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.

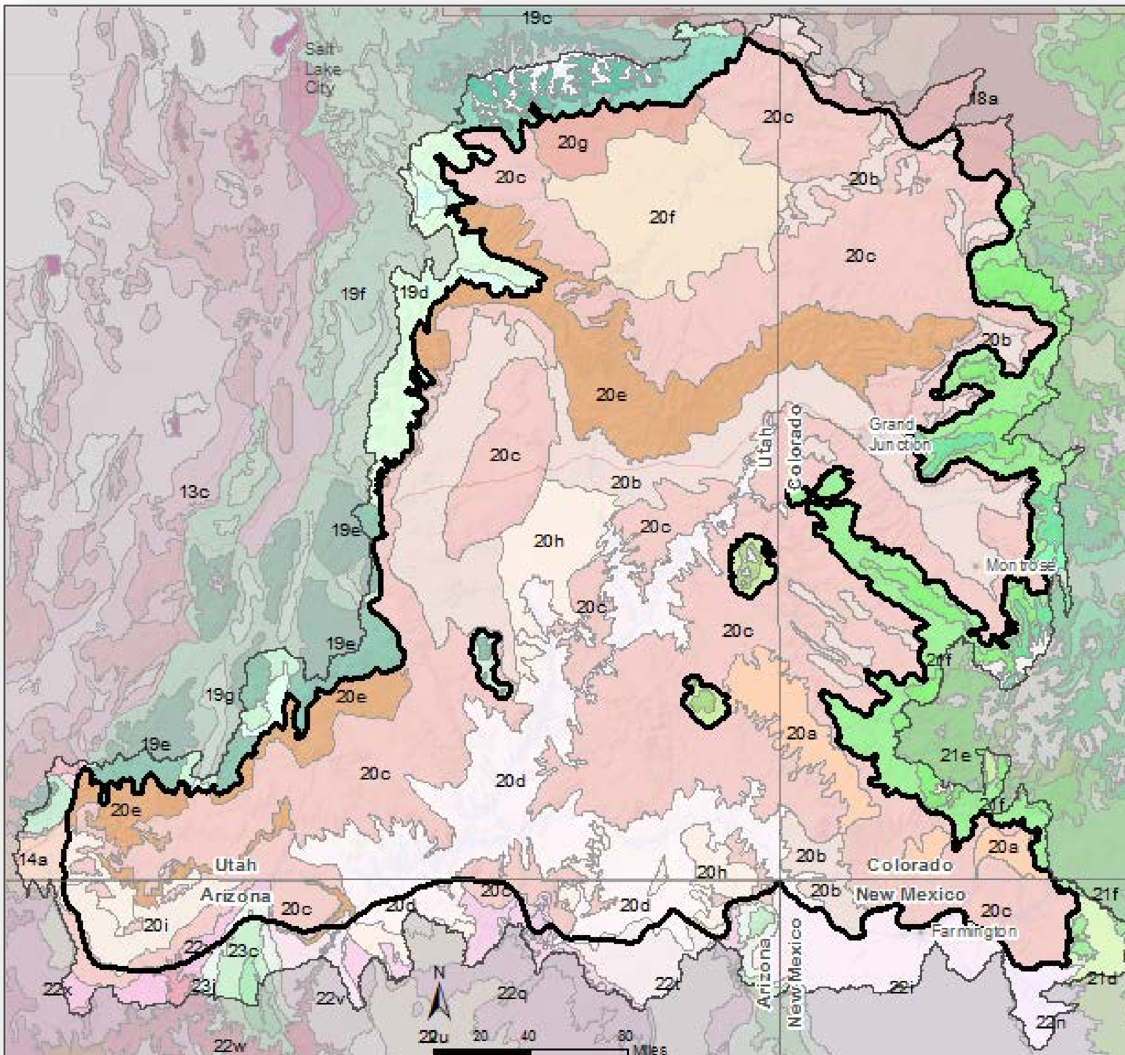


Figure 4-1. Level IV ecoregions of the Colorado Plateau. U.S. EPA, ftp://ftp.epa.gov/wed/ecoregions/

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).

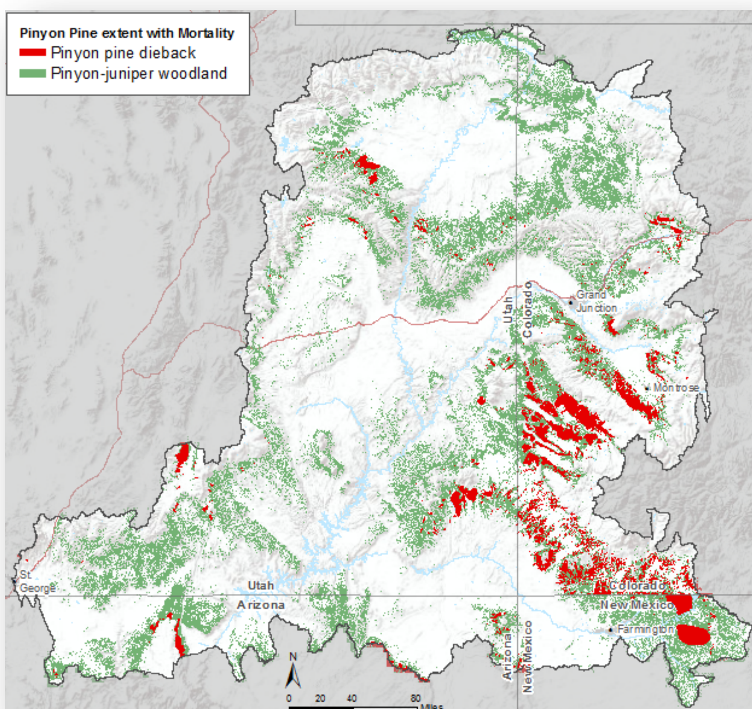


Figure 4-2. Pinyon pine mortality, 2000–2007. U.S. Forest Service Forest Health Technology Enterprise Team

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).

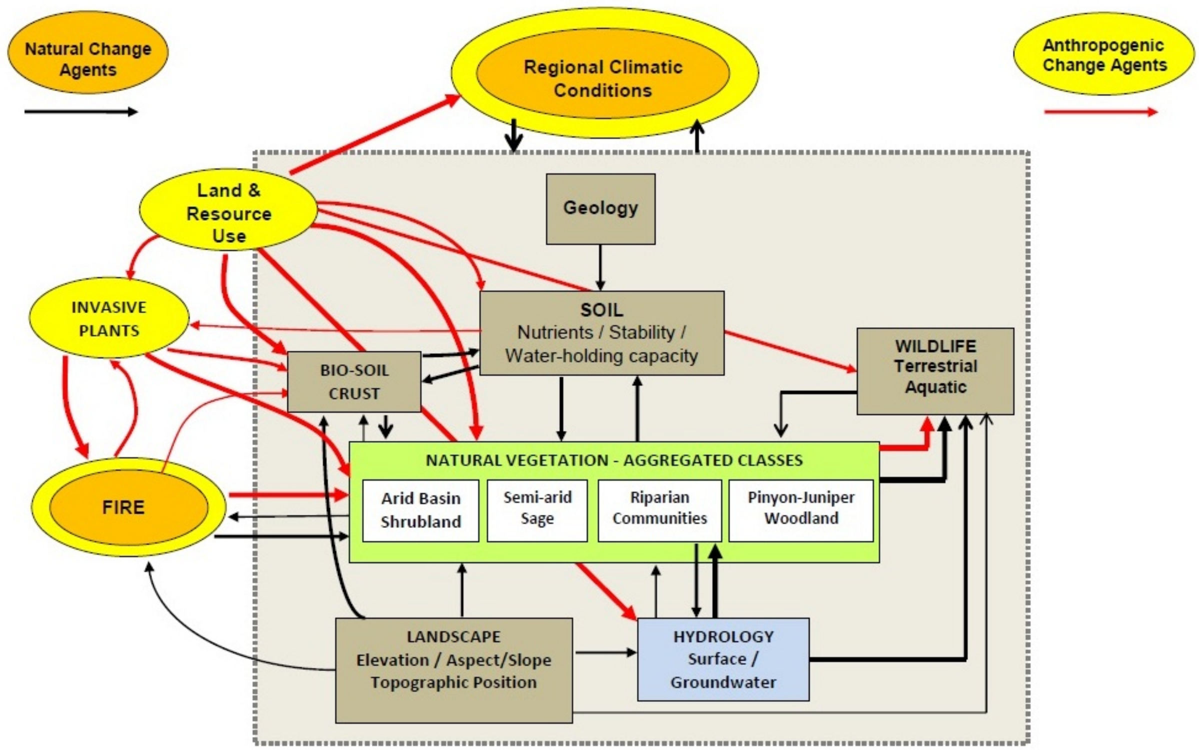


Figure 4-3. Basic ecoregion conceptual model for the Colorado Plateau Ecoregion, with both natural and anthropogenic change agents shown.

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).

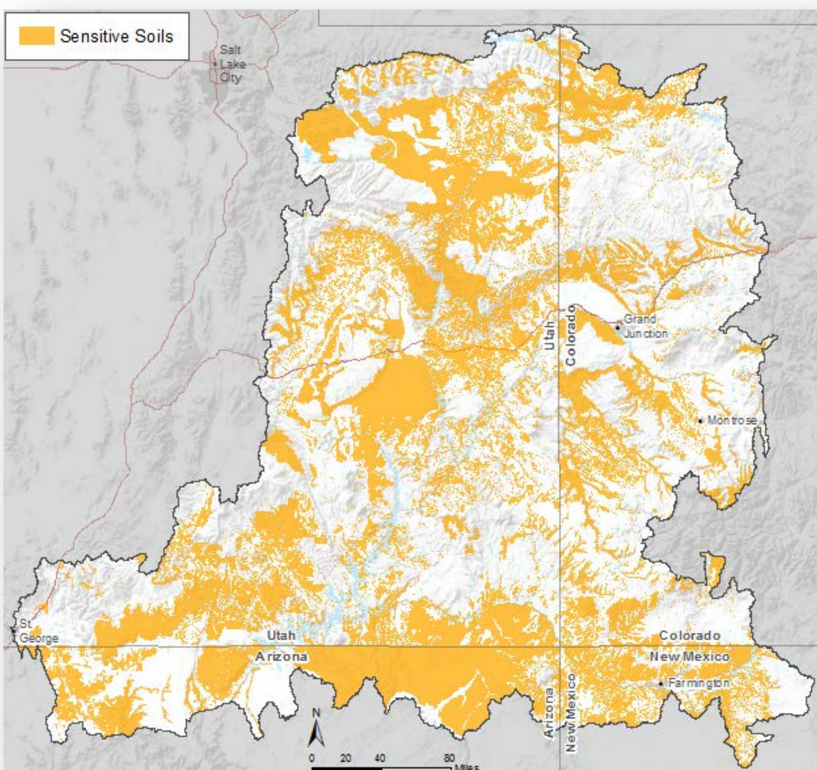


Figure 4-4. Map showing all classes of sensitive soils, including droughty, shallow, hydric, gypsiferous, salty, and high calcium carbonate (calcareous). Large polygon in Arizona reflects availability of only coarser resolution STATSGO soil data. See Appendix A for modeling details.

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.

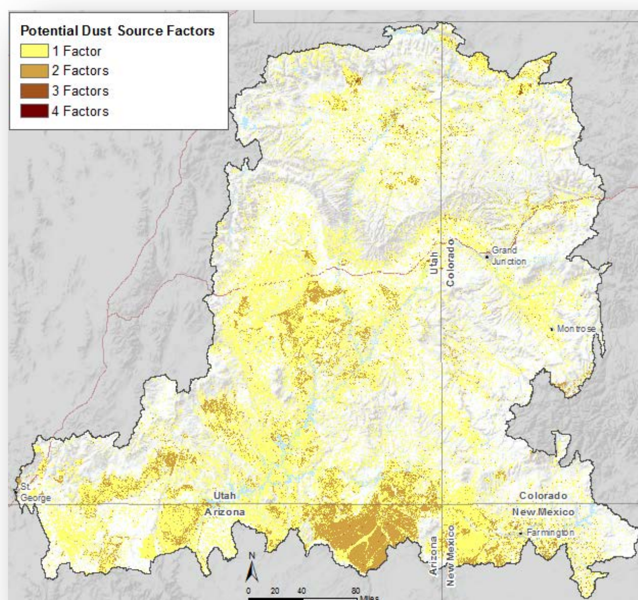


Figure 4-5. Map depicting sources or hotspots producing fugitive dust that may contribute to dust on snow in the southern Rocky Mountains.

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.

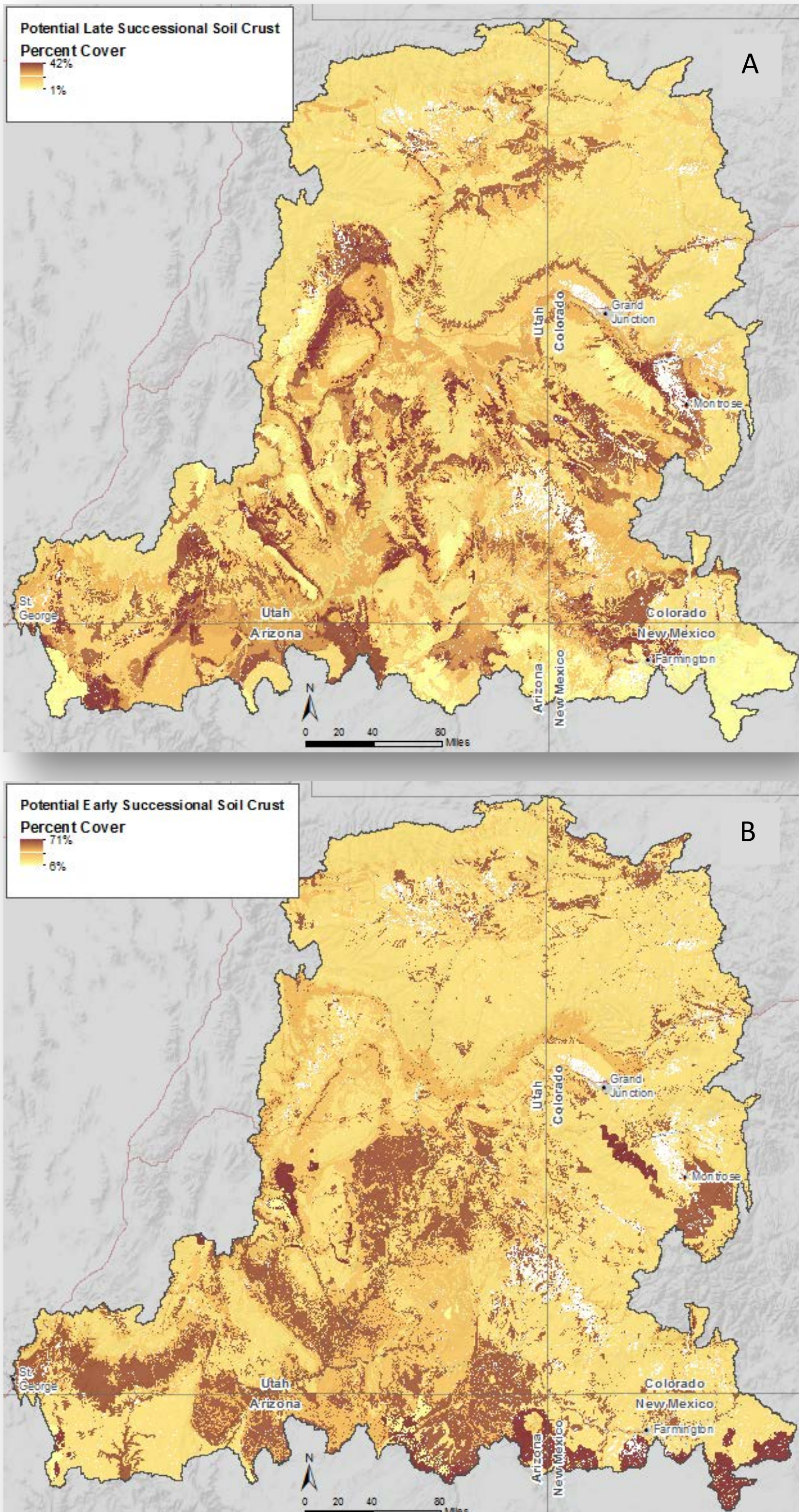


Figure 4-6. Map of late (A) and early (B) successional biological crust for the Colorado Plateau ecoregion. Model and portions of text contributed by M. Bowker and T. Arundel, U.S. Geological Survey.

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?

Photo: Green River, Desolation Canyon. Utah BLM

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.

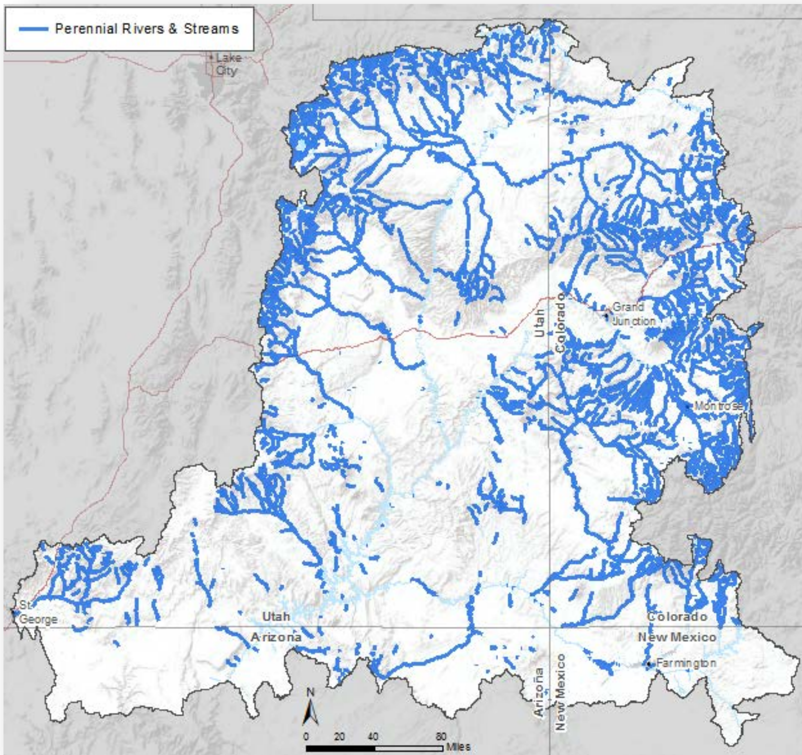


Figure 4-7. Map for management questions B2 shows perennial streams in the Colorado Plateau ecoregion. Mainstem Colorado and Green rivers are in light blue. Data from the National Hydrography Dataset typically over-represent perennial streams because of mapping error or loss of perennial streams over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

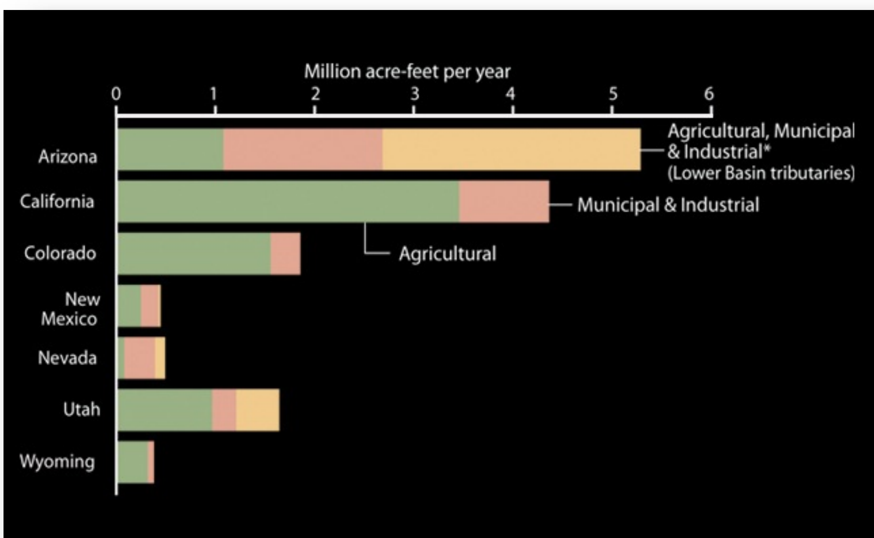


Figure 4-8. Water consumption of states of the upper and lower Colorado River Basin for agriculture (green), municipal and industrial use (pink), and all usage from Colorado River tributaries (yellow, data not recorded by usage class). Data from U.S. Bureau of Reclamation.
National Geographic website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

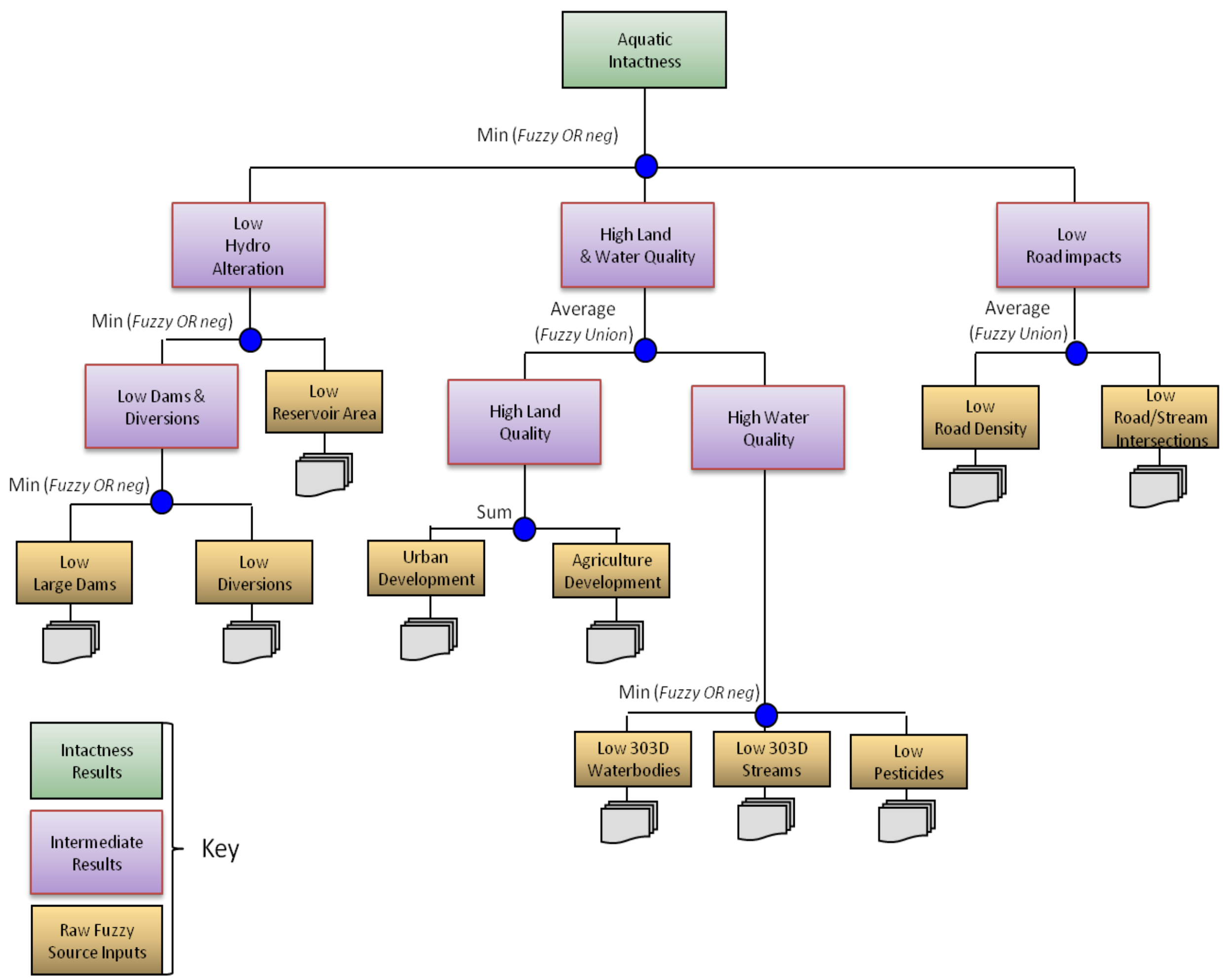


Figure 4-9. Fuzzy logic model for aquatic intactness in the Colorado Plateau ecoregion. Gold boxes represent raw primary data input and purple boxes represent intermediate results (Figure 4-10).

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 5th 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).

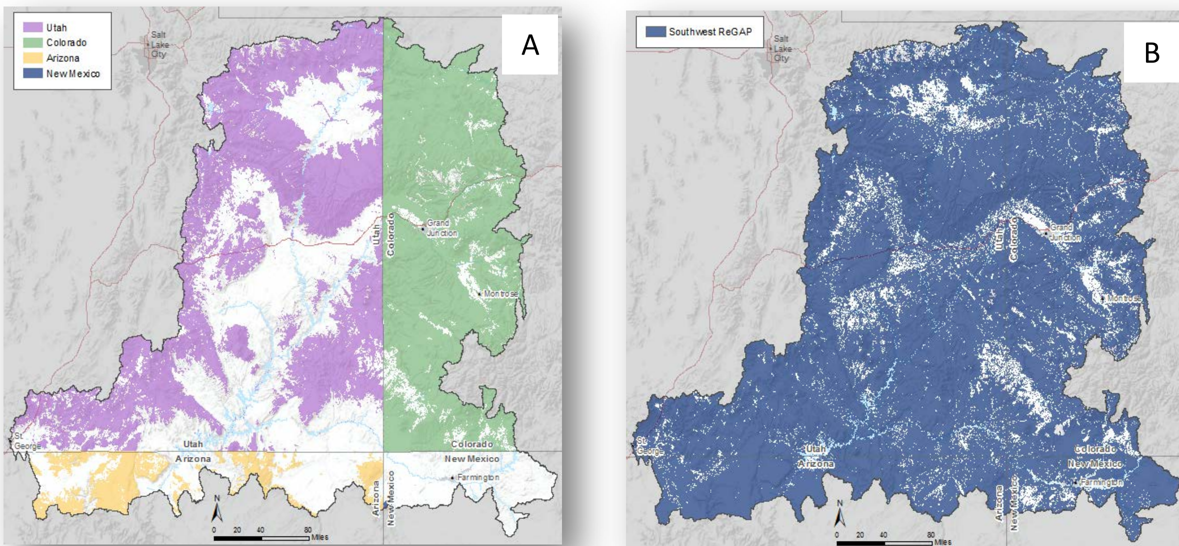


Figure 4-11. (A) Mountain lion distribution acquired from state wildlife agencies or state GAP and (B) Mountain lion distribution according to Southwest ReGAP.

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.

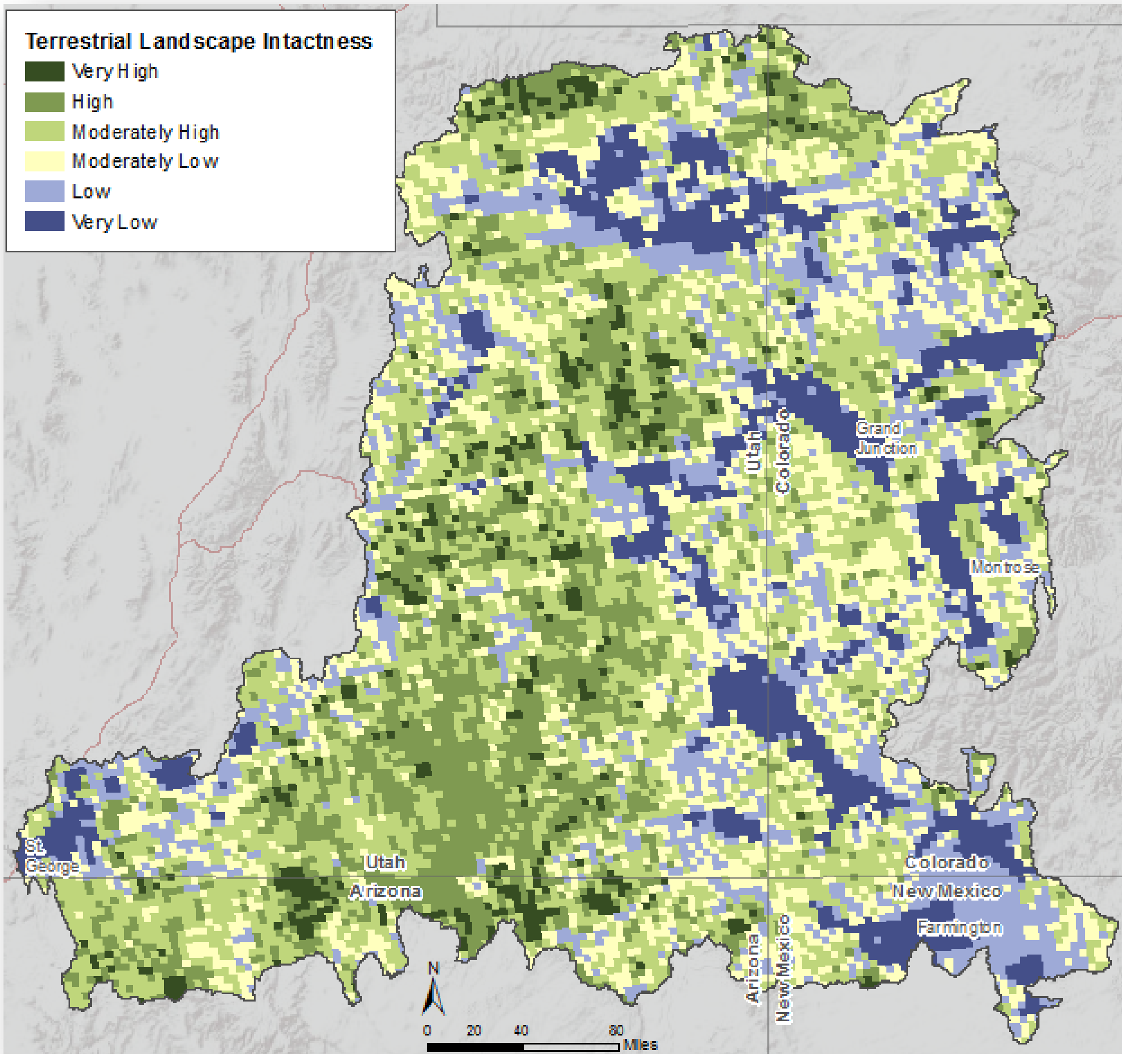
Current terrestrial landscape intactness at 4 km x 4 km resolution (Figure 4-12) and aquatic intactness organized by 5th level HUC (Figure 4-13) for the Colorado Plateau ecoregion show the full range of values from very low to very high intactness and their distribution in the accompanying histogram. 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.



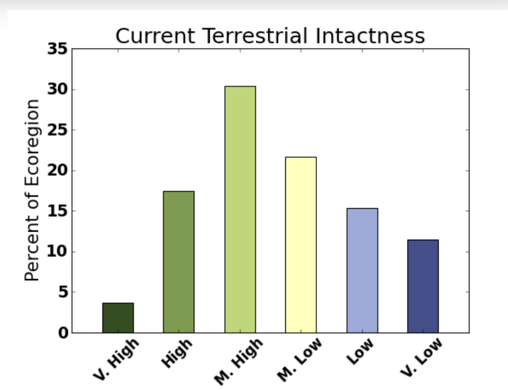
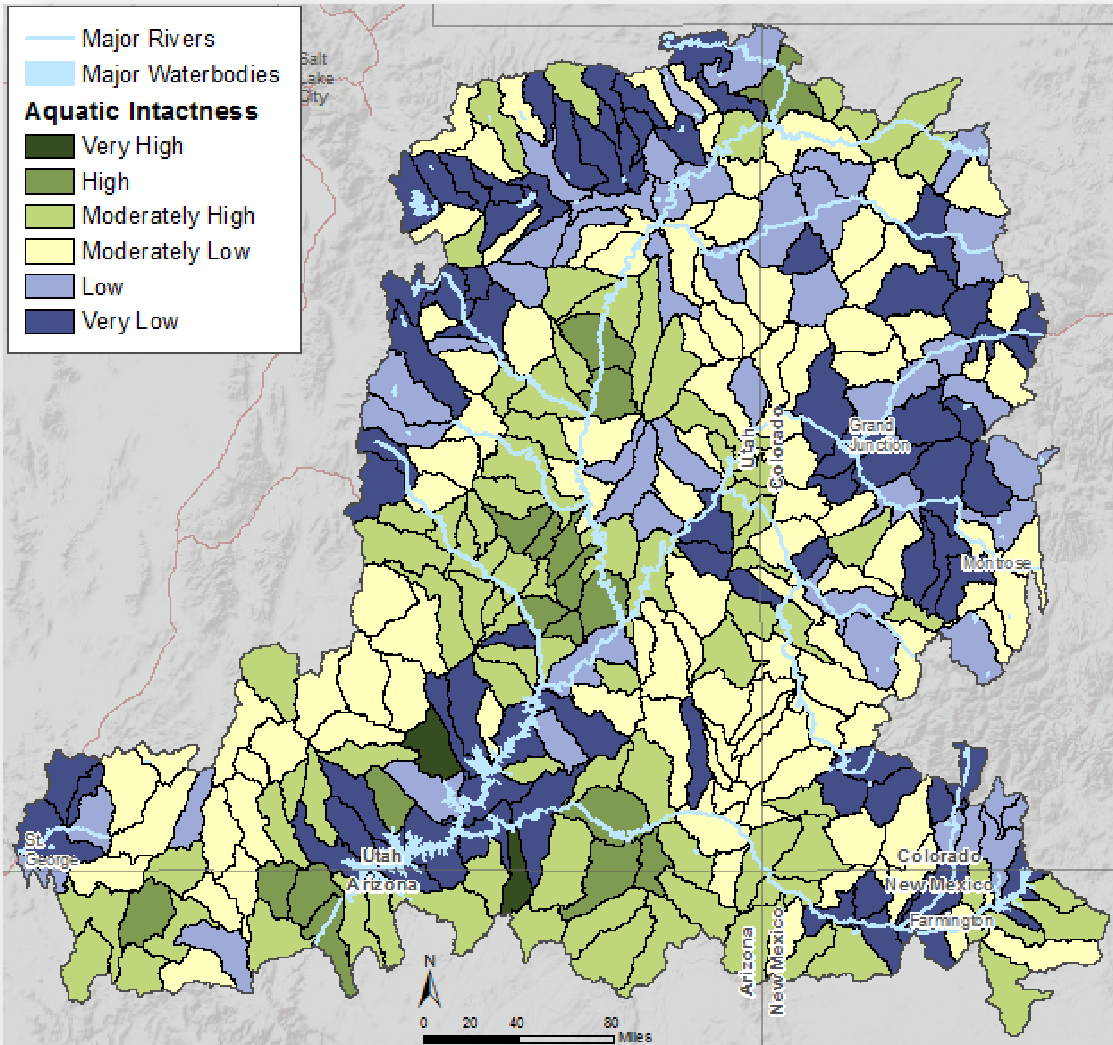


Figure 4-12. Terrestrial landscape intactness results organized in six categories by 4 km X 4 km grid cells for the Colorado Plateau ecoregion with associated histogram.



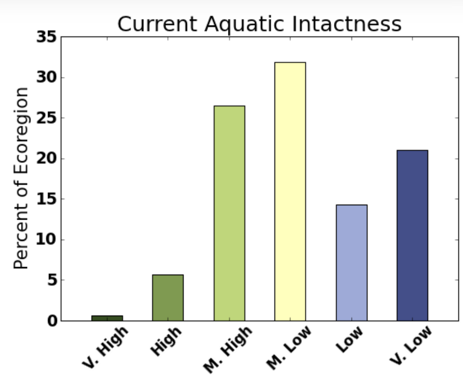


Figure 4-13. Aquatic intactness results organized by 5th level HUCs for the Colorado Plateau ecoregion and associated histogram.

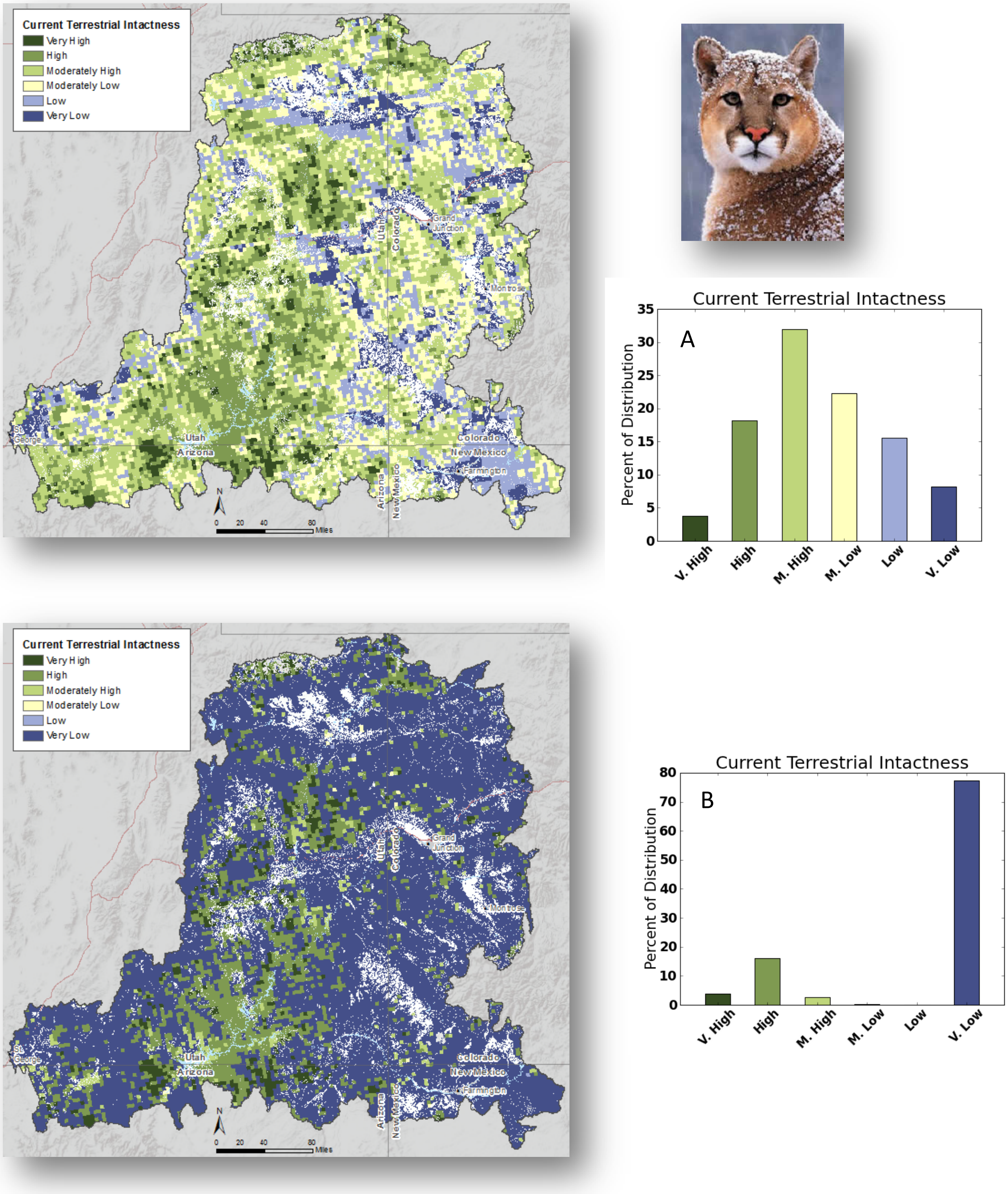
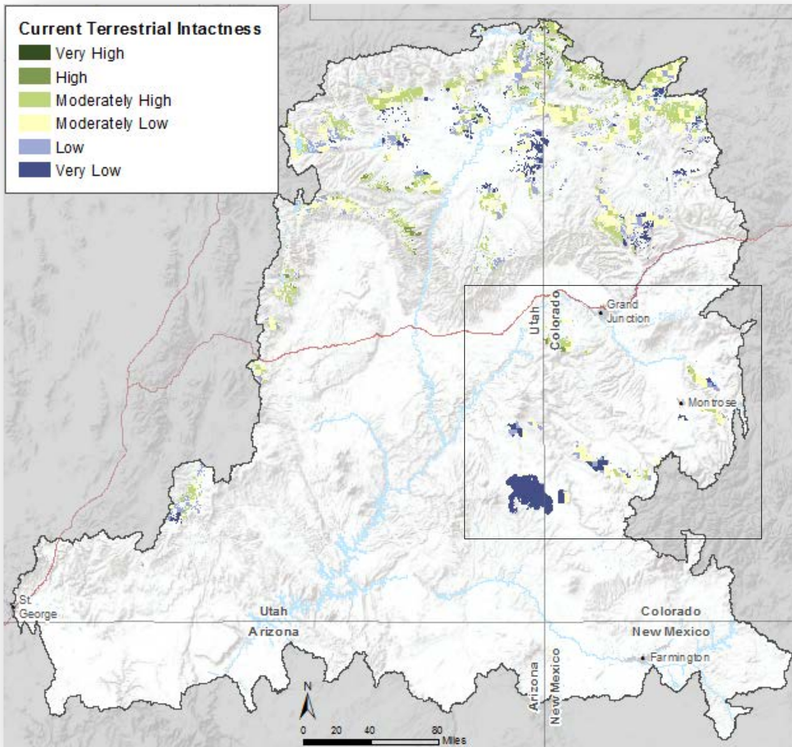
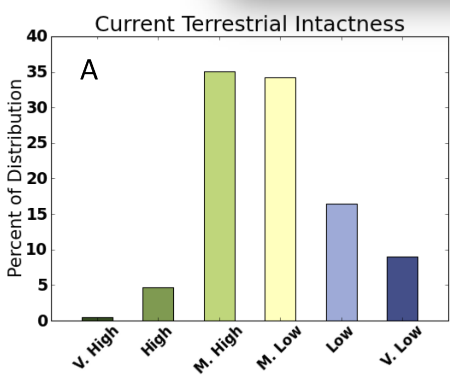
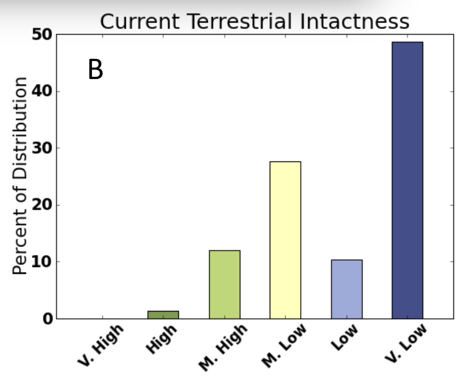


Figure 4-14. Map shows A) mountain lion status created by overlaying current distribution against the general terrestrial intactness model; and B) mountain lion status according to the customized intactness model, with a road density tolerance of 0.6 km/km^2 (Van Dyke et al. 1986), both organized by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







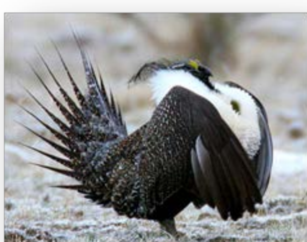


Figure 4-15. Map and histograms show results for greater and Gunnison sage-grouse status, using a threshold for oil and gas well density of >12 well pads per 4 km X 4 km grid cell in the terrestrial landscape intactness model (Doherty [2008], see Sage-grouse Case Study Insert for more details). Map and 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).

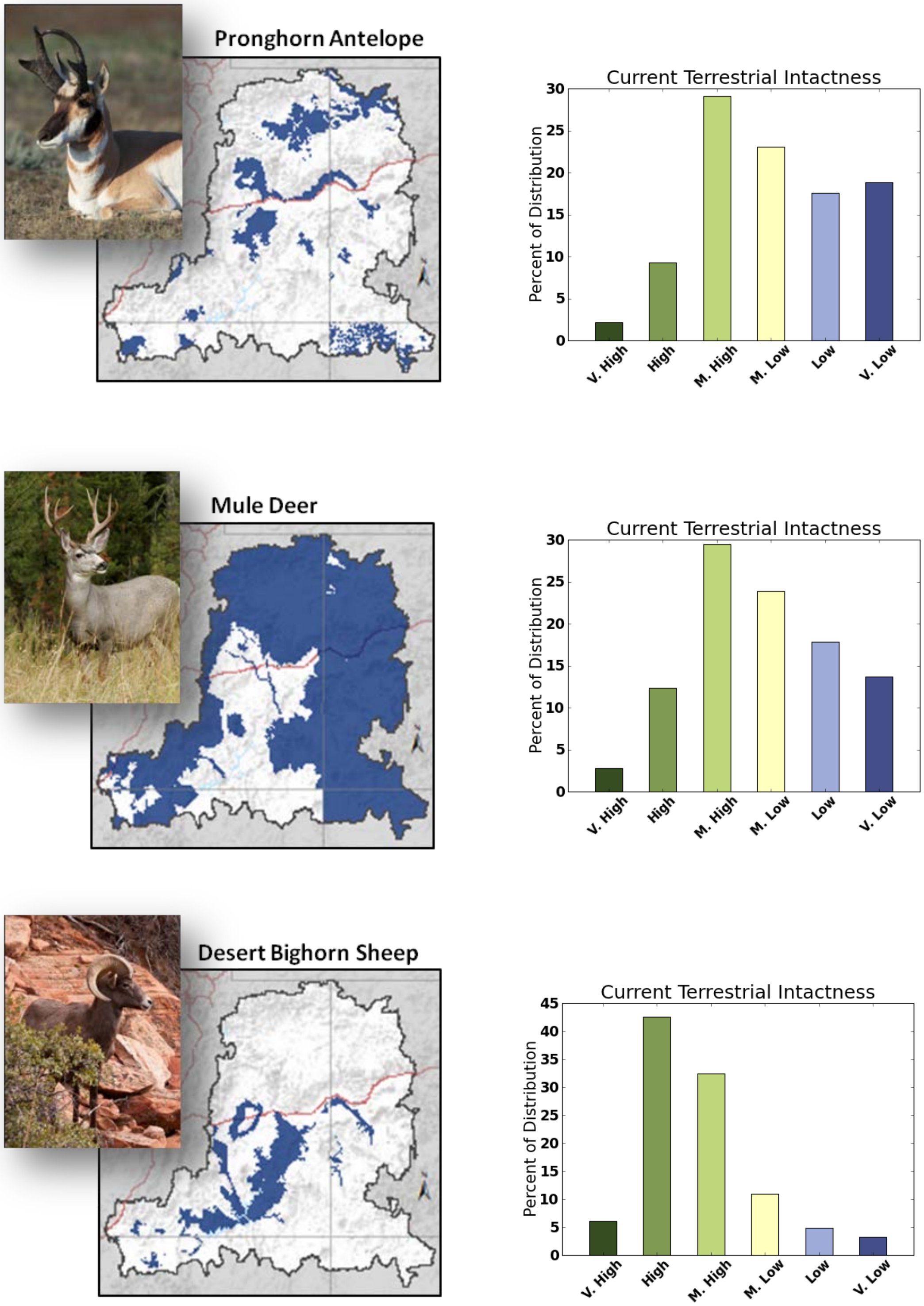


Figure 4-16. Current species distribution (in blue on maps) and conservation element status (histogram) based on current terrestrial intactness model for pronghorn antelope, mule deer, and desert bighorn sheep in the Colorado Plateau ecoregion.

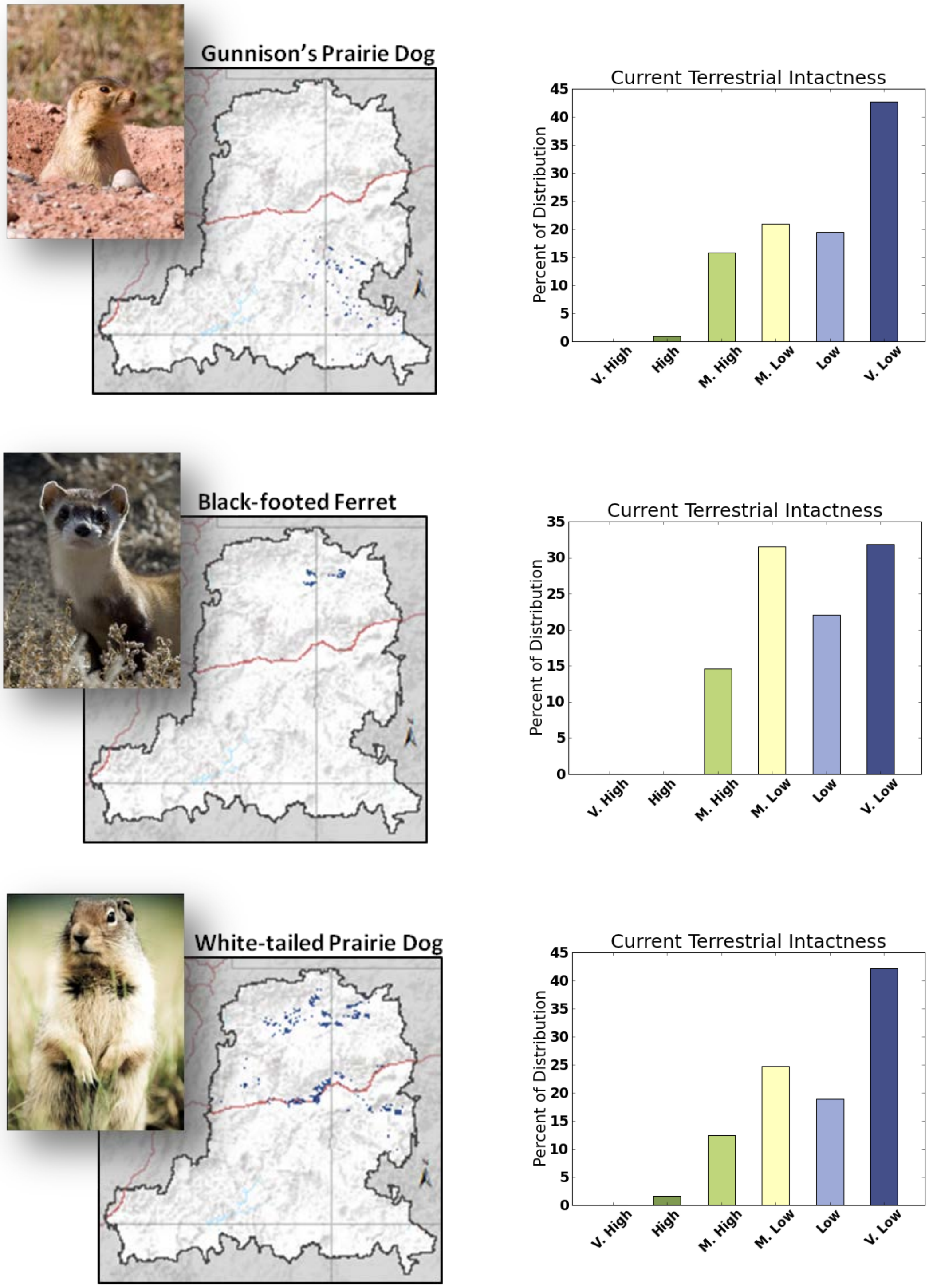


Figure 4-17. Current distribution (in blue on maps) and conservation element status (histogram) based on current terrestrial intactness model for Gunnison's prairie dog, black-footed ferret, and white-tailed prairie dog in the Colorado Plateau ecoregion.

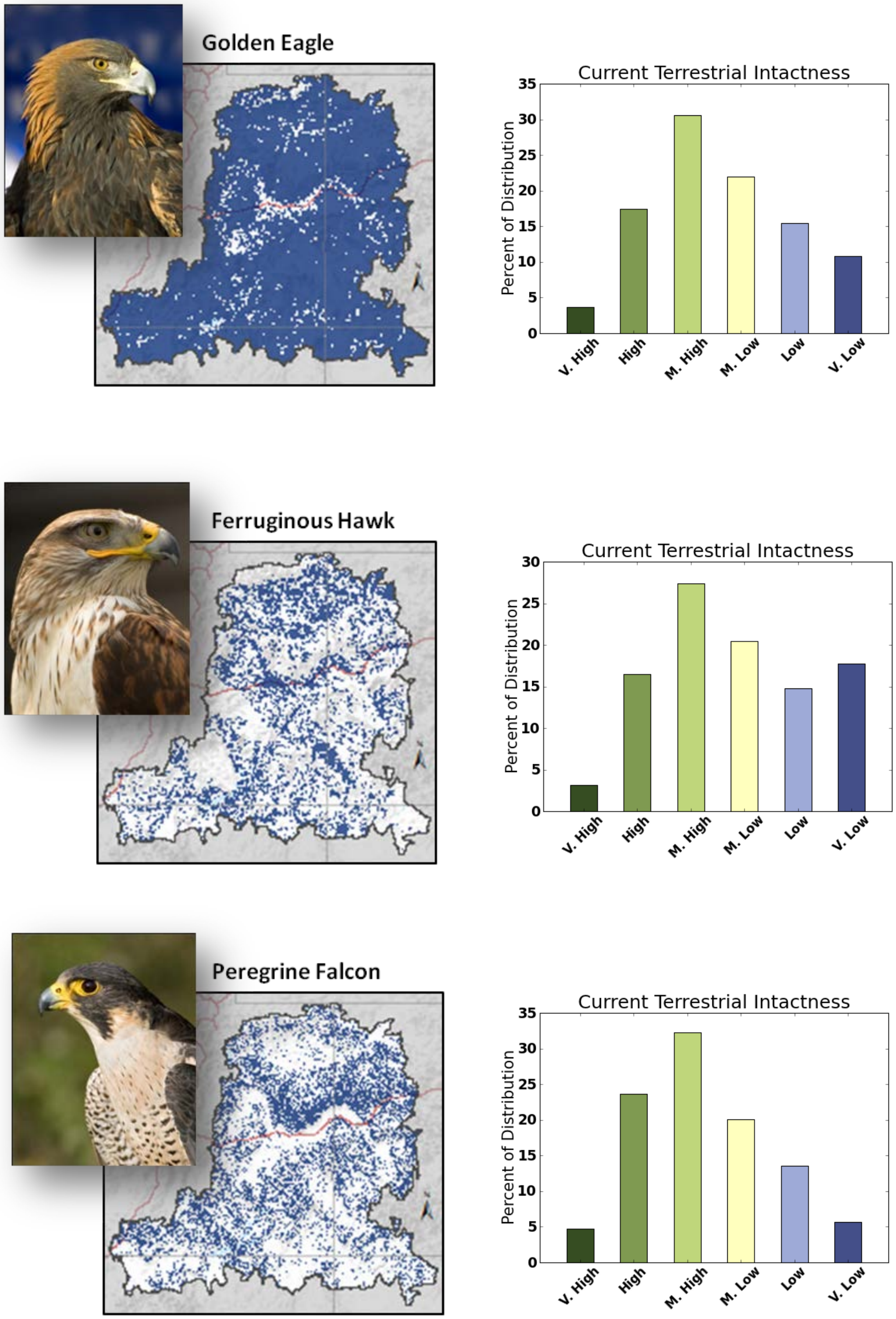


Figure 4-18. Current distribution (in blue on maps) and conservation element status (histogram) based on current terrestrial intactness model for golden eagle, ferruginous hawk, and peregrine falcon in the Colorado Plateau ecoregion.

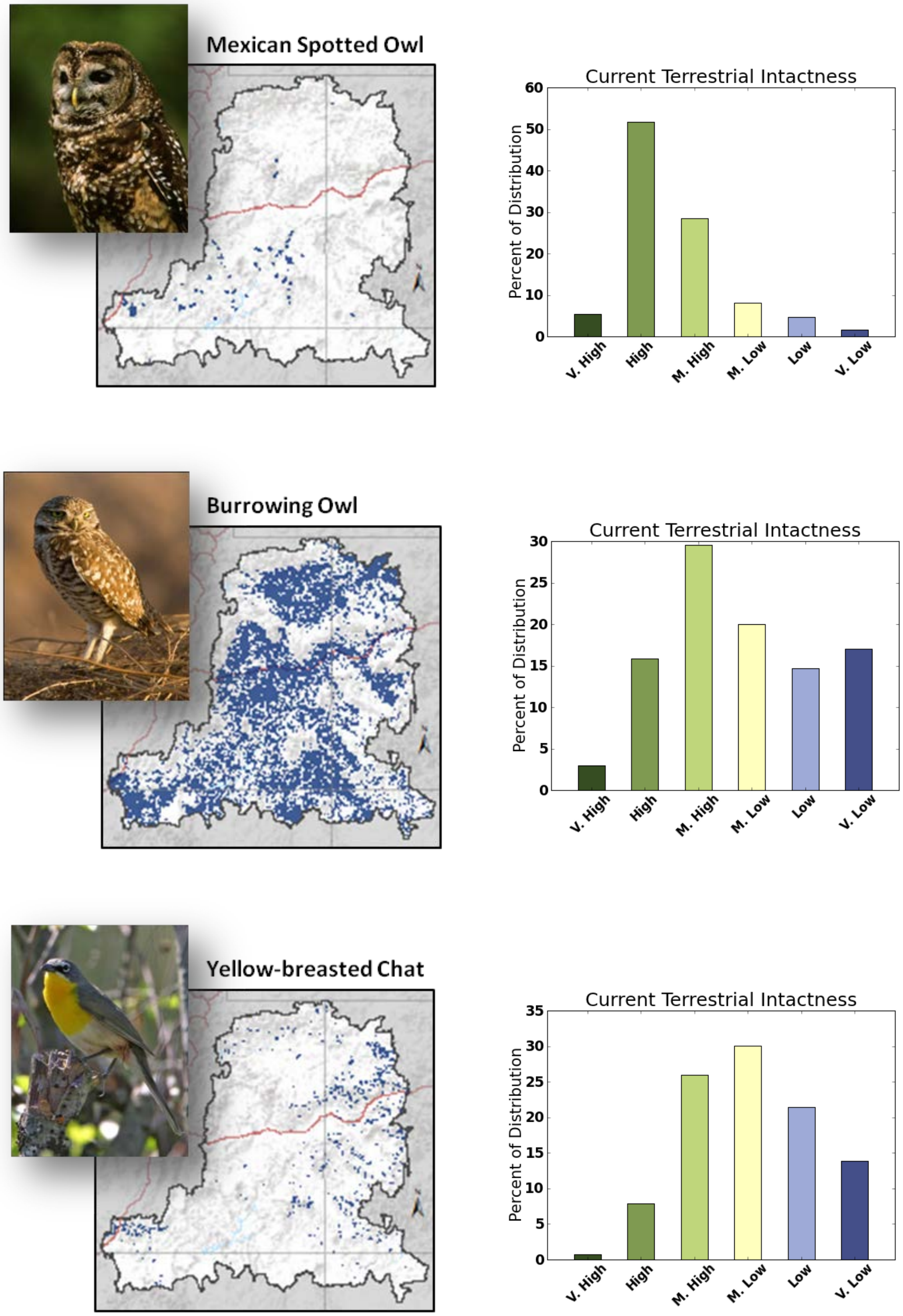


Figure 4-19. Current distribution (in blue on maps) and conservation element status (histogram) based on current terrestrial intactness model for Mexican spotted owl, burrowing owl, and yellow-breasted chat in the Colorado Plateau ecoregion.

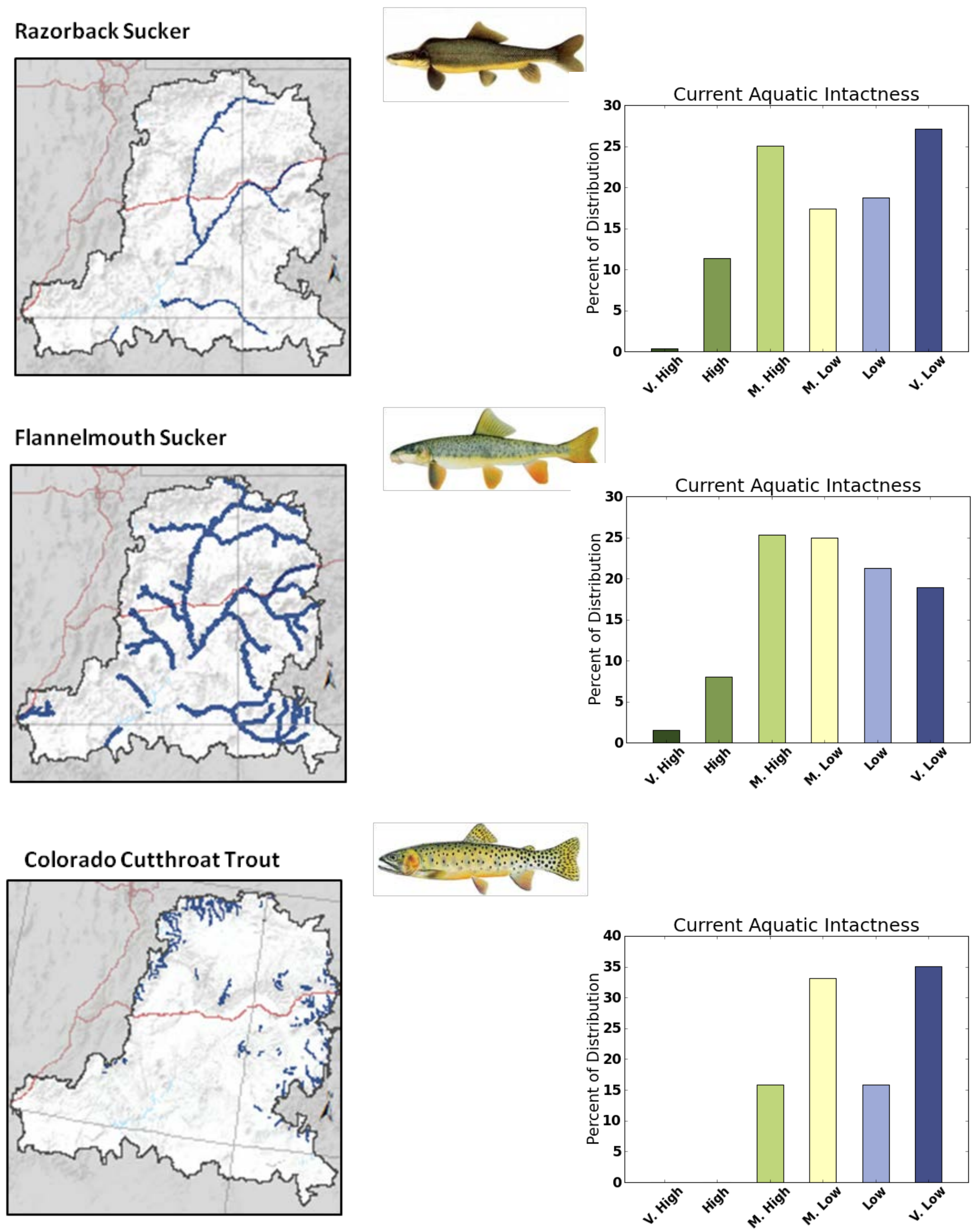


Figure 4-20. Current distribution (in blue on maps) and conservation element status (histogram) based on current aquatic intactness for razorback sucker, flannelmouth sucker, and Colorado River cutthroat trout in the Colorado Plateau ecoregion. See Appendix C for data sources for fish species distributions.

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