (turquoise blue color). In a few areas (greens), both groups of invasives are present at high cover. Based on the compilation of the ILAP data, for this CE's distribution, most of the area showing invasive impacts has high cover of mesquite (rather than non-native grasses or forbs). Where grasslands are experiencing significant mesquite encroachment or conversion to mesquite shrubland, there could be potential for loss of turtle habitat. However, there is not enough information to confirm whether mesquite or other invasives have affected turtle habitat quality.

**Figure F-26. Scores for three indicators for desert box turtle: development indicator (1st map), fire regime departure indicator (2nd map), and invasive species indicator (3rd map) for each 30m pixel.** Yellow (equivalent to 0) indicates high impacts from the CA, dark blue (equivalent to 1) indicates little to no impact from the CA. At the ecoregion scale, many development features are not readily visible (e.g., secondary roads or highways, railroads, small agricultural fields). Only 3 classes of fire regime condition are scored: no to little departure (dark blue), moderate departure and severe departure (lighter blues). For invasives, higher cover of mesquite or invasive exotics have scores between .4 and .6 (light greens), while lower cover has scores between .6 and .8 (light blues).





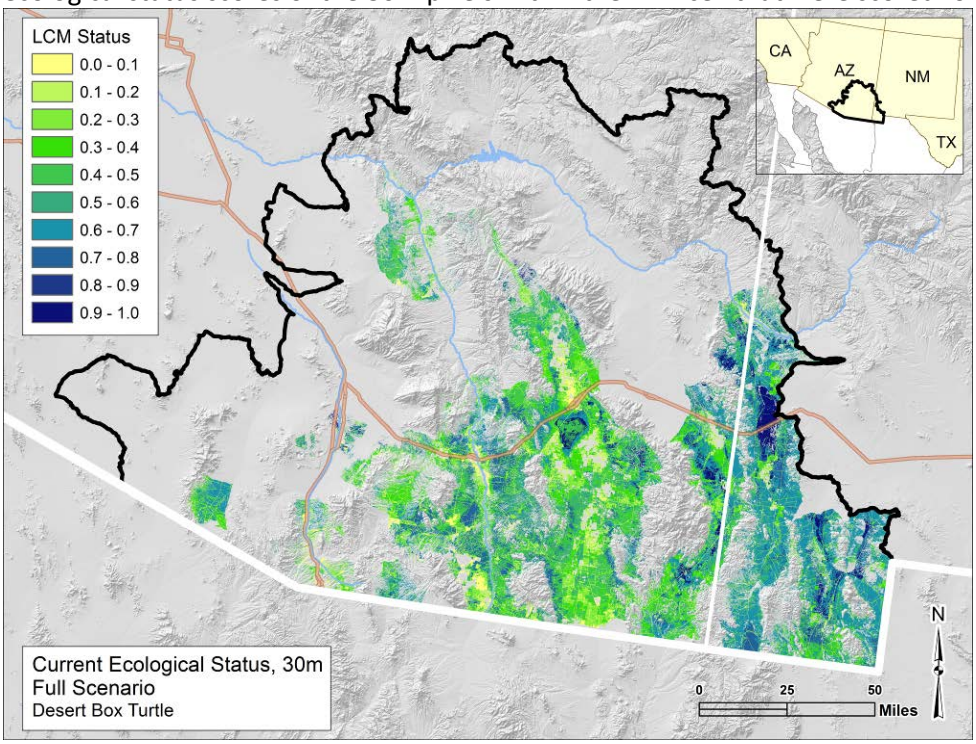

### F-5.3.2 Current Ecological Status: Full Scenario

The results of the three individual status indicators were combined to get an overall ecological status score, per pixel of the CE's distribution, as shown in the first map of Figure F-27 below. The overall, per-

pixel status scores for the CE were then averaged across each of the 4 km grid cells, as shown in the second map of Figure F-27 below.

When the three sets of indicator scores are aggregated to obtain the overall ecological status scores (as shown in the first map of Figure F-27), much of the habitat for this species is in moderate condition, with status scores around 0.4, 0.5 and 0.6 in many areas, and some discrete areas as low as the 0.1 range (Figure F-28). In visually comparing the maps of scores for the three individual indicators (Figure F-26) to the map of the overall status scores (first map of Figure F-27), the cumulative impacts of the change agents reflected by the indicators are clear. The altered fire regime and mesquite cover are both driving down scores in much of this CE's distribution, and development is a contributing factor in a number of areas as well. These same patterns are reflected when the overall status scores are averaged across each 4 km reporting unit, as shown in the second map of Figure F-27 above. Based on the indicators assessed in this REA, altered fire regime and encroachment by native woody increasers are the most widespread change agents in desert box turtle habitat. Aside from direct habitat loss due to development, the effect of these habitat changes on the desert box turtle is unclear.

**Figure F-27. Overall ecological status scores for desert box turtle for all three indicators combined (development, fire regime and invasives) for each 30m pixel (top) and 4km grid cells (bottom).** Yellow scores (equivalent to 0) indicate high impacts from the CAs, dark blue (equivalent to 1) indicate little to no impact from the CAs. In the second map, the score for each 4km cell is an average of the overall ecological status scores of the 30m pixels within the 4km cell that were scored for the CE.



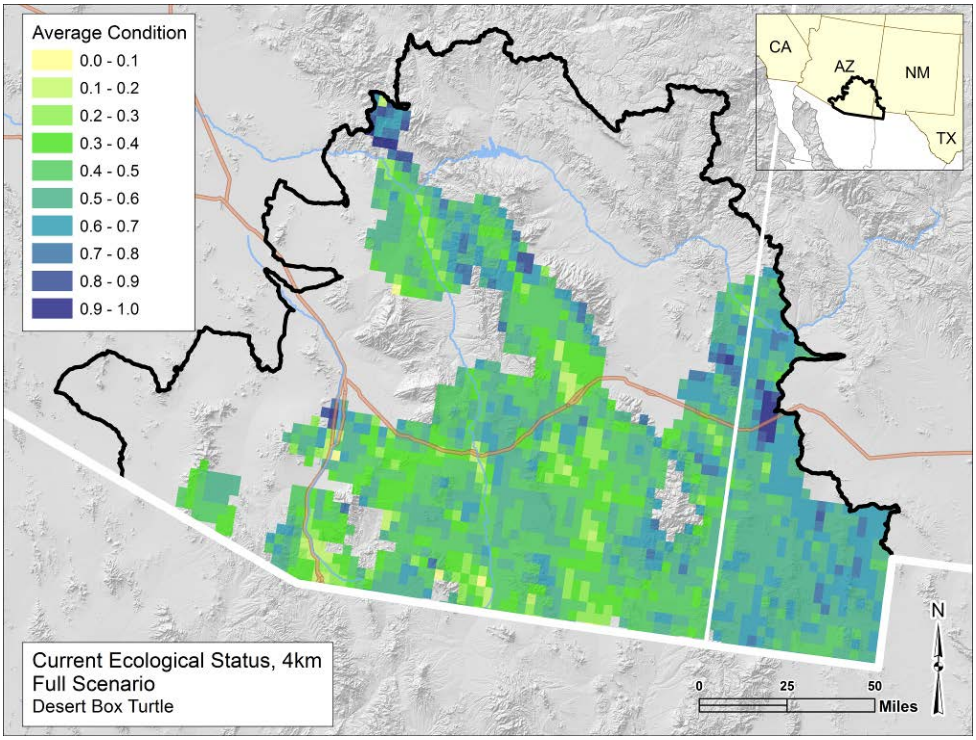
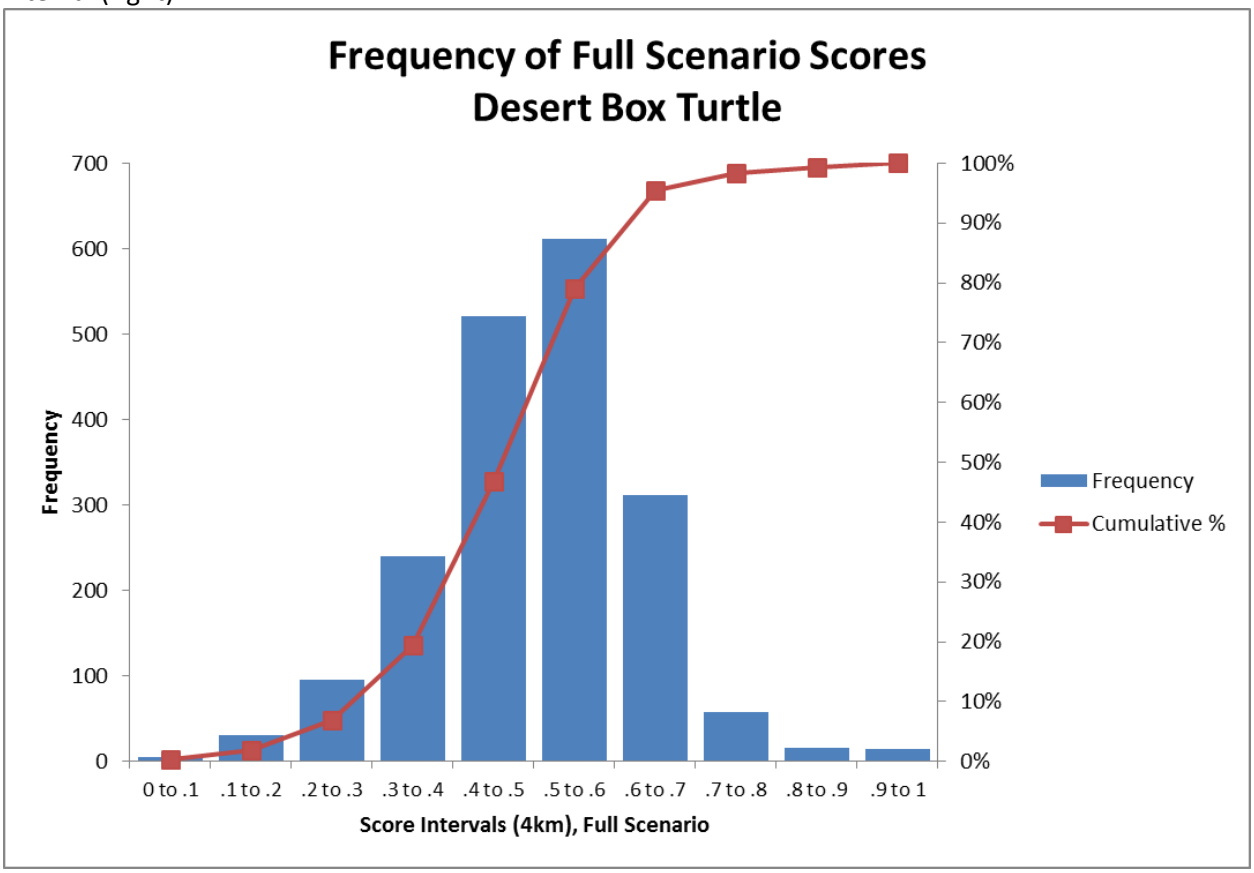



**Figure F-28. Frequency distribution of the 4km ecological status scores for desert box turtle, with cumulative percent.** The x-axis represents the 0.1 increment scoring intervals, while the y-axis shows the number of grid cells in each interval (left) and the cumulative percentage of the grid cells for each interval (right).




## F-5.4 References for the CE


Arizona Game and Fish Department. 2008. Terrapene ornate. Unpublished abstract compiled and edited by the Heritage Data Management System, Arizona Game and Fish Department, Phoenix, AZ. 4 pp.

Betancourt, J. L. 2004. Ecological responses to climate variability in time and space: a Southwestern synopsis. Page 19 *in* Program and Abstracts, Connecting Mountain Islands and Desert Seas, Biodiversity and Management of the Madrean Archipelago II and 5th Conference on Research and Resource Management in the Southwestern Deserts, Tucson, Arizona.

Brennan, T. C. 2013. Reptiles of Arizona. http://www.reptilesofaz.org/Turtle-Amphibs-Subpages/h-t-ornata.html, accessed 17 June 2013

Brennan, T. C., and A. T. Holycross. 2006. *A Field Guide to Amphibians and Reptiles in Arizona.* Arizona Game and Fish Department. Phoenix, AZ

Comer, P., and J. Hak. 2012. Landscape condition model of the western United States. NatureServe, Boulder, CO. In preparation.

Crother, B. I., J. Boundy, F. T. Burbrink, J. A. Campbell, K. Queiroz, D. R. Frost, D. M. Green, R. Highton, J. Iverson, F. Kraus, R. W. McDiarmid, J. R. Mendelson III, P. A. Meylan, R. A. Pyron, T. W. Reeder, M. E. Seidel, S. G. Tilley, D. B. Wake.2012. Scientific and Standard English Names of Amphibians and Reptiles of North America North of Mexico, with Comments Regarding Confidence in Our Understanding, Seventh Edition. Committee on Standard English and Scientific Names.

Dawson, J. 1998. Western Box Turtle. http://www.geocities.com/Heartland/Plains/3550/wbox01.html

Degenhardt, W.G., C.W. Painter and A.H. Price. 1996. Amphibians and Reptiles of New Mexico. University of New Mexico Press, Albuquerque, New Mexico.

Diario Oficial de la Federación. 2010. Norma oficial Mexicana, NOM-059-SEMARNAT-2010, Protección Ambiental-Especies nativas de México de flora y fauna silvestres-Categorías de riesgo y especificaciones para su inclusión, exclusión o cambio-Lista de especies en riesgo. Emitido por la Secretaría del Medio Ambiente y Recursos Naturales. 06 de marzo del 2002. México D.F., p. 80.

Dodd, C.K., Jr. 2001. North American Box Turtles. A Natural History. University of Oklahoma Press, Norman, OK.

Ernst, C. H., J. E. Lovich, and R.W. Barbour. 1994. Turtles of the United States and Canada. Smithsonian Institution Press, Washington, D.C.

Hall, D. H., and R. J. Steidl. 2003. Ecology and Conservation of Desert Box Turtles (*Terrapene ornata luteola*) in Arizona. Arizona Game and Fish Department, Phoeniz, AZ.

Legler, J. M. 1960. Natural history of the ornate box turtle, *Terrapene ornata ornata* Agassiz. University of Kansas Publication of the Museum of Natural History 11:527-669.

Martin B.T., N.P. Berstein, R.D. Birkhead, J. F. Koukl, S. M. Mussmann, J. S. Jr. Placyk. 2013. Sequence-based molecular phylogenetics and phylogeography of the American box turtles (Terrapene spp.) with support from DNA barcoding. Molecular Phylogenetics and Evolution 68: 119-134

McCallum, M. L., J. L. McCallum, and S. E. Trauth. 2009. Predicted climate change may spark box turtle declines. Amphibia-Reptilia 30: 259-264

Moodie, K. B., and T. R. Van Devender. 1978. Fossil box turtles (genus *Terrapene*) from southern Arizona. Herpetologica 34:172-174.

New Mexico Game and Fish Department. 1997. BISON-M Taxonomy. http://www.fw.vt.edu/fishex/nmex_main/species/030415.htm

Patterson, M. W. 1997. Forest fires and drought in the U.S. Southwest. 1997. *In* Impacts of Climate Change on Life and Ecosystems. http://geochange.er.usgs.gov/sw/impacts/biology/fires_drought/

Pima County. 2013. Sonoran Desert Conservation Plan. Multi-Species Conservation Plan. Pima County's Multi-species Conservation Plan: Public Draft. http://www.pima.gov/CMO/SDCP/MSCP/MSCP.html, http://www.pima.gov/CMO/SDCP/species/fsheets/vuln/vuln.html

Plummer, M. P. 2004. Seasonal inactivity of the Desert Box Turtle, *Terrapene ornata luteola* at the species' southwestern range limit in Arizona. Journal of Herpetology, 38: 589–593, 2004

Redder, A. J., C. K. Dodd, Jr., D. Keinath, D. Mcdonald, and Takeshi Ise. 2006. Summary of key components for conservation of the Ornate Box Turtle. Prepared for the USDA Forest Service, Rocky Mountain Region, Species Conservation Project *http://www.fs.fed.us/r2/projects/scp/assessments/ornateboxturtle.pdf*

Southwest Regional Assessment Group. 2000. Preparing for a climate change; the potential for climate variability and change. Report to the U.S. Global Change Research Program. http://www.ispe.arizona.edu/research/swassess/pdf/complete.pdf.

Stebbins, R. C. 2003. A Field Guide to Western Reptiles and Amphibians, Third Edition. Houghton Mifflin Company, Boston, MA.

St. Clair, R.C. 1998. Patterns of growth and sexual size dimorphism in two species of box turtles with environmental sex determination. Oecologica 115:501-507.

The Turtle Shack. 2013. http://turtleshack.com/store/index.php?main_page=advanced_search_result&keyword=ornate+box&categories_id=&inc_subcat=1&manufacturers_id=&pfrom=&pto=&dfrom=&dto=&x=18&y=4&gclid=CL7Djdvq97cCFfBDMgodU2YAgw

The Turtle Source. 2013. http://www.theturtlesource.com/i.asp?id=100200314

Thompson, R. S., K. H. Anderson, and P. J. Bartlein. 1997. Assessment of potential future vegetation changes in the Southwestern United States. *In* Impacts of Climate Change on Life and Ecosystems. http://geochange.er.usgs.gov/sw/impacts/biology/veg_chg_model/

Van Devender, T. R. 1995. Desert Grassland History: Changing climates, evolution, biogeography, and community dynamics. Pp. 68-99 *in* M. P. McClaran and T. R. Van Devender (eds.), The Desert Grassland, University of Arizona Press, Tucson.

van Dijk, P. P., and G. A. Hammerson. 2011. *Terrapene ornata*. In: IUCN 2012. IUCN Red List of Threatened Species. Version 2012.2. <www.iucnredlist.org>. Downloaded on 14 June 2013.

Ward, J. P. 1978. *Terrapene ornata.* Catalogue of American Amphibians and Reptiles No. 217:1-4.

Zug, H.R., Vitt, L.J., Caldwell, J.P. (2001): Herpetology: An introductory biology of Amphibians and Reptiles, 2nd Edition. San Diego, Academic Press

# Amphibians

## F-6 Chiricahua Leopard Frog *(Lithobates chiricahuensis)*

### F-6.1 Conceptual Model

#### F-6.1.1 Classification Comments

Previously the North American leopard frogs (Figure F-29) were classified as *Rana pipens* with subspecies in major geographic areas. Later the subspecies were elevated to full species in the *Rana pipiens* Complex based on morphological and molecular analyses, mating calls, color patterns, and distributions. Additional species of leopard frogs were described subsequently, including *R. blairi* (Plains leopard frog) from the mid-United States west to New Mexico and Arizona (Mecham et al. 1973, Crawford et al. 2013, Rorabaugh 2008a). Frost and Bagnara (1976) described *Rana magnaocularis* (Northwest Mexico leopard frog) in western Mexico from Sinaloa to east-central Sonora. Platz and Mecham (1979) described *R. chiricahuensis* (Chiricahua leopard frog) from southeastern Arizona and adjacent Sonora. Platz and Frost (1984) described *R. yavapaiensis* (lowland leopard frog) from central Arizona and northern Sonora. Platz (1993) described *R. subaquavocalis* (Ramsey Canyon leopard frog) from the Huachuca Mountains, Arizona, but is currently considered a synonym of *R. chiricahuensis* (Goldberg et al. 2004, Crother 2008). Frost et al. (2006) transferred most of the North American *Rana* to the genus *Lithobates.* Preliminary research revealed genetic differentiation between *L. chiricahuensis* along the Mogollon Rim in central Arizona and more southern populations (Platz and Grudzien 1999, Goldberg et al. 2004), suggesting the possibility of two species.

***Species description -*** The Chiricahua leopard frog is recognized by the distinctive pattern on the rear of the thigh consisting of small, raised, cream-colored spots or tubercles on a dark background; dorsolateral folds that are interrupted and deflected medially; stocky body proportions; relatively rough skin on the back and sides; and often green coloration on the head and back. The species also has a distinctive call consisting of a relatively long snore of 1 to 2 seconds in duration. Snout-vent lengths of adults range from approximately 54 to 120 mm (2.1 to 4.7 in).

**Figure F-29. *Chiricahua leopard frog (*Lithobates chiricahuensis*) - Sycamore and Scotia Canyons, Arizona.*** Photo James C. Rorabaugh

## F-6.1.2 Protection Status

The Chiricahua leopard frog was added to the list of category 2 candidate species for protection under the Endangered Species Act in November 1991 (56 FR 58804) and November 1994 (59 FR 58982; Clarkson et al. 1986, Clarkson and Rorabaugh 1989). Category 2 candidates were those taxa for which there was some evidence of vulnerability and threats, but lacked sufficient data to support a listing proposal. In 1994, the Chiricahua leopard frog was elevated to category 1 candidate status. Category 1 candidates were taxa with sufficient information on biological vulnerability and threats to support proposals to list them as endangered or threatened, but for which preparation of listing proposals was precluded by higher priority listing actions. In 1998, the Southwest Center for Biological Diversity submitted a petition to list this species as federally endangered, and followed with two lawsuits (SCBD 1998, 2013).

In 2002 the U.S. Fish and Wildlife Service listed the frog as Threatened (USFWS 2002). The species' recovery priority of 2C indicates a high degree of threat, a high potential for recovery, and a taxonomic classification as a species. A special rule exempted operation and maintenance of livestock tanks on non-Federal lands from the Section 9 take prohibitions of the Endangered Species Act (USFWS 2007). A recovery plan was completed in April 2007 (USFWS 2007). Critical habitat was designated in 2012 (77 FR 16324, March 20, 2012). Critical habitat includes a total of 10,346 acres (4,187 hectares) in Apache, Cochise, Gila, Graham, Greenlee, Pima, Santa Cruz, and Yavapai counties, Arizona, and Catron, Grant, Hidalgo, Sierra, and Socorro Counties, New Mexico. Safe Harbor agreements are in place throughout the range of the species in Arizona and southwestern New Mexico.

## F-6.1.3 Biology and Distribution Status

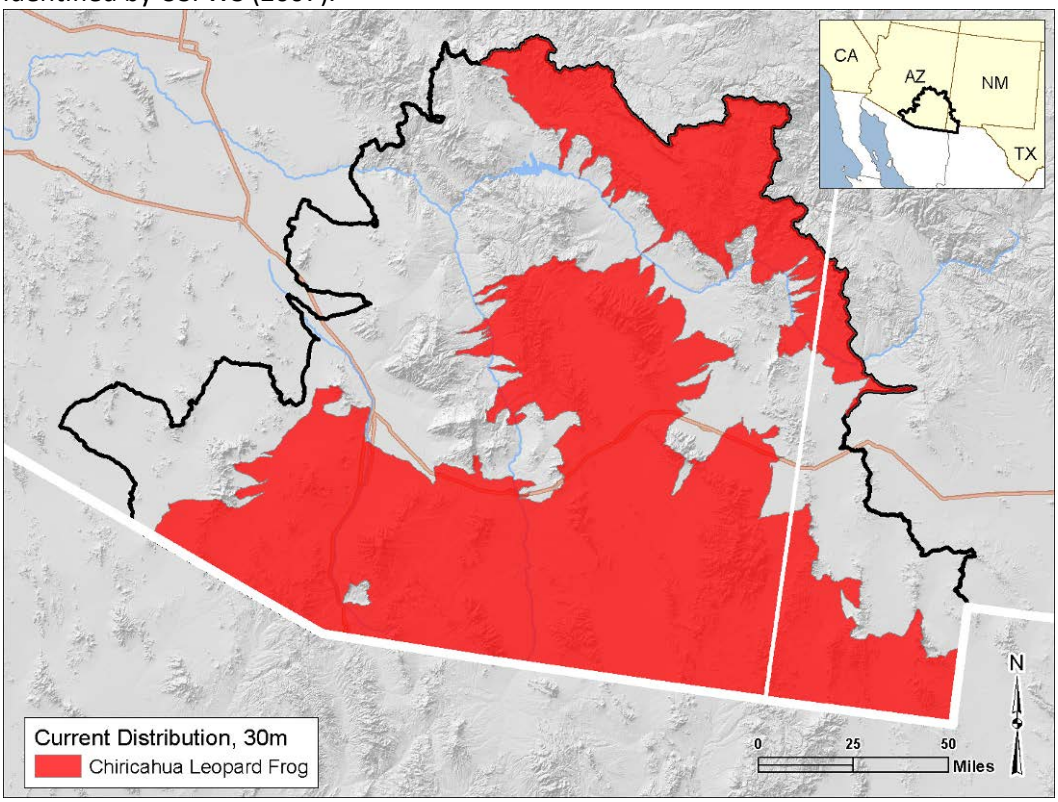
### *F.6.1.3.1 Distribution*

The species occurs in central and southeastern Arizona, west-central and southwestern New Mexico, and the Sky Islands mountain ranges and Sierra Madre Occidental of northeastern Sonora and western Chihuahua, Mexico. More specifically, it occurred historically in the mountains and valleys along the Mogollon Rim east of Camp Verde and the Verde River, but also in southeastern Arizona south of the

Gila River from the Baboquivari Mountains east to the Peloncillo Mountains. Although still fairly well distributed through this range, the species has disappeared from about 88% of its historical localities in Arizona (Rorabaugh 2008b).

Systematic or intensive surveys for Chiricahua leopard frogs have not been conducted in Mexico (USFWS 2007). In northeastern Sonora, it is known in the Madrean Archipelago area from the Sierra San Luis-Sierra Pan Duro complex, the Sierra los Ajos, near Cananea (Sierra Elenita), and in the Río Santa Cruz south of Lochiel (the southern extension of the San Rafael Valley). A specimen from near Yécora in the Sierra Madre Occidental in east-central Sonora likely represents a western extension of the Chihuahua populations. The Chiricahua leopard frog is known in eastern Chihuahua from lowland areas from near Palomas on the New Mexico border south into the Sierra Madre Occidental to Durango (USFWS 2007, Rorabaugh pers. com, 2013).

**Figure F-30. Potential habitat distribution of Chiricahua leopard frog in the MAR ecoregion.** The distribution of potential habitat shows watersheds containing any proportion of recovery units identified by USFWS (2007).




### *F.6.1.3.2 Reproduction*

*Life cycle and reproduction* **-** The life history of the Chiricahua leopard frog has a complex life cycle, consisting of eggs and larvae that are entirely aquatic and adults that are primarily aquatic (USFWS 2007). The species has a distinctive call and males can be temporarily territorial (USFWS 2007). Amplexus is axillary and the male fertilizes the eggs as the female attaches a spherical mass to submerged vegetation. Numbers of eggs in a mass range from 300 to 1,485 and may be correlated with female body size (Jennings and Scott 1991, Sredl and Jennings 2005). Egg masses of Chiricahua leopard frogs have been reported in all months except November to January, but reports of oviposition in June are uncommon (Zweifel 1968, Frost and Bagnara 1977, Frost and Platz 1983, Scott and Jennings 1985).

Eggs hatch in the wild after 8-14 days, depending on water temperature (USFWS 2007). Tadpoles are mainly herbivorous with faster growth rates in warmer conditions. Tadpoles have a long larval period, from three to nine months, and may over winter. Generally activity is between March and October (USFWS 2007).

Males reach sexual maturity at 2.1-2.2 in (5.3-5.6 cm) snout-vent length in less than a year (Sredl and Jennings 2005). Chiricahua leopard frogs presumably experience very high mortality (greater than 90 percent) in the egg and early tadpole stages, high mortality as newly transformed juveniles, and then relatively low mortality when the frogs are adults (Zug et al. 2001, USFWS 2007). Under ideal conditions, Chiricahua leopard frogs may live as long as 10 years in the wild (Platz et al. 1997).

### F-6.1.4 Habitat and Ecology

#### *F.6.1.4.1 Habitat*

The Chiricahua leopard frog is an inhabitant of montane and river valley ciénegas, springs, pools, cattle tanks, lakes, reservoirs, streams, and rivers. The species requires permanent or semi-permanent pools for breeding and water characterized by low levels of contaminants and moderate pH (Watkins-Colwell and Watkins-Colwell 1998). Breeding sites require some open water and, ideally, vegetative cover with some bare substrate. Chiricahua leopard frogs can be eliminated from sites that become overgrown with cattails (Typha sp.) or other emergent plants. Frogs need some emergent or submerged vegetation, root masses, undercut banks, or fractured rock substrates as refugia from predators and extreme climatic conditions (Sredl and Jennings 2005).

Chiricahua leopard frogs also possess an unusual ability among members of the *Rana pipiens* Complex; they can darken their ventral skin under conditions of low reflectance and low temperature (Fernandez and Bagnara 1991, Fernandez and Bagnara 1993), a trait believed to enhance camouflage and escape predation (USFWS 2007). Uplands adjacent to water provide essential foraging and basking sites, particularly in riparian vegetation.

In Arizona, Sredl and Saylor (1998) found a significantly higher proportion (62 percent) of known extant populations in stock tanks as compared to those in riverine habitats (35%), suggesting Arizona populations of this species have fared better in stock tanks than in natural habitats. In some areas, stock tanks replaced natural springs and ciénegas or were developed at spring headwaters and now provide the only suitable habitat available to the Chiricahua leopard frog. Of the nine extant populations of Chiricahua leopard frogs on the Mogollon Rim in Arizona, one is a natural aquatic system and the others are artificial or highly modified aquatic systems (Sredl et al. 1997). The only known localities of the Chiricahua leopard frog in the San Rafael, Altar, and San Bernardino valleys, Buckskin Hills, and the Patagonia Mountains of Arizona are stock tanks. In contrast, many stock tank populations in New Mexico were extirpated, apparently by disease (Painter 2000). Sredl and Saylor (1998) found that stock tanks in Arizona are occupied less frequently by non-native predators (with the exception of American Bullfrogs [*Lithobates catesbeianus*]) than natural sites. There is a high probability that the Chiricahua leopard frog would be extirpated from many more areas if ranchers had not built and maintained stock tanks for livestock production.

Although stock tanks provide refugia for frog populations and are important for this species in many areas, tanks only support small populations and are very dynamic habitats without habitat complexity. Tanks often dry out during drought, and flooding may destroy downstream impoundments or cause siltation, resulting in loss of aquatic communities and extirpation of frog populations. Construction of tanks may destroy natural habitats at or downstream of the tank, and may alter local hydrology.

Periodic maintenance to remove silt from tanks may also cause a temporary loss of habitat and mortality of frogs. Most stock tanks do not provide suitable breeding habitat because they do not regularly hold water long enough for development of larvae to metamorphosis. Sredl and Saylor (1998) caution that stock tank populations are sometimes simply mortality sinks with little reproduction or recruitment.

*Elevation*: Chiricahua leopard frog localities range from 3,281 to 8,890 feet (ft) (1,000 to 2,710 meters [m]) elevation (Platz and Mecham 1979, Sredl et al. 1997, Smith and Chiszar 2003).
*Vegetation*: The Chiricahua leopard frog occurs in a wide variety of permanent and semi-permanent aquatic systems in pine-oak forest, oak woodland, chaparral, plains and desert grasslands, and even desert scrub habitats (Stebbins 1985). Deep areas, root masses, and undercut banks are used when escaping capture. Habitat heterogeneity is likely important, and vegetation adjacent to aquatic habitats is essential for foraging for prey and protection from terrestrial predators, mostly during the summer rainy season.

#### *F.6.1.4.2 Landscape Connectivity*

A metapopulation is a system of local populations connected by dispersing individuals (Hanski and Gilpin 1991). Dispersal habitat provides routes for connectivity and gene flow among local populations within a metapopulation, which enhances the likelihood of metapopulation persistence and allows for recolonization of sites that are lost due to drought, disease, or other factors (Hanski and Gilpin 1991, USFWS 2007). Chiricahua leopard frogs will move rapidly into newly created suitable habitat rapidly near to occupied habitats (Sredl and Jennings 2005). The most likely dispersal routes include combinations of ephemeral, intermittent, and perennial drainages, as well as uplands. Some vegetation cover for protection from predators, and aquatic sites that can serve as buffers against desiccation (drying) and stopovers for foraging (feeding) are desirable along dispersal routes. A lack of barriers that would block dispersal is critical. Features on the landscape likely to serve as partial or complete barriers to dispersal include cliff faces and urban areas (USFWS 2007), reservoirs 20 acres (50 hectares) or more in size that are stocked with sport fishes or other nonnative predators, highways, major dams, walls, agricultural fields, or other structures that physically block movement (Todd and Andrews 2008, Eigenbrod et al. 2009). The effects of highways on frog dispersal can be mitigated with frog fencing and culverts (USFWS 2007). Although detailed studies of dispersal and metapopulation dynamics of Chiricahua leopard frogs have not been conducted, Jennings and Scott (1991) noted that maintenance of corridors used by dispersing juveniles and adults may be critical to conserving populations of frogs.

#### *F.6.1.4.3 Food*

Larval Chiricahua leopard frogs are primarily herbivorous. After metamorphosis, the frogs eat an array of invertebrates and small vertebrates (USFWS 2007). Available food items at one population site included bacteria, diatoms, phytoplankton, filamentous green algae, water milfoil (*Myriophyllum* sp.), duckweed (*Lemna minor*), and detritus (Marti and Fisher 1998). Adults eat invertebrates such as beetles, true bugs, and flies, but also fish and snails (Christman and Cummer 2006).

#### *F.6.1.4.4 Phenology*

Chiricahua leopard frogs are mostly active from March to October in the warm season with hibernation in the winter months and estivation in the arid foresummer (May-June; USFWS 2007). They can be found active both day and night, but adults tend to be active more at night than juveniles (Sredl and Jennings 2005).

#### *F.6.1.4.5 Competitors*

Various other vertebrates potentially compete with the Chiricahua leopard frog for food and habitat. Historically, Chiricahua leopard frogs occurred together with native ranid frogs more frequently (James C. Rorabaugh, pers comm. 2013). Today Chiricahua leopard frogs occur with lowland leopard frogs at Peña Blanca Spring, Peña Blanca Canyon below Peña Blanca Lake, and Thumb Butte Tank. Chiricahua leopard frogs occurred with Plains leopard frogs in the Sulphur Springs Valley, but have not been found together in Arizona since the late 1980s. Both species were still together at Cuchillo Negro Warm Springs in Sierra County, New Mexico as recently as a couple of years ago.

Non-native animals such as American bullfrogs, tiger salamanders, and centrarchid fishes are both predators (see above) and competitors of Chiricahua leopard frogs.

#### *F.6.1.4.6 Demographics*

The Chiricahua leopard frog is rare in suitable habitat, and local abundance appears to fluctuate greatly, and populations in stock tanks generally include fewer than 100 individuals (Santos-Barrera et al. 2004). Historically, it occurred at 212 sites in Arizona, 170 in New Mexico, and 12-13 in Mexico (USFWS 2007). It is now absent from many historical localities and numerous mountain ranges, valleys, and drainages within its former range (USFWS 2007). Where still present, populations often are few, small, and widely scattered (USFWS 2007). Some disappearances from historical sites probably represent natural fluctuations, but in most areas disappearances appear to reflect real, on-going declines caused by human impacts, (USFWS 2007).

Also of importance are degradation and loss of habitat as a result of water diversions and groundwater pumping, livestock management that degrades frog habitat, a history of fire suppression and grazing that has increased the likelihood of crown fires, mining, development, and environmental contamination; disruption of metapopulation dynamics; and increased chance of extirpation or extinction resulting from small numbers of populations and the dynamic nature of frog habitats (USFWS 2002).

#### *F.6.1.4.7 Conservation Actions*

Conservation efforts are being undertaken in Coconino, Tonto, Apache, Sitgreaves, Gila, and Coronado National Forests (Jennings 1995; Sredl et al. 1997). Both the northern and southern populations of *R. chiricahuensis* are listed as threatened under the Endangered Species Act in 2002, and a recovery team was established in 2003. Conservation actions include both short-term interim actions to prevent further deterioration of the species' status, and longer-term planning for eventual recovery of the species. Arizona Game and Fish Commission Order 41 prohibit the collection of this species from the wild in Arizona. It is included as Wildlife of Special Concern in Arizona (Arizona Game and Fish Department 1996). Priority research topics include identification of the importance of disease, pesticides and other contaminants, climate change, UV radiation, fire management, and possibly other threats to the status and recovery potential of the Chiricahua leopard frog. Also, research is needed on key aspects of the frog's status, distribution, and ecology.

### F-6.1.5 Change Agent Characteristics

#### *F.6.1.5.1 Non-native Predators*

Numerous native vertebrate predators commonly coexist with the Chiricahua leopard frog, including fishes, the Sonoran mud turtle (*Kinosternon sonoriense*), other ranid frogs, and garter snakes (Rosen et al. 1996a, Platz and Mecham 1979, USFWS 2007). Tiger salamanders are native to the Chiricahua leopard frog's range in the San Rafael Valley in southeastern Arizona (*A. t. stebbinsi*), and the mountains

of Sonora, Chihuahua, and Durango (*A. rosaceum*). Native fishes, such as trout (*Oncorhynchus*), chub (*Gila*), longfin dace (*Agosia chrysogaster*), and topminnow (*Poeciliopsis*) occur within the range of the Chiricahua leopard frog.

Predation by non-native American Bullfrogs, fishes, crayfish (*Orconectes virilis* and possibly others), and tiger salamanders is implicated as a contributing factor in the decline of ranid frogs in western North America (Moyle 1973, Hayes and Jennings 1986, Bradford et al. 1993, Fernandez and Rosen 1996), and may be the most important factor identified so far in the current decline of the Chiricahua leopard frog (Rosen et al. 1994, 1996a). Barred Tiger Salamander (*Ambystoma mavortium*) is introduced in southeastern Arizona and southwestern New Mexico.

In southeastern Arizona, Rosen et al. (1994, 1996a) documented 13 non-native predaceous vertebrate species in aquatic communities in the range of the Chiricahua leopard frog, including American Bullfrog, Barred Tiger Salamander (*Ambystoma mavortium*), Largemouth Bass (*Micropterus salmoides*), trout, catfish and eight other fish species. Rosen et al. (1994, 1996a) found that when American bullfrogs and centrarchid fish (bass and Green Sunfish, *Lepomis cyanellus*) were present, Chiricahua leopard frogs were absent. Non-native vertebrate predators were absent from 16 of 19 localities where Chiricahua leopard frog localities. Chiricahua leopard frogs are rarely are found with non-native predators in diverse and complex habitats with shallow water, vegetation cover, and other features that provide refuge from predators.
In the San Rafael Valley, Arizona, Chiricahua leopard frogs were only found at sites that lacked non-native fishes and American Bullfrogs (Snyder et al. 1996). In the White Mountains of Arizona, the disappearance of Chiricahua leopard frogs from most historical localities was correlated with the appearance of native tiger salamanders and non-native crayfish (Fernandez and Rosen 1996, Fernandez and Bagnara 1995). Crayfish were found to prey upon Chiricahua leopard frog larvae, metamorphs, eggs, and adults. Crayfish spread to the breeding pond of one of the last and possibly the most robust populations of Chiricahua leopard frogs in the White Mountains, Arizona (USFWS files, Phoenix, AZ; Fernandez and Rosen 1998), and have become very abundant in former Chiricahua leopard frogs habitats on the Blue River, Arizona (J. Rorabuagh, pers. comm. 2011).

**Stresses:** Introduction of non-native predators/competitors in habitats.

**Responses:** Decline or extirpation of population.

#### *F.6.1.5.2 Disease*

Chytridiomycosis is a fungal skin disease caused by the pathogen *Batrachochytrium dendrobatidis* (Bd) that is killing amphibians (mostly anurans) around the globe. Aquatic larval stages contract it, but it only attacks keratin. The only keratin in tadpoles is in the mouthparts, therefore larvae are not substantially affected (Berger et al 1998; Garmyn et al. 2012; James C. Rorabaugh, pers. comm., 2013). After metamorphosis, frogs and toads have keratin in their skin.

The extent of morbidity and mortality in amphibian species worldwide varies within and among populations due to factors such as ecological factors such as climate, elevation, host life history traits and host immunity due to genetic makeup and previous exposure (Savage and Zamudio 2011). Savage and Zamudio's study (2011) highlights the importance of genetic diversity to maintain populations in the face of this disease. Forrest and Schlaepfer (2011) found that lowland leopard frogs (*Rana yavapaiensis*) in Arizona that resided in water sources >30 degrees Celcius (86 degrees Fahrenheit) were significantly protected from infection by Bd. In locations where Bd was known to be present; infection rates in frogs

dropped from 75-100 percent in water <15 degrees Celcius (59 degrees Fahrenheit) to less than 10 percent in water warmer than 30 degrees Celcius. Temperature is also strongly linked to amphibian immune response (Forrest and Schlaepfer 2011).

Waterfowl, particularly wild geese, have been shown to be a carrier of Bd via their keratinous toe tissue and may play a role in dissemination of the disease (Garmyn et al. 2012). Water in which an infected frog is found should be considered contaminated for 7 weeks after removal of the infected individual (Johnson and Speare 2003).

Most die offs in frog populations in Arizona and Sonora occur during the winter months, late fall, or early spring, presumably when the frog's immune system is less able to fight off the Bd fungus. Such die offs called Postmetamorphic Death Syndrome (PDS) are characterized by death of all or nearly all metamorphosed frogs in a short period of time, leaving only tadpoles surviving in the population (Scott 1993). Dead or moribund frogs are often found during or immediately following winter dormancy or unusually cold periods. The apparent PDS of Tarahumara Frog (*Lithobates tarahumare*) populations were documented in southern Arizona and northern Sonora as early as 1974, and by 1983 this species had died out in Arizona (Hale et al. 2005).

Chytridiomycosis was documented in the Chiricahua leopard frog as early as 1992 (Santo-Barrera et al. 2004). Infected populations may exhibit periodic die-offs or be extirpated, but the Chiricahua leopard frog is persisting with the disease. In some areas that lack stressors (i.e., lower predation, higher population numbers, warmer waters, higher pH, and/or lower elevation), the disease appears to have little effect on population viability (USFWS 2007). Chytrid die offs have been seen in Chiricahua leopard frog populations in Catron County, New Mexico, as well as Arizona and Sonora. In some years, very few Chiricahua leopard frogs occurred in the canyons of the Santa Rita and Pajarito mountains in the spring, suggesting that frogs were dying during the winter months.

A number of other diseases may affect Chiricahua leopard frog although few have been documented in Chiricahua leopard frog populations. These include a ranaviruses confirmed present in a dead Chiricahua leopard frog, a bacteria that causes red leg and other pathogenic bacteria, fungal diseases such as Choromycosis, Saprolegniasis, and Phycomycosis, and protozoans and metazoans among others (USFWS 2007).

**Stresses:** Increased chytridiomycosis fungal infections.

**Responses:** More frequent Postmetamorphic Death Syndrome die-offs of adults in the winter with loss of breeding for a year if population recovers (if all adults die overwinter, there is not spring breeding so tadpoles need to transform and breed – either the second or third year). Reduced probability of dispersal between populations.

#### *F.6.1.5.3 Habitat Modification and Livestock Management*

Livestock grazing is the most widespread land management practice in western North America (Fleischner 1994). Intense livestock grazing during the late 1800s and early 1900's was likely a key cause of change in the structure and composition of montane forests, arroyo cutting and loss of ciénegas and riparian systems, increased shrub dominance in grasslands, and altered fire regimes (Hendrickson and Minckley 1984, Swetnam and Baisan 1996), although other factors such as groundwater pumping, logging, mining, loss of beaver populations, and climate change likely contributed (Hereford 1993, Bahre 1995a, b, Geraghty and Miller, Inc. 1995). Livestock are adapted to mesic habitats but are attracted to

the high quality and quantity of forage and select riparian habitats for water, shade, and cooler temperatures, and can adversely affect riparian systems in a number of ways (Fleischner 1994, Belsky et al. 1999, Jones 2000).

As the human population has grown throughout the region and demand for water has intensified, aquatic ecosystems have been greatly altered (Kolar 2003). Severe fragmentation and alteration of aquatic habitats in the southwestern United States has likely constricted many wide ranging aquatic species into isolated pockets, and maintenance of aquatic corridors may be critical in preserving organisms in the arid Southwest (Jennings and Scott 1991). Numerous dams and intensive livestock grazing practices have changed water temperature and flow regimes, usually reducing habitat quality for native fishes (Rinne and Minckley 1991). Since the late 1800s and early 1900s, construction of earthen cattle tanks in upland drainages has been a common range management practice (U.S. General Accounting Office 1988). Livestock tanks provide water and aquatic habitats to many species of wildlife, including amphibians.

Livestock grazing is nearly ubiquitous within the historical range of the Chiricahua leopard frog, but grazing impacts on its populations are not well studied. The effects of livestock grazing on amphibian populations may be positive or negative (Jennings 1988, Rosen and Schwalbe 1998, Sredl and Saylor 1998). However, adverse effects to the Chiricahua leopard frog and its habitat may occur under certain circumstances as a result of livestock activities, including trampling of eggs, tadpoles, and frogs; deterioration of watersheds; erosion and/or siltation of stream courses; elimination of undercut banks that provide cover for frogs; loss of wetland and riparian vegetation and backwater pools; and spread of disease and non-native predators (Gunderson 1968, Arizona State University 1979, Hendrickson and Minckley 1984, Jennings 1988, Ohmart 1995, Jancovich et al. 1997, Belsky and Blumenthal 1997, Bartelt 1998, Belsky et al. 1999, Ross et al. 1999, U.S. Fish and Wildlife Service 2007, Sredl and Jennings 2005). Increased watershed erosion caused by grazing can accelerate sedimentation of deep pools used by frogs (Gunderson 1968). Sediment can alter primary productivity and fill interstitial spaces in streambed materials with fine particulates that impede water flow, reduce oxygen levels, and restrict waste removal (Chapman 1988), or fill pools entirely. In the dry season, low water levels in stock tanks and high detritus loads (including cattle feces), low water levels, high water temperature, and low concentrations of dissolved oxygen can result in high levels of hydrogen sulfide toxic to frogs and cause die-offs (Sredl et al. 1997).

Sredl and Howland (1994) speculated that distribution of extant Chiricahua leopard frog populations in Arizona might be reflective of habitat fragmentation and extinction without recolonization, as well as habitat quality.

Due to the cumulative effects of continued grazing by cattle and elk in central Arizona and west central New Mexico and other anthropomorphic stresses, riparian areas have been deemed the most damaged and threatened ecosystem in the Southwest (Fleischner 1994, Catron et al. 2000).

Other habitat modification taking place in the MAR includes spring development, road development, and channelization of previously pooled habitat.

**Stresses:** Degradation of habitats (increased soil erosion on slopes/siltation, silt removal in stock tanks, reduced water, etc.), degradation of upland habitat, conversion of natural habitats to stock tanks, contamination of water in habitats, increased habitat fragmentation.

**Responses:** Population declines or extirpation, less successful breeding in marginal habitats, increased predation by non-native species, reduced resistance to chytridiomycosis, reduced dispersal and gene flow between populations

### *F.6.1.5.4 Climate Change*

Climate change is an ongoing process in the Southwest with potential effects on all species and biotic associations (Thompson et al. 1997). Mean annual temperatures rose 2.0-3.1°F in the American Southwest in the 20th century, and are predicted to rise 8.1-11.0°F in the 21st century (Southwest Regional Assessment Group 2000). Changes in precipitation are more speculative with contradicting predictions in different models (Southwest Regional Assessment Group 2000, Patterson 1997, Betancourt 2004). The effects of different climatic change scenarios are discussed here.

*Warmer mean annual temperatures* – With annual warming there would be a general upward shift of biotic communities (desertscrub, grassland, oak woodland, pine-oak forest, and mixed-conifer forest). For montane woodlands and forests, there would be reductions in their areas as well. Mixed-conifer forests at the highest elevations would be most vulnerable. There could be a general conversion of plains grassland (San Rafael Valley, Animas Valley) to desert grassland. In desert grassland, shrub dominance should increase as perennial grass and herbaceous perennial dominance declines. Similar shrub increases occurred twice in the last 4,000 years during warm periods before European arrival (Van Devender 1995).

To the south in Sonora, desert grassland is replaced by foothills thornscrub where winter minimum temperatures increase. Tropical plants are very vulnerable to hard freezes. The northern distributional limits of many tropical species are in fingerlike northern tributaries of the Río Bavispe (in the greater Río Yaqui drainage basin) in northeastern Sonora, southeastern Arizona, and southwestern New Mexico. These species would move northward with warmer winter temperatures.

Hurricanes are formed where sea surface temperatures are 79.7°F (26°C) or higher (Data Discover Hurricane Science Center 2013). Today such storms can bring very large amounts of precipitation to Baja California, Sonora, and the Sky Island region in October and November. Storms are either part of the Bermuda high circulation that cross Panama and move northward through the Gulf of California or from the southern Pacific Ocean crossing Baja California to reach the mainland. With increased mean annual and sea surface water temperatures, tropical storms should reach the Madrean Archipelago more frequently with potentially large increases in precipitation.

*Warmer temperatures, reduced winter precipitation, increased summer precipitation* - Reductions in winter precipitation are expected as the North Pacific warms, the Aleutian Low cyclonic circulation pattern weakens, and fewer winter storm tracts cross California to the Sky Island Region. In the Madrean Archipelago region, reduction in winter-spring precipitation lengthens the arid foresummer in May-June with related increases in forest fires and greater post-fire erosion. Shifts to greater summer precipitation dominance would have dramatic impacts on species and vegetation.

Winter precipitation gradually decreases southward in Sonora, and thornscrub and tropical deciduous forest (TDF) in southern Sonora are increasingly dominated by summer precipitation. However, winter drought influences successful spring reproduction of many tropical plants. In contrast, TDF from central Sinaloa southward to Costa Rica has very little or no winter precipitation. In the Sierra Madre Occidental and Sky Islands in northeastern Sonora, winter drought has serious impacts on ‘summer evergreen

Madrean' oaks (*Quercus chihuahuensis, Q. emoryi, Q. oblongifolia,* and others) and is an important abiotic limiting factor for the oak woodland-desert grassland ecotone.

Duration and amount of precipitation in the summer monsoon season increase southward into New World tropics, with more intense monsoon storms.

*Warmer temperatures, reduced winter and summer precipitation* - The same predictions for reduced spring precipitation discussed above apply in this scenario. Less summer rainfall would inhibit the northern expansions of tropical thornscrub species. Reductions of both winter and summer precipitation would increase impacts on desert grassland and montane woodlands and forests. Desert grassland would be even more prone to shrub increases with warm season shrubs like velvet mesquite (*Prosopis velutina*) favored over cool season subshrubs (*Gutierrezia, Isocoma,* other Asteraceae).

*Greater climatic variability* - In recent years, climate seems to be more variable with more frequent extreme events (Karl et al. 2009). Incursions of frigid Arctic air are particularly important in defining the transition between the New World tropics and northern temperate biomes at ca. 28°30' to 30°N latitude in east-central Sonora (Van Devender et al. 2010). Even with increasing mean annual temperatures and summer precipitation, freezes will have catastrophic impacts, including pruning or killing velvet mesquite, creosotebush (*Larrea divaricata*), succulents, etc. in desert grassland; pruning or killing tropical species in foothills thornscrub, limiting their northward dispersal; and dramatic impacts on plant phenology (flowering and seed production), reducing critical spring insect populations for animals.

Climate change is an ongoing process in the Southwest, but contradicting predictions about future precipitation regimes make evaluating threats to the Chiricahua leopard frog and developing recovery strategies difficult. Increasing temperatures have the potential to alter Chiricahua leopard frog breeding phenology, with potential direct effects of earlier reproduction in spring, more rapid development, shorter period of hibernation, longer period of estivation, changes in abilities to find food, spread of infectious disease, and changes in immune function (Blaustein et al. 2001, Beebee 2002). Increasing temperatures may affect the population dynamics of chytridiomycosis because the fungi's growth (Collins et al. 2003, Piotrowski et al. 2004) and effectiveness of antimicrobial peptides on the skin of ranid frogs (Longcore et al. 1999) are temperature dependent.

Increased summer precipitation would improve aquatic habitats and facilitate dispersal of Chiricahua leopard frogs between habitats. Yet, increased precipitation may provide more opportunities for predators to spread and adversely affect frog populations, offsetting any benefits due to more mesic conditions for Chiricahua leopard frogs.

If increased temperatures are coupled with reduced winter and summer precipitation, a variety of indirect effects could occur, including loss and fragmentation of Chiricahua leopard frog habitats. Winter precipitation is important in maintaining water levels in spring and early summer in montane riparian habitats. Reduction in winter rainfall could be especially problematic for the Chiricahua leopard frog, which have been largely relegated to headwater canyons, springs, and stock tanks by the presence of non-natives in the larger aquatic systems. A lot of those smaller systems are at risk of drying out before the monsoons start, which heightens the importance of sites where water levels can be managed (James C. Rorabaugh, pers. com. 2013).

During drought, proximity of suitable drought-resistant habitats may be critical to persistence of each frog population. If Chiricahua leopard frogs cannot disperse from drying habitats and reach suitable

habitat, droughts are likely to produce major population declines. Small drought refugia (crevices in concrete near an overflowing drinker, or accessibility to water storage tanks or drinkers) could become critically important for survival of frogs. Combined winter-summer drought would likely reduce habitat for and invasion by non-native predators.

Changes in interactions with prey, competitors, predators and parasites may be most serious adverse consequences of climate warming on amphibian populations. Drought could result in extirpations of Chiricahua leopard frogs as stock tanks and other marginal habitats dry up. Grazing impacts on habitats could be greater. Dispersal between populations would be more difficult, decreasing the recovery potential and survival of metapopulations.

**Stresses:** Increasing temperatures; reduced winter precipitation; increased summer precipitation; winter and summer drought.

**Responses:** earlier reproduction in spring, more rapid larval development, greater mortality of metamorphs and small adults, shorter hibernation period, extended arid foresummer with longer estivation period and increased forest fires and erosion; increased drying of essential headwater, spring, and stock tank habitats, declines and loss of populations; greater spread of chytridiomycosis fungal disease and non-native predators; limit dispersal and gene flow between populations.

### F-6.1.6 Ecological Status: Key Ecological Attributes and Indicators

This section of the conceptual model addresses Key Ecological Attributes and their potential indicators. The ecological status is a way of describing current status via criteria, functionality, or levels of attributes and asks if they are within the normal range of variation.

#### *F.6.1.6.1 Key Ecological Attributes*

Table F-10 identifies the key ecological attributes for this CE within the Madrean Archipelago ecoregion. A **key ecological attribute** of a focal ecological resource is a characteristic of the resource's biology, ecology, or physical environment that is critical to the resource's persistence in the face of both natural and human-caused disturbance. Alteration of such a characteristic beyond some critical range of variation will lead to the degradation or loss of the resource within decades or less. The KEAs table lists the identified key ecological attributes, with a brief definition, a rationale for why it is important for the CE, and a listing of stressors or change agents that might be affecting the key attribute.

**Table F-10. Key ecological attributes (KEA of the Chiricahua Leopard Frog (CLF) (*Lithobates chiricahuensis*) in the Madrean Archipelago ecoregion.** Indicators for these KEAs can be used to determine the ecological status for this CE; see **Table F-3** for a list of the indicators assessed in this REA.

| ***KEA Class:*** **KEA Name** | **Definition** | **Rationale** | **Stressors** |
|---|---|---|---|
| ***Landscape Context:* Habitat condition and complexity** | This attribute is the intactness of permanent and semipermanent pools with associated upland habitat for foraging. It includes diversity of habitat structure within aquatic and upland areas. | The CLF requires permanent or semi-permanent pools for breeding with some open water and ideally some emergent or submerged vegetation. It also utilizes adjacent uplands for foraging and basking particularly in riparian vegetation. | Factors that affect landscape condition in CLF habitat can be abiotic (drought, fire) or related to human activities. Fire reduces cover in upland habitats and protective cover and food in riparian habitats, and increases erosion and siltation in CLF habitats. Modifications of hydrological systems in watersheds and groundwater pumping can affect the permanency of water in CLF habitats. Stock tank cleaning or abandonment can reduce or eliminate CLF populations. |
| ***Landscape Context:* Intactness of suitable aquatic habitat** | This attribute is the presence of aquatic habitat this is essential for successful breeding and population success. There are key differences in between stock tanks and other artificial waters that provide refugia for frogs but that only support small populations and natural waters with habitat complexity. | Grazing and related surface modifications affect general landscapes and riparian habitats in them. Water sources created or modified for grazing can create new aquatic habitat for the species or diminish the microhabitat quality at previously extant sites. Stock tanks can serve as refugia but can also be temporary habitat due to periodic maintenance, and may destroy natural habitat or dewater downstream habitat by altering local hydrology. | Grazing impacts the upland riparian habitat of CLF through reduced vegetative diversity, increased erosion related to reduced ground cover and siltation in aquatic habitats; and the reduction of cover plants adjacent to upland riparian habit; and loss in food and increased predation. Some aquatic habitats are modified to hold water longer (benefiting CLF). Well-fed stock tanks create new CLF habitat and populations with different annual dynamics than natural populations |

| *KEA Class:* KEA Name | Definition | Rationale | Stressors |
|---|---|---|---|
| ***Landscape Context:*** **Dispersal routes** | This attribute is the presence of intact routes for dispersal that include combinations of ephemeral, intermittent, and perennial drainages as well as uplands and have some vegetation cover to protect against predators and aquatic sites for buffers against desiccation. They also require the absence of physical barriers to movement. | To maintain genetic integrity long-term, CLF populations are dependent on dispersal within metapopulations and from more distant populations. Dispersal corridors are important biological factors related to population genetic integrity, colonization, survival, disease and predation. They allow for gene flow between local populations within a metapopulation. Intact dispersal corridors allow for recolonization of sites when local populations are lost to disease, drought or other factors (USFWS 2007). Chiricahua leopard frogs are reasonably likely to disperse about one mile overland, three miles along intermittent drainages, and five miles along permanent drainages (USFWS 2007). | Disturbance of dispersal corridors stresses CLF populations. Features likely to serve as partial or complete barriers to dispersal include cliff faces, urban areas (USFWS 2007), reservoirs 20 acres (50 hectares) or more in size that are stocked with sport fishes or other non-native predators, highways, major dams, walls, agricultural fields, or other physical structures that block movement (Todd and Andrews 2008, Eigenbrod et al. 2009) Maintenance of corridors benefits CLF, but may also dispersal of non-native predators, some of which are carriers of chytridiomycosis fungal. |
| ***Size/Extent:*** **Population size & demographics** | This attribute is the abundance of populations of CLF within a metapopulation, in addition to the abundance of individuals at a given breeding site. | Population size and demographic structure are direct indicators of the viability and health of the population, and can affect genetic diversity. Where still present, populations are often few, small and widely scattered (USFWS 2007). Of particular concern for CLF is the small numbers of populations within the landscape which increases the chance of extirpation or extinction. | Population size is influenced by various factors. Water quality and permanence, the presence of native and non-native predators (crayfish, fishes, bullfrogs, salamanders, etc.), and the presence of safe sites during drought affect the mortality of larval and adult CLF. In most areas, disappearances reflect real, on-going declines caused by human impacts (USFWS 2007). In stressed populations, CLF adults are more vulnerable to chytridiomycosis fungal infections. Population survival is more likely if there are good corridors connection sites within metapopulations. |
| ***Biotic Condition:*** **Disease** | This attribute is the abundance of chytridiomycosis fungus in the CLF population. In addition to measuring presence of chytrid in populations, the impact of the disease is increased by lack of adequate water and bank cover and presence of non-native predators that stress populations. | Disease can be an important factor in the biological condition of a species, affecting demography (mortality, survivorship). Infected CLF populations may exhibit periodic die-off or be extirpated. | A chytridiomycosis fungus (*Batrachochytrium dendrobatidis*) has been in CLF populations since at least 1992 (Santo-Barrera et al. 2004), but the species is persisting with the disease. The fungus attacks the skin of post-metamorphic frogs, and in stressed populations can cause severe mortality. Frogs are most heavily affected during or immediately following winter dormancy or unusually cold periods. Populations already stressed by predation, higher population density, etc. may be more affected by the disease (USFWS 2007). |

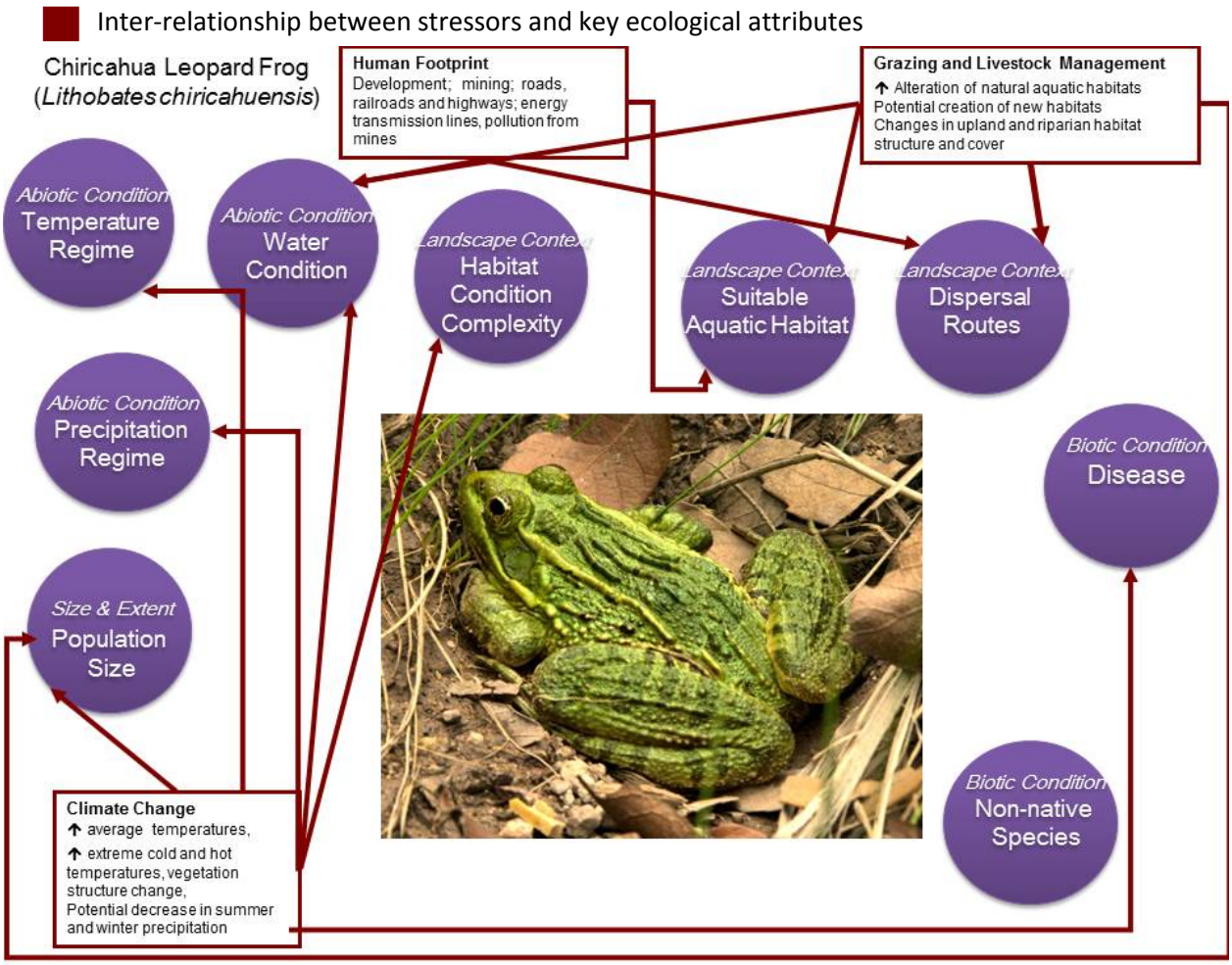
| *KEA Class:* KEA Name | Definition | Rationale | Stressors |
|---|---|---|---|
| ***Biotic Condition:* Non-native species** | This attribute is the abundance of non-native predators that predate and compete with CLF in otherwise suitable CLF habitat. | Non-native species can be invasive with serious impacts on native species and biotic communities. Predation by non-native species is implicated as a contributing factor in the decline of ranid frogs in western North America (Moyle 1973, Hayes and Jennings 1986, Bradford et al. 1993, Fernandez and Rosen 1996), and may be the most important factor identified so far in the current decline of the Chiricahua leopard frog (Rosen et al. 1994, 1996a). | Non-native bullfrogs, fishes, crayfish and tiger slamanders predate CLF. Rosen et al. (1994, 1996a) documented 13 non-native predaceous vertebrate species in aquatic communities in the range of the Chiricahua leopard frog in southeastern Arizona, including American Bullfrog (Lithobates catesbeianus), Barred Tiger Salamander (Ambystoma mavortium), Largemouth Bass (Micropterus salmoides), trout, catfish and 8 other fish species. |
| ***Abiotic Condition:* Precipitation Regime** | This attribute is the amount of precipitation received during the winter and summer seasons in addition to the mean annual precipitation. | The mean annual and seasonal precipitation and fluctuations in them are important abiotic factors in most animal and plant species. Precipitation is a very important in maintaining adequate aquatic habitats for the CLF. Breeding activity is generally between March and October (USFWS 2007). Winter precipitation is important in helping upland stream habitats persist during the arid foresummer in May-July. Summer monsoon precipitation is important because it supports maximum food insect production, and permits dispersal through otherwise dry corridors. | Decreased winter precipitation would lengthen the arid foresummer with related increases in forest fires and greater post-fire erosion. It could also be particularly problematic because CLF populations have been largely relegated to headwater canyons, springs, and stock tanks all of which are at risk of drying out before the monsoons start (J.C. Rorabaugh, pers. com. 2013). Decreased summer precipitation in the future would increase stress on CLF in most aspects of life history. Increased summer precipitation in the future could improve aquatic habitats and facilitate dispersal of CLF but could also facilitate dispersal of predators between habitats. |
| ***Abiotic Condition:* Condition of Water Sources** | This attribute includes the temperature, chemistry, sediment and pollutant load in water in CLF habitat. | Water quality has a direct influence on aquatic animals in riparian habitats. | Pollutants in mountain streams from mining operations likely impact Chiricahua leopard frog populations. Severe wildfires can cause increased sediment deposition, especially below steep slopes (USFWS 2007). In the dry season, low water levels in stock tanks results in high detritus loads (including cattle feces), high water temperature, and low concentrations of dissolved oxygen which can result in high levels of hydrogen sulfide toxic to CLF and cause die-offs (Sredl et al. 1997). |

| *KEA Class:* KEA Name | Definition | Rationale | Stressors |
|---|---|---|---|
| ***Abiotic Conditions:* Temperature Regime** | This attribute is the mean annual temperature, seasonal average temperatures, and extreme highs and lows. | Seasonal temperatures and periodic extreme events control the distributions of most animals and plants. CLF activities including breeding, foraging, hibernation, and estivation periods are directly related to temperature (USFWS 2007). | Increased temperatures coupled with changes in precipitation could lead to loss and fragmentation of CLF habitats. Temperature changes may affect interactions with prey, competitors, predators and parasites which may have serious adverse consequences for CLF. The development of summer monsoonal precipitation is related to temperature. Increasing temperatures may affect the population dynamics of chytridiomycosis because the fungi's growth (Collins et al. 2003, Piotrowski et al. 2004) and effectiveness (Longcore et al. 1999) are temperature dependent. Decreased summer precipitation in the future would increase stress on CLF in most aspects of life history. |

### F-6.1.7 Conceptual Model Diagram

A conceptual model diagram for the CE provides a visual summary representation of the Key Ecological Attributes and stressors that affect the species' persistence (Figure F-31). These diagrams are intended to show how various stressors interact with categories of Key Ecological Attributes to highlight important drivers for species ecological integrity. The arrows indicate relationships between stressors and but do not indicate the nature of the influence (i.e. positive or negative).

**Figure F-31. Conceptual model diagram for the Chiricahua leopard frog, showing key ecological attributes** (by class) for this species, and indicating relationships between stressors and KEAs.




## F-6.2 Considerations and Limitations

The above method for representing the distribution of this CE of necessity over-predicts where it is actually known to occur; however it does provide a watershed and landscape-scale evaluation of potential stressors upon possible reintroduction locations, known critical habitat, or management zones for the species.

Water use scores are the same for all 30 m pixels within each groundwater basin (in Arizona) or county (in New Mexico). These areas are very large relative to the CE distribution. This means that every 30 m pixel within the 6th level (HUC12) watershed gets the same water use impact score. **See Appendix E: Aquatic Systems Status Assessment: Linking CE Conceptual Models to Status Assessment Approach.**

## F-6.3 Ecological Status Assessment Results and Interpretation

This section of the CE summary presents and discusses the results of the ecological status assessment for Chiricahua leopard frog. It addresses each indicator separately, and then addresses the overall assessment, which integrates the results of all individual indicators assessed for the CE. The results are presented using a common framework, in which the status of an indicator – or the combination of all indicators – is scored on a scale from 0.0 to 1.0, where 0.0 indicates a condition of complete replacement of reference ecological conditions due to the impacts of stressors, and 1.0 indicates a condition of no alteration of reference ecological conditions. The same yellow-to-blue color ramp is used for all results, with yellow representing low scores, green moderate scores, and dark blues high scores.

### F-6.3.1 Current Ecological Status: Development, Aquatic Invasives, Recent Severe Burns, and Water Use

The results of the assessments for the four individual indicators for the KEAs for Chiricahua leopard frog are shown in the four maps in Figure F-32 below.

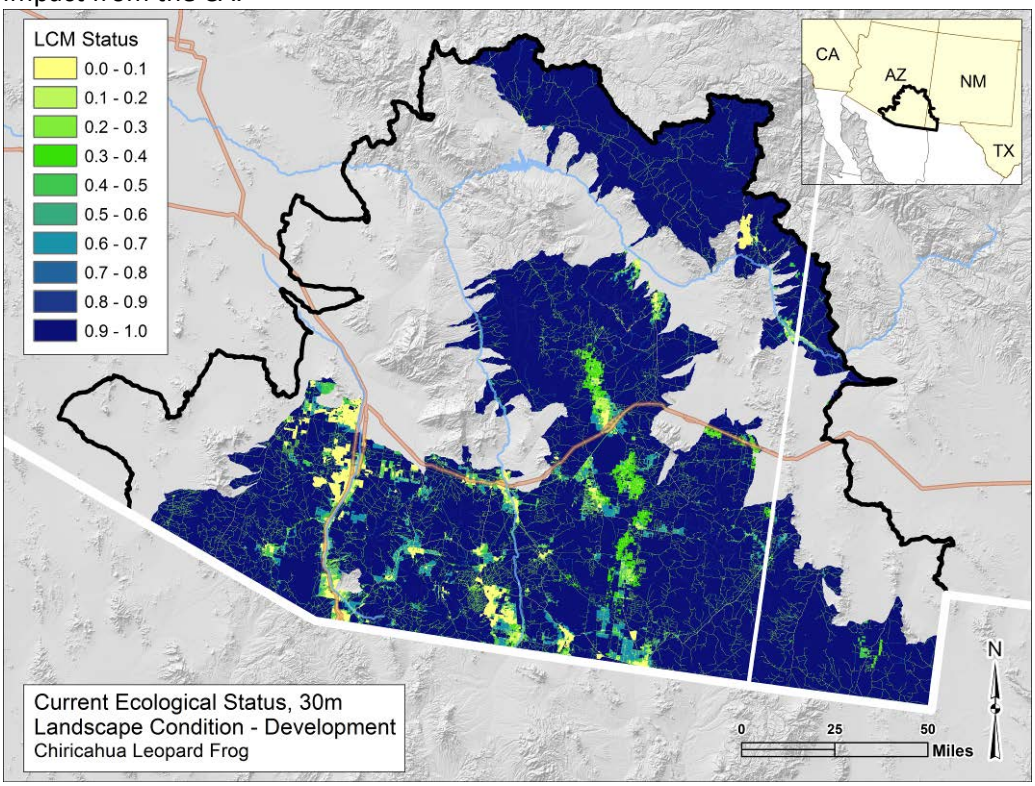
Many development features are not readily visible (e.g., secondary roads or highways, railroads, small agricultural fields) at the ecoregion scale, as shown in the first map of Figure F-32. However, when the development indicator results are viewed at a finer resolution in a GIS, development impacts are especially noticeable in the Tucson metropolitan area; in and around every residential community in the ecoregion; and along corridors associated with interstate highways 10 and 19, US highways 70 and 191, and many other larger roads.

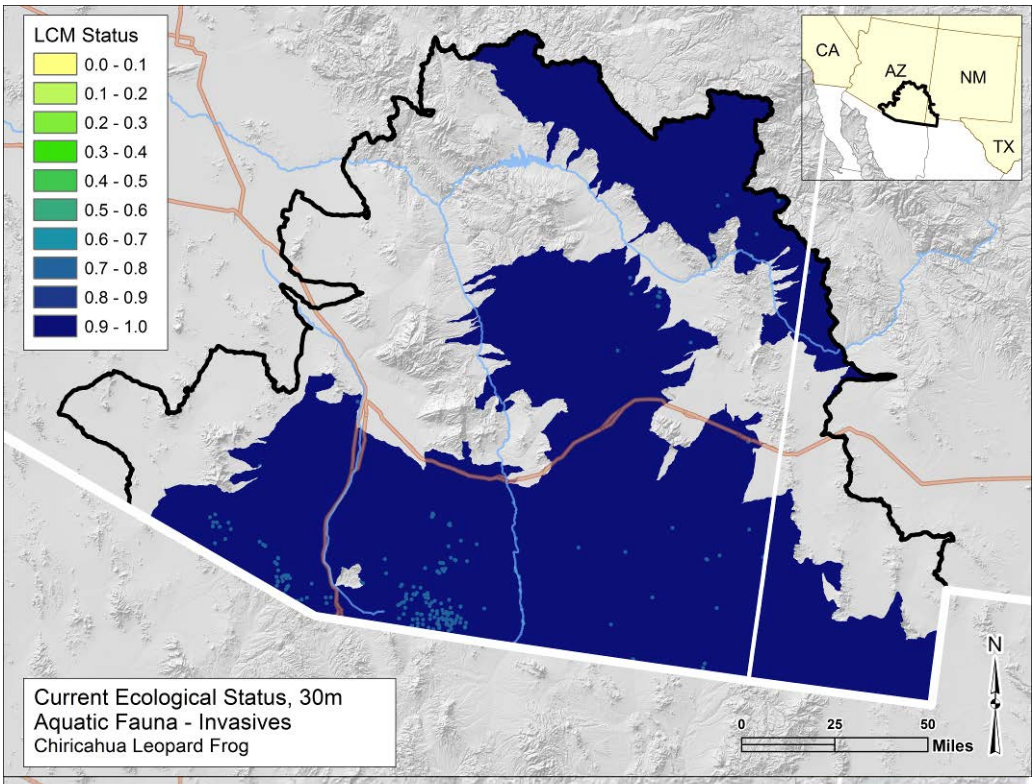
For aquatic invasives, known locations of both highly aggressive species such as bullfrogs and crayfish, and less aggressive species such as salamanders and aquatic plants were each assigned intensity values of 0.7 in the CE response model; lower scores in the invasives indicator results indicate several invasives are present at the same location (second map in Figure F-32). At the ecoregional scale of the maps, it is difficult to see impacted areas. However, it does differentiate infested tributaries from those without invasive species. In the south-central and southwestern parts of the ecoregion, in the upper San Pedro and Santa Cruz watersheds, bullfrog is frequently present (see also Figure E-4 in Appendix E); however, it is unknown whether areas lacking bullfrog simply haven't been surveyed, or have been surveyed and confirmed the absence of bullfrog.

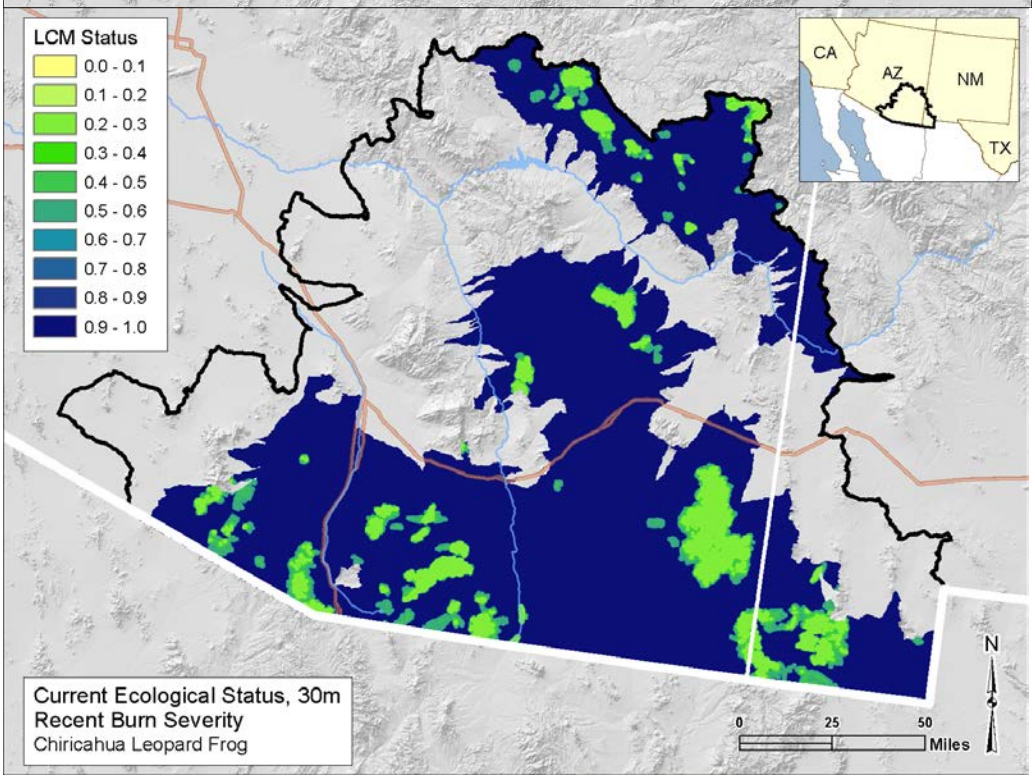
The burn severity map (third map in Figure F-32) shows the locations of the most severe fires over the 15-year period from 1997 through 2011. Fires can result in a pulse of sediment that fills pools and cause local extirpation of frog populations. Where natural fire regimes can be re-established, results indicate watershed and channel improvements benefiting riparian and aquatic ecosystems (USFWS 2007). Areas that see repeat fires may have the higher risk of frog extirpation as these areas have reset the recovery process and recover more slowly than areas that are free from burns for several (>5) or more years. Areas on the map with the lightest colors have seen more severe fires more frequently. These areas may be targeted for population monitoring. However, monitoring non-burn areas will also be needed to see how total frog populations are doing overall (see Luce et al. 2012, Swann 2013, and USFWS 2007 for fire effects on aquatic resources).

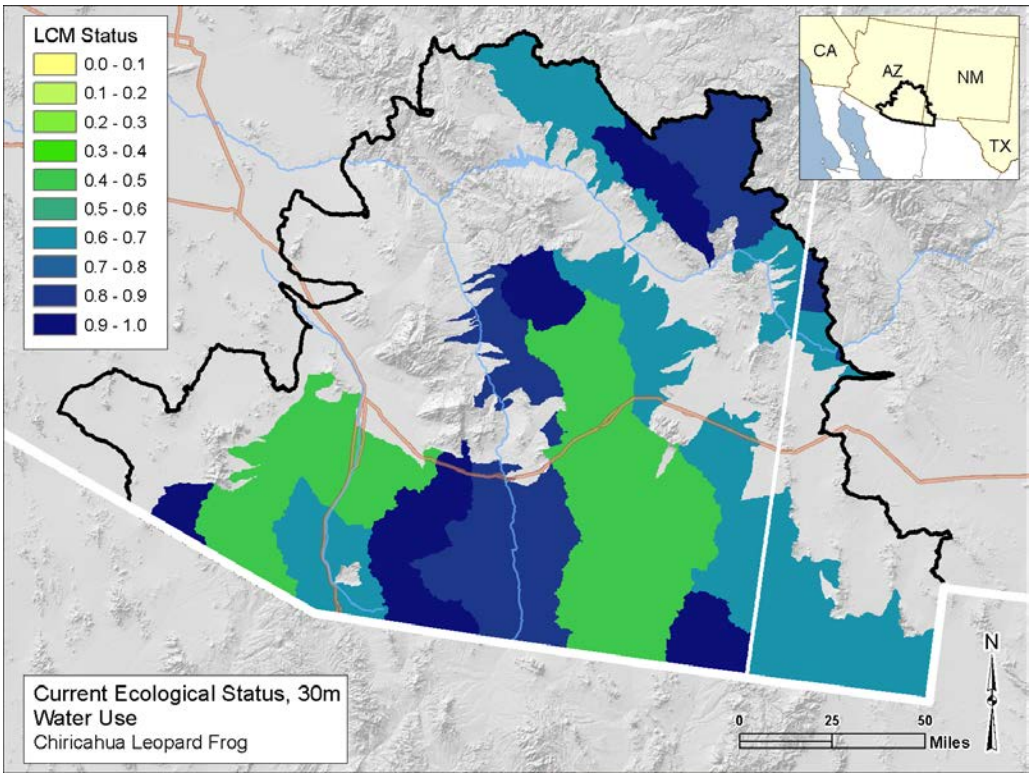
The water use indicator (fourth map in Figure F-32) shows total water use (groundwater and surface water) per acre (by groundwater (in Arizona) or county (in New Mexico). Groundwater basins/counties are quite large, so all the pixels within a 6th-level (HUC12) watershed receive the same water use score. The results show areas with higher water use mainly in the highly populated areas such as Tucson, or high agricultural or high mining water use areas such as the Willcox and Douglas drainages; these are areas where the Chiricahua leopard frog may be stressed from a reduction in the quantity of water available for its habitat.

**Figure F-32. Scores for indicators for Chiricahua leopard frog: development (1st map), aquatic invasive species (2nd map), recent severe burns (3rd map), and water use (4th map) for each 30m pixel.** Yellow (equivalent to 0) indicates high impacts from the CA, dark blue (equivalent to 1) indicates little to no impact from the CA.




LCM Status
0.0 - 0.1
0.1 - 0.2
0.2 - 0.3
0.3 - 0.4
0.4 - 0.5
0.5 - 0.6
0.6 - 0.7
0.7 - 0.8
0.8 - 0.9
0.9 - 1.0
CA
AZ
NM
TX
N
0
25
50
Miles
Current Ecological Status, 30m
Aquatic Fauna - Invasives
Chiricahua Leopard Frog


LCM Status
0.0 - 0.1
0.1 - 0.2
0.2 - 0.3
0.3 - 0.4
0.4 - 0.5
0.5 - 0.6
0.6 - 0.7
0.7 - 0.8
0.8 - 0.9
0.9 - 1.0
CA
AZ
NM
TX
N
0
25
50
Miles
Current Ecological Status, 30m
Recent Burn Severity
Chiricahua Leopard Frog


LCM Status
0.0 - 0.1
0.1 - 0.2
0.2 - 0.3
0.3 - 0.4
0.4 - 0.5
0.5 - 0.6
0.6 - 0.7
0.7 - 0.8
0.8 - 0.9
0.9 - 1.0
CA
AZ
NM
TX
Current Ecological Status, 30m
Water Use
Chiricahua Leopard Frog
0
25
50
Miles
N

### F-6.3.2 Current Ecological Status: Full Scenario

The results of the four individual status indicators were combined to get an overall ecological status score, per pixel of the Chiricahua leopard frog's distribution, shown in the first map of Figure F-33 below. The overall, per-pixel status scores for the leopard frog were then averaged across each 6$^{th}$ level (HUC12) watershed, as shown in the second map of Figure F-33 below.

The overall assessment takes into account the stressor-based indicators of development, aquatic invasive species, recent severe burns, and water use. The results show ecological status scores < 0.5 for approximately half of the pixels (Figure F-34). At the watershed scale (second map of Figure F-33), approximately half of the HUCs have status scores ≤ 0.5, including several HUCs with status scores ≤ 0.3. These contrast with the half of the HUCs having status scores > 0.5, including a subset having status scores in the 0.9-1 range. However, a majority of the HUCs score either 0.4-0.5 or 0.5-0.7. The majority of the distribution of this CE is in moderate to poor condition (Figure F-34). The frequency diagram in Figure F-34 provides a more quantitative measure of HUCs by score interval; some 80% of the HUCs fall at or below the 0.7 score, suggesting only 20% of the leopard frog's possible distribution is in moderately good to excellent condition.

The Gila River corridor downstream from the San Simon River confluence, most of the San Pedro River corridor, and most of the Santa Cruz River corridor south of Tucson show high levels of impact from development, water use, and invasive species. The most altered watersheds containing leopard frog habitat are located in the areas of Animas, NM; and in Safford, Wilcox, and the Tucson metropolitan area in Arizona. The least altered watersheds containing this species occur in the far west-southwestern corner of the ecoregion west and south of Sells, AZ; in the northern third of the lower San Pedro River basin; in the lower San Francisco River basin; and surrounding the San Bernardino National Wildlife Refuge.

**Figure F-33. Overall ecological status scores for Chiricahua leopard frog for all four indicators combined (development, invasives, recent severe burns, and water use) for each 30m pixel (top) and 6th level watersheds (bottom).** Yellow scores (equivalent to 0) indicate high impacts from the CAs, dark blue (equivalent to 1) indicate little to no impact from the CAs. In the second map, the score for each 6th-level watershed is an average of the overall ecological status scores of the CE's 30m pixels within each of those watersheds.

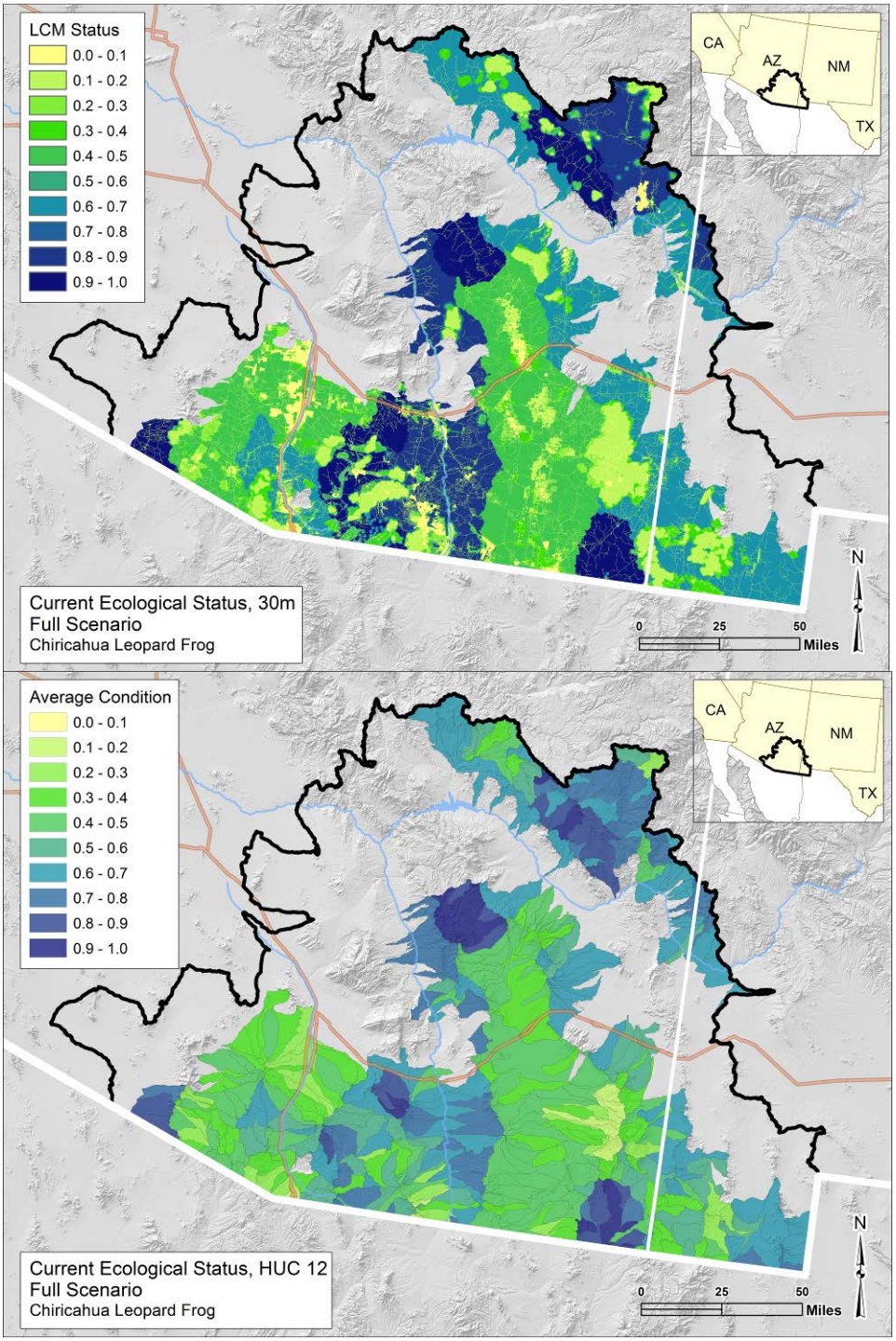
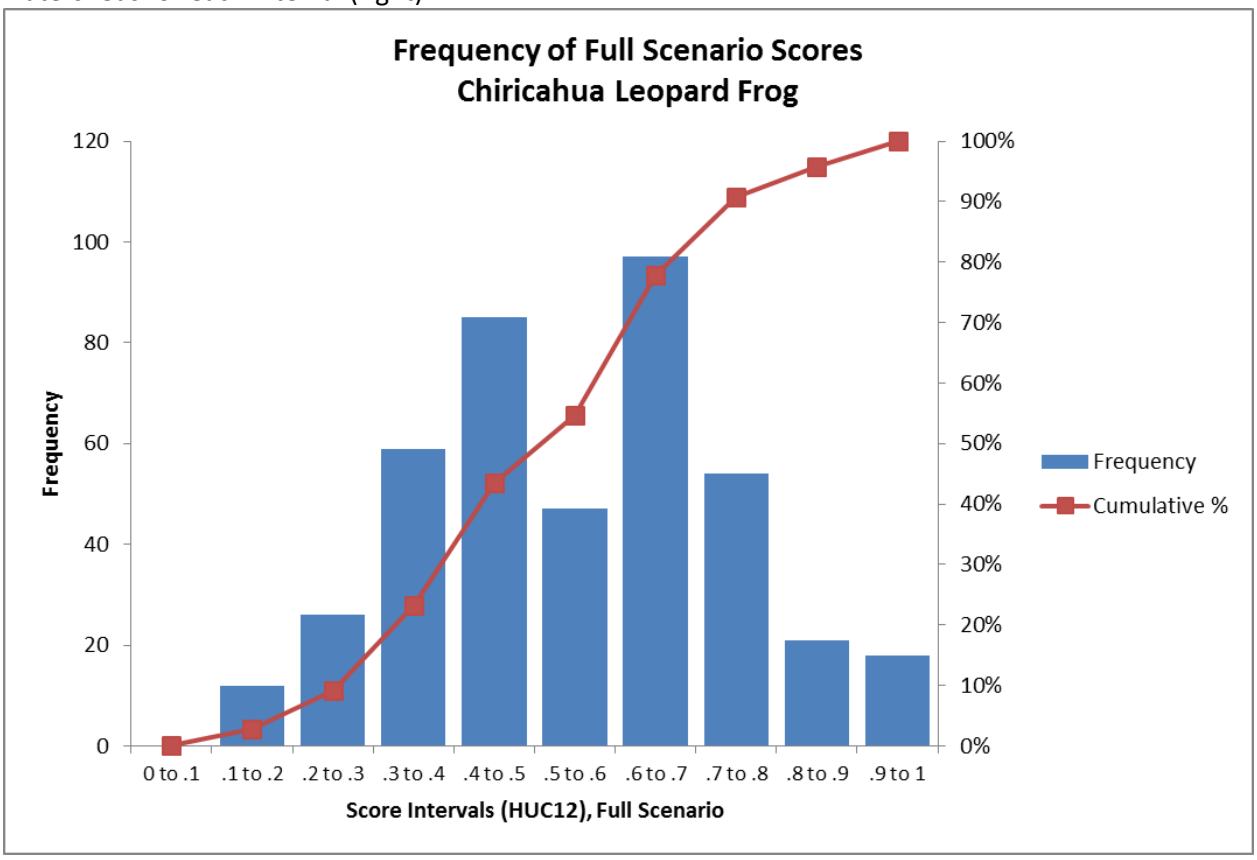


**Figure F-34. Frequency distribution of the 6th-level watershed ecological status scores for Chiricahua leopard frog, with cumulative percent.** The x-axis represents the 0.1 increment scoring intervals, while the y-axis shows the number of watersheds in each interval (left) and the cumulative percentage of the watersheds for each interval (right).




## F-6.4 References for the CE


Arizona State University. 1979. Resource inventory for the Gila River complex, Eastern Arizona. Report to the Bureau of Land Management, Safford District, Safford, Arizona. Contract No. YA-512-CT6-216.

Bahre, C. J. 1995a. Human impacts on the grasslands of southeastern Arizona. Pages 230-264 *in* M. P. McClaran and T. R. Van Devender (eds.), The Desert Grassland, University of Arizona Press, Tucson.

Bahre, C. J. 1995b. Human Disturbance and Vegetation in Arizona's Chiricahua Mountains in1902. Desert Plants 11(4):41-45.

Beebee, T. J. C. 2002. Amphibian phenology and climate change. Conservation Biology 16(6):1454-1455.

Belsky, A. J., and D. M. Blumenthal. 1997. Effects of livestock grazing on stand dynamics and soils in upland forests of the interior west. Conservation Biology 11(2):315-327.

Belsky, A. J., A. Matzke, and S. Uselman. 1999. Survey of livestock influences on stream and riparian ecosystems in the Western United States. Journal of Soil and Water Conservation 54:419-431.

Berger, L., Speare, R., Daszak, P., Green, D. E., Cunningham, A. A., Goggin, C. L., Slocombe, R., Ragan, M. A., Hyatt, A. D., McDonald, K. R., Hines, H. B., Lips, K. R., Marantelli, G., and Parkes, H. 1998. Chytridiomycosis causes amphibian mortality associated with population declines in the rain forests

of Australia and Central America. Proceedings of the National Academy of Sciences, USA. 95: 9031-9036.

Betancourt, J. L. 2004. Ecological responses to climate variability in time and space: a Southwestern synopsis. Page 19 *in* Program and Abstracts, Connecting Mountain Islands and Desert Seas, Biodiversity and Management of the Madrean Archipelago II and 5th Conference on Research and Resource Management in the Southwestern Deserts, Tucson, Arizona.

Blaustein, A. R., L. K. Belden, D. H. Olson, D. M. Green, T. L. Root, and J. M. Kiesecker. 2001. Amphibian breeding and climate. Conservation Biology 15(6):1804-1809.

Bradford, D. F., F. Tabatabai, and D. M. Graber. 1993. Isolation of remaining populations of the native frog, *Rana muscosa*, by introduced fishes in Sequoia and Kings Canyon National Parks, California. Conservation Biology 7(4):882-888.

Catron, J-L. E., S. H. Stoleson, P. L. Stoleson, and D. W. Shaw. 2000. Riparian areas. Pages 281-327 *in* Jemison and Raish (eds.). Livestock Management in the American Southwest: Ecology, Society, and Economics. Elsevier Science B.V. Amsterdam, The Netherlands.

Chapman, D. W. 1988. Critical review of variables used to define effects of fines in redds of large salmonids. Transactions of the American Fisheries Society 117:1-21.

Clarkson, R. W., and J. C. Rorabaugh. 1989. Status of leopard frogs (*Rana pipiens* Complex) in Arizona and southeastern California. Southwestern Naturalist 34(4):531-538.

Clarkson, R. W., J. C. Rorabaugh, and J. E. Platz. 1986. Status and future of ranid frogs in Arizona. Proceedings of the Arizona-Nevada Academy of Sciences 21:21.

Collins, J. P., J. L. Brunner, V. Miera, M. J. Parris, D. M. Schock, and A. Storfer. 2003. Ecology and evolution of infectious disease. Pages 137-151 *in* R.D. Semlitsch, Amphibian Conservation. Smithsonian Books, Washington D.C.

Comer, P., and J. Hak. 2012. Landscape condition model of the western United States. NatureServe, Boulder, CO. In preparation.

Crawford, J. A., L. E. Brown, and C. W. Painter. 2005. *Rana blairi* Mecham, Littlejohn, Oldham, Brown, and Brown, 1973, Plains leopard frog. Pages 532-534 i***n*** M.J. Lannoo (ed), *Amphibian Declines: The Conservation Status of United States Species***.**

Crawford, J. A., L. E. Brown, and C. W. Painter. 2013. *Rana blair**i*** Mecham. AmphibiaWeb: Information on amphibian biology and conservation. [web application]. 2013. Berkeley, California: Amphibia Web. Available: http://amphibiaweb.org/. (Accessed: Mar 27, 2013).

Christman, B. L., and M.,R. Cummer. 2006. Stomach content analysis of Chiricahua leopard frog (*Rana chirircahuensis*) and plains leopard frog (*Rana blairi*) in New Mexico. Report to the New Mexico Department of Game and Fish, Santa Fe, NM.

Crother, B. I. (ed.). 2008. Scientific and Common Names for Amphibians and Reptiles of North America North of México. Society for the Study of Amphibians and Reptiles, Herpetological Circular No. 37:1-84.

Data Discover Hurricane Science Center. 2013. Role of SST in hurricane genesis. http://www.newmediastudio.org/DataDiscovery/Hurr_ED_Center/Hurr_Structure_Energetics/SST/SST.html [Accessed April 2-13]

Eigenbrod, F., S. J. Hecnar, and L. Fahrig. 2009. Quantifying the road-effect zone: threshold effects of a motorway on anuran populations in Ontario, Canada. Ecology and Society 14(1): 24.

Fernandez, P. J., and J. T. Bagnara. 1991. Effect of background color and low temperature on skin color and circulating ά-MSH in two species of leopard frog. General and Comparative Endocrinology 83:132-141.

Fernandez, P. J., and J. T. Bagnara. 1993. Observations on the development of unusual melanization of leopard frog ventral skin. Journal of Morphology 216:9-15.

Fernandez, P. J., and J. T. Bagnara. 1995. Recent changes in leopard frog distribution in the White Mountains of east central Arizona. Page 4 *in* abstracts of the First Annual Meeting of the Southwestern Working Group of the Declining Amphibian Populations Task Force, Phoenix, Arizona.

Fernandez, P. J., and P. C. Rosen. 1996. Effects of the introduced crayfish *Oronectes virilis* on the native aquatic herpetofauna in Arizona. Report to the Arizona Game and Fish Department, Heritage Program, IIPAM Project No. I94054.

Fleischner, T. F.1994. Ecological costs of livestock grazing in western North America. Conservation Biology 8:629-644.

Forrest MJ, Schlaepfer MA (2011) Nothing a Hot Bath Won't Cure: Infection Rates of Amphibian Chytrid Fungus Correlate Negatively with Water Temperature under Natural Field Settings. PLoS ONE 6(12): e28444. doi:10.1371/journal.pone.0028444

Frost, D. R., T. Grant, J. N. Faivovich, R. H. Bain, A. Haas, C. L. F. B. Haddad, R. O. de Sá, A. Channing, M. Wilkinson, S. C. Donnellan, C. J. Raxworthy, J. A. Campbell, B. L. Blotto, P. Moler, R. C. Drewes, R. A. Nussbaum, J. D. Lynch, D. M. Green, and W. C. Wheeler. 2006. The Amphibian Tree of Life. Bulletin of the American Museum of Natural History no. 297: 1-370.

Frost, J. S., and J. T. Bagnara. 1976. A new species of leopard frog (*Rana pipiens* Complex) from northwestern Mexico. *Copeia* 1976: 332-338.

Frost, J. S. and J. T. Bagnara. 1977. Sympatry between *Rana blairi* and the southern form of leopard frog in southeastern Arizona (Anura: Ranidae). Southwestern Naturalist 22:443-453.

Garmyn A, Van Rooij P, Pasmans F, Hellebuyck T, Van Den Broeck W, et al. (2012) Waterfowl: Potential Environmental Reservoirs of the Chytrid Fungus Batrachochytrium dendrobatidis. PLoS ONE 7(4): e35038. doi:10.1371/journal.pone.0035038

Geraghty and Miller, Inc. 1995. Historical flows and conditions in the San Pedro River. Report to the Water Action Task Force, Sierra Vista Economic Development Foundation, Project No. AZ0473.001, Sierra Vista, Arizona.

Goldberg, C. S., K. J. Field, and M. J. Sredl. 2004. Mitochondrial DNA sequences do not support species status of the Ramsey Canyon leopard frog (*Rana subaquavocalis*). Journal of Herpetology 38(3):313-319.

Gunderson, D. R. 1968. Floodplain use related to stream morphology and fish populations. Journal of Wildlife Management 32(3):507-514.

Hale, S. F., P. C. Rosen, J. L. Jarchow, and G. A. Bradley. 2005. Effects of chytrid fungus on the Tarahumara frog (*Rana tarahumarae*) in Arizona and Sonora, Mexico. Pages 407-411 *in* G. J. Gottfried, B. S. Gebow, L. G. Eskew, and C. B. Edminster (compilers), Connecting Mountain Islands and Desert Seas: Biodiversity and Management of the Madrean Archipelago II. Proceedings RMRS-

P-36, Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Hanski, I., and M. Gilpin. 1991. Metapopulation dynamics: brief history and conceptual domain. Biological Journal of the Linnean Society 42:3-16.

Hayes, M. P., and M. R. Jennings. 1986. Decline of ranid frog species in Western North America: are bullfrogs (*Rana catesbeiana*) responsible? Journal of Herpetology 20(4):490-509.

Hendrickson, D. A., and W. L. Minckley. 1984. Cienegas - vanishing climax communities of the American Southwest. Desert Plants 6(3):131-175.

Hereford, R.1993. Geomorphic evolution of the San Pedro River channel since 1900 in the San Pedro Riparian National Conservation Area, southeast Arizona. USDI, Geological Survey, Open File Report 92-339.

Jancovich, J. K., E. W. Davidson, J. F. Morado, B. L. Jacobs, and J. P. Collins. 1997. Isolation of a lethal virus from the endangered tiger salamander *Ambystoma tigrinum stebbinsi*. Diseases of Aquatic Organisms 31:161-167.

Jennings, R. D. 1988. Ecological studies of the Chiricahua leopard frog, *Rana chiricahuensis*, in New Mexico. Report to Share with Wildlife, New Mexico Department of Game and Fish, Santa Fe, New Mexico.

Jennings, R. D., and N. J. Scott. 1991. Global amphibian population declines: insights from leopard frogs in New Mexico. Report to the New Mexico Department of Game and Fish, Albuquerque, New Mexico. 43 pp.

Johnson, M.L. and R Speare. 2003. Survival of Batrachochytrium dendrobatidis in water: Quarantine and Disease Control Implications. Emerging Infectious Diseases. Vol. 9 (8).

Jones, A. 2000. Effects of cattle grazing on North American arid ecosystems: a quantitative review. Western North American Naturalist 60(2):155-164.

Karl, T. R., J. M. Melillo, and T. C. Peterson (eds.). 2009. Global Climate Change Impacts in the United States. Cambridge University Press.

Kolar, C. S. 2003. Summary. Pages 1-6 *in* Dawson and Kolar (eds.). Integrated management techniques to control non-native fishes. U.S. Geological Survey, Upper Midwest Environmental Sciences Center, La Crosse, Wisconsin, December 2003.

Longcore, J. E., A. P. Pessier, and D. K. Nichols. 1999. *Batracytrium dendrobatidis* gen. Et sp. Nov., a chytrid pathogenic to amphibians. Mycologia 91(2):219-227.

Luce, Charles; Morgan, Penny; Dwire, Kathleen; Isaak, Daniel; Holden, Zachary; Rieman, Bruce. 2012. Climate change, forests, fire, water, and fish: Building resilient landscapes, streams, and managers. Gen. Tech. Rep. RMRS-GTR-290. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 207 p.

Marti, E., S.G. Fisher, J.D. Schade, and N.B Grimm. 2000. Flood frequency and stream-riparian linkages in arid lands. Pages 111-136 *in* Jones, J.B. and P.J. Mullholland (eds) Streams and Groundwater. Academic Press, New York.

Mecham, J. S., M. J. Littlejohn, L. E. Brown, R. S. Oldham, and J. R. Brown. 1973. A new species of leopard frog (***Rana pipiens*** complex) from the plains of the central United States. Occasional Papers, Museum, Number 18, Texas Technical University, Lubbock, Texas.

Minckley, W. L., and G. F. Mefee. 1987. Differential selection by flooding in stream-fish communities of the arid American Southwest. Pages 93-104 *in* W. J. Matthews and D. C. Heins (eds.). Community and Evolutionary Ecology of North American Stream Fishes. Univ. Oklahoma Press, Norman, Oklahoma.

Moyle, P. B. 1973. Effects of introduced bullfrogs, *Rana catesbeiana*, on the native frogs of the San Joaquin Valley, California. Copeia 1973:18-22.

Ohmart, R. D. 1995. Ecological condition of the East Fork of the Gila River and selected tributaries: Gila National Forest, New Mexico. Pages 312-317 *in* D. W. Shaw and D. M. Finch (tech. coords.). Desired future conditions for Southwestern riparian ecosystems: bringing interests and concerns together. USDA Forest Service, General Technical Report RM-GTR-272.

Patterson, M. W. 1997. Forest fires and drought in the U.S. Southwest. 1997. *In* Impacts of Climate Change on Life and Ecosystems. http://geochange.er.usgs.gov/sw/impacts/biology/fires_drought/

Piotrowski, J. S., S. L. Annis, and J. E. Longcore. 2004. Physiology of *Batrachochytrium dendrobatidis,* a chytrid pathogen of amphibians. Mycologia 96(1):9-15.

Platz, J. E. 1993. *Rana subaquavocalis*, a remarkable new species of leopard frog (*Rana pipiens* Complex) from southeastern Arizona that calls under water. Journal of Herpetology 27(2):154-162.

Platz, J. E., and J. S. Frost. 1984. *Rana yavapaiensis*, a new species of leopard frog (*Rana pipiens* complex). Copeia 1984:940–948.

Platz, J. E., and T. Grudzien. 1999. The taxonomic status of leopard frogs from the Mogollon Rim country of central Arizona: evidence for recognition of a new species. Proceedings of Nebraska Academy of Sciences 109:51.

Platz, J. E., A. Lathrop, L. Hofbauer, and M. Vradenburg. 1997. Age distribution and longevity in the Ramsey Canyon leopard frog, *Rana subaquavocalis*. Journal of Herpetology 31(4):552-557.

Platz, J. E., and J. S. Mecham. 1979. ***Rana chiricahuensis***, a new species of leopard frog (***Rana pipiens*** complex) from Arizona. Copeia 1979:383–390.

Rinne, J. N., and W. L. Minckley. 1991. Native Fishes of Arid Lands: A Dwindling Natural Resource of the Desert Southwest. USDA Forest Service, General Technical Report RM-206: 1-45. Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.

Rorabaugh, J. C. 2008a. Plains leopard frog *Lithobates blairi.* http://www.reptilesofaz.org/Turtle-Amphibs-Subpages/h-l-blairi.html

Rorabaugh, J. C. 2008b. Chiricahua leopard frog *Lithobates ciricahuensis.* http://www.reptilesofaz.org/Turtle-Amphibs-Subpages/h-l-chiricahuensis.html

Rorabaugh, J.C. 2008. An introduction to the herpetofauna of mainland Sonora, México, with comments on conservation and management. Journal of the Arizona-Nevada Academy of Science 40(1):20-65.

Rosen, P.C. and S. H. Baker. *Manuscript*. Experimental studies of fish predation on bullfrog and leopard frog tadpoles: implications for southwestern leopard frog conservation.

Rosen, P. C., and C. R. Schwalbe. 1998. Using managed waters for conservation of threatened frogs. Pages 180-195 *in* J.M. Feller and D.S. Strouse (eds.), Environmental Economic, and Legal Issues Related to Rangeland Water Developments. The Center for the Study of Law, Science, and Technology, Arizona State University, Tempe, AZ.

Rosen, P. C., C. R. Schwalbe, D. A. Parizek, P. A. Holm, and C. H. Lowe. 1994. Introduced aquatic vertebrates in the Chiricahua region: effects on declining native ranid frogs. Pages 251-261 *in* L. F. DeBano, G. J. Gottfried, R. H. Hamre, C. B. Edminster, P. F. Ffolliott, and A. Ortega-Rubio (tech. coords.), Biodiversity and management of the Madrean Archipelago. USDA Forest Service, General Technical Report RM-GTR-264.

Rosen, P. C., C. R. Schwalbe, and S. S. Sartorius. 1996a. Decline of the Chiricahua leopard frog in Arizona mediated by introduced species. Report to Heritage program, Arizona Game and Fish Department, Phoenix, Arizona. IIPAM Project No. I92052.

Ross, D. A., J. K. Reaser, P. Kleeman, and D. L. Drake. 1999. *Rana luteiventris* (Columbia spotted frog). Mortality and site fidelity. Herpetological Review 30(3):163.

Savage, A.E., Sredl, M.J. and K. R. Zamudio. 2011. Disease dynamics vary spatially and temporally in a North American amphibian. Biol. Conserv. 144(6)1910-1950

Savage and Zamudo. 2011. MHC genotypes associate with resistance to a frog-killing fungus. PNAS vol. 108 (40): 16705–16710

Santos-Barrera, G., G. Hammerson, and M. J. Sredl. 2004. *Lithobates chricahuensis.* In: IUCN 2012. IUCN Red List of Threatened Species. Version 2012.2. http://www.iucnredlist.org/details/58575/0

Scott, N.J. 1993. Post-metamorphic death syndrome. Froglog 7:1-2.

Smith, H. M., and D. Chiszar. 2003. Distributional and variational data on the frogs of the genus *Rana* in Chihuahua, Mexico, including a new species. Bulletin of the Maryland Herpetological Society 39(3):59-66.

Snyder, J., T. Maret, and J. P. Collins. 1996. Exotic species and the distribution of native amphibians in the San Rafael Valley, AZ. Page 6 *in* abstracts of the Second Annual Meeting of the Southwestern United States Working Group of the Declining Amphibian Populations Task Force, Tucson, Arizona.

Southwest Center for Biological Diversity (SCBD). 1998. Petition to list Chiricahua leopard frog *Rana chiricahuensis* as an Endangered Species. Endangered Species Series No. 39. Tucson, Arizona. 28 pages. http://www.biologicaldiversity.org/species/amphibians/Chiricahua_leopard_frog/pdfs/petition-1998.pdf

Southwest Center for Biological Diversity (SCBD). Saving the Chiricahua leopard frog. http://www.biologicaldiversity.org/species/amphibians/Chiricahua_leopard_frog/ [accessed March 2013]

Southwest Regional Assessment Group. 2000. Preparing for a climate change; the potential for climate variability and change. Report to the U.S. Global Change Research Program. http://www.ispe.arizona.edu/research/swassess/pdf/complete.pdf.

Sredl, M. J., and J. M. Howland. 1994. Conservation and management of Madrean populations of the Chiricahua leopard frog, *Rana chiricahuensis*. Arizona Game and Fish Department, Nongame Branch, Phoenix, Arizona.

Sredl, M. J., J. M. Howland, J. E. Wallace, and L. S. Saylor. 1997. Status and distribution of Arizona's native ranid frogs. Pages 45-101 *in* M.J. Sredl (ed). Ranid Frog Conservation and Management. Arizona Game and Fish Department, Nongame and Endangered Wildlife Program, Technical Report 121. Phoenix, Arizona.

Sredl, M. J., and R. D. Jennings. 2005. *Rana chiricahuensis* Platz and Mecham, 1979, Chiricahua leopard frogs. Pages 546-549 *in* M.J. Lanoo (ed), Amphibian Declines: The Conservation Status of United States Amphibians. University of California Press, Berkeley, California. 1094 pp.

Sredl, M. J., and L. S. Saylor. 1998. Conservation and management zones and the role of earthern cattle tanks in conserving Arizona leopard frogs on large landscapes. Pages 211-225 *in* Proceedings of Symposium on Environmental, Economic, and Legal Issues Related to Rangeland Water Developments. November 13-15, 1997, Tempe, Arizona.

Stebbins, R. C. 2003. A Field Guide to Western Reptiles and Amphibians, Third Edition. Houghton Mifflin Company, Boston, Massachusetts.

Swan, Don. 2013. Effects of Wildland Fire on Lowland Leopard Frogs and their Habitat at Saguaro National Park. Recorded webinar epresentation. http://swfireconsortium.org/effects-of-wildland-fire-on-lowland-leopard-frogs-and-their-habitat-at-saguaro-national-park/

Swetnam, T. W., and C. H. Baisan. 1996. Fire histories of montane forests in the Madrean Borderlands. Pages 15-36 *in* P. F. Ffolliott et al. (Tech. Coord.), Effects of fire on Madrean Province ecosystems. USDA Forest Service, General Technical Report, RM-GTR-289.

Thompson, R. S., K. H. Anderson, and P. J. Bartlein. 1997. Assessment of potential future vegetation changes in the Southwestern United States. *In* Impacts of Climate Change on Life and Ecosystems. Http://geochange.er.usgs.gov/sw/impacts/biology/veg_chg_model/

Todd, B.D. and K.M. Andrews. 2008. Response of a reptile guild to forest harvesting. Conservation Biology 22(3):753-761.

U.S. Fish and Wildlife Service (USFWS). 2002. Endangered and Threatened Wildlife and Plants; Listing of the Chiricahua Leopard Frog (*Rana chiricahuensis*); Final Rule. 50 CFR Part 17 RIN 1018–AF41. Federal Register 67 (114): 40790-40811.

U.S. Fish and Wildlife Service (USFWS). 2007. Chiricahua Leopard Frog (*Rana chiricahuensis*) Recovery Plan. Region 2, U.S. Fish and Wildlife Service, Albuquerque, NM. 429 pp.

U.S. Fish and Wildlife Service (USFWS). 2011. Chiricahua Leopard Frog (*Lithobates* [=*Rana*] *chiricahuensis*). 5-Year Review. Phoenix, Arizona.

U.S. General Accounting Office. 1988. Public rangelands: some riparian areas restored by widespread improvement will be slow. GAO/RCED-88-105.

Van Devender, T. R. 1995. Desert Grassland History: Changing climates, evolution, biogeography, and community dynamics. Pp. 68-99 *in* M. P. McClaran and T. R. Van Devender (eds.), The Desert Grassland, University of Arizona Press, Tucson.

Van Devender, T. R., R. S. Felger, M. Fishbein, F. Molina-Freaner, J. J. Sánchez-Escalante, and A. L. Reina-Guerrero. 2010. Biodiversidad de las plantas vasculares. Pp. 229-262 *in* F. Molina-F. and T. R. Van Devender (eds.), Diversidad Biológica de Sonora, Universidad Nacional Autónoma de México, Hermosillo.

Watkins-Colwell, K. A., and G. J.Watkins-Colwell. 1998. Differential survivorship of larval leopard frogs (*Rana pipiens*) exposed to temporary and permanent reduced pH conditions. Bulletin of the Maryland Herpetological Society 7 34:64-75.

Zug, G. R., L. J. Vittand, and J. P. Caldwell. 2001. Herpetology: An Introductory Biology of Amphibians and Reptiles, 2nd edition. Academic Press, San Diego, California.

# Assemblages

## F-7 Grassland Birds

### F-7.1 Conceptual Model

#### F-7.1.1 Introduction

This assemblage is meant to represent grassland birds at a level of detail that will help managers understand how key change agents may broadly affect grassland birds and to inform potential management responses. Vickery et al. 2009 defines grassland birds as "any species that has become adapted to and reliant on some variety of grassland habitat for part or all of its life cycle." Understanding how grassland habitats contribute to the viability of grassland bird species, and how changes to that habitat may affect population heath are critical to developing effective management strategies. Many grassland bird species are undergoing steep, widespread and long-term population declines and the grasslands they depend on are increasingly being lost and degraded through agricultural conversion, conversion to exurban and urban landcover, desertification and shrub encroachment (NRCS 1999, Pool et al. 2012). Some grassland bird species are neotropical migrants, but many are short distance migrants that winter primarily in the United States and Mexico.

Grassland ecosystems in the Madrean Ecoregion have been recognized for their regional biological value, especially for grassland birds (Biodiversity Support Program et al. 1995). Grassland birds have experienced a more consistent and geographically widespread decline in North America than any other guild of birds (Knopf 1994). In the Madrean Ecoregion there are a diversity of birds associated with grassland habitats that depend on this habitat for different aspects of their life histories, including overwintering, migration stopover, and breeding. Generally, species recognized as sensitive or of conservation interest to state and federal agencies managing resources in the Madrean Ecoregion are birds that breed in grassland habitat. The following birds breed in the Madrean Ecoregion: Botteri's sparrow (*Peucaea botterii*), scaled quail (*Callipepla squamata*), western burrowing owl (*Athene cunicularia*), Aplomado falcon (*Falco femoralis*), northern harrier (*Circus cyaneus*), Arizona grasshopper sparrow (*Ammodramus savannarum ammolegus*), loggerhead shrike (*Lanius ludovicianus*), and masked bobwhite (*Colinus virginianus ridgwayi*).

Grassland birds that breed elsewhere but spend winter in the Madrean Ecoregion include Baird's Sparrow (*Ammodramus bairdii*), ferruginous hawk (*Buteo regalis*), and Sprague's pipit (*Anthus spragueii*). Baird's sparrows are concentrated in the Chihuhuan Desert of northern Mexico and the adjacent United States in the winter.

Species of conservation or management concern that are associated with healthy grasslands from the BLM Gila District (USDI-BLM 2010) include the Arizona Botteri's sparrow (*Peucaea botterii arizonae*), Arizona grasshopper sparrow (*Ammodramus savannarum ammolegus*), ferruginous hawk (*Buteo regalis*) (breeding population only), and western burrowing owl (*Athene cunicularia hypogaea*).

Additional grassland-associated birds in the region from lists compiled by Gori et al. (2012) include: Brewer's sparrow, Cassin's sparrow, chestnut-collared longspur, clay-colored sparrow, eastern meadowlark, western meadowlark, golden eagle, horned lark, lark bunting, lark sparrow, long-billed curlew, McCown's longspur, mountain plover, prairie falcon, sandhill crane, short-eared owl, and vesper sparrow.

Each grassland-associated bird has a unique set of habitat requirements and while there are some similarities, habitat management to benefit one specific species may not benefit other species. The actual assemblage of grassland birds will vary with physical habitat structure, disturbances and other factors. Patterns of occurrence for individual species and large populations of wintering grassland-associated birds are highly dynamic from year to year probably due to the variability in rainfall patterns and food production.

Arizona Botteri's sparrow (*Peucaea botterii arizonae*), scaled quail (*Callipepla squamata*), Arizona grasshopper sparrow (*Ammodramus savannarum ammolegus*), and Baird's sparrow (*Ammodramus bairdii)* were selected as examples to characterize and illustrate some key patterns in the habitat requirements and life histories of grassland-associated bird species. In the desert southwest region, all of these birds depend primarily on the semi-arid grassland habitats of the Madrean ecoregion. Additionally, the major change agents for grassland birds are described, drawing upon examples from the above four species, as well as other grassland bird species in the region.

## F-7.1.2 Individual Species Highlights

**NESTING SPECIES**

### *F.7.1.2.1 Botteri's sparrow (Peucaea botterii)*

**Distribution.** The range extends discontinuously from southeastern Arizona (Corman, in Corman and Wise-Gervais 2005), southwestern New Mexico, and extreme southern Texas southward to Nicaragua and Costa Rica (Howell and Webb 1995, Webb and Bock 2012).

In southeastern Arizona, the species is locally common on the Sonoita Plains, eastern Santa Cruz and southwestern Cochise counties, between the Santa Rita and Huachuca Mountains; current range limits are Buenos Aires National Wildlife Refuge, Pima County, to the west; foothills of Mount Fagan south of Vail in Pima County, and Pinery Creek southeast of Dos Cabezas, Cochise County, to the north (not detected near Pinery Creek by Arizona Breeding Bird atlas surveys); and Hay Hollow Wash north of MacDonald Peak, 10 kilometers west of the entrance to Guadalupe Canyon, Cochise County, to the east (Phillips et al. 1964, Monson and Phillips 1981,Webb 1985, Webb and Bock 2012). Elevational range in Arizona is mainly 1,082-1,585 meters (Corman, in Corman and Wise-Gervais 2005). In extreme southwestern New Mexico, small numbers have been reported in recent years in May–August in Hidalgo County (Webb and Bock 2012).

This species is migratory in the United States and northern Mexico, where the species is present primarily from late April-May to September-October (Howell and Webb 1995). Wintering areas of these northern populations are little known (Webb and Bock 2012).

**Habitat.** These sparrows occur in semi-arid grasslands, particularly those with taller grasses, often in swales, floodplains, periodically flooded lower canyon bottoms dominated by giant sacaton (*Sporabolis wrightii*) (Webb and Bock 1990). Also important is proximity to grassy hillsides that do not have as tall of grasses (Webb and Bock 1990). They are most abundant in pockets of healthy ungrazed or lightly grazed grasslands (Webb and Bock 2012). After being mostly extirpated from AZ due to overgrazing in the late 19th century, the species returned with grazing improvements and were then found primarily in sacaton bottoms which were probably the tallest grasses remaining (Ruth personal communication, 2014). Nesting occurs in dense, senescent stands of sacaton (often at upper edge of sacaton-dominated bottoms), and also in upland grasslands with very widely scattered low shrubs such as velvet mesquite (*Prosopis velutina*), whitethorn acacia (*Acasia constricta*) , catclaw acacia (*Acasia gregii*), ocotillo

(*Fouquieria splendens*), and low cacti, and sometimes they nest in rolling grasslands intermixed with open stands of evergreen oaks (Corman, in Corman and Wise-Gervais 2005). In upland habitats, this species is associated with the thickest and tallest grasses, regardless of plant species (Bock and Bock, 1992). On the Audubon Research Ranch, south of Elgin, Arizona, Bock and Bock (1988) and Jones and Bock (2005) found a preference for stands of exotic lovegrass that included Lehmann and Boer, perhaps because at least one of these speices is creaing habitat that is structurally more similar to sacaton habitat than are other upland grasslands dominated by native species. However, it has been observed by Ruth (2014) in the last 10 years of work in the grasslands of southeastern Arizona that Botteri's sparrows are now found more commonly in upland native grasses than they were 10 years ago. They are still more common in sacaton but as grass condition has improved overall, they are using these uplands more frequently (Ruth personal communication, 2014).

Nests are on the ground and generally well concealed by a thick overhanging grass clump (Webb 1985). Foraging occurs mainly in open grassy sites where grasshoppers are more visible (Webb and Bock 1990). Bock et al. (2008) described Botteri's Sparrow response to exurban development noting that nests were usually built on the ground under or deep inside individual grass clumps, while birds in sacaton habitat sometimes nested up to 0.5 m off the ground in tall grass plants (Jones and Bock 2005).

**Diet.** In summer, Botteri's sparrows eat primarily insects, especially Orthoptera, and also seeds (Webb and Bock 2012).

**Reproduction.** In Arizona, nesting peaks between late July and mid-August, with clutches recorded from mid-June to late August (Webb 1985), and nestlings sometimes present into late September (Webb and Bock 1996). Young leave the nest when approximately 10 days old, and adults feed fledglings for 4-5 weeks (Webb 1985).

**Populations and Status.** Prior to the 1900s, this species was more widely distributed in southeastern Arizona (Monson and Phillips 1981). Extended drought and human-caused habitat changes (see Change Agents) have reduced the species' distribution and abundance. Today, U.S. populations of this species are locally abundant, rebound from moderate habitat disturbances, and occupy a variety of grassland types; the species appears stable despite its restricted range (Webb and Bock 2012).

#### *F.7.1.2.2 Grasshopper sparrow (Ammodramus savannarum)*

**Distribution.** This is a broadly distributed grassland bird found in appropriate grasslands throughout southern Canada, the U.S. and south into Mexico, Central America, and the Carribean(Vickery 1996, AOU 1998). There are four subspecies that breed in the U.S. and the subspecies in southern Mexico, Central America, and the Carribbean are resident (non-migratory). The main population is in the Great Plains, from North Dakota south to northern Texas, and east to Illinois (Johnson et al. 1998). The birds breeding in Arizona, New Mexico, Sonoran and Chihuahua populations are a separate subspecies (*Ammodramus savannarum ammolegus*).

The nonbreeding range extends from central California, southern Arizona, southern New Mexico, Texas, central Missouri, Tennessee, and North Carolina south through Mexico and Central America to northern Costa Rica, including the Bahamas and Cuba (Vickery 1996, AOU 1998).

This species is resident in Veracruz, Chiapas, Guatemala, Belize, Honduras, northwestern Costa Rica, Panama, Jamaica, Hispaniola, Puerto Rico, western Colombia, western Ecuador, and the Netherlands Antilles (Vickery 1996, AOU 1998).

In the Madrean ecoregion, the Arizona subspecies nests in southeastern Arizona (e.g., San Bernardino, Babocomari, San Rafael, Sulphur Springs, and Altar valleys, Sonoita Plains, extreme upper San Pedro River drainage), southwestern New Mexico (Williams 2004), and adjacent northern Sonora and Chihuahua, primarily at elevations between 1,000 and 1,600 meters. The Sonoita and San Rafael valleys in Arizona and the Animas Valley in New Mexico are the primary population centers (Ruth 2008). Some of the Arizona subspecies remain in winter, whereas many likely move south into Sonora for the winter (Phillips et al. 1964), especially if grass cover and food resources are scarce (Corman, in Corman and Wise-Gervais 2005). In winter the other subspecies, *perpallidus,* that breeds in the Great Plains is commonly found in the Madrean Ecoregion.

**Habitat.** Breeding habitat generally consists of grasslands with large expanses of intermediate height grass, often with clumped vegetation interspersed with patches of bare ground, and with moderately deep litter and sparse coverage of low woody vegetation (Vickery 1996; Corman, in Corman and Wise-Gervais 2005). Nests are placed on the ground and usually are well hidden at the base of a grass clump, forb, or shrub (Vickery 1996). ). Strong (1988) described the preferred habitat as having grass cover of 42-60 percent, shrub canopy cover of 1-8 percent, and an average grass height of 5 to 20 cm.

Nonbreeding habitats include grass-dominated fields, native prairie (Florida), and grazed pastures (Mexico and Belize) (Vickery 1996

These sparrows forage exclusively on the ground; exposed bare ground is critical for effective foraging (Vickery 1996).

**Diet.** Grasshopper sparrows eat mostly insects (especially grasshopper) in summer. In winter, the diet is primarily seeds (Vickery 1996).
**Reproduction.** In Arizona, most nesting occurs during the typical monsoon season. Adults carrying food to nestlings have been observed from late July to late August (Corman, in Corman and Wise-Gervais 2005).

**Populations and Status.** Breeding Bird Survey data (https://www.pwrc.usgs.gov/bbs/) through 2011 indicates a general declining trend in the western United States. BBS data for the Madrean ecoregion grasslands are too limited for reliable trend determination. Ruth (2008) found that grasshopper sparrows had the largest populations in San Rafael, Sonoita, and Animas valleys, with small remnant populations in all other valleys which is consistent with findings 20-25 years ago (Mills 1982, Strong 1988). There was no evidence that distribution had declined in recent decades. Ruth (2008) also found that there were no significant differences in the mean number of singing males from 2004-2005 suggesting that abundance has not declined in recent decades. In New Mexico, numbers reported in 1992 in the Animas and Playas valleys were 109 and 41, respectively; in 2004, corresponding numbers in the Animas and Playas valleys were 21 and 0 (Williams 2004).

The grasshopper sparrow subspecies (*ammolegus*) is listed as a Bird of Conservation Concern by USFWS in Region 2.

### *F.7.1.2.3 Scaled quail (Callipepla squamata)*

**Distribution.** Scaled quail are resident (nonmigratory) from eastern Arizona, east-central Colorado, and southwestern Kansas south through western Oklahoma, western Texas, and interior Mexico to northeastern Jalisco, Guanajuato, Queretaro, Hidalgo and western Tamaulipas, with a large part of the

range in northeastern Sonora, Chihuahua, Durango, and south to Hidalgo; introduced populations exist in central Washington and eastern Nevada (Howell and Webb 1995, AOU 1998, Dabbert et al. 2009).

**Habitat.** Habitat includes dry open grasslands with scattered low-growing shrubs such as burroweed, snakeweed, cacti, and mesquite, with plenty of open ground (Wise-Gervais, in Corman and Wise-Gervais 2005). In southern Arizona, 89 percent of sightings occurred in mesquite grassland, mixed shrubland, and shrub-dominated washes (Medina 1988).

In Arizona, these quails inhabit open plains, rolling hills, and low ridges and mesas, in open semi-arid grasslands consisting of perennial bunchgrasses scattered with low shrubs and cacti primarily at elevations between 853 and 2134 meters (Brown 1989). Occupied grasslands have a shrub component, as long it is not too dense (Medina 1988, Guthery et al. 2001). King (1998) found that inhabited areas were dominated by perennial bunchgrasses, with 10% woody cover, whereas Bristow and Ockenfels (2006) reported use of areas with ≤ 10% woody cover, ≥ 26% grass cover, and relatively high grass species richness (birds were absent in areas containing ≥ 20% woody cover). In southeastern Arizona, optimum habitat contains low-growing grasses, forbs, and shrubs with a ground cover between 10 and 50% (Goodwin and Hungerford 1977).

In New Mexico, this species prefers shrub-grass communities, using shrubland as available and avoiding grasslands devoid of shrubs; generally it is associated with mesa dropseed, a bunchgrass, three awn, honey mesquite, and broom snakeweed, avoiding sites dominated by stoloniferous black. In the Chihuahuan Desert in New Mexico, a mixture of mid- and late-seral conditions supported the greatest densities of quail (Smith et al. 1996, Nelson et al. 1997, Saiwana et al. 1998, Joseph et al. 2003).

Nests are on the ground, usually in protected, shaded sites in dense vegetation (Dabbert et al. 2009). In New Mexico and Arizona, quail nested under yucca (*Yucca elata*) and small bushes and occasionally in agricultural fields (Bendire 1892). In New Mexico, 66 percent of 14 nests were located in dead Russian-thistle, mixed forbs, soapweed, johnson grass (*Sorghum halepense*), and overhanging rocks (Russell 1932).

**Diet.** The diet includes seeds of forbs, shrubs, and grain; forb seeds and grains main fall and winter food, with insects and herbaceous leaves consumed on a seasonal basis (Dabbert et al. 2009).

**Reproduction.** Nesting generally begins in late April and May, but in dry years may be postponed until the summer monsoon begins in July or August (Brown 1989). Clutches averaging around 13 eggs are incubated for 22-23 days. Downy young have been seen as late as early October (Brown 1989).

**Populations and Status.** Scaled quail appear to be a “boom and bust” species that exhibits wide fluctuations in abundance (Dabbert et al. 2009). Breeding Bird Survey data (https://www.pwrc.usgs.gov/bbs/) through 2011 suggest that declines have occurred in the Arizona-New Mexico region since 1966 and during the most recent decade. Scaled Quail is a WatchList species of continental concern in the Partners in Flight North American Landbird Conservation Plan (Rich et al. 2004).

**WINTERING SPECIES**

### *F.7.1.2.4 Baird's sparrow (Ammodramus bairdii)*

**Distribution.** The breeding range extends from southern Alberta, southern Saskatchewan, and southern Manitoba south to central and eastern Montana, North Dakota, northwestern and north-central South Dakota, and possibly western Minnesota, with unconfirmed records or possible nesting in Wyoming, Wisconsin, and extreme western Ontario (Green et al. 2002). The nonbreeding range extends from southeastern Arizona (Sonoita Plains, Altar and San Rafael valleys, base of Chiricahua, Huachuca, Santa Rita, and Patagonia mountains; Monson and Phillips 1981, Gordon 2000a), southern New Mexico (casual; Otero Mesa, Animas Valley; Hubbard 1978, Howell and Webb 1995, Green et al. 2002), and southwestern Texas south to northeastern Sonora, extreme northwestern Chihuahua, northeastern Durango, and extreme northern Zacatecas (Green et al. 2002).

Baird's Sparrows typically arrive in the Madrean ecoregion between September and mid-October and remain through early April.

**Habitat.** Nesting habitat (which is outside the Madrean ecoregion) includes ungrazed or lightly grazed mixed-grass prairie, prairie with scattered low bushes and matted vegetation (AOU 1998), local pockets of tallgrass prairie, wet meadows, and some types of disturbed habitats.

Nonbreeding habitat includes open grasslands and overgrown fields (AOU 1998). Specific nonbreeding habitat requirements are little known (Green et al. 2002). These sparrows are found mainly in dense stands of grass, usually in extensive grasslands, with a minor shrub component (Green et al. 2002). The birds seem to prefer areas of taller, denser grass, including grasslands dominated by several species of bunchgrasses (*Bouteloua* spp. and *Eragrostris* spp.) and few mesquite (*Prosopis velutina*) as only woody vegetation more than 1 meter in height (Gordon 2000a). They can also be found on south-facing slopes of mixed-oak grassland where the oaks are on the north slope. Baird's sparrows forage on the ground among grass clumps, usually well hidden from view (Green et al. 2002).

**Diet.** During the breeding season, the diet is dominated by various insects (Lane 1968). On the winter range in the Madrean ecoregion, the diet includes primarily the seeds of various grasses and forbs.

**Reproduction.** This species does not breed in the Madrean ecoregion. Most nesting occurs May-July (Green et al. 2002).

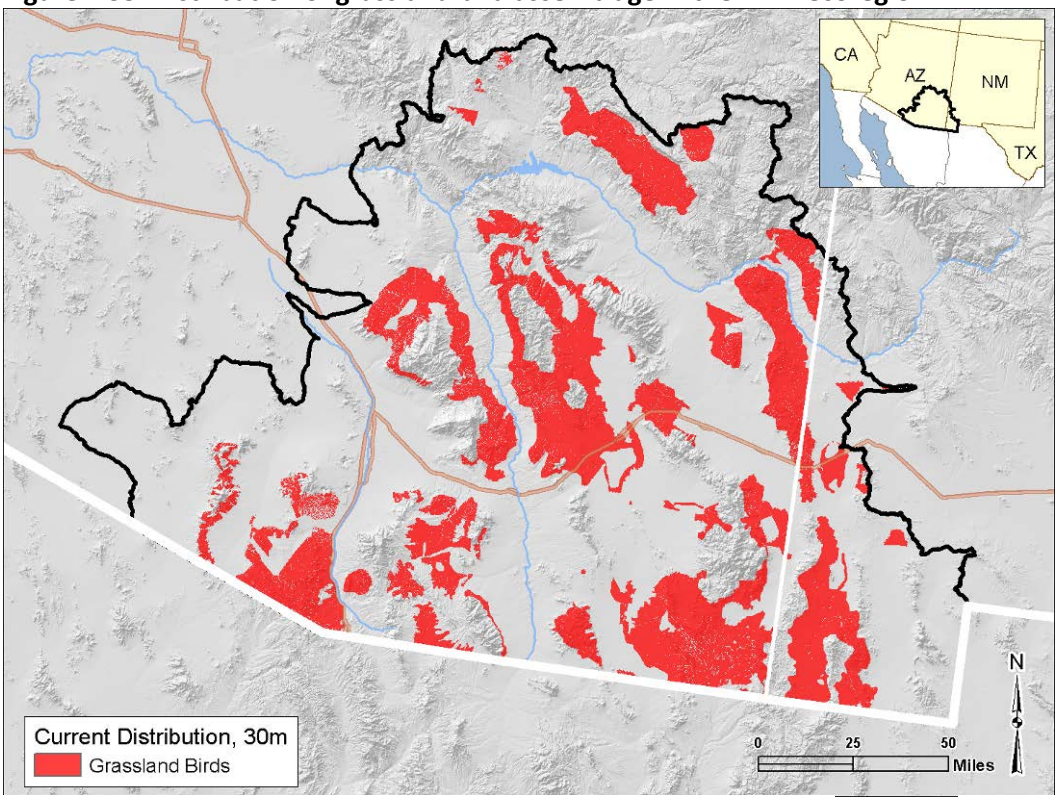
**Populations and Status.** In the breeding range, conversion of native prairie to cropland and exotic vegetation, invasion of native grasslands by exotic plant species, proliferation of shrubs due to fire suppression in moist portions of the range, and poor range management of some remaining tracts have greatly reduced Baird's sparrow populations from presettlement numbers (see Green et al. 2002). Breeding Bird Survey data (https://www.pwrc.usgs.gov/bbs/) indicate a significant decline since 1966 but an increase in abundance over the past decade (2001-2011). Christmas Bird Count data (http://birds.audubon.org/christmas-bird-count) in the winter range show low numbers of detections in Arizona and New Mexico, with no strong trend evident since 1966 or more recently.

## F-7.1.3 Overview of Grassland Bird Habitats

Semi-arid grasslands occur in Arizona, New Mexico, and Texas and southward through Mexico. Most semi-arid grassland habitat is encompassed by privately owned cattle ranches. Some grassland habitats in the Madrean ecoregion are protected in reserves and experimental ranges, but these preserved areas

represent a small percentage of the area dominated by semi-arid grasslands (Pool et al. 2012). For example, only about 8 percent of semi-desert grasslands in the U.S. portion of the Chihuahuan Desert is under some level of protection (Pool et al. 2012). It must also be noted that just because grasslands area is protected by a reserve, it is not necessarily pristine or functioning.

**Figure F-35. Distribution of grassland bird assemblage in the MAR ecoregion.**




Each grassland bird species has a unique set of habitat requirements (Table F-11. For example, Botteri's sparrows favor thick stands of tall grass whereas horned larks depend on expanses of short grass. Morrison (1999) found that in semi-desert grasslands of Arizona and New Mexico, woody plant cover strongly influences the presence and relative abundance of particular bird species. Changes in woody cover may cause recognizable shifts in bird assemblages at some level. Levels of woody cover <10% appeared to support sensitive grassland species such as grasshopper sparrows and chestnut- collared longspurs, and levels >10% support a different and more diverse set of species (Morrison 1999). Habitat conditions and grassland management favorable to one species may exclude other species.

The actual assemblage of grassland birds in a given area varies with physical habitat structure and surrounding landscape charateristics, such as:

- grassland vegetation density and growth form [short, medium, tall], and the presence or absence of woody vegetation (as described above) and other non-grassland features;
- the extent of unbroken grassland habitat; operative disturbance regimes;
- and surrounding land use.

Some grassland birds are specialists with very specific habiat needs. Areas that support a more diverse grassland habitat may have a lot of grassland generalist species that can use a wider range of habitats. However, within grassland habitat, a diverse array of growth forms and successional stages would support different species of grassland birds. Such diversity depends on natural processes that create a patchwork of different types of vegetation across a large landscape.

**Table F-11. Examples of habitat characteristics for grassland bird species.**

| Species | Habitat Characteristics |
|---|---|
| Grasshopper sparrow (*Ammodramus savannarum*) | nearly 75% ungrazed grass cover averaging 30 cm in height, 5% woody cover, and approximately 20% bare ground (SE AZ; Bock and Webb 1984).<br><br>grass cover of 42-60%, shrub cover of 1-8% percent, and average grass height of 5 to 20 cm (Strong 1988). |
| Botteri's sparrow (*Peucaea botterii*) | semi-arid grasslands, taller grasses, in swales, floodplains, periodically flooded lower canyon bottoms dominated by giant sacaton (Webb and Bock 1990).<br><br>most abundant in pockets of healthy ungrazed or lightly grazed grasslands (Webb and Bock 2012).<br><br>nesting: dense, senescent stands of sacaton often at upper-edge of sacaton-dominated bottoms), and upland grasslands with very widely scattered low shrubs; or in rolling grasslands intermixed with open stands of evergreen oaks (Corman, in Corman and Wise-Gervais 2005).<br><br>in uplands is associated with the thickest and tallest grass, regardless of species (Bock and Bock 1992), including monotypic stands of non-native Lehmann lovegrass (Bock and Bock 1988) (structurally more similar to sacaton habitat than are other upland grasslands dominated by native species). |
| Scaled quail (*Callipepla squamata)* | perennial bunchgrasses with scattered low shrubs and cacti (Brown 1989). Typically occupied grasslands have a shrub component, as long it is not too dense (Medina 1988,Guthery et al. 2001).<br><br>dominated by perennial bunchgrasses, with 10% woody cover (King 1998), ≤ 10% woody cover, ≥ 26% grass cover, and relatively high grass species richness (birds were absent in areas containing ≥ 20% woody cover) (Bristow and Ockenfels 2006)<br><br>optimum habitat contains low-growing grasses, forbs, and shrubs with a ground cover between 10 and 50% (SE AZ; Goodwin and Hungerford 1977).<br><br>shrub-grass communities, using shrubland as available and avoiding grasslands devoid of shrubs; avoiding sites dominated by stoloniferous black grama (NM; Smith et al. 1996, Saiwana et al. 1998, Nelson et al. 1999, Joseph et al. 2003). |
| Baird's sparrow (*Ammodramus bairdii* | only winter [nonbreeding] in the Madrean Ecoregion; open grasslands and overgrown fields (AOU 1998).<br><br>mainly in dense stands of grass, usually in extensive grasslands, with a minor shrub component (AZ; Green et al. 2002).<br><br>seem to prefer areas of taller, denser grass, including grasslands dominated by several species of bunchgrasses with few mesquite as only woody vegetation more than 1 meter in height (Gordon 2000a).<br><br>forage on the ground among grass clumps, usually well hidden from view (Green et al. 2002). |

| Species | Habitat Characteristics |
|---|---|
| Masked Bobwhite (*Colinus virginianus ridgwayi*) | open savanna grassland located within river bottoms, level valleys and plains (Kuvlesky et. al. 1997).<br><br>requires high humidity for breeding, typically delays nesting until the late summer monsoon period (Corman and Wise-Gervais 2005). |
| Loggerhead Shrike (*Lanius ludovicianus*) | tall woody plants are key for wintering areas, therefore shrub removal for grassland restoration may adversely affect this species unless groups of tall woody plants are retained (>2m height) which provide perching sites for hunting (Pool et al. 2012). |
| Western Burrowing Owl (*Athene cunicularia hypogaea*) | Associated with abundant burrowing mammals or other burrowing species of appropriate size (especially prairie dogs; also ground squirrels, badgers, marmots, skunks, armadillos, large species of kangaroo rats, or burrowing tortoises) (Haug et al. 1993)<br><br>dry, open, short-grass, treeless plains, steppes, deserts, prairies (Haug et al. 1993)<br><br>vegetation height within 50m of nest burrow best if <15cm<br><br>quantitative vegetative characteristics of nest sites are variable locally, but e.g., % bare ground tends to be high, often >40%, and % grass and other vegetative cover tends to be low, often <40% for grass cover (see Green 1983, MacCracken et al. 1985, Green and Anthony 1989). |

This pattern is affected by annual and long-term climate variations. In the semi-arid grasslands of the Madrean ecoregion, the distribution and abundance of particular bird species are highly dynamic from year to year in accordance with variations in reproductive effort and success that in turn reflect variations in rainfall patterns and productivity of bird food resources (Pool et al. 2012). Long-term climate change exerts an effect through changes in habitat structure and function (e.g., disturbance regimes). It is important to note that grasslands that have low biodiversity may support unique species.

Overall, adequate cover of undisturbed grasslands for nesting and foraging is among the most influential factors affecting grassland bird populations (NRCS 1999). Generally, large blocks of undisturbed grassland can support breeding and feeding behaviors as well as provide winter and migration cover; to support an array of grassland-nesting bird species within an area, NRCS (1999) recommends the presence of contiguous grassland blocks of at least 500 acres in size.

Geographic variation in these habitat factors results in some specific areas in the ecoregion being richer in grassland biodiveristy than other areas. For example, the Huachuca Mountains Grassland Valley Complex and the Sierra San Luis/Peloncillo Mountains, straddling the U.S. - Mexico border, were ranked first and second among conservation areas for species richness and irreplaceability in the Apache Highlands Ecoregional Assessment (Marshall et al. 2004).

### F-7.1.4 Change Agent Characteristics

Change agents for grassland birds do not affect species in a discrete way. The factors that affect bird populations interact in various ways. For example, livestock grazing, fire exclusion, and drought often have interacted in exerting their effects on grassland birds. Livestock grazing and prescribed burning of grasslands often are integrated activities on some rangelands in the Madrean ecoregion. Also, invasion of non-native plant species may affect fire regimes. However, from a plant perspective, fire and grazing

appear to affect species relatively independently (i.e., some species tend to be affected mainly by fire whereas others are more responsive to grazing) (Valone and Kelt 1999).

Grassland habitats in the Madrean Archipelago ecoregion have experienced significant habitat conversion or have lost the basic ecological processes (disturbance regimes) that sustained them. Fire suppression and the removal of native grazers (e.g., black-tailed prairie dog), appear to be particularly important. Many decades of grazing by domestic livestock have altered grassland condition, though light levels of grazing are compatible with grassland bird conservation. Some generalist bird species, and those tolerant of barren or shrub-encroached landscapes, persist in anthropogenic grasslands, but many grassland specialists have declined in the altered landscapes. Grassland birds are increasingly dependent on human management for habitat health. Together with adequate protection and favorable management of grasslands, annual and long-term variation in precipitation patterns plays a major role in the determining the distribution, abundance, and population trends of grassland birds.

#### *F.7.1.4.1 "Permanent" Habitat Loss and Fragmentation*

Some habitat has been converted to intensive farming or urban/residential use and provides little or no habitat for grassland birds. For example, the favored sacaton habitat of Botteri's sparrow was reduced and fragmented during the twentieth century in part as a result of farming and human settlement (Webb and Bock 1990, 2012). In Arizona some privately owned grassland landscapes are being rapidly lost or fragmented as they are converted to vineyards, other row crop agriculture, ranchettes, and other exurban development (Corman, in Corman and Wise-Gervais 2005; Ruth 2008). Some of the former U.S. breeding range of the aplomado falcon (e.g., Animas Valley, New Mexico) has been detrimentally altered by conversion of grassland to farmland.

#### *F.7.1.4.2 Shrub Encroachment*

Declines of scaled quail distribution and abundance in grasslands in southeastern Arizona have been attributed to increases in shrubby vegetation or complete loss of native grass species (Rea 1973; Wise-Gervais, in Corman and Wise-Gervais 2005). Mesquite invasion in grasslands in southeastern Arizona has eliminated breeding populations of grasshopper sparrow in affected areas (Lloyd et al. 1998). In the same region Cassin's sparrows are more common in areas with reduced mesquite density (Maurer 1986) however shrub cover is an important compoenet of Cassin's Sparrow habitat and they are not found in grasslands without it (Bock and Webb, 1984, Ruth, J.M. 2000).
Morrison (1999) and Block and Morrison (2010) found that bird assemblages in semi-desert grasslands in southeastern Arizona and southwestern New Mexico changed greatly as woody plant cover increased; some grassland birds such as grasshopper sparrow (summer), chestnut-collared longspur (winter), and horned lark (both seasons) occurred in greatest numbers where woody plant cover (mainly mesquite and snakeweed) was less than 1 percent, whereas the richness of sparrow species during winter was highest among sites with 6-15 percent woody plant cover.

Some of the former U.S. breeding range of the aplomado falcon in the Madrean ecoregion has been detrimentally altered by an increase in mesquite. Habitat changes have directly impacted the falcon and some (e.g., degradation of grassland habitat by excessive cattle grazing) also have resulted in reductions in its avian prey populations. However, this falcon appears historically to have been a very rare or possibly irregular breeder in southern Arizona and New Mexico (Keddy-Hector 2000), so the degree to which habitat changes have affected it in this region are somewhat speculative.

Studies in Texas and New Mexico indicate that range improvements designed to promote grasses by brush control may remove important food sources for scaled quail (e.g., mesquite, snakeweed, broomweed; Ault and Stormer 1983, Davis et al. 1975, Leif and Smith 1993). An increase in perennial grass cover in the Chihuahuan Desert region of New Mexico between 1961 and 1992 corresponded with a decline in the scaled quail population (Saiwana et al. 1998).

#### *F.7.1.4.3 Habitat Alteration and Livestock Management*

In the semi-arid grasslands of the Madrean ecoregion, overgrazing and drought are thought to have caused a major reduction in grass cover that was accompanied by an increase in shrub density (especially of mesquite) and a reduction in fire frequency resulting from removal of fine fuels that carry fire (Chew 1982, Humphrey 1974, Pyne 1984, Brown and Archer 1989, Swetnam and Baisan 1996). Overgrazing or inappropriate grazing regimes are recognized as a threat to bird species that are associated with semi-desert grassland in the Madrean ecoregion (Latta et al. 1999).

The causes of recent and long-term declines in scaled quail populations in the southwestern part of their distribution (Breeding Bird Survey data) are uncertain but are likely the result of habitat degradation, particularly overgrazing (Dabbert et al. 2009). In Arizona, Bock and Bock (1988) found more scaled quail on grazed sites compared with ungrazed sites, but overgrazing reduces residual grass cover, which is important to overwinter survival of scaled quail (Brown 1978). Grazing on early- and mid-seral condition rangelands appears not to be compatible with supporting scaled quail populations (Saiwana et al. 1998). Grazing during drought conditions may result in loss of cover needed to support quail populations (Saiwana et al. 1998, Joseph et al. 2003, Nelson et al. 1997).

The favored sacaton habitat of the Botteri’s sparrow was reduced and fragmented during the twentieth century primarily as a result of overgrazing and trampling by cattle; water diversion and flood channelization associated with agricultural land use also contributed to the decline (Webb and Bock 1990, 2012). Botteri’s sparrows tolerate moderate cattle grazing (Webb and Bock 1996).

Limited data suggest that Baird’s sparrows have some tolerance of grazing. Gordon (2000b) found higher abundance in summer-grazed pasture than in a nearby area that had been idle for 30 years. However, this species elsewhere shows a preference for minimally grazed areas, and it may be absent from areas receiving more than a moderate amount of grazing (Green et al. 2002).

In one study, grazing on semi-arid grasslands in Arizona was found to exclude grasshopper sparrows (Bock and Webb 1984). However vibrant populations of breeding grasshopper sparrows have been studied on Las Cienegas National Conservation areas for the last 5 years where grazing occurs.In southeastern Arizona, Cassin's sparrow and grasshopper sparrow are common in grasslands from which livestock grazing has been excluded for two decades, but uncommon on heavily grazed pastures (Bock and Bock 1988); Cassin’s Sparrows are most abundant in ungrazed and lightly grazed grasslands with canopy coverage in excess of 50 percent (Bock and Webb 1984; Bock et al. 1986; Bock and Bock, 1988, 1992, 1999). Ruth (2008) notes that effects of grazing vary widely depending on the particular circumstances and there is no reason to conclude that well-managed grazing is incompatible with grasshopper sparrow.

#### *F.7.1.4.4 Planting and Invasion of Non-native Forage Species*

Roadsides are dispersal corridors for non-native grasses and other non-native plant species. These land uses and infrastructure features have facilitated the introduction and past and on-going spread of invasive species into the grasslands of the Madrean Ecoregion. In response to erosion and demand for

cattle forage, species such as Lehmann's lovegrass (*Eragrostis lehmanniana*) and Boer lovegrass (*Eragrostis curvula*) were widely introduced into grasslands in southeastern Arizona and adjacent Sonora, MX, often at the expense of native bunch grasses. As a result, the spread of non-native perennial grasses within grasslands in this ecoregion has been substantial. As documented by Gori et al. (2012), Lehmann's lovegrass and, to a lesser extent, Boer lovegrass are common on at least 1.5 million acres in this ecoregion; non-native grasslands with little to moderate woody increase now comprise 11% of the area's current and former grasslands. Other non-native grasses common in semi-desert grassland include fountaingrass (*Pennisetum setaceum)* and Natalgrass (*Melinis repens* ssp*. repens)*, which have spread more slowly (Van Devender et al. 2007). Buffelgrass (*Pennisetum ciliare*) and soft feather pappusgrass (*Enneapogon cenchroides*) will probably invade further into desert grassland with warmer climates as native mesquites and succulents increase. Native grasses are still present in many invaded areas, but at a lower density.

These non-native forage and erosion control species can negatively affect grassland birds. For example, African lovegrasses (*Eragrostis* sp.) have been planted in an effort to restore defoliated ranges (Cable 1971, Freeman 1979). Scaled quail avoid stands of Lehmann's lovegrass, an exotic species planted extensively in Arizona and dominant at mid- to high elevations (Medina 1988). Compared to ungrazed native grasslands in southeastern Arizona, plantations of Lehmann's lovegrass are ornithologically impoverished, probably because they produce fewer seeds and insects (Bock and Bock 1988). In this study, total birds counted were about twice as many on native transects as compared to exotic transects. For example, Cassin's sparrows breed more abundantly in native Arizona grasslands than in grasslands dominated by exotic lovegrasses (*Eragrostis* spp.) (Bock and Webb 1984, Bock et al. 1986, Bock and Bock, 1988, 1992, 1999). Reduced abundance of birds in unburned grasslands dominated by exotic lovegrasses may be related to thick accumulation of dead grass litter, which may inhibit foraging (Bock and Bock 1992).

#### *F.7.1.4.5 Rodent Control on Rangelands*

Keystone species of mammals (primarily black-tailed prairie dogs in the Madrean Archipelago ecoregion) historically played a role in maintaining open habitats. Prairie dogs constantly clip vegetation to maintain an unobstructed view of the landscape and thereby help prevent the invasion of mesquite and other shrubs into semi-arid grasslands (Koford 1958, Van Auken 2000, Askins et al. 2007). Hence, the eradication of prairie dogs from southeastern Arizona and southwestern New Mexico through poisoning and habitat conversion by humans likely played a significant role in grassland conversion and loss, and has likely affected a number of songbird species that depend on open grasslands. Additionally, burrowing owls—which do not actually burrow—are dependent on burrowing mammals for creation of suitable nest sites, and declines in burrowing owl populations in Madrean semi-arid grasslands certainly are at least in part a result of prairie dog extirpation.

#### *F.7.1.4.6 Fire*

In semi-arid grasslands, fire plays a mixed role for grassland birds, reducing cover in the short term and improving grassland conditions by reducing the density of woody species in the long run.

Bock and Bock (1988) found that fire had no effect on scaled quail numbers in a sacaton (*Sporobolus wrightii*) grassland in southeastern Arizona. Fall counts of scaled quail on burned and unburned grasslands were similar.

Botteri's sparrows tolerate prescribed burning (Webb and Bock 1996). Webb and Bock (2012) noted that most remnant sacaton stands are on private ranchlands, and that ranchers traditionally burn or mow sacaton fields annually in late winter (Feb) (to encourage new growth) and graze them in spring–summer (1 May–15 July) when upland grasses are still dormant (Cox and Morton 1986). Fire is suppressed at the margins of sacaton bottoms to prevent loss of upland grasses, leaving narrow strips of peripheral sacaton habitat intact. Because sparrow nests are concentrated at the edge of sacaton bottoms, only part of the colony is evicted during the first postfire breeding season; by second season, sufficient plant biomass is established to support moderate-sized colonies (Bock and Bock 1978, Webb and Bock 1990).

In Arizona, short-term declines in sparrow numbers may occur following prescribed burning in a mesquite-invaded semi-desert grassland (Kirkpatrick et al. 2002). Botteri's and Grasshopper sparrows will avoid burned areas for one to two growing seasons post fire because of their requirement for heavy cover, but will re-colonize as cover regenerates (Bock and Bock 1982). Cassin's sparrow were most abundant in unburned native grasslands with 50% canopy cover, avoiding burned areas for two years post-fire (Bock and Webb 1984, Bock and Bock 1992).

Small breeding populations of grasshopper sparrows in New Mexico are threatened by shrub encroachment and spread of invasive species that may be a consequence of drought, heavy grazing, and a non-natural regime of fire and fire suppression (New Mexico Partners in Flight; http://nmpartnersinflight.org/grasshoppersparrow.html).

#### *F.7.1.4.7 Drought*

Drought undoubtedly affects scaled quail populations at their range margins in southeastern Arizona (Wise-Gervais, in Corman and Wise-Gervais 2005). Population lows may be attributable primarily to widespread reproductive failures, possibly the result of inadequate rainfall and a lack of succulent foods (Bailey 1928, Wallmo and Uzzell 1958, Campbell et al. 1973, Brown 1989).

The primary factors posing threats to the future of grasshopper sparrows in the region appear to be loss and/or degradation of habitat due to exurban development, overgrazing, and the effects of long-term drought (Ruth 2008).

Based on their study in southeastern Arizona, Bock and Bock (1999) concluded that, due to loss of food and cover, the combined effects of drought and high-density, short-duration grazing left the land in a substantially denuded condition and had a strong negative effect on populations of ground-foraging, seed-eating birds wintering in the study area. Furthermore, one year of moderate rainfall and reduced stocking density only partially ameliorated these effects.

#### *F.7.1.4.8 Disturbance (Nondestructive Intrusion)*

Webb (1985) found that low-flying military aircraft, especially helicopters, and vehicular road traffic disturbed Botteri's sparrow females at nests; birds either flushed repeatedly or delayed returning for up to 1 hour following disturbance. Attendant females exhibited intense alarm responses to bird blinds set up in nest vicinity by abandoning incubation for >1 hour and chipping repeatedly.

### F-7.1.5 Summary

Although Botteri's sparrow, Arizona grasshopper sparrow, scaled quail and Baird's sparrow have differing habitat requirements, a common habitat need for all of them is relatively dense stands of native grass species with a limited woody cover component. The highest recorded tolerance of woody

cover was recorded for scaled quail and it is noted birds were absent in areas containing ≥ 20% woody cover (Bristow and Ockenfels 2006). Generally Botteri's, Baird's and Arizona grasshopper sparrows share a preference for dense stands of taller grasses. The current literature does not thoroughly define "dense" although for Grasshopper sparrow a lower level threshold of 42% grass cover was noted by Strong (1988).

The intensity and distribution of cattle grazing can lead to changes in grassland structure, diversity, density, and forage quality. Changes may be further influenced by precipitation amount and timing.

Connected grassland habitats with different structural characteristics and with surrounding non-grassland habitats may be an important focus for grassland-associated species. Many grassland-associated bird species rely on different habitat types for different components of their life history. The diversity of habitat structure needs represented by this group of four grassland-associated bird species, points to the need to provide a diversity of grassland habitats distributed across the Madrean ecoregion. It is important to note that different combinations of focal grassland-associated species and management objectives will lead to different key components of grassland habitat and structure.

### F-7.1.6 Ecological Status: Key Ecological Attributes and Indicators

This section of the conceptual model addresses Key Ecological Attributes and their potential indicators. The ecological status is a way of describing current status via criteria, functionality, or levels of attributes and asks if they are within the normal range of variation.

#### *F.7.1.6.1 Key Ecological Attributes*

Table F-12 identifies the key ecological attributes for this CE within the Madrean Archipelago ecoregion. A **key ecological attribute** of a focal ecological resource is a characteristic of the resource's biology, ecology, or physical environment that is critical to the resource's persistence in the face of both natural and human-caused disturbance. Alteration of such a characteristic beyond some critical range of variation will lead to the degradation or loss of the resource within decades or less. The KEAs table lists the identified key ecological attributes, with a brief definition, a rationale for why it is important for the CE, and a listing of stressors or change agents that might be affecting the key attribute.

**Table F-12. Key Ecological Attributes (KEA) of the grassland bird assemblage in the Madrean Archipelago ecoregion.** Indicators for these KEAs can be used to determine the ecological status for this CE; see **Table F-2** for a list of the indicators assessed in this REA.

| ***KEA Class:*** **KEA Name** | **Definition** | **Rationale** | **Stressors** |
|---|---|---|---|
| ***Landscape Context:*** **Habitat condition and extent** | This attribute is the amount of unbroken blocks or contiguous patches of grassland habitat that is suitable for grassland bird overwintering and breeding. | Adequate cover of undisturbed grasslands is among the greatest factors affecting grassland bird populations (NRCS 1999). Grassland bird communities display an array of habitat needs. Of the diverse habitat needs of grassland birds examined in this model, the overall extent and intactness of all grassland habitat types is an important indicator for the assemblage. | Continued loss and conversion of grassland breeding and nesting habitats is the largest threat to most grassland birds (NRCS 1999). Factors that affect habitat condition and extent of habitat include human development in the form of urban and suburban development and agriculture, suppression of natural fire regimes, and livestock grazing and management. These stressors can destroy or alter suitable habitat. |
| ***Landscape Context:*** **Diversity of habitat structure** | This is the diversity of habitat features within a block of semi-desert grassland habitat including grass height, interspersion of non-grass habitat features such as woody shrubs or trees, cacti or bare ground. | Wintering grassland bird communities display high variability and composition from winter to winter and show low site fidelity in wintering grounds, highlighting the need to provide habitat distributed across the area of interest (Pool et al. 2012) | Stressors include conversion and loss of habitat due to development, agriculture and shrub encroachment, and reduction of habitat diversity due to livestock grazing. |
| ***Landscape Context:*** **Intact Ecological Processes** | The presence and intactness of ecological processes that drive grassland habitat structure including presence of native grazers and fire return interval. | The structure of grassland habitat including woody cover, grass species composition, grass height and other features is critical to supporting all aspects of life history for a variety of grassland-associated birds. These processes contribute to control of total woody cover and habitat renewal and diversity. | In desert grasslands in the Madrean Ecoregion native grazers including the black-tailed prarie dog are no longer present and fire regimes have been altered by human suppression and management and by changes in grassland fuel load due in part to grazing practices. |
| ***Size/ Extent:*** **Population size & demographics** | This attribute is the abundance of populations of grassland birds as well as the abundance of individuals at a given breeding site. | Population size and demographic structure are direct indicators of the viability and health of the population, and can affect genetic diversity. | Drought, pesticide use and competition with other species that are more tolerant of human caused changes are factors (Yosef 1996). |

| *KEA Class:* KEA Name | Definition | Rationale | Stressors |
|---|---|---|---|
| ***Abiotic Condition:* Precipitation Regime** | This attribute is the amount of precipitation received during the winter and summer seasons in addition to the mean annual precipitation. | The mean annual and seasonal precipitation and fluctuations in them are important to grassland habitat productivity and, in combination with increasing temperatures may drive shrub/tree invasion of grasslands or grassland invasion of woodland habitat. | Decreased winter precipitation would lengthen the arid foresummer with related increases in fires and greater post-fire erosion. Changes in the arrival of monsoon season may adversely affect the phenology of a variety of grassland birds whose breeding is associated with the onset of monsoon and/or the wetness of winter. This may also drive changes in grassland composition that will alter what assemblage of species is supported in a given area. For example grasshopper sparrows and horned larks prefer habitat with <10% shrub cover (Morris 1999). Masked bobwhite are extremely sensitive to fluctuations in precipitation (Camou-Luders et al. 1999). |
| ***Abiotic Condition:* Temperature Regime** | This attribute is the mean annual temperature, seasonal average temperatures, and extreme highs and lows. | | Increased temperatures coupled with changes in precipitation could lead to loss of and change in composition of grassland habitats; as well as temperature stress during breeding and nesting. |

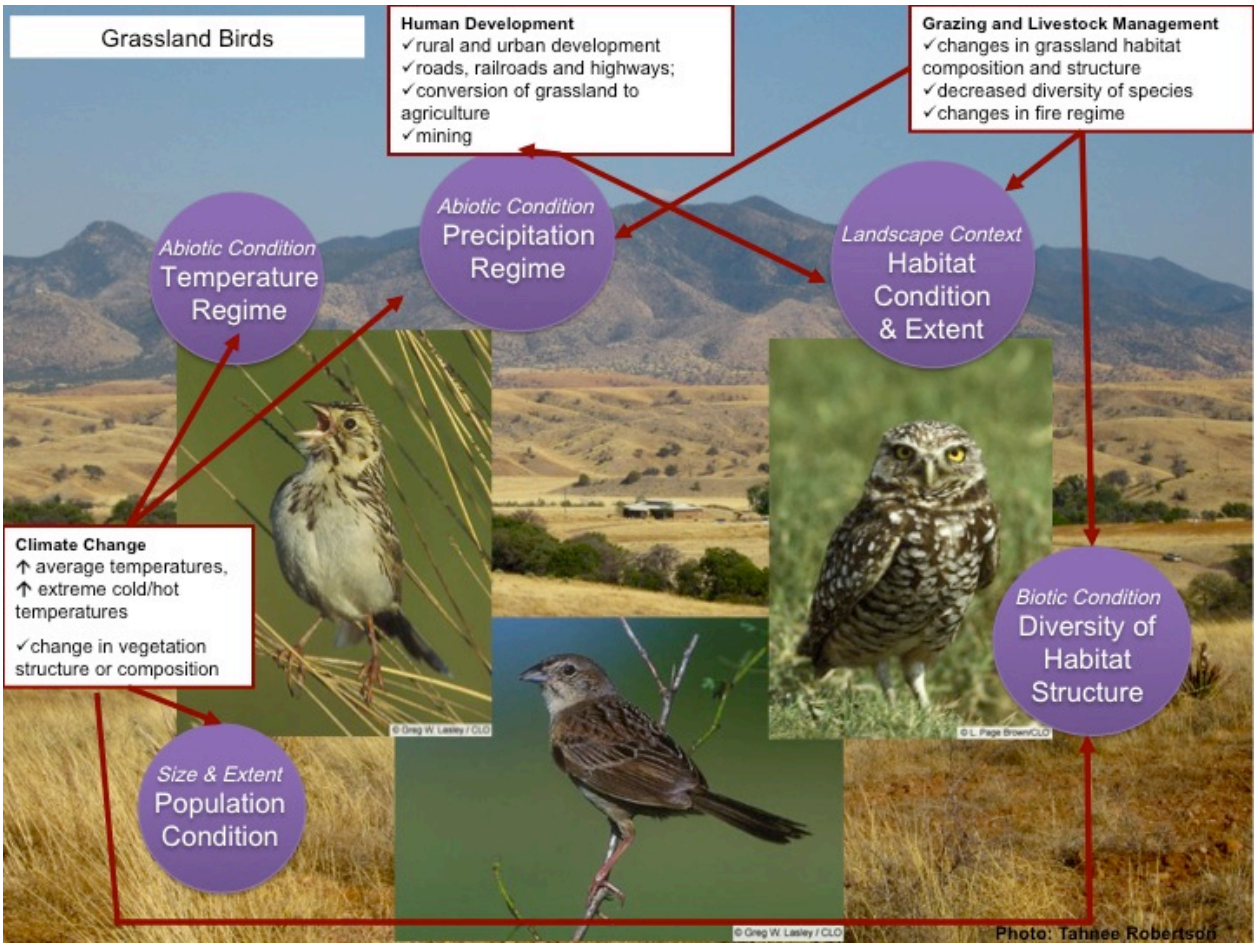
### F-7.1.7 Conceptual Model Diagram

A conceptual model diagram for the grassland bird assemblage provides a visual summary representation of the Key Ecological Attributes and the species comprising the assemblage (Figure F-36).

**Figure F-36. Conceptual model diagram for the grassland bird assemblage, showing key ecological attributes**(by class) for this assemblage of species, and indicating each KEA's influence, positive or negativedepending on the KEA's condition.

Inter-relationship between stressors and key ecological attributes

Key Ecological Attributes




## F-7.2 Considerations and Limitations

The species included in the grassland bird assemblage require a range of habitat structure. In order to best represent the assemblage, distribution and model inputs were designed to have a high level of generality. Consequently, the distribution and ecological status assessment results are landscape-level – that is, they can inform management decisions at the landscape level, but specific species considerations should be taken into account for more site-specific management decisions. Although habitat patch size may be an import limiting factor for grassland birds (NRCS 1999), patch size was not addressed in the analysis because the assessment team did not want to limit the understanding of ecological status or eliminate areas from consideration for management.

## F-7.3 Ecological Status Assessment Results and Interpretation

This section of the CE summary presents and discusses the results of the ecological status assessment for the grassland bird assemblage. It addresses each indicator separately, and then addresses the overall assessment, which integrates the results of all individual indicators assessed for the CE. The results are presented using a common framework, in which the status of an indicator – or the combination of all indicators – is scored on a scale from 0.0 to 1.0, where 0.0 indicates a condition of complete replacement of reference ecological conditions due to the impacts of stressors, and 1.0 indicates a condition of no alteration of reference ecological conditions. The same yellow-to-blue color ramp is used for all results, with yellow representing low scores, green moderate scores, and dark blues high scores.

### F-7.3.1 Current Ecological Status: Development, Fire Regime, Invasives

The results of the assessments for the three individual indicators for the KEAs for the grassland bird assemblage are shown in the three maps in Figure F-37 below.

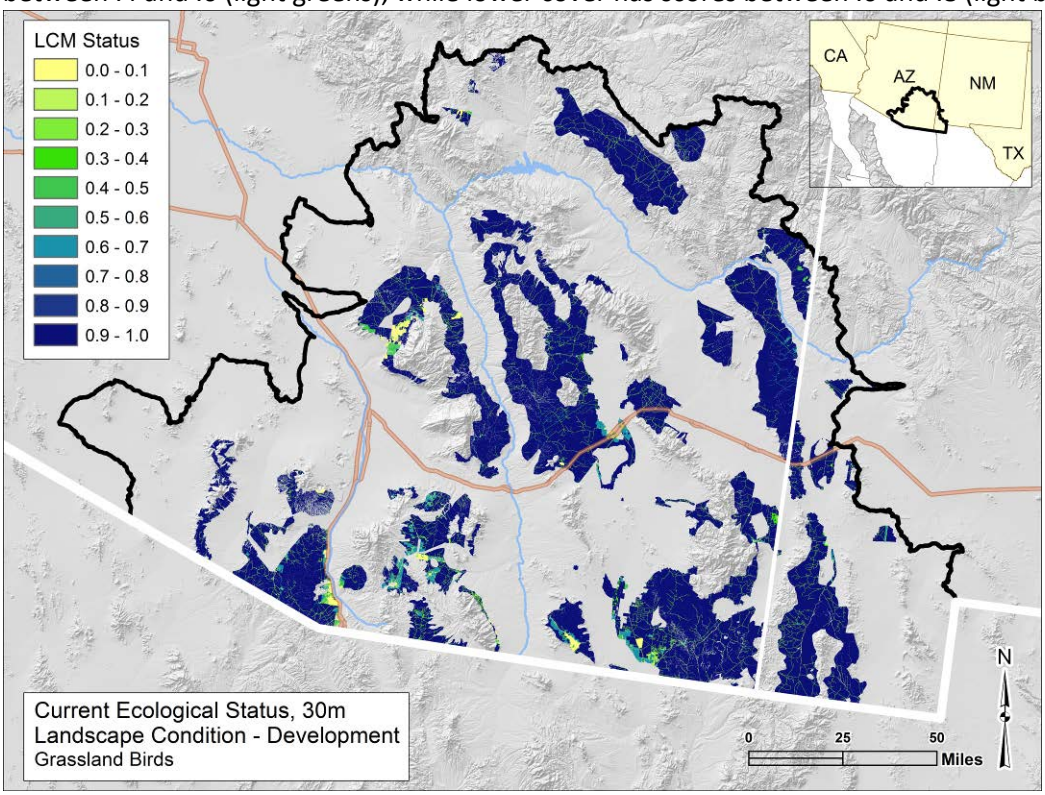
The development indicator results shown in the first map of Figure F-37 reflect a number of large but discrete areas and corridors of intense development (yellows and greens on the map) throughout the ecoregion, representing areas of municipal and agricultural development. These areas of lower development scores in part reflect the presence of communities such as Bisbee, Fort Huachuca, Oro Valley, Portal, and Rio Rico; and corridors associated with interstate highways 10 and 19 and many other larger roads. However, most of the habitat for grassland birds in the MAR ecoregion is experiencing little or no impact (dark blue) from the development features reflected in the landscape condition indicator. Where development is already present or on-going, the primary impact is direct loss of grassland habitat for this species assemblage.

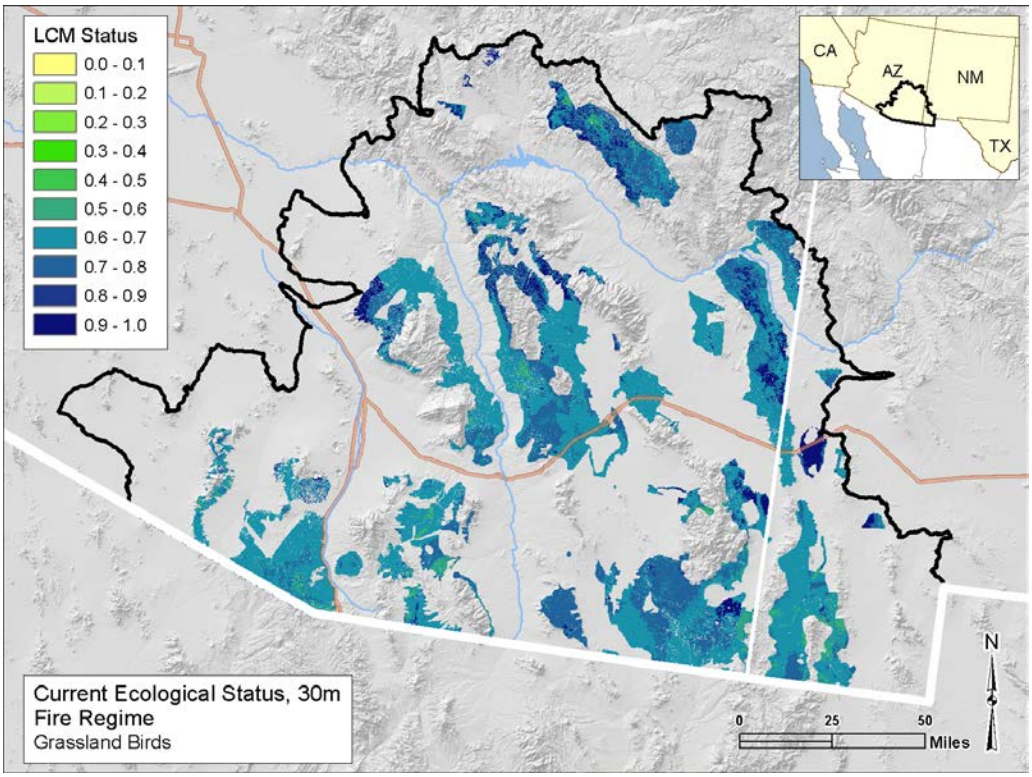
As illustrated in the second map of Figure F-37, much of the habitat for grassland birds is in severe ecological departure in relation to fire regimes throughout the ecoregion. Areas with moderate departure are small and often restricted to higher elevation grasslands. These results are consistent with research documenting the results of fire exclusion in the MAR ecoregion. Historically, the semi-desert grasslands of this ecoregion burned frequently; they were maintained as open grasslands with low shrub cover by fire return intervals of 2.5 to 10 years (Brown and Archer 1999, McPherson 1995, Robinett 1994, Wright 1980). Active and passive fire suppression over the last century has excluded fire from much of this grassland habitat (Gori and Enquist 2003, Schussman 2006a), creating an uncharacteristic fire regime that allows encroachment by woody species (Barton 1999, Gori and Enquist 2003, Muldavin et al. 2002, Turner et al. 2003). This altered fire regime is a key contributor to grasslands becoming dominated by woody vegetation and eventually converted to shrublands or woodlands; conversion to juniper woodlands or mesquite or creosotebush shrublands is common when trees or mesquite exceed 15% cover (Gori and Enquist 2003). While fire can have short-term negative impacts on some grassland birds, fire suppression has overall resulted in loss or degradation of large areas of grassland habitat for grassland bird species.

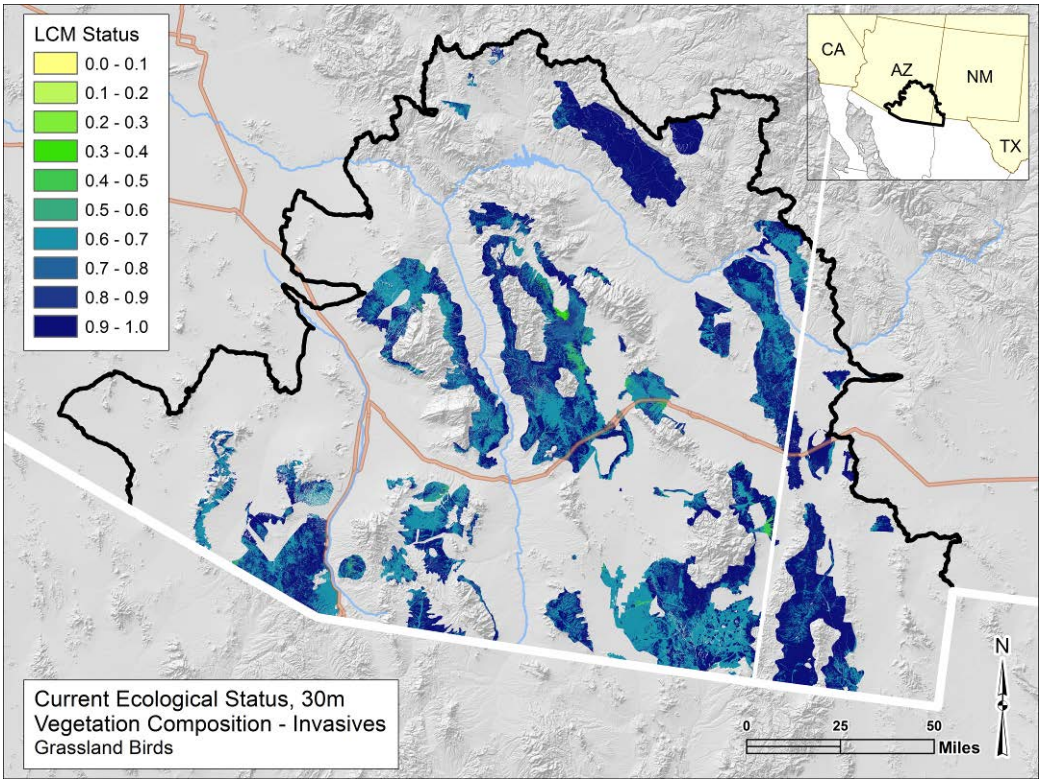
The third map of Figure F-37 indicates moderate (>10 -15%) to high (>25%) cover of exotic grasses and forbs or invasive mesquite in the grassland habitat of the ecoregion. There are significant areas with low or no cover of invasive species in small patches often at higher elevations, and in large patches in Natanes Plateau and Mountains in the north-central part of the ecoregion, as well as the boot heel of New Mexico. The introduction of two invasive non-native, perennial grasses, Lehmann and Boer lovegrasses (*Eragrostis lehmanniana* and *Eragrostis curvula*) has greatly impacted many semi-desert grasslands in this ecoregion (Anable et al. 1992, Cable 1971, Gori and Enquist 2003). Areas with high

cover of **both** non-native grasses and forbs and invasive mesquite are indicated in the green patches east of the Galiuro Mountains around the central part of the ecoregion. There are also fairly large areas with high cover of **either** non-native grasses and forbs, **or** mesquite, distributed throughout most of the ecoregion; these areas have scores between 0.6 and 0.7 and appear in turquoise blue. For grassland birds, the most significant impact of the different categories of invasive species that were assessed is the change in vegetation structure where native woody increasers (mesquite) have invaded and increased in density. However, different grassland bird species have different responses to changes in shrub cover in grasslands, based on their habitat preferences.

**Figure F-37. Scores for three indicators for grassland bird assemblage: development indicator (1st map), fire regime departure indicator (2nd map), and invasive species indicator (3rd map) for each 30m pixel.** Yellow (equivalent to 0) indicates high impacts from the CA, dark blue (equivalent to 1) indicates little to no impact from the CA. At the ecoregion scale, many development features are not readily visible (e.g., secondary roads or highways, railroads, small agricultural fields). Only 3 classes of fire regime condition are scored: no to little departure (dark blue), moderate departure and severe departure (lighter blues). For invasives, higher cover of mesquite or invasive exotics have scores between .4 and .6 (light greens), while lower cover has scores between .6 and .8 (light blues).





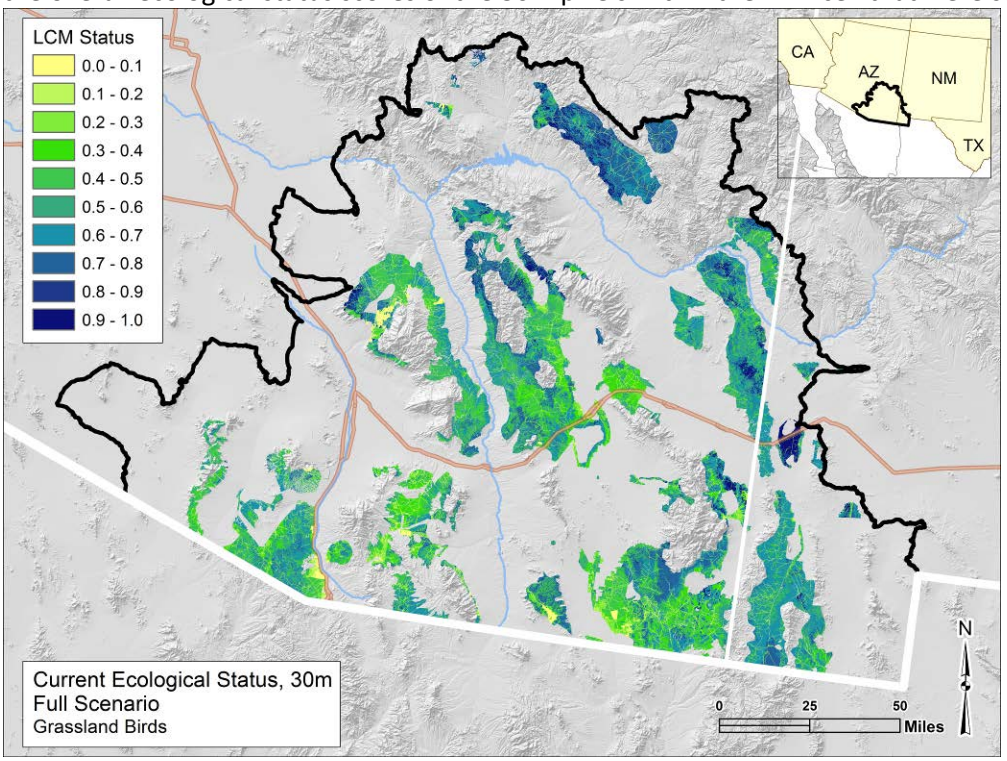

### F-7.3.2 Current Ecological Status: Full Scenario

The results of the three individual status indicators were combined to get an overall ecological status score, per pixel of the CE's distribution, as shown in the first map of Figure F-38 below. The overall, per-

pixel status scores for the CE were then averaged across each of the 4 km grid cells, as shown in the second map of Figure F-38 below.

When the three sets of indicator scores are aggregated to obtain the overall ecological status scores (as shown in the first map of Figure F-38), much of the habitat for this species is in moderate lower condition, with status scores ranging from 0.4 to 0.7 in many areas, and some discrete areas as low as the 0.1 range (Figure F-39). In visually comparing the maps of scores for the three individual indicators (Figure F-37) to the map of the overall status scores, the cumulative impacts of the change agents reflected by the indicators are clear. The altered fire regime and invasives cover are both driving down scores in much of the ecoregion, and scores drop even lower where development is also present. These same patterns are reflected when the overall status scores are averaged across each 4 km reporting unit, as shown in the second map of Figure F-38 above. Based on the indicators assessed in this REA, changes in grassland quality and vegetative structure from altered fire regime and encroachment by native woody increasers are having the most widespread effects on habitat quality for grassland birds. Direct habitat loss (from existing development) is much more localized, but where it occurs, it may result in grassland species being unable to occupy those areas (e.g., urban development, agriculture).

**Figure F-38. Overall ecological status scores for grassland bird assemblage for all three indicators combined (development, fire regime and invasives) for each 30m pixel (top) and 4km grid cells (bottom).** Yellow scores (equivalent to 0) indicate high impacts from the CAs, dark blue (equivalent to 1) indicate little to no impact from the CAs. In the second map, the score for each 4km cell is an average of the overall ecological status scores of the 30m pixels within the 4km cell that were scored for the CE.



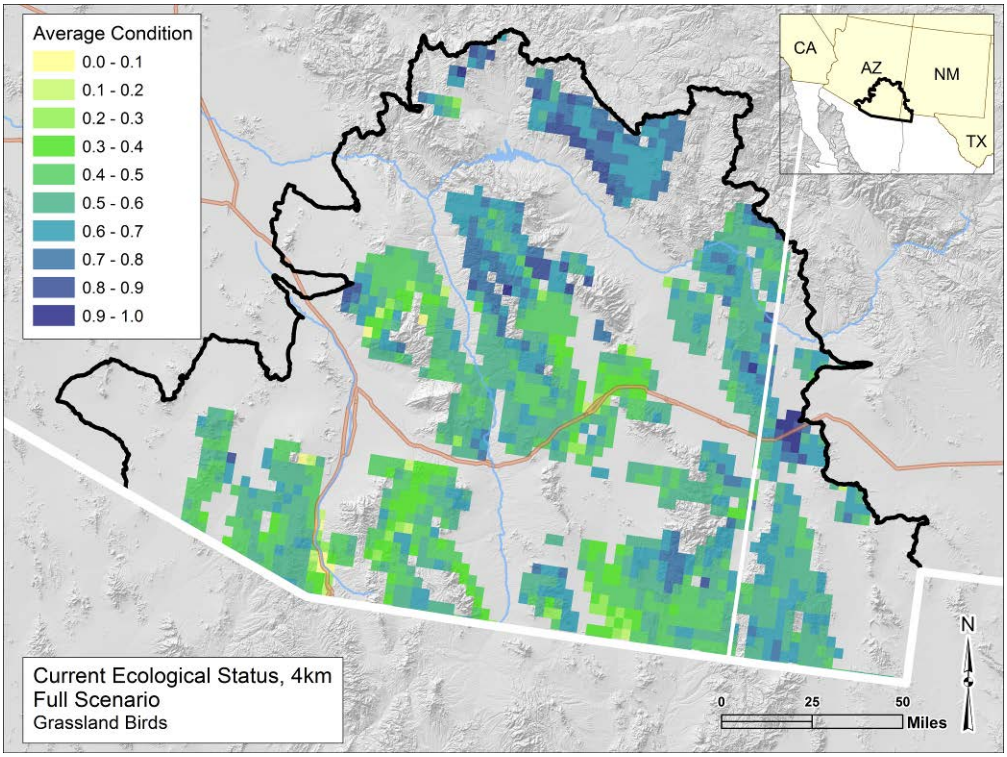
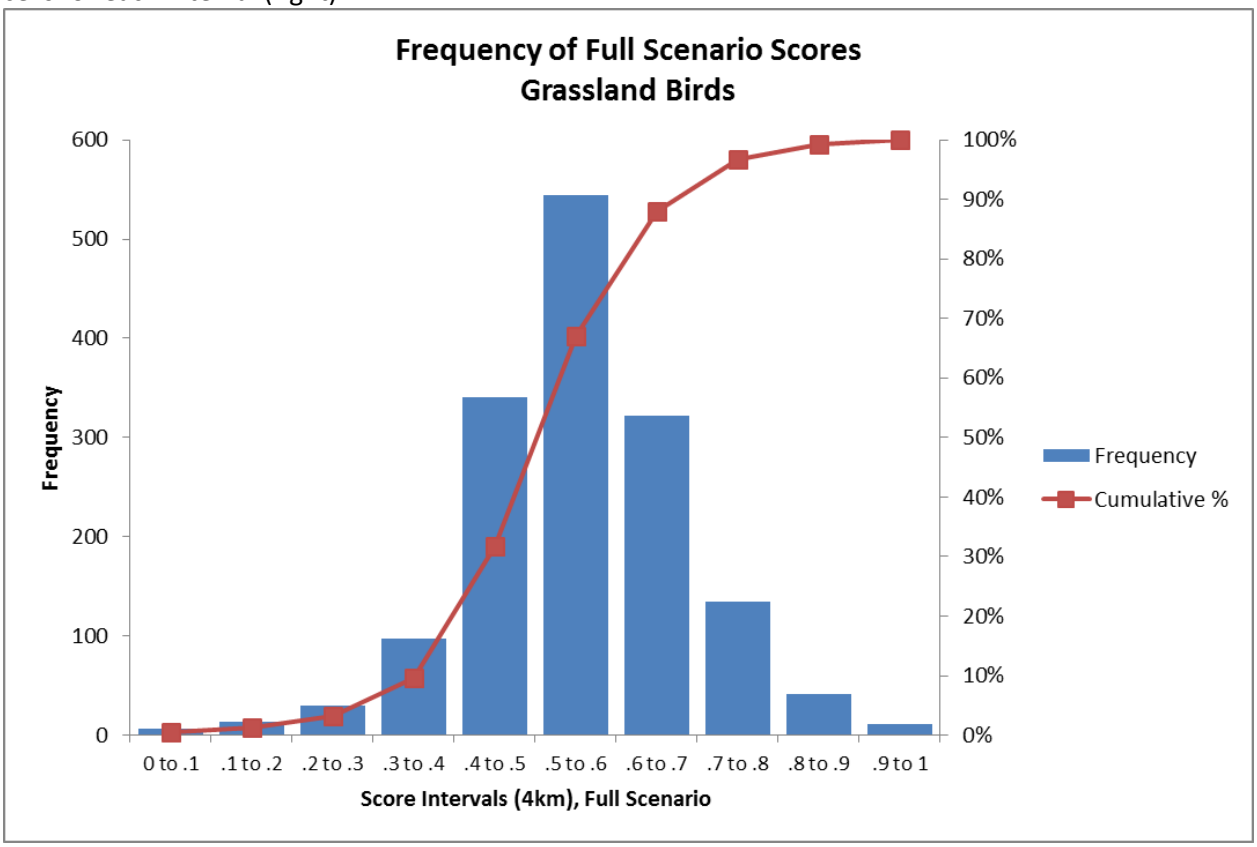



**Figure F-39. Frequency distribution of the 4km ecological status scores for the grassland bird assemblage, with cumulative percent.** The x-axis represents the 0.1 increment scoring intervals, while the y-axis shows the number of grid cells in each interval (left) and the cumulative percentage of the grid cells for each interval (right).




## F-7.4 References for the CE


American Ornithologists' Union (AOU). 1998. Check-list of North American birds. Seventh edition. American Ornithologists' Union, Washington, D.C.

Anable, M. E., M. P. McClaran, and G. B. Ruyle, 1992. Spread of introduced Lehmann lovegrass *Eragrostis lehmanniana* Nees. in southern Arizona, USA. Biological Conservation, 61, 181-188.

Askins, R. A., F. Chavez-Ramirez, B. C. Dale, C. A. Haas, J. R. Herkert, F. L. Knopf, and P. D. Vickery. 2007. Conservation of grassland birds in North America: understanding ecological processes in different regions. Ornithological Monographs 64:1–46.

Ault, S. C. and F. A. Stormer. 1983. Seasonal food selection by scaled quail in northwest Texas. Journal of Wildlife Management 47:222-228.

Bailey, V. 1928. A hybrid Scaled x Gambel's quail from New Mexico. Auk 45:210.

Bendire, C. 1892. Life histories of North American birds, with special reference to their breeding, habits, and eggs. U. S. National Museum Special Bulletin No. I:viii-416.

Best, T. L., T. E. Garrison, and C. G. Schmitt. 1992. Ingestion of lead pellets by scaled quail and bobwhite quail in southeastern New Mexico. Texas Journal of Science 44:99-107.

Block, G., and M. L. Morrison. 2010. Large-scale effects on bird assemblages in desert grasslands. Western north American Naturalist 70:19-25.

Bock, C. E., and J. H. Bock. 1978. Response of birds, small mammals, and vegetation to burning sacaton grasslands in southeastern Arizona. Journal of Range Management. 31(4): 296-300.

Bock, C. E., and J. H. Bock. 1988. Grassland birds in southeastern Arizona: impacts of fire, grazing, and alien vegetation. Pages 43–58 in International Council on Bird Preservation. Technical Report 7, Cambridge, England.

Bock, C. E., and J. H. Bock. 1992. Response of birds to wildfire in native versus exotic Arizona grassland. Southwestern Naturalist. 37(1): 73-81.

Bock, C. E., and J. H. Bock. 1999. Response of winter birds to drought and short-duration grazing in southeastern Arizona. Conservation Biology 13: 1117-1123.

Bock, C. E., J. H. Bock, K. L. Jepson, and J. C. Ortega. 1986. Ecological effects of planting African lovegrasses in Arizona. National Geographic Research 2:456-463.

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

Bock, C.E., Z.F. Jones, and J.H. Bock. 2008. The oasis effect: response of birds to exurban development in a southwestern savanna. Ecological Applications 18:1093-1106.

Bock, C. E., and B. Webb. 1984. Birds as grazing indicator species in southeastern Arizona. J. Wildlife Management 48: 1045-1049.

Brennan, L. A., and W. P. Kuvlesky, Jr. 2005. North American grassland birds: an unfolding conservation crisis? Journal of Wildlife Management 69:1-13.

Bristow, K. D. and R. A. Ockenfels. 2006. Fall and winter habitat use by Scaled Quail in southeastern Arizona. Rangeland Ecology & Management 59(3):308-313.

Brown, D. E. 1978. Grazing, grassland cover and gamebirds. Transactions of the North American Wildlife Conference 53:477-485.

Brown, D. E. 1989. Arizona game birds. Univ. Arizona Press, Tucson.

Brown, J. R., and S. Archer. 1989. Woody plant invasion of grasslands: establishment of honey mesquite (*Prosopis glandulosa* var. *glandulosa*) on sites differing in herbaceous biomass and grazing history. Oecologia 80:19-26.

Cable, D. R. 1971. Lehmann lovegrass on the Santa Rita Experimental Range, 1937-1968. Journal of Range Management 24:17-21

Campbell, H. 1950. Quail picking up lead shot. Journal of Wildlife Management 14:243-244.

Campbell, H., D. K. Martin, P. E. Ferkovich, and B. K. Harris. 1973. Effects of hunting and some other environmental factors on Scaled Quail in New Mexico. Wildlife Monographs 34.

Chew, R. M. 1982. Changes in herbaceous and suffrutescent perennials in grazed and ungrazed desertified grassland in southeastern Arizona, 1958-1978. American Midland Naturalist 108: 159-169.

Corman, T. E. and C. Wise-Gervais. Arizona Breeding Bird Atlas. 2005. University of New Mexico Press. Albuquerque, New Mexico. 636 pp.

Dabbert, C. B., G. Pleasant, and S. D. Schemnitz. 2009. Scaled quail (*Callipepla squamata*). The Birds of North America Online (A. Poole, editor). Ithaca: Cornell Lab of Ornithology. Retrieved from the Birds of North America Online: <http://bna.birds.cornell.edu/bna/species/106>

Davis, C. A., R. C. Barkley, and W. C. Haussamen. 1975. Scaled quail foods in southeastern New Mexico. Journal of Wildlife Management 39:496-502.

Davis, W. A. and S. M. Russell. 1995. Finding birds in southeast Arizona.Tucson Audubon Society. Tucson, AZ.

Dinerstein, E., D. Olson, J. Atchley, C. Loucks, S. Contreras-Balderas, R. Abell, E. Inigo, E. Enkerlin, C. Williams, and G. Castilleja. 2000. Ecoregion-based conservation in the Chihuahuan Desert – a biological assessment. World Wildlife Fund, Comision Nacional para el Conocimiento y Uso de la Biodiversidad (CONABIO), The Nature Conservancy, PRONATURA Noreste, and Instituto Tecnologico y de Estudios Superiores de Monterrey (ITESM).

Engel-Wilson, R. W., and W. P. Kuvlesky, Jr. 2002. Arizona quail: species in jeopardy? Pages 1–7 in S. J. DeMaso, W. P. Kuvlesky, Jr., F. Hernández, and M. E. Berger, editors. Quail V: Proceedings of the Fifth National Quail Symposium. Texas Parks and Wildlife Department, Austin, TX.

Fleasch, A. 1997. Distribution and abundance of breeding grassland birds on the Buenos Aires National Wildlife Refuge: a monitoring program. Report prepared for United States Fish and Wildlife Service, Buenos Aires National Wildlife Refuge, P.O. Box 109, Sasabe, Arizona 85633.

Freeman, D. 1979. Lehmann lovegrass. Rangelands 1:162-163.

Goodwin, Jr., J. G. and C. R. Hungerford. 1977. Habitat used by native Gambel's and Scaled quail and released masked bobwhite quail in southern Arizona. USDA Forest Service Research Paper RM-197.

Gordon, C. E. 2000a. Movement patterns of wintering grassland sparrows in Arizona. Auk 117:748-759.

Gordon, C. E. 2000b. Fire and cattle grazing on wintering sparrows in Arizona grasslands. Journal of Range Management 53:384-389.

Gori, D., G. S. Bodner, K. Sartor, P. Warren, and S. Bassett. 2012. Sky Island Grassland Assessment: Identifying and Evaluating Priority Grassland Landscapes for Conservation and Restoration in the Borderlands. Report prepared by The Nature Conservancy in New Mexico and Arizona. 85 p.

Gori, D. F. and C. A. F. Enquist. 2003. An assessment of the spatial extent and condition of grasslands in central and southern Arizona, southwestern New Mexico and northern Mexico. The Nature Conservancy, Arizona Chapter. Pp29.

Green, G. A. 1983. Ecology and breeding of burrowing owls in the Columbia Basin, Oregon. M. S. thesis, Oregon State University, Corvallis, Oregon. 51 pp.

Guthery, F. S., N. M. King, W. P. Kuvlesky, S. DeStefano, S. A. Gall, and N. J. Silvy. 2001. Comparative habitat use by three quails in desert grassland.Journal of Wildlife Management 65(4):850-860.

Guthery, F. S., M. J. Peterson, J. J. Lusk, M. J. Rabe, S. J. DeMaso, M. Sams, R. D. Applegate, and T. V. Dailey. 2004. Multistate analysis of fixed, liberal regulations in quail harvest management. Journal of Wildlife Management 68(4):1104-1113.

Haug, E. A., B. A. Millsap, and M. S. Martell. 1993. Burrowing owl. Birds of North America 61: 1-20.

Howell, S. N. G. and S. Webb. 1995. The birds of Mexico and northern central America. Oxford University Press, Oxford.

Hubbard, J. P. 1978. Revised check-list of the birds of New Mexico. New Mexico Ornithological Society Publication No. 6.

Humphrey, R. R. 1974. Fire in the deserts and desert grassland areas of North America. Pages 385-400 in Kozlowski, T. T. and Ahlgren, C. E. (editors). Fire and ecosystems. Academic Press, New York. 542 pp.

Jones, Z.F. and C.E. Bock. 2005. The Botteri's Sparrow and exotic Arizona grasslands: an ecological trap or habitat regained? Condor 107:731-741.

Joseph, J., J. L. Holechek, R. Valdez, M. Collins, and M. Thomas. 2003.Research Observation: Effects of rangeland ecological condition on Scaled Quail sightings. Journal of Range Management 56(4):314-318.

Keddy-Hector, D. P. 2000. Aplomado falcon (*Falco femoralis*). The Birds of North America Online (A. Poole, editor). Ithaca: Cornell Lab of Ornithology. Retrieved from the Birds of North America Online: <http://bna.birds.cornell.edu/bna/species/549>

King, N. M. 1998. Habitat use by endangered Masked Bobwhites and other quail on the Buenos Aires National Wildlife Refuge, Arizona. Thesis. University of Arizona, Tucson.

Kirkpatrick, C., S. DeStefano, R. W. Mannan, and J. Lloyd. 2002. Trends in abundance of grassland birds following a spring prescribed burn in southern Arizona. Southwestern Naturalist 47:282-292.

Knopf, F. L. 1994. Avian assemblages on altered grasslands. Pp. 247-257 in J. R. Jehl, Jr. & N. K. Johnson (editors), A century of avifaunal change in western North American. Studies in Avian Biology 15.

Koford, C. B. 1958. Prairie dogs, whitefaces, and blue grama. Wildlife Monographs 3:1-78.

Krueper, D., J. Bart, and T. D. Rich. 2003. Response of vegetation and breeding birds to the removal of cattle on the San Pedro River, Arizona (U.S.A.). Conservation Biology 17:607-615.

Lane, J. 1968. *Ammodramus bairdii* (Audubon). Baird's sparrow. Pages 745-765 *in* Life histories of North American cardinals, grosbeaks, buntings, towhees, finches, sparrows, and their allies. (Austin, Jr., O. L., editor). U.S. National Museum Bulletin 237, Part 2.

Latta, M. J., C. J. Beardmore, and T. E. Corman. 1999. Arizona Partners in Flight bird conservation plan. Version 1.0. Nongame and Endangered Wildlife Program Technical Report 142. Arizona Game and Fish Department, Phoenix, AZ.

Leif, A. P. and L. M. Smith. 1993. Winter diet quality, gut morphology and condition of northern bobwhite and scaled quail in west Texas. Journal of Field Ornithology 64:527-538.

Lloyd, J., R. W. Mannan, S. Destefano, and C. Kirkpatrick. 1998. The effects of mesquite invasion on a southeastern Arizona grassland bird community. Wilson Bulletin 110:403-408.

Long, A. M., W. E. Jensen, and R. S. Matlack. 2012. Effects of prescribed burning on avian nest survival in the southern Great Plains. Journal of Wildlife Management 76:899-906.

MacCracken, J. G., D. W. Uresk, and R. M. Hansen. 1985. Vegetation and soils of burrowing owl nest sites in Conata Basin, South Dakota. *Condor* 87: 152–154.

Macias-Duarte, A., A. O. Panjabi, D. Pool, Erin Youngberg and Greg Levandoski. 2011. Wintering Grassland Bird Density in Chihuahuan Desert Grassland Priority Conservation Areas, 2007-2011. Rocky Mountain Bird Observatory, Brighton, CO, RMBO Technical Report I- NEOTROP-MXPLAT-10-2. 164 pp.

Macias-Duarte, A., A. B. Montoya, W. G. Hunt, A. Lafon-Terrazas, and R. Tafanelli. 2004. Reproduction, prey, and habitat of the Aplomado Falcon (Falco femoralis) in desert grasslands of Chihuahua, Mexico. Auk 121:1081-1093.

Marshall, R. M., D. Turner, A. Gondor, D. Gori, C. Enquist, G. Luna, R. Paredes Aguilar, S. Anderson, S. Schwartz, C. Watts, E. Lopez, and P. Comer. 2004. An ecological analysis of conservation priorities in the Apache Highlands Ecoregion. Prepared by The Nature Conservancy of Arizona, Instituto del Medio Ambiente y el Desarrollo Sustentable del Estado de Sonora, agency and institutional partners. 152 pp.

Maurer, B. A. 1986. Predicting habitat quality for grassland birds using density-habitat correlations. Journal of Wildlife Management 50:556-566.

Medina, A. L. 1988. Diets of Scaled Quail in southern Arizona. Journal of Wildlife Management 32:753-757.

Monson, G. and A. R. Phillips. 1981. Annotated checklist of the birds of Arizona. University of Arizona Press, Tucson.

Morrison, M. L. 1999. Bird habitat relationships in desert grasslands. USDA Forest Service Proceedings RMRS-P-10.

Natural Resources Conservation Service and Wildlife Habitat Council (NRCS). 1999. Grassland birds. Fish and Wildlife Habitat Management Leaflet No. 8.

Nelson, T., J. L. Holechek, R. Valdez, and M. Cardenas. 1997. Wildlife numbers on late and mid seral Chihuahuan Desert rangelands. Journal of Range Management 50(6):593-599.

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.

Phillips, A. R., J. T. Marshall, Jr., and G. Monson. 1964. The birds of Arizona. University of Arizona Press, Tucson.

Pool, D. B., A. Macias-Duarte, A. O. Panjabi, G. Levandoski, and E. Youngberg. 2012. Chihuahuan Desert Grassland Bird Conservation Plan, version 1.0. Rocky Mountain Bird Observatory, Brighton, CO, RMBO Technical Report I-RGJV-11-01. 74 pp.

Pyne, S. J. 1984. Introduction to wildland fire. John Wiley and Sons, New York. 206 pp.

Rea, A. M. 1973. The scaled quail (*Callipepla squamata*) of the southwest: systematic and historical consideration. Condor 75:322–329.

Russel, T. P. 1932. The scaled quail of New Mexico. Master's thesis, University of New Mexico, Albuquerque.

Ruth, J.M. 2000. Cassin's Sparrow status assessment and conservation plan. Biological Technical Publication BTP-R6002-2000. USFWS, Denver, CO

Ruth, J. M. 2008. Distribution of breeding Arizona grasshopper sparrow (*Ammodramus savannarum ammolegus*) in the southwestern United States: past, present, and future. Studies in Avian Biology No. 37: 113-124.

Saab, V. A., C. E. Bock, T. D. Rich, and D. S. Dobkin. 1995. Livestock grazing effects in western North America. Pages 311-353 *in* T. E. Martin and D. M. Finch, editors). Ecology and management of neotropical migratory birds. Oxford University Press, New York.

Saiwana, L., J. L. Holechek, A. Tembo, R. Valdez, and M. Cardenas. 1998. Scaled quail use of different seral stages in the Chihuahuan Desert. Journal of Wildlife Management 62(2):550-556.

Smith, G., J. L. Holechek, and M. Cardenas. 1996. Wildlife numbers on excellent and good condition Chihuahuan Desert rangelands: an observation. Journal of Range Management 49(6):489-493.

Strong, T. R. 1988. Status report: Arizona grasshopper sparrow, *Ammodramus savannarum ammolegus* (H. C. Oberholser). Arizona Game and Fish Department, report to Office of Endangered Species, U.S. Fish and Wildlife Service, Albuquerque, New Mexico.

Swetnam, T., and C. Baisan. 1996. Historical fire regime patterns in the southwestern United States since AD 1700. Pages 11-32 in C.D. Allen, editor. Fire effects in southwestern forests. Proceedings of the 2nd La Mesa Fire Symposium, USDA Forest Service, Rocky Mountain Research Station, General Technical Report RM-GTR-286.

Uhmann, T. V., N. C. Kenkel, and R. K. Baydack. 2001. Development of a habitat suitability index model for burrowing owls in the eastern Canadian prairies. *Journal of Raptor Research* 35: 378–384.

Valone, T. J., and D. A. Kelt. 1999. Fire and grazing in a shrub-invaded arid grassland landscape: independent or interactive ecological effects? Journal of Arid environments 42:15-28.

Van Auken, O. W. 2000. Shrub invasions of North American semiarid grasslands. Annual Review of Ecology and Systematics 31:197-215.

Vickery, P. D. 1996. Grasshopper sparrow (*Ammodramus savannarum*). The Birds of North America Online (A. Poole, editor). Ithaca: Cornell Lab of Ornithology. Retrieved from the Birds of North America Online <http://bna.birds.cornell.edu/bna/species/239>

Wallmo, O. C. and P. B. Uzzell. 1958. Ecological and social problems in quail management in west Texas. Transactions of the North American Wildlife Conference 23:320-327.

Webb, E. A. 1985. Distribution, habitat and breeding biology of the Botteri's Sparrow (*Aimophila botterii arizonae*). Master's thesis. University of Colorado, Boulder.

Webb, E. A. and C. E. Bock. 1990. Relationship of the Botteri's Sparrow to sacaton grassland in southeastern Arizona. Pages 199-209 *in* Krausman, P. R. and N. S. Smith, editors. Managing wildlife in the southwest: proceedings of the symposium. Arizona Chapter of the Wildlife Society. Tucson.

Webb, E. A., and C. E. Bock. 2012. Botteri's sparrow (*Peucaea botterii*). The Birds of North America Online (A. Poole, editor). Ithaca: Cornell Lab of Ornithology. Retrieved from the Birds of North America Online: <http://bna.birds.cornell.edu/bna/species/216>

Williams, S. O. 2004. Trends in Arizona grasshopper sparrow populations on the Gray Ranch. Report to Animas Foundation. New Mexico Department of Game and Fish, Santa Fe, NM.

Yosef, R. 1996. Loggerhead Shrike (Lanius ludovicianus), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu.bnaproxy.birds.cornell.edu/bna/species/231 doi: 10.2173/bna.231.

# *F-8 Nectivorous Bats*

# Lesser long-nosed bat (Leptonycteris yerbabuenae), Mexican long-nosed bat (Leptonycteris nivalis), Mexican long-tongued bat (Choeronycteris mexicana)

## F-8.1 Conceptual Model

### F-8.1.1 Classification Comments

All three species are members of the family Phyllostomidae, which comprises mostly tropical species. All three species have a long snout with a triangular-shaped flap of skin on the nose. They are approximately 2.5-4 inches long, with a wingspan of approximately 13-17 inches. All have relatively large eyes and thus good vision. Long-nosed bats (*Leptonycteris* spp.) lack an external, visible tail (Arita 2005; it is concealed and consists of just a few vertebrae); and *C. mexicana* has a very short tail (Ortega and Arita 2005).

**Lesser long-nosed bat (*Leptonycteris yerbabuenae*):** As summarized by Arizona's Natural Heritage Program (AZGFD 2011), the "…nomenclatural history of this bat and the two other species in the genus has been rather confusing over the years. The currently accepted taxonomy for the lesser long-nosed bat is *Leptonycteris yerbabuenae* (Simmons and Wetterer 2002, Wilson and Reeder 2005, Cole and Wilson 2006); however, it is listed as Endangered under the U.S. Endangered Species Act (ESA) under the name *L. curasoae yerbabuenae* (Wilson and Reeder 2005)."

**Mexican long-nosed bat (*Leptonycteris nivalis*):** Prior to the taxonomic revision by Davis and Carter (1962), specimens of *Leptonycteris yerbabuenae* were reported as *L. nivalis*.

**Mexican long-tongued bat (*Choeronycteris mexicana*):** This is the only species in the genus *Choeronycteris* that comes as far north as the southwestern United States (AZGFD 2006).

### F-8.1.2 Individual Species Highlights

#### *F.8.1.2.1 Lesser Long-nosed Bat (Leptonycteris yerbabuenae)*

**Distribution and migration.** *Global*: The global range extends from southern Arizona, and southwestern New Mexico southward through the drier parts of Mexico (including Baja California) to Guatemala, Honduras, and El Salvador (Cole and Wilson 2006). Elevational range extends from sea level to 2400 meters (Arita 1993, 2005).

*Madrean Archipelago Ecoregion*: The range in southern Arizona extends from the Picacho Mountains southwest to the Agua Dulce Mountains and southeast to the Galiuro and Chiricahua mountains, and continues southward into Mexico and beyond (AZGFD 2011). Two late-summer records of immature individuals from the Phoenix area and one from the Pinaleno Mountains have also been documented (AZGFD 2011). A maternity colony formerly occupied Colossal Cave, southeast of Tucson; however, today all known maternity sites in Arizona are west of Tucson (Wolf, n.d.).

In New Mexico, the species is found in the Animas Mountains, Guadalupe Canyon, and Big Hatchet Mountains in Hidalgo County (Findley et al. 1975, Baltosser 1980, Cook 1986, Hoyt et al. 1994, Bogan 2005). As summarized in the USFWS' five-year review (2007) for this species, "One roost site in the

Peloncillo Mountains on the Arizona-New Mexico border is known. Two roosts were documented in the mid-1990s in the Animas Mountains of Hidalgo County, New Mexico (Altenbach 1995). These include a new day roost that sometimes contains several hundred long-nosed bats, and a historical night roost where both species of long-nosed bat occur. One additional roost was found in the Big Hatchet Mountains in 2005 (Bogan 2005)." Without specifying the species or location, Cryan (2012) reported the recent discovery of a significant new long-nosed bat roost on BLM lands, containing the largest aggregation of these bats found to date in New Mexico.

This species is migratory in the northern extent of its range and is not present in the United States during winter months. Migrants from central Mexico arrive in southwestern Arizona and adjoining portions of Sonora in April, forming large maternity colonies. As described by AZGFD (2011), "During the early part of their stay (late April to late July) pregnant females congregate at traditional roost sites, give birth, and raise their young at lower elevations within the range of columnar cacti. Males and perhaps nonpregnant females do not arrive until sometime in July. By late July, most females and young have dispersed from maternity colonies and some have moved to higher elevations where they are found feeding on agave flowers." Adult females and young of the year form late-summer roosts in southeastern Arizona and southwestern New Mexico. The bats return to central Mexico by September-October (AZGFD 2011).

Lesser long-nosed bats follow two corridors in their northward spring migration (Coastal Lowland Route and Inland Montane Route). At least one corridor is utilized in the southward fall migration (Inland Agave Corridor), however it has not been confirmed that all bats using the Spring Coastal Lowland Route follow this Inland Agave Corridor for the return south in the fall (Wilkinson and Fleming, 1996).

**Reproduction.** As summarized in the USFWS' profile (2013) for this species, **"**Females arrive already pregnant at maternity roosts in Arizona as early as the second week in April. The single young is born in May. Maternity colonies vary in size, from a few hundred to tens of thousands of females. Males maintain separate colonies during this period. Young bats can fly by the end of June, and the maternity colonies break up by the end of July." This discrete, temporally concentrated birthing pattern exists in the Sonoran Desert from southern Sonora, Mexico, to Arizona. Farther south in Mexico, lesser long-nosed bats have a complex life history, with two female "demes," the reproduction of which is temporally and spatially displaced (Medellín 2005).

**Habitat and Ecology.** Habitats include tropical deciduous forest, as well as semi-deciduous thorn scrub, oak-pine, and cloud forests (Arita 2005). In Arizona, the species uses Palo Verde/Saguaro, Semi-desert Grassland, and Oak Woodland (AZGFD 2011). The bats' diet of nectar enables it to be essentially independent of free water (AZGFD 2011). Lesser long-nosed bats often roost deep in a mine or cave (Cryan and Bogan 2003).

**Diet and Pollination.** These bats feed primarily on nectar and pollen from the flowers of the saguaro cactus (*Carnegia gigantea*) in early summer, then focus on the flowers of paniculate agaves from late summer to early fall. The bats may switch to less commonly consumed food plants or shift to distant locations when local food resources fail (USFWS 2007). According to Arizona Game and Fish (2011), "individual bats may land on a panicle of flowers to feed or they may bury their snout in a flower and rapidly lap up nectar while hovering. Feeding at agave flowers may often be done in groups. Long-nosed bats also consume the ripe fruits of saguaro and organ pipe cactus at the end of the flowering season (U.S. Fish and Wildlife Service 2013) and thereby serve as dispersal agents for the seeds. Insects may be ingested incidentally or intentionally while the bats feed on nectar. Lesser long-nosed bats also are

known to feed on sugar water from hummingbird feeders at night; use of feeders may increase when natural food resources are scarce (USFWS 2007). Winter populations in southern Mexico feed on flowers of a variety of plants (Agavaceae, Bombacaceae, Cactaceae, Convolvulaceae, and Leguminosae) based on their temporal availability (see Cole and Wilson 2006).

Long-nosed bats get additional nutrients and protiens not present in nectar by consuming pollen when they grooms themselves. Once the bats feed on nectar, they retire to night roots to rest and groom themselves. The grooming removes pollen stuck to their fur which they then lick off their claws. Night roost may be in different locations than day roosts.

Various agave species, columnar cacti and other Mexican plant species are pollinated by lesser long-nosed bats as they feed. Pollen collects on their heads and shoulders when the reach into flowers to get nectar and is then rubbed off on subsequent plants they visit. It is thought this bat may be an important pollinator of saguaro cactus and other agave species it feeds on in the United States.

**Activities, nightly movements and sociality.** These bats emerge to feed about an hour after sunset. In Mexico, these bats fly up to 30 km each night from their roosts on Isla Tiburon in the Sea of Cortez to their feeding grounds in mainland Sonora (Arizona Game and Fish Department 2011). Wolf (n.d.) indicates that "Long-nosed bats can travel 30 miles to forage, although probably most go no more than 15-20 miles. In southeastern Arizona, radio-transmittered bats were found to spend most of the night foraging, going to the same area each night. Between foraging bouts, they visited night roosts to digest and rest. Night roosts can be in man-made structures such as porches, barns, and bridges as well as caves, mines, and trees." Some observations indicate that lesser long-nosed bats spend about 6 hours a night foraging, alternating about 20 minutes of flying and feeding with about 20 minutes of roosting on plants or rocks and grooming (Howell 1979).

**Abundance and trends.** This species occurs in approximately 40 total roosts across its range in Arizona, New Mexico, and Mexico and at three maternity roosts in the United States (USFWS 2007). Population size is not precisely known but appears to be at least several 10,000s in Arizona, in addition to those in Mexico (100,000s) and New Mexico (several thousand) (USFWS 2007). Based on improved and consistent monitoring protocols in recent years, the overall population at known roosts is believed to be stable or increasing in both the United States and Mexico (USFWS 2007). However, the number of known roost sites has not significantly increased (USFWS 2007).

Over the long term, populations are presumed to have declined significantly (Arizona Game and Fish Department 2011). However, the species is far more numerous than early reports (Wilson 1985) indicated (USFWS 1995). Cockrum and Petryszyn (1991) strongly disputed the reported decline of this species and, in reviewing pertinent data, concluded that little evidence exists to document a long-term decline in Arizona, New Mexico, and Sonora; these authors stated "the various recent reports of disappearance appear to be, at least in part, the result of not looking in the right places at the right times" and further reported that "current populations...are little, if any, decreased from those of a quarter century ago. It even has been suggested that populations have increased in the past century because of more suitable roosts being available as a result of mining activity in the area."

In New Mexico, limited data indicate that lesser long-nosed bats can be locally numerous in late summer. In 1992, Hoyt et al. (1994) netted 150-200 *Leptonycteris* (primarily *L. yerbabuenae* but also about half as many *L. nivalis*), and believed that these represented a portion of a "much larger actual population of both species."

This species is listed as Vulnerable on the IUCN Red List, based on an assessment completed in 2008 (http://www.iucnredlist.org/details/136659/0). It is listed under the U.S. Endangered Species Act as as Endangered (under the name *Leptonycteris curasoae yerbabuenae*), but USFWS (2007) concluded that the species should be proposed for reclassification from endangered to threatened. *Leptonycteris yerbabuenae* was identified as a species of greatest conservation need in the Comprehensive Wildlife Conservation Strategy for New Mexico (NMDGF 2006) and the in State Wildlife Action Plan for Arizona (Arizona Game and Fish Department 2012).

#### *F.8.1.2.2 Mexican Long-nosed Bat (Leptonycteris nivalis)*

**Distribution and migration.** *Global:* The range includes medium to high elevations in northern and central Mexico, southwestern Texas (southern Brewster and Presidio counties), and southwestern New Mexico (Frey 2004), at elevations ranging from approximately 500 to 3,000+ meters. Most occurrences in Mexico are at elevations of 1,000-2,200 meters, but this bat been captured at an elevation of 3,780 meters (see Arita 1991), and the type specimen was caught near snow line at 17,816 feet (5,747 meters) on Mt. Orizaba, in Veracruz, Mexico (USFWS 1994). In Texas, the species is known from the Big Bend National Park and Chinati Mountain area. This species formerly was thought to occupy a much larger area, extending into southern Mexico and Guatemala, but specimens collected from those areas were assigned to *L. yerbabuenae* by Arita and Humphrey (1988). Simmons (in Wilson and Reeder 2005) described the range of *L. nivalis* as extending to southern Mexico and Guatemala, and she also included southeastern Arizona in the range. However, no actual records for Arizona are known, and Arita (1991) showed the range as extending only as far south as Puebla and northern Guerrero.

*Madrean Archipelago Ecoregion*: Two specimens of *Leptonycteris* taken in Hidalgo County, New Mexico (in 1963 and 1967), were determined to be *L. nivalis*. The presence of this species in New Mexico was reconfirmed with numerous captures of *L. nivalis* in the Animas Mountains in late August, Hidalgo County, in 1992; these bats may have been migrants from western Mexico (Hoyt et al. 1994).

The young are born in Mexico during April, May, and early June, then move northward with their mothers.

**Reproduction.** Adult females give birth to a single pup in spring (April-June) in Mexico before females arrive in the United States; no records of pregnant females are known from Texas (Schmidly 1977). In Texas, lactating females have been observed in June-July, flying juveniles in late June. Young are weaned in July or August.

**Habitat and Ecology.** Habitats in the northern part of the range in the United States and northern Mexico include desert scrub, open conifer-oak woodlands, and pine forests, generally in arid areas where agave plants are present (USFWS 1994). In Big Bend National Park, Texas, habitat includes mesquite-acacia (1800-4000 ft.), lechuguilla-creosotebush-cactus (1800-3500 ft.), deciduous woodland (3700-7800 ft.), pinyon-juniper-oak woodland (3700-7800 ft.), and cypress-pine-oak (5800-7200 ft.). As described in the NMDGF's biennial review (2014), "Mexican long-nosed bats inhabit upper desert scrub - pine oak woodlands in or near mountainous areas; characteristic vegetation in these areas includes agaves (*Agave* spp.), junipers (*Juniperus* spp.), oaks (*Quercus* spp.), and Mexican pinyon (*Pinus cembroides*)." According to NMDGF (2014), "Roosting habitats of this species in New Mexico have been recently studied, and all known day roosts are in caves and rock fissures, and an abandoned building near the Animas Mountains is used as a night roost between feeding forays (Bogan et al., 2006a-b;

Cryan, 2007; Goodbar, 2007). They roost in caves, mines, hollow trees, and man-made structures in other portions of their range (Hall and Dalquest, 1963; Novick, 1963; Hensley and Wilkins, 1988)."

**Diet and Pollination.** The diet includes mainly nectar and pollen of cacti (saguaro, organ pipe) and paniculate agaves; also insects associated with flowers, and probably some fruits, especially in the southern part of the range. In one type of forgaging at agaves, the bats crawl down the stalk, thrust the snout into the flowers, and obtain nectar with the long tongue (Harvey et al. 2011).

**Activities, nightly movements and sociality.** These bats emerge relatively late in the evening to feed (Harvey et al. 2011).

**Abundance and trends.** Long-term trend is unclear. Extent of occurrence and area of occupancy probably have not changed much, but the number of occurrences or subpopulations and population size may have dramatically decreased in some locations during the last three decades. Wilson (1985) found that this species was either completely absent or present in reduced numbers in known roosts. The number of bats found represented only a fraction of the total reported in previous studies. For example, in an abandoned mine in Nuevo Leon, Mexico, where an estimated population of 10,000 was observed in 1938, no individuals of *L. nivalis* were found in 1983 (Wilson 1985). Another mine in Nuevo Leon had a ceiling covered with newborn bats in 1967, but only one bat was found in 1983. A few other roosts had reduced numbers of bats compared to findings during previous surveys. These changes could indicate a decline in the overall population, but they might reflect movement of bats among different roosting sites in different years, or they could result from seasonal changes in bat distribution (survey dates varied).

Abundance at one of the two known roosting sites in the United States fluctuates widely from year to year (0 to 10,000+ individuals; Easterla 1972). In 2005, the greatest number of *L. nivalis* (at least 2742-2874, based on thermal imaging data) was present in July (Ammerman et al. 2009). Count data may not reliably reflect true numbers (e.g., some bats either roost in inaccessible crevices or are widely dispersed in the cave and thus cannot be observed and counted; England 2004), and no significant trend can be ascertained from available data. Reasons for the fluctuations are not understood, but they could reflect annual variations in food resources in different parts of the range (USFWS 1994).

In New Mexico, limited data indicate that Mexican long-nosed bats can be locally numerous in late summer. In 1992, Hoyt et al. (1994) netted 150-200 *Leptonycteris* (primarily *L. yerbabuenae* but also about half as many *L. nivalis*), and believed that these represented a portion of a "much larger actual population of both species." Population trend in New Mexico is unknown (Hoyt et al. 1994).

This species is listed as Endangered on the IUCN Red List, based on an assessment completed in 2008 (http://www.iucnredlist.org/details/11697/0). It is listed under the U.S. Endangered Species Act as as Endangered. *Leptonycteris nivalis* was identified as a species of greatest conservation need in the Comprehensive Wildlife Conservation Strategy for New Mexico (NMDGF 2006)

#### *F.8.1.2.3 Mexican Long-tongued Bat (Choeronycteris mexicana)*

**Distribution and migration.** *Global*: The range includes southern California, southern Arizona, southwestern New Mexico, the southern tip of Texas, and much of northern and central Mexico, including the Baja California peninsula, El Salvador, and Honduras in Central America (Arroyo-Cabrales et al. 1987, Ortega and Arita 2005, Arizona Game and Fish Department 2006). Maternity colonies have been found in Arizona and New Mexico. A single individual has been found in Las Vegas, Nevada

(Constantine 1987). In California, most records are from San Diego County in fall and winter, but occurrence records are relatively scarce (Pierson and Rainey, in Brylski et al. 1998). Elevational range extends from 300 to 3,600 meters (Arita 1993, Ortega and Arita 2005).

*Madrean Archipelago Ecoregion*: The Arizona Game and Fish Department (2006) indicates that "the range includes southeastern Arizona, from the Chiricahua Mountains to as far north as the Santa Catalina Mountains and as far west as the Baboquivari Mountains. Unpublished records show them in Pinal, Pima, Graham, Santa Cruz, and Cochise counties."

This species probably migrates seasonally to take advantage of suitable sources of food and populations in the U.S. portion of the ecoregion migrate south across the border into Mexico to spend the winter (AZGFD 2006). However, in the Tucson area, these bats have been observed at hummingbird feeders throughout the winter, so not all migrate to Mexico in the fall. Long-tongued bats have longer tongues than do long-nosed bats and may be able to exploit a wider variety of food plants (Wolf, n.d.).

**Reproduction.** From Arizona Game and Fish Department (2006): "according to Cockrum and Petryszyn (1992), only adult females move north from Mexico into the United States with the males remaining 'in the southern part of the range during the time that young are being nourished by the mothers in the north.' Females typically give birth to one pup between mid to late June and early July. Parturition may be as late as September in Mexico (Harvey et al. 2011). Young grow rapidly and can fly within probably 2-3 weeks. After the young become volant, these bats move about opportunistically in search of food. Females are known to carry their young in flight." Long-tongued bats form small maternity groups, usually of a dozen or less, but occasionally up to 50 adults.

**Habitat and Ecology.** According to Arizona Game and Fish Department (2006), **"**In the northern part of range in the United States, roost sites commonly are associated with mesic areas in oak-conifer woodlands or semi-desert grasslands. Dominant plant species include oaks (*Quercus*), alligator juniper (*Juniperus deppeana*), manzanita (*Arctostaphylos*), yucca, and agave." They also occur at higher elevations in pine-fir forests. A survey of historical roosts found that roosts occupied in 1999 were associated with surface water and riparian vegetation, and were near agaves, which are a major food source. Long-tongued bats commonly are found near roost site entrances (Cryan and Bogan 2003); if disturbed, they tend to fly out into the open rather than deeper into the shelter (Wolf, n.d.). In Arizona and New Mexico, Mexican long-tongued bats exhibit high levels of roost site fidelity (Mumford et al. 1964). They are often found in shallow caves and rock shelters and have been documented in the Cienega Creek Preserve in Arizona roosting in pocketed, eroded clay soil holes such as sink holes, or soil piping caves immediately adjacent to riparian habitat (AZGFD 2006).

**Diet and Pollination.** These bats feed on nectar, pollen, probably insects, and occasionally fruit of columnar cacti (Villa-R. 1967, Alvarez and Gonzalez-Q. 1970). Near Tucson, Arizona, long-tongued bats feed predominantly on cactus and *Agave* species (*Agave schottii* before mid-June, then *A. palmeri*) (Van de Water and Peachey 1997, Arizona Game and Fish Department 2006). They have a bristle-like tongue and lack of lower incisors which allow them to lap up nectar. They have been reported to feed at hummingbird feeders during winter (Arizona Game and Fish Department 2006). Agave species were recored at occupied sites by Cryan and Bogan (2003) with *Agave schottii* observed blooming before mid-June and *Agavie palmeri* after (AZGFD 2006).

Long-nosed and long-tongued bats follow a "nectar corridor" of blossoming plants north from Mexico into the United States each spring (Fleming 2012). Bats are thought to follow sequential blooming of

flowers they feed on as they move northward from central Mexico during spring migration (Arizona Game and Fish Department 2011). Mexican long-tongued bats apparently prefer to feed on *Agave* flowers during northward migration, and they arrive in the southwestern United States later than do the long-nosed bats (not until *Agave* has started blooming).

**Activities, nightly movements and sociality.** Activity peaks 1.5 hours after sunset and extends at low levels until about 3 hours after sunset. According to the Arizona Game and Fish Department (2006), "these bats are less gregarious than other colonial bats and less inclined to roost with other bat species. In roosts, they do not cluster closely together but hang 2.0-5.0 cm apart. Roosts usually consist of 15 or fewer individuals, but when considering roosts in close proximity to each other, population numbers may reach up to 40-50." These bats do not hibernate.

**Abundance and trends.** A 2003 study that surveyed 24 historically occcupied sites in Arizona and New Mexico (Cryan and Bogan) found insufficient evidence to conclude that populations in Arizona and New Mexico have increased or decreased in recent years. Surveys found this species in 18 (75 percent) of the sites. Sites where the species was not found were frequently disturbed, difficult to search, or historically occupied by single individuals.

This species is listed as Near Threatened on the IUCN Red List, based on an assessment completed in 2008 (http://www.iucnredlist.org/details/4776/0).

### F-8.1.3 Summary

*Leptonycteris yerbabuenae* and *Choreonycteris mexicana* both rely on columnar cacti and agave species for nectar feeding but, in the Arizona portion of the Madrean ecoregion, *C. mexicana* arrives later in Arizona when agaves are blooming and this is the primary food source for both *L. yerbabuenae* and *C. mexicana* through the fall. This means that adequate agave foraging habitat in association with suitable daytime and nighttime roosts are important for both. The range of *L. yerbabuenae* and *C. mexicana* overlap slightly, but *C. mexicana* usually occupies higher elevations.

These bats spend most of the time in roosts, which fulfill many important roles in the bats' lives. The most important survival benefits of a suitable roost include: protection from the weather; protection from predators; cheaper thermoregulation; reduced commuting costs to foraging sites; improved mating opportunities; improved maternal care; and competition avoidance, since few other vertebrates make use of most bat roost sites (Nolan, n.d.).

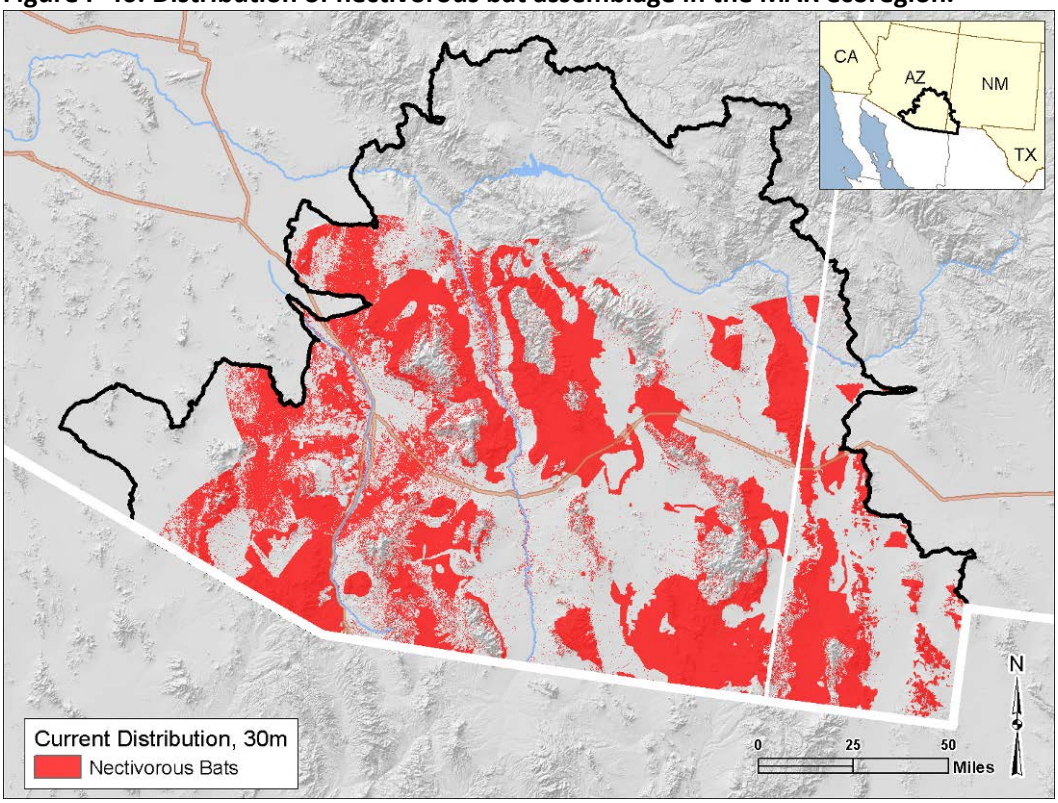
Long-term sustainability of food resources is extremely important to all three species and may be affected by habitat destruction/development, grazing/browsing, and fires (see Change Agents below). The degree of urbanization that a lesser long-nosed bat will tolerate to forage and move through is not well understood (AZGFD 2013).

It is important to note that there is a lack of comprehensive information on the extent of effects of agave harvesting, land clearing, grazing/browsing of agave and roost destruction or disturbance on the persistence and population health of nectar-feeding bats. Because a large portion of the life histories of *Choreonycteris* and *Leptonyteris* spp. are dependent on available roost sites and foraging habitat in Mexico including foraging habitat that supports seasonal migration, management actions taken within the Madrean ecoregion of the US may have limited effect on overall population numbers.

All three species are migratory and rely on corridors with adequate foraging habitat to support their seasonal movements. This is a critical part of their life histories, but because these species are at the northern edge of their range in the Madrean ecoregion, there is a limited extent of "nectar corridor" within the region.

The combined, modeled distribution of the three nectar-feeding bats is shown in Figure F-40; the methods for generating this distribution map are briefly described in the Distribution Mapping Methods section earlier in this appendix, as well as in Appendix C.

**Figure F-40. Distribution of nectivorous bat assemblage in the MAR ecoregion.**




## F-8.1.4 Change Agent Characterization

This section of the conceptual model presents a narrative description of the primary change agents and current knowledge of their effects on nectar-feeding bat populations.

### *F.8.1.4.1 Disturbance and vandalism*

Disturbance, vandalism, and outright killing of bats in roosting sites probably have detrimentally affected all three nectar-feeding species (USFWS 1994). Excessive disturbance may lead to increased mortality (e.g., of neonates) and roost abandonment. Activities associated with attempts to control vampire bats in Mexico have resulted in direct mortality and roost abandonment by long-nosed and long-tongued bats, and suitable alternate roost sites for displaced bats may not be available, even if other caves or abandoned mines are present. These bats have low reproductive rates and can recover only slowly from population declines.

Disturbance impacts and damage to roost sites from human-related activities such as recreation and caving are probably not as great as previously thought, but the potential for such impacts will likely increase in the near future due to increasing development, urbanization, and other land-conversion activities, all of which result in additional human presence in the areas where long-nosed bat roost sites are located (USFWS 2007).

#### *F.8.1.4.2 Closure or gating of cave and mine entrances*

Intentional closing of caves and mines due to human safety concerns has eliminated some roost sites and could eliminate additional sites in the future. Natural cave-ins or gradual fill-in of roost-site entrances or passages also may close off bat access or render some entrances or important passages too small for a large bat colony to pass through without greatly increasing the danger of predation; however, these natural processes are not regarded as a major threat to any of the nectar-feeding bats.

Erection of gates or other semipermeable structures to exclude humans may unintentionally exclude bats from their roosts or degrade or eliminate roost habitats due to changes in air flow and modification of the climate inside the caves or mines. Additionally, modifications that affect bat movements inside or outside caves or mines may result in increased predation on bats. The response of long-nosed bats to various gate designs needs further research (USFWS 2007). State of Texas Mine has been gated and research conducted on the response of lesser long-nosed bats to the gate.

#### *F.8.1.4.3 Loss of foraging habitat*

Destruction of forage plants through excessive human harvesting of native agaves for food and alcohol (e.g., tequila) production, particularly in Mexico, is a threat to these bats (USFWS 1994). Legitimate tequila producers likely have minimal effects on natural forage availability, but bootleg production often utilizes wild agaves and remains a threat (USFWS 2007). Clearing of land, grazing/browsing of agave species, and fires that remove foraging habitat in Mexico may also threaten these species. Recent evidence indicates that lesser long-nosed bats can change their foraging patterns and food sources in response to periodic loss or great reduction in a usually important plant species; this species appears to be more resourceful and resilient than previously thought (USFWS 2007).

Browsing on newly emergent flower stalks of agave species has been suggested as decreasing foraging opportunities (AZGFD 2006). Agave spcies such as *Agave palmeri* grow for approximately 25 years before producing reproductive stalks. When the stalk is browsed before the flower opens, not only does the individual plant not provide nectar for bats, but it will not reproduce sexually thus reducing future availability of nectar resources. For lower-elevation species (*Leptonycteris yerbabuena*) relying on foraging patches in desert and grassland habitats, habitat loss due to development or land use conversion may pose an important threat. For species typically found at higher elevations (*Leptonycteris nivalis*), grazing/browsing of forage plants and loss of forage plants due to fire may pose an important threat. The level of tolerance *Leptonycteris yerbabuenae* has for foraging in suburban or urban habitat patches is not currently known (AZGFD 2013).

#### *F.8.1.4.4 Illegal border activities*

These activities, and associated enforcement actions along the U.S.-Mexico border, affect roosts through disturbance and destruction, and they affect foraging habitat through vegetation damage and increased potential for fire (USFWS 2007). In part as a result of border activities, threats to bat roost sites continue and, in fact, have likely increased in recent years (USFWS 2007).

#### *F.8.1.4.5 Wind energy*

A wind farm project has been proposed on Fort Huachuca, Arizona, an area supporting at least five late-summer lesser long-nosed bat roosts and agave foraging areas. No current wind farms occur in areas occupied by nectar-feeding bats, so the impacts on these bats are uncertain (USFWS 2007). Mortalities at wind turbines have been well documented for other bats species.

#### *F.8.1.4.6 Invasion of non-native plant species*

Expansion of non-native plant species affects the availability of bat food resources over most of their ranges (USFWS 2007). Mesquite has replaced many areas of native grasslands which have agave species as a component (Gori et al. 2012) and may lead to the loss of agaves (see NRCS 2014, ecological site description in the grassland case study above and Appendix D). Invasive grasses, such as the lovegrasses (*Eragrostis* spp.) have also been shown to result in the loss of agave species (Lindsay et al. 2010).

#### *F.8.1.4.7 Increased fire frequency and intensity; drought*

Fires in recent years in the Santa Rita, Catalina, Huachuca, and Chiricahua mountains have impacted large areas of lesser long-nosed bat foraging habitat. Increased human presence in the range of the nectar-feeding bats, and continued drought in the region, in addition to the presence of non-native plant species, may result in more frequent and intense fires and could lead to further negative impacts of bat foraging areas.

Ongoing, long-term drought may result in increased failures in the bats' saguaro and agave food resources (USFWS 2007). On at least a short-term (single season) basis, lesser long-nosed bats appear to be able to shift to other plant species in response to failure of their typical forage species (USFWS 2007). However, the long-term effects of continued drought are uncertain.

### F-8.1.5 Information Needs

For all three species, better information is needed on current distribution and abundance, trends, migration patterns, interactions with and dependence on specific plant species, and characteristics of preferred roost sites. Additionally, better information is needed on gate design and its effects on nectar-feeding bats although State of Texas Mine has been gated and research conducted on the response of lesser long-nosed bats to the gate.

### F-8.1.6 Ecological Status: Key Ecological Attributes and Indicators

This section of the conceptual model addresses Key Ecological Attributes and their potential indicators. The ecological status is a way of describing current status via criteria, functionality, or levels of attributes and asks if they are within the normal range of variation.

#### *F.8.1.6.1 Key Ecological Attributes*

Table F-13 identifies the key ecological attributes for this CE within the Madrean Archipelago ecoregion. A **key ecological attribute** of a focal ecological resource is a characteristic of the resource's biology, ecology, or physical environment that is critical to the resource's persistence in the face of both natural and human-caused disturbance. Alteration of such a characteristic beyond some critical range of variation will lead to the degradation or loss of the resource within decades or less. The KEAs table lists the identified key ecological attributes, with a brief definition, a rationale for why it is important for the CE, and a listing of stressors or change agents that might be affecting the key attribute.

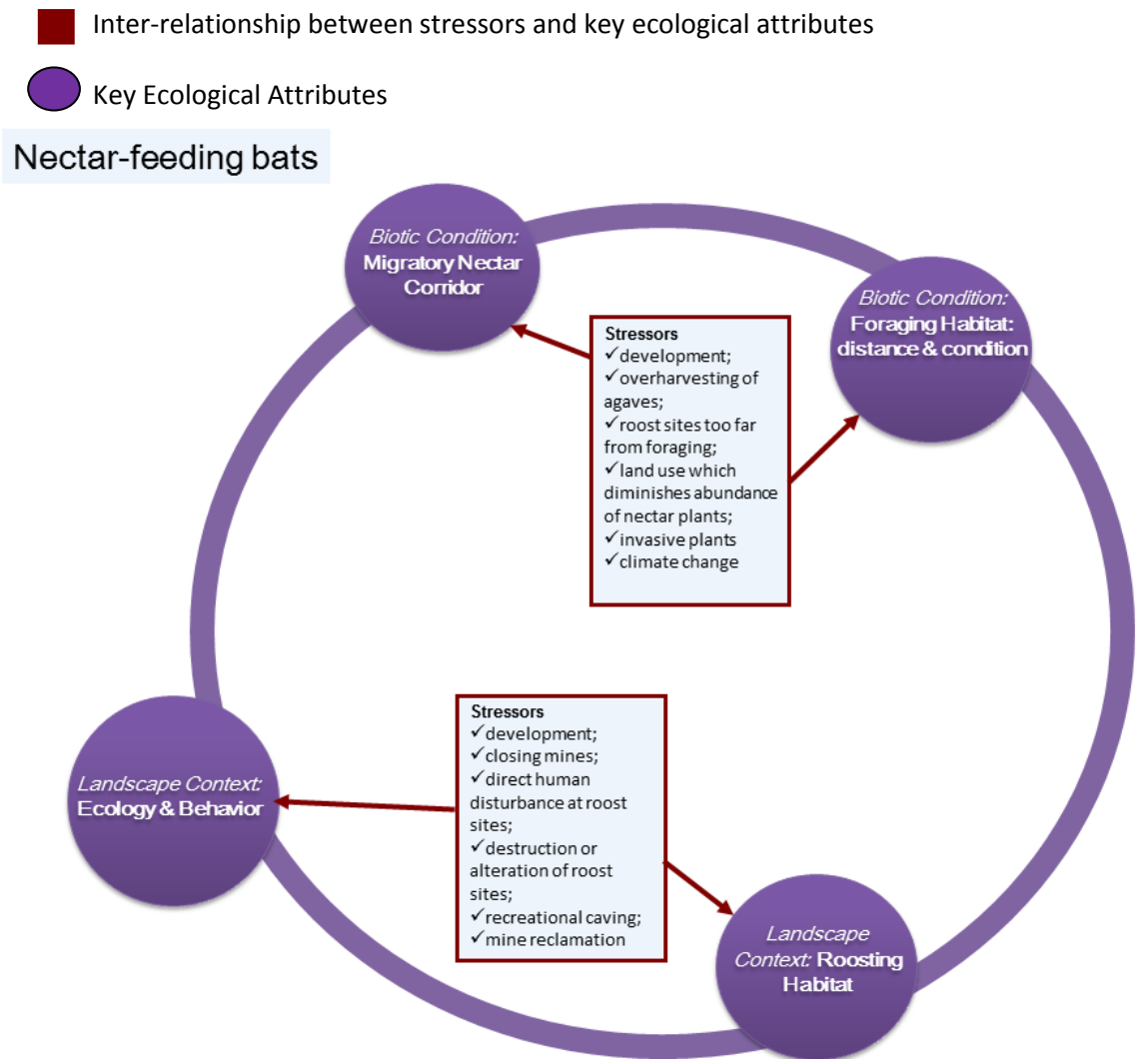
**Table F-13. Key ecological attributes (KEA) of nectar feeding bats: Lesser long-nosed bat (*Leptonycteris yerbabuenae*), Mexican long-nosed bat (*Leptonycteris nivalis*), and Mexican long-tongued bat (*Choeronycteris mexicana*) in the Madrean Archipelago ecoregion.** Indicators for these KEAs can be used to determine the ecological status for this CE; see **Table F-2** for a list of the indicators assessed in this REA.

| ***KEA Class:*** **KEA Name** | **Definition** | **Rationale** | **Stressors** |
|---|---|---|---|
| ***Landscape Context:*** **Roosting Habitat** | The presence of caves, inactive mines, rock crevices and other suitable landscape features for bat roosting. | These species of bat are sensitive to human disturbance at roosting sites. They may have a difficult time finding alternative roosts that meet their requirements. Loss of roosting sites is one of the key factors cited in the population decline of nectar feeding bats (AZGFD 2006; 20011, NMGF 1994, USFWS 2007; 1994). Both daytime and nighttime roost sites from which bats can reach foraging habitat are key to species success in the Madrean ecoregion. | Roosting sites are most threatened by human disturbance and by human alteration of these sites such as the closing of mine shafts or improper gating. Roosting sites may also be compromised or destroyed by natural disasters such as cave collapse. Disturbance at maternity roosts can cause bats to drop their pups to flee from the intruder or cause them to abandon their young altogether (Nolan, n.d.). |
| ***Landscape Context:*** **Ecological / Behavioral Disruption** | The degree to which bats are disturbed by activity in or near roost sites. Mexican long-tongued bats are very sensitive to human disturbance. | Disturbance, vandalism, and outright killing of bats in roosting sites probably have detrimentally affected all three nectar-feeding species (USFWS 1994). Excessive disturbance may lead to increased mortality (e.g., of neonates) and roost abandonment. Activities associated with attempts to control vampire bats in Mexico have resulted in direct mortality and roost abandonment by long-nosed and long-tongued bats, and suitable alternate roost sites for displaced bats may not be available, even if other caves or abandoned mines are present. Bats have low reproductive rates and can recover only slowly from population declines. | Recreational caving, scientific studies in caves or mines, mine reclamation, noise from activity. Disturbance at maternity roosts can cause bats to drop their pups to flee from the intruder or cause them to abandon their young altogether (Nolan, n.d.). |
| ***Biotic Condition:*** **[Distance to & Condition of] Foraging Habitat** | The amount of habitat with columnar cacti and agave nectar plants within foraging distance of known bat roost sites. | Nectar/pollen feeding bats make considerable seasonal and nightly movements. Nightly foraging flights may be as much as 40 km from the roost site, and foraging areas are selected based on past and present signs of high resource availability (many cacti or agaves in an area) and utilized over several nights until the pollen and nectar resources are depleted (U.S. Fish and Wildlife Service 2013). | Destruction of natural habitats and feeding areas due to human development, human commercial use of agave and browsing of agave blooms. Alterations in the floristic composition due to disturbance (e.g., fire, grazing, agave harvest for tequila) |
| ***Biotic Condition:*** **Migratory Nectar Corridors** | The nectar corridor that extends from southern Mexico north to southwestern United States. | In their northward spring migration from Mexico, bats travel along defined corridors - Coastal Lowland Route and Inland Montane Route. During migration they rely on foraging from patches of nectar producing plants to refuel for each subsequent leg of the journey (Arizona-Sonora Desert Museum 2013). | Destruction or modification of habitat supporting nectar plants that bats rely on during migration that would disrupt migratory corridors. |

### F-8.1.7 Conceptual Model Diagram

A conceptual model diagram for each CE provides a visual summary representation of the Key Ecological Attributes and species' life cycle (Figure F-41). These diagrams are intended to show how various stressors interact with categories of Key Ecological Attributes to highlight important drivers for species ecological integrity. The arrows indicate relationships between stressors and but do not indicate the nature of the influence (i.e. positive or negative).

**Figure F-41. Conceptual model diagram for nectar feeding bats, showing key ecological attributes** (by class) for these species, and stressors on the KEAs.




## F-8.2 Considerations and Limitations

There are a number of considerations that were not directly included in the ecological status assessment due to data limitations. Nectar food sources such as agave may be exploited commercially or consumed by livestock. This may be an important management consideration for these species. Bats are very sensitive to human disturbance at roosting sites. They may have a difficult time finding alternative roosts that meet their requirements. Loss of roosting sites is one of the key factors cited in the population decline of nectar feeding bats (AZGFD 2006, 2011, NMGF 2006, USFWS 1994, 2007). Because

of this, ease of access to roost sites and other factors that were not analyzed as part of the ecological status assessment may also be important management considerations. Finally, it is important to note that nectar-feeding bats are migratory and rely on "nectar corridors" to travel from habitat in Mexico to breeding sites in the MAR.

## F-8.3 Ecological Status Assessment Results and Interpretation

This section of the CE summary presents and discusses the results of the ecological status assessment for the nectivorous bat assemblage. It addresses each indicator separately, and then addresses the overall assessment, which integrates the results of all individual indicators assessed for the CE. The results are presented using a common framework, in which the status of an indicator – or the combination of all indicators – is scored on a scale from 0.0 to 1.0, where 0.0 indicates a condition of complete replacement of reference ecological conditions due to the impacts of stressors, and 1.0 indicates a condition of no alteration of reference ecological conditions. The same yellow-to-blue color ramp is used for all results, with yellow representing low scores, green moderate scores, and dark blues high scores.

### F-8.3.1 Current Ecological Status: Development, Fire Regime, Invasives

The results of the assessments for the three individual indicators for the KEAs for the nectivorous bat assemblage are shown in the three maps in Figure F-42 below.

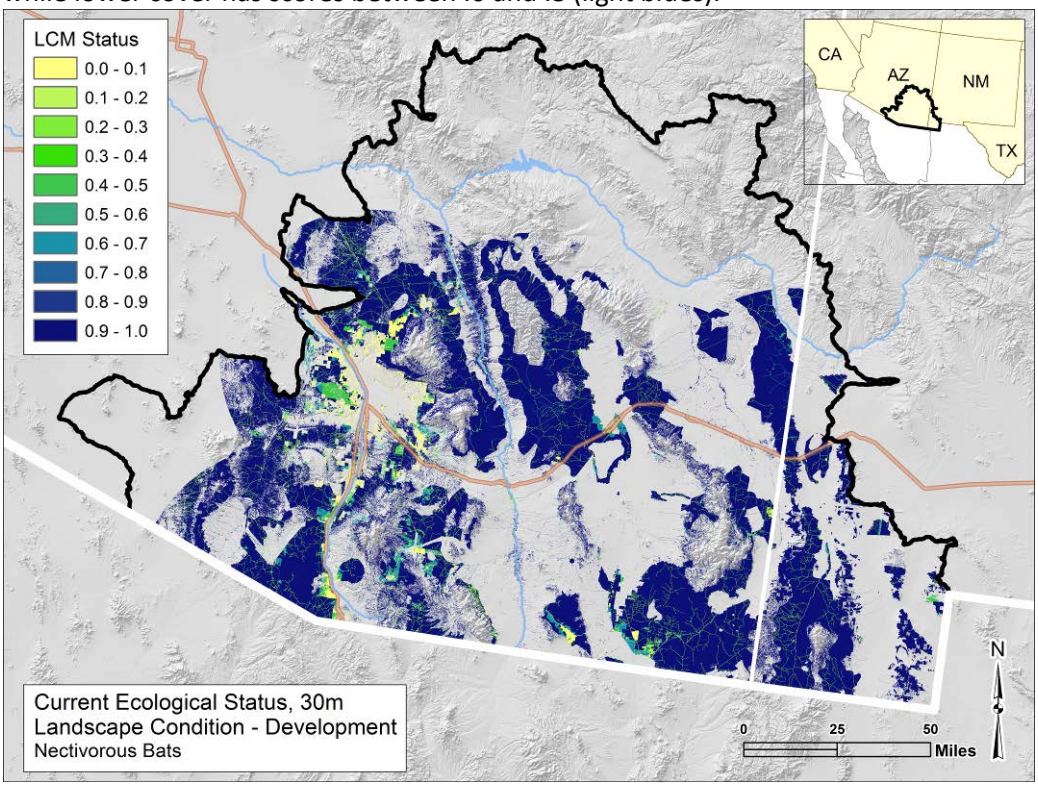
The development indicator results shown in the first map of Figure F-42 reflect a number of discrete areas of development (yellows and greens on the map) across the bat assemblage's current distribution, particularly in the western portions of their distribution. These areas of lower development scores in part reflect a range of development features, including mines, municipalities, and other features. However, most of the bats' distribution in the MAR ecoregion is experiencing little or no impact (dark blue) from the development features reflected in the landscape condition indicator. Where development is present, the primary impacts to bats include direct loss of grassland habitat and potential increased disruption of roosting and other activities due to increased human presence. Transportation features are also visible in the map as networks of crisscrossing lines (yellows and greens) in otherwise blue areas on the map. These are important to note because of the potential for human access within bat habitat to be disruptive to bats.

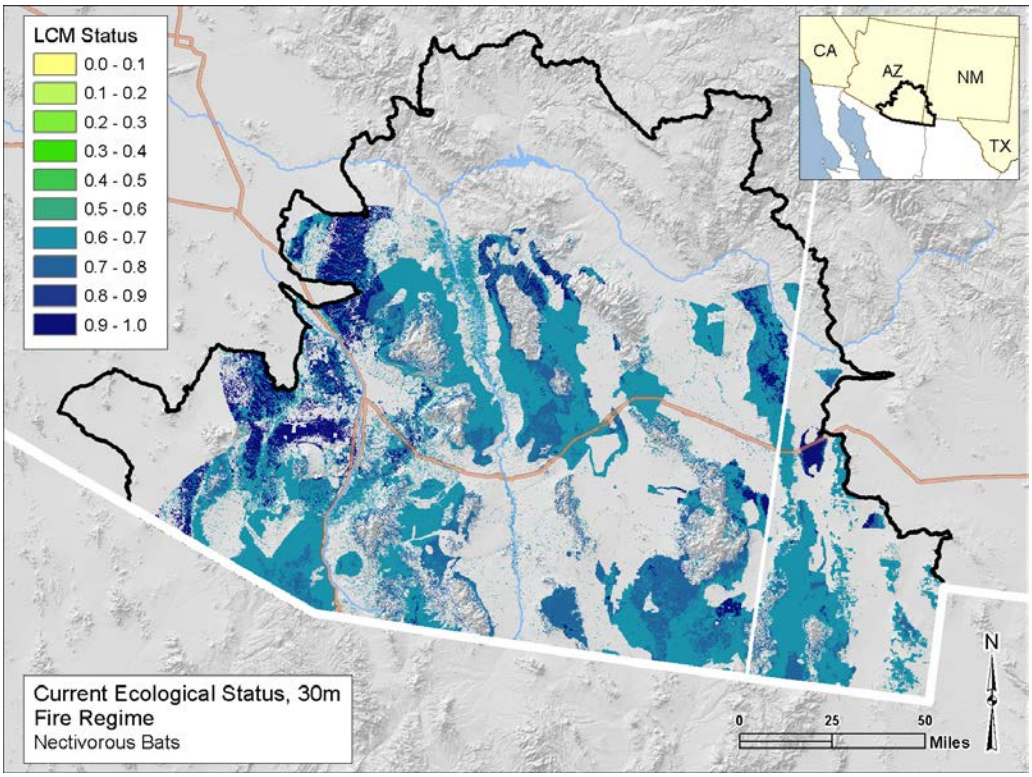
The fire regime indicator results, shown in the second map in Figure F-42, indicate that fire regimes in the ecological systems present in the bat assemblage's distribution are largely split between severe ecological departure (scores of 0.65, shown in turquoise blue) and moderate ecological departure (scores of 0.75); very little of its distribution shows no departure in fire regimes. Alterations in floristic composition within the bat's distribution can occur due to fire. To the degree that fire is maintaining habitat supporting these species' nectar sources (*Agave* spp. and others), the altered fire regime is of potential concern for this assemblage. It may pose a threat of habitat destruction if unnaturally severe fires burn through the area. However, there is not enough information to determine whether altered fire regimes have had any effects on the three bat species.

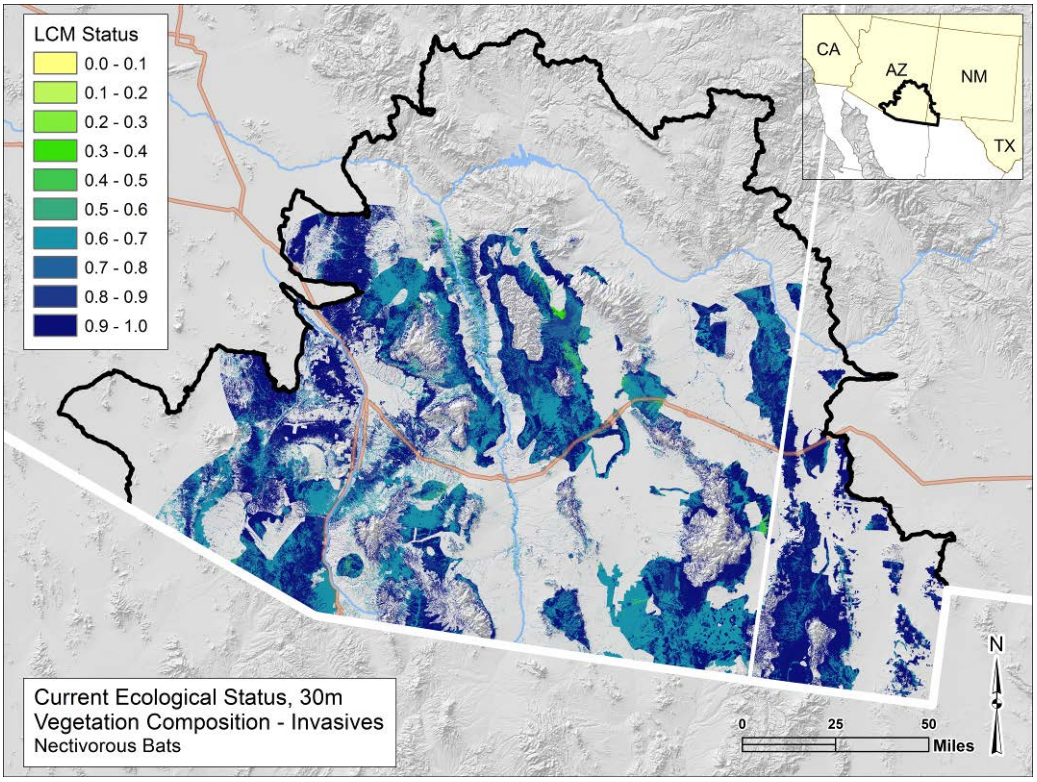
The invasives indicator results (third map in Figure F-42) show that a fair amount of the bat assemblage distribution is impacted either by mesquite or by non-native grasses and forbs (turquoise blue color). In a few areas (greens), both groups of invasives are present at high cover. Based on the compilation of the ILAP data, for this CE's distribution, most of the area showing invasive impacts has high cover of

mesquite (rather than non-native grasses or forbs). Where grasslands that support agave and other forage species for the bats are experiencing significant mesquite encroachment or conversion to mesquite shrubland, there could be potential for loss of the forage species. However, there is not enough information to confirm whether mesquite or other invasives have affected bat habitat quality.

**Figure F-42. Scores for three indicators for nectivorous bats: development indicator (1st map), fire regime departure indicator (2nd map), and invasive species indicator (3rd map) for each 30m pixel.** Yellow (equivalent to 0) indicates high impacts from the CA, dark blue (equivalent to 1) indicates little to no impact from the CA. At the ecoregion scale, many development features are not readily visible (e.g., secondary roads or highways, railroads, small agricultural fields). Only 3 classes of fire regime condition are scored: no to little departure (dark blue), moderate departure and severe departure (lighter blues). For invasives, higher cover of mesquite or invasive exotics have scores between .4 and .6 (light greens), while lower cover has scores between .6 and .8 (light blues).






### F-8.3.2 Current Ecological Status: Full Scenario

The results of the three individual status indicators were combined to get an overall ecological status score, per pixel of the CE's distribution, as shown in the first map of Figure F-43 below. The overall, per-

pixel status scores for the CE were then averaged across each of the 4 km grid cells, as shown in the second map of Figure F-43 below.

When the three sets of indicator scores are aggregated to obtain the overall ecological status scores (as shown in the first map of Figure F-43), much of the habitat for this species is in moderate condition, with status scores around 0.4, 0.5 and 0.6 in many areas, and some discrete areas as low as the 0.1 range (Figure F-44). In visually comparing the maps of scores for the three individual indicators (Figure F-42) to the map of the overall status scores (first map of Figure F-43), the cumulative impacts of the change agents reflected by the indicators are clear. The altered fire regime and mesquite cover are both driving down scores in much of this CE's distribution, and scores drop even lower where development is also present. These same patterns are reflected when the overall status scores are averaged across each 4 km reporting unit, as shown in the second map of Figure F-38 above. Based on the indicators assessed in this REA, changes in habitat quality from altered fire regime and encroachment by native woody increasers are having the most widespread effects on habitat quality for nectivorous bats. Whether these habitat effects are affecting these bat species is unclear. Development features are much more localized, but depending on the type of development, the features may have contributed to direct habitat loss or degradation, as well as increased disruption of bat activities from increased human presence.

**Figure F-43. Overall ecological status scores for nectivorous bat assemblage for all three indicators combined (development, fire regime and invasives) for each 30m pixel (top) and 4km grid cells (bottom).** Yellow scores (equivalent to 0) indicate high impacts from the CAs, dark blue (equivalent to 1) indicate little to no impact from the CAs. In the second map, the score for each 4km cell is an average of the overall ecological status scores of the 30m pixels within the 4km cell that were scored for the CE.

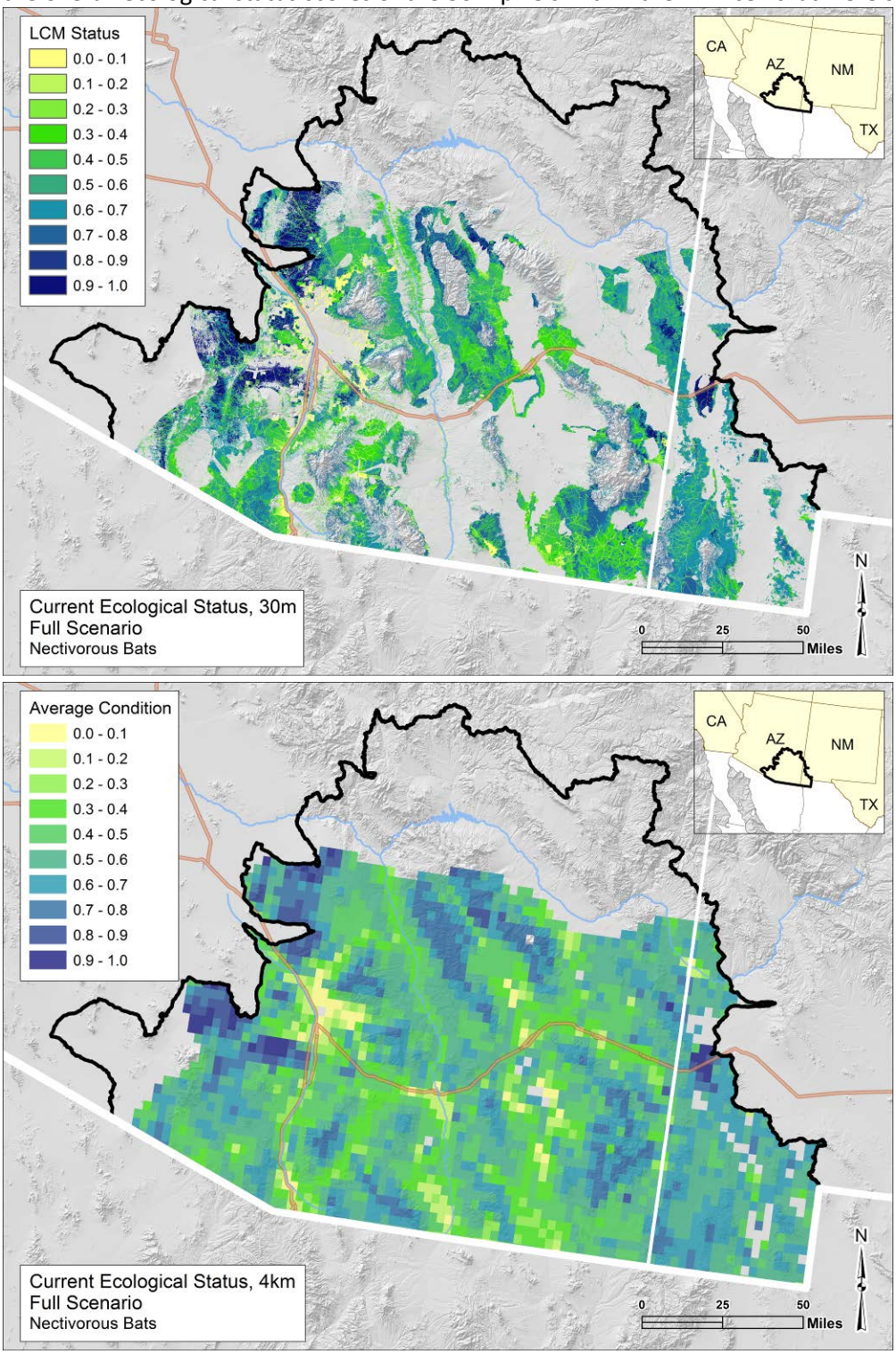
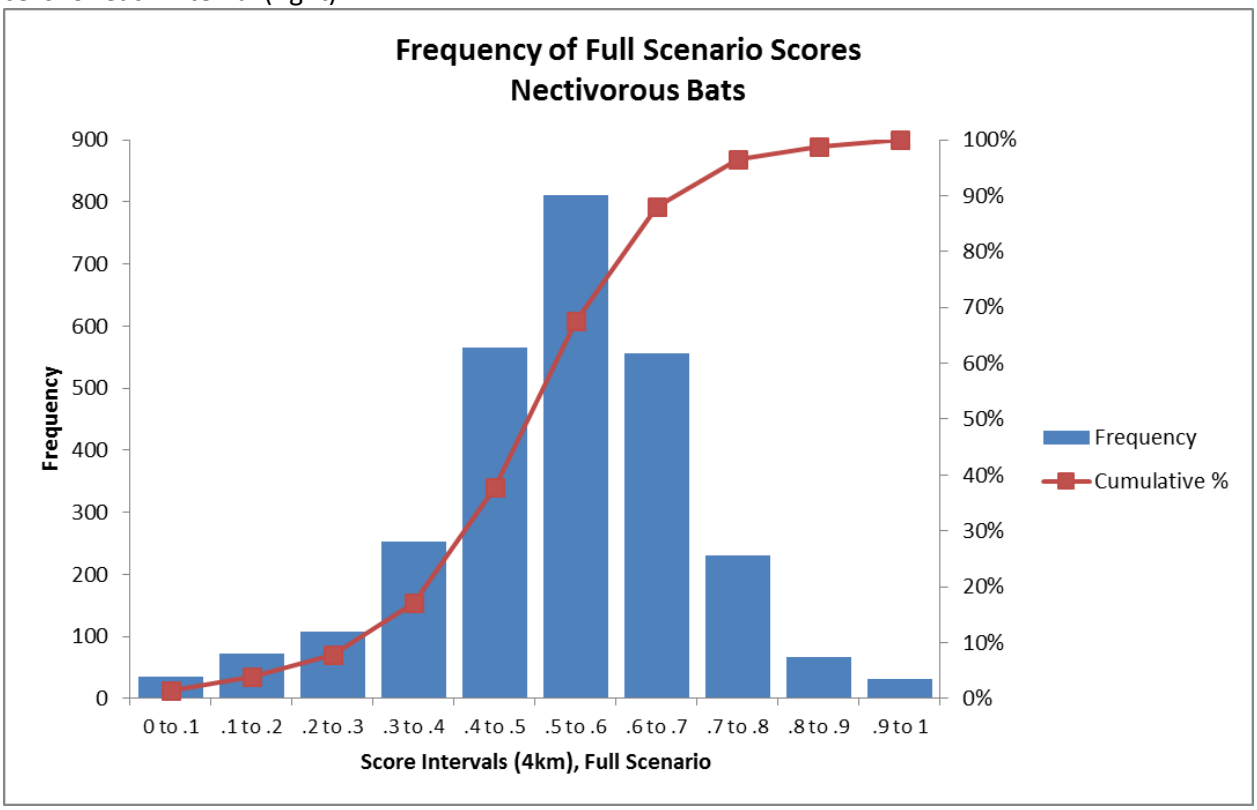


**Figure F-44. Frequency distribution of the 4km ecological status scores for the nectivorous bat assemblage, with cumulative percent.** The x-axis represents the 0.1 increment scoring intervals, while the y-axis shows the number of grid cells in each interval (left) and the cumulative percentage of the grid cells for each interval (right).




## F-8.4 References for the CE


Adams, R. A. 2003. Bats of the Rocky Mountain West: natural history, ecology, and conservation. University Press of Colorado, Boulder, Colorado. xiii + 289 pp.

Altenbach, J. S. 1995. Statement to justify continuation of a permit to study *Leptonycteris curasoae* and *Leptonycteris nivalis* in the Animas Mountains, Hidalgo County, New Mexico. Report submitted to U.S. Fish and Wildlife Service, Albuquerque, New Mexico. 7 pp.

Alvarez, T., and L. Gonzalez-Q. 1970. Análisis polínico del contenido gástrico de murciélagos Glossophaginae de México. Anales del Escuela Nacional Ciencias Biologia, México 18:137-165.

Ammerman, L. K., M. McDonough, N. I. Hristov, and T. H. Kunz. 2009. Census of the endangered Mexican long-nosed bat *Leptonycteris nivalis* in Texas, USA, using termal imaging. Endangered Species Research 8:87-92.

Arita, H. 2005. Murciélago *Leptonycteris curasoae yerbabuenae*. *In*: Ceballos, G and G. Oliva, coords. Los mamíferos silvestres de México. Comisión Nacional para el Conocimiento y Uso de la Biodiversidad. Fondo de Cultura Económica. México. 986pp.

Arita, H. T. 1993. Conservation Biology of the Cave Bats of Mexico. Journal of Mammalogy 74:693–702.

Arita, H. T., and S. R. Humphrey. 1988. Revision taxonomica de los murciélagos magueyeros del genero *Leptonycteris* (Chiroptera: Phyllostomidae). Acta Zoologica Mexicana 29:1-60.

[AZGFD] Arizona Game and Fish Department. 2006. *Choeronycteris mexicana*. Unpublished abstract compiled and edited by the Heritage Data Management System, Arizona Game and Fish Department, Phoenix, AZ. 7 pp.

[AZGFD] Arizona Game and Fish Department. 2011. *Leptonycteris curasoae yerbabuenae*. Unpublished abstract compiled and edited by the Heritage Data Management System, Arizona Game and Fish Department, Phoenix, AZ. 8 pp.

[AZGFD] Arizona Game and Fish Department. 2012. Arizona's State Wildlife Action Plan: 2012-2022. Arizona Game and Fish Department, Phoenix, Arizona.

Arizona-Sonora Desert Museum. 2013. Migratory Pollinators Program. (Online: http://www.desertmuseum.org/pollination/index.php). Accessed July 2013.

[AZGFD] Arizona Game and Fish Department. 2006. *Choeronycteris mexicana*. Unpublished abstract compiled and edited by the Heritage Data Management System, Arizona Game and Fish Department, Phoenix, AZ. 7 pp.

Arroyo-Cabrales, J., R.R. Hollander, and J.K. Jones, Jr. 1987. *Choeronycteris mexicana*. Mammalian Species 291:1-5.

Baltosser, W. H. 1980. A biological inventory of endangered species occurring in Guadalupe Canyon, Hidalgo County, New Mexico. New Mexico Department of Fish and Game Contract 519-68-06. Final report.

Bogan, M. 2005. August Lepto results. Email communication on 9/13/2005 with Angela McIntire, AGFD bat coordinator.

Brylski, P. V., P. W. Collins, E. D. Pierson, W. E. Rainey, and T. E. Kucera.1998. Terrestrial mammal species of special concern in California. Draft final report submitted to California Department of Fish and Game Wildlife Management Division, Nongame Bird and Mammal Conservation Program for Contract No.FG3146WM.

Ceballos, G., T. H. Fleming, C. Chavez, and J. Nassar. 1997. Population dynamics of *Leptonycteris curasoae* (Chiroptera: Phyllostomidae) in Jalisco, Mexico. Journal of Mammalogy 78:1220–1230.

Cocrum, E. L. 1991. Seasonal distribution of northwestern populations of the long-nosed bats, *Leptonycteris sanborni* family Phyllostomidae. Anales del Instituto de Biologia Universidad Nacional Auto´noma de México, Series Zoologica 62:181–202.

Cockrum, E. L., and Y. Petryszyn. 1991. The long-nosed bat, Leptonycteris: an endangered species in the southwest? Occasional Papers Museum Texas Tech University (142):1-32.

Cole, F. R., and D. E. Wilson. 2006. *Leptonycteris yerbabuenae*. Mammalian Species 797:1-7.

Constantine, D. G. 1987. Long-tongued bat and spotted bat at Las Vegas, Nevada. Southwestern Naturalist 32:392.

Cook, J. A. 1986. The mammals of the Animas Mountains and adjacent areas, Hidalgo County, New Mexico. Museum of Southwestern Biology Occasional Papers 4:1-45.

Cruse-Sanders, J. 2005. The pitaya connection: Ferrying cactus pollen across central Mexico. Bat Conservation International. Vol. 23, No. 1.

Cryan, P. 2012. Conservation studies of long-nosed bats in New Mexico. <http://www.fort.usgs.gov/Research/research_tasks.asp?TaskID=2149>

Cryan, P. M., and M. A. Bogan. 2003. Recurrence of the Mexican long-tongued bat (*Choeronycteris mexicana*) at historical sites in Arizona and New Mexico. Western North American Naturalist 63:314-319.

Davis, W. B., and D. C. Carter. 1962. Review of the genus *Leptonycteris* (Mammalia: Chiroptera). Proceedings Biological Society Washington 75:193-197.

De La Mar, S. 2001. Neotropical Nectar-feeding Bats (Family Phyllostomidae) Revisited: Lingual Data Support a Recently-Proposed Molecular Phylogeny. Honors Projects. Paper5. (Online: http://digitalcommons.iwu.edu/bio_honproj/5). Accessed July 2013.

Easterla, D. A. 1972. Status of *Leptonycteris nivalis* (Phyllostomatidae) in Big Bend National Park, Texas. Southwestern Naturalist 17:287-292.

England, A. E. 2004. Radio-tracking of the greater long-nosed bat, *Leptonycteris nivalis*, in Big Bend National Park, Texas. Bat Research News 40:95–96.

Findley, J. S., A. H. Harris, D. E. Wilson, and C. Jones. 1975. Mammals of New Mexico. University of New Mexico Press, Albuquerque. 360 pp.

Fleming, T. H. 2012. Midnight Raiders of Arizona: Monitoring nectar bats at hummingbird feeders. Bat Conservation International. Vol. 30, No. 1.

Frey, J. K. 2004. Taxonomy and distribution of the mammals of New Mexico: an annotated checklist. Occasional Papers, Museum of Texas Tech University Number 240. Lubbock, Texas, USA.

Harvey, M. J., J. S. Altenbach, and T. L. Best. 2011. Bats of the United States and Canada. Johns Hopkins University Press, Baltimore.

Hensley, A. P., and K. T. Wilkins. 1988. Leptonycteris nivalis. Mammalian Species 307:1-4.

Hinman, K. E., and T. K. Snow, editors. 2003. Arizona Bat Conservation Strategic Plan. Nongame and Endangered Wildlife Program Technical Report 213. Arizona Game and Fish Department, Phoenix, Arizona.

Hoffmeister, D. F. 1986. Mammals of Arizona. University of Arizona Press and Arizona Game and Fish Department. 602 pp.

Howell, D.J. 1979. Flock foraging in nectar-feeding bats: advantages to the bats and the host plants. The Amer. Nat. 114(1):23-49.

Hoyt, R. A., J. S. Altenbach, and D. J. Hafner. 1994. Observations on the long-nosed bats (*Leptonycteris*) in New Mexico. Southwestern Naturalist 39:175-179.

Medellín, R.A. 2005. Comments submitted 4/27/05 and 10/5/05, in response to Federal Register Notice of Review (70 FR 5460) for the lesser long-nosed bat (*Leptonycteris curasoae yerbabuenae*).

Moreno-Valdez, A., R. L. Honeycutt, and W. E. Grant. 2004. Colony dynamics of *Leptonycteris nivalis* (Mexican long-nosed bat) related to flowering *Agave* in northern Mexico. Journal of Mammalogy 85:453–459.

Mumford, R. E., L. L. Oakley, and D. A. Zimmerman. 1964. June bat records from Guadalupe Canyon, New Mexico. Southwestern Naturalist 9:43-45.

NMDGF [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.

NMDGF [New Mexico Department of Game and Fish]. 2014. Threatened and Endangered Species of New Mexico: 2014 Biennial Review. New Mexico Department of Game and Fish, Conservation Services Division, Albuquerque, New Mexico. 149 pp.

Nolan, V. P. (n.d.) "Management Initiatives for the Sustainability of Bat Populations" (Online: http://users.erols.com/nolan/vivian/bats.htm). Accessed July 2013.

Ortega, J. and H. Arita. 2005. Murciélago- *Choeronycteris mexicana*. *In*: Ceballos, G and G. Oliva, coords. Los mamíferos silvestres de México. Comisión Nacional para el Conocimiento y Uso de la Biodiversidad. Fondo de Cultura Económica. México. 986pp.

Rojas-Martinez, A., A. Valiente-Banuet, M. del Coro Arizmendi, A. Alcantara-Eguren, and H. T. Arita. 1999. Seasonal distribution of the long-nosed bat (*Leptonycteris curasoae*) in North America: does a generalized migration pattern really exist? Journal of Biogeography 26:1065-1077.

Schmidly, D. J. 1991. The bats of Texas. Texas A & M University Press, College Station, Texas. 188 pp.

Schmidly, D. J. 2004. The mammals of Texas. Revised edition. University of Texas Press, Austin. xviii + 501 pp.

Simmons, N. B., and A. L. Wetterer. 2002. Phylogeny and convergence in cactophilic bats. Pages 87–121 in T. H. Fleming and A. Valiente-Banuet, editors. Columnar cacti and their mutualists: evolution, ecology, and conservation. University of Arizona Press, Tucson.

Texas Parks and Wildlife. 2013. Mexican long-nosed bat (*Leptonycteris nivalis*). http://www.tpwd.state.tx.us/huntwild/wild/species/mexlongnose/ (accessed 29 August 2013).

[USFWS] U.S. Fish and Wildlife Service. 1994. Mexican long-nosed bat (*Leptonycteris nivalis*) recovery plan. U.S. Fish and Wildlife Service, Albuquerque, New Mexico. 91 pp.

[USFWS] U.S. Fish and Wildlife Service. 1995. Lesser long-nosed bat recovery plan. U.S. Fish and Wildlife Service, Albuquerque, New Mexico., New Mexico. 45 pp.

[USFWS] U.S. Fish and Wildlife Service. 2007. Lesser long-nosed bat. Five-year review: summary and evaluation. U.S. Fish and Wildlife Service, Phoenix, Arizona.

[USFWS] U.S. Fish and Wildlife Service. 2013. Lesser long-nosed bat (Leptonycteris curasoae yerbabuenae) species profile. Environmental Conservation Online System. (Online: http://criticalhabitat.fws.gov/speciesProfile/profile/speciesProfile.action?spcode=A0AD). Accessed July, August, 2013.

Van de Water, P. K., and W. D. Peachey. 1997. Dietary analysis of the Mexican long-tongued bat (Choeronycteris mexicana) using pollen analysis of guano collected at Cienaga Creek Natural Preserve. Bat Research News 38:181.

Villa-R., B. 1967. Los murciélagos de México. Instituto de Biología, Universidad Nacional Autónoma de México, Distrito Federal, México, 491 pp.

Wilkinson, G. S., and T. H. Fleming. 1996. Migration routes and evolution of lesser long-nosed bats, Leptonycteris curasoae, inferred from mitochondrial DNA. Molecular Ecology 5:329–339.

Wilson, D. E. 1985. Status report: Leptonycteris nivalis (Saussure) Mexican long-nosed bat. Prepared for: Office of Endangered Species, U. S. Fish and Wildlife Service. 35 pp.

Wolf, S. No date. Nectar eating bats in southern Arizona. (Online: http://wildbirdsonline.com/articles_nectar_eating_bats.html). Accessed July, Aug 2013.

# References for Appendix F


Comer, P. J., and J. Hak. 2009. NatureServe landscape condition model. Internal documentation for NatureServe Vista decision support software engineering, prepared by NatureServe, Boulder, CO.

Comer, P. and D. Faber-Langendoen. 2013. Assessing Ecological Integrity of Wetlands from National to Local Scales: Exploring the Predictive Power, and Limitations, of Spatial Models. National Wetlands Newsletter Special Issue on Wetland Mapping and Assessment. Environmental Law Institute. Washington DC. Vol. 35 No. 3 May/June 2013.

De Leo, G. A., and S. Levin. 1997. The multifaceted aspects of ecosystem integrity. Conservation Ecology [online]1(1): 3. Available from the Internet. URL: http://www.consecol.org/vol1/iss1/art3/

Faber-Langendoen, D., J. Rocchio, M. Schafale, C. Nordman, M. Pyne, J. Teague, T. Foti, and P. Comer. 2006. Ecological Integrity Assessment and Performance Measures for Wetland Mitigation. Final Report to US EPA Office of Water and Wetlands. NatureServe, Arlington, VA.

Harkness, M., M. Reid, P. Crist, L. Misztal, T. Van Devender, G. Kittel, D. Braun, and R. Unnasch. 2013. Madrean Archipelago Rapid Ecoregional Assessment: Pre-Assessment Report. Prepared for the U.S. Department of the Interior, Bureau of Land Management. NatureServe, Boulder, CO.

Holling, C. S. 1973. Resilience and stability of ecological systems. Annual Review of Ecology and Systematics 4: 1-23.

Jenkins, R. E. 1976. Maintenance of natural diversity: approach and recommendations. In: K. Sabol (ed.) Transactions–Forty -first North American Wildlife and Natural Resources Conference. Washington, D. C. March 21-25, 1976. Pp. 441-451.

NatureServe. 2013a. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia. Online:http://www.natureserve.org/explorer. Accessed: 23 March 2013.

NatureServe. 2013b. Terrestrial Ecological Systems of the Conterminous United States. Version 2.9. Completed in cooperation with USGS Gap Analysis Program and inter-agency LANDFIRE. Reflecting early 2000s land cover and MMU approx. 2 hectares. NatureServe, Arlington, VA, USA. Digital map.

Nachlinger, J., K. Sochi, P. Comer, G. Kittel, and D. Dorfman. 2001. Great Basin: an ecoregion-based conservation blueprint. The Nature Conservancy, Reno, NV. 160 pp. + appendices.

Noss, R. F. 1987. From plant communities to landscapes in conservation inventories: A look at The Nature Conservancy (USA). Biological Conservation 41:11-37.

Noss, R. F., C. Carroll, K. Vance-Borland, and G. Wuerthner. 2002. A multicriteria assessment of the irreplaceability and vulnerability of sites in the Greater Yellowstone Ecosystem. Conservation Biology 16(4): 895-908.

Parrish, J. D., D. P. Braun, and R.S. Unnasch. 2003. Are we conserving what we say we are? Measuring ecological integrity within protected areas. Bioscience 53(9): 851-860.

Rollins, M.G., B.C. Ward, G. Dillon, S. Pratt, and A. Wolf. 2007. Developing the LANDFIRE Fire Regime Data Products. Documentation available on-line at: http://www.landfire.gov/documents_vcc.php

State Wildlife Agencies of the Western United States. 2014. West-wide Crucial Habitat Data Set. Western Governors' Crucial Habitat Assessment Tool: Mapping Fish and Wildlife Across the West. Western Governors' Association. Accessed June 2014. http://www.westgovchat.org

Unnasch, R.S., D. P. Braun, P. J. Comer, G. E. Eckert. 2009. The ecological integrity assessment framework: A framework for assessing the ecological integrity of biological and ecological resources of the National Park System**.** Report to the National Park Service.

[USFWS] U.S. Fish and Wildlife Service. 5 August 2008. Designation of critical habitat for the Sierra Nevada bighorn sheep (*Ovis canadensis sierrae*) and taxonomic revision. Federal Register 73(151):45534-45604.

[USFWS] U.S. Fish and Wildlife Service. 2011. Chiricahua Leopard Frog (*Lithobates* [=*Rana*] *chiricahuensis*). 5-Year Review. Phoenix, Arizona.

# Appendix G Ecoregional Conceptual Model and Ecological Integrity Assessment

Version December 8, 2014

# Contents

Appendix G Ecoregional Conceptual Model and Ecological Integrity Assessment ..... 1
G.1 Overview of Appendix G ..... 5
G.2 Ecoregion Conceptual Model ..... 5
G.2.1 Overview of Ecoregion Conceptual Model ..... 5
G.2.2 Assessment Area for the Madrean Archipelago REA ..... 5
G.2.3 Biophysical Controls ..... 9
G.2.4 Ecosystem Processes ..... 15
G.2.5 Ecoregion Conceptual Model Diagrams ..... 20
G.2.6 Biodiversity ..... 26
G.3 Human Context ..... 33
G.3.1 Demographic Overview ..... 33
G.3.2 Land Ownership ..... 34
G.3.3 Land Uses ..... 36
G.3.4 Current Issues in the Madrean Archipelago Ecoregion ..... 36
G.3.5 Considerations for Ecological Integrity ..... 37
G.4 Assessment of Ecological Integrity ..... 39
G.4.1 Methods ..... 39
G.5 Results: Ecological Integrity Assessment ..... 48
G.5.1 Ecological Integrity for the Uplands of the Ecoregion ..... 48
G.5.2 Ecological Integrity for the Aquatic and Wetland Resources of the Ecoregion ..... 67
G.6 References ..... 72

# Tables

Table G-1. Percentage of the Madrean Archipelago ecoregional assessment area in major categories of land ownership. ..... 34
Table G-2. List of Key Ecological Attributes (KEAs) and indicators for ecological integrity of the ecoregion. ..... 38
Table G-3. Life zones defined for ecological integrity assessment of the MAR; their definitions, indicators measured, and reporting units. ..... 40
Table G-4. List of component NatureServe ecological systems for each of the upland Life Zones with total area and percent of MAR. ..... 40

Table G-5. Site intensity values and distance effects used in the Landscape Condition Model for assessing ecological integrity of the upland and aquatic life zones of the MAR. ..... 44

Table G-6. Comparison of historical distribution (pre-European settlement) and current distribution (circa 2000) for upland vegetated ecological systems in the MAR ecoregion. ..... 65

# Figures

Figure G-1. Map of the bi-national Madrean Archipelago ecoregion. ..... 7

Figure G-2. Map of the Madrean Archipelago REA assessment area. ..... 8

Figure G-3. Average monthly temperature and precipitation in the southeastern Arizona Planning Area (see inset), 1930-2002. ..... 11

Figure G-4. Climate diagrams showing average monthly temperature and precipitation from four weather stations in or near the MAR ecoregion. ..... 12

Figure G-5. Maps showing locations of U.S. Climate Divisions that overlap with the MAR assessment area. ..... 13

Figure G-6. Winter (Nov-April) precipitation (20-year moving averages) for southern Arizona dating from 1000 A.D. to 1988, reconstructed from tree rings ..... 14

Figure G-7. Winter (Nov-April) precipitation records relative to normal for 1949-2013, southwestern New Mexico. ..... 15

Figure G-8. Map of major rivers and 5th-level watersheds (shown in yellow outlines) in the Madrean Archipelago assessment area (shown in red outline). ..... 16

Figure G-9. Conceptual diagram for the Madrean Archipelago ecoregion ..... 21

Figure G-10. Conceptual diagram for the montane uplands division of the Madrean Archipelago .. 23

Figure G-11. Conceptual diagram for the valley uplands division of the Madrean Archipelago ..... 24

Figure G-12. Conceptual diagram for the connected stream and wetland division of the Madrean Archipelago ..... 25

Figure G-13. Conceptual diagram for the isolated wetland division of the Madrean Archipelago .... 26

Figure G-14. Map of vegetation classes and other land cover types present in the Madrean Archipelago ecoregion. ..... 28

Figure G-15. Map illustrating land ownership in the Madrean Archipelago ecoregional assessment area. ..... 35

Figure G-16. Watersheds of the MAR used for the terrestrial change in extent analysis. ..... 47

Figure G-17. Distribution of the Terrestrial Desert Scrub life zone at 30 m resolution in the Madrean Archipelago ecoregional assessment area. ..... 49

Figure G-18. Current ecological integrity scores for Terrestrial Desert Scrub Life Zone for all indicators combined (development, fire regime and invasives) for each 30m pixel (top) and 4km grid cells (bottom). ..... 49

Figure G-19. Distribution of the Terrestrial Valley Grassland Life Zone at 30 m resolution in the Madrean Archipelago ecoregional assessment area. 52

Figure G-20. Current ecological integrity scores for Terrestrial Valley Grassland Life Zone for all indicators combined (development, fire regime and invasives) for each 30m pixel (top) and 4km grid cells (bottom). 53

Figure G-21. Distribution of the Terrestrial Montane Forest Life Zone at 30 m resolution in the Madrean Archipelago ecoregional assessment area. 55

Figure G-22. Current ecological integrity scores for Terrestrial Montane Forest Life Zone for all indicators combined (development, fire regime and invasives) for each 30m pixel (top) and 4km grid cells (bottom). 56

Figure G-23. Current ecological integrity scores for all three Terrestrial Life Zones combined into a single map for the ecoregion. 57

Figure G-24. Historical vs. Current Abundance of select terrestrial ecological systems for the entire MAR ecoregion. 59

Figure G-25. Historical vs. Current abundance of select terrestrial ecological systems for the upper Santa Cruz watershed. 60

Figure G-26. Historical vs. Current abundance of select terrestrial ecological sytems for the Lower San Pedro watershed. 61

Figure G-27. Historical vs. Current abundance of select terrestrial ecological sytems for the Upper Gila-San Carlos watershed. 63

Figure G-28. Historical vs. Current abundance of select terrestrial ecological sytems for the Playas Lake, NM watershed. 64

Figure G-29. Distribution of Aquatic Lowland Life Zone; all areas of the MAR below 5000 feet (1524 m). 67

Figure G-30. Ecological integrity scores for Aquatic Lowland Life Zone 69

Figure G-31. Distribution of the Aquatic Montane Life Zone in the Madrean Archipelago ecoregional assessment area. 70

Figure G-32. Ecological integrity scores for Aquatic Montane Life Zone 71

## *G.1 Overview of Appendix G*

This appendix contains the conceptual model for the Madrean Archipelago ecoregion, methods for the assessment of integrity, and the results of the assessment of ecological integrity for the ecoregion. Appendix A describes the methods for selection of the component conservation elements (CEs) and the change agents (CAs) of the ecoregion, as well as the collection and organization of management questions (MQs) of interest to many partners active in this ecoregion. Appendices B and C contain the assessment methods: B contains the methodological approaches to the geospatial assessments, including for the ecological integrity, while C contains the technical GIS documentation. Other appendices contain the conceptual models and ecological status assessment results for the terrestrial CEs (Appendix D), aquatic CEs (Appendix E) and species CEs (Appendix F). Appendix H contains the conceptual model for the Mesquite Upland Scrub, along with the results of the special assessment for that.

The content of this appendix is organized into two major sections. The first section provides the complete ecoregional conceptual model. Much of the content is from Harkness et al. (2013), with some modest revisions. The second section contains the methods for conducting the integrity assessment, and the results for the ecoregion.

## *G.2 Ecoregion Conceptual Model*

### G.2.1 Overview of Ecoregion Conceptual Model

The purpose of the ecoregion conceptual model is to describe the ecoregion's defining physical characteristics and processes, biodiversity and associated processes, anthropogenic context, and the interactions and relationships among these in order to understand the Madrean Archipelago ecoregion as a whole. The ecoregion conceptual model, in conjunction with the characterization of issues currently facing resource managers in the ecoregion (per Appendix A), together provide the context for the selection of conservation elements (CEs) and change agents (CAs).

The ecoregion conceptual model has two components: narrative text describing the ecoregion's features and characteristics, and a series of diagrams that visually illustrate the relationships between these features. This characterization of the ecoregion also includes the identification of key ecological attributes or indicators of the ecological integrity of the ecoregion; these indicators are used to assess the ecological integrity of the Madrean Archipelago ecoregion as a whole.

The ***temporal bounds*** of this conceptual model include the past two centuries, but center on the 20$^{th}$ century and the recent decades. This time period reflects the climatic regimes, ecological patterns and processes, and change agents that are most applicable to this assessment. Although the REA evaluates climate-induced stress and land use scenarios for future time periods, the overarching ecoregional conceptual model is based on knowledge and assumptions up to the present.

The anthropogenic context is a fundamental component of the conceptual model, as previously noted, and some material pertinent to that is contained within the **Human Context** section below, and expanded on in more detail in the **Current Issues** chapter of Harkness et al. (2013).

### G.2.2 Assessment Area for the Madrean Archipelago REA

The Madrean Archipelago (MAR) ecoregion is approximately 7.5 million hectares (18.5 million acres) and spans portions of four states in two countries: southeastern Arizona and extreme southwestern New

Mexico in the United States, and northeastern Sonora and northwestern Chihuahua in Mexico. As defined for North America by the Commission for Environmental Cooperation[1] (CEC 1997), this ecoregion lies to the immediate east of the Sonoran Desert, to the south of the Arizona/New Mexico Mountains, to the west of the Chihuahuan Desert, and to the north of two ecoregions entirely within Mexico: the Sinaloa and Sonora Hills and Canyons with Xeric Shrub and Low Tropical Deciduous Forest, and the Sierra Madre Occidental with Conifer, Oak, and Mixed Forests (Figure G-1).

The conceptual models for the Madrean Archipelago ecoregion (MAR) and the CEs draw on literature for the entirety of this bi-national ecoregion, as appropriate; for example, the CE conceptual model narratives typically discuss the CE across its range. However, the geospatial data and assessments only address the U.S. portion of the ecoregion. Defining the assessment area for the MAR followed the BLM REA standard methods. The ecoregion boundary (CEC 1997) is the core of the assessment area but to ensure that influences affecting the periphery of the ecoregion are included in the assessments, the U.S. portion of the ecoregion was buffered with intersecting 5$^{th}$-level watersheds to define the geographic area assessed in this REA. The REA assessment area (inclusive of the overlapping watersheds) encompasses approximately 6.4 million hectares (15.7 million acres) (Figure G-2).

[1]The CEC was established in 1994 by Canada, Mexico, and the United States to implement the North American Agreement on Environmental Cooperation (NAAEC), the environmental side accord to the North American Free Trade Agreement.

**Figure G-1. Map of the bi-national Madrean Archipelago ecoregion.** The boundary of the Madrean Archipelago ecoregion is shown with the solid, dark green line (this is the unbuffered boundary of the ecoregion); it is located in southeastern Arizona, southwestern New Mexico in the U.S., and northeastern Sonora and northwestern Chihuahua in Mexico.

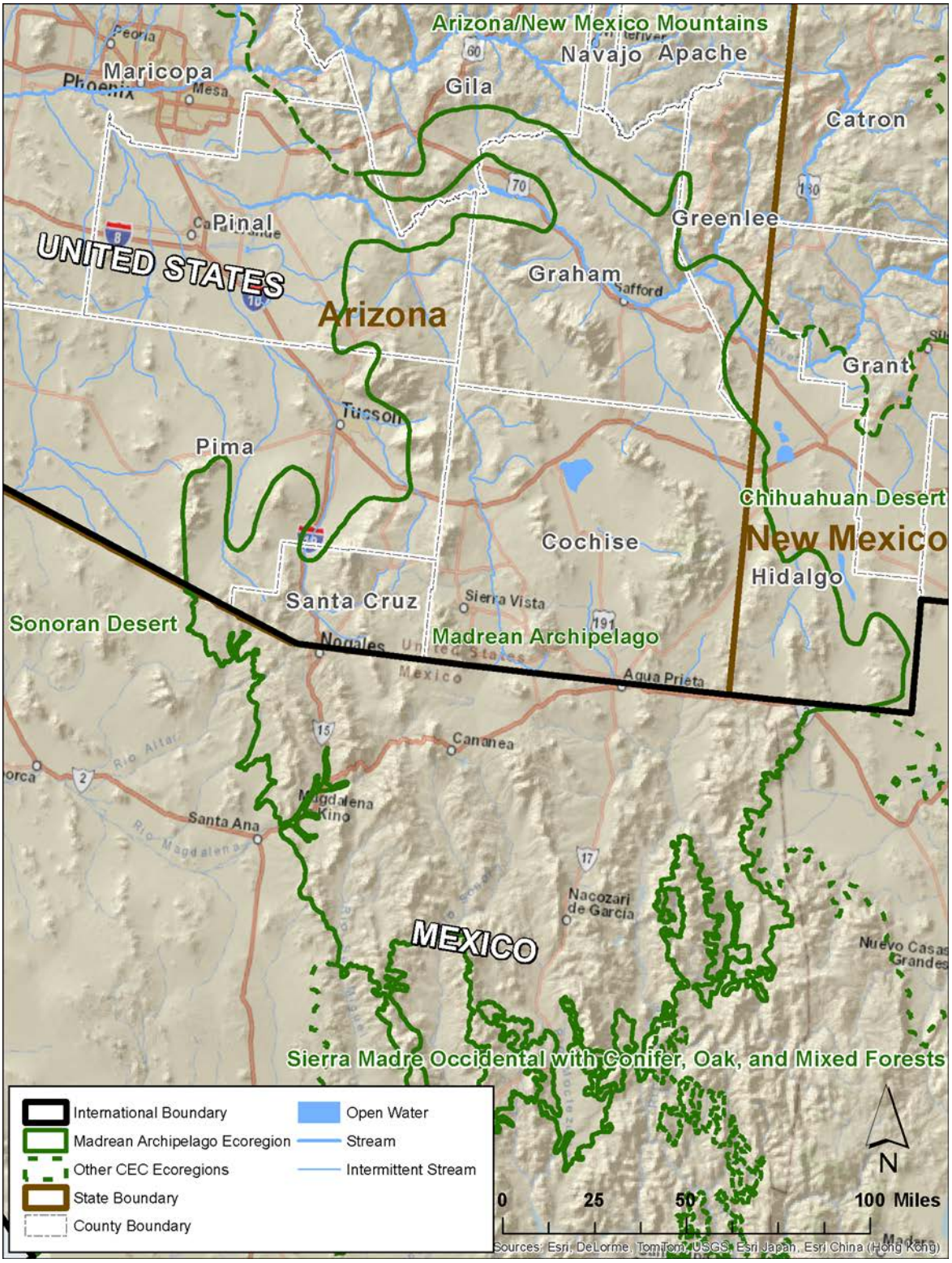
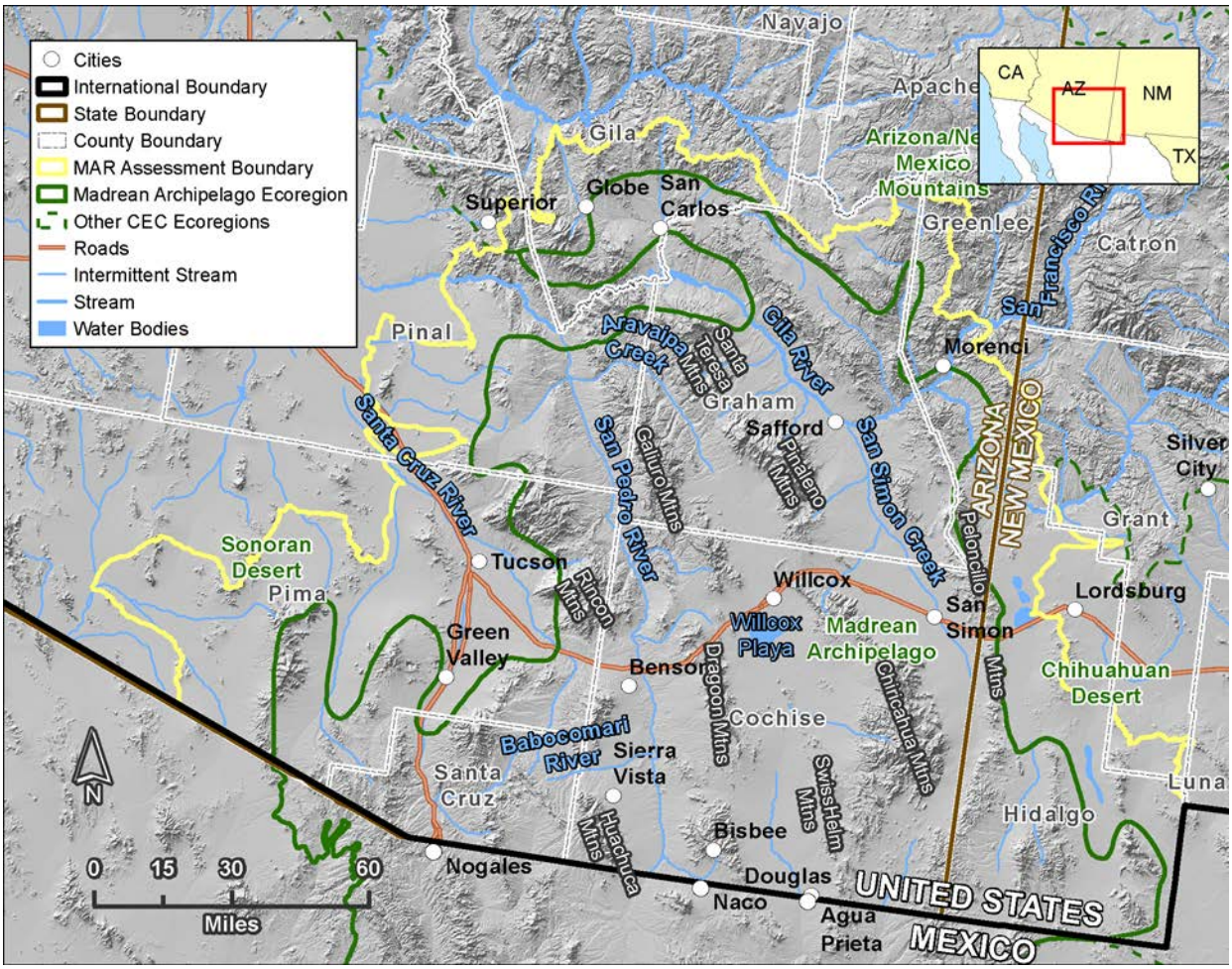


**Figure G-2. Map of the Madrean Archipelago REA assessment area.** The area assessed for this REA is the U.S. portion of the Madrean Archipelago **plus** its intersecting 5th-level watersheds, shown in the yellow outline and by the border between the U.S. and Mexico. The Madrean Archipelago ecoregion is shown by the solid, green line and extends into Mexico beyond the map extent.



## G.2.3 Biophysical Controls

### G.2.3.1 Physiography and Geology

The sky islands of the Madrean Archipelago are unique in many respects, including physiography and geologic history. The mountains of the sky islands are deformational, resulting from extensional forces of continental rifting that began some 13 million years ago. These are "horst/graben" landforms, wherein the mountains did not rise so much as the valleys sank, thereby creating a landscape of parallel sequences of valleys and mountains. This basin and range topography is somewhat unique to North America, covering vast areas of the southwestern United States and south into Mexico (Wiken et al. 2011). Overlaid on this topography are degradational landforms, such piedmonts left by erosion, and constructional landforms, such as alluvial fans.

The "horst/graben" development exposed older rocks derived from a highly diverse geologic past: multiple marine invasions, caldera explosions and lava flows, and metamorphic core complexes (Warshall 1995). The individual mountain ranges of the Madrean Archipelago ecoregion often have a complex mixture of rock types exposed: intrusive igneous rocks (granite), extrusive volcanics (rhyolite, dacite, basalts), metamorphics (gneiss, schists, quartzite), and sedimentary rocks (limestones, shale, conglomerates). Some (such as the Chiricahuas) are individual volcanoes; others (e.g., Huachucas) are predominantly limestones; and still others (e.g., the Pinaleños, the Santa Catalinas) are metamorphics with cores of gneiss and granite.

Most of the lower elevations and valleys of this area are covered by deep alluvium washed down from the adjacent mountains. These deposits of silt, sand, and gravel are very young in the present-day drainages and much older on the valley floors and terraces (NRCS 2006). The spatially and temporally discontinuous deposition of alluvium has resulted in a mosaic of different aged and applied alluvium. Deeply incised landforms result in extreme topographical relief (canyons), or the terraced alluvium of the larger river valleys (McAuliffe and Burgess 1995).

Coblentz and Riitters (2004) found that the high biodiversity of the Madrean Archipelago can be correlated to two factors of physiography: the topographic relief and the northwest to southeast trends of the major mountain ranges. The local-scale relief results in the compression of biotic communities into relatively narrow vertical space (Brown 1982) and also contributes to rapid species turnover (McLaughlin 1994). In addition, the compressed relief encourages the interaction of species that would normally be widely separated (Felger and Wilson 1995). Secondly, the overall northwest-to-southeast orientation of the mountain ranges through the region encourages the movement of both plant and animal species from the southern, neotropical regions into the more northerly temperate zone. These two factors are considered by the authors (Coblentz and Riitters 2004) to be the most important geographical factors contributing to the high biodiversity of the MAR. When combined with the diversity of parent materials (geologic substrates), the physiography results in high diversity in species, ecosystems, and biotic communities.

In addition, there are other geologically derived, climatically shaped habitats that increase biotic diversity. These include aerosol-derived caliche soils, marine clay-rich valleys in Mexico and the San Rafael Valley, remnant sand dunes in the Animas and other valleys, small seeps in the Galiuros, and the morainal-related ciénegas of the Pinaleños from the last glaciation (Warshall 1986, 1995).

### G.2.3.2 Soils

The variability of soils can contribute to the patterns of vegetation found in an area. Geology plays a role by providing the base parent materials in which soils form and hence the soil mineralogy and chemistry,

the structural and textural characteristics, and other properties important to plants. The climate – temperature and precipitation patterns over thousands of years – also influences soil formation. The interactions of climate and parent materials result in a complex mosaic of soils across a landscape, which in turn contribute to the patterns of vegetation at both regional and local scales. Soil characteristics, including depth, moisture, temperature, texture, structure, cation exchange capacity, base saturation, clay mineralogy, organic matter content, and salt content influence the abundance and composition of plant species as well as animals found in the soils.

The soils of this ecoregion have formed under generally dry and warm conditions, but given the diversity of geologic substrates, are highly variable locally. As defined by the U.S. Department of Agriculture's soils classification system, the predominant soil orders in this ecoregion are Aridisols, Entisols, Alfisols, and Mollisols (NRCS 2006). They tend to be shallow and well drained, but in some areas can be very deep, as in the grasslands, or can have a caliche or hardpan layer that impedes drainage, as in the playas.

Within the Madrean Archipelago ecoregion, the patterns of soil origin, development, and chemistry, are important factors in the diversity of ecosystems found here. For example, Whittaker and Niering (1968) found that in the Santa Catalina Mountains, areas of limestone at lower elevations tended to support a more xeric floristic composition, with strong Chihuahuan desert affinities. In contrast, areas of diorite-derived soils (acidic in chemistry) tended to have grasslands or other desert scrub communities with less Chihuahuan composition. Specifics on the characteristics of the soils that are typically associated with each of the ecological system CEs are included in the CE conceptual models.

#### G.2.3.3 Climate

Overall, the climate of this desert ecoregion is hot and dry; in the Arizona portion of the ecoregion, annual average temperature is 61.6° F (16.4° C). The region receives an average of approximately 14.7 inches (373 mm; water equivalent) of precipitation a year. There is some variability within these averages because the sky islands create microhabitats that are cooler and wetter than the ecoregion average. In general, air temperatures and precipitation have strong seasonality: a cool, wet winter season (Nov-March) driven by long-duration cyclonic storms with moisture from the Pacific Ocean; a dry season spanning the spring and early summer (April-June); and a hot, wet monsoon season from late summer into the fall (July-Oct), with heavy, short duration, convective thunderstorms that derive moisture from the Pacific and the Gulf of Mexico and the occasional tropical storm (Serrat-Capdevila et al. 2007). A majority (60%) of the annual rainfall occurs during the monsoon, which is approximately between July and October but centered on the months of July and August, while a second wet season in winter creates a bimodal distribution of precipitation (Figure G-3).

Because the climate is hot and dry, annual rates of potential evapotranspiration far exceed precipitation. Although the majority of the precipitation in the ecoregion falls during the summer monsoon as rain, this season also experiences the greatest variability in the magnitude and location of precipitation as well as extremely high temperatures, resulting in very high rates of evapotranspiration during the summer. Evapotranspiration is lower in the winter, when storms deliver rain and snow, producing more uniform precipitation across the ecoregion.

Compared to summer storms, winter storms produce precipitation of greater duration but lower intensity, with rainfall at lower elevations and snowfall at higher elevations. Slow release of water resulting from high-elevation spring snowmelt and low evaporation rates make winter precipitation the major source of groundwater recharge because there is less runoff and greater gain to streams.

The area receives the majority of its precipitation during summer because of its proximity to the core monsoon region in Mexico, and weather stations in the south document more monsoon precipitation

than weather stations further north (Figure G-4). The monsoon is strongest in northwestern Mexico; Arizona and New Mexico usually only receive the northernmost fringes of precipitation.

**Figure G-3. Average monthly temperature and precipitation in the southeastern Arizona Planning Area (see inset), 1930-2002.** Data are from selected Western Regional Climate Center cooperative weather observation stations (ADWR 2010a).

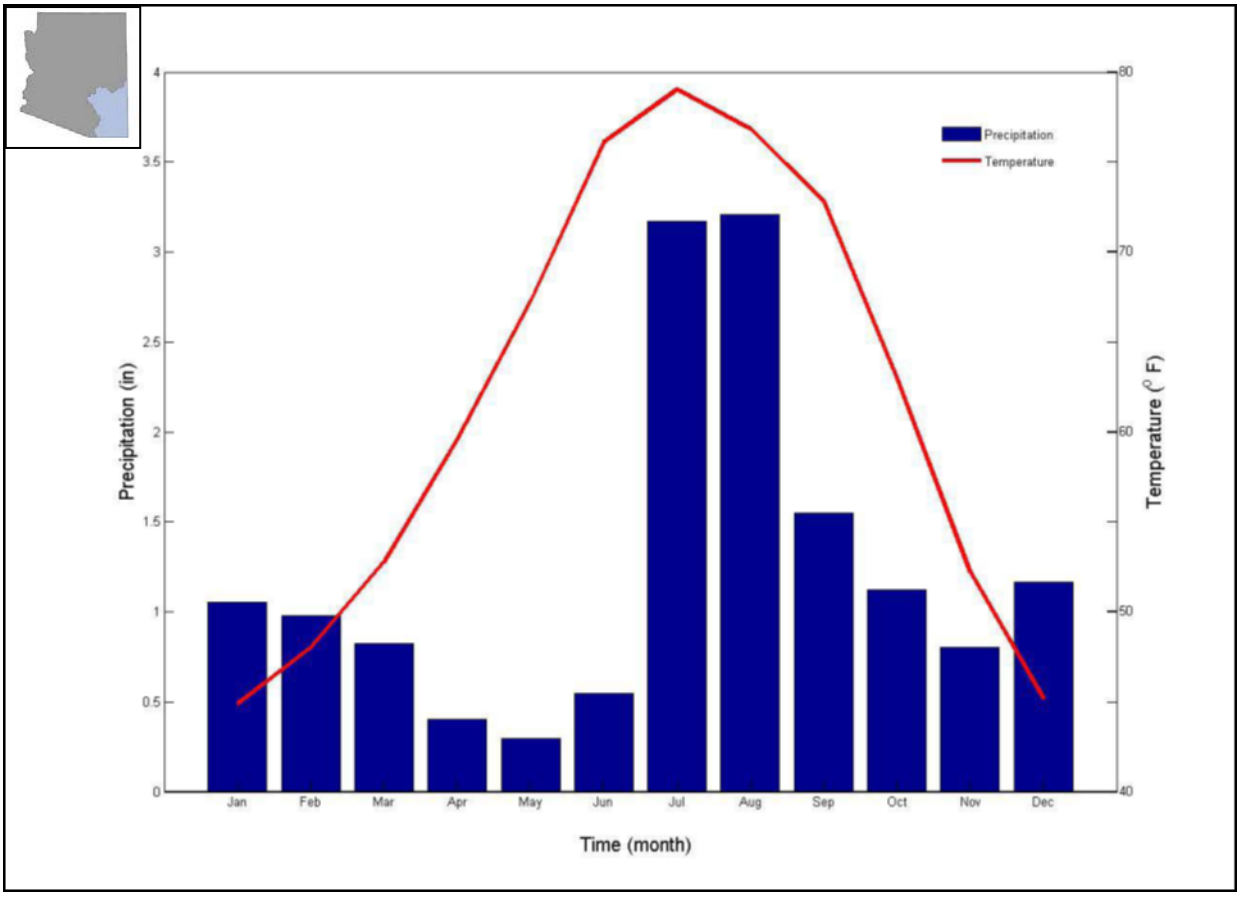
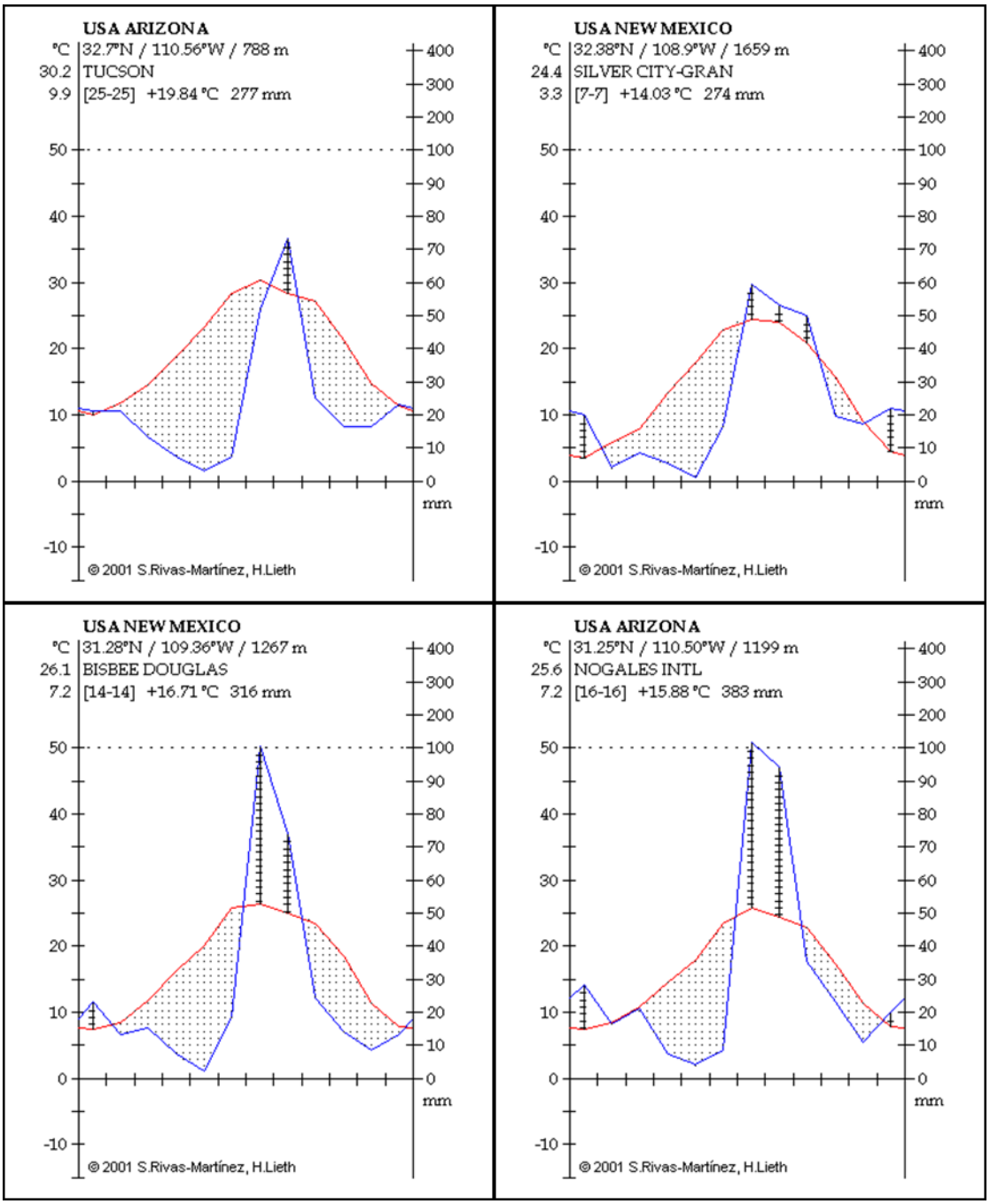


**Figure G-4. Climate diagrams showing average monthly temperature and precipitation from four weather stations in or near the MAR ecoregion.** Tucson (upper left), Silver City NM, (upper right), Bisbee/Douglas, AZ (lower left), and Nogales, MX (lower right). Left axis is temperature in Celsius, right axis is precipitation in mm, and horizontal axis ticks are month start and end; red line is temperature, blue line is precipitation time series, dots indicate water deficit, and verticle bar with cross hatching are areas with water surplus (diagrams from Rivas-Martinez et al. 2002). These diagrams illustrate the stronger summer monsoon pattern in the southern portion of the ecoregion (Bisbee/Douglas and Nogales, in the bottom two graphics). Silver City is just outside the eastern boundary of the MAR assessment area. See Figure G-2 for locations of municipalities where climate stations are located.



Winter precipitation (Nov-April) records (Figure G-6) dating to 1000 A.D., reconstructed from tree rings, show extended periods of above and below average precipitation in every century in the area encompassed in Climate Division 7 (southern AZ) (Figure G-5). Historical records for Climate Division 8 in New Mexico (Figure G-5) also show extended periods of above- and below-average precipitation in the last century and a quarter (Figure G-7). These decadal and shorter time period shifts are related to circulation changes in the Pacific Ocean. On time scales of 10-30 years, precipitation variability is likely related to shifts in Pacific Ocean circulation patterns, such as the El Niño-Southern Oscillation (ENSO) or the Pacific Decadal Oscillation (PDO).

On time scales of 2-7 years, the ENSO, with its phases of El Niño and La Niña, is associated with precipitation variations in the region, most notably during winter months (November-April). During El Niño episodes, there are greater chances for above-average winter precipitation; however, El Niño winters can also produce below-average precipitation. La Niña conditions are generally associated with drought in the region, and particularly with below-average winter precipitation. The ENSO phases also impact precipitation and monsoon strength in the region. For example, long-term records show that the 1950s were a relatively dry decade with an average winter precipitation deficit of -1.46 inches (37 mm), while the 1980s were a relatively wet decade with an average winter precipitation surplus of 1.86 inches (47 mm) (Figure G-7). Pool and Coes (1999) noted that trends in seasonal precipitation at four stations in the southern half of the Upper San Pedro Basin showed a general trend of increasing winter precipitation and decreasing wet-season (summer) precipitation during the period 1956-1997.

**Figure G-5. Maps showing locations of U.S. Climate Divisions that overlap with the MAR assessment area.** Climate Division 7 in Arizona (left) and 8 in New Mexico (right) overlap with the ecoregional assessment area. Maps from the following source: http://www.esrl.noaa.gov/psd/data/usclimdivs/data/map.html.

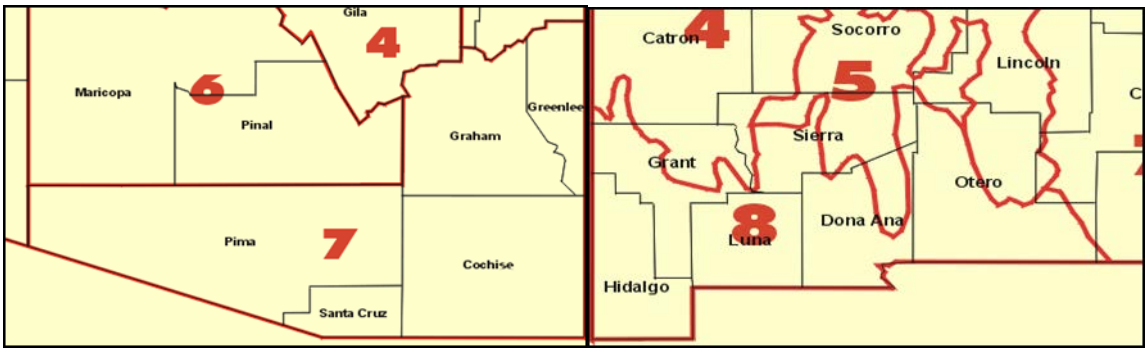


**Figure G-6. Winter (Nov-April) precipitation (20-year moving averages) for southern Arizona dating from 1000 A.D. to 1988, reconstructed from tree rings** show extended periods of above (blue)and below (red) average precipitation in every century in the area encompassed in Climate Division 7 (southern AZ). Data are presented as a 20-year moving average to show variability on decadal time scales. Values shown for each year are centered on a 20-year period. The average winter precipitation for 1000-1988 is 4.9 inches (124 mm). Data: Fenbiao Ni, University of Arizona Laboratory of Tree-Ring Research and CLIMAS. Figure author: CLIMAS (figure and caption from ADWR 2010a).

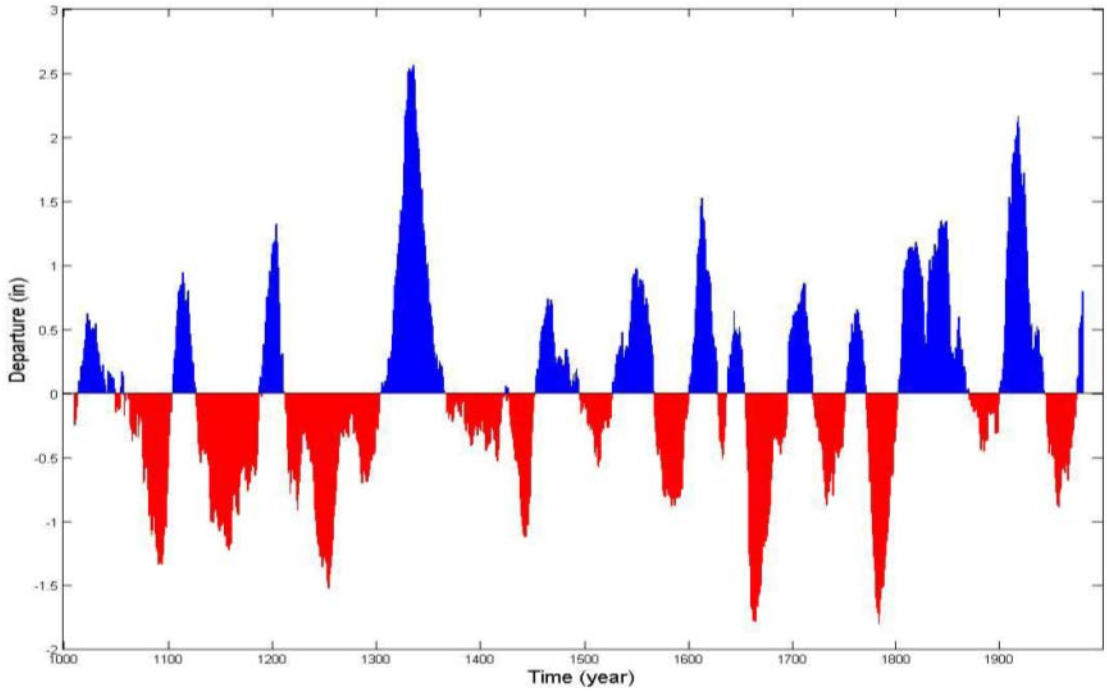
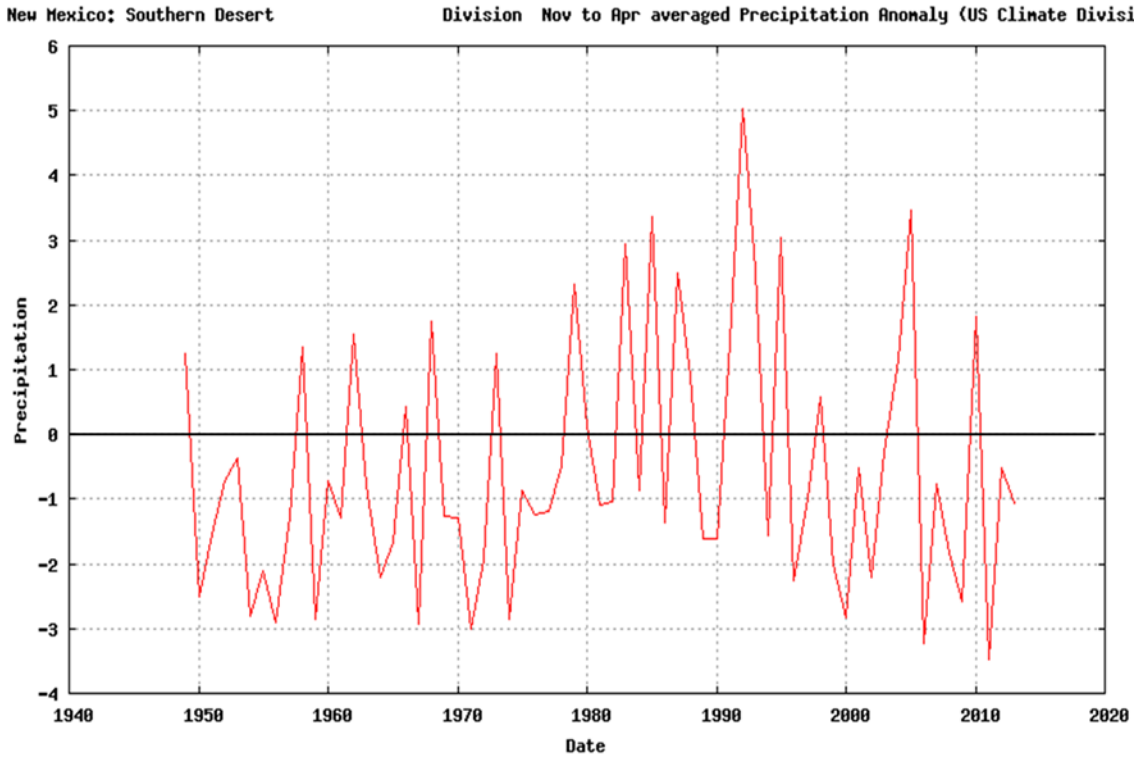


**Figure G-7. Winter (Nov-April) precipitation records relative to normal for 1949-2013, southwestern New Mexico.** Winter months (Nov-April) average precipitation relative to base line (1896-1945 Nov-April average) in inches, for the U.S. Climate Division #8, southwestern NM. Note that 1950s were a relatively dry decade with an average defict of -1.46 inches, while the 1980s were a relatively wet decade with an average winter precipitation surplus of 1.86 inches. Data from the Earth System Research Laboratory, **http://www.esrl.noaa.gov/psd/cgi-bin/data/timeseries/timeseries.pl**.




### G.2.3.4 Biogeography

The Madrean Archipelago straddles the international border and has ecological significance as both a barrier and bridge between two major cordilleras of North America: the Rocky Mountains and the Sierra Madre Occidental. It is the only sky island complex in the world that extends from subtropical to temperate latitudes (approximately 25 to 33 degrees north latitude), and as such, represents a continental-scale ecotone between the tropical and temperate regions of North America (Mau-Crimmins et al. 2005, Warshall 1995). It forms the southernmost edge of many temperate species' ranges and the northernmost edge of many tropical species' ranges. The hot and generally dry desert climate combined with the rich diversity of latitudinal and elevational extremes, topography, geologic substrates, and soils, has resulted in a remarkable suite of biodiversity (Brown 1982, Dinerstein et al. 2000, Mau-Crimmins et al. 2005). In addition, because the woodlands and forests cloaking the mountain ranges are isolated from each other by significantly different ecosystems, genetic interchange is limited and speciation is common; this has resulted in high numbers of endemic species (Bailowitz and Brock 1991, DeBano et al. 1995, Marshall et al. 2004, Mau-Crimmins et al. 2005).

## G.2.4 Ecosystem Processes

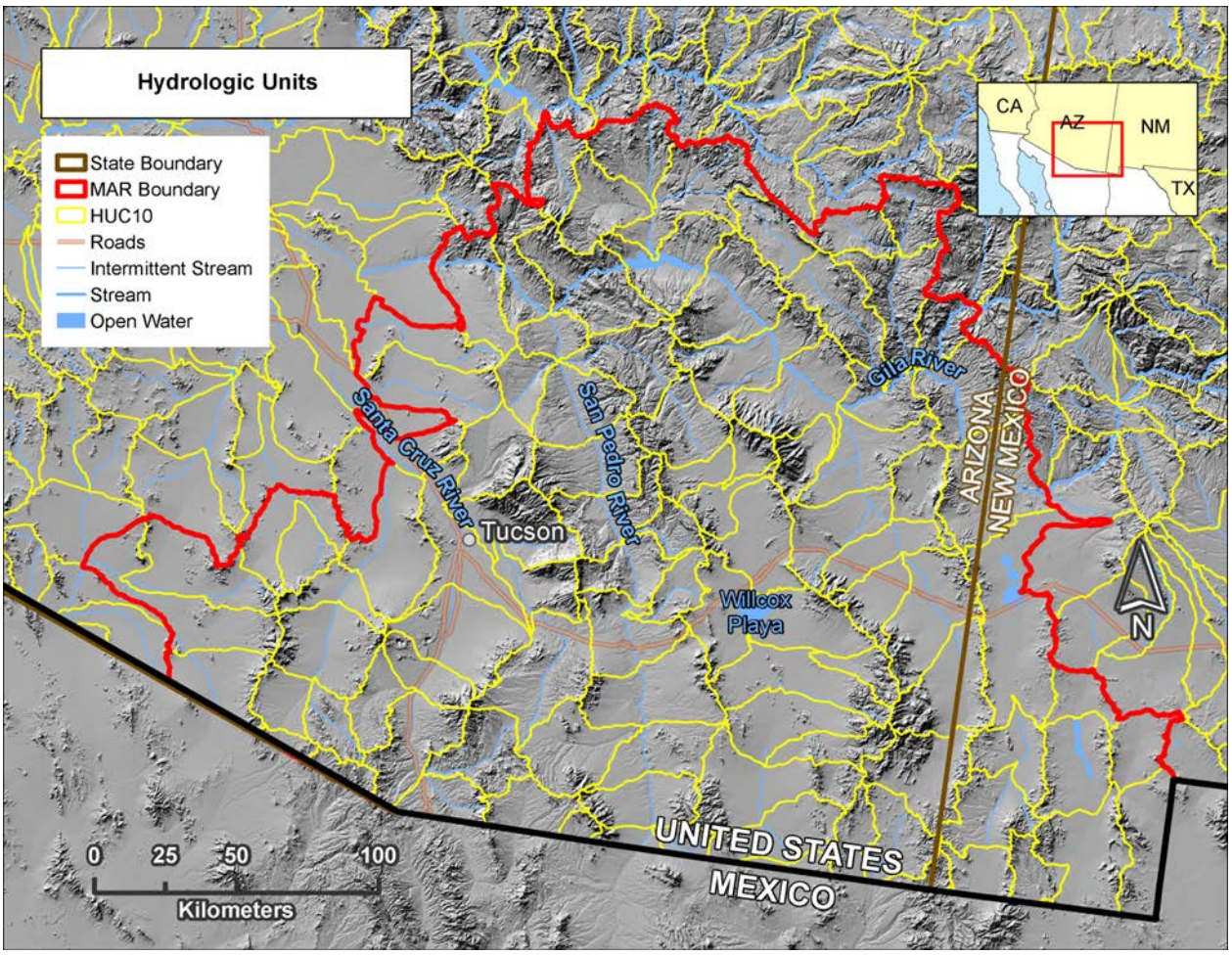
A range of physical and biological processes shape the biodiversity of this ecoregion. This section focuses on describing the natural processes influencing ecosystems and species. Discussion of the anthropogenic

effects on these natural processes and resulting impacts on natural sources is provided in the chapter, **Current Issues in the Madrean Archipelago** Ecoregion, in Harkness et al. (2013).

### G.2.4.1 Hydrology

Major watersheds in the Madrean Archipelago include the Middle Gila, Upper Gila, Sonora, and the Rio Grande-Mimbres (defined by four-digit hydrologic units). Major rivers in the ecoregion are the westward-flowing Gila and its southern tributaries, the San Pedro River and Santa Cruz Rivers (Figure G-8). Other tributaries include the San Francisco and San Simon Rivers, and Aravaipa Creek and Babocomari River, tributaries to the San Pedro River (NRCS 2006). The mainstem San Pedro River originates just across the border in Sonora, Mexico and is formed by the confluence of the Rio Nutrias with other tributaries. The mainstem Santa Cruz River originates in Arizona, but then flows south into Mexico before turning westward around the Sierra San Antonia and then northward to reenter the U.S. A small portion of the ecoregion in extreme southeastern Arizona drains southward into Mexico along Whitewater and Black Draws, tributaries to the Rio De Bavispe, which is in turn a major tributary to the Rio Yaqui. The ecoregion also includes several small terminal (closed) basins: the Willcox basin in AZ, and the Animas Playas and Lordsburg basins in NM.

**Figure G-8. Map of major rivers and 5th-level watersheds (shown in yellow outlines) in the Madrean Archipelago assessment area (shown in red outline).**



Hydrologic regimes are inextricably linked to climate patterns (precipitation and temperature) and the landforms and soils of a region. As briefly noted in the climate summary, although the amount of winter precipitation is lower, the storms are widespread, covering more area with more uniform and gentler rain; in conjunction with lower evapotranspiration rates, winter precipitation contributes more groundwater recharge compared to summer monsoon rains. In addition, more moisture soaks into soils, channels, and bedrock fractures in the mountains and along the mountain fronts (valley margins) during the winter storms, making the mountains and mountain fronts the largest geographic area of groundwater recharge for the ecoregion. Winter precipitation comprises less than half the total annual precipitation on average, but supplies the majority of the annual groundwater recharge (Pool and Coes 1999, Serrat-Capdevila et al. 2007).

**Hydrology and Aquatic Ecosystems**

The hydrologic patterns of a region reflect the relative contributions of surface runoff and groundwater discharge. In the Madrean Archipelago, the stream network exhibits both interrupted perennial and intermittent reaches. Groundwater generally flows from the margins to the central axis of each basin, supporting greater groundwater discharge on average along the central axes than along the valley margins (MacNish et al. 2009). The types of storms associated with different seasons and weather patterns produce different runoff flow magnitudes and durations. Naturally dry washes sustain surface flow during and immediately following precipitation events from both rainfall and snowmelt runoff. Base flow is sustained in stream and river reaches connected with the water table year-round or seasonally (Hirschboeck 2009). The hydrologic regimes of the region's aquatic ecosystems reflect the combined effects of shallow or deep groundwater discharge and watershed runoff (MacNish et al. 2009). While the basic mechanics of water flow remain largely unchanged, perennial stream reaches were longer and more numerous (Logan 2006, Thomas et al. 2006) and groundwater levels were significantly higher overall prior to significant human influences on the region's hydrology. Anthropogenic alterations to hydrology and associated impacts are discussed later in the Current Issues chapter in Harkness et al. (2013), in the section **Water Availability and Altered Hydrology**.

Springs and seeps are present throughout the ecoregion; they depend on groundwater flow and tend to have very stable patterns of discharge (Hendrickson and Minckley 1984). Their discharges may also have unique chemistries due to their origination in different aquifers with long flow paths. Confined groundwater (artesian conditions) can occur within the lower basin fill (ADWR 2010a). Artesian conditions occur in a number of locations, and are a result of localized clay and silt lenses within the basin fill of sand and gravel (MacNish et al. 2009). The ecoregion also contains several small closed basins. Runoff and recharge around the margins of these basins during periods of higher rainfall produce shallow ephemeral (playa) lakes (e.g., Willcox Playa), the chemistry of which can be controlled by near-surface evaporate deposits (Schreiber 1978).

Aquatic ecosystems in the ecoregion that rely entirely on surface water include ephemeral (losing reaches) of stream channels, dry washes, and playas. The biota of these ecosystems evolved to withstand highly dynamic seasonal and decadal changes in the amount and duration of surface flows, from multiple years of no water to extreme flood events. Aquatic ecosystems that rely on perennial sources of water occur along streams with groundwater-fed base flows, but also evolved under and are adapted to very dynamic changes in seasonal flow. For more information on aquatic ecosystems of the region, see the **Ecosystems** summary later in this appendix, as well as the individual conceptual models for the aquatic ecological system CEs (compiled in **Appendix E**) selected for this REA.

#### G.2.4.2 Fire

Natural disturbances are important drivers of change and are defined as any relatively discrete events in space and time that disrupt ecosystem, community, or population structure and change resources, substrate, or the physical environment (White and Pickett 1985). The key components of this definition are that disturbances are discrete in time, in contrast to chronic stress or background environmental variability, and that they cause a notable change, a perturbation, in the state of the system. Within the Madrean Archipelago ecoregion, fire is among the major disturbance agents shaping the ecosystems of the region, whether as a single, discrete event or multiple events comprising a native fire regime.

Fire has shaped, and continues to influence, the ecosystems of this ecoregion. Indeed, fire was likely the most important agent structuring the terrestrial systems within the ecoregion for millennia. With the exception of the desert scrub (creosotebush or the Sonoran Palo Verde-Mixed Cacti Scrub), all of the terrestrial ecosystems of this ecoregion burned frequently – from the grasslands and shrublands in the valley bottoms, to the fringing woodlands, to the coniferous forests high in the mountains.

Early historical records contain many accounts of fires burning millions of acres in the southwest in particular years (McPherson and Weltzin 2000). The vast majority of these fires were likely caused by lightning strikes; this region has among the highest incidence of lightning strikes, and the highest rates of lightning-ignited fires in the U.S. Fire-scar studies have shown that the woodlands and low-elevation forests had a fire-return interval of less than 10 years. The Madrean Archipelago's grasslands likely burned as frequently, if not more so (Bahre 1985, McPherson and Weltzin 2000). The higher, wetter, coniferous forests had comparatively longer, but still remarkably short fire-return intervals of approximately 35 years (Swetnam 1988).

These frequent fires historically burned with low intensity and typically stayed on the ground. Stand-replacing fires were likely extremely rare, as the frequent ground fires kept fuel loads to a minimum and prevented the accumulation of ladder fuels necessary for fire to reach the canopy. Most areas were likely fuel-limited, and fires would occur during periodic droughts following a sequence of wet, productive years. These climate-driven patterns would cause large areas to become susceptible to fire simultaneously, resulting in low-intensity fires spreading across very large areas.

There are major interactions between invasive grasses and fire regimes. Invasive species such as annual red brome (*Bromus rubens*) are often abundant following wet years and pose a fire hazard by creating a continuous fuel layer (Beatley 1966).The perennial invasive buffelgrass (*Pennisetum ciliare*) is expanding into desert scrub areas and drastically changing the fire regime (Brenner and Kanda 2013). Alterations of the fire regimes associated with different ecosystems in the ecoregion and the effects of these alterations are discussed in the Current Issues chapter in Harkness et al. (2013).

#### G.2.4.3 Other Ecosystem Processes

Other ecosystem processes that are not driving the broad patterns of vegetation and biological diversity across the landscape but creating localized patterns of diversity or providing certain functions are also operating in this ecoregion. Nutrient cycling, insect outbreaks, pollination, and herbivory are noteworthy processes, either because of their importance or how they are changing.

Nutrient cycling is especially important in semi-desert regions, where soils are often infertile due to a lack of organic matter, and also often highly alkaline or saline. Invertebrates are important for nutrient cycling, and subterranean species of ants and termites can impact soil properties such as bulk density, infiltration permeability and storage (Whitford et al. 1995). Mau-Crimmins et al. (2005) provide a summation of the importance of nutrient cycling in dryland ecosystems:

"Nutrient cycling involves the input of nutrients (from weathering of rocks, fixation of atmospheric nitrogen, and atmospheric deposition from rain, wind and gases), the loss of nutrients through various ecological processes (such as leaching, emissions, wind erosion, and fire), and the transfer of nutrients between the soils and vegetation within the ecosystem. In arid ecosystems, the spatial pattern of nutrients is highly variable as patches of nutrient-rich soils are often surrounded by a matrix of nutrient-poor soils. These "islands of fertility" are formed as existing vegetation creates a patch of nutrient-rich soil as litter is deposited in the immediate area surrounding the plant. This often allows for the recruitment of other individuals, which perpetuates the process (Aguiar and Sala 1999). The rate at which nutrients are absorbed and utilized is highly dependent upon the species and the nutrient supply. As a result, changes in biotic or abiotic conditions may lead to changes in the nutrient cycling regime of an ecosystem (Chapin et al. 2002)."

As herbivores and disease vectors, insects have significant effects on woodland and forest communities in this ecoregion. Many insects and associated disease pathogens are native to the ecoregion and have natural cycles of outbreaks. However, with changes to fire regimes, increased stress to native organisms due to human activities, changes in air or water quality, and climate change, there is increased potential for more massive and destructive insect or disease outbreaks. For example, the spruce-fir forest on top of the Pinaleño Mountains in the Coronado National Forest is experiencing a massive die-off of mature trees, primarily due to the combined effects of drought, high density of trees and competition, and insect outbreaks (Schussman and Smith 2006). The current insect outbreak involves a variety of species, including the non-native spruce aphid (Lynch 2009). Since 1998, over 90 percent (around 1,800 acres (730 ha)) of the spruce-fir vegetation type on the Coronado National Forest has suffered mature tree mortality due to insect attack and wildfire (Coronado National Forest 2009).

The nectar-feeding bats of the ecoregion play a role in pollination, especially for the agaves (*Agave* spp.) on which they feed. Lesser long-nosed bats (*Leptonycteris yerbabuenae*) pollinate various agave species, columnar cacti, and other Mexican plant species. Mexican long-tongued bat (*Choeronycteris mexicana*) feed on nectar, pollen, probably insects, and occasionally fruit of columnar cacti (Alvarez and Gonzalez-Q. 1970, Villa-R. 1967). Near Tucson, Arizona, long-tongued bats feed predominantly on cactus and *Agave* species (*Agave schottii* before mid-June, then *A. palmeri*) (Arizona Game and Fish Department 2006, Van de Water and Peachey 1997). Both of these bat species follow a "nectar corridor" of blossoming plants north from Mexico into the United States each spring (Fleming 2012), although the phenology for each of them is slightly different (Arizona Game and Fish Department 2011). Many species of butterflies, flies, bees, and moths are important for pollination. Some species such as yucca moths (*Tegeticula yuccasella*) and *Yucca* species have obligate mutualistic pollination relationships (Whitford et al. 1995).

Herbivory in MAR grasslands by native mammals was historically dominated by pronghorn (*Antilocarpa americana*), prairie dogs, and other small mammals. Black-tail prairie dogs (*Cynomys ludovicianus*) once had extensive colonies but were greatly reduced or extirpated from semi-desert grasslands in New Mexico and Arizona by the 1960s and their numbers and impacts are still small (Parmenter and Van Devender 1995). Prairie dogs have been shown in some studies to influence the presence of woody species such as mesquite (*Prosopis* spp.) (e.g., Weltzin et al. 1997; see also discussion in Milchunas 2006). Today, pronghorn and other small mammals are the dominant vertebrate herbivores. Herbivory from native small mammals, such as rodents, is significant as they are the most common mammals in the semi-desert grassland ecosystem. There is also high diversity of these rodents in the MAR, especially ground-dwelling species, and these burrowing rodents have a substantial effect on vegetation composition, soil structure and nutrient cycling (Finch 2004, Parmenter and Van Devender 1995). Grasshoppers feed on grasses and forbs and can consume significant amounts of forage when their

populations are high. However, the intensity and scope of herbivory by native mammals and insects is such that it is not a major driver shaping the mosaic of grasslands and shrublands of the ecoregion. Natural herbivory is noteworthy in relation to livestock grazing; through livestock grazing, large-scale herbivory has been added as a significant ecological process in a region whose biota had not evolved under comparable herbivory processes. Effects of livestock grazing are discussed in the Development and Other Land Uses section in the Current Issues chapter in Harkness et al. (2013).

### G.2.5 Ecoregion Conceptual Model Diagrams

The conceptual diagrams presented in this section were developed to visually illustrate the physical and process drivers, described above, that shape the biodiversity of the ecoregion. These pervasive influences of climatic regimes interacting with the basin and range physiography provide the overarching biophysical controls on MAR ecosystem patterns and processes. Seasonal temperature regimes vary along longitudinal, latitudinal, and elevational gradients, as do seasonal precipitation regimes interacting with rain-shadow effects. Combined, these regimes determine regional patterns in weather, such as monsoons, and movement of water. These in turn shape the distribution of ecological systems across the landscape, as well as the processes and stressors interacting with them. Conceptual diagrams were developed for two levels of organization: the ecoregion as a whole and the primary system "divisions" (broad categories or groupings of systems or habitats) of the ecoregion. The ecological system conservation elements (CEs) are nested within each of the system divisions.

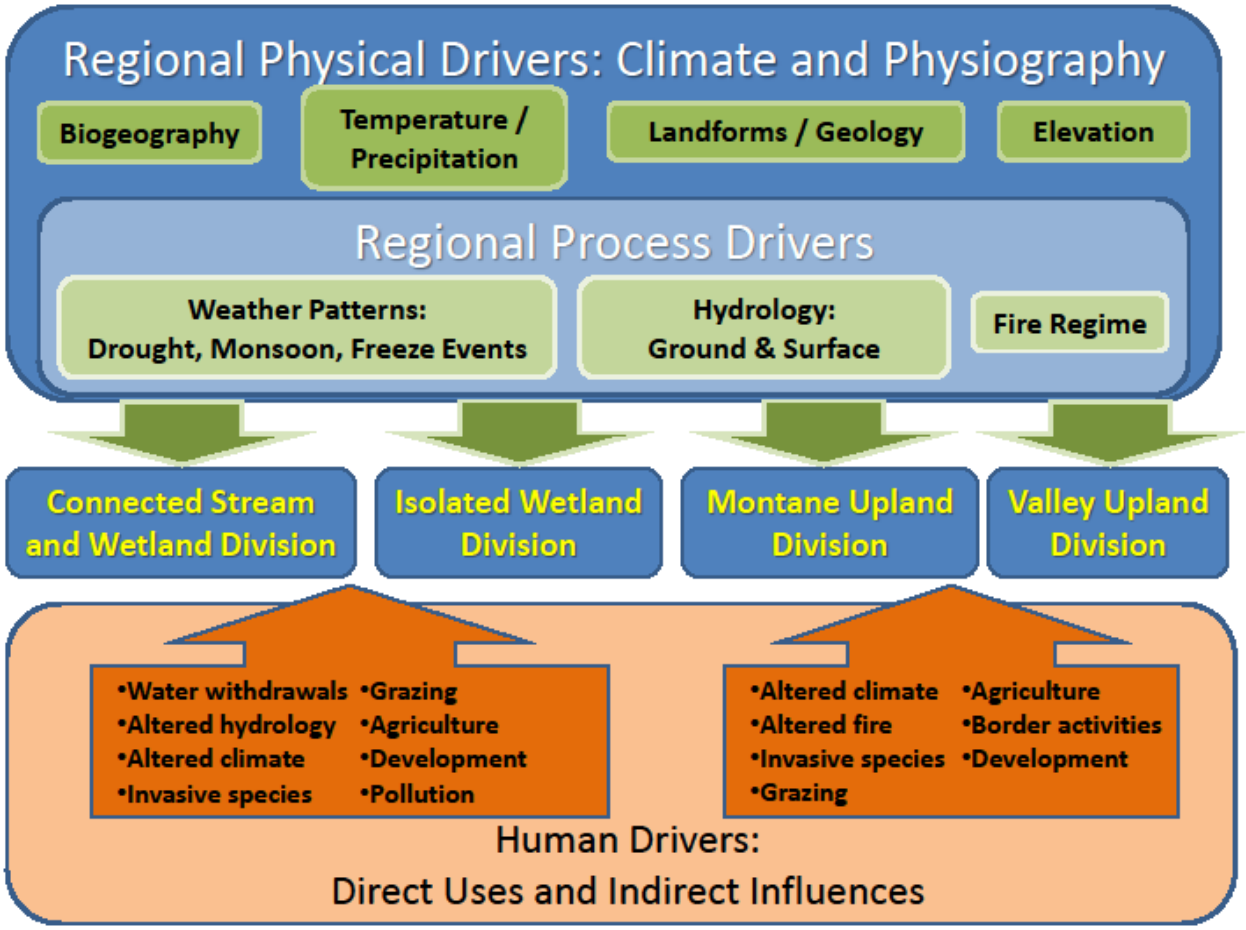
Water plays a critical role in this semi-desert ecoregion, and as an initial division, the overall ecoregion model (Figure G-9) distinguishes uplands, generally driven by water scarcity, from wetlands (aquatic, riparian, and wetland ecosystems) driven by water flow regimes. Within the uplands, elevation is the second defining variable, with montane ecosystems (generally dominated by conifers) distinguished from those found in the foothills and valleys (generally the grasslands, desert scrub, and oak woodlands). Terms such as "basins" or "lowland" often indicate wetlands of various kinds; therefore, the term "valley uplands" is used to denote the low-elevation uplands and distinguish them from the upland ecosystems found in the montane elevations. Within wetlands, isolated, closed-basin wetlands (playas) were separated from the connected stream network and associated wetlands.

Biogeography is one of the major sub-continental drivers of the biotic composition of many of the ecosystems of the MAR. The upland ecosystems include natural drivers of weather patterns, topography and geology, soil characteristics, and natural disturbance dynamics such as fire and insect outbreaks. These vary considerably between higher, cooler montane settings and warmer, low-elevation valley or basin settings. Within the wetland systems (i.e., streams, larger rivers, playa lakes, wetlands, and riparian environments), the driving factors are seasonal water flow regimes and the relative influence and connectivity of surface and groundwater dynamics. Montane rivers and streams are most strongly driven by surface water flow regimes, while those within the valley basins combine surface flow dynamics with groundwater flows and evaporation; these systems are hydrologically connected to varying degrees. The playas are primarily driven by surface water flow and found in closed basins. All of these natural abiotic drivers constrain and influence biotic responses in both plants and animals, such as predator/prey dynamics, herbivory, pollination, migration patterns, and insect and disease outbreaks.

In addition to abiotic drivers, the human dimension is also represented as a distinct model component. Socioeconomic and demographic factors act as drivers or controls of change in land and water use, policy, and activities (Figure G-9). While human uses and activities within and beyond the ecoregion produce many positive values (e.g., economic development, outdoor recreation, and solitude), these uses directly or indirectly alter natural system drivers. For example, fire, herbivory, and biotic soil crust processes are altered through uses or management such as livestock grazing and fire suppression in the

upland systems. Within wetland systems, the human dimension appears through water withdrawals or diversions, water pollution (e.g., human waste, toxic metals from mining, deposition of atmospheric pollutants), wetland conversion, livestock trampling, or introduction of invasive species. Land conversion and introduction of invasive plant species closely follow human land use patterns for settlements, border infrastructure, energy development and mineral extraction, irrigated agriculture, or transportation/communication infrastructure. Air quality is impacted by a variety of pollutants, and deposition of these has detrimental effects on upland and aquatic ecosystems and species. Predator/prey dynamics may be influenced by human/wildlife conflicts, disruption of migration or movement patterns, hunting, habitat alteration by livestock congregation, and resource collecting (e.g., plants).

**Figure G-9. Conceptual diagram for the Madrean Archipelago ecoregion**, showing the most important physical and process drivers for the region as a whole, as well as important human influences and direct uses. The major patterns of ecosystems are represented by four broad divisions (e.g., upland versus wetland), which are shaped or influenced by both the natural and human drivers. While biogeography is not a direct physical driver, it is a major influence on the diversity of the MAR both spatially and temporally.

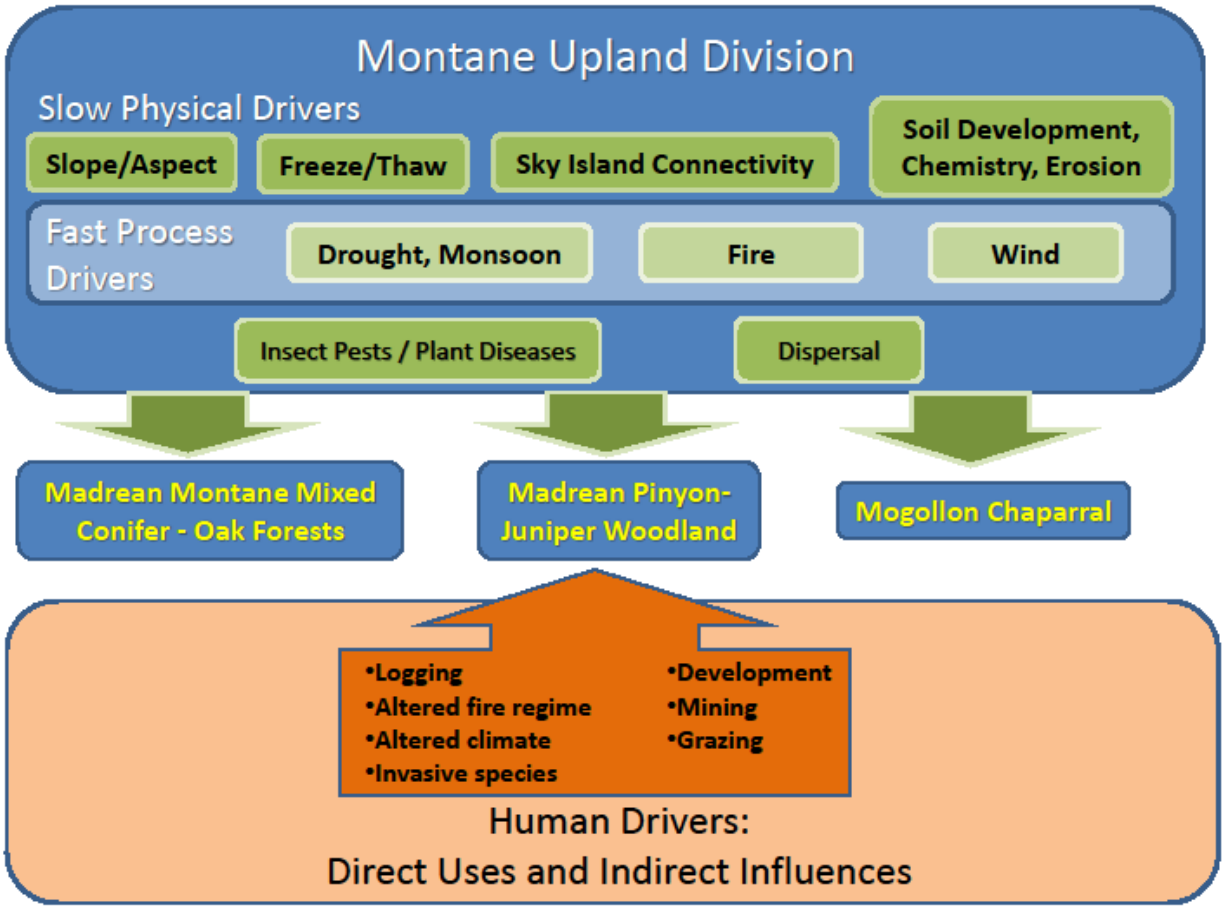


#### G.2.5.1 Division Models

For each of the four broad divisions or groupings of systems (montane uplands, valley uplands, connected wetlands, and isolated wetlands) of the ecoregional conceptual model, associated diagrams introduce additional detail. Natural drivers of patterns are organized in terms of "slow" physical drivers, such as landform and soil development, representing properties and processes that change on decadal and longer timeframes. Drivers of more rapid processes, or "fast" physical drivers, include those such as fire and flooding regimes, soil erosion, and other dynamics that occur over relatively short time frames. Biotic drivers, including the responses and interactions of plants and animals within stated physical bounds and regimes, are also differentiated here. For each of these, the most important drivers of patterns in the MAR landscape are presented; many other drivers of patterns and biotic composition occur on increasingly local scales.

The Montane Upland Division includes three selected conservation elements that encompass landscape pattern, dynamics, and biotic assemblages found in montane mixed conifer forests, pinyon-juniper woodlands, and montane chaparral shrublands (Figure G-10). These are proportionally more limited in extent than Valley Uplands and are primarily found on National Forest lands of the ecoregion.

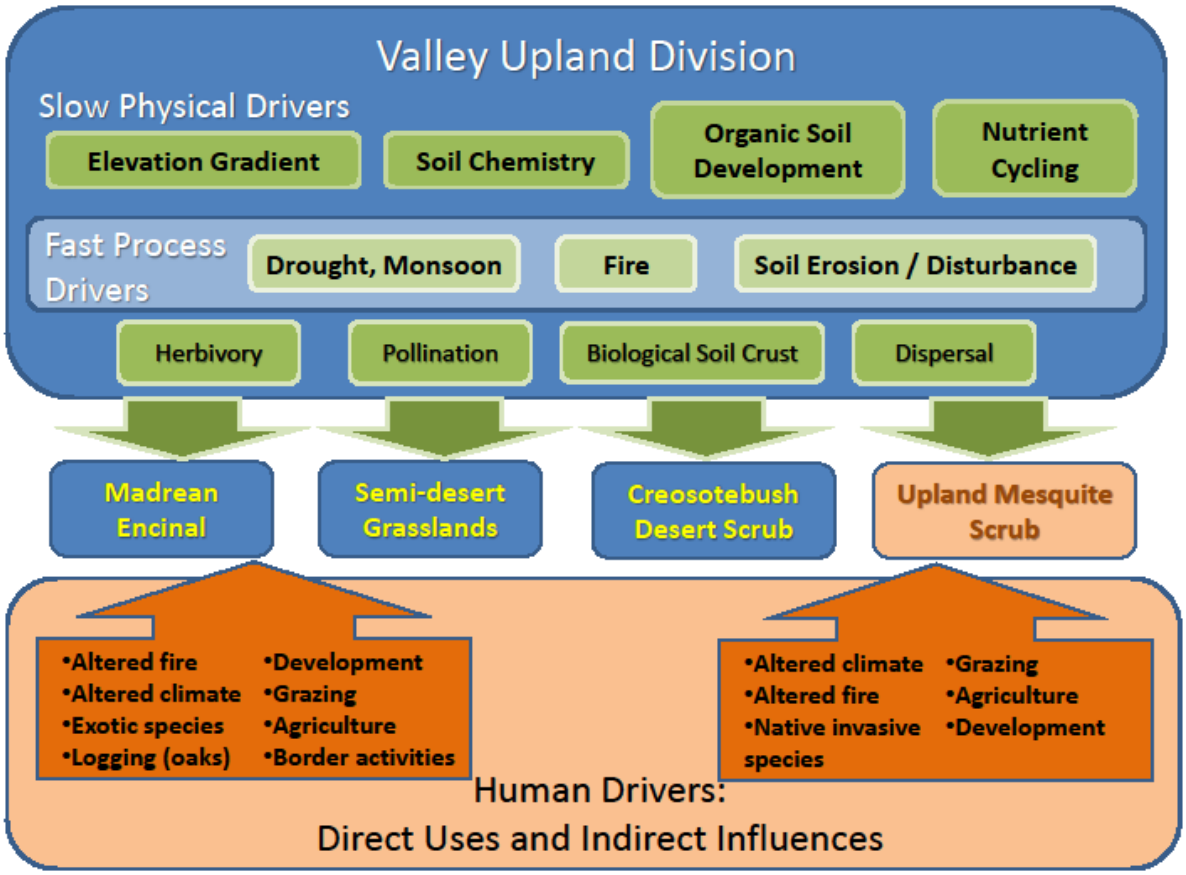
**Figure G-10. Conceptual diagram for the montane uplands division of the Madrean Archipelago**. The diagram highlights the most important physical and process drivers for the montane uplands as a whole, as well as important human influences and direct uses. The major patterns of ecosystems are shown as the three ecological system conservation elements, which are shaped or influenced by both the natural and human drivers.



Encompassing the vast majority of the ecoregion, the Valley Upland Division represents the landscape pattern, dynamics, and biotic assemblages found in encinal oak woodlands, semi-desert grasslands, creosotebush desert scrub, and the mesquite upland scrub (Figure G-11). The ecological systems of the Valley Upland are primarily found on BLM, state, and private lands.

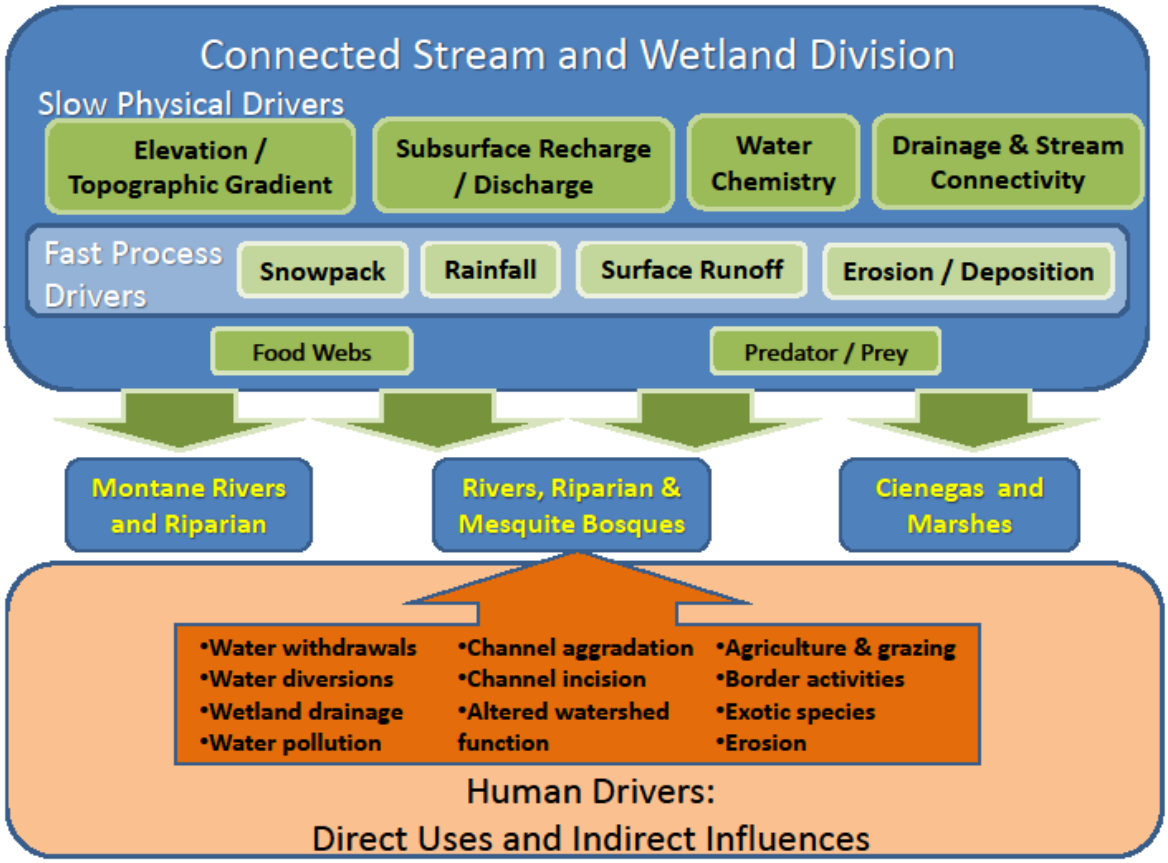
The mesquite upland scrub ecological system is shaped in part by the human activities and influences in the landscape, such as changes in climate and fire regime, historical grazing practices, and development. Although it is not treated as a typical conservation element in this REA, it is an important biotic component of the MAR landscape.

**Figure G-11. Conceptual diagram for the valley uplands division of the Madrean Archipelago**. The diagram highlights the most important physical and process drivers for the valley uplands as a whole, as well as important human influences and direct uses. The major patterns of ecosystems are shown as the three ecological system conservation elements, and one ecological system representing the native increaser, mesquite, all of which are shaped or influenced by both the natural and human drivers.



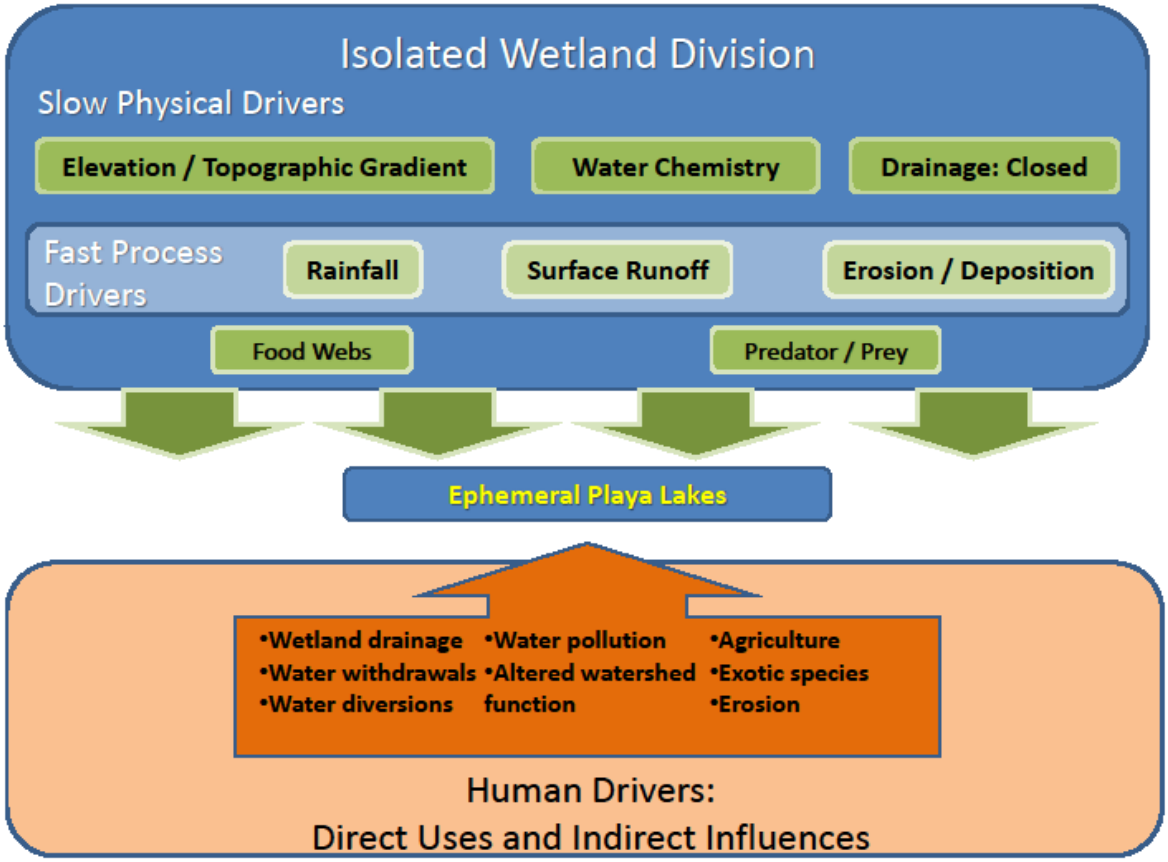
The Connected Stream and Wetland Division includes three ecological system conservation elements that represent landscape pattern, dynamics, and biotic assemblages found in the relatively limited montane and low-elevation rivers, streams, riparian communities, and ciénegas and marshes (Figure G-12). Inherently limited in spatial extent, these systems are found on a range of both public and private lands across the ecoregion.

**Figure G-12. Conceptual diagram for the connected stream and wetland division of the Madrean Archipelago**. The diagram shows the most important physical and process drivers for the connected aquatic, wetland and riparian ecosystems as a whole, as well as important human influences and direct uses. The major patterns of ecosystems are shown as the three ecological system conservation elements, which are shaped or influenced by both the natural and human drivers.



The Isolated Wetland Division includes one ecological system conservation element representing the landscape pattern, dynamics, and biotic assemblages found in the playa lakes (Figure G-13).

**Figure G-13. Conceptual diagram for the isolated wetland division of the Madrean Archipelago**. The diagram shows the most important physical and process drivers for the isolated wetland ecosystems as a whole, as well as important human influences and direct uses. The major pattern of ecosystems is shown as the single ecological system conservation element, which is shaped or influenced by both the natural and human drivers.




## G.2.6 Biodiversity

As described and illustrated in the text and conceptual diagrams above, broad-scale geologic features and landsforms, climate, biogeography, and ecological processes such as hydrology and fire provide the framework that determines the spatial patterns and composition of species and biological community diversity within the ecoregion. These factors are reflected in the biological diversity seen in the ecoregion.

### G.2.6.1 Ecosystems

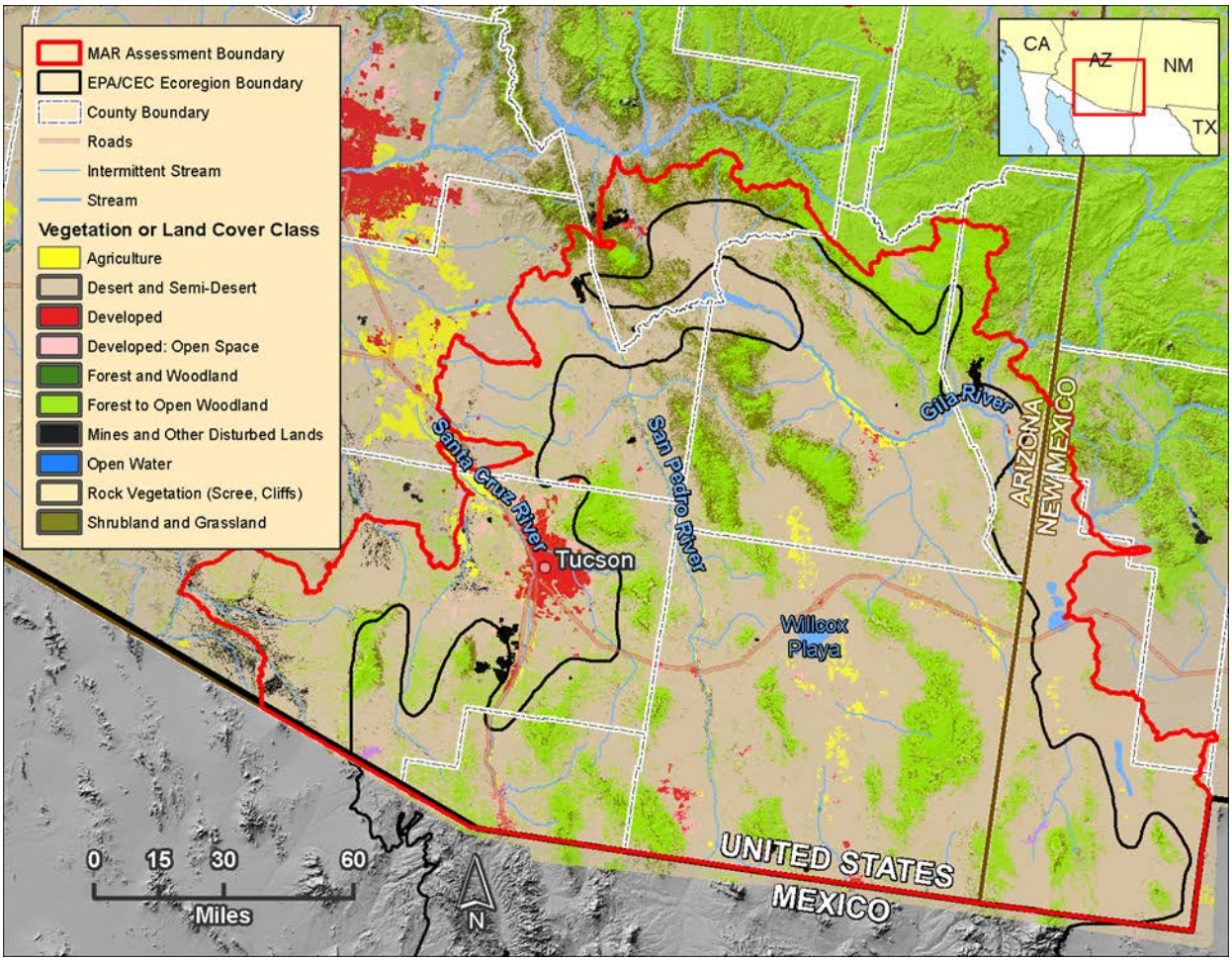
A great diversity of biological communities and ecosystems occur in the Madrean Archipelago ecoregion. Brown (1982) provides a valuable overview of the biotic communities found in southwestern North America, including listings of both plants and animals found in the major vegetation types of this region. The uplands vary from desert scrub or shrublands in the warmest and driest areas to the spruce-fir

forests found in small areas at the highest elevations of a few of the mountain ranges (Brown 1982, Mau-Crimmins et al. 2005). The wetlands and riparian areas vary from montane streams feeding into low-elevation rivers lined with deciduous trees and shrubs, to intermittently flooded playas, and emergent marshes and ciénegas associated with springs.

Below are brief, generalized descriptions of the major upland and wetland ecosystems, starting with the lowest elevation desert scrub and continuing upwards in elevation zones into the sky islands, followed by descriptions of the wetland ecosystems. They are organized by the conceptual model system divisions of Valley Uplands, Montane Uplands, and Streams and Wetlands. Much of the material has been drawn from Brown's (1982) overview of the biotic communities of the southwest.

Figure G-14 is a map illustrating the vegetation classes present in this ecoregion, as mapped by NatureServe (2013). These classes are part of the hierarchical U.S. National Vegetation Classification (Anderson et al. 1998, Grossman et al. 1998); individual ecological systems considered for CEs in this REA are linked to these coarser classes. Because the ecological systems represent a finer-scale unit in the classification hierarchy, the coarser vegetation classes provide a simple, broad overview of the spatial pattern of vegetation in the U.S. portion of the ecoregion.

**Figure G-14. Map of vegetation classes and other land cover types present in the Madrean Archipelago ecoregion.** Ecological systems can be grouped into broader classes in the U.S. National Vegetation Classification (Anderson et al. 1998, Grossman et al. 1998); those classes are shown here, in addition to other land uses, from NatureServe's (2013) data set. In this particular ecoregion, the non-specific disturbed lands class is primarily open-pit mines and is labeled accordingly.



### G.2.6.2 VALLEY UPLANDS

Desert Scrub and Thornscrub

Aridity is the primary determinant of desert vegetation; in the Madrean Archipelago, desert scrub is the driest biotic community and found in the lowest elevations. Net primary productivity is relatively low, although lifeform diversity is high. The plants tend to be widely spaced and typically exhibit leaf adaptations to the hot, dry climate. In the Madrean Archipelago ecoregion, there are two major types of desert scrub: the Sonoran Palo Verde-Mixed Cacti scrub found on the periphery in the western portions, and the Chihuahuan Creosotebush scrub found throughout much of the ecoregion. Important species include acacias (*Acacia* spp.), palo verdes (*Cercidium* spp.), giant saguaro (*Carnegiea gigantea*), and creosote bush (*Larrea tridentata*). Succulents are very common in desert scrub, and to a lesser degree, thornscrub, with agave (*Agave* spp.), yucca (*Yucca* spp.), barrel cactus (*Ferocactus* and *Echinocactus* spp.), hedgehog cactus (*Mammillaria* spp.), and prickly pear and cholla (*Opuntia* spp.) among the most common. Warm- and cool-season annuals, both native (e.g., *Plantago patagonica*) and introduced (e.g., *Bromus rubens*), are common following rainfall. These desert scrub communities are not fire-adapted; the large spacing between plants limits the spread of fire across the landscape. Widespread fire was, therefore, historically a relatively rare occurrence in these communities.

Semi-desert Grasslands

In contrast to the Great Plains to the east, grasslands in the Madrean Archipelago ecoregion are generally semi-desert in nature, although species common in the Great Plains do occur here. These grasslands are typically composed of perennial short- and mid-grass species; annuals and geophytes are also common, with occasional shrubs or trees. Most grasses in semi-desert grasslands use the C4 photosynthetic pathway that provides greater water use efficiency than the C3 photosynthetic pathway of most other plants. Important species in the semi-desert grasslands of this ecoregion include black grama (*Bouteloua eriopoda)*, tobosa (*Pleuraphis mutica*), sideoats grama (*Bouteloua curtipendula)*, blue grama (*Bouteloua gracilis)*, plains lovegrass (*Eragrostis intermedia)*, sand dropseed (*Sporobolus cryptandrus*), vine mesquite (*Panicum obtusum*), curly mesquite *(Hilaria belangeri)*, ephedra (*Ephedra* spp.), sotol *(Dasylirion* spp.), yucca (*Yucca* spp.), ocotillo (*Fouqieria splendens*), many different cacti, and agave (*Agave* spp.). Fire is a relatively common and necessary occurrence in semi-desert grassland, historically burning every five to ten years (Gori et al. 2012, Gori and Enquist 2003). Fire maintains the open structure of the ecosystem, conferring a competitive advantage to graminoids over most woody plants. Fire suppression, intensive grazing, and soil erosion have degraded much of the grassland ecosystem in this region, leading to encroachment by woody species (mesquite, *Prosopis* spp.) and drought-resistant, non-native grasses such as Lehmann's lovegrass (*Eragrostis lehmanniana*) and Boer lovegrass (*Eragrostis curvula),* which were originally introduced to the ecoregion for forage and erosion control. Grassland ecosystems in the Madrean Archipelago have been recognized for their regional biological value, especially for grassland birds (Latta et al. 1999). In the MAR ecoregion there are a diversity of birds associated with grassland habitats that depend on this habitat for different aspects of their life histories, including overwintering, migration stopover, and breeding.

### G.2.6.3 MONTANE UPLANDS

Chaparral

Chaparral is a semi-arid, shrub-dominated biotic community that occurs on the west coast of every continent between 30$^{o}$ and 40$^{o}$N latitude. Chaparral in the interior southwestern U.S. is found along the Mogollon Rim and south into the lower montane zones of many of the sky islands, and Sierra Madre Occidentale of Mexico (Brown 1982, DeBano 1999). Chaparral shrub species have thick, sclerophyllous

leaves containing large quantities of volatile compounds, and have a natural fire regime of intense, fast-moving fires that are often stand replacing. These fire-adapted shrub species sprout vigorously following fire, either from root crowns or from seed banks. The chaparral of the Madrean Archipelago ecoregion is less diverse than the coastal chaparral of California and is composed of dense stands of manzanita (*Arctostaphylos* spp.) and shrub live oak (*Quercus turbinella*), and Toumey oak *(Quercus toumeyi),* with mountain mahogany *(Cercocarpus montanus),* desert ceanothus *(Ceanothus greggii),* buckthorn (*Frangula betulifolia)*, *Purshia* spp., and silktassel (*Garrya* spp.) also present (Carmichael et al. 1978, DeBano 1999).

Madrean Evergreen Woodland

The transitional elevations between semi-desert grassland and the higher-elevation, conifer-cloaked mountains are dominated by Madrean evergreen woodlands. Madrean evergreen woodland is ubiquitous at middle elevations throughout the Madrean Archipelago ecoregion. There are three ecosystems comprising these woodlands: the "encinal" or oak-dominated woodlands; pinyon-juniper woodlands; and lower to mid-montane woodlands of pines, firs, and oaks. The Madrean encinal is characterized by evergreen oaks with thick, sclerophyllous leaves, such as emory oak (*Quercus emoryii*), Arizona white oak (*Quercus arizonica*), gray oak (*Quercus grisea),* silverleaf oak *(Quercus hypoleucoides),* netleaf oak *(Quercus rugosa)*, and Mexican blue oak (*Quercus oblongifolia)*. The Madrean pinyon-juniper woodlands are commonly mixes of pinyon pine species such as Mexican pinyon *(Pinus cembroides),* border pinyon (*Pinus discolor),* or two-needle pinyon (*Pinus edulis)*. The juniper species include red-berry juniper *(Juniperus coahuilensis)* and alligator juniper *(Juniperus deppeana)*. At higher elevations, the oak species mix with Arizona pine (*Pinus arizonica),* Apache pine *(Pinus engelmannii),* Chihuahuan pine *(Pinus leiophylla),* Ponderosa pine *(Pinus ponderosa)*, or southwestern white pine (*Pinus strobiformis).* In places, Douglas-fir (*Pseudotsuga menziesii)* is mixed with the pines and oaks, while Coahuilan fir (*Abies coahuilensis)*, corkbark fir (*Abies lasiocarpa* var. *arizonica*), or white fir (*Abies concolor)* occur in cooler or more mesic settings. Understory grasses and forbs are usually abundant. In general, the Madrean evergreen woodlands are fire-adapted ecosystems; although fire regimes may vary between woodland types, generally they experience frequent, low-intensity fires (Kaib et al. 1996, Schussman 2006).

Coniferous Forests

Dominated by conifers such as pines (*Pinus spp.*), spruces (*Picea spp*.), and firs (*Abies spp.*), coniferous forest occurs in the coolest settings of the Madrean Archipelago. These forests are confined to cooler sites (a function of elevation, aspect, and local geomorphology), generally upslope from the mixes of conifers with evergreen oaks. Most of the conifer forests in the sky islands are dominated by Douglas-fir (*Pseudotsuga menziesii*) and true firs (*Abies* spp.), with spruce (*Picea* spp.) at the highest elevations. These fire-adapted conifer forests have a range of fire regimes, from low-intensity fires occurring every nine to fifteen years in ponderosa pine and mixed-conifer forests (Dimmitt 2000), to very infrequent, stand-replacing fires every 150 to 300 years in the spruce-fir forests at the highest elevations (Margolis et al. 2011).

Temperate Deciduous Forests

Although relatively minor in the Madrean Archipelago ecoregion, temperate deciduous forests are found here and are characterized by cold-tolerant, deciduous, woody plants such as Gambel oak (*Quercus gambellii*), trembling aspen (*Populus tremuloides*), and maples and box elder (*Acer* spp.). These typically occur in micro-climates at high elevations in the sky islands, often on north-facing slopes, and are interspersed with coniferous forest. Cold-tolerant deciduous species are often found in the understory of coniferous forests as well.

### G.2.6.4 STREAMS AND WETLANDS

Freshwater ecosystems are critical components of the Madrean Archipelago's biodiversity. Although occupying only a small proportion of the landscape (<1% of the ecoregion), they support a disproportionately high number of species. Rivers, streams, marshes and ciénegas, and ephemerally wet playas each support a rich diversity of plant and animals species. Riparian corridors along the streams provide migratory birds and pollinating insects and bats with critical travel corridors and resources. Large-scale migrations of waterfowl, shorebirds, raptors, and cranes depend upon shallow playa waters as migratory stopovers. Willcox Playa, Lordsburg Playa, and the upper San Pedro River are critical links for birds on both northern and southern migration routes. Perhaps the critical stopover hotspot of the ecoregion is the riparian woodland of the upper San Pedro, which serves as a corridor for up to four million neotropical migrants and is also important for nesting and wintering habitat. Locally, riparian woodlands help regulate other processes, such as river temperature, flooding intensity, soil retention, and evaporation rates.

Riparian Areas

Riparian corridors and their associated perennially or seasonally flowing streams occur along canyons and across desert valleys of the southwestern United States and adjacent Mexico. Mesquite-dominated (*Prosopis* spp.) sites can also occur along intermittent streams, where higher groundwater levels permit. The vegetation is a mix of riparian woodlands and shrublands, with reaches of herbaceous communities intermixed. Dominant native trees include Fremont's cottonwood (*Populus fremontii*), velvet mesquite (*Prosopis velutina*), Gooding's willow (*Salix gooddingii*), and sycamores (*Platanus racemosa*). Native shrubs include arroyo willow (*Salix lasiolepis*), Geyer's willow (*Salix geyeriana*), seepwillow (*Baccharis* spp.), silver buffaloberry (*Shepherdia argentea*), and coyote willow (*Salix exigua*). Stream reaches alternate between perennial and intermittent flow depending on local alluvial and groundwater patterns as well as local water use. Varying patterns of flooding and drought change the composition and structure of the riparian vegetation and the distribution of aquatic habitats (deep and shallow pools, shading from banks, water temperature and chemistry, extent of the wetted gravel zone and other characteristics); these patterns are driven by the natural timing and amount of flushing flows following winter and summer storms. Over the past century (1900s), the invasion of non-native species such as tamarisk and the increase in native woody species have changed the ratio of woody to herbaceous communities along riparian corridors (Stromberg et al. 2009). Higher-elevation riparian reaches flowing out of the sky islands are typically characterized by bedrock channels, shallower alluvial soils, and lush riparian vegetation, in stark contrast to the adjacent, upslope desert scrub. Cottonwoods (*Populus* spp.) and willows (*Salix* spp.) are common, as well as oaks (*Quercus* spp.), ash (*Fraxinus* spp.) and wingleaf soapberry (*Sapindus saponaria*). Other shrubs include chokecherry (*Prunus virginiana*), alder (*Alnus* spp.) and arroyo willow (*Salix lasiolepis)*. Upper-elevation riparian corridors are an important link between the orographically wetter mountains and the lower, larger riparian reaches, and serve as critically important movement corridors for mammals.

Playas

Playas are closed, shallow drainage pockets or basins that experience intermittent flooding from surface runoff and, in some instances, from shallow groundwater discharge. They are typically barren of vegetation for much of the year or even multiple years. Soils are fine and salt crusts are common, with small saltgrass (*Distichlis spicata*) and alkali sacaton grass (*Sporobolus airoides)* beds in depressions and sparse shrubs around the margins. Large playas found in the Madrean Archipelago ecoregion include Willcox Playa and Lordsburg Playa. Wetting occurs primarily through runoff combined with on-site precipitation, where clay soils or hardpans prevent most downward percolation, although high groundwater levels may have historically contributed to wetting the playas. Madrean Archipelago playas

are ecologically distinct from other types of MAR wetlands in three ways: First, they support a diverse and seasonally changing assemblage of birds, with winter numbers > 5,000 at Willcox Playa alone. During the winter they provide roosting and feeding habitat for large numbers of sandhill cranes and smaller numbers of water birds such as killdeer, snipe, and white-faced ibis, particularly in wet winters. Second, the playas in this ecoregion support a rich and unique assemblage of macroinvertebrates, including some 400 beetle genera from Willcox Playa, and tiger beetles specially adapted to the alkaline chemistry. Numerous crustaceans – particularly branchiopods – are also found here and emerge during wet episodes, providing key food resources for water birds. Third, the playas support several rare plant species.

Ciénegas, Springs, and Marshes

Ciénegas are spring-fed marshes typically composed of lush herbaceous vegetation surrounding pools of water. Ciénegas are sustained by permanent, rarely fluctuating sources of water (springs), and are near enough to headwaters that the probability of scouring from floods is minimal. In shallow pool margins, emergent plants include species of spikerush (*Eleocharis* spp.), sedge (*Carex* spp.) and rushes (*Juncus* spp.). Taller marsh vegetation can be found in adjacent deeper waters, such as cattails (*Typha* spp.), bulrush (*Schoenoplectus* spp., *Scirpus* spp.) and common reed (*Phragmites australis*); all of these species are native to the southwest. Relatively deep water may have floating and submerged aquatic plants. Ciénegas may be ringed by saline soils due to capillary action and evaporation, where salt-grass and alkali sacaton (*Sporobolus airoides*) may be abundant. Ciénegas tend to have deep organic soils and are very productive ecosystems. The sky island geologic faults provide for groundwater recharge along fault lines and are the source for springs and seeps, two important settings for ciénega creation and maintenance. Ciénegas were once much more abundant in the ecoregion, and are now reduced to a small fraction of their former distribution, due to greatly lowered water tables resulting from a complex interaction of drought and increased human and livestock use in the past century (Hendrickson and Minckley 1984). They generally occur as isolated wetlands along the sides of valley floors, separated from the main stream channel, although areas of springs and marsh vegetation do occur within the active channel of the lower San Pedro and other rivers.

#### G.2.6.5 Species Diversity

The Madrean Archipelago has exceptional species richness owing to its complex physiography and its location at the nexus of the Californian, Sonoran, Intermountain, Cordilleran, and Sierra Madrean provinces (Warshall 1995). Many plant and animal species are at the edges of their ranges in this region, particularly tropical species such as trees, orchids, moths, birds and bats (Felger and Wilson 1995), with a high number of endemic as well as threatened and endangered species (Warshall 1995). At least 468 species of birds have been documented in southeastern Arizona in the past 50 years, with approximately 207 species known or thought to breed here, along with 240 butterfly species, and hundreds of species of wood-rotting fungi (Bailowitz and Brock 1991, Corman and Wise-Gervais 2005, Edison et al. 1995, Gilbertson and Bigelow 1998, Marshall et al. 2004). More than 4,000 species of vascular plants and 110 species of mammals, including 23 bat species, have been documented (Felger et al. 1997, Schmidt and Dalton 1995, Simpson 1964). Southeastern Arizona has the greatest mammalian diversity north of Mexico (Turner et al. 1995). This is twice the mammal diversity of Yellowstone National Park and includes narrowly endemic species such as the white-sided jackrabbit (*Lepus callotis*), the Arizona cotton rat (*Sigmodon arizonae*), the Mearn's pocket gopher (*Thomomys bottae mearnsi*), and species at the edges of their ranges such as jaguar (*Panthera onca*) and ocelot (*Leopardus pardalis*). Large mammals with extensive geographic ranges that inhabit this ecoregion include black bear (*Ursus americanus*), mountain lion (*Puma concolor*), bighorn sheep (*Ovis canadensis*), and pronghorn (*Antilocapra*

*americana*). The New Mexico Department of Game and Fish identifies 102 Species of Greatest Conservation need in the New Mexico portion of the MAR ecoregion (NMDGF 2006).

To further illustrate the relative diversity of the ecoregion, at the heart of the MAR in the Peloncillo Mountains of southeastern Arizona and southwestern New Mexico, there are more species of amphibians and reptiles than in any other single mountain range in New Mexico, representing 72% of the 123 species known to occur throughout New Mexico, and Antelope Pass boasts the highest lizard diversity of any comparably-sized area in the U.S. Despite the aridity of this region, this count includes 14 native amphibians (Bodner et al. 2006). In the Arizona portion of the MAR, there are 11 species of amphibians of the 25 known to occur throughout Arizona and 76 species of reptiles, or 71% of the 107 known to occur throughout Arizona (Brennan and Holycross 2006). Insect diversity is largely unexplored but almost 400 species of bees are known from the San Bernardino NWR, with 1,000 species estimated to reside in the Peloncillo Mountain region alone (Bodner et al. 2006). A total of 246 species of skippers and true butterflies have been recorded in southeastern Arizona (Bailowitz and Brock 1991). The MAR is also home to a myriad of narrowly endemic talus and spring snails, and a high number of neotropical butterflies.

## *G.3 Human Context*

Anthropogenic influences are a critical component of the ecoregion conceptual model and are included in the conceptual diagrams in that section. Anthropogenic activities and uses of the ecoregion also shape the issues facing natural resource managers in the ecoregion. This section of the ecoregion conceptual models therefore fills a dual purpose: providing a brief narrative description of the human context that illustrates the anthropogenic role in the ecoregion conceptual model and introducing the subsequent section on natural resources-related issues facing the ecoregion.

### G.3.1 Demographic Overview

The human population in the ecoregional assessment area as a whole (shown in Figure G-2 by yellow outline and international boundary) is estimated to be approaching approximately 1.2 million (compiled from ADWR 2010a, Community by Design 2011, and U.S. Census Bureau). In the U.S. portion of the ecoregion itself (shown in Figure G-2 with solid green outline), the population is estimated to be under 200,000: the population of the surrounding Southeastern Arizona Planning Area was approximately 188,300 in 2000 (as compiled in ADWR 2010a), while Hidalgo County's was approximately 4,900 in 2010 (Community by Design 2011). It is generally concentrated in smaller municipalities, a number of which are located along the Interstate 10 or Interstate 19 corridors. Sierra Vista, Nogales (AZ) and Douglas are the largest municipalities within the U.S. portion of the ecoregion, with circa 2010 populations of approximately 44,000, 21,000, and 17,000 respectively. Other cities and towns in the ecoregion include Safford, Willcox, Benson, and Bisbee in the United States, and Nogales, Agua Prieta, Cananea, Magdalena de Kino, and Nacozari in Mexico. Population densities outside these areas are low, often below five people per square mile (Gorenflo 2003 as summarized by Marshall et al. 2004); for example, Hidalgo County has a circa 2010 density of 1.4 people per square mile (Community by Design 2011).

Lying in the far western portion of the larger ecoregional assessment area (shown in Figure G-2 by the yellow outline and international boundary), Tucson, AZ is the largest city in the assessment area, with a population in 2010 nearing one million in the greater metropolitan area. Tucson is the county seat of Pima County.

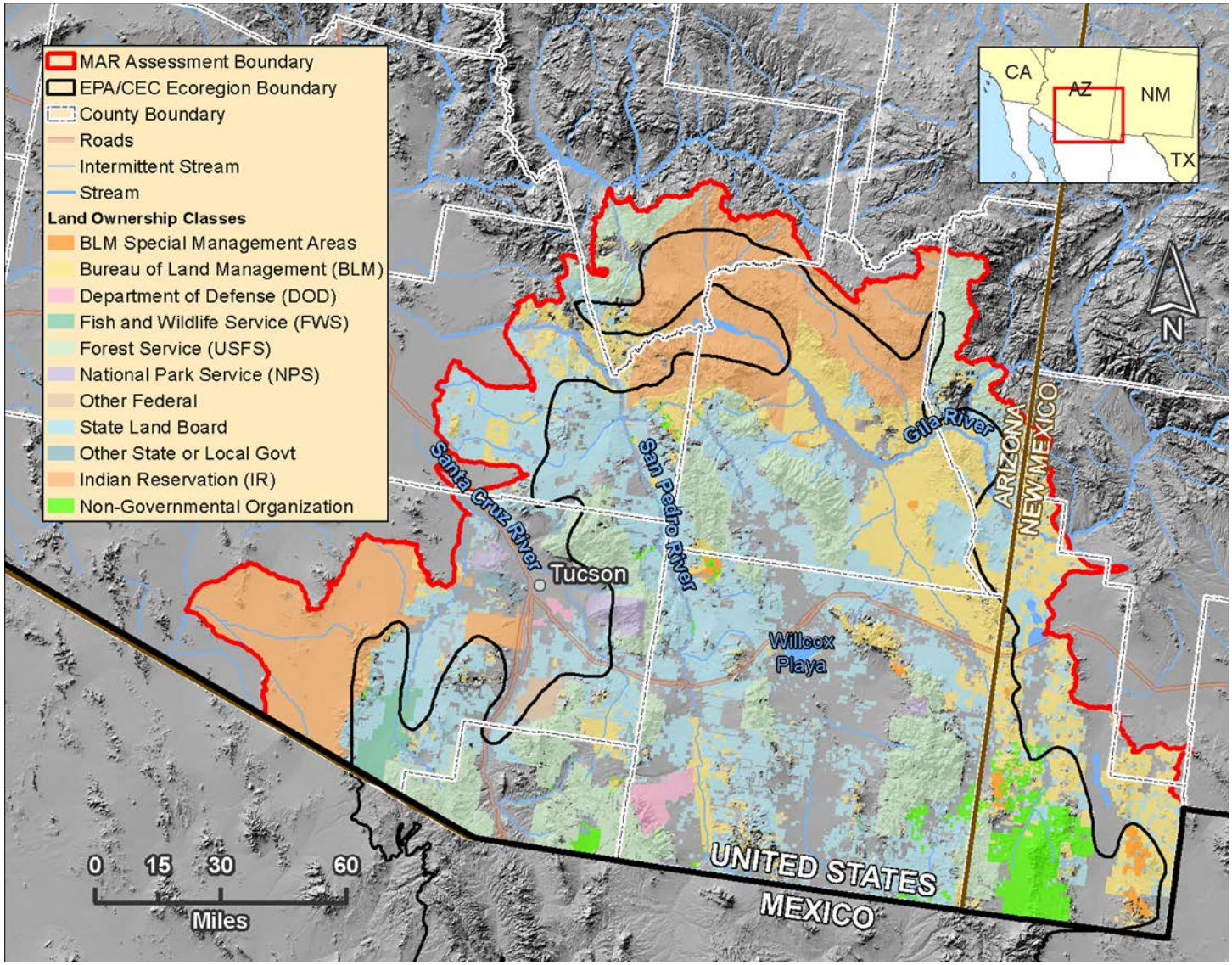
### G.3.2 Land Ownership

As in most of the American West, the majority of the land in this ecoregion is in public ownership (Table G-1). The patterns of land ownership are the result of historical European-American settlement patterns and government agency missions. Private lands tend to be concentrated in the valleys along waterways, in part due to the proximity of water in such settings. U.S. Forest Service lands are located in the sky islands – the mountain ranges that are home to the forest ecosystems in this region; land above approximately 4,000 feet in elevation is generally National Forest. The foothills and lower elevations between the sky islands and the river valleys are predominantly managed either by the Bureau of Land Management or by Arizona's State Land Board. Buenos Aires National Wildlife Refuge is the largest USFWS refuge in the ecoregion. The Tohono O'odham Nation Reservation covers a significant area in the southwestern portion of the Madrean assessment area, and the San Carlos Apache Nation Reservation covers a large portion of the north central part of the ecoregion. The San Xavier District of the Tohono O'odham Nation is on a separate land unit lying to the east of the main area of the Tohono O'odham Nation. The Animas Foundation, which is part of the Malpai Borderlands Group, manages the 321,000-acre (129,900 ha) Diamond A Ranch in the New Mexico portion of the ecoregion. Two large military facilities are located in this ecoregion as well: Fort Huachuca, near the Arizona-Mexico border, and the Willcox Dry Lake Bombing Range on the Willcox Playa. Typical ranching operations are comprised of a small area of private land (on the order of 40-160 acres (16-65 ha)) where the ranch is headquartered and contiguous grazing allotments that are leased from federal or state agencies. The broad ownership patterns of this ecoregion are illustrated in Figure G-15; ownership information is from the USGS' Protected Areas Database of the United States (PAD-US), version 1.3 (USGS 2012). Areas with no ownership information, such as the Willcox Basin around the Willcox Playa, as well as ownership types not fitting in the other categories listed, are grouped as Other/Unspecified.

**Table G-1. Percentage of the Madrean Archipelago ecoregional assessment area in major categories of land ownership.** Based on USGS' Protected Areas Database, v.1.3.

| Land Owner Type | Percentage of Madrean Archipelago assessment area |
|---|---|
| Federal | 32.0% |
| State | 24.9% |
| Other/Unspecified/Private | 24.18% |
| Native American | 15.9% |
| NGO | 2.6% |
| Regional and Local Agencies | 0.41% |

**Figure G-15. Map illustrating land ownership in the Madrean Archipelago ecoregional assessment area.** Map is based on USGS' Protected Areas Database of the United States (PAD-US), version 1.3 (USGS 2012). Land not otherwise designated is assumed to be privately owned.



### G.3.3 Land Uses

The predominant land use in the Madrean Archipelago in terms of spatial extent is cattle grazing; public lands which are not otherwise designated for conservation are generally rangeland. Although estimated to occupy less than 1.5% of the ecoregion (based on LANDFIRE vegetation/land cover data), agriculture is another important land use in this ecoregion; agricultural lands have replaced critical riparian and other lowland habitat and are responsible for a significant portion of cultural water use in the ecoregion. Tucson is the largest metropolitan area in this ecoregion, with smaller municipalities and other developed areas also present. A relatively sparse network of transportation and utility corridors span the ecoregion. Mining is the largest industrial land use in this ecoregion; open-pit copper mines and other large mines appear as substantial areas of "mines and other disturbed land" in Figure G-14. These anthropogenic land uses and associated activities give rise to or are the direct cause of the change agents that are affecting ecological systems and their driving or supporting processes; these land uses and associated CAs are discussed in detail in the Current Issues chapter.

### G.3.4 Current Issues in the Madrean Archipelago Ecoregion

The Madrean Archipelago ecoregion faces many issues relating to its natural resources as a result of the interplay between human activities and influences and the physical and ecological processes shaping the ecoregion. These issues have been summarized in a variety of reports and publications, such as the Heinz Center report on climate change and Arizona wildlife (Heinz Center 2011b), volume 3 of the Arizona Water Atlas (ADWR 2010a), the chapter on this ecoregion in the New Mexico Comprehensive Wildlife Strategy (NMDGF 2006), USGS' publication on U.S.-Mexico borderlands (Updike et al. 2013), and expanded upon in more detail in publications such as the periodic Sky Island/Madrean Archipelago conference proceedings (DeBano et al. 1995, Gottfried et al. 2005, Gottfried et al. 2013). Issues summarized in these reports were also identified in various forms in the REA Development Forums and include the following:

- ✓ Climate change
- ✓ Water availability
- ✓ Invasive species
- ✓ Encroachment of woody species
- ✓ Altered fire regimes and fire suppression
- ✓ Livestock grazing
- ✓ Border control activities and infrastructure
- ✓ Development (residential, industrial, utilities, etc.)
- ✓ Agriculture

These issues inform the change agents (CAs) that were assessed in this REA. For the purpose of having a standard terminology for and shared understanding of the CAs, the issues identified as being critical to this REA are organized into the four broad categories of CAs that are standard for BLM REAs:

A. Climate Change
B. Invasives
   - ✓ Non-native, invasive species
   - ✓ Managed non-native species
   - ✓ Native woody increasers
C. Fire
D. Development

- ✓ Urban/suburban, commercial, industrial development
- ✓ Roads
- ✓ Utilities
- ✓ Mining
- ✓ Energy development
- ✓ Agriculture
- ✓ Livestock grazing
- ✓ Border-related infrastructure, including barriers, roads, lighting, and related features
- ✓ Water usage associated with these activities or infrastructure

The crucial and over-arching issue of water availability is not categorized as its own CA – it is instead reflected within the Development and Climate Change categories. Cultural water use associated with development (e.g., agricultural, municipal, and industrial uses), in conjunction with projected alterations in timing and quantity of precipitation under climate change, are the forces – or CAs – shaping water availability. Regardless of how they are categorized, water availability and hydrologic issues were a key area of assessment in this REA.

These issues and their impacts on the region's natural resources are broadly summarized in Harkness et al. (2013). Note that all of these issues interact with and influence each other to varying degrees, so there is some overlap between the summaries of each of the issues. More details on the specific effects of each of these change agents on individual conservation elements are included in the conceptual models and ecological status assessments for each of the CEs (see **Appendices D, E, and F**).

### G.3.5 Considerations for Ecological Integrity

Characterizing the ecological integrity for the ecoregion as a whole is a key component of the REA. Ecological integrity is defined as the ability of the region's ecosystems to maintain their species composition, structure, spatial patterns or distribution, and ecological functions and processes within natural or acceptable ranges of variation.

The ecoregional conceptual model above provides the foundation for defining ecological integrity. As indicated in the ecoregional conceptual model, the natural features, processes, or drivers that are most important in shaping the biodiversity of the Madrean Archipelago, and are key determinants of ecological integrity, include the following:

- ➢ Elevational gradient
- ➢ Biogeography (nexus of the ecoregion between tropical and temperate)
- ➢ Biodiversity and composition of the ecosystems
- ➢ Physiography: the basin and range (or sky island and desert seas) topography
- ➢ Climate patterns: temperature, timing and patterns of precipitation
- ➢ Hydrologic regime
- ➢ Fire regime
- ➢ Connectivity (relating to basin and range topography)

The elevational gradients and landforms (basin and range topography, with ranges fairly isolated from each other) in conjunction with climate and soils are critical determinants of the overall type and distribution of ecosystems in the ecoregion. Biogeography plays an important role in the particular mix of species found in the ecoregion and is a key reason for the high level of species diversity found in this ecoregion. Fire is a key process in many of the upland ecosystems, further refining the species composition and vegetative structure of these systems; by removing vegetation in key areas of a

watershed, fire can also influence certain aspects of hydrologic regimes (i.e., surface flows). Hydrology, which is primarily determined by the region's climate (precipitation patterns) and landforms, is the main process driver in the aquatic and wetland ecosystems of the region.

Of these biophysical controls and ecological processes, the physiographic characteristics (elevation, landforms) and biogeography are immutable; these characteristics would not be evaluated as part of the assessment of ecological integrity. Climate, hydrology, biological and ecosystem diversity, fire regimes, and soils, on the other hand, may change in response to either natural or anthropogenic influences. Changes in these key variables can then be expected to alter the composition, structure, spatial patterns, and ecological functions of the region's ecosystems. Therefore, ideally the assessment of ecological integrity should address these variables in some fashion.

Following the approach used for individual ecological system CEs (see the CE conceptual models in Appendix D for terrestrial CEs and Appendix E for aquatic CEs), for assessment purposes ecological integrity of the ecoregion can be defined by a series of Key Ecological Attributes (KEAs), which are based on the driving variables listed above, or the resulting expression of these variables through the pattern and composition of the ecological systems themselves. A **key ecological attribute** (KEA) of an ecological resource is a characteristic of the resource's biology, ecology, or physical environment that is critical to the resource's persistence in the face of both natural and human-caused disturbance, e.g., resistance or resilience (De Leo and Levin 1997, Holling 1973, Parrish et al. 2003, Unnasch et al. 2009). Alteration of such a characteristic beyond some critical range of variation will lead to the degradation or loss of the resource within decades or less. Table G-2 lists key ecological attributes identified for the ecoregion and some possible indicators for informing the assessment of ecological integrity.

**Table G-2. List of Key Ecological Attributes (KEAs) and indicators for ecological integrity of the ecoregion.** Those measured for the MAR ecological integrity assessment are in bold.

| KEA Category | KEA | Indicators |
|---|---|---|
| Ecosystem Processes | Climate | ✓ Temperature alterations<br>✓ Precipitation alterations |
| | Hydrology/Climate | ✓ Hydrologic alterations associated with precipitation alterations due to altered climate |
| | Hydrology | ✓ **Water withdrawals for human uses (municipal, industrial, agriculture)**<br>✓ Historical downcutting – altered flow and extent of certain wetlands |
| | Fire | ✓ **Fire regime condition classes** (FRCC) |
| Ecosystem Context | Connectivity | ✓ Overall landscape permeability<br>✓ Fragstats calculation of edges or patchiness (comparing natural vs. non-natural) |
| Ecosystem (Biotic) | Percent natural cover | ✓ % converted to development or |

| KEA Category | KEA | Indicators |
|---|---|---|
| Condition | | agriculture |
| | Landscape condition | ✓ **On-site and off-site impacts resulting from various development features and other infrastructure and land uses** |
| | Condition of ecological systems | ✓ % grazed at x% utilization |
| | Native species composition | ✓ **% covered or significantly affected by invasives** |
| Ecosystem Extent (Size) | Ecosystem Size | ✓ **Change in extent of ecological systems (e.g. shift in extent of ecological systems relative to historical distribution)** |

## *G.4 Assessment of Ecological Integrity*

### G.4.1 Methods

Meaningful assessments of ecological integrity for an ecoregion must consider the diversity of distinctive landscapes contained therein. Athough there is no single accepted method for assessing ecoregional ecological integrity, ecological integrity cannot simply be a function of combining the status results for the individual CEs because these CEs do not cover the extent of the ecoregion. For example, the ecoregion has a number of sky island mountain ranges in the midst of lower elevation grasslands and desert scrub communities. Generally, it was anticipated that the lower elevation regions are more impacted by the pervasive effects of infrastructure, urbanization, mining, and agricultural activities, along with significant abundance of both native and non-native invasive plant species. The characteristics of disturbance regimes (e.g. fire and hydrologic) vary significantly elevationally, as do the human-driven alterations of those regimes. Subsetting the ecoregion into "life zones" allowed ecologically meaningful assessment of conditions that status assessment of the individual CEs (see Appendix D and E for these results) demonstrated are different from one area to another.

These varying characteristics of the ecoregion led to first treating upland integrity separately from aquatic integrity; and secondly dividing each of these (uplands versus aquatic) into major life zones within which the typical disturbance regimes and human stressors would be similar. Five life zones were defined for the integrity assessment (Table G-3): three for the uplands, and two for the wetlands and aquatic features. A distribution dataset for each upland life zone was then created by combining the distributions of the individual ecological systems found in the NatureServe (2013) terrestrial ecological systems map; see Table G-4 for the list of systems in each upland life zone. For the aquatic life zones, all of the 5$^{th}$ level watersheds in the MAR were selected under the assumption that there are, in actuality, aquatic ecosystems in every watershed and thus all watersheds should be assessed. The area of each watershed was then broken into lowland or montane using 1,524 m (5,000 ft) as a break between the two life zones.

Of note: the life zones defined for the integrity assessment are related to, but different from, the major Divisions defined in the ecoregional conceptual model in section G.2.5.1 Division Models. For the

integrity assessment of the uplands, desert scrub ecosystems were separated from the valley grasslands, due to significantly different fire regimes. For the wetlands and aquatic sytems, because the 5th level watersheds were used to represent the distributions, the hydrologic functioning considered for the Divisions could not be represented.

An additional analysis looked at the change in extent from historical pre-European settlement for a selected subset of all upland ecological systems occurring in the MAR. The intent of this is to understand how the abundance and diversity of ecosystems has changed. More specifically what individual ecosystems have been severely reduced or mostly lost and what new ecosystems or land cover types have replaced them.

**Table G-3. Life zones defined for ecological integrity assessment of the MAR; their definitions, indicators measured, and reporting units.**

| Life Zones | Definition | Indicators of Integrity | Reporting Units |
|---|---|---|---|
| **Montane Forest Life Zone** | Forests, woodlands and shrublands in montane areas, selected ecological systems mapped in the montane (generally pinyon-juniper woodlands and higher in elevation) | Development, fire regime departure, invasive plants | 4km |
| **Valley Grassland Life Zone** | Valleys and lower slopes with grasslands or savannas, including oak woodland, juniper savannas, all grassland ecological systems | Development, fire regime departure, invasive plants | 4km |
| **Desert Scrub Life Zone** | Desert regions of the MAR; selected desert scrub ecological systems mapped in the MAR excluding desert grasslands | Development, fire regime departure, invasive plants | 4km |
| **Aquatic Montane Life Zone** | All areas in all watersheds in the MAR above 1,524 m (5,000 ft) elevation | Development, invasive plants & aquatic invasives | 5th HUC |
| **Aquatic Lowland Life Zone** | All areas in all watersheds in the MAR below 1,524 m (5,000 ft) elevation | Development, water use, invasive plants & aquatic invasives | 5th HUC |

**Table G-4. List of component NatureServe ecological systems for each of the upland Life Zones with total area and percent of MAR.** The areas presented in this table are calculated from the NatureServe (2013) map of terrestrial ecological systems.

| Ecosystem name | Hectares | Acres | Percent of MAR |
|---|---|---|---|
| **Desert Scrub Life Zone** | | | |
| Apacherian-Chihuahuan Mesquite Upland Scrub | 1,242,482 | 3,070,241 | 19.419 |
| Chihuahuan Creosotebush Desert Scrub | 841,660 | 2,079,787 | 13.225 |
| Sonoran Paloverde-Mixed Cacti Desert Scrub | 677,741 | 1,674,735 | 10.222 |

| Ecosystem name | Hectares | Acres | Percent of MAR |
|---|---|---|---|
| Chihuahuan Mixed Salt Desert Scrub | 274,821 | 679,099 | 4.367 |
| Sonora-Mojave Creosotebush-White Bursage Desert Scrub | 143,296 | 354,092 | 2.097 |
| Sonoran Mid-Elevation Desert Scrub | 79,974 | 197,620 | 1.261 |
| Chihuahuan Mixed Desert and Thorn Scrub | 51,668 | 127,674 | 0.804 |
| Chihuahuan Stabilized Coppice Dune and Sand Flat Scrub | 46,619 | 115,198 | 0.731 |
| North American Warm Desert Bedrock Cliff and Outcrop | 40,687 | 100,541 | 0.646 |
| Chihuahuan Succulent Desert Scrub | 10,239 | 25,301 | 0.162 |
| North American Warm Desert Volcanic Rockland | 6,471 | 15,990 | 0.103 |
| North American Warm Desert Pavement | 3,141 | 7,761 | 0.050 |
| Sonora-Mojave Mixed Salt Desert Scrub | 2,765 | 6,831 | 0.043 |
| **Valley Grassland Life Zone** | | | |
| Apacherian-Chihuahuan Semi-Desert Grassland and Steppe | 1,158,114 | 2,861,762 | 18.139 |
| Madrean Encinal | 322,799 | 797,654 | 5.103 |
| Madrean Juniper Savanna | 18,323 | 45,277 | 0.290 |
| Chihuahuan Sandy Plains Semi-Desert Grassland | 3,337 | 8,246 | 0.053 |
| Chihuahuan Loamy Plains Desert Grassland | 2,114 | 5,224 | 0.050 |
| Chihuahuan-Sonoran Desert Bottomland and Swale Grassland | 740 | 1,829 | 0.012 |
| Inter-Mountain Basins Semi-Desert Shrub-Steppe | 400 | 989 | 0.004 |
| Inter-Mountain Basins Semi-Desert Grassland | 278 | 688 | 0.004 |
| Inter-Mountain Basins Juniper Savanna | 40 | 99 | 0.001 |
| **Montane Forest Life Zone** | | | |
| Madrean Pinyon-Juniper Woodland | 369,089 | 912,039 | 5.858 |
| Mogollon Chaparral | 303,594 | 750,196 | 4.836 |
| Madrean Lower Montane Pine-Oak Forest and Woodland | 95,118 | 235,042 | 1.512 |
| Southern Rocky Mountain Ponderosa Pine Woodland | 67,363 | 166,457 | 1.073 |
| Madrean Upper Montane Conifer-Oak Forest and Woodland | 12,653 | 31,266 | 0.202 |
| Colorado Plateau Pinyon-Juniper Woodland | 10,329 | 25,524 | 0.165 |
| Southern Rocky Mountain Ponderosa Pine Savanna | 9,363 | 23,136 | 0.149 |
| Colorado Plateau Mixed Bedrock Canyon and Tableland | 7,984 | 19,728 | 0.127 |
| Rocky Mountain Aspen Forest and Woodland | 2,843 | 7,026 | 0.045 |
| Southern Rocky Mountain Dry-Mesic Montane Mixed Conifer Forest and Woodland | 1,066 | 2,634 | 0.017 |
| Southern Rocky Mountain Mesic Montane Mixed Conifer Forest and | 692 | 1,711 | 0.011 |

| Ecosystem name | Hectares | Acres | Percent of MAR |
|---|---|---|---|
| Woodland | | | |
| Rocky Mountain Gambel Oak-Mixed Montane Shrubland | 188 | 465 | 0.003 |
| Southern Rocky Mountain Pinyon-Juniper Woodland | 176 | 436 | 0.003 |
| Rocky Mountain Cliff, Canyon and Massive Bedrock | 6 | 15 | 0.000 |
| Rocky Mountain Subalpine Dry-Mesic Spruce-Fir Forest and Woodland | 4 | 9 | 0.000 |
| Southern Rocky Mountain Montane-Subalpine Grassland | 3 | 8 | 0.000 |

#### G.4.1.1 Geospatial Methods

The ecological integrity assessment used the same geospatial methods as those used for assessing status of the individual CEs. Appendix B provides the methodological approaches to the geospatial assessments, including both the conceptual scientific approach (Appendix B: Rationale for Ecological Status Assessment Approach) to conducting status assessment and a description of the technical approach (Appendix B: Ecological Status Assessment Technical Approach). Some of the material provided here is repetitive with Appendix B, to give the reader an introduction to the methods used, but not at the level of detail provided in Appendix B. As described there, the approach uses a raster-based spatial model that begins with a theoretically perfect condition score of 1.0 for each pixel of a distribution (with zero being lowest condition). From there, a "response model" is applied for how each change agent (CA) is expected to reduce that condition score onsite and in some cases, offsite. Where multiple CAs overlap, the resulting condition scores are multiplied to approximate a cumulative CA effect. This model is called the Landscape Condition Model (LCM) (Comer and Faber-Langendoen 2013, Comer and Hak 2009). It is described in some detail in Appendix B: Ecological Status Assessment.

In summary, a score for a particular CA indicates the degree to which the impact of that CA degrades the ecological condition of the ecosystem occupying the pixel. Further information is provided below about the scoring for individual CAs.

It is important that the spatial status assessment be grounded in what is known about the ecosystem, in this case the life zone- its ecology, dynamic processes, and stressors. The work of developing the conceptual model for the ecoregion above provided the scientific context and current knowledge base from which to identify the key ecological attributes (KEAs) to assess for integrity (see Appendix B: Rationale for Ecological Status Assessment Approach), and Table G-2 which lists key ecological attributes at the ecoregion-scale.

In the MAR, three primary KEAs were the focus for assessment of the upland life zones, and for which spatial data were available: landscape condition (as effected by development), fire regime, and native vegetation [plant species] composition. For these three KEAs, data for direct measures were not available, so indirect, stressor-based measures were applied (more detail on this concept is explained in Appendix B). For each KEA, a scenario (Appendix B: Scenario Generation: Current and Future) was developed to spatially represent the CAs comprising those indirect stressors: development for landscape condition, Landfire Vegetation Condition Class for fire regime, and invasive species for native vegetation composition.

#### G.4.1.2 Landscape Condition Model Inputs

The below material is to aid the reader in understanding the results without having to consult detailed methods available in other appendixes.

As described in Appendix B (Ecological Status Assessment Technical Approach) and above, the KEA indicator scenarios were combined with a "response model" for each life zone. Once a scenario of change agent features was created (e.g. indicators for landscape condition, fire regime and vegetation composition), a response model was needed to tell the geospatial Landscape Condition Model (LCM) how the change agent affects the integrity. The response model was constructed using the same values as for the individual terrestrial or aquatic ecosystem CEs for particular change agents and is represented in a simple table (Table G-5). The same site intensity values were used for the Valley Grassland and Montane Forest life zones, but slightly different ones were used for the Desert Scrub for the invasives change agent. The Aquatic Montane life zone was not assessed for water use, however, the Aquatic Lowland life zone was. The table is provided once, but is referred to in subsequent sections.

The site intensity and distance values were developed and tested previously in two other REAs (for the Central Basin & Range and Mojave Basin & Range), and were also reviewed for the individual CEs status assessments for the MAR REA during the process of completing those geospatial analyses.

Within the Vista tool (see Appendix C—Technical Methods: Documentation of GIS Data Processing for All Assessments for more details), each CA has a site intensity value and a value for the distance at which the impact ends. Intensity values are rated between 0 and 1 with 0 being highest possible impact that removes all ecological integrity completely from that pixel. A value of 0.05 removes 99.95% of the integrity, 0.5 removes 50%, 0.7 30% and so on. Therefore a paved highway has a very high impact value (0.05) as the natural ecosystem has been removed from that 30 m pixel. Other CAs, such as invasive species, can have a large negative ecological impact, but are scored in the 0.5-0.8 range, as even though there is an invasive infestation, it is not the same loss of integrity as from removal by a paved highway for the 30 m pixel. See Appendix B for conceptual information and Appendix C for GIS documentation and application methods.

The five life zones were then assessed using the full scenario generation and status assessment in Vista as described in those process models in Appendix B. This resulted in an output of a raster map for each life zone for integrity for all scenarios combined, as relevant to each life zone (see Table G-3). The results were summarized from 30m analysis units to 4km reporting units for the uplands and 5$^{th}$ level watersheds for the aquatic life zones.

**Table G-5. Site intensity values and distance effects used in the Landscape Condition Model for assessing ecological integrity of the upland and aquatic life zones of the MAR.** Site intensity values are scaled 0.0 - 1.0, are relative to each other, and reflect how much an activity removes ecological integrity. A value of 0.05 removes 99.95% of the integrity, 0.5 removes 50%, 0.7 30% and so on. Where 2 or more activities occur within the same pixel, the values are multiplied together. Distances are in meters. See Appendix B for conceptual information and Appendix C for GIS documentation and application methods. Note on the distance weight: the Vista software requires a minimum distance of 10m as an input, but because the actual pixel size is 30m, no distance effect is calculated in the model unless the distance is >30m.

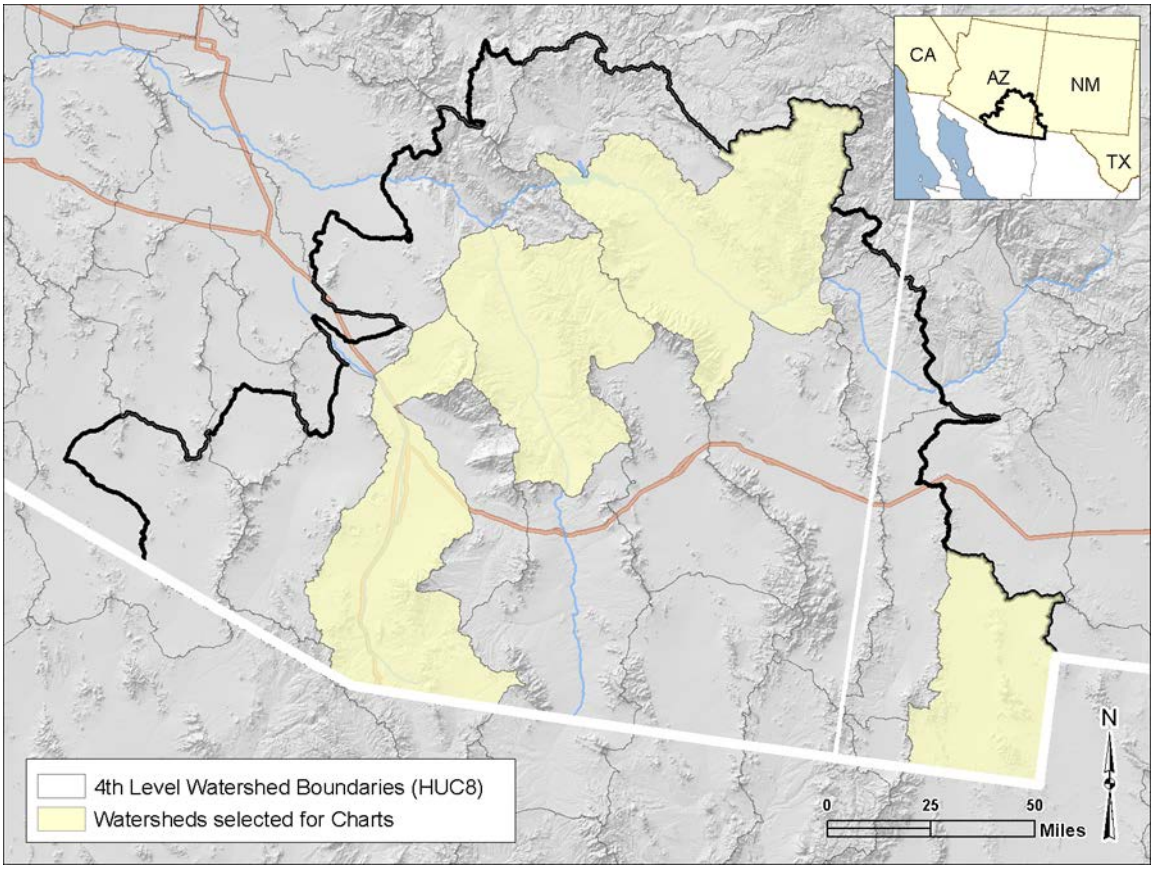
| | Aquatic Lowland Life Zone | | Aquatic Montane Life Zone | | Desert Scrub Life Zone | | Montane Forest & Valley Grassland Life Zones | |
|---|---|---|---|---|---|---|---|---|
| **Land Use Name** | **Site Intensity** | **Distance (m)** | **Site Intensity** | **Distance (m)** | **Site Intensity** | **Distance (m)** | **Site Intensity** | **Distance (m)** |
| Infrastructure | | | | | | | | |
| ----Border Barrier - Pedestrian | 0.5 | 100 | 0.5 | 100 | 0.5 | 10 | 0.5 | 10 |
| ----Border Barrier - Vehicle | 0.6 | 100 | 0.6 | 100 | 0.6 | 10 | 0.6 | 10 |
| ----Communication Towers | 0.3 | 200 | 0.3 | 200 | 0.3 | 10 | 0.3 | 10 |
| ----Below Ground Corridors | 0.7 | 200 | 0.7 | 200 | 0.7 | 10 | 0.7 | 10 |
| ----Above Ground Corridors | 0.5 | 100 | 0.5 | 100 | 0.5 | 10 | 0.5 | 10 |
| Transportation | | | | | | | | |
| ----Dirt & 4-wheel Drive Roads | 0.3 | 200 | 0.3 | 200 | 0.7 | 10 | 0.7 | 10 |
| ----Local/Rural/Private Roads | 0.2 | 500 | 0.2 | 500 | 0.2 | 10 | 0.2 | 10 |
| ----Primary Highways w/ Limited Access | 0.05 | 2000 | 0.05 | 2000 | 0.05 | 10 | 0.05 | 10 |
| ----Primary Highways w/o Limited Access | 0.05 | 1000 | 0.05 | 1000 | 0.05 | 10 | 0.05 | 10 |
| ----Airstrips | 0.5 | 500 | 0.5 | 500 | 0.5 | 10 | 0.5 | 10 |
| ----Railroads | 0.5 | 200 | 0.5 | 200 | 0.5 | 10 | 0.5 | 10 |
| Mining & Landfills | | | | | | | | |
| ----High Impact Mines/Landfills | 0.05 | 200 | 0.05 | 200 | 0.05 | 10 | 0.05 | 10 |

| | Aquatic Lowland Life Zone | | Aquatic Montane Life Zone | | Desert Scrub Life Zone | | Montane Forest & Valley Grassland Life Zones | |
|---|---|---|---|---|---|---|---|---|
| **Land Use Name** | **Site Intensity** | **Distance (m)** | **Site Intensity** | **Distance (m)** | **Site Intensity** | **Distance (m)** | **Site Intensity** | **Distance (m)** |
| ----Medium Impact Mines/Landfills | 0.6 | 50 | 0.6 | 50 | 0.6 | 10 | 0.6 | 10 |
| ----Low Impact Mines/Landfills | 0.9 | 10 | 0.9 | 10 | 0.9 | 10 | 0.9 | 10 |
| Energy | | | | | | | | |
| ----Geothermal Energy | 0.5 | 200 | 0.5 | 200 | 0.5 | 10 | 0.5 | 10 |
| ----Wind Energy | 0.8 | 500 | 0.8 | 500 | 0.8 | 10 | 0.8 | 10 |
| ----Solar Energy | 0.5 | 500 | 0.5 | 500 | 0.5 | 10 | 0.5 | 10 |
| ----Oil & Gas Wells | 0.4 | 500 | 0.4 | 500 | 0.5 | 10 | 0.5 | 10 |
| Recreation: Trails - Hiking/Biking/Horse | 0.7 | 100 | 0.7 | 100 | 0.9 | 10 | 0.9 | 10 |
| Agriculture | 0.3 | 200 | 0.3 | 200 | 0.3 | 0 | 0.3 | 10 |
| Urbanization | | | | | | | | |
| ----Low Density Development | 0.6 | 200 | 0.6 | 200 | 0.6 | 10 | 0.6 | 10 |
| ----Medium Density Development | 0.5 | 200 | 0.5 | 200 | 0.5 | 10 | 0.5 | 10 |
| ----High Density Development | 0.05 | 2000 | 0.05 | 2000 | 0.05 | 10 | 0.05 | 10 |
| Invasives | | | | | | | | |
| ----Aquatic Invasives - High Impact Species | 0.5 | 1200 | 0.5 | 1200 | 0.9999 | 10 | 0.9999 | 10 |
| ----Aquatic Invasives - Low Impact Species | 0.7 | 1200 | 0.7 | 1200 | 0.9999 | 10 | 0.9999 | 10 |
| ----Aquatic - Presense of Tamarisk | 0.7 | 10 | 0.7 | 0 | 0.9999 | 10 | 0.9999 | 10 |
| ----Terrestrial Invasives - Low Cover | 0.9 | 10 | 0.9 | 0 | 0.85 | 10 | 0.9 | 10 |
| ----Terrestrial Invasives - Medium Cover | 0.8 | 10 | 0.8 | 0 | 0.75 | 10 | 0.8 | 10 |
| ----Terrestrial Invasives - High Cover | 0.7 | 10 | 0.7 | 0 | 0.65 | 10 | 0.7 | 10 |

| | Aquatic Lowland Life Zone | | Aquatic Montane Life Zone | | Desert Scrub Life Zone | | Montane Forest & Valley Grassland Life Zones | |
|---|---|---|---|---|---|---|---|---|
| **Land Use Name** | **Site Intensity** | **Distance (m)** | **Site Intensity** | **Distance (m)** | **Site Intensity** | **Distance (m)** | **Site Intensity** | **Distance (m)** |
| Mesquite Cover | | | | | | | | |
| ----Mesquite - Low Cover | | | | | 0.9 | 10 | 0.9 | 10 |
| ----Mesquite - Medium Cover | | | | | 0.8 | 10 | 0.8 | 10 |
| ----Mesquite - High Cover | | | | | 0.7 | 10 | 0.7 | 10 |
| Fire Regime Departure | | | | | | | | |
| ----Moderate Fire Regime Departure | | | | | 0.75 | 10 | 0.75 | 10 |
| ----Severe Fire Regime Departure | | | | | 0.65 | 10 | 0.65 | 10 |
| Dams | | | | | | | | |
| ----Very Large Inundation Area | 0.4 | 1000 | 0.4 | 1000 | | | | |
| ----Large Inundation Area | 0.4 | 500 | 0.4 | 500 | | | | |
| ----Dam Present | 0.4 | 200 | 0.4 | 200 | | | | |
| Water Use | | | | | | | | |
| ----Total Water Use - Low | 0.9999 | 10 | 0.9999 | 0 | | | | |
| ----Total Water Use - Medium | 0.8 | 10 | 0.9999 | 0 | | | | |
| ----Total Water Use - Medium-High | 0.6 | 10 | 0.9999 | 0 | | | | |
| ----Total Water Use - High | 0.5 | 10 | 0.9999 | 0 | | | | |

A second component of measuring ecological integrity for the uplands was a comparison of historical upland ecosystem composition (as represented by areal extent) with current ecosystem composition. The Landfire biophysical settings (BpS) dataset was used to represent historical ecosystem distribution and the NatureServe map (NatureServe 2013) to represent current distribution. Each of these maps was intersected with the map of 8-digit watersheds (4th level HUCs) (Figure G-16) and hectares of extent were generated by watershed within the MAR assessment boundary. Tabular comparisons of areal extent for historical and current distributions of individual ecological systems were then calculated for each watershed and for the MAR ecoregion as a whole. Results for the yellow shaded watersheds in Figure G-16 are provided in the following section, all tabular results have been provided to BLM in an Excel workbook.

**Figure G-16. Watersheds of the MAR used for the terrestrial change in extent analysis.** The light gray boundaries are the 4th level (8 digit) watersheds, also called HUC8. Watersheds shaded in light yellow were selected for comparison of historical and current ecological system distributions by watershed.



## *G.5 Results: Ecological Integrity Assessment*

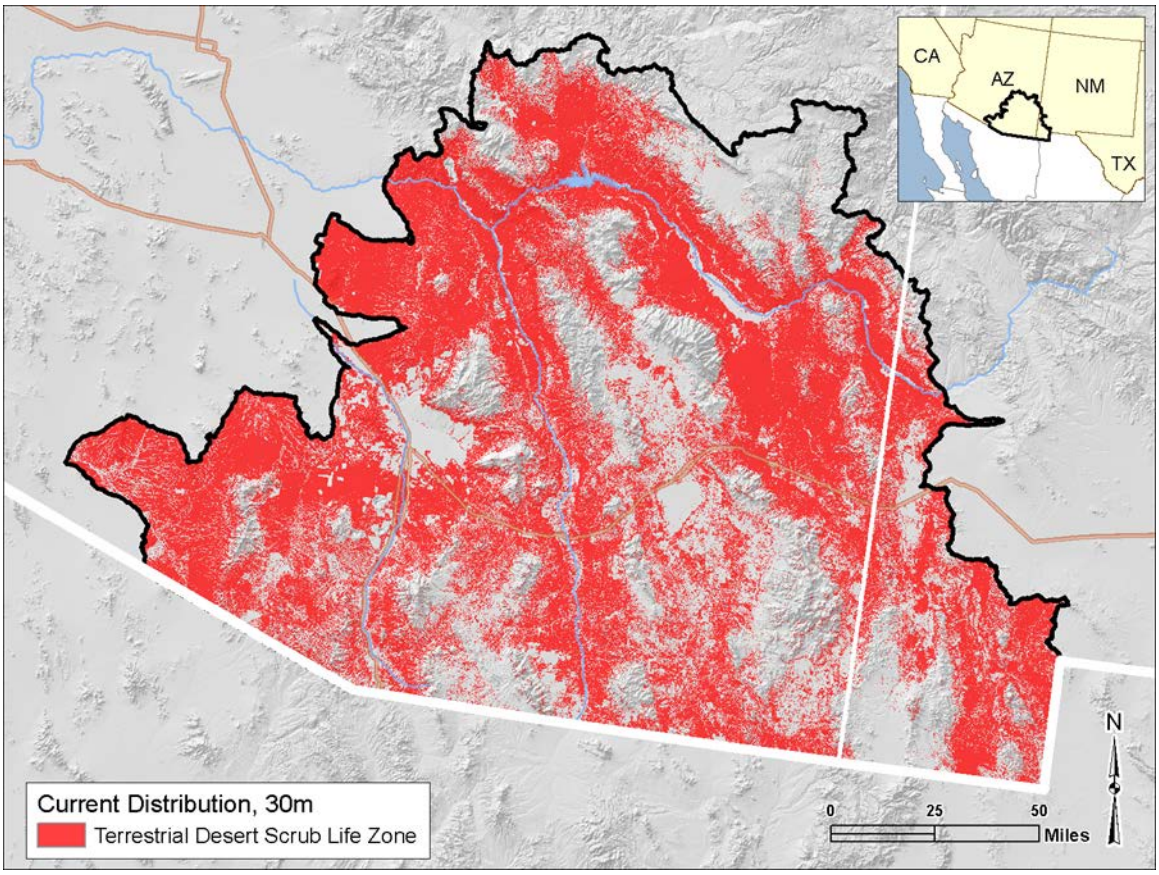
### G.5.1 Ecological Integrity for the Uplands of the Ecoregion

#### G.5.1.1 Terrestrial Desert Scrub Life Zone

The Terrestrial Desert Scrub life zone is composed of thirteen NatureServe ecological systems (Table G-4) and is dominated by three matrix systems: Apacherian-Chihuahuan Mesquite Upland Scrub, Chihuahuan Creosotebush Desert Scrub, and Sonoran Paloverde-Mixed Cacti Desert Scrub, which comprise over 40% of the Madrean Archipelago (Table G-4). The group of desert scrub ecological systems composing this life zone is adapted to the hot, dry climate, periodic drought and poorly developed soils. Stands occur on sand, saline or alkaline clays, gravelly loams, and rock outcrops, commonly at lower elevation sites especially valleys and plains. The vegetation is typically open with large spaces between shrubs with herbaceous cover sparse. The soil surface is mostly bare and sometimes a pebbly desert pavement may form. With such low cover widespread fire was a relatively rare occurrence historically, and most of these communities are poorly adapted to fire. For example, one of the dominant plants, creosotebush (*Larrea tridentata*), is fire-sensitive because of its highly flammable, resinous leaves and limited sprouting ability after burning although it may survive lower intensity fires (Brown and Minnich 1986, Humphrey 1974, Marshall 1995, Paysen et al. 2000). Other plants, such as many of the cacti and other succulents, are eliminated by burning. A few shrubs such as mesquite (*Prosopis* spp.) sprout well following burning. Invasive species such as annual red brome (*Bromus rubens*) are often abundant following wet years and pose a fire hazard by creating a continuous fuel layer (Beatley 1966).The perennial invasive buffelgrass (*Pennisetum ciliare*) is expanding into desert scrub areas and drastically changing the fire regime (Brenner and Kanda 2013).

The distribution of the Terrestrial Desert Scrub life zone throughout the MAR is shown in Figure G-17. Stands are common at lower elevation sites especially valleys and plains. The life zone distribution was created by combining the mapped distributions of the component terrestrial ecological systems. The distributions of component systems were derived from the land cover mapping work completed by SW ReGap (Lowry et al. 2005) and refined by NatureServe (2013).

**Figure G-17. Distribution of the Terrestrial Desert Scrub life zone at 30 m resolution in the Madrean Archipelago ecoregional assessment area.** This map shows the distributions of 13 terrestrial ecological systems which were combined to create the the distribution of this life zone. The Terrestrial Desert Scrub life zone is dominated by three matrix systems: Apacherian-Chihuahuan Mesquite Upland Scrub, Chihuahuan Creosotebush Desert Scrub, and Sonoran Paloverde-Mixed Cacti Desert Scrub.

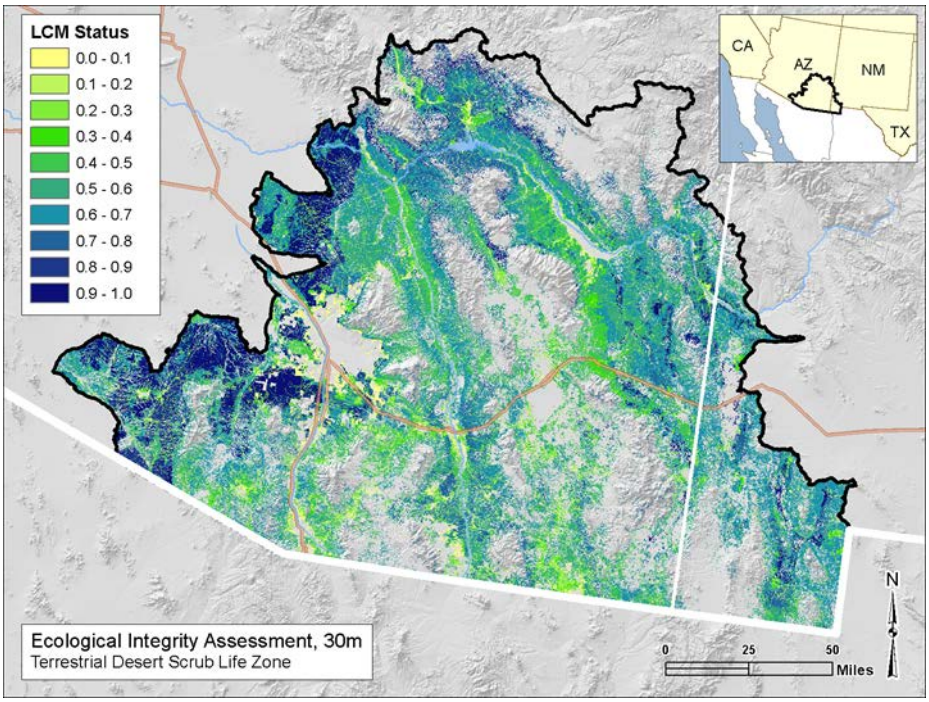


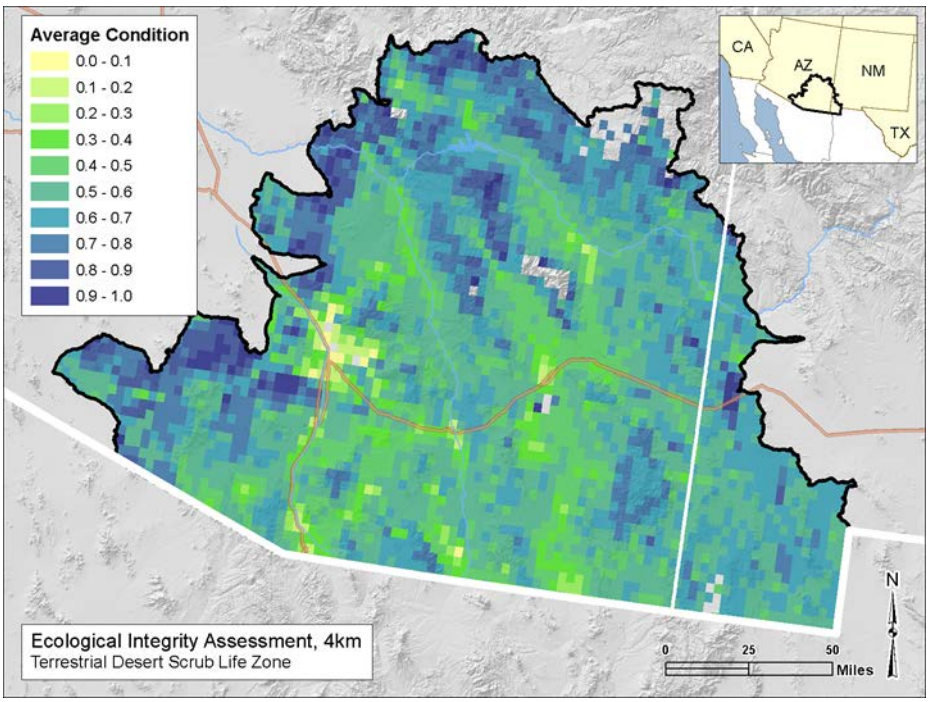

The two maps in Figure G-18 summarize the assessment results for the Desert Scrub life zone. The first map illustrates the results of all of the indicators, landscape condition-development, fire regime departure, and invasive species, which were combined into a single ecological integrity score per pixel of the Terrestrial Desert Scrub life zone distribution. The second map shows the combined integrity score for each pixel summarized to the reporting unit (e.g., 4km grid) by taking the average integrity score from all the pixels of the life zone distribution within the reporting unit. The results indicate that the degradation of the Terrestrial Desert Scrub life zone in the ecoregion is concentrated near larger urban areas, along major transportation and infrastructure routes, and in agricultural areas.

The results of this assessment are largely driven by the landscape condition-development indicator. There are areas of better ecological conditions especially in the western extent and around the periphery of the ecoregion, a result of low levels of development, low or no cover of invasive species, and moderate fire regime departure.

**Figure G-18. Current ecological integrity scores for Terrestrial Desert Scrub Life Zone for all indicators combined (development, fire regime and invasives) for each 30m pixel (top) and 4km grid cells (bottom).** The score for each 4km cell is an average of all 30m pixels that are scored for the life zone. For

example, there are 500 30m pixels of the life zone each with a score for all indicators, those 500 30m pixels are averaged for the 4km grid cell. Yellow scores (equivalent to 0) indicate high impacts from the CAs, dark blue (equivalent to 1) indicate little to no impact from the CAs.




### G.5.1.2 Terrestrial Valley Grassland Life Zone

The Terrestrial Valley Grassland life zone is composed of nine NatureServe ecological systems (Table G-4) and is dominated by the Apacherian-Chihuahuan Semi-Desert Grassland and Steppe and Madrean

Encinal, which comprise over 23% of the Madrean Archipelago (Table G-4). The semi-desert grassland, savanna, and shrub steppe ecological systems in this life zone are adapted to the hot, semi-arid clmate, and periodic drought. The vegetation is characterized by a moderate to dense perennial grass layer with or without scattered shrubs and trees. The life zone developed under fire regime with frequent fires (McPherson 1995, Schussman 2006a, Schussman 2006b). These frequent fires (fire return intervals of 2.5 to 10 years) maintained these open grasslands with low shrub and tree cover (Brown and Archer 1999, McPherson 1995, Robinett 1994, Wright 1980). Active and passive fire suppression over the last century has excluded fire from much of this life zone (Gori and Enquist 2003, Schussman 2006a, Schussman 2006b), and this fire exclusion has allowed increased woody species cover and resulted in an uncharacteristic fire regime in many stands (Barton 1999, Gori and Enquist 2003, Muldavin et al. 2002, Turner et al. 2003). This altered (uncharacteristic) fire regime greatly influences ecosystem processes, resulting in grasslands becoming dominated by woody vegetation and eventually converted to shrublands or woodlands. Conversion to juniper woodlands or mesquite or creosotebush shrublands is common when trees or shrubs exceed 15% cover (Gori and Enquist 2003). less disturbed foothill zone.

Figure G-19 shows the distribution of this life zone concentrated mainly in the southern half of the ecoregion with two additional large areas northeast of the Gila Mountains. The life zone distribution was created by combining the mapped distributions of the component Terrestrial Ecological Systems from the NatureServe (2013) map.

The two maps Figure G-20 summarize the assessment results for the Terrestrial Valley Grassland life zone. The first map illustrates the results of all of the indicators: landscape condition-development, fire regime departure, and invasive species, which were combined into a single ecological integrity score per 30m pixel of the Terrestrial Valley Grassland life zone distribution. The second map shows the combined integrity score for each pixel summarized to the reporting unit (e.g., 4km grid) by taking the average integrity score from all the pixels of the life zone distribution within the reporting unit.

The results in Figure G-20 indicate that degradation of the Terrestrial Valley Grassland life zone in the ecoregion is concentrated near agriculture and urban areas such as Sulfur Springs Valley near Willcox, Fort Huachuca/Sierra Vista area, Sonoita and San Bernardino Valley. The results of this assessment are largely driven by the landscape condition-development indicator, but the fire regime is severely departed across its range and there is some effect of invasives in the southern areas. The areas of better ecological conditions occur in the “Bootheel” of New Mexico, the Animas Valley west of Lordsburg, the large stands northwest of the Gila Mountains and scattered smaller patches in the higher elevation, less disturbed foothill zone.

**Figure G-19. Distribution of the Terrestrial Valley Grassland Life Zone at 30 m resolution in the Madrean Archipelago ecoregional assessment area.** This map shows the distributions of nine terrestrial ecological systems which were combined to create the the distribution of this life zone. The Terrestrial Valley Grassland life zone is dominated by two systems: Apacherian-Chihuahuan Semi-Desert Grassland and Steppe and Madrean Encinal.

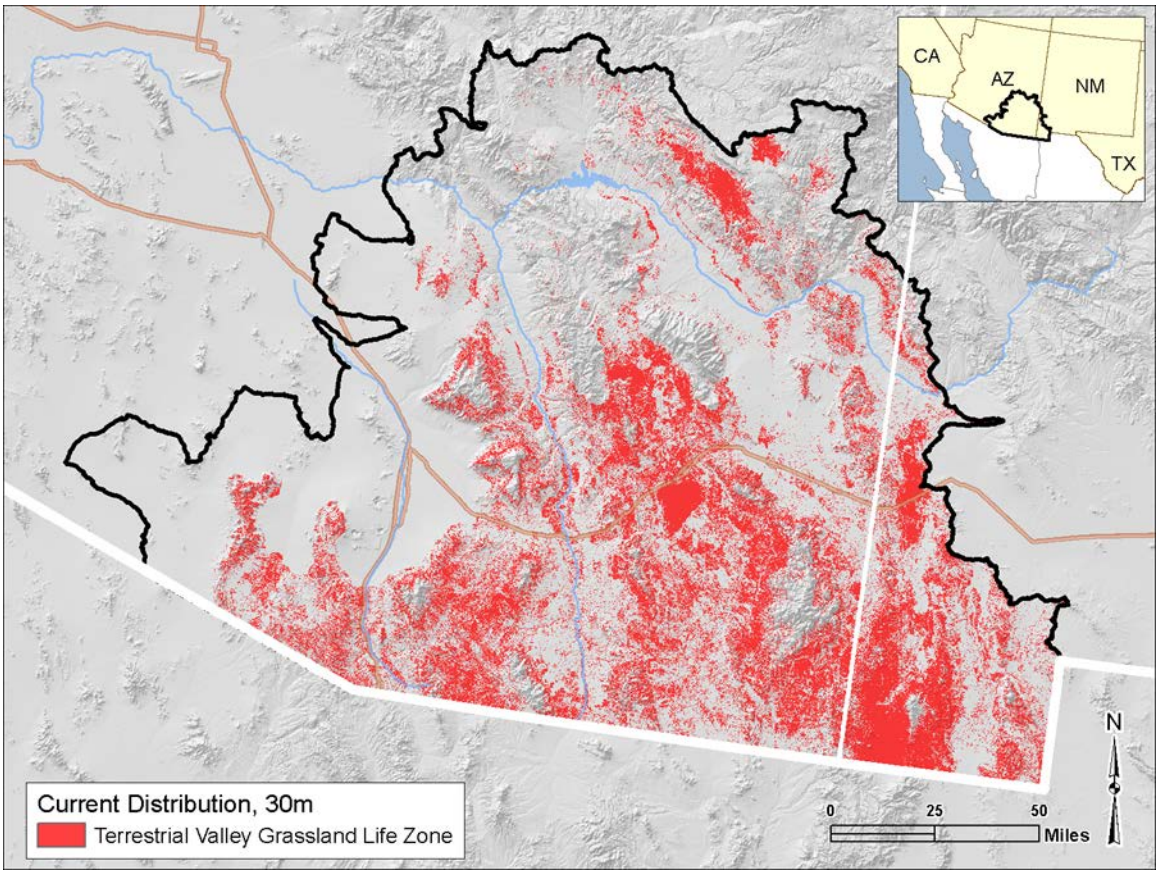
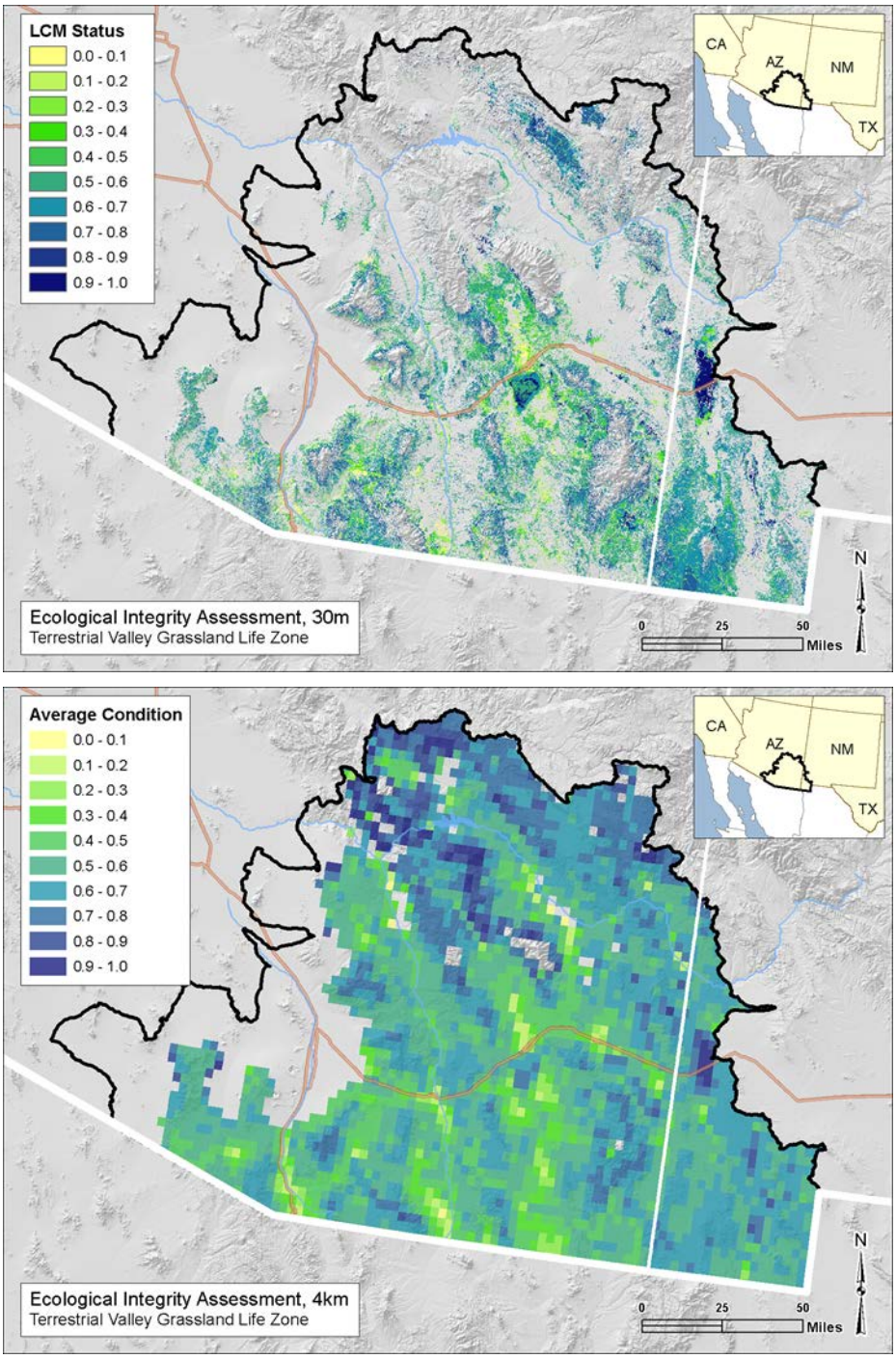


**Figure G-20. Current ecological integrity scores for Terrestrial Valley Grassland Life Zone for all indicators combined (development, fire regime and invasives) for each 30m pixel (top) and 4km grid cells (bottom).** The score for each 4km cell is an average of all 30m pixels that are scored for the life zone. For example, there are 500 30m pixels of the life zone each with a score for all indicators, those 500 30m pixels are averaged for the 4km grid cell. Yellow scores (equivalent to 0) indicate high impacts from the CAs, dark blue (equivalent to 1) indicate little to no impact from the CAs.



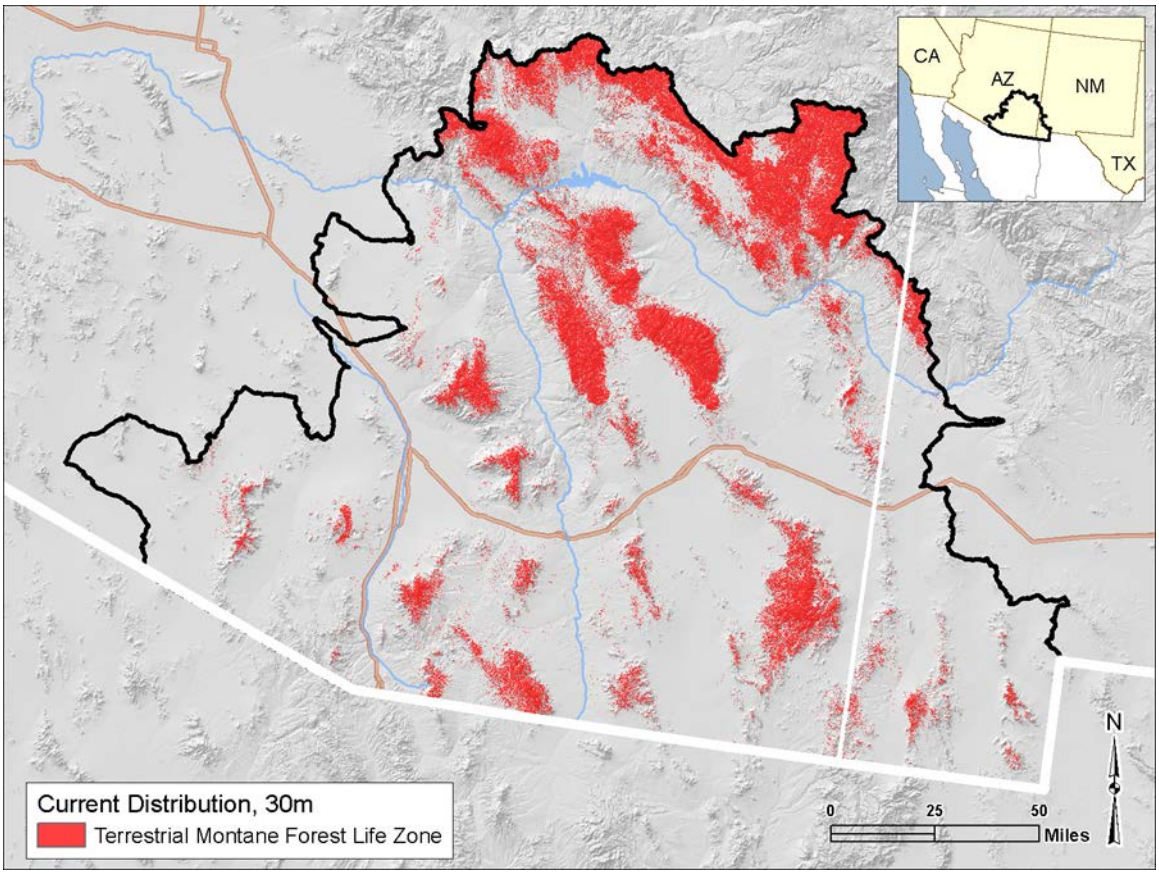
#### G.5.1.3 Terrestrial Montane Forest Life Zone

The Terrestrial Montane Forest Life Zone is composed of sixteen NatureServe ecological systems (Table G-4) and is dominated by two systems: Madrean Pinyon-Juniper Woodland and Mogollon Chaparral that comprise over 10% of the MAR. There are also significant amounts of Madrean Lower Montane Pine-Oak Forest and Woodland and Southern Rocky Mountain Ponderosa Pine Woodland. The remaining components tend to be small patch and/or occur locally, or are restricted to higher elevation mountains.

This montane life zone includes forest, woodland, shrubland, grassland and sparsely-vegetated substrate-driven ecological systems. The vegetation is characterized by open to dense tree and/or shrub layers with a variable perennial grass layer. Historical fire regimes for the systems in this life zone are variable. Mean fire return intervals range from 20-80 years for much of the systems with the more open pine savanna and oak-pine woodland types having more frequent low-severity surface fires that are carried by the herbaceous layer. Other montane systems, especially the sparsely-vegetated and higher elevation forests and chaparral types burned less frequently. Active and passive fire suppression over the last century has excluded fire from much of this life zone (Gori and Bate 2007). In the absence of disturbance such as fire, the woody component increased in density over time and subsequent fires became more severe (Gori and Bate 2007, Swetnam and Baisan 1996, Turner et al. 2003).

The distribution (Figure G-21) of this life zone is concentrated at upper elevation sites in the Sky Island ranges and the northern half of MAR. The life zone distribution was created by combining the mapped distributions of the component terrestrial ecological systems from the NatureServe (2013) map.

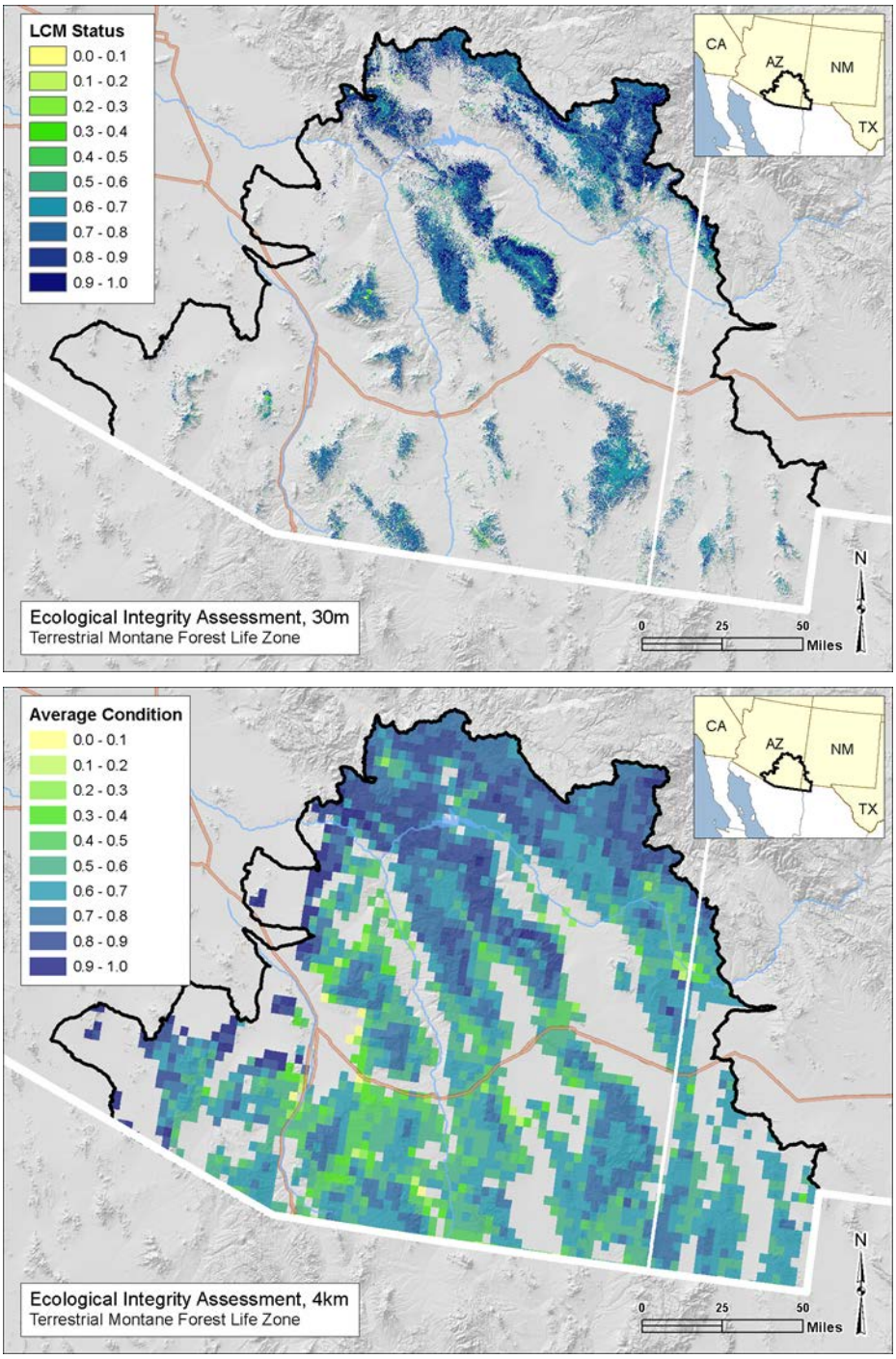
**Figure G-21. Distribution of the Terrestrial Montane Forest Life Zone at 30 m resolution in the Madrean Archipelago ecoregional assessment area.** This map shows the distributions of 16 terrestrial ecological systems which were combined to create the the distribution of this life zone. The Terrestrial Montane Forest life zone is dominated by two systems: Madrean Pinyon-Juniper Woodland and Mogollon Chaparral.




The maps in Figure G-22 summarize the assessment results for the Terrestrial Montane Forest life zone. The first map illustrates the results of all of the indicators: landscape condition-development, fire regime departure, and invasive species, which were combined into a single ecological integrity score per pixel of the Terrestrial Montane Forest life zone distribution. The second map shows the combined integrity score for each pixel summarized to the reporting unit (e.g., 4km grid) by taking the average integrity score from all the pixels of the life zone distribution within the reporting unit.

The results shown in Figure G-22 indicate a general degradation of the Terrestrial Montane Forest life zone in the southern portion of the MAR ecoregion, primarily from severely altered fire regimes. There are local areas of severe degradation driven by poor landscape condition such as in the historic mining district of Bisbee in the Mule Mountains and development in the Santa Catalina Mountains north of Tucson. The mountains in the northern portion of the ecoregion including the northern Sky Island ranges (Galiuro, Gila and Pinaleño mountains) are in the best ecological condition with only moderate fire regime departure and apparently good landscape condition (low development) and low impacts from invasives.

**Figure G-22. Current ecological integrity scores for Terrestrial Montane Forest Life Zone for all indicators combined (development, fire regime and invasives) for each 30m pixel (top) and 4km grid cells (bottom).** The score for each 4km cell is an average of all 30m pixels that are scored for the life zone. For example, there are 500 30m pixels of the life zone each with a score for all indicators, those 500 30m pixels are averaged for the 4km grid cell. Yellow scores (equivalent to 0) indicate high impacts from the CAs, dark blue (equivalent to 1) indicate little to no impact from the CAs.

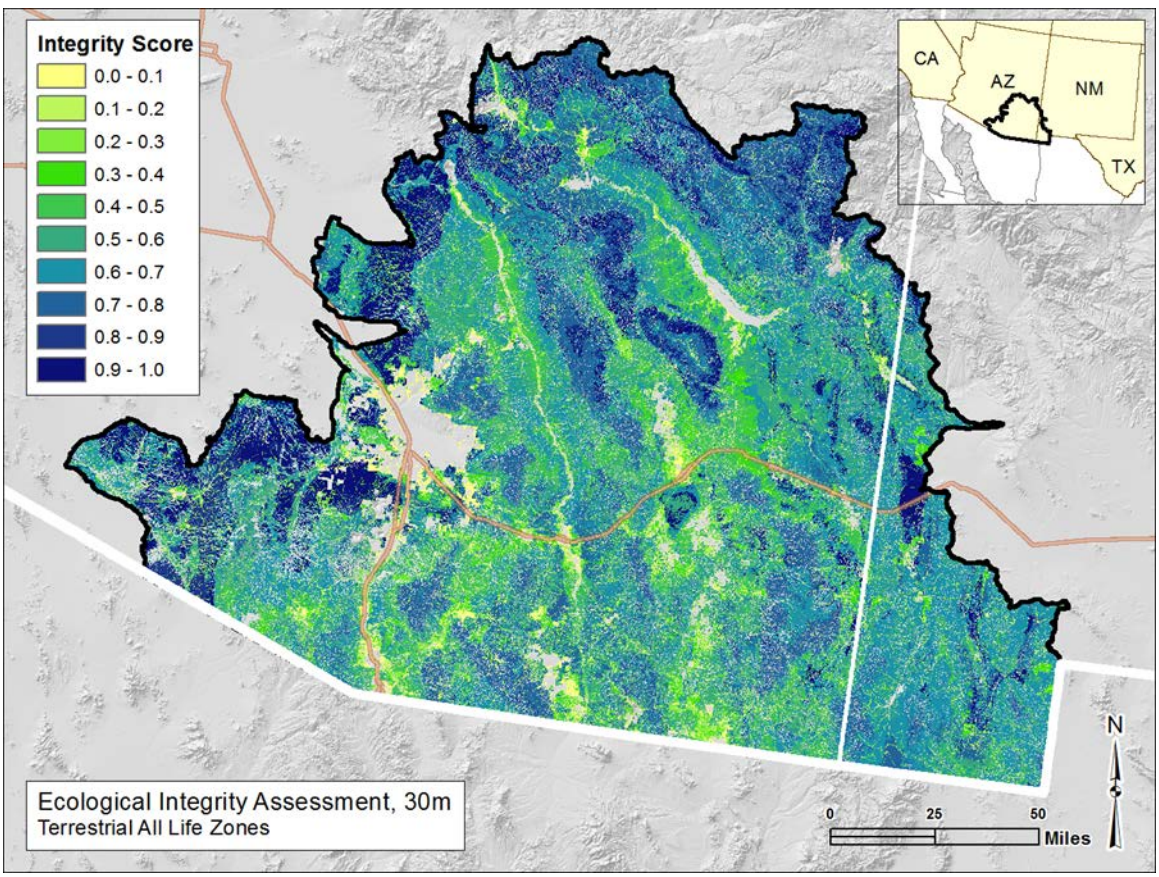


#### G.5.1.4 Ecological Integrity For Terrestrial Life Zones

Because each of the three terrestrial life zones are mutually exclusive in their distributions at a 30m resolution, the ecological integrity scores as presented above can be combined into a single map (Figure G-23). Gray areas in the map are places with development, agricultural activities, or introduced exotic vegetation types. Because the scores for the reporting units ($4km^2$) are coarser, there is a great deal of overlap across the three terrestrial life zones (Figure G-18, Figure G-20, and Figure G-22; lower map in each). The way those scores are calculated means that when 2 or 3 life zones are in the same reporting unit, the integrity score is often different for each of those life zones. Creating a $4km^2$ resolution map is not feasible because of this.

In general, integrity appears to be better across the northern and western portions of the MAR, and also at higher elevations.

**Figure G-23. Current ecological integrity scores for all three Terrestrial Life Zones combined into a single map for the ecoregion.** As with the scores for the individual life zones above, yellow scores (equivalent to 0) indicate high impacts from the CAs, dark blue (equivalent to 1) indicate little to no impact from the CAs. Gray areas are places with development, agricultural activities, or introduced exotic vegetation types. This map is just to visualize terrestrial integrity for the entire ecoregion, the individual life zone datasets have not been combined into a single dataset for delivery.

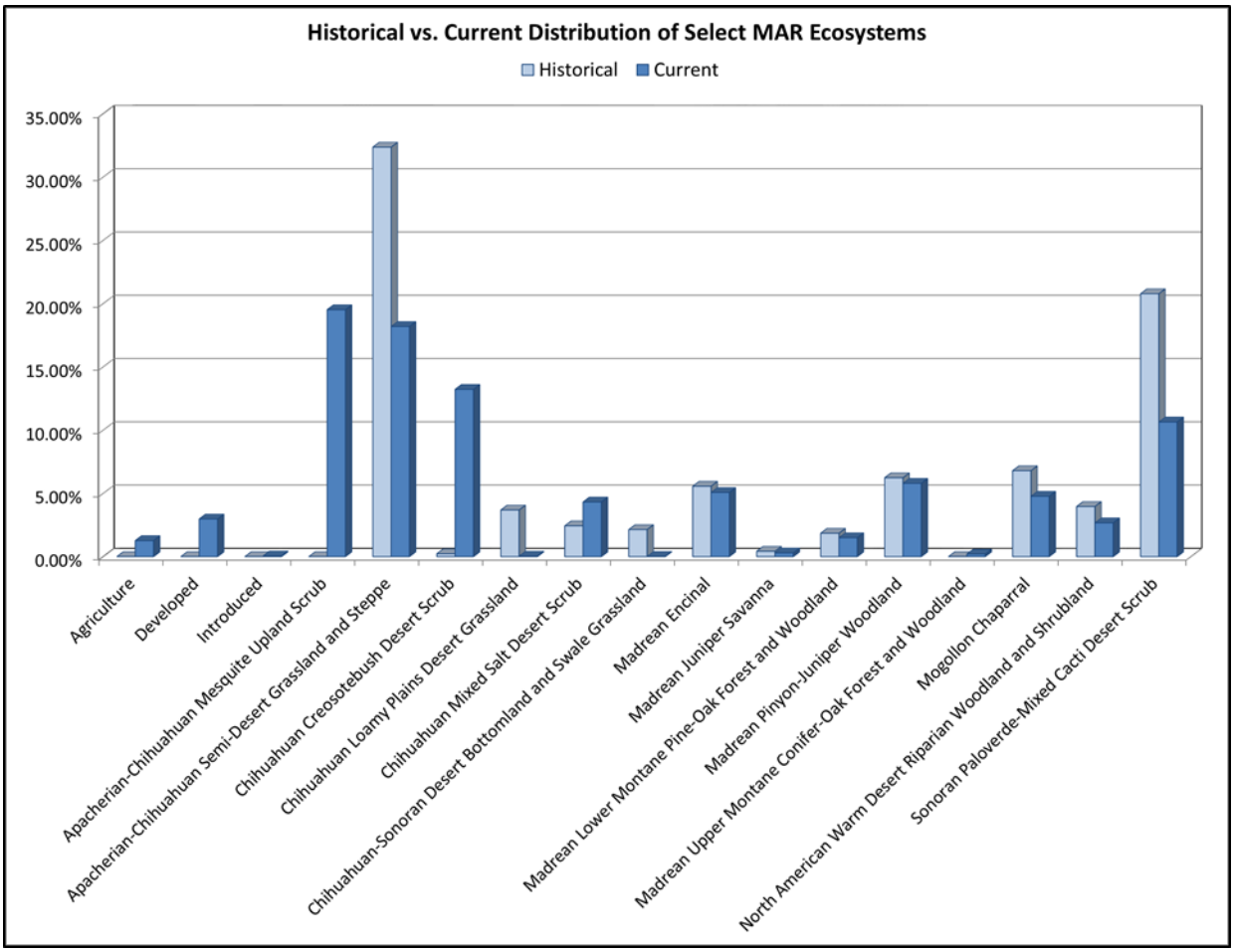


#### G.5.1.5 Change in Extent of the Uplands in MAR

As described previously, this analysis utilizes mapped distributions from the LANDFIRE BpS dataset for historical distribution, and from the NatureServe ecological systems dataset for the current distribution (NatureServe 2013). The NatureServe ecological systems map was derived from the land cover map completed by SW ReGap in 2005 (Lowry et al. 2005). The results should be interpreted with some caution, due to the various issues related to the purpose, spatial resolution, base imagery, and thematic resolution of these different mapping efforts.

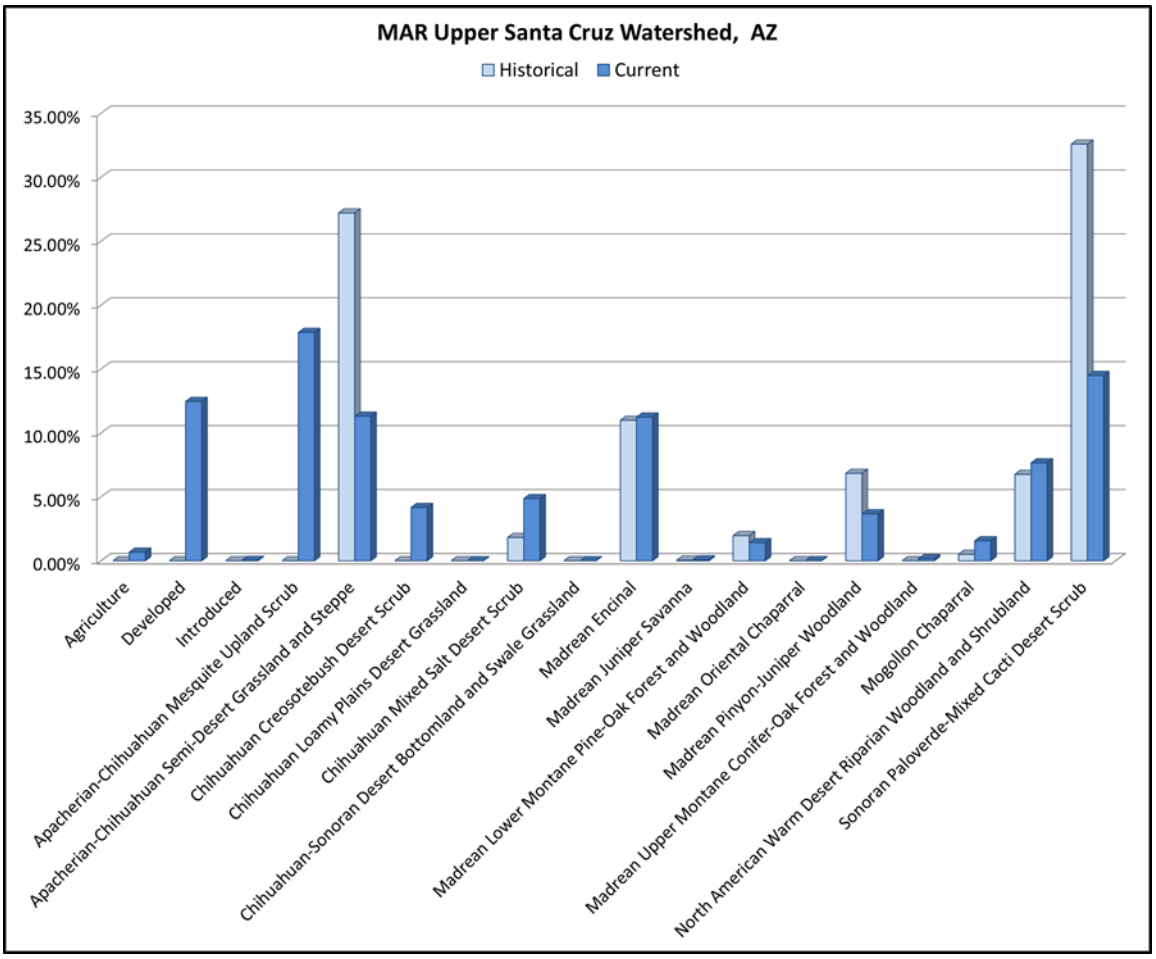
The results of the change in extent analysis from historical to current abundance of select terrestrial ecological systems for the ecoregion (Figure G-24) show obvious increases in post-European settlement impacts such as agriculture (to 1.3%), development (to 3%) and introduced vegetation (to 0.5%) at the expense of ecosystem types. However these changes are a relatively small percentage in extent when compared to the overall area of the ecoregion. Overall, the major changes in extent have occurred with large increases in Mesquite Upland Scrub (0% to 20%) and Chihuahuan Creosotebush Desert Scrub (0.5% to 13%) and large declines in Apacherian-Chihuahuan Semi-desert Grassland and Steppe (32% to 18%), Chihuahuan Loamy Plains Desert Grasslands (4% to nearly 0%), Chihuahuan Bottomland and Swale Grassland (2% to nearly 0%), and Sonoran Paloverde-Mixed Cacti Desert Scrub (21% to 11%). The desertification and conversion of semi-desert grasslands to desert scrub is well documented and has primarily occurred over the last century (McPherson and Weltzin 2000, Wilson et al. 2001). The decline in Sonoran Paloverde-Mixed Cacti Desert Scrub is largely due to urban expansion from Tuscon and surrounding areas and conversion to other desert scrub systems. There has also been a small but notable increase in Chihuahuan Mixed Salt Desert Scrub and substantial decreases in Madrean Encinal, Mogollon Chaparral and North American Warm Desert Riparian Woodland and Shrubland. The net change in other less abundant ecosystems such as Madrean Pinyon-Juniper Woodland appear relatively minor, but there may have been substantial changes locally.

**Figure G-24. Historical vs. Current Abundance of select terrestrial ecological systems for the entire MAR ecoregion.** The y-axis presents the percent of the MAR study area of the mapped historical or current extent of each ecological system or land cover type. Historical distribution was derived from the Landfire biophysical settings map and current distribution from the Natureserve (2013) terrestrial ecological systems map.



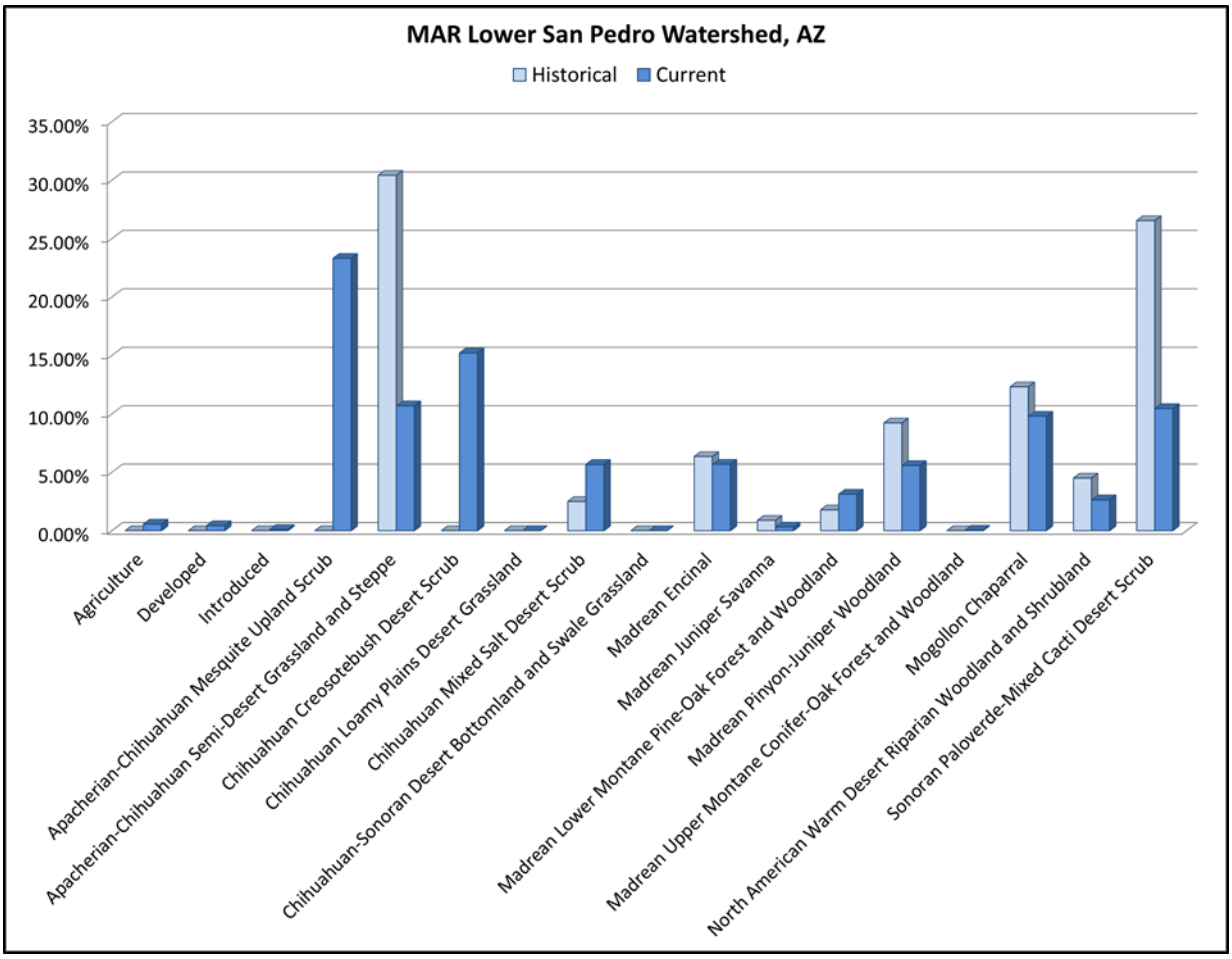
The results for the Upper Santa Cruz Watershed change in extent analysis from historical to current abundance of select terrestrial ecological systems are shown in Figure G-25. This is the watershed containing the urbanized Tucson area. Overall, the major changes in extent show very large increases in Apacherian-Chihuahuan Mesquite Upland Scrub (0 to 18%) and Development (0 to 12%). The major declines in Apacherian-Chihuahuan Semi-desert Grassland and Steppe (27% to 11%) and Sonoran Paloverde Mixed Cacti Desert Scrub (33% to 14%) are largely a result of conversion of semi-desert grassland to mesquite (*Prosopis* spp.) dominated desert scrub and loss of Sonoran Paloverde Mixed Cacti Desert Scrub. These losses are mainly due to development from the urban and suburban growth of Tucson and surrounding areas. Other notable changes include an increase in abundance of Chihuahuan Creosotebush Desert Scrub (0 to 4%) and Chihuahuan Mixed Salt Desert Scrub (2% to 5%) and a decline in Madrean Pinyon-Juniper Woodland extent (7% to 4%). Other small (~1%) changes in extent in minor ecological systems could be a substantial change locally.

**Figure G-25. Historical vs. Current abundance of select terrestrial ecological systems for the upper Santa Cruz watershed.** The y-axis presents the percent of the MAR study area of the mapped historical or current extent of each ecological system or land cover type in the watershed. Historical distribution was derived from the Landfire biophysical settings map and current distribution from the Natureserve (2013) terrestrial ecological systems map, which is based upon the land cover mapping of SW ReGAP (Lowry et al. 2005.



The results for the Lower San Pedro Watershed change in extent analysis from historical to current abundance of select terrestrial ecological systems are shown in Figure G-26. Overall, the major changes in extent show very large increases in extent of Apacherian-Chihuahuan Mesquite Upland Scrub (0 to 23%) and Chihuahuan Creosotebush Desert Scrub (0 to 15%) and large declines in Apacherian-Chihuahuan Semi-desert Grassland and Steppe (30% to 11%) and Sonoran Paloverde Mixed Cacti Desert Scrub (27% to 10%) largely as a result of conversion of of these systems to Apacherian-Chihuahuan Mesquite Upland Scrub and Chihuahuan Creosotebush Desert Scrub (0 to 15%). Other notable changes include an increase of Chihuahuan Mixed Salt Desert Scrub (2% to 6%) and a decline in Madrean Pinyon-Juniper Woodland extent (9% to 6%). Although small in overall extent, the decline from 4.5% to 2.6% of North American Warm Desert Riparian Woodland and Shrubland is very important in part because of the high importance of riparian systems in semi-arid regions. Other changes include ~1% changes in extent in minor ecological systems, which could be a substantial change locally.

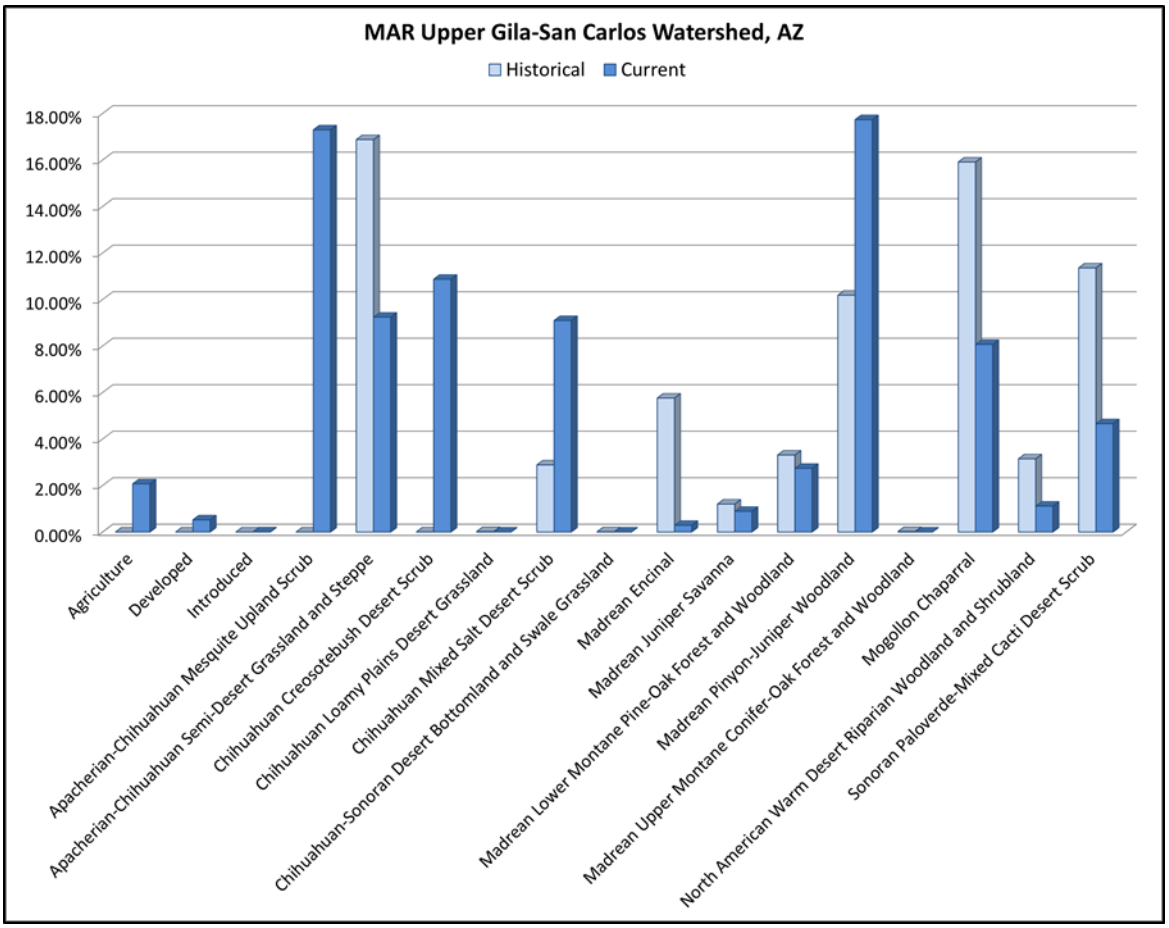
**Figure G-26. Historical vs. Current abundance of select terrestrial ecological sytems for the Lower San Pedro watershed.** The y-axis presents the percent of the MAR study area of the mapped historical or current extent of each ecological system or land cover type in the watershed. Historical distribution was derived from the Landfire biophysical settings map and current distribution from the Natureserve (2013) terrestrial ecological systems map, which is based upon the land cover mapping of SW ReGAP (Lowry et al. 2005).



The results for the Upper Gila-San Carlos Watershed change in extent analysis from historical to current abundance of select terrestrial ecological systems are shown in Figure G-27. Overall, the major changes in extent show large increases in extent of Mesquite Upland Scrub (0 to 17%), Creosote Desert Scrub (0 to 11%), Madrean Pinyon-Juniper Woodland (10% to 18%) and Chihuahuan Mixed Salt Desert Scrub (3% to 9%). These are offset by large declines in Apacherian-Chihuahuan Semi-desert Grassland and Steppe (17% to 9%), Mogollon Chaparral (16% to 8%), Sonoran Paloverde Mixed Cacti Desert Scrub (11% to 5%), and Madrean Encinal (6% to 0). Currently this watershed also has notable amounts of Agriculture (2%) and Development (1%).

The declines in Apacherian-Chihuahuan Semi-desert Grassland and Steppe are mainly a result of conversion of semi-desert grassland to mesquite (*Prosopis* spp.) dominated desert scrub. Although small in overall extent, the decline from 3% to 1 % in North American Warm Desert Riparian Woodland and Shrubland is very important in part because of the high importances of riparian systems in semi-arid regions. Some of this loss is likely because of the San Carlos Reservoir on the main stem of the Gila River and other water developments. The loss of Madrean Encinal and increase in Madrean Pinyon-Juniper Woodland may be because of invasion of pinyon and juniper trees due to to fire suppression.

**Figure G-27. Historical vs. Current abundance of select terrestrial ecological sytems for the Upper Gila-San Carlos watershed.** The y-axis presents the percent of the MAR study area of the mapped historical or current extent of each ecological system or land cover type in the watershed. Historical distribution was derived from the Landfire biophysical settings map and current distribution from the Natureserve (2013) terrestrial ecological systems map, which is based upon the land cover mapping of SW ReGAP (Lowry et al. 2005.

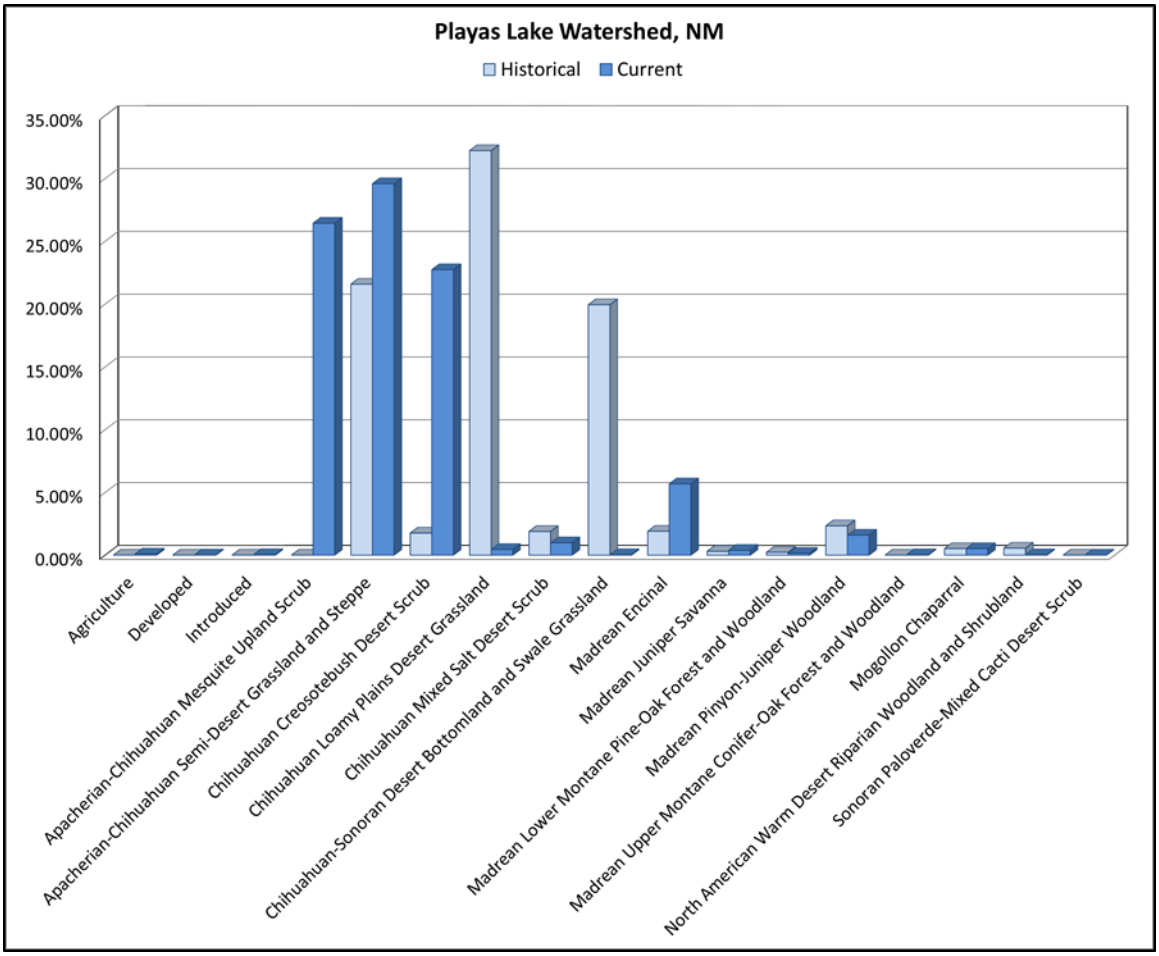



The results for the Playas Lake Watershed change in extent analysis from historical to current abundance of select terrestrial ecological systems within the MAR are shown in Figure G-28. This is not a watershed with significant areas of agriculture or development activities. Overall, the major changes in extent show very large increases in extent of Mesquite Upland Scrub (0 to 26%) and Creosotebush Desert Scrub (2 to 23%). These are offset by the loss of Chihuahuan Loamy Plains Desert Grassland (32% to 0.5%) and near complete loss of Chihuahuan-Sonoran Desert Bottomland and Swale Grassland (20% to 0). These tobosa (*Pleuraphis mutica*) dominated grasslands have been converted to desert scrub.

Other notable changes include an increase of Madrean Encinal (2% to 6%) and a small decline in Madrean Pinyon-Juniper Woodland extent (2.3% to 1.6%). Other changes include ~1% changes in extent in minor ecological systems which could be substantial changes locally. Uncharacteristically in comparison with the MAR in general, rather than a sharp decline in the abundance of Apacherian-Chihuahuan Semi-Desert Grassland and Steppe, it appears to have increased in abundance (22% to 30%)

in this watershed. The loss of North American Warm Desert Riparian Woodland and Shrubland (0.6% to 0) is notable because of the high importances of riparian systems in semi-arid regions.

**Figure G-28. Historical vs. Current abundance of select terrestrial ecological sytems for the Playas Lake, NM watershed.** The y-axis presents the percent of the MAR study area of the mapped historical or current extent of each ecological system or land cover type in the watershed. Historical distribution was derived from the Landfire biophysical settings map and current distribution from the Natureserve (2013) terrestrial ecological systems map, which is based upon the land cover mapping of SW ReGAP (Lowry et al. 2005).




Details for the change in extent for each of the mapped ecological systems and land cover types are provided in Table G-6; the raw data has been provided to BLM in an Excel workbook, with mapped extent in each 4th level watershed for each land cover or ecological system type.

As described above for Figure G-24, there have been substantial to major increases in development, agricultural activities, introduced vegetation (e.g. introduced exotic plants dominating areas), as well as Apacherian-Chihuahuan Mesquite Upland Scrub which is thought to have not occurred here in the past. Other desert scrub systems have increased in distribution, notably the Chihuahuan Mixed Salt Desert Scrub and the Chihuahuan Creosotebush Desert Scrub. Major losses have occurred for all of the grassland ecological systems, as well as for the Encinal, pinyon-juniper ecosystems, and the Sonoran desert scrub ecosystems. These changes can be characterized as loss of grasslands and savannas to

development and the expansion of mesquite and creosotebush into large areas not previously occupied by these types.

**Table G-6. Comparison of historical distribution (pre-European settlement) and current distribution (circa 2000) for upland vegetated ecological systems in the MAR ecoregion.** The top 5 rows are landcover types not found in the ecoregion historically, sorted by most to least abundant. The remaining rows are sorted by historical abundance. Decreases in abundance are shown with parentheses around the number, see e.g. the Apacherian-Chihuahuan Semi-Desert Grassland and Steppe.

| Ecosystem/Land Cover Name | Historical | | Current | | Change | |
|---|---|---|---|---|---|---|
| | Hectares | Acres | Hectares | Acres | Hectares | Acres |
| | | | | | | |
| Apacherian-Chihuahuan Mesquite Upland Scrub | 0 | 0 | 1,242,482 | 3,070,241 | 1,242,482 | 3,070,241 |
| Developed | 0 | 0 | 189,306 | 467,786 | 189,306 | 467,786 |
| Agriculture | 0 | 0 | 80,597 | 199,158 | 80,597 | 199,158 |
| Chihuahuan Stabilized Coppice Dune and Sand Flat Scrub | 0 | 0 | 46,619 | 115,198 | 46,619 | 115,198 |
| Introduced | 0 | 0 | 3,327 | 8,221 | 3,327 | 8,221 |
| | | | | | | |
| Apacherian-Chihuahuan Semi-Desert Grassland and Steppe | 2,061,173 | 5,093,262 | 1,158,114 | 2,861,762 | (903,059) | (2,231,500) |
| Sonoran Paloverde-Mixed Cacti Desert Scrub | 1,323,224 | 3,269,752 | 677,741 | 1,674,735 | (645,483) | (1,595,018) |
| Mogollon Chaparral | 432,414 | 1,068,516 | 303,594 | 750,196 | (128,820) | (318,320) |
| Madrean Pinyon-Juniper Woodland | 397,008 | 981,028 | 369,089 | 912,039 | (27,919) | (68,989) |
| Madrean Encinal | 354,241 | 875,346 | 322,799 | 797,654 | (31,441) | (77,692) |
| Chihuahuan Loamy Plains Desert Grassland | 234,573 | 579,641 | 2,114 | 5,224 | (232,458) | (574,416) |
| Chihuahuan Mixed Salt Desert Scrub | 154,912 | 382,795 | 274,821 | 679,099 | 119,910 | 296,304 |
| Sonora-Mojave Creosotebush-White Bursage Desert Scrub | 154,375 | 381,469 | 143,296 | 354,092 | (11,079) | (27,377) |
| Chihuahuan-Sonoran Desert Bottomland and Swale Grassland | 135,954 | 335,949 | 740 | 1,829 | (135,214) | (334,120) |
| Madrean Lower Montane Pine-Oak Forest and Woodland | 117,699 | 290,840 | 95,118 | 235,042 | (22,581) | (55,799) |
| Chihuahuan Mixed Desert and Thorn Scrub | 112,844 | 278,843 | 51,668 | 127,674 | (61,176) | (151,168) |
| Sonoran Mid-Elevation Desert Scrub | 94,020 | 232,328 | 79,974 | 197,620 | (14,046) | (34,708) |
| Inter-Mountain Basins Semi-Desert Shrub-Steppe | 34,467 | 85,170 | 400 | 989 | (34,067) | (84,181) |
| Madrean Juniper Savanna | 26,960 | 66,619 | 18,323 | 45,277 | (8,637) | (21,342) |

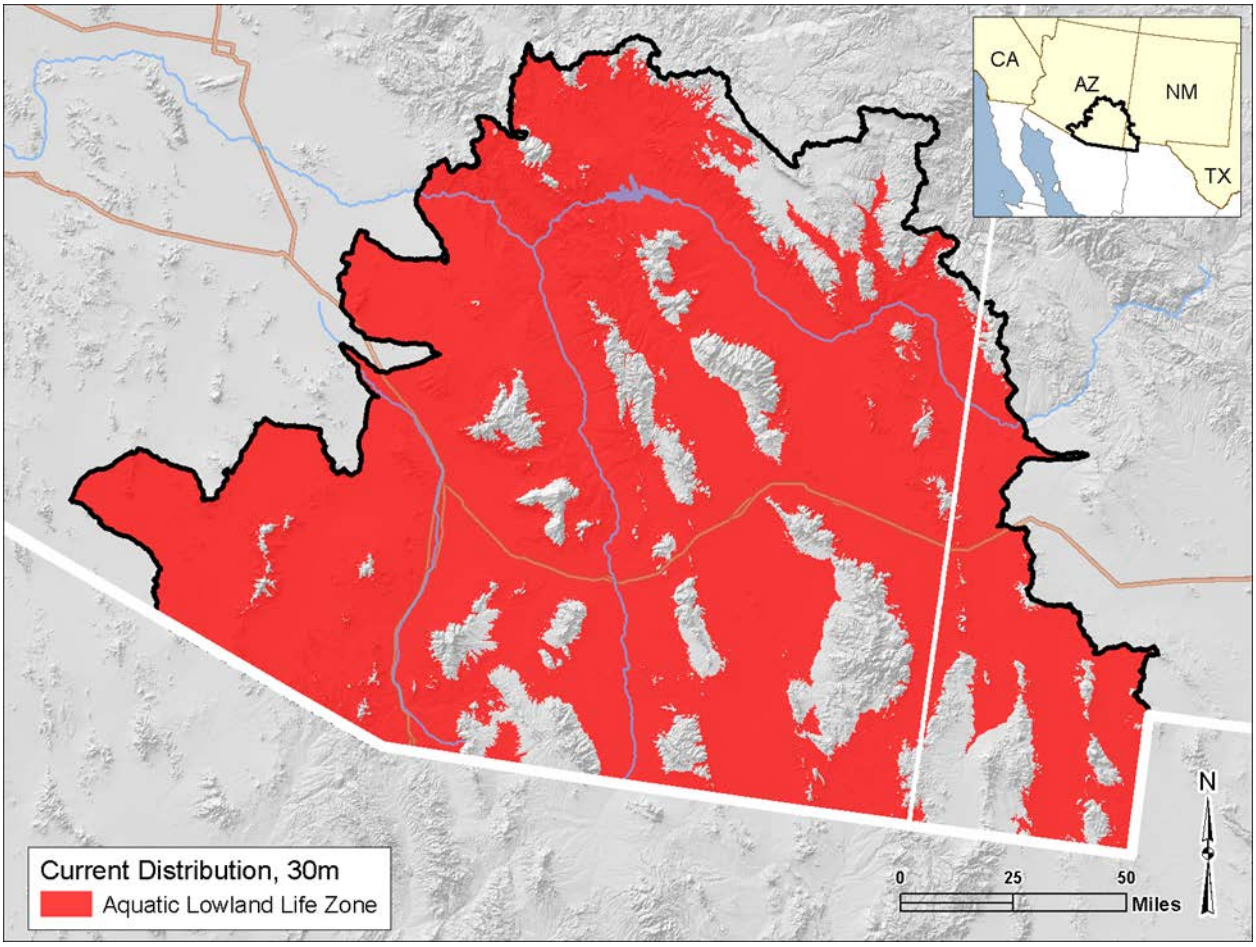
| Ecosystem/Land Cover Name | Historical | | Current | | Change | |
|---|---|---|---|---|---|---|
| | Hectares | Acres | Hectares | Acres | Hectares | Acres |
| Chihuahuan Succulent Desert Scrub | 23,469 | 57,993 | 10,239 | 25,301 | (13,230) | (32,692) |
| Southern Rocky Mountain Ponderosa Pine Woodland | 22,626 | 55,909 | 67,363 | 166,457 | 44,737 | 110,548 |
| Colorado Plateau Pinyon-Juniper Woodland | 22,166 | 54,773 | 10,329 | 25,524 | (11,837) | (29,249) |
| Sonora-Mojave Mixed Salt Desert Scrub | 17,787 | 43,953 | 2,765 | 6,831 | (15,023) | (37,122) |
| Chihuahuan Creosotebush Desert Scrub | 14,464 | 35,741 | 841,660 | 2,079,787 | 827,196 | 2,044,046 |
| Inter-Mountain Basins Semi-Desert Grassland | 12,292 | 30,374 | 289 | 715 | (12,003) | (29,659) |
| Chihuahuan Sandy Plains Semi-Desert Grassland | 10,010 | 24,736 | 3,337 | 8,246 | (6,674) | (16,491) |
| Southern Rocky Mountain Mesic Montane Mixed Conifer Forest and Woodland | 5,754 | 14,219 | 692 | 1,711 | (5,062) | (12,509) |
| Southern Rocky Mountain Dry-Mesic Montane Mixed Conifer Forest and Woodland | 5,676 | 14,026 | 1,066 | 2,634 | (4,610) | (11,392) |
| Inter-Mountain Basins Juniper Savanna | 5,012 | 12,384 | 40 | 99 | (4,972) | (12,285) |
| Madrean Oriental Chaparral | 3,764 | 9,301 | 257 | 636 | (3,507) | (8,665) |
| Southern Rocky Mountain Ponderosa Pine Savanna | 2,782 | 6,875 | 9,363 | 23,136 | 6,581 | 16,262 |
| Rocky Mountain Subalpine Dry-Mesic Spruce-Fir Forest and Woodland | 721 | 1,782 | 4 | 9 | (717) | (1,773) |
| Southern Rocky Mountain Pinyon-Juniper Woodland | 561 | 1,386 | 176 | 436 | (384) | (950) |
| Southern Rocky Mountain Montane-Subalpine Grassland | 551 | 1,362 | 3 | 8 | (548) | (1,354) |
| Madrean Upper Montane Conifer-Oak Forest and Woodland | 266 | 657 | 12,653 | 31,266 | 12,387 | 30,609 |
| Rocky Mountain Gambel Oak-Mixed Montane Shrubland | 253 | 626 | 188 | 465 | (65) | (161) |
| Rocky Mountain Subalpine-Montane Riparian Shrubland | 202 | 499 | 270 | 668 | 68 | 169 |
| Rocky Mountain Aspen Forest and Woodland | 4 | 9 | 2,843 | 7,026 | 2,840 | 7,017 |

## G.5.2 Ecological Integrity for the Aquatic and Wetland Resources of the Ecoregion

### G.5.2.1 Aquatic Lowland Life Zone

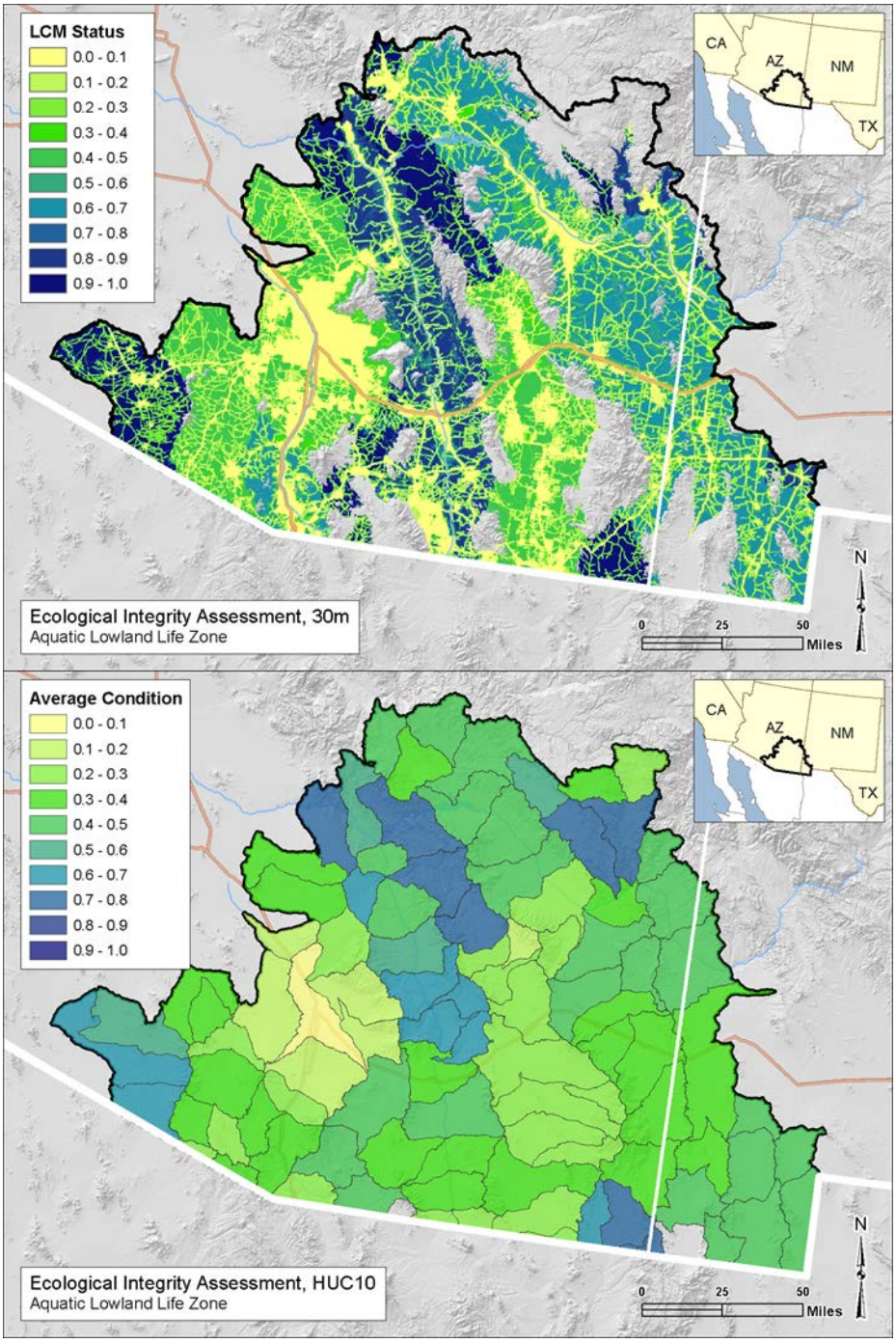
The distribution of the Aquatic Lowland life zone (Figure G-29) illustrates how much of the MAR ecoregion is at lower elevations (below 1,524 m or 5,000 ft). The ecological integrity scores for the Aquatic Lowland Life Zone are reported by by 30m pixel (upper panel) and 5$^{th}$-level watersheds (lower panel, Figure G-30) and take into account the stressor-based indicators for landscape condition-development, aquatic and terrestrial invasive species, and water use. The results show scores of 0.6 or less for most of the watersheds; the majority of this lifezone is in moderate to poor condition.

**Figure G-29. Distribution of Aquatic Lowland Life Zone; all areas of the MAR below 5000 feet (1524 m).** The distribution (in red) shows portions of all 5$^{th}$ level watersheds below 1524 m (5,000 ft) in elevation, and is intended to represent areas that contain any aquatic resource, such as rivers, riparian areas, ciénegas, washes, marshes in this life zone.



Areas surrounding and waters within the Gila River downstream from the San Simon River confluence, most of the San Pedro River and most of the Santa Cruz River south of Tucson show high levels of impact from development, water use, and invasive species. The most altered watersheds are located in the areas of of Safford, Willcox, and the Tucson metropolis, AZ. The least altered watersheds occur in the far west-southwestern corner of the ecoregion west and south of Sells, AZ; in the northern third of the lower San Pedro River basin; in the lower San Francisco River basin; and surrounding San Bernardino National Wildlife Refuge.

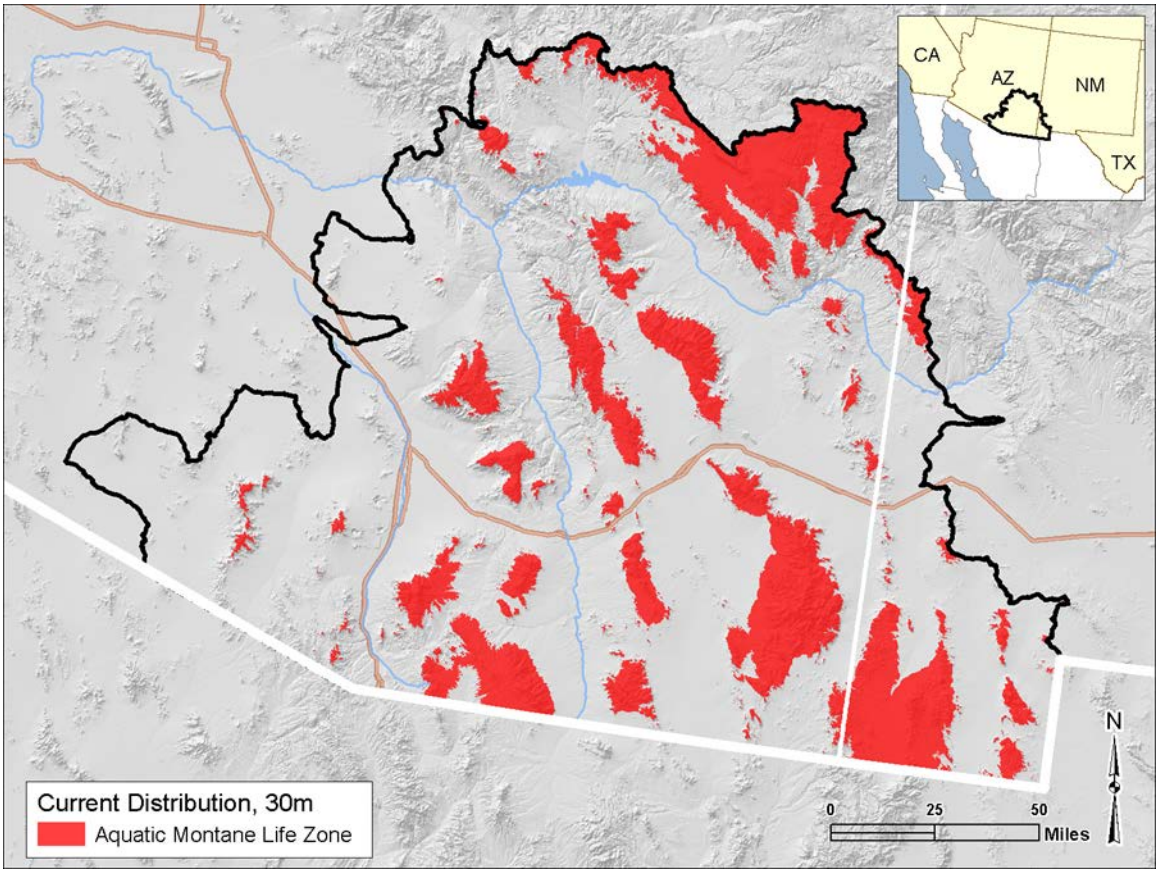
**Figure G-30. Ecological integrity scores for Aquatic Lowland Life Zone** at 30 m pixel scale (upper) and averaged to watersheds (lower). The maps are for the three indicators: landscape condition-development, aquatic and terrestrial invasive species, and water use, which were combined into a single ecological integrity score for each 30 m pixel (upper) or 5th level watershed (lower). Yellow scores (equivalent to 0) indicate high impacts from the CAs, dark blue (equivalent to 1) indicate little to no impact from the CAs.



#### G.5.2.2 Aquatic Montane Life Zone

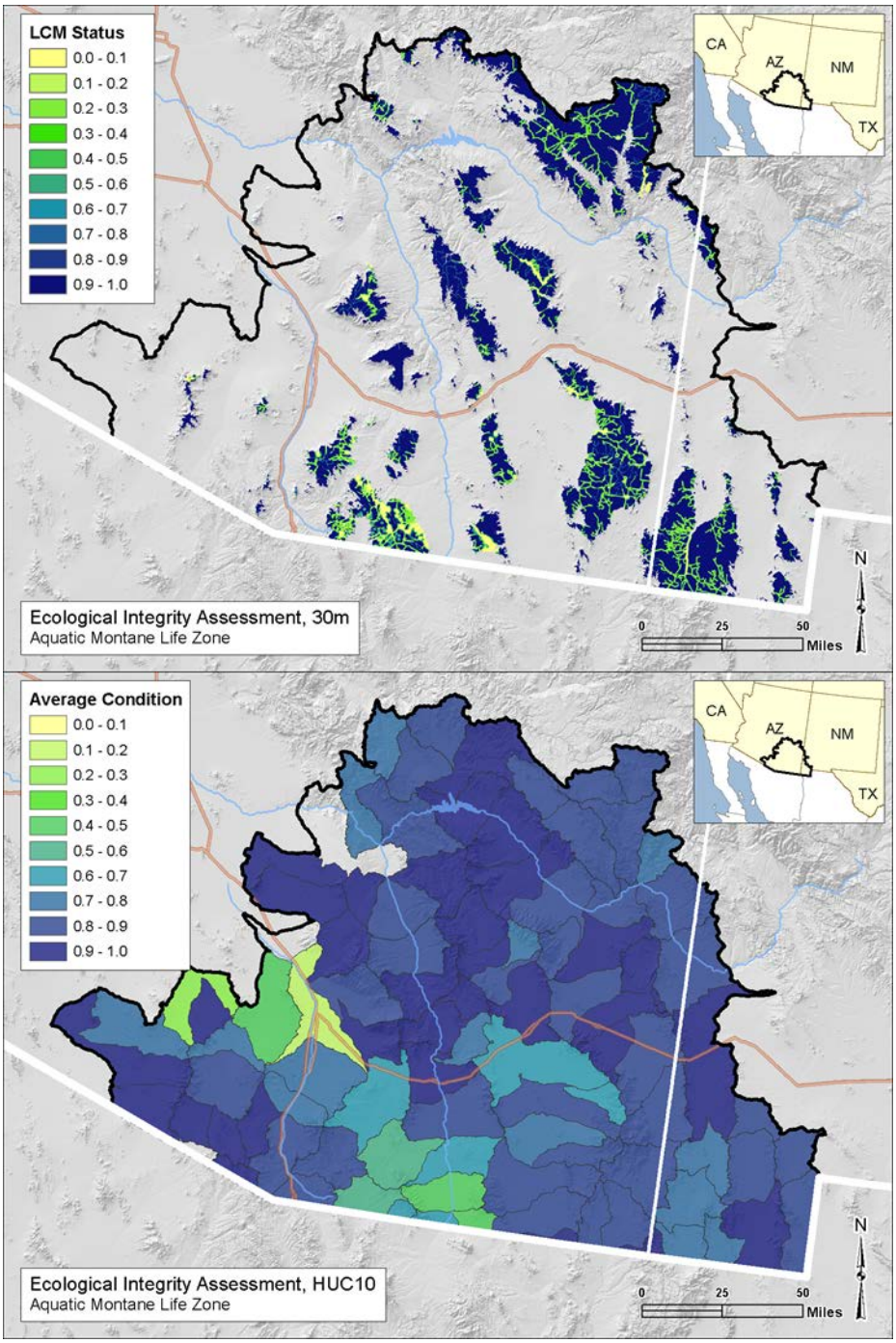
The distribution of the Aquatic Montane life zone (Figure G-31) illustrates the areas of the MAR ecoregion at higher elevations (above 1524 m or 5,000 ft). The ecological integrity scores for the Aquatic Montane Life Zone are reported by the 5th-level watersheds (Figure G-32) and take into account the stressor-based indicators for landscape condition-development, and aquatic and terrestrial invasive species; the water use indicator was not assessed. While large areas of development are not found in this life zone, there are roads, mining and associated dams, and other water manipulation structures that play a role in the degradation of these habitats, along with invasive species. The two indicators used (Landscape Condition-Development) and invasive species (terrestrial plants, aquatic animals and aquatic plants), when combined, show that the life zone integrity varies significantly across the ecoregion. This assessment illustrates that development is one of the most pervasive driving factors in the varying condition of the Aquatic Montane Life Zone across the ecoregion.

**Figure G-31. Distribution of the Aquatic Montane Life Zone in the Madrean Archipelago ecoregional assessment area.** The distribution (in red) shows portions of all 5th level watersheds above, 1524 m (5,000 ft) in elevation, and is intended to represent areas that contain any aquatic resource, such as such as headwater streams and tributaries, riparian areas, ciénegas, washes, or marshes in this life zone.




The southern central portion of the ecoregion has mining, transportation corridors, and towns (e.g. Warren, Bakerville, and Bisbee junction) that occur at similar elevations as this life zone. Compared to the lower elevation aquatic life zone (Figure G-30), the Aquatic Montane Life Zone is faring much better overall in the MAR, with more watersheds having better overall ecological integrity. These areas are not impacted by significant groundwater withdrawal and surface water diversions, nor are they as heavily exposed to development.

**Figure G-32. Ecological integrity scores for Aquatic Montane Life Zone** at 30 m pixel scale (upper) and averaged to watersheds (lower). The maps are for the two indicators: landscape condition-development, and aquatic and terrestrial invasive species, which were combined into a single ecological integrity score for each 30 m pixel (upper) or 5th level watershed (lower). Yellow scores (equivalent to 0) indicate high impacts from the CAs, dark blue (equivalent to 1) indicate little to no impact from the CAs.



## G.6 References


ADWR [Arizona Department of Water Resources]. 2010a. Arizona Water Atlas, Volume 3: Southeastern Arizona Planning Area. Arizona Department of Water Resources, Phoenix, AZ. (Available on-line at http://www.azwater.gov/azdwr/statewideplanning/wateratlas/.)

ADWR [Arizona Department of Water Resources]. 2010b. Arizona Water Atlas, Volume 8: Active Management Area Planning Area. Arizona Department of Water Resources, Phoenix, AZ. (Available on-line at http://www.azwater.gov/azdwr/statewideplanning/wateratlas/.)

Aguiar, M.R. and O.E. Sala. 1999. Patch structure, dynamics, and implications for the functioning of arid ecosystems. Trends in Ecology and Evolution 14:273–277.

Alderson, J., and W. C. Sharp. 1993. Grass varieties in the United States. United States Department of Agriculture, Soil Conservation Service, Agricultural Handbook No. 170, Washington, D.C. Website: http://www.forages.css.orst.edu/Topics/Species/Grasses/Grass_Varieties/

Alvarez, T., and L. Gonzalez-Q. 1970. Análisis polínico del contenido gástrico de murciélagos Glossophaginae de México. Anales del Escuela Nacional Ciencias Biologia, México 18:137-165.

Anable, M. E., M. P. McClaran, and G. B. Ruhle. 1992. Spread of introduced Lehmann lovegrass Eragrostis lehmanniana Nees in Southern Arizona, USA. Biological Conservation 61:181-188.

Anderson, M., P. Bourgeron, M. T. Bryer, R. Crawford, L. Engelking, D. Faber-Langendoen, M. Gallyoun, K. Goodin, D. H. Grossman, S. Landaal, K. Metzler, K. D. Patterson, M. Pyne, M. Reid, L. Sneddon, and A. S. Weakley. 1998. International classification of ecological communities: terrestrial vegetation of the United States. Volume II. The National Vegetation Classification System: list of types. The Nature Conservancy, Arlington, Virginia, USA.

Anderson, R. S., and T. R. Van Devender. 1991. Comparison of pollen and macrofossils in packrat (Neotoma) middens: a chronological sequence from the Waterman Mountains, southern Arizona, U.S.A. Review of Paleobotany and Palynology 68:1-28.

Archer, S., and K. Predick. 2008. Climate Change and Ecosystems of the Southwestern United States, Society for Rangeland Management, Rangelands 30(3):23-28, June 2008, doi: 10.2111/1551-501X(2008)30[23:CCAEOT]2.0.CO;2.

Arndt, D. S., M. O. Baringer, and M. R. Johnson, Eds., (2010), State of the Climate in 2009, Bull. Amer. Meteor. Soc., 91 (7), S1–S224.

Arnold, T. W. and R. M. Zink. 2011. Collision mortality has no discernible effect on population trends of North American birds. PLoS ONE 6(9): e24708. doi:10.1371/journal.pone.0024708

Atwood T.C., J.K. Young, J.P. Beckmann, S.W. Breck, J. Fike, O.E. Rhodes, and K.D. Bristow. 2011. Modeling connectivity of black bears in a desert sky island archipelago. Biological Conservation 144:12, 2851-2862

AZ CCAG [Arizona Climate Change Advisory Group]. 2006. Climate Change Action Plan, Arizona Department of Environmental Quality. http://www.azclimatechange.gov/download/O40F9347. \

AZ FRTF [Arizona Forest Resources Task Force]. 2010. Arizona Forest Resource Assessment, A collaborative analysis of forest related conditions, trend, threats, and opportunities, prepared for Arizona State Forestry Division and U.S. Forest Service, June 18, 2010.

AZGFD [Arizona Game and Fish Department]. 2006. Choeronycteris mexicana. Unpublished abstract compiled and edited by the Heritage Data Management System, Arizona Game and Fish Department, Phoenix, AZ. 7 pp.

AZGFD [Arizona Game and Fish Department]. 2011. Leptonycteris curasoae yerbabuenae. Unpublished abstract compiled and edited by the Heritage Data Management System, Arizona Game and Fish Department, Phoenix, AZ. 8 pp.

AZGFD [Arizona Game and Fish Department]. 2012. Arizona's State Wildlife Action Plan: 2012-2022. Arizona Game and Fish Department, Phoenix, Arizona.

AZGS [Arizona Geological Survey]. 2013. Mineral Resources: Mining in Arizona. http://www.azgs.az.gov/minerals_mining.shtml (accessed June 4, 2013)

Bahre, C. J. 1985. Wildfire in southeastern Arizona between 1859 and 1890. Desert Plants 7:190-194.

Bahre, C. J. 1991. A Legacy of Change: Human Impacts on Vegetation in the Arizona Borderlands. University of Arizona Press, Tucson, AZ.

Bahre, C. J. and C. F. Hutchinson. 2001. Historic vegetation change in La Frontera west of the Rio Grande. Pp. 67-83 in G. L. Webster and C. J. Bahre (eds.). 2001, Vegetation and Flora of La Frontera: Vegetation Change along the United States-Mexican Boundary.

Bailowitz, R.A. and J.P. Brock 1991. Butterflies of southeastern Arizona. Sonoran Arthropod Studies, Inc. Tucson, AZ.

Barlow, P. M. and S. A. Leake. 2012. Streamflow depletion by wells: Understanding and managing the effects of groundwater pumping on streamflow. U.S. Geological Survey Circular 1376, 84 pp. Available at http://pubs.usgs.gov/circ/1376/

Barnett, T., D. W. Pierce, H. G. Hidalgo, C. Bonfils, B D. Santer, T. Das, G. Bala, A. W. Wood, T. Nozawa, A. A. Mirin, D. R. Cayan, M. D. Dettinger. 2008. Human-induced changes in the hydrology of the western United States. Science 319: 1080–1083.

Barton, A.M. 1999. Pines versus oaks: effects of fire on the composition of Madrean forests in Arizona. Forest Ecology and Management 120: 143–156.

Beatley, J. C. 1966. Ecological status of introduced brome grasses *(Bromus* spp.) in desert vegetation of southern Nevada. Ecology. 4 7: 548-554

Bentz, B.J., Régnière J, Fettig CJ, Hansen EM, Hayes, JL, Hicke JA, Kelsey RG, Lundquist J, Negrón J, Progar R, Seybold SJ. Vandygriff JC. 2008. Climate Change and Western US Bark Beetles: Rapid Threat Assessment. Prepared for the Western Wildland Environmental Threat Assessment Center; http://www.fs.fed.us/wwetac/projects/PDFs/RTA_Bark_Beetle.pdf

Bentz, B.J., Régnière, J., Fettig, C.J., Hansen, E.M., Hayes, J.L., Hicke, J.A., Kelsey, R.G., Negrón, J.F., Seybold, S.J., 2010. Climate change and bark beetles of the western United States and Canada: direct and indirect effects. BioScience 60, 602–613.

Beschta, R. L., Donahue, D. L., DellaSala, D. A., Rhodes, J. J., Karr, J. R., O'Brien, M. H., Fleischner, T. L., and Deacon-Williams, C. 2012. Adapting to Climate Change on Western Public Lands: Addressing the Ecological Effects of Domestic, Wild, and Feral Ungulates. Environmental Management. DOI 10.1007/s00267-012-9964-9

Bock, C.E., J.H. Bock, W.R. Kenney, and V.M. Hawthorne. 1984. Responses of birds, rodents, and vegetation to livestock exclosure in a semi-desert grassland site. Journal of Range Management 37(3): 239 – 242.

Bock, C.E., Z. F. Jones, and J. H. Bock. 2008. The oasis effect: response of birds to exurban development in a southwestern savanna. Ecological Applications 18:1093-1106.

Bodner, G.S., J. Atchley Montoya, R. Hanson, and W. Anderson, Editors. 2006. Natural heritage of the Peloncillo Mountain Region: a synthesis of science. World Wildlife Fund and Sky Island Alliance, Tucson, AZ. Available at www.skyislandalliance.org/media/Peloncillo%20Report.pdf

BOR [Bureau of Reclamation]. 2009. Native Aquatic Species of the Gila River Basin in Arizona and New Mexico: Nonnative species impacts. Accessed at http://www.usbr.gov/lc/phoenix/biology/azfish/exoticspecies.html. Accessed 11/5/2013

Bowers, J. E., and S. P. McLaughlin. 1996. Flora of the Huachuca Mountains, a botanically rich and historically significant sky island in Cochise County, Arizona. Journal of the Arizona Nevada Academy of Science 29: 66-107.

Bradley, B. A., M. Oppenheimer, and D. S. Wilcove. 2009. Climate change and plant invasions: restoration opportunities ahead? Global Change Biology 15(6): 1511–1521.

Brennan, T. C. and A. T. Holycross. 2006. Amphibians and Reptiles in Arizona. Arizona Game and Fish Department. Phoenix, Arizona. 150 pp.

Brenner, J.C. and L.L. Kanda. Buffelgrass (Pennisetum ciliare) Invades Lands Surrounding Cultivated Pastures in Sonora, Mexico. Invasive Plant Science and Management. 6(1): 187-195. doi: http://dx.doi.org/10.1614/IPSM-D-12-00047.1

Breshears, D. D., N. S. Cobb, P. M. Rich, K. P. Price, C. D. Allen, R. G. Balice, W. H. Romme, J. H. Kastens , M. L. Floyd, J. Belnap , J. J. Anderson, O. B. Myers, and C. W. Meyer. 2005. Regional vegetation die-off in response to global-change-type drought. Proceedings of the National Academy of Sciences 102: 15144-15148.

Brown, D. E., editor. 1982. Biotic communities of the American Southwest-United States and Mexico. Desert Plants Special Issue 4(1-4):1-342.

Brown, J. R. and S. Archer. 1999. Shrub invasion of grassland: Recruitment is continuous and not regulated by herbaceous biomass or density. *Ecology*, 80, 2386-2396.

Brown, D.E. and R.A. Minnich. 1986. Fire and changes in creosote bush scrub of the western Sonoran Desert, California. The American Midland Naturalist. 116(2): 411-422.

Brusca, R. C., J. F. Wiens, W. M. Meyer, J. Eble, K Franklin, J. T. Overpeck, and W. Moore. 2013. Dramatic response to climate change in the Southwest: Robert Whittaker's 1963 Arizona Mountain plant transect revisited. Ecology and Evolution 3(10): 3307–3319.

Buffington, L. C. and C. H. Herbel. 1965. Vegetational changes on a semidesert grassland range from 1858 to 1963. Ecological Monographs 35(2):139-164.

Burgess, T.L. 1995. Desert grassland, mixed shrub savanna, shrub steppe, or semidesert scrub? The dilemma of coexisting growth forms. In M. P. McClaran and T. R. Van Devender (eds.), The Desert Grassland, pp. 31-67. University of Arizona Press, Tucson.

Cable, Dwight R. 1971. Lehmann lovegrass on the Santa Rita Experimental Range, 1937-1968. Journal of Range Management. 24: 17-21.

Calamusso, B. 2005. Fishes of Southwestern Grasslands-Ecology, Conservation, and Management. In Finch, Deborah M., Editor, Assessment of grassland ecosystem conditions in the Southwestern United States, Volume 2: Wildlife and Fish, pp. 141-168. Rocky Mountain Forest and Range Experiment Station, General Technical Report RMRS-GTR-135-vol. 2. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Callegary, J., J. Langman, J. Leenhouts, and P. Martin. Challenge Theme 2: Assuring Water Availability and Quality in the 21st Century. In Updike, R. G., E. G. Ellis, W. R. Page, M. J. Parker, J. B. Hestbeck, and W. F. Horak (eds). 2013. United States-Mexico Borderlands: Facing tomorrow's challenges through USGS science. U.S. Geological Survey Circular 1380, 336 p., 1 pl.

Carmichael, R. S., O. D. Knipe, C. P. Pase, and W. W. Brady. 1978. Arizona chaparral: Plant associations and ecology. Research Paper RM-202. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO. 16 pp.

Chapin, F., III, P.A. Matson, and H.A. Mooney. 2002. Principles of terrestrial ecosystem ecology. Springer, NY.

Christensen, J.H., B. Hewitson, A. Busuioc, A. Chen, X. Gao, I. Held, R. Jones, R.K. Kolli, W.-T. Kwon, R. Laprise, V. Magaña Rueda, L. Mearns, C.G. Menéndez, J. Räisänen, A. Rinke, A. Sarr and P. Whetton. 2007. Regional Climate Projections, In: Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor and H.L. Miller (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA. http://www.ipcc.ch/publications_and_data/ar4/wg1/en/ch11.html

Coblentz, D.D. and K.H. Riitters. 2004. Topographic controls on the regional-scale biodiversity of the south-western USA. Journal of Biogeography 31: 1125–1138.

Coffin, A. W. 2007. From roadkill to road ecology: a review of the ecological effects of roads. Journal of Transport Geography 15: 396-406.

Comer, P., D. Faber-Langendoen, R. Evans, S. Gawler, C. Josse, G. Kittel, S. Menard, M. Pyne, M. Reid, K. Schulz, K. Snow, and J. Teague. 2003. Ecological Systems of the United States: A Working Classification of U.S. Terrestrial Systems. NatureServe, Arlington, Virginia

Comer, P., P. Crist, M. Reid, J. Hak, H. Hamilton, D. Braun, G. Kittel, I. Varley, B. Unnasch, S. Auer, M. Creutzburg, D. Theobald, and L. Kutner. 2012a. Central Basin and Range Rapid Ecoregional Assessment Report. Prepared for the U.S. Department of the Interior, Bureau of Land Management, Denver, CO.

Comer, P., P. Crist, M. Reid, J. Hak, H. Hamilton, D. Braun, G. Kittel, I. Varley, B. Unnasch, S. Auer, M. Creutzburg, D. Theobald, and L. Kutner. 2012b. Mojave Basin and Range Rapid Ecoregional Assessment Report. Prepared for the U.S. Department of the Interior, Bureau of Land Management, Denver, CO.

[CEC] Commission for Environmental Cooperation. 1997. Ecological regions of North America: toward a common perspective. Commission for Environmental Cooperation, Montreal, Quebec, Canada. 71pp. Map (scale 1:12,500,000).

Community by Design. 2011. Hidalgo County Comprehensive Plan Update: Hidalgo County, New Mexico. Albuquerque, NM.

Corman, T. E. and C. Wise-Gervais. Arizona Breeding Bird Atlas. 2005. University of New Mexico Press. Albuquerque, New Mexico. 636 pp.

Coronado National Forest. 2009. Coronado National Forest Ecological Sustainability Report. United States Department of Agriculture Forest Service. 118 pp.

Coronado National Forest. 2013. Schedule of Proposed Actions (SOPA). http://data.ecosystem-management.org/nepaweb/current-sopa.php?forest=110305#9 (accessed June 4, 2013).

Cox, J. R., and G. B. Ruyle. 1986. Influence of climatic and edaphic factors on the distribution of Eragrostis lehmanniana Nees in Arizona, USA. Journal of the Grassland Society of South Africa. 3: 25-29.

Cox, J. R., G. B. Ruyle, J. H. Fourle, and C. Donaldson. 1988. Lehmann lovegrass--central South Africa and Arizona, USA. Rangelands. 10(2): 53-55.

Cox, J. R., H. L. Morton, T. N. Johnsen, Jr., G. L. Jordan, S. C. Martin, and L. C. Fierro. 1984. Vegetation restoration in the Chihuahuan and Sonoran Deserts of North America. Rangelands 6:112-115.

Cox, J. R., M. H. Martin-R., F. A. Ibarra-F., J. H. Fourie, N. F. Rethman, and D. G. Wilcox. 1988. The influence of climate and soils on the distribution of four African grasses. Journal of Range Management 41:127-139.

Crider, F. J. 1945. Three introduced lovegrasses for soil conservation. Circular 730. U.S. Department of Agriculture, U.S. Printing Office, Washington, D.C.

Crist, P., M. Harkness, T. Robertson, and M. Reid. 2013. Madrean Archipelago Rapid Ecoregional Assessment: Pre-Assessment Work Plan. Prepared for the U.S. Department of the Interior, Bureau of Land Management. NatureServe, Arlington, VA. 34 pages + appendices. Available on-line: http://www.blm.gov/pgdata/etc/medialib/blm/wo/Communications_Directorate/public_affairs/landscape_approach/documents1.Par.26493.File.dat/MAR-1_Pre-assessment_Work_Plan.pdf.

Curtin, C.G. 2002. Livestock Grazing, Rest, and Restoration in Arid Landscapes. Conservation Biology 16(3): 840-842.

Dahl, B. E., and P. F. Cotter. 1984. Management of weeping lovegrass in west Texas. Management Note 5. Texas Tech University, College of Agricultural Sciences, Department of Range and Wildlife Management, Lubbock, Texas.

Dalrymple, R. L. 1970. Weeping Lovegrass establishment and management of first year stands. In : Dalrymple, R. L. (ed./compiler). Proceedings of the First Weeping Lovegrass Symposium (April 28-29, 1970). The Samuel Roberts Noble Foundation, Ardmore, Oklahoma. p. 21-27. Website: http://www.noble.org/ag/Forage/LovegrassSymposium.pdf

DeBano, L. F. 1999. Chaparral shrublands in the southwestern United States. Chapter 7. Pages 83-94 in: P. F. Ffolliott and A. Ortega-Rubio, editors. Ecology and Management of Forests, Woodlands, and Shrublands in Dryland Regions of the United States and Mexico: Perspectives for the 21st Century. Co-edition number 1. University of Arizona-Centro de Investigacione.

DeBano, L.F., G.J. Gottfried, R.H. Hamre, C.B. Edminster, P.F. Ffolliott, and A. Ortega-Rubio, editors. 1995. Biodiversity and management of the Sky Island archipelago. U.S. Department of Agriculture Forest Service, Rocky Mountain Range and Forest Experiment Station, General Technical Report RM-GTR-264. Fort Collins, CO.

De Leo, G. A., and S. Levin. 1997. The multifaceted aspects of ecosystem integrity. Conservation Ecology [online]1(1): 3. Available from the Internet. URL: http://www.consecol.org/vol1/iss1/art3/

DeLoach, C.J., R.I. Carruthers. 2004: Biological control programs for integrated invasive plant management. In: Proceedings of Weed Science Society of America Meeting, Kansas City, MO. Weed Science Society of America (CD-ROM). 17 pp.

DeLoach, C.J., R.I. Carruthers, J.E. Lovich, T.L. Dudley, and S.D. Smith. 2000: Ecological interactions in the biological control of saltcedar (Tamarix spp.) in the United States: toward a new understanding. In N. R. Spencer (ed.), Proceedings of the X International Symposium on Biological Control of Weeds, 4–14 July 1999, Montana State University. Bozeman, Montana, pp. 819–873.

Diffenbaugh, N. and M. Ashfaq. 2010. Intensification of hot extremes in the United States. Geophysical Research Letters 37(15): L15701. doi: 10.1029/2010GL043888.

Dimmitt, M. A. 2000. Biomes and communities of the Sonoran Desert region. Pages 3-18 in S. J. Phillips and P. W. Comus, editors. A natural history of the Sonoran Desert. Arizona-Sonora Desert Museum Press, Tucson, AZ.

Dimmitt, M. A., and T. R. Van Devender. 2009. Sahara mustard (Brassica tournefortii): A new, serious threat to low desert ecosystems in the southwestern United States and northwestern Mexico. Pg. 241-246 in T. R. Van Devender, F. J. Espinosa-García, B. L. Harper-Lore, and T. Hubbard (eds.), Invasive Plants on the Move. Controlling them in North America. Proceedings of Weeds Across Borders 2006 Conference, Hermosillo, Sonora, May 25-28, 2006; Tucson, AZ.

Dinerstein, E., D. Olson, J. Atchley, C. Loucks, S. Contreras-Balderas, R. Abell, E. Inigo, E. Enkerlin, C. Williams, and G. Castilleja. 2000. Ecoregion-based conservation in the Chihuahuan Desert-a biological assessment. World Wildlife Fund, Comisión Nacional para el Conocimiento y Uso de la Biodiversidad (CONABIO), The Nature Conservancy, PRONATURA Noreste, and Instituto Tecnológico y de Estudios Superiores de Monterrey (ITESM).

Dittberner, P. L., and M. R. Olson. 1983. The plant information network (PIN) database: Colorado, Montana, North Dakota, Utah, and Wyoming. FWS/OBS-83/86. U.S. Department of the Interior, Fish and Wildlife Service, Washington, D.C.

Dixon, M. D., J. C. Stromberg, J. T. Price, H. Galbraith, A. K. Fremier, and E. W. Larsen. 2009. Potential Effects of Climate Change on the Upper San Pedro Riparian Ecosystem. Chapter 3. In: Stromberg, J. C. and B. Tellman (eds). 2009. Ecology and Conservation of the San Pedro River, University of Arizona Press.

Dominguez, F., J. Cañon and J. Valdes. 2009. IPCC-AR4 climate simulations for the Southwestern US: the importance of future ENSO projections, Climatic Change, 99:499–514, 10 October 2009, DOI 10.1007/s10584-009- 9672-5.

Dudley TL, Bean DW. Tamarisk biocontrol, endangered species risk and resolution of conflict through riparian restoration. BioControl. 2012; 57:331–347.

Duke, J. A. 1983. Eragrostis curvula (Schrad.) Nees. Handbook of energy crops. Unpublished. At : Purdue University, Center for New Crops and Plants Products. Website: http://www.hort.purdue.edu/newcrop/nexus/Eragrostis_curvula_nex.html

Eby, L.A., W.F. Fagan, and W.L. Minckley. 2003. Variability and dynamics of a desert stream community. Ecological Applications, 13(6): 1566-1579.

Edison, J., M. Malone, R. Ruisinger, R.O. Russell, J. Tweit, R. Tweit, and R. Yetman. 1995. Davis and Russell's finding birds in southeast Arizona. Tucson Audubon Society, Tucson, AZ.

Epple, A. O. 1995. A Field Guide to the Plants of Arizona. The Globe Pequot Press, Guilford, CT.

Felger, R.F., G.P. Nabhan, and R. Rye. 1997. Apachian/Madrean region of southwestern North America as a center of plant diversity. Pages 172-180 in S.D. Davis and others. Centres of plant diversity: A guide and strategy for their conservation – Volume 3: the America. World Wide Fund for Nature and International Union for Conservation and Nature; Cambridge, U.K.

Felger, R.S. and M.F. Wilson, editors. 1995. Northern Sierra Madre Occidental and its Apachian outliers: a neglected center of biodiversity. Pages 36–59, in L. DeBano et al., editors. 1995. Biodiversity and Management of the Madrean Archipelago: the Sky Islands of Southwestern United States and northern Mexico. Gen. Tech. Rep. RM-GTR-264. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest Range Experiment Station, Fort Collins.

Finch, D. M., editor. 2004. Assessment of Grassland Ecosystem Conditions in the Southwestern United States; Volumes 1 and 2. USDA Forest Service Gen. Tech. Rpt. RMRS-GTR-135. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Fleming, T. H. 2012. Midnight Raiders of Arizona: Monitoring nectar bats at hummingbird feeders. BATS Magazine 30(1): 10-12. Bat Conservation International, Austin, TX.

Flesch, A.D., Epps, C.W., Cain, J.W., Clark, M., Krausman, P.R., and J.R. Morgart. 2009. Potential effects of the United States-Mexico border fence on wildlife. Conservation Biology 24, 171–181

Forman, R. T. T., D. Sperling, J. A. Bissonette, A. P. Clevenger, C. D. Cutshall, V. H. Dale, L. Fahrig, R. France, C. R. Goodman, K Heanue, J. A. Jones, F. J. Swanson, T. Turrentine, and T. C. Winter. 2003. Road Ecology: Science and Solutions. Island Press, Washington, D.C.

Frances, A., J. Cordeiro, E. Kabay, L. Oliver, S. Young, and B. Young. 2011. Assessing Chihuahuan Desert Cactus for Vulnerability to Climate Change. Abstract and oral presentation. Botany 2011 Annual Meeting. St. Louis, MO.

Gibbens, R. P., J. M. Tromble, J. T. Hennessy, and M. Cardenas. 1983. Soil movement in mesquite dunelands and former grasslands of southern New Mexico. Journal of Range Management 36(2):145-148.

Gilbertson, R.M. and D.L. Bigelow. 1998. Annotated checklist of wood-rotting basidiomycetes of the sky islands in southeastern Arizona. Journal of the Arizona-Nevada Academy of Science 31:13-36.

Global Invasive Species Database. Brown Trout. http://www.fs.fed.us/outernet/r2/gmug/policy/mis_amend/assessments/brown_trout_MIS_final_12_10_08.pdf [Accessed April 2013]).

Gorenflo, L. J. 2003. Human demography, land use, and conservation in the Apache Highlands Ecoregion, U.S.-Mexico borderlands. Argonne National Laboratory and Conservation International, Washington, D.C.

Gori, D. and J. Bate. 2007. Historical Range of Variation and State and Transition Modeling of Historical and Current Landscape Conditions for Pinyon-Juniper of the Southwestern U.S. Prepared for the U.S.D.A. Forest Service, Southwestern Region by The Nature Conservancy, Tucson, AZ. 141 pp.

Gori, D., G. S. Bodner, K. Sartor, P. Warren, and S. Bassett. 2012. Sky Island Grassland Assessment: Identifying and Evaluating Priority Grassland Landscapes for Conservation and Restoration in the Borderlands. Report prepared by The Nature Conservancy in New Mexico and Arizona. 85 p.

Gori, D. F., and C. A. F. Enquist. 2003. An Assessment of the Spatial Extent and Condition of Grasslands in Central and Southern Arizona, Southwestern New Mexico and Northern Mexico. Prepared by The Nature Conservancy, Arizona Chapter. 28 pp.

Gottfried, G.J., B.S. Gebow, L.G. Eskew, and C.B. Edminster, compilers. 2005. Connecting mountain islands and desert seas: biodiversity and management of the Madrean Archipelago II. 2004 May 11-15; Tucson, AZ. Proceedings RMRS-P-36. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 631 pp.

Gottfried, G. J., P. F. Ffolliott, B. S. Gebow, L. G. Eskew, and L.C. Collins, compilers. 2013. Merging science and management in a rapidly changing world: Biodiversity and management of the Madrean Archipelago III and 7th Conference on Research and Resource Management in the Southwestern Deserts; 2012 May 1-5; Tucson, AZ. Proceedings. RMRS-P-67. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 593 pp.

Gross, J.E. 2005. Developing Conceptual Models for Monitoring Programs. National Park Service, Inventory and Monitoring Program, Fort Collins, CO. On-line. (http://science.nature.nps.gov/im/monitor/docs/Conceptual_Modelling.pdf).

Grossman, D. H., D. Faber-Langendoen, A. S. Weakley, M. Anderson, P. Bourgeron, R. Crawford, K. Goodin, S. Landaal, K. Metzler, K. D. Patterson, M. Pyne, M. Reid, and L. Sneddon. 1998. International classification of ecological communities: terrestrial vegetation of the United States. Volume I. The National Vegetation Classification System: development, status, and applications. The Nature Conservancy, Arlington, Virginia, USA.

Hansen, A.J., R. L. Knight, J. M. Marzluff, S. Powell, K. Brown, P. H. Gude, and K. Jones. 2005. Effects of exurban development on biodiversity: patterns, mechanisms, and research needs. Ecological Applications 15(6): 1893-1905.

Hare, T. and C. Jones. 1995. History, extent, and future of Arizona BLM-managed roadless areas in the Madrean Archipelago. Pages 371-374 in DeBano, L.F., G.J. Gottfried, R.H. Hamre, C.B. Edminster, P.F. Ffolliott, and A. Ortega-Rubio, editors. 1995. Biodiversity and management of the Sky Island archipelago. U.S. Department of Agriculture Forest Service, Rocky Mountain Range and Forest Experiment Station, General Technical Report RM-GTR-264. Fort Collins, CO.

Harkness, M., M. Reid, P. Crist, L. Misztal, T. Van Devender, G. Kittel, D. Braun, and R. Unnasch. 2013. Madrean Archipelago Rapid Ecoregional Assessment: Pre-Assessment Report. Prepared for the U.S. Department of the Interior, Bureau of Land Management. NatureServe, Arlinton, VA. 116 pages + appendices.

Hastings, J. R. 1959. Vegetation Change and Arroyo Cutting in Southeastern Arizona During the Past Century: An Historical Review. In Arid Lands Colloquia 1958-1959.University of Arizona.

Heinz Center. 2011a. Climate Change A Global Stressor: An Overview of Impacts on the American Southwest. The H. John Heinz III Center for Science, Economics and the Environment, Washington, D.C. 28 pp. www.heinzcenter.org/wildlife

Heinz Center. 2011b. Managing and monitoring Arizona's wildlife in an era of climate change: Strategies and tools for success. Report and Workshop Summary, The H. John Heinz III Center for Science, Economics and the Environment, Washington, D.C. 67pp + appendices.

Hendrickson, D.A. and W. L. Minckley. 1984. Ciénegas--Vanishing Climax Communities of the American Southwest. Desert Plants 6(3):129-176.

Hennessy, J. T., R. P. Gibbens, J. M. Tromble, and M. Cardenas. 1983. Vegetation changes from 1935 to 1980 in mesquite dunelands and former grasslands of southern New Mexico. Journal of Range Management 36(3):370-374.

Herbel, C. H., F. N. Ares, and R. Wright. 1972. Drought effects on a semi-desert grassland range. Ecology 53:1084-1093.

Hereford, R. 1993. Entrenchment and widening of the upper San Pedro River, Arizona: Geological Society of America Special Paper 282, 46 p.

Hidalgo County Soil and Water Conservation District Local Work Group. 2012. Local Work Group Summary for the Lordsburg Field Office. Natural Resources Conservation Service. [http://www.nm.nrcs.usda.gov/programs/eqip/2012/Local%20Work%20Group%20Summaries/lordsburg-lwg-summary.html accessed 3/27/2013.]

Hirschboeck, K.K. 2009. Flood Flows of the San Pedro River. Chapter 16. In: Stromberg, Juliet C. and Barbara Tellman (eds). 2009. Ecology and Conservation of the San Pedro River, University of Arizona Press.

Holling, C. S. 1973. Resilience and stability of ecological systems. Annual Review of Ecology and Systematics 4: 1-23.

Horton, J.L., T.E. Kolb, and S.C. Hart. 2001. Responses of riparian trees to inter-annual variation in groundwater depth in a semi-arid river basin. Plant, Cell and Environment 24: 293-304.

Hostetler, S.W., Alder, J.R. and Allan, A.M., 2011. Dynamically downscaled climate simulations over North America: Methods, evaluation and supporting documentation for users: U.S. Geological Survey Open-File Report 2011-1238, 64 pp. [http://pubs.usgs.gov/of/2011/1238/, http://cida.usgs.gov/climate/hostetlerprojections.jsp]

Humphrey, R. R. 1958. The desert grassland: a history of vegetational change and an analysis of causes. Botanical Review 24: 193-252.

Humphrey, R. R. 1970. Arizona range grasses: Their description, forage value and management. Bulletin 298. Tucson, AZ: The University of Arizona, Agricultural Experiment Station. 159 pp

Humphrey, R. R. 1974. Fire in the deserts and desert grassland of North America. Pages 365-400 in: T. T. Kozlowski and C. E. Ahlgren, editors. Fire and Ecosystems. Academic Press, New York.Johnston, M. C. 1963. Past and present grasslands of southern Texas and northeastern Mexico. Ecology 44:456-464.

Humphrey, R. R. and L. A. Mehrhoff. 1958. Vegetation changes on a southern Arizona grassland range. Ecology. 39(4): 720-726.

Hunter, M. 2011. Assessment and guidelines for determining effectiveness and longevity of buffelgrass treatments in southern Arizona. Report to Saguaro National Park.

IPCC [Intergovernmental Panel on Climate Change]. 2007. 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, 976pp.

Kaib, M., C.Baisan, H. D. Grissino-Mayer, and T. W. Swetnam. 1996. Fire history of the Gallery pine-oak forests and adjacent grasslands of the Chiricahua Mountains of Arizona. Pages 253-264 in: Folliott, P. F., D. F. DeBano, D. M. Baker, G. J. Gottfried, G. Solis-Garza, C. B. Edminster, D. G. Neary, L. S. Allen, and R. H. Hamre, eds. 1996. Effects of fire on Madrean province ecosystems-a symposium proceedings. Gen. Tech. Rep. RM-289; 1996 March 11-15; Tucson, AZ. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Experiment Station. 277 p.

Kamp, Dick. April 16, 2013. Sonora mine plan raises questions about water, air impacts. Nogales International. [http://www.nogalesinternational.com/news/sonora-mine-plan-raises-questions-about-water-air-impacts/article_ef1c85cc-a6ad-11e2-b315-001a4bcf887a.html?mode=story (accessed June 2013)]

Katz, G.L., J.C. Stromberg, and M.W. Denslow. 2009. Streamside herbaceous vegetation response to hydrologic restoration on the San Pedro River, Arizona. Ecohydrology 2: 213–225. DOI: 10.1002/eco.62.

Kearney, T. H., and R. H. Peebles. 1964. Arizona Flora. University of California Press, Berkeley, CA.

Kerr, R. A. 2008. Climate Change Hot Spots Mapped Across the United States. Science 321( 5891): 909. doi: 10.1126/science.321.5891.909

Krueper, D.J. 1993. Conservation priorities in naturally fragmented and human-altered riparian habitats of the arid West. USDA Forest Service. General Technical Report RM-43. Available online: http://ww.birds.cornell.edu/pifcapemay/krueper.htm.

Krueper, D.J. 1995. Effects of livestock management on Southwestern riparian ecosystems. In: Desired future conditions for Southwestern riparian ecosystems: Bringing interests and concerns together. D.W. Shaw, and D.M. Finch, tech coords. Sept. 18-22, 1995; Albuquerque, NM. General Technical Report RM-GTR-272. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, 359 p.

Krueper, D., J. Bart, and T. D. Rich. 2003. Response of vegetation and breeding birds to the removal of cattle on the San Pedro River, Arizona (U.S.A.). Conservation Biology 17:607-615.

Kupfer, J.A., Balmat, J. and J. L. Smith. 2005. Shifts in the potential distribution of Sky Island plant communities in response to climate change. In Gottfried, G. J.; Gebow, B. S.; Eskew, L. G.; and Edminster, C. B., compilers. 2005. Connecting mountain islands and desert seas: biodiversity and

management of the Madrean Archipelago II. 2004 May 11-15; Tucson, AZ. Proceedings RMRS-P-36. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 631 p.

Latta, M. J., C. J. Beardmore, and T. E. Corman. 1999. Arizona Partners in Flight bird conservation plan. Version 1.0. Nongame and Endangered Wildlife Program Technical Report 142. Arizona Game and Fish Department, Phoenix, AZ.

Leake, S.A., D.R. Pool, and J.M. Leenhouts. 2008. Simulated effects of ground-water withdrawals and artificial recharge on discharge to streams, springs, and riparian vegetation in the Sierra Vista Subwatershed of the Upper San Pedro Basin, southeastern Arizona. U.S. Geological Survey Scientific Investigations Report 2008-5207, 14 p.

Leenhouts, J.M., J.C. Stromberg, and R.L. Scott, eds. 2006. Hydrologic requirements of and consumptive ground-water use by riparian vegetation along the San Pedro River, Arizona. U.S. Geological Survey Scientific Investigations Report 2005–5163.

Levick, L., J. Fonseca, D. Goodrich, M. Hernandez, D. Semmens, J. Stromberg, R. Leidy, M. Scianni, D. P. Guertin, M. Tluczek, and W. Kepner. 2008. The Ecological and Hydrological Significance of Ephemeral and Intermittent Streams in the Arid and Semi-arid American Southwest. U.S. Environmental Protection Agency and USDA/ARS Southwest Watershed Research Center, EPA/600/R-08/134, ARS/233046, 116 pp.

Lewis, P.A., C.J. Deloach, A.E. Knutson, J.L. Tracy, T.O. Robbins. 2003: Biology of Diorhabda elongata deserticola (Coleoptera: Chrysomelidae), an Asian leaf beetle for biological control of saltcedars (Tamarix spp.) in the United States. Biological Control, 27: 101–116

Lite, S.J. and J.C. Stromberg. 2005. Surface water and ground-water thresholds for maintaining Populus–Salix forests, San Pedro River, Arizona. Biological Conservation 125: 153–167. DOI:10.1016/j.biocon.2005.01.020

Logan, M. F. 2006. The Lessening Stream: An Environmental History of the Santa Cruz River. University of Arizona Press, Tucson, AZ.

Longcore, T., C. Rich and S. A. Gauthreaux, Jr. 2008. Height, guy wires, and steady-burning lights increase hazard of communication towers to nocturnal migrants: A review and meta-analysis. The Auk 125: 485-492.

Lowry, J. H. Jr., R. D. Ramsey, K. Boykin, D. Bradford, P. Comer, S. Falzarano, W. Kepner, J. Kirby, L. Langs, G. Manis, L. O'Brian, J. Prior-Magee, T. Sajway, K. A. Thomas, S. Schrader, D. Schrupp, K. Schulz, B. Thompson, C. Wallace, C. Velasquez, E. Waller, and B. Wolk. 2005. *Southwest Regional Gap Analysis Project: Final Report on Land Cover Mapping Methods*, RS/GIS Laboratory, Utah State University, Logan, Utah.

Lynch, A. M. 2009. Spruce aphid, Elatobium abietinum (Walker): Life history and damage to Engelmann spruce in the Pinaleño Mountains, Arizona. In: Sanderson, H. Reed; Koprowski, John L., editors. The last refuge of the Mt. Graham red squirrel: Ecology of endangerment. Tucson, AZ: University of Arizona Press. p. 318-338. Available at http://www.treesearch.fs.fed.us/pubs/34003.

MacNish, R.; K. J. Baird, and T. Maddock III. 2009. Groundwater Hydrology of the San Pedro River Basin. Chapter 15. In: Stromberg, Juliet C. and Barbara Tellman (eds). 2009. Ecology and Conservation of the San Pedro River, University of Arizona Press.

Margolis, E.Q., T.W. Swetnam, and C.D. Allen. 2011. Historical stand-replacing fire in upper montane forests of the Madrean Sky Islands and Mogollon Plateau, southwestern USA. Fire Ecology 7(3): 88-107.

Marshall, J. T. 1957. Birds of the pine-oak woodland in southern Arizona and Adjacent Mexico. Cooper Ornithological Society, Berkeley, CA.

Marshall, K. A. 1995. *Larrea tridentata*. 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/ [2013, May 13].

Marshall, R.M., D. Turner, A. Gondor, D. Gori, C. Enquist, G. Luna, R. Paredes Aguilar, S. Anderson, S. Schwartz, C. Watts, E. Lopez, and P. Comer. 2004. An ecological analysis of conservation priorities in the Apache Highlands Ecoregion. Prepared by The Nature Conservancy of Arizona, Instituto del Medio Ambiente y el Desarrollo Sustentable del Estado de Sonora, Agency and Institutional partners. 152 pp.

Mau-Crimmins, T., A. Hubbard, D. Angell, C. Filippone, N. Kline. 2005. Sonoran Desert Network Vital Signs Monitoring Plan. Technical Report NPS/IMR/SODN-003. National Park Service. Denver, CO.

McAuliffe, J. R. and R. L. Burgess. 1995. Landscape complexity, soil development, and vegetational diversity within a sky island peidmont. Pages 91-107 in DeBano, L.F., G.J. Gottfried, R.H. Hamre, C.B. Edminster, P.F. Ffolliott, and A. Ortega-Rubio, editors. 1995. Biodiversity and management of the Sky Island archipelago. U.S. Department of Agriculture Forest Service, Rocky Mountain Range and Forest Experiment Station, General Technical Report RM-GTR-264. Fort Collins, CO.

McClaran, M. P., and T. R. Van Devender (eds.). 1995. The Desert Grassland, University of Arizona Press, Tucson.

McCluney, K. E., J. Belnap, A. L. Gonzalez, J. N. Holland, B. P. Kotler, F. T. Maestre, S. D. Smith, S. L. Collins, B. O. Wolf, and E. M. Hagen. 2012. Shifting species interactions in terrestrial dryland ecosystems under altered water availability and climate change. Biological Reviews 87(3): 563-582. doi: 10.1111/j.1469-185X.2011.00209.x

McDonald, C.J. and G.R. McPherson. 2011. Fire behavior characteristics of buffelgrass-fueled fires and native plant community composition in invaded patches. Journal of Arid Environments. 75:1147-1154.

McKechnie, A.E. and B.O. Wolf. 2010. Climate change increases the likelihood of catastrophic avian mortality events during extreme heat waves. Biology Letters 6: 253-256.

McLaughlin, S. P. 1994. An overview of the flora of the sky islands, southeastern Arizona: diversity, affinities, and insularity. Pages 60-70 in DeBano, L.F., G.J. Gottfried, R.H. Hamre, C.B. Edminster, P.F. Ffolliott, and A. Ortega-Rubio, editors. 1995. Biodiversity and management of the Sky Island archipelago. U.S. Department of Agriculture Forest Service, Rocky Mountain Range and Forest Experiment Station, General Technical Report RM-GTR-264. Fort Collins, CO.

McLaughlin, S. P. and J. E. Bowers. 1982. Effects of wildfire on a Sonoran Desert plant community. Ecology 63(1):246-248.

McNab, W.H., D.T. Cleland, J.A. Freeouf, J.E. Keys, Jr., G.J. Nowacki, and C.A. Carpenter, comps. 2007. Description of ecological subregions: sections of the conterminous United States [CD-ROM]. Gen. Tech. Report WO-76B. Washington, DC: U.S. Department of Agriculture, Forest Service. 80 p.

McPherson, G. R. 1995. The role of fire in the desert grasslands. In: M. P. McClaran and T. R. Van Devender (eds.). The Desert Grassland. University of Arizona Press, Tucson, Arizona. pp. 130-151.

McPherson, G. R. and J. F. Weltzin. 2000. Disturbance and climate change in United States/Mexico borderland plant communities: a state-of-the-knowledge review. USDA Forest Service RMRS-GTR-50.

Mendoza, M. G., Y. Salinas-Moreno, A. Olivo-Martínez, and G. Zúñiga. 2011. Factors influencing the geographical distribution of Dendroctonus rhizophagus (Coleoptera: Curculionidae: Scolytinae) in the Sierra Madre Occidental, México. Environmental Entomology 40(3):549-59.

Milchunas, Daniel G. 2006. Responses of plant communities to grazing in the southwestern United States. Gen. Tech. Rep. RMRS-GTR-169. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 126 p.

Miller, G. C., W. B. Lyons, and A. Davis. 1996. Understanding the water quality of pit lakes. Environmental Science and Technology 30: 118-123.

Minckley, W.L. 1973. Fishes of Arizona. Arizona Game and Fish Department, Phoenix, Arizona.

Mondt, R. Personal communication. Special Designations Program Coordinator. Sky Island Alliance.

Mooney, H. A., T. M. Bonnicksen, N. L. Christensen, and others. 1981. Proceedings of the conference: fire regimes and ecosystem properties; 1978 December 11-15; Honolulu, HI. Gen. Tech. Rep. WO-26. Washington, DC: U.S. Department of Agriculture, Forest Service. 594 pp

Muldavin, E., T. Neville, C. McGuire, P. Pearthree, and T. Biggs. 2002. Soils, geology, and vegetation change in the Malpais Borderlands. New Mexico Natural Heritage Program, University of New Mexico, Albuquerque, NM.

Natural Channel Design, Inc. 2010. Monitoring Report: Channel Morphology Related to the Pedestrian Fence, Organ Pipe Cactus National Monument and Coronado National Memorial. National Park Service. 34 pp.

NatureServe. 2013. Terrestrial Ecological Systems of the Conterminous United States. Version 2.9. Completed in cooperation with USGS Gap Analysis Program and inter-agency LANDFIRE. Reflecting early 2000s land cover. MMU approx. 2 hectares. NatureServe, Arlington, VA. Digital map.

NMBGMR [New Mexico Bureau of Geology and Mineral Resources]. 2008. http://geoinfo.nmt.edu/faq/mining/home.html (accessed June 4, 2013)

NMDGF [New Mexico Department of Game and Fish]. 2006. Comprehensive Wildlife Conservation Strategy for New Mexico. New Mexico Department of Game and Fish. Santa Fe, NM. 526 pp + appendices.

NMOSE [New Mexico Office of the State Engineer] 2008. New Mexico Water Use by Categories 2005. Prepared by J.W. Longworth, J.M. Valdez, M.L. Magnuson, E.S. Albury and J. Keller. Technical Report 52, June 2008, Santa Fe, NM. Available at http://www.ose.state.nm.us/PDF/Publications/Library/TechnicalReports/TechReport-052.pdf

Noonan, G. R. 2013. Major Changes in Riparian Habitat Along the Upper San Pedro River and Other Southwestern Waterways as a Result of the Alluvial Cycle. 75 p. Science Quest Technical Paper 1. (http://sciencequest.webplus.net/noonan%20san%20pedro%20river%20papers.html).

NRCS [U.S. Department of Agriculture, Agricultural Research Service, Natural Resources Conservation Service]. 2006. Land Resource Regions and Major Land Resource Areas of the United States, the Caribbean, and the Pacific Basin. United States Department of Agriculture Handbook 296. 663 p.

NRCS [U.S. Department of Agriculture, Agricultural Research Service, Natural Resource Conservation Service], BLM [U.S. Department of the Interior, Bureau of Land Management], and USGS [United States Geological Survey]. 2005. Interpreting Indicators of Rangeland Health, Version 4 - Technical Reference 1734-6.

NWCG [National Wildfire Coordinating Group]. 2006. Glossary of Wildland Fire Terminology. http://www.nwcg.gov/pms/pubs/glossary/f.htm.

Olsson, A.D., J. Betancourt, M.P. McClaran, and S.E. Marsh. 2011. Sonoran Desert ecosystem transformation by a C4 grass without the grass/fire cycle. Diversity and Distributions. 18: 10-21.

Omernik, J.M. and R.G. Bailey. 1997. Distinguishing Between Watersheds and Ecoregions. Journal of the American Water Resources Association 33(5): 935-949.

Parendes, L.A. and J.A. Jones. 2000. Light availability, dispersal, and exotic plant invasion along roads and streams in the H.J. Andrews Experimental Forest, Oregon. Conservation Biology 14: 64-75.

Parmenter, R. R., and T. R. Van Devender. 1995. Diversity, spatial variability, and functional roles of vertebrates in the desert grassland. In: M. P. McClaran and T. R. Van Devender (eds.). The Desert Grassland. University of Arizona Press, Tucson, Arizona. p. 196-229.

Parrish, J. D., D. P. Braun, and R.S. Unnasch. 2003. Are we conserving what we say we are? Measuring ecological integrity within protected areas. Bioscience 53(9): 851-860.

Patrick, C., S. Avila-Villegas, J. Neeley, and L. Misztal. Mapping and Assessing the Environmental Impacts of Border Tactical Infrastructure in the Sky Island Region. Sky Island Alliance, Tucson, AZ. In: Gottfried, G.J., B.S. Gebow, L.G. Eskew, and C.B. Edminster, compilers. 2005. Connecting mountain islands and desert seas: biodiversity and management of the Madrean Archipelago II. 2004 May 11-15; Tucson, AZ. Proceedings RMRS-P-36. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 631 pp.

Paysen, T.E., J.R. Ansley, J.K. Brown, G.J. Gottfried, S.M. Haase, M.J. Harrington, M.G. Narog, S.S. Sackett and R.C. Wilson. 2000. Chapter 6: Fire in western shrubland, woodland, and grassland ecosystems. Pages 121-159 in: J.K. Brown and J. Kapler-Smith, eds. Wildland fire in ecosystems: effects of fire on flora. Gen. Tech. Rep. RMRS-GTR-42-vol. 2. Ogden, UT: USDA Forest Service, Rocky Mountain Research Station. 257 pp.

Pima Association of Governments. 2013. Population Growth by Decade. http://www.pagnet.org/RegionalData/Demographics/PopulationGrowthbyDecade/tabid/123/Default.aspx. Accessed 9/10/2013.

Poff, N. L., B. D. Richter, A. H. Arthington, S. E. Bunn, R. J. Naiman, E. Kendy, M. Acreman, C. Apse, B. P. Bledsoe, M. C. Freeman, J. Henriksen, R. B. Jacobson, J. G. Kennen, D. M. Merritt, J. H. O'Keeffe, J. D. Olden, K. Rogers, R. E. Tharme, and A. Warner. 2010. The ecological limits of hydrologic alteration (ELOHA): A new framework for developing regional environmental flow standards. Freshwater Biology 55:147-170.

Pond, F. W., and D. R. Cable. 1962. Recovery of vegetation following wildfire on a chaparral area in Arizona. Research Note RM-72. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado.

Pool, D. R. and A. L. Coes. 1999. Hydrogeologic investigations of the Sierra Vista Subbasin of the Upper San Pedro River Basin, Cochise Country, Southeast Arizona. USGS Water-Resources Investigations Report 99-4197.

Propst, D. L., K. B. Gido, and J. A. Stefferud. 2008. Natural flow regimes, nonnative fishes, and native fish persistence in arid-land river systems. Ecological Applications 18(5): 1236–1252.

Rauzi, S, 2012, Arizona Oil and Gas Well Location Maps and Report. Arizona Geological Survey Oil and Gas (OG) -12, 32 p. 2 map sheets.

Redford K.H., Coppolillo P., Sanderson E.W., Da Fonseca G.A.B., Dinerstein E., Groves C., Mace G., Maginnis S., Mittermeier R.A, Noss R., Olson D., Robinson J.G., Vedder A., and Wright M. 2003. Mapping the Conservation Landscape. Conservation Biology 17(1): 116-131.

Reina-G., A. L., and T. R. Van Devender. 2005. Floristic comparison of an Arizona 'sky island' and the Sierra Madre Occidental in eastern Sonora: the Huachuca Mountains and the Yécora area. Pp. 154-157 in G. J. Gottfried, B. S. Gebow, L. G. Eskew, and C. B. Edminster (compilers). Connecting Mountain Islands and Desert Seas: Biodiversity and Management of the Madrean Archipelago II. Proc. RMRS-P-36, USDA Forest Service, Rocky Mt. Res. Station, Ft. Collins, CO.

Rhine, John. 1996. The effects of introduced fishes on native fishes: Arizona, Southwestern United States. USDA Forest Service Rocky Mountain Forest and Range Experiment Station. Proceedings of the World Fisheries Congress.

Rivas-Martinez, S., A. Penas, M. A. Luengo, S. Rivas-Saenz. 2002. Worldwide Bioclimatic Classification System, CD-Series: Climate and Biosphere, editor-in-chief H. Lieth, second edition, CD no II. Backhuys Publishers, Leiden/NL/Lambda Digitex GmbH, Osanabrueck/G. ISBN 90.5782.139.7

Robinett, D. 1994. Fire effects on southeastern Arizona plains grasslands. *Rangelands*, 16, 143-148.

Ruth, J. M. Personal communication. Research Ecologist. U.S. Geological Survey, Arid Lands Field Station, University of New Mexico.

Ruyle, G. B. and D. J. Young, (eds.). 1997. Arizona range grasses. Cooperative Extension, publication AZ97105, College of Agriculture, The University of Arizona, Tucson, Arizona. Website: http://ag.arizona.edu/pubs/natresources/az97105/

Salinas-Moreno, Y.; Ager, A.; Vargas, C.F.; Hayes, J.L.; Zuniga, G. 2010. Determining the vulnerability of Mexican pine forests to bark beetles of the genus Dendroctonus Erichson (Coleoptera: Curculionidae: Scolytinae). Forest Ecology and Management. 260: 52-61.

Santos-Barrera, G., G. Hammerson, and M. J. Sredl. 2004. Lithobates chricahuensis. In: IUCN 2012. IUCN Red List of Threatened Species. Version 2012.2. http://www.iucnredlist.org/details/58575/0

Sayre, N.F. 2005. Rangeland degradation and restoration in the “desert seas”: social and economic drivers of ecological change between the Sky Islands. In Gottfried, G. J.; Gebow, B. S.; Eskew, L. G.; and Edminster, C. B., compilers. 2005. Connecting mountain islands and desert seas: biodiversity and management of the Madrean Archipelago II. 2004 May 11-15; Tucson, AZ. Proceedings RMRS-P-36. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 631 p.

Sayre, N.F. and R.L. Knight. 2010. Potential effects of United States-Mexico border hardening on ecological and human communities in the Malpai Borderlands. Conservation Biology 24: 345-348.

Schlesinger, W. H., J. F. Reynolds, G. L. Cunningham, L. F. Huenneke, W. M. Jarrell, R. A. Virginia, and W. G. Whitford. 1990. Biological feedbacks in global desertification. Science 247:1043-1048.

Schmidt, S.L. and D.C. Dalton. 1995. Bats of the Madrean Archipelago (Sky Islands): Current Knowledge, Future Directions. In: Biodiversity and management of the Madrean Archipelago: the Sky Islands of Southwestern United States and Northwestern Mexico. Gen. Tech. Rep. GTR-RM-264. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station.669 p.

Schreiber, Jr. J.F. 1978. Geology of the Willcox Playa, Cochise County, Arizona. New Mexico Geological Society Guidebook, 29th Field Conference, Land of Cochise.

Schussman, H. 2006a. Historical Range of Variation and State and Transition Modeling of Historical and Current Landscape Conditions for Semi-Desert Grassland of the Southwestern U.S. Prepared for the U.S.D.A. Forest Service, Southwestern Region by The Nature Conservancy, Tucson, AZ. 53 pp.

Schussman, H. 2006b. Historical Range of Variation for Madrean Encinal of the Southwestern U.S. Prepared for the U.S.D.A. Forest Service, Southwestern Region by The Nature Conservancy, Tucson, AZ. 16 pp.

Schussman, H. and E. Smith. 2006 Historical Range of Variation and State and Transition Modeling of Historic and Current Landscape Conditions for Potential Natural Vegetation Types of the Southwest. The Nature Conservancy.

Seager, R., M. Ting, I. Held, Y. Kushnir, J. Lu, G. Vecchi, H. Huang, N. Harnik, A. Leetmaa, N. Lau, C. Li, J. Velez, and N. Naiket. 2007. Model projections of an imminent transition to a more arid climate in southwestern North America. Science 316:1181–1184.

Serrat-Capdevilla, A., J. B. Valdes, J. Gonzales Perez, K. Baird, L. J. Mata and T. Maddock III. 2007 Modeling climate change impacts--and uncertainty--on the hydrology of a riparian system: The San Pedro Basin (Arizona/Sonora). Journal of Hydrology 347:48-66.

Shafroth, P.B., J.C. Stromberg, and D.T. Patten. 2000. Woody riparian vegetation response to different alluvial water table regimes. Western North American Naturalist 60(1): pp. 66–76.

Shafroth, P.B., A.C. Wilcox, D.A. Lytle, J.T. Hickey, D.C. Andersen, V.B. Beauchamp, A. Hautzinger, L.E. McMullen, A. Warner. 2010. Ecosystem effects of environmental flows: modeling and experimental floods in a dryland river. Freshwater Biology 55: 68–85. DOI:10.1111/j.1365-2427.2009.02271.x 1.

Simpson, G.G. 1964. Species density of North American mammals. Systematic Zoology 13:57-73.

Smith, S. E., M. G. Mendoza, G. Zúñiga, K. Halbrook, J. L. Hayes, and D. N. Byrne. 2013. Modeling current and future suitable habitat for a seedling-attacking bark beetle and its pine hosts in the Madrean Archipelago. Pp. in G. J. Gottfried, P. F. Ffolliott, B. S. Gebow, and L. G. Eskew (compilers). Merging science and management in a rapidly changing world: biodiversity and management of the Madrean Archipelago III. 2012 May 1-5, Tucson, AZ. Proceedings RMRS-P-67. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Snyder, K.A. and D.G. Williams. 2000. Water sources used by riparian trees varies among stream types on the San Pedro River, Arizona. Agricultural and Forest Meteorology 105: 227–240.

Sogge, M.K., Sferra, S.J., Paxton, E.H., 2008. Tamarix as habitat for birds: implications for riparian restoration in the southwestern United States. Restoration Ecology 16: 146-154.

Soil Conservation Service. 1972. Management and uses of Weeping Lovegrass in Arizona. U.S. Department of Agriculture, Soil Conservation Service. USDA Plants Profile Eragrostis curvula. http://plants.usda.gov/java/profile?symbol=ercu2. Accessed June 2013.

Solomon, S., G. Plattner, R. Knutti, and P. Friedlingstein. 2009. Irreversible climate change due to carbon dioxide emissions. Proceedings of the National Academy of Sciences 106(6): 1704-1709.

Sprouse, T. 2005. Water Issues on the Arizona-Mexico Border: The Santa Cruz, San Pedro and Colorado Rivers. University of Arizona, Tucson, AZ.

Stephens, D. B. and Associates, Inc. 2005. Southwest New Mexico Regional Water Plan. Prepared for Southwest New Mexico Regional Water Plan Steering Committee, City of Deming, New Mexico. Albuquerque, NM.

Strittholt, J.R., S.A. Bryce, B.C. Ward, and D.M. Bachelet. 2012. Sonoran Desert Rapid Ecoregional Assessment Report. Prepared for the U.S. Department of the Interior, Bureau of Land Management, Denver, Colorado.

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

Stromberg, J.C. 2001. Restoration of riparian vegetation in the southwestern United States: importance of flow regimes and fluvial dynamism. Journal of Arid Environments 49: 17-34. DOI:10.1006/jare.2001.0833.

Stromberg, J. C. and B. Tellman (eds). 2009. Ecology and Conservation of the San Pedro River, University of Arizona Press.

Stromberg, J.C., K.J. Bagstad, J.M. Leenhouts, S.J. Lite, and E. Makings. 2005. Effects of stream flow intermittency on riparian vegetation of a semiarid region river (San Pedro River, Arizona). River Research and Applications 21: 925–938. DOI: 10.1002/Rra.858.

Stromberg, J.C., M.D. Dixon, R.L. Scott, T. Maddock III, K.J. Baird and B. Tellman. 2009. Status of the Upper San Pedro River (United States) Riparian Ecosystem. Chapter 20. In: Stromberg, Juliet C. and Barbara Tellman (eds). 2009. Ecology and Conservation of the San Pedro River, University of Arizona Press.

Stromberg, J.C., R. Tiller, and B. Richter. 1996. Effects of groundwater decline on riparian vegetation of semiarid regions: the San Pedro, Arizona. Ecological Applications 6(1): 113-131.

Stromberg, J.C., V.B. Beauchamp, M.D. Dixon, S.J. Lite, and C. Paradzick. 2007. Importance of low-flow and high-flow characteristics to restoration of riparian vegetation along rivers in arid south-western United States. Freshwater Biology 52: 651–679. DOI:10.1111/j.1365-2427.2006.01713.x.

SWCCN [Southwest Climate Change Network]. 2008. Drought and the Environment. http://www.southwestclimatechange.org/impacts/land/drought

Swetnam, T.W. 1988. Fire history and climate in the southwestern United States. In: Krammes, J. S., Tech. Coord. 1990. Effects of Fire Management of Southwestern Natural Resources. Gen. Tech. Rep. RM-191. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 293 p.

Swetnam, T. and C. Baisan. 1996. Historical fire regime patterns in the southwestern United States since AD 1700. In: CD Allen (ed) Fire Effects in Southwestern Forests: Proceedings of the 2nd La Mesa Fire Symposium, pp. 11-32. USDA Forest Service, Rocky Mountain Research Station, General Technical Report RM-GTR-286.

Theobald, D. M., D. M. Merritt, and J. B. Norman, III. 2010. Assessment of threats to riparian ecosystems in the western U.S. Prepared for the Western Environmental Threats Assessment Center, Prineville, OR, by the Department of Human Dimensions of Natural Resources and the Natural Resource Ecology Lab, Colorado State University and USDI Forest Service Watershed, Fish, Wildlife, Air and Rare Plants Staff, Natural Resource Research Center, Ft. Collins, CO. 56 pp.

Thomas, B.E. and D.R. Pool. 2006. Trends in Streamflow of the San Pedro River, Southeastern Arizona, and Regional Trends in Precipitation and Streamflow in Southeastern Arizona and Southwestern New Mexico. U.S. Geological Survey Professional Paper 1712, 79 p.

Thornburg, A. A. 1982. Plant materials for use on surface-mined lands. SCS-TP-157. Washington, DC: U.S. Department of Agriculture, Soil Conservation Service. 88 pp.

Tillman, F.D., J.T. Cordova, S.A. Leake, B.E. Thomas, and J.B. Callegary. 2011. Water availability and use pilot: methods development for a regional assessment of groundwater availability, southwest alluvial basins, Arizona. U.S. Geological Survey Scientific Investigations Report 2011-5071, 118 p.

Tracy, J.L. and T.O. Robbins. 2009: Taxonomic revision and biogeography of the Tamarix-feeding Diorhabda elongata (Brullé, 1832) species group (Coleoptera: Chrysomelidae: Galerucinae: Galerucini) and analysis of their potential in biological control of Tamarisk. Zootaxa, 2101: 1-152.

Treher, A., A. Frances, L. Oliver, and B.E. Young.2012. Vulnerability of Plants on BLM Lands to Climate Change. Technical Report prepared by NatureServe, Arlington, VA.

Turner, Dale S.; Brandes, S.; Fishbein, M.; Hirt, P.W. 1995. Preserve design for maintaining biodiversity in the Sky Island Region. In: Biodiversity and management of the Madrean Archipelago: the Sky Islands of Southwestern United States and Northwestern Mexico. Gen. Tech. Rep. GTR-RM-264.

Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station.669 p.

Turner, R.M., Webb, R.H., and J.E.Bowers. 2003. The Changing Mile Revisited. University of Arizona Press, Tucson. 334 pp.

Unnasch, R.S., D. P. Braun, P. J. Comer, G. E. Eckert. 2009. The ecological integrity assessment framework: A framework for assessing the ecological integrity of biological and ecological resources of the National Park System**.** Report to the National Park Service.

Updike, R. G., E. G. Ellis, W. R. Page, M. J. Parker, J. B. Hestbeck, and W. F. Horak (eds). 2013. United States-Mexico Borderlands: Facing tomorrow's challenges through USGS science. U.S. Geological Survey Circular 1380, 336 p., 1 pl.

URS Corporation and CDM, Inc. 2002. Arid West Water Quality Research Project: Habitat Characterization Study, Final Report to Pima County Wastewater Management Department. Phoenix, AZ.

USCCSP [U.S. Climate Change Science Program] and the Subcommittee on Global Change Research. 2009. Thresholds of Climate Change in Ecosystems, U.S. Climate Change Science Program Synthesis and Assessment Product 4.2, January 2009, http://downloads.climatescience.gov/sap/sap4-2/sap4-2-final-report-all.pdf

USFS [USDA Forest Service]. 2004. Arizona bark beetle epidemics: Fact Sheet and Information Bulletin. January 2004.

USFWS [U.S. Fish and Wildlife Service]. 1999. Designation of Critical Habitat for the Huachuca Water Umbel, a Plant. Department of the Interior Fish and Wildlife Service, 50 CFR Part 17 RIN 1018–AF37, Endangered and Threatened Wildlife and Plants; Designation of Critical Habitat for the Huachuca Water Umbel, a Plant, Final rule. Federal Register /Vol. 64, No. 132 /Monday, July 12, 1999 /Rules and Regulations 37441-37453.

USFWS [U.S. Fish and Wildlife Service]. 2007. Chiricahua Leopard Frog (Rana chiricahuensis) Recovery Plan. Region 2, U.S. Fish and Wildlife Service, Albuquerque, NM. 429 pp.

USGCRP [U.S. Global Change Research Program]. 2009. Global Climate Change Impacts in the United States, Thomas R. Karl, Jerry M. Melillo, and Thomas C. Peterson, (eds.). Cambridge University Press. http://www.globalchange.gov.

USGS [U.S. Geological Survey] and NRCS [U.S. Department of Agriculture, Natural Resources Conservation Service]. 2009. Federal guidelines, requirements, and procedures for the national Watershed Boundary Dataset: U.S. Geological Survey Techniques and Methods 11–A3, 55 p. Available at: http://pubs.usgs.gov/tm/tm11a3/pdf/TM11-A3.pdf.

USGS [United States Geological Survey]. 2012. Protected Areas Database of the United States (PADUS), version 1.3. USGS Gap AnalysisProgram. http://gapanalysis.usgs.gov/padus.

U.S. Government Accountability Office. 2005. Livestock Grazing: Federal Expenditures and Receipts Vary, Depending on the Agency and the Purpose of the Fee Charged. Report to Congressional Requestors. GAO-05-869. 51 pp and appendices.

Van de Water, P. K., and W. D. Peachey. 1997. Dietary analysis of the Mexican long-tongued bat (Choeronycteris mexicana) using pollen analysis of guano collected at Cienaga Creek Natural Preserve. Bat Research News 38:181.

Van Devender, T. R. 1995. Desert Grassland History: Changing climates, evolution, biogeography, and community dynamics. Pp. 68-99 in M. P. McClaran and T. R. Van Devender (eds.), The Desert Grassland, University of Arizona Press, Tucson.

Van Devender, T.R., R.S. Felger, and A.M. Burquez. 1997. Exotic Plants in the Sonoran Desert Region, Arizona and Sonora. California Exotic Pest Plant Council 1997, Symposium Proceedings.

Van Devender, T. R., R S. Felger, A. L. Reina-Guerrero, and J. Jesús Sánchez–Escalante. 2009. Sonora: Non-native and invasive plants. Pp. 85-124 in T. R. Van Devender, F. J. Espinosa-García, B. L. Harper-Lore, and T. Hubbard (eds.), Invasive Plants on the Move. Controlling them in North America. Proceedings of Weeds Across Borders 2006 Conference, Hermosillo, Sonora, May 25-28, 2006; Tucson, AZ.

Van Devender, T. R., R. S. Felger, and M. A. Dimmitt. 2007. Sonoran Desert weed accounts. Pp. 28-29 in Bonnie L. Harper-Lore, M. Johnson, and M. W. Skinner, Roadside Weed Management, U.S. Department of Transportation, Federal Highway Administration.

Van Devender, T. R., R. S. Felger, M. Fishbein, F. Molina-Freaner, J. J. Sánchez-Escalante, and A. L. Reina-Guerrero. 2010. Biodiversidad de las plantas vasculares. Pp. 229-262 in F. Molina-F. and T. R. Van Devender (eds.), Diversidad Biológica de Sonora, Universidad Nacional Autónoma de México, Hermosillo.

Van Devender, T. R., S. Avila-Villegas, M. Emerson, D. Turner, A. D. Flesch, and N. S. Deyo. 2013. Biodiversity in the Madrean Archipelago of Sonora, Mexico. Pp. in G. J. Gottfried, P. F. Ffolliott, B. S. Gebow, and L. G. Eskew (compilers). Merging science and management in a rapidly changing world: biodiversity and management of the Madrean Archipelago III. 2012 May 1-5, Tucson, AZ. Proceedings RMRS-P-67. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Villa-R., B. 1967. Los murciélagos de México. Instituto de Biología, Universidad Nacional Autónoma de México, Distrito Federal, México, 491 pp.

Walsh, R. A. 1994. Eragrostis curvula. In: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. 2002. Fire Effects Information System. Website: http://www.fs.fed.us/database/feis/plants/graminoid/eracur/all.html

Warshall, P. 1986. Biogeography of the high peaks of the Pinaleños. Reprinted from the Environmental Data Book,U S Forest Service,Coronado National Forest, by the Maricopa Audubon Society, Phoenix, Arizona.

Warshall, P. 1995. The Madrean sky island archipelago: a planetary overview. Pages 6-18 in DeBano, L.F., G.J. Gottfried, R.H. Hamre, C.B. Edminster, P.F. Ffolliott, and A. Ortega-Rubio, editors. 1995. Biodiversity and management of the Sky Island archipelago. U.S. Department of Agriculture Forest Service, Rocky Mountain Range and Forest Experiment Station, General Technical Report RM-GTR-264. Fort Collins, CO.

Webb, R.H. and S. A. Leake. 2006. Ground-water surface-water interactions and long-term change in riverine riparian vegetation in the southwestern United States. Journal of Hydrology 320: 302–323. DOI:10.1016/j.jhydrol.2005.07.022.

Webb, R.H., S.A. Leake, and R.M. Turner. 2007. The ribbon of green: Change in riparian vegetation in the southwestern United States. Tucson, University of Arizona Press. 462 p.

Weltzin, J.F., S. Archer, and R.K. Keitschmidt. 1997. Small-mammal regulation of vegetation structure in a temperate savanna. Ecology 78(3): 751-763.

Weltzin, J.F., M.E. Loik, S. Schwinning, D.G. Williams, P.A. Fay, B.M. Haddad, J. Harte, T.E. Huxman, A.K. Knapp, G. Lin, W.T. Pockman, M.R. Shaw, E.E. Small, M.D. Smith, S.D. Smith, D.T. Tissue, and J.C. Zak. 2003. Assessing the response of terrestrial ecosystems to potential changes in precipitation. Bioscience 53(10): 941-952.

White, P. S., and S. T. A. Pickett. 1985. Natural disturbance and patch dynamics: An introduction. Pages 3-13 in: P. S. White and S. T. A. Pickett, editors. The ecology of natural disturbance and patch dynamics. Academic Press, Orlando, FL.

Whitford, W. G., G. S. Forbes, and G. I. Kerley. 1995. Diversity, spatial variability, and functional roles of invertebrates in desert grassland ecosystems. Pages 151-195 in: M. P. McClaran and T. R. Van Devender, editors. The Desert Grassland. University of Arizona Press, Tucson.

Whittaker, R. H., and W. A. Niering. 1968. Vegetation of the Santa Catalina Mountains, Arizona: IV. Limestone and Acid Soils. Journal of Ecology 56:523-544

Wiken, E., F.J. Nava, and G. Griffith. 2011. North American Terrestrial Ecoregions—Level III. Commission for Environmental Cooperation, Montreal, Canada. 149 p.

Williams, B. K., R. C. Szaro, and C. D. Shapiro. 2009. Adaptive Management: The U.S. Department of the Interior Technical Guide. Adaptive Management Working Group, U.S. Department of the Interior, Washington, DC.

Wilson, T.B., R.H. Webb, and T.L. Thompson. 2001. Mechanisms of range expansion and removal of mesquite in desert grasslands of the Southwestern United States. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station General Technical Report, RMRS-GTR-81. Ogden, UT. 23 p.

Wood, C. W., Jr. and T. N. Nash III. 1976. Copper smelter effluent effects on Sonoran Desert vegetation. Ecology 57: 1311-1316.

Wright, H. A. The role and use of fire in the semidesert grass-shrub type. 1980. Ogden, UT, U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station.

Wright, H. A., A. W. Bailey, and R. P. Thompson. 1978. The role and use of fire in the Great Plains: a state-of-the-art review. In: Proceedings: Prairie prescribed burning symposium and workshop. April 25-28, 1978, Jamestown, North Dakota. The Wildlife Society, North Dakota Chapter, VIII-1 to VIII-29. On file with: U.S. Department of Agriculture, Forest Service, Intermountain Research Station, Fire Sciences Laboratory, Missoula, Montana.

York, J. C., and W. A. Dick-Peddie. 1969. Vegetation changes in southern New Mexico during the past hundred years. Pages 157-166 in: W. O. McGinnies and B. J. Goldman, editors. Arid lands in perspective. University of Arizona Press, Tucson.

Zugmeyer, C. and J. Koprowski. 2009. Habitat selection is unaltered after severe insect infestation: Concerns for forest dependent species. Journal of Mammalogy 90(1):175.182.

# Appendix H Mesquite Scrub Expansion: Restoration Opportunities

Version December 8, 2014

# Contents

Appendix H Mesquite Scrub Expansion: Restoration Opportunities ..... 1
H.1 Overview of Appendix H ..... 4
H.2 Overview of the Conceptual Model for Apacherian-Chihuahuan Mesquite Upland Scrub ..... 4
H.2.1 Methods for Characterization of the Ecosystem ..... 5
H.3 Apacherian-Chihuahuan Mesquite Upland Scrub Conceptual Model ..... 6
H.3.1 Classification ..... 7
H.3.2 Summary ..... 7
H.3.3 Species and Mesquite ..... 9
H.3.4 Ecological System Dynamics ..... 10
H.3.4.1 Upland Mesquite Dynamics ..... 10
H.3.4.2 Stressors on Ecological Dynamics Causing Continued Degradation or Change ..... 11
H.3.4.3 Dynamics Models ..... 13
H.3.5 Key Factors Related to Restoration Potential ..... 18
H.4 Assessment Methods ..... 19
H.5 Assessment Results ..... 26
H.6 References ..... 28

# Tables

Table H-1. Apacherian-Chihuahuan Mesquite Upland Scrub ecosystem CE crosswalk with approved Ecological Site Descriptions. ..... 9
Table H-2. Stressors that are likely to affect the successful restoration or management of Apacherian-Chihuahuan Mesquite Upland Scrub in the Madrean Archipelago ecoregion. ..... 11
Table H-3. Key factors for determining the potential for restoration of places dominated by the Apacherian-Chihuahuan Mesquite Upland Scrub ecosystem. ..... 18
Table H-4. The two variables of development and mesquite cover, and their classification into three categories of suitability for restoration of mesquite invaded uplands. ..... 20
Table H-5. Soil Great Groups found in the MAR, and their classification into categories of suitability for grassland restoration ..... 22
Table H-6. Decision matrix used in the Vista evaluation for categories of suitability for each of the three variables. ..... 25

# Figures

Figure H-1. Apacherian-Chihuahuan Mesquite Upland Scrub (http://www.azfirescape.org). ..... 8
Figure H-2. Conceptual state and transition model of the Historic Climax Plant Community (HCPC) and altered dynamics for NRCS ESD R041XA210AZ. ..... 14
Figure H-3. Conceptual state and transition model of the Historic Climax Plant Community (HCPC) and altered dynamics for NRCS ESD R041XA107AZ. ..... 16
Figure H-4. Maps of two of the variables used to determine areas suitable for restoration of mesquite invaded uplands: development (upper), mesquite cover (lower)..... 21
Figure H-5. Map of results of categorizing the soils of MAR into areas suitable for restoration of mesquite invaded uplands..... 24
Figure H-6. Map of suitability that incorporates all three variables (development, mesquite cover, and soils) using the decision matrix found in Table H-6. ..... 25
Figure H-7. Map of the current distribution of Apacherian-Chihuahuan Mesquite Upland Scrub (yellow) combined with the distribution of degraded grasslands (red), and the historical distribution of Apacherian-Chihuahuan Semi-desert Grassland and Steppe & Madrean Encinal ..... 26
Figure H-8. Final results showing areas identified for suitability of restoration of mesquite-invaded uplands to grasslands or other native ecosystems..... 27

## H.1 Overview of Appendix H

This appendix contains the conceptual model and the assessment results for the Apacherian-Chihuahuan Mesquite Upland Scrub. Appendix A describes the methods for selection of the CEs and the change agents (CAs), as well as the collection and organization of management questions (MQs) of interest to many partners active in this ecoregion. Appendices B and C contain the assessment methods: B contains the methodological approaches to the geospatial assessments, while C contains the technical GIS documentation. Appendix B contains the methods for the assessment of mesquite; while Appendix C will contain the detailed documentation of the GIS steps. Other appendices contain the conceptual models and ecological status assessment results for the terrestrial CEs (Appendix D), aquatic CEs (Appendix E) and species (Appendix F). Appendix G contains the ecoregional conceptual model and the results from the ecoregion ecological integrity assessment. Appendix I includes all of the methods and results for the climate change assessments.

## H.2 Overview of the Conceptual Model for Apacherian-Chihuahuan Mesquite Upland Scrub

Mesquite (both *Prosopis velutina* and *P. glandulosa*) has spread into many areas of the ecoregion and, in some locations, has caused a complete type conversion from the natural ecosystems (primarily grasslands and some encinal) to upland mesquite scrub. According to the land cover mapping completed by NatureServe (2013b), the Apacherian-Chihuahuan Mesquite Upland Scrub (hereafter referred to as Mesquite Upland Scrub), as an invasive native shrubland, now covers some 19% of the ecoregion. The assessment for the Mesquite Upland Scrub was aimed at answering a different management question than those addressed by the ecological status assessment for the CEs. The question for Mesquite Upland Scrub was: In areas of mesquite expansion into the uplands, where might there be opportunities for restoration of grasslands that have been invaded by mesquite? Alternatively, one might ask "where might restoration efforts be worth exploring in on-the-ground surveys, given other constraints within the landscape (e.g. development, exotic plants, and soils."

For the Mesquite Upland Scrub, the conceptual model follows a slightly different format than for the CEs described in Appendices D, E, and F, as it is considered a non-natural invasive ecosystem that has resulted from changes in the landscape due to European settlement and recent impacts of drought. Hence the conceptual model contains information pertinent to understanding the change agents that have led to the prominence of Mesquite Upland Scrub in this ecoregion along with some of the variables that could be measured to answer the above question in a rapid spatial assessment.

The information developed is generally intended to cover the full range of distribution, which extends beyond the ecoregion, and but does focus on the characteristics or dynamics as they occur within this ecoregion.

The description includes names of plant species that are characteristic of the ecological system type. In the text sections these names are provided as scientific names. Vascular plant species nomenclature follows the nationally standardized list of Kartesz (1999), with very few exceptions. References are listed that are relevant to the classification, distribution, floristic composition, ecological processes, threats, stressors, or management of the ecosystem, in some cases for portions of the range outside of the ecoregion. This is not an exhaustive literature survey, rather is an accumulation of known references. Some documents may be listed that are not cited in the narrative text.

### H.2.1 Methods for Characterization of the Ecosystem

The ecosystem conservation elements for the MAR REA were selected from NatureServe's classification of terrestrial ecological systems (Comer et al. 2003). Over three dozen ecological systems occur in the MAR, but only a select subset were chosen for the REA; methods for selection are described in Harkness et al. (2013). The descriptive material in the conceptual model builds upon the descriptions for terrestrial ecological systems that NatureServe has been compiling since 2003 when the ecological systems classification was first developed (see http://www.natureserve.org/explorer/index.htm to search and download existing descriptions). For this REA, additional material was added to the description for the Mesquite Upland Scrub ecological system, especially focused on content describing natural and altered vegetation dynamics.

The first section of the conceptual model deals with the classification used, the NatureServe terrestrial ecological systems, as described above. The NatureServe name and tracking code are provided. A second part of the classification section lists the ecological systems that are similar to those in the CE. Similarity might be due to floristic, structural or geographic overlap with the CE; in some cases similar ecological systems are listed because reviewers of the draft conceptual models expressed some confusion about the MAR CE in their comments.

A crosswalk to Ecological Site Descriptions (ESDs) applicable to the ecoregion is provided (https://esis.sc.egov.usda.gov/Welcome/pgESDWelcome.aspx). The crosswalk is only to approved ESDs by NRCS Multiple Resource Land Area (MLRA) that overlap the ecoregion; draft ESDs are being developed in New Mexico that may crosswalk to the mesquite scrub, however because they are draft they are not included in the ESD crosswalk table. The NRCS Site ID in the crosswalk table identifies each type as determined by NRCS. This list is not a complete cross-walk as some MLRAs do not have approved ESDs. Additionally, the user should consider that ESDs are based on landform/soil concepts, so the match between these concepts and the ecological system concept - defined as an integration between biophysical and natural floristic composition - will be imperfect and may vary from type to type.

The vegetation composition and ecosystem dynamics are described in narrative text, with supporting literature cited. A diagrammatic representation of the natural dynamics is not provided for the mesquite scrub, as it is not a natural ecological system and its dynamics are a result of alteration of the Apacherian-Chihuahuan Semi-Desert Grassland and Steppe CE.

Some species of conservation or management concern utilize Mesquite Upland Scrub for foraging, nesting, cover, or other habitat uses. These species are of conservation or management concern due primarily to their relative vulnerability to extinction through alteration of other ecosystems in the MAR, but mesquite scrub fills a valuable habitat need for them. These vulnerabilities stem from their sensitivity to past or current land/water uses, natural rarity, or forecasted vulnerabilities to climate change effects. The species are listed to make users aware of associated species that are of concern.

**Ecological System Dynamics**

This section of the conceptual model presents a narrative description of the primary factors that have resulted in the expansion of mesquite into the uplands in the MAR. The section contains three sub-sections: (1) a description of the altered dynamics and how these may cause the continued decline or degradation of the other ecosystem CEs that have been [mostly] converted to mesquite scrub; 2) a list of the stressors that are the primary agents of these altered dynamics; and (3) state-and-transition models from two Ecological Site Descriptions (ESDs) developed by NRCS.

**Conceptual Model Diagrams**

For uplands, the dynamics are generally represented by state-and-transition diagrams. States (boxes) represent a vegetation community defined by a cover type and structural stage. Transitions link states through processes such as succession, disturbance, and management, and can be either deterministic or probabilistic. Deterministic transitions usually simulate successional changes by defining the number of years until a transition occurs from one successional stage to the next, in the absence of disturbance. Probabilistic transitions specify an annual transition probability of moving from one state to another. Probabilistic transitions represent disturbances (e.g., fire and drought), ecological processes (e.g. tree encroachment and natural recovery), and land management activities (e.g., seeding and prescribed fire).

Two conceptual state and transition models are provided that are representative of the Apacherian-Chihuahuan Mesquite Upland Scrub ecosystem.

**Factors Related to Restoration Potential**

A last section of the conceptual model discusses factors that will affect the ability of land managers to restore areas that have been degraded by mesquite invasion. Literature that was surveyed to develop this information is cited, and a table is provided that summarizes the indicators that might be used in an assessment for measuring (at a landscape-scale) the potential for successful restoration of mesquite-invaded grasslands. These factors cannot be evaluated individually, since there are interacting effects; e.g., the surrounding landscape condition may be favorable to restoration, but the individual mesquite stand may be large, dense, and have a non-native grass understory, making restoration success more difficult without large investments in effort.

## *H.3 Apacherian-Chihuahuan Mesquite Upland Scrub Conceptual Model*

The Apacherian-Chihuahuan Mesquite Upland Scrub ecosystem occurs as upland native woody increaser shrublands as a result of altered dynamics. During the last century, the area occupied by this desert thornscrub type has increased through conversion of semi-desert grasslands (Brown and Archer 1987) and oak woodlands (Turner et al. 2003). Although it is possible that this upland mesquite type may have occurred in minor amounts historically, mesquite was largely confined to mesic drainages until cattle distributed seeds upland from the bosques into grasslands and oak woodlands (Brown and Archer 1987, 1989).

For this assessment, it is considered a novel ecosystem because these *Prosopis* spp. dominated shrublands have replaced large areas of semi-desert grasslands, especially those formerly dominated by *Bouteloua eriopoda*, in Trans-Pecos Texas, southern New Mexico and southeastern Arizona (York and Dick-Peddie 1969, Hennessy et al. 1983). Studies on the Jornada Experimental Range suggest that combinations of drought, overgrazing by livestock, wind and water erosion, seed dispersal by livestock, fire suppression, shifting dunes, and changes in the seasonal distribution of precipitation have caused this recent, dramatic shift in vegetation physiognomy (Buffington and Herbel 1965, Herbel et al. 1972,

Humphrey 1974, McLaughlin and Bowers 1982, Gibbens et al. 1983, Hennessy et al. 1983, Schlesinger et al. 1990, McPherson 1995).

For the MAR REA, there is a desire to understand: a) how invasion of mesquite into the uplands is degrading other, natural ecological systems; b) what are some of the factors managers need to be aware of when attempting to restore natural ecosystems that have been mesquite-invaded; and c) generally where might restoration efforts be worth exploring in on-the-ground surveys, given other constraints within the landscape (e.g. development, exotic plants, soils, and fire regimes).

### H.3.1 Classification

The ecosystem types for the MAR REA were selected from NatureServe's classification of terrestrial ecological systems (Comer et al. 2003). Over three dozen ecological systems occur in the MAR, but only a select subset was chosen for the REA. This system is treated as an altered, non-natural concept in this conceptual model, and includes this NatureServe ecological system type:

- Apacherian-Chihuahuan Mesquite Upland Scrub (CES302.733)

There are other terrestrial ecological systems in the NatureServe classification that also occur in the MAR, or in adjacent ecoregions, which are similar to this CE concept but are not included in this concept. These are listed here to help the reader understand what is not included in this conceptual model; each of these other ecological systems has information that can be searched for and reviewed on NatureServe's on-line website (http://explorer.natureserve.org/) . The first two ecological systems listed below occur peripherally in the MAR, on the eastern edges of the ecoregion, and are extensive further east in the Chihuahuan Desert. The other three ecological systems are grassland types, that when they are degraded, may be similar to Mesquite Upland Scrub, but retain enough of the natural floristic composition and structure so as to be recognizable as degraded grasslands.

- Chihuahuan Stabilized Coppice Dune and Sand Flat Scrub (CES302.737) (mesquite scrub on dunes and sand flats)
- Chihuahuan Mixed Desert and Thornscrub (CES302.734) (might have scattered mesquite, but other desert scrub species such as *Larrea tridentata* are the dominants)
- Degraded Apacherian-Chihuahuan Semi-Desert Grassland and Steppe [CES302.735)
- Degraded Chihuahuan Loamy Plains Desert Grassland (CES302.061) (upland tobosa/blue grama)
- Degraded Chihuahuan Sandy Plains Semi-Desert Grassland [CES302.736) (black grama)

### H.3.2 Summary

This ecological system occurs as upland native woody increaser shrublands that are concentrated in the extensive semi-desert grassland in foothills and piedmonts of the Chihuahuan Desert, extending into the Sky Island region to the west (Figure H-1). Substrates are typically derived from alluvium, and are often gravelly without a well-developed argillic or calcic soil horizon that would limit infiltration and storage of winter precipitation in deeper soil layers. *Prosopis* spp. and other deep-rooted shrubs exploit this deep-soil moisture that is unavailable to grasses and cacti.

Vegetation is typically dominated by *Prosopis glandulosa* or *Prosopis velutina* and succulents. Other desert scrub species that may codominate include *Acacia neovernicosa, Acacia constricta, Juniperus monosperma*, or *Juniperus coahuilensis*. *Larrea tridentata* is typically absent or has low cover. Grass cover is typically low and composed of desert grasses such as *Dasyochloa pulchella (= Erioneuron pulchellum), Muhlenbergia porteri, Muhlenbergia setifolia*, and *Pleuraphis mutica*. During the last century, the area occupied by this system has increased through conversion of semi-desert grasslands as

a result of drought, overgrazing by livestock, and/or decreases in fire frequency. In the Madrean Encinal (oak-dominated woodlands and savannas) similar effects have occurred causing increases in woody species, changing the species composition in some areas from oak dominated woodlands or savanna to mesquite and/or juniper dominated woodlands (Turner et al. 2003).

The Mesquite Upland Scrub is similar to Chihuahuan Mixed Desert and Thornscrub (CES302.734) but is generally found at higher elevations where *Larrea tridentata* and other desert scrub species are not codominant. It is also similar to Chihuahuan Stabilized Coppice Dune and Sand Flat Scrub (CES302.737) but does not occur on aeolian-deposited substrates (sandsheets), although some stands may have evidence of wind erosion and deposition.

The description is based on several references, including Brown (1982b), Dick-Peddie (1993), Gibbens et al. (2005), MacMahon (1988), Muldavin et al. (2002), and NatureServe Explorer (2013).

**Figure H-1. Apacherian-Chihuahuan Mesquite Upland Scrub (http://www.azfirescape.org).**

A crosswalk of this system to approved Ecological Site Descriptions (ESD) by Major Land Resource Areas (MLRA) is provided in Table H-1. There are no approved State and Transition Models developed for approved ESDs in NM that are strongly related to this ecosystem type. However, of the 7 approved NM ESDs, two sites include mesquite in the plant community description and list it as an increaser species; those 2 NM sites are provided below. Altered states of these ESDs with higher densities of mesquite would be included in the Mesquite Upland Scrub ecosystem. The last ESD listed, F041XC310AZ, under

the Historic Climax Plant Community (HCPC) is a large tree mesquite type (occurring in riparian bottomlands), but one altered state is the "mesquite, scrubland" which occurs when the water table is lowered via human development/water pumping. Some areas mapped as Mesquite Upland Scrub in the MAR may well be this altered state of the loamy bottom 12-16" ESD. (For a complete list of ESDs for MLRA 41 see https://esis.sc.egov.usda.gov/Welcome/pgReportLocation.aspx?type=ESD).

**Table H-1. Apacherian-Chihuahuan Mesquite Upland Scrub ecosystem CE crosswalk with approved Ecological Site Descriptions.**

| MLRA | Ecological Site Description Name | Site ID |
|---|---|---|
| 041-Southeastern Arizona Basin and Range | Loamy Upland 8-12" p.z. / *Prosopis glandulosa var. torreyana - Ephedra fasciculata / Pleuraphis mutica – Aristida ( /* honey mesquite - desert Mormon tea / tobosa - Aristida) | R041XB210AZ |
| 041-Southeastern Arizona Basin and Range | Sandy Loam Upland 8-12" p.z. / *Prosopis glandulosa / Yucca elata -Eephedra fasciculata / Muhlenbergia porteri - Bouteloua eriopoda* ( / honey mesquite / soaptree yucca - desert Mormon tea / bush muhly - black grama) | R041XB215AZ |
| 041-Southeastern Arizona Basin and Range | Loamy Bottom 12-16" *Populus fremontii - Juglans nigra / Sporobolus wrightii - Panicum obtusum* (cottonwood - black walnut / giant sacaton - vine-mesquite) | R041XA006NM |
| 041-Southeastern Arizona Basin and Range | Clay Loam Upland 12-16" | R041XA002NM |
| 041-Southeastern Arizona Basin and Range | Loamy Bottom 12-16" p.z. *Prosopis glandulosa var. torreyana / / Sporobolus wrightii* (velvet mesquite - western honey mesquite / / big sacaton) | F041XC310AZ |

### H.3.3 Species and Mesquite

There are many common animal species that utilize Mesquite Upland Scrub as habitat. Honey mesquite (*Prosopis glandulosa*) and velvet mesquite (*P. velutina*) seeds are nutritionally rich and are important food for a large number of wildlife species (Graham 1941, Tull 1987, Steinberg 2001). Honey mesquite seeds form an important part of the diet of mice, kangaroo rats, woodrats, chipmunks, ground squirrels, rock squirrels, cottontail, skunks, quail, doves, ravens, the black-tailed prairie dog, black-tailed jackrabbit, porcupine, raccoon, coyote, collared peccary, javelina, white-tailed deer, mule deer, wild turkey, and mallard (many citations in Steinberg 2001). On the Jornada Experimental Range species of small rodents frequently store whole beans of western honey mesquite in dens or caches, and honey mesquite beans formed the bulk of stored food (Wood 1969). Mesquite flowers are eaten by numerous bird species, and often comprise 10 to 25% of the Gambel's and scaled quails' diets (Davis et al. 1975, DeLoach et al. 1986). Different birds also nest in the tree's canopy. In a southwestern Texas study, honey mesquite fruit comprised 14.9% of the white-tailed deer summer diet, but deer use of any honey mesquite parts during the rest of the year was minimal (Varner and Blankenship 1987). It is an important "honey plant" and bees that forage its flowers produce excellent quality honey (Dayton 1931). It provides a good source of nectar and food for butterfly adults and larvae (Taylor et al. 1997).

Some species of conservation or management concern are associated with this ecological system, and may utilize it for some portions of their life cycle (nesting, foraging, cover, burrows). These species are of conservation or management concern due primarily to their relative vulnerability to extinction through alteration of other ecosystems but Mesquite Upland Scrub may replace lost habitat in some cases. These vulnerabilities stem from their sensitivity to past or current land/water uses, natural rarity, or forecasted vulnerabilities to climate change effects. Although some of the species listed below may be assessed individually (see separate conceptual models for them), most are listed to make users aware of associated species that are of concern.

There is little information in the published literature about sensitive species utilization of Mesquite Upland Scrub, outside of those found in riparian mesquite bosques. One can assume that less-dense

mesquite shrublands, especially those with remaining native grasses, may support some species that are associated with healthy grasslands, especially if the mesquite patches are smaller and are in the vicinity of natural systems. Listed below are selected species of conservation or management concern that are associated with healthy grasslands from the BLM Gila District (USDI-BLM 2010). Ffolliott (1999b) mentioned neotropical birds in general and specifically listed game species important to mesquite ecosystems.

**Birds:** Ferruginous Hawk (*Buteo regalis*) (breeding population only), Mourning dove (*Zenaidura macroura),* Northern Aplomado Falcon (*Falco femoralis septentrionalis*), White-winged dove (*Zenaidura astiatica),* scaled quail (*Callipepla squamata*)

**Mammals:** Banner-tailed Kangaroo Rat (*Dipodomys spectabilis*), Black-tailed Prairie Dog (*Cynomys ludovicianus*), Collared Peccary (*Tayassu tajacu*), Mule deer (*Odocoileus hemionus),* White-tailed deer *(Odocoileus virginianus)*

**Reptiles:** Slevin's Bunchgrass Lizard (*Sceloporus slevini*)

## H.3.4 Ecological System Dynamics

This section of the conceptual model presents a narrative description of the primary factors that have resulted in the expansion of mesquite into the uplands in the MAR. The section contains two sub-sections: (1) a description of the altered dynamics and how these may cause the continued decline or degradation of the other ecosystem CEs that have been [mostly] converted to mesquite scrub; 2) a list of the stressors that are the primary agents of these altered dynamics; and (3) state-and-transition models from two Ecological Site Descriptions (ESDs) developed by NRCS.

For description of this type before conversion to mesquite dominated shrubland refer to the Historic Climax Plant Community (HCPC) and the Reference State section of the below ESD models (Figure H-2). Also refer to the conceptual models for the Apacherian-Chihuahuan Semi-Desert Grassland and Steppe CE and the Madrean Encinal CE contained in Appendix D.

### H.3.4.1 Upland Mesquite Dynamics

The Apacherian-Chihuahuan Mesquite Upland Scrub ecosystem occurs as upland native woody increaser shrublands. During the last century, the area occupied by this system has increased through conversion of semi-desert grasslands as a result of drought, overgrazing and *Prosopis* spp. seed dispersion by livestock, and/or decreases in fire frequency (Buffington and Herbel 1965, Brown and Archer 1987). Season of precipitation is a key environmental variable with periods of strong summer precipitation promoting grasses and periods of summer drought favoring shrub dominance (Burgess 1995, Van Devender 1995). Shrubs such as mesquite and creosote bush have invaded semi-desert grasslands of this region three times in the last 4,000 years (Van Devender 1995). The first two cycles were driven by long-term drought, but the current shrub increase beginning in the 1880s was intensified by human disturbance and cattle (Bahre and Hutchinson 2001) during drought periods. Prior to the most recent invasion, mesquite dominated shrublands rarely occurred in uplands and were largely confined to mesic drainages until cattle distributed seeds upland from the bosques into grasslands (Brown and Archer 1987, 1989). Therefore Apacherian-Chihuahuan Mesquite Upland Scrub is considered a novel ecosystem.

Gori and Enquist (2003) estimate that 84% of the historical (pre-1880s) extent of semi-desert grasslands have some degree of shrub invasion, and 37% has been completely converted to a shrub-dominated system (mesquite or creosote bush). This Mesquite Upland Scrub type is currently estimated to occupy approximately 20% of the ecoregion, based on vegetation mapping by NatureServe (2013). Just east of

the MAR on the Jornada Range, mesquite (*Prosopis* spp.), tarbush (*Flourensia cernua*) and creosote bush occupied approximately 42% of the Jornada Range in the 1850s; by the early 1960s, these shrub species were found through the entirety of the Range (Buffington and Herbel 1965).

Historically in grassland natural-ignition fires were relatively small, probably 10-15 acres in size. Repeated fire is thought to help maintain a general mosaic pattern between open grassland and shrub-dominated areas (Johnston 1963). Wright et al. (1976) found that *Prosopis glandulosa* is very fire-tolerant when only 3 years old. Most plants resprout after being top-killed by fire. Thus, prior to fire and/or drought influences reducing fire frequency, repeated grassland fires probably maintained lower stature of shrubs and prevented new establishment by killing seedlings.

Drought is a relatively common occurrence in this ecoregion, generally occurring every 10-15 years and lasting 2-3 years with occasional long-term drought periods (10-15 years duration). *Prosopis* spp. and other shrubs have extensive root systems that allow them to exploit deep-soil water that is unavailable to shallower rooted grasses and cacti (Burgess 1995). This strategy works well, especially during drought. However, on sites that have well-developed argillic or calcic soil horizons that limit infiltration and storage of winter moisture in the deeper soil layers, *Prosopis* spp. invasion can be limited to a few, small individuals (McAuliffe 1995). This has implications for semi-desert grassland restoration work in the southwestern United States. For example, degraded grasslands on these sites with well-developed argillic or calcic soil horizons could be prioritized for restoration.

#### H.3.4.2 Stressors on Ecological Dynamics Causing Continued Degradation or Change

This novel mesquite-dominated upland shrubland ecological system is the result of several interacting change agents which affected primarily the Apacherian-Chihuahuan Semi-Desert Grassland and Steppe and Madrean Encinal CEs, resulting in degradation and conversion to the Apacherian-Chihuahuan Mesquite Upland Scrub ecosystem. Restoration of these upland mesquite-dominated scrublands will be directly affected by continuous heavy grazing by livestock, direct and indirect wildfire suppression, land development, and non-native plant species invasion. Changes in long-term climate regime or short-term weather patterns are also factors that will affect restoration potential, but are factors that cannot easily be controlled for in restoration efforts. Table H-2 identifies the most likely impacts associated with each of these stressors on ecological dynamics and hence potential for restoration.

**Table H-2. Stressors that are likely to affect the successful restoration or management of Apacherian-Chihuahuan Mesquite Upland Scrub in the Madrean Archipelago ecoregion.**

| Stressor | Impacts on ecological dynamics and restoration potential |
|---|---|
| **Land Use** | |
| Livestock grazing | Grazing of native vegetation by livestock at inappropriate stocking rates, season of use, or duration can be detrimental to grass vigor resulting in decline of grass cover and shifts in species composition to more grazing tolerant or less palatable species (Milchunas 2006). Over time this often results in increased woody cover or bare ground and erosion. Livestock grazing can affect also soil structure and water infiltration, and species diversity (USDA-USFS 2009). Heavy grazing can indirectly decrease fire return intervals by removing fine fuels that carry fire (Swetnam and Baisan 1996). Livestock feed on *Prosopis* spp. seeds and are a major source of dispersal (Brown and Archer 1987). |

| Stressor | Impacts on ecological dynamics and restoration potential |
|---|---|
| Recreation | This mostly relates to off road vehicle use, which creates additional roads and trails that fragment natural ecosystem patches and contribute to increases in soil erosion and compaction and non-native species dispersal (USDA-USFS 2009). |
| **Development** | |
| **Linear Features**<br>Transportation infrastructure<br>Roadways/railways and transmission lines | Fragmentation from transportation infrastructure leads to disruptions in ecological processes such as fire, dispersal of invasive non-native species, and can alter hydrological processes by changing surface flows such as when excessive runoff from roads creates gullies that can lower water tables. Additionally, destruction of wildlife habitat and disruption of wildlife migration patterns can also occur (Bahre 1991, Bock and Bock 2002, Finch 2004, Heinz Center 2011, Marshall et al. 2004, McPherson 1997, Ockenfels et al. 1994, Schussman 2006b). |
| **Site**<br>Suburban/Rural (include Military), Mines/Landfill, Energy (Renewable wind/solar), Oil/Gas | This stress contributes to altered fire regimes (e.g. fire suppression to protect infrastructure), increased erosion, direct habitat loss/conversion, increased groundwater pumping, fragmentation, invasive non-native species dispersal and disruption of wildlife migration patterns (Bahre 1991, Finch 2004, McPherson 1997). |
| **Uncharacteristic Fire Regime** | Fire suppression, both active and passive with livestock removing fine fuels that carry fire, has contributed to the expansion of mesquite dominated shrublands into former semi-desert grasslands and encinal woodlands (Buffington and Herbel 1965, Herbel et al. 1972, Humphrey 1974, McLaughlin and Bowers 1982, Gibbens et al. 1983, Hennessy et al. 1983, Schlesinger et al. 1990, McPherson 1995). The Apacherian-Chihuahuan Mesquite Upland Scrub is also the result, in part, in a change in fire regime from frequent surface fires common in semi-desert grasslands (FRI of 2.5-10 years) (Brown and Archer 1999, Gori and Enquist 2003, McPherson 1995, Robinett 1994, Wright 1980); this results in increasing shrub cover and contributes to the conversion from perennial grasslands to this mesquite dominated desert scrub. |
| **Invasive non-native Species** | Replacement of native vegetation with non-native grass species such as *Eragrostis lehmanniana* and *Eragrostis curvula*. These species are better adapted to frequent fire and increase in relative abundance over native grasses after burning (Anable et al. 1992, Cable 1971, Gori and Enquist 2003, Schussman 2006a). The impact of invasive non-native species on community function of native vegetation is well documented (Anable et al. 1992, Cable 1971, Cox et al. 1988). |

| Stressor | Impacts on ecological dynamics and restoration potential |
|---|---|
| **Soil Erosion** | The condition of the soil/surface substrate directly affects the functioning of natural ecosystems. Studies on the Jornada Experimental Range suggest that wind and water erosion in combination with the other stressors listed here, have caused this recent, dramatic shift in vegetation physiognomy resulting in mesquite dominance (Buffington and Herbel 1965, Herbel et al. 1972, Humphrey 1974, Gibbens et al. 1983, Hennessy et al. 1983, Schlesinger et al. 1990). Loss of ground cover (both vascular and nonvascular plants), livestock trampling, recreational vehicles, and runoff from adjacent developed areas can directly affect soil properties by disturbing soil crusts, compacting pore space that reduces water infiltration and percolation, changing other structural characteristics, and can expose soils to increased erosional forces, leading to degradation of sites. |
| **Climate change** | Alteration of precipitation, evapotranspiration rates, and timing (season) of precipitation, may result in more frequent drought periods and higher intensity precipitation events, which following drought can cause significant erosion of topsoil. Predicted increases in temperatures and effective precipitation will impact restoration efforts. |

#### H.3.4.3 Dynamics Models

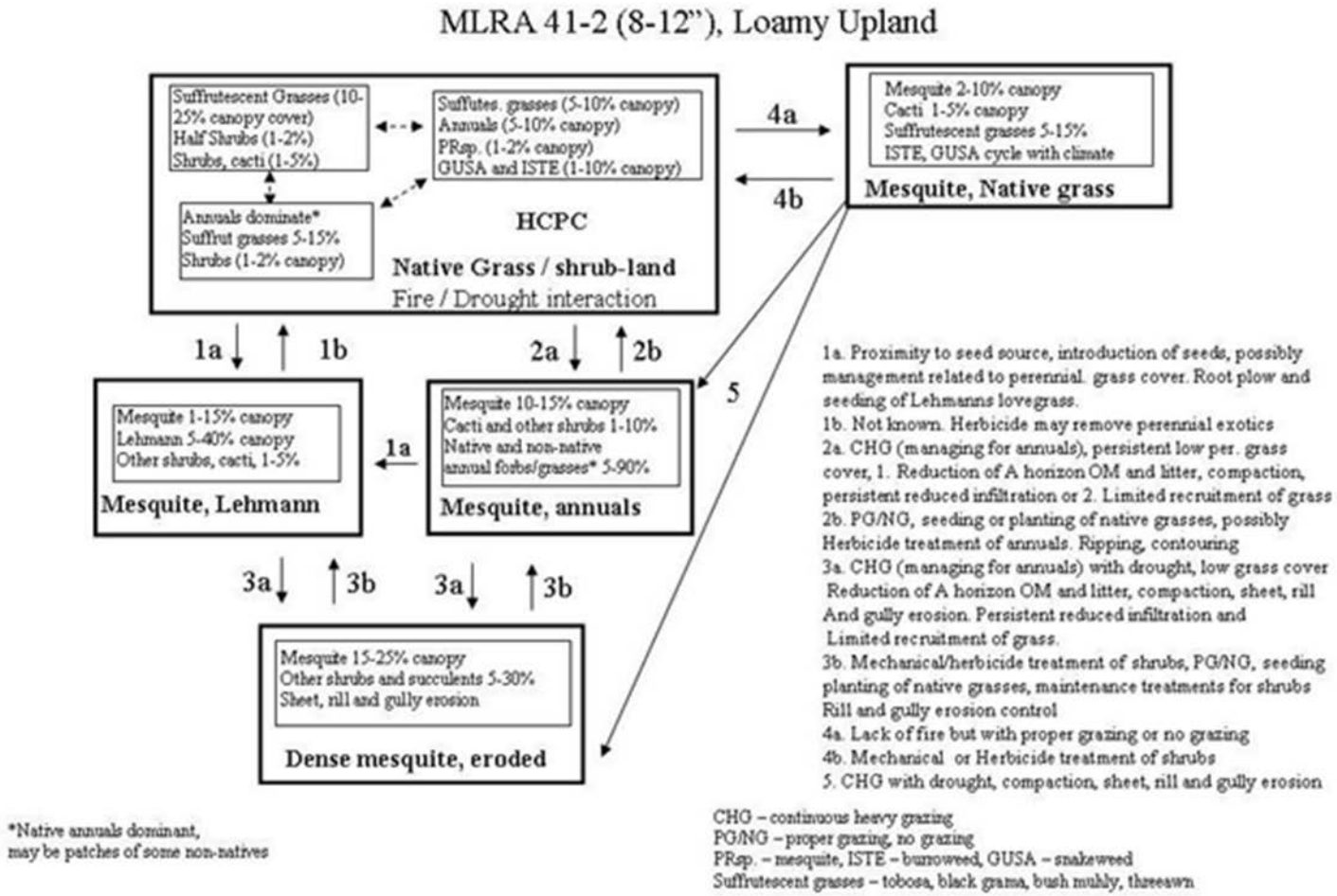
Below are two conceptual state and transition models that are representative of the Apacherian-Chihuahuan Mesquite Upland Scrub ecosystem CA. The model in Figure H-2 (for ESD R041XA210AZ) was developed by the NRCS for southeastern Arizona. This model has four disturbed states: Mesquite, Native grass; Mesquite, Lehmann; Mesquite, annuals; and Dense mesquite, eroded. The undisturbed Historic Climax Plant Community (HCPC) state or reference state that relates directly to the Apacherian-Chihuahuan Semi-Desert Grassland and Steppe CE is part of the model.

The model in Figure H-3 (for ESD R041XA210AZ) was also developed by the NRCS for southeastern Arizona and is an alternative model that is representative of the Apacherian-Chihuahuan Mesquite Upland Scrub ecosystem, as an altered state of the Apacherian-Chihuahuan Semi-Desert Grassland and Steppe CE. The model has three disturbed states: an Exotic Grasses, a Mesquite-Juniper Invaded, and an Eroded Surface. The shrub dominated state relates to the Apacherian-Chihuahuan Mesquite Upland Scrub ecosystem.

For each of these ESD models, the text describing the altered states is provided, along with the ESD state-and-transition models. In general, the transitions from one state to another are influenced by the amount and timing of grazing, the proximity to a seed source for mesquite, the cyclical occurrence of drought, and the fire regime. Mechanical treatments, herbicide applications, or other efforts to manage or control mesquite or invasive exotics can reverse mesquite invasion and degradation of the site, but compaction or disruption of soils by either livestock or equipment can lead to erosion and gullying, and loss of the native perennial grasses.

Additional work on Ecological Site Descriptions is being completed by NRCS and BLM in New Mexico, but are currently in draft form and not available to provide in this conceptual model.

**Figure H-2. Conceptual state and transition model of the Historic Climax Plant Community (HCPC) and altered dynamics for NRCS ESD R041XA210AZ.** This model is representative of the Apacherian-Chihuahuan Mesquite Upland Scrub ecosystem CE. This model is excerpted from the Ecological Site Description (ESD) for R041XA210AZ from the 041-Southeastern Arizona Basin and Range MLRA at: https://esis.sc.egov.usda.gov/Welcome/pgReportLocation.aspx?type=ESD.



**Text directly from ESD R041XA210AZ (Figure H-2):**

**Description of State and Transition Model**

The HCPC portion of this model represents this ecosystem under natural dynamic conditions, a semi-desert grassland that lacks the upland mesquite, a native woody increaser. The Altered Dynamic portions of this community are shown with arrows indicating invasion by mesquite with native grass understory; invasion by mesquite with introduction of non-native forage grasses such as *Eragrostis Lehmanniana* or *Eragrostis curvula*; invasion by mesquite with dominance of non-native and native annual grasses; and an eroded surface with low grass cover (including reduction or loss of A soil horizon, reduced soil infiltration, soil organic material, ground cover, litter, and increased soil compaction, sheet and rill erosion). Descriptions of altered states are excerpted from Ecological Site Description (ESD) for Loamy Upland 8-12" p.z. / *Prosopis glandulosa var. torreyana - Ephedra fasciculata / Pleuraphis mutica – Aristida* below:

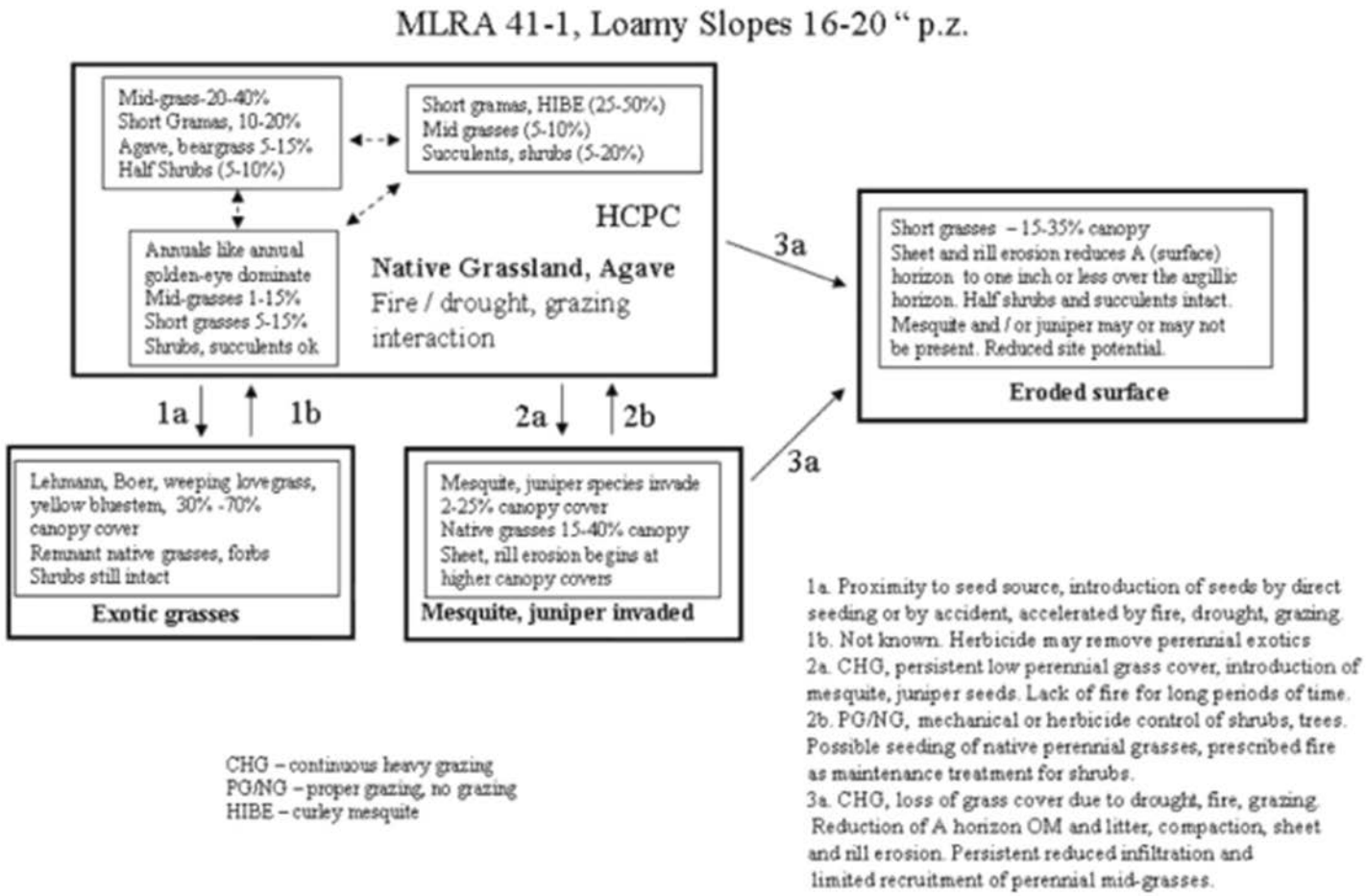
Mesquite, natives
This state occurs where mesquite has increased from between 2 and 10% canopy cover and some cover of native perennial (suffrutescent) grasses and forbs remains. Other shrubs and succulents exist in minor amounts. Annual forbs and grasses (both native and non-native) are very important in their respective (wet) seasons.

Mesquite, Lehmann lovegrass
This state occurs where Lehmann, and in some cases Boers, lovegrass has been seeded; usually in combination with mechanical mesquite control. The cover of Lehmann lovegrass varies widely with climate, ranging from 1-5% canopy in dry years up to 20-40% canopy in years with wet summers. Lehmann never dominates the plant community on this site but does dominate the herbaceous layer of the plant community once established.

Mesquite, annuals
This state occurs where mesquite and other shrubs (creosotebush) and cacti dominate the plant community. Native perennial grasses and forbs have been removed from the plant community and native and non-native annual species dominate the herbaceous layer.

Mesquite, Erosion
This state occurs where mesquite canopy is heavy (15-25%) and the interaction of drought and continuous grazing has resulted in severe sheet, rill and, in some cases, gully erosion on the site. These areas are usually near historic watering locations and are characterized by soil compaction due to trailing and heavy livestock traffic."

**Figure H-3. Conceptual state and transition model of the Historic Climax Plant Community (HCPC) and altered dynamics for NRCS ESD R041XA107AZ.** This model is representative of the Apacherian-Chihuahuan Semi-Desert Grassland and Steppe CE or semi-desert mixed grassland vegetation type. It includes a shrub-invaded state, where either mesquite or juniper have become dominant. This model is excerpted from Ecological Site Description (ESD) for R041XA107AZ Loamy Slopes 16-20" p.z. / *Agave palmeri - Nolina microcarpa / Bouteloua curtipendula - Eragrostis intermedia* from the 041-Southeastern Arizona Basin and Range MLRA at: https://esis.sc.egov.usda.gov/Welcome/pgReportLocation.aspx?type=ESD.



## Text directly from ESD R041XA107AZ (Figure H-3):

**Description of State and Transition Model**

The HCPC portion of this model represents the Apacherian-Chihuahuan Semi-Desert Grassland and Steppe CE ecosystem under natural dynamic conditions. The Altered Dynamics portions of this community are shown with arrows indicating introduction of non-native forage grasses such as *Eragrostis Lehmanniana* or *E. curvula*; invasion by shrubs and small trees (primarily species of *Prosopis* and *Juniperus*) resulting from extended periods of lack of fire; and an eroded surface with low grass cover (including reduction or loss of A soil horizon, reduced soil infiltration, soil organic material, ground cover, litter, and increased soil compaction, sheet and rill erosion).

Descriptions of altered states are excerpted from Ecological Site Description (ESD) for R041XA107AZ below:

"Exotic grasses

This state occurs where non-native lovegrass species or yellow bluestem, have invaded from adjacent areas or roads and ROWs with a seed source. As these species increase to dominate the plant community, native perennial grasses and forbs decrease to remnant amounts. Fire will usually act to increase species like Lehmann lovegrass. The native half shrubs seem to be able to stay in the plant community. It is not known how Agave Palmeri fares under this condition.

Shrub invaded

This state occurs where mesquite, wait a bit mimosa, one-seed juniper and / or alligator juniper have invaded or increased to dominate the plant community. This occurs in the absence of fire for long periods of time, with continuous grazing and in the presence of a seed source of these species. As canopy levels of trees and shrubs approach 30%, sheet and rill erosion can begin to accelerate.

Eroded surface

This state occurs where severe soil compaction and trailing has resulted in loss of plant cover and an increase in runoff. Sheet and rill erosion accelerates and the surface (A) horizon is removed faster than it can be replaced by down-slope soil movement and weathering of the ridgetops. When the subsurface argillic (clayey) horizons are exposed, the site has lost its potential productivity. The plant community will shift from warm season plants to cool season plants and the ratio of runoff to infiltration will increase.

With continuous, heavy grazing, mid-grasses are removed from the plant community and replaced by short grasses such as curly mesquite, slender grama and sprucetop grama. With severe deterioration, shrubby species such as wait-a-bit mimosa, one-seed and alligator juniper, and mesquite can increase to dominate the site. With good management, native mid-grasses will be able to regain their dominance in the plant community, unless soil erosion is severe enough to strip away the surface horizon. Mesquite and Lehmann lovegrass are at the upper limits of their elevation range, but can increase on the site, especially below 5000 feet elevation and on southern exposures. Climatic warming may allow these two species to push higher in elevation as time goes by. Naturally occurring fires in June-August were an important factor in shaping this plant community. Fire-free intervals range from 10-20 years. Without disturbance like grazing or fire, perennial mid-grasses can become decadent and forbs like annual goldeneye, cudweed and camphorweed can increase to dominate the plant community. This site is the principal habitat for the Agave Palmeri in southeastern Arizona, an important food source for the endangered lesser long-nosed bat in June, July, and August. Dense stands of this species occur scattered throughout areas of this site. Nectar production in these stands ranges from 6-10 gallons per acre.

Periodic drought can occur in this LRA and cause significant grass mortality. Droughts in the early 30s, mid 50s, 1975-1976, 88-89, 95-96 and 2002 resulted in the loss of much of the grass cover on this site. The site recovers rapidly, however, due to excellent covers of stone, cobbles and gravel and the favorable climate that prevails in this common resource area."

### H.3.5 Key Factors Related to Restoration Potential

Several interacting factors will affect the ability of land managers to restore areas that have been degraded by mesquite invasion. The models presented above from the NRCS ESDs and the work done by Gori and Enquist (2003) and Gori et al. (2012) to document the historic range of variability of the semi-desert grasslands in this region both provide much useful information. McPherson and Weltzin (2000) provide an overview of the interactions of disturbance, both natural and human-caused, and climate change in the region containing the MAR, and how these have resulted in shifts in plant community composition and physiognomy (e.g. shifts from grasslands to shrublands). In their report, they suggest that prairie dogs probably historically had a role in keeping shrubs, such as mesquite, from dominating areas; they do acknowledge that current scientific understanding of this is poor, and suggest a fruitful area of research would be on the effects of native herbivores on establishment and persistence of woody plants.

Wilson et al. (2001) discuss the expansion of mesquite in this region within the context of the natural history of mesquite species, focusing on the attributes that allowed this expansion. They also provide a useful summary of research that has addressed this expansion, primarily from the decade prior to the report, and a summary of mesquite management practices (such as herbicide application, prescribed burning, and mechanical removal) in the context of several objectives.

Below the information is summarized (Table H-3), and related to the potential for restoration at three levels (high, moderate, low or no potential). Thresholds were adapted from Apache Highland Grassland Assessment condition classes in the project area by Gori et al. (2012) and NRCS ESDs R041XA210AZ. These factors cannot be evaluated individually, since there are interacting effects; e.g., the surrounding landscape condition may be favorable to restoration, but the individual mesquite stand may be large, dense, and have a non-native grass understory, making restoration success more difficult without large investments in effort.

**Table H-3. Key factors for determining the potential for restoration of places dominated by the Apacherian-Chihuahuan Mesquite Upland Scrub ecosystem.**

| Factor | Key indicators of mesquite sites with high potential for restoration to semi-desert grassland | Key indicators of mesquite sites with moderate potential for restoration to semi-desert grassland | Key indicators of mesquite sites with low or no potential for restoration to semi-desert grassland |
|---|---|---|---|
| **Site description** | Open mesquite shrublands (generally <15% cover) with a moderately dense (10-25% cover) native grass layer. | Open to moderately dense (15% to 25% cover) mesquite shrublands with sparse (<10% cover) native grass layer. | Dense mesquite shrublands (>45%) or mesquite shrublands (>25%) with non-native grass layer or bare eroded substrate. |
| **Landscape Context** | Site is large, on public land; or in an area with private partnerships conducive to restoration activities. Fragmentation level (density of roads, transmission lines, urban and exurban development) is | Site is relatively large, on mixture of public and privately owned land. Fragmentation level (density of roads, transmission lines, urban and exurban development) is low | Site is relatively small in a mixture of public and privately owned land. Fragmentation level (density of roads, transmission lines, urban and exurban development) is high. |

| Factor | Key indicators of mesquite sites with high potential for restoration to semi-desert grassland | Key indicators of mesquite sites with moderate potential for restoration to semi-desert grassland | Key indicators of mesquite sites with low or no potential for restoration to semi-desert grassland |
|---|---|---|---|
| | low | | |
| **Livestock Grazing** | No livestock use or at low stocking rate so as not to impact restoration; livestock management allows adjustment of seasonality and stocking rate. | No livestock use or at low stocking rate so as not to impact restoration | Livestock use at moderate to high stocking rate |
| **Vegetation Composition** | Herbaceous layer has moderate to dense cover and is dominated by native perennial grasses. Invasive, non-native grasses like Lehmann lovegrass are absent or very low cover (<1%) | Herbaceous layer has low to moderate cover and dominated by native perennial grasses. Invasive, non-native grasses like Lehmann lovegrass are absent or have low cover (<3%) | Herbaceous layer is sparse and composed of native grasses or herbaceous layer has moderate to high cover of invasive, non-native grasses such as Lehmann lovegrass. |
| **Vegetation Structure** | Open mesquite shrublands with medium woody cover (10-35% total, with cover of mesquite or juniper <15%). Other shrubs can contribute to the total. Perennial native grass cover is >5%, preferably at least 10% cover. | Former grassland and savanna with high woody plant cover (15-25% cover mesquite and juniper combined or 35-45% total woody cover); perennial grass cover <10% but generally <5% and usually <3%). | Shrub cover is dense (>45% total, or with cover of mesquite or juniper >25%). Herbaceous layer is sparse – fine fuels are too low to carry fire; or if grass cover is high, composition is entirely non-natives. |
| **Soil Condition** | Soils well developed, often gravelly surface to limit erosion and with well-developed argillic or calcic soil horizon that would limit infiltration and storage of winter precipitation in deeper soil layers that *Prosopis* spp. and other deep-rooted shrubs can exploit. | Soils well developed enough to support moderately dense grasslands. | Soils shallow, poorly developed/skeletal, sand deposit or top soil is eroded to <10 cm deep. |
| **Fire Regime** | Fire regime restorable with prescribed fire; preferred return intervals of 2-5 yrs. | Fire regime restorable with prescribed fire | Fire regime is severely altered and not restorable |

## *H.4 Assessment Methods*

This assessment seeks to identify areas of the MAR, currently invaded by mesquite in upland settings, where land managers might be able to remove or control mesquite and restore natural grasslands. In order to not limit the assessment to areas now dominated by mesquite (i.e. to identify other places with restoration potential), and to account for possible mapping errors in the land cover data, the inputs to this assessment included the historic distributions of both grasslands and encinal, as well as the current distribution of the Mesquite Upland Scrub.

A “mesquite distribution” map was created by combining the distributions of Apacherian-Chihuahuan Mesquite Upland Scrub (from the NatureServe (2013b) map of ecological systems), the historic distribution for Apacherian-Chihuahuan Semi-desert Grassland and Steppe and Madrean Encinal (from LANDFIRE BpS), and the historic distribution of all grasslands as well as degraded grasslands from the TNC Grassland Assessment (Gori et al. 2012).

The distribution map was used as an input to the assessment in Vista, along with the scenario of landscape condition - development used for the terrestrial ecosystem conservation elements (see Appendices B and C for details about this scenario), data for the percent cover of mesquite (also from ILAP), and data for soil types (SSURGO and STASGO data compiled by the Integrated Landscape Assessment Project, ILAP). Processing of each input for use in the assessment is described below.

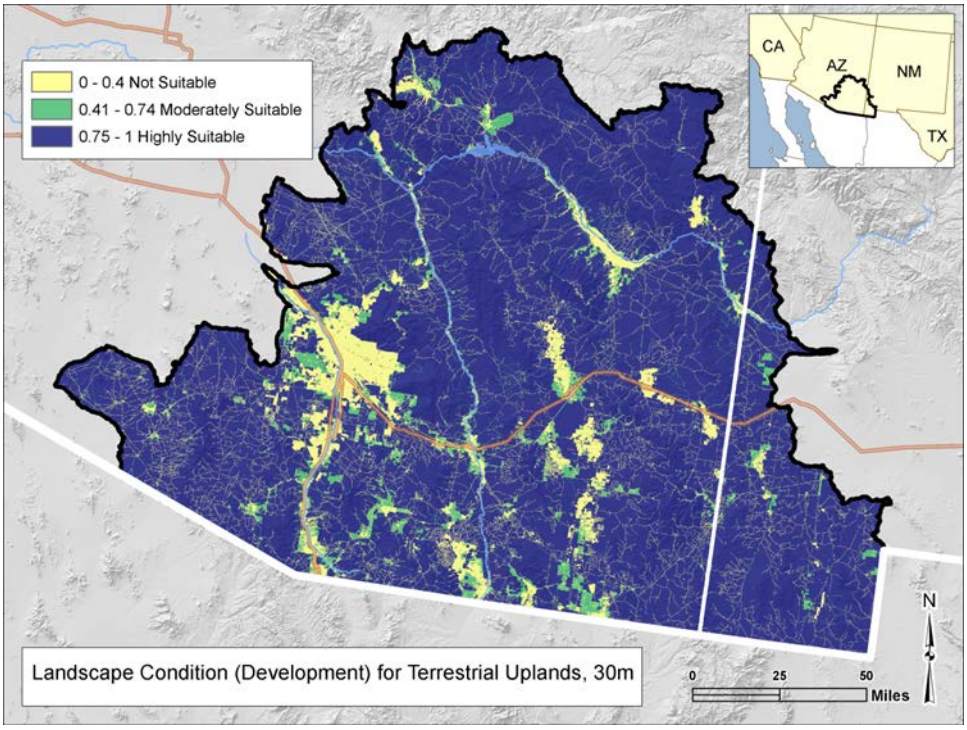
For the development scenario, the response model output for the terrestrial CEs was used as one of the inputs. That response model used site intensity and distance values for each component of development (see Appendix D for a list of those inputs and values), but the output from that response model was then broken into three categories of suitability for this assessment (Table H-4).

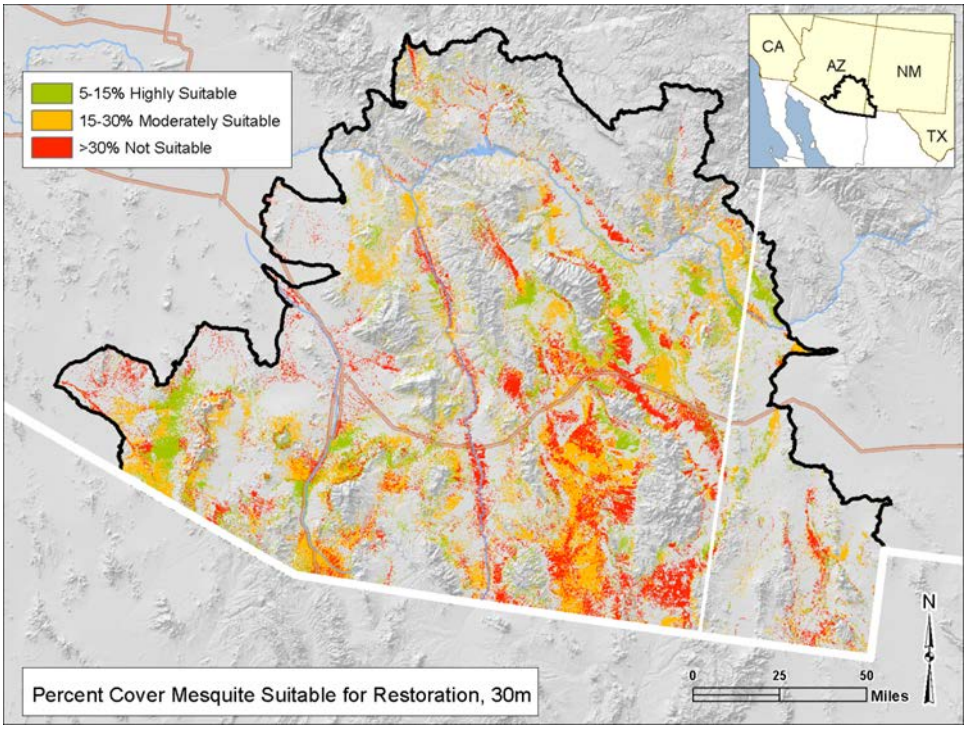
For the percent cover of mesquite- the continuous cover from 0 to over 90% was broken into the same three categories of suitability (Table H-4). These cover breaks are the same as those used to evaluate degradation of grasslands by mesquite and juniper invasion discussed in the conceptual model (Gori et al. 2012; Table H-3). Twenty-five percent cover was identified in Table H-3 as the break between somewhat suitable and not suitable; however, for this assessment the break of 30% is used in order to be more inclusive of possibly suitable areas. The results of these categorizations for development and mesquite % cover are shown in Figure H-4 (development, upper; mesquite cover, lower). Areas with 0-5% cover of mesquite were not considered in the evaluation; in effect this means those areas are highly suitable from the standpoint of mesquite cover.

**Table H-4. The two variables of development and mesquite cover, and their classification into three categories of suitability for restoration of mesquite invaded uplands.**

| Suitability Category | Development Response Model | Mesquite Cover | Comments Relevant to Mesquite Cover |
|---|---|---|---|
| Highly Suitable | .75 to 1 | 5-15% | Includes areas with no mesquite cover |
| Moderately Suitable | .41 to .74 | 15-30% | Even with up to 30% mesquite, the site could still have a grass layer. |
| Not Suitable | 0 to .4 | >30% | With >30% cover - grass layer lost due to shading from scrub |

**Figure H-4. Maps of two of the variables used to determine areas suitable for restoration of mesquite invaded uplands: development (upper), mesquite cover (lower)**. In each map, the variable has been categorized into three levels of suitability: not, moderately, or highly. These maps are intermediate products in the assessment process and were not delivered to BLM.



The soils of the MAR were the third variable selected for use in this assessment, since soil characteristics influence the types of vegetation that can grow. Some soils have higher potential for restoration success than others. A soils dataset was available from ILAP that compiled all SSURGO polygons and then filled in the missing areas with STATSGO data. This combined dataset was then attributed by ILAP with a small subset of the available data on soils from either SSURGO or STATSGO. This ILAP dataset included the soil taxonomy along with a few variables of soil texture and depth. Soil taxonomy is a good way to evaluate soil properties that effect restoration potential. The Great Group level with 40 soil types for the MAR was determined to be the most appropriate scale for ecoregion-wide assessment. Using the Great Group classification level, unsuitable soils such as Natraquolls within the highly suitable soil orders such as Mollisols, can be identified. Great Groups were evaluated using guidance from soils classification references such as Buol et al. (1980) and USDA- Soil Survey Staff (1999). As with the other 2 variables, soil Great Groups were classified as Highly Suitable, Moderately Suitable and Not Suitable for restoration of mesquite invaded uplands. Table H-5 provides a list of the Great Groups in each category of suitability, along with an explanation; and Figure H-5 shows the results of mapping each Great Group soil polygon into one of the three categories.

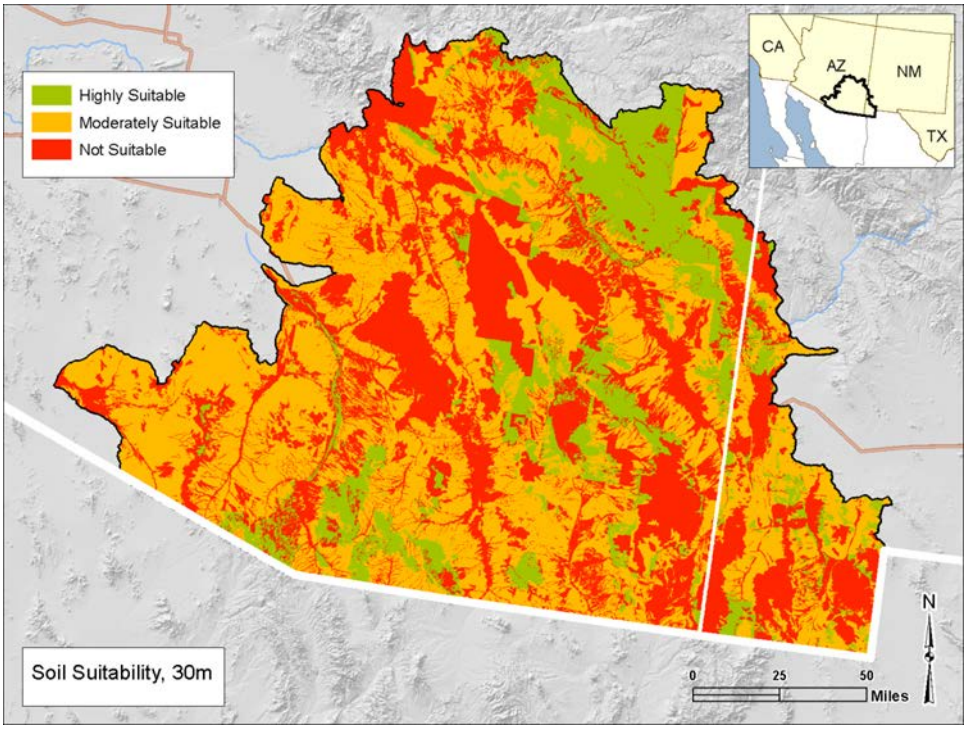
Generally the most suitable soils are Mollisols and selected Aridisols with impermeable subhorizons of clay or calcium carbonate that may restrict water percolation and root penetration of deep rooted shrubs such as mesquite and so favor relatively shallow-rooted grasses (McAuliffe 1995). Moderately suitable soils are most of the remaining Aridisols and Alfisols. Unsuitable soils were generally poorly developed Entisols and Inceptisols, heavy clay Vertisols, and other soils with Gypsic or Natric subhorizons.

**Table H-5. Soil Great Groups found in the MAR, and their classification into categories of suitability for grassland restoration .** Explanations are provided for each suitability rating.

| Soil Order | Great Group | Explanation of Suitability |
|---|---|---|
| **Highly Suitable** | | |
| Mollisols | Argiustolls | Mollisols are highly suitable because they developed under grassland vegetation and are relatively deep and fertile with a significant organic component (Mollic epipedon) that should facilitate grassland restoration. |
| | Calciustolls | |
| | Haplustolls | |
| | Paleustolls | |
| Aridisols | Calciorthids | Aridisols developed under dry conditions and some may be highly suitable especially those with impermeable subhorizons of clay or calcium carbonate that may restrict water percolation and root penetration of deep rooted shrubs such as mesquite and so favor relatively shallow-rooted grasses (McAuliffe 1995). |
| | Durargids | |
| | Durorthids | |
| **Moderately Suitable** | | |
| Aridisols | Argidurids | Aridisols developed under dry conditions; although some may be highly suitable, most are moderately suitable for grassland restoration. Aridisols often have impermeable subhorizons of clay or calcium carbonate that may restrict water percolation and root penetration of deep rooted shrubs such as mesquite and so favor relatively shallow-rooted grasses (McAuliffe 1995). |
| | Calciargids | |
| | Camborthids | |
| | Haplargids | |
| | Haplocalcids | |
| | Haplocambids | |
| | Haplodurids | |
| | Paleargids | |

| Soil Order | Great Group | Explanation of Suitability |
|---|---|---|
| | Petrocalcids | |
| Alfisols | Glossudalfs | Alfisols are moderately suitable for grasslands, except for types with a Natric (sodium) subhorizon. Alfisols have an Argillic (clay) subhorizon. Alfisols typically develop under forests, but are often used for crops so they are assumed to be ok for grassland restoration. They are a minor type in the MAR. |
| | Haplustalfs | |
| | Ochraqualfs | |
| | Paleustalfs | |
| **Not Suitable** | | |
| Aridisols | Calcigypsids | This Aridisol is unsuitable because of a Gypsic horizon |
| Alfisols & Aridisols | Natrustalfs | These Alfisols and Ardisols are not suitable for grassland restoration because of the Natric (sodium) subhorizon. |
| | Nadurargids | |
| | Natrargids | |
| | Paleorthids | |
| Entisols & Inceptisols | Torrifluvents | Inceptisols and Entisols are poorly developed soils. Inceptisols often occur on geomorphic surfaces or parent materials that limit soil development; substrates such as steep slopes, sand, ash, saturated soils that would not be suitable for grassland restoration. Entisols are too recently formed to have significant development such as soils from mass wasting sites (deposition), too cold or too warm for soil development (desert/alpine), fluvial deposition or eolian deposition, weather-resistant bedrock (quartize). These conditions lead to infertility and toxicity, not suitable for grassland restoration. Additionally, some of these soils have Gypsic or Natric horizons, Psamm (sand) texture, or are Torric (hot, dry) |
| | Torriorthents | |
| | Torripsamments | |
| | Ustifluvents | |
| | Ustorthents | |
| | Calciustepts | |
| | Dystrudepts | |
| | Haplustepts | |
| Mollisols | Natraquolls | This Mollisol is unsuitable because of Natric (sodium) subhorizon and possible high wetness (aqu). |
| Vertisols | Calcitorrerts | Vertisols are unsuitable. They are deep, heavy, dark, clay soils composed of shrink-swell clays that crack deeply when dry and would be difficult to restore grasslands on. In addition, some of these soils have Gypsic horizons or are Torric (hot, dry) |
| | Calciusterts | |
| | Chromusterts | |
| | Gypsitorrerts | |
| | Haplotorrerts | |
| | Torrerts | |

**Figure H-5. Map of results of categorizing the soils of MAR into areas suitable for restoration of mesquite invaded uplands**. The categories of suitability in this map correspond to the list of soil Great Groups in each category in Table H-5. This map is an intermediate product in the assessment process and was not delivered to BLM.




Once the categories of restoration suitability for each of the three variables were determined, the spatial data were combined into a single scenario of 30m resolution data, for evaluation in Vista. A categorical response model was used, wherein any pixel deemed not suitable for one variable was assigned not suitable (

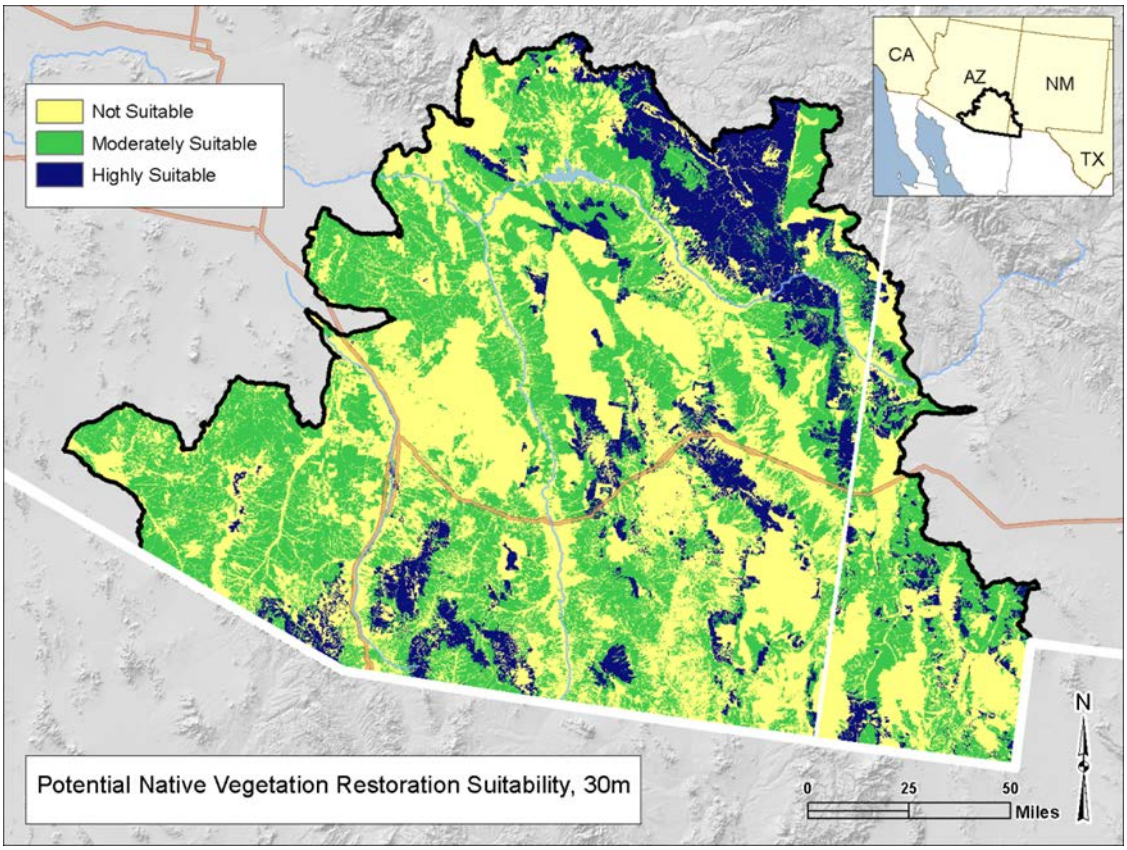
Table H-6) even if the other 2 variables were suitable in that pixel. In sequence, any pixel with a moderately suitable value would override highly suitable in the other 2 variables. Hence, the only pixels assessed as highly suitable are those where all 3 variables were highly suitable (Table H-6). In the case of areas with 0-5% mesquite cover, the suitability categories of the development and soils were the only inputs considered; in effect this means that 0-5% cover of mesquite is suitable or not depending upon the development or soils suitability. A map of the resulting suitability matrix is displayed in

Figure H-6 for the entire MAR area; the actual results are clipped to the distribution dataset described earlier in these methods and are shown below in the results.

**Table H-6. Decision matrix used in the Vista evaluation for categories of suitability for each of the three variables.**

| Development / Soils Suitability | Mesquite Cover Highly Suitable (5-15%) | Mesquite Cover Moderately Suitable (15-30%) | Mesquite Cover Not Suitable (>30%) |
|---|---|---|---|
| Development **or** Soils Not Suitable | Not Suitable | Not Suitable | Not Suitable |
| One Moderately Suitable, the other Highly Suitable | Moderately Suitable | Moderately Suitable | Not Suitable |
| Development **and** Soils Highly Suitable | Highly Suitable | Moderately Suitable | Not Suitable |

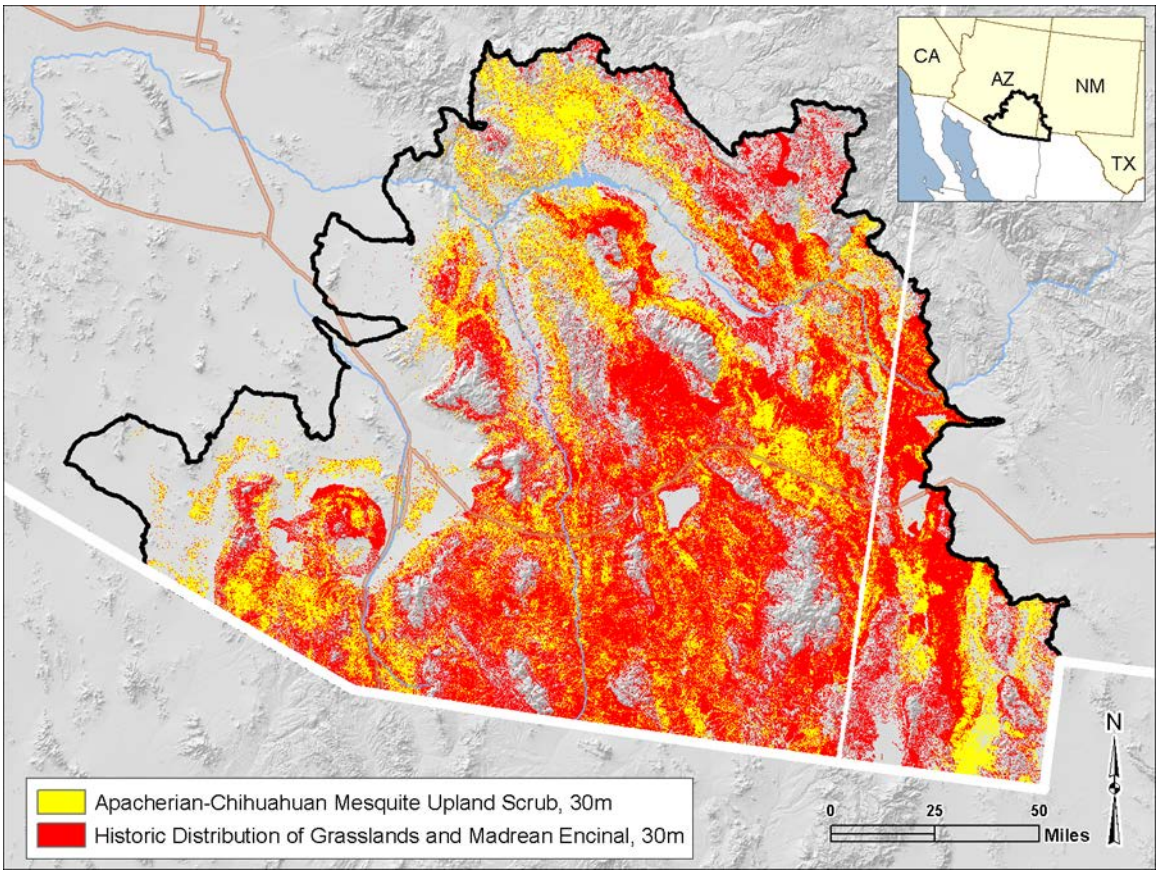
**Figure H-6. Map of suitability that incorporates all three variables (development, mesquite cover, and soils) using the decision matrix found in Table H-6.** This map is an intermediate product in the assessment process and was not delivered to BLM.



## *H.5 Assessment Results*

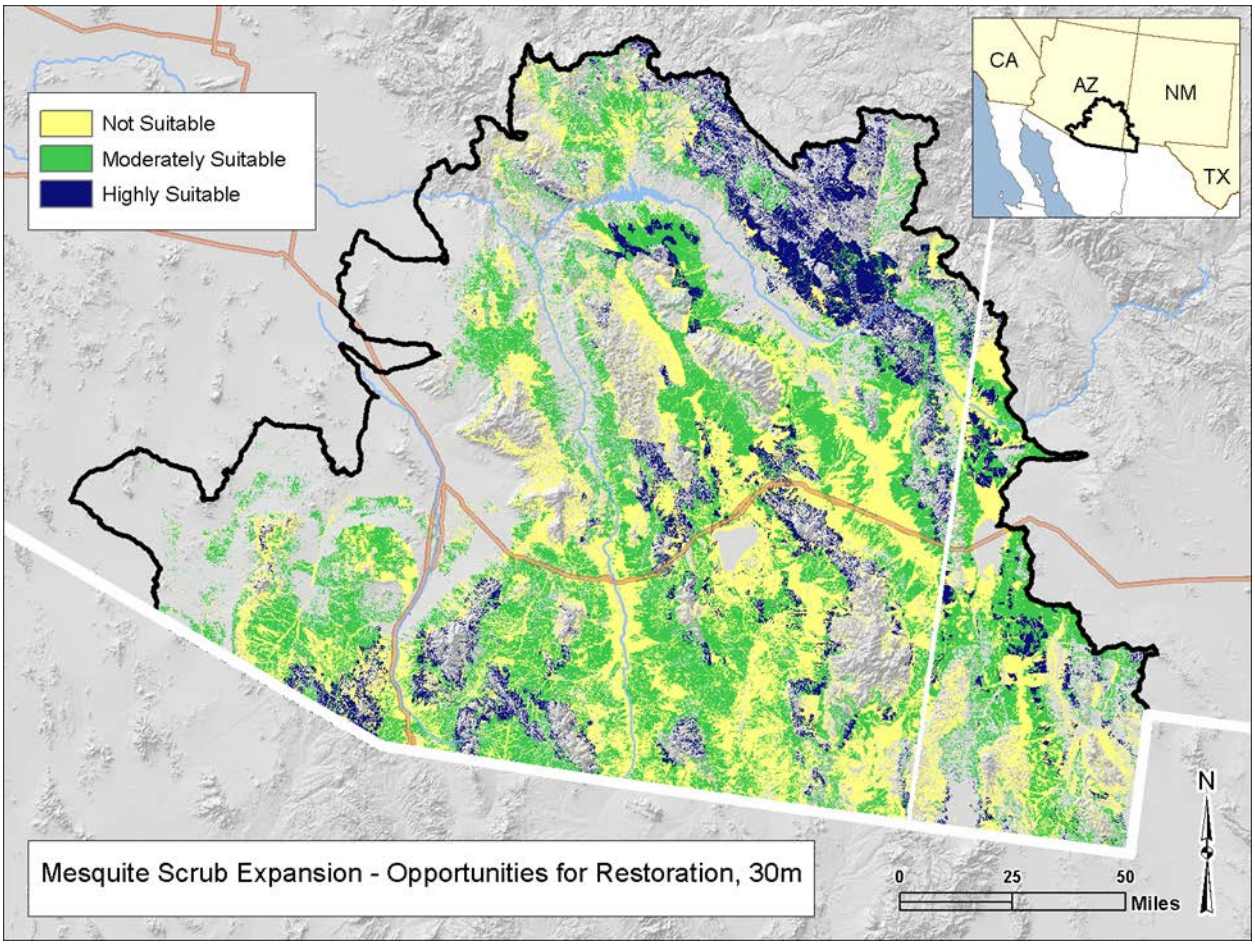
As described above, the area of distribution used in this assessment (Figure H-7) was created by combining the distributions of Apacherian-Chihuahuan Mesquite Upland Scrub (from the NatureServe (2013) map of ecological systems), the historic distribution for Apacherian-Chihuahuan Semi-desert Grassland & Madrean Encinal (from LANDFIRE BpS), and the historic distribution of all grasslands as well as degraded grasslands from the TNC Grassland Assessment (Gori et al. 2012).

**Figure H-7. Map of the current distribution of Apacherian-Chihuahuan Mesquite Upland Scrub (yellow) combined with the distribution of degraded grasslands (red), and the historical distribution of Apacherian-Chihuahuan Semi-desert Grassland and Steppe & Madrean Encinal** (also red) for purposes of assessing possible areas for restoration of mesquite invaded uplands. The combined distribution dataset was delivered to BLM, but not with the attributes of the source dataset (some pixels had overlapping source data, e.g. mapped as mesquite and historic grassland in the same pixel). This map is to help visualize the different source data.



The results of the Mesquite Scrub Expansion: Restoration Opportunities assessment are shown in Figure H-8. The results are a combination of scores from three variables: development, percent cover of mesquite, and the suitability of soils for grassland restoration. The Highly Suitable areas for restoration (blue) mostly occur in the northern portion of the ecoregion around the Gila Mountains and in the Natanes Plateau. Highly suitable areas to the south occur in smaller patches and are restricted to the foothills, piedmont and alluvial plains around the higher elevation desert mountain ranges such as the Pajarito Mountains, Santa Rita Mountains, Patagonia Mountains, Canelo Hills, and Galiuro Mountains. These are areas where grasslands historically occurred on highly suitable Mollisol soils and likely still have some of the better condition remaining grasslands. Moderately suitable areas are found in the middle elevation plateaus, plains and valleys between some of the ranges. The not suitable areas occur in transportation corridors such as from Tucson to Nogales, near some of the smaller urban areas, and in agricultural areas such as the San Simon Valley, Sulphur Springs Valley and Gila Valley below Safford.

**Figure H-8. Final results showing areas identified for suitability of restoration of mesquite-invaded uplands to grasslands or other native ecosystems.** This map combines three variables: development, percent cover of mesquite, and the suitability of soils for grassland restoration. Yellow represents not suitable areas (due to intense development pressure, high cover of mesquite or unsuitable soils).



## *H.6 References*


Anable, M. E., M. P. McClaran, and G. B. Ruhle. 1992. Spread of introduced Lehmann lovegrass *Eragrostis lehmanniana* Nees in Southern Arizona, USA. Biological Conservation 61:181-188.

Bahre, C. J. 1991. A Legacy of Change: Human Impacts on Vegetation in the Arizona Borderlands. University of Arizona Press, Tucson, AZ.

Bahre, C. J. and C. F. Hutchins. 2001. Historic vegetation change in *La Frontera* west of the Rio Grande. Pp. 67-83 *in* G. L. Webster and C. J. Bahre (eds.). 2001, Vegetation and Flora of *La Frontera*: Vegetation Change along the United States-Mexican Boundary.

Brown, D. E., editor. 1982. Biotic communities of the American Southwest-United States and Mexico. Desert Plants Special Issue 4(1-4):1-342.

Brown, J. R., and S. Archer. 1987. Woody plant seed dispersal and gap formation in a North American subtropical savanna woodland: The role of domestic herbivores. Vegetatio 73:73-80.

Brown, J. R., and S. Archer. 1989. Woody plant invasion of grasslands: Establishment of honey mesquite (*Prosopis glandulosa* var. *glandulosa*) on sites differing in herbaceous biomass and grazing history. Oecologia 80:19-26.

Brown, J. R. and S. Archer. 1999. Shrub invasion of grassland: Recruitment is continuous and not regulated by herbaceous biomass or density. *Ecology*, 80, 2386-2396.

Buffington, L. C., and C. H. Herbel. 1965. Vegetational changes on a semidesert grassland range from 1858 to 1963. Ecological Monographs 35(2):139-164.

Buol, S. W., F. D. Hole, and R. J. McCracken. 1980. Soil genesis and classification. Iowa State University, Ames. 404 pp.

Burgess, T. L. 1995. Desert grassland, mixed shrub savanna, shrub steppe, or semidesert scrub. Pages 31-67 in: M. P. McClaran and T. R. Van Devender, editors. The Desert Grassland. University of Arizona Press, Tucson.

Cable, D. R. 1971. Lehmann lovegrass on the Santa Rita Experimental Range, 1937-1968. Journal of Range Management. 24: 17-21.

Comer, P., D. Faber-Langendoen, R. Evans, S. Gawler, C. Josse, G. Kittel, S. Menard, M. Pyne, M. Reid, K. Schulz, K. Snow, and J. Teague. 2003. Ecological systems of the United States: A working classification of U.S. terrestrial systems. NatureServe, Arlington, VA.

Cox, J. R., G. B. Ruyle, J. H. Fourle, and C. Donaldson. 1988. Lehmann lovegrass--central South Africa and Arizona, USA. Rangelands. 10(2): 53-55.

Davis, C.A., R.C. Barkley, and W.C. Haussamen. 1975. Scaled quail foods in southeastern New Mexico. Journal of Wildlife Management 39(3): 496-502.

Dayton, W.A. 1931. Important western browse plants. U.S. Department of Agriculture. Misc. Publ. 101. Washington, DC. 214 p.

DeLoach, C. J., P.E. Boldt, H.A. Cjordo, [and others]. 1986. Weeds common to Mexican and U.S. rangelands: proposals for biological control and ecological studies. Pages 49-68 In: Patton, D.R., V. Gonzales, E. Carlos, A.L. Medina, [and others], technical coordinators. Management and utilization

of arid land plants: Symposium proceedings; 1985 February 18-22; Saltillo, Mexico. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station Gen. Tech. Rep. RM-135. Fort Collins, CO.

Dick-Peddie, W. A. 1993. New Mexico vegetation: Past, present, and future. University of New Mexico Press, Albuquerque. 244 pp.

Finch, D. M., editor. 2004. Assessment of Grassland Ecosystem Conditions in the Southwestern United States; Volumes 1 and 2. USDA Forest Service Gen. Tech. Rpt. RMRS-GTR-135. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Ffolliott, P. F. 1999b. Mesquite Ecosystems in the Southwestern United States. Chapter 8. Pages 95-106 in: P. F. Ffolliott and A. Ortega-Rubio, editors. Ecology and Management of Forests, Woodlands, and Shrublands in Dryland Regions of the United States and Mexico: Perspectives for the 21st Century. Co-edition number 1. University of Arizona-Centro de Investigacione.

Gibbens, R. P., J. M. Tromble, J. T. Hennessy, and M. Cardenas. 1983. Soil movement in mesquite dunelands and former grasslands of southern New Mexico. Journal of Range Management 36(2):145-148.

Gibbens, R. P., R. P. McNeely, K. M. Havstad, R. F. Beck, and B. Nolen. 2005. Vegetation change in the Jornada Basin from 1858 to 1998. Journal of Arid Environments 61(4):651-668.

Goodwin, J.G. and C.R. Hungerford. 1977. Habitat use by native Gambel's and scaled quail and released masked bobwhite quail in southern Arizona. Research Paper RM-197. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO. 8 p.

Gori, D. F. and C. A. F. Enquist. 2003. An assessment of the spatial extent and condition of grasslands in central and southern Arizona, southwestern New Mexico and northern Mexico. The Nature Conservancy, Arizona Chapter. Pp29.

Gori, D., G. S. Bodner, K. Sartor, P. Warren, and S. Bassett. 2012. Sky Island Grassland Assessment: Identifying and Evaluating Priority Grassland Landscapes for Conservation and Restoration in the Borderlands. Report prepared by The Nature Conservancy in New Mexico and Arizona. 85 p.

Graham, E.H. 1941. Legumes for erosion control and wildlife. U.S. Department of Agriculture Misc. Pub. 412. Washington D.C. 153 p.

Heinz Center. 2011b. Managing and monitoring Arizona's wildlife in an era of climate change: Strategies and tools for success. Report and Workshop Summary, The H. John Heinz III Center for Science, Economics and the Environment, Washington, D.C. 67pp + appendices.

Hennessy, J. T., R. P. Gibbens, J. M. Tromble, and M. Cardenas. 1983. Vegetation changes from 1935 to 1980 in mesquite dunelands and former grasslands of southern New Mexico. Journal of Range Management 36(3):370-374.

Herbel, C. H., F. N. Ares, and R. Wright. 1972. Drought effects on a semidesert grassland range. Ecology 53:1084-1093.

Humphrey, R. R. 1974. Fire in the deserts and desert grassland of North America. Pages 365-400 in: T. T. Kozlowski and C. E. Ahlgren, editors. Fire and Ecosystems. Academic Press, New York.Johnston, M. C. 1963. Past and present grasslands of southern Texas and northeastern Mexico. Ecology 44:456-464.

Kingsolver, J. M., C.D. Johnson, S.R. Swier, and A. Teran. 1977. Prosopis fruits as a resource for invertebrates. Pages 108-122 in: Simpson, B. B., ed. Mesquite: Its biology in two desert ecosystems. US/IBP Synthesis Series 4. Dowden, Hutchinson & Ross, Inc., Stroudsburg, PA.

MacMahon, J. A. 1988. Warm deserts. Pages 232-264 in: M. G. Barbour and W. D. Billings, editors. North American terrestrial vegetation. Cambridge University Press, New York.

Marshall, R.M., D. Turner, A. Gondor, D. Gori, C. Enquist, G. Luna, R. Paredes Aguilar, S. Anderson, S. Schwartz, C. Watts, E. Lopez, and P. Comer. 2004. An ecological analysis of conservation priorities in the Apache Highlands Ecoregion. Prepared by The Nature Conservancy of Arizona, Instituto del Medio Ambiente y el Desarrollo Sustentable del Estado de Sonora, Agency and Institutional partners. 152 pp.

McAuliffe, J. R. 1995. Landscape evolution, soil formation, and Arizona's desert grasslands. Pages 100-129 in: M. P. McClaran and T. R. Van Devender, editors. The Desert Grassland. University of Arizona Press, Tucson.

McLaughlin, S. P., and J. E. Bowers. 1982. Effects of wildfire on a Sonoran Desert plant community. Ecology 63(1):246-248.

McPherson, G. R. 1997. Ecology and management of North American Savannas. University of Arizona Press. Tuscon, AZ

McPherson, G. R. 1995. The role of fire in the desert grasslands. Pages 130-151 in: M. P. McClaran and T. R. Van Devender, editors. The Desert Grassland. University of Arizona Press, Tucson.

McPherson, G. R., and J.F. Weltzin. 2000. Disturbance and climate changes in United States/Mexico borderland plant communities: A state-of-the-knowledge review. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station General Technical Report, RMRS-GTR-50. Fort Collins, CO. 24 p.

Milchunas, D.G. 2006. Responses of plant communities to grazing in the southwestern United States. Gen. Tech. Rep. RMRS-GTR-169. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 126 p.

Muldavin E., G. Bell, et al. 2002. Draft ecoregional conservation assessment of the Chihuahuan Desert. Pronatura Noreste. 87 pp.

NatureServe. 2013a. International Ecological Classification Standard: International Vegetation Classification. Central Databases. NatureServe, Arlington, VA.

NatureServe. 2013b. Terrestrial Ecological Systems of the Conterminous United States. Version 2.9. Completed in cooperation with USGS Gap Analysis Program and inter-agency LANDFIRE. Reflecting early 2000s land cover. MMU approx. 2 hectares. NatureServe, Arlington, VA. Digital map.

Ockenfels, R. A., A. Alexander, C. L .D. Ticer, and W. K. Carrel. 1994. Home ranges, movement patterns, and habitat selection of pronghorn in central Arizona. Arizona Game and Fish Department Technical Report Number 13. Phoenix, Arizona

Parmenter, R. R., and T. R. Van Devender. 1995. Diversity, spatial variability, and functional roles of vertebrates in the desert grassland. *In*: M. P. McClaran and T. R. Van Devender (eds.). The Desert Grassland. University of Arizona Press, Tucson, Arizona. p. 196-229.

Pickett, S.T.A., and J.N. Thompson. 1978. Patch dynamics and the design of nature reserves. Biol. Conserv. 13:27-37.

Robinett, D. 1994. Fire effects on southeastern Arizona plains grasslands. *Rangelands*, 16, 143-148.

Schlesinger, W. H., J. F. Reynolds, G. L. Cunningham, L. F. Huenneke, W. M. Jarrell, R. A. Virginia, and W. G. Whitford. 1990. Biological feedbacks in global desertification. Science 247:1043-1048.

Schussman, H. 2006a. Historical Range of Variation and State and Transition Modeling of Historical and Current Landscape Conditions for Semi-Desert Grassland of the Southwestern U.S. Prepared for the U.S.D.A. Forest Service, Southwestern Region by The Nature Conservancy, Tucson, AZ. 53 pp.

Schussman, H. 2006b. Historical Range of Variation for Madrean Encinal of the Southwestern U.S. Prepared for the U.S.D.A. Forest Service, Southwestern Region by The Nature Conservancy, Tucson, AZ. 16 pp.

Shiflet, T. N., editor. 1994. Rangeland cover types of the United States. Society for Range Management. Denver, CO. 152 pp.

Stubbendieck, J., S.L. Hatch, and C.H. Butterfield. 1993. North American Range Plants. 4th ed. Univ. of Nebraska Press, Lincoln, NB. 493 p.

Steinberg, P. 2001. *Prosopis glandulosa*. 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/ [ 2013, December 16].

Swetnam, T.W. and C.H. Baisan. 1996. Fire histories of montane forests in the Madrean borderlands. Effects of fire on Madrean Province ecosystems: A symposium proceedings. RM-GTP-289. December 1996. USDA Forest Service.

Taylor, R.B., J. Zrutledge, and J.G. Herrera. 1997. A field guide to common south Texas shrubs. Texas Parks and Wildlife Press, Austin, TX. p. 106.

Tull, D. 1987. Edible and Useful Plants of Texas and the Southwest. University of Texas Press. Austin, TX. 518 p.

Turner, R.M., R.H. Webb, J.E. Bowers, and J.R Hastings. 2003. The changing mile revisited: An ecological study of vegetation change with time in the lower mile of an arid and semiarid region. University of Arizona Press, Tucson, Arizona.

Unnasch, R.S., D. P. Braun, P. J. Comer, G. E. Eckert. 2008. The Ecological Integrity Assessment Framework: A Framework for Assessing the Ecological Integrity of Biological and Ecological Resources of the National Park System**.** Report to the National Park Service. 46 pp.

USDA- Soil Survey Staff. 1999. Soil taxonomy: A basic system of soil classification for making and interpreting soil surveys. Second edition. Agriculture Handbook Number 436. USDA Natural Resources Conservation Service, Washington, DC. Assessed on line July 2014: http://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/nrcs142p2_051232.pdf

USDA-USFS. 2009. Ecological sustainability report. Coronado National Forest. United States Department of Agriculture. Forest Service. Southwest Region. February 2009. Pp. 118.

Van Devender, T. R. 1995. Desert grassland history. Pages 68-99 in: M. P. McClaran and T. R. Van Devender, editors. The Desert Grassland. University of Arizona Press, Tucson.

Varner, L.W. and L.H. Blankenship. 1987. Southern Texas shrubs--nutritive value and utilization by herbivores. Pages 108-112 In: Provenza, F.D., J.T. Flinders, and E.D. McArthur, compilers. Proceedings--symposium on plant-herbivore interactions; 1985 August 7-9; Snowbird, UT. U.S.

Department of Agriculture, Forest Service, Gen. Tech. Rep. INT-222. Intermountain Research Station, Ogden, UT.

Whitford, W. G., G. S. Forbes, and G. I. Kerley. 1995. Diversity, spatial variability, and functional roles of invertebrates in desert grassland ecosystems. Pages 151-195 in: M. P. McClaran and T. R. Van Devender, editors. The Desert Grassland. University of Arizona Press, Tucson.

Wilson, T.B., R.H. Webb, and T.L. Thompson. 2001. Mechanisms of range expansion and removal of mesquite in desert grasslands of the Southwestern United States. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station General Technical Report, RMRS-GTR-81. Ogden, UT. 23 p.

Wood, J. E. 1969. Rodent populations and their impact on desert rangelands. Bulletin 555. Las Cruces, NM: New Mexico State University, Agricultural Experiment Station. 17 p.

Wright, H. A., S. C. Bunting, and L. F. Neuenschwander. 1976. Effect of fire on honey mesquite. Journal of Range Management 29(6):467-471.

Wright, H. A. The role and use of fire in the semidesert grass-shrub type. 1980. Ogden, UT, U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station.

York, J. C., and W. A. Dick-Peddie. 1969. Vegetation changes in southern New Mexico during the past hundred years. Pages 157-166 in: W. O. McGinnies and B. J. Goldman, editors. Arid lands in perspective. University of Arizona Press, Tucson.

# Appendix I Climate Change: Assessment Methodology and Results

Version December 8, 2014

# Contents


Appendix I Climate Change: Assessment Methodology and Results ..... 1
I.1 Overview of Appendix I ..... 6
I.2 Methods Summary ..... 6
I.2.1 Source Data ..... 6
I.2.2 Recent Climate Trends ..... 6
I.2.3 Future Climate Trends ..... 7
I.3 In-Depth Methodology for Climate Trends ..... 9
I.4 Results: Recent Climate Change ..... 13
I.4.1 Recent Decades versus Baseline ..... 13
I.4.1.1 Precipitation ..... 13
I.4.1.2 Minimum Temperature ..... 15
I.4.1.3 Maximum Temperature ..... 17
I.4.1.4 Climate Change Exposure Index ..... 19
I.4.2 Trends within Recent Decades ..... 20
I.4.2.1 Precipitation ..... 20
I.4.2.2 Minimum Temperature ..... 21
I.4.2.3 Maximum Temperature ..... 22
I.5 Results: Projected Future Climate Change ..... 23
I.5.1 Core Climate Variables ..... 23
I.5.1.1 Precipitation ..... 23
I.5.1.2 Minimum Temperature ..... 25
I.5.1.3 Maximum Temperature ..... 27
I.5.2 Climate Change Exposure Index ..... 29
I.5.3 Derived Climate Variables ..... 30
I.5.3.1 Climatic Moisture Deficit ..... 30
I.5.3.2 Number of Frost-Free Days ..... 32
I.5.3.3 Frost-Free Period ..... 34
I.6 Bioclimatic Envelope Modeling ..... 35
I.6.1 Results ..... 36
I.6.1.1 Apacherian-Chihuahuan Semi-desert Grassland and Steppe ..... 36
I.6.1.2 Chihuahuan Creosotebush Desert Scrub ..... 40

I.6.1.3 Madrean Encinal ..... 43
I.6.1.4 Apacherian-Chihuahuan Mesquite Upland Scrub ..... 46
I.7 Future Climate Change Exposure Index Overlay with CEs ..... 49
I.8 References ..... 68

# Tables

Table I-1. Global climate models used for each variable. ..... 8
Table I-2. Analysis of 5 highest variable contributions to the Apacherian-Chihuahuan Semi-desert Grassland and Steppe bioclimatic envelope model. ..... 39
Table I-3. Analysis of 5 highest variable contributions to the Chihuahuan Creosotebush Desert Scrub bioclimatic envelope model. ..... 42
Table I-4. Analysis of 5 highest variable contributions to the Madrean Encinal bioclimatic envelope model. ..... 45
Table I-5. Analysis of 5 highest variable contributions to the Apacherian-Chihuahuan Mesquite Upland Scrub envelope model. ..... 48

# Figures

Figure I-1. Visualization of a climate delta, for a single pixel for a single variable and month. ..... 10
Figure I-2. Visualization of a climate anomaly or z-score, for a single pixel for a single variable-month. ..... 10
Figure I-3. Visualization of standard Euclidian distance (SED) for a single pixel. ..... 11
Figure I-4. Visualization of Theil-Sen trend for a single pixel for a single variable-month. ..... 12
Figure I-5. MAR monthly total precipitation deltas, 1901-1980 vs. 1981-2012. ..... 13
Figure I-6. MAR monthly total precipitation delta ratios, 1901-1980 vs. 1981-2012. ..... 14
Figure I-7. MAR monthly total precipitation anomalies, 1901-1980 vs. 1981-2012. ..... 14
Figure I-8. MAR monthy average minimum temperature deltas, 1901-1980 vs. 1981-2012. ..... 15
Figure I-9. MAR monthly minimum temperature anomalies, 1901-1980 vs. 1981-2012. ..... 16
Figure I-10. MAR monthly maximum temperature deltas, 1901-1980 vs. 1981-2012. ..... 17
Figure I-11. MAR monthly maximum temperature anomalies, 1901-1980 vs. 1981-2012. ..... 18
Figure I-12. MAR current climate change exposure index. ..... 19
Figure I-13. MAR monthly precipitation trend magnitude and significance, 1981-2012. ..... 20
Figure I-14. MAR monthly precipitation trend ratio magnitude and significance, 1981-2012. ..... 21
Figure I-15. MAR monthly minimum temperature trend magnitude and significance, 1981-2012. ..... 21
Figure I-16. MAR monthly maximum temperature trend magnitude and significance, 1981-2012. ..... 22
Figure I-17. MAR projected monthly total precipitation deltas, 1901-1980 vs. 2040-2069. ..... 24
Figure I-18. MAR projected monthly total precipitation delta ratios, 1901-1980 vs. 2040-2069. ..... 24
Figure I-19. MAR projected monthly total precipitation anomalies, 1901-1980 vs. 2040-2069. ..... 25
Figure I-20. MAR projected monthy average minimum temperature deltas, 1901-1980 vs. 2040-2069. ..... 26
Figure I-21. MAR projected monthly minimum temperature anomalies, 1901-1980 vs. 2040-2069. 26

Figure I-22. MAR projected monthly average maximum temperature deltas, 1901-1980 vs. 2040-2069. ..... 27
Figure I-23. MAR projected monthly average maximum temperature anomalies, 1901-1980 vs2040-2069. ..... 28
Figure I-24. MAR projected future climate change exposure index, 1901-1980 vs. 2040-2069. ..... 29
Figure I-25. MAR projected monthly climatic moisture deficit deltas, 1901-1980 vs. 2040-2069. ..... 30
Figure I-26. MAR projected monthly climatic moisture deficit delta ratios, 1901-1980 vs. 2040-2069. ..... 31
Figure I-27. MAR projected monthly climatic moisture deficit anomalies, 1901-1980 vs. 2040-2069. ..... 31
Figure I-28. MAR projected monthy number of frost-free days deltas, 1901-1980 vs. 2040-2069. .. 32
Figure I-29. MAR projected monthly number of frost-free days anomalies, 1901-1980 vs. 2040-2069. ..... 33
Figure I-30. MAR projected annual frost-free period deltas, 1901-1980 vs. 2040-2069. ..... 34
Figure I-31. MAR projected annual frost-free period anomalies, 1901-1980 vs. 2040-2069. ..... 34
Figure I-32. Modeled current bioclimatic envelope (1961-1990) for Apacherian-Chihuahuan Semi-desert Grassland and Steppe. ..... 37
Figure I-33. Mid-century future modeled suitable bioclimate for Apacherian-Chihuahuan Semi-desert Grassland and Steppe. ..... 38
Figure I-34. Mid-century future modeled suitable bioclimate for Apacherian-Chihuahuan Semi-desert Grassland and Steppe. ..... 39
Figure I-35. Modeled current bioclimatic envelope (1961-1990) for Chihuahuan Creosotebush Desert Scrub. ..... 40
Figure I-36. Mid-century future modeled suitable bioclimate for Chihuahuan Creosotebush Desert Scrub. ..... 41
Figure I-37. Mid-century future modeled suitable bioclimate for Chihuahuan Creosotebush Desert Scrub. ..... 42
Figure I-38. Modeled current bioclimatic envelope (1961-1990) for Madrean Encinal. ..... 43
Figure I-39. Mid-century future modeled suitable bioclimate for Madrean Encinal. ..... 44
Figure I-40. Mid-century future modeled suitable bioclimate for Madrean Encinal. ..... 45
Figure I-41. Modeled current bioclimatic envelope (1961-1990) for Apacherian-Chihuahuan Mesquite Upland Scrub. ..... 46
Figure I-42. Mid-century future modeled suitable bioclimate for Apacherian-Chihuahuan Mesquite Upland Scrub. ..... 47
Figure I-43. Mid-century future modeled suitable bioclimate for Apacherian-Chihuahuan Mesquite Upland Scrub. ..... 48
Figure I-44. Climate Change Exposure Index. ..... 50
Figure I-45. Madrean Montane Conifer-Oak Forest and Woodland Overlay Projected Climate Change Exposure Index. ..... 51
Figure I-46. Mogollon Chaparral Overlay Projected Climate Change Exposure Index. ..... 52
Figure I-47. Madrean Pinyon-Juniper Woodland Overlay Projected Climate Change Exposure Index. ..... 53
Figure I-48. Madrean Encinal Overlay Projected Climate Change Exposure Index. ..... 54
Figure I-49. Apacherian-Chihuahuan Semi-Desert Grassland and Steppe Overlay Projected Climate Change Exposure Index. ..... 55
Figure I-50. Chihuahuan Creosotebush Desert Scrub Overlay Projected Climate Change Exposure Index. ..... 56
Figure I-51. Chiricahua Leopard Frog Overlay Projected Climate Change Exposure Index. ..... 57

Figure I-52. Coues White-tailed Deer Overlay Projected Climate Change Exposure Index. ..... 58
Figure I-53. Bighorn Sheep Overlay Projected Climate Change Exposure Index. ..... 59
Figure I-54. Pronghorn Overlay Projected Climate Change Exposure Index. ..... 60
Figure I-55. Grassland Birds Overlay Projected Climate Change Exposure Index..... 61
Figure I-56. Nectivorous Bats Overlay Projected Climate Change Exposure Index. ..... 62
Figure I-57. Desert Box Turtle Overlay Projected Climate Change Exposure Index. ..... 63
Figure I-58. North American Warm Desert Lower Montane Riparian Woodland, Shrubland and Stream Overlay Projected Climate Change Exposure Index. ..... 64
Figure I-59. North American Warm Desert Ciénega and Marsh Overlay Projected Climate Change Exposure Index..... 65
Figure I-60. North American Warm Desert Playa and Ephemeral Lake Overlay Projected Climate Change Exposure Index..... 66
Figure I-61. North American Warm Desert Riparian Woodland, Shrubland, Mesquite Bosque and Stream Overlay Projected Climate Change Exposure Index. ..... 67

## *I.1 Overview of Appendix I*

This appendix presents methods and results for the Madrean Archipelago region (MAR) climate change analyses. This climate change assessment addresses two broad topics: climate trends, and their projected impacts on conservation elements (CEs).

The recent and future climate trend analysis examines spatial and seasonal patterns of change in various climate variables. It includes an analysis of how much climate change has already occurred in recent decades, and also a distillation of future changes projected by various global climate models.

The bioclimatic envelope modeling assessment, covered in **section I.6** explores how projected future climate change may affect ecological communities of the MAR. Species distribution modeling was used to identify projected future shifts in suitable climate conditions for each of four pre-selected CEs: Apacherian-Chihuahuan semi-desert grassland and steppe, Chihuahuan creosotebush desert scrub, Madrean encinal, Apacherian-Chihuahuan mesquite upland scrub.

## *I.2 Methods Summary*

### I.2.1 Source Data

Two source datasets were used to analyze recent trends and future trends. For recent trends the PRISM (Parameter-elevation Relationships on Independent Slopes) dataset was used (Daly et al. 2008). This dataset is 800m resolution derived by interpolating point-based weather station records onto a gridded surface, a widely-used process that enables the study of climate patterns across an entire landscape but also introduces uncertainty; as such, these results must be viewed as model outputs, rather than empirical measurements. To analyze future trends the Climate Western North American (CWNA) dataset was used (Wang et al. 2012). This is a 4km resolution gridded climate dataset of directly calculated and derived variables. The dataset includes a 20$^{th}$ century time series from 1900-2011 and 6 global climate models (GCMs) under the A2 scenario for three future 30-year time slices: 2010-2039 (near-term), 2040-2069 (mid-century), 2070-2099 (end century). To be most consistent with previous REAs, all analyses of future change will be based on the mid-century time slice. The A2 emissions scenario most closely correlates to the RCP6.0 scenario in the IPCC's 5$^{th}$ assessment report and is closest to the trajectory that we are currently on (Knutti & Sedláček 2013; Peters *et al*. 2012). The A2 emission scenario reflects a future in which there is high population growth, regional economic development, and slow technological changes (Pachauri *et al*. 2007).

### I.2.2 Recent Climate Trends

**Purpose:** To quantify the distribution and magnitude of climate change that has recently occurred in the last 30 years in the Madrean Archipelago region (MAR).

**Summary:** This analysis examines recent climate change in the MAR, highlighting ecologically important geographic and seasonal differences in the magnitude of change. For each grid cell in the MAR, for each month of the year, results show changes for maximum temperature (Tmax), minimum temperature (Tmin), and total precipitation (precip). Climate change exposure indices showing total climate change across all months and variables are also presented for recent time periods.

**Source data:** The source data used in this analysis was conducted using the PRISM dataset comprised of 800m rasters for every month of every year (and for annual averages) from 1901 through 2012, for three climate variables (monthly average minimum temperature, monthly average maximum temperature, and monthly total precipitation).

**Process Steps:**

For each 800m pixel in the MAR boundary:

1. The PRISM 800m time series dataset was divided into a *baseline* time period (1901 through 1980) and a *recent* 32-year time period (1981 through 2012). An 80-year baseline was chosen to capture a large spectrum of historic variability. A thirtytwo year mean was chosen for the recent timeslice because thirty years is considered sufficient duration to capture extreme events and a range of climate variations (IPCC 2007).
2. Deltas were calculated by subtracting the mean climate of the baseline, minus the mean climate of the recent timeslice. This produces an estimated magnitude of change in real climate units for each variable for each month for each pixel.
3. Anomalies were calculated, which is the number of standard deviations the recent 30-year mean is from the baseline. Otherwise known as a z-score, this metric is calculated by dividing the recent delta by the standard deviation of the baseline. This metric measures magnitude of change by representing how unusual the recent timeslice is relative to baseline variability.
4. A Climate change exposure index was generated by aggregating z-scores (process 3) across variables (Tmax, Tmin, and precip) and months to get one index of change across the ecoregion. This method uses standard Euclidean distance as a metric to measure departure in multivariate space between recent and baseline timeslices. Standard Euclidean distance gives equal weight to climate variables when measuring the distance between timeslices in environmental space. The z-scores from each of the 3 monthly variables (totaling 36 z-scores) are squared, those squares are summed, and the square root of that sum is the final index value. Climate change in either direction contributes equally to this index (see section I.3 for details on methodology).
5. Theil-Sen slope was used to calculate magnitude of change within the recent timeframe, a Theil-Sen linear regression line is fit to the 32-year time series. The change in the value of this line across the 32-year period indicates magnitude of climate change.
6. Mann-Kendall p-values were calculated to measure the statistical significance of the magnitude of the 32-year trend. P-values less than 0.05 are considered statistically significant.

**Derived Data:**

For each 800m pixel in the MAR, for each climate variable, and for each month, rasters of the following analyses were produced:

1. Deltas between recent and baseline timeslices.
2. Anomalies of z-score values (# of standard deviations) outside of the baseline mean.
3. Climate change exposure index: a single raster of recent climate change exposure aggregated across variables and months for each pixel in the Madrean Archipelago ecoregion.
4. Trend magnitude and significance: rasters characterizing the magnitude of change in either degrees C or mm where pixels meet the threshold of statistical significance (P-value = .05)

### I.2.3 Future Climate Trends

**Pupose**: To quantify the projected distribution and magnitude of future climate change for the Madrean Archipelago ecoregion.

**Summary:** For this analysis, the years 1900 through 1980 were used as the *baseline*, while the years 2040 through 2069 comprised the *future*. For each 4km pixel in the MAR, for each variable (5 monthly

and 1 annual), the average across future years was calculated for each of the six GCMs, and the median of these GCM values was then calculated. (Two GCMs in the CWNA source data include known preexisting errors in temperature values for the A2 scenario; these two GCMs were therefore not used in the analysis of the temperature-related variables, resulting in a smaller model ensemble for those variables.) The ensemble median was then used to calculate metrics comparing the projected future to the baseline. These metrics were aggregated into a single climate change exposure index (CCEI) across the Madrean Archipelago ecoregion.

**Source Data:** Projected future climate trend results are based on the 4km Climate Western North America dataset (CWNA), which includes historical data analogous to the PRISM dataset, and also the projections of six different global climate models (GCMs) for the A2 greenhouse gas emissions scenario for the mid-centruy timeslice (2040-2069). In addition to the three "core" climate variables (monthly minimum temperature, maximum temperature, and total precipitation also analyzed for recent trends), three variables derived by CWNA from these core variables were also analyzed (monthly average climatic moisture deficit, monthly average number of frost-free days, and annual average frost-free period).

**Table I-1. Global climate models used for each variable.** Variables are: monthly minimum temperature, monthly maximum temperature, monthly total precipitation, monthly climate moisture deficit, monthly number of frost free days, annual frost free period.

| GCM | Run # | Emission Scenario | Time Slice | Tmin | Tmax | Precip | CMD | NFFD | FFP |
|---|---|---|---|---|---|---|---|---|---|
| mri_cgcm232a | 1 | A2 | 2040-2069 | | | X | | | |
| miroc32_medres | 2 | A2 | 2040-2069 | X | X | X | X | X | X |
| gfdl_cm21 | 1 | A2 | 2040-2069 | | | X | | | |
| cccma_cgcm3 | 5 | A2 | 2040-2069 | X | X | X | X | X | X |
| cccma_cgcm3 | 4 | A2 | 2040-2069 | X | X | X | X | X | X |
| bccr_bcm20 | 1 | A2 | 2040-2069 | X | X | X | X | X | X |

**Process Steps:**

For each 4km pixel in the MAR boundary:

1. Deltas were calculated by subtracting the mean climate of the baseline (1901-1980), minus the mean climate of the mid-centry future timeslice (2040-2069). This produces an estimated magnitude of change in real climate units for each variable for each month for each pixel. Deltas are calculated for each GCM and the median value across GCMs is reported.
2. Anomalies were calculated, which is the number of standard deviations the future timeslice is from the 80-year baseline. Otherwise known as a z-score, this metric is calculated by dividing the recent delta by the standard deviation of the baseline. This metric measures magnitude of change by representing how unusual the future timeslice is relative to baseline variability.

3. Climate change exposure index was generated to aggregates z-scores (process 2) across three variables (minimum temperature, maximum temperature, and precipitation) and 12 months to get one index of projected future change across the ecoregion (see section I.3 for details on methodology). Derived variables were not included in this analysis to prevent redundancy of correlated climate variables.

**Derived Data:**

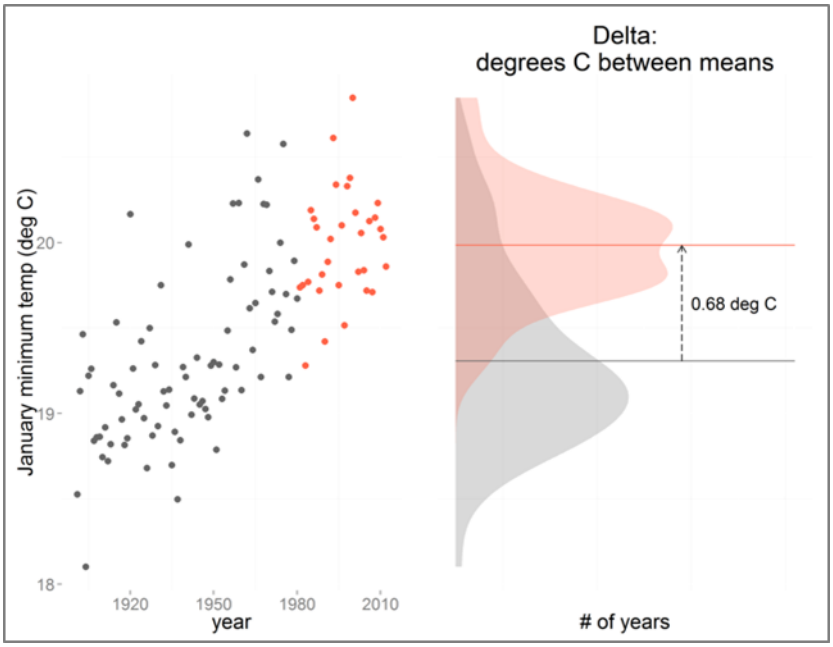
For each 4km pixel in the MAR, for each climate variable, and for each month, rasters of the previous analyses were produced:

1. Deltas: rasters of the geographic distribution of deltas between the future mid-century timeslice (2040-2069) and baseline for each variable, for each month, for each pixel across the Madrean Archipelago ecoregion.
2. Anomalies: rasters of the geographic distribution of z-score values (# of standard deviations) between the future timeslice (2040-2069) for each variable, for each month, for each pixel across the ecoregion.
3. Climate change exposure index: one raster of the future mid-century climate change exposure aggregated across variables and months for each pixel in the Madrean Archipelago ecoregion.

## *I.3 In-Depth Methodology for Climate Trends*

***Deltas*** measure the difference between the mean climates of two time periods, and are the most widely used measure of climate change. For each raster pixel, for a particular variable and month, the mean climate value across the 80 baseline years is calculated, as is the mean across all 32 years of the recent period or all 30 years of the future period. Deltas are then expressed as the difference (either future or recent timeslice – baseline) between these two means, and where relevant they are also presented as a ratio (timeslice/baseline).

**Figure I-1. Visualization of a climate delta, for a single pixel for a single variable and month.** The grey color represents the distribution of baseline values and the red color represents the distribution of values for the recent timeselice.

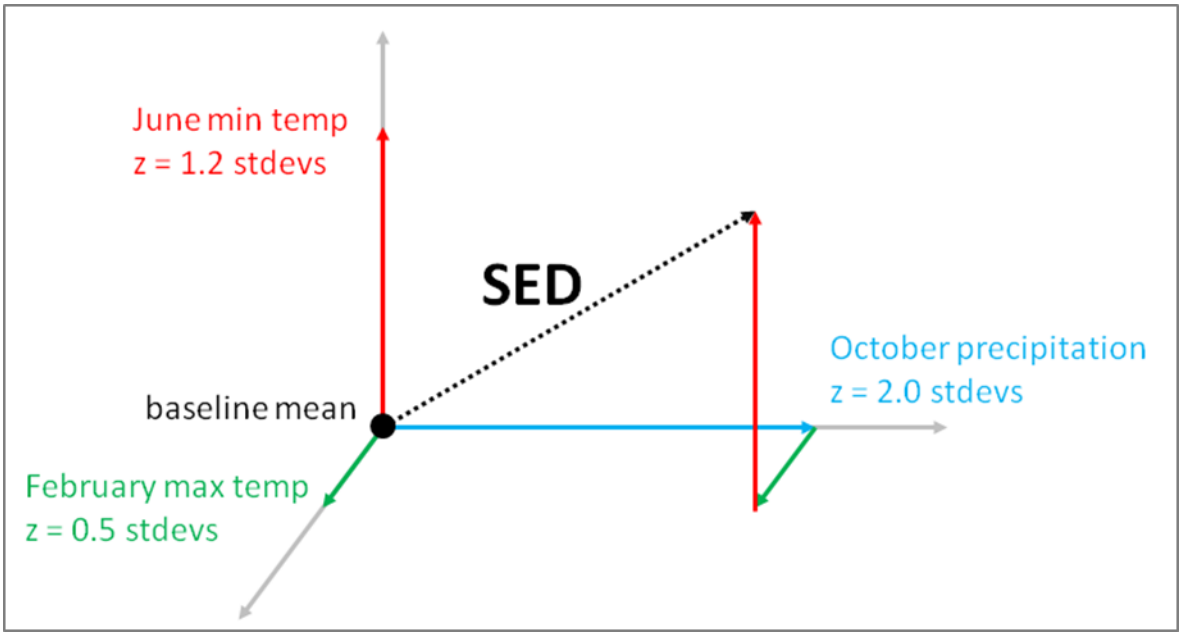



***Anomalies*** (also known as z-scores or standard scores) normalize these deltas by baseline climatic variability, indicating how historically unusual the later time period is. The standard deviation across the 80 baseline years is calculated, and the above delta is divided by this standard deviation, resulting in a standardized measure of the number of baseline standard deviations between the baseline mean and the later mean. This calculation is done for each pixel across the MAR.

**Figure I-2. Visualization of a climate anomaly or z-score, for a single pixel for a single variable-month.**

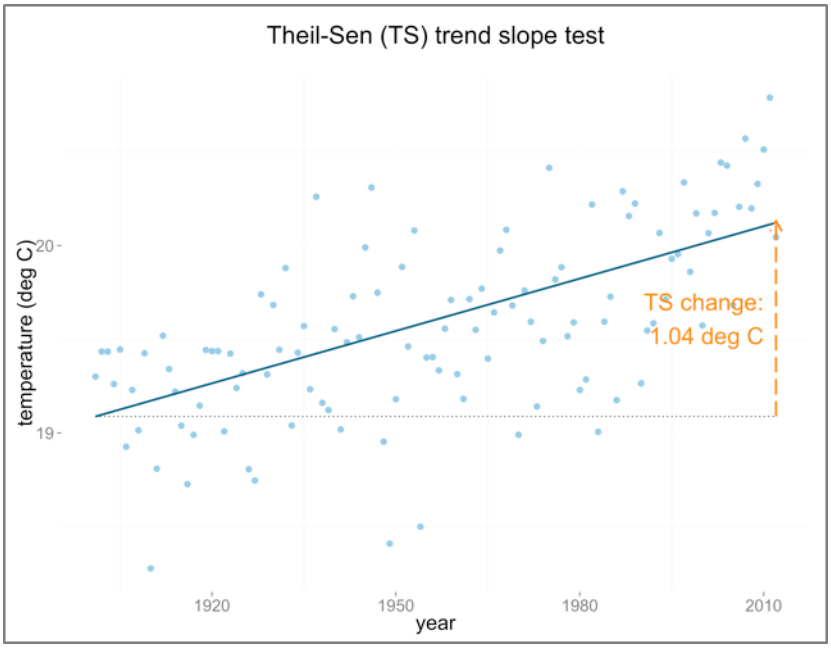
***Standard Euclidian distance (SED)*** is a statistic used to combine these z-scores across different months and climate variables to generate an index of total climate change (Diffenbaugh et al. 2008, Diffenbaugh and Giorgi 2012, Williams et al. 2007), called a climate change exposure index (CCEI) in this report. For both recent and future climate change, z-scores for each of the 12 months for each of the 3 core variables were used as inputs. For each pixel, these z-scores were squared, summed, and square-rooted to calculate the final SED. More information on SED interpretation can be found in the results section.

**Figure I-3. Visualization of standard Euclidian distance (SED) for a single pixel.** A 36-dimensional version of the 3-dimensional version shown here was used to generate the climate change exposure indices in this report.




***Trends*** measure the magnitude and statistical significance of climate change *within* a single time period (in this case, only used for the recent 1981-2012 timeslice), using a combination of two tests that are frequently used in tandem to quantify trends in non-normally distributed climate data (e.g. Skansi et al. 2013). For each variable-month, trend magnitude was estimated for each pixel using the Theil-Sen test, which fits a linear regression line to the time series; the change in the value of this regression line was reported as the estimated change in climate. Statistical significance is estimated using the Mann-Kendall test (specifically, a pre-whitened variation proposed by Yue et al. (2002) and implemented in the R zyp package (Bronaugh and Werner 2013)); change was only deemed statistically significant in places where the Mann-Kendall p-value was less than 0.05.

**Figure I-4. Visualization of Theil-Sen trend for a single pixel for a single variable-month.**



## *I.4 Results: Recent Climate Change*

### I.4.1 Recent Decades versus Baseline

The following maps depict the magnitude of climate change between the baseline and the most recent 32 years for each 800-m pixel in the MAR, for each month, for each of three climate variables: monthly total precipitation, monthly average maximum temperature (the mean of all daily maximum temperatures during the month), and monthly average minimum temperature (the mean of all daily minimum temperatures during the month).

#### I.4.1.1 Precipitation

Average precipitation in the MAR across the years since 1981 differed moderately from baseline precipitation in certain months and locations, with both increases and decreases observed. The majority of individual months saw both increases and decreases in different locations, although most changes were less than 25% of baseline means. The greatest precipitation rises were 50% to 100% increases that occurred in May across the eastern MAR, and in July, November, and December across the far eastern edge of the region -- these corresponded to average recent climates 0.4 to 0.8 standard deviations wetter than the baseline. Rainfall declined by roughly 50% across the northwestern portion of the MAR in June (modest z-scores of -0.25 to -0.5 suggest this remains well within historic variability however), and also saw more modest declines in March, April, July, and November across significant portions of the region.

**Figure I-5. MAR monthly total precipitation deltas, 1901-1980 vs. 1981-2012.**

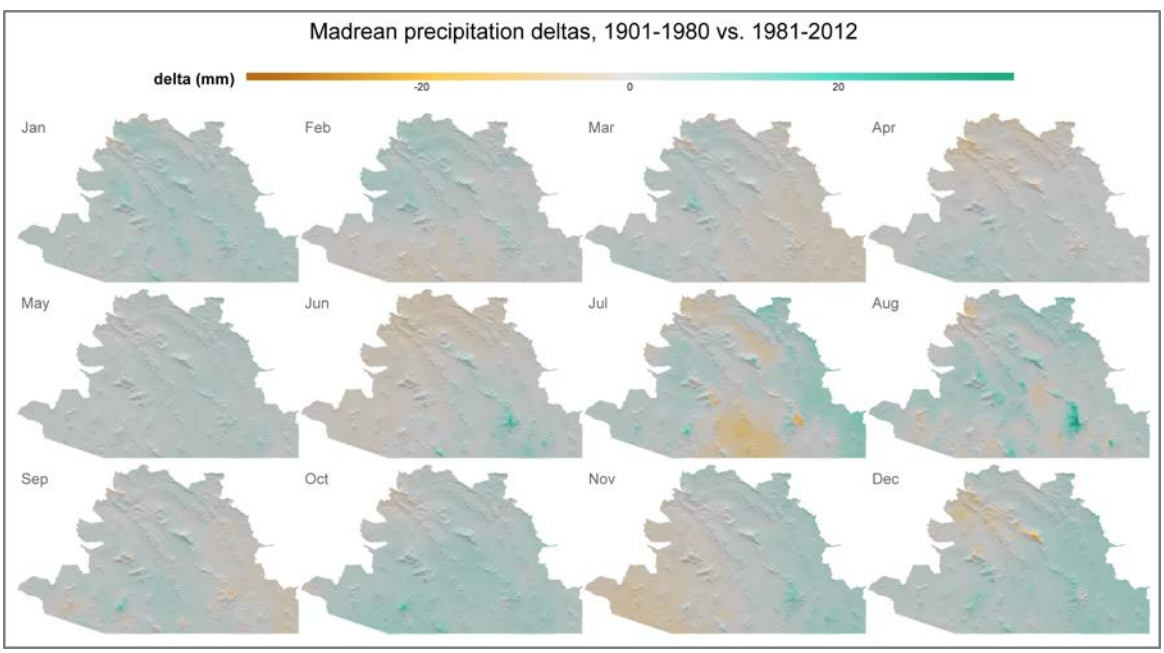


**Figure I-6. MAR monthly total precipitation delta ratios, 1901-1980 vs. 1981-2012.**

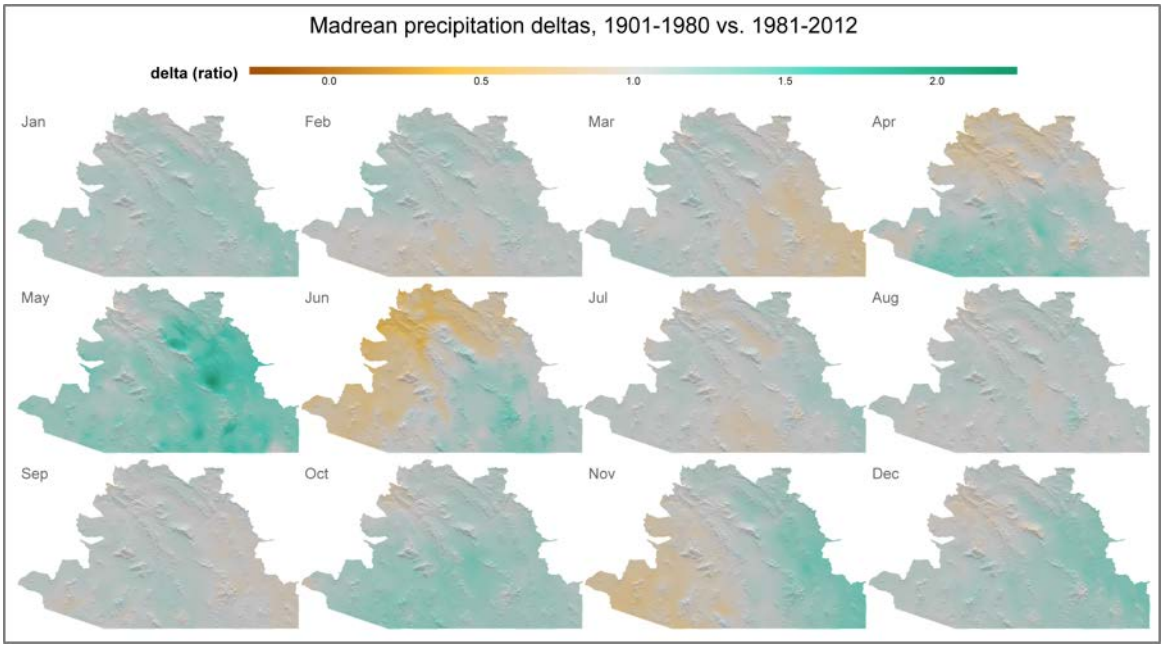
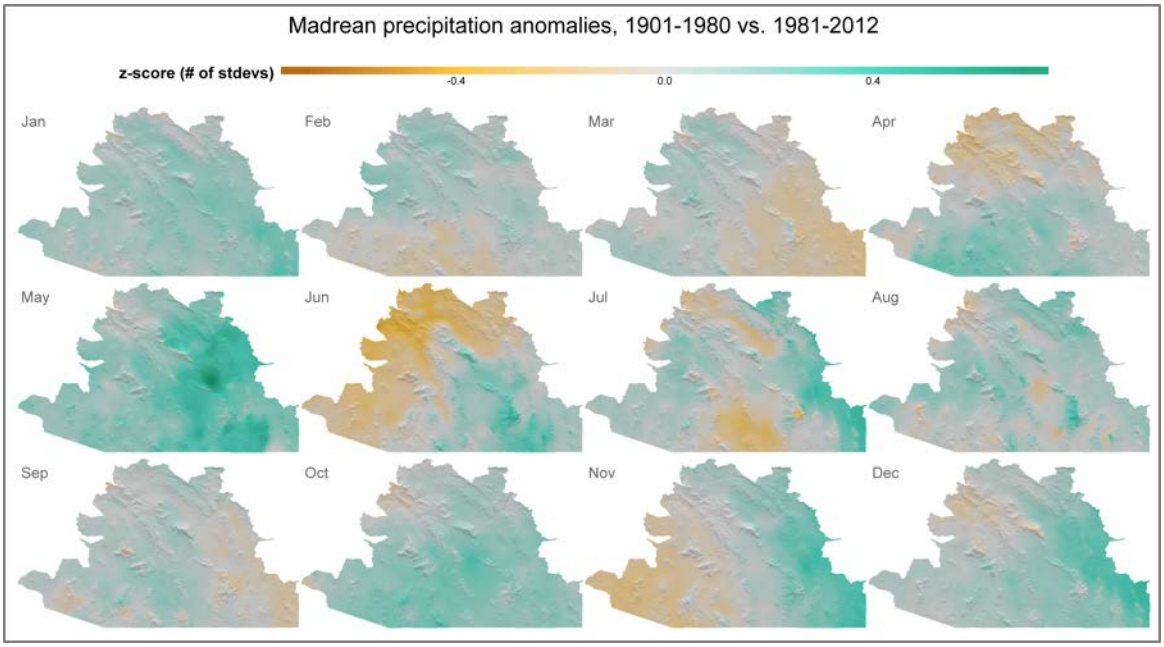



**Figure I-7. MAR monthly total precipitation anomalies, 1901-1980 vs. 1981-2012.**



#### I.4.1.2 Minimum Temperature

Every month of the year exhibited minimum temperature increases across the majority of the ecoregion between the baseline and recent periods. The greatest increases approached 4 degrees Celsius and occurred during the spring months in low-lying western and northern parts of the MAR; April, May, June, and August had minimum temperature z-scores of 1.5 to 2.0 in these locations, suggesting a notable departure from baseline variability.

However, nearly every month also saw pockets of minimum temperature stability and even modest decreases on the order of 1 degree Celsius; this occurred predominantly in mountainous locations dispersed across the region.

**Figure I-8. MAR monthy average minimum temperature deltas, 1901-1980 vs. 1981-2012.**

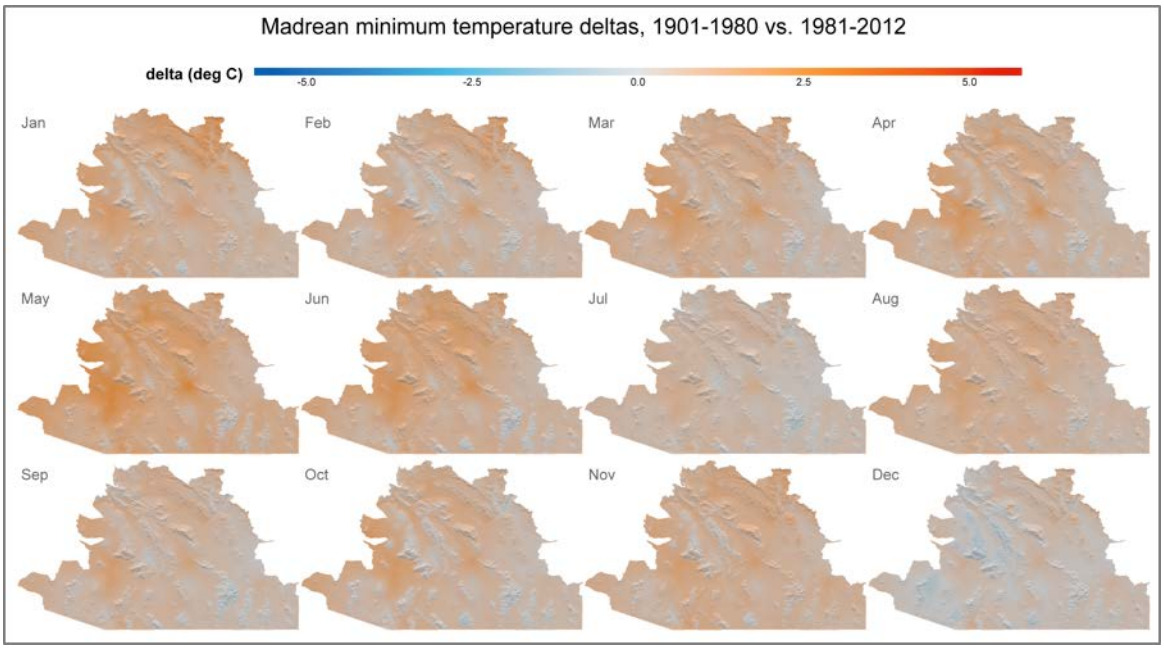
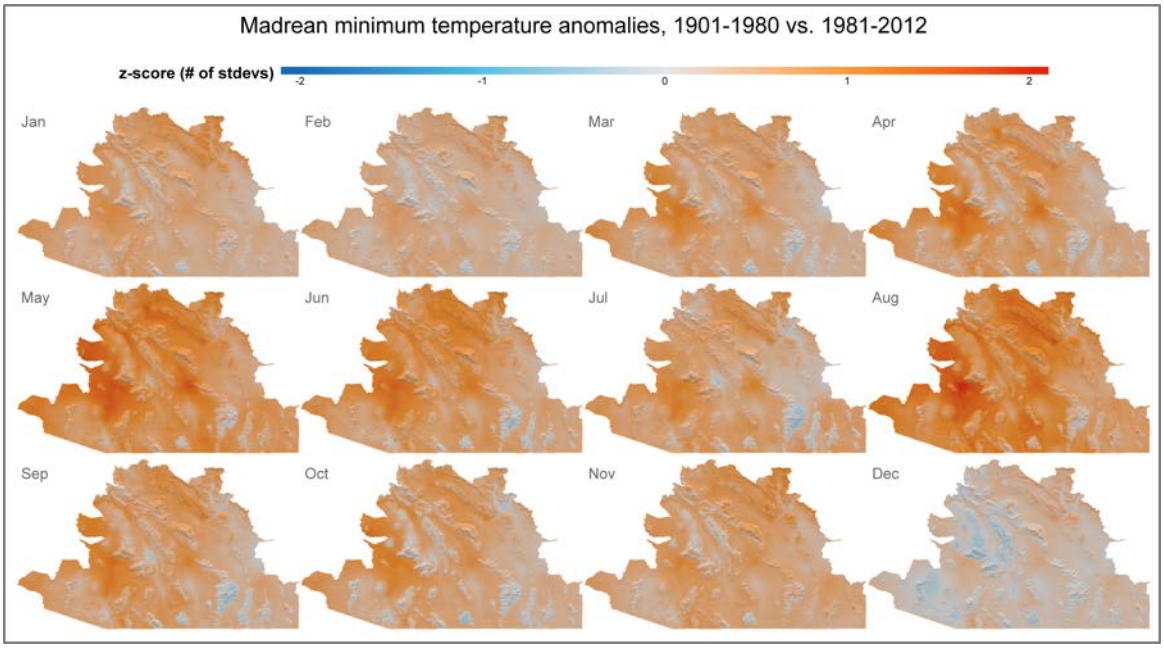


**Figure I-9. MAR monthly minimum temperature anomalies, 1901-1980 vs. 1981-2012.**



### I.4.1.3 Maximum Temperature

Compared to minimum temperature, maximum temperature changes between the baseline and recent periods exhibited similar patterns across the calendar year and across space, but showed smaller increases and larger decreases. The greatest maximum temperature increases, around 2 degrees Celsius, occurred in January, March, April, and May across most of the region, and were approximately one standard deviation higher than the baseline mean.

Maximum temperature declines were greatest in high elevation areas across the northern MAR from November through February, with declines topping 1 degree Celsius in these places and making the recent climate 0.25 to 0.75 standard deviations cooler than the baseline. Smaller but more widespread maximum temperature decreases were also noted in July through September.

With maximum temperatures rising more slowly than minimum temperatures, the magnitude of diurnal temperature fluctuation has declined.

**Figure I-10. MAR monthly maximum temperature deltas, 1901-1980 vs. 1981-2012.**

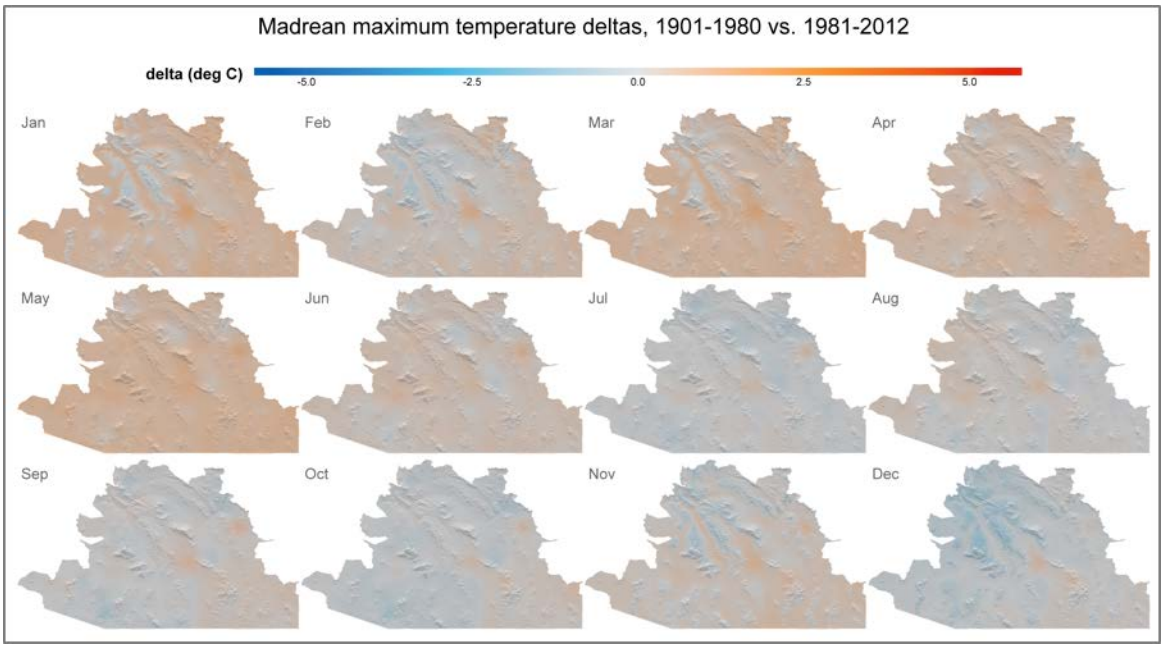
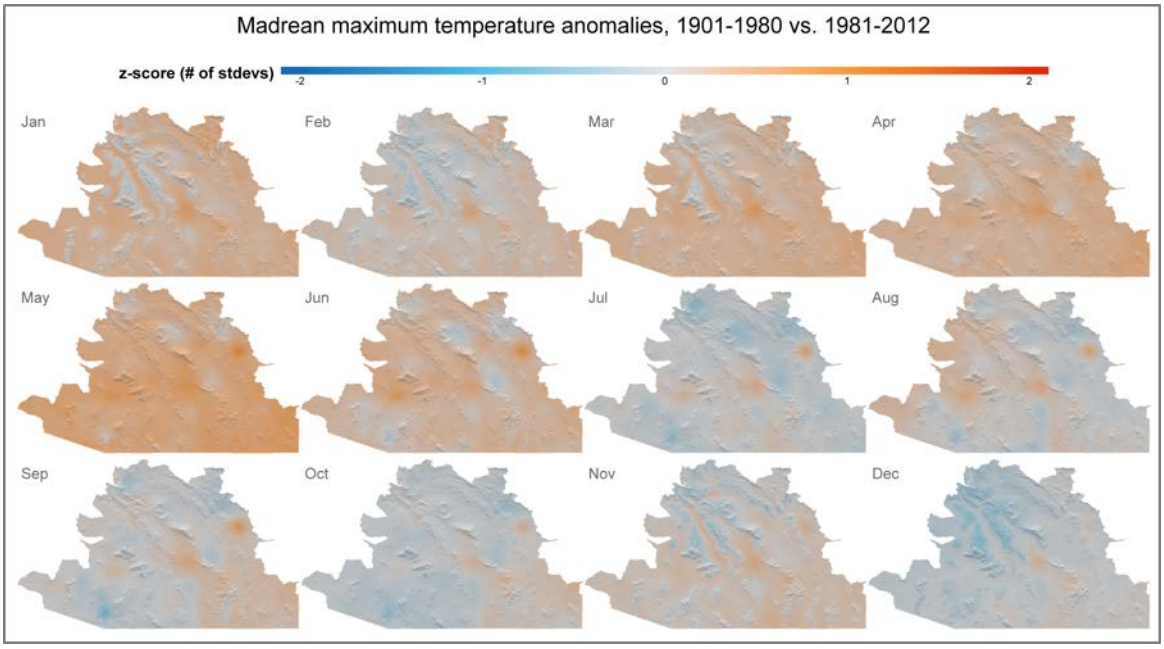


**Figure I-11. MAR monthly maximum temperature anomalies, 1901-1980 vs. 1981-2012.**

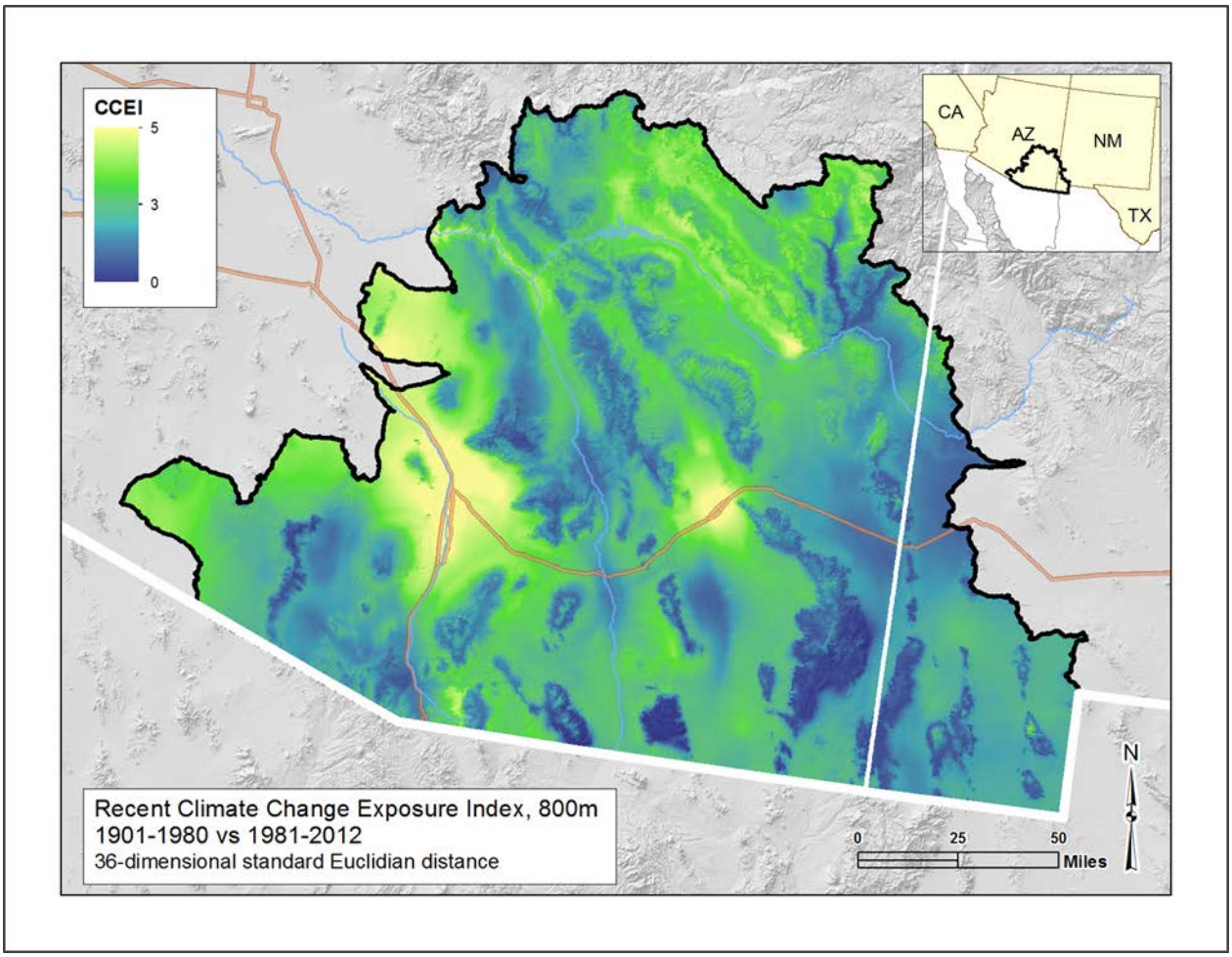


#### I.4.1.4 Climate Change Exposure Index

The recent climate change exposure index combines the above anomalies across all twelve months and all three climate variables, presenting an overall measure of how much recent climate has strayed from baseline variablity. For each pixel, these 36 z-scores are squared, summed, and square-rooted, resulting in a single metric known as standard Euclidian distance (SED) -- this is an index of total climate change across the MAR between the baseline and the last 32 years. An increase or decrease in any of the variables contributes equally to the index.

The result suggests that lower elevation areas within the MAR experienced the most overall climate change between the baseline and the recent 32-year period, while mid- to high elevation climates remained relatively more stable. This highest overall change occurred in the area around Tuscon.

**Figure I-12. MAR current climate change exposure index.** Blue colors represent less overall climate change exposure and yellow represents high exposure. The CCEI is in units of standard deviation showing depature of recent climate from historic variability.



## I.4.2 Trends within Recent Decades

This analysis addresses the magnitude and statistical significance of climate trends *within* the recent 32-year timeslice (as opposed to comparing this period to the baseline, as above). Trend magnitude is estimated for each pixel using the Theil-Sen test, which fits a linear regression line to the time series; in the maps below, color indicates the change in value of that regression line between 1981 and 2012 (temperature changes are expressed in degrees Celsius, while precipitation changes are expressed separately both in mm and as ratios). Statistical significance is estimated using the Mann-Kendall test; in the maps below, pixels where this test results in a p-value greater than 0.05 are grayed out, indicating that recent trends in those places are not statistically significant.

### I.4.2.1 Precipitation

During the 32 years from 1981 to 2012, precipitation saw widespread statistically significant declines and no significant increases. July and August are the peak monsoon months accounting for the most precipitation; July saw no significant change, while August exhibited precipitation declines of roughly 30% across the western half of the MAR. The data suggest that October and November (months of intermediate overall precipitation levels), experienced the most spatially widespread and greatest percent decreases, with declines approaching 100% in some areas.

**Figure I-13. MAR monthly precipitation trend magnitude and significance, 1981-2012.**

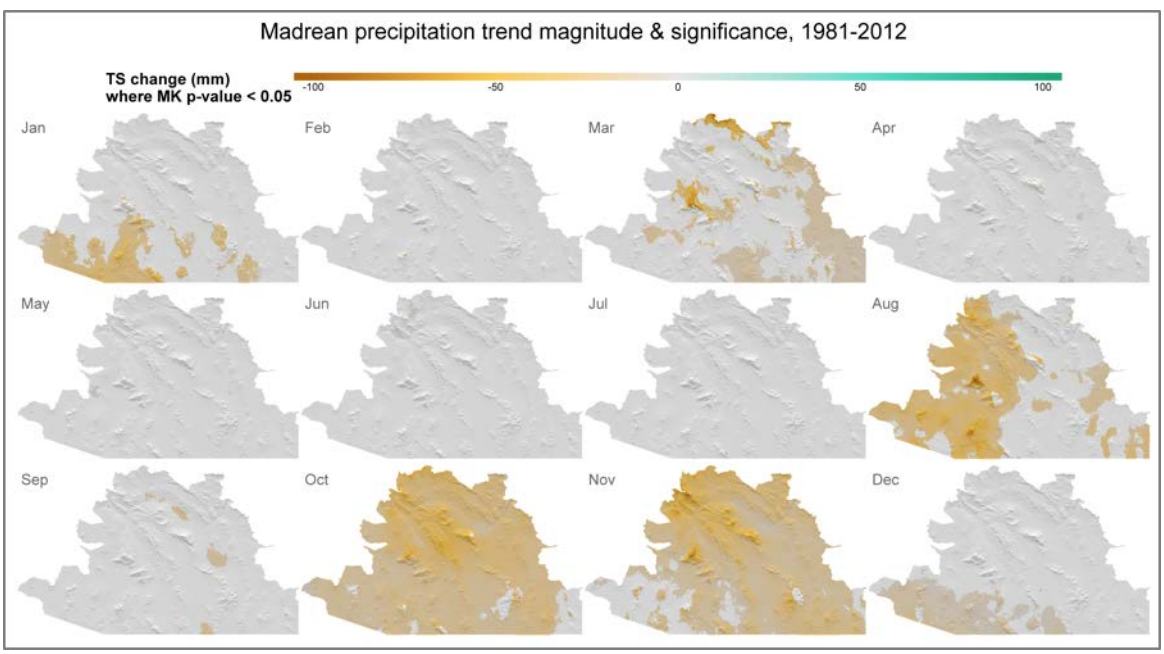
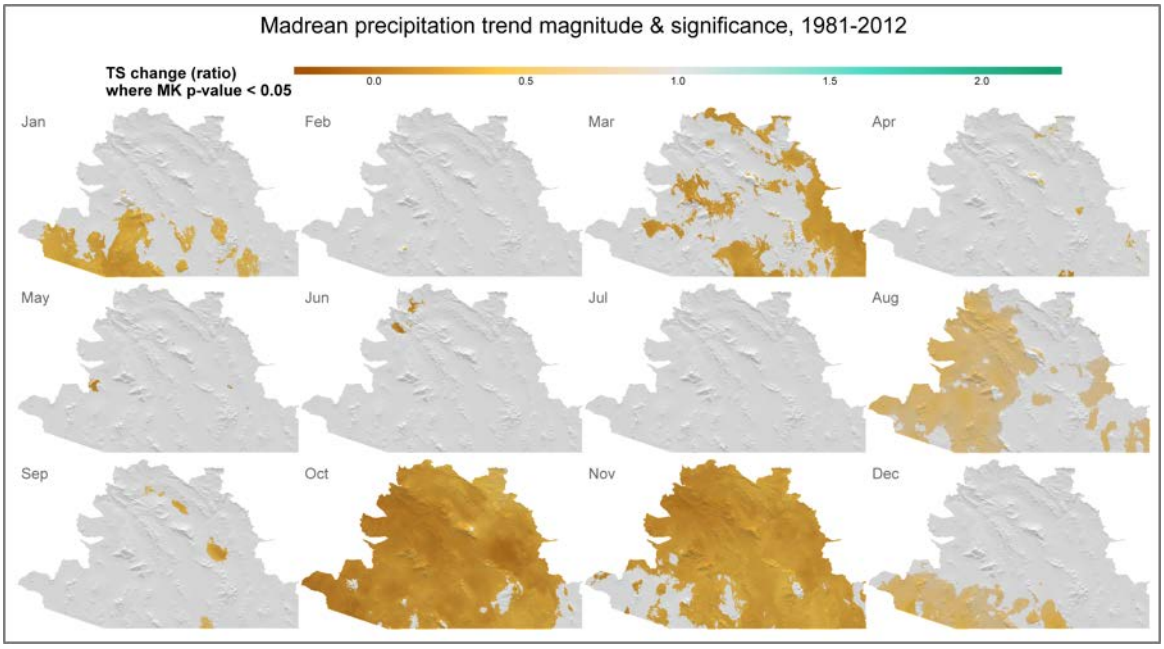


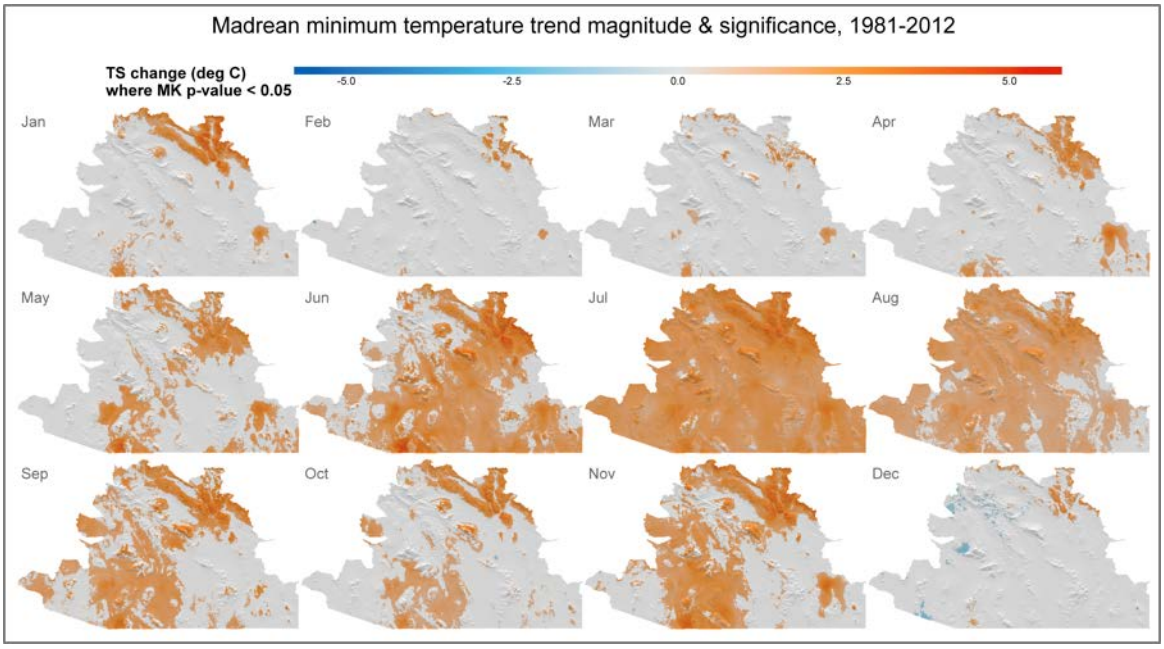
**Figure I-14. MAR monthly precipitation trend ratio magnitude and significance, 1981-2012.**




### I.4.2.2 Minimum Temperature

During the years 1981 to 2012, significant minimum temperature increases were most widespread across the ecoregion during summer months, but high elevations in the Northwest consistently experienced significant increases across all months. Most of these increases were in the 1 to 3 degree Celsius range, but in some high elevation locations in the northeastern corner of the MAR minimum temperatures increased by more than 4 degrees C. Relatively few statistically significant minimum temperature decreases were detected.

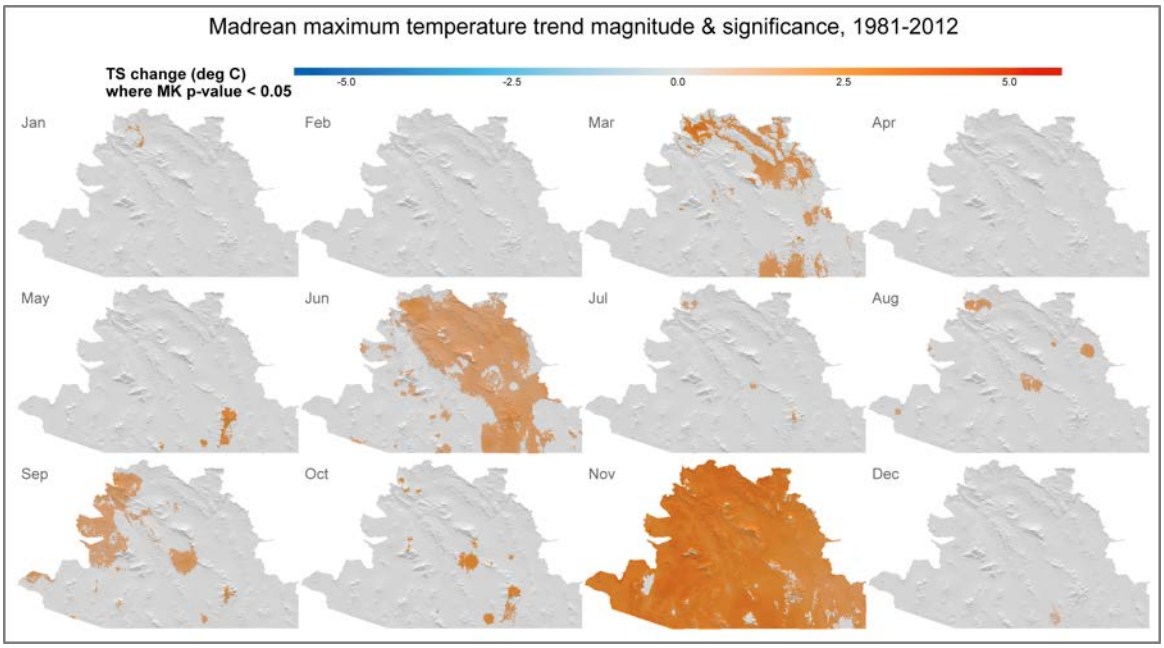
**Figure I-15. MAR monthly minimum temperature trend magnitude and significance, 1981-2012.**



### I.4.2.3 Maximum Temperature

Statistically significant maximum temperatures trends were less pervasive than for minimum temperature, with the majority of months exhibiting no widespread trends. November showed the most significant change, with maximum temperatures increasing 2 to 4 degrees C across the MAR during this 32-year period. July and March also saw more limited maximum temperature increases in the western and northern portions of the ecoregion. No pockets of significant maximum temperature decline were detected.

**Figure I-16. MAR monthly maximum temperature trend magnitude and significance, 1981-2012.**



## *I.5 Results: Projected Future Climate Change*

This analysis summarizes projected future climate change across the Madrean Archipelago ecoregion. For each 4km pixel in the MAR, for each climate variable, for each month of the year, for each model in an ensemble of four to six global circulation models (GCMs) run under the A2 emissions scenario, average climate projections across the years 2040 through 2069 were combined into a single ensemble median representing the best-guess projection for that period. This was then compared to the average climate for 1901-1980, the same baseline used in the recent climate change analysis presented above, to estimate projected change between baseline and future climates.

GCM results include uncertainty about initial conditions, climate processes, and emissions scenarios. And because they operate at a relatively coarse spatial resolution and do not capture localized microclimates, the fine-scale spatial heterogeneity expressed in the recent trends (800m) is not present in future trends (4km). As such, these future projections are more useful for understanding broad regional spatial climate change gradients, and for understanding seasonal climate change patterns, as opposed to identifying likely differences between proximate localities.

### I.5.1 Core Climate Variables

#### I.5.1.1 Precipitation

By the mid 21$^{st}$ century, MAR rainfall is projected to increase by up to 40% of baseline levels, or decrease by as much as 60% of baseline levels, depending on the month and location.

The greatest percent increases are projected for October and December, while June and September are also forecast to see small precipitation increases across large portions of the MAR. None of these projected increases entail a mean future climate that is more than 0.5 standard deviations wetter than the baseline mean, however.

In contrast, the drying that is forecast at other times of year indicates future a climate 1 to 2 standard deviations drier than the baseline mean. The largest percentage decreases in precipitation are projected in April and May, while July and August (the two heaviest monsoon months) as well as January and February, are all also projected to see precipitation declines of up to 20% across the MAR.

Among months where a mix of increased and decreased rainfall is projected for different locations (March, June, September, and November), the one commonality is that the southwestern quadrant of the MAR is projected to become somewhat drier.

**Figure I-17. MAR projected monthly total precipitation deltas, 1901-1980 vs. 2040-2069.**

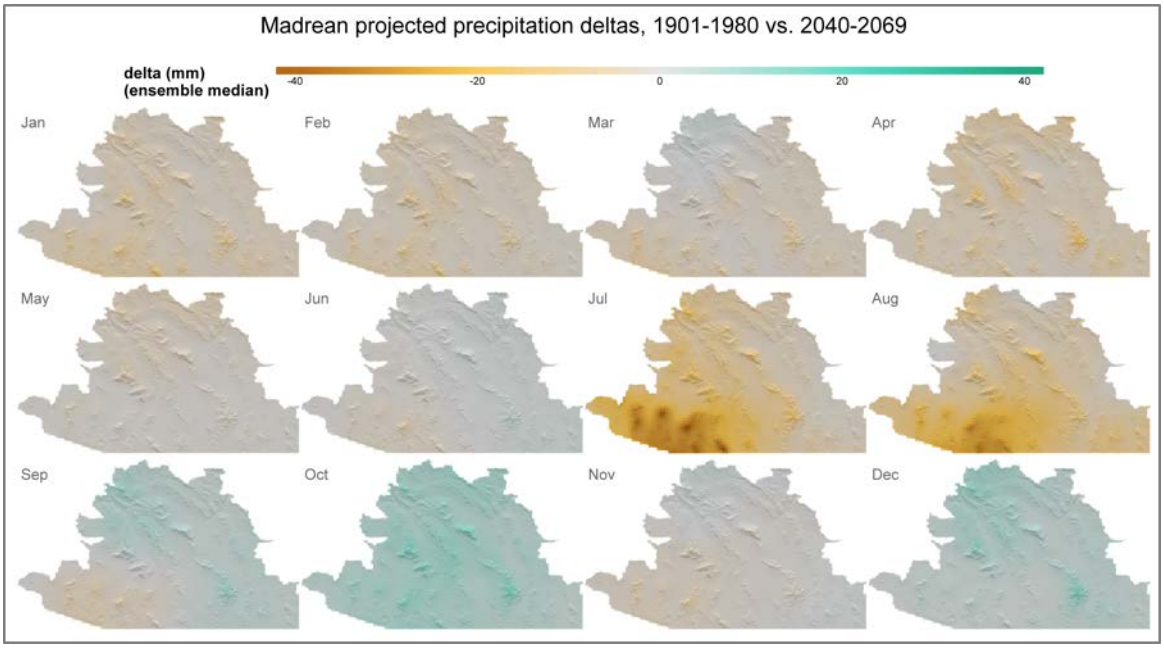



**Figure I-18. MAR projected monthly total precipitation delta ratios, 1901-1980 vs. 2040-2069.**

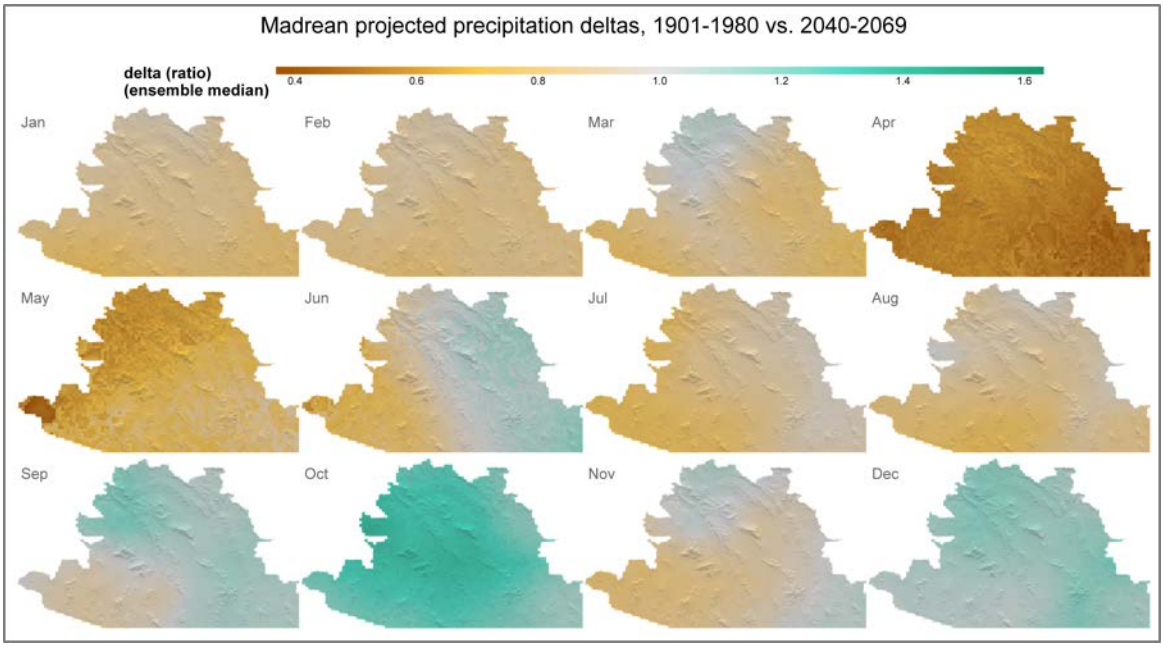
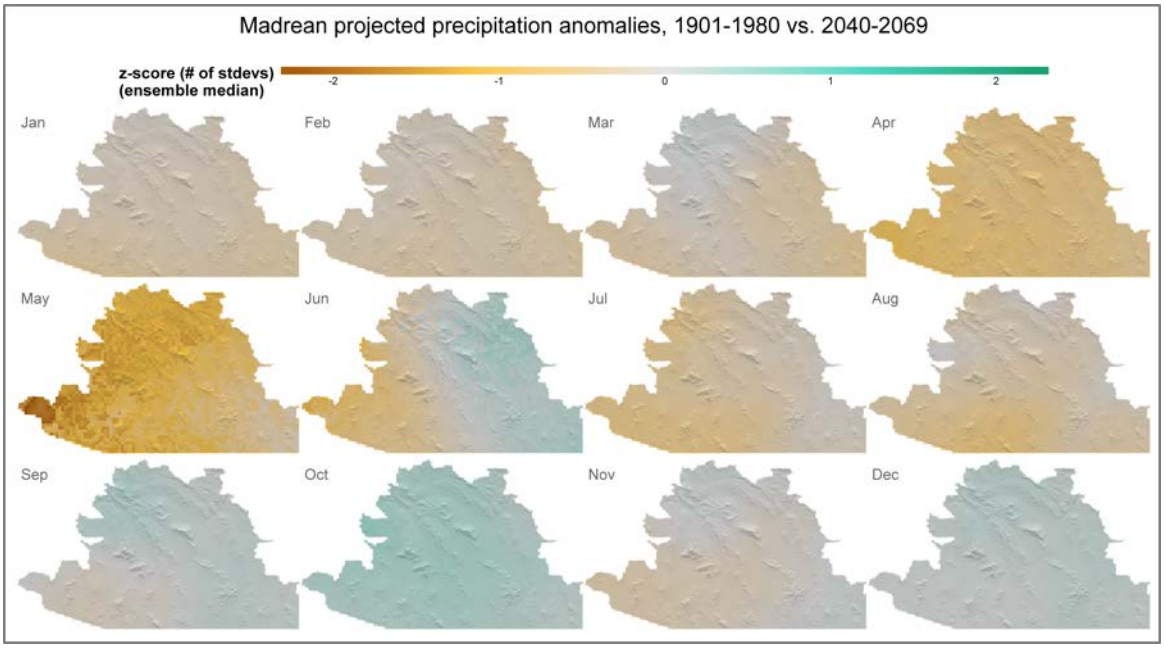


**Figure I-19. MAR projected monthly total precipitation anomalies, 1901-1980 vs. 2040-2069.**




### I.5.1.2 Minimum Temperature

Compared to the baseline, climate models predict that by mid-century the average minimum temperature will increase by 1 to 4 degrees C throughout the MAR, across all months of the year. The winter months (December through March) are forecast to see increases of 1 to 2.5 degrees C, while May through October have projected increases of 3 to 4 degrees C across the region; the shoulder months of April and November have intermediate projected increases of 2.5 to 3.5 degrees C. While the spatial differentiation of these projections is low, most months exhibit relatively lower projected change magnitudes in the southeastern MAR and relatively more warming in the northwestern MAR.

While these deltas are relatively uniform across space and time, they result in far less uniform anomalies. The mean mid-century climate in the winter months is projected to exceed baseline mean minimum temperatures by no more than 1.5 standard deviations, while for July, August, and September minimum temperature anomalies across the southern MAR are forecast to exceed the baseline mean by between 4 and 5.5 standard deviations.

**Figure** I-20**. MAR projected monthy average minimum temperature deltas, 1901-1980 vs. 2040-2069.**

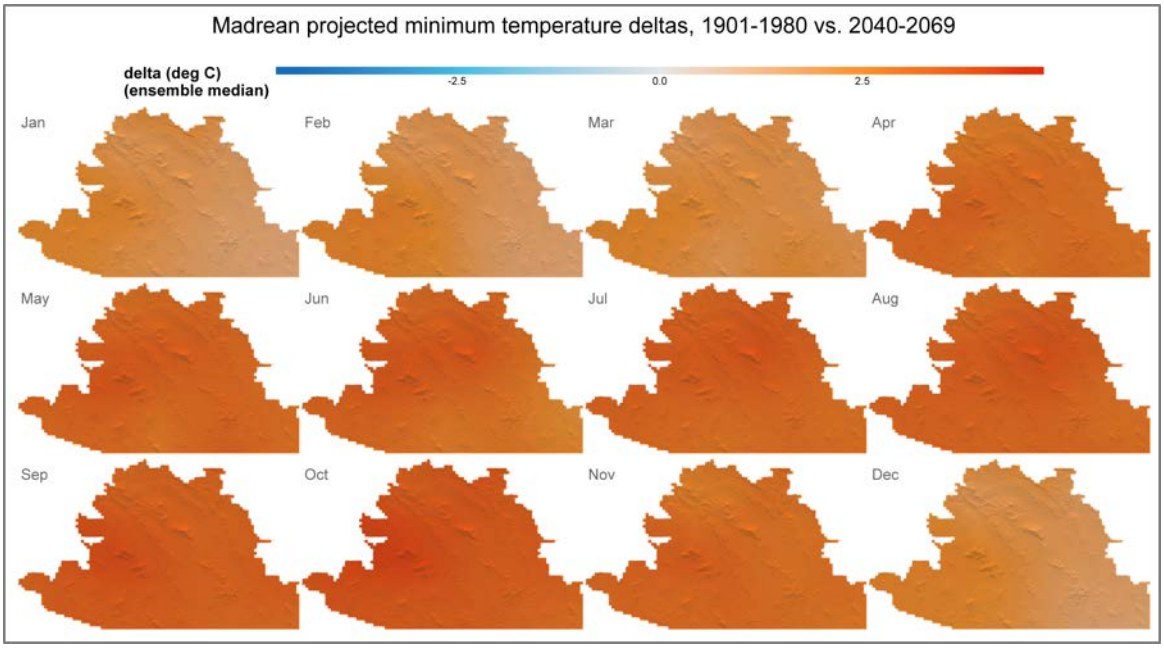
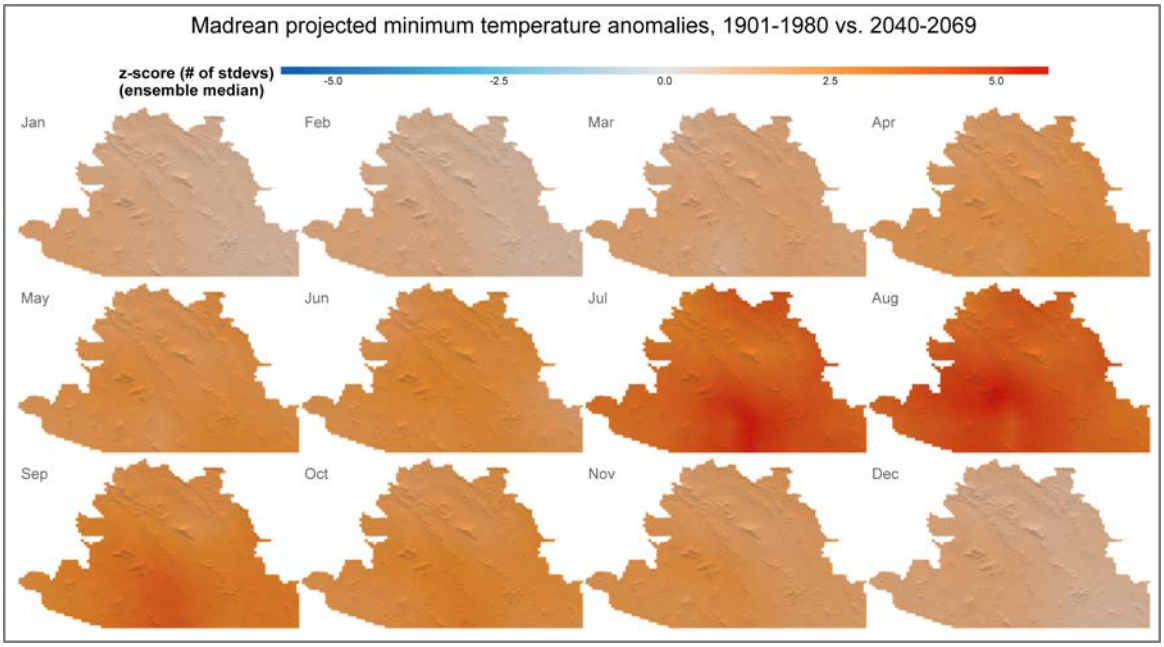



**Figure I-21. MAR projected monthly minimum temperature anomalies, 1901-1980 vs. 2040-2069.**



### I.5.1.3 Maximum Temperature

Like minimum temperatures, maximum temperature deltas are forecast to increase least in the winter and more in other months. From November through March, monthly maximum temperature is projected to rise by 1 to 3 degrees C compared to the baseline, with mean mid-century anomalies of 0.5 to 1.5 standard deviations. Greater increases are projected for April through October, with deltas of roughly 3 to 4.5 degrees C, and anomalies peaking in July and August at more than 3 standard deviations warmer than the baseline mean.

**Figure I-22. MAR projected monthly average maximum temperature deltas, 1901-1980 vs. 2040-2069.**

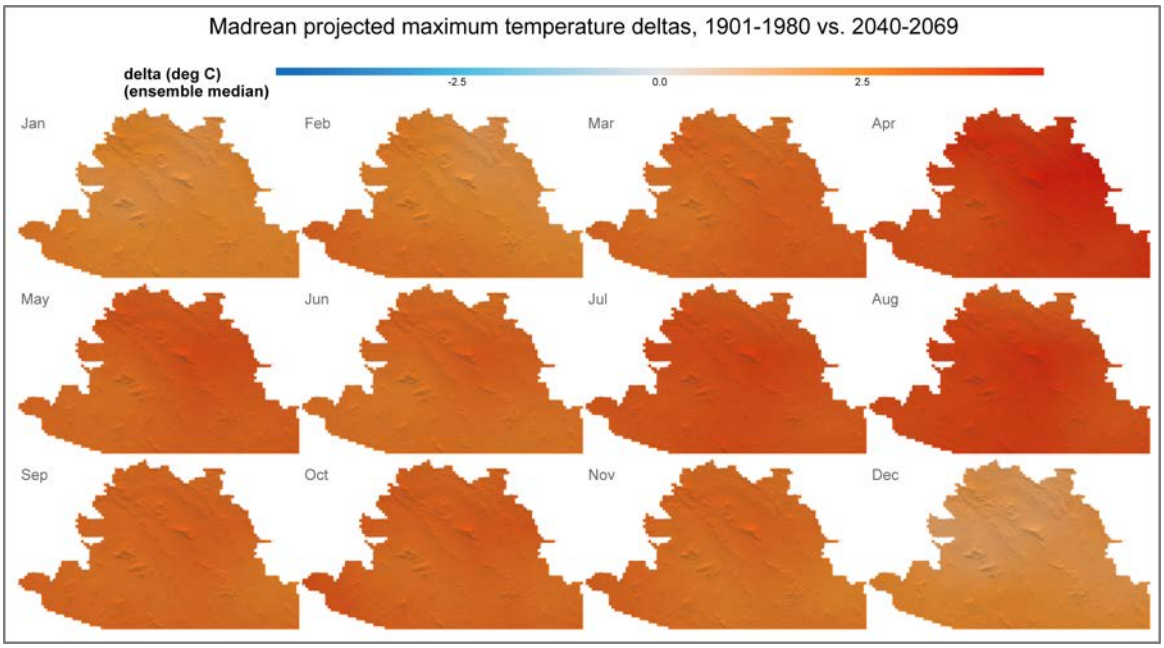
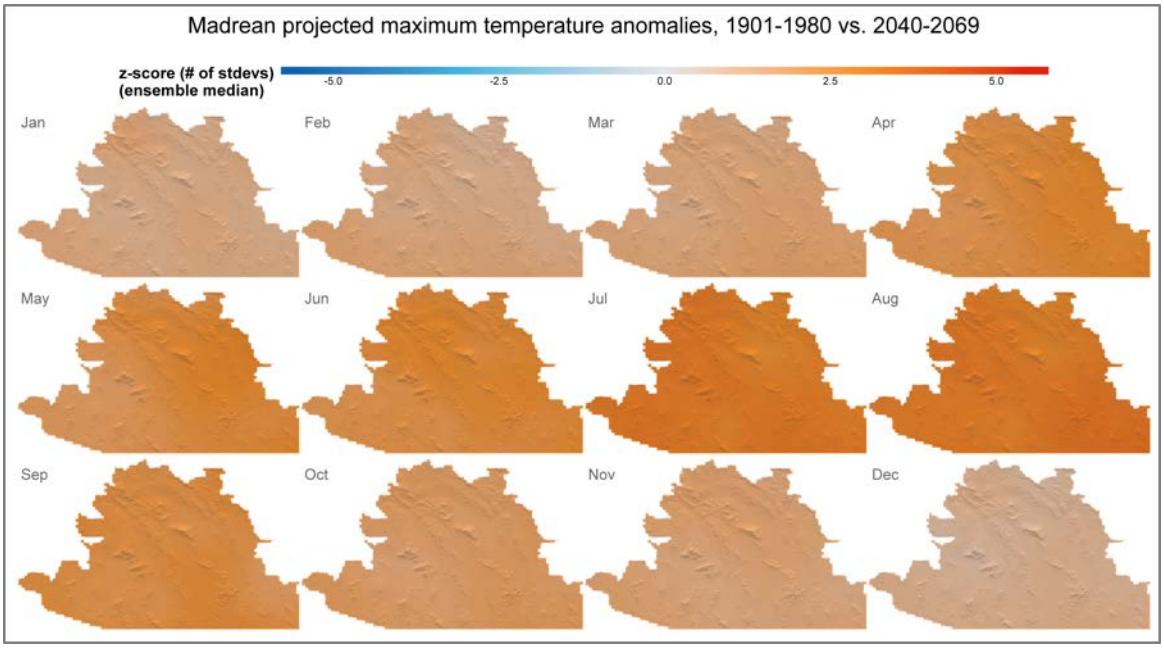


**Figure I-23. MAR projected monthly average maximum temperature anomalies, 1901-1980 vs2040-2069.**

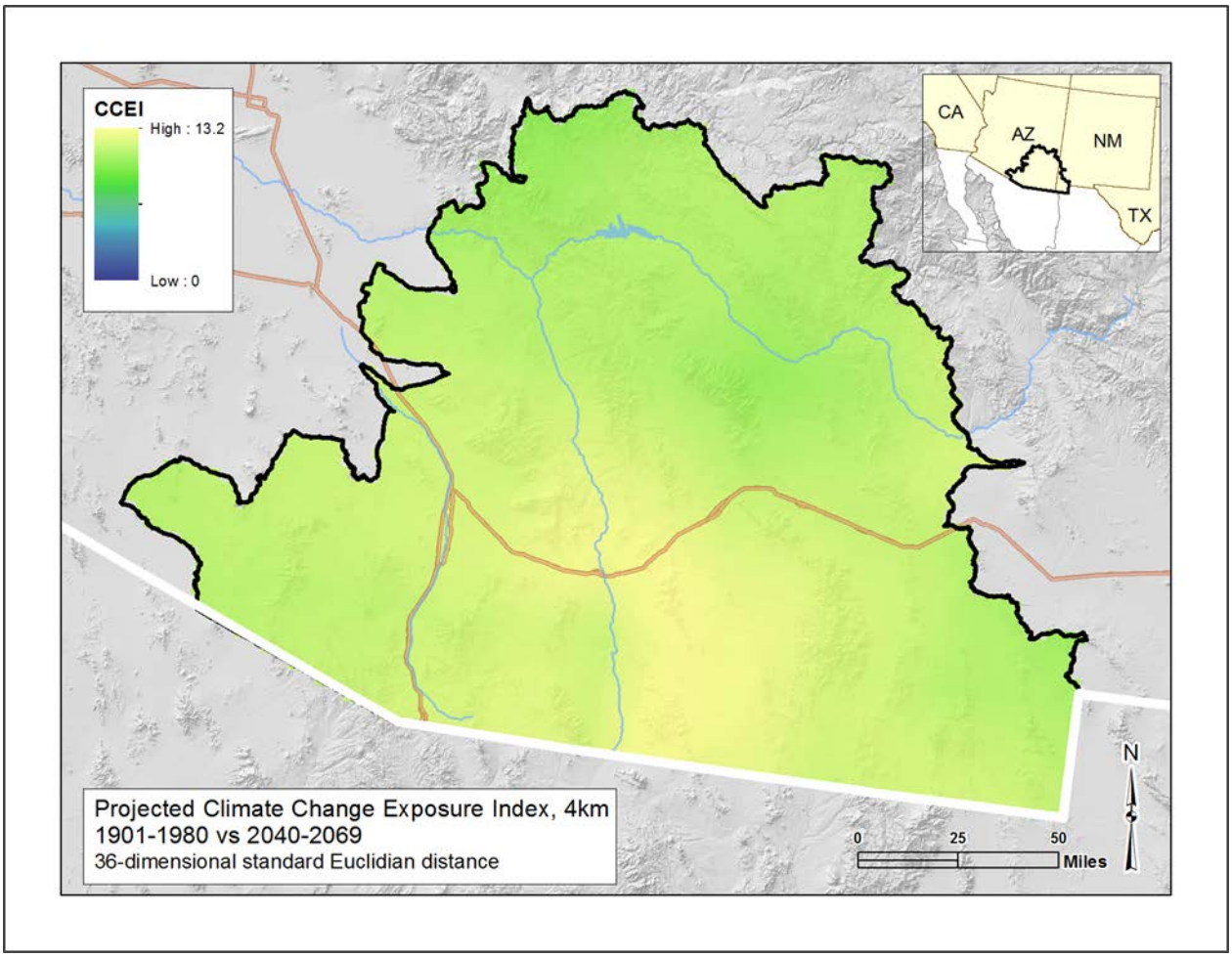


## I.5.2 Climate Change Exposure Index

The projected climate change exposure index combines anomalies across all twelve months and all three core climate variables (precipitation, minimum temperature, and maximum temperature) using the SED statistic, presenting an overall measure of how future climate is forecast to stray from baseline variability. An increase or decrease in any of the variables contributes equally to the index, with an index of 0 indicating that the future mean climate is identical to the baseline climate, and higher values indicating greater overall climate departure.

Results show that across the MAR, climate is projected to be quite different by mid-century than it was during the 1901-1980 baseline. Relatively speaking, the northern MAR is projected to experience less overall climate change than the central and southern regions of the MAR. These differences are driven by spatial patterns both in projected change magnitude and in natural climate variability.

**Figure I-24. MAR projected future climate change exposure index, 1901-1980 vs. 2040-2069.** Blue colors represent less overall climate change exposure and yellow represents high exposure. The CCEI is in units of standard deviation showing depature of projected future climate from historic variability.



## I.5.3 Derived Climate Variables

### I.5.3.1 Climatic Moisture Deficit

Climatic moisture deficit (CMD) is a measure of dryness derived from temperature and precipitation, and can serve as a useful indicator of drought stress. Increasing CMD is associated with hotter and/or drier conditions, while decreasing CMD indicates greater moisture availability due to precipitation increases and/or temperature declines.

Climate models project that CMD will decrease throughout the high-elevation areas of the MAR in December, and in more isolated high-elevation areas in November, January, and February. These increases in moisture availability are 0.5 to 1 standard deviations from the baseline mean.

CMD is forecast to double in July and August in the mountains of the southern MAR, and to experience increases of smaller percentages throughout the MAR in all months from March through September. Z-scores greater than 2 indicate that the April, May, and June drying trends represent the most significant departures from baseline conditions, and that the large percentage increases in July and August in fact fall closer to 1 standard deviation from baseline means.

**Figure I-25. MAR projected monthly climatic moisture deficit deltas, 1901-1980 vs. 2040-2069.**

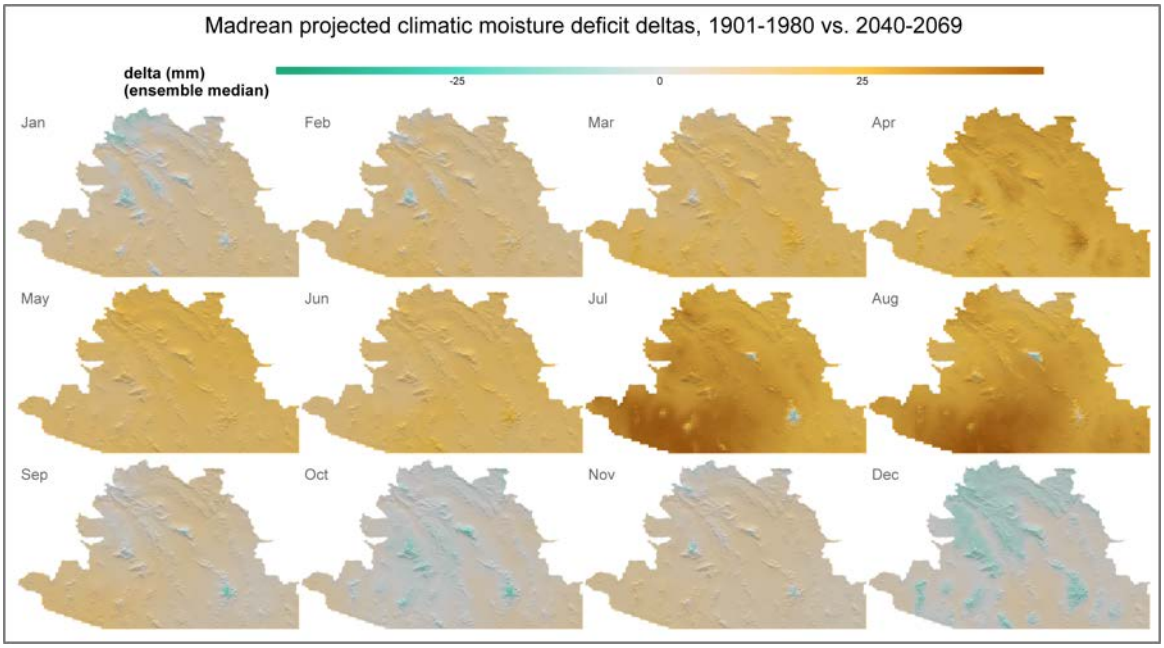


**Figure I-26. MAR projected monthly climatic moisture deficit delta ratios, 1901-1980 vs. 2040-2069.**

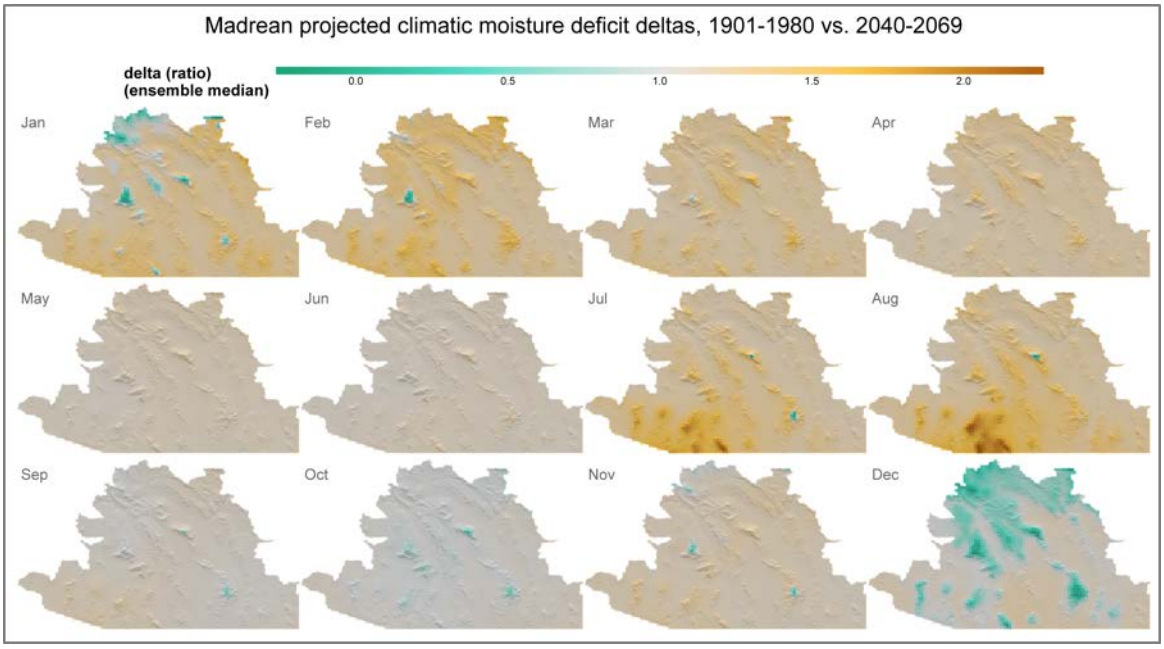
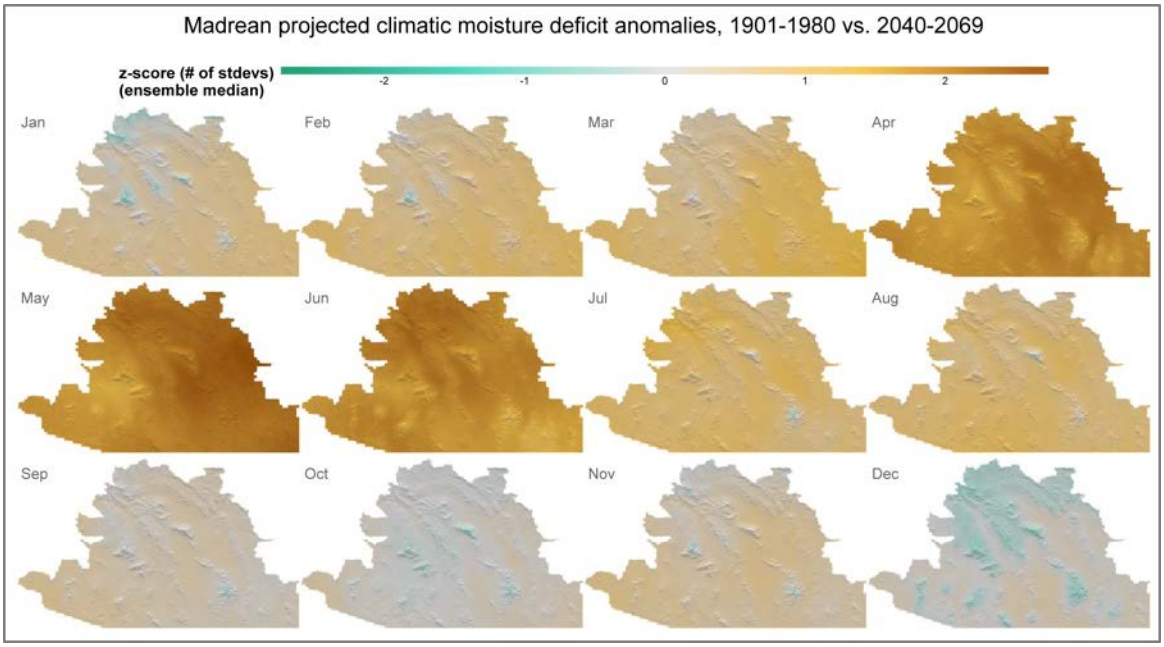



**Figure I-27. MAR projected monthly climatic moisture deficit anomalies, 1901-1980 vs. 2040-2069.**



### I.5.3.2 Number of Frost-Free Days

Minimum temperature data can be used to calculate changes in the number of frost-free days (NFFD) in each month of the year. The warmest months (June through September) had no frost during the baseline for most MAR locations, so NFFD for those months could not increase, and anomalies could not be calculated. All other months are projected to see increases in NFFD, with the greatest changes occurring in the shoulder season months of April and November, where high elevations are projected to see up to 10 fewer nights of temperatures falling below 0 degrees C. From December through March, NFFD increases are forecast to occur throughout the MAR, with delta values approaching 8 days in the western portions of the region and lesser change in the east. Z-scores for these changes are relatively uniform across time space, suggesting that NFFD will be roughly 2 standard deviations higher by the middle of this century than during the 20$^{th}$ century baseline.

**Figure I-28. MAR projected monthy number of frost-free days deltas, 1901-1980 vs. 2040-2069.**

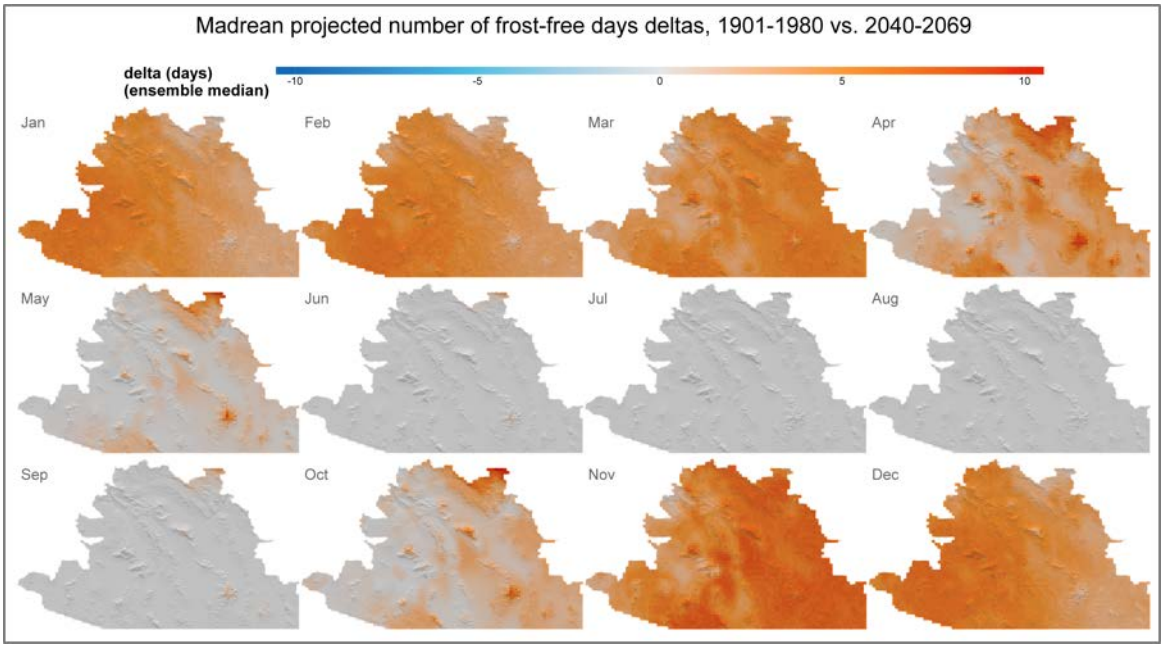
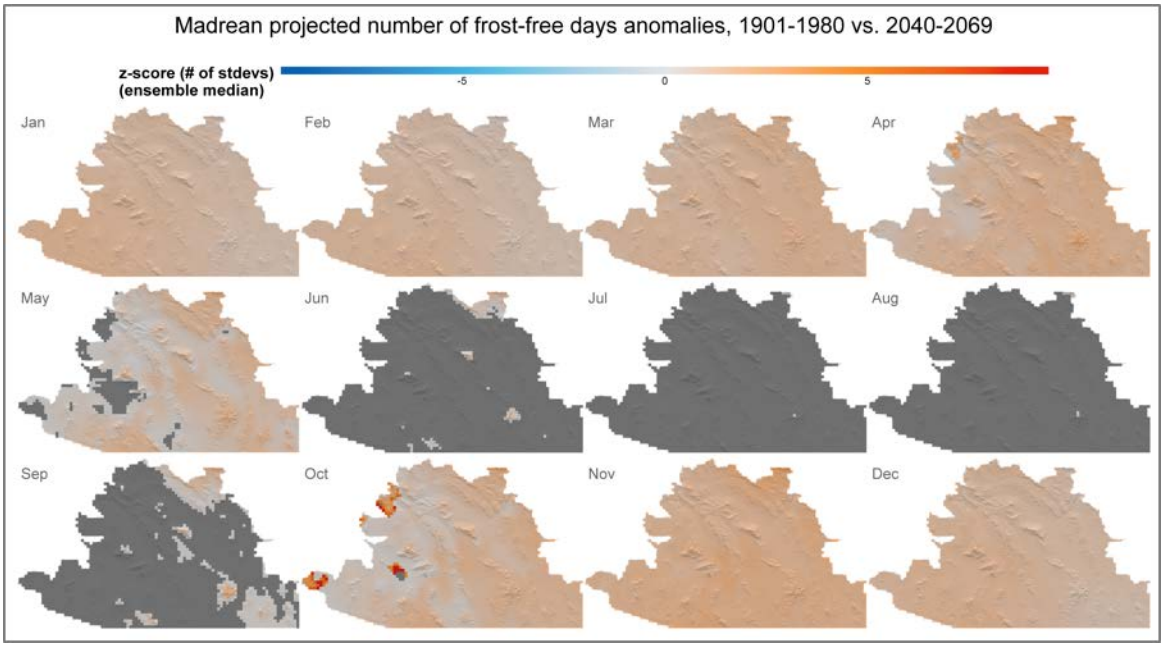


**Figure I-29. MAR projected monthly number of frost-free days anomalies, 1901-1980 vs. 2040-2069.** The dark grey areas represent no data because these are warm months that do not have frost, therefore anomalies cannot be calculated.



### I.5.3.3 Frost-Free Period

Frost-free period (FFP) is an annual variable derived from minimum temperature data, representing the number of consecutive days between the last frost of the spring and the first frost of the fall. Climate models project that the average length of the warm season will increase dramatically in the MAR by the mid 21st century, with FFP increasing by anywhere between 25 days (in the eastern MAR) and 55 days (in the west). These changes represent anomalies of 2 to 4 standard deviations from the baseline mean.

**Figure I-30. MAR projected annual frost-free period deltas, 1901-1980 vs. 2040-2069.**

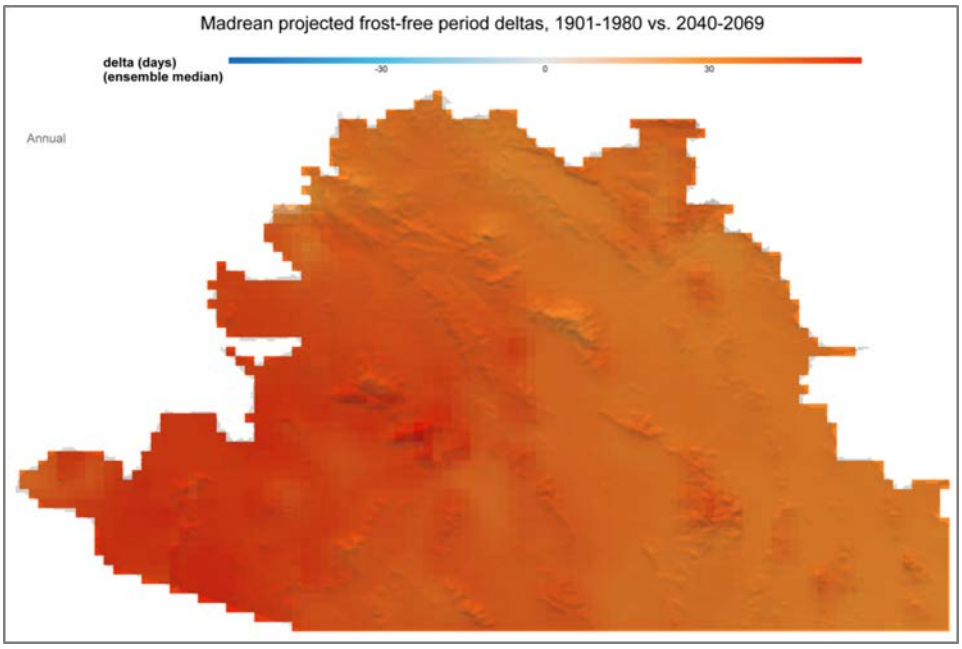
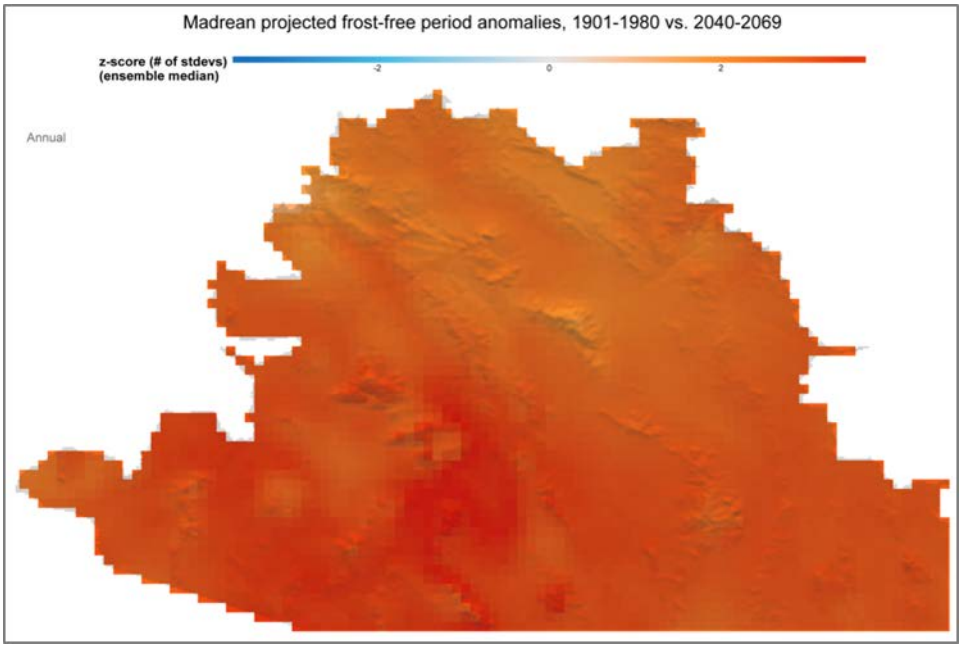



**Figure I-31. MAR projected annual frost-free period anomalies, 1901-1980 vs. 2040-2069.**



## *I.6 Bioclimatic Envelope Modeling*

**Purpose:** Bioclimatic envelope modeling was used to help understand the potential impacts of climate change for CEs of interest by analyzing projected shifts in suitable climate conditions.

**Summary:** The process used in this study characterizes a CE's suitable bioclimate by relating known current CE localities to current climate variables, and then comparing where that combination of variables occurs in the present day versus in various models of the future. While vegetation communities are unlikely to migrate smoothly in the fashion projected by these models, the range shift maps can be useful in management prioritization by suggesting which portions of the current CE range may experience the greatest and least climate stress in future decades.

Bioclimatic envelope modeling was performed for four CEs, which were pre-selected from the full suite of CEs based on their relative importance and suitability to this modeling approach:

1. Apacherian-Chihuahuan semi-desert grassland and steppe
2. Chihuahuan creosotebush desert scrub
3. Madrean encinal
4. Apacherian-Chihuahuan mesquite upland scrub

**Source Data:** The 4km Climate Western North America dataset (Wang et al. 2012) was used for each of 12 months for each of three climate variables: minimum temperature, maximum temperature, and total precipitation in order to define a current and projected bioclimate. NatureServe provided distribution locations for each of the CEs as inputs for the envelope models, from their Terrestrial Ecosystems map (NatureServe 2013).

**Process Steps:**

1. In order to predict how climate change may shift the suitable climatic conditions for a vegetation assemblage, its bioclimactic niche is defined by correlating its current range with current climatic conditions. A current bioclimatic envelope was modeled for each CE using point localities for each CE, 36 baseline spatial climate data layers (the 1961-1990 mean for each of 12 months for each of three climate variables: minimum temperature, maximum temperature, and total precipitation), and the species distribution modeling algorithm MaxEnt (Elith et al. 2011, Philips et al. 2006). The 1961-1990 baseline was chosen because this is the baseline that was used in the Climate Western North America dataset to downscale raw GCMs outputs to a 4km grid.
2. This current bioclimate envelope was then projected into the future mid-century timeslice (2040-2069) for each of the 6 GCMs under the A2 emissions scenario (for climate models used see Table I-1). These results were compiled to produce a final map of suitable bioclimate in 2050 with a classified color scale representing model agreement.
3. In order to summarize change in bioclimate for a given CE, a change layer was created by calculating the difference between modeled current bioclimate and projected future suitability. In order to calculate change, 2050 model agreement maps were reclassified to a presence/absence layer with presence defined as locations where at least 2 out of 6 GCMs agree that there will be suitable conditions in 2050. The resulting change summary layer was generated representing areas of projected future stability, contraction, and expansion for each CE's bioclimatic envelope.

**Derived Data:**

1. Raster of modeled baseline bioclimatic envelope for each CE.

2. Raster of modeled future projected bioclimatic envelope with degree of model agreement for each CE.
3. Summary raster representing the areas of stability, contraction, and expansion for each CE's bioclimatic envelope based on a threshold of model agreement (2 out of 6 models).

**MaxEnt and Species Distribution Modeling**

Maxent is a machine learning algorithm that uses the principle of maximum entropy to estimate a set of functions that relate environmental variables and species known occurrences in order to approximate species' niche and potential geographic distribution. Maxent was chosen because of its established performance with presence-only data relative to alternative niche modeling techniques, and its built-in capacity to deal with multi-colinearity in the environmental variables (Elith et al. 2006, Elith and Leathwick 2009). Maxent calculates a surface of probability across geographic space, where each cell has a value of the probability that a species niche will occur there at a given time. Maxent focuses on how the environment where the species is known to occur relates to the environment across the rest of the study area. The model does not identify the species fundamental niche; but only that part of the niche defined by the observed records.

**Threshold Selection**

In order to translate the raw Maxent probability distribution into estimates of species presence or absence, a specific threshold had to be selected, a necessary post-processing step when using multiple GCMs. The threshold used in this analysis is the "equal training sensitivity plus specificity" threshold. This threshold maximizes the agreement between observed and predicted distributions, a choice that has proven to produce the most accurate predictions (Jimenes-Valverde and Lobo 2007)

**Model Evaluation**

Model evaluation was performed using the area under the curve (AUC) of the receiver operating characteristic (ROC) plot analysis. Twenty percent of occurrence points for a given conservation element were withheld from the model to be used as independent test data in calculating the AUC. The AUC is a widely accepted, threshold-independent metric of species distribution model performance that provides an overall picture of how well the data fits the model and has previously been used in comprehensive SDM evaluations (Elith et al. 2006).

## I.6.1 Results

### I.6.1.1 Apacherian-Chihuahuan Semi-desert Grassland and Steppe

The bioclimatic envelope of the Apacherian-Chihuahuan grassland community is currently widely distributed across the southern and eastern MAR, but is projected to contract significantly by mid-century both within the MAR and throughout the entire distribution. Large regions of projected expansion also exist, but these fall almost entirely outside the MAR boundary. Of the four CE bioclimates modeled, the Apacherian-Chihuahuan grassland was projected to have the largest area of contraction as a percentage of its total current distribution, with relatively little overlap between modeled current and future distributions. Based on projected shifts, foothills and mid-elevation habitats may be the most stable in the future. There is relatively high confidence in model performance for this vegetation assemblage with an AUC of 0.98. The most influential climate variables to the model projections are winter and summer maximum temperatures and spring precipitation (see Table I-2 for specific percent contributions).

**Figure I-32. Modeled current bioclimatic envelope (1961-1990) for Apacherian-Chihuahuan Semi-desert Grassland and Steppe**.

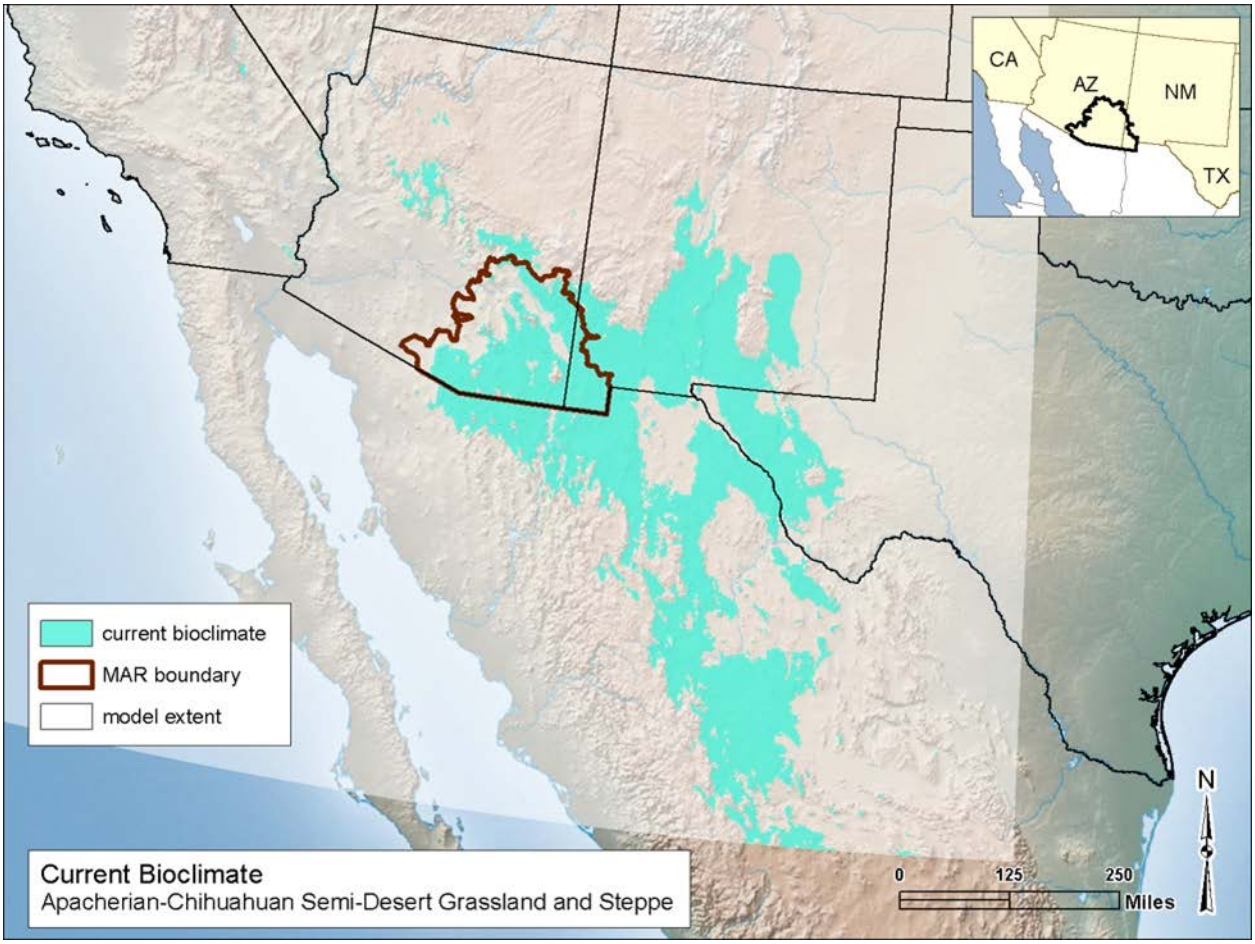


**Figure I-33. Mid-century future modeled suitable bioclimate for Apacherian-Chihuahuan Semi-desert Grassland and Steppe**. Areas in blue indicate high model agreement and therefore greater confidence in future suitable bioclimate. Foothills and mid-elevations are areas of high model agreement that there will be suitable conditions in the future.

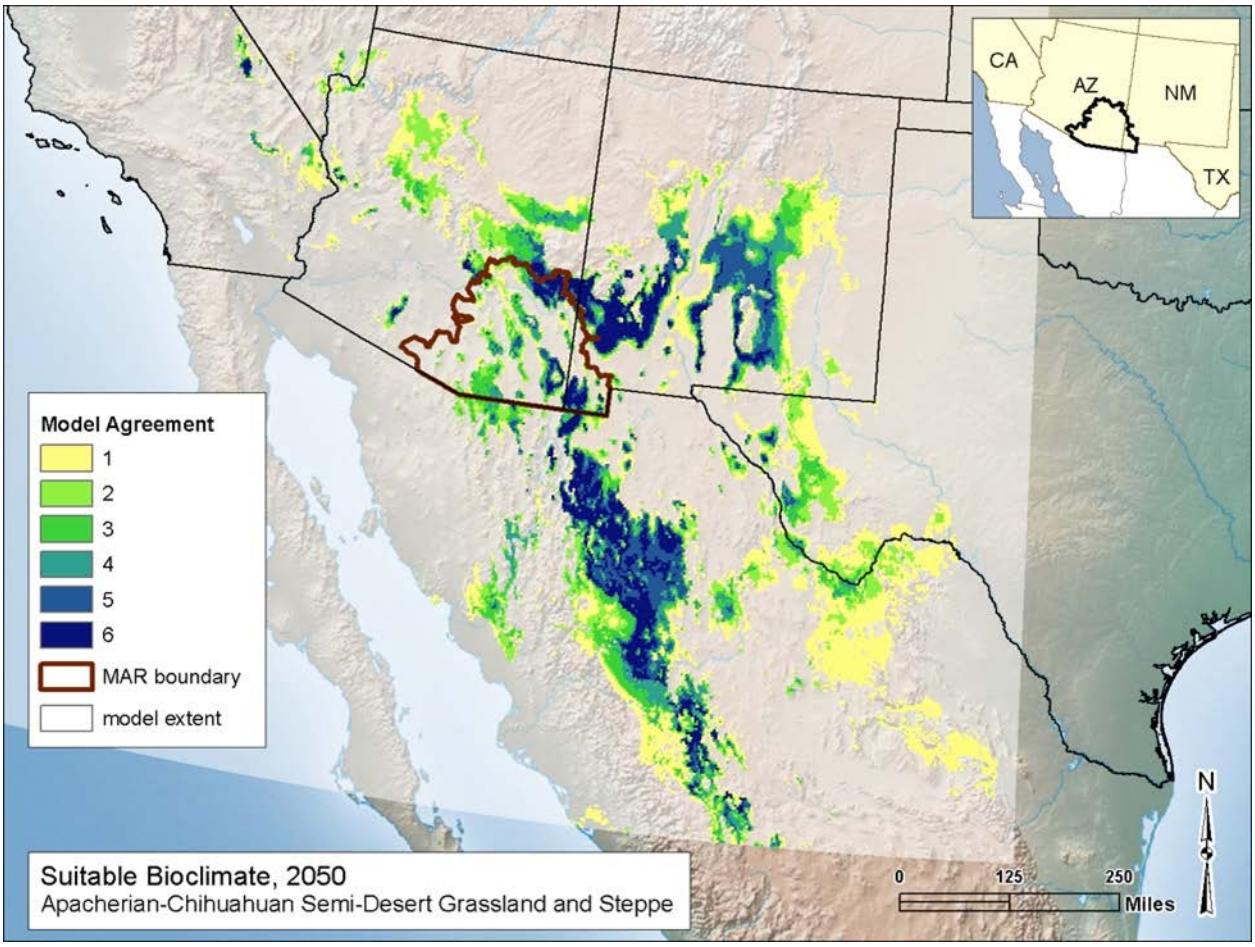


**Figure I-34. Mid-century future modeled suitable bioclimate for Apacherian-Chihuahuan Semi-desert Grassland and Steppe.** This map depicts how current suitable bioclimate is projected to shift by showing areas of contraction, expansion, and overlap (stability). Projections indicate that there will be significant contraction of suitable conditions within the MAR, with some mid-elevation areas of stability.

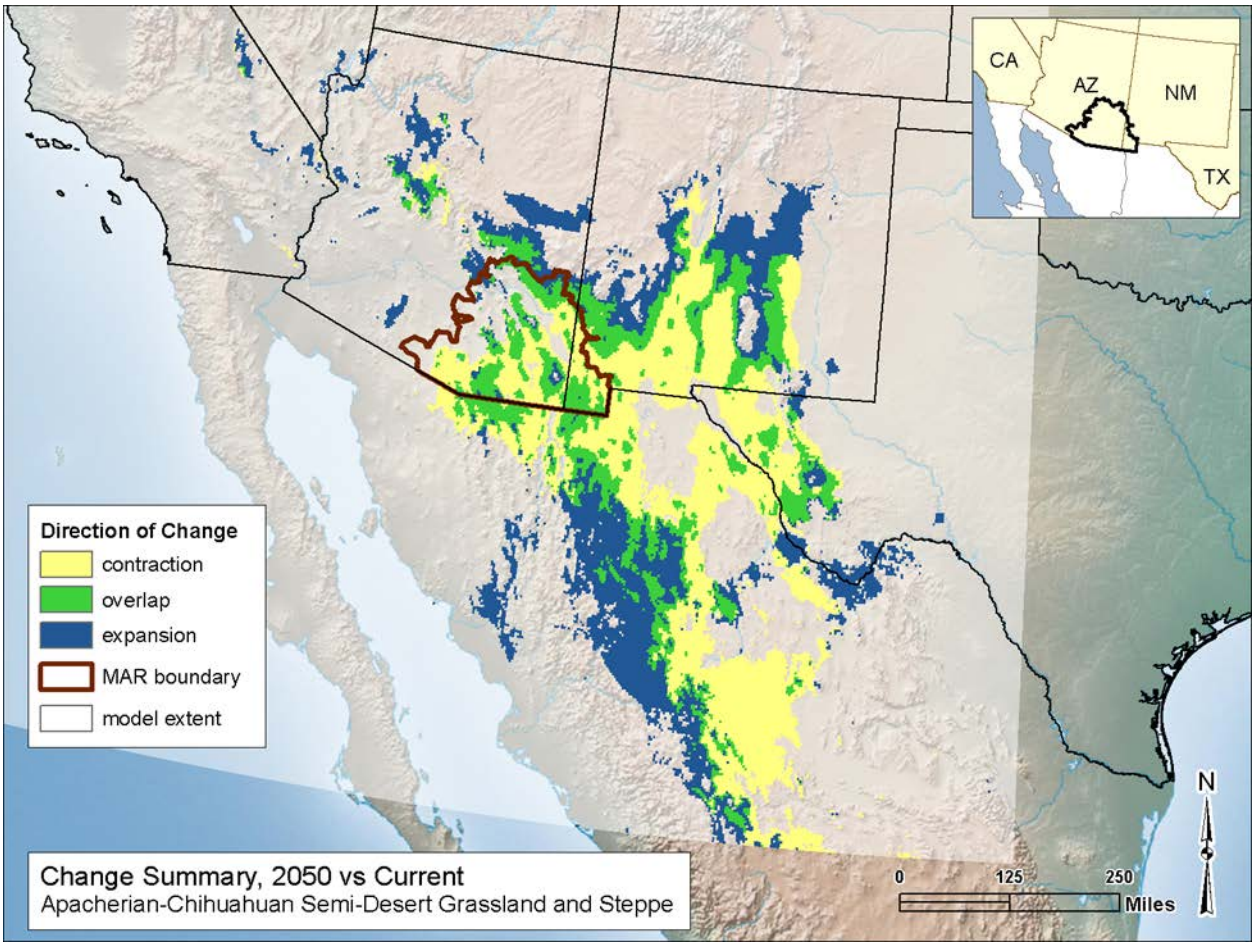



**Table I-2. Analysis of 5 highest variable contributions to the Apacherian-Chihuahuan Semi-desert Grassland and Steppe bioclimatic envelope model.**

| Variable | Percent contribution |
|---|---|
| Tmax 12 | 29.8 |
| Tmax 06 | 26.7 |
| Precip 04 | 7.8 |
| Tmax 01 | 6 |
| Precip 08 | 5.7 |

#### I.6.1.2 Chihuahuan Creosotebush Desert Scrub

The MAR lies at the fringes of the modeled current Chihuahuan Creosotebush Desert Scrub bioclimate, and while this bioclimate is projected largely to remain stable or expand, it is projected to contract significantly within the MAR. Contraction is projected for central and northeastern parts of the MAR, while stability is projected for the southeastern MAR. Spring maximum temperature contributed most to model projections, with May and April maximum temperatures comprising 59% of variable contributions (Table I-3). Model performace was validated with an AUC of 0.973.

**Figure I-35. Modeled current bioclimatic envelope (1961-1990) for Chihuahuan Creosotebush Desert Scrub**.

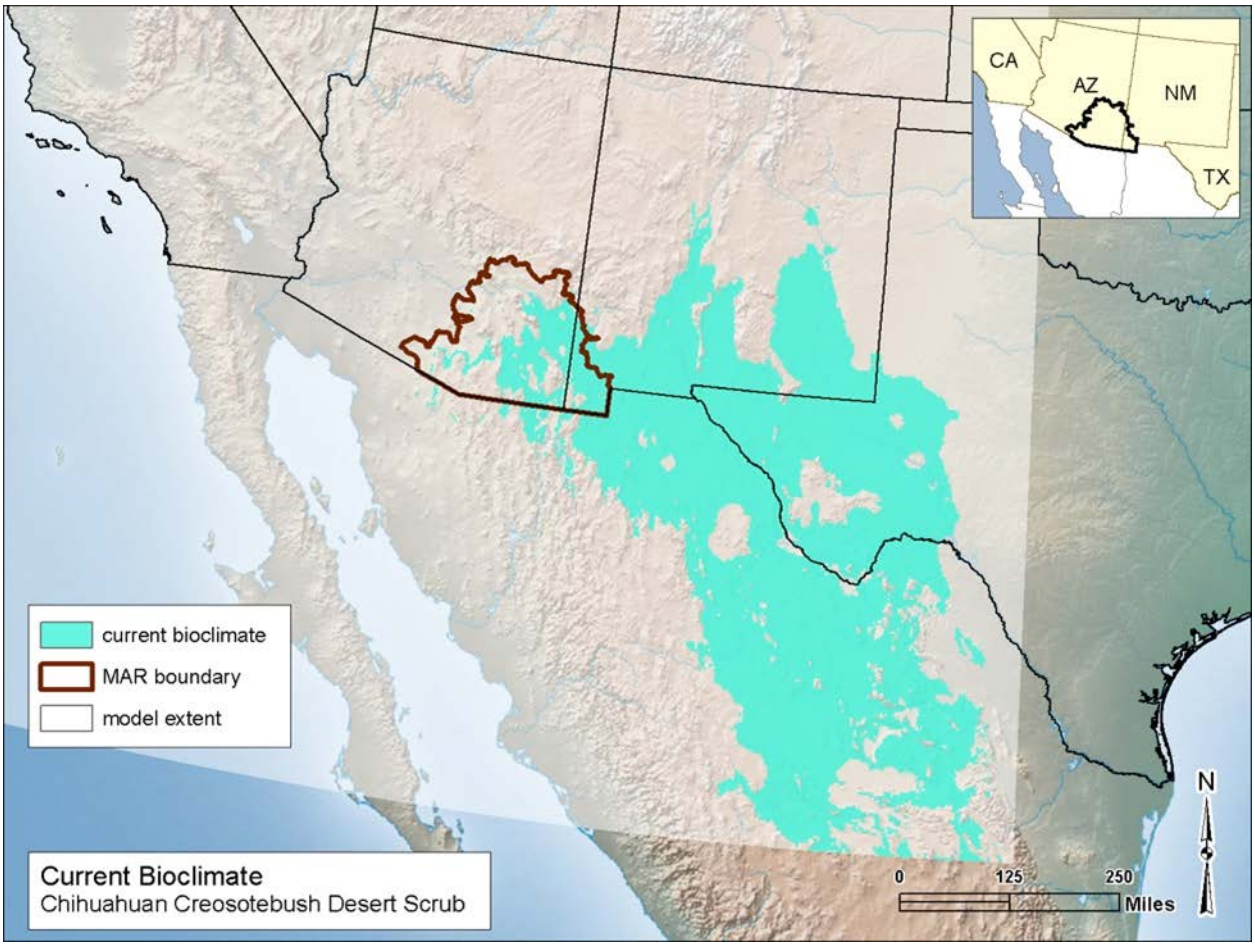


**Figure I-36. Mid-century future modeled suitable bioclimate for Chihuahuan Creosotebush Desert Scrub**. Areas in blue indicate high model agreement and therefore greater confidence in future suitable bioclimate.

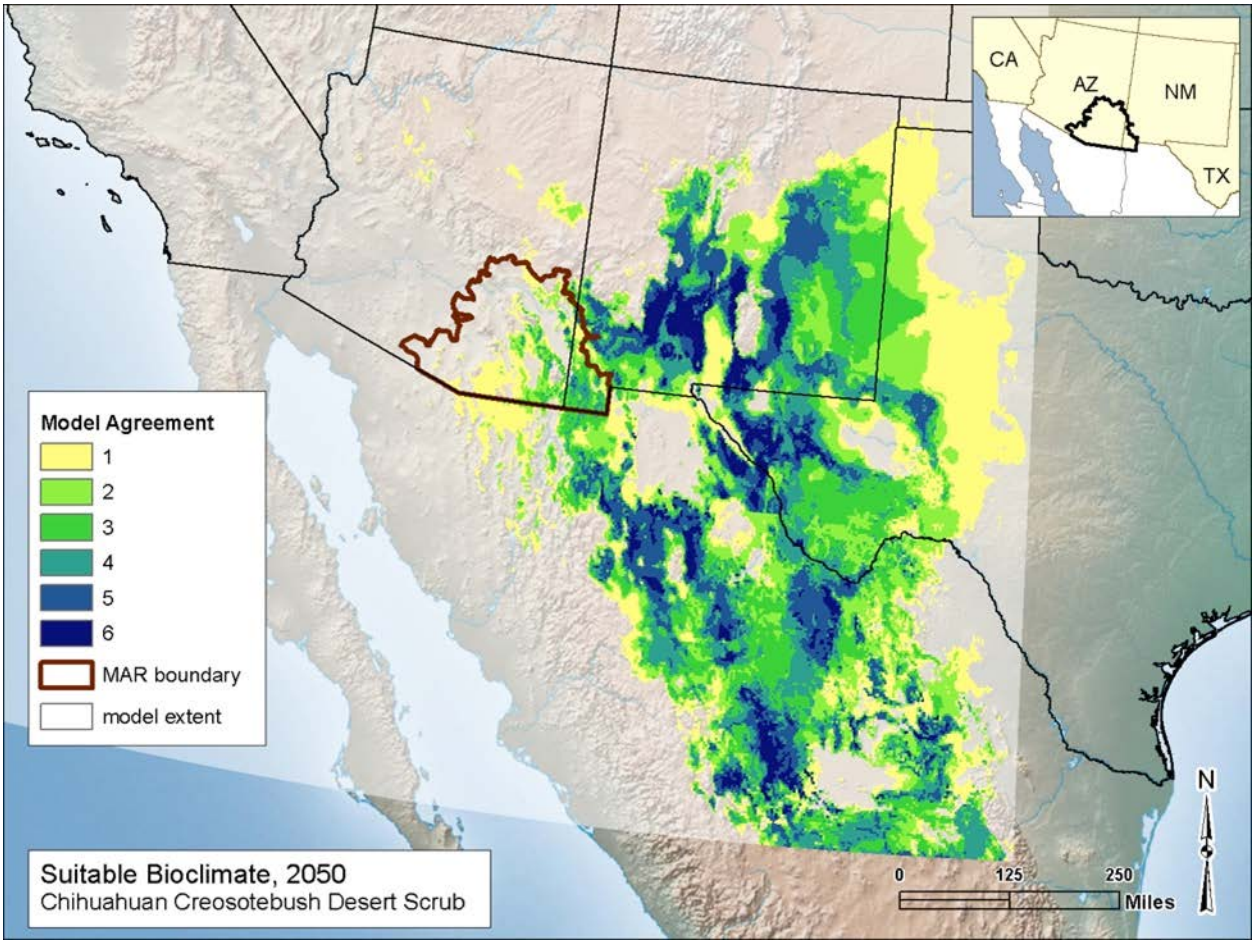


**Figure I-37. Mid-century future modeled suitable bioclimate for Chihuahuan Creosotebush Desert Scrub**. This map depicts how current suitable bioclimate is projected to shift by showing areas of contraction, expansion, and overlap (stability).

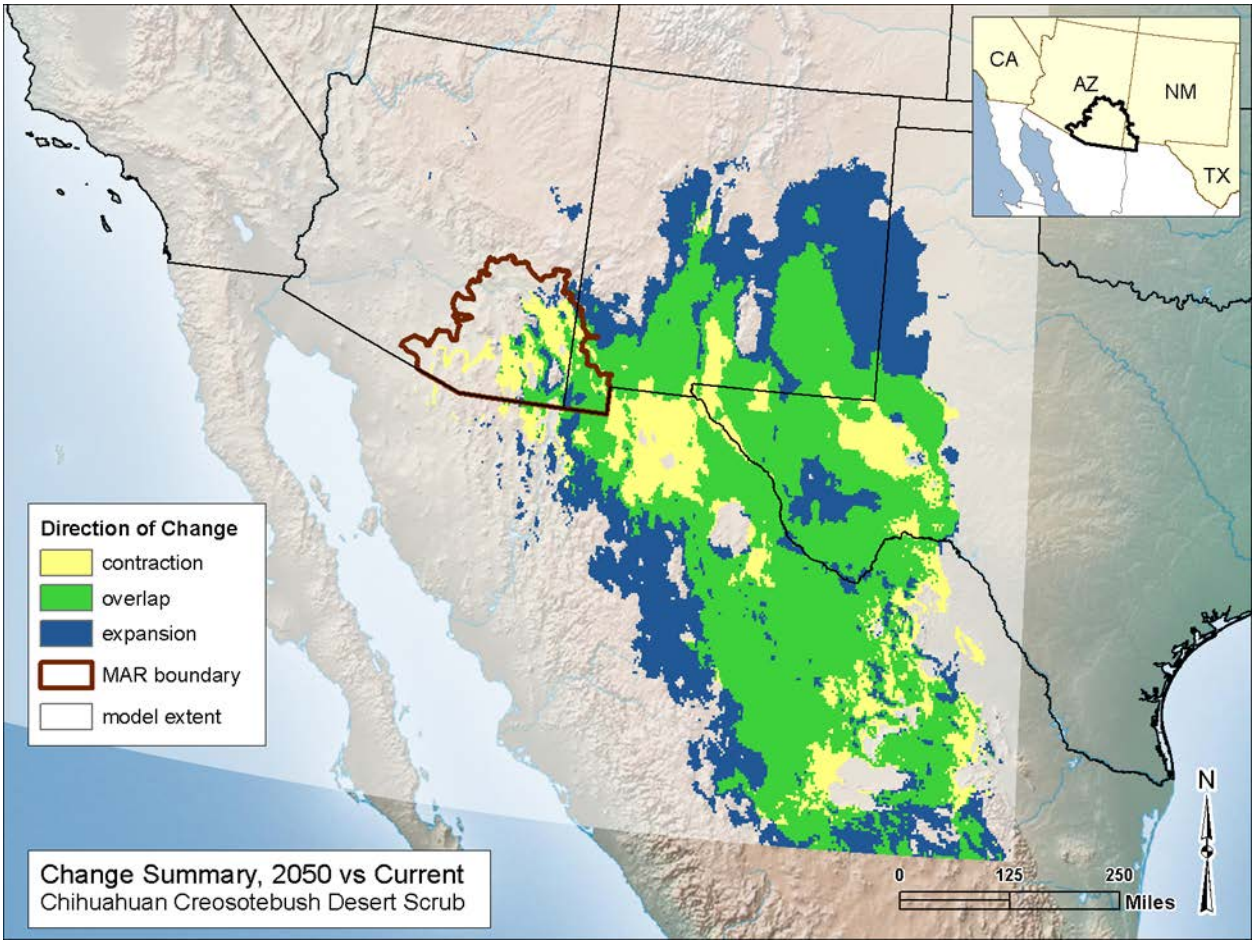



**Table I-3. Analysis of 5 highest variable contributions to the Chihuahuan Creosotebush Desert Scrub bioclimatic envelope model.**

| Variable | Percent contribution |
|---|---|
| Tmax 05 | 43.1 |
| Tmax 04 | 16 |
| Tmax 02 | 6.3 |
| Precip 03 | 6 |
| Precip 06 | 5.8 |

### I.6.1.3 Madrean Encinal

Across its entire current distribution, suitable bioclimate for Madrean Encinal is projected to contract at lower elevations and move uplslope, while mid-elevations are projected to remain stable. A majority of the area within the MAR boundary is contracting (rather than remaining stable or expanding). High model agreement of future suitable conditions exists mainly at high elevations. Changes in fall maximum temperature and summer precipitation contribute significantly to model projections (Table I-4). Model performance is relatively robust with an AUC of 0.985.

**Figure I-38. Modeled current bioclimatic envelope (1961-1990) for Madrean Encinal**.

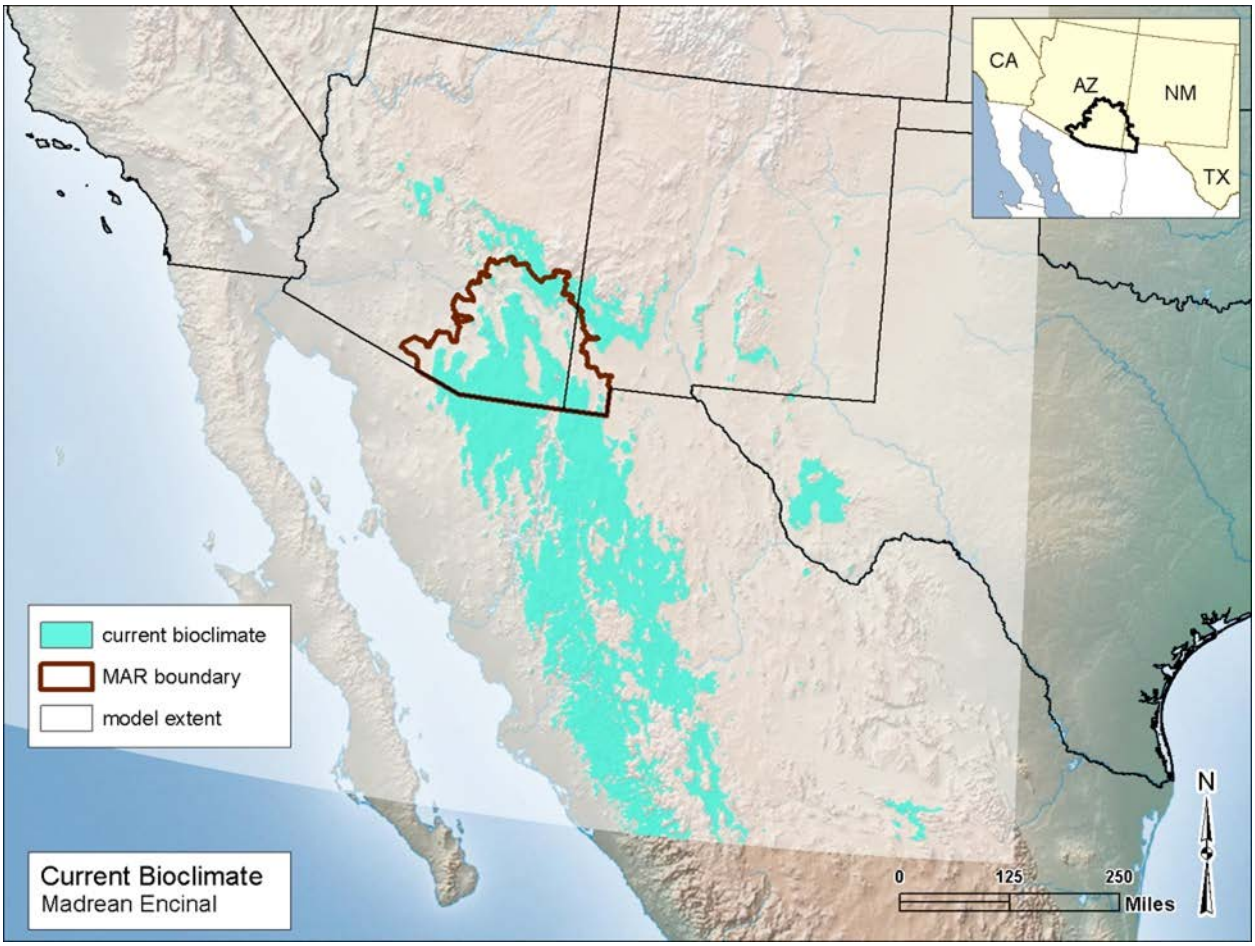


**Figure I-39. Mid-century future modeled suitable bioclimate for Madrean Encinal**. Areas in blue indicate high model agreement and therefore greater confidence in future suitable bioclimate. Model agreement is highest in mountainous regions especially within the MAR boundary.

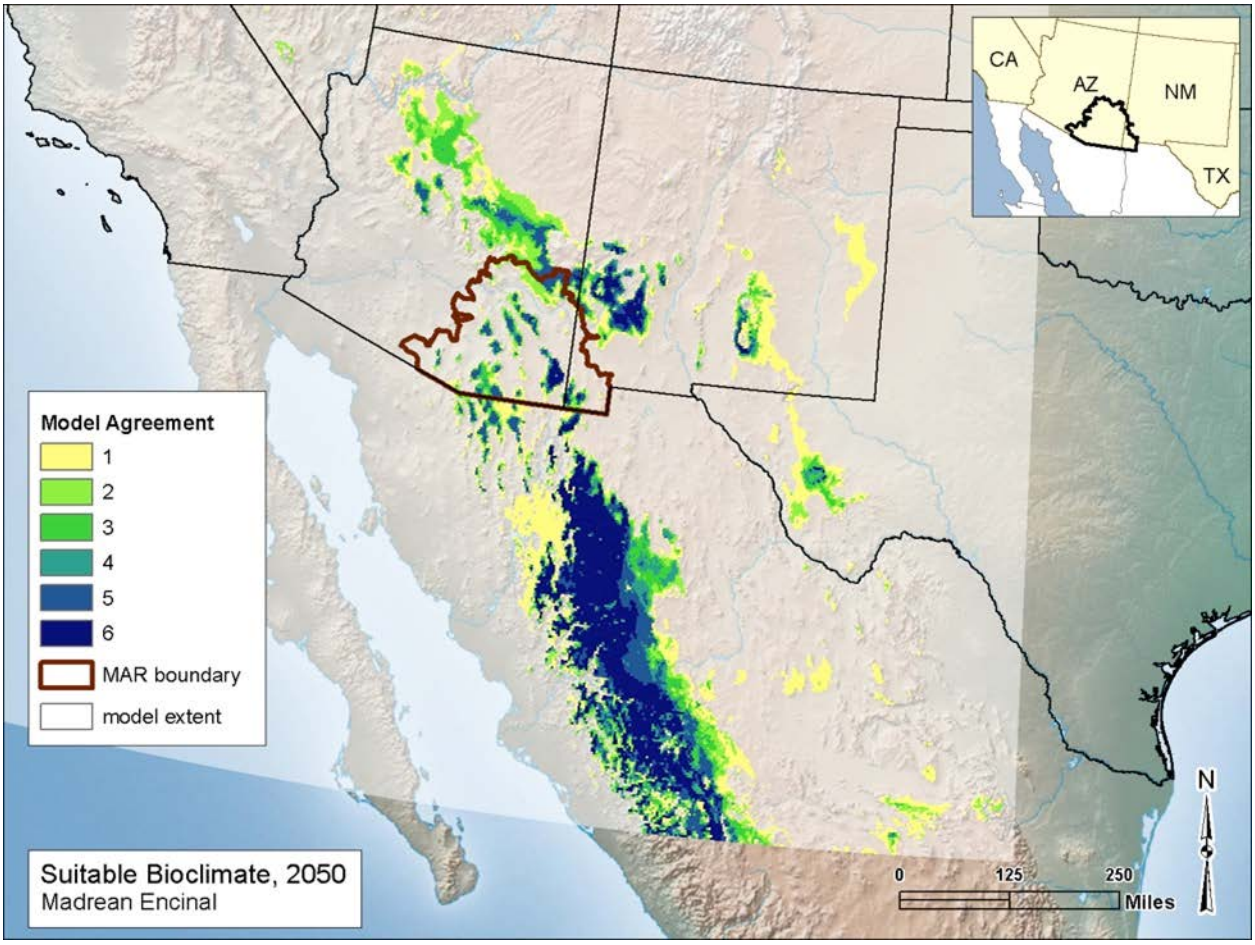


**Figure I-40. Mid-century future modeled suitable bioclimate for Madrean Encinal.** This map depicts how current suitable bioclimate is projected to shift by showing areas of contraction, expansion, and overlap (stability). Future bioclimate is projected to shift upslope and mid-elevations are potential areas of stability.

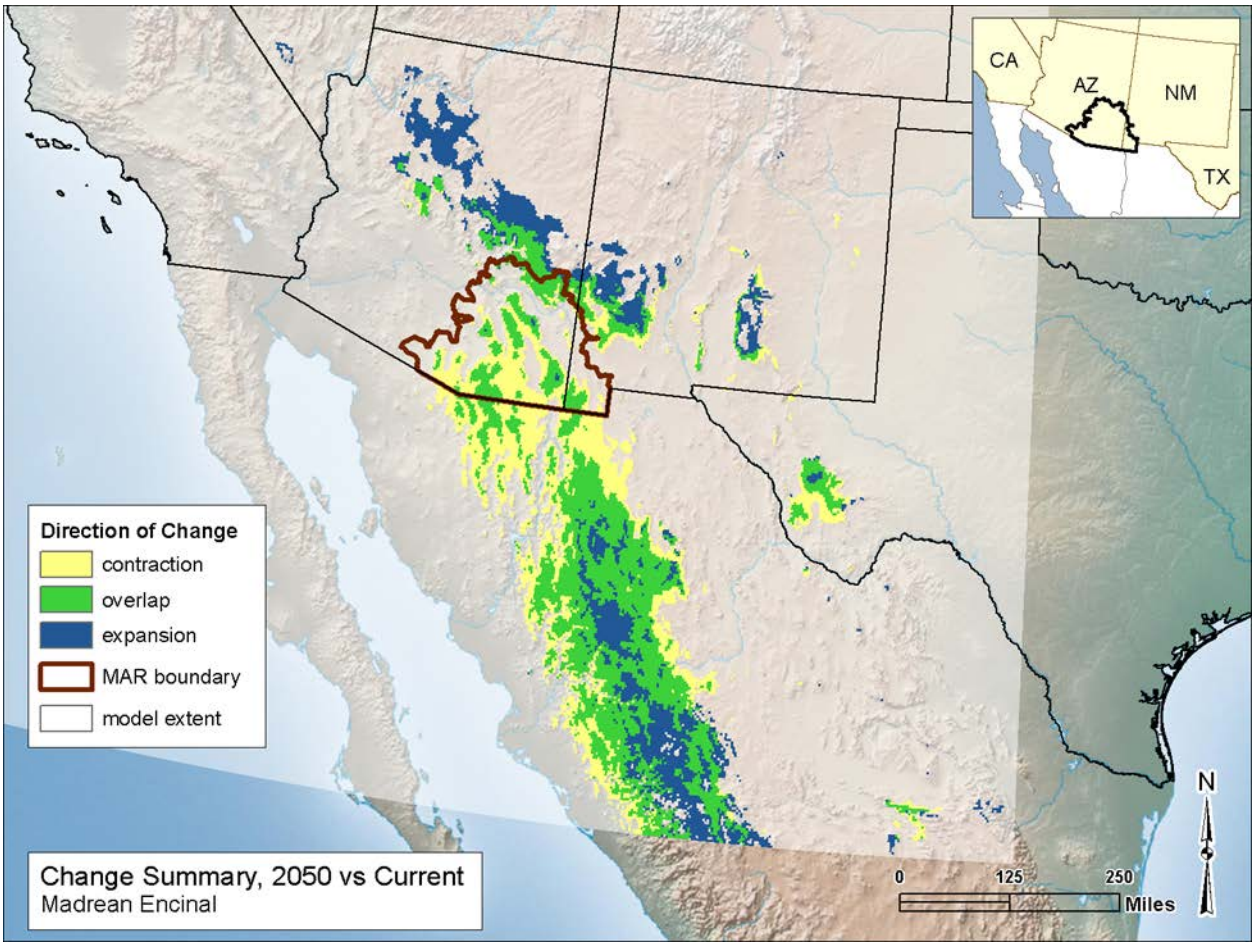



**Table I-4. Analysis of 5 highest variable contributions to the Madrean Encinal bioclimatic envelope model.**

| Variable | Percent contribution |
|---|---|
| Tmax 11 | 29.4 |
| Precip 07 | 18 |
| Tmax 01 | 13.5 |
| Precip 08 | 10.3 |
| Tmin 07 | 4.8 |

#### I.6.1.4 Apacherian-Chihuahuan Mesquite Upland Scrub

Of the four CEs modeled, Apacherian-Chihuahuan Mesquite Upland Scrub had the most favorable projection for future bioclimate stability and expansion both within the MAR boundary and across its entire range. Bioclimate for the Mesquite Upland Scrub is projected to persist and expand in every direction except some areas of contraction in Texas. Within the MAR boundary some small shifts toward higher elevations are projected. Bioclimate for mesquite upland shrub was projected to remain favorable in virtually all the MAR locations where Apacherian-Chihuahuan grassland bioclimate was projected to contract, suggesting that mesquite may continue to expand into desert grasslands in the future. Model agreement of suitable future bioclmate is high within the MAR boundary. Summer maximum temperature is the main predictor variable driving model projections (Table I-5) and winter maximum and minimum temperatures are also significant contributors. Model performance is validated with an AUC of 0.972.

**Figure I-41. Modeled current bioclimatic envelope (1961-1990) for Apacherian-Chihuahuan Mesquite Upland Scrub**.

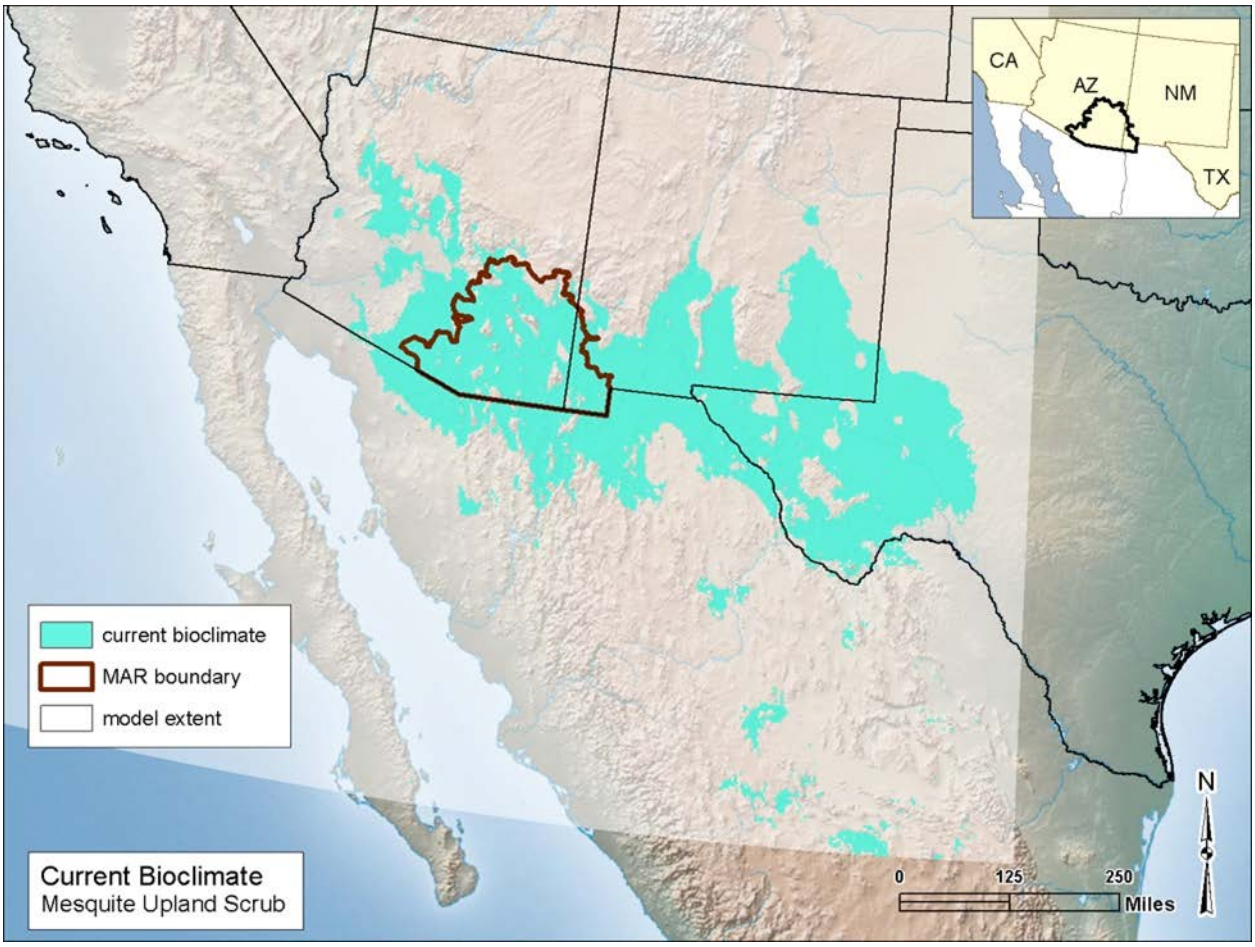


**Figure I-42. Mid-century future modeled suitable bioclimate for Apacherian-Chihuahuan Mesquite Upland Scrub**. Areas in blue indicate high model agreement and therefore greater confidence in future suitable bioclimate.

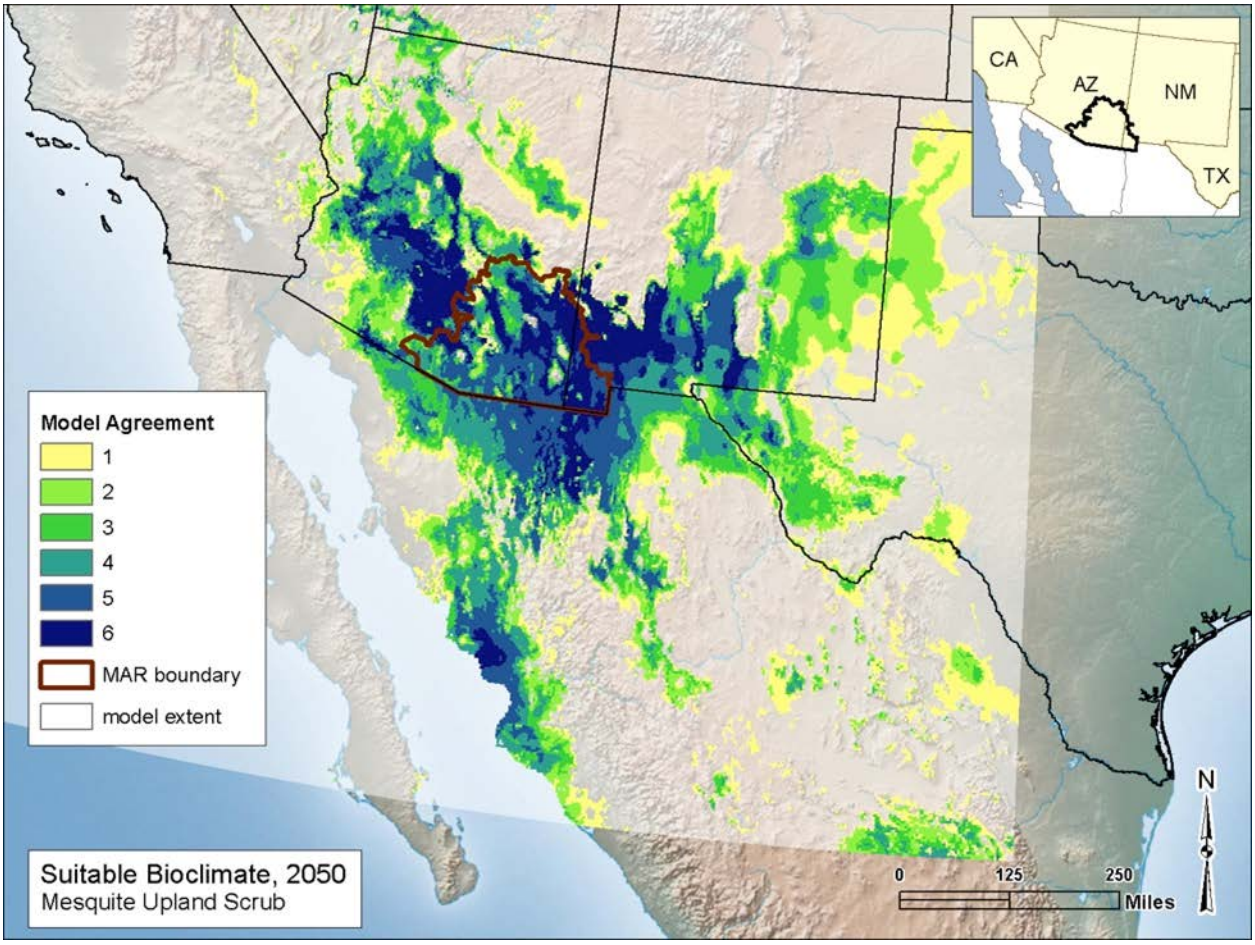


**Figure I-43. Mid-century future modeled suitable bioclimate for Apacherian-Chihuahuan Mesquite Upland Scrub**. This map depicts how current suitable bioclimate is projected to shift by showing areas of contraction, expansion, and overlap (stability). The bioclimate for this vegetation assemblage is projected to remain stable and increase in some areas. Within the MAR boundary some pixels are highlighted as shifting upslope.

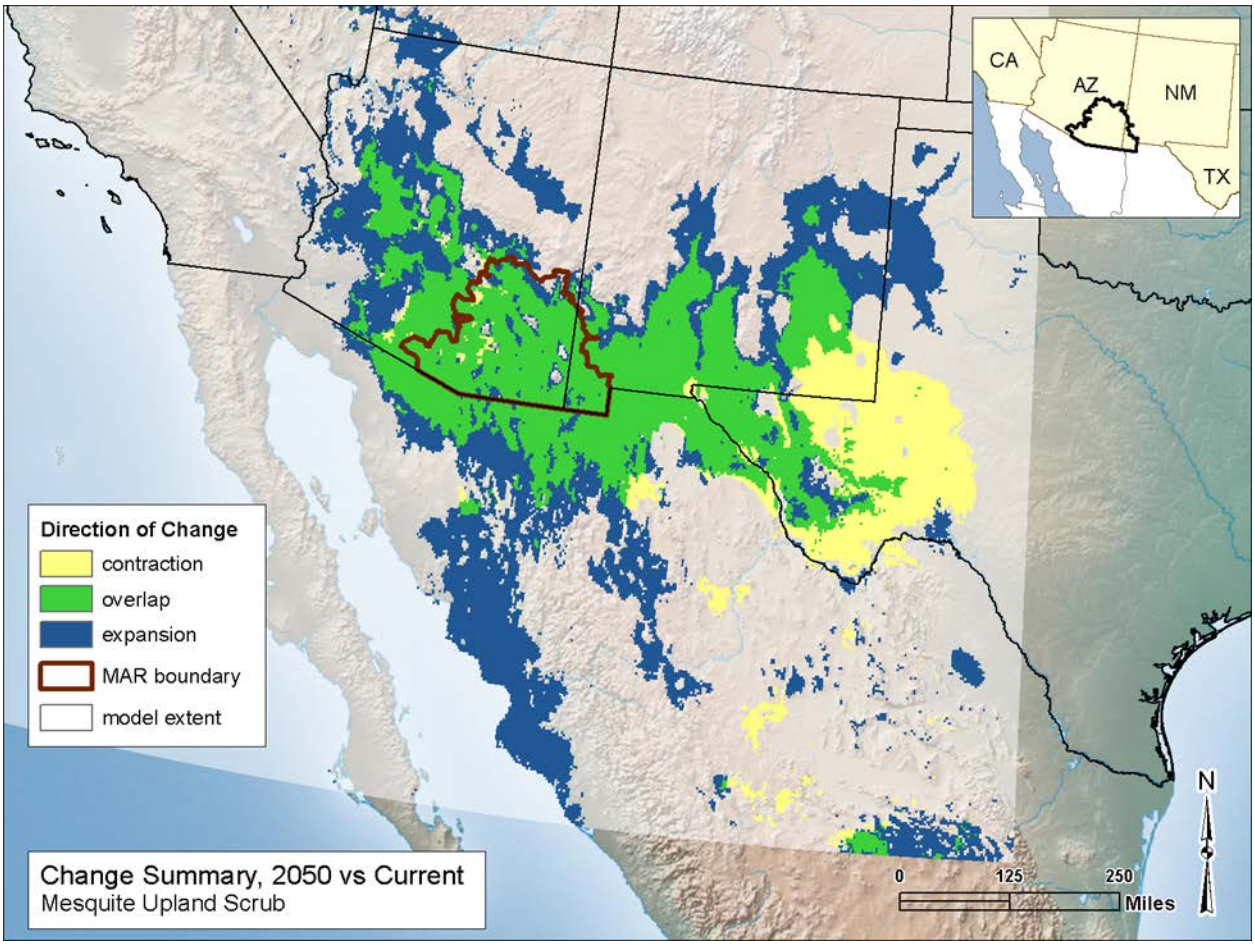



**Table I-5. Analysis of 5 highest variable contributions to the Apacherian-Chihuahuan Mesquite Upland Scrub envelope model.**

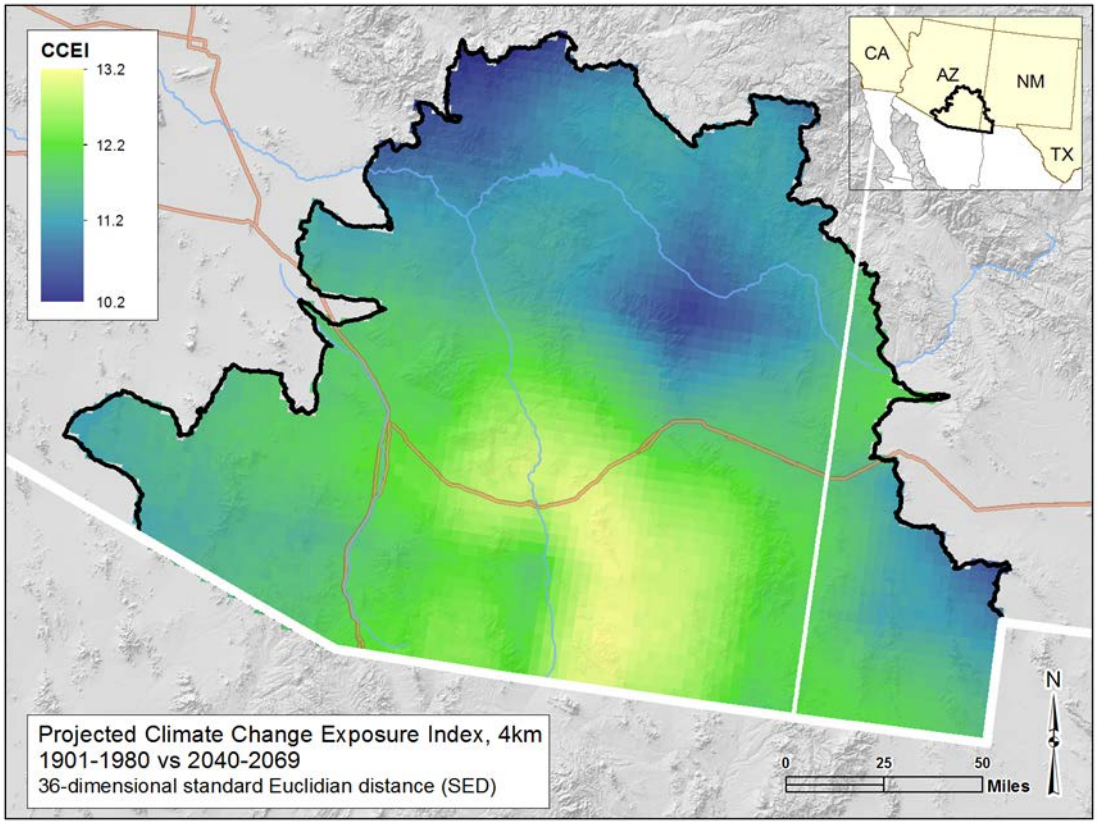
| Variable | Percent contribution |
|---|---|
| Tmax 06 | 47.1 |
| Tmax 02 | 31.9 |
| Tmin 02 | 6.6 |
| Precip 06 | 3.4 |
| Precip 08 | 3.1 |

## *I.7 Future Climate Change Exposure Index Overlay with CEs*

This assessment seeks to inform the relative degree of climate change the conservation elements (CEs) may be exposed to in the future. This is accomplished with a simple overlay of the CE distribution maps on the climate change index map (section I.5.2). As seen in the projected future Climate Change section, ecologically the greatest climate change is projected to be decreases in spring precipitation, with May showing the greatest decrease, especially in the western portion of the ecoregion and an increase in precipitation in the fall across the entire ecoregion (Figure I-19). In addition, it is interesting to note that the future projected precipitation change indicates a reversal of the observed recent drying trend (compare recent anomalies Figure I-14 with projected anomalies Figure I-19). This may be due to an increase in the number of El Niño events that create conditions for increased winter rains and therefore increased stream runoff (Hirshboeck 2009). Minimum (night-time) temperatures are projected to increase across all months across the entire ecoregion (Figure I-21) with the greatest increases in July and August. This means that the areas will not cool off at night, but remain warm. Maximum (day-time) temperatures are also projected to increase across the MAR (Figure I-23) with greatest increases projected to occur in July and August, as well in the Spring and Fall months. Together these projected climate changes mean significant changes to the entire MAR ecoregion. The climate change index summarizes these changes into a single value per pixel, which clearly shows no area of the MAR is going to escape climate change impact (Figure I-24).

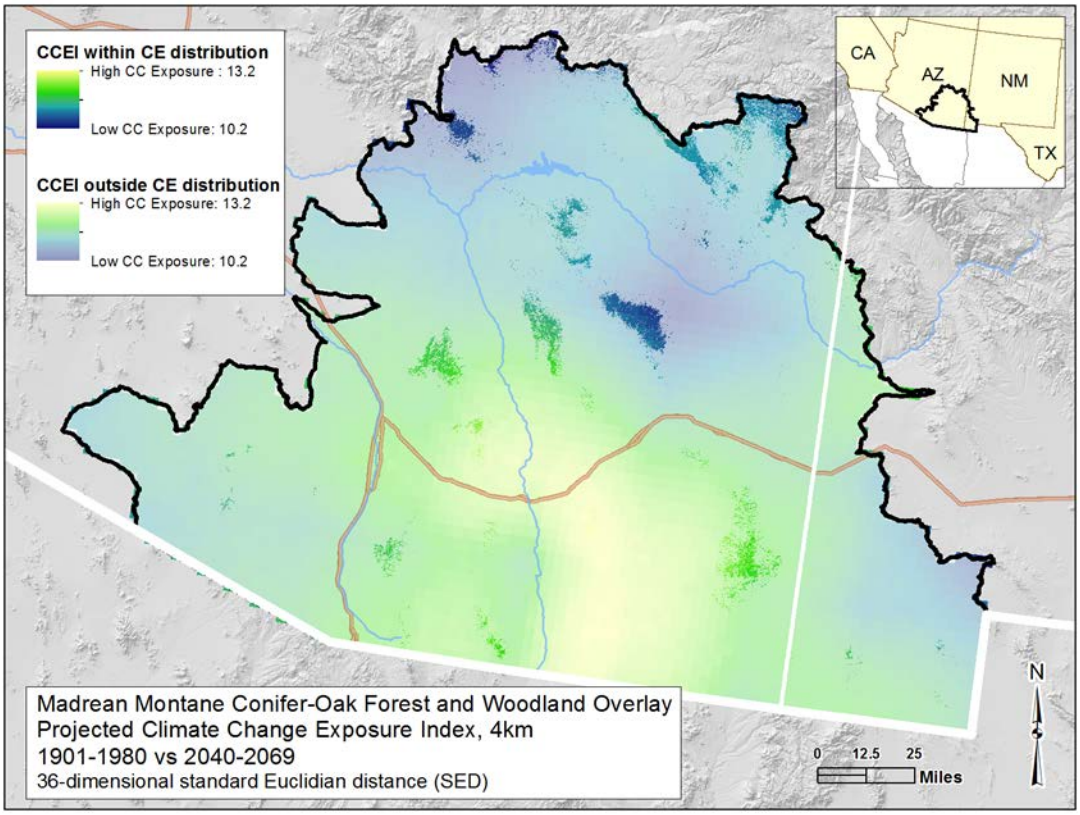
In the following figures, the index values are scaled to show just the upper quartile of the index in order to display those differences spatially. Lighter colors indicate fewer variables are significantly different, and darker colors indicate many if not all of 36 climate variables are projected to be significantly different from the baseline period. However, the darker colors should not be perceived as “low climate change” because all areas are expected to experience significant change. These figures illustrate that the south-central portions of the ecoregion have not only significant minimum and maximum temperatures increases for every month, but also the highest magnitude of change for July and August minimum temperatures (Figure I-44 and see Figure I-21).

**Figure I-44. Climate Change Exposure Index.** CCEI is in units of standard deviation of projected future departure from historic variability. Scale shows upper values from 10.2-13.2 (instead of 0-13.2) to show variation of exposure across the landscape.



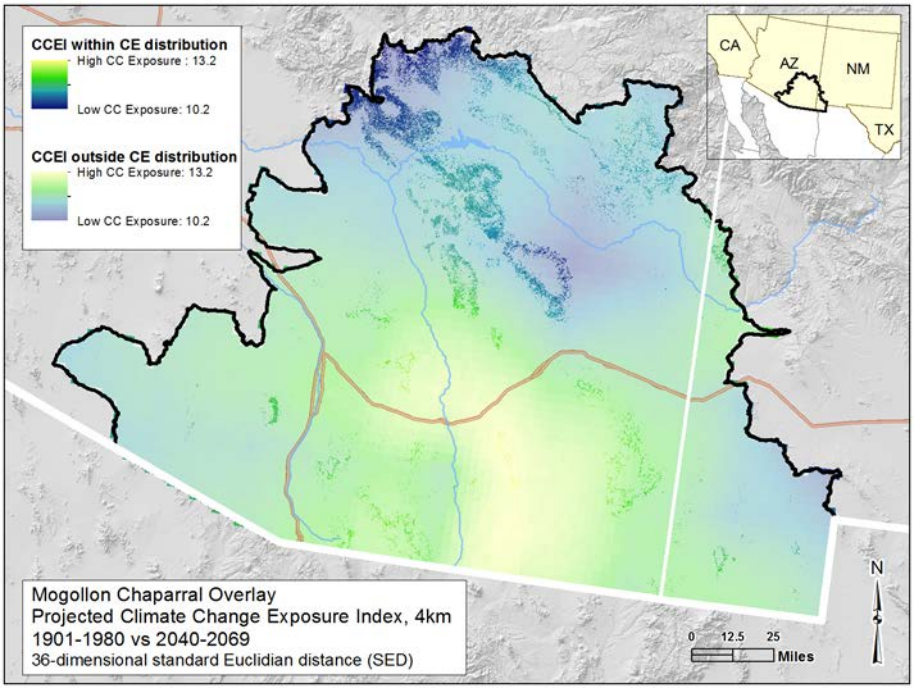

On top of this climate index, conservation element (CE) distributions are overlain as 4 km grid. In each figure, areas that do not include a conservation element (CE) distribution are masked in order to see the climate change index across the CE's distribution. Keep in mind that for all of these figures, climate change exposure is significant and occurs across the entire ecoregion.

**Figure I-45. Madrean Montane Conifer-Oak Forest and Woodland Overlay Projected Climate Change Exposure Index**.



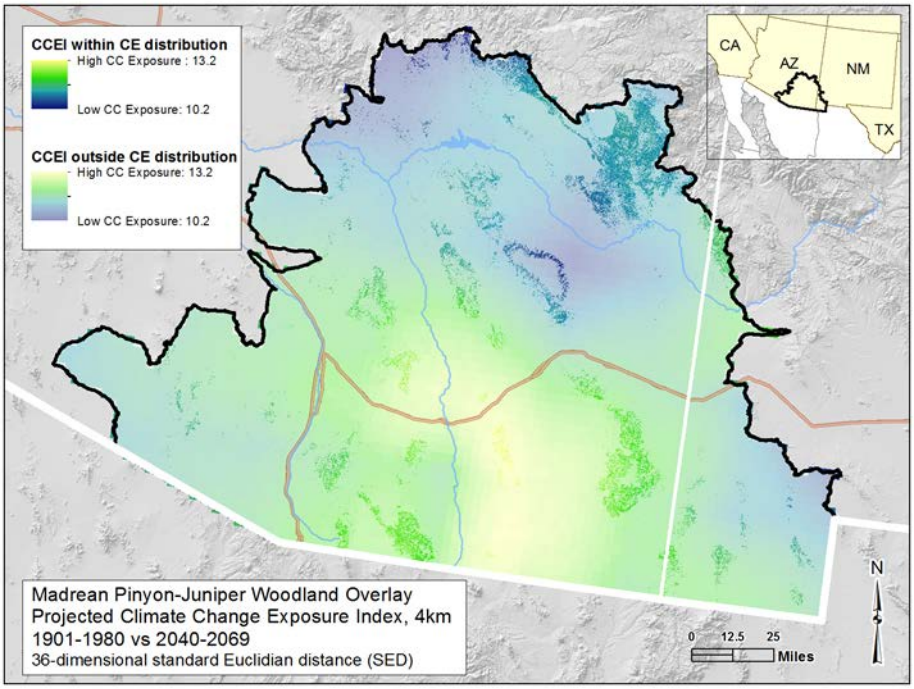

The Madrean Montane Conifer-Oak Forest and Woodland ecosystem occurs throughout the central portion of the ecoregion and is mostly exposed to extreme climate change in the southeastern portion of its range.

**Figure I-46. Mogollon Chaparral Overlay Projected Climate Change Exposure Index**.



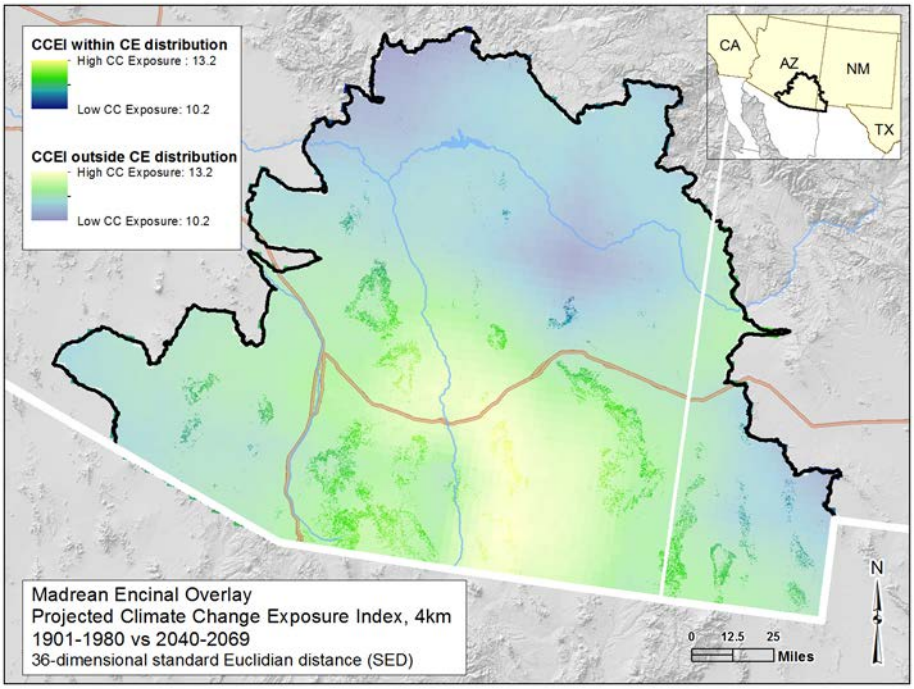

Much of the Mogollon Chaparral occurs in the northern portion of the MAR, which is projected to have less climate change exposure than the south. About a third of the distribution occurs in the southern half of the ecoregion.

**Figure I-47. Madrean Pinyon-Juniper Woodland Overlay Projected Climate Change Exposure Index**.



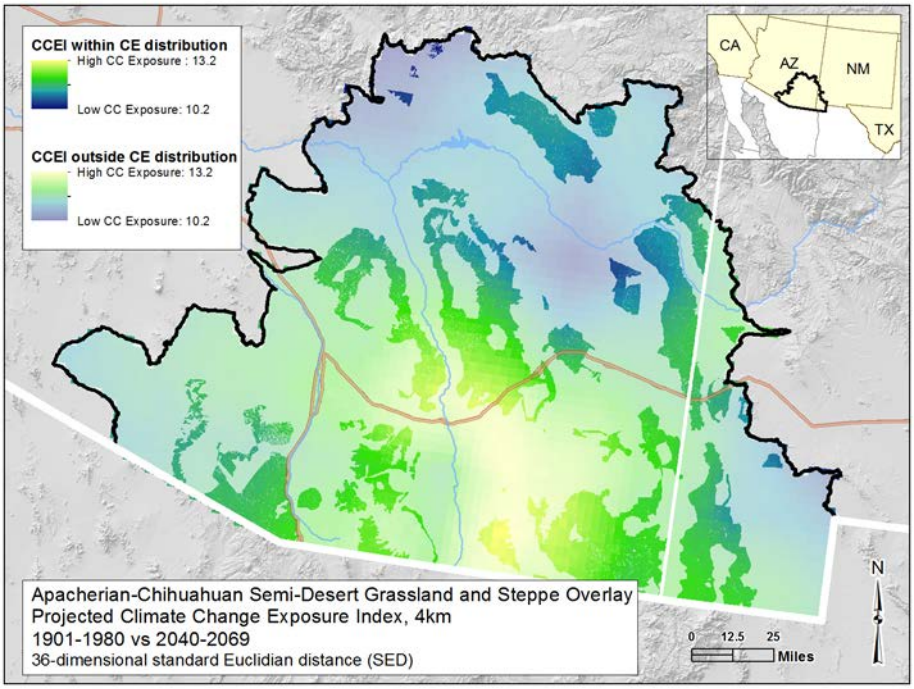

The Madrean Pinyon-Juniper Woodland ecosystem occurs throughout the ecoregion and is mostly exposed to extreme climate change in the south central portion of its range.

**Figure I-48. Madrean Encinal Overlay Projected Climate Change Exposure Index**.



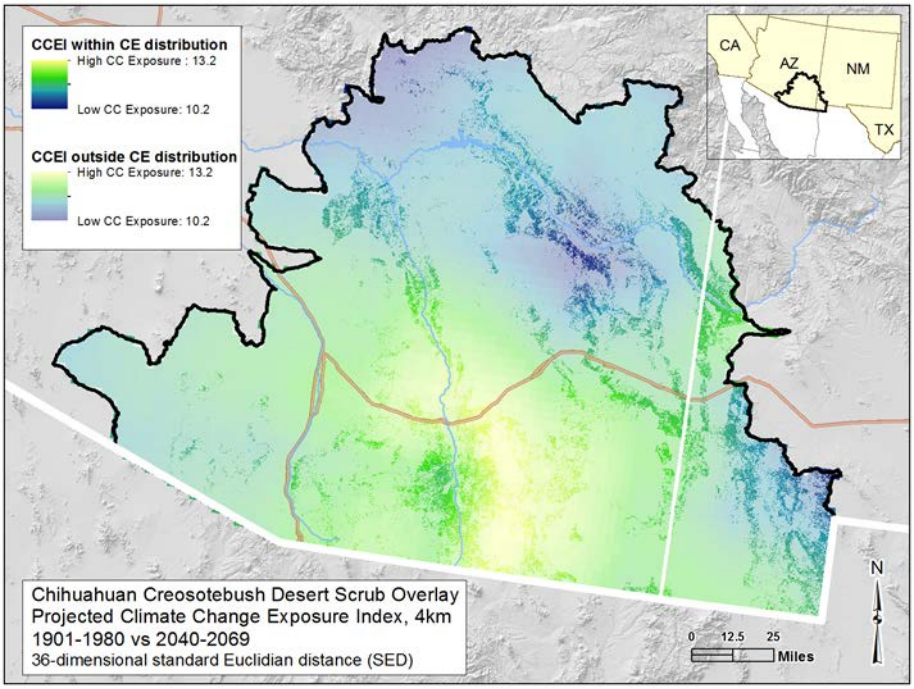

The Madrean Encinal occurs mostly in the southern half of the ecoregion, with western and eastern most portions of its range lying in areas to be less impacted by climate change.

**Figure I-49. Apacherian-Chihuahuan Semi-Desert Grassland and Steppe Overlay Projected Climate Change Exposure Index**.



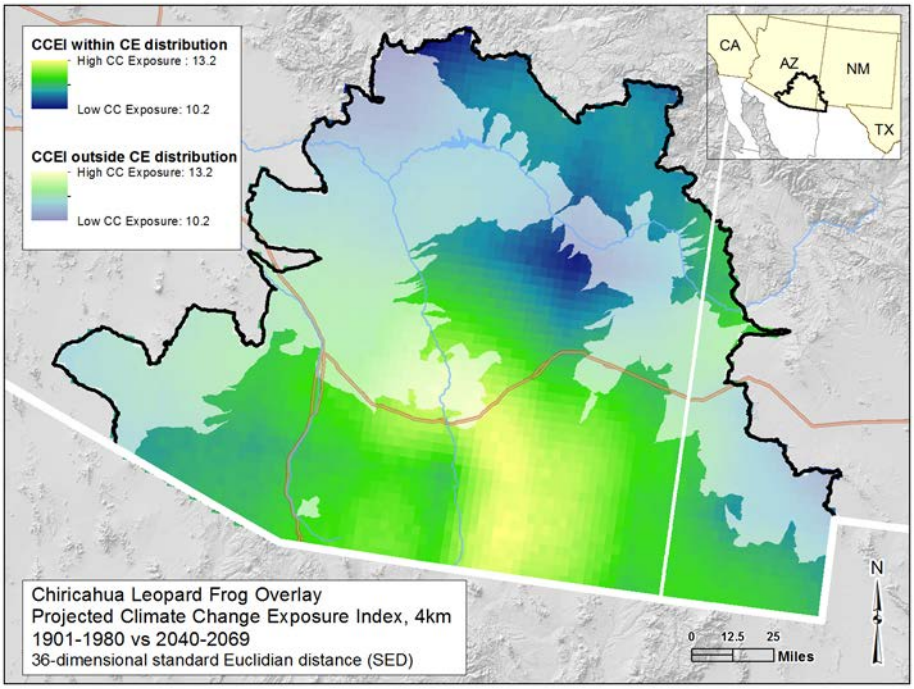

The Apacherian-Chihuahuan Semi-Desert Grassland conservation element (CE) is distributed throughout the ecoregion and about a quarter of the distribution co-occurs with areas projected to have the highest climate change exposure.

**Figure I-50. Chihuahuan Creosotebush Desert Scrub Overlay Projected Climate Change Exposure Index**.



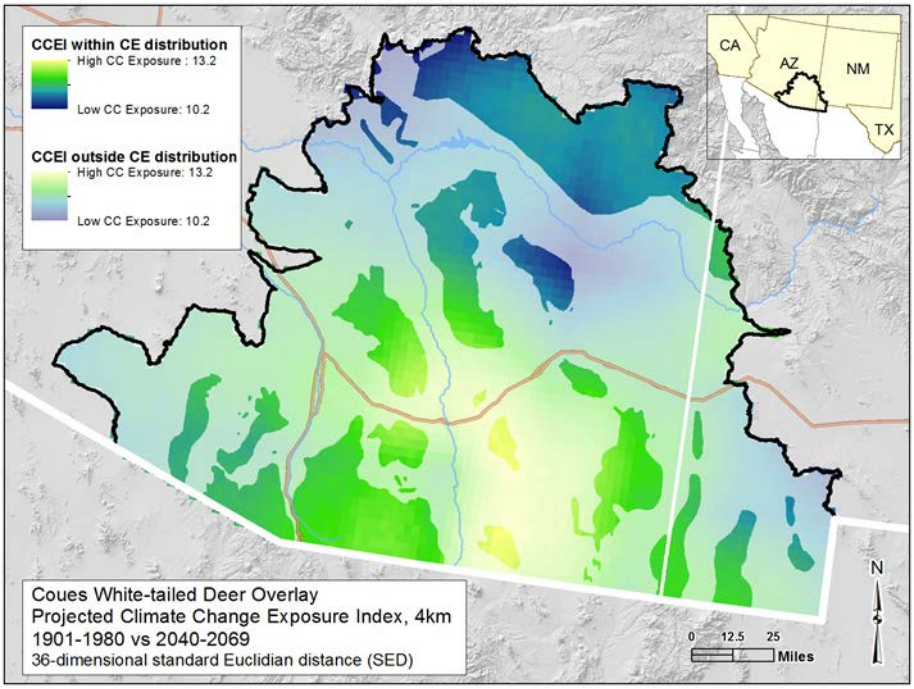

The south central portion of the Chihuahuan Creosotebush Desert Scrub occurs within the greatest climate change exposure areas projected for the MAR, however it also extends to the lowest exposure areas as well in the east and in the north and north west.

**Figure I-51. Chiricahua Leopard Frog Overlay Projected Climate Change Exposure Index**.



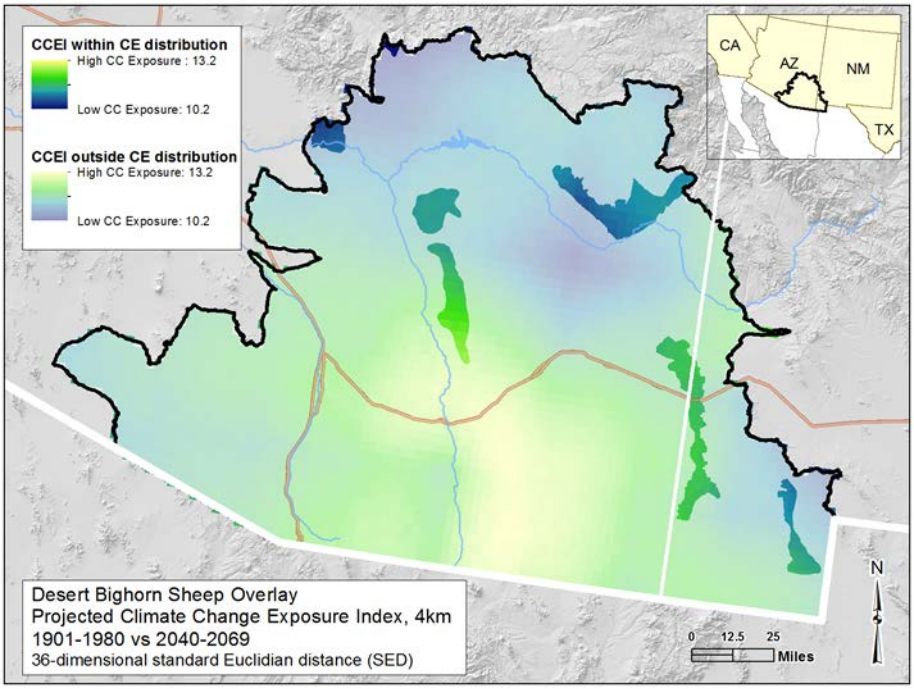

Much of the Chiricahua Leopard Frog distribution coincides with the greatest climate change exposure areas for the MAR. However, the northern and eastern-most portions of its distribution occur in areas of lower climate change exposure.

**Figure I-52. Coues White-tailed Deer Overlay Projected Climate Change Exposure Index**.



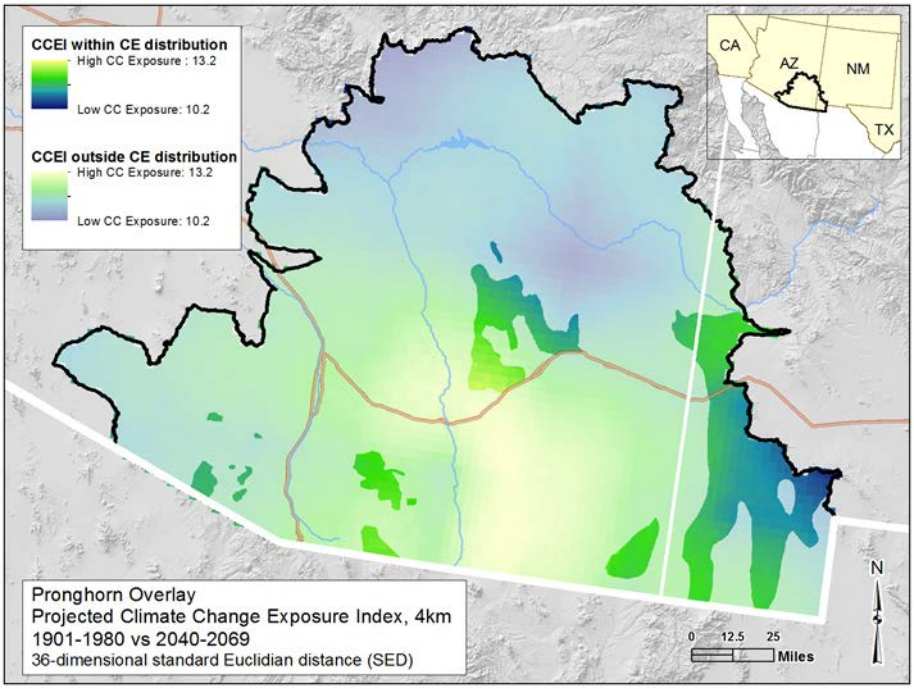

Only a small portion of the Coues white-tailed deer distribution occurs within the highest climate change exposure areas.

**Figure I-53. Bighorn Sheep Overlay Projected Climate Change Exposure Index**.



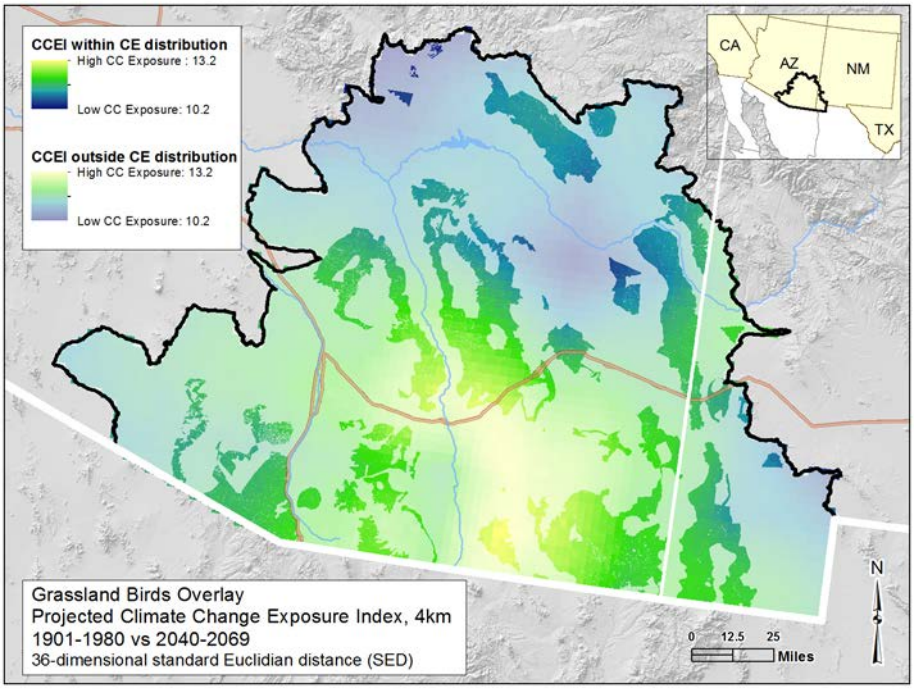

Bighorn sheep occur mostly in the northern and eastern sections of the MAR where climate change exposure will be less than in the south. However, the western-central extent of its range does coincide with some areas of greatest climate change exposure.

**Figure I-54. Pronghorn Overlay Projected Climate Change Exposure Index**.



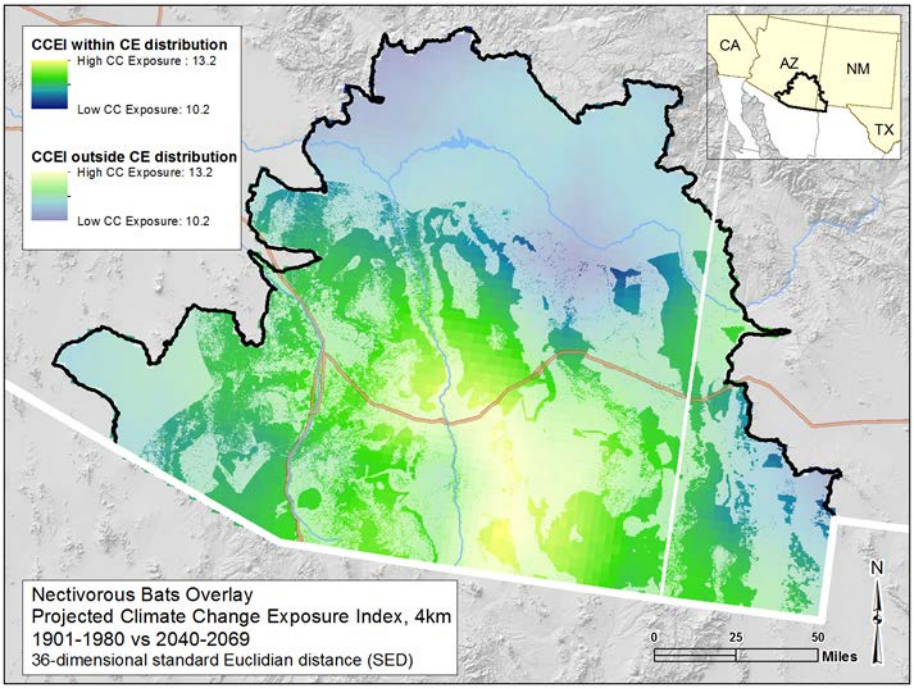

The central portions of the Pronghorn distribution within the areas of highest projected climate exposure. The eastern portion of its distribution may have the least climate exposure of all MAR pronghorn distribution, but note this area is still expected to see significant exposure.

**Figure I-55. Grassland Birds Overlay Projected Climate Change Exposure Index**.



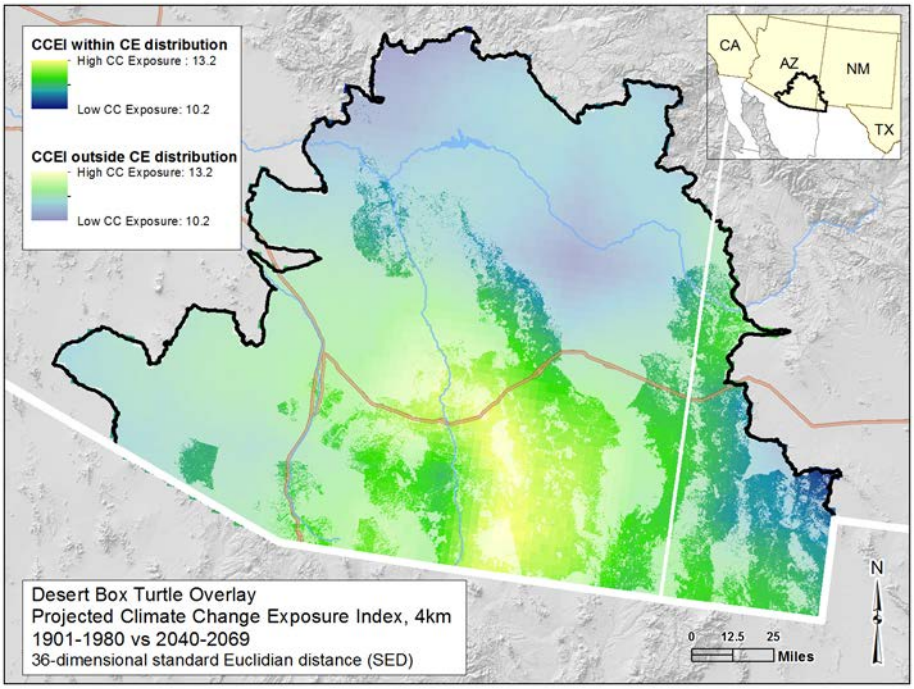

The Grassland Bird Assembledge conservation element (CE) is distributed throughout the ecoregion and about a quarter of the distribution co-occurs with areas projected to have the highest climate change exposure.

**Figure I-56. Nectivorous Bats Overlay Projected Climate Change Exposure Index**.



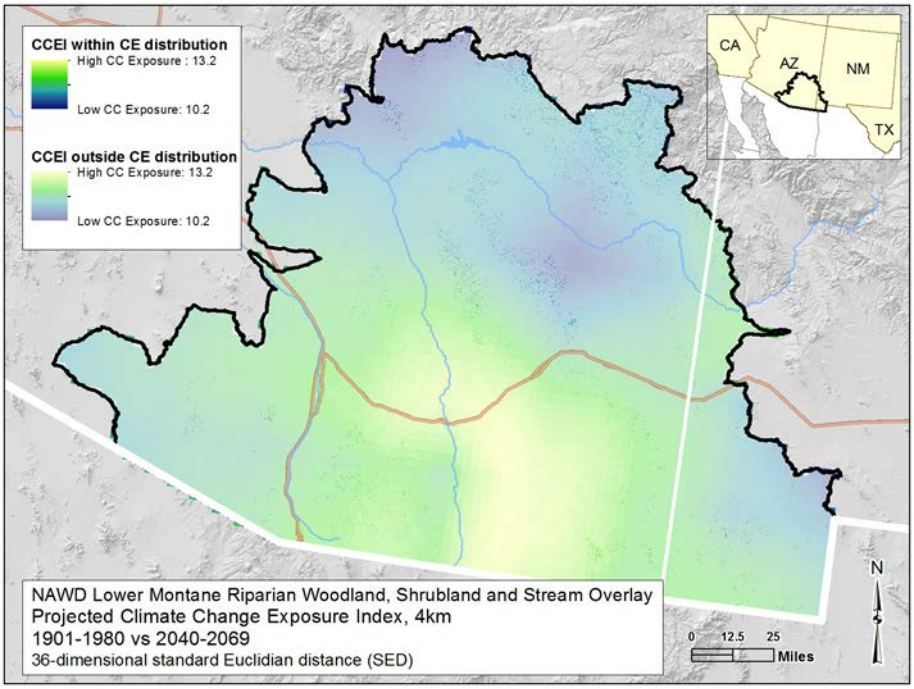

The central portions of the bats assemblage distribution are within the areas of highest projected climate exposure. The eastern portion of the distribution may have the least climate exposure of all, but note this area is still expected to see significant exposure.

**Figure I-57. Desert Box Turtle Overlay Projected Climate Change Exposure Index**.



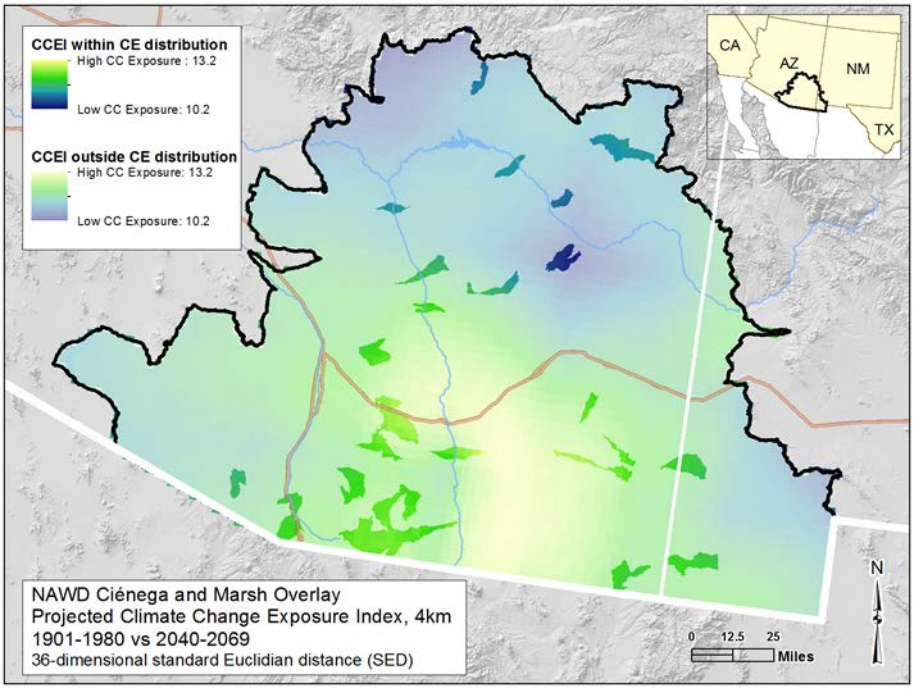

The Desert Box turtle conservation element (CE) occurs mostly in the southern and eastern portions of the MAR, and approximately a third to a half of its distribution occurs within the area projected to have the greatest climate change exposure.

**Figure I-58. North American Warm Desert Lower Montane Riparian Woodland, Shrubland and Stream Overlay Projected Climate Change Exposure Index.**



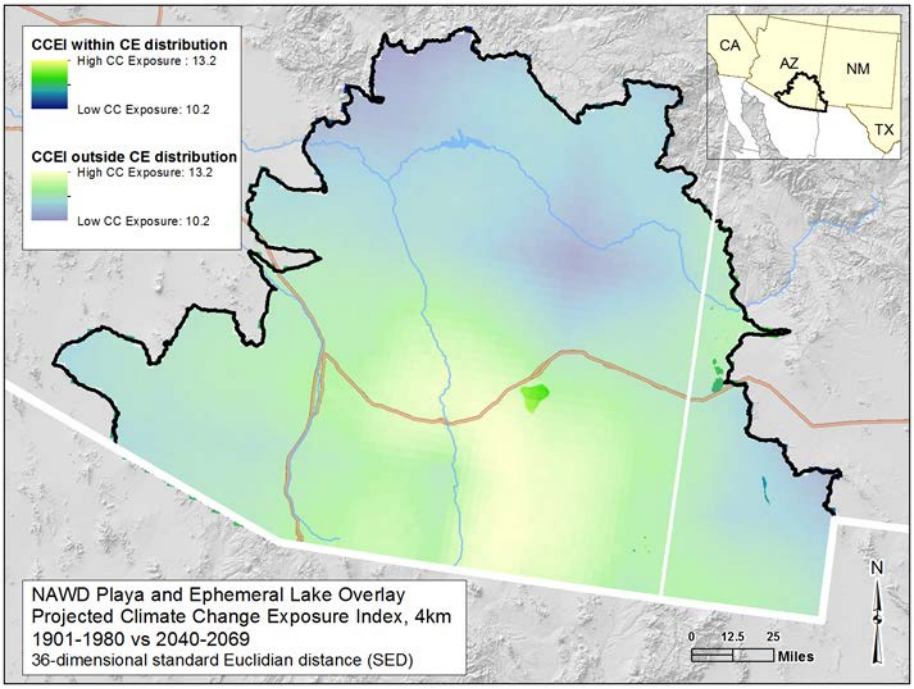

Much of the distribution of this riparian conservation element (CE) occurs with in the south central portion of the ecoregion projected to have the greatest climate change exposure.

**Figure I-59. North American Warm Desert Ciénega and Marsh Overlay Projected Climate Change Exposure Index**.



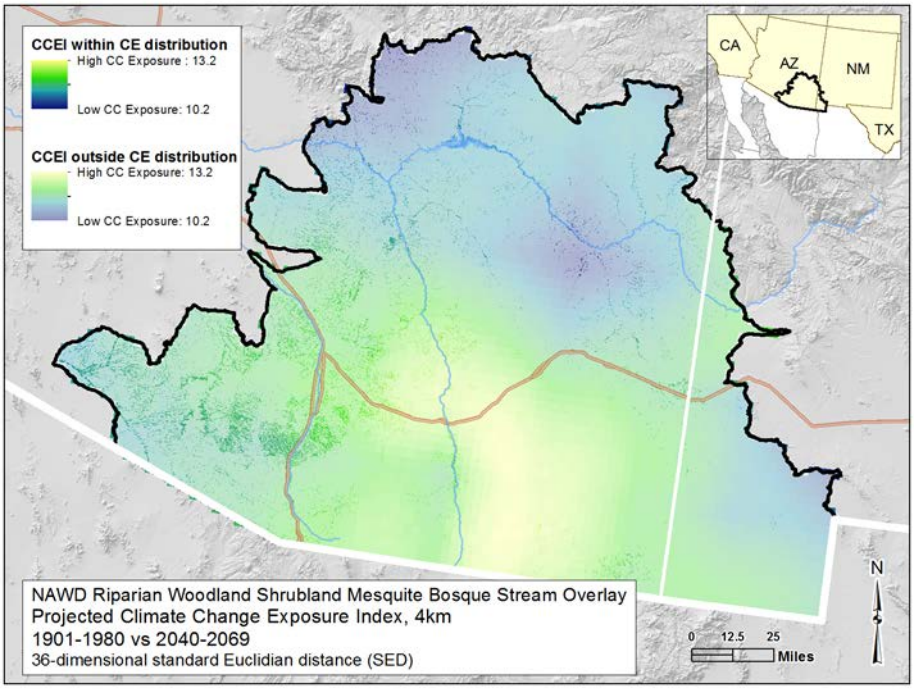

6th level watersheds that contain ciéngas are distributed throughout the ecoregion, particularly in the central portion of the ecoregion. Approximately a third of the distribution appears to co-occur with the area projected to have the greatest climate change exposure.

**Figure I-60. North American Warm Desert Playa and Ephemeral Lake Overlay Projected Climate Change Exposure Index**.




The Wilcox Playa occurs in the heart of the greatest exposure to projected climate change. The Lordsburg and other playas further east occur where climate change exposure is still significant, but lie outside the areas of greatest projected change.

**Figure I-61. North American Warm Desert Riparian Woodland, Shrubland, Mesquite Bosque and Stream Overlay Projected Climate Change Exposure Index.**




Much of this lower elevation riparian CE lies within areas of the MAR that will see significant climate change exposure, but the bulk of the distribution occurs outside of the areas projected to see the greatest July and August temperature increases.

## *I.8 References*


Bronaugh, D. and A. Werner. 2013. Trends package for the Pacific Climate Impacts Consortium. R Package version 0.10-1. http://CRAN.R-project.org/package=zyp.

Daly, C. et al. 2008. Physiographically sensitive mapping of climatological temperature and precipitation across the conterminous United States. International Journal of Climatology 28:2031-2064.

Diffenbaugh, N. S., F. Giorgi, and J.S. Pal. 2008. Climate change hotspots in the United States. Geophysical Research Letters 35.

Diffenbaugh, N. S. and F. Giorgi. 2012. Climate change hotspots in the CMIP5 global climate model ensemble. Climatic Change 114:813–822.

Elith, J., C. Graham, R.P. Anderson, M. Dudık, S. Ferrier, A. Guisan, and N.E. Zimmermann. 2006. Novel methods improve prediction of species' distributions from occurrence data. ECOGRAPHY (29):129-151.

Elith, J., S.J. Phillips, T. Hastie, M. Dudík, Y.E. Chee, and C.J. Yates. 2011. A statistical explanation of MaxEnt for ecologists. Diversity and Distributions, 17:43-57.

Hirschboeck, Katherine K. 2009. Flood Flows of the San Pedro River. Pages 300-312 in: Juliet C. Stromberg and Barbara Tellman, editors. Ecology and Conservation of the SanPedro River. Universtiy of Arizona Press, Tucson.

Intergovernmental Panel on Climate Change (IPCC). (2007). Climate Change 2007: impacts, adaptation and vulnerability: contribution of Working Group II to the fourth assessment report of the Intergovernmental Panel on Climate Change (Vol. 4). Cambridge University Press.

Jiménez-Valverde, A. and M. Lobo. 2007. Threshold criteria for conversion of probability of species presence to either–or presence–absence. Acta oecologica, 31(3): 361-369.

Knutti, R., & Sedláček, J. (2013). Robustness and uncertainties in the new CMIP5 climate model projections. *Nature Climate Change*, *3*(4), 369-373.

NatureServe. 2013. Terrestrial Ecological Systems of the Conterminous United States. Version 2.9. Completed in cooperation with USGS Gap Analysis Program and inter-agency LANDFIRE. Reflecting early 2000s land cover. MMU approx. 2 hectares. NatureServe, Arlington, VA. Digital map.

Pachauri, R. K. & Reisinger, A. (2007). Climate Change 2007: Synthesis Report. Contribution of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. *Intergovernmental Panel on Climate Change* **1.**

Phillips, S.J., R.P. Anderson, and R.E. Schapire. 2006. Maximum entropy modeling of species geographic distributions. Ecological Modelling 190:231-259.

Peters, G. P. *et al.* (2012). The mitigation challenge to stay below two degrees. *Nature Climate Change* **3**.

Skansi, M. de los M., M. Brunet, J. Sigró, E. Aguilar, J.A.A. Groening, O.J. Bentancur, Y.R.C. Geier, et al. 2013. Warming and Wetting Signals Emerging from Analysis of Changes in Climate Extreme Indices over South America. Global and Planetary Change 100:295–307.

Wang, T., A. Hamann, D.L. Spittlehouse, and T.Q. Murdock. 2012. ClimateWNA—High-Resolution Spatial Climate Data for Western North America. Journal of Applied Meteorology and Climatology 51:16–29.

Williams, J. W., et al. 2007. Projected distributions of novel and disappearing climates by 2100AD. Proc. Natl. Acad. Sci. USA 104:5738–5742.

Yue, S., P. Pilon, B. Phinney, and G. Cavadias. 2002. The influence of autocorrelation on the ability to detect trend in hydrological series. Hydrological Processes 16:1807–1829.