

Appendix C – Species Conservation Elements

Organization of Appendix C

For each conservation element, we provide some background information, a conceptual model, description of the analytical process (including source data) and/or a Process Model for each management question, and results in the form of maps and other supporting graphics. Access to a data portal to examine the results in greater detail is available at: <http://www.blm.gov/wo/st/en/prog/more/climatechange.html>.

Species Conceptual Models

Conceptual models attempt to organize and articulate the relationship between the various change agents and natural drivers for each conservation element. Not all of the relationships identified lend themselves well to measurement or monitoring, but they are still important to include, as they add to our general understanding of complex interactions.

All conceptual models include a series of change agents (depicted with yellow boxes) and natural drivers (cyan boxes). Specifics regarding some of the factors are presented in blue text. Arrows represent relationships between the various change agents and natural drivers on the community overall and, where appropriate, on the dominant species more directly. More specific information is provided by the orange text. Thicknesses of the arrows **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.

Species Process Models

Two basic management questions were addressed for each species conservation element. The first question pertained to current distribution and status. The second question referred to potential impact on the species from near-term (2025) future change, impact from potential energy development, and long-term potential-for-change (2060) from climate change. The basic method for each species was similar, but, for current distribution, input data varied in source and quality. Source data for each is provided in the introduction for each species. Current status was determined by overlaying current distribution against terrestrial landscape intactness (Chapter 4) for terrestrial species.

For potential future condition, current distribution was evaluated in a similar fashion against potential energy development (Chapter 5, Section 5.2), near-term (2025) terrestrial landscape intactness (Chapter 5, Section 5.3), and climate change model results (Chapter 5, Section 5.4).



Bell's vireo is represented in the Sonoran REA as two distinct subspecies, Arizona Bell's vireo (*Vireo bellii arizonae*) and least Bell's vireo (*Vireo bellii pusillus*). In the Sonoran Desert ecoregion, Arizona Bell's vireo occurs east of the Colorado River in southern Arizona, and in California it is confined to the west bank and floodplain of the lower Colorado River. Least Bell's vireo occurs on the western edge of the ecoregion in California, in riparian areas of streams draining the lower slopes of the San Jacinto and Santa Rosa mountains (Patten 1998). Subspecies *arizonae* is state-listed as endangered in California (not listed in Arizona), and *pusillus* is both state and federally listed as endangered in California. Least Bell's vireo experienced a steady decline in California throughout the mid- to late- 20th century. The species was extirpated from the San Joaquin and Sacramento valleys by the mid-1950s (Goldwasser 1978). North American Breeding Bird Survey trend results are not reliable for least Bell's vireo because of its low numbers; but results for Arizona Bell's vireo have higher confidence and indicate a 2.0 % yearly percentage decline in Bell's vireo abundances in Arizona from 1966–2009 (Sauer et al. 2011).

Bell's vireo inhabits early successional riparian thickets. Depending on location, the dominant vegetation may be cottonwood (*Populus fremontii*), willow (*Salix* spp.), mesquite (*Prosopis* spp.), elderberry (*Sambucus* spp.), or seep willow (*Baccharis salicifolius*). Although many report that Arizona Bell's vireo will use invasive tamarisk (*Tamarix* spp.) for nesting (Brown and Trosset 1989, Patten 1998, GBBO 2010, CEC 2011), Averill-Murray and Corman (2005) state that “few” Bell's vireos were found in tamarisk thickets during 7 years of canvassing for the Arizona Breeding Bird Atlas. In Arizona, besides moist riparian areas, vireos inhabit drier mesquite bosques and desert washes containing blue paloverde (*Parkinsonia floridum*), ironwood (*Olneya tesota*), and netleaf hackberry (*Celtis reticulata*, Averill-Murray and Corman 2005). Dense, low shrubby habitat structure, independent of vegetation type, is important to the vireo because it nests and forages for insects close to the ground (0.6–3 m, Goldwasser 1978). Loss of riparian (and xeroriparian) shrub cover is a major threat to the continued productivity of both vireo subspecies; riparian habitat loss has occurred and continues due to agricultural practices, livestock grazing, stream channelization, flood control projects, conversion of riparian habitat to parks and golf courses, water diversion, and lowering of groundwater levels from groundwater pumping (Patten 1998, USFWS 2006, GBBO 2010).

However, depletion of riparian habitat is not the only disturbance affecting the survival of Bell's vireo. In a survey of 21 streams in 7 southern California counties, Goldwasser (1978) found that just 19% of potential nesting habitat was occupied and suggested that other factors, such as nest predation and brood parasitism by brown-headed cowbird (*Molothrus ater*)—accounting for 27% and 46%, respectively, of the nest failures recorded in the study—were responsible for the species' precipitate decline. Although cowbirds are not as numerous in the range of Arizona Bell's vireo (Clark 1988, Patten 1998), they still have a negative impact on the species, as they do on other common cowbird hosts, yellow warbler (*Dendroica petechia*), yellow-breasted chat (*Icteria virens*), and willow flycatcher (*Empidonax traillii*). During the Arizona Breeding Bird Atlas survey, Bell's vireo was recorded as the top cowbird host (Averill-Murray and Corman 2005).

Bell's vireo readily responds to habitat restoration as well as to brown-headed cowbird control efforts. Goldwasser (1978) noted that male birds sang in newly-established riparian habitat (< 8 years old), suggesting that the birds might respond fairly quickly to riparian vegetation restoration efforts. Recently, multiple county and state agencies collaborated in the Santa Ana River basin to remove giant reed (*Arundo donax*), which had formed a riparian monoculture along streams and rivers. Following giant reed removal, nesting vireo pairs in the basin increased from only 19 in 1986 to 413 in 2004 (IWAC 2006). Other riparian restoration and cowbird removal efforts in San Diego County on the Santa Margarita River and in Riverside County in the Prado Basin also resulted in significant increases in breeding least Bell's vireo populations (Patten 1998). Overall, in southern California, there has been a 10 fold increase in Least Bell's vireo abundance since listing in 1986; the US Fish and Wildlife Service (2006) determined that the successes resulted from a combination of riparian restoration and cowbird control that may need to continue into the foreseeable future to sustain the species. However, the success of the least Bell's vireo recovery effort in California indicates that similar efforts in the lower Colorado River basin may help to reverse the decline of Arizona Bell's vireo. As an example from a recent study, riparian shrub density increased and Arizona Bell's vireo abundance increased from 0.91 to 2.69 detections/km four years after cattle were removed from a section of the San Pedro River in Arizona (Krueper et al. 2003).

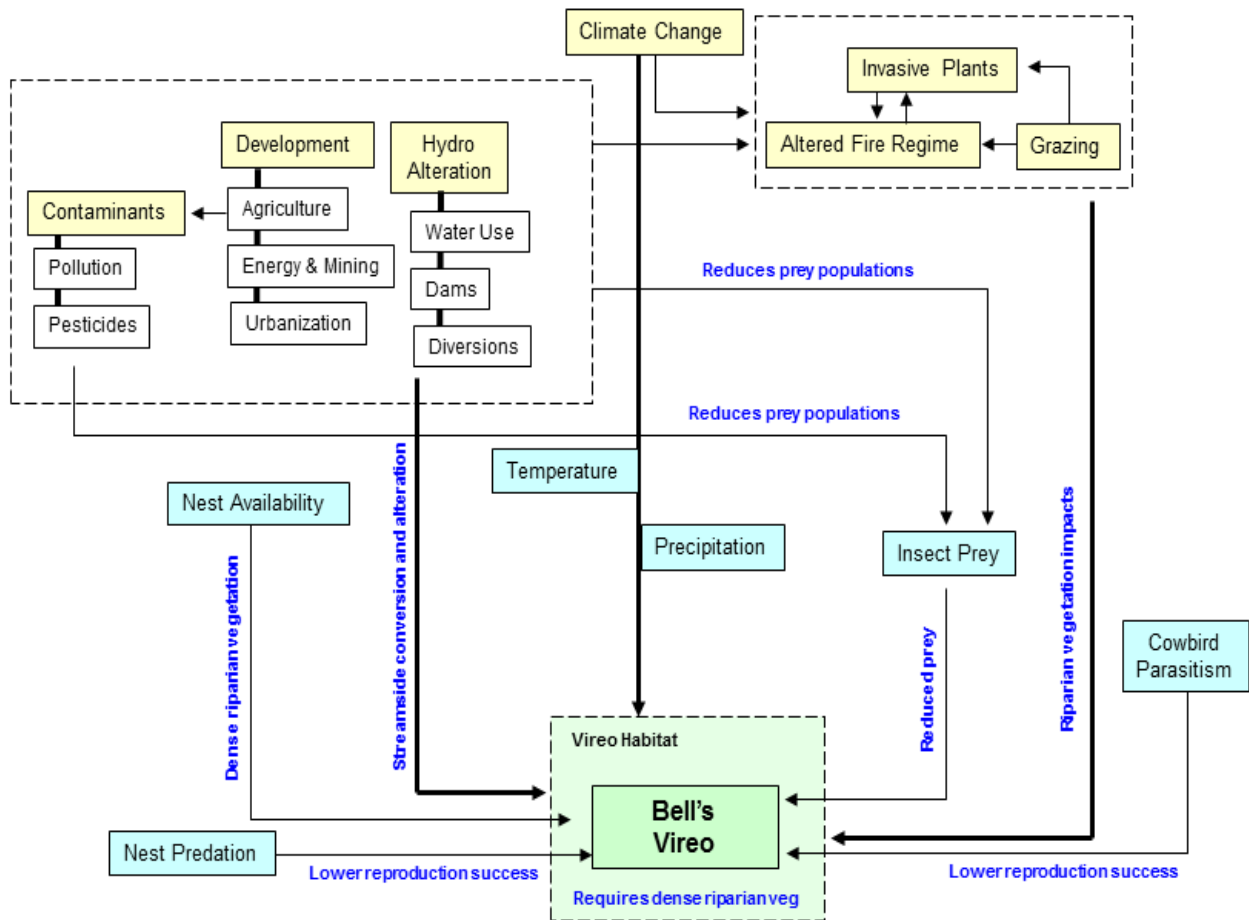
Attributes and Indicators

Attribute	Indicator	Indicator Rating				Citation
		Poor	Fair	Good	Very Good	
habitat	riparian vegetation	No shrub cover			Dense shrub understory up to 3 m [10 ft] high; tree overstory either relatively open or absent	Averill-Murray et al. 1999
habitat degradation	water diversion - distance to water				< 1,000 m [0.6 mi] from water; standing water is an important habitat element	Kus et al. 2010
parasitism	cowbird abundance	increased parasitism with decreased density of understory vegetation			parasitism decreases with increased density of understory vegetation	Kus et al. 2010

References Cited

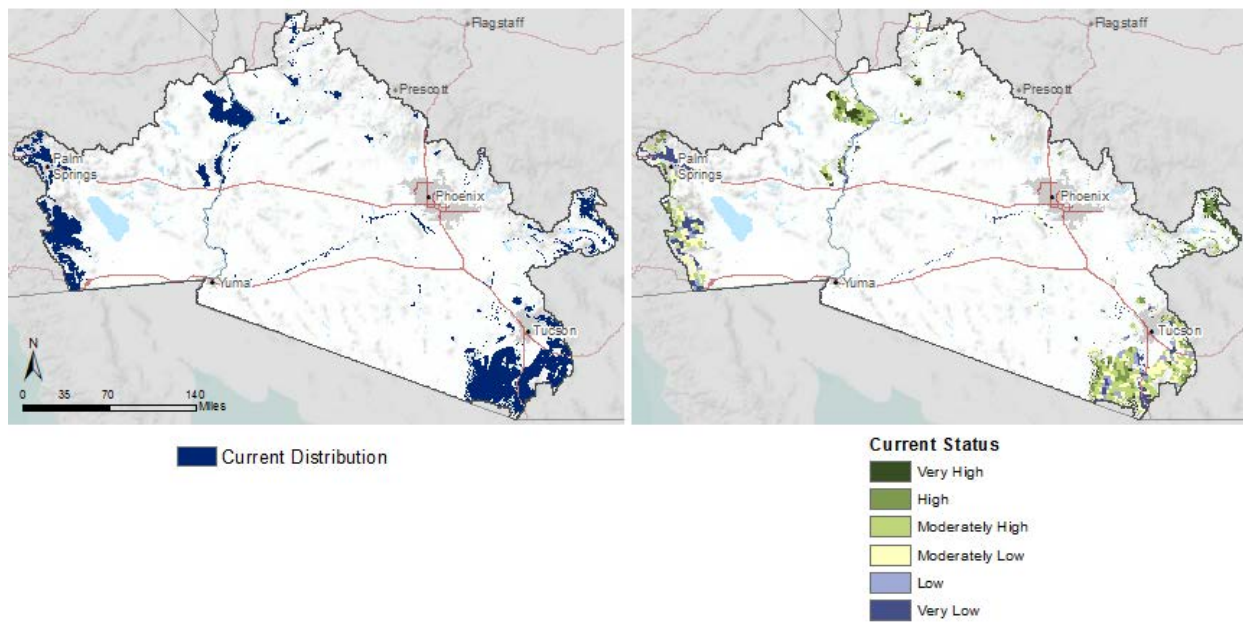
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Bell's Vireo Conceptual Model



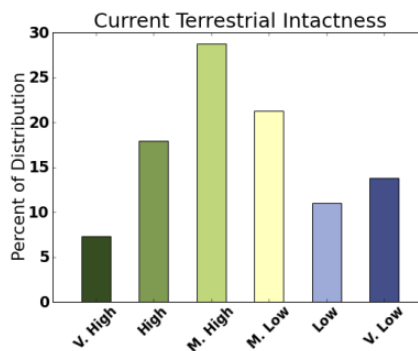
Thick arrows denote components with available spatially explicit data – NOT importance

MQ D1. What are the current distribution and status of Bell's vireo (and historic occupied habitat if available), seasonal and breeding habitat, and movement corridors (as applicable)?



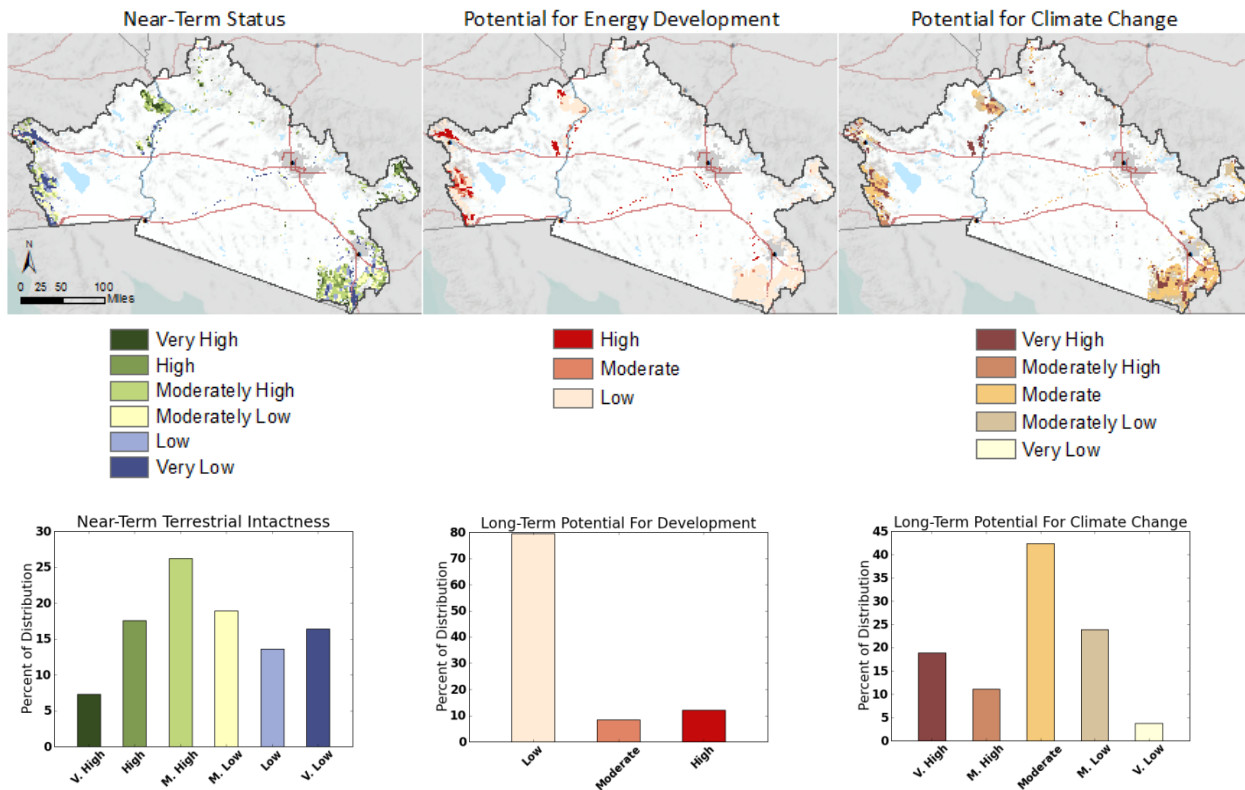
Data Sources:

Arizona GAP and California GAP

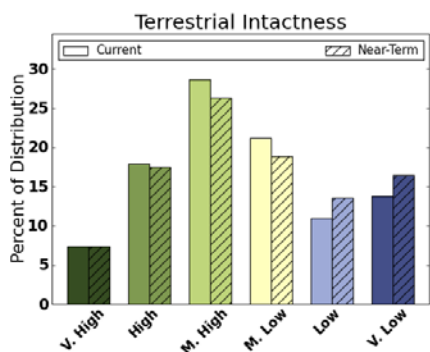


MQ D6. What sites and movement corridors are vulnerable to change agents in the near-term horizon, 2025 (development, fire, invasive species) and long-term change horizon, 2060 (climate change)?

Bell's Vireo Potential for Change



Current & Near-term Intactness



Desert Bighorn Sheep – *Ovis canadensis nelsoni*



Attributes and Indicators

Attribute	Indicator	Indicator Rating				Citation
		Poor	Fair	Good	Very Good	
habitat	cover & terrain	forest/thick brush; lack of precipitous escape terrain			visually open with steep, rocky slopes	Sierra Nevada Bighorn Sheep Foundation; Beecham et al 2007
Disease	Proximity to domestic livestock				a minimum of 13.5 km between sheep & domestic livestock	Beecham et al, 2007; Singer et al, 2001
Habitat	Habitat fragmentation	Increased human disturbance			Little to no human disturbance	Beecham et al, 2007; King and Workman 1985
Climate	effect on vegetation	higher temperatures - decreased precipitation			normal to higher levels of rainfall	Beecham et al, 2007

References

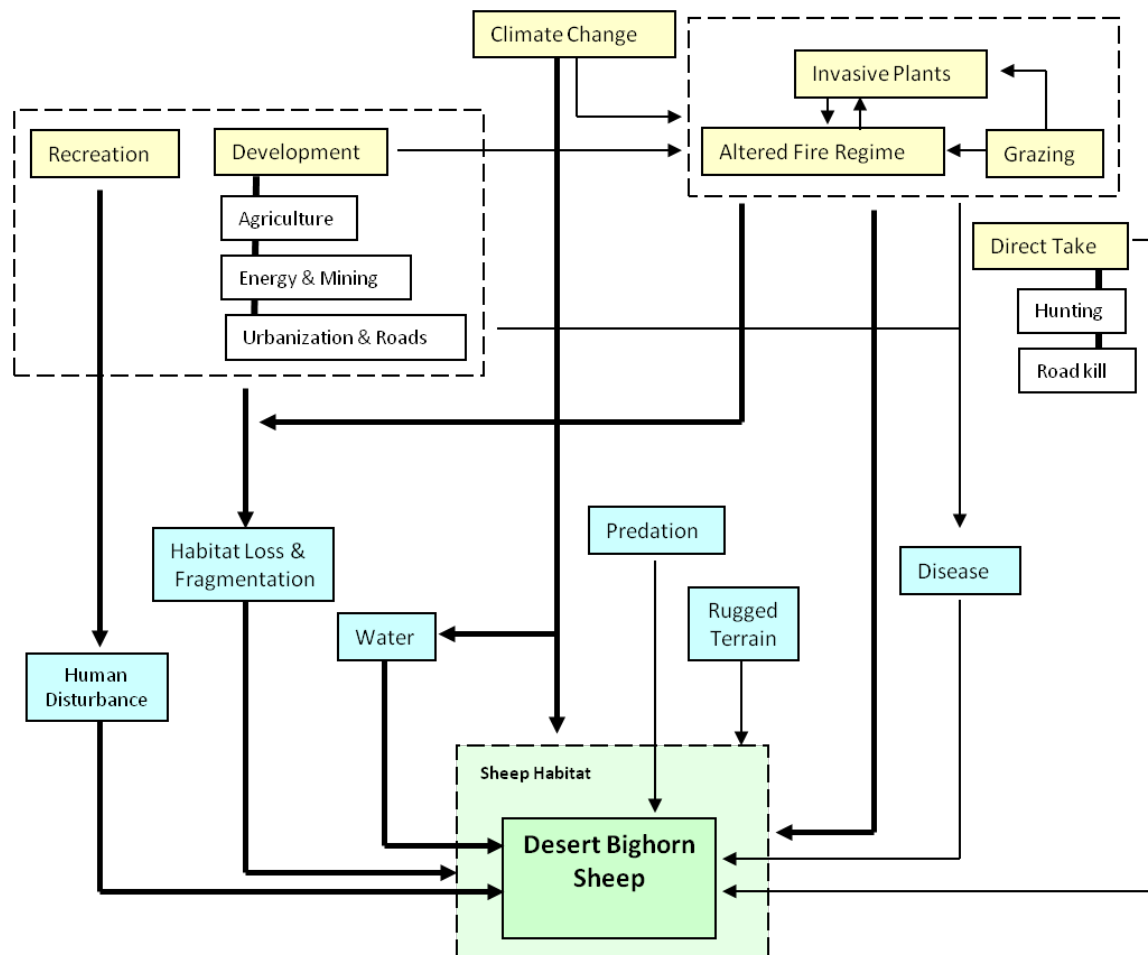
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King, M.M. and G.W. Workman. 1985. Response of desert bighorn sheep to human harassment: Management implications. *Transactions of the 51st North American Wildlife and Natural Resources Conference* 51: 74–85.

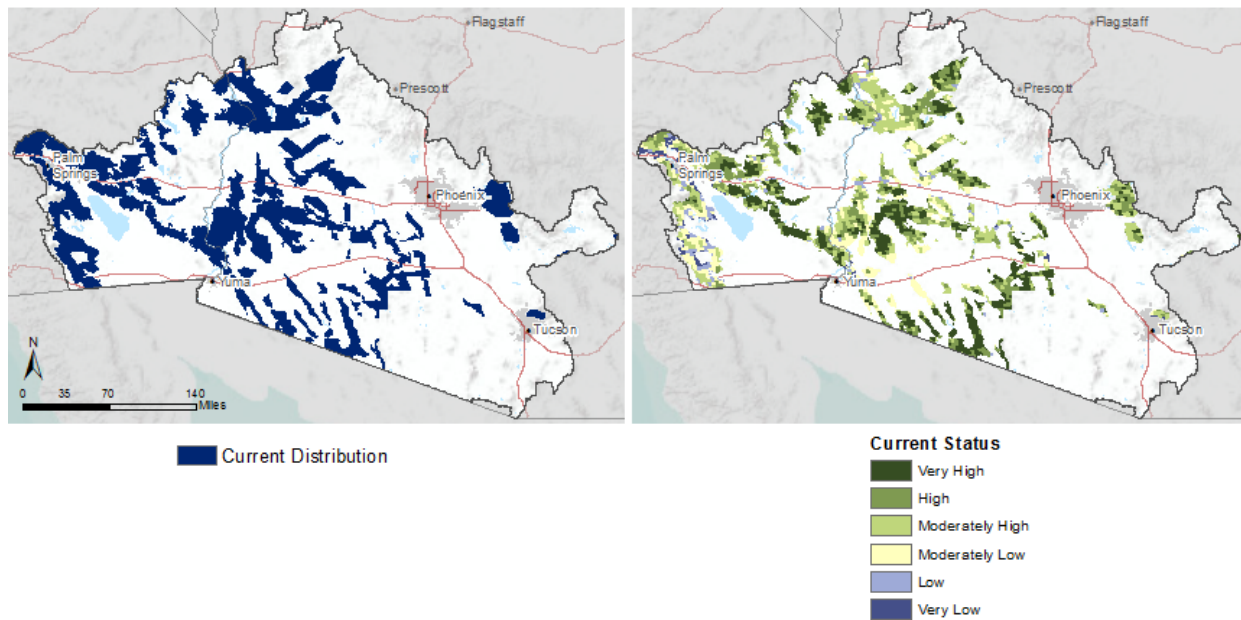
Sierra Nevada Bighorn Sheep Foundation; <http://www.sierrabighorn.org/Pages/S-NHistory.htm>

Singer, F. J., L.C. Zeigenfuss, and L. Spicer. 2001. Role of patch size, disease, and movement in rapid extinction of bighorn sheep. *Conservation Biology* 15(5): 1347–1354.

Desert Bighorn Sheep Conceptual Model

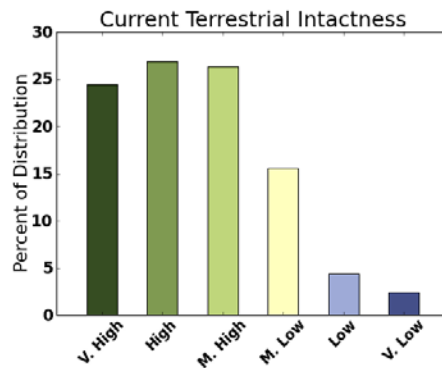


MQ D1. What's the current distribution and status of desert bighorn sheep (and historic occupied habitat if available), seasonal and breeding habitat, and movement corridors (as applicable)?



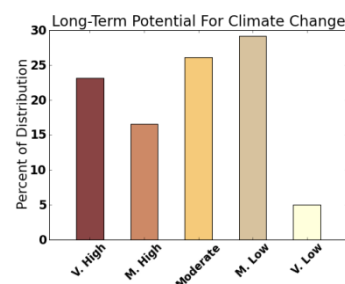
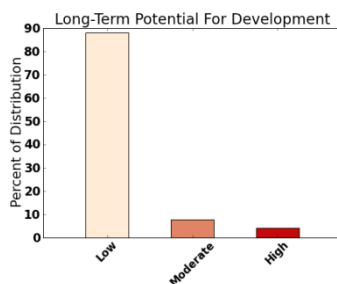
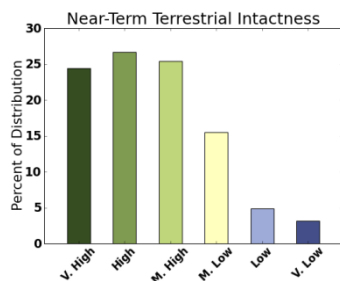
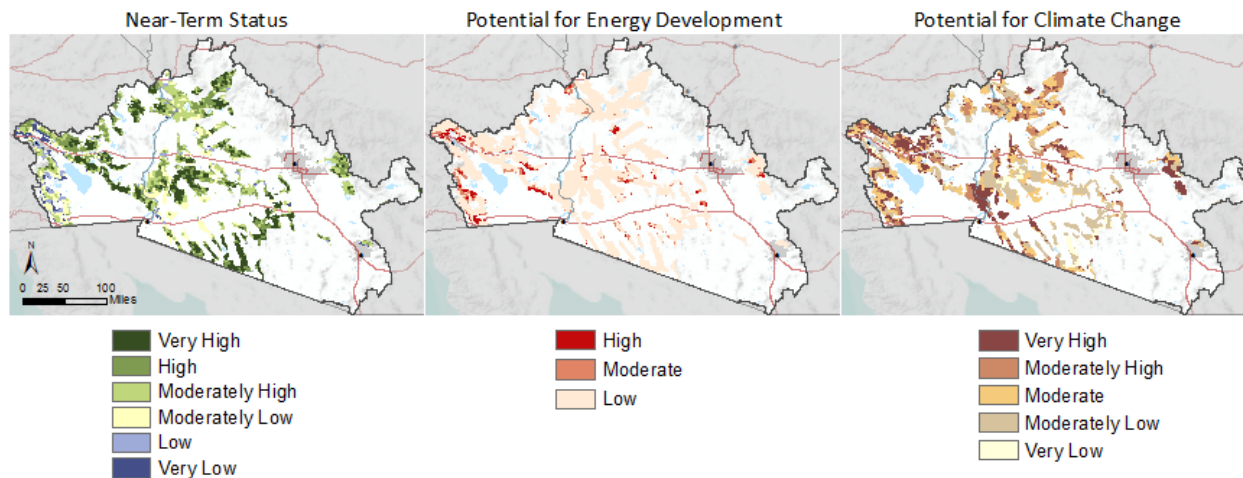
Data Sources:

USFWS Critical Habitat, BLM Wildlife Movement Corridors, California and Arizona Wildlife Agencies (compiled by USFS)

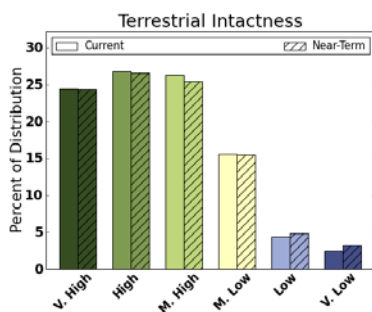


MQ D6. What sites and movement corridors are vulnerable to change agents in the near-term horizon, 2025 (development, fire, invasive species) and long-term change horizon, 2060 (climate change)?

Desert Bighorn Sheep Potential for Change



Current & Near-term Intactness



Current (solid color) and Near-term (cross-hatched) Intactness

Mojave Desert tortoise (*Gopherus agassizii*)

For full detailed account of both desert tortoise species see Desert Tortoise Case Study Insert.



The desert tortoise was selected as a core conservation element for the Sonoran Desert REA because it is an iconic species of the region that reflects inter-regional variability in climate, landform, and vegetation. The tortoise is a good indicator of desert condition because it is widely distributed across the ecoregion and, at the same time, sensitive and vulnerable to multiple disturbance factors. The desert tortoise inhabits desert environments in the Mojave and Sonoran deserts in southern California, southern Nevada, Arizona, southwestern Utah, and northwestern Mexico. Once recognized as a single species (*Gopherus agassizii*) with two recognized populations, it has recently been split into two species (Averill-Murray 2011). The Mojave desert tortoise occurs north and west of the Colorado River and retains the Latin name

Gopherus agassizii. It was listed as threatened in 1990 and, 22 years after listing, the species is still declining, particularly in the western portion of its range in California (Brussard et al. 1994, Tracy et al. 2004, USFWS 2008, 2011). The Sonoran population is now called *Gopherus morafkai*, distinguished from *G. agassizii* by its physical features, different habitat, life history traits, and DNA evidence (Murphy et al. 2011). The Sonoran desert tortoise occurs east and south of the Colorado River, from Arizona into Mexico. REA results produced maps for current status and future condition for the two desert tortoise species.

The Mojave desert tortoise occurs mainly in creosote bush (*Larrea tridentata*) flats, but it is also found in salt desert scrub and on sloping terrain on alluvial fans or foothills. It forages mostly on annual plants produced by winter rains. The yearly life cycle of the Mojave desert tortoise is heavily influenced by the annual precipitation pattern in the western Sonoran (and Mojave) Desert—precipitation that mainly falls in the winter and early spring with little or no summer precipitation (Van Devender 2002, Dickinson et al. 2002). As a result, most Mojave tortoise activity takes place in the spring when winter annuals and spring grasses are readily available (Nagy and Medica 1986, Brussard et al. 1994). Mojave tortoise hatchlings may overwinter in their nest and may not eat fresh forage until the following winter or spring. In years of low winter rainfall, Mojave tortoises may feed on introduced annual grasses in the absence or scarcity of winter annuals (Esque 1994), and while it is known that a diet of invasive grasses will keep tortoises alive, it is unknown if over time such a diet will keep them fit (Esque et al. 2002).

The species faces the prospect of annual summer drought; in the hot summer months and through the winter, the tortoises spend many months of inactivity in burrows in estivation or hibernation without eating or drinking. Mojave tortoises actively dig their own burrows in the friable soils of the western Sonoran Desert's basins and alluvial fans; they have the opportunity to alter the depth and extent of burrows to provide optimal thermal refuge and proper nest temperatures. Mojave desert tortoises typically burrow under shrubs in coarse sandy or loamy soils; they will also burrow under rocks, layers of caliche (as in the photo below), or even cement slabs in disturbed areas (Andersen et al. 2000, Lovich and Daniels 2000). Tortoises use multiple burrow sites that may vary in aspect throughout the year; burrows are often located under shrubs for shade, thermal cover, and protection from predation (eggs and juveniles, Lovich and Daniels 2000).

Because the species is at the northern limit of the overall range of desert tortoise species and because of their dietary restraints and restricted access to water, the Mojave desert tortoise may be more vulnerable to mortality from drought, loss of condition, and other stressors than the Sonoran desert tortoise (Peterson 1996, Oftedal 2002). The harsher conditions of the western Sonoran Desert ecoregion are reflected in the demographic characteristics of Mojave tortoises: individuals mature earlier reproductively and have a shorter

life span than the Sonoran tortoises (Curtin et al. 2009). Curtin et al. (2009) admit that relatively fast growth and early reproduction in a harsh environment may be counterintuitive, but that such a life history strategy may have a selection advantage in populations with high juvenile mortality and shorter overall life span.

Although similar threats and disturbances affect both tortoise species, there are differences related to their varying life histories and habitats (Curtin et al. 2009). For example, as a lowland tortoise, Mojave tortoise inhabits more developable flatlands and basins in fast-developing areas of California's Sonoran Desert; as a result, it is more directly threatened by displacement from urban, agricultural, and energy development than the Sonoran tortoise that frequents the rocky slopes of the Arizona Upland (Hunter et al. 2003, also see development section below). The fragmentation of habitat through rural housing and energy development affect tortoise populations not just through direct alteration of habitat but also through providing infrastructure and amenities that benefit predators of juvenile tortoises (Doak et al. 1994, Boarman 2003). Residential development, roads, and landfills favor tortoise predators such as ravens, coyotes, and feral and domestic dogs. For example, during a 25-year period in the late 20th century, some Mojave and California Sonoran raven (*Corvus corax*) populations in recently developed areas increased by 450-1000% (Boarman 2003). Piles of tortoise shells (incriminating evidence) have been found under raven nests (Boarman 2003). In contrast, Boarman and Coe (2002) found that raven densities were low in the roadless portions of Joshua Tree National Park.

Desert tortoises in the Mojave Desert suffer more than Sonoran desert tortoises from the upper respiratory tract disease (URTD) mycoplasmosis. Losses from this disease were one of the reasons for listing the Mojave species as threatened under the Endangered Species Act in 1990 (Van Devender 2002, USFWS 2008). For the Mojave tortoise, the frequency and intensity of URTD may be influenced by the effects of other disturbances. Habitat degradation, drought stress, food shortages, and crowding may all affect the onset and severity of URTD infections (Tracy et al. 2004)

Declines in Mojave desert tortoise continue even though tortoise management areas have been established and some of the major disturbances in those areas have been excluded. Prospects for recovery of Mojave desert tortoise are bleak if threats to both adult and juvenile segments of the population are not reduced. Doak et al. (1994) found that the rate of desert tortoise population growth was most sensitive to the survival of large adult females, and they proposed that improving survival of adult females could reverse population declines. Tracy et al. (2004) observed that the threats to desert tortoise are interactive and synergistic, and that recovery management required attention to factors affecting other age classes as well, such as the increase in predation on juvenile tortoises.

Attributes and Indicators

Attribute	Indicator	Indicator Rating				Citation
		Poor	Fair	Good	Very Good	
habitat	size	<200 sq mi	200-500 sq mi	500-1,000 sq mi	>1,000 sq mi	Brussard et al. (1994)
predation	common ravens	abundant	fairly common	rare	absent	Brussard et al. (1994)
habitat degradation	exotic ephemerals	abundant, ineradicable	fairly common and widespread	scarce and patchy	none	Brussard et al. (1994)

References Cited

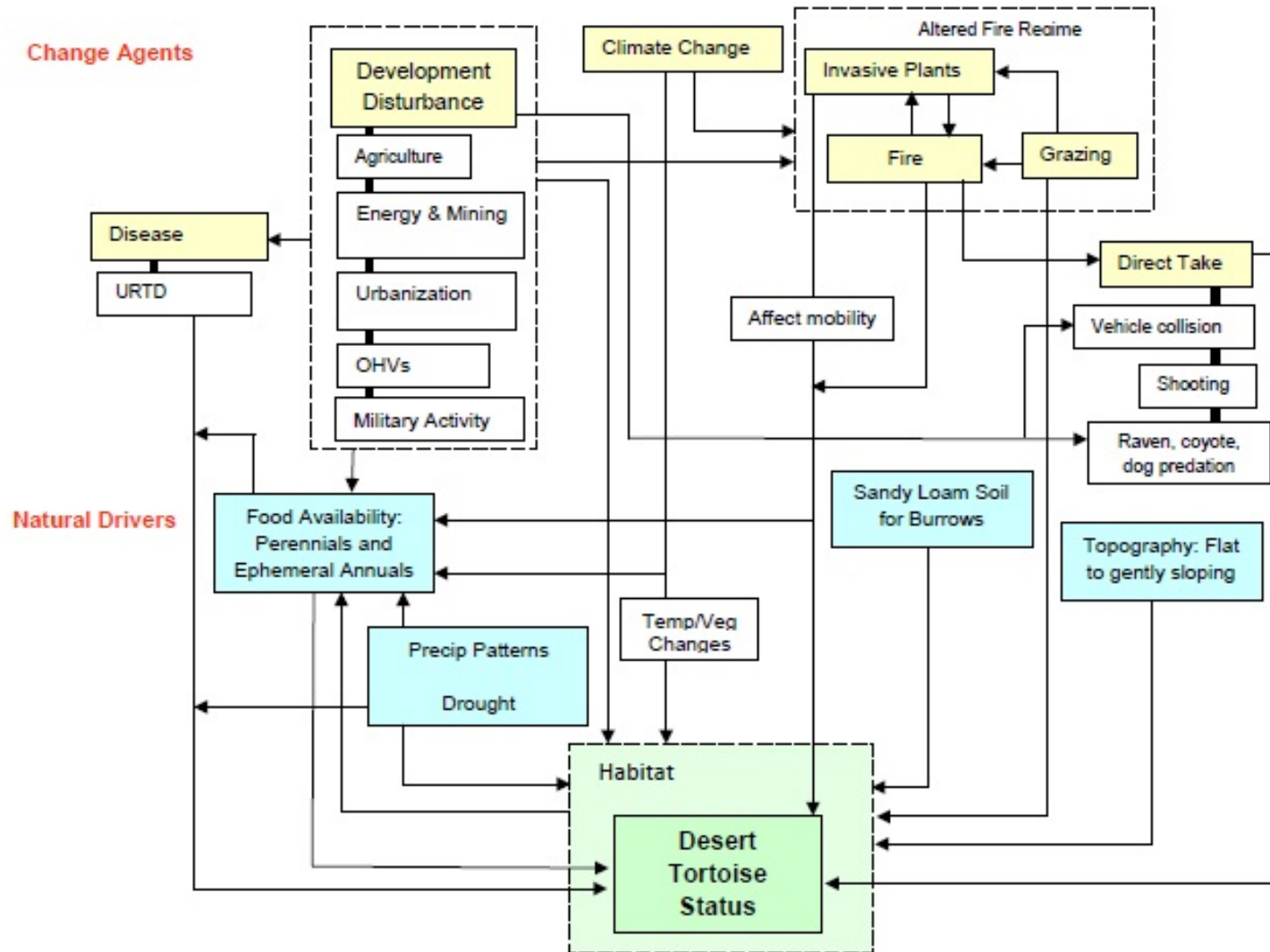
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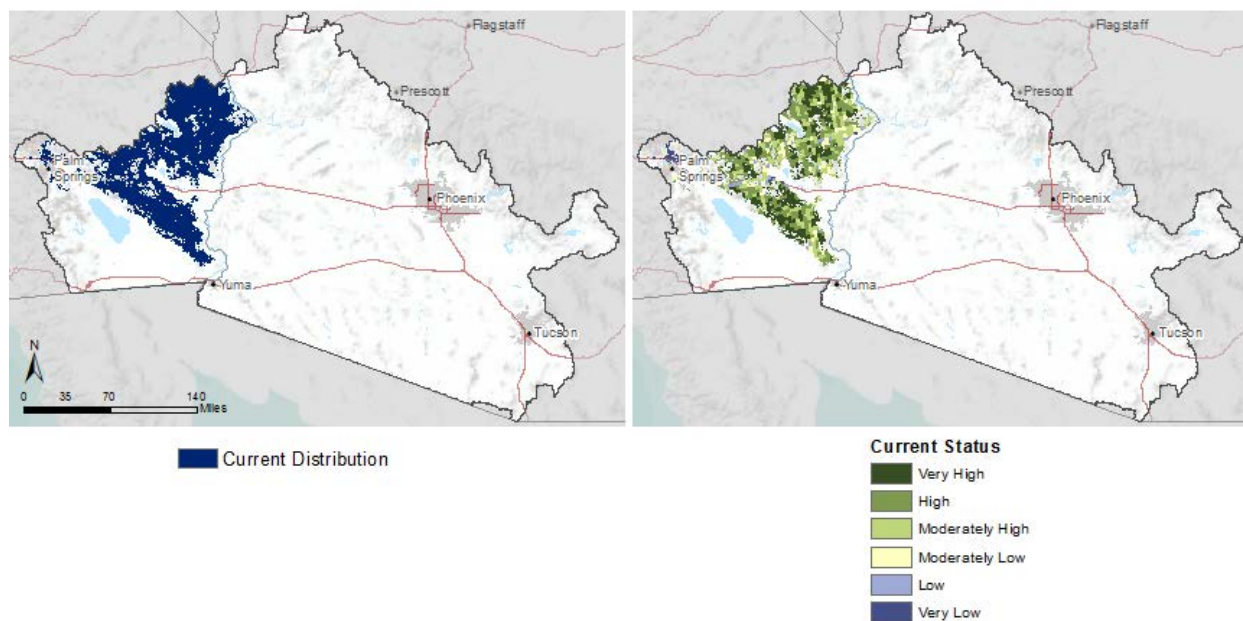


Photo: Mojave tortoise in its burrow. S. Schwarzbach, U.S. Geological Survey

Mojave Desert Tortoise Conceptual Model

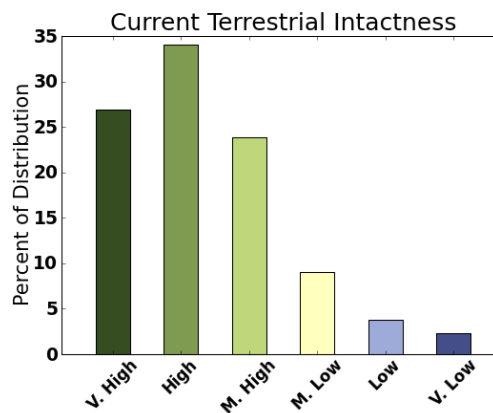


MQ D1. What's the current distribution and status of Mojave desert tortoise (*Gopherus agassizii*, and historic occupied habitat if available), seasonal and breeding habitat, and movement corridors (as applicable)?



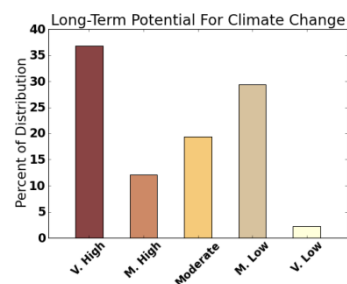
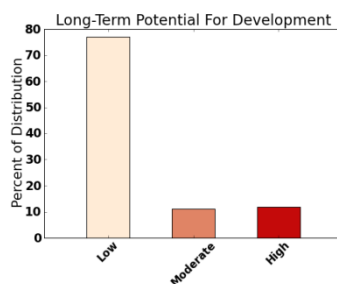
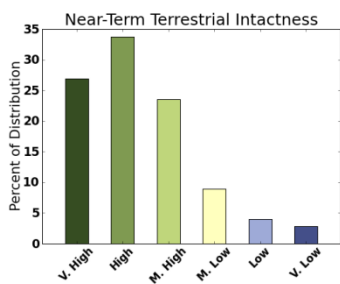
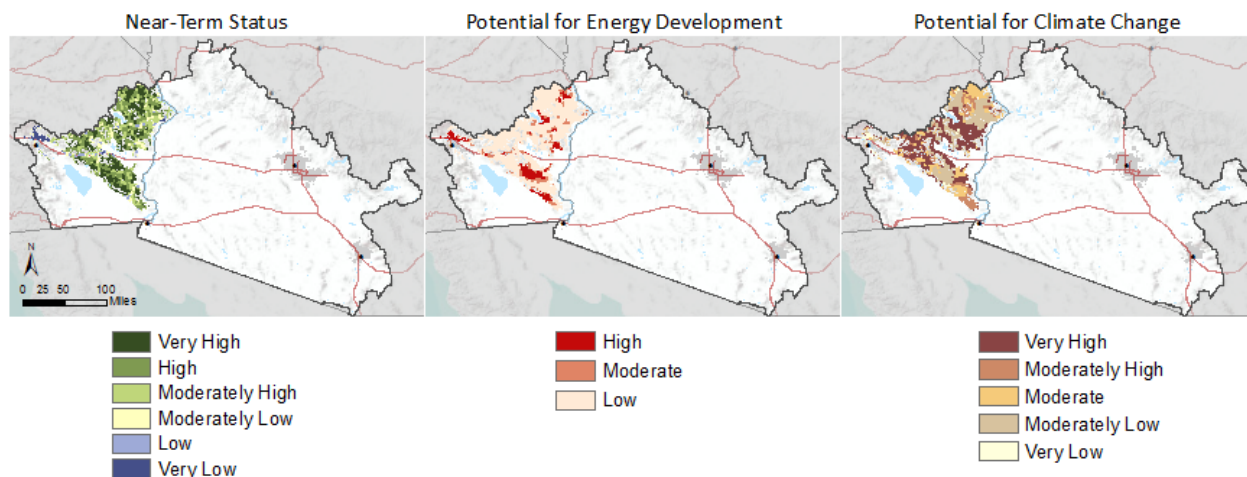
Data Sources:

Probability model from Nussear et al. 2009
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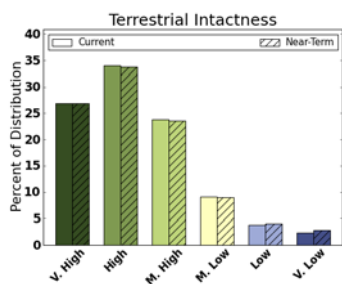


MQ D6. What sites and movement corridors are vulnerable to change agents in the near-term horizon, 2025 (development, fire, invasive species) and long-term change horizon, 2060 (climate change)?

Mojave Desert Tortoise Potential for Change



Current & Near-term Intactness



Current (solid color) and
Near-term (cross-
hatched) Intactness



Sonoran desert tortoises live on the rocky slopes and bajadas of Arizona east of the Colorado River in the Arizona Uplands and northwestern Mexico. There is a wide range in tortoise densities across the Sonoran Desert depending on habitat conditions and food availability; Sonoran tortoise populations may range from 15–100 adults/mi² (Averill-Murray et al. 2002). Home range sizes also vary, but a typical female tortoise home range in Arizona is 10 ha; males' territories may be larger, overlapping the range of several females (Van Devender 2002, Averill-Murray et al. 2002). The species does occur on occasion and in low densities in the valleys (USFWS 2010), but the frequency of dispersal of young or adults between mountain ranges is unknown. It appears that the Sonoran desert tortoise, with its patchy distribution, may have fewer opportunities for maintenance of genetic diversity and dispersal than the Mojave tortoise, which has greater continuity among populations across the broad basins of the Colorado Desert (disregarding fragmentation and human disturbance factors, Van Devender 2002, Hagerty et al. 2011).

Sonoran desert tortoises construct burrows under shrubs and rocks or in caliche caves; the tortoise may expand existing crevices under rocks, but the rocky soil does not permit the extent of burrowing that occurs in the more friable soils of the Colorado Desert. Desert washes are important to this species as they provide exposed banks with variable aspects, exposed caliche caves for locating burrows, and xeroriparian vegetation for thermal cover (Riedle et al. 2008). Unlike the Mojave tortoise that estivates in its burrow during the summer drought, the Sonoran tortoise is active in the summer during the monsoon season when fresh forage is available. Eggs usually hatch at the end of the summer rainy season, meaning that hatchlings have more access than Mojave tortoise hatchlings to fresh forage in most years (Averill-Murray et al. 2002). Besides summer annual forbs, the Sonoran tortoise feeds on warm season grasses such as big galleta (*Pleuraphis rigida*), bush muhly (*Muhlenbergia porteri*), and threeawns (*Aristida* spp.). These grasses become sparser to the west where the summer monsoon rains dwindle; as a result, Sonoran tortoises living on the drier mountain ranges closer to the Colorado River subsist on alternate food sources more similar to those available to Mojave tortoises (Van Devender 2002).

The eggs and young of both species of tortoise are subject to heavy predation by a range of mammal and bird species as well as other reptiles (e.g., Gila monsters). With their soft shells, the young are rather defenseless, and they also must spend a greater proportion of their time foraging, exposing them to predation (Morafka 1994). Raven predation, however, may not be as high for tortoises in Arizona as it is in California; the increases in raven populations subsidized by development have not (yet) occurred to the same extent. Bird predation on tortoises in general may be less in much of tortoise habitat in Arizona because of the greater cover provided by denser upland vegetation (USFWS 2010).

The greatest human-induced threats to Sonoran desert tortoise are urban and exurban development, associated road building and highway upgrading, and the increasing demands of a larger population on outdoor recreation. Throughout the 1990s the urban fringe in Phoenix advanced outward at the pace of ½ mile per year (Rex 2005). Population projections for the Phoenix areas for the next 5 decades envision a 1–1.5 million increase per decade (assuming sufficient water availability, Rex 2005). Although urban development in lowland areas may not directly convert tortoise habitat on slopes and bajadas, it puts human influence and activities in closer proximity to tortoise habitat, increasing overall access, recreation use, harassment, and pet predation. Even if valley dispersal among populations is not common, it may be important to genetic diversity; barriers from development between mountain ranges create closed populations that, if degraded or damaged, will not have the ability to recover through recruitment from other

populations (USFWS 2010). In 2010 the US Fish and Wildlife Service found that listing the Sonoran population of the desert tortoise was warranted, but that listing was precluded by higher priority actions (USFWS 2010). As a result, the Sonoran population of the desert tortoise was added to the candidate species list, where its status will be reconsidered annually.

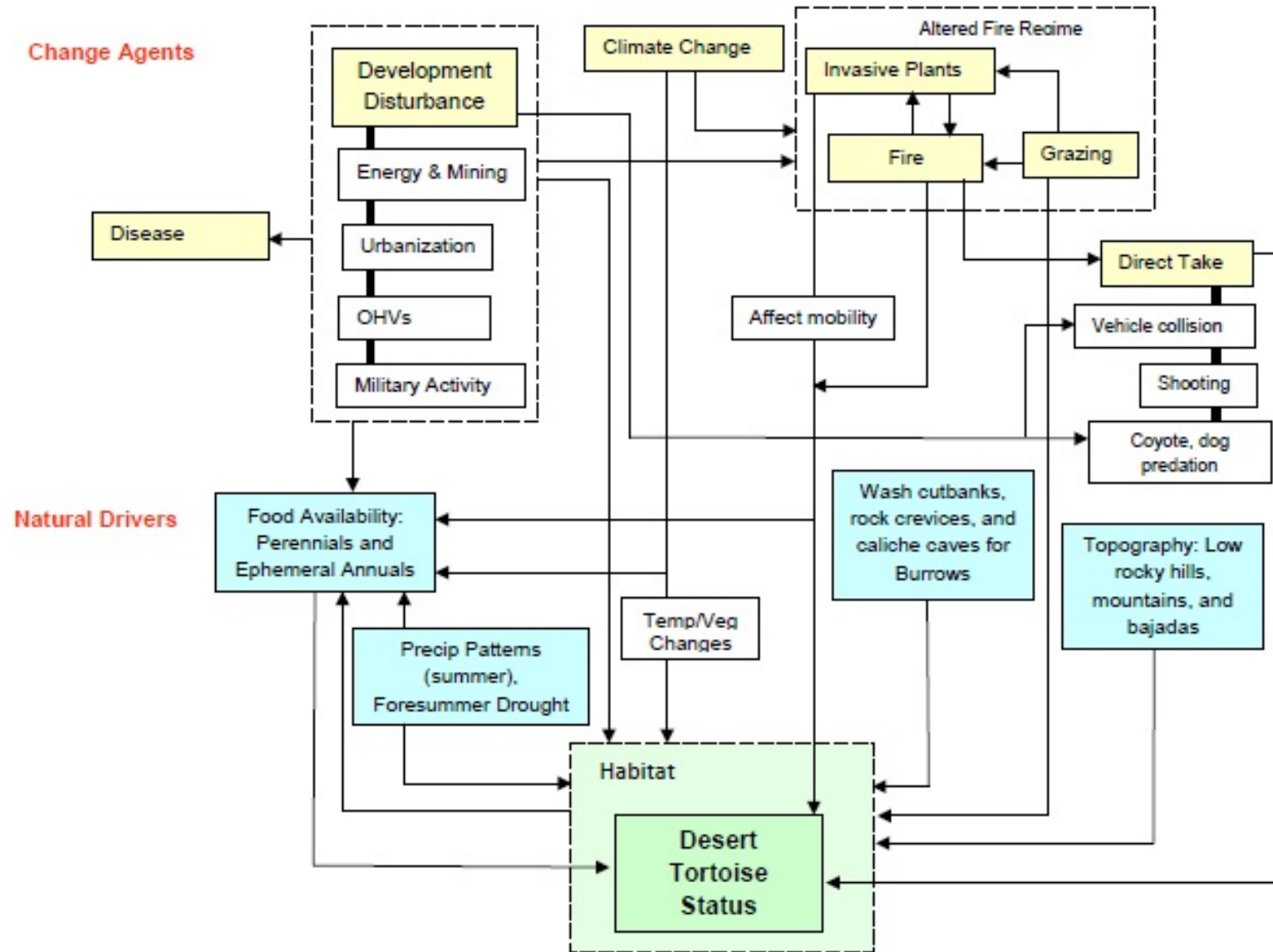
Attributes and Indicators

Attribute	Indicator	Indicator Rating				Citation
		Poor	Fair	Good	Very Good	
habitat	size	<200 sq mi	200-500 sq mi	500-1,000 sq mi	>1,000 sq mi	Brussard et al. (1994)
habitat degradation	exotic ephemerals	abundant, ineradicable	fairly common and widespread	scarce and patchy	none	Brussard et al. (1994)

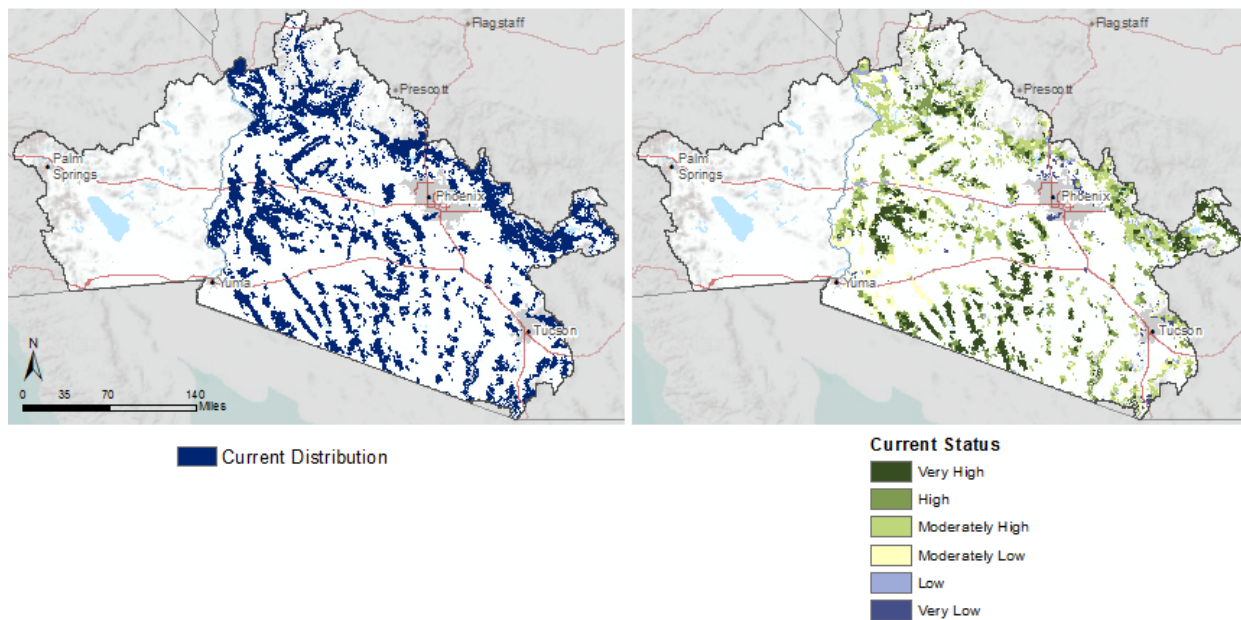
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Sonoran Desert Tortoise Conceptual Model

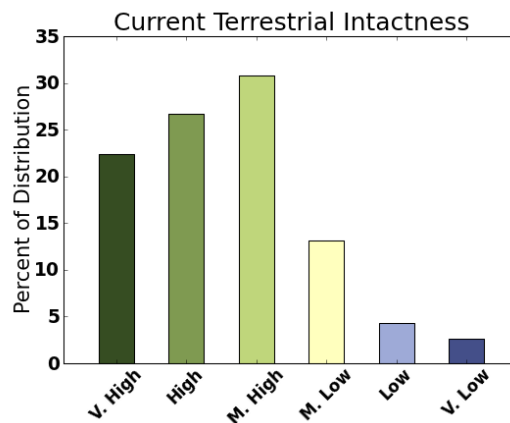


MQ D1. What's the current distribution and status of Sonoran desert tortoise (G, morafkai, and historic occupied habitat if available), seasonal and breeding habitat, and movement corridors (as applicable)?



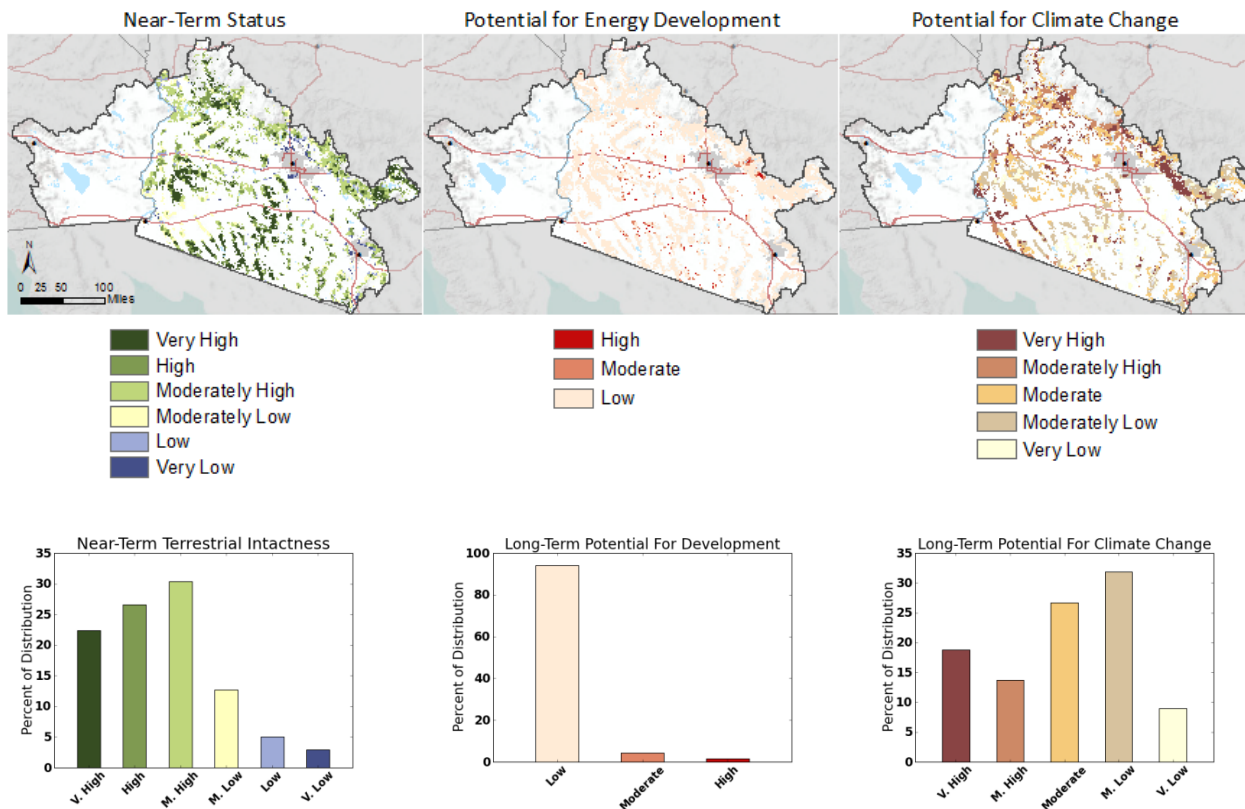
Data Sources:

Arizona GAP (clipped to remove ag/urban in LANDFIRE)

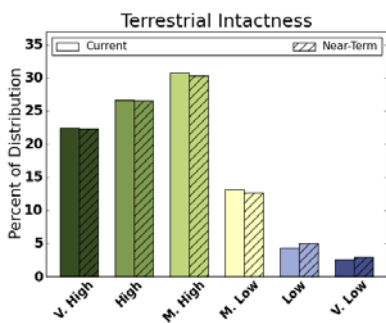


MQ D6. What sites and movement corridors are vulnerable to change agents in the near-term horizon, 2025 (development, fire, invasive species) and long-term change horizon, 2060 (climate change)?

Sonoran Desert Tortoise Potential for Change



Current & Near-term Intactness



Current (solid color) and Near-term (cross-hatched) Intactness

Golden Eagle – *Aquila chrysaetos*



Golden eagles hunt over open spaces in western North America, often in the vicinity of cliffs and ridges where the birds prefer to nest (Kochert et al. 2002). The eagles feed primarily on small to medium-sized mammals, principally hares and rabbits (Olendorff 1976). In a sample of prey remains from 9 nests in central Arizona, Eakle and Grubb (1986) found that black-tailed jackrabbit (*Lepus californicus*) was the dominant prey item followed by ground squirrel (*Spermophilus* spp.); bird prey included other raptors such as red-tailed hawk and great-horned owl. Stahlecker et al. (2009), in their survey of 191 nests in the Four Corners region of the southwestern U.S., confirmed the preference for jackrabbit and noted that ravens were the most common avian prey.

Golden eagles usually nest on cliffs, although in more developed areas they will also utilize human-made structures such as windmills, electrical transmission towers, and nesting platforms (Kochert et al. 2002). Golden eagles are typically short- to medium-distance partial migrants with individuals from northern breeding areas migrating longer distances. Eastern golden eagles nesting in northern Quebec have been tracked to wintering grounds as far south as West Virginia and the Georgia border and Alaskan eagles have been detected as far south as Kansas (Brodeur et al. 1996). Eagles in more moderate climates migrate shorter distances or remain as winter residents. Canvassers for the Arizona Breeding Bird Atlas found golden eagles sparsely distributed in the Sonoran Desert; 15% of all golden eagles detected over the 7 year survey were recorded in Sonoran desert scrub (Driscoll 2005). As with other desert inhabitants, golden eagles breeding success in the Sonoran Desert depends on a combination of favorable weather and prey availability.

Although eagles and their nests have been protected since 1962 by the Bald and Golden Eagle Protection Act, long-term surveys indicate population declines in portions of the western U.S. (Kochert and Steenhof 2002). Eagles are vulnerable to environmental change, especially from human development and changes to habitat. Breeding Bird Survey trend results show a 0.4% yearly percentage increase between 1966 and 2009 for the Mojave and Sonoran deserts and a 1.1% yearly percentage decline for Arizona for the same time period. However, these trend results carry substantial caveats since they reflect the detection difficulties and small sample size of a wide-ranging species with low abundance (Sauer et al. 2011).

The major reasons for the decline of golden eagles are direct take and habitat destruction through development. Humans cause over 70% of recorded deaths, either directly or indirectly, through collisions with vehicles, power lines, and wind turbines, electrocution on power poles, poisoning, and shooting (Franson et al. 1995). Although they are protected under the Bald and Golden Eagle Protection Act, golden eagles are sometimes illegally shot when suspected of killing livestock.

Habitat destruction due to land development has led to large-scale population declines in some areas (Kochert and Steenhof 2002). Alteration of open shrubland or grassland habitat through development or conversion to agriculture has a negative effect on eagle populations because it reduces prey populations. Eagles are often the victims of secondary poisoning when they consume prey that have been killed or sickened by pesticides, herbicides, or rodenticides (Franson et al. 1995). Eagles may also survive with elevated blood-lead levels from consuming prey that are contaminated with lead or from directly ingesting lead shot (Pattee et al. 1990, Kramer and Redig 1997). Wildfires affect golden eagles in the sagebrush community in the western U.S. through the loss of shrub habitat and resident prey. Kochert et al. (1999) found that golden eagles in sagebrush areas in Idaho avoided previously burned areas and that eagle fledging success declined with an increasing extent of burned area in the vicinity of the nest. It is unknown how

burned-over areas will affect golden eagles in the Sonoran Desert as wildfire, once rare in desert scrub, becomes more common with the expansion of invasive annuals.

Infrastructure such as power lines and wind turbines are also responsible for eagle mortality. In the Altamont Pass Wind Resource Area in west-central California, Smallwood and Thelander (2008) estimated 67 golden eagle fatalities per year due to wind turbines; sub-adults and floaters appeared to be affected disproportionately (Hunt 2002). Golden eagle fatalities were correlated with turbine height, location, and topography with the majority of deaths associated with shorter turbines (e.g. Type 13), end of row and second from the end turbines, and favored aerial pathways through dips and notches in topography (Curry and Kerlinger 1998, Hunt 2002). Although it has been reported that fatalities are much lower from newer wind farms with more recent turbine designs, there is no clear relationship between pre-construction risk assessment planning and reduced mortality (Lynn and Auberle 2009). While, on one hand, Smallwood and Karas (2009) estimated that newer turbines at Altamont could reduce mean annual fatality rates by 54% for raptors (while more than doubling annual wind-energy generation), eagle deaths tallied at a new (2 year old) wind farm north of Los Angeles, showed an annual death rate per turbine to be three times higher than at the older Altamont facility (Sahagan 2011). Potential risk assessments conducted prior to permitting wind facilities evaluate topography, weather patterns, and vegetation type, the presence of flyways and migration corridors, the numbers of birds potentially flying in the risk zones near the rotors, the possible presence of species of concern, the distance to important nesting areas and roost sites for birds and bats, and the potential for prey species such as ground squirrels to inhabit the site (Lynn and Auberle 2009).

Golden eagles populations are sustained by the conservation of large areas of intact desert habitat. Eagle home ranges are large, but they vary considerably in size depending on region, prey availability, and season from a few thousand to tens of thousands of hectares. Eagle management is inseparable from management of prey populations and their habitat, and shrub patch size is an important element; in sagebrush communities to the north, a management rule of thumb is to avoid fragmentation of shrub habitats below the mean patch size of 5000 ha shown to support healthy jackrabbit populations (Marzluff et al. 1997). If they do not already exist, establishing similar guidelines for golden eagle prey species' habitat patches in the Sonoran Desert may be a useful research objective to accompany eagle monitoring.

In the Sonoran Desert, golden eagles benefit from established protected areas, such as the Kofa and Cabeza Prieta National Wildlife Refuges, National Parks and Wilderness areas, and some military lands such as Barry M. Goldwater Air Force Range. In southeastern California, the Desert Renewable Energy Conservation Plan reports that the golden eagle has 72% of its predicted breeding habitat and 48% of its predicted foraging habitat already within protected areas (CEC 2011). With the advent of additional energy development in potential solar and wind areas of the Sonoran Desert, planning for golden eagles should include protecting nest sites and minimizing activity in eagle nesting areas (CEC 2011), eagle-sensitive turbine selection and placing (Curry and Kerlinger 1998, Hunt 2002, Smallwood and Thelander 2007), and raptor-safe electrical transmission lines and poles with widely spaced conductors, perch guards, or perches installed above the conductors (BLM 2005).

Attributes and Indicators

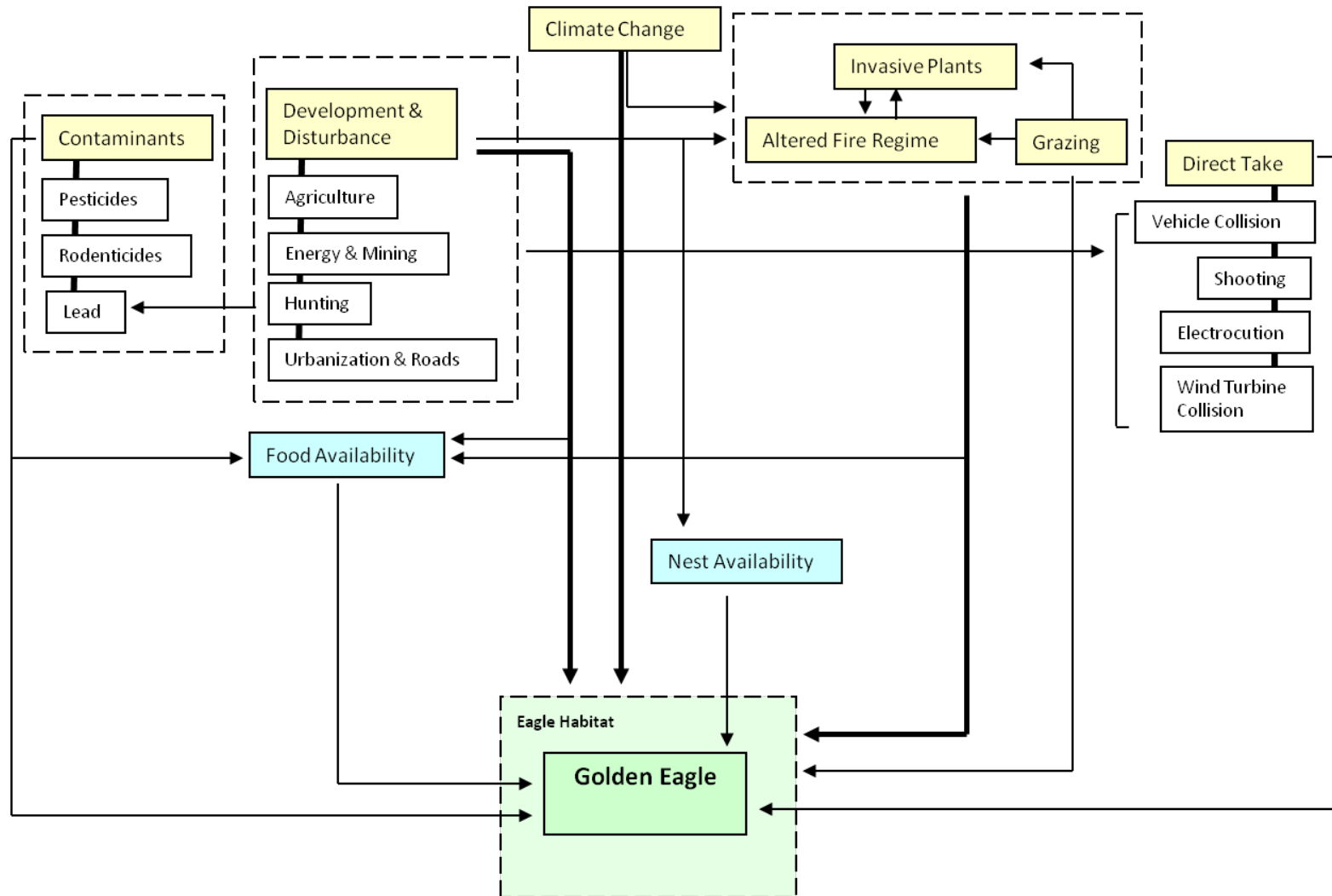
Attribute	Indicator	Indicator Rating				Citation
		Poor	Fair	Good	Very Good	
habitat loss or degradation	urban development	present	--	minimal	absent	Kochert and Steenhof (2002)
habitat degradation	livestock grazing and agriculture	existing or planned	--	--	absent	Beecham and Kochert (1975)
habitat degradation	fire	>40,000 ha of shrublands burned	--	burned territory; adjacent vacant unburned	unburned territories	Kochert et al. (1999)
habitat degradation	mining and energy development	present	--	--	absent	Phillips and Beske (1982)
habitat	vegetation	disturbed areas, grasslands, agriculture			shrubland/open grassland	Marzluff et al. (1997), Peterson (1988)
habitat/nest sites	topography	--	--	--	cliffs within 7 km of shrubland	Menkens and Anderson (1987), Cooperrider et al. (1986)
mortality	infrastructure (roads, power lines, wind turbines)	--	--	--	infrastructure absent	Franson et al. (1995)
Illness mortality	poisoning from pesticides and other toxins	high levels of contaminants	--	--	low/no contaminants	Franson et al. (1995), Harmata and Restani (1995), Kramer and Redig (1997), Pattee et al. (1990)
habitat loss or degradation	urban development	present	--	minimal	absent	Kochert and Steenhof (2002)

References Cited

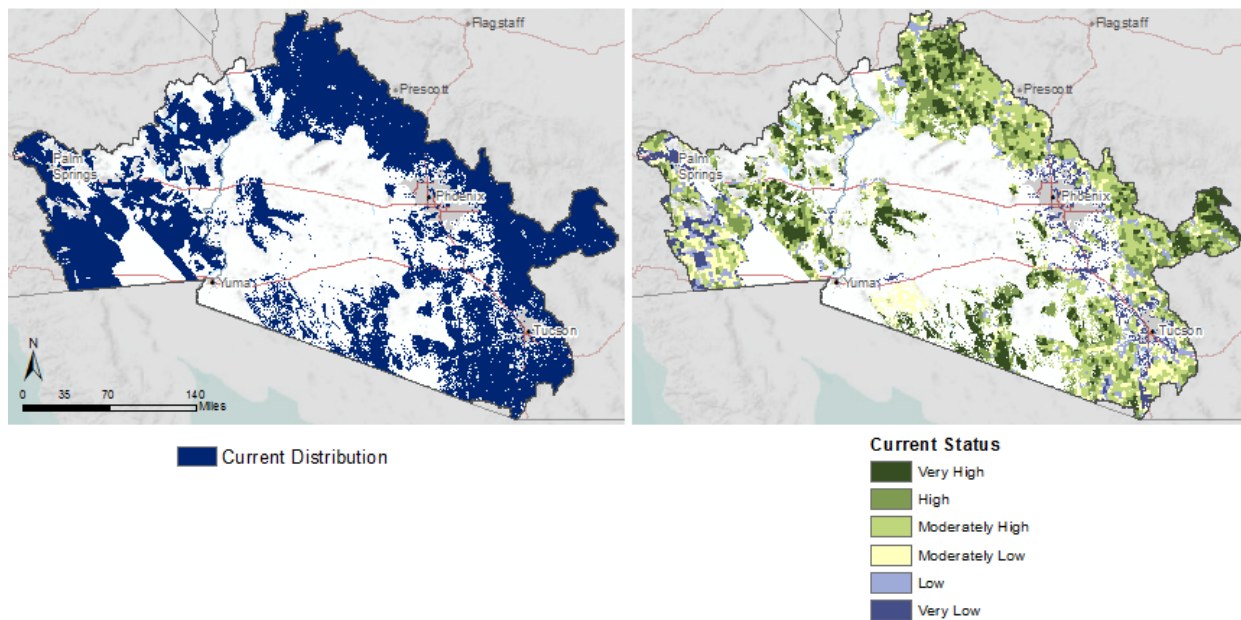
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Golden Eagle Conceptual Model

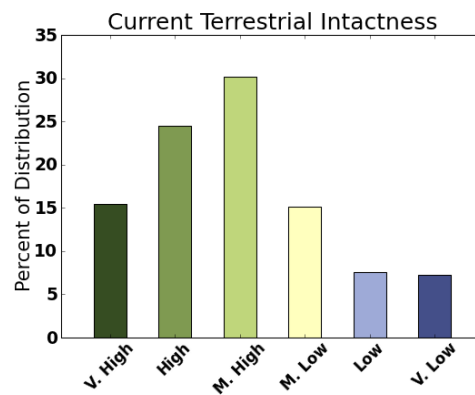


MQ D1. What's the current distribution and status of golden eagle (and historic occupied habitat if available), seasonal and breeding habitat, and movement corridors (as applicable)?



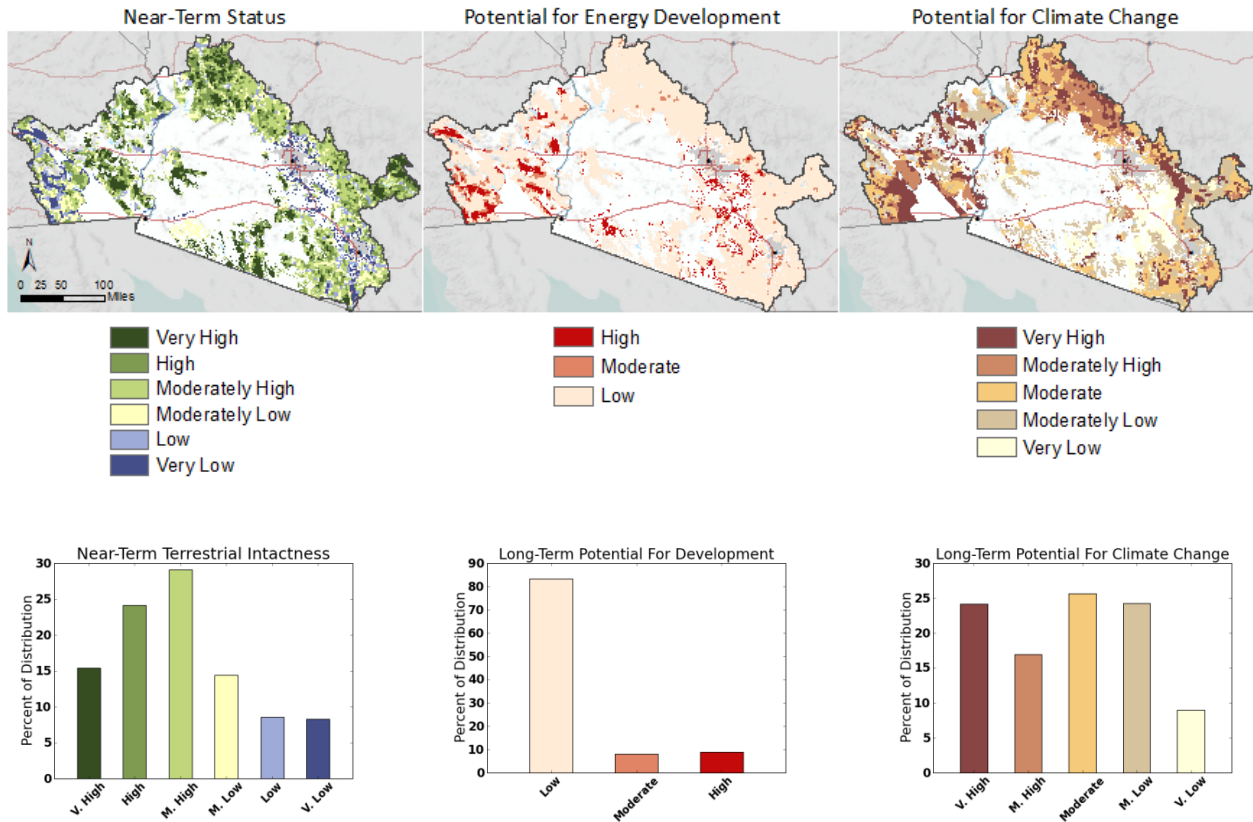
Data Sources:

California GAP and Arizona Game and Fish Department

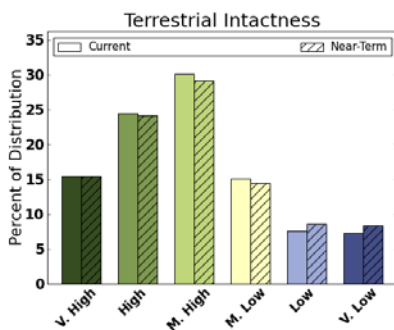


MQ D6. What sites and movement corridors are vulnerable to change agents in the near-term horizon, 2025 (development, fire, invasive species) and long-term change horizon, 2060 (climate change)?

Golden Eagle Potential for Change



Current & Near-term Intactness



Le Conte's Thrasher (*Toxostoma lecontei*)



Le Conte's thrasher (*Toxostoma lecontei*) is on the Audubon Society's Watch List as a species that is at risk because of its low abundance (historic and present) and relative lack of mobility. Even in optimal habitat, densities may be as low as five pairs or fewer per square mile (National Audubon Society 2011). Breeding Bird Survey trend estimate data for the Sonoran Desert ecoregion shows a 3.2 % per year decline for the period 1966–2009 and a 2.6% per year decline for the period 1999–2009 (n=detections on 47 BBS routes, Sauer et al. 2011). Habitat loss has resulted in local extirpations in the Coachella and Imperial valleys of California (Weigand and Fitton 2008). The thrasher population in the Gila River valley in Arizona is also in danger of extirpation due to agriculture and expanding urban and exurban sprawl (Corman 2005). Thrashers previously reported in the Avra Valley in northern Pima County, Arizona, have not been detected since the 1980s (Corman 2005).

The species is a year-round resident in the Colorado Desert that covers the sparsely vegetated, lower-elevation portions of the Sonoran Desert ecoregion in southeastern California and southwestern Arizona. Le Conte's thrashers frequent somewhat rolling, well-drained areas for foraging (the toeslopes of bajadas or alluvial fans), avoiding the more poorly drained valley floors (playas) that may be flooded in winter or highly alkaline in the dry season (Shuford and Gardali 2008). Optimal foraging areas have a mixture of bare ground and scattered litter. The thrashers forage on the ground by digging and probing in the soil, flipping bits of debris with their stout curved bills to search for insects beneath. They also glean other prey from vegetation or pursue insects and lizards on the ground (Weigand and Fitton 2008, National Audubon Society 2011). Le Conte's thrashers hide their nest low to the ground in the thick vegetation of thorny bushes or small trees such as saltbush (*Atriplex* sp.), mesquite (*Prosopis* sp.), desert-thorn (*Lycium* sp.), lotebush (*Ziziphus obtusifolia*), blue palo verde (*Parkinsonia floridum*), ironwood (*Olneya tesota*), Joshua trees (*Yucca brevifolia*), cholla cacti (recorded only in California, *Cylindropuntia* sp.), and ocotillo (in Arizona, *Fouquieria splendens*, Corman 2005, Weigand and Fitton 2008, CalPIF 2009). These favored nesting shrubs and small trees occur in basins and washes in the creosotebush (*Larrea tridentata*)-white bursage (*Ambrosia dumosa*) vegetation community class common to the alkaline basins and lowlands of the Sonoran Desert. As with other desert species, during periods of drought, Le Conte thrasher nesting may fail or be deferred, while, in better years, there may be multiple broods.

The increasing incidence of fire in the desert shrub habitats of the Sonoran Desert poses a serious threat to Le Conte's thrasher. Many desert shrub species do not sprout readily and are slow to reestablish. Burned areas may be replaced by invasive annual plants that burn more frequently. Following fire, thrashers may be displaced from the burned area for decades because they lose nesting shrubs and, if annuals establish post-burn, they also lose the bare ground and surface litter that they require for foraging (Shuford and Gardali 2008). Grazing also has been implicated in the gradual conversion of shrublands to non-native annual grasses and forbs. Saltbush shrub habitat may be managed for grazing to avoid negatively affecting Le Conte's thrasher by avoiding stocking in drought years and by limiting grazing to early season use to preclude browsing on shrubs later in the season (Shuford and Gardali 2008).

The long-term sustainability of Le Conte's thrasher depends on unfragmented expanses of desert shrubland and connectivity between patches, since the species is relatively sedentary and does not disperse widely. Le Conte's thrashers apparently will not inhabit fragments of shrub habitat within areas undergoing development due to increased disturbance and lack of connectivity (Weigand and Fitton 2008). Since they nest close to the ground, thrashers experience increased nest predation by wild and domestic predators in fragmented areas. Concentrating development-related disturbances in the thrasher's range will help to limit

fragmentation (Weigand and Fitton 2008). For example, siting a solar energy field in former agricultural land or near existing infrastructure is preferable to carving it out of remote, intact desert shrubland. Thrashers are also vulnerable to disturbance and death from OHV activity in basins and desert washes. Limiting OHV use to designated routes would conserve thrasher habitat. Le Conte's thrasher benefits from refuges that are created to protect other inhabitants of the Sonoran Desert; intact breeding populations of Le Conte's thrashers occur within the Desert Tortoise Research Natural Area in California and Cabeza Prieta National Wildlife Refuge and the Barry M. Goldwater Range in Arizona (National Audubon Society 2011, Corman 2005).

Attributes and Indicators

Attribute	Indicator	Indicator Rating				Citation
		Poor	Fair	Good	Very Good	
habitat degradation	invasive grasses				revegetation at disturbed sites of desert thorn (<i>Lycium</i>) and saltbush (<i>Atriplex</i>) species;	Weigand and Fitton (2008)
habitat	Habitat fragmentation -	>10 km distance between habitat fragments			maximum of 2 km distance between habitat fragments	Weigand and Fitton (2008)

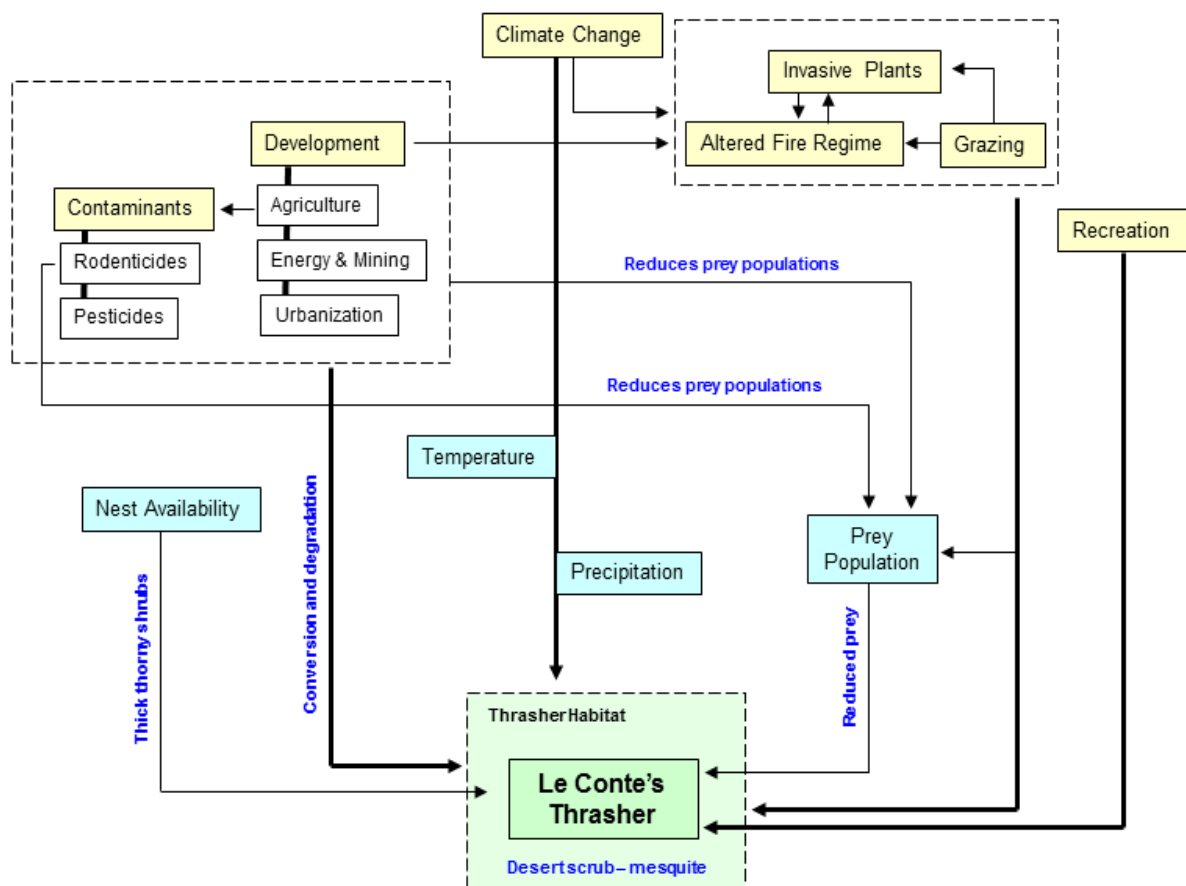
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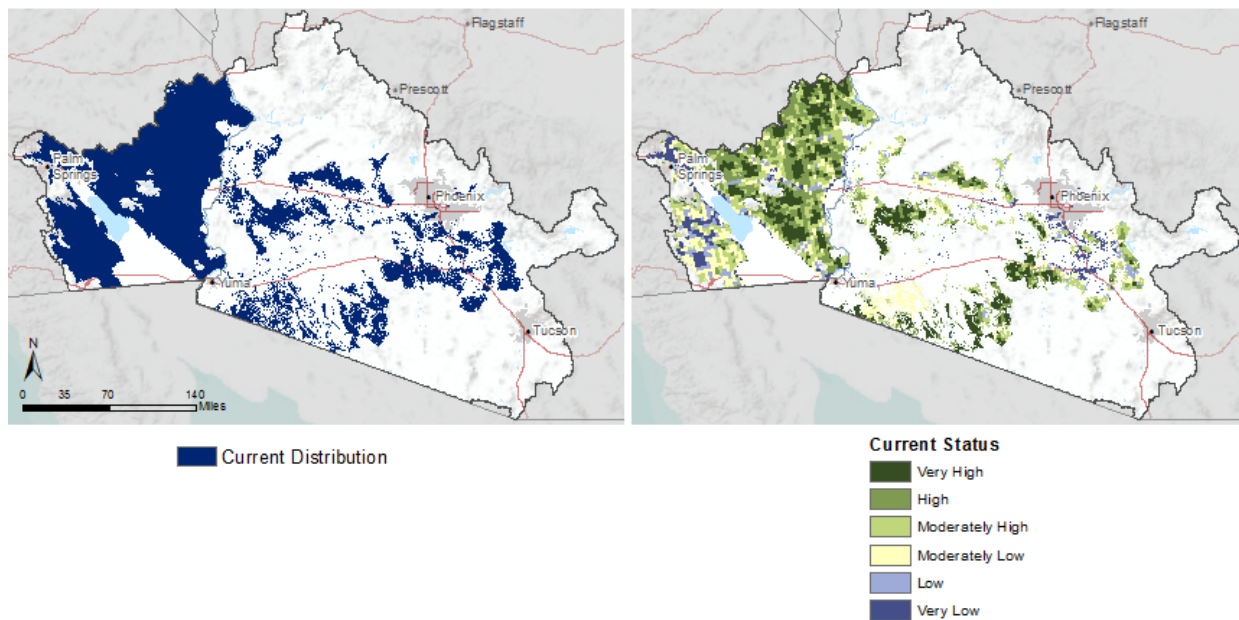
Weigand, J., and S. Fitton. 2008. Le Conte's thrasher (*Toxostoma lecontei*) in Draft desert bird conservation plan: A strategy for reversing the decline of desert-associated birds in California. California Partners in Flight. <http://www.prbo.org/calpif/html/docs/desert.html>

Le Conte's Thrasher Conceptual Model



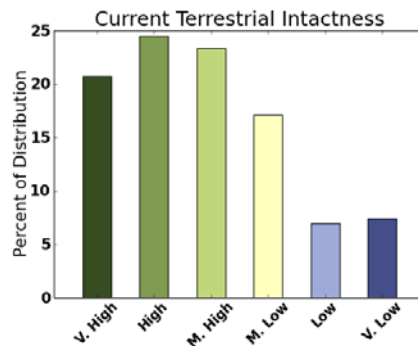
Thick arrows denote components with available spatially explicit data – NOT importance

MQ D1. What's the current distribution and status of Le Conte's thrasher (and historic occupied habitat if available), seasonal and breeding habitat, and movement corridors (as applicable)?



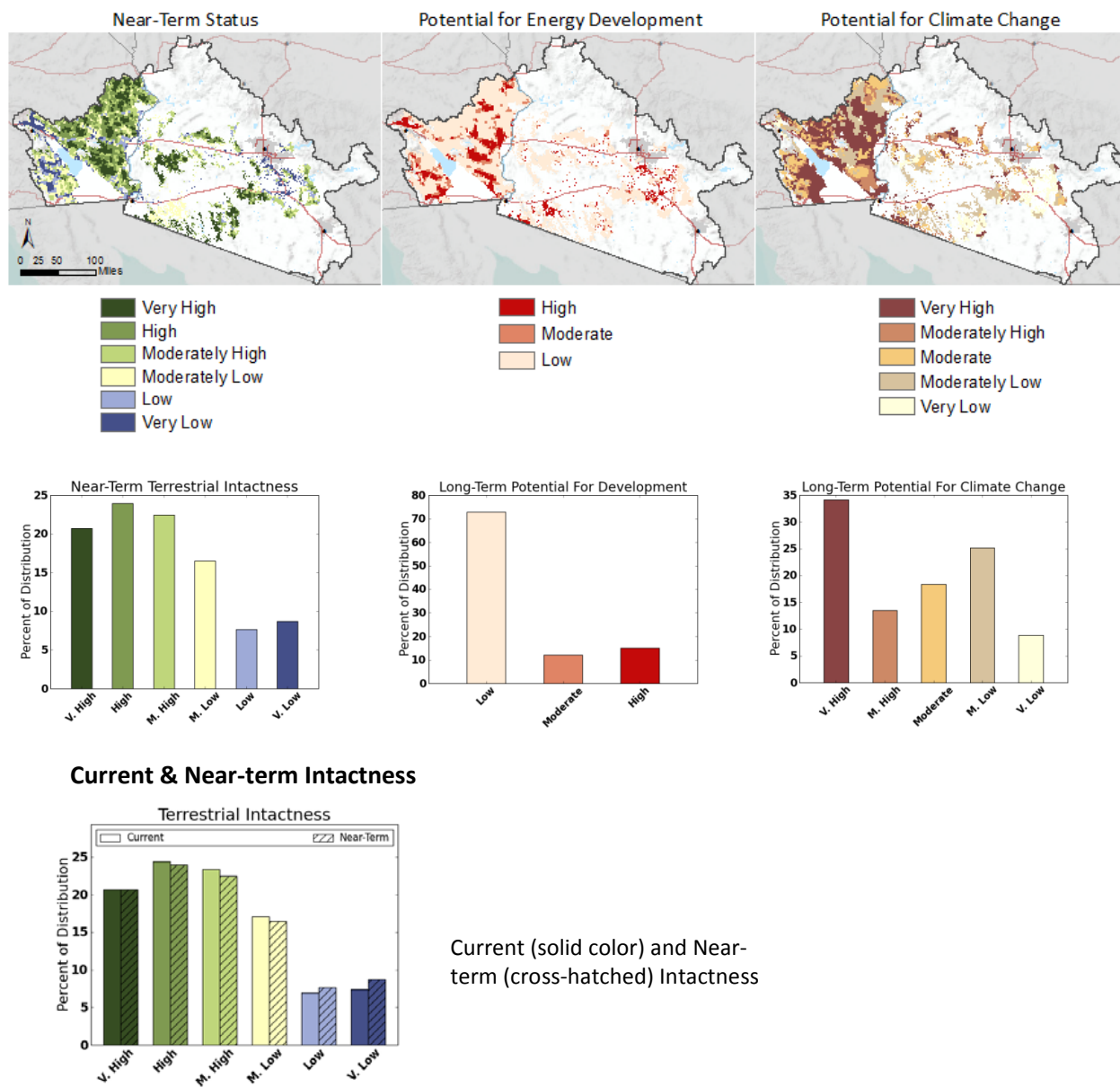
Data Sources:

California GAP and Arizona Game and Fish Department



MQ D6. What sites and movement corridors are vulnerable to change agents in the near-term horizon, 2025 (development, fire, invasive species) and long-term change horizon, 2060 (climate change)?

Le Conte's Thrasher Potential for Change



Lowland Leopard Frog (*Lithobates yavapaiensis*)



Attributes and Indicators

Attribute	Indicator	Indicator Rating				Citation
		Poor	Fair	Good	Very Good	
habitat	elevation	>8,200 ft	6,400 - 8,200 ft	-	<6,400 ft	AZGFD (2006)
predation	American bullfrog	present	-	-	absent	Jennings (1994)
habitat	water development	present			absent	Center for Biological Diversity & S. Utah Wilderness Alliance- Petition to list

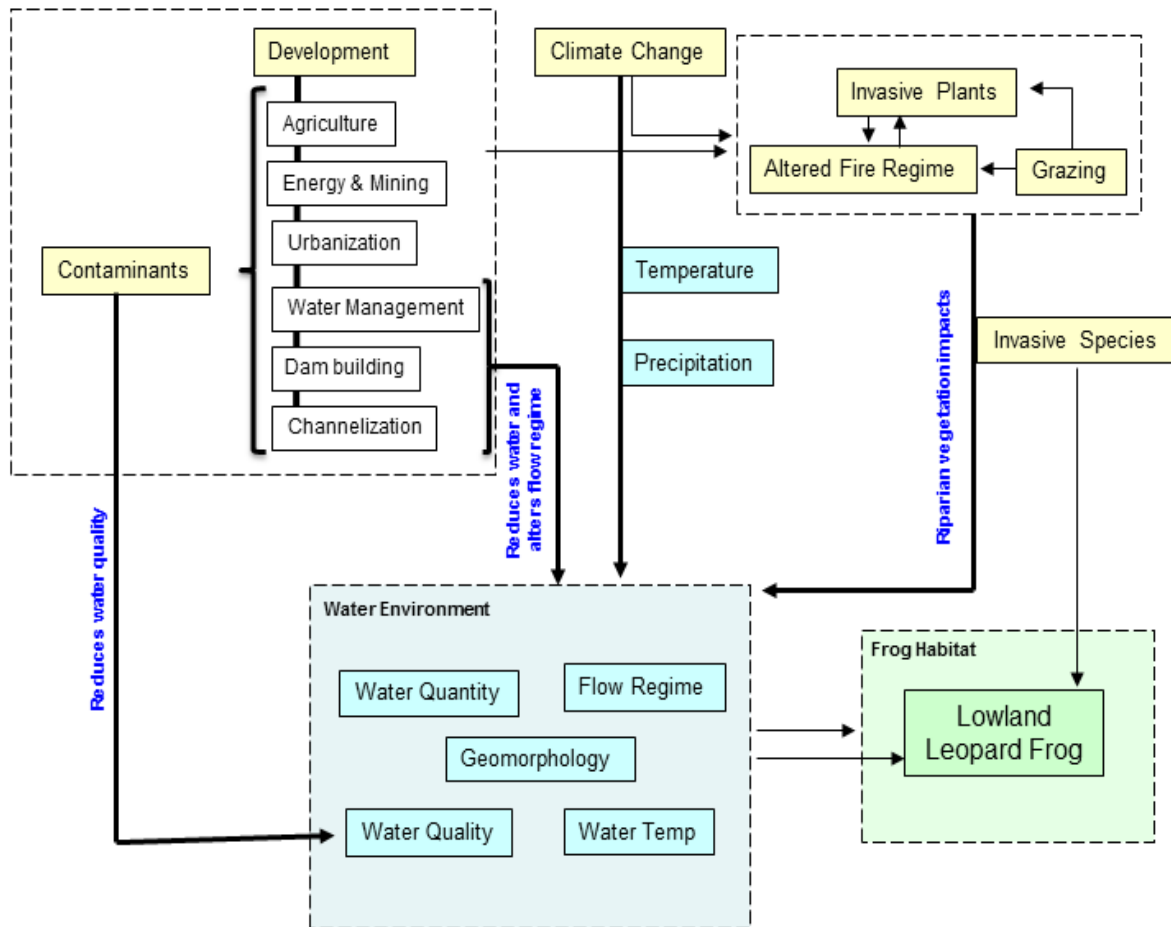
References Cited

(AZGFD) Arizona Game and Fish Department. 2004. Heritage Data Management System. 10 pp.

Center for Biological Diversity and Southern Utah Wilderness Alliance http://www.biologicaldiversity.org/species/amphibians/relict_leopard_frog/pdfs/petition.pdf.

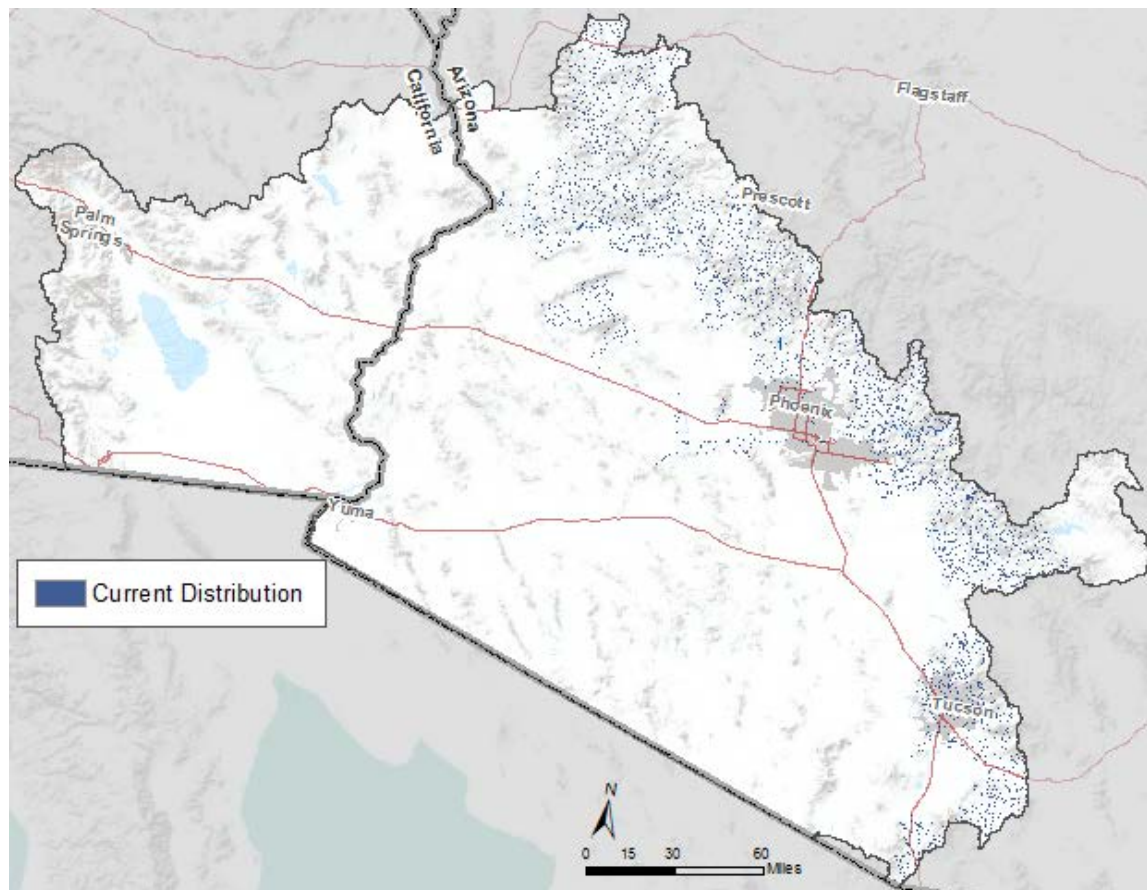
Jennings, M.R., and M.P. Hayes. 1994. Decline of native ranid frogs in the desert southwest. Pages 183–211 in P.R. Brown and J.W. Wright (eds.), Herpetology of the North American deserts. Southwestern Herpetologists' Society, Special Publication No. 5.

Lowland Leopard Frog Conceptual Model



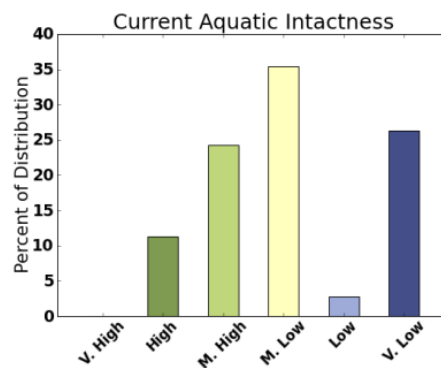
Thick arrows denote components with available spatially explicit data – NOT importance

MQ D1. What's the current distribution and status of lowland leopard frog (and historic occupied habitat if available), seasonal and breeding habitat, and movement corridors (as applicable)?



Data Sources:

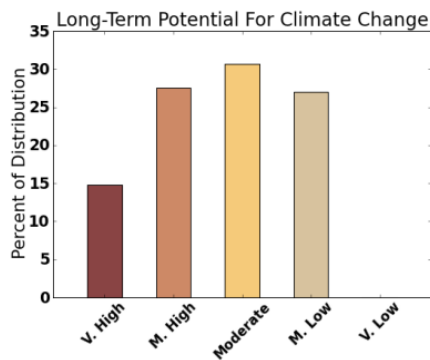
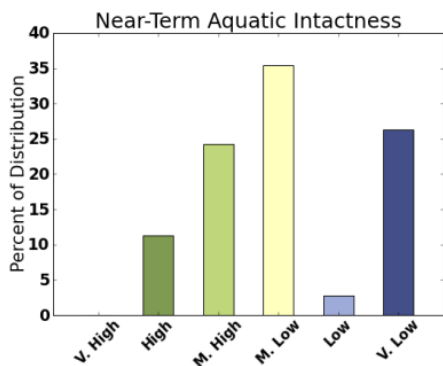
SW ReGAP clipped to watersheds containing post-1980 element occurrence data from Arizona Natural Heritage Program



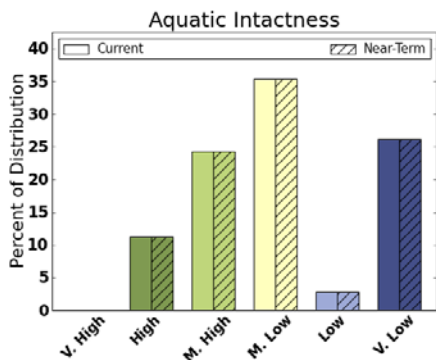
Lowland Leopard Frog Status

MQ D6. What sites and movement corridors are vulnerable to change agents in the near-term horizon, 2025 (development, fire, invasive species) and long-term change horizon, 2060 (climate change)?

Lowland Leopard Frog Potential for Change



Current & Near-term Intactness



Current (solid color) and Near-term (cross-hatched) Intactness



The sustainability of Lucy's warbler populations is directly linked to the availability of functioning riparian woodland. The warbler is considered a species of special concern in California, where it was once considered common early in the 20th century (Garrett 2008). In the Sonoran Desert ecoregion in California, Lucy's warbler is confined to the west bank and tributary washes of the lower Colorado River in addition to a remnant population in Anza-Borrego State Park (Garrett 2008). Although most of the honey mesquite (*Prosopis glandulosa*) bosques along the Colorado River were cleared during the 1990s, pockets of remaining habitat may be found near Fort Mojave and between Blythe, California and Yuma, Arizona (National Audubon Society 2011). Lucy's warbler is somewhat more common in riparian areas of southern Arizona.

Breeding Bird Survey (BBS) trend estimate data for the Sonoran Desert-Mojave Desert combined ecoregion shows a 1.3 % per year increase for the period 1966–2009 and a 2.7% per year increase for the period 1999–2009 (n=detections on 32 BBS routes, Sauer et al. 2011). However, these trend results carry the caveat of small sample size of a species occurring in a habitat not well-represented in the BBS (Sauer et al. 2011).

Lucy's warbler is an insect gleaner that is dependent for foraging on a diverse riparian structure with both a tree canopy and shrubby understory. In optimal riparian habitat in southwestern New Mexico, breeding territory densities of Lucy's warbler have been estimated as high as 1.7 to 3.3 territories per ha (Stoleson et al. 2000). The species is a cavity nester that requires trees of adequate diameter for nesting (McCreedy 2011). It will also use hollow limbs, loose bark, crevices in cliffs or stream banks, abandoned cliff swallow nests, or suspended flood debris (Corman 2005). Although brown-headed cowbirds will parasitize Lucy's warbler nests, the impacts are not as great as they are for Bell's vireo, perhaps because cavity nesting offers a deterrent to cowbird egg laying (Corman 2005, Garrett 2008). The species' preferred habitats vary regionally: in California and along the lower Colorado River, Lucy's warbler frequents mesquite (*Prosopis* spp.) bosques; elsewhere in Arizona and southwestern New Mexico, it may nest in cottonwood-willow (*Populus fremontii*-*Salix gooddingii*) or sycamore-ash (*Platanus-Faxinus* spp.) associations (Stoleson et al. 2000, Garrett 2008, CalPIF 2009). Where mesquite bosques have been degraded or eliminated along the lower Colorado River, Lucy's warbler nests have been found in drier tributary washes in blue palo verde (*Parkinsonia floridum*), foothills palo verde (*Parkinsonia microphylla*) and ironwood (*Olneya tesota*) trees (average diameter at ground level 55.6 cm [22 in], McCreedy 2011). Lucy's warbler has also adapted to nesting in tamarisk (*Tamarix* spp.), and the species has increased in abundance in the tamarisk thickets of the Grand Canyon (Yard et al. 2004). Yard et al. (2004) found that a non-native leafhopper (*Opsius stactagolus*, occurring only on tamarisk) composed 49% of the warbler's diet at these Grand Canyon sites. The adaptability of Lucy's warbler to exploit alternative nesting trees and food resources suggests that the species will respond readily to restoration management in riparian areas across the ecoregion.

The loss of riparian and desert wash habitats is a major threat to Lucy's warbler populations. Lucy's warbler is not found in urban, suburban, or agricultural areas (Rosenberg et al. 1987, Corman 2005, Garrett 2008). Agricultural and urban development has eliminated Lucy's warbler from its historic nesting habitats in the Imperial and Coachella valleys in California (Garrett 2008). McCreedy (2011) noted that the depletion of xeric woodland (e.g., mesquite and palo verde) to just 7% of wash corridor surface area at study sites in western Arizona represented an ecological bottleneck for woodland nesting species. Riparian habitats are continually reduced by cumulative disturbances such as direct habitat destruction through clearing, flood control projects, damming, channelization, agricultural conversion, grazing, invasion of exotic annuals, OHV activity, urbanization, groundwater pumping, and more recently in the Sonoran Desert, fire. Flow regulation, groundwater pumping, and concurrent salt buildup threaten areas with existing native riparian and

xeroriparian vegetation. Depth to groundwater is a limiting factor that affects the distribution of native plant species within the riparian zone (Lite and Stromberg 2005, Shafroth et al. 2010). Cottonwood trees lose their vigor if groundwater levels fall below 3 m (10 ft.), and even deep-rooted mesquite will decline if groundwater levels drop to 12 m (40 ft.). Groundwater withdrawals for human use put native species at risk and promote the spread of invasives such as tamarisk and invasive annual grasses (Stromberg et al. 2007, Garrett 2008).

Fire is increasing in frequency in riparian areas of the southwestern U.S. for a number of reasons: a lack of flood flows in regulated river systems with a subsequent buildup of litter and woody debris, increased human ignitions, an increase in fire-adapted invasive species, combined with typical or climate-change-induced drought cycles (Ellis 2001). In a study of the lower Colorado River, Busch (1995) found that wildfire could be expected to burn over 20% of the riparian vegetation along the lower Colorado River each decade. Xeroriparian species growing in desert washes are not fire-adapted and they re-populate very slowly following fire (Esque and Schwalbe 2002). McCreedy (2011) identified fire as the greatest threat to the persistence of riparian and xeroriparian species and recommended protection of native mesquite and palo verde woodlands to reduce annual invasives and minimize fire danger.

The adaptability of Lucy's warbler to various nest substrates and alternate food sources suggests that the species will respond readily to restoration efforts. Management in riparian areas and desert washes that provides for mature native trees and understory shrubs will benefit the warbler and other riparian and xeroriparian woodland species. Opportunities exist on public lands in the lower Colorado River valley to enhance existing mesquite bosques and restore lost riparian cottonwood and willow in areas of adequate available groundwater (McCreedy 2011, National Audubon Society 2011). Both grazing and OHV use interfere with tree and shrub regeneration and also promote the invasion of invasive species. Management actions such as the removal of feral burros, limiting grazing or removing livestock, and re-routing OHV trails away from desert washes can reverse the trend of woodland decline (Garrett 2008, McCreedy 2011). For example, the removal of cattle from the San Pedro National Conservation Area (NCA) in Arizona resulted in a rapid recovery of riparian and associated mesquite and with it the return of several bird species of concern. Four years after cattle were removed from the NCA, understory vegetation recovered and Lucy's warbler abundance increased from 13.8–20.81 detections/km (Kreuper et al. 2003). Other bird species responded as well: insectivores, granivores and omnivores all increased significantly, with insectivores showing the strongest response. For restoration efforts that require vegetation clearing to reduce the presence of invasives, Yard et al. (2004) warned that managers should consider the availability of alternate habitats for species such as Lucy's warbler that use tamarisk for nesting before proceeding with broad-scale tamarisk clearing projects.

Attributes and Indicators

Attribute	Indicator	Indicator Rating				Citation
		Poor	Fair	Good	Very Good	
habitat	Loss & degradation of riparian mesquite habitat	extensive development			no development	Otahal, C.D., 2006, Johnson et al., 1997
interspecific interactions	brood parasitism	prevalent			not present	Otahal, C.D., 2006; Johnson et al., 1997
habitat	Overgrazing of mesquite scrub	present			not present	Otahal, C.D., 2006; Johnson et al., 1997

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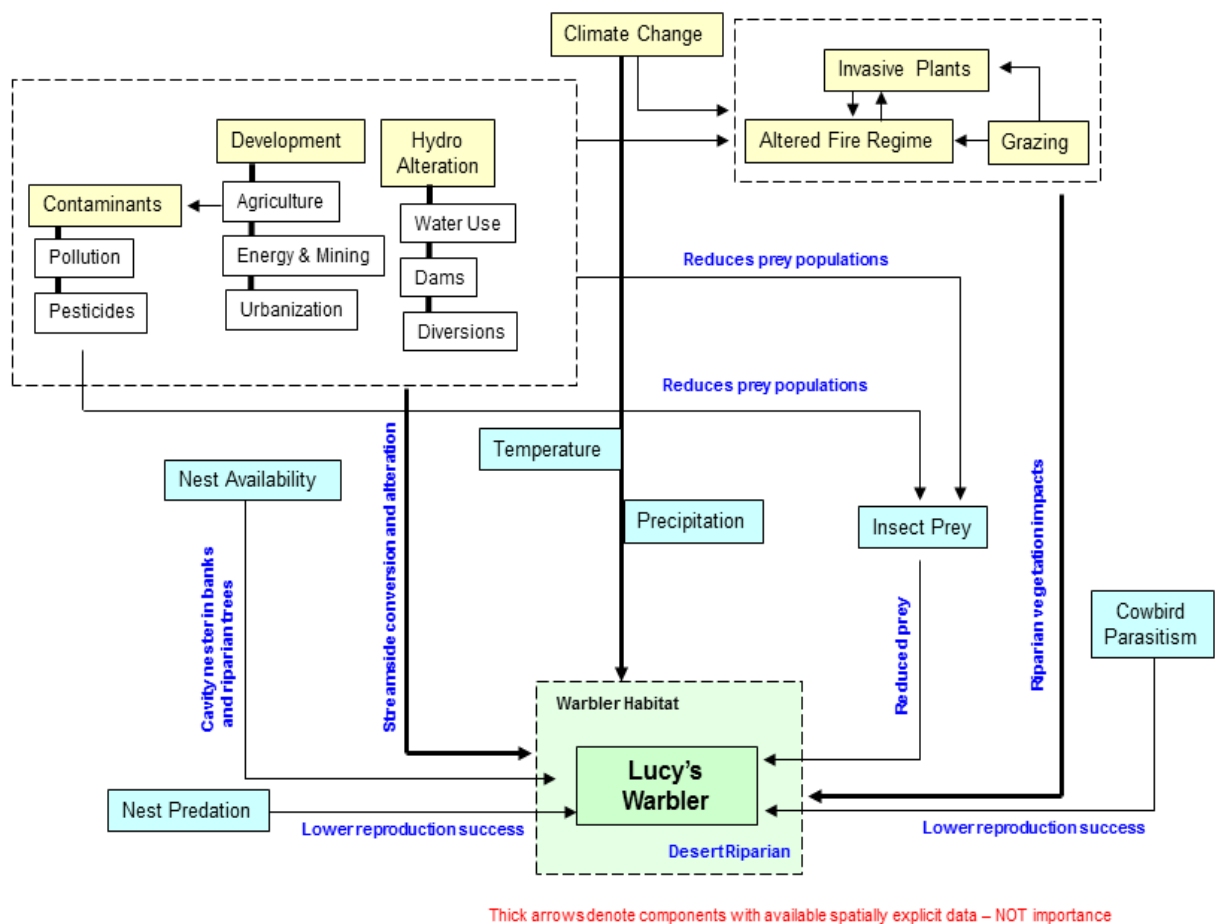
Shafroth, P.B., C.A. Brown, and D.M. Merritt (eds.). 2010. Saltcedar and Russian olive control demonstration act science assessment: U.S. Geological Survey Scientific Investigations Report 2009–5247, U.S. Geological Survey, Reston, Virginia. 143 p.

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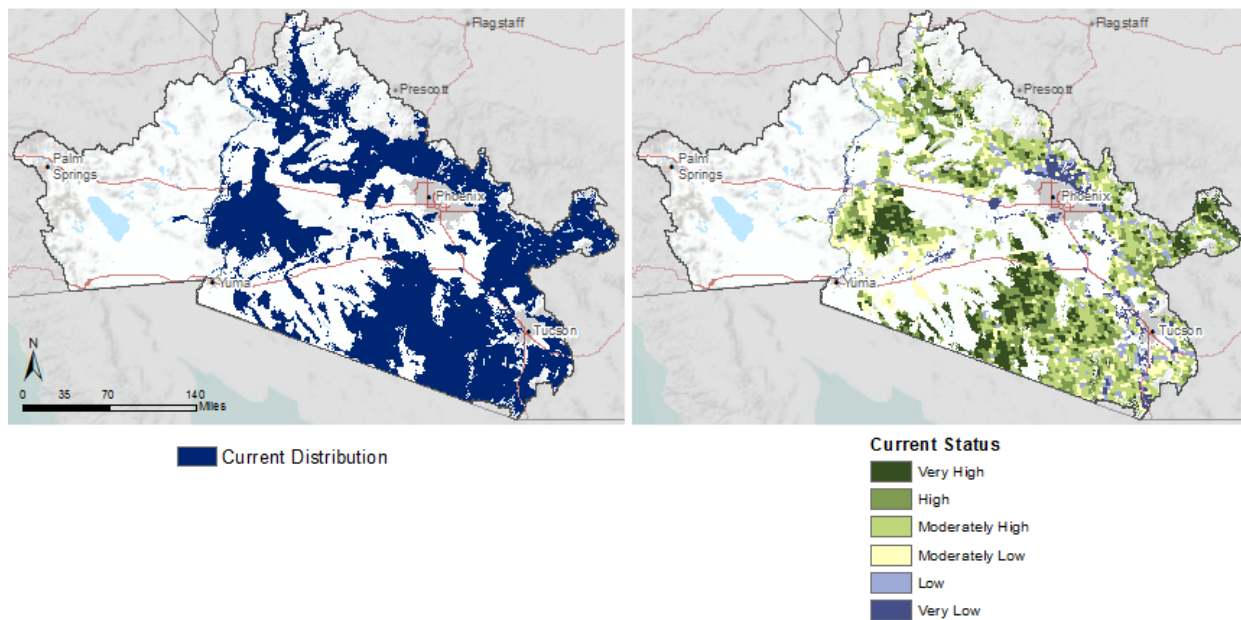
Stromberg, J.C., V.B. Beauchamp, M.D. Dixon, S.J. Lite, and C. Paradzick. 2007. Importance of low-flow and high-flow characteristics to restoration of riparian vegetation along rivers in arid south-western United States. *Freshwater Biology* 52:651–679.

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Lucy's Warbler Conceptual Model

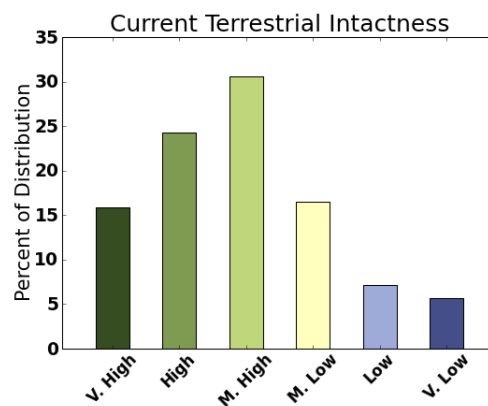


MQ D1. What's the current distribution and status of Lucy's warbler (and historic occupied habitat if available), seasonal and breeding habitat, and movement corridors (as applicable)?



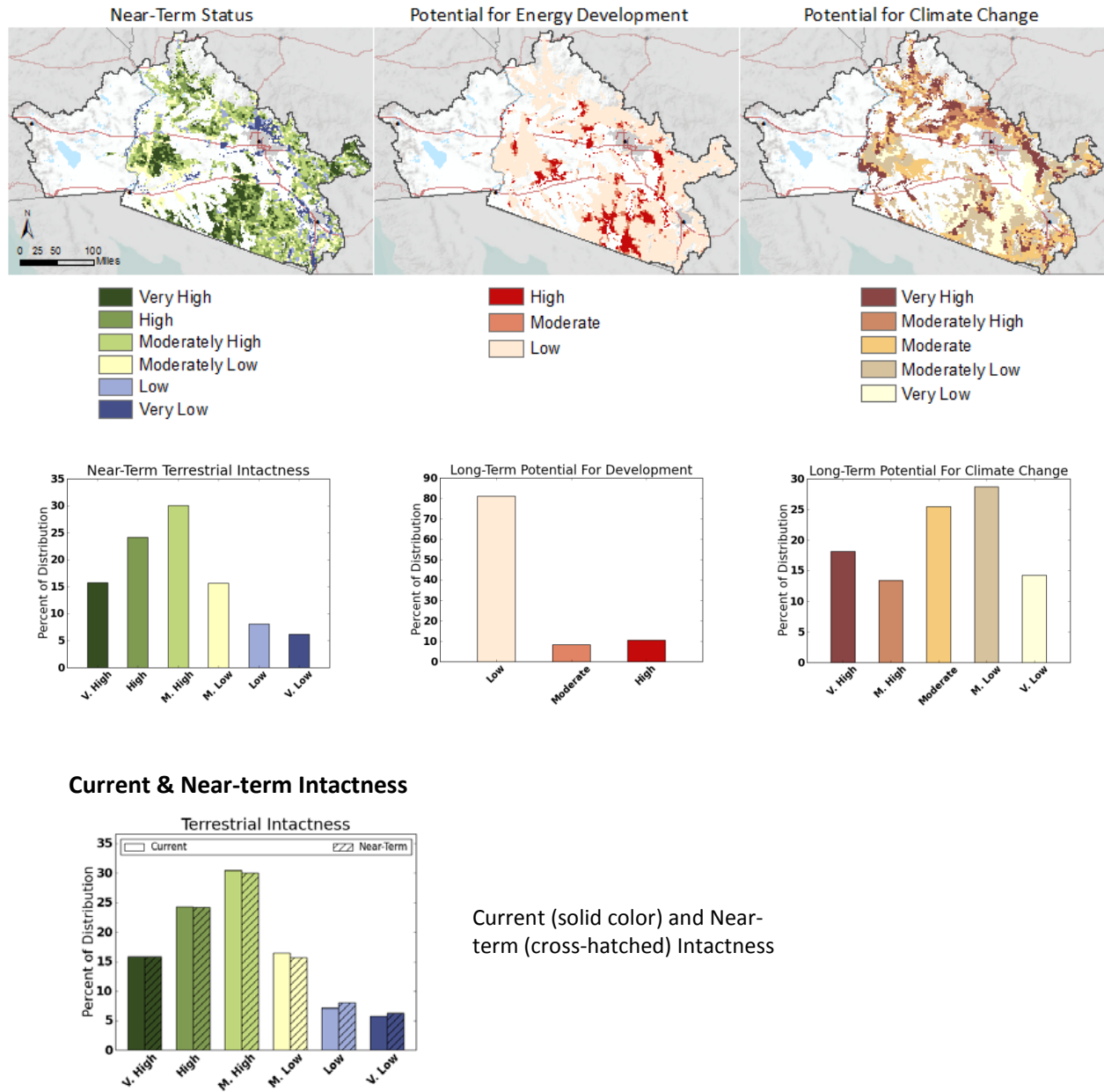
Data Sources:

Arizona GAP and California Wildlife Habitat Relations



MQ D6. What sites and movement corridors are vulnerable to change agents in the near-term horizon, 2025 (development, fire, invasive species) and long-term change horizon, 2060 (climate change)?

Lucy's Warbler Potential for Change



Mountain Lion – *Puma concolor*



Mountain lions are habitat generalists and have adapted to a wide range of environmental conditions (Weaver et al. 1996). The three main components defining high quality mountain lion habitat is abundance of prey species (e.g., mule deer, elk, and bighorn sheep), steep, rugged terrain, and vegetative cover to allow for the successful stalking of prey (Hornocker 1970, Koehler and Hornocker 1991). Mountain lions can inhabit all elevations but prefer open mixed hardwood and coniferous forest vegetation zones below timberline. Terrain ruggedness is a better predictor than vegetation cover in some landscapes such as the Sonoran Desert, and the species is fairly widespread throughout the ecoregion. However, availability of abundant prey (especially in winter) is the most important factor in supporting a strong lion population. Mountain lions are solitary predators, highly territorial, and display a wide variability in home range sizes (males 25->500 sq mi and females 8->400 sq mi). Territory size, which often shifts seasonally, is determined by a number of ecological and allometric factors including abundance of prey – higher prey densities often result in smaller home ranges (Grigione et al. 2002). Hemker et al. (1984) reported some of the largest known home range sizes for mountain lions in southern Utah with males occupying up to 513 sq mi and females up to 426 sq mi. A typical mountain lion population consists of resident males and females in occupied territories, transient males and females moving across the landscape looking to establish their own territories, and dependent kittens of resident females (Lynch 1989).

At the ecoregion level, mountain lions require fairly large home ranges with ample food and cover (provided by vegetation cover and/or rugged terrain). They also require the ability to disperse widely in search of prey and new territories as this is important component of their life history. Mountain lions can tolerate significant human disturbance (Weaver et al. 1996); however, they do avoid developed and semi-developed areas unless dispersing to new territories, which is normally conducted at night when under more stressful circumstances (Beier 1995). The most important threat to mountain lions in the ecoregion is overall habitat degradation due to residential development, recreational development, and road building. For example, Van Dyke et al. (1986) reported road densities > 0.6 km/sq km as poor for mountain lion due to avoidance behavior and direct mortality through increased conflict with humans.

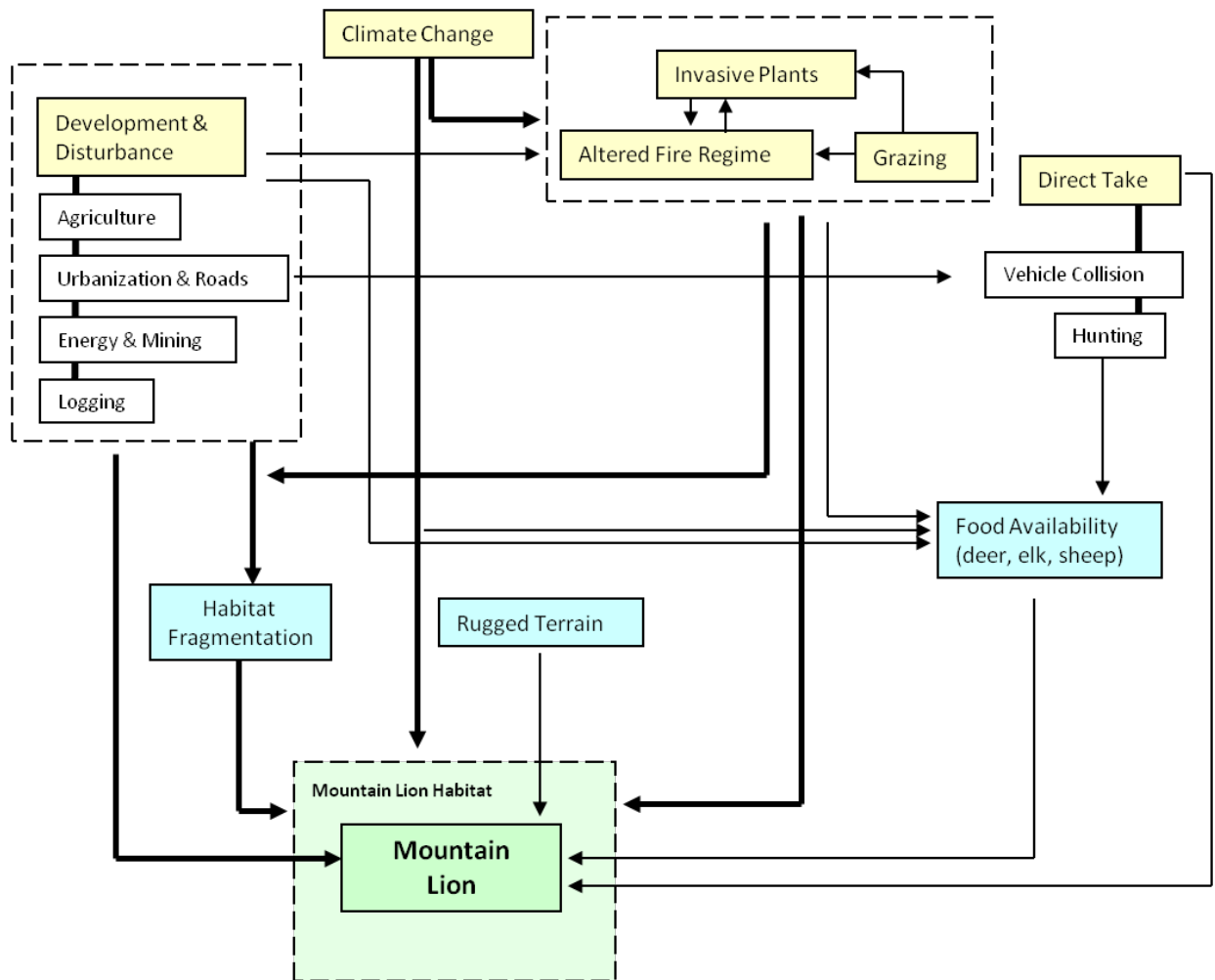
Attributes and Indicators

Attribute	Indicator	Indicator Rating				Citation
		Poor	Fair	Good	Very Good	
prey	ungulate density	low	medium	high	very high	Julander and Jeffrey (1964)
habitat degradation	road density	.6 km/sq km	0.4	0.2	0	Van Dyke et al. (1986)
habitat	cover & terrain	very dense or open cover	-	-	rugged terrain with mixed cover	Riley (1998)
habitat degradation	human development	Highly developed	moderate developed	low developed	no development	Van Dyke et al. (1986)

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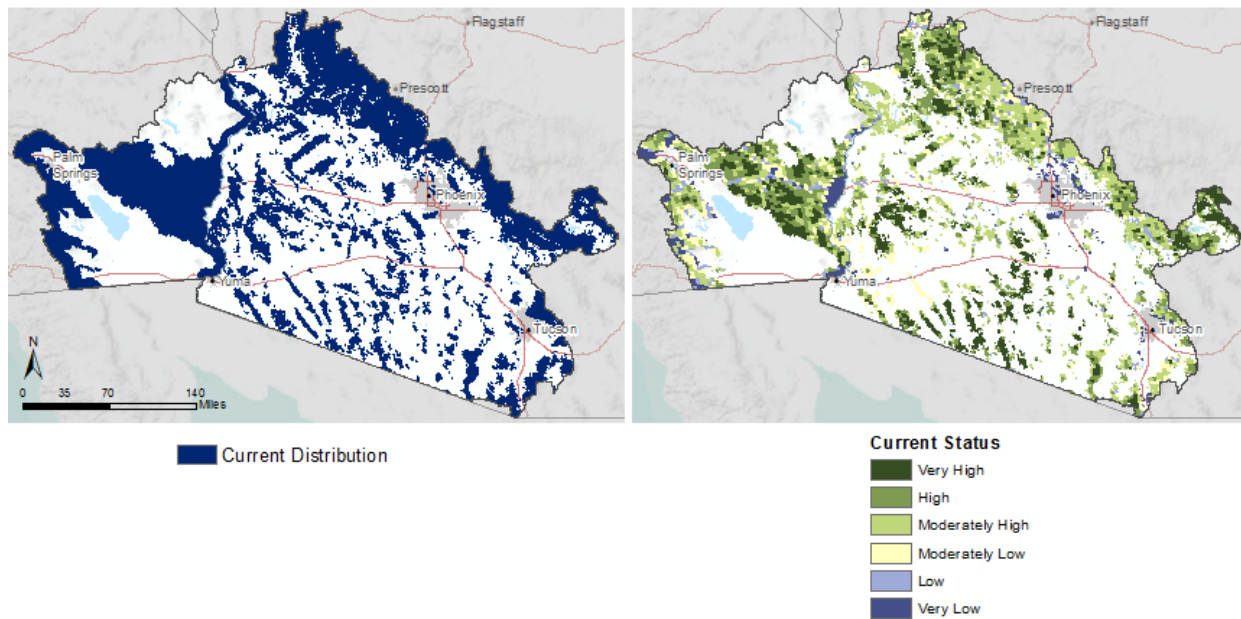
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Mountain Lion Conceptual Model



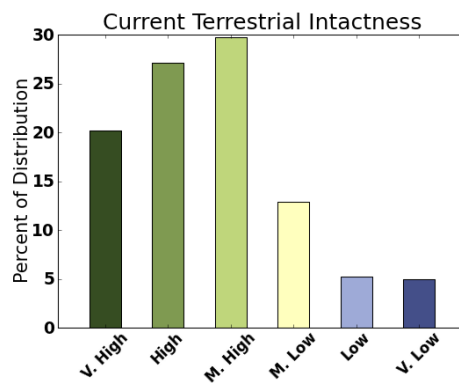
Thick arrows denote components with available spatially explicit data – NOT importance

MQ D1. What's the current distribution and status of mountain lion (and historic occupied habitat if available), seasonal and breeding habitat, and movement corridors (as applicable)?



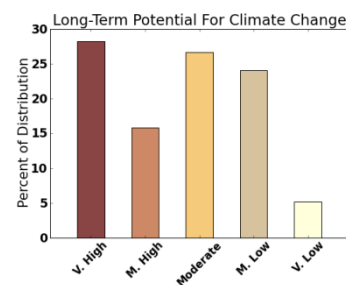
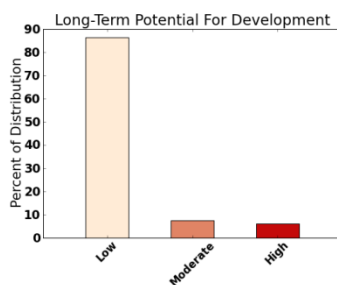
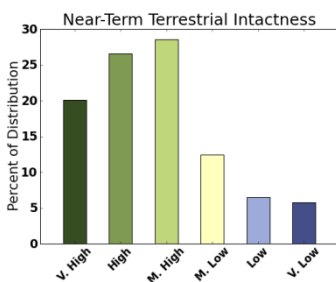
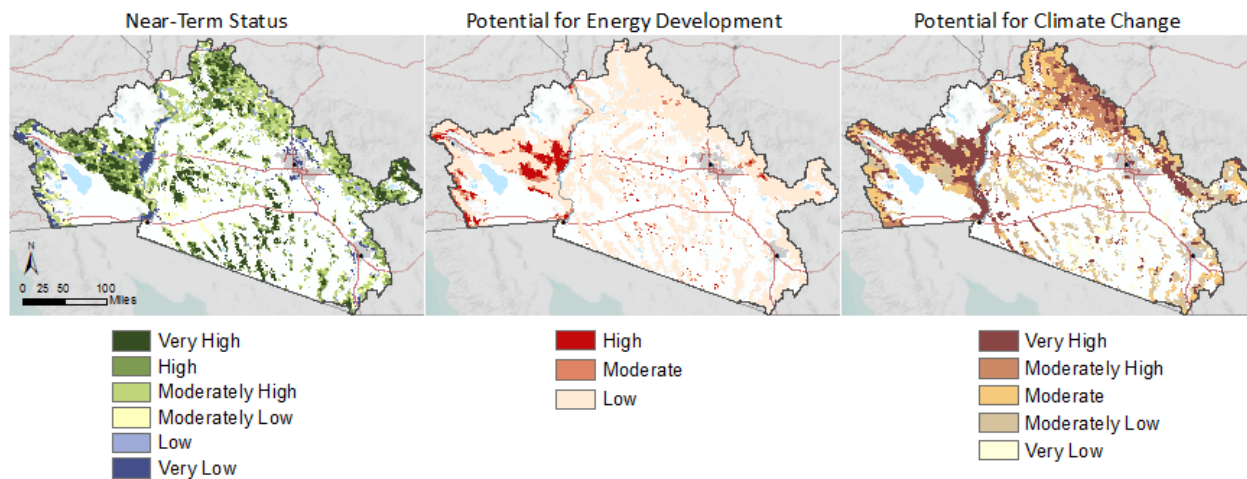
Data Source:

Arizona GAP and California GAP

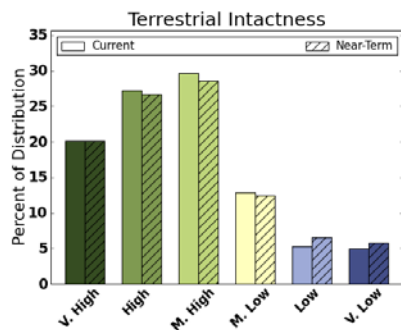


MQ D6. What sites and movement corridors are vulnerable to change agents in the near-term horizon, 2025 (development, fire, invasive species) and long-term change horizon, 2060 (climate change)?

Mountain Lion Potential for Change



Current & Near-term Intactness



Current (solid color) and Near-term (cross-hatched) Intactness

Mule Deer – *Odocoileus hemionus*



Mule deer have the ability to occupy a diverse set of habitats but are most commonly associated with sagebrush communities (Mule Deer Working Group 2003; Theodore Roosevelt Conservation Partnership 2011). Shrub communities are important to mule deer for food and shelter, and the connection of such seasonal habitats is critical to the survival of mule deer populations (Theodore Roosevelt Conservation Partnership 2011). Like most deer, mule deer are browsers and rely on a diverse range of plants for their nutrition. In late spring to early fall, mule deer eat mostly forbs and grasses, while in late fall they eat the leaves and stems of brush species, and in winter to early spring they must survive on just twigs and branches (Theodore Roosevelt Conservation Partnership 2011).

So, while mule deer forage on a wide variety of plant species, they also have very specific seasonal foraging requirements, and variety and high nutritional content across seasons is imperative to the survival of populations (Watkins et al. 2007). Ideally, mule deer consume around 3 quarts of water per day for every 100 lbs of body weight (Wallmo 1978 and Wallmo 1981). Mountain lions are the top predators in the ecoregion.

Despite their adaptability, mule deer populations have been decreasing in numbers since the latter third of the 20th century. In Utah, the 2007 post-hunting season population was estimated to be 302,000 deer well below the long-term management objective of 426,000 individuals (Utah Division of Wildlife Resources, Statewide Management Plan for Mule Deer 2008). There are a myriad of stressors on mule deer, but the most significant threats involve how their habitat is changing and disappearing (Theodore Roosevelt Conservation Partnership 2011). The vegetative species composition has been modified extensively with the invasion of non-native plants such as cheatgrass (Watkins et al. 2007). Cheatgrass out-competes most native plant species in a moisture-limited environment and changes the site-specific fire ecology, resulting in a loss of important shrub communities (Watkins et al. 2007). Plant species composition has also changed due to excessive livestock foraging, successional changes caused by fire suppression, and the disturbance and conversion of habitat (Watkins et al. 2007). In addition to the change in plant species composition, active fire suppression has changed the vegetation structure to result in the accumulation of unnaturally high fuel loads that can lead to massive fires (Watkins et al. 2007; Mule Deer Working Group 2011). Other factors that contribute to the decline of mule deer populations include habitat fragmentation due to gas, mineral, and oil exploration and increased competition with elk when habitat is poor or limited (Mule Deer Working Group 2011). Numerous diseases impact mule deer but two viral hemorrhagic diseases - epizootic hemorrhagic disease (EHD) and bluetongue (BTV) - are recognized as the most devastating. In recent years, chronic wasting disease and tuberculosis are emerging (Mule Deer Working Group 2003).

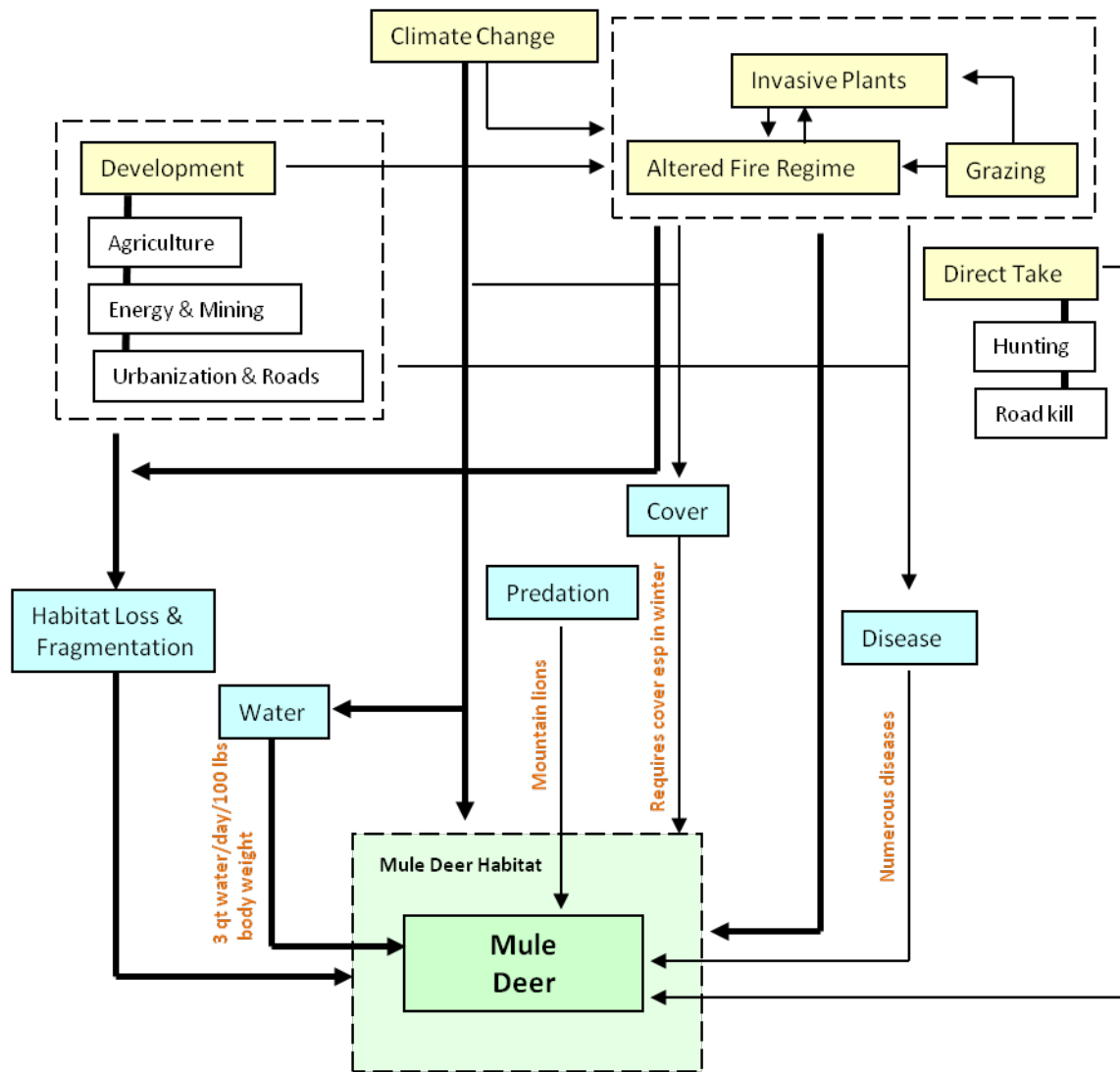
Attributes and Indicators

Attribute	Indicator	Indicator Rating				Citation
		Poor	Fair	Good	Very Good	
habitat degradation	distance from oil wells	<2.7 km	-	-	>3.7 km	Sawyer et al. (2006)
habitat degradation	distance from roads	>200m	-	-	>500 m	
habitat	loss, fragmentation, drought, fire, low quality					http://www.ndow.org/wild/animals/facts/mule_deer.shtml
habitat	vegetation/food preference as associated with fire suppression	Large, hot fires			small, cool, frequent fires (early successional plants)	Mule Deer Working Group - Western Assoc. of Fish & Wildlife agencies, 2003
habitat	variety of vegetation	homogenous			mosaic of early successional habitat (food) & tree-dominated habitats (cover)	Mule Deer Working Group - Western Assoc. of Fish & Wildlife agencies, 2003

References Cited

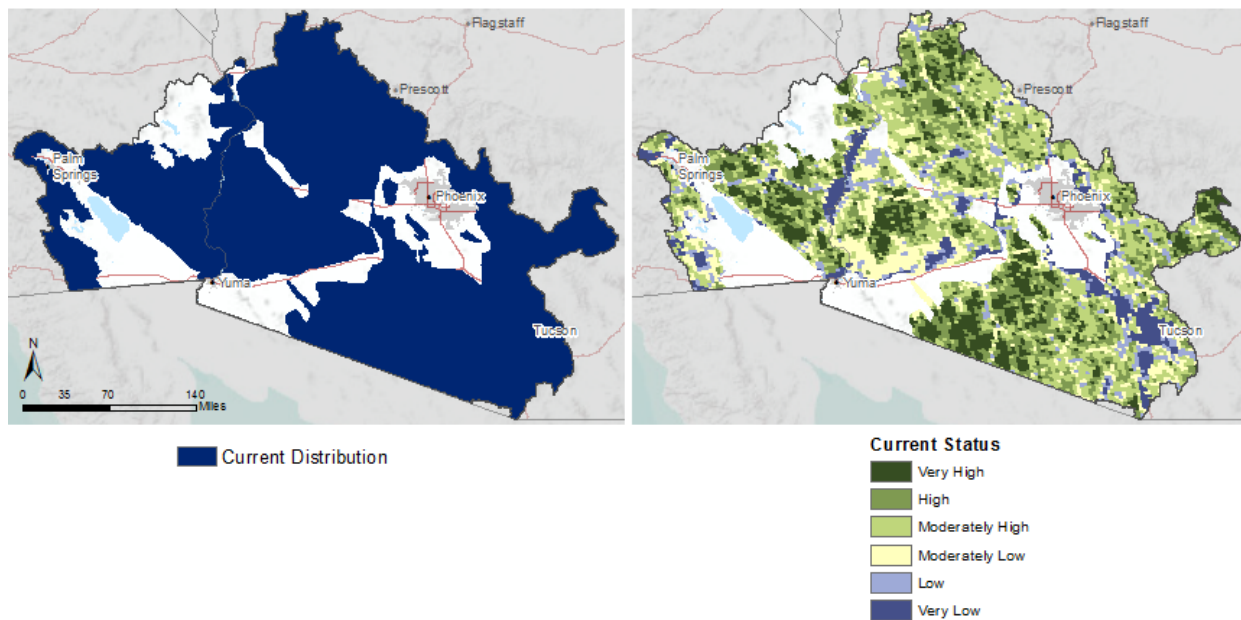
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Mule Deer Conceptual Model



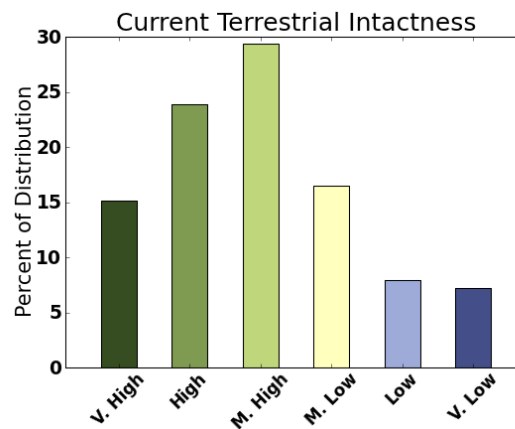
Thick arrows denote components with available spatially explicit data – NOT importance

MQ D1. What's the current distribution and status of mule deer (and historic occupied habitat if available), seasonal and breeding habitat, and movement corridors (as applicable)?



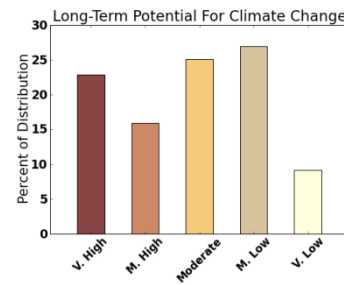
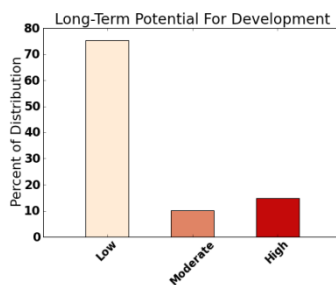
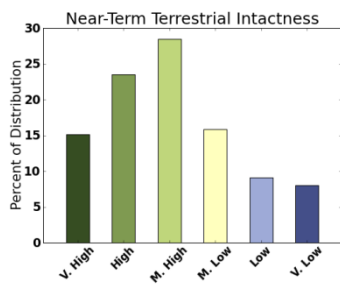
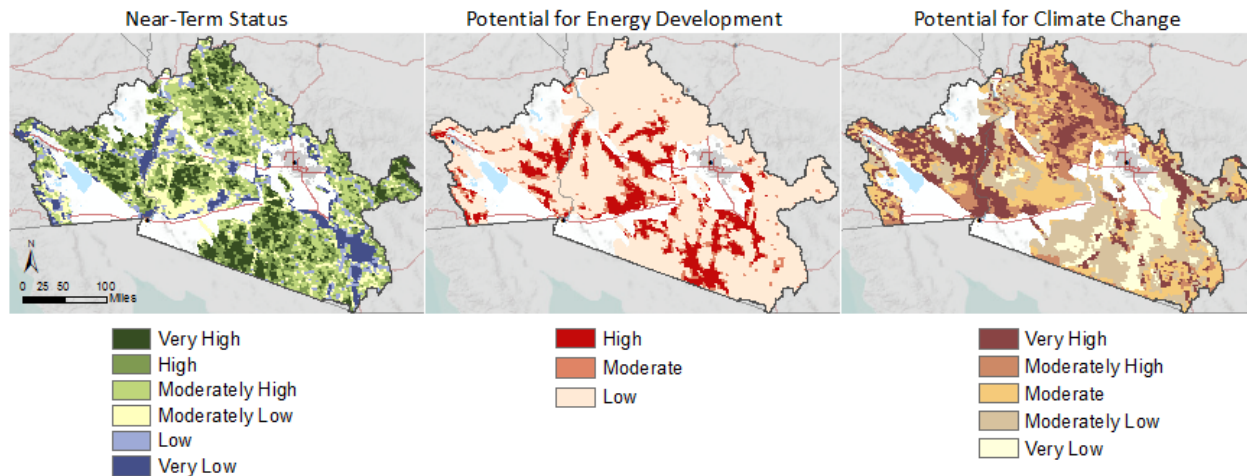
Data Sources:

Arizona Game and Fish Department, California Wildlife Habitat Relations, Mule Deer Habitat of North America

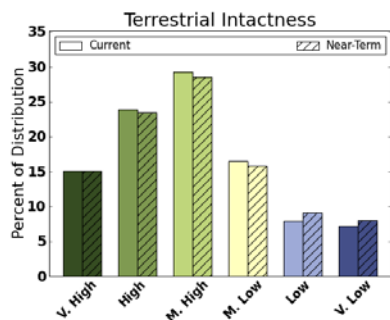


MQ D6. What sites and movement corridors are vulnerable to change agents in the near-term horizon, 2025 (development, fire, invasive species) and long-term change horizon, 2060 (climate change)?

Mule Deer Potential for Change



Current & Near-term Intactness



Current (solid color) and Near-term (cross-hatched) Intactness

Southwestern Willow Flycatcher (*Empidonax traillii extimus*)



The southwestern willow flycatcher was state listed as endangered by California in 1990 and the subspecies *extimus* was federally listed as endangered in 1995 (USFWS 2002). Range-wide, flycatcher experts estimate the species overall population at 1300 to 2000 individuals (McCarthy 2005, Sogge et al. 2005). Once common in riparian willow thickets, the flycatcher is now found in the California Sonoran Desert only along San Felipe Creek on the far western edge of the ecoregion; it is also recorded as a probable breeder at several sites on the west bank of the lower Colorado River. In Arizona, breeding territories for the flycatcher are more often recorded on Colorado River tributaries, rather than along the river's mainstem where suitable habitat is limited. However, mainstem Colorado River riparian habitats are critically important to and more heavily used by migrating flycatchers in the spring and fall (USFWS 2011). The largest concentrations of breeding territories (> 500) occur in Arizona on the Gila and lower San Pedro rivers and along the Salt River and Tonto Creek inflows to Roosevelt Lake (USFWS 2011). The Arizona Game and Fish Department manages another important flycatcher breeding habitat (30 territories) at Alamo Lake State Wildlife Area, La Paz County, Arizona.

Southwestern willow flycatchers are most often found near low gradient streams with broad floodplains and riparian thickets of the proper density. Thickets of tree saplings and shrubs, such as Goodding willow (*Salix gooddingii*), mesquite (*Prosopis* spp.), seepwillow (*Baccharis salicifolia*), arrowweed (*Tessaria sericea*), tamarisk (*Tamarix* spp.), and blackberry (*Rubus* spp.) may be overtopped by a canopy of mature cottonwood (*Populus fremontii*, USFWS 2002). Most willow flycatchers occur below 1219 m (4000 ft.); a smaller segment of the population nests at higher elevations above 2438 m (8000 ft.) in Geyer willow (*Salix geyeriana*, McCarthy 2005). The southwestern willow flycatcher has adapted to life in the dynamic riparian zone where nesting habitat is in constant flux (USFWS 2011). In a study at Roosevelt Lake, Arizona, Cardinal and Paxton (2005) found that, even when mature native riparian vegetation was available, it was not used by radio-tagged nesting flycatchers, because of its rarity (1% of total). Instead, the birds used mixed mature (a mix of native and non-native vegetation at least 5 years old, 28% of available habitat) for nesting and woodland of all ages for foraging. Canvassers for the Arizona Breeding Bird Atlas found that 54% of 59 atlas records for the flycatcher were recorded in native vegetation or native vegetation mixed with tamarisk and 29% of survey detections originated in tamarisk or tamarisk-dominated riparian growth (McCarthy 2005). Tamarisk has similar structural characteristics to native riparian shrubs and woodland. At Roosevelt Lake, flycatchers choose tamarisk nesting habitat even when native patches are still available (Sogge et al. 2005). Researchers there also observed flycatchers moving into younger riparian growth that filled flood-scoured areas or receding lake margins (Hatten and Paradzick 2003). Although Drost et al. (2001) found a significant difference in diet among flycatchers in native vs. tamarisk habitats, Sogge et al. (2005) could find no differences in physiological condition, nesting success, or survivorship among flycatchers nesting in either vegetation type. Yard et al. (2004) found that tamarisk patches provided an alternative food source, a non-native leafhopper (*Opsiurus stactagolus*), that occurs only on tamarisk, which is readily utilized by insectivorous birds.

A major threat to the sustainability of southwestern willow flycatcher populations is the management of water resources in the arid southwestern (Marshall and Stoleson 2000, USFWS 2002). During the 20th century, urban and agricultural developments created increasing demands for water use, resulting in the

damming and manipulation of most of the region's streams and rivers. Flow alterations below dams, groundwater pumping, and reductions in flooding and the scouring of stream channels all contribute to losses of native riparian vegetation (for more details see flow regime overview in Invasive Species, Tamarisk, Section XX). For example, the flow regime of the lower Colorado River has been so altered that it meets none of the requirements for native plant regeneration: perennial or near-perennial flow, periodic spring floods, sediment delivery, salinity levels < 4g/l NaCl, and shallow depth to groundwater for seedling and sapling survival (Stromberg et al. 1996, Glenn et al. 1998). As a result, the Lower Colorado Management Unit for flycatcher critical habitat, where the species was once common, has just 13% of known southwestern willow flycatcher breeding territories (USFWS 2005) because of the loss of suitable flycatcher habitat.

Loss of connectivity between suitable riparian habitats negatively affects flycatcher recovery. As appropriate habitats become more rare and fragmented, dispersing flycatchers lose opportunities for colonization and recruitment. Although most known populations of southwestern willow flycatchers are concentrated in localized groups of breeding territories, non-nesting, pre-nesting, and post-nesting males use larger territories than nesting birds, and they move between nesting territories and even between drainages (Cardinal and Paxton 2005, USFWS 2011).

Fire is increasing in frequency in riparian areas of the southwestern U.S. for a number of reasons: increased human ignitions, the spread of fire-adapted invasive species, a buildup of litter and woody debris from a lack of flooding, and lowered water tables, in addition to typical or climate-change induced drought cycles (Busch 1995, Busch and Smith 1995, Ellis 2001). Increased fire risk in tamarisk-dominated riparian areas is also one of the greatest threats to willow flycatcher breeding sites (USFWS 2002). Marshall and Stoleson (2000) noted the loss of several southwestern willow flycatcher breeding sites due to fire in the mid- to late-1990s in riparian areas along the San Pedro and Gila rivers in Arizona. Presumably these habitats are lost to nesting flycatchers until suitable riparian vegetation (of at least 5 years of age) has re-established.

Livestock grazing and the incidence of associated brown-headed cowbirds both negatively affect the success of southwestern willow flycatchers. The effects of livestock grazing on riparian areas, both historically and currently, have been well-documented (Marshall and Stoleson [2000] provide a thorough literature review). Over time, a regime of livestock grazing in riparian areas during the late spring and summer can eliminate the regeneration of cottonwood and willow saplings and shift the abundance of plant species from wetland obligate species to more upland species or to invasives such as tamarisk (Ohmart 1996, Dobkin et al. 1998). Removal of livestock from the upper San Pedro River National Conservation Area (managed by BLM) in the late 1980s allowed the regeneration of riparian understory species and resulted in the subsequent confirmation of southwestern willow flycatcher as a breeding species there within a decade of the elimination of grazing (McCarthy et al. 1998).

Brown-headed cowbirds occur in association with grazed systems and row crop agriculture, and their parasitism of southwestern willow flycatcher nests contributes to flycatcher decline and limits flycatcher recovery (Whitfield and Sogge 1999, USFWS 2002, Brodhead et al. 2007). In a study conducted on the Kern River in California, Uyehara et al. (2000) found that cowbird parasitism explained 44% of the variation in flycatcher population growth rates. Cowbirds parasitize flycatcher nests at various rates depending on region of occurrence, surrounding land use, and the configuration of nesting habitat (Whitfield and Sogge 1999, Uyehara et al. 2000). Small, isolated willow flycatcher populations in smaller habitat patches suffer much

higher rates of nest parasitism. Whitfield and Sogge (1999), using pooled data from the Kern River in California, the Grand Canyon, and other sites in Arizona, found that the majority of parasitized nests fail and that flycatcher fledging rate in parasitized nests was significantly lower than in un-parasitized nests, 11% vs. 47% respectively. In heavily parasitized areas, cowbird trapping can be an effective means to increase the flycatcher's reproductive success. Uyehara et al. (2000) noted that a restoring a self-sustaining flycatcher population may be possible at their Kern River, California study site if parasitism levels are maintained below 8.4% with the help of continued cowbird control.

The Southwestern Willow Flycatcher Recovery Plan (USFWS 2002), critical habitat designation documents (USFWS 2005, 2011), and Chapter 10 in Finch and Stoleson (2000) all give thorough overviews of management options for restoring flycatcher populations rangewide. In managing for flycatcher habitat, it is of primary importance to manage for or select a stream length with a flow regime and groundwater level that supports riparian woodland and shrubland. Existing riparian habitats can be classified and ranked into potential, suitable, or occupied habitats for willow flycatcher (Finch et al. 2000). Potential habitats lack essential flycatcher habitat elements and may require more restoration cost and effort; suitable unoccupied habitats may only require minimal management effort in the form of recreation or grazing restrictions to eventually attract nesting flycatchers. Because willow flycatcher dispersal is more common within drainages than among drainages (Cardinal and Paxton 2005), management effort directed toward enhancing or restoring new flycatcher habitat may have the best chance for success if it is focused on areas within the dispersal distance of existing nesting populations (about 15 km, USFWS 2002) to increase the chance of colonization or to connect isolated populations. Management options recommend the exclusion of livestock in flycatcher-occupied sites (or in *potential* sites adjacent to or near occupied sites), but dormant season grazing may be allowed if high-quality shrub cover can be maintained (Finch et al. 2000). Since flycatchers use tamarisk for nesting, tamarisk control and clearing should be done on a site-by-site basis after monitoring for southwestern willow flycatcher presence. A more gradual or patch replacement of tamarisk (in patches not exceeding 25% of the total such as might occur with scouring floods), may ensure that sufficient riparian woodland remains available for nesting (Finch et al. 2000). Cowbird trapping may be considered if monitoring reveals that nest parasitism exceeds about 8–10% (Uyehara et al. 2000, Finch et al. 2000).

Attributes and Indicators

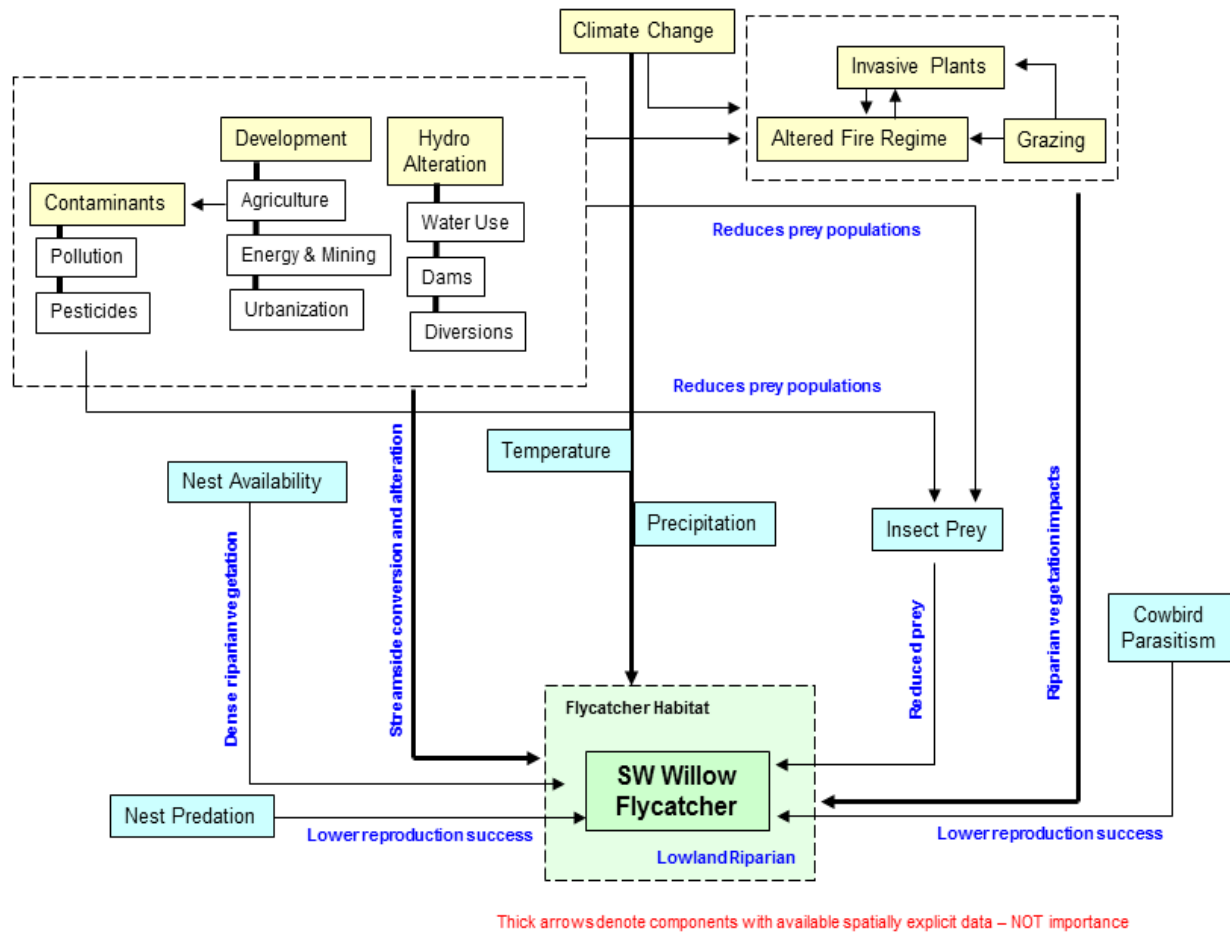
Attribute	Indicator	Indicator Rating				Citation
		Poor	Fair	Good	Very Good	
habitat	proximity to surface water	>100 m	50-100 m	25-50 m	0-25 m	Sogge and Marshall (2000)
connectivity	distance between occupied sites	>30 km	15-30 km	2-15 km	<2 km	Finch et al. (2002)
habitat degradation	recreation	present in high intensity	present in moderate intensity	present in low intensity	absent	Marshall and Stoleson (2000)

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Southwestern Willow Flycatcher Conceptual Model

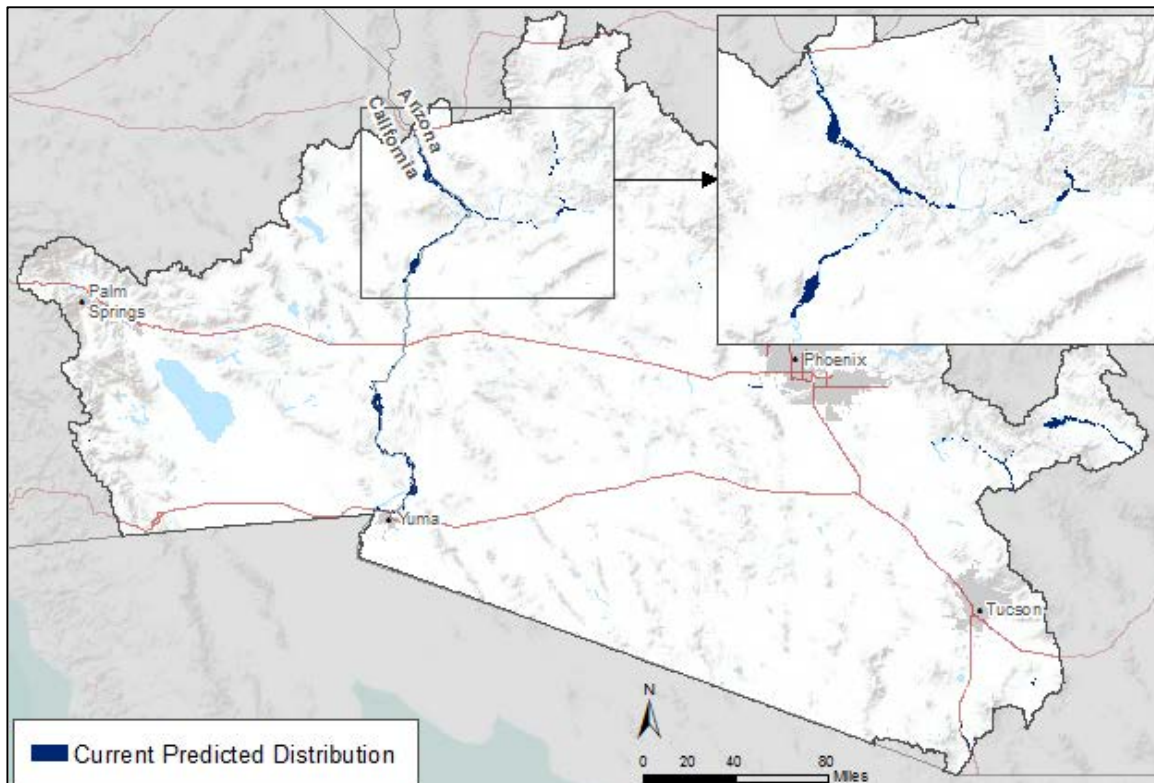


Access to a data portal to examine the results in greater detail is available at:

<http://www.blm.gov/wo/st/en/prog/more/climatechange.html>.

MQ D1. What's the current distribution and status of southwestern willow flycatcher (and historic occupied habitat if available), seasonal and breeding habitat, and movement corridors (as applicable)?

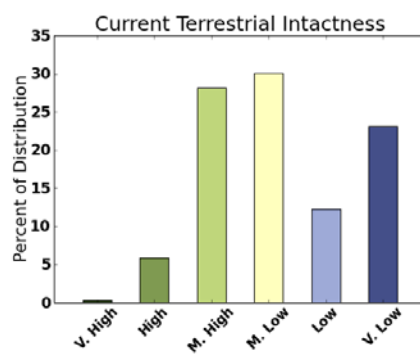
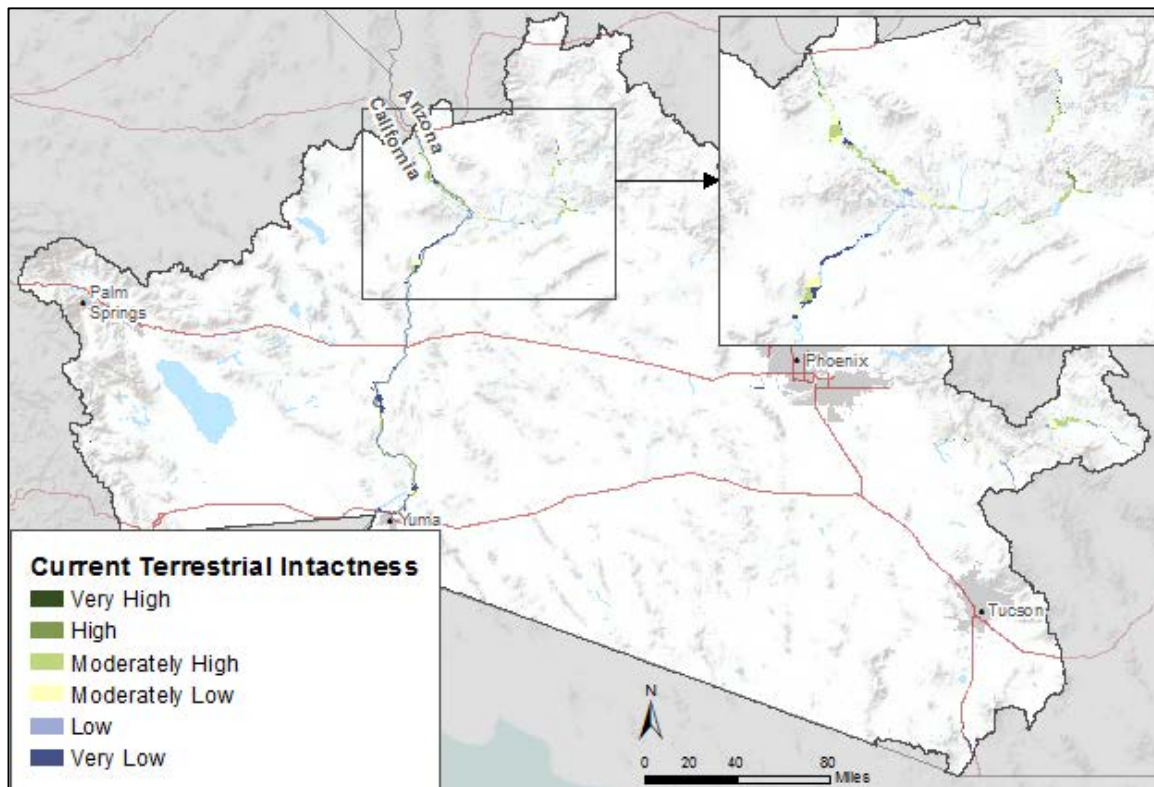
Current Distribution



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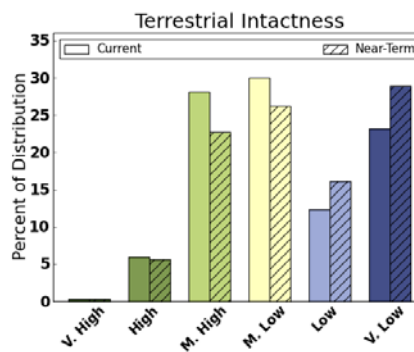
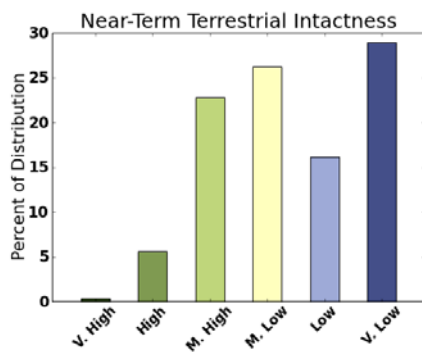
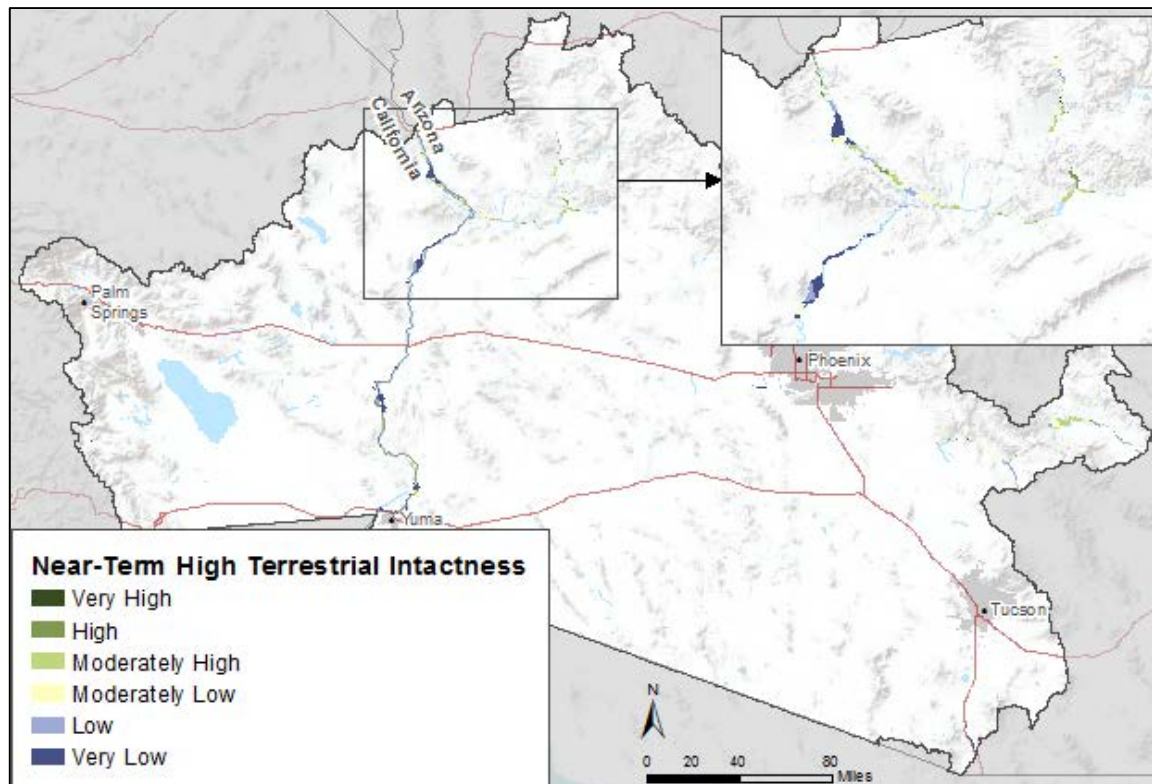
USFWS Critical Habitat

Southwestern Willow Flycatcher Status

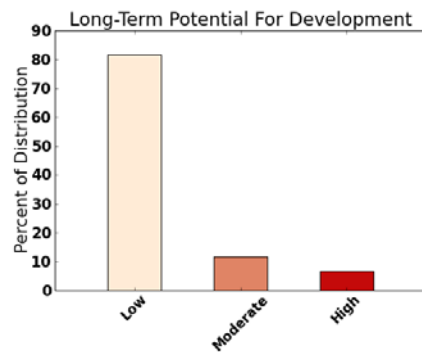
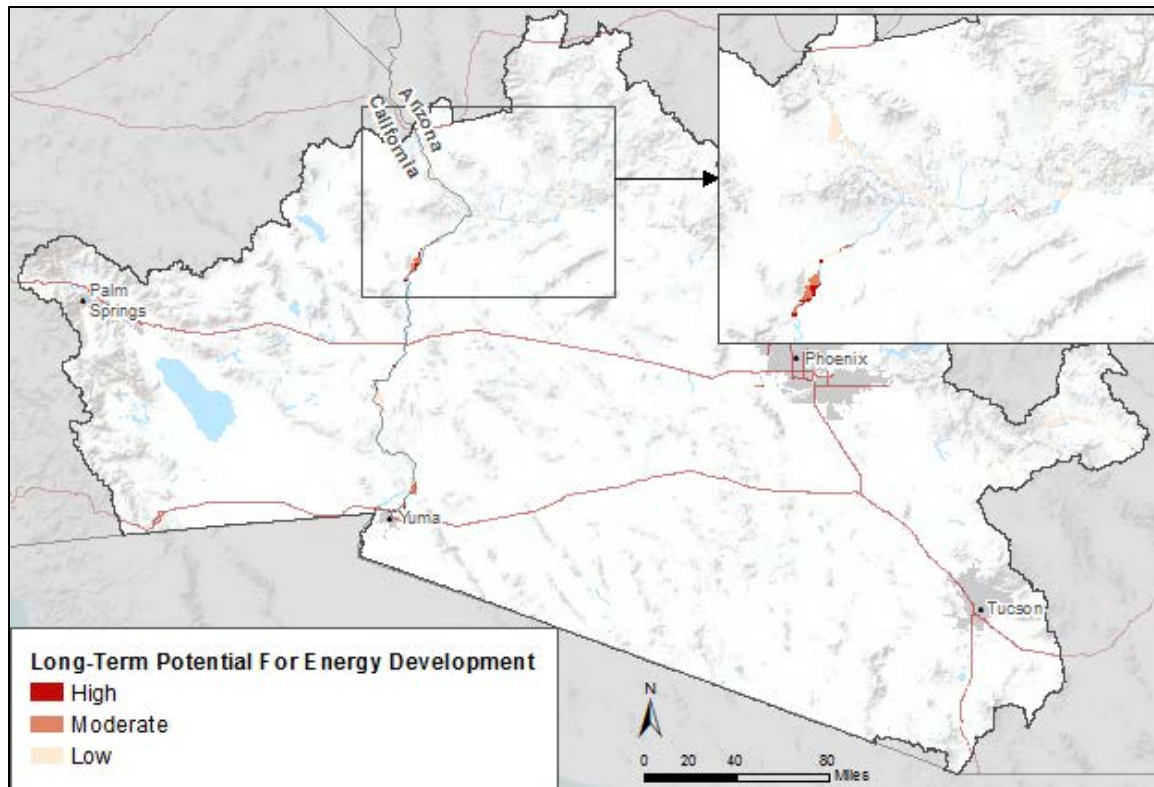


MQ D6. What sites and movement corridors are vulnerable to change agents in the near-term horizon, 2025 (development, fire, invasive species) and long-term change horizon, 2060 (climate change)?

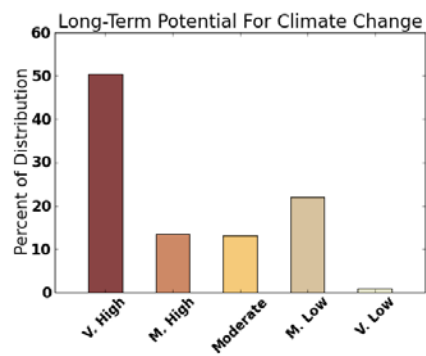
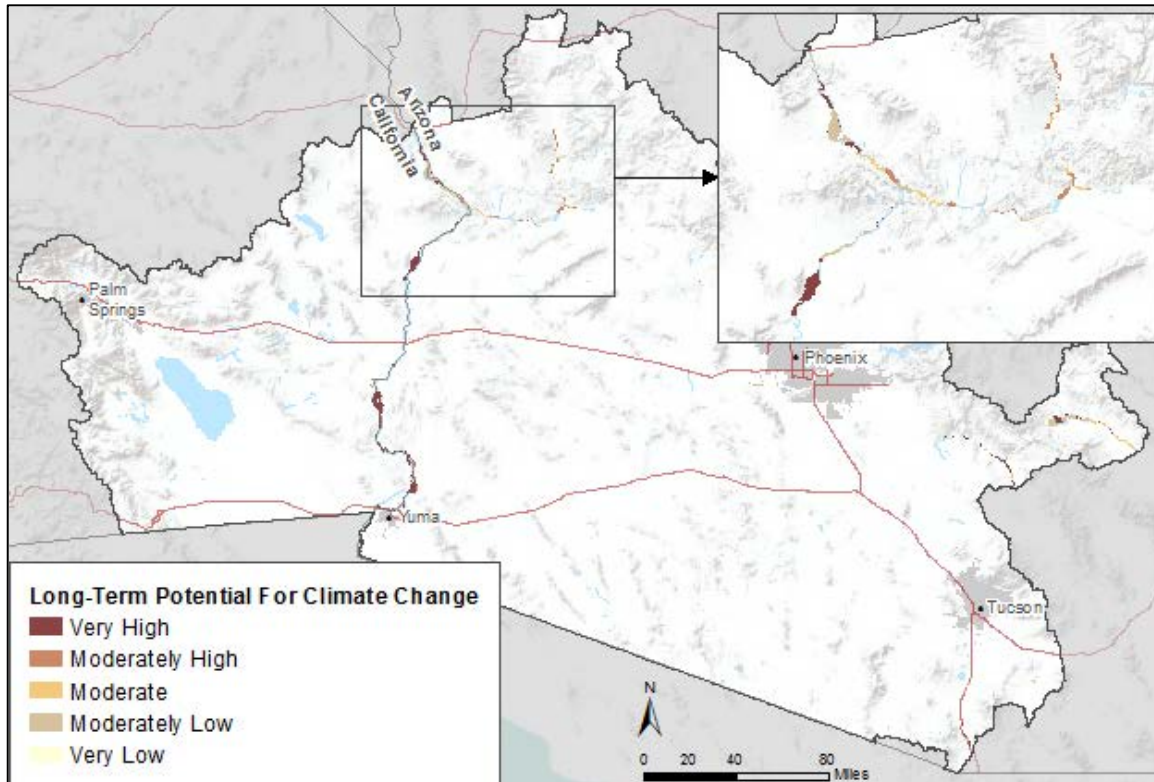
Southwestern Willow Flycatcher Potential for Change: Near-Term (2025) Development



Southwestern Willow Flycatcher Potential for Change: Maximum (Long Term) Energy Development



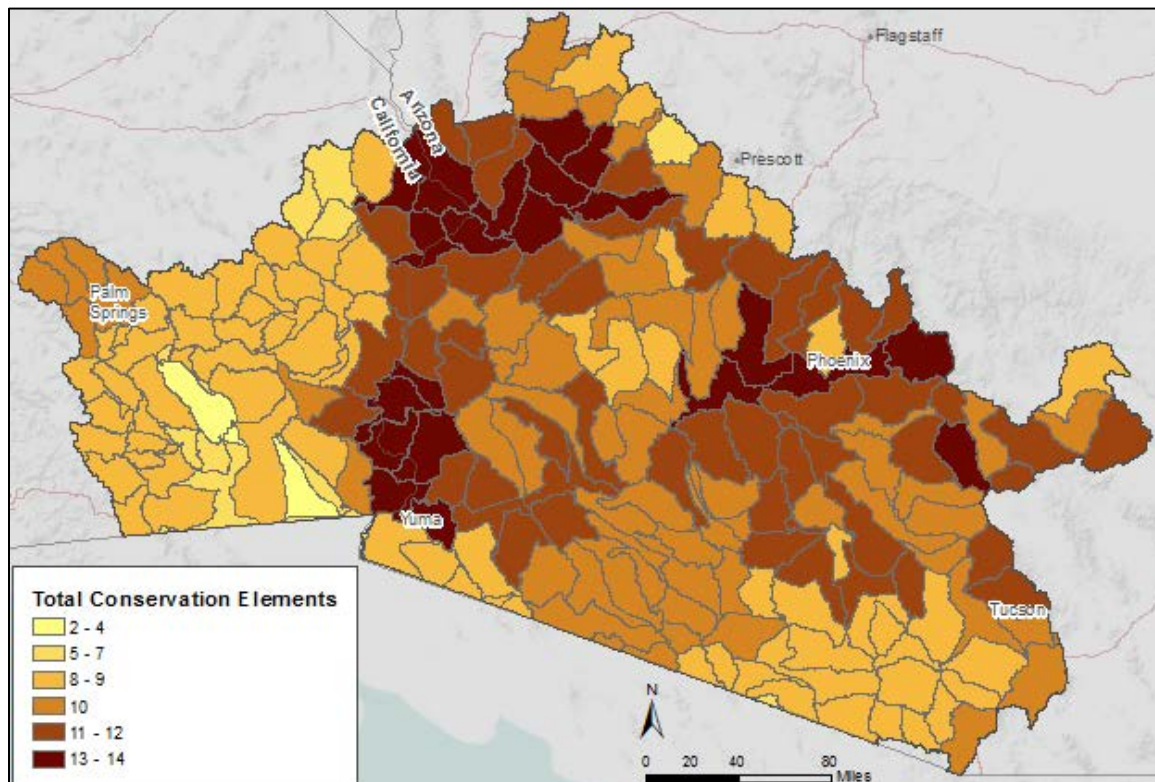
Southwestern Willow Flycatcher Potential for Change: Potential for Climate Change



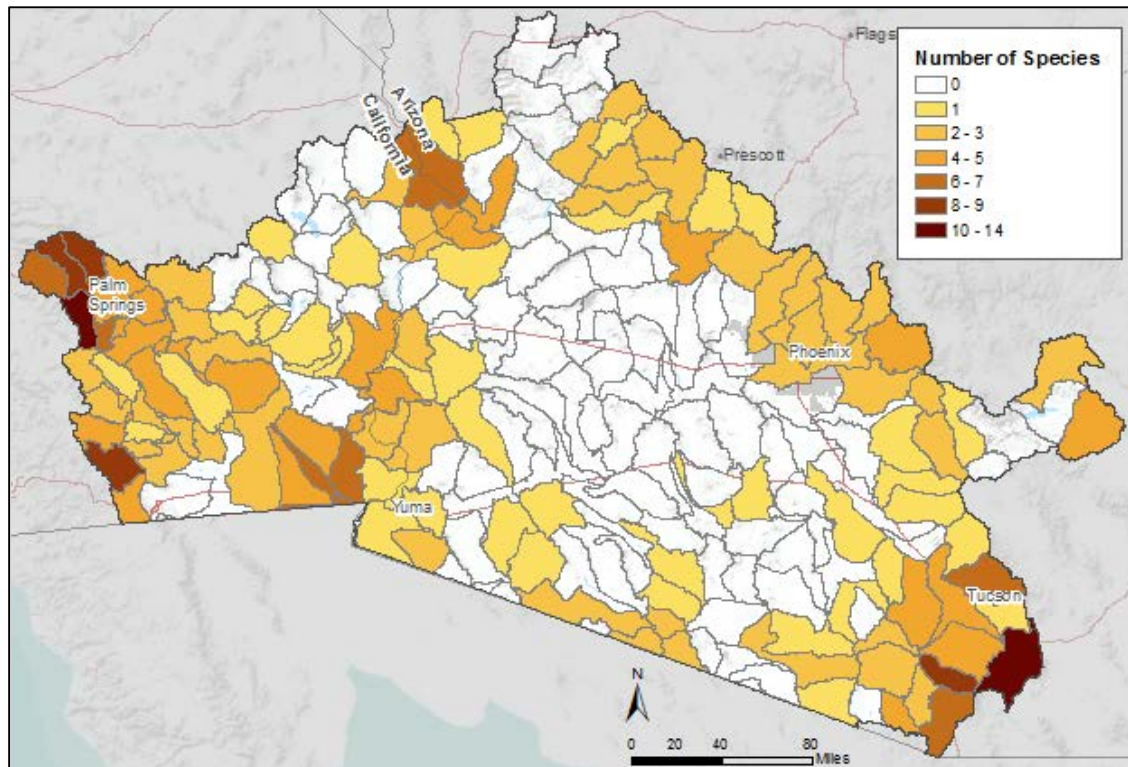
NatureServe Element Occurrence Data

BLM acquired species element occurrence data NatureServe Natural Heritage data enumerated by HUC for the REA. From the data, which was organized by 5th level HUC, four different map-based products were generated, including (1) number of all species, (2) number of globally critically imperiled and imperiled species (G1 and G2 species), (3) number of globally critically imperiled, imperiled, and vulnerable species (G1-G3 species), and (4) number of USFWS listed threatened and endangered species.

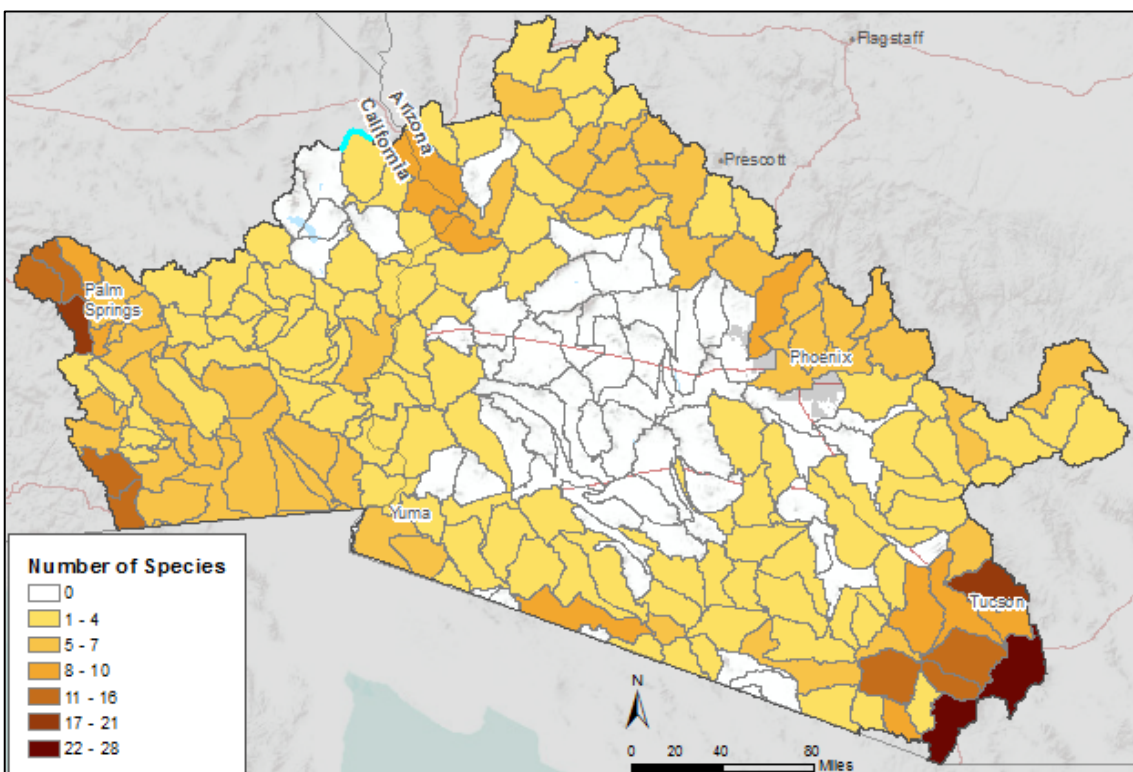
Number of All Species



Number of G1 and G2 Species



Number of G1–G3 Species



Number of USFWS Listed Species

