Colorado Plateau

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 Colorado Plateau 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 select 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 Colorado Plateau.

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 Colorado Plateau 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 Colorado Plateau 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 logic model development, the ecological integrity concept and geoprocessing models in Section 5.

Detailed conceptual models that accompany each example methods module in Section 4 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 Colorado Plateau 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*.

At Workshop 2, Data Identification and Evaluation, 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 the flannelmouth sucker (*Catostomus latipinnis*) as a representative of mid-elevation streams and the ferruginous hawk (*Buteo regalis*), a sensitive raptor and associate species of prairie dog towns (one of the Colorado Plateau REA major species assemblages)..

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 species-of-concern richness-summary map layer representing recorded G1 through G3 species occurrence data available from State Natural Heritage Programs. We have the option of organizing subsets of these data in different ways to include biodiversity hotspots and endemics. These richness function map layers are limited in that they only represent locations from which occurrences have been recorded, 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 list 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 complete Memo 3, the AMT is considering the recommendations and will soon finalize the full list of management questions.

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 Colorado Plateau 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: 2020 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 with the inspection and approval of approaches and methods during Tasks 3 and 4.

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 was linked to 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 stage 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 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 optimum 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 of these gaps have been filled since Workshop 2.

2.2.4 Results of Workshop 2, October 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 were strongly in favor of using the GAP data, which they considered more accurate. Fire specialists naturally preferred LANDFIRE for fire-related questions. Two 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. 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 a means of using both frameworks and integrating them into a common layer when necessary (See Section 5.3.2).

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

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3 CONSERVATION VALUES OF THE COLORADO PLATEAU

3.1 Terrestrial & 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 (1966) in describing the role starfish played in intertidal environments. Power et al. (1996) provided a more specific definition of the term as pertaining to those species that have a disproportionately large impact on ecosystems relative to their abundance or biomass. Species like the American beaver and American alligator are classic examples of keystone species having far greater impact on their ecosystems compared to their abundance. "Keystone ecosystems" is an extension of that concept to apply to features other than species. For example, salt licks, serpentine communities, old-growth forest groves, and caverns are good examples. For the arid Colorado Plateau ecoregion, water ecosystems—wetlands, springs, and river washes—and the linkages between aquatic and terrestrial landscapes are particularly important.

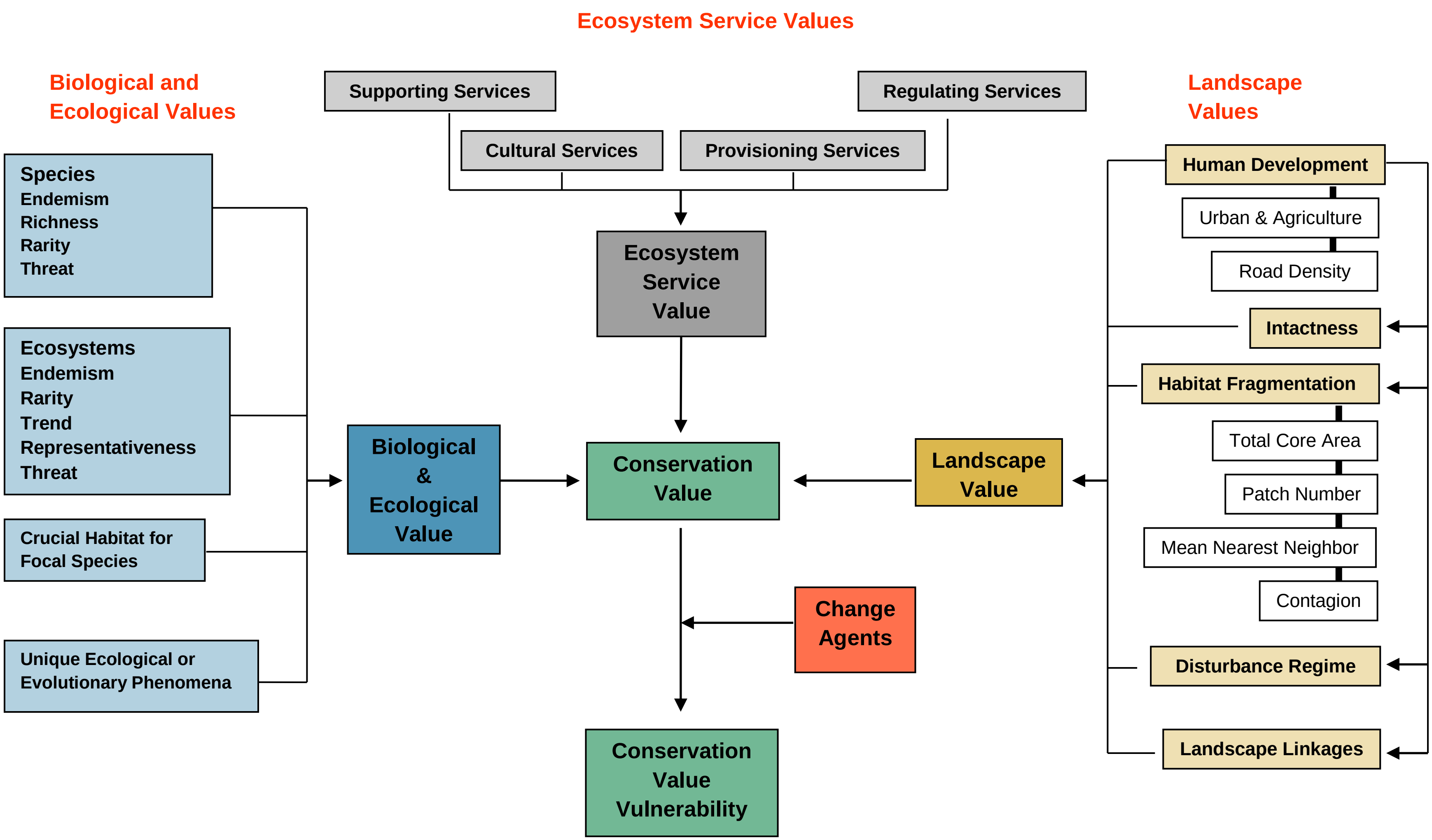

Figure 3-1. Conceptual diagram for assessing terrestrial conservation values featuring the major components that comprise overall conservation value and vulnerability to change agents.

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 make up 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 <u>not</u> 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 Capitol 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 – Soil Stability 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.

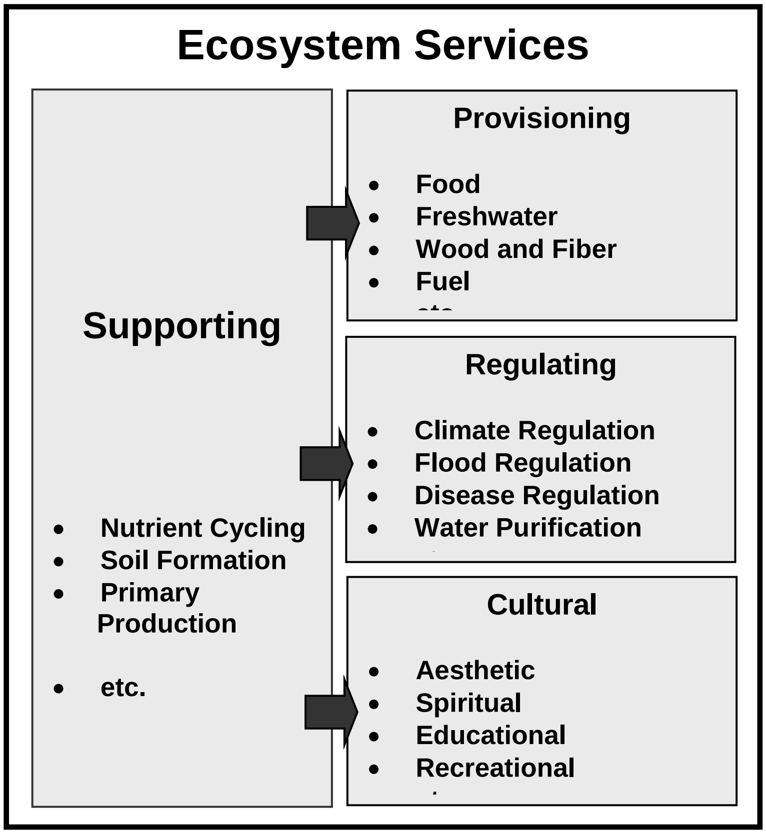

Figure 3-2. Diagram depicting the four classes of ecosystem services as described by The Millennium Assessment (2005).

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 Colorado Plateau 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.

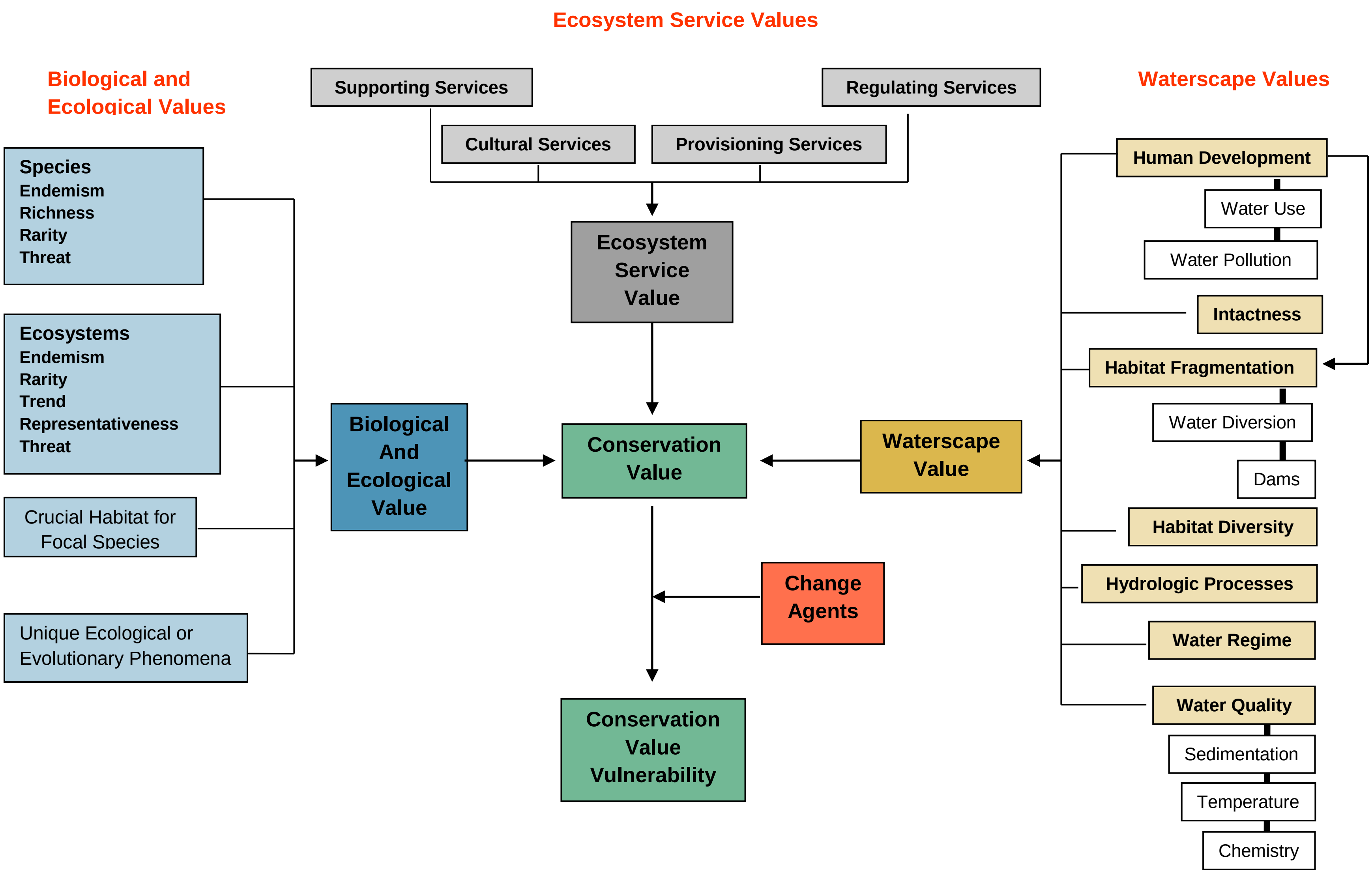

Figure 3-3. Conceptual diagram for assessing aquatic conservation values outlining the major components that makeup overall conservation value and vulnerability due to change agents.

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 3-4). 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.

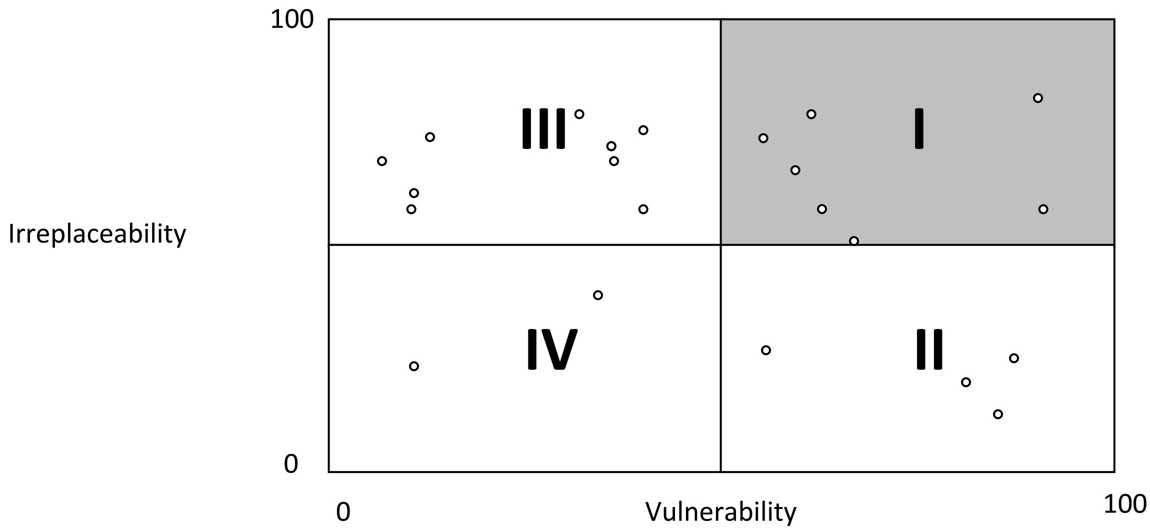

Figure 3-4. Irreplaceability and vulnerability matrix diagram identifying various categories of conservation priority of specific sites (denoted as small circles).

Our assumption is that for the Colorado Plateau 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).

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4 PRIMARY CHANGE AGENTS OF THE COLORADO PLATEAU

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, Figure 4-1). Forecasting future conditions is complex as change agents are often stochastic and uncertain.

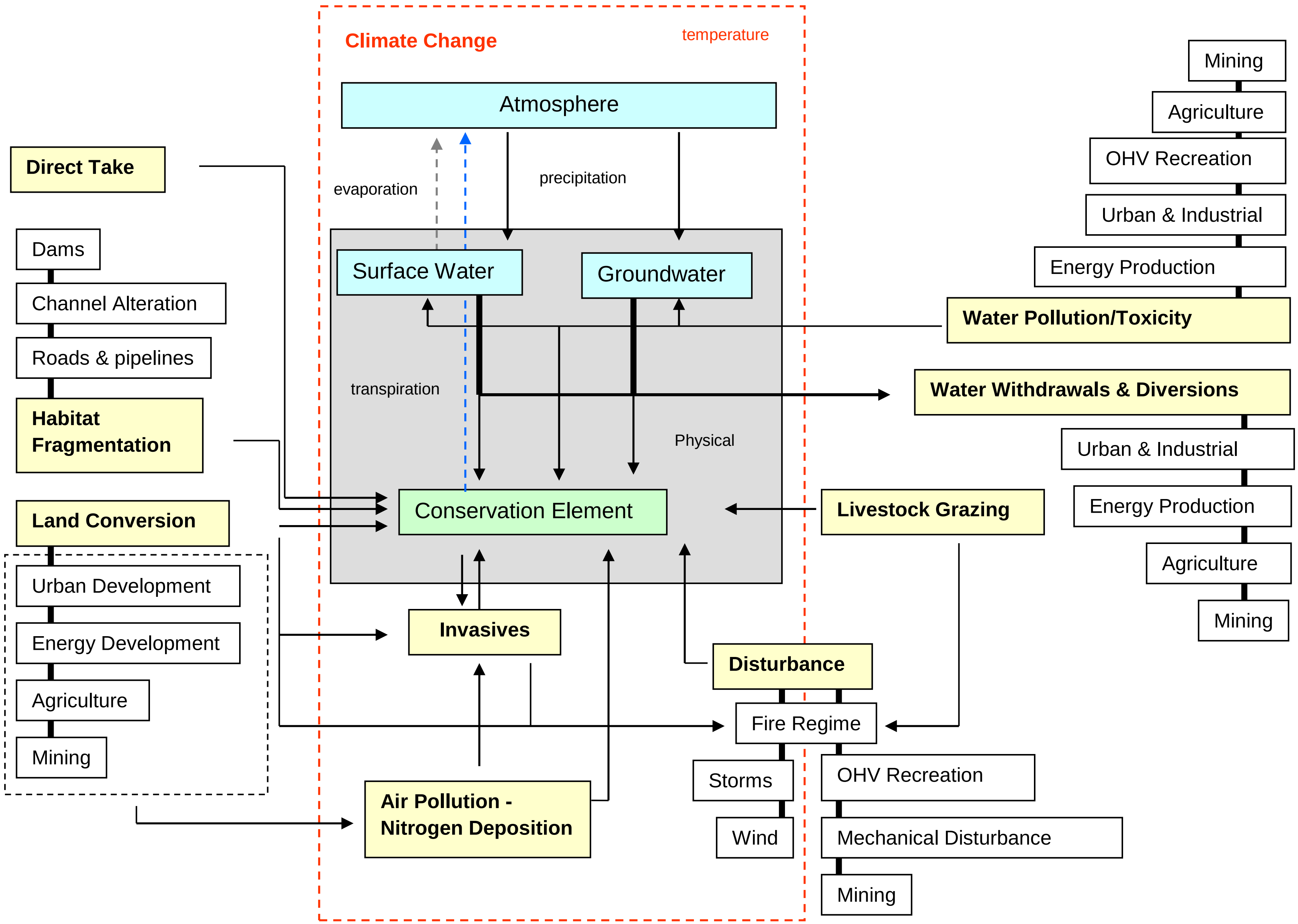

Figure 4-1. General conceptual model showing the influence of change agents on conservation elements.

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 the Fire Regime Condition Classifications (or what areas are most departed from their reference condition fire regimes?)
4. Where is fire adverse to ecological communities, features, and resources of concern?

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 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 Colorado Plateau includes a wide variety of vegetation types and associated fire regimes, ranging from conifer forest types with frequent fire (e.g., ponderosa pine, mixed conifer) to semi-desert shrubland types with rare fire under pre-settlement reference conditions. Periods of vegetation establishment (and subsequent effects on composition) and fire occurrence across the region have been quite dynamic over the period of record (Brown et al. 1999), and are in part related to periodic cycles in climate (Brown & Wu 2005). Recent alterations to fire regime include the following factors:

- Increased vegetation density and fuel loading in vegetation types with historically high fire frequency due to effective fire suppression
- Woodland encroachment into areas historically dominated by shrublands and subsequent effects to fuel load and fire severity
- Invasion by exotic plants and subsequent increases in fine fuels and connectivity

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

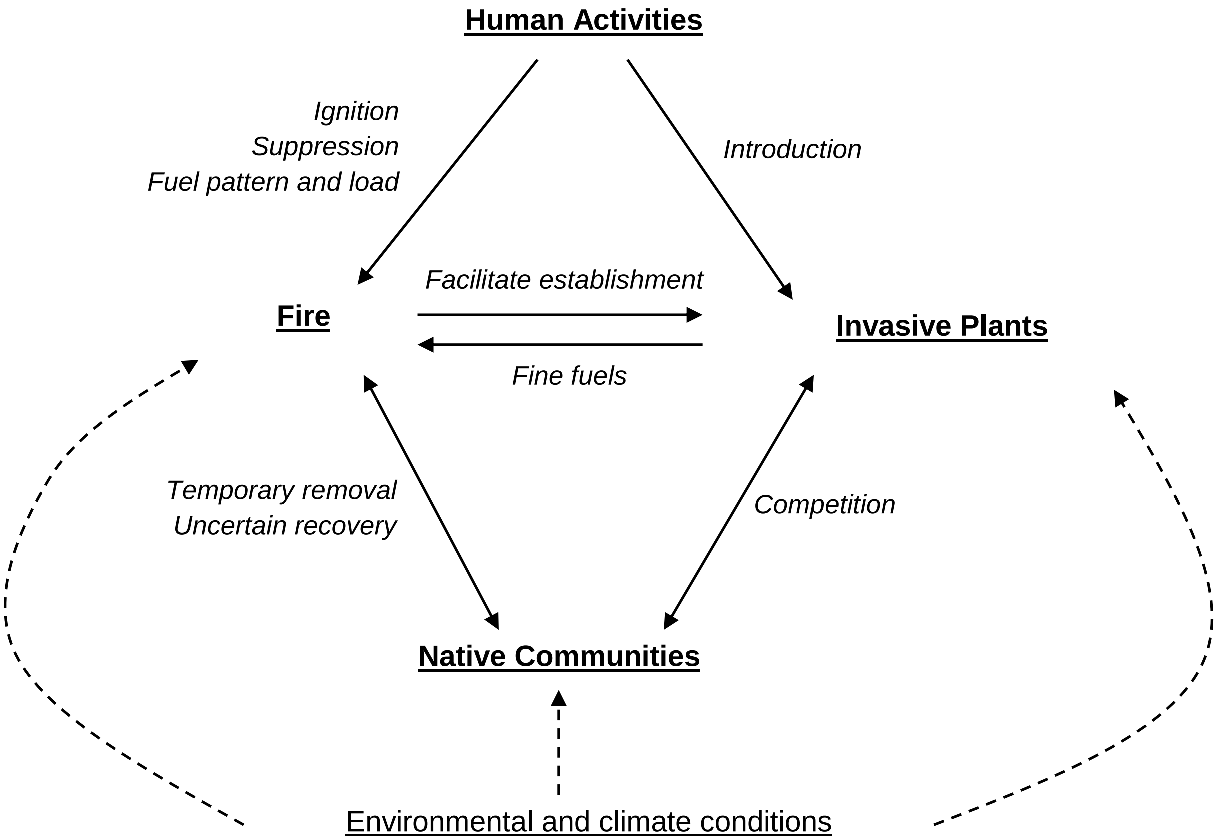

Figure 4-2. Conceptual diagram illustrating the interaction of drivers in wildfires.

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
Cheatgrass / 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)
Cheatgrass current and future distribution	1-4
Red brome current and future distribution	1-4 Occurs in S. Utah
Tamarisk 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 fire-intolerant communities in the Colorado Plateau, 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 in the Colorado Plateau), 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).

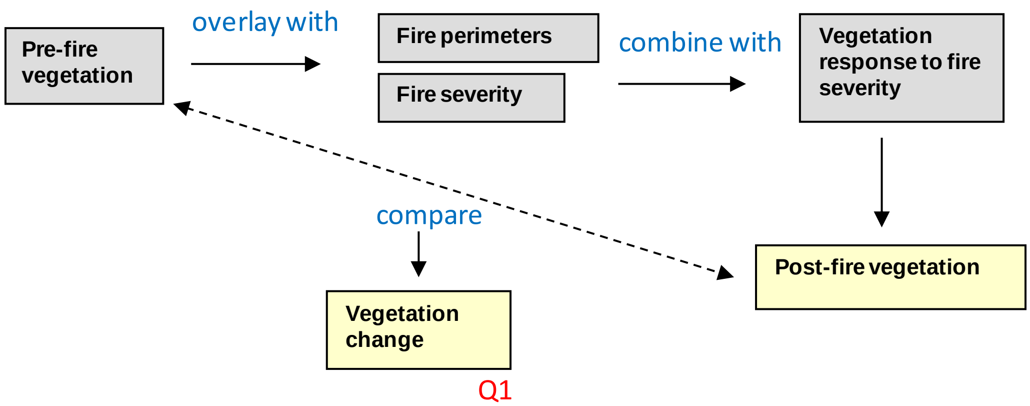

Figure 4-3. Process model for areas of recent change due to fire.

Management Question 2: Where are the 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 fire-intolerant communities in the Colorado Plateau, 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 therefore fire occurrence can be treated more simplistically than in vegetation communities in which fire was a more frequent disturbance under reference conditions.

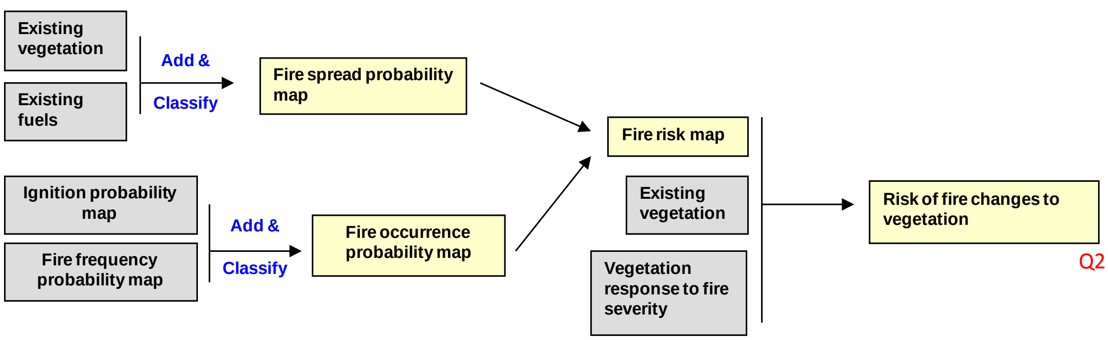

Figure 4-4. Process model for identifying areas with potential for alteration by fire.

Management Question 3: Where are the Fire Regime Condition Classifications or What areas are most departed from their reference condition fire regimes?

Identifying the areas most departed from their reference condition fire regimes will first require defining the reference condition fire regimes and their associated vegetation components (succession classes) that can be used as a basis of comparison. Then current fire regimes or their vegetation component proxies must be defined. Current conditions would then be compared to reference conditions within an appropriate landscape summary unit, and differences calculated. Previous efforts (e.g., LANDFIRE) have attempted to characterize the degree of departure of current conditions compared to reference conditions with the intent of characterizing changes in fire regime. However, such applications have been limited because data for historical fire regime are often lacking or not available for large areas, and current fire regime cannot be adequately expressed using only the most recent observations of fire location and severity. In the absence of comprehensive estimates of current fire regime, it is necessary to estimate the current conditions using the proportions of the areas that would have been occupied by a vegetation community under reference conditions (e.g., LANDFIRE BpS) that are now occupied by a series of succession classes (e.g., LANDFIRE SCLASS) defined in the context of those reference conditions (Figure 4-5). These proportions indicate where those succession classes are more or less abundant than would have been expected under reference conditions, and in particular cases can serve as a proxy for changes in fire regime (e.g., an overabundance of later-seral vegetation can indicate decreases in fire frequency for fire-tolerant vegetation communities).

Available estimates of current succession classes would need to be supplemented with additional information about recent fires and invasion by exotic plants where such information is available. Capturing the effect of recent fires is particularly important in fire-sensitive and fire-intolerant vegetation communities. Existing estimates of fire regime condition class, such as LANDFIRE FRCC, should be used with caution due to the potential under-representation of the influence of invasive plants and recent fires among other factors; however, some of the input data produced by LANDFIRE for calculating FRCC may be corrected to produce better estimates of ecological and fire regime departure.

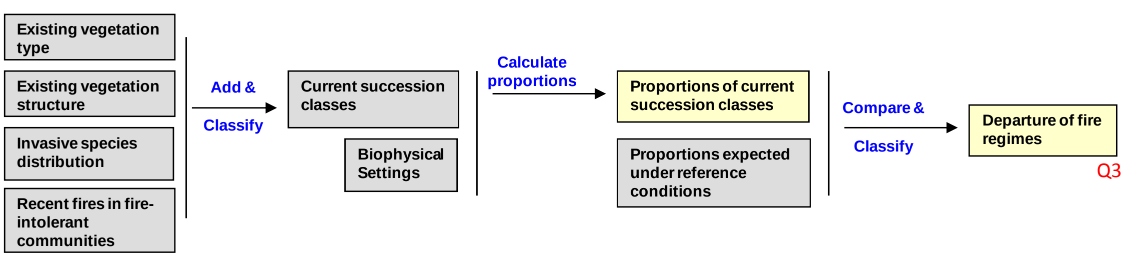

Figure 4-5. Process model for identifying areas with high departure from reference condition fire regimes.

Management Question 4: Where is fire adverse to ecological communities, features, and resources of concern?

To determine where vegetation communities may be adversely affected by fire, it is necessary to identify those communities which are sensitive or intolerant to increases in fire frequency or severity (Figure 4-6). For vegetation communities in which fire has historically been rare or absent, 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.

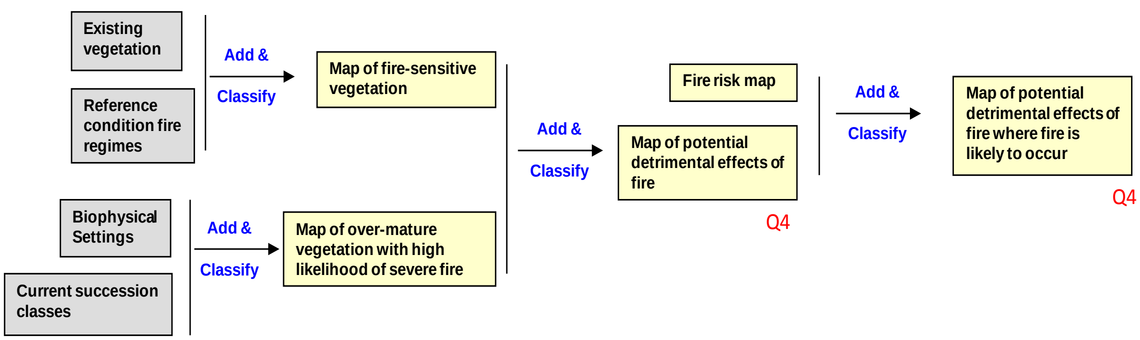

Figure 4-6**.** Process model for identifying areas where fire is adverse to vegetation communities.

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4.2 Change Agent: Invasive Species

Invasive species are considered change agents because they alter ecosystem processes and adversely affect natural resources in the region; they have the potential to expand and/or shift their ranges in the future with continued land cover disturbance and projected global 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). Two invasive plant species of concern-have been selected for the Colorado Plateau REA: cheatgrass (*Bromus tectorum*) and tamarisk (*Tamarix* spp.). Both species have been implicated in changes in the fire regime—even tamarisk, which typically grows in riparian areas. Nonnative species tend to dominate post-fire landscapes and fuel recurrent and more frequent fire (Figure 4-7); the cumulative evidence to date supports the assumptions of D'Antonio and Vitousek's (1992) grass/fire model (Brooks 2008).

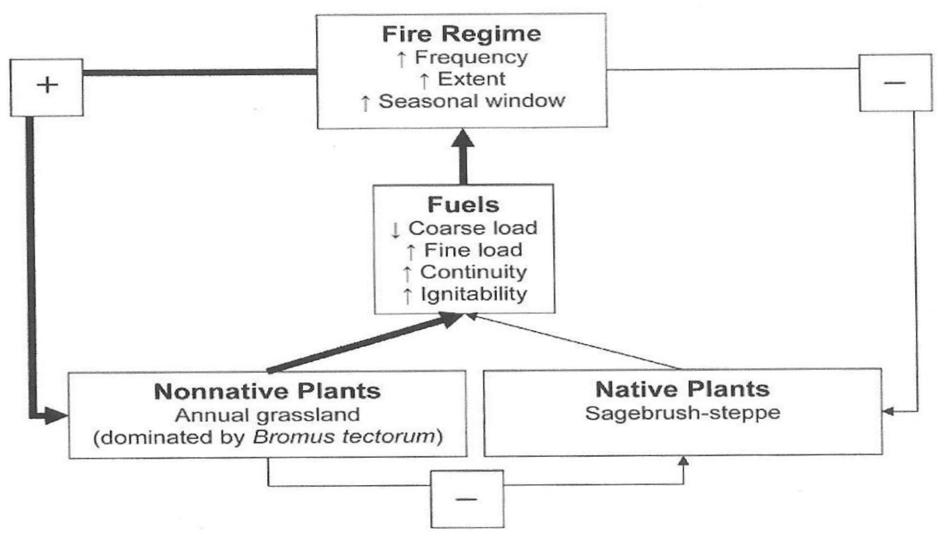

Figure 4-7. Nonnative annual plants create a grass/fire cycle in invaded areas by producing large amounts of litter that contribute to increased fire frequency, intensity, and extent, resulting in the eventual dominance of invasive annual cheatgrass and a type conversion from native shrub or woodland habitat (Brooks 2008).

Continued changes in the fire cycle combined with projected changes from global climate change raise the possibility of widespread type conversion to low-diversity nonnative annual grasslands with major effects on ecosystem function and the abundance of wildlife (Figure 4-8, Dukes and Mooney 2004).

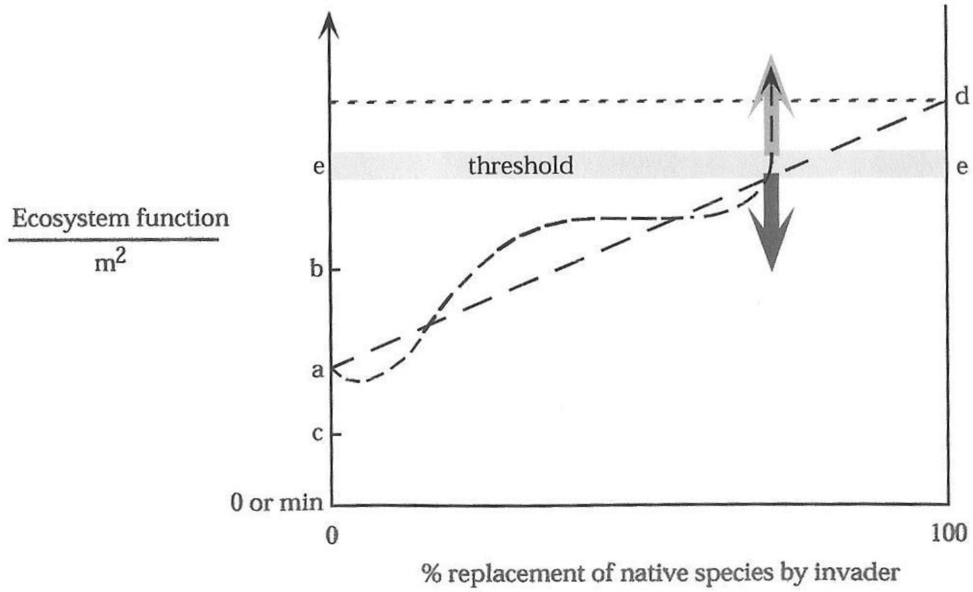

Figure 4-8. An invasive species' composition within an ecosystem may vary between *b* and *c* over time (wavy dashed line). The hypothetical invader in this scenario has on average a higher ecosystem function than the native species (*d*). Eventually a threshold may be crossed that may cause irreversible changes in the ecosystem with a type conversion to annual grassland (Dukes and Mooney 2004).

Four key management questions relate to invasive species:

1. Where are areas dominated by tamarisk and cheatgrass?
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 criteria, and (3) future potential habitat, based on species-specific criteria in conjunction with predicted climate change scenarios.

In the following sections, each invasive species is described with regard to its general characteristics, biological and physical criteria, effects on conservation elements, and data requirements and approach for addressing management questions.

4.2.1 Cheatgrass (*Bromus tectorum*)

Note: BLM has suggested that we use existing models or maps if possible. We present a standard approach to species modeling in these invasive species sections. However, we are evaluating existing cheatgrass remote sensing products and may use one (or both) if it is ready within the timeframe of the REA, covers the entire ecoregion, or may be easily adapted to the full ecoregional extent.

Two options exist for cheatgrass maps:

1. USGS has proposed extending a project from a single county in Utah (proposal by Terence Arundel, Appendix 14) to create a seamless cheatgrass map for the entire Colorado Plateau ecoregion. Ultimately, the final cheatgrass map will be converted from a raster format to a vector format with both sets of data allowing for further analysis in either image analysis software (ENVI) or a geographic information system software (ArcGIS).
2. Douglas Ramsey of Utah State University has developed yearly models of cheatgrass greenup over a ten year period using MODIS EVI (Moderate Resolution Imaging Spectroradiometer Enhanced Vegetation Index). The output maps are generalized into three categories: a minor cheatgrass component (having a low positive temporal normalized index value), a medium cheatgrass component, and a high cheatgrass component (having the highest index value).

4.2.1.1 Introduction

Cheatgrass is an annual grass that is native to Europe, northern Africa, and southwestern Asia (Novack and Mack 2001). This species was introduced accidentally to North America in the mid- to late-1800s (Young 2000, Mack 1981), and by the early 1900s, it had occupied much of its present range (Novack and Mack 2001, Mack 1981). It is particularly invasive in the western U.S. due, in part, to heavy grazing (Mack 1981). This species is a successful colonizer in both disturbed and undisturbed habitat. The ability to persist and dominate disturbed sites and to invade undisturbed habitat, makes this species particularly problematic in the West, where it displaces native vegetation, outcompetes native species, alters fire and hydrological regimes, and encourages topsoil erosion (Boxell and Drohan 2009, Young 2000, Knapp 1996). It currently dominates shrublands in the Intermountain West (Pellant and Hall 1994), occupying at least 40,000 km^2 in Nevada and Utah alone (Bradley and Mustard 2005). Cheatgrass is most prevalent in sagebrush shrub and steppe communities; it also occurs in salt-desert scrub, blackbrush scrub, and pinyon-juniper shrublands and woodlands (Dukes and Mooney 2004, Zouhar 2003, Young 2000). Cheatgrass has replaced native cool- and warm-season grasses, such as Indian ricegrass (*Achnatherium hymenoides*), James galleta (*Pleuraphis jamesii*), blue grama (*Bouteloua gracilis*), sand dropseed (*Sporobolus cryptandrus*), and needle-and-thread grass (*Hesperostipa cornata*), all important forage plants, but also essential to maintaining soil stability, wind and water erosion control, and natural fire regimes (USU Cooperative Extension). To the south, cheatgrass is replaced in dominance by another invasive, red brome (*Bromus rubens*); however, in southern Utah, their ranges overlap.

Cheatgrass occurs primarily between 2,000–5,000 ft (610–1524 m) elevation, but in recent years, it appears to be expanding into higher elevations (Zouhar 2003, Brown and Rowe no date). It can be found on all exposures, although it tends to be most invasive on south- and west-facing slopes. The species prefers deep, loamy, or coarse-textured soils (Young 2000). It does well on both low-fertility soils (Young 2000, Klemmedson and Smith 1964) and higher fertility soils where competition has been reduced (Dakheel et al. 1993). Elevated nitrogen levels can enhance cheatgrass growth (Sperry et al. 2006, Ziska et al. 2005, Smith et al. 2000, D'Antonio 2000, Lowe et al. 2002, Young et al. 1995).

Rainfall and temperature appear to be limiting factors in cheatgrass distribution (Bradley 2008). Cheatgrass grows under a range of climatic conditions, but it is most successful in areas where the rainfall ranges between 6–22 inches (15–56 cm, Young 2000). The species preferred moisture range and germination schedule may affect prospects for restoring native species in cheatgrass-invaded areas. Seeds typically germinate in response to autumn precipitation, although recruitment can occur through late spring within a few days after rain (Young 2000, Mack and Pyke 1983, Klemmedson and Smith 1964). Seeds germinate at temperatures just above freezing and below 86°F (30° C); however, optimal germination occurs between 50–68°F (10–20° C, Zouhar 2003, Hulbert 1955). By germinating early and utilizing available water sources, cheatgrass gains a competitive advantage over native shrubs that remain dormant until spring (Figure 4-9, Rice et al. 1992). Cheatgrass germination is inhibited on sites with well-developed biological soil crusts (which prevent seed burial) and low plant litter (Zouhar 2003). Seed production by cheatgrass is prolific (Hulbert 1955). Seed dispersal mechanisms include gravity, short-distance dispersal by wind or water, or long-distance dispersal by humans, animals, vehicles, or equipment (Hulbert 1955, Young 2000). Cheatgrass does not form a persistent soil seed bank, although seeds may persist in litter or soil for up to a year (Hulbert 1955).

4.2.1.2 Conceptual Models

Cheatgrass was selected as a change agent in the Colorado Plateau REA because it affects multiple conservation elements, including regional Ecological Systems (vegetation communities), forage resources, soil stability, and wildlife habitat. The conceptual models (Figure 4-9 and 4-10) illustrate the causal relationships between cheatgrass invasion and expansion and the subsequent effects on physical and biological systems. Cheatgrass is introduced into disturbed sites through a variety of anthropogenic sources that remove native vegetation and disturb the soil surface, thus creating gaps or areas of bare soil. Cheatgrass readily invades disturbed areas, such as rangelands, pastures, cultivated fields, waste areas, and roadsides (Zouhar 2003, Young 2000). In the Colorado Plateau, disturbances that damage biological soil crusts are particularly beneficial to the establishment of this species (Figure 4-10, Brooks and Pyke 2001, Meyer et al. 2001, Belnap et al. 1999, D'Antonio and Vitousek 1992).
Cheatgrass invasion results in loss of wildlife habitat and wildlife species diversity by eliminating forage and altering vegetation structure (Brooks and Pyke 2001). These changes, in turn, can result in increased predation, decreased prey bases, direct mortalities (e.g., from fire), and impediments to small mammal and avian movement (Brooks et al. 2004, Kochert et al. 1999). Two REA conservation elements, sage grouse and mule deer, may be directly affected by cheatgrass invasion through loss of cover and forage (Zouhar 2003, USFWS 2003).

Cheatgrass dispersion is enhanced by grazing and fire events that clear vegetation and provide bare soil for seed germination (Hulbert 1955). Cheatgrass alters the natural fire regime of Colorado Plateau ecosystems, resulting in more frequent and intense fires that favor its spread and persistence (Figure 4-9 and 4-10). Cheatgrass creates a grass/fire cycle in invaded areas (Stewart and Hull 1949, Mack 1981, Young et al. 1987, D'Antonio and Vitousek 1992) that is particularly detrimental to fire-sensitive plant communities (Chambers et al. 2009). The large amounts of litter produced by this species provide a persistent fine fuel source that contributes to increased fire frequency, intensity and extent (Figure 4-7 and 4-9, Brooks and Pyke 2001, D'Antonio 2000, Young and Evans 1978, Beatley 1966, Stewart and Hull

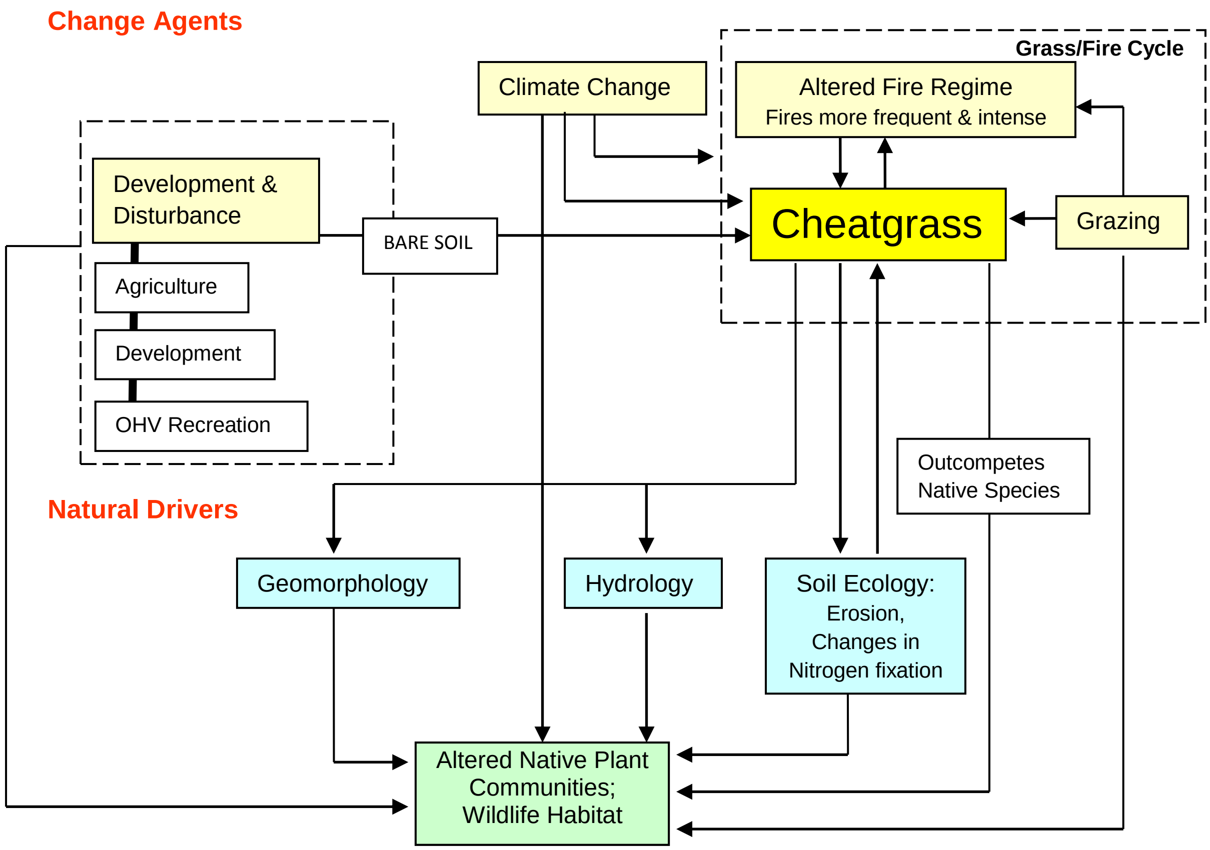

Figure 4-9. Conceptual model for cheatgrass in the Colorado Plateau ecoregion.

1949). Although cheatgrass is killed by fire, it quickly re-establishes from on- or offsite seed sources, thus, perpetuating the cycle. This grass/fire cycle results in type conversion of native shrub habitats (Figure 4-8), and it displaces native plant species by reducing or eliminating suitable habitat, consuming resources (e.g., water), and altering soil ecology, nitrogen dynamics, and geomorphology (Boxell and Drohan 2009, Sperry et al. 2006, Zouhar 2003, Melgoza and Novak 1991, Melgoza et al. 1990). Fire also kills biologic soil crusts (Figure 4-10, Belnap et al. 2001), which further opens sites for cheatgrass invasion (Callison et al. 1985).

Climate change is expected to result in shifts in cheatgrass distribution, based on modeling that considers both temperature and precipitation and changes in fire regime; the species will likely experience both range expansions and contractions in the future (Bradley et al. 2009). Projected distributional shifts northward may result in contractions or loss of climatically suitable habitat in Nevada and Utah, although these lands may then be susceptible to invasion by other species, such as red brome (Bradley 2009, Bradley et al. 2009). Other aspects of climate change, such as increasing atmospheric $C0_2$ levels and nitrogen deposition, are expected to enhance cheatgrass growth, leading to increased biomass and fuels (Sperry et al. 2006, Ziska et al. 2005, Smith et al. 2000).

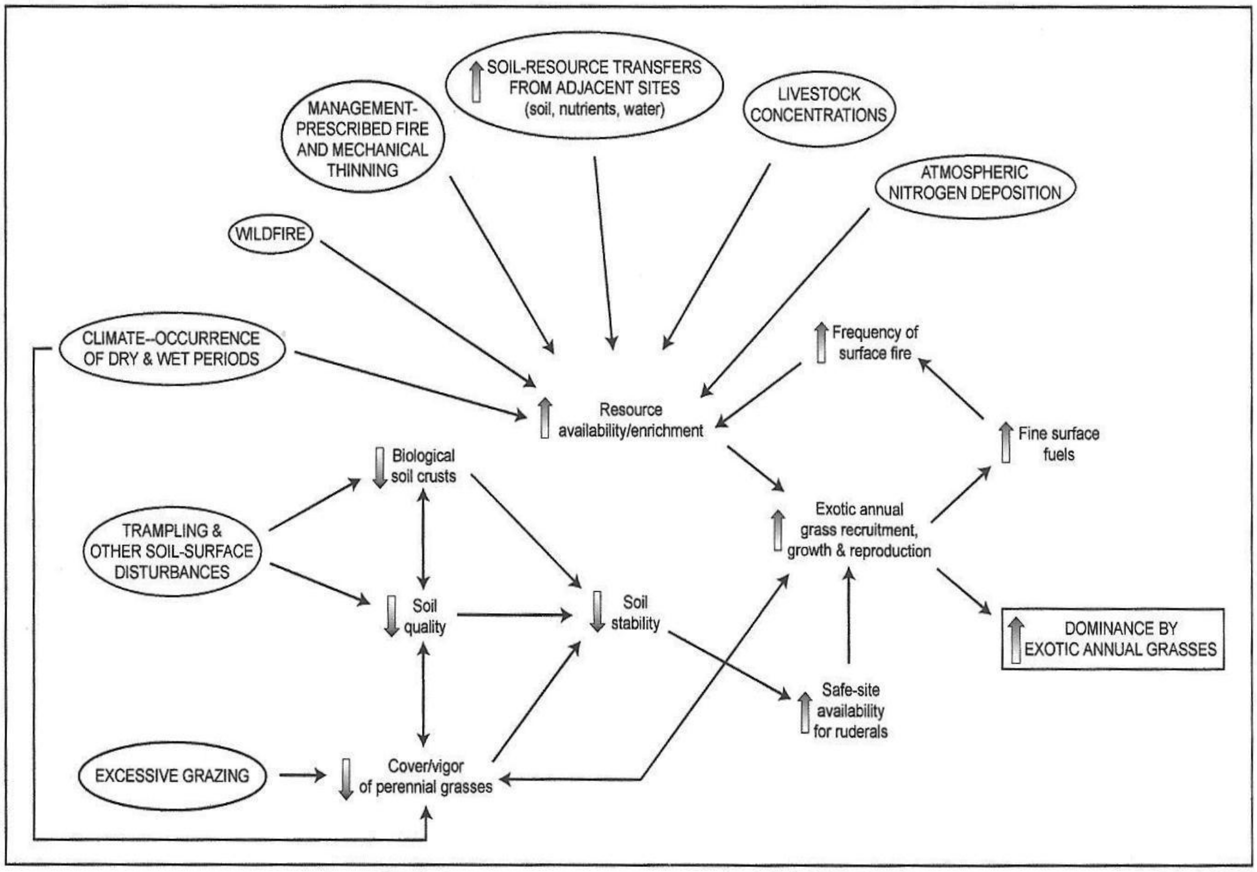

Figure 4-10. Conceptual model illustrating the processes by which climate and human disturbance factors can lead to increasing dominance by exotic annual grasses, in this case, cheatgrass (Miller 2005).

4.2.1.3 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 modeling cheatgrass distribution.

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	1,3,4
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 cheatgrass.

Dataset	Source	Location	Management Questions (s)
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	1,3,4
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

4.2.1.4 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 environmental attributes in the modeling for cheatgrass

4.2.1.5 Model Assumptions

There are several assumptions on which the cheatgrass models are based. These include 1) cheatgrass out-competes and displaces native species; 2) cheatgrass colonizes disturbed and undisturbed habitats; 3) cheatgrass alters fire and hydrological regimes, thus perpetuating its own persistence; 4) cheatgrass tends to be most invasive on south- and west-facing slopes; 5) cheatgrass prefers deep, loamy, or coarse-textured soils; cheatgrass is most prevalent in arid shrublands; 6) rainfall and temperature are limiting factors in cheatgrass distribution; 7) cheatgrass dispersions is enhanced by grazing and fire; and 8) cheatgrass is expected to undergo range expansions and contractions as a result of climate change.

4.2.1.6 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 model (Figures 4-11) 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 cheatgrass. 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 4-11). 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 4-11).

The example model (Figure 4-11) addresses future distribution questions (MQ 2) by identifying areas of potential future encroachment by cheatgrass, based on species-specific attributes. Disturbed sites from grazing and burning are extremely important drivers for cheatgrass, and they will be modeled using available spatial datasets. Suitable physical habitat for cheatgrass is comprised of its preferred topography, soils, vegetation, slope, temperature and precipitation. An example of the proposed method/analysis flow we would take to answer management question two (MQ2 in Figure 4-11) would be to combine physical parameters, vegetation, and recently disturbed sites (grazing, fires). The resulting dataset would indicate areas of suitable cheatgrass expansion by HUC5s.

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

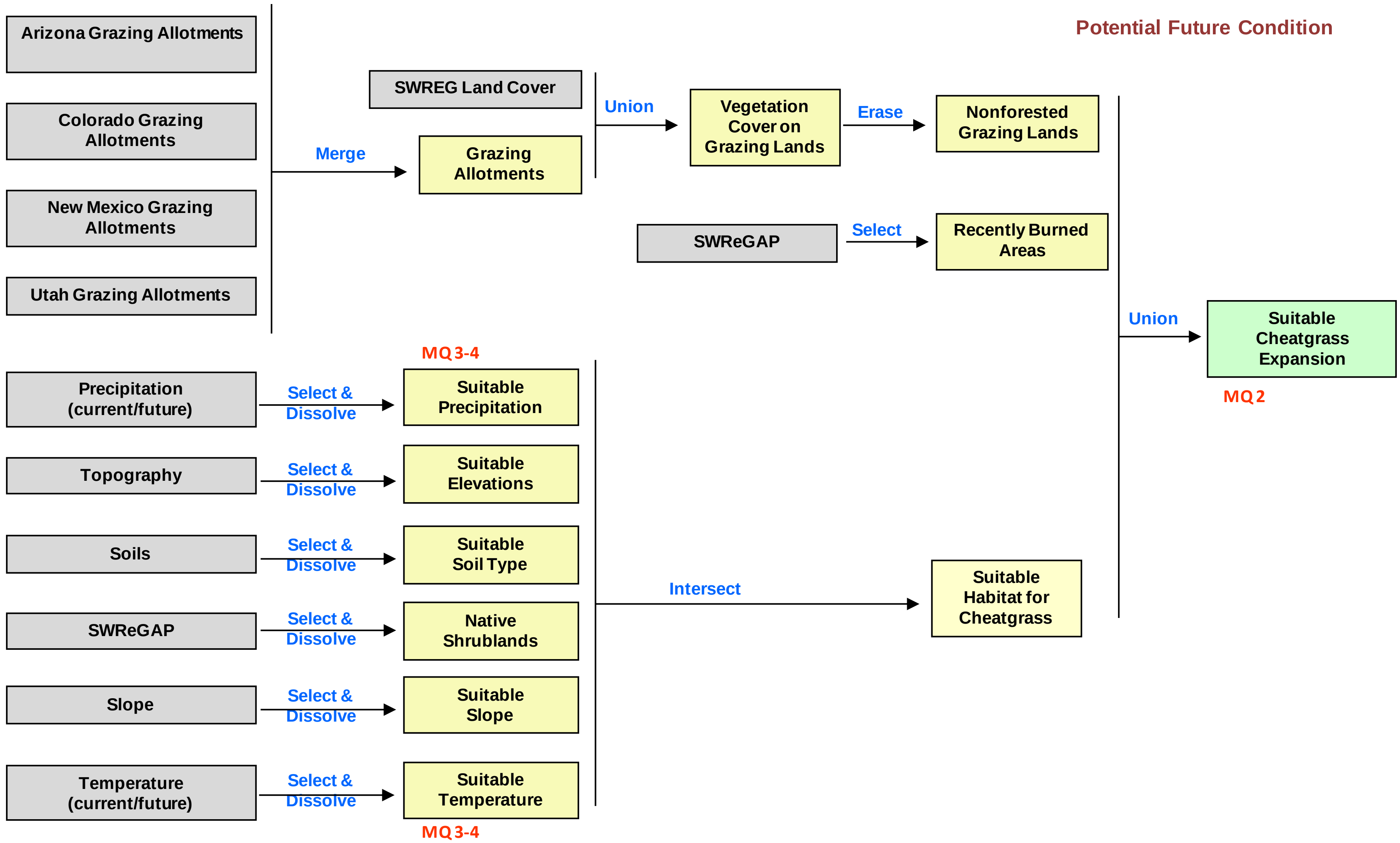

Figure 4-11. *Example* process model for cheatgrass potential habitat, restoration areas, and expansion in the Colorado Plateau ecoregion.

4.2.1.7 References Cited

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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). We will also incorporate recent mapped data on the progress of the tamarisk leaf beetle (*Diorhabda carinulata*) from the Tamarisk Coalition.

4.2.2.1 Background

Tamarisk (or salt cedar) is an invasive shrub that has been designated as a change agent in the Colorado Plateau 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 4-12 and Figure 4-13) 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 4-12). 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.

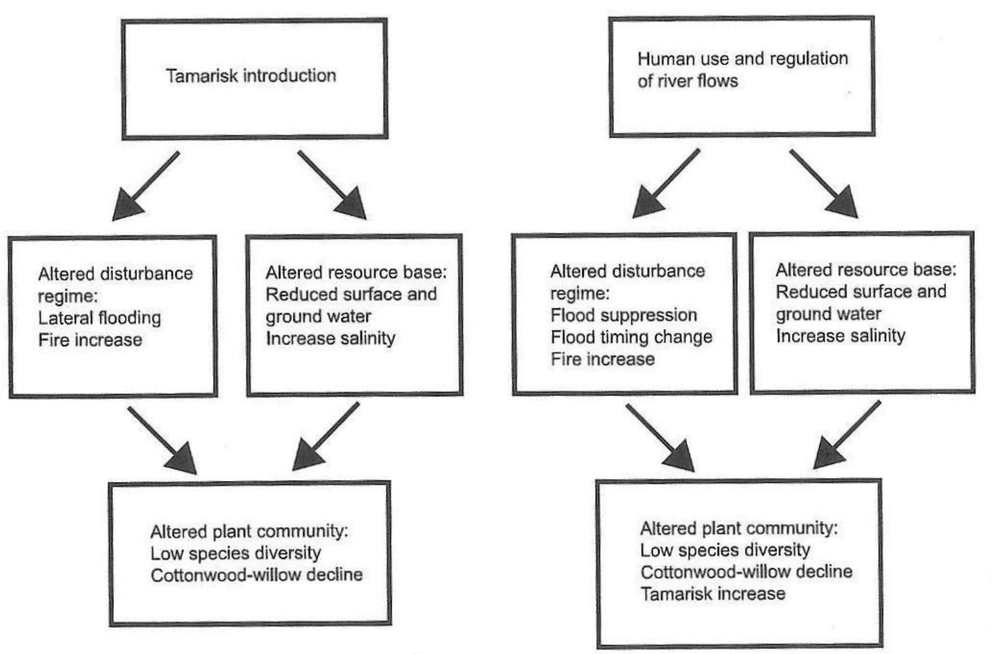

Figure 4-12. Tamarisk invasion and human disturbances as two major influences on the conversion of southwestern riparian systems to nonnative monocultures (Stromberg et al. 2005)

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 4-13 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 4-13, 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 4-13, 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 tamarisk-dominated sites. Yellow-breasted chat (*Icteria virens*), a Colorado Plateau REA species conservation element, is another species that will regularly use tamarisk for nesting habitat. 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 4-13, Altered Fire Regime, Climate Change).

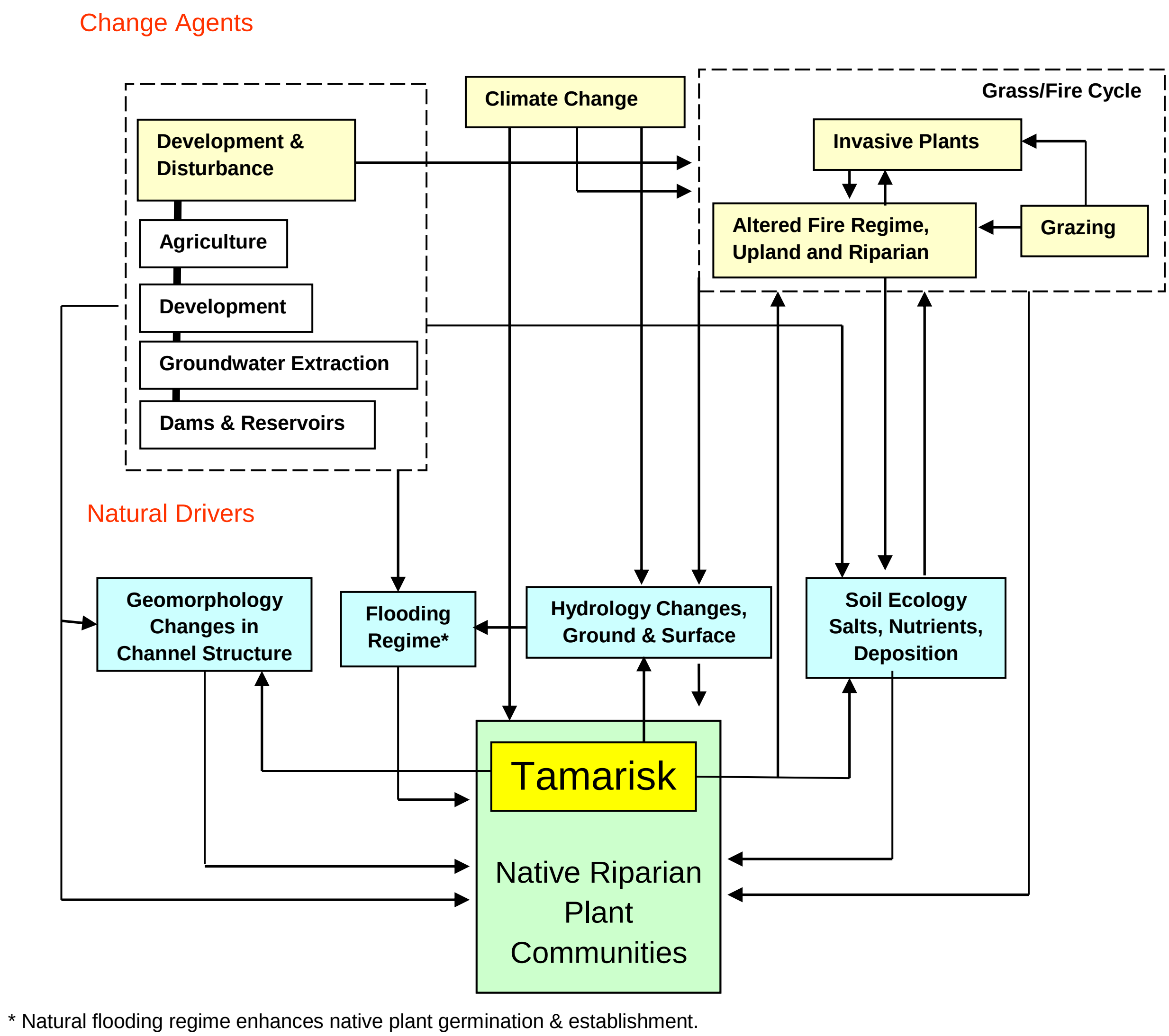

Figure 4-13. Conceptual model for tamarisk in the Colorado Plateau ecoregion.

4.2.2.3 Required Input Data Layers

We compiled a list of datasets we think are most relevant for answering the management questions.

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 Colorado Plateau 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 conceptual 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 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 4-14). 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. Although we have not provided all combinations for our proposed methodological approach, we have provided an example (Figure 4-14).

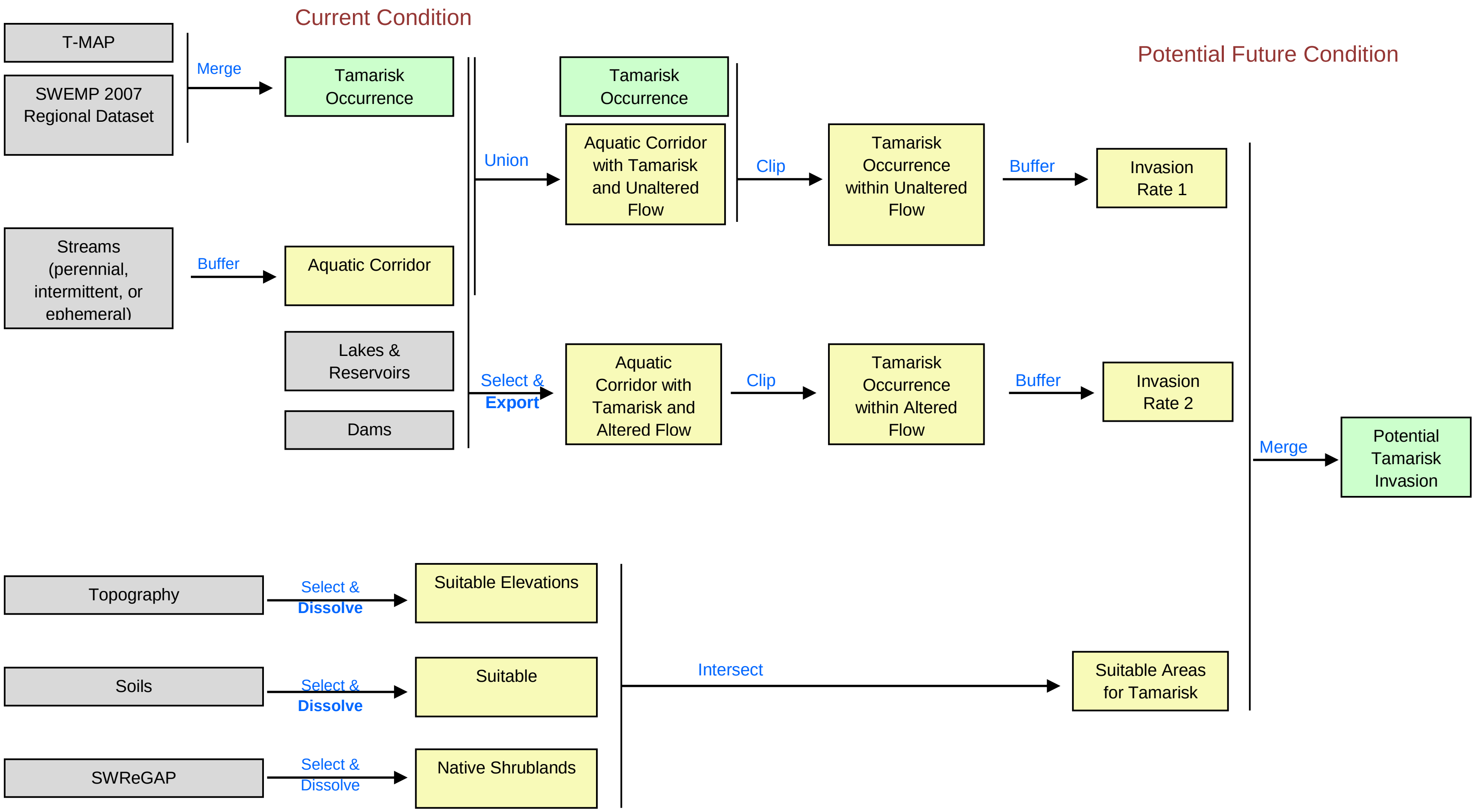

Figure 4-14. Application model for current and future conditions of tamarisk in the Colorado Plateau ecoregion.

The example model (Figure 4-14) 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 4-14) 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 4-14) also addresses areas of potential tamarisk invasion (MQ 2 in Figure 4-14). 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 4-14) 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.

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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 4-8).

Table 4-8. Emission scenarios used for AR4 (IPCC 2007) with information extracted and simplified from Nakicenovic et al. 2000.

SRES Emission Scenarios used for AR4	Description	CO_2-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 southern portion of the CO plateau and 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:

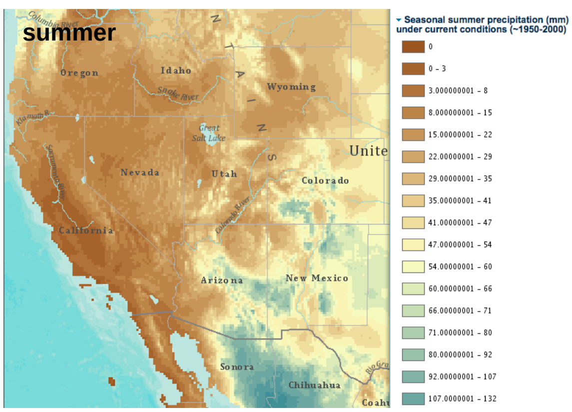
1. *What aquatic and terrestrial species conservation elements are vulnerable to change agents in the near term horizon, 2020 (development, fire, invasive species) and a long-term change horizon, 2060 (climate change)? Where are these species located?*

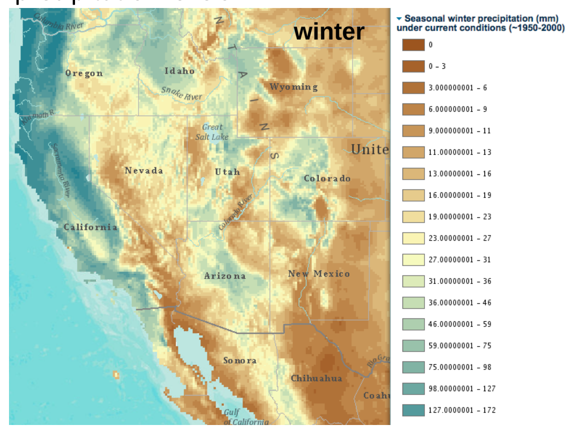
2. *Where/how will the distribution of dominant native plant and invasive species be vulnerable to or have potential to change from climate change in 2060?*

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

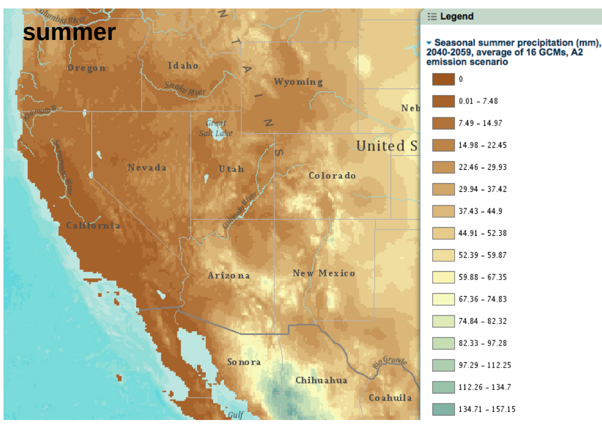
Figure 4-15. Comparison between current seasonal precipitation levels and projected under the A2 emission scenario averaged over 16 climate models results using WorldClim historical baseline. Data provided by California Academy of Sciences.

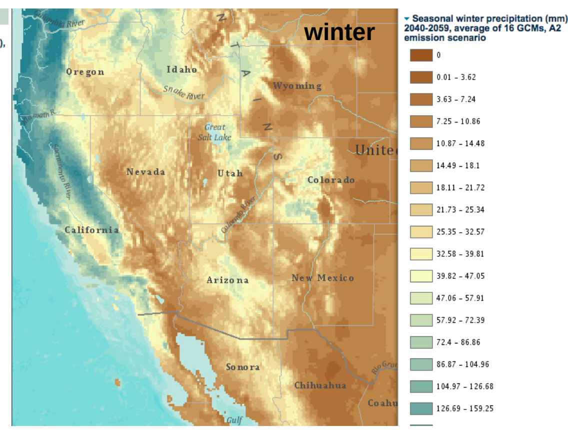
For current conditions (1950-2000), average seasonal precipitation levels



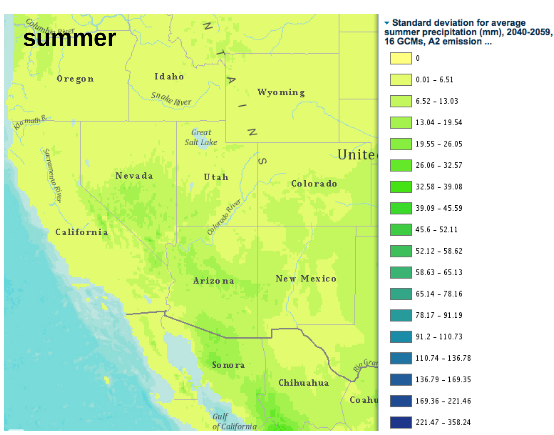


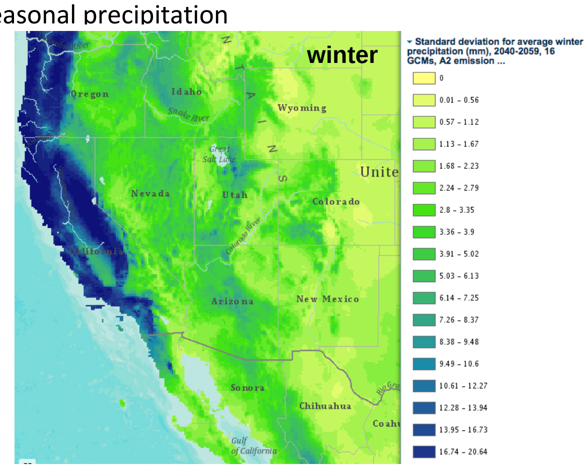
For the period 2040-2060, average seasonal precipitation





For the period 2040-2060, standard deviation for seasonal precipitation





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.

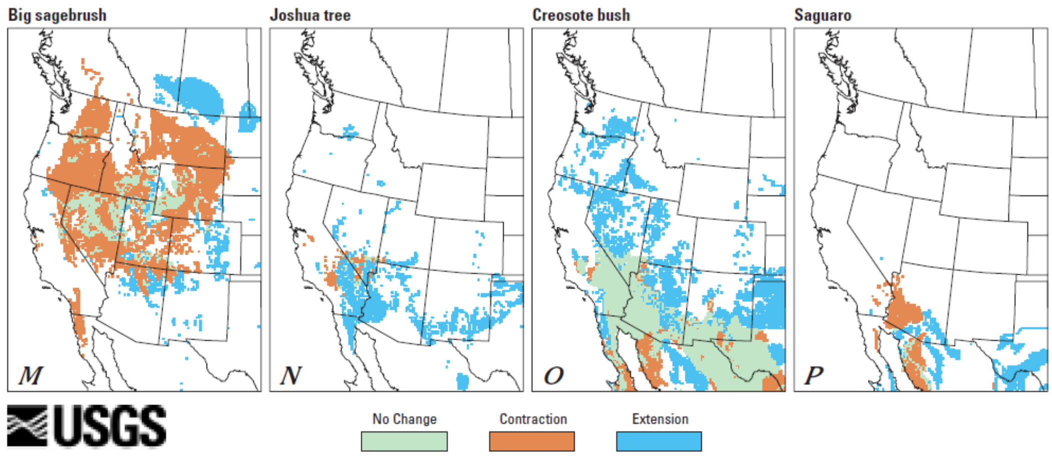


Figure 4-16. MaxEnt modeled output of current (base period) and future potential distribution based on key bioclimatic variables. Areas of CONTRACTION are comparable to areas of VULNERABILITY (sensitivity + exposure), as emphasized by BLM for the REAs (Thompson et al. 1998).

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.

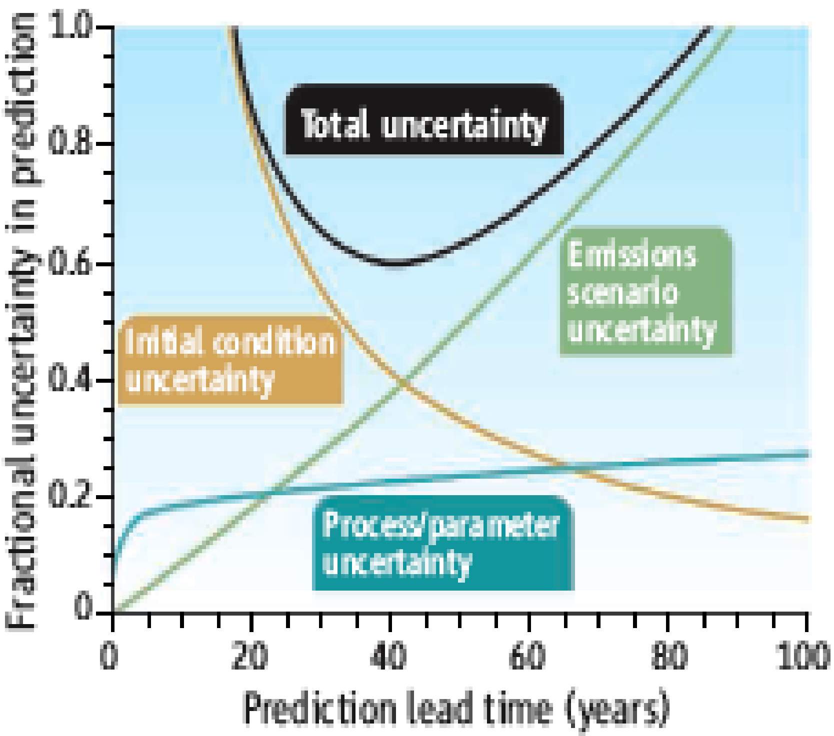


Figure 4-17. Graphic representation of the uncertainty associated with today's climate projections (from Cox & Stephenson 2007).

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.

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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. The Dynamac team is permitted to use EMDS as long as the final products are delivered in Model Builder. We propose to use EMDS for some aspects of the work (budget and time permitting) to address some of the more complex questions (e.g., assessing ecological integrity) 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)

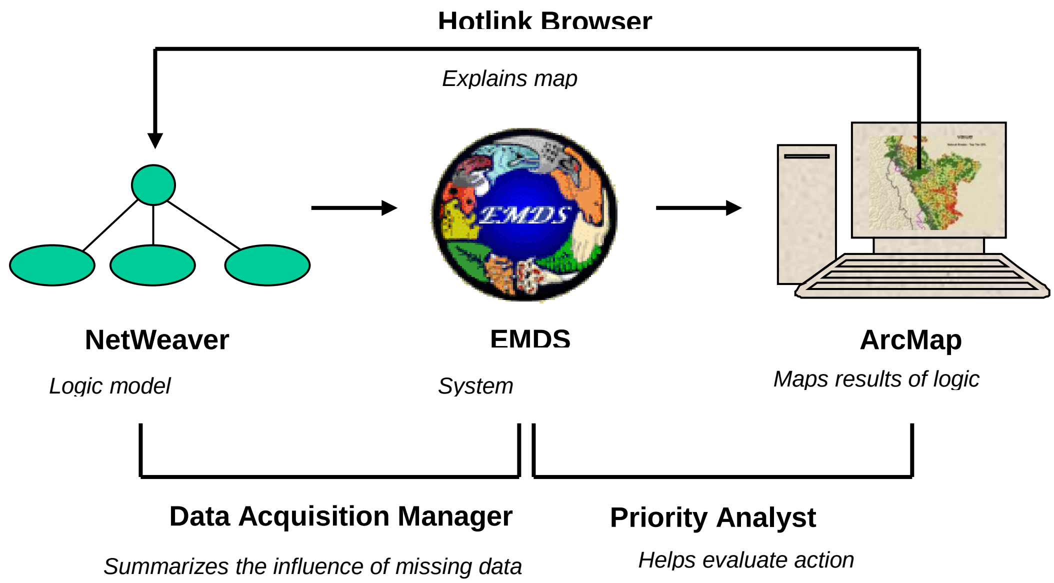

Figure 5-1. Diagram showing the EMDS-software interaction.

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.

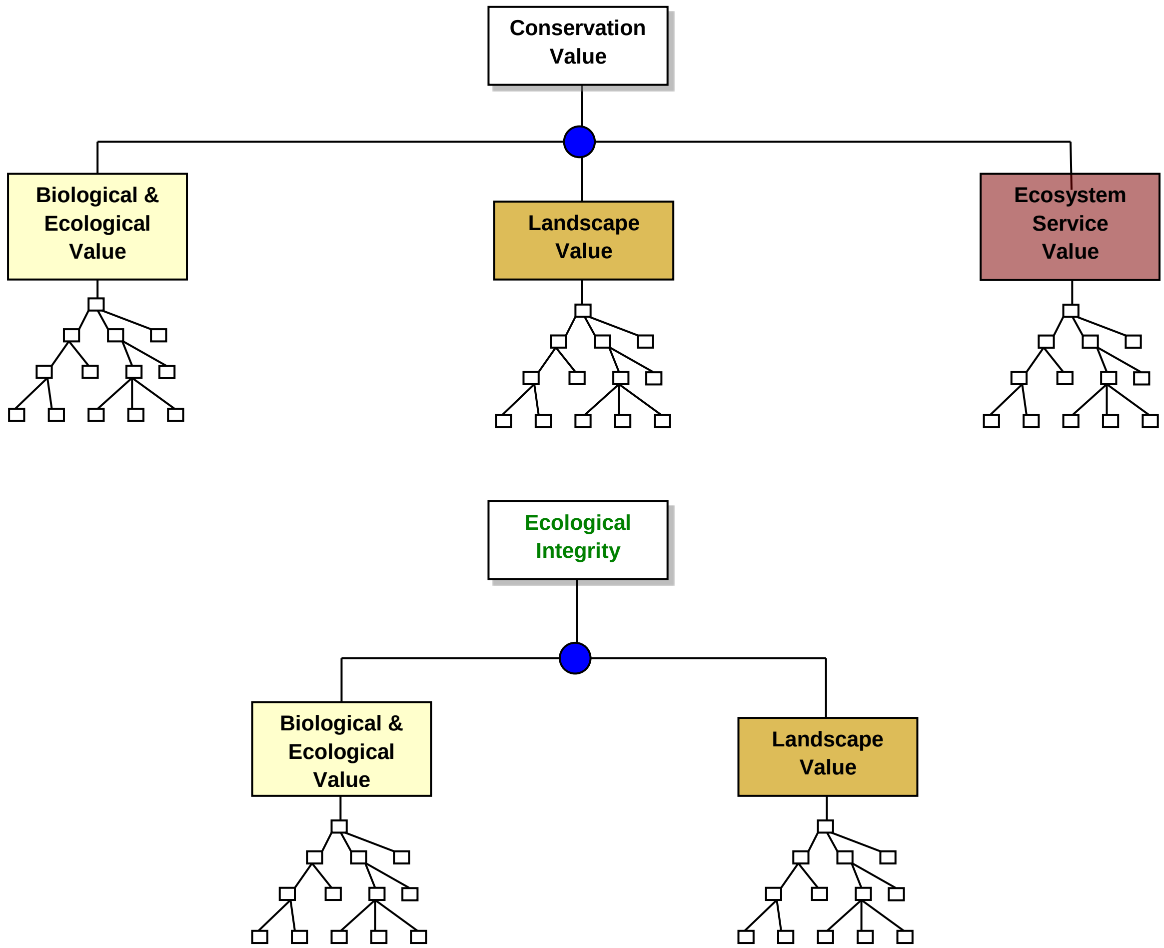

Figure 5-2. Diagram showing simple EMDS expression of Conservation Value and Ecological Integrity.

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 aide in orientation: Tier I is the highest order topic and tier VIII is the lowest.

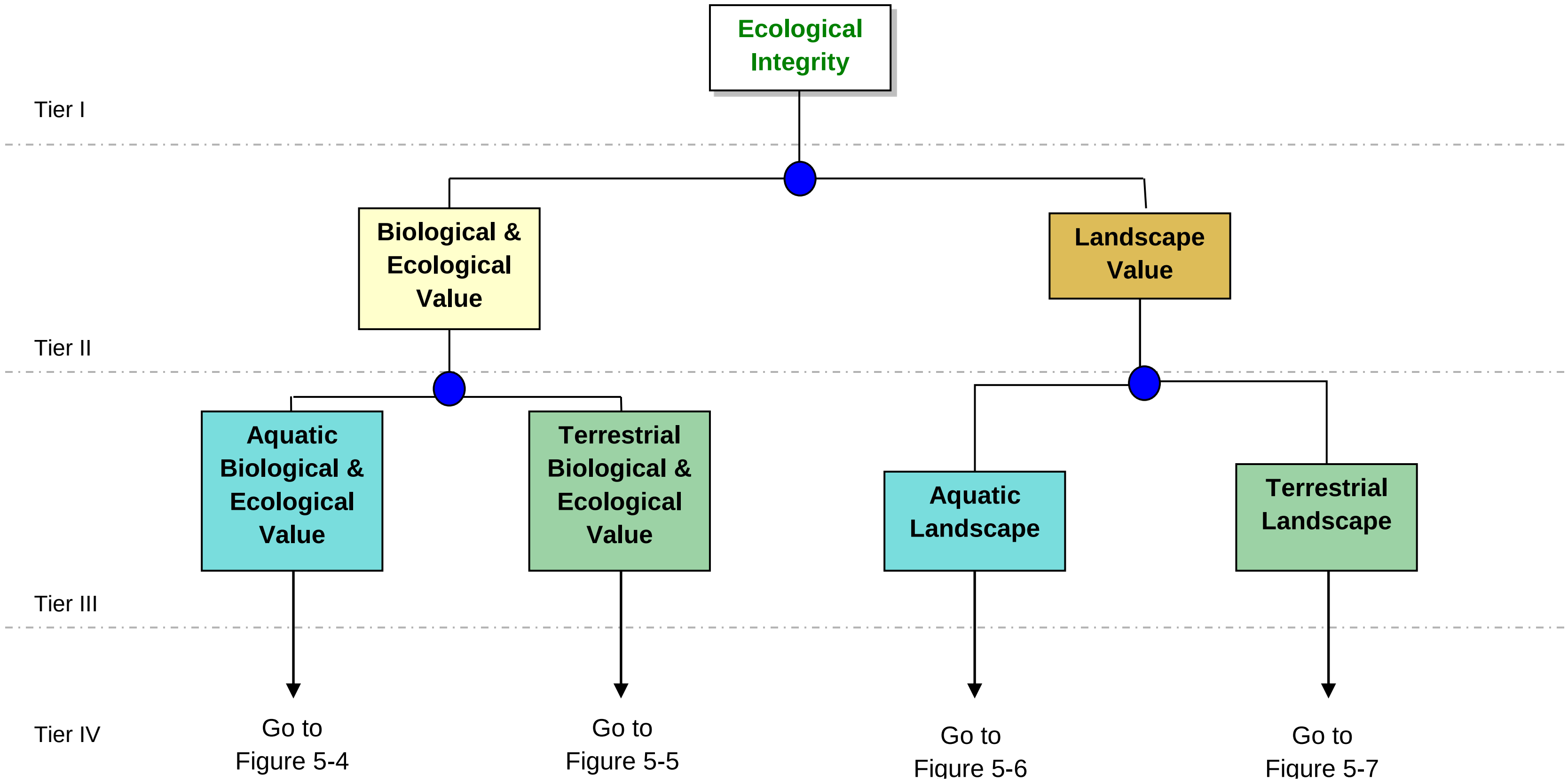

Figure 5-3. Draft EMDS logic diagram showing high level topics for Ecological Integrity modeling.

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 & Ecological Value. Important Aquatic Communities is formed by combining 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. The only aquatic animals of interest are fish species, including razorback sucker, flannelmouth sucker, and Colorado River cutthroat trout. Important sites for these species include a combination of known locations and suitable habitat. Aquatic High Value Species Composite is made up of data on Aquatic Species Endemism, Aquatic Threatened and Endangered Species, and Aquatic Species Richness, which in turn is composed of datasets on fishes, aquatic 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 get 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 cryptogamic crust, mountain sagebrush, Wyoming big sagebrush, pinyon-juniper woodlands and shrublands, intermountain montane sagebrush steppe, intermountain big sagebrush shrubland, gambel oak-mixed montane shrubland, blackbrush-morman tea shrubland, mixed salt desert scrub, and bedrock canyon and tablelands.

The Terrestrial Species topic is derived from topics on Terrestrial Plant Species of Interest (including pinyon pine, gambel oak, Utah juniper, littleleaf mahogany, shadscale, and blackbrush). Terrestrial Animal Species of Interest include Birds and Mammals. For the topic Bird Composite, the model will combine all important sites for each species of interest (including golden eagle, burrowing owl, Mexican spotted owl, peregrine falcon, greater sage grouse, Gunnison's sage grouse, ferruginous hawk, and yellow-breasted chat). For the bird species, the logic model is showing two different ways data could be entered. Species with good location and habitat needs data will be treated like the golden eagle example. Those with a red star will be modeled using Maxent or other species distribution modeling tools to generate high 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.

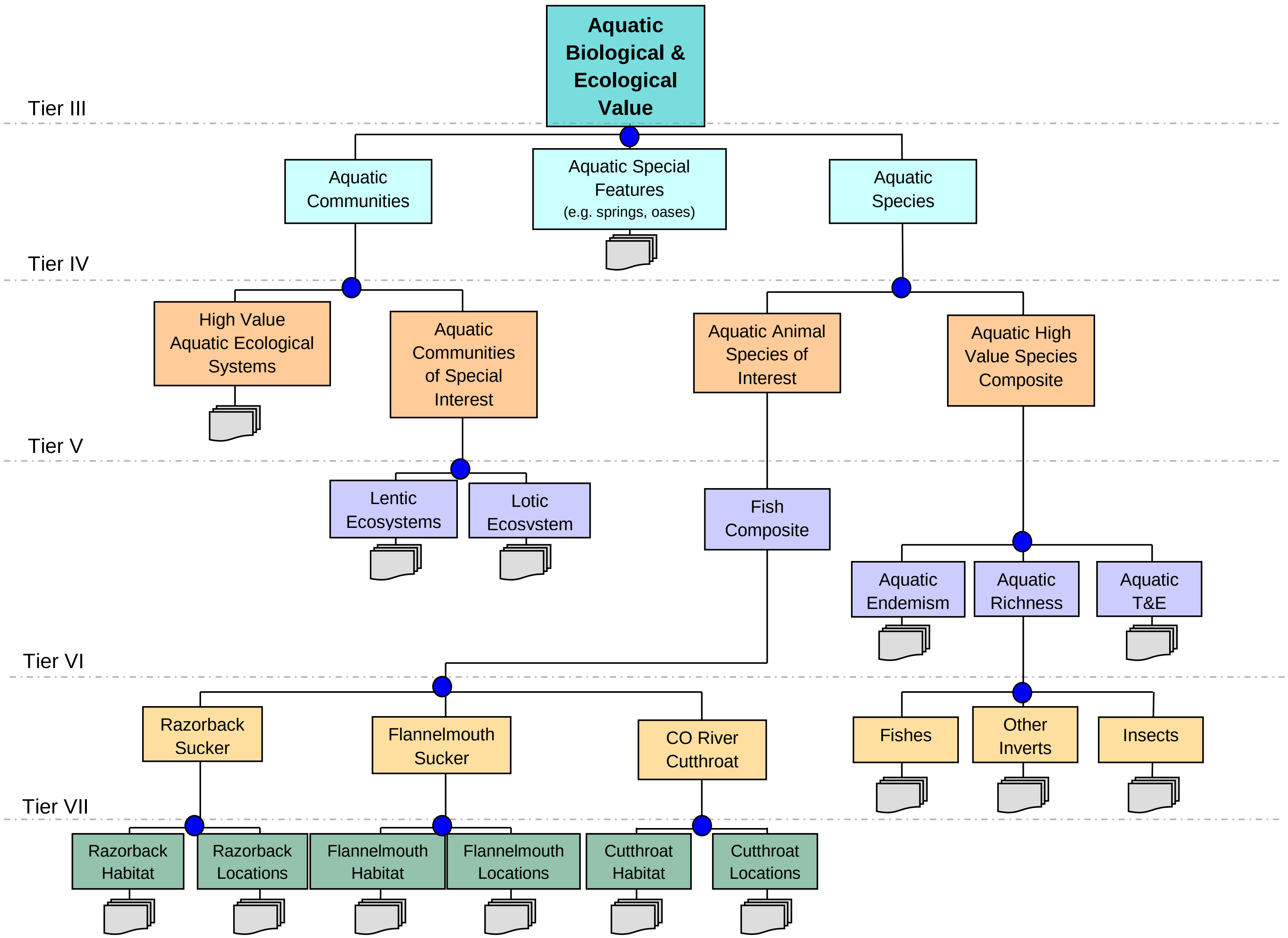

Figure 5- 4. Draft EMDS model showing model logic for Aquatic Biological and Ecological Value.

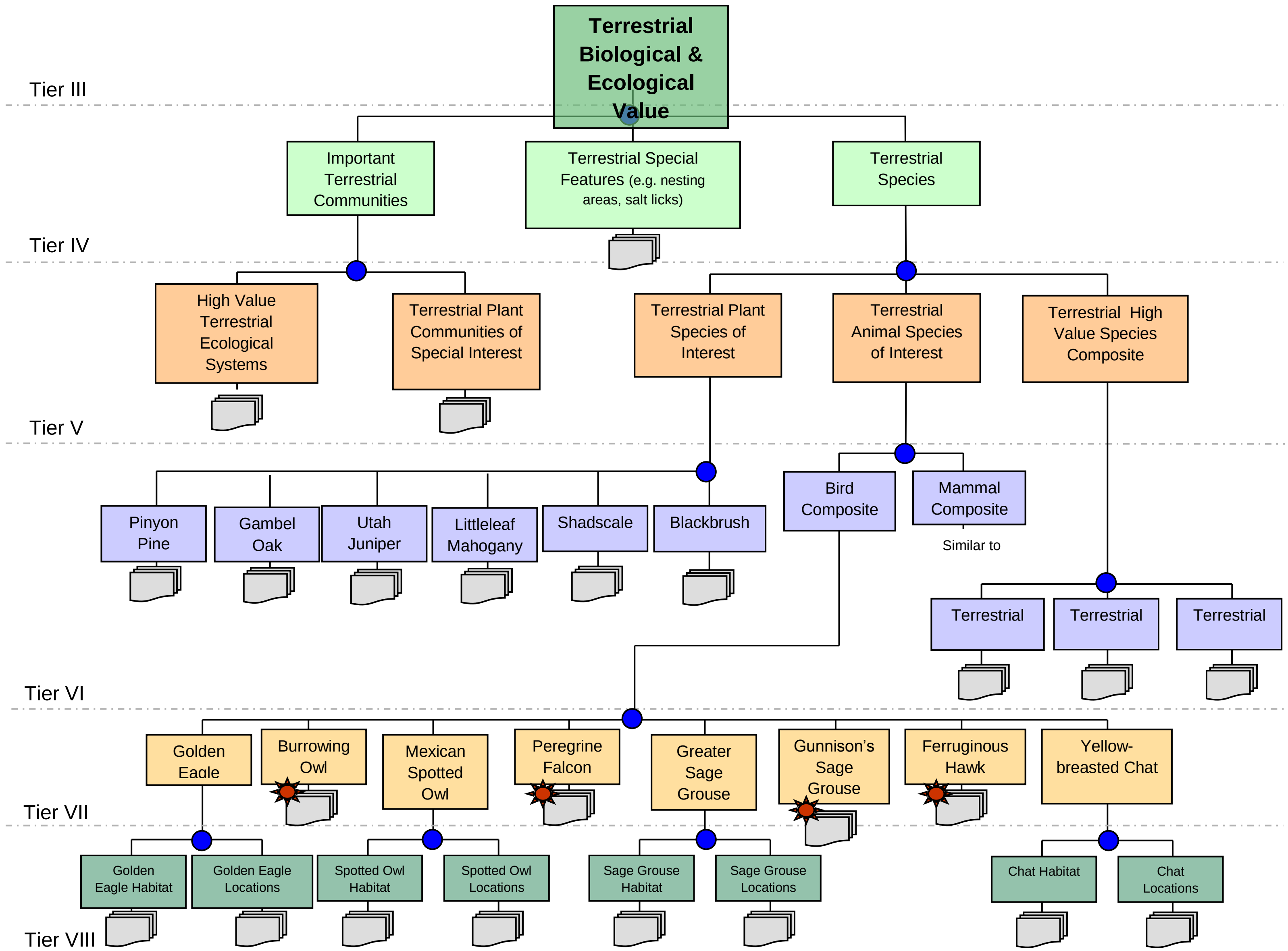

Figure 5-5. Draft EMDS model showing model logic for Terrestrial Biological and Ecological Value.

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 (5th field watersheds) are irregular in shape and size. Invasive inputs will be based on available distribution data for tamarisk and cheatgrass for the Colorado Plateau. Developed Area will include Urban, Agriculture, and Energy footprints.

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 so as to avoid redundancy. Based on previous experience (see Staus et al. 2010), three metrics

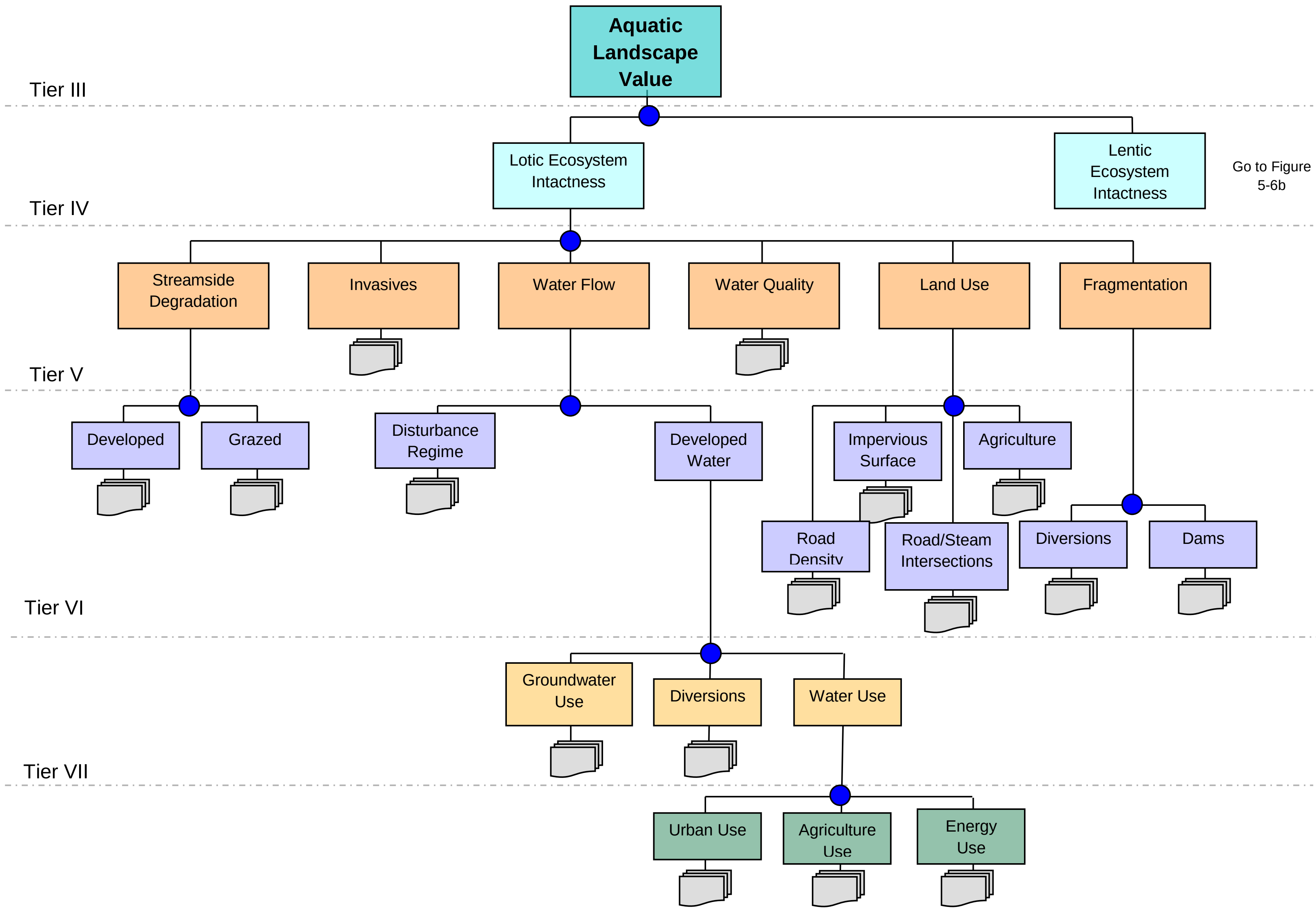

Figure 5-6a. Draft EMDS logic model for Aquatic Landscape Value – Lotic Ecosystem Intactness.

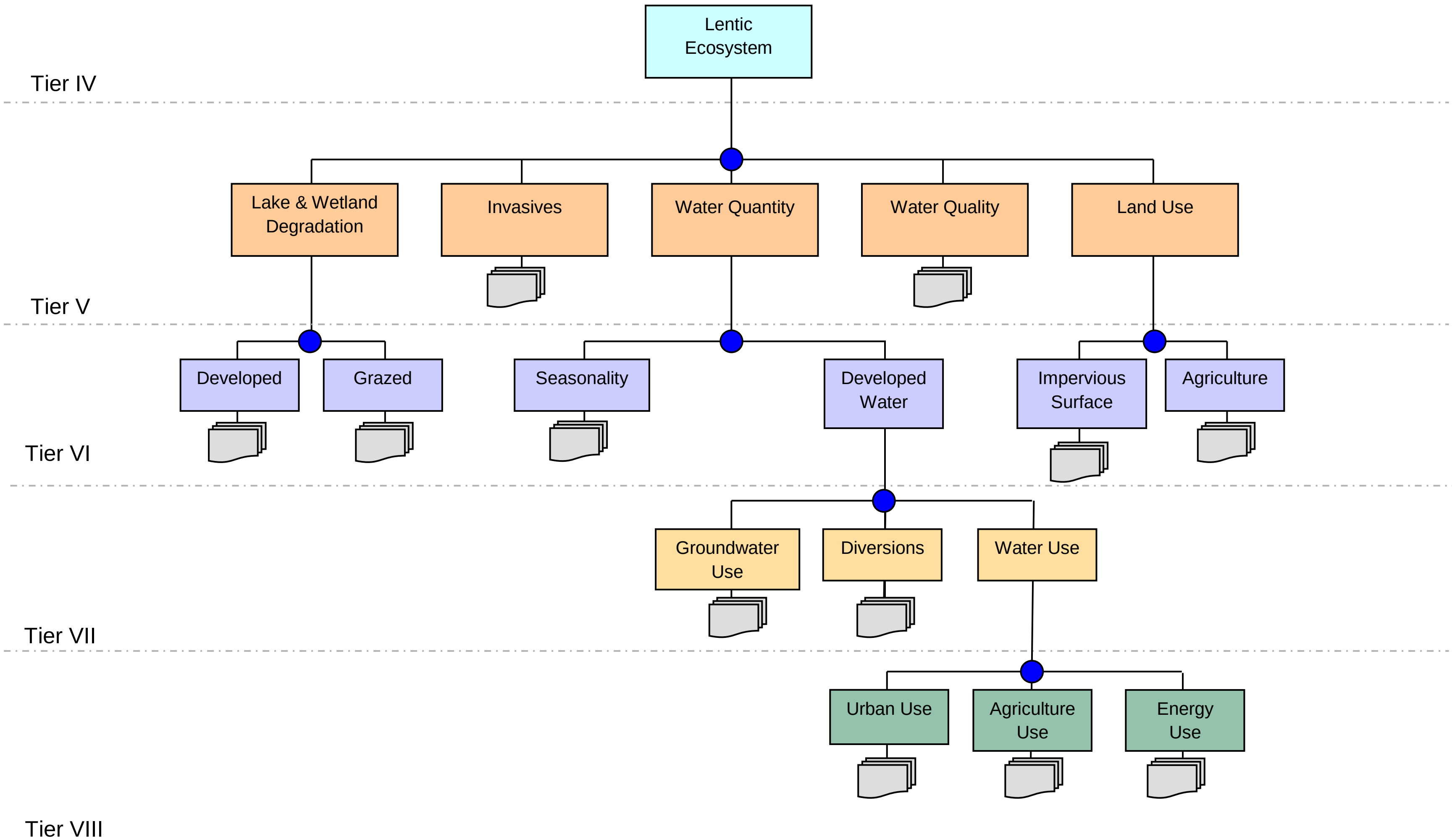

Figure 5-6b. Draft EMDS logic model for Aquatic Landscape Value – Lentic Ecosystem Intactness.

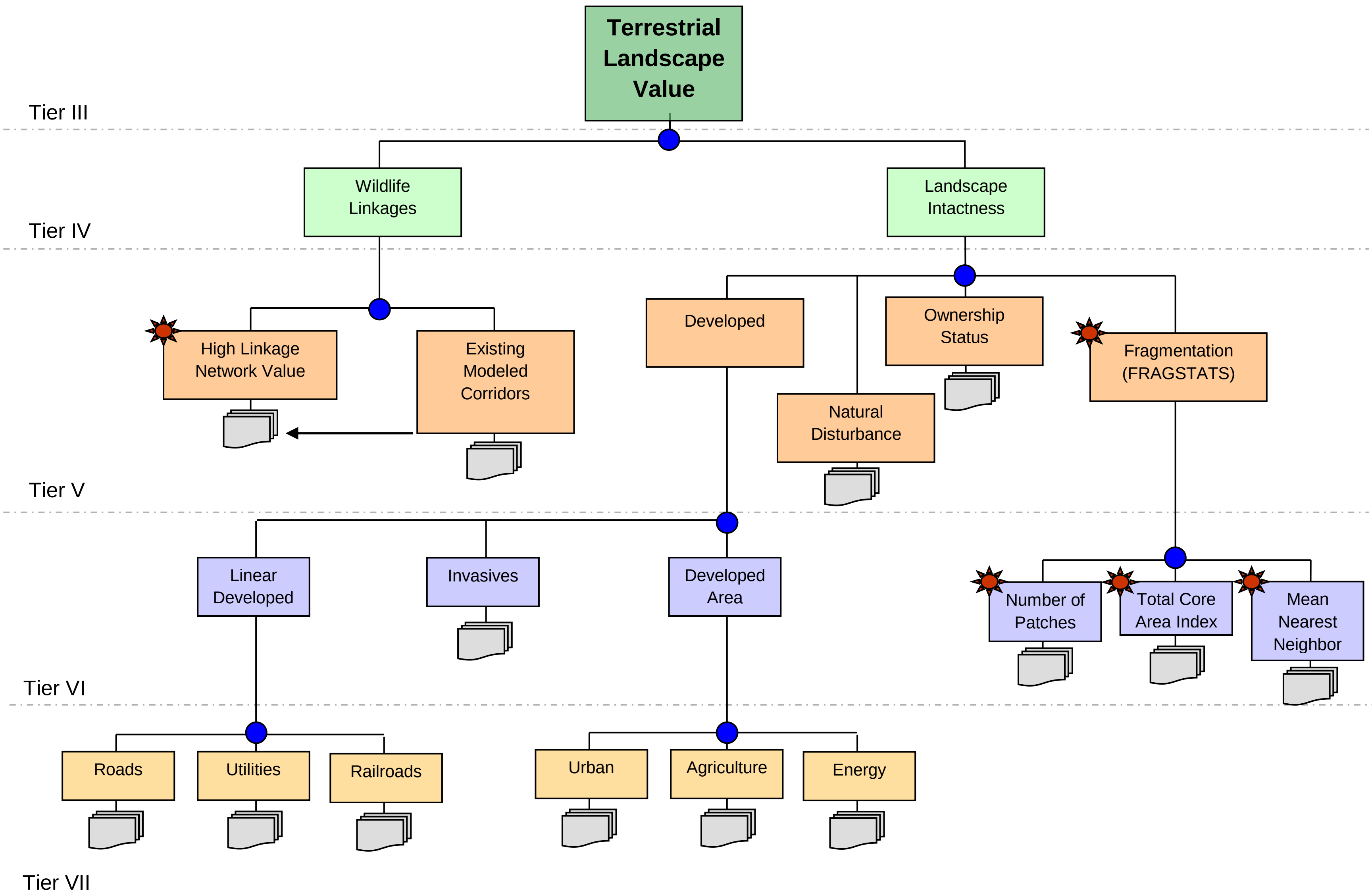

Figure 5-7. Draft EMDS logic model for Terrestrial Landscape Value.

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 affect 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, five ecosystem services were listed as topics of interest. Two are supporting services – Soil Stability 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 drinking water means less instream flow for aquatic wildlife. 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

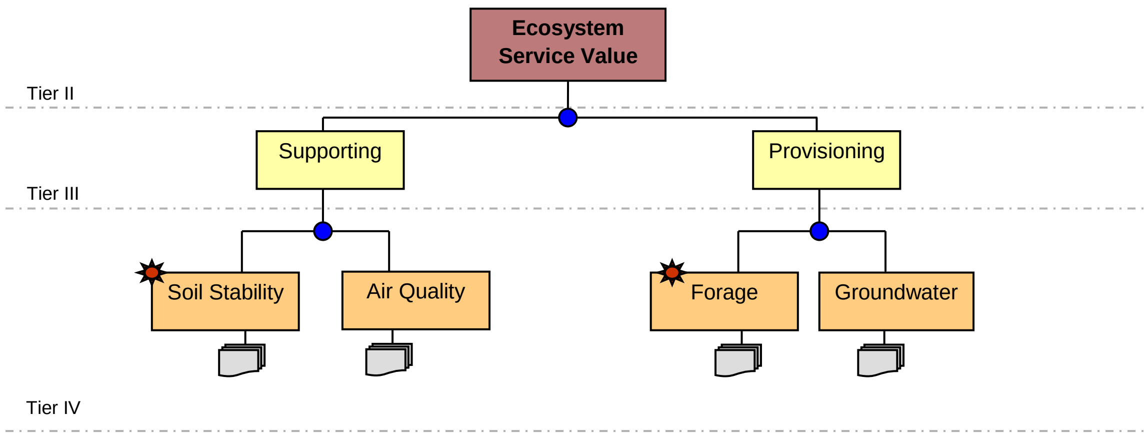

Figure 5-8. Draft EMDS diagram for Ecosystem Service Values.

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 current disturbances as well as 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 can then be applied to any of the topics from the previous model. For example, if you are interested in where Greater Sage Grouse are likely to be affected by fire in the future, you could overlay the topic called Greater Sage Grouse 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.

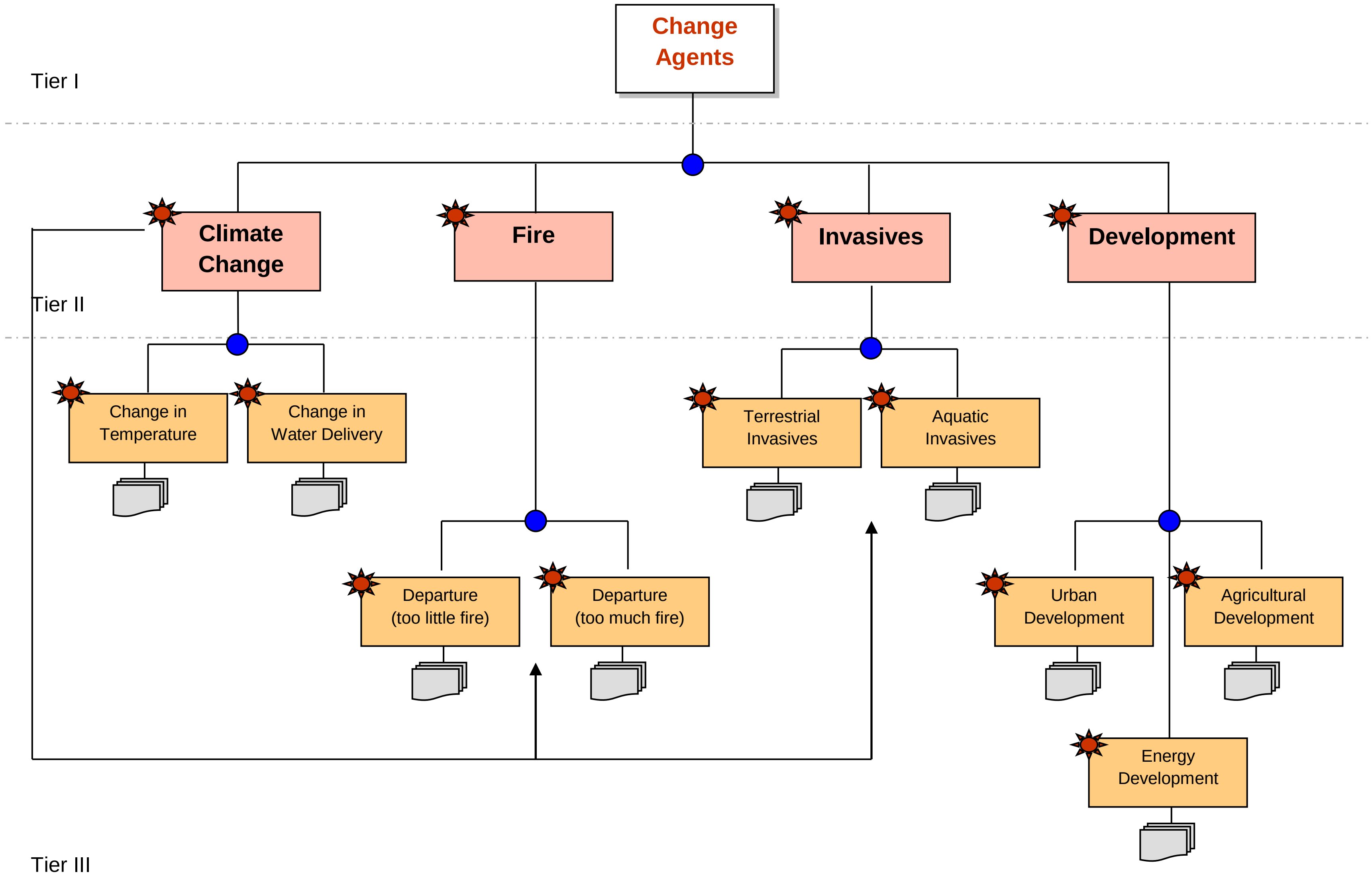

Figure 5-9. Draft EMDS model for change agents of interest. Note that input data for all topics are generated outside of EMDS.

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 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.

<u>**Note**: BLM approved the use of Data Basin on 3-1-11. It will provide a venue of group review of sets of models as they are completed. Folders will be organized in classes of terrestrial and aquatic conservation elements, change agents, and ecological integrity.</u>

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.

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.

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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 ethodology 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).

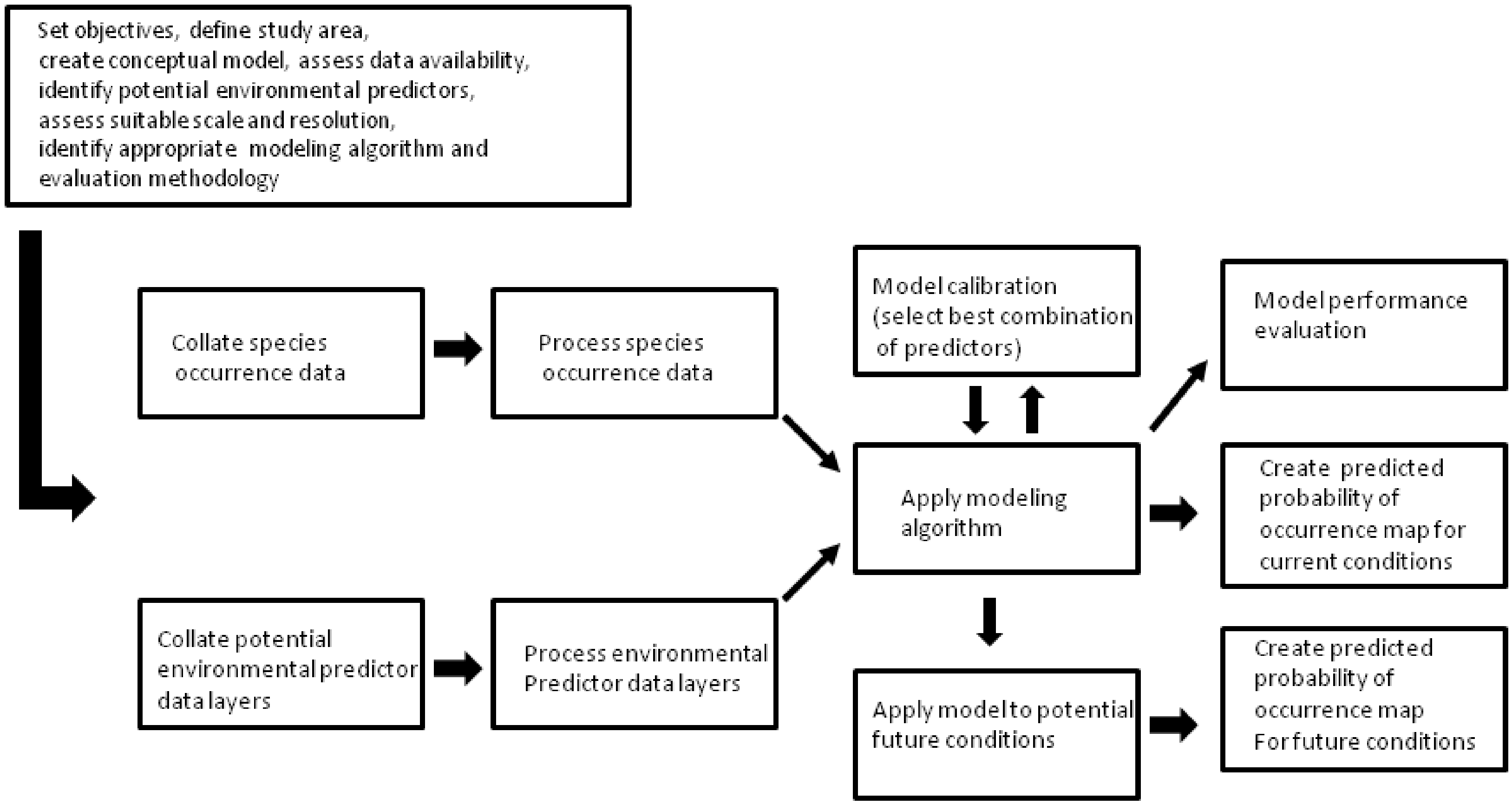

Figure 6-1. Flow diagram of species distribution modeling process (adapted from Pearson 2007).

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).

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

Table 6-1. Some available published species distribution modeling packages

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).

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

absence data is lacking; they are easy to interpret, able to incorporate interactions among predictors, and indicate the relative importance of predictors (Beauvais et al. 2006).

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 (terrestrial animal species), razorback sucker (aquatic animal species), pinyon pine (plant species), and cryptogamic crust (plant commuity). Additional examples are provided in the Appendix.

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

Golden eagles occur worldwide, but they have recently begun to decline in the western U.S. 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 Colorado Plateau 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 Colorado Plateau ecoregion. The following species-related management questions were identified by the Assessment Management Team (AMT) for inclusion in the Colorado Plateau 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 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/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). 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). In two coal-mining counties in eastern Utah, Bates and Moretti (1994) found active eagle nests in four different habitats: in trees on saltbush flats or low elevation riparian areas, on cliffs and escarpments in pinyon-juniper and talus territories, and in prominent trees in the aspen-conifer zone. 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 post-burn (Kochert et al. 1999) and it is likely that post-burn results in the Colorado Plateau would be similar to those in Idaho since they share similar vegetation communities.

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 67golden 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, location, and topography with the majority of deaths associated with shorter turbines (e.g. Type 13), end of row and second from the 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 in burned areas (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).

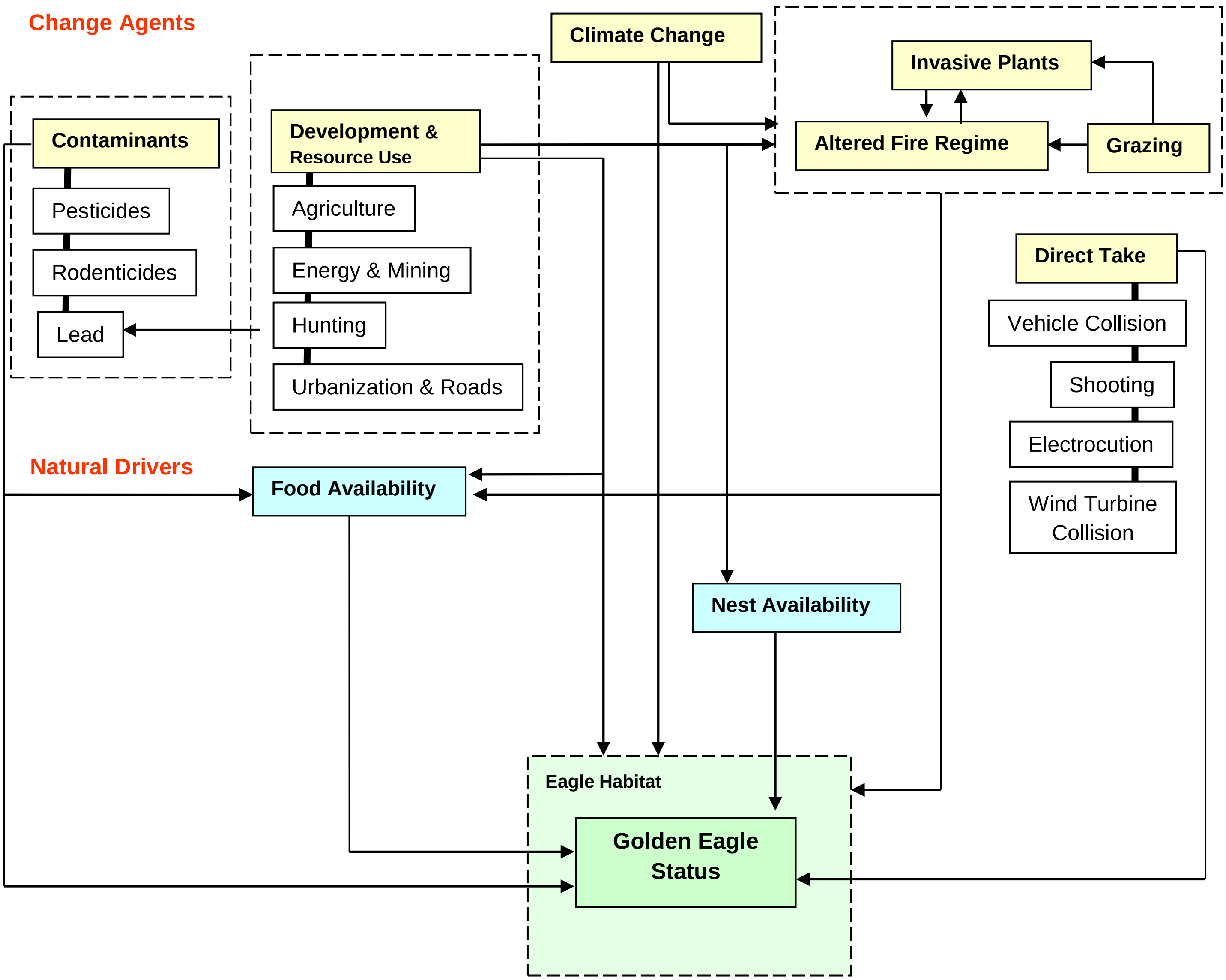

Figure 6-2. Principle interactions among population drivers and change agents for golden eagles in the Colorado Plateau.

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, SLOPE	**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 flyways	HABITAT	**1, 4**
*Identified seasonal habitats	HABITAT	**1**
Fire perimeters	HABITAT	**1, 3**
SWreGAP vegetation	VEGETATION	**1, 3**
Grazing pressure	**RESOURCE USE**	**1, 3**

6.9.4 Model Assumptions

In our modeling approach 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 Table 6-3 below that could be utilized for addressing management questions 1–3 and 5 and for parameterizing the habitat model. (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 developed to answer this management question.

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

   Transect locations marking the survey plots of Good et al. (2004).

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. What/where is the potential for future change to this terrestrial species in the near-term horizon, 2020 (development) and a long-term 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 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 management question 4 about movement corridors and connectivity, which may not be appropriate for this species.

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, typical territory size, 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.

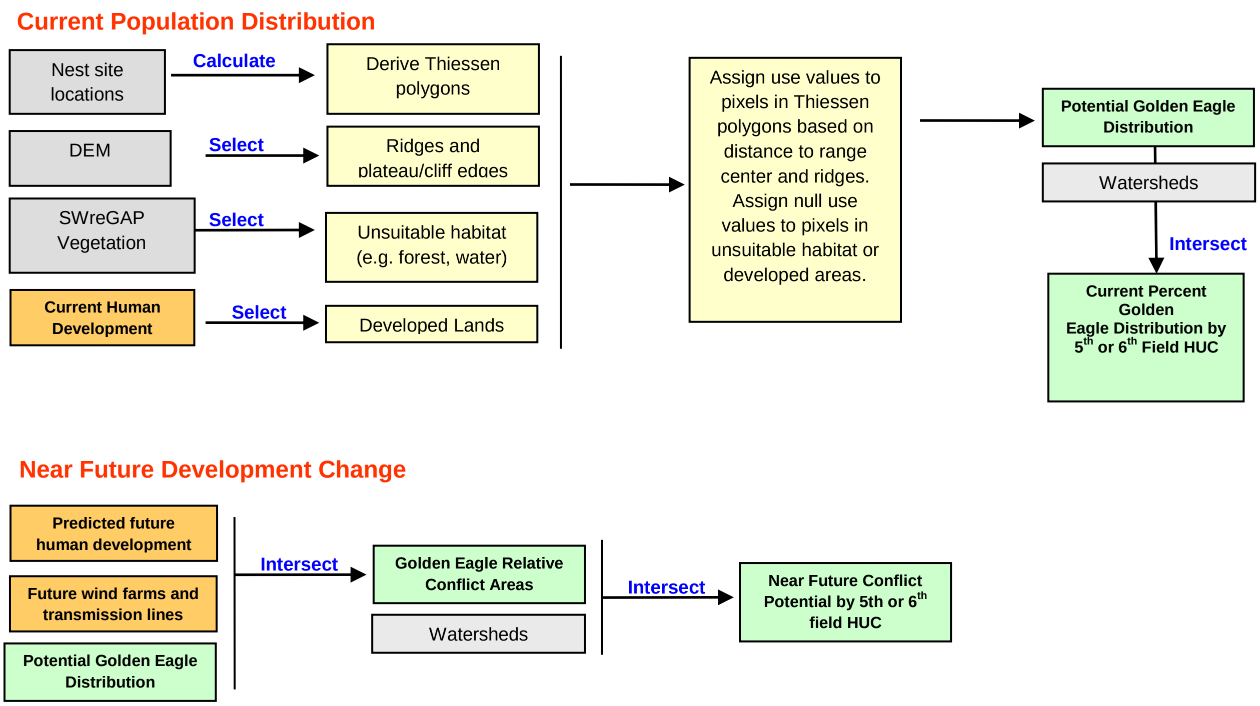

Figure 6-3. Datasets and processing steps for development of a golden eagle distribution model.

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6.10 Aquatic Wildlife Species Conservation Element: Razorback Sucker (*Xyrauchen texanus*)

6.10.1 Introduction

Information gained from this assessment will help provide the basis for future management planning across multiple spatial scales and jurisdictional boundaries and help direct future research in areas where knowledge gaps are identified. This assessment is not designed to capture every potential change agent, nor ecological function or service. Rather, it is intended to capture the elements generally considered to be most important to maintaining proper functioning conditions for razorback suckers. The rapid and general nature of this assessment precludes detailed, in-depth modeling.

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. This module of the REA attempts to gauge the resource requirements needed to address the full complement of management questions in a manner that has utility for the BLM for future planning purposes. Management questions for this aquatic wildlife species conservation element include:

Current Habitat

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

Habitat Restoration

3. Where are potential habitat restoration areas?
4. Where are potential areas to restore connectivity?

Forecast Future Conditions

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

Major change agents affecting the razorback sucker form the foundation of status and future condition forecasts for this REA. For razorback suckers, the most important change agents include:

Invasive Species	*non-native fishes (non-native crayfishes in some areas)
Development	*dam building
	*channelization
Resource Use	*water management – change in water regime and loss of connectivity
Climate Change	*change in form/amount/timing/duration of precipitation

The purpose of this Colorado Plateau REA species module is to assess the current status of the razorback sucker (*Xyrauchen texanus*) at the ecoregional scale and to investigate how their status may change in the future as a result of future development and climate change. The razorback sucker (also known as the humpback sucker or buffalo fish) was federally listed in 1991 as an endangered species (USFWS 1991; also IUCN Red-listed as Endangered, IUCN 2010). Endemic to and historically distributed throughout the

Colorado and Gila River basins (Minckley et al. 1991, NatureServe 2010, Schooley and Marsh 2007), the species has been nearly extirpated from Arizona and now occurs naturally only in lakes Mohave and Mead and in small populations in the Yampa and Green rivers of Utah and Colorado (Lanigan and Tyus 1989, Minckley et al. 1991, Mueller et al. 2000, Tyus and Karp 1989). Hatchery-reared fish have been reintroduced into Lake Havasu, the Colorado River below Parker Dam, and the Verde River (Douglas and Marsh 1998, Modde et al. 1996, Minckley et al. 1991, Tyus and Karp 1989). Critical habitat was designated by the USFWS in 1994 and a Recovery Plan finalized by the USFWS in 1998. General habitat types currently utilized by razorback suckers include wetlands, permanent rivers, streams, creeks and lakes, artificial impoundments, and irrigation channels.

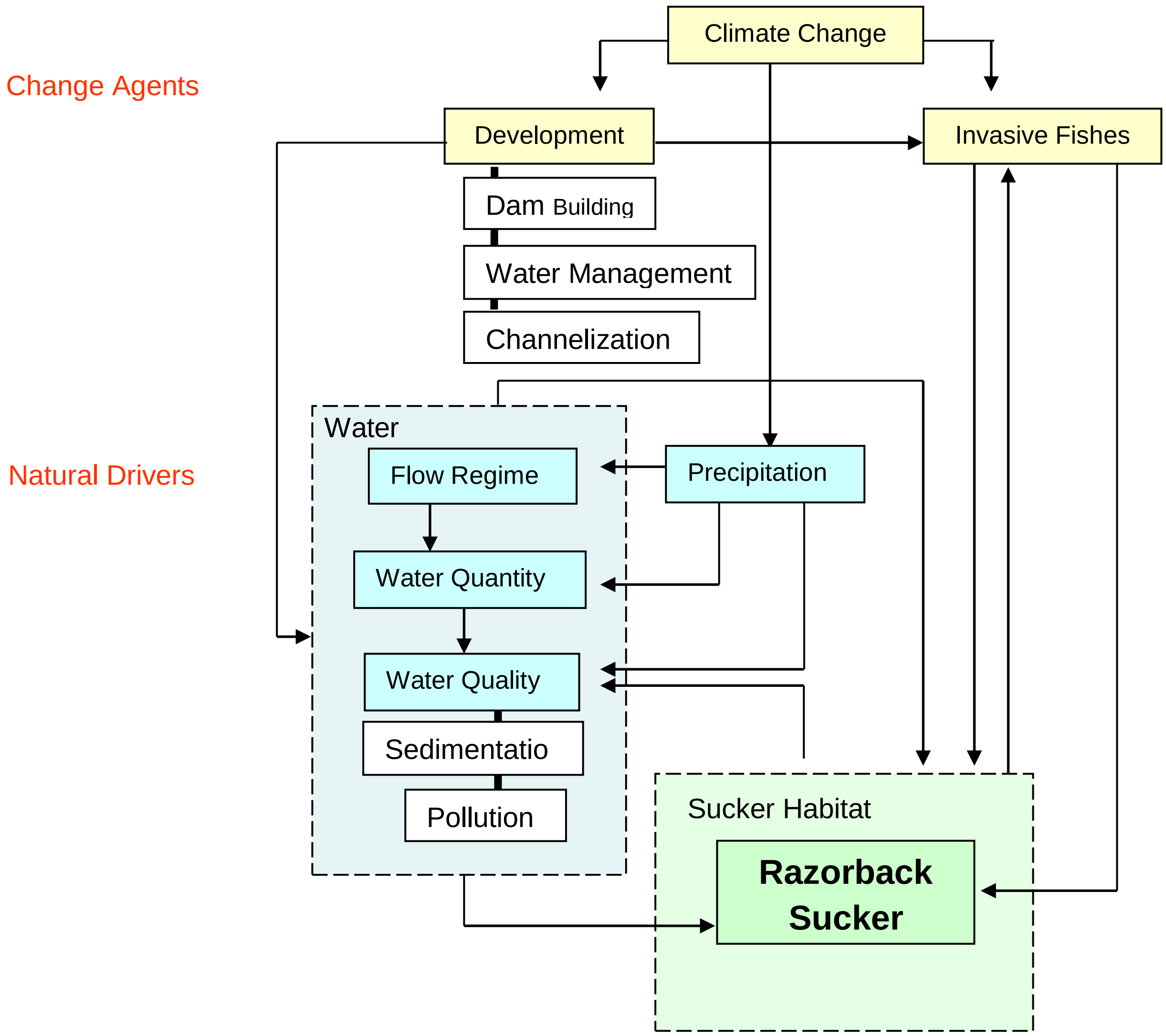
Given the relative interest in razorback sucker restoration over the past couple of decades, there is a surprising paucity of quantitative razorback sucker models found in primary literature. The only razorback sucker models we could find were related to preferred habitats (i.e., habitat associations, Gurtin et al. 2003), mathematical stocking survival (Schooley and Marsh 2007), population dynamics (Crowl and Bouwes 1997) and genetic diversity (Dowling et al. 1996, Dowling and McKinley 1993). Numerous relationships, however, between razorback suckers and various biological and physical parameters have been established in the literature. We have adapted these relationships (denoted in the figures below by connecting lines) into conceptual and application/method models that can be run in GIS.

6.10.2 Conceptual Model

The razorback sucker conceptual model (Figure 6-4) includes major change agents, natural drivers, and the pathways through which they influence the primary conservation element (i.e., razorback sucker). This conceptual model is not intended to capture all potential threats and change agents affecting razorback sucker currently or in the future. Rather, it is intended to capture examples of some of the major factors that influence razorback sucker populations. Primary threats to the razorback sucker include interactions with non-native fishes and human alteration of riverine habitats.

Habitat alterations of primary interest resulting from human development and resource use include dam operations that have substantially restricted the amount of suitable habitat available for the species during multiple life stages (Figure 6-4). Alterations caused by dam-building and subsequent flow management trigger detrimental changes in timing, magnitude and duration of winter and spring flows, altered river temperatures (Clarkson and Childs 2000), reduced flooding (USFWS 1990, Hedrick et al. 2009), and abatement of sedimentation (Johnson and Hines 1999) and gravel bar accretion. Channelization for agricultural or highway projects further reduces the amount of gravel bar and slow backwater areas necessary for nesting and fry nurseries. Detailed changes to razorback sucker habitats are described in the USFWS proposal to federally list the species (1990) and in the recovery plan (USFWS 1998).

Figure 6-4. Conceptual model for razorback suckers of the Colorado Plateau ecoregion.



Recruitment of larvae and young has been very low (or absent), despite protracted hatchery intervention practices and costly habitat restoration projects (Hedrick et al. 2009). Besides a lack of recruitment from loss of backwater habitat, it is limited primarily from the pervasiveness of predatory non-native fishes (Clarkson et al 2005, Jelks et al. 2008, Johnson et al. 1993, Marsh et al. 2003; see Table 1 in USFWS 1998 for a more detailed list). The presence of nonnative, invasive fish species can directly and indirectly influence razorback suckers by limiting the space available for razorback sucker to occupy (indirect), competing for razorback sucker food sources (prey; indirect), predating on eggs/larvae/juvenile razorback suckers (direct), or exhibiting aggressive behavior toward razorback suckers (direct, Figure 6-4). Lenon and others (2002) also noted that competition with and predation by non-native crayfishes may be a problem in some areas. Hybridization with other sucker species also occurs (Tyus and Karp 1990, Minckley et al. 1991).

In addition to human development and pressures from invasive species, climate change may have an additional impact on flow regimes that are important to the razorback sucker (Figure 6-4). Climate change will have a direct impact on the type, amount, and timing of precipitation. Because stream flows are closely related to precipitation patterns in the region (e.g., Christensen and Lettenmaier 2007, but see discussion in Hoerling et al. 2009), climate change will affect the aquatic environment through influencing the flow regimes, water quality, and water quantity, all of which are important drivers of razorback sucker populations.

6.10. 3 Required Input Data Layers

We compiled a list of datasets we think are most relevant for answering the management questions (Tables 6-4 and 6-5). In some cases, the datasets exist but may have varying levels of usability and/or availability; we won't be able to determine usability until we have the datasets in hand. In other cases, we were uncertain of the dataset availability ("location" not identified in the tables) or even if the datasets exist ("source" not identified in the tables). Datasets are divided into raw (unprocessed) datasets (Table 6-3) and previously processed datasets (Table 6-4).

6.10.3.1Raw Data

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

Dataset	Source	Location	Management Question(s)
Wetlands (current and historic, by state) AZ, CO, NM, UT	NWI (USFWS)	http://www.fws.gov/wetlands/	3-5
30-arc second Digital Elevation Model (DEM)	USGS-Gotopo30	http://eros.usgs.gov/#/Find_Data/Products_and_Data_Available/gtopo30_info	1, 3-5
Stream locations	NHD (USGS)	http://nhd.geo.usgs.gov/viewer.htm	1, 3-5
303(d) waters	EPA (federal)	http://www.data.gov/geodata/g599873/	3-5
Selenium-bearing waters	State EPA offices?		3,5
Critical habitat	USFWS	http://criticalhabitat.fws.gov/docs/crithab/crithab_all/crithab_all_layers.zip http://www.data.gov/raw/1852	All
Dam locations (major/minor)	USACE	http://nationalatlas.gov/mld/dams00x.html	All
Razorback sucker distribution	NatureServe	http://www.natureserve.org/explorer/servlet/NatureServe?searchName=Xyrauchen+texanus	All
Native fish locations	Western Native Fishes (AFS) database	CBI offices, 2DVDs	All
Razorback sucker distributions	Respective state agencies (AZ, CO, NM, UT)		All
Fish locations (native and nonnative)	Natural history programs	Get from BLM since they are a subscriber	All

6.10.3.2 Previously Processed Data

Table 6-5. Previously processed datasets proposed for use in evaluating key management questions related to razorback suckers.

Dataset	Source	Location	Management Question(s)
Estimated water use, by county	USGS	http://www.blm.gov/nils/GeoComm/home_services.html	3-5
Human Footprint of the West	USGS	http://sagemap.wr.usgs.gov/	3-5
Aquatic Conservation Status (threats) to freshwater ecoregions of the world	WWF	http://www.feow.org/downloads.php?PHPSESSID=67d7d203216b980841a8747e26a58e30	3-5
Priority conservation areas	TNC-DOI	http://azconservation.org/downloads/category/ecoregional_assessment/	3-5
Priority conservation areas	CEC	http://www.cec.org/atlas/	3-5
Fish locations (native)	TNC-AZ 2010 Freshwater Assessment	In-house (CBI)	All
Potentially suitable habitat	NMG&F		2-5
Impoundment backwater locations	USBOR, CSU	http://cedb.asce.org/cgi/WWWdisplay.cgi?137316	All
Projected precipitation/runoff for the Colorado River Basin	VIC (Variable Infiltration Capacity) model		5

6.10.4 Attributes, Indicators, and Metrics

There are a number of potential attributes that could be utilized for addressing the management questions. Oliver and Tuhy (2010) provide a list far more detailed than necessary to answer the questions. Regardless, we have provided the full list, below, with proposed attributes/indicators/metrics for use in this assessment noted with bolded text. Key indicators include length of free-flowing rivers and size of reservoirs, aquatic habitat types (e.g., pool, riffle, backwater, etc.), water temperatures, presence of non-native, invasive fishes, and alteration of natural flow regimes.

Table 6-6. Full list of Attributes, Indicators, and Metrics that may be useful for evaluating the status and condition of razorback suckers. Table after Oliver and Tuhy 2010.

Species	Key Ecological Attribute	Indicator	Indicator rating				Basis for Indicator Rating	Comments
			Poor	Fair	Good	Very Good		
Razorback sucker	**spawning migrations and other movements**	**length of suitable aquatic habitat**	**<20 km in rivers or <8 km in reservoirs**	**20–60 km in rivers or 8–14 km in reservoirs**	**60–100 km in rivers or 14–20 km in reservoirs**	**>100 km in rivers or >20 km in reservoirs**	**Valdez et al. (2002) and sources cited therein**	**In both rivers and reservoirs some razorback suckers make very long migratory movements for spawning, although at other times they tend to be relatively sedentary. For example, Tyus (1987) found that 28 of 52 tagged individuals moved an average of 59.3 km in □8 years, the longest movement being 206 km in 5 years, and other studies have documented movements of □100 km in rivers. Reported movements and apparent spawning migrations in reservoirs have involved distances of 8–24 km.**
Razorback sucker	population ("minimum viable population")	no. of individuals (adults)	<5,763	5,763	>5,763	»5,763	Valdez et al. (2002)	Calculations of minimum viable populations are based on assumptions and best guesses.
Razorback sucker	**general habitat**	**water body***	**other (e.g., irrigation canals)**	**small rivers, reservoirs**	**medium rivers**	**large rivers**	**various authors**	**Although aging adults survive in the rated aquatic situations ("fair"–"very good"), larvae and juveniles do not, presumably because of predation by nonnative fishes.**
Razorback sucker	**breeding habitat**	**river feature (April–June)***	**rapids, riffles**	**slow runs, fast runs, eddies, shorelines**	**pools, off-channel flooded gravel pits**	**backwaters**	**Osmundson et al. (1995)**	
Razorback sucker	**habitat**	**river feature (July)**	**fast runs, rapids, eddies, off-channel flooded gravel pits**	**riffles, shorelines**	**slow runs, pools**	**backwaters**	**Osmundson et al. (1995)**	
Razorback sucker	**habitat**	**river feature (August–October)**	**fast runs, rapids, riffles, shorelines, backwaters, off-channel flooded gravel pits**	**eddies**	**pools**	**slow runs**	**Osmundson et al. (1995)**	
Razorback sucker	**habitat**	**river feature (November–March)**	**fast runs, rapids, riffles, shorelines, off-channel flooded gravel pits**	**backwaters**	**slow runs, eddies**	**pools**	**Osmundson et al. (1995)**	

Table 6-6 (continued**).** Full list of Attributes, Indicators, and Metrics that may be useful for evaluating the status and condition of razorback suckers. Table after Oliver and Tuhy 2010.

Species	Key Ecological Attribute	Indicator	Indicator rating				Basis for Indicator Rating	Comments
			Poor	Fair	Good	Very Good		
Razorback sucker	**habitat (activity, feeding, survival, maturation, reproduction, etc.)**	**water temperature (summer)***	**<29 °C or <12 °C**	**26.9–29 °C or 12–17.5 °C**	**24.8–26.9 °C or 17.5–22.9 °C**	**22.9–24.8 °C**	**Bulkley and Pimentel (1983)**	**Tailwaters of impoundments are typically much too cold. It is likely that the drop in water temperature below the dam after impoundment of the Green River to create Flaming Gorge Reservoir has caused the disappearance of *X. texanus* for 105 km downstream to the confluence of the Yampa River, where warmer water enters the Green River (Bulkley and Pimentel 1983).**
Razorback sucker	**spawning**	**water temperature***	**<6 °C or >22 °C**	**6–9.5 °C or 19–22 °C**	**9.5–15 °C or 15–19 °C**	**15 °C**	**Tyus (1987), Valdez et al. (2002) and sources cited therein**	
Razorback sucker	**hatching success**	**water temperature***	**<10 °C or >30 °C**	**10–15 °C or 25–30 °C**	**15–20 °C or 20–25 °C**	**20 °C**	**Marsh and Minckley (1985), Valdez et al. (2002) and sources cited therein**	
Razorback sucker	spawning and nursery habitat	aquatic feature*	—	mouths of tributaries to rivers	shoals, shallow near-shore areas (e.g., on outwash fans) of reservoirs; low-velocity shoreline habitats in alluvial reaches of rivers	off-channel flooded gravel pits, backwaters, inundated floodplains, broad alluvial flatwater areas of rivers	Valdez et al. (2002) and sources cited therein	
Razorback sucker	spawning and nursery habitat	substrate	mud, silt, fines, sediment	—	coarse sand	cobble, gravel	Valdez et al. (2002) and sources cited therein	
Razorback sucker	spawning and nursery habitat	water velocity	»1.0 m/s	≥1.0 m/s	<1.0 m/s	0 m/s or «1.0 m/s	Valdez et al. (2002) and sources cited therein	
Razorback sucker	spawning and nursery habitat	water depth	»1.0 m	—	—	<1.0 m	Valdez et al. (2002) and sources cited therein	

Table 6-6 (continued**).** Full list of Attributes, Indicators, and Metrics that may be useful for evaluating the status and condition of razorback suckers. Table after Oliver and Tuhy 2010.

Species	Key Ecological Attribute	Indicator	Indicator rating				Basis for Indicator Rating	Comments
			Poor	Fair	Good	Very Good		
Razorback sucker	**survival of eggs, larvae, and, fry (i.e., recruitment)**	**nonnative fishes***	**present**	**—**	**—**	**absent**	**Minckley et al. (1991), Valdez et al. (2002), and others**	**Presence or absence of nonnative fishes is the most important single factor in the ecology of this species. If nonnative fishes are present, the likelihood of successful recruitment is very low or nonexistent. However, nonnative fishes are so well established throughout the Colorado River system that to eradicate them from any water body suitable for *X. texanus*, other than off-channel pools or ponds not connected to rivers, may be a practical impossibility. Attempts to control nonnative fishes in rivers and reservoirs for the benefit of *X. texanus* thus far have not restored recruitment.**
Razorback sucker	hybridization	white sucker (*Catostomus commersoni*) and possibly other species of suckers	present in large numbers	present in moderate numbers	present in small numbers	absent	Valdez et al. (2002) and sources cited therein	Hybridization with the flannelmouth sucker is known to have occurred historically, presumably under completely natural conditions. Hybridization with the nonnative white sucker is considered to be a potential threat in the upper Colorado River basin (Valdez et al. 2002).
Razorback sucker	water quality (survival, food sources)	pollution (from oil and gas extraction, mining, agricultural runoff, industrial and municipal effluents, etc.)	existing	—	—	none	Valdez et al. (2002) and sources cited therein	
Razorback sucker	**spawning, nursery, and adult habitats, water temperatures, food sources**	**alteration of natural flow regimes (impoundments, channelizations, diversions, levees, etc.)***	**existing or planned**	**—**	**—**	**none**	**Valdez et al. (2002) and sources cited therein**	**Restoration and maintenance of high spring flows resulting in inundation of floodplains is of importance to all life stages and especially for reproduction. “These [flooded] areas provide warmwater temperatures, low-velocity flows, and increased food availability” (Valdez et al. (2002).**

6.10.5 Model Assumptions

In this razorback sucker process model, we make the following assumptions:

1) The current distribution of razorback suckers does not necessarily reflect preference for particular types of habitats (e.g., impoundments),
2) razorback suckers utilize different habitats during different life stages,
3) razorback suckers require a variety of complex aquatic habitats to thrive (e.g., floodplains, wetlands, backwaters, large free-flowing river systems, complex riffle/pool habitats, etc.),
4) razorback suckers are not well-adapted to living in habitats where non-native, invasive fishes abound, and
5) razorback suckers need access to habitats with unaltered flow regimes.

6.10.6 Methods and Tools

Given the relative dearth of existing razorback sucker models in the literature, our general approach is to develop conceptual ecological and process/application models based on literature-established relationships between razorback suckers and various threats/drivers (both direct and indirect) of their existing populations. 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 6-5 and 6-6) 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 razorback suckers. Output datasets will be displayed at the 5th field Hydrologic Unit Code (HUC), where appropriate. In general, existing and output datasets address 1) populations and 2) threats/stressor locations. Two *example* process/application models are provided, below (Figures 6-5 and 6-6). 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 several examples (Figures 6-5 and 6-6).

The first example model (Figure 6-5) addresses current habitat management questions (MQ 1 and 2). An example of the proposed method/analysis flow we would take to answer management question one (Q1 in Figure 6-5) would be to compare razorback sucker distributions from several different sources (respective State agencies) to two "global distribution" datasets (e.g., NatureServe and Western Native Fishes database). If there were no unique records (i.e., records that didn't overlap with one of the "global distribution" datasets), we'd simply use one of the global distribution datasets and intersect it with the 5th field HUCs file. The resulting dataset would indicate the current razorback sucker distribution by HUC5s. If there were unique records, we would "merge" the various files and proceed as noted above.

The second example model (Figure 6-6) addresses population status, habitat restoration management, and potential climate change-related questions (3-5). An example of the proposed method/analysis flow we would take to answer management questions three and four (MQs 3-4 in Figure 6-6) would be to intersect current razorback sucker locations with water quality-limited streams. We would then select records based on individual water quality parameters of interest (e.g., presence of selenium, high water temperatures, etc.), and intersect the resulting file with the 5th field HUCs file. The resulting dataset would indicate locations where razorback sucker habitats are threatened by the water quality parameter of interest (at the HUC5-level), locations where populations may be at risk, and locations that may benefit from restoration actions. As an example of how we could start answering management question five (MQ5 in Figures 6-5 and 6-6), we would intersect current razorback sucker habitats affected by a change agent of interest with the projected change in precipitation (for a given future year). We would then select areas where there are projected decreases in precipitation and intersect those records with 5th field HUCs. The resulting dataset(s) would indicate locations (at the HUC5-level) where razorback sucker habitats are potentially at risk from decreased precipitation and/or increased water temperatures (e.g., Christensen and Lettenmaier 2007, Christensen et al. 2004, Hendrick et al. 2009). To explore how development may potentially impact razorback suckers in the near-term horizon (2020; MQ5 in Figure 6-5), an example approach would be to merge various water source datasets into a single file, perform a union of the resulting file with one or more of the change agent datasets (e.g., invasive fish distributions, dam/diversion locations (development), 303(d)-listed streams, etc.), then select areas of overlap, export those records to a new file and intersect the resulting file with the 5th field HUCs file. The resulting dataset would indicate the current razorback sucker habitats (by HUC5s) potentially affected by development.

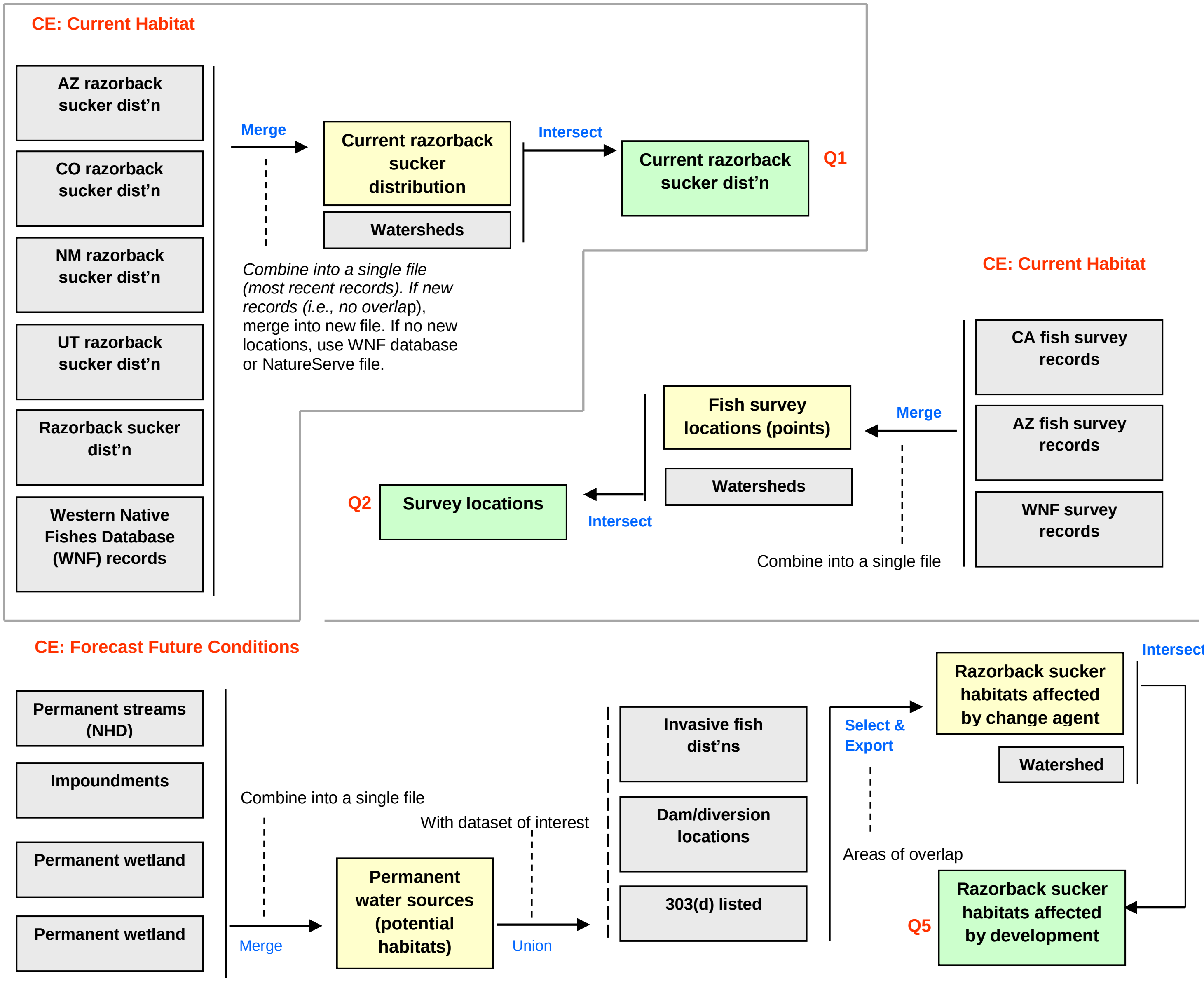


Figure 6-5. Process model illustrating relationships between conservation elements, datasets, and process steps utilized to arrive at answers to specific management questions for razorback suckers of the Colorado Plateau ecoregion.

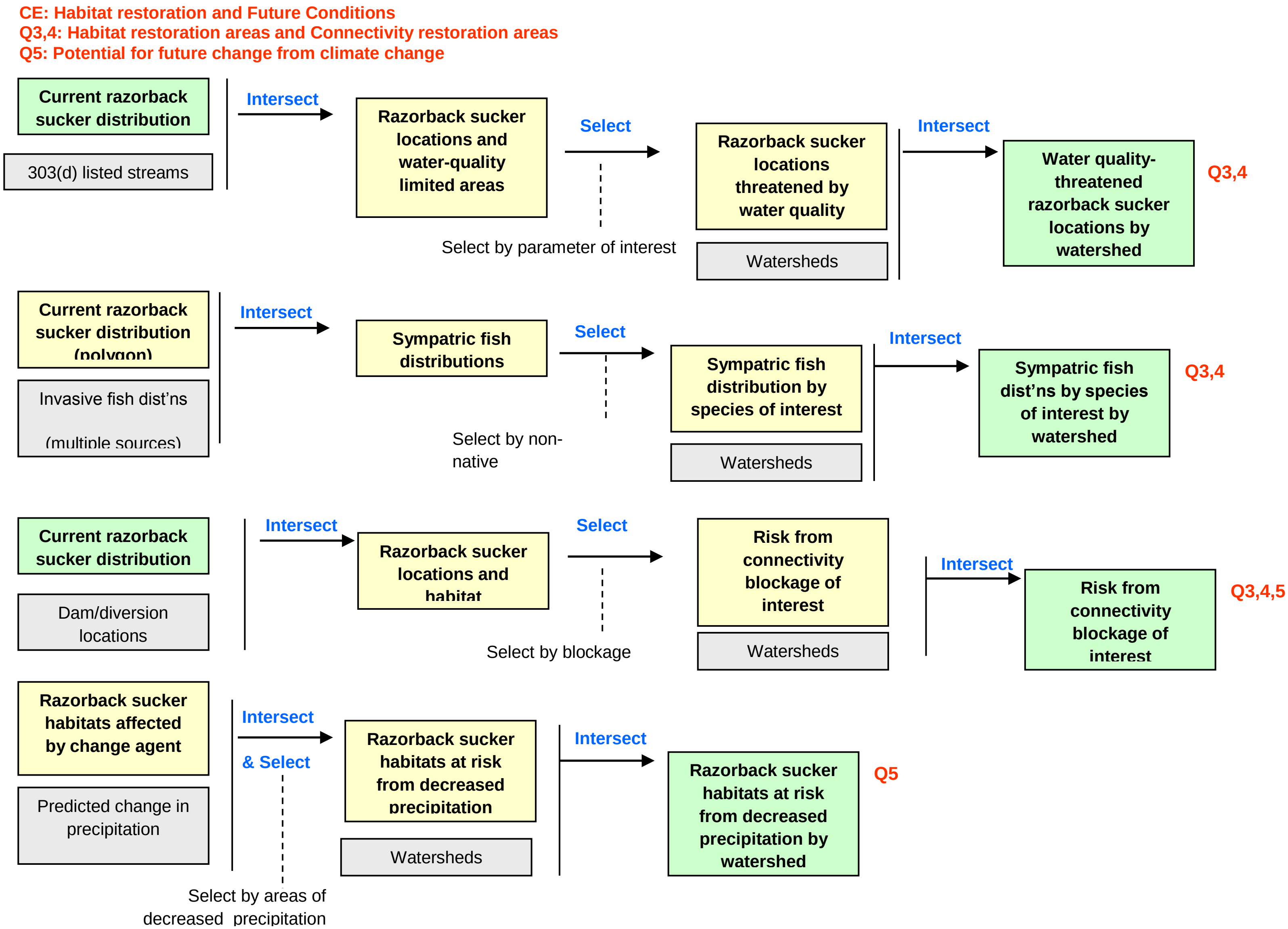


Figure 6-6. Process model illustrating relationships between conservation elements, datasets, and process steps utilized to arrive at answers to specific management questions for razorback suckers of the Colorado Plateau ecoregion.

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6.11 Plant Species Conservation Element: Pinyon Pine (*Pinus edulis*)

6.11.1 Introduction

Note: A standard MaxEnt approach using FIA inventory plot data is proposed in this module. Recently, models for pinyon pine have been completed for all of the Colorado Plateau. The Dynamac team will work with Jacob Gibson, Utah State University, and Tom Edwards, USGS, to incorporate the new models into the Colorado Plateau REA.

This module of the Rapid Ecological Assessment (REA) for the Colorado Plateaus ecoregion investigates the relationships between the plant species conservation element pinyon pine (*Pinus edulis*) and the change agents climate, grazing, insects, and fire.

The purpose of this species module is to assess the current status of pinyon pine at the ecoregional scale and to investigate how its status may change in the future as a result of future development and climate change. 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. This methods module attempts to gauge the resource requirements needed to address the management questions in a manner that has utility for the BLM for future planning purposes. Management questions for this terrestrial plant species conservation element include:

1. What is the most current distribution of this species (and historic distribution if available)?
2. Where are 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. What 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 and sites located?

Pinyon pine covers approximately 14.9 million acres in Utah, Colorado, New Mexico, and Arizona with sporadic occurrences in Texas, Oklahoma, Wyoming, California, and Mexico. The optimal range of precipitation for this species is 250 mm–560 mm (10–22 inches) at elevations between 1,370 and 2,440 m (4495–8005 feet, Burns and Honkala 1990). Pinyon pine is often associated with juniper forming pinyon pine-juniper woodlands. Pinyon pine-dominated stands at higher elevations tend to form more closed-canopied woodland conditions than at lower elevations, including a significant shrub component of oaks and alderleaf mountain mahogany (*Cercocarpus montanus*) and limited grasses.

Because of the physical growth requirements, it is not uncommon to see this species shift its range and extent of cover based on these environmental factors. In some areas, pinyon pine will be in rapid decline, while in others, expansion. Studies of fossils from the 13th and 14th centuries show that pinyons have historically expanded in response to increases in precipitation and declined in response to prolonged periods of drought on an approximately decadal timescale (Gray et al. 2006). Because the cones are often dispersed by birds and humans, they can be carried long distances and establish where conditions are the most favorable.

Decadal expansion and contraction of pinyon ranges can be seen as fortuitous or problematic, depending on the perspective of the local stakeholders. Pinyon-juniper woodland is valued for providing food and habitat diversity to wildlife such as desert bighorn sheep, mule deer and a wide variety of birds, mammals and reptiles (Weisberg 2009). Woodland cover also reduces soil loss and retains snow cover and soil moisture. Pinyon-juniper expansion into shrublands can threaten sage grouse and pygmy rabbit habitats. Encroachment of pinyon-juniper woodlands onto grasslands leads to rangeland recovery efforts in which pinyon-juniper woodlands are thinned or burned.

Pinyon-juniper woodlands can either be carbon sources or sinks, depending on whether their range is expanding or declining. Neff et al. (2009) found that soil carbon and nitrogen pools accumulate at higher rates during periods of pinyon-juniper expansion due to the decaying woody biomass beneath the increased canopy cover. Low residence time and physical instability of the particulate organic matter (POM) lead to rapid losses of soil carbon and nitrogen during decline or thinning (Neff et al. 2009). Disturbance of soil biomass accumulated within the past century risks releasing carbon and nitrogen stocks that have not been stabilized in the mineral soil fraction.

6.11.2 Conceptual Model

The main driver of change in pinyon pine populations is climate (Figure 6-7, Gray et al. 2006, McDowell et al. 2008, Barger et al. 2009, Allen et al. 2010). Historically, pinyon pine range has expanded when temperatures were relatively cool and wet for the region (Barger et al. 2009). Throughout its range, pinyon-juniper woodland has expanded to approximately 10 times its pre-settlement range, due to changes in fire regimes, grazing, climate change, and increased atmospheric CO_2 (Miller and Rose 1999). At higher elevations, pinyons and junipers have expanded into ponderosa pine habitats, and at lower elevations, they are increasingly occupying grassland and shrubland (Miller and Tausch 2002). The increase in woodland has altered fire regimes from a frequent low-intensity fire regime to a low frequency high-severity fire regime; because the pinyon-juniper woodlands can now support crown fires, the overall pattern of pinyon pine distribution may change in the future.

In some regions, there has been a marked decline in pinyon pine. Over recent years, temperatures have increased and precipitation has decreased resulting in overall water stress. Stressed trees are then more susceptible to insect infestation and fires: an outbreak of bark beetles created a pinyon pine die-off in the region between 2000 and 2007. Drought-stressed pines are more vulnerable to large scale beetle attacks, and Raffa et al. (2008) showed that drought in the early 2000s exceeded the impacts and extent of droughts during the 1930s and 1950s.

Environmental factors such as changes in water availability from increased tree density may help explain why the pines have shown evidence of drought stress in recent years. Groundwater movement, retention, and absorption are all determined by soil properties. As a major contributor to plant health, soils are an indicator of where pinyon pines are likely to succeed or fail. Physical and chemical properties of soils determine plant available moisture and nutrients. Cobb et al. (1997) showed that soil texture and mineral composition are linked to pinyon pine stress and insect susceptibility. Soils affect available moisture, resin production, and nutrient availability, which in turn affect insect resistance.

Standing dead trees caused by increased mortality from insect infestation increase fire risk in pinyon-juniper woodlands (Figure 6-7). Fire regimes in the region are also influenced by anthropogenic factors. Prescribed fires are used to convert woodland to grassland. Grazing affects the fire return interval as well; it increases pinyon pine populations by reducing soil resource competition from grasses. Historic grazing led to a decrease in cool season grasses and an increase in grazing-resistant species (increasers). In some regions, over-grazing caused desertification of the understory. The resulting bare spaces may have served as a fire suppressant, but they also created erosive watershed conditions with increased runoff and permanent productivity losses (Campbell1954, Ellison 1960, Burkhardt and Tisdale 1976, Miller and Rose 1999).

Humans directly affect pinyon-juniper woodlands through land conversion to rangelands and through land treatments such as fuels treatment, mastication, biofuel production, or restoration thinning. Historically chaining was employed to remove large areas of pinyon-juniper for land conversion to rangelands. More recently, vegetation manipulation is implemented for restoration and removal of pinyon-juniper from areas of expansion through prescribed fire, mechanical treatments, and also harvest for Christmas trees, firewood, and biofuel production. Energy companies such as Apollo Bioenergy are evaluating the use of pinyon-juniper biomass in biofuel production (Scott 2010). The “WECHAR” bill to develop biochar technology suggests that pinyon-juniper woodlands are invasive species which can be used in biochar production (Reid et al. 2009).

Bird species, such as pinyon jay and Clark’s nutcrackers, disseminate pinyon pine seeds. The seeds within the cones are large and wingless, and therefore they can’t be disseminated by the wind. Pinyon jays are the most significant distributors of pinyon seeds, becaue they harvest from both green and mature cones, cache seeds in hospitable environments, transport seeds the greatest distance (up to 12 km), and distribute them evenly across the landscape (Ronco, 1990).

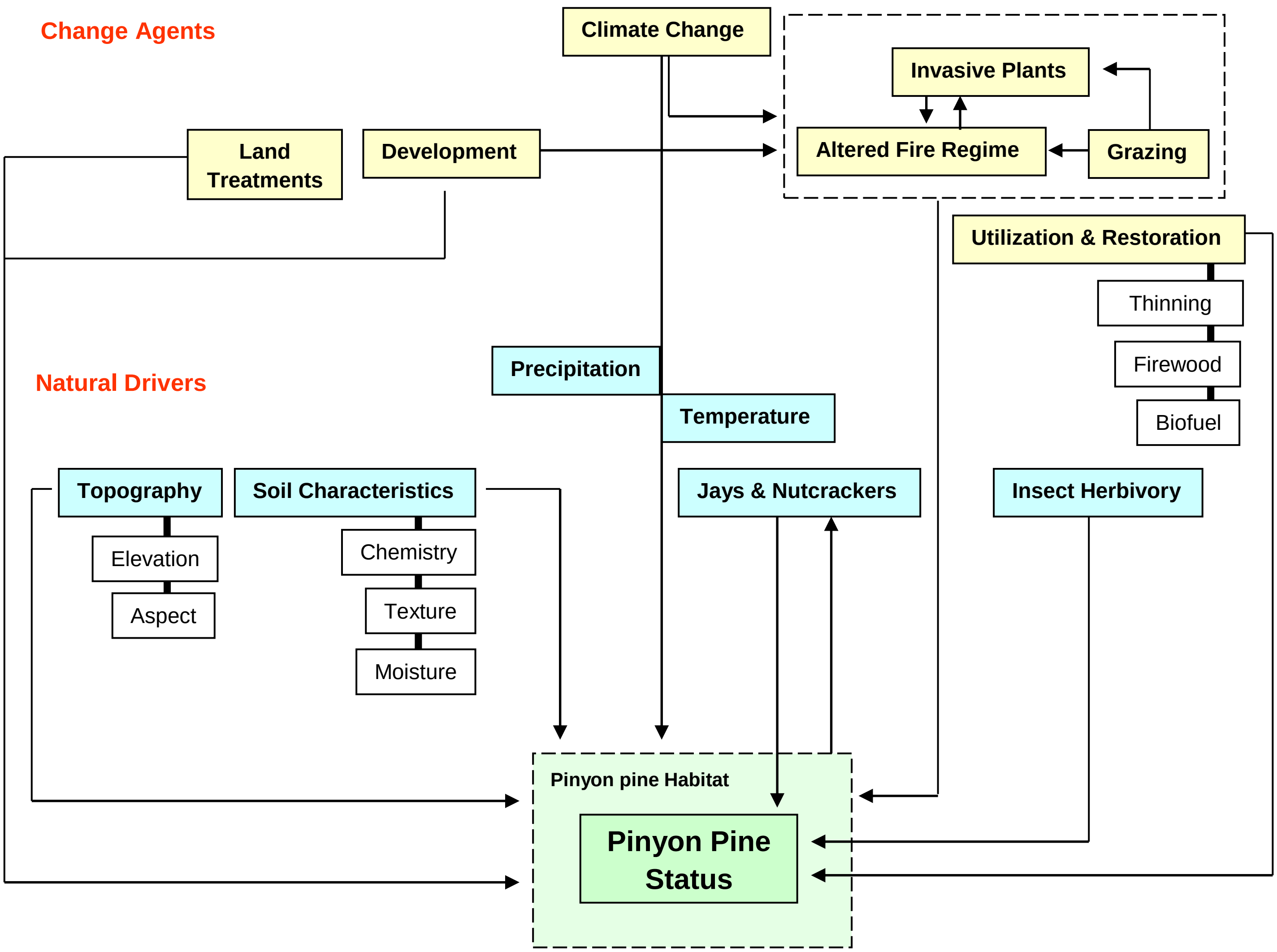


Figure 6-7. Conceptual model for Pinyon pine in the Colorado Plateau ecoregion

6.11.3 Required Input Data Layers

To make a complete model of the influences of all of the above factors, one would need datasets representing pinyon pine distribution and relative density, detailed soils data representing soil orders, climate regimes, textures and mineralogies, grazed lands, temperature and precipitation change projections, fire forecasts, insect distributions, data on tree mortality, and understory ground cover. Most of these data are available and are listed with sources and associated management question below (Table 6-7)

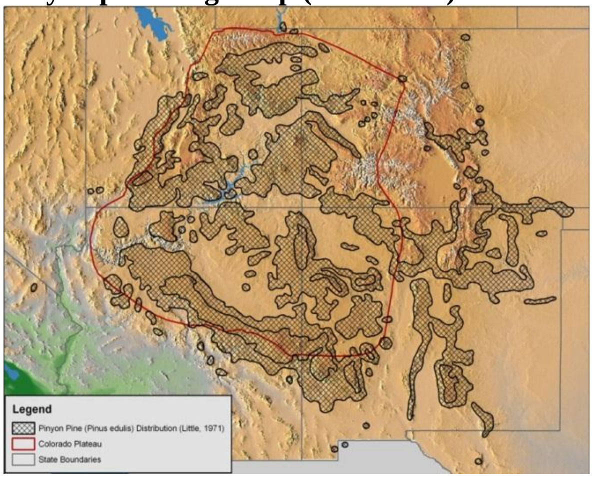
Table 6-7. Tentative data needs associated with management questions related to pinyon pine.

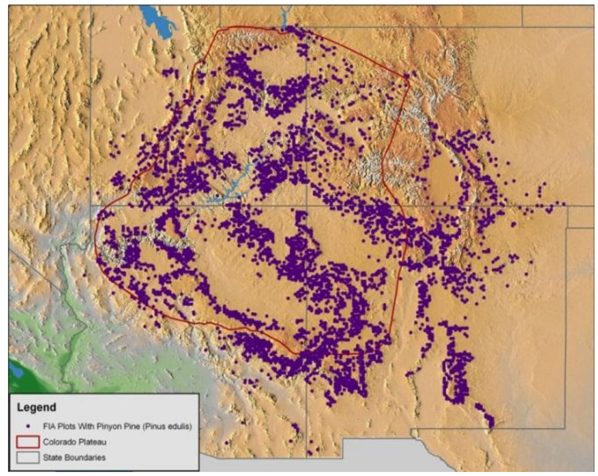
Dataset	Source	MQs
STATSGO or SSURGO	NRCS	1, 3, 4
Wildland fire potential	USFS	4
Grazed lands	BLM	4
FIA pinyon pine inventory plots	Forest Inventory and Analysis Program (FIA), USFS http://www4.nau.edu/direnet/data/index.html	1, 2
Pinyon range map	USGS (Little, 1971) http://esp.cr.usgs.gov/data/atlas/little/	1, 2
NED	USGS	4
Pinyon Ips beetle distribution	Univ. of Wisconsin	4
Historic precipitation change (1961–1990)	VEMAP	4
Historic temperature change (1961–1990)	VEMAP	4
Pinyon pine mortality datasets: 2000-2007	USFS http://www4.nau.edu/direnet/data/index.html	1,3,4

Table 6-8. Previously processed datasets used for modeling pinyon pine.

PREVIOUSLY PROCESSED DATA	LOCATION
Pinyon Pine Inventory Plots	Forest Inventory and Analysis Program (FIA)

Pinyon pine range map (Little 1971).





FIA pinyon pine inventory plots.

Figure 6-8. Little's 1971 range map of pinyon pine in a four state area of the Southwest on the left and, on the right, a corresponding map of Forest Inventory and Analysis (FIA) inventory plots that can be treated as occurrence data in the modeling of pinyon pine distribution.
http://www.usgs.nau.edu/global_change/RangeMaps.html
http://www.mpcer.nau.edu/pjwin/pinyon_pine.html

6.11.4 Attributes and Indicators

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. We will use attributes and indicators related to climate, soil, beetle distribution, and fire regime as ranges of optimal environmental response in the MaxEnt modeling for pinyon pine (Table 6-9).

Table 6-9. Key ecological attributes and associated indicator measures for pinyon pine in the Colorado Plateau ecoregion.

Species	Key Ecological Attribute	Indicator	Indicator rating				Basis for Indicator Rating	Comments
			Poor	Fair	Good	Very Good		
Pinyon pine	Habitat	Elevation	<1400 m	---	1400 m – 2700 m	2100 m – 2400 m	Cronquist et al. (1972)	
Pinyon pine	Dispersal	Pinyon jays, Clark's nutcracker					Chambers (1999)	
Pinyon pine	Competition	Grazing	none	-----	Non-degrading	---------	Barger et al. (2009)	
Pinyon pine	Competition	Vegetation	Grasses, Ponderosa Pine				Barger et al. (2009)	
Pinyon pine	Mortality	Fire return interval	<100 years		100 to 300 years	>300 years	Keeley (1981)	
Pinyon pine	Mortality	Restoration thinning	----	-----	----	-----	Neff et al. (2009)	
Pinyon pine	Mortality	Bark beetles	----	---	----	----	Bentz et al. (2010)	
Pinyon pine	Mortality	Firewood harvest	---	---	---	---		
Pinyon pine	Mortality	Biomass harvest	---	---	---	---	Reid et al. (2009), Scott (2010)	
Pinyon pine	Climate	Precipitation	<102 mm		102 mm – 520 mm		Ffolliott (1974)	
Pinyon pine	Climate	Temperature	18° C > , < 24° C				Anderson (2002)	
Pinyon pine	Climate	Frost free days	>120		<120		Ronco (1990)	
Pinyon pine	Water Availability	Soil moisture regime	Torric/ ustic				Peterman (personal communication, 2011)	Correlation analysis performed with GIS by Wendy Peterman, Soil Scientist.
Pinyon pine	Water Availability	Cemented calcium carbonates in soils	calcic		None		Peterman (personal communication, 2011)	Correlation analysis performed with GIS by Wendy Peterman, Soil Scientist.
Pinyon pine	Water and Nutrient Availability	Soil particle size	Medium to coarse		Fine		Peterman (personal communication, 2011)	Correlation analysis performed with GIS by Wendy Peterman, Soil Scientist.
Pinyon pine	Water and Nutrient Availability	Soil Order	Entisols		Mollisols		Peterman (personal communication, 2011)	Correlation analysis performed with GIS by Wendy Peterman, Soil Scientist.

6.11.5 Methods and Tools

MaxEnt is a species distribution modeling method that is designed to deal with presence-only species occurrence data, useful 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 listed below and in Figure 6-9:

1. Process species occurrence data (Figure 6-8). For use in MaxEnt, the species occurrence data (FIA inventory plot data) will be converted to a .csv table with x- and y- coordinates. 75% of the species occurrence points will be used for model development, with the remaining 25% to be reserved for model validation and occurrence threshold definition.

2. Process environmental data (Table 6-7), derive needed predictor layers (soil moisture, fire regime, soil particle size, soil order, soil chemistry, elevation, precipitation, temperature, grazed lands, vegetation, harvest/restoration plots) with GIS, and convert to ASCII format for use in MaxEnt

   Environmental variables that are unknowable in the future (such as beetle distribution, fire regime, pinyon mortality areas, or land treatments) will not be used for the distribution modeling. MaxEnt models of beetle kill and mortality can be run separately.

3. Run MaxEnt, specifying the logistic model output format.

4. Convert MaxEnt output from ASCII format to grid, convert logistic probability output to predicted potential habitat by reclassification based on a threshold corresponding to 95% of the occurrence points. This operation will answer management question 1: What is the most current distribution of this species? The FIA inventory plot locations contribute to answering management question 2: Where are areas known to have been surveyed and what areas have not known to have been surveyed (i.e., data gap locations)?

5. For predicting future distribution of pinyon pine, project the MaxEnt model into the future using outputs of climate and development modeling (Figure 6-9). Overlaying the resulting mapped layers of pinyon pine current potential habitat and probable future potential habitat resulting from likely climate change and human development will answer management question 4, 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)? To address vulnerability to climate change, the environmental response curves generated by the MaxEnt software will be used to parameterize the Climate Change Vulnerability Index (CCVI) and compared with an existing CCVI (if one already exists) based on best professional judgment. **Note**: We have recently acquired a modeled climate change map for pinyon pine from Tom Edwards USGS.

6. For management question 3 (Where are potential habitat restoration areas?) we will locate disturbed or degraded areas of pinyon-juniper woodland (e.g., extensive burns, beetle-killed areas) in areas with optimal environmental attributes (soil, climate, elevation, topography).

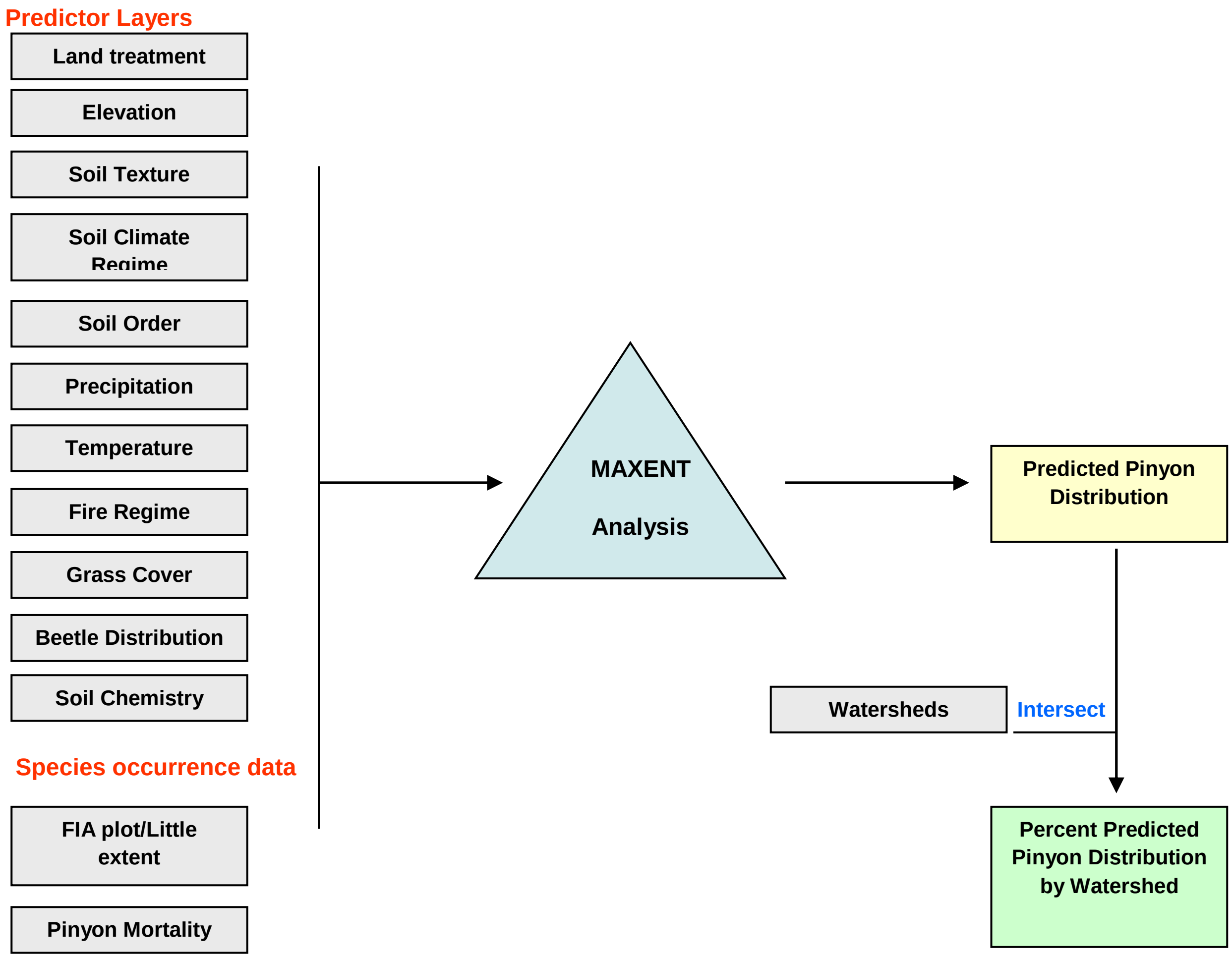


Figure 6-9. Example MAXENT process model used to determine predicted pinyon pine distribution.

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6.12 Plant Community Conservation Element: Cryptogamic Soil Crusts

The Dynamac Team will collaborate with USGS to develop a map of potential biological crust distribution across the Colorado Plateau (Matt Bowker, Section 6.12.3).

6.12.1 Introduction

The conservation element cryptogamic (or biological) soil crust was selected for the REA because of its important role in maintaining ecosystem function in the Colorado Plateau ecoregion. This community of algae, mosses, fungi, cyanobacteria, and lichens can comprise up to 70% of live ground cover in North American deserts (Belnap 1994). Their ability to photosynthesize only when wet and suspend respiration when dry, makes them well-adapted to the extreme climates of hot deserts and arctic tundra (Belnap et al. 2001).

The following management questions relating to cryptogamic soil crusts were identified by the Assessment Management Team (AMT) for inclusion in the Colorado Plateau Rapid Ecoregional Assessment (REA):

1. Where are soils that have potential to have cryptogamic soil crusts?
2. What/where is the potential for future change to the cryptogamic crusts?
3. Where are hotspots producing fugitive dust that may contribute to accelerated snow melt in the Colorado Plateau?

Change agents affecting soil crust and considered in this analysis include: wildland fire, human development, resource uses, invasive plant species, and climate change.

6.12.2 Conceptual Model

The conceptual model (Figure 6-10) shows the relationships between the natural drivers that both support and benefit from the cryptogamic soil crusts. While climate is an important driver, soils have the greatest impact on where cryptogamic soil crusts form. Favorable soil characteristics for the highest richness in soil crust species are stable, finely-textured soils with high levels of calcium carbonate on north or east facing surfaces, situated below 1000m elevation (Hansen et al. 1999). Soil crusts of filamentous lichen can be found on coarser, less stable soils on steeper surfaces and at slightly higher elevations.

Intact cryptogamic soil crusts exist in areas of low disturbance. Soil crust cover is essential to preserving soil stability in areas with sparse vegetation. Biological soil crusts serve a variety of ecosystem functions. They enhance soil properties such as structure, stability, moisture, and carbon/nitrogen fixation (DRECP 2010). Soil crusts slow the spread of invasive weeds and provide nutrients for microorganisms, vascular plants, and foraging animals. The dark color of the crusts reduces surface reflectance and increases soil temperature (Belnap 1994). Many plants and animals depend on this temperature control for the timing of their reproduction and foraging activities (Nash 1996, Doyen and Tschinkel 1974; Crawford 1991).

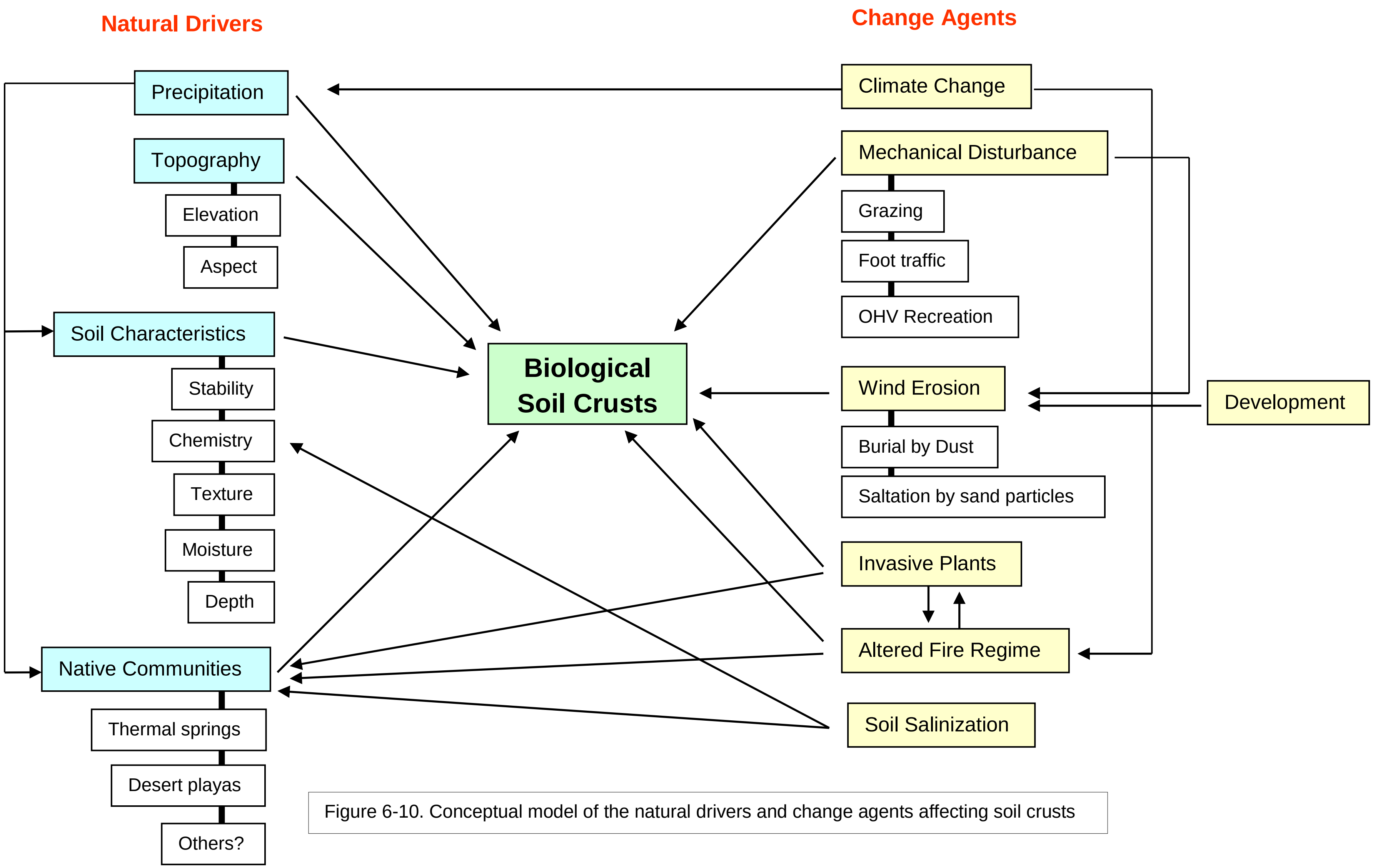


Figure 6-10. Conceptual model of the natural drivers and change agents affecting soil crusts

The erosion control properties of soil crusts reduce the spread of dust. Airborne dust reduces air quality and visibility. Dust in the desert creates negative feedbacks for plants, soil hydrology, and burrowing animals (DRECP 2010). Accumulated dust also has serious impacts on water supply; dust from desert sources of disturbance accumulate on mountain snow, reducing its reflectance and causing early snowmelt and subsequent water shortages (DRECP 2010). In 2009, a drought year with a dry late winter and subsequent low late-winter annual production, the mountains east of the Colorado Plateau saw a dramatic increase in soil exposure and dust production. Recent research indicates that in 2009 snowmelt was earlier than average by 45–50 days (J. Deems, University of Colorado, presentation at Workshop 3), and that dust forcing actually overwhelms the estimated impact of early snowmelt from predicted temperature increases from climate change (5–15 days earlier than average).

The change agents on the right side of the conceptual model are predominantly external disturbances that are inter-related. Soil crust populations are degraded when mechanical disturbances such as vehicular traffic, trampling by concentrations of grazing mammals, or land clearing for energy development disturb the soil surface. While any of these disturbances may not directly eliminate soil crusts, repeated disturbance fragments and degrades crust cover and reduces its stabilizing function. Dust and sand stirred up by traffic and construction bury soil crusts or abrade their surfaces. Land surface disturbances also create seedbeds for invasive plants. Invasive plants compete for available soil moisture and create a dense ground cover that is inhospitable for crusts; invasive annual species change the chemistry of the soil and provide fuel for more frequent and intense fires. Soil crusts may survive some fires and provide surface stability during post-fire recovery. However, the greater frequency, intensity, and extent of fires driven by the increased litter of invasive annual plants degrades soil crust and exposes it to replacement by invasive annuals.

6.12.3 USGS Proposal - Mapping Potential Biological Crust Abundance on the Colorado Plateau (submitted by Matt Bowker)

6.12.3.1 Introduction

The proposed work will produce a data layer indicating the *potential* quantitative cover of biological crusts, and major constituents (mosses, lichens, dark cyanobacterial crusts) across the entire Colorado Plateau. The product is intended to assist BLM and its contractor, Dynamac Inc., in treating biological crusts as a conservation element in the Colorado Plateau Rapid Ecoregional Assessment. It is also highly relevant to the soil stability conservation element.

At the scale of the entire Colorado Plateau, we will provide a spatially explicit estimate of the crust abundance that would likely exist if the site were in a “least-disturbed” state. Least-disturbed indicates an ecosystem state existing under current or recent climate conditions, that has been as minimally affected by disturbance as possible, given the context of widespread current and historical grazing. This state may or may not be equivalent to a historical reference condition; there is simply no information to know. Examples of least-disturbed sites include: 1) sites in National Parks where grazing has been excluded for some time, 2) never-grazed relicts, 3) range exclosures, 4) Sites within grazed landscapes that are distant from water and/or high quality forage, or are geographically isolated.

This work will be useful for regional scale analyses but may or may not provide a reliable basis for determining the status of a particular location (e.g. a hectare plot). Due to time and budgetary constraints, we are forced to partially rely on relatively low resolution model inputs (e.g. PRISM climate data). This may compromise the accuracy of model predictions at finer spatial scales.

This work will estimate and map the potential crust abundance, but it will not map the current, existing crust abundance. Remote sensing is the only practical way to conduct the latter at such a large scale.

The results of this project will be useful for regional scale analyses but it will not provide a reliable basis for determining the status of a particular location (e.g. a hectare plot). Due to time and budgetary constraints, we are forced to rely on relatively low resolution model inputs. This compromises the accuracy of model predictions at finer spatial scales.

6.12.3.2 Materials & methods

Using existing data, we will prepare classification and regression tree models which estimate potential abundance of biological crusts on the Colorado Plateau landscape. Model outputs will be mapped in raster grids, and delivered to BLM as a GIS.

Model inputs:

Annual precipitation and seasonality: The PRISM model will provide information at a 800m grid cell size regarding annual average precipitation. We will also use the monthly averages to compute the proportion of the total that falls from July – September, an index of the relative import of the summer monsoon.

Annual maximum and minimum temperature: The PRISM model will provide information at a 4km grid cell size regarding annual average maximum and minimum temperature, the July maximum, and the January minimum.

Soils data: Dynamac has overseen the production of an ecoregional soil survey map based upon NRCS SSURGO data. This entailed joining numerous individual surveys into a single shapefile and database. Gaps in coverage were filled using lower resolution STATSGO soil data. We have extracted and mapped 6 soil property indicators from this database: CaCO3, gypsum, sodium adsorption ratio, % sand, % clay, and the plasticity index. These predictors will be rasterized for use in spatial models.

Crust Abundance and Soil Stability Data: An integrated dataset of samples from around the Colorado Plateau, and its Northern, Southern and Eastern ecotones has been assembled (Table 6-10). All sites are in least-disturbed condition. Seven data sources were used. Sites from datasets by Bowker et al. 2006 were carefully selected based upon known or inferred disturbance history. Other data sources are from currently ungrazed areas in NPS units. In addition to being ungrazed we screened out sites which may be in a persistent annualized state (>5% exotic annuals) and interviewed data collectors about the reasonability of including these sites. All told, there are 682 total records. 593 contain data on total crust cover, and 502 contain data on soil stability (aggregate stability in water). In addition 259 contain primary soil data collected in association with the crust surveys; these data include soil texture, $CaCO_3$ and gypsum content.

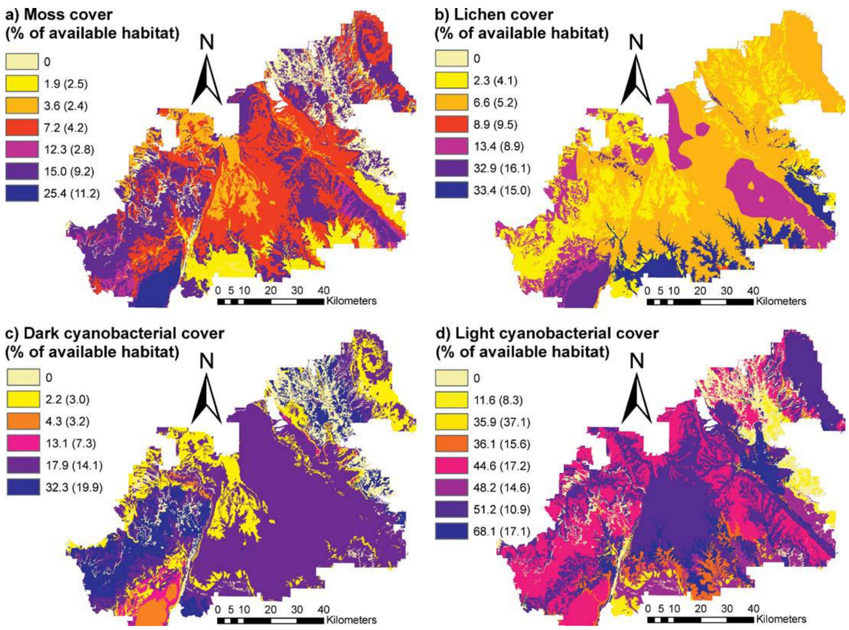
Modeling and validation approach

All predictor rasters will be snapped to a common grid to avoid topological problems. We will overlay the coordinates of the above sites on our predictor grids, and obtain the corresponding grid values. We will use primary soil data when available and fill in missing data from the grids. Using these input data, we will construct CART models of at least total crust cover, but will attempt them for specific groups of crust biota as well as chlorophyll *a* (a cyanobacterial biomass indicator). These models will be bootstrap validated, and their accuracy will determined by plotting model predicting and observed values using linear regressions. A "perfect" model would be expected to have an R^2 of 1.0, a slope of 1, and a y-intercept of 0. Model outputs will be mapped to raster grids using the raster calculator in the Spatial Analyst extension of ArcGIS 9.2 (Bowker et al. 2006). We conducted a similar modeling exercise for Grand Staircase-Escalante National Monument (Bowker et al. 2006), using a reclassification of soils (Figure 6-11).

Table 6-10. Summary of integrated dataset of quantitative biological crust data. Numbers refer to number of samples in each category. (M) denoted that soils stability values were modeled based on crust characteristics; otherwise they are measured on site.

Data source	Location	Soil data	Dk. cyano	Moss	Lichen	Total crust	Chlor. a	Soil stability
Bowker et al. 2006	Grand Staircase-Escalante NM & vicinity	114	114	114	114	114	113	113(**M)**
Bowker & Belnap 2007	Walnut Cyn NM & vicinity	11	11	11	11	11	11	11
	Wupatki NM	25	25	25	25	25	25	25
	Sunset Crater NM	4	4	4	4	4	4	4
	Verde Valley, Arizona	11	11	11	11	11	11	11
	Other N. Arizona	13	13	13	13	13	13	13
Bowker et al. 2005	Canyonlands & vicinity	38		38	38			
	Dinosaur NM	8		8	8			
	Natural Bridges NM	8		8	8			
	Glen Canyon NRA	23		23	23			
	Other (Hovenweep NM Arches NP)	4		4	4			
Coles et al. 2010	Arches NP		90	90	90	90		
Miller et al. unpub	Canyonlands NP					101		101
NPS I&M								
NCPN	Canyonalnds NP		62	62	62	62		62
	Capitol Reef NP (retired allot's)		21	21	21	21		21
	Black Canyon/Curecanti (retired allot's)		17	17	17	17		17
	Dinosaur NM (retired allot's)		16	16	16	16		16
SCPN	Chaco Cyn CP		16	16	16	16		16
	Mesa Verde NP		20	20	20	20		20
	Petrified Forest NP		62	62	62	62		62
	Grand Canyon NP		10	10	10	10		10
Totals		259	492	573	573	593	177	502

Figure 6-11. Output maps from a similar modeling exercise at the scale of Grand Staircase-Escalante National Monument (Bowker et al. 2006), using a reclassification of soils.



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

The Dynamac team provides the following review after examining both SWreGAP and Landfire. 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: Fuel models are applied based on expert interpretation of existing vegetation, cover, height, and site potential. They are 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 SWReGAP, possibly a hybrid of SWReGAP and Landfire EVT for use as vegetation/landcover data. We will use Landfire products to address fire-related management questions.

APPENDIX 2. Terrestrial Wildlife Conservation Element: Burrowing Owl (*Athene cunicularia*)

Note: Since burrowing owls can be found in agricultural areas and other disturbed areas where burrows may be available (e.g., culverts), our modeling approach will treat the burrowing owl as part of a prairie dog town assemblage, including other core and desired species selected for the Colorado Plateau such as the Gunnison's and white-tailed prairie dog, black-footed ferret, and ferruginous hawk.

The burrowing owl (*Athene curicularia*) is a small raptor that inhabits open prairie grasslands primarily in midwestern and western U.S. and Canada. Populations have been declining for several years, mostly due to habitat destruction and the loss of sciurid burrows in which they nest. This module describes methodology, as part of the BLM's Colorado Plateau Ecoregional Assessment, to assess the current distribution of potential burrowing owls, change agents affecting this distribution, and where the owls are at risk.

Introduction

The purpose of this assessment is to design a technical approach to address the status of burrowing owls (*Athene cunicularia*) in the Colorado Plateau ecoregion. The following species-related management questions were identified by the Assessment Management Team (AMT) for inclusion in the Colorado Plateau Rapid Ecoregional Assessment (REA):

- What is the most current distribution of available occupied habitat (and historic occupied habitat if available), seasonal and breeding habitat, and movement corridors (as applicable)?
- What areas are known to have been surveyed and what areas have not been surveyed (i.e., data gap locations)?
- Where are potential habitat restoration areas?
- Where are potential areas to restore connectivity?
- 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?

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

Conceptual Model

The burrowing owl inhabits open prairie grassland habitat and desert ecosystems primarily throughout the midwestern and western U.S. and Canada, although there are some populations in Florida, Mexico, Central America, and some Caribbean islands (Haug et al. 1993). Although populations in North America have been declining, the U.S. Fish and Wildlife Service has not formally listed the species. However, burrowing owls are included on an informal federal list as "Species of Concern," and listed as endangered or threatened in a number of other states (Sheffield 1997). Although most North American burrowing owl populations are migratory, little information exists on migration routes, times, or wintering areas. In addition, there is little information about juvenile dispersal from nest sites (Haug et al. 1993).

Western burrowing owls prefer open habitats, usually with short grasses and sparse shrubs, and they avoid areas near trees or with tall, thick vegetation (Plumpton & Lutz 1993). However, the owls are quite adaptable and will forage in agricultural fields and grazed pastures within disturbed habitats. Although burrowing owls prefer to nest in prairie dog or ground squirrel burrows, they also will nest in nest boxes, irrigation pipes, and culverts if no natural sites are available. Availability of nest sites appears to be a major population limiting factor, especially in disturbed or more developed habitats.

The major reason for the decline of burrowing owls is the destruction and alteration of their habitat, and the subsequent loss of suitable nest sites (Sheffield 1997). Although the birds prefer open grassland habitat, little is known about finer-scale preferences such as how habitat fragmentation affects distribution and/or reproductive success. A study in the Northern Great Plains suggested that fragmentation is not important as long as nest sites are plentiful (Restani et al. 2008). Nothing is known of dispersal or migration patterns and consequently connectivity issues are not easily addressed. Food availability also appears to be a less important factor as the owls are opportunistic feeders and are able to forage in a variety of disturbed habitats (e.g. pastures, golf courses). However, in drought years, food may be an important limiting factor (Rosenberg et al. 2009).

The conceptual model (Figure 1) illustrates the relationships among natural population drivers, change agents, and burrowing owl populations. Changes caused by development and resource use, climate change, and altered fire regime affect burrowing owl habitat. Although the owls are relatively adaptable, alteration of grassland habitat through development or conversion to agricultural land has a negative effect on burrowing owl populations due to its effect on nest availability, an increase in predator populations (wild and domestic), increased exposure to toxic contaminants, and food availability. Farmers and ranchers often destroy prairie dogs and ground squirrels leading to a scarcity of suitable nesting burrows in many areas, and thus the burrowing owls' decline parallels that of prairie dog colonies. Although most of the identified change agents are predicted to have negative effects on burrowing owl populations, occasionally habitat changes have the opposite effect. For example, development or agricultural conversion of grassland habitat is generally negative, but in some cases human activities such as mowing and wetland drainage have increased the species' range (Haug et al. 1993).
Contaminants and human-caused owl deaths affect owl individuals and populations directly. The rodenticides used to destroy ground squirrels and prairie dogs negatively affect owls by causing decreased body mass, decreased breeding success, and death (James et al. 1990). Agricultural anti-cholinesterase insecticides cause severe reproductive effects in some owl populations (Fox et al. 1989). Habitat alteration has also led to an increase in owl predators such as coyotes and foxes (White 1994).

It is not known how climate change will affect this species. However, potential consequences are changes in fire regime and invasive plants which could alter food availability either by directly affecting prey populations, or altering foraging habitat. In addition, if climate change affects sciurids, it may change availability of nest burrows.

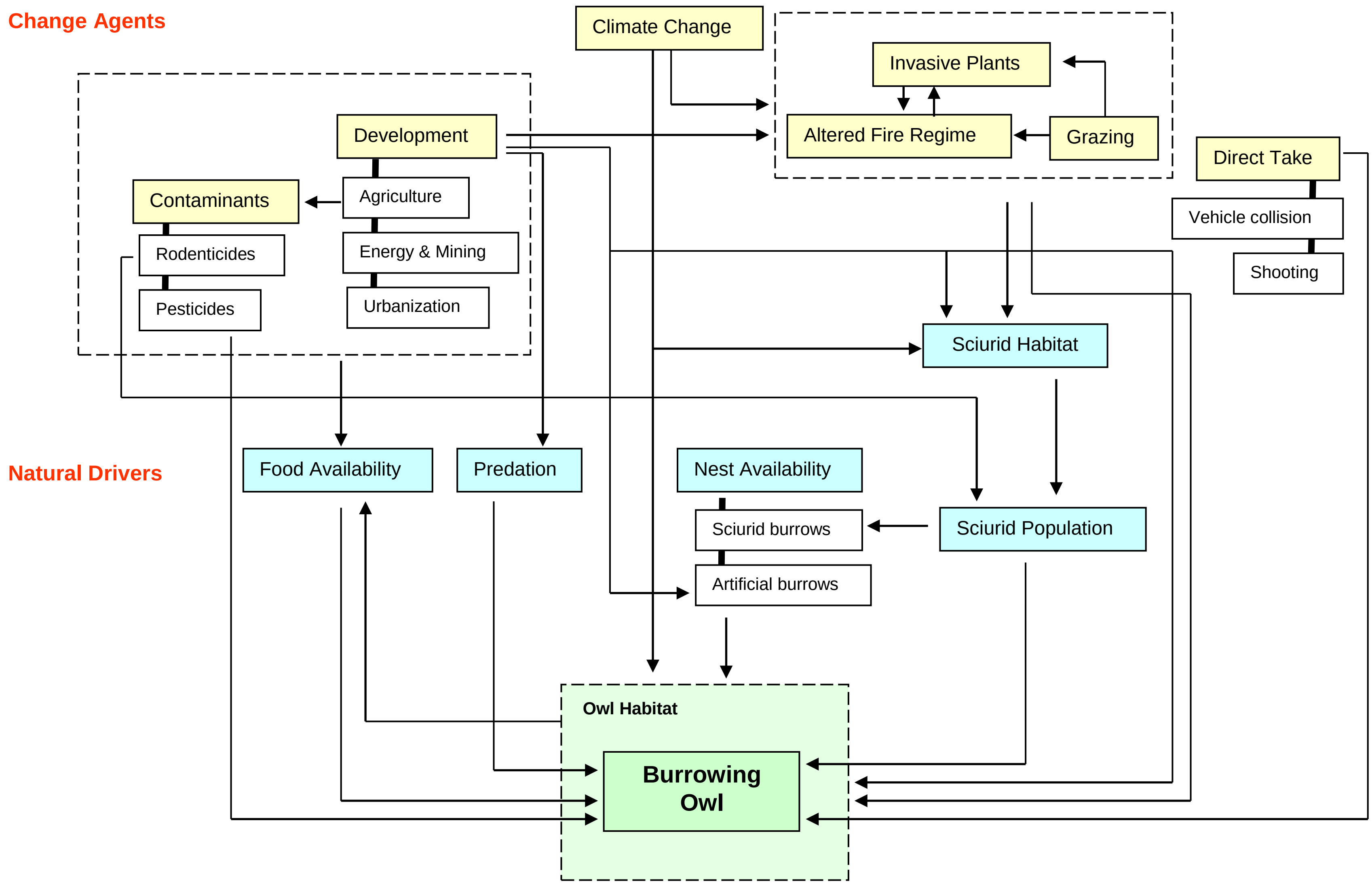

Figure 1. Diagram of principle interactions among population drivers and change agents for burrowing owls in the Colorado Plateau.

Required Data Layers

To answer the management questions, we need to know where burrowing owls are located now, and where they are likely to be located in the future. Because of their reliance on sciurid burrows for nesting, their distribution will be tied closely to that of prairie dogs and ground squirrels. In addition, they prefer open areas with short grassland vegetation for foraging, so it is important to locate the distribution of this vegetation type in relation to nest burrows and to determine where this vegetation will be located in the future to predict future species distribution. Known data gaps include migration routes and dispersal patterns, so questions about connectivity needs are not easily answered.

Table 1 indicates the raw data layers necessary for answering the conservation management questions for burrowing owls. Data layers with asterisks indicate data that may not be readily available and for which surrogates may need to be developed. **Table 2** lists previously processed data for model inclusion.

TENTATIVE DATA NEEDS	DATA CLASS
NHP EO’s	SPECIES OCCURRENCES
Current climate (PRISM, DAYMET)	CLIMATE – RECENT
Future climate (2060 downscaled model)	CLIMATE – FUTURE
Drought	CLIMATE – RECENT
Human footprint (Development)	DEVELOPMENT
Road Density	DEVELOPMENT
Land use planning areas	DEVELOPMENT – FUTURE
Population growth projections	DEVELOPMENT – FUTURE
Identified movement corridors	HABITAT
Identified seasonal habitats	HABITAT
SWreGAP vegetation	VEGETATION
Grazing pressure	RESOURCE USE
STATSGO	SOILS
*Toxic contaminant levels	POLLUTANTS
*Direct mortality records	MORTALITY

Table 2. Processed data sets required for inclusion in the model.

PREVIOUSLY PROCESSED DATA	LOCATION
Gunnison’s prairie dog distribution	SWreGAP
White-tailed prairie dog distribution	SWreGAP
Black-tailed prairie dog distribution	SWreGAP

General Approach

Previous habitat suitability studies have indicated that nest availability and foraging habitat are the most important factors in predicting burrowing owl occurrences (Uhmann et al., 2001). Although the effects of human disturbance are not well understood, they can lead to direct mortality and loss of nesting and foraging habitat, so these areas need to be eliminated as potential habitat. Little is known about migratory patterns and habitats so these are not included in the model. In addition, the model does not include variables for toxic contaminants which may be important but for which datasets are not readily available.

Future burrowing owl distribution will depend on how development and climate change affect the distribution of prairie habitat and sciurid populations. The same model could be used to investigate these changes if modeled future foraging habitat and sciurid distributions replaced the current datasets and predicted human footprint replaced the current one.

Approach Assumptions

This model uses sciurid distribution as a surrogate for burrow locations and therefore may not be entirely accurate. Due to the rapid nature of the assessment, existing sciurid distribution maps will be used rather than developing more fine-scale versions. Although some human disturbance may actually benefit burrowing owls (e.g. mowing, golf courses), we are assuming that development is mostly negative for the purposes of this course scale analysis.

Methods and Tools

The model will be developed in ArcGIS utilizing existing GIS datasets (Figure 2). We will use a habitat distribution software package such as Maxent to develop a model of current burrowing owl distribution within the Colorado Plateau. The two major natural drivers of this population are availability of foraging habitat and nest burrows, so data sets related to these drivers will be the inputs for the model. Foraging habitat is generally mixed and short grass prairies (Haug et al., 1993) so these vegetation types will be selected from the SWreGAP vegetation dataset. Burrowing owls generally do not construct their own burrows, but nest in unused sciurid burrows. Because a dataset of burrows does not exist, we will use sciurid distribution maps as a surrogate for this information. In addition, burrows in loamy soil tend to be more suitable than those in sandy soil, so we will select appropriate soil types from the STATSGO data set (Desmond et al., 2000). Finally, owls are often negatively affected by human disturbance including roads, so these areas will be eliminated from consideration for good potential owl habitat.

This model can be used to estimate future burrowing owl distribution in the following manner. To account for changes due to climate change, modeled prairie and sciurid locations would replace the current datasets in the model. To examine potential changes due to development, a modeled future human footprint dataset would replace the current one to indicate where populations may be at risk.

Prior to modeling, all datasets clipped to ecoregion extent

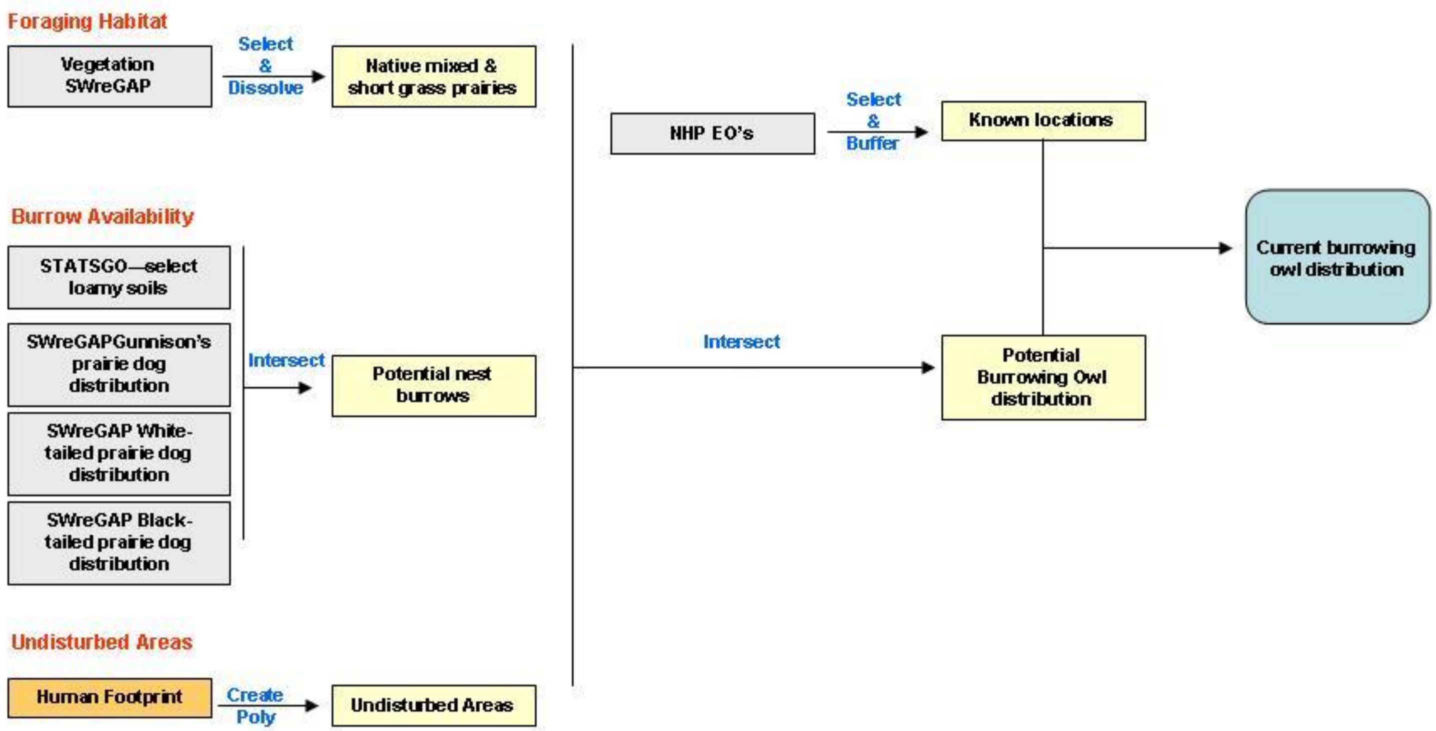

Figure 2. Datasets and processing steps for development of a burrowing owl distribution model.

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Appendix 3. Terrestrial Wildlife Conservation Element: Pronghorn (*Antilocapra americana*)

Introduction

The habitat suitability model focuses on landscape level variables that relate to long-term persistence of populations. Suitability values reflect the probability that a given area will be occupied by Pronghorn. Scores of 0.0 denote unsuitable habitat. Increasing probabilities of occurrence correspond to higher values (maximum = 1.0).

General Approach

The habitat suitability model is patterned after earlier models (Boykin et al. 2009; Allen et al. 1984). The recommended model is comprised of two components: (1) a Landscape component predicts potential habitat suitability based on proximity to free water sources; and (2) a Habitat component that incorporates vegetation structure and landform slope. These components are combined to provide an overall index of habitat suitability (Figure 1).

The Habitat component identifies potential habitat as being comprised of a wide variety of vegetation types, including shrubland, grassland, savannas, and cropland. The modeling efforts of Boykin et al. 2009 as well as the Southwest ReGap analysis identify the following list of vegetation types used by pronghorns:

S054 Inter-Mountain Basins Big Sagebrush Shrubland
S055 Great Basin Xeric Mixed Sagebrush Shrubland
S071 Inter-Mountain Basins Montane Sagebrush Steppe
S074 Southern Rocky Mountain Juniper Woodland and Savanna
S075 Inter-Mountain Basins Juniper Savanna
S079 Inter-Mountain Basins Semi-Desert Shrub Steppe
S085 Southern Rocky Mountain Montane-Subalpine Grassland
S090 Inter-Mountain Basins Semi-Desert Grassland
S096 Inter-Mountain Basins Greasewood Flat
N80 Agriculture
D02 Recently Burned

These vegetation types inherently differ with regards to vegetation structure, especially height and cover. Pronghorn prefer short vegetation, generally shorter that 18 inches (Allen et al. 1984), and they avoid vegetation taller than 24 inches and rarely use vegetation taller than 30 inches (North Dakota Game and Fish Department 2006). Moderate vegetation cover seems to be preferred and optimum conditions are 15-30% shrub cover, and 10-40% herbaceous cover (Allen et al. 1984). These guidelines can be used to initially assign a suitability index to each of the potentially suitable vegetation types based on their expected condition, or the model (Figure 1) can be modified by explicitly including simple suitability functions on vegetation height and cover.

Pronghorns typically use sites having slopes of less than 10%, but slopes greater than 20% are generally avoided (Allen et al. 1984, North Dakota Game and Fish 2006). The model contemplated in Figure 1 uses a simple suitability threshold: at or below 20% slope is suitable; > 20% slope is unsuitable.

Moderately rolling terrain may be beneficial, however. Ridges, rims, and depressions are used as thermal and escape cover and may contribute to greater diversity in food resources (Allen et al. 1984). Hence, further review may suggest that it would be desirable to include a more complex algorithm that recognizes that terrain that averages about 9–25 % would be a better representation of habitat (Allen et al. 1984). This could be accomplished by using a moving window analysis in which the proportion of pixels with a slope between 9-25% is quantified within a 3-5 km radius surrounding each pixel. Each focal pixel would be assigned a score ranging from 0 to 1.0 depending on the proportion of the surrounding pixels in the window that had the appropriate slope.

Water obtained from forage, especially cacti and succulent forage plants may be sufficient to meet the water requirements for adult survival, but may not meet lactation needs (North Dakota Game and Fish 2006). The optimum spacing of water sources is not consistently agreed upon however. Some studies cited by North Dakota Game and Fish 2006 indicate that herds occupying rangelands with drinking water every 1-3 miles had higher densities compared to areas with scant drinking water. Other studies suggest that the spacing of water sources every 4 miles would be considered to be sufficient. At the sparse end of the scale, pronghorns have been observed more than 40 miles from water, although it is not clear that populations could persist for any length of time under those conditions. We will initially use a simple suitability function with three levels of suitability: 1) < 1 mile is optimum (i.e., index of 1.0); 2) 1 mile < distance <= 10 miles is moderately suitable (i.e., 0.5 index); and 3) distance > 10 miles is unsuitable.

Approach Assumptions

1) We assume that LANDFIRE data or Southwest ReGap vegetation data will be used for application of this habitat model. There are at least two reasons why such an approach might be beneficial for purposes of continuing to model the relationship between change agents and wildlife habitat. First, the LANDFIRE vegetation succession and disturbance histories are being modeled for each of the ecological system classes. Second, LANDFIRE provides a consistent mapping protocol across broad scales so it provides opportunities for consistency of analyses across ecological and jurisdictional boundaries. This will facilitate analyses at multiple scales.
2) There is good information in the ecological literature concerning pronghorn preferences for certain conditions of sagebrush height, % canopy cover, and other habitat characteristics. The model outlined herein (Figure 1) assumes that the various vegetation classes that would be potentially used by pronghorn can be assigned a basic suitability value based on their expected vegetation structure (e.g., vegetation height and vegetation cover) and floristics. Consequently, this will require that change agents and management actions be described in terms of associated changes in vegetation class, rather than how such actions would affect vegetation structure and floristics, per se. If this approach does not provide the necessary resolution, the model can be modified to directly incorporate additional variables related to vegetation structure.

Required Input Data Layers

1. Vegetation Types
2. (Optional) Vegetation height, vegetation cover.
3. Landform slope
4. Surface water features (e.g., lakes, streams, springs)

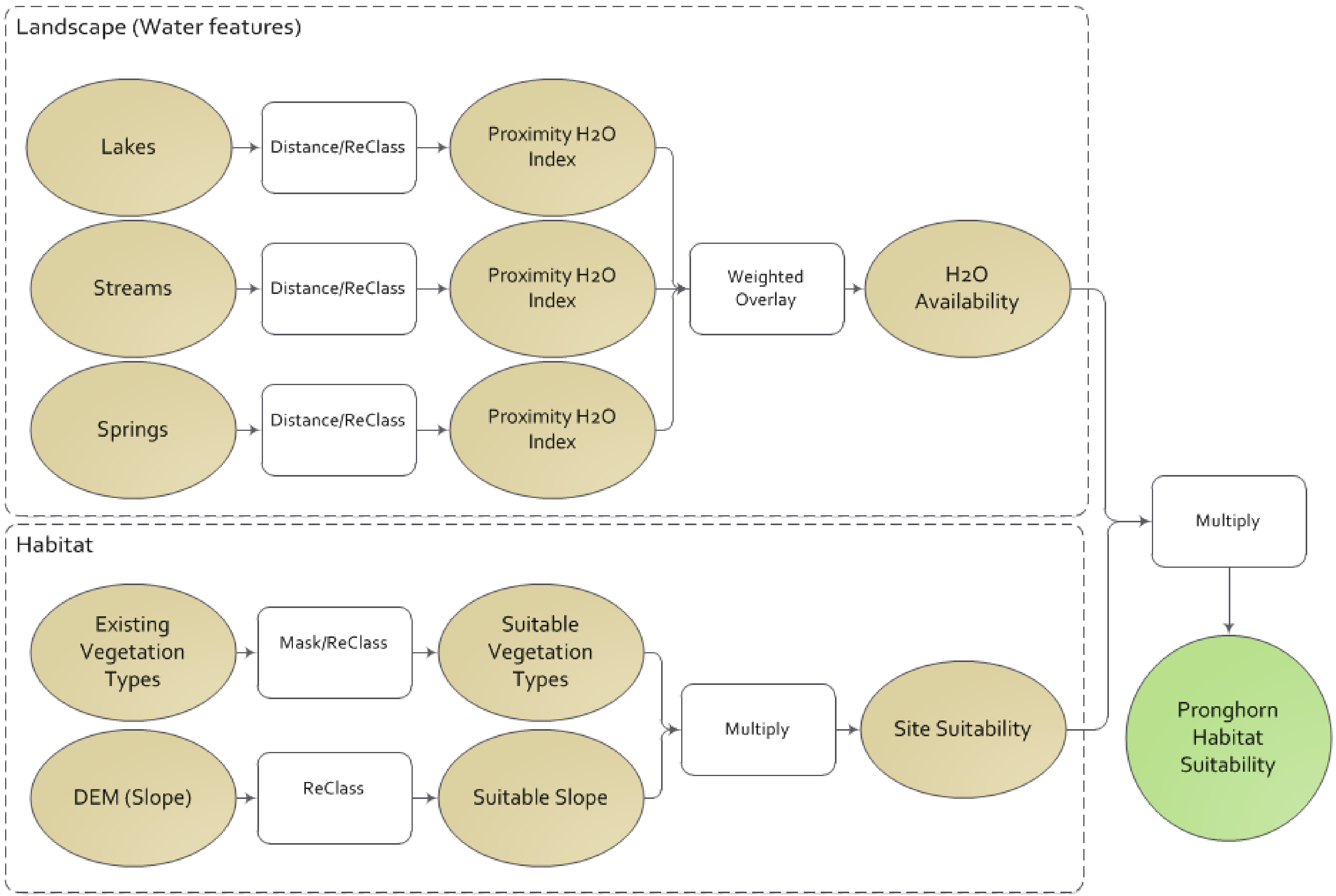

Figure 1. Conceptual habitat suitability model for the Pronghorn.

Methods and Tools

1) Habitat (Vegetation and slope)
   a) Existing Vegetation Types (*Mask/Reclassify*).
      - Set data cells located outside boundary of Vegetation Type grid to NoData
      - Set identified vegetation types to **value = (relative suitability based on height and cover)**; other types = NoData
   b) Slope (*Reclassify*).
      - Set cells with slope <= 20% to **value = 1.0**
      - Set cells with slope >20% to **value = 0.0**
   c) (Optional) Terrain/Slope (*Focal Statistics/Reclassify*).
      - Set cells with 9% <= slope <= 20% to **value = 1.0;** set other cells to 0.0.
      - Moving circular window (3-5 km in radius); set focal cell to **value = sum**.
      - Rescale cell values to values in range {0.0-1.0}.
   d) *Multiply* values from a and b (or c) above to produce habitat suitability.

2) Landscape (Water Features)
   a) Disturbance Types (*Distance/Reclassify*).
      - Calculate distance from focal cell to nearest cell with defined water category. Repeat for all defined water categories (Three categories shown in Figure 1).
      - Set cell **value = Proximity Index** based on distance.
   b) Proximity H_20 Index (*Weighted Overlay/Reclassify*).
      - Overlay and **Sum** index from all water categories.
      - Set cell **value = minimum{1.0, Sum}.** Constrain value to {0, 1}.
3) Habitat Suitability
   a) Site Suitability; H_20 Availability (*Multiply*).
      - Calculate Habitat Suitability on a cell-by-cell basis by multiplying Site Suitability and H_20 Availability:

HABITAT SUITABILITY = **Site Suitability * $H_2$0 Availability**

APPENDIX 4. Coarse Filter Ecological System Conservation Elements for the Colorado Plateau.

FOREST & WOODLAND CLASSES (31.2%)

Percent of Ecoregion	Code	Ecological System
3.13%	S023	Rocky Mountain Aspen Forest and Woodland
0.01%	S024	Rocky Mountain Bigtooth Maple Ravine Woodland
0.00%	S025	Rocky Mountain Subalpine-Montane Limber-Bristlecone Pine Woodland
1.50%	S028	Rocky Mountain Subalpine Dry-Mesic Spruce-Fir Forest and Woodland
0.66%	S030	Rocky Mountain Subalpine Mesic Spruce-Fir Forest and Woodland
0.47%	S031	Rocky Mountain Lodgepole Pine Forest
0.85%	S032	Rocky Mountain Montane Dry-Mesic Mixed Conifer Forest and Woodland
0.61%	S034	Rocky Mountain Montane Mesic Mixed Conifer Forest and Woodland
2.55%	S036	Rocky Mountain Ponderosa Pine Woodland
0.01%	S038	Southern Rocky Mountain Pinyon-Juniper Woodland
20.39%	S039	Colorado Plateau Pinyon-Juniper Woodland
0.35%	S040	Great Basin Pinyon-Juniper Woodland
0.67%	S042	Inter-Mountain West Aspen-Mixed Conifer Forest and Woodland Complex

APPENDIX 4 (Continued). Coarse Filter Ecological System Conservation Elements for the Colorado Plateau.

SHRUB / SCRUB CLASSES (37.3%)

Percent of Ecoregion	Code	Ecological System
0.04%	S043	Rocky Mountain Alpine Dwarf-Shrubland
2.03%	S045	Inter-Mountain Basins Mat Saltbush Shrubland
4.49%	S046	Rocky Mountain Gambel Oak-Mixed Montane Shrubland
0.66%	S047	Rocky Mountain Lower Montane-Foothill Shrubland
0.02%	S050	Inter-Mountain Basins Mountain Mahogany Woodland and Shrubland
6.34%	S052	Colorado Plateau Pinyon-Juniper Shrubland
9.14%	S054	Inter-Mountain Basins Big Sagebrush Shrubland
0.00%	S055	Great Basin Xeric Mixed Sagebrush Shrubland
0.68%	S056	Colorado Plateau Mixed Low Sagebrush Shrubland
0.19%	S057	Mogollon Chaparral
6.32%	S059	Colorado Plateau Blackbrush-Mormon-tea Shrubland
0.13%	S060	Mojave Mid-Elevation Mixed Desert Scrub
5.37%	S065	Inter-Mountain Basins Mixed Salt Desert Scrub
0.23%	S069	Sonora-Mojave Creosotebush-White Bursage Desert Scrub
0.00%	S070	Sonora-Mojave Mixed Salt Desert Scrub
0.01%	S128	Wyoming Basins Low Sagebrush Shrubland
1.06%	S136	Southern Colorado Plateau Sand Shrubland

APPENDIX 4 (Continued). Coarse Filter Ecological System Conservation Elements for the Colorado Plateau.

GRASSLANDS (9.1%)

Percent of Ecoregion	Code	Ecological System
0.15%	S081	Rocky Mountain Dry Tundra
0.35%	S083	Rocky Mountain Subalpine Mesic Meadow
0.26%	S085	Southern Rocky Mountain Montane-Subalpine Grassland
1.71%	S090	Inter-Mountain Basins Semi-Desert Grassland
3.91%	S071	Inter-Mountain Basins Montane Sagebrush Steppe
0.13%	S075	Inter-Mountain Basins Juniper Savanna
0.00%	S078	Inter-Mountain Basins Big Sagebrush Steppe
2.57%	S079	Inter-Mountain Basins Semi-Desert Shrub Steppe

WOODY WETLAND & RIPARIAN CLASSES (2.4%)

Percent of Ecoregion	Code	Ecological System
0.00%	S014	Inter-Mountain Basins Wash
0.00%	S020	North American Warm Desert Wash
0.11%	S091	Rocky Mountain Subalpine-Montane Riparian Shrubland
0.00%	S092	Rocky Mountain Subalpine-Montane Riparian Woodland
0.49%	S093	Rocky Mountain Lower Montane Riparian Woodland and Shrubland
0.00%	S094	North American Warm Desert Lower Montane Riparian Woodland and Shrubland
1.79%	S096	Inter-Mountain Basins Greasewood Flat
0.01%	S097	North American Warm Desert Riparian Woodland and Shrubland
0.00%	S098	North American Warm Desert Riparian Mesquite Bosque
0.00%	S118	Great Basin Foothill and Lower Montane Riparian Woodland and Shrubland

EMERGENT HERBACEOUS WETLAND CLASSES (0.2%)

Percent of Ecoregion	Code	Ecological System
0.01%	S100	North American Arid West Emergent Marsh
0.20%	S102	Rocky Mountain Alpine-Montane Wet Meadow

APPENDIX 4 (Continued). Coarse Filter Ecological System Conservation Elements for the Colorado Plateau.

SPARSELY VEGETATED / BARREN CLASSES (13.8%)

Percent of Ecoregion	Code	Ecological System
0.00%	S001	North American Alpine Ice Field
0.35%	S002	Rocky Mountain Alpine Bedrock and Scree
0.09%	S004	Rocky Mountain Alpine Fell-Field
0.61%	S006	Rocky Mountain Cliff and Canyon
0.00%	S009	Inter-Mountain Basins Cliff and Canyon
10.55%	S010	Colorado Plateau Mixed Bedrock Canyon and Tableland
1.17%	S011	Inter-Mountain Basins Shale Badland
0.86%	S012	Inter-Mountain Basins Active and Stabilized Dune
0.08%	S013	Inter-Mountain Basins Volcanic Rock and Cinder Land
0.02%	S016	North American Warm Desert Bedrock Cliff and Outcrop
0.01%	S019	North American Warm Desert Volcanic Rockland
0.05%	N31	Barren Lands, Non-specific
0.00%	S015	Inter-Mountain Basins Playa
0.00%	S022	North American Warm Desert Playa

OPEN WATER (0.7%)

Percent of ecoregion	Code	Ecological System
0.71%	N11	Open Water

CRYPTOGAMIC CRUST

Cryptogamic crust	NA	Ecological System

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 5. Plant Species Representative of Major Ecological Systems.

ECOLOGICAL SYSTEM	% OF ECOREGION	REPRESENTATIVE SPECIES	SCIENTIFIC NAME
Colorado Plateau Pinyon-Juniper Woodland	20.4%	Pinyon Pine	*Pinus edulis*
Inter-Mountain Basins Big Sagebrush Shrubland	9.1%	Wyoming Big Sagebrush	*Artemisia tridentate wyomingensis*
Inter-Mountain Basins Montane Sagebrush Steppe	3.9%	Mountain Sagebrush	*Artemisia tridentata ssp. vaseyana*
Colorado Plateau Mixed Bedrock Canyon and Tableland	10.6%	Littleleaf Mountain Mahogany	*Cercocarpus intricatus*
Rocky Mountain Gambel Oak-Mixed Montane Shrubland	4.5%	Gambel Oak	*Quercus gambelii*
Colorado Plateau Pinyon-Juniper Shrubland	6.3%	Utah Juniper	*Juniperus osteosperma*
Colorado Plateau Blackbrush-Mormon-Tea Shrubland	6.3%	Blackbrush	*Coleogyne ramosissima*
Inter-Mountain Basins Mixed Salt Desert Scrub	5.4%	Shadscale	*Atriplex confertifolia*
TOTAL AREA	66.5%		

Appendix 6. Candidate Landscape Species and Coppolillo Screening Scores for the Colorado Plateau.

SPECIES	SCIENTIFIC NAME	AREA	HETEROGENEITY	VULNERABILITY	FUNCTIONALITY	SOC. ECON. SIGNIF	SCORE
Mountain lion	*Puma concolor*	1.00	0.77	0.25	0.50	1.00	3.52
American peregrine falcon	*Falco peregrinus*	1.00	0.57	0.75	0.50	0.40	3.22
Big free-tailed bat	*Nyctinomops macrotis*	1.00	0.69	0.75	0.00	0.40	2.84
Golden eagle	*Aquila chrysaetos*	1.00	0.45	0.25	0.50	0.60	2.80
Bighorn sheep	*Ovis canadensis*	0.75	0.42	0.50	0.50	0.60	2.77
Gunnison sage-grouse	*Centrocercus minimus*	1.00	0.09	1.00	0.00	0.60	2.69
Bobcat	*Lynx rufus*	1.00	0.55	0.00	0.50	0.60	2.65
Kit fox	*Vulpes macrotis*	0.50	0.36	0.50	1.00	0.20	2.56
Burrowing owl	*Athene cunicularia*	0.25	0.34	0.50	1.00	0.40	2.49
Gunnison's prairie dog	*Cynomys gunnisoni*	0.00	0.19	0.50	1.00	0.60	2.29
White-tailed prairie dog	*Cynomys leucurus*	0.00	0.12	0.50	1.00	0.60	2.22
Black-footed ferret	*Mustela nigripes*	0.00	0.12	1.00	0.50	0.60	2.22
Greater sage-grouse	*Centrocercus urophasianus*	1.00	0.09	0.50	0.00	0.60	2.19
Mule deer	*Odocoileus hemionus*	0.25	1.00	0.00	0.50	0.40	2.15
Pinyon jay	*Gymnorhinus cyanocephalus*	0.25	0.22	0.25	1.00	0.40	2.12
Mexican spotted owl	*Strix occidentalis lucida*	0.25	0.11	0.75	0.50	0.40	2.01
Pronghorn	*Antilocapra americana*	1.00	0.16	0.25	0.00	0.40	1.81
Southwestern willow flycatcher	*Empidonax traillii extimus*	0.00	0.12	1.00	0.00	0.60	1.72
Razorback sucker	*Xyrauchen texanus*	0.00	0.00	1.00	0.00	0.60	1.60
Canyon treefrog	*Hyla arenicolor*	0.00	0.00	0.50	0.50	0.20	1.20
Arizona toad	*Bufo microscaphus*	0.00	0.04	0.75	0.00	0.40	1.19
White-tailed jackrabbit	*Lepus townsendii*	0.00	0.17	0.25	0.50	0.20	1.12
Sage sparrow	*Amphispiza belli*	0.00	0.41	0.50	0.00	0.20	1.11
Olive-sided flycatcher	*Contopus cooperi*	0.00	0.22	0.50	0.00	0.20	0.92
Colorado River cutthroat	*Oncorhynchus clarkii pleuriticus*	0.00	0.00	0.50	0.00	0.40	0.90
Northern leopard frog	*Rana pipiens*	0.00	0.10	0.50	0.00	0.20	0.80
Black-throated sparrow	*Amphispiza bilineata*	0.00	0.23	0.25	0.00	0.20	0.68
Yellow-breasted chat	*Icteria virens*	0.00	0.12	0.25	0.00	0.20	0.57
Sage thrasher	*Oreoscoptes montanus*	0.00	0.36	0.00	0.00	0.20	0.56
Juniper titmouse	*Baeolophus ridgwayi*	0.00	0.17	0.00	0.00	0.20	0.37

Appendix 7. Final Selection of Landscape Species for the Colorado Plateau 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	0.77	0.25	0.50	1.00	3.52
American peregrine falcon	1.00	0.57	0.75	0.50	0.40	3.22
Big free-tailed bat	1.00	0.69	0.75	0.00	0.40	2.84
Desert Bighorn sheep	0.75	0.42	0.50	0.50	0.60	2.77
~~Bobcat~~	1.00	0.55	0.00	0.50	0.60	2.65
~~Kit fox~~	0.50	0.36	0.50	1.00	0.20	2.56
Burrowing owl	0.25	0.34	0.50	1.00	0.40	2.49
Yellow-breasted chat	0.00	0.12	0.25	0.00	0.20	0.57
Razorback sucker	0.00	0.00	1.00	0.00	0.60	1.60
Colorado River cutthroat	0.00	0.00	0.50	0.00	0.40	0.90

Appendix 8. Desired Species Conservation Elements for the Colorado Plateau Ecoregion.

SPECIES	AREA	HETEROGENEITY	VULNERABILITY	FUNCTIONALITY	SOCIO-ECONOMIC SIGNIFICANCE	SPECIES SCORE
Golden eagle	1.00	0.45	0.25	0.50	0.60	2.80
Gunnison sage-grouse	1.00	0.09	1.00	0.00	0.60	2.69
Gunnison's prairie dog	0.00	0.19	0.50	1.00	0.60	2.29
White-tailed prairie dog	0.00	0.12	0.50	1.00	0.60	2.22
Black-footed ferret	0.00	0.12	1.00	0.50	0.60	2.22
Greater sage-grouse	1.00	0.09	0.50	0.00	0.60	2.19
Mule deer	0.25	1.00	0.00	0.50	0.40	2.15
Mexican spotted owl	0.25	0.11	0.75	0.50	0.40	2.01
Pronghorn	1.00	0.16	0.25	0.00	0.40	1.81
Flannelmouth sucker	----	----	----	-----	-----	-----
Ferruginous hawk	----	----	----	-----	-----	-----

Appendix 9. Sites of Conservation Concern selected for the Colorado Plateau 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 • Historic Districts • 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 10. Ecological Functions and Services Selected for the Colorado Plateau.

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 11. Change agents selected for the Colorado Plateau 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 grazing (proposed by Dynamac) — Wild horse and burro grazing (proposed by AMT) — Wildlife grazing (proposed by AMT) — Groundwater and Surface Water Extraction, Development, and Transportation — Recreational Uses — Pollution (Air Quality) • Climate change

APPENDIX 12. 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 http://www.blm.gov/wo/st/en/prog/more/climatechange/reas.html.

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 1). This grouping proved useful not only for the data needs assessment, but later in data gap identification as well.

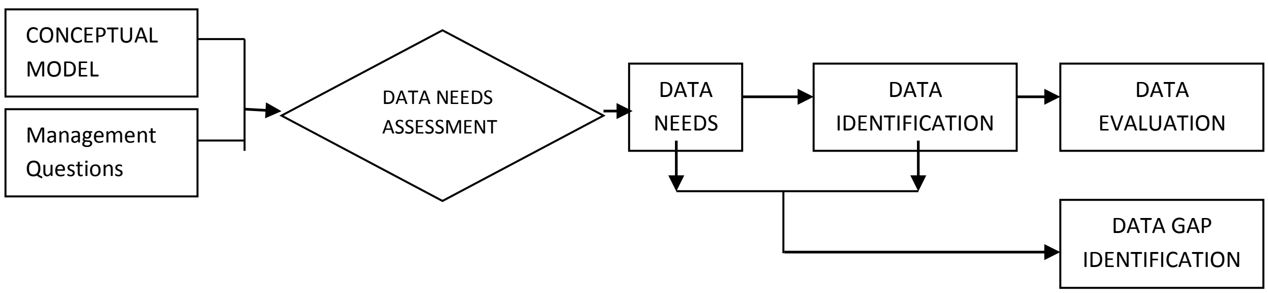

Figure 1**.** Process of data needs assessment through data evaluation and data gap identification.

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.

APPENDIX 13

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 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).

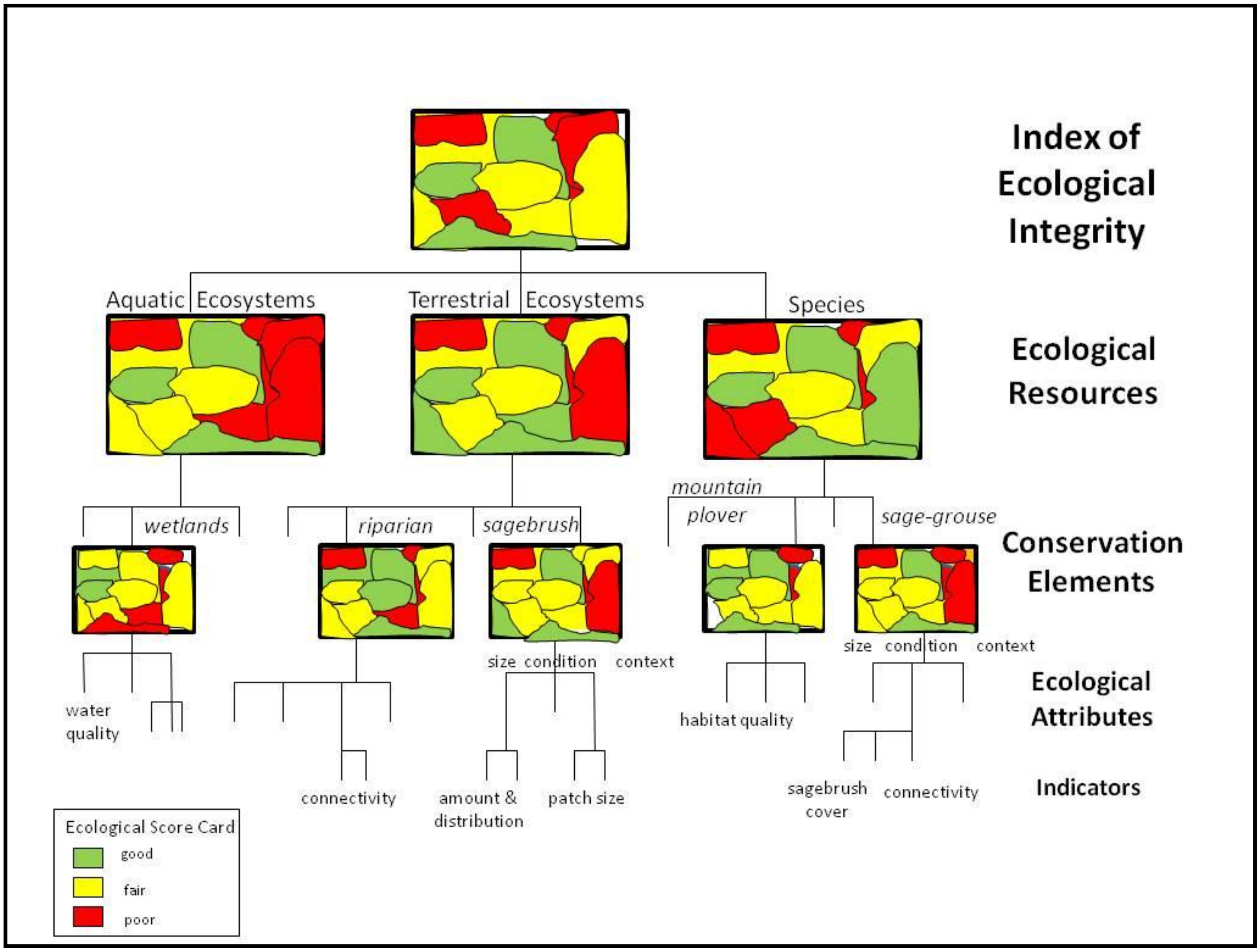

Figure 1. Hypothetical Ecological Integrity Assessment Framework. Conceptual models are used to identify key ecosystem components (conservation elements) for three ecological resource values: aquatic ecosystems, terrestrial ecosystems, and species. Assessments for individual conservation elements are 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).

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).

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.

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).

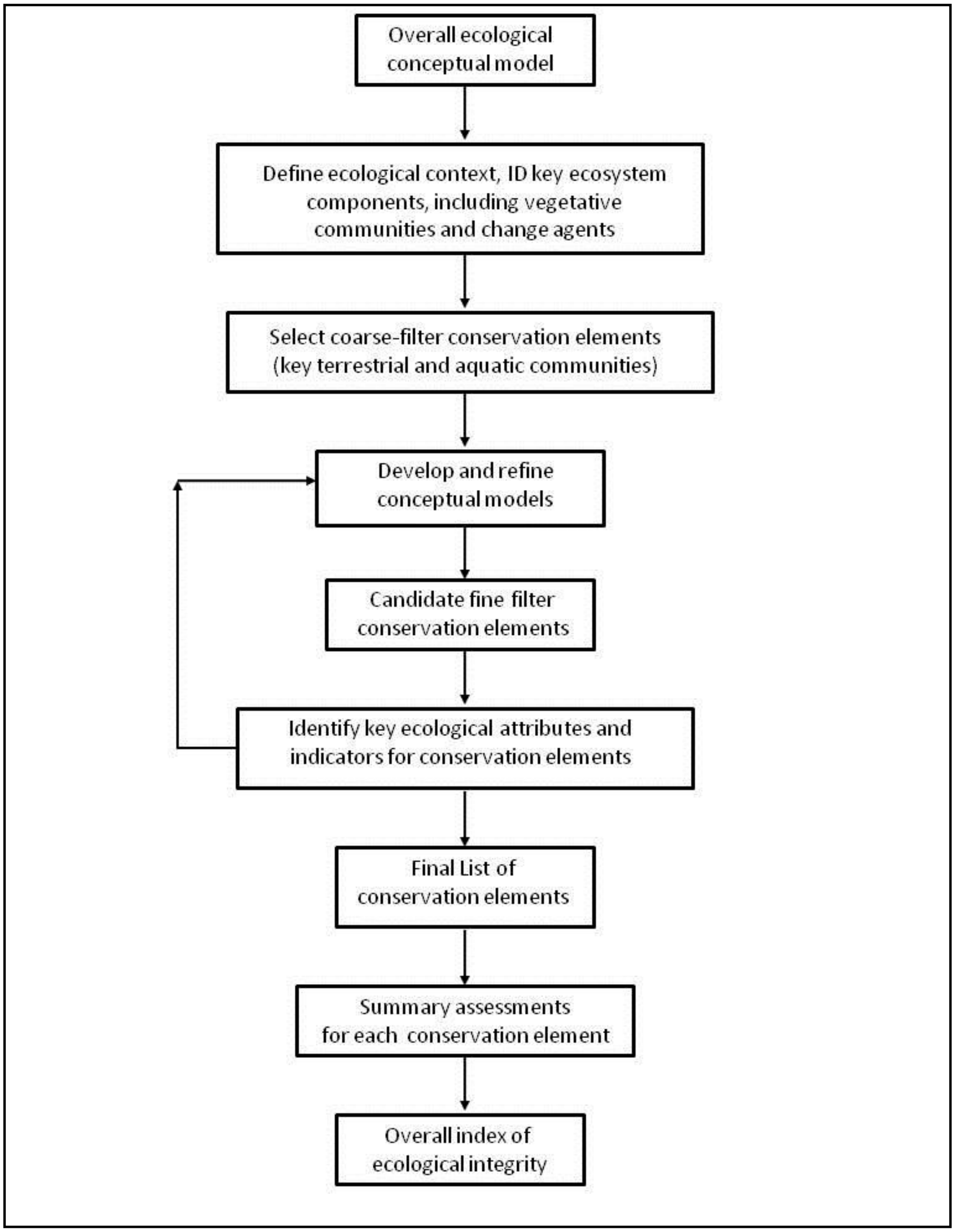

Figure 2. Ecological Integrity Assessment Framework. The process begins with the development of ecological models, which provides the ecological context and will help identify critical components of the ecosystem to be assessed. The process is iterative and contains many feedbacks to refine and identify 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.

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.

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APPENDIX 14. USGS Proposal to Map Cheatgrass across the Colorado Plateau

Date: 15 March 2011

Title: Development of a Cheatgrass Map for the Colorado Plateau

Principal Investigator: Terence Arundel

The purpose of this proposal is to outline the objectives for developing an invasive annual plant map for the Bureau of Land Management (BLM) Colorado Plateau Rapid Ecoregional Assessment. Invasive annual plants (red brome, cheatgrass, and African mustard) are known to have significant impacts on vast areas of dryland environments. Potential impacts of invasive annual grasses include influences on habitat structure, disturbance regimes, and nutrient cycling. These impacts can degrade habitat quality for wildlife and native grass species, as well as having possible socioeconomic impacts on human-valued systems. Identifying the extent and distribution of invasive annual plant species on the Colorado Plateau can assist managers in addressing the potential impacts of invasive annual grasses on the ecoregion.

To identify and address this information need for BLM and National Park Service managers, the U.S. Geological Survey recently conducted a multiyear analysis of mapping invasive annual grasses in Washington County, Utah. This project examined landscape-level dynamics of invasive annual grasses through the analysis of satellite imagery (Landsat 5 and 7) over a 10 year time period (2001-2010). We used a mapping algorithm that was initially developed for a cheatgrass study near Canyonlands National Park. The mapping algorithm toolset is part of a software system, Detection of Early Season Invasives (DESI), which works in conjunction with the ENVI image analysis and processing software. The DESI toolset uses Landsat Thematic Mapper(TM) and Enhanced Thematic Mapper (ETM) to detect early-season invasive grasses (cheatgrass). Specifically, DESI compares spring and summer Landsat images and models the abundance of cheatgrass, other brome grasses, and spring weeds as a function of seasonal differences in vegetation greenness. Results of this mapping effort can provide resource managers, stakeholders, and partners in the ecoregion and at the local level for the management of fuels, fire, sensitive plant and animal populations, wildland ecosystems, and socioeconomic issues.

The project will consist of a sequence of phases requiring the analysis and acquisition of optimal Landsat imagery, development of the necessary analysis images from the Landsat images using the DESI toolset, mosaicking the 16 seasonal (Spring and Summer) Landsat images into a series of analysis images required for the DESI toolset, and applying the DESI analysis algorithm to create a seamless cheatgrass map for the Colorado Plateau ecoregion. Ultimately, the final cheatgrass map will be converted from a raster format to a vector format with both sets of data allowing for further analysis in either image analysis software (ENVI) or a geographic information system software (ArcGIS). Based on the results of the Washington County project where we determined the peak distribution of cheatgrass for the 10 year study occurred during 2005 (see Figure 1), we propose using Landat imagery from 2005 to create a cheatgrass map for the Colorado Plateau ecoregion.

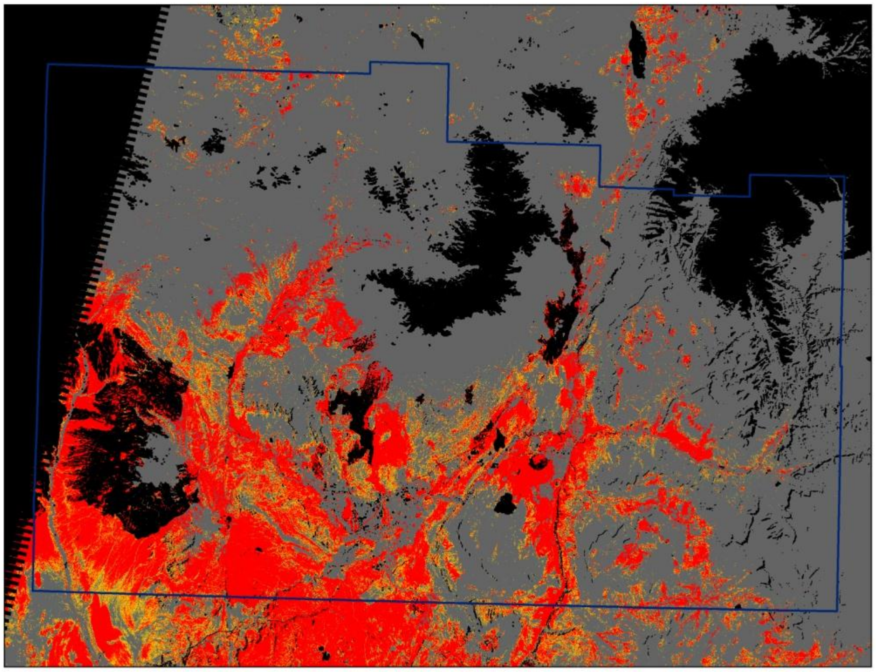

Figure 1. Map illustrating the extent and distribution of cheatgrass in 2005.

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APPENDIX 15. Attribute and Indicator Table for Conservation Elements

Conservation Element	Key Ecologcal 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)
Peregrine Falcon	Actively breeding peregrine falcons	Number of active nests	1 breeding pair (3 year running average)	2 - 4 breeding pairs (3 year running average)	5 -10 breeding pairs (3 year running average)	10 breeding pairs (3 year running average)	
	Breeding habitats	Distance from human disturbance			>1km		GBBO
	Breeding habitats	Cliff height	<12m		200+ meters		GBBO

Conservation Element	Key Ecologcal Attribute	Indicator	Indicator		Rating		Citation
Ferruginous Hawk	abundance of main prey	jackrabbit density	<10 per sq km	10-30 per sq km	30-50 per sq km	>50 per sq km	Howard and Wolfe (1976)
	habitat suitability	size of contiguous cropland	>16 ha	8-16 ha	1-8 ha	none	Jasikoff (1982)
	habitat degradation	livestock density	present in large number	present in moderate numbers	present in small numbers	absent	Olendorff (1993)
Big Free-tailed Bat	habitat	plant community	other	montane conifer or mixed forest	temperate woodland	lowland riparian, desert shrub	Milner et al. (1990), Oliver (2000)
	habitat	roosts	other	tree cavities	buildings, caves	rock crevices in cliffs	Milner et al. (1990)
	habitat	elevation	>9,200 ft	7,550-9,200 ft	5,900-7,550 ft	<5,900 ft	Milner et al. (1990), Oliver (2000)
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
Yellow-breasted Chat	Population size & dynamics	Abundance	0 -0.31 birds/ha	0.31 -0.62 birds/ha	0.62 -0.93 birds/ha	>0.93 birds/ha	Golet 2011
	Population size & dynamics	Shrub density	Low			High	
	Habitat	Elevation			<1600m		

Conservation Element	Key Ecologcal Attribute	Indicator	Indicator Rating				Citation
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)
	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)

Conservation Element	Key Ecologcal Attribute	Indicator	Indicator		Rating		Citation
	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)
	mortality	shooting	Occurs	--	--	doesn't occur	Beans (1996)
	population	surveys					Good et al. (2004)
Gunnison Sage Grouse	habitat	plant communties	developed	agricultural fields	grasslands	sagebrush, riparian, wet meadows	Lupis (2005)
	habitat degradation	sagebrush loss from leks	<0.6 mi of active lek	0.6-4.0 mi from active lek	4.0-6.0 mi from active lek	none in vacinity	GSRSC (2005)
	disturbance	development footprint	<0.6 mi of active lek	0.6-4.0 mi from active lek	>4.0 mi from active lek	none in vacinity	GSRSC (2005)
Greater Sage Grouse	general habitat	cover type	cultivated fields	scrub-willow; sagebrush savannas	small sagebrush; forb rish mosaics	tall sagebrush	Schroeder et al. (1999) Connelly et al. (2004)

Conservation Element	Key Ecologcal Attribute	Indicator	Indicator		Rating		Citation
	habitat	invasive conifers (e.g. junipers)	abundant and encroaching	present but not encroaching	few and not encroaching	absent	Connelly et al. (2000)
	nest sites	mean sagebrush canopy cover	<15% or >38%	15-23%	23-30%	30-38%	Connelly et al. (2000)
Gunnison's Prairie Dog	forage	available foods	shrubs	insects	forbs	grasses	Shalaway and Slobodchikoff (1988)
	habitat	elevation	<4,500 ft or >11,000 ft	4,500-5,000 ft or 10,000-11,000 ft	5,000-6,000 ft or 8,500-10,000 ft	6,000-8,500 ft	Longhurst (1944), Pizzimenti and Hoffman (1973)
	habitat	slope	>15%	5-15%	2-5%	0-2%	Fitzgerald and Lechleitner (1974)
White-tailed Prairie Dog	habitat	elevation	<4,160 ft or >9,630 ft	8,525 - 9,630 ft	7,640 - 8,525 ft	4,160 - 7,640 ft	Utah Natural Heritage Program
	habitat	slope	>10 degreess	5-10 degrees	0-5 degrees	0 degrees	Collins and Lichvar (1986)
	habitat	max vegetation height	>92 cm	62-92 cm	31-62 cm	<31 cm	Collins and Lichvar (1986)
Black-footed Ferret	prey	prarie dog density	<3.63 per ha	3.63-5 per ha	5-7 per ha	>7 per ha	Houston et al. (1986), Biggins et al. (1993)
	prey	total area of prairie dog colonies	<800 ha	800-1,900 ha	1,900-3,000 ha	>3,000 ha	Houston et al. (1986), Biggins et al. (1993)

Conservation Element	Key Ecologcal Attribute	Indicator	Indicator		Rating		Citation
	dispersal	prairie dog intercolony distance	>4.3 km	3.2-4.3 km	2.1-3.2 km	<2.1 km	Minta and Clark (1989)
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	
Mexican Spotted Owl	habitat	forest type	spruce-fir, pinyon-juniper, low elevation riparian	ponderosa pine, gambel oak, AZ cypress	Madrean pine-oak, evergreen oak, high elevation riparian	mixed-conifer (Douglas-fir and/or white fir)	Ganey and Balda (1989), Ganey and Dick (1995)
	habitat	canopy closure	<55%	55-67%	68-80%	>80%	Ganey and Balda (1989), Ganey and Dick (1995)
	habitat	physiography	-	-	mountain slopes	narrow, steep-walled deep canyons below 2,300 ft	Ganey and Balda (1989), Ganey and Dick (1995)
Pronghorn	habitat	distance to water	>6.5 km	4.5-6.5 km	4.5-1.5 km	<1.5 km	Yoakum et al. (1996)
	movement	barriers	abundant	common	few	none	Jaeger and Fahrig (2004)

Conservation Element	Key Ecologcal Attribute	Indicator	Indicator		Rating		Citation
	habitat	diet	woody vegetation	single food type - grass or shrub	somewhat mixed food type	well-mixed food - forbs, grass, and shrubs	Yoakum et al. (1996), Martinka (1967)
Burrowing Owl	thermal biology	elevation	>9,000 ft	7,500-9,000 ft	5,500-7,500 ft	<5,500 ft	Utah Natural Heritage Program (2007)
	mortality	proximity to roads	<0.5 mi	0.5-1.0 mi	1.0-1.5 mi	>1.5 mi	Haug et al. (1993)
	habitat	aridity/openness of habitat	other	golf courses, fairgrounds, some ag land		dry, open short-grass prairies and steppes	Haug et al. (1993)
Flannelmouth Sucker	habitat	summer water temperature	<10 degrees C or >30 degrees C	10-17.5 degrees C or 29-30 degrees C	17.5-25 degrees C or 27-29 degrees C	25-27 degrees C	Bezzerides and Bestgen (2002)
	habitat	water depth	<0.5 m or >2.5m	0.5-1.0 m or 2.0-2.5 m	1.5-2.0 m	1.0-1.5 m	Beyers et al. (2001)
	dispersal	unblocked linear extent	<1.6 km	1.6-10 km	10-20 km	>20 km	Bezzerides and Bestgen (2002)
Colorado River Cutthroat	habitat	avg max water temperature	<4 degrees C or >20 degrees C	4-6.5 degrees C or 19-20 degrees C	6.5-12 degrees C or 14-19 degrees C	12-14 degrees C	Binns and Eiserman (1979), Hickman and Raliegh (1982)

Conservation Element	Key Ecologcal Attribute	Indicator	Indicator		Rating		Citation
	habitat	avg min dissolved oxygen	<6.3 mg/L	6.3-7.2 mg/L	7.2-9 mg/L	>9 mg/L	Hickman and Raliegh (1982)
	flow regime	avg daily base flow	<25%	25-37.5%	37.5-50%	>50%	Binns and Eiserman (1979), Hickman and Raliegh (1982)
Razorback Sucker	habitat	water bidy	irrigation canals	small rivers, reservoirs	medium rivers	large rivers	Valdez et al. (2002)
	breeding habitat	river feature	rapids, riffles	slow runs, eddies	pools, off-channel flooded pits	backwaters	Osmundson et al. (1995)
	habitat	summer water temperature	>29 degrees C or <12 degrees C	26.9-29 degrees C or 12-17.5 degrees C	24.8-26.9 degrees C or 17.5-22.9 degrees C	22.9-24.8 degrees C	Buckley and Pimentel (1983)
Plant Species							
Pinyon Pine	Habitat	elevation	<1400 m	---	1400 m - 2700 m	2100 m - 2400 m	Cronquist (1972)
	mortality	Fire return interval	<100 years		100 to 300 years	>300 years	Keeley (1981)
	Climate	precipitation	<102 mm		102 mm - 520 mm		Ffolliott (1974)

APPENDIX 16. Final AMT-Approved Colorado Plateau REA Management Questions 3-17-11: TOTAL 45

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)?
3. Which HMAs and allotments may experience significant effects from change agents including climate change?
4. Where are soils that have potential to have cryptogamic soil crusts?
5. What/where is the potential for future change to the cryptogamic crusts?
6. Where are hotspots producing fugitive dust that may contribute to accelerated snow melt in the Colorado Plateau?

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. What are seasonal discharge maxima and minima for the Colorado River and major tributaries at gaging stations?
4. Where are the alluvial aquifers and their recharge areas (if known)?
5. What is the condition of these various aquatic systems defined by PFC?
6. Where are aquatic systems listed on 303d for degraded water quality or low macroinvertebrate diversity?
7. What is the location/distribution of these aquatic biodiversity sites?
8. Where are the areas of high and low groundwater potential?

C. ECOLOGICAL SYSTEMS MANAGEMENT QUESTIONS

1. Where are existing vegetative communities?
2. Where are vegetative communities 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), seasonal and breeding habitat, and movement corridors (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. Where are HMAs located?

E. WILDFIRE MANAGEMENT QUESTIONS

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 the Fire Regime Condition Classifications?
4. Where is fire adverse to ecological communities, features, and resources of concern?

F. INVASIVE SPECIES MANAGEMENT QUESTIONS

1. Where are areas dominated by tamarisk and cheatgrass, and where are 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 of suitable biophysical setting (precipitation/soils, etc.) with restoration potential?

G. FUTURE DEVELOPMENT MANAGEMENT QUESTIONS

1. Where are areas of planned development (e.g., plans of operation, urban growth, transmission corridors, governmental planning)?
2. 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 type of allotment?
5. Where are the areas of potential woody biomass for energy utilization?

I. AIR QUALITY MANAGEMENT QUESTIONS

1. Where are the viewsheds adjacent to 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 for fragmentation as a result of climate change in 2060?
3. Where are areas of species conservation elements distribution change between 2010 and 2060?
4. Where are aquatic/riparian areas with potential to change from climate change?