Sonoran Desert

Rapid Ecoregional Assessment

Final Memorandum I-3-c: Models, Methods, and Tools

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EXECUTIVE SUMMARY

The primary objective of the Rapid Ecoregional Assessment program is to characterize the current status and to forecast future condition of regional resources of concern. This report describes a set of transparent, repeatable, defensible, and rapid approaches, methods, and tools recommended for the Sonoran Desert Rapid Ecoregional Assessment (REA). Our approach has been to develop a logical, hierarchical organizational framework and management tool for assessing condition of regional resources of conservation concern. Rather than providing a static set of output products, our approach has been to develop a flexible set of resource management tools which will provide BLM managers with the tools necessary to address current and future sustainable resource management challenges at the regional scale. Our approach goes beyond the specific needs of this REA to provide an efficient means to update data, probe relationships, drill down to conduct sub-assessments, and conduct 'what-if' scenarios to aid in strategic planning.

The Ecosystem Management Decision Support (EMDS) tool in conjunction with a logical modeling front-end NetWeaver were selected to meet these requirements. These tools have been successfully applied to goal-driven ecoregional conservation management projects elsewhere over very different geographic extents – global to local scales. This model represents both a conceptual model of relationships between ecosystem components, and a processing framework to address the core questions of resource status assessment and trends. This addresses the need for an overarching model to depict ecological features, processes, and interactions from the coarse scale of the ecoregion down to the finer scale of the conservation element.

The logic model containing the conservation elements and change agents identified were prepared during Tasks I-1 and I-2 in this REA. This model provides a clear picture of how indicators, selected to represent attributes of conservation elements, are rolled up into a set of tiered condition assessments to characterize the current state, and forecast future condition, of biological, landscape, and ecosystem service values in the Sonoran Desert.

<u>Note: BLM denied the use of EMDS on March 1, 2011. However, the Dynamac team is free to use EMDS as long as the final products are delivered in Model Builder/Python. We have retained the EMDS sections in Memo 3 for this reason. The EMDS logic tree diagrams are also useful to visualize how the individual elements are aggregated to create the summary assessments that will be made for each ecoregion.</u>

The REA is designed to provide two distinctly different output products. The first is an assessment of the integrity of resources of regional concern summarized across multiple spatial scales. A second, higher resolution set of output products will address specific management questions identified as of regional importance with respect to a select set of conservation elements and change agents. These two products represent the core of the final deliverable at project's end. The ecoregion logic model helps to identify what information is needed to address the general questions related to resource status. A set of conceptual sub-models and specific

geoprocessing methods were developed to address the specific management questions. These will be continue to be developed with stakeholder input over the course of this project to address the full suite of management questions, conservation elements, and change agents.

Transparency is extremely important for acceptance of an REA approach, processes and products. The logical models and sub-models are readily visible to facilitate expert review through BLM workshops or other means. This transparent engagement is critical to the continuous improvement process of the data, the models and their applications.

Flexibility is another key to the utility and success of a resource management tool. The logic model is designed to incorporate changes that will allow exploration of alternative strategies and priorities. This analytical flexibility often reveals important ecosystem drivers and results that might not otherwise have been anticipated. In addition, sub-models can be easily swapped in and out to evaluate new data sets or alternate sub-model approaches. This feature also supports an efficient process to identify and fill information gaps, and to generate updated output through rapid re-analysis of condition assessments.

The approach presented here is built to perform rapid comprehensive conservation value and condition assessments. Our experience shows that resource managers need the tools that will allow them to creatively explore available information and compare the resulting scenarios. The base layers and final products of the decision support tools provided will be housed in Data Basin (www.databasin.org) allowing BLM resource managers to respond to conservation and development challenges both during and following this initial Sonoran Desert REA project.

1 INTRODUCTION

1.1 Rapid Ecological Assessment (REA) Approach

Rapid Ecological Assessments (REAs) are a product of the BLM's evolution toward a landscape approach to land management. The broad regional extent of the landscape approach addresses issues that transcend administrative boundaries, such as renewable energy development, the spread of invasive species, and projected climate change. Using the landscape approach, the BLM hopes to integrate available scientific data and information from BLM field offices, other federal and state agencies, and public stakeholders to develop shared responses and collaborative management efforts across administrative boundaries. The data collected for the REAs will comprise a baseline from which to evaluate the results of adaptive management.

A central purpose of the Sonoran Desert Rapid Ecological Assessment (REA) is to document the current status of selected ecological resources (conservation elements) at the ecoregional scale and to investigate how this status may change over several future time horizons. REA assessments are expected to identify areas with high ecological integrity and elements of high biological and ecological value to provide a better understanding of key ecosystem processes and potential impacts of future changes. REAs do not involve original research, but they use existing data, modeling, and GIS analyses to answer a broad range of management questions. REAs are also timely in supporting planning, management, and mitigation of impacts anticipated from various climate change scenarios. Intensive data collection required to conduct an REA will also reveal knowledge gaps and highlight areas for future ecosystem monitoring and research.

The three tasks comprising Phase I of the REA that were prerequisite to the final components of the REA Workplan included: 1) the selection of management questions and conservation elements; 2) the collection and evaluation of data layers necessary to conduct the assessment; and 3) the recommended approach to analyses, i.e., methods, models, and tools.

1.2 Task I-3 Objectives

During Tasks 1 and 2, the Dynamac team, the BLM, and workshop participants refined management questions, selected conservation elements, and collected hundreds of candidate data layers. Task 3, Models, Methods, and Tools, focuses on the models and analysis methods required to address groups of management questions and conservation elements.

The first objective of Task I-3 was to develop an overarching organizational approach for making status and vulnerability assessments. When assembled, this higher-order logic model will integrate the attributes and indicators derived from lower-order models of specific conservation elements to assess biotic and landscape condition. We also present examples of approaches to various classes of conservation elements in separate sections, or methods modules, which include background text, conceptual models, and geoprocessing models developed to address specific sets of management questions. The individual conceptual models at the lowest level of organization assist in developing geoprocessing methods to derive metrics for inclusion in the higher-order integrity and status/vulnerability assessments.

Conceptual Models. The REA process has emphasized conceptual models. Most of the attention to date has been focused on lower-order models of specific conservation elements. For this task we adopt a top-down approach and focus first on the organizational structure to address the core questions in the Statement of Work (SOW). One advantage of a top-down approach is the identification of gaps in elements required for a comprehensive assessment. It is also part of the iterative process to assemble the attributes and indicators generated from the conceptual modeling of individual conservation elements. In Task I-3, higher order conceptual or logic models play a central role. These models describe how the various evaluations are combined for the final status and vulnerability assessments, and they characterize ecological condition as a surface across the landscape. We describe the ecological integrity concept, geoprocessing models, and logic model development in Section 5.

Detailed conceptual models that accompany each example methods module relate to individual conservation elements, groups of management questions, and lists of attributes and indicators necessary to assess the status and condition of conservation elements. The Dynamac team has approached the REA conceptual models with a strategy of increasing detail and documentation with each iteration of the Pre-Assessment from the broad scale basic ecoregion model presented in Task 1 to the detailed models that accompany the modeling and mapping approaches in Task 3. The conceptual models developed for Task 2 were at an intermediate level of detail and resolution; they illustrated the mechanisms and relationships that assisted Dynamac staff in the data needs evaluation. To avoid duplication of effort, we planned that a full literature review would accompany the models to be developed for Task 3.

The conceptual models developed to date for the REA process are stressor models that illustrate the mechanisms and pathways of the sources of stress and the key, typical, or known responses of ecosystem attributes (conservation elements). The conceptual models developed for Task 3 are more detailed and specific to individual management questions and related conservation elements. Literature citations and text support the conceptual linkages within the model and explain the use of the model to depict current status and potential for change.

Process Models. Process models represent a schematic description of the methods and tools used to address a management question or to generate a metric for use in status assessments and an index of ecological integrity. A description of the required data and parameters are also included. We describe various types of process models used at different levels of organization of REA components. In theory, a separate process model could be created for each combination of management question, conservation element, or change agent.

Methods. Each approach, proposed for groups of management questions or individual conservation elements, specifies the method of analysis to address a specific set of management questions for a specific conservation element and one or more change agents. The process models described above summarize the methods. For efficiency, we grouped sets of similar management questions related to an individual conservation element and prepared a separate section addressing the pertinent set of questions. This approach was useful for independent preparation of conceptual models and approaches by topical experts, and it was considered a helpful format for reviewers as well. In each case, the methods modules provide a rationale for method selection as well as the selection of appropriate datasets for each group of management questions or conservation elements.

Tools: We present a number of tools in recommended approaches, often in various combinations. These tools range from software to pre-existing process models. MaxEnt, FRAGSTATS, and MAPSS are all described as contributing to the final deliverable. A featured tool is the Ecosystem Management Decision Support (EMDS®) system, in concert with NetWeaver® software is proposed to help develop a logic model for integrating the various components of the REA that will then be produced in Model Builder to the degree possible.

Output products: The application of methods and tools will result in a collection of output products: textual, tabular, and spatial. Ecoregional assessment analysis will cover conservation elements and change agents relative to their current status and potential for future change. According to the SOW, "…current status is the existing state or cumulative condition that has resulted from all past changes imposed upon the prior historical condition. Status is characterized by attributes and indicators for size, condition, landscape context, and trend." Describing status for various conservation elements and resource values assumes that specific characteristics of a resource can be identified and mapped.

Potential for change predicts how status may change in the future; potential for change consists of attributes and indicators for direction, magnitude, likelihood, and certainty of change. For example, to estimate the vulnerability of biological soil crust to disturbance, we must show the relative likelihood of exposure to mechanical disturbance and the likelihood of resource resilience. Products displaying potential-for-change help clarify how current evidence of cumulative impacts may be projected into the future and help identify potential trade-offs, alternatives, and mitigation strategies for BLM planning purposes.

Another REA product of interest to BLM is the location of areas with high potential for renewable energy development. Current and potential development data layers overlaid on mapped results for conservation elements, change agents, and ecological condition will produce composite maps that reveal potential areas for renewable energy development. Taking a broad ecoregional view of important conservation areas and wildlife corridors will presumably facilitate the choice of prospective renewable energy development sites having the fewest environmental effects.

Specific output products will include:

(1) Status – Conservation Values (biological/ecological values + landscape values + ecosystem service values)
(2) Status – Ecological Integrity (biological values + landscape values)
(3) Status – Biological and Ecological Values
(4) Status – Landscape Values
(5) Status – Ecosystem Services Values
(6) Status – Wildlife Species Conservation Elements (from the pre-selected suite of core and desired species)
(7) Status – All species (richness and endemism metrics)
(8) Status – Change agents (locations and magnitude)
(9) Future – Change agents (locations and magnitude)
(10) Vulnerability – Conservation Values
(11) Vulnerability – Ecological Integrity
(12) Vulnerability – Biological and Ecological Values
(13) Vulnerability – Landscape Values
(14) Vulnerability – Ecosystem Services Values
(15) Vulnerability – Wildlife Species Conservation Elements (from the pre-selected suite of core and desired species)

2 BACKGROUND

2.1 Review of Task I-1: Refining Management Questions and Selecting Conservation Elements

In addition to this brief overview of Task 1, a more detailed version can be found in Appendix 1 in this document. The full Sonoran Desert Task 1 Memorandum I-1-c is located at BLM Programs Rapid Ecoregional Assessments website: http://www.blm.gov/wo/st/en/prog/more/climatechange/reas.html.

2.1.1 Objectives of Task 1

The objectives of the first phase of the REA process were to identify the subjects of the assessment, develop a basic ecoregional model, and produce a finalized list of ecoregion-specific management questions. The REA will assess the current status and future condition of the ecoregion's natural resources by examining the relationships between a set of *conservation elements* and disturbance factors or *change agents.* The REA Task Order defines core conservation elements as biotic constituents (wildlife and plant species and assemblages) or abiotic factors (e.g., soil stability) of regional significance in major ecosystems and habitats across the level III ecoregion. This limited suite of conservation elements represents all renewable resources and values within the ecoregion and may serve as surrogates for ecological condition across the ecoregion. Through the individual or interactive effects of change agents, the condition of conservation elements may depart from a model of a minimally- or least-disturbed *reference condition* and thus depart from a state of ecological or biological integrity (Frey 1977, Karr and Dudley 1981).

2.1.2 Selection of Conservation Elements

Following Workshop 1, the Assessment Management Team (AMT) recommended separating wildlife conservation elements into categories: sensitive species, which would be mapped as a richness-function (indicating species diversity hotspots); up to a dozen landscape wildlife species; and a set of desired species. BLM suggested that the landscape species be screened using the Coppolillo method (Coppolillo et al. 2004) because it is systematic and fairly objective. Participants in the REA process continued to suggest additional wildlife species of unrepresented taxa or habitats throughout Tasks 1 and 2. AMT guidance during and following Workshop 2 indicated that wildlife species conservation elements may be considered for inclusion throughout the Pre-Assessment phase. USGS review comments following Workshop 2 suggested that species selection should focus on identifying species that are vulnerable to change agents. The Dynamac team agreed that the selection of disturbance-sensitive species will provide the best representation of status and condition at the ecoregional level with respect to habitat alteration, displacement, and human stressors. However, the Dynamac team felt constrained to retain the full list of species selected using the Coppolillo screening suggested after Workshop 1 because too many species substitutions threaten to invalidate the entire screening process requiring us to start again. Dynamac proposed that any other species added to the list of conservation elements after Workshop 1be considered *desired species*.

Note: At Workshop 3, Methods, Models, and Tools, participants suggested additional species of unrepresented taxa or habitats to serve as conservation elements. At the workshop, the AMT and workshop participants agreed to add Le Conte's thrasher *(Toxostoma lecontei,)*, Bell's vireo (*Vireo bellii*), and lowland leopard frog (*Rana yavapaiensis*) and drop burrowing owl (*Athene cunicularia*) and razorback sucker (*Xyrauchen texanus*).

2.1.3 Biodiversity

To address ecoregional biodiversity, the AMT indicated that Dynamac will receive G1 through G3 species occurrence data generalized to the level of the 5th level hydrologic units (HUCs), one of the landscape reporting units specified in the REA Statement of Work. The intent is to present a generalized richness-summary map layer based on occurrence data of G1 through G3 species from the Arizona Natural Heritage Programs and the California Natural Diversity Database. We have the option to organize subsets of these data in different ways to depict biodiversity hotspots and endemics. These richness-function map layers are limited as they only represent known locations of occurrences, rather than where the species currently occurs. In addition, the age of the records needs to be considered as well. The BLM required a coarse expression of the data because of the prohibitive costs associated with acquiring spatially-explicit occurrence data as well as concerns about mapping detailed occurrences for vulnerable species.

2.1.4 Management Questions

Other major requirements of Task 1 were to finalize the lists of management questions and change agents. The Dynamac team evaluated each management question to determine whether they could be feasibly answered during the short timeframe of the REA. Participants at Workshop 1 helped to refine or delete various management questions. The Dynamac team accepted the change agents identified by the AMT as clearly important to ecological resources at the ecoregional scale, and we suggested an additional change agent, grazing, for AMT consideration (Appendix 9). After group discussion at the first workshop and subsequent AMT direction, grazing was accepted as a change agent if it included grazing by all herbivores, i.e., wildlife, wild horses and burros, and livestock.

Note: The USGS conducted another review of the management questions in January 2011 to clarify the language and suggest deletion of additional questions that were unclear or outside the scope of the REA. As we completed Memo 3, the AMT finalized the full list of management questions (3-18-11). We have updated the questions wherever possible in this memo and have put a list of the finalized management questions in Appendix 13.

2.2 Review of Task 2: Data Identification and Evaluation

In addition to this brief overview of Task 2, a more detailed version can be found in Appendix 10 in this document. The full Sonoran Desert Task 2 Memorandum I-2-c, Data Identification and Evaluation (with all of the conceptual models and data needs, data evaluation, and data gaps tables included) is located at BLM Programs Rapid Ecoregional Assessments website:
http://www.blm.gov/wo/st/en/prog/more/climatechange/reas.html .

2.2.1 Objectives of Task 2

The objectives of Task 2 were to identify, evaluate, and ultimately recommend datasets required to assess current status of a suite of ecological systems, species, sites, and ecological function and service conservation elements and to forecast changes in status at two future time horizons: 2025 and 2060.

In this second stage of the process, the Dynamac team conducted a data needs assessment, located and identified extant data layers from a variety of sources for consideration, identified data gaps, and solicited additional data layers from Workshop 2 participants. Data acquisition and evaluation is an ongoing and iterative process; the results of Workshop 2 and the accompanying memo marked the beginning of a data identification process that will continue to the Work Plan Preparation stage. Additional data needs may arise after inspection and approval of approaches and methods during Task 3.

2.2.2 Data Needs Assessment

To identify general data needs to address specific management questions, the Dynamac team grouped management questions into subject classes. Using a conceptual model of conservation elements, change agents, and influential processes as a guide, we identified data layers needed to address each question within the management question group. A tentative analysis approach accompanied each management question to provide a rationale for the related data needs assessment. We organized the results of the data needs assessment into sets of tables for each group of related management questions. The conceptual models presented in Task 2 are stressor models that illustrate the mechanisms and pathways of the sources of stress and the key, typical, or known responses of ecosystem conservation elements. The conceptual models developed for Task 3 are more detailed and specific to individual management questions pertaining to each conservation element.

2.2.3 Data Identification and Evaluation

Data identification and evaluation is a continuation of the process that began with the review and evaluation of the lists of management questions provided by the AMT during Task 1. The object of the data evaluation is to match potential data layers with identified data needs and to assess the utility of the datasets to map key attributes of conservation elements and to address classes of management questions.

The linear nature of tasks and deliverables complicated the data search, since the needed data is largely dependent on the methods to be used, and methods will not be identified and approved until Task I-3. The large number of acquired datasets to evaluate delayed the selection of a final set of useful data layers to address the groups of management questions. Including the required and recommended datasets listed by BLM, several hundred candidate data layers were acquired before Workshop 2. The SOW called for each dataset to be evaluated according to 11 quality criteria listed in the Data Management Plan (for example, criteria such as spatial accuracy, thematic accuracy, and precision) and given a confidence score to aid in choosing the optimal data layer in each thematic class. During the data evaluation process, the Dynamac team also noted major data gaps in a series of tables to help focus the discussion for Workshop 2 participants. Some data gaps have been filled since Workshop 2.

2.2.4 Results of Workshop 2, November, 2010

Data Identification and Evaluation. Because of the large number of data layers, it became apparent that completion of the data identification and evaluation step was not realistic within the time and level-of-effort constraints inherent in the REA process. As a result, the AMT agreed to extend the data identification and evaluation stage through Task 3 and 4 of the REA and to delay the formal evaluation of data layers until they were formally accepted for the modeling effort. Memo I-2-c therefore represents a status report of data evaluations conducted through 18 October 2010.

A lesson learned from these early REAs for BLM is the possibility of funding a pre-assessment to have groups of similarly-themed data layers evaluated to choose the best ones and then provide the best basic layers, such as energy development or agriculture, in the required or recommended list.

Choice of Vegetation Data Layers. A major theme at the workshop was the accuracy of the major vegetation data layers, SW ReGAP and LANDFIRE. The Dynamac team showed an example of the differences between the two frameworks in extent and attribution of various riparian vegetation classes for the same location. Some workshop participants favored of using the GAP data, which they considered more accurate. Fire specialists naturally preferred LANDFIRE for fire-related questions. Two possible solutions discussed were 1) to use SW ReGAP for all vegetation questions and LANDFIRE for fire-related questions with the risk of having incomparable results or 2) perform a cross-walk between SW ReGAP and LANDFIRE. The crosswalk would require rewriting the code for LANDFIRE using biophysical information from SW ReGAP. We expect that the second option would be too time-consuming to be accomplished within the REA framework. **Note**: This issue was resolved after Workshop 3 with a proposal from the Dynamac team for an approach to using both frameworks (see Section 5.3.2 this memo).

Climate Data. The AMT advised the Dynamac team that climate change data would be forthcoming from USGS. These data were provided after Workshop 2. Because of this, there was no systematic search for climate change data before Workshop 2.

Aquatic and Terrestrial Sites of High Biodiversity. Natural Heritage sites and sites noted in State Wildlife Action Plans were deleted from the list of Sites of Conservation Concern because of a lack of mappable data.

2.3 References Cited

Coppolillo, P., H. Gomez, F. Maisels, and R. Wallace. 2004. Selection criteria for suites of landscape species as a basis for site-based conservation. Biological Conservation 115: 419–430.

Frey, D. 1977. Biological integrity of water: an historical approach. Pages 127–140 in R.K. Ballentine and L.J. Guarraia (editors).The Integrity of Water.Proceedings of a Symposium, March 10–12, 1975, U.S. Environmental Protection Agency, Washington, D.C.

Karr, J.R., and D.R. Dudley. 1981. Ecological perspective on water quality goals. *Environmental Management* 5:55–68.

3 CONSERVATION VALUES OF THE SONORAN DESERT

3.1 Terrestrial and Aquatic Conservation Values

A major challenge for ecological assessments at any scale is how to assign relative conservation values for planning and management purposes. How important are endemic species or overall species richness? Are some elements more important than others? Are certain geographic locations more important than others? How much of a particular element is required for long-term survival? Ecological assessments attempt to answer these and other questions by evaluating the ecological value of terrestrial and aquatic systems. The practice of conservation planning has grown rapidly over recent decades with increasing levels of sophistication in both the conceptual basis of the field as well as the tools used to carry it out the work (see Noss and Cooperrider 1994; Noss et al. 1997; Soulé and Terborgh 1999; and Groves 2003). Because conservation is often about *place*, mapping is a major cornerstone of the discipline, especially as computer mapping technology of GIS and remote sensing techniques continue to improve.

3.2 Overall Conservation Value

Conceptually, overall conservation value consists of three major components – biological and ecological values, landscape values, and ecosystem service values (The Millennium Assessment 2005; Figure 3-1). Biological values have historically focused on individual species; especially game species or species that are endangered or threatened. More modern assessments of conservation value integrate natural communities, ecosystems, or even natural processes into these analyses. This expansion of element types, which is a reflection of our understanding of biodiversity itself, emphasizes the need to think, evaluate, and plan at multiple spatial and temporal scales. This multi-faceted approach has become a core issue in conservation planning over the last decade (Wiens 2002).

The concept of High Conservation Value (HCV) is a widely accepted concept internationally. First introduced as High Conservation Value Forests (or HCVF) by the Forest Stewardship Council (FSC) as part of their forest certification process (Jennings 2004), the concept now includes many different ecosystem types including grasslands (Cousins et al. 2003) and aquatic environments (Boon 2000). Jennings (2004) defined six main types of HCVs (see below) with HCV1 and 3 pertaining to biological values; HCV2 pertains to landscape values; and HCV4–6 pertaining to ecosystem service values. Most HCV assessments to date have focused on HCV1–3 using traditional conservation planning datasets and techniques and apply HCV4–6 through consultation with various stakeholders for each region.

> ***HCV1 –** Areas containing globally, regionally or nationally significant concentrations of biodiversity values (e.g., endemism, endangered species, refugia)*
>
> ***HCV2** – Globally, regionally or nationally significant large landscape-level areas where viable populations of most if not all naturally-occurring species exist in natural patterns of distribution and abundance*
>
> ***HCV3** – Areas that are in or contain rare, threatened or endangered ecosystems.*
>
> ***HCV4** – Areas that provide basic ecosystem services in critical situations (e.g., watershed protection, erosion control)*
>
> ***HCV5 –** Areas fundamental to meeting basic needs of local communities (e.g., subsistence, health)*

HCV6 *– Areas critical to local communities' traditional cultural identity (areas of cultural, ecological, economic or religious significance identified in cooperation with such local communities)*

3.3 Biological and Ecological Values

Because species are not evenly distributed across the landscape, two components are commonly used in assessing biological value of regions: high species richness and/or high species endemism (e.g., see Ricketts et al. 1999). Areas that possess these qualities are often referred to as "hotspots"; however, these findings are very sensitive to scale. For example, Stoms (1992) demonstrated that as the mapping unit changes, so does the location of areas identified as centers of richness or endemism for vertebrates. If an ecoregion is the mapping unit, some entire ecoregions will be identified as hotspots. However, examination at a finer scale will show that there are actually hotspots within hotspots. Throughout the arid west, the greatest species richness and diversity are in riparian areas. If minimal mapping units are too coarse, riparian areas may be overlooked and other areas may appear to possess higher values, thus misrepresenting reality on the ground. In addition, sampling history and intensity may inadvertently identify hotspots in close proximity to roads, trials, and waterways – obviously places within easy access to humans.

Another important component to determining overall biological value is the location and concentration of rare species. The assessment team needs to determine what constitutes rarity and how to treat it. For example, is a particular element rare in the world, country, or region? The answer might lead one to weight different elements by their relative importance. Globally rare is of greater importance than regionally rare. Rarity can be over-emphasized in planning biological value leading to skewed or even flawed results. Rarity should not dominate the process.

Crucial habitat for selected wildlife species conservation elements is another important consideration. Wildlife species conservation elements should include a variety of species that characterize a region. A useful list of 4 to12 species is often comprised of some combination of area-limited species, dispersal-limited species, resource-limited species, process-limited species, and keystone species (Lambeck 1997, Noss et al. 1997, Coppolillo et al. 2004). Predators, for example, are good indicators of ecosystem health. Another issue is the definition of "crucial habitat" for each species chosen (NOT to be confused with the legal definition of "designated critical habitat" according to ESA guidelines). Habitat requirements for consideration include composition, structure, area, and configuration.

The "keystone species" concept was first described by Paine (1969) in describing the role starfish played in intertidal environments. Power et al. (1996) provided a more precise definition of the term as species with a disproportionately large impact on ecosystems relative to their abundance or biomass. Species such as the American beaver and American alligator are classic examples of keystone species having far greater impact on their ecosystems compared to their abundance. Various authors have suggested representative keystone species for the Sonoran ecoregion: ironwood (*Olneya tesota,* Suzán et al. 1996), mesquite (*Prosopis spp*., Carillo-Garcia et al. 1999), saguaro (*Carnegiea gigantea*, Drezner 2008), and cicadas (*Diceroprocta apache,* Andersen 1994). "Keystone ecosystems" are an extension of that concept that applies to features, landscapes, and ecosystems rather than species. For example, salt licks, serpentine communities, old-growth forest groves, and caverns are good examples. Sonoran keystone species such as ironwood, mesquite, and saguaro may be considered members of Sonoran keystone ecosystems because they provide tall structure and biomass in the desert and serve as hosts and habitat for a number of other plant and animal species. For the arid Sonoran Desert ecoregion, water ecosystems—oases, springs, river washes, and ciénagas—and the linkages between aquatic and terrestrial landscapes are particularly important.

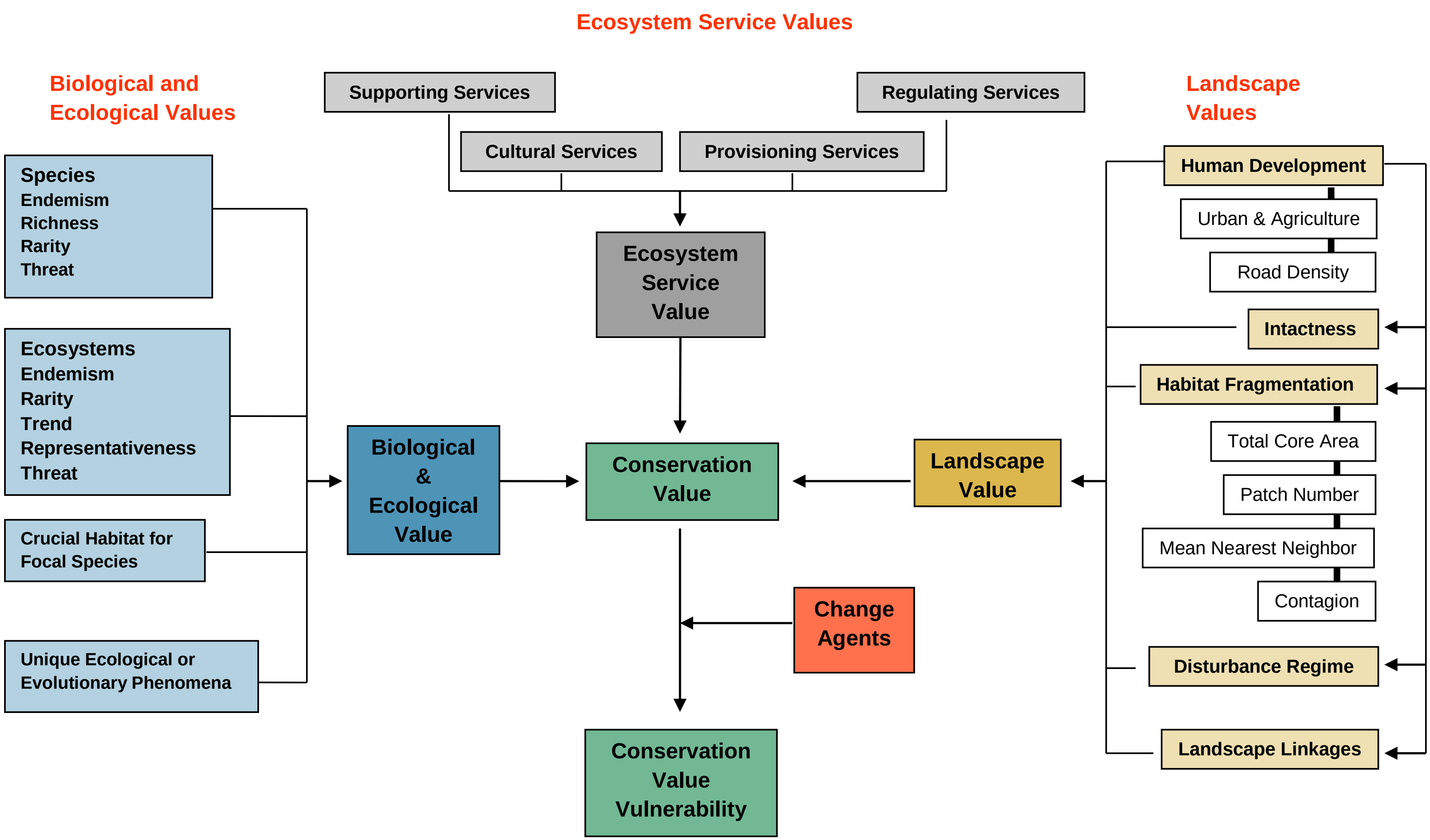


Figure 3-1. Conceptual diagram for assessing terrestrial conservation values featuring the major components that comprise overall conservation value and vulnerability to change agents.

Perhaps the most comprehensive of all conservation criteria is the concept of ecological representation (Noss and Cooperrider 1994). As early as 1926, when the Committee of the Preservation of Natural Conditions of the Ecological Society of America attempted to assess the protection status of biomes in the U.S., ecologists have examined the question of representation (Shelford 1926). Planning for representation means "capturing the full spectrum of environmental variation with the understanding that this variation is dynamic" (Noss and Cooperrider 1994). The national Gap Analysis Program has provided valuable datasets and techniques to address this issue in a standardized fashion throughout the U.S. (Jennings 2000), but scale and the detail at which natural communities are defined play a significant role in the outcomes of representation analysis. In the REA context, it is important to include those natural communities at a meaningful level of taxonomic detail that researchers know to be underrepresented in the existing protected areas network. In some cases, they will include matrix communities —those that cover large areal extents naturally—but more often, they will include patch communities (Anderson et al. 1999).

Rare ecological and evolutionary phenomena are also important considerations in assessing relative conservation value in a region. These elements can be many things such as intact predator-prey systems, prime migratory stopover areas, or spawning areas. If not explicitly noted for inclusion in an REA, important ecological values might be overlooked.

When combined with the appropriate weightings, all of the biological value components comprise a raw biological value data layer (or intermediate map).

3.4 Landscape Values

No place on Earth remains unaffected by modern humans (Vitousek 1997), but some regions have been more directly and severely affected than others. We know that natural landscapes lose components and functionality as human uses expand and continue over time. Some ecosystem changes can be quite gradual (e.g., loss of interior forest habitat over time) while others are punctuated (e.g., loss of a keystone species). Intactness is not a binary (yes/no) quality, but one of degree: a continuum of intactness from a pristine environment on one end to a totally developed environment on the other. Quantifiable and replicable indices and scales of measurement are needed to score landscapes on this continuum. Although significant progress is being made (Anderson 1991, Angermeier 2000), this area of applied research remains quite young. Nevertheless, although ranking natural landscapes by relative intactness may be imperfect, it need not be arbitrary.

Landscape intactness as it applies to forested landscapes is more developed than for other ecosystem types (due to Forest Stewardship Council forest certification and the Global Forest Watch network), but many of the same principles apply to any natural landscape. The footprint of human development and the linear infrastructure (roads, railroads, and utilities) surround and delineate potential blocks of natural landscapes. In general, larger blocks are more highly valued as they are more likely to possess more intact ecosystem composition, structure, and function. However, individual block size is insufficient to determine quality alone. Level of habitat fragmentation within each natural landscape block, detailed spatial information on human activity, and the spatial arrangement of all blocks help to further define relative intactness. Roadless areas are described and mapped at this level of assessment (Strittholt and DellaSala 2001). It is important to note that these landscape blocks should not be thought of as static entities. An ecosystem with a high level of intactness maintains its biodiversity and ecosystem functionality over time – not in any fixed, quantitative sense, but rather as a dynamic property (O'Neill et al. 1986, Holling 1992).

Consideration of natural disturbance regimes is also important when assessing relative landscape value. Different natural systems require different areal extents to accommodate the dominant natural disturbance agent(s) characteristic of a disturbance regime. For planning purposes, the objective is not to determine the minimum size that a single natural landscape must be to accommodate every conceivable natural disturbance event; rather, it is to establish a size threshold for mapping purposes that reasonably reflects the scale of the dominant disturbance agent. For the arid west, fire is one of the important disturbance agents that can impact large areas. Human activity has considerably altered the fire history of the ecoregions of the southwestern U.S. (Allen 1996), and the resulting disturbance pattern will need to be considered when trying to determine the minimum dynamic area – the area necessary to ensure survival or re-colonization of disturbed sites (Pickett and Thompson 1978).

Overall, habitat loss and fragmentation is the most important factor leading to the loss of native species (Wilcox and Murphy (1985). Habitat fragmentation is the process of subdividing a continuous habitat type into smaller patches, which results in the loss of original habitat, a reduction in patch size, and the increasing isolation of patches (Andrén 1994). To counter the negative effects of habitat fragmentation, promoting functional connectivity between existing patches of native habitat is fundamentally important. Functional connectivity can be achieved by identifying and preserving actual landscape linkages (narrow bands of native habitat between existing protected areas or other core natural habitats) or by planning for an effective level of landscape permeability. Both of these methods should target area- and habitat-sensitive species. Landscape level connectivity is also important for maintaining broader scale ecological processes (e.g., aquatic-terrestrial interaction, natural plant and animal dispersal, predator-prey interactions, and species migration). To the degree possible, regional planning should strive to identify important locations where these processes can be supported. Wildlife connectivity is currently a priority of the Western Governors' Association and many states are identifying important landscape linkages and crucial habitats (WGA 2008).

The assemblage of various landscape level components generates a composite landscape value data layer (or intermediate map).

3.5 Ecosystem Service Values

Ecosystem service value is the newest category to evaluating conservation value. The Millennium Assessment (2005) identified four basic categories of ecosystem services that benefit humans: (1) supporting, (2) provisioning, (3) regulating, and (4) cultural (Figure 3-2). Supporting services provided by natural ecosystems include such things as producing oxygen, rebuilding soils, and providing nutrients. Provisioning services are those that directly support human needs for water, food, and fiber. Regulating values are those stabilize water supplies, prevent disease, and mitigate the effects of climate change. And finally, cultural services include the educational, recreational, aesthetic and spiritual values people find in natural landscapes. Ecosystem service values are related to High Conservation Value area types 4 to 6 (e.g., watershed protection, subsistence and cultural values) discussed in Section 3.2.

Although ecosystem services make logical sense to include, quantifying and assessing them is challenging as assets embodied in natural ecosystems are often poorly understood, rarely monitored and undergoing rapid degradation (Heal 2000). Scientists are racing to create an ecosystem services framework that is credible, replicable, saleable, and sustainable (Dailey et al. 2009). Through initiatives such as the Natural Capital Project—a partnership between Stanford University, World Wildlife Fund and the Nature Conservancy—significant progress is being made.

For the REA, five ecosystem services were listed as topics of interest. Two are supporting services – Soils and Air Quality – and three are provisioning services – Forage and Surface Water and Groundwater.

In the case of supporting services, ecological integrity and human needs are positively served (e.g., good air quality is good for both humans and natural ecosystems). Provisioning services, however, can be seen as often conflicting—more surface water for agriculture and urban water supply means less in-stream surface water for fish and wildlife. The integration of conflicting outcomes needs further development to allow the land manager and policy maker to consider different perspectives and tradeoff options to optimize societal benefit from ecosystem services.

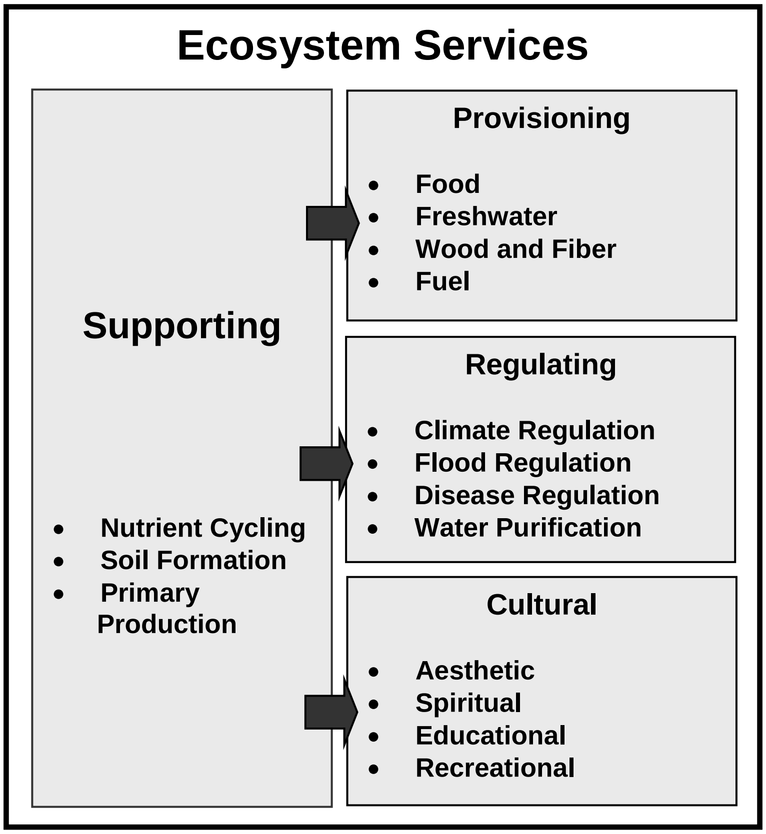


Figure 3-2. Diagram depicting the four classes of ecosystem services as described by The Millennium Assessment (2005).

3.6 Aquatic Conservation Value

The aquatic conservation value conceptual diagram is very similar to the terrestrial one except characteristics of waterscapes replace components of landscape values (Figure 3-3). Because aquatic organisms are so dependent upon the quantity, quality, and dynamics of water, the waterscape condition is closely tied to aquatic biological values (Moyle and Randall 1998). In fact, Harding et al. (1998) showed that landscape history from 50 years ago was an excellent predictor of fish and macroinvertebrate diversity.

Dunn (2000) identified about 50 different components for assessing the conservation value of riverine systems in Australia. These include about 35 elements in the Biological Values category under the headings of Representativeness, Diversity and Richness, Rarity, and Special Features as well as 15 landscape (or waterscape) level factors (Table 3-1). This table offers examples of aquatic elements that may aid in developing a list of attributes for the Sonoran Desert ecoregion. Note the similarity that exists between this list of aquatic elements and terrestrial values previously described

Some differences exist between the landscape value and waterscape value components of the two conceptual diagrams. Human development is a major driver in both, but it is represented slightly differently in the two models. Human development on the terrestrial side causes direct conversion of land to other land uses, impacts overall landscape intactness, and habitat fragmentation (primarily through the configuration of developed land and linear features on the landscape (i.e., roads, railroads, pipelines, and utilities). Human development has also impacted the natural disturbance regimes and, in some locations, degraded natural landscape linkages.

Within the aquatic domain (or waterscapes), human development has had some direct conversion of aquatic habitats, but has had far reaching impacts on water quantity and quality throughout the West. Water management has diverted and altered natural water regimes, seriously degraded many aquatic environments, and caused considerable aquatic habitat fragmentation. Water pollution has changed the quality of the water negatively impacting many aquatic biological values.

3.7 Conservation Element Selection

As part of the REA process, Biological and Ecological Values, Landscape Values, and Ecosystem Service Values from both the terrestrial and aquatic domains will be aggregated into a meaningful framework for evaluating likely impacts from the various change agents. Each component, or conservation element, will be assessed separately, weighted as needed, and incorporated into a common decision support framework.

Normally, conservation element selection is an extremely important step in conservation planning, which is iteratively conducted during a data and information review process (Groves 2003). Good conservation element candidates are both important ecologically and have enough credible data upon which to base a scientific evaluation. For a rapid ecological assessment, responsive and informative conservation elements may be one of the important products.

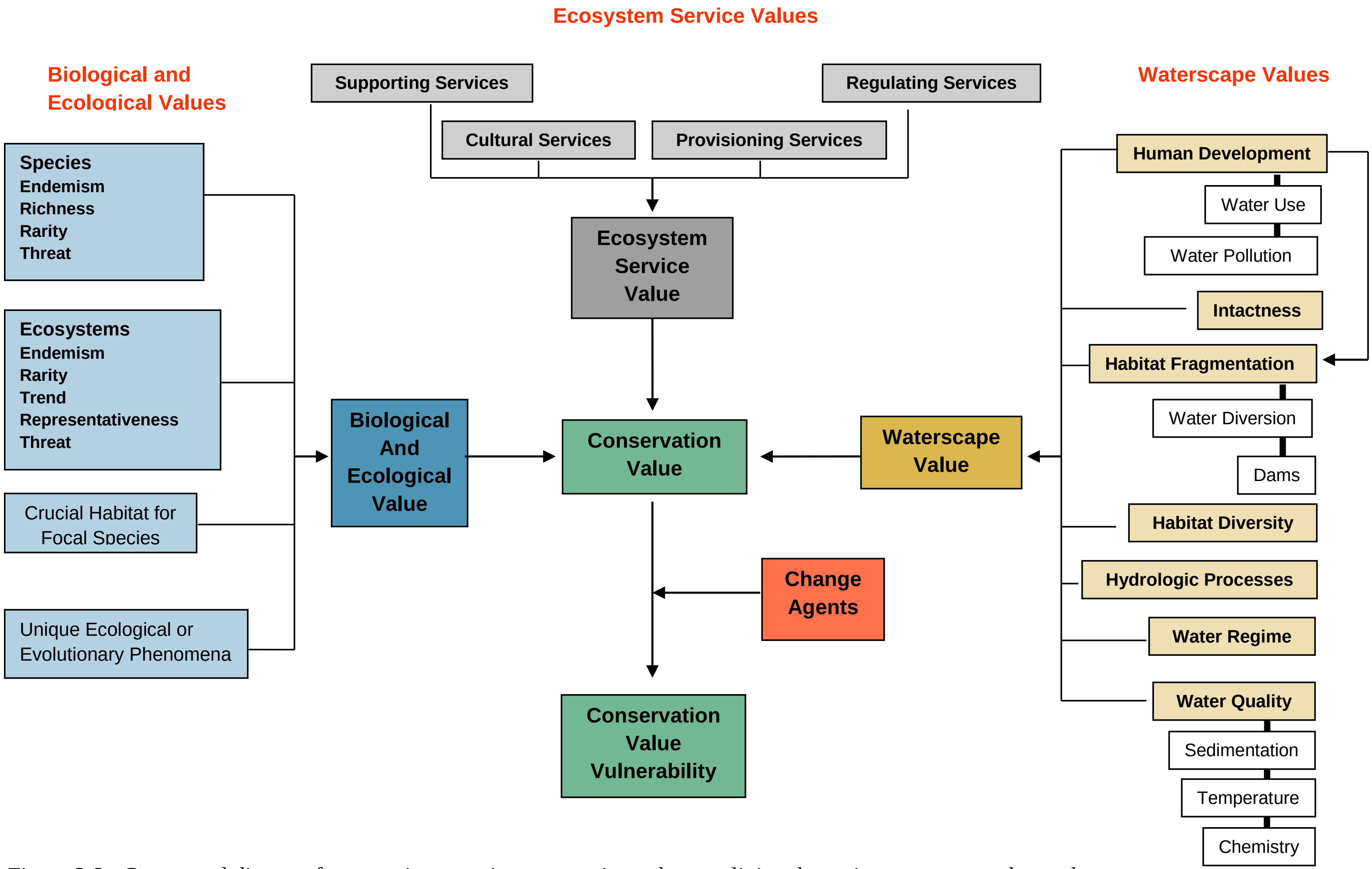


Figure 3-3. Conceptual diagram for assessing aquatic conservation values outlining the major components that makeup overall conservation value and vulnerability to change agents.

Table 3-1. List of criteria and attributes for riverine conservation value according to Dunn (2000).

Criteria	Attributes
1 Naturalness	1.1 undisturbed catchment 1.2 unregulated flow 1.3 unmodified flow 1.4 unmodified river/channel features 1.5 natural water chemistry 1.6 absence of inter-basin water transfer 1.7 intact and interconnected river elements 1.8 natural temperature regimes 1.9 natural processing of organic matter 1.10 natural nutrient cycling process 1.11 intact native riparian vegetation 1.12 absence of exotic flora or fauna 1.13 habitat corridor 1.14 natural in-stream faunal community composition 1.15 natural ecological processes, including energy base and energy flow in food webs
2 Representativeness	2.1 representative river system or section 2.2 representative river features 2.3 representative hydrological processes 2.4 representative aquatic macroinvertebrate communities 2.5 representative instream flora or riparian communities 2.6 representative fish communities or assemblages
3 Diversity and richness	3.1 diversity of rock types or substrate size classes 3.2 diversity of instream habitats, for example, pools, riffles, meanders, rapids 3.3 diversity of channel, floodplain (including wetland) morphologies 3.4 diversity of native flora or fauna species 3.5 diversity of instream or riparian communities 3.6 diversity of floodplain and wetland communities 3.7 diversity of endemic flora or fauna species 3.8 important bird habitat
4 Rarity	4.1 rare or threatened geomorphological features 4.2 rare or threatened ecological processes 4.3 rare or threatened geomorphological processes 4.4 rare or threatened hydrological regimes 4.5 rare or threatened invertebrate fauna 4.6 rare or threatened fish or other vertebrates 4.7 rare or threatened habitats 4.8 rare or threatened fauna 4.9 rare or threatened communities or ecosystems 4.10 rivers with unusual natural water chemistry

5 Special features	5.1 karst, including surface features 5.2 significant ephemeral floodplain wetlands 5.3 dryland rivers with no opening to ocean 5.4 important for the maintenance of downstream or adjacent habitats such as floodplain or estuary 5.5 important for the maintenance of karst system of features 5.6 important for migratory species or dispersal of terrestrial species 5.7 drought refuge for terrestrial or migratory species 5.8 habitat for important indicator or keystone taxa 5.9 habitat for flagship taxa 5.10 refuge for native species and communities in largely altered landscapes

Of course, IF conservation targets are well-defined, a more sophisticated assessment of site irreplaceability and vulnerability could be performed to prioritize conservation action for specific locations within an ecoregion (see Pressey and Nichols 1989; Margules and Pressey 2000; and Pressey and Taffs 2001). This technique involves assigning relative irreplaceability and relative vulnerability scores to individual sites and graphing them together on a single matrix (Figure 3-4).

Essentially, irreplaceability is a measure assigned to an area that reflects its relative importance in the context of a planning domain (e.g., biome, ecoregion, or site) for achieving a set of regional conservation targets (Cowling and Pressey 2001). Vulnerability is defined as the risk that human activity will transform a planning unit or site (e.g., intact natural landscape block or watershed, Margules and Pressey 2000). Irreplaceability and vulnerability can be considered in various proportions according to conservation priorities **(**Figure 3-7). Level I sites (in the shaded quadrant) score high on the irreplaceability scale and are under the highest threat levels from change agents. These sites will require the most immediate conservation attention. Level II sites are areas with high vulnerability but lower values of irreplaceability, perhaps because their targets are already conserved elsewhere or there is the potential to conserve them in other places. Level III sites are those with high irreplaceability scores but lower risk of being altered over the short-term. Level IV sites are not presently thought to be vulnerable and are generally replaceable (targets are already conserved elsewhere or other choices exist). Noss et al. (2002) give an example of how this technique was applied to the greater Yellowstone region of the western U.S.

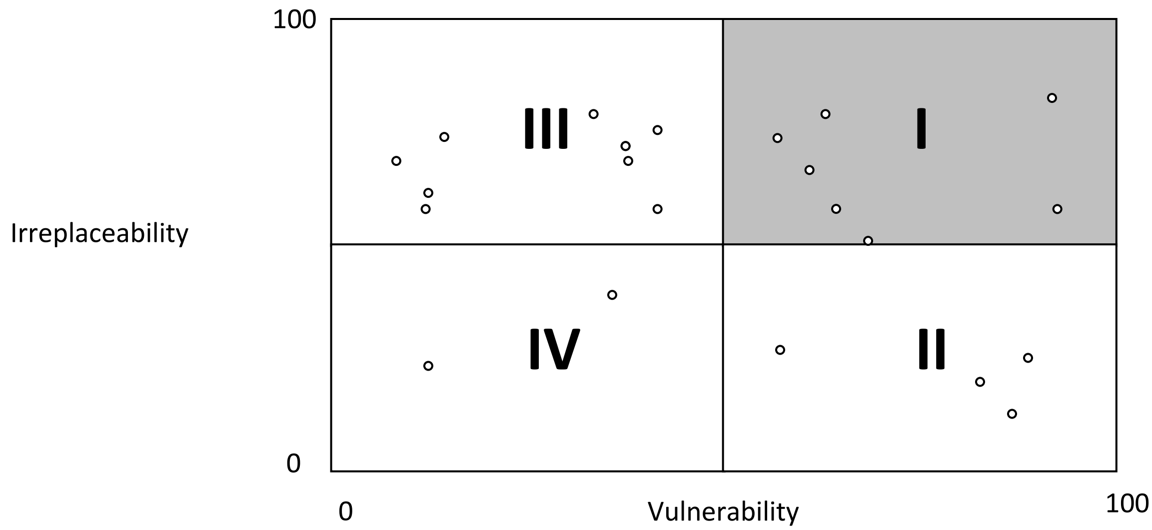


Figure 3-4. Irreplaceability and vulnerability matrix diagram identifying various categories of conservation priority of specific sites (denoted as small circles).

Our assumption is that for the Sonoran Desert REA, well-defined targets will be one of the final outcomes of the initial assessment making this level of assessment at this time inappropriate. However, it will be possible and important to attempt to quantify vulnerability of terrestrial and aquatic conservation elements present in specific locations to a wide range of impacts—urban development, agricultural expansion, energy development—that will provide important information at the ecoregional scale (see Change Agents sections).

3.8 References Cited

Allen, C.D. (technical editor). 1996. Fire effects in southwestern forests: proceedings of the second La Mesa fire symposium. U.S. Forest Service General Technical Report RM-GTR-286, U.S. Forest Service, Reno, Nevada. 216pp.

Andersen, D.C. 1994. Are cicadas (Diceroprocta apache) both a "keystone" and a "critical link" specie in lower Colorado River riparian communities? *The Southwestern Naturalist* 39(1):26–33.

Anderson, J. E. 1991. A conceptual framework for evaluating and quantifying naturalness. *Conservation Biology* 5(3):347–352.

Anderson, M., P. Comer, D. Grossman, C. Groves, K. Poiani, M. Reid, R. Schneider, B. Vickery, and A. Weakley. 1999. Guidelines for representing ecological communities in ecoregional conservation plans. The Nature Conservancy, Arlington, Virginia. Available online: www.conserveonline.org.

Andrén H. 1994. Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat: a review. *Oikos* 71(1994):355–366.

Angermeier, P.L. 2000. The natural imperative for biological conservation. *Conservation Biology* 14 (2):373–381.

Boon, P.J. 2000. The development of integrated methods for assessing river conservation value. Hydrobiologia 422/423: 413–428.

Carillo-Garcia, A., J.L. LeondelaLuz, Y. Bashan, and G.J. Bethlenfalvay. 1999. Nurse plants, mycorrhizae, and plants establishment in a disturbed area of the Sonoran Desert. *Restoration Ecology* 7:321–335.

Coppolillo, P., H. Gomez, F. Maisels, and R. Wallace. 2004. Selection criteria for suites of landscape species as a basis for site-based conservation. Biological Conservation 115: 419–430.

Cousins, S.A.O., S. Lavore, and I. Davies. 2003. Modeling the effects of landscape pattern and grazing regimes on the persistence of plant species with high conservation value in grasslands in south-eastern Sweden. *Landscape Ecology* 18:351–332.

Cowling, R.M. and R.L. Pressey. 2001. Rapid plant diversification: planning for an evolutionary future. Proceedings of the National Academy of Sciences of the U.S. 98(10):5452–5457.

Dailey, G.C., S. Polasky, J. Goldstein, P.M. Kareiva, H. Mooney, L. Pejchar, T. Ricketts, J.E. Salzman, and R. Shallenberger. 2009. Ecosystem services in decision making: time to deliver. *Frontiers in Ecology and the Environment* 7(1):21–28.

Drezner, T. D., and B.L. Lazarus. 2008. The population dynamics of columnar and other cacti: a review. *Geography Compass* 2: 1–29.

Dunn, H. 2000. Identifying and protecting rivers of high ecological value. LWRRDC Occasional Paper 01/2000.

ForestEthics, Greenpeace, Natural Resources Defense Council, and Rainforest Action Network. 2006. Ecological Components of Endangered Forests. Available at: http://forestethics.org/downloads/EFDefinitions_April_2006_2.pdf (April 2007).

Groves, C.R. 2003. Drafting a Conservation Blueprint. Island Press, Washington, D.C. 457pp.

Harding, J.S., E.F. Benfield, P.V. Bolstad, G.S. Helfman, and E.B.D. Jones III. 1998. Stream biodiversity: a ghost of land past. Proceedings of the National Academy of Science 95:14843–14847.

Heal, G. 2000. Valuing ecosystem services. *Ecosystems* 3:24–30.

Holling, C.S. 1992. Cross-scale morphology, geometry, and dynamics of ecosystems. *Ecological Monographs* 62:447–502.

Jennings, S. 2000. Gap analysis: concepts, analysis, and recent results. *Landscape Ecology* 15:5–20.

Jennings, S. 2004. The High Conservation Value Forest Toolkit. Edition 1, 2003. Proforest. Available at: www.fscoax.org/principal.htm (April 2004).

Lambeck, R.J. 1997. Focal species define landscape requirements for nature conservation. *Conservation Biology* 11:849–856.

Margules, C. and R.L. Pressey. 2000. Systematic conservation planning. *Nature* 405:243–253.

Moyle, P.B. and P.J. Randall. 1998. Evaluating the biotic integrity of watersheds in the Sierra Nevada, California. *Conservation Biology* 12:1318–1326.

Noss, R.F. and A. Cooperrider. 1994. Saving Nature's Legacy: Protecting and Restoring Biodiversity. Defenders of Wildlife and Island Press, Washington, D.C. 416 pp.

Noss, R.F., M.A. O'Connell, and D.D. Murphy. 1997. The Science of Conservation Planning: Habitat Conservation Under the Endangered Species Act. Island Press, Washington, D.C. 246 pp.

Noss, R.F., C. Carroll, K. Vance-Borland, and G. Wuerthner. 2002. A multicriteria assessment of irreplaceability and vulnerability of sites in the greater Yellowstone Ecosystem. *Conservation Biology* 16:895–908.

O'Neill, R.V., D.L. Deangelis, J.B. Waide, and G.E. Allen. 1986. A hierarchical concept of ecosystems. Princeton University Press, Princeton, New Jersey.

Paine, R.T. 1969. A note on trophic complexity and community stability. *American Naturalist* 100:65–75.

Pickett, S.T.A. and J.N. Thompson. 1978. Patch dynamics and the design of nature reserves. *Biological Conservation* 13:27–37.

Power, M.E., D. Tillman, J.A. Estes, B.A. Menge, W.J. Bond, L.S. Mills, G. Dailey, J.C. Castilla, J. Lubchenco, and R.T. Paine. 1996. Challenges in the quest for keystones. *Bioscience* 46:609–620.

Pressey, R.L. and A.O. Nichols. 1989. Efficiency in conservation evaluation—scoring versus iterative approaches. *Biological Conservation* 50:199–218.

Pressey, R.L. and K.H. Taffs. 2001. Scheduling conservation action in production landscapes: priority areas in western New South Wales defined by irreplaceablity and vulnerability to vegetation loss. *Biological Conservation* 100:355–376.

Ricketts T.H., E. Dinerstein, D.M. Olson, C.J. Loucks, W. Eichbaum, D. DellaSalla, et al. 1999. Terrestrial Ecoregions of North America: A Conservation Assessment. Island Press, Washington, D.C. 485 pp.

Shelford, V.E. 1926. Naturalist's Guide to the Americas. Williams and Wilkins, Baltimore, Maryland.

Soulé, M.E., and J. Terborgh. 1999. Continental Conservation: Scientific Foundations of Regional Reserve Networks. Island Press, Washington, D.C. 227 pp.

Stoms, D.M. 1992. Effects of habitat map generalization I biodiversity assessment. *Photogrammetric Engineering and Remote Sensing* 58:1587–1591.

Strittholt, J.R. and D. A. DellaSala. 2001. Importance of roadless areas in biodiversity conservation in forested ecosystems: A case study—Klamath-Siskiyou Ecoregion, U.S.A. *Conservation Biology* 15(6): 1742–1754.

Suzán, H., G.P. Nabhan, and D.T. Patten. 1996. The importance of *Olneya tesota* as a nurse plane in the Sonoran Desert. *Journal of Vegetation Science* 7(5):635–644.

The Millennium Assessment. 2005. Millennium Ecosystem Assessment Synthesis Report. Available at: http://www.millenniumassessment.org/en/index.aspx (April 2005).

Vitousek, P.M., J.D. Aber, R.W. Howarth, G.E. Likens, P. A. Matson, D.W. Schindler, W.H. Schlesinger, and D.G. Tilman. 1997. Human alteration of the global nitrogen cycle: sources and consequences. *Ecological Applications* 7 (3):737–750.

Western Governors' Association. 2008. Wildlife Corridors Initiative. 142 pp.

Wiens, J. 2002. Predicting species occurrences: progress, problems, and prospects. Pages 739–749 *in* Scott, J.M., P.J. Heglund, M.L. Morison, Jr., J.B. Haufler, M.G. Raphael, W.A. Wall, and F.B. Samson (eds.). Predicting Species Occurrences: Issues of Accuracy and Scale. Island Press, Washington, D.C.

Wilcox, B.A. and D.D. Murphy. 1985. Conservation strategy: the effects of fragmentation on extinction. *American Naturalist* 125:879-887.

4 PRIMARY CHANGE AGENTS OF THE SONORAN DESERT

There were five major change agents of interest in the REA – Resource Use, Development, Invasives, Fire, and Climate Change. Change agents as used in the REA scope of work pertain to both current conditions (cumulative disturbance) and probable future conditions. For each of the conservation elements in question, the response to change has many dependencies – sensitivity to each particular change agent, impact of multiple change agents on the element, and even the interaction of the change agents themselves (e.g., the relationship of climate change, invasives, and fire, see Figure 4-1). Forecasting future conditions is complex as change agents are often stochastic and uncertain.

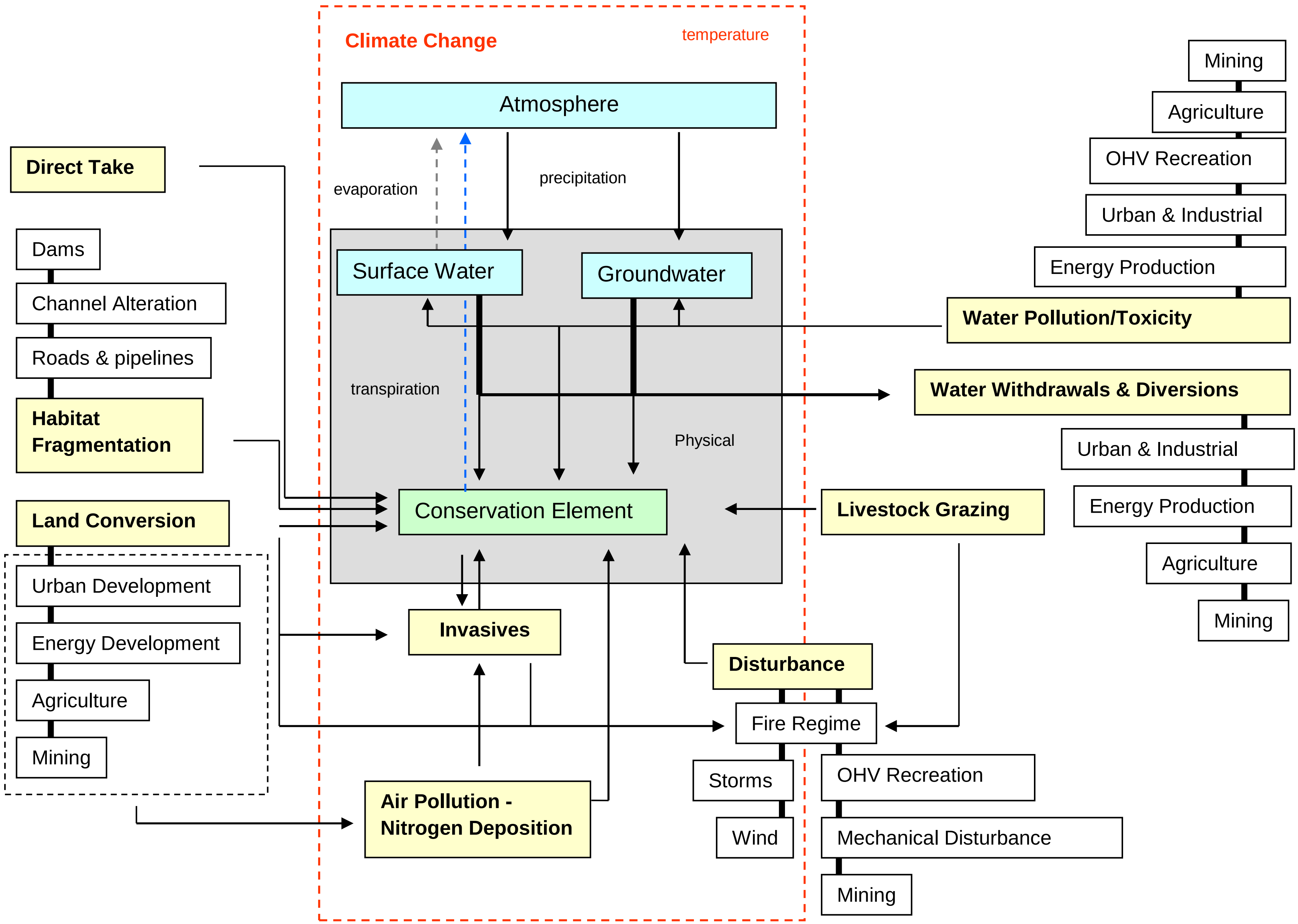


Figure 4-1. General conceptual model showing the influence of change agents on conservation elements.

4.1 Change Agent: Wildfire (Altered Fire Regimes)

4.1.1 Introduction

Fire is a natural ecosystem process in many regions. In any given region, species are typically adapted to a particular fire regime, which can be characterized in terms of fire frequency, seasonality, severity, and size (Pausas and Keeley 2009). The degree to which fire may become an ecologically significant change agent is related to the extent to which the fire regime has been altered compared to reference conditions and the associated effects of the altered fire regime on the vegetation community. For example, certain plant communities adapted to frequent, low-intensity fire are threatened by the consequences of decades of effective fire suppression, which can increase the potential for large, high-severity fires. In contrast, other communities adapted to very infrequent or absent fire are now threatened by increases in fire frequency due to invasive plants and increasing human ignitions.
Key management questions for fire include:

1. Where are the areas that have been changed by wildfire between 1999 and 2009?
2. Where are the areas with potential to change from wildfire?
3. Where are fire-adapted communities?

4.1.2 Background

Fire regimes have been altered in many Southwestern ecosystems compared to reference conditions that would have been present prior to Euro-American settlement. In recent decades, invasive species and human activities (e.g., grazing, urbanization, fire suppression), as well as other sources of human ignitions, have altered fire regimes in many fire-adapted ecosystems and introduced fire to other ecosystems that historically never experienced fire. Some widely-distributed invasive species, such as cheatgrass and red brome, increase fire frequency, size, and duration of the fire season by increasing fine fuel loads and continuity, thus allowing fires to spread into areas that were once fuel-limited (Hunter 1991, Brooks 1999, Brooks and Pyke 2001). These alterations to fire regime can promote further species invasion and thus create a tight feedback loop of increasing fire frequency (Mack and D'Antonio 1998). In the western US, the source of invasions has been linked to various anthropogenic disturbances, including but not limited to grazing, transportation (roads and trains), logging, and residential development (Kemp and Brooks 1998). Just as exotic species are likely to spread from these areas, human-caused ignitions are also likely to increase in areas with higher levels of human presence (Syphard et al. 2007, 2008).

In many ecosystems where fire historically served an important ecological function, several decades of effective fire suppression, combined with alterations to fuel load and pattern by anthropogenic land use and management practices, has led to conversions in vegetation type (e.g., shrub encroachment in semi-desert grasslands or pinyon-juniper woodland encroachment into sagebrush communities) or structure (e.g., increased canopy density as well as surface and canopy fuel loads, McPherson 1995, Van Auken 2000, Keane et al. 2002). Unless fuel loads are reduced, or unless fire occurs under non-severe weather conditions, fires in many of these communities may now become abnormally large and severe, which can result in dramatic reduction in aboveground live biomass, leading to cascading ecological impacts (DellaSalla et al. 2004, Lehmkuhl et al. 2007, Hurteau and North 2009).

Different species may be differentially affected by changes in fire regime, and over different spatial and temporal scales. Fire regimes can be highly variable over space and time—even among vegetation types within close proximity to one another (Pyne et al. 1996, Wells et al. 2004). The direction of change of a fire regime and associated effects to the vegetation community may also vary widely over short distances, depending in part on landscape position, disturbance history, and recent rapid changes in vegetation composition or structure, such as invasion by exotic plant species. Thus care must be exercised in calculating and interpreting measures of alteration of fire regimes and subsequent effects to vegetation communities. Unfortunately, comprehensive data describing fire regimes under reference conditions or current conditions are often lacking, particularly in deserts where fire historically has not occurred, thus necessitating the use of available data that may be widely different in spatial and temporal scale as well sampling density.

4.1.3 Conceptual Model Description

The Sonoran Desert is dominated by desert and semi-desert vegetation types that rarely if ever supported fire under pre-settlement reference conditions. These systems were typically fuel-limited with relatively small fires occurring infrequently, often following years of high moisture availability (Brooks et al. 2007, Brooks & Matchett 2006). Vegetation communities are often fire-intolerant or fire-sensitive, and are typically slow to recover after fire, if recovery is even possible. Reference condition fire regimes are poorly documented, in part because the physiology of the major vegetation species does not support methods well-established in the field of dendrochronology. In particular, these species often experience high mortality from fire rather than surviving and recording multiple fire events. Furthermore, because fire rarely occurred historically, there is little fire record before recent years.
Recent alterations to fire regime include the following factors:

- Invasion by exotic plants and subsequent increases in fine fuels and connectivity (Brooks et al. 2007)
- Increases in human ignitions

The conceptual model (Figure 4-2) illustrates the interaction of human activities, invasive species, fire, and native communities.

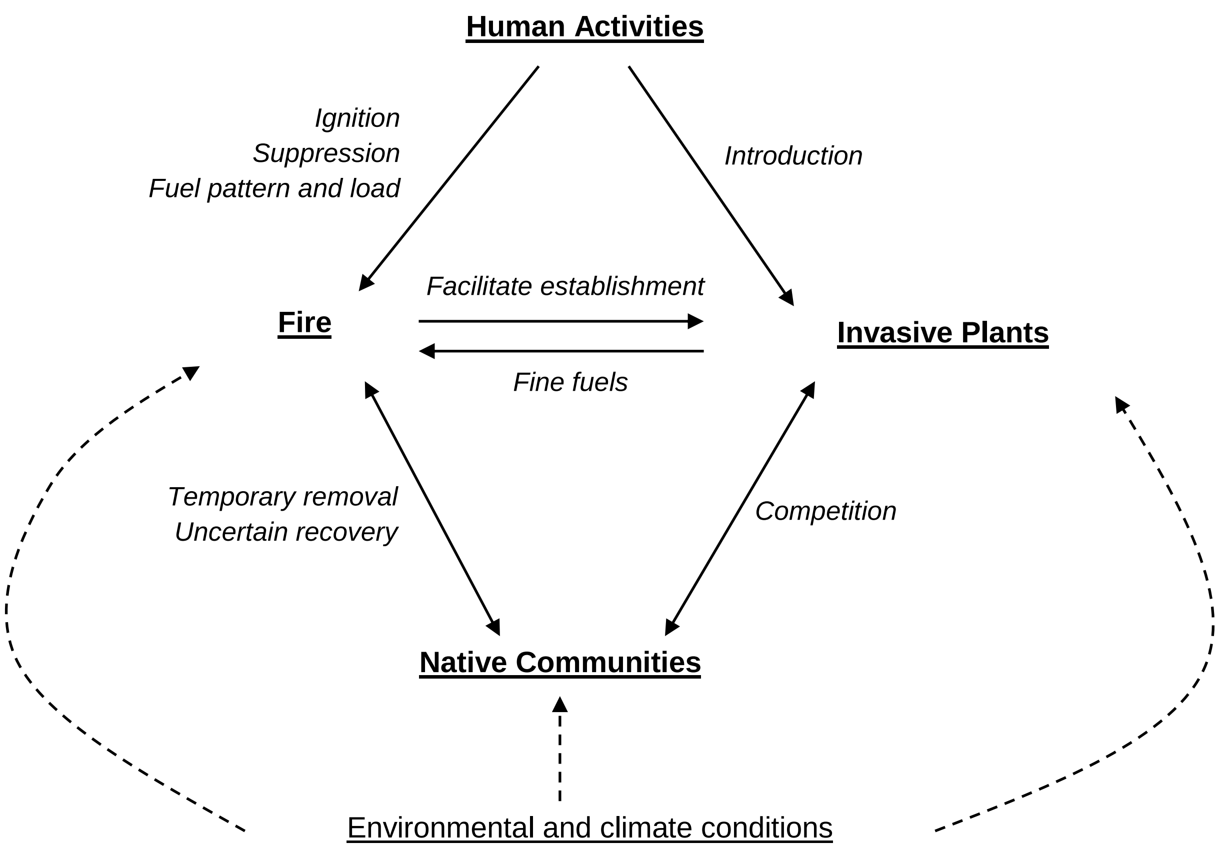


Figure 4-2. Conceptual diagram illustrating the interaction of drivers in wildfires.

4.1.4 Required Input Data Layers

Data needs required to address the management questions include the following:

- Reference condition fire regime characteristics
    - Frequency
    - Severity
- Recent fire locations and boundaries (1999-2009)
- Current vegetation type and structure
- Reference condition vegetation type
- Current succession classes
- Estimates of ecological / fire regime departure between reference and current conditions
- Fire regime condition class
- Current fuel characteristics
- Locations of fire-mediating invasive plant species
- Potential invasion locations for fire-mediating invasive plants
- Fire ignition probability
- Fire occurrence probability
- Wildland urban interface

This list reflects an ideal set of data that we would use to answer the management questions given the scope and duration of this project. However, not all of these data will be available, usable, or in an appropriate temporal or spatial scale. Some of these data also reflect derived products created or obtained through other parts of this project, in particular the distributions of invasive plants. Fire ignition or probability maps will also be derived as part of this project and combined with other data to answer management questions.

Raw Data

Table 4-1**.** Raw datasets proposed for use in evaluating key management questions related to fire.

Dataset	Source	Location	MQs
Lightning strike locations	BLM		2
Invasive plant locations	SWEPIC?	http://sbsc.wr.usgs.gov/research/projects/swepic/swemp/swempA.asp	2,4
Roads			2
Urban areas			2,4
Fuel treatment areas	BLM?		2

Previously Derived Data Layers

Table 4-2**.** Previously derived datasets proposed for use in evaluating key management questions related to fire.

Dataset	Source	Location	MQs
Reference condition fire frequency	LANDFIRE	http://www.landfire.gov	2-4
Reference condition fire severity	LANDFIRE	http://www.landfire.gov	2-4
Reference condition fire regime group	LANDFIRE	http://www.landfire.gov	2-4
Biophysical settings (reference condition vegetation)	LANDFIRE	http://www.landfire.gov	2-4
Existing vegetation type	LANDFIRE	http://www.landfire.gov	1,2,4
Existing vegetation structure	LANDFIRE	http://www.landfire.gov	1,2,4
Existing vegetation type (Refresh)	LANDFIRE	http://www.landfire.gov	1,2,4
Existing vegetation structure (Refresh)	LANDFIRE	http://www.landfire.gov	1,2,4
Anderson Fire Behavior Fuel Models	LANDFIRE	http://www.landfire.gov	2
Existing vegetation type	SW ReGAP	http://fws-nmcfwru.nmsu.edu/swregap/	1,2,4
Succssion classes	LANDFIRE	http://www.landfire.gov	All
Fire regime condition class	LANDFIRE	http://www.landfire.gov	3
Fire regime departure index	LANDFIRE	http://www.landfire.gov	3
Disturbance 1999-2008	LANDFIRE	http://www.landfire.gov	1
Large fire perimeters (1999-2007)	LANDFIRE	http://www.landfire.gov	1
Fire perimeters and locations (2000-2009)	USGS RMGSC	http://rmgsc.cr.usgs.gov/outgoing/GeoMAC/historic_fire_data/	1

Dataset	Source	Location	MQs
Red brome distribution	USGS	Availability unknown	2,4
Wildland urban interface	Hammer et al. 2004	http://silvis.forest.wisc.edu/old/Library/HousingData.php	4
Wildland urban interface	BLM?		4

Table 4-3. Datasets developed by other components of this project for use in evaluating key management questions related to fire.

Dataset	Management Question(s)
Sahara Mustard current and future dist.	1-4 Newly added species 3-15-11
Red brome current and future distribution	1-4
Tamarisk current and future distribution	1-4
Buffelgrass current and future distribution	1-4
Current and future distributions of fire-sensitive and fire-intolerant conservation elements	4

4.1.5 Model Assumptions

We make the following assumptions in the modeling approach discussed below:

1. Required input data will be available at the time of the analysis
2. Fire occurrence probability can be estimated using available data, including lightning density and proxies for human ignitions (distance to human infrastructure)
3. Reference condition fire regimes are suitably well defined to permit comparison with current conditions
4. Current vegetation composition and structure can be approximated using available data
5. Effects to vegetation based on fire severity can be inferred using available data
6. Recent fires can be approximated using available data for fire perimeters, locations, and severity
7. Distributions of fire-mediating invasive species can be adequately captured, and can be used to approximate areas of higher risk for fire spread
8. The effect of fire can be treated simplistically for fire-intolerant communities

4.1.6 Methods and Tools

Management Question 1: Where are the areas that have been changed by wildfire between 1999 and 2009?

Identifying areas that have been altered by wildfire in the recent past will require combining available fire locations for this time period with measures of fire severity and pre- and post-fire vegetation, along with information describing a vegetation community's response to various severities of fire (Figure 4-3). Change in this context is taken to mean an ecologically significant alteration of the vegetation community composition or structure. For vegetation communities in much of the Sonoran Desert, any fire likely caused a large change in vegetation composition and structure due to the low physiological tolerance of species to fire. Furthermore, any fire may create opportunity for invasion by exotic plants, which would begin a feedback loop that would continue to increase fire frequency and changes in community composition toward uncharacteristic conditions. Thus, a simple overlay of fire perimeters in these fire-sensitive and fire-intolerant communities likely indicates areas that have been changed by fire, and it may not be necessary to determine post-fire vegetation in these communities. For vegetation communities that range from fire-sensitive to fire-adapted (primarily near the ecoregion buffer areas), it would be necessary to determine vegetation response to the severity of fire that was observed to determine the degree of post-fire vegetation change. In some cases, it may be possible to approximate both pre- and post-fire vegetation directly using available data (e.g., LANDFIRE EVT and LANDFIRE EVT Refresh).

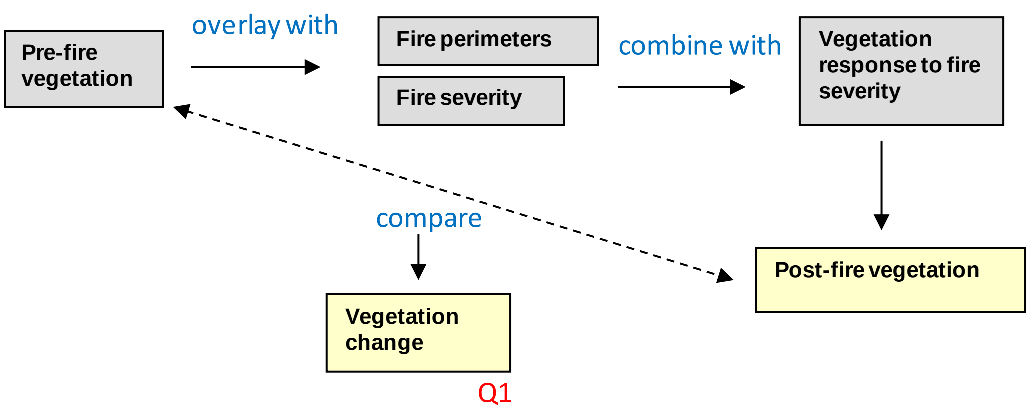


Figure 4-3. Process model for areas of recent change due to fire.

Management Question 2: Where are areas with potential to change from wildfire?

To determine the areas that have strong potential for alteration by wildfire, it will be necessary to map the areas with the highest risk for fire occurrence and spread, and then determining how fire in those areas would change vegetation composition or structure (Figure 4-4). The degree to which fire occurrence and spread probability can be approximated will depend on available data, which in many cases may be sparse or widely variable in scale. In some vegetation communities, fire history or reference condition fire regime could be used to inform fire occurrence probability. However, in many areas of the Sonoran Desert, fire history data are largely unavailable and likely non-informative with respect to current fire occurrence probabilities. Additional proxies of will be required to estimate areas of high fire occurrence probability, including locations of previous fires, distance from human infrastructure (e.g., roads), and locations of invasive plant species. Furthermore, in these vegetation communities, the occurrence of any fire likely constitutes a larger impact to current vegetation composition and structure, and can therefore fire occurrence can be treated more simplistically than in vegetation communities in which fire was a more frequent disturbance under reference conditions.

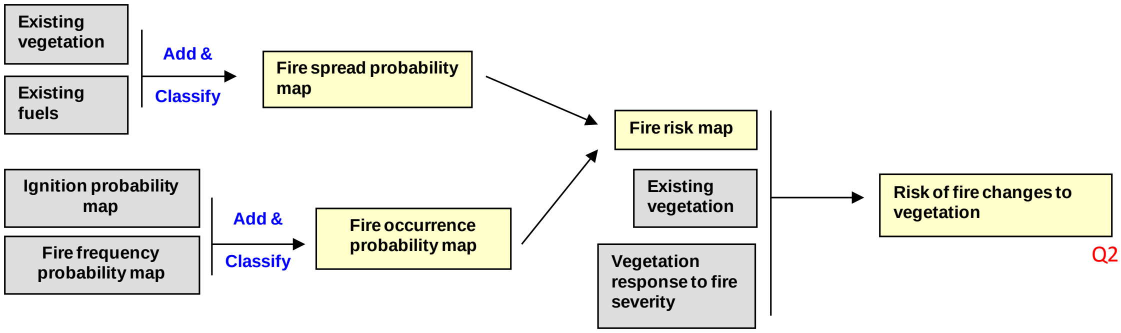


Figure 4-4. Process model for identifying areas with potential for alteration by fire.

Management Question 3: Where are fire-adapted communities?

To determine where vegetation communities may be located that are more tolerant to fire, it is necessary to identify those communities which are sensitive or intolerant to increases in fire frequency or severity (Figure 4-5). For vegetation communities in which fire is assumed to be rare or absent (much of the Sonoran Desert), fire severity (removal of aboveground biomass) and subsequent mortality is usually high due to the low physiological tolerance of the component species. Furthermore, the high mortality of the vegetation community provides an opportunity for colonization by invasive plants, which increases the likelihood of repeated fires in that area. Thus nearly any fire in these communities is likely to be detrimental. In contrast, high fuel loads in vegetation communities that would have experienced more frequent fires under reference conditions may lead to higher severity fires and higher subsequent mortality. In these cases, the reintroduction of fire and subsequent high mortality may cause large shifts in species composition and also create opportunities for invasion by exotic species.

To address this management question, we propose to use existing vegetation, reference condition fire regimes, biophysical settings (and their corresponding predicted proportions of succession classes), and current proportions of those succession classes to identify the following classes of fire sensitivity:

1. Fire-sensitive or intolerant vegetation communities (risk of increased frequency).
2. Fire-sensitive to fire-tolerant with altered composition or structure due to human activities (fire suppression), with higher likelihood of severe fires (risk of increased severity).
3. Fire-tolerant with composition and structure similar to reference conditions (low risk).

Fire occurrence in classes 1) or 2) would likely indicate a detrimental effect to the vegetation community. It is important to note that fire in class 3) may serve a valuable ecological function, and that it is the effective suppression of fires that is detrimental to the vegetation community; this effect is not identified here. These classes can then be combined with areas of high likelihood of fire occurrence and spread (fire risk) developed to address management question 2 above, to identify those areas that are sensitive to increases in fire frequency likely to experience fire and areas that are sensitive to increases in fire severity likely to experience fire. The small areas of fire-adapted communities that remain once fire-sensitive communities are identified will likely be located on foothills and interior mountain ranges in the transition to woodland.

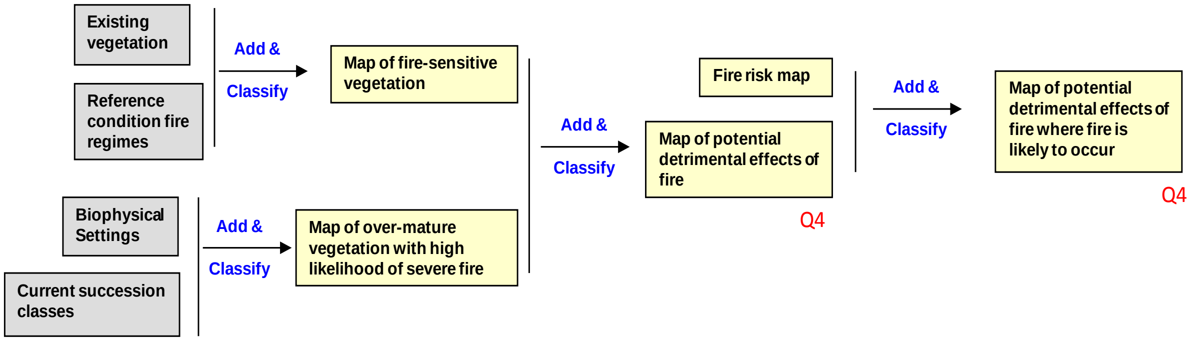


Figure 4-5**.** Process model for identifying areas of fire-adapted vegetation communities.

4.1.7 References Cited

Brooks, M.L. 1999. Alien annual grasses and fire in the Mojave Desert. *Madroño*. 46: 13–19.

Brooks, M.L., T.C. Esque, and T. Duck. 2007. Fuels and fire regimes in creosotebush, blackbrush, and interior chaparral shrublands. Pages 97–110 *in* General Technical Report RMRS-GTR-202, U.S. Forest Service, Washington, D.C.

Brooks, M.L. and J.R. Matchett. 2006. Spatial and temporal patterns of wildfires in the Mojave Desert, 1980-2004. *Journal of Arid Environments* 67:148–164.

Brooks, M.L., and D.A. Pyke. 2001. Invasive plants and fire in the deserts of North America. Pages 1–14 *in* Galley, K.E.M., and T.P. Wilson (eds.). Proceedings of the Invasive Species Workshop: the Role of Fire in the Control and Spread of Invasive Species. Fire Conference 2000: the First National Congress on Fire Ecology, Prevention, and Management. Miscellaneous Publication No. 11, Tall Timbers Research Station, Tallahassee, FL.

DellaSalla D.A., J.E. Williams, C.D. Williams, J.F. Franklin. 2004. Beyond smoke and mirrors: a synthesis of fire policy and science. *Conservation Biology* 18: 976–986.

Hunter, R.B. 1991. Bromus invasions on the Nevada Test Site: present status of *B. rubens* and *B. tectorum* with notes on their relationship to disturbance and altitude. *Great Basin Naturalist* 51: 176–182.

Hurteau, M., and M. North. 2009. Fuel treatment effects on tree-based carbon storage and emissions under modeled wildfire scenarios. *Frontiers of Ecology and the Environment* 7: 409–414.

Kemp, P., and M.L. Brooks. 1998. Exotic Species of California Deserts. Fremontia 26:30–34.

Keane, RE, T. Veblen, K.C. Ryan, J. Logan, C. Allen, and B. Hawkes. 2002. The cascading effects of fire exclusion in the Rocky Mountains. Pages 133–153 *in* Rocky Mountain futures: An ecological perspective, Island Press, Washington, DC.

Lehmkuhl J.F., M. Kennedy, E.D. Ford, P.H. Singleton, W.L. Gaines, and R.L. Lind. 2007. Seeing the forest for the fuel: Integrating ecological values and fuels management. *Forest Ecology and Management* 246:73–80.

Mack, M. C., and C. M. D'Antonio. 1998. Impacts of biological invasions on disturbance regimes. *Trends in Ecology and Evolution* 13:195–198.

McPherson, G.R. 1995. The role of fire in desert grasslands. Pages 130–151 i*n* McClaran, M.P., and T. R.Van Devender (eds.). The desert grassland, University of Arizona Press, Tucson, Arizona.

Pausas, J.G., and J.E. Keeley. 2009. A burning story: The role of fire in the history of life. *Bioscience* 59:593–601.

Pyne, S.J., P.L. Andrews, and R.D. Laven. 1996. Introduction to wildland fire. WileyPress, New York.

Syphard, A.D., V.C. Radeloff, J.E. Keeley, T.J. Hawbaker, M.K. Clayton, S.I. Stewart, and R.B. Hammer. 2007. Human influence on California fire regimes. *Ecological Application*s 17:1388–1402.

Syphard, A.D., V.C. Radeloff, N.S. Keuler, R.S. Taylor, T.J. Hawbaker, S.I. Stewart, and M.K. Clayton. 2008. Predicting spatial patterns of fire on a southern California landscape. *International Journal of Wildland Fire* 17:602–613.

Van Auken, O.W. 2000. Shrub invasions of North American semiarid grasslands. *Annual Review of Ecological Systems* 31:197–215.

Wells, M.L., J.F. O'Leary, J. Franklin, J. Michaelsen, and D.E. McKinsey. 2004. Variations in a regional fire regime related to vegetation type in San Diego County, California. *Landscape Ecololgy* 19: 139–152.

4.2 Change Agent: Invasive Species

Invasive species are considered change agents because they alter ecosystem processes and adversely affect natural resources in a region; they have the potential to expand and/or shift their ranges in the future with continued land cover disturbance and projected climate change. Invasive annuals outcompete native annuals by using soil nutrients and water at a greater rate or earlier in the season and thus the invasives regularly produce greater biomass (DeFalco et al. 2003, DeFalco et al. 2007). Three invasive plant species of concern have been identified for the Sonoran Desert ecoregion: red brome (*Bromus madritensis* subsp. *rubens*), buffelgrass (*Pennisetum ciliare*), and tamarisk (*Tamarix* spp.). All three species have been implicated in changes in the fire regime—even tamarisk, which typically grows in riparian areas. [**Note**: Saharan mustard (*Brassica tournefortii*) was added as an important Sonoran Desert invasive species in March 2011.] Nonnative species tend to dominate post-fire landscapes and fuel recurrent and more frequent fire (Figure 4-6). For example, historical photographic and anecdotal evidence indicate that since red brome invaded the southwestern US early in the 20th century, fire return intervals have shortened to 20 years compared to historical estimates of >100 years. The cumulative evidence supports the assumptions of D'Antonio and Vitousek's (1992) grass/fire model (Brooks 2008).

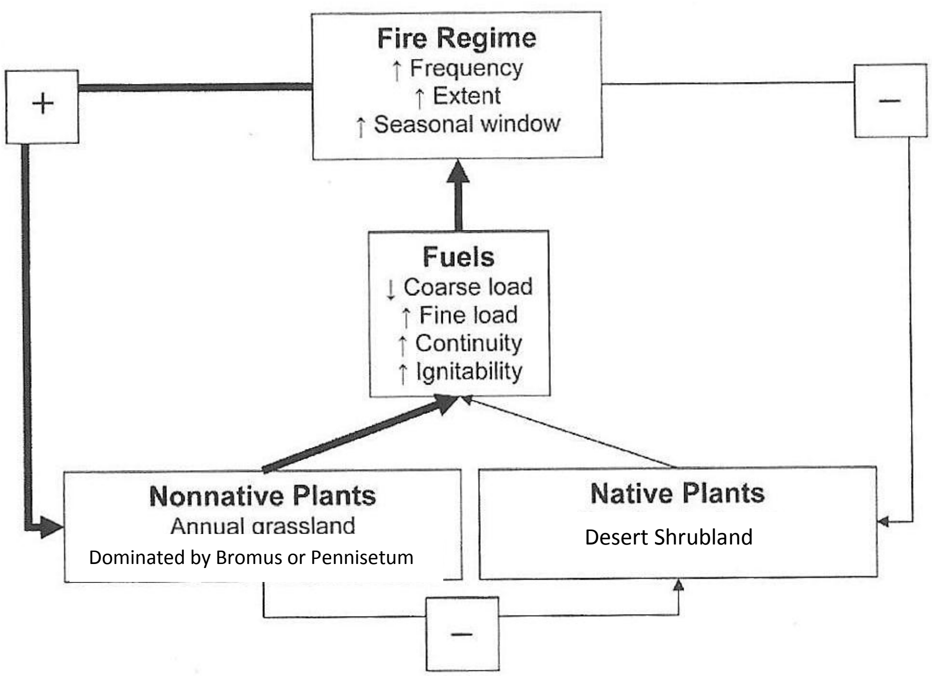


Figure 4-6. Nonnative annual plants create a grass/fire cycle in invaded areas by producing large amounts of litter that contribute to increased fire frequency, intensity, and extent (Brooks 2008). Even tamarisk introduces more frequent fire to riparian areas through a buildup of litter and a reduction of soil moisture.

Continued changes in fire cycle combined with projected changes from global climate change raise the possibility of widespread type conversion of desert shrublands to low-diversity nonnative annual grasslands with major effects on ecosystem function and the abundance of desert wildlife (Figure 4-7, Smith et al. 2000, Dukes and Mooney 2004).

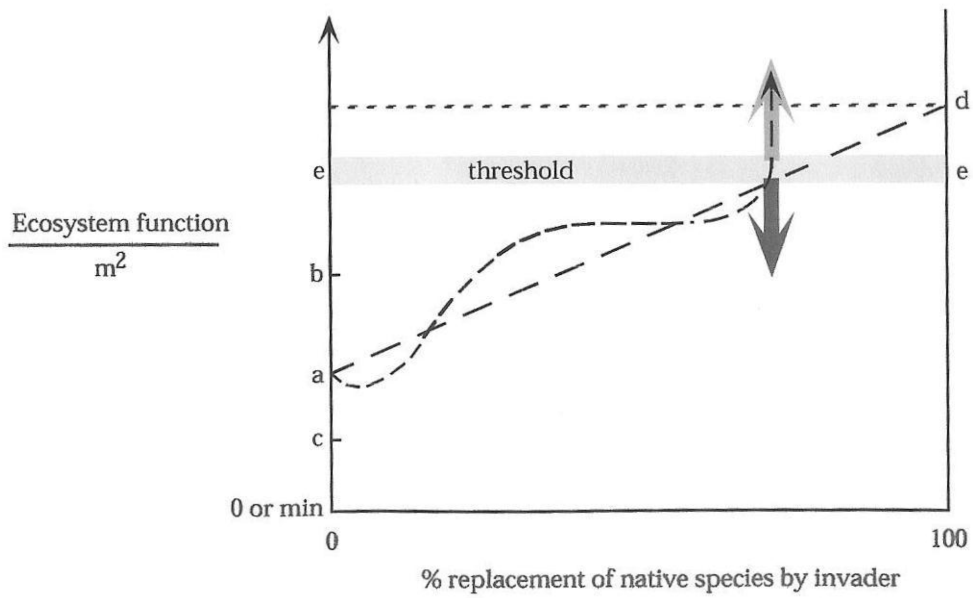


Figure 4-7. An invasive species' composition within an ecosystem may vary between *b* and *c* over time (wavy dashed line). The hypothetical invader in this scenario has on average a higher ecosystem function than the native species (*d*). Eventually a threshold may be crossed that may cause irreversible changes in the ecosystem with a type conversion, in this case from desert shrubland to annual grassland (Dukes and Mooney 2004)

Four key management questions relate to invasive species:

1. Where are areas dominated by this invasive species?
2. Where are the areas of potential future encroachment from this invasive species?
3. Where are areas of suitable biophysical setting (precipitation/soils, etc.) with restoration potential?
4. Where/how will the distribution of dominant native plant species and invasive species change from climate change?

In order to address these management questions, it will be necessary to identify (1) existing occupied habitat, (2) existing potential habitat, based on species-specific growth criteria, and (3) future potential habitat, based on species-specific growth criteria in conjunction with predicted climate change scenarios.

In the following sections, each change agent is described with regard to general characteristics, biological and physical criteria, effect(s) on conservation elements, and data requirements and approach for addressing management questions.

4.2.1 Red Brome (*Bromus madritensis* subsp. *rubens*)

Red brome is a nonnative, annual grass from the Mediterranean region that was introduced into the western United States in the mid-1800s (Salo 2005, Newman 2001). Initially evaluated for use as a forage plant, this species escaped cultivation to inhabit disturbed and cultivated sites before spreading into natural areas (Reid et al. 2008, Burgess et al. 1991). This ability to invade undisturbed habitat makes this species particularly problematic in the Southwest, where it alters fire regimes, thereby threatening native plant communities and associated wildlife species. It currently occupies vast areas in the Arizona Upland of the Sonoran Desert (ASDM 2010, Turner and Brown 1982).

Red brome typically occurs below 5,000 feet elevation on gentle to moderate slopes, often in shallow, sandy loam or clayey soils where it is tolerant of high salt and pH conditions (Wu and Jain 1978 *in* Newman 2001, Sampson et al. 1951 *in* Newman 2001). Red brome readily invades disturbed areas (Newman 2001), and it is enhanced by grazing and fire (Hulbert 1955). Red brome also invades undisturbed habitats (Burgess et al. 1991, Beatley 1966), including scrub communities and mesquite bosques in the Sonoran Desert (Simonin 2001).

Red brome is a prolific seed producer (Wu and Jain 1979). Seed is dispersed by wind, small mammals, and water (Drezner et al. 2001, Hulbert 1955). Red brome does not form a persistent soil seed bank; but it germinates earlier and with less rainfall than native annual species, and it may displace natives in wet years (Reid et al. 2008, Salo 2005, Newman 2001, Beatley 1966). On the other hand, red brome populations may be adversely affected by drought (Salo 2004).

4.2.1.1 Conceptual Model

The conceptual model illustrates the causal relationships between red brome invasion and expansion and the subsequent effects on physical and biological systems (Figure 4-8). Rainfall may be a limiting factor in red brome distribution. The species grows densely only in wetter years. Under conditions of above-average winter rainfall, this species establishes and produces high biomass (Reid et al. 2008, Bowers 1987). DeFalco et al. (2007) found that red brome reduced the growth of Mojave Desert perennials (particularly perennial grasses) when winter rains allowed red brome establishment during winter; there was less effect when both native perennials and red brome germinated in the spring. Optimal germination occurs between 68–77°F (20–25C) and greater than 0.4 inches (1 cm) of precipitation (Hammouda and Bakr 1969). Red brome is frost-sensitive and killed by temperatures below 32°F (0° C, Hulbert 1955).

Red brome alters the natural fire regime of Sonoran desert ecosystems, resulting in more frequent, intense, and larger fires that favor this species' spread and persistence, at least in above-average winter rainfall years. Red brome creates a grass/fire cycle in invaded areas by producing large amounts of litter which contribute to increased fire frequency, intensity, and extent (Figure 4-7 and 4-8, Brooks 2008, Esque and Schwalbe 2002, Brooks 1999, McClaran and Brady 1994, D'Antonio and Vitousek 1992, Hunter 1991, Beatley 1966). This grass/fire cycle can result in type conversion of native shrub habitats, and it displaces native species by reducing or eliminating suitable habitat, consuming resources (e.g., water), and/or altering soil ecology (Figure 4-8). Native plant species that are particularly vulnerable are stem succulents, non-fire adapted shrubs and subshrubs, and annuals (Reid et al. 2008, Salo 2004, Rogers 1985). Saguaro cacti (*Carnegia gigantea*) are vulnerable to fire and individuals can be eliminated outright during or following a grass-carried burn (Esque et al. 2006). Although existing red brome is killed by fire, it quickly re-establishes from seed sources (Simonin 2001), thereby perpetuating the grass/fire cycle. Native wildlife species that are adversely affected by changes due to this invasive include insects, desert tortoise, raptors, and small mammals (Newman 2001). Red brome directly reduces forage quality and habitat structure (shrub thermal cover) for desert tortoise (Brooks and Esque 2002)

The persistent litter produced by red brome also alters soil ecology by increasing soil nitrogen levels. Increased soil nitrogen in a low-fertility system provides red brome with a competitive advantage over some native species (Salo et al. 2005, Brooks 2003, Burgess et al. 1991, Hunter 1991, Kay 1971, Hulbert

1955, Beatley 1966). Red brome may also indirectly alter geomorphology through increased wind and water erosion following fire (Figure 4-8).

Red brome is expected to expand its distribution in response to climate change. Warming temperatures may result in latitudinal and elevation shifts (Hernandez et al. 2010). Increases in atmospheric CO_2 levels and nitrogen deposition will enhance plant growth while decreasing water needs, and provide red brome additional competitive advantages over native species (Reid et al. 2008, Salo 2005, Smith et al. 2000). Smith et al. (2000) found that when subjected to increasing CO_2, red brome had substantially higher total plant biomass and seed production than native plants, although individual seed mass and nitrogen content decreased (Huxman et al. 1999). The reduction in seed quality for red brome at elevated CO_2 resulted in a pronounced decline in growth rate of subsequent seedlings (Huxman et al. 1998). Therefore, red brome appears to respond to elevated CO_2 with the production of many more seeds, but those seeds may be of lower quality. A climate change scenario that could result in a reduction in the distribution of red brome would include extended drought and reduced winter precipitation.

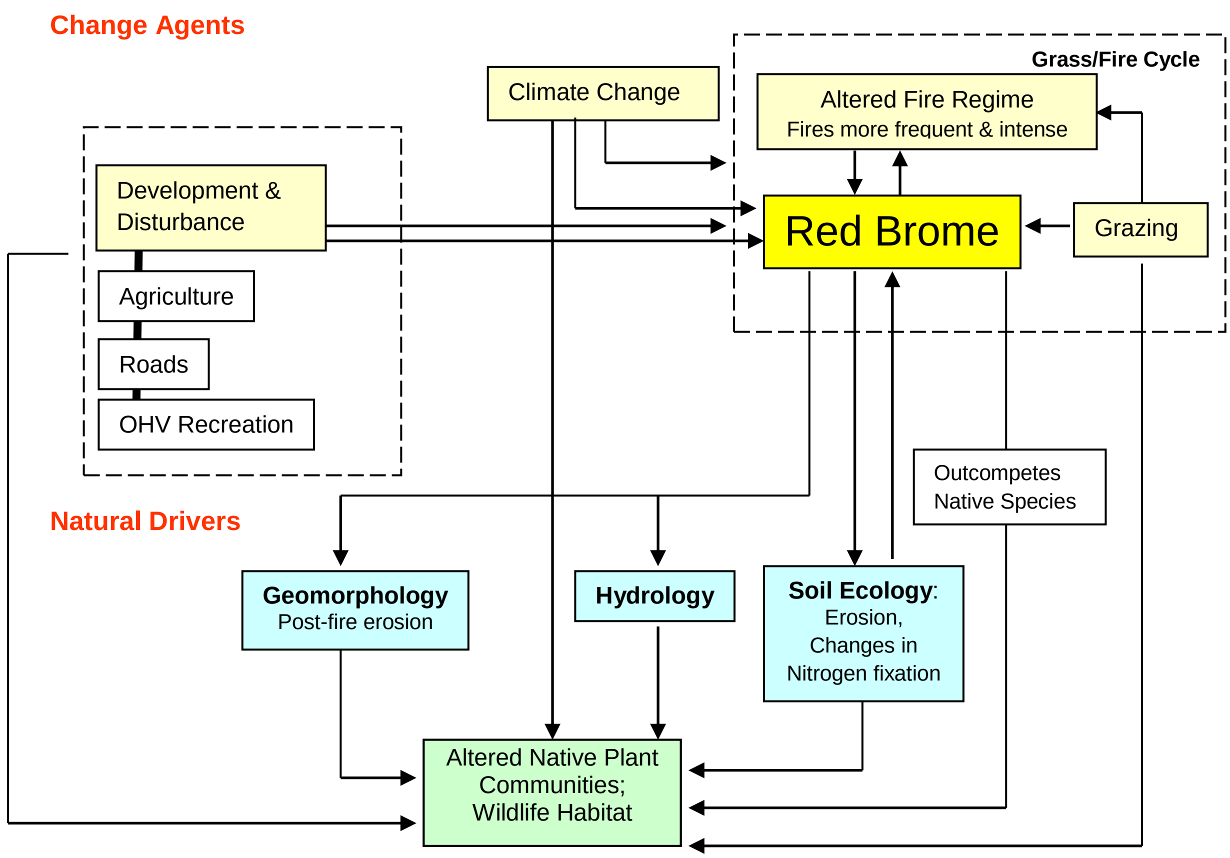


Figure 4-8. Conceptual model for red brome in the Sonoran Desert ecoregion.

4.2.1.2 Required Input Data Layers

We compiled a list of datasets we think are most relevant for answering the management questions (Tables 4-4 and 4-5). Datasets are divided into raw (unprocessed) datasets and previously processed datasets.

Raw Data

Table 4-4. Raw (unprocessed) datasets proposed for use in evaluating key management questions related to red brome.

Dataset	Source	Location	Management Questions (s)
Occurrence Records	Southwest Exotic Mapping Program (SWEMP)	http://sbsc.wr.usgs.gov/research/projects/swepic/swemp/swempA.asp	1-4
Soils	U.S. Department of Agriculture, NRCS STATSGO or SSURGO	http://soildatamart.nrcs.usda.gov	1-3
Hydrology	USGS Hydrology NHD	http://waterdata.usgs.gov/nwis	1-3
Topography	USGS	http://edc2.usgs.gov/geodata/index.php	1-3
Vegetation	SWReGAP or LANDFIRE	http://earth.gis.usu.edu/swgap/landcover.html and http://www.landfire.gov/NationalProductDescriptions23.php	1-4
Fire History	Recently Burned data from SWReGAP or LANDFIRE	http://earth.gis.usu.edu/swgap/landcover.html and http://www.landfire.gov/	1-3

Previously Processed Data

Table 4-5. Previously processed datasets proposed for use in evaluating key management questions related to red brome.

Dataset	Source	Location	Management Questions (s)
Occurrence Records	Southwest Exotic Mapping Program (SWEMP) 2007	http://sbsc.wr.usgs.gov/research/projects/swepic/swemp/swempA.asp	1-4
Precipitation	Oak Ridge National Laboratory Distributed Active Archive Center	http://daac.ornl.gov/ and http://app.databasin.org/app/pages/datasetPage.jsp?id=a6127300bf904831b7d647f7d966e87c	1-4
Temperature	Oak Ridge National Laboratory Distributed Active Archive Center	http://daac.ornl.gov/ and http://app.databasin.org/app/pages/datasetPage.jsp?id=a6127300bf904831b7d647f7d966e87c	1-4
Grazing Allotments	BLM	http://app.databasin.org/app/pages/galleryPage.jsp?id=bb9de27783a949fc8600078384558733	2

4.2.1.3 Attributes, Indicators, and Metrics

Ecological attributes are traits or factors that are necessary to maintaining a fully functioning species population, assemblage, community, or ecosystem. On a species level, they are traits that are necessary for species survival and long-term viability. Indicators are measurable aspects of ecological attributes. In the REAs, attributes and indicators are key elements used to answer management questions, parameterize models, and help explain the expected range in status and condition of individual conservation elements. We propose using attributes and indicators related to elevation, temperature, precipitation, and soils as a potential collection of environmental attributes in the modeling for red brome.

4.2.1.4 Model Assumptions

There are several assumptions on which the red brome models are based. These include 1) red brome out-competes and displaces native species; 2) red brome colonizes disturbed and undisturbed habitats; 3) red brome alters fire regimes and soil ecology, thus perpetuating its own persistence; 4) red brome prefers sandy loam or clay soils; red brome is prevalent in scrub communities and mesquite bosques; 5) rainfall is a limiting factor in red brome distribution; 6) red brome distribution is enhanced by grazing and fire; and 7) red brome distribution is expected to expand as a result of climate change.

4.2.1.5 Methods and Tools

The general approach for answering the management questions involves analyzing existing datasets using standard analytical tools in a Geographic Information System (GIS). We plan to use ESRI's ArcMap and ArcInfo to conduct the process/application model. Outputs from the conceptual process/application models (Figures 4-9) identify specific management questions by referencing their corresponding number in this REA module (see "Management Questions" section, above).

This process/analysis utilizes standard ArcGIS software tools. Using a combination of Intersect, Select, Merge, Dissolve, Export, etc. tools, we will utilize existing datasets to analyze and create new datasets that identify areas of primary REA concern for red brome. Output datasets will be displayed at the 5th field Hydrologic Unit Code (HUC), where appropriate. In general, existing and output datasets address 1) occurrences and 2) physical attributes that contribute to invasion success. One *example* process/application model is provided, below (Figures 4-9). Raw datasets are represented by gray boxes. Previously-processed datasets are represented by yellow boxes. Green boxes represent datasets that answer specific management questions (indicated by the management question [MQ] number in red). Lines and arrows indicate the process steps taken in the GIS to arrive at specific answers. Red lettered text generally indicates the management question addressed by the particular analysis. Although we have not provided all combinations for our proposed methodological approach, we have provided an example (Figure 4-9).

The example model (Figure 4-9) addresses future distribution questions (MQ 2) by identifying areas of potential future encroachment by red brome, based on species-specific attributes. Disturbed sites from grazing and burning are extremely important drivers for red brome, and they will be modeled using available spatial datasets. Suitable physical habitat for red brome is comprised of its preferred topography, soils, vegetation, temperature and precipation. An example of the proposed method/analysis flow we could take to answer management question two (MQ2 in Figure 4-9) would be to combine physical parameters, vegetation, and recently disturbed sites (grazing, fires). The resulting dataset would indicate areas of suitable red brome expansion by HUC5s.

Identifying areas of suitable biophysical setting (e.g., precipitation/soils) with restoration potential (MQ 3) will require mapping areas of existing, occupied habitat that could be considered for future management intervention through site manipulation and reseeding based on site characteristics and precipitation amounts, and that will no longer be suitable to support red brome in the future based on predicted climate change scenarios. Alternatively, sites with restoration potential may include areas where future seed germination is poor where habitats may become unsuitable into the future (compared to current conditions) due to changes in temperature and precipitation patterns.

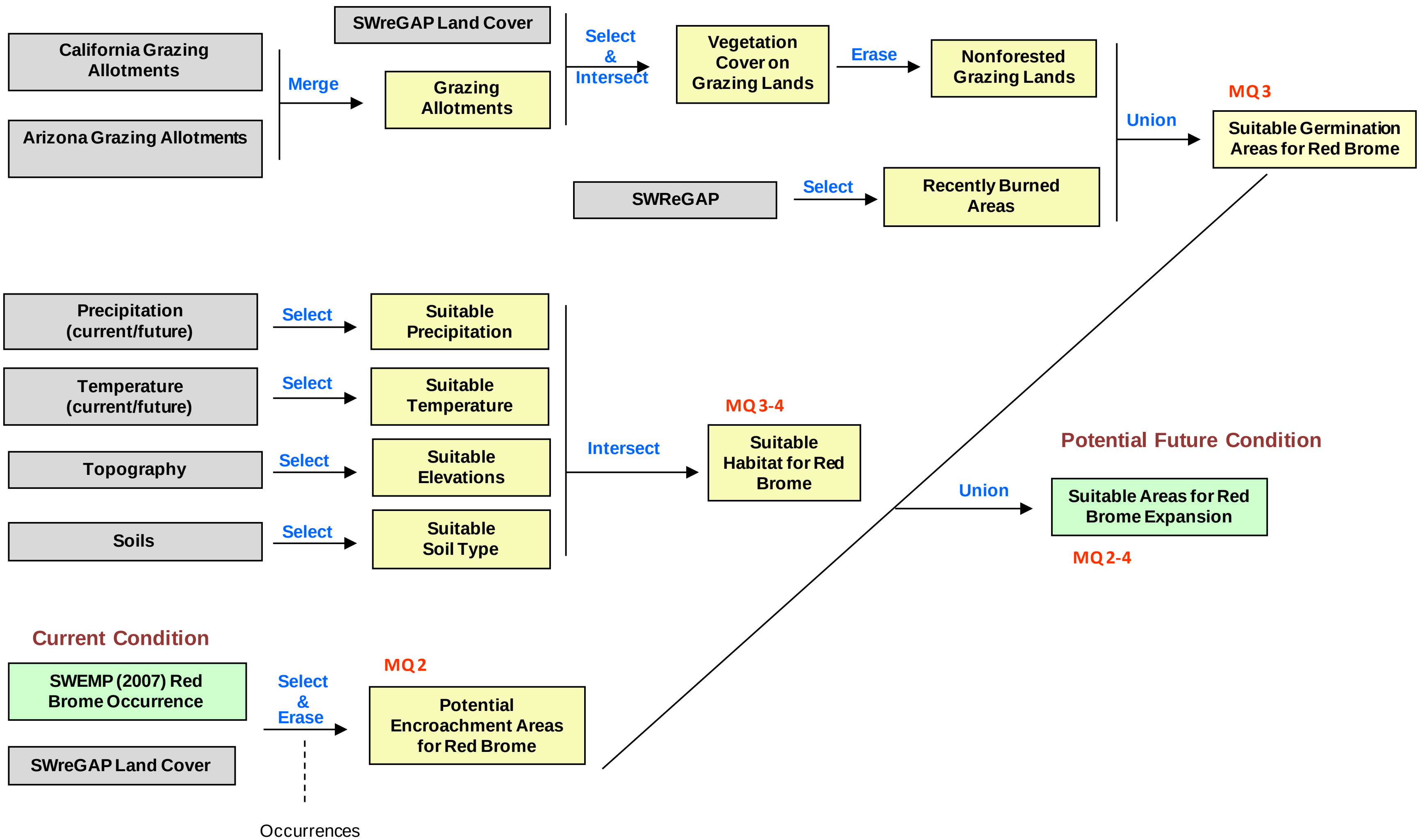


Figure 4-9. *Example* process model for possible red brome habitat, expansion, and restoration areas in the Sonoran Desert ecoregion.

4.2.1.6 References Cited

ASDM (Arizona-Sonora Desert Museum). 2010. Invasive species in the Sonoran Desert region. http://www.desertmuseum.org/invaders/vol_handbook/Invasive_Species_in_the_Sonoran_Desert Region.pdf

Beatley, J.C. 1966. Ecological status of introduced brome grasses (*Bromus* spp.) in desert vegetation of southern Nevada. *Ecology* 47(4):548–554.

Bowers, M.A. 1987. Precipitation and the relative abundances of the desert winter annuals: a 6-year study in the northern Mohave Desert. *Journal of Arid Environments* 12:141–149.

Brooks, M.L. 1999. Alien annual grasses and fire in the Mojave Desert. *Madroño* 46:13–19.

Brooks, M.L. 2003. Effects of increased soil nitrogen on the dominance of alien annual plants in the Mojave Desert. *Journal of Applied Ecology* 40:344–353.

Brooks, M.L. 2008. Plant invasions and fire regimes. Chapter 3 *in* Zouhar, K., J.K. Smith, S. Sutherland, and M.L. Brooks. 2008. Wildland fire in ecosystems: fire and nonnative invasive plants. General Technical Report RMRS-GTR-42-vol. 6. U.S. Forest Service, Rocky Mountain Research Station, Ogden, Utah. 355 p.

Brooks, M. L., and T.C. Esque. 2002. Alien plants and fire in desert tortoise (*Gopherus agassizii*) habitat of the Mojave and Colorado deserts. *Chelonian Conservation and Biology*. 4(2): 330–340.

Burgess, T.L., J.E. Bowers, and R.M. Turner. 1991. Exotic plants at the desert laboratory, Tucson, Arizona. *Madroño* 38:96–114.

D'Antonio C.M., and P.M. Vitousek. 1992. Biological invasions by exotic grasses, the grass/fire cycle, and global change. *Annual Review of Ecology and Systematics* 23:63–87.

DeFalco, L.A., D.R. Bryla, V. Smith-Longozo, and R.S. Nowak. 2003. Are Mojave Desert annual species equal? Resource acquisition and allocation for the invasive grass *Bromus madritensis* subsp. *rubens* (Poaceae) and two native species. *American Journal of Botany* 90:1045–1053.

DeFalco, L.A., G.C.J. Fernandez, and R.S. Nowak. 2007. Variation in the establishment of a non-native annual grass influences competitive interactions with Mojave Desert perennials. *Biological Invasions* 9:293–307.

Drezner, T.D., P.L. Fall, and J.C. Stromberg. 2001. Plant distribution and dispersal mechanisms at the Hassayampa River Preserve, Arizona, USA. *Global Ecology and Biogeography* 10(2):205–217.

Dukes, J.S., and H.A. Mooney. 2004. Disruption on ecosystem processes in western North America by invasive species *in* Bradshaw, G.A., P.A. Marquet, and H.A. Mooney (eds.), How landscapes change: Human disturbance and ecosystem disruptions in the Americas, Springer-Verlag, New York.

Esque, T.C. and C.R. Schwalbe. 2002. Alien annual grasses and their responses to fire and biotic change in Sonoran desertscrub. Pages 165–194 *in* Tellman, B.J., (ed.), Invasive exotic species in the Sonoran region. University of Arizona Press, Tucson, AZ.

Esque, T. C., C. R. Schwalbe, J. A. Lissow, D. F. Haines, D. Foster, and M. Garnett. 2006. Buffelgrass fuel loads in Saguaro National Park, Arizona, increase fire danger and threaten native species. *Park Science* 24: 33–37.

Hammouda, M. and Z. Bakr. 1969. Some aspects of germination of desert seeds. *Phyton* 13(3–4): 183–201.

Hernandez, L., R. Nowak, and L. Salto. 2010. Research poster: physiological responses of two invasive annual grasses, cheatgrass and red brome, in the Great Basin. 2010 Annual Nevada NSF EPSCoR Climate Change Conference, paper 11. http://digitalcommons.library.unlv.edu/epscor/2010/feb02/11

Hulbert, L. 1955. Ecological studies of *Bromus tectorum* and other annual bromegrasses. *Ecological Monographs* 25(2):181–213.

Hunter, R. 1991. *Bromus* invasions on the Nevada Test Site: Present status of *B. rubens* and *B. tectorum* with notes on their relationship to disturbance and altitude. *Great Basin Naturalist* 51:176–182.

Huxman, T.E., E. P. Hamerlynck, and S.D. Smith. 1999. Reproductive allocation and seed production in *Bromus madritensis* ssp. *rubens* at elevated atmospheric CO_2. *Functional Ecology* 13:769–777.

Huxman, T.E., E.P. Hamerlynck, D.N. Jordan, K.J. Salsman, and S.D. Smith. 1998. The effects of parental CO_2 environment on seed quality and subsequent seedling performance in *Bromus rubens*. *Oecologia* 114:202–208.

Kay, B. 1971. Atrazine and simazine increase yield and quality of range forage. *Weed Science* 19(4): 370–371.

McClaran, M.P., and W.W. Brady. 1994. Arizona's diverse vegetation and contributions to plant ecology. *Rangelands* 16(5):208–217.

Newman, D. 2001. Element stewardship abstract for *Bromus rubens*. The Nature Conservancy, Arlington,VA.

Reid, C.R., S. Goodrich, and J.E. Bowns. 2008. Cheatgrass and red brome: the history and biology of two invaders. Pages 27–32 *in* Kitchen, S.G., R.L. Pendleton, T.A. Monaco, and J. Vernon (compilers). Proceedings-Shrublands under fire: disturbance and recovery in a changing world, June 6–8, 2006, Cedar City, UT. Proceedings RMRS-P-52, U.S. Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado.

Rogers, G.F. 1985. Mortality of burned *Cereus giganteus*. *Ecology* 66(2):630–632.

Salo, L.F. 2004. Population dynamics of red brome (*Bromus madritensis* subsp. *rubens*): Times for concern, opportunities for management. *Journal of Arid Environments* 57:291–296.

Salo, L.F. 2005. Red brome (*Bromus rubens* subsp. *madritensis*) in North America: Possible modes for early introductions, subsequent spread. *Biological Invasions* 7:165–180.

Salo, L.F., G.R. McPherson, and D.G. Williams. 2005. Sonoran desert winter annuals affected by density of red brome and soil nitrogen. *American Midland Naturalist* 153(1):95–109.

Sampson, A., A. Chase, and D. Hedrick. 1951. California grasslands and range forage grasses. California Agricultural Experiment Station Bulletin 724:3–130.

Simonin, K.A. 2001. *Bromus rubens, Bromus madritensis in* U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. Available: http://www.fs.fed.us/database/feis/ [Accessed 2010, December 29].

Smith, S.D., T.E. Huxman, S.F. Zitzer, T.N. Charlet, D.C. Housman, J.S. Coleman, L.K. Fenstermaker, J.R. Seemann, and R.S. Nowak. 2000. Elevated CO_2 increases productivity and invasive species success in an arid ecosystem. *Nature* 408:79–82.

Turner, R. M., and D. E. Brown. 1982. Sonoran desertscrub *in* Brown, D.E. (ed.), Biotic communities of the American Southwest: United States and Mexico. *Desert Plants* 4: 181–221.

Wu, K. and K. Jain. 1978. Genetic and plastic responses in geographic differentiation of *Bromus rubens* populations. *Canadian Journal of Botany* 56:873–879.

Wu, K. and S. Jain. 1979. Population regulation in *Bromus rubens* and *Bromus mollis*: Life cycle components and competition. *Oecologia* 39(3):337–357.

4.2.2 Tamarisk (*Tamarisk* spp.)

Note: We present a standard approach to species modeling in these three invasive species sections. However, we are evaluating existing invasive species models and may use one if it is readily available, covers the entire ecoregion, or may be easily adapted to the full ecoregional extent. Several potential tamarisk distribution models have been developed. We are considering three tamarisk models that have westwide U.S. coverage: Morisette et al. 2006, Cord et al. 2010, and Friedman et al (2005).

4.2.2.1 Background

Tamarisk (or salt cedar) is an invasive shrub that has been designated as a change agent in the Sonoran Desert REA because it negatively affects surface and groundwater aquatic resources, aquatic sites of conservation concern, and native riparian systems. The term *tamarisk* refers to a number of related species in the genus *Tamarix* (e.g., *T. chinensis*, *T. gallica*, and *T. ramosissima*) that are similar in appearance and that hybridize freely (Gaskin and Shafroth 2005). Tamarisk may have been introduced into North America by the Spaniards, but it did not became widely distributed until the 1800s, when it was planted as an ornamental plant, for windbreaks, and for shade; it is now found throughout nearly all western and southwestern states (Lovich 2000). Tamarisk is a concern because its dense and rapid growth allows it to out-compete native plant species. In addition, it is extremely drought resistant, has high fecundity, produces salts that inhibit the germination and growth of native species, alters fire regimes, and uses large amounts of water (California State Parks 2005). Tamarisk impacts native wildlife by changing the composition of forage plants and the structure of native riparian systems and by desiccating surface water sources.

Tamarisk tolerates a range of soil types, but it is most commonly found in alkaline and saline soils that are seasonally saturated at the surface (Brotherson and Field 1987). A mature tamarisk can produce hundreds of thousands of seeds that are easily dispersed by wind and water (Sudbrock 1993). Seeds can germinate while floating on water, and seedlings may grow up to a foot per month in early spring (Sudbrock 1993). Tamarisks occur mostly in low-lying areas: riparian habitats, washes, and playas are most threatened by tamarisk invasion. Tamarisk can completely replace a more diverse native riparian plant assemblage (what once were common riparian trees and shrubs such as Fremont cottonwood [*Populus fremontii*], Goodding willow [*Salix gooddingii*], and narrowleaf willow [*Salix exigua*]).

4.2.2.2 Conceptual Models

The conceptual models (Figure 4-10 and Figure 4-11) below illustrate the causal relationships between tamarisk invasion and expansion, and its effects on physical and biological systems. Although this species was originally introduced intentionally, it is an opportunistic invader that spreads easily into suitable habitat by means of copious seed production. Tamarisk's opportunism is implicated as one of the main reasons for widespread conversion of southwest riparian systems to a nonnative monoculture (Figure 4-10). In addition, man-made modifications to the natural environment provide habitat for colonization and enhance the spread of this species, as do associated anthropogenic changes to flow regimes. Tamarisk perpetuates its own survival in a number of ways, including altering fire regimes, soil salinity, and geomorphology and lowering groundwater levels. Under all of these changed conditions, tamarisk is able to out-compete native species, eventually forming dense, monotypic stands along rivers, lakes, and other waterbodies.

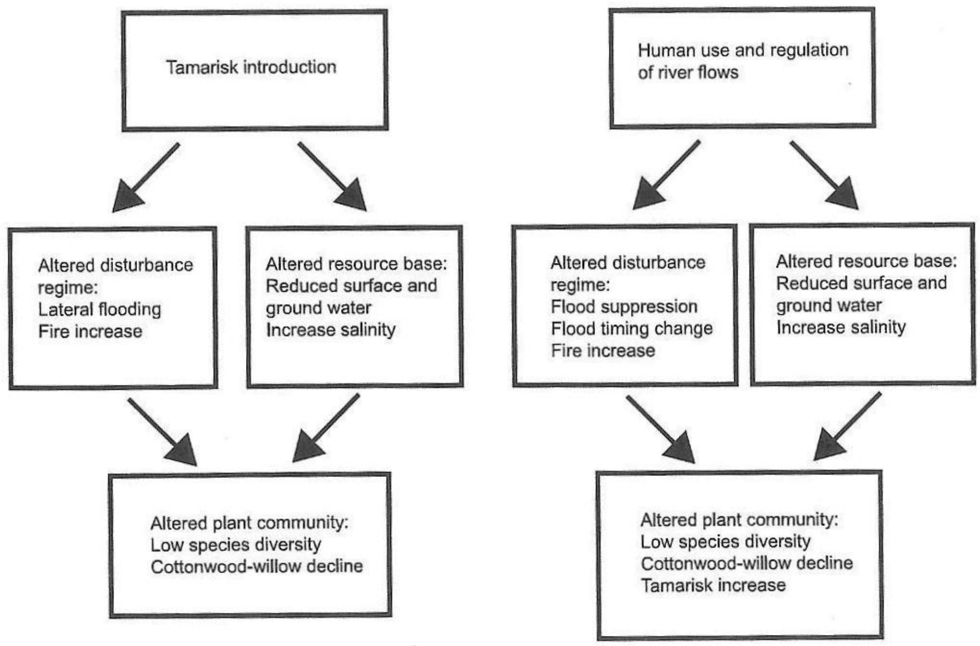


Figure 4-10. Tamarisk invasion and human disturbances as two major influences on the conversion of southwestern riparian systems to nonnative monocultures (Stromberg et al. 2005)

The creation of dams, lakes, and reservoirs has enhanced tamarisk establishment and survival by altering the frequency, timing, and velocity of flows and thus providing new substrates for tamarisk colonization (Shafroth et al. 2002, Zouhar 2003). Additional anthropogenic stressors that facilitate tamarisk establishment include grazing, groundwater pumping, agriculture, and urban development (Figure 4-11 Development and Disturbance, Stromberg 1998, Zouhar 2003), although tamarisk can establish in the absence of disturbance, as well (DiTomaso 1998).

Tamarisk outcompetes and replaces native riparian vegetation through a variety of pathways. For example, it 1) draws down groundwater and it is more tolerant of low groundwater than native species; 2) it reduces seedling recruitment of natives through salt deposition and an increased litter layer; and 3) it increases fire frequency and is more fire-tolerant than native riparian species (Watters 2005, Zouhar 2003). Dense stands of tamarisk result in overbank flooding that alters stream channel structure and sediment deposition (Figure 4-11, Hydrology, Flooding Regime, and Geomorphology, Lovich 2000, Dudley et al. 2000, Cooper et al. 2003). Tamarisk also alters the breakdown of organic materials in desert streams (Kennedy and Hobbie 2004) and creates large deposits of salt above and below the ground that inhibit other plants (Figure 4-11, Soil Ecology, Brotherson and Field 1987, Sudbrock 1993). Tamarisk‘s deep root system enables it to draw down the water table, resulting in drier floodplains and lower flows in streams and rivers (Brotherson and Field 1987). A buildup of leaves and duff under thick riparian growth increases fire frequency in riparian areas dominated by tamarisk (Busch and Smith 1993, Watters 2005, Zouhar 2003).

Tamarisk reduces the value of critical habitat for some wildlife species dependent on specific native riparian habitats (Kennedy et al. 2005, Chen 2001, Johnson et al. 1999, Hunter et al. 1988, Johnson 1986, Cohan et al. 1978), but it does provide habitat value for other species (D'Antonio 2000, Dudley et al. 2000). For example, the southwestern willow flycatcher, a listed endangered species, occurs in southern Utah. Tamarisk thickets have similar structural characteristics to the native vegetation that the birds use in preferred native breeding habitat—located in mesic areas or near surface water and having dense structure, high canopy cover, and tall stature (Sogge et al. 2005). Sogge et al. (2005) found that across the southwestern states approximately 25 percent of southwestern willow flycatcher breeding sites, supporting one-third of the roughly 1,300 known flycatcher territories, are in saltcedar-dominated sites. Brown and Trosset (1989) found that five species nested regularly in tamarisk along the Colorado River in the Grand Canyon; the species with >10 nest sites that they recorded in tamarisk for the Grand Canyon sites were Bell's vireo (*Vireo bellii*), yellow warbler (*Dendroica petechia*), yellowthroat (*Geothlypis trichas*), yellow-breasted chat (*Icteria virens*), and Bullock's oriole (*Icterus bullockii*). Ellis (1995) found that species use of the two habitats had a seasonal element, with similarity between native vegetation and tamarisk use greatest in the fall. On the other hand, Ellis (1995) found at Bosque del Apache National Wildlife Refuge, New Mexico, that some species were never found in tamarisk and preferred cottonwood groves in all seasons (e.g., summer tanager [*Piranga rubra*], bark gleaners [white-breasted nuthatch {*Sitta carolinensis}*, Northern flicker {*Colaptes auratus}*, and other woodpeckers], and cavity nesters).

Instream habitats and species are affected by tamarisk as well. Tamarisk removal at a spring in Ash Meadows National Wildlife Refuge in Nevada resulted in an increased density of Ash Meadows pupfish, because the shade produced by the dense tamarisk thickets reduced the algae necessary to sustain the pupfish (Kennedy et al. 2005). In an Arizona perennial stream, Bailey et al. (2001) found a two-fold decrease in aquatic macroinvertebrate richness and a four-fold decrease in total abundance of macroinvertebrates on tamarisk leaf packs vs. native Fremont cottonwood leaf packs.

Tamarisk has a higher drought tolerance than many native riparian species (Glenn and Nagler 2005). Climate change models predict that rising temperatures are unlikely to adversely affect tamarisk distribution, with the majority of habitat remaining suitable and only a small percentage of currently invaded lands becoming climatically unsuitable by 2100 (Bradley et al 2009). The effects of climate change, such as warming temperatures and increased fire frequency and intensity, are hypothesized to enhance tamarisk invasion and expansion, while limiting native riparian plant communities and their dependent species (Figure 4-11, Altered Fire Regime, Climate Change).

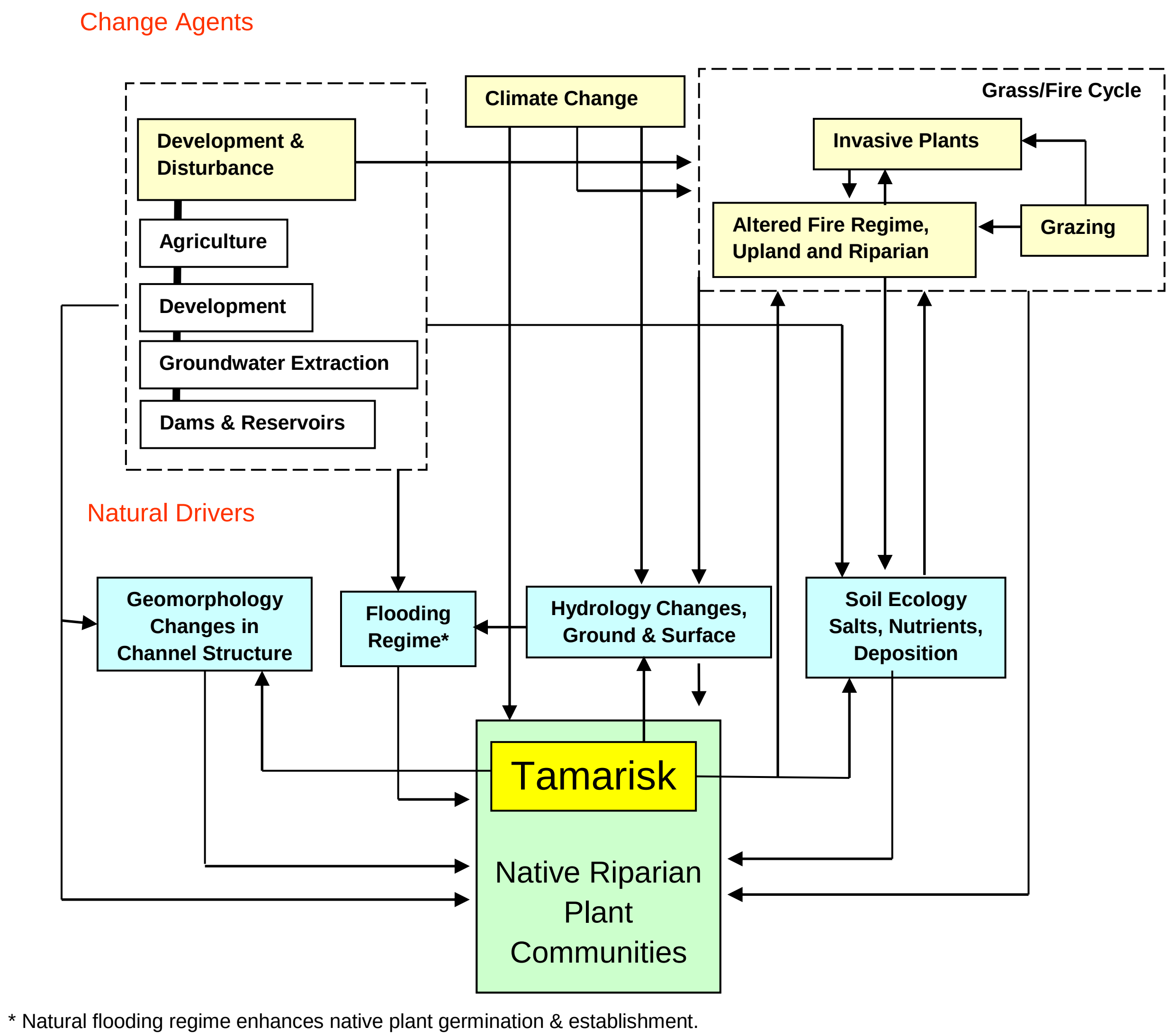


Figure 4-11. Conceptual model for tamarisk in the Sonoran Desert ecoregion.

4.2.2.3 Required Input Data Layers

We compiled a list of datasets we think are most relevant for answering the management questions (Tables 4-6 and 4-7). Dataests are divided into raw (unprocessed datasets; Table 4-6) and previously processed datasets (Table 4-7).

Existing data are available that address management questions (see previously processed data, below). Refer to Cord et al. 2010 and Morisette et al. 2006 for modeling tamarisk distribution and habitat suitability.

Raw Data

Table 4-6. Raw (unprocessed) datasets proposed for use in evaluating key management questions related to tamarisk.

Dataset	Source	Location	MQs
Occurrence Records	Tamarix Cooperative Mapping Initiative Occurrence Data 2009-2010	http://www.tamariskmap.org/	1-4
Occurrence Records	National Institute of Invasive Species Science database (NIISS)	http://www.niiss.org/cwis438/Browse/Organism/OrganismInfo_List.php?WebSiteID=1 and http://www.niiss.org/cwis438/Browse/TiledMap/Scene_Basic.php?WebSiteID=1	1-4
Soils	U.S. Department of Agriculture, NRCS STATSGO or SSURGO	http://soildatamart.nrcs.usda.gov	1-3
Hydrology	USGS Hydrology NHD	http://waterdata.usgs.gov/nwis	1-3
Topography	USGS	http://edc2.usgs.gov/geodata/index.php	1-3
Vegetation	SWReGAP or LANDFIRE	http://earth.gis.usu.edu/swgap/landcover.html and http://www.landfire.gov/NationalProductDescriptions23.php	1-4
Fire History	Recently Burned data from SWReGAP or LANDFIRE	http://earth.gis.usu.edu/swgap/landcover.html and http://www.landfire.gov/	1-3

Previously Processed Data

The sources in Table 4-7 have synthesized tamarisk distributional and ecological attributes, and produced datasets that may be adequate for addressing the management questions.

Table 4-7. Previously processed datasets proposed for use in evaluating key management questions related to tamarisk.

Dataset	Source	Location	MQs
Occurrence Records	National Institute of Invasive Species Science database (NIISS)	http://www.niiss.org/cwis438/Browse/Organism/OrganismInfo_List.php?WebSiteID=1 and http://www.niiss.org/cwis438/Browse/TiledMap/Scene_Basic.php?WebSiteID=1	1-4
Habitat Suitability	NASA Habitat Suitability for Tamarisk Invasion (using MODIS)	http://www.nasaimages.org/luna/servlet/detail/NSVS~3~3~7107~107107:National-Map-Showing-Habitat-Suitab	2-4
Tamarisk Mapping	USGS Mapping Invasive Tamarisk (Landsat EMT+, Maximum Entropy model "Maxent")	http://www.fort.usgs.gov/products/publications/pub_abstract.asp?PubID=22697	2-4
Precipitation	Oak Ridge National Laboratory Distributed Active Archive Center	http://daac.ornl.gov/ and http://app.databasin.org/app/pages/datasetPage.jsp?id=a6127300bf904831b7d647f7d966e87c	1-4
Temperature	Oak Ridge National Laboratory Distributed Active Archive Center	http://daac.ornl.gov/ and http://app.databasin.org/app/pages/datasetPage.jsp?id=a6127300bf904831b7d647f7d966e87c	1-4

4.2.2.4 Attributes, Indicators, and Metrics

Ecological attributes are traits or factors that are necessary to maintaining a fully functioning species population, assemblage, community, or ecosystem. On a species level, they are traits that are necessary for species survival and long-term viability. Indicators are measurable aspects of ecological attributes. In the REAs, attributes and indicators are key elements used to answer management questions, parameterize models, and help explain the expected range in status and condition of individual conservation elements. We propose using attributes and indicators related to elevation, temperature, precipitation, and soils as a potential collection of optimal environmental response in the modeling for Sonoran Desert tamarisk.

4.2.2.5 Model Assumptions

There are several assumptions on which the tamarisk models are based. These include 1) tamarisk outcompetes native species; 2) tamarisk prefers alkaline and saline saturated soils, and is generally found in low-lying areas such as floodplains, watercourses, and lake margins; 3) tamarisk alters fire regimes and soil chemistry, thus perpetuating its own persistence; 4) tamarisk is an opportunistic invader that is expected to thrive with continued or new anthropogenic disturbance; and 5) tamarisk is a drought-tolerant species that is expected to maintain or even expand its current distribution under climate change scenarios.

4.2.2.6 Methods and Tools

The general approach for answering the management questions involves analyzing existing datasets using standard analytical tools in a Geographic Information System (GIS). We plan to use ESRI's ArcMap and ArcInfo to conduct the process/application model. Outputs from the process/application models (Figure 4-12) identify specific management questions by referencing their corresponding number in this REA module (see "Management Questions" section, above).

This process/analysis utilizes standard ArcGIS software tools. Using a combination of Intersect, Select, Merge, Dissolve, Export, etc. tools, we will utilize existing datasets to analyze and create new datasets that identify areas of primary REA concern for tamarisk. Output datasets will be displayed at the 5th field Hydrologic Unit Code (HUC), where appropriate. In general, existing and output datasets address 1) occurrences and 2) physical attributes that contribute to invasion success. One *example* process/application model is provided, below (Figure 4-12). Raw datasets are represented by gray boxes. Previously-processed datasets are represented by yellow boxes. Green boxes represent datasets that answer specific management questions (indicated by the question number). Lines and arrows indicate the process steps taken in the GIS to arrive at specific answers. Red lettered text generally indicates the management question addressed by the particular analysis. Although we have not provided all combinations for our proposed methodological approach, we have provided an example (Figure 4-12).

The example model (Figure 4-12) addresses current and future distribution questions (MQ 1 and 2). An example of the proposed method/analysis flow we could take to answer management question one (Q1 in Figure 4-12) would be to compare tamarisk distributions from different sources (T-map, SWEMP 2007 dataset). If there were no unique records (i.e., records that didn't overlap), we'd simply use one of the datasets and intersect it with the 5th field HUCs file. The resulting dataset would indicate the current tamarisk distribution by HUC5s. If there were unique records, we would "merge" the various files and proceed as noted above.

This model (Figure 4-12) also addresses areas of potential tamarisk invasion (MQ 2 in Figure 4-12). Mapping potential future expansion of this species will require assessing suitable habitat based on hydrology, topography, soils, and native shrublands data. Potential invasion includes information on where the species currently exists on both altered and unaltered stream flow sites. Invasion on altered stream flow sites is much higher than on unaltered sites. Outputs for the two rates combined with overall suitable habitat yields a potential invasion map.

Identifying areas of suitable biophysical setting (e.g., precipitation/topography/soils) with restoration potential (MQ 3 in Figure 4-12) will require mapping areas of existing, occupied habitat that could be considered for future management intervention through site manipulation and replanting based on site characteristics and precipitation amounts. Alternatively, sites with restoration potential (MQ 3 in Figure 4-12) may include areas where future habitats may become unsuitable (compared to current conditions) due to changes in temperature, precipitation and/or soil patterns.

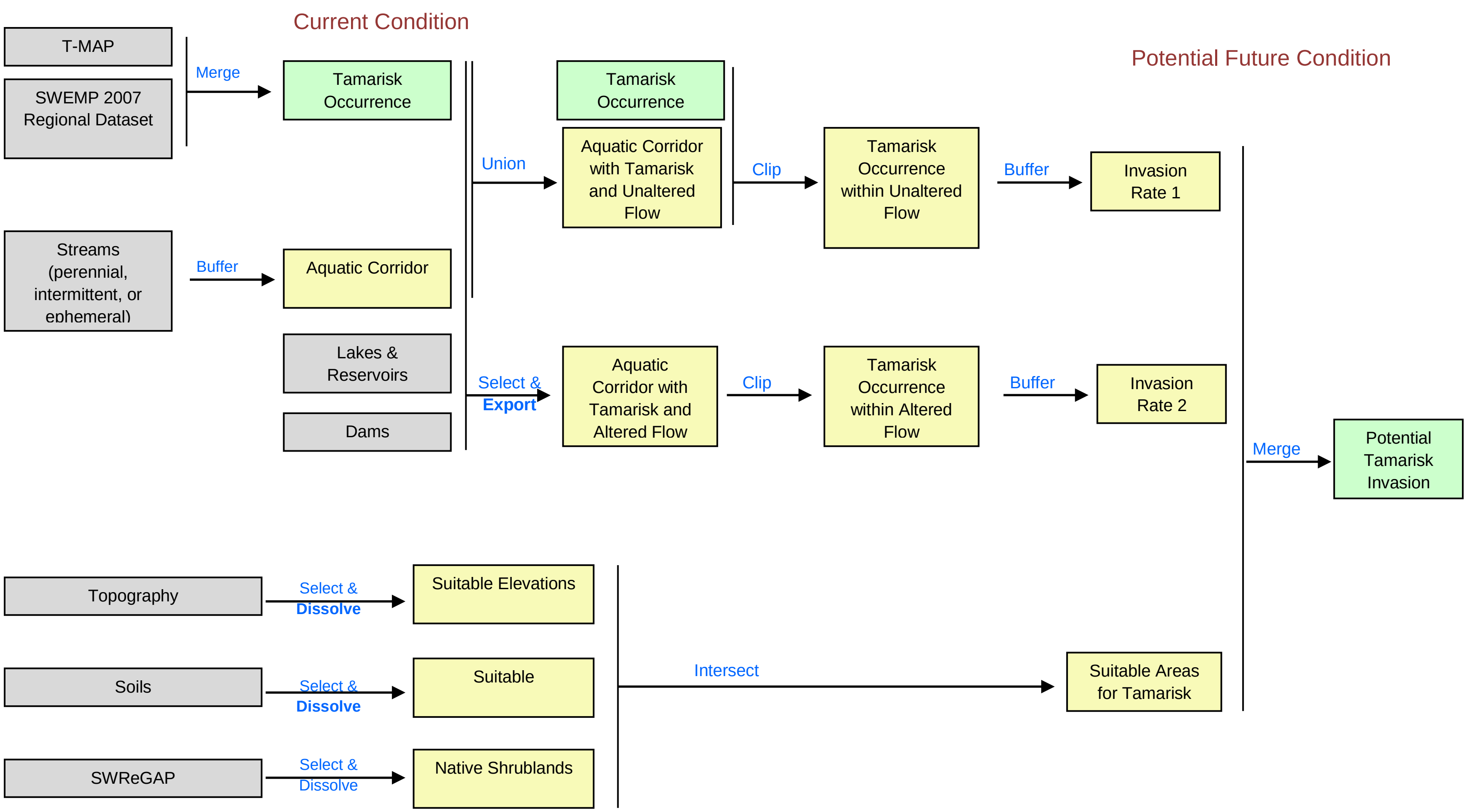


Figure 4-12. Application model for current and future conditions of tamarisk in the Sonoran Desert ecoregion.

4.2.2.7 References Cited

Bradley, B.A., M. Oppenheimer, and D.S. Wilcove. 2009. Climate change and plant invasion: restoration opportunities ahead? *Global Change Biology* 15:1511–1521.

Brotherson, J.D. and D. Field. 1987. *Tamarix*: impacts of a successful weed. *Rangelands* 9(3):110–112.

Busch, D.E. and S.D. Smith. 1993. Effects of fire on water salinity relations of riparian woody taxa. *Oecologia* 94:186–194.

California State Parks. 2005. Anza-Borrego Desert State Park: final general plan and environmental impact report (EIR). SCH # 2002021060. California State Parks, Sacramento, California.

Chen, L.Y. 2001. Cost savings from properly managing endangered species habitats. *Natural Areas Journal* 21(2):197-203.

Cohan, D.R., B.W Anderson, and R.D. Ohmart. 1978. Avian population responses to salt cedar along the lower Colorado River. Pages 371–381 *in* Johnson, R.R and J.F. McCormick (technical coordinators). Strategies for protection and management of floodplain wetlands and other riparian ecosystems: Proceedings of the symposium; December 11–13, Callaway Gardens, GA. General Technical Report WO-12. U.S. Forest Service, Washington, D.C.

Cooper, D.J, D.C. Anderson, and R.A. Chimner. 2003. Multiple pathways for woody plant establishment on floodplains at local to regional scales. *Journal of Ecology* 91:182–196.

Cord, A., D. Klein, and S. Dech. 2010. Remote sensing time series for modeling invasive species distribution: a case study of *Tamarix* spp. in the U.S. and Mexico. Swayne, D.A., W. Yang, A.A. Voinov, A. Rizzoli, and T. Filatova (eds.). International Environmental Modeling and Software Society (iEMSs), 2010 International Congress on Environmental Modeling and Software, Modeling for Environment's Sake, Fifth Biennial Meeting, Ottawa, Canada.

D'Antonio, C.M. 2000. Fire, plant invasions, and global changes. Pages 65–93 *in* Mooney, H.A. and R.J. Hobbs (eds.). Invasive species in a changing world. Island Press, Washington, DC.

DiTomaso, J.M. 1998. Impact, biology, and ecology of saltcedar (*Tamarix* spp.) in the southwestern United States. *Weed Technology* 12:326–336.

Dudley, T.L., C.J. DeLoach, J.E. Lovich, and R.I. Carruthers. 2000. Saltcedar invasion of western riparian areas: impacts and new prospects for control. Pages 345–381 *in* New insights and new incites in natural resource management: Transactions, 65th North American wildlife and natural resources conference; March 24-28, Rosemont, IL. Wildlife Management Institute, Washington, D.C.

Friedman, J.M., G.T. Auble, P.B. Shafroth, M.L. Scott, M.F. Merigliano, M.D. Preehling, and E.K. Griffin. 2005. Dominance of non-native riparian trees in western USA. *Biological Invasions* 7: 747–751.

Gaskin, J.F. and P.B. Shafroth. 2005. Hybridization of *Tamarix ramosissima* and *T. chinensis* (saltcedars) with *T. aphylla* (athel) (Tamaricaceae) in the southwestern USA determined from DNA sequence data. *Madroño* 52(1):1–10.

Glenn, E.P. and P.L. Nagler. 2005. Comparative ecophysiology of *Tamarix ramosissima* and native trees in western U.S. riparian zones. *Journal of Arid Environments* 61:419–446.

Hunter, W.C., R.D. Ohmart, and B.W. Anderson. 1988. Use of exotic saltcedar (*Tamarix chinensis*) by birds in arid riparian systems. *The Condor* 90: 113–123.

Johnson, S. 1986. Can salt cedar be controlled? The Nature Conservancy News, Oct.-Nov.:19−20. The Nature Conservancy, Arlington, Virginia.

Johnson, K., P. Mehlhop, C. Black, and K. Score. 1999. Reproductive failure of endangered southwestern willow flycatchers on the Rio Grande, New Mexico. *Southwestern Naturalist* 44:226−231.

Kennedy, T.A. and S.E. Hobbie. 2004. Saltcedar (*Tamarix ramosissima*) invasion alters organic matter dynamics in a desert stream. *Freshwater Biology* 49:65–76.

Kennedy, T.A., J.C. Finlay, and S.E. Hobbie. 2005. Eradication of invasive *Tamarix ramosissima* along a desert stream increases native fish density. *Ecological Applications* 15:2072–2083.

Lovich, J. 2000. *Tamarix ramosissima/Tamarix chinensis/Tamarix gallica/Tamarix parviflora*. Pages 312–317 *in* Bossard, C.C.; J.M. Randall, and M.C. Hoshovsky (eds.), Invasive plants of California's wildlands. University of California Press, Berkeley, California.

Morisette, J.T., C.S. Jarnevich, A. Ullah, W. Cai, J.A. Pedelty, J.E. Gentle, T.J. Stohlgren, and J.L. Schnase. 2006. A tamarisk habitat suitability map for the continental United States. *Frontiers in Ecology and the Environment* 4(1):11–17.

Shafroth, P.B., J.C. Stromberg, and D.T. Patten. 2002. Riparian vegetation response to altered disturbance and stress regimes. *Ecological Applications* 12:107−123.

Stromberg, J.C. 1998. Functional equivalency of saltcedar (*Tamarix chinensis*) and Fremont cottonwood (*Populus fremontii*) along a free-flowing river. *Wetlands* 18(4):675–686.

Sudbrock, A. 1993. Tamarisk control. 1. Fighting back: an overview of the invasion, and a low-impact way of fighting it. *Restoration and Management Notes* 11(1):31–34.

Watters, K. 2005. *Tamarix chinensis*: plant assessment form. AZ-WIPWG, Version 1. 14 pp. August.

Zouhar, K. 2003. *Tamarix* spp. *in* Fire Effects Information System, U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. Available: http://www.fs.fed.us/database/feis

4.2.3 Buffelgrass: (*Pennisetum ciliare*)

4.2.3.1 Introduction

Buffelgrass is a drought-tolerant, warm-season, perennial bunchgrass native to Africa, Asia, and the Middle East. The species is grown for forage in many arid and semi-arid regions of the world, and it was introduced to the United States as livestock forage in the 1930s. Since then it has also been used for erosion control and soil stabilization (SABCC 2010a). Buffelgrass is invasive in North and South America, Australia, and many oceanic islands. It is considered a noxious weed in Arizona (Arizona Department of Agriculture 2006); it is particularly problematic in the Sonoran Desert of southern Arizona and northern Mexico (SABCC 2010b), where it alters fire regimes, soil ecology, hydrology, and geomorphology, thereby threatening native plant communities and associated wildlife species.

The Assessment Management Team (AMT) defined a set of management questions in the Statement of Work (SOW) for this REA that were refined and finalized by the AMT and participants in two workshops. Management questions for this invasive species include:

1. Where are areas dominated by this invasive species?
2. Where are the areas of potential future encroachment from this invasive species?
3. Where are areas with restoration potential?
4. Where/how will the distribution of dominant native plant and invasive species be vulnerable to or have potential to change from climate change in 2060?

The process model described below in Section 4.2.3.6 (Figure 4-14) incorporates the management questions and shows how the management questions will be answered. We will also investigate using existing models for buffelgrass. There is a model by Frid et al. 2010 for buffelgrass in southern Arizona, including a climate change scenario for 2060. We will see if it is adaptable to the full Sonoran ecoregion.

In the Sonoran Desert, buffelgrass occurs primarily in disturbed sites, along roadsides, in desert washes, and on rocky hillsides (Búrquez-Montijo et al. 2002, Rutman and Dickson 2002, Burgess et al. 1991). In Arizona, it has invaded upland desert scrub habitat and it is also considered a threat to desert grassland, chaparral, and oak woodland (Van Devender and Dimmitt 2006).

Temperature and precipitation appear to be limiting factors in buffelgrass distribution. The species thrives where winter temperatures are above 41°F (5°C) and annual rainfall is between 13–22 inches (33–56 cm, Búrquez-Montijo et al. 2002, Williams and Baruch 2000, Ibarra-F. et al. 1995, Cox et al. 1988). The elevational range of this species is generally between about 66 and 2300 feet (20–701 m), although it has been found above 2950 feet (899 m) in Arizona (Ibarra-F. et al. 1995).

Buffelgrass spreads aggressively by seed, but it can also spread vegetatively by rhizomes (Arriaga et al. 2004, Williams and Baruch 2000). This species germinates with relatively low amounts of precipitation (Ward et al. 2006) and establishes readily in disturbed areas (e.g., roads, trails, riparian corridors, mines, landfills). Once established, it can spread into adjacent, undisturbed habitats (Búrquez-Montijo et al. 2002). Dispersal agents include wind, water, animals, and humans (Arriaga et al. 2004). Under natural conditions, seed remains viable in the soil seedbank for only 2 to 4 years (Winkworth 1971).

4.2.3.2 Conceptual Model

The conceptual model (Figure 4-13) illustrates the causal relationships between buffelgrass invasion and expansion and the subsequent effects on physical and biological systems. Although buffelgrass benefits from any activity that disturbs the soil surface, roadsides play an important role in the introduction and expansion of this species in the region by providing corridors for colonization, as well as supplemental water (e.g., runoff) for establishment. Roads also provide a platform for subsequent invasion into adjacent, undisturbed habitat (Brenner 2009, Van Devender and Dimmitt 2006). Buffelgrass displaces native species by reducing or eliminating suitable habitat (e.g., type conversion), consuming resources (e.g., water), and/or altering soil ecology and geomorphology.

Buffelgrass is a fire-tolerant species that has introduced fire into Sonoran desert plant communities that are not pre-adapted to fire (Van Devender and Dimmitt 2006). This species creates large amounts of litter, thus increasing fuel loads and fire frequency and intensity (Figure 4-13), Rutman and Dickson 2002). It also recovers quickly from fire, thereby perpetuating a grass/fire cycle (D'Antonio and Vitousek 1992, Brooks et al. 2004). Adverse effects of this altered fire regime include displacement of native species, alteration of plant community structure and composition, and reductions in plant and animal biodiversity (Van Devender and Dimmitt 2006, Búrquez-Montijo et al. 2002, Williams and Baruch 2000, Marshall et al. 2000, Burgess et al. 1991).

Buffelgrass can occur on a variety of substrates, but it prefers low-fertility, well-drained sandy loam soils (Van Devender and Dimmitt 2006, Ibarra-F. et al. 1995). It does not flourish in soils with a sand content below 40% and it is generally excluded from soils with less than 30% sand and more than 50% clay and 60% silt (Barrera 2008). Buffelgrass alters soil ecology by increasing the amount of litter and reducing or depleting soil nutrients (Búrquez-Montijo et al. 2002, Ibarra-Flores et al. 1999). Buffelgrass also out-competes native species for water (Van Devender and Dimmitt 2006) and indirectly alters geomorphology through increased wind and water erosion following fire (Logan Simpson Design, Inc. 2009).

Because of its drought tolerance and ability to out-compete native species, buffelgrass is expected to expand northward and into higher elevations with warming temperatures associated with climate change (Kurc 2008). An increase in atmospheric CO_2 levels and nitrogen deposition may accelerate invasion and dominance (Williams and Baruch 2000).

Buffelgrass is considered a serious threat to several key desert species including the saguaro cactus, foothill palo verde, and desert tortoise (Southern Arizona Buffelgrass Strategic Plan 2008, Esque et al. 2007, Esque et al. 2004). Additional wildlife threatened directly or indirectly by buffelgrass invasion include lizards, lesser long-nosed bats, cactus ferruginous pygmy-owls, and mule deer (Morales-Romero and Molina-Freaner 2008, Southern Arizona Buffelgrass Strategic Plan 2008, Esque et al. 2007, Flanders et al. 2006, Clarke et al. 2005, Esque et al. 2003, Brooks and Esque 2002, Búrquez-Montijo et al. 2002, Williams and Baruch 2000, Burgess et al. 1991). Natural communities most at-risk in the Sonoran Desert include upland desert scrub, desert grassland, chaparral, and oak woodland.

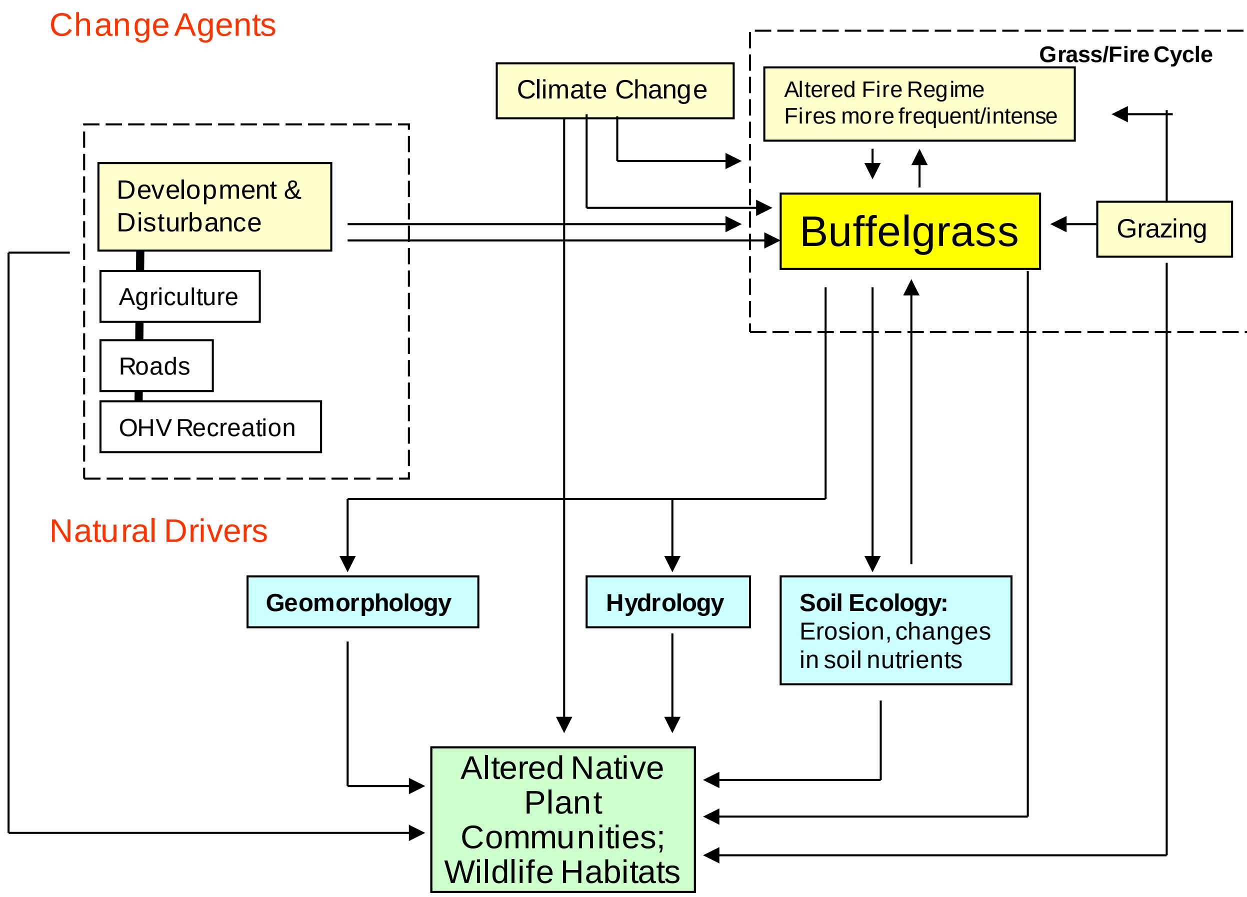


Figure 4-13. Conceptual model for buffelgrass in the Sonoran Desert ecoregion.

4.2.3.3 Required Input Data Layers

We compiled a list of dataset we think are most relevant for answering the management questions (Tables 4-8 and 4-9). Datasets are divided into raw (unprocessed) datasets and previously processed datasets.

Raw Data

Table 4-8. Raw (unprocessed) datasets proposed for use in evaluating key management questions related to buffelgrass.

Dataset	Source	Location	Management Questions (s)
Occurrence Records	Southwest Exotic Mapping Program (SWEMP) 2007 Dataset	http://sbsc.wr.usgs.gov/research/projects/swepic/swemp/swempA.asp	1-4
Occurrence Records	Desert Museum Survey (2004)	http://www.desertmuseum.org/invaders/invaders_buffelgrass.php	1-4
Occurrence Records	National Institute of Invasive Species Science database (NIISS)	http://www.niiss.org/cwis438/Browse/Organism/OrganismInfo_List.php?WebSiteID=1 and http://www.niiss.org/cwis438/Browse	1-4

Dataset	Source	Location	Management Questions (s)
		/TiledMap/Scene_Basic.php?WebSiteID=1	
Soils	U.S. Department of Agriculture, NRCS STATSGO or SSURGO	http://soildatamart.nrcs.usda.gov	1-3
Hydrology	USGS Hydrology NHD	http://waterdata.usgs.gov/nwis	1-3
Topography	USGS	http://edc2.usgs.gov/geodata/index.php	1-3
Vegetation	SWReGAP or LANDFIRE	http://earth.gis.usu.edu/swgap/landcover.html and http://www.landfire.gov/NationalProductDescriptions23.php	1-4
Fire History	Recently Burned data from SWReGAP or LANDFIRE	http://earth.gis.usu.edu/swgap/landcover.html and http://www.landfire.gov/	1-3

In portions of the study area, data may be available that address specific management questions (e.g., Logan Simpson Design, Inc. 2010, Southern Arizona Buffelgrass Strategic Plan 2008, Olsson 2006, Van Devender and Dimmitt 2006).

Previously Processed Data

Table 4-9. Previously processed datasets proposed for use in evaluating key management questions related to buffelgrass.

Dataset	Source	Location	Management Questions (s)
Occurrence Records	Southwest Exotic Mapping Program (SWEMP) 2007 Dataset	http://sbsc.wr.usgs.gov/research/projects/swepic/swemp/swempA.asp	1-4
Occurrence Records	Desert Museum Survey (2004)	http://www.desertmuseum.org/invaders/invaders_buffelgrass.php	1-4
Occurrence Records	National Institute of Invasive Species Science database (NIISS)	http://www.niiss.org/cwis438/Browse/Organism/OrganismInfo_List.php?WebSiteID=1 and http://www.niiss.org/cwis438/Browse/TiledMap/Scene_Basic.php?WebSiteID=1	1-4
Precipitation	Oak Ridge National Laboratory Distributed Active Archive Center	http://daac.ornl.gov/ and http://app.databasin.org/app/pages/datasetPage.jsp?id=a6127300bf904831b7d647f7d966e87c	1-4
Temperature	Oak Ridge National Laboratory Distributed Active Archive Center	http://daac.ornl.gov/ and http://app.databasin.org/app/pages/datasetPage.jsp?id=a6127300bf904831b7d647f7d966e87c	1-4

4.2.3.4 Attributes, Indicators, and Metrics

Ecological attributes are traits or factors that are necessary to maintaining a fully functioning species population, assemblage, community, or ecosystem. On a species level, they are traits that are necessary for species survival and long-term viability. Indicators are measurable aspects of ecological attributes. In the REAs, attributes and indicators are key elements used to answer management questions, parameterize models, and help explain the expected range in status and condition of individual conservation elements. We propose using attributes and indicators related to suitable elevation, temperature, precipitation, and soil type as a potential collection of optimal environmental response in the modeling for Sonoran Desert buffelgrass.

4.2.3.5 Model Assumptions

There are several assumptions on which the buffelgrass models are based. These include 1) roadsides are an important conduit in the establishment and spread of buffelgrass because of direct disturbance of the substrate, but perhaps also because of concentrated water drainage at road edges; 2) buffelgrass outcompetes and displaces native species; 3) buffelgrass increases fuel loads and fire intensity and frequency; 4) buffelgrass perpetuates and benefits from a fire/grass cycle; 5) buffelgrass prefers low-fertility, well-drained sandy loam soils; 6) buffelgrass is drought-tolerant and is expected to expand its current distribution under climate change scenarios.

4.2.3.6 Methods and Tools

The general approach for answering the management questions involves analyzing existing datasets using standard analytical tools in a Geographic Information System (GIS). We plan to use ESRI's ArcMap and ArcInfo to conduct the process/application model. Outputs from the process/application models (Figure 4-14) identify specific management questions by referencing their corresponding number in this REA module (see "Management Questions" section, above).

This process/analysis utilizes standard ArcGIS software tools. Using a combination of Intersect, Select, Merge, Dissolve, Export, etc. tools, we will utilize existing datasets to analyze and create new datasets that identify areas of primary REA concern for buffelgrass. Output datasets will be displayed at the 5th field Hydrologic Unit Code (HUC), where appropriate. In general, existing and output datasets address 1) occurrences and 2) physical attributes that contribute to invasion success. An example process/application model is provided, below (Figure 4-14). Raw datasets are represented by gray boxes. Previously-processed datasets are represented by yellow boxes. Green boxes represent datasets that answer specific management questions (indicated by the question number). Lines and arrows indicate the process steps taken in the GIS to arrive at specific answers. Red lettered text generally indicates the management question addressed by the particular analysis.

The example model (Figure 4-14) addresses current and future distribution questions (MQs 1 and 2). An example of the proposed method/analysis flow we would take to answer management question one (Q1 in Figure 4-12) would be to compare buffelgrass distributions from different sources (Desert Museum, SABCC road survey, SWEMP 2007 dataset). If there were no unique records (i.e., records that didn't overlap), we'd simply use one of the datasets and intersect it with the 5th field HUCs file. The resulting dataset would indicate the current buffelgrass distribution by HUC5s. If there were unique records, we would "merge" the various files and proceed as noted above.

This model (Figure 4-14) also addresses areas of potential buffelgrass invasion (MQ 2). Mapping potential future expansion of this species first requires assessing suitable habitat based on topography and soils. Disturbed sites from roads and burning are extremely important drivers for buffelgrass and modeled using available spatial datasets. Combining these sites makes up lands where buffelgrass expansion is most probable.

Identifying areas of suitable biophysical setting (e.g., precipitation/topography/soils) with restoration potential (MQ 3 in Figure 4-14) will require mapping areas of existing, occupied habitat that could be considered for future management intervention through site manipulation and replanting based on site characteristics and precipitation amounts. Alternatively, sites with restoration potential (MQ 3 in Figure 4-14) may include areas where future habitats may become unsuitable (compared to current conditions) due to changes in temperature, precipitation and/or soil patterns.

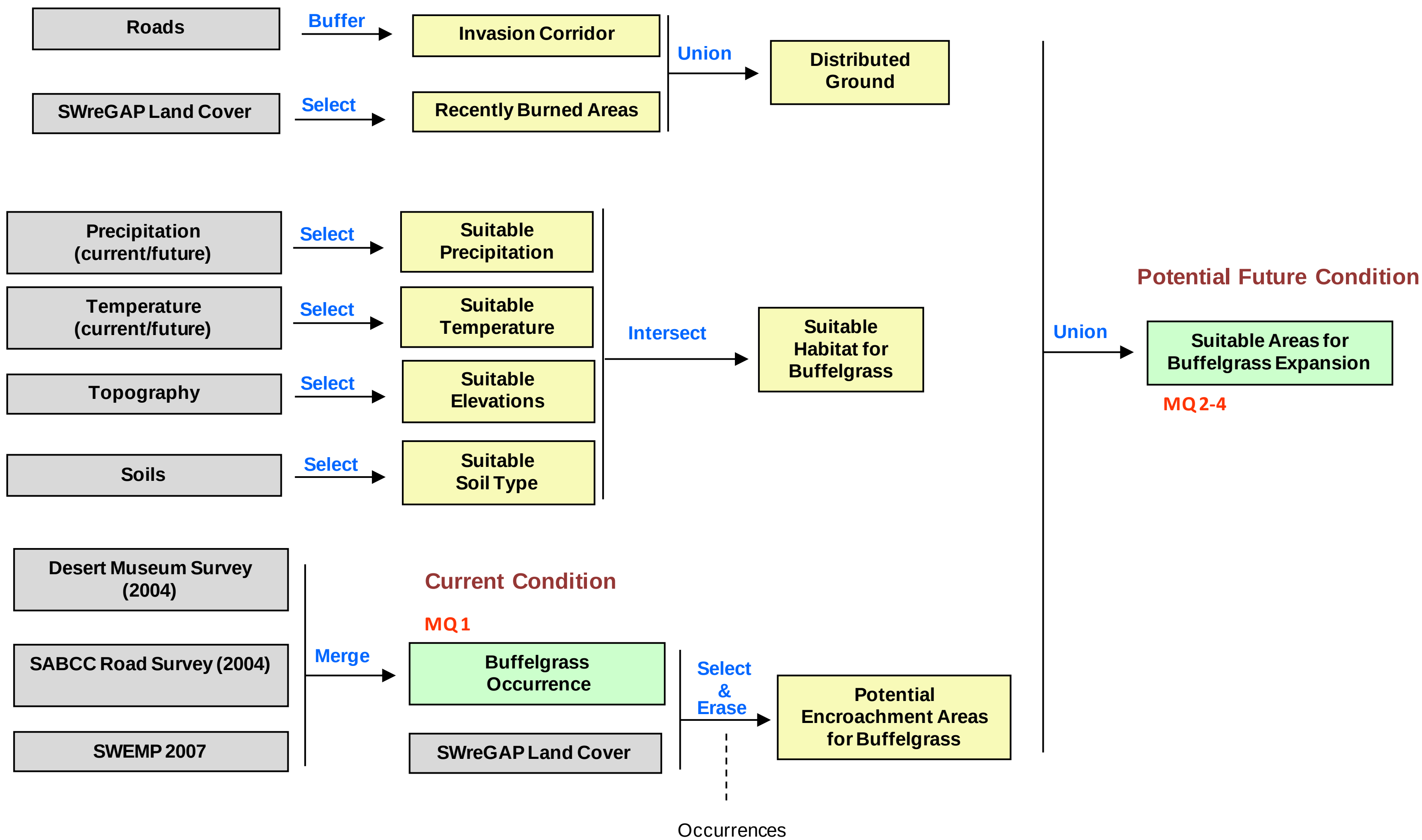


Figure 4-14. Process model for current condition and potential future condition of buffelgrass occurrence.

4.2.3.7 References Cited

Arizona Department of Agriculture . 2006. Prohibited, regulated, and restricted noxious weeds. http://www.azda.gov/psd/quarantine5.htm

Arriaga, L., V. Castellanos, E. Alejandro, E. Moreno, and J. Alarcon. 2004. Potential ecological distribution of alien invasive species and risk assessment: a case study of buffelgrass in arid regions of Mexico. *Conservation Biology* 18(6):1504–1514.

Barrera, E. 2008. Recent invasion of buffel grass (*Cenchrus ciliaris*) of a natural protected area from the southern Sonoran Desert. *Revista Mexicana de Biodiversidad* 79: 385–392,

Brenner, J.C. 2009. Structure, agency, and the transformation of the Sonoran Desert by buffelgrass (*Pennisetum ciliare*): an application of land change science. Ph.D. Dissertation. Clark University Graduate School of Geography. Worcester, Massachusetts.

Brooks, M.L., C.M. D'Antonio, D.M. Richardson, J.B. Grace, J.E. Keeley, J.M. DiTomaso, R.J. Hobbs, M. Pellant, and D. Pyke. 2004. Effects of invasive alien plants on fire regimes. *BioScience* 54(7):677–688.

Brooks, M.L., and T.C. Esque. 2002. Alien plants and fire in desert tortoise (*Gopherus agassizii*) habitat of the Mojave and Colorado deserts. *Chelonian Conservation and Biology* 4:330–340.

Burgess, T.L., J.C. Bowers, and R.M. Turner. 1991. Exotic plants at the Desert Laboratory, Tucson, Arizona. *Madroño* 38(2):96–114.

Búrquez-Montijo A., M.E. Miller, and A. Martinez-Yrizar. 2002. Mexican grasslands, thornscrub, and the transformation of the Sonoran Desert by invasive, exotic buffelgrass (*Pennisetum ciliare*). Pages 26–146 *in* B. Tellman, (ed.). Invasive exotic species in the Sonoran region, Arizona-Sonora Desert Museum and University of Arizona Press, Tucson, Arizona.

Clarke, P.J., P.K. Latz, and D.E. Albrecht. 2005. Long-term changes in semi-arid vegetation: invasion of an exotic perennial grass has larger effects than rainfall variability. *Journal of Vegetation Science* 16:237–248.

Cox J.R., M.H. Martin-R, F.A. Ibarra-F, J.H. Fourie, N.F.G. 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.

D'Antonio, C.M., and P.M. Vitousek. 1992. Biological invasions by exotic grasses, the grass/fire cycle, and global change. *Annual Review of Ecology and Systematics* 23:63–87.

Esque, T.C., C.R. Schwalbe, D.F. Haines, and W.L. Halvorson. 2004. Saguaros under siege: invasive species and fire. *Desert Plants* 20:49–55.

Esque, T.C., C.R. Schwalbe, L.A. DeFalco, T.J. Hughes, and R.B. Duncan. 2003. Effects of wildfire on small desert vertebrates, especially desert tortoises (*Gopherus agassizii*). *The Southwestern Naturalist* 48:103–110.

Esque, T.C., C.R. Schwalbe, J.A. Lissow, D.F. Haines, D.Foster, and M.C. Garnett. 2007. Buffelgrass fuel loads in Saguaro National Park, Arizona, increase fire danger and threaten native species. *Park Science* 24(2):1–11.

Flanders, A.A., W.P. Kuvlesky, Jr., D.C. Ruthven, III, R.E. Zaiglin, R.L. Bingham, T.E. Fulbright, F. Hernández, and L.A. Brennan. 2006. Effects of invasive exotic grasses on South Texas rangeland breeding birds. *The Auk* 123:171–182.

Frid, L., K. Bryan, A. Hall, T. Holcombe and A. Olsson. 2010. Developing a decision support model for buffelgrass management in Southern Arizona. Report prepared by ESSA Technologies Ltd., Vancouver, British Columbia. 33 pp. + appendices. http://www.essa.com/documents/Buffelgrass-Report.pdf.

Ibarra-F., F.A., J.R. Cox, M.H. Martin-R., T.A. Crowl, and C.A. Call. 1995. Predicting buffelgrass survival across a geographical and environmental gradient. *Journal of Range Management* 48(1):53–59.

Ibarra-Flores, F., J.R. Cox, M. Martin-Rivera, T.A. Crowl, B.E. Norton, R.E. Banner, and R.W. Miller. 1999. Soil physicochemical changes following buffelgrass establishment in Mexico. *Arid Soil Research and Rehabilitation* 13:39–52.

Kurc, S. 2008. Extreme makeovers: crossing critical thresholds into desertification. *Aridlands Newsletter* 60:4–11.

Logan Simpson Design, Inc. 2010. Tucson Basin buffelgrass assessment. Prepared for Southern Arizona Buffelgrass Coordination Center.

Marshall, R.M., S. Anderson, M. Batcher, P. Comer, S. Cornelius, R. Cox, A. Gondor, D. Gori, J. Humke, R. Paredes Aguilar, I.E. Parra, and S. Schwartz. 2000. An ecological analysis of conservation priorities in the Sonoran Desert ecoregion. Prepared by The Nature Conservancy Arizona Chapter, Sonoran Institute, and Instituto del Medio Ambiente y el Desarrollo Sustentable del Estado de Sonora with support from Department of Defense Legacy Program, Agency and Institutional Partners. 146 pp.

Morales-Romero, D., and F. Molina-Freaner. 2008. Influence of buffelgrass pasture conversion on the regeneration and reproduction of the columnar cactus, *Pachycereus pectin-aboriginum*, in northwestern Mexico. *Journal of Arid Environments* 72:228–237.

Olsson, A. 2006. Modeling the potential distribution of buffelgrass in southern Arizona: a comparison of logistic regression and artificial neural networks. Unpublished M.S. thesis, University of Arizona, Tucson.

Rutman, S., and L. Dickson. 2002. Management of buffelgrass on Organ Pipe Cactus National Monument, Arizona. Pages 311–318 *in* B. Tellman, (ed.), Invasive exotic species in the Sonoran region, Arizona-Sonora Desert Museum and University of Arizona Press, Tucson, Arizona.

SABCC (Southern Arizona Buffelgrass Coordination Center). 2010a. History and distribution. http://www.buffelgrass.org/node/11. [Accessed December 17, 2010].

SABCC (Southern Arizona Buffelgrass Coordination Center). 2010b. Why is it a threat? http://www.buffelgrass.org/node/13. [Accessed December17, 2010].

Southern Arizona Buffelgrass Strategic Plan. 2008. Prepared by Buffelgrass Working Group. http://www.desertmuseum.org/invaders/splan/buffelgrass_strategic_plan.pdf

Van Devender, T.R., and M.A. Dimmitt. 2006. Final report on conservation of Arizona upland Sonoran desert habitat: Status and threats of buffelgrass (*Pennisetum ciliare*) in Arizona and Sonora. Project #2004-0013-003). Arizona-Sonora Desert Museum, Tucson, AZ.

Ward, J.P., S.E. Smith, and M.P. McClaran. 2006. Water requirements for emergence of buffelgrass (*Pennisetum ciliare*). *Weed Science* 54(4):720–725.

Williams, D.G., and Z. Baruch. 2000. African grass invasion in the Americas: ecosystem consequences and the role of ecophysiology. *Biological Invasions* 2:123–140.

Winkworth, R.E. 1971. Longevity of buffel grass seed sown in an arid Australian range. *Journal of Range Management* 24(2):141–145.

4.3 Change Agent: Climate Change

4.3.1 Choosing Climate Models and Scenarios

When examining potential impacts of climate change, land managers are searching for reliable future climate projections and likely emission scenarios. The various IPCC reports (http://www.ipcc.ch/) have been the repository of the state-of-the-art information on climate modeling and climate impacts projections. In the last IPCC report (AR5), climate projections from 23 climate models (from 17 modeling teams) were provided for 8 emission scenarios. For the BLM REA assessments, Steve Hostetler's (USGS) dynamical downscaled climate data (15km) was selected to support the REA effort for the following reasons: (1) dynamic downscaling is more appropriate for fine scale assessments than statistical downscaling of GCM results; (2) up to 60 climate variables are provided, far more than the number that GCM groups have been archiving for impacts assessments; (3) IPCC AR5 projections were used; (4) the data are provided by DOI's science agency and are also being used by some LCCs; and (5) it will be provided in ARCGRID format. Three AR4 emission scenarios were used to run the Regional Climate Model (RCM) including B1, A2, and A1B (Table 4-10).

Table 4-10. Emission scenarios used for AR4 (IPCC 2007) with information extracted and simplified from Nakicenovic et al. 2000.

SRES Emission Scenarios used for AR4	Description	CO2-equiv. in ppm by 2100	Temperature Change (in deg. C 2090-99 relative to 1980-99)	Sea Level Rise in meters at 2090-99 relative to 1980-99 (conservative)
A1B	Rapid economic growth, global population peaks mid-century (9 billion in 2050), rapid introduction of new and more efficient technologies: **balance across all energy sources**	~850ppm	1.7-4.4 (2.8)	0.21-0.48
B1	Global environmental sustainability, global population peaks mid-century (9 billion in 2050), service and information economy, introduction of clean and resource-efficient technologies	~600ppm	1.1-2.9 (1.8)	0.18-0.38
A2	Regionally oriented economic development, continuously increasing population (15 billion people in 2100), slow technological change	~1250ppm	2.0-5.4 (3.4)	0.23-0.51

Another important aspect about climate projections is the fact that many publications focus on annual averages while seasonal patterns are most important to examine potential impacts on species. An exception is the global climate dataset provided by the CA Academy of Sciences available in Data Basin (www.databasin.org , Figure 4-15). Seasonal averages of temperature and precipitation have been calculated with standard deviations to highlight seasonal changes and give an estimate of the variability of the climate projections. However, there are caveats that need to be taken into account when using the datasets. We use precipitation as an example because it is one of the most difficult climate variables to measure accurately, let alone simulate. In the CA Academy of Sciences dataset, winter precipitation is shown with high standard deviations due to large inter-annual natural variability in winter precipitation and snowfall. Because the amount of summer precipitation is limited, standard deviations for the summer period are small. Yet convective storms and the Arizona monsoon drive the acquisition of moisture in the Sonoran desert and they are very difficult to predict both spatially and temporally. The patchiness of summer storms and high evaporation rates make it difficult for recording stations to describe current rainfall reliably. Annual averages are not ideal for predicting individual species or biological community response because much of species phenology is linked to seasonal conditions. Consequently, we will pay close attention to seasonal variations in climate over time.

In assessing climate change in the REA, we will rely on NCEP 1968-1999 data to report on baseline climate conditions while also using the full PRISM time series 1895-2009 to extend historical and current coverage. We intend to statistically downscale the USGS Hostetler data from 15km to 4km to match the resolution of the baseline data. At a minimum, we intend to consider two time slices (2015-2030) and (2045-2060) for future conditions based on the three emission scenarios: B1, A2, and A1B. We further propose to organize climate projections by season to better estimate biological impacts on conservation targets.

4.3.2 Assessing Management Questions

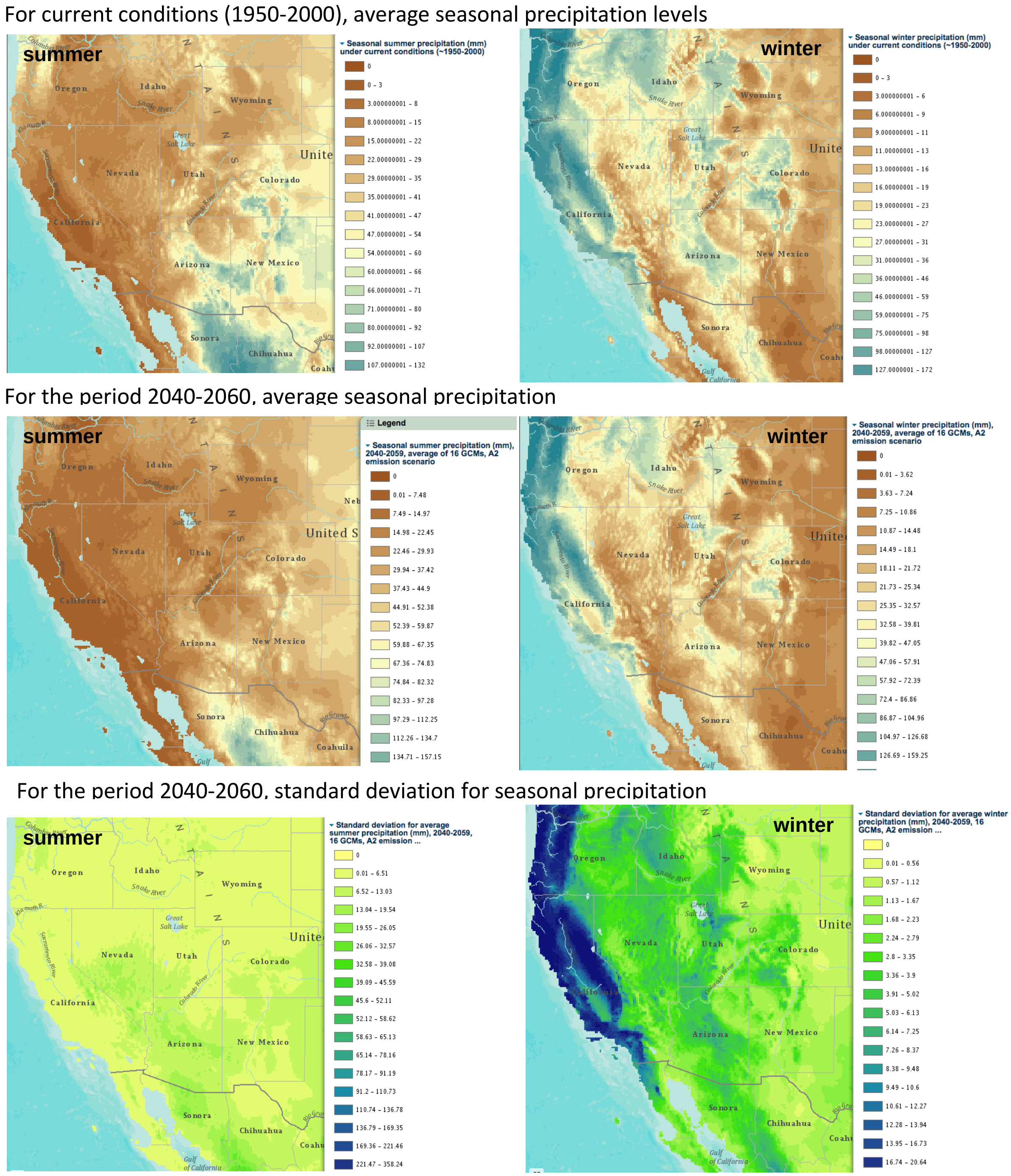
The management questions pertaining to climate change are intended to assess the overall impact of climate change in a spatially explicit fashion on each of the conservation elements of interest whether they are individual species, communities (aquatic and terrestrial), or sites characterized as having high conservation value. Management questions pertaining to species individually or collectively include:

1. *What aquatic and terrestrial species conservation elements are vulnerable to change agents in the near term horizon, 2020 (development, fire, invasive species) and a long-term change horizon, 2060 (climate change)? Where are these species located?*

2. *Where/how will the distribution of dominant native plant and invasive species be vulnerable to or have potential to change from climate change in 2060?*

3. *Where are areas of potential species conservation elements distribution change between 2010 and 2060?*

Figure 4-15. Comparison between current seasonal precipitation levels and projected under the A2 emission scenario averaged over 16 climate models results using WorldClim historical baseline. Data provided by California Academy of Sciences.



For communities and sites with high values, the management questions are:

1. *Where are aquatic/riparian areas with potential to change from climate change?*
2. *Where are surface water flows likely to increase or decrease in the near-term, 2025 (development), and long-term, 2060 (climate change)?*
3. *What high (aquatic and terrestrial) biodiversity sites and movement corridors are vulnerable to change agents in the near term horizon, 2020 (development, fire, invasive species) and a long-term change horizon, 2060 (climate change)? Where are these sites located?*

Conservation element vulnerability, a function of exposure and sensitivity, is an important aspect of many of the management questions. At a minimum, we intend to overlay modeled future climate with current conditions to quantify the change in exposure. We can then superimpose the areas where change is occurring onto the range of the conservation elements and determine their sensitivity based on protocols and procedures defined by NatureServe's Climate Change Vulnerability Index (CCVI) tool, which is based on best professional judgment (Young et al. 2010). The combination of change in exposure and species sensitivity will provide an estimate of conservation element vulnerability.

Time and resources permitting, we would also like to examine vulnerability using two additional approaches. First, we propose to model (or obtain model output for) changes in the climate envelopes for each species (or a subset of species) that are based on environmental response curves for key bioclimatic variables as they pertain to each individual species. This can be done using MaxEnt modeling software. MaxEnt output for climate change scenarios typically shows future potential distributions of individual species with categories of *no change*, *contraction*, and *expansion* of the species' distribution (Figure 4-16). Changes in climate envelopes will be acquired or calculated by Conservation Biology Institute (CBI) staff using the approved input climate models data and scenarios. Areas of current conservation element distribution will be classified as "vulnerable" where the future climate is outside the current observed climate envelope, and "not vulnerable or less vulnerable" for conservation element distribution within the climate envelope, as defined by the base period. We will then use the output developed through this inductive process and compare them to those developed by best professional judgment in the Climate Change Vulnerability Index (CCVI) process.

The second modeling approach does not focus on individual species but rather on water availability for plants at the community level. The MAPSS model (Neilson 1995) is an equilibrium biogeography model that includes a set of biogeography rules that determine climatic zone, life form, and plant type as a function of temperature thresholds and water availability. MAPSS models 52 different vegetation types. It also includes a mechanistic hydrology module that calculates water fluxes through the plant and the soil profile to determine the available water for plant uptake. For more details on the MAPSS model, see www.fs.fed.us/pnw/mdr/mapss/about/index.html.

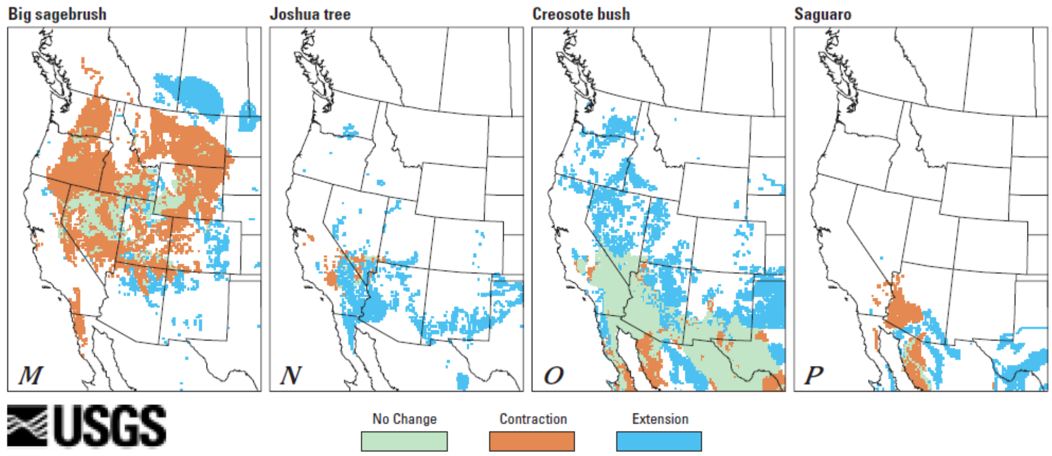


Figure 4-16. MaxEnt modeled output of current (base period) and future potential distribution based on key bioclimatic variables. Areas of CONTRACTION are comparable to areas of VULNERABILITY (sensitivity + exposure), as emphasized by BLM for the REAs (Thompson et al. 1998).

MAPSS calculates the maximum potential LAI (leaf area index) a site can support using a 30-year average monthly climate dataset, assuming the vegetation will use all the available soil water during the driest month of the year. In the model, grasses and trees have different rooting depths and compete for available soil water, while shading by trees limits grass growth. It simulates a CO_2-induced increase in water use efficiency by reducing stomatal conductance by 35% at double the present CO_2 concentration (Eamus 1991). Vegetation classification in MAPSS is based on climatic thresholds and the presence/absence and LAI values of three life forms—trees, shrubs, and grasses—with differing leaf characteristics, thermal affinities, and seasonal phenology.
Since the MAPSS software is built with a mechanistic hydrologic model, it also provides information on changes in hydrology due to climate change and projects changes in habitat characteristics affecting the response of communities that depend on it. These results based on process simulations can then be compared to the results of envelope models based on simple correlations.

4.3.3 Estimating Uncertainty

Understanding uncertainty associated with climate model projections is extremely important as one applies it to management decisions. Predictability declines at the local scale due to the inherent coarse spatial resolution of climate models that generalize diverse vegetation cover and complex topography so important to land managers. Downscaling techniques (statistical or dynamic) bring climate model results to the management scale, but accuracy is limited to that of the original projection. Furthermore, feedbacks from the biosphere to the atmosphere continue to be woefully under-represented in global models and regional model feedbacks to the GCMs have not even been developed yet.

The uncertainty of climate model projections result from the imperfect knowledge of 1) initial conditions such as sea surface temperatures that are difficult to measure, 2) the levels of future anthropogenic emissions which are unknowable since they are dependent on current and future political decisions and social choices and not on physical laws of nature, and finally 3) general system behavior (such as clouds, ice sheet melt) that continues to be the subject of basic climate research and constitute the “known unknowns" of the climate system (Figure 4-17, Cox and Stephenson 2007). In this figure, climate model initialization affects the near term model results, but its impact decreases with time. The uncertainty of emissions scenarios increases exponentially as our guesses about future societal choices become less certain. The uncertainty associated with the current model structure, parameter values, and number of processes included in the model increases with time as new observations cause future model improvements.

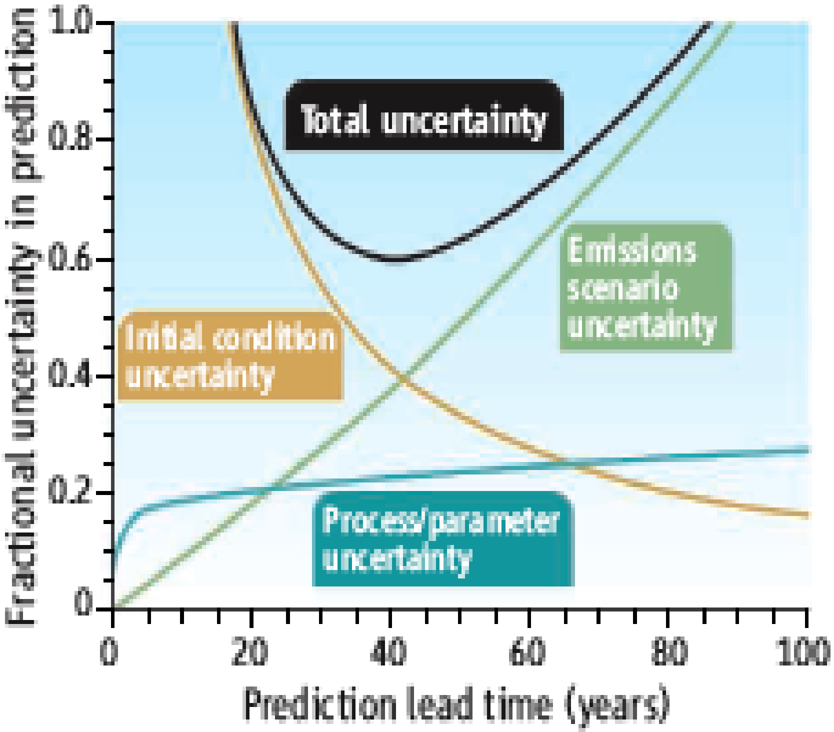


Figure 4-17. Graphic representation of the uncertainty associated with today's climate projections (from Cox & Stephenson 2007).

As the climate is changing, surprises such as the unexpectedly rapid collapse of the Larsen B ice shelf on the Antarctic Peninsula, an ice sheet the size of Rhode Island, routinely bring climate scientists back to the drawing board to improve existing models. Extreme events (long, intense droughts, flood, and hurricanes/typhoons) are difficult to predict. They pose a challenge to policy makers and managers who are more comfortable thinking about chronic linear change rather than abrupt and unpredictable change.

Some researchers have looked at climate prediction uncertainty using an ensemble approach that brings together several climate models and emission scenarios: in other words, locations identified as subject to change in multiple approaches are described as more likely to have that change actually occur. At a minimum, we will provide this level of information.

Another way to consider uncertainty, which is particularly valuable in the context of the REA process, is to determine and communicate relative uncertainty based on site "quality." All climate model projection uncertainty is influenced by a number of factors:1) the distribution and density of meteorological stations whose data have been interpolated to create the historical baseline for the site or region of interest, 2) terrain complexity, which may include areas where local climate decouples from regional climate, and 3) the influence of water bodies and riparian corridors that buffer regional drought stress. We propose to combine these three factors and generate an uncertainty surface data layer that can be superimposed over all other climate change-related outputs to help provide important guidance as to where the models are more or less likely to be accurate across the ecoregional landscape. Proximity to urban areas (urban heat islands), agricultural management such as irrigation, or natural resource extraction that modifies land cover, all greatly affect local environmental conditions and should also be considered as a source of uncertainty.

4.3.4 References Cited

Cox P., and D. Stephenson. 2007. A changing climate for predictions. *Science* 317(5835): 207–208.

Eamus, D. 1991. The interaction of rising CO2 and temperatures with water use efficiency. *Plant, Cell, and Environment* 14:843–852.

Nakicenovic, N. et al. 2000. Special Report on Emissions Scenarios (SRES): A Special Report of Working Group III of the Intergovernmental Panel on Climate Change, Cambridge University Press, Cambridge, U.K., 599 pp. Available online at: http://www.grida.no/climate/ipcc/emission/index.htm and http://sres.ciesin.org/final_data.html .

Neilson, R.P. 1995. A model for predicting continental scale vegetation distribution and water balance. *Ecological Applications* 5: 362–385.

Thompson, R.S., S. W. Hostetler, P.J. Bartlein, and K. H. Anderson. 1998. A strategy for assessing potential future changes in climate, hydrology, and vegetation in the Western United States. U.S. Geological Survey Circular 1153, U.S. Geological Survey, Denver, Colorado.

Young, B., E. Byers, K. Gravuer, K. Hall, G. Hammerson, and A. Redder. 2010. Guidelines for using the NatureServe Climate Change Vulnerability Index. Release 2.0.27. NatureServe, Arlington, VA. 45pp.

5 DECISION SUPPORT MODELING APPROACH

One major challenge of a rapid ecological assessment is an efficient and effective mechanism to consider all conservation elements, change agents, and management questions collectively. A good software candidate should be: (1) spatially explicit, (2) computationally powerful, (3) illustrative of model interactions, (4) transparent, (5) flexible, and (6) easily designed and understood from a logic perspective.

EMDS (Ecosystem Management Decision Support) was selected as the most appropriate software to accomplish the high-level decision support functions required for the REA process. It was presented by the Dynamac team at Workshop 3 on January 24–25, 2011 in Denver, Colorado. BLM evaluated EMDS and rendered a decision on 3-1-11 to drop the use of EMDS as an overarching decision support system and reporting medium for results. EMDS is permitted as long as the final products are delivered in Model Builder. We propose to use EMDS for some aspects of the work (e.g. assessing ecological integrity) to address some of the overarching questions that will help inform a Model Builder solution and facilitate reviewer understanding.

5.1 Ecosystem Management Decision Support (EMDS)

The U.S. Forest Service originally developed EMDS (Reynolds 1999), but the Redlands Institute of the University of Redlands currently maintains it. EMDS runs within ArcMap where it integrates the logic engine of NetWeaver (Rules of Thumb, Inc.) to perform landscape evaluations. The decision modeling engine called Criterium DecisionPlus (InfoHarvest, Inc.) is used for evaluating management priorities (also known as Priority Analyst). Spatial data is the primary input and maps are the primary output – all in ArcMap. Two other plug-ins to EMDS include Hotlink Browser that explains mapped results by automatically tying the map to the logic diagram and Data Acquisition Manager that helps explain the relative importance of missing data (Figure 5-1).

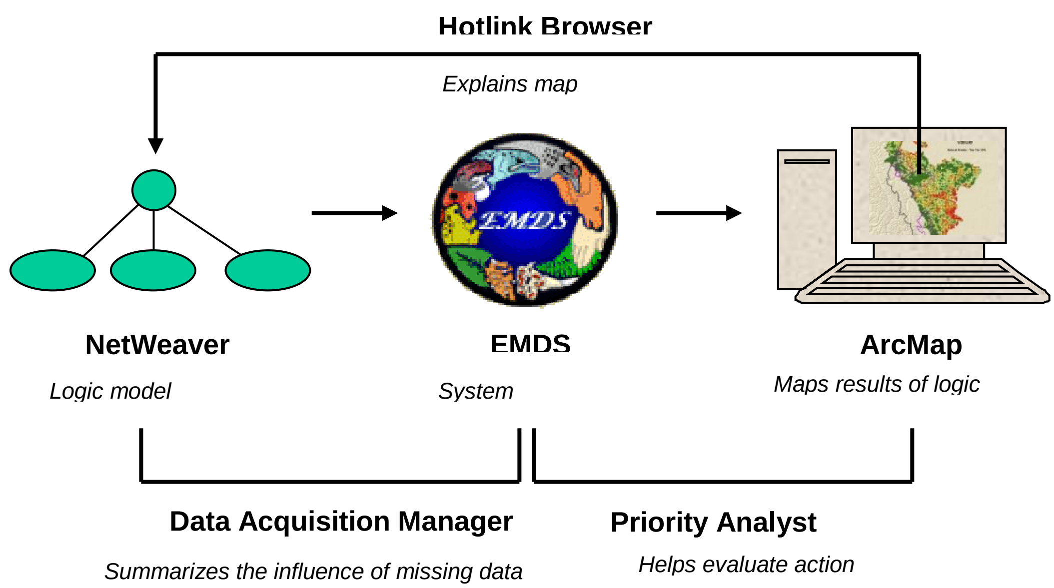


Figure 5-1. Diagram showing EMDS software interaction.

NetWeaver is logic-based software developed to address questions that rely upon spatial data. Unlike conventional GIS applications that use Boolean logic (1,0) or scored input layers, NetWeaver relies on fuzzy logic. Individual spatial data layers are assembled into a hierarchical logic framework to address a particular question. NetWeaver provides easy-to-use tools to form a logical representation of how to evaluate map-based information essentially forming a mental map about a problem or question. There are many advantages of this approach: (1) it is interactive and works well individually or for groups; (2) the graphic design makes it easy to visualize thought processes; (3) the logic components are modular making it easy to include or exclude pieces of the logic design; and (4) numerous and diverse topics can be included into a single integrated analysis.

One of the more powerful aspects of this software as opposed to conventional GIS operations is in the fuzzy logic. Simply put, fuzzy logic allows the user to assign shades of gray to thoughts and ideas rather than assigning values to thoughts and ideas as black (false) and white (true). All data inputs (regardless of the type of number inputs being used – ordinal, nominal, or continuous) are assigned relative values between -1 (false) and +1 (true) up to three decimal places.

For every data input, the user determines how to assign the range of values along a truth continuum. Suppose we are trying to determine and map the most suitable habitat for conserving red wolves. A roads layer is one of several important data inputs, and we know from field research how roads impact wolves. The greater the road density to a threshold of 1.5km/sq km, the greater the negative impact on wolves. Wolves are essentially eliminated from landscapes with road densities above 1.5km/sq km. The logic framework to address wolf habitat considers each layer along a true/false continuum based on a proclamation – "high road densities are bad for wolves." For this example, places with no roads get assigned a value of -1 (false). In other places, as road density increases the closer the assigned value approaches +1 (true). Since we know that wolves respond to a road density threshold (1.5km/sq km), we can assign a +1 value for all places with road densities >1.5km/sq km. Logic trees constructed in NetWeaver assign every input layer in similar fashion allowing for very detailed and transparent ways of thinking about spatial data.

The way in which the data are assembled is controlled by a number of logic operators (e.g., AND, OR, UNION, etc.). **EMDS** reads the Netweaver file and translates it into mapped results within **ArcMap**. Finally, the interactive linking of the various software packages allows the user to view how a particular outcome was derived for maximum transparency—this is <u>not</u> a black box solution. Through the **Hotlink Browser**, users can query a map result and link automatically back to the logic diagram for a graphical explanation of the results. The graphical interface is intuitive and effective in explaining results to broad audiences.

Another key feature of the EMDS environment is the ability to evaluate the influence of missing information on the logical completeness of an assessment. The **Data Acquisition Manager**, in conjunction with the NetWeaver logic engine and the EMDS Project Environment, summarizes the influence of that missing information, given the information that is currently available, and assists the user with establishing priorities for obtaining the missing data to improve the logical completeness of an assessment in the most efficient way. Finally, **Priority Analyst** is a planning component that assists with setting priorities for management activities within an assessment area given results of a NetWeaver logic model. Whereas NetWeaver results show current state of an assessment area, Priority Analyst addresses issues about where to direct management actions for the best outcome. For most applications, maintaining this distinction is important because the landscape elements in poorest condition may not be the best candidates for particular management activities

EMDS has been used in a variety of settings to address a wide range of questions. Analytical units can be almost anything – watersheds, grid cells, ownership polygons, etc. It has been used to predict fire hazards and fuel treatment in central Utah (Hessburg et al. 2007); develop a basin-wide watershed restoration strategy for the Sandy River Basin in northwest Oregon (Johnson et al. 2007); and forest ecosystem sustainability (Reynolds 2001).

EMDS has been used to model high conservation value (HCV) in the central Sierra Nevada ecoregion (White and Strittholt, In Review), Alberta Foothills (Strittholt et al. 2007), and western Oregon (Staus et al. 2010). Finally, EMDS was ranked very highly for resource applications in a comprehensive review of modeling tools by Gordon et al. (2004). See http://www.spatial.redlands.edu/emds/ for more information.

5.2 Proposed Decision Support Modeling Approach

For a rapid ecological assessment, our recommendation is to pursue an EMDS modeling solution. It provides the necessary functionality to guide users to gain a better understanding of current conditions, helps decision makers identify important elements to emphasize and track, and easily incorporates other aspects of any decision making process. The logic aspect of REAs, which is fundamentally important to the process, is completely transparent and easy for non-technical users to participate using EMDS. Of the three software packages we evaluated, it also has the greatest flexibility and has been used more than the others to assess high conservation value specifically in a number of geographic settings.

An EMDS modeling approach can be reasonably assembled over a period of a few months and the resulting logic models and products can be easily reviewed throughout the process and can be operational well-beyond the life of the initial REA. Users can always build more logic components, add new and improved datasets to the existing models, and edit the existing framework as new information emerges.

5.3 Draft EMDS Model

An initial draft of an EMDS logic model, which integrates all of the various conservation elements and change agents listed in the BLM scope of work for the REA, is provided through a series of figures and descriptions throughout this section. The details furnish a starting point – the entire model is subject to review and revision. The basic design and functionality of how EMDS is used as an integrating framework for the REA process is important to communicate. However, using this framework DOES NOT mean all modeling and analytical work is carried out in this single software. Rather, EMDS will take outputs from other modeling efforts reviewed in this document and will integrate them into this important decision support framework. For example, we propose to use FRAGSTATS to describe landscape fragmentation. FRAGTSTATS is run completely outside of EMDS. In this case, outputs from FRAGSTATS would be used as inputs to EMDS.

5.3.1 Modeling Ecological Integrity

The concept of ecological integrity is complex and a great deal has been written about it in the literature (e.g. Angermeier and Karr 1994 and Pimentel et al. 2000). Other terms often used interchangeably with integrity include ecosystem health, resilience, resistance, and stability. In almost all treatments of ecological integrity, the focus has been on the 'ecosystem' not specific species or even plant communities. As Karr and Dudley (1981), working in aquatic systems, describe it – ecological integrity is the sum of all physical, chemical and biological integrity. Karr and Chu (1995) later define integrity as "the capacity to support and maintain a balanced, integrated, adaptive biological system having the full range of elements (genes, species, assemblages) and processes (mutation, demography, biotic interactions, nutrient and energy dynamics, metapopulation processes) expected in the natural habitat of a region."

More simply stated ecological integrity is the degree to which all ecosystem components and their interactions are represented and functioning.

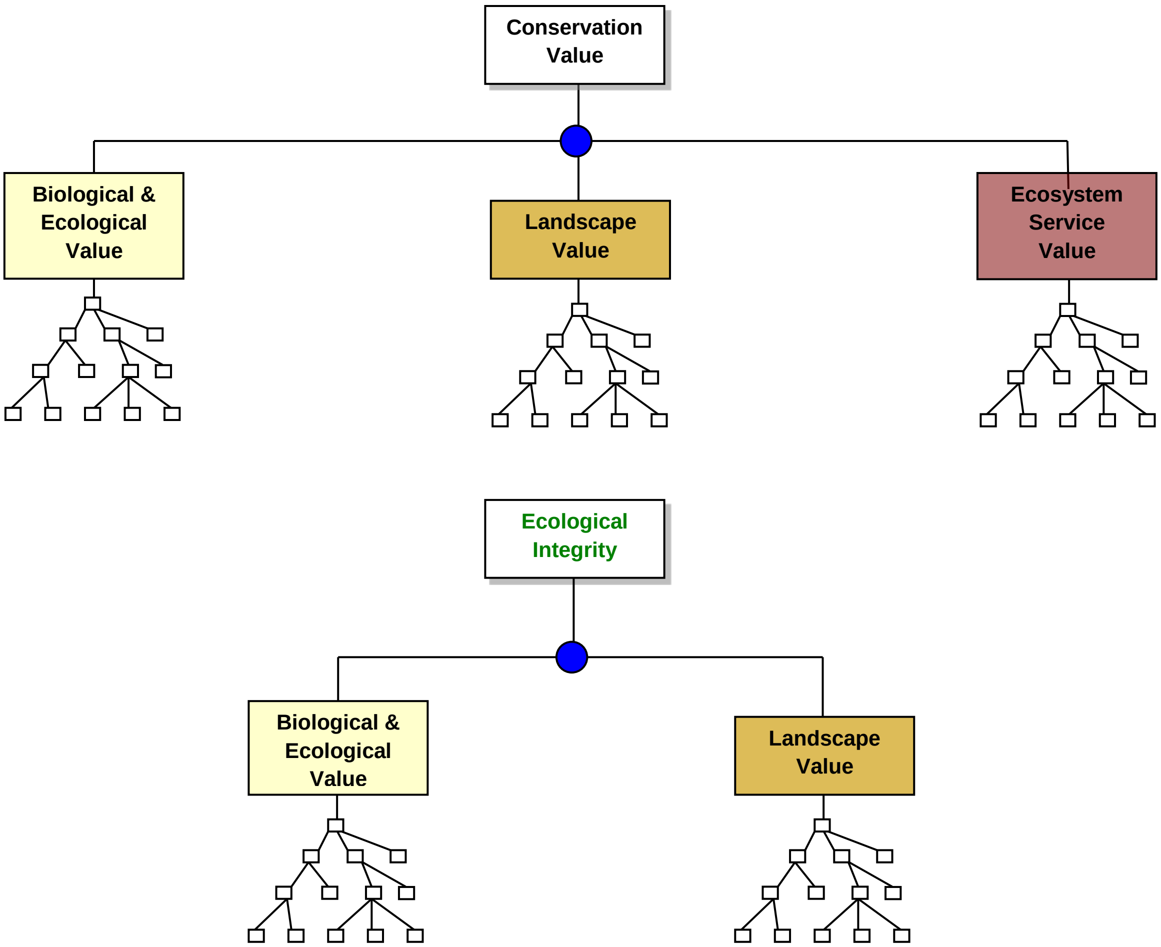


Figure 5-2. Diagram showing simple EMDS expression of Conservation Value and Ecological Integrity.

As described in the previous section, Conservation Value is a function of Biological and Ecological Value, Landscape Value, and Ecosystem Service Value. The majority of the REA scope of work involves biological/ecological and landscape values, although a few ecosystem service values listed as well (e.g. soil stability and groundwater). For our proposed approach, Ecological Integrity is a combination of Biological/Ecological Value and Landscape Value (Figure 5-2). In this simple graphic, the small modeling tree under each topic heading means there is a logic diagram under each one. Blue dots represent that a specific EMDS operator controls the logic for the ways that the various pieces combine to determine the outcome of the next highest topic.

The data and information supporting the Biological and Ecological Value topic is aimed to address more of the components side of the definition while the Landscape Value topic addresses more of the function side. One may think of the various inputs under each topic as attributes and indicators that are combined together to define relative Ecological Integrity. We have proposed this integration approach as it is superior to alternative methods such as the use of indices. Indices are simple constructs of complex ideas but quickly become difficult to interpret and understand. The method we are proposing is more discriminating as it represents results along a more data driven continuum and is far more transparent - users can query our modeling approach and easily obtain the rationale for a particular outcome. Indices are usually more difficult to understand – this location got a score of a "6.5" and this other one a score of "4.2" but users typically have no way of knowing why.

Ecological Integrity is generated by combining Biological/Ecological Value and Landscape Value. Biological/Ecological Value is comprised by the combination of Aquatic and Terrestrial Biological/Ecological Values. Overall Landscape Value is the combination of Aquatic Landscape (or Waterscape) Value and Terrestrial Landscape Value (Figure 5-3).

Please note that throughout the next series of figures the blue dots represent a logic operator and the stack of gray files under a topic means one or more spatial datasets. For each topic (colored box) in the logic diagram, a map will be generated after each logic model run. We have also added tier breaks to the figures to aid in orientation – Tier I is the highest order topic and tier VIII is the lowest.

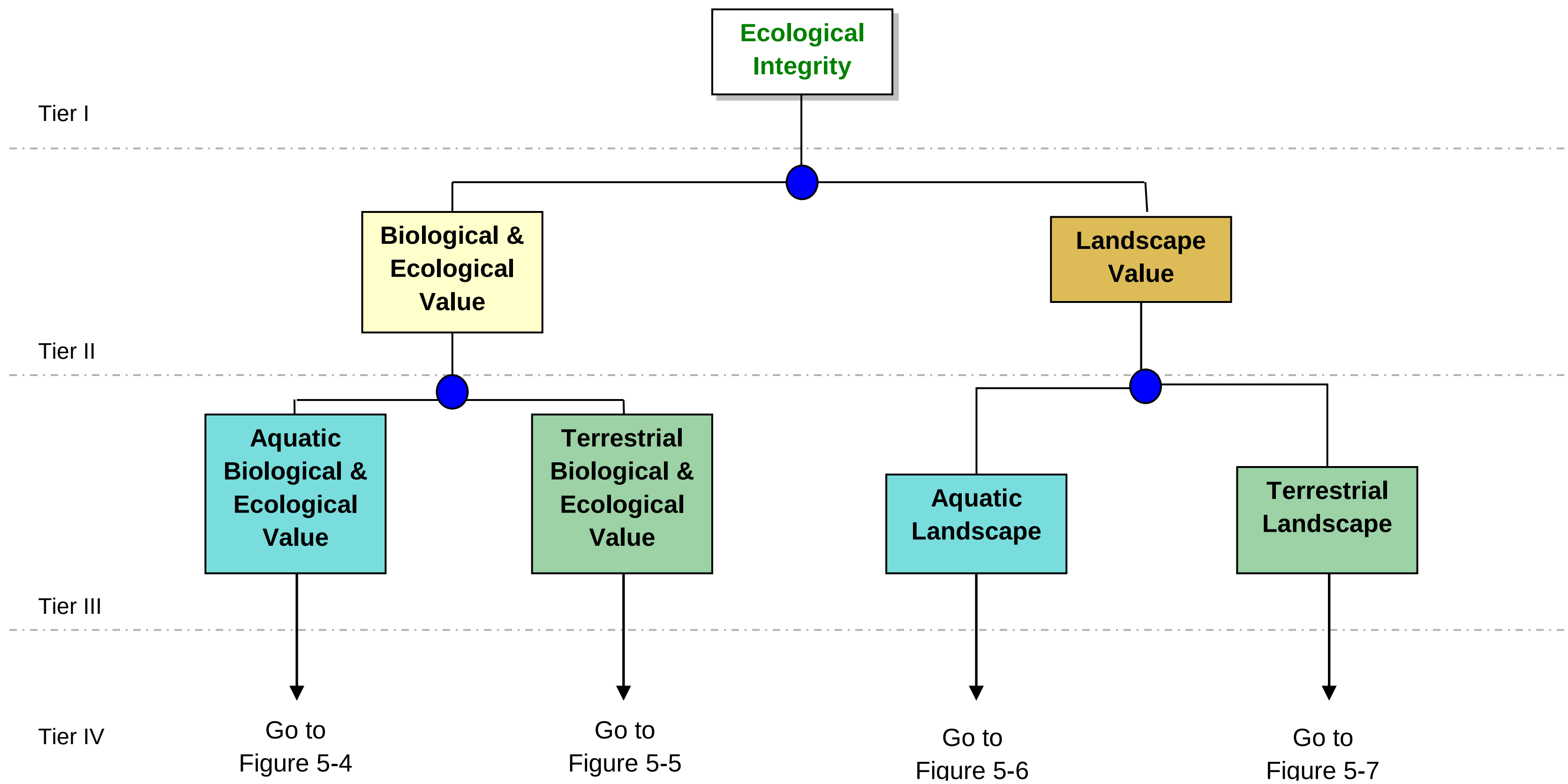


Figure 5-3. Draft EMDS logic diagram showing high level topics for Ecological Integrity modeling.

Aquatic Biological and Ecological Value is defined by Important Aquatic Communities, Aquatic Special Features (e.g., springs, palm oases, etc.), and Aquatic Species (Figure 5-4). All tier IV topics have equal influence on Aquatic Biological and Ecological Value. The value of Important Aquatic Communities combines High Value Aquatic Ecological Systems based on rare Ecological Systems from the national standard and Aquatic Communities of Special Interest requested from BLM. These special interest communities are comprised of Lotic (including rivers and streams) and Lentic (including lakes, ponds, and wetlands) Ecosystems. Aquatic Species is a composite of Aquatic Animal Species of Interest and Aquatic High Value Species Composite. "Important sites" is a combination of known locations and suitable habitat. The Aquatic High Value Species Composite consists of data on Aquatic Species Endemism, Aquatic Threatened and Endangered Species, and Aquatic Species Richness, which in turn is composed of datasets on Fishes, Insects, and Other Invertebrates. It may be desirable to emphasize locations of endangered species by weighting this topic. Doing so will preserve these sites as they are incorporated into higher level topics.

Terrestrial Biological and Ecological Value is defined by combining Important Terrestrial Communities, Terrestrial Special Features (e.g., nesting areas, slat licks, etc.), and Terrestrial Species (Figure 5-5). Important Terrestrial Communities is a composite of High Value Terrestrial Ecological Systems based on rare Ecological Systems from the national standard and Terrestrial Plant Communities of Special Interest (including Sonoran Paloverde-Mixed Cacti Desert Scrub and Sonoran-Mojave Creosotebush-White Bursage Desert Scrub.

The Terrestrial Species topic is derived from topics on Terrestrial Plant Species of Interest (including Saguaro Cacti and Creosotebush). Terrestrial Animal Species of Interest include Birds, Mammals, and Reptiles. For the topic Bird Composite, the model will combine all important sites for each species of interest (including golden eagle, Lucy's warbler, Le Conte's thrasher, Bell's vireo, and Southwest willow flycatcher). For the bird species, the logic model is showing two different ways data could be entered. Species that have good data on their locations and habitat needs would be treated as in the golden eagle example. For those with a red star, these would be modeled using Maxent or other species distribution modeling tools to generate high habitat suitability for the species (See species modeling section). PLEASE NOTE: These are just examples. Each species would be evaluated to determine the best modeling approach. Regardless of the process pursued, the outcome would be high occurrence probability.

Note: According to BLM guidance we are to 1) use existing models wherever they are available with a full ecoregional extent or wherever they may readily be extended from a portion of the assessment region; 2) if existing models are not available, but occurrence data are available, we will use a modeling approach such as MaxEnt; 3) if occurrence data are lacking, we will use existing SW ReGAP models.

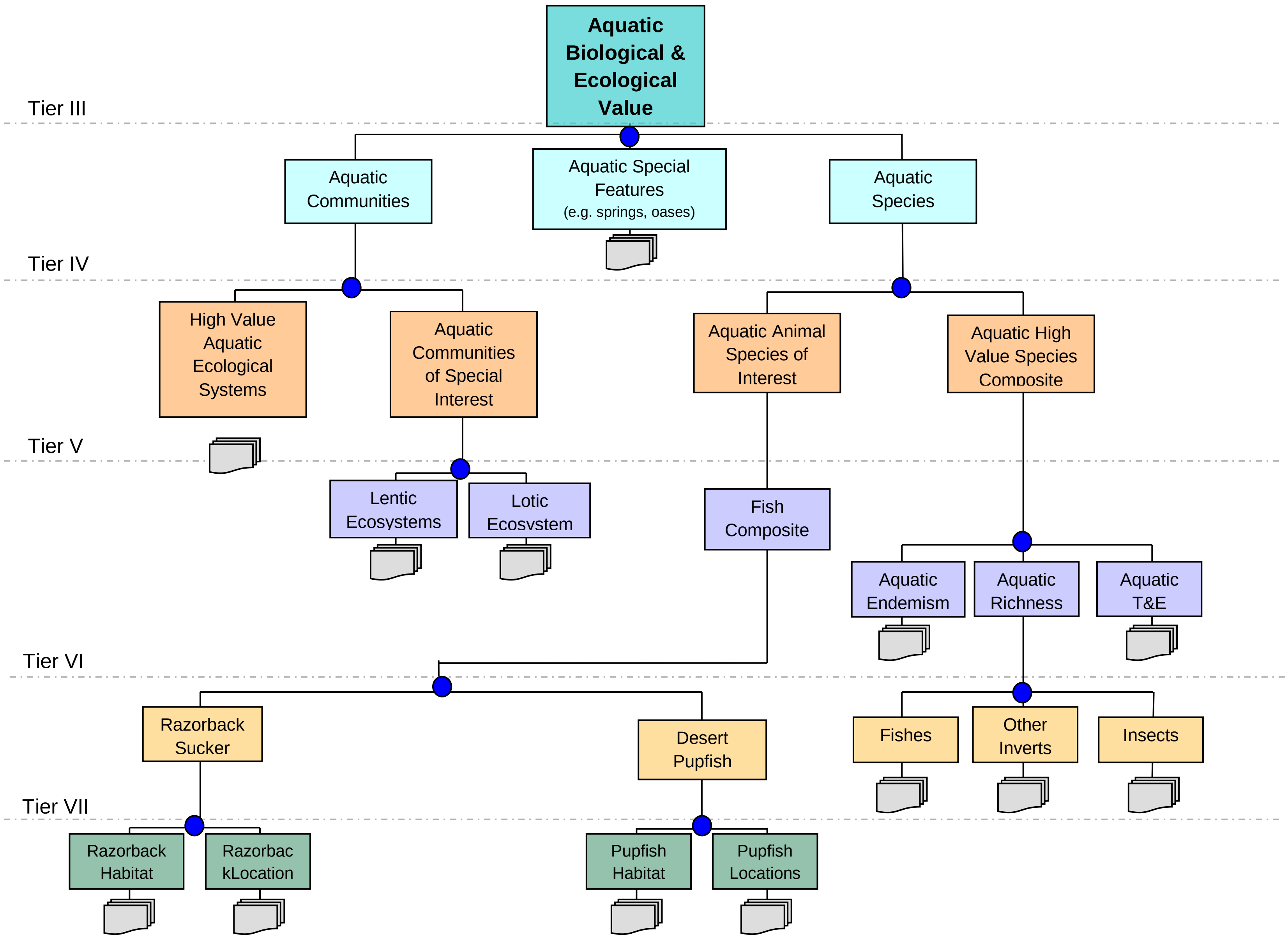


Figure 5- 4. Draft EMDS model showing model logic for Aquatic Biological and Ecological Value.

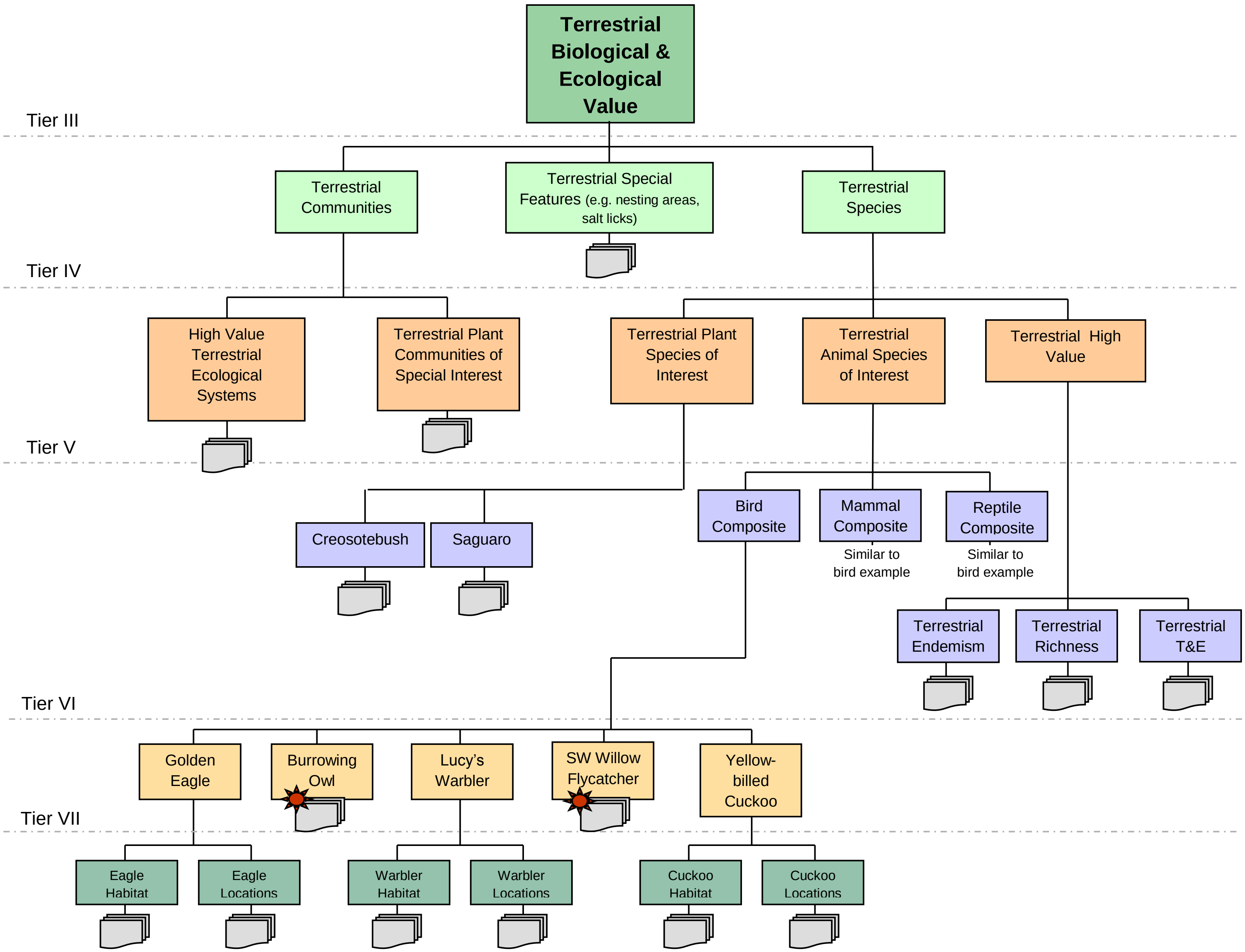


Figure 5-5. Draft EMDS model showing model logic for Terrestrial Biological and Ecological Value.

Overall Landscape Value is defined by Aquatic Landscape (or Waterscape) Value and Terrestrial Value. Aquatic Landscape Value combines overall Lotic Ecosystem Intactness (Figure 5-6a) and Lentic Ecosystem Intactness (Figure 5-6b).

Lotic Ecosystem Intactness is based upon results from several topics – Streamside Degradation, Invasives, Water Flow, Water Quality, Land Use, and Fragmentation. Two of these topics – Invasive and Water Quality – are based on available spatial datasets. The remaining four topics require greater development of the logic model. Streamside Degradation is dependent upon datasets on Developed and Grazed lands. Water Flow is influenced by Disturbance Regime (i.e., degree of departure from natural flooding events) and Developed Water. Developed Water combines Groundwater Use, Water Diversions, and Water Use with Water Use further based upon impacts from Urban, Agriculture, and Energy users. Land Use is driven by four other topics – Road Density, Road/Stream Intersections, Impervious Surface, and Agriculture. Existing datasets support all of these topics. Fragmentation is based on the degree of water diversions and number of dams. The Lentic Ecosytem Intactness logic model topic (Figure 5-6b) is very similar to the lotic ecosystems model only from the standpoint of non-flowing water bodies.

Terrestrial Landscape Value consists of two topic inputs – Wildlife Linkages and Landscape Intactness (Figure 5-7). Wildlife Linkages is made up of Existing Modeled Corridor datasets provided by the state wildlife agencies. If unavailable, we propose to generate potential linkages for the ecoregion using the approach described by Spencer et al. (2010) for designing essential wildlife connectivity for the State of California. In their approach, they defined a series of 'natural landscape blocks' and then designed a network of links using least-cost-path analysis based on a developed base friction layer and a series of operating rules.

We propose an additional assessment evaluating the relative importance of defined wildlife linkages using software called the Connectivity Analysis Toolkit (version 1.1) is proposed. The Connectivity Analysis Toolkit evaluates regional connectivity using a number of different tools that help define and discriminate the importance of modeled linkages. Based on circuit theory, one of its functions is to assign relative importance of each segment of a connectivity network. We propose to use it to answer the question: What is the relative importance of known or modeled wildlife linkages throughout the ecoregion?

The Connectivity Toolkit was designed by Carlos Carroll and Brad McRae, and the software was engineered by Allen Brookes, Kevin Djang, Nathan Schumaker, and Carlos Carroll. The Connectivity Toolkit is based in part upon portions of the following software modules – Hexsim, LEMON, NetworkX, and Python. Please see McRae and Beier 2007 and McRae et al. 2008.

Landscape Intactness is based on topics called Developed, Natural Disturbance, Ownership Status, and Fragmentation. The development footprint is derived from the combination of Linear Development, Invasives, and Developed Area. Linear Developed depends upon Roads, Utilities, and Railroads data from which are expressed as densities since our analytical units (5^{th} field watersheds) are irregular in shape and size. Invasive inputs will be based on available distribution data for tamarisk and cheatgrass for the Sonoran Desert. Developed Area will include Urban, Agriculture, and Energy footprints.

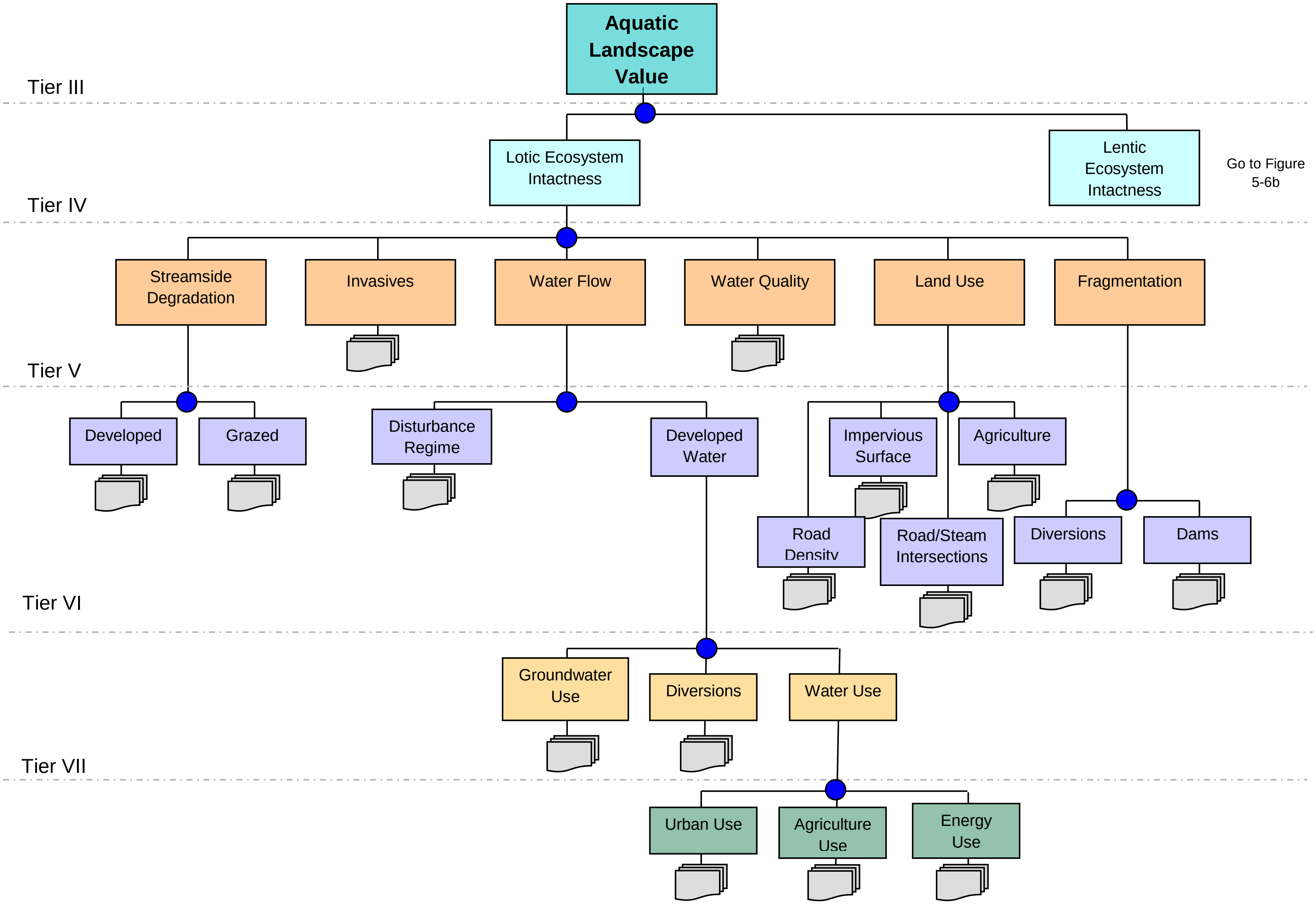


Figure 5-6a. Draft EMDS logic model for Aquatic Landscape Value – Lotic Ecosystem Intactness.

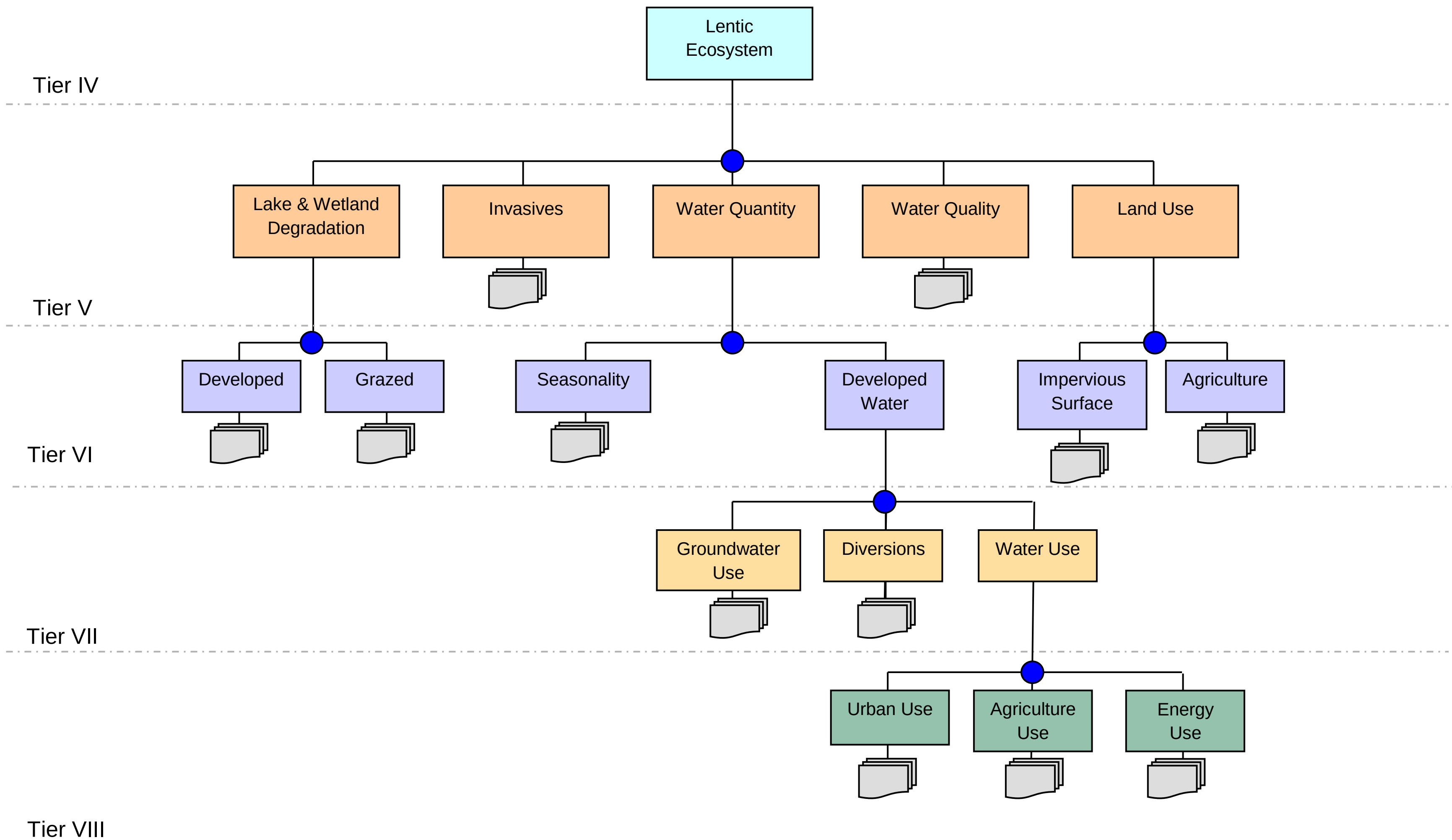


Figure 5-6b. Draft EMDS logic model for Aquatic Landscape Value – Lentic Ecosystem Intactness.

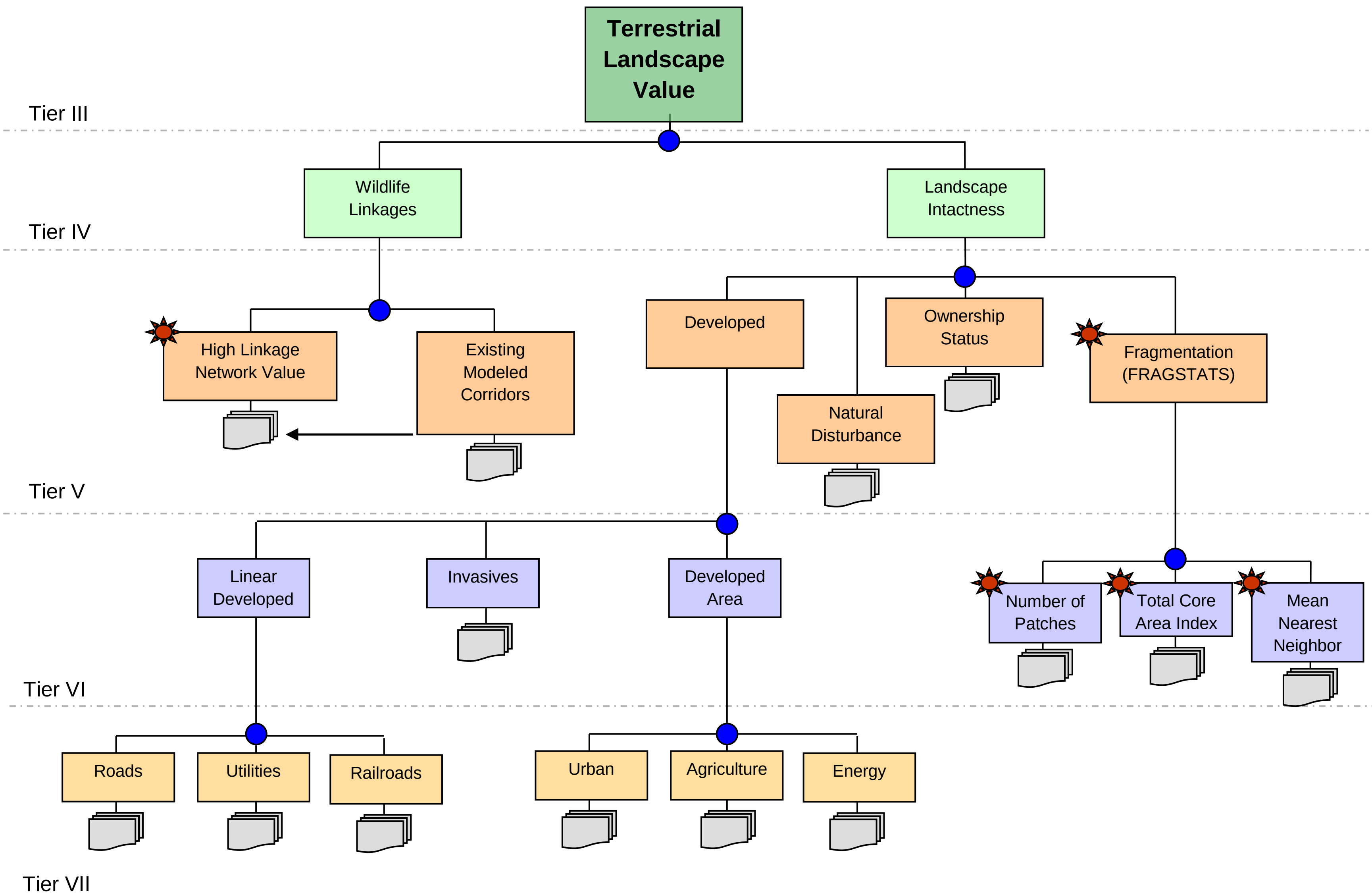


Figure 5-7. Draft EMDS logic model for Terrestrial Landscape Value.

The fragmentation topic will be handled using FRAGSTATS (see McGarigal and Marks 1995). Using SWreGAP data as the primary input, the ecoregion will reclassified into three classes – natural cover, unnatural cover, and water. FRAGSTATS will be run with each watershed as a defined landscape. Using FRAGSTATS terminology, a select number of 'patch', 'class', and 'landscape' level metrics will be used to define overall fragmentation. It is important to choose from the lengthy list of metrics generated by the software as to avoid redundancy. Based on previous experience (see Staus et al. 2010), three metrics provide a solid foundation for assessing fragmentation – (1) index of distance to between natural patches in a landscape – mean nearest neighbor (MNN), (2) number of patches of natural cover (NP), and (3) area-corrected index for amount of interior natural habitat - Total Core Area Index (TCAI). Depending on the data, we may elect to add or substitute metrics for the best performance. Other possible metrics include largest patch size, total edge, and landscape contagion.

5.3.2 Reference Condition and Historic Range of Variability (HRV)

The concept of reference condition and indicators for reference condition have been applied by regulatory and management agencies for 25 years. Initial applications to aquatic ecosystems were introduced in the mid-1980s (Hughes 1985, Hughes et al. 1986, Whittier et al. 1987). In those studies, stream test (disturbed) sites were compared against multiple reference sites of similar size and in the same ecoregion (Omernik 1987) that had least-disturbed catchments. In much of the U.S., reference conditions are based on least-disturbed conditions because of the extensive and intensive disturbance by humans; in other words, few reference sites are minimally-disturbed, let alone pristine (Hughes 1995, Stoddard et al. 2006, Whittier et al. 2007b). The indicators of aquatic condition were typically multimetric indices (MMI) based on field collections of fish, macroinvertebrate, or diatom assemblages (e.g., the Index of Biological Integrity [IBI], Karr 1981, Karr et al. 1986, Roset et al. 2007). A multimetric index aggregates scores from multiple variables or metrics into a single number that can be easily graphed, explained to decision-makers, and tracked through time and space to depict status and trends (Stoddard et al. 2008, Whittier et al. 2007a). Such metrics typically include standardized scores incorporating taxa richness and abundance as well as various guilds (tolerance, trophic, habitat, reproductive, life history). More recently, large survey data sets have been developed at regional, national and continental scales. From those data sets, least-disturbed reference sites have been used for developing predictive models for MMI (Pont et al. 2006, 2009, Moya et al. 2011) or taxonomic richness (Paulsen et al. 2008). Coupled with probability surveys producing regionally representative data, such models have been used for assessing the ecological condition of all mapped streams in the conterminous U.S. (Paulsen et al. 2008) or western conterminous USA (Pont et al. 2009) with known confidence intervals. In all these latter cases, reference conditions are described by normal curves depicting the distribution of index scores across the reference sites and thereby representing regional variability in expected scores at least-disturbed sites against which scores for disturbed sites can be assessed. Both the MMI and aquatic reference conditions are founded on the concept of biological integrity, which is the objective of the Clean Water Act of 1972 (USGPO 1989) and was described by Frey (1977) and Karr and Dudley (1981) as "the ability to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitat of the region."

Terrestrial ecologists also have a goal of using ecosystem integrity as a benchmark, defining or modeling a system that maintains its structure and function through time by being resistant and resilient to natural disturbance regimes (e.g., Whitford and deSouza 1999). Because of their focus on the spatially extensive pattern and long-term temporal dynamics of multiple types of vegetation assemblages, terrestrial ecologists have been faced with a far more complex task of assessing reference condition than have aquatic ecologists. In addition, terrestrial ecologists lack a widely used indicator with which they can track status and trends across landscapes, regions, or continents. Consequently, forest and range ecologists have been longer in developing adequate landscape assessment tools and now tend to use historical range of variability (HRV, Kean et al. 2009, Schussman and Smith 2006) of plant associations. HRV incorporates components of vegetation composition and structure along a range of potential conditions, including the area and distribution of vegetation cover types; therefore, it is typically displayed in digital map format or in distribution and abundance probabilities for a particular delineated area (e.g., basin or ecoregion). HRV maps and distribution probabilities are based on data and maps of landscape characteristics (e.g., topography, soil, climate), knowledge of historical trends and disturbance events, and computer modeling. Of course, accurate data and knowledge are limited at regional spatial extents (Keane et al. 2009, Swetnam et al. 1999).

5.3.2.1 Operational Reference Condition

In the Rapid Ecoregional Assessments, the status of a conservation element will be expressed by comparing measures of current conditions to reference conditions, and determining the degree to which those conditions differ within appropriate landscape summary units. However, we are faced with the challenge of assessing reference condition spatially over a continuous landscape. Vegetation community is a key element for estimating current and reference conditions for conservation elements; thus it is necessary to obtain vegetation community datasets for both current and reference conditions. Current vegetation conditions can be expressed using either the Southwest Regional Gap (SW RegGAP) land cover dataset (2004) or the LANDFIRE Existing Vegetation Type (EVT; National released 2006, Refresh 2008 to be released 2nd quarter 2011). The only dataset that is available over the entire region that attempts to map reference condition is the LANDFIRE Biophysical Settings (BpS) dataset; it depicts the vegetation communities that may have been dominant on the landscape prior to Euro-American settlement (www.landfire.gov) and thus provides the best available representation of vegetation community reference conditions. All vegetation communities are described in terms of NatureServe's Ecological Systems classification and are mapped using a combination of vegetation plot data, biophysical gradients, vegetation dynamics models, and other information as available. The BpS units are coupled with reference condition vegetation dynamics models, which describe the primary succession classes (e.g., post-fire vegetation, old growth forest) and their state-transition probabilities, including rates of fire, which would most likely have occurred under pre-settlement conditions. These probabilities are integrated to estimate the proportion of each BpS unit that would be occupied by each succession class averaged across *time and space*. These values are averages and *do not* express ranges of variability (HRV, as discussed above), nor can the locations of the BpS be used to express a spatial range of variability or patch characteristics.

It is important to note that the BpS units describe a spatially-dynamic mosaic of succession classes over time as the landscape experiences disturbances, such as fire, and vegetation succession in the absence of disturbance. Thus care must be exercised when comparing these reference vegetation conditions to current vegetation conditions, which represent a single point-in-time estimate of vegetation communities on the landscape. Typically, this comparison is performed by aggregating current vegetation type and structure (e.g., LANDFIRE EVT, vegetation canopy cover, and vegetation canopy height products) into the succession classes defined for the Biophysical Settings on which those combinations fall, or additional states that represent conditions that would not have occurred under reference condition dynamics (e.g., invasive vegetation types). The percent of a BpS occupied by each succession class and uncharacteristic condition is then calculated within an appropriate landscape summary unit (e.g., 5th level HUC or 4th level ecological region) and compared to the percentages of those succession classes that would have been expected under reference conditions.

5.3.2.2 Current Vegetation Conditions

LANDFIRE EVT will be used wherever possible to estimate current vegetation condition because it minimizes errors of comparison when used alongside the LANDFIRE BpS. This is because both products were produced using similar input data and methods and they have been rectified against each other as part of the LANDFIRE mapping process. However, to determine potential errors and uncertainties in the LANDFIRE EVT, it will be overlaid on the SW ReGAP land cover dataset and areas of significantly different vegetation communities will be highlighted. It is essential to use the latest available LANDFIRE EVT because it incorporates disturbance effects to vegetation communities from recent fires missing from the original LANDFIRE EVT along with other refinements. Many of the significant ecological differences between the LANDFIRE EVT and SW ReGAP may have been addressed in creation of these newer versions. Where significant differences exist between LANDFIRE EVT and SW ReGAP, these areas will be evaluated to determine a) if any differences would impact the distribution of the conservation element and b) if those differences are related to recent disturbances not captured by SW ReGAP. Where SW ReGAP is deemed to better capture current vegetation in these areas of high difference, the LANDFIRE EVT will be corrected using SW ReGAP.

In summary, the proposed approach for assessing change in vegetation and habitat for conservation elements will follow the three steps listed below:

1. We will evaluate the most current LANDFIRE EVT against SW ReGAP data and summarize the differences as it pertains to each conservation element.

2. For those species where the comparison shows large agreement (and with species that are habitat generalists), we will use LANDFIRE EVT and LANDFIRE BpS to avoid introducing additional error from using different data sources.

3. For those species where the comparison shows large disagreement (most likely with habitat specialists), we will hybridize the two current vegetation layers (LANDFIRE EVT and SW ReGAP) in order to represent the best predictor of current habitat. This hybrid file will then be used in conjunction with LANDFIRE BpS to explain relationship to reference condition.

5.3.3 Modeling Ecosystem Services

For the REA, four ecosystem services were listed as topics of interest. Two are supporting services – Soil Stability and Air Quality – and two are provisioning services – Forage and Groundwater. In the case of supporting services, ecological integrity and human needs are positively served (e.g., good air quality is good for both humans and natural ecosystems). Provisioning services, however, can be seen as often conflicting – more forage for livestock means less forage for wild animals. EMDS is extremely flexible and can allow for different interpretations of the same data inputs and models. The simple EMDS expression of all the ecosystem services is provided in Figure 5-8.

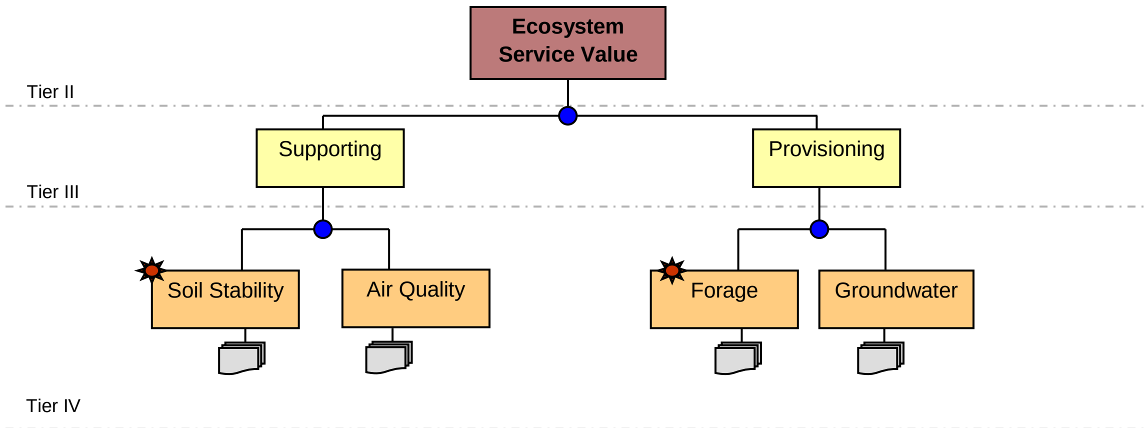


Figure 5-8. Draft EMDS diagram for Ecosystem Service Values.

5.3.4 Modeling Change Agents

There are four major change agents of interest in the REA –Development, Invasive Species, Fire, and Climate Change. Unlike modeling for Landscape Intactness, which is a reflection of current condition, change agents as used in the REA scope of work pertain to probable future conditions. For each conservation element in question, the response to change has many dependencies – sensitivity to each particular change agent, impact of multiple change agents on the element, and even the interaction of the change agents themselves (e.g., the relationship of climate change, invasives, and fire). Forecasting future conditions is complex as change agents are often stochastic and uncertain.

While recognizing the shortcomings in the selected models, we believe that we can generate useful spatially explicit information on an ecoregional scale to provide guidance as to where change might have the most effect on identified conservation elements. We propose to build a separate EMDS model for users to consider the combined impact from the different change agents or to consider them individually (Figure 5-9). The inputs into the EMDS model are generated outside of EMDS (note the red stars above many of the topics), but as with modeling Ecological Integrity, we can pull the derived inputs together for each component (or topic) to create an individual potential change surface for each agent as well as a cumulative potential change surface that we can then be applied to any of the topics from the previous model. For example, if you are interested in where Sonoran Desert woodlands are likely to be affected by fire in the future, you could overlay the topic called Sonoran Desert woodlands with the topic from the Change Agent EMDS called Fire. If you want to know how energy development might impact Aquatic Landscape Value, you could overlay the Aquatic Landscape Value topic with the Energy Development topic in the Change Agent EMDS model. Alternatively in this case, you could substitute the current energy development footprint with a modeled future energy footprint in the Ecological Integrity model and run it again. Both approaches would be informative.

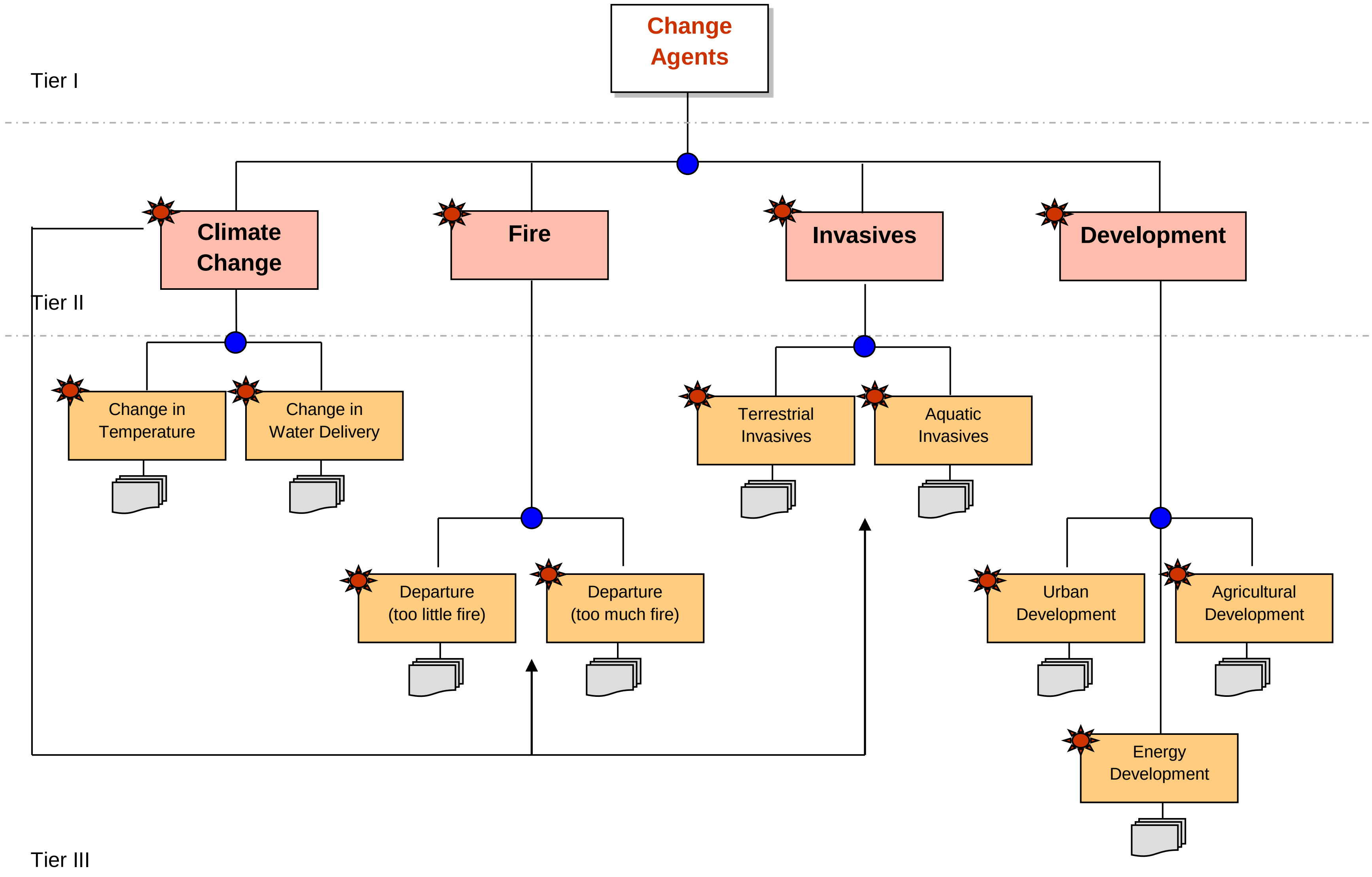


Figure 5-9. Draft EMDS model for change agents of interest. Note that input data for all topics are generated outside of EMDS.

5.4 Data Basin: Decision Support and Data Management System

We propose to use Data Basin (www.databasin.org) as the overall data management and decision support system for integrating the different components of this complex project. Data Basin is an innovative web-based mapping system that connects users to conservation spatial datasets, numerous mapping and analytical tools, and scientific expertise. Individuals and institutions can explore and download thousands of conservation spatial datasets, upload their own datasets, connect to other external data sources, and produce customized maps that can be easily shared. Users can also gain quick access to experts, group functions, and specialized analytical tools. The Conservation Biology Institute developed Data Basin in partnership with the foundation community and the Esri Corporation. Publicly launched in July 2010, Data Basin currently has over 1,200 registered users (individual membership is free) and is housing over 3,000 spatial conservation datasets (adding approximately 50 per week).

This advanced web-based technology will upload, integrate and manage the numerous datasets needed to implement the BLM REA modeling and analytical processes and various final products. Data Basin will support the review process for this input data, modeled results and alternative management scenarios. The system will also allow resource managers to upload new data, update existing data, and adjust weighting factors to reflect the dynamic and changing nature of the landscape and associated management challenges. In summary, the development of this decision support application for the BLM REA will ensure that this effort will result in the sustained ability for BLM to address future management challenges, rather than create one set of analytical results that might lose their relevance over time.

5.4.1 Using Data Basin for Review and Product Delivery

Upon approval by BLM, we propose to set up either a 'closed' or 'by request' group in Data Basin where we will post spatial datasets for each ecoregion as well as draft outputs from EMDS along with attached supporting documentation. Closed groups consist of an approved list of members. A designated group administrator manages the by-request groups. The administrator is responsible for approving or denying Data Basin access to prospective users who express interest in joining the group. This operates like a public group.

Reviewers will have easy access to the group and receive instructions and tools for conducting a review. The easy-to-use functions and features in Data Basin allow non-GIS users to access conservation spatial data and to contact GIS professionals. In this way, users can work from their own computers at their own time and pace, once users meet established review milestones.

5.4.2 Presentation of Final Products

In addition to the list of deliverables as outlined in the scope of work, we propose to provide all of the input spatial datasets and final EMDS model results as published galleries in Data Basin for access by the BLM and, as permitted by data sharing agreements, the rest of the Data Basin community. This will allow the entire body of work easily accessible to users via the Internet without needing to acquire GIS software.

It may be possible to host EMDS within Data Basin itself for ongoing investigation by BLM staff, but that would require additional programming and system support, which are out of scope for this REA.

In addition to the list of deliverables as outlined in the scope of work, we propose to provide all of the input spatial datasets and final EMDS model results as published galleries in Data Basin for access by the BLM and, as permitted by data sharing agreements, the rest of the Data Basin community. This will make the entire body of work easily accessible to users via the Internet without needing to acquire GIS software.

<u>Note: Although the Dynamac team may use EMDS for various aspects of the project, we will deliver all final products in Model Builder as specified by BLM.</u>

5.5 References Cited

Angermeier, P.L. and J.R. Karr. 1994. Biological integrity versus biological diversity as policy directives. *BioScience* 44:690–697.

Aplet, G., and W.S. Keeton. 1999. Application of historical range of variability concepts to biodiversity conservation. Pages 71–86 *in* R.K. Baydack, H. Campa, and J.B. Haufler (Editors) Practical approaches to the conservation of biological diversity. Island Press, New York.

Frey, D.G. 1977. Biological integrity of water - an historical approach. Pages 127–140 in R.K. Ballentine and L. I. Guarria. The integrity of water. Report 335. U.S. Environmental Protection Agency. Washington, DC.

Gordon, S.N., K.N. Johnson, K.M. Reynolds, P. Crist, and N. Brown. 2004. Decision support systems for forest biodiversity: Evaluation of current systems and future needs. Final Report – Project A10 National Commission on Science and Sustainable Forestry. Available at www.ncssf.org (April 2007).

Hessburg, P.F., K.M. Reynolds, R.E. Keane, K.M. James, and R.B. Salter. 2007. Evaluating Wildland fire danger and prioritizing vegetation and fuels treatments. *Forest Ecology and Management* 247:1–17.

Hessburg, P.F., B.G. Smith, R.B. Salter, R.D. Ottmar, and E. Alvarado. 2000. Recent changes (1930's–1990's) in spatial patterns of interior northwest forests, USA. *Forest Ecology and Management* 136:53–83.

Hughes, R.M. 1985. Use of watershed characteristics to select control streams for estimating effects of metal mining wastes on extensively disturbed streams. *Environmental Management* 9:253–262.

Hughes, R. M. 1995. Defining acceptable biological status by comparing with reference conditions. Pages 31–47 in W.S. Davis and T. P. Simon (editors). Biological assessment and criteria: tools for water resource planning and decision making for rivers and streams. Lewis Publishers, Boca Raton, Florida.

Hughes, R.M., D.P. Larsen, and J.M. Omernik. 1986. Regional reference sites: A method for assessing stream potentials. *Environmental Management* 10:629–635.

Johnson, K.N., S. Gordon, S. Duncan, D. Lach, B. McComb, K. Reynolds. 2007. Conserving creatures of the forest: a guide to decision making and decision models for forest biodiversity. Oregon State University, College of Forestry, Corvallis, Oregon. 88 pp.

Karr, J.R. 1981. Assessment of biotic integrity using fish communities. *Fisheries* 6(6): 21–27.

Karr, J.R., and D.R. Dudley. 1981. Ecological perspectives on water quality goals. *Environmental Management* 5:55–68.

Karr, J. R., K. D. Fausch., P. L. Angermeier, P. R. Yant, and I. J. Schlosser. 1986. Assessing biological integrity in running waters: a method and its rationale. Illinois Natural History Survey, Champaign, Illinois.

Karr, J.R., and E.W. Chu. 1995. Ecological integrity: reclaiming lost connections. In L. Westra and J. Lemons (eds.) Perspectives on Ecological Integrity, pp. 34–48. Kluwer Academic Publishers, Doredrecht, Netherlands.

Keane, R.E., R. Parsons, and P. Hessburg. 2002. Estimating historical range and variation of landscape patch dynamics: limitations of the simulation approach. Ecological Modelling 151:29–49.

Keane, R.E., P.F. Hessburg, P.B. Landres, and F.J. Swanson. 2009. The use of historical range and variability (HRV) in landscape management. *Forest Ecology and Management* 258:1025–1037

McGarigill, K., and B.J. Marks. 1995. FRAGSTATS: Spatial pattern analysis program for quantifying landscape structure. U.S. Forest Service General Technical Report PNW-GTR-351, U.S. Forest Service, Pacific Northwest Research Station, Portland, Oregon. 122pp.

McRae, B.H., and P. Beier. 2007. Circuit theory predicts gene flow in plant and animal populations. *Proceedings of the National Academy of Sciences of the USA* 104:19885–19890.

McRae, B.H., B.G. Dickson, T.H. Keitt, and V.B. Shah. 2008. Using circuit theory to model connectivity in ecology and conservation. *Ecology* 10:2712–2724.

Morgan, P., G.H. Aplet, J.B. Haufler, H.C. Humphries, M.M. Moore, and W.D. Wilson. 1994. Historical range of variability: a useful tool for evaluating ecosystem change. *Journal of Sustainable Forestry* 2:87–111.

Moya, N., R.M. Hughes, E. Dominguez, F-M Gibon, E Goita, and T. Oberdorff. 2011. Macroinvertebrate-based multimetric predictive models for measuring the biotic condition of Bolivian streams. *Ecological Indicators* 11:840–847.

Omernik, J. M. 1987. Ecoregions of the conterminous United States. *Annals of the Association of American Geographers* 77:118–125.

Paulsen, S. G., A. Mayio, D.V. Peck, J.L. Stoddard, E. Tarquinio, S.M. Holdsworth, J. Van Sickle, L.L. Yuan, C.P. Hawkins, A. Herlihy, P.R. Kaufmann, M.T. Barbour, D.P. Larsen, and A.R. Olsen. 2008. Condition of stream ecosystems in the US: an overview of the first national assessment. *Journal of the North American Benthological Society* 27:812–821.

Pimentel, D., L. Westra, and R. F. Noss. 2000. Ecological Integrity: Integrating Environment, Conservation, and Health. Island Press, Washington, DC.

Pont, D., B. Hugueny, U. Beier, D. Goffaux, A. Melcher, R. Noble, C. Rogers, N. Roset, and S. Schmutz. 2006. Assessing river biotic condition at the continental scale: a European approach using functional metrics and fish assemblages. *Journal of Applied Ecology* 43:70–80.

Pont, D., R.M. Hughes, T.R. Whittier, and S. Schmutz. 2009. A predictive index of biotic integrity model for aquatic-vertebrate assemblages of western U.S. streams. *Transactions of the American Fisheries Society* 138:292–305.

Reynolds, K.M. 1999. NetWeaver for EMDS version 2.0 user guide: a knowledge base development system. U.S. Forest Service General Technical Report PNW-GTR-471, U.S. Forest Service, Pacific Northwest Research Station, Portland, Oregon.

Reynolds, K.M. 2001. EMDS: Using a logic framework to assess forest ecosystem sustainability. *Journal of Forestry* 99(6) 26–30.

Roset ,N., G. Grenouillet, D. Goffaux, D. Pont, and P. Kestemont . 2007. A review of existing fish assemblage indicators and methodologies. *Fisheries Management and Ecology* 14:393-405.

Schussman, H., and E. Smith. 2006. Southwest forest assessment project. The Nature Conservancy. Denver, Colorado.

Spencer, W.D., P. Beier, K. Penrod, K. Winters, C. Paulman, H. Rustigian-Romsos, J. Strittholt, M. Parisi, and A. Pettler. 2010. California Essential Habitat Connectivity Project: A Strategy for Conserving a Connected California. Prepared for the California Department of Transportation, California Fish and Game, and Federal Highway Administration.

Staus, N.L., J.R. Strittholt, and D.A. DellaSala. 2010. Evaluating high conservation value in a portion of the Northwest Forest Plan: A practical application of a decision support model. *Conservation Biology* 24(3): 711–720.

Stoddard, J. L., D.P. Larsen, C.P. Hawkins, R.K. Johnson, and R.H. Norris. 2006. Setting expectations for the ecological condition of streams: the concept of reference condition. *Ecological Applications* 16:1267–1276.

Stoddard, J. L, A.T. Herlihy, D.V. Peck, R.M. Hughes, T.R. Whittier, and E. Tarquinio. 2008. A process for creating multi-metric indices for large-scale aquatic surveys. *Journal of the North American Benthological Society* 27:878–891.

Strittholt, J.R., N.L. Staus, G. Heilman, and J. Bergquist. 2007. Mapping high conservation value and endangered forests in the Alberta Foothills in using spatially explicit decision support tools. Report to Limited Brands. 66 pp.

Swetnam, T.W., C.D. Allen, and J.L. Betancourt. 1999. Applied historical ecology: using the past to manage for the future. *Ecological Applications* 9:1189–1206.

USGPO. 1989. Federal Water Pollution Control Act (33 U.S.C. 1251 et seq.) as amended by a P.L. 92-500. In Compilation of Selected Water Resources and Water Pollution Control Laws. U.S. Government Printing Office, Washington, DC.

White, M.D. and J.R. Strittholt. In review. Using EMDS in Forest Conservation Planning. Book Chapter**.**

Whitford, W.G.R.D.J., and A.G. deSoyza. 1999. Using resistance and resilience measurements for ‘fitness’ tests in ecosystem health. *Journal of Environmental Management* 57:21–29.

Whittier, T.R., D.P. Larsen, R.M. Hughes, C.M. Rohm, A.L. Gallant, and J.M. Omernik. 1987. The Ohio stream regionalization project: A compendium of results. EPA/600/3-87/025. Corvallis, OR: U.S. Environmental Protection Agency, Environmental Research Laboratory.

Whittier, T.R., R.M. Hughes, J.L. Stoddard, G.A. Lomnicky, D.V. Peck, and A.T. Herlihy. 2007a. A structured approach to developing indices of biotic integrity: three examples from western USA streams and rivers. *Transactions of the American Fisheries Society* 136:718–735.

Whittier, T.R., J.L. Stoddard, D.P. Larsen, and A.T. Herlihy. 2007b. Selecting reference sites for stream biological assessments: best professional judgment or objective criteria. *Journal of the North American Benthological Society* 26:349–360.

6 SPECIES AND COMMUNITY MODELING

6.1 Introduction

Note: According to BLM guidance we are to 1) use existing models wherever they are available with a full ecoregional extent or wherever they may readily be extended from a portion of the assessment region; 2) if existing models are not available, but occurrence data are available, we will use a modeling approach such as MaxEnt; 3) if occurrence data are lacking, we will use existing SW ReGAP models.

Species and community (referred to as simply species from here out) distribution models quantify associations between environmental variables and occurrence records to identify environmental conditions where a species is likely to be found (Pearson 2007). Species distribution modeling has become an important part of conservation planning and a valuable tool to assess impacts of environmental changes. A wide variety of techniques are available and commonly used. Steps in the species distribution modeling process, which will be briefly discussed, include: conceptualization, data preparation, model fitting, model evaluation, and spatial predictions (Figure 6-1).

6.2 Conceptualization

The first steps of species distribution modeling involve setting clear objectives, defining the study area, and creating a conceptual model based on the ecology of the species of interest. Potential environmental predictors must be identified, data availability assessed, and appropriate scale and resolution evaluated. Lastly, the most appropriate modeling algorithm needs to be identified and model evaluation methodology should be decided on.

6.3 Data Preparation

Species occurrence and environmental data must be collated and processed for input into species distribution models. Occurrence data may be from a single systematic survey or opportunistically collected from multiple sources (museum collection records, online data portals). Species occurrence data may include detections only (presence) or both detections and non-detections (presence-absence). Non-detection records may be difficult to obtain if no systematic survey data are available, and may be unreliable (species may be present but not detected). Pseudo-absence data may be generated from randomly located points across the study area if non-detection data are unavailable (Beauvais 2007). If the species of interest is known to vary habitat use by seasons or by gender, occurrence data may be divided by season or sex and modeled separately.

Potential sources of bias or error in occurrence data include species misidentification, inaccurate spatial referencing, historical records, and uneven sampling effort (Beauvais et al. 2006, Pearson 2007). When using occurrence data from sources other than systematic surveys, sampling bias will likely exist, where parts of the study area may be sampled intensively while others not at all and records may be clustered in easily accessible areas. Occurrence records may be filtered so that no two points are within a minimum user-defined species-specific distance (such as the radius of a typical home range) to avoid biasing model results towards heavily sampled environments (Beauvais et al. 2006).

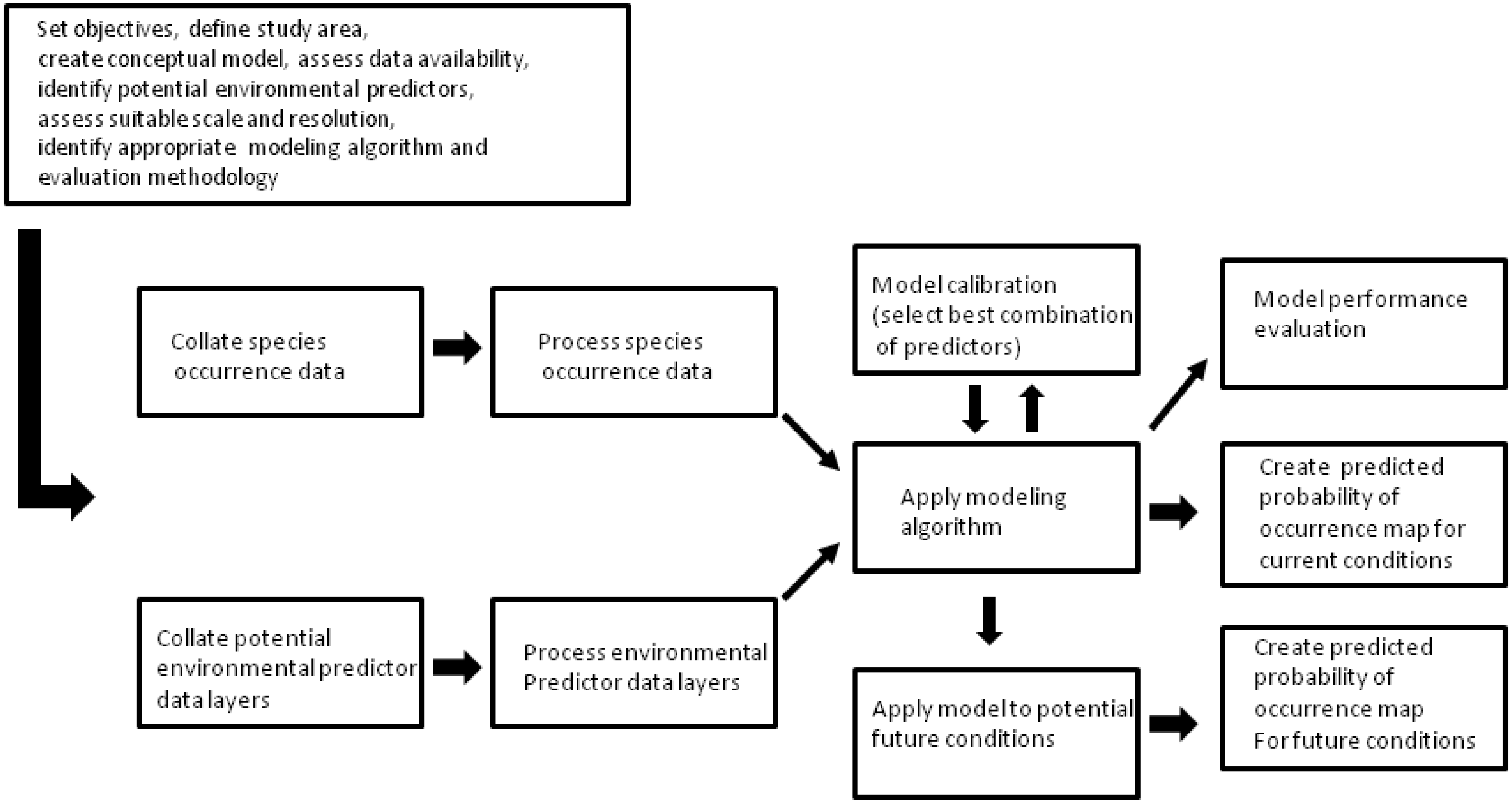


Figure 6-1. Flow diagram of species distribution modeling process (adapted from Pearson 2007).

Environmental predictor data layers should cover the study area extent of interest completely and correspond temporally with the occurrence data (Beauvais et al. 2006). The resolution (cell size or minimum mapping unit) of the species distribution model is often constrained by the environmental predictor data sources used (climate data is often available at 1 km, while land cover is often 30 m). Other considerations include the intended use of model output (site specific management or regional conservation planning), as well as processing constraints due to study area size and computing power.

The most commonly used types of environmental predictors in species distribution modeling are climatic, topographic, soils, and land cover (Pearson 2007). Environmental predictors can be divided into two classes: direct (having physiological influence on a species, such as temperature, water, or prey distribution) and indirect (lack a physiological effect, such as slope or elevation). It is preferable to base species distribution models on predictors which have a direct causal influence on species occurrence, however GIS data for these are uncommon and often less precise, resulting in the use of indirect variables as surrogates (Guisan and Zimmerman 2000).

Environmental data may be either continuous, such as temperature or precipitation, or categorical, such as land cover or soil type (Pearson 2007). Some modeling algorithms require continuous environmental predictors, requiring the transformation of categorical data. Categorical data may be summarized to create continuous data by using neighborhood functions in a GIS (percent forest), distance measures (for distance to the nearest stream), density measures (road density), or landscape metrics (largest patch index, edge density, proximity index). Summarizing environmental predictors in these ways has some advantages: mobile organisms may be responding to what is within their home range rather than what is found exactly where they were detected and small inaccuracies in the GIS data may be smoothed out; but the moving window size selection should be appropriate for the species of interest.

A priori knowledge of species ecology should be used to identify and combine key environmental predictor variables driving species distribution to construct candidate models. There are often multiple possible predictors which must be whittled down to a smaller subset of the most relevant variables. Visual assessment of variable maps, boxplots, and scatterplots are be useful in predictor selection as may be more formal variable reduction procedures such as principle components analysis (Beauvais et al. 2006).

6.4 Model Fitting

Many species distribution modeling algorithms are commonly used, these can be divided into several families: environmental envelopes, classification techniques, regression, maximum entropy, and other algorithms including ecological niche factor analysis, Bayesian techniques, ordination techniques and artificial neural networks (Table 6-1, Guisan and Zimmerman 2000, Guisan and Thuiller 2005, Beauvais et al. 2006, Pearson 2007). These methods differ in their data requirements, ease of use, transparency, interpretability, and predictive performance (Hernandez et al. 2006).

Environmental envelopes, such as BIOCLIM and DOMAIN, define a multidimensional environmental box for the set of occurrences and use presence-only data (Beauvais et al. 2006). These algorithms, while easy to explain and implement, are unable to incorporate interactions among predictors, weight all predictors equally, have no procedures for variable selection, and have been found to be sensitive to outliers and sampling bias (Hernandez et al. 2006).

Table 6-1. Some available published species distribution modeling packages. Classification techniques, such as CART, TREE, and Random Forests, use a discriminant process to successfully split presence and absence points resulting in a dichotomous tree showing a series of cut-points on environmental variables leading to suitable and unsuitable habitat (Beauvais et al. 2006). These approaches require either presence-absence occurrence data or the generation of pseudo absences if absence data is lacking, and are easy to interpret, able to incorporate interactions among predictors, and indicate the relative importance of predictors (Beauvais et al. 2006).

Method	**Software**	**Example References**	**URL**
Environmental Envelopes	BIOCLIM, DOMAIN	Lindenmayer et al. 1991, Beaumont et al. 2005; Carpenter et al. 1993	http://arcscripts.esri.com/details.asp?dbid=13745 http://www.cifor.cgiar.org/online-library/research-tools/domain.html
Classification Techniques (CART, Random Forests)	R	Vayssieres et al. 2000, Cutler et al. 2007	http://www.r-project.org/
Regression (GLM, GAM)	R	Guisan et al. 2002	http://www.r-project.org/
Maximum Entropy	MaxEnt	Phillips et al. 2006, Elith et al. 2011	http://www.cs.princeton.edu/~schapire/maxent/
Ecological Niche Factor Analysis	BIOMAPPER	Hirzel et al. 2001, Hirzel et al. 2002	http://www2.unil.ch/biomapper/
Multiple Methods	BIOMOD	Thuiller 2003	http://www.will.chez-alice.fr/Software.html

Regression has been used extensively to predict species distribution due to its strong ecological foundation and strong performance (Elith et al. 2006, Albert and Thuiller 2008). Multiple logistic regressions relate a binary response variable (presence or absence) to a combination of environmental predictors and represent the relationship as a linear function. Generalized additive models (GAM) are more flexible due to the use of non-parametric smoothers to fit non-linear functions, making them more capable of modeling complex ecological responses (Elith et al. 2006). While both logistic regression and GAMs are easy to implement, GAMs are more difficult to interpret. Both can incorporate interactions, have variable reduction procedures, and allow for the investigation of variable importance. Absence data are required and results are sensitive to the ratio of presence to absence points.

Maximum Entropy (MaxEnt) is a statistical mechanics approach that makes predictions from incomplete information (Hernandez et al. 2006). MaxEnt "minimizes the relative entropy between two probability densities (one estimated from the presence data and one, from the landscape) defined in covariate space" (Elith et al. 2011). It uses presence-only data, is very easy to implement, incorporates interactions, evaluates variable importance, and has been found to be high performing, even with small sample sizes (Elith et al. 2006, Hernandez et al 2006). However, it is difficult to interpret and provides no procedure for variable selection.

6.5 Model Evaluation

Many methods of evaluating or validating the predictive performance of a species distribution model are found. The most appropriate method will depend on the modeling method used, data availability, and project goals (Pearson 2007). Ideally, data to test the model is collected independently from the data used to build the model, but often this is not possible. Another option is to divide the species occurrence data

before fitting, with 75% of the records used as the training set and the other 25% reserved for use as a test set. Iterative methods, such as K-fold partitioning are a useful approach to model evaluation, especially when the occurrence dataset is small. This involves dividing the dataset into *k* parts (often 5 or 10) and each part is used as a test set while the model is built from the other *k*-1 sets. Validation statistics are reported as the mean and standard deviation from the set of *k* tests (Pearson 2007). For very small datasets (less than 20), jackknifing, or 'leave one out' partitioning is recommended, where each individual occurrence record is excluded from model building during one partition (Pearson 2007).
There are many metrics used to quantify model performance. Most, such as accuracy, sensitivity, specificity, and Kappa, require the selection of a threshold value to divide the model output into 'suitable' and 'unsuitable' habitat (Beauvais et al. 2006). A commonly used threshold-independent metric is the area under the receiver operating characteristic curve (ROC AUC), which provides a single measure of predictive performance across the full range of possible thresholds (Pearson 2007).

6.6 Spatial Predictions

Methods used to convert species distribution modeling outputs into spatial predictions (GIS layers of predicted probability of occurrence) vary depending on the algorithm and software used. Some, such as BIOCLIM and MaxEnt, produce a GIS layer as part of the process, while others, such as GAM, do not and require an amount of GIS expertise to create. There is similar variability in ease of projecting model outputs onto future conditions (for example, reflecting land use or climate change).

6.7 Limitations and Caveats

There is no one single best approach or method to species distribution modeling. The most appropriate method depends on the data available and project assumptions and goals (Segurado and Araujo 2004). Several factors have been found to influence model performance. Data quality of both species occurrence locations and environmental predictors strongly affects model performance ("garbage in, garbage out") and small errors can propagate, resulting in larger unpredictable errors. The selection of appropriate spatial scale and resolution affect model performance (Thuiller et al. 2003) as do the number of occurrence records, and the range size and ecological niche breadth of the organism of interest (Hernandez et al. 2006). The adequacy of the predictor data layers used also influence model performance, with poorly predictive models resulting if key driving factors of species distribution patterns are overlooked (Hernandez et al. 2006). The conceptual basis for the model must reflect ecological reality for the model to be successful.

Outputs of species distribution models carry a high potential for misuse. Maps of predicted probability of occurrence are spatial models or estimates with associated uncertainty, not direct representations of 'truth'. Quantifying and effectively communicating the uncertainty in model results as well as appropriate uses given the resolution of the model predictions is highly important to prevent model misinterpretation and misuse (Beauvais et al. 2006).

Many of the conservation elements identified as of interest in the REA process are species or communities. Where available, we will include species that have been mapped using the types of modeling approaches reviewed in this section. If unavailable, we will choose a representative sample of species where occurrence records are abundant to derive a draft occurrence dataset.

The following examples are provided to review the types of considerations and modeling approaches that may be used to address species and community-level issues in the REA – golden eagle (mammal species), desert bighorn sheep (mammal species), and desert tortoise (reptile species).

6.8 References Cited

Albert, C.H., and W. Thuiller. 2008. Favourability functions versus probability of presence: advantages and misuses. *Ecography*: 31, 417–422.

Beaumont, L.J., L. Hughes, and M. Poulsen. 2005. Predicting species distributions: use of climatic parameters in BIOCLIM and its impact on predictions of species' current and future distributions. *Ecological Modeling* 186: 250–269.

Beauvais, G.P., D.A. Keinath, P. Hernandez, L. Master, and R. Thurston. 2006. Element distribution modeling: a primer. http://www.natureserve.org/prodServices/pdf/EDM_white_paper_2.0.pdf. Accessed 18 December 2007.

Carpenter G., A.N. Gillson, and J. Winter. 1993. DOMAIN: a flexible modeling procedure for mapping potential distributions of plants and animals. *Biodiversity and Conservation* 2: 667–680.

Cutler, D.R., T.C. Edwards, K.H. Beard, A. Cutler, K.T. Hess, J. Gibson, and J.J. Lawler. 2007. Random forests for classification in ecology. *Ecology* 88:2783–2792.

Elith, J., C. H. Graham, R.P. Anderson, M. Dudik, S. Ferrier, A. Guisan, R.J. Hijmans, F.Huettmann, J.Leathwick, A. Lehmann, J.Li, L.G. Lohmann, B.A. Loiselle, G. Manion, C. Moritz, M. Nakamura, Y. Nakazawa, J. M. Overton, A.T. Peterson, S.J. Phillips, K.S. Richardson, R. Scachetti-Pereira, R.E. Schapire, J.Soberon, S. Williams, MS. Wisz, and N.E. Zimmermann . 2006. Novel methodsimprove prediction of species' distributions from occurrence data. *Ecography* 29:129–151.

Elith, J., S.J. Phillips, T.Hastie, M. Dudik, Y.E. Chee, and C. J. Yates. 2011. A statistical explanation of MaxEnt for ecologists. *Diversity and Distributions* 17:43–57.

Guisan, A., T.C. Edwards, and T. Hastie. 2002. Generalized linear and generalized additive models in studies of species distributions: setting the scene. *Ecological Modeling* 157: 89–100.

Guisan, A., and W. Thuiller. 2005. Predicting species distribution: offering more than simple habitat models. *Ecology Letters* 8:993–1009.

Guisan, A., and N.E. Zimmerman. 2000. Predictive habitat distribution models in ecology. *Ecological Modeling* 135:147–186.

Hernandez, P.A., C.H. Graham, L.L. Master, and D.L. Albert. 2006. The effect of sample size and speciescharacteristics on performance of different species distribution modeling methods. *Ecography* 29:773–785.

Hirzel, A.H.; J. Hausser; D. Chesser; and N. Perrin. 2002. Ecological-Niche factor analysis: How to compute habitat-suitability maps without absence data? *Ecology* 83: 2027–2036.

Hirzel, A.H.; V. Helfer; and F. Metral. 2001. Assessing habitat-suitability models with a virtual species. *Ecological Modeling* 145:111–121.

Lindenmayer, D.B., H.A. Nix, J.P. McMahon, M.F. Hutchinson, and M.T. Tanton. 1991. The conservation of Leadbeater's possum, Gymnobelideus leadbeateri (McCoy): A case study of the use of bioclimatic modelling. *Journal of Biogeography* 18:371–383.

Pearson, R.G. 2007. Species' distribution modeling for conservation educators and practitioners. Synthesis. American Museum of NaturalHistory. <http://biodiversityinformatics.amnh.org/files/SpeciesDistModelingSYN_1-16-08.pdf>. Accessed 14 December 2010.

Phillips, S.J., R.P. Anderson, and R.E. Schapire. 2006. Maximum entropy modeling of species geographic distributions. *Ecological Modeling* 190: 231–259.

Segurado, P., and M.B. Araujo. 2004. An evaluation of methods for modeling species distributions. *Journal of Biogeography* 31:1555–1568.

Thuiller, W. 2003. BIOMOD - optimizing predictions of species distributions and projecting potential future shifts under global change. *Global Change Biology* 9:1353–1362.

Vayssieres, M.P., R.E. Plant, and B.H. Allen-Diaz. 2000. Classification trees: An alternative non parametric approach for predicting species distributions. *Journal of Vegetation Science* 11: 679–694.

6.9 Terrestrial Wildlife Species Conservation Element: Golden Eagle (*Aquila chrysaetos*)

Golden eagles occur across all of North America, in western Europe, Eurasia, and parts of Africa, but they have recently begun to decline in the western U.S. and other areas experiencing increased population growth and development. They are vulnerable to environmental change, especially from human development and changes to habitat. This module describes a methodology to assess the current distribution of potential golden eagle habitat in the Sonoran Desert ecoregion, change agents affecting habitat distribution, and areas where eagles are at risk both in the near- and long-term.

6.9.1 Introduction

The purpose of this assessment is to design a technical approach to address the status of golden eagles (*Aquila chrysaetos*) in the Sonoran Desert ecoregion. The following species-related management questions were identified by the Assessment Management Team (AMT) for inclusion in the Sonoran Desert Rapid Ecoregional Assessment (REA):

1. What is the most current distribution of available occupied habitat (and historic occupied habitat if available), including breeding, seasonal habitat, and movement corridors (as applicable)?
2. What areas known to have been surveyed and what areas have not been surveyed (i.e., data gap locations)?
3. Where are potential habitat restoration areas?
4. Where are potential areas to restore connectivity?
5. What/where is the vulnerability to change of species to change agents in the near-term horizon, 2020 (development, fire, invasive species) and a long-term change horizon, 2060 (climate change)?

Change agents selected for the REA and considered in this analysis include: wildland fire, human development, resource uses, invasive plant species, and climate change.

The golden eagle inhabits open spaces in western North America that provide hunting habitat, often near cliffs and ridges where the birds prefer to nest (Kochert et al. 2002). The birds feed primarily on small to medium-sized mammals, principally hares and rabbits (Olendorff 1976, Eakle and Grubb 1986, Glinski 1998). Golden eagles breed and forage in open and semi-open habitats including grasslands and shrublands, and they avoid heavily forested areas (Poole and Bromley 1988). They usually nest on cliffs although they will utilize human-made structures such as windmills, electrical transmission towers, and nesting platforms (Kochert et al. 2002). Golden eagles are short- to medium-distance partial migrants with individuals from northern breeding areas migrating longer distances than southern nesters (Mead 1973). Little is known of migratory routes.

Long-term surveys indicate population declines in the western U.S. (Kochert and Steenhof 2002). Although the golden eagle is not listed as threatened or endangered, the eagles and their nests have been protected since 1962 by the Bald and Golden Eagle Protection Act. However, Native Americans are permitted to take and possess eagles and their parts for religious purposes (Kochert et al. 2002).

6.9.2 Conceptual Model

The conceptual model (Figure 6-2) illustrates the relationships between natural population drivers, change agents, and golden eagle populations. Changes caused by development and resource use, climate change, and altered fire regime affect eagle habitat. Alteration of open shrubland or grassland habitat through development or conversion to agriculture has a negative effect on eagle populations due to its effect on nest availability and prey populations, particularly black-tailed jackrabbit.

The major reason for the decline of golden eagles is habitat destruction through development and direct take; humans cause over 70% of recorded deaths, either directly or indirectly, through collisions with vehicles and power lines, electrocution, poisoning, and shooting (Franson et al. 1995). Habitat destruction due to land development has led to large-scale population declines in some areas (Kochert and Steenhof 2002). Although fire in forested ecoregions of the southeastern U.S. has enhanced the prey base for golden eagles and increased their hunting efficiency (Landers 1987), wildfires in the western U.S. have led to loss of shrubs and resident black-tailed jackrabbit (*Lepus californicus*), a major food item for golden eagles in this area (Kochert et al. 1999). Large-scale shrub loss due to wildfires in southwestern Idaho led to lower golden eagle reproductive success for 4-6 years postburn (Kochert et al. 1999).

Contaminants and human-caused eagle deaths affect golden eagle individuals and populations directly. Eagles are often the victims of secondary poisoning when they consume prey that have been killed or sickened by pesticides, herbicides, or rodenticides (Franson et al. 1995). Eagles may also survive with elevated blood-lead levels from consuming prey items that are contaminated with lead or from directly ingesting lead shot and bullet fragments.

Human-made infrastructure such as power lines and wind turbines are also responsible for eagle mortality. In the Altamont Pass Wind Resource Area in west-central California, Smallwood and Thelander (2007) estimated 67 golden eagle fatalities per year due to wind turbines; sub-adults and floaters appeared to be affected disproportionately (Hunt 2002). Golden eagle fatalities were correlated with turbine height and location, and topography with the majority of deaths associated with shorter turbines (e.g. Type-13), end of row and second from end turbines, and dips and notches in topography (Curry and Kerlinger 1998, Hunt 2002). Collision with vehicles, power lines and other structures is the leading cause of known deaths followed by electrocution when landing on power poles (Franson et al. 1995). Although they are protected under the Bald and Golden Eagle Protection Act, golden eagles are sometimes illegally shot when suspected of killing livestock.

It is not known how climate change will affect this species. The potential consequences are related to how climate change may directly affect shrub and grassland habitats or indirectly affect them through altered fire regimes and distribution of invasive plants, both of which may affect prey populations. For example, if climate change leads to more widespread fire, this could lead to the loss of shrubs and a decline in small mammal populations which could negatively affect eagle populations (Kochert et al. 1999). However, its broad latitudinal range in North America (from Mexico to the Arctic) and generalist habits make the golden eagle a poor candidate to model the effects of climate change. For this reason, we will concentrate on the near-term development effects on golden eagles (2025).

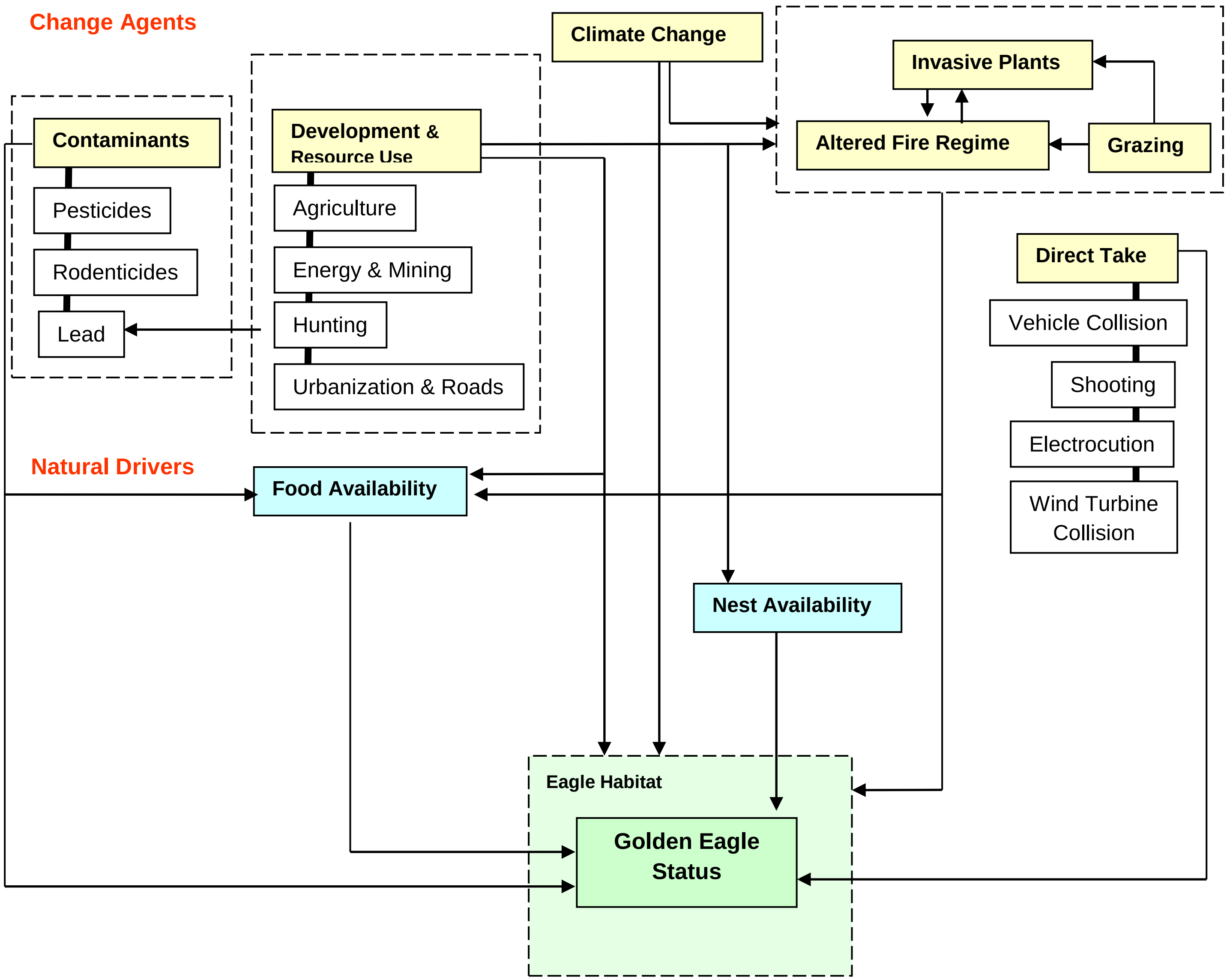


Figure 6-2. Principle interactions among population drivers and change agents for golden eagles in the Sonoran Desert.

6.9.3 Required Input Data Layers

To answer the management questions, we need to know the current potential distribution of golden eagles and where they are likely to be located in the future. Data gaps exist for nest site locations, migration routes, and dispersal patterns.

Table 6-2. Raw data layers necessary for answering the conservation management questions for golden eagles. Data layers with asterisks indicate data that may not be readily available.

TENTATIVE DATA NEEDS	DATA CLASS	MANAGEMENT QUESTION
NHP EO's	SPECIES OCCURRENCES	1, 2
DEM	TOPOGRAPHY	1, 3, 4
*Golden Eagle nest sites	SPECIES OCCURRENCES	1
Human footprint	DEVELOPMENT	1
Road Density	DEVELOPMENT	1
Survey locations	SPECIES OCCURENCES	2
Land use planning areas	DEVELOPMENT – FUTURE	5
Human Population growth	DEVELOPMENT – FUTURE	5
*Identified movement corridors	HABITAT	1, 4
*Identified seasonal habitats	HABITAT	1
Fire perimeters	HABITAT	1, 3
SWreGAP vegetation	VEGETATION	1, 3
Grazing pressure	RESOURCE USE	1, 3

BLM plans to conduct surveys for golden eagles in Arizona during 2011, with data becoming available by summer (discussion at Sonoran REA Task 3 meeting, 27 January 2011). These surveys will include a substantial portion of the Sonoran ecoregion. If these data are available, they will probably provide the best input data for eagle distribution maps and models that are prepared for the Sonoran Desert ecoregion. In the California Sonoran Desert data may be available from surveys conducted to monitor the suitability of renewable energy project locations.

6.9.4 Model Assumptions

In our process model, we make the following model assumptions:

1. Eagles tend to center their hunting territory around their nest sites;
2. because eagles need thermals for soaring flight, they are found in the neighborhood of ridges and mountain ranges;
3. eagles are sensitive to human disturbance and avoid disturbed areas and agricultural lands;
4. eagles use shrubland habitat preferentially.

6.9.5 Attributes and Indicators

There are a number of potential attributes in the table below that could be utilized for addressing management questions 1–3 and 5. (Connectivity question, MQ #4, is not appropriate for golden eagle.):

1. What is the current distribution of occupied habitat, including breeding and seasonal habitat, and movement corridors?

   Attributes in the table that deal with habitat and nest sites will populate the model that will answer this management question.

2. What areas known to have been surveyed and what areas have not been surveyed (i.e., data gap locations)?

   BLM plans surveys for golden eagles in Arizona during 2011, with data becoming available by summer.

3. Where are potential habitat restoration areas?

   Areas where habitat has been degraded could be considered for potential restoration. For example, in the Altamont area of California, habitat for golden eagles was restored by rescinding the poisoning of ground squirrels on various parcels of land to increase ground squirrel populations and lure eagles away from concentrations of wind turbines. However, solutions such as these are often at a scale that is finer than the ecoregional scale of the REAs.

4. MQ #5. What/where is the vulnerability to change of species to change agents in the near-term horizon, 2020 (development, fire, invasive species) and a long-term change horizon, 2060 (climate change)?

   Human development (including agriculture), fire, and energy development are the main drivers of potential changes in eagle populations in the next decade.

Species	Key Ecological Attribute	Indicator	Indicator rating				Basis for Indicator Rating	Comments
			Poor	Fair	Good	Very Good		
Golden Eagle	habitat loss or degradation	urban development	present	--	minimal	absent	Kochert and Steenhof (2002)	Large-scale declines of nesting eagles in San Diego County between 1956-80 were related to extensive residential development.
Golden Eagle	habitat degradation	livestock grazing and agriculture	existing or planned	--	--	absent	Beecham and Kochert (1975)	Extensive agricultural development reduces jackrabbit populations and makes areas less suitable for nesting and wintering eagles
Golden Eagle	habitat degradation	fire	>40,000 ha of shrublands burned	--	burned territory; adjacent vacant unburned	unburned territories	Kochert et al. (1999)	Large-scale shrub loss due to wildfires in southwestern Idaho led to lower golden eagle reproductive success for 4-6 years postburn; fire suppression is recommended in areas where much shrub habitat has been lost to fire
Golden Eagle	habitat degradation	mining and energy development	present	--	--	absent	Phillips and Beske (1982)	Mining and various types of energy development occur in eagle nesting and wintering habitat. Surface coal mines threaten limited nesting sites in Wyoming (Phillips and Beske 1982).
Golden Eagle	habitat	vegetation	disturbed areas, grasslands, agriculture			shrubland	Marzluff et al. (1997), Peterson (1988)	Eagles in SW Idaho selected shrub habitats and avoided disturbed areas, grasslands, and agriculture; in n. Utah, nests mainly in grass, shrub, and juniper (*Juniperus* spp.) habitats
Golden Eagle	habitat/nest sites	topography	--	--	--	cliffs within 7 km of shruband	Menkens and Anderson (1987), McGrady et al. (2002), Cooperrider et al. (1986)	Usually nests on cliffs in proximity to hunting grounds; over 98% of eagle observations in western Scotland were <6 km from the nest site; During the nesting season the golden eagle usually forages within 4.4 miles (7 km) of the nest
Golden Eagle	mortality	infrastructure (roads, power lines, wind turbines	--	--	--	infrastructure absent	Franson et al. (1995)	Humans cause over 70% of recorded deaths, either directly or indirectly, through collisions with vehicles and power lines, electrocution, poisoning, and shooting
Golden Eagle	illness/mortality	poisoning from pesticides and other toxins	high levels of contaminants	--	--	low/no contaminants	Craig and Craig (1998), Franson et al. (1995), Harmata and Restani (1995), Kramer and Redig (1997), Pattee et al. (1990)	Eagles are often the victims of secondary poisoning when they consume prey that have been killed or sickened by pesticides, herbicides, or rodenticides (Franson et al. 1995). Elevated blood-lead levels (>0.20 ppm) occurred in 36% of 162 eagles from s. California, 1985–1986 (Pattee et al. 1990), 46% of 281 wintering eagles from Idaho, 1990–1997 (Craig and Craig 1998), and 56% of 86 spring migrants in Montana, 1985–1993 (Harmata and Restani 1995). Chronic subclinical lead exposure may weaken eagles and predispose them to injury, predation, starvation, disease, or reproductive failure (Kramer and Redig 1997, Craig and Craig 1998).
Golden Eagle	mortality	shooting	Occurs	--	--	doesn't occur	Beans (1996)	Traditionally shot in parts of North America where depredation of domestic sheep was suspected. In 1971, >500 killed in Colorado and Wyoming by helicopter gunmen hired by sheep ranchers (Beans 1996). Illegal shooting continues to occur; no information on recent trends or levels.

6.9.6 Methods and Tools

A species distribution model will be created based on the PAT (Predicting Aquila Territory) model developed for golden eagles in western Scotland (McLeod et al. 2002a). The model predicts suitable habitat based on weighted mean nest site location in up to the last ten years, Thiessen polygons based on distance between neighboring range centers or 6 km radius if this isn't known, a preference for ridge features, and avoidance of areas with human activity or unsuitable habitat. This model appeared to be robust based on a good agreement between observed and predicted use patterns (McLeod et al. 2002b). If nest site locations are unavailable, it may be possible to predict habitat use based on the intersection of ridges and shrub habitat within 6 km. The results of this model will answer part of management question 1 regarding the current distribution of potentially occupied habitat, although it will not answer the part of the question about movement corridors.

Figure 6-3 illustrates the process model for golden eagles that will be developed in ArcGIS utilizing existing GIS data sets. If nest site locations are not available, it may be possible to develop the model based solely on terrain and suitable habitat.

To identify areas of near-term risk due to human development, we will use datasets that depict population projections and land use planning areas so we can estimate areas of future development and predicted locations of future energy development such as wind farms or transmission corridors. Eagle populations will be at risk of decline where areas of future development overlap current golden eagle distribution. This mapping exercise will answer management question 5 regarding potential for future change to the population in the near term. Potential changes in populations due to climate change will not be examined based on the reasons discussed in Section 6.9.2.

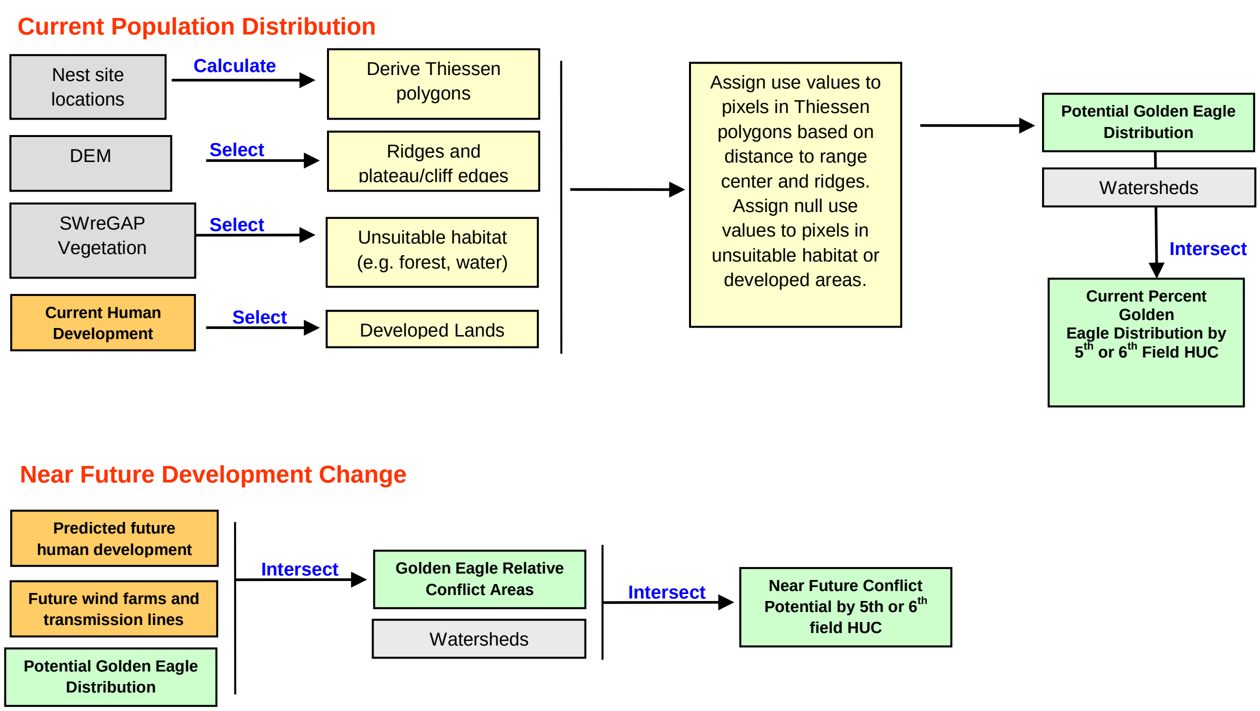


Figure 6-3. Datasets and processing steps for development of a golden eagle distribution model

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6.9.7 References Cited

Beans, B. E. 1996. Eagle's plume. Scribner, New York.

Beecham, J. J., and M. N. Kochert. 1975. Breeding biology of the Golden Eagle in southwestern Idaho. *Wilson Bulletin* 87:506–513.

Cooperrider, A.Y., R.J. Boyd, and H.R. Stuart (eds.). 1986. Inventory and monitoring of wildlife habitat. U.S. Bureau of Land Management, Service Center, Denver, Colorado. 858 p.

Craig, E. H., and T. H. Craig. 1998. Lead and mercury levels in golden and bald eagles and annual movements of golden eagles wintering in east central Idaho 1990–1997. U.S. Bureau of Land Management, Idaho State Office, Boise, Idaho.

Curry, R. C., and P. Kerlinger. 1998. Avian mitigation plan: Kenetech model wind turbines, Altamont Pass Wind Resource Area, California. Prepared for Kenetech Windpower, San Francisco, California.

Eakle, W.L., and T.G. Grubb. 1986. Prey remains from Golden Eagle nests in central Arizona. *Western Birds* 17: 87–89.

Glinski, R. L. 1998. The Raptors of Arizona. University of Arizona Press, Tucson, Arizona.

Hunt, G. 2002. Golden eagles in a perilous landscape: Predicting the effects of mitigation for wind turbine blade-strike mortality. Public Interest Energy Research P500-02-043F, California Energy Commission, Sacramento, California. 72 p.

Franson, J. C., L. Sileo, and N. J. Thomas. 1995. Causes of eagle deaths. Page 68 *in* Our living resources, LaRoe, E. T., G. S. Farris, C. E. Puckett, P. D. Doran, and M. J. Mac (eds.). U.S. Department of the Interior, National Biological Service, Washington, D. C.

Harmata, A. R., and M. Restani. 1995. Environmental contaminants and cholinesterase in blood of vernal migrant bald and golden eagles in Montana. *Intermountain Journal of Science*. 1:1–15.

Kochert, M. N., and K. Steehnoff. 2002. Golden Eagles in the U.S. and Canada: Status, trends, and conservation challenges. *Journal of Raptor Research* 36 (supplement):33–41.

Kochert, M. N., K. Steenhof, L. B. Carpenter, and J. M. Marzluff. 1999. Effects of fire on Golden Eagle territory occupancy and reproductive success. *Journal of Wildlife Management* 63:773–780.

Kochert, M. N., K. Steenhof, C. L. Mcintyre, and E. H. Craig. 2002. Golden Eagle (*Aquila chrysaetos*), The Birds of North American Online, A. Poole (ed.), Cornell Laboratory of Ornithology, Ithaca, New York. http://bna.birds.cornell.edu.proxy.library.oregonstate.edu/bna/species/684 .

Kramer, J.L., and P. T. Redig. 1997. Sixteen years of lead poisoning in eagles, 1980–95: an epizootiologic view. *Journal of Raptor Research* 31:327–332.

Landers, J.L. 1987. Prescribed burning for managing wildlife in southeastern pine forests. Pages 19–27 *in* Dickson, J.G., and O.E.Maughan (eds.). Managing southern forests for wildlife and fish: a proceedings; U.S. Forest Service General Technical Report SO-65. U.S. Forest Service, Southern Forest Experiment Station, New Orleans, LA.

Marzluff, J. M., S. T. Knick, M. S. Vekasy, L. S. Schueck, and T. J. Zarriello. 1997. Spatial use and habitat selection of golden eagles in southwestern Idaho. *Auk* 114:673–687.

McGrady, M.J., J.R. Grant, I.P. Bainbridge, and D.R.A. McCleod. 2002. A model of golden eagle (*Aquila chrysaetos*) ranging behavior. *Journal of Raptor Research* 36 (1 Supplement):62–69.

McLeod, D.R.A., D.P. Whitfield, A.H. Fielding, P.F. Haworth, and M.J. McGrady. 2002a. Predicting home range use by golden eagles (*Aquila chrysaetos)* in western Scotland. *Journal of* ***Avian Science*** 2:183–198.

McLeod, D.R.A., D.P. Whitfield, and M.J. McGrady. 2002b. Improving prediction of golden eagle (*Aquila chrysaetos*) ranging in western Scotland using GIS and terrain modeling. *Journal of RaptorResearch* 36 (1 Supplement):70–77.

Mead, C. J. 1973. Movements of British raptors. *Bird Study* 20:259–286.

Menkens, Jr., G. E., and S. H. Anderson. 1987. Nest site characteristics of a predominantly tree-nesting population of golden eagles. *Journal of Field Ornithology* 58:22–25.

Olendorff, R. R. 1976. The food habits of North American Golden Eagles. *American Midland Naturalist* 95:231-236.

Pattee, O. H., P. H. Bloom, J. M. Scott, and M. R. Smith. 1990. Lead hazards within the range of the California Condor. *Condor* 92:931–937.

Phillips, R. L., and A. E. Beske. 1982. Golden Eagles and coal development in the eastern Powder River Basin of Wyoming, Annual Report for 1982. U.S. Fish and Wildlife Service. Sheridan, Wyoming.

Peterson, D. L. 1988. Nesting and habitat parameters for selected raptors in the desert of northern Utah. Master’s Thesis, Utah State Univ., Logan, UT.

Poole, K. G., and R. G. Bromley. 1988. Interrelationships within a raptor guild in the central Canadian arctic. *Canadian Journal of Zoology* 66:2275–2282.

Smallwood, K. S., and C. Thelander. 2008. Bird mortality in the Altamont Pass Wind Resource Area, California. *Journal of Wildlife Management* 72(1):215–223.

6.10 Terrestrial Wildlife Species Conservation Element: Desert Bighorn Sheep (*Ovis canadensis* spp.)

6.10.1 Introduction

This module describes methodology to assess the current status and future condition of potential desert bighorn sheep (*Ovis canadensis spp.)* habitat in the Sonoran Desert, change agents affecting present and future distribution, where the population is at risk, and where opportunities for restoration may exist.

Note: California Department of Fish and Game has a desert bighorn sheep model in progress for the California Sonoran Desert, but the details are not available at this time. The BLM National Operation Center (NOC) has a bighorn sheep data layer that we have not yet received. It is unknown what the NOC data include (e.g., existing model, activity polygons, or occurrence data). The data used in Figure 6-5 below includes SW ReGAP habitat suitability models (that under-represent occupied habitat in the Arizona Sonoran Desert), critical habitat data in California (a subset of occupied habitat) and polygons developed from field knowledge and expert judgment in the Arizona Sonoran Desert. The modeling process described for the desert bighorn in this module is a standard MaxEnt approach. We will use the proposed approach if occurrence data is available and if other bighorn sheep models are not available.

Desert bighorn sheep were selected as a species conservation element because of the species' large area requirements, vulnerability to anthropogenic disturbance or threats associated with change agents, functional contributions to the ecological system, and relative socio-economic importance.

The BLM is conducting Rapid Ecoregional Assessments (REAs) across a number of ecoregions in the western U.S. to assess the current status of selected ecological resources and possible alterations to future conditions. Spatial analysis will address the following species-related management questions that were identified by the Assessment Management Team (AMT) for inclusion in the Sonoran Desert Rapid Ecoregional Assessment (REA):

1. What is the most current distribution of available occupied habitat (and historic occupied habitat if available), including breeding, seasonal habitat, and movement corridors and bottlenecks (as applicable)?
2. What areas are known to have been surveyed and what areas have not been surveyed (i.e., data gap locations)?
3. Where are potential habitat restoration areas?
4. Where are potential areas to restore connectivity?
5. What terrestrial species are vulnerable to change agents in the near term horizon, 2020 (development, fire, invasive species) and a long-term change horizon, 2060 (climate change)? Where are these species located?

Change agents selected for the REA and considered in this analysis include: wildland fire, human development, resource uses including recreation, invasive plant species, and climate change.

Desert bighorn sheep are habitat specialists, closely associated with rugged, mountainous terrain. In desert regions they occupy insular mountain ranges separated by intermountain flats (Singer et al. 2000). Following large declines and numerous local extinctions over the past century, bighorn sheep have disappeared from much of their former range in the western US (Buechner 1960, Johnson and Swift 2000); presently desert bighorns occur in small, isolated populations. Desert bighorn sheep are characterized by small population sizes, low dispersal rates, and naturally fragmented distributions (Epps et al. 2004). The small and isolated nature of bighorn sheep populations makes them particularly vulnerable to environmental change. Connectivity is considered crucial for the persistence of desert bighorn sheep, to facilitate daily and seasonal movements within subpopulations, required due to the region's limited water and forage, as well as larger movements between subpopulations to allow for genetic exchange and colonization (Bunn et al. 2007). The Sonoran Desert is home to the federally endangered Peninsular Range bighorn sheep, a distinct segment of the larger desert bighorn sheep population which is restricted to the Peninsular mountain ranges of California (Bunn et al. 2007). Peninsular bighorn sheep numbers have been in steady decline since the 1970s due to habitat loss, degradation and fragmentation, disease, and predation (Bunn et al. 2007).

6.10.2 Conceptual Model

The conceptual model (Figure 6-4) depicts the relationships between natural population drivers, change agents, and desert bighorn sheep populations. Desert bighorn sheep generally occur in small fragmented groups (subpopulations) characterized as metapopulations (Schwartz et al. 1986, Bleich et al. 1990, Bleich et al. 1996). They require anthropogenic-barrier-free corridors between summer and winter ranges (and between populations). Anthropogenic barriers, such as roadways, canals, or fencing are known to block the movement of desert bighorn sheep between seasonal ranges and subpopulations (Bleich et al. 1996, Epps et al. 2005).

Desert bighorn sheep prefer to feed in open areas with low-growing vegetation in close proximity to steep and rugged terrain, which is used for escape from predators, lambing, and bedding (California Department of Fish and Game 2010). Desert bighorns forage on a wide variety of plants, but they prefer succulent grasses and forbs in open habitats such as rocky barrens, meadows, and low, sparse brush, in close proximity to rugged terrain (California Department of Fish and Game 2010). Males and females maintain separate herds except during the mating season, with female bighorn sheep using steeper and more rugged terrain than male bighorn sheep (Bleich et al. 1996, Sappington et al. 2007). Desert bighorn sheep migrate between summer and winter ranges (California Department of Fish and Game 2010).

Human development (urban, commercial, residential, and energy), including associated activities and infrastructure such as roads, livestock grazing, recreation, fire suppression, and water extraction, threatens Sonoran desert bighorn sheep across its entire range. Critical factors for persistence of desert bighorn sheep include access to dependable water sources, protection from disease and excessive predation or hunting, and maintenance of connectivity within and between subpopulations. Several change agents are included in the conceptual model that can influence these factors (Figure 6-4):

1. Human development and resource use
    a. Agriculture, urban, exurban, and rural development and energy development are associated with increases in infrastructure (fences, highways, canals, aqueducts, rural residential, recreation facilities) resulting in habitat disturbance, loss and fragmentation, as well as direct mortality from road kill (Bleich et al. 1996, Dunn 1996, Krausman et al. 2004, Epps et al. 2005).
    b. Agriculture, urban, exurban, and rural development are associated with increases in water extraction, resulting in reduced availability of spring water, increased competition for surface water, and decreased viability (Bunn et al. 2007).
    c. Grazing increases the potential for contact with domestic livestock, subsequent disease transmission, and increased mortality (Foreyt and Jessup 1982, Jessup 1985, Smith et al. 1991, Gross et al. 2000, Singer et al. 2000).
    d. Grazing may increase competition for forage (Hansen 1982).

e. Recreation, such as off-highway vehicle use, rock climbing, and hiking, are associated with habitat disturbance (Thompson et al. 2007). Krausman et al. (2004) report recreation use as one of the reasons for the extirpation of desert bighorn sheep from the Santa Catalina Mountains.

f. Development is associated with wildfire suppression which has resulted in the encroachment of denser taller vegetation into bighorn sheep habitat at higher elevations in desert mountain ranges, reducing their ability to detect and avoid predators (Peek et al. 1979, Singer et al. 2000, Bunn et al. 2007). Previous to two large burns in the Santa Catalina Mountains in the early 2000s, fire suppression was estimated to create a 0.5% per year reduction in visibility in bighorn sheep habitat (Krausman et al. 1996).

2. Climate

   a. Changes in seasonal precipitation and increased drought are associated with reduced availability of forage and dependable freshwater springs (Douglas and Leslie 1986, Wehausen et al. 1987, Epps et al. 2004).

6.10.3 Required Input Data Layers

Predicting potential habitat for desert bighorn sheep in the Sonoran Desert ecoregion requires GIS data layers which describe environmental conditions thought to influence desert bighorn sheep occurrence: topography and elevation, vegetation, water, human development. Spatially explicit desert bighorn sheep occurrence data are also necessary, preferably recent, accurate, and from a systematic sample design. Predictors used in previous efforts to predict potential desert bighorn sheep habitat include: slope, elevation, incident radiation, terrain ruggedness, perennial water sources, roads, human development (urban, commercial, residential, and agriculture), and vegetation cover type (Sappington et al. 2007, Rubin et al. 2009).

Other datasets necessary to evaluate the proposed management questions include: known locations of desert bighorn sheep movement corridors, known locations of all desert bighorn sheep surveys (state and federal agencies, universities, NGOs), probable locations of human development, and probable changes in perennial water sources and vegetation cover resulting from climate change (Table 6-3).

Raw Data

Table 6-3. Raw data input layers.

Data Layer	Potential Sources	Location	MQs
Elevation	NED (USGS)	http://seamless.usgs.gov/website/seamless/viewer.htm	1, 3-6
Vegetation	Landfire (USFS), SWReGAP (USGS)	http://landfire.cr.usgs.gov/viewer/ http://earth.gis.usu.edu/swgap/landcover_download.html	1, 3-6
Canopy cover	NLCD Canopy Cover (USGS),	http://www.mrlc.gov/nlcd2001_downloads.php	1, 3-6
Water (including springs and seeps)	NHD (USGS), NWI (USFWS)	http://nhdgeo.usgs.gov/viewer.htm http://www.fws.gov/wetlands/	1, 3-6

Data Layer	Potential Sources	Location	MQs
		Data/DataDownload.html	
Wildlife water developments	Arizona Game and Fish Dept.		1, 3-6
Roads	StreetMap (ESRI), National Atlas	http://www.esri.com/data/data-maps/data-and-maps-dvd.html http://www.nationalatlas.gov/atlasftp.html?openChapters=%2Cchptrans#chptrans	1, 3-6
Human footprint	NLCD Impervious Surfaces	http://www.mrlc.gov/nlcd2006_downloads.php	1, 3-6
Development	NLCD Land Cover (USGS), Landfire (USFS), SWReGAP (USGS)	http://landfire.cr.usgs.gov/viewer/ http://earth.gis.usu.edu/swgap/landcover_download.html	1, 3-6
Agriculture	NLCD Land Cover (USGS), Landfire (USFS), SWReGAP (USGS)	http://landfire.cr.usgs.gov/viewer/ http://earth.gis.usu.edu/swgap/landcover_download.html	1, 3-6
Land protection status and ownership	PAD-US & NCED	http://databasin.org/protected-center/features/PAD-US-CBI/download	1, 3-6
Grazing allotments			1, 3-6
Desert bighorn sheep occurrences	Unknown		1-6
Desert bighorn sheep movement corridors	Unknown		1, 3-6
Desert bighorn sheep survey locations	Unknown		1-6

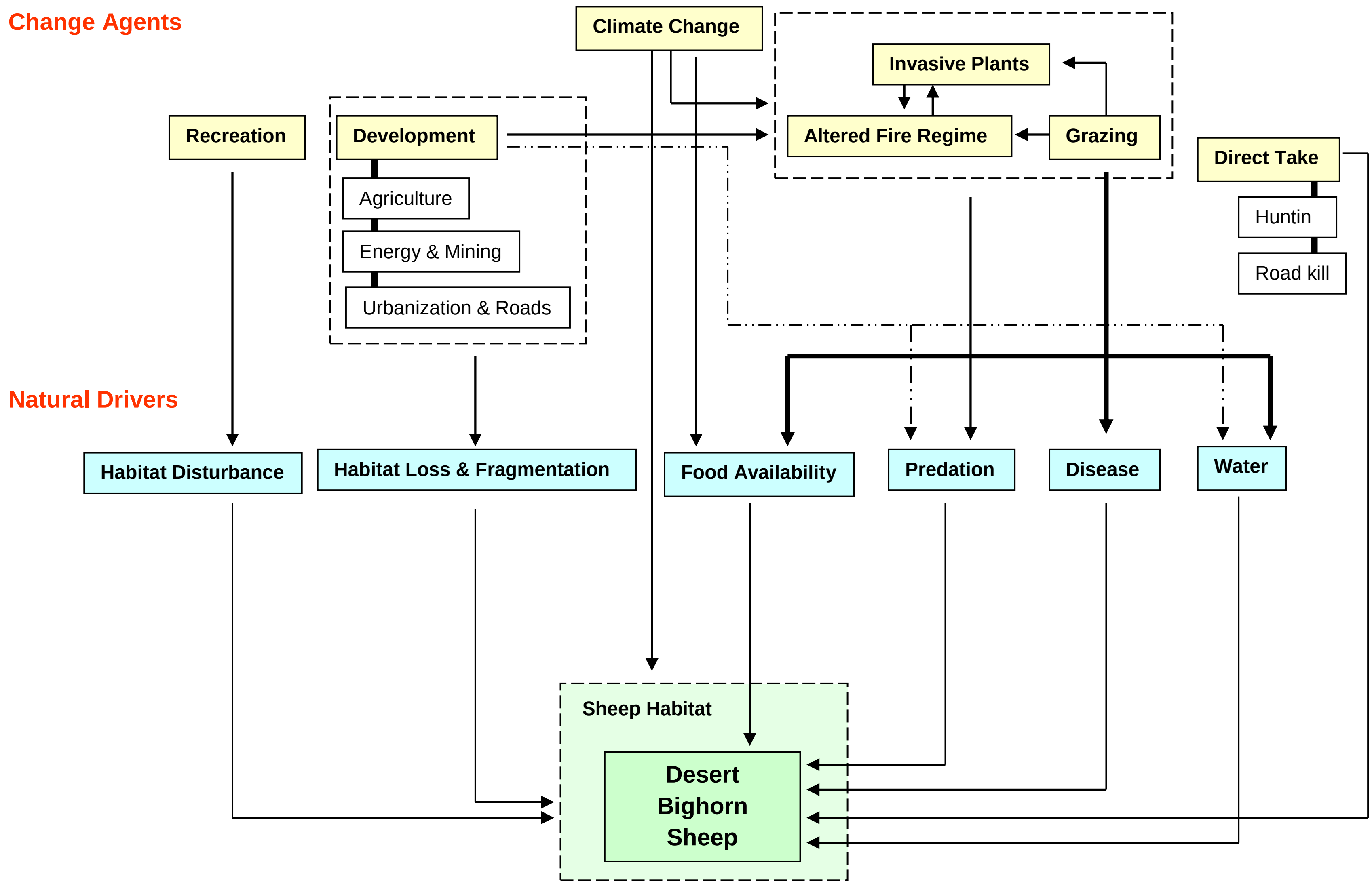


Figure 6-4. Conceptual model for Desert bighorn sheep.

Previously Processed Data

Processed data layers useful for predicting desert bighorn sheep potential habitat include slope, terrain ruggedness (vector ruggedness measure of terrain (VRM), Sappington et al. 2007), incident radiation, distance to suitable vegetation types, proportion of suitable vegetation types, distance to perennial water sources, road density, distance to nearest road, distance to development, distance to agriculture, and distance to grazing (Table 6-4). Some of these data layers may not have been previously processed and will be need to be created for this effort.

Table 6-4. Processed data layers.

Derived Data Layer	Derived from	Location	MQs
Slope	Elevation	http://seamless.usgs.gov/website/seamless/viewer.htm	1, 3-6
Terrain ruggedness (VRM, Sappington et al. 2007)	Elevation	http://seamless.usgs.gov/website/seamless/viewer.htm	1, 3-6
Incident radiation	Elevation	http://seamless.usgs.gov/website/seamless/viewer.htm	1, 3-6
Distance to suitable vegetation types	Vegetation	http://landfire.cr.usgs.gov/viewer/ http://earth.gis.usu.edu/swgap/landcover_download.html	1, 3-6
Proportion of suitable vegetation types	Vegetation	http://landfire.cr.usgs.gov/viewer/ http://earth.gis.usu.edu/swgap/landcover_download.html	1, 3-6
Distance to perennial water	Water	http://nhdgeo.usgs.gov/viewer.htm http://www.fws.gov/wetlands/Data/DataDownload.html	1, 3-6
Distance to wildlife water developments	Water	Arizona Game and Fish Dept.	
Distance to roads	Roads	http://www.esri.com/data/data-maps/data-and-maps-dvd.html http://www.nationalatlas.gov/atlasftp.html?openChapters=%2Cchptrans#chptrans	1, 3-6
Road density	Roads	http://www.esri.com/data/data-maps/data-and-maps-dvd.html http://www.nationalatlas.gov/atlasftp.html?openChapters=%2Cchptrans#chptrans	1, 3-6
Distance to development	Developmt	http://www.mrlc.gov/nlcd2006_downloads.php	1, 3-6
Distance to agriculture	Agriculture	http://landfire.cr.usgs.gov/viewer/ http://earth.gis.usu.edu/swgap/landcover_download.html	1, 3-6
Distance to grazing	Grazing		1, 3-6
Future percent tree	Climate		6

Derived Data Layer	Derived from	Location	MQs
cover predicted by climate change models	change modeling		
Future vegetation type predicted by climate change models	Climate change modeling		6
Future water predicted by climate change models	Climate change modeling		6
Future human footprint predicted by development models	Developmt modeling		6

6.10.4 Attributes, Indicators, and Metrics

Ecological attributes are traits or factors that are necessary to maintaining a fully functioning species population, assemblage, community, or ecosystem. On a species level, they are traits that are necessary for species survival and long-term viability. Indicators are measurable aspects of ecological attributes. In the REAs, attributes and indicators are key elements used to answer management questions, parameterize models, and help explain the expected range in status and condition of individual conservation elements. We propose using attributes and indicators related to habitat, disease and predation risk, forage competition, and seasonal ranges as a potential collection of optimal environmental responses in the modeling for Sonoran Desert bighorn sheep (Table 6-5).

Table 6-5. An example list of Attributes, Indicators, and Metrics that may be useful for evaluating the status and condition of bighorn sheep. Table patterned after Oliver and Tuhy, 2010. NOTE: *Many attributes vary by location, sex, and season.

Species	Key Ecological Attribute	Indicator	Indicator rating				Basis for Indicator Rating	Comments
			Poor	Fair	Good	Very Good		
Bighorn sheep	habitat	slope						Varies by season
Bighorn sheep	habitat	elevation						Varies by location
Bighorn sheep	habitat	ruggedness					Sappington et al. 2007	Associated with rugged terrain
Bighorn sheep	visibility of predators	vegetation height and density	Dense understory, brushy areas, mean vegetation height > 0.5 m		Open areas, sparse understory, mean vegetation height < 0.5 m		Smith et al. 1991	Avoid areas with poor visibility
Bighorn sheep	habitat	distance to perennial water	>3.2 km		≤ 3.2 km		Smith et al. 1991 Turner et al. 2004	
Bighorn sheep	disease, direct forage competition,	contact with domestic livestock	< 5km		≥ 5 km		Smith et al. 1991, Zeigenfuss	

Species	Key Ecological	Indicator	Indicator rating		Basis for Indicator	Comments
	social intolerance				et al. 2000	
Bighorn sheep	habitat	human activity	areas within 150m of residential, commercial, industrial development, highways, roads, and structures		Zeigenfuss et al. 2000	
Bighorn sheep	habitat	area			Smith et al. 1991	17 km^2 (6.5 mi^2) of habitat to support minimum viable population
Bighorn sheep	winter range	distance to escape terrain	> 300 m	≤ 300 m	Smith et al. 1991	
Bighorn sheep	winter range	snowpack depth	> 25 cm	≤ 25 cm	Smith et al. 1991	
Bighorn sheep	winter range	aspect	Non-southern exposures	SW, S, or SE exposure	Smith et al. 1991	
Bighorn sheep	winter range	area			Smith et al. 1991	6.5 km^2 (2.5 mi^2) of winter range to support minimum viable population
Bighorn sheep	escape terrain (provides antipredator protection, bedding and lambing areas, areas of reduced snowpack)	slope	< 60%	> 60%	Smith et al. 1991	Presence of rock outcroppings also important
Bighorn sheep	lambing terrain	slope	< 60%	> 60%	Smith et al. 1991	
Bighorn sheep	lambing terrain	aspect	< 90º, > 270º	90 - 270º	Smith et al. 1991	
Bighorn sheep	lambing terrain	horizontal visibility			Smith et al. 1991	Select against areas with poor visibility
Bighorn sheep	lambing terrain	dstance to water	> 1 km	≤ 1 km	Smith et al. 1991	Water sources need to be within or adjacent to lambing areas.
Bighorn sheep	lambing terrain	patch area	< 2 ha (5 acres)		Smith et al. 1991	

6.10.5 Model Assumptions

A major assumption behind this approach is the ready availability of necessary data, including desert bighorn sheep occurrences, current human footprint, probable future human footprint, and future vegetation type and percent cover and surface water under likely climate change scenarios.

Current patterns of desert bighorn sheep detections will be assumed to represent areas suitable for supporting desert bighorn sheep, but it is unknown how current habitat quality relates to historic habitat quality (Spencer et al. 2008).

All models are necessarily oversimplifications of reality. Statistical models rely on correlations between variables to make predictions, often without full understanding of the cause and effect relationships underlying these correlations. Fitting a model to characterize the distribution of a species over a large and heterogeneous region, such as the Sonoran Desert ecoregion, risks oversimplifying how the species actually selects habitat on a finer scale, which may vary somewhat from place to place (Spencer et al. 2008).

Models are only as good as the data used to construct them. The accuracy of the output of this effort will largely depend on the quality of the data available.

While predicted potential suitable desert bighorn sheep habitat produced by this approach will likely be at the same resolution as the data used to derive it (30m raster data), data can be rescaled up to 5^{th} level HUCs for the purposes of reporting regional results.

6.10.6 Methods and Tools

In order to answer the management questions listed above, it is first necessary to map current potential desert bighorn sheep habitat. Many previous efforts to map potential bighorn sheep habitat have been based on expert opinion (Smith et al. 1991, Dunn 1996, USFWS 2000, Boykin 2007). This approach can be problematic due to the fragmented nature of bighorn sheep distribution, where habitat may vary from population to population. Furthermore, this approach is often subjected to criticism due to its subjective nature and lack of repeatability (Rubin et al. 2009).

Smith et al. (1991) developed an expert opinion based habitat evaluation procedure for Rocky Mountain bighorn sheep in the intermountain west, and cites several previous efforts for desert bighorn sheep published between 1970 and 1988. This approach has been tested in various locations by Zeigenfuss et al. (2000) and Johnson and Swift (2000). Johnson and Swift (2000) tested the accuracy of the Smith et al. (1991) procedure in predicting translocation success at several national parks in Colorado and found its performance low. Zeigenfuss et al. (2000) tested the ability of a modified version of the Smith et al. (1991) procedure in predicting the success of translocations at national parks in the Rocky Mountains, Colorado Plateau, desert, and prairie badlands of six states and found its accuracy to vary from 23 to 97%. The following criteria were used to identify suitable habitat:

1. Delineate all areas with slope between 27 and 85 degrees as ‘escape terrain’.
2. Delineate core habitat as all areas within 300m of escape terrain or within 1000m of escape terrain when an area is bordered by escape terrain on more than 2 sides.
3. Remove from core habitat all areas that
    a. Are more than 3.2 km from perennial water sources.
    b. Contain movement barriers: impassable fences, highways, housing developments, reservoirs, high activity areas, large or swift moving bodies of water, impassable cliffs, and large patches of dense vegetation.
    c. Have less than 55% horizontal visibility (indicative of vegetation density, based on field measurements in applicable vegetation communities).
    d. Are developed (residential, commercial, or industrial development; roads and highways; and structures; and land within 150 m of these areas.
4. Delineate summer range as all areas within suitable core habitat with slopes less than 27 degrees.
5. Delineate winter range as all areas within suitable core habitat with southern aspects, slope between 27 and 85 degrees, and snowpack less than 25 cm.

6. Delineate lambing period habitat as all suitable core habitat with area greater than 2 ha, southern, western, or eastern aspects, slopes between 27 and 85 degrees, and within 1000m of water.

USFWS (2000) used expert opinion to delineate critical habitat for the federally endangered population of bighorn sheep in California's Peninsular Range (western Sonoran Desert), characterizing areas with or within 0.8 km of slopes equal to or greater than 20% as critical habitat, and removing several vegetation associations and developed areas (Figure 6-5). The SW ReGAP program (USGS GAP Analysis Program 2005) used expert opinion to model the habitat distribution of bighorn sheep in the Arizona portion of the Sonoran Desert ecoregion, with the resulting model combining elevation and ecological systems. The California GAP program (Davis et al. 1998) used the California Wildlife Habitat Relationships database, an expert opinion-based habitat model, to map bighorn sheep suitability according to vegetation type (Figure 6-5). Polygons in the western Arizona Sonoran Desert originate from hand-drawn maps of known bighorn sheep occupied habitats.

Quantitative or statistical models of species distribution may help identify potential habitat overlooked by experts or not represented by current habitat usage (Rubin et al. 2009). Several examples of statistical models of bighorn sheep habitat can be found in the literature, but most have very small spatial extents. Extrapolating the results of a model constructed with data from a small localized area to a broad geographic extent often results in low performance and high inaccuracy.

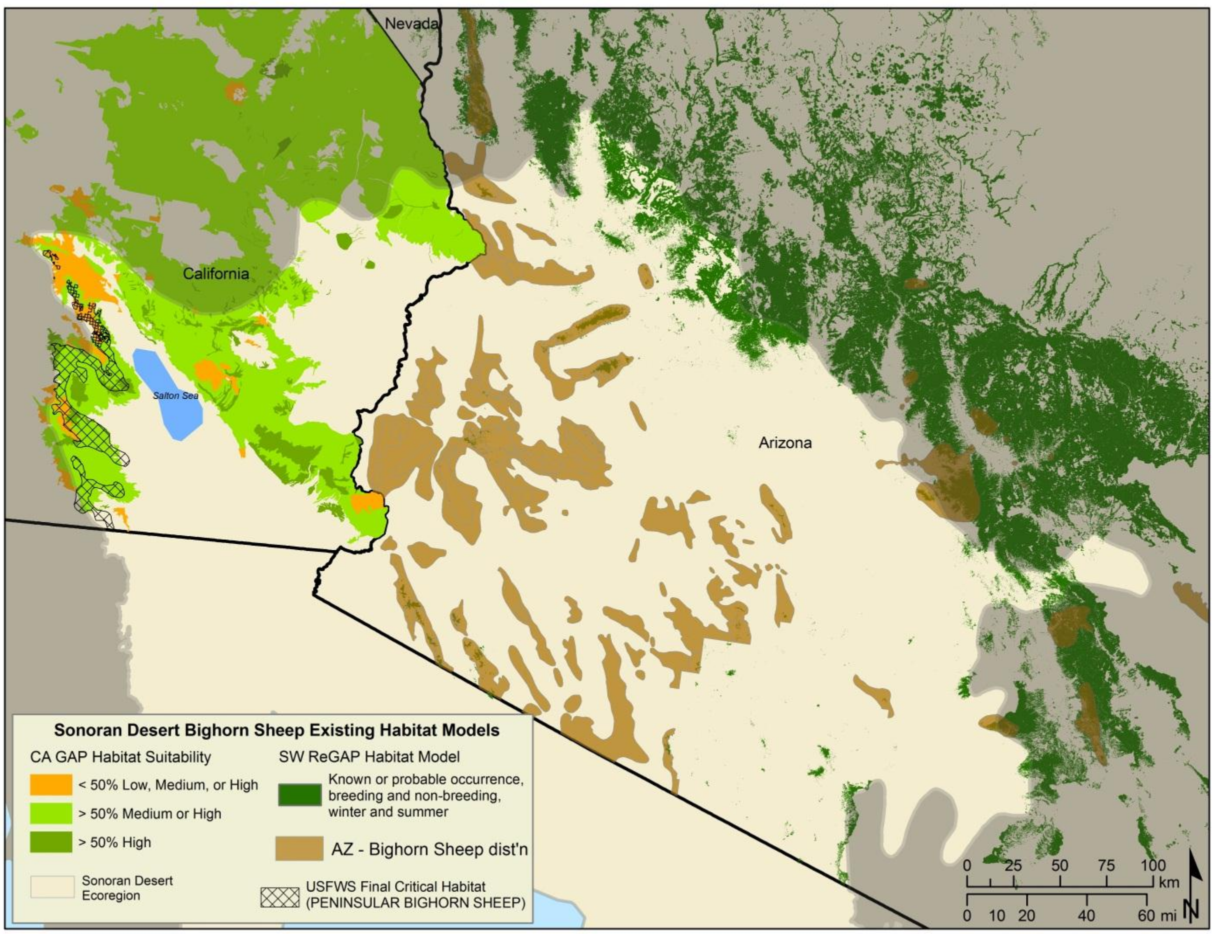


Figure 6-5. Existing desert bighorn sheep habitat data in the Sonoran Desert ecoregion.

Thompson et al. (2007), Sappington et al. (2007), and Longshore et al. (2009) constructed logistic regression models to predict desert bighorn sheep occurrence in the Mojave Desert. Thompson et al. (2007) used location data from 10 ewes in a 300 km^2 area at Joshua Tree National Park to create separate models of female lambing, summer, and winter habitat with slope, ruggedness, elevation, and distance to water as predictors. Longshore et al. (2009) also constructed logistic regression models to predict female summer habitat around Joshua Tree National Park, finding slope, distance to water, and ruggedness to all be significant predictors of desert bighorn sheep presence, with distance to water being most important. Sappington et al. (2007) modeled springtime female desert bighorn sheep habitat in three Mojave mountain ranges using logistic regression and slope, distance to water, and ruggedness as predictors.

Ruggedness and distance to water were significant in all three study areas, while slope was significant in two of the study areas.

Shannon et al. (2008) and Olson et al. (2008) modeled bighorn sheep habitat in Utah. Olson et al. (2008) constructed logistic regression models to predict female bighorn sheep habitat use at two study areas in north-central Utah, 45 km apart. Winter, summer, lambing, and year-round habitat use were modeled separately using slope, ruggedness, area with tree cover, and solar radiation as predictors. The best model for all seasons and year round use in one study area contained slope only, while for the second study area, the best year round model contained slope and area with tree cover and best seasonal models used all four variables. Shannon et al. (2008) tested the accuracy of using existing logistic regression models constructed for Mount Nemo in the Uinta National Forest on Mount Timpanogos, 60 km away. The models predicted lambing and winter habitat using elevation, slope, aspect, and cover type. Shannon et al. (2008) found the winter model to accurately predict 14% of bighorn sheep locations, and the lambing model to predict 18% of bighorn sheep locations. The addition of a 100m buffer increased model accuracy to 43% for winter and 36% for lambing. Shannon et al. (2008) attributed the poor performance of these models when extrapolated to a site 60 km away to differences in habitats used by bighorn between study areas.

Turner et al (2004) and Rubin et al. (2009) created predictive models of Peninsular Range bighorn sheep in the Sonoran Desert. Turner et al. used elevation, ruggedness, proximity to water, proximity to escape habitat, and slope aspect and logistic regression to predict habitat use. While Turner et al. (2004) does provide values of these predictors by habitat quality, their approach has been characterized as flawed (Ostermann-Kelm et al. 2005). Rubin et al. (2009) used bioclimatic envelopes and the GARP species distribution modeling program to predict Peninsular Range desert bighorn sheep habitat, using slope, elevation, incident radiation, distance to water, distance to high ruggedness, ruggedness, distance to roads, distance to forest, distance to hardwood woodland, distance to conifer woodland, distance to shrub, distance to desert shrub, distance to desert woodland, distance to urban or agriculture, and Normalized difference vegetation index (NDVI, reflectance value from satellite data, index of vegetation cover). The results were similar to those found by USFWS (2000), an expert opinion model.

California Department of Fish and Game also is reported to have a desert bighorn sheep model for the Sonoran Desert, but the details are not available at this time.

While many models of bighorn sheep habitat exist, most are:

1. Developed for areas outside the Sonoran Desert ecoregion and quite local in extent (Thompson et al. 2007, Sappington et al. 2007, Longshore et al. 2009, Shannon et al. 2008, Olson et al. 2008). These models likely will not accurately predict habitat over the broad geographic region of the Sonoran Desert.
2. Include the Sonoran Desert, but only the Peninsular Range population along the western portion of the ecoregion (USFWS 2000, Turner et al. 2004, Rubin et al. 2009). The accuracy of these models if extrapolated outside the Peninsular Range has not been evaluated.

3. Include the Sonoran Desert, but rely on expert opinion and vegetation type (Davis et al. 1998, USGS GAP Analysis Program 2005), while most habitat models have shown strong associations with abiotic variables such as slope, ruggedness, and proximity to water. Furthermore, these efforts show general year-round habitat, while lambing and winter habitats are more limiting.
4. Rely on expert opinion but were designed for regions outside the Sonoran Desert (Smith et al 1991, Zeigenfuss et al. 2000, Johnson and Swift 2000). The performance of these models has not been evaluated in the Sonoran Desert ecoregion.

For these reasons, if occurrence data were available across the region, it would be advisable to construct a new model to predict desert bighorn sheep habitat across the Sonoran Desert ecoregion. The approach used here predicts suitable desert bighorn sheep habitat using a statistical species distribution model (MaxEnt) which is easy to use and quickly interfaces with ArcGIS.

MaxEnt is a species distribution modeling method that uses presence-only species occurrence data, valuable when systematic formal survey data is lacking, which is often the case. MaxEnt has been used extensively and found to be high performing and robust to small sample sizes (Elith et al. 2006, Hernandez et al. 2006, Elith et al. 2011).

Inputs for MaxEnt include species occurrence data and environmental predictor layer for the study area of interest. Steps for the species distribution modeling process are (Figure 6-6):

1. Process species occurrence data – filter by sex (male and female desert bighorn sheep are known to use habitat differently and should be modeled separately), season (habitat use varies between winter and summer and should be modeled separately), date (discarding records which are too old or filter out records which are too close together in time for temporal independence), and distance (filter records so that the minimum nearest neighbor distance is not less than a home range diameter for spatial independence and to remove clustering bias). For use in MaxEnt, the species occurrence data must be converted to a .csv table with x- and y- coordinates.
2. Process environmental data – derive needed predictor layers with GIS, check for correlation between predictors using a Pearson correlation test and eliminate correlated variables, and convert to ASCII format for use in MaxEnt.
3. Run MaxEnt.
4. Convert MaxEnt output from ASCII format to grid, convert continuous probability output to predicted potential habitat by reclassification.
5. Project MaxEnt model into the future using outputs of climate and development modeling.
6. Using resulting layers of desert bighorn sheep current potential habitat and probable future potential habitat resulting from likely climate change and human development, evaluate management questions with a GIS.

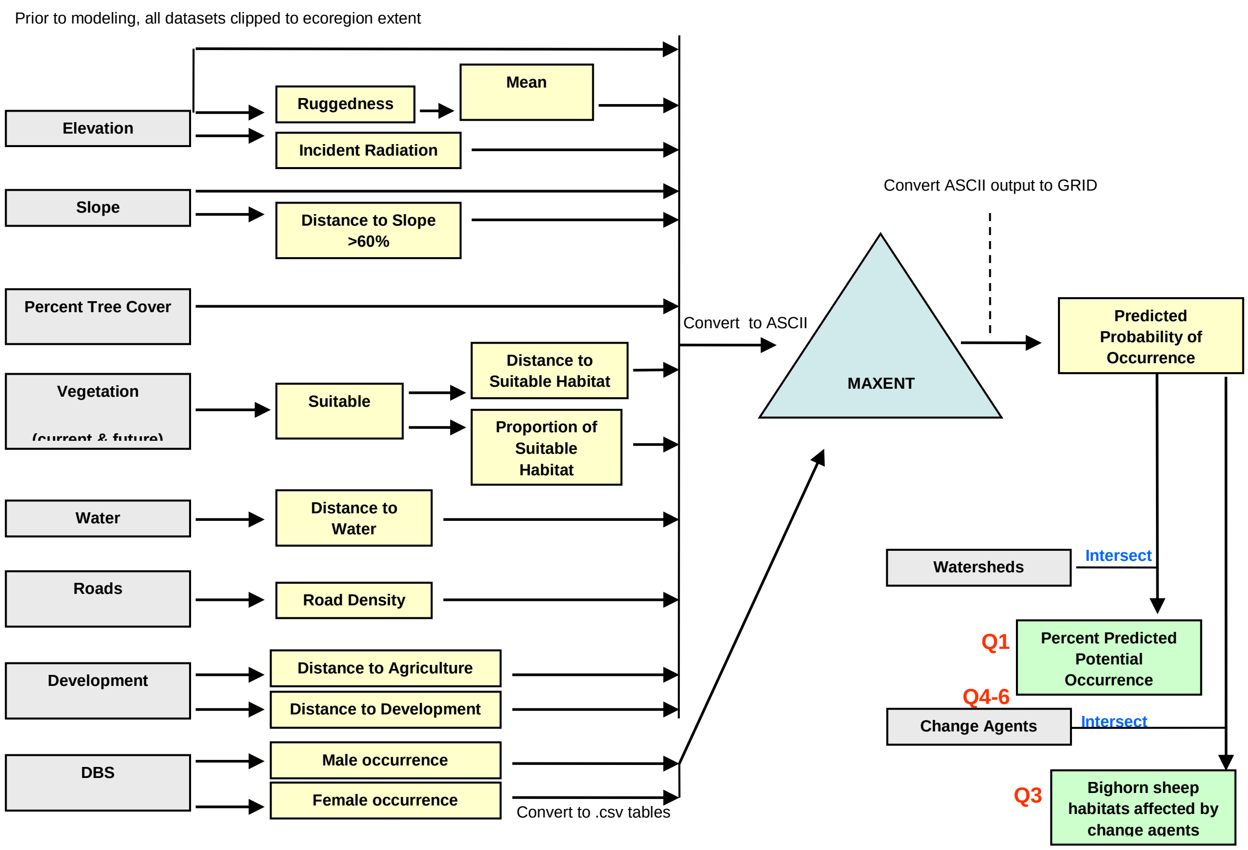


Figure 6-6. Process model for predicting desert bighorn sheep occurrence (current and future).

6.10.7 References Cited

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.

Bleich, V.C., J.D. Wehausen, R.R. Ramey II, and J.L. Rechel. 1996. Metapopulation theory and mountain sheep: implications for conservation. Pages 353–373 *in* D.R. McCullough, editor. Metapopulations and wildlife conservation. Island Press, Covelo, California.

Boykin, K.G. 2007. Ecoregional animal-habitat models for the southwestern United States. New Mexico Cooperative Fish and Wildlife Research Unit. Las Cruces, New Mexico.

Bunn, D., A. Mummert, M. Hoshovsky, K. Gilardi, and S. Shanks. 2007. California wildlife: conservation challenges (California's Wildlife Action Plan). California Department of Fish and Game, Sacramento, California.

Buechner, H.K. 1960. The bighorn sheep in the United States: its past, present, and future. *Wildlife Monographs* 4. 174 pp.

California Department of Fish and Game. 2010. Bighorn sheep life history account. *In* Zeiner, D.C., Laudenslayer, W.F., Jr., K.E. Mayer, and M. White (eds.). 1988-1990. California's Wildlife. Vol. I-III. California Depart. of Fish and Game, Sacramento, California. http://www.dfg.ca.gov/biogeodata/cwhr/cawildlife.aspx. Accessed 20 December 2010.

Davis, F. W., D. M. Stoms, A. D. Hollander, K. A. Thomas, P. A. Stine, D. Odion, M. I. Borchert, J. H. Thorne, M. V. Gray, R. E. Walker, K. Warner, and J. Graae. 1998. The California Gap Analysis Project--Final Report. University of California, Santa Barbara, CA. http://www.biogeog.ucsb.edu/projects/gap/gap_rep.html. Accessed 20 December 2010.

Douglas, C.L., and D.M. Leslie. 1986. Influence of weather and density on lamb survival of desert mountain sheep. *Journal of Wildlife Management* 50:153–156.

Dunn, W.C. 1996. Evaluating bighorn sheep habitat: a landscape approach. Bureau of Land Management Technical Note 395, U.S. Department of Interior, Santa Fe,, New Mexico.

Elith, J., C. H. Graham, R.P. Anderson, M. Dudik, S. Ferrier, A. Guisan, R.J. Hijmans, F.Huettmann, J. Leathwick, A. Lehmann, J.Li, L.G. Lohmann, B.A. Loiselle, G. Manion, C. Moritz, M. Nakamura, Y. Nakazawa, J. M. Overton, A.T. Peterson, S.J. Phillips, K.S. Richardson, R. Scachetti-Pereira, R.E. Schapire, J.Soberon, S. Williams, MS. Wisz, 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. Dudik, Y.E. Chee, and C. J. Yates. 2011. A statistical explanation of MaxEnt for ecologists. *Diversity and Distributions* 17:43–57.

Epps, C.W., D.R. McCullough, J.D. Wehausen, V.C. Bleich, and J.L. Rechel. 2004. Effects of climate change on population persistence of desert-dwelling mountain sheep in California. *Conservation Biology* 18:102–113.

Epps, C.W., P.J. PalsbØll, J.D. Wehausen, G.K. Roderick, R.R. Ramey II, and D.R. McCullough. 2005. Highways block gene flow and cause a rapid decline in genetic diversity of desert bighorn sheep. *Ecology Letters* 8:1029–1038.

Foreyt, W.J., and D.A. Jessup. 1982. Fatal pneumonia of bighorn sheep following association with domestic sheep. *Journal of Wildlife Diseases* 18:163–168.

Gross, J.E., F.J. Singer, and M.E. Moses. 2000. Effects of disease, dispersal, and area on bighorn sheep restoration. *Restoration Ecology* 8:25–37.

Hansen, C.G. 1982. Desert bighorn sheep: another view. *Wildlife Society Bulletin* 10:133–140.

Jessup, D.A. 1985. Diseases of domestic livestock which threaten bighorn sheep populations. *Desert Bighorn Council Transactions* 29:29–33.

Johnson, T.L., and D.M. Swift. 2000. A test of a habitat evaluation procedure for Rocky Mountain bighorn sheep. *Restoration Ecology* 8:47–56.

Krausman, P.R., G. Long, and L. Tarango. 1996. Desert bighorn sheep and fire, Santa Catalina Mountains, Arizona. Pages 162–168 *in* P. F. Ffolliott, L. Debano, M. B. Baker, Jr. G. J. Gottfried,

G. Soils-Garza, C. B. Edminster, D. G. Neary, L. S. Allen, and R. H. Hamre (technical coordinators), Effects of fire on the Madrean Province ecosystems. General Technical Report RM-289, U.S. Forest Service, Washington, D.C.

Krausman, P.R., J.W. Cain III, and H.E. Johnson. 2004. Impact of the Bullock and Aspen fires on desert bighorn sheep habitat in the Santa Catalina Mountains, Arizona. Report to Coronado National Forest, U.S. Forest Service, Tucson, Arizona.

Longshore, K.M., C. Lowrey, and D.B. Thompson. 2009. Compensating for diminishing natural water: Predicting the impacts of water development on summer habitat of desert bighorn sheep. *Journal of Arid Environments* 73:280–286.

Oliver, G.V., and J. Tuhy. 2010. Ecological Integrity Tables for Utah Animals of Conservation Concern. Report to the Utah Division of Wildlife Resources. 30June2010.

Olson, D., S. Petersen, R. Larsen, J. Flinders, J. Shannon, and J. Whiting. 2008. A robust approach for modeling bighorn sheep habitat.Proceedings of the 28th Annual ESRI International User Conference. http://proceedings.esri.com/library/userconf/proc08/papers/papers/pap_1162.pdf Accessed 16 March 2011.

Ostermann-Kelm, S.D., E. S. Rubin, J. D. Groom, J.R. DeForge, G. Wagner, P. Sorensen, S.G. Torres, M. C. Jorgensen, A.J. Byard, and O. Ryder. 2005. Flawed model has serious conservation implications: Response to Turner et al. *Wildlife Society Bulletin* 3:1456–1464.

Peek, J.M., R.A. Riggs, and J.L. Lauer. 1979. Evaluation of fall burning on bighorn sheep winter range. *Journal of Range Management* 32:430–432.

Rubin, E.S., C.J. Stermer, W.M. Boyce, and S.G. Torres. 2009. Assessment of predictive habitat models for bighorn sheep in California's Peninsular Ranges. *Journal of Wildlife Management* 73: 859–869.

Sappington, J. M., K.M. Longshore, and D. B. Thompson. 2007. Quantifying landscape ruggedness for animal habitat analysis: a case study using bighorn sheep in the Mojave Desert. *Journal of Wildlife Management* 71:1419–1426.

Schwartz, O.A., V.C. Bleich, and S. A. Holl. 1986. Genetics and the conservation of mountain sheep *Ovis canadensis nelsoni*. *Biological Conservation* 37:179–190.

Shannon, J.M., D.D. Olson, S.L. Petersen, J.C. Whiting, and J.T. Flinders. 2008. Can GIS models predict habitat use of reintroduced bighorn sheep? Proceedings of the 28th Annual ESRI International User conference.http://proceedings.esri.com/library/userconf/proc08/papers/papers/pap_1573 Accessed 16 March 2011.

Singer, F.J., V.C. Bleich, and M.A. Gudorf. 2000. Restoration of bighorn sheep metapopulations in and near western national parks. *Restoration Ecology* 8: 14–24.

Smith, T.S., J.T. Flinders, and D.S. Winn. 1991. A habitat evaluation procedure for Rocky Mountain bighorn sheep in the Intermountain West. *Great Basin Naturalist* 51:205–225.

Spencer, W.D., H.L. Rustigian, R.M. Scheller, A. Syphard, J. Strittholt, and B. Ward. 2008. Baseline evaluation of fisher habitat and population status, and effects of fires and fuels management on fishers in the southern Sierra Nevada. Unpublished report prepared for USDA Forest Service, Pacific Southwest Region. June 2008. 133 pp + appendices.

Thompson, D., K. Longshore, and C. Lowrey. 2007. The impact of human disturbance on desert bighorn sheep (*Ovis canadensis nelson*) in the Wonderland of Rocks/Queen Mountain region of Joshua Tree National Park, California. Unpublished Final Report Submitted to Joshua Tree National Park. 43 pp.

Turner, J.C., C.L. Douglas, C.R. Hallum, P.R. Krausman, and R.R. Ramey. 2004. Determination of critical habitat for the endangered Nelson's bighorn sheep in southern California. *Wildlife Society Bulletin* 32:427–448.

USFWS (U.S. Fish and Wildlife Service). 2000. Recovery plan for bighorn sheep in the Peninsular Ranges, California. U.S. Fish and Wildlife Service, Portland, Oregon, USA.

USGS GAP Analysis Program . 2005. Southwest Regional Gap Analysis Project: Final Report. http://fws-nmcfwru.nmsu.edu/swregap/pubs.htm Accessed 16 March 2011.

Wehausen, J.D., V.C. Bleich, B. Blong, and T.L. Russi. 1987. Recruitment dynamics in a southern California mountain sheep population. *Journal of Wildlife Management* 51:86–98.

Zeigenfuss, L.C., J.J. Singer, and M.A. Gudorf. 2000. Test of a modified habitat suitability model for bighorn sheep. *Restoration Ecology* 8:38–46.

6.11 Wildlife Species Conservation Element: Desert Tortoise (*Gopherus agassizii*), Mojave Population

6.11.1 Introduction

In this section we present a technical approach to address the current status and future condition of the Mojave population of the desert tortoise (*Gopherus agassizii*). Spatial analysis will incorporate and address the following species-related management questions that were identified by the Assessment Management Team (AMT) for inclusion in the Sonoran Desert Rapid Ecoregional Assessment (REA):

6. What is the most current distribution of available occupied habitat (and historic occupied habitat if available), including breeding, seasonal habitat, and movement corridors and bottlenecks (as applicable)?
7. What areas are known to have been surveyed and what areas have not been surveyed (i.e., data gap locations)?
8. Where are potential habitat restoration areas?
9. Where are potential areas to restore connectivity?
10. What terrestrial are vulnerable to change agents in the near term horizon, 2020 (development, fire, invasive species) and a long-term change horizon, 2060 (climate change)? Where are these species and sites located?

To answer the first two management questions, we will use model developed by Nussear et al. (2009) to represent the potential habitat of the Mojave population of the desert tortoise located in the Colorado Desert of California (west of the Colorado River) and parts of the western Sonoran Desert in Arizona (just east of the Colorado River). A separate model will be proposed in the near future for the Sonoran desert tortoise population found in the Arizona Sonoran Desert east of the Colorado River. Disturbance overlays and climate change modeling will help answer the remaining management questions. Change agents selected for the REA and considered in this analysis include: wildland fire, human development, resource uses, invasive plant species, and climate change.

The desert tortoise (*Gopherus agassizii)* inhabits desert environments in the Mojave and Sonoran deserts in southern California, southern Nevada, Arizona, southwestern Utah, and northwestern Mexico. The Mojave population of the desert tortoise occurs north and west of the Colorado River; it is listed as threatened, and 17 years after listing, the Mojave population is still declining, particularly in the western portion of its range in California (Brussard et al. 1994, Doak et al. 1994, Tracy et al. 2004). Declines continue even though tortoise management areas have been established and some of the major disturbances in those areas have been excluded.

Tracy et al. (2004) note that the original recovery plan prescribed that desert tortoises composed a single large population that required large blocks of habitat, but that it is also possible that tortoises exist in metapopulation patches connected by corridors; over the long term, patches may be vacant or densely populated. If the metapopulation theory is correct, the habitat fragmentation created by the continuing expansion of road networks, suburban and rural development, and renewable energy development makes the preservation of movement corridors difficult and may require facilitation of tortoise movement (translocations, road underpasses, Tracy et al. 2004). Another obstacle to tortoise recovery is the difficulty of managing a species that is sensitive, slow-moving, long-lived, late-maturing, and vulnerable to multiple threats to their existence. A tortoise generation is similar in length to a human generation, making it difficult to maintain funding and staffing continuity to track population response to recovery efforts. Threats can be reduced (for example, by building tortoise-proof fencing to reduce road mortality), but data must be collected and analyzed for longer time periods to document population-level recovery (Boarman and Kristan 2006).

The Mojave desert tortoise occurs mainly on creosote bush (*Larrea tridentata*) flats, but it is also found on sloping terrain on alluvial fans or foothills. It forages on herbaceous perennials and ephemeral annual plants; most tortoise activity (including breeding) takes place in the spring when annual plants and grasses are readily available (Nagy and Medica 1986, Brussard et al. 1994). Although Mojave desert tortoises burrow under shrubs in coarse sandy or loamy soils, they will also burrow under rocks or even cement slabs in disturbed areas (Andersen et al. 2000, Lovich and Daniels 2000). Burrows typically face north or northeast and are located under shrubs for shade, thermal cover, and protection from predation (eggs and juveniles, Lovich and Daniels 2000).

Tortoises are directly threatened by humans in a myriad of ways including conversion of tortoise habitat by development, degradation of habitat by road networks and ORVs, intentional killing of tortoises and direct mortality due to collisions with vehicles on roads and ORV trails. Note: We are considering the use of two threat diagrams developed by Tracy et al. (2004) and Darst and others at the Desert Tortoise Recovery Office. Once we have chosen a diagram in a useful format and with documentation, we will include it in future tortoise methods sections. Habitat destruction due to land development has led to large-scale population declines in some areas (Doak et al. 1994), and intentional killing of tortoises can account for up to 14% of all known mortality (Berry 1986). In addition, humans indirectly affect tortoise populations through grazing practices that create competition for the tortoise's herbaceous food plants. Development, road building, and grazing practices also promote the spread of invasive annual plant species and alter fire regimes. The invasion of alien grasses contributes to the increased number of fires that negatively affect tortoises (Brooks and Esque 2002, Esque et al. 2003).

6.11.2 Conceptual Model

Figure 6-7 depicts the relationships between natural population drivers, change agents, and desert tortoise populations. Alteration of desert habitats through development or conversion to agriculture has a negative effect on tortoise populations through direct mortality and habitat alteration. Habitat fragmentation and barriers to movement can severely limit desert tortoise populations (Edwards et al. 2004). Off-road vehicles (ORVs) destroy and degrade habitat, crush burrows, and kill tortoises. Although both habitat damage and direct mortality may occur, habitat damage is the most strongly established effect of ORV use (Bury and Luckenbach 2002).

Resource extraction and energy development affect tortoise populations not just through direct alteration of habitat but also through providing infrastructure and amenities that benefit predators of juvenile tortoises (Doak et al 1994, Boarman 2003). Residential development, roads, and landfills favor tortoise predators such as ravens, coyotes, and feral and domestic dogs. During a 25 year period in the late 20th century, some Mojave and California Sonoran raven (*Corvus corax*) populations increased by 450–1000% (Boarman 2003). Piles of shells, incriminating evidence, have been found under raven nests (Boarman 2003). In contrast, Boarman and Coe (2002) found that raven densities were low in the roadless portions of Joshua Tree National Park.

Prospects for recovery are bleak if threats to both adult and juvenile segments of the population are not reduced. Doak et al. (1994) found that the rate of desert tortoise population growth was most sensitive to the survival of large adult females, and they proposed that improving survival of adult females could reverse population declines. They urged that major sources of adult mortality should be the primary focus of management strategies. Tracy et al. (2004) countered that changes in the years since listing (such as the increase in predators on juvenile tortoises) may require rethinking how threats to tortoise survival should be managed. They stressed that threats are interactive and synergistic. If reproductive-aged female tortoises are removed from the population, then raven predation on juvenile life stages may have an even greater effect on tortoise numbers because it affects a larger proportion of juveniles in the population.

Similarly, the effects of disease on desert tortoise populations cannot be assessed in isolation. The frequency and intensity of upper respiratory disease, URTD, are mixed with the effects of other disturbances. Habitat degradation, drought stress, food shortages, and crowding may all affect the severity of URTD infections (Tracy et al. 2004}. The links of disease with development and drought stress and food availability are indicated in the conceptual model (Figure 6-7).

Invasive annual plant species can lead to higher fire frequencies that result in direct mortality, a loss of forage plants, shrub cover, and thermal refugia. Intense and extensive fire is increasing in both the Sonoran and Mojave deserts that have not evolved with fire. Direct effects of fire in desert habitats included animal mortality and loss of vegetation cover. Though tortoises may escape in underground burrows, direct mortality from fire has been documented in the Sonoran Desert (Esque et al. 2003). Esque and others (2003) estimated that 11% of adult desert tortoises present in the area of a fire at Saguaro National Park near Tucson, Arizona had died. Indirect effects of fire on tortoises include increased predation and loss of thermal cover.

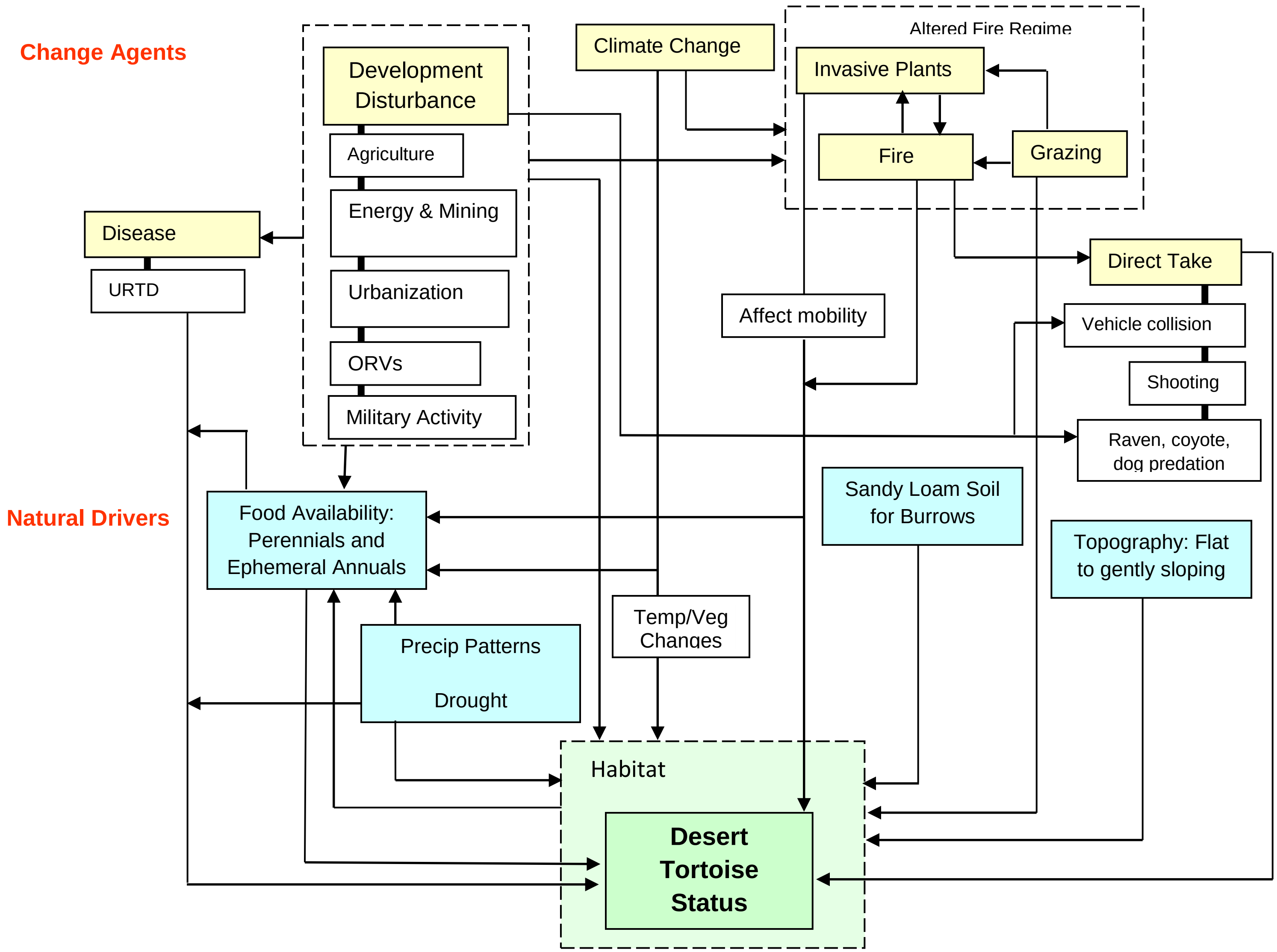
Because the desert tortoise exhibits temperature-dependent sex determination of hatchlings, loss of thermal cover can lead to skewed sex ratios that could affect future populations (Baxter et al. 2008). If the loss of thermal refugia is extreme, it could lead to direct mortality because tortoises are sensitive to extreme temperatures. On the other hand, skewed sex ratios are not found exclusively in stressed turtle populations (Lovich and Gibbons 1990), and tortoises have survived other periods of temperature extremes in their long evolutionary history. Patterns of hibernation and the use and placement of burrows play an important role in tortoise response to temperature extremes.

Climate change will likely affect tortoise populations in a variety of ways. If it leads to an increase in invasive plant populations, it could reduce the quality and/or abundance of annual plant forage. In addition, if there is an increase in fire frequency, there will be a decrease in suitable forage and shrub thermal cover, which could lead to increased heat- and drought-related mortality. Also, higher temperatures (estimated to be 2˚C in 2050) may cause changes in the suitable elevation range for the species, possibly shrinking the species' distribution in its present range (Bare et al. 2009).

6.11.3 Attributes and Indicators

Table 6-6 below lists key ecological attributes of the desert tortoise (Mojave population) that are important for the species' distribution, survival, productivity, and future viability. Ecological attributes are traits or factors that are necessary to maintaining a fully functioning species population, assemblage, community, or ecosystem. On a species level, they are traits that are necessary for species survival and long-term viability. Indicators are measurable aspects of ecological attributes. In the REAs, attributes and indicators are key elements used to answer management questions, parameterize models, and help explain the status and condition of individual conservation elements. The indicators listed are measurable components of the ecosystem that will be used to assess the current and future status of the Mojave desert tortoise. We will use attributes and indicators related to distance from urban areas, paved roads, agricultural areas, and areas of energy development and military activity overlaid on an existing model for the potential habitat of the Mojave desert tortoise to indicate near-term (2020) risk to the desert tortoise

Figure 6-7. Conceptual model for the Mojave population of the desert tortoise.



Species	Attribute Class	Key Ecological Attribute	Indicator	Range of Variation				Basis for Indicator Rating
				Poor	Fair	Good	Very Good	
Desert tortoise – Mojave	Size	Habitat	Size and distribution	<200 mi^2 (<518 km^2)	200–500 mi^2 (518–1295 km^2)	500–1,000 mi^2 (1295–2590 km^2)	>1,000 mi2 (>2590 km^2)	Brussard et al. (1994)
Desert tortoise – Mojave	Landscape Context	Habitat	Physiography	—	—	Rolling hills, mountainous toeslopes, rock outcrops, badlands, sand dunes,	Flats, valleys, alluvial fans, bajadas, rocky slopes, washes	Brussard et al. (1994), Barrett et al. (1994), Luckenbach (1982)
Desert tortoise – Mojave	Condition	Burrow and nest sites	Substrate type, bulk density, depth to bedrock, %rock	—	—	Rocky	Sand, sandy gravel, sandy loam, light gravel–clay, heavy gravel	Brussard et al. (1994), Barrett et al. (1994), Luckenbach (1982)
Desert tortoise – Mojave	Landscape Context	Climate, habitat	Elevation	>4,000 ft (>1219 m)	—	—	<4,000 ft (<1219 m)	Barrett et al. (1994), Luckenbach (1982), Stebbins (1985)
Desert tortoise – Mojave	Condition	Habitat degradation	% Cover Nonnative annual species such as red brome (*Bromus rubens*) and Sahara mustard (*Brassica tournefortii*)	Abundant, ineradicable	Fairly common and widespread	Scarce, patchy, potentially eradicable	None	Brussard et al. (1994), California Invasive Plant Council
Desert tortoise – Mojave	Condition	Fragmentation of habitat and populations, direct mortality	Distance to roads (dirt, paved, highways, freeways), railroads; Road density	<2 mi (<3.2 km)	2–5.5 mi (3.2–8.9 km)	5.5–7 mi (8.9–11.3 km)	>7 mi (>11.3 km)	Brussard et al. (1994)
Desert tortoise – Mojave	Condition	Habitat degradation, mortality	% Area Energy and mineral exploration and extraction	Existing (or planned), taking place in spring, summer, and fall (March–October) or year-round	Existing (or planned), taking place in winter (November–February)	None	None	Brussard et al. (1994)
Desert tortoise - Mojave	Condition	Mortality, habitat fragmentation, favoring of predators (ravens)	% Area Utility and energy corridors	Existing or planned	—	—	None	Brussard et al. (1994)
Desert tortoise – Mojave	Condition	Mortality (shooting, road kills, domestic predators, ravens), habitat alteration and elimination	Proximity of towns and cities	<30 mi (<48.3 km)	30–40 mi (48.3–64.4 km)	40–50 mi (64.4–80.5 km)	>50 mi (>80.5 km)	Brussard et al. (1994)
Desert tortoise – Mojave	Condition	Mortality, habitat degradation	Distance to military operations	<30 mi (<48.3 km)	30–40 mi (48.3–64.4 km)	40–50 mi (64.4–80.5 km)	>50 mi (>80.5 km)	Brussard et al. (1994), Barrett et al. (1994)

Table 6-6. Attributes and indicators for the Mojave population of the desert tortoise.

from human development (Table 6-6). The ranges of variation for each indicator help establish condition classes for that element. This is a representative (not complete) list of attributes and indicators; several of these indicators are key elements of the conceptual model, the potential habitat model, and the mapping of risk to tortoise potential habitat.

6.11.4 Modeling Methods

To depict the predicted habitat distribution of the Mojave population of the desert tortoise in the Sonoran Desert ecoregion, we will use an existing MaxEnt model developed by Nussear et al. (2009) for a wider region (the Mojave Desert and parts of the Sonoran Deserts of California, Nevada, Utah, and Arizona).

Presence records used for the model included data representing 15, 311 locations from 23 different data collection efforts dated from1970 through 2008 (most of the data were collected after 1990, Figure 6-8). This compilation of presence records will answer management question 2: Where are areas known to have been surveyed and not surveyed?

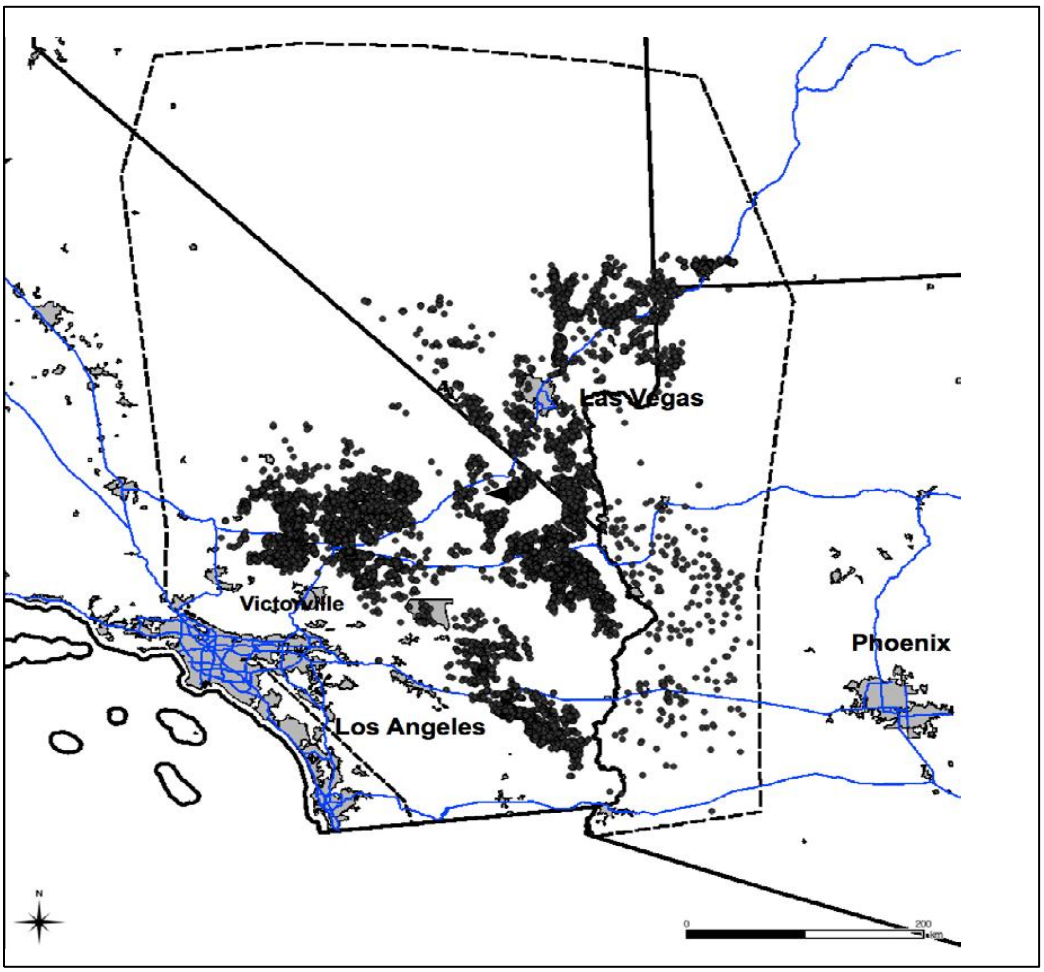


Figure 6-8. Distribution of desert tortoise (*Gopherus agassizii*) presence observations at sites in the Mojave Desert and parts of the Sonoran Desert of California, Nevada, Utah, and Arizona. *Solid circles* indicate records of one or more observations of live or dead tortoises (Nussear et al. 2009).

Environmental data layers used in the modeling were selected from the literature and best professional judgment as attributes that defined and influenced desert tortoise habitat:

- Elevation
- Percent smooth topography
- Average surface roughness
- Mean dry season precipitation for 30 yr normal period
- Mean wet season precipitation for 30 yr normal period
- Average soil bulk density
- Percent area with depth to bedrock > 1 m
- Percent of soil mass with rocks>254 mm (B-axis intermediate diameter)
- Total perennial plant cover modeled using MODIS EVI*
- Annual plant growth potential**

* Moderate Resolution Imaging Spectroradiometer (MODIS) Enhanced Vegetation Index (EVI) collected by the MODIS satellite and composited over 16-day intervals (Wallace et al. 2008)
**Annual growth potential derived by calculating the difference in greenness between two highly contrasting years of annual plant production (very wet and very dry year).

Nussear et al. (2009) modeled potential habitat using the MaxEnt algorithm (version 3.2.19, Phillips and others, 2006) and in their report they describe the method:

> Maxent uses a maximum entropy probability distribution to compare samples of occurrence data with background environmental data. Each of the included predictor variables were assessed using a jackknife test of variable importance and percent contribution (Phillips and others 2006). We used the logistic model output to represent an index of the potential of the habitat in a cell given the training data (Phillips and Dudik, 2008).
>
> The Maxent model produced a map of potential desert tortoise habitat for parts of the Mojave and Sonoran Deserts (Fig. 6 in Nussear et al. 2009). This model had a high AUC test score (0.93) and had a significant Pearson's correlation coefficient of 0.74 ($p < 0.01$), indicating a substantial agreement between the predicted habitat and the observed localities of desert tortoises. The model produced output with habitat-potential scores ranging from 0 to 1 (Fig. 7), plus an area that was not estimable because environmental data were not available for one or more layers. These scores were placed in 12 different bins (*0–1 by 0.10 increments*) to provide an index of habitat potential. Tortoises were present in 1-km^2 cells that spanned the entire range of model outputs. The mean model score for all tortoise presence cells was 0.84, and 95% of the cells with known presence had a model score greater than 0.7.

The MaxEnt results predict desert tortoise habitat, but they do not account for existing areas of development or disturbance factors that may have already altered or erased habitat. We will use indicators related to distance from urban areas, paved roads, agricultural areas, and areas of energy development and military activity overlaid on the existing MaxEnt potential habitat model to indicate current risk to the desert tortoise from human development (disturbance data layers listed in Table 6-7).

DISTURBANCE DATA LAYERS	DATA CLASS
Current climate (PRISM, DAYMET)	CLIMATE – RECENT
Future climate (2060 downscaled model)	CLIMATE – FUTURE
Human footprint (Development)	CURRENT DEVELOPMENT
Road Density	DEVELOPMENT
Land use planning areas	DEVELOPMENT – FUTURE
Population growth projections	DEVELOPMENT – FUTURE
Fire perimeters	FIRE
Grazing	ALLOTMENTS

Table 6-7. Disturbance data layers needed to estimate current and potential future risk to Mojave desert tortoise.

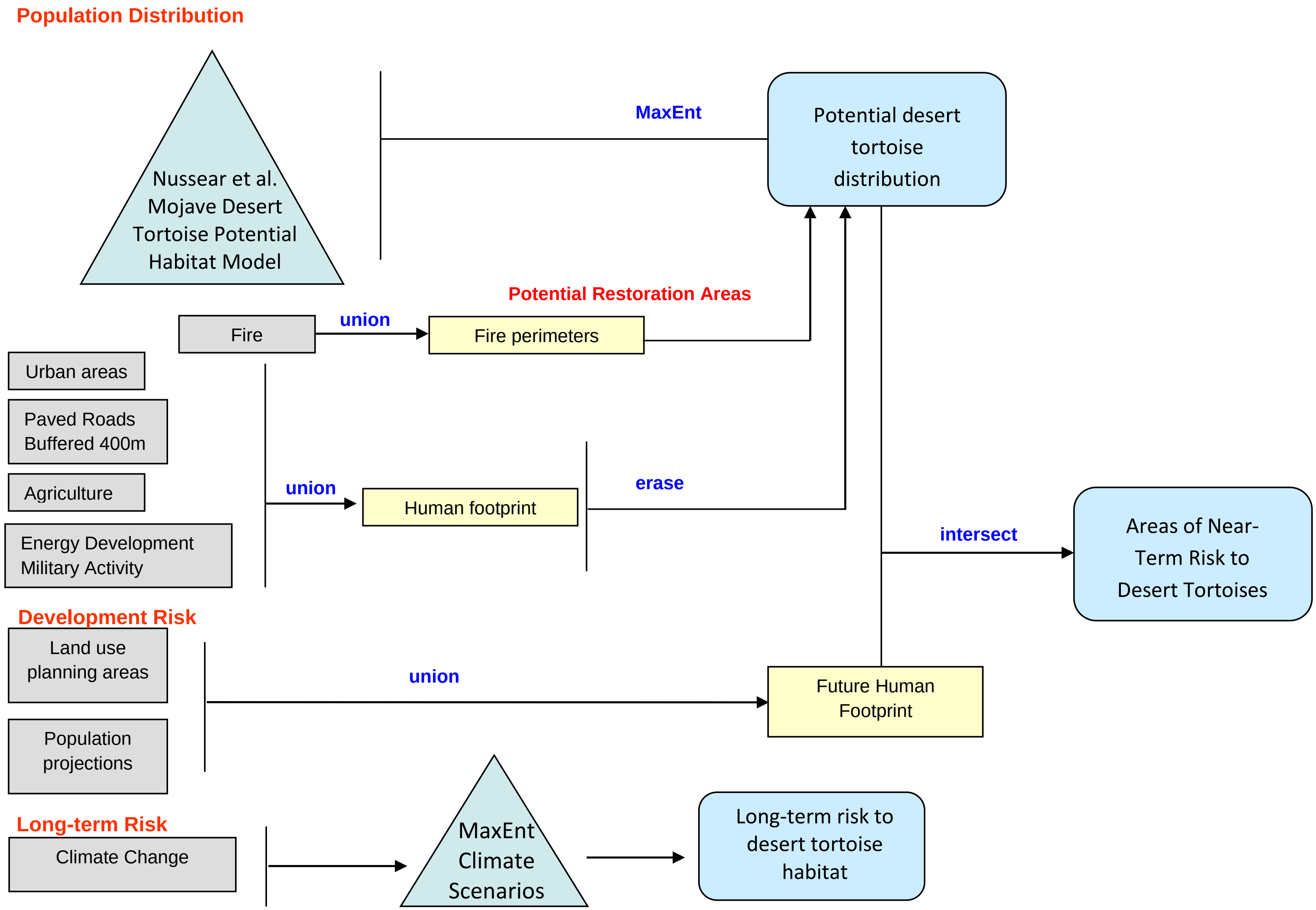
Figure 6-9 illustrates a process model for desert tortoises that will be developed in ArcGIS utilizing existing GIS data sets. In the process model (Figure 6-9) raw datasets are represented by gray boxes and previously-processed datasets are represented by yellow boxes. The MaxEnt potential habitat map will be overlaid with the human footprint layers to depict current potential Mojave desert tortoise distribution and answer management question 1: What is the most current distribution of available occupied habitat?

To identify areas of near-term risk from human development (management question 5), we will use data layers that depict population projections, land use planning areas, and areas of planned traditional and renewable energy development to delineate predicted areas of near term (2020) future development. The areas of projected development overlapping current desert tortoise distribution represent the predicted areas where tortoise populations will be at risk of decline or extirpation.

We can use fire perimeters (pictured upper center in Figure 6-9) to begin to answer management question 3: Where are potential habitat restoration areas? Other potential restoration areas may be delineated by identifying known grazing exclusions in Desert Tortoise Management Areas.

Desert tortoise vulnerability to climate change (management question 5) is a function of exposure and sensitivity. To determine effects of long-term risks due to climate change, we intend to overlay modeled future climate with current conditions to quantify the change in exposure. We can then superimpose the areas where change is occurring onto tortoise habitat and determine their sensitivity based on NatureServe's Climate Change Vulnerability Index (CCVI) tool (Young et al. 2010). The combination of change in exposure and species sensitivity will provide an estimate of vulnerability. In addition, as described in Section 4.4, we propose to use MaxEnt to model changes in the climate envelopes for each species based on environmental response curves for key bioclimatic variables of each individual species. MaxEnt output for climate change scenarios typically shows future potential distributions of individual species with categories of *no change*, *contraction*, and *expansion* of the species' distribution (Figure 4-16 in Section 4.3). Changes in climate envelopes will be acquired or calculated using the approved input climate models data and scenarios. Areas of current tortoise distribution will be classified as "vulnerable" where the future climate is outside the current observed climate envelope, and "not vulnerable or less vulnerable" for tortoise distribution within the climate envelope, as defined by the base period.

Figure 6-9. Processing steps for development of mapped areas of near-term and long-term risk to Mojave Desert tortoise using an existing desert tortoise potential habitat model for the Sonoran Desert of California and far western Arizona (Nussear et al. 2009) and disturbance data layers.



6.11.5 References Cited

Andersen, M. C., J. M. Watts, J. E. Freilich, S. R. Yool, G. I. Wakefield, J. F. McCauley, and P. B. Fahnestock. 2000. Regression-tree modeling of desert tortoise habitat in the central Mojave Desert. *Ecological Applications* 10(3):890-900.

Bare, L., T. Bernhardt, T. Chu, M. Gomez, C. Noddings, and M. Viljoen. 2009. Cumulative impacts of large-scale renewable energy development in the West Mojave. Group project report, Donald Bren School of Environmental Science and Management. Retrieved 1/5/2011 http://fiesta.bren.ucsb.edu/~westmojave/images/Wemo_Final.pdf.

Baxter, P. C., D. S. Wilson, and D. J. Morafka. 2008. The effects of nest date and placement of eggs in burrows on sex ratios and potential survival of hatchling desert tortoises, *Gopherus agassizii*. *Chelonian Conservation Biology* 7:52-59.

Berry, K. H. 1986. Incidence of gunshot deaths in desert tortoises in California. *Wildlife Society Bulletin* 14:127-132.

Berry, K. H., and L. L. Nicholson. 1984. The distribution and density of tortoise populations in California in the 1970's. Chapter 2 *in* K. H. Berry, editor. The status of the desert tortoise (*Gopherus agassizii*) in the United States. Desert Tortoise Council Report to U.S. Fish and Wildlife Service, Sacramento, California, USA.

Boarman, W.I. 2003. Managing a subsidized predator population: reducing common raven predation on desert tortoises. Environmental Management 32:205–217.

Boarman, W.I., and S.J. Coe. 2002. An evaluation of the distribution and abundance of common ravens at Joshua Tree National Park. Bulletin of the Southern California Academy of Science 101:86–102.

Boarman, W.I., and Kristan, W.B., 2006, Evaluation of evidence supporting the effectiveness of desert tortoise recovery actions. USGS Scientific Investigations Report 2006–5143, U.S. Geological Survey, Reston, Virginia.

Brooks, M. L. and Esque, T. C. 2002. Alien plants and fire in desert tortoise (Gopherus agassizii) habitat of the Mojave and Colorado Deserts. *Chelonian Conservation and Biology* 4(2), 330–340.

Brussard, P.F., K.H. Berry, M.E. Gilpin, E.R. Jacobson, D.J. Morafka, C.R. Schwalbe, C.R. Tracy, and F.C. Vasek. 1994. Desert tortoise (Mojave population) recovery plan. U.S. Fish and Wildlife Service, Portland, Oregon.

Bury, R.B., and R.A. Luckenback. 2002. Comparison of desert tortoise (*Gopherus agassizii*) populations in an unused and off-road vehicle area in the Mojave Desert. *Chelonian Conservation and Biology* 4(2):457–463.

Doak, D., P. Kareiva, and B. Klepetka. 1994. Modeling population viability for the desrt tortoise in the Western Mojave desert. *Ecological Applications* 4(3):446-460.

Edwards, T., C. R. Schwalbe, D. E. Swann, and C. S. Goldberg. 2004. Implications of anthropogenic landscape change on inter-population movements of the desert tortoise (*Gopherus agassizii*). *Conservation Genetics* 5:485-499.

Esque, T. C., C. R. Schwalbe, L. A. DeFalco, R. B. Duncan, and T. J. Hughes. 2003. Effects of desert wildfires on desert tortoise (*Gopherus agassizii*) and other small vertebrates. *The Southwestern Naturalist* 48:103-111.

Lovich, J. E. and R. Daniels. 2000. Environmental characteristics of desert tortoise (*Gopherus agassizii*) burrow locations in an altered industrial landscape. Chelonian Conservation and Biology 3(4):714-721)

Lovich, J. E. and J. W. Gibbons. 1990. Age at maturity influences adult sex ratio in the turtle *Malaclemys terrapin*. Oikos 59:126-134.

Nagy, K., and P. Medica. 1986. Physiological ecology of desert tortoises in southern Nevada. *Herpetologica* 42:73-92.

Nussear, K.E., Esque, T.C., Inman, R.D., Gass, Leila, Thomas, K.A., Wallace, C.S.A., Blainey, J.B., Miller, D.M., and Webb, R.H., 2009, Modeling habitat of the desert tortoise (*Gopherus agassizii*) in the Mojave and parts of the Sonoran Deserts of California, Nevada, Utah, and Arizona: U.S. Geological Survey Open-File Report 2009-1102, 18 p.

Tracy, C.R., R. Averill-Murray, W.I.Boarman, D. Delehanty, J. Heaton, E. McCoy, D. Morafka, K. Nussear, B. Hagerty, and P. Medica. 2004. Desert Tortoise Recovery Plan Assessment, U.S. Fish and Wildlife Service, Reno, Nevada.

Wallace, S.A., Webb, R.H. and Thomas, K.A. 2008. Estimation of perennial vegetation cover distribution in the Mojave Desert using MODIS-EVI data. *GIScience & Remote Sensing*, 45(2), 167–187.

APPENDICES

APPENDIX 1. SWReGAP vs. LANDFIRE Vegetation/Land Cover Data Review

We reviewed both SWreGAP and Landfire and provide the following review. SWreGAP contains only one main attribute field – land cover. The number of unique cover fields is 76. Landfire has many more attributes – including Biophysical Settings, Environmental Site Potential, Existing Vegetation Type (comparable to SWreGAP), Existing Vegetation Canopy Cover, Existing Vegetation Canopy Height, Fire Regime Condition Class, Fire Regime Condition Class Departure, Mean Fire Return Interval, Fire Regime Groups, Percent Low Severity Fire, Percent Mixed Severity Fire, Percent Replacement Severity Fire, Succession Classes (SClass), and Fuel Models. A short critique of each one is provided below and written by one of the main technical staff from Landfire (who now works at CBI) who was on the team which built the Landfire datasets for the southwest.

Dataset Evaluations:

Biophysical Settings (BpS): This dataset is probably the best available representation of where many communities would have been historically under "natural variability." Accuracy is higher for more easily categorized types (e.g., Pinyon-Juniper communities) than for types underrepresented by plots (barren types) or difficult to map from coarse site info alone (specific soil-based communities or riparian types). These data are accompanied by state-transition models that describe historical conditions & drivers. Quality level: probably reasonable for this objective. More info and download state-transition models/descriptions at: http://www.landfire.gov/NationalProductDescriptions20.php

Environmental Site Potential (ESP): This dataset is a variant of Biophysical Settings but targeted at what sites could support that community. It may have utility for characterizing which areas in the landscape can support communities that are not currently observed . Quality level: probably good enough for an REA.

Existing Vegetation Type (EVT): This dataset depicts where vegetation types occur now. These data are more strongly influenced by spectral signature than site conditions, so some communities that have similar signatures are mapped out incorrectly in places. Quality level: decent, but recommend comparison with SWreGAP. For many areas, the two datasets are very similar. For some classes (e.g., barren) SWreGAP is more detailed. We also believe SWreGAP had more external review and edits made as a result making for a slightly better product overall. **Recommendation**: *Use SWreGAP or a hybrid of the two for existing vegetation and use Landfire for all fire-related modeling*.

Existing Vegetation Canopy Cover: This dataset breaks canopy cover into 10% cover classes, based strongly on imagery and minimally corrected to reasonable levels for the mapped vegetation type (and thus this may indicate higher or lower cover than really appropriate). Quality level: not great, use with caution. Binning into broader categories would help. Under-represents distribution of annual grasses & other invasives; often poorly represents canopy co-dominants/subdominants.

Existing Vegetation Canopy Height: This dataset is intended to reflect height classes, minimally corrected to be reasonable for vegetation type. Quality level: not good, use with more caution. Binning may help, but I'd probably avoid this one unless really required.

Fire Regime Condition Class (FRCC): Three categories that describe how far out of sync the current conditions are from expected historical conditions. This dataset has received considerable criticism, and should be used with caution (the methods used were not appropriate for the scale or scope of work). It also involves an arbitrary landscape summary unit that introduces bias.

Fire Regime Condition Class Departure: Similar as FRCC but using a 100% scale, and so is more open to interpretation. Departure is measured by comparing percent in each current succession class to percentage of landscape expected in each class from state transition models. This dataset should also treat this with caution.

Mean Fire Return Interval: This dataset represents the best landscape scale estimation of mean fire return interval. During development, a respectable separation of high fire from low fire systems was observed, although there are edge effects (esp. between LANDFIRE zones or between vastly different vegetation types). The numbers lack precision, so should be treated as rough estimates.

Fire Regime Groups: A binning of Mean Fire Return Interval. Quality level: decent.

Percent Low Severity Fire, Percent Mixed Severity Fire, Percent Replacement Severity Fire: This dataset represents a summary from historical simulations using state transition models of how many fires were in different severity categories. It was intended to divide out surface fire from canopy fire systems. This dataset should be with caution, since interpreting severity vary widely with scale of analysis (and thus varied widely in the models we were feeding in)

Succession Classes (SClass): These data represent a binning of existing vegetation type, cover, and height into one of up to 5 states from the state transition models. It represents an attempt to identify which areas are recovering from disturbance vs. over-mature. Also includes areas affected by invasive vegetation (mapped out specifically above and beyond the EVT using methods I developed). Quality level: somewhat rough, but could be decent depending on use. Interpretation is contingent on knowing the BpS underneath each SClass.

Fuel Models: Applied based on expert interpretation of existing vegetation, cover, height, and site potential. Often reviewed by fire managers in each area and hand-corrected. One of the most highly reputed products out of LANDFIRE (most highly demanded), and given the most leeway to correct things using arbitrary rules. Quality level: probably decent, depending on the model system used, but I was pretty removed from development of these. These were intended to be used as inputs to models like FARSITE.

Additional general comments of Landfire products include:

- FBFM 13: probably decent
- FBFM 40: probably decent
- Forest Canopy Cover / Forest Canopy Height: corrected to be more reasonable for vegetation type & fuel models
- Forest Canopy Bulk density: rough, treat with caution
- Forest Canopy Base height: even more rough, treat with more caution
- FLCC fuelbeds (this was under development and not really being mapped while I was there, not sure of quality or availability)
- Fuel Loading Models (this was under development and not really being mapped while I was there, not sure of quality or availability)

REA Coarse-filter data: We will use a combination of CA GAP and SWReGAP, possibly a hybrid of CA GAP, SWReGAP and Landfire EVT for use as vegetation/landcover data. We will use Landfire products to address fire-related management questions.

APPENDIX 2. Coarse-Filter Ecological Systems for the Sonoran Desert Ecoregion

FOREST AND WOODLAND CLASSES (1.0%)

Percent of ecoregion	Code	Ecological System
0.06%	S035	Madrean Pine-Oak Forest and Woodland
0.00%	S036	Rocky Mountain Ponderosa Pine Woodland
0.17%	S039	Colorado Plateau Pinyon-Juniper Woodland
0.15%	S040	Great Basin Pinyon-Juniper Woodland
0.07%	S051	Madrean Encinal
0.56%	S112	Madrean Pinyon-Juniper Woodland

SHRUB /SCRUB CLASSES (82.4%)

Percent of ecoregion	Code	Ecological System
1.21%	S057	Mogollon Chaparral
2.13%	S058	Apacherian-Chihuahuan Mesquite Upland Scrub
0.34%	S060	Mojave Mid-Elevation Mixed Desert Scrub (*including the Joshua Tree anomaly*)
0.01%	S061	Chihuahuan Succulent Desert Scrub
0.25%	S062	Chihuahuan Creosotebush, Mixed Desert and Thorn Scrub
33.48%	S063	Sonoran Paloverde-Mixed Cacti Desert Scrub (*including Saguaro communities*)
0.01%	S068	Chihuahuan Stabilized Coppice Dune and Sand Flat Scrub
42.36%	S069	Sonora-Mojave Creosotebush-White Bursage Desert Scrub
0.97%	S070	Sonora-Mojave Mixed Salt Desert Scrub
0.22%	S116	Chihuahuan Mixed Salt Desert Scrub
1.46%	S129	Sonoran Mid-Elevation Desert Scrub

APPENDIX 2(Continued). Coarse-Filter Ecological Systems

GRASSLAND / HERBACEOUS CLASSES (0.5%)		
Percent of ecoregion	Code	Ecological System
0.20%	S075	Inter-Mountain Basins Juniper Savanna
0.25%	S077	Apacherian-Chihuahuan Piedmont Semi-Desert Grassland and Steppe
0.00%	S113	Chihuahuan Sandy Plains Semi-Desert Grassland
0.03%	S115	Madrean Juniper Savanna
WOODY WETLAND / RIPARIAN CLASSES (3.4%)		
Percent of ecoregion	Code	Ecological System
2.77%	S020	North American Warm Desert Wash
0.01%	S094	North American Warm Desert Lower Montane Riparian Woodland and Shrubland
0.18%	S097	North American Warm Desert Riparian Woodland and Shrubland
0.49%	S098	North American Warm Desert Riparian Mesquite Bosque
EMERGENT HERBACEOUS WETLAND CLASSES (0.0%)		
Percent of ecoregion	Code	Ecological System
0.00%	S100	North American Arid West Emergent Marsh
SPARSELY VEGETATED / BARREN CLASSES (1.7%)		
Percent of ecoregion	Code	Ecological System
0.00%	S010	Colorado Plateau Mixed Bedrock Canyon and Tableland
0.13%	S016	North American Warm Desert Bedrock Cliff and Outcrop
0.89%	S018	North American Warm Desert Active and Stabilized Dune
0.02%	S019	North American Warm Desert Volcanic Rockland
0.67%	N31	Barren Lands, Non-specific
OPEN WATER (1.1%)		
Percent of ecoregion	Code	Ecological System
1.05%	N11	Open Water

Classes adapted from: Lowry, J. H, Jr., R. D. Ramsey, K. Boykin, D. Bradford, P. Comer, S. Falzarano, W. Kepner, J. Kirby, L. Langs, J. Prior-Magee, G. Manis, L. O'Brien, T. Sajwaj, K. A. Thomas, W. Rieth, S. Schrader, D. Schrupp, K. Schulz, B. Thompson, C. Velasquez, C. Wallace, 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.

APPENDIX 3. Representative Plant Species Conservation Elements of principle Ecological Systems in the Sonoran Desert.

ECOLOGICAL SYSTEM	% OF ECOREGION	FINE FILTER SPECIES	SCIENTIFIC NAME
Sonoran Paloverde-Mixed Cacti Desert Scrub	33.5%	Saguaro	*Carnegia gigantean*
Sonora-Mojave Creosotebush-White Bursage Desert Scrub	42.3%	Creosotebush	*Larrea tridentata*
TOTAL AREA	75.8%		

Appendix 4. Candidate Landscape Species and Scores for the Sonoran Desert Ecoregion using a modified version of the Coppolillo et al. (2004) approach.

SPECIES	SCEINTIFIC NAME	AREA	HETEROGENEITY	VULNERABILITY	FUNCTIONALITY	SOCIO-ECONOMIC SIGNIFICANCE	SPECIES SCORE
Mountain lion	*Puma concolor*	1.00	0.84	0.25	0.50	0.80	3.39
American Peregrine Falcon	*Falco peregrinus*	1.00	0.74	0.50	0.50	0.20	2.94
Burrowing owl	Athene cunicularia	0.63	0.39	0.50	1.00	0.40	2.91
Peninsular Bighorn	Ovis canadensis pop. 2	0.38	0.10	1.00	0.50	0.80	2.77
Mexican long-tongued bat	Choeronycteris mexicana	0.13	0.29	0.75	1.00	0.60	2.77
Kit fox	Vulpes macrotis	0.75	0.55	0.25	1.00	0.20	2.75
Bobcat	Lynx rufus	0.88	0.68	0.00	0.50	0.60	2.65
Big free-tailed bat	Nyctinomops macrotis	1.00	0.94	0.50	0.00	0.20	2.64
Mule deer	Odocoileus hemionus	0.63	1.00	0.00	0.50	0.40	2.53
Desert tortoise – Mojave Population	Gopherus agassizii pop. 1	0.63	0.19	0.75	0.50	0.40	2.47
Desert bighorn (not Peninsular)	*Ovis Canadensis nelsoni*	0.38	0.48	0.50	0.50	0.60	2.46
Cactus ferruginous pygmy-owl	Glaucidium brasilianum cactorum	0.38	0.16	1.00	0.50	0.40	2.44
Lesser Long-nosed bat	Leptonycteris yerbabuenae	0.88	0.19	0.75	0.00	0.40	2.22
Antelope jackrabbit	Lepus alleni	0.50	0.29	0.50	0.50	0.20	1.99
Desert tortoise – Sonoran Population	Gopherus agassizii pop. 2	0.63	0.19	0.25	0.50	0.40	1.97
Gilded Flicker	Colaptes chrysoides	0.38	0.23	0.25	0.50	0.40	1.75
Southwestern Willow Flycatcher	Empidonax traillii extimus	0.00	0.13	1.00	0.00	0.60	1.73
Desert pupfish	Cyprinodon macularius	0.00	0.00	1.00	0.00	0.60	1.60
Razorback sucker	Xyrauchen texanus	0.00	0.00	1.00	0.00	0.60	1.60
Flat-tailed horned lizard	Phrynosoma mcallii	0.25	0.06	0.75	0.00	0.40	1.46
Lucy's warbler	Vermivora luciae	0.38	0.39	0.50	0.00	0.20	1.46
Sage thrasher	Oreoscoptes montanus	0.50	0.52	0.00	0.00	0.20	1.22
Gila topminnow	Poeciliopsis occidentalis	0.00	0.00	0.75	0.00	0.40	1.15
Lowland leopard frog	*Rana yavapaiensis*	0.38	0.00	0.25	0.00	0.20	0.83
Arizona toad	Bufo microscaphus	0.00	0.10	0.50	0.00	0.20	0.80
Canyon treefrog	Hyla arenicolor	0.38	0.00	0.00	0.00	0.20	0.58

Appendix 5. Final Selection of Landscape Species for the Sonoran Desert Ecoregion identified using a modified version of the Coppolillo et al. (2004) approach.

SPECIES	AREA	HETEROGENEITY	VULNERABILITY	FUNCTIONALITY	SOCIO-ECONOMIC SIGNIFICANCE	SPECIES SCORE
Mountain lion	1.00	0.84	0.25	0.50	0.80	3.39
~~Burrowing owl~~	0.63	0.39	0.50	1.00	0.40	2.91
Kit fox	0.75	0.55	0.25	1.00	0.20	2.75
Mule deer	0.63	1.00	0.00	0.50	0.40	2.53
Desert bighorn sheep	0.38	0.48	0.50	0.50	0.60	2.46
Lucy's warbler	0.38	0.39	0.50	0.00	0.20	1.46
~~Gila topminnow~~	0.00	0.00	0.75	0.00	0.40	1.15
~~Razorback sucker~~	0.00	0.00	1.00	0.00	0.60	1.60

Appendix 6. Desired Species Conservation Elements for the Sonoran Desert Ecoregion.

SPECIES	AREA	HETEROGENEITY	VULNERABILITY	FUNCTIONALITY	SOCIO-ECONOMIC SIGNIFICANCE	SPECIES SCORE
Desert tortoise – Sonoran	0.63	0.19	0.25	0.50	0.40	1.97
Desert tortoise- Mojave	0.63	0.19	0.75	0.50	0.40	2.47
Golden eagle						
Lowland leopard frog						
SW willow flycatcher						
Le Conte's thrasher						
Bell's vireo						

Appendix 7. Sites of Conservation Concern as Conservation Elements selected for the Sonoran Desert Ecoregion.

SITE CLASSES

Terrestrial Sites of High Biodiversity:

- TNC portfolio sites
- Important bird areas (Audubon)
- Areas recognized by Partners-In-Flight

Terrestrial Sites of High Ecological and/or Cultural Value:

- Historic and Nationally Designated Trails
- Wilderness Areas
- Wilderness Study Areas
- National Wildlife Refuges
- Monuments
- National and State Parks
- NCAs
- ACECs
- Forest Service Research Natural Areas
- State Wildlife Management Areas
- Suitable Wild and Scenic Rivers
- Designated Recreation Management Areas
- Sensitive Air Quality and Smoke Impact Receptors

Aquatic Sites of High Biodiversity:

- TNC portfolio sites
- EMAP-West Reference Sites

Appendix 8. Ecosystem Functions and Services of Conservation Concern selected for the Sonoran Desert Ecoregion.

SITE CLASSES

Terrestrial Functions of High Ecological Value:

- Soil stability
- Forage

Surface and Subsurface Water Availability:

- Aquatic systems of streams, lakes, ponds, etc.
- Springs/seeps/wetlands
- Riparian areas
- High quality and impaired waters
- Groundwater protection zones, sole source aquifers

Appendix 9. Change agents selected for the Sonoran Desert Ecoregion.

CHANGE AGENTS
• Wildland Fire • Invasive Species • Land and Resource Use — Urban and Roads Development — Oil, Gas, and Mining Development — Renewable Energy Development (i.e., solar, wind, geothermal, including transmission corridors) — Agriculture — Livestock, wild horse and burro, and wildlife grazing — Groundwater and Surface Water Extraction, Development, and Transportation — Recreational Uses — Pollution (Air Quality) • Climate change

APPENDIX 10. Overview of Task 2, Memorandum I-2-c: Data Identification and Evaluation

Appendix 10 provides a brief overview of Task 2, Memorandum I-2-c. The full memorandum with data tables, conceptual models, and appendices may be found at Website.

To identify general data needs to address specific management questions, the Dynamac team grouped management questions into subject classes and, using a conceptual model of conservation elements, change agents, and influential processes as a guide, we identified data layers needed to address each question within the group (Figure 3). This grouping proved useful not only for the data needs assessment, but later in data gap identification as well.

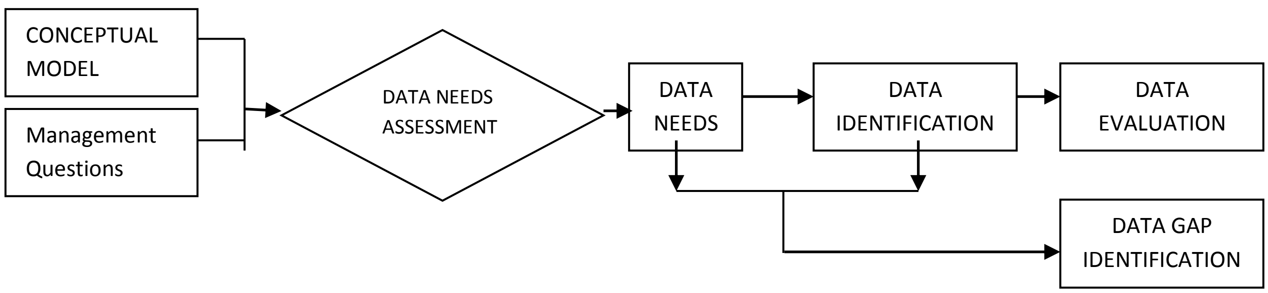


Figure 3. Process of data needs assessment through data evaluation and data gap identification.

Identification of the data needs related to groups of management questions first required consideration of the general approaches, methods, and tools by which each question might be answered. At this stage it is premature to assume that any particular approach or method will be approved, since decisions on approaches will not be made until the conclusion of Task 1.3. However, some assumptions had to be made to focus our data needs assessments. In general, the approaches will take the form of assessments of status or of potential for change, depending on the nature of the question and the availability of the data. Current status can be defined in spatially explicit terms. The footprint of oil and gas wells, the network of service roads, or locations of habitat corridors can be accurately described. Many questions related to future condition or potential for change lack this spatial specificity. Oil, gas, and renewable energy lease areas, or areas identified as having high potential for future development are simply zones in which measurable footprints, or even approximate locations, cannot be determined. Nor, for example, can we predict patterns of connectivity of vegetation under a climate change scenario and a change in disturbance frequency and severity. Logical areas may be set aside in which to preserve connectivity, but actual spatial configurations, patch size frequency distributions, and inter-patch distances can only be estimated. Successful comparison of current with future forecast conditions require output products that can be directly compared.

Data Needs by Management Question Group

Management questions were reorganized into groups for data needs evaluation and gap assessments. Each management question was reviewed and a tentative approach identified to provide a rationale for the data needs assessment. The rationale and data needs assessment by management question are summarized in Appendix 10 of the full document.

The conceptual models developed for Task 2 are at an intermediate level of detail and resolution. The focus of this task was data and data acquisition; the conceptual models illustrate the mechanisms and relationships that assisted Dynamac staff in the data needs evaluation. To avoid duplication of effort, we planned that a full literature review would accompany the models to be developed for Task 3, Methods and Models. The conceptual models developed for Task 3 will be more detailed and specific to individual management questions pertaining to each conservation element. The conceptual models used to date in the REA process are stressor models that illustrate the mechanisms and pathways of the sources of stress and the key, typical, or known responses of ecosystem attributes (conservation elements). Up and down arrows are commonly used to indicate the hypothesized response of particular ecosystem elements.

Data Identification and Evaluation

Data identification and evaluation is a continuation of the process that began with the review and evaluation of the lists of management questions provided by the AMT during the pre-assessment phase. The object of the data evaluation stage was to match potential data layers to the identified data needs (outlined in Section III and Appendix 10 of Memorandum I-2-c) and assess the utility of the datasets to map key attributes of conservation elements and to address classes of management questions.

Hundreds of datasets have already been acquired. The Dynamac team began the data evaluation by examining the data layers provided by BLM and classifying them into groups matching classes of management questions and sub-models of the basic ecoregional conceptual model. Evaluation efforts will be ongoing for some time and not confined to the pre-workshop timeframe.

Each dataset was to have been evaluated according to 11 quality criteria listed in the Data Management Plan (for example, criteria such as spatial accuracy, thematic accuracy, and precision) and given a confidence score. Confidence scores allow data layers within the same thematic class to be compared and the most suitable one chosen. Data evaluation tables and scores were meant to assist the AMT in making decisions on the choice of datasets to use in the assessment phase. However, evaluating the huge number of data layers was very time-consuming and complicated by redundant data layers. Many additional promising data layers were suggested by the participants in Workshop 2 and they remain to be incorporated and evaluated.

As a result of the challenges described, it became apparent that completion of the data identification and evaluation step was not realistic within the time and level-of-effort constraints that characterize the REA process. As a result, the AMT agreed to extend the data identification and evaluation stage through Task 3 and 4 of the REA and to delay the formal evaluation of data layers until they were formally accepted for the modeling effort. Memo I-2-c therefore represents a status report of data evaluations conducted through 18 October, 2010. A lesson learned from these early REAs might be for BLM to fund a sub-assessment to have groups of similarly-themed data layers evaluated to choose the best ones and then provide the best of the basic layers, such as energy development or agriculture, in the required or recommended list.

Data Gap Identification

In this section we review the data required to address specific conservation elements and change agents but not yet acquired. We have denoted clear data gaps under the EVALUATION column as “DATA GAP”. These represent high priority data needs. We identified some possible data sources for conservation elements through the workshop process, but species distribution data represents a major remaining data gap. A number of data layers and sources of layers have been identified which will likely fill many of the other data gaps, but they are yet to be evaluated. Tables 15 through 31 in Memorandum I-2-c define the specific conservation elements and change agents and list files or links which have been identified as possible data sources, and identify specific gaps that must be filled. Ecological Systems are not shown, since they will be defined based on either LANDFIRE or SW ReGAP.

Discussion

Attribution Accuracy

A common theme at both workshops was the accuracy of the major vegetation data layers, SW ReGAP and LANDFIRE. The Dynamac team showed an example of the differences in extent and attribution of various riparian vegetation classes for the same location. Some workshop participants were strongly in favor of using the GAP data, which they considered more accurate. Fire specialists naturally preferred LANDFIRE for fire related questions. The possible solutions are 1) to use SW ReGAP for all vegetation questions and LANDFIRE for fire-related questions with the risk of having incomparable results or 2) perform a cross-walk between SW ReGAP and LANDFIRE. The crosswalk would require rewriting the code for LANDFIRE using biophysical information from SW ReGAP. This would presumably be far too time-consuming to be accomplished within the REA framework. This issue is extremely important to resolve, as it will influence our proposed approaches, methods, and tools, as well as time estimates for Task I-3 related to ecological systems, fire, invasive species, and species habitat mapping.

Other attribution issues involve the accuracy of large nationwide data layers and our need to use them without alteration. The National Hydrography Dataset (NHD) is a basic required data layer that we will use for the REA. The NHD is a full-coverage digital data layer representing surface water features of the United States. A set of embedded attributes provides specialized information such as stream network or flow direction and links to related data such as discharge, habitat, or fish data. Because of its complexity, there are errors in the NHD. For example, in areas dense with canals crossing natural stream channels, we have experienced flow arrows pointing at each other or pointing uphill. The possibility of these errors influencing the outcome of the REA must be noted, although the SOW specifies that we are not to correct errors in data layers because of time limitations.

Data at Multiple Scales

One of the biggest challenges in the REA besides the sheer number of datasets will be the range in scale of the various data layers, ranging from coarse climate data interpolated onto a15 km grid to 30m resolution raster data to species occurrence data that may be spatially explicit or generalized. Limitations in the ability to overlay disparate data will influence the kinds of questions we will be able to answer. Many of the management questions are very specific, but the available data may not be specific enough to answer some questions.

Registration Errors

Overlaying different data layers from various sources may expose differences in registration. For example, when examining riparian vegetation as habitat, corridor, or to assess condition, it will be necessary to overlay the NHD dataset with a layer depicting vegetation, such as Landfire. We may want to buffer stream networks to calculate what proportion of stream miles contains riparian vegetation. There will be cases where the registration will be off and the stream blue line and areas of riparian vegetation will not match.

APPENDIX 11

The Ecological Integrity Assessment Framework: Application to BLM's Rapid Ecoregional Assessments

Purpose

This document provides a general framework and guidance on analytical and integrated approaches to assess resource values for Bureau of Land Management (BLM) Rapid Ecoregional Assessments (REAs). The motivation for this document is to provide additional information and clarification of the ecological integrity assessment framework described in the REA Statement of Work (SOW) developed by BLM. The primary objective is to help ensure that REAs: include appropriate conceptual models to address ecological integrity and identify key ecosystem components, are relevant to management issues identified by BLM, and support planning and management decisions within and among field offices. This additional guidance is also intended to help maintain consistency among REAs, thereby facilitating assessments across multiple ecoregions.

The Ecological Integrity Assessment Framework

Ecological integrity is a foundational concept in the REAs and is defined in the SOW as: "*The ability of ecological systems to support and maintain a community of organisms that have the species composition, diversity, and functional organization comparable to those of natural habitats within the ecoregion range (or area)*." In this definition, "functional organization" refers to the dominant ecological characteristics and processes that "occur within their natural (or acceptable) ranges of variation and can withstand and recover from most perturbations" (Parrish et al. 2003). Ecological integrity can also be viewed as the ecological condition or health of ecosystems.

A major impetus for using the concept of ecological integrity in the REAs is recognition that focusing on individual species (e.g., umbrella species, species of concern, game species): 1) will not adequately represent the complexity and dynamics of ecosystems, and 2) may not provide protection for species with habitat requirements or responses to stressors that differ from a selected set of indicator species. The ecological integrity approach addresses multiple levels of the system (species, communities, ecosystems), and includes coarse and fine filter components (Noss 1987). The coarse filter component emphasizes the management of dynamic and intact communities and ecosystems (Poiani et al 2000), and is based on the premise that intact and functioning systems are more resistant and resilient to stressors, thereby providing suitable habitat for most species (Noss 1987). Assessments at this level typically focus on the structure and composition of dominant or regionally important plant communities or ecosystems.

The coarse filter component serves as a safety net for most species. It is also recognized that some species may require greater specificity in habitat conditions than can be assessed by the coarse filter component and these species represent the fine filter component of the ecological integrity approach. The fine filter component consists of rare or specialized species, which would not adequately be protected by the coarse filter component, and are selected to represent unique contributions to the integrity of a system (Poiani et al 2000). Such species may require localized or limited habitats, or may already be at risk and require active management to prevent further population declines. The REAs will focus on species of regional importance.

Species-level or fine filter elements are commonly used by BLM in planning and management assessments. Thus, assessment of the coarse filter and fine filter components across an ecoregion will augment current BLM approaches. By evaluating coarse filter elements along with individual species or biotic composition, REAs may facilitate early detection of threats and changes and help to ensure that crucial aspects of ecological integrity are managed for the entire system (Parrish et al. 2003).

Ecological systems are complex, and the myriad of interactions and feedbacks are often poorly understood. The ecological integrity assessment framework and the development of conceptual ecological models can help to address this complexity. Conceptual ecological models are the foundation for this approach and provide an organizing structure to help identify key ecological elements for a given ecoregion and also inform selection of a suite of attributes and indicators that can be used to assess the condition of these elements (Parrish et al. 2003). This multi-level approach helps to address uncertainty by including different kinds of information and degrees of understanding. The ecological integrity assessment framework also provides a systematic and transparent process to develop measures that are scientifically defensible, practical, comparable across areas, and replicable over time (Parrish et al. 2003).

Integrated Assessment of Ecological Integrity

Ecoregional assessments are not exhaustive compilations of all resource information for a given ecoregion. Rather, they focus on regionally significant ecological resources that are relevant to BLM:

1. Terrestrial ecological features, functions, and services.
2. Aquatic ecological features, functions, and services.
3. Native fish, wildlife, or plant species.

A hierarchical framework for evaluating ecological integrity is presented in Figure 1. This framework is centered on coarse and fine filter conservation elements (Table 1) and builds from quantifiable indicators for important ecological attributes of these conservation elements. For assessing ecological integrity, coarse and fine filter conservation elements are selected based on conceptual ecological models. Assessments for conservation elements are aggregated into component scores for the ecological resources of the REAs, as well as an overall index of ecological integrity.

Management Applications

An assessment of ecological integrity across an ecoregion has a number of advantages over assessments of priority species at the project or field office level. The ecological integrity assessment framework provides information on potential cumulative effects of stressors across

jurisdictional boundaries (Tierney et al. 2009). The assessments also provide a frame of reference for naturally dynamic conditions and can serve as a benchmark for comparing the

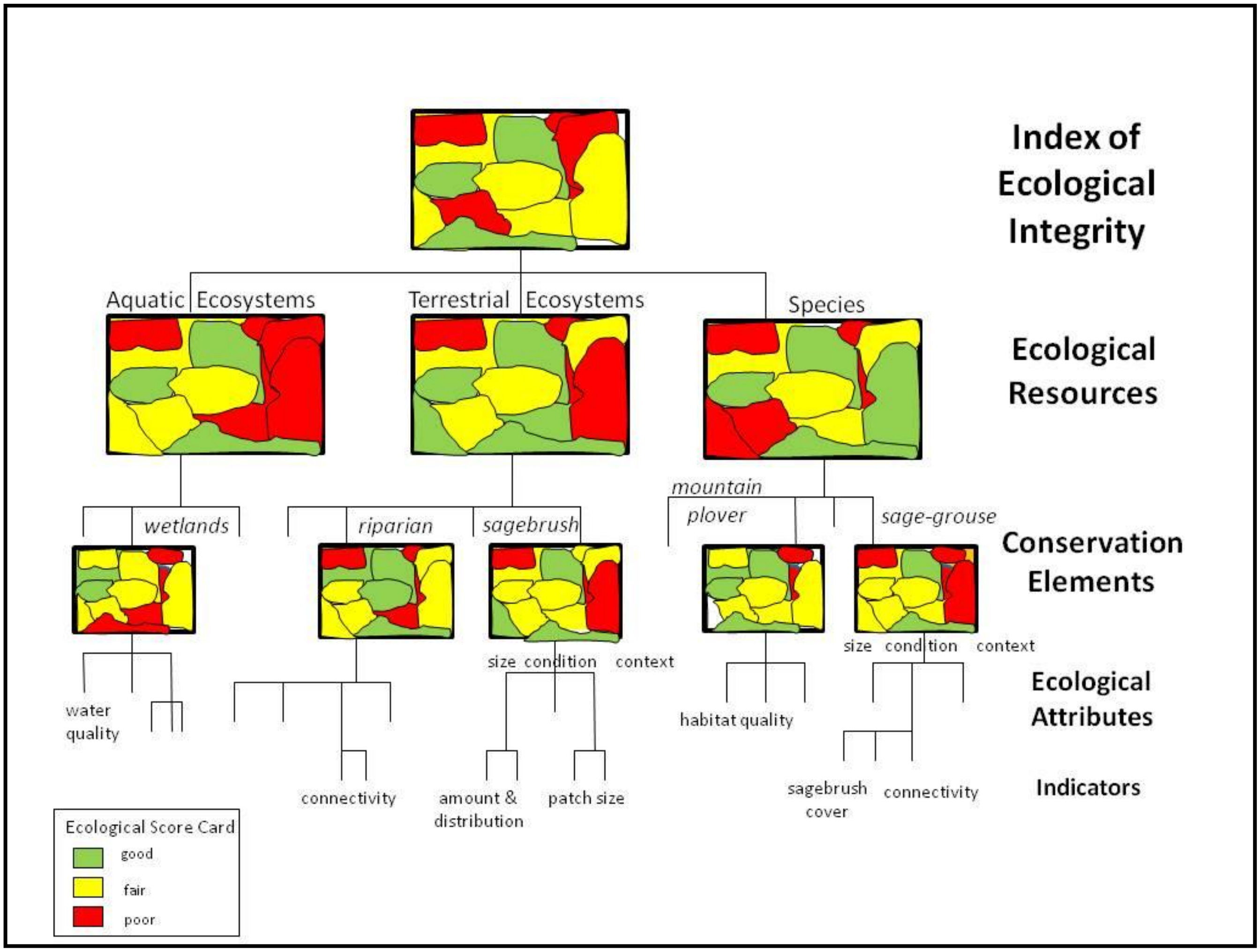


Figure 1. Hypothetical Ecological Integrity Assessment Framework. Conceptual models are used to identify key ecosystem components (conservation elements) for three ecological resource values: aquatic ecosystems, terrestrial ecosystems, and species. Assessments for individual conservation elements are aggregated into component scores for priority ecological resources and for an overall index of ecological integrity. Spatially explicit status assessments are provided for multiple levels in the hierarchy (see Table 1 for definitions of each level). For simplicity these maps are provided only for the upper levels of the hierarchy in this figure. Additionally, only a few conservation elements, ecological attributes, and indicators are identified at each level in this example, but multiple factors may be identified for each hierarchical branch. Change agents (e.g., fire, invasive species, development, climate change) are evaluated for each conservation element. Ecological attributes for each conservation element correspond to size, condition, and landscape context. For each indicator, acceptable ranges of variation are determined (represented by the ecological score card), thereby providing a spatially explicit evaluation system. In each map, polygons represent the 5th level HUC, which is the reporting unit for the REA. See Table 2 for additional examples of each hierarchical level.

Table 1. Definitions of hierarchical components of ecological integrity*

Term	Definition
Index of Ecological Integrity	A complementary, integrated suite of *Conservation Elements* that collectively represent important ecological components of an ecosystem.
Conservation Elements	A limited number of *species and communities* that represent critical components of ecosystems.
Coarse filter Conservation Elements	*Aquatic and Terrestrial Communities or Ecosystems* that collectively represent the ecological integrity of the ecosystem and are presumed to represent the habitat requirements of most plant and animal species of the ecoregion.
Fine filter Conservation Elements	*Species* whose health and population dynamics vary in response to critical agents of change, This may include sensitive or specialized, but regionally significant species, which are not protected by coarse filter elements.
Ecological Attributes	Defining characteristics of *Conservation Elements* that are especially pivotal, influence other characteristics of the Conservation Element, and affect long-term persistence or viability.
Indicators	Measurable components of a system whose characteristics are used to assess the condition of *Ecological Attributes*

*Adapted from SOW and Parrish et al. (2003).

effects of anthropogenic changes across an ecoregion (Tierney et al. 2009). The assessments of ecological integrity can help managers identify landscape configurations that balance natural and cultural goals, as well as address potential effects of stressors (Tierney et al. 2009).

In addition to the overall index of ecological integrity, information is summarized for multiple organizational levels (Figure 1). This allows flexibility in the application of assessment products for step-down assessments at local levels, where more finely scaled evaluations and additional information (e.g., riparian or allotment assessments) may be relevant to site-level planning. The index of ecological integrity thus provides the larger information context for addressing cumulative impacts or regional planning and monitoring across jurisdictional boundaries. Assessments of ecological integrity will be compatible with, but will not replace, site-level information.

A goal of an REA is to provide information that will facilitate the decision-making process related to regional resource values and uses. An index of ecological integrity can be used to:

- Help decision-makers identify priority areas for conservation, restoration, or monitoring activities.
- Provide information for adaptation and mitigation planning in response to climate change and environmental compliance.
- Provide information for proposed resource management strategies and actions, and cumulative impact assessments under the National Environmental Policy Act (NEPA).
- Establish baseline information for long-term monitoring of regional ecological conditions.

- Provide integrated landscape-scale information, understanding, and awareness to inform planning and decision-making on public lands.
- Provide an interdisciplinary currency (i.e., ecological integrity) to promote effective and efficient collaboration and cooperation among resource managers and other interested parties.
- Initiate more detailed sub-assessments (i.e., inventories, research, monitoring).

Ecological Integrity Assessment Framework- Details and Examples

In this section we expand on the overall framework presented above and describe the basic steps to assess ecological integrity for the REAs. It is anticipated that differences among ecoregions in ecological contexts and available data will result in some variation in specific approaches to assess ecological integrity. Consequently, it is especially important to include full documentation of the models and criteria used for each of the basic steps as applied for each REA. The overall framework and steps described below are intended to improve common understanding and to facilitate consistency across ecoregions.

Step I. Develop overall ecological models for the ecoregion to identify key change agents, conservation elements, and essential ecosystem characteristics and functions, and then describe interactions between each of the components. These models should address the primary system components and functions and include individual models for the ecoregion, terrestrial, and aquatic systems (e.g., Britten et al. 2007 pp 28-34.) and help to identify key vegetation communities. In addition, the models should identify assumptions and specify relevant spatial/temporal scales.

The overall models will serve as a guide for developing additional conceptual models for selected conservation elements. Conceptual models represent our current understanding of the system and help to ensure that key ecological processes and patterns are addressed by guiding the selection of appropriate conservation elements and associated ecological attributes. Published conceptual models should be used in the REAs, or models should be adapted/developed based on published literature.

Step II. Identify potential Conservation Elements. Conservation elements for assessing ecological integrity represent essential ecosystem components within an ecoregion. Conservation elements include both coarse and fine filter elements. The selection of suitable conservation elements and associated ecological attributes and indicators is a challenging process (Dale and Beyeler 2001, Doren et al. 2009), yet the usefulness of the REA is conditional on the validity of this selection process. **Establishing and documenting criteria are essential** for selection of appropriate conservation elements. In addition, it is important to document how the criteria were applied to refine the list of conservation elements. A manageable number of conservation elements for an index of ecological integrity is approximately 12 (Unnasch et al. 2008 p22).

Potential criteria are provided below. Conservation elements:

- Are regionally significant and can be evaluated at the 5th-level HUC scale.
- Include both terrestrial and aquatic components.
- Are sensitive to change agents and respond to them in a predictable manner; signify existing or impending changes relevant to the entire ecological system.
- Are complementary and integrative, whereby the full suite of conservation elements provides measures of key gradients with minimal duplication.
- Respond to variability at scales that makes it applicable to a large portion of the entire system.

Coarse filter: The dominant and/or regionally significant communities will serve as conservation elements, but these elements should be inclusive rather than exclusive of subclasses of vegetation cover types or rare communities. Thus, broad cover classes or community types should be selected. Uncommon cover types that have high ecological values (e.g., riparian and wetlands, high amounts of endemism) should also be considered. Collectively, the coarse-filters should represent all major ecosystem types in the ecoregion.

Fine filter: Regionally significant species will serve as potential fine filter elements. Potential species should be compared against coarse filter conservation elements such that only species not adequately represented by coarse filter elements will be included as fine filter elements. In some cases, species assemblages may be selected, such as species that individually may not be regionally significant, but collectively represent important ecological attributes of the system (e.g., native cold-water fishes). Selection criteria should be established and documented for all candidate fine filter species. The preliminary list of species should be revised in relation to coarse filter conservation elements and ecological models (Figure 2). In later steps, the list of fine filter conservation elements might be refined based on availability of data.

Step III. Develop conceptual models for each Conservation Element. Conceptual models for individual conservation elements are used to identify ecological attributes and provide greater details and resolution than provided in the overall ecoregion conceptual models (e.g., see Miller et al. 2010). Conceptual models for conservation elements should include important characteristics, relevant change agents, relevant spatial scales, and linkages to the ecoregion conceptual model. It should be determined whether the level of knowledge is sufficient for establishing ecological attributes and indicators for each conservation element (see Table 4 in Doran et al. 2009). In addition to the primary change agents identified by the SOW, the conceptual models may identify additional change agents (e.g., pathogens such as bark beetles, flood regimes) that are important drivers of the system or may introduce new stresses to the system especially if they are altered by human activities.

Although the concept of ecological integrity includes fundamental ecological processes, these processes are often difficult to assess, especially given the time constraints and broad spatial scales of an REA. Processes or functions include colonization/extinction, disturbance, succession, patch dynamics, and erosion/flooding. To select indicators of ecosystem function, ecological models can help identify relevant structure and composition characteristics of the ecosystem that reflect the state of the underlying processes; for example, habitat connectivity (Noon 2003).

Table 2. Examples of potential Conservation Elements and associated Ecological Attributes/Indicators.

Conservation Element	Key Ecological Attribute	Indicators	Potential Basis for Rating.[1]
Terrestrial communities (e.g., aspen, sagebrush, riparian)	Amount and distribution (*size*)	Mapped occurrence (e.g., presence/absence of community by pixel).	Provides baseline maps for evaluating condition and landscape context. Comparisons to historic maps can be used to evaluate condition.
	Patch size (*size*)	Size of patches (by community type)	Size distribution of patches relative to baseline maps under reference conditions.
	Composition (*condition*)	Percent cover of invasive species	Relative ranking based on percent cover of invasive species
	Connectivity (*landscape context*)	Area of community connected within a specified distance (e.g., 500 m).	Relative ranking based on total area.
	Landscape pattern (*landscape context*)	Land use intensity	Relative ranking based on road density
Aquatic : Ground Water Surface Water	Hydrologic function (*condition*)	Priority aquifers Fish habitat 303D listings	Ranking of quality waters/habitats for cold water fish species.
Terrestrial sensitive species (e.g., greater sage-grouse, mountain plover)	Habitat amount and distribution (*size*)	Presence/absence based on species distribution maps	Size distribution of patches relative to baseline habitat maps.
	Habitat quality (*condition*)	Habitat suitability index	Relative ranking based on species abundance.
Aquatic sensitive species (e.g., Razor-backed sucker, CO cutthroat trout)	Habitat amount and distribution (*size*)	Presence/absence based on species distribution maps	Provides baseline maps for evaluating condition and landscape context. Comparisons to historic maps can be used to evaluate condition.
	Hydrologic regime (*condition*)	Habitat suitability index based on flow regimes.	Relative ranking based on departure from historic flows.

[1] Ratings are based on acceptable ranges of variation for a given assessment scale (e.g., 5th level HUC). Status is characterized by attributes and indicators for size, condition, and landscape condition (indicated in italics).

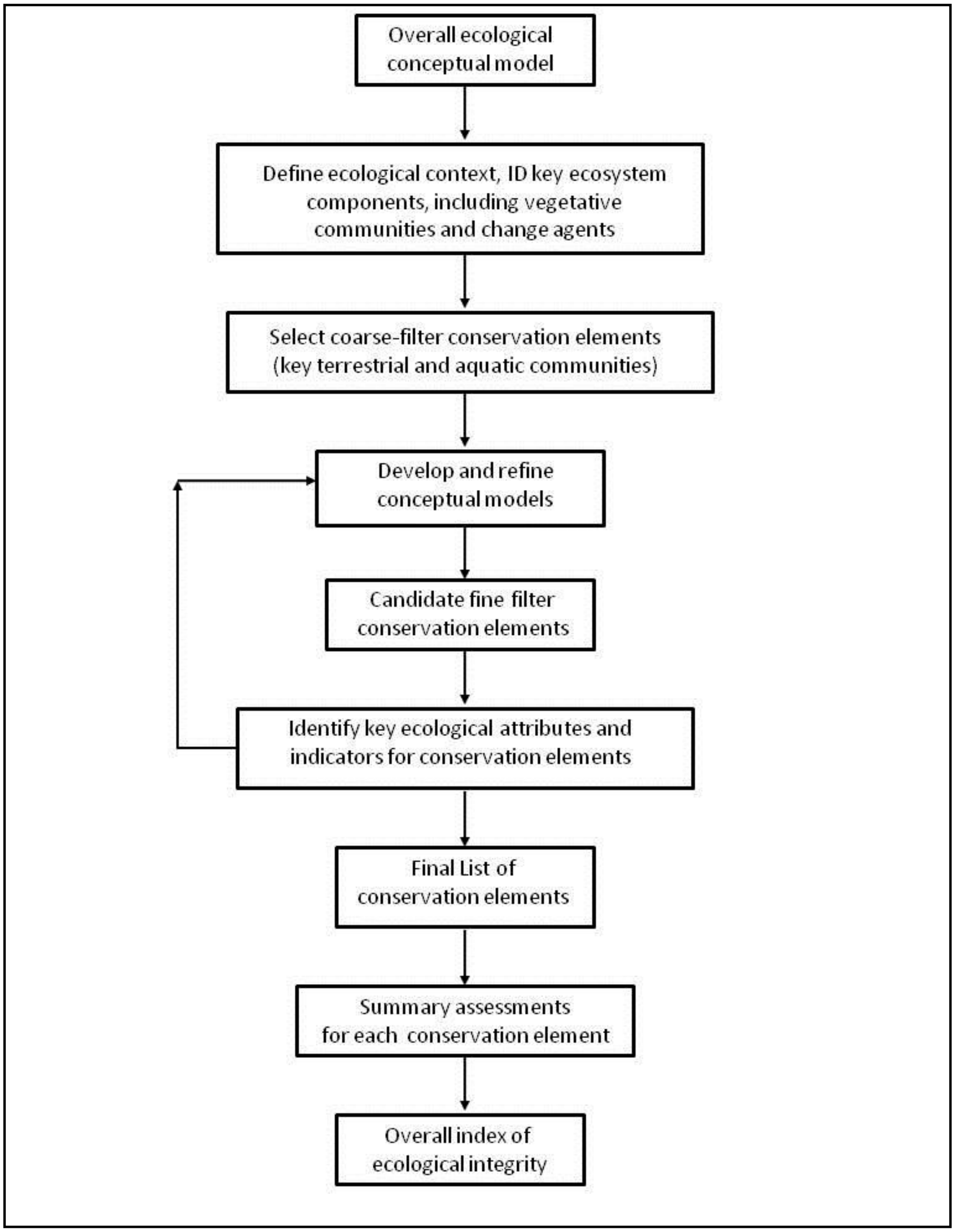


Figure 2. Ecological Integrity Assessment Framework. The process begins with the development of ecological models, which provides the ecological context and will help identify critical components of the ecosystem to be assessed. The process is iterative and contains many feedbacks to refine and identify the final list of ~12 conservation elements representing key ecosystem components, while limiting duplication among elements. To create a transparent and scientifically based process, it is essential to provide documentation for conceptual models, criteria used for selecting conservation elements, and application of selection criteria to justify inclusion/exclusion of potential conservation elements.

Step IV. Identify a limited suite of key ecological attributes, indicators, and acceptable range of variation for each indicator. The conceptual models for each conservation element guide the selection of key ecological attributes. Ecological attributes and associated indicators, at both fine and coarse filter levels, should reflect **size, condition, and landscape context**, and may include biological characteristics, ecological processes, environmental regimes, and aspects of landscape structure that sustain the conservation element (Table 2). Measurable indicators and an acceptable range of variation for each ecological attribute are identified for assessing status and trends.

Possible criteria for selection of indicators include (adapted from Unnasch et al. 2008):

- Are easily measured in a reliable, repeatable, and accurate fashion.
- Are unambiguously associated with ecological attributes.
- Are sensitive to change agents or other stressors at relevant spatial and temporal scales.
- Are comprehensive and complementary.
- Are scientifically defensible and interpretable in common language.

If there is more than one indicator, they are compiled for each ecological attribute, and attributes in turn are compiled for each conservation element. To address the differences in magnitude of indicator values, it may be useful to standardize indicator values to range between 0 and 1 before compiling.

For each conservation element, acceptable ranges of variation will be summarized at the 5th HUC level. Because of uncertainty and information gaps, an acceptable range of variation is often difficult to determine and in many cases will be based on our best, but limited, understanding of the system. The resulting ecological scorecard (Figure 1) is therefore a qualitative evaluation of resource condition derived from quantitative assessments. Possible approaches for integrating indicators and defining acceptable ranges of variation can be found in Parrish et al. (2003) and Section IV, Unnasch et al. (2008), with the caveat that not all examples may be relevant for ecoregion assessments.

Individual indicators should be evaluated at appropriate spatial scales, although they can also be summarized at the 5th HUC level. This is important because ecosystem processes and constraints that affect conservation elements can operate at different spatial scales (Parrish et al. 2003). For example, sage-grouse habitat might be quantified relative to home range size (e.g., Kotliar et al. 2008). Indicators can be summarized across a range of relevant spatial scales, including the 5th level HUC. Assessments at multiple scales will facilitate analysis of cumulative impacts at a particular spatial scale.

Step V. Develop an index of ecological integrity: Ecological integrity is summarized for multiple levels of biological organization in a hierarchical fashion (Figure 1). At the upper levels of the hierarchy, qualitative assessments (based on the ecological score card) for conservation elements are summarized at two levels: 1) for each of three priority ecological resources (aquatic and terrestrial communities, species) and 2) integrated into an overall index of ecological integrity. The qualitative assessments at these levels allow integration of multiple components of ecological integrity in a format that is easy to interpret and provides a spatially explicit evaluation framework for the ecoregion. Qualitative assessments are derived from underlying quantitative assessments (based on indicators at the original scales of evaluation) for each conservation element or attribute. This approach provides ready access to the quantitative basis for assessment of ecological resource condition and the overall index of ecological integrity

Although the index of ecological integrity is an important assessment product, individual spatially explicit assessments for each level of the hierarchy are retained to provide maximum flexibility for end users. This will permit identification of conservation elements that may have the greatest influence on the overall score, but will also highlight areas with extreme scores that are overshadowed by averaged composite scores. In particular, correspondence in high (or low) scores among all levels of the hierarchy may indicate areas of high conservation (or restoration) potential. The underlying data layers can also provide step-down data summaries for specific management issues or to inform species management, whereas the composite scores provide an overall view of the condition of ecological resources and represent cumulative effects of stressors.

One benefit of the hierarchical approach is that status assessments at different levels of the hierarchy are necessary to view the system holistically. This is particularly important for evaluating vegetation communities, which are not static entities, but rather comprise a shifting mosaic. While it is useful to assess the status for individual cover types to determine if one is particularly vulnerable or degraded, it is also important to view each cover type in the context of the ecosystem as a whole. Many processes occur across vegetation patch types (e.g., migration, colonization, fire). Thus, ecosystem-level assessments are important for understanding larger-scale processes.

Management Questions

The REAs will address priority management issues for BLM. This is accomplished by using management questions, which were developed by resource managers and decision-makers for each ecoregion and were framed with respect to the three ecological resources and four change agents. The management questions provided in the SOW largely address priority information needs for individual species (fine filter), which is a subset of ecological integrity. Thus, the ecological integrity assessment framework and the use of conceptual models provide the broader ecological context to address priority management issues. The development of linkages between conceptual models and management questions helps to ensure that important ecosystem components at multiple levels of organization are evaluated as a part of the REA. Scores for each level of the assessment framework may be useful in identifying important factors to monitor or manage, and consequently provide information that may be used to evaluate success of management actions.

Summary of Key Points:

- This document provides a general framework and guidance on the ecological integrity assessment framework for the REAs.
- The REAs address three ecological resources (aquatic ecosystems, terrestrial ecosystems, native species) and four change agents (fire, development, invasive species, climate change).
- The ecological integrity assessment framework is based on conceptual ecological models, which inform the selection of coarse filter and fine filter conservation elements.
- The conceptual models identify and develop linkages among key ecosystem components and change agents.
- Conservation elements are complementary, address the primary drivers/stressors for the system, and represent a range of spatial/temporal scales.
- To create a transparent and scientifically based process, it is essential to provide documentation for conceptual models, criteria used for selecting conservation elements, and how the criteria were applied.
- A hierarchical framework for evaluating ecological integrity for the REA is centered on conservation elements and builds from quantifiable indicators of important ecological attributes for these conservation elements.
- Quantitative assessments of indicators are used to develop ecological scorecards for conservation elements, which are aggregated into summary scores for the ecological resources and integrated into an overall index of ecological integrity.
- Information is summarized for multiple organizational levels to allow flexibility in the application of assessment products for step-down assessments at local levels, where more finely scaled evaluations and additional information may be relevant to site-level planning.
- The ecological integrity assessment framework provides the regional context for addressing cumulative impacts, informs planning and monitoring across jurisdictional boundaries, and is compatible with site-level information.

Literature Cited

Britten, M.E., Schweiger, W., Frakes, B., Manier, D., and Pilmore, D. 2007. Rocky Mountain Network Vital Signs Monitoring Plan. Natural Resource Report NPS/ROMN/NRR-2007/010. National Park Serivice, Fort Collins, CO. 146 p.

Dale, V.H., and Beyeler, S.C. 2001. Challenges in the development and use of ecological indicators. Ecological Indicators 1: 3-10.

Dale, V.H., Peacock, A.D., Garten, C.T., Jr., Sobeck, E., and Wolfe, A.K. 2008. Selecting indicators of soil, microbial, and plant conditions to understand ecological changes in Georgia pine forests. Ecological Indicators 8:818-827.

Doren, R.F., Trexler, J.C., Gottlieb, A.D, et al. 2009. Ecological indicators for system-wide assessment of the Greater Everglades Ecosystem restoration program. Ecological Indicators 9S: S2-S16.

Kotliar, N.B., Bowen, Z.H., Ouren, D.S., and Farmer A.H. 2008. A regional approach to wildlife monitoring related to energy exploration and development in Colorado: US Geological Survey Open-File Report 2008-102, 66 p. (http://pubs.usgs.gov/of/2008/1024/pdf/OF08-1024_508.pdf)

Noon, B.R. 2003. Conceptual issues in monitoring ecological resources. In: Busch, D.E., Tresler, J.C. (eds). Monitoring Ecosystems: Interdisciplinary Approaches for Evaluating Ecoregional Initiatives. Island Press, Covelo, CA. pp 27-72.

Noss, R.F. 1987. From plant communities to landscapes in conservation inventories: a look at The Nature Conservancy (USA). Biological Conservation 41:11-37.

Miller, D.M., S.P. Finn, A. Woodward, A. Torregrosa, M.E. Miller, D.R. Bedford, and A.M. Brasher. 2010. Conceptual Ecological Models to Guide Integrated Landscape Monitoring of the Great Basin. U.S. Geological Survey Scientific Investigations Report 2010:5133, 134 p.

Parrish, J.E., Braun, D.P., and Unnasch. R.S. 2003. Are we conserving what we say we are? Measuring ecological integrity within protected areas. BioScience 53: 851-860.

Poiani, K.A., Richter, B.D., Anderson, M.G., and Richter H E. 20000. Biodiversity conservation at multiple scales: Functional sites, landscapes, and networks. BioScience 50:133-46.

Tierney, G. L., Faber-Langendoen, D., Mitchell, B.R., Shriver, W.G., and Gibbs, J.P. 2009. Monitoring and evaluating the ecological integrity of forest ecosystems. Frontiers in Ecology 7:308-316.

Unnasch, R.S., Braun, D.P., Comer, P.J., Eckert, G.E. 2008. The ecological integrity assessment framework: a framework for assessing the ecological integrity of biological and ecological resources of the national park system, version1.0. Unpublished report to the National Park Service. Accessed December 2010 from http://www.natureserve.org/publications/NPS_EcologicalIntegrityFramework_January09.pdf

APPENDIX 12. Attribute and Indicator Table for Sonoran Desert Conservation Elements

Conservation Element	Key Ecological Attribute	Indicator	Indicator Rating				Citation
			Poor	Fair	Good	Very Good	
Wildlife and Desired Species							
Mountain Lion	prey	ungulate density	low	medium	high	very high	Julander and Jeffrey (1964)
	habitat	cover & terrain	very dense or open cover	-	-	rugged terrain with mixed cover	Riley (1998)
	habitat degradation	human development	high human development	moderate human development	low human development	no human development	Van Dyke et al. (1986)
Desert Bighorn Sheep	habitat	distance to perennial water	>3.2 km		< 3.2 km		Smith et al. 1991, Turner et al. 2004
	winter range	snowpack depth	> 25 cm		< 25 cm		Smith et al. 1991
	summer range	area	< 227 km2		> 227 km2		Zeigenfuss et al. 2000
Golden Eagle	habitat loss or degradation	urban development	present	--	minimal	absent	Kochert and Steenhof (2002)
	habitat degradation	livestock grazing and agricultural development	existing or planned	--	--	absent	Beecham and Kochert (1975)

Conservation Element	Key Ecological Attribute	Indicator	Indicator		Rating		Citation
	habitat degradation	fire	>40,000 ha of shrublands burned	--	burned territory with adjacent vacant unburned territory	unburned territories	Kochert et al. (1999)
	habitat degradation	mining and energy development	present	--	--	absent	Phillips and Beske (1984)
	habitat	vegetation	disturbed areas, grasslands, agriculture			shrubland	Marzluff et al. (1997), Peterson (1988)
	habitat/nest sites	topography	--	--	--	cliffs within 7 km of shruband	Menkens and Anderson (1987), McGrady et al. (2002), Cooperrider et al. (1986)
	mortality	infrastructure (roads, power lines, wind turbines	--	--	--	infrastructure absent	Franson et al. (1995)
	illness/mortality	poisoning from pesticides and other toxins	high levels of contaminants	--	--	low/no contaminants	Craig and Craig (1998), Franson et al. (1995), Harmata and Restani (1995), Kramer and Redig (1997), Pattee et al. (1990)

Conservation Element	Key Ecological Attribute	Indicator	Indicator	Rating			Citation
	mortality	shooting	Occurs	--	--	doesn't occur	Beans (1996)
Mule Deer	habitat degradation	distance from wells	<2.7 km	-	-	>3.7 km	Sawyer et al. (2006)
	habitat degradation	distance from roads	>200m	-	-	>500 m	
Kit Fox	habitat	plant community	dense brush, woodlands, forest	barren	-	shrib, shrub-grass	McGrew (1979), Cypher (2003)
	habitat	terrain	steep, rugged >20% slope	rolling 11-20% slope	slight slope 5-10%	flat <5% slope	Cypher (2003)
	thermal biology	freeze-free season	<100 days	100-119 days	-	>120 days	McGrew (1977)
Lucy's Warbler	Breeding Habitat	Density of mesquite			high		Johnson et al. 1997
	Nesting Habitat	Density of nesting cavities			high	very high	
	Habitat	elevation	<20m		<2,133m		
Southwest Willow Flycatcher	habitat	proximity to surface water	>100 m	50-100 m	25-50 m	0-25 m	Sogge and Marshall (2000)
	connectivity	distance between occupied sites	>30 km	15-30 km	2-15 km	<2 km	Finch et al. (2002)
	habitat degradation	recreation	present in high intensity	present in moderate intensity	present in low intensity	absent	Marshall and Stoleson (2000)
LeConte's Thrasher	habitat	invasive					

Conservation Element	Key Ecological Attribute	Indicator	Indicator		Rating		Citation
	degradation	grasses					
	habitat	plant community				desert flats, washes and alluvial fans with sandy soils	Grinnel and Miller (1944)
	habitat degradation	OHV, grazing, energy development					
Bell's Vireo	habitat	riparian vegetation					
	habitat degradation	water diversion					
	parastism	cowbird abundance					
Desert Tortoise - Sonoran	habitat	size	<200 sq mi	200-500 sq mi	500-1,000 sq mi	>1,000 sq mi	Brussard et al. (1994)
	predation	common ravens and other predators	abundant	fairly common	rare	absent	Brussard et al. (1994)
	habitat degradation	exotic ephemerals	abundant, ineradicable	fairly common and widespread	scarce and patchy	none	Brussard et al. (1994)
Desert Tortoise - Mojave	habitat	size	<200 sq mi	200-500 sq mi	500-1,000 sq mi	>1,000 sq mi	Brussard et al. (1994)
	predation	common ravens and other predators	abundant	fairly common	rare	absent	Brussard et al. (1994)

Conservation Element	Key Ecological Attribute	Indicator	Indicator Rating				Citation
	habitat degradation	exotic ephemerals	abundant, ineradicable	fairly common and widespread	scarce and patchy	none	Brussard et al. (1994)
Lowland Leopard Frog	habitat	elevation	>8,200 ft	6,400 - 8,200 ft	-	<6,400 ft	AZ Game and Fish (2006)
	predation	American bullfrog	present	-	-	absent	Jennings (1994)
	habitat	distance to roads	<100 m	100-200m	200-300m	>300 m	RLFCT (2004)

APPENDIX 13. Final AMT-Approved Sonoran Desert REA Management Questions 3-18-11

A. SOILS, BIOLOGICAL CRUSTS, AND FORAGE MANAGEMENT

1. Where are soils susceptible to wind and water erosion?
2. Where are sensitive soils (including saline, sodic, gypsiferous, shallow, low water holding capacity) and highly productive (higher clay content, hydric) soils?
3. Which HMAs and allotments may experience significant effects from change agents including climate change?

B. SURFACE AND GROUNDWATER MANAGEMENT QUESTIONS

1. Where are lotic and lentic surface waterbodies and livestock and wildlife watering tanks and artificial water bodies?
2. Where are perennial streams and stream reaches?
3. Where are the alluvial aquifers and their recharge areas (if known)?
4. Where are aquatic systems listed on 303(d) with degraded water quality or low macroinvertebrate diversity?
5. Where are surface water flows likely to increase or decrease in the near-term, 2025 (development), and long-term, 2060 (climate change)?
6. What is the location/distribution of these aquatic biodiversity sites?
7. What are seasonal maxima and minima discharges for the Colorado River and major tributaries at gaging stations?

C. ECOLOGICAL SYSTEMS MANAGEMENT QUESTIONS

1. Where are existing vegetative communities?
2. Where are vegetative communities likeliest to vulnerable to change agents in the future?
3. What change agents have affected existing vegetation communities?

D. SPECIES CONSERVATION ELEMENT MANAGEMENT QUESTIONS

1. What is the most current distribution of available occupied habitat (and historic occupied habitat if available), including breeding, seasonal habitat, and movement corridors and bottlenecks (as applicable)?
2. What areas known to have been surveyed and what areas have not known to have been surveyed (i.e., data gap locations)?
3. Where are potential habitat restoration areas?
4. Where are potential areas to restore connectivity?
5. What is the location/distribution of terrestrial biodiversity sites?
6. What aquatic and terrestrial species CEs and high biodiversity sites and movement corridors are vulnerable to change agents in the near term horizon, 2020 (development, fire, invasive species) and a long-term change horizon, 2060 (climate change)? Where are these species and sites located?
7. What is the location/distribution of aquatic biodiversity sites?
8. Where are HMAs located?

E. WILDFIRE MANAGEMENT QUESTIONS

1. Where are the areas that have been changed from wildfire from 1999–2009?
2. Where are the areas with potential to change from wildfire?
3. Where are fire-adapted communities?

F. INVASIVE SPECIES MANAGEMENT QUESTIONS

1. Where are tamarisk, buffelgrass, red brome, Sahara mustard, quagga and zebra mussel, and Asiatic clam present?
2. Where are the areas of potential future encroachment from this invasive species?
3. Where are areas with restoration potential?

G. FUTURE DEVELOPMENT MANAGEMENT QUESTIONS

1. Where are current locations of these development types?
2. Where are areas of planned development (e.g., plans of operation, urban growth, transmission corridors, governmental planning)?
3. Where are areas of potential development (e.g., under lease), including renewable energy sites and transmission corridors and where are potential conflicts with CEs?

H. RESOURCE USE MANAGEMENT QUESTIONS

1. Where are high-use recreation sites, developments, roads, infrastructure or areas of intensive recreation use located (including boating)?
2. Where are areas of concentrated recreation travel (OHV and other travel) located?
3. Where are permitted areas of intensive recreation use (permit issued)?
4. Where are allotments and where is most forage available?

I. AIR QUALITY MANAGEMENT QUESTIONS

1. Where are the viewsheds adjacent to designated scenic conservation areas?
2. Where are the viewsheds most vulnerable to change agents?
3. Where are the designated non-attainment areas and Class I PSD areas?

J. CLIMATE CHANGE MANAGEMENT QUESTIONS

1. Where/how will the distribution of dominant native plant and invasive species be vulnerable to or have potential to change from climate change in 2060?
2. Where are areas of potential species (conservation elements) distribution change between 2010 and 2060?
3. Where are aquatic/riparian areas with potential to change from climate change?
4. Where are areas of potential surface water flow change?

Total: 41 Management Questions