Colorado Plateau

Rapid Ecoregional Assessment Report

U.S. Department of Interior
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Photo: Rough mule's ear (*Wyethia scabra*). Arches National Park, N. Herbert

Executive Summary

Rapid Ecoregional Assessments: Purpose and Scope

Rapid Ecoregional Assessments (REAs) are a product of the evolution of the Bureau of Land Management (BLM) toward a landscape approach to land and resource management. Using the landscape approach, the BLM hopes to integrate available scientific data from BLM field offices, other federal and state agencies, and public stakeholders to develop collaborative management efforts across administrative boundaries. Regional-scale information and assessment analyses on current and future condition will be used by the BLM and its partners to assist with land use planning, developing best-management practices, authorizing uses, and establishing conservation and restoration priorities. REAs are informational tools, not decision documents.

The regional scope of the Colorado Plateau REA and the assessment of its numerous conservation elements and their interactions with change agents produced a massive volume of results that can only be summarized within the constraints of a report of reasonable length. Major highlights of the results appear in the body of the report and appendices provide more detailed information on methods and models. Several key aspects of the REAs highlight their utility to the BLM:

Management Questions: Management questions are the foundation and catalyst for the REAs because they determine the scope of data requirements and analyses. BLM land managers and partners provided a broad range of management questions to the REA to frame regional issues and data needs (full list in Section 2.4.1). The regionally-significant management questions developed for each REA match the scale of the assessment. The 34 management questions prepared for the Colorado Plateau REA refer to native and invasive flora and fauna, disturbance factors or change agents that affect present and future resource status, and significant (designated) sites and ecological functions and services.

Ecoregional Scale: Region-wide analyses explaining the association of native species, aquatic and terrestrial resources, and environmental change agents provide the BLM with another scale of consideration beyond the field office level. REAs thus inform future management planning across multiple spatial scales and jurisdictional boundaries to prioritize resource uses. They also provide a management mechanism for ensuring species' access to seasonal habitats and migration corridors by maintaining connectivity among populations. At the same time, while REAs are scaled at the ecoregional level, they also provide conceptual and geoprocessing models that can be reworked at the state or field office levels using more refined data.

Data Compilation: One of the more important components of the REA process is data compilation in topical areas that are regionally significant. REAs do not involve original research, but they use existing data, modeling, and geographic information system (GIS) analyses to answer a broad range of management questions. The REA effort provides a baseline of information and results built on spatial data that was publicly available during the 2010–2012 time frame. In all, 217 data layers were used to create hundreds of final derived results and maps. The intensive collection and organization of spatial data in itself is of value to the BLM as a library or atlas of spatial data for use in future agency investigations.

Assessing Current Condition: The evaluation of the current status of regionally-significant biotic elements (wildlife and plant species) and abiotic factors (e.g., soils, water resources) was a key aspect of the REA. Eight characteristic vegetation communities of the Colorado Plateau represented the coarse-filter component (Table 2-2, Section 2.4.2). Fine filter elements were represented by 18 wildlife species conservation elements as well as a list of designated sites and essential ecosystem functions and services (e.g., aquatic systems, riparian areas, and soil stability).

Because of the spatial nature of the REAs, describing status for various conservation elements and resource values requires the ability to identify and map specific characteristics of that resource. As a result, REA results and the regional assessments, while valuable, must always be considered incomplete: some important elements will be absent because their effects were not visible or because data to represent them were not available.

Status is the current condition of various conservation elements resulting from all stressors and changes imposed on a prior historical condition or benchmark reference condition.

Projecting Future Condition: REAs also evaluate the potential of change agents—including wildland fire, invasive species, development, and climate change—to affect ecoregion condition. Assessment output products documenting potential-for-change demonstrate how current evidence of cumulative impacts may be projected into the future to identify potential trade-offs, alternatives, and mitigation strategies for BLM planning purposes. A development-related REA product of interest to the BLM is the location of areas with high potential for traditional or renewable energy development. REA results contain current and potential development data layers that were merged with mapped distributions for the various conservation elements to identify how and where the elements may be affected by various planned and potential energy development areas.

Application to Adaptive Management: REAs are timely in supporting planning, management, and mitigation strategies for impacts anticipated from rapidly-developing issues related to traditional and renewable energy development, the spread of invasive species, changing fire regimes, and climate change. REAs provide a foundation for an adaptive management approach that will allow implementation strategies to be adjusted for new information and changing conditions. REAs represent a baseline condition from which to evaluate the results of adaptive management and to characterize potential trends in resource condition both in the near-term (2025)—as a consequence of development activities—and in the long-term (2060) as a result of climate change. The final chapter of this REA report (Chapter 6) provides examples showing how the data and results may be arranged and manipulated using mapped and tabular results, for all land ownerships and BLM-lands only, for areas of intact habitats, resource value hotspots, and opportunities for connectivity with existing designated lands.

REA Products and Results

Landscape Intactness

Intactness is a quantifiable estimate of naturalness measured on a gradient of anthropogenic influence and based on available spatial data.

The BLM and other participants in the Colorado Plateau REA agreed to emphasize the concept of *intactness* for the mapping of ecological condition. As defined and used here, intactness is a measure of naturalness as well as an attribute that can be defensibly supported by existing geospatial datasets, mapped, and reasonably tracked through time. Because vegetative cover represents wildlife habitat, it serves as a surrogate to estimate the status of species that depend on that habitat, particularly since spatial data for the pre-disturbance distribution or abundances of various wildlife species are typically not available. For example, in the Colorado Plateau, representative areas of canyons and tablelands may be placed along a gradient of intactness (or conversely, along a gradient of disturbance) with sites that are experiencing increasing levels of disturbance considered to have lower intactness. The lowest intactness levels occur in areas completely converted from their original character. Terrestrial (Figure 1) and aquatic intactness models were created for the entire ecoregion. Intactness models serve as the basis for assessing conservation element status for current and future condition.

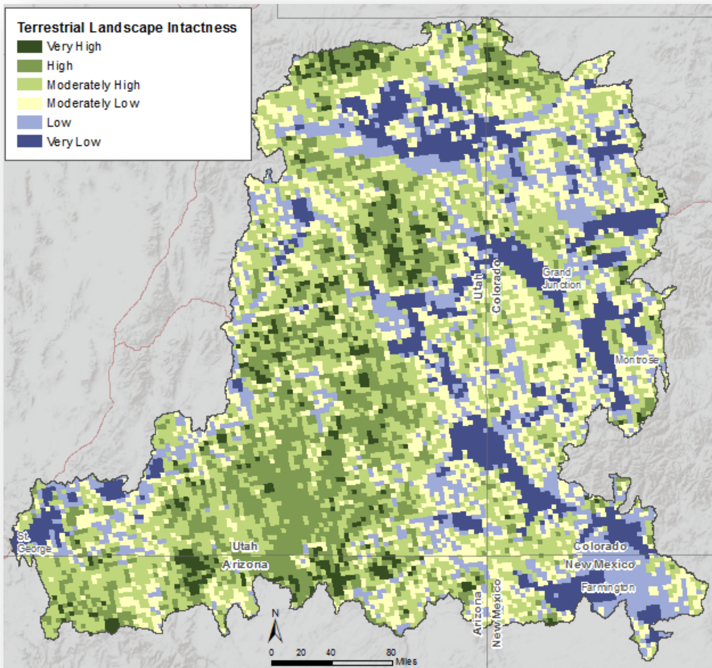


Figure 1. Colorado Plateau terrestrial landscape intactness in six classes from High (relatively undisturbed in dark green) to Very Low (highly disturbed from agriculture, resource development, or urbanization in dark blue) depicted with a 4 km X 4 km grid cell. Intactness is a critical element for assessing the status of conservation elements for current as well as near-term future (2025) condition.

Change Agents Current and Future

The status or condition of various conservation elements cannot be discussed without first examining the risks that these elements experience from a collection of regional disturbances or change agents. Natural and anthropogenic disturbance factors are represented in the REA by four change agents: land and resource use (development), climate change, invasive species, and wildland fire. The major change agents and their effects on conservation elements are considered in the current time frame and projected over the near-term future (2025) for development and the longer term future (2060) for climate change. Land and resource use is the largest change agent class, encompassing urbanization and road density, oil, gas, and mining, renewable energy development, agriculture, grazing, ground and surface water extraction, and recreation.

REA results include mapped and tabular products describing historical and recent (within the last 20 years) change to major vegetation communities from disturbances such as urbanization and roads development, agriculture, invasive species, fire, and mechanical treatments. The greatest amount of total area changed (nearly 2.5 million acres or 30% of ecoregion area) was in big sagebrush shrubland, with maximum acres altered for urbanization and road development (565,000 acres), agriculture (495,000 acres), and invasive species (about 846,000 acres). Another 572,000 acres of sagebrush habitat were converted to uncharacteristic vegetation, for example, from pinyon-juniper expansion into sagebrush shrubland. The sustainability of the greater sage-grouse depends entirely on intact expanses of sagebrush. Sage-grouse distribution has been reduced to 56% of its former range in the West during the last century because of the fragmentation and conversion of sage habitats. Oil and gas drilling is the most pressing current and future threat to the sustainability of the sage-grouse in the Colorado Plateau (see Sage-grouse Case Study Insert). Across the West, more than 17 million acres of public lands—or 44% of the lands that the federal government leases for oil and gas development—have been authorized for drilling within the distribution of the greater sage-grouse. REA analyses produced future status results for sage-grouse (and each of the other conservation elements) relative to near-term (2025) development and potential energy development (a longer term scenario based on mapped energy reserves and renewable energy potential, Figure 2).

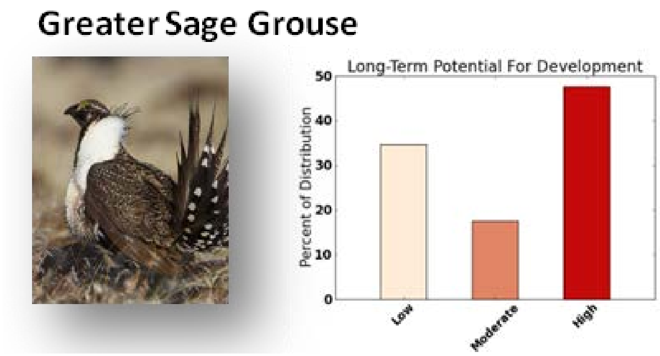


Figure 2. Histogram representing the overlay of sage-grouse distribution with a model of long term potential energy development that included data for oil and gas leases, wind and solar potential, and oil shale and tar sands reserves. Nearly 50% of current sage grouse distribution falls within the High Risk category and almost 20% falls within the Moderate Risk category for potential energy development.

Two invasive plant species of concern, cheatgrass (*Bromus tectorum*) and tamarisk (*Tamarix* spp.), were selected for the Colorado Plateau REA because they are considered significant change agents in the region. These species alter ecosystem processes, such as nutrient cycling and fire regimes. They have the potential to expand their distributions in spite of human and natural disturbances and to adapt and shift their ranges in response to climate change. Invasive annuals out-compete native species by using soil nutrients and water at a greater rate or earlier in the season and by regularly producing greater biomass. The models produced for current and near-term future distribution of invasive species for the REA used multiple models and mapped sources, but the results likely underestimate the total distribution of invasive vegetation in the ecoregion (Sections 4.3.1 and 5.3). Invasive species, such as cheatgrass, increase fire frequency and size and the duration of the fire season by increasing fine fuel loads and continuity, thus allowing fires to spread into areas that were once fuel-limited. The degree to which fire may become an ecologically significant change agent relates to the extent to which the fire regime has been altered compared with reference conditions and the associated effects of the altered fire regime on the vegetation community. Four fire-related management questions were addressed in the REA related to fire occurrence within the past decade, areas with potential to change from wildfire, and areas of fire regime departure from expected frequencies (Section 4.3.2).

A major portion of the report dedicated to future conditions on the Colorado Plateau covers projections of climate change for mid-century (circa 2060, Section 5.4). Three different future climate projections were investigated for the REA; but the ECHAM5-driven RegCM3 climate projections were selected for the body of the report to evaluate potential impact on the various conservation elements. ECHAM5 has been identified as one of the better models to represent natural climate variability, and the regional RegCM3 model represents the North American Monsoon (summer rainfall pattern) which is important to Colorado Plateau vegetation dynamics (see Climate Change Scenario below).

Conservation Element Status

Current status for each species and conservation element was derived by overlaying conservation element distribution with the overall intactness model (Figure 1). The intactness model provides a regional perspective of vegetation condition, habitat quality, development, and natural habitat fragmentation patterns. Not all species demonstrate the same level of tolerance to the various model inputs, but the overall intactness model provides a standard baseline from which to explore specific species or areas where tolerances to various components may vary. The regional intactness model may be rerun with new or higher resolution data to test specific thresholds for individual species.

Of the wildlife species, Gunnison sage grouse had the lowest overall status with almost 50% of its distribution in the Very Low intactness category and >85% of its distribution in the three lowest categories (Figure 3). Other species with low status signatures were the Colorado River cutthroat trout, Gunnison's prairie dog, and white tailed prairie dog. Prairie dogs are a species that may have a greater tolerance to disturbed landscapes, but many colonies are in conflict with human activities and are under considerable stress. Mexican spotted owl and desert bighorn sheep had relatively high status signatures. The owl's distribution is limited, but its status score reflects the fact that the species' prime (and remaining) activity centers are concentrated in highly intact areas of the landscape, i.e., in protected National Parks and Monuments. A similar situation exists for desert bighorn, inhabiting (and being reintroduced) into steep, remote habitats.

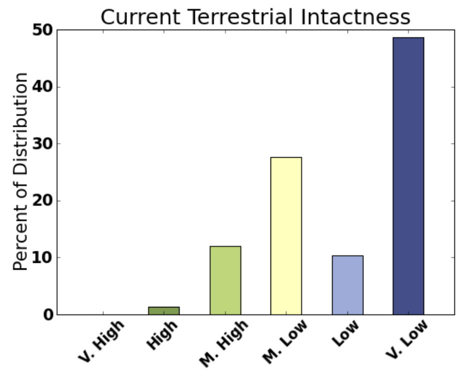


Figure 3. Histogram represents status for Gunnison sage grouse in 6 intactness classes with 50% of its distribution in the Very Low intactness class.

Climate Change Scenario

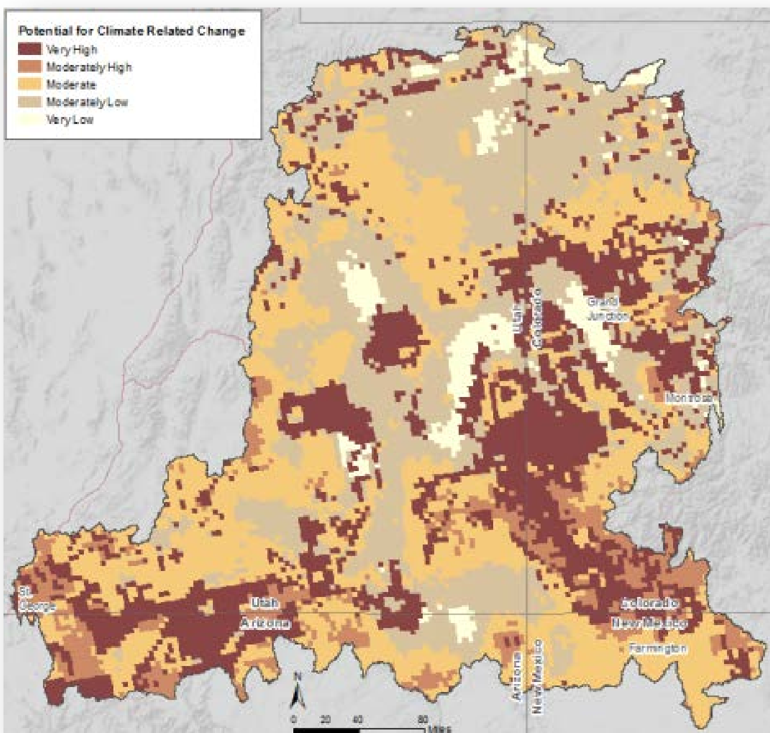


Figure 4. Map shows overall potential for climate change expressed in five classes from Very High (dark red-brown color) to Very Low (off-white). The southwest, west-central, and eastern portions of the ecoregion have the highest potential for climate change.

To simplify the complex and numerous future climate projections, a number of the key findings were selected from the analyses and assembled into an overall relative climate change map (Figure 4). The model inputs included potential for summer temperature change and potential for winter temperature change averaged into a single factor, potential for runoff change, potential for precipitation change, and potential for vegetation change. Five output classes from Very High to Very Low represent the potential for risks from climate-related change. The exposure of species, habitats, and sites to predicted climate change is represented by overlaying the climate model with the distribution of each conservation element to identify areas potentially affected by climate change. Gunnison's prairie dog and Gunnison sage-grouse showed the highest exposure with 70% of their current distributions expected to be under Very High climate stress by 2060. Yellow-breasted chat, a riparian species, also showed High exposure to changing climate in the region. Of the vegetation communities, those showing the most area under High climate change potential include the shrublands (especially big sagebrush and blackbrush-Mormon-tea communities), riparian vegetation, and pinyon-juniper woodland. Climate change challenges the standard management practice of setting aside threatened species activity areas or critical habitats relative to areas deemed developable, when vegetation community, ecosystem, and even ecoregion boundaries will be in constant flux under climate change.

Application of Results

The vast amount of information produced by this REA can and must be examined in multiple ways and at multiple scales. Chapter 6 provides examples of applications of the results by manipulating maps and data tables in various planning scenarios using concentrations of conservation elements (or hotspots) for energy planning, and protected area or connectivity planning. The examples given in Chapter 6 are for hotspots over all lands, all lands minus developed and designated lands, and BLM-only lands. In the example below (Figure 5), one can see where high concentrations of conservation elements and areas of high intactness exist in BLM lands shaded in dark pink. A map of this kind highlights areas of potential conflict, restoration, or mitigation.
The examples in Chapter 6 show the utility of examining the data in detail and becoming familiar with the strengths and weaknesses of the models and the underlying data sources. The models will acquire ecological meaning as they are calibrated with finer scale data and groundtruthing. It is highly likely that higher resolution data and analyses may modify REA results locally, but they will remain valid at the regional scale at which they were produced.

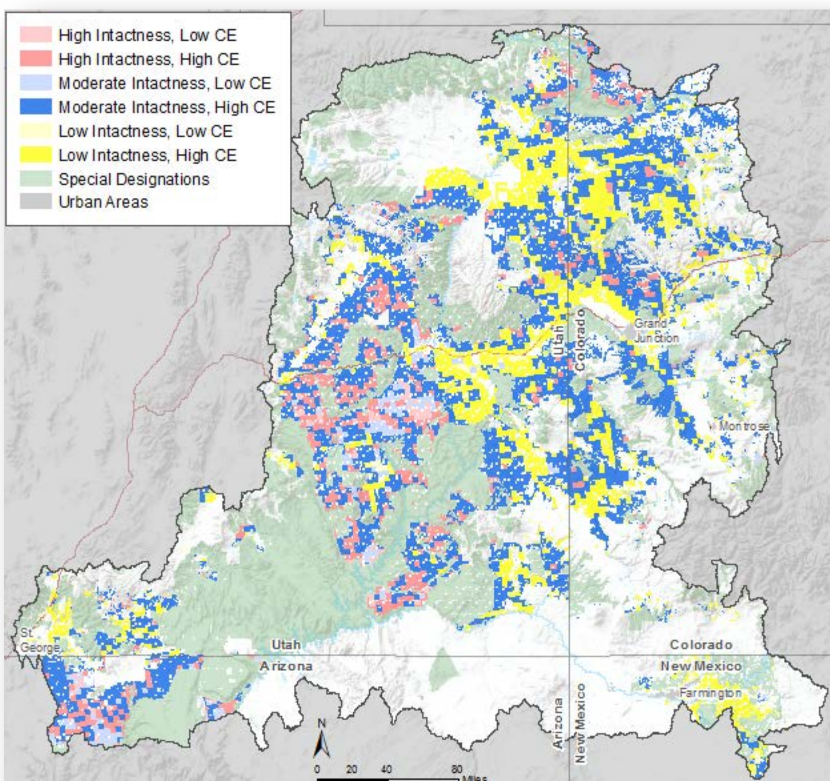


Figure 5. Map shows BLM-managed land areas of various intactness classes in the Colorado Plateau intersected with low and high concentrations of conservation elements (CEs). Designated protected areas are shown in green; white areas are non-BLM lands. Darker pink areas represent the intersection of high concentrations of conservation elements and areas of high intactness.

I. BLM's Approach to Ecoregional Direction and Adaptive Management

Assessments help managers address problems by providing information that can be integrated into future management action. The success of this Rapid Ecoregional Assessment (REA) ultimately depends on how well it helps inform management decisions (Johnson and Herring 1999): 1. Was it contextual? Did it significantly improve understanding about the conditions of the resources being studied within the ecoregion and the consequences of particular actions? 2. Was it integrated? Was that understanding integrated into managers' thinking to guide future action? 3. Was it pragmatic? Did the assessment lead to potential solutions for the management questions?

The contract for this assessment clearly requests information designed to be integrated into specific management approaches. However, the contract stops short of actually integrating the findings into management actions. The Bureau of Land Management (BLM) chose to retain responsibility for all aspects of integrating the assessment into management actions and decisions. The process presented here is conceptual; no process has yet been established as a commitment or accepted as a responsibility by the BLM.

This proposed process helps address the environmental changes the West is experiencing. To be effective in addressing these regional challenges, the process must address them at multiple scales and across multiple jurisdictions. All BLM programs can contribute to this effort. The BLM is exploring innovative approaches to a process in landscape direction across programs and geographic scales. The following paragraphs briefly describe a systematic approach to these ecoregional challenges:

Managing resources at multiple scales: Traditionally, the BLM has undertaken resource management project by project, permit by permit, and land use plan by land use plan, without systematically assessing landscape scale effects. To effectively address the projected environmental changes in the West, resource managers will have to develop the capacity to evaluate effects at multiple geographic scales.

Managing resources across ownerships and jurisdictions: Traditionally, resource managers have focused on activities within their own administrative units. To effectively address the environmental changes the West is experiencing, resource managers will have to develop the institutional and technical capacity to work across ownerships and jurisdictions.

Managing resources across programs: Traditionally, resource management has been defined by programs (e.g. wildlife, range, minerals). To address the environmental changes the West is experiencing, resource managers will have to more effectively integrate activities across programs by inter-disciplinary management.

Standardizing and integrating data: The ability to collect, synthesize and share geospatial information about resource conditions, change agents such as wildland fire, and on-the ground management activities is a critical part of this effort. Without the ability to compile and correlate such information within and outside of BLM, it is extremely difficult to achieve conservation, restoration, and adaptation strategies and to evaluate the effectiveness of such strategies once implemented.

Systematic integration requires some fundamental shifts in the BLM's traditional management practices. Although project-focused work and traditional practices will still be part of BLM's management strategy, the REAs will help the BLM to identify what processes are appropriate for the broader scale landscape approach (Table 1).

Table 1-1. Comparison of differences between aspects of BLM's traditional management practices and the landscape approach represented in the Rapid Ecoregional Assessments.

Traditional Practice	Landscape Approach
Project Focus	Landscape Focus
Program/Functional Direction	Integrated Direction Across Programs
Unit Decision Making	Cross Jurisdictional Decision Making
Unit Priorities	Collaborative and Partnership Priorities
Program Accomplishments	Integrated Accomplishments Across Programs
Authorize Uses and Mitigate Ecological Values	Ecological values and Use Authorizations Considered Equally
Ecological Component (Individual Species)	Ecological Function and Service
Agency Funding	Partnership Leveraged Funding

Many of the landscape approach activities listed in the table above have been part of BLM's approach at the land use planning scale. BLM is undertaking the following activities at the regional scale to deal with environmental changes:

Rapid Ecoregional Assessments

Working with agency partners, BLM is conducting rapid ecological assessments like this one, covering approximately 450 million acres of public and non-public lands in ten ecoregions in the American West to identify potential priority areas for conservation and development. Over time, the BLM anticipates collaboration with the Landscape Conservation Cooperatives (LCCs, public-private partnerships for adaptive management grounded in science) to periodically update ecoregional assessments and identify science needs.

Ecoregional Direction

BLM is developing a standard ecoregion-scale process for conserving or developing priority areas and for incorporating REA results into land use planning, environmental impact assessments, use authorizations, conservation and restoration project planning, and acquisition of conservation easements.

Ecoregional direction uses the information from the REAs, along with input from partner agencies, stakeholders, and Tribal agencies to develop a broad scale management strategy for an ecoregion's BLM-managed lands. This broad scale management strategy will identify focal areas on BLM-managed lands for conservation and development, including areas for conserving wildlife habitats and migration corridors and for potential energy development and urban growth. Ecoregional direction will also provide a blueprint for coordinating and implementing these priorities at the BLM's state and field-office levels. Ecoregional direction links REAs and the BLM's Resource Management Planning and other on-the-ground decision making processes. It also helps integrate existing initiatives and facilitates coordination across programs, offices, and partnerships. Ecoregional direction establishes a regional roadmap for reviewing and updating Resource Management Plans, developing multi-year projects for identified priority conservation and development areas, establishing Best Management Practices for authorized use, designing regional adaptation and mitigation strategies, and developing conservation land acquisitions.

> Ecoregional direction uses the information from the REAs and stakeholders to develop a broad scale management strategy for an ecoregion's BLM-managed lands.

Ecoregional direction development begins with conversations among regional partners about stepping the REAs down into management. Partners that guide the step-down process will likely include BLM State Directors (or their representatives) and equivalent peers from other federal, state, and Tribal agencies and entities.

The partners will review the completed REA and other assessments to evaluate proposed findings and recommendations and:

- Delineate a schedule, process and expected products;
- Identify proposed and ongoing activities within the REA region. Such activities may include proposed or on-going assessments, planning efforts, National Environmental Policy Act (NEPA) analyses, or special area evaluations;
- Communicate with organizations knowledgeable about the REA or potentially affected by it; and
- Conduct partnership and stakeholder outreach.

Individual partners will develop their own respective direction to implement the agreements. In the case of the BLM, this will be in the form of ecoregional direction. In developing ecoregional direction, the proposed findings and recommendations will be discussed with:

- The affected BLM's State Management Teams;
- The leadership of local, state, federal and Tribal partners; and
- The Washington Office if there are potential national policy and coordination issues.

After reviewing the proposed findings and recommendations and discussing them with the leadership of potentially affected partners, the BLM State Director(s) may issue ecoregional direction outlining what the BLM will do over the next 3–5 years to incorporate the Rapid Ecoregional Assessments into management activities. If desired, the partners may coordinate the implementation of ecoregional direction among the participating entities.

Monitoring and Adaptive Management

Adaptive management is a systematic process for continually improving management policies and practices by learning from the outcomes of previously employed policies and practices. Ecoregional assessments help to move adaptive management from a concept to an applied approach; if rapid ecoregional assessments reoccur every 5 to 10 years as planned, they will serve as a monitoring and evaluation process for the effectiveness of adaptive management. Working with partners, BLM employs a national Assessment, Inventory and Monitoring (AIM) strategy that identifies core indicators of terrestrial and aquatic condition, performance indicators for fish and wildlife action plans, and scalable sampling designs to help integrate and focus BLM's monitoring activities and facilitate adaptive management.

> Adaptive management is a systematic process for continually improving management policies and practices by learning from the outcomes of previously employed policies and practices.

1.1 References Cited

Johnson, K. N., and M. Herring. 1999. Understanding bioregional assessments. Pages 341–376 *in* Johnson, K. N., F. Swanson, M. Herring, and S. Greene (eds.), Bioregional assessments: Science at the crossroads of management and policy, Island Press, Washington, D.C.

II. INTRODUCTION

2.1 Why Conduct Rapid Ecoregional Assessments?

> The gap between conservation at the species and community level and planning at the landscape level has not been bridged.
>
> — Noss 1987

Rapid Ecological Assessments (REAs) are a product of the Bureau of Land Management's evolution toward a landscape approach to land and resource management. 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. Another objective of the REAs is to assess the current status of selected ecological resources (conservation elements) at the ecoregional scale and to investigate how this status may change in the future across several time horizons. For these assessments, *status* represents the current condition of the various conservation elements resulting from all stressors and changes imposed on a prior historical condition or benchmark reference condition. The stressors are defined as change agents—natural phenomena or human activities that influence the status of conservation elements. REA results identify areas with high ecological integrity and high biological and ecological value—conservation areas, biological hotspots, and wildlife corridors—to provide a better understanding of key ecosystem processes and the potential impacts of future changes. REAs are timely in supporting planning, management, and mitigation strategies for impacts anticipated from various climate change scenarios as well as rapidly developing issues related to renewable energy development, the spread of invasive species, and changing fire regimes.

The knowledge gained from these assessments will inform future management planning across multiple spatial scales and jurisdictional boundaries. Part of the reason for the continuing decline in many species of concern relates to the scale at which many of our land management practices occur. Because of the pattern of ownerships and administrative districts across a region, management actions directed at any particular issue or species are often implemented in piecemeal fashion. To successfully maintain rangewide species and habitat viability requires managers to coordinate local efforts at a regional scale by practicing cross-jurisdictional planning, involving federal and state management agencies, non-governmental organizations, and citizen working groups. For example, whether a regional species issue is desert bighorn, desert tortoise, sage grouse, or northern spotted owl, pooling information across ownerships is necessary to prioritize resource uses, allow species' access to seasonal habitats and migration corridors, and provide connectivity between productive and less productive populations.

Rapid ecoregional assessments assist regional management by compiling, organizing, and maintaining a comprehensive source of regional datasets and analyses and making them available to land managers and the public to query and reassemble in issue- and project-specific ways. REAs are not meant to allocate resource uses or make management decisions. One of the more important components of the REA process is data compilation in topical areas that are regionally important. REAs, being *rapid* assessments, do not involve original research, but they use existing data, modeling, and GIS analyses to answer a broad range of management questions. The intensive data collection required to conduct an REA reveals knowledge gaps and highlights areas for future ecosystem monitoring and research. REAs also provide a baseline condition from which to evaluate the results of adaptive management and to characterize potential trends in resource condition both in the near-term (2025)—as a consequence of development activities—and in the long-term (2060) as a result of climate change. While REAs are scaled at the ecoregional level, they provide conceptual and geoprocessing models that can be reworked at the state or field office levels using more refined data.

2 .2 The Spatial Nature of REAs

2.2.1 Mapping and Modeling

Because an REA is a rapid assessment, not research, the analyses and results are limited by available spatial data. The REA effort provides a baseline of information and results built on spatial data that was publicly available during the 2010–2012 time frame. The intensive collection and organization of spatial data in itself provides value to the BLM to serve as a library or atlas of spatial data for use in future agency investigations. In addition, the use of the spatial information to produce analyses explaining the association of native species, aquatic and terrestrial resources, and environmental change agents across the whole ecoregion provides BLM with another scale of consideration beyond the field office level that will assist in the coordination of regional issues among various BLM Field Offices (and between the BLM and other state and federal agencies dealing with the same issues). Regional-scale information and assessment analyses on current and future condition will be used by the BLM to assist with land use planning, developing best-management practices, authorizing uses, and establishing conservation and restoration priorities.

To digest the vast amount of material produced by the assessment, it is important to become familiar with the spatial analysis and modeling tools that made up the core of the REA. As a starting point, conceptual models were created for each conservation element and change agent (natural or human-influenced disturbance) to aid in our understanding of complex interactions between each specific subject and the relevant natural drivers and human-induced changes. To assist in the replication of analyses, process analytical models were developed that detail actual mapping and modeling steps. The more complex analyses required logic modeling to help organize and communicate the process and findings. While most analyses were carried out using ArcGIS Model Builder or python scripts, additional specialized software was utilized, including FRAGSTATS (to evaluate habitat fragmentation), MaxEnt (to build probability surfaces), NetCDF Climate Operator software (to manage climate input data), and MAPSS (Mapped Atmosphere-Plant-Soil System to predict vegetation and runoff response to climate variables).

Although the REA focused on the ecoregion extent, data collection had to be conducted within political boundaries, most prominently at the state level. For example, the Colorado Plateau ecoregion included areas inside four different states—Utah, Colorado, Arizona, and New Mexico. Significant differences existed among the states in what features were routinely mapped, the regularity of mapping techniques used, and attributes assigned to spatial datasets leading to inconsistencies along political boundaries from numerous standpoints in both geometry and content. For the entire ecoregion, all data collection, analysis, and reporting was conducted within the outer boundary of all 5th level hydrologic units (HUC5s) that intersected the Colorado Plateau ecoregion boundary. This buffer was created to mitigate edge effects during spatial analyses and provide an area of overlap for edge-matching between data layers generated for REAs in neighboring ecoregions. All datasets were projected to USA Contiguous Albers Equal Area projection (USGS version) for mapping and modeling.

Assessments of species status, potential for change due to change agents, and ecological integrity were performed using landscape reporting units. These units provided a uniform framework for summarizing detailed information to a higher level that allowed integration across multiple disparate factors. The reporting units used for this REA were 1) a 4 km X 4 km grid for current and near-term status and potential for change of terrestrial conservation elements, terrestrial intactness, long-term climate potential for change, and current, near-term, and long term development change; and 2) 5th level hydrologic units (HUC5s) for current and near-term status and potential for change of aquatic conservation elements and ecological integrity (intactness, defined in Section 2.4.4).

2.2.2 Using Existing Data and Determining Data Gaps

One of the overarching requirements of the REA was to use pre-existing data as inputs to the modeling process. Data acquisition, review, and pre-processing occurred throughout the REA process, even though the original intent of the REA was to identify and evaluate all relevant datasets prior to the onset of modeling. Acquisition of existing datasets presented a number of challenges:

- Existing, centralized, and easily accessible datasets are often older, whereas very recently developed datasets often require significant outreach effort to discover and obtain.
- Datasets actively used for BLM planning often became obsolete as soon as they were acquired (e.g., renewable energy priority projects), necessitating multiple acquisitions over the course of this REA.
- Data developed by BLM field offices (e.g. grazing land health evaluations) were generally not available for this REA, including data recently developed for Resource Management Plans because of consistency of data standards and level of effort.
- Existing data on particular themes (e.g., wildlife habitat) tend to vary widely in data quality, accuracy, methodology, thematic resolution, and timeliness across sources, which made it quite difficult to create a seamless dataset across the ecoregion of uniform quality.

For example, although grazing was identified as a change agent in the Colorado Plateau, a lack of consistent data limited assessment products related to grazing. After some discussion, the consensus of Workshop 1 participants was that 1) grazing should be addressed as a change agent that includes all herbivores; 2) grazing data sources should be evaluated; and 3) the Assessment Management Team (AMT) would compile a set of grazing questions. The grazing management questions were added and remained until the end of Pre-assessment Task 3 (March 2011) when BLM determined that no region-wide, readily available spatial data existed for grazing on federal or private land and that the timeframe of the assessment precluded converting BLM's hard-copy records for their grazing allotments into electronic spatial data. As a result, although grazing remained as a change agent and is included in literature review where applicable throughout the assessment report, the grazing management questions were not specifically addressed and were deferred as a possible post-REA sub-assessment. Lack of consistent, region-wide, quality data affected the REA in this and other resource areas, such as recreation and off-highway-vehicle (OHV) routes.

All existing source datasets were evaluated for data quality, and outstanding issues noted. Many more datasets were pre-screened and evaluated than were actually used in modeling, because it was often necessary to compare several datasets for a particular theme to determine those that were most appropriate for the modeling effort. In total, 217 data layers were used to create final derived results and maps for the Colorado Plateau REA. Several key data gaps became apparent during this REA:

- High quality, locally-accurate, and seamless data across the entire ecoregion for most themes.
- High quality and uniform wildlife habitat maps across state boundaries for the species evaluated in this REA.
- Current and detailed grazing allotment use and status datasets for federal and private lands.
- Uniform projections of urban growth, change in agriculture area, and potential development of oil, gas, and renewable energy sources.
- Existing assessments of where species have been surveyed for presence/absence.
- Uniformly developed, detailed maps of soil characteristics (datasets exist but are not complete within ecoregion).
- Consistent recreation data, including OHV routes.

The modeling method used to answer conservation element management questions depended on the data available for species occurrence locations and environmental predictors. Because of the short time frame of the REAs and the stipulation to avoid research, existing models were considered most appropriate. An order of preference for modeling was agreed on by participants in the REA process to use 1) existing high quality models that covered the full ecoregional extent or that could be readily be extended from a portion of the assessment region to cover the desired areal extent; 2) a modeling approach such as MaxEnt (or related software) if enough occurrence data were available, and 3) southwest regional gap analysis (SW ReGAP, Prior-Magee et al. 2007) models if both (more-detailed) existing models and occurrence data were lacking. No new MaxEnt models for potential species distributions were created for this REA because adequate occurrence data were not available for any conservation element species (Table 2-3, Section 2.4.2). State wildlife distribution data were available for many species and generally more detailed than SW ReGAP models, which typically overestimate species distributions; however, in an ecoregion composed of multiple states, edgematching disparate state data at state boundaries was a common problem. Where more detailed state data were not available, or where edgematching issues in data from multiple states could not be resolved, SW ReGAP models were used for species distributions, meaning that distributions for some species like mountain lion and golden eagle were generalized to cover most of the area of the ecoregion. Finally, since correcting or updating datasets was beyond the scope of the REA, any gaps in distribution data were reflected in the results. For example, for desert bighorn sheep, a 55,000 acre area surrounding the Dolores River canyon in Colorado is not represented in the species data used for this REA because the introductions were recent (2010 and 2011) and the spatial data had not been updated.

Regional spatial datasets are constantly evolving; rarely is a dataset of proper extent and quality that exactly fits a project's needs available to pluck off the shelf. At various points in the REA process, participants and the BLM in particular were required to make choices and decisions about various data layers—for example, to allow the use of a dataset with limited extent but high value or one of a coarser scale than specified in the Statement of Work. Typically, if a dataset required a significant amount of alteration or correction or if it existed as hard-copy records only, it was excluded from this rapid assessment and treated as a data gap.

2.2.3 Assessing the Present—Projecting the Future

Assessment of the current status and future condition of the ecoregion's natural resources occurs by examining the relationships between a set of *conservation elements* and disturbance factors or *change agents.* Selected core conservation elements may be biotic elements (wildlife and plant species or assemblages) or abiotic factors (e.g., soils, water resources) of regional significance in major ecosystems and habitats of the ecoregion. REAs assess current status—or the existing state resulting from all past changes imposed on the prior historical condition—for each of the conservation elements. Because of the spatial nature of the REAs, describing status for various conservation elements and resource values requires that specific characteristics of that resource can be identified and mapped.

REAs also assess for each conservation element the potential for change from four change agents: fire, development, invasive species, and climate change. Potential for change predicts how status may change in the future in direction, magnitude, likelihood, and certainty. Assessment output products documenting potential-for-change demonstrate how current evidence of cumulative impacts may be projected into the future to identify potential trade-offs, alternatives, and mitigation strategies for BLM planning purposes. A development-related REA product of interest to BLM is the location of areas with high potential for renewable energy development—REA results contain current and potential development data layers that were merged with mapped distributions for the various conservation elements to identify the elements that may be affected by various renewable energy development forecasts.

In summary, REAs establish baseline ecological data to gauge the effect and effectiveness of future management actions. In this way, REAs provide a foundation for an adaptive management approach that enables implementation strategies to be adjusted for new information and changing conditions. REAs assess both the current and future scenarios by:

- identifying and answering important regional management questions;
- documenting key resource values, or conservation elements, with a focus on regionally-significant terrestrial habitats, aquatic habitats, and species of concern;
- describing current and projected future influences from four environmental change agents: climate change, wildfire, invasive species, and development;
- identifying and mapping opportunities for resource conservation, restoration, and development;
- identifying science gaps and data needs; and
- providing a baseline to evaluate and guide future management actions.

The regional scope of the Colorado Plateau REA, its numerous conservation elements and their interactions with change agents, produced a massive volume of results that can only be summarized within the constraints of a report of reasonable length. The body of this Colorado Plateau REA report contains highlights of major topics and case studies of key individual conservation elements. Appendices provide more detailed information on methods and models and specific results for all management questions, conservation elements, and change agents. The maps and data may also be examined in greater detail on the data portal (access at http://www.blm.gov/wo/st/en/prog/more/climatechange.html).

2.3 REA Process and Workflow

An Assessment Management Team (AMT) composed of BLM managers, partner agencies and technical specialists from within the ecoregion monitored the progress of each REA. At the beginning of the REA process, other federal and state agencies were invited as partners to the Assessment Management Team, including representatives of the Western Governors Association and Landscape Conservation Cooperatives. Members of the U.S. Geological Survey were retained as peer reviewers of REA products. The AMT guided the assessment and directed the work of the contractors.

REAs progress in two phases (Figure 2-1). In the first phase, the *pre-assessment*, participants refined the management questions, identified the data available for analysis, and agreed to methods and modeling approaches. The *assessment* phase followed agreement on the formal terms of a workplan; in the assessment phase, the contractors conducted the analyses and prepared the assessment report, maps, and supporting documents. The BLM, recognizing the importance of participation and input from agency partners and stakeholders, planned workshops near the end of each task for an interdisciplinary group to discuss and review the REA products. Workshop 4 marked the end of the pre-assessment phase, where a formal workplan was accepted to direct the rest of the REA process throughout the assessment phase. A peer review panel of USGS scientists monitored and commented on REA products at the completion of each task. For the review of mapped results, a private group was established on the data portal, Data Basin (Conservation Biology Institute, http://www.databasin.org/), where analyses and map results were posted weekly over a three month time period. Teams of reviewers viewed maps, component data layers, process models, and attachments, and entered review comments for products within their topical area of expertise. The AMT also produced a suggested outline for the final report and the AMT and USGS reviewers reviewed and commented on two drafts of the final report and appendices.

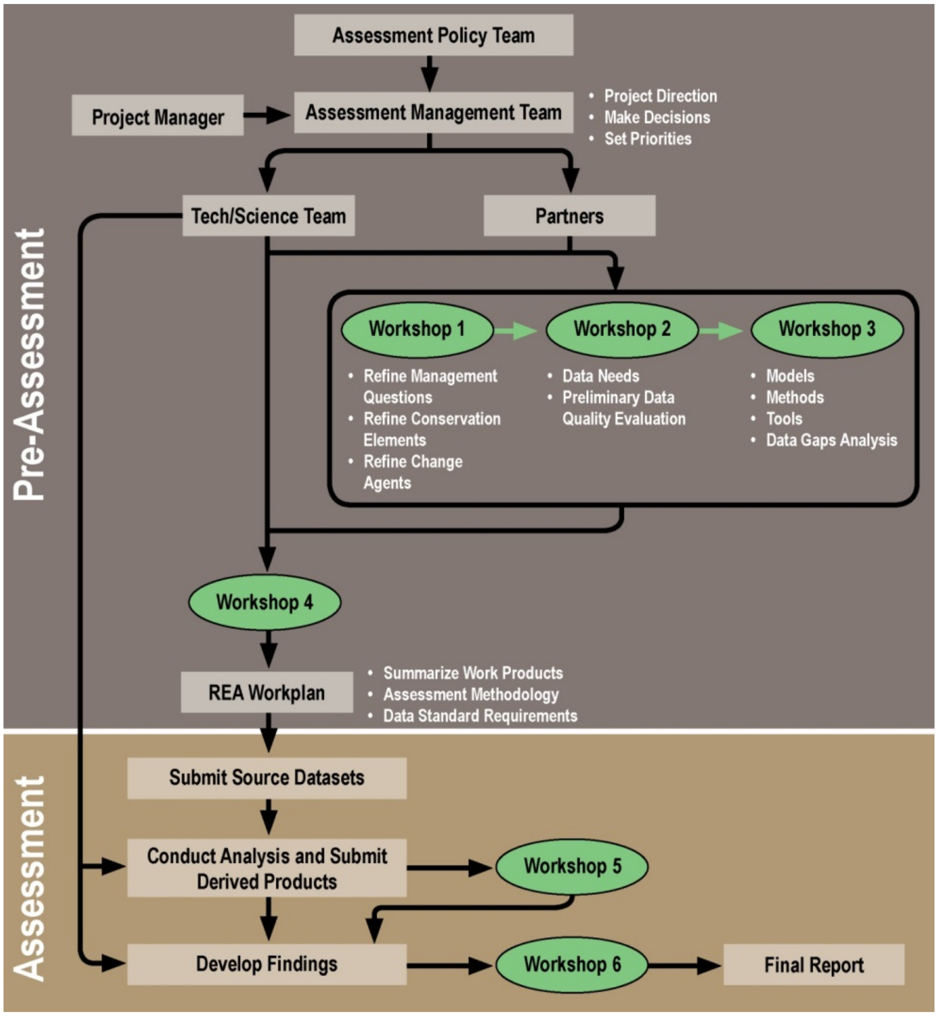


Figure 2-1. REA workflow divided into pre-assessment and assessment phases with regular workshops. Contents of each of the first three workshops listed beneath each workshop symbol in white text. Workshop 4 marked the preparation of a workplan with formal timelines, workflow, and review process. Workshops 5 and 6 provided forums for presenting analyses and products described in the final report.

2.4 REA Elements

2.4.1 Management Questions

BLM land managers provided a broad range of management questions to the REA to frame regional issues and data needs for land use planning, refining best management practices, and setting priorities for conservation, development, and restoration (Table 2-1). Management questions are the foundation and catalyst for the REAs in that they determine the scope of data requirements and analyses. The management questions developed for each ecoregion match the scale of the assessment because the issues captured by the questions are considered regionally significant. The management questions prepared for the Colorado Plateau REA refer to native and invasive flora and fauna, significant sites and ecological functions and services, and disturbance factors or change agents that affect present and future resource status.

Throughout the Pre-Assessment phase, BLM staff, REA contractors, and workshop participants weighed the time and resource requirements needed to address the full complement of management questions in the short time frame of the REA and in a manner that would have utility for BLM for future planning purposes. All participants suggested revisions, clarifications, and additions to the core list of management questions. USGS peer reviewers evaluated the questions with reference to the clarity of the language and the availability of data required to answer them. After the evaluation, 34 management questions in 10 topical classes (e.g., wildlife, invasive species, wildfire, and development) remained for the Colorado Plateau REA.

Table 2-1. Final AMT-Approved Colorado Plateau REA Management Questions. There are 34 management questions; labels out of order indicate deletion of various questions from redundancy or lack of adequate data. Results presented in the body of the report will denote appropriate management question. All management questions are presented with their results in the Appendices.

A. SOILS, BIOLOGICAL CRUSTS, AND FORAGE MANAGEMENT

MQ A1. Where are soils susceptible to wind and water erosion?

MQ A2. Where are sensitive soils (including saline, sodic, gypsiferous, shallow, and low water holding capacity)?

MQ A3. Which HMAs and allotments may experience significant effects from change agents, including climate change?

MQ A4. Where are soils that have potential to have cryptogamic soil crusts?

MQ A5. What/where is the potential for future change to the cryptogamic crusts?

MQ A6. Where are hotspots producing fugitive dust that may contribute to accelerated snow melt in the Colorado Plateau?

B. SURFACE AND GROUNDWATER MANAGEMENT QUESTIONS

MQ B1. Where are lotic and lentic surface waterbodies and livestock and wildlife watering tanks and artificial water bodies?

MQ B2. Where are perennial streams and stream reaches?

MQ B3. What are seasonal discharge maxima and minima for the Colorado River and major tributaries at gaging stations?

MQ B4. Where are the alluvial aquifers and their recharge areas (if known)?

MQ B6.Where are aquatic systems listed on 303d with degraded water quality or low macroinvertebrate diversity?

MQ B7. What is the location/distribution of these aquatic biodiversity sites?

C. ECOLOGICAL SYSTEMS MANAGEMENT QUESTIONS

MQ C1. Where are existing vegetative communities?

MQ C2. Where are vegetative communities vulnerable to change agents in the future?

MQ C3. What change agents have affected existing vegetation communities?

D. SPECIES CONSERVATION ELEMENT MANAGEMENT QUESTIONS

MQ D1. 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)?

MQ D5. What is the location/distribution of terrestrial biodiversity sites?

MQ D6. What aquatic and terrestrial species CEs and high biodiversity sites and movement corridors are vulnerable to change agents in the near term horizon, 2025 (development, fire, invasive species) and a long-term change horizon, 2060 (climate change)? Where are these species and sites located?

MQ D7. Where are HMAs located?

E. WILDFIRE MANAGEMENT QUESTIONS

MQ E1. Where are the areas that have been changed by wildfire between 1999 and 2009?

MQ E2. Where are the areas with potential to change from wildfire?

MQ E3. Where are the Fire Regime Condition Classifications?

MQ E4. Where is fire adverse to ecological communities, features, and resources of concern?

F. INVASIVE SPECIES MANAGEMENT QUESTIONS

MQ F1. Where are areas dominated by tamarisk and cheatgrass, and where are quagga and zebra mussel and Asiatic clam present?

MQ F2. Where are the areas of potential future encroachment from this invasive species?

G. FUTURE DEVELOPMENT MANAGEMENT QUESTIONS

MQ G1. Where are areas of planned development (e.g., plans of operation, urban growth, transmission corridors, governmental planning)?

MQ G2. 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

MQ H1. Where are high-use recreation sites, developments, roads, infrastructure or areas of intensive recreation use located (including boating)?

MQ H2. Where are areas of concentrated recreation travel (OHV and other travel) located?

MQ H3. Where are allotments and type of allotment?

I. AIR QUALITY MANAGEMENT QUESTIONS

MQ I3. Where are Class I PSD areas?

J. CLIMATE CHANGE MANAGEMENT QUESTIONS

MQ J1. 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?

MQ J3. Where are areas of species (conservation elements) distribution change between 2010 and 2060?

MQ J4. Where are aquatic/riparian areas with potential to change from climate change?

Although the management questions selected for the REAs were regionally significant, the scale of the data available to answer the questions often did not match the scale of the questions. That is, the management questions were conceived by BLM managers, but field office data were not available to the REA effort because of data consistency and level-of-effort issues. Often, publicly-available data gathered at the state or ecoregional scale did not match the detail necessary to answer some of the management questions. In many cases, data of the proper extent and detail to address the wildlife species and management issues found in Resource Management Plans at the field office level were not available at all. Although this was a limitation, it was also a revelation in that it revealed the limitations and gaps in the myriad data sources available to a project of this kind.

2.4.2 Conservation Elements

Coarse Filter Elements. The BLM planned that condition assessments within the REA framework follow a coarse-filter/fine-filter approach. A coarse filter approach employs elements such as vegetation communities, ecosystems, or land classes for planning and management across landscape- and regional-level management units (Noss 1987, Haufler et al. 1996, Desmet and Cowling 2004). Vegetation communities compose the habitat that supports the region's wildlife species. An assumption of the coarse filter approach is that blocks of naturally functioning communities will protect a diverse collection of flora and fauna. Within this paradigm, a top-down or "umbrella" approach is considered a more realistic and economical management system than one that attempts to address a host of species individually. The Nature Conservancy planned that its state-by-state coarse filter heritage network would preserve 85–90% of a state's species (Noss 1987). Noss (1987) noted, however, that coarse filter frameworks are typically based on dominance or homogeneity and that an optimal coarse filter would also incorporate food webs, species seasonal use, disturbance regimes, and hydrology. The REAs included some of these additional elements, such as seasonal use and disturbance regimes (e.g., for fire), where spatial information was available.

Characteristic vegetation communities of the Colorado Plateau, specifically the vegetation types (Table 2-2, Ecological Systems) defined in the Southwest Regional GAP Analysis Project (SWReGAP, Prior-Magee et al. 2007), represented the coarse-filter component of the REA. Because of the ecoregional scope of the REA, eight of the largest vegetation communities were selected that together cover 66.5% of the Colorado Plateau ecoregion. Vegetation management questions addressed the communities' current distribution, the effects of change agents on particular vegetation types, and areas where communities may be vulnerable to change agents in the future.

Table 2-2. ECOLOGICAL SYSTEMS	**% OF ECOREGION**
Colorado Plateau Pinyon-Juniper Woodland	20.4%
Inter-Mountain Basins Big Sagebrush Shrubland	9.1%
Inter-Mountain Basins Montane Sagebrush Steppe	3.9%
Colorado Plateau Mixed Bedrock Canyon and Tableland	10.6%
Rocky Mountain Gambel Oak-Mixed Montane Shrubland	4.5%
Colorado Plateau Pinyon-Juniper Shrubland	6.3%
Colorado Plateau Blackbrush-Mormon-Tea Shrubland	6.3%
Inter-Mountain Basins Mixed Salt Desert Scrub	5.4%
TOTAL AREA	66.5%

Although the coarse filter-fine filter approaches are meant to be complementary, limitations in species distribution datasets often force the use of coarse-filter surrogates to assess condition (Desmet and Cowling 2004). Because vegetative cover provides wildlife habitat, it can serve as a surrogate to estimate the status of species that are dependent on those habitats. As stated previously, status is the current condition of various conservation elements resulting from all stressors and changes imposed on a prior historical condition or benchmark reference condition. To express present status in terms of a gradient of condition, it is necessary to describe how far a conservation element has departed from a model of its minimally-disturbed reference condition and thus from a state of ecological or biological integrity (Frey 1977, Karr and Dudley 1981). Since spatial information for the presettlement distribution or abundances of various wildlife species is not available, the coarse filter vegetation communities are used to estimate changes over time. However, using vegetation communities to estimate historical reference condition requires a spatial dataset that is continuous across the entire ecoregion. While current vegetation conditions can be expressed using either the NatureServe national landcover dataset (version 2.7, 2009) or the LANDFIRE Existing Vegetation Type data (EVT; revised 2011, www.landfire.gov), the only dataset that maps (or models) reference condition over the entire region is the LANDFIRE Biophysical Settings (BpS) dataset. LANDFIRE BpS models the vegetation communities that may have been dominant on the landscape prior to Euro-American settlement. All vegetation communities are mapped using a combination of vegetation plot data, biophysical gradients, and 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 that would most likely have occurred under pre-settlement conditions.

The current distribution of existing vegetation communities were presented using both the NatureServe National Landcover and LANDFIRE existing vegetation (EVT) datasets because REA participants had definite preferences for one dataset or the other. However, to show change over time, the LANDFIRE BpS dataset was used for historic reference condition to compare with LANDFIRE EVT ("apples to apples"), an approach that minimized errors of comparison since both products were produced using similar input data and methods.

Fine Filter Elements. The fine filter approach is meant to complement the coarse filter by targeting species with requirements that will not be met through the broad brush of dominant vegetation communities—rare, threatened or endangered species, wildlife species of management interest, or those species that consistently use ecotones or multiple habitats on a diurnal or seasonal basis. Two variants of the fine filter approach are the focal species and landscape species approaches. Under the focal species approach, species are grouped according to susceptibility to regional threats or disturbances, and the species with the highest sensitivity needing the most comprehensive management response is selected for each threat category; the rationale for species selection is that if the most sensitive species' requirements are met, then so will the needs of the full complement of species dependent on the ecosystem in question (Lambeck 1997, Noss et al. 1999, Hess and King 2002).

Table 2-3. WILDLIFE SPECIES CONSERVATION ELEMENTS

Black-footed ferret (*Mustela nigripes*)
Desert bighorn sheep (*Ovis canadensis nelson*)
Gunnison's prairie dog (*Cynomys gunnisoni*)
Mountain lion (*Puma concolor*)
Mule Deer (*Odocoileus hemionus*)
Pronghorn antelope (*Antilocapra americana*)
White-tailed prairie dog (*Cynomys leucurus*)
American peregrine falcon (*Falco peregrinus*)
Burrowing owl (*Athene cunicularia*)
Ferruginous hawk (*Buteo regalis*)
Golden eagle (*Aquila chrysaetos*)
Greater sage-grouse (*Centrocercus urophasianus*)
Gunnison sage-grouse (*Centrocercus minimus*)
Mexican spotted owl (*Strix occidentalis lucida*)
Yellow-breasted chat (*Icteria virens*)
Colorado River cutthroat trout (*Oncorhynchus clarki pleuriticus*)
Flannelmouth sucker (*Catostomus latipinnis*)
Razorback sucker (*Xyrauchen texanus*)

Landscape species, on the other hand, are chosen according to a scoring system that incorporates multiple criteria (Sanderson et al. 2002, Coppolillo et al. 2004). The BLM suggested the landscape species approach of Coppolillo et al. (2004) for wildlife species selection for the Colorado Plateau REA. Using this approach, species are selected that capture a range of important attributes characterizing their environment, such as heterogeneity in habitat use, large home range area, vulnerability to anthropogenic disturbance, functional contributions to the ecological system, and relative socio-economic importance (Coppolillo et al. 2004). Species are ranked by aggregate scores for each of these attributes and selected based on the highest aggregate score and minimum overlap in the major vegetation communities (Ecological Systems) used, until all Ecological Systems are accounted for. A cross check is then made to ensure that all change agent threats are accounted for as well. A set of 25–30 species were selected and scored from the State Wildlife Action Plan lists in addition to the core species identified by the BLM. The screening process resulted in ten wildlife species with the highest scores representing the minimum overlap in habitats. Those species identified by the BLM that did not score high enough to make it on the final landscape species list were retained and included in the assessment.

The Statement of Work requested an objective screening process to select wildlife species conservation elements, or landscape species. It was also apparent that to provide the best representation of status and condition at the ecoregional level with respect to habitat alteration, displacement, and human stressors, it was important to select species that were vulnerable to the selected change agents. Although the group at Workshop 1 agreed to a species selection process based on Coppolillo et al. (2004) that produced an initial list of landscape species, REA participants continued to suggest additional wildlife species of unrepresented taxa or habitats throughout Tasks 1, 2, and 3 of the pre-assessment phase (Table 2-3).

In addition to the list of wildlife landscape species, the selection of fine filter elements also included 1) special status plant or animal species (sensitive, threatened and endangered) enumerated by 5th level hydrologic unit and mapped as species richness or species diversity hotspots and 2) a range of terrestrial and aquatic sites of conservation concern (Table 2-4) and ecosystem functions and services (Table 2-5).

The terrestrial and aquatic sites of conservation concern range from Nature Conservancy portfolio sites, National Parks, Wildlife Refuges, National Conservation Areas, and wilderness areas, all of which have various levels of protection (Table 2-4). Both current and future threats were assessed for these sites. Mapping the sites with surrounding ownership status will provide opportunities for interagency cooperation in management. Some of these sites may lose the function or features for which they were designated as a result of interactions among climate change and other change agents such as fire and invasive species. Are there cross-jurisdictional opportunities to create an additional buffer of protection around sites of conservation concern? Establish corridors between sites? Plan for future refugia from climate change? Are diverse ecosystems at all elevations well-represented? These questions can be addressed by the BLM through ecoregional direction (see Chapter 1).

The list of ecological functions and services focuses on aquatic features such as springs, seeps, and riparian areas, recognizing the importance of water availability in an arid environment (Table 2-5); REA participants added the terrestrial function of soil stability to the list to of ecosystem functions and services because of concerns over soil erosion, dust on snow, and the sustainability (and possible loss) of biological soil crust.

Table 2-4. SITES OF CONSERVATION CONCERN

Terrestrial Sites

- TNC portfolio sites
- Important bird areas (Audubon)
- Historic and Nationally Designated Trails
- Wilderness Areas
- Wilderness Study Areas
- Historic Districts
- National Wildlife Refuges
- Monuments
- National and State Parks
- National Conservation Areas
- BLM Areas of Critical Environmental Concern
- Forest Service Research Natural Areas
- State Wildlife Management Areas
- Wild and Scenic Rivers
- Designated Recreation Management Areas

Aquatic Sites

- TNC portfolio sites

Table 2-5. ECOSYSTEM FUNCTIONS AND SERVICES

Terrestrial Functions:

- Soil stability
- Biological soil crust

Surface and Subsurface Water Availability:

- Aquatic systems (streams, lakes, ponds)
- Springs/seeps/wetlands
- Riparian areas
- High quality and impaired waters
- Groundwater aquifers

2.4.3 Change Agents

An assessment of the status of conservation elements must be conducted with reference to both natural and anthropogenic disturbance factors. The status or condition of various conservation elements cannot be discussed without examining the risks that these resources experience from a collection of regional disturbances or change agents. Human disturbances represent the change agents of interest in the REA process (Table 2-6). Although the same change agent may threaten one organism and benefit another, the change agents selected for the REAs typically affect habitat negatively and degrade the productivity and sustainability of the selected conservation elements.

Many effects of change agents are directly apparent, representing changes in land use during development, agriculture, resource extraction, such as logging and mining, and energy development. While normally not as destructive as urbanization, various forms of recreation are expanding throughout the region each with a unique set of impacts, from increased hiking and mountain biking to OHV use, which can result in habitat fragmentation, connectivity loss, soil erosion, and wildlife disturbance (Papouchis et al. 2001, Belnap 1995, Brooks and Lair 2005, Ouren et al. 2007, Schwinning et al. 2008).

Other effects are more diffuse, such as the changes in plant species dominance created by prolonged grazing (Belsky and Gelbard 2000, Krueper et al. 2003, Miller et al. 2011) or the synergy of livestock grazing, invasive species introduction, and fire (D'Antonio and Vitousek 1992, Brooks et al. 2004). Fire is a natural disturbance agent, but when it deviates from expected frequencies, it can be considered a form of anthropogenic change agent. Fire often deviates from its characteristic regime, through fire suppression, increased ignition frequencies, and changes in characteristic fuels and fuel loads (D'Antonio and Vitousek 1992, Keane et al. 2002, Brooks et al. 2004). Perhaps the most overarching and profound change agent of all is climate change. As indicated by recent evidence and robust predictive models, climate change has the potential to change the landscape over the next 50 years in fundamental ways with direct impacts on natural systems while increasing the influence of many of the other change agents. For example, projected climate change influences fire regimes, alters invasive plant species competition, affects hydrologic regimes and water yields, and changes basic soil properties (Seager et al. 2007, Schwinning et al. 2008, Munson et al. 2011).

Table 2-6. CHANGE AGENTS

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

2.4.4 Index of Ecological Integrity

The concept of ecological integrity is complex and a great deal has been written about it in the literature (Angermeier and Karr 1994, 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 communities. As Karr and Dudley (1981) described it—ecological integrity is the sum of all physical, chemical, and biological integrity. Karr and Chu (1995) later defined 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.

A number of strategies have been devised to conduct assessments of ecological condition, from data-driven indices of biological integrity or IBIs, to more qualitative, conservation guidance approaches such as those discussed by Parrish et al. (2003) and Unnasch et al. (2008). Approaches such as these differ in rigor and defensibility, and they also differ in terms of their potential application in products such as Rapid Ecoregional Assessments. Indices of biotic integrity (IBIs), as developed over the last 3 decades for aquatic ecosystems, use systematically-collected species abundance data to develop metrics representing taxonomic richness, trophic categories, or sensitivity to disturbance. Candidate metrics are screened for responsiveness to disturbance, low variability, and lack of redundancy (Hughes et al. 1998, Mebane et al. 2003, Whittier et al. 2007). Metric values at minimally- or least-disturbed sites serve as a reference model against which to compare indicator metric values at disturbed sites (Hughes et al. 1986, Hughes 1995, Whittier et al. 2007). Few indices of *terrestrial* ecological integrity have been developed using the approach described above. Development of terrestrial integrity indices present even greater challenges than aquatic indices of biointegrity, and terrestrial applications of indices of biotic integrity are limited in the scientific literature (O'Connell et al. 1998, Bradford et al. 1998, Bryce et al. 2002, Bryce 2006, Mattson and Angermeier 2007).

The development of data-driven indicators of ecological integrity is beyond the scope of the REA process because it would require a major research effort. REAs are defined in the Statement of Work as "assessments only, evaluating status and potential changes in status for selected core conservation elements." Thus, the approach to regional ecological integrity within the REAs represents an early iteration of a process that will continue to evolve. Concurrently with these first REAs, BLM and agency partners have considered various more qualitative approaches to characterize landscape-level ecological integrity or condition based on existing geospatial data.

For this REA, the group agreed to emphasize the mapping of ecological condition by focusing on *intactness*, an attribute that could be defensibly supported by existing geospatial datasets and reasonably tracked through time. No place on Earth remains unaffected by modern humans (Vitousek 1997), but some regions have been more directly and severely affected than others. 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.

> Intactness is a quantifiable estimate of naturalness measured on a gradient of anthropogenic influence and based on available spatial data.

The origin of the intactness concept can be traced to the concept of naturalness. Machado (2004) provides a thorough review of the history and use of the term "naturalness" and how it has been applied to conservation planning throughout the world. There has been a mostly philosophical and semantic debate regarding the concept of naturalness as it pertains to a conservation value. Less confusion and debate surrounds the concept as it applies to its use as a parameter or state descriptor of ecosystems (Grumbine 1994). The term "landscape intactness", which is used as a quantifiable state descriptor, has been largely applied to forested landscapes (Lee et al. 2002, Heilman et al. 2002, Strittholt et al. 2006, Potapov et al. 2008), but many of the same principles apply to any natural landscape. Choosing the canyons and tablelands ecosystem as an example specific to the Colorado Plateau, the most intact canyonlands are those with the least influence from anthropogenic change agents. Representative areas of canyons and tablelands may be placed along a gradient of intactness (or conversely a gradient of disturbance) with sites that are experiencing increasing levels of disturbance considered to have lower intactness. Even within a group of protected areas, a wilderness area with no known disturbances will have higher intactness than another protected area that retains evidence or scars of the grazing or mining that occurred before it was established. Across a region, intactness levels decrease with the increasing intensity and extent of various land uses—grazing, rowcrop agriculture, energy development, and urbanization; the lowest intactness levels occur in areas completely converted from their original character.

Thus, intactness may be mapped as a quantifiable estimate of naturalness according to the level of anthropogenic influence based on available spatial data. Intactness considers an assemblage of spatially explicit indicators that helps define the condition of the natural landscape. Different species may possess different tolerances to these conditions, but natural assemblages of species and natural patterns and processes are increasingly compromised as human influences intensify. For this REA, terrestrial and aquatic intactness models were created for the entire ecoregion (see Methods, Chapter 3) and they served as the foundation against which conservation element status was assessed based on current condition as well as future projections.

Presence or absence of particular species, species richness, or species rarity did not factor into any metric of integrity. First and foremost, high species richness or concentration of rare or endemic species is not indicative of high ecological integrity. Areas with high species endemism or high species richness may be important from a conservation or management perspective, but regions with these species are not necessarily better from an ecological integrity perspective. Species do not naturally arrange themselves equally across the landscape even under pristine conditions. Natural concentrations of species are driven by many factors. For example, vertebrate species richness is often higher at middle elevations (McCain 2003, McCain 2007) or in warmer river and stream systems (Mebane et al. 2003, Hughes et al. 2004). Species numbers typically increase with moderate disturbance (Odum et al. 1979, Odum 1985). Ecosystem condition can sometimes even decline as species diversity (even native species diversity) increases (Scott and Helfman 2001). Areas with high species endemism or high species richness should be evaluated separately and they were for this REA.

2.5 REA Assumptions and Limitations

As previously stated, the REA was not intended to be a research project; however, at numerous times throughout the project, that is what was needed in order to generate a useful assessment. There was inadequate time and funding to allow full development of every topic identified by the assessment team or outside reviewers, but several major areas were explored that could be classified as work above and beyond a typical rapid assessment. Of all the issues and management questions addressed, a significant amount of research time was dedicated to the following topics, which resulted in a more useful set of products:

- using logic models to help aggregate and synthesize large concepts using numerous, disparate data inputs
- refining the concept of intactness and how it could be used to assess current and future status in a repeatable and scientifically defensible fashion
- instituting the 4km resolution as one of the primary reporting units
- including natural habitat fragmentation as an important metric for assessing intactness
- modification and improvement of fire modeling
- utilization of both LANDFIRE EVT v 1.1 and NatureServe Landcover v 27 in the assessment
- integration of STATSGO and SURRGO soils data in assessing a variety of soils management questions
- inclusion of MAPSS in the climate change component of the project to extend our understanding of vegetation responses to predicted changes in temperature and precipitation
- inclusion of seasonality in climate change projections

The REA was also not a specific planning exercise, which typically requires higher levels of project definition with measurable goals and objectives against which a rigorous analytical treatment is devised and carried out. The REA took on a much broader approach focusing more on how many topics could be addressed at once rather than an in-depth exploration of a smaller subset of the issues. It was the intent of the BLM to use the REA to obtain a regional context with analyses that would help them later prioritize or focus on particular areas of need or special interest in a series of step-down efforts.

With any spatial analysis, especially for a large geographic area such as an ecoregion, the most fundamental limitation is the availability and quality of the spatial data. For this REA, even after exhaustive searches and time-intensive data compilations, acquiring and assembling useful spatial datasets to address specific issues or management questions often proved challenging. The inability to acquire datasets such as specific point locations for species, OHV tracks, recreation areas, and grazing history and current intensity either limited our ability to address specific questions or prevented us from meaningfully addressing them at all. For most issues, the scale/resolution of acquired datasets allowed for a reliable coarse level assessment, but the datasets were generally insufficient to allow for site-specific management applications (e.g. restoration of invasive grass patches). However, for the purposes of a regional ecoregional assessment, the datasets assembled and analyzed resulted in very useful contextual information on top of which local analyses and management prescriptions could be explored and implemented.

Spatial data accuracy (geometry and attribution) was highly variable for different themes and often between subregions (e.g. states) for the same theme. Even for the most authoritative datasets, errors are commonplace. For example, the National Hydrography Dataset stream flow status attribute currently has a high rate of error in arid ecoregions. In a recent stream survey (EMAP-West 2000–2004) conducted by the Environmental Protection Agency (Stoddard et al. 2005), many streams identified as perennial were in fact not perennial when visited in the field. Both LANDFIRE EVT v1.1 and NatureServe Landcover v2.7 are recognized as authoritative, yet significant differences occur between them. In reality, they both possess errors, meaning that more detailed vegetation data are needed to carry out site-specific planning and management. With data inputs of variable quality, analyzing complex ecological systems, and trying to forecast into the future, spatial modeling possesses a fairly high degree of uncertainty. Initially, data confidence maps were planned to accompany each result to help the user identify areas of uncertainty. This proved too difficult to do except for portions of the climate change modeling. However, the review process helped our team and the external reviewers to identify problem areas; in addition, qualitative levels of confidence in each data source and in model results have been included in tables in Appendix E.

Throughout the project, the data portal Data Basin (www.databasin.org) was used to solicit regular feedback from outside reviewers on the data inputs, analytical approaches conducted, and final results through a private working group created in the online system. Customized commenting tools helped reviewers pose spatially explicit or general comments and questions. Having all of the spatial datasets and attached processing models and notes easily available via the Internet, Data Basin enhanced numerous webinars for subsets of reviewers to explore specific topical areas or problem areas. Although generating batches of mapped results on a regular schedule for posting on Data Basin created more work than the original scope of work outlined, Data Basin proved to be an extremely valuable tool for managing the review process, improving the assessment in numerous ways through an improved suite of products and better overall understanding.

2.6 References Cited

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

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

Angermeier, P.L., and J.R. Karr. 1994. Biological integrity versus biological diversity as policy directives: Protecting biotic resources. *BioScience* 44(10):690–697.

Belnap, J. 1995. Surface disturbances: Their role in accelerating desertification. *Environmental Monitoring and Assessment* 37:39–57.

Belsky, A.J., and J.L. Gelbard. 2000. Livestock grazing and weed invasions in the arid west*.* Oregon Natural Desert Association. Bend, Oregon.

Bradford, D.F., S.E. Franson, A.C. Neale, D.T. Heggem, G.R. Miller, and G.E.Canterbury. 1998. Bird species assemblages as indicators of biological integrity in Great Basin rangeland. *Environmental Monitoring and Assessment* 49:1–22.

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

Brooks, M.L., and B. Lair. 2005. Ecological effects of vehicular routes in a desert ecosystem. Report prepared for the United States Geological Survey, Recoverability and Vunerabilty of Desert Ecosystems Program, U.S. Geological Survey, Las Vegas, Nevada. 23 pp.

Bryce, S.A. 2006. Development of a bird integrity index: Measuring avian response to disturbance in the Blue Mountains of Oregon, USA. *Environmental Management* 38(3):470–486.

Bryce, S.A., R.M. Hughes, and P.R. Kaufmann. 2002. Development of a bird integrity index: Using bird assemblages as indicators of riparian condition. *Environmental Management* 30(2):294–310.

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

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

Desmet, P., and R. Cowling. 2004. Using the species-area relationship to set baseline targets for conservation. *Ecology and Society* 9(2): 11. http://www.ecologyandsociety.org/vol9/iss2/art11

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

Grumbine, R.E. 1994. What is ecosystem management? *Conservation Biology* 8:27–38.

Haufler, J.B., C.A. Mehl, and G.J. Roloff. 1996. Using a coarse-filter approach with species assessment for ecosystem management. *Wildlife Society Bulletin* 24(2):200–208.

Heilman, G.E., J.R. Strittholt, N.C. Slosser, and D.A. DellaSala. 2002. Forest fragmentation of the conterminous United States: Assessing forest intactness through road density and spatial characteristics. *Bioscience* 52(5): 411–422.

Hess, G.R., and T.J. King. 2002. Planning open spaces for wildlife: Selecting focal species using a Delphi survey approach. *Landscape and Urban Planning* 58(1):25–40.

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

Hughes, R.M., S. Howlin, and P.R. Kaufmann. 2004. A biointegrity index (IBI) for coldwater streams of western Oregon and Washington. *Transactions of the American Fisheries Society* 133:1497–1515.

Hughes, R.M., P.R. Kaufmann, A.T. Herlihy, T.M. Kincaid, L. Reynolds, and D.P. Larsen. 1998. A process for developing and evaluating indices of fish assemblage integrity. *Canadian Journal of Fisheries and Aquatic Sciences* 55: 1618–1631.

Hughes, R.M., D.P. Larsen, and J.M. Omernik. 1986. Regional reference sites: A method for assessing stream potentials. *Environmental Management* 10(5):629–635.

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

Karr, J.R., and E.W. Chu. 1995. Ecological integrity: Reclaiming lost connections. Pages 34–48 *in* L. Westra and J. Lemons (eds.), Perspectives in Ecological Integrity, Kluwer Academic, Dordrecht, Netherlands.

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

Krueper, D. J. Bart, and T.D. Rich. 2003. Response of vegetation and breeding birds to the removal of cattle on the San Pedro River, Arizona, USA. *Conservation Biology* 17(2):607–615.

Lambeck, R.J. 1997. Focal species: A multi-species umbrella for nature conservation. *Conservation Biology* 11(4):849–856.

Lee, P., D. Aksenov, L. Laestadius, R. Nogueron, and W. Smith. 2002. Canada's large intact forest landscapes. Global Forest Watch Canada, Edmonton, Canada. http://www.globalforestwatch.org/english/canada/pdf/Canada_LIFL-Text_Section.pdf

Machado, A. 2004. An index of naturalness. *Journal of Nature Conservation* 12(2004):95–110.

Mattson, K.M., and P.L. Angermeier. 2007. Integrating human impacts and ecological integrity into a risk-based protocol for conservation planning. *Environmental Management* 39:125–138.

McCain, C.M. 2003. North American desert rodents: A test of the mid-domain effect in species richness. *Journal of Mammology* 84:967–980.

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

Mebane, C.A., T.R. Maret, and R.M. Hughes. 2003. An index of biological integrity (IBI) for Pacific Northwest rivers. *Transactions of the American Fisheries Society* 132:239–261.

Miller, M.E., R.T. Belote, M.A. Bowker, and S.L. Garman. 2011. Alternative states of a semiarid grassland ecosystem: Implications for ecosystem services. *Ecosphere* 2(5):art55. doi:10.1890/ES11-00027.1

Munson, S., J. Belnap, C. Schelz, M. Moran, T. Carolin. 2011. On the brink of change: Plant responses to climate on the Colorado Plateau. *Ecosphere* 2(6), Article 68. Available: digitalcommons.usu.edu/cgi/ viewcontent.cgi?article=1340.

Noss, R.F. 1987. From plant communities to landscapes in conservation inventories: A look at The Nature Conservancy (USA). *Biological Conservation* 41:11–37.

Noss, R.F., J. Strittholt, K. Vance-Borland, C. Carroll, and P. Frost. 1999. Conservation plan for the Klamath-Siskiyou ecoregion. *Natural Areas Journal* 19(4):392–411.

O'Connell, T.J., L.E. Jackson, and R.P. Brooks. 1998. A bird community index of biotic integrity for the mid-Atlantic highlands. *Environmental Monitoring and Assessment* 51(1-2):145–156.

Odum, E.P., J.T. Finn, and E.J. Franz. 1979. Perturbation theory and the subsidy-stress gradient. *BioScience* 29(6):349–352.

Odum, E.P. 1985. Trends expected in stressed ecosystems. *BioScience* 35(7):419–422.

Ouren, D.S., C. Haas, C.P. Melcher, S.C. Stewart, P.D. Ponds, N.R. Sexton, L. Burris, T. Fancher, and Z. Bowen. 2007. Environmental effects of off-highway vehicles on Bureau of Land Management lands: A literature synthesis, annotated bibliographies, extensive bibliographies, and internet resources. U.S. Geological Survey, Open-File Report 2007-1353, U.S. Geological Survey, Reston, Virginia. 225 p.

Papouchis, C.M., F.J. Singer, and W.B. Sloan. 2001. Responses of desert bighorn sheep to increased human recreation. *The Journal of Wildlife Management* 65(3): 573–582.

Parrish, J.E., D.P. Braun, and R.S. Unnasch. 2003. Are we conserving what we say we are? Measuring ecological integrity within protected areas. *BioScience* 53:851–860.

Pimentel, D., L. Lach, R. Zuniga, and D. Morrison. 2000. Environmental and economic costs of nonindigenous species in the United States. *Bioscience* 50(1):53–65.

Potapov P., A. Yaroshenko, S. Turubanova, M. Dubinin, L. Laestadius, C. Thies, D. Aksenov, A. Egorov, Y. Yesipova, L. Glushkov, M. Karpachevskiy, A. Kostikova, A. Manisha, E. Tsybikova, and I. Zhuravleva. 2008. Mapping the world's intact forest landscapes by remote sensing. *Ecology and Society*, 13(2): http://www.ecologyandsociety.org/vol13/iss2/art51/

Prior-Magee, J.S., K.G. Boykin, D.F. Bradford, W.G. Kepner, J.H. Lowry, D.L. Schrupp, K.A. Thomas, and B.C. Thompson (Eds). 2007. Southwest Regional Gap Analysis Project final report. U.S. Geological Survey, Gap Analysis Program, Moscow, Idaho.

Sanderson, E.W., K.H. Redford, A. Vedder, P.B. Coppolillo, and S.E. Ward. 2002. A conceptual model for conservation planning based on landscape species requirements. *Landscape and Urban Planning* 58(1):41–56.

Scott, M.C., and G.S. Helfman. 2001. Native invasions, homogenization, and the mis-measure of integrity of fish assemblages. *Fisheries* 26(11):6–15.

Schwinning, S., J. Belnap, D.R. Bowling, and J.R. Ehleringer. 2008. Sensitivity of the Colorado Plateau to change: Climate, ecosystems, and society. *Ecology and Society* **13**(2): 28. http://www.ecologyandsociety.org/vol13/iss2/art28/

Seager, R., M. Ting, I. Held, Y. Kushnir, J. Lu, G. Vecchi, H. Huang, N. Harnik, A. Leetmann, N. Lau, C, Li, J. Velez, and N. Naik. 2007. Model projections of an imminent transition to a more arid climate in southwestern North America. *Science* 316:1181–1184.

Stoddard, J.L., D.V. Peck, S.G. Paulsen, J. Van Sickle, C.P. Hawkins, A.T. Herlihy, R.M. Hughes, P.R. Kaufmann, D.P. Larsen, G. Lomnicky, A.R. Olsen, S.A. Peterson, P.L. Ringold, and T.R. Whittier. 2005a. An ecological assessment of western streams and rivers. EPA 620/R-05/005, U.S. Environmental Protection Agency, Washington, DC.

Strittholt, J.R., R. Nogueron, M. Alvarez, and J. Bergquist. 2006. Mapping undisturbed landscapes in Alaska. World Resources Institute, Washington, DC. http://www.globalforestwatch.org/english/us/pdf/GFW-Alaska_report_final.pdf

Unnasch, R.S., D.P. Braun, P.J. Comer, and G.E. Eckert. 2008. The ecological integrity assessment framework: A framework for assessing the ecological integrity of biological and ecological resources of the National Park system, version1.0. Unpublished report to the National Park Service.

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

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

Photo: Wild horse near Dugway, Utah. BLM Utah

III. SUMMARY OF METHODOLOGY

3.1 Data Management

The majority of data processed for this REA were handled according to the BLM Data Management Plan (DMP), except in specific cases where guidance was not sufficiently detailed, not feasible according to schedule and budget constraints, or where specific characteristics of the data or processing required a special approach. In nearly all cases, additional guidance was provided by the NOC Data Management Team and the AMT to address these specific cases. In particular, the data processing workflow specified by the DMP required substantial modification during this REA. While it was originally intended by the REA workflow that data would be acquired, fully evaluated, and approved by the AMT prior to the modeling phases, this proved infeasible, and it resulted in the early acquisition and evaluation of many datasets that subsequently were not used for modeling. Midway through the REA, a workflow more tightly coupled to the modeling process was adopted, which included acquisition and pre-evaluation of datasets as part of the modeling effort. As such, dataset collection activities were targeted to very specific themes and pre-screened to determine appropriateness for a particular analysis. Additional datasets were identified during workshops and the iterative review process managed using the data portal, Data Basin (www.databasin.org). Thus, although initially over 400 datasets were collected and considered for the REA, 217 datasets were ultimately used in analyses for the Colorado Plateau. After source datasets were successfully used in modeling efforts, they were evaluated according to 11 criteria as specified in the DMP; these included criteria such as non-duplication, spatial accuracy, and thematic accuracy. Data were scored using narrative descriptions for each criterion to highlight potential data quality issues; earlier efforts to use a numeric scoring system proved too time-consuming and less informative.

The analytical extent for this ecoregion was the outer boundary of all 5th level hydrologic units (HUCs) that intersect the Environmental Protection Agency's (EPA's) Level III Ecoregion boundary of the Colorado Plateau (CEC 1997, Figure 3-1). All datasets were clipped to this extent and re-projected to USA Contiguous Albers Equal Area Projection (USGS Version) as specified by the DMP. Prior to delivery to BLM, all spatial data were standardized into ArcGIS File Geodatabase Feature Class and ArcGRID file formats. This included conversion of quasi-spatial datasets (e.g., spreadsheets with coordinates, print maps) into these formats through format conversion and digitization. Digitization of published materials was used as a last resort for essential datasets when original spatial data could not otherwise be obtained.

Climate data were developed at a 4km resolution from the native 15km resolution for the western U.S., and processed primarily in NetCDF format due to the temporal nature of such data. (NetCDF is a file format ideal for climate data because it can accommodate multiple dimensions in a single file.) The outer extent of all 4km grid cells within the ecoregion/5th level watershed boundary was used as the analytical extent for these data. Derived results, such as annual average temperature for 2015–2030, were extracted into ArcGRID format.

All datasets required development of FGDC compliant metadata per BLM specifications. In many cases, full FGDC metadata were not available for all original source datasets, and often available information was insufficiently detailed to achieve all BLM desired metadata elements. The Dynamac team exerted considerable effort to populate missing metadata elements. The substantial effort involved in achieving full compliance with FGDC and BLM metadata standards deterred delivery of any datasets to BLM other than those used directly in the modeling and analysis process; thus, several datasets of potential interest but no direct application in this REA were excluded.

Most datasets were processed using ArcGIS ModelBuilder and python scripts delivered as ArcGIS tools, per BLM requirements. Many of these models were developed in such a way as to permit other users beyond this

REA to modify the input and processing methods and rerun the tools. Specifically, the terrestrial and aquatic intactness models are likely to be of high value to end-users. A few non-ArcGIS analysis tools were used to generate some of the results developed in this REA, including MaxEnt and FRAGSTATS.

A number of data-related issues were encountered during this REA:

- some existing thematic data were not available for use by the Dynamac team due to proprietary restrictions (e.g., Natural Heritage data, some oil and gas data);
- data may have existed in digital form for some published materials (e.g., maps presented in a report), but data was not always obtainable in a timely fashion from authors. In specific cases, this required that the Dynamac team digitize these data directly from the published materials;
- some data specifically developed by the BLM and other agencies as part of their planning processes, for example BLM Field Office data, were not available to the Dynamac team. BLM had asked that field office data not be gathered that was not already in national datasets because of consistency, data standards and level of effort;
- versioning of datasets for continually updated themes (e.g., BLM renewable energy projects datasets) presented challenges by becoming available late in the REA or requiring rectification as new versions became available;
- many source datasets were developed at the state level (e.g., wildlife habitat), and presented numerous challenges when combining these at the ecoregion level, such as edge-matching between states, thematic resolution, spatial scale, attribution, and data standards.

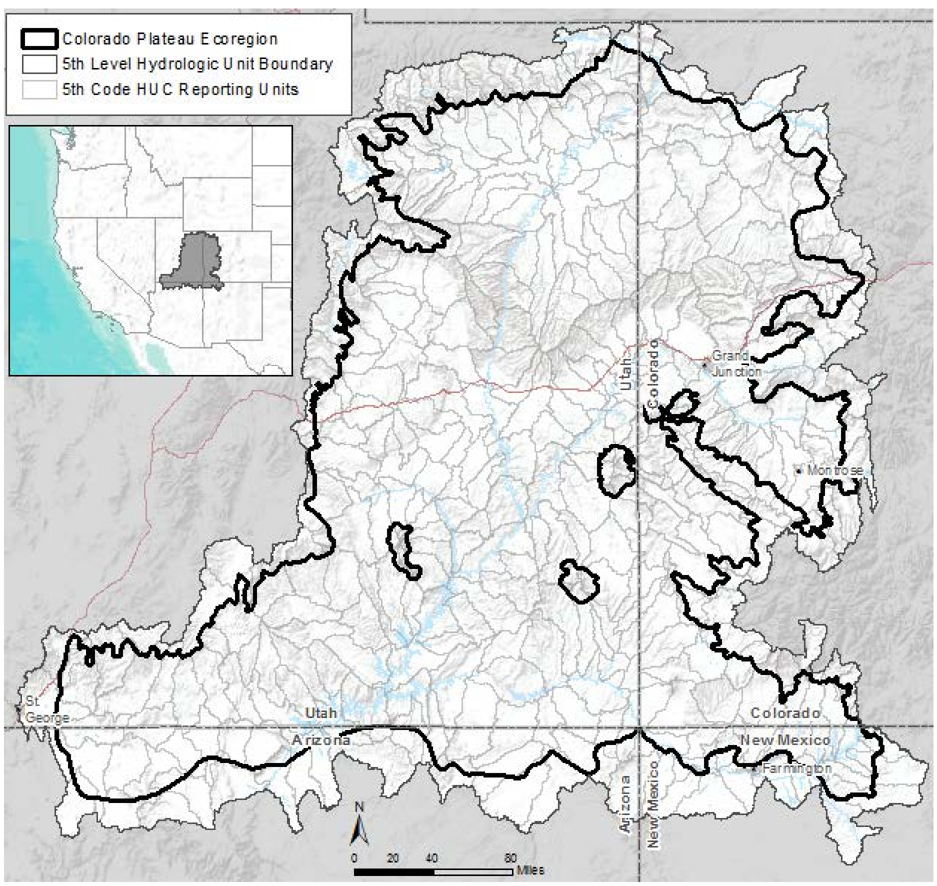


Figure 3-1. Map of the Colorado Plateau ecoregion showing EPA Level III ecoregion boundary, hydrologic unit boundaries, and analytical extent of buffer. The three holes in the coverage are mountainous outliers of adjacent level III ecoregions (Southern Rockies and Wasatch and Uinta Mountains).

3.2 Models, Methods, and Tools

Throughout the REA process, numerous types of models were developed and analysis tools used to address the various management questions and overarching issues of interest. This section discusses the development of ecological conceptual models, process and logic models, and habitat fragmentation, connectivity, fire, and climate change modeling.

3.2.1 Conceptual Models

Conceptual models graphically depict the interactions between a conservation element, the biophysical attributes of its environment, and the change agents that drive ecosystem character. The boxes and arrows that make up the conceptual model represent the state of knowledge about the subject and its relationships to these attributes (Figure 3-2). Conceptual models are also supported and referenced by scientific literature. REA conceptual models were developed at three levels. At the ecoregion level, an overarching model was developed that outlined the interactions of the major ecological features, processes, and change agents. Since change agents are a major focus of the REAs, a comprehensive change agent conceptual model was also produced. Finally, individual conceptual models were created for each conservation element with particular attention paid to the potential impacts from the various change agents. Some conceptual models were adapted from Miller (2005) and Miller et al. (2010).

Conceptual models for conservation elements were standardized by including all change agents (yellow boxes) and natural drivers (cyan boxes) with close attention paid to those attributes and indicators that could be used to help assess current and future status. Specifics regarding some of the components (when known) are presented in blue text. Arrows represent relationships between the various change agents and natural drivers acting on the conservation element from the standpoint of the natural community or habitat as well as on one or more individual species. Specific information about the flows between components is provided in orange text. It is important to note that not all of the relationships identified in the conceptual models lend themselves well to measurement or monitoring because adequate spatial data does not exist in many cases or there is a lack in scientific knowledge to intelligently quantify a particular indicator. In spite of these shortcomings, all important components are included as they aid in our general understanding of complex interactions.

Unlike many published conceptual models, thicknesses of the arrows in these models **DO NOT** represent degree of importance. Rather, bold lines represent those factors that are tracked or modeled to varying degrees of certainty throughout the REA analysis. The conceptual models as presented in this report, therefore, provide information in several ways—they provide information on: (1) ecological interactions; (2) what spatial data are available to track changes over time; and (3) where there are spatial data gaps.

For example, in the conceptual model for Inter-Mountain Basins Montane Sagebrush Steppe (Figure 3-2), there are six primary natural drivers (cyan boxes) for this ecological system including topography, soil characteristics, precipitation, temperature, insects and disease, and animal herbivory (details in blue text in conceptual model, NatureServe 2009, Tart 1996, LANDFIRE 2007). Mountain sagebrush (*Artemsia tridentata* ssp*. vaseyana*) is the signature species for this ecosystem and it is affected by a number of factors. Climatic events such as periods of excessive moisture (Sturges et al. 1984) as well as droughts impact this and related species (Anderson and Inouye 2001). The Aroga moth (*Aroga websteri*) and leaf beetles *(Trirhabda pilosa*) can cause significant sagebrush mortality (Pringle 1960, Gates 1964). Mechanical removal or burning of this community to improve grazing conditions can have negative ecological consequences (Harniss and Murray 1973, Blaisdell et al. 1982, Hormay 1970). Mechanical removal or burning of this community can also promote invasive grasses, altering the system even further.

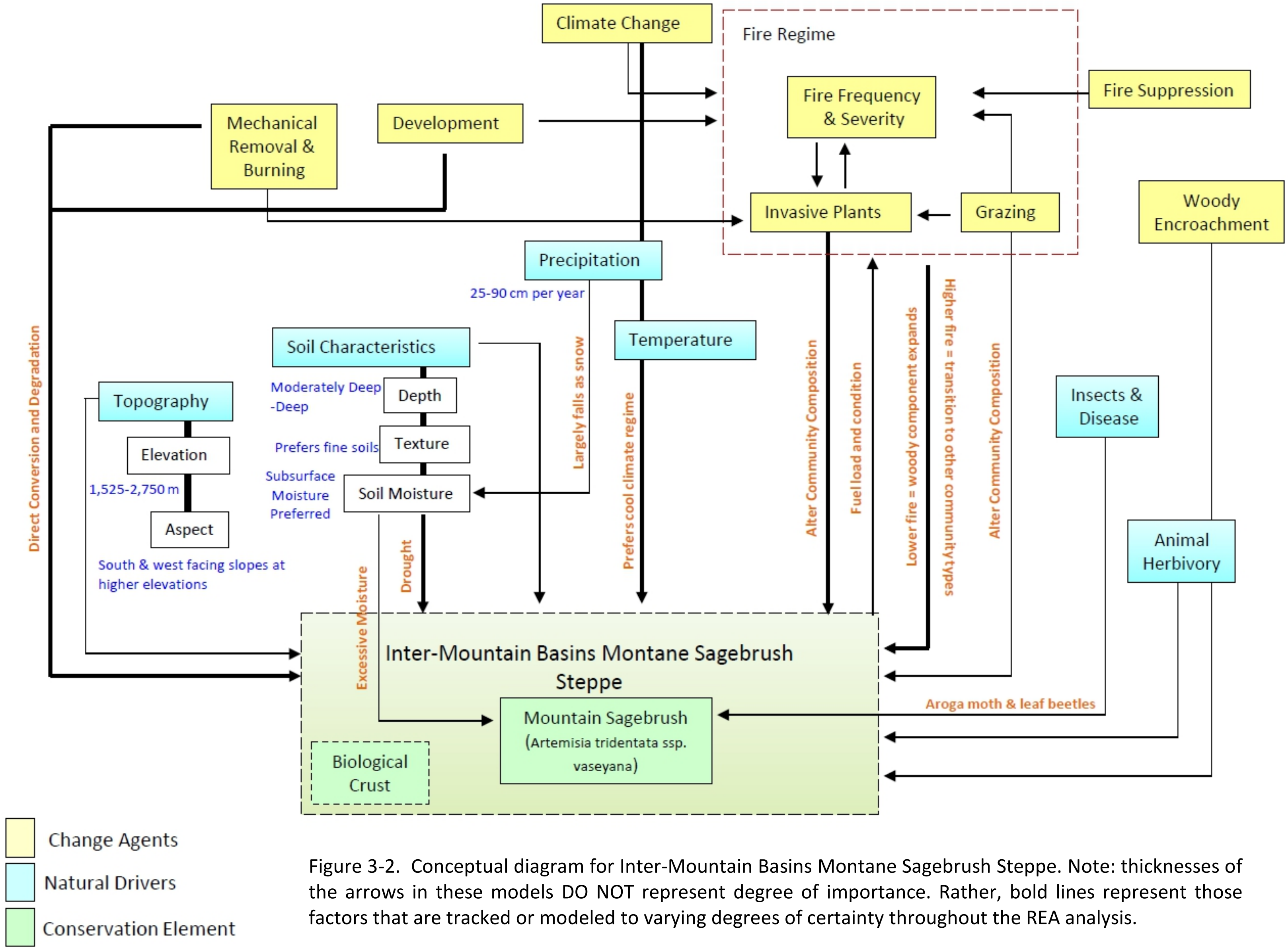


Figure 3-2. Conceptual diagram for Inter-Mountain Basins Montane Sagebrush Steppe. Note: thicknesses of the arrows in these models DO NOT represent degree of importance. Rather, bold lines represent those factors that are tracked or modeled to varying degrees of certainty throughout the REA analysis.

Fire regime (components within the red, dash-lined box) is influenced by a complex interaction of factors—fuel load and condition, grazing, invasive species, and fire frequency (natural [a function of climate] and human-caused [a function of development]). Fire suppression is another influencing factor on the fire regime. Native ecosystems can also be directly affected by invasive species and grazing. Climate change and development affects the entire complex and all of its components. Because Inter-Mountain Basins Montane Sagebrush Steppe occupies many different kinds of physical zones, the natural fire regime for this community is complex. Historically, it experienced stand replacing fire with a mean of 10 years at the ponderosa pine ecotone, 40 or more years at the Wyoming big sagebrush ecotone, and up to 80 years where low sagebrush makes up a high proportion of the landscape. LANDFIRE (2007) reported a replacement fire return interval for this community at 40-80 years (with a mean of 50 years) with the scale of fire disturbance historically ranging from <10 acres to >1,000 acres. Besides fire frequency, seasonality of fire is also important. Sagebrush generally responds favorably to spring fires, but fall fires tend to cause significant mortality in sagebrush. Fire suppression, livestock grazing, and the introduction of invasive species have altered this vegetation community throughout the Colorado Plateau (D'Antonio and Vitousek 1992, Belsky and Gelbard 2000, NatureServe 2009). In locations where fire suppression has been successful, woody encroachment (e.g. juniper and pinyon pine) has been significant. Due to the dynamic nature and interaction of many Colorado Plateau natural ecological systems and the challenge of accurately mapping vegetation using remote sensing, it is extremely difficult to track woody encroachment on this community over large geographic areas. In addition, having more detailed data on grazing history and intensity would greatly improve assessing the overall status of this community type. Although both woody encroachment and grazing intensity are reported to be extremely important for this community, data do not exist to reliably assess and map their impacts.

Change agents affecting this ecological system accounted for in the REA process include development (based on current and projected future extent of urban land cover) and recent disturbance (1999–2008) from mechanical removal, fires, and insects and disease. Mechanical removal or disturbance of this community can also promote invasive grasses altering the system in significant ways. Overall landscape intactness, which includes development from all sources (urban, agriculture, energy, and roads), invasive species, and habitat fragmentation, is used to describe the regional environment that contains this ecosystem type and thus infer its status. Climate change projections (including precipitation and temperature changes as well as MAPSS modeling outputs) are also used to predict where this conservation element may be under significant climate stress.

Following this format, select conceptual models are presented in later sections in this document and all conceptual models for each of the conservation elements are provided in Appendix B for vegetation communities and Appendix C for wildlife species.

3.2.2 Process Models

With conceptual models in-hand to inform the relationships between components, drivers, and processes, individual *process models* were generated to address each stated management question. Process models are diagrams that map out data sources, GIS analyses, and workflow. These models were not intended to attempt to replicate all of the interactions of the concept models. Rather, they were created to inform the user about the spatial analysis details to address each management question, providing important analytical transparency and allowing for repeatability of the same or similar model in the future (perhaps including new input data for a key variable). Each model could be viewed as the analysis recipe including information about data sources, specific GIS operations, and data and map workflows highlighting all intermediate and final map results.

Some management questions required only a series of simple GIS operations (see Figure 3-3 for an example). More sophisticated analyses required developing a more complex, customized approach requiring the construction and implementation of Model Builder/Python scripts and, in some cases, the inclusion of non-ArcGIS software (e.g. MaxEnt, MAPSS, and FRAGSTATS). A separate process model is provided in Appendix A for each management question.

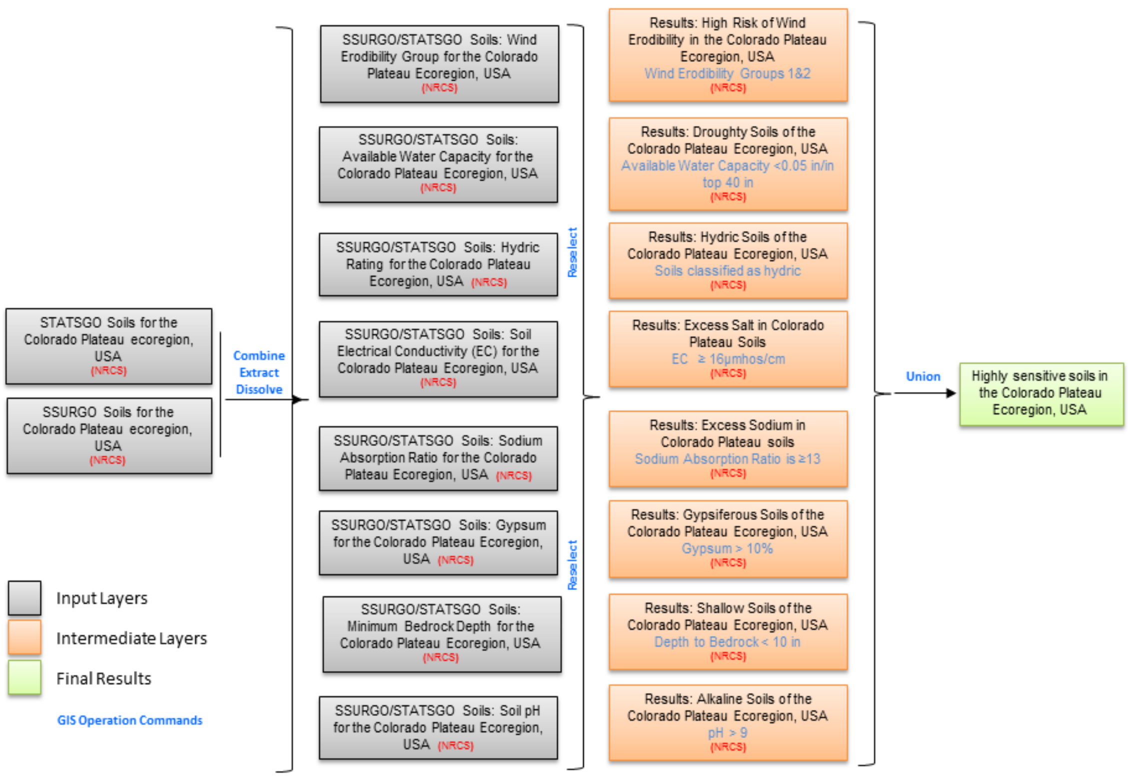


Figure 3-3. Process model diagram for soil sensitivity in the Colorado Plateau ecoregion: Management Question, *Where are sensitive soils (including saline, sodic, gypsiferous, shallow, and low water holding capacity?*)

3.2.3 Logic Models

The most complex questions, such as terrestrial landscape intactness, aquatic intactness, cumulative development, and summarizing climate modeling results, were assessed using the EMDS (Ecosystem Management Decision Support) modeling approach (Reynolds 1999, Reynolds 2001), but all of the modeling operations were conducted using ArcGIS Model Builder and Python scripts, with additional inputs provided by approved outside analyses such as FRAGSTATS. *Logic models* were constructed to help communicate how the various data inputs were used in a spatial modeling environment. A logic model is a cognitive map (Jensen et al. 2009) that presents networks of various spatial data components and their logical relationships to explain the process used to evaluate a complex topic such as landscape intactness. Logic models were constructed in a hierarchical fashion relying on symbols, colors, labels, and the physical arrangement of components to communicate how a series of spatial datasets were assembled and analyzed to answer a particular question. Using the terrestrial landscape intactness logic model as an example (Figure 3-4), the spatial data layers are arranged in a hierarchy to answer a primary question that is located at the top of the diagram. In this case, the question is, what is the level of terrestrial landscape intactness for the ecoregion? Data and analysis flows from the bottom up.

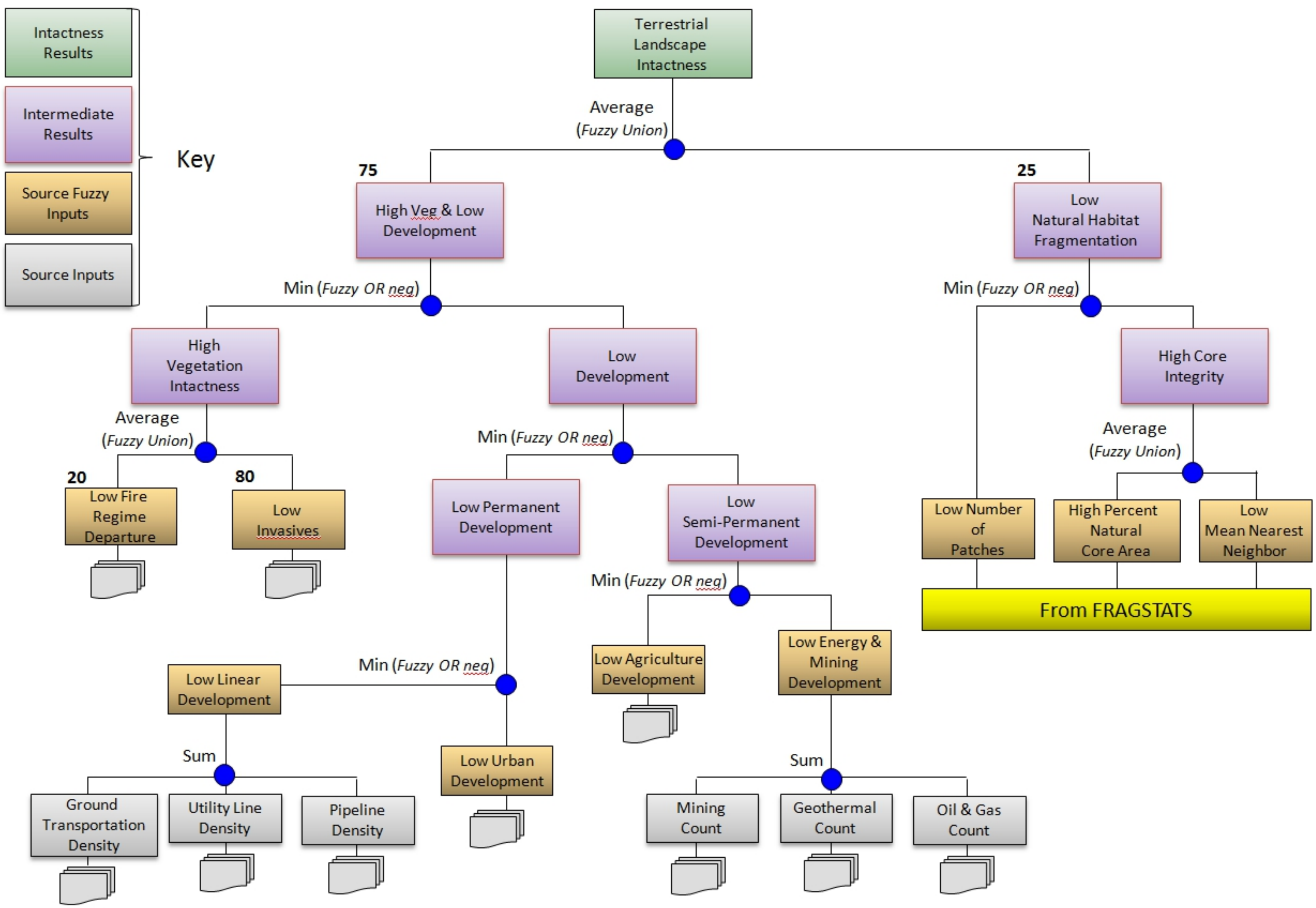


Figure 3-4. Logic model for terrestrial landscape intactness for the Colorado Plateau ecoregion.

Unlike conventional GIS applications that use Boolean logic (1s and 0s) or scored input layers, logic models rely on fuzzy logic. Simply put, fuzzy logic allows the user to assign shades of gray to thoughts and ideas rather than being restricted to black (false) and white (true) determinations. All data inputs (regardless of the type—ordinal, nominal, or continuous) are assigned relative values between -1 (false) and +1 (true) up to six decimal places. There are many advantages of this modeling approach: (1) it is highly interactive and flexible; (2) it is easy to visualize thought processes; (3) the logic components are modular making it easy to include or exclude pieces of the logic design; (4) the logic can be managed using a number of different mechanisms; and (5) numerous, diverse topics can be included into a single integrated analysis.

Raw spatial data source inputs (gold boxes) are populated by one or more GIS data layers (indicated by the stack of gray files). Moving up the diagram, these data are arranged and analyzed to form intermediate map products (purple boxes), which are then arranged and analyzed to generate the final results (green box). One way the user controls the logic of the information is the arrangement of the various data inputs and intermediate products—the higher up in the diagram, the greater the influence on the final result.

Using fuzzy logic as the core modeling principle, logic model performance is achieved in several ways. For every spatial data input, the user determines how to assign the range of values along a truth continuum. As an example, to determine and map the most suitable habitat relative to road density for wildlife—one might consider the greater the road density, the greater the risk to wildlife through habitat degradation and direct mortality. In our example, road density ranges from 0 km/km^2 to 24.5 km/km^2. To assign a fuzzy logic continuum for this range of values, one could assign a -1 to the high value (this value is totally bad for wildlife or false) and a +1 to the lowest value (this value is totally good for wildlife or true, red line in Figure 3-5). However, mountain lion research has shown that mountain lion populations have a low probability of persistence in areas with road densities > 0.6 km/km^2 (Van Dyke et al. 1986). A more meaningful alternative then for setting fuzzy thresholds for this parameter would be that a road density of > 0.6 km/km^2 is totally false (-1) and 0 remains totally true (+1, green line in Figure 3-5). Of course, not all wildlife species have the same sensitivity to roads, but this example illustrates how the logic in the model can be altered for known thresholds.

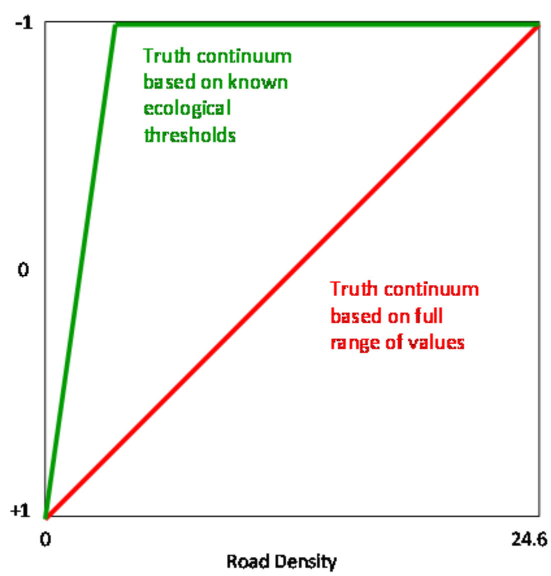


Figure 3-5. Diagram of two treatments of road density in fuzzy logic modeling that illustrate important model control options, one based on a full range of values (red line) and the other based on a threshold from the literature (Van Dyke et al. 1986) for negative effects of road density (> 0.6 km/km^2 is totally false [-1], green line) on mountain lion.

Individual thresholds used for each component in the terrestrial landscape intactness logic model shown in Figure 3-4 are provided in Table 3-1. In this example, there are 12 primary inputs to the model, but two components (Low Linear Development and Low Energy and Mining Development) were created by summing several input values together before applying any fuzzy thresholds. Taking this into account, only nine primary inputs in the logic model required threshold setting.

Table 3-1. List of data inputs for the terrestrial landscape intactness logic model for the Colorado Plateau ecoregion showing data type, range of values, and true and false modeling thresholds for each item at the 4 km x 4 km resolution.

Item	Data Type	Data Range	True Threshold	False Threshold
Fire Regime	Percent Area	0–100	13[1]	98
Invasive Grasses & Tamarisk	Percent Area	0–88	0[3]	33
Linear Development	Density	0–18	0[1]	2.5
Urban Percent	Percent Area	0–99	0[3]	15
Agriculture Percent	Percent Area	0–90	0[3]	20
Energy & Mining Development	Number	0–37	0[2]	1.25
Number of Patches	Number	1–1,455	1[4]	700
Mean Nearest Neighbor	Distance	60–272	60[1]	180
Percent Natural Core Area	Percent Area	.56–95	100[3]	20

1. Used full range or full range with outliers ignored; 2. Skewed data range: 1 Standard Deviation from the mean;
3. Skewed data range: 2 Standard Deviations from the mean; 4. Skewed data range: 2.5 Standard Deviations from the mean.

Spatial data are integrated together using one of several logic 'operators', including Sum, Average (or *Fuzzy Union*), Minimum (or *Fuzzy Or neg*), and Maximum (or *Fuzzy Or*). The Sum operator simply combines similar data into a single file before assigning fuzzy thresholds. For example, Low Linear Development is the fuzzy expression of three linear feature densities—ground transportation, utility lines, and pipelines. Average (or *Fuzzy Union*) simply averages all of the fuzzy inputs to form a new output. Minimum (or *Fuzzy Or neg*) has the lowest value dominate in the resultant map between two or more inputs. For example, in producing the High Veg and Low Development intermediate file, cells that are the lowest in either input get reflected in the resulting map.

Lastly, the logic models produced for the REA contain some weighting of inputs. In the logic model (Figure 3-4), weighting was used in the High Vegetation Intactness intermediate layer (80% for the Low Invasives input and 20% for Low Fire Regime Departure). Weighting was also used in the final combination of High Vegetation and Low Development and Low Natural Habitat Fragmentation (75% and 25%, respectively). Weighting was applied to keep less important factors from dominating the resulting model. If all factors are considered of equal influence, weights may be avoided altogether, or weights can be applied and adjusted on successive model runs to balance the components and test the outcome.

All intermediate and final map results are rendered as fuzzy outputs, which range from -1.000000 (totally false) to +1.000000 (totally true). Interpretation of the range of values for a given map can be organized and interpreted in many ways using standard GIS binning—Natural Breaks, Equal Area, etc. For the terrestrial landscape intactness results, ecologically meaningful results were estimated using a careful selection of operators, thresholds, and input data. A modified EMDS classification was used to characterize intactness and assign six intactness classification descriptions—Very Low, Low, Moderately Low, Moderately High, High, and Very High (Table 3-2). This way, the degree of intactness could be evaluated against multiple conservation values and easily compared to potential future conditions based on updated raw inputs (e.g. new urban development projections) using a standard scale.

Table 3-2. Intactness value ranges and legend descriptions.

Intactness Value	Legend
-1.000 to -0.750	Very Low
-0.750 to -0.500	Low
-0.500 to 0.000	Moderately Low
0.000 to 0.500	Moderately High
0.500 to 0.750	High
0.750 to 1.000	Very High

3.2.4 Habitat Fragmentation Modeling

The three inputs to the Natural Fragmentation component in the terrestrial landscape intactness logic model (number of patches, average mean nearest neighbor, and percent natural core area) were generated using FRAGSTATS (McGarigal and Marks 1995). FRAGSTATS produces a series of metrics that are focused at the individual patch, class, and landscape levels. All three fragmentation indicators chosen were class-level metrics. Prior to running FRAGSTATS, the entire landscape was mapped into one of three classes – natural vegetation, invasive species, and other (including developed, agriculture, and water, Figure 3-6). For this exercise, spatial details on fragmentation of different natural communities were not of primary interest, meaning that differentiating various vegetation communities (e.g. sagebrush shrubland from woodlands) was not needed. Two classes would have sufficed—natural vegetation cover and un-natural vegetation cover (developed land, agriculture); however, having a third class of fragmentation information on invasive species may prove useful in the future as part of a step-down assessment. See specific details on how the master layer was generated in Appendix E. Two of the functions (Percent Natural Core Area and Average Mean Nearest Neighbor) were averaged together to create an intermediate layer called High Core Integrity. This intermediate layer was then combined with the Number of Natural Patches using a Min (or fuzzy Or neg) operator to generate the final Low Natural Habitat Fragmentation component in the model (Figure 3-7).

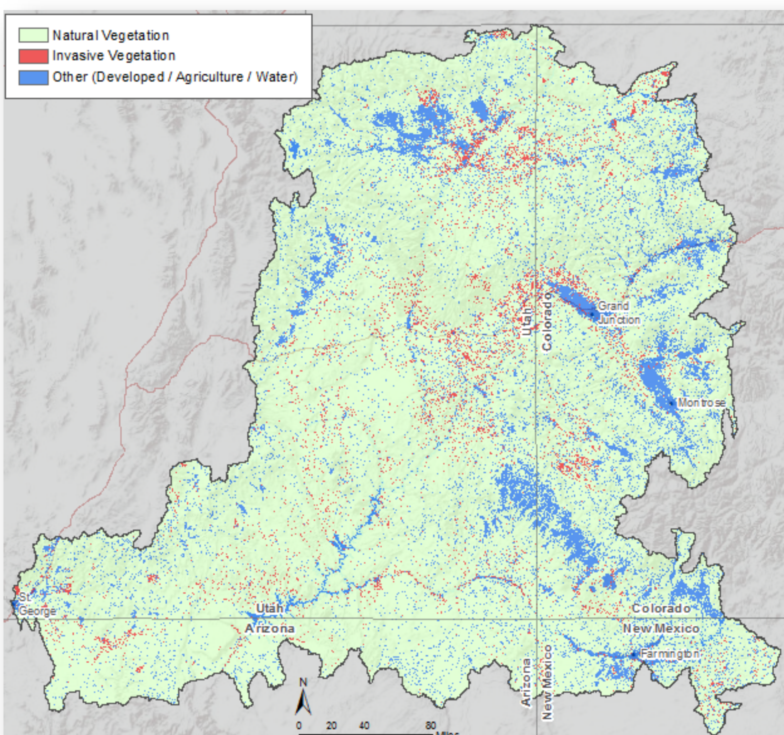


Figure 3-6. Initial FRAGSTATS fragmentation classification showing natural (light green), invasive (red), and un-natural land cover (other=water, developed, and agriculture, blue).

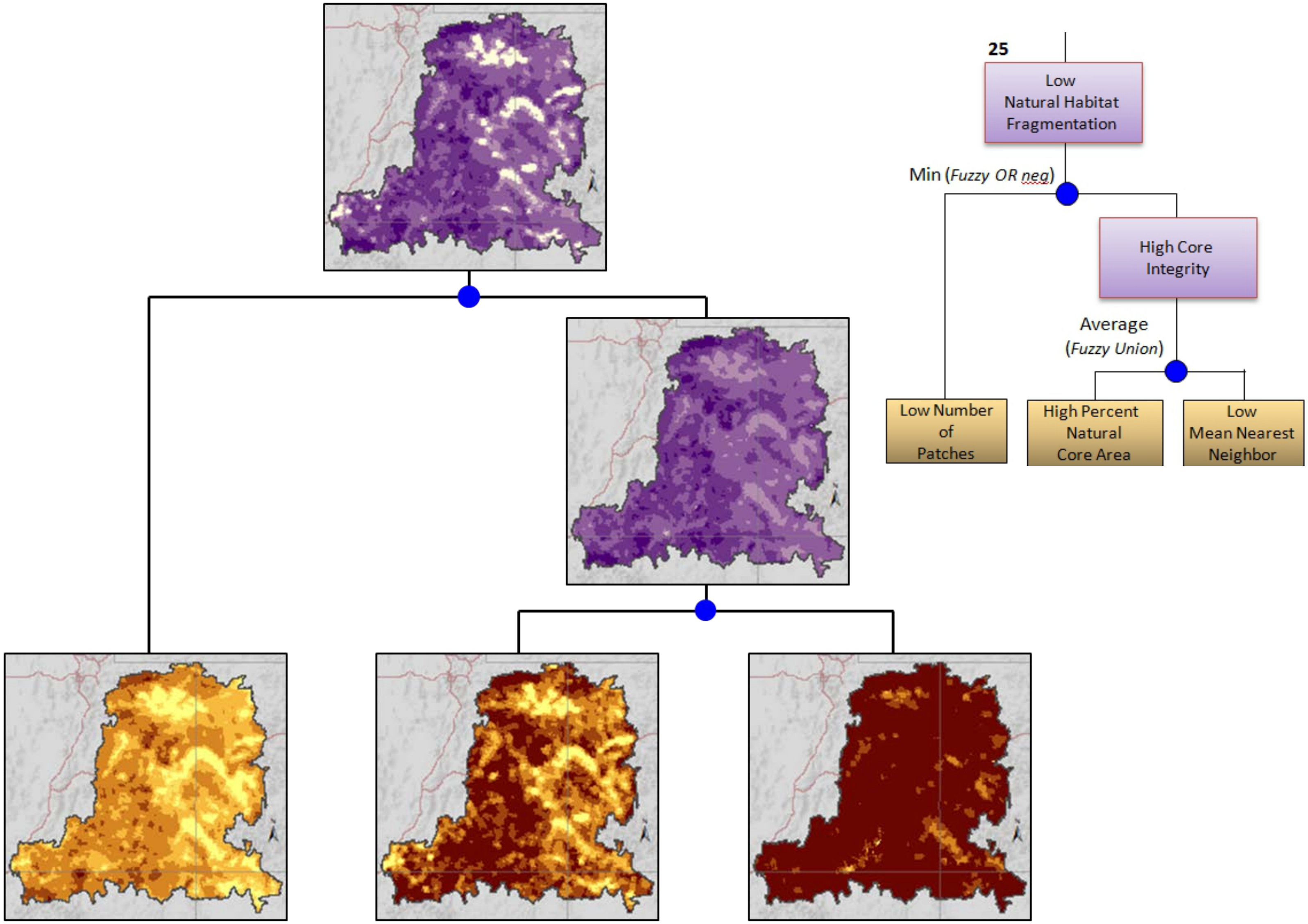


Figure 3-7. FRAGSTATS-based fragmentation inputs into the terrestrial landscape intactness model at 4km resolution for the Colorado plateau ecoregion.

3.2.5 Connectivity Modeling

Habitat connectivity was modeled for the Colorado Plateau using a slightly modified version of connectivity modeling by Spencer et al. (2010). First, natural landscape blocks were mapped for the ecoregion; then, a natural landscape template was constructed starting with the natural cover from the FRAGSTATS analysis layer (see Appendix C for details). Starting with larger blocks (>5,000 ac), an assumption consistent with the block size used in Spencer et al. (2010), natural landscape nodes were delineated throughout the ecoregion.

A cost surface was created following Spencer et al. (2010) combining landcover and protection status costs. In general, water and highest intensity developed classes from LANDFIRE EVT received the highest costs; agriculture and lower intensity developed classes received moderately high costs; invasive species received moderate costs; and natural vegetation received the lowest costs. Costs were also derived from protected areas, such that more highly protected areas (e.g., wilderness) received lower costs and less protected areas received higher costs. A 25-meter buffer around major highways (converted to 30m raster) and a 30m raster of all roads (BLM ground transportation database) were used to assign road costs (among the highest overall). Potential linkages were hand drawn between neighboring natural landscape blocks by connecting each one using a system of drawn sticks (centroid to centroid). Blocks separated only by a major road were connected using "road sticks" while those separated by larger distances were connected by "corridor sticks" (Figure 3-8). Road sticks were excluded from further analysis as these areas would require road mitigation measures to improve wildlife movement. For each pair of blocks connected by a corridor stick, a 5 kilometer neighborhood extent was selected around the pair for least-cost modeling. The cost surface was clipped to this extent and the standard ArcGIS tool "Cost Distance" was used for each block in the pair. The results from each of these cost paths were input to the "Corridor" ArcGIS tool. The corridor was sliced into 20 equal width classes and the lowest 5% of the cost corridor was extracted and mosaicked across all pairs of blocks.

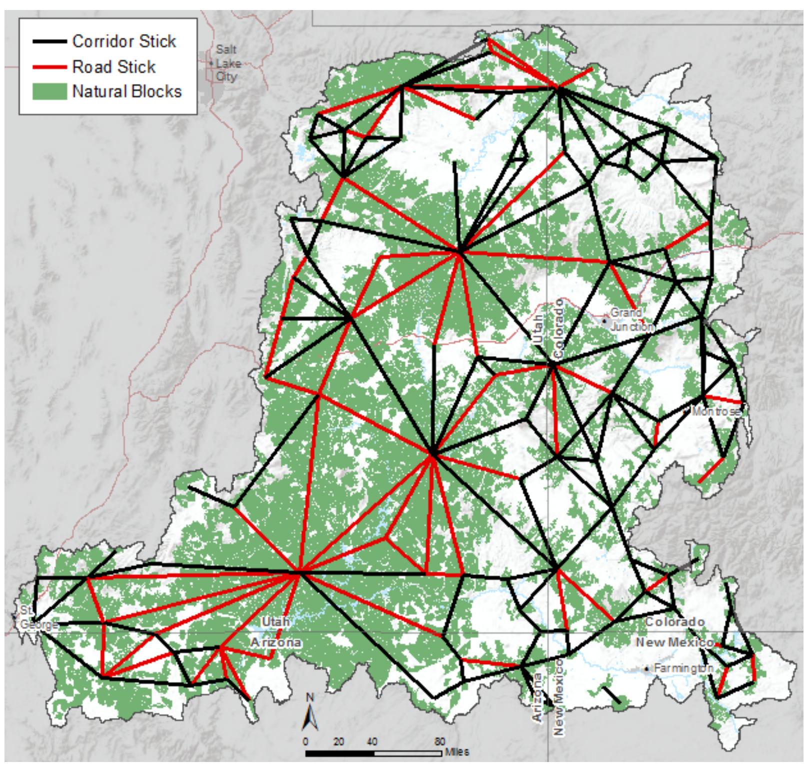


Figure 3-8. Natural Landscape Blocks and connectivity sticks (corridor and road) for the Colorado Plateau ecoregion cost surface connectivity modeling.

3.2.6 Fire Modeling

To assess areas changed by fire (1999–2010), fire location and severity were extracted from LANDFIRE disturbance layers (1999–2008) and wildland fire perimeters (2000–2010) for the Colorado Plateau ecoregion. The degree to which vegetation changed could not be assessed due to the lack of accurate pre- and post-fire vegetation maps. Instead, the focus was on highlighting the severity of the fires, where information was available, because the degree of ecological changes likely increases with increasing severity.

To assess areas with potential to change from wildfire, models were developed to predict the probability of human- and naturally-caused fire occurrences. Thirty years of fire occurrence data (Figure 3-9) were used to develop two MaxEnt models (Elith et al. 2011) to predict human and natural fire occurrences. A series of input surfaces were used as the basis for prediction, including elevation, fuel type, vegetation type, climate variables (e.g., average summer temperature, average summer precipitation, average winter precipitation, and average annual precipitation), distance to major roads, distance to all roads and trails, distance to urban areas, and lightning density. Areas of high probability of occurrence were then extracted from the human and natural model results and combined into a single dataset to express areas likely to experience fires due to humans, natural causes, or both.

A combination of existing data and expert opinion was then used to identify areas of high fire regime departure. LANDFIRE Fire Regime Departure Index (v1.0) was used as an estimate of departure of current vegetation conditions compared to reference vegetation conditions. Reference condition vegetation conditions describe the proportions of various successional stages of a given Biophysical Setting that would be expected to occur across space and time under the influence of unaltered disturbance regimes. Current conditions were tabulated from existing vegetation type and structure and compared to these reference conditions to determine vegetation departure.

Measures of current fire regime (frequency and severity) were obtained from fire experts familiar with the ecoregion for the 40 most extensive Biophysical Settings. These values were compared against reference condition fire regime estimates derived from LANDFIRE Mean Fire Return Interval and Percent Replacement Severity. Measures of fire frequency and severity departure were calculated according to FRCC Guidebook (Barrett et al. 2010) methods, using the average of the minimum and maximum departure values obtained from comparing each range of fire frequency and severity from current estimates to reference condition estimates. Lastly, the maximum departure between vegetation departure and fire frequency and severity departure were extracted to use as our overall measure of fire regime departure.

Photo: Fire in Wyoming big sagebrush steppe. BLM

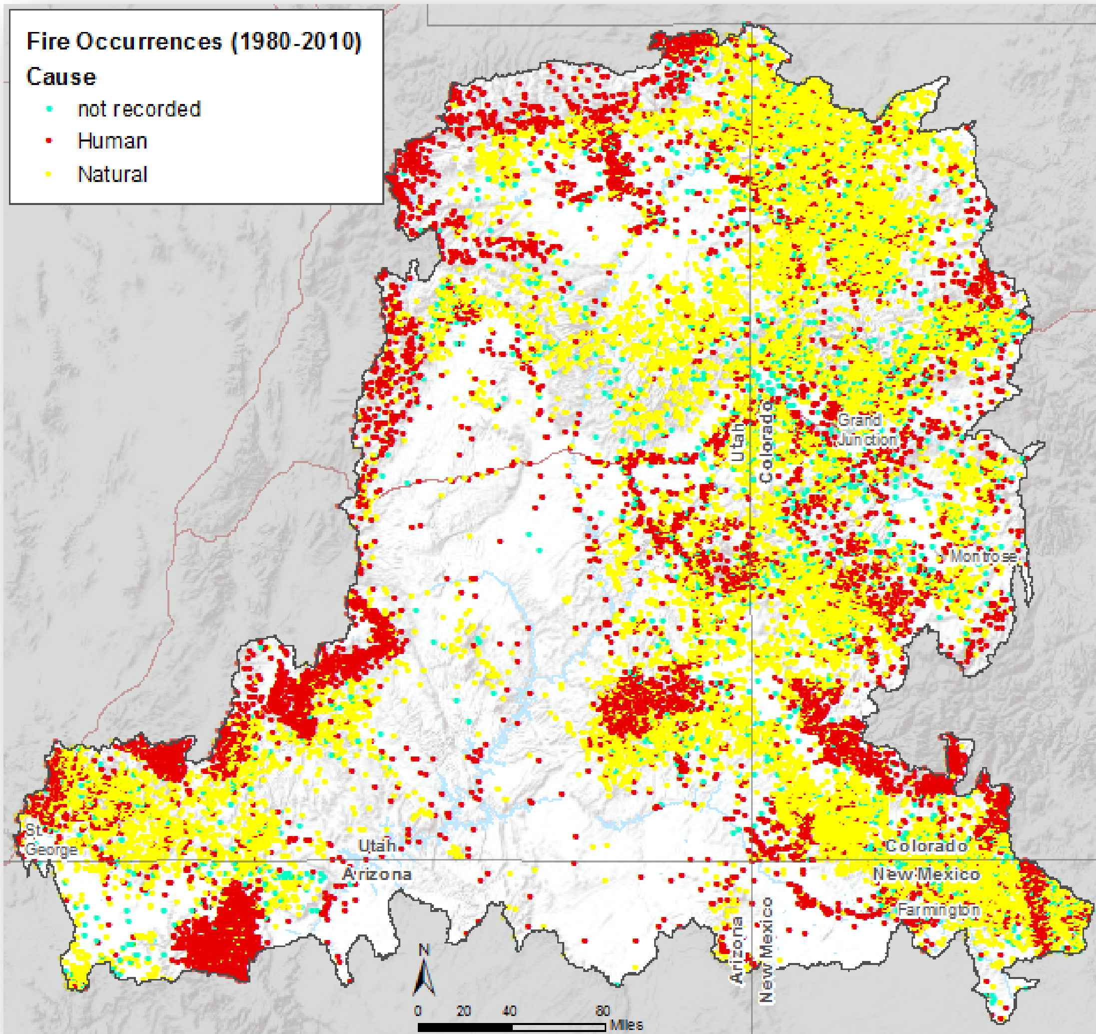


Figure 3-9. Fire occurrences for the Colorado Plateau ecoregion between 1980 and 2010 according to cause of ignition.

To assess areas where fire may be adverse to ecological communities and resources of concern, areas from the LANDFIRE Fire Regime Groups and Succession Classes datasets were extracted to capture the following conditions:

- historically rare fire systems (fires that occur may result in high severity, and may be uncharacteristically frequent if caused by human ignitions).
- historically frequent fire systems (fires may produce potentially uncharacteristic fire behavior due to legacy effects of fire suppression).
- uncharacteristic native vegetation composition or structure (fires may produce uncharacteristic behavior due to uncharacteristic fuel conditions).
- invasive vegetation (fire frequency, severity, and size may be altered by presence of invasives, especially annual grasses).

3.2.7 Climate Modeling

The climate change modeling required extensive exploration and several major processing steps best communicated with a diagram (Figure 3-10). Eight major steps were taken to generate a final potential climate change impact map for the ecoregion.

The base input data into the modeling process was RegCM3: a regional climate model run at 15km spatial resolution. A dataset developed at Oregon State University was selected because it uses dynamical downscaling with RegCM3 and it reflects conditions in the southwestern United States including the North American summer monsoon (Hostetler et al. 2011). It also allowed consistency for REAs across the west. Regional Climate Models have been developed based on the concept of one-way nesting, in which large scale meteorological fields from General Circulation Model (GCM) runs provide initial and time-dependent meteorological lateral boundary conditions (LBCs) for high resolution Regional Climate Model (RCM) simulations, with no feedback from the RCM to the driving GCM. The Regional Climate Model system RegCM, originally developed at the National Center for Atmospheric Research (NCAR) in Colorado, is maintained in the Earth System Physics section of the International Center for Theoretical Physics in Italy. The first version of the model, RegCM1, was based on the NCAR-Pennsylvania State University (PSU) Mesoscale Model version 4 (MM4) (Dickinson et al. 1989, Giorgi 1989). Since then the model has undergone major updates including RegCM2 based on NCAR's Community Climate Model version 2 (CCM2, Hack et al. 1993) and the mesoscale model MM5 (Grell et al. 1994). Further development based on the Community Climate Model version 3 (CCM3, Kiehl et al. 1996) gave rise to RegCM2.5 and RegCM3 that include the effect of three additional types of greenhouse gases (NO_2, CH_4, CFCs), atmospheric aerosols, and cloud ice as well as a prognostic equation for cloud water used in the cloud radiation calculations. RegCM3 includes further improvements that are described in detail elsewhere (Giorgi et al. 2003). In these later models, the USGS Global Land Cover Characterization and Global 30 Arc-Second Elevation datasets are now used to define topography. In addition, NCEP (National Center for Environmental Protection, part of the U.S. National Weather Service) and ECMWF (European Centre for Medium-Range Weather Forecasts) global reanalysis climate datasets are used for initial and boundary conditions.

Input data was first re-projected to the 4km Albers Equal-Area projection using the proj4 library. Elevation data and anomalies for temperature, precipitation, and vapor pressure were re-projected from the 15km Lambert projection (original RegCM3 resolution and projection) and interpolated using bilinear interpolation. Variables examined throughout this assessment included annual average temperature, average annual total precipitation as well as seasonal averages for both temperature and precipitation (winter, spring, summer and fall).

A number of boundary conditions were based on NCEP records and three different GCMs (ECHAM5, GFDL, and GENMOM). To establish the historic baseline, historic model runs were examined using the different GCMs and compared to NCEP and PRISM, which rely on observed weather data over the 1968–1999 time period. PRISM was believed to be the more reliable dataset as it takes into account more information such as elevation and other terrain influences. All GCM-influenced historic model-runs projected wetter conditions than the actual data supported, so the historic baseline was defined using the PRISM-based results. This decision required that anomalies (differences) be calculated between PRISM historic and future time steps based on the various GCMs. The final future climate projections were generated by adding (for temperature variables) or multiplying (for precipitation variables) the model differences to the PRISM historic baseline. It was decided, after review of the future output results and after consultation with climate model experts, to use only the ECHAM5-based future potential climate impacts on the conservation elements. The ECHAM5-based results were then fed into MAPSS (Mapped Atmosphere-Plant-Soil System modeling software, Neilson 1995). Results from MAPSS and ECHAM5 climate projections were integrated into a fuzzy logic model in order to evaluate potential climate change impacts on conservation elements (Figure 3-11).

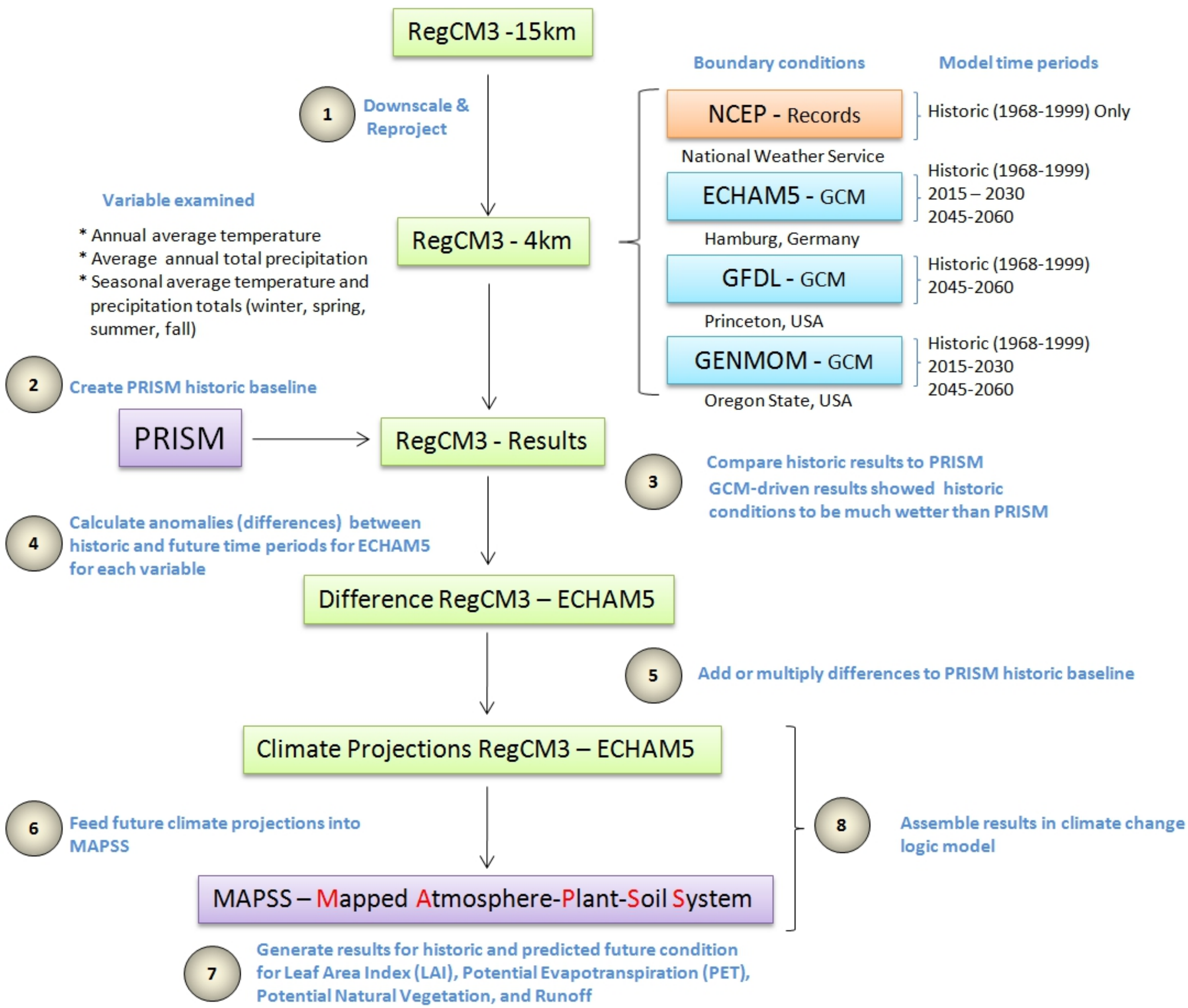


Figure 3-10. Climate change processing workflow: Eight major steps were taken to generate a final potential climate change impact map for the ecoregion.

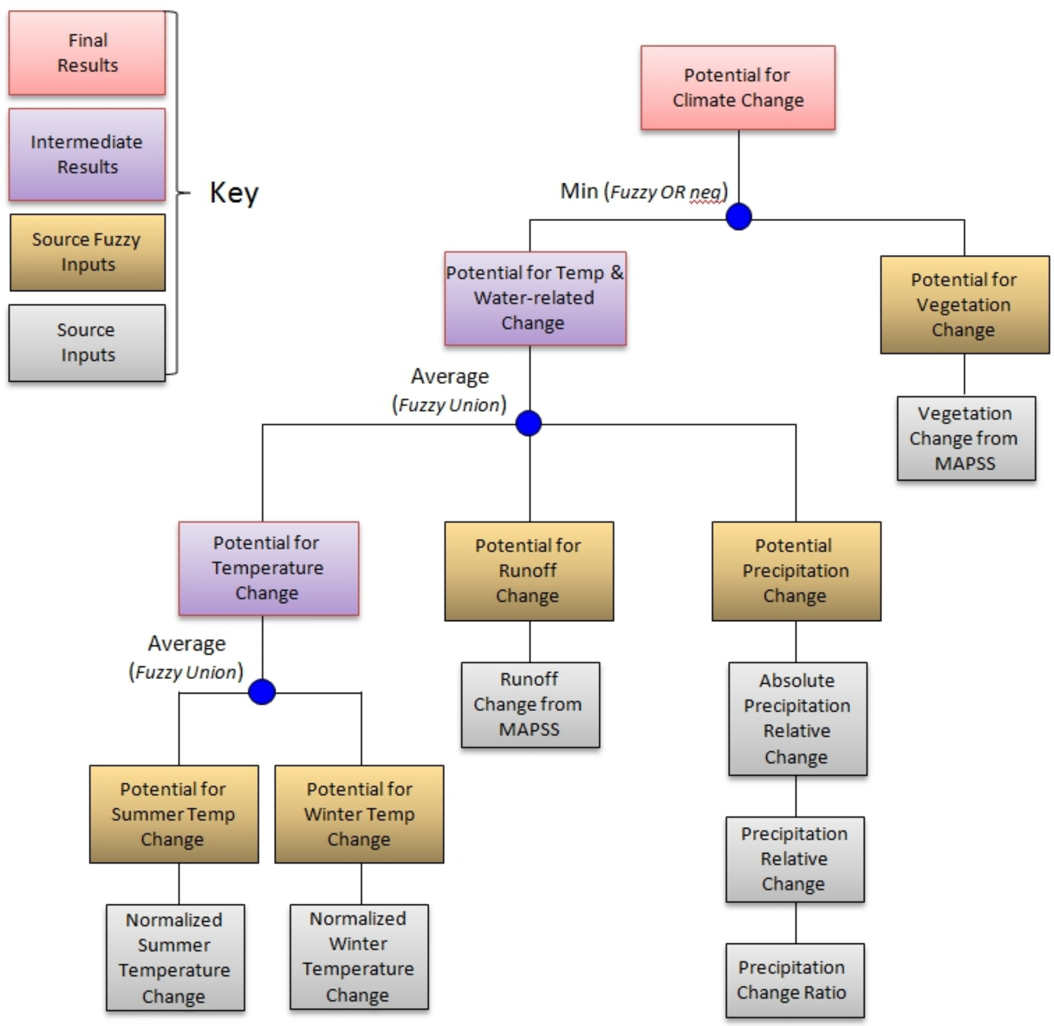


Figure 3-11. Logic diagram assembling key climate variables into an overall potential climate change surface.

3.3 References Cited

Anderson, J.E., and R.S. Inouye. 2001. Landscape-scale changes in plant species abundance and biodiversity of sagebrush steppe over 45 years. *Ecological Monographs* 71:531–556.

Barrett, S., D. Havlina, J. Jones, W. Hann, C. Frame, D. Hamilton, K. Schon, T. Demeo, L. Hutter, and J. Menakis. 2010. Interagency Fire Regime Condition Class Guidebook. Version 3.0. www.frcc.gov. Accessed 10/15/2011.

Belsky, A.J., and J.L. Gelbard. 2000. Livestock grazing and weed invasions in the arid west. Oregon Natural Desert Association. Bend, Oregon.

Blaisdell, J.P., R.B. Murray, and E.D. McArthur. 1982. Managing Intermountain rangelands: Sagebrush-grass ranges. General Technical Report INT-134. U.S. Forest Service, Intermountain Forest and Range Experiment Station, Ogden, Utah. 41 p.

CEC (Commission for Environmental Cooperation). 1997. Ecological regions of North America: Toward a common perspective. Commission for Environmental Cooperation, Montreal, Quebec, Canada. 71p. Map scale 1:12,500,000. Revised 2006.

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

Dickinson, R.E., R.M. Errico, F. Giorgi, and G T. Bates. 1989. A regional climate model for the western United States. *Climatic Change* 15:383–422.

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

Gates, D.H. 1964. Sagebrush infested by leaf defoliating moth. *Journal of Range Management* 17:209–210.

Giorgi, F. 1989. Two-dimensional simulations of possible mesoscale effects of nuclear war fires. *Journal of Geophysical Research* 94:1127–1144.

Giorgi, F., R. Francisco, and J. S. Pal. 2003. Effects of a subgrid-scale topography and land use scheme on the simulation of surface climate and hydrology. Part 1: Effects of temperature and water vapor disaggregation. *Journal of Hydrometeorology* 4:317–333.

Grell, G.A., J. Dudhia, and D.R. Stauffer. 1994. Description of the fifth generation Penn State/NCAR Mesoscale Model (MM5). Technical Report TN-398+STR. National Center for Atmospheric Research, Boulder, Colorado. 121 p.

Hack, J.J., B.A. Boville, B.P. Briegleb, J.T. Kiehl, P.J. Rasch, and D.L. Williamson. 1993. Description of the NCAR community climate model (CCM2). Technical Report NCAR/TN-382+STR, National Center for Atmospheric Research, Boulder, Colorado.

Harniss, R.O., and R.B. Murray. 1973. Thirty years of vegetal change following burning of sagebrush-grass range. *Journal of Range Management* 26:322–325.

Hormay, A.L. 1970. Principles of rest-rotation grazing and multiple-use land management. USDI/USDA Handbook. 25 pp.

Hostetler, S.W., J.R. Alder, and A.M. Allan. 2011. Dynamically downscaled climate simulations over North America: Methods, evaluation, and supporting documentation for users. U.S. Geological Survey, Open-File Report 2011-1238.

Jensen, M., K. Reynolds, U. Langner, and M. Hart. 2009. Application of logic and decision models in sustainable ecosystem management. Proceedings of the 42nd Hawaii International Conference on Systems Sciences. Waikoloa, Hawaii. 5–8 January 2009.

Kiehl, J.T., J.J. Hack, G.B. Bonan, B.A. Boville, B.P. Breigleb, D. Williamson, and P. Rasch. 1996. Description of the NCAR community climate model (CCM3). Technical Report NCAR/TN-420+STR, National Center for Atmospheric Research, Boulder, Colorado.

LANDFIRE. 2007. Biophysical Setting Model. Wildland Fire Science, Earth Resources Observation and Science Center, U.S. Geological Survey, Sioux Falls, South Dakota.

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

Miller, D.M., S.P. Finn, A. Woodward, A. Torregrosa, M.E. Miller, D.R. Bedford, and A.M. Brasher, A.M. 2010. Conceptual ecological models to guide integrated landscape monitoring of the Great Basin, USGS Scientific Investigations Report 2010-5133, U.S. Geological Survey, Reston, Virginia. 134 pp.

Miller, M.E. 2005. The structure and functioning of dryland ecosystems: Conceptual models to inform long-term ecological monitoring. USGS Scientific Investigations Report 2005-5197, U.S. Geological Survey, Reston, Virginia. 73 pp.

NAST (National Assessment Synthesis Team). 2000. Climate change impacts on the United States: The potential consequences of climate variability and change, US Global Change Research Program, Washington DC.

NatureServe. 2009. International ecological classification standard: Terrestrial ecological classifications. NatureServe Central Database, Arlington, Virginia.

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

Pringle, W.L. 1960. The effect of a leaf feeding beetle on big sagebrush in British Columbia. *Journal of Range Management* 13:139–142.

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

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

Spencer, W.D., P. Beier, K. Penrod, K. Winters, C. Paulman, H. Rustigian-Romsos, J. Strittholt, M. Parisi, and A. Pettler. 2010. California essential habitat connectivity project: A strategy for conserving a connected California. Prepared for the State of California and Federal Highways Administration.

Sturges, D.L., and D.L. Nelson. 1984. Snow depth and incidence of a snowmold disease on mountain big sage. Symposium: The biology of *Artemisia* and *Chrysothamnus,* July 9–13, 1984. General Technical Report INT-200, U.S. Forest Service, Intermountain Forest and Range Experimental Station, Provo, Utah.

Tart, D.L. 1996. Big sagebrush plant associations of the Pinedale Ranger District, Pinedale, WY. U.S. Forest Service, Bridger-Teton National Forest, Jackson, Wyoming. 97 p.

Van Dyke, F.G., R.H. Brocke, H.G. Shaw, B.B. Ackerman, T.P. Hemker, and F.G. Lindzey. 1986. Reactions of mountain lions to logging and human activity. *The Journal of Wildlife Management* 50(1):95–102.

IV. EXISTING CONDITIONS IN THE COLORADO PLATEAU

4.1. Colorado Plateau Resources of Concern

4.1.1 Ecoregion Character

> …the strangeness and wonder of existence are emphasized here, in the desert, by the comparative sparsity of the flora and fauna: life not crowded upon life as in other places but scattered abroad in spareness and simplicity, with a generous gift of space for each herb and bush and tree,… so that the living organism stands out bold and brave and vivid against the lifeless sand and barren rock.
>
> — Edward Abbey
> *Desert Solitaire*

Photo: Newberry's twinpod (*Physaria newberryi*), Arches National Park, Neal Herbert

The Colorado Plateau is an elevated tableland situated between the Wasatch Range and Aquarius Plateau to the west and the Southern Rockies in the east. It has a broad latitudinal range, from the Uinta Basin in the north to the arid canyonlands near the Arizona and New Mexico border. The region is an erosional landscape with wind and water working on layer upon layer of sedimentary rock. The Colorado Plateau receives winter precipitation from the Pacific Ocean and variable amounts of summer rain—the summer monsoon—arriving as sporadic storm cells from the south. The summer monsoon is not as reliable as it is in the Sonoran Desert, but it differentiates the Colorado Plateau from the Great Basin, which typically receives little to no summer precipitation (Schwinning et al. 2008). The summer monsoon reaches as far north as the escarpment of the Book Cliffs that separates the southern 2/3 of the region from the Uinta Basin. The Uinta Basin is transitional to the Wyoming Basin in climate and vegetation. The overall climate of the Colorado Plateau, influenced by the El Niño Southern Oscillation (ENSO) climate pattern, is variable from year to year and decade to decade, with periodic droughts of varying length and degree (Swetnam and Betancourt 1998, Cayan et al. 1999).

The major subregions of the Colorado Plateau reflect elevation, moisture availability, and broad vegetation classes (Figure 4-1, Woods et al. 2001, Chapman et al. 2006). At the lowest elevations, 975–1372 m (3200–4500 feet) and 12.7–20.3 cm (5–8 in/yr) of precipitation, the Arid Canyonlands region (20d) delineates the inner gorge of the Colorado River and its tributaries, where steep canyon walls separate the region from the higher plateaus and benches above. Valleys and broad basins with low relief and similar low annual precipitation levels occur at mid-elevations (shale and sand deserts, 20b and 20h). The signature canyon landscapes of the region incorporate the exposed bedrock outcrops, mesas, benches, and rimrock at elevations of 1524–2286 m (5000–7500 feet, 20c). This region, the Semi-arid Benchlands and Canyonlands, receives more precipitation, about 20.3–40.6 cm (8–16 in/yr, up to 20–22 in/yr at the highest elevations of pinyon-juniper). Warm season grasses (e.g. galleta [*Pleuraphis* spp.] and blue grama [*Bouteloua gracilis*]) and big sagebrush (*Artemisia tridentata ssp. wyomingensis*) grow in deeper aeolian soils on the benches and pinyon-juniper woodland covers broad expanses of more rugged terrain. Pinyon-juniper spans a broad elevational range—at the lower, drier end, juniper dominates and tree density is savanna-like; pinyon pine (*Pinus edulis* and *Pinus monophylla*) overlaps juniper distribution at higher elevations, where increased moisture creates more of a closed woodland canopy.

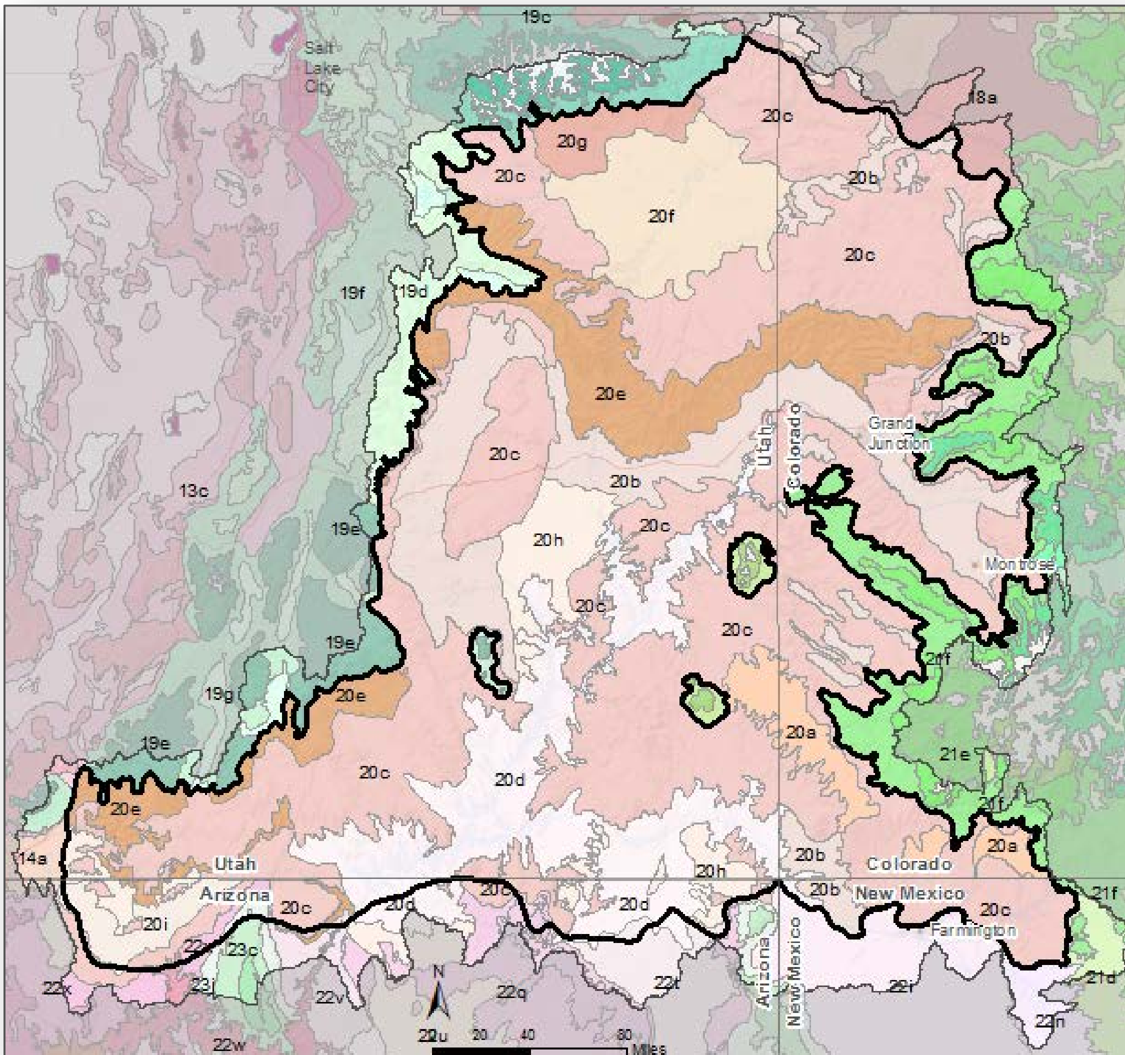


Figure 4-1. Level IV ecoregions of the Colorado Plateau. U.S. EPA, ftp://ftp.epa.gov/wed/ecoregions/

Eight Ecological Systems (vegetation communities) were selected as conservation elements for the REA; they are listed in Table 2-2 in Section 2.4.2, in the discussion of the ecoregion conceptual model below, and in map results discussed in Section 4.2.2. The vegetation communities selected represent the regional range in elevation and aridity. Rocky Mountain Gambel Oak-Mixed Montane Shrubland occurs at the highest elevations of the region in the transition to the mountainous inclusions of the La Sal, Abajo, and Henry mountain ranges (where precipitation levels increase enough to support scattered ponderosa pine). Characteristic vegetation communities of higher elevation mountains (e.g., Rocky Mountain ponderosa pine woodland or subalpine spruce-fir) are not included in the REA (they appear as “doughnut holes” in the map results). However, the mapped distributions of wide-ranging wildlife species such as mountain lion or mule deer do include the mountain ranges.

Periodic drought and human land use influence plant mortality, insect outbreaks, and fire frequency that over time modify species interactions and distributions (Allen and Breshears 1998, Schwinning et al. 2008). Pinyon-juniper woodland, for example, is in constant flux, with juniper expanding into finer soils at the lower end of its elevation range and woodland becoming generally denser and less savanna-like as grasses and forbs are eliminated or reduced by grazing. With the elimination of fine fuels that carried frequent low severity fires, fire in pinyon-juniper is evolving toward more infrequent stand-replacing burns at all elevations. Where pinyon-juniper has been invaded by non-native annuals such as cheatgrass, the opposite may occur, with fire becoming more frequent and invasive grasses becoming dominant (Getz and Baker 2008, Brooks 2008). Pinyon is also capable of rapid upslope movement to replace ponderosa pine killed by drought (Allen and Breshears 1998). In a study of pinyon-juniper populations in western Colorado, Shinneman and Baker (2009) linked woodland species age structure to ocean-atmospheric fluctuations (ENSO). They confirmed that juniper is more drought resistant than pinyon pine and noted some areas of juniper expansion during times of drought. Pinyon pine, on the other hand, experienced major setbacks during periods of drought, and the species appeared to require above-average moisture periods for recovery. The most recent drought (1998–2005) resulted in broad areas of pinyon pine mortality related to both drought and subsequent insect outbreaks (pinyon ips beetle [*Ips confusus*], Figure 4-2).

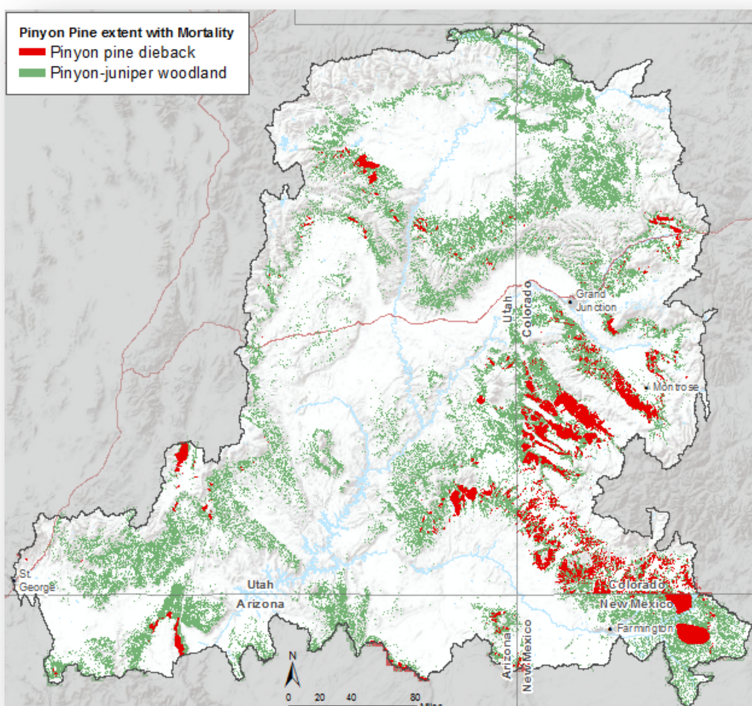


Figure 4-2. Pinyon pine mortality, 2000–2007. U.S. Forest Service Forest Health Technology Enterprise Team

Within the last 50 years, the large blocks of intact vegetation that characterized the Colorado Plateau have been fragmented by energy, recreation, and rural home development, road building, and expanding off-road vehicle usage. Two pressing issues affecting the near-term future land management in the region are oil and gas leasing and a renewed interest in uranium mining (after a 10X uranium price increase between 2002 and 2007, Harding 2007). Approximately 12,500 new oil and gas wells are predicted in the San Juan River basin in northwestern New Mexico over the next 10 years, increasing the density around the 18,000 existing wells by 50% (NMDGF 2006, from BLM Farmington Resource Management Plan [2003]). A similar issue exists in the northeastern Colorado Plateau (Uinta and Piceance Basins) in sagebrush communities where oil and gas leasing projections and management strategies for candidate-listed sage grouse must be resolved by 2014. Region-wide stressors and their effects on biota are covered in Sections 4.1.2 through 4.3—terrestrial and aquatic resources, change agents, distribution and status of conservation elements—and Chapter V, potential future conditions.

Photo: Aerial view of oil and gas wells at the base of Roan Plateau near Grand Junction, Colorado. Photo courtesy of Skytruth, Ecoflight, 2007

4.1.2 Ecoregional Conceptual Model

Conceptual models help to visualize the factors that affect, both positively and negatively, the current and future condition of resources of conservation concern and to define the relationships between conservation elements and associated change agents. The expression of known relationships in conceptual models forms the basis for the development of management questions and the selection of associated data layers and analyses. The ecoregion conceptual model provides a broad scale overview of the region, denoting important natural drivers and anthropogenic change agents. It served as the source for more detailed conceptual models that were delineated to relate individual conservation elements to topical information gleaned through literature review and to identify how much of that information was accessible as spatial data.

In the ecoregional conceptual model for the Colorado Plateau (Figure 4-3), boxes represent abiotic attributes and conservation elements, ovals the classes of change agents, and arrows their direct and indirect effects (threats, stresses, or positive effects) on ecosystem components. Regional climatic conditions represent the dominant natural change agent (orange oval) with natural fire regime and cyclical drought secondary. Human activities (yellow oval marked *land and resource use*) cover urban and industrial development, surface and groundwater extraction, recreation, agriculture, grazing, and the introduction of invasive plants. A yellow concentric oval surrounds regional climate and fire to indicate ongoing human-induced climate change and changes in fire regime. Across the ecoregion, variability in geology, physiography, elevation, aspect, ground and surface water availability, and soil (texture, depth, and water-holding capacity) is reflected in patterns of vegetative cover. Wildlife occurrence and abundance is dependent on interactions with all the abiotic factors (such as climate, fire regime, and water availability) and the vegetation classes (representing major habitats).

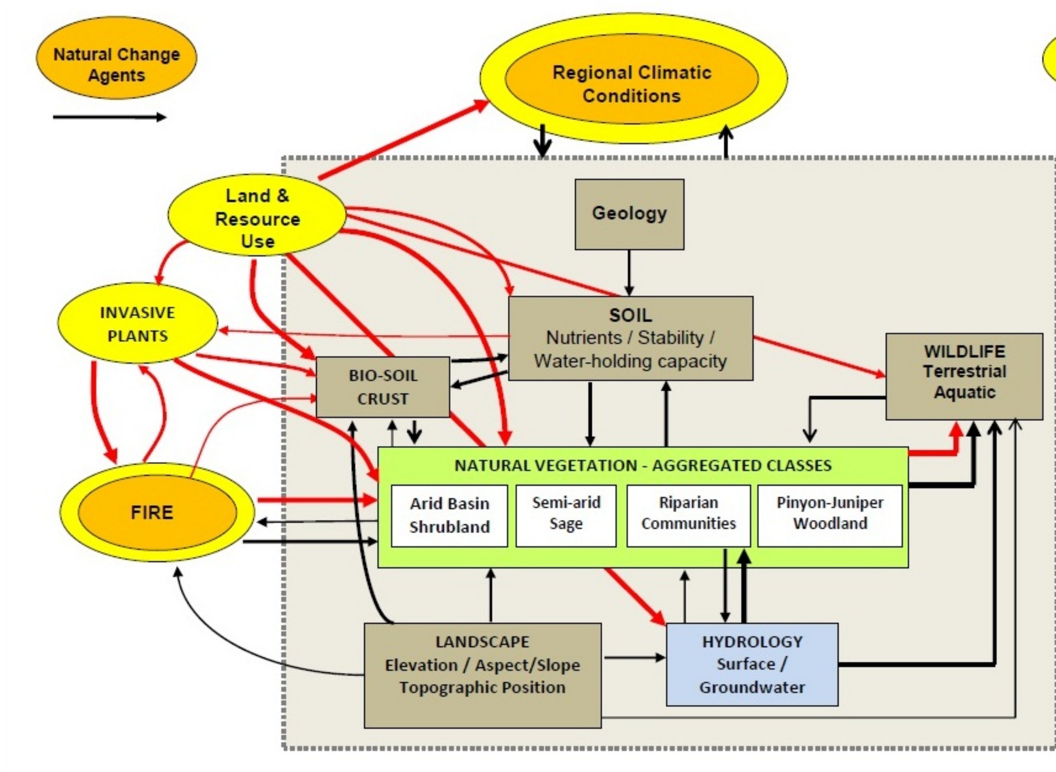


Figure 4-3. Basic ecoregion conceptual model for the Colorado Plateau Ecoregion, with both natural and anthropogenic change agents shown.

Four representative natural vegetation classes are centrally located in the ecoregion conceptual model. The boxes for vegetation classes are depicted according to elevational and moisture differences; they represent various aggregations of the coarse filter conservation element classes (Table 2-2, Chapter 2, SWReGAP, Prior-Magee et al. 2007):

- Upland Forest and Woodland class mainly includes pinyon-juniper woodland, but it may also cover small inclusions of other woodland and mesic shrubland vegetation types, such as Rocky Mountain Aspen Forest and Woodland or Gambel Oak-Mixed Montane Shrubland, in the transition to neighboring higher elevation ecoregions or mountainous inclusions (such as the slopes of the La Sal Mountains)
- Riparian Communities contains the coarse filter classes Woody Wetland and Riparian Communities and Emergent Herbaceous Wetlands.
- Semi-Arid Sage class covers the Shrub/scrub and Semi-arid Grasslands vegetation classes in areas with annual precipitation ranges of 8–13 in/yr.
- Arid Basin Shrubland represents mainly the Inter-Mountain Basins Mixed Salt Desert Scrub and Southern Colorado Plateau Sand Shrubland.

The signature canyonlands, dunes, playas, bedrock, and cliffs of the Colorado Plateau are represented by the Sparsely-Vegetated and Barren class (not pictured as a vegetation class). Although biological (cryptogamic) soil crust might logically fall into several of the coarse filter vegetation classes, it is shown separately in the conceptual model to highlight its importance as a key conservation element. Soil crusts serve as intermediaries between soil and vegetation, with important soil stabilization and nitrogen-fixing roles to play (Belnap 2002, Housman et al. 2006).

4.1.3 Terrestrial Resources of Concern

4.1.3.1 Soil Stability

Soils Management Questions

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. *Where are hotspots producing fugitive dust that may contribute to accelerated snow melt in the Colorado Plateau?*

Photo: Dust storm from Milford Flat Burn area, eastern Bonneville Basin, Utah. M. Miller, U.S. Geological Survey

Soil stability was selected as a terrestrial function of high ecological value for the Colorado Plateau REA. Soils of the region are relatively undeveloped, having formed in residuum from sedimentary rocks weathering-in-place (Bowker and Belnap 2008). Aridisol and Entisol soil orders are dominant across the Colorado Plateau with soil temperatures ranging from thermic to mesic depending on elevation and aspect. Calcium carbonate commonly precipitates out in soils to produce a *caliche* layer that restricts the downward movement of water (Boettinger, 2012). Colorado Plateau soils are fragile—being generally shallow, with low organic content, and sparse vegetative cover—and exposed to erosion by a number of natural and anthropogenic change agents. Persistent wind and wind erosion of soil are natural phenomena in desert ecosystems, but human activities, including mining, energy and urban development, agriculture, recreation, and grazing, all disturb the soil surface, affecting protective crusts, and exposing underlying soils to wind and water erosion. Fine-textured soft shales, mudstones, and siltstones (such as Mancos shale; photo, right), besides being susceptible to mechanical disturbance, are also particularly vulnerable to water erosion. After storm events, these soils deliver excess sediment, salt, and sometimes toxic elements (mercury, arsenic, and selenium) to runoff that affects the Colorado River and its tributaries, such as the Dirty Devil and Paria rivers that carry heavy sediment loads (Voigt et al. 1997, Waring 2011, Jackson 2005). Mitigation of disturbances to saline soils is essential for the BLM to comply with the Colorado Basin Salinity Control Act (BLM 1987). Soils with unique chemical and physical properties develop from the varied geological formations of the Colorado Plateau—e.g., calcareous (limey or chalky) or gypsiferous (high in gypsum) soils—which in turn support a number of rare and endemic plant species. The Colorado Plateau ecoregion has the largest number of endemic plant species in North America (Waring 2011); many of the region's endemics are restricted to growing on a single geologic type (e.g., gypsum, limestone, Davis 2011).

Photo: Mancos shale deposit. T. McCabe, U.S. NRCS

Soils in minimally-disturbed arid and semi-arid systems maintain stability and resist erosion through a complex interaction of plants (shrubs and a sparse cover of grasses and forbs), biological soil crusts, and a network of filamentous, subsurface root symbionts or arbuscular mycorrhizal (AM) fungi. Chaudhary et al. (2009) used structural equation modeling to estimate the contribution of each of these elements to soil stability. Their model explained 35% of the variation in soil stability; biological soil crusts made the largest contribution, followed by plants and AM fungi. They found no difference in stability between shrub-protected soils and soil in the inter-shrub spaces because of the protection offered by soil micro-communities above and below ground. Chaudhary et al. (2009) concluded that aridland managers should expend a greater proportion of their funding and effort on preserving and restoring biological soil crust (and associated AM fungi) than on plant cover.

Soils that have characteristics that make them extremely susceptible to impacts and difficult to restore or reclaim are considered sensitive soils. Ranges in soil properties may be partitioned into classes of vulnerability to site degradation (Table 4-1, Bill Ypsilantis, BLM via Lisa Bryant, Utah BLM). Known values and predicted thresholds for local soil properties can be used to manage within acceptable ranges and protect vulnerable sites from accelerated erosion, compaction, or invasion by alien annual grasses or noxious weeds. Managers have the option to avoid locating disturbances in areas with high-risk sensitive soils and to incorporate best management practices to mitigate negative impacts. Management strategies will vary by the cause of sensitivity. REA component maps produced using STATSGO and higher resolution SSURGO data, where available, depict wind and water erodibility, individual classes of sensitive soils as listed in Table 4-1 (plus hydric and gypsiferous soils), and a composite map of potentially sensitive soils (Figure 4-4).

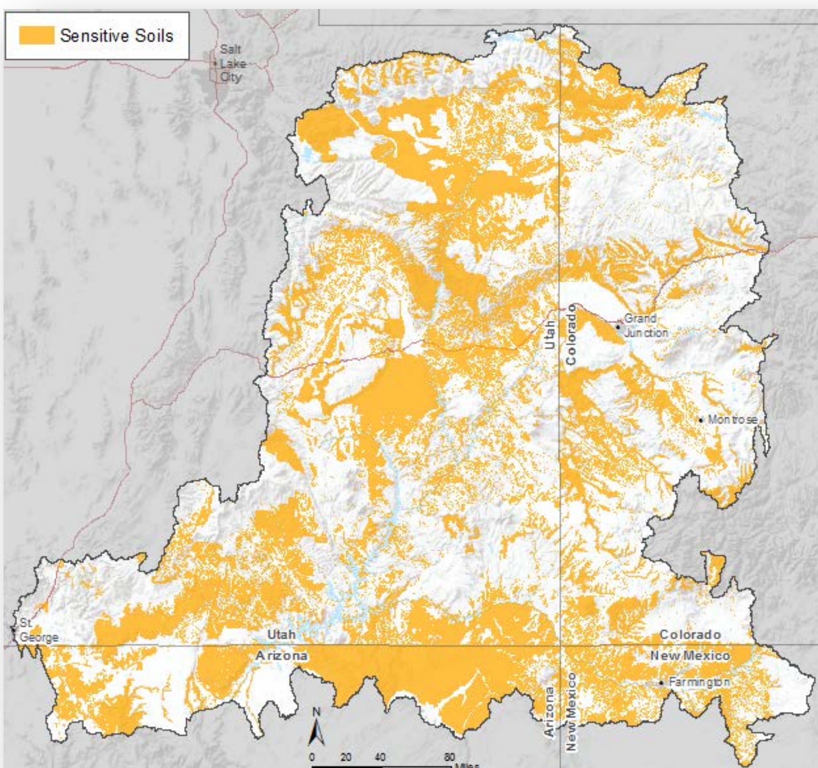


Figure 4-4. Map showing all classes of sensitive soils, including droughty, shallow, hydric, gypsiferous, salty, and high calcium carbonate (calcareous). Large polygon in Arizona reflects availability of only coarser resolution STATSGO soil data. See Appendix A for modeling details.

Table 4-1. Soil vulnerability to site degradation depicts ranges of soil properties with low, moderate, and high risk of degradation. Other properties mapped but not listed include hydric and gypsiferous soils.

PROPERTIES	LOW	MODERATE	HIGH	RESTRICTIVE FEATURE
SLOPE (Pct) Kw < 0.20[1,2] Kw 0.20 – 0.36[1,2] Kw >0.36[1,2]	<20 <15 <10	20–40 15–35 10–25	>40 >35 >25	Steep Slopes Water Erosion
WIND ERODIBILITY GROUP (Surface Layer)	5, 6, 7, 8	3,4, 4L	1, 2	Wind Erosion Hazard
AVAILABLE WATER CAPACITY[2] (Ave. to 40 in. or limiting layer; in/in)	>0.10	0.05–0.10	<0.05	Droughty Soils
SALINITY[2] Surface Layer (µmhos/cm)	<8	8–15.9	≥16	Excess Salt
SODIUM ADSORPTION RATIO[2] Surface Layer	<8	8–12.9	≥13	Excess Sodium
DEPTH TO BEDROCK/ CEMENTED PAN[2] (Inches)	>20	10–20	<10	Rooting Depth
ALKALINITY pH (mol/L)	Slightly alkaline 7.4–7.8	7.9–9	>9	High Alkalinity

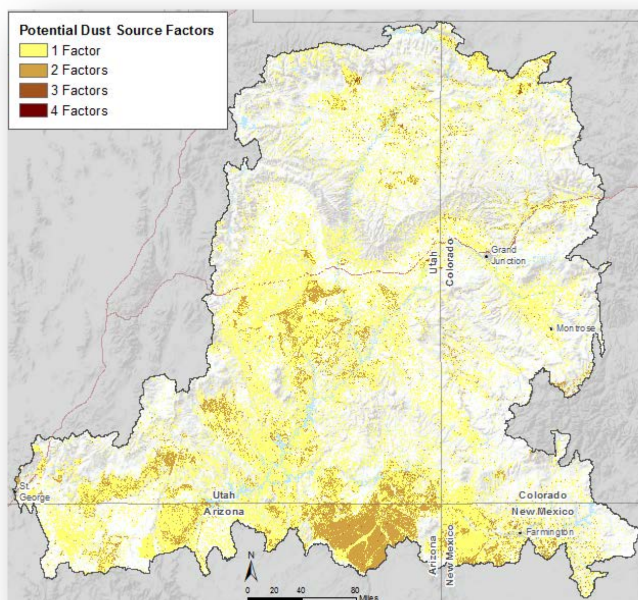
1 K Factor of surface layer adjusted for effect of rock fragments (Kw).
2 The representative value for the range in soil properties

4.1.3.2 Wind Erodibility and Dust on Snow

Wind erosion removes nutrients and growing medium from shallow desert soils and semi-arid agricultural areas. Airborne soil particles affect air quality and visibility, nutrient balance, and spring snowmelt in mountainous areas downwind, and blowing dust creates a health and safety hazard for the region's residents (Neff et al. 2008, Munson et al. 2011). Evidence suggests that accelerated wind erosion has occurred since Euro-American settlement and may increase in the future with increasing drought predicted under future climate change. Neff et al. (2008) found that the dust load in several alpine lakes in the San Juan Mountains east of the Colorado Plateau increased 6X following settlement of the ecoregion in the 19th century and it persists at 5X natural levels to the present day. The dust loading peaked in the early 20th century when unrestricted grazing was practiced across the ecoregion and stabilized following passage of the Taylor Grazing Act of 1934. Grazing pressure has declined somewhat, but grazing continues along with energy development, road building, agricultural activities, and off-road motorized recreation that all add to soil disturbance and dust generation.

Dust production varies by soil type, amount of disturbance, plant cover, drought cycles, and extreme wind events. Clearly, vegetative cover is a deterrent to wind erosion in a region with shallow, undeveloped soil and recurrent drought. Well-developed biological soil crust prevents soil movement in high winds (Belnap and Gillette 1998), and shrubs with soil crust covering inter-shrub spaces provide the best protection against wind erosion. Munson et al. (2011) modeled wind erosion under various vegetation scenarios and found that taller shrubs such as sagebrush (*Artemisia tridentata*) and blackbrush (*Coleogyne ramosissima*) had low modeled sediment movement even without a protective cover of soil crust between the plants. Areas with lower stature shrubs, such as saltbush (*Atriplex* spp.) growing under more hostile conditions, resisted erosion if soil crust was present. Munson et al. (2011) detected a threshold of 10% perennial shrub canopy cover—when shrub cover fell below 10%, wind erosion increased substantially. Levels of wind erosion also varied among grassland types with grass-bare areas (perennial grasses and bare ground) consistently emitting dust and annualized-bare areas (invasive annual grasses and forbs plus bare ground) particularly vulnerable to severe wind erosion when drought conditions reduced the cover of annual plants (Miller et al. 2011). In a nine-year study of emissions from plots of varying disturbance regimes, Belnap et al. (2009) found that a grazed plot with annual grass cover produced 41 times more airborne sediment over the course of the study than a never-grazed site with few invasive annuals. A grazed site with perennial grass cover and a site withdrawn from grazing for 45 years produced 4–4.6 times the sediment as the never-grazed site. Extreme drought years maximize the losses from wind erosion; during the severe drought year of 2002, the annual grass plot produced 334 times the sediment of the never-grazed plot (Belnap et al. 2009). A combination of drought, soil erosion, and nutrient loss negatively influence rangeland sustainability in the region (Neff et al. 2005).

One of the farthest reaching implications of wind-borne sediment is its effect on snowpack in downwind mountain ranges and ultimately, on water yield to the Colorado River and its tributaries. Airborne dust that collects on mountain snow decreases snow reflectance and accelerates spring snowmelt. For example, in 2009, the San Juan Mountains experienced heavy fallout from spring dust storms; even though the snow pack was average, spring snow melt out was the earliest on record at 50 days earlier than normal (J. Deems, REA Workshop 3 presentation). Painter et al. (2010) modeled the impacts of dust on snow to estimate its contribution to changes in runoff in the Upper Colorado River Basin during the timeframe 1916–2003. They found that while modeled natural flow peaked in June and produced runoff into July, post-disturbance (present day) runoff increased in April, peaked in May, and dropped off in June. Their models indicate that dust is reducing the flow on the Colorado River by 5% (two times the annual allotment for Las Vegas). Early snowmelt from accumulated dust (26–50 days) is greater than that predicted for temperature and precipitation changes from climate change (5–15 days). The authors believe that regional efforts at dust abatement and soil stabilization could have a real mitigating effect on the runoff response of the Upper Colorado River as well as future regional impacts of climate change.



REA map results answer the management question—Where are hotspots producing fugitive dust that may contribute to accelerated snow melt in the Colorado Plateau? The map shows potential sources of dust, which may contribute to accelerated snowmelt in the ecoregion (Figure 4-5). In particular, this dataset shows a number of factors that may contribute to dust production at a location. These factors include areas around mines and oil/gas wells, low vegetation cover or invasive annual vegetation, recent disturbances (since 2005), unpaved roads, and soils with high potential for wind erosion. Note that the roads factor should be treated with the least certainty because the dataset used for this analysis does not fully distinguish paved from unpaved roads. The combination of factors at a location may produce a non-linear response with respect to dust production: each factor alone may have varying magnitude depending on location, local wind and topography, and degree of disturbance. Factors may combine such that the net effect is greater than the sum of the factors taken independently. See Appendix A for full treatment of each management question, modeling approach, data sources, and other component maps.

4.1.3.3 Biological Soil Crust

Crust Management Questions

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?*

Photo: Well-developed and minimally disturbed biological soil crust at Canyonlands National Park. N. Herbert, NPS

Cryptogamic (or biological) soil crust was selected as a conservation element because of its key role in maintaining ecosystem function in the Colorado Plateau ecoregion (see Appendix A for conceptual model). Biological soils crusts are comprised of cyanobacteria, fungi, and lichen growing in a symbiotic relationship on the soil surface. Soil crusts can cover up to 70% of live ground cover in the region (Belnap 1994). Soil crust species richness varies by soil type and parent material, with species richness higher on gypsiferous soils, non-calcareous sandy soils, and limestone-derived soils and lower (or minimal) on fine shale-derived soil (Bowker and Belnap 2008). Soil crusts are useful ecological indicators of desert condition because they are not only sensitive to disturbance but they respond to disturbances in predictable and quantifiable ways (Bowker et al. 2008).

Some of soil crusts' essential functions have been discussed in earlier sections on soil stability and wind erodibility. Semi-arid and arid landscapes with sparse vegetation and soil crust cover lack redundancy in function—when crust is eliminated so too are the essential functions of nitrogen fixation, carbon storage, the capture of dust and airborne nutrients, moisture retention, and the provision of microsites for native plant germination (Miller et al. 2011). Soil crusts provide the largest (natural) nitrogen input to soil in the Colorado Plateau. Estimates for annual nitrogen fixation range from 1–9 kg/ha/yr depending on soil crust composition and cover (Belnap 2002).

Most soil-crust nitrogen fixation occurs during the cooler seasons of the year, peaking in the spring when the nitrogen becomes available to vascular plants for the new growing season (Schwinning et al. 2008). Desert nutrient cycling is particularly prone to disturbance and loss with the degradation of soil crust, because a high proportion of nutrients in desert soil occur in surficial fines that are easily carried away by the wind when unprotected by crust (Neff et al. 2005).

Soil crust populations are degraded when mechanical disturbances such as vehicular traffic, land clearing, or trampling disturb the soil surface. While any of these disturbances may not directly eliminate soil crusts, repeated disturbance degrades and fragments crust cover and may keep it in an early successional state (Belnap et al. 2001). Land surface disturbances also create seedbeds for invasive alien plants. Invasive plants compete for available soil moisture and light and create a continuous ground cover that eventually out-competes soil crust. Continuous fuels carried by invasive annual plant litter promote more intense and frequent fires in the low elevation vegetation communities that historically did not often burn (Schwinning et al. 2008). Soil crusts may survive some lower intensity fires and provide surface stability during post-fire recovery (Belnap et al. 2001). However, the greater frequency, intensity, and extent of fires driven by the increased litter of invasive annual plants degrade soil crust and expose it to replacement by invasive annuals.

4.1.3.4 Mapping Potential Biological Crust Abundance on the Colorado Plateau

Maps of potential crust abundance indicate the *potential* quantitative cover of biological crusts and major crust constituents (mosses, lichens, dark cyanobacterial crusts) across the Colorado Plateau. This modeling effort is an expansion to the entire region of a similar model done for the Grand Staircase-Escalante National Monument (Bowker et al. 2006). The work is relevant to both soil crust and soil stability as important REA conservation elements. A biological crust predictive model enables land managers to compare observed crust distribution with potential distribution, which serves as a surrogate for reference condition (Bowker et al. 2006). Such comparisons suggest appropriate management strategies as well as areas for preservation or restoration. The model provides a spatially explicit estimate of the crust abundance that would potentially 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 little affected by disturbance as possible. A least-disturbed state may or may not be equivalent to a historical reference condition; there is simply no information available to corroborate their similarity. The model will be useful for regional scale analyses, but it may or may not provide a reliable basis for determining the status of a particular location (e.g. a hectare plot). The map results estimate and map potential crust abundance, rather than current, existing crust abundance. Remote sensing techniques are currently being developed that may be able to capture information on existing crust cover at a regional scale.

Using existing field data, classification and regression tree models were prepared to estimate potential abundance of biological crusts across the Colorado Plateau. Model inputs included annual and seasonal precipitation, annual maximum and minimum temperature, 6 soil property indicators extracted from STATSGO and SSURGO soil data, field data on total crust cover from 593 sites, and field data for soil stability from 502 sites. The 6 soil property indicators were CaCO3, gypsum, sodium adsorption ratio, % sand, % clay, and plasticity index. Field data representing least-disturbed sites included: 1) sites in National Parks where grazing has been excluded for some time, 2) never-grazed relict sites, 3) range exclosures, 4) sites within grazed landscapes that are distant from water and/or high quality forage, or are geographically isolated. Sites with more than 5% exotic annuals were eliminated from the sample.

Using these inputs, Classification and Regression Tree (CART) models were constructed for specific groups of crust biota (total mosses, total lichens, dark cyanobacterial crusts, and early successional and late successional crust). These models were bootstrap validated, and their accuracy determined by plotting model predicting and observed values using linear regressions (as in Bowker et al. 2006). More details of the methodology and figures of regression tree models may be found in Appendix A.

Model outputs were generated at 800 m resolution. Modeled percent area estimates of total late successional biological crust (including biocrust lichens, mosses and dark cyanobacteria) ranged from less than 1 to slightly over 48 percent (Figure 4-6A). A companion early successional crust (i.e. light cyanobacterial and some physical crust cover) model showed results ranging from nearly 7 to slightly over 71 percent (Figure 4-6B). See Appendix A for maps of early and late successional crust cover relative to classes of landscape intactness.

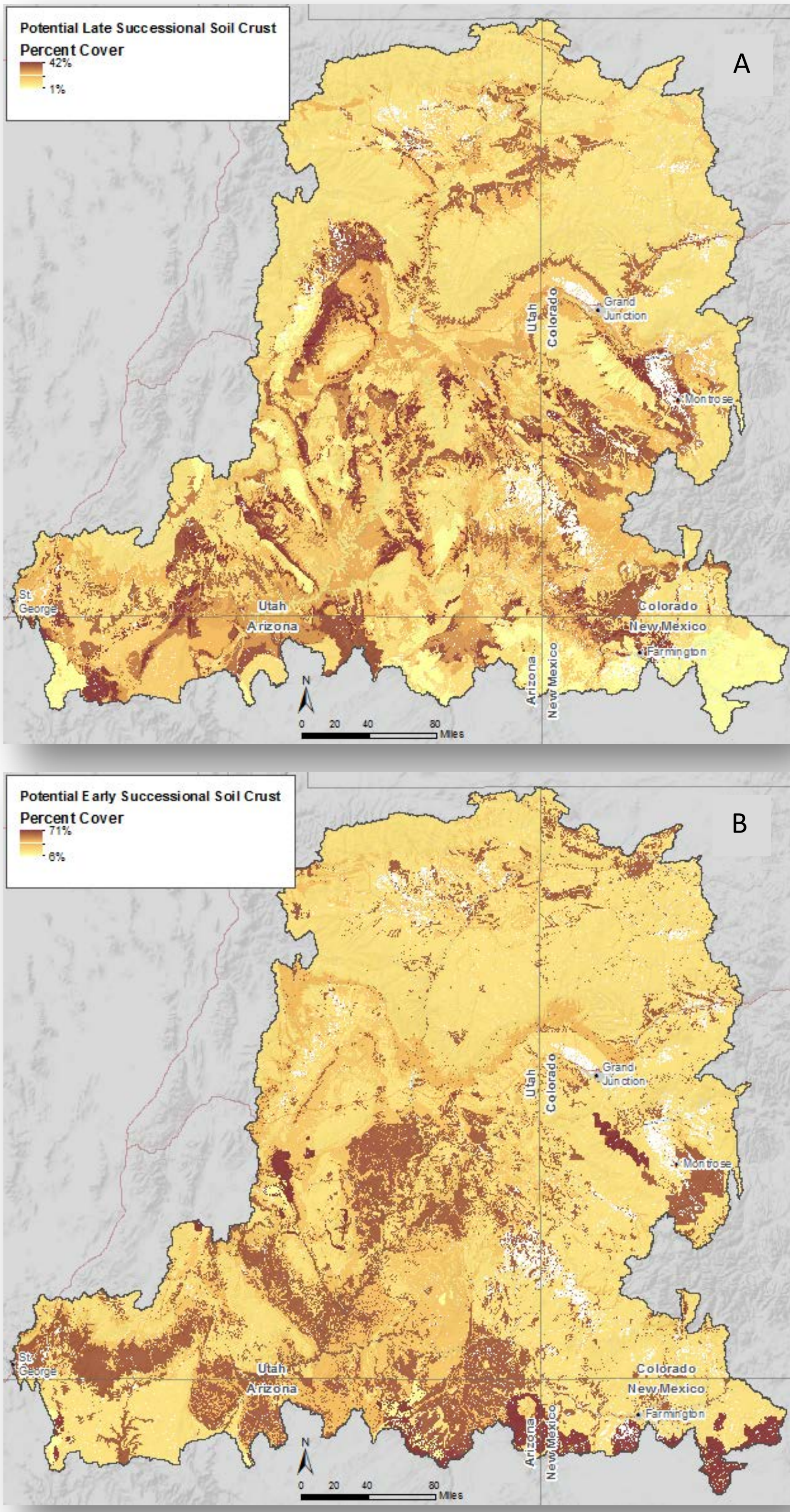


Figure 4-6. Map of late (A) and early (B) successional biological crust for the Colorado Plateau ecoregion. Model and portions of text contributed by M. Bowker and T. Arundel, U.S. Geological Survey.

4.1.3.5 Soil Crust Restoration

Restoration of soil crust in highly disturbed areas is known to be extremely slow, taking as long as 100s of years for recovery (Belnap et al. 2001). Soil crust must go through a succession process with cyanobacteria establishing first and cyanolichens arriving years later after the slow development of the microtopography favorable to lichen recruitment (Belnap et al. 2001, Davidson et al. 2002). Neff et al. (2005) observed that at sites that had been retired from grazing for 30 years there was still only spotty distribution of cyanobacteria with as yet little lichen or moss development. Bowker et al. (2006) suggest that recovery time may be shortened if restoration occurs in the cool, moist season and if crust organisms are provided with additional moisture, specific nutrients, and shade, taking care to avoid conditions that would promote the invasion of exotic annuals. As noted earlier, soil crust species richness is higher in gypsiferous soils, non-calcareous sandy soils, and limestone-derived soils and lower (or minimal) in fine shale-derived soil (Bowker and Belnap 2008); restoration efforts are more likely to be successful in the former soil types.

4.1.4 Aquatic Resources of Concern

Surface and Groundwater Management Questions

MQ B1 Where are lotic and lentic surface waterbodies, livestock and wildlife watering tanks, and artificial water bodies?

MQ B2 Where are perennial streams and stream reaches?

MQ B3 What are seasonal maximum and minimum discharges for the Colorado River and major tributaries at gaging stations?

MQ B4 Where are the alluvial aquifers and their recharge areas (if known)?

MQ B6 Where are aquatic systems listed on 303(d) with degraded water quality or low macro-invertebrate diversity?

MQ B7 What is the location/distribution of aquatic biodiversity sites?

MQJ4 Where are aquatic/riparian areas with potential to change from climate change?

The value of water resources to desert dwellers is obvious and inestimable. The importance of water resources to the Colorado Plateau REA process is reflected in the number of water-related management questions (see callout box above) and the selection of three fish species conservation elements, razorback sucker, flannelmouth sucker, and Colorado River cutthroat trout (discussed in Section 4.2.1), to represent the region's aquatic ecosystems. In addition, aquatic resources were represented in REA data and results as aquatic sites of conservation concern (TNC portfolio sites) and ecosystem functions and services: springs and seeps, lakes and artificial waterbodies, wetlands, and riparian areas. Natural lake habitats are limited in the region, but presently, 400 dams and reservoirs on the Colorado River and its tributaries have created permanent standing water habitat (Pool et al. 2010). Results for management questions MQ B2 and MQ B3 are presented below; results for the rest of the aquatic management questions may be found in Appendix A.

In arid and semi-arid regions, streams experience extreme variations in water flow, permanence, and sediment transport that produce braided, meandering, or anastomosing channels (Hughes et al. 2011). Stream flows range from perennial (mountain source or spring-fed), to spatially intermittent (flowing only where bedrock forces ground water to the surface), temporally intermittent (flowing only during the wet or snow melt seasons), and ephemeral (flowing only during major storm events). Because of this natural variability, cumulative impacts such as human water consumption and channel dewatering, climate change, or simple mapping error, a high proportion (>70%) of stream length in arid and semi-arid regions in the western U.S. that was historically mapped as permanent is now temporary (Stoddard et al. 2005b, Figure 4-7, management question B2). Statewide, 79% of Utah streams and 68% of Colorado streams are intermittent or ephemeral (Levick et al. 2008). Carlisle et al. (2011) also reported, in an assessment of streamflow alteration (covering a time period of 1980–2007), that >50% of the stream length in arid USA regions experienced reduced base and flood flows relative to historic levels. Diminished flow was the primary predictor of biological integrity for aquatic species with the likelihood of impairment increasing as flows diminished. In an assessment of stream resources in 12 western states, Stoddard et al. (2005a) estimated that 50% and 48% of stream length in the xeric portions of these states had highly disturbed aquatic vertebrate and macroinvertebrate condition, respectively. Climate change is projected to result in mean air temperature increases, increased drought conditions, earlier and smaller spring peak flows, and lower summer flows (Cayan et al. 2001, Seager et al. 2007). As discussed earlier in the discussion of dust on snow, changes in spring snowmelt and peak flows from climate change will be added to those already occurring in the southern Rocky Mountains from wind-borne dust on snow (Painter et al. 2010). Although fluctuating flows, high turbidity, and periodic flooding and drought are important natural processes in streams draining arid and semi-arid regions, the increasing amplitude and variability of these processes created by climate change and continued human pressures threaten to reduce and fragment aquatic habitats even further and stress native species beyond their ability to adapt.

Because of the region's aridity and high demand for water, most lotic and lentic ecosystems in the Colorado Plateau ecoregion have been degraded by humans to some degree. The entire region is drained by the Colorado River, one of the most-altered drainages in North America (Ohmart et al. 1988, Hughes et al. 2005, Wegner 2008). Thirty million people in the upper and lower Colorado River Basin depend on the Colorado River and its tributaries for their water supply; fluctuations in water yield occur from variability in precipitation, runoff, snow pack, spring snow melt (Table 4-2, management question B3). The river and its tributaries are highly regulated and the water over-allocated. The original Colorado River Compact of 1922 allocated 17.5 million acre-feet of water each to the Upper and Lower Colorado River Basins. However, the long-term mean gaged flow at Lees Ferry (1906–2004) is about 15.1 million acre-feet, resulting in a chronic over-allocation, the effects of which have been delayed because the Upper Basin states do not claim their full allocation (NOAA 2012). The extra water is delivered downstream to the Lower Basin states except in severe drought years. According to the Upper Colorado Basin Compact of 1948, of the 7.5 million acre-feet of water allotted to the upper basin, Colorado receives, 51.75%, Utah 23%, and New Mexico 11.25% (Figure 4-8). In each state in the Colorado Plateau ecoregion, more water is consumed by agriculture for irrigation than municipal or industrial uses; any irrigation water that is returned to the rivers or streams is laden with leached salts and agricultural chemicals. In a study examining the effects of agriculture on fish in the western U.S., Moore et al. (1996) reported that the number of fish species listed under the Endangered Species Act per county was positively correlated with the level of irrigated agriculture in that county.

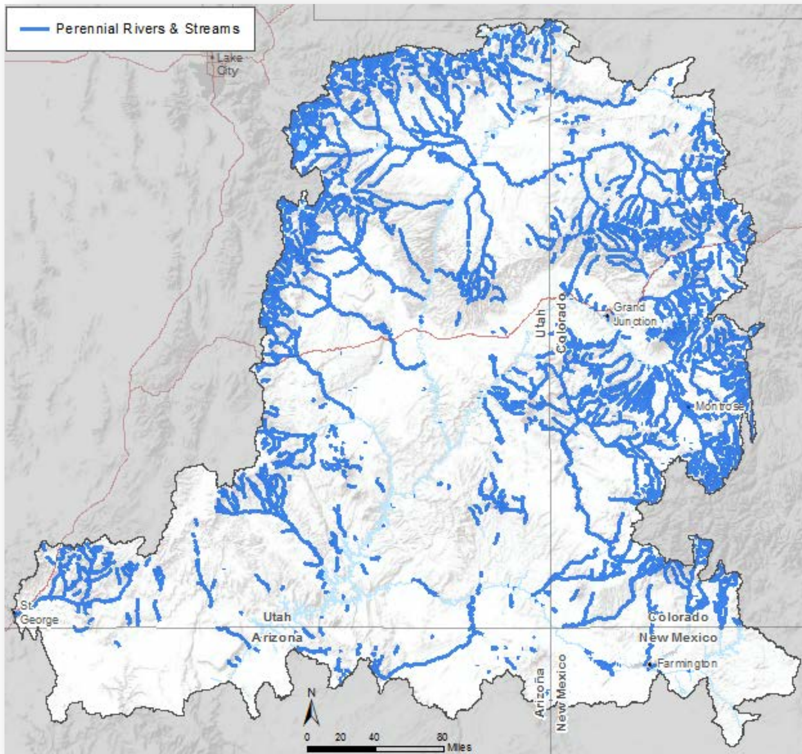


Figure 4-7. Map for management questions B2 shows perennial streams in the Colorado Plateau ecoregion. Mainstem Colorado and Green rivers are in light blue. Data from the National Hydrography Dataset typically over-represent perennial streams because of mapping error or loss of perennial streams over time (water consumption, climate change).

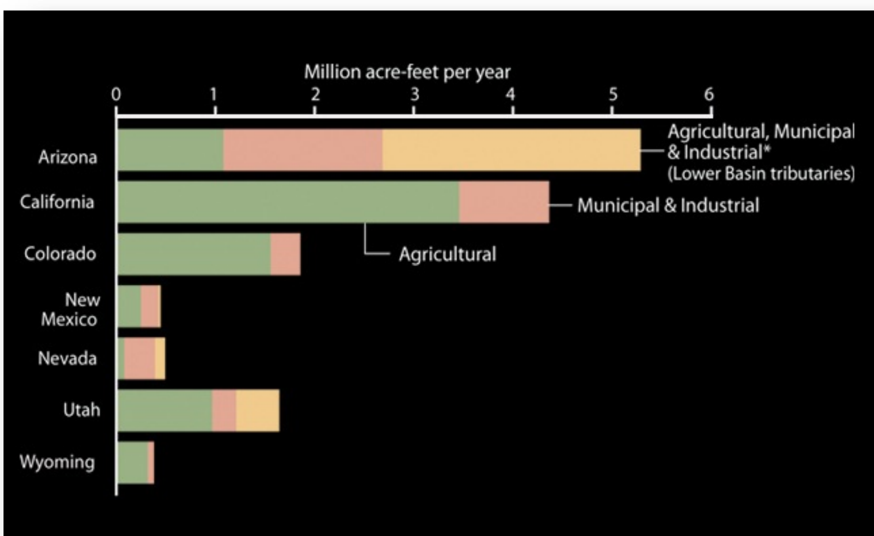


Figure 4-8. Water consumption of states of the upper and lower Colorado River Basin for agriculture (green), municipal and industrial use (pink), and all usage from Colorado River tributaries (yellow, data not recorded by usage class). Data from U.S. Bureau of Reclamation.
National Geographic website http://www.savethecolorado.org/map.php

Table 4.2. Average seasonal maxima and minima for gaging stations on the Colorado River and major tributaries recording 7–102 years of records from various stations through 9-30-2010 (Source weblink: http://waterdata.usgs.gov/nwis. Figures in cubic feet/second (cfs) rounded to the nearest cfs. Table answers management question MQ B3: *What are seasonal maximum and minimum discharges for the Colorado River and major tributaries at gaging stations?*

Gaging Station Location	SPMN	SPMX	SUMN	SUMX	FMN	FMX	WMN	WMX
GREEN RIVER NEAR JENSEN, UT	2481	23991	559	11378	430	5089	604	6220
YAMPA RIVER AT DEERLODGE PARK, CO.	1670	15381	56	4485	161	1392	224	1643
DUCHESNE RIVER NEAR RANDLETT, UT	19	4570	7	2930	31	1560	47	1264
WHITE RIVER NEAR WATSON, UTAH	301	3581	79	2886	207	1135	190	1280
PRICE RIVER AT WOODSIDE, UT	8	1646	1	1299	11	731	13	271
COLO RIVER NR PALISADE CO	945	13246	161	9551	839	2621	1130	2500
SAN RAFAEL RIVER NEAR GREEN RIVER, UT	4	1768	0	1391	3	885	11	449
GUNNISON RIVER GRAND JUNCTION, CO.	541	18088	174	9474	361	3671	498	3859
COLORADO RIVER NEAR CISCO, UT	2041	43002	991	25958	1565	9093	1704	7086
DOLORES RIVER NEAR CISCO, UT	110	6132	16	1617	94	895	91	591
DIRTY DEVIL R NR HANKSVILLE, UT	9	562	0	1218	21	1434	36	342
VIRGIN RIVER NEAR BLOOMINGTON, UT	25	1938	10	644	42	722	56	1997
PARIA RIVER AT LEES FERRY, AZ	3	165	2	939	5	502	6	354
SAN JUAN RIVER AT FOUR CORNERS, CO	536	9613	283	6978	518	3853	537	3994
MANCOS RIVER NEAR TOWAOC, CO.	0	700	0	465	0	264	2	153
ANIMAS RIVER AT FARMINGTON, NM	124	5806	8	4292	108	2042	142	861

SPMN=spring minimum; SPMX=spring maximum; SUMN=summer minimum; SUMX=summer maximum; FMN=fall minimum; FMX=fall maximum; WMN=winter minimum; WMX=winter maximum.

Metal and coal mining occurs over relatively small areas of the region compared to irrigated agriculture; however, mining also requires large quantities of water. Mining increases sediment loads to streams, alters channel structure and flow regimes, and frequently delivers highly toxic effluents to surface waters (Woody et al. 2010). Renewed interest in uranium mining occurred recently in the ecoregion when the price of uranium climbed rapidly from $9.70 to over $90 per pound from 2002–2007 (Harding 2007). Presently, the U.S. Nuclear Regulatory Commission requires that mining companies file an approved financial assurance plan to ensure cleanup of a uranium mining site prior to commencing operation. However, a publicly-financed cleanup process continues on the millions of tons of uranium tailings remaining in the region from earlier abandoned mines. Various cleanup operations have occurred over the last 25 years to remove tailings from the Atlas Mine near Moab, Utah (USNRC 2011) and mines near Monticello and Uravan in Colorado. Data for existing and authorized uranium mines were included in the REA for the development models.

In recent years, oil companies have increased the use of hydraulic fracturing or fracking in the region to extract oil and gas from formations previously seen as unprofitable or difficult to drill. During fracking, water and chemicals are pumped into the gas or oil-bearing rock to break the formation to release the gas or oil. Fifty thousand to 350,000 gallons of water may be required to fracture a single well in a coalbed formation while two to five million gallons of water may be necessary to fracture one horizontal well in a shale formation. Fresh water from local sources is generally used for fracking and this water is lost to other uses in the drilling process. Besides concern over the heavy use of water for fracking in arid and semi-arid regions, the public has expressed concerns that the injected chemicals—and naturally occurring elements such as local metals and radionuclides—may subsequently seep into groundwater and drinking water supplies (Kargbo et al. 2010, USEPA 2010). While the chemicals used in fracking are proprietary, lists of chemicals known to have been used at various stages of the fracking process have been developed by state agencies and other interested parties (Earthworks 2011). The Environmental Protection Agency plans to release a study on the safety of water supplies in oil and gas drilling regions in 2012.

Mining for oil shale has been a latent resource issue since the 1980's. Oil shale beds exist in the Uinta Basin in Utah and the Piceance Basin in northwestern Colorado. Oil shale production uses large amounts of water; for an oil shale field producing 2.5 million barrels per day, water use is estimated at between 105 and 315 million gallons per day for direct industry use and 58 million gallons per day for industry-related municipal use (DOE 2012). In 2011, the Secretary of the Interior called for a review of oil shale plans based on latest research and information on water supply and demand. Oil shale lease data (dated 2008) were used in the REA in models for potential energy development (Section 5.2); newer data became available early in 2012, too late to be incorporated into this REA.

Besides diminished instream flow in streams, altered flow regimes created by dams, channelization, canal systems, and water diversions are associated with increased homogenization of fish assemblages through extirpations of native fishes coupled with increased dominance by alien fishes (Williams et al. 1985, Stanford 1994, Hughes et al. 2005, Olden et al. 2006, Poff et al. 2007). Native fish species in the region have declined in range and abundance since the early 20th century. Of the 52 fish species that occur in the upper Colorado River Basin, just 13 species are native (USFWS 2011). Two of the selected REA fish species, the razorback sucker (*Xyrauchen texanus*) and flannelmouth sucker (*Catostomus latipinnis*), have similar habitat requirements in larger rivers and tributaries, although the flannelmouth sucker has a somewhat broader elevational range than the razorback sucker. Both species are adapted to seasonal spring flooding and use of backwater habitats for spawning. Today river flow regulation, channelization, levees, and dikes have eliminated spring flooding, and dams have created barriers to upstream movement (Chart and Bergerson 1992, Rees et al. 2005, USFWS 2011). Cold water releases from reservoirs reduce recruitment and larval growth (Clarkson and Childs 2000). Predation by nonnative fish such as northern pike and smallmouth bass has a devastating effect on recruitment, reducing razorback sucker populations to mostly older adults (USFWS 2011).

Alien invasive species have been ranked as the second or third most important threat to the biodiversity of native fishes (Miller et al. 1989, Hughes et al. 2005, Reed and Czech 2005). Lomnicky et al. (2007) estimated that alien aquatic vertebrates occurred in 74 ±14% of Utah streams, 86 ±8% of Colorado streams, and westwide, in 83 ±6% of large rivers. Aliens affect native fish assemblages through competition (Carpenter 2005) and predation (Li and Moyle 1981, Meffee 1984, Dunham et al. 2004). Nonnative predators may entirely eliminate a native fish assemblage in a particular catchment—even in an otherwise unmodified watershed—if the native fish are stressed or experiencing low recruitment, as during a drought (Probst et al. 2008). Alien invasive aquatic macroinvertebrates can be problematic as well. Stoddard et al. (2005a) estimated that alien crayfish occurred in 7 ±3% and Asian clam occurred in 6 ±3% of the stream length in xeric regions of the western U.S. Although their occurrence probabilities were low, when present, the crayfish and clam were associated west-wide with a doubling or tripling of the risk of having poor vertebrate and macroinvertebrate biological integrity scores (Stoddard et al. 2005a).

Thus, while the retention or mimicking of natural hydrologic regimes is essential for maintaining native fish assemblages (Stanford 1994, Poff et al. 1997, Probst and Gido 2004), a reduction in competition from nonnative species is just as important (Eby et al. 2003, Mueller 2005, Propst et al. 2008). A natural flow regime allows connectivity and genetic diversity, but it also allows nonnative fish easy access to native refugia (Propst et al. 2008). Recovery activities for native aquatic species includes managing water releases from dams to benefit native species life cycles, acquisition of bottomlands and easements, breaching of levees, stocking hatchery-raised threatened and endangered species, managing nonnative species introductions, and conducting targeted nonnative species control (Rees et al. 2005, Mueller 2005, USFWS 2011).

Markedly altered flow regimes may also eliminate native riparian vegetation (Rood and Mahoney 1990, Lytle and Merritt 2004), change riparian community composition (Busch & Smith 1995, Merritt and Wohl 2006, Stromberg et al. 2007, Merritt & Poff 2010), species richness (Nilsson et al. 1991) and productivity (Stromberg and Patten 1990, Molles et al. 1998). Although historically riparian habitats composed about 1% of the land area of the western states, ground water pumping and a broad range of human disturbances have resulted in the loss of >90% of the region's wetlands and native riparian woodlands (Krueper 1996). As much as 80% of all vertebrates use the remaining riparian habitats for cover and foraging, and over 50% of southwestern bird species use riparian woodland and shrubland for nesting (Knopf et al. 1988, Krueper 1996). Xeroriparian habitats also attract nesting birds (Levick et al. 2008). For a full discussion of riparian issues, see the Tamarisk Case Study Insert.

A fuzzy logic model was developed for aquatic intactness (reported by 5th level HUC) similar to the one done for terrestrial landscape intactness (in Chapter 3, Section 3.2.3) that is later used to assess status for aquatic conservation elements. The model includes 10 primary inputs with three major contributors—hydrologic alteration, land & water quality, and road impacts, represented as intermediate results in purple below (Figure 4-9).

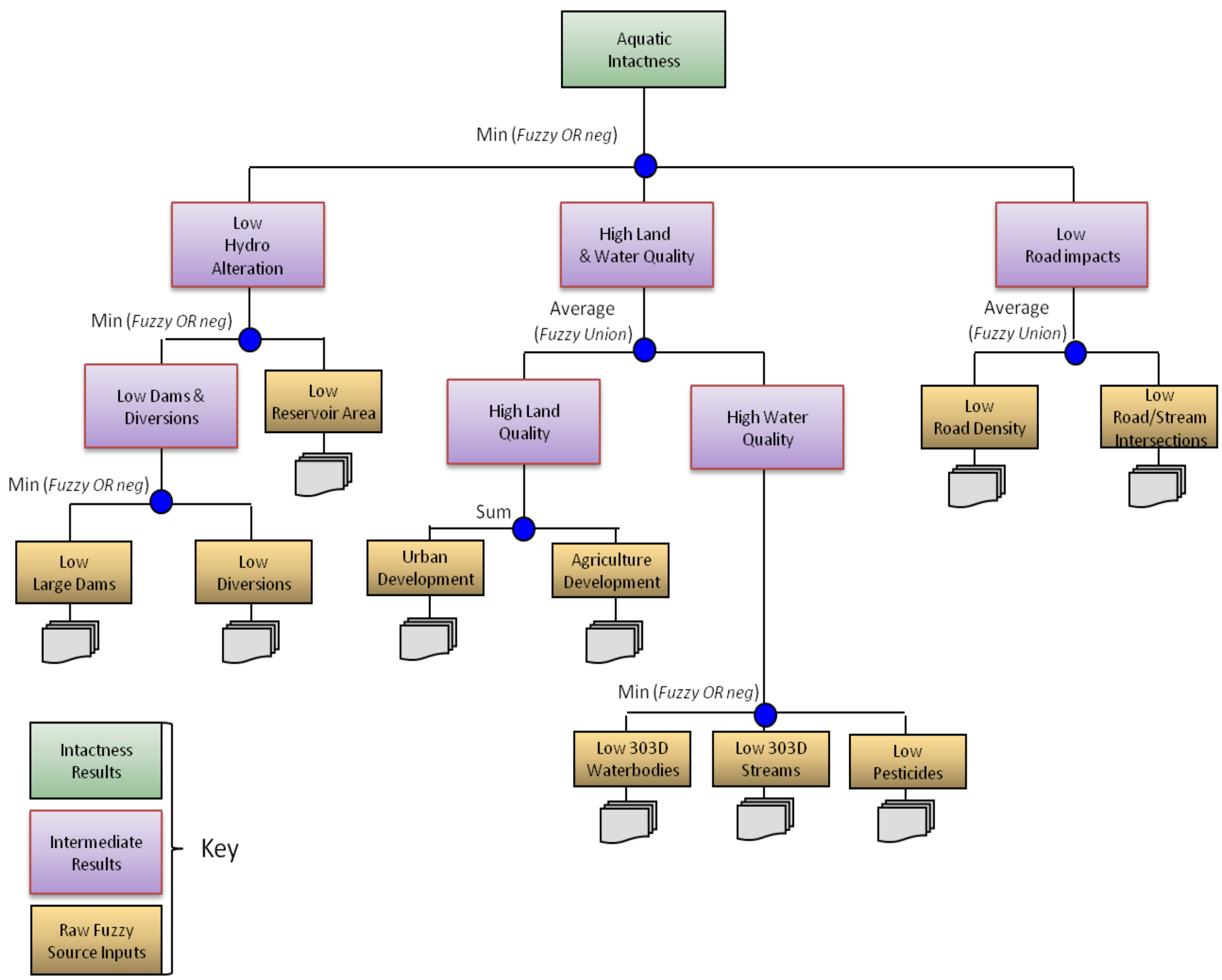


Figure 4-9. Fuzzy logic model for aquatic intactness in the Colorado Plateau ecoregion. Gold boxes represent raw primary data input and purple boxes represent intermediate results (Figure 4-10).

Intermediate result maps for the 3 major contributors highlighting the aquatic degradation drivers show widespread aquatic impacts throughout the ecoregion (Figure 4-10). Darker color is higher on a relative scale, meaning A) higher hydrologic alteration, B) higher land and water quality, and C) higher road impacts. Final aquatic intactness results are provided in Section 4.2.1. Appendix A contains specific results for each stated aquatic management question listed at the beginning of this section.

Figure 4-10. Intermediate results maps for (A) hydrologic alteration, (B) land & water quality, and (C) road impacts for aquatic intactness in the Colorado Plateau ecoregion. Darker color is higher on a relative scale.

4.1.5 References Cited

Allen, C.D., and D.D. Breshears. 1998. Drought-induced shift of a forest-woodland ecotone. Rapid landscape response to climate variation. *Proceedings of the National Academy of Sciences* 95:14,839–14,842.

BLM (Bureau of Land Management). 1987. Salinity control on BLM-administered public lands in the Colorado River Basin: A report to Congress. BLM/Y A/PT-87/019+7000, Washington, D.C.

Belnap, J. 1994. Cryptobiotic soil crusts: Basis for arid land restoration (Utah). *Restoration and Management Notes* 12(1):85–86.

Belnap, J. 2002. Nitrogen fixation in biological soil crusts from southeast Utah, USA. *Biology and Fertility of Soils* 35:128–135.

Belnap, J., and D.A. Gillette. 1998. Vulnerability of desert biological soil crusts to wind erosion: The influences of crust development, soil texture, and disturbance. *Journal of Arid Environments* 39:133–142.

Belnap, J., J.H. Kaltenecker, R. Rosentreter, J. Williams, S. Leonard, and D. Eldridge. 2001. Biological soil crusts: Ecology and management. Bureau of Land Management, National Science and Technology Center. Technical reference 1730-2. http://www.soilcrust.org/crust.pdf. Accessed 3/21/11.

Belnap, J., R.L. Reynolds, M.C. Reheis, S.L. Phillips, F.E. Urban, and H.L. Goldstein. 2009. Sediment losses and gains across a gradient of livestock grazing and plant invasion in a cool, semi-arid grassland, Colorado Plateau, USA. *Aeolian Research* 1:27–43.

Boettinger, J.L. 2012. Soils of Utah. Utah State University, http://extension.usu.edu/utahrangelands/files/uploads/RRU_Section_Six.pdf. Accessed 1/12.

Bowker, M.A., and J. Belnap. 2008. A simple classification of soil types as habitats of biological soil crusts on the Colorado Plateau, USA. *Journal of Vegetation Science* 19: 831–840.

Bowker, M.A., J. Belnap, and M.E. Miller. 2006. Spatial modeling of biological soil crusts to support rangeland assessment and monitoring. *Rangeland Ecology and Management* 59(5):519–529.

Bowker, M.A., M.E. Miller, J. Belnap, T.D. Sisk, and N.C. Johnson. 2008. Prioritizing conservation effort through the use of biological soil crusts as ecosystem function indicators in an arid region. *Conservation Biology* 22: 1533–1543.

Brooks, M.L. 2008. Plant invasions and fire regimes. Pages 33–46 *in* Zouhar, K., J. Kapler Smith, S. Sutherland, and M.L. Brooks, Wildland fire in ecosystems: Fire and non-native invasive plants. USDA Forest Service General Technical Report RMRS-GTR-42-vol. 6, U.S. Forest Service, Rocky Mountain Research Station, Ogden, Utah. 355 p.

Busch, D.E., and S.D. Smith. 1995. Mechanisms associated with decline of woody species in riparian ecosystems of the southwestern U.S. *Ecological Monographs* 65:347–370.

Carlisle, D.M., D.M. Wolock, and M.R. Meador. 2011. Alteration of streamflow magnitudes and potential ecological consequences: A multiregional assessment. *Frontiers in Ecology and the Environment.* 9:264–270.

Carpenter, J. 2005. Competition for food between an introduced crayfish and two fishes endemic to the Colorado River Basin. *Environmental Biology of Fishes* 72:335–342.

Cayan, D.R., K.T. Redmond, and L.G. Riddle. 1999. ENSO and hydrologic extremes in the western United States. *Journal of Climate* 12:2881–2893.

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

Chaudhary, V.B., M.A. Bowker, T.E. O'Dell, J.B. Grace, A.E. Redman, M.C. Rillig, and N.C. Johnson. 2009. Untangling the biological contributions to soil stability in semiarid shrublands. *Ecological Applications* 19(1):110–122.

Chapman, S.S., G.E. Griffith, J.M. Omernik, A.B. Price, J. Freeouf, and D.L. Schrupp. 2006. Ecoregions of Colorado, U.S. Geological Survey, Reston, Virginia (map scale 1:1,200,000).

Chart, T.E., and E.P. Bergersen. 1992. Impact of mainstream impoundment on the distribution and movements of the resident flannelmouth sucker (Catostomidae: *Catostomus latipinnis*) population in the White River, Colorado. *Southwestern Naturalist* 37(1):9–15.

Clarkson, R.W., and M.R. Childs. 2000. Temperature effects of hypolimnial-release dams on early life stages of Colorado River basin big-river fishes. *Copeia* 2:402–412.

Davidson, D.W., M. Bowker, D. George, S.L. Phillips, and J. Belnap. 2002. Treatment effects on performance of N-fixing lichens in disturbed soil crusts of the Colorado Plateau. *Ecological Applications* 12(5):1391–1405.

Davis, J. 2011. What are the roots of geobotany? Utah Geological Survey, Utah.gov Services, Salt Lake City. http://geology.utah.gov/surveynotes/gladasked/gladgeobotany.htm. Accessed 1/12.

DOE (Department of Energy). 2012. Fact sheet: Oil shale water resources. Department of Energy, Office of Petroleum Reserves, Strategic Unconventional Fuels, Washington, D.C. http://fossil.energy.gov/programs/reserves/npr/Oil_Shale_Water_Requirements.pdf

Dunham, J.B., D.S. Pilliod, and M.K. Young. 2004. Assessing the consequences of nonnative trout in headwater ecosystems in western North America. *Fisheries* 29(6):18–26.

Earthworks. 2011. Hydraulic fracturing 101. http://www.earthworksaction.org/issues/detail/hydraulic_fracturing_101. Accessed 1/12.

Eby, L.A., W.F. Fagan, and W.L. Minckley. 2003. Variability and dynamics of a desert stream community. *Ecological Applications* 13:1566–1579.

Getz, H.L., and W.L. Baker. 2008. Initial invasion of cheatgrass into burned pinyon-juniper woodlands in western Colorado. *American Midland Naturalist* 159:489–497.

Harding, J. 2007. Economics of new nuclear power and proliferation risks in a carbon-constrained world. Nonproliferation Policy Education Center, Arlington, Virginia. http://www.npolicy.org/files/20070600-Harding-EconomicsNewNuclearPower.pdf.

Housman, D.C., H.H. Powers, A.D. Collins, and J. Belnap. 2006. Carbon and nitrogen fixation differ between successional stages of biological soil crusts in the Colorado Plateau and Chihuahuan Desert. *Journal of Arid Environments* 66(4):620–634.

Hughes, R.M., Kaufmann, P.R., and M.H. Weber. 2011. Strahler order versus stream size. *Journal of the North American Benthological Society* 30:103–121.

Hughes, R.M., J.N. Rinne, and B. Calamusso. 2005. Historical changes in large river fish assemblages of the Americas: A synthesis. Pages 603–612 *in* Rinne, J.N., R.M. Hughes, and B. Calamusso (eds.), Historical changes in large river fish assemblages of the Americas, Symposium 45, American Fisheries Society, Bethesda, Maryland.

Jackson, L. 2005. Mancos shale literature review on the Colorado Plateau. Resource Note No. 80, Bureau of Land Management, Moab Field Office, Moab, Utah.

Kargbo, D.M., R. Wilhelm, and D.J. Campbell. 2010. Natural gas plays in the Marcellus Shale: Challenges and potential opportunities. *Environmental Science and Technology* 44:5679–5684.

Knopf, F.L., R.R. Johnson, T. Rich, F.B. Samson, and R.C. Szaro. 1988. Conservation of riparian ecosystems in the United States. *Wilson Bulletin* 100:272–284.

Krueper, D.J. 1996. Effects of livestock management on southwestern riparian ecosystems *in* Desired future conditions for southwestern riparian ecosystems: Bringing interests and concerns together. Shaw, D.W., and D.M. Finch (technical coordinators), Sept. 18–22, 1995, Albuquerque, New Mexico. General Technical Report RM-GTR-272, U.S. Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado. 359 p.

Levick, L., J. Fonseca, D. Goodrich, M. Hernandez, D. Semmens, J. Stromberg, R. Leidy, M. Scianni, D.P. Guertin, M. Tluczek, and W. Kepner. 2008. The ecological and hydrological significance of ephemeral and intermittent streams in the arid and semi-arid American Southwest. U.S. Environmental Protection Agency and USDA/ARS Southwest Watershed Research Center, EPA/600/R-08/134, ARS/233046, Tucson, Arizona. 116 pp.

Li, H.W., and P.B. Moyle. 1981. Ecological analysis of species introductions into aquatic ecosystems. *Transactions of the American Fisheries Society* 110:772–782.

Lomnicky, G.A., T.R. Whittier, R.M. Hughes, and D.V. Peck. 2007. Distribution of nonnative aquatic vertebrates in western U.S. streams and rivers. *North American Journal of Fisheries Management* 27:1082–1093.

Lytle, D.A., and D.M. Merritt. 2004. Hydrologic regimes and riparian forests: A structured population model for cottonwood. *Ecology* 85:2493–2503.

Meffee, G.K. 1984. Effects of abiotic disturbance on coexistence of predator-prey fish species. *Ecology* 65:1525–1534.

Merritt, D.M., and N.L. Poff. 2010. Shifting dominance of riparian *Populus* and *Tamarix* along gradients of flow alteration in western North American rivers. *Ecological Applications* 20:135–152.

Merritt, D.M., and E.E. Wohl. 2006. Plant dispersal along rivers fragmented by dams. *River Research and Applications* 22:1–26.

Miller, M.E., R.T. Belote, M.A. Bowker, and S.L. Garman. 2011. Alternative states of a semiarid grassland ecosystem: Implications for ecosystem services. *Ecosphere* 2(5):1–18.

Miller, R.R., J.D. Williams, and J.E. Williams. 1989. Extinctions of North American fishes during the past century. *Fisheries* 14(6): 22–38.

Molles, M.C., C.S. Crawford, L.M. Ellis, H.M. Valett, and C.N. Dahm. 1998. Managed flooding for riparian ecosystem restoration. *BioScience* 48:749–756.

Moore, M.R., A. Mulville, and M. Weinberg. 1996. Water allocation in the American West: Endangered fish versus irrigated agriculture. *Natural Resources Journal* 36:319–357.

Mueller, G. A. 2005. Predatory fish removal and native fish recovery in the Colorado River mainstem: What have we learned? *Fisheries* 30:10–19.

Munson, S.M., J. Belnap, and G.S. Okin. 2011. Responses of wind erosion to climate-induced vegetation changes on the Colorado Plateau. *Proceedings of the National Academy of Sciences* 108:3854–3859.

NMDGF (New Mexico Department of Game and Fish). 2006. Comprehensive Wildlife Conservation Strategy for New Mexico. New Mexico Department of Game and Fish, Santa Fe. 526 p. + appendices.

Neff, J.C., A.P. Ballantyne, G.L. Farmer, N.M. Mahowald, J.L. Conroy, C.C. Landry, J.T. Overpeck, T.H. Painter, C.R. Lawrence, and R.L. Reynolds. 2008. Increasing eolian dust deposition in the western United States linked to human activity. *Nature Geoscience* 1(3):189–195.

Neff, J.C., R.L. Reynolds, J. Belnap, and P. Lamothe. 2005. Multi-decadal impacts of grazing on soil physical and biogeochemical properties in southeast Utah. *Ecological Applications* 15(1):87–95.

Nilsson, C., A. Ekblad, M. Gardfjell, and B. Carlberg. 1991. Long-term effects of river regulation on river margin vegetation. *Journal of Applied Ecology* 28:963–987.

NOAA (National Oceanic and Atmospheric Administration). 2012. Western water assessment: The Compact and Lee's Ferry. http://wwa.colorado.edu/treeflow/lees/compact.html.

Ohmart, R.D., B.W. Anderson, and W.C. Hunter. 1988. The ecology of the lower Colorado River from Davis Dam to the Mexico-United States international boundary: A community profile. U.S. Fish and Wildlife Service Biological Report 85 (7.19). Center for Environmental Studies, Arizona State University, Tempe, Arizona. 296 p.

Olden, J. D., N. L. Poff, and K. R. Bestgen. 2006. Life-history strategies predict fish invasions and extirpations in the Colorado River Basin. *Ecological Monographs* 76:25–40.

Painter, T.H., J. Deems, J. Belnap, A. Hamlet, C.C. Landry, and B. Udall. 2010. Response of Colorado River runoff to dust radiative forcing in snow. Proceedings of the National Academy of Sciences, published September 20, 2010,doi:10.1073/pnas.0913139107.

Poff, N.L., J.D. Allan, M.B. Bain, J.R. Karr, K.L. Prestegaard, B.D. Richter, R.E. Sparks, and J.C. Stromberg. 1997. The natural flow regime: A paradigm for river conservation and restoration. *BioScience* 47:769–784.

Poff, N.L., J.D. Olden, D.M. Merritt, and D.M. Pepin. 2007. Homogenization of regional river dynamics by dams and global biodiversity implications. *Proceedings of the National Academy of Sciences of the United States of America* 104:5732–5737.

Pool, T.K., J.D. Olden, J.B. Whittier, and C.P. Paukert. 2010. Environmental drivers of fish functional diversity and composition in the lower Colorado River Basin. *Canadian Journal of Fisheries and Aquatic Sciences* 67:1791–1807.

Prior-Magee, J.S., K.G. Boykin, D.F. Bradford, W.G. Kepner, J.H. Lowry, D.L. Schrupp, K.A. Thomas, and B.C. Thompson (eds). 2007. Southwest Regional Gap Analysis Project final report. U.S. Geological Survey, Gap Analysis Program, Moscow, Idaho.

Propst, D. L., and K. B. Gido. 2004. Responses of native and nonnative fishes to natural flow regime mimicry in the San Juan River. *Transactions of the American Fisheries Society* 133:922–931.

Propst, D.L., K.B. Gido, and J.A. Stefferud. 2008. Natural flow regimes, nonnative fishes, and native fish persistence in arid-land river systems. *Ecological Applications* 18:1236–1252.

Reed, K.M., and B. Czech. 2005. Causes of fish endangerment in the United States, or the structure of the American economy. *Fisheries* 30(7):36–38.

Rees, D.E., J.A. Ptacek, R.J. Carr, and W.J. Miller. 2005. Flannelmouth Sucker (*Catostomus latipinnis*): A technical conservation assessment. U.S. Forest Service, Rocky Mountain Region. Available: http://www.fs.fed.us/r2/projects/scp/assessments/flannelmouthsucker.pdf. Accessed 1/12.

Rood, S.B., and J.M. Mahoney. 1990. Collapse of riparian poplar forests downstream from dams in western prairies: Probable causes and prospects for mitigation. *Environmental Management* 14:451–464.

Schwinning, S., J. Belnap, D.R. Bowling, and J.R. Ehleringer. 2008. Sensitivity of the Colorado Plateau to change: Climate, ecosystems, and society. *Ecology and Society* 13(2): 28. [online] URL: http://www.ecologyandsociety.org/vol13/iss2/art28/.

Seager, R., M. Ting, I. Held, Y. Kushnir, J. Lu, G. Vecchi, H. Huang, N. Harnik, A. Leetmaa, N. Lau, C. Li, J. Velez, and N. Naik. 2007. Model projections of an imminent transition to a more arid climate in southwestern North America. *Science* 316:1181–1184.

Shinneman, D.J., and W.L. Baker. 2009. Historical fire and multidecadal drought as context for piñon–juniper woodland restoration in western Colorado. *Ecological Applications* 19:1231–1245.

Stanford, J.A. 1994. Instream flows to assist the recovery of endangered fishes of the Upper Colorado River Basin. Biological Report 24. U.S. Fish and Wildlife Service, Denver, Colorado.

Stoddard, J.L., D.V. Peck, S.G. Paulsen, J. Van Sickle, C.P. Hawkins, A.T. Herlihy, R.M. Hughes, P.R. Kaufmann, D.P. Larsen, G. Lomnicky, A.R. Olsen, S.A. Peterson, P.L. Ringold, and T.R. Whittier. 2005a. An ecological assessment of western streams and rivers. EPA 620/R-05/005, U.S. Environmental Protection Agency, Washington, DC.

Stoddard, J.L., D.V. Peck, A.R. Olsen, D.P. Larsen, J. Van Sickle, C.P. Hawkins, R.M. Hughes, T.R. Whittier, G. Lomnicky, A.T. Herlihy, P.R. Kaufmann, S.A. Peterson, P.L. Ringold, S.G. Paulsen, and R. Blair. 2005b. Environmental Monitoring and Assessment Program (EMAP) western streams and rivers statistical summary. EPA 620/R-05/006, Office of Research and Development, U.S. Environmental Protection Agency, Washington, DC.

Stromberg, J.C., S.J. Lite, R. Marler, C. Paradzick, P.B. Shafroth, D. Shorrock, J.M. White, and M.S. White. 2007. Altered stream-flow regimes and invasive plant species: The *Tamarix* case. *Global Ecology and Biogeography* 16:381–393.

Stromberg, J.C., and D.T. Patten. 1990. Riparian vegetation instream flow requirements: A case study from a diverted stream in the eastern Sierra Nevada, California. *Environmental Management* 14:185–194.

Swetnam, T.W., and J.L. Betancourt 1998. Mesoscale distribution and ecology response to decadal climatic variability in the American Southwest. *Journal of Climate* 11:3128–3147.

USEPA (U.S. Environmental Protection Agency). 2010. Hydraulic fracturing research study. Science in Action Press Release, EPA/600/F-10/002, Office of Research and Development, U.S. Environmental Protection Agency, Washington, D.C. www.epa.gov/ord . Accessed 1/12.

USFWS (U.S. Fish and Wildlife Service). 2011. Revision: Upper Colorado River Endangered Fish Recovery Program Plan (RIPRAP). U.S. Fish and Wildlife Service, Lakewood, Colorado. http://www.coloradoriverrecovery.org/documents-publications/foundational-documents/recovery-action-plan.html**.** Accessed 1/12.

USNRC (U.S. Nuclear Regulatory Commission). 2011. Fact sheet on uranium mill tailings. http://www.nrc.gov/reading-rm/doc-collections/fact-sheets/mill-tailings.html. Accessed 1/12.

Voigt, C., T. Bozorth, B. Carey, E. Janes, and S. Leonard. 1997. Sediment related issues and the public lands: Expanding sediment research capabilities in today's USGS: A Bureau of Land Management overview. Proceedings of a USGS sediment workshop, February 4–7, 1997. http://water.usgs.gov/osw/techniques/workshop/voigt.html.

Waring, G.L. 2011. A natural History of the intermountain west: Its ecological and evolutionary story. The University of Utah Press, Salt Lake City.

Wegner, D. 2008. New ideas for old dams: Developing solutions for a shrinking Colorado River. *Golden Gate University Environmental Law Journal* 2:69–95.

Williams, J.E., D.B. Bowman, J.E. Brooks, A.A. Echelle, R.J. Edwards, D.A. Hendrickson, and J.T. Landye. 1985. Endangered aquatic ecosystems in North American deserts with a list of vanishing fishes of the region. *Journal of the Arizona-Nevada Academy of Science* 20(1):1–61.

Woods, A.J., D.A. Lammers, S.A. Bryce, J.M. Omernik, R.L. Denton, M. Domeier, and J.A. Comstock. 2001. Ecoregions of Utah, U.S. Geological Survey, Reston, Virginia (map scale 1:1,175,000).

Woody, C.A., R.M. Hughes, E.J. Wagner, T.P. Quinn, L.H. Roulsen, L.M. Martin, and K. Griswold. 2010. The U.S. General Mining Law of 1872: Change is overdue. *Fisheries* 35:321–331.

4.2 Distribution and Status of Conservation Elements

> Species Management Questions
>
> 1. What is the current species distribution and status?
> 2. Where are potential areas to restore connectivity?

Conservation elements were organized into three categories—wildlife species, ecological systems, and designated sites. For the Colorado Plateau ecoregion, analyses were conducted on 18 species (7 mammals, 8 birds, and 3 fishes, Table 4-3) and nine ecological systems that included eight coarse filter vegetation communities plus riparian vegetation (Table 4-4). Sites of ecological and management concern included designated sites, high biodiversity sites, and herd management areas (HMAs). In addition, natural heritage species data organized by 5^{th} level HUCs was provided by NatureServe. Natural heritage data summaries included number of species, number of U.S. Fish and Wildlife Service threatened and endangered species, and number of globally critically imperiled, imperiled, and vulnerable species.

Table 4-3. List of wildlife species conservation elements (CEs) examined in the Colorado Plateau ecoregion.

Species CEs	
Black-footed Ferret (*Mustela nigripes*)	Gunnison's Prairie Dog (C*ynomys gunnisoni)*
Burrowing Owl (*Athene cunicularia*)	Mexican Spotted Owl (*Strix occidentalis lucida*)
Colorado Cutthroat Trout (*Oncorhynchus clarki*	Mountain Lion (*Puma concolor*)
Desert Bighorn Sheep (*Ovis canadensis nelsoni)*	Mule Deer (*Odocoileus hemionus)*
Ferruginous Hawk (*Buteo regalis*)	Peregrine Falcon (*Falco peregrinus*)
Flannelmouth Sucker (*Catostomus latipinnis*)	Pronghorn Antelope (*Antilocapra americana*)
Golden Eagle (*Aquila chrysaetos*)	Razorback Sucker (*Xyrauchen texanus*)
Greater Sage Grouse (*Centrocercus urophasianus)*	White-tailed Prairie Dog (C*ynomys leucurus)*
Gunnison Sage Grouse (*Centrocercus minimus)*	Yellow-breasted Chat (*Icteria virens*)

Table 4-4. List of conservation elements (CEs) examined: ecological systems (vegetation communities with dominant species listed) and classes of sites.

Ecological Systems CEs
Colorado Plateau Blackbrush-Mormon Tea Shrubland (Blackbrush)
Colorado Plateau Mixed Bedrock Canyon and Tableland (Littleleaf Mountain Mahogany)
Colorado Plateau Pinyon-Juniper Shrubland (Utah Juniper)
Colorado Plateau Pinyon-Juniper Woodland (Pinyon Pine)
Inter-Mountain Basins Big Sagebrush Shrubland (Wyoming Big Sagebrush)
Inter-Mountain Basins Mixed Salt Desert Scrub (Shadscale)
Inter-Mountain Basins Montane Sagebush Steppe (Mountain Sagebrush)
Riparian Vegetation
Rocky Mountain Gambel Oak-Mixed Montane Shrubland (Gambel Oak)
Sites CEs
Designated Sites
Biodiversity Sites – Terrestrial and Aquatic
HMAs

4.2.1 Evaluating Wildlife Species Distribution and Current Status

Current distribution data for the wildlife species conservation elements were derived from state GAP, Southwest ReGAP, or compilations of state agency spatial data. Emphasis was placed on state wildlife agency data, but often it was impossible to reconcile boundary issues between the different states. Original species distribution mapping was not possible due to lack of detailed occurrence records necessary to adequately conduct MaxEnt modeling. Therefore, many of the distribution results are based on either state GAP or Southwest ReGAP data that typically overestimate distribution. For example, mountain lion data was obtained from each of the state wildlife agencies for the ecoregion, but it was impossible to reconcile the obvious boundary issues. With no occurrence data available, Southwest ReGAP data was selected to represent current distribution of this species (Figure 4-11).

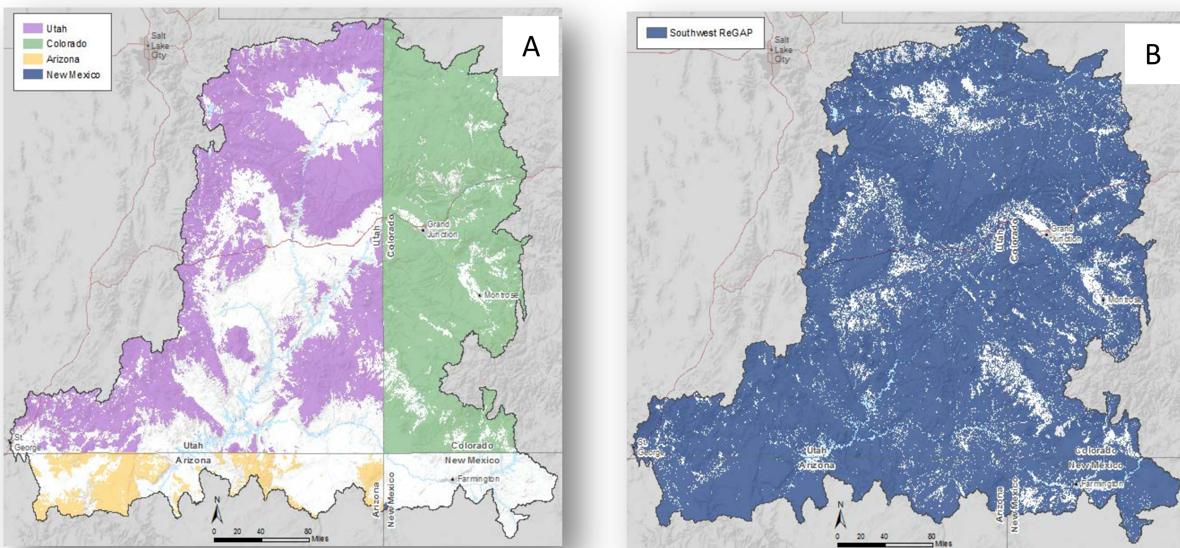


Figure 4-11. (A) Mountain lion distribution acquired from state wildlife agencies or state GAP and (B) Mountain lion distribution according to Southwest ReGAP.

The total area examined in the ecoregion was 44.8 million acres (18 million hectares). Current distributions for the terrestrial species based on the spatial distribution data ranged from about 100,000–41,190,000 acres (Table 4-5). The three fish species were mapped according to total stream length (Table 4-6).

Species status was evaluated in two ways—a review of background information (discussed in individual species profiles in Appendix C) and an overlay of current distribution with intactness: that is, terrestrial intactness at a 4 km X 4 km grid cell resolution for terrestrial species and aquatic intactness organized by 5th level hydrologic unit (HUC) for the three fish species. This model of intactness is fundamental to assessing the status of all conservation elements in the REA.

Terrestrial landscape intactness was mapped following the methods described in Chapter 3, Sections 3.2.3 and 3.2.4. In this model, numerous species-level attributes and indicators were examined (Appendix D), particularly known change agents that provide the most important information related to likely changes in species status over time. Unfortunately, the scientific literature does not provide many quantifiable indicators, and when it does, spatial data is typically not available for that indicator.

Table 4-5. Total current distribution area (in 1000s of acres) for terrestrial species conservation elements for the Colorado Plateau.

Species CEs	Total Distribution Area	Percent of Ecoregion
Black-footed Ferret (*Mustela nigripes*)	100	0.2
Burrowing Owl (*Athene cunicularia*)	18,733	41.8
Desert Bighorn Sheep (Ovis canadensis nelsoni**)**	4,719	10.5
Ferruginous Hawk (*Buteo regalis*)	13,746	30.7
Golden Eagle (*Aquila chrysaetos*)	41,190	91.9
Greater Sage Grouse (*Centrocercus urophasianus)*	1,998	4.5
Gunnison Sage Grouse (*Centrocercus minimus)*	443	1
Gunnison's Prairie Dog (C*ynomys gunnisoni)*	219	0.5
Mexican Spotted Owl (*Strix occidentalis lucida*)	572	1.3
Mountain Lion (*Puma concolor*)	39,756	88.7
Mule Deer (*Odocoileus hemionus)*	32,127	71.7
Peregrine Falcon (*Falco peregrines*)	15,221	34
Pronghorn Antelope (*Antilocapra americana*)	6,182	13.8
White-tailed Prairie Dog (C*ynomys leucurus)*	653	1.5
Yellow-breasted Chat (*Icteria virens*)	1,857	4.1

Table 4-6. Total current distribution stream length (1000s of miles) for fish species conservation elements.

Species CEs	Total Distribution (Length) (miles)
Colorado Cutthroat Trout (*Oncorhynchus clarki*	21
Flannelmouth Sucker (*Catostomus latipinnis*)	57
Razorback Sucker (*Xyrauchen texanus*)	3

For example, golden eagle and ferruginous hawk status is closely tied to prey density (especially jackrabbits according to Howard and Wolfe [1976]). Prey density would be a strong indicator for predicting status for this species, but data were not available to create a spatial model. Even if data for this indicator could be generated, it would still be challenging to use for this purpose because of its inherent dynamism—many prey species such as jackrabbits display boom and bust population cycles every 7 to 10 years (Gross et al. 1974).

Some of the more common status indicators for species pertain to one or more types of human development (including urban, agriculture, mining, recreation and roads): in other words, minimal human development generally indicates intact habitat conditions for a species and high levels of development indicate degraded conditions. For this reason, status for each species was derived by overlaying species distributions against the overall intactness model, which provides the best regional perspective of vegetation condition and habitat quality, development profile, and natural habitat fragmentation patterns. Not all species demonstrate the same level of tolerance to the various model inputs, but an overall intactness model provides a standard baseline from which to explore specific species or regions where tolerances to various components may vary. With an overall intactness model in-hand, it is relatively easy to test specific thresholds for individual species.

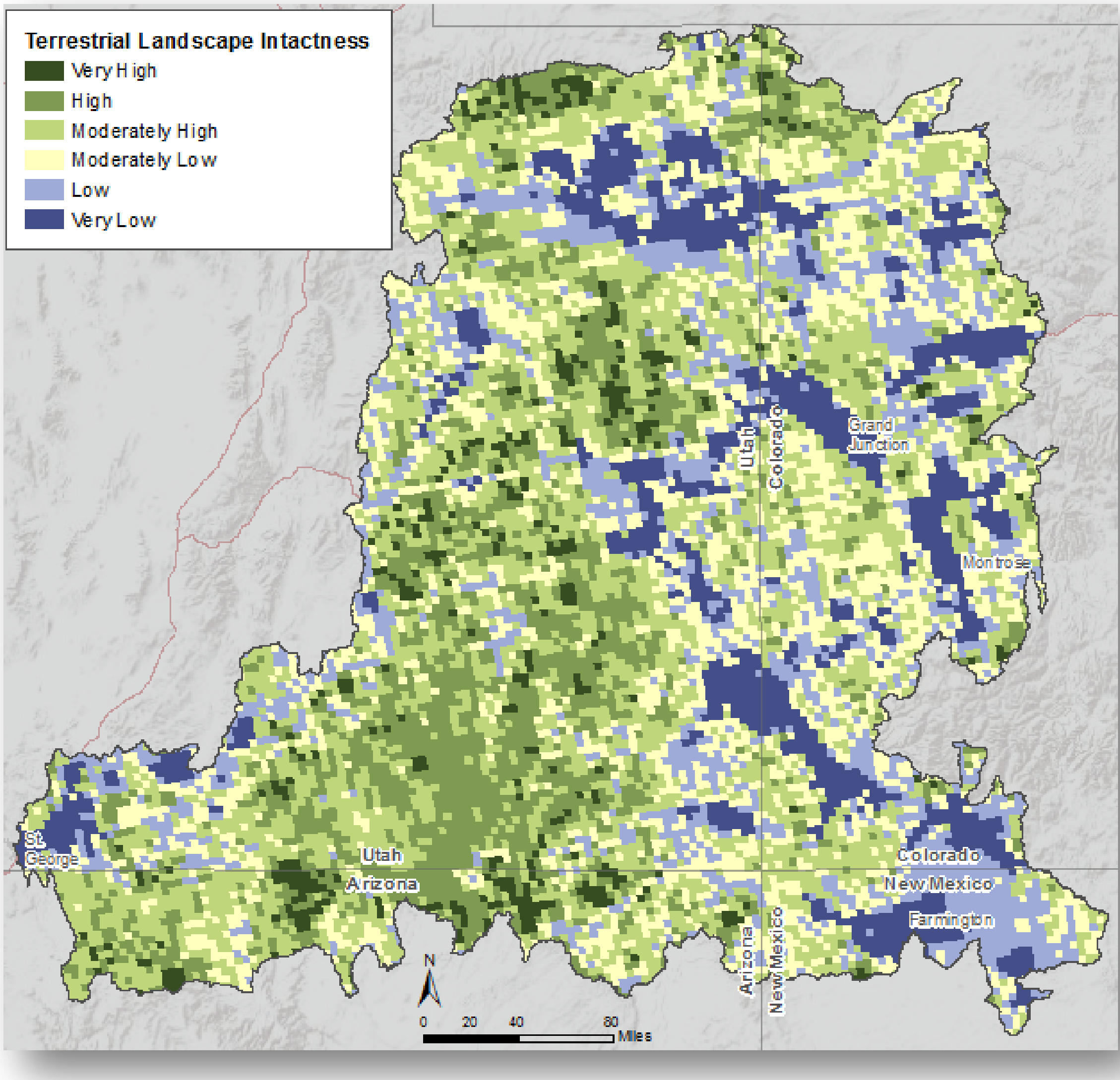
Current terrestrial landscape intactness at 4 km x 4 km resolution (Figure 4-12) and aquatic intactness organized by 5th level HUC (Figure 4-13) for the Colorado Plateau ecoregion show the full range of values from very low to very high intactness and their distribution in the accompanying histogram. The results for the terrestrial intactness model showed 1.6 million acres in the Very High intactness class and 7.8 million acres in the High class. For aquatic intactness, 400,000 acres were recorded for Very High intactness and 2.7

million acres for High intactness. When terrestrial and aquatic resources are considered at a regional scale, one gets the impression that some terrestrial highly-intact refugia remain, but that aquatic refugia are fewer.

In cases where more quantifiable thresholds have been reported and can be tested, the logic model is easily modified. For example, Figure 4-14 presents two terrestrial intactness results for mountain lion. Map 4-14A shows the overall intactness model results overlaid by mountain lion distribution to provide a status profile and map 4-14B shows the same mountain lion distribution over a customized version of the intactness model that includes a road density tolerance threshold of 0.60 km/km^2 reported by Van Dyke et al. (1986) for their study in southern Utah. One can easily see the difference a reported threshold can have on the results. The histograms show a dramatic decline of suitable mountain lion habitat when this threshold is enforced in the model. Map 4-14B clearly shows islands of high quality mountain lion habitat based on noted attributes and indicators for this species (Appendix D). A few of these blocks are very large while others are small and somewhat isolated from one another. Mountain lions could occur over most of the ecoregion according to the Southwest ReGAP distribution data, but in areas of low or very low intactness quality, mountain lions are expected to come into regular contact with human activities, often with negative consequences. Prey density (especially mule deer) is another important indicator of high quality mountain lion habitat. While spatially explicit information for primary prey species density was not available, the status results using the reported road density threshold can be compared with current distributions of mule deer, bighorn sheep, and pronghorn antelope to observe the overlap with mountain lion distribution. Interestingly, the largest blocks of prime mountain lion habitat based on terrestrial landscape intactness indicators coincide with bighorn sheep distributions, but they are largely outside occupied mule deer and pronghorn antelope habitat. However, more local scale information is needed to verify this.

A second example pertains to sage grouse and its tolerance to oil and gas well development. Doherty (2008) reported a well density of >12 wells per 4 km grid cell as limiting to greater sage-grouse on winter habitat (Figure 4-15). Incorporating this threshold into the intactness model resulted in adding 2% to greater sage-grouse habitat in the Very Low category (Figure 4-15A). Regardless of which current intactness model is used, Gunnison sage-grouse status based on habitat intactness is considerably lower than the status profile for greater sage-grouse (Figure 4-15B). Only about 14% of habitat occupied by Gunnison sage grouse is in the Moderately High category or above compared to roughly 45% for greater sage grouse areas in the northern portion of the ecoregion. Current distribution maps and status histograms for the 15 remaining wildlife species conservation elements are provided in Figure 4-16–Figure 4-20. Note that in these figures species distribution is indicated in blue on the distribution maps for each of the 15 species and intactness is represented in the histograms only. Live maps may be viewed on the data portal for panning, zooming, or combining with other data layers.

For the three ungulate CEs in the Colorado Plateau, desert bighorn sheep occupies more intact portions of the landscape than the other two species (Figure 4-16). Perhaps this can be partially explained by the choice of reintroduction sites for this species. The pronghorn antelope status histogram profile is skewed to the low end of the spectrum because its habitat is fragmented by human disturbances. Pronghorn is also subject to the same exposure to oil and gas drilling areas as the sage-grouse. The two prairie dogs, especially Gunnison's, also occur in habitat that is skewed very much to the low end of the intactness spectrum (Figure 4-17). Prairie dogs may have greater tolerance to low intactness and disturbed landscapes, but according to these results, many colonies are under considerable stress. Lack of intactness has direct effects on species, but low intactness also serves as a meaningful surrogate for other impacts not directly mapped such as shooting, poisoning, and plague (Lupis et al. 2007). The limited current distribution of black-footed ferrets is quite precarious according to the status profiles as expected (Figure 4-17). Overall, white-tailed prairie dog status is low and ferret status is affected by the limited number of large prairie dog colonies needed to support a sustainable ferret population and available for reintroduction. There is a notable bump in status for black-footed ferret in the Medium Low category, which may reflect reintroduction efforts.



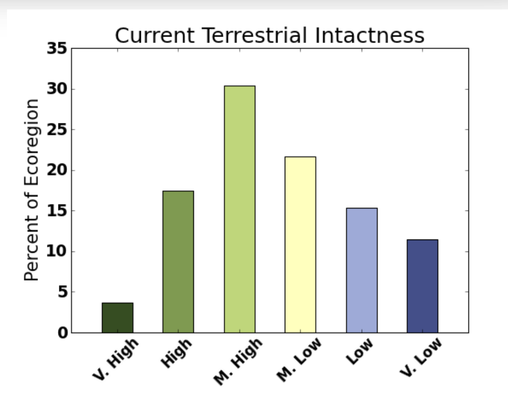
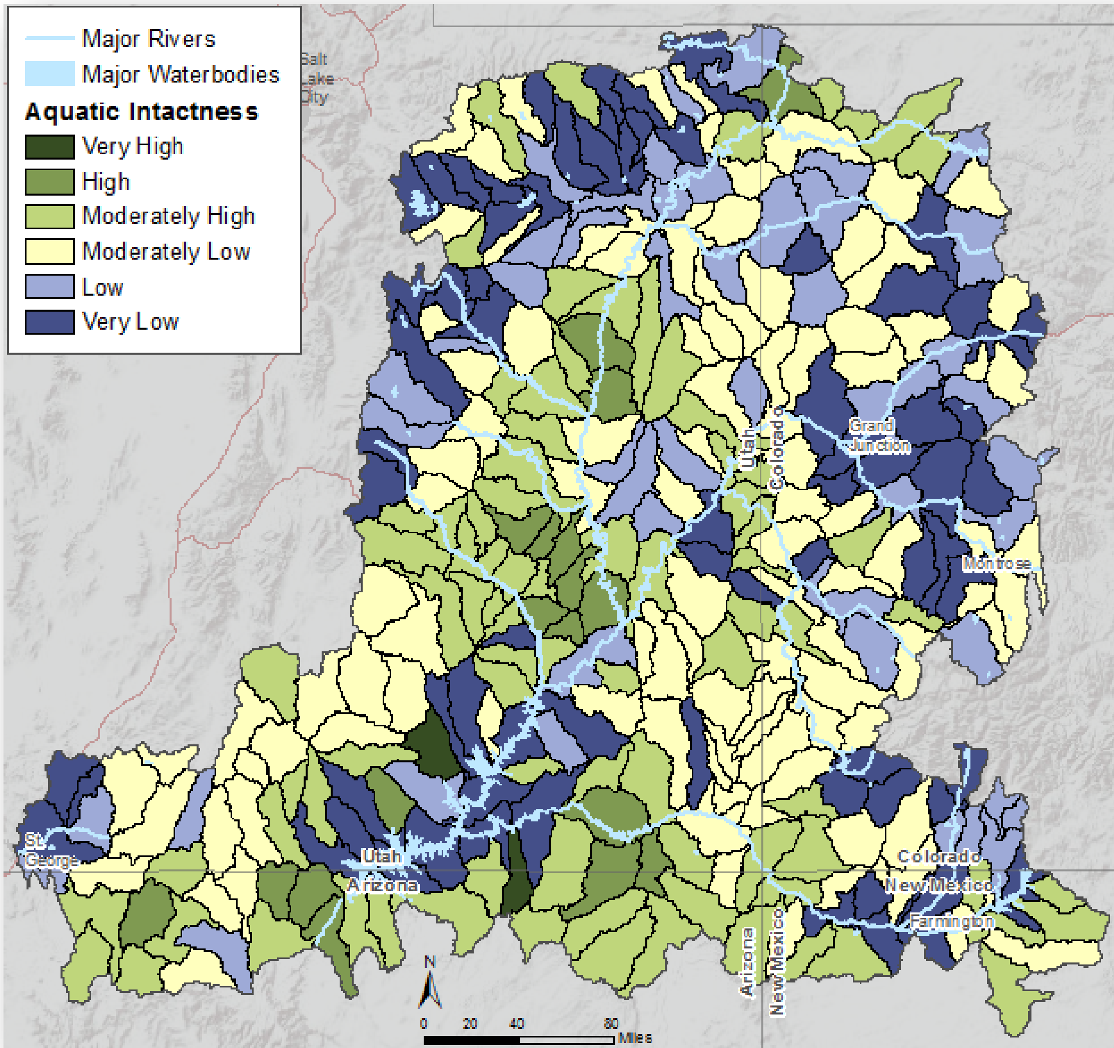


Figure 4-12. Terrestrial landscape intactness results organized in six categories by 4 km X 4 km grid cells for the Colorado Plateau ecoregion with associated histogram.



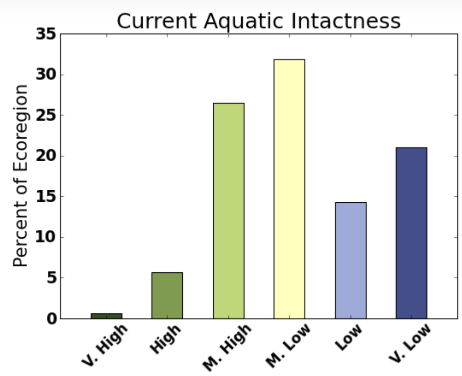


Figure 4-13. Aquatic intactness results organized by 5th level HUCs for the Colorado Plateau ecoregion and associated histogram.

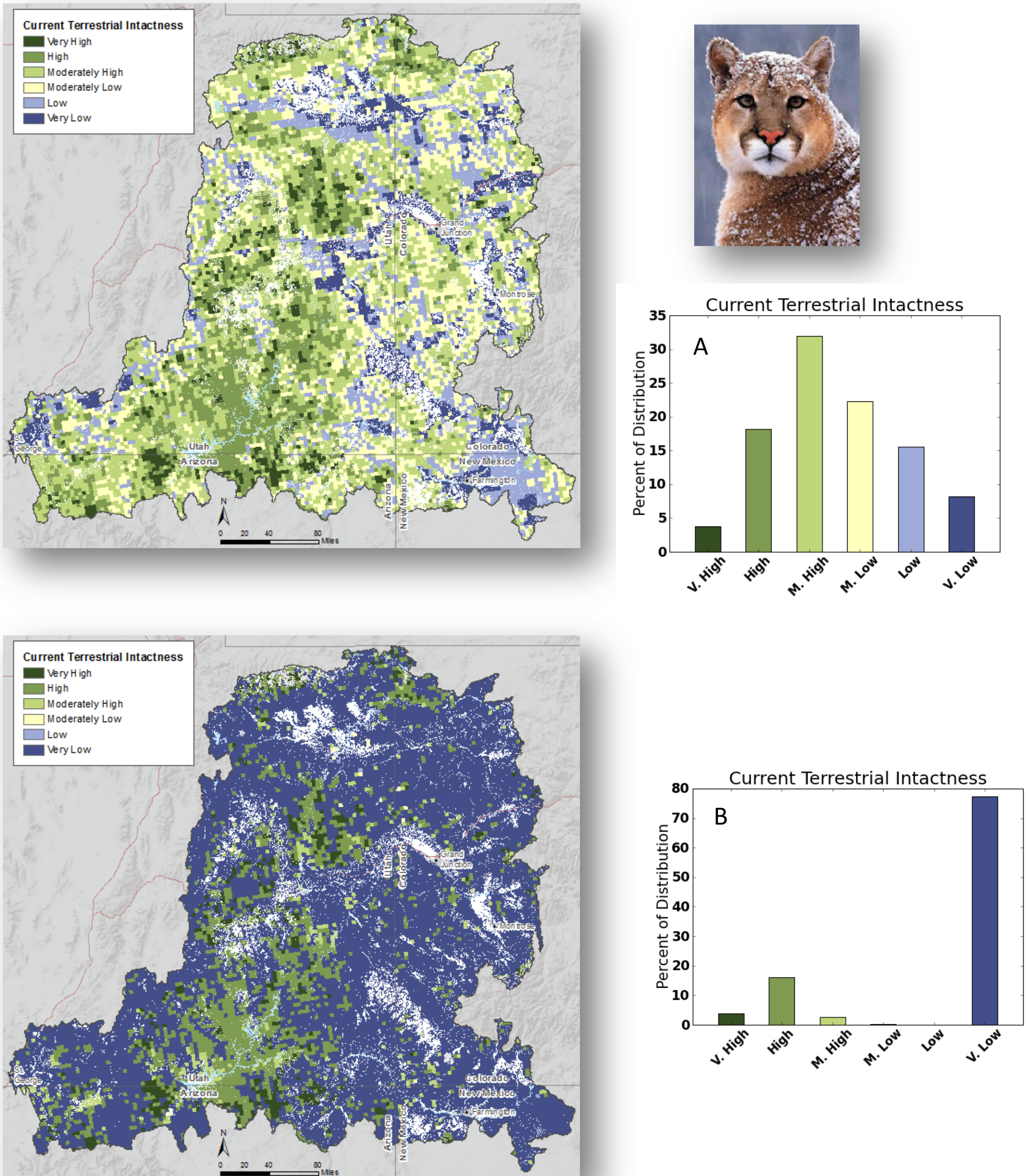


Figure 4-14. Map shows A) mountain lion status created by overlaying current distribution against the general terrestrial intactness model; and B) mountain lion status according to the customized intactness model, with a road density tolerance of 0.6 km/km^2 (Van Dyke et al. 1986), both organized by 4km X 4 km grid cells for the Colorado Plateau ecoregion.

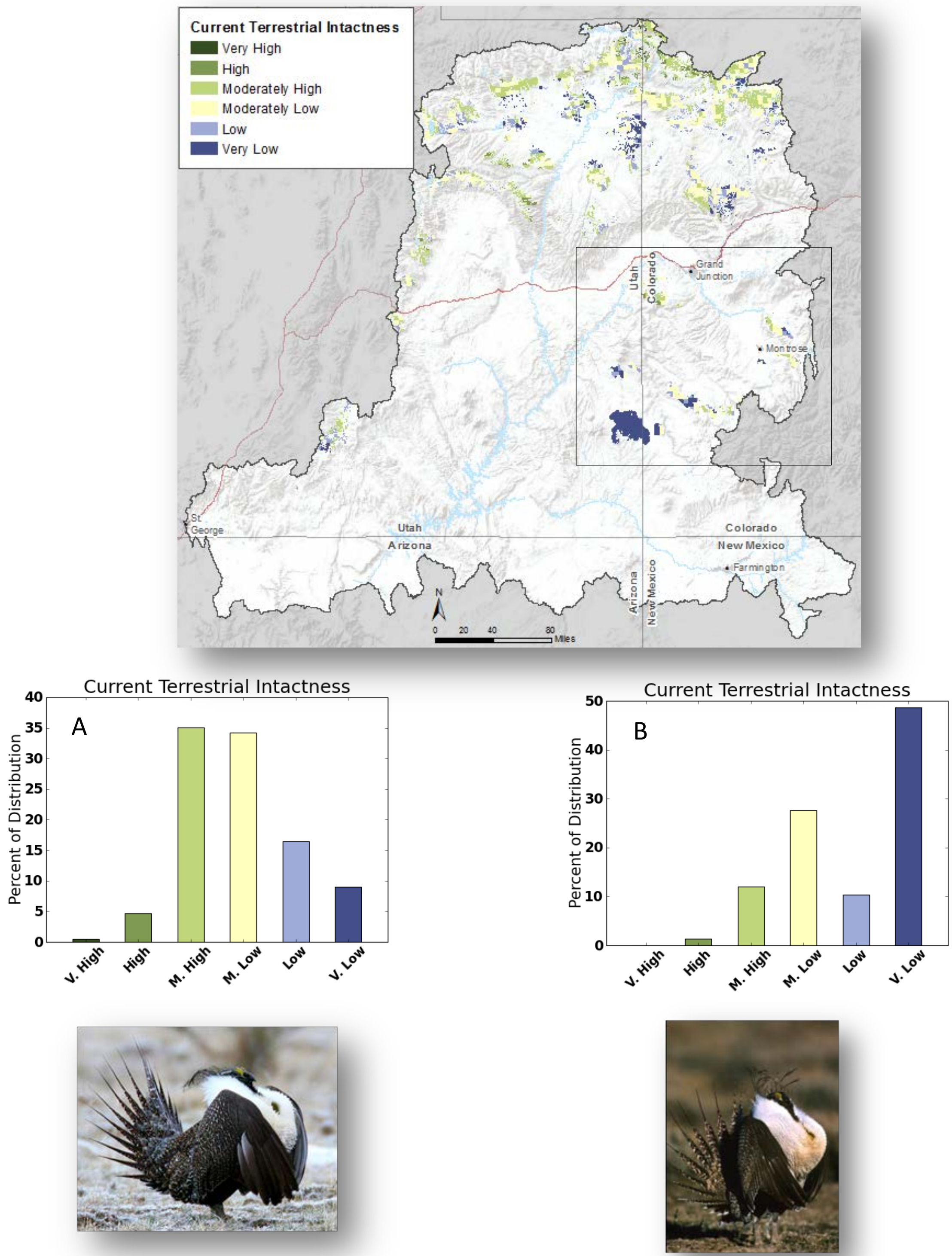


Figure 4-15. Map and histograms show results for greater and Gunnison sage-grouse status, using a threshold for oil and gas well density of >12 well pads per 4 km X 4 km grid cell in the terrestrial landscape intactness model (Doherty [2008], see Sage-grouse Case Study Insert for more details). Map and histograms both show (A) status for greater sage-grouse and (B) Gunnison sage-grouse in six intactness classes. Status for both species is shown on the same map (Gunnison sage-grouse distribution and status inside box on map).

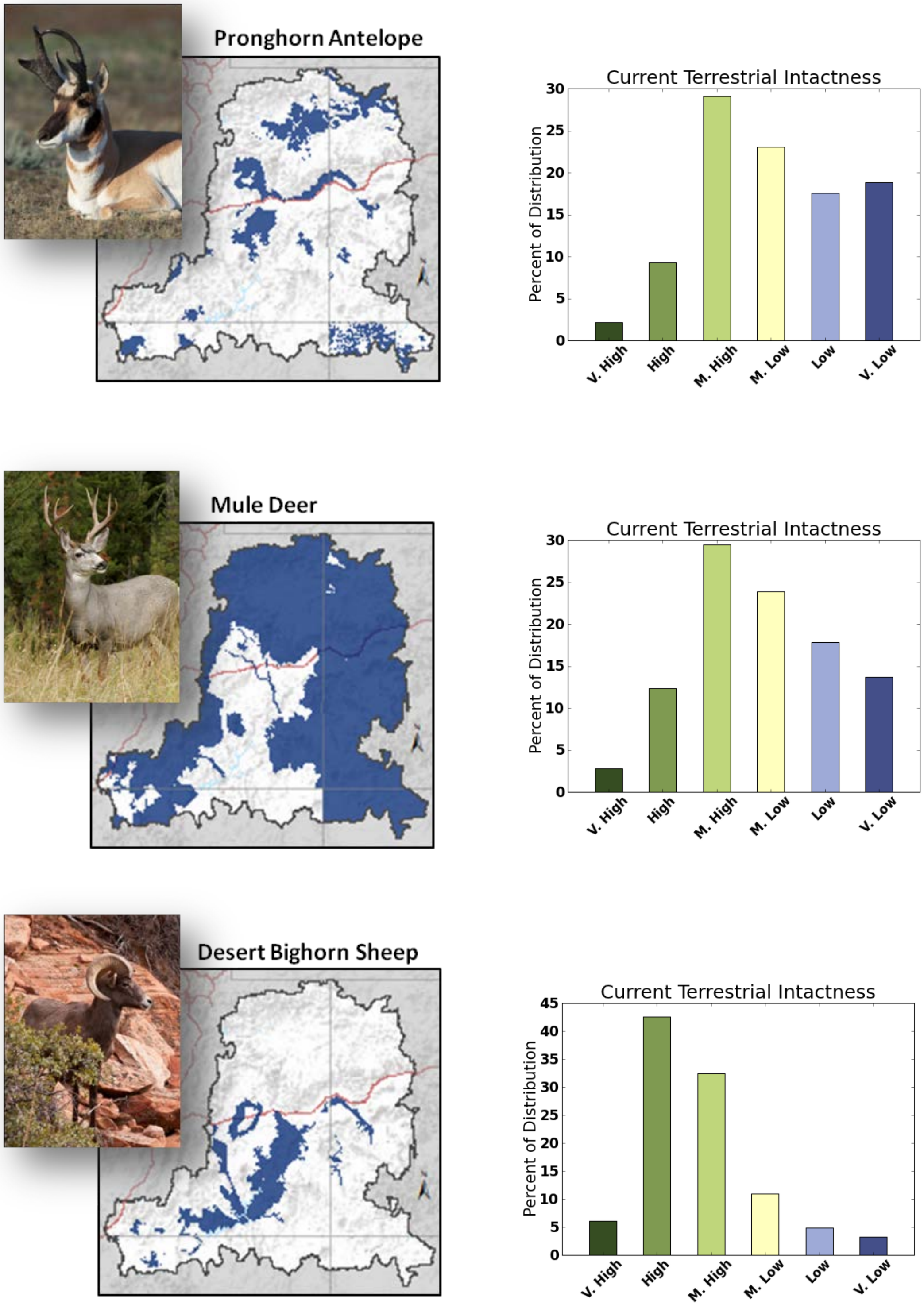


Figure 4-16. Current species distribution (in blue on maps) and conservation element status (histogram) based on current terrestrial intactness model for pronghorn antelope, mule deer, and desert bighorn sheep in the Colorado Plateau ecoregion.

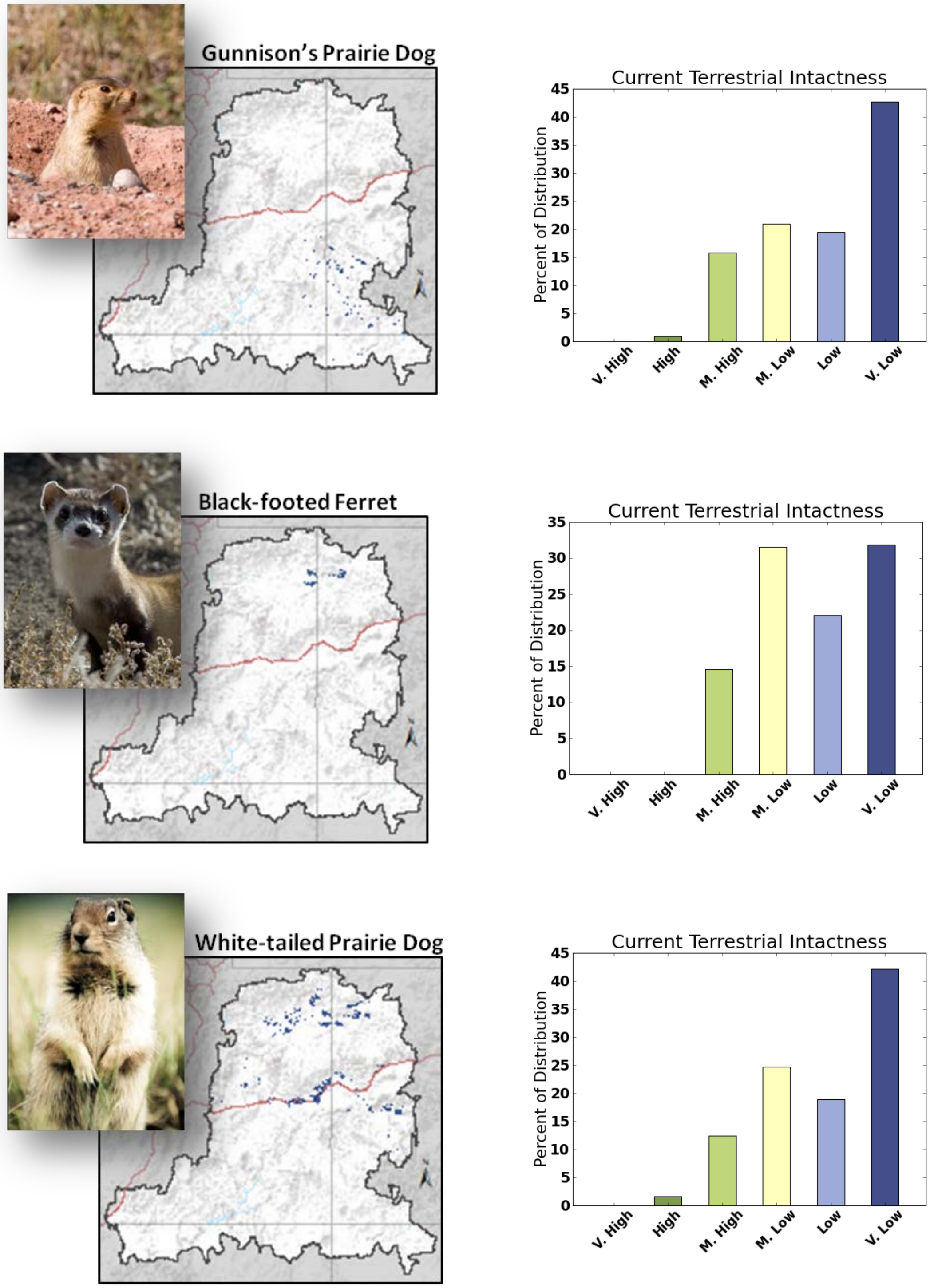


Figure 4-17. Current distribution (in blue on maps) and conservation element status (histogram) based on current terrestrial intactness model for Gunnison's prairie dog, black-footed ferret, and white-tailed prairie dog in the Colorado Plateau ecoregion.

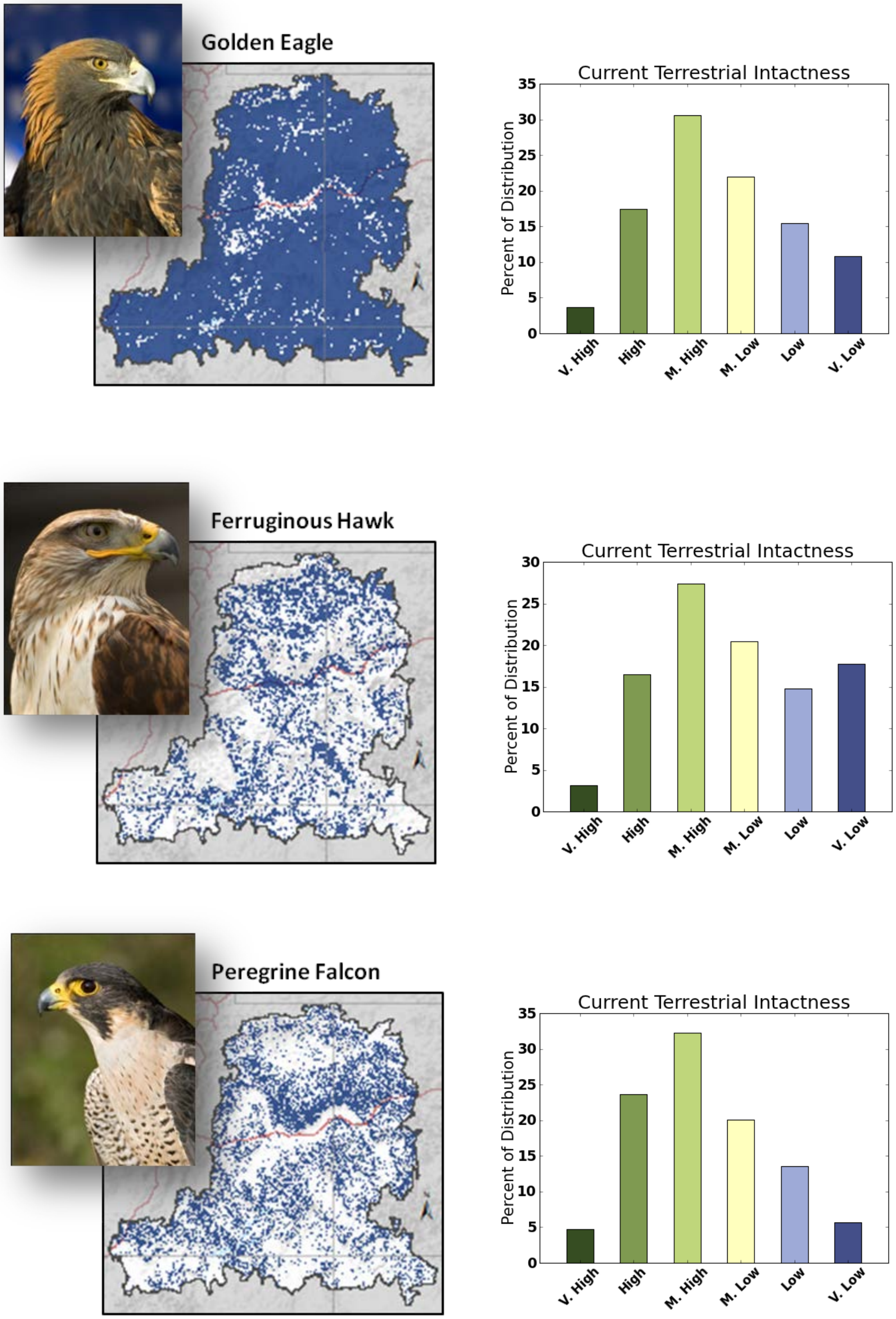


Figure 4-18. Current distribution (in blue on maps) and conservation element status (histogram) based on current terrestrial intactness model for golden eagle, ferruginous hawk, and peregrine falcon in the Colorado Plateau ecoregion.

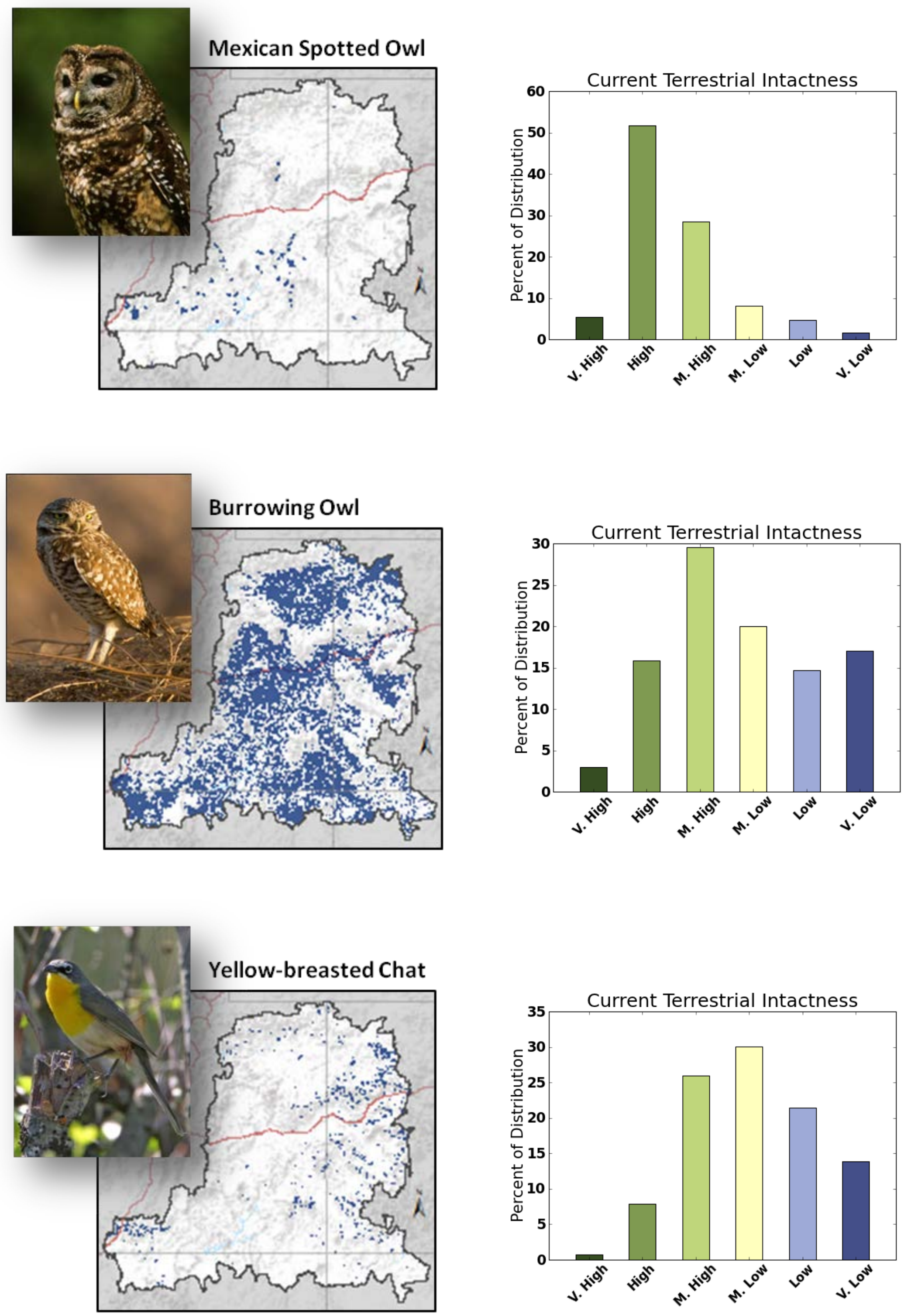


Figure 4-19. Current distribution (in blue on maps) and conservation element status (histogram) based on current terrestrial intactness model for Mexican spotted owl, burrowing owl, and yellow-breasted chat in the Colorado Plateau ecoregion.

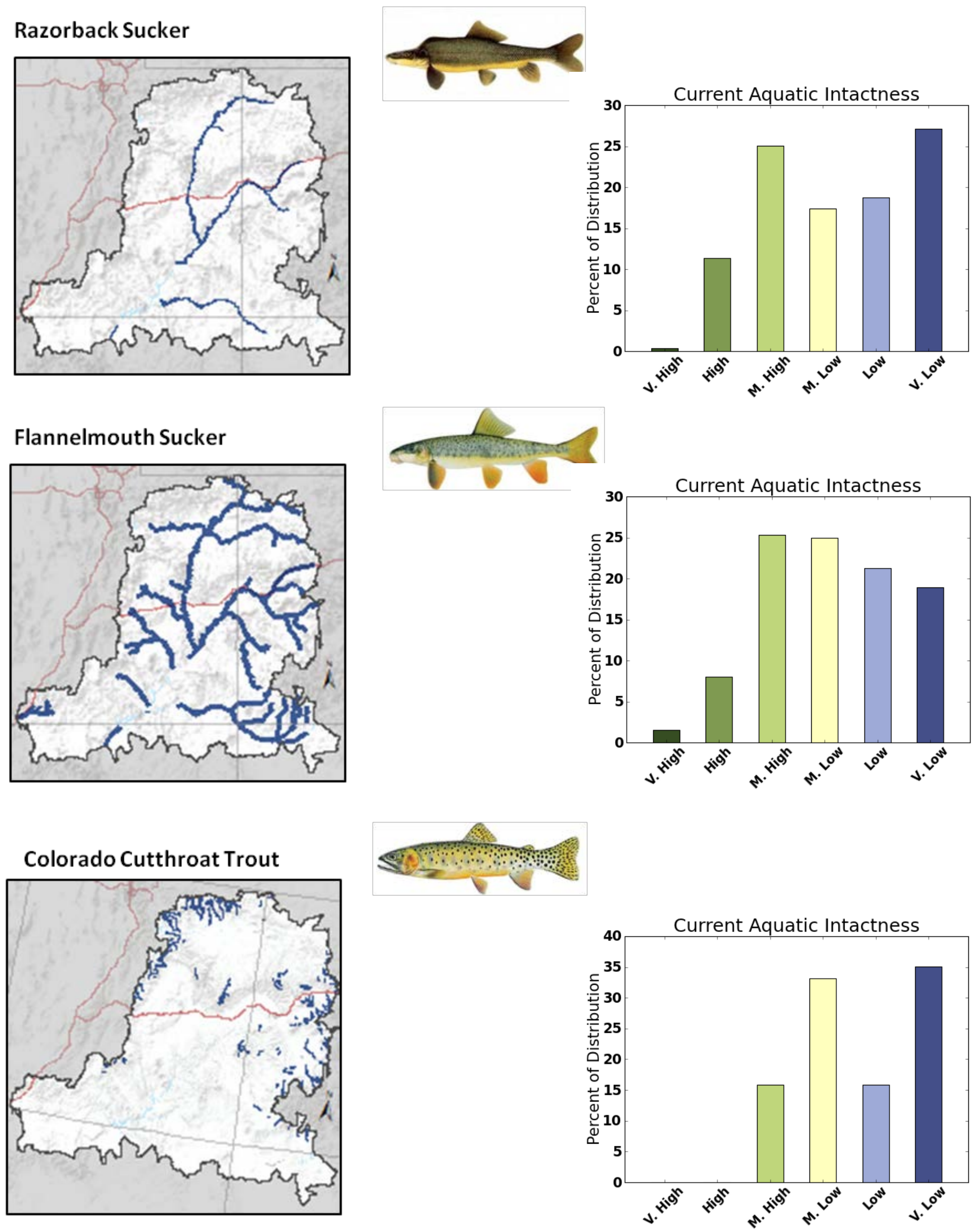


Figure 4-20. Current distribution (in blue on maps) and conservation element status (histogram) based on current aquatic intactness for razorback sucker, flannelmouth sucker, and Colorado River cutthroat trout in the Colorado Plateau ecoregion. See Appendix C for data sources for fish species distributions.

Golden eagle, ferruginous hawk, and peregrine falcon showed similar patterns for general status (Figure 4-18). Ferruginous hawk status was skewed to the low side of the spectrum more than the other two species. Golden eagle distribution was so widespread and generalized that the status histogram was almost the same as the overall intactness statistics.

Status profiles for Mexican spotted owl, burrowing owl, and yellow-breasted chat were all quite different (Figure 4-19). Mexican spotted owl had a relatively high status signature; the owl's distribution is limited, but its status score reflects the fact that the species' prime (and remaining) activity centers are concentrated in highly intact areas of the landscape, i.e., in protected National Parks and Monuments. Burrowing owl is more widespread and its status profile peaks in the moderately high category with a good portion of its habitat in the low classes, including 17% in the very low category. Burrowing owl's situation parallels that of the prairie dog species': they occur in lower elevations where human activity is also high. If the model is indicative of habitat quality for this species, burrowing owl populations in the Low and Very Low intactness classes should be under considerable stress. Yellow-breasted chat status profile is centered on the middle categories with a skewing to the low side of the spectrum; this reflects the general condition of riparian areas and the limited area of dense riparian shrub canopy that is optimal for nest habitat for chat. Yellow-breasted chat will use tamarisk thickets for nesting (Livingston and Schemnitz 1996, Sogge et al. 2008), which should be a consideration in tamarisk clearing and riparian restoration efforts. Having no other nesting options, chat will also be negatively affected by tamarisk defoliation and mortality from tamarisk beetle damage.

Unlike the other species, the three fish species were evaluated against the aquatic intactness results organized by 5th level HUC (Figure 4-20). Based on the status map results, Colorado cutthroat trout are found largely in stream systems where water entering the region from bordering mountain ranges is quickly diverted for other uses. Flannelmouth suckers are skewed heavily to the low side as well, but they do exist in some HUCs that scored in the higher intactness categories. Status for razorback suckers, primarily occupying the main stem rivers, showed heavy skewing to the low side of the intactness spectrum as expected. However, the aquatic intactness model did not represent all of the main stem impacts, which could affect some of these results. Also, the 5th level HUCs are extremely large, making it difficult to expose the details of the underlying data. The same model run at a finer HUC-based resolution would provide a more detailed and useful picture of status for these and other aquatic species.

Photo: Razorback sucker. M. Fuller, U.S. Fish and Wildlife Service

Greater Sage-Grouse
(*Centrocercus urophasianus*)

Photo: U.S. Fish and Wildlife Service

The sustainability of the greater sage-grouse (*Centrocercus urophasianus*) is entirely dependent on intact expanses of sagebrush. The sage-grouse is one of over 350 plant and animal species that are sagebrush obligates; a high proportion of these are endemic, threatened, or endangered, because the sagebrush community is one of the most-altered vegetation classes in the western states (Connelly et al. 2004). Over the last century, the sage-grouse has been reduced to 56% of its former range westwide. The U.S. Fish and Wildlife Service (USFWS) recently gave the greater sage-grouse candidate status rather than listing it as threatened or endangered—stating that it warrants protection, but that other species, facing greater and more immediate threats, take precedence (USFWS 2010). A court ruling in 2011 followed a number of law suits filed against the USFWS for delaying full Endangered Species Act protection for the grouse; it gave the USFWS until 2015 to decide the bird's status. In the interim, the BLM will review Resource Management Plans throughout the range of the greater sage-grouse and revise or amend them if necessary to incorporate sage-grouse conservation measures (BLM 2011a).

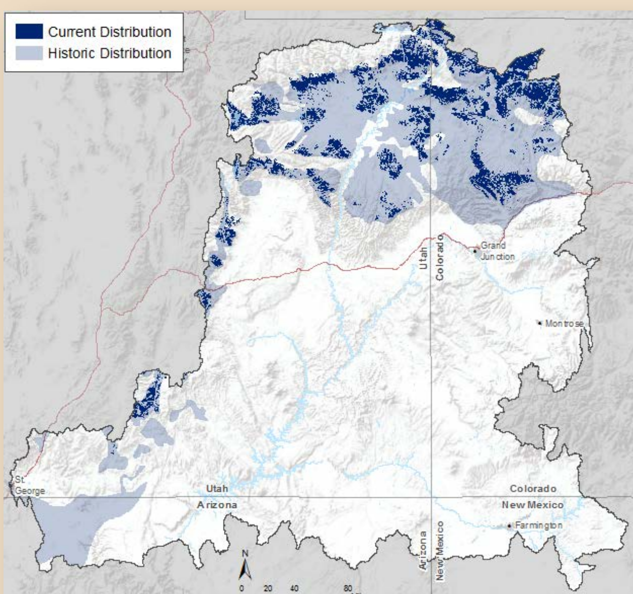


Figure 1. Map shows historic (light blue) and current (dark blue) distribution of greater sage-grouse in the Colorado Plateau.

Across the species' range, trend results from research and monitoring of sage-grouse populations indicate general declines, but results vary depending on the region and the scale of the investigation. Breeding Bird Survey trend estimate data for the Southern Rockies-Colorado Plateau ecoregion showed a 7.1% per year decline for the period 1966–2009 and a 5.2% per year decline for the period 1999–2009 (Sauer et al. 2011). However, these trend results carry a caveat, since they reflect detection difficulties on existing Breeding Bird Survey routes and a small sample size (<14). Local trends differ when examined at a regional level. Utah and northwestern Colorado represent the southeastern-most extent of the species' current distribution, which has contracted to the north (Figure 1), based on evidence of historic distributions. Greater sage-grouse populations in northwestern Colorado still maintain some connectivity with sage-grouse strongholds in Wyoming and Montana. Colorado populations are relatively stable and have been increasing (about 1% per year) over the last 17 years (Connelly et al. 2004). Sage-grouse habitat in Utah connects to these northern populations through the Uinta Basin where sage habitats are heavily fragmented. Sage-grouse populations are small and scattered along the western border of the Colorado Plateau ecoregion, and several small populations have been recently extirpated from former leks in southern Utah (Connelly et al. 2004). Annual rates of change in Utah populations indicate a long-term decline from levels of the late 1960s and early 1970s, when populations were approximately 2-3 times higher than current numbers (Connelly et al. 2004). The number of males per lek has decreased significantly and lek size has also decreased since the late 1960s, although there was a gradual increase in number of males per lek between 1997 and 2005 (UDWR 2009). In an examination of available data, Connelly at al. (2004) determined that sage-grouse populations declined at an overall rate of 0.35% per year in Utah from 1965 to 2003.

Thousands of pages have been written about sage-grouse functional requirements and threats to their future productivity; for a detailed review of greater sage-grouse related population ecology, data, study results, and literature, see Connelly et al. (2004) and Knick and Connelly (2011). Sage-grouse need large contiguous patches of sagebrush habitat because their functional habitat requirements differ by season and are quite specific, based on percent sagebrush cover and height, percent herbaceous cover and height, distance to other seasonal habitat types, and topographic position (Connelly et al. 2000). Access to several types of seasonal habitats for lekking, nesting, brood-rearing, and wintering is important for reproductive success, chick survival, and recruitment. Sagebrush patches used for nesting and brooding may be under 100 ha and located within a few kilometers of leks, but distances traveled by male grouse from lek to summer habitat and for all grouse between summer and winter ranges may be as much as 35–50 km (Connelly et al. 2004).

The species is sensitive and easily disturbed by land use activities that subdivide the landscape, disrupt the birds' site fidelity to traditional lekking and nesting areas, and ultimately isolate remnants of the population. Widespread degradation and conversion of sagebrush communities has occurred over the last century with broad scale agricultural conversion in irrigable areas, sagebrush treatments to increase forage for livestock on rangelands, the introduction of invasive annual species, and subsequent changes in fire regimes. In somewhat higher and more mesic areas, a cycle of grazing, leading to a decrease in fire frequency, has resulted in pinyon and juniper encroachment into sage grouse habitat and a reduction in ground cover perennials and forbs. Elsewhere, the invasion of cheatgrass (*Bromus tectorum*) and an associated increase in fire frequency has resulted in extensive loss of sagebrush stands that may take several decades to recover (Connelly *et al.* 2004, Crawford et al. 2004). Agricultural fields and irrigation canals affect 32% of sagebrush habitat in 9 western states (Connelly et al. 2004). In recent decades, exurban growth, expressed as rural small parcel development, has increased the fragmentation of sage habitat in former rangelands. The subsequent expansion of road networks, even low-volume secondary roads, negatively affects sage grouse. Recent studies have indicated that minimal road traffic (1–12 vehicles/day) reduces female grouse nest initiation (Lyon and Anderson 2003) and the number of breeding males displaying at leks (Holloran 2005). Powerlines and communications towers increase the pressure from predators and provide perches for raptors as do fences, which also cause direct mortality of sage grouse through collision and entanglement. Fences within 1.25 miles of active leks and fence densities > 1.6 miles/mile^2 of fence have been shown to increase risks for sage-grouse (thresholds listed in BLM [2011b], adopted from a study by Stevens [2011]).

Oil and gas drilling is the most pressing current and future threat to the sustainability of the sage-grouse in the Colorado Plateau. Increasing demand, a desire for energy security, favorable pricing, and recent extraction methods (e.g., fracking, see Section 4.1.4, Aquatic Resources of Concern) that retrieve oil and gas once thought too difficult and expensive to extract have created intense pressure to drill on public land

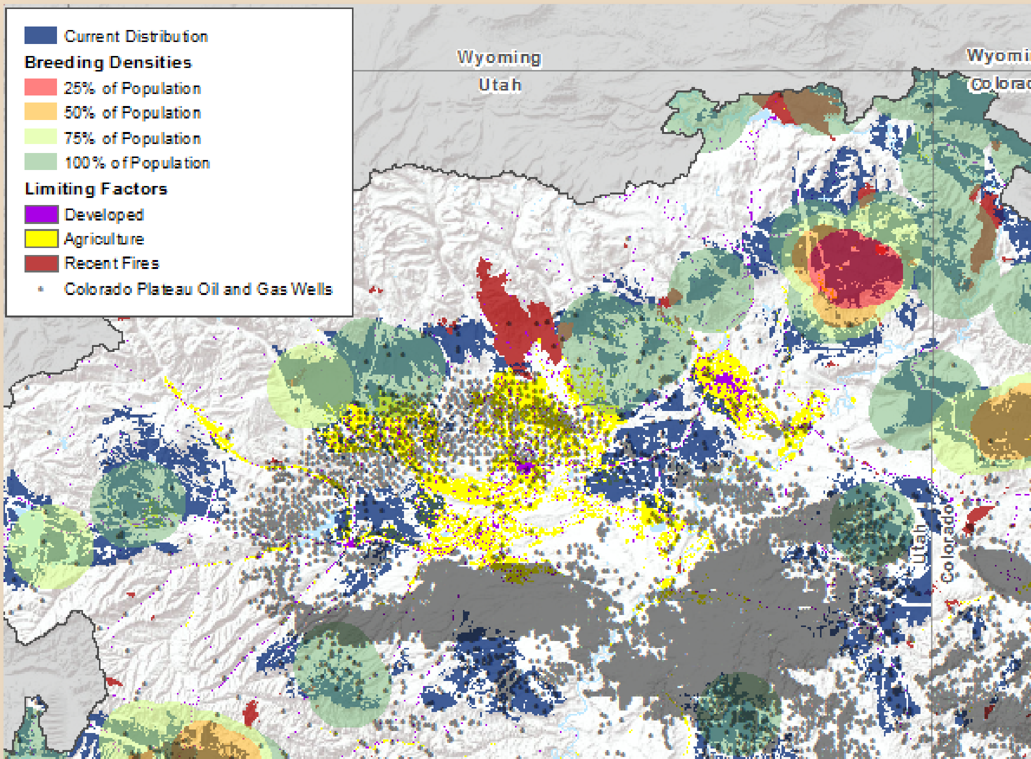


Figure 2. Map indicates sage-grouse activity areas and major stressors in the Uinta Basin: sage-grouse current distribution (blue areas), active leks (red, orange and green circles), oil wells (gray areas), agricultural areas (yellow), recent fires (irregular red polygons), and urban areas (purple).

in sagebrush habitats. Westwide, seven million hectares (~17,300,000 acres) of public lands—or 44% of the lands that the federal government controls for oil and gas development—have been authorized for drilling within distribution of the greater sage-grouse (Naugle et al. 2011). The sage-grouse has already been marginalized to the edges of the Uinta basin by oil and gas fields (gray areas in Figure 2) and other change agents (wildfire, urban and agricultural areas, Figure 2). Several long-term studies of sage-grouse response to oil and gas development in Wyoming have shown that the birds are sensitive to road density, traffic volume, noise, distance to wells, and well density (Holloran 2005, Walker et al. 2007, Doherty 2008, Harju et al. 2010). Walker et al. (2007) found that current management practices do not prevent impacts to the number of males attending sage grouse leks. In a 12-year study of 702 leks in Wyoming, Harju et al. (2010) found that impacts began occurring at well-pad densities as low as 0.396 well pads/km^2 (1 well pad/mile^2) and 0.772 well pads/km^2 (2 well pads/mile^2). Harju et al. (2010) also recorded that common well pad densities of 1.54 and 3.09 well pads/km^2 (4 and 8 well pads/mile^2) were associated with lek attendance declines ranging from 13.0% to 74.0% and 76.6% to 79.4%, respectively. Other seasonal habitats, such as winter habitat, are very important to sustain sage-grouse populations, but winter habitats are not regulated in terms of well pad densities. Doherty (2008) found in a winter habitat study that sage-grouse were 1.3 times more likely to occupy sagebrush habitats that lacked wells within a 4-km^2 area, compared to sage habitats that had a maximum density of 12.3 wells/4 km^2 (8 wells/mile^2).

Any attempt to strike a balance between conservation and energy development must have science-based tools to apply the information to a range of alternative solutions. The map of range-wide breeding densities is one such tool (Figure 3)—it can assist in cross-jurisdictional planning among federal and state agencies and local working groups (Doherty et al. 2010). Similar maps exist for sage-grouse management zones and individual states. The four colors of mapped dots represent the smallest area necessary to contain 25%, 50%, 75%, and 100% of nesting sage-grouse populations range-wide. The red and orange dots represent the highest densities of breeding males and the highest priority leks for protection where development may be restricted. Blue and green dots may be leks supporting smaller populations that are candidates for restoration or to maintain as nodes of connectivity with more productive sites (Doherty 2008). Some proportion of these lower productivity sites may be sacrificed to development. A map such as this provides a focus for regional planning and coordination among various agencies; local areas identified for possible development will require additional scrutiny for other important aspects of sage-grouse ecology such as chick rearing and winter habitat, seasonal migration corridors, and connectivity with other populations. The BLM and state agencies have collaborated on developing priority habitat areas that include breeding, brood-rearing, and winter concentration areas. Planning proposals may limit human disturbance in priority habitat. One proposal suggests that human-caused disturbance in priority habitats would be limited to less than 2.5% of the species' total habitat within that priority area (BLM 2011c); however, this is not a final determination—many proposals will be discussed over the next three years before the 2015 sage-grouse listing decision.

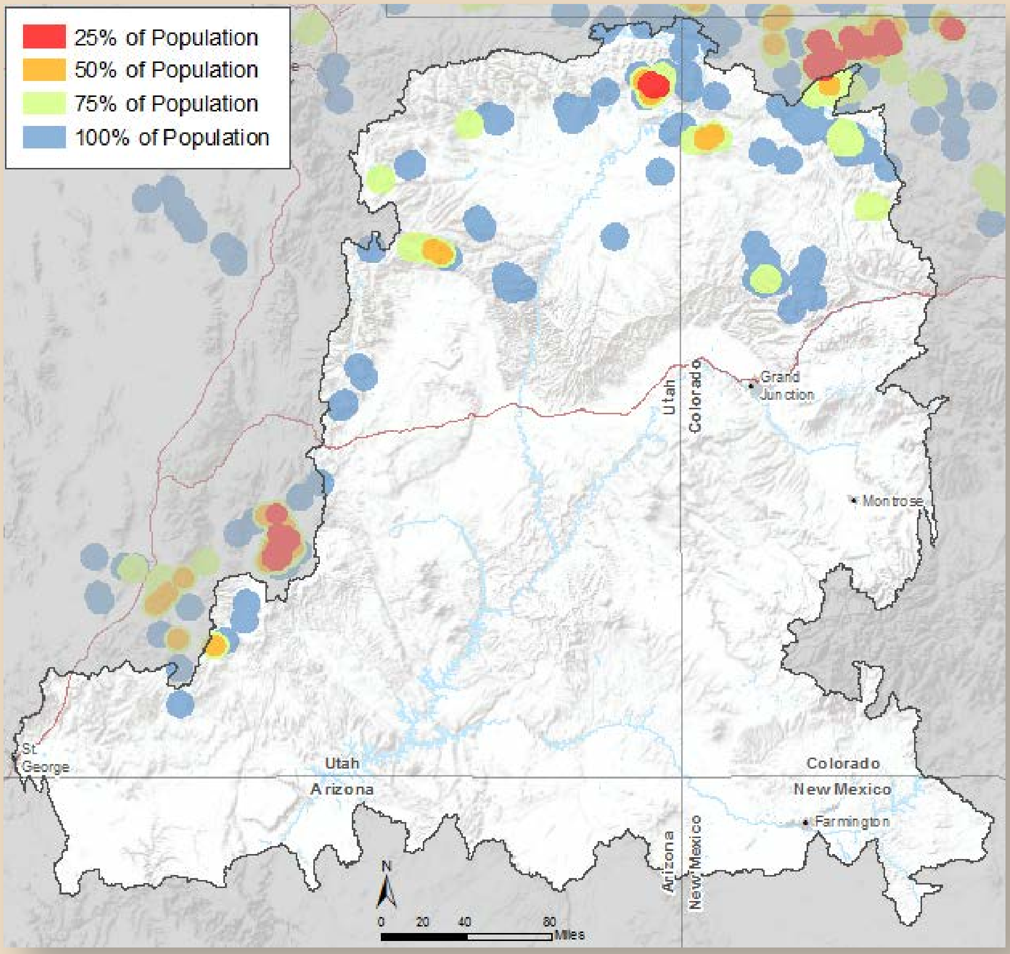


Figure 3. Sage-grouse breeding bird density map identifying buffered lek areas with red and orange symbols supporting the highest density of breeding males (from Doherty et al. 2010).

Status

Current distribution was evaluated for each wildlife species conservation element against the overall intactness model, which provides a regional perspective of vegetation condition, habitat quality, development profile, and natural habitat fragmentation patterns. It is relatively easy to test specific thresholds for individual species by altering the intactness model. As an example, for sage grouse winter habitat, the logic model was constrained with a threshold of >12 wells per 4 km^2 grid cell (>12 wells being false or unacceptable), a well density and activity level known to be limiting to sage grouse on their wintering grounds as discussed above (Doherty 2008); this minor adjustment to the model put 2% more sage grouse habitat into the very low intactness category (Figure 4). This is not a prescription, but an example to demonstrate how the model can be modified to test various management scenarios.

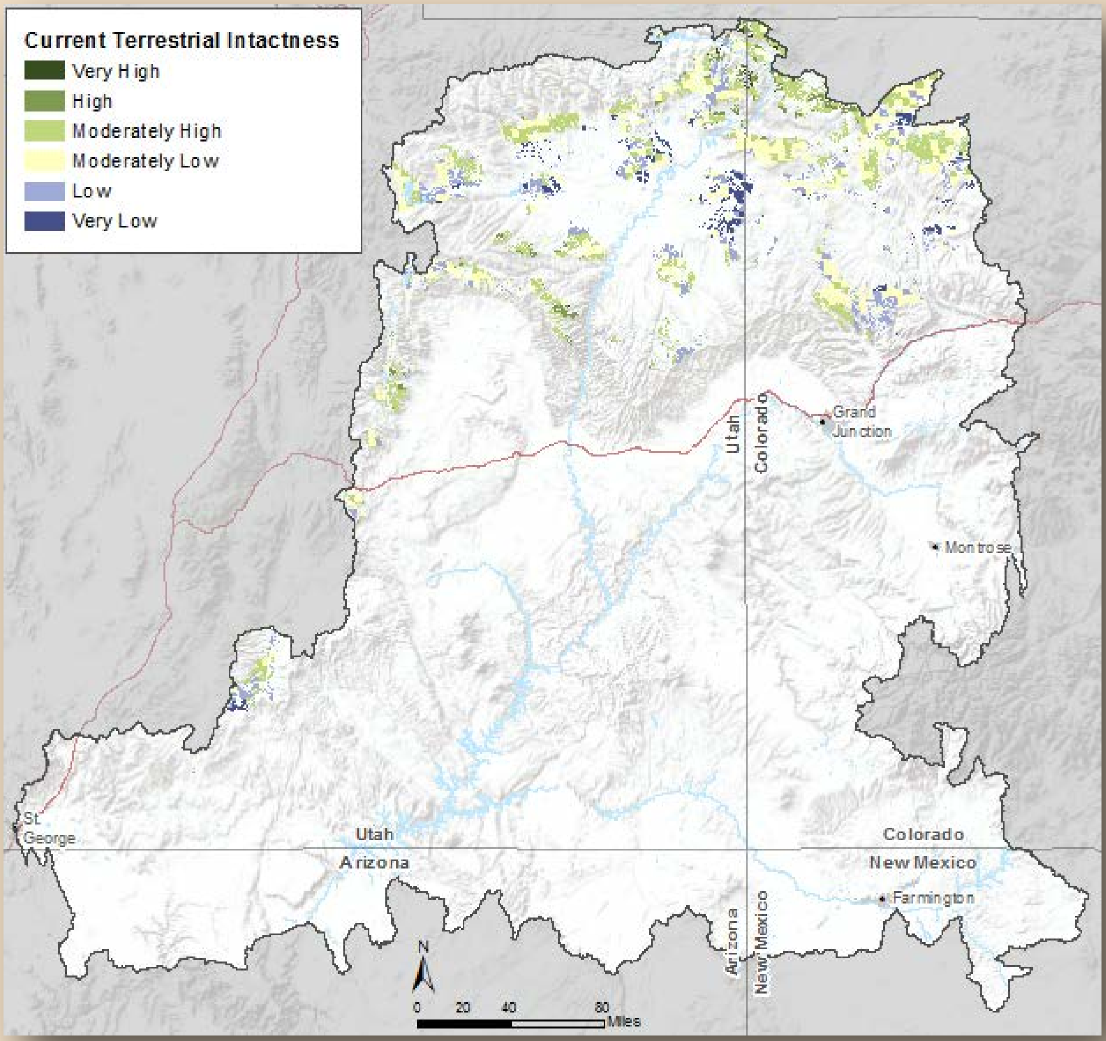


Figure 4. Current status for sage-grouse obtained from overlaying current distribution and the current landscape intactness model. A threshold of >12 wells/4 km^2 (8 wells/mi^2) was applied to the model to represent a known disturbance affecting sage-grouse populations.

Development Scenario (2025)

MQ D6 What terrestrial species are vulnerable to change agents in the near term horizon, 2025? Where are these species and sites located?

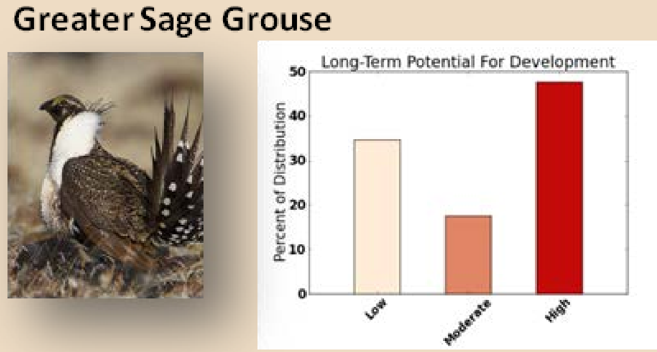


Figure 5. Histogram shows risk to sage-grouse from potential energy development with nearly 50% of its distribution in the High category.

As discussed above, oil and gas drilling is the most pressing current and future threat to the sustainability of the sage-grouse in the Colorado Plateau. Figure 2 showed the current situation for sage grouse in relation to human disturbances, and oil and gas in particular, in the Uinta Basin. The pressures on sage-grouse and its sagebrush habitat from oil and gas are increasing. When sage-grouse distribution was compared to the 4 km results for potential energy development, sage-grouse showed the highest risk of any conservation element from potential energy development with nearly 50% of its existing distribution in the high category with about another 18% in the Moderate category (Figure 5).

Copeland et al. (2009) created a model of oil and gas potential using geological and geophysical predictor variables, and they developed two build-out scenarios—anticipated and unrestrained—based on leasing history, recent increases in leasing based on increased demand, and agency (BLM) projections (Figure 6 below for the Uinta/Piceance Basin). Based on 2007 lek counts, Copeland et al. (2009) predicted a 7% sage-grouse population decline in the anticipated scenario and a 19% decline in the unrestricted scenario rangewide (added to declines that have already occurred).

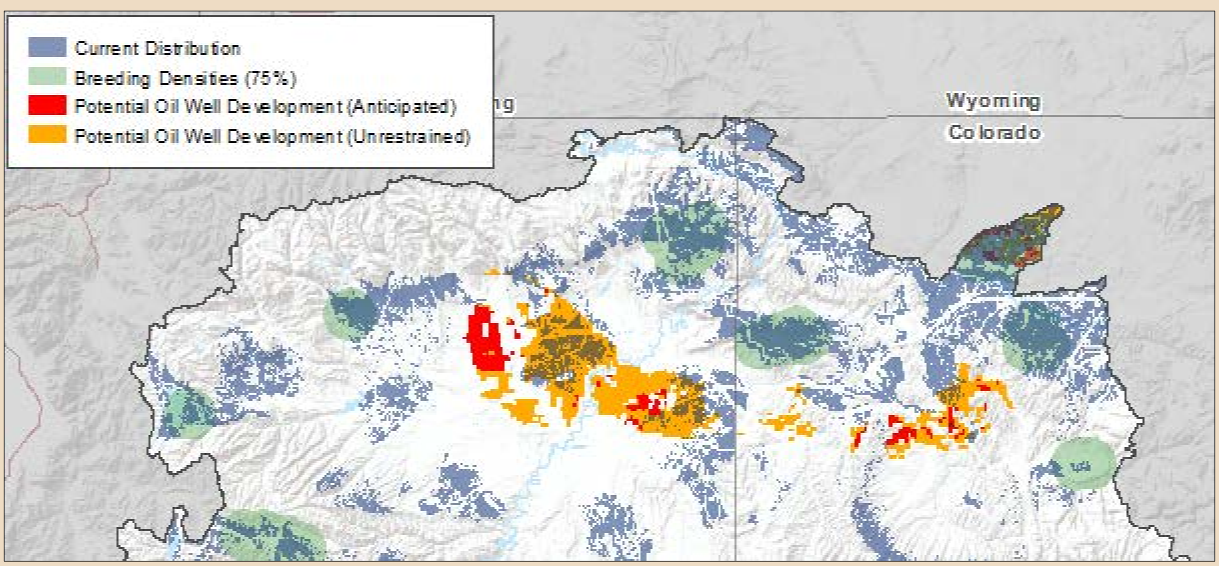


Figure 6. Map adapted from Copeland et al. (2009) for the Uinta/Piceance Basin showing coincidence of greater sage-grouse distribution and productive leks with two near-term future energy development scenarios. Map symbols represent current sage-grouse distribution (blue areas), productive sage-grouse leks (green symbols), and anticipated and unrestrained oil well development (red and orange areas).

In addition to sage-grouse, other sagebrush-dependent species are affected by the proliferation of drill pads; two of these species, pronghorn (*Antilopcapra americana*) and mule deer (*Odocoileus hemionus*) are REA core conservation elements. A model such as this offers an important tool for sage grouse conservation to be used with information on other major stressors and seasonal habitats. Note when comparing the map in Figure 6 to the previous map of lek densities (Figure 3) that the 75% breeding density symbols in the future development scenario (green circles) subsume the red, orange, and green symbols in the previous map, indicating the most productive sage-grouse leks. A number of blue symbols, representing less-productive leks that appeared in the previous map have been omitted from this future development scenario (Figure 6).

Climate Change Scenario (2060)

MQ D6. What terrestrial species are vulnerable to climate change in the long-term change horizon (2060)?

Key elements from the complex collection of climate change MAPSS results, such as potential for seasonal temperature and precipitation change and potential for vegetation change, were combined to create an overall relative climate change map (Section 5.4). The distribution of sage grouse was then overlaid on the climate change potential map to represent sage-grouse exposure to climate change (Figure 7).

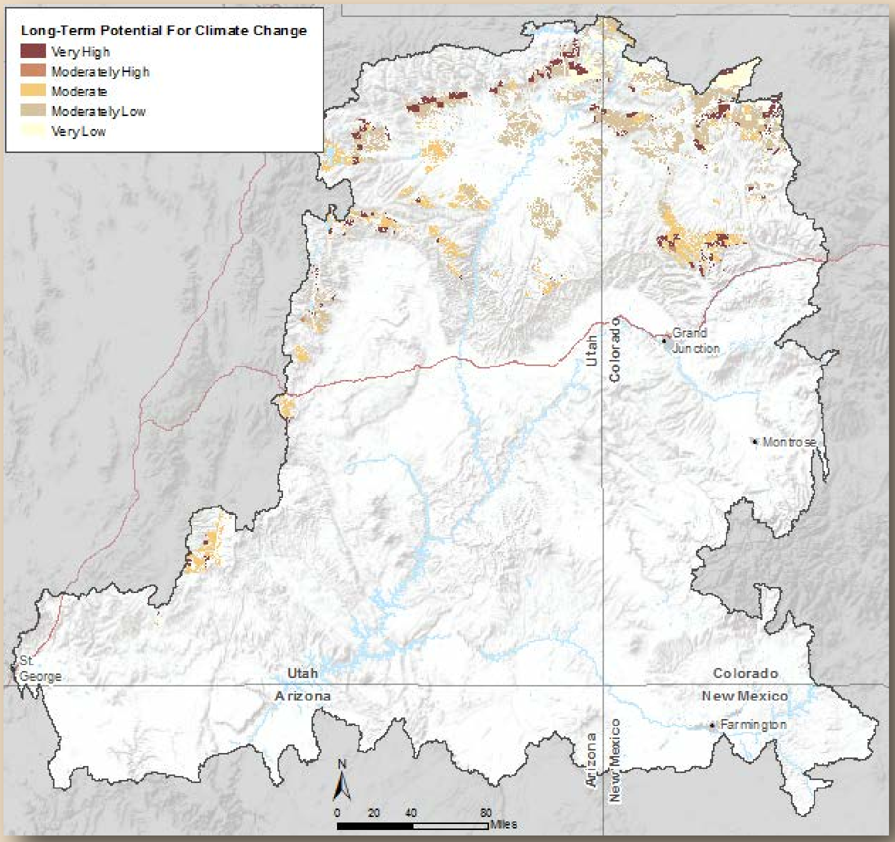


Figure 7. Map shows long term potential (2060) for climate change overlaid with the distribution of greater sage-grouse to produce a map representing sage-grouse exposure to climate change.

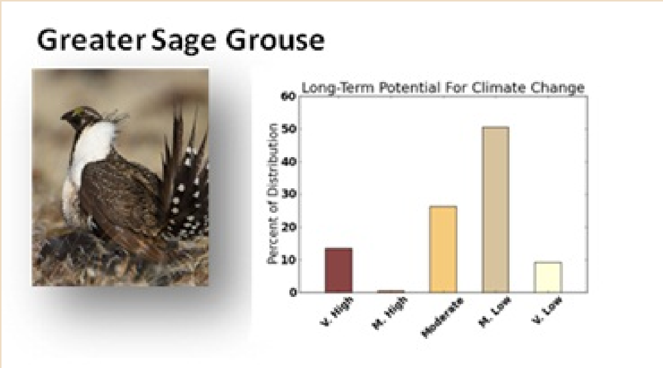
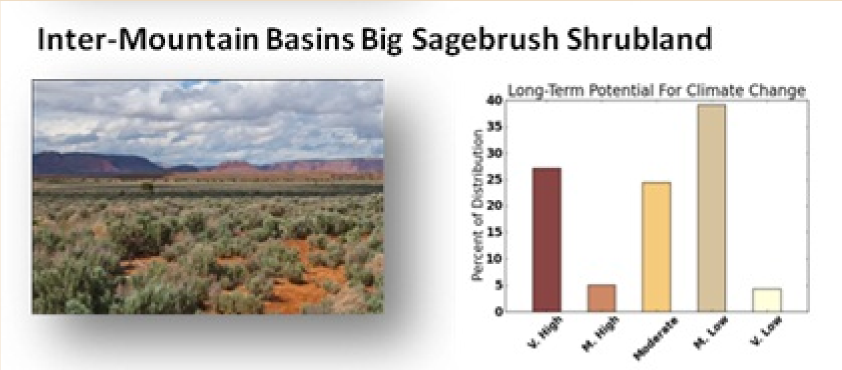


Figure 8. Histogram shows potential vulnerability to climate change for sage grouse with about 85% of its distribution in the moderate to very low range.

The amount of area in each class in the climate change map, when summarized in a histogram, indicates that about 85% of sage grouse distribution is in the moderate to very low categories (Figure 8). However, being a sagebrush obligate species, sage-grouse is very much tied to the condition of its sagebrush habitat. Of the vegetation communities, those showing the highest exposure to climate change in this analysis include the shrublands, particularly the Intermountain Basins Big Sagebrush and Montane Sagebrush Steppe (Figure 9).



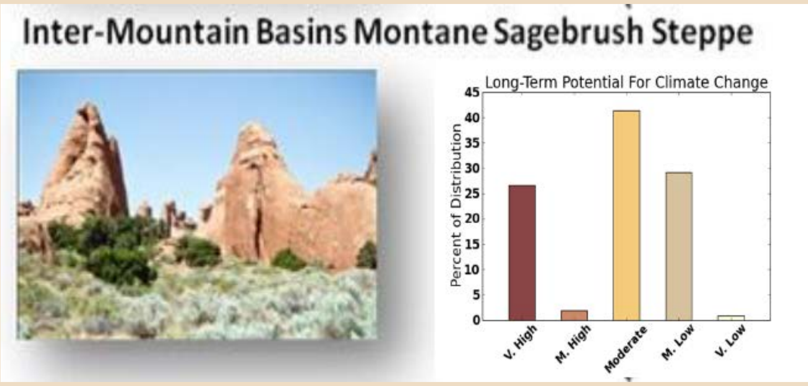


Figure 9. (Top): Histogram showing exposure to climate change for vegetation community big sagebrush shrubland and (Bottom): montane sagebrush steppe community exposure to climate change.

Almost 30% of the distribution of these two sage communities is in the High to Moderately High potential for climate change. Other estimates project that about 12% (or 87,000 km^2) of the current distribution of sagebrush will be lost with each 1° C increase in temperature (Neilson et al. 2005). However, any prediction is subject to innumerable conflicting variables and possible outcomes. For example, the largest areas of sagebrush in the Colorado Plateau occur in the northernmost portions, in the Uinta and Piceance Basins. This portion of the ecoregion is north of the influence of the summer monsoon; it may also be considered transitional to the mid- and northern latitudes, where climate change predictions may differ from those for the southwestern region. For example, some models predict that winters in mid-latitudes will be wetter as well as warmer (Miller et al. 2011). Increasing temperatures and increased atmospheric carbon dioxide favor invasive annual grasses like cheatgrass and also create an increasing incidence of fire that will favor the continued expansion of invasive annuals (Miller et al. 2011). Sagebrush communities may be further squeezed between saltbush incursion at lower elevations (that become climatically inhospitable to sagebrush) and woody vegetation infilling montane sagebrush habitats at higher elevations. Every encroachment into and fragmentation of sagebrush habitat reduces sage-grouse distribution and abundance. Thus, although climate change was not a major factor in determining candidate status for listing the greater sage-grouse (USFWS 2010), climate change will interact with other change agents (e.g., oil and gas development, invasive species, and fire) that have already degraded and reduced sage-grouse habitat to further threaten the sustainability of the species. Agencies that adopt a management strategy that withdraws core sage grouse areas from development must face the prospect that climate change may make these areas unsuitable for sage grouse. A core area strategy that works for today may have fewer options for future sage grouse conservation if the distribution of sagebrush habitats changes significantly (Smith et al. 2011).

References Cited

BLM (Bureau of Land Management). 2011a. National greater sage-grouse planning strategy charter. http://www.blm.gov/pgdata/etc/medialib/blm/wo/Communications_Directorate/public_affairs/sage-grouse_planning/documents.Par.2415.File.dat. Accessed 1/12.

BLM (Bureau of Land Management). 2011b. Greater sage-grouse interim management policies and procedures, Instruction Memorandum No. 2012-043. http://www.blm.gov/wo/st/en/info/regulations/Instruction_Memos_and_Bulletins/national_instruction/2012/IM_2012-043.html. Accessed 1/12.

BLM (Bureau of Land Management). 2011c. BLM issues greater sage-grouse conservation guidance. Bureau of Land Management News Release. http://www.blm.gov/wo/st/en/info/newsroom/2011/december/R_12_27_2011.html.

Connelly, J.W., S.T. Knick, M.A. Schroeder, and S.J. Stiver. 2004. Conservation assessment of greater sage-grouse and sagebrush habitats. Western Association of Fish and Wildlife Agencies. Unpublished Report. Cheyenne, Wyoming.

Connelly, J.W., M.A. Schroeder, A.R. Sands, and C.E. Braun. 2000. Guidelines to manage sage grouse populations and their habitats. *Wildlife Society Bulletin* 28:967–985.

Copeland, H.E., K.E. Doherty, D.E. Naugle, A. Pocewicz, and J.M. Kiesecker. 2009. Mapping oil and gas development potential in the U.S. intermountain west and estimating impacts to species. PLoS ONE 4(10) e7400, www.plosone.org. Accessed 1/12.

Crawford, J.A., R.A. Olson, N.E. West, J.C. Mosley, M.A. Schroeder, T.D. Whitson, R.F. Miller, M.A. Gregg, and C.S. Boyd. 2004. Ecology and management of sage-grouse and sage-grouse habitat. *Journal of Range Management* 57:2–19.

Doherty, K. 2008. Sage-grouse and energy development: Integrating science with conservation planning to reduce impacts. PhD dissertation, University of Montana, Missoula. 125 pp.

Doherty, K.E., J.D. Tack, J.S. Evans and D.E. Naugle. 2010. Mapping breeding densities of greater sage-grouse: A tool for range-wide conservation planning. Report prepared for BLM.

Harju, S.M., M.R. Dzialak, R.C. Taylor, L.D. Hayden-Wing, and J.B. Winstead. 2010. Thresholds and time lags in effects of energy development on greater sage-grouse populations. *Journal of Wildlife Management* 74(3): 437–448.

Holloran, M.J. 2005. Greater sage-grouse (*Centrocercus urophasianus*) population response to natural gas field development in western Wyoming. PhD dissertation. University of Wyoming, Laramie. 211 pp.

Knick, S.T., and J.W. Connelly (eds.). 2011. Greater sage-grouse: Ecology and conservation of a landscape species and its habitat. Studies in Avian Biology, Volume 38, University of California Press, Berkeley, California.

Lyon, A.G., and S.H. Anderson. 2003. Potential gas development impacts on sage grouse nest initiation and movement. *Wildlife Society Bulletin* 31: 486–491.

Miller, R.F., S.T. Knick, D.A. Pyke, C.W. Meinke, S.E. Hanser, M.J. Wisdom, and A.L. Hild. 2011. Characteristics of sagebrush habitats and limitations to long-term conservation. Pp. 145–184 *in* Knick, S.T., and J.W. Connelly (eds.), Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology (Volume 38), University of California Press, Berkeley, California.

Naugle, D.E., K.E. Doherty, B.L. Walker, M.J. Holloran, and H.E. Copland. 2011. Energy development and greater sage-grouse. Number 21 *in* Knick, S.T., and J.W. Connelly (eds.). Greater sage-grouse: Ecology and conservation of a landscape species and its habitats. Studies in Avian Biology, Volume 38, University of California Press, Berkeley, California.

Neilson, R.P., J.M. Lenihan, D. Bachelet, and R.J. Drapek. 2005. Climate change implications for sagebrush ecosystems. Pages 145–159 *in* Transactions of the 70th North American Wildlife and Natural Resources Conference, March 16–19, Arlington, Virginia, Wildlife Management Institute, http://www.wildlifemanagementinstitute.org/.

Sauer, J.R., J.E. Hines, J.E. Fallon, K.L. Pardieck, D.J. Ziolkowski, Jr., and W.A. Link. 2011. The North American Breeding Bird Survey, results and analysis 1966–2009. Version 3.23.2011, U.S. Geological Survey Patuxent Wildlife Research Center, Laurel, Maryland. http://www.mbr-pwrc.usgs.gov/bbs/bbs.html.

Smith, J., J. Vogel, K. Carney, and C. Donovan. 2011. Adaptation case studies in the western United States. Georgetown Climate Center, Gerogetown Law, Washington, D.C.

Stevens, B.S. 2011. Impacts of fences on greater sage-grouse in Idaho: Collision, mitigation, and spatial ecology. Master's Thesis, University of Idaho, Moscow, Idaho.

UDWR (Utah Division of Wildlife Resources). 2009. Utah greater sage-grouse management plan. Utah Department of Natural Resources, Division of Wildlife Resources, Publication 09-17, Salt Lake City.

USFWS (U.S. Fish and Wildlife Service). 2010. 12-month findings for petitions to list the greater sage-grouse (*Centrocercus urophasianus*) as threatened or endangered. Proposed rules, March 4, 2010. *Federal Register* 1–107.

Walker, B.L., D.E. Naugle, and K.E. Doherty. 2007. Greater sage-grouse population response to energy development and habitat loss. *Journal of Wildlife Management* 71:2644–2654.

4.2.2 Vegetation Communities: Distribution and Current Status

> Vegetation Communities
> Management Questions
>
> 1. Where are existing vegetative communities? What is their status?
> 2. What change agents have affected existing vegetation communities?

There were nine coarse filter vegetation communities evaluated for the Colorado plateau ecoregion—eight matrix vegetation communities plus riparian vegetation. For the specific vegetation communities, two different sources of data were compiled (LANDFIRE EVT v1.1 and NatureServe Landcover v2.7) to depict current distribution (Figure 4-21 A and B). All of the vegetation communities were distinct classes in the NatureServe Landcover dataset, but only six communities were mapped in LANDFIRE EVT—pinyon-juniper shrublands were not differentiated and the bedrock canyon and tableland class was combined with other barren lands in the LANDFIRE product.

Besides the differences in classes mapped, area covered for each vegetation community type according to the two classifications differed to varying degrees (Table 4-7**)**. While a visual inspection of maps of the two data sources presents each vegetation community in approximately the same general locations, the actual pixel-to-pixel agreement is generally poor, ranging in percent overlap from 0 to nearly 50 percent.

Comparison map results for the two classifications for each vegetation community for each data source are provided in Appendix B. Even though there are significant differences between the two classification systems, participants agreed that it is more appropriate to acknowledge the differences and choose the one most meaningful for a particular purpose than to attempt to hybridize the two into a single product.

Table 4-7. Comparison of area (in 1000s of acres) between NatureServe Landcover v2.7 and LANDFIRE EVT v1.1 for selected vegetation communities.

Vegetation Community	NatureServe Only	LANDFIRE Only	Both	Percent Overlap
Colorado Plateau Pinyon-Juniper Woodland	2,595	3,665	6,079	49.3
Colorado Plateau Pinyon-Juniper Shrubland	2,694	In PJ woodlands	0	0.00
Colorado Plateau Blackbrush-Mormon-tea Shrubland	1,293	2,568	1,460	27.4
Inter-Mountains Basins Big Sagebrush Shrubland	1,543	3,970	2,370	30.1
Inter-Mountains Basins Mixed Salt Desert Scrub	1,645	1,964	681	15.9
Rocky Mountain Gambel Oak-Mixed Montane Shrubland	1,424	634	660	24.3
Inter-Mountain Basins Montane Sagebrush Steppe	1,551	61	115	6.7
Colorado Plateau Mixed Bedrock Canyon and Tableland	4,598	Not mapped	0	0.00

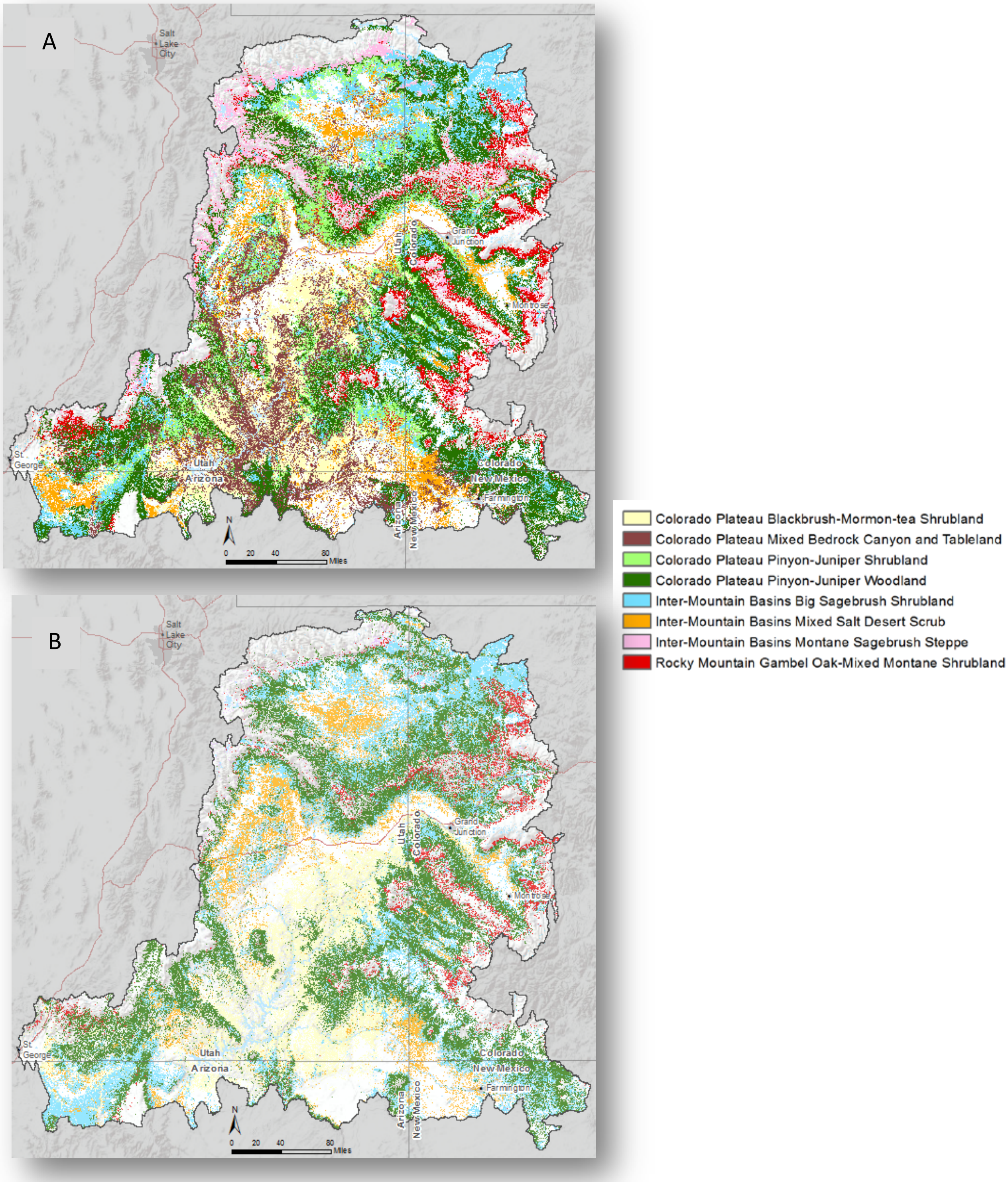


Figure 4-21. Maps show (A) NatureServe Landcover v2.7 and (B) LANDFIRE EVT v 1.1 for the matrix vegetation communities in the Colorado Plateau ecoregion. Eight vegetation communities were distinguished in the NatureServe Landcover dataset, but only six communities were mapped in LANDFIRE EVT. Pinyon-juniper shrublands were not differentiated and the bedrock canyons and tablelands class was combined with other barren lands in the LANDFIRE product.

Evaluating current status for each vegetation community is challenging in several ways. First, many of these vegetation communities are dynamic over time and space demonstrating a degree of fluidity, especially along ecotonal boundaries, driven by the pattern and timing of fire, climate, and human disturbance (Miller 2005, Miller et al. 2010). Specific plant communities are not fixed on the landscape; individual site histories and competition among species dictate what community is expressed at a particular time period. For example, some portions of a sagebrush community in the absence of periodic fire will transition into pinyon-juniper woodland or shrubland. Over time, these two communities can shift in distribution and abundance. Remotely sensed imagery, informed by physical environmental variables, limited training sites, and different levels of interpretation and expert opinion, produce different mapping outcomes such as those seen in Figure 4-21.

The LANDFIRE Biophysical Settings (BpS) data served as the reference condition to address questions of historic change. Biophysical settings provide a spatially explicit estimate of which vegetation communities would likely occur in a specific location based on physical conditions (e.g. soils, elevation, aspect, moisture, and natural fire regime). BpS is a model and a strict alignment with current distribution (i.e. LANDFIRE EVT) should not be expected. For example, the BpS and EVT maps for Inter-Mountain Basins Big Sagebrush Shrubland show considerable overlap but also some differences (Figure 4-22). It is reasonable to assume that some of these differences are the result of conversion of this community type to other land uses.

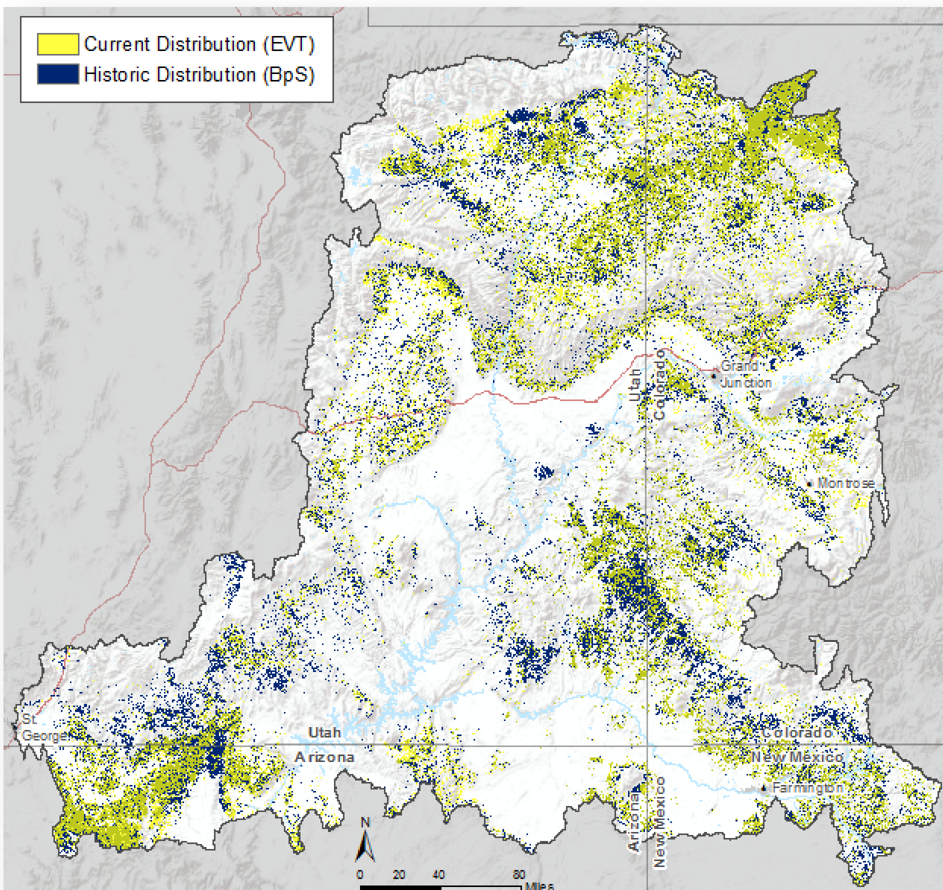


Figure 4-22. Comparison between LANDFIRE current distribution (EVT) and historic distribution (BpS) for Inter-Mountain Basins Big Sagebrush Shrublands. Differences between the two datasets represent conversion of this community type to other land uses.

Overlaying current urban and agriculture land uses, roads, invasive intrusion, and uncharacteristic native vegetation on the historic distribution of big sagebrush highlights areas of change from historic (reference) condition (Figure 4-23A). More recent disturbances (from the last 10–20 years) such as fire, mechanical treatment, and other disturbances were also obtained and overlaid in the same way (Figure 4-23B). Current distribution, historic change, and recent disturbance maps for each vegetation community are provided in Appendix B.

A total of 5.6 million acres (~22%) of the natural vegetation communities in the ecoregion as mapped by LANDFIRE BpS (representing reference condition) were affected by historic change (Table 4-8). Changes due to invasive species conversion and uncharacteristic native vegetation changes dominated the results for historic change, each affecting over 1.7 million acres (Table 4-8). Conversion from urbanization and roads altered over 1.3 million acres and intensive agriculture (excluding grazing) influenced over 760,000 acres. The greatest amount of total area changed (nearly 2.5 million acres or 30% of total BpS area) was for Inter-mountain Basins Big Sagebrush Shrubland, and this community led with maximum acres altered for urban and roads, agriculture, and invasives. Loss to invasive grasses was particularly noteworthy (~846,000 acres) for this community type. The large area of uncharacteristic native vegetation for Inter-mountain Basins Big Sagebrush Shrubland was mainly due to pinyon-juniper expansion.

Colorado Plateau Pinyon-Juniper Woodland, the second-largest vegetation community in the ecoregion, has also been affected by human land use conversion, but more significantly by invasive grasses (~273,000 acres) and uncharacteristic native vegetation conditions (~635,000 acres), which in this case is likely due to the uncharacteristic density of the pinyon-juniper trees from years of fire suppression.

Data for recent disturbance was acquired from datasets for fire perimeters for 2000–2010, LANDFIRE disturbance datasets (1999–2008), and BLM pinyon-juniper vegetation treatments (1958–2008). A total of about 822,000 acres (~3% of the combined area) were recently disturbed in the ecoregion (Table 4-9), mostly by fire (~453,000 acres) followed by mechanical treatment (~366,000 acres). As in the previous summary table, Inter-mountain Basins Big Sagebrush Shrubland was altered the most (>370,000 acres), followed by Colorado Plateau Pinyon-Juniper Woodland (>266,000 acres). One prominent figure is acres of Inter-mountain Basins Big Sagebrush Shrubland mechanically treated (~231,000 acres). Caution must be taken when interpreting this value as the purpose of the management action (e.g. removal of sagebrush to improve grazing or removal of woody intrusion to help restore sagebrush) are not differentiated in the dataset. The majority of approximately 72,000 acres of mechanical treatment in Colorado Plateau Pinyon-Juniper Woodland is likely from thinning operations.

In addition to evaluating historic and recent disturbance to the matrix vegetation communities, which provides some insight into loss and the types of recent disturbances, the existing setting in which these communities currently occur was also evaluated. For each community, the current LANDFIRE distribution was overlaid against the current terrestrial landscape intactness model results. The assumption is that each natural vegetation community is affected in various ways based on the overall intactness of its immediate neighborhood. Intactness maps and profiles for each matrix vegetation community are provided in Appendix B. The profile is a histogram of intactness versus percent of the total distribution. An example of current status for Inter-Mountain Basins Big Sagebrush Shrublands is provided in Figure 4-24A and B, with the results for NatureServe Landcover v2.7 represented in Figure 4-24A and LANDFIRE EVT v1.1 in Figure 4-24B.

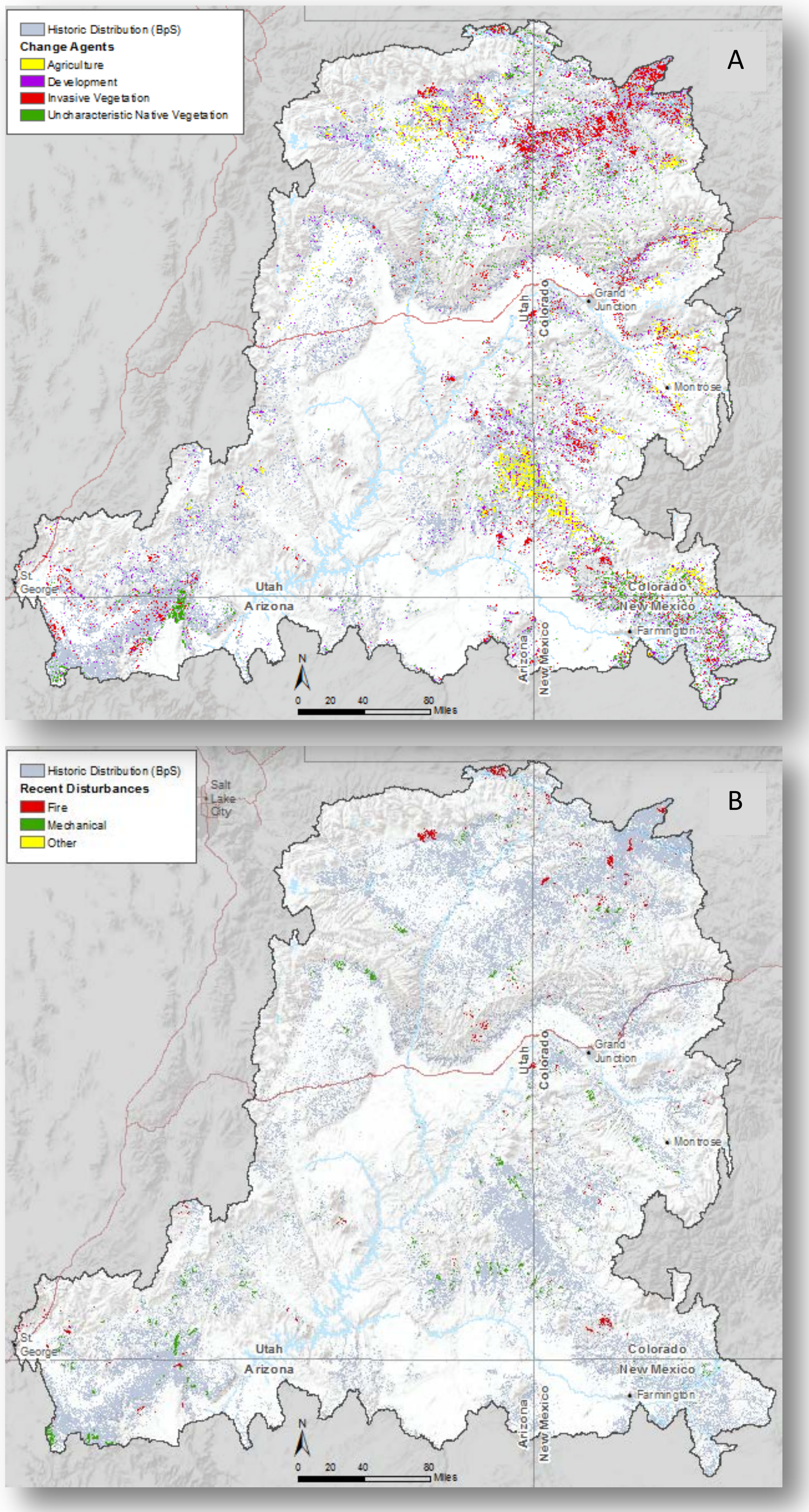


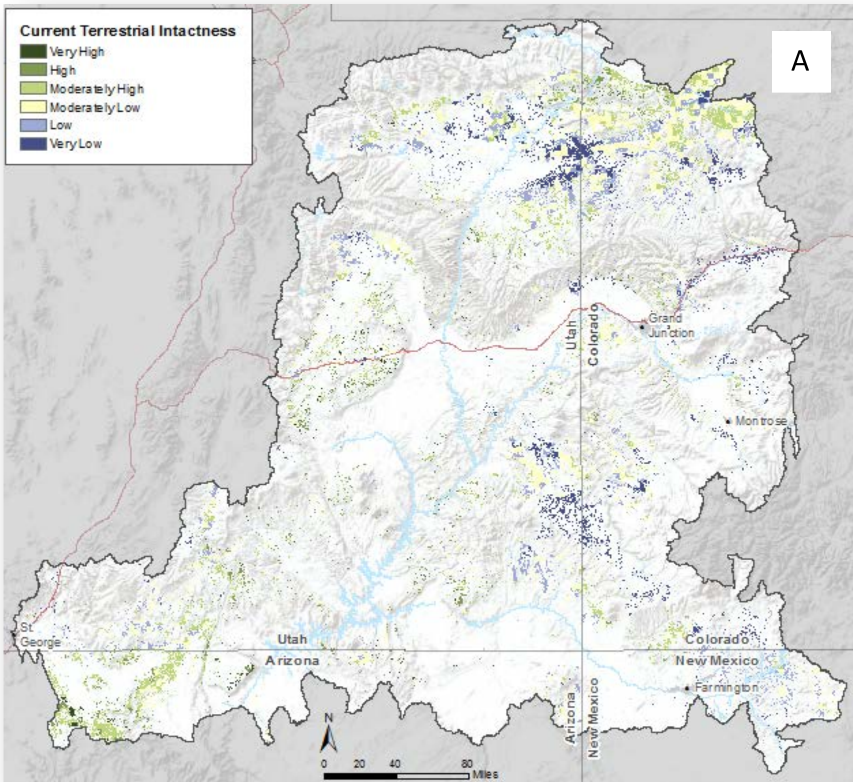
Figure 4-23. (A) Historic change and (B) recent disturbance of Inter-Mountain Basins Big Sagebrush Shrublands. See more detail by examining the live map on the data portal.

Table 4-8. Summary of area (in 1000s of acres) of historic change for each vegetation community, comparing existing vegetation to LANDFIRE BpS (representing reference condition).

Vegetation Community	Total BpS Area	Urban & Roads	Agriculture	Invasives	Unchar Native Veg	Total Changed	Percent
Colorado Plateau Blackbrush-Mormon-tea Shrubland	3,124	132	4	176	7	319	10.2%
Inter-Mountain Basins Big Sagebrush Shrubland	8,228	565	495	846	572	2,477	30.1%
Rocky Mountain Gambel Oak-Mixed Montane Shrubland	2,039	131	89	29	335	585	28.7%
Inter-Mountain Basins Montane Sagebrush Steppe	1,030	77	18	26	38	160	15.5%
Colorado Plateau Pinyon-Juniper Shrubland	94	5	2	21	10	38	40.4%
Colorado Plateau Pinyon-Juniper Woodland	7,515	229	46	273	635	1,183	15.7%
Inter-Mountain Basins Mixed Salt Desert Scrub	3,155	178	109	403	117	807	25.6%
Totals	**25,185**	**1,317**	**763**	**1,774**	**1,714**	**5,569**	

Table 4-9. Summary of area (in 1000s of acres) of recent disturbances (~10–20 years) for each matrix vegetation community.

Vegetation Community	Total BpS Area	Fire	Mechanical	Other	Total Disturbed	Percent
Colorado Plateau Blackbrush-Mormon-tea Shrubland	3,124	9	2	0	11	0.4%
Inter-Mountain Basins Big Sagebrush Shrubland	8,228	139	231	0.1	370	4.5%
Rocky Mountain Gambel Oak-Mixed Montane Shrubland	2,039	75	31	1	108	5.3%
Inter-Mountain Basins Montane Sagebrush Steppe	1,030	29	14	0.2	43	4.1%
Colorado Plateau Pinyon-Juniper Shrubland	94	0.8	0.8	0	2	1.8%
Colorado Plateau Pinyon-Juniper Woodland	7,515	194	72	0.8	267	3.6%
Inter-Mountain Basins Mixed Salt Desert Scrub	3,155	6	15	.01	21	0.7%
Totals	**25,185**	**453**	**366**	**2**	**822**	



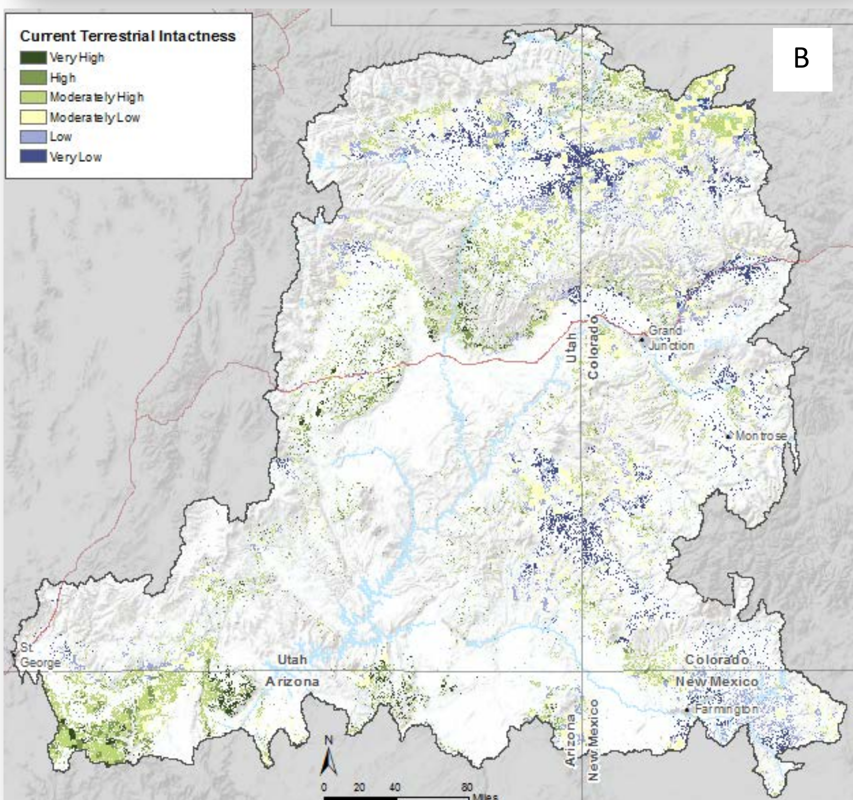


Figure 4-24. Current status for Inter-Mountain Basins Big Sagebrush for the Colorado Plateau ecoregion for A) NatureServe Landcover and B) LANDFIRE EVT from overlay of distribution with terrestrial intactness.

4.2.2.1 Riparian Vegetation

Riparian ecological systems have undergone significant physical and biological changes throughout the ecoregion because of direct conversion to other uses; changes in the natural flow regimes and suppression of fluvial processes (Busch and Smith 1995, Stromberg 2001, Stromberg et al. 2007a); livestock grazing (Armour et al. 1994); and alien species invasion, e.g., tamarisk (Horton 1977, Graf 1978, Stromberg et al. 2007b). As much as 90% of pre-settlement riparian ecosystems have been lost (LUHNA 2011). Livestock grazing has damaged approximately 80% of stream and riparian ecosystems in the western US (Belsky et al. 1999). Grazing alters streamside morphology, increases sedimentation, degrades riparian vegetation through trampling and consumption and causes nutrient loading to the system. Invasive plants such as tamarisk often successfully out-compete native species, because tamarisk produces seeds multiple times in a year; it is also more tolerant of drought and flow alterations than natives (Stromberg et al. 2007a, Merritt and Poff 2010). Riparian issues are covered in depth in the tamarisk case study insert.

Mapping riparian systems is difficult to do using satellite remote sensing. The narrow linear nature of the community makes it difficult to delineate with high levels of accuracy. NatureServe Landcover (v2.7) was used for the REA assessment to assess current distribution. Status was evaluated using the terrestrial landscape intactness results at 4km resolution. According to the NatureServe Landcover data, about 1,735,000 acres of riparian vegetation currently exist in the ecoregion. Status results, based on the terrestrial landscape intactness model, show that the dominant category is moderately high with the rest of the results skewed to the lower intactness classes (Figure 4-25). Although a 4 km X 4 km grid cell is an appropriate reporting unit for a region-wide assessment, it is less discriminating in characterizing linear communities.

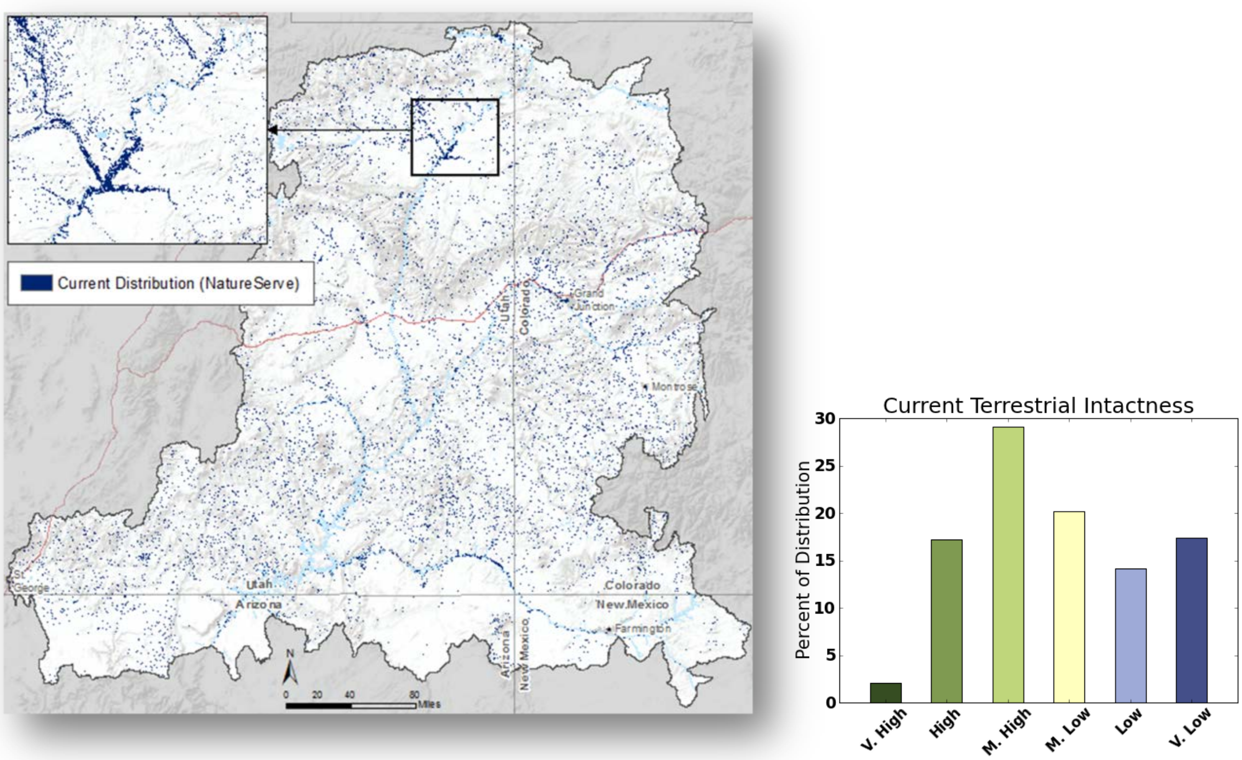


Figure 4-25. Detail of riparian vegetation distribution (in blue) based on NatureServe Landcover v2.7 for the Colorado Plateau ecoregion and general status histogram based on the terrestrial intactness model.

4.2.3 Evaluating Designated Sites: Distribution and Current Status

Approximately 28% (~12.4 million acres) of the Colorado Plateau ecoregion is currently under federal, state, local government or private conservation land designation, including conservation easements (Figure 4-26). These data are limited to designated protected lands and do not include other conservation lands under current land management plans by the various agencies. In some instances, these land designations are nested and ranked, in which case the more protective designation is displayed over the top of another (e.g. wilderness area above a national recreation area). Approximately 1,400 miles of wild and scenic rivers and national trails are also included in the map.

Status of these lands was evaluated by overlaying the designated lands polygons on terrestrial landscape intactness and summarizing the results (Figure 4-27, Table 4-10). Wilderness Study Areas made up the largest proportion of the protected areas. Other categories occupying over 1 million acres included Designated Roadless Areas, Other Protected Lands, National Monuments, National Recreation Areas, and Areas of Critical Environmental Concern. Wilderness Areas and National Parks accounted for somewhat less than 1 million acres each and all other classes combined (National Conservation Areas, State Wildlife Management Areas, State Parks, and National Wildlife Refuges) made up just less than 500,000 acres.

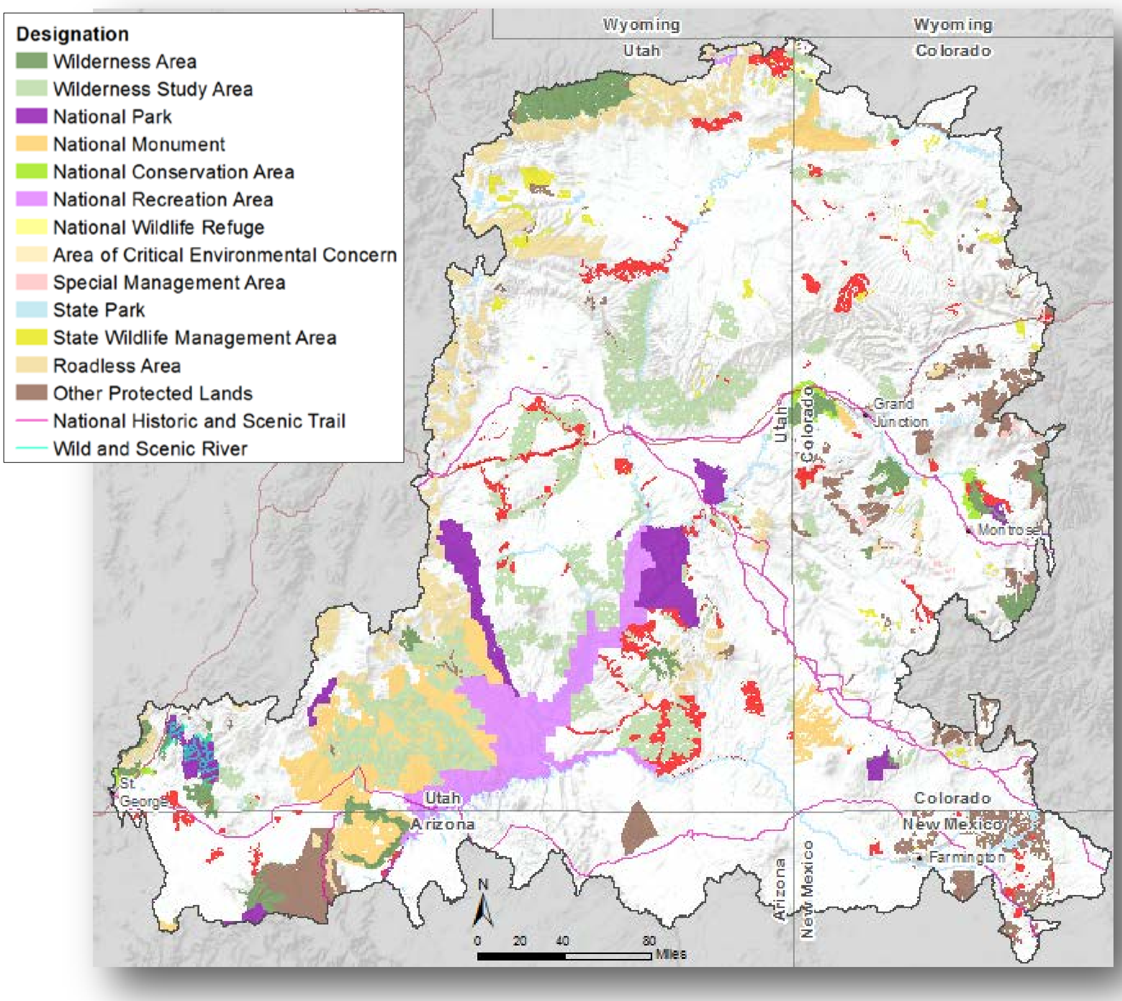
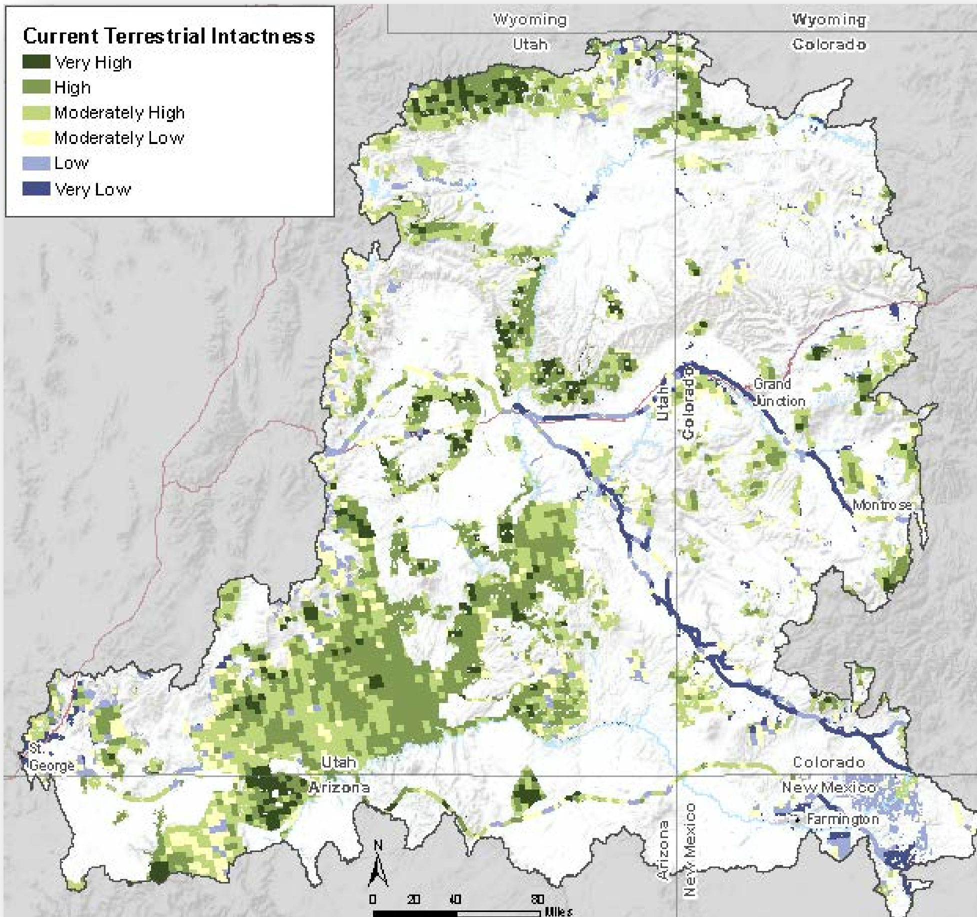


Figure 4-26. Map of designated lands in the Colorado Plateau ecoregion.

In general, terrestrial landscape intactness for special designated lands was heavily skewed (>75% of the area) towards more intact landscapes; however, not all designation classes scored equally (Figure 4-28). Wilderness Areas, Wilderness Study Areas, National Monuments, and National Recreation Areas showed the best intactness profiles. National Parks did well, but they had significant areas in low intactness classes, which may be surprising. However, several of the parks (e.g. Bryce Canyon and Arches) are not large and they are surrounded by various classes of development. Designations such as National Conservation Areas, State Parks, State Wildlife Management Units, and National Wildlife Refuges showed lower overall intactness.



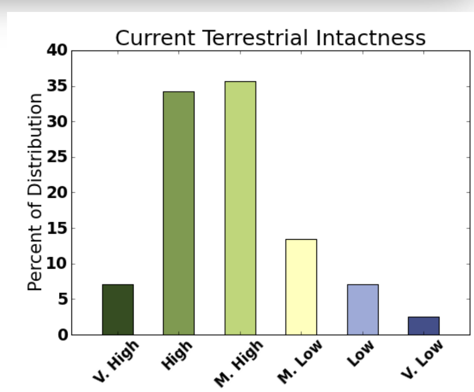
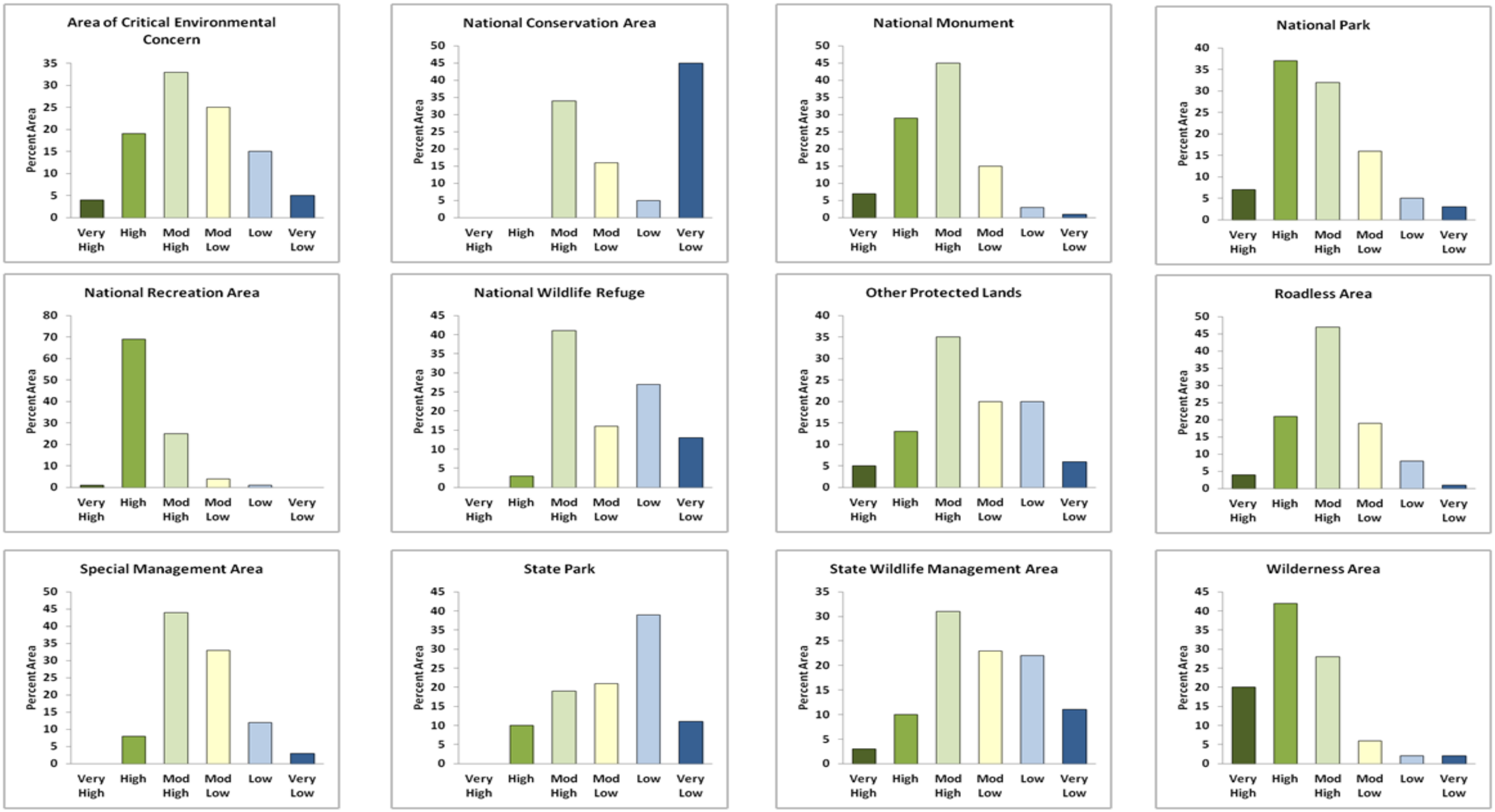


Figure 4-27. Status of designated lands in the Colorado Plateau ecoregion based on current terrestrial landscape intactness.

Table 4-10. Total area (in 1000s of acres) in each status category for all designated lands in the Colorado Plateau ecoregion.

Designation Category	Very High	High	Moderately High	Moderately Low	Low	Very Low	Total Area (acres)
Area of Critical Environmental Concern	38	195	336	258	151	50	1,028
National Conservation Area	0	0	36	17	5	48	106
National Monument	113	461	724	239	44	11	1592
National Park	65	347	304	147	50	27	940
National Recreation Area	11	874	313	48	8	2	1,256
National Wildlife Refuge	0	1	8	3	6	2	20
Other Protected Lands	78	216	564	327	329	99	1,613
Roadless Area	70	361	803	320	138	9	1,701
Special Management Area	0	4	21	16	6	1	48
State Park	0	5	10	11	21	6	53
State Wildlife Management Area	8	29	90	65	63	31	286
Wilderness Area	183	399	267	58	19	16	942
Wilderness Study Area	316	1,367	963	163	51	10	2,870
Total	**882**	**4,259**	**4,439**	**1,672**	**891**	**312**	**12,455**



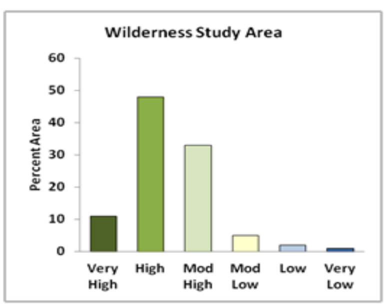


Figure 4-28. Terrestrial landscape intactness profiles for each designated land class. Note that the y-axis (percent area) varies for each histogram.

4.2.4 Connectivity

Least-cost path analysis for the Natural Landscape Blocks as described in the methods section (Chapter 3, Section 3.2.5) provided a number of key linkage zones for the ecoregion (Figure 4-29). Potential linkages were hand drawn between neighboring natural landscape blocks by connecting each one using a system of drawn sticks (centroid to centroid, as pictured in Section 3.2.5). Sticks identified the pairs of blocks to evaluate; the ArcGIS tools Cost Distance and Corridor determined the final least-cost corridors. Natural blocks included the designated lands. Most of the linkage corridors were concentrated in the eastern third of the study area where much of the human disturbance is located. Corridors do not exist where human disturbance is most heavily concentrated, e.g., in the central Uinta or San Juan basins.

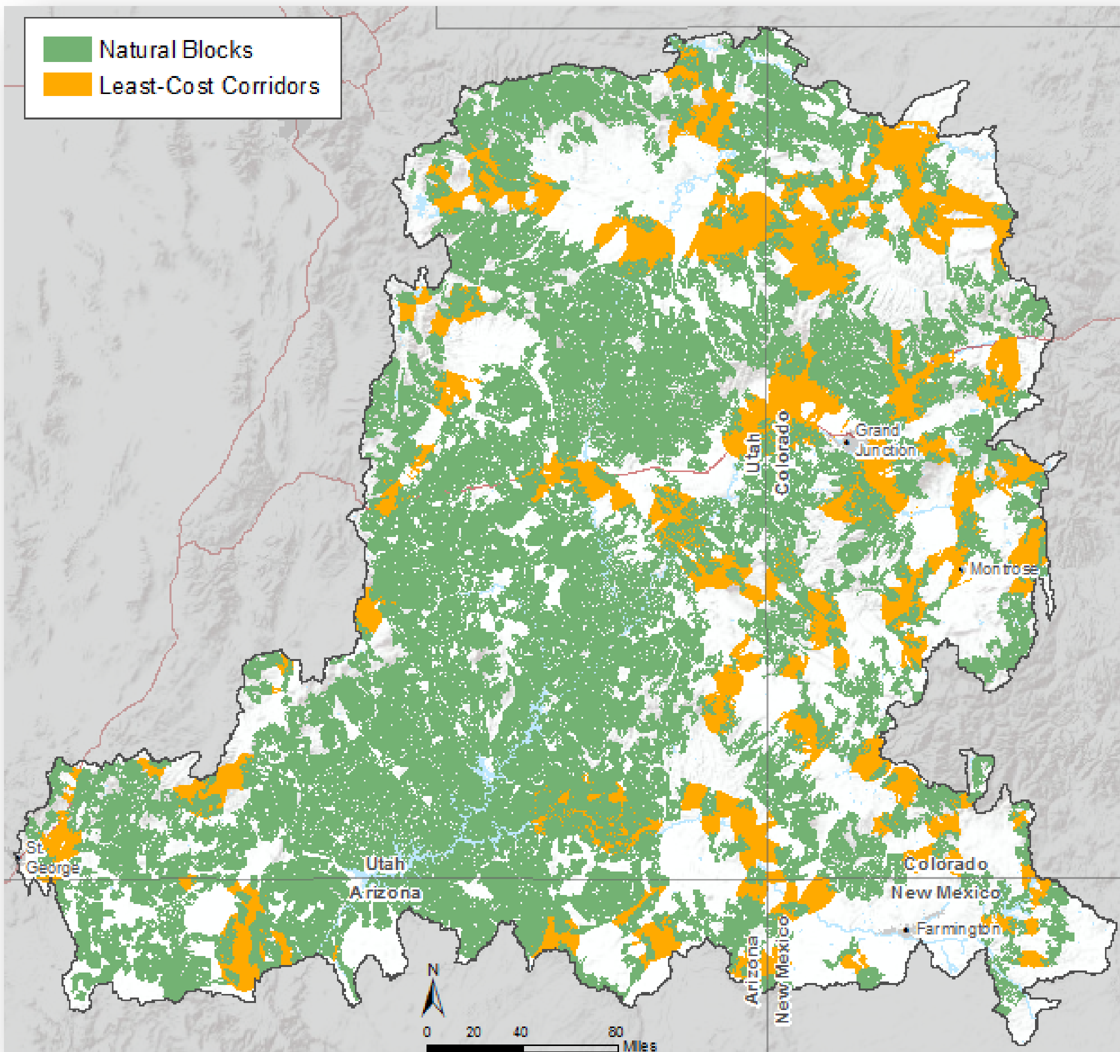


Figure 4-29. Landscape connectivity results based on generic (non-species specific) least-cost path analysis for the Colorado Plateau ecoregion.

4.2.5 References Cited

Armour, C., D. Duff, and W. Elmore. 1994. The effects of livestock grazing on western riparian and stream ecosystems. *Fisheries* 19(9): 9–12.

Belsky, A.J., A. Matzke, and S. Uselman. 1999. Survey of livestock influences on stream and riparian ecosystems in the western United States. *Journal of Soil and Water Conservation* 54:419–431.

Busch, D.E., and S.D. Smith. 1995. Mechanisms associated with the decline of woody species in riparian ecosystems of the Southwestern U.S. *Ecological Monographs* 65:347–370.

Doherty, K. 2008. Sage-grouse and energy development: Integrating science with conservation planning to reduce impacts. PhD dissertation, University of Montana, Missoula. 125 pp.

Graf, W.F. 1978. Fluvial adjustment to the spread of tamarisk in the Colorado Plateau region. *Geological Society of America Bulletin* 89**:** 1491–1501.

Gross, J.E., L.C. Stoddart, and F.H. Wagner. 1974. Demographic analysis of a northern Utah jackrabbit population. *Wildlife Monographs* 40:3–68.

Horton, J.S. 1977. The development and perpetuation of the permanent tamarisk type in the phreatophyte zone of the southwest. Pages 124–127 *in* Importance, preservation and management of riparian habitat: A symposium. General Technical Report RM-43, U.S. Forest Service, Washington, D.C.

Howard, R.P., and M.L. Wolfe. 1976. Range improvement practices and ferruginous hawks. *Journal of Range Management* 29:33–37.

Livingston, M.F., and S.D. Schemnitz. 1996. Summer bird/vegetation associations in tamarisk and native habitat along the Pecos River, southeastern New Mexico. Pages 171–180 *in* Shaw, D.W., and D.M. Finch (technical coordinators), Desired future conditions for southwestern riparian ecosystems: Bringing interests and concerns together. General Technical Report RM-GTR-272, U.S. Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado.

LUHNA. 2011. Land Use History of North America: Colorado Plateau. US Geological Survey, http://cpluhna.nau.edu/Biota/riparian_communities.htm

Lupis, S. G., K. D. Bunnell, T. A. Black, and T. A. Messmer. 2007. Utah Gunnison's prairie dog and white-tailed prairie dog conservation plan: Draft #5. Utah Division of Wildlife Resources, Salt Lake City, Utah.

Merrit, D.M., and N.L. Poff. 2010. Shifting dominance of riparian *Populus* and *Tamarix* along gradients of flow alteration in western North American rivers. *Ecological Applications* 20(1):135–152.

Miller, M.E. 2005. The structure and functioning of dryland ecosystems—Conceptual models to inform long-term ecological monitoring. USGS Scientific Investigations Report 2005-5197, U.S. Geological Survey, Reston, Virginia. 73 pp.

Miller, D.M., S.P. Finn, A. Woodward, A. Torregrosa, M.E. Miller, D.R. Bedford, and A.M. Brasher, A.M. 2010. Conceptual ecological models to guide integrated landscape monitoring of the Great Basin, USGS Scientific Investigations Report 2010-5133, U.S. Geological Survey, Reston, Virginia. 134 pp.

Sogge, M.K., S.J. Sferra, and E.H. Paxton. 2008. *Tamarix* as habitat for birds: Implications for riparian restoration in the southwestern United States. *Restoration Ecology* 16(1):146–154.

Stromberg, J. C. 2001. Restoration of riparian vegetation in the southwestern United States: Importance of flow regimes and fluvial dynamism*. Journal of Arid Environments* 49:17–34.

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

Stromberg, J.C., S.J. Lite, R. Marler, C. Paradzick, P.B. Shafroth, D. Shorrock, J.M. White, and M.S. White. 2007b. Altered stream-flow regimes and invasive plant species: the *Tamarix* case. *Global Ecology and Biogeography* 16:381–393.

Van Dyke, F.G., R.H. Brocke, H.G. Shaw, B.B. Ackerman, T.P. Hemker, and F.G. Lindzey. 1986. Reactions of mountain lions to logging and human activity. *The Journal of Wildlife Management* 50(1):95–102.

Photo: National Park Service, N. Herbert.

4.3 Change Agent Distribution and Intensity

Current Change Agent Management Questions

MQF1. Where are areas dominated by tamarisk and cheatgrass?

MQE1. Where are areas that have been changed by wildfire between 1999 and 2009?

MQE2. Where are areas with potential to change from wildfire?

MQE3. Where are the Fire Regime Condition Classes?

MQE4. Where is fire adverse to ecological communities, features, and resources of concern?

MQG1. Where are areas of planned development?

An assessment of the status of conservation elements must be conducted with reference to both natural and anthropogenic disturbance factors. Although the current distribution and status of REA conservation elements were presented together in Section 4.2 to economize on presentation space, the status or condition of various conservation elements should not be discussed without examining the risks that these resources experience from a collection of regional disturbances or change agents. The primary change agents affecting the region were introduced in Chapter II Introduction, Section 2.4.3 (Table 2-4). Those change agents related to current conditions are presented in this section: invasive vegetation, wildfire, and current development. Change agents associated with future conditions, near-term future (2025) development and intactness, potential energy development, and climate change, are presented in Chapter V Potential Future Conditions in the Colorado Plateau.

4.3.1 Invasive Vegetation

While there are multiple invasive species in the Colorado Plateau, two invasive plant species of concern, cheatgrass (*Bromus tectorum*) and tamarisk (*Tamarix* spp.), have been selected for the Colorado Plateau REA because they are considered significant change agents in the region. These species alter ecosystem processes, such as fire regimes; they have the potential to expand their distribution in spite of human and natural disturbances and to adapt and shift their range in response to climate change. Invasive annuals out-compete native species by using soil nutrients and water at a greater rate or earlier in the season and regularly producing greater biomass (DeFalco et al. 2007). As these species expand in distribution and dominance on the landscape, native species and communities become increasingly marginalized, which over time may seriously degrade the function of these ecosystems.

Cheatgrass (*Bromus tectorum*) is one of the key invasive species in the Colorado Plateau due to its strong potential to mediate a feedback cycle that can dramatically change the natural fire regime of ecologically significant vegetation communities, such as sagebrush. It is an annual grass native to Europe, northern Africa, and southwestern Asia that was accidentally introduced to North America in the mid- to late-1800s (Mack 1981, Young 2000, Novack and Mack 2001). It had occupied much of its present range by the early 1900s (Novack and Mack 2001, Mack 1981). It is particularly invasive in the western U.S. due, in part, to grazing (Mack 1981). Its 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 comata*), which are not only important forage plants, but also essential to maintaining soil stability, wind and water erosion control, and natural fire regimes (USU Cooperative Extension 2011).

Another key invasive species in the region is tamarisk, with multiple species and hybrids present (e.g., *Tamarix chinensis, T. gallica,* and *T. ramosissima*). Tamarisk became widely distributed in the 1800s, when it was planted as an ornamental plant; it is now found throughout nearly all western and southwestern states (Lovich 2000). Tamarisk is of particular 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, and alters fire regimes (Busch and Smith 1995, Glenn et al. 1998). Tamarisk affects native wildlife by changing the composition of forage plants and the structure of native riparian systems. For more discussion about riparian ecosystems and tamarisk, see the Tamarisk Case Study Insert.

Accurately mapping the full distribution of major invasive vegetation species is quite difficult due to a general lack of systematically sampled occurrences, the difficulty in distinguishing low seasonal abundance within the satellite imagery often used to create land cover classifications, and the requirement of carefully calibrated satellite imagery time series to capture the particular phenology of the invasive species, such as early season green-up. Invasives may be difficult to detect where they are co-dominants, present in the understory, or not actively growing during the season of imagery. Results from multiple mapping efforts were combined to estimate the extent of major invasive vegetation species in the Colorado Plateau (Figure 4-30).

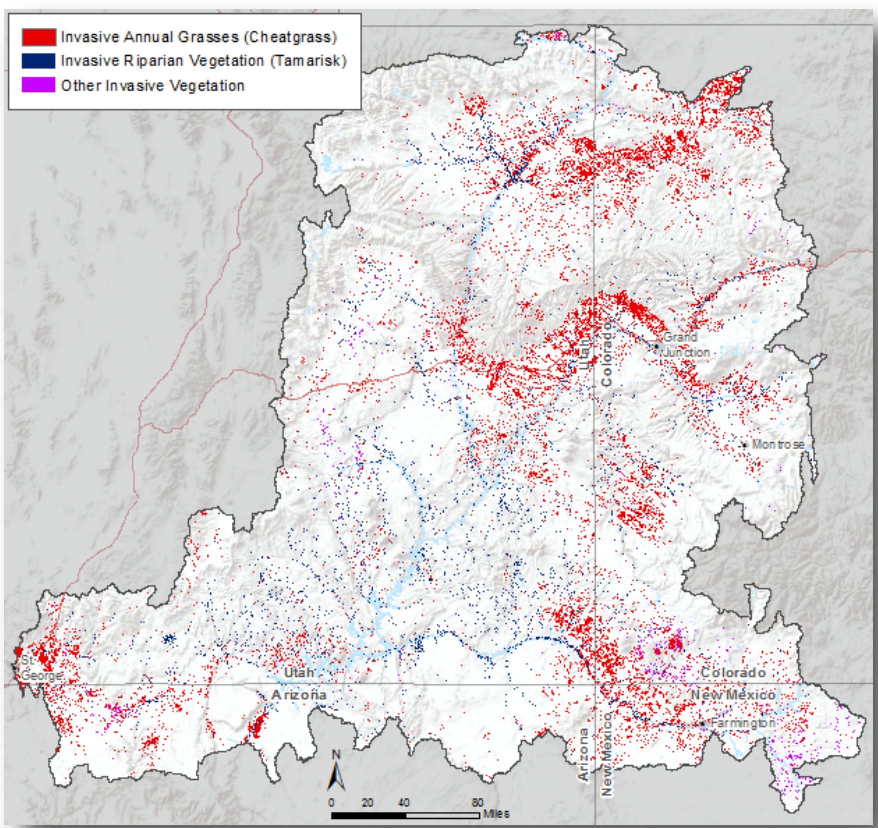


Figure 4-30. Distribution of major invasive vegetation species, including cheatgrass and tamarisk.

To create the map, invasive annual grass classes were extracted from LANDFIRE Existing Vegetation Type (EVT v1.1) and NatureServe Landcover (v2.7, classes include cheatgrass, red brome, and other species) and combined with the results of a modeled distribution of early season invasives (including cheatgrass, red brome, and Sahara mustard) for the Colorado Plateau (created by J. Hansen and T. Arundel of USGS). Similarly, invasive riparian vegetation classes were extracted from LANDFIRE and NatureServe (classes include tamarisk species [*Tamarix* spp.] and Russian olive [*Elaeagnus angustifolia*]) and combined with a tamarisk probability map (Jarnevich et al. 2011) and available tamarisk occurrence data. Other invasive vegetation classes mapped by LANDFIRE and NatureServe were also extracted. These data and models likely underestimate the total distribution of invasive vegetation, because most methods used remotely-sensed imagery and required dominance of a site by these species to be detectable. Where these species occur as less dominant components of the vegetation community, they may expand and dominate quickly due to disturbance, land use, and climate change.

4.3.2 Changes in Fire Regime

Fire is a natural ecosystem process in many regions, including the Colorado Plateau. 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 vegetation 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 (Schoennagel and Nelson 2010). In contrast, other communities adapted to very infrequent fire are now threatened by increases in fire frequency due to invasive plants and human ignitions.

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 rarely 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, Brooks et al. 2004). 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. 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, have 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 (DellaSala et al. 2004, Lehmkuhl et al. 2007, Hurteau and North 2009).

For the management question, *Where are the Fire Regime Condition Classes?,* current fire regime departure compared to reference conditions was estimated using a combination of existing measures of vegetation departure (LANDFIRE Fire Regime Condition Class Departure Index v1.0) and calculated departure of fire frequency and severity from expert estimates of current fire regime parameters (Figure 4-31). Vegetation departure describes the degree to which the proportions of various successional stages of a particular community are similar to the proportions that would be expected to occur over space and time under reference conditions. Vegetation departure increases with increasing abundance of invasive vegetation or in response to greater proportions of later or earlier successional vegetation than would have been expected under reference conditions.

Current estimates of fire regime (fire frequency and severity) were estimated for the 40 most abundant Biophysical Settings (from LANDFIRE Biophysical Settings v1.0) and applied to the full distribution of each system within the ecoregion (see Appendix A). Typically, estimates of fire regime are developed for smaller landscape reporting units tied to the reference condition fire regime characteristics of frequency and size (larger, infrequent fire regimes require larger reporting units); however, this was not feasible within the scope of this REA. It is very difficult to estimate both current and reference condition fire regimes with high confidence; this is due in large part to incomplete knowledge of fire history for each system within each unique landscape in the ecoregion and the relatively short period over which current estimates are drawn. Vegetation communities with historically frequent fires (Fire Regime Groups I and II; Table 4-11) can be described in terms of the number of fire cycles missed in recent decades, due in part to effective fire suppression.

Table 4-11. Fire Regime Group Characteristics

Fire Regime Group	Fire Return Intervals
I	≤ 35 year fire return interval, low and mixed severity
II	≤ 35 year fire return interval, replacement severity
III	35–200 year fire return interval, low and mixed severity
IV	35–200 year fire return interval, replacement severity
V	> 200 year fire return interval, any severity

In contrast, communities with historically infrequent fire are more difficult to estimate, because the period of analysis must be longer than is available for current estimates. Therefore, these estimates of current fire regimes should be treated with some degree of caution; while these are based on the best available information and expert understanding of the systems, they may under- or over-estimate actual fire regime departure. These estimates are also conflated due to necessity of summarizing results at the ecoregion scale because averaging across larger areas tends to drive estimates of departure toward the middle.

The analysis of reference condition fire regimes was extended to answer the management question, *Where is fire adverse to ecological communities, features, and resources of concern?* Systems were selected with historically longer fire return intervals (≥35 years, Fire Regime Groups III, IV, and V, Table 4-10) from the LANDFIRE Fire Regime Groups dataset where they intersected invasive vegetation mapped for this REA (Figure 4-29) and uncharacteristic exotics and uncharacteristic native vegetation classes from the LANDFIRE Succession Classes dataset (Figure 4-32). While fire may not always be adverse to these systems, the presence of invasives or uncharacteristic native vegetation increases the likelihood of negative post-fire vegetation response. In particular, fire may be particularly adverse to long fire return interval systems (Fire Regime Group V) occupied by invasives because native species may take longer to recover post-fire, whereas invasives may greatly expand in distribution and dominance.

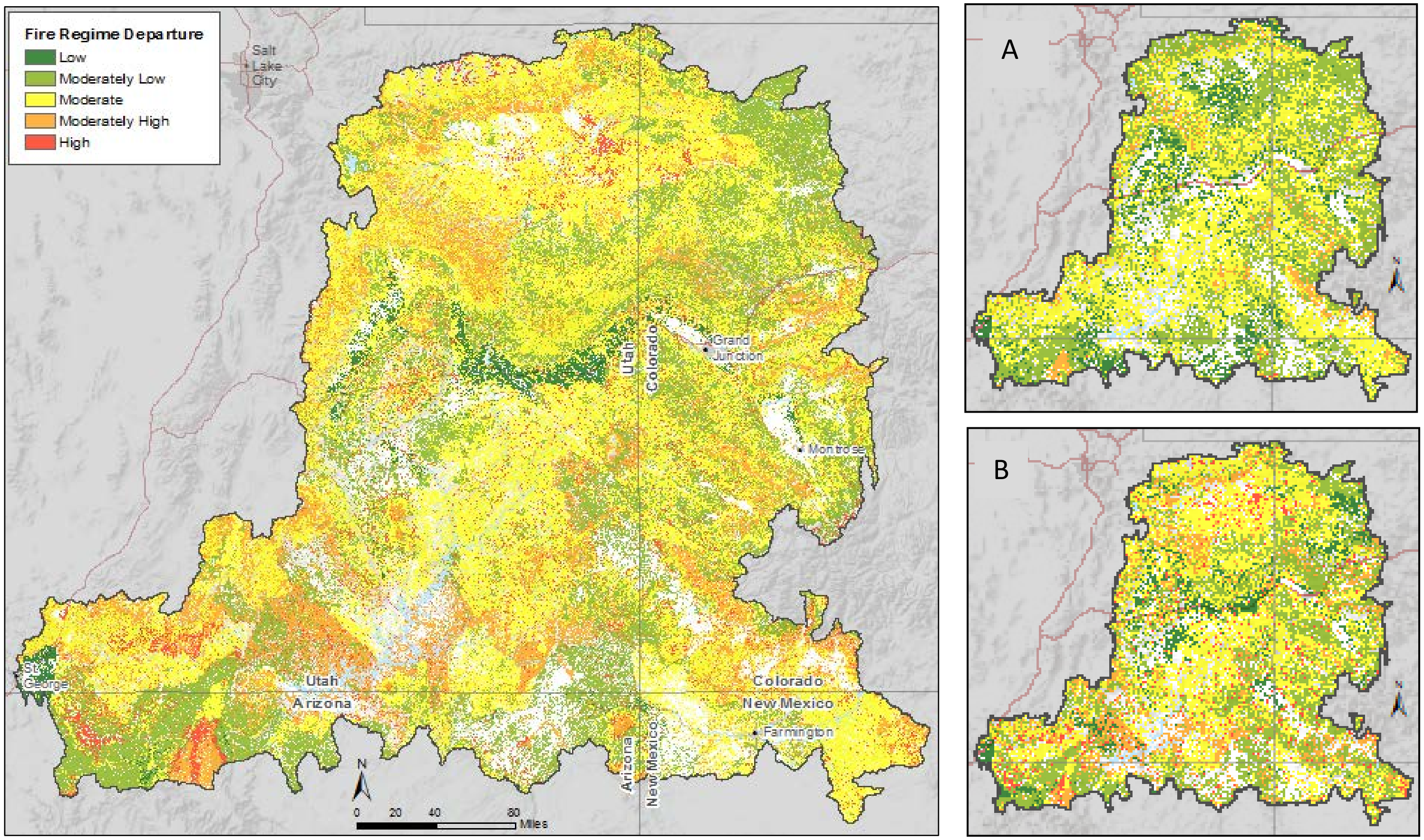


Figure 4-31. To answer the management question, *Where are the Fire Regime Condition Classes?,* map on the left depicts fire regime departure showing the maximum departure value between (A) existing measures of vegetation departure (LANDFIRE Fire Regime Condition Class Departure Index v1.0) and (B) calculated departure of fire frequency and severity from expert estimates of current fire regime parameters for the Colorado Plateau ecoregion.

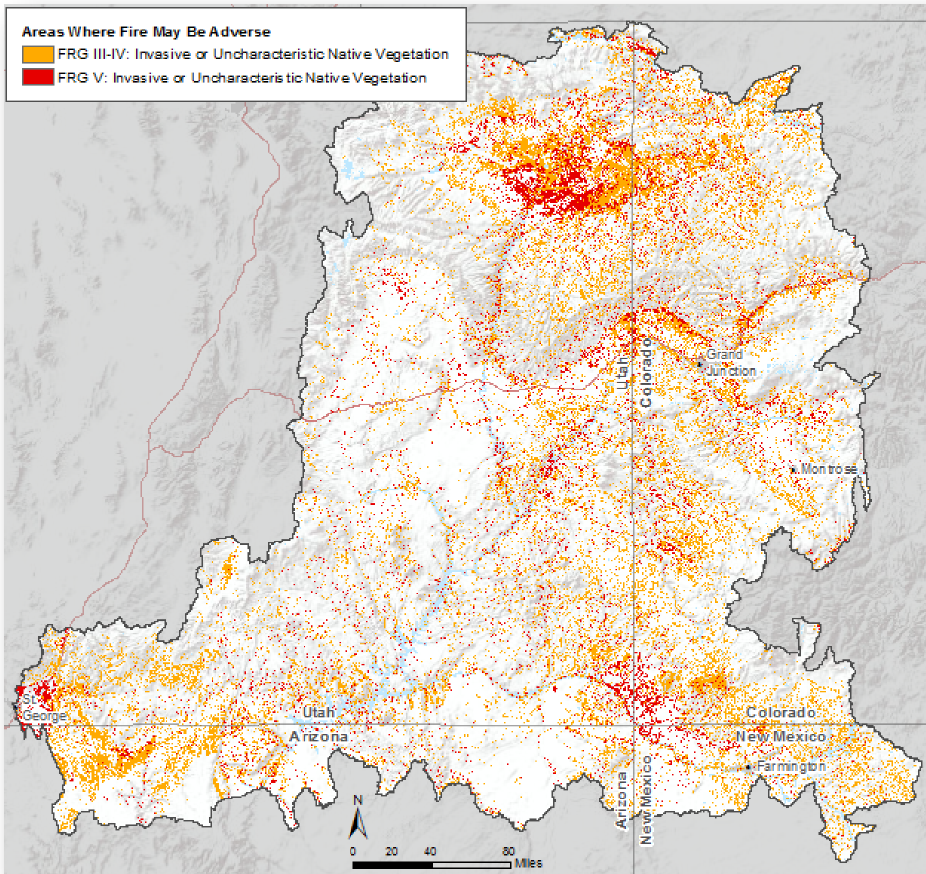


Figure 4-32. Areas where fire may be adverse to vegetation communities in the Colorado Plateau ecoregion. Systems were selected with historically longer fire return intervals (≥35 years, Fire Regime Groups III, IV, and V, Table 4-10) from the LANDFIRE Fire Regime Groups dataset where they intersected invasive vegetation mapped for this REA and uncharacteristic exotics and uncharacteristic native vegetation classes from the LANDFIRE Succession Classes dataset.

Areas changed by recent (1999–2010) wildfires were estimated using fire perimeters (GeoMAC 2000–2010, http://www.geomac.gov/index.shtml) supplemented with estimates of fire severity (LANDFIRE Disturbance datasets 1999–2008) where available (Figure 4-33). While efforts were made to compile the most complete dataset of fires during this period, some fires may be absent from both the fire perimeters dataset and the LANDFIRE disturbance dataset. LANDFIRE estimates of fire severity should be interpreted with caution in shrub and grassland systems because methods and definitions of fire severity were developed primarily for forested systems. Any area that has experienced fire has been changed by it to a degree that generally increases with increasing severity. High severity fires tend to result in early successional vegetation states followed by a recovery period during which characteristic species recolonize the site. However, areas with uncharacteristically high severity (due in part to fire suppression and fuel buildup) may transition to a different vegetation state, such as persistent invasive vegetation. It is not possible to evaluate the underlying change in vegetation resulting from fire because of the lack of accurate regional maps of pre- and post-fire vegetation. While the most recent version of LANDFIRE Existing Vegetation (v1.1) has been updated in areas of disturbance, the updates are not necessarily an accurate reclassification of the post-fire vegetation, but instead appear to be the result of applying a rule set based on pre-fire vegetation type and fire severity coupled with a systematic update of the entire product to correct areas of major inaccuracy.

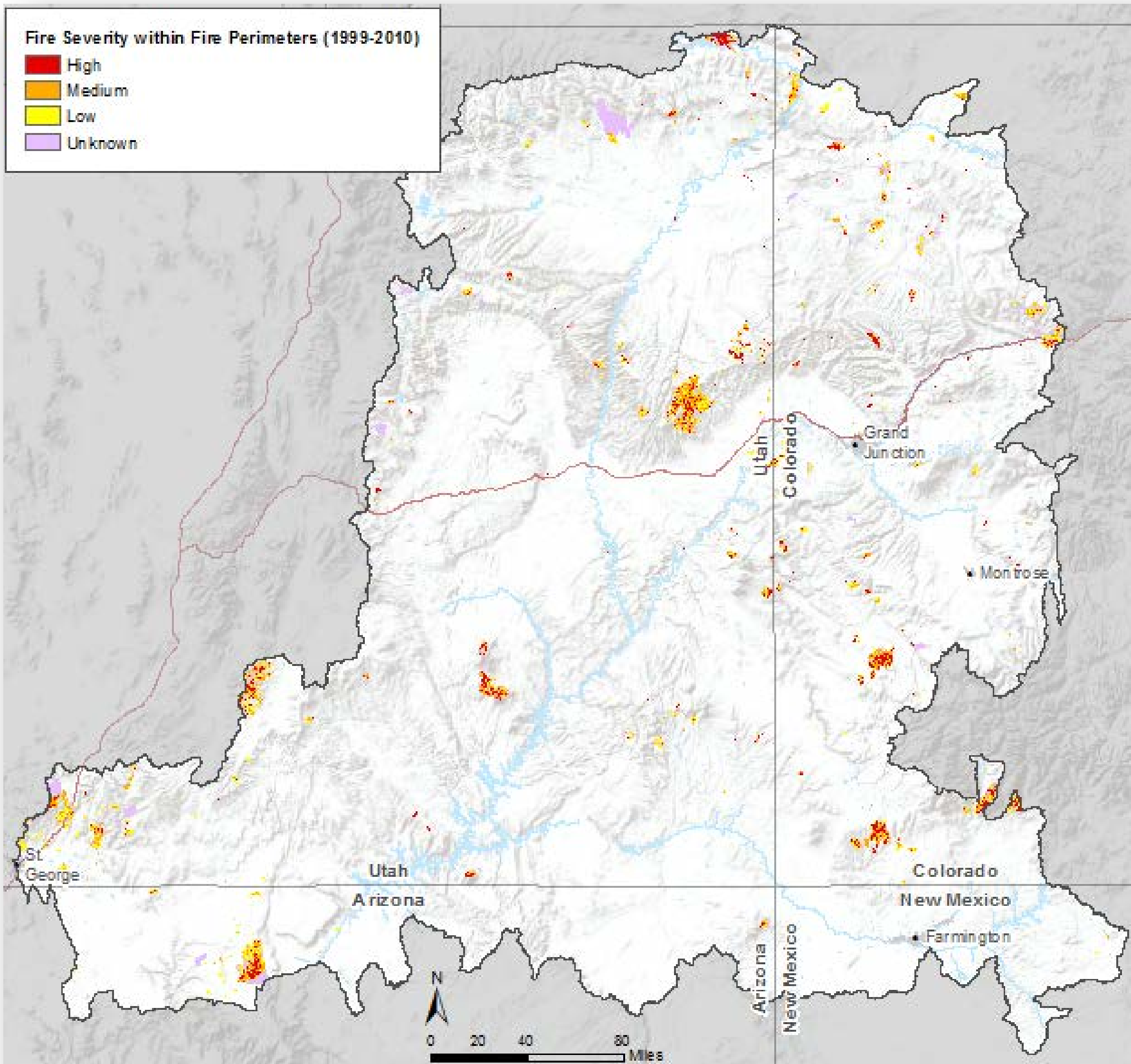


Figure 4-33. Map of fire perimeters annotated by severity (where available) answering the management question, *Where are the areas that have been changed by wildfire between 1999 and 2009*?

MaxEnt models of potential fire occurrence were developed to answer the final fire-related question (*Where are the areas with potential to change from wildfire?* Figure 4-34). In reality, fire has the potential to cause a greater or lesser magnitude of change due to fine scale fuel conditions, local fire behavior, fire weather, and pre-fire vegetation sensitivity to fire disturbance along with many other factors. It is not possible given existing data to evaluate these factors at the ecoregion scale. Instead, the focus was on predicting where fires are likely to occur based on the premise that this would provide a meaningful context for more detailed, local assessments of potential impacts due to fire.

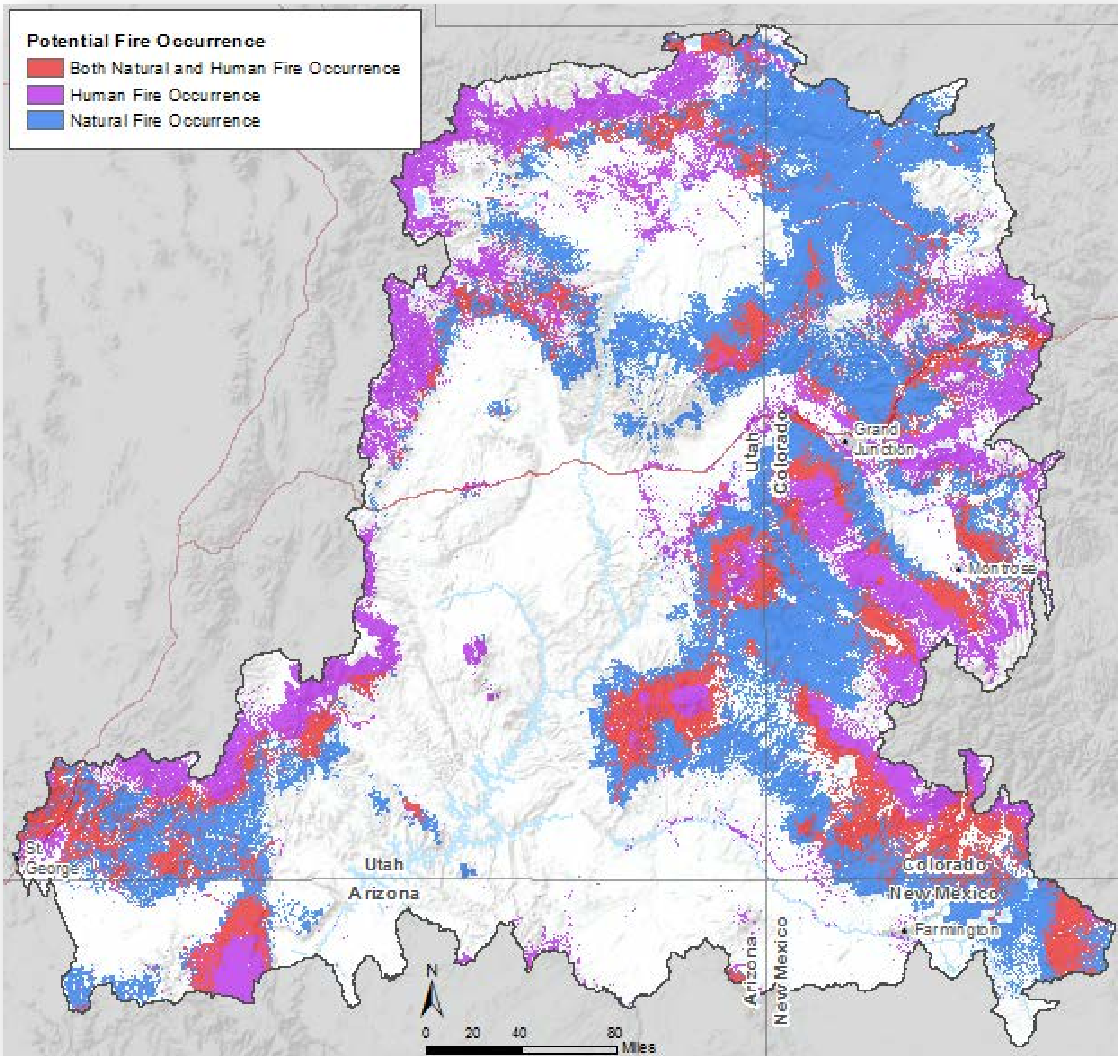


Figure 4-34**.** Potential fire occurrence map from combined human and natural fire occurrence MaxEnt models for the Colorado Plateau ecoregion answers the management question *Where are the areas with potential to change from wildfire?*

Thirty years of fire occurrence data were subdivided into human and naturally-caused fires (11,971 human caused fires and 23,716 naturally-caused fires) and a separate MaxEnt model was developed for each because of the very disparate relationship between fire cause and underlying geographic and environmental variables. Both models performed somewhat poorly (human model *Area Under Curve* or AUC: 0.678 and natural model AUC: 0.618). The results of these models should be interpreted with caution due to somewhat poor accuracy and because the models represent the likelihood of fire occurrence based on point-based fire occurrence data. Many ecologically significant fires may spread over large areas due to fuel and fire weather characteristics not captured by these models, and also may affect much larger areas than the occurrence points used to depict them. Thus, some fires shown in Figure 4-33 are not predicted as having high probability of fire occurrence in these models (Figure 4-34).

The most influential factors in the human model include: distance to recreation areas, distance to roads and highways, and annual and summer precipitation. The most influential factors in the natural model include: annual precipitation, summer temperature, and existing vegetation type. Even though the density of fire-season lightning events (1990–2009) was included in the natural model, it was the least important factor. In general, fire potential increases moving west to east at higher elevations, with the highest overall areas located in Colorado. Significant areas of overlap occur between the human and natural fire models, indicating that these areas may be at higher risk of fire occurrence in the future. It is important to note that fires may cause significant impacts to vegetation communities where they occur outside the areas of higher fire occurrence potential, because fires that do occur may be uncharacteristically severe, may occur in areas not generally adapted to fire disturbance, or may transition to invasive vegetation.

4.3.3 Current Development

Four major components of development were assessed for the ecoregion—energy, agriculture, urbanization (including roads), and recreational development. A dozen major inputs derived from multiple original datasets were compiled using a fuzzy logic model (Figure 4-35) to produce a single development footprint for the ecoregion. Reliable spatial data was available for all but the recreation input data, which proved to be very difficult to acquire. A subset of the recreation data that had been compiled and analyzed to address more specific recreation management questions was used for the composite model (see Appendix A for more details on recreation).

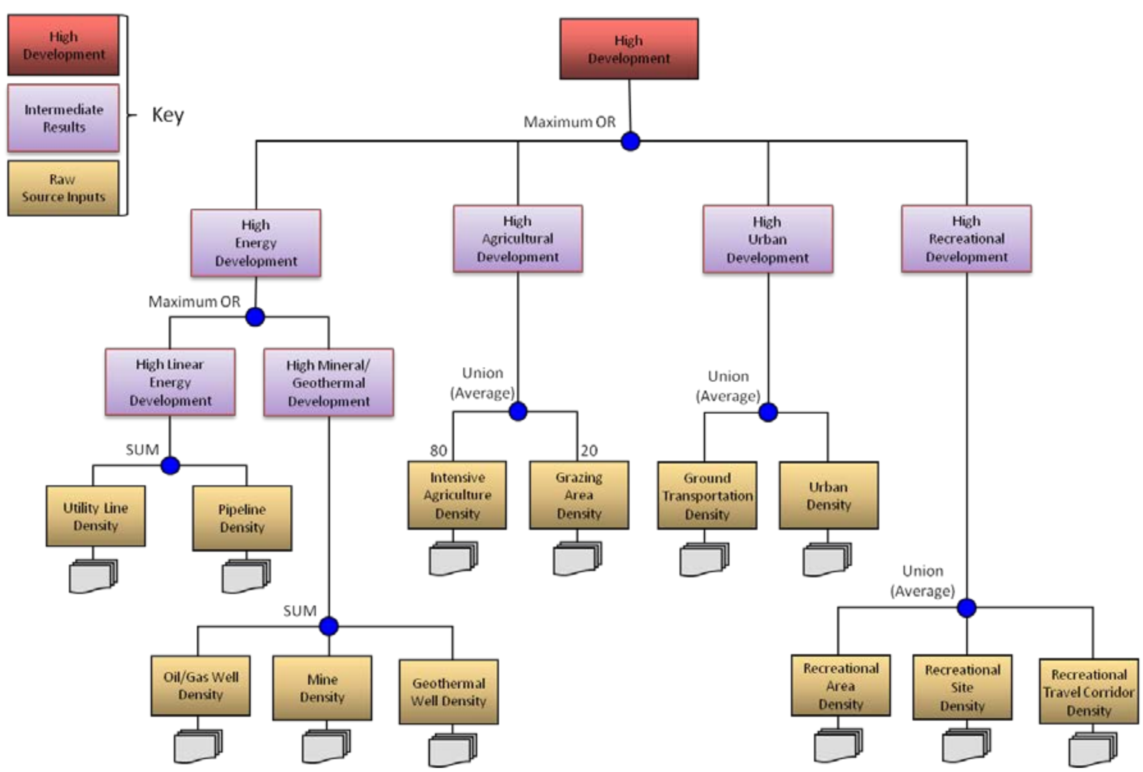


Figure 4-35. Current development fuzzy logic model for the Colorado Plateau ecoregion. Raw data inputs are in gold color, intermediate results for energy development, agriculture, urbanization, and recreation are in purple, and the final development footprint represented in red.

The recreation data used for the composite development model focused on land recreation only and included point, line, and polygon inputs (Figure 4-36D). Current energy development comprised spatial data on linear features (utility lines and pipelines) and point features (oil/gas wells, mines, and geothermal wells) and the data were aggregated using a *Maximum OR* logic operator (Figure 4-36A). The urban development component of the fuzzy logic model averaged urban landcover density and road density based on the ground transportation linear features dataset provided by BLM (Figure 4-36B). No weighting or special treatment of roads was conducted as the dataset was too inconsistently attributed (did not distinguish paved from unpaved) to allow for more detailed treatment of the road infrastructure, which ranged from OHV dirt paths to interstate highways.

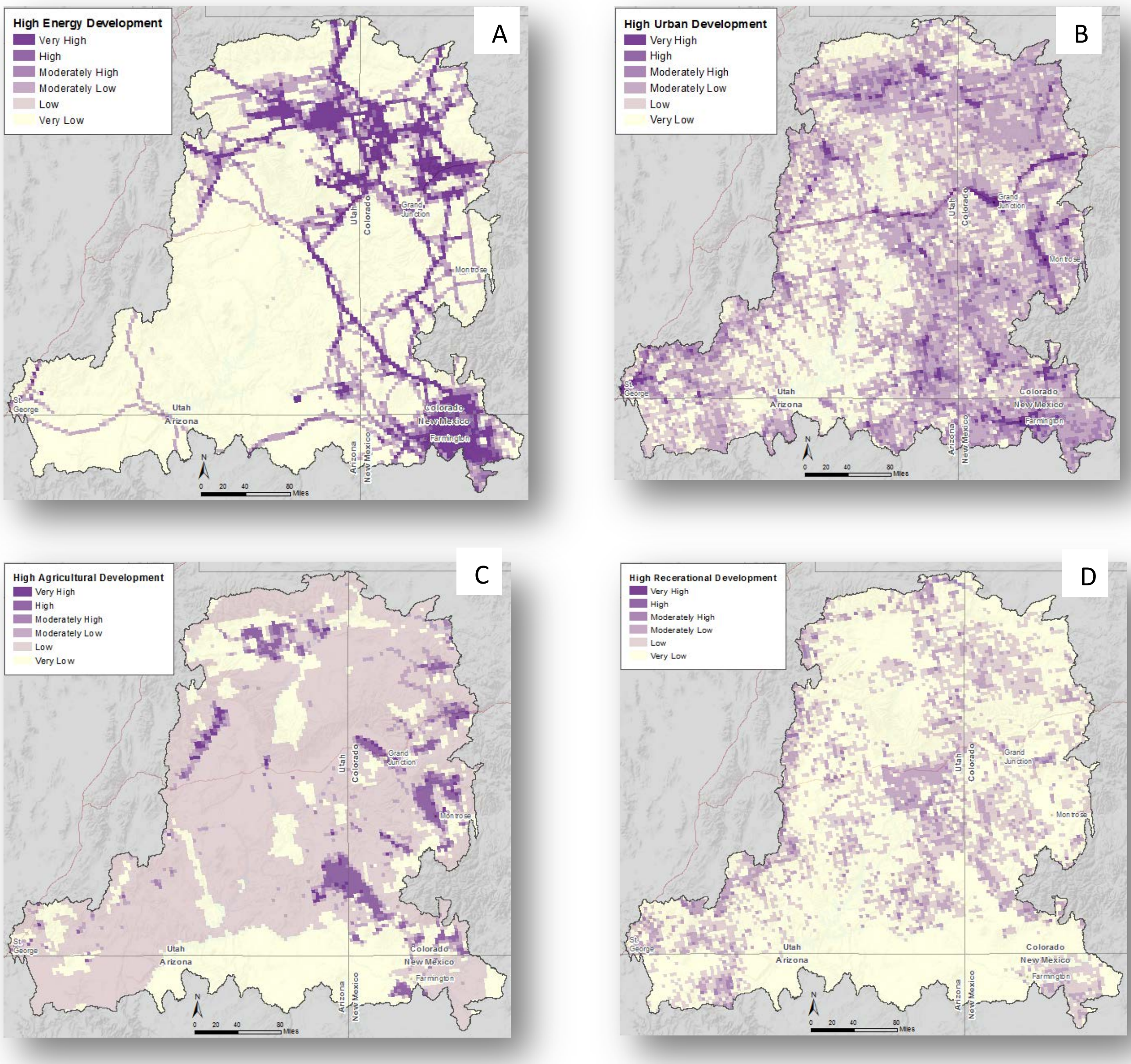


Figure 4-36. Intermediate results of the current development fuzzy logic model showing (A) current energy development, (B) urban development, (C) agriculture development, and (D) recreation development.

Agricultural development was derived from agriculture landcover data and grazing allotment data using an *Average* (or *Union*) logic operator and weighting converted agricultural land to grazing lands by 80/20 (Figure 4-36C). Agriculture contributes to the final map; however, lack of data on recent livestock density and overall range condition is a serious data gap in the model. In addition, there were no data for the large Navajo Nation in the southern portion of the ecoregion. Recreational development data is also substandard and the model would do a better job of incorporating recreation impacts with more detailed and complete data for the wide array of recreational activities (both active and passive). Filing these data gaps would enhance the development model as well as both the terrestrial and aquatic intactness models.

The full development footprint for the Colorado Plateau shows the highest development in the northern and eastern portions of the ecoregion where traditional energy development (oil and gas) and urbanization is concentrated (Figure 4-37). Future development scenarios are presented in Chapter V, Potential Future Conditions in the Colorado Plateau Ecoregion.

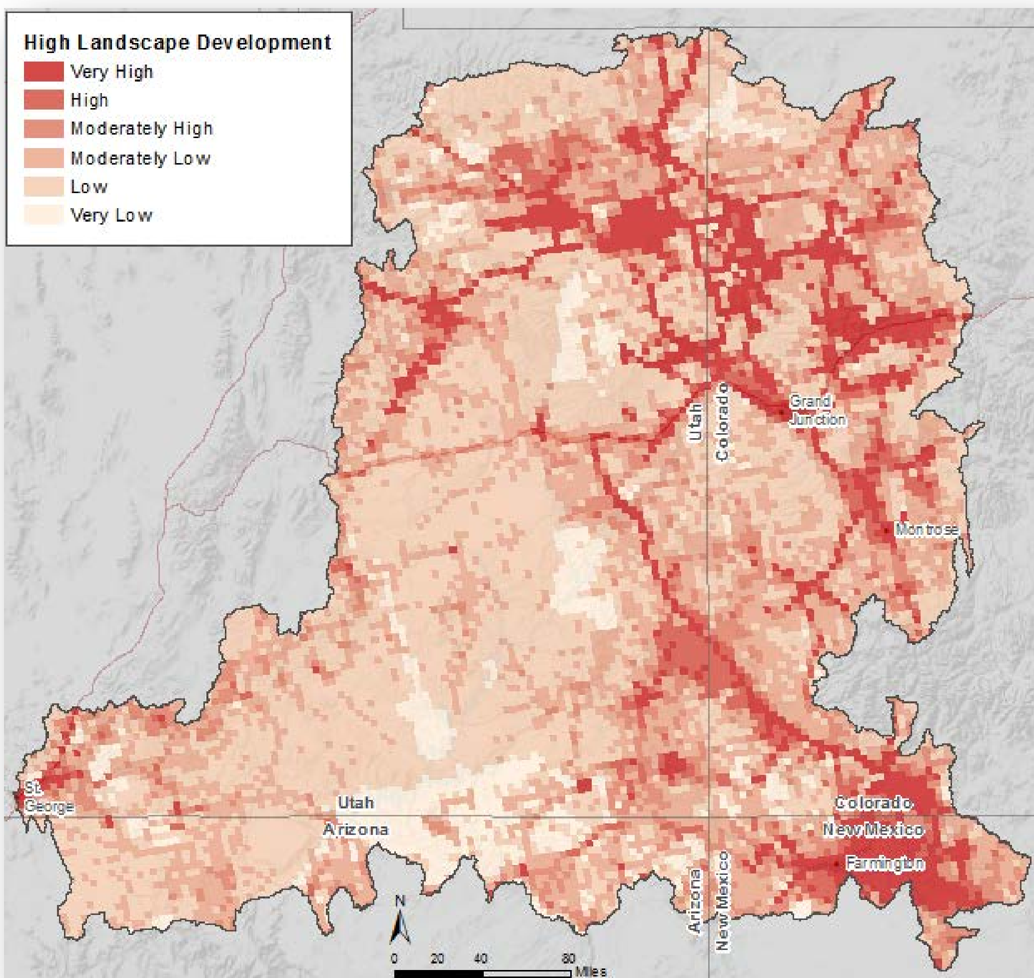


Figure 4-37. Composite map of current development in the Colorado Plateau ecoregion.

4.3.4 References Cited

Boxell, J., and P.J. Drohan. 2009. Surface soil physical and hydrological characteristics in *Bromus tectorum* L. (cheatgrass) versus *Artemisia tridentata* Nutt. (big sagebrush) habitat. *Geoderma* 149(3-4):305–311.

Bradley B.A., and J.F. Mustard. 2005. Identifying land cover variability distinct from land cover change: Cheatgrass in the Great Basin. *Remote Sensing of Environment* 94:204–213.

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

Brooks, M.L., and D.A. Pyke. 2001. Invasive plants and fire in the deserts of North America. Pages 1–14 *in* Galley, K.E.M., and T.P. Wilson (eds.). Proceedings of the invasive species workshop: The role of fire in the control and spread of invasive species. Fire Conference 2000. Miscellaneous Publication No. 11, Tall Timbers Research Station, Tallahassee, FL.

Busch, D.E., and S.D. Smith. 1995. Mechanisms associated with the decline of woody species in riparian ecosystems of the Southwestern U.S. *Ecological Monographs* 65:347–370.

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

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

Dukes, J.S., and H.A. Mooney. 2004. Disruption of ecosystem processes in North America by invasive species. *Revista Chilena de Historia Natural* 77:411–437.

Glenn, E., R. Tanner, S. Mendez, T. Kehret, D. Moore, J. Garcia, and C. Valdes. 1998. Growth rates, salt tolerance, and water use characteristics of native and invasive riparian plants from the delta of the Colorado River, Mexico. *Journal of Arid Environments* 40:281–294.

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

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

Jarnevich, C.S., P. Evangelista, T.J. Stohlgren, and J. Morisette. 2011. Improving national-scale invasion maps: Tamarisk in the western United States. *Western North American Naturalist* 71(2): 164–175.

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

Knapp, P.A. 1996. Cheatgrass (*Bromus tectorum* L) dominance in the Great Basin Desert: History, persistence, and influences to human activities. *Global Environmental Change* 6(1):37–52.

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

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

Mack, R.N. 1981. Invasion of *Bromus tectorum* L. into western North America: an ecological chronicle. *Agro-Ecosystems* 7:145–165.

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

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

Novak, S.J., and R.N. Mack. 2001. Tracing plant introduction and spread: Genetic evidence from *Bromus tectorum* (cheatgrass). *BioScience* 51(2):114–122.

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

Pellant, M., and C. Hall. 1994. Distribution of two exotic grasses on intermountain rangelands: status in 1992. Pages 109–112 *in* Monsen, S.B. and S.G. Kitchen (compilers). Proceedings: Ecology and management of annual rangelands, May 18–22, 1992, Boise, ID. General Technical Report INT-GTR-313, U.S. Forest Service, Intermountain Research Station, Ogden, Utah.

Schoennagel, T., and C.R. Nelson. 2010. Restoration relevance of recent National Fire Plan treatments in forests of the western United States. *Frontiers in Ecology and the Environment*: 9(5): 271–277.

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

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

USU (Utah State University) Cooperative Extension. 2011. Range plants of Utah: Grasses and grass-likes. http://extension.usu.edu/rangeplants/htm/grasses-and-grasslikes .

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

Young, J. 2000. *Bromus tectorum* L. Pages 76–80 *in* Bossard, C.C., J.M. Randall, and M.C. Hoshovsky (eds). Invasive plants of California's wildlands. University of California Press, Berkeley, California.

Zouhar, K. 2003. *Bromus tectorum in* Fire Effects Information System, [Online]. U.S. Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory, Fort Collins, Colorado. http://www.fs.fed.us/database/feis.

Tamarisk (*Tamarix* spp.)

This is one of two case studies that demonstrate how the data collected during the REA process can be applied to management issues of concern. Case studies delve into greater detail to cover the underlying ecological and human influences affecting the selected conservation element or change agent and to articulate the nature of regional issues and associated management questions. Case studies also demonstrate how REA data and results can be applied to land use planning and resource management. Tamarisk was selected for a case study because it represents a key invasive-species, but it also relates to discussions of river regulation, flow regime changes, groundwater, and changes in native riparian species distribution and biodiversity.

Photo: Columbia University Invasive Species Summary Project

The history of the expansion of tamarisk throughout the riparian areas of the southwestern U.S. parallels the development and allocation of water resources in arid and semi-arid ecosystems in the 20th and 21st centuries. Tamarisk (or saltcedar) is an invasive shrub that has been designated as a change agent in the Colorado Plateau REA because it affects native riparian ecosystems and aquatic sites of conservation concern. The name *tamarisk* refers to a number of related species in the genus *Tamarix* (e.g., *T. ramosissima* and *T. chinensis*) that are similar in appearance and that hybridize freely (Gaskin and Shafroth 2005). The species did not become widely distributed in the U.S. until the 1800s. It is presently found throughout nearly all western and southwestern states (Lovich 2000). In a survey of 475 gaging stations across the western U.S., Friedman et al. (2005) found tamarisk to be the third most frequently-occurring riparian woody plant in the West. Tamarisk is widely distributed across the Colorado Plateau ecoregion (Figures 1 and 2). Any depiction of its distribution derived from remotely-sensed data is likely to underestimate its actual distribution as the species is not always distinguishable when mixed with native vegetation.

Tamarisk occurs in low-lying areas such as riparian habitats, washes, and playas. It tolerates a range of soil types, but it is most commonly found in alkaline and saline soils that are seasonally saturated (Brotherson and Field 1987). Although tamarisk can spread in the absence of disturbance (DiTomaso 1998, Cooper et al. 2003, Merritt and Poff 2010), human activities enhance the establishment of this species, through the damming of free-flowing rivers (with subsequent changes to flow regimes and seasonal flooding cycles), groundwater pumping, grazing, agriculture, irrigation, and urban development (Figure 3, Conceptual Model, Development and Disturbance). All of these activities have negatively affected native riparian vegetation and resulted in the conversion of many diverse southwestern riparian zones to nonnative monocultures. Tamarisk exerts competitive pressure on native riparian vegetation through a variety of pathways: it 1) tolerates a greater depth to groundwater than native species; 2) outcompetes native species in saline conditions; 3) reduces seedling recruitment of natives through its prodigious seed production, dense cover, and underlying litter layer; and 4) increases riparian zone fire frequency (Zouhar 2003, Busch and Smith 1995, Lite and Stromberg 2005). Dense stands of tamarisk also create overbank flooding that alters stream channel structure and sediment deposition (Figure 3, Geomorphology, Flooding Regime, and Hydrology Changes, Lovich 2000, Dudley et al. 2000, Cooper et al. 2003).

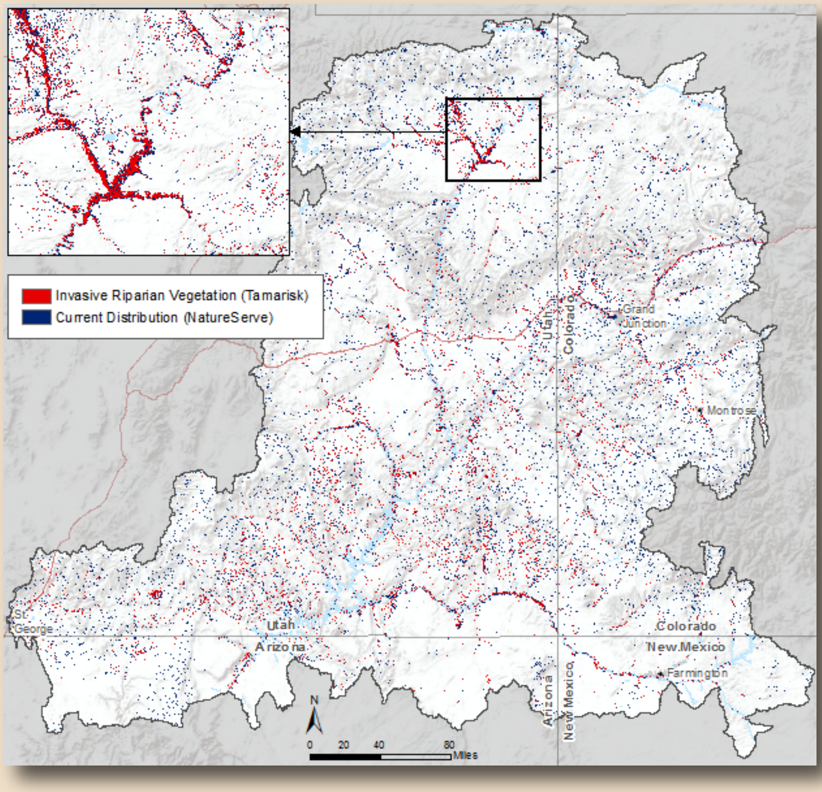


Figure 1. Distribution of tamarisk (in red) relative to the distribution of other riparian vegetation (NatureServe landcover dataset).

Figure 2. Detail of current distribution of tamarisk (in blue) near Fort Duchesne, Uinta Basin, Utah as mapped for the REA. See Appendix A for modeling approach and region-wide results.

Flow Alteration. Although it is likely that native riparian species would have declined with the extensive flow alteration of western U.S. streams and rivers regardless of the presence of invasive species (Merritt and Poff 2010), flow regulation has facilitated the spread of tamarisk. The creation of dams and reservoirs has enhanced tamarisk establishment and survival by altering the frequency, timing, and velocity of flows, reducing the frequency of seasonal flooding, and providing stable substrates for colonization (Figure 4, Shafroth et al. 2002, Zouhar 2003, Lite and Stromberg 2005, Stromberg et al. 2007b, Merritt and Poff 2010). Even slight modifications in flow regime affect cottonwood recruitment (Merritt and Poff 2010). While native riparian species produce seeds during a narrow germination period that corresponds to a former spring flooding time frame, tamarisk produces hundreds of thousands of seeds over the entire growing season; in regions with summer rainfall, tamarisk seeds may germinate late in the season following monsoonal storm events (Shafroth et al. 1998, Stromberg et al. 2007b).

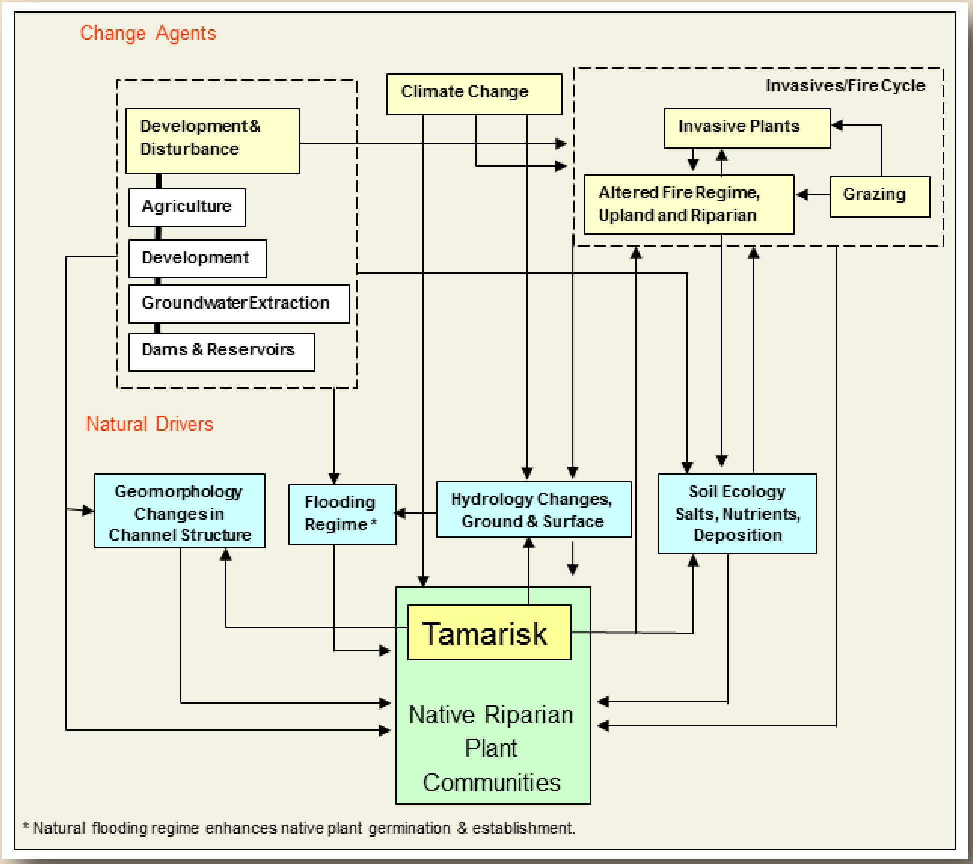


Figure 3. Conceptual model for tamarisk in the Colorado Plateau ecoregion.

Flow regulation also increases stream and riparian soil salinity in arid aquatic systems by eliminating regular flooding and subsequent flushing of accumulated salts from natural sources and irrigation return water (Busch et al. 1992, Merritt and Poff 2010). Tamarisk has greater salt tolerance than native species; it alters the breakdown of organic materials in desert streams (Kennedy and Hobbie 2004) and concentrates salt in leaf litter, inhibiting other species' germination and growth (Figure 3, Soil Ecology, Brotherson and Field 1987, Glenn et al. 1998, Busch and Smith 1995, Vandersande et al. 2001). Busch et al. (1992) compared reaches along the Bill Williams River in Arizona having intact native riparian vegetation to disturbed reaches along the Colorado River that were dominated by tamarisk; they found that, where cottonwood and willow competed successfully with tamarisk, soil salinity levels were 1–3 g/l NaCl compared to 6–8 g/l NaCl where invasive tamarisk was dominant. Glenn et al. (1998) supported these field results with a greenhouse experiment, concluding that a native cottonwood-willow association is not competitive with tamarisk above about 4 g/l NaCl.

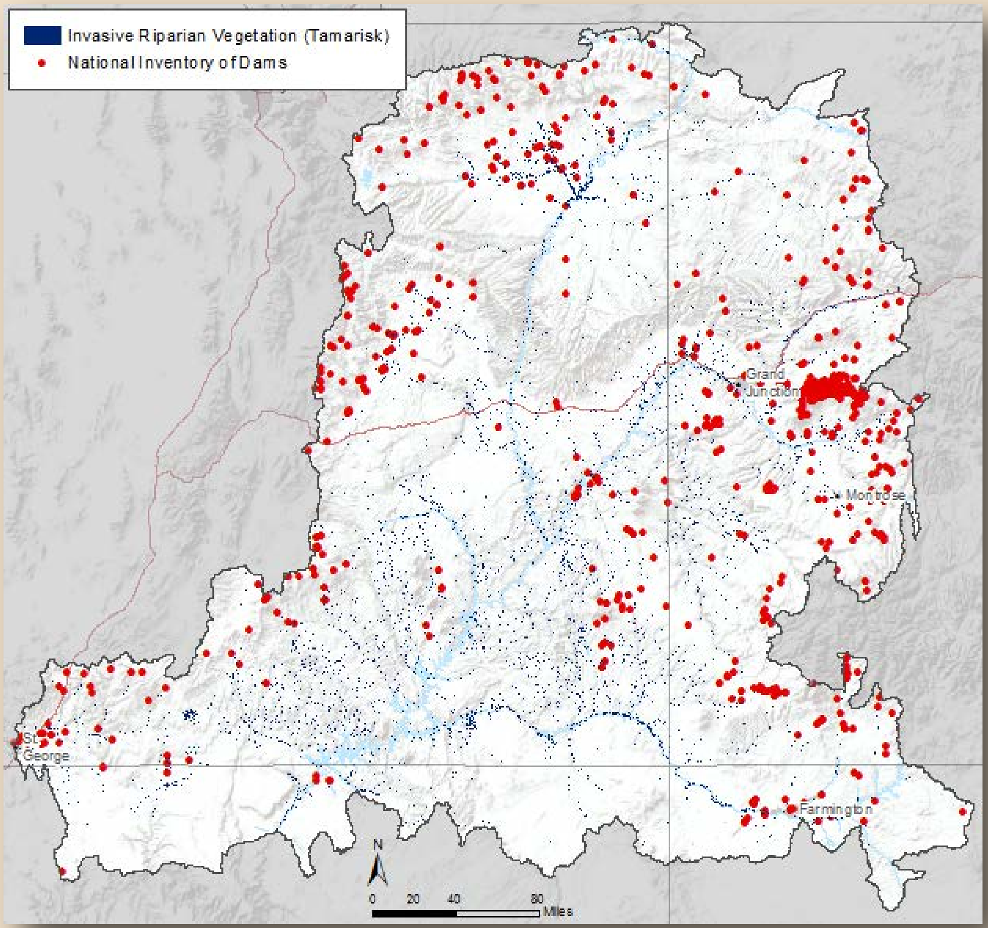


Figure 4. Distribution of tamarisk relative to the distribution of dams in the Colorado Plateau.

Thus, although natural flow conditions do not deter the recruitment of tamarisk on the Colorado Plateau, managing to imitate natural flow conditions and flooding regimes to promote native species allows natives to compete more successfully with tamarisk (Cooper et al. 2003, Birken and Cooper 2006, Merritt and Poff 2010).

Depth to Groundwater. Groundwater withdrawals for human use put native species at risk and promote the spread of invasives such as tamarisk. In semiarid and arid aquatic ecosystems, permeable floodplain substrates do not retain moisture, and shallow groundwater serves as a more reliable source of water than surface water for riparian plant communities. Depth to groundwater is a limiting factor that affects the distribution of native plant species within the riparian zone (Stromberg et al. 1996, Lite and Stromberg 2005, Nagler et al. 2009). Stromberg et al. (1996) found in a study of riparian vegetation on the San Pedro River in Arizona, that optimal groundwater levels were <0.25 m for obligate wetland herbaceous species, < 1 m for cottonwood and willow seedlings, and < 3 m for mature cottonwood. Tamarisk tolerates a wide range of groundwater depths as a seedling and adult (up to a depth of 10 m) and thus it can out-compete other more sensitive native species (Stromberg et al. 1996, Stromberg et al. 2007a). Lite and Stromberg (2005) discussed the need to 1) refine the hydrologic thresholds that indicate a shift in composition between native and exotic riparian vegetation and 2) determine the groundwater levels at which drought-tolerant species tend to assert dominance. Over a two-year study period, Lite and Stromberg (2005) found that where surface flow persisted >75% of the time, with inter-annual groundwater fluctuation < 0.5 m, and average maximum depth to groundwater < 2.6 m, native cottonwood and willow remained dominant over tamarisk. At increasing groundwater depths between 2.5 and 3.5 m and groundwater fluctuations between 0.5 and 0.8 m annually, cottonwood persisted alongside tamarisk, but willow, which requires shallower groundwater levels, declined sharply.

Fire in Riparian Zones. Fire is increasing in frequency in riparian areas of the southwestern U.S. for a number of reasons in addition to typical or climate change-induced drought cycles: increased human ignitions, a lack of flood flows, a buildup of litter and woody debris, lowered water tables, and the increasing dominance of fire-adapted invasive species (Ellis 2001). Unlike native riparian vegetation that lacks fire adaptations to resist burn damage or to repopulate burned areas, tamarisk readily re-sprouts from the roots after fire, and it is better able to utilize remaining post-fire soil moisture (Busch and Smith 1993, Busch 1995). A buildup of leaves and litter under dense growth increases fire frequency in riparian areas dominated by tamarisk; fire risk is magnified in regulated systems that lack regular flood flows to flush out accumulated litter (Figure 3, Altered Fire Regime, Busch and Smith 1993, Busch and Smith 1995, Ellis et al. 1998, Ellis 2001, Zouhar 2003).

Effects on Wildlife Habitat. Tamarisk affects native wildlife by changing the composition of forage plants and the structure of native riparian systems. Tamarisk reduces the value of critical habitat for some wildlife species dependent on specific native riparian habitats, particularly those that require mature canopy trees (Cohan et al. 1978, Hunter et al. 1988, Johnson et al. 1999, Chen 2001), but it does provide some habitat value for other species (D'Antonio 2000, Dudley et al. 2000, van Riper et al. 2008). For example, the southwestern willow flycatcher, a listed endangered species that occurs as far north as southern Utah, will use tamarisk for nesting (McCarthey 2005, Cardinal and Paxton 2005, Sogge et al. 2005, Sogge et al. 2008). However, increased fire risk in tamarisk-dominated riparian areas is also one of the greatest threats to willow flycatcher breeding sites (USFWS 2002). Brown and Trosset (1989) found that, besides willow flycatcher, five other species nested regularly in tamarisk along the Colorado River in the Grand Canyon: Bell's vireo (*Vireo bellii*), yellow warbler (*Dendroica petechia*), yellowthroat (*Geothlypis trichas*), yellow-breasted chat (*Icteria virens*), and Bullock's oriole (*Icterus bullockii*). Many songbirds, woodpeckers, and other cavity nesters are never found in tamarisk and prefer cottonwood groves in all seasons (Ellis 1995).

Tamarisk also affects instream habitats and aquatic species. 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 studies examining the response of aquatic macroinvertebrates to exotic riparian vegetation, Bailey et al. (2001) found a two-fold decrease in macroinvertebrate richness and a four-fold decrease in total abundance of macroinvertebrates on tamarisk leaf packs vs. native Fremont cottonwood leaf packs placed in an Arizona perennial stream; and Moline and Poff (2008) noted that native leaf packs remained in the stream longer than leaves from tamarisk, making the leaves available longer to macroinvertebrate leaf shredders.

Restoration of Native Riparian Species

Present riparian restoration efforts to reverse the spread of tamarisk (and other riparian exotics such as Russian olive [*Elaeagnus angustifolia*]) cover a management spectrum from the restoration or imitation of fluvial processes that favor the natural establishment of native species to mechanical and chemical tamarisk clearing operations and irrigated native tree planting. Tamarisk removal may be a lower priority or even unnecessary on perennial free-flowing rivers where fluvial processes remain more intact and native species can compete with invasives (Stromberg et al. 2007b). Stabilizing groundwater levels by limiting groundwater withdrawals (Stromberg et al. 1996) and managing to reduce salinity levels to < 4 g/l NaCl (Busch et al. 1992, Glenn et al. 1998) protect existing native riparian plant communities. In areas of tamarisk dominance, clearing and planting efforts are not likely to be successful without a concurrent restoration of accessible shallow groundwater. If tamarisk clearing is pursued, a more gradual or patch replacement of tamarisk, such as might occur with scouring floods, may ensure that enough tamarisk woodland remains available during a transitional period for bird species that use tamarisk for nesting. Bateman and Paxton (2009) provide a thorough review of wildlife use of tamarisk and likely wildlife responses to tamarisk control.

Restoration of native riparian vegetation with a return to natural fluvial processes requires active management to allow (or mimic) regional hydrologic regimes with characteristic perennial stream flows, flood timing and intensity, and available shallow groundwater. Native species recruitment may occur in sections of rivers below dams if larger flood flows exceed the storage capacity of the dam or if flood flows are managed through spring water releases (Shafroth et al. 1998). Outcomes will vary with flood timing and intensity; high volume spring flooding may scour the stream channel, rearrange sediments, and provide a seedbed for native species early in the season. Summer water releases for irrigated agriculture in reaches below dams, on the other hand, may favor tamarisk dominance because tamarisk is able to take advantage of moist summer seedbeds (Shafroth et al. 1998, Stromberg et al. 2007a, b).

Rivers that retain more of their natural flow regime as well as available groundwater reserves provide a better opportunity for recovery of native vegetation following riparian fire. Although mature cottonwood tree mortality is very high following moderate to severe riparian burns, cottonwoods do respond with stem and root sprouts and root suckering following lighter fires (Smith et al. 2009). Native cottonwood seeds may sprout after a riparian fire if managed post-fire flooding is employed during the spring cottonwood seed dispersal period (Ellis 2001, Smith et al. 2009). Finally, as a preventative measure, reducing fuel loads and litter in riparian zones through mechanical removal or through re-establishing flooding regimes could reduce the incidence of riparian fires in mature riparian canopies (Ellis 2001).

Tamarisk dominance on perennial free-flowing streams and rivers where native species should be competitive may indicate past or present heavy grazing pressure and suggest a need for a change in grazing management (Stromberg et al. 2007b). Livestock selectively forage on the shoots of native species and find tamarisk to be less desirable than native species. Hughes (2000) found on the Arizona Strip that when livestock were restricted to winter use and kept out of riparian areas in the spring and summer, native species were able to compete with tamarisk.

Tamarisk Beetle. During the late 2000s, the U.S. Department of Agriculture (USDA) allowed tamarisk control using defoliating *Tamarix* leaf beetles (*Diorhabda carinulata*) north of the 38th parallel to avoid conflict with southwest willow flycatcher nesting territories to the south. When a later release of a different subspecies of beetle near St. George, Utah threatened to allow beetle invasions southward into Arizona, a lawsuit prompted the USDA to ban the release or interstate transport of the *Diorhabda* beetle in 2010 (Center for Biological Diversity 2009, Lamberton 2011). Field studies in the Colorado Plateau to monitor yearly beetle infestations (Figure 5) and subsequent tamarisk mortality suggest that tamarisk is not weakened as much as had been hoped by beetle defoliation; shrubs re-sprout yearly and the amount of shrub mortality varies by location and post-defoliation conditions (Tamarisk Coalition 2010, Nagler et al. 2011). Cooperating land management agencies have the opportunity to respond promptly with comprehensive restoration plans should the extent of tamarisk mortality expand widely enough to create candidate areas for riparian restoration. Areas of beetle-killed tamarisk may present atypical soil and site conditions that may require different management techniques to avoid colonization by other noxious weeds (Dennison et al. 2009, Hultine et al. 2010).

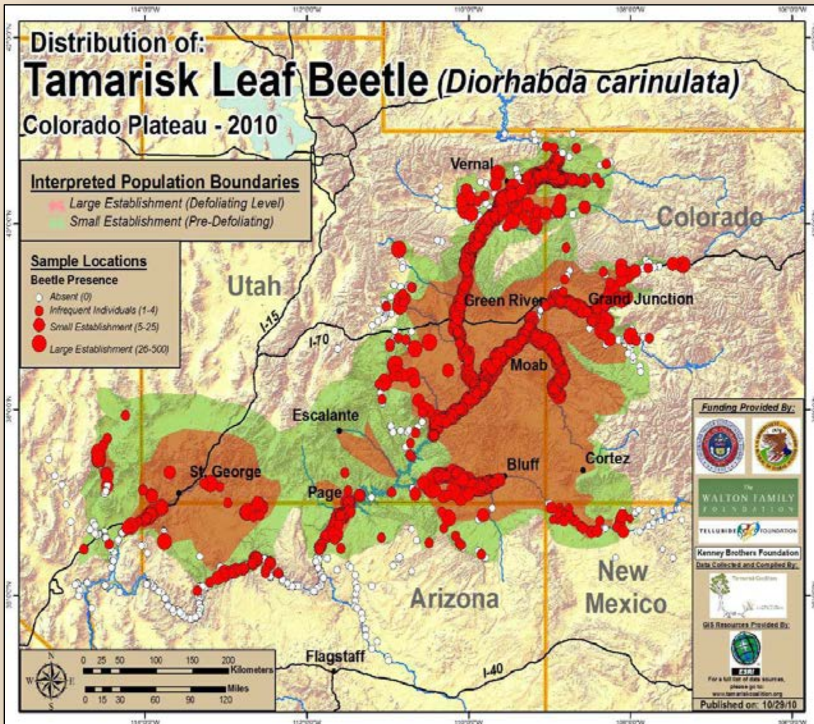


Figure 5. Map of tamarisk beetle distribution across the Colorado Plateau (Tamarisk Coalition 2010). Brown areas show areas of defoliation and green areas indicate beetle presence with low defoliation.

Climate Change

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 even more than currently (Figure 3, Altered Fire Regime, Climate Change, Merritt and Poff 2010, Seager et al. 2007). Climate change projections predict declining river flows (with maximum spring flows coming earlier in the season), more frequent droughts, and increasing human water consumption with its pressures on groundwater levels; all of these conditions will make it more difficult for native species to reproduce and compete with invasives in riparian areas (Smith et al. 2009).

References Cited

Bailey, J.K., J.A. Schweitzer, and T.G. Whitham. 2001. Saltcedar negatively affects biodiversity of aquatic macroinvertebrates. *Wetlands* 21(3):442–447.

Bateman, H.L., and E.H. Paxton. 2009. Saltcedar and Russian olive interactions with wildlife. Pages 51–63 *in* Shafroth, P.B., C.A. Brown, and D.M. Merritt (eds.), Saltcedar and Russian olive control and demonstration act science assessment, U.S. Geological Survey Scientific Investigations Report 2009–5247, U.S. Geological Survey, Washington, D.C. 143 p.

Birken, A.S., and D.J. Cooper. 2006. Processes of *Tamarix* invasion and floodplain development along the lower Green River, Utah. *Ecological Applications* 16:1103–1120.

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

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

Brown, B.T., and M.W. Trosset. 1989. Nesting-habitat relationships of riparian birds along the Colorado River in Grand Canyon, Arizona. *The Southwestern Naturalist* 34(2):260–270.

Busch, D.E. 1995. Effects of fire on southwestern riparian plant community structure. *The Southwestern Naturalist* 40(3):259–267.

Busch, D.E., N. Ingraham, and S. Smith. 1992. Water uptake in woody riparian phreatophytes of the southwestern United State: A stable isotope study. *Ecological Applications* 2:450–459.

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

Busch, D.E., and S.D. Smith. 1995. Mechanisms associated with the decline of woody species in riparian ecosystems of the Southwestern U.S. *Ecological Monographs* 65:347–370.

Cardinal, S.N., and E.H. Paxton. 2005. Home range, movement, and habitat use of the southwestern willow flycatcher, Roosevelt Lake, AZ, 2004. U.S. Geological Survey report to the U.S. Bureau of Reclamation, Phoenix. 26 pp.

Center for Biological Diversity. 2009. Press Release: Agriculture Department forced to re-examine tamarisk leaf-eating beetle program that hurts endangered songbird. http://www.biologicaldiversity.org/news/press_releases/2009/southwestern-willow-flycatcher-06-17-2009.html. Accessed 12/11.

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

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

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

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

Dennison, P.E., P.L. Nagler, K.R. Hultine, E.P. Glenn, and J.R. Ehleringer. 2009. Remote monitoring of tamarisk defoliation and evapotranspiration following saltcedar leaf beetle attack. *Remote Sensing of Environment* 113:1462–1472.

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

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

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

Ellis, L.M. 2001. Short term responses of woody plants to fire in a Rio Grande riparian forest, central New Mexico, U.S.A. *Biological Conservation* 97:159–170.

Ellis, L.M., C.S. Crawford, and M.C. Molles. 1998. Comparison of litter dynamics in native and exotic riparian vegetation along the middle Rio Grande of central New Mexico, U.S.A. *Journal of Arid Environments* 38:283–296.

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

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

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

Glenn, E., R. Tanner, S. Mendez, T. Kehret, D. Moore, J. Garcia, and C. Valdes. 1998. Growth rates, salt tolerance, and water use characteristics of native and invasive riparian plants from the delta of the Colorado River, Mexico. *Journal of Arid Environments* 40:281–294.

Hughes, L.E. 2000. Tamarisk: Maybe not invincible. *Rangelands* 21(5):11–14.

Hultine, K.R., J. Belnap, C. van Riper III, J.R. Ehleringer, P.E. Dennison, M.E. Lee, P.L. Nagler, K.A. Snyder, S.M. Uselman, and J.B. West. 2010. Tamarisk biocontrol in the western United States: Ecological and societal implications. *Frontiers in Ecology and the Environment* 8(9): 467–474.

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

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

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

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

Lamberton, M.L. 2011. The thirsty tree: Confronting invasive salt cedar in the American Southwest. Terrain.org 27: http://www.terrain.org/articles/27/lamberton.htm. Accessed 12/11.

Lite, S.J., and J.C. Stromberg. 2005. Surface water and ground-water thresholds for maintaining *Populus-Salix* forests, San Pedro River, Arizona. *Biological Conservation* 125:153–167.

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

McCarthey, T. 2005. Southwest willow flycatcher. Pages 302–303 *in* Arizona breeding bird atlas, Corman, T.E., and C. Wise-Gervais (eds.), University of New Mexico Press, Albuquerque, New Mexico. 636 p.

Merrit, D.M., and N.L. Poff. 2010. Shifting dominance of riparian *Populus* and *Tamarix* along gradients of flow alteration in western North American rivers. *Ecological Applications* 20(1):135–152.

Moline, A.B., and N.L. Poff. 2008. Growth of an invertebrate shredder on native (*Populus*) and non- native (*Tamarix, Elaeagnus*) leaf litter. *Freshwater Biology* 53:1012–1020.

Nagler, P.L., T. Brown, K.R. Hultine, C. van Riper III, D.W. Bean, P.E. Dennison, R.S. Murray, and E.P. Glenn. 2011. Regional-scale impacts of *Tamarix* leaf beetles (*Diorhabda carinulata*) on leaf phenology and water use of *Tamarix* spp. on western U.S. rivers. *Remote Sensing of Environment* 118:227–240.

Nagler, P.L., E.P. Glenn, C.S. Jarnevich, and P.B. Shafroth. 2009. Distribution and abundance of saltcedar and Russian olive in the western United States. Pages 7–32 *in* Shafroth, P.B., C.A. Brown, and D.M. Merritt (eds.), Saltcedar and Russian olive control and demonstration act science assessment, U.S. Geological Survey Scientific Investigations Report 2009–5247, U.S. Geological Survey, Washington, D.C. 143 p.

Seager, R., M. Ting, I. Held, Y. Kushmir, J. Lu, G. Vecchi, H. Huang, N. Harnik, A. Leetmaa, N. Lau, C. Li, J. Velez, and N. Naik. 2007. Model projections of an imminent transition to a more arid climate in southwestern North America. *Science* 316:1181–1184.

Shafroth, P.B., G.T. Auble, J.C. Stromberg, and D.T. Patten. 1998. Establishment of woody riparian vegetation in relation to annual patterns of streamflow, Bill Williams River, Arizona. *Wetlands* 18(4):577–590.

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

Smith, D.M., D.M. Finch, C. Gunning, R. Jemison, and J.F. Kelly. 2009. Post-wildfire recovery of riparian vegetation during a period of water scarcity in the southwestern USA. *Fire Ecology* 5(1):38–55.

Sogge, M.K., E.H. Paxton, and A. Tudor. 2005. Saltcedar and southwestern willow flycatchers: Lessons from long-term studies in central Arizona *in* Aguirre-Bravo, C., et. al. (eds.), Monitoring Science and Technology Symposium: Unifying knowledge for sustainability in the Western Hemisphere; September 20–24, 2004, Denver, Colorado. Proceedings RMRS-P-37-CD, U.S. Forest Service, Rocky Mountain Research Station, Ogden, Utah.

Sogge, M. K., S. J. Sferra, and E. H. Paxton. 2008. Tamarix as habitat for birds: Implications for riparian restoration in the southwestern United States. *Restoration Ecology* 16:146–154.

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

Stromberg, J.C., S.J. Lite, R. Marler, C. Paradzick, P.B. Shafroth, D. Shorrock, J.M. White, and M.S. White. 2007b. Altered stream-flow regimes and invasive plant species: The *Tamarix* case. *Global Ecology and Biogeography* 16:381–393.

Stromberg, J.C., R. Tiller, and B. Richter. 1996. Effects of groundwater decline on riparian vegetation of semiarid regions: The San Pedro, Arizona. *Ecological Applications* 6:113–131.

Tamarisk Coalition. 2010. 2010 Tamarisk leaf beetle monitoring report. Tamarisk Coalition, Grand Junction, Colorado. http://www.tamariskcoalition.org/

USFWS (U.S. Fish and Wildlife Service). 2002. Southwestern willow flycatcher recovery plan. U.S. Fish and Wildlife Service, Albuquerque, New Mexico.

Vandersande, M.W., E.P. Glenn, and J.L. Walworth. 2001. Tolerance of five riparian plants from the lower Colorado River to salinity, drought, and inundation. *Journal of Arid Environments* 49:147–159.

van Riper, C. III, K.L. Paxton, C. O'Brien, P.B. Shafroth, and L J. McGrath. 2008. Rethinking avian response to *Tamarix* on the lower Colorado River: A threshold hypothesis. *Restoration Ecology* 16:155–167.

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

Photo: Desolation Canyon riparian vegetation. BLM

V. Potential Future Conditions of the Colorado Plateau

Predicting potential future conditions for the Colorado Plateau was approached in several ways that relied heavily on fuzzy logic modeling for near-term development, potential energy development, near-term terrestrial landscape and aquatic intactness, and potential mid-century (2060) climate change impacts. Results for each analysis were then assessed against the distribution of each of the conservation elements to illustrate their potential future status. Lack of source data for future projections was a common issue resulting in underestimates of what is likely to occur, including in the near-term 2025 time frame. Maximum potential (or long term) energy development has an indeterminate time frame; it is based on broad scale polygons representing energy zones rather than specific leases or applications. For this reason, maximum potential energy development, as discussed in Section 5.2, when overlaid on conservation elements' distributions may overestimate the impacts to species, habitats, and sites. The potential energy development analysis considered all potential known traditional (Copeland et al. 2009) and renewable energy projections. Projecting into the future is a challenging endeavor and the results should be viewed critically as they possess many uncertainties; they should not be relied upon for detailed site-level planning and management without additional information and analysis. The results do provide one view of what to expect for the ecoregion in the coming years and how the predicted changes are likely to affect the various conservation elements of interest.

5.1 Projected Near-term Future (2025) Development

Several management questions (MQ C2, D6, G1, and G2) required an assessment of the future effects of change agents, especially development, on conservation elements. The year 2025 was selected for this analysis. Projected near-term (2025) development was built from the current development fuzzy logic model, which is comprised of four major development components—energy development, agricultural development, urban and road development, and recreational development (Figure 5-1). In reality, all of these factors are likely to change, but there was a lack of predictive data available to use that provided meaningful projections into the future. Although there were no data for near-term expansion of linear utilities, data did exist for projected near-term oil and gas development (Copeland et al. 2009). The only data provided for projected renewable energy development was a small area of potential wind development in the southwest corner slightly outside the ecoregion boundary. There were no datasets for projected future for either intensive agriculture or grazing. Given climate change results, will agriculture begin to decline in the region? If so, then where? Current recreation data was difficult to acquire and assemble so any future projections based on it would likely be poor as well; as a result, there was no change made in recreation for the near-term. Lastly, future projections for urban development were based on model results from Theobald (2010), but there were no accompanying data on projected road building, which is a noteworthy deficiency as road development precedes urbanization and road impacts on intactness and many wildlife species are well-known. Even with the lack of important topical data, some measurable changes were observed (Table 5-1). The Very High development class increased by 2% and both High and Moderately High classes gained approximately .5%; in all, the development footprint increased by about 1.5 million acres. All of the results from the development model were incorporated into the near-term intactness models except the projected oil and gas development footprint, because the near-term intactness models had already been completed when a new oil and gas dataset arrived in January of 2012 (Copeland et al. 2009). However, the projected oil and gas development footprint was applied to sage grouse because of its importance to that species' future (see sage grouse insert). The potential impact on conservation elements from near-term future development was examined by applying the near-term intactness modeling described in Section 5.3.

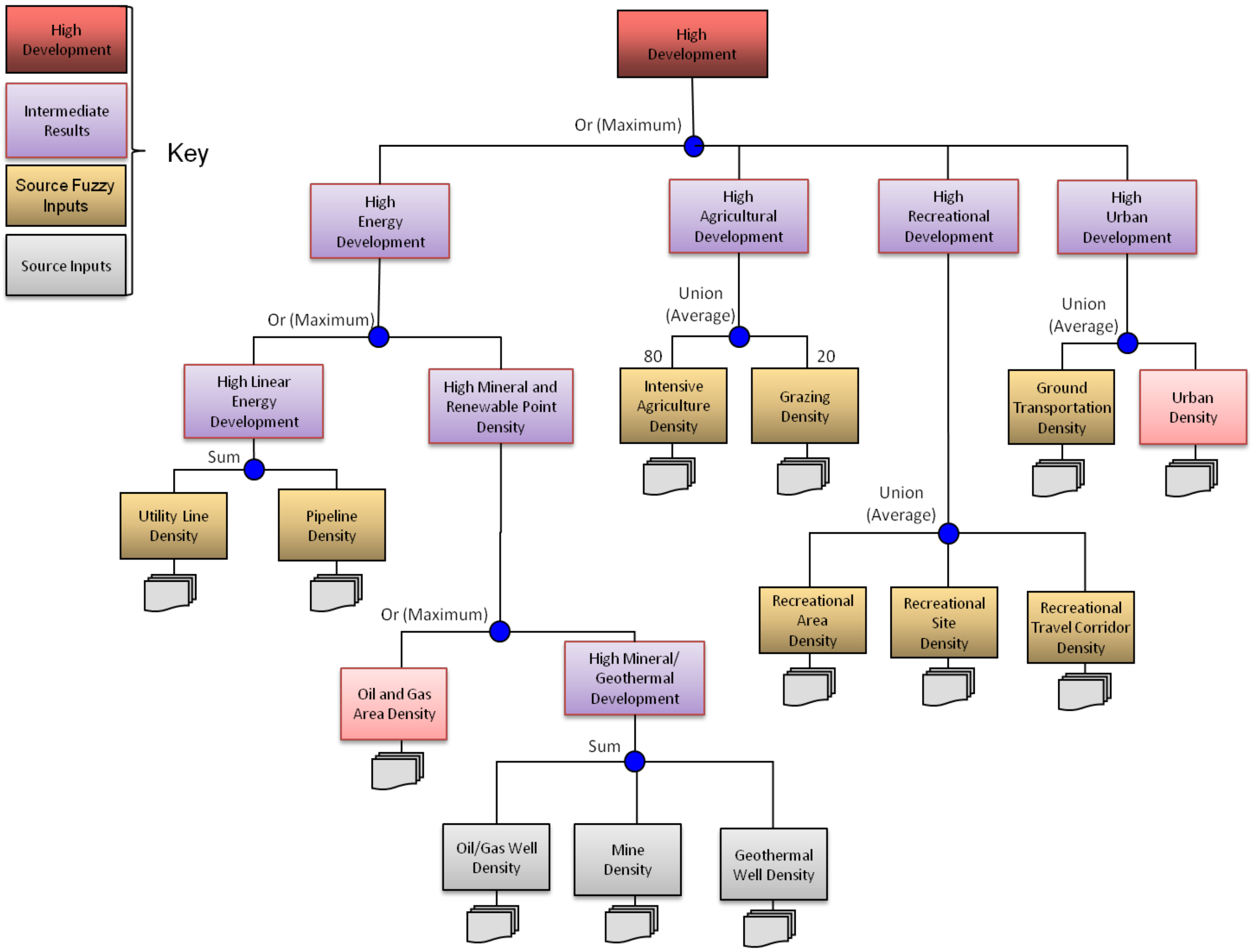


Figure 5-1. Fuzzy logic model for future near-term (2025) development for the Colorado Plateau ecoregion. Projection data existed only for projected near-term oil and gas development and urban expansion (Theobald 2010, pink boxes).

Table 5-1. Modeled change in land area (in 1000s of acres) from increased development 2011–2025. The Very High development class increased by 2% and both High and Moderately High classes gained approximately .5%; in all, the development footprint increased by about 1.5 million acres.

Category	Current	Percent	Near-term	Percent	Change
Very High	4,779	10.7%	5,698	12.7%	+2.1%
High	2,270	5.1%	2,553	5.7%	+0.6%
Moderately High	4,740	10.6%	5,004	11.2%	+0.6%
Moderately Low	13,200	29.5%	12,313	27.5%	-2.0%
Low	16,706	37.3%	16,184	36.1%	-1.2%
Very Low	3,109	6.9%	3,052	6.8%	-0.1%

5.2 Potential Energy Development

Maximum potential (long-term) energy development was also handled with a fuzzy logic model that included three major components—traditional oil and gas, wind energy, and solar energy (Figure 5-2). Potential oil and gas development data included numerous sources—Intermountain West oil and gas potential (Copeland et al. 2009), BLM oil and gas leases, allowable leasing footprints for oil shale and tar sand extraction (2008 data from BLM Oil Shale and Tar Sands Preliminary Environmental Impact Statement), and Department of Energy producing oil and gas fields, mapped by buffering existing active wells by 1.4 km (Figure 5-3). Two data sources comprised potential wind development—Utah BLM wind energy priority Areas and National Renewable Energy Laboratory (NREL) wind power density classes 3 and above at 50 m high (Figure 5-4). Solar resource potential (>5.5 kW/m^2) was obtained from NREL as well (Figure 5-5).

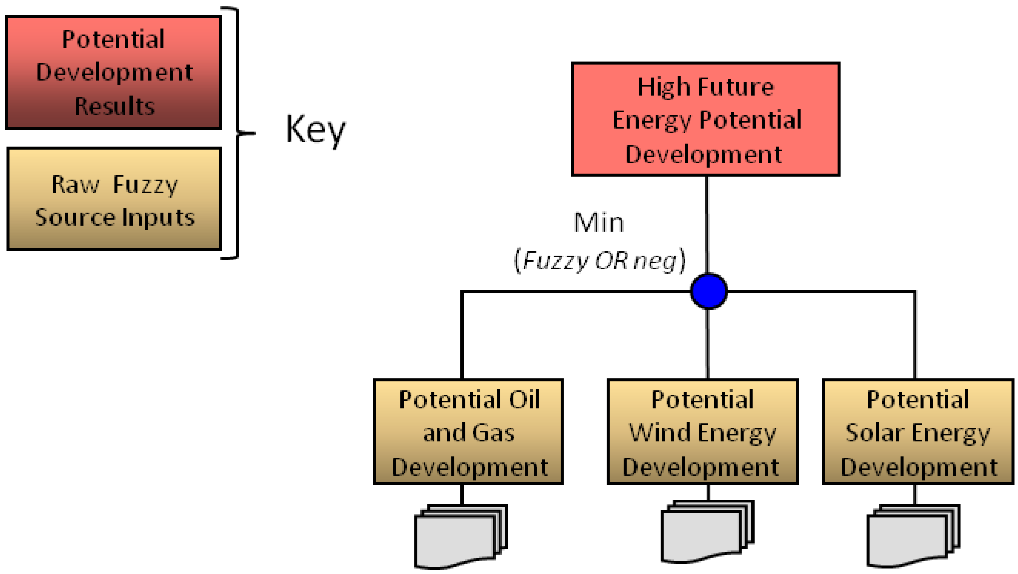


Figure 5-2. Fuzzy logic model diagram for potential energy development in the Colorado Plateau ecoregion.

Summarized at 4km resolution, the final composite map for all three energy components covered a large area of the ecoregion, particularly in the northern and eastern portions (Figure 5-6). Values from the fuzzy logic model were divided into three basic classes (High: 1 to 0.33, Moderate: 0.33 to -0.33, and Low: -0.33 to -1) instead of the six classes that have been used in other fuzzy logic models (such as the intactness models and the model for near-term [2025] development); finer differentiation was not depicted or warranted because the subject data covered broad areas and were more speculative (that is, not based on actual plans for development). For the ecoregion, about 12,500,000 acres (28%) were classified as having High potential, about 5,000,000 acres (12%) Moderate potential, and the rest, over 27,000,000 acres (60%) Low potential. These results were overlaid with the distribution maps for all of the conservation elements to evaluate the potential impact. Designated lands were not included in this part of the analysis because most energy development is prohibited from these areas.

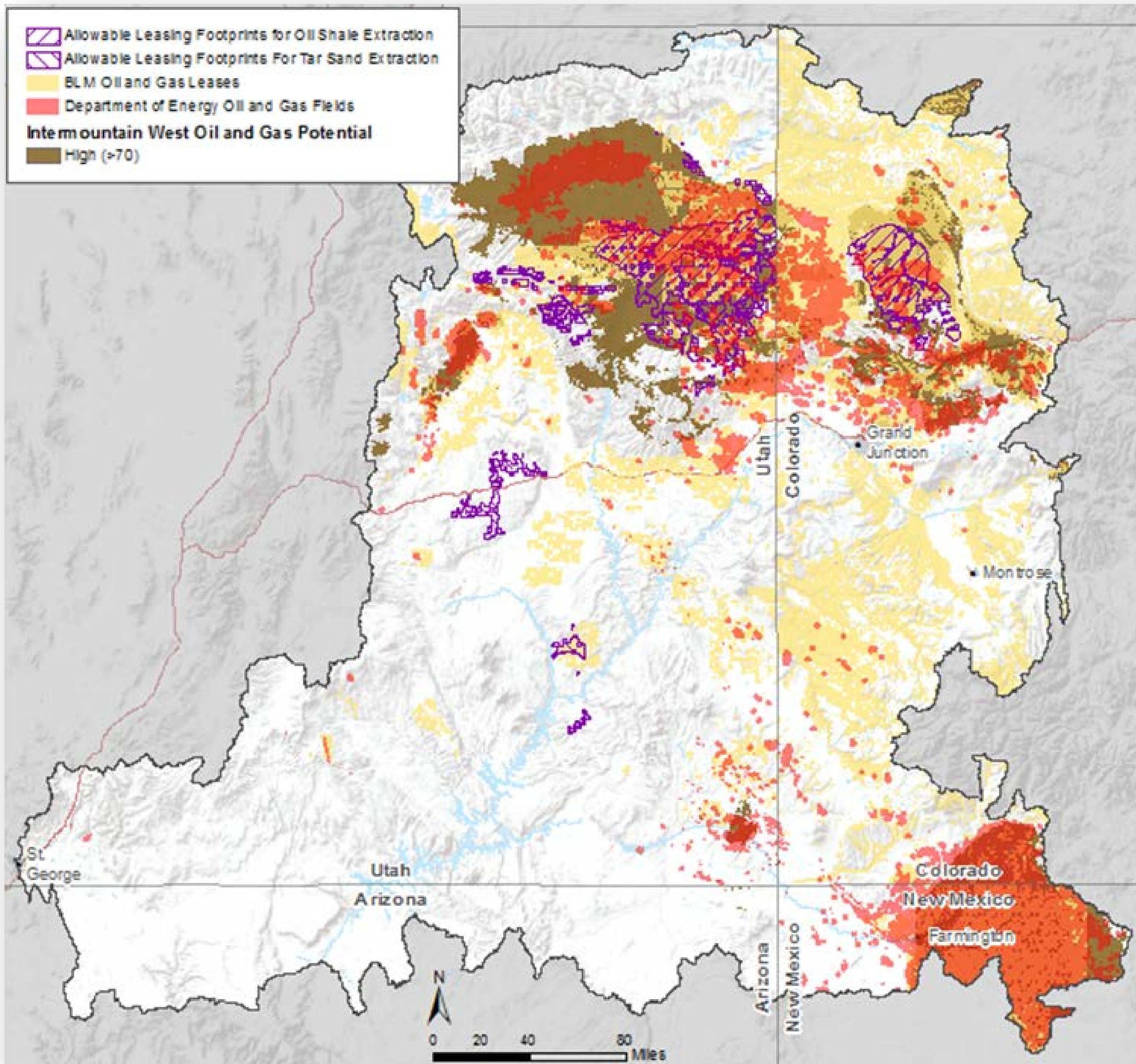


Figure 5-3. Map showing data sources for potential oil and gas development including Intermountain West oil and gas potential (Copeland et al. 2009, brown areas), BLM oil and gas leases (yellow), allowable leasing footprints for oil shale and tar sand extraction (2008 data from BLM Oil Shale and Tar Sands Preliminary Environmental Impact Statement, purple hatched areas), and Department of Energy producing oil and gas fields (orange areas), mapped by buffering existing active wells by 1.4 km.

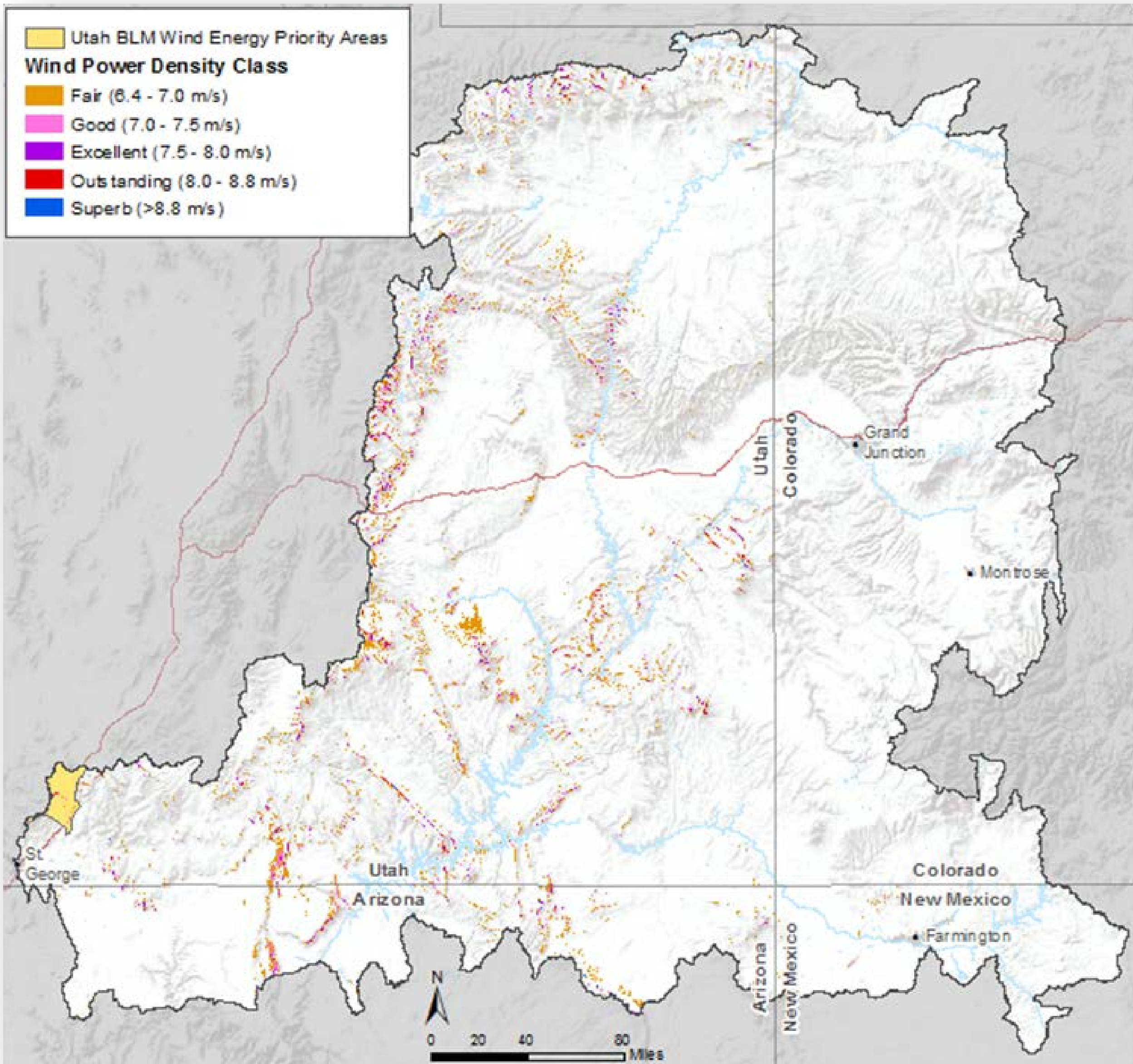


Figure 5-4. Map showing data sources for potential wind development comprised of Utah BLM wind energy priority Areas and National Renewable Energy Laboratory (NREL) wind power density classes 3 and above at 50 m high.

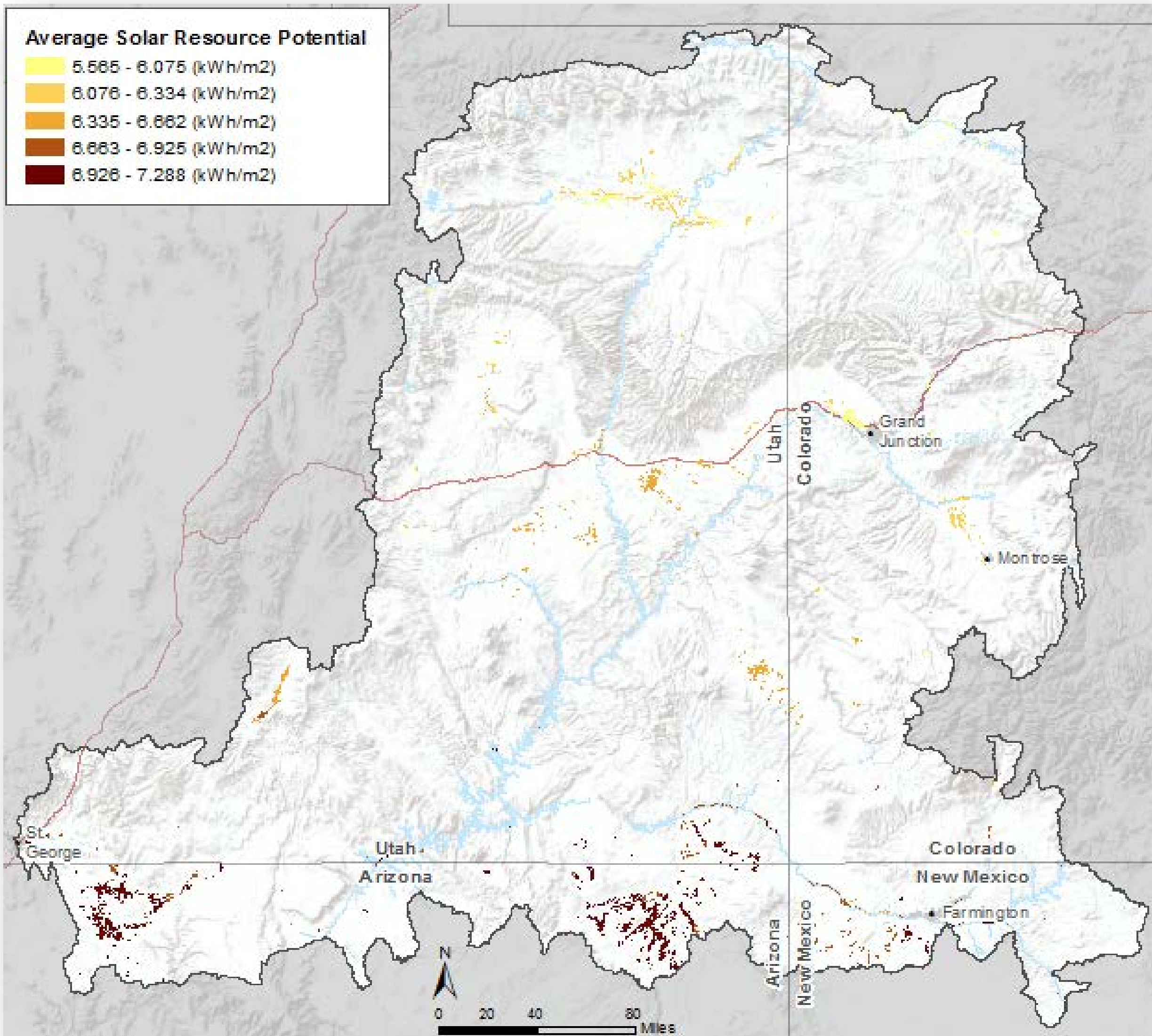


Figure 5-5. Map of solar resource potential (>5.5 kW/m^2) obtained from the National Renewable Energy Laboratory (NREL). The highest solar resource potential is in the southern portion of the ecoregion.

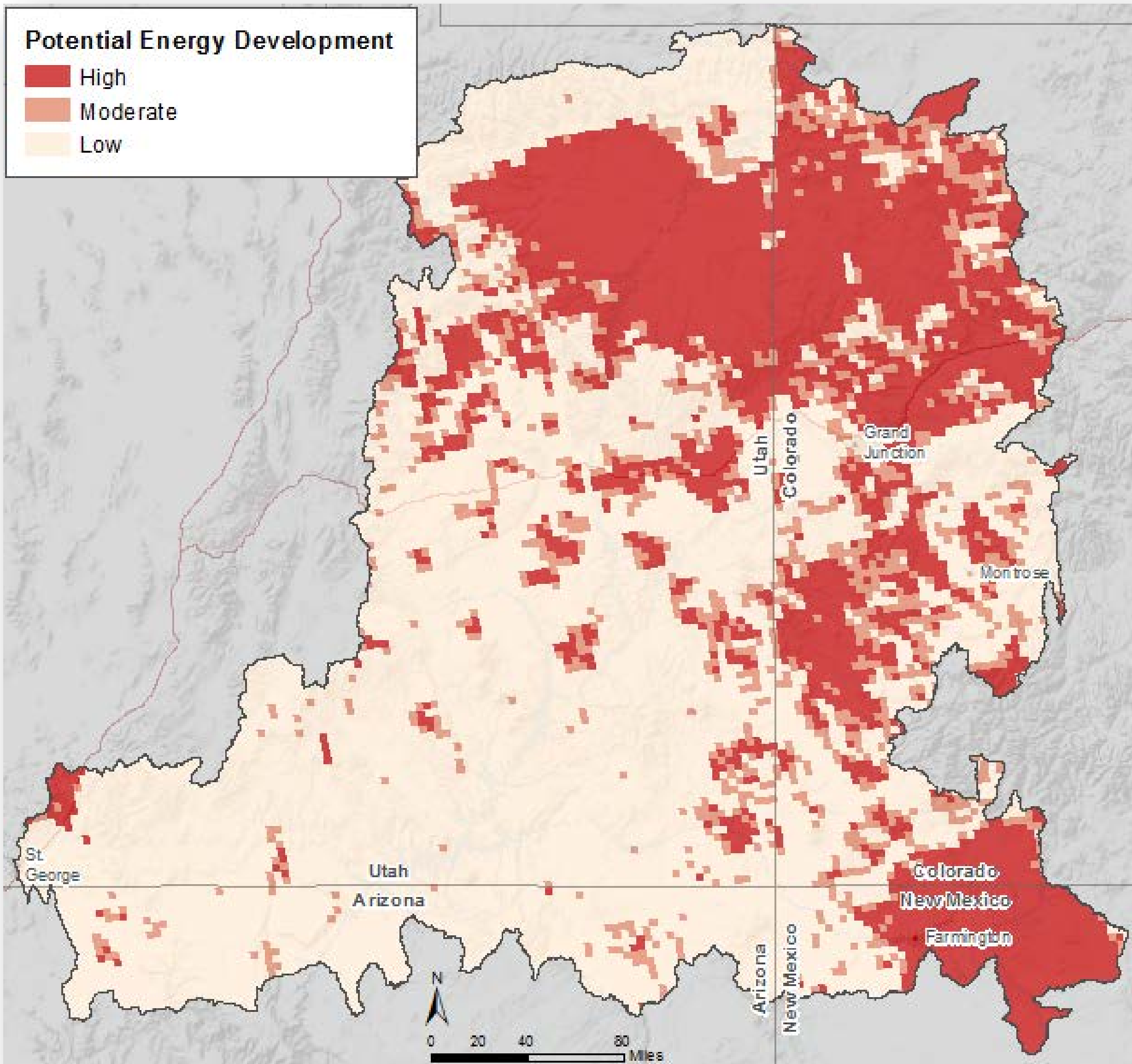


Figure 5-6. Map of potential energy development for all three energy components (oil and gas, wind, and solar energy) in the Colorado Plateau ecoregion. Values from the fuzzy logic model were divided into three basic classes (High 1 to 0.33, Moderate 0.33 to -0.33, and Low -0.33 to -1).

5.2.1 Impact of Potential Energy Development on Wildlife Species

Potential impact on species conservation elements from potential (or long term) energy development varied greatly. Of the seven mammal species examined (Figure 5-7), all of the animals with habitat concentrated in the northern portions of the ecoregion showed large potential impacts. Nearly 90% of the extremely limited current black-footed ferret distribution fell within the High risk class. The white-tailed prairie dog, the main prey item for ferrets, had 60% of its current distribution within the High risk class. Pronghorn antelope showed nearly 50% of its current distribution in the High risk class. All of the other species had approximately 30% of their habitat within the High risk category.

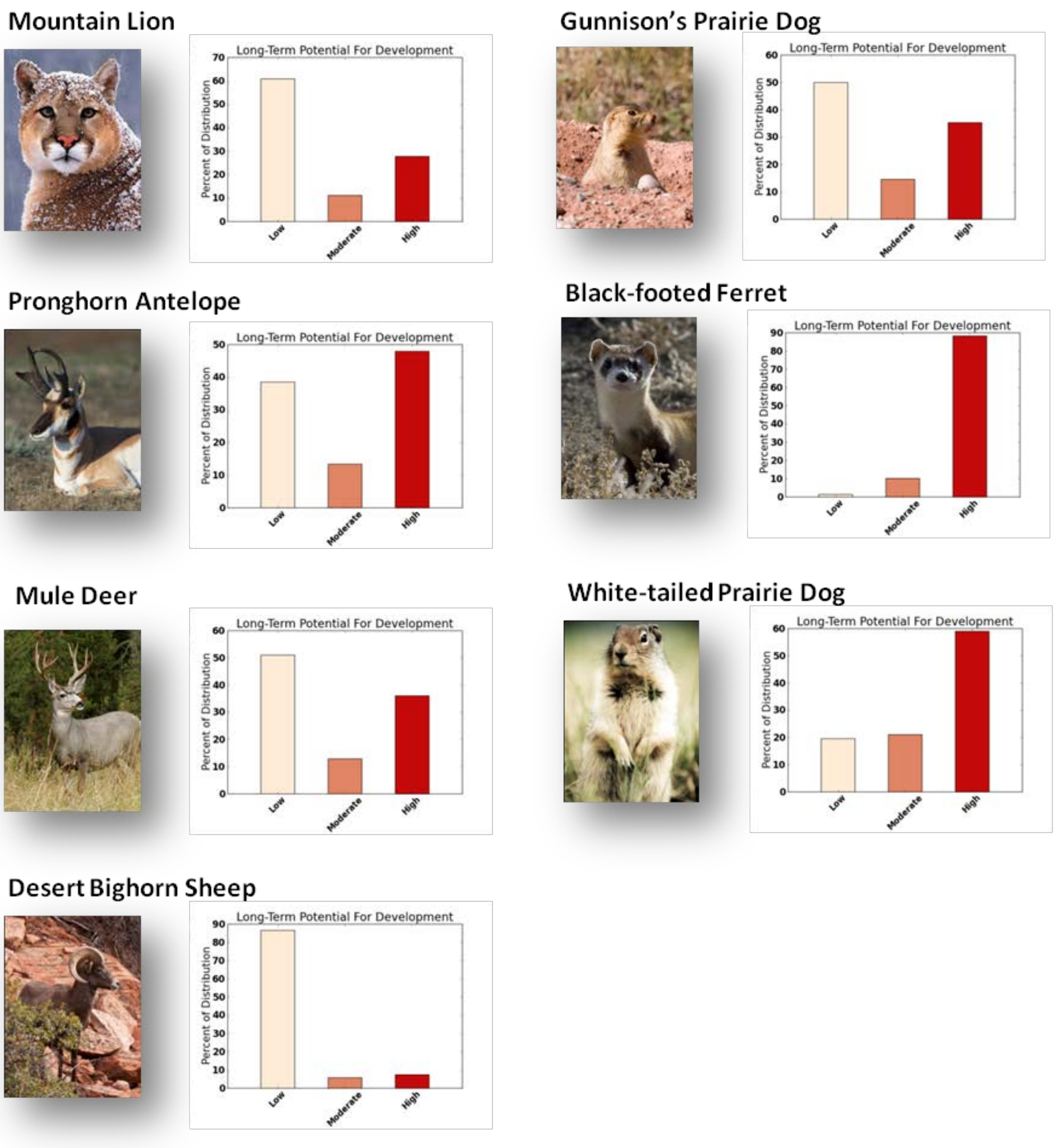


Figure 5-7. Potential impact from energy development for the mammal conservation elements of the Colorado Plateau ecoregion.

Birds showed a variable pattern as well (Figure 5-8). Not surprisingly, greater sage grouse showed the highest risk from potential energy development with nearly 50% of its existing distribution in the High category with another 15% in the Moderate category. Yellow-breasted chat showed high risk as well with 40% High and 18% Moderate, respectively. All of the birds of prey showed similar results with around 30% of their distributions in the High potential class except Mexican spotted owl, which had very little overlap in distribution with energy development. Although Gunnison sage-grouse appears to be less threatened by energy development than greater sage-grouse with about 40% in High and Moderate risk categories, just 1/4 of its distribution (about 100,000 acres) occurs on BLM land subject to energy development. The species has a much more limited distribution than greater sage-grouse, and it is subject to other impacts (rowcrop agriculture, grazing, and some urbanization) on the 2/3 of its distribution that is on private land.

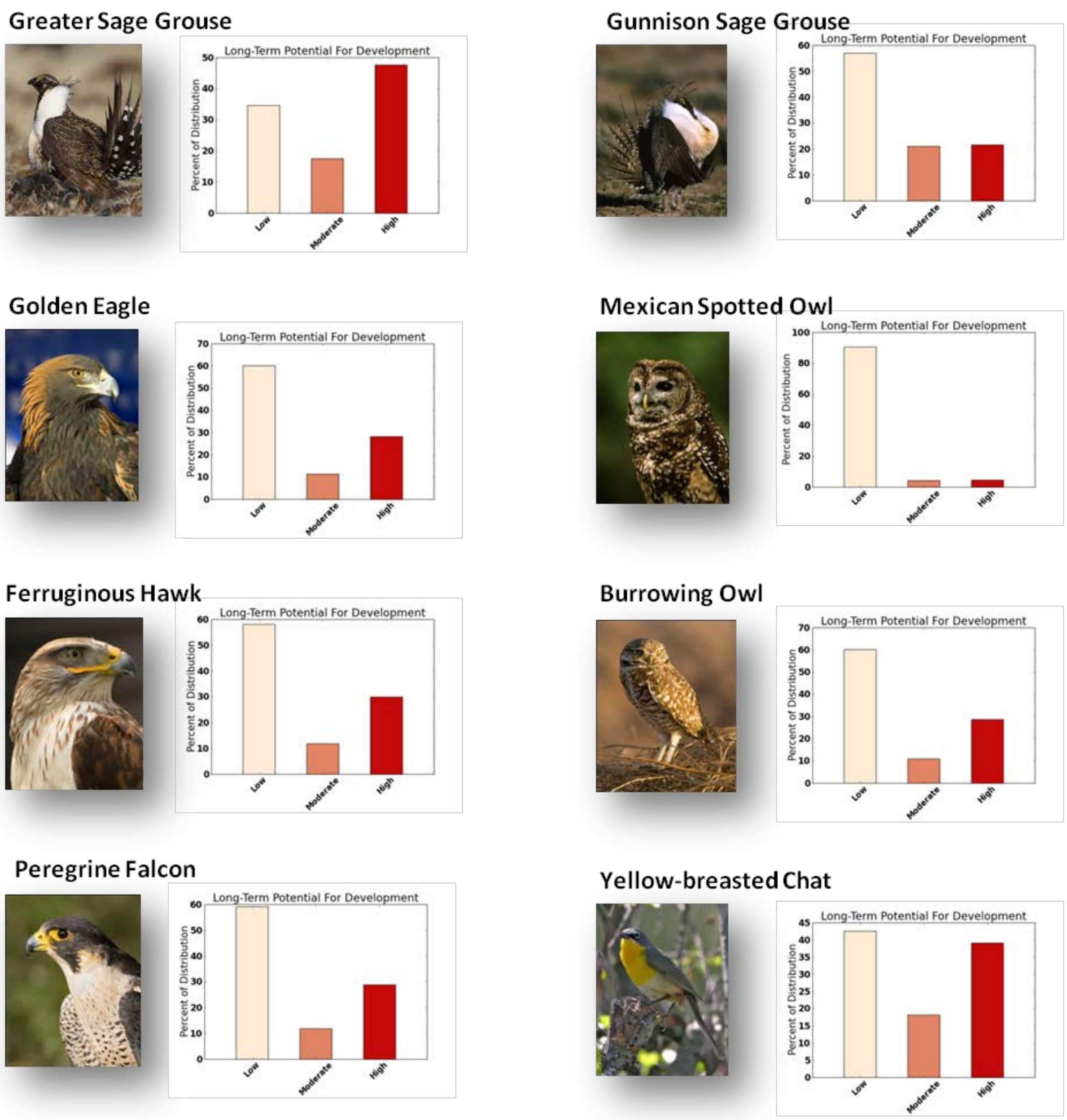


Figure 5-8. Potential impact from energy development for the birds of the Colorado Plateau ecoregion.

5.2.2 Potential Energy Development Impact on Vegetation Communities

Of the nine vegetation communities examined, intermountain basins big sagebrush shrubland and pinyon-juniper woodland showed the highest potential for change. Inter-mountain Basins Big Sagebrush Shrubland and Colorado Plateau Pinyon-Juniper Woodland showed over 40% of their current distributions within the High risk class (Figure 5-9). Colorado Plateau Blackbrush-Mormon-tea Shrubland and Mixed Bedrock Canyon and Tableland were least affected by energy development. All other communities had 20-30% of their current distributions at High risk.

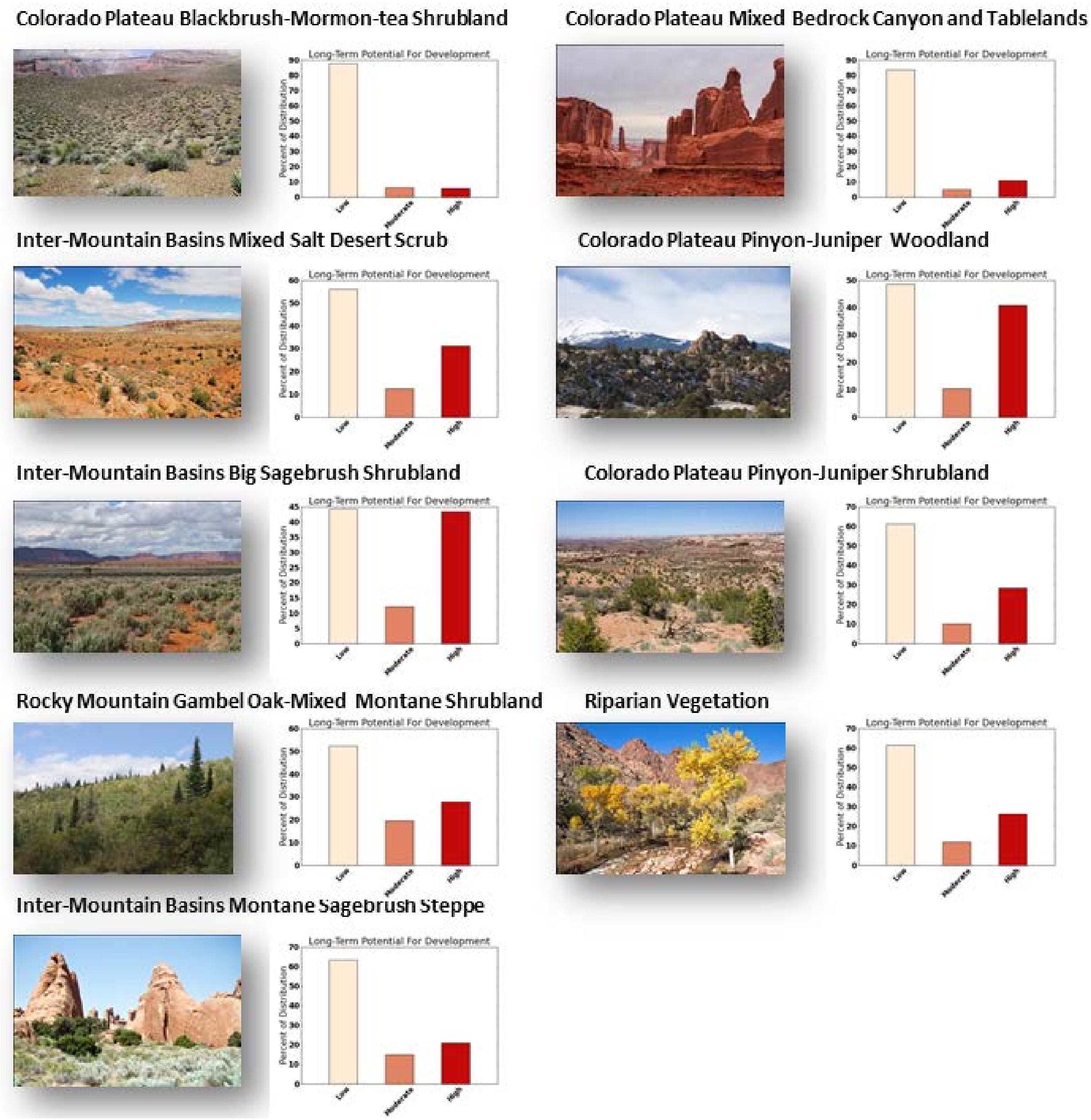


Figure 5-9. Potential impact from energy development for the vegetation communities of the Colorado Plateau ecoregion.

5.3 Near-term Future (2025) Terrestrial Landscape Intactness

Near-term (2025) terrestrial landscape intactness models (at both 4km and HUC5 reporting units) consisted of the same components and construction as the current intactness models with available projection datasets used in place of current condition (Figure 5-10). Urban area and invasive projections (pink boxes in logic model) were updated for the terrestrial landscape intactness model. Invasives species projection data was added to current invasive data (LANDFIRE and NatureServe invasives classes, a predictive model of tamarisk distribution [Jarnevich et al. 2011], and historic tamarisk polyline data). Projections of invasive spread were based on LANDFIRE succession class data, which included all invasive species, and US Geological Survey data on early seasonal invasives (Figure 5-11, J. Hansen, T. Arundel, and R. Kokaly, model created in 2011 for this REA). Process models and source maps may be viewed in Appendix A. FRAGSTATS was not rerun because of the coarse resolution of some of the updated invasives data which would have added additional uncertainty to the results. Overall, conditions in the ecoregion showed a decline with modest decreases in Very High and High intactness area countered by slight increases in the other four categories (Figure 5-12). After completing the intactness model for the near-term (2025) future, distribution data was evaluated for each of the conservation elements and their change in status predicted from the near-term change agents for which data existed.

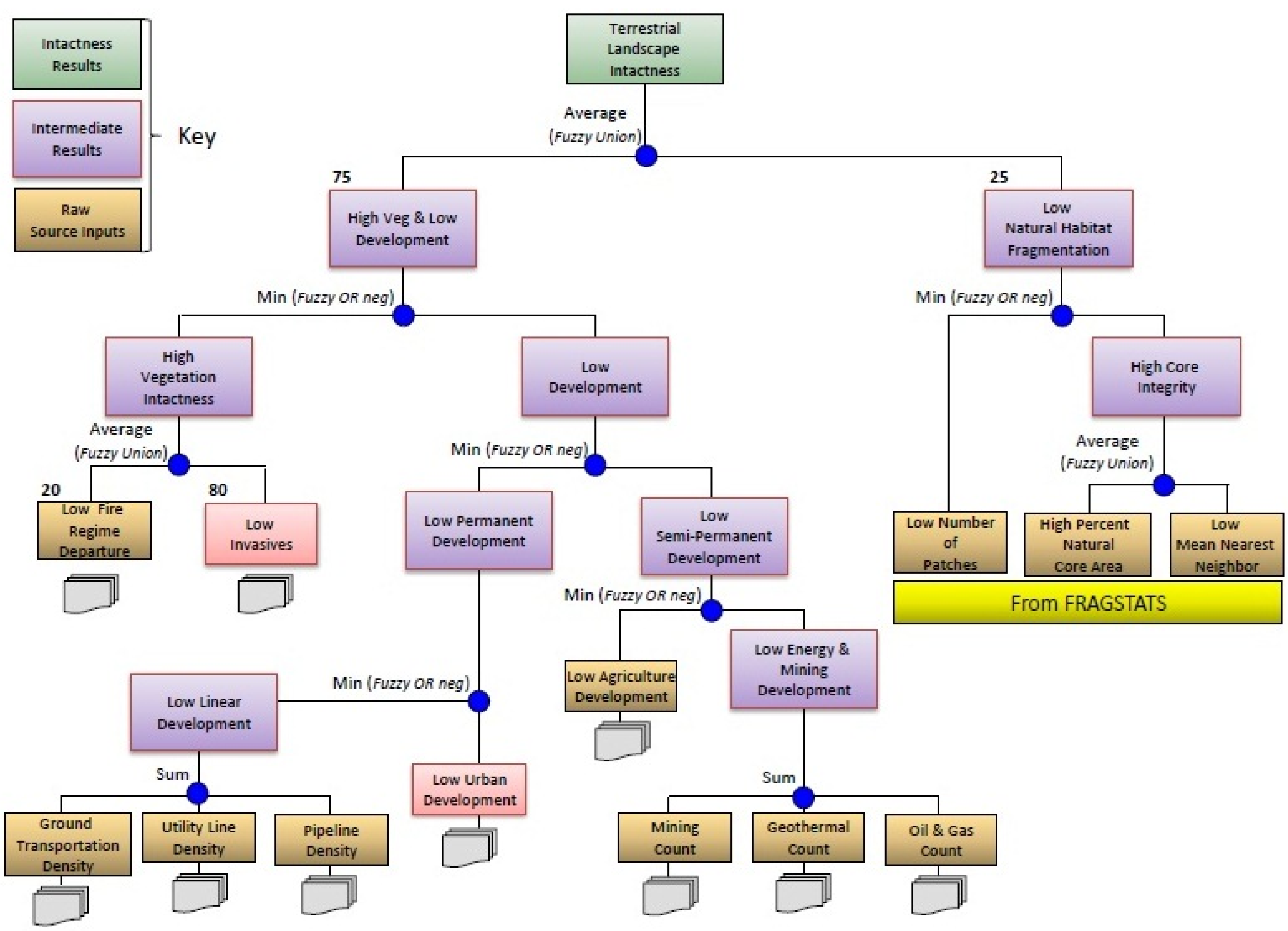


Figure 5-10. Near-term future terrestrial landscape intactness fuzzy logic model.

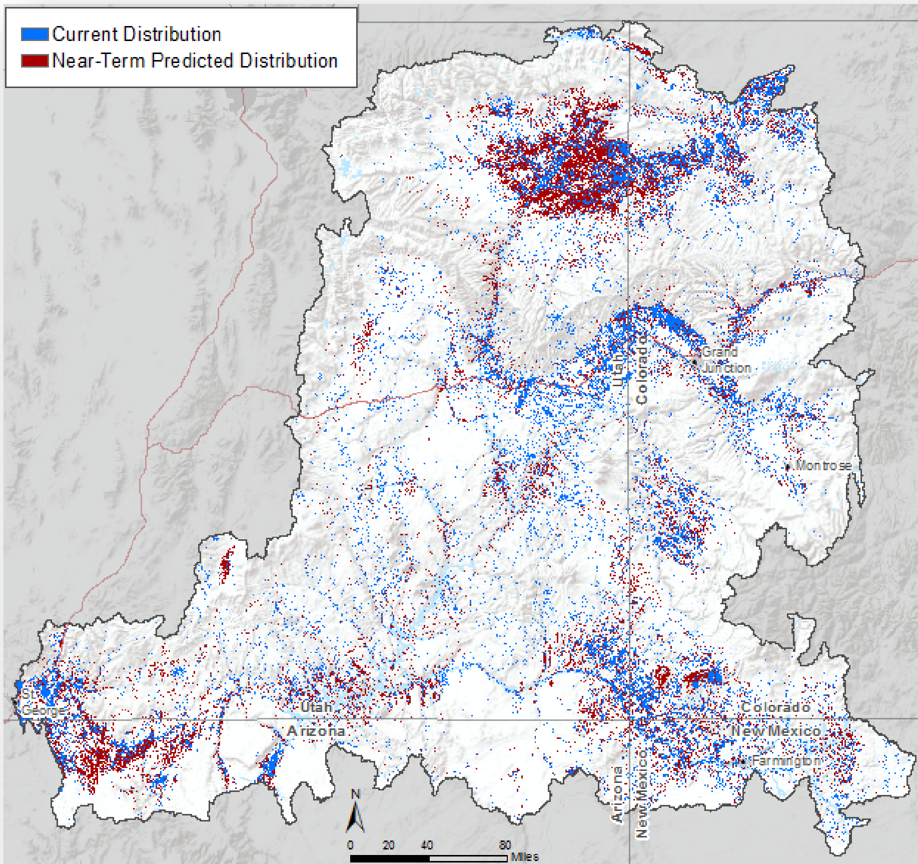


Figure 5-11. Current and near-term future (2025) predicted distribution of invasive species.

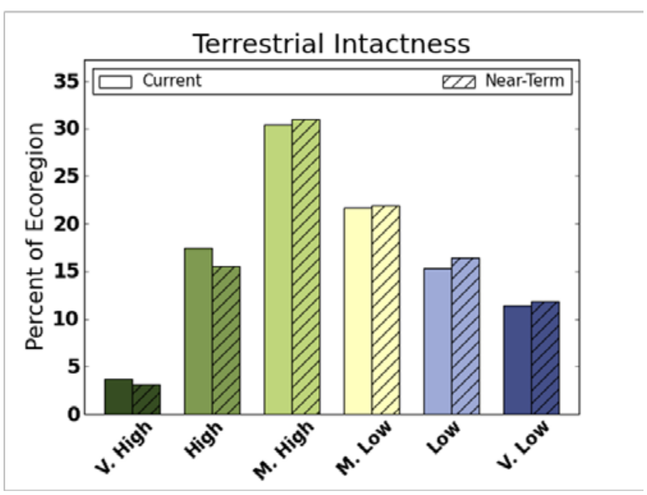


Figure 5-12. Histogram comparing current (solid color bars) and near-term future (hatched bars) terrestrial landscape intactness for the Colorado Plateau showing decreases in Very High and High intactness area and slight increases in the other four categories.

5.3.1 Near-Term Future Status for Terrestrial Wildlife Species

Current and near-term status for each conservation element was based on the terrestrial landscape intactness models for the two time periods using the 4km resolution grid. Results pertain to the area of actual occurrence at the finest scale (1:24,000) or resolution (30m pixels) available for that element. Changes are not dramatic; although the model is well-constructed, it is lacking data for some important elements, such as new recreation, roads, agriculture, and grazing. Thus, although changes are small, they are in a negative (lower intactness) direction, as one would expect.

All mammals experienced some declines in status (Figure 5-13), although some species' habitats were in poorer condition based on current conditions. All of the mammal species that inhabit low elevation, open landscapes showed further declines. Black-footed ferret started out with none of its habitat in Very High or High condition and the next class (Moderately High) indicated a drop of approximately two-thirds. Both prairie dog species showed almost all of their remaining High intactness areas eliminated, and pronghorn antelope also had some declines in all three positive classes. Desert bighorn sheep showed the least amount of impact in 2025 even though it too had declines at the highest intactness classes. Mule deer and mountain lion showed similar response to near-term change (Figure 5-13 and Figure 5-14A) when using the same thresholds for the model variables. When the road threshold was applied to the model for mountain lion as described in Chapter 4 (0.60 km/km^2, Van Dyke et al. 1986), the declines in mountain lion viability are more dramatic (Figure 14A, histogram on the right). The declines are evident not from the addition of potential roads data (projections on roads were not available) but because road densities representing true (or +1 in fuzzy logic) are constrained in the model to a level that does not negatively affect mountain lion (0.60 km/km^2 according to Van Dyke et al. [1986], Figure 5-14A). This is one example of the flexibility of the modeling process that allows quantifiable threshold information to be inserted as it becomes available.

Greater sage-grouse showed the most notable declines in habitat quality of all the bird species, especially when using the oil and gas well density threshold of > 12.3 wells/4 km^2 (8 wells/$mile^2$, Figure 5-14B). The most recent Copeland et al. (2009) data on projected oil and gas potential was included for this one species only. The oil and gas projection data [Copeland et al. 2009] was not applied to Gunnison sage-grouse because none of the projections included in the Copeland et al. (2009) study occurred within its distribution.

Gunnison sage-grouse showed minimal change (Figure 5-14C); however, of all the bird species, Gunnison sage-grouse presently inhabits the least intact habitats, comparable in quality to the habitats of black-footed ferret and the two prairie dog species. Two-thirds of the distribution of Gunnison sage-grouse is on private agricultural or grazing lands that are not at high risk of major transformation in the near-term future except for possible oil and gas development on private land for which there was no projection data. The other bird species, all with a wider range of more intact habitat classes in the present time period, showed consistent declines in the higher quality intactness classes with matching increases in the lower classes in the near-term future (Figure 5-15).

5.3.2 Near-Term Future Status for Aquatic Wildlife Species

The only change made in the aquatic intactness model was the addition of new urban areas for the 2025 time frame. No other data was available to populate the model whether it was planned dams and diversion changes, road construction, or chemical discharge and pesticide application changes. All of these have tremendous impact on aquatic systems for which there was no mechanism to predict into the future. All three fish species primarily showed areas shifting from Low to Very Low aquatic intactness, which by our definition affects their overall status (Figure 5-16).

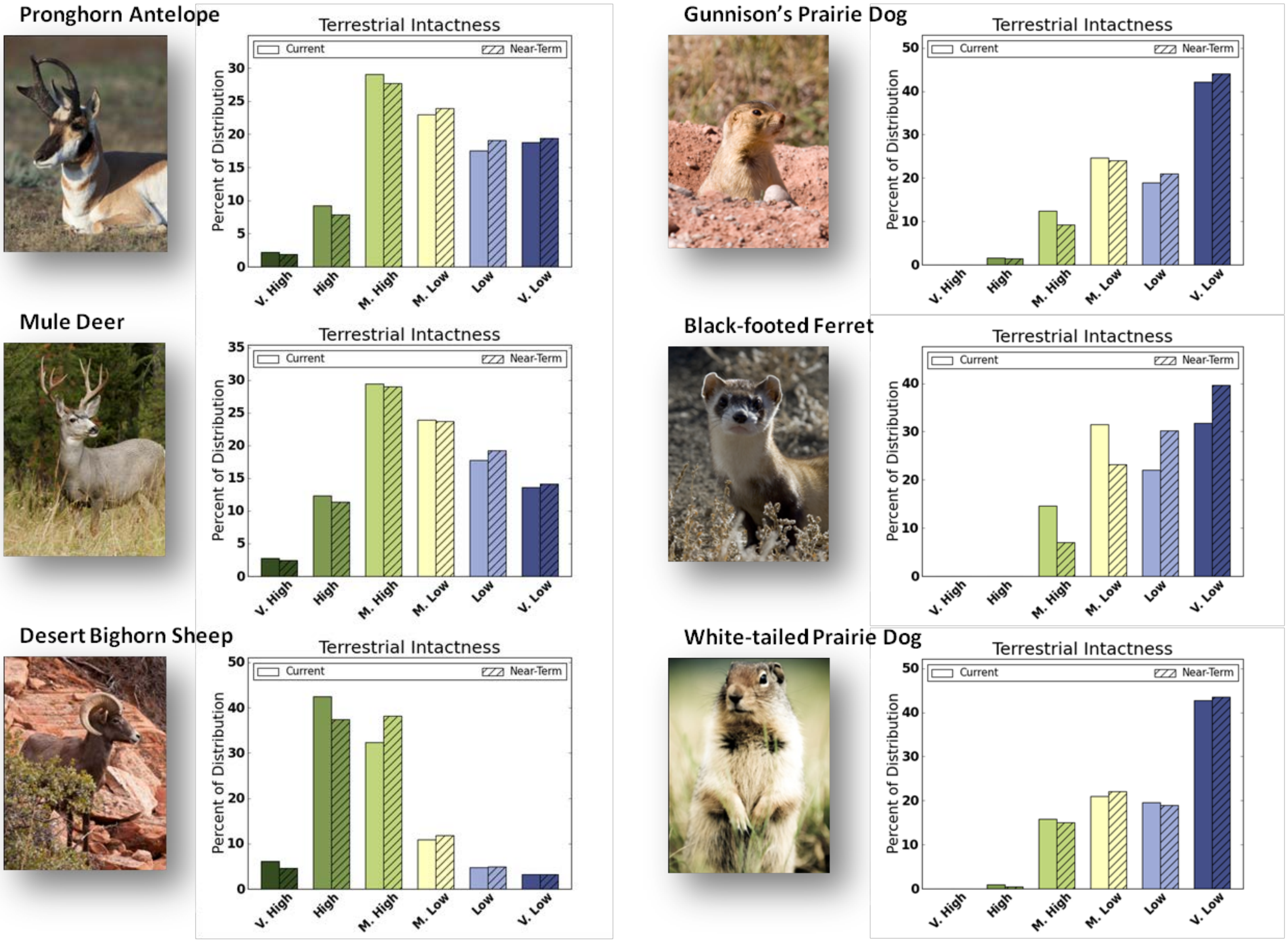


Figure 5-13. Comparison between current (solid) and near-term future (crosshatched) status for mammals.

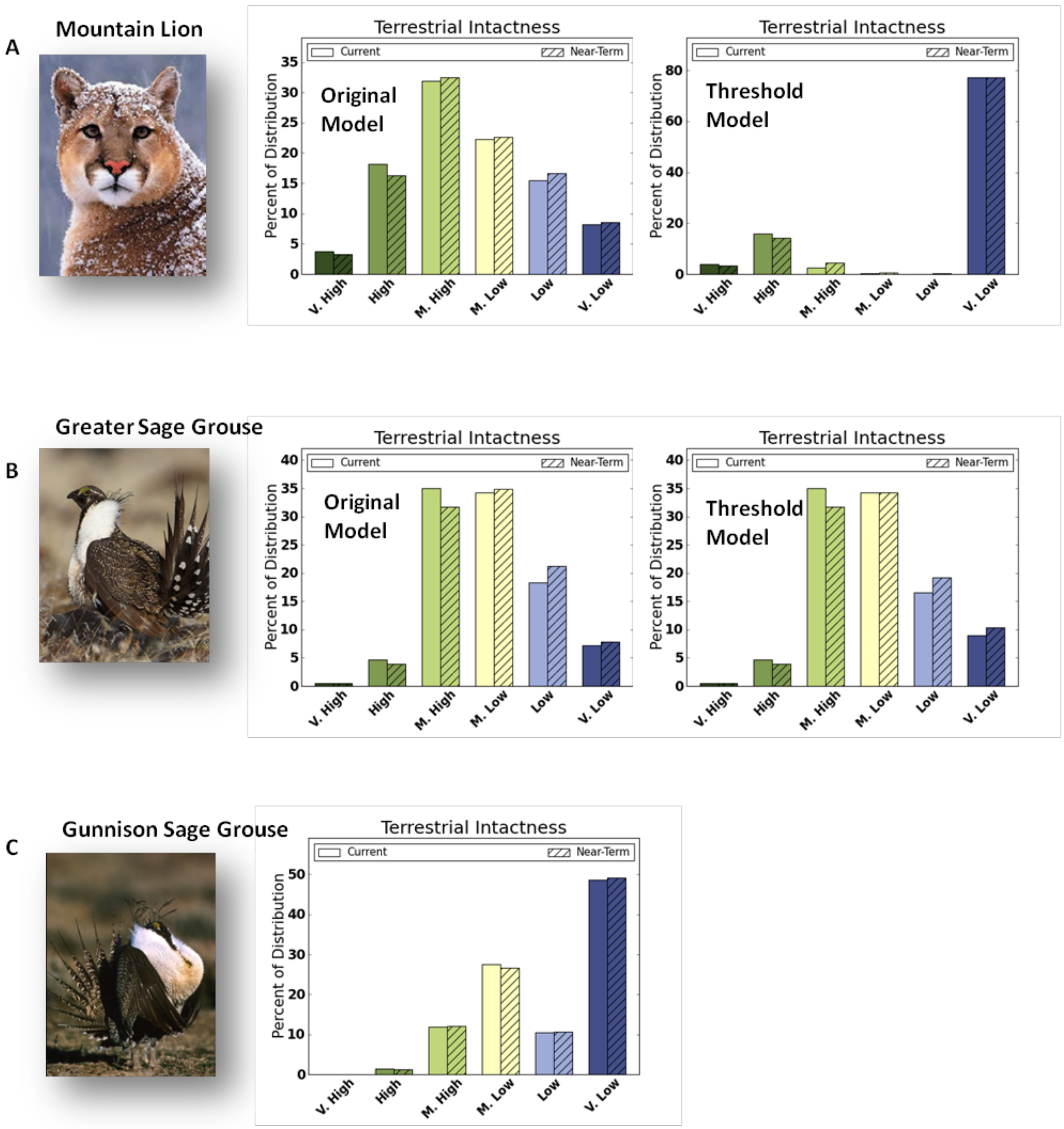


Figure 5-14. A) Comparison of current and near-term future status for mountain lion based on terrestrial landscape intactness for both original and road density threshold (0.60 km/km^2, Van Dyke et al. 1986) model versions; (B) greater sage-grouse current and near-term future status based on original model and near-term future version using additional energy projection data (Copeland et al. 2009) and the >12.3 wells/4 km^2 (8 wells/mile^2) well-density threshold for greater sage-grouse winter habitat (Doherty 2008); and (C) Gunnison sage-grouse current and near-term future status based on original intactness model. (No recently acquired oil and gas data [Copeland et al. 2009] occurred within the distribution of this species.)

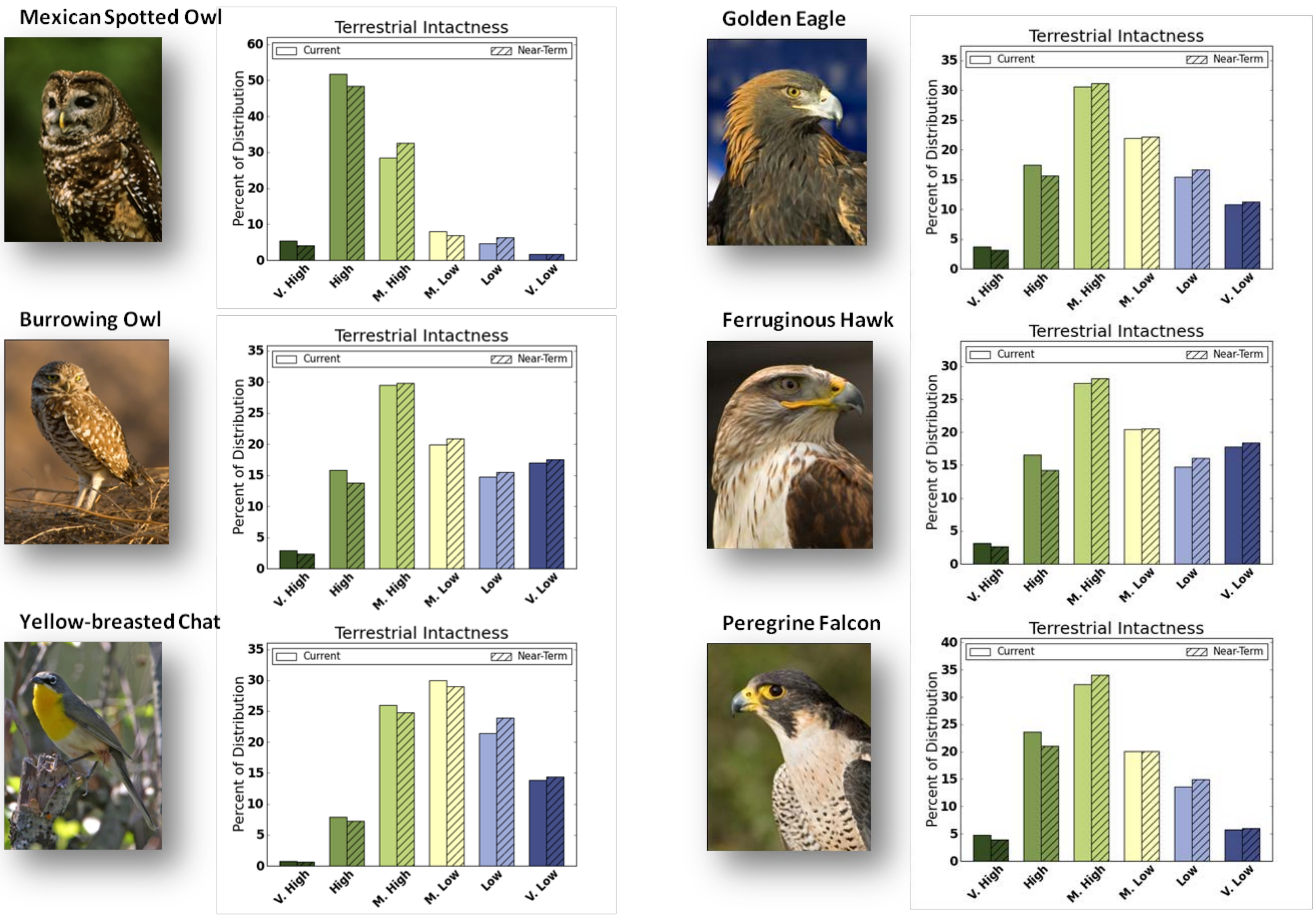


Figure 5-15. Comparison between current (solid bars) and near-term future (2025, crosshatched bars) status for birds.

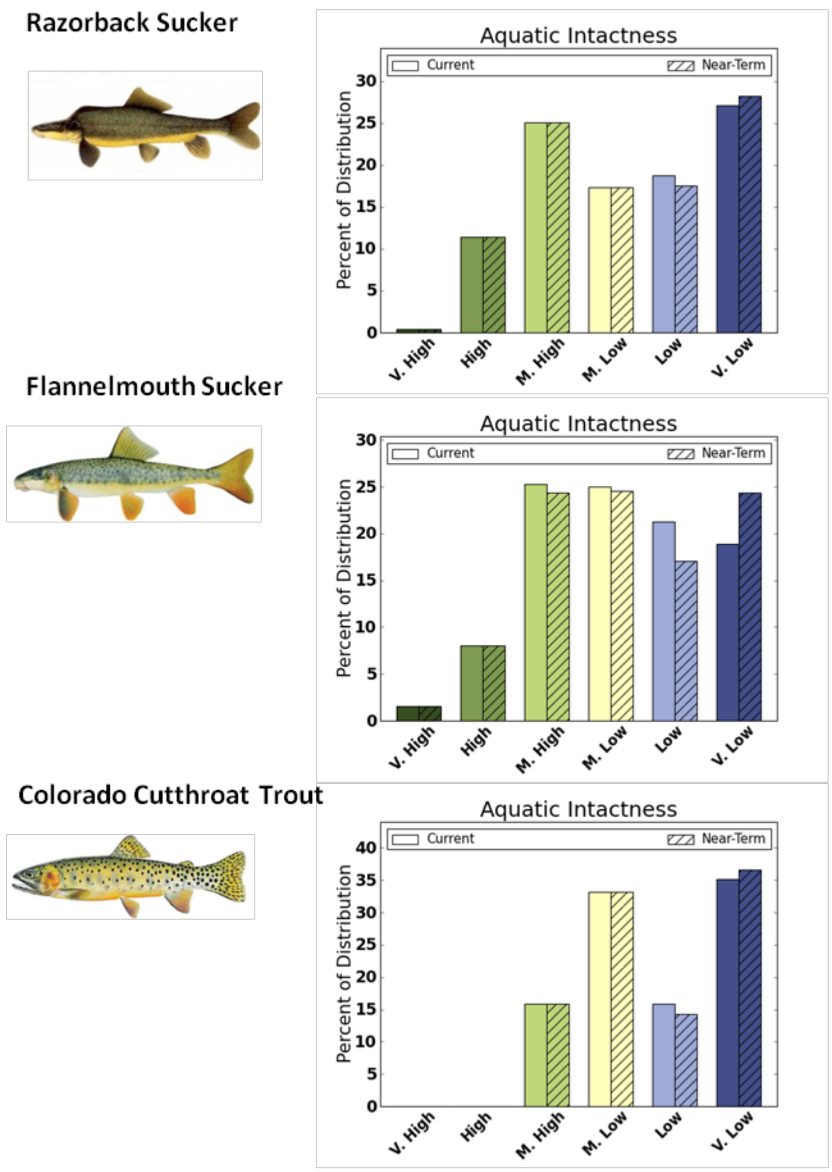


Figure 5-16. Comparison between current (solid bars) and near-term (crosshatched bars) status for fishes based on aquatic intactness.

5.3.3 Near-term Future Status for Designated Lands

Changes in near-term future intactness showed small percentage changes in the status of the existing designated lands in the Colorado Plateau ecoregion (Figure 5-17). Most of this is due to the projected increase in invasive species although some designated lands are already located near developed areas, some of which are expected to expand over time, further degrading lands around these sites. Information on the predicted near-term change in status for the remaining sites conservation elements (e.g., biodiversity sites, herd management areas) can be found in Appendix A.

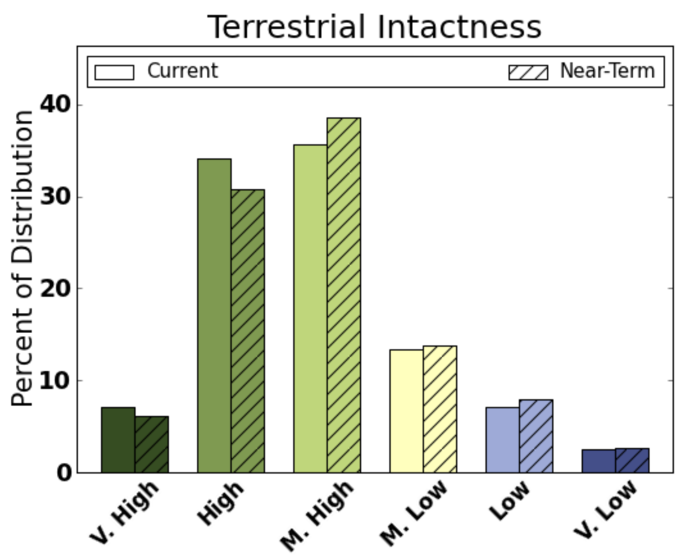
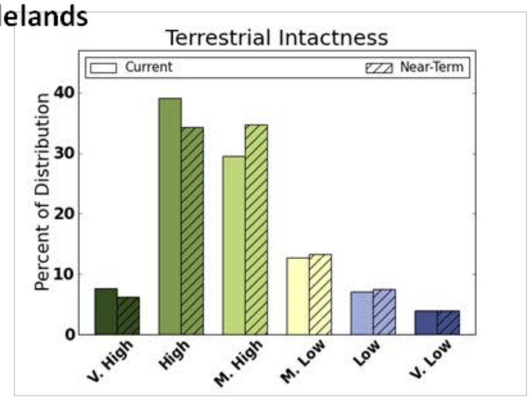


Figure 5-17. Current and near-term (2025) future for status of designated lands in the Colorado plateau ecoregion.

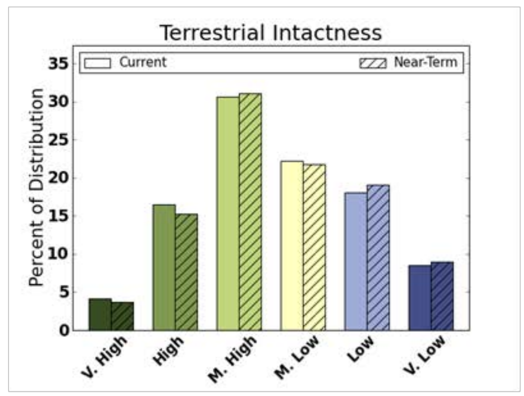
5.3.4 Near-term Future Status for Vegetation Communities

Changes in status from current to near-term future for vegetation communities are presented in Figure 5-18 through Figure 5-20. Near-term terrestrial intactness results showed habitat quality declines in all vegetation communities with the greatest declines observed for the two dominant communities—Colorado Plateau Pinyon-Juniper Woodland and Inter-Mountain Basins Big sagebrush Shrubland (Figure 5-18 and 5-19). The least change was observed in the more sparsely vegetated community types such as Colorado Plateau Mixed Bedrock Canyon and Tablelands (Figure 5-18) and Colorado Plateau Blackbrush-Mormon-tea Shrubland (Figure 5-20).

Colorado Plateau Mixed Bedrock Canyon and Tablelands



Colorado Plateau Pinyon-Juniper Woodland



Colorado Plateau Pinyon-Juniper Shrubland

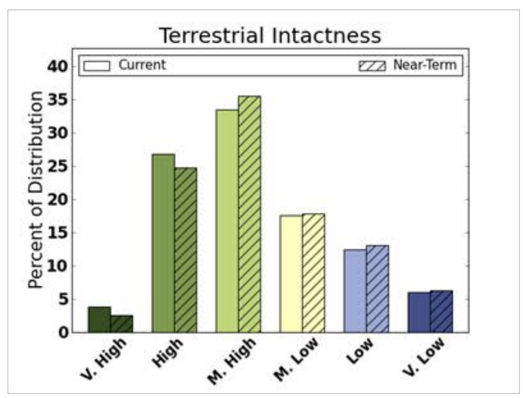
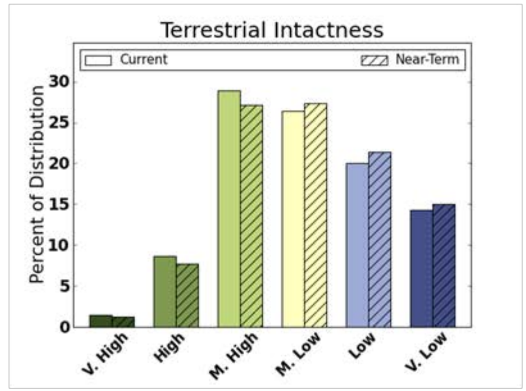
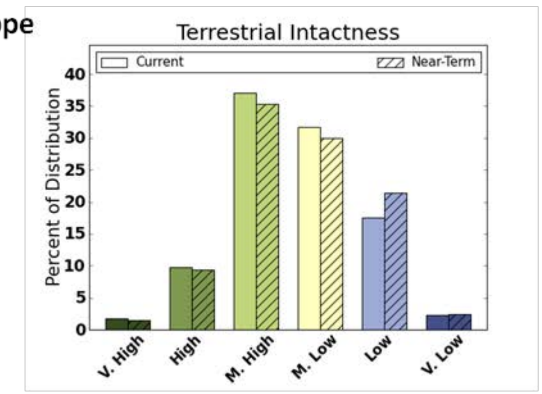


Figure 5-18. Comparison between current (solid) and near-term (crosshatched) status for Colorado Plateau Mixed Bedrock Canyon and Tablelands, Colorado Plateau Pinyon-Juniper Woodland, and Colorado Plateau Pinyon-Juniper Shrubland according to NatureServe Landcover v 2.7 for the Colorado Plateau ecoregion.

Inter-Mountain Basins Big Sagebrush Shrubland



Inter-Mountain Basins Montane Sagebrush Steppe



Riparian Vegetation

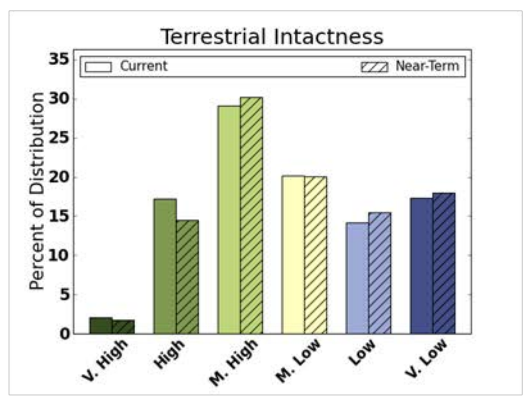
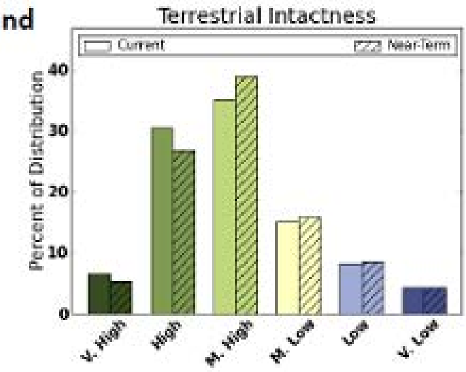
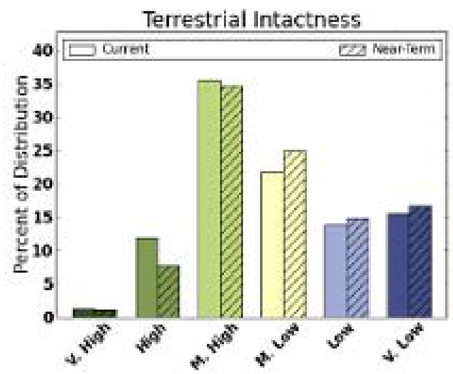


Figure 5-19. Comparison between current (solid) and near-term (crosshatched) status for Inter-Mountains Basins Big Sagebrush Shrubland, Inter-Mountain Basins Montane Sagebrush Steppe, and Riparian vegetation according to NatureServe Landcover v 2.7 for the Colorado Plateau ecoregion.

Colorado Plateau Blackbrush-Mormon-tea Shrubland



Inter-Mountain Basins Mixed Salt Desert Scrub



Rocky Mountain Gambel Oak-Mixed Montane Shrubland

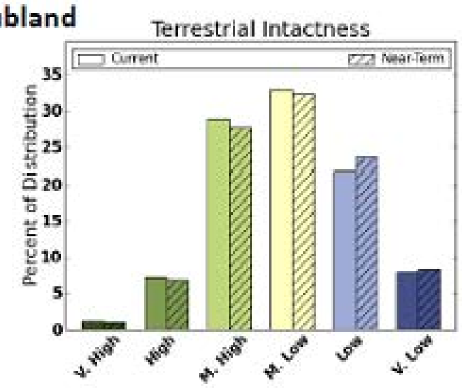


Figure 5-20. Comparison between current (solid) and near-term (crosshatched) status for Colorado Plateau Blackbrush-Mormon-tea Shrubland, Inter-Mountains Basins Mixed Salt Desert Scrub, and Rocky Mountain Gambel Oak-Mixed Montane Shrubland according to NatureServe Landcover v 2.7 for the Colorado Plateau ecoregion.

5.4 Climate Change

Climate change results for the Colorado Plateau ecoregion are extensive and complex. Although three different future climate projections were investigated, the ECHAM5-driven RegCM3 climate projections were selected for the body of the report to evaluate potential impact on the various conservation elements. ECHAM5 is the fifth generation of the ECHAM global general circulation model (GCM) developed at the Max Planck Institute in Hamburg, Germany; it has been identified as one of the better models to represent natural climate variability (Mote et al. 2010, Garfin et al. 2010). The other two projections, the GFDL- and GENMOM-driven RegCM3, had results that were wetter overall than many of the other published climate projections (IPCC 2007). The regional RegCM3 model had been chosen because of its representation of the North American Monsoon (Hostetler et al. 2011) which is important to Colorado Plateau vegetation dynamics.

5.4.1 Climate Projections

As explained in detail in Chapter 3, Methodology, the climate model data provided by Hostetler et al. (2011) were assembled for two time periods (2015–2030 and 2045–2060), and data from the 2045–2060 time period were used to evaluate the conservation elements, which are presented later in this section. For both temperature and precipitation results, water bodies were left as holes in the modeled data because they create their own local climate conditions and are thus areas of greater uncertainty for the climate model to simulate. Weather stations are rarely placed near water bodies because they would be skewed towards these very local conditions and would not represent broad patterns over the landscape.

Differences in temperature projections—average annual temperature (Figure 5-21), seasonal summer temperature (July–September; Figure 5-22), and winter temperature (January–March; Figure 5-23)—were calculated between historical (1968–1999) and future time periods (2015–2030 and 2045–2060) as simulated by the ECHAM5-driven RegCM3 model. The differences were then used to modify the PRISM historic baseline for 1968–1999. Results show that the ecoregion is expected to undergo general warming over the entire region with as much as 2° Celsius increase by 2060 in some locations, particularly in the southern portion of the ecoregion. Average summer temperatures are expected to increase, but even greater increases are simulated for the winter months. Downscaled climate modeling for the southern Colorado Plateau by Garfin et al. (2010) predicted even greater warming of 4.7° Celsius by the end of the century.

It is generally accepted that climate models are less reliable in simulating precipitation than temperature. There was a large bias in the RegCM3 simulations of historical precipitation for this region. Consequently we had to bias-correct the climate model results to provide more realistic climate input to the vegetation model. We generated future climate projections (precipitation maps Figures 5-25 through 5-27, top row maps 2 and 3) by calculating the ratios between future and historical precipitation values simulated by RegCM3 and multiplying them by the historical PRISM baseline. Under RegCM3 projections, precipitation is expected to decline throughout much of the year during the 2015–2030 time period (with the exception of a couple months in the fall) with severe drought likely to occur in some areas (graph in Figure 5-24 and Figure 5-25). The 2045–2060 time period remains drier (or comparable to historic conditions) during most of the year, but sporadic wetter months (e.g., February, June, and October in Figure 5-24) result in some areas expressing overall projected increases in annual precipitation (Figure 5-25). Considerable variability can be seen when the graphed data is expressed spatially in the precipitation map figures (Figures 5-25 to 5-27). For the seasonal results, summer (Jul–Sep, Figure 5-26) showed more spatial variability in precipitation than did the winter season (Jan–Mar, Figure 5-27).

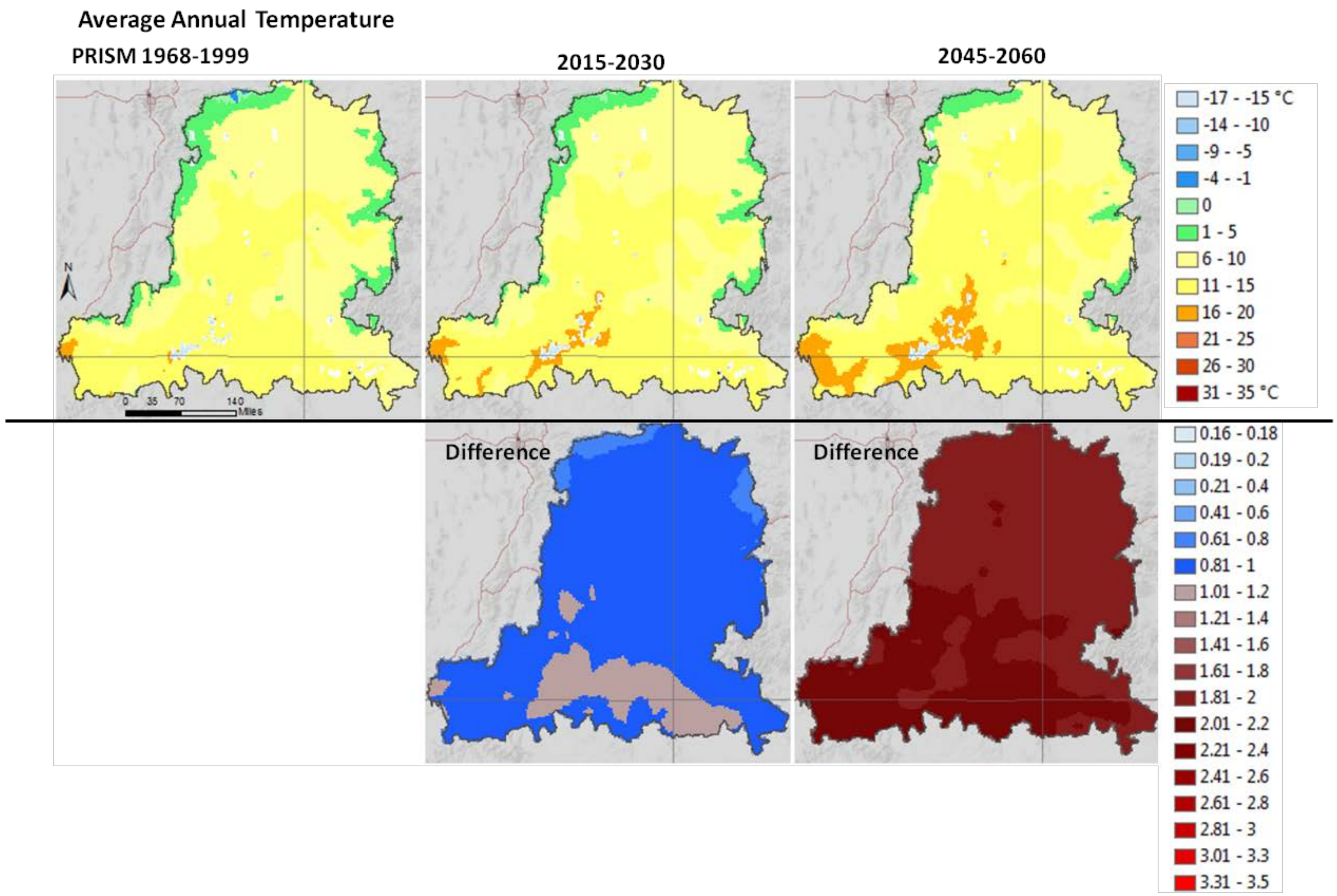


Figure 5-21. Map results for change in raw average annual temperature. Top Row: 1) Observed average annual temperature from PRISM averaged over the historical period (1968–1999 baseline) for the Colorado Plateau ecoregion.; 2-3) Bias-corrected future temperature using the ECHAM5-driven RegCM3 regional climate model deltas modifying the PRISM baseline (1) and averaged for two future time periods. Bottom row: Simulated ECHAM5-driven RegCM3 regional climate model differences between historical (1968–1999) and future (2015–2030; 2045–2060) average annual temperature. All colors on the difference maps are warmer than historic. Note: Bias correction was applied to the climate model results for more realistic climate input to the vegetation model. Future climate projections (top row 2-3) were generated by calculating the differences between future and historical temperature values simulated by RegCM3 (bottom row) and adding them to the historical PRISM baseline (top row).

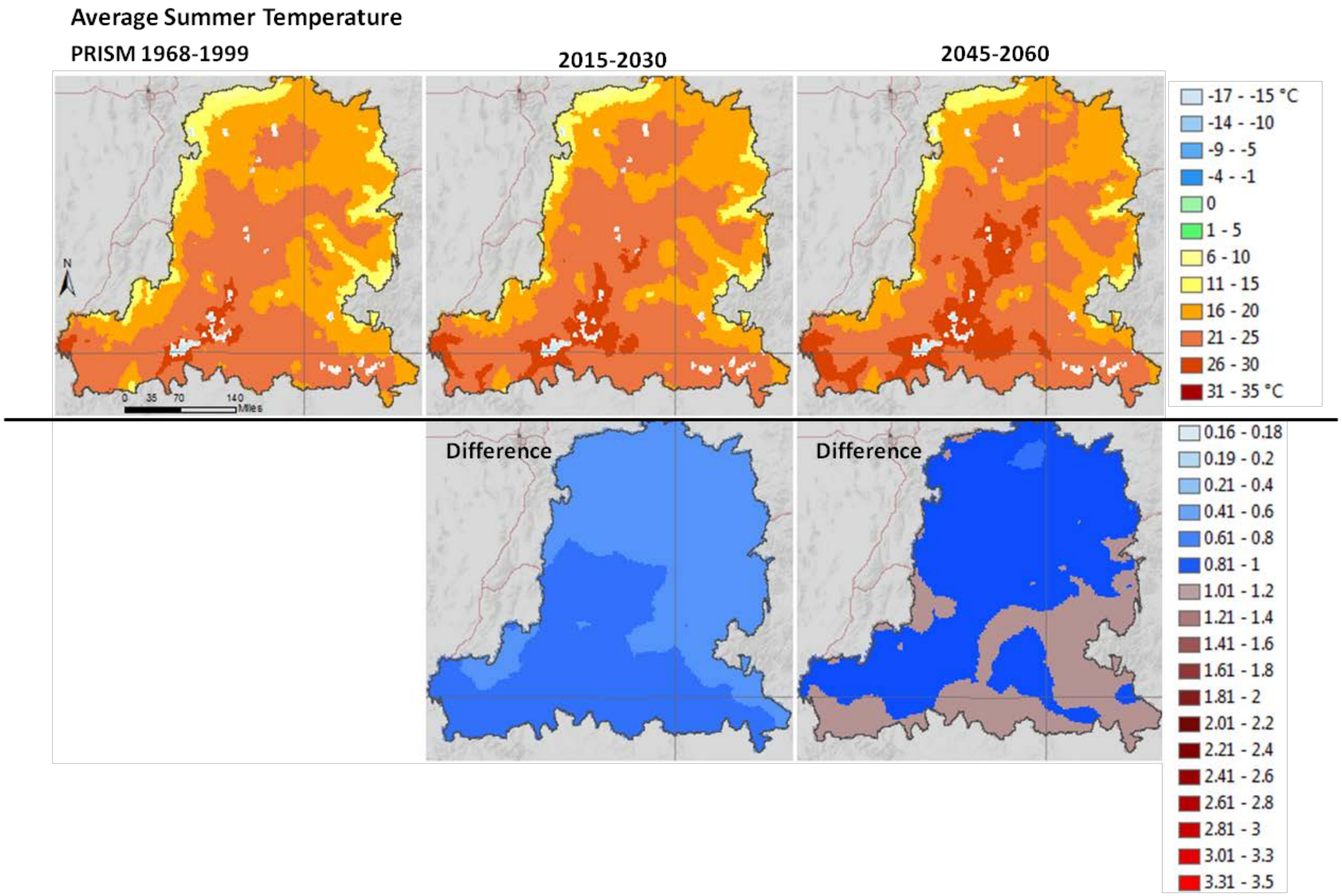


Figure 5-22. Map results for change in raw average summer temperature. Top Row: 1) Observed average summer (July–September) temperature from PRISM averaged over the historical period (1968–1999 baseline) for the Colorado Plateau ecoregion.; 2-3) Bias-corrected future summer temperature using the ECHAM5-driven RegCM3 regional climate model deltas modifying the PRISM baseline (1), and averaged for two future time periods. Bottom row: Simulated ECHAM5-driven RegCM3 regional climate model differences between historical (1968–1999) and future (2015–2030; 2045–2060) average summer temperature. All colors on the difference maps are warmer than historic. Note: Bias correction was applied to the climate model results for more realistic climate input to the vegetation model. Future climate projections (top row 2-3) were generated by calculating the differences between future and historical temperature values simulated by RegCM3 (bottom row) and adding them to the historical PRISM baseline (top row).

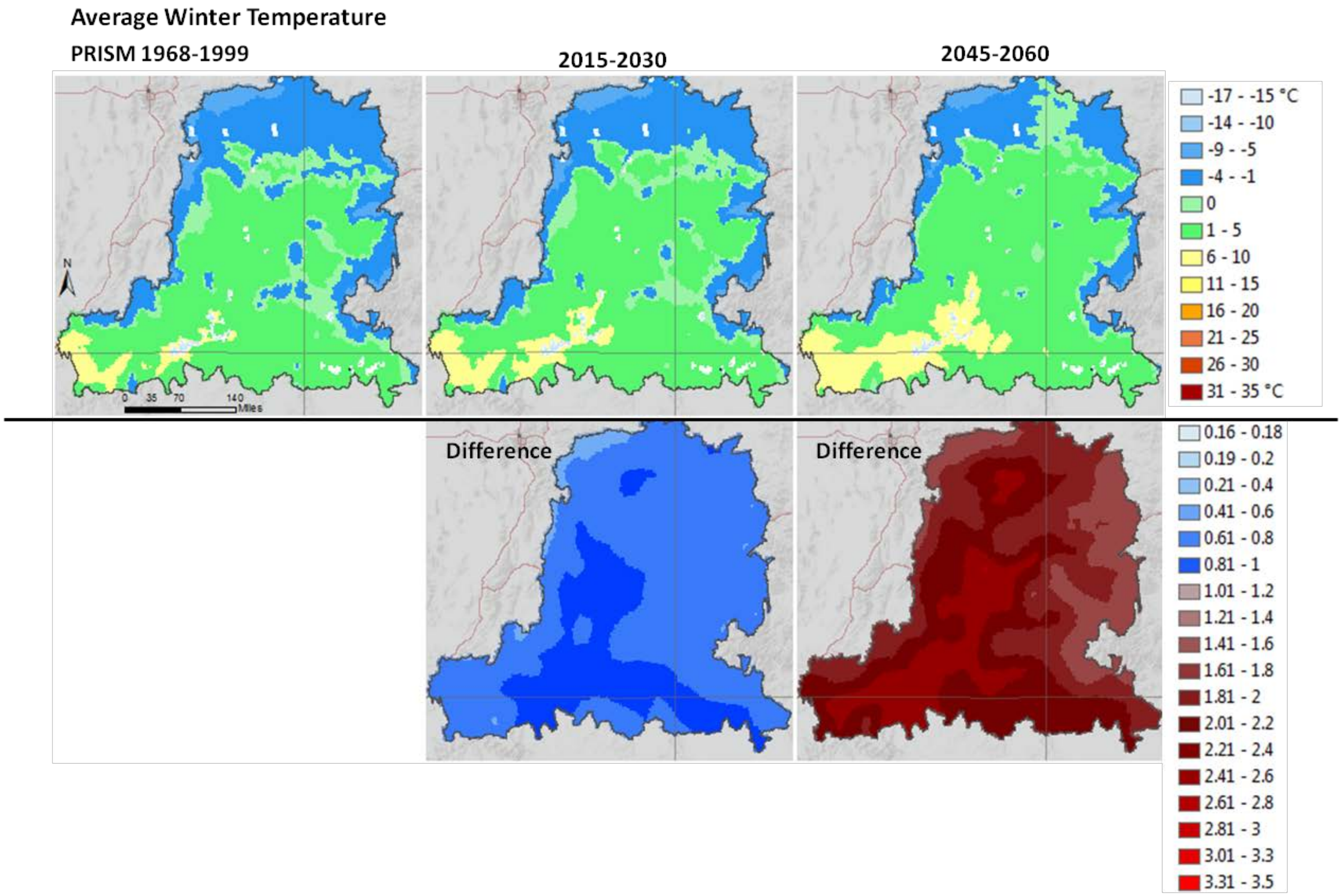
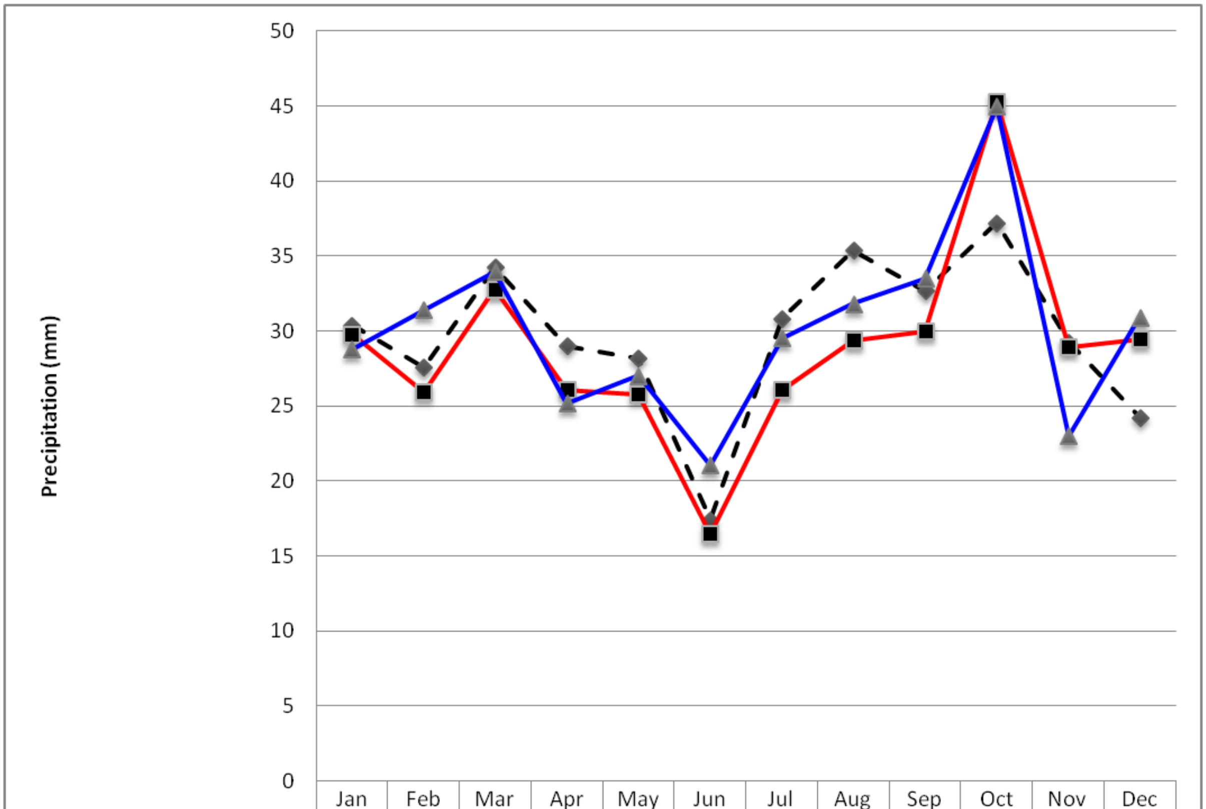


Figure 5-23. Map results for change in raw average winter temperature. Top Row: 1) Observed average winter (January–March) temperature from PRISM averaged over the historical period (1968–1999 baseline) for the Colorado Plateau ecoregion; 2-3) Bias-corrected future winter temperature using the ECHAM5-driven RegCM3 regional climate model deltas modifying the PRISM baseline (1), and averaged for two future time periods. Bottom row: Simulated ECHAM5-driven RegCM3 regional climate model differences between historical (1968–1999) and future (2015–2030; 2045–2060) average winter temperature. All colors on the difference maps are warmer than historic. Note: Bias correction was applied to the climate model results for more realistic climate input to the vegetation model. Future climate projections (top row 2–3) were generated by calculating the differences between future and historical temperature values simulated by RegCM3 (bottom row) and adding them to the historical PRISM baseline (top row).



	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
PRISM	30.39	27.57	34.25	28.99	28.19	17.42	30.84	35.38	32.69	37.22	29.25	24.23
ECHAM5 2015-2030	29.79	25.94	32.79	26.05	25.76	16.47	26.09	29.37	29.97	45.27	28.9	29.44
ECHAM5 2045-2060	28.77	31.39	33.94	25.18	27.01	21.04	29.51	31.8	33.51	44.97	22.98	30.92

Figure 5-24. Graph of average precipitation for each month for each evaluated time period (PRISM=historic 1968–1999) and two future time windows (2015–2030 and 2045–2060) based on the RegCM3 using ECHAM5 boundary conditions.

Using ECHAM-5 GCM projections, Garfin et al (2010) found a 30% decrease in summer precipitation for the southern Colorado Plateau in 2050, as compared to only a 6.5% decline in annual precipitation using an ensemble approach of general circulation model (GCM) projections. Seager et al (2007), using the ensemble mean of 19 GCMs (from the Intergovernmental Panel on Climate Change [IPCC] Fourth Assessment Report [AR4] for the 20th and 21st centuries) and looking at the difference between projected precipitation and evaporation in the Southwest region, warned of future droughts more intense than those recorded during the Dust Bowl of the 1930s and the U.S. droughts of the 1950s.

This (regional) RegCM3 result does not necessarily agree precisely with (global) GCMs. In the past, assessments have used coarse general circulation model projections statistically downscaled to the landscape of interest. Regional climate models are run at finer scales and take into account local processes that are not detected by global climate models. Disagreement or differences in the magnitude of changes between the two types is to be expected. A full interpretation of which is correct is beyond the scope of this assessment but available for debate and comparison to observed records during the historical period.

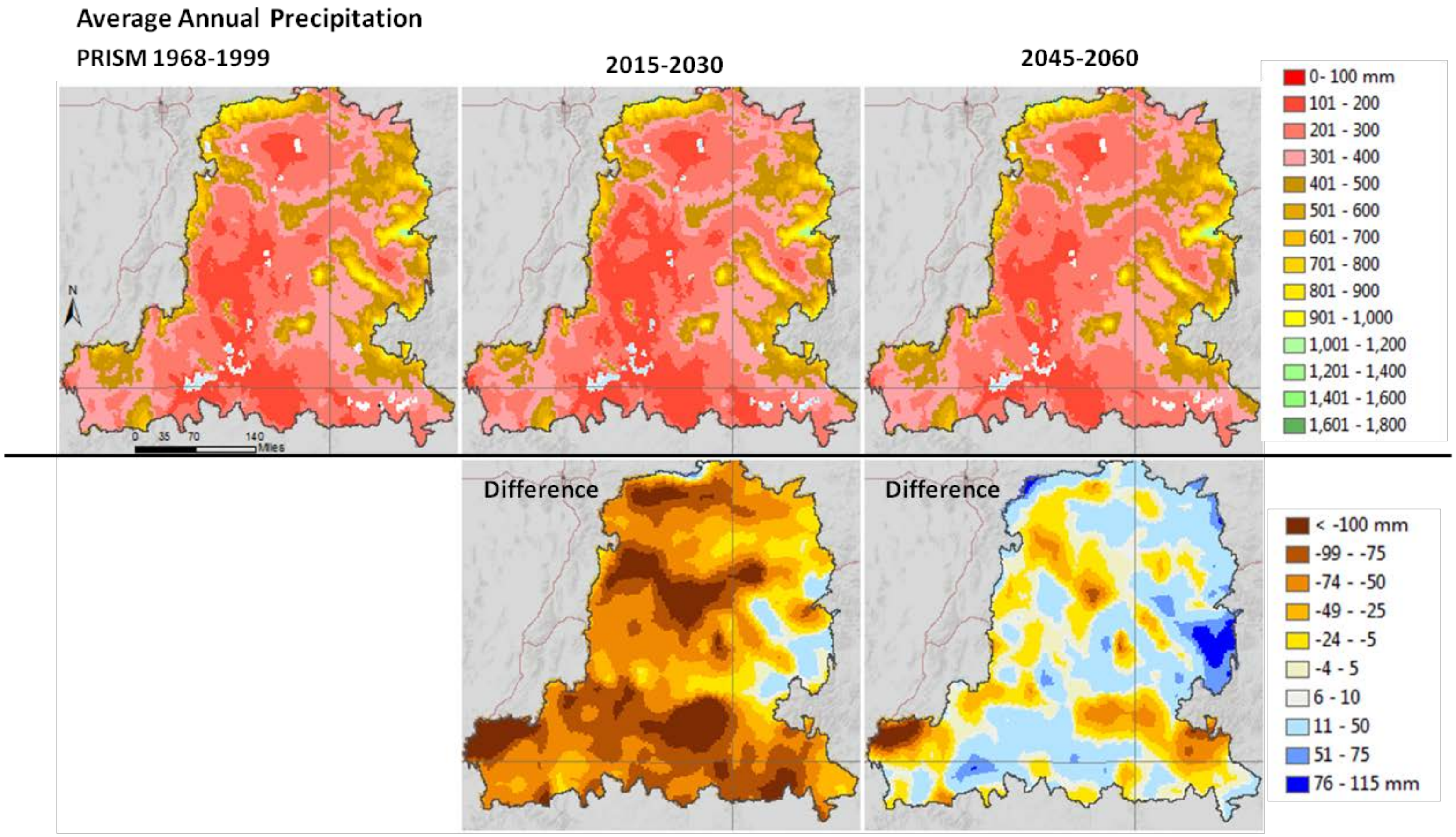


Figure 5-25. Map results for change in average annual precipitation. Top Row: 1) Observed average annual precipitation from PRISM averaged over the historical period (1968–1999 baseline) for the Colorado Plateau ecoregion.; 2-3) Bias-corrected future precipitation using the ECHAM5-driven RegCM3 regional climate model deltas modifying the PRISM baseline (1), and averaged for two future time periods. Bottom row: Simulated ECHAM5-driven RegCM3 regional climate model differences between historical (1968–1999) and future (2015–2030; 2045–2060) average annual precipitation. For the difference maps, brown color tones represent drier conditions and blue colors represent wetter conditions. Note: There was a large bias in the RegCM3 simulations of historical precipitation for this region. Consequently, the climate model results were bias corrected to provide more realistic climate input to the vegetation model. Future climate projections (top row 2-3) were generated by calculating the ratios between future and historical precipitation values simulated by RegCM3 and multiplying them by the historical PRISM baseline.

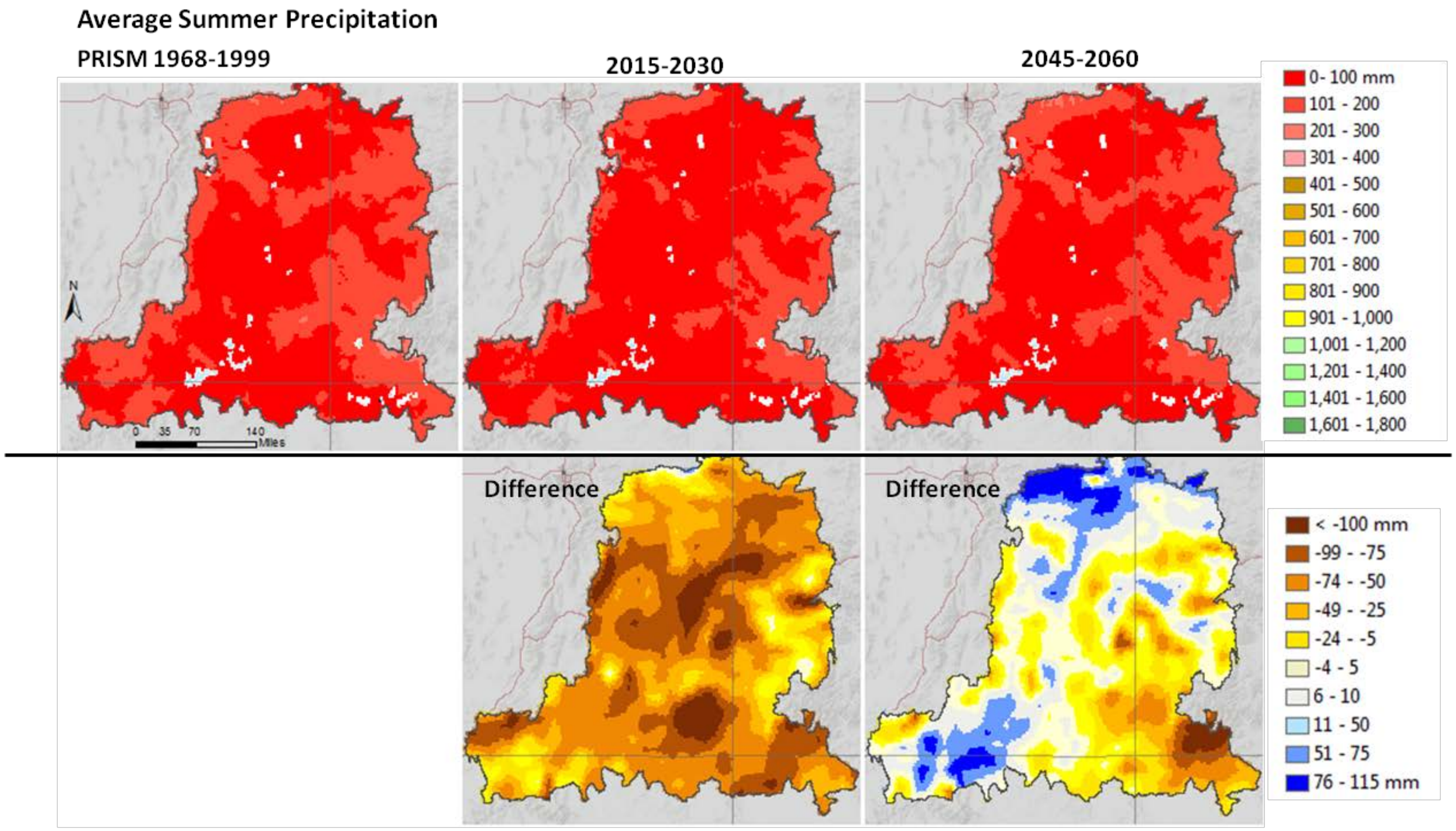


Figure 5-26. Map results for change in average annual summer precipitation. Top Row: 1) Observed summer precipitation (July–September) from PRISM averaged over the historical period (1968–1999 baseline) for the Colorado Plateau ecoregion.; 2-3) Bias-corrected future precipitation using the ECHAM5-driven RegCM3 regional climate model deltas modifying the PRISM baseline (1), and averaged for two future time periods. Bottom row: Simulated ECHAM5-driven RegCM3 regional climate model differences between historical (1968–1999) and future (2015–2030; 2045–2060) average summer precipitation. In difference maps, brown colors represent drier conditions and blue colors represent wetter conditions. Note: There was a large bias in the RegCM3 simulations of historical precipitation for this region. Consequently, the climate model results were bias corrected to provide more realistic climate input to the vegetation model. Future climate projections (top row 2-3) were generated by calculating the ratios between future and historical precipitation values simulated by RegCM3 and multiplying them by the historical PRISM baseline.

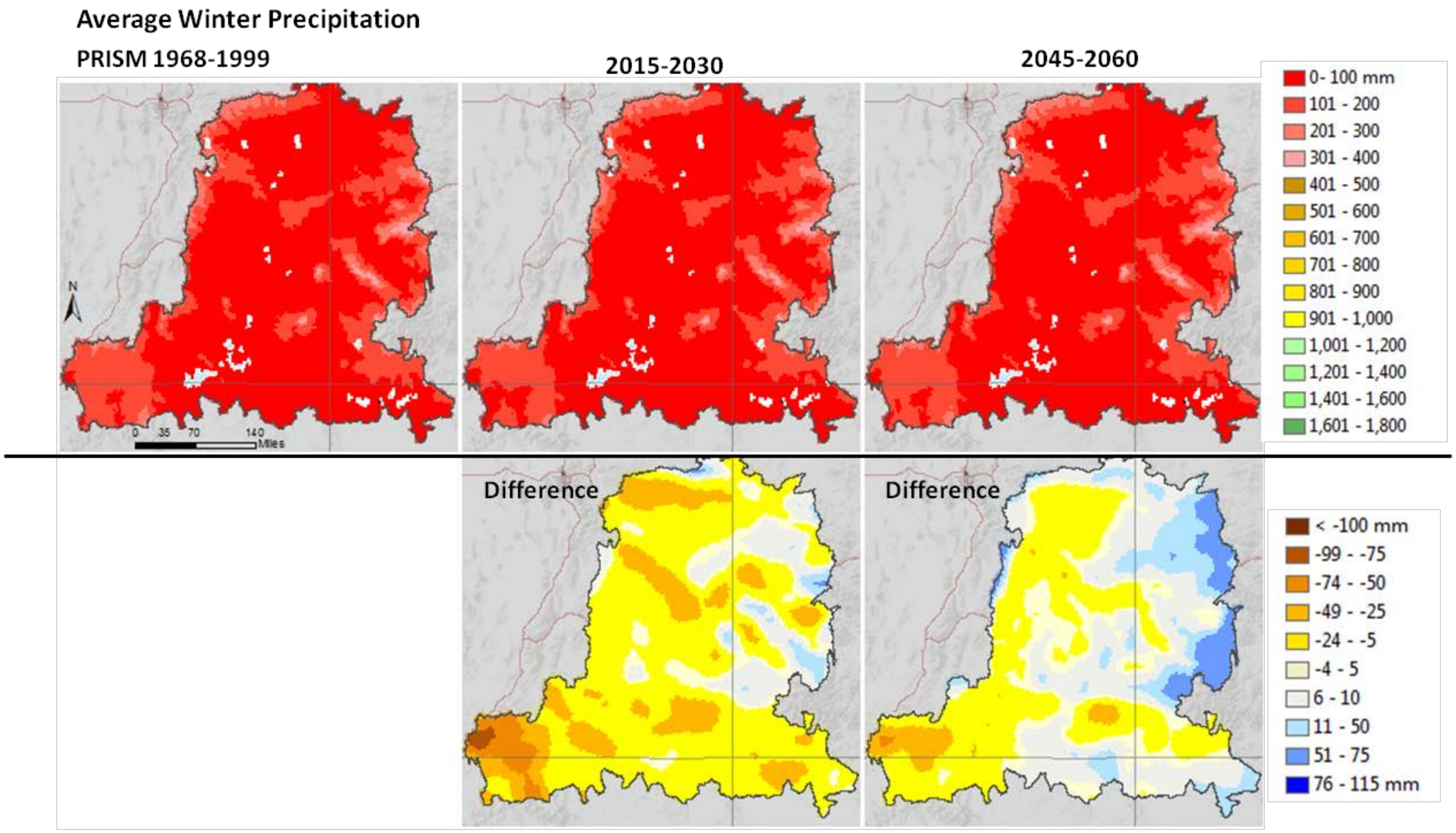


Figure 5-27. Map results for change in average annual winter precipitation. Top Row: 1) Observed winter precipitation (January–March) from PRISM averaged over the historical period (1968–1999 baseline) for the Colorado Plateau ecoregion.; 2-3) Bias-corrected future precipitation using the ECHAM5-driven RegCM3 regional climate model deltas modifying the PRISM baseline (1), and averaged for two future time periods. Bottom row: Simulated ECHAM5-driven RegCM3 regional climate model differences between historical (1968–1999) and future (2015–2030; 2045–2060) average winter precipitation. For the difference maps, brown color tones represent drier conditions and blue colors represent wetter conditions. Note: There was a large bias in the RegCM3 simulations of historical precipitation for this region. Consequently, the climate model results were bias corrected to provide more realistic climate input to the vegetation model. Future climate projections (top row 2-3) were generated by calculating the ratios between future and historical precipitation values simulated by RegCM3 and multiplying them by the historical PRISM baseline.

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5.4.1.1 MAPSS Modeling Results

Four different Mapped Atmosphere Plant Soil System (MAPSS) modeling outputs (see Chapter 3 Methods) were generated for the REA—Leaf Area Index (LAI), Potential Evapotranspiration (PET), Runoff, and Potential Vegetation distribution. Simulated LAI slightly declined overall in most areas, meaning that higher mortality causing canopy thinning or a shift to sparser, more drought-resistant vegetation may cause a change in vegetation communities in these locations (Figure 5-28). Only a few areas at higher elevations displayed any increase in LAI (green pixels on the difference maps). Changes in Potential Evapotranspiration (Figure 5-29) indicated an overall drying of the soil and, with the decline in LAI, suggested a probable decline in vegetation growth over most of the ecoregion (green areas on the map) with only a limited spattering of more moist conditions for vegetation (in purple, Figure 5-29). Runoff showed a slight decline over most of the ecoregion except for the eastern portion, which is expected to experience more runoff from higher elevations in the future (Figure 5-30).

One of the main projections from the MAPSS model is the potential shift in major vegetation types through time based on changes in plant functional groups. The model uses a historic climate baseline (PRISM) to predict the types of vegetation that would be supported under the given set of environmental conditions (see Methods, Climate Modeling, for more details). MAPSS does not take into account human management of natural landscapes (e.g. water management, logging, or grazing). It only uses the raw environmental variables (climate and soil) to predict vegetation. With a long history of human use in the ecoregion, the PRISM historic starting point should not be expected to reflect exactly what is on the ground today.

Considerable change in vegetation is predicted between 1968–1999 and 2045–2060 (Table 5-2 and Figures 5-31 and 5-32). Also, the MAPSS model is a static vegetation model that is run independently for each of the two time periods; therefore, one result does not affect the other. Normally, any dry or wet periods have repercussions on the following year's vegetation response. In this case, the static vegetation model just simulates what the instantaneous climate data can support.

The MAPSS model predicts that climate conditions will change to favor more grasses and shrubland subtropical xeromorphic (e.g., Gambel oak and western juniper). MAPSS results show small to moderate increases in C_3 grasses, tall and mid-sized C_3C_4 mixed grasses and larger increases in short C_4 and C_3C_4 mixed grasses (Table 5-2). Potential declines were reported in the vegetation communities shrub savanna evergreen, pinyon-juniper savanna continental, savanna evergreen needle continental, and forest evergreen needle continental. Munson et al. (2011), in a study examining historic climate-induced changes to vegetation plots in protected National Parks, predicted declines in C_3 grasses, no change in C_4 grasses, and increases in shrubs and woodland juniper. A partial explanation for the differences between the two results, particularly in the projection of the distribution of grasses, is that Munson et al. (2011) were looking at actual vegetation in parks that had experienced some human disturbance before the areas were protected and the MAPSS model focuses on projecting future potential vegetation reflecting no human influences and no community changes from agents such as invasive species.

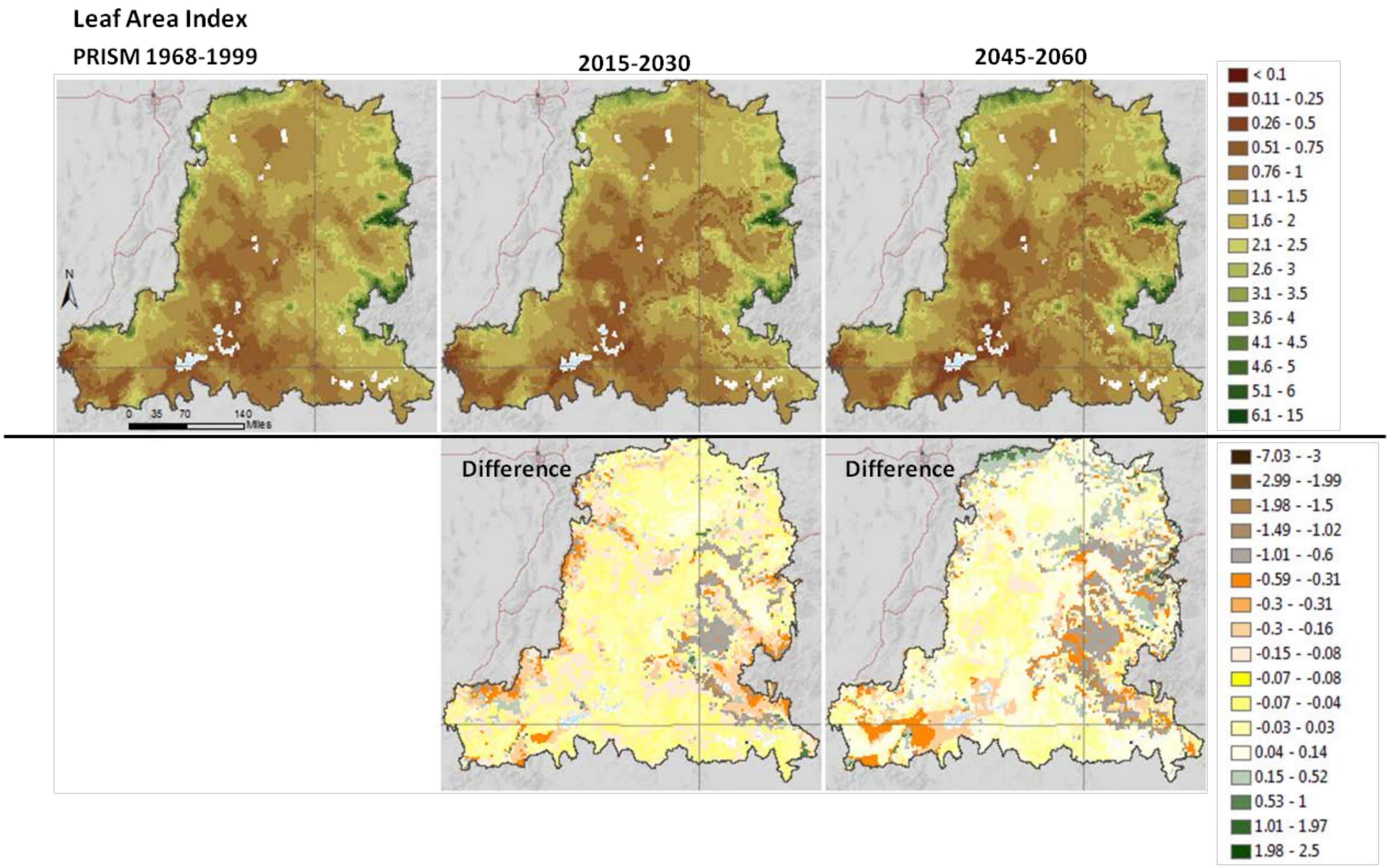


Figure 5-28. Map results for change in Leaf Area Index (LAI) based on MAPSS modeling for the Colorado Plateau ecoregion for 2015–2030 and 2045–2060.

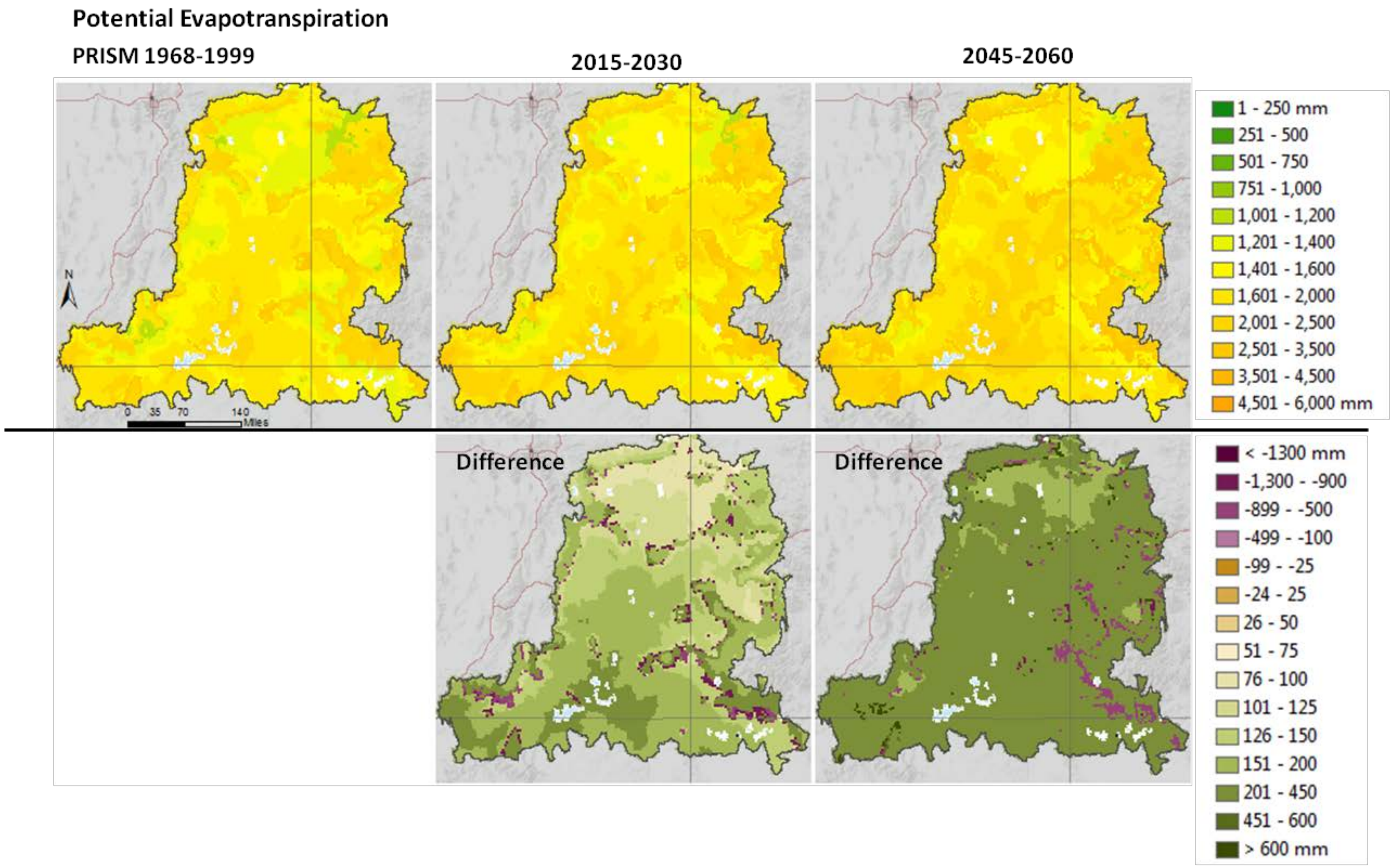


Figure 5-29. Map results for change in Potential Evapotranspiration (PET) based on MAPSS modeling for the Colorado Plateau ecoregion for 2015–2030 and 2045–2060.

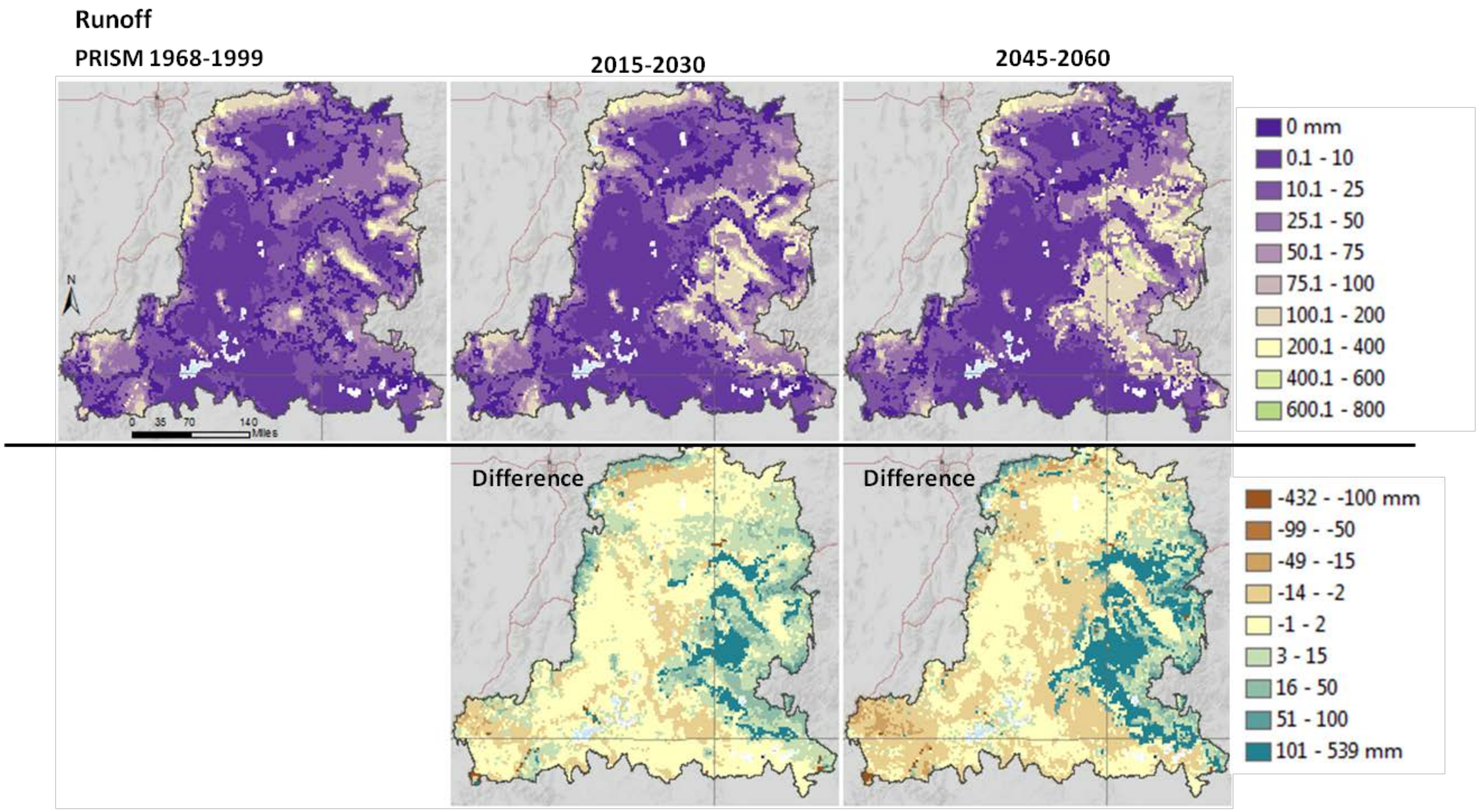


Figure 5-30. Map results for change in runoff based on MAPSS modeling for the Colorado Plateau ecoregion for 2015–2030 and 2045–2060.

Table 5-2. Change in major vegetation type (in 1000s of acres) according to MAPSS modeling for the Colorado Plateau ecoregion.

PRISM	2045–2060	Potential Change	Vegetation Type	Example Species
1,317	1,095	-221	Forest Evergreen Needle Continental	Douglas-fir, lodgepole pine
0	71	71	Tree Savanna Deciduous Broadleaf	Gambel oak
5,844	5,353	-490	Tree Savanna Evergreen Needle Continental	ponderosa pine
0	127	127	Tree Savanna Mixed	mixed hardwoods, pines
7,089	6,286	-803	Tree Savanna PJ Continental	pinyon pine, western juniper
4	225	221	Shrub Savanna Deciduous Broadleaf	blackbrush, greasewood, sagebrush
29,593	22,319	-7,275	Shrub Savanna Evergreen	sagebrush, saltbrush
1,139	2,432	1,293	Shrubland Subtropical Xeromorphic	Gambel oak, western juniper
43	134	91	Shrubland Subtropical Mediterranean	mountain mahogany
0	4	4	Grass Tall C3	Canada wildrye, needleandthread grass
16	154	138	Grass Mid C3	bluebunch and thickspike wheatgrass, Indian ricegrass
4	51	47	Grass Short C3	Sandberg bluegrass
0	12	12	Grass Tall C3C4	wheatgrass, spike dropseed
28	162	134	Grass Mid C3C4	wheatgrass, ricegrass
257	2,167	1,910	Grass Short C3C4	bluegrass, grama
0	8	8	Grass Mid C4	sideoats grama, James' galleta
158	4,290	4,132	Grass Short C4	sandhill muhly, blue grama
482	1,091	609	Grass Semi Desert C4	blue grama
20	12	-8	Desert Subtropical	creosotebush

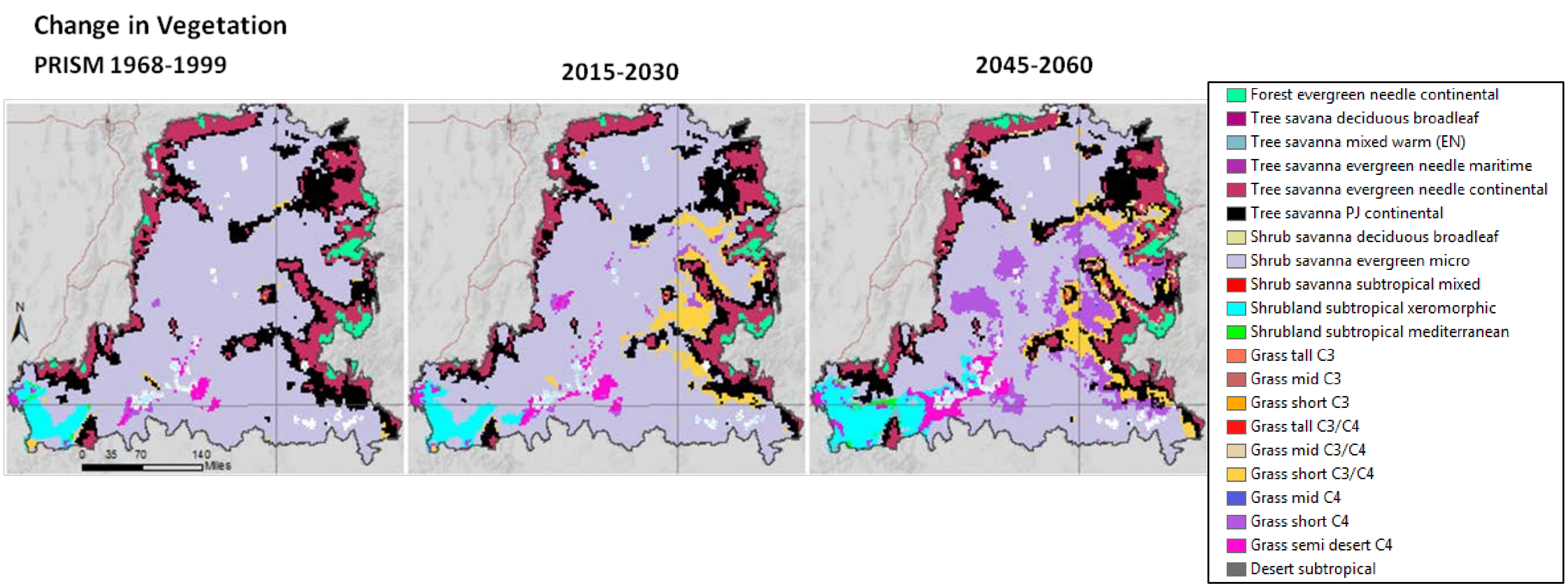


Figure 5-31. Map results for change in vegetation based on MAPSS modeling for the Colorado Plateau ecoregion from historic baseline (1968–1999) to 2015–2030 and 2045–2060.

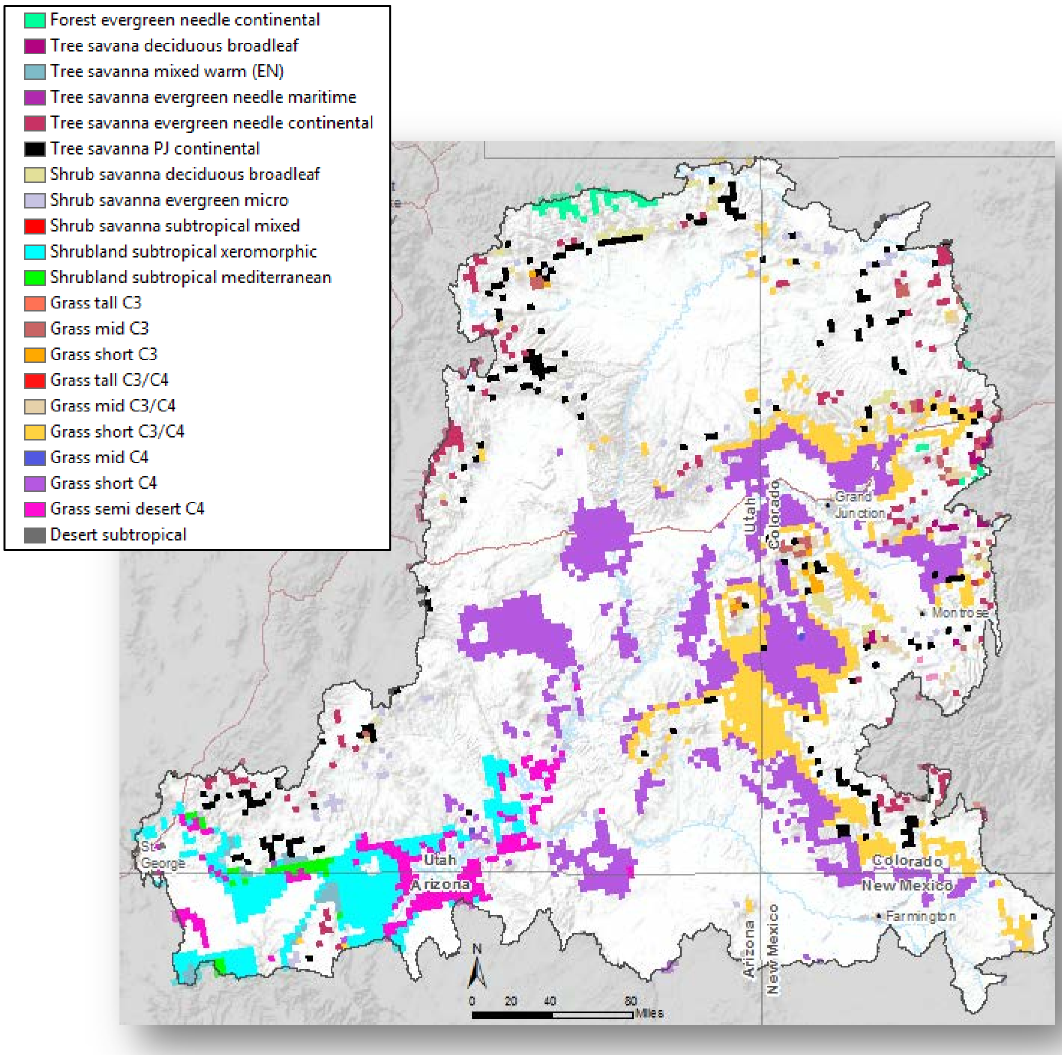


Figure 5-32. Map results showing just the pixels that changed to different vegetation types between historic baseline (1968–1999) and 2045–2060 based on MAPSS modeling for the Colorado Plateau ecoregion.

Potential vegetation change depicted here is simulated using climatic and soil information. The results do not mean the potential vegetation type will necessarily be established during a particular time period, only that climate conditions would be optimal for their development there at that time period if seed sources were available and human intervention did not occur to destabilize soils or modify its hydrological properties. Many other factors will affect future vegetation type such as human-caused fire, invasive species introduction, or dispersal factors. The projections may also indicate trends where vegetation mortality may occur.

The RegCM3 climate model projects increasing temperatures in all seasons and for both time scenarios. For 2015–2030, the model shows less precipitation annually, in winter and especially in summer (reduction in the monsoon); for 2045–2060, the model shows a slight increase in annual precipitation particularly during winter months. Regional differences can be found such as an increase in summer (monsoonal) precipitation in Utah and an increase in winter precipitation in Colorado. Winter precipitation is critical to perennial native plants and biological soil crusts and it enhances annual productivity especially for C_3 plants. If both winter and summer precipitation is reduced, trees, especially pinyon pine and biological soil crusts may be the biggest losers in this century (Schwinning et al 2008) while shrubs (e.g. blackbrush) are likely to continue to expand (Munson et al 2011).

The extremely warm and dry period in 2015–2030 may exacerbate the decline of native C_3 perennial grasses (Munson et al, 2011) and cause some tree mortality in many areas while favoring semidesert grasses in Western Colorado. By 2060 the model's big losers are some of the drier shrublands (sagebrush in particular), savanna pinyon-juniper, and some evergreen forest. Gains are expected in the grasses, especially short C_4 and short C_3C_4 mixes. Elevated CO_2 is expected to slightly favor C_3 over C_4 plants, which are less sensitive to warmer temperatures, and thus the increased projections in CO_2 may mitigate some of the effects of projected warming on the competitive advantage of C_4 grasses. This was confirmed in a recent experiment with C_3 and C_4 semi-arid steppe grasses (Morgan et al. 2011). Morgan et al. (2011) found that elevated CO_2 favored C_3 grasses and warming favored C_4 grasses. They also noted that the *combination* of warming and CO_2 enrichment stimulated the growth of C_4 grasses. Their overall results indicated that productivity in semi-arid grasslands may be higher than previously projected under climate change scenarios.

For both the 2015–2030 and 2045–2050 time periods, the seasonality and intensity of precipitation will be a key factor. Biological soil crusts are vulnerable to an increased frequency of summer rains (Belnap et al. 2004). Summer rain creates short wet-dry cycles that result in carbon and nitrogen losses to soil crusts and that could cause increased mortality or extirpations of some crust species (Evans et al. 2001). If the trend is toward wetter winters or springs, the invasive C_3 grasses such as cheatgrass or red brome will spread and will burn in the summer and fall, reinforcing their persistence over larger areas. If multiple wet years occur, grasses may have the advantage over shrubs in establishment and survival (Peters 2011).

In summary, land managers should begin to prepare for changes in the present ecoregion character (and boundaries) that could be expressed as shifts in vegetation composition, diversity and growth, declines in net primary production, intensification of the hydrologic cycle (more intense runoff), reductions in streamflow, declines in native fish diversity, increases in soil erosion, increases in nonnative species populations, and increased frequency and intensity of fire (Archer et al, 2008).

5.4.1.2 Uncertainty

Uncertainty can be examined in different ways and from different perspectives. First, impacts models depend on the reliability of the climate data that they use. It is important to note that while climate projections diverge after 2040, models generally agree for the first half of the century and the choice of a particular climate model or scenario is less important if the management goal is limited to the next 2 or 3 decades. Beyond 2040, it becomes critical to rely upon experts who can select climate models based on less than perfect criteria. For example, it is common to choose climate models that best simulate past climate dynamics, particularly paying attention to the most important local climate feature (as was done for this REA with the choice of the RegCM3 model that recognizes the summer monsoon for the U.S. Southwest). Three GCMs driven by the RegCM3 regional model were analyzed for this project: ECHAM-5, GFDL and GENMOM. The data portal contains the results of each model, including associated MAPSS results. Users can delve into these models to gain a deeper understanding of the range of potential results from various models.

Model verification is obviously impossible for future projections and one is reduced to putting one's confidence in the ability of climate models to reproduce faithfully past climatic changes. However, there is no guarantee that a model that reproduces the past well will simulate the future accurately. Current models include our current understanding of past climate dynamics that may change drastically as atmospheric and stratospheric composition change as well as the planet's albedo. General circulation models (GCMs) were designed to simulate the planet's climate and their results compare well to climate observations at the global scale. The accuracy of global models declines at the local scale due to their inherent coarse spatial resolution that averages diverse vegetation cover and complex topography so important to conservation practitioners. Downscaling techniques (statistical or dynamic) bring GCM results to the scale of concern, but their 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 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 that constitutes the "known unknowns" of the climate system. Finally, while surprises have been projected for the biological world, as the climate is changing, surprises such as the unexpected Larsen B ice shelf rapid collapse, the "unknown unknowns", also bring climate scientists back to the drawing board to improve existing models. It is important to understand that as change occurs (e.g. ice free poles, glaciers disappearance, new wind patterns, change in ocean currents), the basic assumptions at the core of the climate models may become obsolete, reminding us again that there is no assurance that a model that reproduces the past well is going to be reliable when projecting the future. Climate scientists learn constantly from every new observation and they update their models accordingly as new observations bring new knowledge.

Extreme events (e.g. long, intense droughts, flood, hurricanes or typhoons) are also difficult to predict by general circulation models. The 2007 report from the IPCC warns about the increased risk of more intense, more frequent and longer-lasting heat waves as exemplified by the European heat wave of 2003 that killed several thousand people. Along with a greater risk of drought, there is an increased chance of intense precipitation and flooding due to the greater water-holding capacity of a warmer atmosphere such that both wet and dry extremes should become more severe. These extreme events, while unpredictable, are often what shape our landscapes. Past extreme events such as the drought of the 1930s that caused the Dust Bowl in the USA certainly affected natural ecosystems and human land use. Recently, records of extreme events have been increasing. For example, the drought of 1999–2002 that spawned fires, dust storms, and pinyon pine mortality across the southwestern states may have been an indication of climate destabilization. A drought in the summer of 2010 that caused crop failure and huge fires in Russia occurred at the same time as record rainfall that caused extensive flooding and loss of lives in both China and Pakistan. These extremes are consistent with what climate scientists have been expecting. Extreme events certainly pose a challenge to land managers who are typically more comfortable thinking about chronic linear change rather than abrupt and unpredictable change. Yet these events should be considered when trying to prepare for change. In the past, the reliability of models was tested in part by simulating large disturbances and observing the simulated system's response. It may be of interest to practitioners to focus now more on disturbance simulation to fully explore the resilience of their system.

There is also inherent natural variability in the expression of climate (e.g. cold air drainage, inversions in deep valleys), which is often influenced by the complexity of the regional terrain. At a fine scale, this means localized climate refugia–narrow swales, moist draws, etc. Close examination of a reasonable resolution (30 m) digital elevation model (DEM) can provide some insight as to locations that are more likely to provide refugia (Figure 5-33). These sites are found at a much finer scale than the analytical grid of the climate change work. At a coarser level, places on the landscape in and around rugged terrain will experience higher natural levels of climate variability.

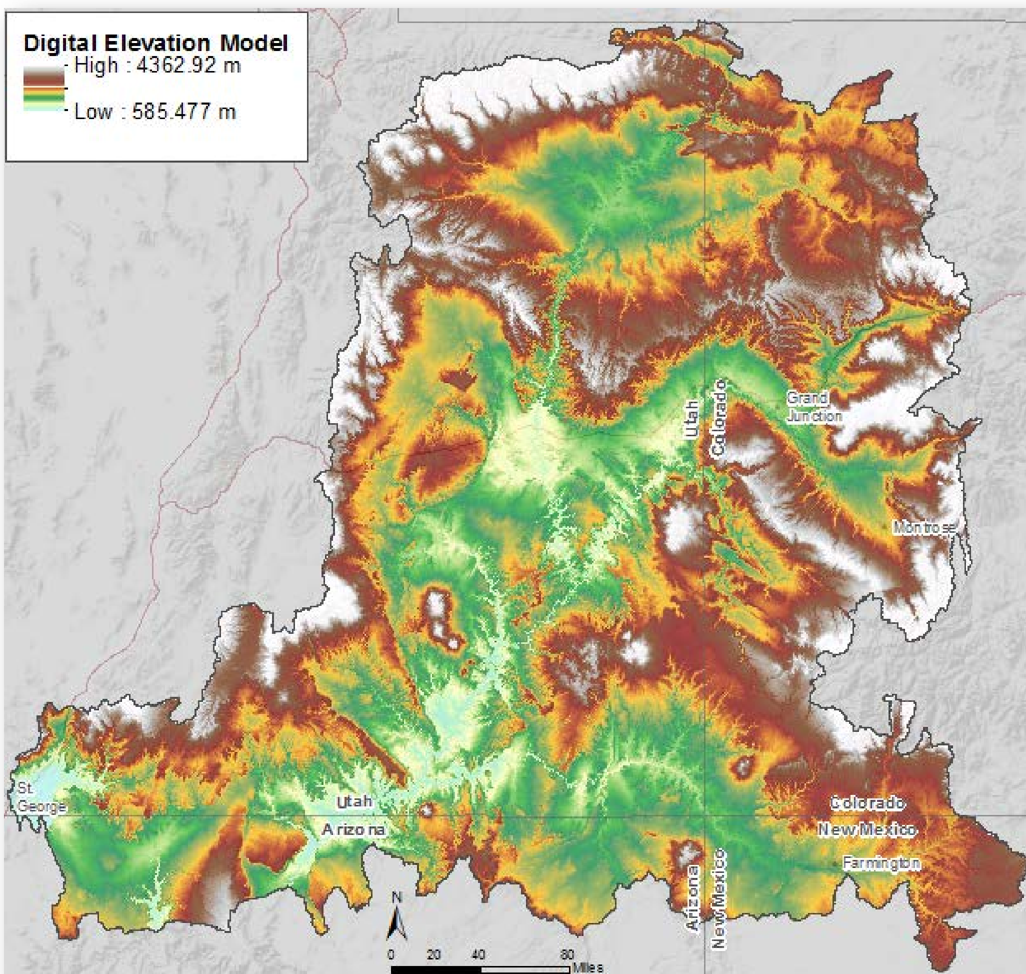


Figure 5-33. Digital elevation model (DEM) for the Colorado Plateau ecoregion.

Calculating the pixel standard deviation of annual average temperature and annual average precipitation separately based on the PRISM historic climate data provides map products that highlight areas on the landscape that are prone to more variability for these primary climate variables (Figure 5-34). The natural variability of precipitation for this arid landscape is quite small at lower elevations, but the range of variability increases to a modest degree as elevation increases. The range of variability is far more pronounced for the temperature data. Here, the valleys express higher levels of temperature variability from year-to-year (areas that are orange). These areas are highly influenced by the close proximity of the various mountains and benches. These results allow us to state two things. First, plants and animals living in areas that are more naturally variable in their climate have evolved mechanisms to help cope in that setting. It also suggests that the climate forecasts in these areas will tend to be less reliable compared to other locations in the region.

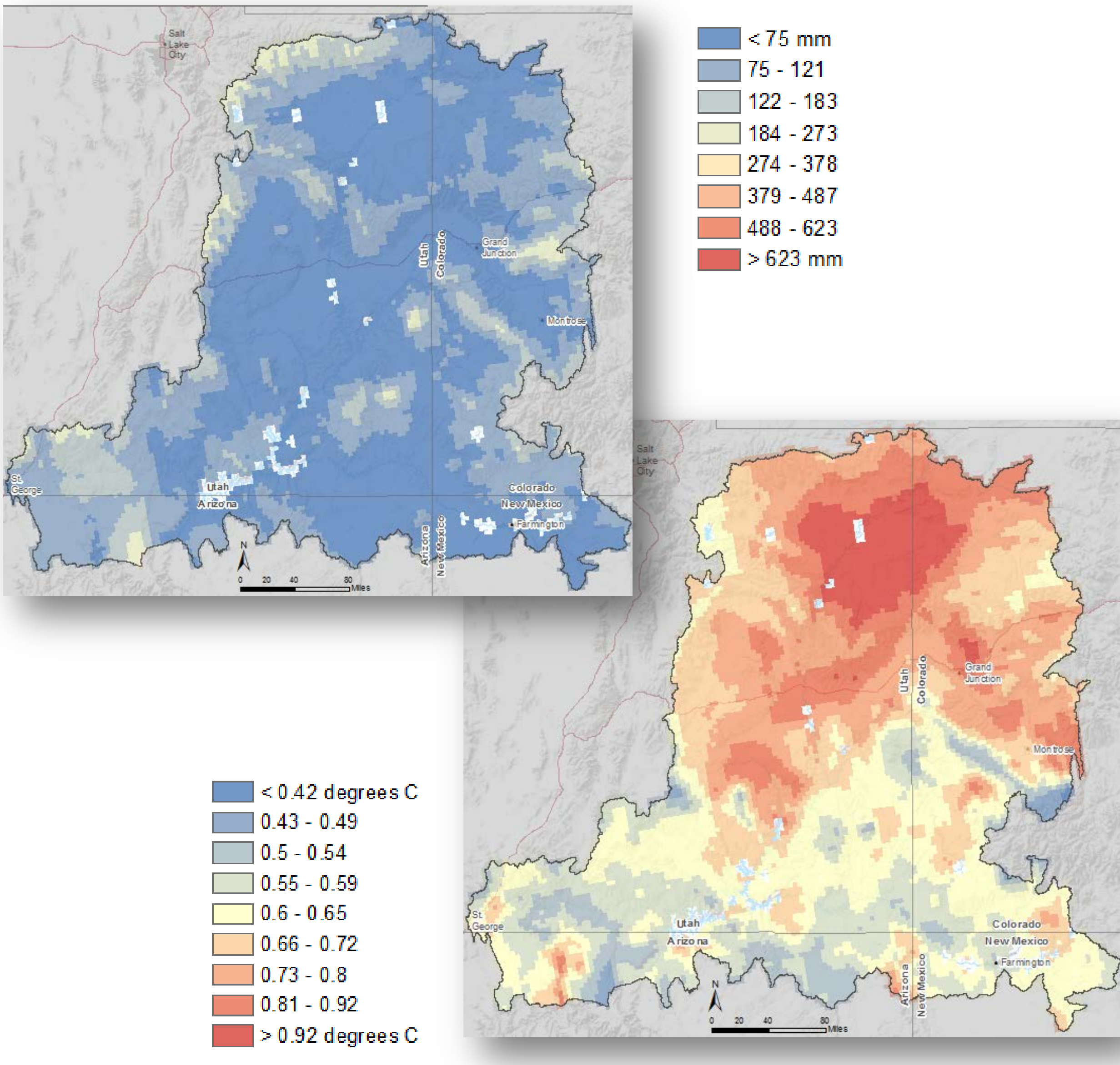


Figure 5-34. Uncertainty depicted as standard deviation of precipitation (A) and temperature (B) data from PRISM historic conditon (1968–1999).

5.4.1.3 Assessing Conservation Elements' Exposure to Climate Change

To simplify the numerous future climate projections and MAPSS modeling results, a number of key findings from these analyses were selected and assembled into an overall relative climate change map. The different classes of potential for climate change were then overlaid on the distributions of specific conservation elements to assess the relative impact. The fuzzy model inputs included potential for summer temperature change and potential for winter temperature change averaged into a single factor, potential for runoff change from MAPSS modeling, potential for precipitation change, and potential for vegetation change again from MAPSS modeling (Figure 5-35). For this purpose the, direction of the change was not important—only degree of departure from historic measures. Details regarding change in temperature by degrees or actual predicted changes in precipitation can easily be assessed from the additional datasets provided in the body of the text or in the appendices. The model logic states that all 4km pixels with potential to change in primary vegetation type get the highest change score with the average of the other factors filling in the rest of the landscape.

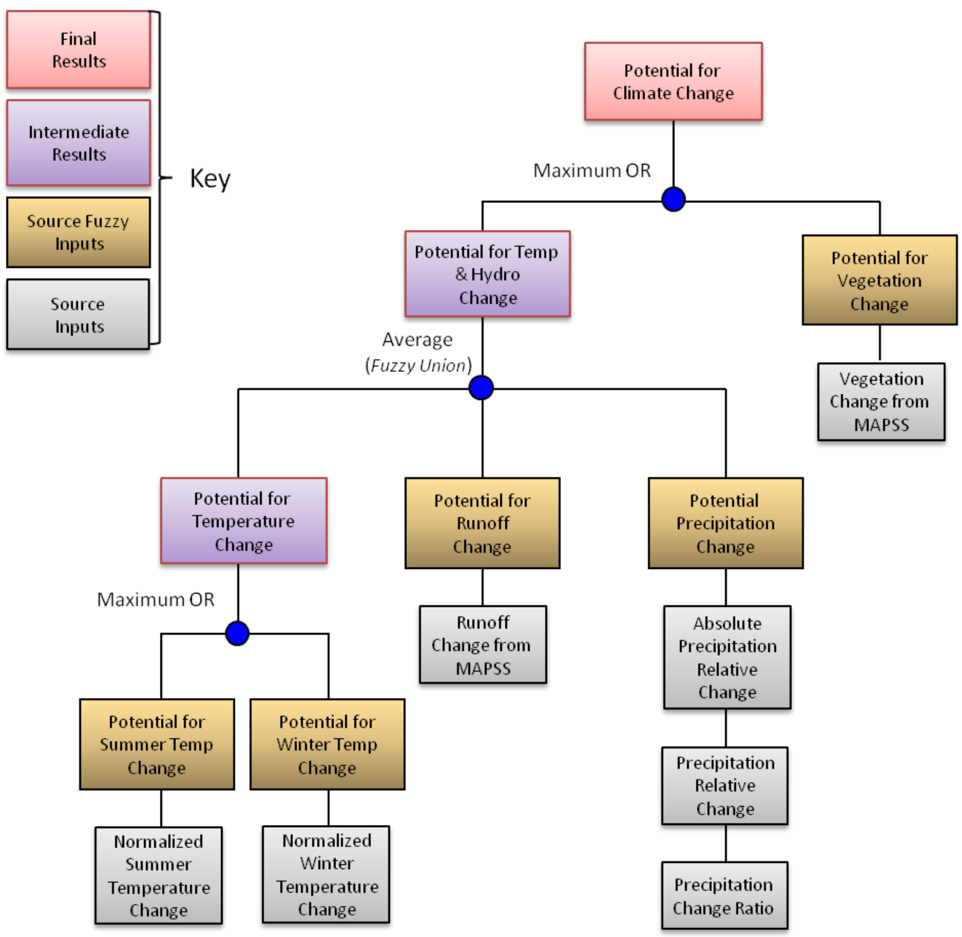


Figure 5-35. Fuzzy logic model for integrating climate change impacts to assess potential exposure of conservation elements to climate change.

Resultant maps from the fuzzy logic model show the contributions made by the various components (Figure 5-36). Areas likely to show the most change are those that either are predicted to change in their vegetation type or as a combination of all the other factors.

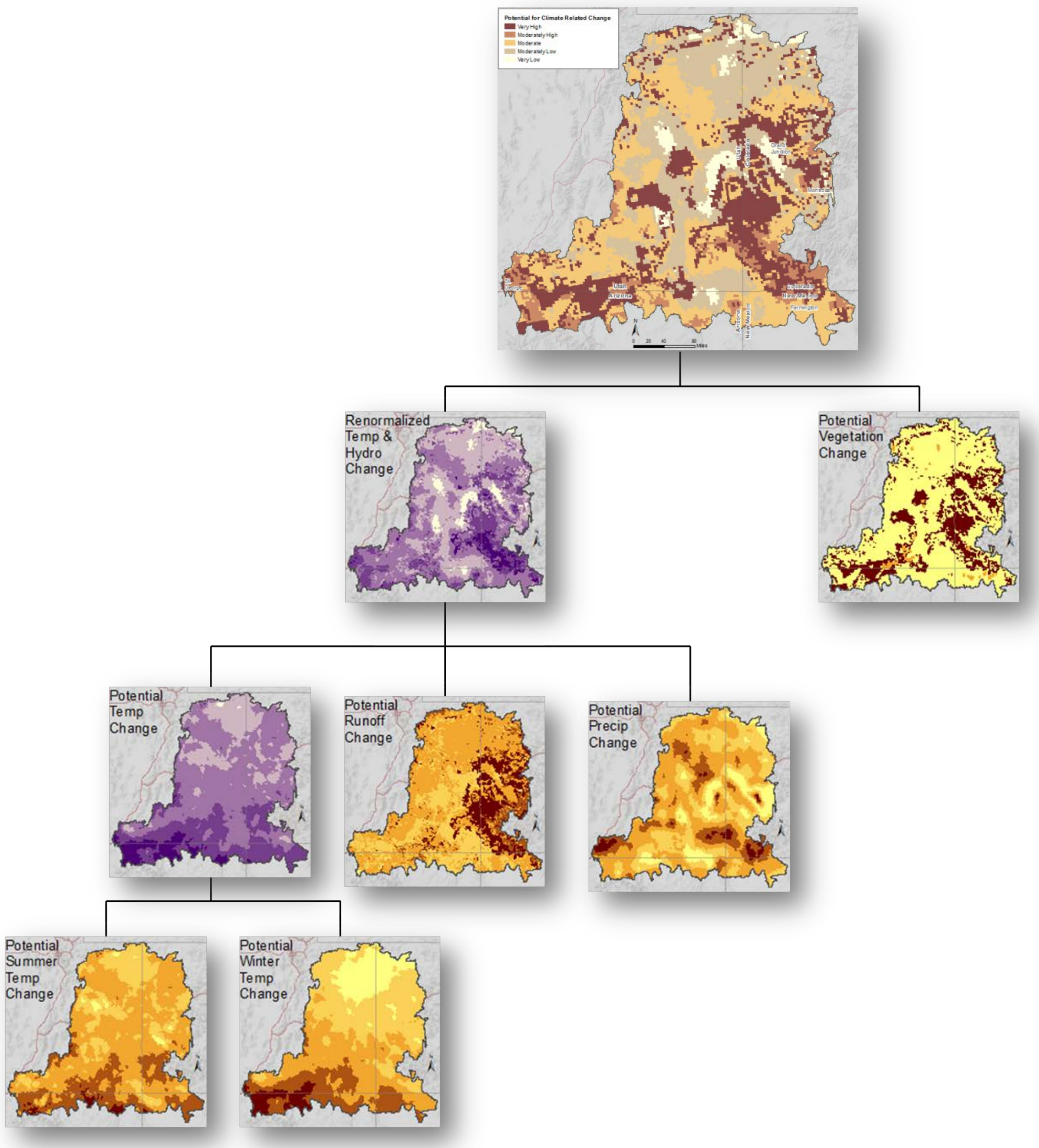


Figure 5-36. Map outputs for each step in the climate change fuzzy logic model for the Colorado Plateau ecoregion.

The climate change model results were then overlaid with species' and vegetation communities' distribution maps to assess their exposure to climate change. Exposure is just one aspect of ecosystem and species' vulnerability to climate change. Vulnerability is defined by the United Nations' Intergovernmental Panel on Climate Change (IPCC 2001) as…"(t)he degree to which a system is susceptible to, or unable to cope with, adverse effects of climate change, including climate variability and extremes. Vulnerability is a function of the character, magnitude, and rate of climate variation to which a system is exposed, [*as well as*] its sensitivity and its adaptive capacity." See also definition in Glick et al. (2011). The sensitivity of a species or system to climate change can be considered in terms of a "dose-response" relationship describing its exposure, resulting impacts, and its response (decline or adaptation, Füssel and Klein 2006). The development of vulnerability indices requires the implementation of species-specific indicators of sensitivity and species response or capacity to adapt, along with thresholds of impact that may lead to species decline (Carter et al. 2007). Füssel (2007) notes that time must be factored in as well; sensitivity represents immediate or short-term effects on a system or species, while resilience or adaptation must be considered over a longer time frame to assess or project the species' ability to maintain basic functions and possibly return to its original state. Although no readily-available metrics yet exist to quantitatively describe the vulnerability of an ecosystem or species to climate change (Füssel and Klein 2006, Adger 2006, Carter et al. 2007), the pressing need to identify vulnerable species and to manage for mitigation under various climate change scenarios has prompted the development of more qualitative approaches to project species' vulnerability (Glick et al. 2011, Young et al. 2011).

The REA climate change results presented here are modeled from available spatial data and focus mostly on exposure. However, some non-spatial sensitivity information was also available for many of the REA wildlife and vegetation community conservation elements from a Climate Change Vulnerability Index (CCVI) developed for the Nevada/Mojave region and the Gunnison Basin in western Colorado (NNHP 2011, TNC 2011). CCVI is a product of assessment teams employing literature review, professional judgment, and expert review through workshops (Young et al. 2011). All of the REA wildlife species conservation elements were classified in the two available CCVIs as *Presumed Stable*, except for Gunnison sage-grouse, greater sage-grouse, and Colorado River cutthroat trout, all of which were listed as *Highly Vulnerable* to significant decreases in abundance or distribution with projected climate change (NNHP 2011, TNC 2011). Razorback sucker was in the class *Increase Likely* within the Nevada/Mojave region. Five of our REA species were not listed: mountain lion, white-tailed prairie dog, black-footed ferret, pronghorn, and Mexican spotted owl. Vegetation communities were assessed only for the Gunnison Basin (of which the lower portions through the pinyon-juniper community occur within the Colorado Plateau ecoregion); of these, montane sage is in the category *Moderately Increase* and the big sage and pinyon-juniper communities are in the *Presumed Stable to Moderately Increase* categories (TNC 2011).

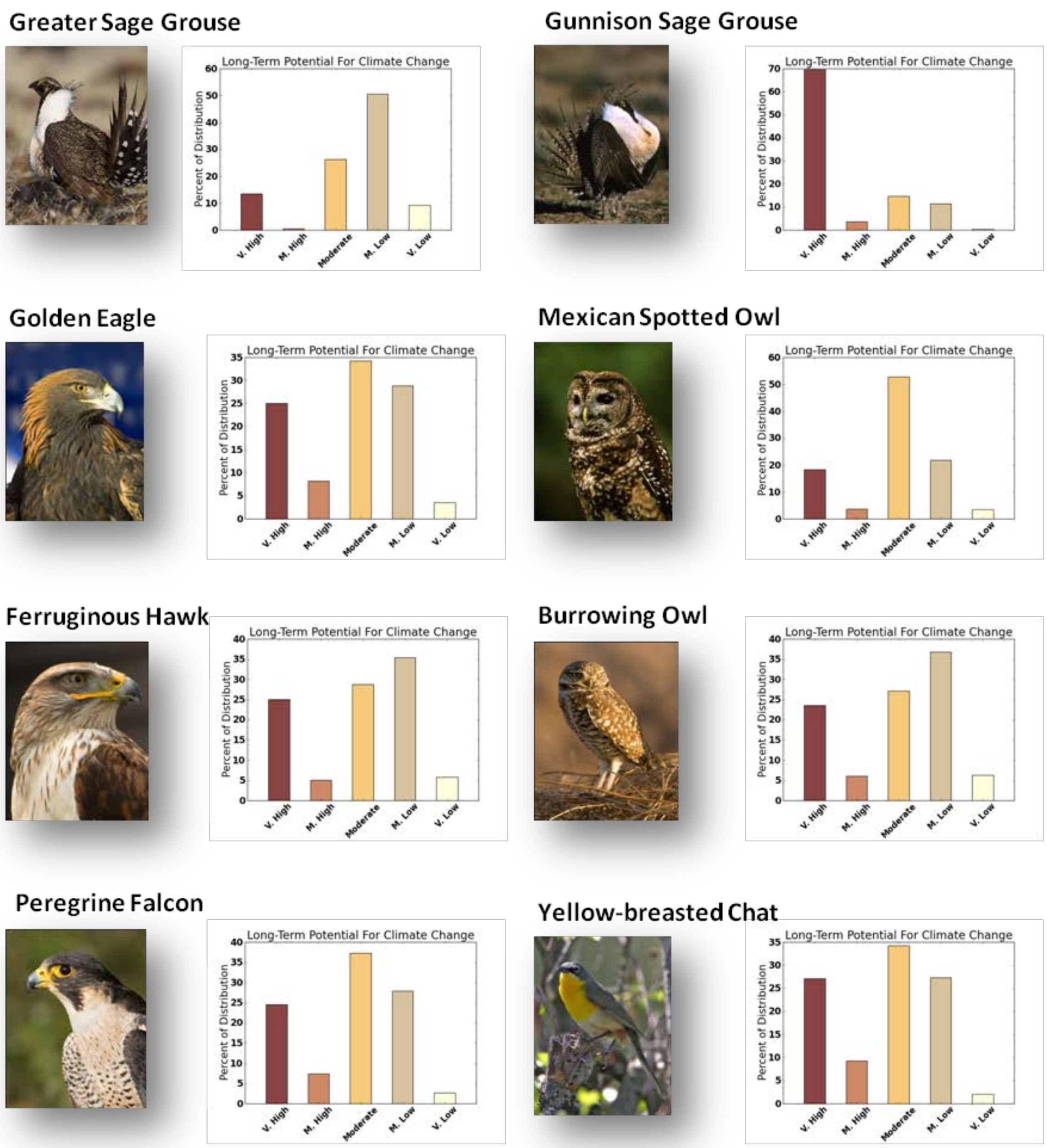
For the body of this report, results were posted in histograms as five climate change exposure classes for 2060: Very High, High, Moderate, Moderately Low and Low for the potential for an area to be affected by climate change as defined in the fuzzy logic model. One can portray this aspect of potential vulnerability (exposure) by overlaying the conservation element's distribution map (Section 4.2) with the climate change exposure map (top map in Figure 5-36). An overlay map for each conservation element relative to climate change exposure can be found in Appendices B and C; the maps and data may also be examined in greater detail on the data portal (access at http://www.blm.gov/wo/st/en/prog/more/climatechange.html). Bringing additional species sensitivity information to this analysis, such as that for the Highly Vulnerable species listed above, will allow the identification of locations where the species may experience various degrees of vulnerability to climate change as well as locations of possible refugia.

Each of the mammal species showed a unique signature to the climate model results (Figure 5-37). Gunnison's prairie dog showed the highest exposure of any of the mammals with 70% of its current distribution in the Very High category. The severely-limited black-footed ferret population scored low in exposure; however, when considering its potential reintroduction, the profile for the white-tailed prairie dog is just as important; it showed somewhat greater potential impacts from both climate change and the potential for energy development discussed in Section 5.2. Desert bighorn sheep, inhabiting somewhat higher elevations, is in a lower exposure category. Mountain lion and two prey species (especially mule deer) showed similar potential exposure with about a third of their populations under Very High or High climate change conditions.

Figure 5-37. Potential exposure to climate change for mammals of the Colorado Plateau ecoregion. Note: Percent distribution (y-axis) differs for each species. Gunnison's prairie dog showed the highest exposure of any of the mammals with 70% of its current distribution in the Very High category.

The bird species also showed species-specific patterns (Figure 5-38). Gunnison sage-grouse, like its prairie dog companion, is the species with the highest exposure with about 70% of its current distribution affected by climate change by 2060. Yellow-breasted chat, a riparian species, also showed high exposure to changing climate in the region. Greater sage-grouse, on the other hand, showed a very different pattern with most of its current populations in the lower exposure classes. Mexican spotted owl fared better than the remaining bird species, and all of the others shared a similar profile expressing only subtle differences. Like the mammals, around a third of the distributions for these species are predicted to be under Very High or High climate change exposure in the mid-term future.

Figure 5-38. Potential exposure to climate change for birds of the Colorado Plateau ecoregion. Note: Percent



distribution (y-axis) differs for each species.

Of the fish species, flannelmouth sucker showed the highest exposure from climate change compared to the other two species (Figure 5-39). Colorado River cutthroat trout experienced somewhat less exposure than flannelmouth sucker with almost 30% in the Very High and Moderately High classes. Wenger et al. (2011), using a composite model of multiple climate projections, found a decline in stream length with suitable habitat for Colorado cutthroat of 28% in the 2040s and 58% in a 2080s scenario from a combination of temperature increases and interactions with other introduced trout species. The authors estimate that interactions with introduced species decrease current cutthroat stream length by 33% and future 2080s scenario stream length by 26%. See more details on fish species in Appendix C.

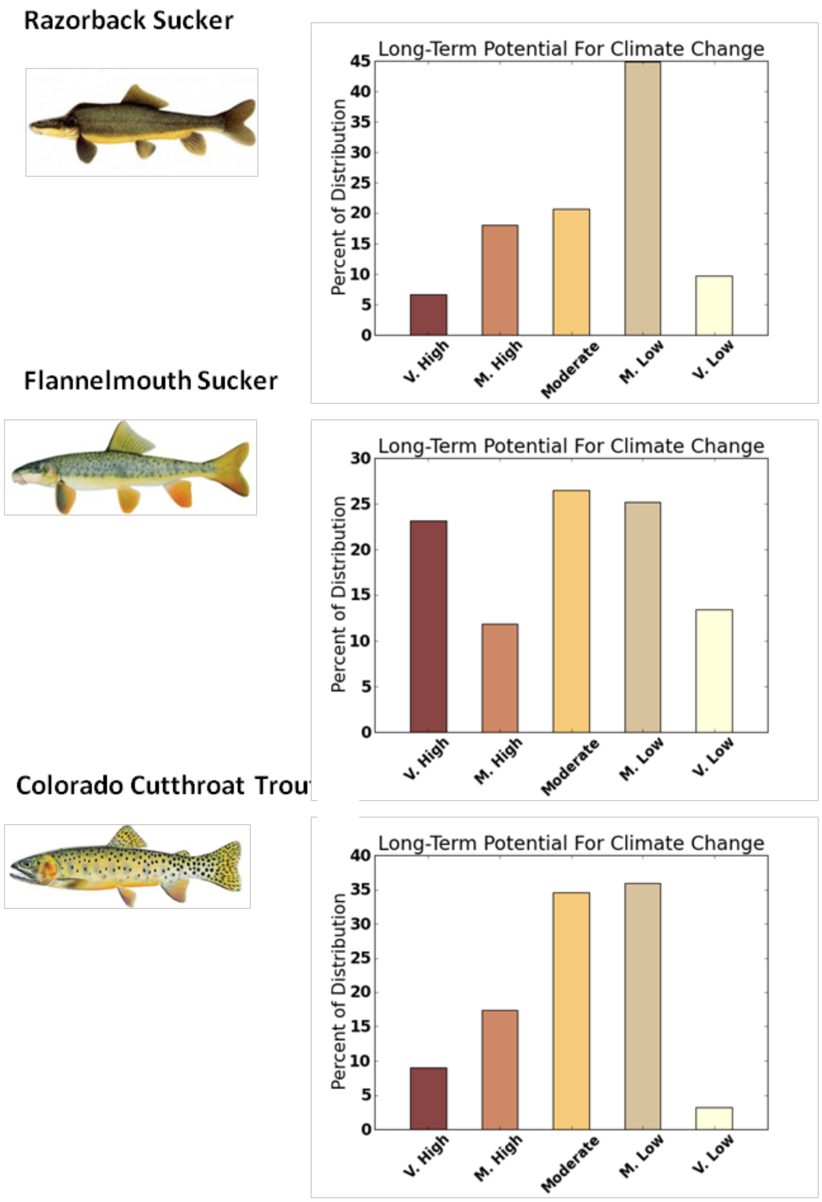


Figure 5-39. Histograms show potential exposure to climate change for the fishes of the Colorado Plateau ecoregion. Of the fish species, flannelmouth sucker showed the highest potential exposure to climate change compared to the other two species.

The vegetation communities (Figure 5-40) showing the most area under high climate change exposure include the shrublands (especially big sagebrush and blackbrush-Mormon-tea communities), riparian vegetation, and pinyon-juniper woodland, which is consistent with the MAPSS results presented in Table 5-2. With the vegetation communities, note that when interpreting these results high exposure does not definitively mean decline; it means higher probability of change. Insects and disease will play a collateral role with the effects of climate change in altering the dominance and distribution of various vegetation species.

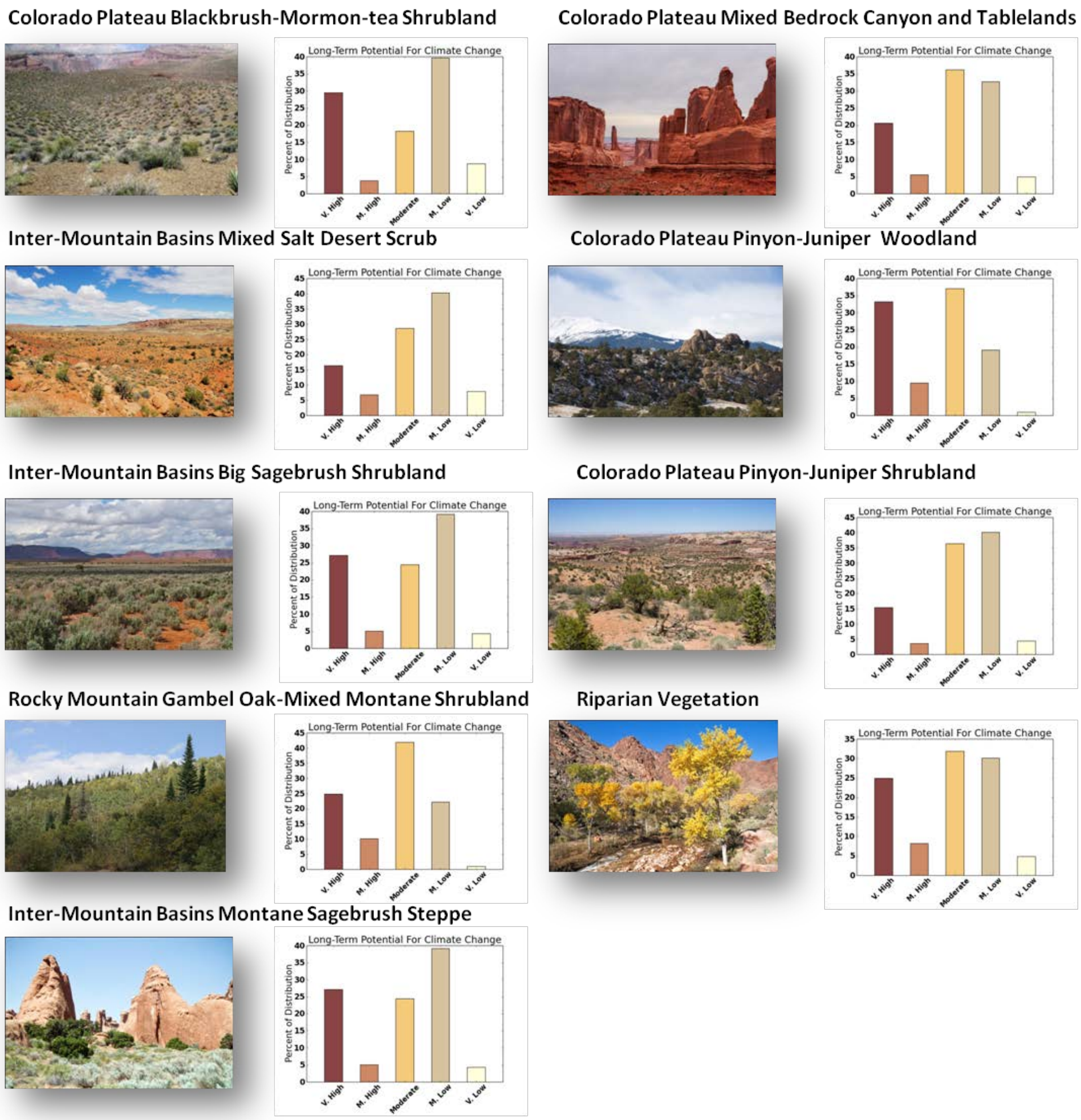


Figure 5-40. Histograms show potential exposure to climate change for the vegetation communities of the Colorado Plateau ecoregion. The big sagebrush and blackbrush-Mormon-tea communities show the most area under high climate change exposure in addition to riparian vegetation and pinyon-juniper woodland, which is consistent with the MAPSS results presented in Table 5-2.

Finally, existing designated sites show fairly high exposure to climate change by 2060 with 30% of land area under Very High or High and nearly another 40% under Moderate vulnerability (Figure 5-41).

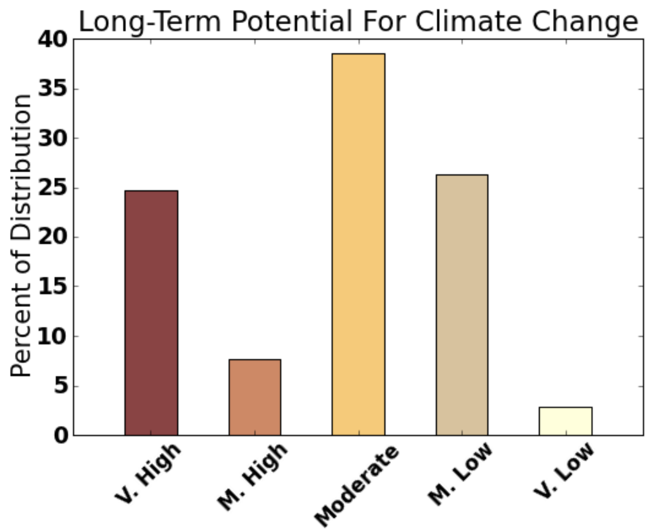


Figure 5-41 Potential exposure to climate change for the designated sites of the Colorado Plateau ecoregion with 30% of land area under Very High or High and nearly another 40% under Moderate potential for climate change.

5.5 References Cited

Adger, W.N. 2006. Vulnerability. *Global Environmental Change* 16(3): 268–281.

Archer, S.A. and K.I. Predick. 2008. Climate change and ecosystems of the Southwestern United States. *Rangelands* 30(3): 23-38.

Belnap, J., S. L. Phillips and M. E. Miller (2004). Response of desert biological soil crusts to alterations in precipitation frequency. *Oecologia* 141(2): 306-316.

Carter, T.R., R.N. Jones, X. Lu, S. Bhadwal, C. Conde, L.O. Mearns, B.C. O'Neill, M.D.A. Rounsevell, and M.B. Zurek. 2007. New assessment methods and the characterisation of future conditions. Pages 133–171 *in* Parry, M.L., O.F. Canziani, J.P. Palutikof, P.J. van der Linden, and C.E. Hanson (eds.), Climate change 2007: Impacts, adaptation and vulnerability, Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, Cambridge University Press, Cambridge, United Kingdom.

Copeland, H.E., K.E. Doherty, D.E. Naugle, A. Pocewicz, and J.M. Kiesecker. 2009. Mapping oil and gas development potential in the US intermountain west and estimating impacts to species. PLoS ONE 4(10): e7400.

Doherty, K. 2008. Sage-grouse and energy development: Integrating science with conservation planning to reduce impacts. PhD dissertation, University of Montana, Missoula. 125 pp.

Evans, R.D., J. Belnap, F. Garcia-Pichel, and S.L. Phillips. 2001. Global change and the future of biological soil crusts. Pages 417–429 *in* Belnap, J., and O.L. Lange (eds.), Biological soil crust: Structure, function, and management. Springer, New York.

Füssel, H.M. 2007. Vulnerability: A generally applicable conceptual framework for climate change research. *Global Environmental Change* 17:155–167.

Füssel, H.M., and R.J.T. Klein. 2006. Climate change vulnerability assessments: An evolution of conceptual thinking. *Climatic Change* 75 (3):301–329.

Garfin, G., J. Eischeid, M. Lenart, K. Cole, K. Ironside, and N. Cobb. 2010. Downscaling climate projections to model ecological change on topographically diverse landscapes of the arid southwestern United States. Pages 21–44 *in* Van Riper, C., III, B.F. Wakeling, and T.D. Sisk (eds.), The Colorado Plateau IV, Proceedings of the 9th Biennial Conference on Colorado Plateau Research, October, 2007. University of Arizona Press, Tucson.

Glick, P., B.A. Stein, and N.A. Edelson. 2011. Scanning the conservation horizon: A guide to climate change vulnerability assessment. National Wildlife Federation, Washington D.C.

Hostetler, S., J. Alder, and A. Allan. 2011. Dynamically downscaled climate simulations over North America: Methods, evaluation, and supporting documentation for users. Open-File Report 2011–1238, U.S. Geological Survey, Reston, Virginia.

IPCC (Intergovernmental Panel on Climate Change). 2001. Climate change 2001: Synthesis report. A Contribution of Working Groups I, II, and III to the Third Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom.

Jarnevich, C.S., P. Evangelista, T.J. Stohlgren, and J. Morisette. 2011. Improving national-scale invasion maps: Tamarisk in the western United States. *Western North American Naturalist* 71(2):164–175.

Morgan, J.A., D.R. LeCain, E. Pendall, D.M. Blumenthal, B.A. Kimball, Y. Carrillo, D.G. Williams, J. Heisler-White, F.A. Dijkstra, and M. West. 2011. C_4 grasses prosper as carbon dioxide eliminates desiccation in warmed semi-arid grassland. *Nature* 476: 202–206.

Mote, P.W., D. Gavin, and A. Huyer. 2010. Climate change in Oregon's land and marine environments. Chapter 1 in Dello, K.D., and P.W. Mote (eds.), Oregon Climate Assessment Report. College of Oceanic and Atmospheric Sciences, Oregon State University, Corvallis, Oregon.

Munson, S., J. Belnap, C. Schelz, M. Moran, T. Carolin. 2011. On the brink of change: Plant responses to climate on the Colorado Plateau. *Ecosphere* 2(6), Article 68. digitalcommons.usu.edu/cgi/viewcontent.cgi?article=1340

NNHP (Nevada Natural Heritage Program). 2011. Climate Change Vulnerability Index (Release 2.01). Nevada Natural Heritage Program, Carson City, Nevada.

Peters, D.P.C., J. Yao, O.E. Sala, and J. Anderson. 2011. Directional climate change and potential reversal of desertification in arid and semiarid ecosystems. *Global Change Biology* (in press) DOI: 10.1111/j.1365-2486.2011.02498.x.

Schwinning, S., J. Belnap, D.R. Bowling, and J.R. Ehleringer. 2008. Sensitivity of the Colorado Plateau to change: Climate, ecosystems, and society. *Ecology and Society* 13(2): 28. http://www.ecologyandsociety.org/vol13/iss2/art28/

Seager, R., M. Ting, I. Held, Y. Kushnir, J. Lu, G. Vecchi, H. Huang, N. Harnik, A. Leetmann, N. Lau, C, Li, J. Velez, and N. Naik. 2007. Model projections of an imminent transition to a more arid climate in southwestern North America. *Science* 316:1181–1184.

TNC (The Nature Conservancy). 2011. Gunnison Basin preliminary climate vulnerability assessment for the Gunnison Climate Working Group. The Nature Conservancy, Southwest Climate Change Initiative, Colorado Natural Heritage Program, Fort Collins, Colorado.

Theobald, D.M. 2010. Estimating natural landscapes changes from 1992 to 2030 for the conterminous United States. *Landscape Ecology* 25(7):999–1011.

Van Dyke, F.G., R.H. Brocke, and H.G. Shaw. 1986. Use of road track counts as indices of mountain lion presence. *Journal of Wildlife Management* 50(1):102–109.

Wenger, S.J., D.J. Isaak, C.H. Luce, H.M. Neville, K.D. Fausch, J.B. Dunham, D.C. Dauwalter, M.K. Young, M.M. Elsner, B.E. Rieman, A.F. Hamlet, and J.E. Williams. 2011. Flow regime, temperature, and biotic interactions drive differential declines of trout species under climate change. *Proceedings of the National Academy of Sciences,* Published online before print August 15, 2011, doi: 10.1073/pnas.1103097108.

Young, B., E. Byers, K. Gravuer, K. Hall, G. Hammerson, and A. Redder. 2011. Guidelines for using the NatureServe Climate Change Vulnerability Index, Release 2.1. NatureServe, Arlington, VA.

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Photo: Four Corners Powerplant from Lake Powell, National Park Service.

VI. Summary Findings and Applications

This chapter presents REA findings designed to help managers visualize the REA products and how they may be used at various scales (ecoregional, state, field office). The focus of this example is on BLM lands not currently protected, but the models are flexible enough to analyze all areas at the ecoregion, state, or field office scales. This summary presents ways to use the integrity/intactness results with composite species information to provide an overview of key regional issues as an introduction to more local step-down management or planning. Understanding the relationship of these data provides basic ecoregion-level information to begin to identify broad areas of opportunity for development, restoration, conservation, or connectivity that may be examined at multiple scales, both regional and local.

6.1 Using REA Results for Regional Planning

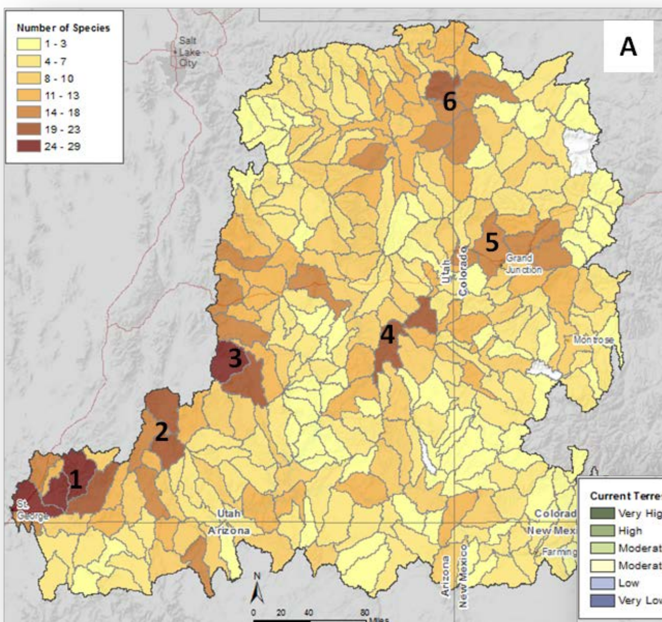
The REA Statement of Work required an assessment of ecological integrity (condition or health). As defined in the Statement of Work, ecological integrity is "the ability of ecological systems to support and maintain a community of organisms that have the species composition, diversity, and functional organization comparable to those of natural habitats within the ecoregion (Karr and Dudley 1981)." The wildlife species selected as core conservation elements for the REA were envisioned to be wide-ranging species that represent other species and multiple habitats and serve as indicators of the condition of the ecoregion. Besides having broad representation, some of the selected indicator species should be habitat specialists that express site fidelity for breeding, nesting, or wintering (to reduce interannual variability in sampling) and also sensitive and responsive to a range of disturbances. The ecoregion-wide scope in these REAs did not lend itself well to accommodate an approach using indicator species. Perhaps reducing the size of the region to more homogeneous subunits such as Environmental Protection Agency (EPA) level IV ecoregions (Woods et al. 2001, Chapman et al 2006) and selecting assemblages or species guilds (e.g., sagebrush obligates) across sites with a range of disturbances within these smaller units would produce a useful biological component to add to the spatial measure of terrestrial ecological integrity developed for this REA.

There are few measurable indicators and metrics available as spatial data for individual species to incorporate into such an effort. For this REA, our present state of knowledge required the use of the condition of vegetation communities, habitats, or landscapes as surrogates for the condition of the species and ecological processes in the region. With BLM approval, the REA focused on landscape intactness, an attribute that could be defensibly supported by existing geospatial datasets and reasonably tracked through time. Although different species may possess different tolerances to regional habitat conditions, species assemblages and natural patterns and processes are typically increasingly compromised by the cumulative effects of the change agents that affect their habitats. Terrestrial and aquatic landscape intactness models served as the foundation against which to assess current and future conservation element status.

This reliance on landscape intactness to represent ecological integrity meant that the presence or absence of a particular species, species rarity, or species richness did not factor into any metric of integrity. High species richness or concentrations of rare or endemic species do not indicate high ecological integrity (Odum 1985, Scott and Helfman 2001). Richness is limited by the partitioning of energy among species (Currie 1991, Hawkins et al. 2003); some of our most valued and intact landscapes support few species (Currie 1991, Hughes et al. 2004). On the other hand, although areas of high species richness or endemism should be evaluated separately from integrity or intactness, they are still important for conservation and management decision making. Much of the BLM's management and planning is species-centric. This chapter examines the use of regional concentrations or hotspots of species and resource values as one avenue to regional planning that identifies regional areas of interest for closer examination at a local scale.

6.1.1 NatureServe Natural Heritage Elements

NatureServe summarized Natural Heritage data for the ecoregion by 5th level HUCs, enumerating all G1–G3 species (Master 1991, Master et al. 2000) and threatened and endangered species occurring within each HUC. The map identifies specific areas that are species-richness hotspots for these sensitive fine-filter elements within the ecoregion (Figure 6-1A). The richness function map layers represent locations from which occurrences have been recorded, rather than where the species currently occurs. The greatest concentration of these species is along the western border of the ecoregion along the boundary with the high elevation Aquarius Plateau, but other concentrations can be observed in the central and northeastern portions of the ecoregion as well. Comparing these species concentrations to the same areas on the terrestrial landscape intactness map shows that many of the HUCs with high concentrations of sensitive species do not coincide with areas of High or Very High landscape intactness (green areas in Figure 6-1B). This is not unexpected when one considers that human activities tend to put species at risk, but it is interesting to see the regional pattern.



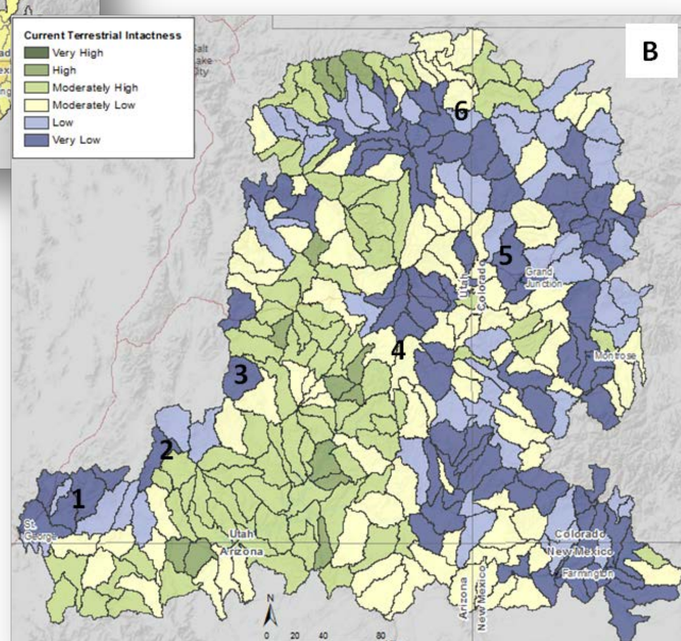
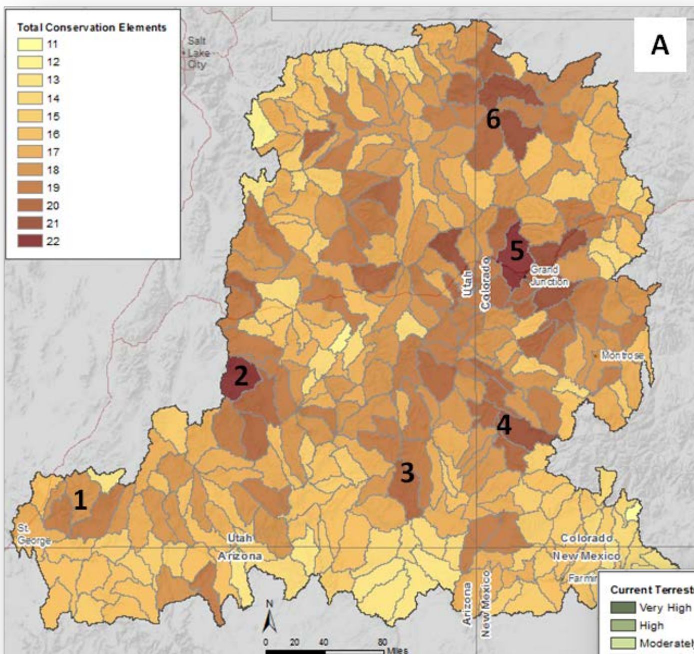


Figure 6-1. (A) Map shows number of G1–G3 species from NatureServe Heritage data for the Colorado Plateau ecoregion organized by 5th level HUC and (B) current terrestrial landscape intactness model results by HUC. Numbers in bold relate areas of high concentrations of sensitive species with levels of landscape intactness in the same locations. Summary maps for NatureServe data for all species included in the Heritage database, G1–G3 and threatened and endangered species are provided in Appendix C.

6.1.2 Concentrations of Conservation Elements

As was done in the previous section for the heritage data, the collection of REA conservation elements (CEs) was combined by HUC to create CE concentrations or hotspots to compare against regional terrestrial landscape intactness. The list of 29 conservation elements included 18 species, 9 ecological systems, and Herd Management Areas (HMAs). The number of conservation elements contained within a single HUC ranged from 11–22. As before, although the map results do not mimic the NatureServe data exactly, the areas with high concentrations of conservation elements (19–22) were located in the lower-scoring intactness landscapes (Figure 6-2A and B) and no HUCs containing the highest CE concentrations were classified as having High or Very High intactness.



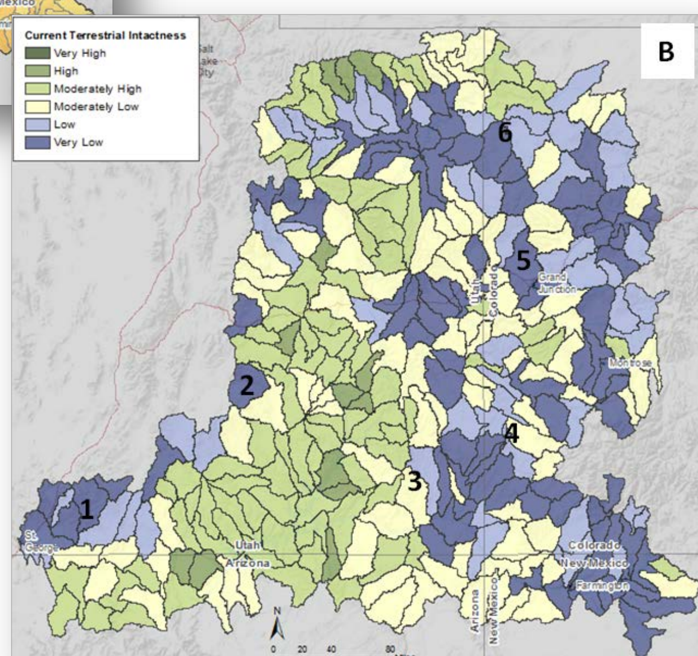


Figure 6-2. Map (A) shows number of conservation elements for the Colorado Plateau ecoregion organized by 5th level HUC and (B) current terrestrial landscape intactness model results. Numbers in bold relate areas of high concentration of conservation elements to corresponding levels of landscape intactness.

Mapping the conservation element composite at the 4 km reporting unit reveals an improvement in spatial detail with the increase in resolution of the reporting unit (Figure 6-4B). The most apparent difference at the 4 km scale is the ability to detect some of the stream networks and with them the contribution of the aquatic conservation elements to the CE concentrations. The 4 km resolution shows a more textured result when mapped and compared to landscape intactness reported by HUC; that is, groups of 4 km grid cells within low scoring HUCs will show a wider range of intactness classes, picking up the complete range (1–18, compared to the 11–22 elements observed using HUCs). The 4 km results are at a scale and detail that more closely matches recognizable topographic changes and areas of management interest. When 4 km results such as these are compared to regional intactness mapped at the 4 km unit (as in Figure 6-5A in Section 6.2.1 below), management may be aimed at grid cells with higher levels of intactness or neighboring grid cells of lower intactness that might be candidates for restoration.

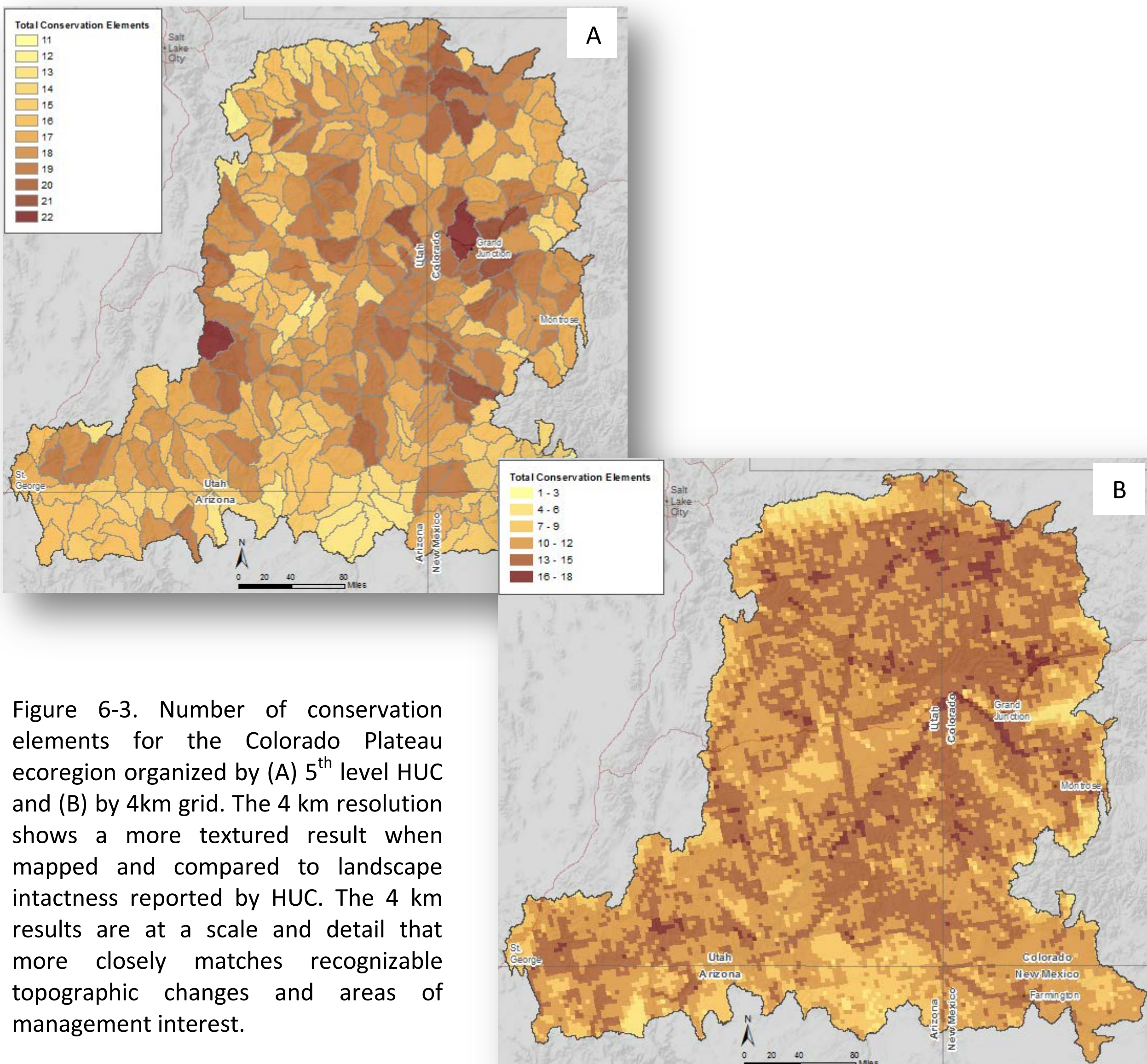


Figure 6-3. Number of conservation elements for the Colorado Plateau ecoregion organized by (A) 5th level HUC and (B) by 4km grid. The 4 km resolution shows a more textured result when mapped and compared to landscape intactness reported by HUC. The 4 km results are at a scale and detail that more closely matches recognizable topographic changes and areas of management interest.

6.2 Regional View of Landscape Intactness: Current and Future Risk to Conservation Elements

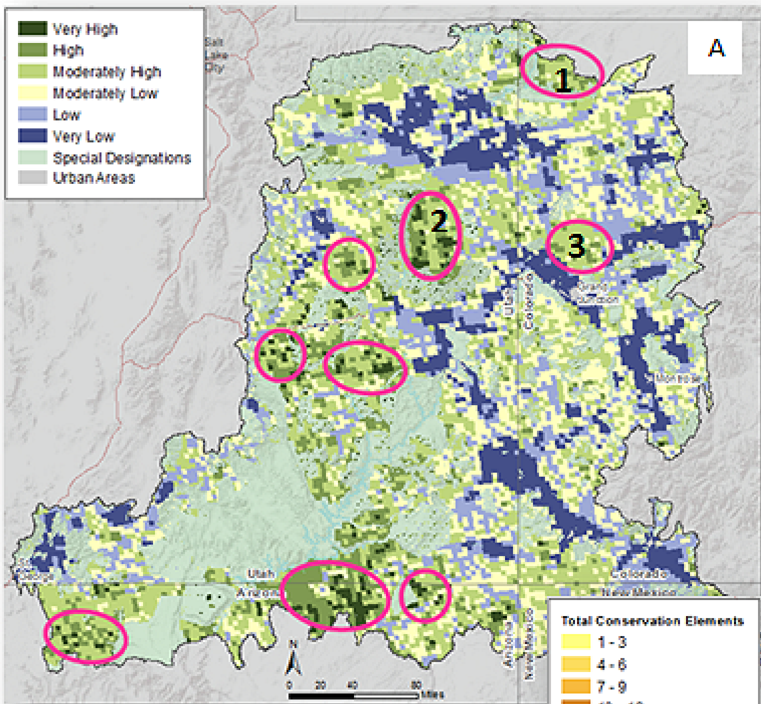
6.2.1 Comparing Concentrations of Conservation Elements with Regional Levels of Intactness

Three different maps were considered to represent the concentrations of resource values and to reveal patterns across the region—1) REA conservation elements enumerated by 4 km grid cell (Figure 6-4A); 2) the number of globally critically imperiled and vulnerable species (G1–G3 by 5th level HUC, Figure 6-4B); and 3) the number of USFWS threatened and endangered species recorded by 5th level HUC (Figure 6-4C). Additional areas of interest were added to map 6-4A from maps 6-4B and 6-4C to create one map for purposes of discussion (Figure 6-4D), used when comparing concentrations of conservation elements with intactness maps and maps of future condition found in the following sections. All three maps share areas surrounding Dinosaur National Monument in the northeastern portion of the ecoregion, Castle Valley and the Colorado River corridor between Moab and Canyonlands in the center of the map, and the Zion National Park-Uinkaret Plateau area in the southwestern corner. Maps 6-4A and 6-4B share an area in the vicinity of the Roan Plateau and Colorado National Monument in the northeast. Other concentrations of globally imperiled species in map 6-4B occur along the western boundary in the transition to the Sevier Plateau near Bryce Canyon and Capitol Reef. Finally, a concentration of threatened and endangered species in map 6-4C just south of the Uinta Basin marks an area on the west Tavaputs Plateau straddling the Green River.

To compare these concentrations of conservation elements (CEs) to the condition of surrounding habitats at the 4 km grid scale, areas of moderately high to high intactness have been outlined (in pink) on the intactness map (Figure 6-5A) and the higher concentrations of CEs outlined in royal blue on the map in Figure 6-5B. A comparison of the two maps identifies some broad areas of interest between the two layers, particularly in the north. Area 1 (black numbers in bold in Figure 6-5A) marks the Douglas Mountain area north of Dinosaur National Park; area 2 represents a concentration of threatened and endangered species and REA conservation elements near the Winter Ridge Wilderness Study Area and Green River, western Tavaputs Plateau; and area 3 covers an area common to maps 6-4A and 6-4B near Grand Junction that includes the Roan Cliffs and Roan Plateau. In planning situations, of course, there may be valid reasons for restoring or protecting areas of lower intactness or lower numbers of resource values. Areas 4 and 5 (white numbers, Figure 6-5B) represent areas with high concentrations of CEs that occur in moderate to lower intactness classes. Existing protected areas tend to occur in rugged or high elevation terrain that experiences lower development pressure than lowland areas where heavy resource use makes it more difficult to establish conservation areas. For example, Gunnison sage-grouse is an REA species of concern that is in serious decline, an inhabitant of sagebrush habitats that are more highly developed, privately owned, and difficult to conserve. When Gunnison sage- grouse distribution is compared to designated protected areas (Figure 6-6), one can see that there is little overlap. BLM manages 23% of the species' distribution (101,000 acres); 10% of the species distribution (46,000 ac) is protected by various designations, leaving 67% (295,000 acres) unprotected.

The vast amount of information produced by this REA can and must be examined in multiple ways and at multiple scales. To accompany the spatial mapped results, it will be useful for managers to have tabular summaries of conservation elements and areas in various intactness classes. Table 6-1 shows the results for all lands across the Colorado Plateau. The matrix is organized into six different categories; the colored panels indicate High, Medium, and Low intactness classes (red, blue, and yellow, respectively) from left to right with increasing numbers of conservation elements from top to bottom (darker color tones for the higher concentrations of conservation elements). An accompanying map using the same color scheme is provided in Figure 6-7.

Figure 6-4. Maps of (A) concentrations of conservation elements, (B) globally imperiled species, and (C) USFWS-listed threatened and endangered species with highest concentrations circled; (D) map A with additional areas of interest (represented by 3 additional blue ellipses) added to it from maps B and C. Protected areas are masked out in light green.



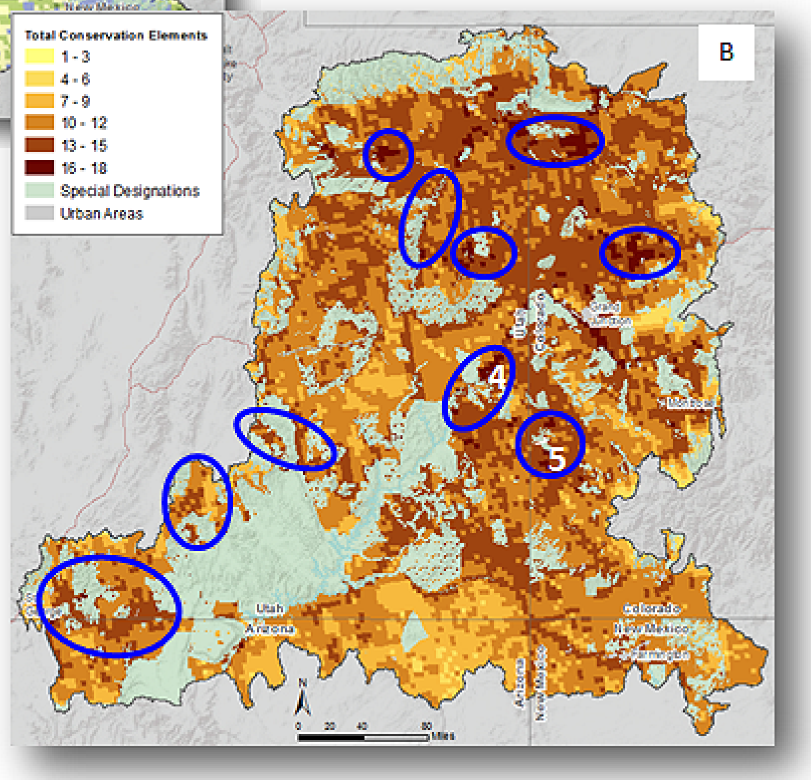


Figure 6-5. Maps show (A) terrestrial intactness for the Colorado Plateau ecoregion and (B) concentrations of all species and other conservation elements of interest mapped using 4 km grid with designated protected areas in light green. Black numbers (map A) mark broad areas of interest between the two maps. White numbers (map B) denote high CE concentrations in lower intactness areas that may be worthy of protection or restoration.

Acres within each category in Table 1 may be viewed in different ways to assess management options and to inform policy decisions. For example, areas in dark red—those locations that contain high concentrations of conservation elements and that demonstrate the highest levels of landscape intactness—can be viewed as places of high potential conflict or high protection value. Future development may be more acceptable in areas in the light yellow category (low intactness and low concentrations of conservation elements) assuming specific issues (protection of a sensitive species) are assessed and properly managed. Areas in dark blue (high concentrations of conservation elements combined with moderate intactness) may be the best locations for restoration to get the greatest return on investment.

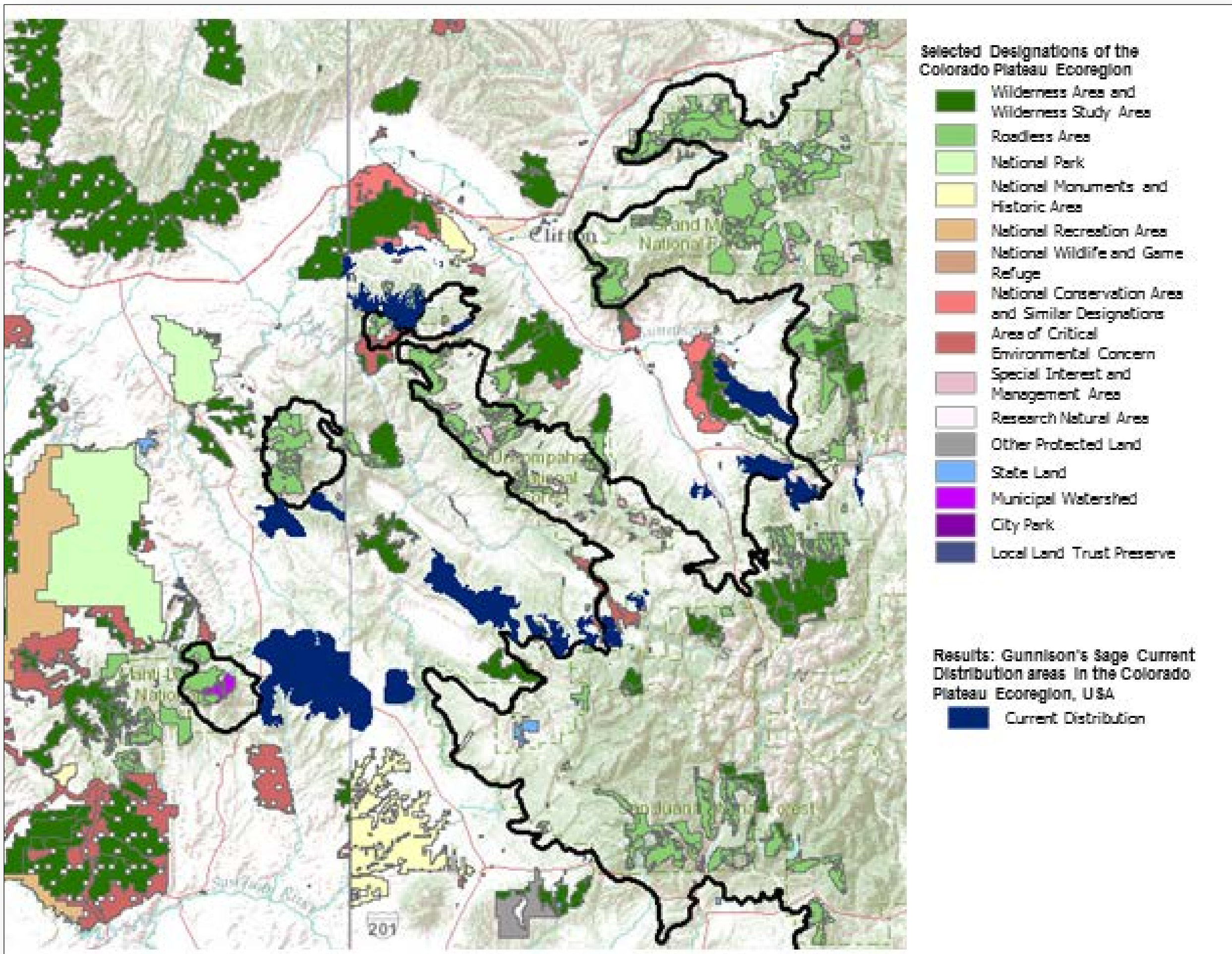


Figure 6-6. Map shows classes of designated protected areas in various colors with the distribution of Gunnison's sage-grouse in dark blue. There is little overlap of protected areas with the distribution of this threatened, though not yet federally listed, species except just north of Black Canyon of the Gunnison (tri-colored area near eastern boundary).

Table 6-1 is one example of how the matrix table could be organized. Depending on the circumstances and issues to be addressed, managers could organize the same data in different ways (Figure 6-8). The standard model presented here (Figure 6-8A) could be changed by increasing (Figure 6-8B) or decreasing (not shown) the threshold for conservation element concentrations. A simpler grid could be applied to the data using a 4 panel instead of a 6 panel organization (Figure 6-8C). Finally, the number of categories could be increased based on the range of conservation element concentrations or number of management options (Figure 6-8D). Managers could also take into account rare species information by adding the heritage findings (the globally imperiled or threatened and endangered species shown in Figures 6-1B and C) to the list of conservation elements and incorporating them into the matrix diagram. In addition to creating a useful matrix table, one could improve the approach by grouping species into guilds, ranking species by sensitivity to disturbance, working at various scales (both regional and local) or working within relatively homogeneous landscape areas (such as EPA level IV ecoregions).

	AREA IN ACRES FOR ALL COLORADO PLATEAU LANDS BY NUMBER OF CONSERVATION ELEMENTS AND INTACTNESS CLASSES						
NUMBER OF CONSERVATION ELEMENTS	Very High	High	Moderately High	Moderately Low	Low	Very Low	Totals
0							
1							
2		325					325
3	2	1,044		39	867	15	1,966
4	308	2,634	197	1,082	1,244	18	5,483
5	41,932	78,555	101,008	95,350	50,978	3,954	371,777
6	72,732	188,071	233,112	130,455	67,398	1,902	693,670
7	75,490	173,751	462,173	310,476	202,505	32,823	1,257,219
8	82,251	264,931	659,448	455,730	243,903	68,004	1,774,268
9	207,748	461,112	1,008,241	694,964	485,681	130,156	2,987,903
10	179,732	841,176	1,536,486	1,116,386	772,118	361,426	4,807,325
11	218,433	1,147,886	1,863,389	1,326,415	1,068,169	787,067	6,411,359
12	262,209	1,620,275	2,485,117	1,774,932	1,256,474	1,212,609	8,611,617
13	268,180	1,503,916	2,546,286	1,851,837	1,186,985	1,184,497	8,541,701
14	170,005	905,374	1,730,262	1,220,270	924,792	727,319	5,678,021
15	43,490	418,739	654,945	498,003	403,267	379,546	2,397,989
16	7,907	154,190	249,077	166,051	159,265	185,819	922,310
17	3,954	63,258	71,165	55,350	39,536	47,443	280,706
18		7,907	23,722	7,907	7,907	11,861	59,304
Totals	1,634,374	7,833,144	13,624,628	9,705,248	6,871,091	5,134,459	44,802,943

Table 6-1 lists all lands for all ownerships across the Colorado Plateau with the number of conservation elements on the y-axis and columns for area of lands in 6 intactness classes. The colored panels indicate High, Moderate, and Low intactness classes (red, blue, and yellow, respectively) from left to right and lower and higher numbers of conservation elements (CEs) from top to bottom (lighter and darker colors).

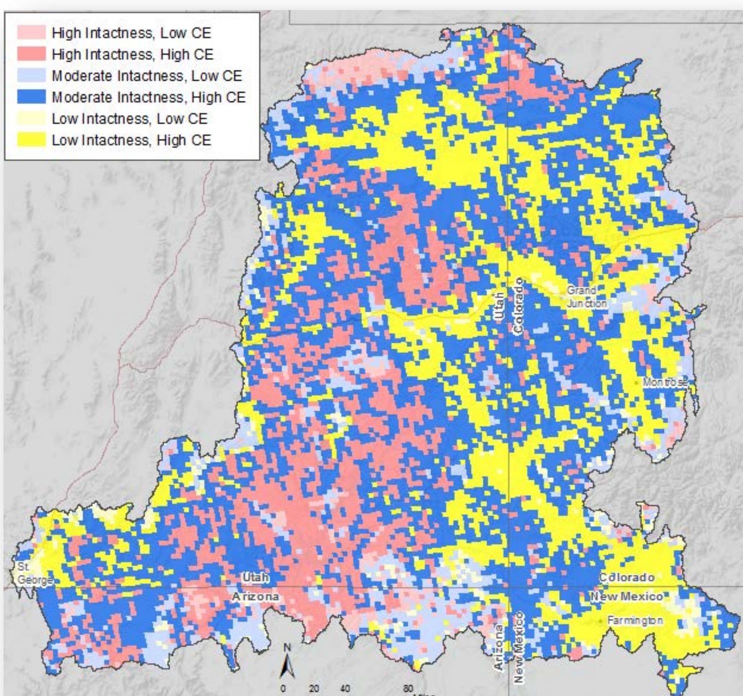


Figure 6-7. Map to accompany Table 6-1 showing 6 classes of intactness by number of CEs. Colors in map match color panels in Table 6-1.

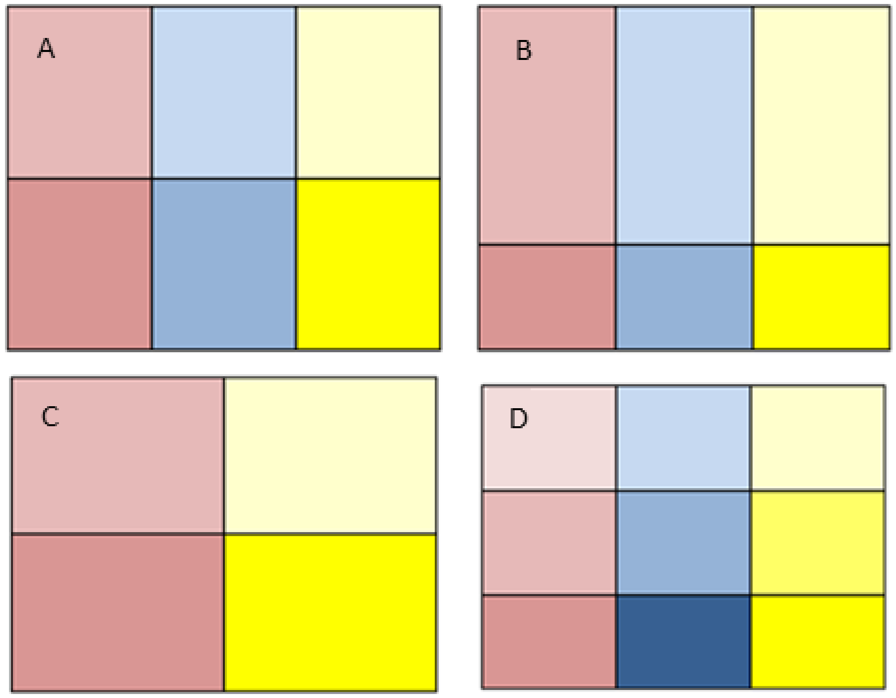


Figure 6-8. Different options for organizing data (area information in acres as presented in Table 6-1) comparing concentrations of conservation elements (y-axis) and groupings of classes of relative landscape intactness (x-axis). Colors correspond to different categories for the combinations and match colors in Table 6-1.

Using the example provided by Table 6-1 and its associated map (Figure 6-7), the analysis was rerun, this time excluding all specially designated lands and urban areas. The resulting matrix table (Table 6-2) and companion map (Figure 6-9) emphasize land areas in play across multiple ownerships and reduces the amount of land area from the "all lands" view by approximately 28 percent.

Although BLM managers will be pursuing a landscape approach to management that stresses cooperative planning across agencies and ownerships, they will also want to examine REA results for BLM lands only (Figure 6-10, maps of intactness and concentrations of conservation elements with designated lands excluded; maps are the same as those in Figure 6-5A and 6-5B but for BLM lands only). Table 6-3 and companion map (Figure 6-11) present the land area information (for BLM lands only outside of designated lands) with the number of conservation elements on the y-axis and, across the table, six columns with acreage totals of area of BLM lands in the various intactness classes.

AREA IN ACRES FOR ALL LANDS MINUS DESIGNATED SITES AND URBAN AREAS

NUMBER OF CONSERVATION ELEMENTS	Very High	High	Moderately High	Moderately Low	Low	Very Low	Totals
0							
1							
2		122					122
3				39	798	15	852
4	26	126	197	592	1,099	18	2,057
5	6,359	4,190	36,343	33,790	27,398	3,954	112,034
6	9,544	26,932	133,560	101,880	57,198	646	329,759
7	38,928	98,061	299,887	223,372	165,249	19,282	844,779
8	41,916	136,620	397,361	369,330	211,830	48,166	1,205,223
9	143,779	337,058	822,095	596,874	379,410	111,317	2,390,531
10	98,300	533,111	1,136,860	916,274	647,323	334,184	3,666,053
11	116,486	585,514	1,298,366	1,119,992	916,874	713,744	4,750,976
12	126,932	709,864	1,648,555	1,484,415	1,110,455	1,118,341	6,198,563
13	103,747	668,880	1,757,000	1,577,248	1,057,591	1,098,130	6,262,596
14	52,918	319,149	1,112,105	1,008,368	838,234	666,643	3,997,417
15	8,891	114,230	380,216	414,642	367,666	361,540	1,647,185
16	3,954	31,811	113,629	142,211	150,540	171,455	613,599
17		9,116	33,357	34,424	34,039	39,112	150,048
18		75	18,172	7,880	7,774	4,528	38,429
Totals	751,780	3,574,858	9,187,701	8,031,330	5,973,479	4,691,074	32,210,222

Table 6-2 lists all lands minus areas of designated sites and urban lands across the Colorado Plateau with the number of conservation elements on the y-axis and six columns for area of lands in various intactness classes with acreage totals.

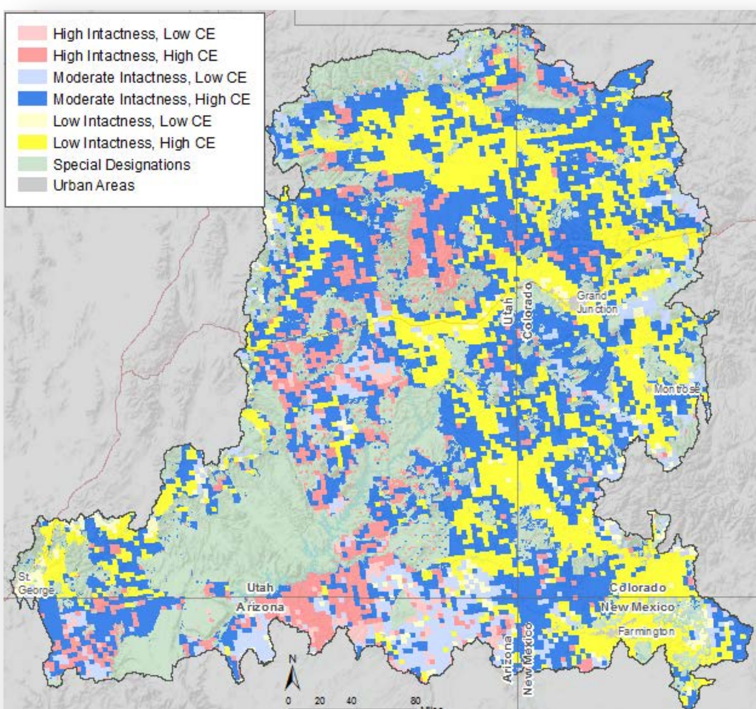
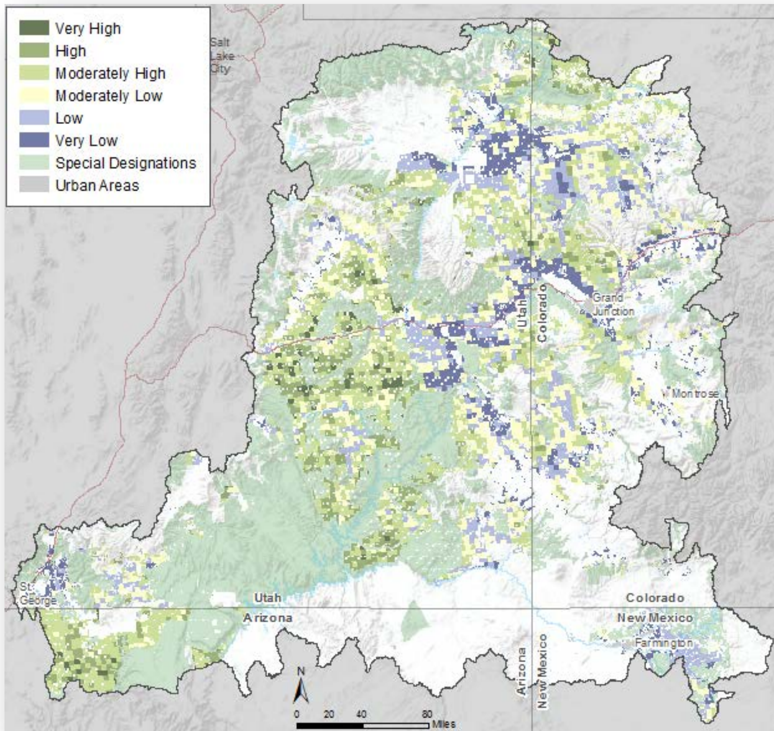


Figure 6-9. Map to accompany Table 6-2 showing 6 classes of intactness by high or low number of CEs. Colors in map match color panels in Table 6-2.



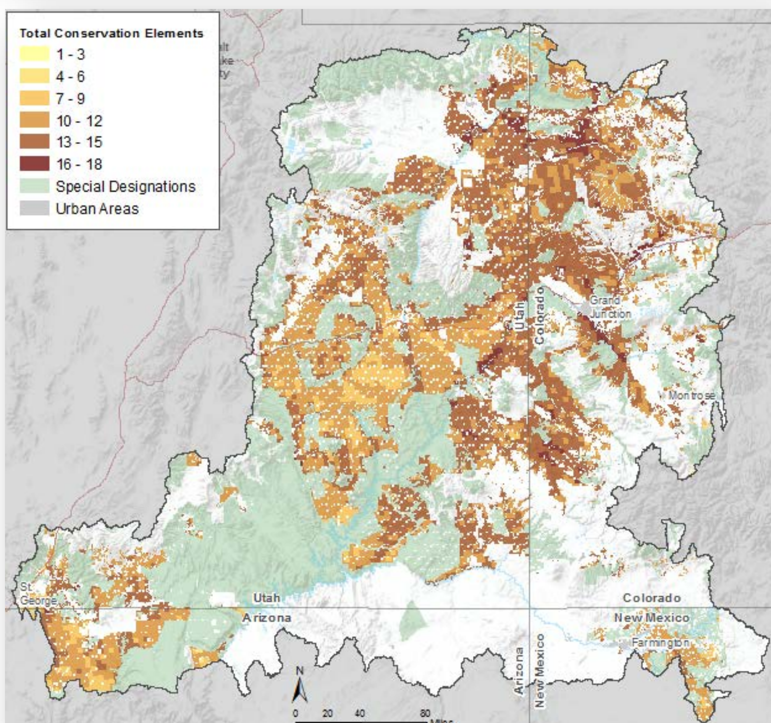


Figure 6-10. Top: Map of intactness for BLM lands outside of designated areas (light green). Bottom: Map of concentrations of conservation elements for BLM lands outside of designated areas (light green). These maps reproduce Figure 6-2A and B for BLM lands only.

AREA IN ACRES FOR BLM LANDS MINUS DESIGNATED AND URBAN AREAS

NUMBER OF CONSERVATION ELEMENTS	Very High	High	Moderately High	Moderately Low	Low	Very Low	Totals
4	7		197				**204**
5			1,151	1,374	1,913	10	**4,448**
6	59	2,646	4,348	1,794	3,357	91	**12,295**
7	8,192	13,291	20,011	11,652	20,337	859	**74,341**
8	24,067	42,906	84,095	14,025	24,991	5,153	**195,239**
9	23,452	88,286	261,824	68,049	51,423	18,581	**511,615**
10	27,006	202,761	375,175	222,689	133,158	86,081	**1,046,870**
11	51,942	278,677	536,640	383,509	308,140	187,473	**1,746,382**
12	37,733	334,219	857,830	678,580	497,175	302,426	**2,707,963**
13	40,041	364,065	996,388	797,590	536,856	336,634	**3,071,573**
14	17,732	192,241	654,400	576,989	470,881	256,826	**2,169,069**
15	1,842	49,516	239,021	243,921	186,167	166,667	**887,134**
16		12,864	71,744	98,806	56,650	73,244	**313,308**
17		2,076	24,439	25,264	14,961	16,933	**83,672**
18		2	12,112	6,480	7,316	2,349	**28,260**
Totals	**232,071**	**1,583,550**	**4,139,375**	**3,130,723**	**2,313,326**	**1,453,328**	**12,852,373**

Table 6-3 lists all BLM lands minus areas of designated and urban lands for the Colorado Plateau with the number of conservation elements on the y-axis and six columns of area of lands in the various intactness classes with acreage totals.

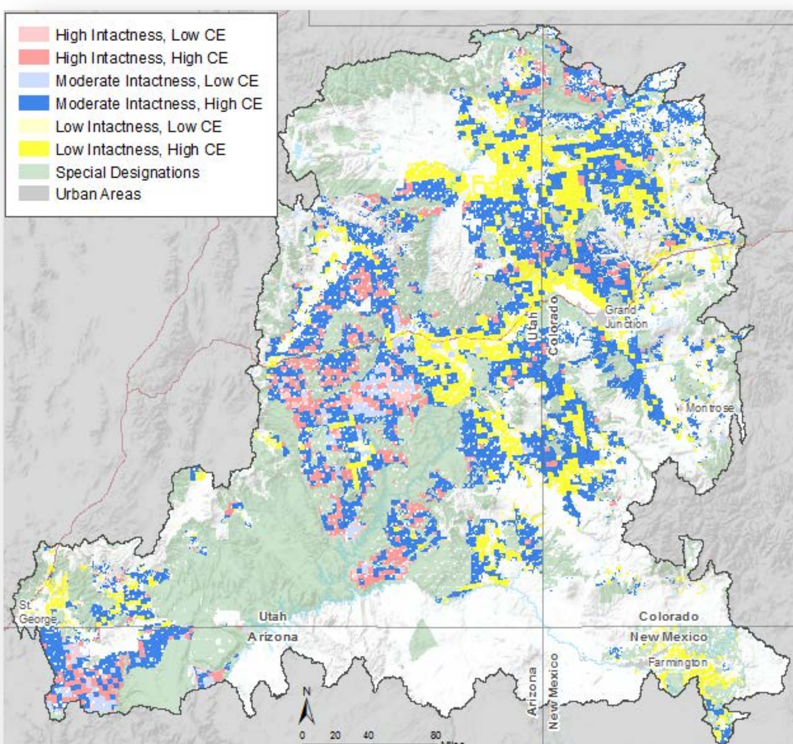


Figure 6-11. Map to accompany Table 6-3 showing 6 classes of intactness by high or low number of CEs for BLM lands only. Colors match color panels in Table 6-3.

6.2.2 Exposure of CE Concentrations to Change Agents

6.2.2.1 Current and Near-Term Future (2025) Development

The relationship of region-wide concentrations of conservation elements and three development scenarios are presented in Figure 6-12A–D. The current and near-term development models appear very similar (Figures 6-12A and 6-12B), with changes occurring mostly in the Uinta Basin and Grand Valley from oil and gas development and the spread of invasive species. The near-term future development (2025) model was built from the logic model presented in Section 5.1, which contains four major development components—energy, agriculture, urban and roads, and recreational development. Little predictive data were available for future projections; the REA relied on data for projected near-term oil and gas development (Copeland et al. 2009), the spread of invasives, and urban expansion (Theobald 2010). The only projected near-term renewable energy development was a small area of potential wind development in the southwest corner slightly outside the ecoregion boundary (Figure 6-12B). The third map, maximum potential energy development (Figure 6-12C), is more speculative—that is, not based on actual plans for development—with a longer term time frame; the results there are shown in three classes. Potential oil and gas development data included numerous sources—oil and gas potential data (Copeland et al. 2009), BLM oil and gas leases, allowable leasing footprints for tar sand and oil shale extraction, and Department of Energy producing oil and gas fields, mapped by buffering existing active wells by 1.4 km. Two data sources comprised potential wind development—Utah BLM wind energy priority Areas and National Renewable Energy Laboratory (NREL) wind power density classes 3 and above at 50 m high. Solar resource potential (>5.5 kW/m^2) was obtained from NREL as well. Summarized at 4km resolution, the final composite map for all energy components covered a large area of the ecoregion in the northern and eastern portions.

Just as the status of individual conservation elements was determined relative to current and near-term future development (2025) in Chapters 4 and 5, concentrations of CEs and high resource values can be identified that are at risk from current, near-term, and longer term potential energy development (Figure 6-12A–D). Urban expansion and renewable energy are not high-ranking development issues in the Colorado Plateau ecoregion—traditional oil and gas is the top terrestrial development issue in the region. The high concentrations of conservation elements (circled in royal blue in Figures 6-12A–D) are most at risk for change from near-term future development (2025) in the Uinta Basin, Farmington, St. George, and Grand Junction areas. Development pressures in the Uinta Basin and Grand Valley affect many of the core REA conservation elements: sagebrush obligates, particularly greater sage-grouse, and species associated with white-tailed prairie dog colonies such as black footed ferret, burrowing owl, golden eagle, and ferruginous hawk. See more detailed results for the distribution and status of each of these species in Appendix C.

Circled areas of concentrations of CEs in the central portion of the region are at less risk for change from development (Figure 6-12A–D)—particularly areas previously discussed marked 2, 4, and 5 in Figure 6-2A and B. Although it is difficult to see the changes between the current and the near-term development map (Figure 6-12A and B), the Very High development class grew by 2%, and both High and Moderately High classes gained approximately .5%; in all, the development footprint increased by about 1.5 million acres for the near-term 2025 scenario.

Summary tables for future energy development (both near-term [2025] and maximum potential [longer term] energy development) accompany the mapped results (Tables 6-4 and 6-5). Areas in acres for both categories of land area were assessed using the intersection of the additional area of future energy developments, the 4 km intactness surface, and the total concentration of conservation elements per 4 km grid cell.

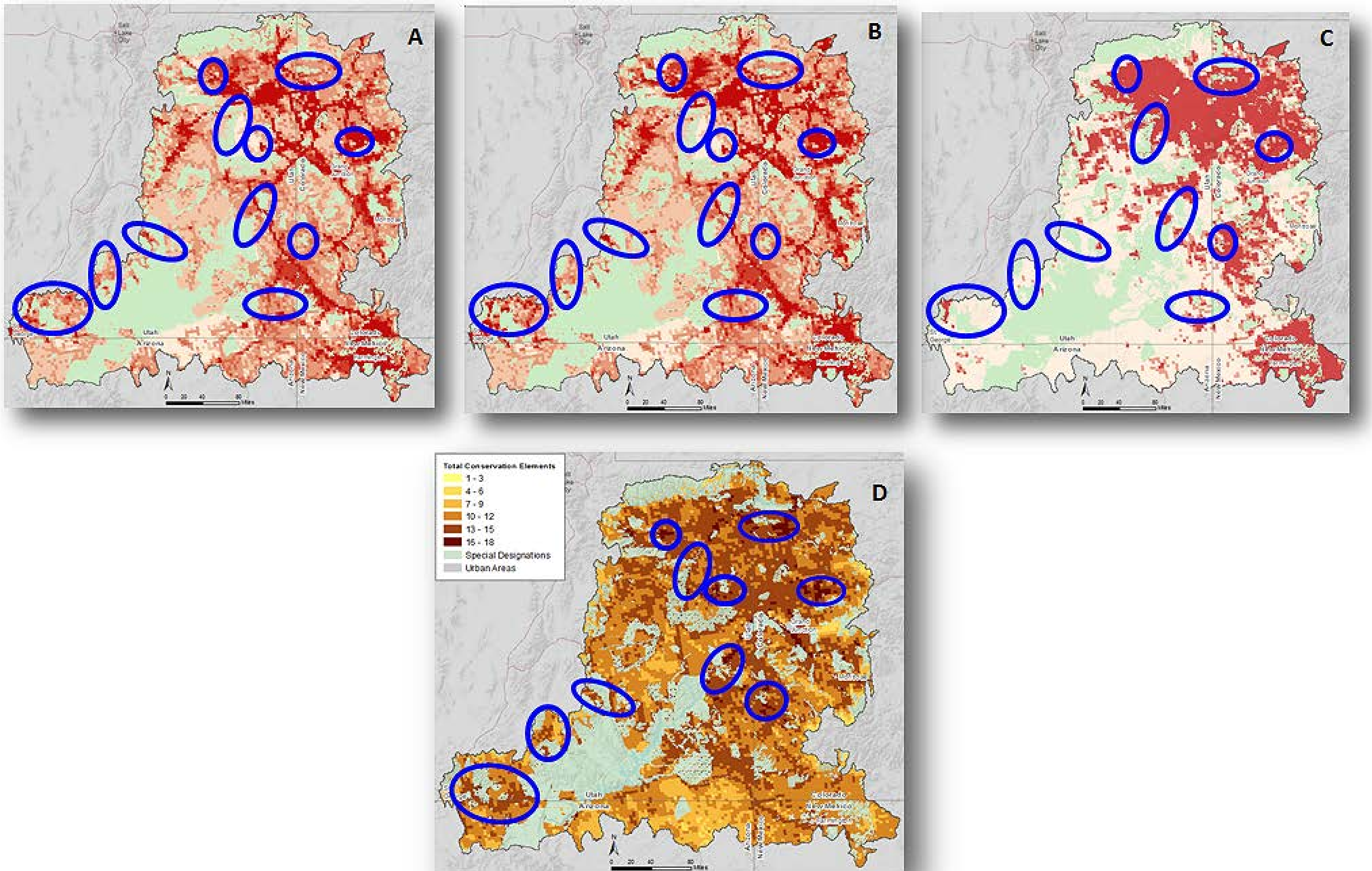


Figure 6-12. Maps arranged to compare patterns show (A) current development footprint in the Colorado Plateau ecoregion, (B) near-term future (2025) development, (C) longer term maximum potential energy development, and (D) concentrations of conservation elements with highest concentrations circled in royal blue in each map. Designated sites are masked out in light green on all maps.

AREA IN ACRES OF LAND SURFACE AFFECTED BY NEAR-TERM (2025) ENERGY DEVELOPMENT BY NUMBER OF CONSERVATION ELEMENTS AND INTACTNESS CLASS

# of CONSERVATION ELEMENTS	Very High	High	Moderately High	Moderately Low	Low	Very Low	Totals
9					1,832		1,832
10			742	5,715	2,528	4,060	13,045
11		409	691	4,218		9,989	15,307
12		230	5,023	6,256	862	26,213	38,585
13		2,880	4,180	1,453	14	14,732	23,259
14				1,776	655	2,054	4,485
15				422	136	2,061	2,620
16				414	499		914
17				26			26
Totals	0	3,519	10,636	20,281	6,526	59,110	100,072

Table 6-4. Land area in acres in various intactness classes and number of conservation elements affected by near-term (2025) energy development. It is useful to know that there is little acreage in the Very High and High categories affected by near-term energy development. Closer inspection may reveal if acreage in the High category should be off-limits to development. It is also useful to note that most of the land area affected by possible near-term energy development contains high concentrations of conservation elements.

AREA IN ACRES OF LAND SURFACE AFFECTED BY POTENTIAL ENERGY DEVELOPMENT BY NUMBER OF CONSERVATION ELEMENTS AND INTACTNESS CLASS

NUMBER OF CONSERVATION ELEMENTS	Very High	High	Moderately High	Moderately Low	Low	Very Low	Totals
0							
2		179					**179**
3	2	617			349	15	**982**
4	184	690	6	584	219	18	**1,701**
5	9,199	12,169	11,841	14,986	7,496	1,098	**56,788**
6	10,887	26,895	41,840	24,672	17,898		**122,191**
7	12,193	24,712	70,496	69,281	35,161	2,286	**214,129**
8	13,620	47,741	114,439	85,012	58,191	15,830	**334,834**
9	17,113	49,894	225,466	176,711	242,293	20,343	**731,820**
10	18,213	88,546	356,938	342,981	396,093	204,058	**1,406,829**
11	29,595	139,184	448,881	528,942	599,121	375,095	**2,120,819**
12	34,332	246,484	655,276	705,143	700,515	668,596	**3,010,346**
13	33,835	248,098	902,613	885,330	641,888	656,360	**3,368,124**
14	26,133	190,674	630,215	620,558	518,868	431,410	**2,417,858**
15	11,651	111,519	281,143	280,884	265,114	266,312	**1,216,622**
16	37	32,191	77,931	102,911	92,450	129,744	**435,265**
17	101	8,096	26,654	35,252	26,390	19,599	**116,092**
18		1,136	12,759	6,579	6,741	4,287	**31,502**
Totals	**217,095**	**1,228,824**	**3,856,500**	**3,879,825**	**3,608,788**	**2,795,051**	**15,586,082**

Table 6-5. Land area in various intactness classes and number of conservation elements affected by maximum potential development. Longer term maximum potential energy development occurs in all classes and concentrations of conservation elements.

6.2.2.2 Current and Future Risk from the Spread of Invasive Species

Urban area and invasive projections were updated for the near-term future (2025) terrestrial landscape intactness model (see logic model Section 5.3). The change in urban area relative to concentrations of conservation elements was covered in the previous development section (6.2.2.1). The only other future projection data available for near-term future terrestrial intactness was that for the spread of invasive species. Invasives species projection data was added to current invasives data (LANDFIRE and NatureServe invasives classes, a predictive model of tamarisk distribution [Jarnevich et al. 2011], and historic tamarisk polyline data). Projections of invasive spread were based on LANDFIRE succession class data, which included all invasive species, and US Geological Survey data on early seasonal invasives (J. Hansen, T. Arundel, and R. Kokaly, model created in 2011 for this REA). The near-term change attributed to the spread of invasives shows the most impacts in the Uinta Basin, the southwestern corner of the ecoregion, and the San Juan River basin-Farmington region (Figure 6-13A). About half of the CE concentrations are located in dense areas of invasives, particularly in the northeastern, central, and southwestern portions of the ecoregion.

6.2.2.3 Future Risk from Climate Change

The MAPSS climate results were used to predict changes in temperature, precipitation, potential evapotranspiration, and runoff; a number of the key findings from these analyses were selected to assemble into an overall relative climate change map showing different levels of climate change potential that could then be used to assess relative impacts on the specific conservation elements (Section 5.4). The fuzzy model inputs included potential for summer temperature change and potential for winter temperature change averaged into a single factor, potential for runoff change from MAPSS modeling, potential for precipitation change, and potential for vegetation change, again from MAPSS modeling. Direction of the change was not important—only degree of departure from historic measures. Areas most likely to show the most extensive changes were those that either were predicted to change in their vegetation type or as a combination of all the other factors (temperature, precipitation, and runoff). Results were mapped in five separate classes: Very High, High, Moderate, Moderately Low and Low for the potential for an area to be affected by climate change as defined in the fuzzy logic model (Figure 6-14A). Individual species and vegetation communities' response to climate were presented in Section 5.4 as histograms. Histograms and maps for the exposure to climate change of all conservation elements (distributions overlaid with the climate change potential map) may also be viewed in Appendices B and C. Of the vegetation communities, those showing the most area under High climate exposure include the shrublands (especially big sagebrush and blackbrush-Mormon-tea communities) and pinyon-juniper woodland. This pattern is apparent when the dark brown high-climate-change exposure areas in Figure 6-14A are compared with the vegetation community landcover maps (Section 4.2.2). The areas of Very High exposure in the south and central portions of the ecoregion occur in concentrations of pinyon-juniper and big sagebrush. On the other hand, these same vegetation communities, pinyon-juniper and sagebrush, experience Moderately Low exposure to climate change farther north in the Uinta Basin. When the climate change map is compared to the concentrations of conservation elements in Figure 6-14B, the potential for climate-related change is projected to be moderate for the circled areas in the higher elevations surrounding the Uinta Basin and along the western edge of the ecoregion. The two circled areas on the eastern side of the region in the Grand Junction and Dolores River areas and the group of conservation elements along the Utah/Arizona border in the southwestern portion of the ecoregion have the highest exposure to climate change.

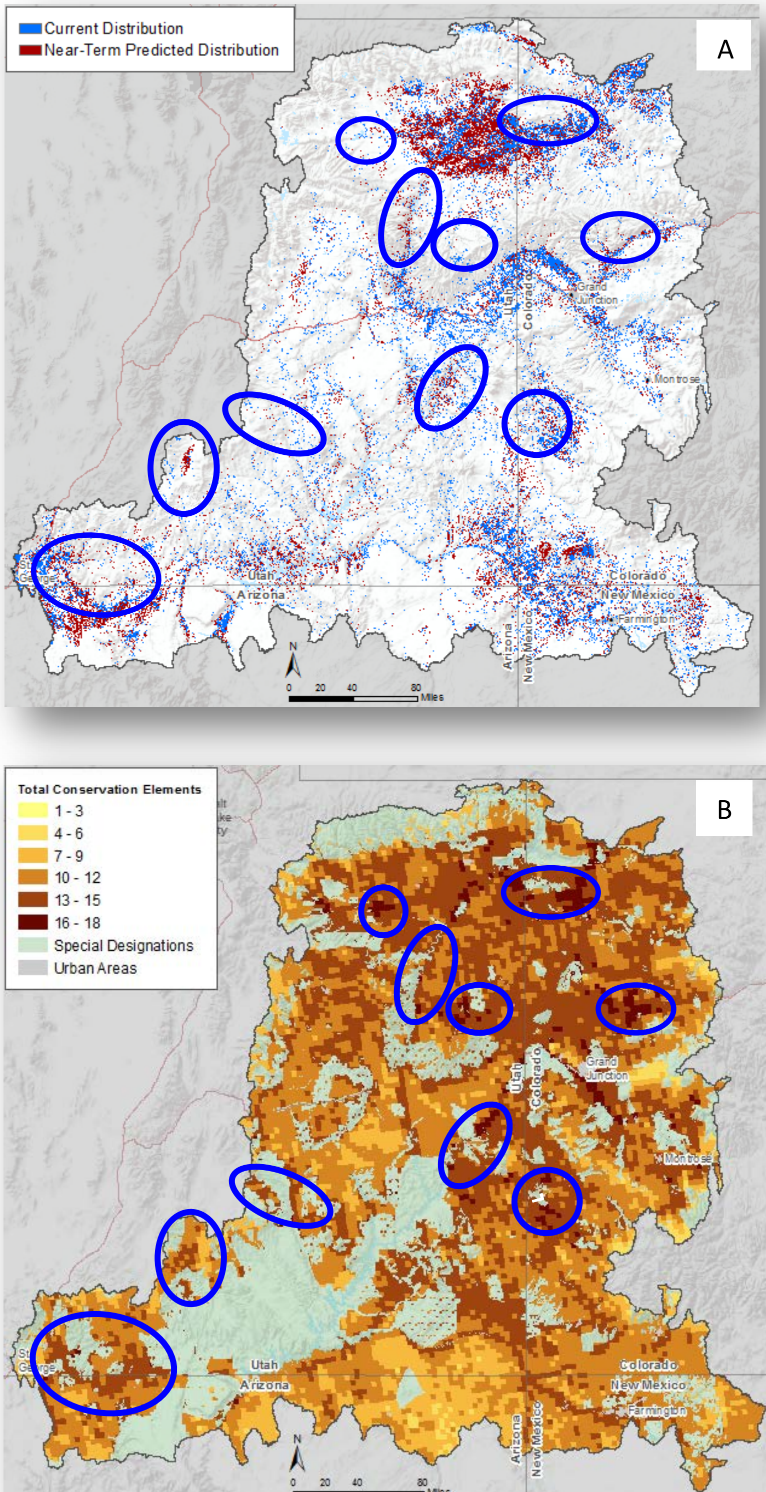


Figure 6-13. Maps for (A) current (in blue) and near-term future (2025, in red) predicted distribution of invasive species, and (B) concentrations of conservation elements with designated sites shown in green.

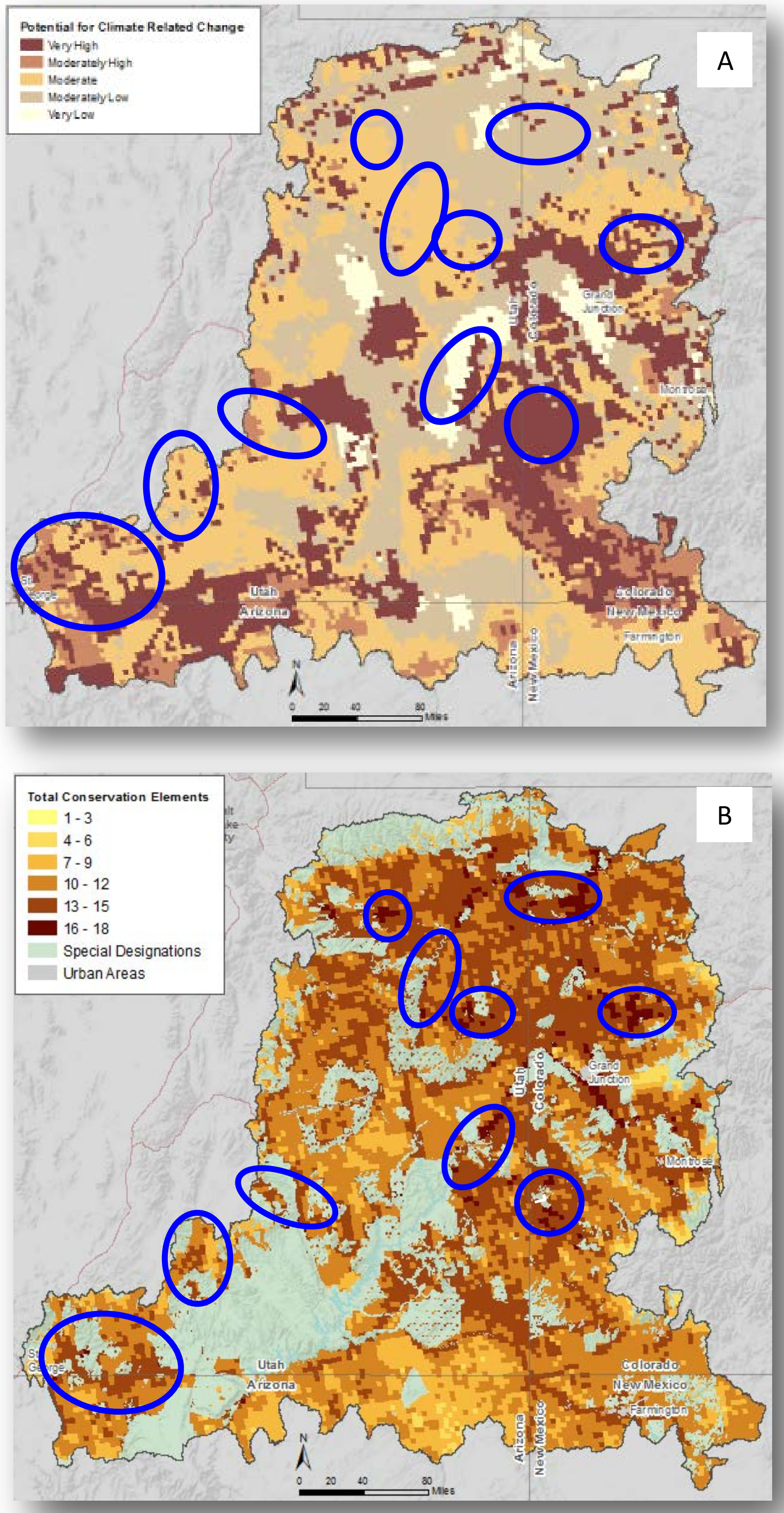


Figure 6-14. (A) Map of relative climate change potential in five classes with areas depicting concentrations of CEs circled; (B) concentrations of conservation elements. Designated sites masked out in green on map (B).

6.3 Conclusion

The examples presented in this chapter offer a few of the many ways this wealth of REA data and maps may be examined depending on project objectives, area of interest, species of concern, and present or future time frames. All that is required of the user is an understanding of the relatively coarse resolution of the results and an ability to translate the results between scales, from regional to local. Application of the results of the current and near-term future intactness models and conservation element status determinations also depend on an understanding of the limitations of a rapid ecoregional assessment of this kind. The effort is fundamentally limited by available spatial data and ecological thresholds so important to tailoring the logic models. These aspects are only likely to improve in the future as the geospatial technology and science evolve.

This REA will serve as a baseline for future efforts in the Colorado Plateau ecoregion. This REA effort provided the opportunity to inventory available information, to collect and archive an atlas of useful spatial data, and to produce hundreds of mapped products. Users may find information about access to the data at http://www.blm.gov/wo/st/en/prog/more/climatechange.html. The models are well documented and are flexible enough to be modified and improved with the addition of new data. Using the baseline current scenario, the REA components are designed for periodic updating to track the ecological status of Colorado Plateau conservation elements as they respond to landscape change and adaptive management in the coming years.

6.4 References Cited

Chapman, S.S., G.E. Griffith, J.M. Omernik, A.B. Price, J. Freeouf, and D.L. Schrupp. 2006. Ecoregions of Colorado, U.S. Geological Survey, Reston, Virginia (map scale 1:1,200,000).

Copeland, H.E., K.E. Doherty, D.E. Naugle, A. Pocewicz, and J.M. Kiesecker. 2009. Mapping oil and gas development potential in the US intermountain west and estimating impacts to species. PLoS ONE 4(10): e7400.

Currie, D.J. 1991. Energy and large-scale patterns of animal and plant species richness. *The American Naturalist* 137(1):27–49.

Hawkins, B.A., R. Field, H.V. Cornell, D.J. Currie, J-F. Guégan, D.M. Kaufman, J.T. Kerr, G.G. Mittelbach, T. Oberdorff, E.M. O'Brien, E.E. Porter, and J.R.G. Turner. 2003. Energy, water, and broad-scale geographic patterns of species richness. *Ecology* 84:3105–3117.

Hughes, R.M., S. Howlin, and P.R. Kaufmann. 2004. A biointegrity index (IBI) for coldwater streams of western Oregon and Washington. *Transactions of the American Fisheries Society* 133:1497–1515.

Jarnevich, C.S., P. Evangelista, T.J. Stohlgren, and J. Morisette. 2011. Improving national-scale invasion maps: Tamarisk in the western United States. *Western North American Naturalist* 71(2):164–175.

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

Master, L.L. 1991. Assessing threats and setting priorities for conservation. *Conservation Biology* 5: 559–563.

Master, L.L., B.A. Stein, L.S. Kutner, and G.A. Hammerson. 2000. Vanishing assets: Conservation status of U.S. species. Pages 93–118 in Stein, B.A., L.S. Kutner, and J.S. Adams (eds.), Precious heritage: The status of biodiversity in the United States. Oxford University Press, New York.

Odum, E.P., J.T. Finn, and E.J. Franz. 1979. Perturbation theory and the subsidy-stress gradient. *BioScience* 29(6):349–352.

Scott, M.C., and G.S. Helfman. 2001. Native invasions, homogenization, and the mis-measure of integrity of fish assemblages. *Fisheries* 26(11):6–15.

Theobald, D.M. 2010. Estimating natural landscape changes from 1992 to 2030 for the conterminous United States. *Landscape Ecology* 25(7):999–1011.

Woods, A.J., D.A. Lammers, S.A. Bryce, J.M. Omernik, R.L. Denton, M. Domeier, and J.A. Comstock. 2001. Ecoregions of Utah, U.S. Geological Survey, Reston, Virginia (map scale 1:1,175,000).

Photo: Petroglyphs, BLM

Glossary and Acronym List

Adaptive Management: Adaptive management is a systematic process for continually improving management policies and practices by learning from the outcomes of previously employed practices.

ArcGRID: A raster GIS file format developed by Esri**.** The grid defines geographic space as an array of equally-sized square grid points arranged in rows and columns. Each grid point stores a numeric value that represents a geographic attribute for that unit of space. Each grid cell is referenced by its xy coordinate location.

Areas of Critical Environmental Concern (ACEC): Areas within the public lands where special management attention is required to protect and prevent irreparable damage to important historic, cultural, or scenic values, fish and wildlife resources or other natural systems or processes.

Assessment Management Team (AMT): A group of BLM managers that provides overall direction and guidance to the REA and makes decisions regarding ecoregional goals, resources of concern, conservation elements, change agents, management questions, tools, methodologies, models, and output work products.

C_3: A plant in which carbon dioxide is first fixed into a compound containing three carbon atoms before completing the photosynthesis cycle.

C_4: A plant in which carbon dioxide is first fixed into a compound containing four carbon atoms before entering the photosynthesis cycle.

C**hange Agent:** An environmental phenomenon or human activity that can alter or influence the future status of resource condition. Some change agents (e.g., roads) are the result of direct human actions or influence. Others (e.g., climate change, wildland fire, and invasive species) may involve natural phenomena or be partially or indirectly related to human activities.

Coarse Filter: A focus of ecoregional analysis that is based upon conserving resource elements that occur at coarse scales, such as ecosystems, rather than upon finer scale elements, such as specific species. The concept behind a coarse filter approach is that preserving coarse-scale conservation elements will also preserve elements occurring at finer spatial scales.

Conceptual models: Conceptual models graphically depict the interactions between a conservation element, the biophysical attributes of its environment, and the change agents that drive ecosystem character. The boxes and arrows that make up the conceptual model represent the state of knowledge about the subject and its relationships to these attributes. Conceptual models are also supported and referenced by scientific literature.

Conservation Element: A renewable resource object of high conservation interest.

Development: A type of change (change agent) resulting from urbanization, industrialization, transportation, mineral extraction, water development, or other human activities that occupy or fragment the landscape or that develop renewable or non-renewable resources.

Ecological Integrity: The ability of an ecological system 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.

Ecoregion: An ecological region or ecoregion is defined as an area with relative homogeneity in ecosystems. Ecoregions depict areas within which the mosaic of ecosystem components (biotic and abiotic as well as terrestrial and aquatic) differs from those of adjacent regions.

Ecoregional Direction: Ecoregional direction uses the information from the Rapid Ecoregional Assessments and stakeholders to develop a broad scale management strategy for an ecoregion's BLM-managed lands.

Fine Filter: A focus of ecoregional analysis that is based upon conserving resource elements that occur at a fine scale, such as specific species. A fine-filter approach is often used in conjunction with a coarse-filter approach (i.e., a coarse filter/fine-filter framework) because coarse filters do not capture every management concerns, such as management of endemic species.

Geographic Information System (GIS): A computer system designed to collect, manage, manipulate, analyze, and display spatially referenced data and associated attributes.

Habitat: A place where an animal or plant normally lives for a substantial part of its life, often characterized by dominant plant forms and/or physical characteristics.

Hydrologic Unit: An identified area of surface drainage within the U.S. system for cataloging drainage areas. The drainage areas are delineated to nest in a multilevel, hierarchical arrangement.

Intactness: Intactness may be mapped as a quantifiable estimate of naturalness according to the level of anthropogenic influence based on available spatial data. Intactness considers an assemblage of spatially explicit indicators that helps define the condition of the natural landscape.

Invasive Species: Species that are not part of (if exotic non-natives) or are a minor component of (if native), an original community that have the potential to become a dominant or co-dominant species if their future establishment and growth are not actively controlled by management interventions, or that are classified as exotic or noxious under state or federal law.

Landscape Species: Landscape species use large, ecologically diverse areas. The species often have significant impacts on the structure and function of natural ecosystems.

Logic Model: A logic model is a cognitive map that presents spatial data components and their logical relationships to explain the process used to evaluate a complex topic. Logic models are constructed in a hierarchical fashion relying on symbols, colors, labels, and the physical arrangement of components to communicate how a series of spatial datasets are assembled and analyzed to answer a particular question.

Management Questions: Questions from decision-makers that usually identify problems and request how to fix or solve those problems.

Model: Any representation, whether verbal, diagrammatic, or mathematical, of an object or phenomenon. Natural resource models typically characterize resource systems in terms of their status and change through time

Native Species: Species that historically occurred or currently occur in a particular ecosystem that were not introduced.

Population: Individuals of the same species that live, interact, and migrate through the same niche and habitat.

Process Models: Process models are diagrams that map out data sources, GIS analyses, and workflow. Process models present the spatial analysis details and allow for repeatability of the same or similar model in the future

Rapid Ecoregional Assessment (REA): The methodology used by the BLM to assemble and synthesize regional-scale resource information, which provides the fundamental knowledge base for devising regional resource goals and priorities on a relatively short time frame (less than 2 years).

Status: The condition of a criterion (biological or socio-economic resource values or conditions) within a geographic area (e.g., watershed, grid). A rating (e.g., low, medium, or high) or ranking (numeric) is assigned to specific criteria to describe status.

Step-Down: A step-down is any action related to regionally-defined goals and priorities discussed in the REA that are acted upon through actions by specific State and/or Field Offices. These step-down actions can be additional inventory, a finer-grained analysis, or a specific management activity.

Acronyms

AM	Arbuscular Mycorrhizal
AMT	Assessment Management Team
AUC	Area Under the Curve
ArcGIS	Arc Geographic Information System
BpS	Biophysical Setting
BLM	Bureau of Land Management
CO_2	Carbon Dioxide
CE	Conservation Element
DEM	Digital Elevation Model
ECHAM5	European Centre Hamburg, Version 5
EMDS	Ecosystem Management Decision Support
EPA	Environmental Protection Agency
ENSO	El Nino Southern Oscillation
EVT	Existing Vegetation Type (LANDFIRE)
FGDC	Federal Geographic Data Committee
FRAGSTATS	Fragmentation Statistics software
FRCC	Fire Regime Condition Classification
G-1, G-3	Globally Imperiled-Globally Vulnerable
GCM	Global Circulation Model
GFDL	Geophysical Fluid Dynamics Laboratory

GENMOM	GENesis-Modular Ocean Model
GIS	Geographical Information System
HMAs	Herd Management Areas
HUC	Hydrologic Unit Classification
IPCC AR4	Intergovernmental Panel on Climate Change Fourth Assessment Report
LAI	Leaf Area Index
LANDFIRE	LANDscape FIRE and Resource Management Planning Tools Project
MAPSS	Mapped Atmosphere Plant Soil System
MaxEnt	Maximum Entropy model
MQ	Management Question
NCAR	National Center for Atmospheric Research
NCEP	National Centers for Environmental Prediction
NetCDF	Network Common Data Form
NHD	National Hydrography Dataset
NREL	National Renewable Energy Laboratory
OHV	Off-Highway Vehicles
PET	Potential Evapotranspiration
PFT	Plant Functional Type
PRISM	Parameter-elevation Regressions on Independent Slopes Model
REA	Rapid Ecoregional Assessment
RegCM3	Regional Climate Model Version 3
RMP	Resource Management Plan
SSURGO	Soil Survey Geographic database
STATSGO	State Soil Geographic
SOW	Statement of Work
SW ReGAP	Southwest Regional Gap Analysis Project
TNC	The Nature Conservancy
USDA	U.S. Department of Agriculture
USFWS	U.S. Fish and Wildlife Service
USGS	U.S. Geological Survey