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Natural Resource Stewardship and Science

Natural Resource Condition Assessments for Six Parks in the Mojave Desert Network

Natural Resource Report NPS/MOJN/NRR—2019/1959



ON THE COVER
Death Valley National Park, November 2011
Photograph courtesy of Erica Fleishman

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Prologue

Publisher's Note: Changes in publishing requirements, and in some cases scientific delays, resulted in several NRCA reports not being published in a timely manner. These publications reported on studies initiated in the 2013–16 timeframe. Since Natural Resource Condition Assessments provide a snapshot-in-time evaluation of park resource conditions, it is important to note that data discovery and analyses for this study was conducted a few years prior to publication. Thus, park conditions reported in this document pertain to that time period. Please see the Publisher's Note at the beginning of the Executive Summary or Chapter 2 for dates specific to this report.

Executive Summary

Publisher's Note: Natural Resource Condition Assessments provide a snapshot-in-time evaluation of park resource conditions. For this report, most or all of the data discovery and analyses occurred during the period of 2014 to 2015. Thus, park conditions reported in this document pertain to that time period. Due to revised publishing requirements and/or scientific delays, this report was not published until 2019.

This Natural Resources Condition Assessment (NRCA) for six parks in the Mojave Desert Network—Death Valley National Park, Grand Canyon–Parashant National Monument, Joshua Tree National Park, Lake Mead National Recreation Area, Manzanar National Historic Site, and Mojave National Preserve—is intended to provide scientific information on park resources that was requested by park managers and will be applicable to their planning and decision-making. Region-level and park-unit level objectives for the NRCA, and associated priorities for assessment, largely were identified by managers from these six parks. For cases in which few data were available to directly assess the conditions of high-priority resources, park managers and our assessment team agreed that we would review relevant literature and outline data requirements and analytical methods that likely would allow such an assessment. Because the distributional ranges of several iconic species, and a high proportion of stressors to natural resources, occur over much of the Mojave Desert, park managers and our assessment team agreed that wherever possible, we would assess resources at the regional level. We agreed to supplement the regional assessment with evaluations of the condition of a small number of resources that were of interest to a particular park or subset of parks.

High-priority natural resources across the Mojave Desert Network include elements of the three components of biological or ecological diversity: structure, composition, and function. National Park Service (NPS) personnel indicated that management of areas under their jurisdiction might best be informed by a focus on assessment of processes rather than individual species. On the basis of our scoping and discussions with Mojave Desert Network personnel, we agreed to emphasize the following resources and stressors: climate change, land use, air quality, fire, the distribution and dynamics of non-native plants, habitat quality for Mojave desert tortoise (*Gopherus agassizii*), connectivity and adaptive capacity of desert bighorn sheep (*Ovis canadensis nelsoni*), water availability for native wildlife, feral burros, bats, and mule deer (*Odocoileus hemionus*) in Grand Canyon–Parashant.

The responses of different species, ecological processes, and non-climate stressors to ongoing and projected changes in climate by 2100 will not be uniform; some responses likely will be considered negative, whereas others may be considered positive. As summarized in a recent, publicly available technical assessment of climate change in the southwestern United States, the scientific community has high confidence that annual and seasonal average temperatures, the length of the frost-free season, and the number and intensity of heat waves have increased and will continue to increase. Additionally, the scientific community has high confidence that mountain snowpack will decrease. There is medium-high confidence that spring precipitation will decrease and that drought severity will increase.

Land use and its effects of land cover both within and outside of parks affect natural resources within parks. Data on land use are highly relevant to analyses of the human footprint, which generally is defined as the effects on the environment or natural resources of human presence and actions. We examined both the physical effect area of the human footprint (the actual area covered by human-created features) and the ecological effect area (the area within which species or ecological processes are affected by human-created features). We found that publicly available data on human-created features often were inconsistent with data collected by park units or with data derived from high-resolution, remotely sensed images. As a result, if these inconsistencies are not addressed, estimates of trends in physical and ecological effect areas across the Mojave Desert Network may be inaccurate and may not reflect responses of natural resources to management actions. Changes in land use and land cover from 1990 through 2006 were limited, and the change process per se likely is not having a substantial effect on most native species that inhabit the Mojave Desert Network.

There is a strong seasonal cycle of air quality in the Mojave Desert. Air quality typically is lowest during summer, especially near the mountain passes that connect the Mojave Desert with upwind sources of urban, industrial, and agricultural pollutants. In autumn and winter, regional air quality is good to excellent. From 2008–2012, air quality on the basis of ozone concentration criterion was poor in Death Valley, Joshua Tree, Manzanar, and Mojave National Preserve and moderate to poor in Lake Mead. Air quality on the basis of nitrogen and sulfur was good in Death Valley, Joshua Tree, and Manzanar and good to moderate in Lake Mead and Mojave National Preserve. Air quality on the basis of the visibility criterion was moderate at Death Valley, Joshua Tree, Lake Mead, Manzanar, and Mojave National Preserve.

The number of fires in the Mojave Desert is increasing as the distribution and density of non-native invasive plants increases and climate changes. To inform expectations and management of contemporary fire risk, we explored recent and projected future changes in fire probability, including changes in land cover and the timing and amount of precipitation. We quantified the probability of fire occurrence by relating whether locations burned to a suite of environmental variables, including upland fuel type. We also quantified the suite of factors that might cause a fire to become relatively large (> 20 ha) or lead to a high probability of a relatively large fire. We found strong negative relations between the modeled probability of fire occurrence and distance from roads. The probability of fire occurrence increased as the productivity of invasive grasses (e.g., red brome [*Bromus rubens*]) and topographic roughness increased. The probability of fire occurrence was greatest in the desert montane ecological zone. The modeled probability of relatively large fires was strongly and positively associated with ignition by lightning and with early dates of the maximum productivity of invasive grasses in the preceding calendar year. High topographic roughness and increasing distances from roads also were strongly and positively associated with the probability of a relatively large fire. Large fires were most likely in middle elevation shrubland.

Because field data were limited, we used climate-based models to assess the distribution and abundance of red brome and Sahara mustard (*Brassica tournefortii*), two common, non-native invasive species in the Mojave Desert. At the regional level, these were the only two species for which occurrences across extensive climatic gradients were recorded and reported in sufficient

quantity to facilitate bioclimatic envelope modeling. Throughout the Mojave Desert Network, current climate is consistent with the climate in locations where red brome is known to occur. Model projections indicated that each park contains areas that could support high abundance of red brome, especially in southern Nevada and western Arizona. Climatic habitat for Sahara mustard is present throughout all of the parks except the eastern portion of Grand Canyon–Parashant. Climatic habitat that could support high abundance of Sahara mustard is present throughout Lake Mead, in the drier areas of Joshua Tree and Mojave National Preserve, and along the southern and eastern edges of Death Valley. As minimum temperature increased, the probability of presence of red brome increased considerably. As maximum temperature increased, the probability of presence of Sahara mustard increased substantially. As minimum temperature and spring precipitation increased, the probability of high abundance of red brome increased. By contrast, as summer precipitation increased, the probability of high abundance of Sahara mustard increased.

At the suggestion of experts on Mojave desert tortoise, we explored relations among habitat quality for desert tortoise, habitat quality for three species that prey on desert tortoise (Common Raven [*Corvus corax*], Golden Eagle [*Aquila chrysaetos*], and coyote [*Canis latrans*]), and habitat quality for black-tailed jackrabbit (*Lepus californicus*), an alternative source of prey for predators of desert tortoise. Habitat quality for desert tortoises and black-tailed jackrabbits was not highly correlated. Areas where habitat quality for at least one predator was high covered about 46% of the habitat for desert tortoise. Areas in which habitat quality for all predators was high overlapped with about 3% of desert tortoise habitat and 8% of habitat for black-tailed jackrabbits.

Desert bighorn sheep are a high priority for management throughout the Mojave Desert. We measured the condition of desert bighorn sheep within individual parks and across the Mojave Desert Network in three ways. First, we assessed the connectivity and genetic diversity of desert bighorn sheep populations in Joshua Tree National Park and Mojave National Preserve. Second, we assessed the strength of gene flow among populations in Death Valley National Park and Lake Mead National Recreation Area. Third, we assessed the potential role of different populations and corridors in maintaining genetic connectivity and demographic connectivity. We found that genetic diversity (a measure of adaptive capacity) of populations of desert bighorn sheep in or near Death Valley was moderate to high. Our assessment suggested that over time, populations in the southern end of Death Valley have been smaller or more isolated than populations in northern Death Valley. It appears that there is little gene flow between eastern and western Death Valley. Genetic diversity of most populations in Lake Mead was relatively high and similar to that of populations in Death Valley, Mojave National Preserve, and Joshua Tree. We found considerable differences between genetic and demographic connectivity of bighorn sheep populations; levels of both short-term and long-term genetic connectivity were much higher than levels of demographic connectivity. Therefore, managing to maintain genetic connectivity among desert bighorn sheep populations would not necessarily maintain the potential for natural recolonization. We identified locations where restoration of genetic or demographic connectivity via mitigation of anthropogenic barriers might be particularly effective for restoring either gene flow or colonization potential.

Small, isolated montane springs are the primary natural source of water across much of the Mojave Desert Network, and the distribution and flow of these montane springs is associated with the distribution and viability of many native species of animals. A substantial proportion of montane springs in the Mojave Desert have been excavated, fitted with diversion structures, or otherwise developed. Groundwater recharge and stores likely will decrease as climate continues to change and human demands for water increase. Therefore, the trend in condition of montane springs at the level of the Mojave Desert Network is likely to be stable or decreasing.

The condition of most native species of wildlife is likely to decrease as water availability decreases, but the magnitude of the change will vary considerably among taxonomic groups. For example, although individual desert bighorn sheep can survive for many days without free-standing water, the presence of surface water increases habitat quality for bighorn sheep and is associated with the distribution of the species. Moreoever, as precipitation becomes more variable, the abundance of bighorn sheep is likely to become more variable. Decreases in water availability may reduce fecundity of desert tortoises, and wurvival of adult desert tortoise is highly dependent on annual production of plant biomass and on not only the amount but the timing of precipitation. Kangaroo rats, by contrast, are better adapted to seasonal or short-term drought than many other taxonomic groups native to the Mojave Desert.

The Wild Free-Roaming Horses and Burros Act (Public Law 92–195) gives the NPS jurisdiction to remove wild horses and burros from the lands they manage, but other federal land management agencies do not have this option. Competition between feral burros and wildlife species in the Mojave Desert could occur through dietary overlap or displacement from high-quality habitat or water sources. Replicated, long-term exclosure studies or control-impact studies could provide insights to the biological significance of interactions between wildlife and burros.

Twenty-three of the 45 species of bats that occur in the United States occur in parks within the Mojave Desert Network. Abandoned mines provide habitat for many of the bat species that occur in the Mojave Desert, particularly because land use has reduced the extent and quality of natural habitat. Bat gates are believed to reduce the high energetic costs to bats associated with anthropogenic disturbance during roosting or hibernation, increase habitat stability by stabilizing the mine entrance, and minimize human disturbances to the mine. However, bats have abandoned mines at which gates were installed poorly, and energetics may change after gate installation. Two species of bats in the Mojave Desert were affected by white-nose syndrome in the eastern United States. Climate data from caves and mines in the Mojave Desert Network and across the western United States indicate that conditions could support growth of the disease-causing fungus. However, it is unclear how the fungus will respond to the aridity that is typical of the region.

Our ability to assess the condition of many high-priority resources was constrained by data availability and by the lack of explicit management targets. Our assessment suggested that collection or updating of publicly available data and metadata from within the Mojave Desert Network parks on roads; air quality; distributions and abundances of non-native invasive plants across time and environmental gradients; and the distributions, movements, and vital rates of high-priority species would be especially informative. The ability of any ecological element to respond to climate change

is greater when land use does not create barriers to movement of individuals or resources. We hope that the NRCA has strengthened the basis for collaboration among the research and management communities and for resource stewardship by NPS and the public.

Chapter 1. Background

Natural Resource Condition Assessments have been commissioned nationwide to evaluate current conditions of high-priority natural resources within lands managed by the National Park Service (NPS), to elucidate phenomena that are known or hypothesized to affect the condition of park resources, and to identify gaps in data and knowledge that constrain assessment and management of park resources. The aim of this Natural Resources Condition Assessment (NRCA) for six parks in the Mojave Desert Network is to provide scientific information on park resources that was requested by and will be applicable to planning and decision-making by park managers. Example processes to which the information might be relevant include but are not limited to general management plans, resource stewardship strategies, and implementation plans.

Region-level and park-unit level objectives for the NRCA, and associated priorities for assessment, largely were identified by managers from Death Valley National Park, Grand Canyon–Parashant National Monument, Joshua Tree National Park, Lake Mead National Recreation Area, Manzanar National Historic Site, and Mojave National Preserve. Early in the NRCA process, there was recognition by NPS managers and the assessment team that in many cases, few data exist with which to directly assess the conditions of high-priority resources. We agreed that where data are insufficient to allow strong inference to status and trend, the NRCA would outline data requirements and analytical methods that likely would allow such inference. Resource specialists and science coordinators were instrumental in evaluating data sources and reliability and in providing data where possible.

Early in the NRCA process, there also was recognition that consensus that resources and stressors for which assessment was a high priority rarely were restricted to one or two parks. Instead, for example, the distributional ranges of iconic species such as desert bighorn sheep and Mojave desert tortoise cover much of the Mojave Desert, including multiple parks in the Mojave Desert Network. Similarly, a high proportion of stressors—any physical, chemical, or biological event or process that has undesirable effects on a given natural resource—to natural resources in Mojave Desert Network parks are not restricted to one park, but are regional. Accordingly, we agreed to assess resources at the regional level wherever possible, and to supplement the regional assessment with evaluations of the condition of a small number of resources that were of interest to a particular park or subset of parks.

Chapter 2. Introduction and Resource Setting

Publisher's Note: Natural Resource Condition Assessments provide a snapshot-in-time evaluation of park resource conditions. For this report, most or all of the data discovery and analyses occurred during the period of 2014 to 2015. Thus, park conditions reported in this document pertain to that time period. Due to revised publishing requirements and/or scientific delays, this report was not published until 2019.

2.A. Introduction

2.A.1. Enabling legislation

The parks in the Mojave Desert Network were established over more than 35 years. Lake Mead, the first National Recreation Area in the United States, was established by Public Law 88-639 on 8 October 1964. Its designation was intended to support public recreation and to preserve the area's scenic, historic, and natural features. Manzanar National Historic Site was designated on 3 March 1992 by Public Law 102-248 to "provide for protection and interpretation of historical, cultural, and natural resources associated with the relocation of Japanese Americans during World War II." Death Valley National Park, Joshua Tree National Park, and Mojave National Preserve were established by the California Desert Protection Act on 31 October 1994 to preserve vistas, geologic formations, wildlife, and diverse ecosystems; preserve historical and cultural elements associated with Native American cultures and western exploration and settlement; provide opportunities for public recreation and for interpretation of natural and cultural features; promote public understanding and appreciation of wilderness and desert; and provide opportunities for scientific research. Grand Canyon–Parashant National Monument was established by Presidential Proclamation 7265 on 11 January 2000. The designation was intended to protect features including but not limited to geologic formations, fossils, archaeological sites, and diverse ecosystems and species.

2.A.2. Geographic setting

The Mojave Desert Network covers parts of three states (Arizona, California, Nevada) and five ecoregions (Colorado Desert, Colorado Plateau, Great Basin, Mojave Desert, and Sonoran Desert) (Figure 1).

Death Valley is the largest national park in the conterminous United States. Death Valley's 3,396,192 ac (13,744 km^2) encompass the 156-mi (251 km)-long valley itself, the Panamint Range to the west, and the Amargosa Range to the east. More than 90% of the park (3,102,456 ac [12,155 km^2]) is designated as wilderness. Elevations in the park range from 282 ft (86 m) below sea level in the Badwater Basin to 11,049 ft (3368 m) at the summit of Telescope Peak. Death Valley lies within Inyo and San Bernadino Counties in California and Nye and Esmeralda Counties in Nevada. It generally is considered to lie within the Great Basin and Mojave Desert. The park often is accessed from Las Vegas via Pahrump, Nevada. Beatty, Nevada and Ridgecrest and Bishop, California also are access points. Death Valley includes five inland sand dunes: Eureka, Ibex, Mesquite Flat, Panamint, and Saline Valley. The park has four perennial streams and 629 springs and seeps.



Figure 1. The six parks in the Mojave Desert Network that are addressed in this Natural Resources Condition Assessment (Esri, NPS).

Grand Canyon–Parashant's 1,048,321 ac (4242 km^2) lie in Mohave County in northwestern Arizona. About two-thirds of the park falls within the Colorado Plateau rather than the Mojave Desert. Administration is shared among the National Park Service's Lake Mead National Recreation Area (208,449 ac [844 km^2]), the Arizona Strip District of the Bureau of Land Management (810,661 ac [3281 km^2]), Arizona State Trust (23,206 ac [94 km^2]), and private land owners (6,005 ac [24 km^2]). About 300,000 ac (1214 km^2) are designated or eligible for designation as wilderness, and livestock grazing is allowable on about 791,017 ac (3201 km^2). The park spans the Colorado Plateau, Mojave Desert, and Great Basin. Elevations range from 1201 to 8028 ft (366 to 2447 m). The closest towns to Grand Canyon–Parashant, all more than two hours' drive away, are Littlefield, Beaver Dam, Scenic, Fredonia, Colorado City, and Centennial, Arizona; Mesquite and Bunkerville, Nevada; and St. George, Utah. There are no paved roads or visitor services within the monument. Grand Canyon–Parashant has 206 springs and seeps. Joshua Tree covers 792,623 ac (3208 km^2), of which 591,624 (2394 km^2) is designated wilderness, within the Mojave and Colorado Deserts. The park lies along the transverse ranges of the Little San Bernadino Mountains in Riverside and San Bernardino Counties, California. Five additional mountain ranges, the Cottonwood, Hexie, Pinto, Eagle, and Coxcomb Mountains, are within the park. Elevations within the park range from sea level to 5814 ft (1772 m). Joshua Tree is within 140 to 225 mi (225 to 362 km) of four major cities: Los Angeles, San Diego, Las Vegas, and Phoenix. The park's southern boundary is the northern edge of the Coachella Valley and its northern boundary is the Morongo Basin. The park contains five fan palm (*Washingtonia filifera*) oases. Joshua Tree has no perennial streams, ponds, or lakes. However, the park has four small reservoirs and 109 springs and seeps.

Lake Mead, the United States' first national recreation area, covers 1,500,000 ac (6070 km^2) of the Mojave Desert in Clark County, Nevada and Mohave County, Arizona over an elevation gradient from 499 to 5640 ft (152 to 1719 m). Lakes Mead and Mohave were created by damming of the Colorado River. The park contains nine designated wilderness areas. There are nine major access points to the park. The closest town to the visitor center and main entrance to the park, Boulder City, Nevada, is less than an hour from Las Vegas. Thermal and non-thermal springs occur throughout the park. Rodgers Spring in the Overton Arm of the lake is known for its warm temperature (86˚F [30˚C]) and relatively consistent, high flow (720 gallons [2727 liters] per minute). Downstream of Hoover Dam, Black Canyon contains numerous seeps and warm and cool springs. Lake Mead has one perennial stream, two large reservoirs, and 89 springs or seeps.

Manzanar lies on the east side of the Sierra Nevada in Owens Valley, California. The historic site covers 814 ac (3.3 km^2) at an elevation of 3799 ft (1158 m). The historical site has no surface waters. Manzanar is about 210 mi (338 km) north of Los Angeles and 50 mi (80 km) south of Bishop in Inyo County, California; the closest towns are Independence to the north and Lone Pine to the south. Manzanar is located at the southern end of the Great Basin or northern end of the Mojave Desert. Paiute Indians occupied the Owens Valley for about 1500 years before the early 1860s, when miners and ranchers colonized the valley and forcibly displaced many of the Paiute. The area was an agricultural community from about 1910 to 1930. About 120,000 U.S. citizens and residents of Japanese ancestry were interned at Manzanar during World War II.

Mojave National Preserve encompasses 1,600,000 ac (6475 km^2) of the Great Basin and Mojave and Sonoran Deserts in San Bernardino County, California. Almost half of the park (695,200 ac [2813 km^2]) is designated as wilderness. Mojave National Preserve is the third largest national park in the conterminous United States. Elevations within Mojave National Preserve range from 880 ft (268 m) to 7929 ft (2417 m). The closest towns are Barstow and Baker, California. The park is four hours' drive from Los Angeles and 90 minutes from Las Vegas. More than 240 natural springs and seeps have been recorded in Mojave National Preserve, and the park has one perennial stream. Mojave National Preserve contains the Kelso Dunes, part of an inland sand-dune system; volcanic rock formations at Hole-in-the-Wall; the granite Cima Dome; and the limestone Mitchell Caverns.

2.A.3. Number of visitors

We obtained data on the annual number of visitors for each of the Mojave Desert Network parks and tested whether there was a linear trend in the number of visitors over the last 10 years for which visitor data were publicly available (2004 through 2013).

The average annual number of visitors to Death Valley from 2004 through 2013 was 858,219 ± 104,609 (sd). The increase in the annual number of visitors during that period of time was statistically significant. However, the average annual number of visitors from 2004 through 2013 was well below that for the preceding 10 years (1,084,537 ± 129,413). The average number of visitors to Joshua Tree from 2004 through 2013 was 1,348,176 ± 66,448 per year. The increase in the annual number of visitors during that period of time was statistically significant, and the number of visitors from 2004 through 2013 was greater than that during any previous decade for which data were available. The average annual number of visitors to Lake Mead (7,229,046 ± 644,556) decreased significantly from 2004 through 2013 and was lower than the average annual number of visitors during the previous decade (8,778,311 ± 710,822). From 2004 through 2013, the average annual number of visitors to Manzanar was 79,734 ± 5255. There was no trend in the number of visitors during that time. Similarly, there was no trend in the average annual number of visitors to Mojave National Preserve over the past decade (567,494 ± 37,926). To the best of our knowledge, data on the number of visitors to Grand Canyon–Parashant are not collected.

2.B. Natural resources

2.B.1. Ecosystems

We characterized vegetation within each park on the basis of 2012 data from the Landscape Fire and Resource Management Planning Tools (LANDFIRE) project. Thirteen vegetation types cover at least 1% of Death Valley (Table 1). Shrublands cover about two-thirds of the park. Dominant species include creosote bush (*Larrea tridentata*), white bursage (*Ambrosia dumosa*), blackbrush (*Coleogyne ramosissima*), California or eastern Mojave buckwheat (*Eriogonum fasciculatum*), Mormon tea or Nevada jointfir (*Ephedra nevadensis*), spiny hopsage (*Grayia spinosa*), wolfberry (*Lycium* spp.), spiny menodora (*Menodora spinescens*), beargrass (*Nolina* spp.), buckhorn cholla (*Opuntia acanthocarpa*), bladder sage (*Salazaria mexicana*), Parish's goldeneye (*Viguiera parishii*), Joshua tree (*Yucca brevifolia*), and Mojave yucca (*Yucca schidigera*).

Table 1. Vegetation types in Death Valley National Park that cover ≥ 1% of the park's area (LANDFIRE).

Vegetation type	Area (km^2)	Proportion of park area
Sonora-Mojave Creosotebush-White Bursage Desert Scrub	5417	0.39
Mojave Mid-Elevation Mixed Desert Scrub	3361	0.24
North American Warm Desert Sparsely Vegetated Systems	1871	0.14
Barren	601	0.04
North American Warm Desert Riparian Systems	554	0.04
Inter-Mountain Basins Sparsely Vegetated Systems	450	0.03
Great Basin Pinyon-Juniper Woodland	317	0.02
Sonora-Mojave Mixed Salt Desert Scrub	317	0.02
Inter-Mountain Basins Mixed Salt Desert Scrub	239	0.02
Great Basin Xeric Mixed Sagebrush Shrubland	228	0.02
Inter-Mountain Basins Big Sagebrush Shrubland	129	0.01
Introduced Upland Vegetation-Annual Grassland	90	0.01
Inter-Mountain Basins Montane Riparian Systems	75	0.01

Woodlands cover a greater proportion of Grand Canyon–Parashant than of the other parks in the Mojave Desert Network (Table 2). These woodlands are dominated by pinyon (*Pinus edulis*, *P. monophylla*) and juniper (*Juniperus osteosperma*, *J. monosperma*, and *J. scopulorum*) trees. Among the 14 other vegetation types that cover at least 1% of the park are shrublands dominated by sagebrush (*Artemisia tridentata*), creosote bush, or white bursage and grasslands dominated by non-native annual species.

Table 2. Vegetation types in Grand–Parashant National Monument that cover ≥ 1% of the park's area (LANDFIRE).

Vegetation type	Area (km^2)	Proportion of park area
Inter-Mountain Basins Big Sagebrush Shrubland	787	0.19
Colorado Plateau Pinyon-Juniper Woodland	709	0.17
Introduced Upland Vegetation-Annual Grassland	557	0.13
Great Basin Pinyon-Juniper Woodland	533	0.13
Sonora-Mojave Creosotebush-White Bursage Desert Scrub	429	0.10
Mojave Mid-Elevation Mixed Desert Scrub	328	0.08
Coleogyne ramosissima Shrubland Alliance	313	0.07
North American Warm Desert Riparian Systems	85	0.02
Southern Rocky Mountain Ponderosa Pine Woodland	79	0.02
Barren	71	0.02
North American Warm Desert Sparsely Vegetated Systems	70	0.02
Quercus turbinella Shrubland Alliance	62	0.01

Table 2 (continued). Vegetation types in Grand–Parashant National Monument that cover ≥ 1% of the park's area (LANDFIRE).

Vegetation type	Area (km^2)	Proportion of park area
Mogollon Chaparral	57	0.01
Inter-Mountain Basins Semi-Desert Grassland	45	0.01
Sonoran Mid-Elevation Desert Scrub	44	0.01

The majority of the vegetation in Joshua Tree is shrublands (Table 3). Dominant species include creosote bush, narrowleaf or interior goldenbush (*Ericameria linearifolia*), California or eastern Mojave buckwheat (*Eriogonum fasciculatum*), ocotillo (*Fourqueria splendens*), canotia (*Canotia holacantha*) (on limestone or granitic soils), jojoba (*Simmondsia chinensis*) (on rhyolitic soils), and white bursage.

Table 3. Vegetation types in Joshua Tree National Park that cover ≥ 1% of the park's area (LANDFIRE).

Vegetation type	Area (km^2)	Proportion of park area
Sonoran Mid-Elevation Desert Scrub	1063	0.33
Sonora-Mojave Creosotebush-White Bursage Desert Scrub	1031	0.32
North American Warm Desert Sparsely Vegetated Systems	342	0.11
Barren	207	0.06
Mojave Mid-Elevation Mixed Desert Scrub	206	0.06
North American Warm Desert Riparian Systems	148	0.05
Sonoran Paloverde-Mixed Cacti Desert Scrub	94	0.03
Introduced Upland Vegetation-Annual Grassland	88	0.03

About 85% of Manzanar and about half of Mojave National Preserve is covered by mixed desert scrub (Tables 4, 5). Dominant species include blackbrush (*Coleogyne ramosissima*), California or eastern Mojave buckwheat, Mormon tea or Nevada jointfir (*Ephedra nevadensis*), spiny hopsage (*Grayia spinosa*), wolfberry (*Lycium* spp.), spiny menodora (*Menodora spinescens*), beargrass (*Nolina* spp.), buckhorn cholla (*Opuntia acanthocarpa*), bladder sage (*Salazaria mexicana*), Parish's goldeneye (*Viguiera parishii*), Joshua tree (*Yucca brevifolia*), and Mojave yucca (*Yucca schidigera*). Playas with sparse iodinebush (*Allenrolfea occidentalis*), seepweed (*Suaeda* spp.), saltgrass (*Distichlis spicata*), saltbush (*Atriplex* spp.), and other halophytes also occur in Mojave National Preserve.

Shrubland dominated by creosote bush or white bursage is the most common vegetation type across Lake Mead. However, more than half of the park is covered by open water or is sparsely vegetated (Table 6).

Table 4. Vegetation types in Manzanar National Historical Site that cover ≥ 1% of the park's area (LANDFIRE).

Vegetation type	Area (km^2)	Proportion of park area
Mojave Mid-Elevation Mixed Desert Scrub	2.77	0.85
North American Warm Desert Riparian Systems	0.34	0.10
Developed-Upland Shrubland	0.09	0.03
Developed-Roads	0.03	0.01

Table 5. Vegetation types in Mojave National Preserve that cover ≥ 1% of the park's area (LANDFIRE).

Vegetation type	Area (km^2)	Proportion of park area
Mojave Mid-Elevation Mixed Desert Scrub	3628	0.56
North American Warm Desert Sparsely Vegetated Systems	853	0.13
Sonora-Mojave Creosotebush-White Bursage Desert Scrub	650	0.10
Barren	540	0.08
North American Warm Desert Riparian Systems	357	0.06
Introduced Upland Vegetation-Annual Grassland	246	0.04
Great Basin Pinyon-Juniper Woodland	113	0.02

Table 6. Vegetation types in Lake Mead National Recreation Area that cover ≥ 1% of the park's area (LANDFIRE).

Vegetation type	Area (km^2)	Proportion of park area
Sonora-Mojave Creosotebush-White Bursage Desert Scrub	2419	0.46
Open Water	695	0.13
North American Warm Desert Sparsely Vegetated Systems	691	0.13
Barren	230	0.04
Mojave Mid-Elevation Mixed Desert Scrub	226	0.04
Coleogyne ramosissima Shrubland Alliance	190	0.04
Inter-Mountain Basins Big Sagebrush Shrubland	177	0.03
North American Warm Desert Riparian Systems	132	0.03
Introduced Upland Vegetation-Annual Grassland	102	0.02
Mogollon Chaparral	76	0.01
Sonoran Mid-Elevation Desert Scrub	63	0.01
Introduced Riparian Vegetation	39	0.01
Colorado Plateau Pinyon-Juniper Woodland	30	0.01
Inter-Mountain Basins Mixed Salt Desert Scrub	29	0.01
Sonoran Paloverde-Mixed Cacti Desert Scrub	26	0.01

2.B.2. Resource descriptions

High-priority natural resources across the Mojave Desert Network include elements of the three components of biological diversity: structure, composition, and function. In general, the parks aim to maintain native land cover with minimal fragmentation or other disturbances by authorized and unauthorized human activities. Authorized activities include road building, air traffic, urbanization, and some recreational uses. Park personnel noted that expansion of non-native invasive plant species, which not only change vegetation composition but can change fire dynamics, and high abundance of feral burros widely are potential stressors to high-priority natural resources. The parks also are concerned about direct and indirect effects on natural resources of human population growth in and around Los Angeles, California; Las Vegas, Nevada; and Phoenix, Arizona. Increases in human populations are likely to increase the number of visitors to the parks. Visitors create opportunities for outreach and public engagement, but can place additional stress on native species and ecological functions and increase the probability of dispersal of non-native invasive species. Growth of human populations across the southwestern United States also increases demands for the region's limited surface water and groundwater. Moreover, increases in human populations increase night lighting and levels of anthropogenic sound, which the parks consider undesirable.

Desert bighorn sheep and desert tortoise are perhaps the most iconic species within the Mojave Desert Network and the greatest focus of management at the species level. Human appropriation of groundwater and changes in fire dynamics are considered threats to native species resources by personnel across the Mojave Desert Network. Park personnel also noted diseases that affect wild animals, from pneumonia in bighorn sheep to the potential emergence of white-nose syndrome in bats, as potential stressors to native species. All of the Mojave Desert Network parks aim to maintain or increase the status of species that are listed or candidates for listing under the U.S. Endangered Species Act.

At the level of individual parks, Death Valley's 2002 General Management Plan noted that the park was a class II area under the Clean Air Act, which would allow increases in some air pollutants, and that NPS intended to seek class I designation for the park (i.e., smaller allowable increases in sulfur dioxide and particulates).

The Manager's 2013 Annual Report for Grand Canyon–Parashant identified eight major known or potential stressors to the park's natural resources: invasive species, especially bromes; uncharacteristic fire; loss of habitat for threatened and endangered species; potential emergence of white nose syndrome in the park's bats; climate change; population growth in St. George, Utah and Mesquite, Nevada; increases in illegal use of off-highway vehicles, and increases in air traffic.

The 2011 Joshua Tree Foundation Statement listed six fundamental resources and values related primarily to natural resources: Joshua trees, oases and other riparian areas, habitat for the desert tortoise, connectivity of land cover, overall biological diversity, and groundwater.

Manzanar's purpose is to protect and interpret resources associated with the relocation of Japanese-Americans during World War II. Therefore, some non-native invasive plants that otherwise might be considered undesirable are maintained because of their historical context. Manzanar plans to

maintain an irrigated orchard that predated the relocation center and plant and rock gardens that were established by individuals who were relocated.

Long-term goals related to preservation of park resources that were described in Lake Mead's 2001–2005 strategic plan included containment of non-native plants and feral burros and increasing the status of species listed under the U.S. Endangered Species Act. The strategic plan listed a mission goal of basing management decisions about resources and visitors on "adequate scholarly and scientific information," and indicated that information on ranges of variation in resource status would be quite useful. Resource-management goals highlighted in the 1987 Land Protection Plan for Lake Mead included improvement of air and water quality, maintenance and, in some cases, restoration of natural ecosystems and native species, control of non-native species, and understanding and management of off-road vehicle use and other visitor activities.

Mojave National Preserve's 2013 foundation document characterized Joshua tree, desert tortoise, and desert bighorn sheep as iconic species of the Mojave Desert and fundamental resources. The foundation document also attributed value to stable populations of the full range of native species characteristic of the eastern Mojave Desert and to knowledge of environmental changes and potential adaptations of native systems and species to fire, climate change and livestock grazing.

We compiled approved records from NPSpecies for species classified as present in the park, not classified as vagrants, and for which there was at least one observation or voucher.

As of February 2015, NPSpecies indicated that two species listed as endangered under the US Endangered Species Act, Devils Hole pupfish (*Cyprinodon diabolis*) and Eureka Valley dune grass (*Swallenia alexandrae*), occur in Death Valley. Devils Hole pupfish was classified as a management priority. NPSpecies also classified hoary bat (*Lasiurus cinereus*), a species of undefined status; desert tortoise (*Gopherus agassizii*), a species listed as threatened; and Panamint alligator lizard (*Elgaria panamintina*), the status of which is under review, as present in Death Valley and as management priorities. Seven additional species of concern were classified as management priorities: Californian myotis (*Myotis californicus*), fringed myotis (*Myotis thysanodes*), pallid bat (*Antrozous pallidus*), silver-haired bat (*Lasionycteris noctivagans*), western pipistrelle (*Pipistrellus hesperus*), Willow Flycatcher (*Empidonax traillii*), and common chuckwalla (*Sauromalus ater*). Management priorities also included two non-native species, American bullfrog (*Lithobates catesbeianus*) and southern alligator lizard (*Elgaria multicarinata*), and 11 native species: Mexican free-tailed bat (*Tadaria brasiliensis*), Townsend's big-eared bat (*Corynorhinus townsendii*), western small-footed myotis (*Myotis ciliolabrum*), Bell's Vireo (*Vireo bellii*), Least Bittern (*Ixobrychus exilis*), Gilbert's skink (*Plestidon gilberti*), Mojave fringe-toed lizard (*Uma scoparia*), King's eyelash grass (*Blepharidachne kingii*), red grama (*Bouteloua trifida*), Rothrock's penstemon (*Keckiella rothrockii rothrockii*), and sulfur flower buckwheat (*Eriogonum umbellatum*). A further 71 species of concern were listed as present in Death Valley (Table 7).

Table 7. Species of concern that were not designated as management priorities, and which NPSpecies indicated were present in each park, not classified as vagrants, and for which there was at least one observation or voucher; indicated by 1s in the table.

Taxonomic group	Species	Death Valley	Grand Canyon-Parashant	Joshua Tree	Lake Mead	Mojave National Preserve
Mammals	American badger (*Taxidea taxus*)	1	–	1	1	1
	bighorn sheep (*Ovis canadensis*)	1	1	–	1	–
	California leaf-nosed bat (*Macrotus californicus*)	–	–	1	–	–
	coyote (*Canis latrans*)	1	1	–	1	1
	ringtail (*Bassariscus astutus*)	1	–	1	1	–
	San Diego pocket mouse (*Chaetodipus fallax*)	–	–	1	–	1
Birds	American Bittern (*Botaurus lentiginosus*)	1	–	1	1	–
	American White Pelican (*Pelecanus erythrorhynchos*)	1	–	–	1	–
	Band-tailed Pigeon (*Patagioenas fasciata*)	–	1	–	–	–
	Bank Swallow (*Riparia riparia*)	1	–	–	1	–
	Black Swift (*Cypseloides niger*)	1	–	–	–	–
	Black Tern (*Chlidonias niger*)	1	–	–	1	–
	Black-crowned Night Heron (*Nycticorax nycticorax*)	1	1	1	1	–
	Brewer's Sparrow (*Spizella breweri*)	1	1	1	1	–
	Brown Creeper (*Certhia americana*)	1	1	–	1	–
	Brown-crested Flycatcher (*Myiarchus tyrannulus*)	1	–	–	–	–
	Cactus Wren (*Campylorhynchus brunneicapillus*)	1	1	1	1	–
	Common Loon (*Gavia immer*)	1	–	1	1	–
	Common Tern (*Sterna hirundo*)	–	–	–	1	–
	Cooper's Hawk (*Accipiter cooperii*)	1	1	1	1	–
	Crissal Thrasher (*Toxostoma crissale*)	–	–	–	1	–
	Double-crested Cormorant (*Phalacrocorax auritus*)	1	–	–	1	–
	Downy Woodpecker (*Picoides pubescens*)	–	1	–	–	–

Table 7 (continued). Species of concern that were not designated as management priorities, and which NPSpecies indicated were present in each park, not classified as vagrants, and for which there was at least one observation or voucher; indicated by 1s in the table.

Taxonomic group	Species	Death Valley	Grand Canyon-Parashant	Joshua Tree	Lake Mead	Mojave National Preserve
Birds (continued)	Golden Eagle (*Aquila chrysaetos*)	1	1	–	1	–
	Grasshopper Sparrow (*Ammodramus savannarum*)	–	–	–	1	–
	Gray Vireo (*Vireo vicinior*)	1	1	1	1	–
	Great Blue Heron (*Ardea herodias*)	1	1	1	1	–
	Le Conte's Thrasher (*Toxostoma lecontei*)	1	–	–	1	1
	Lesser Nighthawk (*Chordeiles acutipennis*)	1	–	1	1	–
	Lesser Scaup (*Aythya affinis*)	1	–	1	1	–
	Lincoln's Sparrow (*Melospiza lincolnii*)	1	1	1	1	–
	Loggerhead Shrike (*Lanius ludovicianus*)	1	1	1	1	–
	Long-eared Owl (*Asio otus*)	1	–	1	1	–
	MacGillivray's Warbler (*Oporornis tolmiei*)	–	–	–	1	–
	Merlin (*Falco columbarius*)	1	–	1	1	–
	Northern Harrier (*Circus cyaneus*)	1	1	1	1	–
	Olive-sided Flycatcher (*Contopus cooperi*)	1	1	1	1	–
	Osprey (*Pandion haliaetus*)	1	–	1	1	–
	Peregrine Falcon (*Falco peregrinus*)	1	–	–	1	–
	Phainopepla (*Phainopepla nitens*)	1	1	1	1	–
	Prairie Falcon (*Falco mexicanus*)	1	–	1	1	–
	Purple Martin (*Progne subis*)	–	–	1	1	–
	Red Crossbill (*Loxia curvirostra*)	1	1	–	1	–
	Red-shouldered Hawk (*Buteo lineatus*)	1	–	–	–	–
	Rough-legged Hawk (*Buteo lagopus*)	–	–	1	1	–
	Rufous Hummingbird (*Selasphorus rufus*)	1	1	1	1	–
	Sagebrush Sparrow (*Amphispiza belli*)	–	1	1	1	–

Table 7 (continued). Species of concern that were not designated as management priorities, and which NPSpecies indicated were present in each park, not classified as vagrants, and for which there was at least one observation or voucher; indicated by 1s in the table.

Taxonomic group	Species	Death Valley	Grand Canyon-Parashant	Joshua Tree	Lake Mead	Mojave National Preserve
Birds (continued)	Sharp-shinned Hawk (*Accipiter striatus*)	1	1	1	1	–
	Short-eared Owl (*Asio flammeus*)	1	–	–	–	–
	Summer Tanager (*Piranga rubra*)	1	1	1	1	–
	Tree Swallow (*Tachycineta bicolor*)	1	–	1	1	–
	Turkey Vulture (*Cathartes aura*)	1	1	1	1	–
	Vermilion Flycatcher (*Pyrocephalus rubinus*)	1	–	–	1	–
	Western Bluebird (*Sialia mexicana*)	1	1	1	1	–
	Western Grebe (*Aechmophorus occidentalis*)	1	–	–	1	–
	White-faced Ibis (*Plegadis chihi*)	1	–	1	1	–
	Williamson's Sapsucker (*Sphyrapicus thyroideus*)	1	–	–	–	–
	Willow Flycatcher (*Empidonax traillii*)	–	–	–	1	–
	Wilson's Warbler (*Wilsonia pusilla*)	–	–	–	1	–
	Yellow-breasted Chat (*Icteria virens*)	1	1	1	1	–
Reptiles	common chuckwalla (*Sauromalus ater*)	–	1	1	1	1
	desert iguana (*Dipsosaurus dorsalis*)	1	–	1	1	1
	glossy snake (*Arizona elegans*)	–	–	–	1	1
	long-nosed leopard lizard (*Gambelia wislizenii*)	1	1	1	1	1
	ring-necked snake (*Diadophis punctatus*)	1	–	–	1	1
	sidewinder (*Crotalus cerastes*)	1	1	1	1	1
	speckled rattlesnake (*Crotalus mitchellii*)	1	–	1	1	1
	western groundsnake (*Sonora semiannulata*)	1	–	–	1	1
Plants	Beaver dam breadroot (*Pediomelum castoreum*)	–	–	–	1	–

Table 7 (continued). Species of concern that were not designated as management priorities, and which NPSpecies indicated were present in each park, not classified as vagrants, and for which there was at least one observation or voucher; indicated by 1s in the table.

Taxonomic group	Species	Death Valley	Grand Canyon-Parashant	Joshua Tree	Lake Mead	Mojave National Preserve
Plants (continued)	black milkvetch (*Astragalus funereus*)	1	–	–	–	–
	bush seepweed (*Suaeda nigra*)	1	–	–	–	–
	common phacelia (*Phacelia distans*)	–	–	1	1	–
	curve-podded Mojave milk-vetch (*Astragalus mohavensis hemigyrus*)	–	1	–	–	–
	Death Valley beardtongue (*Penstemon fruticiformis amargosae*)	1	–	–	–	–
	Death Valley sandpaper plant (*Petalonyx thurberi gilmanii*)	1	–	–	–	–
	desert paintbrush (*Castilleja applegatei martinii*)	1	–	–	–	–
	Gilman's milkvetch (*Astragalus gilmanii*)	1	–	–	–	–
	Grand Canyon rose (*Rosa stellata abyssa*)	–	1	–	1	–
	Hanaupah rock daisy (*Perityle villosa*)	1	–	–	–	–
	July gold (*Dedeckera eurekensis*)	1	–	–	–	–
	Las Vegas bearpoppy (*Arctomecon californica*)	–	–	–	1	–
	Little San Bernardino Mountains gilia (*Gilia maculata*)	–	–	1	–	–
	Mojave thistle (*Cirsium mohavense*)	1	–	–	1	–
	Mt. Trumbull beardtongue (*Penstemon distans*)	–	1	–	–	–
	napkinring buckwheat (*Eriogonum intrafractum*)	1	–	–	–	–
	Panamint daisy (*Enceliopsis covillei*)	1	–	–	–	–
	Panamint liveforever (*Dudleya saxosa saxosa*)	1	–	–	–	–
	Panamint Mountains buckwheat (*Eriogonum microthecum panamintense*)	1	–	–	–	–

Table 7 (continued). Species of concern that were not designated as management priorities, and which NPSpecies indicated were present in each park, not classified as vagrants, and for which there was at least one observation or voucher; indicated by 1s in the table.

Taxonomic group	Species	Death Valley	Grand Canyon-Parashant	Joshua Tree	Lake Mead	Mojave National Preserve
Plants (continued)	Panamint Mountains lupine (*Lupinus magnificus magnificus*)	1	–	–	–	–
	Parish phacelia (*Phacelia parishii*)	–	–	–	1	–
	ringstem (*Anulocaulis leiosolenus*)	–	–	–	1	–
	Rusby's desert mallow (*Sphaeralcea rusbyi eremicola*)	1	–	–	–	–
	Stephens' beardtongue (*Penstemon stephensii*)	–	–	–	–	1
	sticky buckwheat (*Eriogonum viscidulum*)	–	–	–	1	–
	threecorner milkvetch (*Astragalus geyeri triquetrus*)	–	–	–	1	–
	tufted lovegrass (*Eragrostis pectinacea*)	–	–	–	1	–
	two-tone rosy beardtongue (*Penstemon bicolor roseus*)	–	–	–	1	–
	white bear poppy (*Arctomecon merriamii*)	1	–	–	–	–
	wild heliotrope (*Phacelia distans*)	1	–	–	–	1
	Wildrose Canyon buckwheat (*Eriogonum eremicola*)	1	–	–	–	–
	Yuma ditaxis (*Ditaxis serrata*)	–	–	1	–	–

According to NPSpecies, one endangered species, California condor (*Gymnogyps californianus*); one threatened species, desert tortoise; and one candidate species, relict leopard frog (*Lithobates onca*); all management priorities, occur in Grand Canyon–Parashant. The delisted Bald Eagle (*Haliaeetus leucocephalus*) was listed as a management priority, as were five other taxa— pallid bat, Ferruginous Hawk (*Buteo regalis*), Peregrine Falcon (*Falco peregrinus*), Prairie Falcon (*Falco mexicanus*), and Willow Flycatcher. An additional 30 species of concern, all native, are believed to occur in the park (Table 7).

NPSpecies suggested that the endangered triple-ribbed milkvetch (*Astragalus tricarinatus*), a management priority; two species that are threatened and management priorities [desert tortoise and Parish's daisy (*Erigeron parishii*)]; and Yellow-billed Cuckoo (*Coccyzus americanus*), a species proposed for listing as threatened, occur in Joshua Tree. Additional species of concern that NPSpecies tagged as management priorities in Joshua Tree were big brown bat (*Eptesicus fuscus*), Californian myotis, fringed myotis, long-legged myotis (*Myotis volans*), pallid bat, spotted bat

(*Euderma maculatum*), western pipistrelle, coyote (*Canis latrans*), American White Pelican (*Pelecanus erythrorhynchos*), Golden Eagle (*Aquila chrysaetos*), Le Conte's Thrasher (*Toxostoma lecontei*), Willow Flycatcher, alkali mariposa lily (*Calochortus striatus*), and Robison's monardella (*Monardella robisonii*). Mountain Quail (*Oreortyx pictus*), Las Animas colubrina (*Colubrina californica*), pinyon rock-cress (*Arabis dispar*), and San Jacinto prickly-phlox (*Leptodactylon jaegeri*) also were listed as management priorities. Forty-three additional species of concern were listed as occurring in the park (Table 7).

NPSpecies indicated that Lake Mead is occupied by at least three endangered species: California condor, bonytail (*Gila elegans*), and razorback sucker (*Xyrauchen texanus*). California condor and razorback sucker were flagged as management priorities. Desert tortoise occurs in the park and is a management priority. Bald Eagle, Yellow-billed Cuckoo, and relict leopard frog also are believed to occur in Lake Mead; relict leopard frog is a management priority. Hoary bat also is a management priority. Additional species of concern that were classified by NPSpecies as present in Lake Mead and as management priorities were Allen's big-eared bat (*Idionycteris phyllotis*), big brown bat (*Eptesicus fuscus*), Californian leaf-nosed bat (*Macrotus californicus*), Californian myotis, pallid bat (*Antrozous pallidus*), silver-haired bat (*Lasionycteris noctivagans*), western pipistrelle (*Pipistrellus hesperus*), western red bat (*Lasiurus blossevillii*), and Yuma myotis (*Myotis yumanensis*). NPSpecies listed another 70 species of concern within the park (Table 7).

According to NPSpecies, four species of concern occur in Manzanar: Loggerhead Shrike (*Lanius ludovicianus*), Turkey Vulture (*Cathartes aura*), glossy snake (*Arizona elegans*), and long-nosed leopard lizard (*Gambelia wislizenii*).

NPSpecies suggested that Mojave National Preserve is occupied by 19 species that are management priorities: the threatened desert tortoise, dark-nosed small-footed myotis (*Myotis melanorhinus*), hoary bat, Townsend's big-eared bat (*Corynorhinus townsendii*), North American porcupine (*Erethizon dorsata*), black-tailed jackrabbit (*Lepus californicus*), Mojave tui chub (*Siphateles bicolor mohavensis*), northern Pacific treefrog (*Pseudacris regilla*), California kingsnake (*Lampropeltis californiae*), gila monster (*Heloderma suspectum*), western threadsnake (*Rena humilis*), and eight species of concern: bighorn sheep (*Ovis canadensis*), big brown bat, California myotis, fringed myotis (*Myotis thysanodes*), long-eared myotis (*Myotis evotis*), long-legged myotis (*Myotis volans*), pallid bat, and western pipistrelle. Fourteen additional species of concern were listed as present by NPSpecies (Table 7).

2.B.3. Resource management priorities

During our scoping process, we discovered considerable overlap in resource-management priorities among managers of the Mojave Desert Network parks. For example, among the highest network-level priorities were assessing the population structure, status, and trend of desert bighorn sheep and the connectivity of habitat for desert bighorn sheep. A second high priority was assessing potential shifts in the distribution and connectivity of populations of non-native invasive plants.

As noted above, across the network, managers expressed interest or concern in how human population growth and changes in land use across the southwestern United States might affect natural

resources in the parks. Examples of potential changes in land use included the density of roads and the proportion of different classes of roads, development of renewable-energy infrastructure, and increases in anthropogenic light and sound. Managers requested information about projected changes in climate means, variability, and extremes (e.g., drought and flood) and how natural resources may respond to such changes. In particular, there was considerable interest in responses of wildlife to changes in the availability and quality of natural and anthropogenic sources of water. Additionally, NPS personnel asked how measures of air quality, such as nitrogen deposition, ozone, and particulates, might respond to climate change, and how topography affects air flow and quality. Regional changes in fire dynamics were a high priority across the network. Managers wished to examine how fire size and frequency might respond to changes in the timing of precipitation and to spatially extensive changes in vegetation.

Also as noted above, a major network-level priority was assessing potential shifts in the distribution and connectivity of populations of non-native invasive plants. NPS staff asked about the current distributions of non-native bromes (e.g., red brome and cheatgrass) and other species for which sufficient data exist to model occurrence patterns. Furthermore, they requested guidance on the design of monitoring plans for non-native invasive plants that assess responses of plants to changes in climate, land use, and major events.

Elements of the biology of desert bighorn sheep that were emphasized by NPS personnel during the scoping process included demographic and genetic measures of population or metapopulation dynamics and the current and potential future amount, configuration, and connectivity of habitat. Furthermore, managers asked how other wild animals, from big game to small nongame species, might respond to changes in the provision of guzzlers or water impoundments. NPS personnel requested that if the assessment identified gaps in information, then suggestions be made for the design of studies that might fill the gaps.

There was considerable interest across the Mojave Desert Network in current and projected connectivity of habitat for desert tortoise. There also was interest in the effects of human infrastructure on connectivity of desert tortoise habitat and on predation of desert tortoise.

Managers asked how bats in the Mojave Desert may respond to closure of mines, potential expansion of white-nose syndrome, and energy development (e.g., wind turbines). Moreover, there was widespread interest in status and trend of feral burros.

At the level of individual parks, personnel in Death Valley identified their lack of access to major internet search engines (e.g., Web of Knowledge) and online journals as a substantial impediment to their ability to apply the best science available to their stewardship. Among the park's priorities for assessment were potential responses of wildlife to presence or removal of big-game guzzlers, methods for data synthesis and archiving, and the distribution of human-created infrastructure such as fences.

During the scoping process, staff at Grand Canyon–Parashant inquired about the current ability of meteorologists to project probabilities and characteristics of major storms. Among their high

priorities for assessment were the status and trends in abundance of mule deer (*Odocoileus hemionus*) and potential responses of that species to changes in vegetation structure.

A high priority of Joshua Tree staff was options for fire management in different elevation or vegetation zones and covariates of density and distribution of recreational or social trails, such as distance from roads. Park personnel regarded air pollution, fragmentation of native land cover adjacent to the park, lack of baseline data on the park's natural resources, and effects of recreation on natural resources as potential threats and priorities for management.

Staff at Lake Mead expressed considerable interest in the potential for high-resolution mapping of individual species of plants, including non-native plants in uplands. They also asked about the potential effectiveness of mapping covariates of expansion of non-native plants as a surrogate for mapping the species. Lake Mead's strategic plan noted increases in damage to natural resources from unauthorized use of off-road vehicles.

Staff at Manzanar asked about the feasibility of restoration of native vegetation given much vegetation on site is non-native and the potential effects of removal of vegetation on migratory birds.

Mojave National Preserve asked how one might assess the response of vegetation, especially near seeps, springs, and riparian areas, to removal of cattle in 2001 and to removal of burros around the same time. The preserve's foundation document expressed concern about the effects on natural resources of transportation corridors, new development both within and outside the preserve, development of solar and wind energy, and water limitations resulting from climate change and groundwater pumping.

2.C. Resource stewardship

2.C.1. Management directives and planning guidance

The 2008 vital signs report listed the four dominant stressors across the Mojave Desert Network as invasive species, alteration of water quantity, land-use change and development, and reduction in air quality. Moderate stressors were altered disturbance regimes, recreation and visitation, climate change, reduction in water quality, and soil alteration. Livestock grazing, resource extraction, increases in populations of native wildlife species, habitat fragmentation, nutrient availability, and disease were noted as low stressors.

We found virtually no quantitative targets (i.e., desired conditions, measurement indicators, or short-term or long-term management target values) for the status of fundamental or other high-priority natural resources.

2.C.2. Status of supporting science

The Vital Signs Monitoring Program identified 20 vital signs for the Mojave Desert Network that are believed to represent the condition of park resources, effects of stressors, and elements on which humans place a high value: ozone, wet and dry deposition, visibility and particulate matter, basic meteorology, soil erosion and deposition, soil disturbance, soil chemistry and nutrient cycling, soil hydrologic function, changes in upland vegetation, biological soil crusts, changes in riparian vegetation, riparian bird communities, invasive/exotic plants, fire and fuel dynamics, groundwater

dynamics and chemistry, surface water dynamics of groundwater and springs, surface water chemistry of groundwater and springs, surface water dynamics of streams and lakes, surface water chemistry of streams and lakes, landscape dynamics, small mammal communities, reptile communities, and at-risk populations (Chung-MacCoubrey et al. 2008). Preliminary monitoring objectives for some of those vital signs have been identified. Pilot studies for integrated upland monitoring have been developed for all of the parks addressed in this document except Manzanar. A monitoring protocol for streams and lakes has been developed (Caudill et al. 2012), and pilot data were collected in 2009 and 2010. Additionally, two large springs have been included in pilot studies. To the best of our knowledge, no science-based reference conditions and values have been identified to date.

Death Valley has evaluated the park's natural quality, or the extent to which the park is "substantially free from the effects of modern civilization" (Tricker et al. 2012). Measures of natural quality were plant and animal species and communities (land cover type [whether native], presence of non-native species of plants and animals [burros]), physical resources (annual average concentration of ozone [ppm]), total annual wet deposition of nitrate and ammonium, presence of mining sites, alteration of springs, anthropogenic sky brightness), and biophysical processes (livestock grazing, presence of guzzlers; the park hoped to address departure of fire regime from natural and climate change, but data on these elements were insufficient) (Tricker et al. 2012).

The Grand Canyon–Parashant Manager's annual report for fiscal year 2013. Grand Canyon – Parashant National Monument indicated that natural resources are in good condition and that the trend is stable, but the data or other information on which the status and trend estimates were based was unclear.

Chapter 3. Study Scoping and Design

3.A. Preliminary scoping

The lead investigators met with representatives from Death Valley National Park, Grand Canyon–Parashant National Monument, Joshua Tree National Park, Lake Mead National Recreation Area, Manzanar National Historic Site, and Mojave National Preserve and from the Mojave Desert Network's Inventory and Monitoring Program to launch the NRCA process. We discussed region-level and park-level objectives for the NRCA and priorities for assessment. We also discussed alternative ways to organize the NRCA and to address high-priority resources for which relatively few data exist. For example, we agreed that where data are insufficient to allow strong inference to status and trend, the investigators would aim to outline data requirements and analytical methods that might allow such inference. Additionally, we reviewed the anticipated phases of the NRCA and associated timetables and responsibilities.

Our initial meeting suggested a number of resources and stressors for which assessment was a high priority for a majority of the park units. About 50 resources and stressors identified during the meeting formed the basis for more detailed discussions during site visits to individual park units.

The investigators visited each of the six park units to discuss assessment priorities, data availability, and data sources in greater depth. These interactions allowed for identification of resources and stressors most amenable to assessment at the regional level (i.e., across much of the Mojave Desert and adjacent lands) versus the park-unit level. During each site visit, we learned about potential sources of information on high-priority resources that were available from the park units, Mojave Desert Network, Inventory and Monitoring Program, other NPS offices, or other investigators or institutions. To the extent possible, we gathered information while visiting park units. Where it was not possible to gather information during site visits, we identified high-quality sources of data and obtained those data in subsequent weeks and months.

The investigators worked with the Mojave Desert Network to draft a preliminary outline for a region-level assessment. The Mojave Desert Network indicated that management might best be informed by a focus on assessment of processes rather than individual species and highlighted water availability, fire dynamics, and distribution and dynamics of non-native invasive plants as priorities. At the species level, the Mojave Desert Network requested assessments of connectivity and quality of habitat for desert bighorn sheep (*Ovis canadensis nelsoni*): connectivity and quality of habitat (Death Valley National Park, Joshua Tree National Park, Lake Mead National Recreation Area, Mojave National Preserve, and network-wide); habitat quality for predators of desert tortoise (*Gopherus agassizii*); potential responses of bats to changes in land use and disease; feral burros; and mule deer (*Odocoileus hemionus*) in Grand Canyon–Parashant.

A theme throughout our scoping discussions with NPS personnel, and therefore throughout this NRCA, is the extent to which remote sensing can provide reliable information on the condition and trend of resources that is not feasible to obtain in the field. For example, monitoring can inform efforts to prevent colonization of a given location or region by non-native invasive species of plants. Monitoring also can inform eradication efforts. Yet eradication is practical only during the early

stages of invasion, when there are relatively few individuals in a relatively small area. Therefore, monitoring of invasive plants is most likely to be useful to NPS when implemented over extensive areas. However, field surveys across hundreds to thousands of square kilometers are not feasible. Accordingly, NPS asked us to assess whether remote sensing could be used to identify the current locations of invasive plants, to assess the probability of future invasion, or to document temporal trends in the distribution or abundance of invasive plants.

As an additional illustration, remotely sensed vegetation indexes such as the normalized difference vegetation index (NDVI) are potential measures for monitoring the diet quality of ungulates at high resolution over long time periods and large spatial extents. Therefore, we validated the relation between NDVI and diet quality for desert bighorn sheep.

3.B. Study design

On the basis of our scoping and discussions with Mojave Desert Network personnel, we agreed to emphasize the following resources and stressors: climate change, land use, air quality, fire dynamics, the distribution and dynamics of non-native plants, habitat quality for Mojave desert tortoise, connectivity and adaptive capacity of desert bighorn sheep, water availability for native wildlife, feral burros, bats, and mule deer in Grand Canyon–Parashant. Where data allowed, we reported at the level of individual parks and at the ecoregion level.

Chapter 4. Climate Change

The climate of the Mojave Desert and other deserts across the southwestern United States (Arizona, California, Colorado, Nevada, New Mexico, and Utah) is highly variable, which complicates identification of current climate and trends in climate. Nevertheless, some changes in climate already are being observed, and information from both weather stations and climate models are sufficient to indicate changes that have a relatively high likelihood of occurring during the twenty-first century. The responses of different species, ecological processes, and non-climate stressors to ongoing and projected changes in climate will not be uniform; some responses likely will be considered negative, whereas others may be considered positive.

Native species in the Mojave are adapted to a variable and often severe climate. Compared to other ecosystems in the United States, there is relatively little human infrastructure to impede movements of animals and plants. However, the speed of contemporary changes in climate is fairly high, and, over evolutionary time, species may not have been exposed to some of the projected absolute changes in climate (e.g., particular temperature extremes). We recommend that future evaluations of climate change by NPS and its partners be framed in terms of change rather than vulnerability.

Observed and projected changes in climate across the Southwest were assessed in detail in 2013 by more than 100 scientists in a manner roughly analogous to that used by the Intergovernmental Panel on Climate Change (Garfin et al. 2013). This assessment of climate change is available online at no cost. Among the many resources included in the assessment are descriptions of present and projected future mean and extreme climate (and explanations of the methods and assumptions on which climate summaries or projections were based); potential effects of climate change on ecosystems, economic sectors, and human communities; and options for characterizing and communicating uncertainty.

The group of scientists agreed that there is considerable scientific evidence that the region's climate is changing and will continue to change. The changes observed to date are consistent with anthropogenic emissions of greenhouse gases. As described below, there is medium-high to high scientific confidence in many observed and projected changes in climate across the Southwest. Medium-high confidence reflects moderate consensus and evidence, such as multiple sources of information and generally consistent results. High confidence reflects high consensus and strong evidence, such as established theory, numerous sources of information, and well documented and accepted methods. Although the magnitude of some future changes in climate remains uncertain, the types of changes that have been observed and projected allow for development of scenarios that can inform management planning and actions by NPS and other resource management jurisdictions across the Mojave Desert.

There is high confidence that average annual temperature has increased and will continue to increase. Average annual temperatures across the Southwest increased by 1.6 ± 0.5˚F (mean ± SD) from 1901–2010. Climate-model projections suggest that average annual temperatures will increase by another 1–4°F from 2021–2050, 1–6°F from 2041–2070, and 2–9°F from 2070–2099.

There is high confidence that temperatures have increased and will continue to increase in all seasons, with greatest increases in summer and autumn. Relative to 1971–2000, climate models project mean increases in summer temperatures of 3.5°F from 2021–2050 and mean increases in summer temperatures of 9°F from 2070–2099.

There is high confidence that the length of the freeze-free season has increased and will continue to increase. The duration of the freeze-free season in the Southwest was about 7% (17 days) longer from 2001–2010 than from 1901–2000. Model projections that were based on a high-emissions scenario suggested that from 2041–2070, the duration of the frost-free season across most of the Southwest will increase by a minimum of another 17 days.

There is high confidence that the number of heat waves has increased and will continue to increase. A greater number of heat waves occurred over the Southwest from 2001–2010 than from 1901–2000. Model projections suggest that the number and intensity of heat waves will continue to increase.

There is medium-high confidence that the number of cold snaps is decreasing and will continue to decrease. The number of cold waves that occurred over the Southwest from 2001–2010 was less than the average from 1901–2000. Climate models project that the number of cold snaps, although not necessarily their intensity, will continue to decrease through 2100. Annual and decadal variability in the number and intensity of cold snaps likely will be high.

There is medium-high confidence that spring precipitation across the Southwest will decrease. The great majority of climate models project that spring precipitation will decrease by 2050. The median projected decrease by 2070–2099, depending on the emissions scenario, is 9–29%.

There is high confidence that mountain snowpack will decrease, and this change is occurring in some parts of the Southwest. Model projections suggest that late winter and spring snowpack will decrease across the mountains of the Southwest throughout the twenty-first century, primarily because temperature is decreasing rather than because precipitation is decreasing. There also is high confidence that snowmelt and peak streamflow are occurring, and will continue to occur, earlier in the year. There is medium-high confidence that climate change will further limit water availability across much of the Southwest.

There is medium-high confidence that drought severity increased from 1901–2010 and will continue to increase. However, the most severe and sustained droughts during the past century were neither as severe nor as long as some droughts that occurred during the preceding 2000 years. Decreases in precipitation during observed droughts have been exacerbated by anomalously warm summer temperatures. Climate models project that summer temperatures will continue to increase, which in turn will continue to exacerbate future droughts.

Chapter 5. Land Use

5.A. Rationale for and information provided by the assessment

5.A.1. The human footprint

Land use and its effects of land cover both within and outside of parks affect natural resources—both ecological processes and species—within parks. Data on land use are highly relevant to analyses of the human footprint, which generally is defined as the effects on the environment or natural resources of human presence and actions (Sanderson et al. 2002). Human activity has both direct and indirect effects on natural resources. For instance, human settlements, recreation, and other activities may support synanthropic species, that is, native species that exploit human activity. Full synanthropes (e.g., House Sparrow [*Passer domesticus*]) require human settlements for survival, whereas casual synanthropes (e.g., House Finch [*Carpodacus mexicanus*]) exploit but do not depend on human development. Tangential synanthropes (e.g., Bullock's oriole [*Icterus bullockii*]) are those that occasionally exploit human development. Syanthropic species may be considered desirable or undesirable depending on the species and context. For example, Common Raven (*Corvus corax*), which typically is classified as synanthropic because its density is relatively high in areas with high-intensity land use (Kristan and Boarman 2003, 2007), often is considered to be undesirable in natural areas in the Mojave Desert because it is a predator of Mojave desert tortoise (*Gopherus agassizii*). Human activity also can increase the probability of colonization by non-native invasive plants, which can have substantial effects on vegetation composition and, in the Mojave Desert and adjacent ecosystems, fire frequency and size.

The human footprint can be modeled with respect to physical effects or ecological effects. The physical effect area is the actual area covered by human-created features such as roads, well pads, houses, or power lines. The physical effect area can be calculated as the proportion of a pixel covered by the features (Theobald 2013). The ecological effect area is the area within which species or ecological processes are affected by human-created features. The ecological effect area can be modeled either independent of species or processes or with respect to a given species or process (e.g., a species of mammal that is known to avoid roads). As a measure of potential threats to the status of natural resources, we estimated the areas in which habitat quality for a large suite of vertebrate species is affected by anthropogenic features and land use.

The human footprint can be calculated at multiple points in time to examine trends, and it can be used in scenario assessments to explore the potential environmental effects of changes in management. Estimates of the human footprint typically include multiple measures of land cover or land use, human access, human settlement, and energy infrastructure (Trombulak et al. 2010). Different human footprint models have included, for example, housing density, traffic volume, distance to or density of roads, and presence of power lines.

Spatial data included in a human-footprint model may be points (e.g., landfills or picnic areas), linear features (e.g., roads or irrigation canals), or polygons. Inputs commonly are derived from remotely sensed data, whether satellite images or aerial photographs, and often from spatial data layers that are public. For example, land cover might be delineated on the basis of data from the Landscape Fire and

Resource Management Planning Tools Project (LANDFIRE; www.landfire.gov), whereas primary roads (typically interstate highways with four or more lanes), secondary roads (typically main arteries such as U.S. highways without a median, state highways, or county highways), railroads, and power lines might be derived from **T**opologically **I**ntegrated **G**eographic **E**ncoding and **R**eferencing (TIGER)/Line data from the U.S. Census Bureau. Roads can affect dust production; soil density, temperature, and water content; and patterns of runoff and sedimentation (Trombulak and Frissell 2000).

Estimates of land use that provide the foundation for human-footprint models typically are derived from publicly available spatial data, but common sources of publicly available data may not accurately represent actual extents or intensities of land use (Leu et al. 2008). Because development of a human footprint model requires that the data on land use be comparable across the assessment area, one cannot use diverse sources or resolutions of data to assess land-use change. As a result, it may be necessary to exclude data for a subset of the assessment area that have high resolution or accuracy from an assessment of land-use change over a more extensive area. However, the higher-resolution data can be used to evaluate the lower-resolution data and estimate potential sources of bias or uncertainty. Therefore, before assessing effect zones for vertebrates, we examined potential biases in the application of publicly available spatial data to human footprint models across the assessment area.

5.A.2. Anthropogenic light

Light from human activities can reduce humans' ability to observe stars, planets, and other features characteristic of the night sky. Anthropogenic light can disrupt the sleep, circadian rhythms, or phenology of plants and animals, including humans (Navara and Nelson 2007, Holker et al. 2010a). On the basis of sky-brightness measurements from international astronomical observatories, it has been estimated that sky brightness increased worldwide by an average of 3–6 % per year since the mid-1900s (Davis 2001, Schwarz 2003, Holker et al. 2010a). At the request of NPS, we assessed potential responses of natural resources in the Mojave Desert Network to anthropogenic light. The NPS has a mandate to protect the visibility of the night sky from anthropogenic sources of light (Duriscoe et al. 2007), and the NPS Night Sky Team has monitored and sky brightness since 2001.

5.A.3. Anthropogenic sound

Air traffic, energy development, visitors, and urbanization are increasing sound levels in the Mojave Desert Network. Anthropogenic sounds can reduce the condition and fitness of animals and, in extreme cases, cause temporary or permanent loss of hearing or shifts in hearing thresholds. Chronic and impulsive anthropogenic sounds also may cause increases in levels of stress hormones and changes in behaviors associated with movement, communication, and foraging. For example, anthropogenic sound may decrease the ability of mammals to recognize either conspecific sounds or sounds used to perceive the environment, find prey, or avoid predators (Ellison et al. 1987, George et al. 1989, Clark et al. 2009). At the request of NPS, we assessed potential responses of natural resources in the Mojave Desert Network to anthropogenic sound. The National Parks Overflights Act of 1987 mandated that the NPS assess responses of wildlife to sound.

5.B. Data and methods

5.B.1. Land use and land cover change

We used Hydrologic Unit Code (HUC) 8 watersheds to delineate the area within which we assessed land-use and land-cover change. We included in the assessment area all watersheds that either overlapped with, were adjacent to, or were between parks. The assessment area is sufficiently large to capture most of the effects of land uses and anthropogenic features located outside of Mojave Desert Network parks on ecological processes and species within parks.

We assessed land-cover change from 1992–2001 and 2001–2006 on the basis of the National Land Cover Database (NLCD; Appendix 1), which is considered to be one of the most definitive sources of information on land cover across the conterminous United States. Data from previous years are not available from the NLCD, and the 2011 product (released in spring 2014) was not available at the time we conducted the assessment. Because different methods were used by the NLCD to classify land-cover change during each time period, we could not generate (and the NLCD recommends that one not generate) a single estimate of land-cover change from 1992–2006.

We omitted the ice/snow class because this land-cover type is rare in the assessment area (<< 1% of the region). We reclassified the 2001–2006 land-cover classes to allow direct comparison with the 1992–2001 classes. We reclassified cultivated land and hay/pasture as agriculture; deciduous, evergreen, and mixed forest as forest; herbaceous and shrub/scrub as grassland/shrub; developed (low, medium, and high intensity) and urban-open space as urban; and emergent herbaceous and woody wetlands as wetland.

5.B.2. Accuracy of data on land use

We assessed the accuracy of publicly available spatial data on anthropogenic features (Appendix 1) by comparing the extents of those features in the public data to the extents estimated from hand-digitized aerial photographs (Bing Maps Aerial, included with ArcGIS 10.0 [ESRI, Redlands, California, 2013]). Outside of the parks, we randomly selected 25 test areas of 25 km^2 from a grid covering the assessment area. Eight of the 25 test areas contained no anthropogenic features. Therefore, we compared publicly available spatial data to hand-digitized data for 17 test areas (Figure 2). Within parks, we randomly placed a total of 15 additional test areas of 25 km^2, three in each of Death Valley, Grand Canyon–Parashant, Joshua Tree, Lake Mead, and Mojave National Preserve (Figure 3). For Manzanar, we restricted our analysis to the 3.3 km^2 area of the park.

We selected a test-area size of 25 km^2 because that extent was sufficient to capture variation in the distribution of roads and power lines but not large enough to make heads-up digitizing intractable. Heads-up digitizing refers to delineation by hand, and incorporation as shapefiles within a Geographic Information System (GIS), of spatial features recognizable on aerial photographs. Although the human eye may overlook features on aerial photographs, heads-up digitizing is the most accurate method, after digitizing features in the field, to evaluate publicly available data on anthropogenic features.

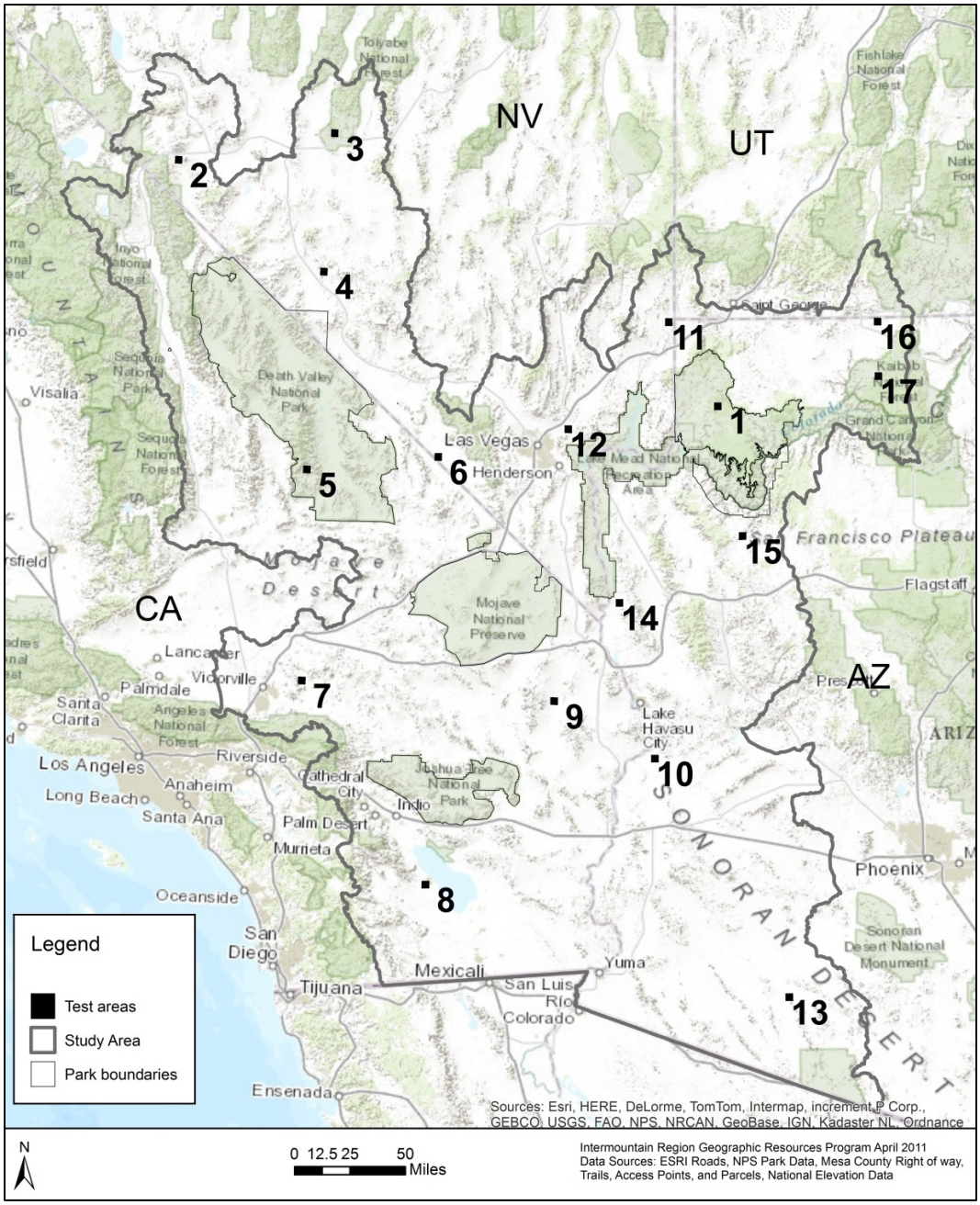


Figure 2. Locations of 17 test areas of 25 km^2 outside the Mojave Desert Network in which we compared publicly available data [Topologically Integrated Geographic Encoding and Referencing (TIGER) data and National Park Service data] on roads, all-terrain vehicle trails, and power lines to data digitized from aerial photographs.

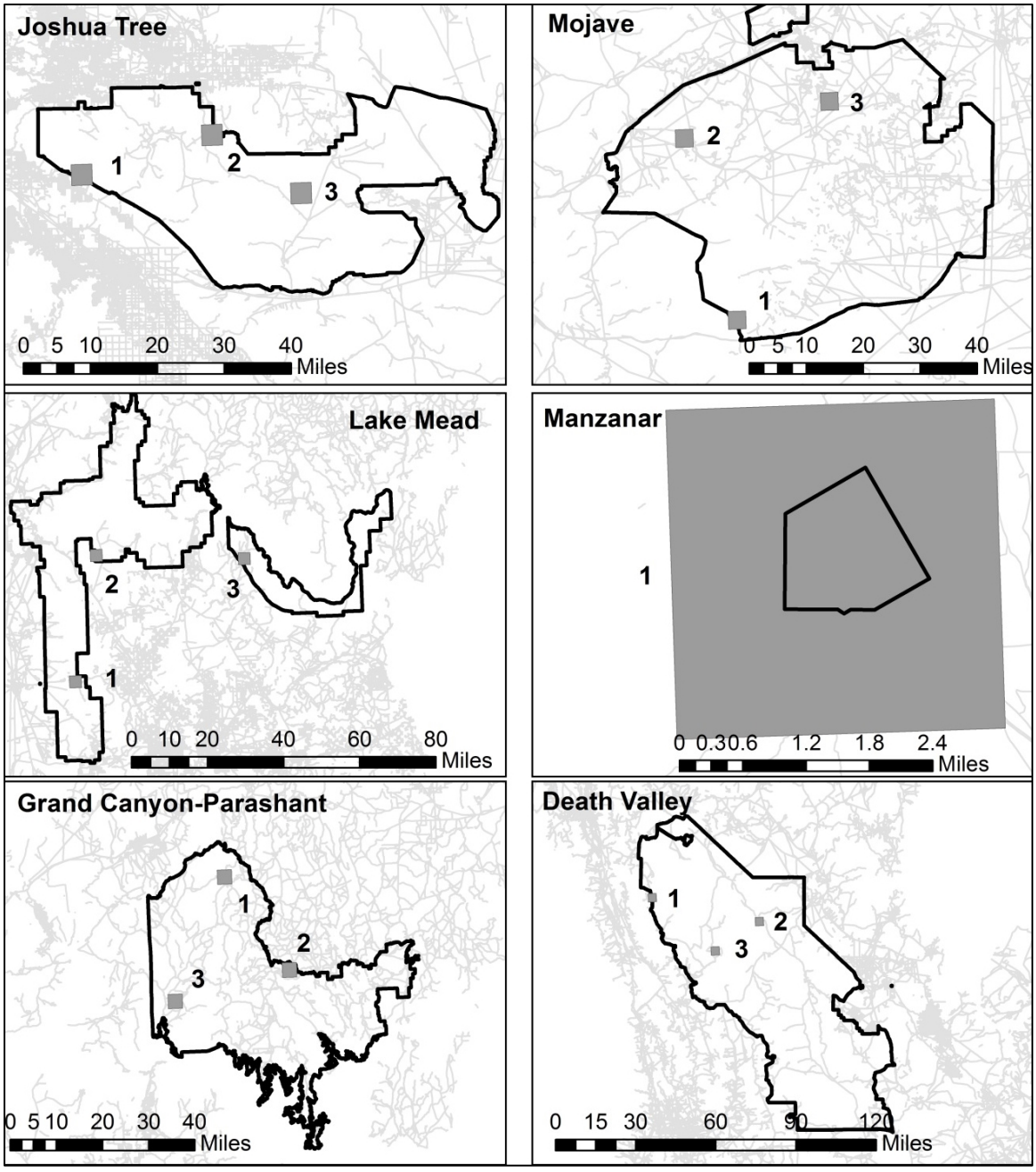


Figure 3. Locations of 16 test areas within the Mojave Desert Network in which we compared publicly available data [Topologically Integrated Geographic Encoding and Referencing (TIGER) data and National Park Service data] on roads and all-terrain vehicle trails to data hand-digitized from aerial photographs. Test areas were 25 km^2 except within Manzanar, where we restricted our analyses to the park's area of 3.3 km^2.

5.B.2.a. Roads and trails

To assess the accuracy of publicly available data on roads and all-terrain vehicle trails outside of the Mojave Desert Network parks, we obtained TIGER data and merged the data among counties (Appendix 1). From these data we extracted primary highways (interstate highways; Master Address

File [MAF]/TIGER Feature Class Code [MTFCC] S1100), secondary highways (federal and state highways; MTFCC S1200), local roads (arterial and non-arterial roads, including interstate off-ramps, that connect to secondary and primary highways; MTFCC S1400 and S1630), other roads (e.g., private paved and dirt roads, service roads, and parking areas; MTFCC S1640, S1730, S1740, and S1780), all-terrain vehicle (ATV) trails (mainly on public lands; MTFCC S1500), and non-motorized vehicle trails (MTFCC S1710, 1820, and 1830). Definitive identification of ATV trails and inferences about apparent ATV trails can be challenging. Use of ATVs is legal in Grand Canyon–Parashant. Moreover, closed roads or trails, or historic roads or trails, erroneously may appear to be ATV trails. Because local knowledge or evaluation data were scarce, we encourage *ATV trails* to be interpreted simply as a class of features with similar appearance and potentially similar effects on movements of species, but unknown status.

Outside of the parks, we digitized secondary highways, local roads that appeared to be either gravel or paved, other roads, and ATV trails from aerial photographs. To ensure complete digitizing, we subdivided each 25 km^2 test area into 25 1-km^2 areas. We conducted all digitizing at the 1:2000 cartographic scale, the finest possible resolution without pixelation of the photographs.

We also compared the extent of the road network as estimated from hand-digitized aerial photographs to road data provided to us by each of the Mojave Desert Network parks. We found that the NPS road classification schemes, and the proportion of roads classified, were inconsistent among parks. For example, Death Valley classified road surfaces as primary, secondary, or tertiary. Grand Canyon–Parashant classified road surfaces as primary paved or unpaved, secondary paved or unpaved, tertiary unpaved, single track, or reclaimed. Joshua Tree, by contrast, classified road surfaces as paved, dirt, or dirt 4x4; 27% of the roads were classified as closed or possibly closed. Lake Mead classified road surfaces as secondary highway, major road, local road, or four-wheel drive. Road surfaces in Manzanar were classified as highway or unimproved road. Mojave National Preserve's classification included surfaces as paved, oil, gravel, dirt, or 4x4, and 6% of roads were not classified. The inconsistencies among classifications make it difficult to compare the potential effects of roads on ecological processes among parks, or to estimate the effects of roads on ecological processes at the level of the Mojave Desert Network.

Because classifications used by most of the parks were more highly resolved than the classes we applied to hand-digitized roads (paved, unpaved, or all-terrain vehicle [ATV]) or the TIGER/Line road classes (primary highways, secondary highways, local roads, other roads, ATV trails, and non-motorized vehicle trails), we lumped all road classes to compare data sets. To compare the TIGER/Line, NPS, and hand-digitized data, we converted all sets of data to raster. Because exact road locations were not perfectly aligned among the three data sets, we identified where roads in the three sets of data did not overlap within 60 m.

5.B.2.b. Power lines

We obtained information on major power-line corridors from both TIGER/Line data and the Interior Columbia Basin Ecosystem Management Project (ICEBMP) (Appendix 1). Neither source of data includes power-line grids for local distribution of electricity (Leu et al. 2008). From the TIGER/Line road data we extracted all segments with MTFCC L4020. We conducted all digitizing from aerial

photographs at the 1:1000 cartographic scale, the finest possible resolution at which the shadows of power line posts could be discerned.

5.B.2.c. Railroads

We extracted TIGER/Line data with MTFCCs R1011, R1051, and R1052 to produce a seamless spatial data layer on railroads.

5.B.2.d. Mines

We obtained publicly available data on active mines and mineral processing plants as of 2003 from the U.S. Geological Survey (Appendix 1). These data are point locations rather than polygons corresponding to the footprints of the mines. We hand-delineated the extent of mines on the basis of aerial photographs. We zoomed in on each mine and delineated the mine's footprint, which in most cases was clearly visible, on the basis of changes in land cover at a 1:2,000 cartographic scale. If the USGS data indicated a point location but no mine footprint was visible at the corresponding location on the aerial photographs, we searched for mines within 500 m. If we did not find a mine within 500 m of the putative location, we considered the record invalid.

5.B.2.e. Other land uses

We extracted the extent of agricultural land from the NLCD. We used the NLCD to delineate four classes of housing: open space (defined in the NLCD as areas with some constructed materials, but mostly lawn with impervious surfaces < 20% of total cover), low-intensity housing (single-family units with impervious surfaces 20–49% of total cover), intermediate-intensity housing (single-family units with impervious surfaces 50–79% of total cover), and high-intensity housing (highly developed areas with impervious surfaces 80–100% of total cover).

Because proposed development of solar, wind, and geothermal energy across the Mojave Desert is highly dynamic, and sources of data are not comparable among parks and political jurisdictions, we reviewed the literature on potential responses of natural resources to these forms of land use.

5.B.2.f. Urbanization

Land use affects ecological processes across both local and regional extents (Scott et al. 2001, Leu et al. 2008). We used Bierwagen et al.'s (2010) decadal projections of impervious-surface cover and housing density at 1 km resolution (i.e., pixels that are 1 km per side, or 1 km^2) to project land cover across the assessment area in 2050. Bierwagen et al. (2010) projected the distribution of five housing-density classes: commercial/industrial, urban (<0.1 ha/housing unit), suburban (0.1–0.68 ha/unit), exurban (0.68–16.18 ha/unit), and rural (>16.18 ha/unit). Their projections were based on five scenarios of trends in human demography, economics, technology, and land use from 2000 to 2100. Scenarios varied with respect to human reproductive rates, domestic and international human migration, household size, and time required to travel away from urban cores; data on which the scenarios were based are available from the U.S. Environmental Protection Agency's Integrated Climate and Land-Use Scenarios project (Appendix 1). We averaged changes in impervious-surface cover and housing density across the five scenarios.

5.B.3. Ecological effects of land use

To estimate effect zones, we identified anthropogenic features and land-cover types that typically reduce use of habitat by vertebrate species: primary and secondary highways (Forman 2000, Rowland et al. 2000, Rudolph et al. 1999, Stoms 2000), urban land cover (Vistnes and Nellemann 2001, Robel et al. 2004), agricultural land cover (Vander Haegen et al. 2002), and power lines (Robel et al. 2004, Silva et al. 2010). We also included active mines and railroads because these features reduce habitat quality for vertebrates via alteration of land cover (Nielsen et al. 2011).

We defined habitat quality as intensity of use (or probability of species presence) by vertebrates. We assumed that all else being equal, habitat quality increases nonlinearly (Nielsen et al. 2009) as distance from an anthropogenic feature increases (Maestas et al. 2003, Doherty et al. 2008, Kociolek et al. 2011).

We estimated habitat quality as

$$\textbf{min value}\left(\sum_{\mathbf{i=1}}^{\mathbf{n}}(\mathbf{1}-[\mathbf{exp}(\mathbf{Euclidean\ distance}/-\boldsymbol{\theta}_{\mathbf{i-n}})]/\mathbf{n}\right),$$

where Euclidean distance is the shortest two-dimensional distance from any 30-m cell to the nearest perimeter of an anthropogenic feature. We calculated Euclidean distances in ArcGIS 10.0 (ESRI 2013). Θ is a scalar that defines the asymptote where habitat quality approaches either 0 or 1 at a distance equivalent to the radius of a mean species-specific circular home range. For example, if θ=15, then habitat quality approaches 1 at 149 m, equivalent to the radius of a 7-ha circular home range.

To estimate habitat quality, we first converted shapefiles of anthropogenic features into raster files (grids) with 30-m resolution. Second, we calculated the distance from each cell to the nearest anthropogenic feature (cell value = 1) with the Eucdistance function in Spatial Analyst (ArcGIS, ESRI, 2013). We defined mean home range sizes on the basis of an extensive review of the literature on 63 vertebrate species (34 birds, 9 reptiles, and 20 mammals) that occur in the assessment area (Appendix 2). We grouped species' home ranges into 17 size classes on the basis of breaks in the frequency distributions of home range sizes (Table 8). For each size class, we used Raster Calculator in Spatial Analyst to apply the habitat-quality function to each of the seven anthropogenic features. We then averaged the 17 sets of habitat-quality values for each anthropogenic feature. We selected the minimum of the seven habitat-quality values to estimate the human footprint.

We examined the relation between values generated by the habitat-quality model and the incidence rate (IR; number of years a species was reported present from 1993–2011) for four guilds of birds classified on the basis of degree of synanthropy (Appendix 3.1 in Johnston 2001): full (requires human development for survival; e.g., House Sparrow), casual (exploits but does not depend on human development; e.g., House Finch), tangential (occasionally exploits human development; e.g., Bullock's Oriole), and non-synanthrope (avoids human development; e.g., Greater Sage-Grouse [*Centrocercus urophasianus*]). To calculate IRs, we used data from the North American Breeding Bird Survey (BBS) (www.pwrc.usgs.gov/bbs/index, accessed 10 November 2013). The BBS data are collected annually along 39.4 km (24.5 mi) routes by volunteers; 39 routes coincided with the study

area. Volunteers record any bird species detected within a 250-m radius of point locations spaced at 0.8 km (0.5 mi) intervals along the route. We estimated IRs for each guild, route, and year. We buffered each BBS route by 1 km and derived its mean habitat-quality value. We used linear regression implemented in R (version 2.13.2; R Development Core Team 2011) to examine the relation between IR and habitat quality.

Table 8. Mean home-range sizes (rounded to nearest ha) of 63 vertebrate species (65 home-range estimates).

Mean (SD) home range (ha)	Range of home-range sizes (ha)	Number of species
0.5*	0.02–0.5	10
1.0 (0.3)	1.0–1.8	6
2.0 (0.3)	2.0–2.8	7
4.0 (0.3)	3.3–3.8	3
7.0 (0.9)	6.4–8.0	3
13 (3)	10.8–18.5	5
65	–	1
148 (7.8)	143–154	2
233 (19.5)	205–248	4
312 (8.9)	295–317	5
416 (30.5)	391–450	3
728 (157)	522–895	4
1418.0 (398)	1147–1875	3
2365 (120)	2280–3400	2
7995 (2415)	5361–9760	3
24739 (10085)	13788–32296	3
238450	–	1

*Not averaged

5.B.4. Anthropogenic light and sound

Because no data on the responses of wildlife in the Mojave Desert Network to anthropogenic light or sound were available, we used literature reviews to explore potential responses. Where available, we summarized existing data on levels of anthropogenic light and sound in the Mojave Desert Network parks.

C. Assessment results

5.C.1. Land use and land cover change

Changes in land use and land cover from 1990 through 2006 were limited, and the change process per se likely is not having a substantial effect on most species that inhabit the Mojave Desert Network. Land cover changed across 0.7% of the assessment area from 1992–2001 (Table 9). The percentage of urban land cover changed the least. More than 20% of grassland/shrub and forest, and 10–20% of agriculture and barren, changed to another class. The greatest percentages of change were

from forest to grassland/shrub (20.0%), grassland/shrub to urban (15.0%), grassland/shrub to barren (14.4%), agriculture to grassland/shrub (11.1%), grassland/shrub to agriculture (9.6%), and barren to grassland/shrub (8.7%). The percentage of change from wildland (i.e., barren, forest, grassland/shrub, open water, and wetland) to urban (17.2%) was greater than from wildland to agriculture (12.0%). Increases in urban land cover were greatest in the regions around Las Vegas, Nevada; Lancaster, California; and Yuma, Arizona (Figure 4).

Table 9. Percentage of land-cover change across the Mojave Desert from 1992–2001. Data derived from the National Land Cover Database.

From	To	Number of 30-m cells	Change (%)
Agriculture	Barren	47207	1.8
Agriculture	Forest	1289	0.1
Agriculture	Grassland/shrub	284980	11.1
Agriculture	Open water	8726	0.3
Agriculture	Urban	32658	1.3
Agriculture	Wetland	17691	0.7
Barren	Agriculture	24018	0.9
Barren	Forest	2246	0.1
Barren	Grassland/shrub	221783	8.7
Barren	Open water	4639	0.2
Barren	Urban	36138	1.4
Barren	Wetland	3880	0.2
Forest	Agriculture	16027	0.6
Forest	Barren	12903	0.5
Forest	Grassland/shrub	511480	20.0
Forest	Open water	2013	0.1
Forest	Urban	15097	0.6
Forest	Wetland	8833	0.3
Grassland/shrub	Agriculture	244776	9.6
Grassland/shrub	Barren	367993	14.4
Grassland/shrub	Forest	34875	1.4
Grassland/shrub	Open water	55641	2.2
Grassland/shrub	Urban	385162	15.0
Grassland/shrub	Wetland	71947	2.8
Open water	Forest	1681	0.1
Open water	Agriculture	2172	0.1
Open water	Barren	14099	0.6
Open water	Grassland/shrub	26050	1.0
Open water	Urban	2923	0.1

Table 9 (continued). Percentage of land-cover change across the Mojave Desert from 1992–2001. Data derived from the National Land Cover Database.

From	To	Number of 30-m cells	Change (%)
Open water	Wetland	4031	0.2
Urban	Agriculture	4963	0.2
Urban	Barren	3004	0.1
Urban	Forest	343	0.0
Urban	Grassland/shrub	23012	0.9
Urban	Open water	4137	0.2
Urban	Wetland	1253	0.0
Wetland	Agriculture	21143	0.8
Wetland	Barren	752	0.0
Wetland	Forest	294	0.0
Wetland	Grassland/shrub	11368	0.4
Wetland	Open water	22406	0.9
Wetland	Urban	4173	0.2

Land cover changed across 0.6% of the assessment area from 2001–2006 (Table 10). The percentage of forest changed the least (0.8%). More than 20% of grassland/shrub; 10–20% of barren, urban, and wetland; and <10% of agriculture changed. The greatest percentages of change were from grassland/shrub to forest (25.8%), barren to grassland/shrub (12.7%), urban to grassland/shrub (9.7%), wetland to grassland/shrub (6.4%), agriculture to grassland/shrub (6.1%), grassland to barren (5.9%), and wetland to open water (5.0%). The percentages of wildland that changed to urban and agriculture were 0.5% and 7.0%, respectively. North of Lake Mead National Recreation Area and in the Owens Valley, large areas of barren and agricultural land changed to grassland/shrub, probably as a result of natural succession (Figure 5). Agricultural land cover increased in the greater Palm Springs, California area (Figure 5).

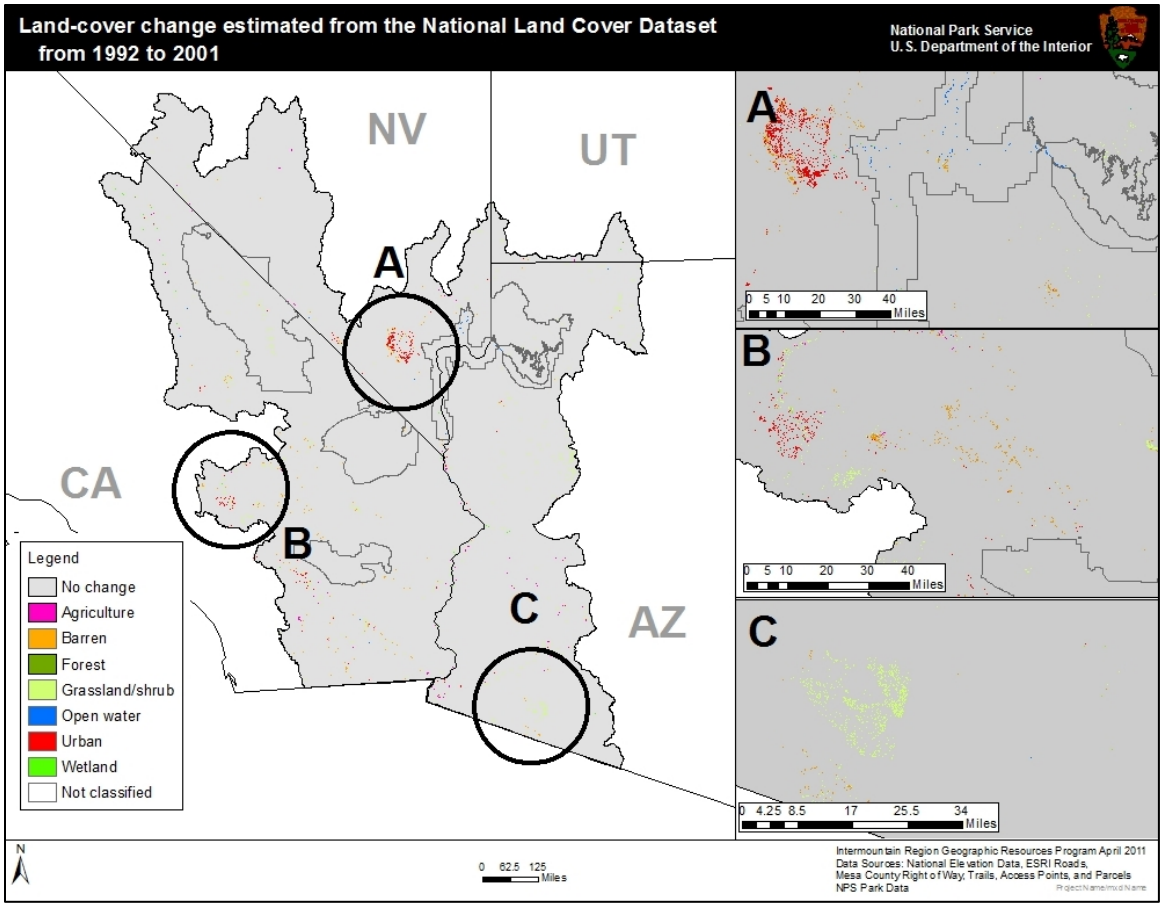


Figure 4. Estimated changes in the distribution of seven classes of land cover, as classified by the National Land Cover Database, from 1992–2001.

Table 10. Percentage of land-cover change across the Mojave Desert from 2001–2006. Data derived from the National Land Cover Database.

From	To	Number of 30-m cells	Change (%)
Agriculture	Barren	12819	0.83
Agriculture	Forest	260	0.02
Agriculture	Grassland/Shrub	90110	5.81
Agriculture	Open water	463	0.03
Agriculture	Urban	2225	0.14
Agriculture	Wetland	3122	0.20
Barren	Agriculture	9221	0.59
Barren	Forest	3110	0.20
Barren	Grassland/shrub	186585	12.03
Barren	Open water	57828	3.73
Barren	Urban	492	0.03
Barren	Wetland	4726	0.30

Table 10 (continued). Percentage of land-cover change across the Mojave Desert from 2001–2006. Data derived from the National Land Cover Database.

From	To	Number of 30-m cells	Change (%)
Forest	Agriculture	253	0.02
Forest	Barren	868	0.06
Forest	Grassland/shrub	9323	0.60
Forest	Open water	173	0.01
Forest	Urban	170	0.01
Forest	Wetland	727	0.05
Grassland/shrub	Agriculture	77108	4.97
Grassland/shrub	Barren	86789	5.60
Grassland/shrub	Forest	380463	24.54
Grassland/shrub	Open water	40268	2.60
Grassland/shrub	Urban	5365	0.35
Grassland/shrub	Wetland	16385	1.06
Open water	Agriculture	8117	0.52
Open water	Barren	15101	0.97
Open water	Forest	1207	0.08
Open water	Grassland/shrub	34648	2.23
Open water	Urban	241	0.02
Open water	Wetland	5823	0.38
Urban	Agriculture	20221	1.30
Urban	Barren	69545	4.49
Urban	Forest	59	0.00
Urban	Grassland/shrub	143704	9.27
Urban	Open water	219	0.01
Urban	Wetland	698	0.05
Wetland	Agriculture	8778	0.57
Wetland	Barren	7079	0.46
Wetland	Forest	1458	0.09
Wetland	Grassland/shrub	94445	6.09
Wetland	Open water	73713	4.75
Wetland	Urban	1453	0.09

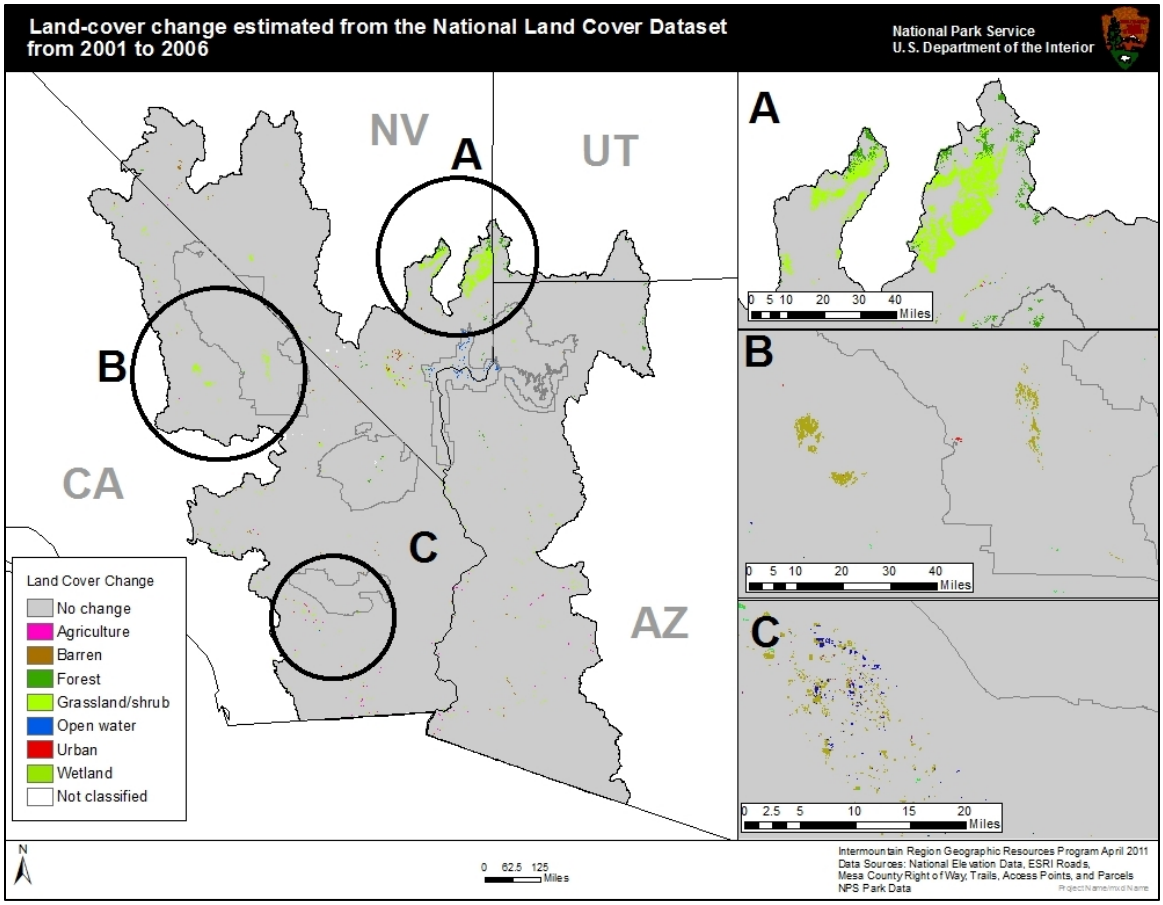


Figure 5. Estimated changes in the distribution of seven classes of land cover, as classified by the National Land Cover Database, from 2001–2006.

Utility-scale solar energy production—generation of electricity that is delivered into the transmission grid, generally with capacities greater than 20 MW (USDOI 2012)—is the dominant type of renewable energy development in the southwestern United States. Research on changes in land cover in the Mojave Desert as a result of recent and proposed energy development has not been conducted (Lovich and Ennen 2011, Cameron et al 2012). However, the literature suggests potential effects (Copeland et al. 2011, Pocewicz et al. 2011, Cameron et al. 2012). Land in the footprint of a utility-scale solar facility is cleared to maximize capture of sunlight; the amount of solar energy produced is directly proportional to the area covered (McDonald et al. 2009). The utility-scale solar energy footprint may be comparable to that of coal (Muller et al. 2011). Clearing and grading of land for utility-scale solar energy production can cover 405–60,703 ha (1,000–150,000 ac) (DiDonato 2012). Because vegetation growth and succession in the Mojave Desert generally is slow, clearing and grading may have long-term effects. Shifts in microtopography and drainage patterns due to grading for facility and road construction may affect water infiltration and erosion rates, soil density, secondary plant succession, invasion by non-native plant species, and the stability of cryptobiotic soil crusts (Lovich and Bainbridge 1999, Webb et al. 2009). Energy production and maintenance of energy facilities may create additional demands on surface and groundwater, and dust suppressants

and herbicides used to maintain areas from which vegetation has been cleared may affect local or regional air quality (USDOI 2012).

5.C.2. Accuracy of data on land use

5.C.2.a. Roads and trails

Roads are one of the most substantial drivers of fragmentation of natural land cover. The discrepancies we found among publicly available data on roads, NPS data on roads, and hand-digitization of roads suggest that it currently is quite challenging to draw strong inferences on the location and extent of these features and their potential ecological effects.

Across the assessment area, densities of roads were high near major urban and agricultural areas (Figure 6). Within the assessment area, the TIGER/Line data indicated 2,879 km of primary highways, 6,443 km of secondary highways, 151,627 km of local roads, 2,950 km of other roads, 15,682 km of ATV trails, and 79 km of non-motorized vehicle trails (Figure 7).

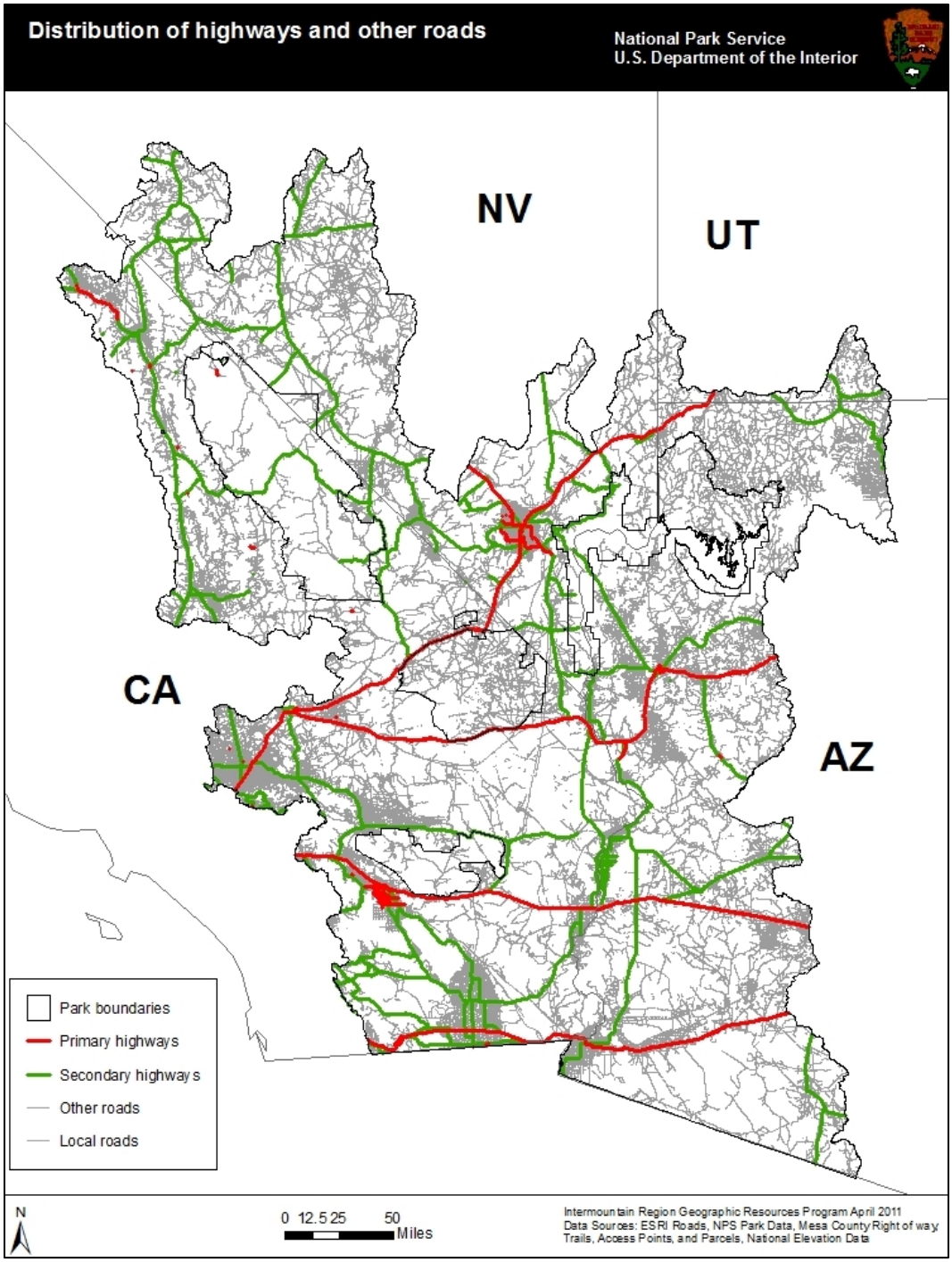


Figure 6. Delineation of roads as indicated by publicly available Topologically Integrated Geographic Encoding and Referencing (TIGER) data.

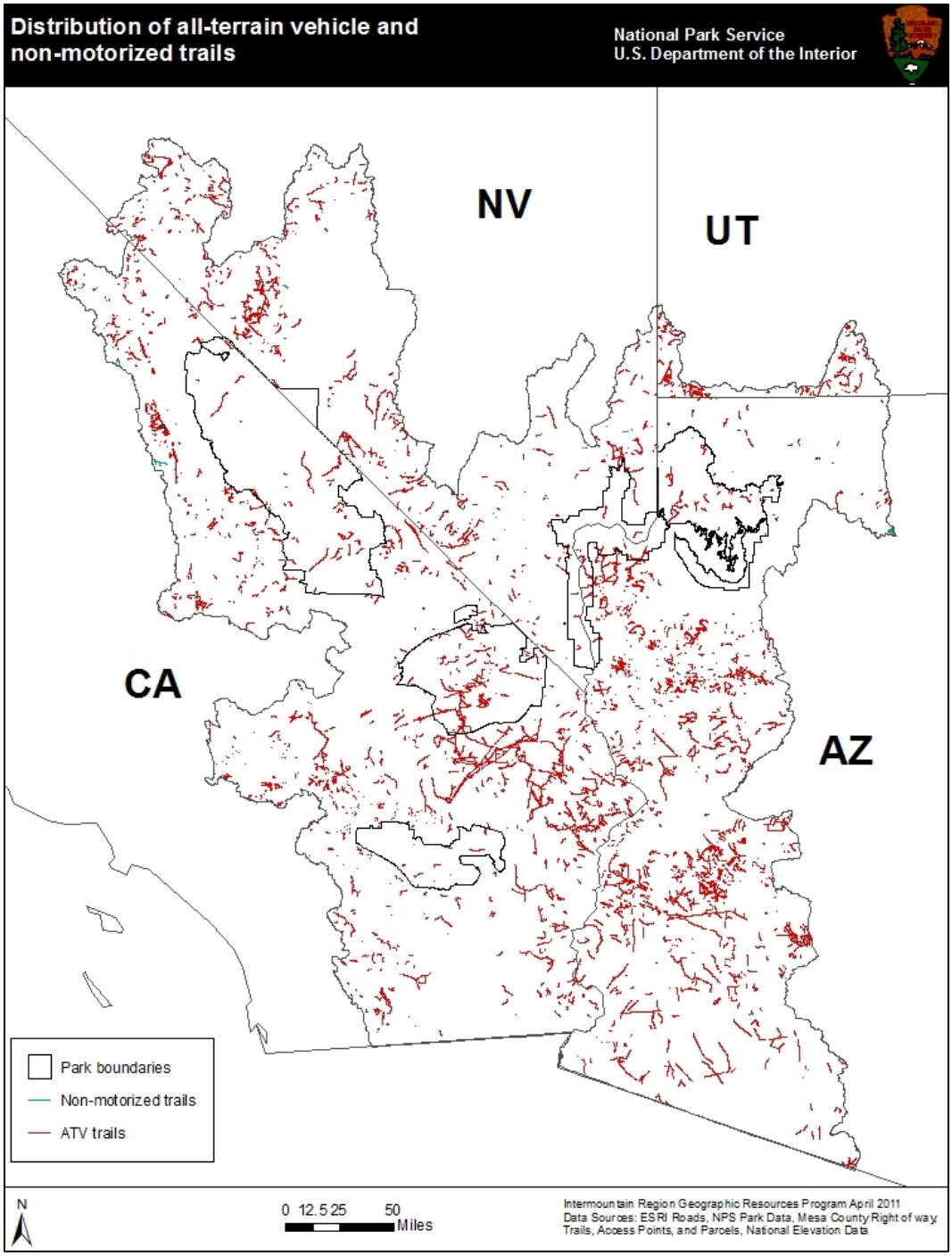


Figure 7. Delineation of all-terrain vehicle and non-motorized vehicle trails as indicated by publicly available Topologically Integrated Geographic Encoding and Referencing (TIGER) data.

Outside of the Mojave Desert Network, we found that secondary highways, local and other roads, and ATV trails were underrepresented in the TIGER/Line data (Figure 8). Agreement between data sets was higher for secondary highways (95.0 ± 7.1% [mean ± SD], range 84.9–100.0%) than for

local and other roads (76.7 ± 30.1%, range 0–100%) (Table 11). Mean agreement between data sets for ATV trails was 8.0% ± 25.2% (range 0.0–100.0%) (Table 12).

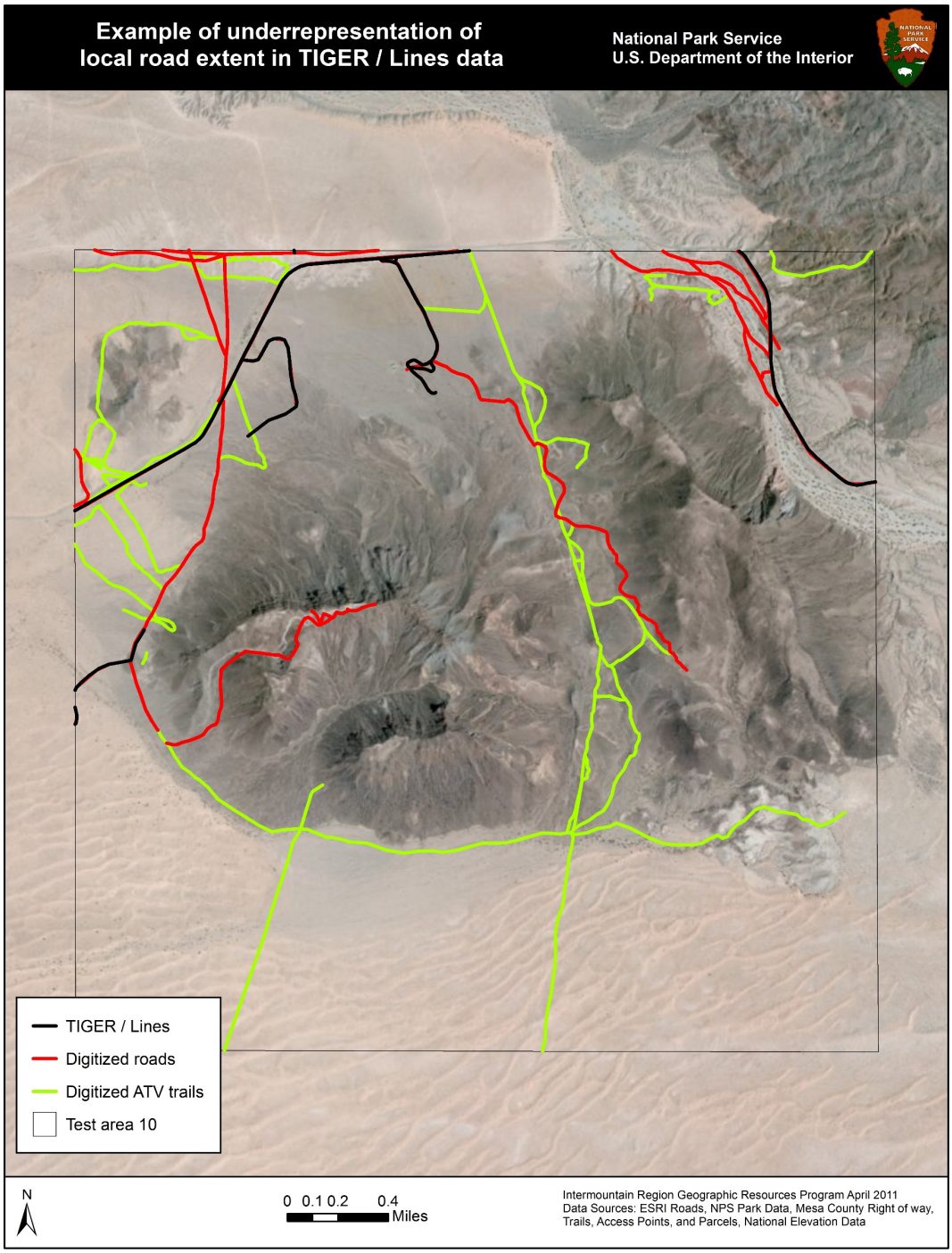


Figure 8. Example of the underrepresentation of the extent of secondary highways, paved or gravel local roads, and all-terrain vehicle trails or closed roads in publicly available Topologically Integrated Geographic Encoding and Referencing (TIGER) data for area 10 (Figure 2). The TIGER data did not indicate the presence of all-terrain vehicle trails, which are legal in Arizona, or closed roads.

Table 11. Agreement between TIGER/ Line data and heads-up digitized data on local roads (arterial and non-arterial roads connecting to secondary and primary highways, including interstate off-ramps), other roads (e.g., private paved and dirt roads, service roads, and parking areas), and secondary highways (federal and state highways) for 17 test areas of 25 km^2 (Figure 4). Number of cells reflects that linear data on roads were converted to raster. Blank cells in the table indicate no data on roads.

	Local and other roads Number of cells			Secondary highways Number of cells		
Test area	Digitized	TIGER	Agreement (%)	Digitized	TIGER	Agreement (%)
1	324	246	75.9	–	–	–
2	–	–	–	106	106	100.0
3	323	323	100.0	–	–	–
4	337	327	97.0	–	–	–
5	160	159	99.4	–	–	–
6	1362	1298	95.3	–	–	–
7	885	841	95.0	120	120	100.0
8	2582	2434	94.3	232	197	84.9
9	39	0	0.0	–	–	–
10	629	320	50.9	–	–	–
11	189	184	97.4	–	–	–
12	509	369	72.5	–	–	–
13	463	427	92.2	–	–	–
14	1161	217	18.7	–	–	–
15	356	218	61.2	–	–	–
16	355	314	88.5	–	–	–
17	621	550	88.6	–	–	–

Table 12. Agreement between TIGER/Line data and heads-up digitized data on all-terrain vehicle trails for 17 test areas of 25 km^2 (Figure 4).

	Number of cells		
Test area	Digitized	TIGER	Agreement (%)
1	47	0	0.0
2	295	0	0.0
3	392	0	0.0
4	200	0	0.0
5	43	0	0.0
6	992	0	0.0
7	495	0	0.0
8	1895	0	0.0

Table 12 (continued). Agreement between TIGER/Line data and heads-up digitized data on all-terrain vehicle trails for 17 test areas of 25 km^2 (Figure 4).

	Number of cells		
Test area	Digitized	TIGER	Agreement (%)
9	522	187	35.8
10	573	0	0.0
11	96	96	100.0
12	2779	0	0.0
13	56	0	0.0
14	203	0	0.0
15	39	0	0.0
16	198	0	0.0
17	295	0	0.0

In 56.2% of the test areas, the extent of roads (paved and unpaved roads and ATV trails) as measured by the number of 60-m raster cells was highest in the NPS data (Table 13). The percentages of roads present in a given data source that were unique to that source were 5.9 ± 5.8% (mean ± SD), 10.6 ± 17.1%, and 8.3 ± 9.9% for the TIGER/Line, hand digitized, and NPS data, respectively (Table 13). In contrast to the hand digitized and TIGER/Line data, the NPS data for test area 3 within Lake Mead did not indicate the presence of any roads (Figure 3); the hand-digitized data indicated that the extent of roads was ten times greater than suggested by the TIGER/Line data. The mean percentage of overlap between two of the three sources of data was greatest for the hand digitized and NPS data (18.0 ± 20.8%) and lowest for the TIGER/Line and NPS data (2.2 ± 3.4%). The percentage of overlap between the TIGER/Line and hand-digitized data was 8.4 ± 13.9%. Mean agreement among the three sources of data was 46.6 ± 26.3%. Accounting for both the overlap between two of three sources and the overlap among the three sources, mean agreement between the hand digitized and NPS data, hand digitized and TIGER/Line data, and TIGER/Line and NPS data was 64.6 ± 24.7%, 57.2 ± 25.6%, and 48.8 ± 27.6%, respectively (Table 13).

Table 13. Road extent (number of 60-m raster cells in which roads were present), percentage of cells that appeared to contain roads or all-terrain vehicle (ATV) trails in one of the three data sources (unique), percentage of cells that appeared to contain roads or ATV trails in two of the three data sources, and percentage of agreement among hand digitized (HD), **T**opologically **I**ntegrated **G**eographic **E**ncoding and **R**eferencing (TIGER [T]), and National Park Service (NPS) data for 16 test areas in Mojave Desert Network parks (Figure 3). Test areas were 25 km2 in all parks except Manzanar, in which we restricted analyses to the 3.3 km^2 area of the park. Grand Canyon and Mojave refer to Grand Canyon–Parashant and Mojave National Preserve, respectively. The total percentage overlap between two data sources is the sum of the percentage overlap between two of three data sources and the percentage overlap among all data sources.

	Road extent			Percentage unique			Percentage overlap between two of three data sources			Percentage overlap among all data sources
Test area	T	HD	NPS	T	HD	NPS	T, HD	T, NPS	HD, NPS	
Death Valley 1	159	197	100	3.9	1.5	0.0	46.1	0.5	21.8	26.2
Death Valley 2	118	117	116	1.7	0.0	1.7	2.5	0.0	0.8	93.4
Death Valley 3	433	464	458	3.5	1.6	4.7	5.5	1.2	9.2	74.4
Grand Canyon 1	303	447	456	2.2	15.5	16.6	0.2	0.7	13.3	51.5
Grand Canyon 2	141	214	242	7.2	1.5	0.0	14.0	5.3	25.7	46.4
Grand Canyon 3	481	342	490	3.4	14.6	16.7	1.6	1.0	12.6	50.0
Joshua Tree 1	32	101	117	6.0	24.4	33.3	0.0	0.6	23.2	12.5
Joshua Tree 2	361	506	620	6.8	8.0	22.5	0.7	1.8	20.3	39.9
Joshua Tree 3	12	75	75	2.6	0.0	2.6	0.0	0.0	79.2	15.6
Lake Mead 1	425	483	525	3.2	1.6	9.8	1.8	1.1	13.0	69.6
Lake Mead 2	335	293	322	9.7	1.3	1.6	4.5	11.8	9.2	61.9
Lake Mead 3	69	209	0	6.7	69.2	0.0	24.1	0.0	0.0	0.0
Manzanar 1	194	558	583	0.6	7.4	11.0	0.6	0.9	51.3	28.2
Mojave 1	109	104	66	12.3	7.2	11.6	32.6	0.7	2.2	33.3
Mojave 2	347	364	348	0.5	12.3	0.3	0.3	8.3	0.8	77.8
Mojave 3	303	253	240	23.9	3.9	0.0	0.6	0.6	5.7	65.4

Most differences in classification between the hand digitized and NPS data reflected roads or ATV trails in alluvial fans or ATV trails below the tree canopy that were not captured during the heads-up digitization process. In addition, in some cases, trails or abandoned roads were classified as ATV trails during the heads-up digitization process. On the whole, NPS's spatial data on road networks within the Mojave Desert Network parks were consistent with road networks as detected by hand digitization.

5.C.2.b. Power lines

The power line network across the Mojave Desert is extensive (Figure 9). TIGER/Line and ICEBMP data indicated 4,730 and 9,203 km of power lines, respectively. Agreement between the ICEBMP and TIGER/Line data varied spatially. For example, in the southwestern area of the assessment, the ICEBMP delineated an extensive power-line network that was not represented in the TIGER/Line data.

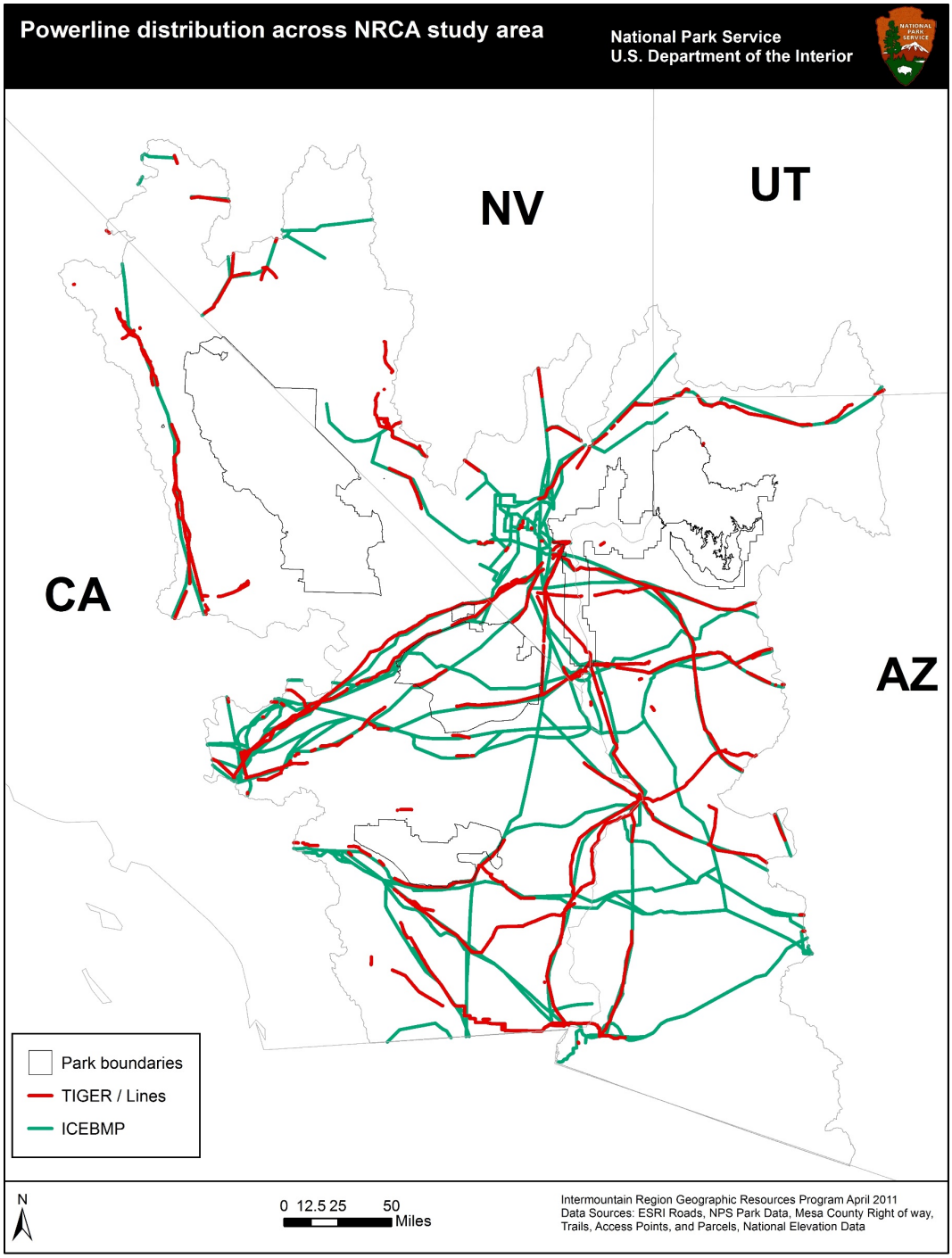


Figure 9. Distribution of power lines as indicated by publicly available **T**opologically **I**ntegrated **G**eographic **E**ncoding and **R**eferencing (TIGER) data and by data from the Interior Columbia Basin Ecosystem Management Project (ICEBMP).

The TIGER/Line power line data were more similar than the ICEBMP data to hand-digitized data. Our digitizing documented the presence of power lines in six of the 17 test areas that included anthropogenic features (Table 14). The TIGER/Line data indicated that power lines were present in

three of those six test areas whereas the ICEBMP indicated that power lines were present in five of the six test areas. The ICEBMP data also indicated that power lines were present in a test area for which neither hand-digitized nor TIGER/Line data suggested power lines were present.

Table 14. Agreement among sources of data on the extent of power lines for 17 25-km2 test areas across the Mojave Desert. Digitized, heads-up hand digitization from aerial photographs; TIGER, **T**opologically **I**ntegrated **G**eographic **E**ncoding and **R**eferencing data; ICEBMP, Interior Columbia Basin Ecosystem Management Project (ICEBMP) data. Test areas 18 and 19 are within Joshua Tree National Park and Mojave National Preserve, respectively (Figure 4). Percent agreement is based on the number of 30-m cells. Blank cells indicate no power line data.

Test area	Digitized	TIGER	Agreement (%)	Digitized	ICEBMP	Agreement (%)
1	–	–	–	–	–	–
2	–	–	–	–	–	–
3	–	–	–	–	–	–
4	–	–	–	–	–	–
5	–	–	–	–	–	–
6	201	0	0.0	–	–	–
7	127	0	0.0	–	135	0
8	1254	100	8.0	–	271	0
9	–	–	–	–	–	–
10	476	210	44.1	–	156	0
11	–	–	–	–	–	–
12	6	0	0.0	–	524	0
13	–	–	–	–	–	–
14	–	–	–	–	469	0
15	96	96	100	30	209	14.4
16	–	–	–	–	–	–
17	–	–	–	–	–	–

The extent of power lines across the Mojave Desert was greatly underestimated in the TIGER/Line data. Mean agreement between the digitized and TIGER/Line data on power lines was 25.3% (SD 39.9%). The low agreement probably reflects absence of low-voltage power lines in the TIGER/Line data. The ICEBMP data also greatly underestimated the extent of power lines and were not spatially accurate. Mean agreement between the digitized and ICEBMP data on power lines was 2.4% (SD 5.4%) (Table 14). This low agreement is difficult to explain given the ICEBMP data indicated power lines were present in five of six test areas where digitizing recorded presence of power lines. Heads-up digitizing may not fully represent the extent of power lines if aerial photographs were taken at noon, when shadows are minimal.

5.C.2.c. Railroads

The TIGER/Line data indicated 3,366 km of railroads within the study area (Figure 10). The presence of railroad fragments in the northwest corner of the study area suggested the TIGER/Line data are incomplete.

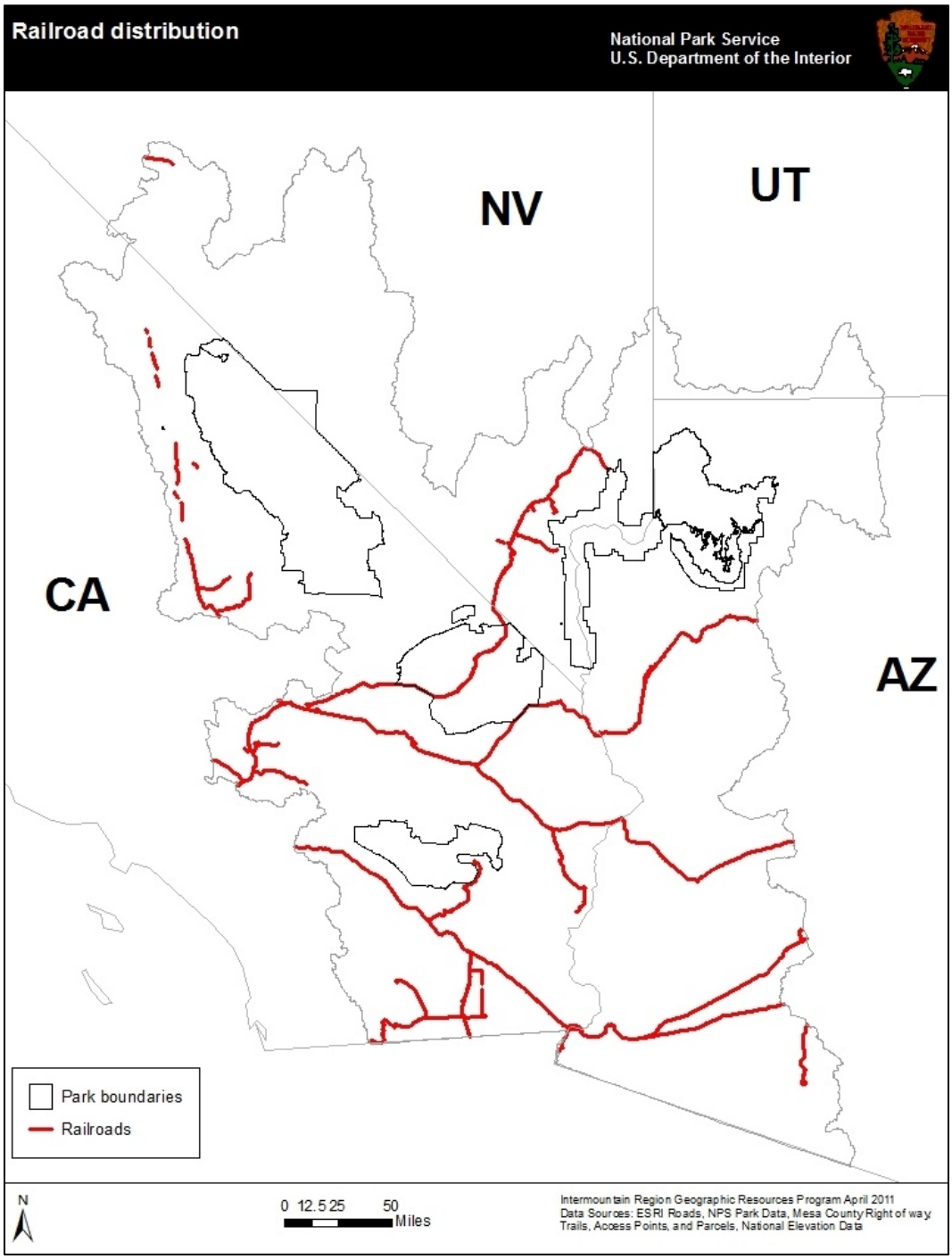


Figure 10. Distribution of railroads as indicated by publicly available **T**opologically **I**ntegrated **G**eographic **E**ncoding and **R**eferencing (TIGER) data.

5.C.2.d. Mines

Publicly available point-location data indicated 148 mines in the assessment area, with lowest densities in Arizona. The areas of mine footprints ranged from < 1 – 150.3 km^2 (4.2 ± 16.2 km^2, mean ± SD). We found no photographic evidence of 37 of the mines (25%) that were identified in the USGS data (Figure 11). In addition, we detected 74 mines from the aerial photographs that were not included in the public data (Appendix 1).

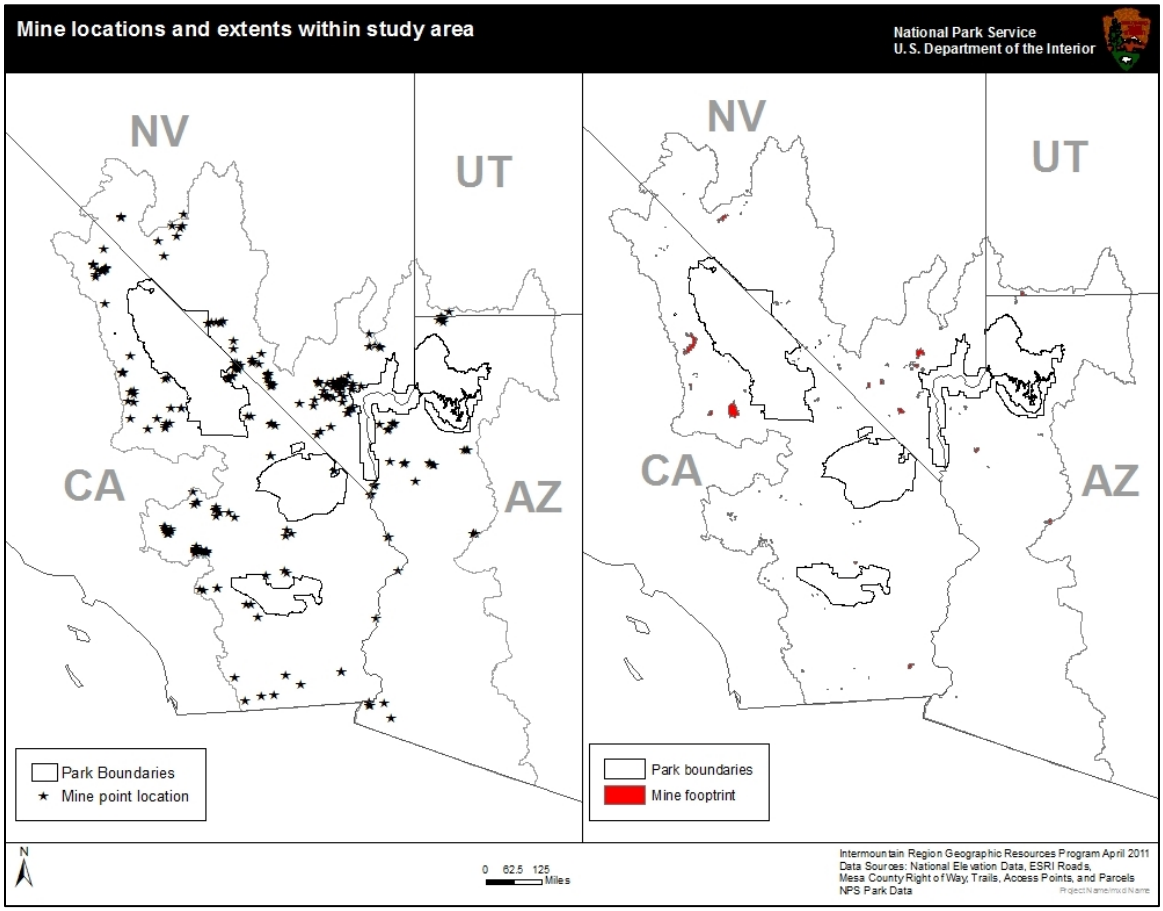


Figure 11. Left panel: point locations of 148 mines identified by the US Geological Survey (Appendix 1). Right panel: extents of 185 mines identified on the basis of aerial photographs.

5.C.2.e. Other land uses

Cultivated crops (NLCD class 82) covered 1.1% (23,199 km^2) of the Mojave and pasture or hay (class 81) covered 0.8% (17,478 km^2). Agriculture was concentrated along the Colorado River and south of the Salton Sea (Figure 12). Housing covered 1.8% of the study area (Figure 13): 0.7% (15,859 km^2) urban-open space, 0.6% (13,888 km^2) low intensity, 0.3% (6,946 km^2) intermediate intensity, and 0.08% (1,646 km^2) high intensity.

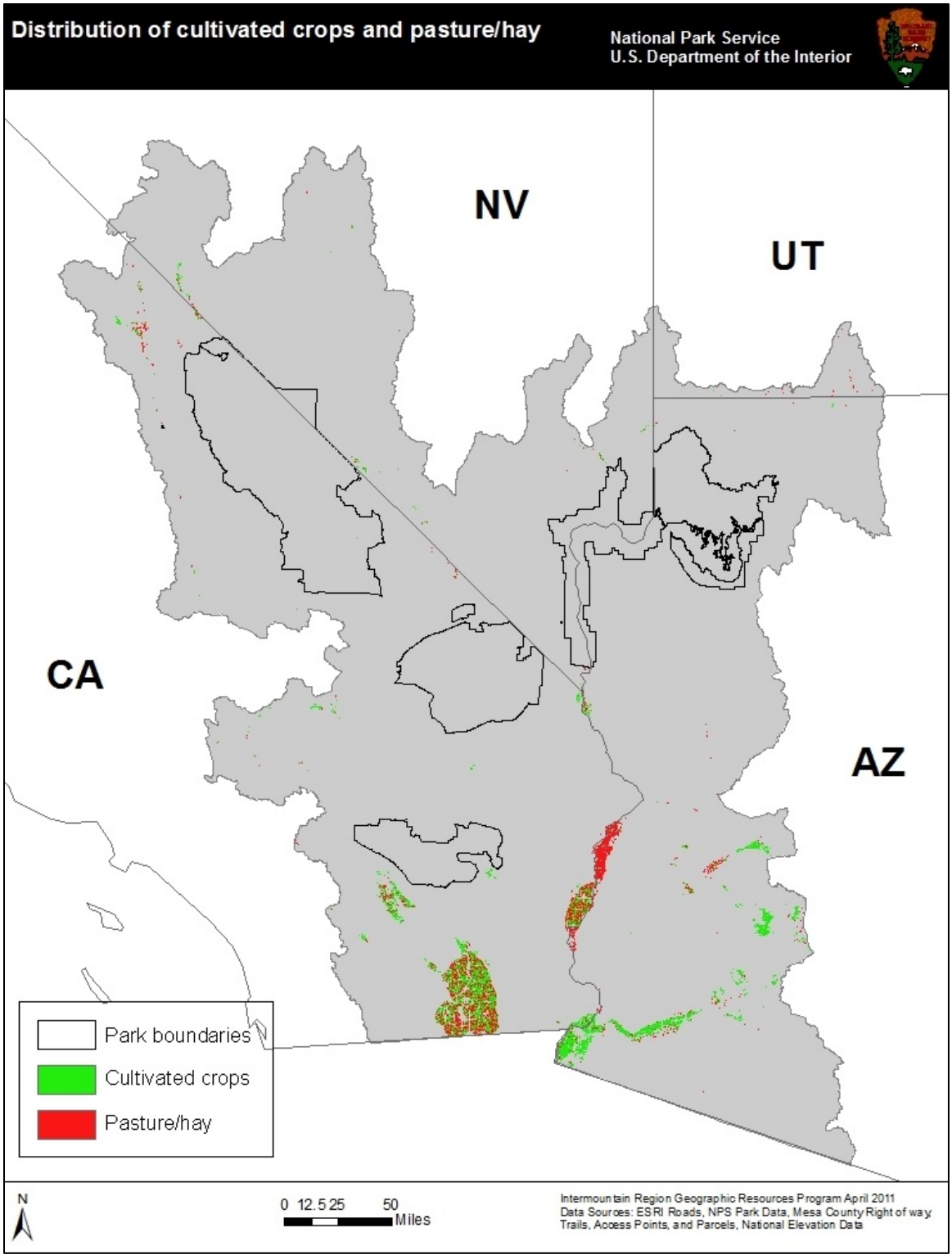


Figure 12. Distribution of cultivated crops and pasture or hay as represented in the National Land Cover Database.

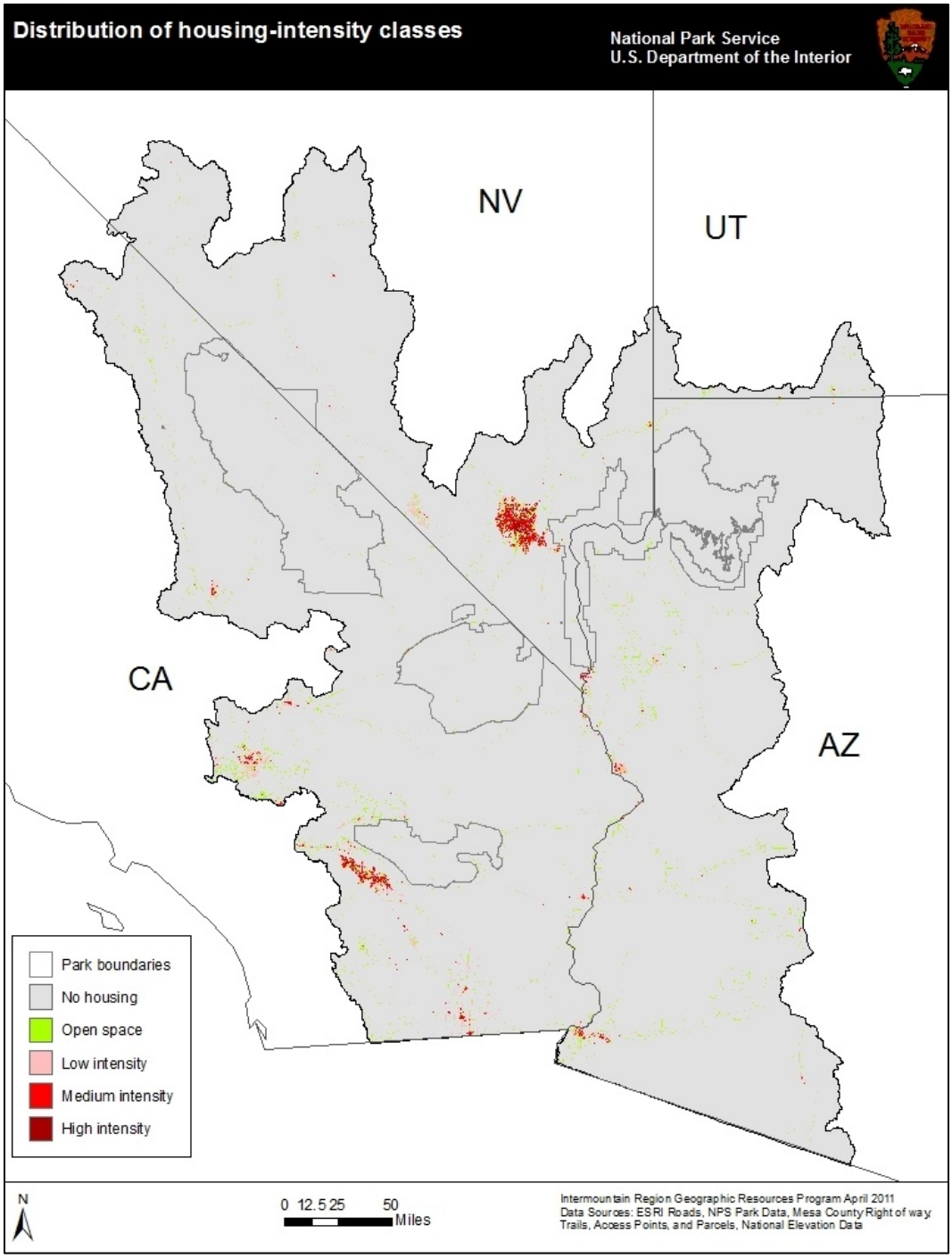


Figure 13. Distribution of housing-intensity classes as represented in the National Land Cover Database. Open space, some constructed materials, but mostly lawn with impervious surfaces < 20% of total cover; low-intensity, single-family units with impervious surfaces 20–49% of total cover; intermediate-intensity, single-family units with 50–79% impervious cover; high-intensity, highly developed areas with 80–100% impervious cover.

Spatial data from the Federal Communications Commission (Appendix 1) indicated that 3,872 communication towers occur within the study area (Figure 14).

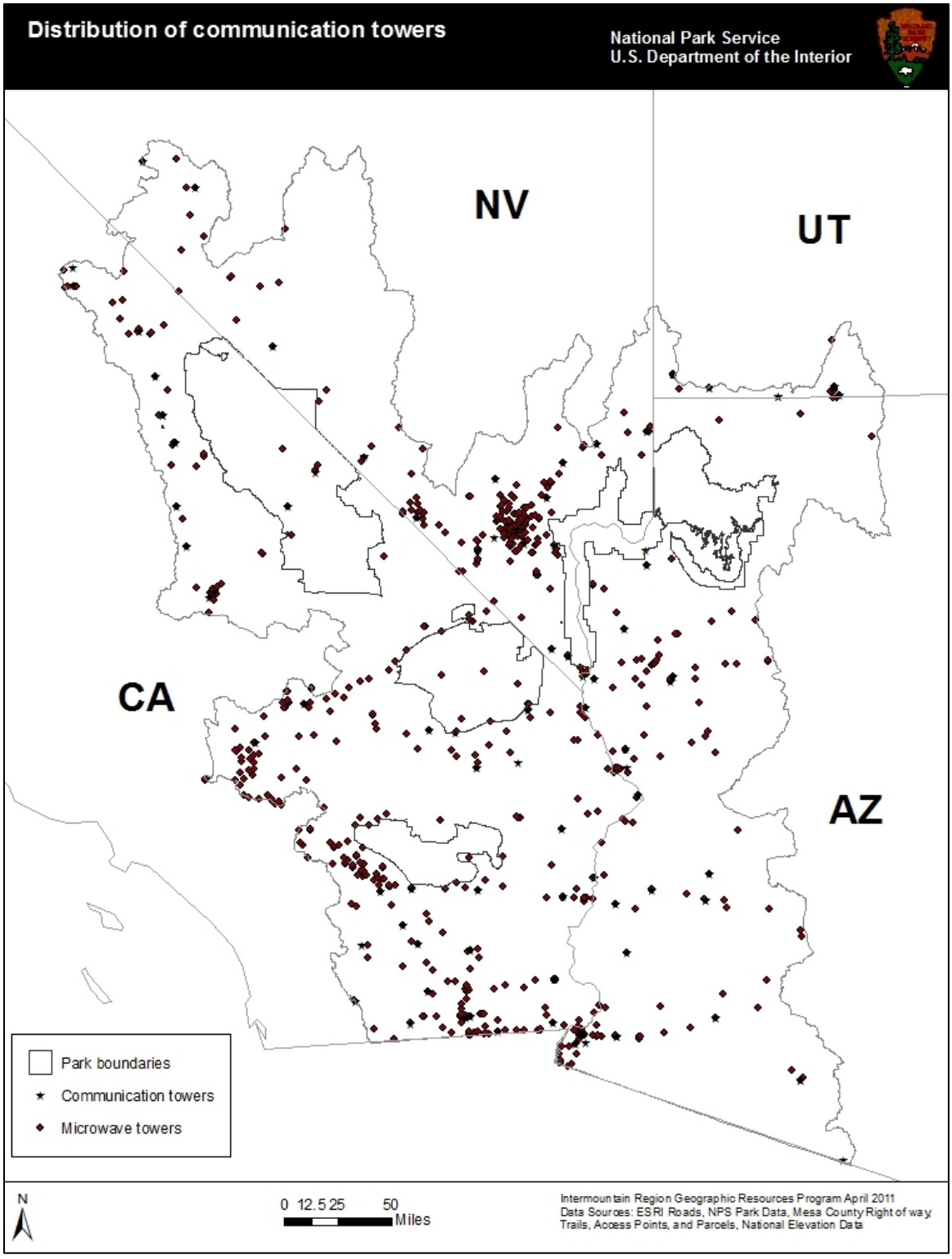


Figure 14. Distribution of communication towers as indicated by data from the Federal Communications Commission.

In 2005, the federal Energy Policy Act (Public Law 109–58) established a target of generating 10,000 megawatts (MW) of approved non-hydropower energy per year within 10 years from facilities on federal land. In California, 33% of retail electricity must be produced with from renewable sources by 2020 (Executive Order 2008), whereas Nevada and Arizona aim to produce 25% of their energy from renewables by 2025 (Arizona 2007, Nevada 2011). The recent Solar Programmatic Environmental Impact Statement (PEIS) (USDOI 2012) included PEISs for wind energy, geothermal energy, and transmission corridors, all of which were driven by demands for efficient permitting of renewable-energy development on federal lands (USDOI 2005, 2008, 2012).

Solar energy

Thermal-concentrating central solar receiver facilities use computer-controlled, sun-tracking heliostats to collect and concentrate sunlight. Vegetation may be mowed to a height of 30–46 cm (12–18") under the heliostats rather than cleared completely. However, temperature in the shadow of the heliostats decreases and wind speed and evapotranspiration change (Pimentel 1994). These changes in microclimate can change soil moisture and soil nutrient availability. Photosynthetic activity of plants with crassulacean acid metabolism (CAM), such as Mojave yucca (*Yucca schidigera*), barrel cactus (*Ferocactus wislizenii*), and cholla (*Opuntia* sp.), could be reduced by shading. Mowing also increases the probability of colonization by non-native invasive species.

Concentrating solar facilities can increase the albedo of nearby lands by 30–56% (Mihlmester et al. 1980), potentially increasing local temperatures and decreasing local precipitation by transporting unused heat downwind (Harte and Jassby 1978, Pimentel 1994). Extensive solar energy development in arid regions can increase dust production through loss of vegetation, cleaning of dust from solar arrays to maintain energy-collection efficiency, increased off-road vehicle traffic, and construction of transmission lines and associated access roads (Field et al. 2009). Photosynthesis, gas exchange, and water usage of Mojave Desert shrubs can be affected by dust deposition (Sharifi et al. 1999). Increases in wind erosion and dust emissions caused by clearance of vegetation for facilities and roads can affect soil fertility and water retention and can physically damage plants through root exposure, burial, and abrasions to leaves and stems (Lovich and Ennen 2011).

Dust suppression is employed in utility-scale solar energy production to reduce road emissions and to keep solar arrays clear of materials that decrease efficiency of mirrors and panels. Dust commonly is suppressed in areas with exposed soil that surround the facility, such as roads, graded areas, and areas from which vegetation has been cleared (Lovich and Ennen 2011). Dust primarily is suppressed with water. Salts and brines also are used to suppress dust. Magnesium chloride, a common chemical used for dust suppression because it extracts moisture from the air and keeps roads damp by resisting evaporation, can damage plants both directly and, via transportation in runoff, indirectly (Goodrich et al. 2009).

Wind Energy

Little information is available in the peer-reviewed literature on responses of land cover to wind-energy development, including not only turbines but infrastructure such as such as roads, transformers, substations, and maintenance facilities (Lovich and Ennen 2013). However, much research has been conducted on potential effects of wind energy production on wildlife, especially

birds and bats, and some information on land-cover change can be gleaned from these studies. Wildlife and land cover in the Mojave Desert may be affected by soil compaction from roads and by construction of culverts to divert water and reduce erosion around turbines in steep terrain. Other effects of roads are discussed above. Surface temperatures can increase in the vicinity of wind farms, which potentially could affect water availability and some plants (Zhou et al. 2012).

Geothermal Energy

The potential for geothermal energy development in the Mojave Desert region is less than for development of solar or wind energy. Although the footprints of geothermal plants can vary considerably among reservoirs and site-specific discharge of waste streams, they are typically 20 and 50 times smaller than the footprints of solar thermal and solar photovoltaic footprints, respectively (MIT 2006). However, increased construction, use, and maintenance of roads, pipelines, and transmission lines are likely. Effects of such infrastructure are discussed above. Additionally, some increases in shrub mortality in the vicinity of geothermal plants have been observed. The mortality was attributed to reinjection of hot fluids.

5.C.2.f. Urbanization

We projected little change in the percent cover of impervious surfaces across the assessment area from 2010 to 2050. We projected that the cover of impervious surfaces will decrease across 9.3% of the region (i.e., the percentage of impervious surfaces in 9.3% of the 1-km pixels will decrease), increase across 17.0% of the region, and will not change across 73.8% of the region. We projected that the percent cover of impervious surfaces in most rural areas will decrease, whereas the cover of impervious surfaces in urban areas will increase (Figure 15). Increases in impervious-surface cover were most pronounced around Las Vegas, Nevada and Victorville, California and along the Colorado River. In these three locations, the changes corresponded to an increase in suburban housing.

We projected no change in housing density in 43.4–100.0% of the area currently covered by each of the five housing-density classes (Table 15). We projected no change in housing density in the areas currently classified as commercial/industrial, whereas we projected housing density would change across more than 56% of land that was classified as suburban in 2010. Housing density was projected to increase across 0–47.3% of each class and to decrease across 0–15.5% of each class.

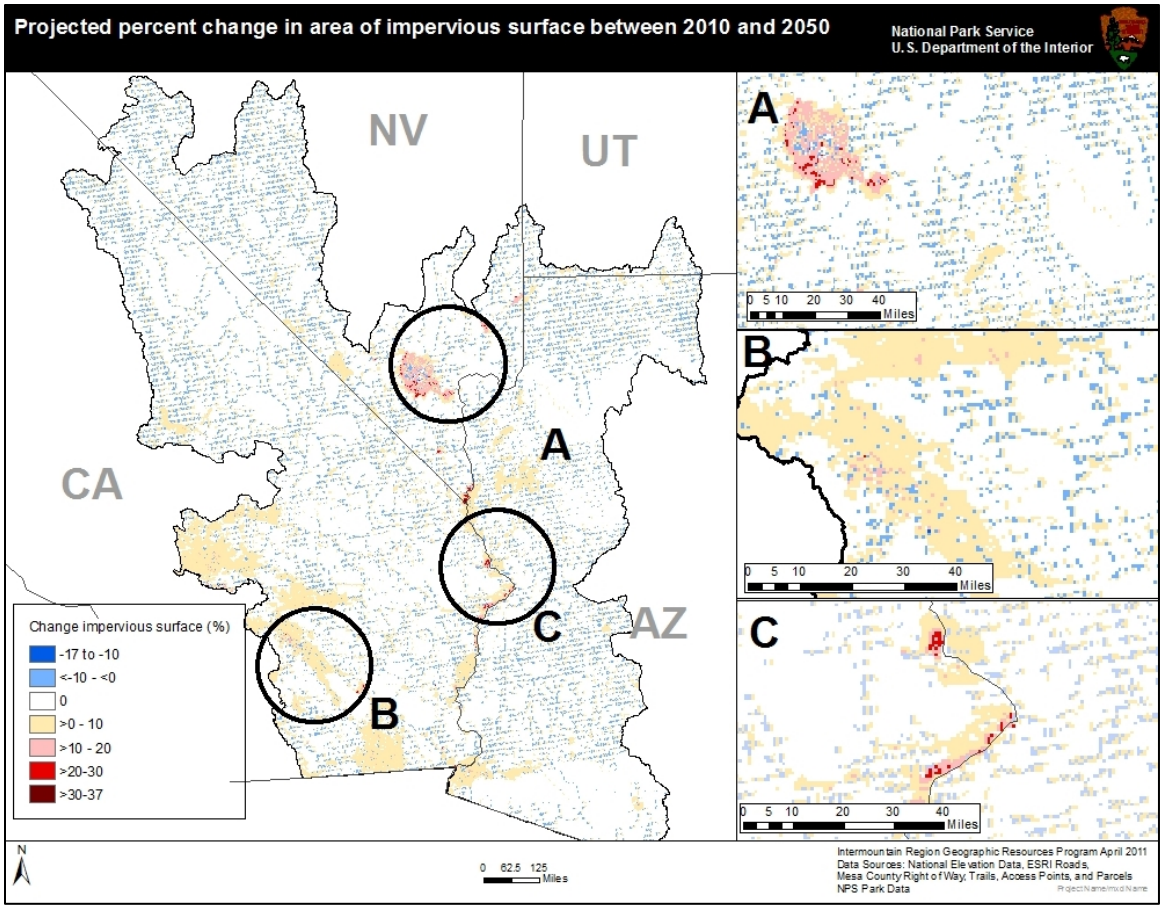


Figure 15. Projected change in the area of impervious surfaces from 2010–2050 on the basis of five scenarios developed by Bierwagen et al. (2010). Detail: (A) Las Vegas, Nevada, (B) Victorville, California, (C) Colorado River.

Table 15. Mean projected percentage of change in the area of five housing-density classes across the Mojave Desert from 2010–2050. Projections based on five scenarios developed by Bierwagen et al. (2010). Standard deviations in parentheses.

Housing-density class	Decrease	No change	Increase
Rural	8.4 (4.7)	91.6 (4.7)	0 (0)
Exurban	15.5 (2.8)	54.4 (12.2)	21.2 (22.8)
Suburban	9.3 (2.1)	43.4 (8.3)	47.3 (8.8)
Urban	0 (0)	56.3 (5.8)	43.7 (5.8)
Commercial/industrial	0 (0)	100.0 (0)	0 (0)

Projected changes in the distribution of housing-density classes varied spatially (Figure 16). We projected decreases in the extent of rural housing across the assessment area, especially from the Salton Sea south. In the latter area, we also projected an increase in suburban housing.

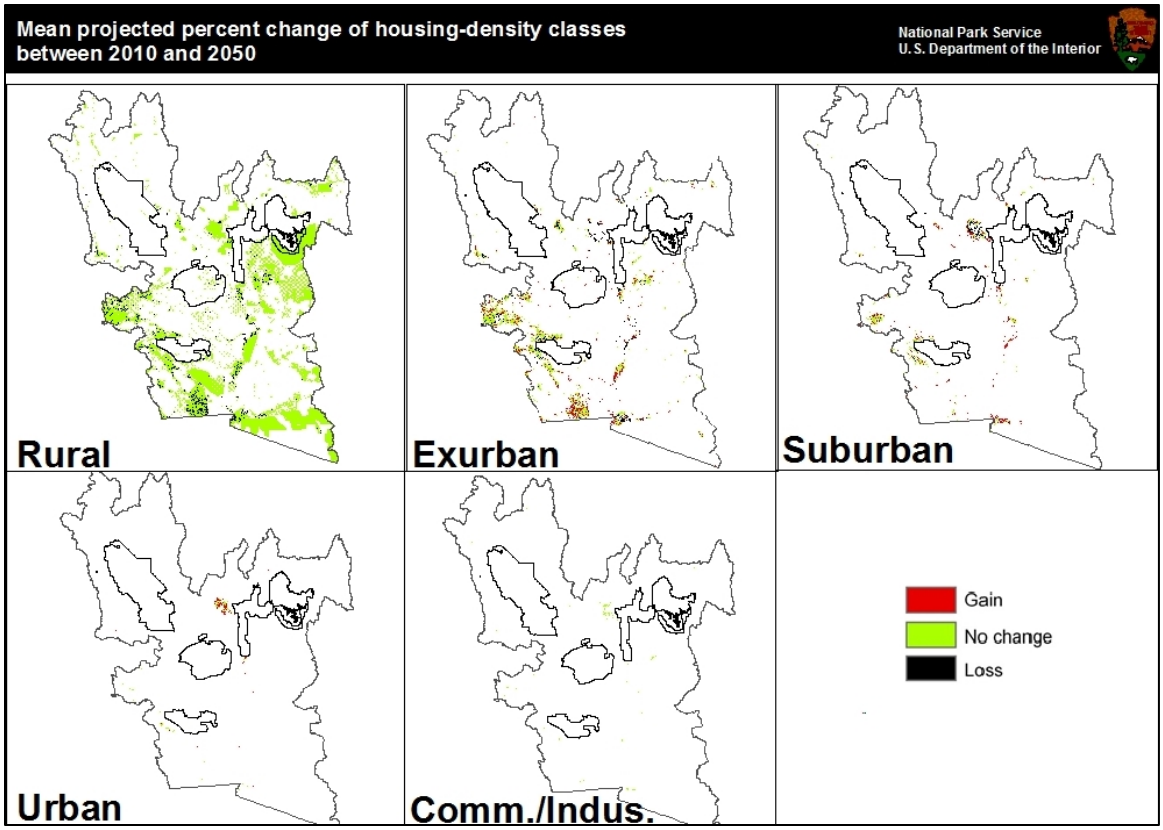


Figure 16. Projected changes in the area of five housing-density classes from 2010–2050 on the basis of scenarios developed by Bierwagen et al. (2010). Urban, <0.1 ha/housing unit; suburban, 0.1–0.68 ha/unit; exurban, 0.68–16.18 ha/unit; rural, >16.18 ha/unit. Comm. /Indus., commercial and industrial.

5.C.3. Ecological effects of land use

Mean habitat quality for vertebrates varied greatly across the study area (Figure 17). About 80% of the cells had values of 1.0, indicating no human feature within home ranges of any size. Cells with habitat quality values of 0 (4.1%) were centered around major urban and agricultural centers. About 3.3% of cells had values of 0.31–0.60, whereas 12.7% of cells had values of 0.61–0.90.

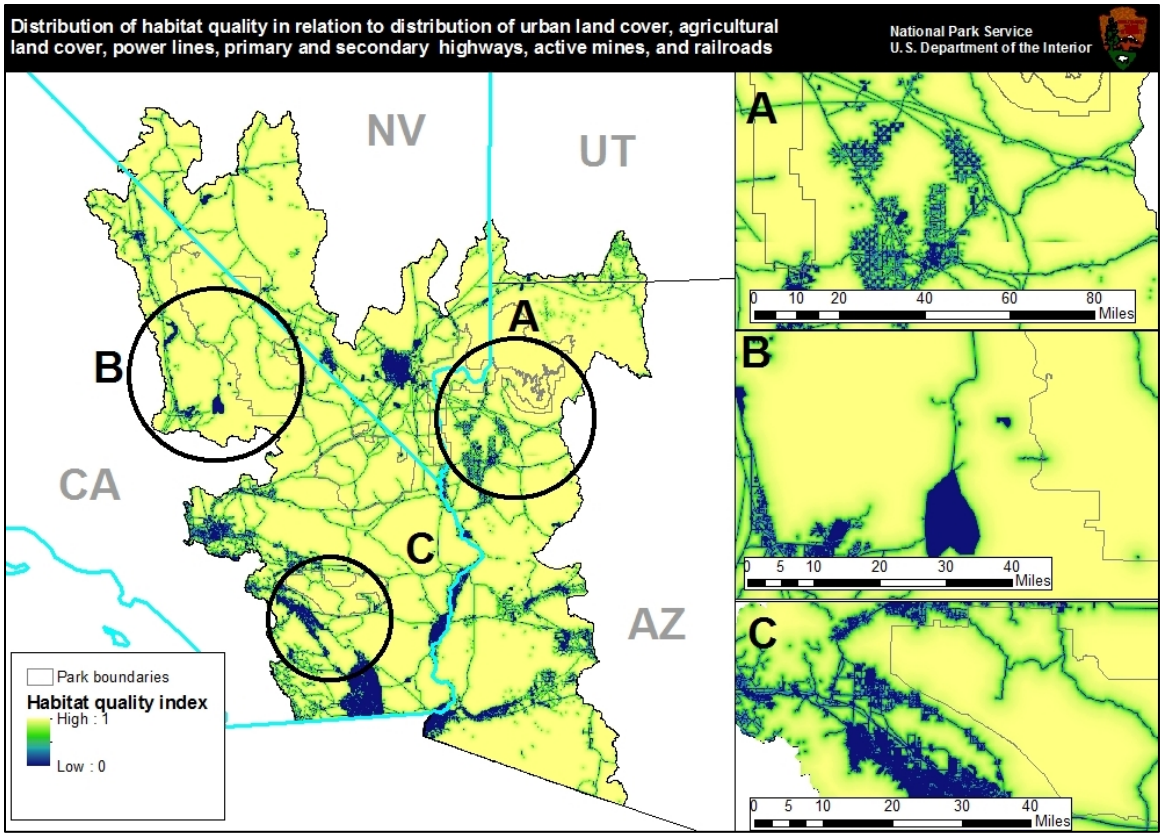


Figure 17. Mean habitat quality for vertebrates as a function of intensity of human use. Estimates of land-use intensity were based on the distribution of urban land cover, agricultural land cover, power lines, primary and secondary highways, active mines, and railroads. Habitat quality values were averaged across the home ranges of 65 vertebrate species (NLCD, TIGER, USCB, USGS).

The IRs for full and casual synanthropes were negatively associated with habitat quality as measured by distance from anthropogenic features; these species were associated with human activity (full synanthrope: $F_{1, 37} = 14.82$, $p < 0.001$; casual synanthrope: $F_{1, 37} = 4.40$, $p = 0.04$) (Figure 18). Incidences of tangential synanthropes and non-synanthropes were not significantly associated with habitat quality (tangential synanthrope: $F_{1, 37} = 0.17$, $p = 0.68$; non-synanthrope: $F_{1, 37} = 2.33$, $p = 0.14$).

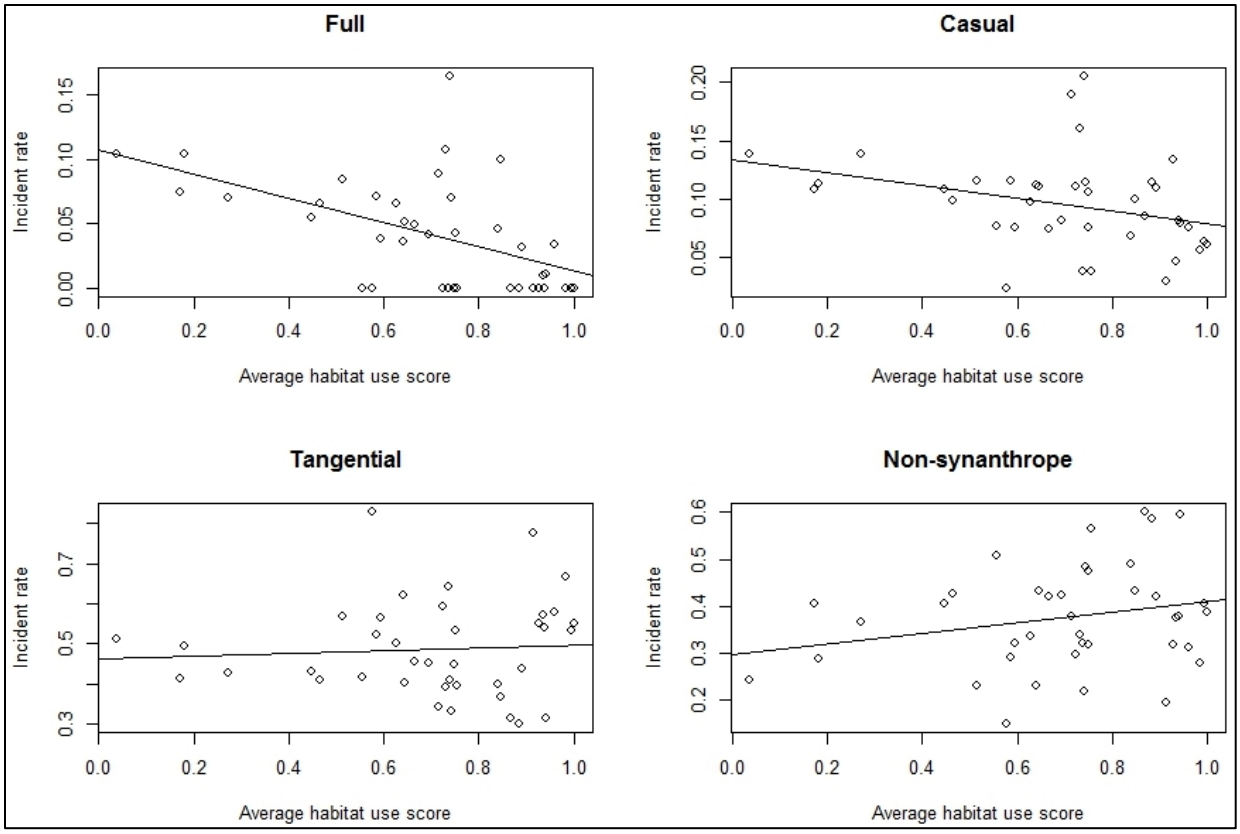


Figure 18. Relation between habitat quality within a 1-km buffer and incidence rates from 1993–2011 for full, casual, and tangential syanthropes and non-synanthropes (Johnston 2001) recorded on 39 Breeding Bird Survey routes that coincided with the assessment area.

5.C.4. Anthropogenic light

Artificial sources of light include buildings, from homes to stadiums; streetlights, bridges, and other infrastructure; and motorized vehicles on land and water. Most light comes from streetlights and commercial lights (Schwarz 2003), and has wavelengths from 589 nm (sodium) to 486 nm (mercury). Anthropogenic light can affect the behavior and health of wild animals via its effects on melatonin and endocrine hormones. For example, artificial light can disorient animals, reduce foraging and metabolic efficiency, reduce immunity, change the timing of reproduction and migration, and affect communication (Longcore and Rich 2004, Navara and Nelson 2007, Horvath 2009, Holker et al. 2010b). Artificial light also can increase the exposure of some animals to predators (Longcore and Rich 2004). At least in the short term, some predators (e.g., bats, spiders, and diurnal birds and reptiles) may benefit from increases in artificial light. Moreover, insects and migratory birds that use polarized light for orientation and navigation may be affected negatively by polarized light that is reflected from artificial surfaces, such as buildings, asphalt roads, and black plastic sheets that are used in agriculture (Horvath et al. 2009). Particulates amplify atmospheric air glow.

The NPS Night Sky team uses a method developed by Duriscoe et al. (2007) to monitor night skies. The method includes four major steps. First, a high-resolution, charge-coupled device camera

attached to a laptop computer with robotic arm is used to capture images of as much of the full horizon as possible. The images (45 or 104, depending on the camera's format) are compiled into a mosaic. Acquisition of all images usually can be completed in within 35 minutes. Second, images are calibrated to stars of known brightness, and the brightness of the sky is measured. Third, sources of light are identified. Fourth, natural and artificial sources of light are differentiated.

The NPS typically reports three measures of sky brightness. The first is the surface brightness of the zenith and the darkest and brightest portions of the sky (in mag arcsec^{-2}). Skies without artificial light tend to have brightness values no lower than 21.3–22.0 mag arcsec^{-2} (Walker 1988, Krisciunas and Schaefer 1991, Schwarz 2003, Duriscoe et al. 2007). The second is the total integrated brightness of the full sky, with the exception of stars. The third is the total integrated brightness of light domes produced by towns and cities.

Additionally, field technicians use the Bortle Dark Sky Scale to make qualitative observations of the night sky (Bortle 2001). The scale ranges from virtually no anthropogenic light (Bortle Class 1) to high amounts of anthropogenic light (e.g., inner cities; Bortle Class 9). Light levels in many national parks in the western United States correspond to Bortle Class 3 (typical of rural areas) or 4 (typical of the interface between rural and suburban areas).

In 2004 and 2005, night-sky brightness was recorded at three locations in Death Valley (Furnace Creek, Dante's View, and Rogers Peak), Lake Mead (Overton Beach, Temple Bar, and Government Wash), and Mojave National Preserve (Sunrise Rock) (Duriscoe et al. 2007). Brightness values less than 19 mag arcsec^{-2} are considered to impair human night vision considerably (Duriscoe et al. 2007). In Death Valley, the brightness of the lightest sample point was 19.3 ± 0.9 (mean ± SD) mag arcsec^{-2}, and brightness of the darkest sample point was 21.9 ± 0.2 mag arcsec^{-2}. Thus, skies in Death Valley can be considered relatively dark. In Lake Mead, the brightness of the lightest sample point was 16.6 ± 0.9 mag arcsec^{-2}, and the brightness of the darkest sample point was 21.1 ± 0.9 mag arcsec^{-2}. The light dome from the greater Las Vegas metropolitan area dominated the western portion of the sky at these three sampled locations. The brightest sample point at Sunrise Rock in Mojave National Preserve was 17.5 mag arcsec^{-2}, and the darkest sample point was 21.5 mag arcsec^{-2}.

5.C.5. Anthropogenic sound

Chronic exposure to sound has been associated with depression and aggression, decreased memory, reduced immunity, and delayed neural development in humans and other animals (Kight and Swaddle 2011, Pijanowski et al. 2011). Taxonomic groups that rely on acoustic communication, including as birds, bats, anurans, and crickets, and marine mammals, may be particularly sensitive to sound (Barber et al. 2009, Holt et al. 2011, Blackwell et al. 2013).

Mechanisms by which individual animals may respond chronic and impulsive sound also include displacement and chronic stress. In the context of anthropogenic sound, displacement refers to the distance by which or direction in which animals change their location in response to one or more sources of sound. Displacement might range from a relatively modest shift in location to exclusion from habitat (e.g., feeding grounds) (Elgar 1986, Wilkinson and Boughman 1998). Effects of displacement on survival and reproduction may vary among life stages. Nondisplacement is difficult

to interpret because it may reflect either habituation to sound (about which little is known) or tolerance of sound. Tolerance encompasses both absence of a response to sound and the necessity of enduring sound.

Chronic stress refers to long-term elevated levels of stress hormones that result from constant sources of anthropogenic sound. There is some evidence that chronic sound can elevate stress and translate into effects on fitness or survival (Barber et al. 2009, Kight and Swaddle 2011, Rolland et al. 2012). If animals are affected by sound but cannot leave a given area, their level of stress may increase, reducing their resistance to disease. Hormonal responses to stress are both short-term and long-term. Animals might experience a given sound while aggregated, but individuals' responses to sound might occur over the longer term, while animals are dispersed. Long-term responses are more likely to affect survival and reproduction.

The NPS Natural Sounds Program collects information on sound pressure level (SPL, in decibels [db]) and sound frequency (pitch; in hertz [Hz]). Sampling is conducted over 25 days. Metrics reported by the program include the median ambient sound level (L_{50}), the median ambient sound level in the absence of anthropogenic sources of sound (L_{NAT}), and the percentage of time that anthropogenic sounds are audible, with 5% considered as a reference condition (Miller 2007). The NPS and Federal Aviation Administration also define ambient sound without commercial air tours and without aircraft of any kind. Sound propagation typically decreases as the density and structural complexity of vegetation increases. Therefore, NPS differentiates sound levels in different land-cover types as defined by the National Land Cover Database.

Within the Mojave Desert and the Mojave Desert Network, major sources of anthropogenic sound include transportation networks; construction, maintenance, and operation of solar and wind energy (Lovich and Ennen 2011); motorized recreation, and military operations. Air traffic accounts for a considerable proportion of anthropogenic sounds in national parks (Lynch et al. 2011). Transportation networks in the greater Las Vegas area and southern California can elevate ambient sound levels late at night in parts of Death Valley, Joshua Tree, Lake Mead, and Mojave National Preserve (Lynch et al 2011). Edwards and Nellis Air Force Bases, March Air Reserve Base, Camp Pendleton, the China Lake Naval Air Weapons Station, and the Marine Corps Yuma Air Station frequently fly missions over the Mojave Desert, and a new Marine Corps Air Ground Combat Center has been proposed in Twentynine Palms, California.

Ambient sound levels and the percentage of time that sounds are audible were measured in Mojave National Preserve in 2005 and in Death Valley in 2008 (Table 16). In all cases, sound levels exceeded reference conditions.

Table 16. The percentage of time that anthropogenic sounds were audible, the median ambient sound level in the absence of anthropogenic sources of sound (Lnat), and the median ambient sound level (L50) in two Mojave Desert Network Parks. Land cover defined by the National Land Cover Database.

		Percentage of time audible			Ambient sound level	
Park	Land-cover type	Aircraft	All anthropogenic sound	No anthropogenic sound	L_{nat} (dB)	L_{50} (dB)
Death Valley	Developed	2.5	80.6	19.4	32.7	38.3
	Barren	16.3	21.7	78.3	26.5	20.8
	Shrubland	18.3	20.7	79.3	15.5	17.8
	Forest	10.1	37.7	62.3	23.9	28.0
Mojave National Preserve	Mountain juniper shrubland	36.3	36.3	63.7	26.6	28.6
	Joshua tree shrubland	19.4	38.5	61.5	21.7	23.8
	Mountain creosote shrubland	13.4	15.3	84.7	24.2	25.1

5.D. Data gaps and confidence in the assessment

5.D.1. Land use and land cover change

Our assessment indicated that for many areas within the Mojave Desert that affect the Mojave Desert Network, publicly available data on roads, a primary cause of fragmentation of natural land cover, are incomplete. It would be feasible to update the TIGER data in two stages. First, heads-up digitization and digital imagery could be used to update the data. Second, NPS staff with local knowledge could use this experience and, as necessary, field visits to evaluate the accuracy of the stage-one outputs.

Trombulak et al. (2010) outlined some of the limitations of human-footprint models. Because the models generally are based on existing data, one cannot draw inferences at resolutions finer than those at which features were mapped. If land-cover data are derived from Landsat images, the finest resolution of the model will be 30 m. Data for a given input must be available for the full extent of the analysis area. Higher-resolution data for a subset of the area cannot be incorporated, and either the native or resampled resolution of all data sources must be the same. Additionally, there often is a lag between the year of analysis and the most recent year for which data are available (e.g., National Land Cover Data for 2011 were released in 2014). Moreover, the most recent year may differ among data inputs.

Assessment of ecological effects of human presence or actions on one or more species is especially challenging when many human activities occur over large areas or long time periods. Even when biological information is ample, projections of effects are complicated by variation in responses to human activities and to stressors created by those activities among, for example, species, sexes, life stages, and behaviors. Furthermore, physiological or behavioral responses may have negligible, sublethal, or lethal effects, and individual-level effects may not propagate to the population. We

define stressors as entities or processes that have negative effects on individuals or populations (e.g., reduce probabilities of survival, reproduction, or persistence) within a given location and time period. Thus, stressors are defined relative to response variables. An activity is not a stressor in and of itself. Instead, some attribute of the activity is a stressor. For example, domestic cats and dogs that are maintained in human settlements may prey on or affect the behavior of wild animals such as songbirds and small mammals. Also, although ecological effects of multiple human activities or features typically are assumed to be additive, the effects may be additive, multiplicative, synergistic, or antagonistic.

5.D.2. Anthropogenic light

Levels of anthropogenic light have not been measured in Grand Canyon–Parashant, Joshua Tree, and Manzanar. There is considerably uncertainty about whether responses of wild animals or plants to various forms of anthropogenic light are sufficiently predictable to develop defensible management guidelines.

5.D.3. Anthropogenic sound

Beyond the dearth of data on ambient sound and responses of wildlife to sound across the Mojave Desert Network, empirical data for assessment of population-level effects of anthropogenic sounds via robust statistical models are limited. Responses to sound often are assessed on the basis of observed changes in behavior of individuals, but these may not affect population growth rates or vital rates. The potential non-lethal effects of sound are quite difficult to estimate. Also, sources of sound that in isolation may not affect vital rates may in the aggregate affect a population's status and trend.

Chapter 6. Air Quality

6.A. Rationale for and information provided by the assessment

There are both regulatory and biological reasons why air quality is relevant to an assessment of the condition of a given park's natural resources. Air quality in national parks is regulated in part by the Clean Air Act, one purpose of which is "to preserve, protect, and enhance the air quality in national parks, national wilderness areas, national monuments, national seashores, and other areas of special national or regional natural, recreational, scenic, or historic value." Before the 1950s, the Mojave Desert largely was uninhabited, concentrations of pollutants were low, and air quality was high. During the last 70 years, urban populations within and upwind of the Mojave Desert have grown considerably, and major irrigation projects created extensive agricultural areas. As a result of these shifts in land use, concentrations of pollutants have increased and air quality has decreased. The Environmental Protection Agency (EPA) maintains National Ambient Air Quality Standards for six pollutants: carbon monoxide, lead, nitrogen dioxide, ozone, particulates, and sulfur dioxide.

The Clean Air Act defines two classes of protection for natural areas, class I and class II. Class I areas include national parks larger than about 25 km^2 (6000 ac) that were authorized before 7 August 1977. Joshua Tree has a class I designation. Congress "declare[d] as a national goal the prevention of any future, and the remedying of any existing, impairment of visibility" in class I areas that resulted from human sources of air pollution. Other parks in the Mojave Desert Network have a class II designation. Air quality in class I areas is regulated more heavily than in class II areas.

Assessment of air quality provides information on visibility and the potential condition or competitive ability of some species of plants. Because the natural turbidity of the air in the Mojave Desert is relatively low, even low concentrations of particles can reduce visibility substantially (Trijonis et al. 1988, Malm 1999).

Responses to ozone vary among plant species. A primary mechanism by which ozone affects vegetation is oxidation within leaf stomata (Tingey and Hogsett 1985). Laboratory and field experiments indicate that within a given species, damage to plants is roughly proportional to their cumulative ozone exposure during the growing season. NPS (2003, 2006) classified a number of plant species in the Mojave Desert as sensitive to ozone. Species designated as sensitive occur in Death Valley, Joshua Tree, Lake Mead, and Mojave National Preserve (Table 17).

Although largely outside the scope of this document, assessment of air quality also is relevant to the health of human visitors and to human populations outside the parks. As noted in chapter two, ground-level ozone, wet and dry deposition of nitrogen and sulfur, and visibility and particulate matter were identified as vital signs for the Mojave Desert Network.

Table 17. Plant species in the Mojave Network identified by the National Park Service (2003, 2006) as sensitive to ozone. 1, observed in a given park.

Species	Common name	Family	Death Valley	Joshua Tree	Lake Mead	Mojave	Grand Canyon–Parashant
Ailanthus altissima (non-native)	tree of heaven	Simaroubaceae	1	–	–	–	–
Amelanchier alnifolia	Saskatoon serviceberry	Rosaceae	1	–	–	–	1
Apocynum cannabinum	Indian hemp	Apocynaceae	–	–	1	1	–
Artemisia douglasiana	California mugwort	Asteraceae	1	–	–	–	–
Artemisia ludoviciana	silver wormwood	Asteraceae	1	1	1	–	–
Pinus ponderosa	ponderosa pine	Pinaceae	1	1	1	–	1
Populus tremuloides	quaking aspen	Salicaceae	–	–	–	–	1
Prunus virginiana	chokecherry	Rosaceae	–	–	1	–	–
Rhus trilobata	skunkbush	Anacardiaceae	–	1	1	1	1
Robinia pseudoacacia	black locust	Fabaceae	1	1	–	1	–
Salix gooddingii	Gooding's willow	Salicaceae	1	1	1	1	1
Sambucus mexicana	blue elderberry	Caprifoliaceae	1	1	–	1	–

6.B. Data and methods

The NPS Inventory and Monitoring Program provides estimates of ground-level ambient ozone, wet deposition of nitrogen and sulfur compounds, and visibility on the basis of air quality monitors operated by NPS, the EPA, states, and other entities. The data are interpolated across the conterminous United States, and the observed or interpolated value closest to the center of each park is reported.

The NPS estimates current ozone concentrations as the five-year average of the annual fourth-highest eight-hour ozone concentration. Ozone concentrations in areas between measurement stations are interpolated. If the five-year average is ≥ 76 ppb then the condition is considered poor. Five-year average concentrations of 61–75 ppb (concentrations > 80% of the standard) are considered moderate, and concentrations < 61 ppb are considered good.

The NPS also estimates ozone concentrations relative to assumed responses of plants. The W126 index measures cumulative ozone concentrations over each year's peak three-month concentration period (growing season) (Lefohn et al. 1988). W126 for each day is calculated as the sum of sigmoidally weighted hourly concentrations from 08:00–20:00 local time. The daily values are summed to monthly values, and the peak three-month period for each year is identified. The index

typically varies among years as weather varies. Values of 7–13 ppm/hr. (moderate) are assumed to allow growth of tree seedlings, whereas values of 5–9 ppm/hr. are assumed to be low enough that foliage is not damaged (Heck and Cowling 1997, EPA 2007). W126 values > 13 ppm/hr. and < 7 ppm/hr. are considered indicative of poor and good condition, respectively.

The NPS interpolates values for atmospheric deposition of nitrogen and sulfur compounds on the basis of wet deposition because dry-deposition data generally are not available. The NPS estimates annual wet deposition within the conterminous United States by multiplying the nitrogen or sulfur concentrations in precipitation by the average annual precipitation from 1981 through 2010 [derived from the Parameter-elevation Regressions on Independent Slopes Model (PRISM]).

Estimates of total (wet and dry) natural background deposition of either nitrogen or sulfur are 0.25 kilograms per hectare per year in the western United States, which is about equal to wet deposition of 0.13 kg/ha/yr. Wet deposition < 1 kg/ha/yr., 1–3 kg/ha/yr., and > 3 kg/ha/yr. are considered by NPS to be indicative of good, moderate, and poor condition, respectively.

The NPS measures visibility on the basis of a haze index, which is derived from estimates of light extinction and reported in deciviews (dv). As the haze index increases, visibility decreases. Visibility is estimated as the five-year average current visibility minus the estimated average natural visibility, where average visibility is the mean of the visibility observations from the 40th through 60th percentiles. Visibility measures are interpolated as necessary. Visibilities < 2 dv above natural conditions are considered good. Visibilities 2–8 dv above natural conditions and > 8 dv above natural conditions are considered moderate and poor, respectively.

6.C. Assessment results

There is a strong seasonal cycle of air quality in the Mojave Desert. Air quality typically is lowest during summer, especially near the mountain passes that connect the Mojave Desert with upwind sources of urban, industrial, and agricultural pollutants. In autumn and winter, regional air quality is good to excellent, although particulates may be trapped in Las Vegas Valley and other basins. In this section, we address air quality in five of the six parks included in the NRCA. The NPS does not currently report air quality data for Grand Canyon–Parashant.

6.C.1. Sources of reduction in air quality

Anthropogenic pollution that is generated within the Mojave Desert tends to be local, concentrated around population centers or trapped in isolated basins. The principal urban and industrial air pollution sources within the Mojave Desert are its two major population centers, the Las Vegas area and the southwestern boundary of the Mojave, where groups of small cities and unincorporated communities are clustered. These clusters include Antelope Valley north of Soledad Pass; Victor Valley north of Cajon Pass; and the cluster of communities northeast of San Gorgonio Pass and just west of Joshua Tree (Morongo Valley, Yucca Valley, Joshua Tree, and Twentynine Palms). A number of rural activities also can affect regional or local air quality in the Mojave Desert. The power plants clustered around Las Vegas are major contributors to that area’s ozone production.

Anthropogenic air pollution also flows into the Mojave Desert from the Los Angeles metropolitan area and from urban and agricultural areas in the San Joaquin Valley. Flows from the Los Angeles area are carried over Soledad, Cajon and San Gorgonio (Banning) Passes. Although the Sierra Nevada and Tehachapi Mountains block most air flows from the San Joaquin Valley into the Mojave Desert, Walker and Tehachapi Passes allow some flow of pollutants from the populated areas upwind. Sources of particulate and gaseous pollution in Southern California include surface and marine transportation networks, petroleum production and refining, manufacturing, and domestic emissions from a population of around 20 million.

Large wildfires in the Pacific Coast Ranges, Sierra Nevada, or Cascade Range occasionally spread smoke plumes across the Mojave. Such events are sufficiently rare that they do not substantially affect long-term air quality.

Most of the interior western United States is exposed to persistent, low concentrations of particulates and gases, including carbon monoxide, ozone, and nitrogen oxides (NO_x) (Fiore et al. 2002, Jaegle et al. 2003, Jaffe et al. 2004), that originate in Asia (VanCuren and Cahill 2002, VanCuren 2003). Asian emissions of most pollutants, including nitrous oxides, are expected to increase (Zhang et al. 2012), but it appears that sulfur dioxide (SO_2) emissions are stabilizing as China implements sulfur controls (Lu et al. 2010).

6.C.2. Ozone

From 2008–2012, air quality on the basis of the average ozone concentration criterion was poor in Death Valley (76.8 ppb), Joshua Tree (84.3 ppb), Manzanar (88.3 ppb), and Mojave National Preserve (79.0 ppb) and moderate in Lake Mead (73.1 ppb) (NPS ARD 2014a). On the basis of the W126 ozone criterion, air quality was poor in all Mojave Network parks for which data were provided by NPS [Death Valley (22.5 ppm/hr.), Joshua Tree (23.8 ppm/hr.), Lake Mead (15.1 ppm/hr.), Manzanar (36.5 ppm/hr.), and Mojave National Preserve (21.7 ppm/hr.)] (NPS ARD 2014a). Long-term trends due to changes in emissions or air quality regulations and annual variation in weather are both evident in multiyear W126 data (Table 18). From April through October 2014, ozone exceeded the National Ambient Air Quality Standard (currently 75 ppb as a daily maximum 8-hour average) on one day at the Park Village station in Death Valley, on 41, 14, and 20 days at the Black Rock, Cottonwood Canyon, and Pinto Wells stations in Joshua Tree, and on 5 days at the Kelso Mountains Station in Mojave National Preserve (www.nature.nps.gov/air/Monitoring/exceed.cfm).

Most ozone (precursor emissions) in Death Valley originates locally. The southern Sierra Nevada prevents efficient transport of ozone to the Death Valley area; Manzanar also is substantially sheltered from ozone transport. From late autumn through early spring, most ozone in the Mojave Desert is generated through natural processes, and ozone levels peak around noon. In Death Valley, for example, soil denitrification when temperatures are high may supply nitrogen oxides and enable local ozone production (McCalley and Sparks 2009). During late spring and summer there is strong ozone transport into the Mojave from upwind source areas, and ozone levels are highest in late afternoon. Models that were validated by aerial measurements over California and the eastern Pacific

Ocean also indicate that Asian sources increase ozone levels in the Mojave Desert (Huang et al. 2010).

Table 18. Three-month W126 ozone concentration index (parts per million per hour) for parks in the Mojave Desert Network. Cells with – indicate missing data. Mesquite, Nevada is the ozone-monitoring location closest to Grand Canyon–Parashant.

Park	Survey months	2007	2008	2009
Death Valley	Jan-Feb-Mar	4.2	3.4	3.7
	Feb-Mar-Apr	8.3	9.6	7.6
	Mar-Apr-May	21.4	15.8	11.4
	Apr-May-Jun	30.6	20.9	13.6
	May-Jun-Jul	32.5	24.6	13.8
	Jun-Jul-Aug	28.0	23.9	15.3
	Jul-Aug-Sep	20.0	19.9	12.7
	Aug-Sep-Oct	15.3	10.2	9.2
	Sep-Oct-Nov	6.5	4.2	3.7
	Oct-Nov-Dec	2.9	1.3	1.5
Joshua Tree	Jan-Feb-Mar	5.5	–	5.1
	Feb-Mar-Apr	11.1	–	11.3
	Mar-Apr-May	18.5	–	17.9
	Apr-May-Jun	23.6	–	20.1
	May-Jun-Jul	20.1	–	17.5
	Jun-Jul-Aug	13.6	–	15.2
	Jul-Aug-Sep	7.0	–	11.8
	Aug-Sep-Oct	6.1	–	9.6
	Sep-Oct-Nov	5.4	–	5.4
	Oct-Nov-Dec	2.7	–	3.6
Mesquite, Nevada	Jan-Feb-Mar	1.7	1.5	1.8
	Feb-Mar-Apr	4.0	6.3	4.3
	Mar-Apr-May	6.7	11.3	6.2
	Apr-May-Jun	8.9	13.7	7.0
	May-Jun-Jul	8.2	11.4	6.3
	Jun-Jul-Aug	7.8	8.2	5.8
	Jul-Aug-Sep	6.3	6.0	4.5
	Aug-Sep-Oct	5.2	3.7	3.0
	Sep-Oct-Nov	2.7	1.9	1.4
	Oct-Nov-Dec	1.0	0.7	0.8
Mojave National Preserve	Jan-Feb-Mar	–	–	–
	Feb-Mar-Apr	–	–	–

Table 18 (continued). Three-month W126 ozone concentration index (parts per million per hour) for parks in the Mojave Desert Network. Cells with – indicate missing data. Mesquite, Nevada is the ozone-monitoring location closest to Grand Canyon–Parashant.

Park	Survey months	2007	2008	2009
Mojave National Preserve (continued)	Mar-Apr-May	–	–	–
	Apr-May-Jun	–	–	–
	May-Jun-Jul	29.4	33.8	15.9
	Jun-Jul-Aug	30.8	39.3	16.6
	Jul-Aug-Sep	30.5	36.3	19.0
	Aug-Sep-Oct	–	31.8	19.8
	Sep-Oct-Nov	–	–	–
	Oct-Nov-Dec	–	–	–

6.C.3. Nitrogen and sulfur

Air quality from 2008–2012 on the basis of the average wet deposition criterion was good in Death Valley, Joshua Tree, and Manzanar for ammonium (NH_4), nitrate (NO_3), sulfate (SO_4), total nitrogen, and total sulfur (all < 1 kg/ha/yr.). In Lake Mead, nitrate (1.9 kg/ha/yr.) and sulfate (1.1 kg/ha/yr.) were moderate, but ammonium and total nitrogen and sulfur were good. Nitrate in Mojave National Preserve was moderate (1.0 kg/ha/yr.), but all other components of wet deposition were good (NPS ARD 2014b).

The nitrogen compounds in clouds and dry air in the Mojave Desert that come from southern California and the Central Valley are dominated by combustion-derived nitrous oxides and by ammonia (NH_3) from upwind automotive catalysts, farms, dairies, and feed lots. Nitrogen compounds are delivered to soils in the Mojave Desert in rainwater (wet deposition) and by direct surface exchange of gases and settling of aerosols (dry deposition). The amount of dry nitrogen deposition in the Mojave Desert is much greater than the amount of wet deposition. Dry, warm conditions drive ammonium nitrate from its neutralized aerosol form into gaseous ammonium and nitrate ions (Neuman et al. 2003, Tonnesen et al. 2007).

Modeling of total nitrogen deposition in the western United States identified a corridor of strong nitrogen-rich air extending from Southern California through the Mojave Desert to Las Vegas and into Utah (Fenn et al. 2003). Death Valley and Manzanar appeared to be sheltered by the Sierra Nevada from nitrogen deposition (Fenn et al. 2003). Emissions of nitrous oxides from Asia, arriving in the western United States as long-lived peroxyacetyl nitrate, likely contribute to the level of nitrogen that is deposited.

In California, emissions of nitrous oxides, including those in the Los Angeles area and the San Joaquin Valley, have fallen substantially since 2002 and are projected to continue dropping through 2020. Emissions of nitrous oxides in Las Vegas are much less than those in California, but also are projected to decline due to ozone control requirements. However, the California Air Resources Board does not project considerable reduction in ammonia emissions in the near future.

It is unlikely that vegetation in the Mojave Desert Network will be affected by exposure to sulfur dioxide (SO_2). SO_2 is measured at two sites in the Mojave Desert, Victorville and Trona. Because Victorville and Trona are close to local sources of SO_2 (motor vehicles in Victorville, industrial production in Trona), concentrations at those sites likely are among the highest in the Mojave Desert. Nevertheless, concentrations at Victorville and Trona generally are less than 5 parts per billion by volume (ppbv), although concentrations in some hours approach 10 ppbv (CARB 2009). These concentrations are below the thresholds above which plants may be injured, which ranges from 30 to 100 ppbv (Hindawi 1970).

Atmospheric sulfuric acid and sulfate at the concentrations measured in the Mojave Desert are unlikely to affect soils in the region, but may impair visibility. Sulfate concentrations, and thus sulfur deposition, in the Mojave Desert likely will continue to be above the default natural target levels set by NPS, but are unlikely to increase in the near future. Most sulfur emissions in California come from fuels, and levels have been relatively low since the 1990s. Analysis of the combustion-related materials traveling with Asian dust in hundreds of IMPROVE samples suggested that sulfate is a common constituent (VanCuren 2003). Asian sulfate likely is a large component of the total sulfate measured in Death Valley.

6.C.4. Visibility and primary particles

Air quality from 2008–2012 on the basis of the visibility criterion was moderate at Death Valley (5.5 dv), Joshua Tree (6.8 dv), Lake Mead (4.1 dv), Manzanar (7.7 dv), and Mojave National Preserve (6.0 dv) (NPS ARD 2014c).

Prescribed fire in upwind areas, particularly in the southern Sierra Nevada, can generate particulates and reduce visibility. Agricultural dust, especially from fallow or plowed fields, is generated in Antelope Valley in the far western Mojave and along the Virgin and Colorado Rivers in the far eastern Mojave. Mining dust is generated in isolated areas within the Mojave Desert in California, including Trona (salt) and open-pit mines at Boron (borate) and Mountain Pass (rare earth). Combustion emissions from industrial facilities and energy production and dust from disturbed ground at energy facilities can reduce air quality. Off-road vehicle use at military training facilities (for example, Fort Irwin and Twentynine Palms) can generate dust, as can use of the wide network of primitive roads and trails throughout the Mojave.

In areas of the Mojave with few local sources of dust, including many uplands within the Mojave Desert, Asian dust often is a major fraction of the total atmospheric dust burden unless strong local winds are driving local dust emissions. From 1990–2000, the most recent period reported, the proportion of Asian fine dust (PM2.5) in IMPROVE samples from Death Valley was roughly equal to that of local fine dust (A. VanCuren personal communication).

The largest single dust source in the United States is the bed of Owens Lake, which was exposed by diversion of water for urban use. Dust from the lake bed has been identified as a major source of dust at Manzanar (Reheis et al. 2002, GBUAPCD 2008). However, large, regional Owens Lake dust events are rare (GBUAPCD 2008).

In undisturbed areas of the Mojave Desert, much of the soil surface is stabilized by either soil crusts, a coarse pebble surface layer, or widespread stable vegetative cover (Belnap 2001). Where soil crusts are disturbed, mineral dust can loft more readily. The large dune systems in the Mojave Desert, such as the Kelso Dunes in Mojave National Preserve and the Mesquite Dunes in Death Valley, are open to dust transport from multiple directions and are positioned in areas where drifting sediment is dropped as winds encounter barriers or exit confined troughs. The presence of the multiple dune systems in the Mojave Desert is evidence that wind-blown dust is natural in the ecosystem.

6.D. Data gaps and confidence in the assessment

Most of the air quality measures for the Mojave Desert Network parks are based on interpolation across extensive areas rather than on field data. Few air-quality monitoring stations are located in the Mojave Desert, let alone in the Mojave Desert Network parks. There currently is one ozone monitoring site in Death Valley (Park Village, California) and three ozone monitoring sites in Joshua Tree (Black Rock, Cottonwood Canyon, and Pinto Wells, California). Ozone is measured with a portable continuous analyzer during the summer at Mojave National Preserve. Wet deposition of nitrogen (ammonium, nitrate, and nitrogen) and sulfur (sulfate and sulfur) also is measured in Death Valley, and both dry and wet deposition of nitrogen and sulfur (dry deposition includes nitric acid, nitrate, ammonium, sulfate, and sulfur dioxide) are measured in Joshua Tree. The reliance on interpolation leads us to gauge confidence in the assessment as moderate.

There is some uncertainty in the extent to which concentrations of sulfur compounds in the Mojave are generated by natural sources. Creosote is the strongest known plant source of dimethyl sulfide (DMS) (Jardine et al. 2010). DMS is a biogenic product usually associated with marine phytoplankton, and is a major source of non-sea-salt sulfate over the ocean. Creosote DMS may increase the uncertainty of the natural background SO_4 concentration targets assigned by EPA and IMPROVE to class I areas. Creosote emissions may highlight plant and soil processes that have not been recognized as sources of aerosol sulfate measured in remote sites in the Mojave and Sonoran deserts.

The projected future sulfur fraction of Asian emissions also is highly uncertain. Combustion of coal in Asia coal is a major global source of sulfur. Use of coal in Asia is increasing, although some countries are adopting sulfur controls in an effort to limit acid rain and protect human health. Models based on projected emissions controls indicated that SO_2 emissions in China will decrease by about 16% by 2015 (Wang et al. 2013).

Neither the current nor the future effects emissions from Las Vegas on nitrogen deposition at Lake Mead and Grand Canyon–Parashant are known. Ammonia trends also are uncertain.

6.E. Sources of Expertise

The National Park Service's Air Resources Division oversees the national air resource management program for the NPS. Together with parks and NPS regional offices, they monitor air quality in park units, and provide air quality analysis and expertise related to all air quality topics. For current air quality data and information for this park, please visit the NPS Air Resources Division website at https://www.nps.gov/subjects/air/index.htm.

Chapter 7. Environmental associations with fire probability and size

7.A. Rationale for and information provided by the assessment

The number of fires in the Mojave Desert is increasing (Brooks and Berry 2006), which is inconsistent with the National Park Service's aim of restoring and maintaining natural fire regimes. The increase has been attributed to the spread of non-native invasive plant species (Brooks and Pyke 2000, Balch et al. 2013) and changes in climate (Abatzoglou and Kolden 2011). Non-native invasive annual grasses (e.g., *Bromus rubens*, *B. tectorum*, *Schismus* spp.) colonized the Southwest about a century ago (Hunter 1991) and can become dominant in many parts of the region (Brooks 2000). A growing human population and activities such as road construction, urbanization, and off-highway vehicle use have facilitated the expansion of non-native annual grasses (Lovich and Bainbridge 1999, Brooks and Pyke 2000, Brooks and Berry 2006). These plants compete with native species (Brooks 2000) and are associated with increases in the number, intensity, and size of fires (Whisenant 1990, Brooks and Matchett 2006, Balch et al. 2013). Because the non-native grasses are both highly flammable and have a high probability of colonization after fire, native plant communities often are converted into those dominated by non-natives (Brooks et al. 2004a).

As precipitation in the Southwest decreases and temperatures increase, the moisture content of native vegetation may decrease while fine-fuel loads composed of drought-tolerant, non-native invasive grasses remain high (Brooks and Esque 2002). Additionally, the distributions of some native plant species may shift to higher elevations, which, in conjunction with increased fire frequency, could modify the vegetation communities of the Mojave Desert (Hughson et al. 2011, Balch et al. 2013).

The Mojave Desert historically had sparse vegetation, low fuel biomass, and a low frequency of fire, particularly at relatively low elevations (Brown and Minnich 1986, Brooks and Pyke 2000, Brooks and Matchett 2006). In contrast to some tree species in heavily forested ecosystems that evolved in the presence of fire and are dependent upon fire for seed germination (Keeley 1991), many native plant species in the Mojave are poorly adapted to fire and recover slowly from fire (Brown and Minnich 1986, Brooks 1999).

To inform expectations and management of contemporary fire risk, we explored recent and projected future changes in fire probability, including changes in land cover and the timing and amount of precipitation, within the Mojave Desert Network. We used two different approaches to assess fire probability on the basis of fire ignitions from 1992 through 2011. First, we quantified the probability of fire occurrence by relating whether locations burned to a suite of environmental variables (Castedo-Dorado et al. 2011, Hawbaker et al. 2013). Second, we quantified the suite of factors that might cause a fire to become relatively large (> 20 ha) or lead to a high probability of a large fire.

7.B. Data and methods

7.B.1. Assessment area

Our assessment area included five management units within the Mojave Desert Network: Death Valley, Joshua Tree, Lake Mead, Mojave National Preserve, and Grand Canyon–Parashant. We did

not include Manzanar in our analyses given its small size. We restricted our analyses to the perimeter of each park given the limited control that the National Park Service has over activities that occur outside of the parks.

7.B.2. Data

We developed two data sets, one for fire occurrence and one for large fires, by associating the point of origin of a fire with six environmental or social variables: topographic roughness, aspect, distance to roads, land cover, phenology, and cause of the fire. For fires that occurred from 1992 through 2011, we obtained fire occurrence data from a national database that included location of ignition, size, cause, and year in which the fire occurred (i.e., fire year) as reported by federal, non-federal, and interagency systems (Short 2013). To account for location error in the fire occurrence data and to associate point-location data with environmental attributes that extended beyond the point of origin, we summarized environmental data at a larger spatial resolution (described below) before associating them with the fire data.

To enable comparisons between locations where fires were present and absent, we developed a shapefile of pseudo-fire absence (hereafter, pseudo-absence) point locations. We randomly located pseudo-absences within the boundaries of each park, but excluded areas that were defined as water in the National Land Cover Database (NLCD) (Fry et al. 2011). We also excluded a 14-m buffer (7 m on either side), approximating the typical width of a road, around lines indicating the locations of primary and secondary roads (U.S. Census Bureau 2012). We assigned a fire year to each pseudo-absence location by sampling from the distribution of known fire locations (hereafter, true fires) by year in each park. To ensure that locations of pseudo-absences were not the same as those of true fires in the same year, we assumed that true fires were circular (this was a necessary simplification for modeling; we realize that in reality, fires are not circular) and used the corresponding radius to buffer around the true-fire locations. We removed any pseudo-absence locations that fell within the radius during the same year as a true fire.

Following a literature review, we derived six spatial data layers (environmental variables) to include in models of fire occurrence and large fires as described below (Table 19). From a 90-m digital elevation model (DEM), we derived topographic roughness and aspect (northness). High topographic roughness has been associated with both fire occurrence and the size of large fires (Dickson et al. 2006), whereas north aspects tend to have higher fuel moisture and vegetation productivity than other aspects (Rollins et al. 2002). We used a Euclidean distance raster layer, which we calculated in a geographic information system (GIS), to obtain the distance from point locations of fires to all primary and secondary roads (U.S. Census Bureau 2012) at 90-m resolution. Road density has been positively associated with fire occurrence (Brooks et al. 2004b), whereas the size of fires increases as the distance from roads increases (Dickson et al. 2006, Syphard et al. 2007).

Table 19. Variables used to model probabilities of fire across five Mojave Network parks (Death Valley, Grand Canyon–Parashant, Joshua Tree, Lake Mead, and Mojave National Preserve) from 1992–2011. All continuous variables were standardized to have a mean of zero and unit variance.

Name	Type	Description
Park[1]	Categorical	Death Valley, Joshua Tree, Lake Mead, Mojave National Preserve, or Grand Canyon–Parashant
Cause of fire[2]	Categorical	Lightning, human, or unknown
Topographic roughness	Continuous	Topographic roughness calculated from the standard deviation of slope
Distance from road	Continuous	Euclidian distance from the nearest road
Ecological zone	Categorical	Ecological zones defined on the basis of elevation and dominant vegetation (low elevation shrub, middle elevation shrub, high elevation woodland, desert montane)
Maximum NDVI	Continuous	Maximum value of the normalized difference vegetation index (NDVI) in the year prior to the fire
Date of maximum NDVI	Continuous	Julian day of the year prior to the fire corresponding to the maximum annual NDVI

[1] Park was included as a random effect in models, whereas all other variables were included as fixed effects

[2] Included in the model of probability of large fires but not the model of fire occurrence

We classified land cover as one of four ecological zones that corresponded to the major upland fuel types in the Mojave Desert as defined by Brooks and Matchett (2006): desert montane, high elevation woodland, middle elevation shrubland, and low elevation shrubland (Table 20). We obtained two measures of phenology (hereafter, *phenometrics*) at 1-km resolution from the USGS Earth Resources Observation and Science Center (phenology.cr.usgs.gov): the maximum value of the Normalized Difference Vegetation Index (NDVI) (hereafter, *maximum NDVI*) and the Julian day of the year on which the maximum NDVI occurred (*date of maximum NDVI*). The maximum NDVI is correlated with the maximum level of photosynthetic activity. Phenometrics and precipitation typically are strongly correlated (Wang at el. 2003) and reflect both precipitation and the response of vegetation, in the form of biomass, to precipitation (Wallace and Thomas 2008). We originally used 250-m resolution phenometric data but, within the Mojave Desert, 1-km resolution data were more complete. Given the temporal availability of the 1-km phenometrics (1989–2011) and the years for which fire data were available (1992–2011), we selected all fires that occurred from 1992 through 2011 to allow for a one-year lag in phenometric values (i.e., values during the year prior to each fire). Although we explored the inclusion of several different phenometric lag-effects, including phenometric values during the fire year, one year prior to the fire year, and cumulative values from two and three years prior to the fire year, we selected the phenometric values for one year prior to the fire year on the basis of a pilot analysis and use of model selection criteria (methods described below). The results of the pilot analysis were similar to studies suggesting that precipitation and corresponding plant growth in the year prior to the fire year is directly associated with fuel conditions (Knapp 1998, Brooks and Matchett 2006).

Table 20. Mean elevation (standard deviation in parentheses) and the dominant plant species in each of four ecological zones in Death Valley, Grand Canyon–Parashant, Joshua Tree, Lake Mead, and Mojave National Preserve as described by Brooks and Matchett (2006).

Ecological zone	Mean elevation (m)	Dominant plant species
Low elevation shrub	476 (347)	*Larrea tridentata*, *Ambrosia dumosa*
Middle elevation shrub	1107 (338)	*Coleogyne ramosissima*, *Larrea tridentata*, *Yucca brevifolia*, *Achnatherum* spp., *Pleuraphis* spp.
High elevation woodland	1638 (350)	*Juniperus osteosperma*, *Pinus monophylla*, *Artemisia* spp., *Ceanothus* spp., *Purshia* spp.
Desert montane	2110 (340)	*Pinus ponderosa*, *P. longaeva*

Because the resolution of our data ranged from 1 km to 90 m, we resampled the original data layers to a consistent resolution of 90 m. We used focal statistics (i.e., a neighborhood of cells surrounding fire and non-fire locations) to summarize each of the six environmental variables. We calculated all focal statistics for a 900 m radius around the centroid of each cell with a circular moving window operation in the GIS. In a pilot analysis, we found that focal statistics for moving windows of three different extents (300, 600, 900 m) were not qualitatively different. We used the largest extent to reduce noise discrepancies between data on points of origin of fires and the locations of burn perimeters and to match the extents of large fires and focal statistics.

We used a spatially and temporally balanced method to include pseudo-absence locations in the fire occurrence data. We combined all true fires and an equal number of pseudo-absences into a single data set (King and Zeng 2001) and then extracted the values of all environmental variables. After we removed fires for which data on environmental variables did not exist, the fire occurrence data included 1,747 true fires and 1,747 pseudo- absences. The data did not include fire cause and fire size because these covariates were only available for the true fires.

For the large-fire model, we divided the true fire data into large fires (> 20 ha or 10% of all true fires; $n = 165$) versus small fires (< 1 ha or 80% of all true fires) and removed all fires of intermediate size (1–20 ha), which allowed us to better discriminate between small and large fires. Although the majority of wildfires in the Mojave Desert are suppressed regardless of size (Joshua Tree Fire Captain A. Kendall, personal communication), the 20 ha threshold for large fires reflects fires with a relatively high probability of growing even larger. Fires larger than 20 ha are more likely than smaller fires to be affected by environmental variables that act over extensive areas (Neuenschwander et al. 2000, Dickson et al. 2006). Because there were far more small fires than large fires, we randomly selected 165 small fires from the full set of true fires.

7.B.3. Statistical modeling

We used mixed-effects multiple logistic regression (Gelman and Hill 2007) and multi-model inference (Burnham and Anderson 2002) to assess whether the probability of fire occurrence or a large fire was associated with a candidate set of covariates (i.e., fixed effects) that we identified from the literature (Table 19). Prior to analysis, we standardized all continuous covariates to have a mean of zero and unit variance. We included park as a subject-level random effect to account for

heterogeneity in fire occurrence among parks (Faraway 2006). To account for any residual spatial autocorrelation in the fire occurrence data, we computed the variance-covariance matrix of the fixed-effects with the asymptotically consistent sandwich estimator (Diggle et al. 1994). We implemented all analyses with PROC GLIMMIX in SAS (v.9.3; SAS Institute, Cary, North Carolina) and the R statistical environment (R Development Core Team 2012).

We used Akaike's Information Criterion (AIC) to evaluate the extent to which our models approximated the data and to rank and compare the relative strength of association of covariates with response variables (Burnham and Anderson 2002). For both the fire occurrence and large-fire models, we used an intercept-only model (including the random effect for park) and the difference in AIC values (ΔAIC) to evaluate how well each model fit the data (Anderson 2008). We considered models with ΔAIC values that were > 4.0 AIC units from the intercept-only model to approximate the data well (Anderson 2008). We calculated and summed AIC weights across all possible models (n = 256 for fire occurrence, n = 1,024 for large fires) in which a given covariate occurred (Burnham and Anderson 2002). We used these cumulative AIC weights (w_+) to evaluate the strength of evidence that a given covariate was associated with the response variable, and considered values ≥ 0.50 strong evidence for a positive or negative association with the response variable (Barbieri and Berger 2004). In addition, we calculated the model averaged parameter estimate

$$\left(\tilde{\beta}\right)$$

and unconditional standard error for each covariate to accommodate bias and model selection uncertainty (Burnham and Anderson 2002).

Before implementing our fire probability models, we assessed the correlations between all continuous covariates with Pearson's correlation coefficient and collinearities among covariates with variance inflation factors (VIFs). We removed covariates that had correlations > 0.6 and inspected the VIFs of the reduced data set to ensure that all remaining covariates had a VIF < 10.0 (Neter et al. 1996).

7.B.4. Fire probability mapping

To derive spatially explicit surfaces (maps) of the fire occurrence and large-fire model outputs, we implemented our final statistical models in the GIS and classified the output of each model into quintiles. The outputs presented in these maps are estimates of the relative probability of at least one fire occurring in a given pixel from 1992–2011. To compare fire probabilities between years with different precipitation and vegetation growth and to evaluate differences in fire probability over time, we selected multiple years that represented different phenometric values for mapping different probabilities of fire occurrence and large fires, excluding years with a high number of missing pixels. In addition, we mapped areas where the estimated probabilities in the 80th percentile of the fire occurrence and large fires coincided to indicate areas with the highest probabilities of fire.

7.C. Assessment results

From 1992–2011, 1,747 fires occurred in the five parks. The total area burned was 157,246 ha, with a mean burned area per fire of 90 ha (SD 930). The largest fire was the lightning-caused Wildhorse

Fire in Mojave National Preserve (part of the Hackberry Complex), which occurred in 2005 and burned 25,672 ha. The proportion of park area that burned from 1992–2011 was greatest in Grand Canyon–Parashant (0.21) and least in Death Valley (< 0.01) (Table 21). The number of fires and area burned each year varied among years and did not appear to follow a trend (Figure 19). The greatest area burned in 2005 (74,820 ha) and the greatest number of fires occurred in 2006 (185). Across years, the majority of fires (1,629) occurred between May and September (Figure 20). Most fires (1,337) were lightning caused, followed by 307 human caused and 103 with unknown cause.

Table 21. Number of fires and area and proportion of park burned in Death Valley, Grand Canyon–Parashant, Joshua Tree, Lake Mead, and Mojave National Preserve from 1992–2011. The true number of fires is greater than the number of fires included in the models because true fires for which values of any covariates were missing were removed from the data. Data from Short (2013).

Park	True number of fires	Park area burned (ha)	Proportion of park burned
Death Valley	36	11,712	< 0.01
Grand Canyon–Parashant	1,037	88,493	0.21
Joshua Tree	119	16,245	0.05
Lake Mead	218	7,423	0.01
Mojave National Preserve	337	33,372	0.05
Total	**1,747**	**157,245**	**–**

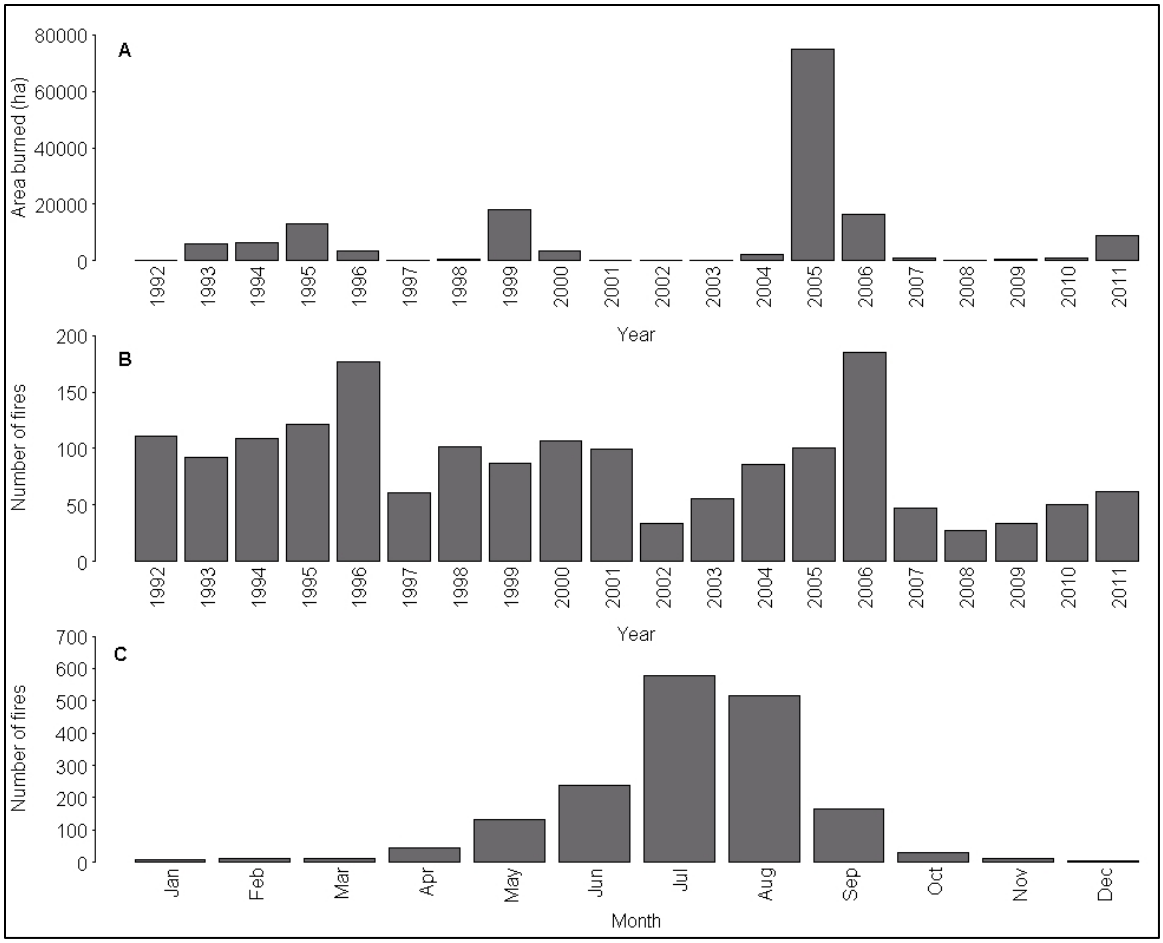


Figure 19. Area burned per year (A), number of fires per year (B), and cumulative monthly number of fires (C) in five national parks in the Mojave Desert Network. Data include 1,747 fires for which data on environmental variables were available. Data from Short (2013).

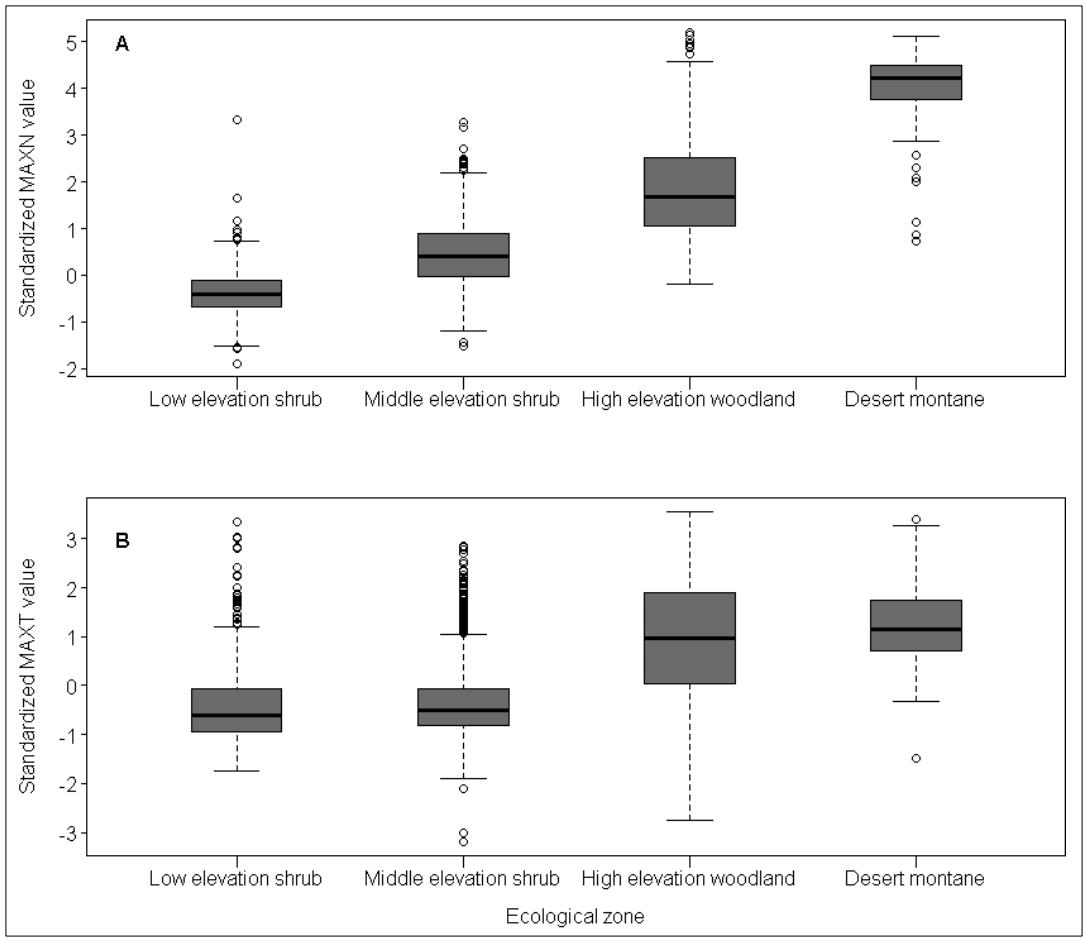


Figure 20. Relation between the elevational gradient of four ecological zones in the Mojave Desert as described by Brooks and Matchett (2006) and (A) maximum normalized difference vegetation index (NDVI) (MAXN) and (B) date of maximum NDVI (MAXT). The ends of the boxes are the upper and lower quartiles, the bar in the middle of the box is the median, the bars at the end of the dashed lines are the maximum and minimum values (excluding outliers), and the dots indicate outliers. The median elevation of each ecological zone increases from left to right across the x-axis.

The proportional distribution of ecological zones varied among the parks and from east to west (Table 22). In the central Mojave Desert, for example, Death Valley and Joshua Tree have similar percentages of middle elevation shrub (53% in Death Valley and 62% in Joshua Tree) and low elevation shrub (39% in Death Valley and 34% in Joshua Tree). The latitude of Mojave National Preserve is between that of Death Valley and Joshua Tree, but the park is further east and has the highest percentage of middle elevation shrub among the five parks (77%). Further east, Lake Mead has the highest percentage of low elevation shrub (60%) and the second highest percentage of high elevation woodland (15%). Grand Canyon–Parashant is located just outside of the Mojave Desert and its distribution of ecological zones is quite different from that of the other four parks. Grand Canyon–Parashant has the highest percentage of high elevation woodland (72%) and desert montane (2%).

Table 22. Proportional cover of ecological zones within five national parks in the Mojave Desert. Ecological zones were described by Brooks and Matvhett (2006). The proportional cover of desert montane in Death Valley, Lake Mead, and Mojave National Preserve was too small to estimate accurately.

Ecological zone	Death Valley	Grand Canyon–Parashant	Joshua Tree	Lake Mead	Mojave National Preserve
Low elevation shrub	0.39	0.01	0.34	0.60	0.16
Middle elevation shrub	0.53	0.25	0.62	0.25	0.77
High elevation woodland	0.08	0.72	0.04	0.15	0.07
Desert montane	–	0.02	–	–	–

The number of fires and the cause of fires differed among ecological zones (Table 23). For example, the desert montane ecological zone covered less than 1% of the five parks, but had the greatest density of fires from 1992–2011 (0.0078/ha). The ecological zone with the largest proportion of area burned during the study period was the high elevation woodland (1.88×10^{-1} ha/ha), whereas zone with the lowest proportion of area burned was the low elevation shrubland (7.64×10^{-3} ha/ha). The greatest number of human-caused fires occurred in the low elevation shrubland and the greatest number of lightning-caused fires occurred in the high elevation woodland. Among the ecological zones, maximum NDVI and date of maximum NDVI decreased as elevation decreased (Figure 20).

Table 23. Number and cause of fires in each ecological zone in Death Valley, Grand Canyon–Parashant, Joshua Tree, Lake Mead, and Mojave National Preserve from 1992–2011. Ecological zones were described by Brooks and Matchett (2006). Fire density measured as number of fires per hectare.

Ecological zone	Number of fires included in models	Area burned (ha)	Fire density	Area burned (ha/ha)	Ratio of human- to lightning-caused fires
Low elevation shrub	207	7,705	2.05×10^{-4}	7.64×10^{-3}	3.07
Middle elevation shrub	579	48,788	3.55×10^{-4}	2.99×10^{-2}	0.35
High elevation woodland	862	100,136	1.62×10^{-3}	1.88×10^{-1}	0.04
Desert montane	99	616	7.76×10^{-3}	4.83×10^{-2}	0.14
Total	**1,747**	**157,245**	**–**	**–**	**–**

7.C.1. Probability of fire occurrence

The fire occurrence model was 198 AIC units lower than the null model, indicating that this model approximated the data quite well (Burnham and Anderson 2002). This model indicated a strong (w_+ = 1.00) negative relation between the probability of fire occurrence and distance from roads (Table 24). The probability of fire occurrence also increased as maximum NDVI (0.99) or topographic roughness (0.53) increased. There was a strong positive relation (0.99) between the probability of fire occurrence and the desert montane ecological zone.

Table 24. Cumulative Akaike's Information Criterion weights (w+), model-averaged regression coefficients (β), and unconditional standard errors (in parentheses) for variables included in the fire occurrence model. The intercept term includes low elevation shrubland as a constant for comparison.

Variable	w_+	$\left(\underset{\beta}{\simeq}\right)$
Intercept	na	-0.40 (0.14)
Distance from road	1.00	-0.71 (0.18)
Maximum NDVI	1.00	0.29 (0.05)
Desert montane	0.99	0.93 (0.19)
Topographic roughness	0.53	0.03 (0.05)
Aspect (northness)	0.47	0.03 (0.03)
High elevation woodland	0.36	0.04 (0.12)
Middle elevation shrub	0.31	-0.01 (0.12)
Date of maximum NDVI	0.28	-0.003 (0.01)

Across the study period, the average annual maximum NDVI was 0.0125. We used the fire occurrence model to derive maps for a year with higher than average (2010; 0.0129) and lower than average (2001; 0.0121) maximum NDVI values (and precipitation) in the preceding year. However, because many of the covariates, such as distance from road and ecological zone, vary little among years, and there was little variation in annual maximum NDVI, there was little difference in the probability of fire occurrence between years with low and high maximum NDVI (Table 25). Thus, we used maximum NDVI in 2010 to estimate and map the average probability of fire occurrence in each park during 2011 (i.e., a one-year lag between high NDVI and the fire year). Because our models were based on a one-year lag between the phenometric values and the year for which fire occurrence was modeled, this map represents the probability of fire occurrence in 2011 (Figure 21). This model indicated that the average probability of fire occurrence was 0.62 in Grand Canyon–Parashant, 0.50 in Mojave National Preserve, and < 0.50 in Lake Mead, Joshua Tree, and Death Valley (Table 25).

Table 25. Mean (SD) probability of fire occurrence. Estimates are based on a year in which maximum NDVI in the preceding year was lower than average (2001) or higher than average (2010).

Park	Low maximum NDVI	High maximum NDVI
Death Valley	0.34 (0.15)	0.34 (0.15)
Grand Canyon–Parashant	0.62 (0.10)	0.62 (0.11)
Joshua Tree	0.39 (0.11)	0.40 (0.11)
Lake Mead	0.40 (0.14)	0.40 (0.14)
Mojave National Preserve	0.51 (0.09)	0.50 (0.08)

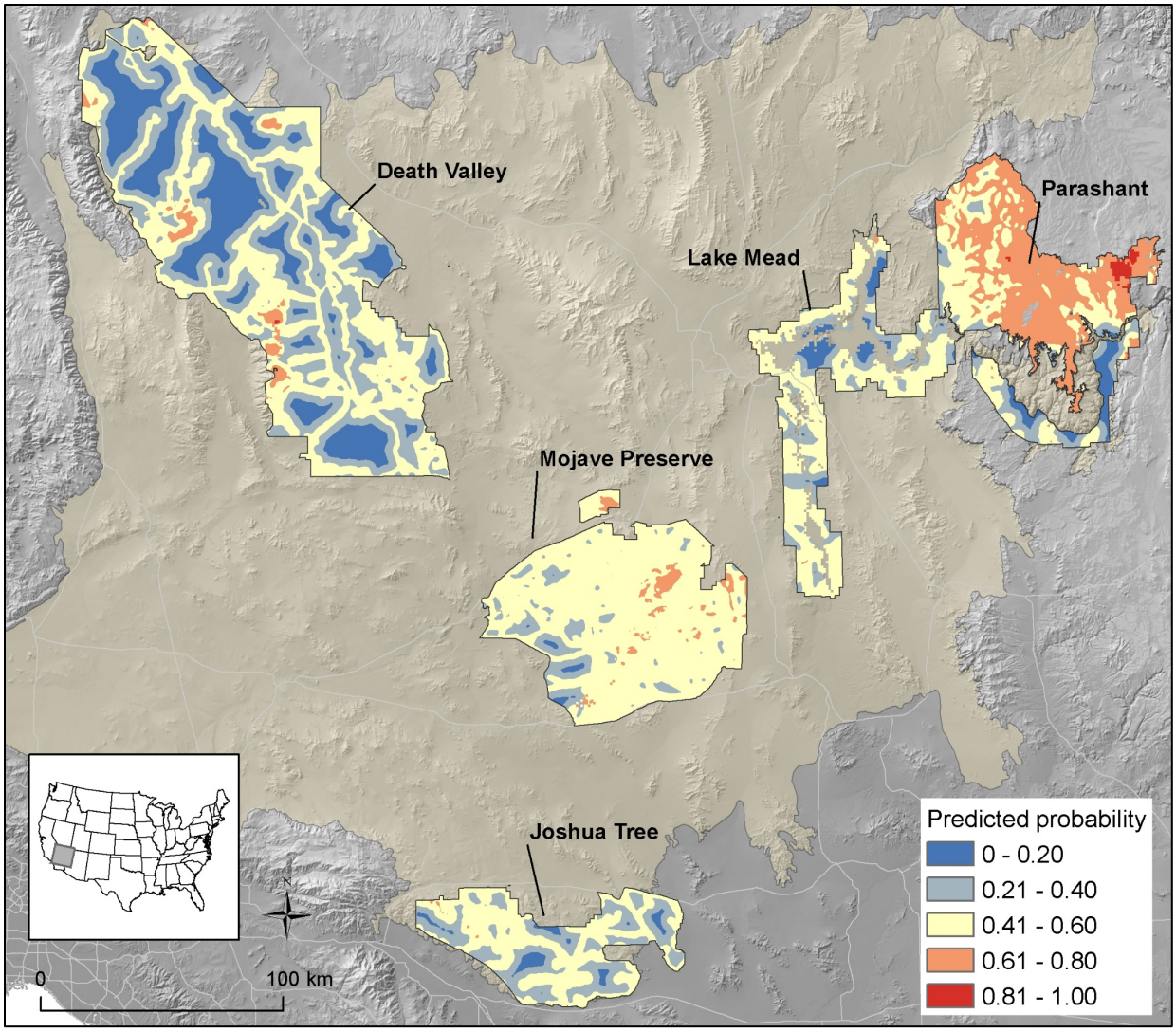


Figure 21. Relative probability of fire occurring at least once in a given pixel from 1992–2011. Estimates were based on a fire occurrence model for 2011 and values of the normalized difference vegetation index (NDVI) for 2010, which were greater than the average from 1992–2011. Extent of the Mojave Desert (beige) defined on the basis of the U.S. Environmental Protection Agency's level IV ecoregions.

7.C.2. Probability of a large fire

The large fire model approximated the data well compared to the null model ($\Delta AIC < 76$). Ignition by lightning was strongly ($w_+ = 1.00$) associated with large fires, and lightning-ignited fires usually were larger than human-caused fires (Table 26). As the date of the maximum NDVI in the preceding calendar year became earlier, the probability of large fires increased. High topographic roughness (0.96) and increasing distance from roads (0.59) also were strongly and positively associated with the probability of a large fire. The only ecological zone that was strongly and positively associated with large fires was middle elevation shrubland (0.91).

Table 26. Cumulative Akaike's Information Criterion weights (w+), model averaged regression coefficients (β), and unconditional standard errors (in parentheses) for variables included in the model of probability of large fires. The intercept term includes the low elevation shrubland ecological zone and human caused fires as constants for comparison.

Variable	w_+	$\left(\underset{\beta}{\simeq}\right)$
Intercept	na	-1.85 (0.72)
Lightning caused	1.00	2.02 (0.51)
Date of maximum NDVI	1.00	-0.55 (0.12)
Topographic roughness	0.96	0.44 (0.18)
Middle elevation shrub	0.91	0.88 (0.48)
Distance from road	0.59	0.30 (0.31)
High elevation woodland	0.44	0.24 (0.49)
Maximum NDVI	0.32	0.03 (0.10)
Desert montane	0.31	-0.07 (0.45)
Unknown cause	0.30	0.15 (0.38)
Aspect (northness)	0.28	0.02 (0.09)

Models of large-fire probability as a function of annual precipitation indicated that the probability of large fires was greatest in a year following one with early maximum NDVI (e.g., high probability of large fires in 2006, following maximum NDVI in March of 2005; by contrast, the maximum NDVI in 1999 occurred in August) (Figure 22). Across management units, the maximum predicted probability of a large fire occurring during the study period, given NDVI values in 2005, was 0.99 (0.52 ± 0.30, mean ± SD). The maximum predicted probability given NDVI values in 1999 was 0.99 (0.37 ± 0.26). Regardless of the year modeled, the probability of large fires was greatest in Grand Canyon–Parashant, followed by Mojave National Preserve, Joshua Tree, Death Valley, and Lake Mead. For all parks except Death Valley, the mean probability of a large fire was 20% higher in 2006 than 2000 (Table 27).

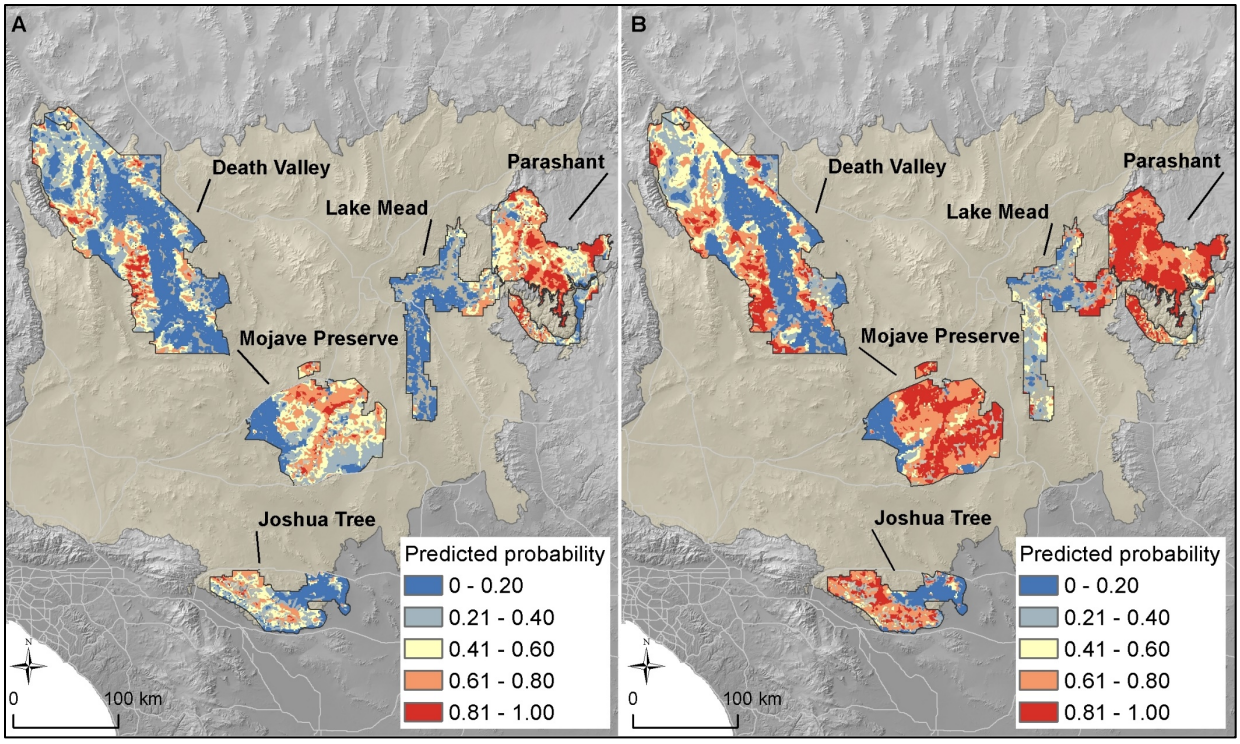


Figure 22. Predicted probability of a large fire occurring at least once in a given pixel from 1992–2011. Estimates were based on a fire occurrence model for 2000, a year in which the maximum normalized difference vegetation index (NDVI) in the preceding year was later than average (A), and for 2006, a year in which the maximum NDVI in the preceding year was earlier than average. Extent of the Mojave Desert (beige) defined on the basis of the U.S. Environmental Protection Agency's level IV ecoregions.

Table 27. Mean (SD) probability of a large fire. Estimates are based on a year in which maximum NDVI in the preceding year was earlier than average (2000) or higher than average (2006).

Park	Late date of maximum NDVI	Early date of maximum NDVI
Death Valley	0.30 (0.24)	0.40 (0.28)
Grand Canyon–Parashant	0.62 (0.20)	0.81 (0.12)
Joshua Tree	0.35 (0.23)	0.51 (0.30)
Lake Mead	0.27 (0.24)	0.45 (0.26)
Mojave National Preserve	0.45 (0.23)	0.66 (0.27)

By comparing areas with high probabilities of both fire occurrence and large fires (Figure 23), we found that fire probability was greatest in Grand Canyon–Parashant, followed by Mojave National Preserve. Relatively small areas with high probability of both fire occurrence and large fires occurred in Death Valley, Joshua Tree, and Lake Mead.

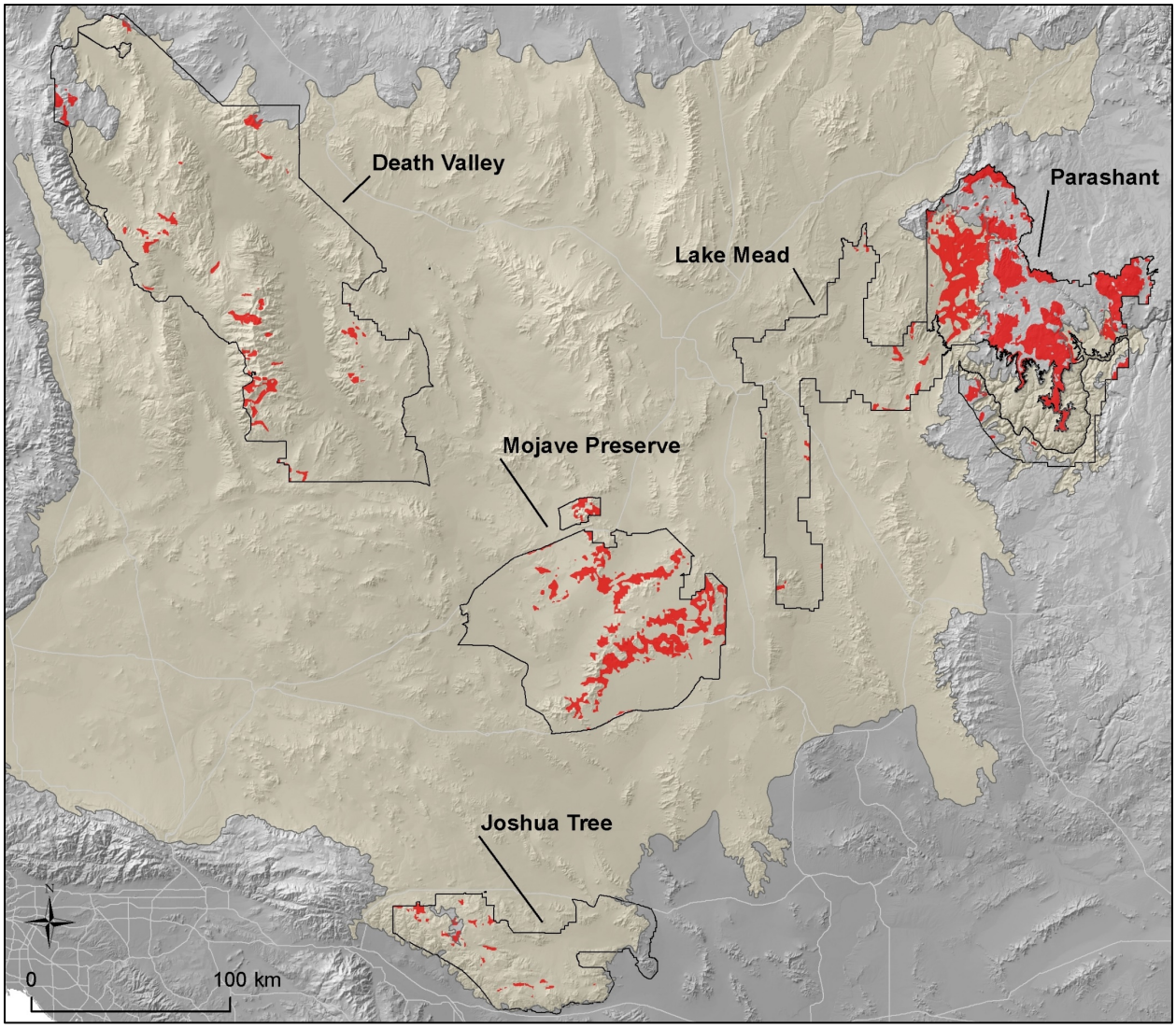


Figure 23. Areas with a high probability (80th percentile) of both fire occurrence and a relatively large fire (red). Results are based on a year with a high probability of fire occurrence (2011) and a year with a high probability of large fires (2006) relative to other years from 1992–2011. Extent of the Mojave Desert (beige) defined on the basis of the U.S. Environmental Protection Agency's level IV ecoregions.

7.C.3. Inferences and applications

Our results revealed patterns and potential drivers of fire occurrence within the past two decades that can inform assessments of future probability of fire across the Mojave Desert. We found that covariates representing phenometrics derived for the year preceding the fire, ecological zone, distance from roads, fire cause, and topographic roughness describe variation in the probability of both fire occurrence and large fires across this ecosystem. There were differences in the relative strength of association between a given variable and fire occurrence versus probability of a large fire.

Above-average vegetation growth occurred in years with high precipitation (1992, 1998, 2005, and 2010). Areas with high maximum NDVI in a given year had a high probability of fire occurrence in the following year. Locations where the maximum NDVI occurred relatively early in the calendar

year (e.g., March or April as opposed to August or September) were associated with a high probability of large fires in the following year. In these areas, plant growth was relatively low in the following spring given relatively low precipitation in the autumn (Beatley 1974, Miao et al. 2012) and, by extension, a relatively long period of dry conditions (see next section). Slow decomposition of non-native invasive grasses, which have a higher NDVI in wet years than native grasses (Bradley and Mustard 2006), contribute to increased fuel loads and higher number of fires (Knapp 1995, Brooks 1999, Brooks et al. 2004a). Phenometrics and precipitation are often highly correlated (Wang et al. 2003), and we found that certain phenometric data may be a surrogate for coarser-resolution measures of precipitation in the Mojave Desert.

Our model results indicated that ecological zone was associated with both the probability of fire occurrence and the probability of a large fire, although different ecological zones were strongly associated with each response variable. The desert montane ecological zone was positively associated with fire occurrence, whereas the middle elevation zone was positively associated with large fires. Like Brooks and Matchett (2006), we found that the highest proportion of fires occurred in the desert montane ecological zone despite its limited percent cover of vegetation (see next section).

Despite the relatively high proportion of fires near roads, such fires rarely were large. The latter pattern may reflect factors controlling fire occurrence or behavior, such as fuels, weather, and topography (Finney 2005), which also can vary among ecological regions (Syphard et al. 2007). Ignitions adjacent to roads typically are human caused (Syphard et al. 2007, Balch et al. 2013), and a higher probability of fires in these areas may be due in part to the association between human activity and occurrence of non-native invasive plants (Hawbaker et al. 2013), which in turn increases fuel loads (Bradley and Mustard 2006). Nevertheless, the data we compiled indicated that large fires generally occurred in relatively remote areas, which has been observed throughout the Mojave Desert (Brooks and Matchett 2006) and in other ecosystems in the western United States (e.g., Dickson et al. 2006, Hawbaker et al. 2013).

From 1992–2011, the largest fires in the Mojave Desert Network were caused by lightning as opposed to humans or unknown causes. Similar to the patterns noted for the 1980s and 1990s by Brooks and Esque (2002), we observed clusters of small, human-caused fires near roads and large, lighting-caused fires in more remote areas. Moreover, Brooks and Matchett (2006) observed that patterns of fire cause were congruent with their delineation of ecological zones: lightning-caused fires were more common in high elevation woodland than other ecological zones. In our study, which was based on a similar derivation of ecological zones, the majority of human-caused fires occurred in low elevation shrubland and the majority of lightning-caused fires occurred in high elevation woodland.

Our models revealed a strong positive relation between topographic roughness and both fire occurrence and large fires, although this relation was stronger for large fires. This result is consistent with other studies that have found that the number of fires does not vary as a function of slope (Heyerdahl et al. 2001), but fire size can increase as topographic roughness increases by bringing flames nearer to upslope fuels (Whelan 1995). The positive association of topographic roughness (or its proxies) and fire occurrence has been attributed to higher flammability at lower elevations due to

differences in temperature and relative humidity (Iniguez et al. 2008). Large fires are associated with topographic roughness through increased fire spread rate on steep slopes (Finney 2005) and the difficulty in suppressing fires in rough terrain (Dickson et al. 2006).

As climate changes, temperature and precipitation patterns in the Southwest are expected to shift and become more dynamic and could modify dominant land cover throughout the region (Lenihan et al. 2003, Archer and Predick 2008). The number and size of fires could increase as precipitation (Brooks and Esque 2002, Brooks and Matchett 2006, Abatzoglou and Kolden 2011), the proportion of dead and dry native biomass increases (Hereford et al. 2006), and the persistence of drought-tolerant, non-native invasive plants changes little (Brooks and Esque 2002). Although future changes in land cover are difficult to predict, expansion of grassland and a reduction of woody species (Lenihan et al. 2003, Balch et al. 2013) and increases in the abundance of non-native annual grasses (Archer and Predick 2008) have the potential to increase fire probability throughout the Mojave Desert. With or without major shifts in land cover, coincident increases in temperature and decreases in precipitation are likely to increase the length of the fire season and the sizes of fires (Keeley et al. 2011).

7.D. Data gaps and confidence in the assessment

Our confidence in this assessment is relatively high, especially given that we quantified uncertainty in the outputs of our analyses. On the basis of our assessment, we suspect that periods of high fire probability in the Mojave Desert Network are related to increases in herbaceous growth and fuel loads that occur in conjunction with unusually wet periods (Brown and Minnich 1986, Knapp 1998, Brooks and Matchett 2006, Greenville at al. 2009).

We believe that the occurrence of fires in the desert montane ecological zone may reflect dominance of ponderosa pine (*Pinus ponderosa*) and bristlecone pine (*P. longaeva*) and the position of this ecological zone in the east-central Mojave Desert, where elevations are relatively high and lightning strikes relatively common (Brooks and Matchett 2006). The relation between the middle elevation ecological zone and large fires may be affected by the abundance of non-native invasive grasses in this zone (Brooks and Matchett 2006). The latter grasses have the potential to increase fuel loads and continuity (Davies and Nafus 2013).

Aspect was the only covariate in our models that had weak relations with both fire occurrence and the probability of large fires. We anticipated that aspect would be strongly associated with both response variables because fuels on north-facing slopes tend to have relatively high moisture content and relatively low flammability (Rollins et al. 2002, Iniguez et al. 2008). Although some previous research found that a greater proportion of fires occur on south-facing slopes than on slopes with other aspects (Taylor and Skinner 1998, Heyerdahl et al. 2001, Syphard et al. 2008), other studies found either no relation (Iniguez et al. 2008, Castedo-Dorado et al. 2011) or positive relations between fire occurrence and both north and south aspects (Rollins et al. 2002). Aspect may not be as strongly associated with fire probability in the Mojave Desert as in forested ecosystems given to the high spatial heterogeneity of fuel loads across the Mojave (Vann Linn et al. 2013). Furthermore, the effects of non-native invasive grasses may be greater than the natural patterns of fuel loading and flammability between north- and south-facing slopes.

Chapter 8. Distribution and dynamics of non-native invasive plants: distribution and abundance of red brome and Sahara mustard

8.A. Rationale for and information provided by the assessment

The ranges of non-native plants across the United States continue to expand and fill in. In the Mojave Desert Network and across the Mojave Desert as a whole, managers are concerns about shifts in the distributions of non-native invasive plants. Increases in the distribution, abundance, and density of non-native invasive plants can lead to decreases in the species richness of native plants and animals (Vitousek et al. 1996, Vilà et al. 2011) and in nutrient availability (Ehrenfeld 2003). In the Mojave Desert, invasive plants have been associated with increases in fire frequency (D'Antonio and Vitousek 1992, Brooks et al. 2004) and decreases in water availability (Zavaleta 2000). Here, we define a non-native invasive (henceforth, *invasive*) plant as one that colonizes or is introduced to a new location, establishes a self-sustaining population, and spreads into surrounding, uncolonized locations (Lockwood et al. 2007).

We assessed the distribution and abundance of red brome (*Bromus rubens*), a grass, and Sahara mustard (*Brassica tournefortii*), a forb. Although these are not the only invasive species in the Mojave Desert (others include, for example, Mediterranean grass [*Schismus* spp.], redstem filaree [*Erodium cicutarium*], and tamarisk [*Tamarix* spp.]), they are two of the most common. At the regional level, these were the only two species for which occurrences across extensive climatic gradients were recorded and reported in sufficient quantity to facilitate bioclimatic envelope modeling. Other species that are present or expanding into the Mojave Desert also are likely to affect the status of natural resources currently and as climate changes.

Red brome is an annual grass native to southern Europe. It was first recorded in the southwestern United States in the 1880s (Salo 2005). Most native species of plants in the Mojave Desert germinate following precipitation from late September through early December (Beatley 1974), and require cumulative precipitation of ≥ 25 mm to germinate. By contrast, red brome can germinate following 10 mm of precipitation (Beatley 1966). After germination, red brome grows until it flowers and produces fruits in April and May (Beatley 1966). We conducted complementary assessments of the distribution of red brome as a function of either climate (this chapter) or topography, land cover, and phenology (next chapter). Because data requirements for these two assessments differed, the assessments were conducted separately. However, inferences largely were similar, and the assessments suggested clear next steps to decrease uncertainty in projections of future distributions and dominance of non-native invasive plants in the Mojave Desert Network.

Projected changes in climate are likely to affect trends in the distribution and abundance of red brome and associated trends in the condition of natural resources. High winter precipitation increases the ability of red brome to outcompete native plants (Salo 2005), and germination and growth of red brome decreases during periods of extended drought (Salo 2004). In years with precipitation near the current climate normal (30 year average), red brome germinates earlier than most native species and

sequesters much of the available water and nitrogen (Brooks 2000), both of which are limiting resources in the Mojave Desert (Rundel and Gibson 1996). In the Sonoran Desert, the biomass of native species was lower in areas where density of red brome was high than in areas where density of red brome was low (Salo et al. 2005).

Sahara mustard is an annual forb that thrives in coarse, sandy soils and has colonized many vegetation types in the Mojave Desert (Bangle et al. 2008). It is native to the semi-arid and arid regions of northern Africa and the Mediterranean regions of southern Europe, and was first collected in the United States in 1927 (Minnich and Sanders 2000). Germination of Sahara mustard is initiated by autumn precipitation. The species typically flowers and produces fruits in early March, earlier than native species, and persists in the seed bank (Bangle et al. 2008). High levels of aboveground biomass of Sahara mustard can affect the condition of natural resources because they increase the probability of fire.

The ability to maintain natural resources depends in part on distinguishing between where invasive species are likely to be present and where they are likely to be abundant. Therefore, we assessed the current and potential future (to 2050) probabilities of presence and of relatively high abundance of red brome and Sahara mustard across the Mojave Desert. Our assessments suggested clear next steps to increase confidence in projections of future distributions and dominance of non-native invasive plants in the Mojave Desert Network.

Monitoring can inform efforts to prevent new invasions and to eradicate recent invasions of non-native species (Moody and Mack 1988). Eradication is practical only during the early stages of invasion, when there are relatively few individuals in a relatively small area (Rejmanek et al. 2005). Therefore, monitoring of invasive plants is most likely to be useful to NPS when implemented over extensive areas. However, field surveys across hundreds to thousands of square kilometers are not feasible. Accordingly, NPS asked us to assess whether remote sensing could be used to identify the current locations of invasive plants, to assess the probability of future invasion, or to document temporal trends in the distribution or abundance of invasive plants. This assessment is provided as Appendix 3.

As we describe in Appendix 3, remote sensing only is effective across extensive areas after the abundance of invasive plants becomes high. High abundance allows for detection on the basis of spectral, textural, or temporal characteristics. Use of remote sensing is unlikely to identify early colonizations directly, but models at landscape and regional extents (such as those presented below) can incorporate remotely mapped results to identify areas where probability of colonization may be high, and targeted monitoring may be warranted. Even widespread invasive plants have not filled in all habitat (Bradley et al. 2015). Therefore, identifying areas that may be colonized currently and as environmental conditions change may support effective, long-term control efforts.

8.B. Data and methods

8.B.1. Data

We compiled information on the distribution of red brome and Sahara mustard from regional data (California Invasive Plant Council 2012, CalFlora 2014), surveys conducted by the Bureau of Land

Management and Mojave Desert Network, the records of local and regional biologists, and herbaria (Appendix 4). Most of the data were presence-only, but some documented quantitative or qualitative abundance. We classified locations as *high abundance* if the cover of the species was 10% or greater or if the data qualitatively described the cover of the species as continuous.

We included in the compilation our data from two field surveys that we conducted in March 2011 and 2012 for a project that was separate from the NRCA. For that separate project, we collected data on presence and abundance of red brome and Sahara mustard along about 1500 km of primary roads in southern Nevada, southern California, and Arizona. We classified the abundance of either species as high if its cover was continuous well beyond the road corridor (typically hundreds of meters).

We aggregated the available information to create separate sets of data on presence and high abundance of each species. Records for both species were restricted to the southwestern United States (i.e., the region in which red brome and Sahara mustard are invasive). Therefore, we limited the extents of the study regions to those circumscribed by the presence locations of each species. Absence data comparable in extent to the presence and high abundance data were not available for either species. To remove duplicate entries, reduce sampling bias, and make the resolution of the species data consistent with the resolution of the climate data (see below), we resampled each of the four datasets to include only one point per 2.5 arcminute (~5 km). If abundance data were available for more than one point within a grid cell, we retained the maximum abundance value. Although higher resolution (~1 km) climate interpolations are available for the region, all gridded climate layers are limited by the dearth of weather stations across the Mojave Desert. In sparsely gauged regions such as the Mojave Desert, high-resolution climate data can give a false sense of precision. Therefore, we elected to construct regional models with 5 km interpolated data.

We obtained estimates of current global climate and projected future global climate from Worldclim (www.worldclim.org). Worldclim provides these data as interpolated surfaces of mean monthly temperature and mean monthly precipitation at a spatial resolution of 2.5 arcminutes (Hijmans et al. 2005). We used estimates for 1950–2000 to represent current climate. We used an ensemble of climate-model projections from the Fifth Assessment Report of the Intergovernmental Panel on Climate Change (IPCC) to represent future climate in 2050. Details on our selection of climate projections are in Appendix 5.

8.B.2. Models of presence and abundance

We used bioclimatic envelope models (BEMs) to assess the current and potential future distribution of climatic habitat for red brome and Sahara mustard across the Mojave Desert. Climate is one of the many dimensions of habitat for any species, and is relevant to assessing potential drivers of a species' distribution across large areas and long periods of time. BEMs are useful for exploring the relations between the locations where a species occurs and the climate at those locations (Franklin 2009). Identification of climatic and other environmental attributes consistent with presence or high abundance of an invasive species may be used to identify other areas within which the species may establish or become abundant, thus to prioritize management actions. BEMs often have been applied to map areas that can support presence of a given species, but rarely to map areas associated with high abundance of a given species (Bradley 2013).

Because the red brome and Sahara mustard may occur in areas with climatic attributes that were not sampled in this analysis, we do not assume that climatic habitat is synonymous with the current or future distribution of either species. The data we compiled for this analysis represent much of the climatic gradient across the Mojave Desert. However, data limitations still could result in underestimation of potential ranges in which the species are present or have high abundance. Therefore, we have relatively high confidence in BEM projections of areas in which likelihood of presence or high abundance is high, but lower confidence in projections of areas in which likelihood of presence or high abundance is low.

For each species, we used two BEM methods, Maxent (Phillips et al. 2006) and Bioclim, to project the current and future locations of climatic habitat consistent with presence and high abundance. Both methods are based on presence-only data, but Maxent generates pseudo-absences to construct probabilistic relations between environmental variables and species presence (or high abundance) whereas Bioclim uses thresholds associated with each variable to define areas consistent with presence (Pearson and Dawson 2003). We applied Maxent and Bioclim to calculate the total area of current climatic habitat consistent with the species' presence and high abundance within the region circumscribed by the data (Figure 24). We then identified areas in which climatic habitat for red brome and Sahara mustard may expand, remain stable, or contract in the future.

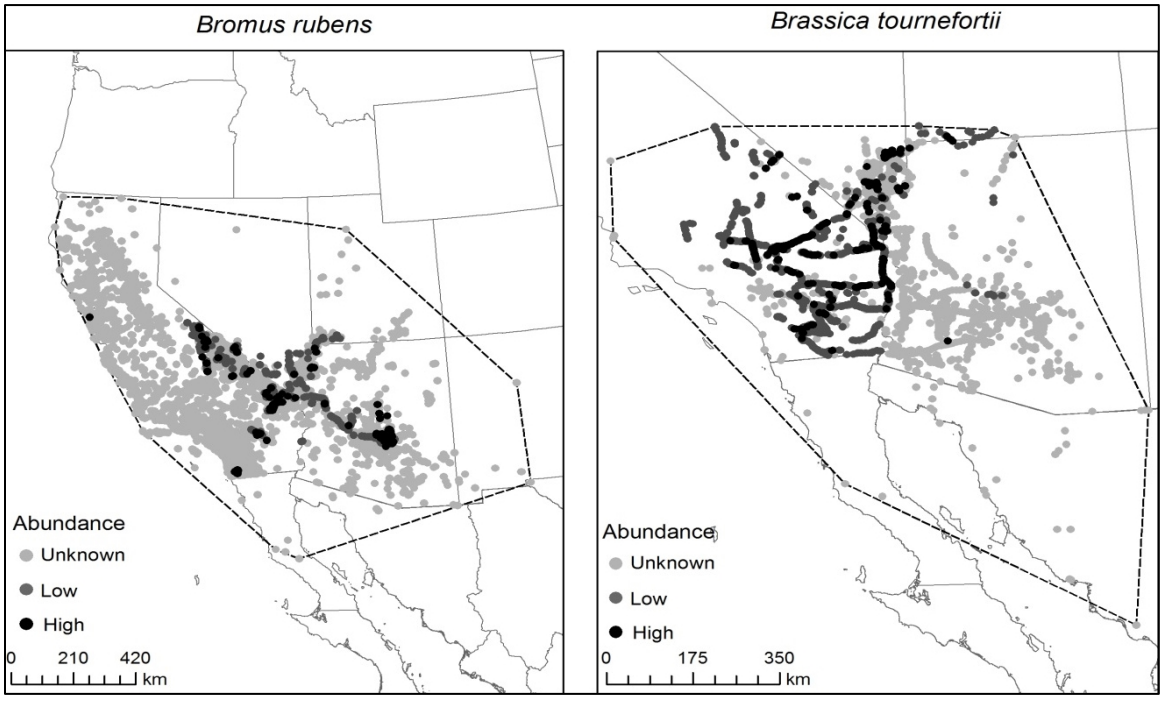


Figure 24. Locations at which red brome (*Bromus rubens*) and Sahara mustard (*Brassica tournefortii*) were recorded by one or more sources (Appendix 4). We classified a record as *high abundance* if the cover of the species was 10% or greater, if the cover of the species was described qualitatively as continuous, or if the cover of the species was continuous well beyond the road corridor. We classified a record as *low abundance* if the latter criteria were not met but some description of abundance was associated with the record. We classified a record as *unknown* if it did not include a description of abundance.

Details on the assumptions of each method and the process we followed to map current and future climatic habitat are in Appendix 6. In brief, we estimated current habitat on the basis of observed climate and the combined projections of the two BEM methods. We estimated future habitat assuming two different trends or pathways of carbon dioxide concentrations. In the main NRCA, we report results from a pathway that is considered to be medium-low (representative concentration pathway [RCP] 4.5), which corresponds to atmospheric carbon dioxide (CO_2) concentrations of 650 ppm by 2100. Results from a pathway that is considered to be high (RCP 8.5), which corresponds to atmospheric CO_2 concentrations of 900 ppm by 2100, are in Appendix 7. For each CO_2-concentration pathway, we used each BEM to estimate future climatic habitat on the basis of outputs from five Atmosphere-Ocean General Circulation Models (AOGCMs). Thus, we obtained 10 model projections for each CO_2-concentration pathway. Estimating future habitat or species' distributions on the basis of an ensemble of climate models and multiple BEMs reduces the bias inherent in any one climate model or BEM (Araújo and New 2007).

We applied two thresholds to project where climatic habitat for each invasive species may remain stable, expand, or contract. The lower threshold included all areas classified as future habitat by at least one of the ten model projections. The higher threshold included all areas classified as future habitat by six or more model projections. Regardless of which threshold we applied, we defined areas of stability as those classified as habitat by models of both current habitat and future habitat. We defined areas of expansion as those classified as habitat by models of future habitat but not models of current habitat. We defined areas of contraction as those classified as habitat by models of current habitat but not by models of future habitat.

8.C. Assessment results

Data on red brome presence (3303 records at a 2.5 arcminute spatial resolution) and high abundance (110 records) were distributed throughout the Mojave Desert (Figure 24). Points at which abundance of red brome was high were recorded in each of the Mojave Desert Network parks. Data on Sahara mustard presence (1855 records) and high abundance (218 records) also were spatially distributed. Points at which Sahara mustard was abundant were recorded in all of the Mojave Desert Network parks except Grand Canyon–Parashant.

8.C.1. Current habitat

Our assessment suggested that current climatic habitat for both red brome and Sahara mustard is widespread (Figure 25). With the exception of Badwater Basin in Death Valley, we projected that the full extent of all of the parks could support red brome. This does not mean that red brome currently is present throughout the parks, or that habitat quality is homogenous. Instead, it means that throughout the Mojave Desert Network, current climate is consistent with the climate in locations where red brome is known to occur. Our projections also indicated that each park contains areas that could support high abundance of red brome, although climatic habitat consistent with high abundance appeared to be more extensive in southern Nevada and western Arizona than in southern California.

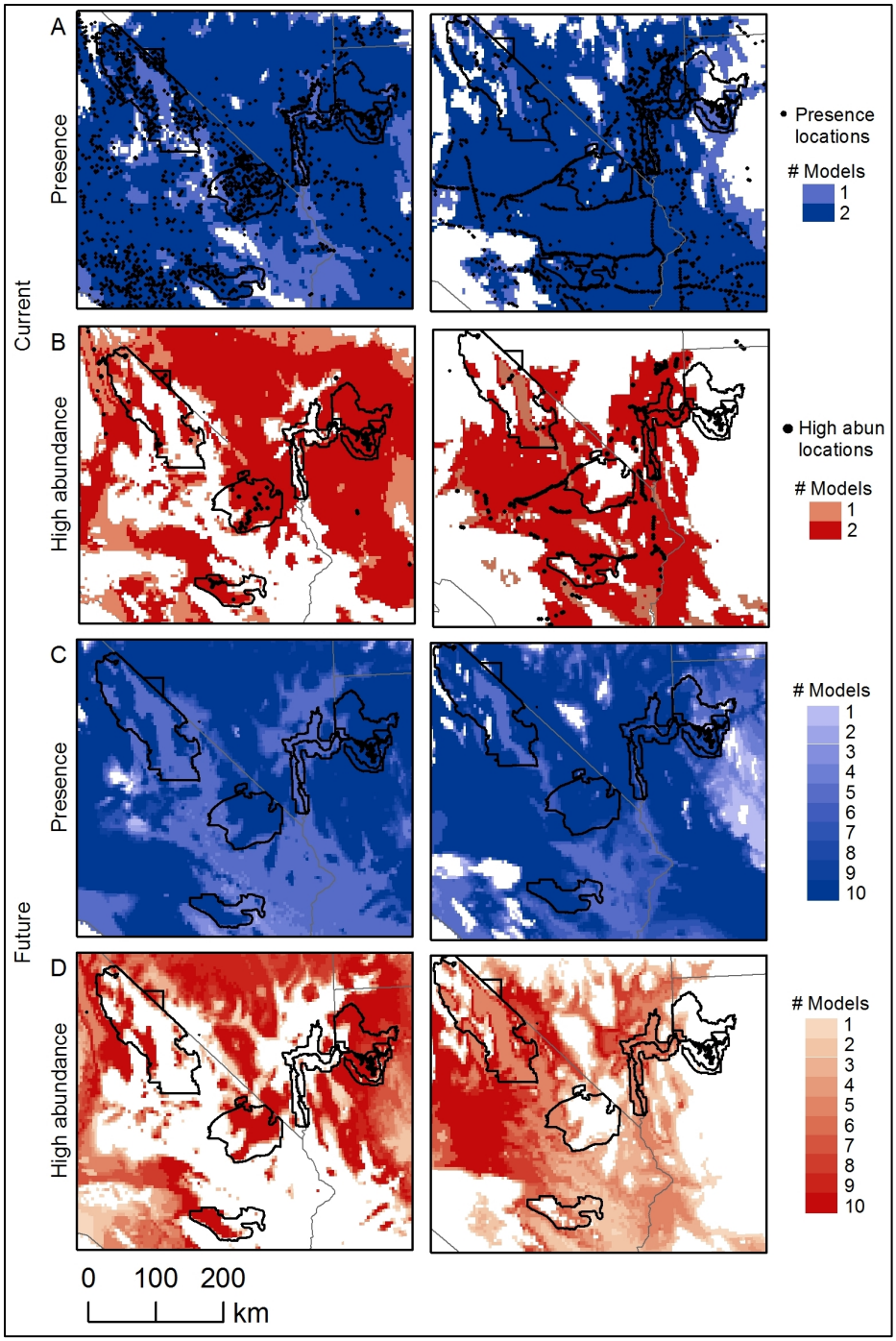


Figure 25. Projected current and future distributions of climatic habitat for red brome (left column) and Sahara mustard (right column). Black dots in A and B indicate where data were collected. (A, B) We summed the Maxent and Bioclim projections (# Models) of current climatic habitat to estimate the maximum extent of current presence and high abundance of each species. (C, D) Models of future climatic habitat were based on representative concentration pathway (RCP) 4.5 (see Appendix 7 for outputs of models based on RCP 8.5). The process we used to create ensemble models of future climatic habitat is described in Appendix 6. The color ramp indicates how many of the ten models (two envelope models x five climate projections) in the ensemble projected that future climate at a given location would be consistent with presence or high abundance.

Our models projected that climatic habitat for Sahara mustard is present throughout all of the parks except the eastern portion of Grand Canyon–Parashant. We projected climatic habitat that would support high abundance of Sahara mustard throughout Lake Mead, in the drier areas of Joshua Tree and Mojave National Preserve, and along the southern and eastern edges of Death Valley.

The results of our Maxent models suggested that different aspects of climate are associated with presence and high abundance. Temperature was by far the variable most strongly associated with presence of both red brome and Sahara mustard, although minimum temperature was strongly associated with presence of red brome and maximum temperature was strongly associated with presence of Sahara mustard. Our results were consistent with experiments that suggested that intolerance of cold temperatures may limit survival of red brome during the winter (Bykova and Sage 2012). Our results also were consistent with measurements of the heat tolerance of Sahara mustard; both our results and those measurements suggested that Sahara mustard tends to become established during periods in which temperature is relatively high (Suazo et al. 2012).

By contrast, minimum temperature and spring precipitation were strongly associated with high abundance of red brome. The variable most strongly associated with high abundance of Sahara mustard was summer precipitation. Both species are believed to have higher population growth rates (Beatley 1974, Salo 2004) and a competitive advantage over native species (e.g., Barrows et al. 2009) during periods of relatively high precipitation. Thus, although temperatures may limit the geographic ranges of red brome and Sahara mustard, precipitation may limit the abundance of either species within its geographic range.

8.C.2. Future habitat

Our models suggested that future climate in all of the Mojave Desert Network parks will be consistent with presence of red brome (Figure 26, Appendix 7). Again, this does not mean that red brome will spread throughout each park by 2050. Our models projected that future climate in at least some areas within each park will be consistent with high abundance of red brome (Figure 3). Our models suggested that future climate could become less consistent with high abundance of red brome in drier portions of all of the parks (Figure 3), including the eastern portion of Joshua Tree and the southeastern portion of Mojave National Preserve. Our projections also suggested that future climate will become less consistent with high abundance of red brome in Death Valley than in other parks by 2050. We projected future climatic habitat only along the western edge of Death Valley.

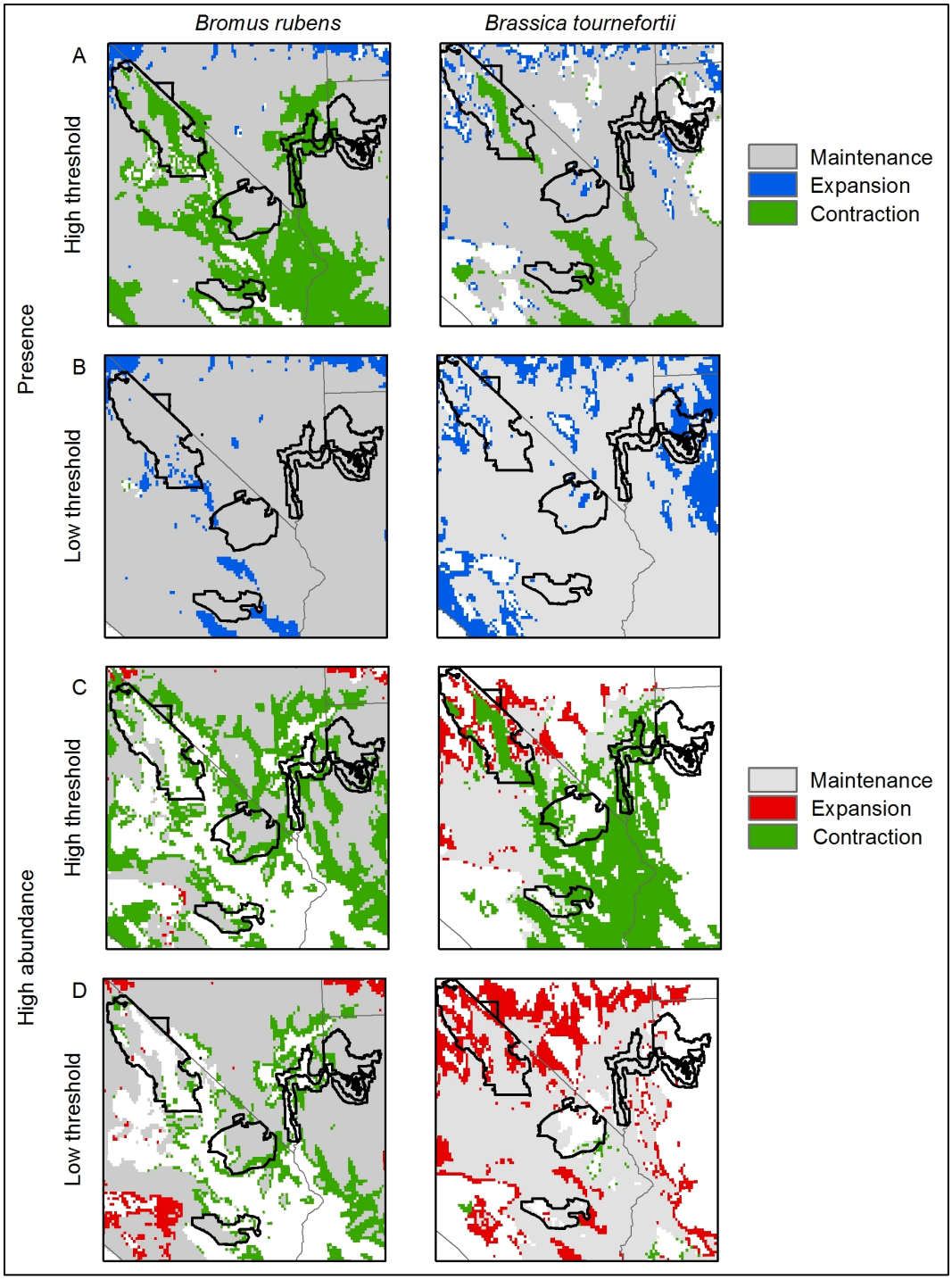


Figure 26. Projected areas in which climatic habitat for red brome (*Bromus rubens*) and Sahara mustard (*Brassica tournefortii*) will be stable, expand, or contract by 2050 assuming RCP 4.5. Stable, location was classified as habitat by models of both current habitat and future habitat. Expansion, location was classified as habitat by models of future habitat but not models of current habitat. Contraction, location classified as habitat in models of current habitat but not in models of future habitat. High threshold, locations classified as future habitat by six or more of the ten model projections. Low threshold, locations classified as future habitat by one or more of the ten model projections.

Each park had areas in which a relatively high number of models projected widespread habitat consistent with presence of Sahara mustard by 2050 (Figure 26). Our models projected future climatic habitat consistent with high abundance of Sahara mustard in the eastern portion of Joshua Tree, the western edge of Mojave National Preserve, and throughout much of Death Valley and Lake Mead. However, with the exception of some high-elevation areas of Death Valley, projections of different models varied. As a result, there is high uncertainty in the future distribution of climatic habitat consistent with high abundance of Sahara mustard in the Mojave Desert Network.

Projections of expansion of climatic habitat consistent with high abundance of Sahara mustard illustrate the latter uncertainty (Figure 26). If we base identification of future climatic habitat on agreement among six or more of the ten models, then large portions of the Southwest, including relatively warm areas of some parks, may not support high abundance of Sahara mustard by 2050.

However, if we base identification of future climatic habitat on the results of one or more of the ten models, then the extent of climatic habitat consistent with high abundance is stable is most areas and will increase at high elevations in Death Valley by 2050. The differences among models of future climatic habitat for Sahara mustard were driven by whether the models assumed that high abundance is limited by high temperature. If we assume that Sahara mustard can tolerate increases in summer temperature (i.e., changes in temperature consistent with current projections of climate change), then the extent of habitat consistent with high abundance is likely to be stable or increase (Figure 26).

8.D. Data gaps and confidence in the assessment

Many efforts to project the future distribution of invasive species rely on presence-only data. Presence-only data are useful for estimating the area in which an invasive species may establish, but do not allow reliable projections of abundance (Bradley 2013); abundance is more closely associated with the condition of natural resources. By basing projections on two BEMs, five AOGCMs, and two emissions pathways rather than on single models or scenarios, we reduced uncertainty in the identification of areas that likely can support presence or high abundance of red brome and Sahara mustard (Araújo and New 2007, Diniz-Filho et al. 2010). As the proportion of models that were consistent increases, so does our confidence in the projections.

Interpretations of ensemble models of the future distributions of red brome and Sahara mustard depend on the threshold between habitat and not habitat (i.e., the number of models with similar outputs). There is no consensus on selection of a threshold. Use of a higher threshold (i.e., greater consistency among models is necessary to identify an area as potential habitat) will increase confidence in projections of areas of expansion, but also will increase the probability of false negatives (erroneously predicting no expansion). It is likely that our lower threshold overestimated the probability of establishment and high abundance of red brome and Sahara mustard by 2050. If one is prioritizing management actions, then we suggest that identification of locations with relatively high likelihood of future invasion be based on relatively high agreement among models. However, data limitations may lead to underestimation of the range of climatic conditions consistent with invasion and establishment. We have relatively high confidence that invasion is most likely in areas where six or more models projected expansion of climatic habitat. We have less confidence that invasion is unlikely in areas where six or more models projected contraction of habitat.

The empirical data that we compiled on species presence and abundance were collected opportunistically, mostly along roadsides. Although the data encompass regional climatic gradients, they do not represent gradients of disturbance, soil type, and topography. Ideally, our results should be treated as the coarsest-resolution layer of a hierarchical model that combines regional projections of climatic habitat with regional and local projections of habitat modeled on the basis of disturbance, soil type, and topography (e.g., Pearson et al. 2004, Bradley 2010). Even if climate is broadly consistent with invasion, environmental factors such as soil type and competition with native plants could prevent local establishment and increases in abundance. Therefore, our results should be interpreted as indicating only climatic conditions consistent with invasion.

Confidence in the model outputs would be increased by collection of field data in areas that have not been well sampled. Information on the relative abundance of a given invasive species, even if qualitative, is more strongly associated with the potential ecological effects of that invasive species than are data on the presence of the species. Absence data also are useful for increasing confidence in model outputs. Collection of abundance data across the climatic or latitudinal gradients in the Mojave Desert will be the most useful for modeling relations between abundance and climate. However, if the ultimate goal is to project future distributions at spatial resolutions finer than 5 km, then data collection should be stratified not only across climatic gradients but across gradients of land use or topography. For example, sampling could be conducted along a climatic gradient from Joshua Tree to Death Valley and along a gradient of distance from roads.

Chapter 9. Distribution and abundance of non-native invasive plants: distribution and abundance of red brome as a function of topography, land cover, and phenology

9.A. Rationale for and information provided by the assessment

We aimed to identify variables that likely would be useful for projecting the occurrence and cover of red brome (*Bromus rubens*) and other invasive plants across the Mojave Desert Network, and to use models that proved effective in mapping species distributions and habitat quality in previous research. Therefore, we focused not only on model outputs, but on methods. In this assessment, we consolidated field data on the occurrence (presence or absence) and cover (abundance) of red brome; projected the occurrence and cover of the species across the Mojave Desert, including but not limited to the Mojave Desert Network, and evaluated relations between environmental variables and occurrence and cover of red brome.

9.B. Data and methods

9.B.1. Consolidation of data and selection of response variables

We evaluated multiple sources of data on the occurrence, abundance, and biomass of red brome on the basis of whether they contributed reliable information to train and test predictive models of the species' distribution (Table 28). We concluded that three sources met our criteria given the similarity of their objectives, sampling methods, and spatial and temporal scales: the NPS (Mojave Desert Network Inventory & Monitoring Program and National Vegetation Mapping Program), a group of individuals affiliated with diverse organizations who are collaborating on research in the Mojave Desert supported by the Department of Defense's Strategic Environmental Research and Development Program, and researcher Scott Abella (University of Nevada, Las Vegas).

Table 28. Sources of data on the distribution of red brome in the Mojave Desert.

Data type	Spatial extent	Years collected	Volume of data	Limitations	Owner or custodian
Presence only, points	California	1929–2011. New records occasionally are added to the online database.	Varies among counties	Survey effort may not have been spatially consistent	CalFlora
	Arizona, California, Idaho, Nevada, New Mexico, Utah, Mexico	Not specified	12,127 points	Compiled from numerous public sources	Bethany Bradley and Caroline Curtis
Presence only, polygons	Death Valley National Park, California	1970–2012	905 polygons	Collected during treatments to remove invasive species and field surveys	Jane Cipra
	Joshua Tree National Park, California	Not specified	57 polygons	Species of *Bromus* not identified	NPS

Table 28 (continued). Sources of data on the distribution of red brome in the Mojave Desert.

Data type	Spatial extent	Years collected	Volume of data	Limitations	Owner or custodian
Percent cover (abundance)	Arizona, California, Nevada	Not specified	134 points	Observations had to be ≥ 10% cover to be included	Bethany Bradley and Caroline Curtis
Presence, absence, and percent cover (abundance)	Arizona, Nevada	2008–2009	126 points	Includes data on both dead and live plants	Scott Abella
	Arizona, California, Nevada, Utah	2009–2012	618 points	–	Robert Klinger
	California	1997–1999	1,242 points	Non-native grasses not identified at the species level	Kathryn Thomas
	Arizona, New Mexico	Not specified	12,070 points	Percent cover was recorded, but many of the data records have been corrupted. 38 points are of red brome.	Arizona GAP Analysis Program
	Arizona, California, Idaho, Nevada, Utaho, New Mexico, Mexico	Not specified	10,999 points	Abundance rank was recorded at several points, although most abundances are zero; compiled from multiple sources.	Lori Pelech
	Arizona, California, Nevada: Death Valley, Lake Mead, and Mojave National Preserve	2010–2011	1,665 points	Data limited to national parks	NPS
	Arizona, Colorado, New Mexico, Nevada, Utah	1911–2005	62,001 points	Of the total points, 1216 are red brome; only two records of abundance	Southwest Exotics Plant Mapping Program

Data from the two NPS programs were collected to classify and map vegetation. Data were collected from 1,660 plots that were sampled during 2010 and 2011. Plots were 1,000 m^2. Most were 31.6 m on a side, although some plots that were constrained by terrain had different dimensions. The second set of data was collected to support research on the distribution of non-native invasive plant species and wildfire in the Mojave Desert given projected future changes in climate. Data were collected from 618 plots, each 1,000 m^2 (~32 m on a side), in 2009, 2011, and 2012. Ocular estimates of cover of all vascular plants were recorded at multiple resolutions; to increase consistency with cover estimates from NPS, we used only cover estimates at the 1,000 m^2 resolution. Abella's data were collected to assess biological and physical correlates of the presence of red brome. Data were collected from 126 plots, each 900 m^2, in and around Lake Mead National Recreation Area from autumn 2008 through spring 2009 (Abella et al. 2012). We considered the difference between 1000 m^2 and 900 m^2 plots to be negligible.

9.B.2. Derivation of environmental variables

We developed a set of spatial data layers to represent multiple environmental variables that were likely to be associated strongly with occurrence or cover of red brome. Data on all of the variables were available for the full extent of the Mojave Desert. With the exception of the geological and phenological variables, we processed and summarized all variables at their native resolution (in most cases, 30–90 m pixels) before resampling them to a common resolution (250 m, the resolution of the NDVI data described below). The final set of variables included measures of elevation, topographic roughness, aspect, land cover, distance from the nearest primary or secondary road, geology, and phenology. We selected phenology variables (Table 29) on the basis of hypotheses about phenological differences between invasive species (e.g., red brome) and native species (e.g., Wolkovich and Cleland 2011) or previously demonstrated associations between metrics derived from vegetation indexes and cover of plants in arid systems (e.g., Peterson 2005, Wallace and Thomas 2008). To facilitate projections, we generated phenology data for 2005 (a relatively wet, productive year) and 2008 (a relatively dry year) (Figure 27). We conducted all geoprocessing in a geographic information system (GIS; ArcGIS v10.1, Esri, Redlands, California) and in R v3.0 (R Core Team 2013, www.R-project.org). Details of the derivation are in Appendix 8.

Table 29. All phenology variables derived from smoothed eMODIS time-series data. We derived values of the variables from 15 September of the previous year through 1 June in the current year for each growing season in the time series from 2001 through 2013.

Variable	Interpretation	Derivation
Start of season	Beginning of measurable photosynthesis in the vegetation canopy	Day of year identified as that with the local minima of NDVI nearest to but before the peak NDVI for the season
End of season	End of measurable photosynthesis in the vegetation canopy	Day of year identified as that with the local minima of NDVI nearest to but after the peak NDVI for the season
Duration of green-up	Duration of photosynthetic activity from the start of season through the day of peak greenness	Number of days (expressed as a fraction of a calendar year) from start of season through the day in the growing season with maximum NDVI
Time-integrated NDVI	Photosynthetic activity in the canopy across the growing season	Daily (interpolated) integration of NDVI above zero across the growing season
Season-integrated NDVI	Photosynthetic activity in the canopy across the growing season as determined by the start and end of measurable photosynthesis	Daily (interpolated) integration of NDVI above zero across the growing season (start of season to end of season)

* Variables affected by the fact that in many pixels, there was measurable photosynthetic activity both early and late in the growing season (i.e., dual local maxima). These variables were further refined to reflect growth both early and late in the season when such patterns were detected by the algorithm.

Table 29 (continued). All phenology variables derived from smoothed eMODIS time-series data. We derived values of the variables from 15 September of the previous year through 1 June in the current year for each growing season in the time series from 2001 through 2013.

Variable	Interpretation	Derivation
Seasonal maximum NDVI*	Maximum level of photosynthetic activity in the canopy	Maximum NDVI within the growing season
Difference between seasonal maximum NDVI and the lowest 10th percentile of NDVI over the time series*	Maximum increase in canopy photosynthetic activity above the baseline for the pixel	Difference between seasonal maximum NDVI and the baseline established by the lowest 10th percentile of observations in the NDVI time series
Difference between seasonal maximum NDVI and the median of NDVI observations over the entire time series*	Difference between canopy photosynthetic activity and a baseline established by a temporally varying baseline for the pixel	Difference between seasonal maximum NDVI and a baseline calculated as the median NDVI for all years on the day on which seasonal maximum NDVI was observed
Difference between seasonal maximum NDVI and mean local NDVI on the same day*	Difference between canopy photosynthetic activity and a baseline established by photosynthetic activity in the local neighborhood	Difference between seasonal maximum NDVI and a baseline calculated as the mean NDVI for the 24-pixel neighborhood surrounding the focal pixel on the day on which seasonal maximum NDVI was observed
Difference between seasonal maximum NDVI and mean global NDVI on the same day*	Difference between canopy photosynthetic activity and a baseline established by mean photosynthetic activity at the extent of the study area	Difference between seasonal maximum NDVI and a baseline calculated as the median NDVI for a random sample of 100,000 pixels across the study area on the day on which seasonal maximum NDVI was observed
Rate of green-up*	Rate of green-up	Slope of the line formed by connecting values of NDVI at the start of season and at the time of peak greenness

* Variables affected by the fact that in many pixels, there was measurable photosynthetic activity both early and late in the growing season (i.e., dual local maxima). These variables were further refined to reflect growth both early and late in the season when such patterns were detected by the algorithm.

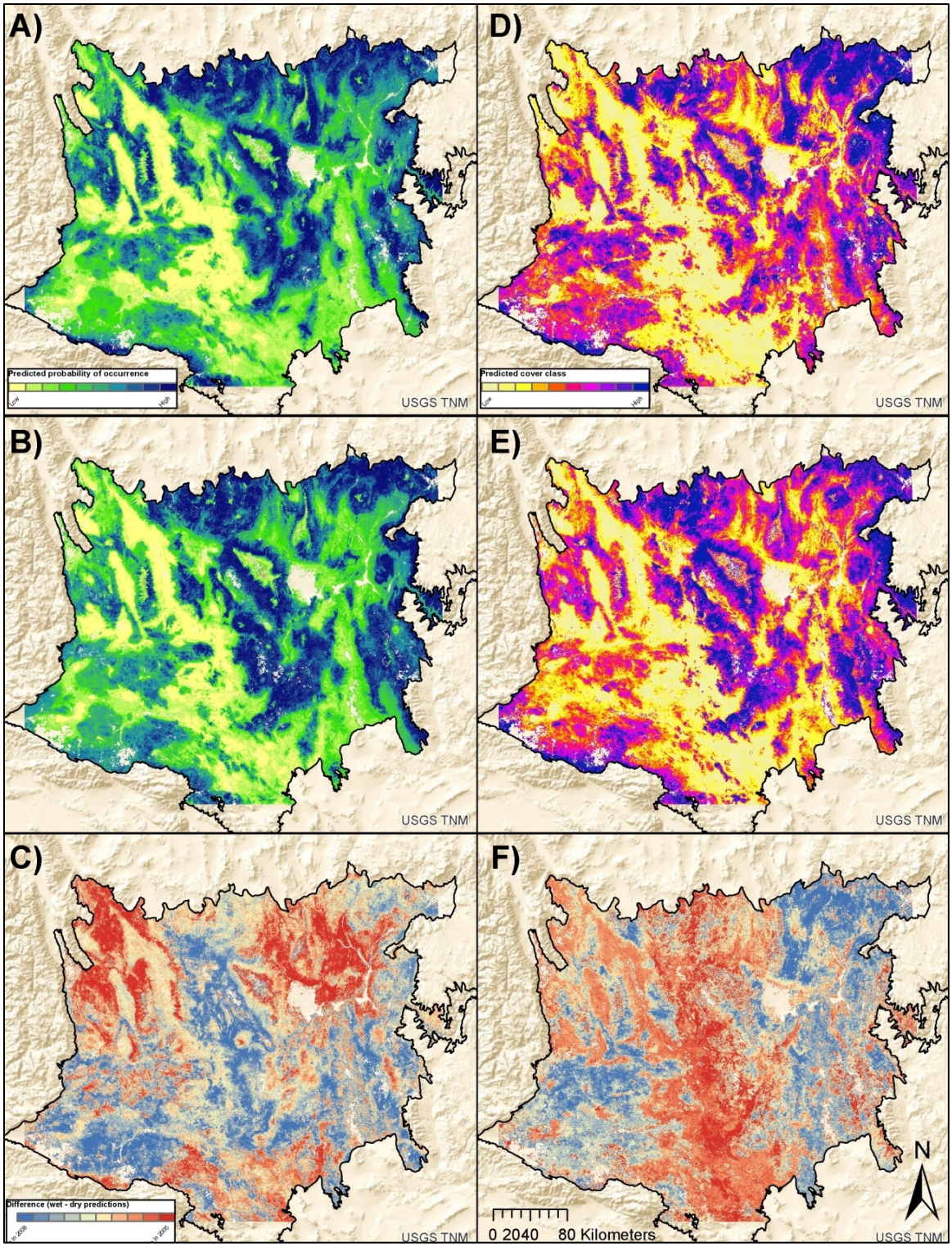


Figure 27. Predicted probabilities of occurrence (A, B) and cover class (D, E) of red brome in a wet year (2005; A, D) and a dry year (2008; B, E) year, and the difference between wet and dry years (C, F). Abundance classified into 10 quantiles.9.B.3. Model fitting, prediction, and evaluation.

We used the environmental variables described above to fit both a model of probability of occurrence and a model of percent cover of red brome. We classified cover as [0–0.1), [0.1–1), [1–2), [2–5), [5–10), [10–25), [25–50), [50–75), [75–95), and [95–100]. We used three methods to fit the models: support vector machine (SVM; Vapnik 1998), random forest (Breiman 2001), and gradient boosted machine learning (Friedman 2001) (Appendix 9). We evaluated the accuracy of each model (Appendix 9) before building the final models of occurrence and cover across the Mojave Desert during 2005 and 2008. Because our focus was on projection rather than explanation or description, we did not evaluate relative strength of association or the direction and magnitude of associations between the environmental variables and the response variables.

9.C. Assessment results

9.C.1. Consolidation of data

Data on occurrence and cover of red brome have been collected throughout the Mojave Desert. The same variables were measured on each plot in each of the three sets of data we used. The data were collected during or just after annual green-up of vegetation, and include estimates of both occurrence and cover of red brome. In each year, approximately 66% of plots contained red brome (range 52–81%), whereas annual mean percent cover ranged from 0–80%. Cover values varied annually, with the highest observed levels of cover occurring in 2012. Most (~90%) of the plots containing red brome had low cover, from trace (< 0.1%) to 9.5%.

The field data we used were collected across the Mojave Desert on the basis of stratified random sampling and spanned major environmental gradients. Many other sources of data, by contrast, are opportunistic (e.g., roadside) samples (Table 28). Additionally, all three data sources included observed absences in addition to presences of red brome, which often increases the accuracy of distribution models (Franklin 2009).

9.C.2. Occurrence and cover of red brome

Our results suggested that habitat for red brome is distributed across much of the Mojave Desert. For example, projected occurrence probabilities for 2005 for 25% of the region were ≥ 0.83, and projected occurrence probabilities for about half of the region were ≥ 0.64. Occurrence probabilities for 2008 were similar. The projected cover of red brome generally was low across the Mojave Desert in both wet and in dry years; mean projected cover was 2.6 (SD 1.0) and 2.1 (0.78) in 2005 and 2008, respectively.

However, projected habitat quality was not uniform, and projected probabilities of occurrence and cover of red brome varied among parks (Figure 27a, b). For example, mean projected occurrence probabilities for 2005 were 0.48 (SD 0.30), 0.67 (0.23), 0.52 (0.19), 0.66 (0.27), and 0.80 (0.08) for Death Valley, Joshua Tree, Lake Mead, Mojave National Preserve, and Grand Canyon–Parashant, respectively. Mean projected cover for 2005 was 2.13 (0.95), 2.47 (0.77), 2.63 (0.72), 2.47 (0.92), and 4.11 (0.80) for Death Valley, Joshua Tree, Lake Mead, Mojave National Preserve, and Grand Canyon–Parashant, respectively.

The spatial distribution of projected occurrence and cover changed as a function of time and possibly as a function of annual precipitation (Figure 27c, f). For example, projected probabilities of

occurrence in 2005 and 2008 were highest in the northeast and southwest, respectively. Projected cover in 2005 was highest in the northeast and southwest, whereas projected cover in 2008 was highest in a large, north-south oriented swath in the center of the region.

In a dry year (2008), the areas in which occurrence probability was high also were those in which projected cover of red brome was high (e.g., Figure 27b, e). In a wet year (2005), however, the areas with a relatively high projected probability of occurrence were not necessarily projected to have relatively high cover of the species (Figure 27c, f).

9.D. Data gaps and confidence in the assessment

The field data we used were collected from plots ≤ 50 m on a side, which is much smaller than the resolution of a MODIS pixel (250 m). Model accuracy might be increased if the areal extent of future field data collection is matched to the native resolution of the remote sensing imagery on which the models are conditioned (Casady et al. 2013).

All five parks are either completely or partially located within the Mojave Desert. However, Joshua Tree is partially located in the Sonoran Desert and Grand Canyon–Parashant is located primarily within the Colorado Plateau. We attempted to account for these biogeographic differences by including land cover as a variable. Future modeling efforts might consider alternative vegetation classifications or fitting region-specific models.

There are multiple sources of data on the occurrence, abundance, cover, and biomass of red brome in the Mojave Desert (Table 28). The sampling period, objectives, and protocols differ among data sources, and data collectors or custodians are diverse. To our knowledge, there has not been an effort to consolidate all of the data on red brome. Additionally, most of these data owners are also pursuing, or have pursued, various analyses to map the distribution, cover, or biomass of the species. Our impression is that at least some of the groups are working independently of one another and, in some cases, are duplicating data-collection and analysis efforts. A synthesis may identify data or knowledge gaps and suggest directions for future work. Obtaining permissions to synthesize the data may require a small number of informal or formal working agreements.

Chapter 10. Relations between habitat quality for Mojave desert tortoise and habitat quality for its predators

10.A. Rationale for and information provided by the assessment

Experts on Mojave desert tortoise (*Gopherus agassizii*, henceforth desert tortoise) and Mojave Desert Network staff had relatively high confidence in existing models of habitat quality for desert tortoise. They suggested that assessment of the species' status and management of the species would benefit from exploring relations among habitat quality for desert tortoise, habitat quality for its predators, and habitat quality for alternate prey. It was necessary to focus on habitat quality rather than species occurrence or abundance because occurrence records for predators and alternate prey are sparse and have high uncertainty. We based our models on published data and related those data to a limited set of geospatial information that was available for the full extent of the study area.

We modeled habitat quality for three species that prey on desert tortoises: Common Raven (*Corvus corax*), Golden Eagle (*Aquila chrysaetos*), and coyote (*Canis latrans*) (Kristan and Boarman 2003, 2007, Esque et al. 2010). We also modeled habitat quality for black-tailed jackrabbit (*Lepus californicus*), an alternative source of prey for predators of desert tortoises. Abundance of black-tailed jackrabbit is inversely related to intensity of predation on desert tortoises (Esque et al. 2010).

Common Raven is considered to be a synanthropic species because its density is relatively high in areas where the intensity of land use is high (Kristan and Boarman 2003, 2007). Abundance of Common Ravens increased across the southwestern United States from 1966 through 2004 (Sauer et al. 2005). Common Ravens prey on desert tortoises even when densities of desert tortoises are low (Kristan and Boarman 2003).

Prior to banning of dichloro-diphenyl-trichloroethane (DDT) within the United States in 1972, Golden Eagles were in sharp decline. The species recovered rapidly after DDT was banned (Pagel et al. 2010), but as human activity throughout the western United States increased, some Golden Eagle populations again declined (Kochert et al. 2002). Golden Eagles hunt in shrublands and nest in trees or cliffs (Marzluff et al. 1997c). Black-tailed jackrabbits are a large proportion of the diet of Golden Eagles (Marzluff et al. 1997c, Kochert et al. 1999, 2002). The proportion of Golden Eagle pairs that laid eggs and the hatching dates of those eggs were associated with densities of black-tailed jackrabbit (Steenhof et al. 1997). Golden Eagles may establish territories and concentrate foraging in areas in which black-tailed jackrabbits have a high probability of occurrence (Marzluff et al. 1997c). Golden Eagles also prey on rodents (Marr and Knight 1983), ungulates (Tjernberg 1981), and desert tortoises (Ernst and Lovich 2009).

In many parts of the United States, coyotes are top predators (Mezquida et al. 2006). Coyotes are generalists. They consume small mammals, birds, amphibians, reptiles, and plants (Andelt and Gipson 1997). Their populations have grown and expanded over most of the conterminous United States over the past 150 years despite efforts to reduce their abundance (Nowak 1979, Crête and Lemieux 1996, McClennen et al. 2001). Coyotes are major predators of desert tortoises (Peterson 1994).

10.B. Data and methods

We conducted an extensive literature review to identify environmental covariates that may be positively or negatively associated with habitat quality for Common Ravens, Golden Eagles, coyotes, and black-tailed jackrabbits. We assumed that species occurrence has a nonlinear relation with these covariates (Nielsen et al. 2009, Hanser et al. 2011). Depending on the species, covariates positively associated with habitat quality included proportion and configuration of different land-cover classes, including anthropogenic classes such as agriculture and anthropogenic features such as power lines. The literature review indicated that abiotic covariates positively associated with habitat quality included aspect, slope, elevation, terrain ruggedness (Nielsen et al. 2004), and soil depth. Depending on the species, covariates negatively associated with habitat quality included proportion and configuration of anthropogenic land cover classes (e.g., agriculture) and distance to linear or other anthropogenic features (e.g., roads with high traffic volumes). We incorporated these covariates into our models of habitat quality (Appendix 10).

10.B.1. Habitat quality for desert tortoises

The current geographic range of desert tortoises includes the Mojave Desert, Sonoran Desert, and southern Great Basin (USFWS 1990). Desert tortoises are associated with undisturbed creosote (*Larrea tridentata*), white bursage (*Ambrosia dumosa*), and paloverde (*Cercidium microphyllum*) (Barrett 1990, Nussear et al. 2009) in southwest-facing areas that have loamy soil and are close to washes (Andersen et al. 2000, Zylstra and Steidl 2009). One study found that occurrence of desert tortoises was low within 400 m of major roads, highways, and railroads (Boarman and Sazaki 2006). Desert tortoises also tend to be absent near developed and agricultural areas (Andersen et al. 2000). Power lines may reduce habitat quality for desert tortoises because predators such as Common Ravens and Golden Eagles sometimes perch on power lines (Steenhof et al. 1993). Power lines are associated with habitat quality for Greater Sage-Grouse (*Centrocercus urophasianus*); distance to the nearest power line is much greater in areas occupied by Greater Sage-Grouse than in areas from which the species has been extirpated (Wisdom et al. 2011). Common Ravens prey on Greater Sage-Grouse nests (Coates et al. 2008) and Golden Eagles prey on adult Greater Sage-Grouse (Schroeder et al. 1999). Therefore, we assumed that probability of predation on desert tortoise increases as distance to a power line decreases.

We used a model of habitat quality for desert tortoise developed by Nussear et al. (2009). This model was based on application of Maxent (Phillips and Dudík 2008) to empirical data on desert tortoise presence and to data on 16 abiotic and biotic environmental covariates. The covariates included soils (average soil bulk density, depth to bedrock, and percentage of soil with large rocks), precipitation (mean and coefficient of variation from November through April and from May through October), topography (average surface roughness, percent smooth, percent rough, slope, aspect [northness and eastness], and elevation), and vegetation (perennial plant cover, annual plant growth potential).The output of the model was a projection of habitat quality at 1-km resolution across a portion of the assessment area (Figure 28). Nussear et al. (2009) randomly assigned 80% of their presence records to model training and 20% to model testing. Eastness, northness, the coefficients of variation of precipitation, percent roughness, and slope were not retained in the final model. Pearson's correlation

coefficient between modeled habitat presence and observed presence of desert tortoises was 0.74, which was statistically significant ($p < 0.01$) (Nussear et al. 2009).

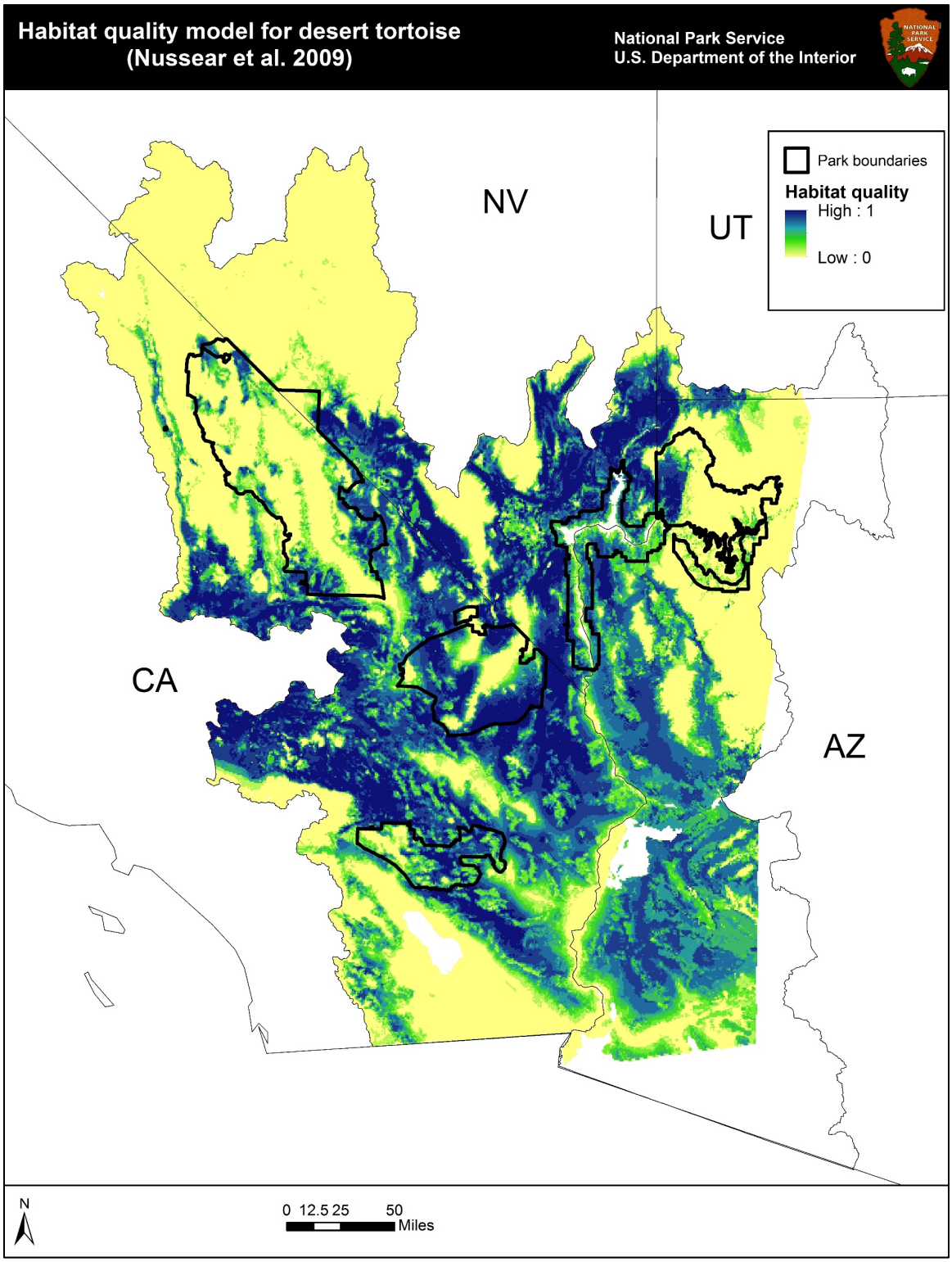


Figure 28. Modeled habitat quality for Mojave desert tortoise (Nussear et al. 2009).

Habitat quality for black-tailed jackrabbits was highest in the northwestern, northeastern, and southwestern portions of the Mojave Desert (Figure 29). Mean modeled park-level habitat quality in descending order was 0.66 ± 0.30 for Grand Canyon-Parashant, 0.16 ± 0.26 for Death Valley, 0.08 ± 0.20 for Joshua Tree, 0.08 ± 0.22 for Mojave National Preserve, 0.07 ± 0.18 for Lake Mead, and 0.03 ± 0.02 for Manzanar.

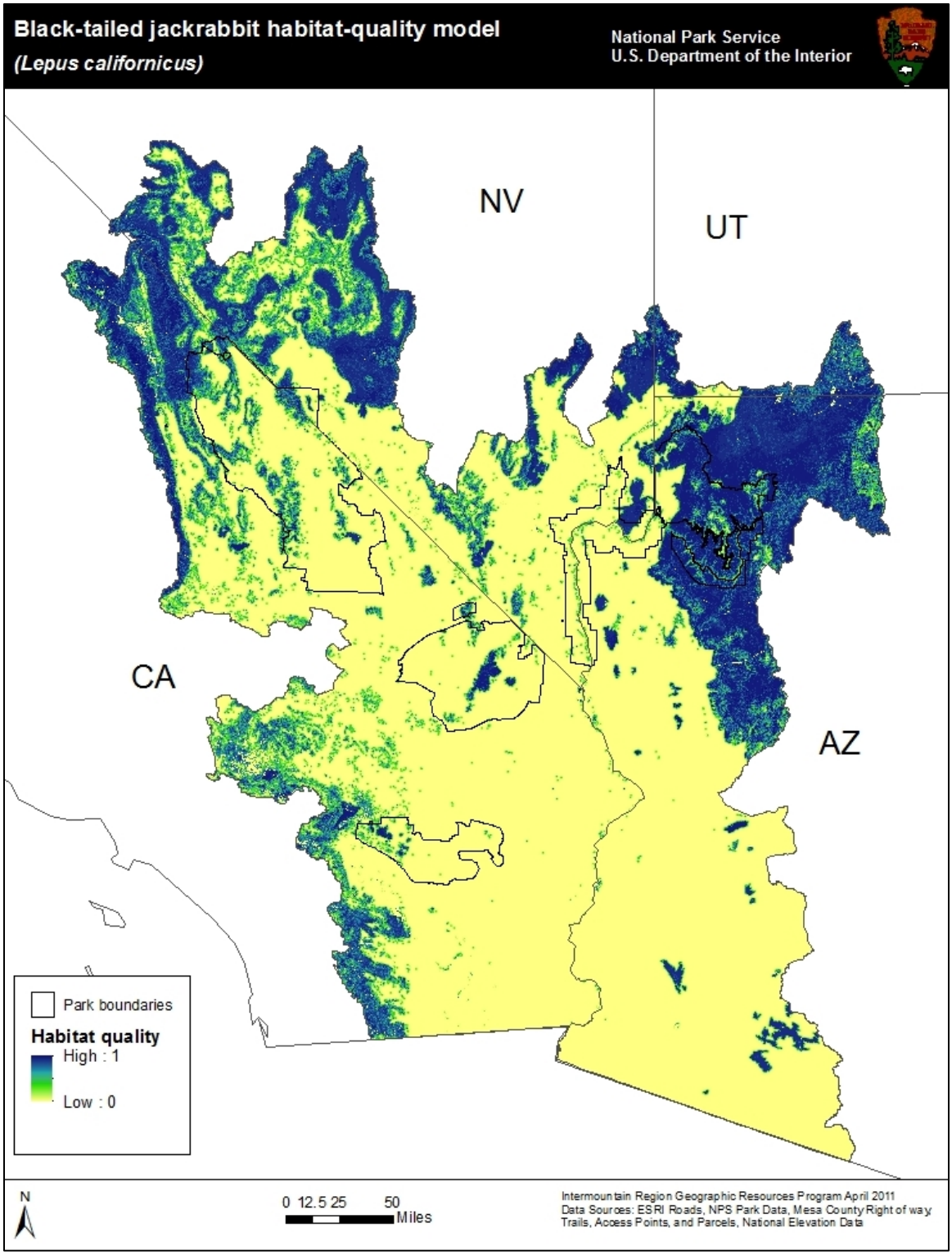


Figure 29. Modeled habitat quality for black-tailed jackrabbit.

10.B.2. Habitat quality for black-tailed jackrabbits

Black-tailed jackrabbits are the main prey of mesopredators and raptors in the southwestern United States. The species occurs in grassland and shrubland or scrubland throughout its range (Currie and Goodwin 1966, Alipayou et al. 1993, Knick and Dyer 1997, Marín et al. 2003). In periods of drought, the abundance of black-tailed jackrabbits decreases (Esque et al. 2010). Black-tailed jackrabbits typically do not occur in agricultural, developed, or recently burned areas (Fagerstone et al. 1980, Knick and Dyer 1997). Details on models of habitat quality for black-tailed jackrabbits are in Appendix 10.

10.B.3. Habitat quality for predators of desert tortoise

In rural areas, Common Ravens are associated with anthropogenic features (Kristan and Boarman 2003, 2007; Kristan et al. 2004). Occurrence of Common Ravens is positively associated with power lines, which provide nesting platforms (Steenhof et al. 1993), perches (Knight and Kawashima 1993), and roost sites (Engel and Young 1992). Reliable and often abundant sources of food for Common Ravens are associated with rural developments (Tewksbury et al. 1998), urban areas (Kristan and Boarman 2007), campgrounds (Marzluff and Neatherlin 2006), landfills (Kristan et al. 2004), and roads (Case 1978, Rolley and Lehman 1992, Knight and Kawashima 1993, Kristan et al. 2004).

Common Raven populations generally include both breeding territorial birds and nonbreeding birds that aggregate in flocks (Boarman and Heinrich 1999). The spatial distributions of breeding and nonbreeding individuals differ in the Mojave Desert (Kristan and Boarman 2003). Breeding pairs are spaced evenly, but nonbreeding birds aggregate at sites with abundant food. We developed separate models of habitat quality for breeding and nonbreeding populations (Appendix 10).

Golden Eagles have different habitat requirements during the breeding and non-breeding seasons. Golden Eagles adjust their home ranges and foraging behavior on the basis of the spatial distribution and abundance of prey (Kochert and Steenhof 2002). Power-line towers are used by Golden Eagles as nesting platforms (Steenhof et al. 1993). The species rarely is associated with grasslands, recently disturbed areas, human development, railroads, primary roads, and agricultural lands.

Although coyotes are generalists, they rarely occur in urban (Quinn 1997a, b; Riley et al. 2003; Morey 2004) or agricultural areas (Atwood et al. 2004), perhaps reflecting distributions of prey and human activity. Prey distributions differ between agricultural areas and wildlands. Gehring (2000) found lower densities of small mammals in row crops than in wildlands. Abundances of small mammals typically increase in years with above-mean precipitation (Windberg et al. 1997).

Details on models of habitat quality for predators are in Appendix 10.

10.C. Assessment results

10.C.1. Habitat quality for desert tortoises, black-tailed jackrabbits, and their predators

Modeled habitat quality for desert tortoises as modeled by Nussear et al. (2009) varied considerably across the study area (Figure 28). At the level of national parks, mean habitat quality was highest within Manzanar National Historical Site (0.60 ± 0.17), followed by Mojave National Preserve (0.56 ± 0.33), Joshua Tree National Park (0.51 ± 0.29), Lake Mead National Recreation Area (0.46 ± 0.26),

Death Valley National Park (0.22 ± 0.27), and Grand Canyon–Parashant National Monument (0.22 ± 0.33).

Our model suggested that habitat quality for Common Ravens was relatively high in the northern and southwestern parts of the study area and relatively low in the southeast (Figure 30). Mean habitat quality was > 0.6 for all parks. Habitat quality was lowest in Death Valley National Park (0.65 ± 0.37) and highest in Manzanar National Historical Site (0.99 ± 0.003). Habitat quality for the other four parks was 0.86 ± 0.27 (Mojave National Preserve), 0.73 ± 0.23 (Lake Mead National Recreation Area), 0.73 ± 0.32 (Joshua Tree National Park), and 0.75 ± 0.13 (Grand Canyon–Parashant National Monument).

Habitat for Golden Eagles was uniformly distributed across the study area (Figure 31). Mean habitat quality at the park level was 0.68 ± 0.19 (Death Valley National Park), 0.64 ± 0.20 (Joshua Tree National Park and Mojave National Preserve, 0.62 ± 0.23 (Lake Mead National Recreation Area), 0.56 ± 0.29 (Grand Canyon-Parashant National Monument), and 0.40 ± 0.12 (Manzanar National Historical Site).

Mean habitat quality for coyotes varied locally, but habitat of intermediate quality was uniformly distributed (Figure 32). Mean park-level habitat quality was 0.79 ± 0.05 (Grand Canyon–Parashant National Monument), 0.70 ± 0.06 (Mojave National Preserve), 0.69 ± 0.06 (Death Valley National Park and Joshua Tree National Park, 0.68 ± 0.08 (Lake Mead National Recreation Area), and 0.66 ± 0.07 (Manzanar National Historical Site).

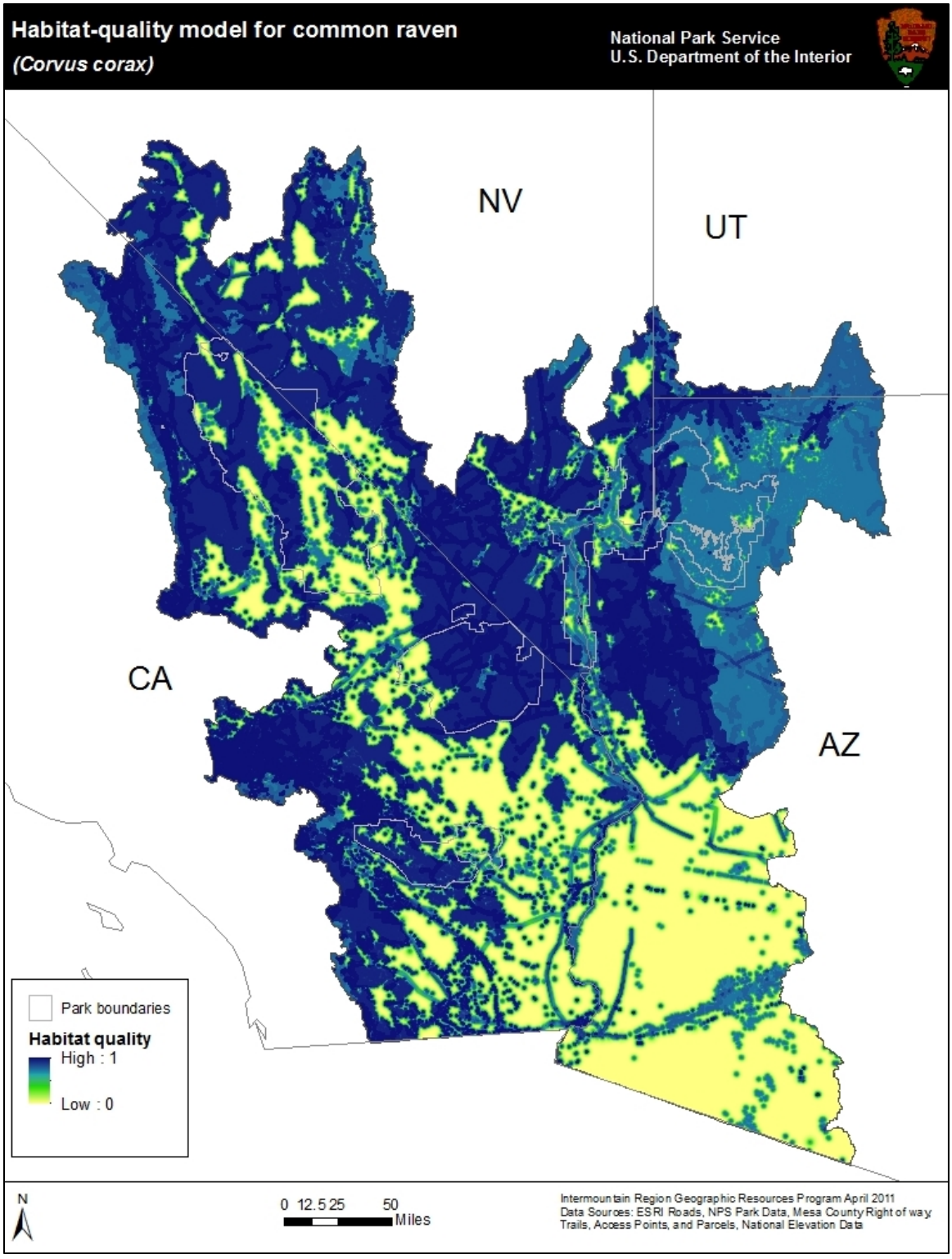


Figure 30. Modeled habitat quality for breeding and nonbreeding Common Ravens.

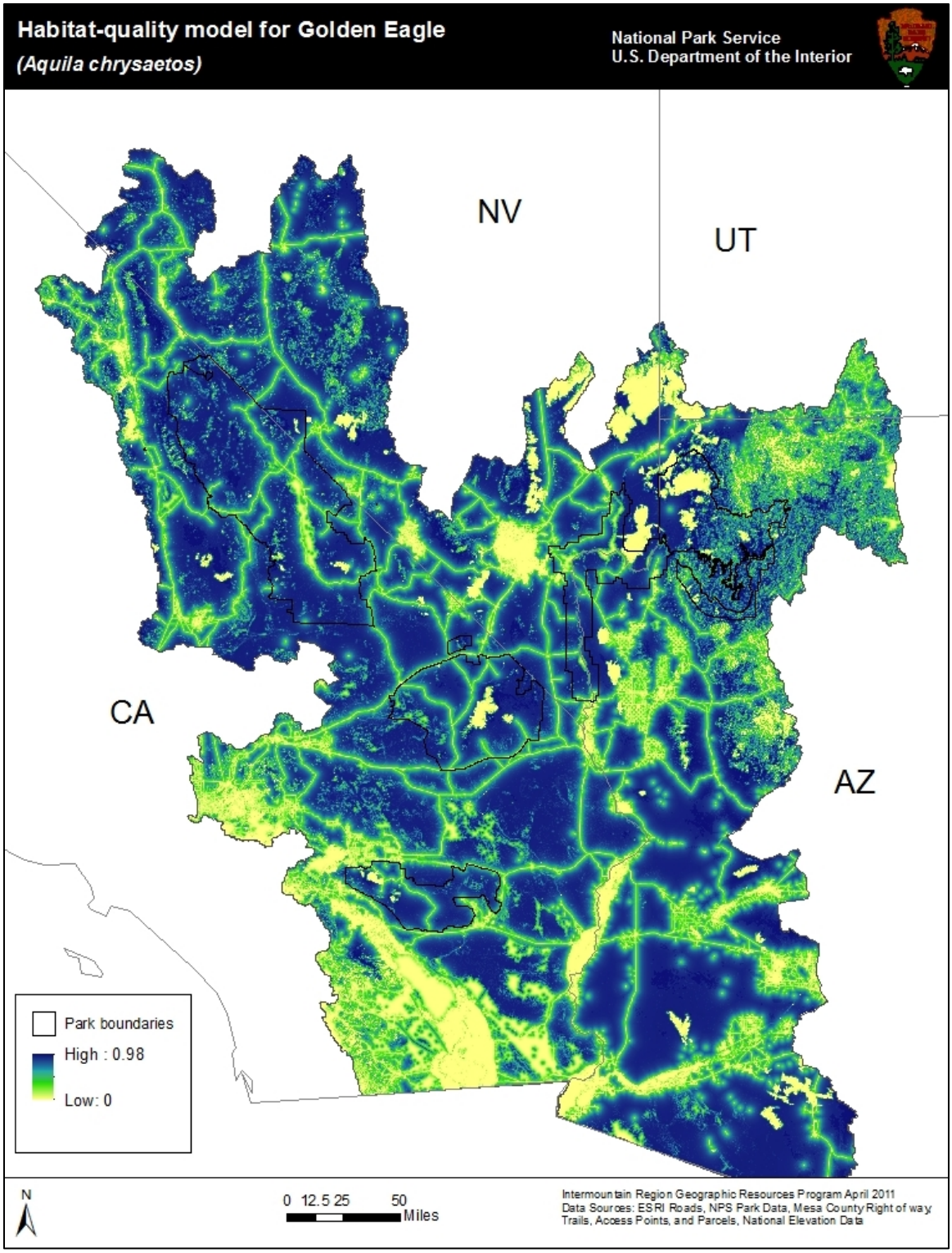


Figure 31. Modeled habitat quality for Golden Eagles.

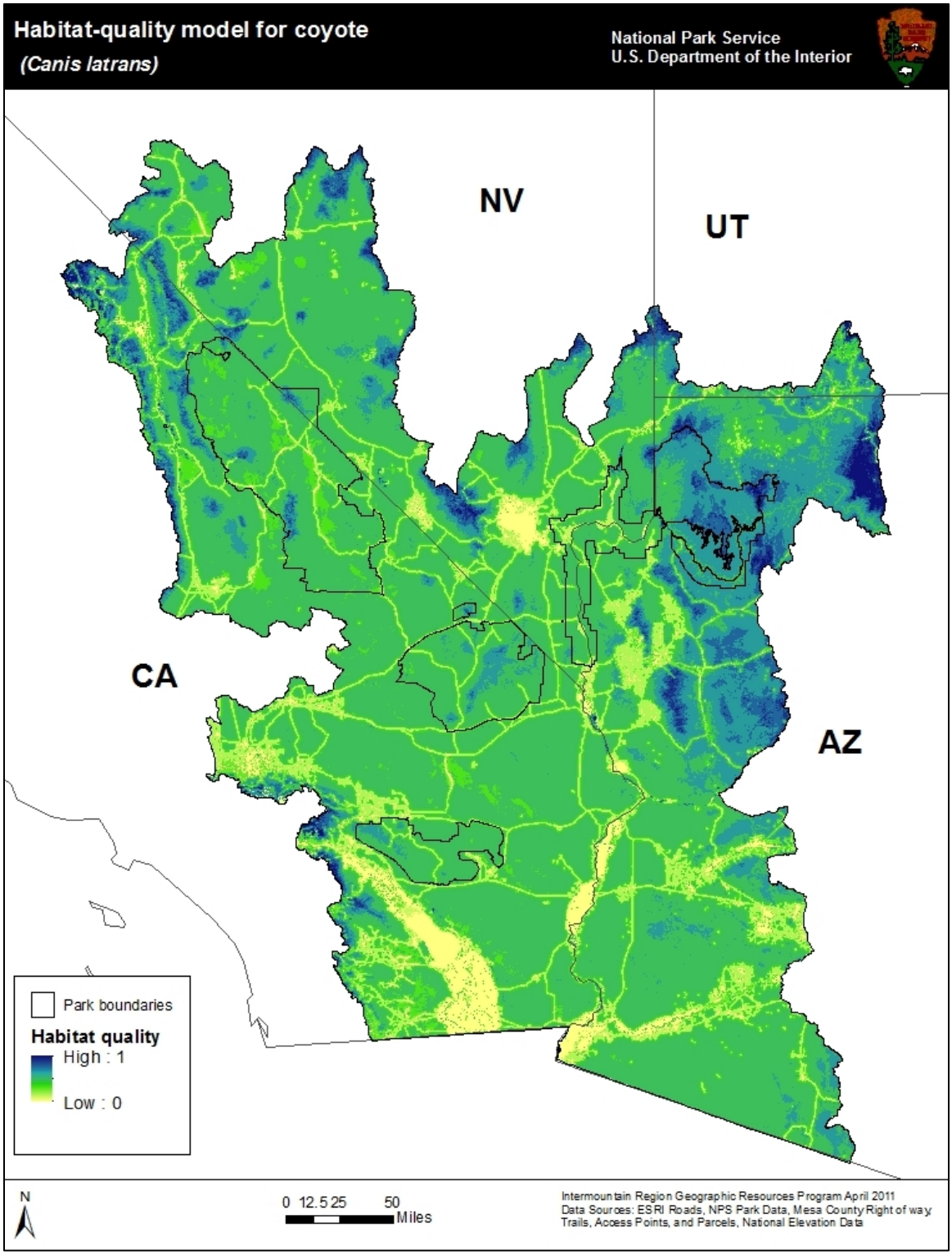


Figure 32. Modeled habitat quality for coyotes.

10.C.2. Overlap of habitat for desert tortoises, black-tailed jackrabbits, and their predators

We compared overlap of modeled habitat for desert tortoises and for black-tailed jackrabbits with modeled habitat for their predators. We evaluated overlap on the basis of two thresholds of habitat quality: ≥ 0.75 and ≥ 0.50. We assumed that a given species often would occur in areas with habitat-quality values ≥ 0.75 and sometimes would occur in areas with habitat-quality values ≥ 0.50.

Habitat quality for desert tortoises and black-tailed jackrabbits was not highly correlated (Figure 33). Habitat quality for both species was ≥ 0.75 in 0.6% of the area in which habitat quality for desert tortoises was modeled (Nussear et al. 2009). Habitat quality for both species was ≥ 0.50 in 2.6% of that area.

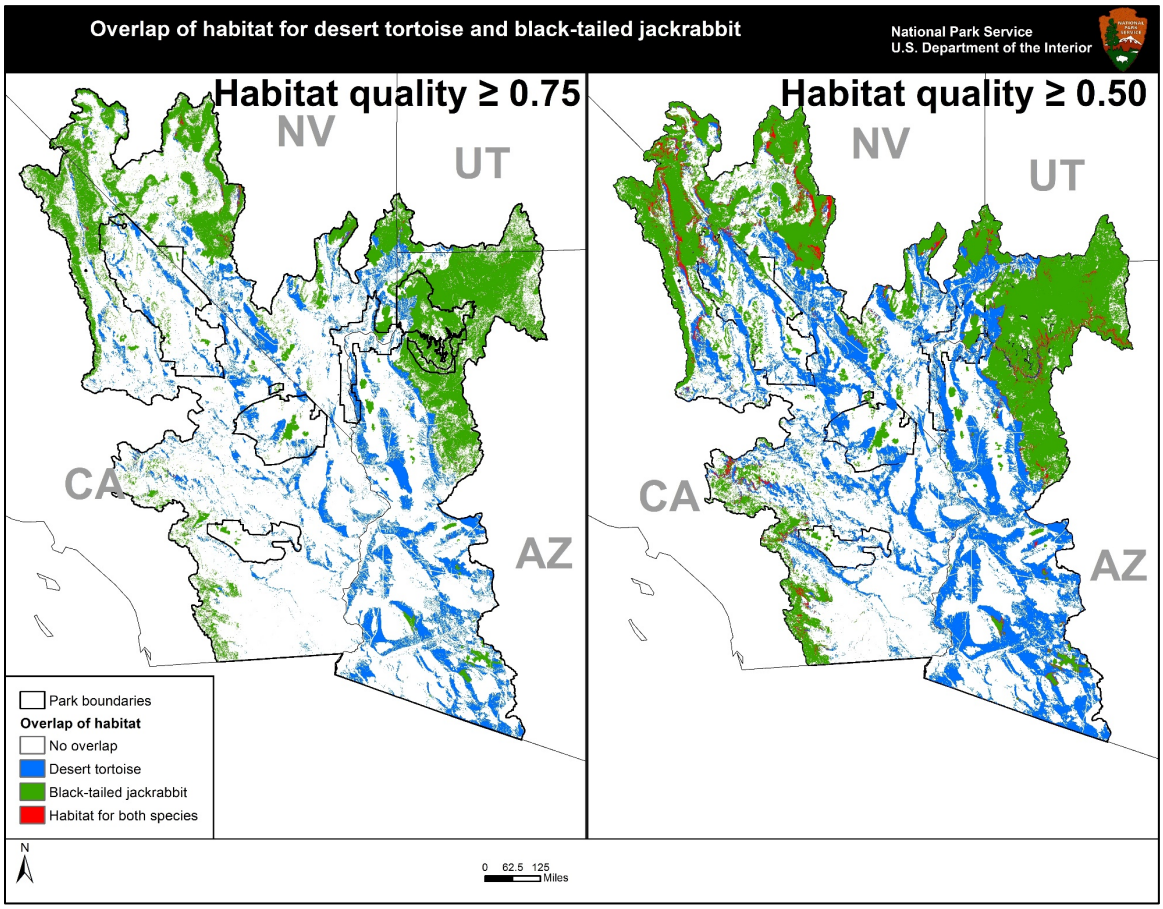


Figure 33. Overlap in modeled habitat quality for desert tortoises and black-tailed jackrabbits.

We used the resample cubic function in ArcGIS to resample our models of habitat quality for predators (native resolution 30 m) to the 1-km resolution of Nussear et al.'s (2009) model of habitat quality for desert tortoises. We then explored overlap in habitat quality for Common Ravens, Golden Eagles, and coyotes with habitat for desert tortoises and black-tailed jackrabbits. Areas where habitat quality for at least one predator was ≥ 0.75 or ≥ 0.50 covered 45.5% and 95.2%, respectively, of the area in which habitat quality for desert tortoises was modeled.

Areas in which habitat quality for all predators was ≥ 0.75 or ≥ 0.50 overlapped with 3.1% and 67.3%, respectively, of desert tortoise habitat. Areas in which habitat quality for all predators was ≥ 0.75 or ≥ 0.50 overlapped with 7.7% and 63.9%, respectively, of habitat for black-tailed jackrabbits. Areas in which habitat quality for all predators was ≥ 0.75 overlapped with 0.7% of desert tortoise habitat with quality ≥ 0.75; areas in which habitat quality for all predators was ≥ 0.50 overlapped with 70.1% of desert tortoise habitat with quality ≥ 0.50 (Figure 34). Areas in which habitat quality for all predators was ≥ 0.75 overlapped with 15.5% of black-tailed jackrabbit habitat with quality ≥ 0.75; areas in which habitat quality for all predators was ≥ 0.50 overlapped with 52.3% of black-tailed jackrabbit habitat with quality ≥ 0.50.

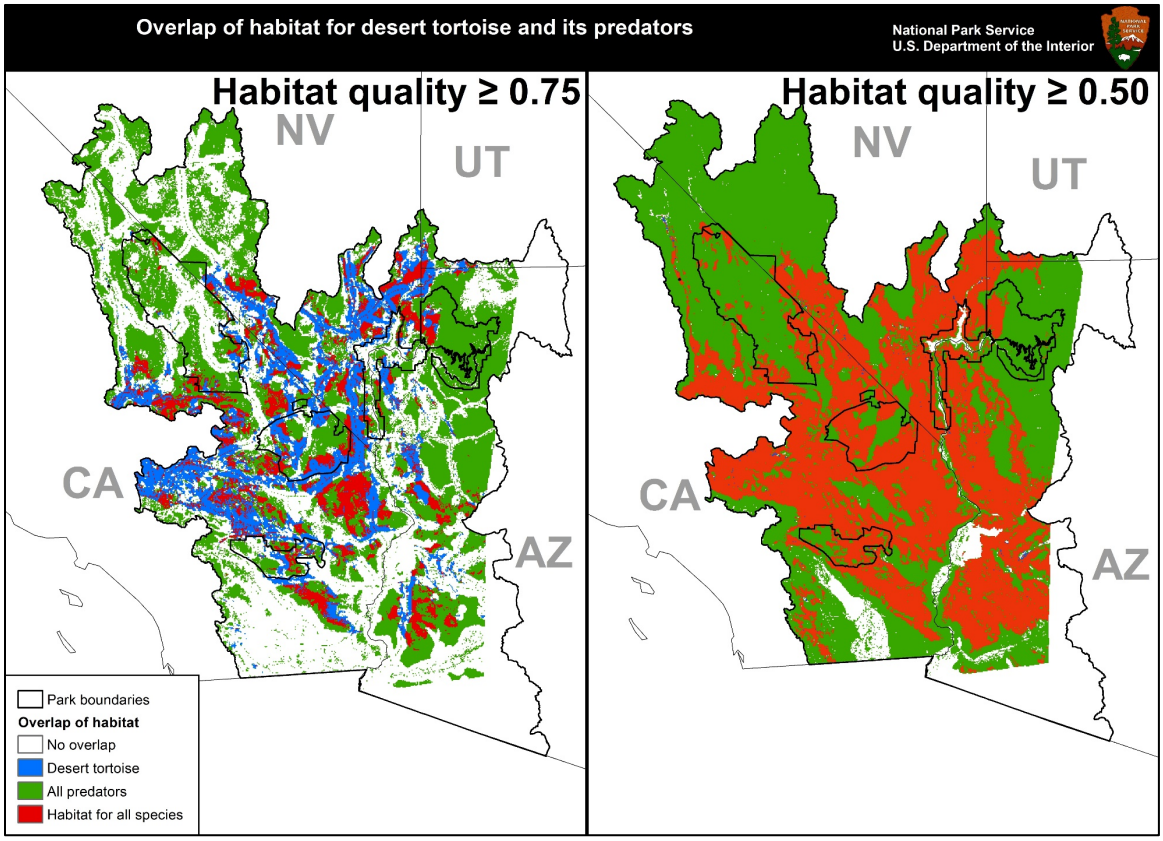


Figure 34. Overlap in modeled habitat quality for desert tortoises and three of their predators (Common Ravens, Golden Eagles, and coyotes).

10.D. Data gaps and confidence in the assessment

The habitat quality models that we developed should be considered as a first step to inform additional sampling and modeling. Because we based our models on published data and related those data to a limited set of geospatial information that was available for the full extent of the study area, our habitat quality models have two biases. First, most of the published data were collected from outside the region we modeled. This may introduce error because one cannot assume that habitat quality relations inferred from one geographic area are transferable to another geographical area. Second, data on environmental variables associated with habitat quality may only be available at coarse

resolution or may not be available across the full extent of the study area. For example, although desert tortoise space use overlaps with loamy soils (Andersen et al. 2000, Zylstra and Steidl 2009), we could not find spatial data set delineating loamy soils across the study area. Instead, we used soil depth as a proxy for loamy soil.

The projections of the habitat quality models presented here should be evaluated with data on occurrence, abundance, or demography (Rittenhouse et al. 2010). The species data will be most informative if collected across the gradient of habitat quality. Survey designs developed to examine how shrubsteppe fauna and flora in the Wyoming Basin respond to abiotic and biotic variables, including human land use (Leu et al. 2011), may be applicable to the Mojave Desert.

Chapter 11. Connectivity and adaptive capacity of desert bighorn sheep

11.A. Rationale for and information provided by the assessment

Desert bighorn sheep (*Ovis canadensis nelsoni*) are a high priority for management throughout the Mojave Desert. The species once was common throughout the region, but its abundance decreased considerably as a result of habitat loss and fragmentation, unsustainable levels of hunting, and disease. Evaluation of connectivity and adaptive capacity of desert bighorn sheep became a higher priority following an outbreak of respiratory disease that was detected at Old Dad Peak (Mojave National Preserve) in May 2013. This outbreak caused the death of at least several dozen adult bighorn sheep. One pathogen implicated in pneumonia in bighorn sheep, *Mycoplasma ovipneumoniae,* subsequently was detected in ten populations in or near the Mojave National Preserve in 2013 and 2014 (California Department of Fish and Wildlife, unpublished data). The source of the pathogens in this outbreak is unclear. Domestic livestock typically are associated with initial outbreaks of pneumonia in bighorn sheep (Besser et al. 2012), but subsequent spread may be caused by movements or dispersal of bighorn sheep. The effects of pathogens on different populations of bighorn sheep may be affected by genetic diversity at immune system genes (Epps, unpublished data). Thus, understanding connectivity among populations is relevant for predicting the spread of respiratory pathogens and potentially for maintaining the status of populations that are exposed to the pathogens. Other outbreaks recently have recently been detected in Nevada and Arizona, highlighting the potential utility of evaluating connectivity and the likelihood of disease spreading across the entire Mojave.

It is well established that conservation of a given species is more likely when patches of its habitat are not highly isolated from each other (Hanski 1991) and when individuals and genes move among populations. Connectivity generally refers to the effects of vegetation structure, configuration, and composition on the ability of animals to survive in and move throughout a given region. Assessments of connectivity may be relevant to identifying areas through which individuals can move readily, thus facilitating gene flow and colonization or recolonization of habitat patches. Maintenance of genetic diversity and gene flow generally are associated with the ability of populations or species to adapt to short-term and long-term environmental change.

11.B. Data and methods

We measured the condition of desert bighorn sheep within individual parks and across the Mojave Desert Network in three ways. First, we assessed the connectivity and genetic diversity of desert bighorn sheep populations in Joshua Tree National Park and Mojave National Preserve. Second, we assessed the strength of gene flow among populations in Death Valley National Park and Lake Mead National Recreation Area. We did not include Grand Canyon–Parashant in our analyses because data on bighorn sheep in that park were not available. Third, we assessed the potential role of different populations and corridors in maintaining genetic connectivity and demographic connectivity.

Genetic connectivity refers to the potential for gene flow among habitat patches—in this case, mountain ranges capable of supporting a population of desert bighorn sheep. Demographic

connectivity refers to the potential for recolonization of vacant habitat patches. These forms of connectivity are not equivalent for desert bighorn sheep in the Mojave Desert because males disperse further than females. Patches may be connected genetically by male dispersal, but female dispersal may be necessary to establish new populations. We considered four management actions that might maximize connectivity of the network of populations: protecting occupied habitat patches, protecting intact dispersal corridors, restoring desert bighorn sheep populations to unoccupied habitat patches, and restoring broken dispersal corridors by removing barriers.

11.B.1. Connectivity of bighorn sheep populations in and near Death Valley National Park

We capitalized on an ongoing assessment of genetic diversity and connectivity of desert bighorn sheep populations across California (except the Peninsular Range) and parts of Nevada, Arizona, and Utah that is supported by the National Park Service (PMIS 162673). From 2011 through 2013, we collected feces and tissues from carcasses at known water holes in the vicinity of Death Valley National Park (Figure 35). We did not attempt to sample bighorn sheep reported to occur on Tucki Mountain because the site is remote and genetic samples from bighorn sheep just south of that area were available.

Populations in western Death Valley and to the west of Death Valley were sampled non-invasively by J. Wehausen from 2003–2010 and by the California Department of Fish and Game from 1995–2004. Genotypes for the latter samples were generated by J. Wehausen. We pooled these data with those gathered from 2011–2013 to assess genetic diversity and structure of desert bighorn sheep populations across northern California, with a focus on connectivity within Death Valley.

We used GENEPOP (Rousset 2008) to evaluate whether loci in each population met the assumption of random mating (i.e., were in Hardy-Weinberg equilibrium) and to test whether alleles at different loci that are in close proximity were non-randomly associated (i.e., to assess evidence of linkage disequilibrium). We estimated genetic diversity (number of alleles) in each population with FSTAT (Goudet 1995). We used ARLEQUIN (Schneider et al. 2000) to estimate average expected heterozygosity (a standard measure of genetic diversity that is calculated on the basis of the number and proportional representation of different alleles) for each population.

We used STRUCTURE (Pritchard et al. 2000) to assess the genetic structure of desert bighorn sheep populations within Death Valley and surrounding areas. We used FSTAT to assess genetic differentiation among populations (F_{ST}) as a measure of long-term gene flow. We used BIMr (Faubet 2007) to estimate recent rates of migration among populations (Epps et al. 2013a, b). We contrasted long-term and recent estimates to elucidate whether genetically similar populations are currently linked by gene flow or whether they historically may have either exchanged individuals or served as sources and recipients of colonists.

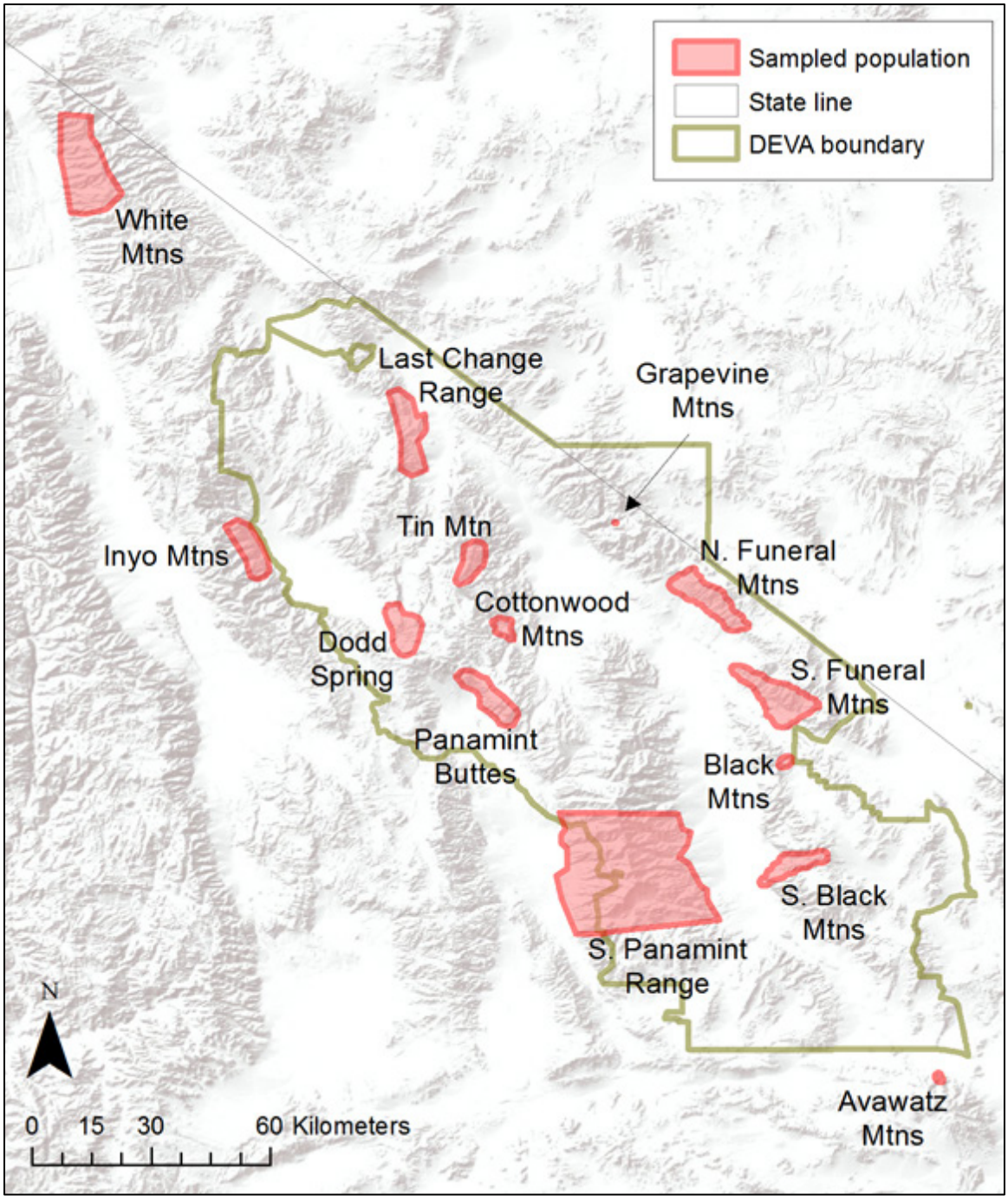


Figure 35. Populations of desert bighorn sheep within and near Death Valley National Park (DEVA) that were included in assessments of genetic structure. Red polygons represent the approximate area from which samples were collected. Samples from Dodd Spring were not included in assessments because they likely originated from transient rams.

11.B.2. Connectivity of bighorn sheep populations in and near Lake Mead National Recreation Area

Jaeger and Wehausen (2012) assessed genetic diversity (nuclear DNA, 16 microsatellite loci; 515 bp of mitochondrial DNA control region), genetic structure, and recent gene flow (Jaeger and Wehausen 2013) among desert bighorn sheep populations in southern Nevada, including Lake Mead (see Jaeger and Wehausen 2012, page 5). Their methods for collecting and analyzing genetic samples and data are described in Appendix 11.

11.B.3. Connectivity of desert bighorn sheep populations in and near Joshua Tree National Park and Mojave National Preserve and the metapopulation in the Mojave Desert

Again, we capitalized on an ongoing assessment of genetic diversity and connectivity of desert bighorn sheep populations across California (except the Peninsular Range) and parts of Nevada, Arizona, and Utah that is supported by the National Park Service (PMIS 162673). Detailed methods and results for measure of the condition of desert bighorn sheep are described in full in Creech et al. (2014). Our analysis had four steps (for more detail, see Appendix 12). First, we estimated connectivity on the basis of effective distance between habitat patches (mountain ranges that can support a population) in the Mojave Desert. Effective distance reflects both geographic distance and relative resistance to dispersal and is a more accurate estimator of realized population connectivity than straight-line distance. Areas with > 15% slope are ten times less resistant to dispersal of desert bighorn sheep than areas with < 15% slope because steep terrain allows escape from predators (Epps et al. 2007).

Second, we combined among-patch connectivity estimates with genetic estimates of dispersal thresholds to build models of genetic and demographic connectivity across the network of populations. Networks included nodes, which represented occupied habitat patches, and edges, which represented connections among patches.

Third, we estimated how the addition or loss of habitat patches and corridors affected estimated network-level genetic and demographic connectivity. We calculated two network-level metrics after adding or removing each patch or corridor from the network. The first metric, mean weighted closeness, is an index of the average amount of gene flow expected between any two patches in the network along the most efficient path between them. The second metric, effectively connected pairs, is the number of pairs of patches connected by a total effective distance less than the estimated maximum effective dispersal distance, and reflects the short-term potential for connectivity between a given patch and neighboring patches. Additions and removals that yielded the largest increases in connectivity identified the patches and corridors that might be highest priorities for management action.

Fourth, we explored sensitivity of rankings to connectivity type, network metric, and modeling error. For each management action, we calculated the correlation between ranks from the genetic and demographic networks to determine how the type of connectivity affected the prioritization process. Similarly, we calculated the correlation between ranks based on mean weighted closeness and effectively connected pairs to determine the effect of network metric.

11.C. Assessment results

11.C.1. Connectivity of bighorn sheep populations in and near Death Valley National Park

We found no evidence of non-random mating (violations of Hardy-Weinberg equilibrium) across loci or populations. Genetic diversity of populations in or near Death Valley was moderate to high in comparison with other published estimates of genetic diversity for desert bighorn sheep (Table 30).

Table 30. Genetic diversity (average expected heterozygosity, He) of desert bighorn sheep populations within or near Joshua Tree National Park, Mojave National Preserve, Death Valley National Park, and Lake Mead National Recreation Area. Estimates were derived from 12–16 putatively neutral microsatellite markers (nuclear DNA). Loci overlapped but were not identical among populations. Sample size refers to number of individuals; each population was within or near the National Park Service unit or units listed.

National Park Service unit(s)	Mountain range or population	H_e	Sample size	Number of loci	Source
Lake Mead	Black Mountains (Arizona)	0.62–0.63	14–24	16[1]	Jaeger & Wehausen 2012
Lake Mead	Muddy	0.59	34	16[1]	Jaeger & Wehausen 2012
Lake Mead	Black Mountains (Nevada)	0.52	24	16[1]	Jaeger & Wehausen 2012
Lake Mead	River Mountains	0.63	46	16[1]	Jaeger & Wehausen 2012
Lake Mead	Eldorado–Gold Strike	0.63	20	16[1]	Jaeger & Wehausen 2012
Lake Mead	Eldorado–Black Canyon	0.66	21	16[1]	Jaeger & Wehausen 2012
Lake Mead	Eldorado–Huse Spring	0.66	21	16[1]	Jaeger & Wehausen 2012
Lake Mead	Newberry	0.66	15	16[1]	Jaeger & Wehausen 2012
Lake Mead	Highland	0.66	21	16[1]	Jaeger & Wehausen 2012
Lake Mead and Mojave	McCullough	0.66	43	16[1]	Jaeger & Wehausen 2012
Mojave	Castle	0.62	19	16[1]	Jaeger & Wehausen 2012
Mojave and Death Valley	Kingston	0.61	8	16[1]	Jaeger & Wehausen 2012
Mojave	Clark	0.59	17	16[1]	Jaeger & Wehausen 2012
Mojave	Spring Mountains–Little Devil	0.57	20	16[1]	Jaeger & Wehausen 2012
Lake Mead, Mojave, and Death Valley	Spring Mountains– Brownstone Basin	0.61	17	16[1]	Jaeger & Wehausen 2012

[1] Loci included: AE16, MAF65, TGLA387, OarFCB266, MAF209, MAF48, HH47, MAF36, HH62, OarFCB11, OarFCB304, CP20, OarFCB193, MAF33, AE129, TCRBV62, MMP9, BM4513, SOMA, OarJMP29. References for these loci in Jaeger & Wehausen (2012).

[2] Loci included: AE16, MAF65, TGLA387, OarFCB266, MAF209, MAF48, MAF36, HH62, OarFCB11, OarFCB304, OarFCB193, MAF33, AE129, TCRBV62, OarJMP29.

[3] Loci included: AE16, MAF65, OarFCB266, MAF209, MAF48, HH47, MAF36, HH62, OarFCB11, OarFCB304, CP20, MAF33, FCB128, DS52

Table 30 (continued). Genetic diversity (average expected heterozygosity, He) of desert bighorn sheep populations within or near Joshua Tree National Park, Mojave National Preserve, Death Valley National Park, and Lake Mead National Recreation Area. Estimates were derived from 12–16 putatively neutral microsatellite markers (nuclear DNA). Loci overlapped but were not identical among populations. Sample size refers to number of individuals; each population was within or near the National Park Service unit or units listed.

National Park Service unit(s)	Mountain range or population	H_e	Sample size	Number of loci	Source
Mojave and Death Valley	Avawatz	0.66	12	15[2]	C. Epps, unpublished data
Death Valley	Black Mountains (Death Valley, California)	0.63	12	15[2]	C. Epps, unpublished data
Death Valley	Cottonwood Mountains	0.62	14	15[2]	C. Epps, unpublished data
Death Valley	Grapevine Mountains	0.69	23	15[2]	C. Epps, unpublished data
Death Valley	northern Funeral Mountains	0.68	30	15[2]	C. Epps, unpublished data
Death Valley	southern Funeral Mountains	0.68	23	15[2]	C. Epps, unpublished data
Death Valley	southern Panamint Range	0.62	26	15[2]	J. Wehausen, unpublished data
Death Valley	Panamint Buttes	0.57	12	15[2]	J. Wehausen, unpublished data
Death Valley	southern Black Mountains	0.59	25	15[2]	Epps & Wehausen, unpublished data
Death Valley	Tin Mountain	0.67	22	15[2]	J. Wehausen, unpublished data
Death Valley	Last Chance Range	0.68	21	15[2]	J. Wehausen, unpublished data
Death Valley	Inyo Mountains	0.61	31	15[2]	J. Wehausen, unpublished data
Death Valley	White Mountain	0.54	25	15[2]	J. Wehausen, unpublished data
Joshua Tree	Coxcomb Mountains	0.62	7	14[3]	Epps et al. 2005

[1] Loci included: AE16, MAF65, TGLA387, OarFCB266, MAF209, MAF48, HH47, MAF36, HH62, OarFCB11, OarFCB304, CP20, OarFCB193, MAF33, AE129, TCRBV62, MMP9, BM4513, SOMA, OarJMP29. References for these loci in Jaeger & Wehausen (2012).

[2] Loci included: AE16, MAF65, TGLA387, OarFCB266, MAF209, MAF48, MAF36, HH62, OarFCB11, OarFCB304, OarFCB193, MAF33, AE129, TCRBV62, OarJMP29.

[3] Loci included: AE16, MAF65, OarFCB266, MAF209, MAF48, HH47, MAF36, HH62, OarFCB11, OarFCB304, CP20, MAF33, FCB128, DS52

Table 30 (continued). Genetic diversity (average expected heterozygosity, He) of desert bighorn sheep populations within or near Joshua Tree National Park, Mojave National Preserve, Death Valley National Park, and Lake Mead National Recreation Area. Estimates were derived from 12–16 putatively neutral microsatellite markers (nuclear DNA). Loci overlapped but were not identical among populations. Sample size refers to number of individuals; each population was within or near the National Park Service unit or units listed.

National Park Service unit(s)	Mountain range or population	H_e	Sample size	Number of loci	Source
Joshua Tree	Eagle Mountains–Buzzard Spring	0.65	17	14[3]	Epps et al. 2005
Joshua Tree	Eagle Mountains–Lost Palms Oasis	0.63	14	14[3]	Epps et al. 2005
Joshua Tree	Little San Bernardino Mountains	0.63	12	14[3]	Epps et al. 2005
Joshua Tree	Queen Mountains	0.59	11	14[3]	Epps et al. 2005

[1] Loci included: AE16, MAF65, TGLA387, OarFCB266, MAF209, MAF48, HH47, MAF36, HH62, OarFCB11, OarFCB304, CP20, OarFCB193, MAF33, AE129, TCRBV62, MMP9, BM4513, SOMA, OarJMP29. References for these loci in Jaeger & Wehausen (2012).

[2] Loci included: AE16, MAF65, TGLA387, OarFCB266, MAF209, MAF48, MAF36, HH62, OarFCB11, OarFCB304, OarFCB193, MAF33, AE129, TCRBV62, OarJMP29.

[3] Loci included: AE16, MAF65, OarFCB266, MAF209, MAF48, HH47, MAF36, HH62, OarFCB11, OarFCB304, CP20, MAF33, FCB128, DS52.

However, genetic diversity for the samples collected in the southern end of Death Valley (southern Panamint Range, Black Mountains, and southern Black Mountains) was lower than for populations to the north (Grapevine Mountains and Funeral Mountains), suggesting that over time, populations in the south have been smaller or more isolated.

On the basis of recent estimates of gene flow, we found substantial genetic differentiation between desert bighorn sheep populations in eastern and western Death Valley (eastern: Black Mountains, southern Black Mountains, southern Funeral Mountains, northern Funeral Mountains, and Grapevine Mountains; western: White Mountains, Inyo Mountains, Cottonwood Mountains, Last Chance Range, southern Panamint Range, Panamint Butte, Tin Mountain) (Figure 36a). Some populations were related to both the eastern and western Death Valley clusters, including Avawatz Mountains (more strongly associated with eastern Death Valley populations), southern Black Mountains (strongly associated with eastern Death Valley populations but related to western populations), and Cottonwood Mountains (more strongly associated with western Death Valley populations but related to eastern populations). The assignments to eastern and western clusters suggested that gene flow has occurred around the northern and southern ends of Death Valley. However, the clear division between east and west suggested that little gene flow has occurred across Death Valley itself (Figure 36a).

The White Mountains, Inyo Mountains, and southern Panamint Range populations each formed distinct clusters, as did the combination of Grapevine, northern Funeral Mountains, and southern Funeral Mountains (Figure 36b). The Last Chance Range also was relatively distinct. The Cottonwood Mountains and Panamint Buttes formed a distinctive cluster, although individuals partially were assigned to the southern Panamint Range, indicating gene flow to the south, and Tin Mountain individuals were assigned to the Cottonwood/Panamint Butte Cluster, and to a lesser extent, the Last Chance Range. Two individuals in the Cottonwood Mountains had strong partial assignment to the southern Panamint Range, indicating that they could be descended from recent migrants from that population or possibly from the unsampled Tucki Mountain population. The southern Black Mountains population was relatively distinct. Ancestry of individuals in the Black Mountains population included both the southern Black Mountains and Funeral Mountains clusters, indicating that gene flow occurs between the Black Mountains and Funeral Mountains. The Avawatz Mountains population was most closely related to the southern Black Mountains cluster and also related to the southern Panamint Range population.

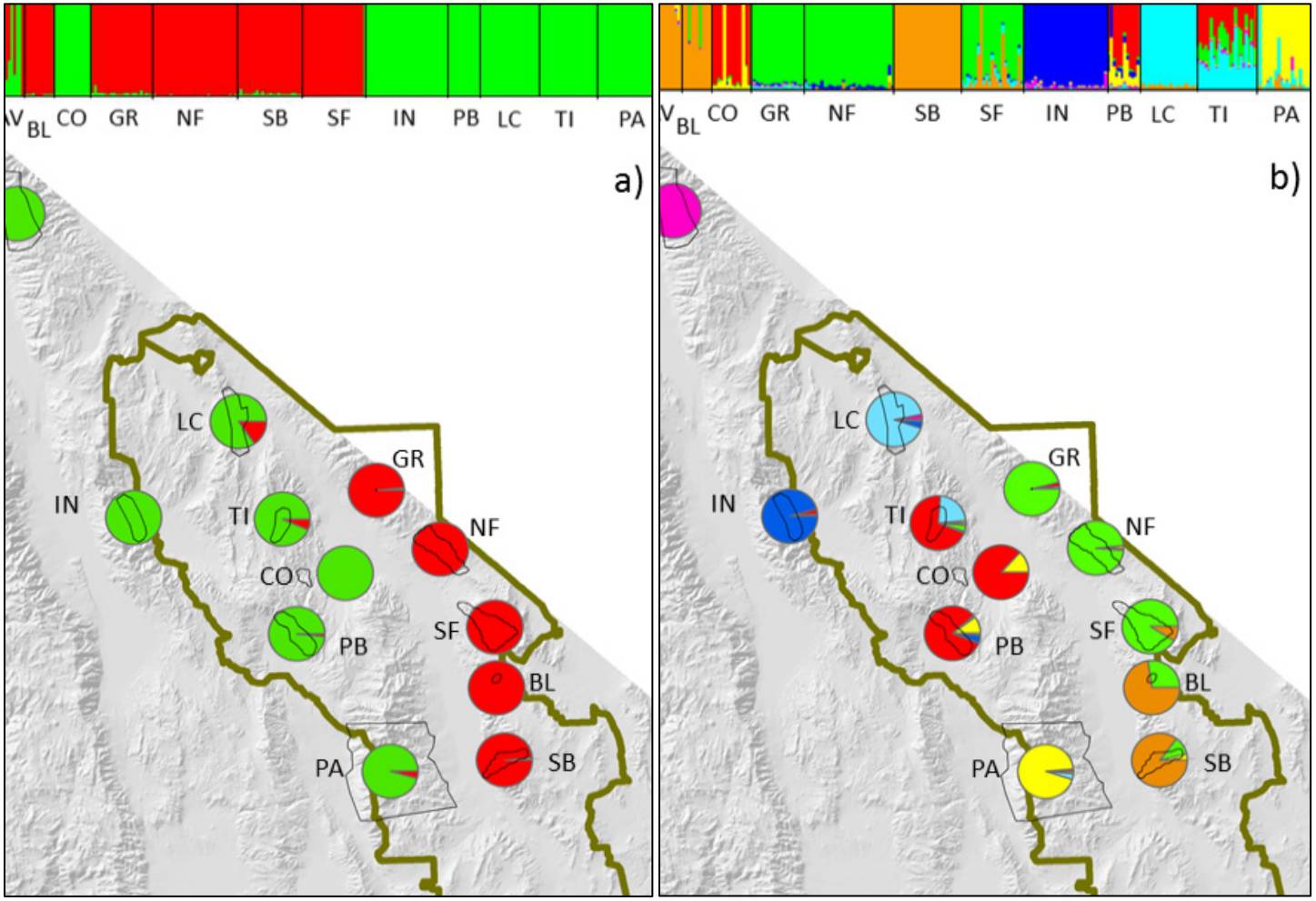


Figure 36. Clustering of genotypes of desert bighorn sheep from populations in or near Death Valley National Park. Thirteen populations genotyped at 15 loci, (a) two clusters (b) seven clusters. Colors represent clusters of individuals. Each individual is represented by a single bar in each figure; the bar represents the probability of assignment to each cluster. Single colors indicate strong assignment to a given cluster. Sampling locations (populations) are indicated on the x-axis and separated by black lines. AV, Avawatz Mountains; BL, Black Mountains; CO, Cottonwood Mountains; GR, Grapevine Mountains; IN, Inyo Mountains; LC, Last Chance Range; NF, Northern Funeral Mountains; PA, southern Panamint Range; PB, Panamint Buttes; SB, Southern Black Mountains; SF, Southern Funeral Mountains; TI, Tin Mountain; WH, White Mountains.

We found evidence of strong gene flow and dispersal between the Grapevine and northern and southern Funeral Mountains populations and between the Black and southern Black Mountains populations, and fairly strong gene flow and dispersal between the Black and Funeral Mountains. Our analyses suggested that bighorn sheep move in a stepwise manner between Cottonwood Mountains and Tin Mountain and between Tin Mountain and the Last Chance Range in western Death Valley. However, the Panamint Buttes population was relatively strongly differentiated from populations in the southern Panamint Range and the Cottonwood Mountains, all in western Death Valley (Table 31).

Our analyses indicated high, recent gene flow between the Grapevine and northern Funeral Mountains and northern Funeral to southern Funeral Mountains, more restricted recent gene flow between the Black and Funeral Mountains, and negligible recent gene flow among other populations (Table 31).

Table 31. Genetic structure of desert bighorn sheep populations in northern portions of their range in California, including Death Valley National Park, on the basis of 15 microsatellite loci[1] (above diagonal), and estimates of recent gene flow (migration rate, m) among a subset of those populations (below diagonal; blank cells represent connections that were not estimated). Sample sizes (number of individuals) in boldface on the diagonal. AV, Avawatz Mountains; BL, Black Mountains; CO, Cottonwood Mountains; GR, Grapevine Mountains; IN, Inyo Mountains; LC, Last Chance Range; NF, northern Funeral Mountains; PA, southern Panamint Range; PB, Panamint Buttes; SB, southern Black Mountains; SF, southern Funeral Mountains; TI, Tin Mountain; WH, White Mountains.

Subset	AV	BL	CO	GR	IN	LC	NF	PA	PB	SB	SF	TI	WH
AV	**12**[2]	0.070	0.120	0.110	0.130	0.100	0.090	0.110	0.130	0.070	0.090	0.100	0.160
BL	<0.010	**12**[2]	0.150	0.070	0.150	0.100	0.060	0.130	0.200	0.010	0.060	0.080	0.190
CO	<0.010	<0.010	**14**[2]	0.110	0.120	0.110	0.120	0.10	0.130	0.170	0.110	0.050	0.210
GR	<0.010	<0.010	<0.010	**23**[2]	0.120	0.090	0.020	0.120	0.150	0.120	0.020	0.090	0.220
IN	–	–	–	–	**31**[2]	0.080	0.130	0.110	0.110	0.160	0.120	0.080	0.200
LC	<0.010	<0.010	<0.010	<0.010	–	**21**[2]	0.100	0.110	0.120	0.110	0.090	0.050	0.160
NF	<0.010	<0.010	<0.010	0.033	–	<0.010	**30**[2]	0.100	0.170	0.100	0.020	0.090	0.190
PA	<0.010	<0.010	<0.010	<0.010	–	<0.010	<0.010	**26**[2]	0.140	0.140	0.080	0.080	0.210
PB	<0.010	<0.010	<0.010	<0.010	–	<0.010	<0.010	<0.010	**12**[2]	0.180	0.140	0.090	0.220
SB	<0.010	<0.010	<0.010	<0.010	–	<0.010	<0.010	<0.010	<0.010	**25**[2]	0.090	0.110	0.180
SF	<0.010	<0.010	<0.010	<0.010	–	<0.010	0.102	<0.010	<0.010	<0.010	**23**[2]	0.080	0.200
TI	<0.010	<0.010	<0.010	<0.010	–	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	**22**[2]	0.150
WH	–	–	–	–	–	–	–	–	–	–	–	–	**25**[2]

[1] Loci included: AE16, MAF65, TGLA387, OarFCB266, MAF209, MAF48, MAF36, HH62, OarFCB11, OarFCB304, OarFCB193, MAF33, AE129, TCRBV62, OarJMP29.

[2] Sample sizes (number of individuals) are bold.

11.C.2. Connectivity of bighorn sheep populations in and near Lake Mead National Recreation Area

Genetic diversity of most populations in Lake Mead was relatively high and similar to that of populations in Death Valley, Mojave National Preserve, and Joshua Tree (Table 30). However, there is evidence that genetic diversity in the River Mountains has begun to decrease in response to restriction of gene flow between the River and Eldorado Mountains by Highway 93 (Jaeger and Wehausen 2012, 2013). Although desert bighorn sheep still are reported to cross Highway 93, genetic data suggested that these crossings reflect movements of desert bighorn sheep from the Eldorado Mountains rather than interaction between animals from the River and Eldorado Mountains. The McCullough Range population also may be becoming isolated.

11.C.3. Connectivity of desert bighorn sheep populations in and near Joshua Tree National Park and Mojave National Preserve and the metapopulation in the Mojave Desert

There were considerable differences between networks that were based on genetic connectivity and networks that were based on demographic connectivity. Levels of both short-term and long-term connectivity were much higher in the genetic network. Therefore, managing to maintain genetic connectivity among desert bighorn sheep populations would not necessarily maintain the potential for natural recolonization. Similarly, strong connections for gene flow might be overlooked if only colonization potential was considered. Our results suggested that long, multi-step dispersal paths, which maximize connectivity over the long term, are not necessarily those that maximize local dispersal and thus short-term connectivity. However, we were able to identify locations where restoration of genetic or demographic connectivity via mitigation of anthropogenic barriers might be particularly effective for restoring either gene flow or colonization potential.

11.C.4. Inferences and applications

11.C.4.a. Connectivity of bighorn sheep populations in and near Death Valley National Park

Unlike the most isolated populations in the Mojave Desert (Appendix 2 in Epps et al. 2005), no population in Death Valley or the vicinity had extremely low levels of genetic diversity. This suggests that no groups of desert bighorn sheep that we sampled have been isolated for long periods of time. However, strong and hierarchical genetic structure across the system suggested that distance, topography, and possibly local extinction and colonization have created many distinct populations of desert bighorn sheep in the region.

We detected strong gene flow and weak genetic structure among populations in eastern Death Valley, particularly between the Grapevine and Funeral Mountains, where genetic diversity was highest. Gene flow was more restricted and genetic diversity was slightly lower between the Funeral and Black Mountains. In western Death Valley, gene flow among populations appeared to be lower; genetic structure in some cases was much greater than among populations in eastern Death Valley that are separated by comparable distances. However, even some distant populations (e.g. southern Panamint Range and Last Chance Range, separated by ~123 km) had moderate genetic structure. This suggests that desert bighorn sheep historically ranged through the relatively continuous habitat from the Panamint Range north to the Last Chance Range.

Nevertheless, genetic structure within some pairs of populations within the cluster of populations in western Death Valley was high, suggesting that local genetic structure has developed in response to population or extinction-colonization dynamics (e.g., extirpation followed by recolonization) or recent restriction of gene flow. Results suggested that populations in Inyo Mountains, Tin Mountain, Last Chance Range, and southern Panamint Range may have been present and relatively stable for a long period of time, whereas the second set of populations may have been extirpated and recolonized more frequently. There was no evidence of recent gene flow between White Mountains and Death Valley (Table 31, Figure 35), and White Mountains had the lowest genetic diversity reported for any population in the northern range of desert bighorn sheep in California (i.e., north of Interstate 15).

Gene flow across Death Valley appears to be rare. However, gene flow likely occurred historically around the north and south ends of the valley. For instance, the population in Avawatz Mountains to the south was related to both the southern Panamint and South Black populations (Figs. 35, 36). At the north end of the valley, there was little evidence of recent interaction between the Grapevine Mountains population and the Last Chance Range or Tin Mountain populations. Historically, desert bighorn sheep would have had access to water sources in the vicinity of Scotty's Castle at the north end of the Grapevine Mountains, which would increase the probability of gene flow around the north end of Death Valley, but our field studies in 2012 did not yield evidence of recent use. Prolonged human activity in that area, fencing, and historical presence of livestock may have resulted in abandonment of that water source by desert bighorn sheep. However, occasional or future use of water at the site by desert bighorn sheep is possible, and we may not have located all sources of water in the immediate area.

Given that this system has no large fenced highways or canals to limit gene flow (Epps et al. 2005), the primary means of maintaining genetic diversity and gene flow is to maintain the desert bighorn sheep populations themselves. This requires maintaining access by desert bighorn sheep to water sources throughout the system and limiting exposure to domestic livestock, particularly domestic sheep and goats, which can be the source of fatal respiratory illness (Wehausen et al. 2011). Additionally, monitoring would be necessary to determine whether respiratory illness observed in bighorn sheep in the Kingston Range southeast of Death Valley in January 2014 (R. Abella, California Department of Fish and Wildlife, personal communication) eventually could spread into the Death Valley system.

11.C.4.b. Connectivity of bighorn sheep populations in and near Lake Mead National Recreation Area

Jaeger and Wehausen (2012, 2013) recommended three actions to conserve connectivity of bighorn sheep populations in and near Lake Mead National Recreation Area. First, facilitate natural movement of desert bighorn sheep between the Eldorado Mountains and River Mountains, possibly by improving crossings structures. Second, build a crossing structure across Interstate 515/Highway 93 at Railroad Pass to facilitate movement of desert bighorn sheep between the River and McCullough Mountains. Third, preserve the genetic link between the McCullough Range and the Eldorado Mountains that is maintained by movement of desert bighorn sheep through small patches

of habitat in the Highland Range and the hills at the south end of Eldorado Valley, across Highway 95.

11.C.4.c. Connectivity of desert bighorn sheep populations in and near Joshua Tree National Park and Mojave National Preserve and the metapopulation in the Mojave Desert

Habitat patches and corridors associated with Mojave National Preserve and Joshua Tree were among the highest-ranked features for maintenance of connectivity in our analysis. Many of the highest-priority patches were in Mojave National Preserve (Table 32, Figs. 37–41). Most of the highest-priority corridors were associated with patches in Mojave National Preserve (Table 33, Figs. 38–41). Several restorable corridors associated with patches in Mojave National Preserve or Joshua Tree received high ranks (Table 34, Figs. 42–45). The Pinto Mountains patch, which lies along the northern boundary of Joshua Tree, was the highest-priority restorable patch for both network metrics and both connectivity types (Table 35, Figs. 46–49).

Table 32. Ranks of patches in the genetic and demographic networks of desert bighorn sheep on the basis of effectively connected pairs (ECP; a measure of short-term connectivity) and mean weighted closeness (MWC; a measure of long-term connectivity). Delta indicates the change in connectivity associated with adding or removing the patch from the network. Patch abbreviations in Figure 37.

Genetic Network ECP			Genetic Network MWC			Demographic Network ECP			Demographic Network MWC		
Patch	**Δ**	**Rank**	**Patch**	**Δ**	**Rank**	**Patch**	**Δ**	**Rank**	**Patch**	**Δ**	**Rank**
PRO[1]	0.205	1	COX[2]	0.162	1	GRA[1]	0.231	1	PRO[1]	0.203	1
GRA[1]	0.180	2	IRO	0.142	2	PRO[1]	0.215	2	GRA[1]	0.190	2
PCC[1]	0.172	3	PRO[1]	0.134	3	NBR	0.200	3	PCC[1]	0.161	3
NBR	0.156	4	OWO	0.130	4	CAD	0.138	4	NBR	0.154	4
CAD	0.115	5	PCC[1]	0.108	5	PCC[1]	0.123	5	CAD	0.105	5
COX[2]	0.082	7	PRG	0.077	6	KME	0.108	6.5	KME	0.099	6
KME	0.082	7	GRA[1]	0.074	7	WHA[1]	0.108	6.5	CSS	0.082	7
SHE	0.082	7	SHE	0.067	8	CSS	0.092	8	WHA[1]	0.064	8
AVA	0.074	9.5	NBR	0.064	9	COX[2]	0.077	11	SHE	0.054	9
WHA[1]	0.074	9.5	EMO[2]	0.054	10	EMO[2]	0.077	11	SSO[1]	0.045	10
CSS	0.066	13	MAR	0.052	11	QUE[2]	0.077	11	EMO[2]	0.038	11
EMO[2]	0.066	13	QUE[2]	0.045	12	SHE	0.077	11	AVA	0.038	12
LSB[2]	0.066	13	KME	0.045	13	SSO[1]	0.077	11	PRG	0.035	13
PRG	0.066	13	LSB[2]	0.044	14	BUL	0.046	16.5	LSB[2]	0.032	14
SSO[1]	0.066	13	CAD	0.044	15	IRO	0.046	16.5	COX[2]	0.031	15
BUL	0.057	17.5	TUR	0.040	16	LSB[2]	0.046	16.5	QUE[2]	0.030	16
IRO	0.057	17.5	BUL	0.039	17	MAR	0.046	16.5	BUL	0.025	17
OKM[1]	0.057	17.5	WHA[1]	0.037	18	OKM[1]	0.046	16.5	IRO	0.025	18
QUE[2]	0.057	17.5	CSS	0.036	19	PRG	0.046	16.5	DEA	0.023	19
OWO	0.049	20	LMA	0.028	20	AVA	0.031	23.5	OKM[1]	0.022	20

[1] Patches within or overlapping Mojave National Preserve are shaded in red

[2] Patches within or overlapping Joshua Tree National Park are shaded in blue

Table 32 (continued). Ranks of patches in the genetic and demographic networks of desert bighorn sheep on the basis of effectively connected pairs (ECP; a measure of short-term connectivity) and mean weighted closeness (MWC; a measure of long-term connectivity). Delta indicates the change in connectivity associated with adding or removing the patch from the network. Patch abbreviations in Figure 37.

Genetic Network ECP			Genetic Network MWC			Demographic Network ECP			Demographic Network MWC		
Patch	**Δ**	**Rank**	**Patch**	**Δ**	**Rank**	**Patch**	**Δ**	**Rank**	**Patch**	**Δ**	**Rank**
SGO	0.041	21	SGO	0.028	21	CHU	0.031	23.5	ORO	0.015	21
DEA	0.033	25	NSB	0.023	22	CLI	0.031	23.5	MAR	0.014	22
ECR	0.033	25	SSO[1]	0.022	23	DEA	0.031	23.5	LMA	0.012	23
LMA	0.033	25	WHI	0.019	24	LMA	0.031	23.5	WCH	0.011	24
MAR	0.033	25	OKM[1]	0.019	25	ORO	0.031	23.5	ECR	0.000	25
NOR	0.033	25	CLI	0.019	26	SBR	0.031	23.5	SBR	0.000	26
NSB	0.033	25	AVA	0.016	27	WCH	0.031	23.5	CLI	-0.001	27
WCH	0.033	25	SBR	0.015	28	ECR	0.015	29.5	NSB	-0.002	28.5
CLI	0.025	29.5	NOR	0.011	29	NSB	0.015	29.5	SGO	-0.002	28.5
ORO	0.025	29.5	DEA	0.011	30	OWO	0.015	29.5	OWO	-0.002	30
CHU	0.016	33	WCH	0.009	31	SGO	0.015	29.5	CHU	-0.005	31
ECH	0.016	33	ORO	0.005	32	CHE	0.000	34.5	CHE	-0.023	34.5
SBR	0.016	33	ECR	0.004	33	ECH	0.000	34.5	ECH	-0.023	34.5
TUR	0.016	33	CHE	-0.003	34	NOR	0.000	34.5	NOR	-0.023	34.5
WHI	0.016	33	CHU	-0.006	35	SGA	0.000	34.5	SGA	-0.023	34.5
CHE	0.008	36	ECH	-0.013	36	TUR	0.000	34.5	TUR	-0.023	34.5
SGA	0.000	37	SGA	-0.023	37	WHI	0.000	34.5	WHI	-0.023	34.5

[1] Patches within or overlapping Mojave National Preserve are shaded in red

[2] Patches within or overlapping Joshua Tree National Park are shaded in blue

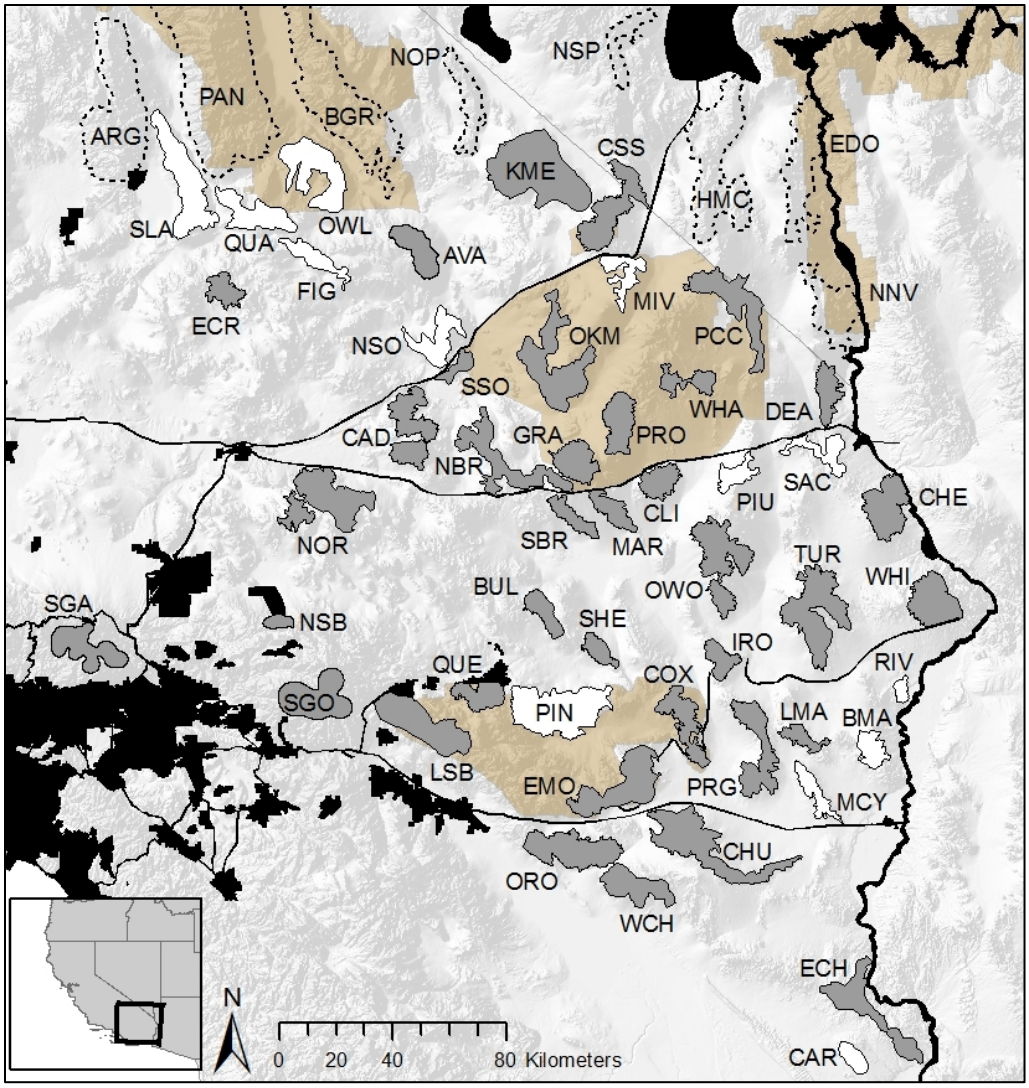


Figure 37. Patches of habitat for desert bighorn sheep. Gray, occupied; white, unoccupied. Polygons with dotted outlines represent populations outside the assessment area that were not ranked. Black polygons or features represent barriers to dispersal. Brown polygons are National Park Service lands. ARG, Argus Mts; AVA, Avawatz Mts; BMA, Big Maria Mts; BGR - Black Mts/Greenwater Range; BUL, Bullion Mts; CAD, Cady Mts; CAR, Cargo Muchacho Mts; CHE, Chemehuevi Mts; CHU, Chuckwalla Mts; CSS, Clark Mts/South Spring Range; CLI, Clipper Mts; COX, Coxcomb Mts; DEA, Dead Mts; ECR, Eagle Crags; EMO, Eagle Mts; ECH, East Chocolate Mts; EDO, Eldorado Mts; FIG, Fort Irwin Granite Mts; GRA, Granite Mts; HMC, Highland Range/McCullough Range; IRO, Iron Mts; KME, Kingston Mts/Mesquite Mts; LMA, Little Maria Mts; LSB, Little San Bernardino Mts; MAR, Marble Mts; MCY, McCoy Mts; MIV, Mescal Range/Ivanpah Mts; NNV, Newberry Mts; NOR, Newberry Mts/Ord Mts/Rodman Mts; NOP, Nopah Mts; NBR, North Bristol Mts; NSB, North San Bernardino/Cushenbury Mts; NSO, North Soda Mts; NSP, North Spring Range; OKM, Old Dad Peak/Kelso Mts/Marl Mts/Club Peak/Indian Spring; OWO, Old Woman Mts; ORO, Orocopia Mts; OWL, Owlshead Mts; PRG, Palen Mts/Riverside Granite Mts; PAN, Panamint Mts; PIN, Pinto Mts; PIU, Piute Mts; PCC, Piute Range/Castle Peaks/Castle Mts; PRO, Providence Mts; QUA, Quail Mts; QUE, Queen Mountain; RIV, Riverside Mts; SAC, Sacramento Mts; SGA, San Gabriel Mts; SGO, San Gorgonio Mts; SHE, Sheephole Mts; SLA, Slate Range; SBR, South Bristol Mts; SSO, South Soda Mts; TUR, Turtle Mts; WCH, West Chocolate Mts; WHI, Whipple Mts; WHA, Woods Mts/Hackberry Mts.

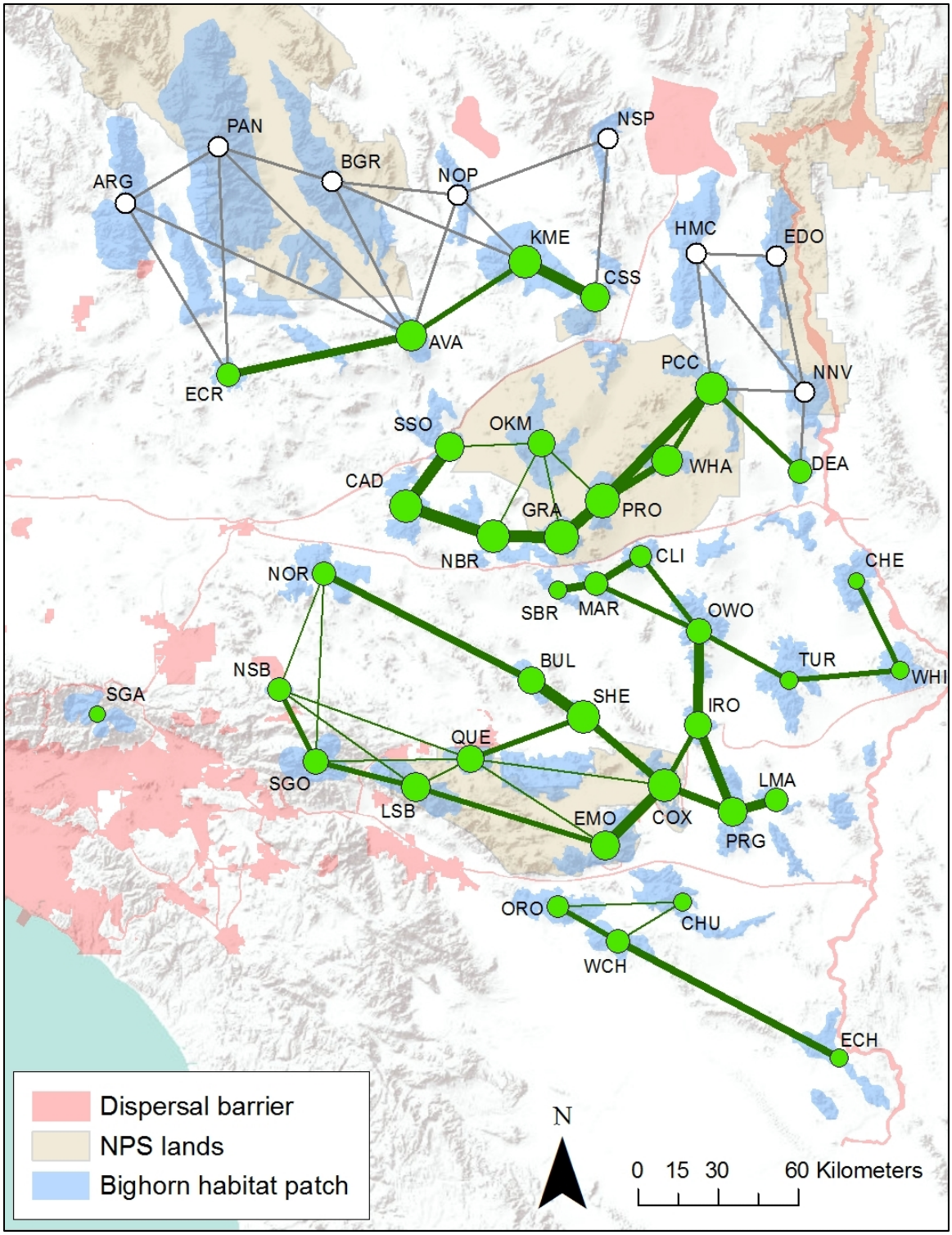


Figure 38. Prioritization of patches and corridors in the genetic network on the basis of effectively connected pairs (ECP). Green circles and lines represent existing patches and corridors included in the assessment; circle size and line width are positively correlated with priority. White circles and gray lines represent patches and associated corridors that were not ranked. Patches are labeled as in Figure 37.

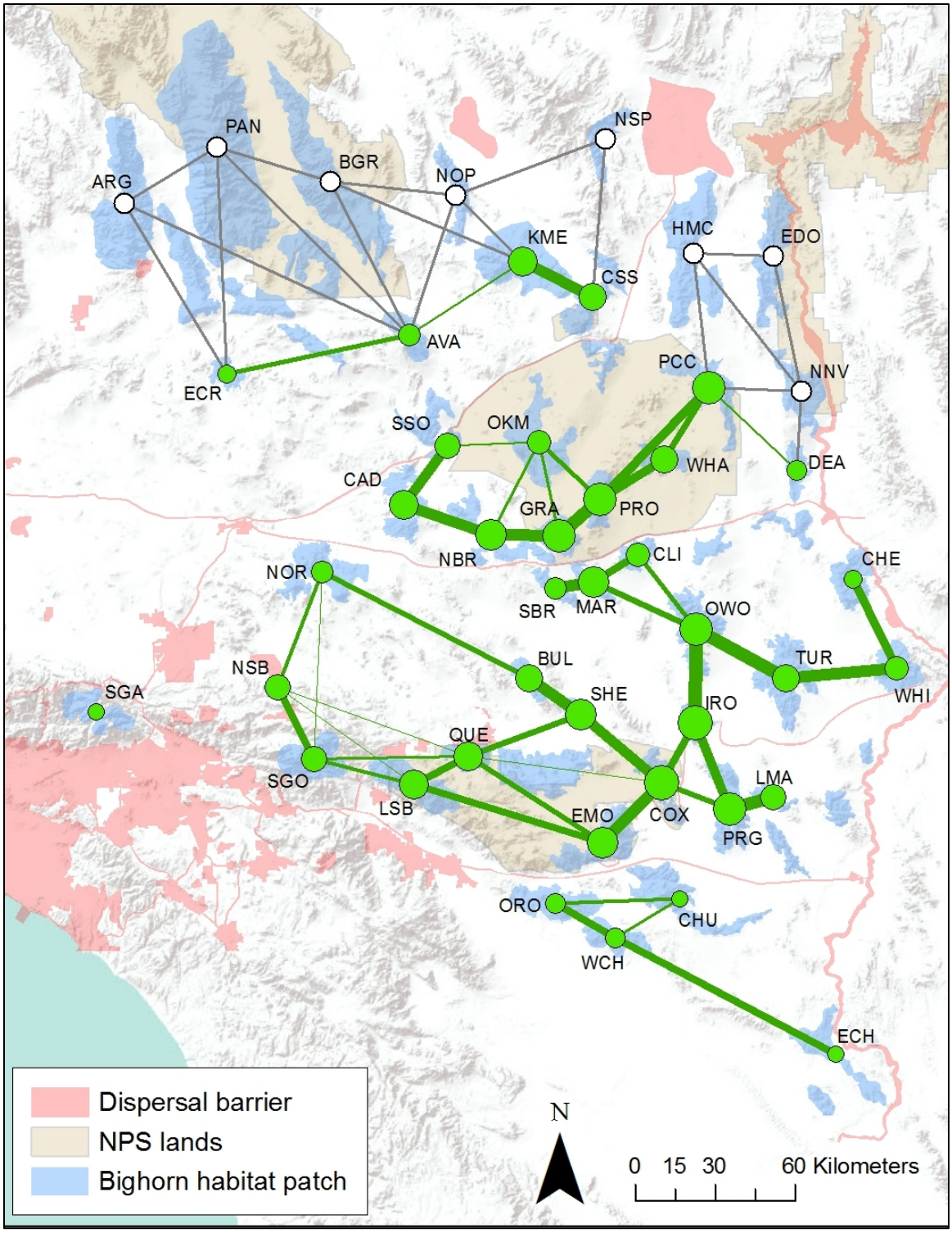


Figure 39. Prioritization of patches and corridors in the genetic network on the basis of mean weighted closeness (MWC). Green circles and lines represent existing patches and corridors included in the assessment; circle size and line width are positively correlated with priority. White circles and gray lines represent patches and associated corridors that were not ranked. Patches are labeled as in Figure 37.

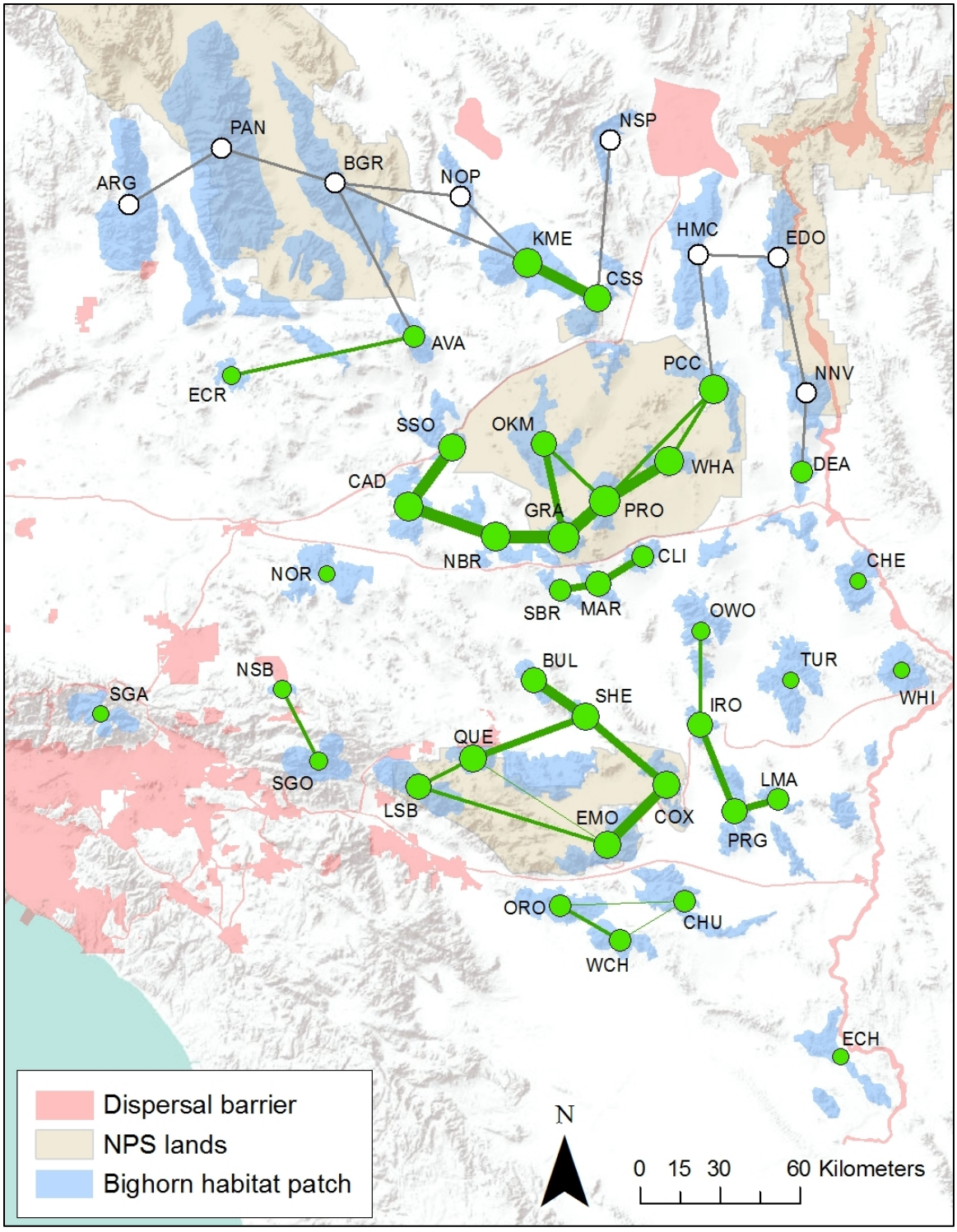


Figure 40. Prioritization of patches and corridors in the demographic network on the basis of effectively connected pairs (ECP). Green circles and lines represent existing patches and corridors included in the assessment; circle size and line width are positively correlated with priority. White circles and gray lines represent patches and associated corridors that were not ranked. Patches are labeled as in Figure 37.

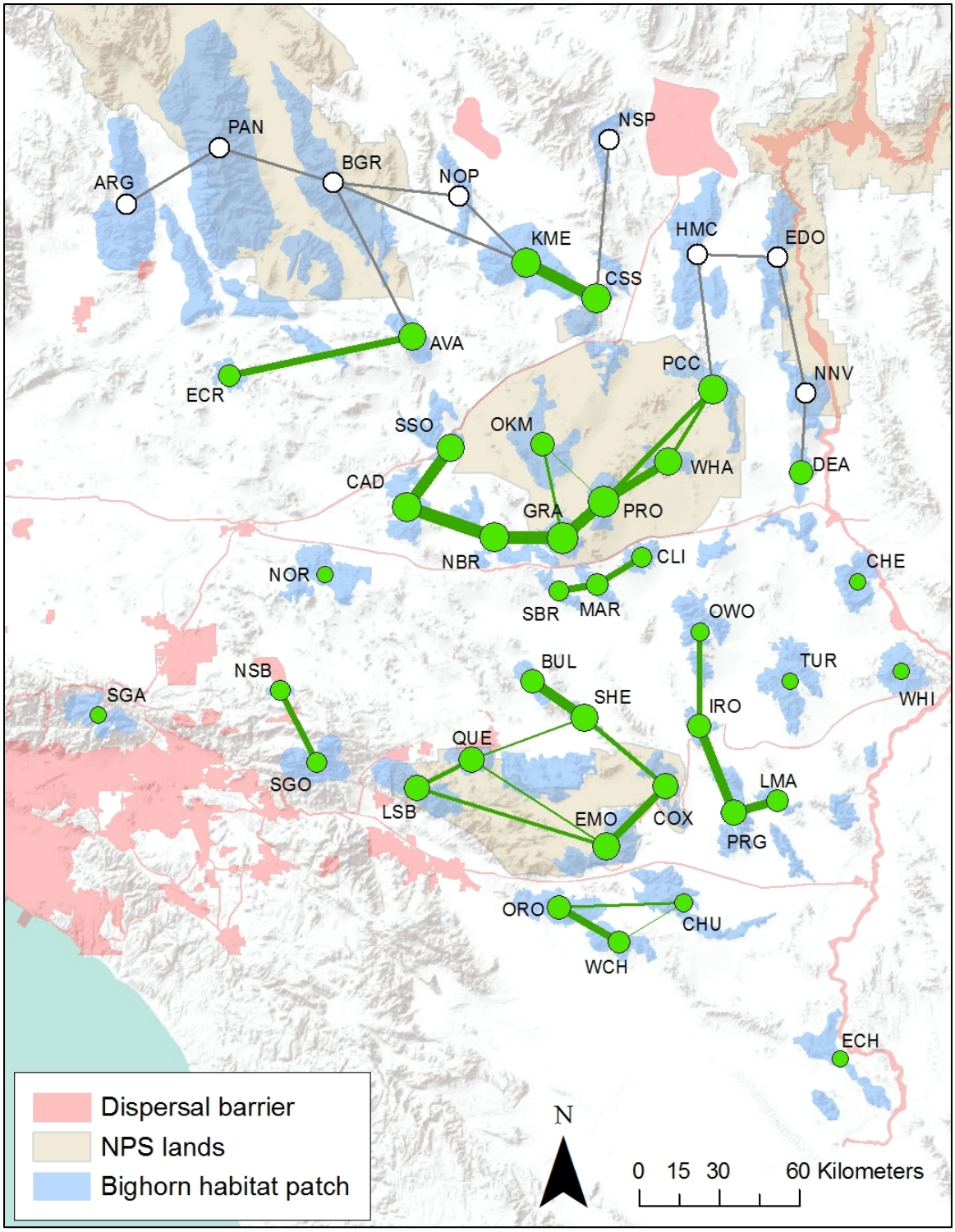


Figure 41. Prioritization of patches and corridors in the demographic network on the basis of mean weighted closeness (MWC). Green circles and lines represent existing patches and corridors included in the assessment; circle size and line width are positively correlated with priority. White circles and gray lines represent patches and associated corridors that were not ranked. Patches are labeled as in Figure 37.

Table 33. Ranks of corridors for protection in the genetic and demographic networks of desert bighorn sheep on the basis of effectively connected pairs (ECP; a measure of short-term connectivity) and mean weighted closeness (MWC; a measure of long-term connectivity). Delta indicates the change in connectivity associated with adding or removing the patch from the network. Patch abbreviations in Figure 37.

Genetic Network ECP			Genetic Network MWC			Demographic Network ECP			Demographic Network MWC		
Corridor	**Δ**	**Rank**	**Corridor**	**Δ**	**Rank**	**Corridor**	**Δ**	**Rank**	**Corridor**	**Δ**	**Rank**
GRA-PRO[1]	0.148	1	IRO-OWO	0.121	1	GRA-NBR[1]	0.169	1	GRA-NBR[1]	0.146	1
GRA-NBR[1]	0.131	2	OWO-TUR	0.055	2	GRA-PRO[1]	0.154	2	CAD-NBR	0.102	2
CAD-NBR	0.098	3	LMA-PRG	0.050	3	CAD-NBR	0.123	3	GRA-PRO[1]	0.091	3
CAD-SSO[1]	0.057	4.5	GRA-PRO[1]	0.047	4	CAD-SSO[1]	0.077	5	CSS-KME	0.084	4
CSS-KME	0.057	4.5	GRA-NBR[1]	0.044	5	CSS-KME	0.077	5	CAD-SSO[1]	0.066	5
BUL-SHE	0.049	6	BUL-SHE	0.040	6	PRO-WHA[1]	0.077	5	BUL-SHE	0.047	6
IRO-PRG	0.033	7.5	TUR-WHI	0.037	7	BUL-SHE	0.046	7.5	IRO-PRG	0.037	7
LMA-PRG	0.033	7.5	MAR-SBR	0.037	8	COX-EMO[2]	0.046	7.5	LMA-PRG	0.034	8
COX-EMO[2]	0.025	10	CAD-NBR	0.034	9	CLI-MAR	0.031	12	PRO-WHA[1]	0.027	9
IRO-OWO	0.025	10	CSS-KME	0.034	10	COX-SHE[2]	0.031	12	COX-EMO[2]	0.024	10
PCC-PRO[1]	0.025	10	COX-EMO[2]	0.028	11	GRA-OKM[1]	0.031	12	ORO-WCH	0.023	11
AVA-ECR	0.016	15.5	IRO-PRG	0.025	12	IRO-PRG	0.031	12	AVA-ECR	0.022	12
BUL-NOR	0.016	15.5	COX-SHE[2]	0.025	13	LMA-PRG	0.031	12	MAR-SBR	0.022	13
CLI-MAR	0.016	15.5	CAD-SSO[1]	0.023	14	MAR-SBR	0.031	12	CLI-MAR	0.021	14
COX-PRG[2]	0.016	15.5	CHE-WHI	0.019	15	QUE-SHE[2]	0.031	12	NSB-SGO	0.020	15
COX-SHE[2]	0.016	15.5	PRO-WHA[1]	0.017	16	AVA-ECR	0.015	20	IRO-OWO	0.020	16
ECH-WCH	0.016	15.5	PCC-PRO[1]	0.016	17	EMO-LSB[2]	0.015	20	PCC-PRO[1]	0.020	17
MAR-SBR	0.016	15.5	ORO-WCH	0.014	18	IRO-OWO	0.015	20	COX-SHE[2]	0.017	18
PRO-WHA[1]	0.016	15.5	COX-IRO[2]	0.013	19	LSB-QUE[2]	0.015	20	LSB-QUE[2]	0.017	19
AVA-KME	0.008	26.5	LSB-QUE[2]	0.011	20	NSB-SGO	0.015	20	PCC-WHA[1]	0.010	20
CHE-WHI	0.008	26.5	NSB-SGO	0.011	21	OKM-PRO[1]	0.015	20	EMO-LSB[2]	0.009	21

[1] Patches within or overlapping Mojave National Preserve are shaded in red

[2] Patches within or overlapping Joshua Tree National Park are shaded in blue

Table 33 (continued). Ranks of corridors for protection in the genetic and demographic networks of desert bighorn sheep on the basis of effectively connected pairs (ECP; a measure of short-term connectivity) and mean weighted closeness (MWC; a measure of long-term connectivity). Delta indicates the change in connectivity associated with adding or removing the patch from the network. Patch abbreviations in Figure 37.

Genetic Network ECP			Genetic Network MWC			Demographic Network ECP			Demographic Network MWC		
Corridor	**Δ**	**Rank**	**Corridor**	**Δ**	**Rank**	**Corridor**	**Δ**	**Rank**	**Corridor**	**Δ**	**Rank**
CLI-OWO	0.008	26.5	CLI-MAR	0.011	22	ORO-WCH	0.015	20	CHU-ORO	0.006	22
COX-IRO[2]	0.008	26.5	EMO-LSB[2]	0.010	23	PCC-PRO[1]	0.015	20	GRA-OKM[1]	0.004	23
DEA-PCC[1]	0.008	26.5	ECH-WCH	0.009	24	PCC-WHA[1]	0.015	20	QUE-SHE[2]	0.003	24
EMO-LSB[2]	0.008	26.5	QUE-SHE[2]	0.008	25	CHU-ORO	0.000	26	EMO-QUE[2]	0.003	25
LSB-SGO[2]	0.008	26.5	MAR-OWO	0.007	26	CHU-WCH	0.000	26	OKM-PRO[1]	0.003	26
MAR-OWO	0.008	26.5	PCC-WHA[1]	0.007	27	EMO-QUE[2]	0.000	26	CHU-WCH	0.000	27
NSB-SGO	0.008	26.5	BUL-NOR	0.007	28	–	–	–	–	–	–
OWO-TUR	0.008	26.5	COX-PRG[2]	0.007	29	–	–	–	–	–	–
ORO-WCH	0.008	26.5	EMO-QUE[2]	0.005	30	–	–	–	–	–	–
PCC-WHA[1]	0.008	26.5	AVA-ECR	0.005	31	–	–	–	–	–	–
QUE-SHE[2]	0.008	26.5	OKM-PRO[1]	0.004	32	–	–	–	–	–	–
TUR-WHI	0.008	26.5	CLI-OWO	0.004	33	–	–	–	–	–	–
CHU-ORO	0.000	40.5	CHU-ORO	0.004	34	–	–	–	–	–	–
CHU-WCH	0.000	40.5	LSB-SGO[2]	0.003	35	–	–	–	–	–	–
COX-QUE[2]	0.000	40.5	NOR-NSB	0.002	36	–	–	–	–	–	–
EMO-QUE[2]	0.000	40.5	GRA-OKM[1]	0.002	37	–	–	–	–	–	–
GRA-OKM[1]	0.000	40.5	QUE-SGO[2]	0.002	38	–	–	–	–	–	–
LSB-NSB[2]	0.000	40.5	NBR-OKM[1]	0.001	39	–	–	–	–	–	–
LSB-QUE[2]	0.000	40.5	CHU-WCH	0.001	40	–	–	–	–	–	–
NOR-NSB	0.000	40.5	OKM-SSO[1]	0.001	41	–	–	–	–	–	–

[1] Patches within or overlapping Mojave National Preserve are shaded in red

[2] Patches within or overlapping Joshua Tree National Park are shaded in blue

Table 33 (continued). Ranks of corridors for protection in the genetic and demographic networks of desert bighorn sheep on the basis of effectively connected pairs (ECP; a measure of short-term connectivity) and mean weighted closeness (MWC; a measure of long-term connectivity). Delta indicates the change in connectivity associated with adding or removing the patch from the network. Patch abbreviations in Figure 37.

Genetic Network ECP			Genetic Network MWC			Demographic Network ECP			Demographic Network MWC		
Corridor	**Δ**	**Rank**	**Corridor**	**Δ**	**Rank**	**Corridor**	**Δ**	**Rank**	**Corridor**	**Δ**	**Rank**
NOR-SGO	0.000	40.5	DEA-PCC[1]	0.001	42	–	–	–	–	–	–
NBR-OKM[1]	0.000	40.5	AVA-KME	0.000	43	–	–	–	–	–	–
NSB-QUE[2]	0.000	40.5	LSB-NSB[2]	0.000	44	–	–	–	–	–	–
OKM-PRO[1]	0.000	40.5	COX-QUE[2]	0.000	46	–	–	–	–	–	–
OKM-SSO[1]	0.000	40.5	NOR-SGO	0.000	46	–	–	–	–	–	–
QUE-SGO[2]	0.000	40.5	NSB-QUE[2]	0.000	46	–	–	–	–	–	–

[1] Patches within or overlapping Mojave National Preserve are shaded in red

[2] Patches within or overlapping Joshua Tree National Park are shaded in blue

Table 34. Ranks of corridors for restoration of connectivity among populations of desert bighorn sheep on the basis of effectively connected pairs (ECP; a measure of short-term connectivity) and mean weighted closeness (MWC; a measure of long-term connectivity). Delta indicates the change in connectivity associated with adding or removing the patch from the network. Patch abbreviations in Figure 37.

Genetic Network ECP			Genetic Network MWC			Demographic Network ECP			Demographic Network MWC		
Corridor	**Δ**	**Rank**	**Corridor**	**Δ**	**Rank**	**Corridor**	**Δ**	**Rank**	**Corridor**	**Δ**	**Rank**
GRA-MAR[1]	0.246	1	GRA-MAR[1]	0.342	1	GRA-MAR[1]	0.231	1.5	AVA-SSO[1]	0.196	1
NBR-SBR	0.197	2	NBR-SBR	0.308	2	NBR-SBR	0.231	1.5	GRA-MAR[1]	0.196	2
EMO-ORO[2]	0.123	3	CLI-PRO[1]	0.280	3	LSB-SGO[2]	0.108	3	NBR-SBR	0.185	3
CHU-EMO[2]	0.115	4	BUL-CAD	0.225	4	EMO-ORO[2]	0.092	4	CLI-PRO[1]	0.110	4
CLI-PRO[1]	0.090	5.5	BUL-NBR	0.217	5	CHU-EMO[2]	0.077	5.5	EMO-ORO[2]	0.101	5
LSB-ORO[2]	0.090	5.5	CAD-NOR	0.205	6	LSB-NSB[2]	0.077	5.5	CHU-EMO[2]	0.096	6
ORO-QUE[2]	0.066	7	AVA-CAD	0.171	7	LSB-ORO[2]	0.062	7.5	LSB-SGO[2]	0.094	7
AVA-CAD	0.049	8.5	AVA-SSO[1]	0.166	8	ORO-QUE[2]	0.062	7.5	COX-PRG[2]	0.090	8
AVA-SSO[1]	0.049	8.5	CHE-DEA	0.159	9	NSB-SGA	0.031	9.5	LSB-ORO[2]	0.084	9
ORO-SHE	0.033	10	EMO-ORO[2]	0.155	10	SGA-SGO	0.031	9.5	LSB-NSB[2]	0.072	10
BUL-CAD	0.025	11.5	CHU-EMO[2]	0.148	11	AVA-SSO[1]	0.015	13	ORO-QUE[2]	0.071	11
SGA-SGO	0.025	11.5	KME-OKM[1]	0.147	12	CHE-DEA	0.015	13	COX-IRO[2]	0.071	12
CAD-NOR	0.016	15.5	CSS-PRO[1]	0.142	13	CLI-PRO[1]	0.015	13	CHE-DEA	0.028	13
CHE-DEA	0.016	15.5	LSB-ORO[2]	0.141	14	COX-IRO[2]	0.015	13	NSB-SGA	0.020	14
CSS-PRO[1]	0.016	15.5	CSS-WHA[1]	0.141	15	COX-PRG[2]	0.015	13	SGA-SGO	0.018	15
CSS-WHA[1]	0.016	15.5	ORO-QUE[2]	0.132	16	–	–	–	–	–	–
KME-OKM[1]	0.016	15.5	ORO-SHE	0.094	17	–	–	–	–	–	–
NSB-SGA	0.016	15.5	CHU-PRG	0.081	18	–	–	–	–	–	–
BUL-NBR	0.008	20	SGA-SGO	0.041	19	–	–	–	–	–	–
CHU-PRG	0.008	20	NSB-SGA	0.039	20	–	–	–	–	–	–
NOR-SGA	0.008	20	NOR-SGA	0.022	21	–	–	–	–	–	–

[1] Patches within or overlapping Mojave National Preserve are shaded in red

[2] Patches within or overlapping Joshua Tree National Park are shaded in blue

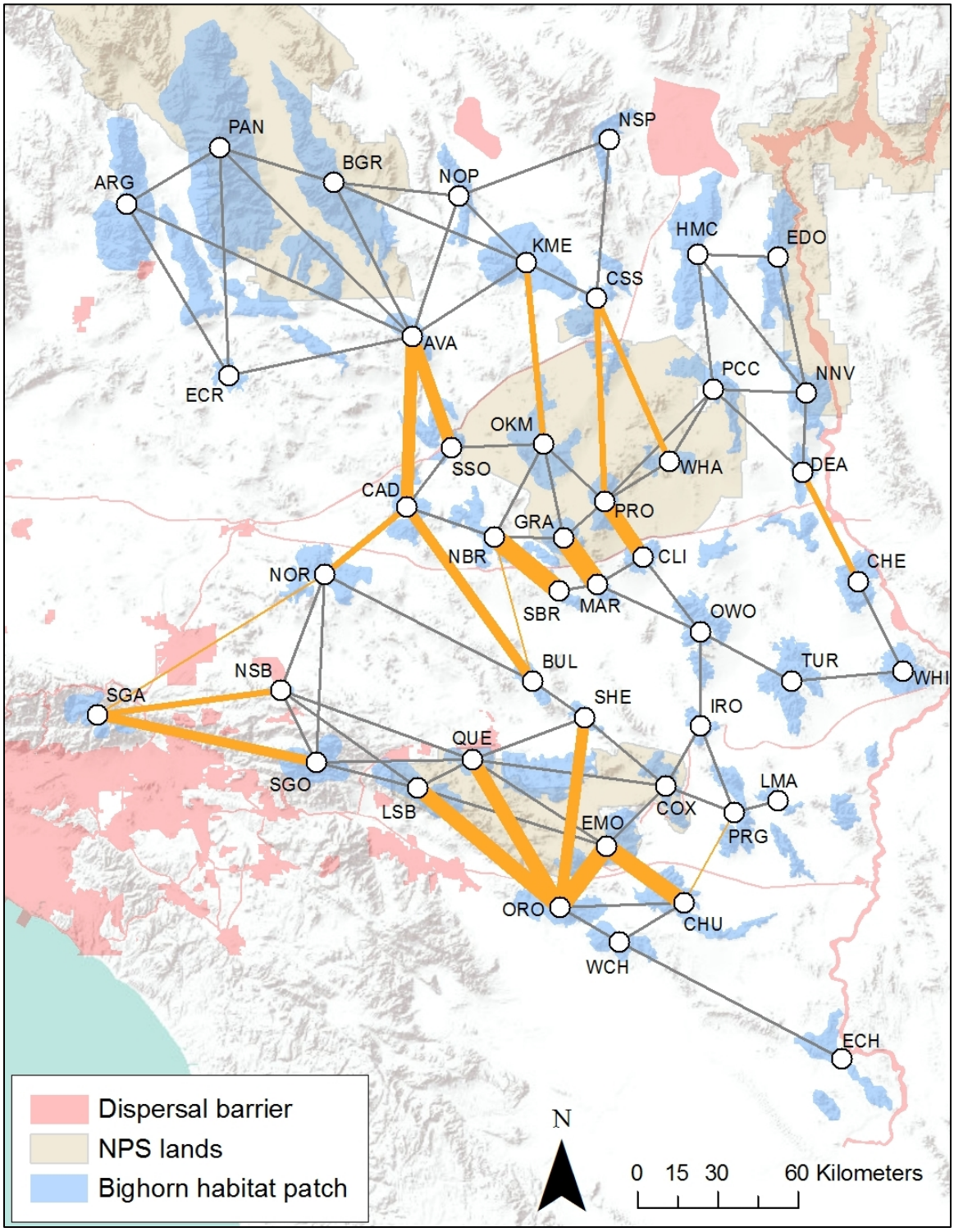


Figure 42. Prioritization of restorable corridors in the genetic network on the basis of effectively connected pairs (ECP). Grey lines, existing corridors. Orange lines, potentially restorable corridors; line width is positively correlated with priority. Patches are labeled as in Figure 37.

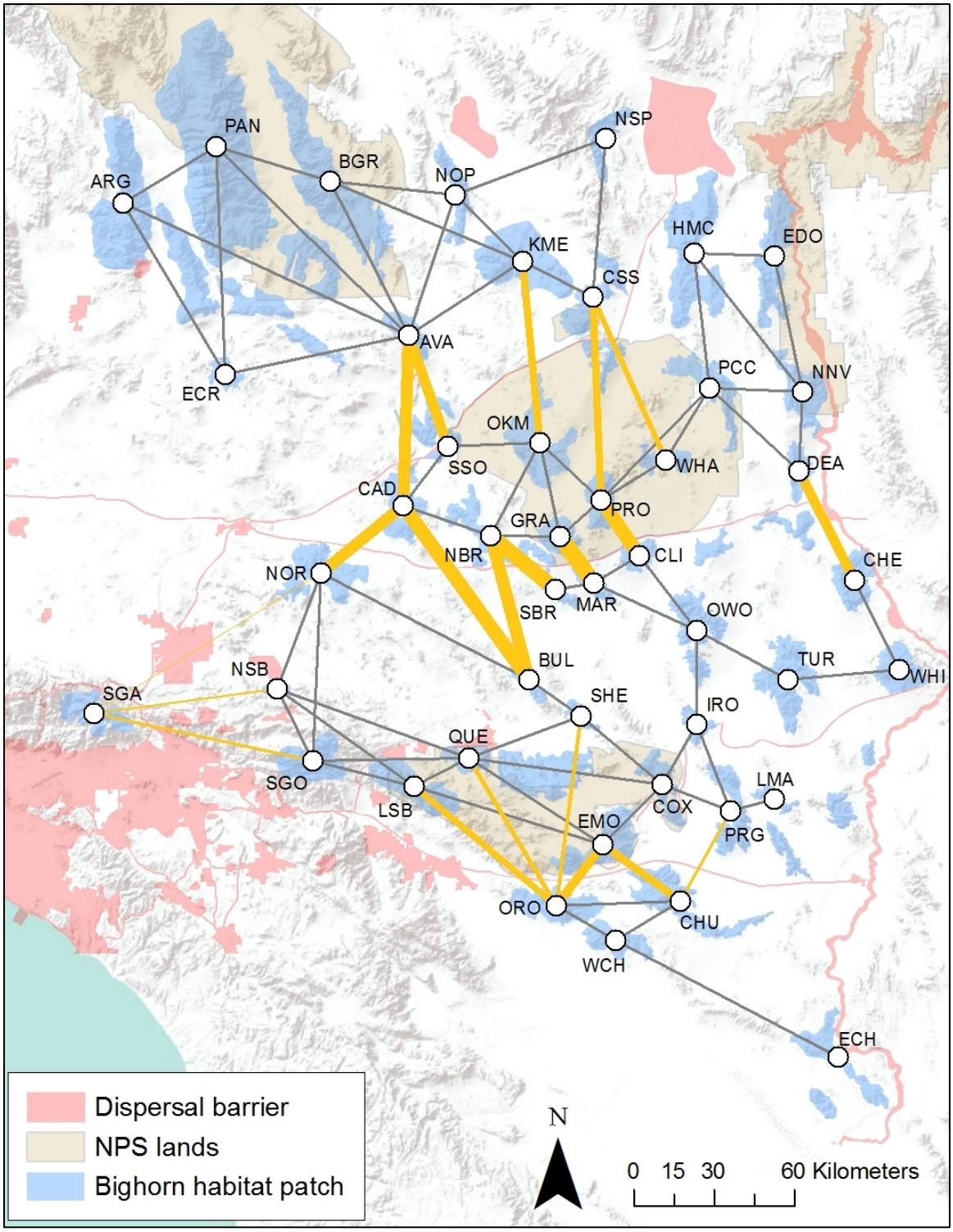


Figure 43. Prioritization of restorable corridors in the genetic network on the basis of mean weighted closeness (MWC). Grey lines, existing corridors. Orange lines, potentially restorable corridors; line width is positively correlated with priority. Patches are labeled as in Figure 37.

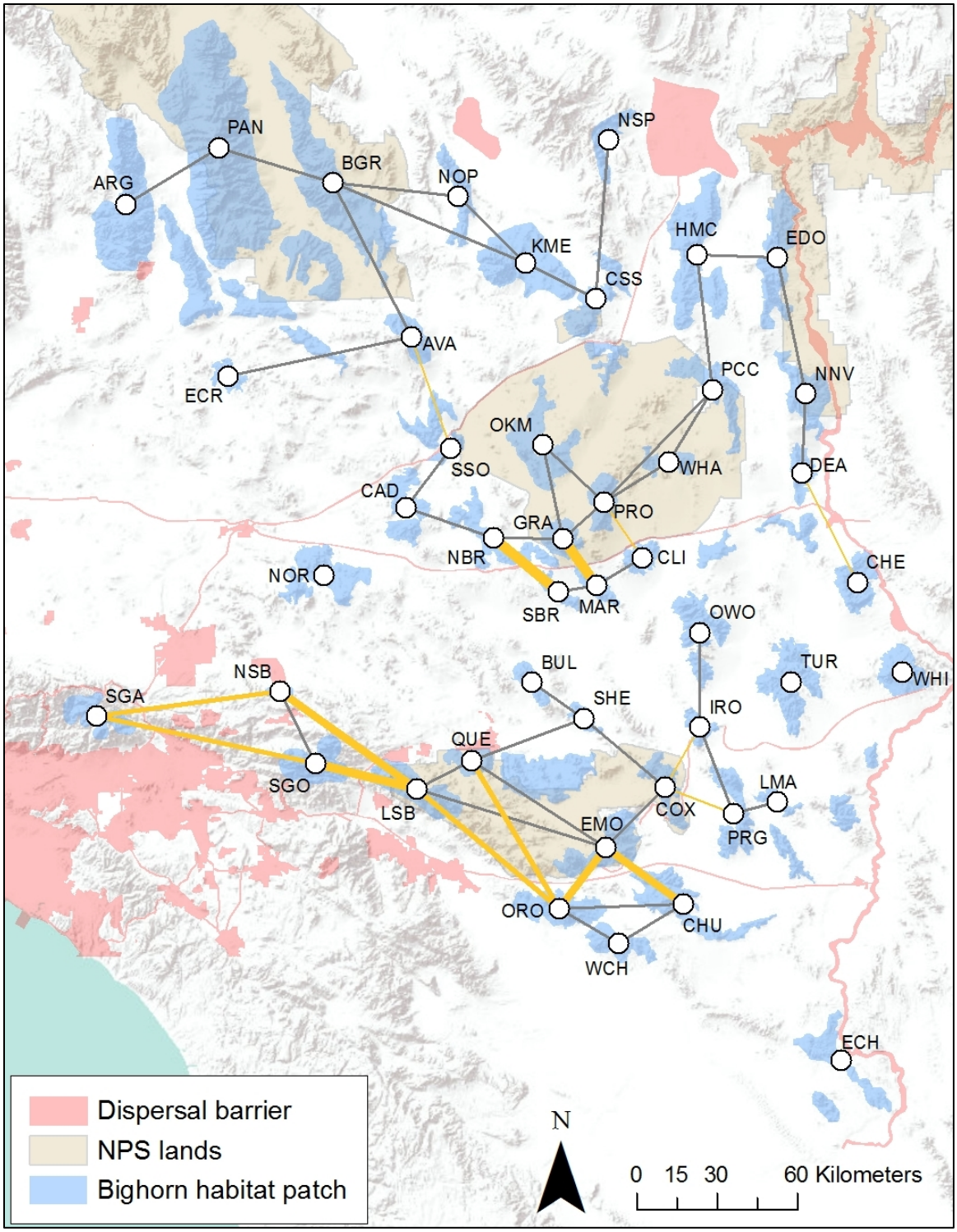


Figure 44. Prioritization of restorable corridors in the demographic network on the basis of effectively connected pairs (ECP). Grey lines, existing corridors. Orange lines, potentially restorable corridors; line width is positively correlated with priority. Patches are numbered as in Figure 37.

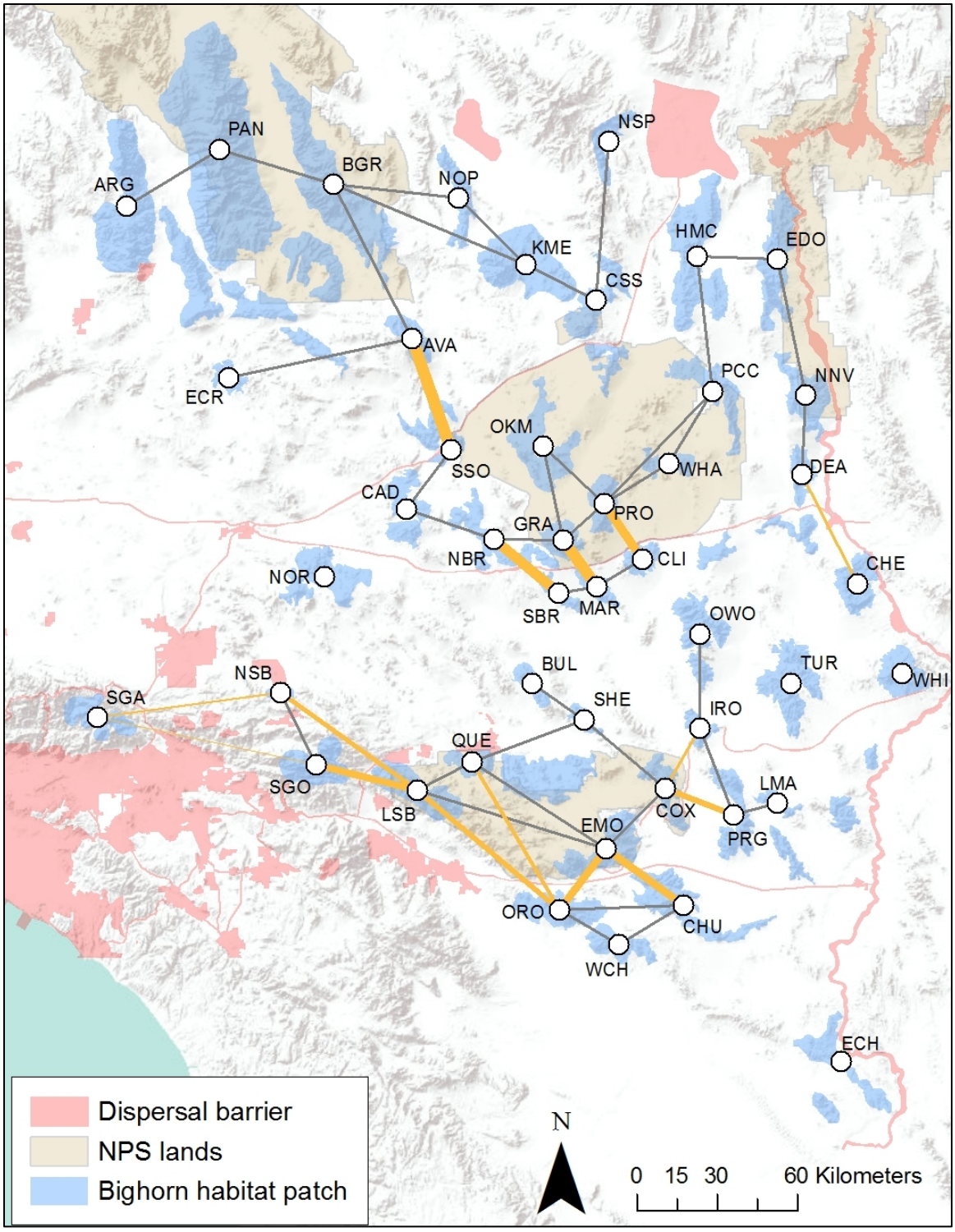


Figure 45. Prioritization of restorable corridors in the demographic network on the basis of mean weighted closeness (MWC). Grey lines, existing corridors. Orange lines, potentially restorable corridors; line width is positively correlated with priority. Patches are labeled as in Figure 37.

Table 35. Ranks of patches for restoration in the genetic and demographic networks of desert bighorn sheep on the basis of effectively connected pairs (ECP; a measure of short-term connectivity) and mean weighted closeness (MWC; a measure of long-term connectivity). Delta indicates the change in connectivity associated with adding or removing the patch from the network. Patch abbreviations in Figure 37.

Genetic Network ECP			Genetic Network MWC			Demographic Network ECP			Demographic Network MWC		
Patch	**Δ**	**Rank**	**Patch**	**Δ**	**Rank**	**Patch**	**Δ**	**Rank**	**Patch**	**Δ**	**Rank**
PIN[2]	0.098	1	PIN[2]	0.068	1	PIN[2]	0.154	1	PIN[2]	0.082	1
OWL	0.090	2	QUA	0.040	2	QUA	0.108	2.5	OWL	0.072	2
FIG	0.066	3	OWL	0.039	3	SLA	0.108	2.5	QUA	0.070	3
MIV[1]	0.057	4.5	SLA	0.036	4	OWL	0.092	4	SLA	0.065	4
SLA	0.057	4.5	FIG	0.030	5	FIG	0.077	5	FIG	0.061	5
QUA	0.049	6	MCY	0.030	6	BMA	0.046	6.5	BMA	0.020	6
BMA	0.041	7	BMA	0.029	7	MCY	0.046	6.5	NSO	0.019	7
MCY	0.033	9	PIU	0.023	8	CAR	0.015	9.5	MCY	0.015	8
NSO	0.033	9	MIV[1]	0.012	9	NSO	0.015	9.5	PIU	0.003	9
PIU	0.033	9	NSO	0.011	10	PIU	0.015	9.5	SAC	-0.007	10
RIV	0.016	11	RIV	0.009	11	SAC	0.015	9.5	CAR	-0.015	11
CAR	0.008	12.5	SAC	0.005	12	–	–	–	–	–	–
SAC	0.008	12.5	CAR	-0.009	13	–	–	–	–	–	–

[1] Patches within or overlapping Mojave National Preserve are shaded in red

[2] Patches within or overlapping Joshua Tree National Park are shaded in blue

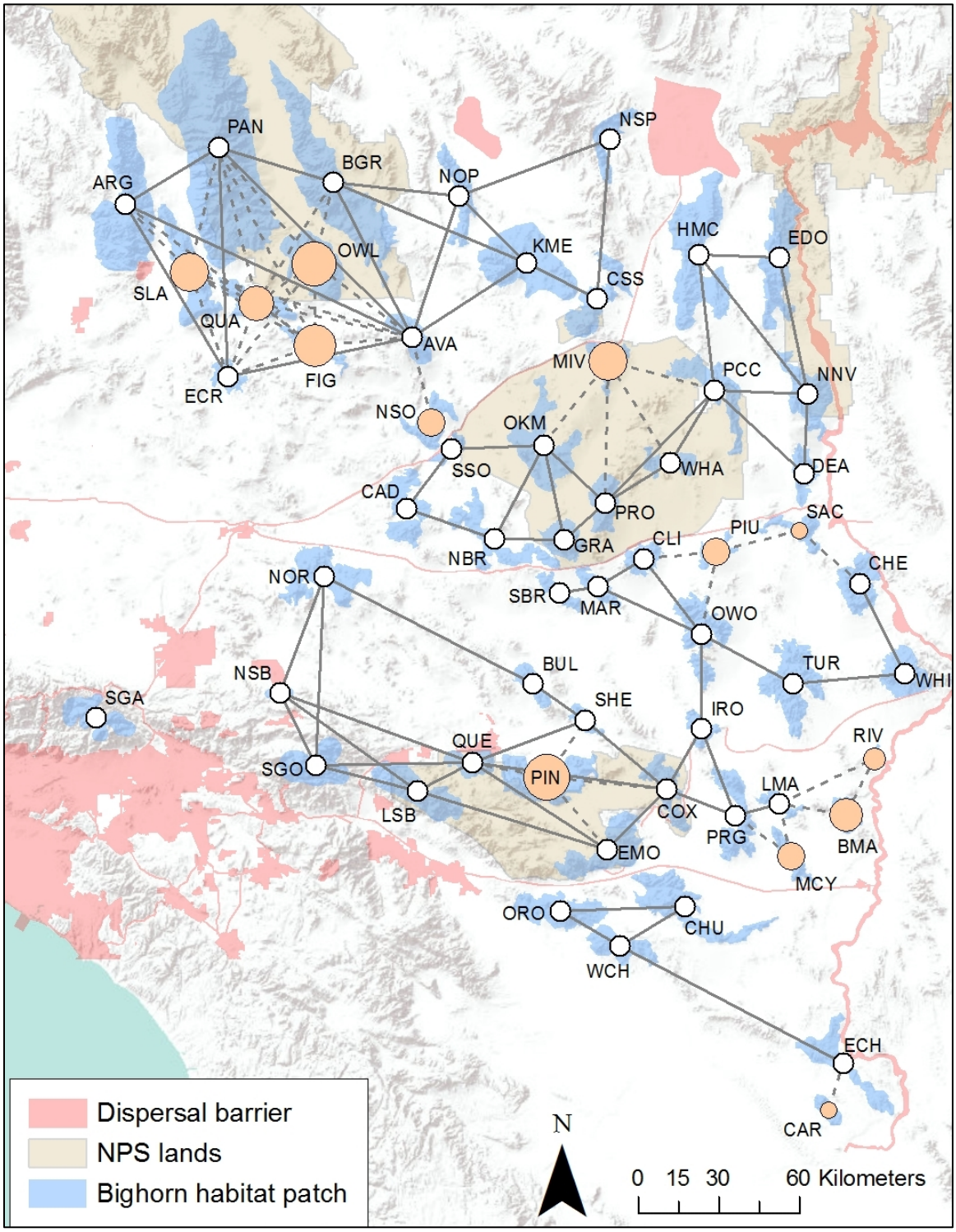


Figure 46. Prioritization of restorable patches in the genetic network on the basis of effectively connected pairs (ECP). Orange circles, potential reintroduction sites; circle size is positively correlated with priority. Solid lines, existing network edges. Dashed lines, edges associated with reintroductions. Patches are labeled as in Figure 37.

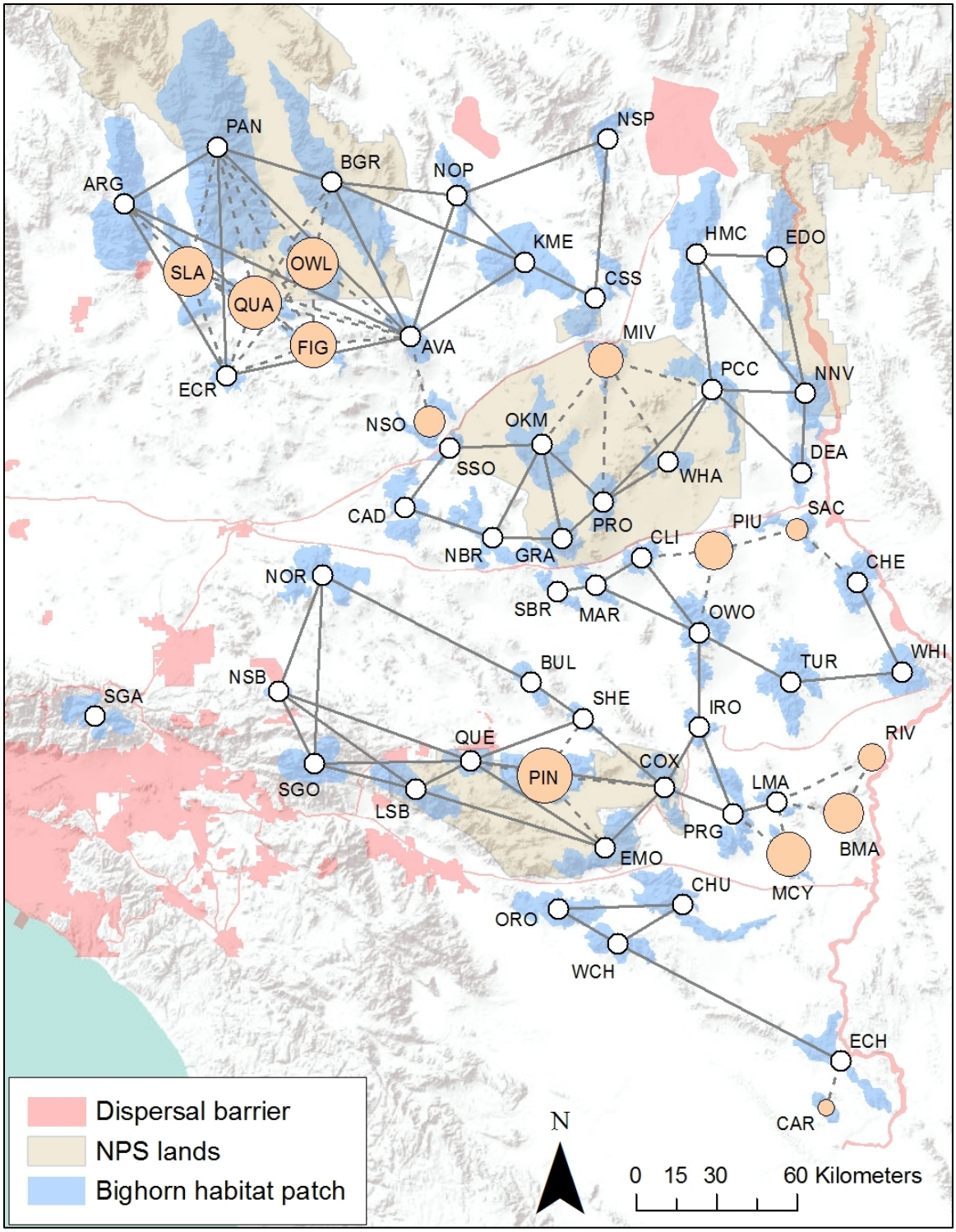


Figure 47. Prioritization of restorable patches in the genetic network on the basis of mean weighted closeness (MWC). Orange circles, potential reintroduction sites; circle size is positively correlated with priority. Solid lines, existing network edges. Dashed lines, edges associated with reintroductions. Patches are labeled as in Figure 37.

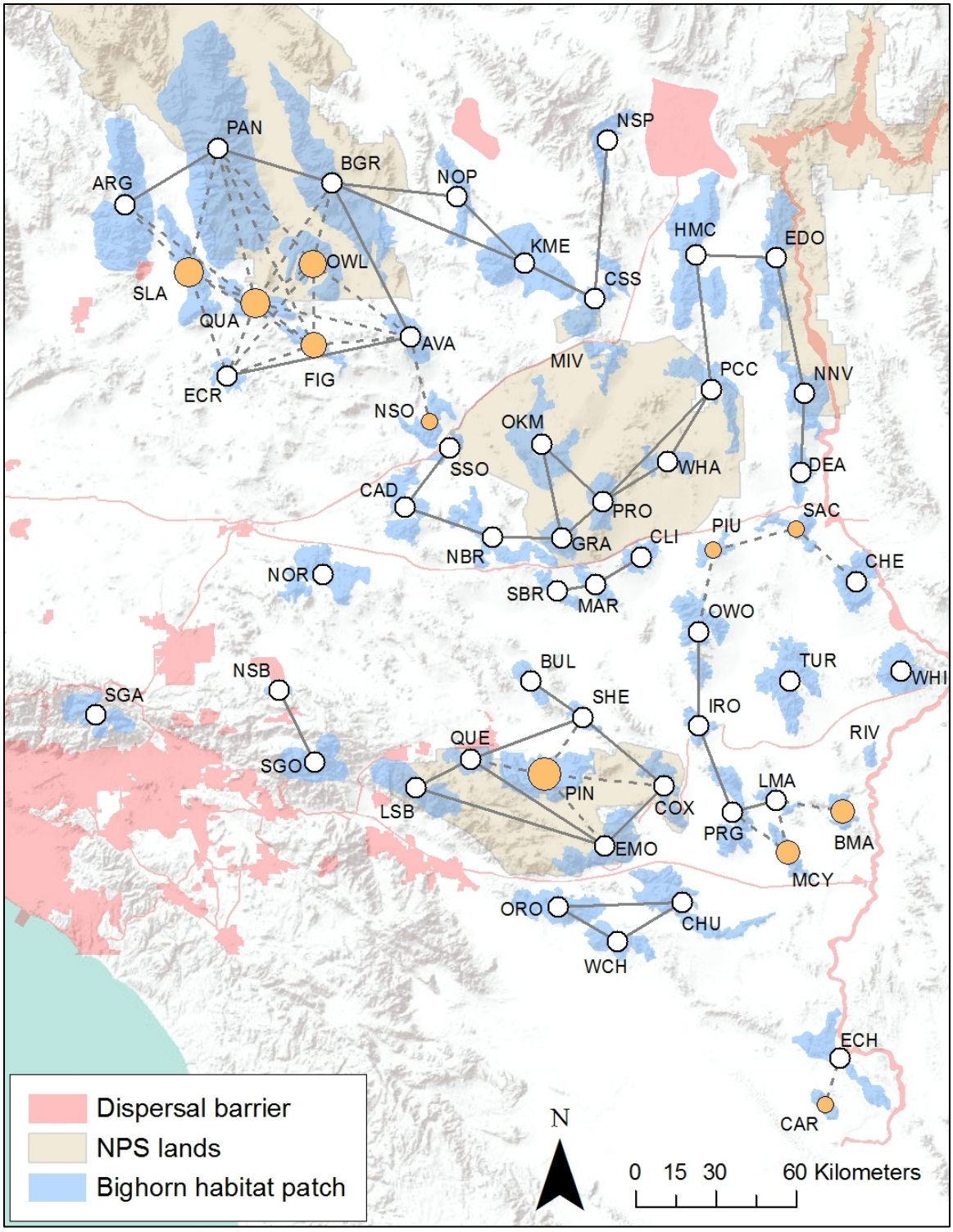


Figure 48. Prioritization of restorable patches in the demographic network on the basis of effectively connected pairs (ECP). Orange circles, potential reintroduction sites; circle size is positively correlated with priority. Solid lines, existing network edges. Dashed lines, edges associated with reintroductions. Patches are labeled as in Figure 37.

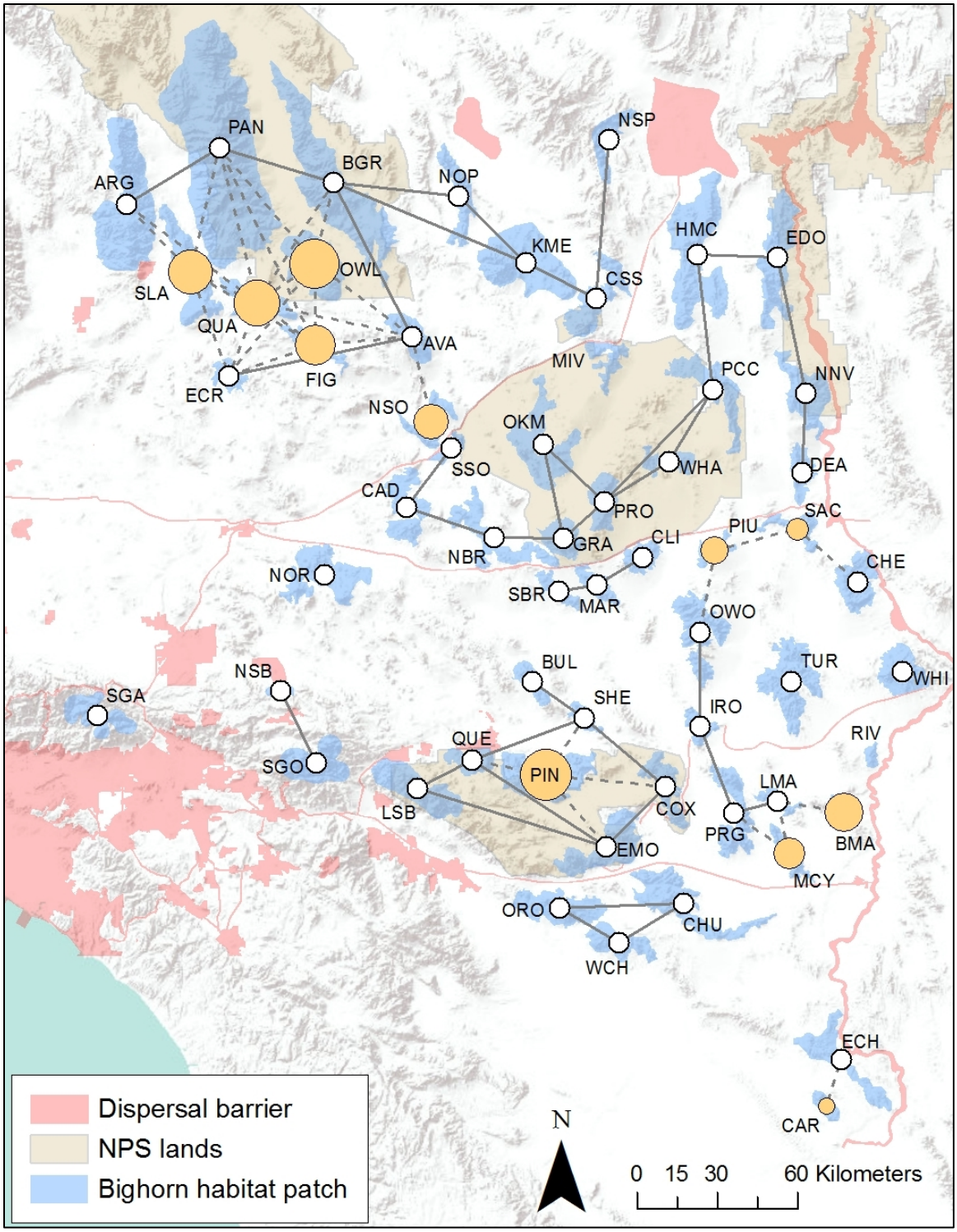


Figure 49. Prioritization of restorable patches in the demographic network on the basis of mean weighted closeness (MWC). Orange circles, potential reintroduction sites; circle size is positively correlated with priority. Solid lines, existing network edges. Dashed lines, edges associated with reintroductions. Patches are labeled as in Figure 37.

The Granite Mountains and Providence Mountains were extremely strong contributors to regional connectivity across management actions, network metrics, and connectivity types (Table 36). The Coxcomb Mountains and Eagle Mountains, and their associated corridors, were generally the highest-priority features in Joshua Tree, although their ranks were lower than those of patches in Mojave National Preserve in most instances. Our analyses suggested that genetic and demographic connectivity could be more than doubled by restoring one corridor across each interstate highway that currently fragments the area occupied by the metapopulation. All three such highways are adjacent to Mojave National Preserve (I-15 along the north boundary, I-40 along the south boundary) or Joshua Tree (I-10 along the south boundary), and the most effective locations for constructing highway crossing structures would be adjacent to patches such as the Granite Mountains, Providence Mountains, Soda Mountains, and Eagle Mountains. Thus, habitat in and around both national parks may play a major role in maintaining and augmenting regional connectivity. Detailed rankings of existing and restorable patches and corridors are presented in Creech et al. (2014).

Table 36. High priority patches and corridors in the genetic and demographic networks of desert bighorn sheep. High priority features are defined as those with ranks among the top five (✓) in at least two of the four combinations of connectivity type (genetic, demographic) and network metric (effectively connected pairs [ECP; a measure of short-term connectivity] or mean weighted closeness [MWC; a measure of long-term connectivity]). Patch protection, protecting occupied habitat patches; patch restoration, restoring desert bighorn sheep populations to unoccupied habitat patches; corridor protection, protecting intact dispersal corridors; corridor restoration, removal of barriers. Patch abbreviations in Figure 37.

Management action	Patch or corridor	Genetic network			Demographic network		
		ECP	MWC	n[1]	ECP	MWC	n
Patch protection	PRO[2]	✓	✓	37	✓	✓	37
	PCC[2]	✓	✓	37	✓	✓	37
	GRA[2]	✓	–	37	✓	✓	37
	NBR	✓	–	37	✓	✓	37
	CAD	✓	–	37	✓	✓	37
Patch restoration	PIN[3]	✓	✓	13	✓	✓	11
	OWL	✓	✓	13	✓	✓	11
	FIG	✓	✓	13	✓	✓	11
	SLA	✓	✓	13	✓	✓	11
	QUA	–	✓	13	✓	✓	11
Corridor protection	GRA-PRO[2]	✓	✓	47	✓	✓	27
	GRA-NBR[2]	✓	✓	47	✓	✓	27
	CAD-NBR	✓	–	47	✓	✓	27
	CSS-KME	✓	–	47	✓	✓	27
	CAD-SSO[2]	✓	–	47	✓	✓	27

[1] Number of features associated with the given management action in the genetic or demographic network

[2] Populations or corridors within or linking directly to Mojave National Preserve are shaded in red

[3] Populations or corridors within or linking directly to Joshua Tree National Park are shaded in blue.

Table 36 (continued). High priority patches and corridors in the genetic and demographic networks of desert bighorn sheep. High priority features are defined as those with ranks among the top five (✓) in at least two of the four combinations of connectivity type (genetic, demographic) and network metric (effectively connected pairs [ECP; a measure of short-term connectivity] or mean weighted closeness [MWC; a measure of long-term connectivity]). Patch protection, protecting occupied habitat patches; patch restoration, restoring desert bighorn sheep populations to unoccupied habitat patches; corridor protection, protecting intact dispersal corridors; corridor restoration, removal of barriers. Patch abbreviations in Figure 37.

Management action	Patch or corridor	Genetic network			Demographic network		
Corridor restoration	GRA-MAR[2]	✓	✓	21	✓	✓	15
	NBR-SBR	✓	✓	21	✓	✓	15
	EMO-ORO[3]	✓	–	21	✓	✓	15
	CLI-PRO[2]	–	✓	21	–	✓	15

[1] Number of features associated with the given management action in the genetic or demographic network

[2] Populations or corridors within or linking directly to Mojave National Preserve are shaded in red

[3] Populations or corridors within or linking directly to Joshua Tree National Park are shaded in blue.

11.D. Data gaps and confidence in the assessment

More-extensive data on diet quality for desert bighorn sheep in the Mojave could facilitate both retrospective and prospective analyses of the species' population dynamics (Seager et al. 2007). A theme throughout our scoping discussions with NPS personnel, and therefore throughout this NRCA, is the extent to which remote sensing can provide reliable information on the condition and trend of resources that is not feasible to obtain in the field. As described in Appendix 13, we assessed whether the Normalized Difference Vegetation Index (NDVI) could be used as a surrogate measure of diet quality of desert bighorn sheep in the Mojave Desert. We believe that at the seasonal level and in a limited geographic area, NDVI indeed may be a reliable surrogate measure of diet quality. Use of NDVI to augment or replace measures of fecal nitrogen could dramatically improve the efficiency of assessments of diet quality of desert bighorn sheep in the Mojave Desert. Our results (Appendix 13) also support the retrospective application of NDVI to infer temporal trends in diet quality and associated probabilities of population persistence. The ability to efficiently characterize diet quality could facilitate the identification of promising locations for bighorn sheep reintroduction that remain unoccupied for reasons other than low-quality forage (e.g., anthropogenic disturbances, disease, low recolonization potential). NDVI may not support comparison of diet quality among geographically distant populations.

Chapter 12. Responses of natural resource conditions to water availability: water availability in montane springs

12.A. Rationale for and information provided by the assessment

Small, isolated montane springs are the primary natural source of water across much of the Mojave Desert Network, and the distribution and flow of these montane springs is associated with the distribution and viability of many native species of animals (Dekker and Hughson 2014). Flows of montane springs generally depend on groundwater (Patten et al. 2008). Personnel in the Mojave Desert Network requested an assessment of the condition of natural, ephemeral sources of water and potential trends in condition as climate changes. Such an assessment may inform management of anthropogenic sources of water throughout the Network.

12.B. Data and methods

Although the locations of many springs throughout the Mojave Desert are known (Appendix 14), their sources of flow are poorly understood. Monitoring of spring stability, water quality and other dynamic attributes, and short-term disturbances (e.g., grazing by domestic livestock) throughout the southwestern United States has been inconsistent in space and time (Earman and Dettinger 2008). Locations of areas that have been designated as wetlands by the federal government and delineated by the National Wetlands Inventory are available for all parks in the Mojave Desert Network, and additional information is available on wetlands and springs in Arizona (Appendix 14). However, the paucity of consistent data and historical data on springs and groundwater, lack of understanding about the regional hydrologic network, and considerable variation in size, microclimate, topography, and other attributes of individual springs makes it quite difficult to define reference conditions at the regional level. Given the lack of empirical data, we focused our assessment on the best information available in the literature.

12.C. Assessment results

A substantial proportion of montane springs in the Mojave Desert (e.g., 51% of the 238 surveyed in Mojave National Preserve [Dekker and Hughson 2014], 78% of 45 surveyed in the Spring Mountains [Sada et al. 2005]) have been excavated, fitted with diversion structures, or otherwise developed. Groundwater recharge and stores likely will decrease as climate continues to change and human demands for water increase. Therefore, the trend in condition of montane springs at the level of the Mojave Desert Network is likely to be stable or decreasing.

12.C.1. Responses of montane springs to climate change

The primary drivers of the condition and trend of montane springs are climate and human appropriation of water. These drivers have both direct and indirect effects on springs. For example, humans not only divert water from some springs but pump groundwater from aquifers that may discharge to springs. Responses of groundwater to climate at annual to decadal scales generally have not been the focus of substantial study, in part because it was assumed that surface water is more strongly affected by changes in climate than groundwater, and are not well understood (Earman and Dettinger 2011, R.G. Taylor et al. 2013). Earman and Dettinger (2011) reviewed the potential response of groundwater to climate change. They identified four mechanisms by which climate

affects the quantity and quality of groundwater: amount of precipitation, timing of precipitation, proportions of precipitation that fall as snow and rain, and evaporation and transpiration. Decreases in precipitation are likely to reduce groundwater recharge. Yet because temperature in the Mojave Desert is increasing, groundwater recharge likely will decrease even if the amount of precipitation remains constant or increases moderately. Moreover, wet years in the southwestern United States can induce unusual levels of vegetation growth, increasing total transpiration (RG Taylor et al. 2013). However, the increases in atmospheric carbon dioxide that contribute to increases in temperature may decrease transpiration in many plants (Earman and Dettinger 2011, RG Taylor et al. 2013). Evaporation and transpiration also are affected by cloud cover, wind, and humidity, all of which are likely to be affected by ongoing climate change (Earman and Dettinger 2011).

In the coming decades, the frequency of wet days and the precipitation amounts on those wet days are projected to decrease and increase, respectively, over the southwestern United States (Polade et al. 2014; also see Chapter 4). Even if the total amount of precipitation is constant, a decrease in the frequency of storms or warmer temperatures during periods of precipitation may decrease groundwater recharge because water availability at a given time will be insufficient to infiltrate or to persist in all soil layers (Earman and Dettinger 2011).

Because mountains in the western United States typically are wetter and cooler than basins, an estimated 50–90% of groundwater recharge in the region is supplied by precipitation that falls in the mountains (Earman and Dettinger 2008). Snowmelt is more likely to recharge groundwater than rain (Earman and Dettinger 2011). The snowpack typically holds precipitation from more than one storm; infiltrates the soil slowly and steadily; and is absorbed vertically into the soil rather than flowing horizontally over land. Additionally, snow begins to melt when temperatures and evaporation and transpiration rates are moderate (Earman and Dettinger 2011). The timing of maximum groundwater levels becomes earlier, and the duration of low streamflow increases, as the initiation of snowmelt becomes earlier and the proportion of rain to snow increases, and if rain falls on snow (RG Taylor et al. 2013).

Climate change may affect the quality of groundwater. On the one hand, dissolved subsurface material is diluted by precipitation. On the other hand, if increased recharge flushes salts from unsaturated soils, groundwater salinity may increase (Earman and Dettinger 2011). Decreases in precipitation, increases in temperature, and population growth in the southwestern United States all will increase human demands for surface water and groundwater in the Mojave Desert.

12.C.2. Interactions between groundwater and surface water

Groundwater typically contributes to surface water during periods with low flow; even during periods with high flow, groundflow contributes 25–75% of streamflow (Earman and Dettinger 2011). Increases in depth to groundwater that reflect decreases in recharge or storage of groundwater can have considerable effects on streamflow because streams typically receive groundwater from the upper part of the aquifer (Earman and Dettinger 2011). In Mojave National Preserve and perhaps elsewhere in the Mojave Desert, springs that are associated with bedrock outcrops and topography that constricts stream channels are more likely to persist through dry years than springs that are associated with geologic contacts or faults, which typically are fed by local precipitation and surface

flow (Dekker and Hughson 2014). Groundwater in the area around Lake Mead is relatively shallow; its levels, and in turn spring flow, may reflect local precipitation (Patten et al. 2008). It has been estimated that the water table in the Death Valley regional flow basin fluctuated by more than 10 m in the first half of the twentieth century as a result of natural variability in climate and groundwater withdrawals for agriculture.

12.D. Data gaps and confidence in the assessment

There remains considerable uncertainty about the mechanisms that affect groundwater in the Mojave Desert and relations between groundwater and surface water. Earman and Dettinger (2008) suggested that decadal commitments to monitoring recharge in the montane western United States are necessary to reduce those uncertainties. Installation and maintenance of clusters of monitoring wells and of stream gauges likely is feasible and would provide reliable long-term data. A monitoring network that is based on multiple methods, and that not only collects data but allows for research on the methods themselves, may have the greatest likelihood of success. Monitoring of springs is likely to be more informative than monitoring of unpumped wells.

Chapter 13. Responses of wildlife to changes in water availability: responses of natural resource conditions to water availability

13.A. Rationale for and information provided by the assessment

The condition and trend of wildlife throughout the Mojave Desert Network is responsive to changes in the availability of water, whether as surface water or as water in vegetation. Therefore, personnel in the Mojave Desert Network requested an assessment of potential responses of wildlife to changes in water availability. We focused on assessing how changes in water availability may affect the condition of these taxa rather than assessing their current condition per se. This assessment, like that of water availability in montane springs, may inform management of anthropogenic sources of water throughout the Network.

13.B. Data and methods

We examined peer-reviewed scientific literature that assessed or projected the effects on wildlife of recent and anticipated changes in the availability of water across the Mojave Desert. We use three examples to illustrate these relations: desert bighorn sheep (*Ovis canadensis nelsoni*), Mojave desert tortoise (*Gopherus agassizii*), and kangaroo rats (*Dipodomys* spp.). We selected these taxa because they are well-studied and represent a gradient of reliance on water. Moreover, desert bighorn sheep and Mojave desert tortoise are high priorities for management across the Mojave Desert. Our assessment does not account for potential adaptation (plasticity or evolution) of the taxa over years to decades.

Native populations of desert bighorn sheep occur throughout the Mojave Desert (Epps et al. 2003, Drost and Hart 2008). The species is considered by the NPS's Inventory and Monitoring Program to be common (i.e., relatively abundant) in Death Valley and Joshua Tree National Parks, Mojave National Preserve, and Lake Mead National Recreation Area (Drost and Hart 2008, NPS 2013) (although an outbreak of pneumonia in 2013 and 2014 killed at least several dozen adults) and uncommon in Grand Canyon–Parashant National Monument (NPS 2013). None of the subspecies or population segments of desert bighorn sheep that occupy the Mojave Desert Network are listed under the U.S. Endangered Species Act. However, the population of desert bighorn sheep (*O. c. nelsoni*) in the Peninsular Range adjacent to Joshua Tree and the subspecies of bighorn sheep (*O. c. sierrae*) in the Sierra Nevada adjacent to Death Valley (Epps et al. 2003) are listed as threatened.

The distinct population segment of desert tortoise in the Mojave Desert (north and west of the Colorado River; USFWS 2011) is listed as threatened under the U.S. Endangered Species Act, although populations in the Sonoran Desert are not. The Mojave desert tortoise is considered by the NPS's Inventory and Monitoring Program to be common in Mojave National Preserve (Persons and Nowak 2007) and Lake Mead, uncommon in Grand Canyon–Parashant, Joshua Tree, and Death Valley (Persons and Nowak 2006), and absent from Manzanar. Designated critical habitat overlaps with Lake Mead, Grand Canyon–Parashant, Joshua Tree, and Mojave National Preserve. Critical habitat is defined within the US Endangered Species Act as "specific areas within the geographical

area occupied by the species at the time of listing, if they contain physical or biological features essential to conservation, and those features may require special management considerations or protection; and outside the geographical area occupied by the species if the agency determines that the area itself is essential for conservation."

Although kangaroo rats are not a current focus of management within the Mojave Desert Network, granivorous rodents such as kangaroo rats are thought to play a primary role in structuring arid and semiarid ecosystems. For example, in the Chihuahuan Desert, species including banner-tailed kangaroo rat (*D. spectablis*), Merriam's kangaroo rat (*D. merriami*), and Ord's kangaroo rat (*D. ordii*) affect species diversity of plants and biogeochemical processes through seed predation and soil disturbance (Brown and Heske 1990). Research has been conducted on diverse species of kangaroo rat (Kelt 2011) and especially on endemic species in the Mojave Desert within California (Goldingay et al. 1997). At least two species of kangaroo rat are present in each of the Mojave Desert parks, and some are designated as rare or uncommon by the NPS Inventory and Monitoring Program (NPS 2013). The chisel-toothed kangaroo rat (*D. microps*) occurs in Death Valley, Lake Mead, Manzanar, Grand Canyon–Parashant, Joshua Tree, and Mojave National Preserve; it is considered rare in the latter two parks (NPS 2013). Panamint kangaroo rat (*D. panamintinus*) occurs in Death Valley, Manzanar, and Mojave National Preserve, whereas Ord's kangaroo rat (*D. ordii*) occurs in Death Valley, Lake Mead, and Grand Canyon–Parashant (NPS 2013). Merriam's kangaroo rat occurs in all six park units. Desert kangaroo rat (*D. deserti*) occurs in Death Valley, Lake Mead, Mojave National Preserve, and Grand Canyon–Parashant and is considered uncommon in Joshua Tree (NPS 2013).

Many data sources are relevant for assessing relations between wildlife, sources of water, and climate across the Mojave Desert. The Weather and Climate Inventory for the Mojave Desert Network (NPS 2007) assessed data from stations within and near parks in the network (except Grand Canyon–Parashant). The stations were members of any of 24 weather monitoring networks. The density of stations was sparse except near visitor centers or other locations with high levels of traffic (NPS 2007). Stations in populated areas near some parks, such as Joshua Tree, can supplement data from within the parks, but there are few surrogates to compensate for the lack of weather stations in parts of Death Valley and Lake Mead (NPS 2007). Historic data on temperature and precipitation from the Parameter-elevation Regressions on Independent Slopes Model (PRISM), Western Regional Climate Center, and National Climatic Data Center were used in a number of the publications we cite in this section, as were downscaled global climate models (GCMs) from the Fourth Assessment Report of the Intergovernmental Panel on Climate Change or the North American Regional Climate Change Assessment Program (which was based on GCMs included in the Third Coupled Model Intercomparison Project [CMIP3]; Garfin et al. 2013).

Artificial sources of water for wildlife are constructed and maintained in Arizona, Nevada, and California to supplement natural sources of water. The Arizona Game and Fish Department tracks the locations of artificial sources of water via its HabiMap program, whereas Nevada documents such sources in its Water Development Atlas. We did not find public, spatial data on artificial sources of water for wildlife in California. Cameras with infrared triggers in Mojave National Preserve have collected data on animals that visit a number of springs and water-storage tanks (guzzlers), but these

data do not allow inference to the potential effects of modification or loss of individual water sources.

Previous resource condition assessments for each park in the Mojave Desert Network yielded spatial data on wetlands, including riparian areas and streams. The BLM produced spatial data on riparian areas and semi-wet meadows for the Arizona Strip. Data from The National Map (nationalmap.gov/small_scale/) also document the locations of streams, water bodies, and wetlands.

Data on seeps, springs, and riparian vegetation across the Mojave Desert are maintained by the Museum of Northern Arizona Spring Stewardship Institute's Springs Inventory database. The Desert Renewable Energy Conservation Plan mapped vegetation in the western Mojave Desert at high resolution. The Vegetation Drought Response Index provides biweekly models of responses of vegetation to drought at 1 km resolution. The Landscape Fire and Resource Management Planning Tools Project (LANDFIRE) described environmental site potentials, or vegetation that could occur in a given location given the location's biophysical attributes.

We compiled peer-reviewed journal articles, agency reports, book chapters, and conference proceedings on water requirements of desert bighorn sheep, Mojave desert tortoise, and kangaroo rats and on phenomena that affect the availability of water across the Mojave Desert Network (Appendix 14).

Spatially explicit estimates of the distribution of habitat for bighorn sheep, desert tortoise, and kangaroo rats are publicly available (Appendix 14). For example, the Arizona Game and Fish Department and California Department of Fish and Wildlife estimate distributions of bighorn sheep and kangaroo rats across the state. However, these spatially extensive data may not be useful at the level of national parks. Previous park-level resource condition assessments included estimates of habitat for listed species such as Mojave desert tortoise, Sierra Nevada bighorn sheep, and Peninsular bighorn sheep. Spatial data on habitat for desert bighorn sheep and desert tortoise also were published by the BLM for the Arizona Strip, including Lake Mead and Grand Canyon–Parashant. Descriptions of population distributions at the park level that were not spatially explicit were developed by USGS via the Mojave Desert Mammal Inventory (for desert bighorn sheep and kangaroo rats) and an assessment of sensitive plant and animal species in the western Mojave Desert (for desert tortoise). Additionally, temporally extensive projections of the distributions of bighorn sheep and kangaroo rats are available through the Faunmap project.

13.C. Assessment results

13.C.1. Potential responses of desert bighorn sheep to changes in water availability

The condition of desert bighorn sheep across the Mojave Desert Network is likely to decrease given projected changes in precipitation and water availability. Desert bighorn sheep are well-adapted to short-term water limitation. Individuals can survive without free-standing water for up to ten days (Turner and Weaver 1980, Krausman et al. 1985, Alderman et al. 1989) through oxidative metabolism or consumption of moisture-rich vegetation such as saguaro (*Carnegiea gigantea*) fruits, barrel cacti (*Ferocactus* and *Echinocactus* spp.; Turner and Weaver 1980, Warrick and Krausman 1989, Oehler et al. 2003), and littleleaf paloverde (*Cercidium microphyllum*). However, desert

bighorn sheep have relatively high water requirements in an absolute sense. Some studies have indicated that bighorn sheep require at least 4% of their body weight in water per day (Turner 1970), although the percentage varies as a function of age, sex, and body size (Turner 1970, Krausman and Czech 1998, Whiting et al. 2010). The probability of extirpation of desert bighorn sheep populations in the Mojave Desert was strongly and negatively correlated with total precipitation (Epps et al. 2004).

The responses of desert bighorn sheep to changes in the number of anthropogenic water sources (sometimes referenced as developed waters) or the relative proportion of natural and anthropogenic water sources are difficult to project with confidence. The presence of surface water increases habitat quality for bighorn sheep and is associated with the distribution of the species (Krausman et al. 1985, Bleich et al. 2010). Some populations of bighorn sheep depend on anthropogenic water sources (Leslie and Douglas 1979, Sappington et al. 2007; but see Broyles 1997, Cain et al. 2006, Krausman et al. 2006). Nevertheless, desert bighorn sheep are thought to prefer natural to artificial sources of water (Hailey 1967), and in some cases the behavior of desert bighorn sheep has not changed in response to newly available sources of developed water (Krausman and Etchberger 1995) or variation in the reliability of those sources (Cain et al. 2008).

Responses of desert bighorn sheep to changes in water availability may vary among life stages. Large, adult herbivorous mammals that are reproductive, including bighorn sheep, are more resilient to density dependence and inclement weather than are juveniles (and possibly older generations) (Gordon et al. 2004). Survival of pre-weaning young and female fecundity of large herbivores often is positively correlated with the amount of winter and spring precipitation (Gaillard et al. 2000; McKinney et al. 2001, 2006; Colchero et al. 2009). Both abundance and lamb-to-ewe ratios of bighorn sheep decreased during extended periods (e.g., ≥ 30 months) of low precipitation (Wehausen et al. 1987, Douglas 2001, Marshal et al. 2009). Decreases in the amount of precipitation or increases in the variability of precipitation also can affect both the quantity and the quality of forage, including its water content (Wehausen et al. 1987, McKinney et al. 2006). In environments similar to that of the Mojave Desert, such as central Arizona's Mazatzal Mountains, winter precipitation was associated with the nutritional status of desert bighorn sheep via increased quantity of forage and levels of phosphorous, selenium, and zinc in vegetation (McKinney et al. 2006).

As precipitation becomes more variable, the abundance of bighorn sheep also is likely to become more variable. Increases in the variability of precipitation may affect the phenological overlap of bighorn sheep with green-up, especially in populations that move seasonally to forage (Leslie and Douglas 1979, Festa-Bianchet 1988, Berger 1991). Historically, seasonal movements by desert bighorn sheep were associated with pluvial cycles that caused changes in phenology (Leslie and Douglas 1979). Rapid changes in the phenology of plants on which ungulates feed are associated with declines in the health of juveniles (Pettorelli et al. 2007).

13.C.2. Potential responses of Mojave desert tortoise to changes in water availability

The status of Mojave desert tortoise is likely to decrease given projected changes in precipitation and temperature. Mojave desert tortoises are active at body temperatures from 20–38°C and actively seek shelter to avoid higher temperatures (Barrows 2011). The species requires food and drinking water to

maintain activity and growth throughout the year (Henen et al. 1998). Desert tortoises can aestivate during dry periods in late spring and early summer (Nagy and Medica 1986) and construct and maintain multiple burrows simultaneously (Medica et al. 1980). Some studies have indicated that the primary water source for desert tortoises is moisture-rich vegetation and that land tortoises generally can survive without free-standing water (Medica et al. 1980). However, desert tortoises drink and store surface water when it is available (Henen et al. 1998), whether in naturally occurring puddles or in human-constructed water-collection holes (Medica et al. 1980). Desert tortoises can drink the equivalent of 11–28% of their body mass and retain 58–93% of that water for multiple days after drinking (Nagy and Medica 1986). Desert tortoises have large urinary bladders to store water and waste, and reduce their consumption during dry periods. Therefore, if local precipitation allows, desert tortoises can maximize water intake while minimizing potentially toxic levels of dietary ionic potassium (K+) that occur in water-stressed plants (Medica et al. 1980, Oftedal 2002).

Decreases in water availability may reduce fecundity of desert tortoises. Water can be a limiting resource for desert tortoises, especially for egg production (Henen et al. 1998). Water intake during summer and early autumn can affect whether females produce eggs during the following spring (Henen et al. 1998). Incubation temperatures regulate sex ratios, and an increase in temperature may result in an increase in the proportion of female hatchlings (Lewis-Winokur and Winokur 1995, Barrows 2011). Moreover, hatching occurs later in relatively cool years than in relatively warm years (Lovich et al. 2012).

The physiology of Mojave desert tortoises can change substantially among seasons as the water content of vegetation changes. For example, desert tortoise activity during July declines and individuals lose body water and body mass because forbs and grasses do not provide the animals with sufficient water (Nagy and Medica 1986). Weights of individual desert tortoises can decrease by ≤5% over 6–8 weeks without rain (Medica et al. 1980). Peak metabolic rates measured in the field were higher in years with above-average rainfall than in years with average rainfall (Henen et al. 1998). Survival of desert tortoise also has been directly linked to forage quality (Nagy and Medica 1986). Short, intense precipitation events allow desert tortoises to drink and, when precipitation exceeds 25 mm, stimulate vegetation production (Beatley 1974, Nagy and Medica 1986).

Survival of adult desert tortoise is highly dependent on annual production of plant biomass and on not only the amount but the timing of precipitation. Droughts of 1–3 years in which little or no biomass was produced were associated with ≤ 30% decreases in survival (Longshore et al. 2003). Winter rains are the major source of water for Mojave desert tortoises in the western Mojave Desert whereas both winter and monsoonal summer rains affect desert tortoises in the eastern Mojave (Henen et al. 1998). Oftedal (2002) outlined potential responses of desert tortoises in the Mojave and Sonoran Deserts to four scenarios of water availability: a year of drought, a year of low winter rainfall, a year of high winter rainfall, and, in the eastern Mojave, a year of high summer rainfall. Condition of desert tortoise may be stable during droughts of one or two years' duration, which have occurred regularly during the past several centuries. During such droughts, little food is available for desert tortoise because annuals may not germinate and herbaceous perennials may not grow (Oftedal 2002). In these circumstances, desert tortoises reduce their activity and remain in their burrows,

rarely emerging to forage (Nagy and Medica 1986, Henen et al. 1998). If and when rains arrive, desert tortoises can become active at relatively low temperatures (9–15°C) because rainfall after a dry period allows tortoises to emerge and rehydrate, and facilitates the consumption of dry forage (Medica et al. 1980, Oftedal 2002).

The condition of desert tortoise generally decreases during years with low winter rainfall. When winter rainfall in the Mojave Desert is < 25 mm, most winter annuals and perennials do not grow (Beatley 1974). However, a small number of non-native species (e.g., filaree [*Erodium cicutarium*] and red brome [*Bromus rubens*]) and a limited set of native herbaceous perennials (e.g., fiddleneck [*Amsinckia tessellata*] and woolly plantain [*Plantago ovata*]) are productive. During these periods, the nutritional quality of forage for desert tortoises decreases and it becomes more challenging for the animals to meet their requirements for water and protein without reaching high, and potentially toxic, potassium levels that impede nitrogen retention (Oftedal 2002).

Condition of desert tortoise may be fairly good during years with high winter rainfall, which allow for substantial germination of annual and perennial plant species on which desert tortoises forage (Oftedal 2002). During these periods, desert tortoises may be relatively active and consume relatively large amounts of plants with high water content (Oftedal 2002).

Condition of desert tortoise also may be fairly good during years with intense and localized spring and summer storms (typically occurring March through November) in the eastern Mojave Desert. These precipitation patterns lead to local germination of annual plants and growth of perennial plants, and to an increase in surface activity of desert tortoise (Medica et al. 1980). Although desert tortoises obtain most of their water from winter annuals and spring-active herbaceous perennials, free-standing water during the summer allow for greater consumption of senescent and dry grasses and greater deposition of body fat, and increases resiliency to future drought (Henen et al. 1998, Oftedal 2002). Summer rains in the western Mojave Desert have been comparatively rare (Oftedal 2002).

13.C.3. Potential responses of kangaroo rats to changes in water availability

The condition of kangaroo rats across the Mojave Desert Network is likely to decrease as climate-driven water availability decreases. However, responses to climate change likely will vary considerably among species, and kangaroo rats are better adapted to seasonal or short-term drought than many other taxa.

Desert rodents are nocturnal and lose relatively little water through their lungs, skin, and excretions (Yousef et al. 1974, Goldingay et al. 1997). Compared to other species of rodents, kangaroo rats use little water per unit of oxygen consumed (Yousef et al. 1974). Projected increases in the variability of precipitation and temperature may reduce growth of kangaroo rats. For example, body size of the Dulzura kangaroo rat (*D. simulans*), which may occur in Joshua Tree National Park, is smaller in areas with substantial seasonal changes in precipitation and temperature than in areas with less variation among seasons (Sullivan and Best 1997). In some cases, changes in short-term and long-term precipitation patterns also may affect elevation ranges of kangaroo rats. For instance, the minimum elevation at which the Panamint kangaroo rat occurs was associated with precipitation-related food availability (Sullivan and Best 1997). Changes in the distribution of a given species of

kangaroo rat may have cascading effects on the distributions of other species. The upper elevation range of Merriam's kangaroo rat, perhaps the most abundant and ubiquitous native rodent in the Mojave Desert (Chew and Butterworth 1964), was associated with the lower limit of the distribution of the Panamint kangaroo rat (Sullivan and Best 1997).

It is difficult to project future trends in the abundance of kangaroo rats, which is positively associated with the amount of winter and summer precipitation (Price et al. 2000), given the considerable uncertainty about the future seasonal distribution of precipitation. The distributions of some species, including the chisel-toothed kangaroo rat, may be restricted to locations with relatively high precipitation-to-temperature ratios. Because shrub cover reduces evaporative loss of water from the soil, the probability of occurrence of some kangaroo rat species increases as shrub cover increases (Beatley 1976). Merriam's kangaroo rat can persist in areas with little to no shrub cover and relatively high temperatures. By contrast, chisel-toothed kangaroo rat typically is absent from locations where shrub cover is less than 23% and occurs where temperatures are relatively low.

Kangaroo rats can survive without drinking free-standing water and can oxidize food to obtain water from vegetation (Yousef et al. 1974, Soholt 1977, Frank 1988). Kangaroo rats that occur in the Mojave Desert persist mostly on dry seeds such as those of creosote bush (*Larrea tridentata*) during late spring to early or mid-winter. Seeds are collected along the desert floor and cached in burrows where absorption from high-humidity air increases the water content of the seeds (Nagy and Gruchacz 1994). Seeds of some Mojave Desert plants can absorb up to 25% of their mass as water from vapor exchange from humid air (Morton and Macmillen 1982, Nagy and Gruchacz 1994). In late winter to early spring, kangaroo rats primarily consume green vegetation but supplement their diets with arthropods, such as ants and beetles (Nagy and Gruchacz 1994). Some studies have indicated that the diet of Merriam's kangaroo rat, which includes foods with low lipid content that maximize net metabolic water gain (Frank 1988), allows for limited stress in animals of most life stages during the summer (Nagy and Gruchacz 1994). An exception is lactating females, which typically lose body mass during the summer (Nagy and Gruchacz 1994). The consumption of vegetation by reproductive female kangaroo rats tends to increase during periods of highest reproductive activity; increased moisture intake from green vegetation facilitates milk production (Soholt 1977). Increases in water intake are associated with increases in litter size (Soholt 1977).

13.D. Data gaps and uncertainty

Uncertainty about the potential responses of wildlife to changes in water availability across the Mojave Desert Network might be reduced substantially by collection of additional field data that are amenable to statistical modeling. In our experience, rigorous statistical models of habitat quality (of which water availability is a major component) for wildlife require a minimum of 30 occurrence records from each of a minimum of a dozen individuals that were collected in a systematic and comprehensive manner (Noon et al. 2009). Data might be gathered through camera trapping, telemetry, or other systematic aerial or ground-based surveys. Given such data on species occurrence and data on water availability, there are numerous methods for deriving spatially explicit models of habitat quality and connectivity within a park or across the Mojave Desert. The outputs of such models can be presented as maps. Spatial extent and resolution may affect the outcome of a model

and the associated inferences for management (Wiens 1989, Flather et al. 2009). Therefore, the methods used to quantify habitat quality and connectivity require one to consider how different species interact with water sources and other environmental features during periods of stasis or movement (Tischendorf and Fahrig 2000), and how these behaviors or resources are affected by management actions (Noon et al. 2009).

Commonly derived models of wildlife habitat quality and connectivity typically are implemented within a geographic information system (GIS; e.g., ArcGIS, QGIS) with R software (www.r-project.org) or other freely available software packages. If the number or types of occurrence records from within individual parks or across the Mojave Desert Network are too sparse to permit rigorous statistical modeling), then the empirical data from the parks could be coupled with data obtained in adjacent jurisdictions, or from areas with similar environmental attributes. When empirical data on occurrence are limited, models of habitat quality also may be derived through expert elicitation (Johnson et al. 2012, Martin et al. 2012, Dickson et al. 2013a).

For example, to model and map habitat quality for mule deer (*Odocoileus hemionus*) within Grand Canyon–Parashant, Horncastle et al. (2013) used a GIS to estimate the Euclidean distance between more than 5750 radio-telemetry locations from 10 animals and reliable water sources (e.g., trick tanks). The locations of water sources were derived from records of the Bureau of Land Management and Arizona Game and Fish Department (Appendix 14) and verified in the field. The statistical model generated by Horncastle et al. (2013) indicated that mule deer were moderately more likely to use areas near reliable water sources than areas further away. Intensity of space use was strongly associated with slope (negative relation), thinning and burning treatments (positive relation), and areas dominated by ponderosa pine (positive relation).

As another illustration, for 41 desert bighorn sheep (*O. c. mexicana*) in the western Sonoran Desert of Arizona, Hoglander et al. (2015) developed statistical models of seasonal space use (i.e., habitat quality) that integrated information on the Euclidian distance from 103 artificial water sources to locations of bighorn sheep recorded by global positioning system collars. These water sources included apron catchments, drainage dams, tanks, and wells and human-modified ponds, springs, and tinajas. The models also included estimates of the normalized difference vegetation index (NDVI), which permitted an analysis of the relative association of space use with both forage availability and preformed water sources (that is, water in vegetation). Although the results of these models indicated that escape terrain was the environmental variable most strongly associated with space use, forage and preformed water sources were strongly associated with space use in the non-summer season.

The methods for data acquisition, specification of environmental variables, and statistical modeling used by Horncastle et al. (2013) and Hoglander et al. (2015) are transferable to estimation and mapping of habitat quality for other wide-ranging species that occur within the Mojave Desert Network. Provided sufficient occurrence data are available, similar methods could be used to estimate habitat quality for those species with more limited or restricted distributions. Alternative model structures for projecting habitat quality, including occupancy and niche modeling methods, also can integrate spatially explicit data on water availability (Noon et al. 2009, Dickson et al. 2013b).

The fundamental ecological processes of long-distance adult movement and juvenile dispersal can be projected with spatially extensive models of habitat connectivity (Dickson et al. 2013b). The availability of water is often a major contributor to connectivity of habitat for species in the Mojave Desert and other arid regions (Dickson et al. 2013a, Creech et al. 2014). Spatial data on natural lakes, reservoirs, rivers, and canals can obtained from local, regional, or national sources of hydrographic data and incorporated into connectivity models (Dickson et al. 2013b) (Appendix 14). Spatial data on the distance from a given location (a pixel, for example) to a water source can be combined with other spatial data (e.g., terrain, vegetation, climate, natural or artificial barriers to movement) to derive estimates of a region's resistance to movement of a given species (Dickson et al. 2013a, LaPoint et al. 2013). The resistance surface is a common input for multiple models of connectivity, including those based on least-cost path analysis (Singleton et al. 2002), circuit theory (McRae et al. 2008), graph theory (Bunn et al. 2000), and flow paths (Carroll et al. 2012). The outputs of connectivity models (typically rasters or grids) can range from quantitative estimates of movement likelihood or cost to simple measures of the quality of habitat for movement. When summarized in map form, model outputs can help identify relatively high levels of habitat quality or connectivity, highlighting locations where water availability and associated management actions may affect conditions for one or more co-occurring species (Dickson et al. 2013b).

Chapter 14. Effects of feral burros on natural resources

14.A. Rationale for and information provided by the assessment

Burros were introduced to the Mojave Desert in the early nineteenth century. They were used extensively by miners and other early settlers as work animals because they are well-adapted to desert environments. Abandoned burros formed feral populations that gradually increased in size over the next several decades. The Mojave Desert Network was interested in an assessment of how burros might affect soils and vegetation in the region, and whether burros may compete with wildlife and domestic livestock. This assessment might inform management of burros to the extent feasible given legal protection of the species.

14.B. Data and methods

Because recent surveys of burros in Mojave Desert Network parks have not been conducted, we used anecdotal information on abundance and locations of burros and previously published information to conduct the assessment.

14.C. Assessment results

Burros were introduced to the Mojave Desert in the early nineteenth century. They were used extensively by miners and other early settlers as work animals because they are well-adapted to desert environments. Abandoned burros formed feral populations that gradually increased in size over the next several decades. Because feral burros were perceived to damage soils and vegetation and to compete with wildlife and domestic livestock, killing of burros was unregulated for many decades. In 1971, the Wild Free-Roaming Horses and Burros Act (Public Law 92–195) was signed in response to public desire to protect these species. The act prevents the killing or harassment of feral horses and burros on federal lands with the exception of national parks and wildlife refuges. Thus, although the NPS has jurisdiction to remove wild horses and burros from their lands, other federal land management agencies, such as the BLM and USFS, do not and are required to manage them as part of the ecosystem. Burro population sizes have increased substantially since the 1970s.

Numerous observations of reductions in vegetation cover, soil disturbance, and disturbance to springs as a result of long-term grazing by burros have been recorded, but quantitative data are limited (Seegmiller and Ohmart 1981, Abella 2008). Reduced soil-surface stability has been related to presence of domestic livestock and burros in Mojave National Preserve, with soil stability increasing as the time since cessation of grazing increases (Beever et al. 2006). Existing quantitative information on burro diet and effects of burros on Mojave Desert vegetation were reviewed by Abella (2008). The review concluded that burros reduce native perennial grass cover. The review also suggested that in the Mojave Desert, grasses and forbs are a higher proportion of the diet of burros than shrubs.

Competition between feral burros and domestic livestock or wildlife species in the Mojave Desert could occur through dietary overlap or displacement from high-quality habitat or water sources. Habitat (Weaver 1974) and dietary (Ginnett and Douglas 1982) overlap between feral burros and desert bighorn sheep has been observed in the Mojave Desert, but no studies have been conducted that explicitly identified competitive interactions between the two species. In the Sonoran Desert,

burro abundance had a significantly negative association with abundance of bighorn sheep abundance, suggesting that the two species compete. However, the mechanism by which they might compete could not be identified (Marshal et al. 2008). Another study in the Sonoran Desert indicated that habitat overlap between mule deer (*Odocoileus hemionus*) and burros is minimal and that dietary overlap was biologically significant during the summer but not during other seasons (Marshal et al. 2012). Whether burros and mule deer compete or simply occupy different areas was inconclusive.

In the late 1990s and early 2000s, many burros (exact numbers are not available) were removed from Death Valley, Mojave National Preserve, and Lake Mead. However, lack of fencing infrastructure and maintenance allowed burros to trespass into the parks from neighboring public lands and to persist tin the parks. Current abundances in Mojave National Preserve and Lake Mead are estimated to be in the hundreds (Robert Bryson and Ross Haley, personal communication). In Grand Canyon–Parashant, trespass burros also are present. If fencing to exclude burros is being considered as a management option, a cadastral survey would be necessary to identify boundaries of the monument. Due to its relative isolation from areas occupied by burros, burros are not present in Joshua Tree. We were unable to obtain data on burros from Death Valley.

14.D. Data gaps and uncertainty

The long-term responses of wildlife, vegetation, and soils in the Mojave Desert to grazing by burros are difficult to identify with high certainty given that effects analyses were not initiated until after burros and other livestock had been present for more than a century (Abella 2008). Replicated, long-term exclosure studies that control for soil type, vegetation type, and burro density have potential to provide strong inference of effects of burros on soils and vegetation. Control-impact studies that compare current demographics of wildlife populations in areas with and without burros, or before and after removal of burrow, could provide insights to the biological significance of interactions between wildlife and burros.

Chapter 15. Responses of bats to environmental stressors

15.A. Rationale for and information provided by the assessment

Of the 45 species of bats that occur in the United States, 23 occur in parks within the Mojave Desert Network (Table 37). A number of these species have been listed as priorities by the Western Bat Working Group, a partner in the Coalition of North American Bat Working Groups that includes agencies, organizations, and individuals interested in bat research, management, and conservation from 13 western states, British Columbia, Alberta, and northern Mexico (Table 37). Habitat loss, climate change, the emerging disease white-nose syndrome, and development of wind power have been hypothesized as causes of rapid declines in the distribution and abundance of bats in the Mojave Desert. However, the magnitude and extent of these declines are not well estimated given that most species are cryptic and understanding of the population dynamics and ecology of bats is limited (Rodhouse et al. 2012).

Table 37. Bat species that occur in Death Valley National Park, Manzanar National Historical Site, Mojave National Preserve, Joshua Tree National Park, and Lake Mead National Recreation Area4, selected life history traits, and status as assigned by the Western Bat Working Group.

Family	Scientific name	Common name	Hibernates[4]	Known to roost in mines[5]	Status[6]
Molossidae	*Eumops perotis*[1]	western mastiff (or western bonneted bat)	–	–	high
	Nyctinomops femorosaccus[2]	pocketed free-tailed bat	–	–	medium
	Nyctinomops macrotis[1]	big free-tailed bat	–	–	medium high
	Tadarida brasiliensis	Brazilian free-tailed bat	–	yes	medium
Phyllostomidae	*Macrotus californicus*	California leaf-nosed bat	–	yes	high
Vespertilionidae	*Antrozous pallidus*[1]	Pallid Bat	yes	yes	high
	Corynorhinus townsendii[1]	Townsend's Big-eared Bat	yes	yes	high
	Eptesicus fuscus	big brown bat	yes [3]	yes	low
	Euderma maculatum[1]	spotted bat	yes	–	high
	Idionycteris phyllotis	Allen's big-eared bat	yes	–	high

[1] Documented occurrence in Grand Canyon–Parashant (Herder 2000, Wynne et al. 2005, BLM 2008, Wynne and Voyles 2011)

[2] Unconfirmed but potentially occurs in these parks

[3] The fungus that causes white-nose syndrome has been isolated from individuals of this species (Rogall and Verant 2012)

[4] Source: Cryan 2012

[5] Source: AMEC 2011a

[6] Source: Western Bat Working Group 1998

Table 37 (continued). Bat species that occur in Death Valley National Park, Manzanar National Historical Site, Mojave National Preserve, Joshua Tree National Park, and Lake Mead National Recreation Area4, selected life history traits, and status as assigned by the Western Bat Working Group.

Family	Scientific name	Common name	Hibernates[4]	Known to roost in mines[5]	Status[6]
Vespertilionidae	*Lasionycteris noctivagans*	silver-haired bat	–	–	medium
	Lasiurus blossevillii	western red bat	–	–	high
	Lasiurus cinereus	hoary bat	–	–	medium
	Lasiurus xanthinus	western yellow bat	–	–	high
	Myotis californicus	California myotis	yes	yes	low
	Myotis ciliolabrum[1]	western small-footed myotis	yes	yes	medium
	Myotis evotis[1,2]	long-eared myotis	yes	yes	medium
	Myotis lucifugus[2]	little brown myotis	yes [3]	yes	medium
	Myotis thysanodes[1]	fringed myotis	yes	yes	high
	Myotis velifer[2]	cave myotis	yes	yes	medium
	Myotis volans[1]	long-legged myotis	yes	yes	high
	Myotis yumanensis	Yuma myotis	yes	yes	medium low
	Parastrellus hesperus	western pipistrelle	yes	yes	low

[1] Documented occurrence in Grand Canyon–Parashant (Herder 2000, Wynne et al. 2005, BLM 2008, Wynne and Voyles 2011)

[2] Unconfirmed but potentially occurs in these parks

[3] The fungus that causes white-nose syndrome has been isolated from individuals of this species (Rogall and Verant 2012)

[4] Source: Cryan 2012

[5] Source: AMEC 2011a

[6] Source: Western Bat Working Group 1998

15.B. Data and methods

Because data are limited, we used a literature review to assess whether and how the status of bats in the Mojave Desert Network may be affected by three potential stressors: closures of abandoned mines, development of wind power, and infection with white-nose syndrome. We identified information gaps and suggested ways to fill the gaps.

15.C. Assessment results

15.C.1. Potential effects of closing abandoned mines

Abandoned mines provide habitat for many of the bat species that occur in the Mojave Desert (Altenbach and Pierson 1995), particularly because land use has reduced the extent and quality of natural habitat and led to population declines throughout the western United States (Tuttle and Taylor 1998). In 2011, although inventories for Death Valley National Park and Mojave National Preserve

were not yet complete, the NPS estimated that the Mojave Network included approximately 914 mine sites and 5185 mine features—34% of sites and 57% of features within the National Park System (Table 38) (NPS 2011). Fourteen of the bat species that occur in the Mojave Network roost in mines (AMEC 2011b; Table 37). Some of these species are cave or mine obligates, whereas others use caves or mines opportunistically (Burghardt 2003). Bats were present at a minimum of 57% of the 228 mine features sampled in Death Valley National Park, 28% of the 205 mine features sampled in Mojave National Preserve, and all 22 of the mine features sampled in Joshua Tree National Park (ECORP 2010, AMEC 2011b, MACTEC 2011).

Table 38. Abandoned mines and abandoned mine features in the Mojave Desert Network (NPS 2011).

Park	Number of abandoned mines	Number of abandoned mine features
Death Valley	143	3000
Grand Canyon–Parashant	3	11
Joshua Tree	132	395
Lake Mead	340	844
Manzanar	not reported	not reported
Mojave	636	1779

From 1978 through 2000, thousands of mines were closed across the southwestern United States. Although bats are estimated to occupy 30–70% of the mines in the United States, an estimated 5% of mines were closed with methods that do not exclude bats (Meier and Garcia 2000). In 1984, the NPS initiated the Abandoned Mineral Lands Program to minimize toxicants and risks to public safety, preserve historic features, and maintain habitat for wild animals, including bats (Burghardt 2003).

Bat gates are commonly used to meet both public safety and habitat conservation objectives associated with mine closures. Gates are designed and installed to exclude humans without restricting bat flight or altering microclimate (temperature and humidity) (Burghardt 2003). There is substantial guidance on design and installation of bat gates (Vories and Throgmorton 2002, Fant et al. 2009, BCI 2013). Nevertheless, there have been few scientifically robust, quantitative studies on the effects of mine closures on bat populations. Bat gates are believed to reduce the high energetic costs to bats associated with anthropogenic disturbance during roosting or hibernation (Speakman et al. 1991), increase habitat stability by stabilizing the mine entrance, and minimize human disturbances to the mine (Burghardt 2003). However, bats have abandoned mines at which gates were installed poorly (Sherwin et al. 2002). Flight behavior may change after gate installation; for example, circling may increase (Grandison et al. 2000; Spanjer and Fenton 2005). Additionally, bats that are sensitive to the size of roost openings may abort attempts to enter and exit the mine opening after gates are installed (Pugh and Altringham 2005, Slade and Law 2008). These changes in behavior can have energetic costs (Lee and McCracken 2001). Biologically significant differences in the microclimates of gated versus ungated mines and caves have not been detected in the limited studies to date (King 2005; Martin et al. 2006).

15.C.2. Potential effects of development of wind energy

Bat mortalities associated with wind turbines are well documented (Arnett et al. 2008, Cryan 2011, Ellison 2012). Reviews and syntheses have identified some generalizations about traits of species most likely to be killed by wind turbines, the primary causes of mortality, and operations that may minimize probability of mortality.

At the national level, the majority of bat mortalities associated with wind turbines are of migratory, tree-roosting species, especially hoary bat (*Lasiurus cinereus*) and to a lesser extent silver haired bat (*Lasionycteris noctivagans*) and eastern red bat (*Lasiurus borealis*; Arnett et al. 2008). In California, mortality of long-legged myotis (*Myotis volans*), little brown myotis (*Myotis lucifugus*), western red bat (*Lasiurus blossevillii)*, big brown bat (*Eptesicus fuscus*), silver-haired bat, Brazilian free-tailed bat (*Tadarida brasiliensis*), western long-eared myotis (*Myotis evotis*), hoary bat, and big free-tailed bat (*Nyctinomops macrotis*) has occurred at wind farms (BLM 2012). All of the latter species occur in the Mojave Desert Network parks (Table 1). Within the Mojave Desert, Brazilian free-tailed bats, big free-tailed bat, and western mastiff bat (*Eumops perotis*) may be particularly vulnerable to mortality given their tendency to forage at high altitudes (CBWG 2006). No association has been detected between bat mortalities and lighting of wind facilities (Arnett et al. 2008) or siting of wind turbines (Arnett et al. 2008, Cryan 2011). However, bat mortality increased as the size of turbines increased (Barclay et al. 2007), and mortality was greater during mating periods, autumn migration (Willis et al. 2010), periods with low wind speeds (Horn et al. 2008, Arnett et al. 2008), and ahead of and immediately following storm fronts (Arnett et al. 2008).

The few studies that examined whether bat mortalities at wind turbines were caused primarily by blunt-force trauma or barotrauma indicated that all bats experienced severe blunt force trauma (Baerwald et al. 2008, Grodsky et al. 2011, Rollins et al. 2012). Some of the latter bats also had possible signs of barotrauma, but etiological evidence suggested that blunt force was the major cause of mortality (Rollins et al. 2012).

15.C.3. Effects of white-nose syndrome on bats

White-nose syndrome, which is caused by the fungus *Geomyces destructans*, has substantially reduced the abundance of hibernating bats in the northeastern United States (Foley et al. 2011). Although the mechanism of mortality is not fully understood, severity of infection in hibernating bats is positively correlated with the frequency of arousals from torpor. Arousals increase body temperatures and reduce energy stores, ultimately leading to starvation and death (Reeder et al. 2012). It has been hypothesized that *G. destructans* increases arousal frequency by dehydrating membrane surfaces (Willis et al. 2011).

G. destructans occurs in environments with high humidity and can survive at temperatures above 3˚ C. Depending on the strain of the fungus, growth is optimal from 12.5–15.8˚ C, with an upper limit of 19–20˚ C (Verant et al. 2012). Growth rates of the fungus are sensitive to small changes in temperature, translating into differences in severity of infection on the basis of small changes in microclimate within a hibernaculum (Verant et al. 2012). Bats in relatively cold regions may be more susceptible to mortality than bats in relatively warm regions given the higher energetic costs

associated with disruption of torpor (Boyles and Willis 2009) and that longer durations of hibernation increase the likelihood of exhausting energy stores (Flory et al. 2012, Hallam and Federico 2012).

White-nose syndrome was detected in the northeastern United States in 2006. It spread steadily and by early 2013 had been detected as far west as Missouri (Butchkoski 2013). Although the fungus can be dispersed via animals, wind, or water (Hayes 2012), transmission is thought to occur primarily through contact between bats (Lorch et al. 2011). The fungus can persist in cave substrates in the absence of bats (Lorch et al. 2013), and transmission through contact with cave environments in which the fungus is present also is possible (Lindner et al. 2011, Puechmaille et al. 2011). Density-dependent declines in abundance of solitary bats have been observed. By contrast, species that cluster in a hibernaculum are linked to high transmission rates regardless of population size (Langwig et al. 2012). Migration also is thought to drive the spread of white-nose syndrome (Frick et al. 2010), and the presence of migratory bats within a hibernaculum may increase probability of infection (Sullivan et al. 2012).

Fourteen of the 23 species of bats that occur in the Mojave Desert Network generally are classified as hibernating species (Table 1) and may have the highest probabilities of infection with white-nose syndrome (Cryan 2012). Two of these species, big brown bat and especially little brown myotis, were affected by white-nose syndrome in the eastern United States (Foley et al. 2011). Climate data from caves and mines in the Mojave Desert Network and across the western United States indicate that conditions could support *G. destructans* growth. However, it is unclear how *G. destructans* will respond to the lower humidity typical of the arid and semi-arid western United States.

15.D. Data gaps and uncertainty

A small number of studies in Death Valley National Park, Mojave National Preserve, and Joshua Tree National Park that included one or two visits before and after mine closure indicated that bats continued to use all abandoned mine features within seven months after closure (AMEC 2011a, b; MACTEC 2011). It is unknown whether these or other closures may affect behavior, population size, or habitat quality for different species. Effects ultimately will depend on interactions between the closure design and installation and the species present, type of roost, and shape, size, and orientation of the mine feature. Pre- and post-closure monitoring is necessary to identify changes in use, behavior, or microclimate that may affect population dynamics of bats. Monitoring ideally should be initiated at least one year before to closure because the effects of installing bat gates cannot be determined without pre-treatment data (Herder 2002). Microclimate monitoring allows one to determine whether gating has affected this aspect of habitat quality (Kennedy 2002). Ideally, monitoring of microclimate should be conducted at a fine enough temporal resolution and over a sufficiently long duration to capture climatic variability. Such monitoring typically can be achieved at low cost with automatic data loggers (Kennedy 2002). Bat species composition, abundance, timing of use, and emergence can be monitored with a variety of methods, including remote detection or physical surveys (Grandison et al. 2000, Herder 2002).

Most peer-reviewed studies of the effects of wind-energy development on bats have been conducted in the eastern and central United States, and it is unclear whether the inferences are transferable to the Mojave Desert. It may be possible to adjust operations to reduce the probability of mortality, but

it remains unclear whether siting affects probability of mortality (Cryan 2011). Whether bats are attracted to wind turbines or randomly or coincidentally collide with turbines, and the proximate causes of mortality, have generated considerable debate. Because bats are cryptic, estimation of population sizes is challenging (Willis et al. 2010). By extension, it is difficult to determine whether bat mortality is proportional to population size. Limited observations of bat behavior at turbines suggest that bats congregate around moving and stationary turbines, feed on insects that swarm at turbines, and actively investigate both moving and stationary turbines, but the reasons for these behaviors are not well understood (Horn et al. 2008).

Research on bat occurrence at potential wind development sites may fill gaps in information about whether siting of wind-energy facilities affects mortality. High-priority variables include proximity to known roosts, the composition and abundance of bats in the area, and the timing of use; these data will facilitate assessment of whether the site may lie along a migration route or serve as foraging habitat. Potential methods for data collection include radar, thermal imaging, mist netting, roost surveys, or acoustic surveys (CBWG 2006).

Little is known about what may attract bats to wind turbines. Behavioral studies would contribute to determining whether bats are attracted to lights, sounds, motions, or structures of the wind facility or whether bats are attracted to wind developments for mating, gathering, or feeding. Alternative hypotheses, predictions, and potential methods for addressing behavioral hypotheses are outlined in Cryan and Barclay (2009). Sharing of data from pre- and post-construction monitoring and the use of standardized protocols for measuring effects of wind development on bats also would be useful for identifying responses of bats to wind development (Piorkowski et al. 2012).

Identifying species most likely to be affected by white-nose syndrome would be facilitated by information on the temperature and humidity of roosts, overwintering ecology, social behavior, and rates of contact among hibernating bats, and how these correspond to growth and transmission dynamics of *G. destructans* (Kunz et al. 2010). It may be useful to monitor the seasonal climate and microclimate of caves and mines within the Mojave Desert Network and relate these data to optimal conditions for growth, survival, and transmission of *G. destructans*. Internal-cave surface temperatures of large cave systems tend to correspond to the average annual temperature of the land surface, which may suggest which caves or mines may fall in this range of conditions, but cave climate is modified by size, location, configuration, airflow, and water infiltration (Perry 2012). At much greater spatial extents, information on connectivity among populations would facilitate estimation of pathways and rates of spread. For example, modeling across the northeastern United States indicated that the pattern of spread in the northeastern United States since 2006 was associated with winter temperatures and the number of caves within a given county that might serve as hibernacula (Maher et al. 2012).

Chapter 16. Mule deer in Grand Canyon–Parashant National Monument

16.A. Rationale for and information provided by the assessment

Mule deer (*Odocoileus hemionus*) are an economically important game species (deVos and McKinney 2007, BLM 2013) and are iconic in the western United States (Heffelfinger et al. 2006, Watkins et al. 2007, BLM 2013). The abundance of mule deer, including herds within Grand Canyon–Parashant (BLM 2013), on the nearby Kaibab Plateau in northern Arizona, and on the Three Bar Wildlife Area in central Arizona (Watkins et al. 2007), has decreased since a relative peak in the mid to late twentieth century (Clements and Young 1997). Grand Canyon–Parashant provides prime mule deer habitat in the Arizona Strip region (BLM 2013). Among the high priorities for assessment by staff at Grand Canyon–Parashant were the status and trends in abundance of mule deer and potential responses of that species to changes in vegetation structure.

16.B. Data and methods

Although mule deer are widespread in the United States, knowledge about resident mule deer within the Grand Canyon–Parashant region is relatively limited. Accordingly, our assessment of current patterns of abundance and space use of mule deer within the Grand Canyon–Parashant region, the species' vegetation requirements, and potential responses of those attributes of vegetation to regional climate change largely was based on a literature review. We focused on inferences specific to the Mt. Trumbull Wilderness Area and neighboring Kaibab Plateau, but also referenced inferences in ecologically similar regions of the southwestern and intermountain western United States.

16.C. Assessment results

16.C.1. Mule deer abundance

Mule deer abundance in the western United States, although highly variable (Mackie et al. 2003), generally has declined since its relative peak in the mid to late twentieth century in response to loss of habitat and reductions in habitat quality (Clements and Young 1997, Peek et al. 2002). Early in the twentieth century, mule deer were relatively scarce due to unrestricted hunting and conversion of mule deer habitat to agriculture (Clements and Young 1997, Mackie et al. 2003). However, hunting restrictions and disturbance, including killing of predators (mountain lions [*Puma concolor*], wolves [*Canis lupus*], and coyotes [*Canis latrans*]; Rasmussen 1941) and logging, increased the abundance of plants on which mule deer forage and reduced probabilities of predation and competition (Clements and Young 1997, Mackie et al. 2003). Estimates of mule deer abundance before the late 1970s are imprecise (Clements and Young 1997). Nevertheless, it is believed that mule deer abundance increased in the 1940s and 1950s until carrying capacities were reached, resulting in sharp declines in the 1960s and 1970s (e.g. on the Kaibab Plateau; Mackie et al. 2003). During the 1970s, deer densities at Three Bar Wildlife Area in central Arizona and on the Kaibab Plateau summer range in northern Arizona were estimated to be 2–5 individuals per square kilometer (6–13 per square mi) and 3–7 individuals per square kilometer (8–17 per square mi), respectively (Heffelfinger 2006). Thereafter, populations stabilized and abundances increased until the mid-1990s (Mackie et al. 2003). According to the BLM, the Arizona Game and Fish Department (AGFD) estimated that the

average number of mule deer in the Arizona Strip during the 1990s was 1,900–2,150. The current mule deer population in the Arizona Strip is less than 2,000 individuals, as compared with approximately 4,600 individuals in the late 1970s (BLM 2013).

Historic estimates of mule deer abundance for Grand Canyon–Parashant are limited. However, annual population modeling by the AGFD provides some insight into trends from 2003–2013. Grand Canyon–Parashant stretches across two AGFD Game Management Units, 13A (eastern Grand Canyon–Parashant, including the ponderosa pine, Great Basin, and Mojave-Great Basin transition ecological zones) and 13B (western Grand Canyon–Parashant, including all four ecological zones). In unit 13A, modeled pre-hunt estimates increased from a low of about 450 in 2002 to about 1,140 in 2013, averaging about 700 individuals for 2002–2013. Similarly, in unit 13B, modeled pre-hunt population estimates increased from a low of about 780 in 2003 to about 1,060 in 2013, averaging about 870 for 2002–2013. In 2013, post-hunt ratios of bucks to does were estimated at 0.43:1 and 0.44:1 for units 13A and 13B, respectively. Male to female mule deer ratios are typically 0.5:1 in natural populations, although heavily hunted populations may be more heavily skewed toward females (Mackie et al. 2003). Although the current three-year population trends are considered stable or increasing by AGFD, abundances are lower than estimated for the mid- to late-twentieth century.

16.C.2. Habitat, habitat use, and migration patterns

Mule deer habitat includes vegetation that provides cover and forage (see below), and often has a high ratio of edge between canopy and openings (Wallmo and Schoen 1981, Kie et al. 2002, Watkins et al. 2007). Mule deer habitat ranges from high-elevation mixed conifer forests to mid to lower elevation pinyon-juniper woodlands (Hungerford et al. 1981, Heffelfinger 2006). BLM-defined ecological zones within Grand Canyon–Parashant in which mule deer are considered a dominant species are ponderosa pine, Great Basin, Mojave-Great Basin transition, and interior chaparral (BLM and NPS 2007). Ponderosa pine typically occurs in relatively cool, moist areas. The Great Basin ecological zone is characterized by pinyon (*Pinus edulis*) and juniper (*Juniperus* spp.) woodlands on the mountains and plateaus and by grass- and shrub-dominated valleys. The Mojave Desert ecological zone is dominated by shrubs and forbs. The Mojave-Great Basin transition is characterized by plant species from the surrounding zones, including creosote (*Larrea tridentata*), cacti, sagebrush (*Artemisia* spp.), pinyon and juniper, Gambel oak (*Quercus gambelii*), and manzanita (*Arctostaphylos* spp.) (BLM and NPS 2007).

In most cases, male and female mule deer are spatially segregated except during the breeding season, and their vegetation associations vary among seasons (Ordway and Krausman 1986, Bellantoni and Krausman 1991). Male mule deer generally occupy areas that are more open and have more-diverse but lower-quality forage than the areas occupied by females (Ordway and Krausman 1986). Especially during pregnancy and fawning, females often move to higher elevations, where cover and ability to avoid predators generally is greater than at lower elevations (Bellantoni and Krausman 1991, Marshal et al. 2006).

Mule deer herds can include both migratory and non-migratory individuals (Nicholson et al. 1997, Kie et al. 2002). The size of home ranges is affected more by the absence of areas with little cover and forage (e.g., areas near human development) than by the presence of high quality cover and

forage (e.g., meadows and riparian areas; Nicholson et al. 1997). For example, home ranges are larger in grasslands than in montane areas dominated by conifers or pinyon and juniper (Bender et al. 2007).

Migration of individual mule deer generally occurs in response to local weather patterns (Monteith et al. 2011). During the late summer, both male and female mule deer shift to relatively mesic habitats, which often occur at relatively high elevations and have intermediate slopes and northern aspects (Ordway and Krausman 1986, Bellantoni and Krausman 1991). These movements likely are associated with the onset of summer rains and subsequent vegetation growth, and may occur earlier in drought years than in non-drought years (Bellantoni and Krausman 1991). Summer home ranges typically are larger than winter ranges. Individual mule deer remain on summer ranges as long as possible, although delayed autumn or winter migration may increase the probability of entrapment by winter storms, predation, or other sources of mortality (Monteith et al. 2011). However, individuals that migrate late may be able to capitalize on the relatively high productivity of forage during autumn (Monteith et al. 2011).

Autumn or winter migration in montane areas is associated with decreases in temperature and increases in precipitation or snow depth that reduce forage availability and increase energetic costs of movement and thermoregulation (Hungerford et al. 1981, Nicholson et al. 1997, Heffelfinger 2006, Monteith et al. 2011). In parts of central Arizona where most precipitation falls as winter rain, mule deer often occur at high elevations where forage remains relatively abundant (Fox and Krausman 1994).

During spring, migration is associated with increasing temperatures, decreasing precipitation, reductions in the quantity and quality of winter-range forage, and spring green-up (Nicholson et al. 1997, Monteith et al. 2011). Movement to higher elevations during this time may allow mule deer to forage on vegetation at an earlier phenological stage, with higher protein content and digestibility (Monteith et al. 2011). The nutritional requirements of gestating females increase during spring (Monteith et al. 2011).

16.C.3. Forage requirements

Mule deer are well adapted to locating and consuming vegetation that is readily digestible, includes high levels of protein and of nutrients such as phosphorus, and has a balanced ratio of phosphorus to calcium (Urness et al. 1971, deVos and McKinney 2007). Plant abundance, density, and forage quality can be highly correlated with mule deer distribution, abundance, and habitat quality (Leopold and Krausman 1991). Reproductive cycles of mule deer fluctuate as precipitation fluctuates throughout the year, although fawning generally occurs from August through October (Fox and Krausman 1994, Marshal et al. 2006). Fawn health is directly dependent on doe health during gestation and parturition, and low recruitment has been linked with low habitat quality and protein and nutrient deficiencies (Urness et al. 1971, Zwank 1976, Severson 1981). This relation can become stronger during drought (Severson 1981).

Although mule deer consume a large number of plant species, a small number of species comprise ≥ 1% of their diet (Urness et al. 1971, Severson 1981). Unlike other ruminant ungulates that have large

rumens adapted to digesting plants with a high concentration of cellulose (e.g., grass), such as domestic cattle (*Bos taurus*) or elk (*Cervus elaphus*), mule deer have small rumens adapted to digesting plants with high cell solubility or high lignin (Hanley 1982, deVos and McKinney 2007, Watkins et al. 2007). Woody plants such as sagebrush, cliff rose (*Cowania* spp.), and antelope bitterbrush (*Purshia tridentata*) can comprise as much as 77% of the diet of mule deer in Grand Canyon–Parashant (BLM 2013). Shrubs are considered to be the primary forage in all ecological zones (Severson 1981, Bender et al. 2007), although the dominance may be driven by abundance rather than preference (Severson 1981). Mule deer also eat cacti, mushrooms, and the fruits of shrubs and trees (Heffelfinger 2006). Although mule deer are dependent on shrubs for a major portion of their diet, especially during periods of low precipitation (Heffelfinger 2006), forbs provide supplements of protein and phosphorus (Urness et al. 1971, Heffelfinger et al. 2006). Grasses and forbs were estimated to be 17% and 6% of mule deer diets, respectively (BLM 2013), but the percentages increase when grasses and forbs are germinating or growing, such as during spring and early summer (Severson 1981, Heffelfinger 2006). Poor nutrition during the transition to winter ranges, whether from a lack of high-quality summer forage or increased energetic costs of reproduction, can affect reproduction and survival (Monteith et al. 2013).

Mule deer diets also can change along elevation gradients and among ecoregions. In the Kaibab Plateau and Arizona Strip region, summer diets often are dominated by aspen (*Populus tremuloides*), white fir (*Abies concolor*), mountain dandelion (*Agoseris aurantiaca*), non-native orchardgrass (*Dactylis glomerata*), milk vetch (*Astragalus* spp.), and Oregon grape (*Mahonia aquifolium*). Autumn diets may include cliff rose, penstemon (*Penstemon* spp.), buckwheat (*Eriogonum* spp.), and sagebrush, whereas winter diets may be dominated by cliff rose, sagebrush, Apache plume (*Fallugia paradoxa*), Utah juniper (*J. utahensis*), and pinyon pine (Heffelfinger 2006). In all seasons, aspen was considered to be the most important browse species (Heffelfinger 2006).

16.C.4. Cover requirements

Mule deer use vegetation and topography as cover for their bed sites, which relieve heat or cold stress and provide protection from predators (Wallmo and Schoen 1981, Parker and Gillingham 1990). Both thermal and escape cover can be provided by topography (e.g., overhanging cliffs or rugged terrain), the vegetation canopy, or ground cover (Mysterud and Østbye 1999). Escape cover includes both dense foliage that provides concealment and open areas with high visibility and many escape routes (Smith et al. 1986, Mysterud and Østbye 1999). For example, female mule deer in southern Arizona and Texas rarely forage in relatively flat, creosote-dominated areas except during the spring (Bellantoni and Krausman 1991, Leopold and Krausman 1991, Fox and Krausman 1994). Similarly, mule deer in arid shrublands in New Mexico rarely are present in areas dominated by grasses (Bender et al. 2007).

Adults often seek escape cover and bed sites in dense ground vegetation (Smith et al. 1986). For fawns, thermal cover is vegetation ≥ 1 m in height (Fox and Krausman 1994). In arid shrublands, these sites generally occur on steep slopes, in shrub-dominated flat areas, or in washes (Fox and Krausman 1994).

In montane areas, mule deer often bed under or near conifers or in aspen. These areas have a relatively cool microclimate and provide concealment (Smith et al. 1986). Mule deer typically bed along or near a deer trail because the energetic costs of escape along established trails are lower than through dense brush (Smith et al. 1986). Bed and cover sites typically have steeper slopes in xeric areas, where vegetation is relatively short, than in mesic areas where vegetation is taller (Smith et al. 1986).

16.C.5. Potential effects of climate change on habitat quality for mule deer

The BLM and NPS have classified Grand Canyon–Parashant into crucial and general summer ranges, crucial winter ranges, yearlong ranges, and areas of limited use by mule deer. Most of the crucial summer habitat is in the ponderosa pine ecological zone, whereas most of the crucial winter habitat is in the Mojave–Great Basin transition and Great Basin ecological zones. The majority of general summer and yearlong ranges is in the Mojave–Great Basin transition and Great Basin ecological zones. Smaller proportions of summer and yearlong habitat are in the Mojave Desert and interior chaparral ecological zones (BLM and NPS 2007).

Fire dynamics will affect whether crucial summer mule deer habitat persists in the ponderosa pine ecological zone. Surface fires in ponderosa pine typically have four-year cycles, and stand-replacing fires are rare (BLM and NPS 2007). In ponderosa pine forests, ladder fuels and the probability of crown fires have increased whereas understory productivity has decreased (Watkins et al. 2007). Within Grand Canyon–Parashant, thinning and prescribed fire may increase the quality of foraging habitat by opening forest canopy and encouraging production by woody and herbaceous understory species, but may reduce the number of day-bed sites (Horncastle et al. 2013). In the Mt. Trumbull area of the Grand Canyon–Parashant, treated ponderosa pine forests had 50–100% less day-bed microhabitat than untreated forest, but 30–60% more forage habitat (Germaine et al. 2004). Almost all beds in treated forest were under small stands of oak, which may not conceal beds as effectively as the conifers in untreated forests (Germaine et al. 2004). Areas treated only with thinning were used less intensively by mule deer than areas with thinning and fire treatments (Horncastle et al. 2013). This may reflect a reduction in cover and forage production immediately following thinning; a substantial increase in forage may not occur until after a prescribed fire (Horncastle et al. 2013).

In the Great Basin and Mojave–Great Basin transition zones, both increases and decreases in woody-shrub cover can affect mule deer habitat quality (Heffelfinger et al. 2006). An increase in the cover of woody perennials can increase canopy cover, reduce understory vegetation and forage (Heffelfinger et al. 2006), limit mule deer movement, and impede establishment of escape routes (Smith et al. 1986). Additionally, the palatability and nutritional quality of woody species may decrease as they mature, or woody species can make forage inaccessible (Heffelfinger et al. 2006). However, decreases in woody vegetation could reduce the amount of forage, thermal cover, or cover from predators (Heffelfinger et al. 2006).

Pinyon-juniper woodlands throughout Grand Canyon–Parashant are in various stages of succession and thus have different fuel loads, habitat quality for mule deer, and treatment responses (Miller and Tausch 2000). In heavy pinyon-juniper, accumulation of fine fuels is limited (Clements and Young 1997, Miller and Tausch 2000).

Grazing allotments on Grand Canyon–Parashant are monitored for compliance with the Arizona Standards for Rangeland Health and Guidelines for Grazing Administration (BLM 2011). Grazing is allowed across about 80% of Grand Canyon–Parashant and, as of 2007, nearly 38,000 Animal Unit Months (AUMs), each representing approximately 363 kg (800 pounds) of forage, were authorized (BLM and NPS 2007). Of the grazing allotments in Grand Canyon–Parashant that were reviewed, 38% met those standards and 62% were "progressing toward meeting standards" (BLM 2011).

As climate changes, mortality of drought-deciduous plants (e.g., *Ambrosia* spp.) on which mule deer feed may be greater than mortality of other shrubs (McAuliffe and Hamerlynck 2010). A relatively high proportion of the eastern and northeastern Mojave Desert, including the Arizona Strip, burned from 1980–2004 (Brooks and Matchett 2006). The proportion of fires in mid-elevation shrublands has been greater than in other ecoregions in the Mojave Desert and generally has increased as precipitation has increased (Brooks and Matchett 2006).

16.D. Data gaps and uncertainty

Relatively few data on mule deer and their habitat are available from within the boundaries of Grand Canyon–Parashant. The extent to which modeled trends in abundance for larger areas (ADFG Game Management Units) accurately reflect trends in the park is uncertain. Uncertainty about the status and trends of mule deer in Grand Canyon–Parashant would be reduced by design and implementation of a monitoring program for the species that focused not only on distribution and abundance but on survival, reproduction, and the amount and quality of habitat for the species. Tracking of males and females at different life stages (e.g., with GPS tags) would provide insight into movement and space-use patterns, especially in conjunction with field data or field-validated, remotely sensed data on vegetation composition and structure. Installation of weather stations across climatic gradients within the park would help to reduce uncertainty in projected responses of mule deer and their habitat to short-term variation in weather and to longer-term climate change.

Chapter 17. Discussion

In response to guidance from region-level and park-level managers, we assessed the condition of high-priority natural resources, and their known or potential responses to environmental stressors, within Death Valley National Park, Grand Canyon–Parashant National Monument, Joshua Tree National Park, Lake Mead National Recreation Area, Manzanar National Historic Site, and Mojave National Preserve. Whenever possible, we assessed resources at the regional level, recognizing that parks are not isolated but affect and are affected by processes outside their jurisdictional boundaries. Moreover, despite its name, the Mojave Desert Network spans five desert and semiarid ecoregions.

Our ability to assess the condition of many high-priority resources was constrained by data availability and by the lack of explicit management targets. NRCAs are not intended to be exhaustive or to suggest what management targets should be. Our assessment suggested that collection or updating of publicly available data and metadata from within the Mojave Desert Network parks on roads; air quality; distributions and abundances of non-native invasive plants across time and environmental gradients; and the distributions, movements, and vital rates of high-priority species would be especially informative. For cases in which few data existed, we conducted literature reviews and outlined the data and analyses that likely would be necessary to reduce uncertainty in the assessments.

The management community increasingly is expected to account for ongoing, anthropogenic climate change, which has or will affect the status and trend of myriad species and processes. Society may consider some ecological responses to climate change desirable, whereas others likely will be considered undesirable. The ability of any ecological element to respond to climate change is greater when land use does not create barriers to movement of individuals or resources. Therefore, land use is highly relevant to the current and future condition of national parks.

Framing management actions in terms of hypotheses, and then using field data to evaluate the hypotheses and inform changes in management, may help fill data gaps and increase the probability that management goals will be achieved. For example, monitoring the ecological effects of anthropogenic or natural environmental change may be most effective when informed by a theory of change—a set of hypotheses about the mechanisms by which a given change might be expected to have a given, measurable ecological effect, whether negative or positive (e.g., Stem et al. 2005). We hope that the NRCA has strengthened the basis for collaboration among the research and management communities and for resource stewarship by NPS and the public.

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Appendix 1. Sources of spatial data on land use.

Table A-1.1. Sources of spatial data on land use.

Data description	Year	Producer	URL	Date accessed
California Gap analysis	1998	University of California, Santa Barbara	www.biogeog.ucsb.edu/projects/gap/gap_home.html	15 July 2012
Census Blocks 2000	2009	US Census Bureau	www.census.gov/geo/www/tiger/	15 July 2012
Census Blocks 2009	2009	US Census Bureau	www.census.gov/geo/www/tiger/	15 July 2012
Communication towers	2010	Federal Communications Commission	wireless.fcc.gov/geographic/index.htm	1 July 2013
Digital elevation model	2001	US Geological Survey	data.geocomm.com/dem/	15 November 2013
Housing Density	2007	Hammer et al. 2004	silvis.forest.wisc.edu/Library/HousingDataDownload.asp?state=United%20States&abrev=US	15 July 2012
Integrated Climate and Land-Use Scenarios	2010	Environmental Protection Agency	www.epa.gov/ncea/global/iclus/	5 November 2013
Landfills	2003	US Geological Survey	sagemap.wr.usgs.gov/HumanFootprint.aspx	20 November 2013
Landfire Existing Vegetation Type	2008	Wildland Fire Leadership Council	www.landfire.gov/	15 July 2012
Mines	2005	US Geological Survey	minerals.usgs.gov/minerals/	15 July 2013
National Land Cover Database	2001 2006	US Geological Survey	www.mrlc.gov/finddata.php	15 July 2012
Normalized Difference Vegetation Index (250 m resolution)	2000–2010	National Aeronautics and Space Administration	modis.gsfc.nasa.gov/data/dataprod/dataproducts.php?MOD_NUMBER=13	15 July 2012

Table A-1.1 (continued). Sources of spatial data on land use.

Data description	Year	Producer	URL	Date accessed
Pipelines	–	US Census Bureau	www.census.gov/geo/www/tiger/	–
Population growth	2010	Environmental Protection Agency	www.epa.gov/ncea/global/iclus/	5 November 2013
Power lines	2009	US Census Bureau	www.census.gov/geo/www/tiger/	15 July 2012
Power lines	2003	Interior Columbia Basin Ecosystem Management Project (ICEBMP)	sagemap.wr.usgs.gov/HumanFootprint.aspx	15 July 2012
Railroads	2009	US Census Bureau	www.census.gov/geo/www/tiger/	15 July 2012
Rest areas	2003	US Geological Survey	sagemap.wr.usgs.gov/HumanFootprint.aspx	10 November 2013
Roads and trails	2009	US Census Bureau	www.census.gov/geo/www/tiger/	15 July 2012
Southwest Regional Gap Analysis project	2009	US Geological Survey	swregap.nmsu.edu/	15 July 2012
Soil depth	2003	US Geological Survey	sagemap.wr.usgs.gov/HumanFootprint.aspx	10 November 2013
Solar and wind energy installations	2009	Bureau of Land Management	www.blm.gov/ca/st/en/prog/energy/solar.html	15 July 2012
Housing density	2009	US Census Bureau	www.census.gov/geo/www/tiger/; www2.census.gov/census_2000/datasets/Summary_File_1/	15 July 2012

Appendix 2. Home ranges of 63 vertebrate species.

For many species, the maximum (max) and minimum (min) are based on the average of home ranges for males and females. BT, breeding territory; FA, feeding area; H, home range; SH, summer home range; WH, winter home range; YH, year-round home range.

Table A-2.1. Home ranges of 63 vertebrate species.

Class	Species	Scientific Name	Life stage	Min (ha)	Max (ha)	Mean (ha)	Citation
Reptiles	Desert tortoise	*Gopherus agassizii*	H	7.2	29.8	18.5	O'Connor et al. 1994
	Collared lizard	*Crotaphytus collaris*	H	0.01	0.16	0.09	Eifler and Fogarty 2006
	Long-nosed leopard lizard	*Gambelia wislizenii*	H	2.3	3.2	2.8	Rose 2004
	Common chukwalla	*Sauromalus ater*	H	0.17	0.57	0.4	Johnson 1965
	Desert iguana	*Dipsosaurus dorsalis*	H	0.15	0.16	0.2	Kreakorian 1976
	Zebra-tailed lizard	*Callisaurus draconoides*	H	0.35	0.65	0.5	Tanner and Krogh 1975
	Texas horned lizard	*Phrynosoma cornutum*	H	0.1	4.7	2.4	Wone and Beauchamp 2003
	Side-blotched lizard	*Uta stansburiana*	H	0.1	0.03	0.02	Castañeda et al. 2007
	Gopher snake	*Pituophis catenifer* spp.	H	0.10	0.29	0.2	Rodriguez-Robles 2003
Birds	Turkey Vulture	*Cathartes aura*	BT	5218	122750	32296	Coleman and Fraser 1989
	Cooper's Hawk	*Accipiter cooperii*	BT	13	1518	766	Boal and Mannon 2000
	Red-tailed Hawk	*Buteo jamaicensis*	H	145	480	313	Bloom et al. 1993
	Golden Eagle	*Aquila chrysaetos*	BT	190	8330	2280	Marzluff et al. 1997b,c
	American Kestrel	*Falco sparverius*	BT	66	242	154	Bird and Palmer 1988
	Prairie Falcon	*Falco mexicanus*	BT	5900	31500	9670	Marzluff et al. 1997a, Steenhof 1998
	Gambel's Quail	*Callipepla gambelii*	BT	0.5	6.7	3.6	Brown et al. 1998
	Mourning Dove	*Zenaida macroura*	BT	100	800	450	Howe and Flake 1988
	Greater Roadrunner	*Geococcyx californianus*	BT	33	454	244	Kelley et al. 2011
	Barn Owl	*Tyto alba*	BT	294	750	522	Martin et al. 2010

Table A-2.1 (continued). Home ranges of 63 vertebrate species.

Class	Species	Scientific Name	Life stage	Min (ha)	Max (ha)	Mean (ha)	Citation
Birds (continued)	Long-eared Owl	*Asio otus*	H	97	374	236	Henrioux 2000
	Great Horned Owl	*Bubo virginianus*	H	115	699	407	Bennett and Bloom 2005
	Burrowing Owl	*Athene cunicularia*	H	14	481	248	Haug et al.1993
	Western Screech-Owl	*Megascops kennicottii*	H	65	65	65	Davis and Weir 2010
	Common Poorwill	*Phalaenoptilus nuttallii*	BT	0.9	4.0	2.5	Stiles 1980
	Anna's Hummingbird	*Calypte anna*	BT	0.9	1.3	1.1	Clark and Russell 2012
	Loggerhead Shrike	*Lanius ludovicianus*	BT	2.7	25	14	Collister 1994, Woods 1994, Yosef 1996
	Western Scrub-jay	*Aphelocoma californica*	BT	0.7	3.2	2.0	Carmen 1988
	Common Raven	*Corvus corax*	BT	102	512	307	Dorn 1972, Linz et al. 1992, Steenhof et al. 1993
	Verdin	*Auriparus flaviceps*	H	3.6	12.5	8.0	Webster 1999
	Bushtit	*Psaltriparus minimus*	BT	1.0	2.0	1.5	Sloane 1992
	Bewick's Wren	*Thyromanes bewickii*	BT	0.2	4.8	2.5	Stafford 1983, Kroodsma 1973
	Cactus Wren	*Campylorhynchus brunneicapillus*	BT	1.0	6.7	3.8	Anderson and Anderson 1973
	Rock Wren	*Salpinctes obsoletus*	BT	0.9	3.3	2.1	Merola 1995
	Canyon Wren	*Catherpes mexicanus*	BT	0.5	1.6	1.0	Mirsky 1976
	Black-tailed Gnatcatcher	*Polioptila melanura*	BT	0.8	2.7	1.8	Hensley 1954
	Blue-gray Gnatcatcher	*Polioptila caerulea*	BT	0.5	3.0	1.8	Root 1969, Ellison 1991
	Northern Mockingbird	*Mimus polyglottos*	BT	0.4	2.5	1.5	Howard 1974
	California Thrasher	*Toxostoma redivivum*	BT	1.6	2.5	2.1	Cody 1998
	Le Conte's Thrasher	*Toxostoma lecontei*	BT	3.5	18	10.8	Sheppard 1996
	Phainopepla	*Phainopepla nitens*	BT	0.03	0.4	0.2	Walsberg 1977
	California Towhee	*Melozone crissalis*	BT	0.4	6.2	3.3	Purcell and Verner 1998
	Brewer's Blackbird	*Euphagus cyanocephalus*	H	800	31400	16100	Williams 1952, Horn 1968
	Great-tailed Grackle	*Quiscalus mexicanus*	BT	–	317	317	Johnson and Peer 2001

Table A-2.1 (continued). Home ranges of 63 vertebrate species.

Class	Species	Scientific Name	Life stage	Min (ha)	Max (ha)	Mean (ha)	Citation
Mammals	Merriam's shrew	*Sorex merriami*	FA	0.03	0.5	0.3	Reid 1997
	Townsend's big-eared bat	*Corynorhinus townsendii*	FA	530	3220	1875	Dobkin et al. 1995
	Spotted bat	*Euderma maculatum*	H	11300	465600	238450	Wai-Ping and Fenton 1989
	Black-tailed jackrabbit	*Lepus californicus*	H	100	310	205	Smith 1990
	White-tailed jackrabbit	*Lepus townsendii*	H	4.0	777	391	Clark and Stromberg 1987
	Pygmy rabbit	*Brachylagus idahoensis*	SH	2.7	20.2	11.5	Gahr 1993
	White-tailed prairie dog	*Cynomys leucurus*	H	5.9	6.9	6.4	Clark et al. 1971
	Round-tailed ground squirrel	*Xerospermophilus tereticaudus*	H	–	0.3	0.3	Drabek 1973
	White-tailed antelope squirrel	*Ammospermophilus leucurus*	H	1.40	9.40	6.70	Bradley and Deacon 1965
	Great Basin pocket mouse	*Perognathus parvus*	H	0.1	0.3	0.2	Schreiber 1978
	Porcupine	*Erethizon dorsatum*	H	5.6	20.7	13.1	Sweitzer 2003
	Gray fox	*Urocyon cinereoargenteus*	H	605	738	671	Harrison 2002
	Coyote	*Canis latrans*	H	280	18530	9405	Windberg and Knowlton 1988
	Raccoon	*Procyon lotor*	H	110	480	295	Gehrt and Fritzell 1997
	Ringtail	*Bassariscus astutus*	H	258	343	301	Harrison 2012
	Striped skunk	*Mephitis mephitis*	H	19	268	143	Bixler and Gittleman 2000
	American badger	*Taxidea taxus*	H	200	27375	13788	Quinn 2008
	Bobcat	*Lynx rufus*	H	192	10530	5361	Elizalde-Arellano et al. 2012
	Pronghorn	*Antilocapra americana*	SH	260	2033	1147	O'Gara 1978, Hoskinson and Tester 1980
	Pronghorn	*Antilocapra americana*	WH	650	1140	895	Bayless 1969
	Pronghorn	*Antilocapra americana*	YH	165	2300	1233	Clark and Stromberg 1987
	Bighorn sheep	*Ovis canadensis*	H	1170	3730	2450	Shackleton 1985

Appendix 3. Remote detection of non-native invasive plants

This appendix is a summary of Bradley, B. 2014. Remote detection of invasive plants: a review of spectral, textural and phenological approaches. Biological Invasions 16:1411–1425.

Monitoring can inform efforts to prevent new invasions and to eradicate recent invasions of non-native species (Moody and Mack 1988). Eradication is practical only during the early stages of invasion, when there are relatively few individuals in a relatively small area (Rejmanek et al. 2005). Therefore, monitoring of invasive plants is most likely to be useful to NPS when implemented over extensive areas. However, field surveys across hundreds to thousands of square kilometers are not feasible. Accordingly, NPS asked us to address whether remote sensing could be used to identify the current locations of invasive plants, to assess the probability of future invasion, or to document temporal trends in the distribution or abundance of invasive plants.

Before an assessment of invasion probability can be conducted across large areas, information about current associations between a given invasive plant and environmental attributes (e.g., natural disturbances, land use, soil type and structure, or fragmentation of land cover) is needed (see Vilà and Ibáñez 2011). In some cases, such information can be derived from field data, but surveys must be designed to capture gradients of the environmental attributes of interest. Herbarium and ad hoc data often are not representative of regional environmental gradients and tend to be biased toward relatively accessible areas (e.g., adjacent to roads). Herbarium and ad hoc data also tend to overrepresent areas in which abundance of the invasive plant is low rather than high (e.g., because herbarium collections typically focus on relatively rare species rather than relatively common species; Marvin et al. 2009, Bradley 2013). Accordingly, probability of invasion across large areas may be difficult to infer on the basis of ad hoc data. Contemporary data that are collected systematically across environmental gradients, including via methods designed to train or validate remotely sensed data, are more likely to facilitate achievement of management objectives.

Although mapping invasive plants on the basis of spectral features is the most common method for remote detection, it is increasingly feasible to use texture or phenology to identify invaded locations (see Table 3.1). Here, we identify circumstances in which remote detection is feasible and cost effective. We also provide guidance on collection of field data that will be the most complementary to remotely sensed images.

Table A-3.1. Common terms in remote sensing.

Term	Definition
Absorption	Inverse of reflectance, calculated as 1–reflectance
Hyperspectral	Imagery of the same region that contains many spectral bands, typically hundreds, in visible and near-infrared wavelengths
Multispectral	Imagery of the same region that contains multiple spectral bands, typically 4–10, in visible and near-infrared wavelengths
Near-infrared (NIR)	Energy at slightly longer wavelengths than visible, typically referring to wavelengths 0.7–1 μm, in which photosynthetic vegetation has high reflectance
Phenology	The timing of seasonal events in the life cycles of plants and animals. Time series of remotely sensed imagery, typically derived vegetation indexes, can identify phenological stages such as the start and end of photosynthetic activity.
Pixel	The smallest unit of measure of a satellite or aerial image, typically expressed in terms of the length of one side of a square (e.g., a 30 m pixel is a 900 m^2 square)
Reflectance	A unitless ratio of outgoing energy to incoming energy at a given wavelength. All sensors discussed in this section use reflectance spectroscopy.
Short wavelength infrared (SWIR)	Energy at slightly longer wavelengths than near infrared, typically 1–2.5 μm
Spectral band	A range of wavelengths within which reflectance data are acquired by the sensor (e.g., the near infrared band for Landsat 5 is 0.76–0.90 μm)
Spectrum	Reflectance of a material across a series of wavelengths
Texture	Variation in reflectance between neighboring pixels
Vegetation index	A ratio of near-infrared to visible red that highlights photosynthetic vegetation. The most common is the Normalized Difference Vegetation Index (NDVI): (NIR – VIS)/(NIR + VIS)
Visible (VIS)	Wavelengths of visible (light) energy: 0.4–0.7 μm

A. Sources of remotely sensed images

Remotely sensed images are acquired by both public and private satellites and by diverse airborne sources of aerial photographs and imagery (Table 3.2). There are trade-offs among the spatial extent, spatial resolution, spectral resolution, and frequency of acquisition of remotely sensed images. Images that cover relatively large areas allow for spatially extensive mapping of invasive plants and ultimately provide more data to inform models of invasion probability. Images with relatively fine spatial resolution (e.g., images from the National Agricultural Imagery Program [NAIP], which have a spatial resolution of 1 m) increase the probability of detecting individual species and early invasions. High spectral resolution increases the number of options for differentiating plant pigments and chemistry. Infrared wavelengths have proven especially useful for identification of invasive plants. Frequent data acquisition facilitates identification of vegetation phenology and detection of vegetation change. Frequency of acquisition tends to be correlated positively with spatial extent, but negatively with spatial and spectral resolution.

Table A-3.2. Types and sources of remotely sensed data for the United States.

Type	Imagery	Swath width (km)	Spatial resolution[1]	Temporal availability	Acquisition frequency	Data sources	Cost[3]
Multispectral	Advanced very high resolution radiometer (AVHRR)	2600	1.1 km	1978–present	7+ day composite	Earth Explorer	free
	Moderate resolution imaging spectroradiometer (MODIS)	2330	250–1000 m	2000–present	7+ day composite	Earth Explorer, NASA Reverb	free
	Landsat Thematic Mapper (TM), Enhanced Thematic Mapper + (ETM+)	185	30 m	1982–2011	16 days	Earth Explorer	free
	Landsat 8	185	30 m	2013–present	16 days	USGS Earth Resources Observation and Science (EROS) Data Center	free
	Advanced spaceborne thermal emission and reflection radiometer (ASTER)	60	15–30 m	2000–present	tasked	Earth Explorer	free
	Satellite pour l'observation de la terre (SPOT)	60	2.5–20 m	1986–present	tasked	Earth Explorer (1986–1998), USGS EROS Data Center	free
	IKONOS	11	1.8–4 m	1999–present	tasked	various vendors	$$
	Quickbird	17	2.5 m	2001–present	tasked	various vendors	$$
	Worldview-2	16	1.8 m	2009–present	tasked	various vendors	$$
	GeoEye-1,2	15	1.65 m	2008–present	tasked	various vendors	$$
	Pleiades-1,2	20	0.5 m	2011–present	tasked	various vendors	$$
Hyperspectral	Hyperion	7.7	30 m	2000–present	tasked	Earth Explorer, Data Acquisition Requests (new tasks)	Free
	Airborne visible/infrared imaging spectrometer (AVIRIS)[2]	2–11	4 – 20 m	1992–present	tasked	AVIRIS Flight Data Locator (2006–2011 only)	Free

[1] The length of one side of a square pixel

[2] Airborne rather than satellite platform

[3] $: Aerial photographs typically cost $30 per image to scan at high resolution. $$: Most commercial satellite images cost on the order of $10–20 per km^2, sometimes with discounts for academic uses.

Table A-3.2 (continued). Types and sources of remotely sensed data for the United States.

Type	Imagery	Swath width (km)	Spatial resolution[1]	Temporal availability	Acquisition frequency	Data sources	Cost[3]
Aerial photographs[2]	National Aerial Photography Program (NAPP)	9	1:40,000	1987–2007	years	Earth Explorer (so-called medium resolution scanned images are free)	$
	National High Altitude Photography (NHAP)	–	1:80,000	1980–1989	years	Earth Explorer (medium resolution scanned images are free)	$
	Aerial photographs	varies	various	various	years	Earth Explorer (single frames from historic imagery)	$
	National Agriculture Imagery Program (NAIP)	mosaic	1 m	2001–present	years	datagateway.nrcs.usda.gov	$
	USGS digitial orthophoto quadrangle (DOQ)	mosaic	1 m	location dependent	years	Earth Explorer	free

[1] The length of one side of a square pixel

[2] Airborne rather than satellite platform

[3] $: Aerial photographs typically cost $30 per image to scan at high resolution. $$: Most commercial satellite images cost on the order of $10–20 per km^2, sometimes with discounts for academic uses.

Images with relatively high spatial resolution includes aerial photographs, images from the airborne visible/infrared imaging spectrometer (AVIRIS), and images from a number of satellite sensors recently launched by private companies such as DigitalGlobe. Spectral differentiation implies that the target invasive species has one or more light absorption or reflectance features in either visible or near-infrared wavelengths that are not present in native vegetation. Aerial photographs may be grayscale or visible color, and sometimes have an additional near-infrared band. Recent high-resolution satellite images (e.g., IKONOS and Worldview-2) typically have four spectral bands, three in the visible and one in the near-infrared. The cost of satellite images from private vendors can be prohibitive for management agencies and universitie: on the order of $10–20 per square kilometer, with minimum purchase amounts in the hundreds to thousands of dollars.

The most widespread and freely available satellite images are acquired by multispectral sensors, typically with 4–10 spectral bands in the visible and near infrared, at moderate spatial resolutions. Examples include Landsat (30 m resolution, 185 km swath width), Aster (15–30 m resolution, 60 km swath width), and Satellite pour l'observation de la terre (SPOT; 10 m resolution, 60 km swath width). Landsat has a regular return interval (acquisition frequency) of 16 days, and the near-continuous 25-year record of Landsat imagery for the United States facilitates both change detection and time-series measurements of phenology. All Landsat data are freely available.

Images from the Advanced Very High Resolution Radiometer (AVHRR) and the Moderate Resolution Imaging Spectroradiometer (MODIS) have spatial resolutions from 250–1000 m and wide swaths (Table 3.2). Both AVHRR and MODIS take daily images of Earth's entire surface. These daily images are used to create weekly, 10 day, or biweekly composites that minimize cloud cover. The AVHRR archive extends back to the 1980s, whereas MODIS data were first acquired in 2000. Images from AVHRR and MODIS are free.

For an invasive plant species to be detectable with remote sensing, it must have a spectral, textural, or phenological signal that is different from that of the surrounding vegetation. The invasive species also must have high percent cover relative to the spatial resolution of the sensor. For example, the species must be abundant within an area of 900 m^2 to be detected by Landsat, whereas single trees might be detectable in the 4 m^2 pixels of aerial photographs or high-resolution (1–4 m^2 pixel size) commercial images. Invasive plants tend to establish at high densities and often form near-monotypic stands.

B. Differentiation of invasive plants

B.1. Spectral differentiation

Most of the successful applications of remote sensing to detection of invasive plants have relied on spectral differences between the invasive and the surrounding native vegetation (see reviews by Lass et al. 2005, Huang and Asner 2009, He et al. 2011). Spectral differences are easiest to identify with hyperspectral images (Figure 3.1a, b), which have hundreds of spectral bands. Hyperspectral analyses typically target either specific wavelengths or ratios of two wavelengths that highlight differences in plant pigmentation (distinctive coloration of leaves or flowers during one or more life stages), water content, or leaf chemistry (e.g., Underwood et al. 2003, Asner et al. 2008). Spectral

differentiation of invasive plants requires that the invasive species be at the top of the canopy. Accordingly, spectral detection of invasive plants is most feasible in grasslands, shrublands, and aquatic systems without overstory vegetation.

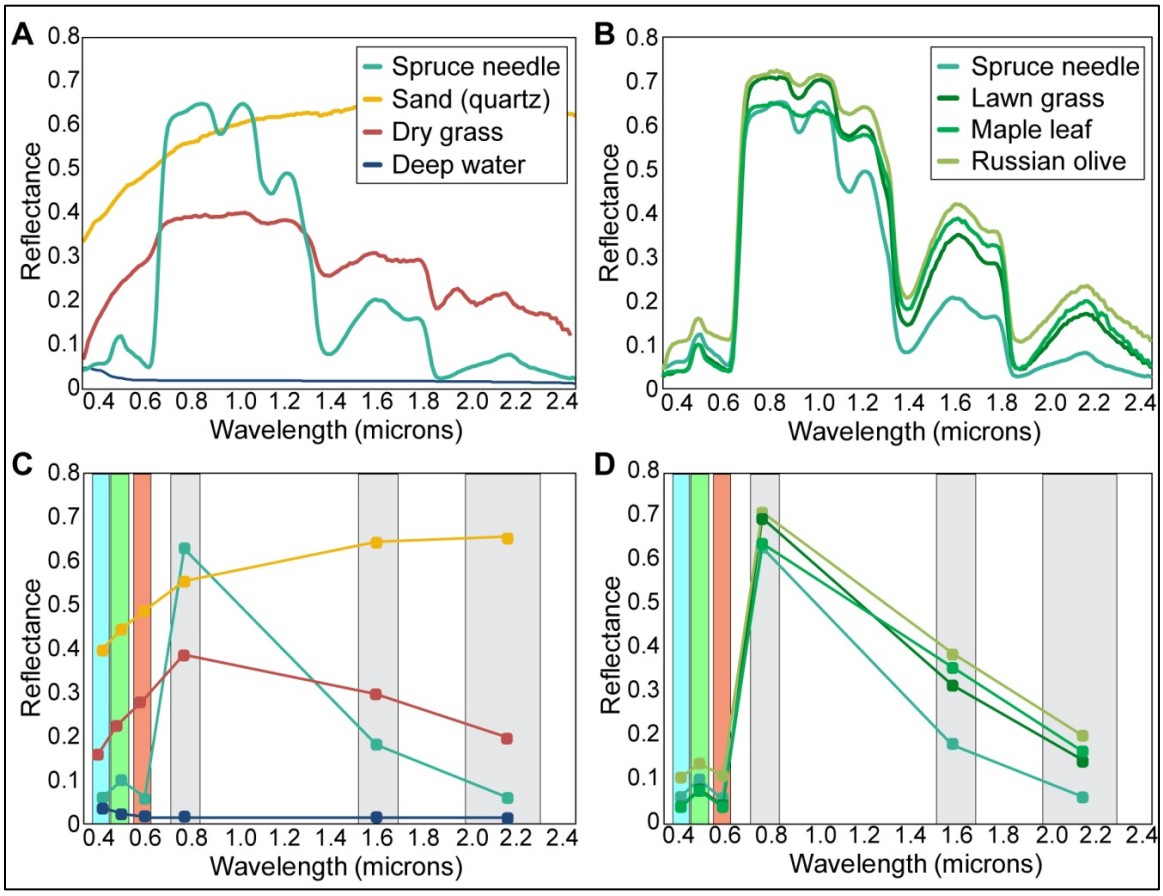


Figure A-3.1. Examples of pure (unmixed) spectral signatures from hyperspectral and multispectral sensors. Vertical bars indicate the approximate wavelengths measured by Landsat and Moderate Resolution Imaging Spectroradiometer (MODIS) sensors, with the blue, green and red bars representing the wavelengths of visible light. (A) These four spectrally distinct materials are easily separated on the basis of hyperspectral data. (B) These four photosynthetic species have similar spectra, but may be possible to separate on the basis of hyperspectral data. (C) The materials in (A) remain easy to identify on the basis of multispectral data. (D) The four photosynthetic species in (B) appear nearly identical in multispectral data.

Use of multispectral images to identify invasive plants is most likely to be successful where the invasive plant changes the dominant land cover. Floating aquatic invasive plants have been identified remotely on the basis of the strong spectral distinction between photosynthetic vegetation and water. One such species, water hyacinth [*Eichhornia crassipes* (Albright et al. 2004, Everitt and Yang 2007, Hestir et al. 2008, Cavalli et al. 2009)], has established in Mexico (Gutiérrez et al. 1996) and Texas (Everitt & Yang 2007) and is a potential invasive in reservoirs in the Mojave Desert.

Differences in plant pigmentation may be more apparent in flowers than in leaves (Andrew and Ustin 2006, Miao et al. 2006, Mirik et al. 2013). For example, leafy spurge (*Euphorbia esula*), which

invades grasslands and shrublands throughout the western United States, has characteristic yellow flowers that bloom in the early summer. This distinct pigmentation enables remote detection on the basis of either hyperspectral data (Williams and Hunt 2002, Glenn et al. 2005) or color aerial photographs (Everitt et al. 1995). However, even species with distinct flowers can be misclassified if native species are flowering at the same time. For instance, in relatively diverse ecosystems, the prolific white flowers of perennial pepperweed (*Lepidium latifolium*) can be confused with the white flowers of other plants (Andrew and Ustin 2008).

B.2. Textural differentiation

Identification of invasive plants on the basis of texture relies on a difference in the shape, functional type, or growth pattern of the invasive species relative to co-occurring native or established non-native species. Textures most often are differentiated with aerial photographs or high-resolution multispectral images, and images generally are classified on the basis of visual interpretation. Textural differentiation is most feasible when the invasive species grows at high densities or in monocultures, which makes the invasive appear homogeneous relative to more-diverse native communities.

An alternative to visual interpretation of textural differences is object-based classification, a computer-based method that identifies patches with similar color (Blaschke 2010). Invasive trees are the easiest targets for object-based classification because individual trees are larger than the pixels in the image. Object-based classification can be particularly effective for identifying expansion of trees (invasive or native) into surrounding shrubland or grassland (Pearlstine et al. 2005, Xie et al. 2008, Gil et al. 2013; Figure 3.2). For example, object-based classification of native pinyon (*Pinus monophylla*), juniper (*Juniperus osteosperma*), and ponderosa pine (*Pinus ponderosa*) was effective for measuring expansion of these trees into surrounding shrubland on the basis of historical aerial photographs (Mast et al. 1997, Weisberg et al. 2007).

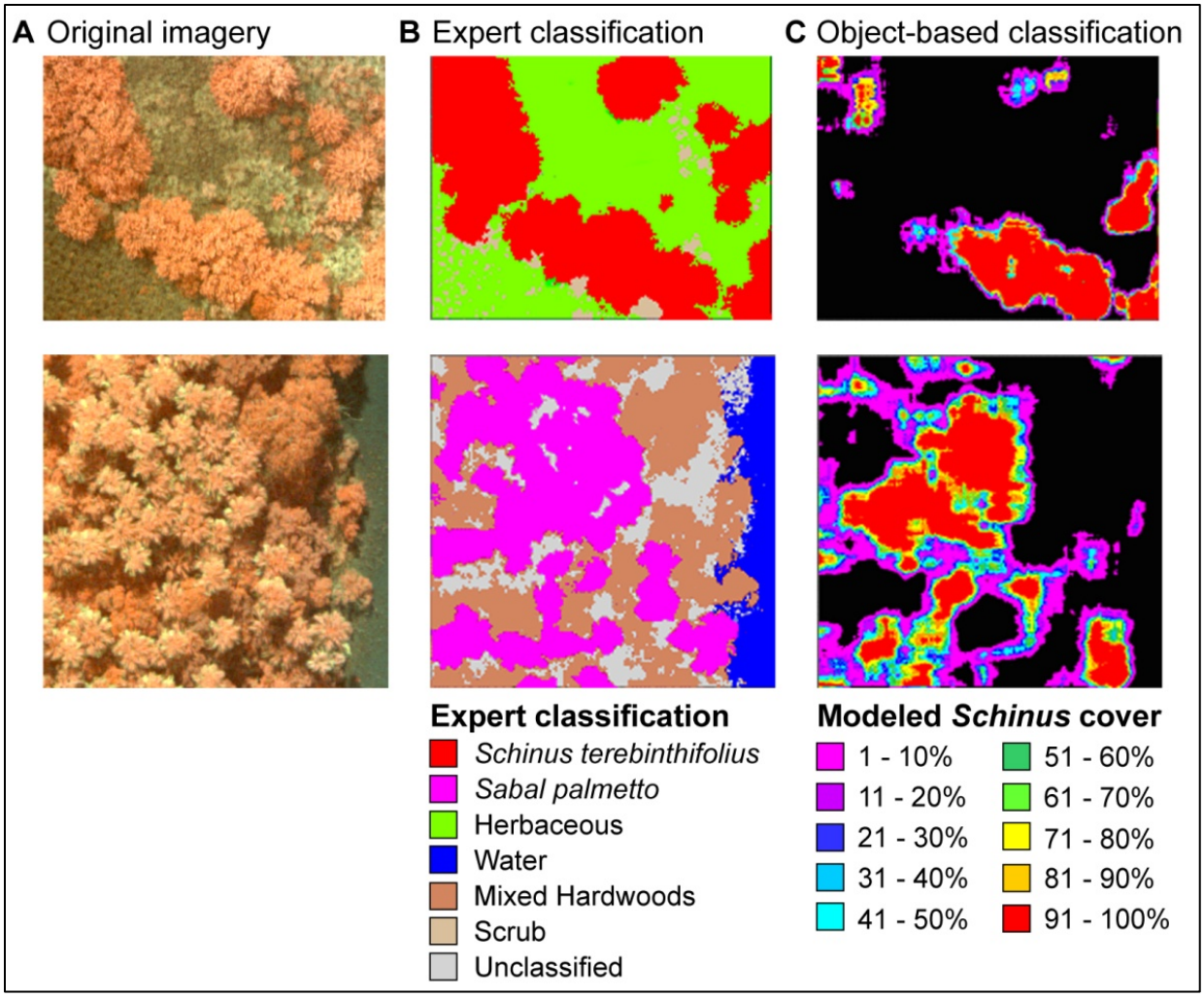


Figure A-3.2. Textural analysis of non-native *Schinus terebinthifolius* in south Florida. (A) The original false color imagery; red indicates high levels of photosynthesis. The top image is of *S. terebinthifolius* and the bottom image is of native *S. palmetto*. (B) Maps classified by experts on the basis of visual interpretation coupled with substantial field data have high classification accuracy, but are time consuming to create. (C) An object-based classification algorithm based on texture effectively differentiated cover of invasive trees from native herbaceous vegetation, but misclassified *S. palmetto* as *S. terebinthifolius*. Figure adapted from Pearlstine et al. 2005. Reprinted with permission from the American Society for Photogrammetry and Remote Sensing, Bethesda, Maryland, www.asprs.org.

Texture-based mapping of invasive plants generally requires that the pixel size of the imagery be substantially smaller than the invasive plant or aggregation of plants. Texture-based mapping most often is used to identify individual trees with canopies larger than 1 m^2. Texture-based mapping of invasive plants also requires that the shape of individual plants or multiple plants be different from that of native species. If an invasive plant species grows in monocultures, then textural analysis is likely to identify areas in which the density of the invasive already is high rather than the earliest stages of invasion.

B.3. Phenologic differentiation

Identifying an invasive species on the basis of phenology implies that the species has a seasonal or annual growth pattern different than that of native species. Frequent images are necessary to assemble time series of data on phenology (typically vegetation indexes). This requirement precludes the use of most aerial photographs, hyperspectral data, and multispectral data with high spatial resolution (but see Somers and Asner 2013). The most viable options for high-frequency image acquisition are Landsat, MODIS, and AVHRR (Table 3.2), which have spatial resolutions from 30 m – 1 km.

Differences in phenology often are associated with functional differences. For example, an invasive plant might be phenologically distinct if it is an evergreen species invading a native deciduous community, if it becomes green earlier or stays green longer than native species, or if its annual signal, such as its annual variability, is different from that of natives. Even invasive plants in the understory can be detected if their phenology is distinct from that of native overstory species. For example, bush honeysuckle (*Lonicera maackii*) invasions in forest understory in the midwestern United States can be identified by targeting images taken in early spring and late autumn – time periods when honeysuckle is green but the tree canopy is not (Figure 3.3) (Resasco et al. 2007, Wilfong et al. 2009, Shouse et al. 2013).

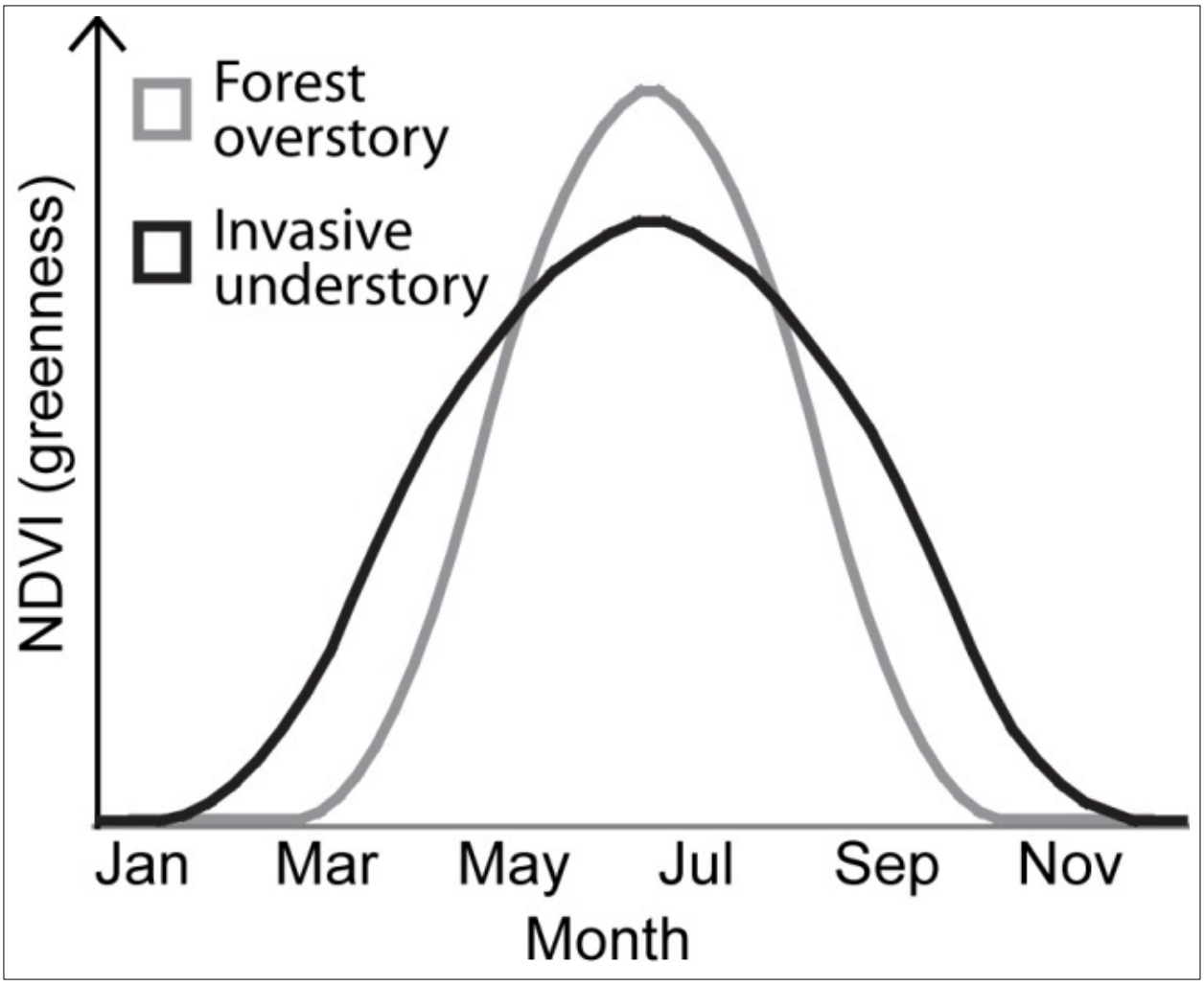


Figure A-3.3. The longer growing season of invasive non-native understory species in forests provides opportunities for remote detection in the early spring and late autumn, when tree leaves are not photosynthetically active.

Because cheatgrass becomes green earlier in spring than native plants, it can be identified on the basis of differences in photosynthetic greenness in Landsat images taken during early spring and early summer (Peterson 2005, Singh and Glenn 2009, Clinton et al. 2010). Cover and biomass of cheatgrass also are more responsive to changes in winter and spring precipitation than cover and biomass of many native species (i.e., cheatgrass has higher annual variability; Figure 3.4), and the range of cheatgrass is extensive enough that the species can be detected with both Landsat and AVHRR (Bradley and Mustard 2005). Another invasive grass, Lehmann lovegrass (*Eragrostis lehmanniana*), was detected in southern Arizona on the basis of annual variability in greenness as recorded by MODIS (Huang & Geiger 2008). As we illustrate in the NRCA, annual phenology also has been used to map invasive annual plants such as red brome (*Bromus rubens*) and Sahara mustard (*Brassica tournefortii*) (Wallace and Thomas 2008).

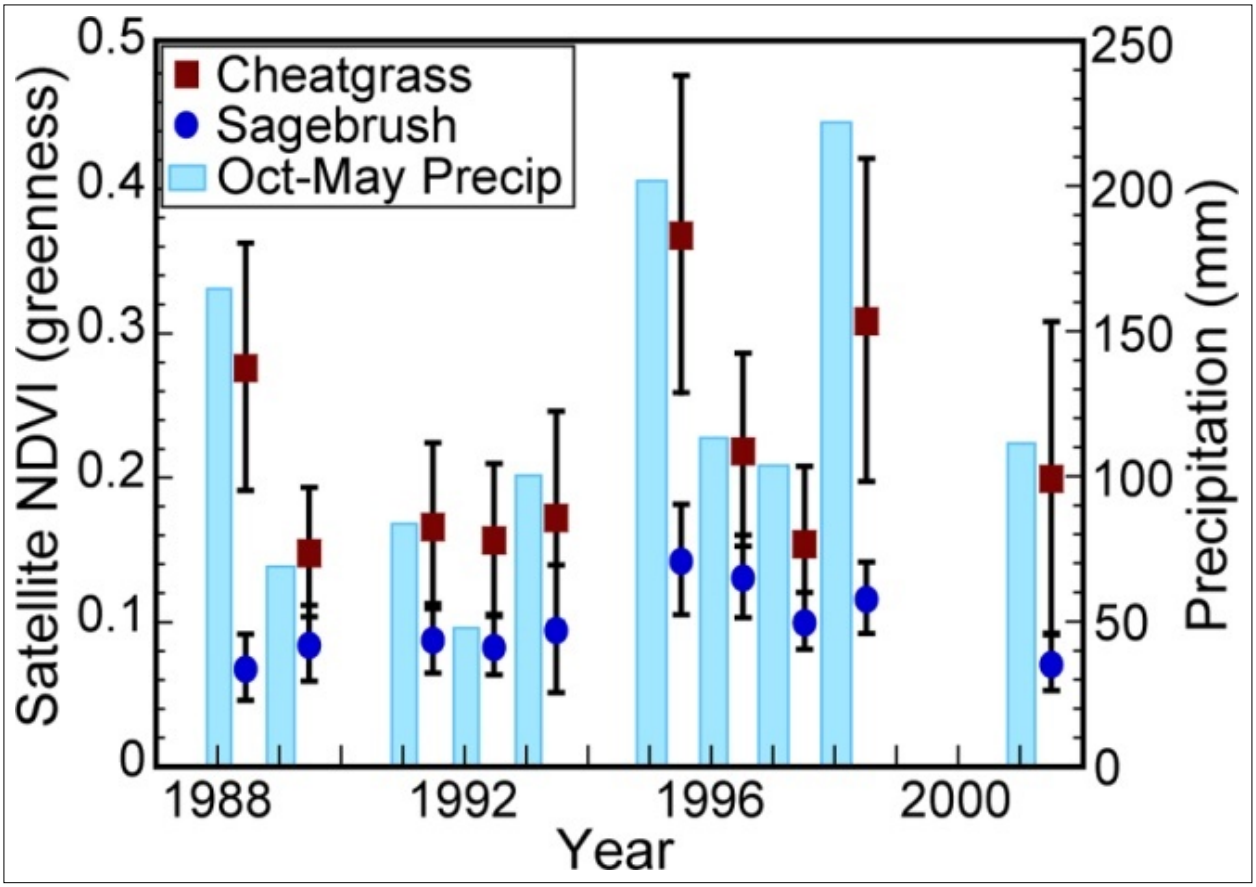


Figure A-3.4. Following high-precipitation events in central Nevada in 1988, 1995, and 1998 (blue bars, measured from a local rain gauge), vegetated greenness (measured by the normalized difference vegetation index, NDVI) of areas dominated by cheatgrass (*Bromus tectorum*) was substantially higher than that of areas dominated by sagebrush (*Artemisia tridentata*). This pattern of annual variability makes it possible to identify cheatgrass-dominated areas across large regions.

Phenology-based mapping requires that the cover or density of the invasive plant be high in both an absolute sense and relative to pixel size. Accordingly, phenology-based mapping generally cannot be used for early detection of invasive plants. However, maps of locations with high densities of invasive plants can be used to identify associated environmental attributes (e.g., disturbance, soils, topography) and to model future probability of invasion (e.g., Bradley and Mustard 2006).

C. Linking remotely sensed images and field data

Field data can be invaluable for training and validation of remotely sensed classifications of invasive plants. However, the field data must be comparable to what is recorded in remotely sensed images. Because remotely sensed images are pixel-based, field surveys that collect percent-cover data within a defined square are most useful for comparison to images. The size of the square depends on the resolution of the analysis. If the analysis will be based on aerial photographs or images with high spatial resolution, then field sampling should be conducted within 1–4 m^2 squares, and the center of each square in which data are collected should be recorded with a GPS. If Landsat is the target sensor, then cover should be estimated within 900 m^2 squares, which could be achieved through

radial transects or ocular estimates in plots within the square. If multiple sensors with different spatial resolutions are used in an analysis, a nested design is most appropriate. For example, randomly selected 1 m^2 plots (the number of plots depends on the desired confidence interval) within a 900 m^2 square could be used to estimate cover at the resolution of both aerial photographs and Landsat pixels (Figure 3.5).

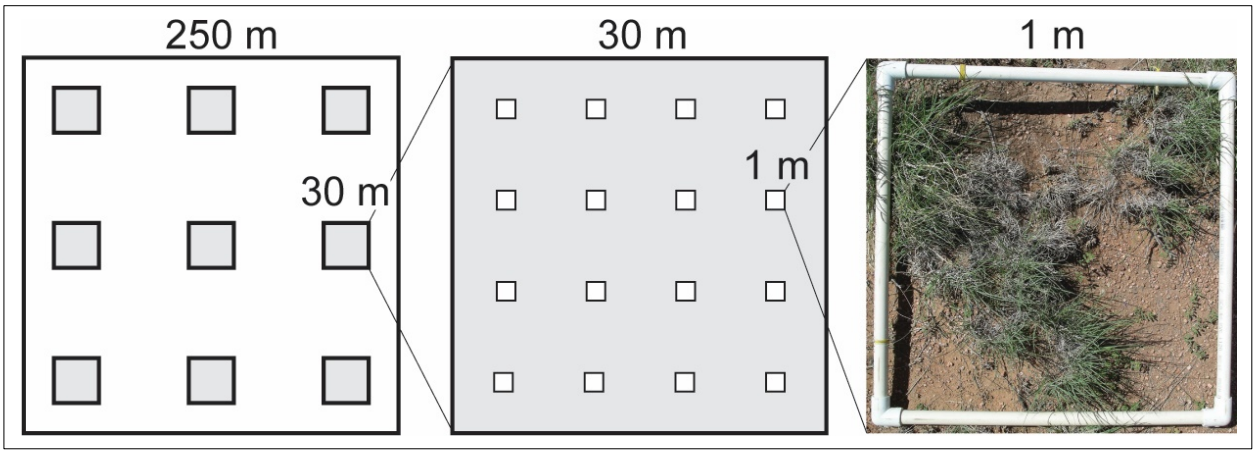


Figure A-3.5. Example of a nested field sampling design based on ocular estimates. A MODIS-sized pixel (left, 250 x 250 m) contains an array of nine Landsat-sized pixels (center, 30 x 30 m), which in turn contains an array of 16 1 m^2 pixels that can be classified in the field as soil, woody vegetation, or green vegetation associated with the target dominant species (right panel).

The spectral signatures of senescent and woody plants are distinctly different from those of photosynthetic plants (compare, for example, dry grass to green lawn grass; Figure 3.1a, b) and their cover should be classified separately, even if the senescent and photosynthetic individuals are members of the same species. Cover of bare soil should be measured because bare ground and soil contribute substantially to spectral and phenological signals.

Simultaneous collection of field data and image acquisition is ideal. At minimum, field data should be collected during seasons or growing conditions similar to those during image acquisition. Simultaneous field and sensor observations are especially important for classifications that rely on phenology or on spectra from a specific time period (e.g., flowering).

Appendix 4. Sources of data on presence and high abundance of red brome (*Bromus rubens*) and Sahara mustard (*Brassica tournefortii*).

Table A-4.1. If percent cover was unknown or recorded as <10%, the data were retained as presence points.

Source of data	Data type	Number of data points, red brome		Number of data points, Sahara mustard		Notes
		P[1]	HA[2]	P	HA	
Regional botanists (1)	Percent cover	318	0	15	0	Regional botanists (S. Abella, University of Nevada, Las Vegas and Cindy Salo, Sage Ecosystem Science)
Southwest Regional GAP Training Sites Databases (2)	Percent cover	38	8	28	1	Southwest Regional Gap training sites databases (earth.gis.usu.edu/swgap/trainingsites.html)
National Park Service botanists (3)	Percent cover	1169	49	67	2	National Park Service botanists. Data were provided by the National Park Service Mojave Desert Network Inventory & Monitoring Program, the NPS National Vegetation Mapping Program, and regional botanists (J. Cipra, Death Valley National Park; K. Kain, Joshua Tree National Park; and C. Norman, Lake Mead National Recreation Area).
National Park Service botanists (Lake Mead National Recreation Area) (3)	• Distribution characterizations • Under shrub • Scattered • Isolated patch • Isolated individual • Gradient • Continuous cover • Clumped	0	0	5041	469	National Park Service botanists. Data were provided by the National Park Service Mojave Desert Network Inventory & Monitoring Program, the NPS National Vegetation Mapping Program, and regional botanists (J. Cipra, Death Valley National Park; K. Kain, Joshua Tree National Park; and C. Norman, Lake Mead National Recreation Area).
National Park Service botanists (Joshua Tree National Park) (3)	Percent cover	0	0	443	63	National Park Service botanists. Data were provided by the National Park Service Mojave Desert Network Inventory & Monitoring Program, the NPS National Vegetation Mapping Program, and regional botanists (J. Cipra, Death Valley National Park; K. Kain, Joshua Tree National Park; and C. Norman, Lake Mead National Recreation Area).
National Park Service botanists (Death Valley National Park) (3)	• Qualitative descriptions. Examples: • Very dense, creating ground cover in most areas • Dense in places, creating ground cover, thinning quickly southward up canyon remnants 1+ yrs • Locally dense beneath shrubs and on banks; mostly new growth, few remnants • Abundant plant population; field survey; plants outside property and surrounding areas	42	0	15	6	National Park Service botanists. Data were provided by the National Park Service Mojave Desert Network Inventory & Monitoring Program, the NPS National Vegetation Mapping Program, and regional botanists (J. Cipra, Death Valley National Park; K. Kain, Joshua Tree National Park; and C. Norman, Lake Mead National Recreation Area).
Roadside surveys in California, Nevada, and Arizona (4)	Ranked abundance 0: None detected 1: Low abundance 2: High abundance along road corridors 3: High abundance extending away from road corridors	234	55	79	13	Roadside surveys in southern California, southern Nevada, and Arizona conducted in 2011 and 2012 by L. Pelech and B. Bradley (University of Massachusetts, Amherst)

[1] Points at which presence was recorded
[2] Points at which high abundance was recorded

Table A-4.1 (continued). If percent cover was unknown or recorded as <10%, the data were retained as presence points.

Source of data	Data type	Number of data points, red brome		Number of data points, Sahara mustard		Notes
Herbaria (5)	not applicable	864	0	0	0	Data were compiled from the University of Arizona herbarium, Northern Arizona University Deaver herbarium, Arizona State University Plant herbarium, Northern Great Plains herbarium, University of California Riverside herbarium, University of New Mexico herbarium, Herbarium of the University of Sonora, Mexico, Utah State University Intermountain herbarium, Grand Canyon National Park herbarium, and Desert Botanical Garden herbarium collection.
California Invasive Plant Council (6)	Ranked abundance 0: None detected 1: Low abundance 2: Moderate abundance 3: High abundance	0	0	9475	269	www.cal-ipc.org
CalFlora (7)	Percent cover	6342	156	0	0	www.calflora.org
Vegbank (8)	Percent cover	71	8	0	0	vegbank.org/vegbank/index.jsp
SWEMP (9)	not applicable	772	0	0	0	Southwest Exotic Plant Information Clearinghouse: Southwest Exotic Mapping Program (sbsc.wr.usgs.gov/research/projects/swepic/swemp/swempa.asp).
US Geological Survey (10)	not applicable	733	0	0	0	www.usgs.gov
Bureau of Land Management, Arizona (11)	not applicable	6	0	0	0	www.blm.gov/az/st/en.html

[1] Points at which presence was recorded
[2] Points at which high abundance was recorded

Appendix 5. Selection of projections of future climate.

Layers of data on future climate that are available through Worldclim were based on atmosphere ocean general circulation models (AOGCMs) from the Fifth Assessment Report (AR5) of the Intergovernmental Panel on Climate Change (IPCC). The baseline data for downscaling of these models by the IPCC were the Worldclim 1.4 data on current climate (Hijmans et al. 2005). For the NRCA, we selected the climate-model projections with the lowest errors in prediction of temperature and precipitation in the Pacific Northwest (Rupp et al. 2013) because a comparable error assessment has not been conducted for the southwestern United States. We selected AOGCMs to include in our ensemble that best modeled past temperature and precipitation patterns in the western United States (Rupp et al. 2013) and were available in a high-resolution downscaled format from Worldclim. We included AOGCM projections in our ensemble that were based on two representative concentration pathways (RCPs): RCP4.5 and RCP8.5. RCPs, which were used in AR5, replace the emissions scenarios that were used in previous IPCC assessment reports (Nakicenovic and Swart 2000). RCP4.5 is considered to be a medium-low pathway and is characterized by a stabilization of radiative forcing at 4.2 W/m^2 by 2100, which corresponds to atmospheric carbon dioxide (CO_2) concentrations of 650 ppm. RCP8.5 is considered to be a high pathway and projects a stabilization of radiative forcing at 8.3 W/m^2 by 2100, which corresponds to atmospheric CO_2 concentrations of 900 ppm (Stocker et al. 2013). Our ensemble included climate projections that were based on five AOGCMs from the following modeling groups and models: National Center for Atmospheric Research (CCSM4); National Centre for Meteorological Research (CNRM-CM5); Met Office Hadley Centre (HadGEM2-ES); Atmosphere and Ocean Research Institute, National Institute for Environmental Studies, and Japan Agency for Marine-Earth Science and Technology (MIROC5); and EarthClim (NorESM1-M).

We derived four covariates from the climate data: mean monthly temperature for the coldest (January) and warmest (July) months of the year and accumulated precipitation for two quarters (March through May, June through August). We did not include precipitation for the winter quarter (December through February) and autumn quarter (September through November) because they were highly correlated with spring and summer precipitation, but were not strongly associated with non-native species in earlier Maxent models that included all variables. We assumed that the covariates we derived would encompass climatic conditions that might be associated with the growth and reproduction of both native and non-native invasive species. When we began our assessment, there was little information available on strength of association between climatic variables and growth of Sahara mustard. By contrast, there was some evidence that red brome plants can be killed by cold winter temperatures (Bykova and Sage 2012) and that growth is limited by low autumn precipitation (Beatley 1966, Salo 2004). Both species likely compete with native perennial species that respond to spring and summer precipitation, which could lead to an indirect effect of climate on range extents that is mediated by competition.

Appendix 6. Modeling habitat consistent with presence and high abundance given current and projected future climate.

Modeling habitat given current climate

Many methods for building bioclimatic envelope models (BEMs; also referred to as species distribution models, habitat suitability models, or environmental niche models) have been developed. BEMs are used to identify the relations between the locations where a given species occurs and the climatic conditions at those locations (Franklin 2009). A model of climate consistent with the presence of the species then can be projected spatially to estimate the locations where the species may become established or abundant. One also can project the locations of future climatic habitat for the species on the basis of projected future climate. We used two BEM methods, Maxent and Bioclim, to project the current and future locations of habitat consistent with presence and high abundance of *B. rubens* and *B. tournefortii*.

We used Maxent (Version 3.3.3k), an implementation of maximum entropy modeling (Phillips et al. 2006), to model climatic habitat consistent with presence and high abundance. Maxent relies on presence-only data, but draws pseudo-absences from the study area to construct probabilistic relations between climate and the species' distribution. Accordingly, Maxent models are sensitive to the extent of the area from which pseudo-absences are drawn (VanDerWal et al. 2009). We used a mask in Maxent to restrict selection of pseudo-absence points to a convex hull around each species' occurrences. To account for uneven sampling of the species and therefore uneven distribution of the data for each species (Kramer-Schadt et al. 2013), we included a bias file for each of the species that was based on the presence of national parks and distance to roads. We evaluated the fit of each Maxent model with ten-fold cross-validation (the default setting). Maxent creates a different function for each climatic variable that reflects the extent to which values of the variable are associated with locations at which the species has been detected. This process generates a spatial model with continuous values of climatic habitat quality. We transformed this continuous model into a binary map of habitat and not-habitat on the basis of a threshold value that encompassed 95% of the locations at which the species was detected. Use of this threshold assumes that the species' presence or absence at almost all of the locations was classified correctly, which was consistent with our goal of characterizing likelihood of invasion at a coarse scale (fine-resolution characterization was not possible given the empirical data).

To reduce potential bias introduced by use of a single BEM, we also created Bioclim models of climatic habitat quality for *B. rubens* and *B. tournefortii.* Bioclim identifies thresholds for each climatic variable that encompass the distribution data (Busby 1991, Pearson and Dawson 2003). We used ArcGIS 10.1 to extract the values of the four climate variables for all of the locations at which the species had been recorded (Appendix 4), and then calculated the ranges of values of each variable that encompassed 95% of the records. That is, we excluded the highest and lowest 2.5% of values of each climate variable that were associated with the presence or high abundance records. We calculated climatic habitat within Bioclim as areas identified as consistent with presence or high

abundance in all four climate layers. We combined the Maxent and Bioclim results to identify areas that were climatic habitat in either model; we assumed that these areas could support the species.

We evaluated the Maxent models on the basis of area under the curve (AUC) values, which are widely applied to determine agreement between predicted species distributions and empirical records of occurrence (Fielding and Bell 1997, Thuiller 2003, Pearson et al. 2006). AUC values are based on the receiver operating curve (ROC), which plots the proportion of true positive predictions (sensitivity) against the proportion of false positive predictions (specificity). AUC values range from 0.5 (no better than random) to 1 (perfect agreement between model predictions and empirical data).

Modeling habitat given projected future climate

After identifying the ranges of values of climate variables currently consistent with presence or high abundance of *B. rubens* and *B. tournefortii*, we identified locations within those ranges as projected by models of 2050 climate. We repeated this process for each of the five AOGCM projections.

We created ensemble models of future habitat consistent with presence and high abundance of *B. rubens* and *B. tournefortii* by summing the set of binary maps (i.e., those created by Maxent and Bioclim for each AOGCM); a given location was identified as habitat in zero to ten models. Combining models generated with multiple BEMs and AOGCMs (Araújo and New 2007) reduces the effect of any single model or scenario, and the degree of model overlap or agreement serves as a measure of confidence. As the number of models that made the same prediction increases, uncertainty decreases. We created separate ensemble models for each relative concentration pathway.

To identify locations where future climate may be consistent with invasion by *B. rubens* or *B. tournefortii*, we identified areas in which climatic habitat may expand, be stable, and contraction. We assumed that current climatic habitat was present at any location where one or more Maxent or Bioclim models identified current climate as consistent with presence or high abundance.

We applied two thresholds to project where habitat for each invasive species may remain stable, expand, or contract. The lower threshold included all areas classified as future habitat by at least one of the ten model projections. The higher threshold included all areas classified as future habitat by six or more model projections. For both the lower threshold and higher threshold analyses, we defined areas of stability as those classified as habitat by models of both current habitat and future habitat. We defined areas of expansion as those classified as habitat by models of future habitat but not models of current habitat. We defined areas of contraction as those classified as habitat in models of current habitat but not in models of future habitat.

Appendix 7. Projected current distributions of red brome and Sahara mustard and projected future distribution of climatic habitat for the species given representative concentration pathway (RCP) 8.5.

Table A-7.1. Projected increases in distributional range (thousands of km^2) within the study area on the basis of a low threshold (climatic habitat identified by one or more models) and high threshold (climatic habitat identified by six or more models).

Species	Abundance	Time period	Low threshold		High threshold	
			Area	% Change	Area	% Change
Red brome	Presence	Current	736	–	736	–
		Future	1254	70	404	-45
	High abundance	Current	475	–	475	–
		Future	806	70	317	-33
Sahara mustard	Presence	Current	410	–	410	–
		Future	557	36	256	-38
	High abundance	Current	135	–	135	–
		Future	231	71	64	-53

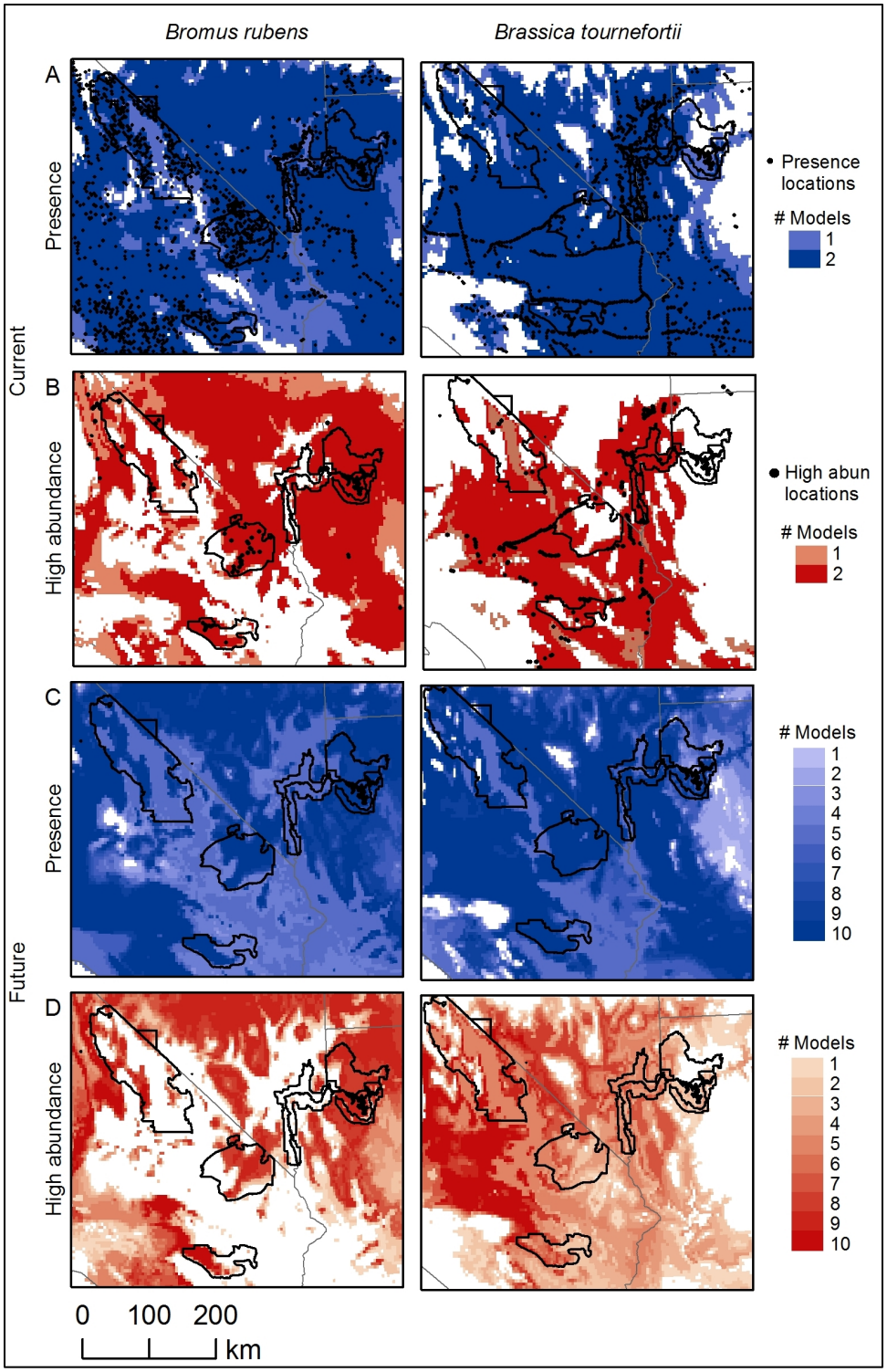


Figure A-7.1. Projected current and future distributions of climatic habitat for red brome (*Bromus rubens*) and Sahara mustard (*Brassica tournefortii*). Points (black dots in A and B) indicate where presence and abundance data were collected. We combined the Maxent and Bioclim projections of current climatic habitat to estimate the maximum extent of current presence and high abundance of each species (A, B). In C and D, the color ramp indicates how many of the ten models in the ensemble projected that future climate at a given location would be consistent with presence or high abundance.

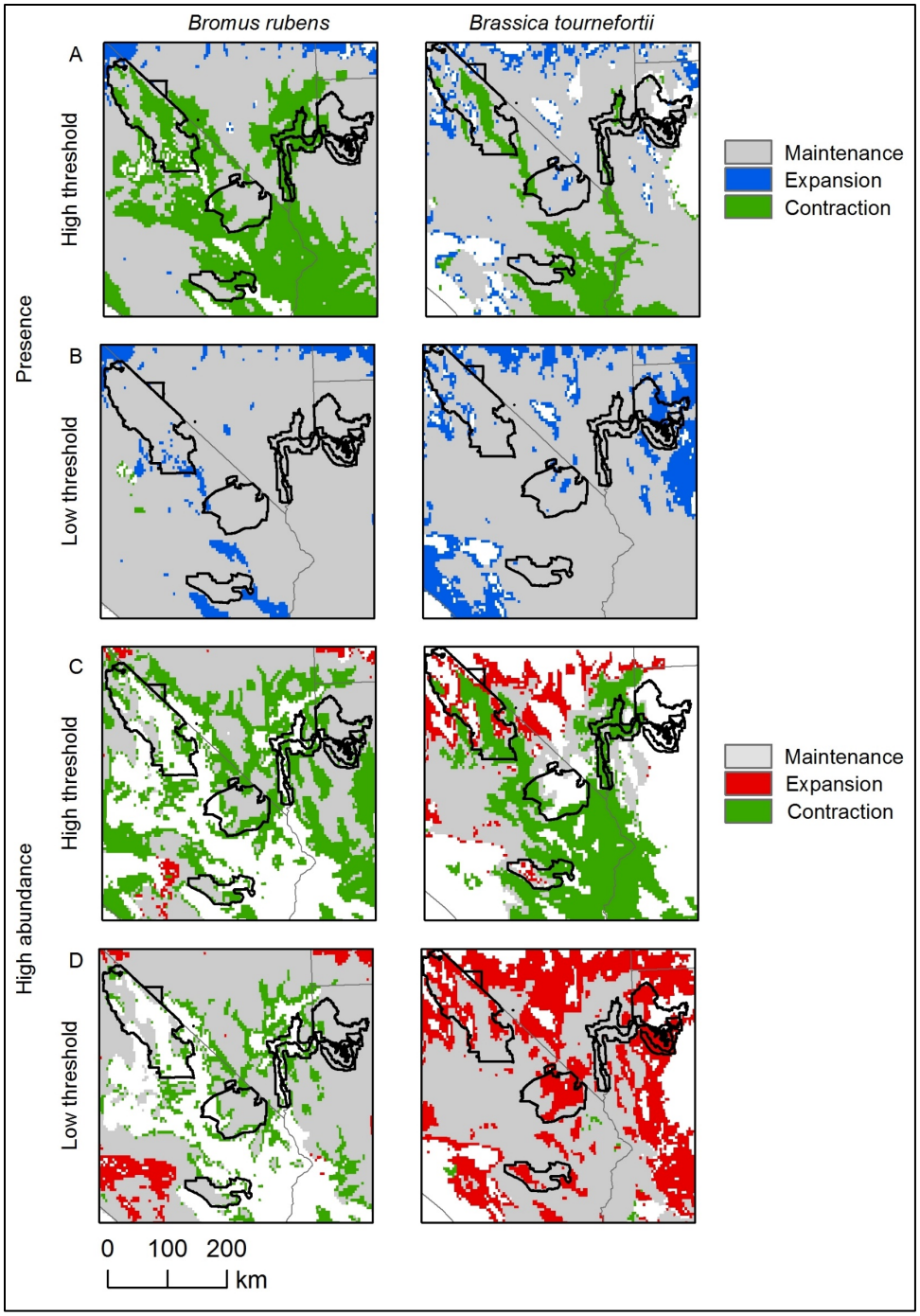


Figure A-7.2. Projected areas in which climatic habitat for red brome (*Bromus rubens*) and Sahara mustard (*Brassica tournefortii*) will be stable, expand, or contract by 2050 assuming representative concentration pathway (RCP) 8.5. Stable, location was classified as habitat by models of both current habitat and future habitat. Expansion, location was classified as habitat by models of future habitat but not models of current habitat. Contraction, location classified as habitat in models of current habitat but not in models of future habitat. High threshold, locations classified as future habitat by six or more of the ten model projections. Low threshold, locations classified as future habitat by one or more of the ten model projections.

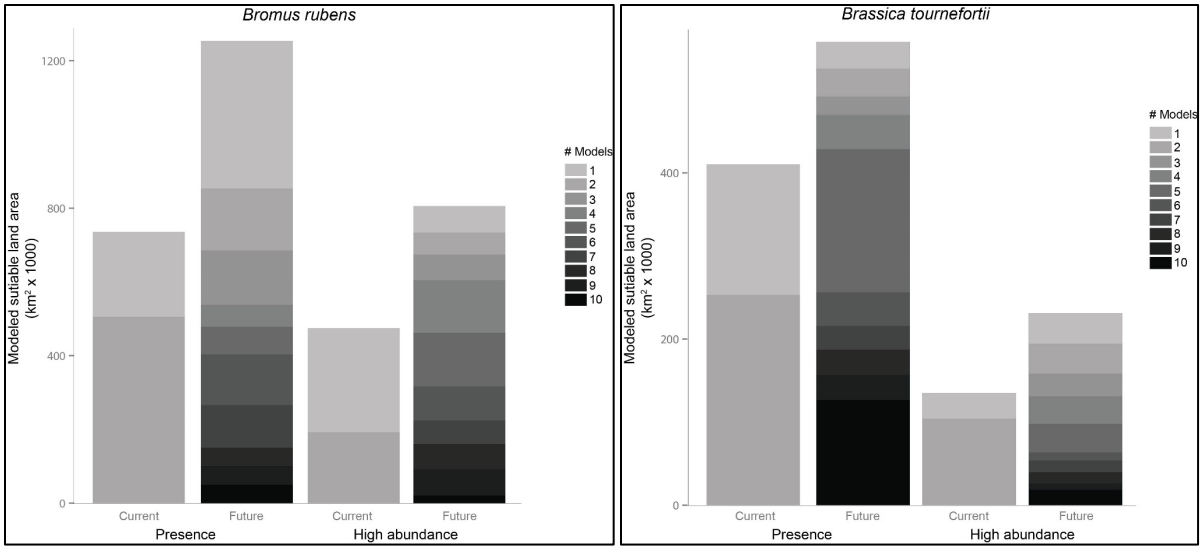


Figure A-7.3. Predicted area in which climate will be consistent with presence or high abundance of red brome and Sahara mustard currently and by 2050 assuming representative concentration pathway (RCP) 8.5.

Appendix 8. Derivation of environmental variables to assess the distribution and abundance of red brome as a function of topography, land cover, and phenology.

We derived elevation, topographic roughness, and aspect from a 90-m DEM (Gesch et al. 2002). Red brome tends to occur at about 500–1,500 m elevation (Beatley 1966, Hunter 1991, Brooks and Berry 2006, Abella et al. 2012). We used focal statistics (Miller et al. 1999) to calculate topographic roughness as the standard deviation of slope. Topographic roughness and associated hydrographic channels were associated with germination of the closely related species cheatgrass (*Bromus tectorum*) (Bradley and Mustard 2006). We included aspect (northness and eastness, calculated as the cosine and sine of aspect, respectively) because both red brome and cheatgrass occur most often on west- and northwest-facing slopes, which has been attributed to prevailing wind patterns and rain shadow effects (Bradley and Mustard 2006).

We used the NLCD (NLCD2006; Fry et al. 2011), which includes 16 land-cover classes across the conterminous United States at 30 m resolution, to derive land-cover variables. We clipped the NLCD2006 to the boundary of the Mojave Desert as defined by the US Environmental Protection Agency's Level III Ecoregions, and reclassified the data to include six classes. In order of greatest to least percent cover within the study area, the six were shrub/scrub, forest, grassland, barren, agriculture, and wetland. We reclassified as *no data* those pixels classified in the NLCD2006 as water or developed.

In the Mojave Desert, the density of or distance from roads is correlated with the cover of many non-native species (Brooks and Lair 2009). The fact that evidence for an association between roads and presence or abundance of red brome is conflicting (e.g., Johnson et al. 1975, Bradley and Mustard 2006, Brooks and Berry 2006, Craig et al. 2010) likely reflects differences in scales of analysis and the metrics used to quantify the relation. Our analysis included all primary and secondary roads from the 2011 TIGER/Line data (U.S. Census Bureau 2012). We did not include county roads to avoid introducing bias into model projections as a result of known errors in these data, including those documented in the text. For each pixel, we calculated distance from roads as the Euclidean distance to the nearest primary or secondary road at 90 m resolution.

Abella et al. (2012) found relations between the presence of red brome, soil great group (taxonomic units used to group soils with common characteristics), and parent material (the unaltered material from which the soil profile develops). Presence of red brome was less common on gypsum soil parent material than on limestone-sandstone and mixed alluvium. As a measure of geologic formation, we used data from the Mojave Desert Ecosystem Initiative (mojavedata.gov) that were derived from published state geological maps. We used the five classes of geologic rock type and age in those data: unconsolidated Quaternary and Pliocene deposits (0 to 6 Ma), volcanic and sedimentary Tertiary rocks (6 to 66 Ma), intrusive Cenozoic and Mesozoic rocks (~14 to 250 Ma), Mesozoic, Paleozoic, and Proterozoic strata (~65 to 1,400 Ma), and intrusive and metamorphic Precambrian rocks (~1,400 to 1,700 Ma).

In some cases, invasive species' phenological signatures are distinct from those of native species (Appendix 3). Land surface phenology refers to phenological signals aggregated (sometimes among land cover types) at the spatial resolution of a given satellite sensor. Land surface phenology metrics (e.g., the timing of the start and end of the growing season, the duration of the growing season, or the magnitude of productivity in a given season) are typically based on time series of vegetation indexes (e.g., NDVI). There are multiple methods for deriving these phenological metrics (de Beurs and Henebry 2010), and the effectiveness of a given method may vary geographically (White et al. 2009). Most land surface phenology metrics are derived from MODIS data.

Two primary sources of national land surface phenology data are available online: NASA's MODIS for the North American Carbon Program (MODIS–for–NACP) data products (www.accweb.nascom.nasa.gov) and the USGS Earth Resources Observation and Science (EROS) Center's remote sensing phenology data (phenology.cr.usgs.gov). The NASA annual phenology variables were derived with a curve-fitting algorithm (an asymmetric Gaussian smoothing function) to fit the time-series data and to derive dates of the start and end of the growing season (Tan et al. 2011). EROS calculates phenological metrics from MODIS time-series NDVI data on the basis of a curve derivative method, which employs a backward-looking or delayed moving average (Reed et al. 1994).

There are two obstacles to use of the NASA and EROS data sources. First, the statistical methods used to derive phenology variables for a given year, t, require data from years t-1 and t+1. As a result, data on current-year growing conditions and probabilities of disturbances (e.g., fire) cannot be calculated. Second, because these approaches are poorly calibrated to detect photosynthesis in arid ecoregions (White et al. 2009) (e.g., an NDVI time series with low-amplitude seasonal changes), there often are large gaps in the data. For example, phenology data were missing for more than 30% of plots in 2009 and 2011 in the MODIS–for–NACP data.

To address these limitations, we derived phenology variables for the Mojave Desert on the basis of the best-available NDVI time series data: the eMODIS products generated by EROS (Jenkerson et al. 2010). eMODIS products are generated over the conterminous United States, and records are available from 2000 through the present. eMODIS delivers a seven-day rolling composite, created daily from the most recent seven days of acquisition, which allows users to monitor real-time vegetation conditions.

First, we filtered the best-quality data in the NDVI time series on the basis of the data quality information that accompanies the images in the eMODIS data bundle. We removed pixels flagged by EROS as affected by clouds on the day of image acquisition and replaced them with the average of the NDVI values for that pixel from reliable data captured on the closest possible dates before and after the target day. Second, we smoothed the data with a Savitzky-Golay filter (Chen et al. 2004) to remove noise caused by atmospheric interference. Third, we used the smoothed time series to derive a suite of phenology variables (Table 29 in the main text).

We selected phenology variables on the basis of hypotheses about phenological differences between invasive species (e.g., red brome) and native species (e.g., Wolkovich and Cleland 2011) or

previously demonstrated associations between metrics derived from vegetation indexes and cover of plants in arid systems (e.g., Peterson 2005, Wallace and Thomas 2008). We generated data for all phenology variables at 90 m resolution. Additionally, to facilitate projections, we generated phenology data for 2005 (a relatively wet, productive year) and 2008 (a relatively dry year) (Figure 8.1).

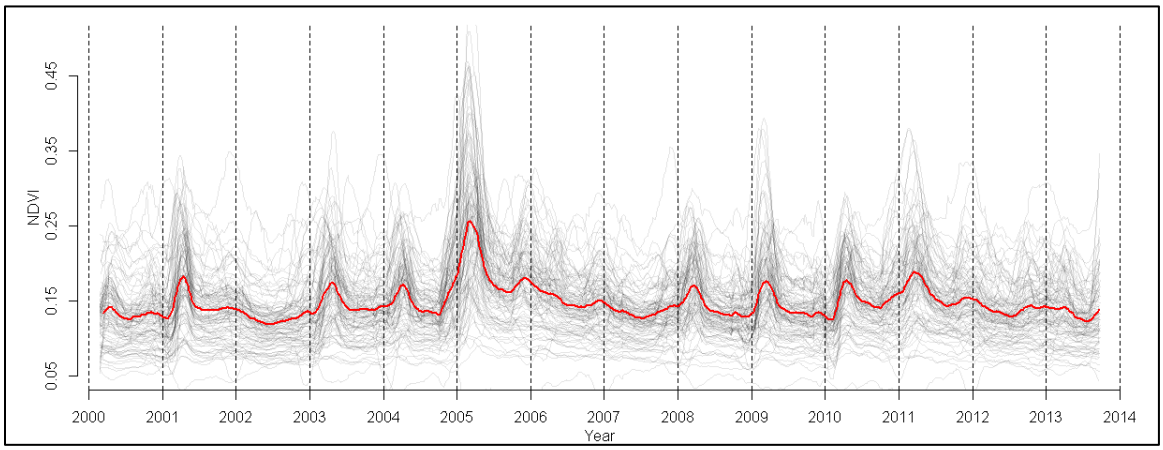


Figure A-8.1. Time series of NDVI for 100 pixels (gray lines) sampled at random from within the assessment area and the mean of the 100 pixels (red line).

Appendix 9. Modeling fitting, projections, and evaluation of models to assess the distribution and abundance of red brome as a function of topography, land cover, and phenology.

Methods

We used *k*-fold ($k = 10$) cross-validation to evaluate model accuracy. We created 10 partitions of the data, where each partition contained a subset of data for training and subset of data for testing. We created partitions by random sampling (without replacement) from the full set of data. We then fit and tested the model ten times. In each run, the records corresponding to the test subset were used to evaluate the model, whereas the other four groups were used to fit the model. The overall accuracy is estimated as the average of the ten partitions.

We used the Area Under the Receiver Operator Curve (AUC) to evaluate the accuracy of the occurrence models. The AUC is a measure of rank correlation. High AUC indicates that locations with high modeled occurrence values tend to be areas where the species is known to be present and locations with low modeled occurrence values tend to be areas where the species is known to be absent. AUC values of 0.5 indicate that the modeled occurrence values are no more accurate than randomly selected values. AUC scores of 0.6–0.7 indicate a poor fit of the model to the data, whereas values of 0.7–0.8, 0.8–0.9, and 0.9–1.0 indicate fair, good, and excellent fit, respectively. To evaluate the accuracy of the continuous cover class models, we quantified explained variation on the basis of the coefficient of determination, R^2, between modeled and observed cover class.

We built the final models of occurrence and cover across the Mojave Desert during 2005 and 2008 on the basis of optimal tuning parameters. We conducted basic parameter tuning in the R package *caret* on the basis of 10-fold cross-validation. For a gradient boosted machine learning model, there are three main tuning parameters: number of iterations (i.e., trees), complexity of the tree, and learning rate (i.e., how quickly the algorithm adapts). We conducted a grid-search of tuning-parameter values. We selected the tuning parameters that resulted in models with the largest AUC value and the highest coefficient of determination as the optimal parameters for the occurrence and cover models, respectively.

Results

The accuracy of the three machine learning algorithms was comparable (Table 9.1, 9.2). In general, random forest and gradient boosted approaches were more accurate than the support vector machine models. Because spatial predictions and the accuracy of predictions generated by the former two approaches were so similar, we report only the outputs from the gradient boosted model.

The optimal values for number of trees, tree complexity, and learning rate for the occurrence model were 500, 16, and 0.005, respectively. The optimal values for number of trees, tree complexity, and learning rate for the cover model were 1000, 5, and 0.01, respectively.

Table A-9.1. Accuracy of predictions of occurrence of red brome generated by three different machine learning algorithms.

Occurrence of red brome Algorithm	AUC range for independent folds		Mean AUC
	Minimum	Maximum	
Support vector machine	0.690	0.813	0.728
Random forest	0.708	0.828	0.761
Gradient boosted model	0.699	0.827	0.765

Table A-9.2. Accuracy of percent cover of red brome generated by three different machine learning algorithms.

Cover of red brome Algorithm	Range of variance explained for independent folds		Mean variance explained
	Minimum	Maximum	
Support vector machine	43%	62%	52%
Random forest	47%	64%	56%
Gradient boosted model	49%	67%	57%

Appendix 10. Relations between habitat quality for Mojave desert tortoise and habitat quality for its predators

We incorporated environmental covariates, identified on the basis of an extensive literature review, into our models of habitat quality for Common Ravens, Golden Eagles, coyotes, and black-tailed jackrabbits. We delineated land cover on the basis of data from the Landscape Fire and Resource Management Planning Tools Project (LANDFIRE). For coyotes, we adjusted non-anthropogenic land cover by above-ground productivity as measured by the normalized difference vegetation index (NDVI). We derived primary roads (typically interstate highways with four or more lanes) and secondary roads (typically main arteries such as U.S. highways without a median, state highways, or county highways), railroads, and power lines from TIGER/Line county-level data, and communications towers data from data made available by the Federal Communications Commission.

We used ArcGIS Spatial Analyst (v. 10.1, Esri, Redlands, California) to reclassify seamless (covering the full extent of the modeled area) spatial data for each covariate to binary raster data at 30-m resolution and to calculate Euclidean distances between cells, or patches of cells, containing a given land-cover class or linear feature. We assumed that within the perimeter of a patch of a given land-cover class or linear feature, habitat quality was either zero if the covariate was negatively associated with habitat quality or one if the covariate was positively associated with habitat quality. We further assumed that as distance from the patch or feature increased, habitat quality increased exponentially if the covariate was negatively associated with habitat quality or decreases exponentially if the covariate was positively associated with habitat quality. We applied the following exponential increase and decay functions:

> exponential increase: habitat quality = 1 – [exp (Euclidean distance from land-cover patch or feature/- θ)]
>
> exponential decay: habitat quality = exp (Euclidean distance from land-cover patch or feature/- θ),

where θ is a scalar that defines the asymptote where habitat quality approaches either 0 or 1 at a distance equivalent to the radius of a mean species-specific circular home range. Exponential increase and decay functions for individual species are below.

We included soil depth in models of habitat quality for fossorial species; activity of fossorial vertebrates increases as soil depth increases (Steers 2010). Soil depth across the study area ranged from 0–60 cm (38.7 ± 22.0, mean ± SD). We converted soil depth (rounded to the nearest 10 cm) to a measure of habitat quality by assigning locations with 40-cm deep soil a habitat-quality value of 0.50 and assigning habitat-quality values of 0, 0.05, 0.20, 0.35, 0.75, and 1.0 to locations with soil depths of 0, 10, 20, 30, 50, and 60 cm, respectively. Unless otherwise stated, we calculated the geometric mean of the 30-m cells within each land-cover patch or feature to estimate values of habitat quality.

We adjusted values of habitat quality that were derived from covariates positively associated with habitat quality on the basis of the mean monthly NDVI from November through March of 2000

through 2010. Mean NDVI values were based on measurements associated with all non-anthropogenic land cover classes (i.e., excluding agriculture, developed, and developed open land and low-density and high-density urban areas) (Table 10.1). We converted NDVI values to habitat quality values by assigning locations with mean NDVI a habitat quality of 0.5 and increasing or decreasing habitat-quality values by 0.1 for each standard deviation.

Table A-10.1. Covariates positively or negatively associated with habitat quality for desert tortoise, Common Raven, Golden Eagle, coyote, and black-tailed jackrabbit.

Species	Positive associations		Negative associations
	Abiotic	**Land cover and anthropogenic features**	
Desert tortoise	• desert washes • aspect • soil depth	• creosote-white bursage • Sonoran paloverde-mixed cacti desert scrub • NDVI	• primary and secondary roads • railroads • power lines • developed lands • agricultural lands
Common Raven breeding	N/A	• Joshua tree • deciduous canopy • evergreen canopy • mixed evergreen-deciduous open canopy • mixed evergreen-deciduous sparse canopy • towers • power lines • primary and secondary roads	• landfills • campgrounds • rest stops
Common Raven nonbreeding	N/A	• rest stops • landfills • campgrounds	N/A
Golden Eagle	N/A	• shrubland • power lines	• developed lands • agricultural lands • disturbed areas • railroads • primary roads • grassland
coyote	N/A	NDVI	• developed lands • agricultural lands
black-tailed jackrabbit	N/A	• grassland • shrubland • NDVI	• developed lands • agricultural lands

We merged spatial data in each of the species-level habitat quality models by selecting maximum habitat-quality values for covariates positively associated with habitat quality and by selecting minimum habitat quality values for covariates negatively associated with habitat quality. We selected maximum and minimum habitat-quality values with the Cell Statistics function in ArcGIS. We calculated the geometric mean of habitat-quality values associated with abiotic covariates because we had no empirical basis on which to weight these covariates. Final estimates of habitat quality for each species are the geometric mean of habitat-quality values based on covariates positively and negatively associated with habitat quality.

Habitat quality for Common Ravens

We estimated the mean home range of breeding Common Ravens as 5.1 km^2, corresponding to a circular home range with 1,274-m radius ($\theta = 160$) (Linz et al. 1992).

Covariates positively associated with habitat quality of breeding Common Ravens included those that we assumed were surrogate measures of nesting platforms, food associated with human activity, and natural food sources (Table 10.1). In the Mojave Desert, 57% of observed nests were built in Joshua trees (*Yucca brevifolia*), 27% in artificial structures such as towers and poles, and 13% in trees (Kristan and Borman 2007). We used data from LANDFIRE to delineate surrogate measures of nesting habitat: Joshua tree and deciduous, evergreen, and mixed evergreen-deciduous forests [characterizations consistent with those in LANDFIRE] (Table 10.1, 10.2). We used towers and power lines as measures of artificial nesting platforms (Table 10.2). We buffered nesting habitat by 1,274 m and assigned habitat quality values of 0.57, 0.27, and 0.13 to Joshua trees, artificial structures, and patches of the other land-cover types, respectively (Kristan and Borman 2007). We used exponential decay functions to model decreases in habitat quality as distance from patches of nesting habitat increased and adjusted habitat quality at distance 0 m, depending on nesting habitat, between 0.13 and 0.57:

> Joshua tree: [exp (Euclidean distance from Joshua trees/-160)]/1.73
>
> towers and power lines: [exp (Euclidean distance from towers and power lines/-160)]/3.7
>
> patches of forest land-cover classes: [exp (Euclidean distance from patches of forest /-160)]/7.8

Table A-10.2. Crosswalk of land cover variables derived from the LANDFIRE and features derived from TIGER/Line data.

Species	Land-cover type	Corresponding LANDFIRE or TIGER/Line classes
desert tortoise	Creosote-white bursage	LANDFIRE: 2082, 2087
desert tortoise	Sonoran paloverde-mixed cacti desert scrub	LANDFIRE: 2109
desert tortoise, Golden Eagle, coyote, black-tailed jackrabbit	developed lands	LANDFIRE: 13, 14, 15, 16, 17, 20, 21, 22, 23, 24
desert tortoise, Golden Eagle, coyote, black-tailed jackrabbit	agricultural lands	LANDFIRE: 80, 81, 82, 83, 84, 85
desert tortoise, Common Raven	primary (interstate highways) and secondary roads (main arteries such as United States, state, and county highways)	TIGER/Line: S1100, S1200
Golden Eagle	primary roads	TIGER/Line: S1100
desert tortoise, Common Raven, Golden Eagle	power lines	TIGER/Line: L4020
desert tortoise, Golden Eagle	railroads	TIGER/Line: R1011, R1051, R1052
Common Raven	Joshua tree	LANDFIRE: 2082
Common Raven	deciduous forest	LANDFIRE: 2011, 2012
Common Raven	evergreen forest	LANDFIRE: 2014, 2015, 2016, 2017, 2019, 2020, 2023, 2024, 2025, 2027, 2028, 2029, 2030, 2031, 2032, 2033, 2034, 2044
Common Raven	mixed evergreen-deciduous forest	LANDFIRE: 2061, 2062, 2113, 2114, 2118, 2152, 2154, 2159, 2180
Golden Eagle, black-tailed jackrabbit	shrublands	LANDFIRE: 2062, 2064, 2066, 2070, 2071, 2078, 2079, 2080, 2086, 2093, 2107, 2210, 2211, 2214, 2214, 2216, 2217, 2220, 2545
Golden Eagle	disturbed areas	LANDFIRE: 78, 2541, 2542, 2543, 2544, 2545, 2549
Golden Eagle, black-tailed jackrabbit	grasslands	LANDFIRE: 2121, 2123, 2130, 2131, 2135, 2138, 2146, 2181, 2182, 2184, 2191

Pairs of Common Ravens with nestlings typically forage within about 400 m of their nests (Sherman 1993). Concentrations of activity of breeding Common Ravens vary temporally and spatially. Nest-site fidelity ranged from 18% (on the basis of 54 nests that were observed over five years) to 62% (on the basis of 302 nests that were surveyed from one through five years) (Kristan and Borman 1997). To incorporate temporal variation in nest-site fidelity, we assigned habitat-quality values for breeding birds at different distances from all nesting habitat combined as

$$\text{habitat quality}_{0\text{-}400\text{m}} = 0.62$$

habitat quality$_{401\text{-}2877m}$ = [exp (Euclidean distance from patches of nesting habitat/-1700)]/1.61.

Because 59% of nests were within 2 km of roads, we assigned a habitat quality value of 0.59 to any area ≤ 2 km from a primary or secondary road and a value of 0.41 to any area > 2 km from a primary or secondary road (Table 10.2). Because about 62% of Common Ravens breed > 2 km from point sources of food (Kristan and Borman 2007), such as rest stops, campgrounds, and landfills, we assigned any area directly adjacent to a point-source of food a habitat quality value of 0.38, and areas > 2 km from a point-source of food habitat quality values > 0.60. We estimated habitat quality values between a point-source of food and areas up to 2 km away from a point source as

2.4 – [[exp (Euclidean distance from food source/-250)]/3.6]

We developed the model of habitat quality for breeding Common Ravens by taking the geometric mean of habitat quality values for all input layers.

We based the habitat quality model for nonbreeding Common Ravens on spatial data on landfills, campgrounds, and rest stops, which we assumed were surrogates for anthropogenic sources of food (Table 10.1, 10.2). Because nonbreeding Common Ravens are gregarious, not all anthropogenic sources of food are used at all times (Kristan and Borman 2007). Therefore, we developed an algorithm that incorporated both temporal variation in anthropogenic food-source use and distances from those sources: [exp (Euclidean distance from food source/-1700)]/1.25. Kristan and Borman (2007) found that nonbreeding Common Ravens concentrate within 2 km of anthropogenic food sources. Habitat quality at 0 m and 2000 m from an anthropogenic food source was 0.80 and 0.25, respectively.

We combined habitat quality models for nonbreeding and breeding Common Raven by selecting the maximum value in both models. To standardize habitat quality with other species, we divided all values by the highest habitat quality value, 0.6, to express habitat quality as a value between 0 and 1.

Habitat quality for Golden Eagles

We estimated mean home range at any time of year as 2,280 ha, corresponding to a circular home range with 2,694-m radius ($\theta = 350$).

Covariates positively associated with habitat quality for Golden Eagles were shrubland and power lines (Table 10.1, 10.2). We included developed and agricultural lands, disturbed areas, railroads, primary roads, and grassland as covariates negatively associated with habitat quality (Table 10.1, 10.2).

Habitat quality for coyotes

We estimated the mean home range of coyotes as 94.1 km^2, corresponding to a circular home range with 5,471-m radius ($\theta = 690$).

Mean NDVI, which we assumed was a surrogate measure of precipitation and, by extension, prey abundance, was positively associated with habitat quality for coyotes. We included developed and agricultural lands as covariates negatively associated with habitat quality for the species.

Habitat quality for black-tailed jackrabbits

We estimated the mean home range of black-tailed jackrabbits as 2.05 km^2, corresponding to a circular home range with 807-m radius ($\theta = 105$).

Covariates positively associated with habitat quality for black-tailed jackrabbits were grassland and shrubland. Covariates negatively associated with habitat quality for black-tailed jackrabbits were developed and agricultural lands (Table 10.1, 10.2).

Appendix 11. Connectivity and adaptive capacity of desert bighorn sheep: sampling methods

Water holes in locations around Death Valley (locations at which desert bighorn sheep feces and tissues were collected) were identified by Welles and Welles (1961) on the basis of observed use by desert bighorn sheep in the 1950s and by our own observations. Samples were collected in summer and autumn, and opportunistically year-round. We analyzed genetic data at the level of putative populations, which we assumed comprise individuals that interact and breed with each other. We defined populations on the basis of topography and distance: desert bighorn sheep are largely restricted to steep slopes (>15%) (Epps et al. 2007) and most gene flow occurs between populations separated by < 15 km (Epps et al. 2005). Therefore, we identified clusters of samples containing at least 10 individuals not separated by > 15 km within montane areas, and used ArcMap 10.0 (ESRI, Redlands California) to draw simple convex polygons that encompassed sample locations for each population (Figure 36 in main text).

Genotypes for this assessment were analyzed in two different laboratories (Epps lab, Oregon State University; Wehausen lab, Bishop, California) and were based on somewhat different sets of microsatellite loci. Thus, we restricted analyses to the loci that were included in both sets of data and realigned allele sizes to account for the different sizes of primers and allele-calling strategies used by the two laboratories. We identified differences in reported allele sizes for each locus by comparing genetic data generated by both laboratories for the Willow Spring population in the Black Mountains, and genotyping five samples in both laboratories. We translated all allele sizes to those used by the Epps laboratory to allow for consistent analysis across the entire data set. We genotyped samples collected from 2011–2013, and some samples from 2005–2007, in the Epps laboratory. We genotyped those samples at six microsatellite loci, identified and removed duplicate genotypes originating from the same individuals, and then genotyped samples at ten additional microsatellite loci. Samples collected from 1995–2010 from the western and southeastern portion of the study area (Figure 36 in main text) were genotyped at up to 20 microsatellite loci in the Wehausen population genetics laboratory in Bishop California. The two datasets had 15 loci in common (Table 30 in main text).

Appendix 12. Connectivity and adaptive capacity of desert bighorn sheep: patch connectivity metrics

We considered four connectivity variables from a previous analysis of bighorn sheep population connectivity in the Mojave Desert (Creech et al. 2014) as potential covariates in the regression of genetic diversity on NDVI. Creech et al. constructed network models of genetic connectivity (the potential for gene flow among populations) and demographic connectivity (the potential for colonization of empty habitat patches). Creech et al. (2014) used a genetic-based landscape resistance model (Epps et al. 2007) to estimate the strength of connections between populations in the networks. Bighorn sheep exhibit sex-biased dispersal, with males moving between patches more frequently and over greater distances than females; thus, genetic and demographic connectivity are not equivalent. Genetic connectivity is limited by the maximum effective dispersal distance (ED_{MAX}) of males, whereas demographic connectivity is limited by the shorter ED_{MAX} of females, as both sexes must disperse in order to colonize an empty habitat patch.

Two network metrics, weighted closeness and effectively connected patches, were then calculated, in each network, to characterize the degree to which each patch is connected to other network patches. Weighted closeness (WC) estimates the distance from a focal patch to every other patch in a network and accounts for differences in the strength of connections among patches. The WC reflects the long-term potential for transfer of genes or individuals to and from the focal patch because it considers all connections, including connections with distant patches that would require numerous dispersal steps.

Effectively connected patches (ECP) is the number of patches connected to a single focal patch along a path (single or multi-step) of effective distance < ED_{MAX}. ECP describes the short-term potential for genetic or demographic connectivity between a focal patch and neighboring patches.

The four variables we to characterize patch connectivity were demographic network WC, genetic network WC, demographic network ECP, and genetic network ECP. We calculated the correlation between each of these variables and our genetic diversity indices, allelic richness (A_r) and expected heterozygosity (H_e). Demographic WC was most highly correlated with both A_r and H_e (Table 12.1); therefore, we used it as our patch connectivity variable in the analysis of associations between NDVI and genetic diversity.

Table A-12.1. Pearson's correlations between connectivity metrics and genetic diversity indices.

Index	Demographic WC	Genetic WC	Demographic ECP	Genetic ECP
A_r	0.67	0.60	0.63	0.49
H_e	0.63	0.56	0.59	0.44

Appendix 13. Connectivity and adaptive capacity of desert bighorn sheep: use of NDVI for predicting diet quality and genetic diversity of desert bighorn sheep in the Mojave Desert

A. Rationale for and information provided by the assessment

NDVI is correlated with net primary productivity, plant biomass, and leaf area index, which are relevant to the quantity and quality of food for ungulates (Pettorelli et al. 2011). Moreover, NDVI has been related to individual- and population-level characteristics of ungulates such as body condition (Ryan et al. 2012) and body mass (Herfindal et al. 2006, Mysterud et al. 2008), which affect rates of survival and reproduction, and to conception rate (Rasmussen et al. 2006, Trimble et al. 2009) and breeding phenology (Wittemyer et al. 2007). However, because diet quality depends on forage quality (i.e., nutrient content), it is spatially and temporally variable. Additionally, the relation between NDVI and diet quality differs among species and environments. Accordingly, we validated the relation between NDVI and diet quality with long-term data on fecal nitrogen from five populations of desert bighorn sheep in the Mojave Desert. Fecal nitrogen has been used as a measure of diet quality for bighorn sheep (Wehausen 1992, 2005; Irwin et al. 1993, Rubin et al. 2002, Blanchard et al. 2003), but obtaining data on fecal nitrogen still requires extensive field sampling.

We considered four potential complications to using NDVI to assess diet quality for bighorn sheep and, potentially, many other herbivores. First, the relations between NDVI and diet quality may differ among populations of a given species (Martinez-Jauregui et al. 2009). Plant composition, and resulting diet quality or quantity, may vary dramatically among locations with similar NDVI values. Second, it is unclear how best to summarize the pixel-level NDVI values within a habitat patch to most accurately reflect the degree to which animals may use areas with different forage quality. Third, not all variation within the range of observed NDVI values may be relevant to diet quality. Fourth, although NDVI can be summarized for a particular habitat patch at a variety of temporal resolutions (e.g., daily, seasonally, across multiple years), NDVI may not provide much information about bighorn sheep diet quality at all of these resolutions.

Nevertheless, even the ability to distinguish between years with unusually high and low diet quality would make NDVI quite useful for monitoring population dynamics of bighorn sheep. If NDVI is strongly associated with diet quality at the resolution of a growing season, it may be possible to use NDVI to assess the long-term effect of patch-level forage quality on bighorn sheep populations. Such information would be highly relevant for assessing potential reintroduction sites or parameterizing models of habitat quality (DeYoung et al. 2000). Demographic data such as population sizes or recruitment rates would allow the most direct test of the long-term information content of NDVI, but such demographic data are rare for bighorn sheep populations. We used genetic data from 25 bighorn sheep populations in the Mojave Desert to test whether long-term NDVI explained variation in genetic diversity after accounting for connectivity among populations.

B. Methods

Our assessment included 25 currently occupied patches of habitat in southern California and Nevada for which genetic data were available (Epps et al. 2005, 2006). Three patches (Eagle Mountains, Little San Bernardino, Queen Mountains) were within or overlapped Joshua Tree, and six patches (Clark Mountains/South Spring Range, Granite Mountains, Old Dad Peak, Piute Range/Castle Peaks/Castle Mountains, Providence Mountains, Wood Mountains/Hackberry Mountains) were within or overlapped Mojave National Preserve (Figure 13.1). Patch boundaries were delineated along the margins of mountain ranges where steep slopes transition to flat valleys; we used 15% slope as a cutoff (Epps et al. 2007). In most cases, *patch* and *population* are interchangeable because each population of Mojave Desert bighorn sheep corresponds to a single habitat patch (Table 13.1). We used NDVI data from 2000 through 2011 that were derived from MODIS images with 250 m resolution (Gao et al. 2008, Sesnie et al. 2011) and fecal nitrogen data for 293 samples that were collected from five populations from 2000 through 2011. We examined the relation between fecal nitrogen and NDVI at two temporal resolutions: the sample (essentially a snapshot in time) and the winter-spring growing season [1 October through 30 June, the period during which the vast majority of precipitation in the Mojave Desert occurs (Beatley 1974)].

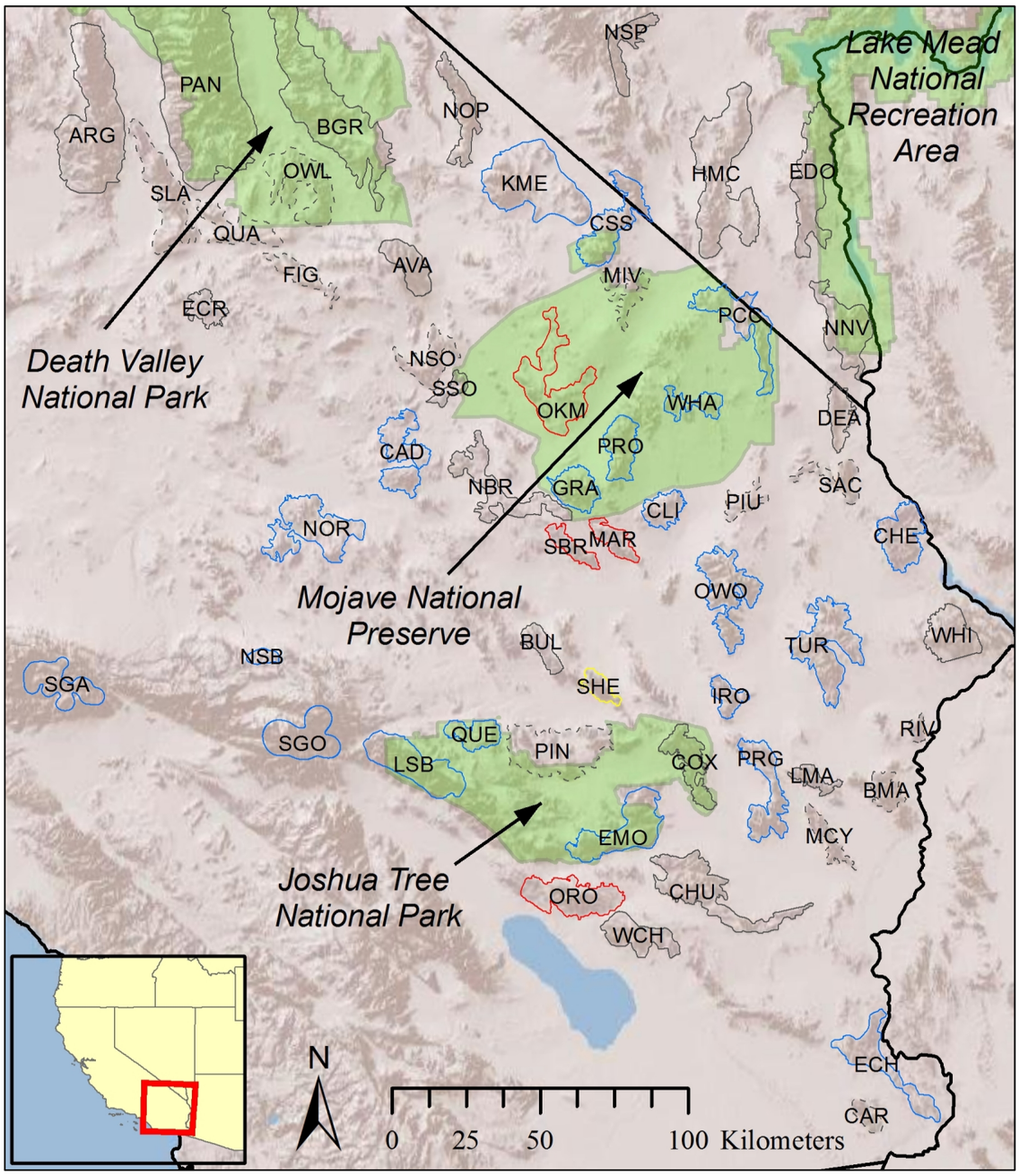


Figure A-13.1. Bighorn sheep populations and habitat patches for which data on fecal nitrogen (yellow outline), genetics (blue outline), or both (red outline) were available. Solid gray and dashed gray outlines indicate other occupied and unoccupied habitat patches, respectively, in the region. Green polygons are national parks. ARG, Argus Mountains; AVA, Avawatz Mountains; BGR, Black Mountains/Greenwater Range; BMA, Big Maria Mountains; BU, Bullion Mountains; CAD, Cady Mountains; CAR, Cargo Muchacho Mountains; CHE, Chemehuevi Mountains; CHU, Chuckwalla Mountains; CLI, Clipper Mountains; COX, Coxcomb Mountains; CSS, Clark Mountains/South Spring Range; DEA, Dead Mountains; ECH, East Chocolate Mountains; ECR, Eagle Crags; EDO, Eldorado Mountains; EMO, Eagle Mountains; FIG, Fort Irwin Granite Mountains; GRA, Granite Mountains; HMC, Highland Range/McCullough Range; IRO, Iron Mountains; KME, Kingston Mountains/Mesquite Range; LMA, Little Maria Mountains; LSB, Little San Bernardino Mountains; MAR, Marble Mountains; MCY, McCoy Mountains; MIV, Mescal Range/Ivanpah Mountains; NBR, North Bristol Mountains; NNV, Newberry Mountains; NOP, Nopah Mountains; NOR, Newberry Mountains/Ord Mountains; NSB, North San Bernardino Mountains; NSO, North Soda Mountains; NSP, North Spring Range; OKM, Old Dad Peak/Kelso Mountains/Marl Mountains/Club Peak/Indian Spring; ORO, Orocopia Mountains; OWL, Owlshead Mountains; OWO, Old Woman Mountains; PAN, Panamint Mountains; PCC, Piute Range/Castle Peaks/Castle Mountains; PIN, Pinto Mountains; PIU, Piute Mountains; PRG, Palen Mountains/Riverside Granite Mountains; PRO, Providence Mountains; QUA, Quail Mountains; QUE, Queen Mountain; RIV, Riverside Mountains; SAC, Sacramento Mountains; SBR, South Bristol Mountains; SGA, San Gabriel Mountains; SGO, San Gorgonio Mountains; SHE, Sheephole Mountains; SLA, Slate Range; SSO, South Soda Mountains; TUR, Turtle Mountains; WCH, West Chocolate Mountains; WHA, Woods Mountains/Hackberry Mountains; WHI, Whipple Mountains.

Table A-13.1. Characteristics of bighorn sheep habitat patches in the Mojave Desert region. Water, elevation, and precipitation data were used by Epps et al. (2004) to predict extinction probability of Mojave Desert populations, which may be tied strongly to diet quality.

Patch name	Patch abbreviation	Populated[1]	Area (km^2)	Water[2]	Maximum elevation (m)	Maximum precipitation[3] (mm)
Argus Mountains	ARG	Yes	865.3	Yes	2685	317.5
Avawatz Mountains	AVA	Yes	172.5	Yes	1875	190.5
Black Mts/ Greenwater Range	BGR	Yes	1021.3	Yes	1940	254
Big Maria Mountains	BMA	No	105.0	No	1029	127
Bullion Mountains	BUL	Yes	109.1	Yes	1271	254
Cady Mountains	CAD	Yes	280.0	Yes	1401	254
Cargo Muchacho Mountains	CAR	No	70.5	No	677	127
Chemehuevi Mountains	CHE	Yes	250.2	Yes	1125	190.5
Chuckwalla Mountains	CHU	Yes	437.2	Yes	1371	254
Clipper Mountains	CLI	Yes	124.6	Yes	1399	190.5
Coxcomb Mountains	COX	Yes	201.4	No	1340	254
Clark Mountains/ South Spring Range	CSS	Yes	300.1	Yes	2414	317.5
Dead Mountains	DEA	Yes	133.0	Yes	1085	190.5
East Chocolate Mountains	ECH	Yes	283.8	Yes	664	127
Eagle Crags	ECR	Yes	103.6	Yes	1676	190.5
Eldorado Mountains	EDO	Yes	428.1	Yes	1543	190.5
Eagle Mountains	EMO	Yes	352.1	Yes	1624	190.5
Ft Irwin Granite Mountains	FIG	No	122.1	Yes	1613	190.5
Granite Mountains	GRA	Yes	171.1	Yes	2058	317.5

[1] Yes = bighorn sheep population currently occupying patch; No = currently unoccupied, but evidence of previous occupation

[2] Yes = permanent surface water source within patch (e.g., natural spring or seep, artificial guzzler); No = no permanent surface water

[3] Maximum of average annual precipitation (1961–1990) among all pixels completely or partially within patch boundary, derived from 400-ha resolution PRISM climate data (PRISM Climate Group, Web site: http://www.prism.oregonstate.edu)

[4] Bighorn sheep sign recently has been observed in the Owlshead Mountains by biologists at Death Valley National Park; genetic analyses will determine if both sexes are present in this patch

Table A-13.1 (continued). Characteristics of bighorn sheep habitat patches in the Mojave Desert region. Water, elevation, and precipitation data were used by Epps et al. (2004) to predict extinction probability of Mojave Desert populations, which may be tied strongly to diet quality.

Patch name	Patch abbreviation	Populated[1]	Area (km^2)	Water[2]	Maximum elevation (m)	Maximum precipitation[3] (mm)
Highland Range/ McCullough Range	HMC	Yes	746.8	Yes	2139	317.5
Iron Mountains	IRO	Yes	97.5	Yes	1013	127
Kingston Mountains/ Mesquite Mountains	KME	Yes	637.5	Yes	2231	317.5
Little Maria Mountains	LMA	Yes	76.2	No	916	127
Little San Bernardino Mountains	LSB	Yes	357.5	Yes	1770	317.5
Marble Mountains	MAR	Yes	112.0	Yes	1170	190.5
McCoy Mountains	MCY	No	106.5	No	865	127
Mescal Range/ Ivanpah Mountains	MIV	No	131.0	Yes	1981	190.5
North Bristol Mountains	NBR	Yes	322.3	Yes	1238	190.5
Newberry Mountains	NNV	Yes	308.1	Yes	1712	190.5
Nopah Mountains	NOP	Yes	216.6	Yes	1944	254
Newberry Mountains/ Ord Mountains/ Rodman Mountains	NOR	Yes	406.3	Yes	1925	190.5
North San Bernardino/Cushenbury Mountains	NSB	Yes	43.3	Yes	2517	698.5
North Soda Mountains	NSO	No	181.9	No	1115	254
North Spring Range	NSP	Yes	157.9	Yes	2481	317.5
Old Dad Peak/ Kelso Mountains/ Marl Mountains/ Club Peak/ Indian Spring	OKM	Yes	433.2	Yes	1505	190.5
Orocopia Mountains	ORO	Yes	323.6	Yes	1163	127
Owlshead Mountains	OWL	No[4]	339.5	Yes	1345	254

[1] Yes = bighorn sheep population currently occupying patch; No = currently unoccupied, but evidence of previous occupation

[2] Yes = permanent surface water source within patch (e.g., natural spring or seep, artificial guzzler); No = no permanent surface water

[3] Maximum of average annual precipitation (1961–1990) among all pixels completely or partially within patch boundary, derived from 400-ha resolution PRISM climate data (PRISM Climate Group, Web site: http://www.prism.oregonstate.edu)

[4] Bighorn sheep sign recently has been observed in the Owlshead Mountains by biologists at Death Valley National Park; genetic analyses will determine if both sexes are present in this patch

Table A-13.1 (continued). Characteristics of bighorn sheep habitat patches in the Mojave Desert region. Water, elevation, and precipitation data were used by Epps et al. (2004) to predict extinction probability of Mojave Desert populations, which may be tied strongly to diet quality.

Patch name	Patch abbreviation	Populated[1]	Area (km^2)	Water[2]	Maximum elevation (m)	Maximum precipitation[3] (mm)
Old Woman Mountains	OWO	Yes	341.6	Yes	1620	190.5
Panamint Mountains	PAN	Yes	2230.1	Yes	3367	444.5
Piute Range/ Castle Peaks/ Castle Mountains	PCC	Yes	306.1	Yes	1783	190.5
Pinto Mountains	PIN	No	474.8	No	1405	254
Piute Mountains	PIU	No	91.7	Yes	1277	190.5
Palen Mountains/ Riverside Granite Mountains	PRG	Yes	264.5	Yes	1324	127
Providence Mountains	PRO	Yes	186.1	Yes	2181	317.5
Quail Mountains	QUA	No	214.5	Yes	1563	190.5
Queen Mtn	QUE	Yes	136.4	Yes	1730	190.5
Riverside Mountains	RIV	No	36.5	No	686	127
Sacramento Mountains	SAC	No	122.2	Yes	1010	190.5
South Bristol Mountains	SBR	Yes	96.3	Yes	1063	190.5
San Gabriel Mountains	SGA	Yes	250.6	Yes	3068	1079.5
San Gorgonio Mountains	SGO	Yes	311.0	Yes	3505	1079.5
Sheephole Mountains	SHE	Yes	78.9	Yes	1404	254
Slate Range	SLA	No	386.8	Yes	1706	190.5
South Soda Mountains	SSO	Yes	62.3	Yes	741	254
Turtle Mountains	TUR	Yes	421.8	Yes	1311	190.5

[1] Yes = bighorn sheep population currently occupying patch; No = currently unoccupied, but evidence of previous occupation

[2] Yes = permanent surface water source within patch (e.g., natural spring or seep, artificial guzzler); No = no permanent surface water

[3] Maximum of average annual precipitation (1961–1990) among all pixels completely or partially within patch boundary, derived from 400-ha resolution PRISM climate data (PRISM Climate Group, Web site: http://www.prism.oregonstate.edu)

[4] Bighorn sheep sign recently has been observed in the Owlshead Mountains by biologists at Death Valley National Park; genetic analyses will determine if both sexes are present in this patch

Table A-13.1 (continued). Characteristics of bighorn sheep habitat patches in the Mojave Desert region. Water, elevation, and precipitation data were used by Epps et al. (2004) to predict extinction probability of Mojave Desert populations, which may be tied strongly to diet quality.

Patch name	Patch abbreviation	Populated[1]	Area (km^2)	Water[2]	Maximum elevation (m)	Maximum precipitation[3] (mm)
West Chocolate Mountains	WCH	Yes	211.0	Yes	928	254
Woods Mountains/ Hackberry Mountains	WHA	Yes	111.3	Yes	1886	317.5
Whipple Mountains	WHI	Yes	245.5	Yes	1255	190.5

[1] Yes = bighorn sheep population currently occupying patch; No = currently unoccupied, but evidence of previous occupation

[2] Yes = permanent surface water source within patch (e.g., natural spring or seep, artificial guzzler); No = no permanent surface water

[3] Maximum of average annual precipitation (1961–1990) among all pixels completely or partially within patch boundary, derived from 400-ha resolution PRISM climate data (PRISM Climate Group, Web site: http://www.prism.oregonstate.edu)

[4] Bighorn sheep sign recently has been observed in the Owlshead Mountains by biologists at Death Valley National Park; genetic analyses will determine if both sexes are present in this patch

We included connectivity between populations as a covariate in our analyses of the association between NDVI and genetic diversity (allelic richness and expected heterozygosity) because genetic diversity in this system is strongly affected by exchange of alleles between neighboring populations (Epps et al. 2005).

C. Assessment results

The peak magnitude and seasonal pattern of NDVI varied considerably among years from 2000 through 2011 (Figure 13.2). The timing and relative magnitude of peaks in NDVI were largely consistent among patches within the region. In Marble Mountains and Old Dad Peak, the timing of peaks and troughs in the fecal nitrogen curve was roughly consistent with those in the NDVI curve (Figure 13.3).

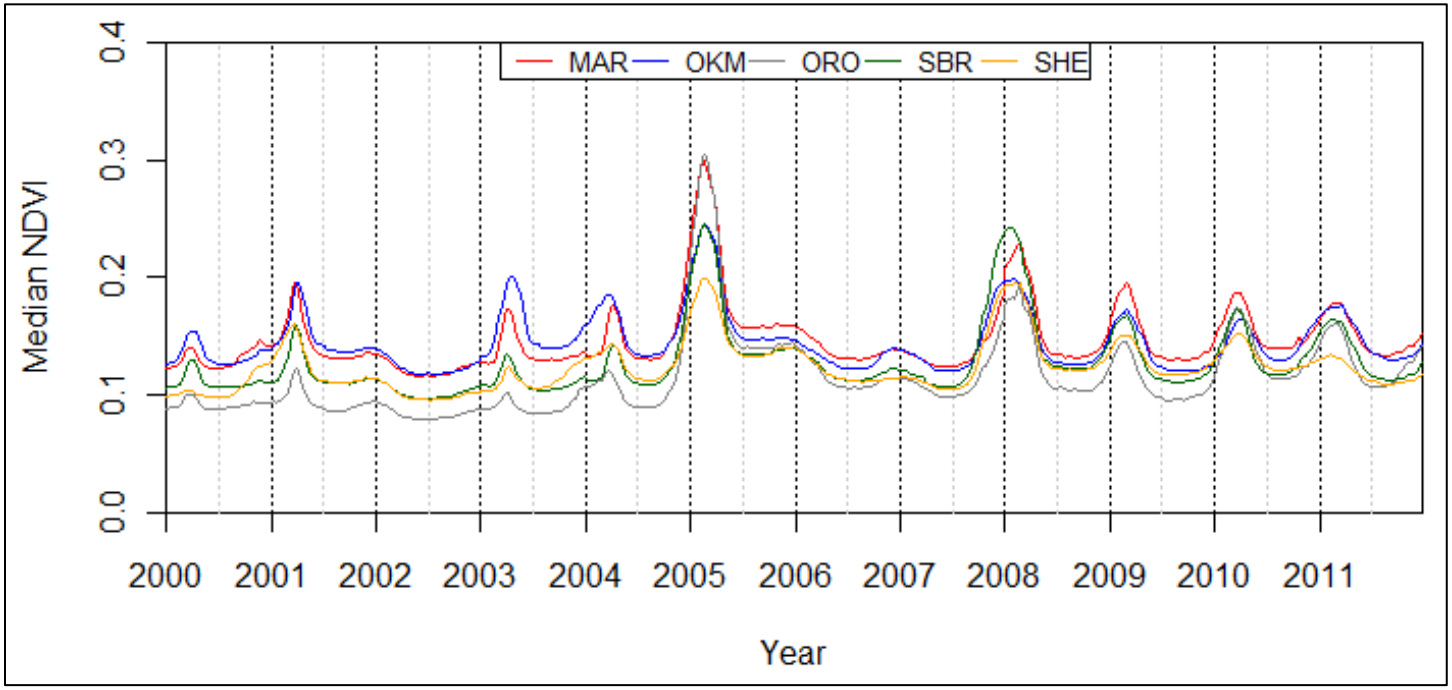


Figure A-13.2. Time series of the median Normalized Difference Vegetation Index (NDVI) from 2000 through 2011 for five patches of habitat for desert bighorn sheep in the Mojave Desert: Marble Mountains (MAR), Old Dad Peak (OKM), Orocopia Mountains (ORO), South Bristol Mountains (SBR), and Sheephole Mountains (SHE). Dotted black and gray vertical lines represent 1 January and 1 July, respectively, of each year.

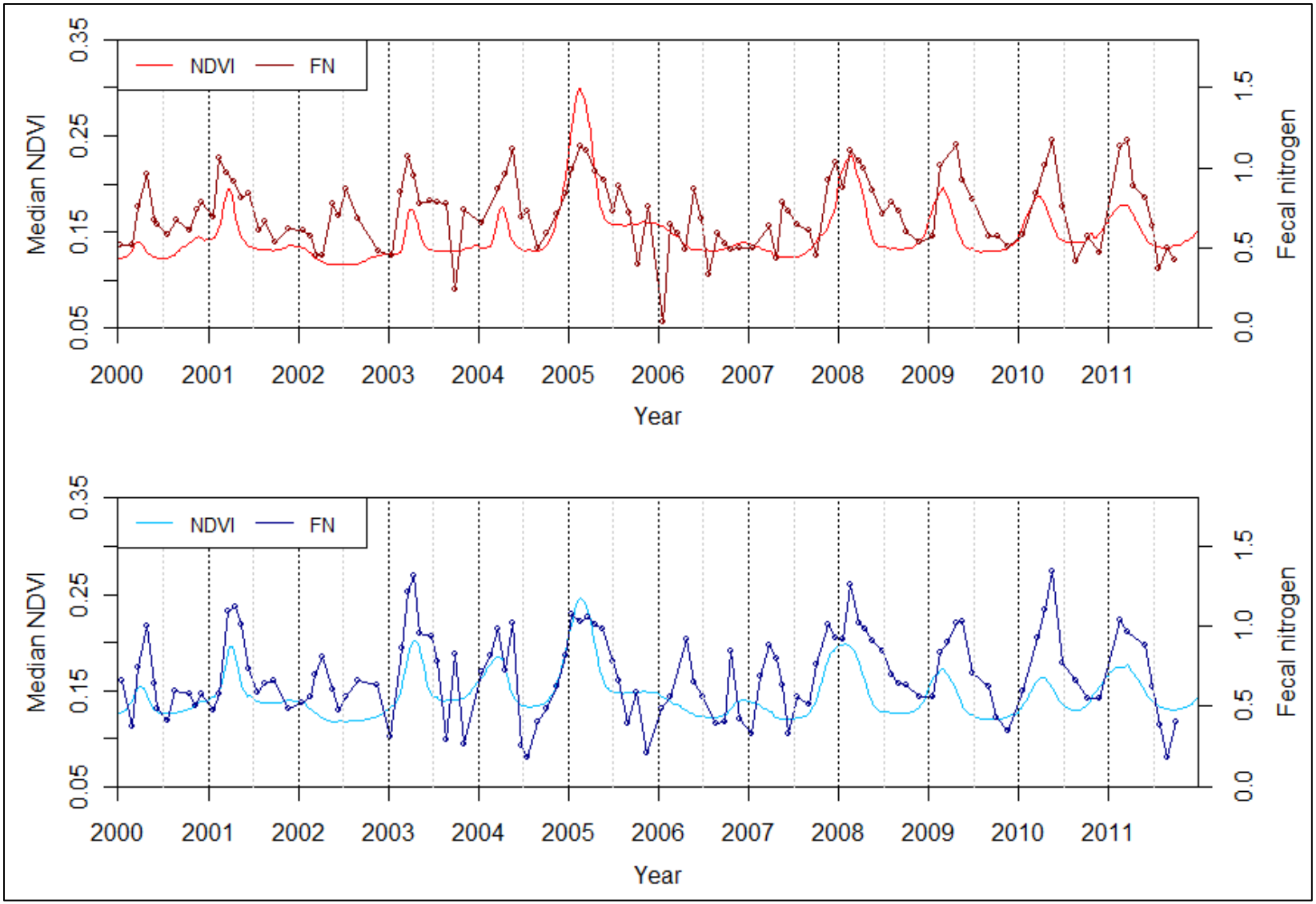


Figure A-13.3. Median Normalized Difference Vegetation Index (NDVI) and fecal nitrogen (FN) for the Marble Mountains (top) and Old Dad Peak (bottom) patches of habitat for or populations of desert bighorn sheep. Dotted black and gray vertical lines represent 1 January and 1 July, respectively, of each year.

We found that NDVI had limited potential to distinguish between high- and low-quality diets at the sample level. However, NDVI was strongly associated with fecal nitrogen at the season level, particularly during the window of peak NDVI. Our results suggested that at the season level, NDVI could reliably distinguish between high- and low-quality diets (Figure13. 4).

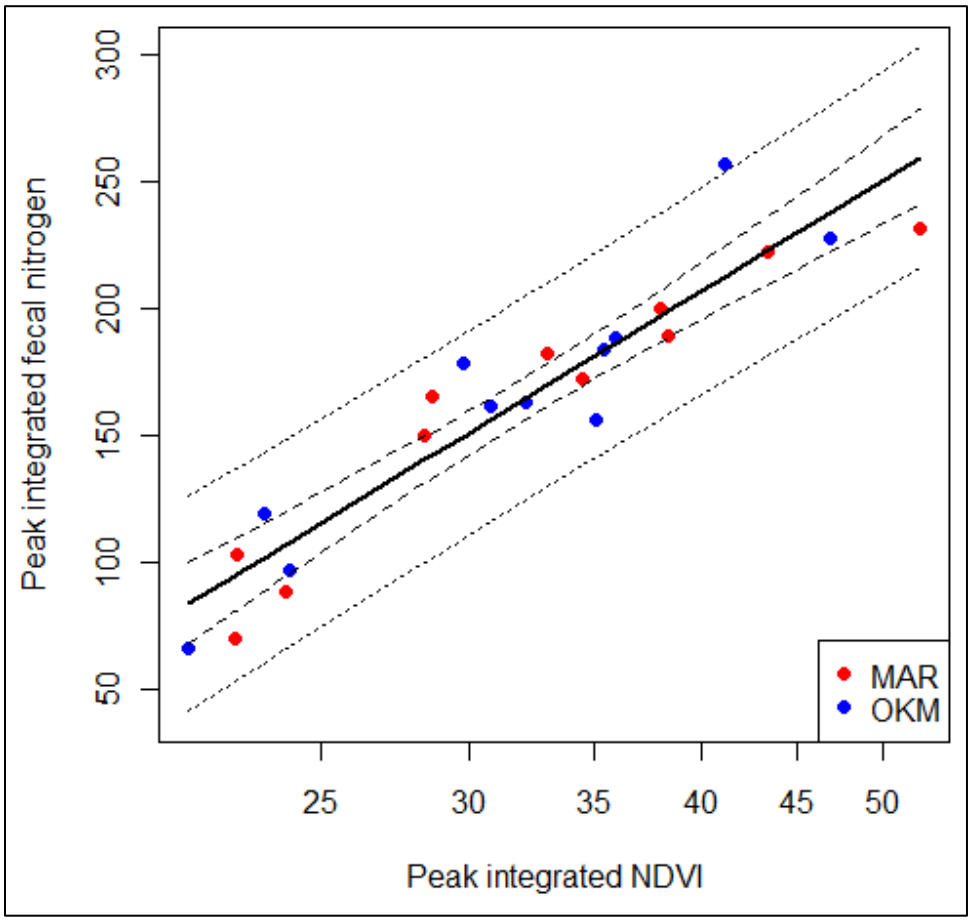


Figure A-13.4. Relation between integrated fecal nitrogen and the integrated Normalized Difference Vegetation Index (NDVI) during the peak of the growing season for the Marble Mountains (MAR) and Old Dad Peak (OKM) populations from 2001 through 2011. Regression line (solid line) represents output of a model in which slope and intercept did not vary between populations. Dashed and dotted lines are 95% pointwise confidence and prediction intervals, respectively.

After accounting for connectivity, both allelic richness and expected heterozygosity were positively related to growing-season NDVI from 2001–2011 (Table 13.2, Figure 13.5). For both measures of genetic diversity indices, connectivity was more strongly associated with genetic diversity than NDVI. However, models that included both covariates were better supported by the data than single-factor models.

Table A-13.2. Relations between genetic diversity and long-term NDVI (median of NDVI integrated over the growing season from 2001 through 2011) for 22 bighorn sheep populations in the Mojave Desert. Connectivity was measured as demographic weighted closeness. *p*-values reflect significance from F-tests of β=0 for a given covariate or all model covariates.

Response	Covariates	Covariate *p*	Model *p*	R^2	ΔAIC_c
Allelic richness	Median NDVI	0.15	0.01	0.38	11.94
	Square of median NDVI	0.26	–	–	–
	Log (connectivity)	<0.001	<0.001	0.43	7.04
	Median NDVI	0.01	<0.001	0.69	0
	Square of median NDVI	0.01	–	–	–
	Log (connectivity)	<0.001	–	–	–
Expected heterozygocity	Median NDVI	0.24	0.02	0.34	9.77
	Square of median NDVI	0.39	–	–	–
	Log (connectivity)	0.001	0.001	0.42	3.90
	Median NDVI	0.03	0.001	0.64	0
	Square of median NDVI	0.04	–	–	–
	Log (connectivity)	0.001	–	–	–

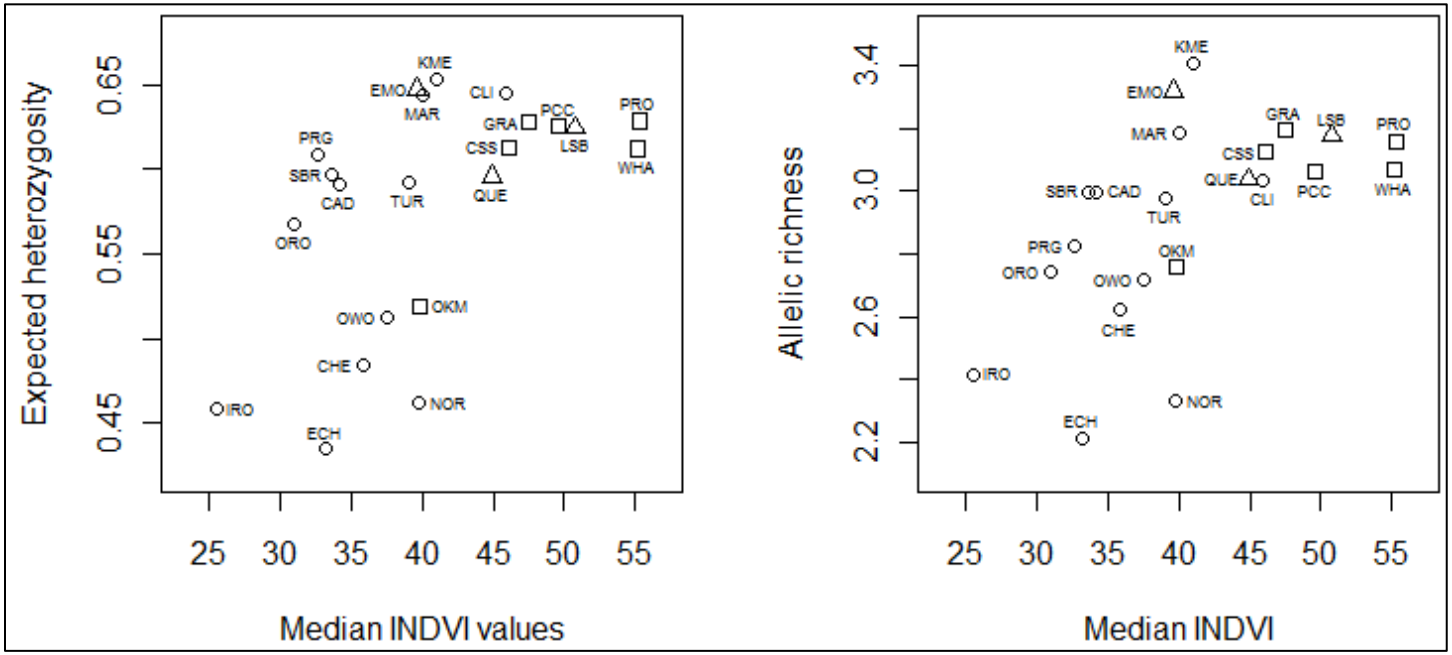


Figure A-13.5. Relation between genetic diversity (expected heterozygosity [H_e], left; allelic richness [A_r], right) and the Normalized Difference Vegetation Index (NDVI) for 22 populations of desert bighorn sheep in the Mojave Desert. NDVI was calculated as the median of growing-season integrated NDVI from 2001 through 2011. Patch abbreviations are in Figure 1. Patches that overlap Mojave National Preserve and Joshua Tree National Park are represented with squares and triangles, respectively; patches outside these parks are represented with circles. Three outliers (SGA, SGO, NSB) are not shown.

D. Data gaps and confidence in the assessment

On the basis of our results, we believe that at the seasonal level and in a limited geographic area, the Normalized Difference Vegetation Index may be a reliable surrogate measure of diet quality for desert bighorn sheep in the Mojave Desert. Use of NDVI to augment or replace measures of fecal nitrogen could dramatically improve the efficiency of assessments of diet quality of desert bighorn sheep in the Mojave Desert. Our results also support the retrospective application of NDVI to infer temporal trends in diet quality and associated probabilities of population persistence. The ability to efficiently characterize diet quality could facilitate the identification of promising locations for bighorn sheep reintroduction that remain unoccupied for reasons other than low-quality forage (e.g., anthropogenic disturbances, disease, low recolonization potential).

Relations between NDVI and fecal nitrogen were much weaker at the sample level than at the season level. At least two explanations are possible. First, random error in individual fecal-nitrogen samples may be reduced when samples are grouped at the season level. Second, fecal nitrogen samples were typically from a single group of bighorn sheep within a patch, and may only represent forage quality in the limited area used by that group shortly before sample collection, whereas our NDVI metrics were calculated for the entire patch.

Our results suggested that low levels of fecal nitrogen have limited resolution as an index of diet quality, and that NDVI may not support comparison of diet quality among geographically distant populations. Genetic diversity indexes had a considerably stronger association with connectivity than with NDVI, but several limitations of the analysis may have prevented detection of a stronger association with NDVI. The duration of our NDVI data may not have been long enough to fully characterize the long-term average and variability of NDVI in habitat patches. Eleven years represents less than two generations of bighorn sheep, too few to strongly affect genetic diversity, and our assumption that recent NDVI was representative of longer-term NDVI may not be warranted given regional climate change.

Three populations were clear outliers in our analysis of associations between NDVI and genetic diversity: San Gorgonio Mountains, San Gabriel Mountains, and North San Bernardino Mountains. These patches had much higher NDVI values than all other patches, but the populations had relatively low genetic diversity. All three are isolated, high-elevation patches at the periphery of the Mojave Desert, and consequently have different plant composition than patches in the center of the region. Chaparral and dense conifers in these three patches may increase NDVI but not bighorn sheep diet quality because these plant species are not used as forage. Moreover, bighorn sheep avoid such areas because of greatly increased predation risk (Risenhoover and Bailey 1985, Wakelyn 1987), so habitat in those patches is quite limited (Sawyer et al. 2011). Thus, in these patches, NDVI had a different relation with diet quality than in the central Mojave.

The relations we observed between fecal nitrogen and NDVI were saturated at both the sample and season levels: beyond a threshold, further increases in NDVI were associated with negligible increases in fecal nitrogen. This saturation might imply that bighorn sheep are able to maximize their diet quality even at intermediate levels of NDVI. However, this pattern also could reflect consumption of browse by bighorn sheep in the Sonoran-influenced Orocopia Mountains that was

high in phenolic compounds, which reduce protein digestion and inflate fecal nitrogen. The effects of climate change on bighorn sheep may be mitigated if the species can maximize diet quality at intermediate NDVI. Increased aridity may not decrease probabilities of population persistence in patches that currently have relatively high NDVI. Conversely, if precipitation in some areas increases, probabilities of persistence may not increase if bighorn sheep already have maximized diet quality at current NDVI.

The genetic diversity analysis provided more striking evidence that very high NDVI may not necessarily benefit bighorn sheep. Wehausen (2005) documented that the lamb:ewe ratio and spring diet quality were positively related to diet quality up to a threshold, and negatively related beyond the threshold. He suggested that abundant precipitation increases both diet quality of bighorn sheep and abundances of insect populations that serve as disease vectors, such that years with highest diet quality also have highest disease incidence. Alternatively, areas with relatively high average NDVI may support other prey species (e.g., mule deer) that attract predators on bighorn sheep, such as mountain lions. Another possibility is that areas with relatively high NDVI historically were used intensively by domestic livestock that transmitted diseases to bighorn sheep. These parasite- or predator-mediated increases in the mortality of bighorn sheep could have reduced genetic diversity.

The relations we found may not be applicable beyond the range of NDVI values or the spatial extent included in our analysis. Additionally, comparisons of NDVI among distant populations with different climates and plant communities may lead to erroneous inferences about relative diet quality. Thus, we recommend that among-patch comparisons on the basis of NDVI be restricted to habitat patches in Mojave National Preserve and Joshua Tree and nearby patches. Additional analyses would be needed to assess whether the same relations apply to Lake Mead or Death Valley

Appendix 14. Sources of data relevant to effects of climate change and drought on water sources for desert bighorn sheep, desert tortoises, and kangaroo rats in the Mojave Desert Network

Except where noted, all sources are publicly available and data are spatial. AGDF, Arizona Game and Fish Department; CDFW, California Department of Fish and Wildlife; NPS, National Park Service; BLM, Bureau of Land Management; USFWS, United States Fish and Wildlife Service; USGS, United States Geological Survey. N/A, not applicable.

Table A-14.1. Sources of data relevant to effects of climate change and drought on water sources for desert bighorn sheep, desert tortoises, and kangaroo rats in the Mojave Desert Network.

Resource	Data Source	Description	Spatial extent	Originator	URL	Type	Last updated	Temporal extent of data
Bighorn sheep or desert bighorn sheep	California Wildlife Habitat Relationships System	Map of the distribution of bighorn sheep	California	CDFW	www.dfg.ca.gov/biogeodata/cwhr/cwhr_downloads.asp#CWHR_GIS_Data	vector	1995	1988–1990
	Death Valley National Park Resource Condition Geodatabase	Habitat for desert tortoise and Sierra Nevada bighorn sheep as designated by the USFWS (polygon)	Death Valley	NPS	irma.nps.gov/App/Reference/DownloadDigitalFile?code=454321&file=ResourceCondition_DEVA.zip	vector	2011	N/A
	Death Valley National Park Resource Condition Geodatabase	Projected habitat for desert bighorn sheep. Based on ground-truthed topographic maps from before 1988 that identified elevations over 2000 m (3600') where bighorn sheep were present	Arizona Strip	BLM	www.blm.gov/az/st/en/prog/maps/gis_files.html	vector	2007	N/A
	Faunmap	Generalized national distribution of desert bighorn sheep during the late Quaternary (Burns et al. 2003)	Mojave Desert Network	Illinois State Museum	www.museum.state.il.us/research/faunmap/	.txt	1995	N/A
	HabiMap Arizona	Predicted distribution of desert bighorn sheep	Arizona	AGFD	habimap.org/habimap/	map only	2011	N/A
	Joshua Tree National Park Resource Condition Geodatabase	Habitat for desert tortoise and peninsular bighorn sheep as designated by the USFWS	Joshua Tree	NPS	irma.nps.gov/App/Reference/DownloadDigitalFile?code=454321&file=ResourceCondition_JOTR.zip	vector	2011	N/A
	Manzanar National Historic Site Resource Condition Geodatabase	Habitat for Sierra Nevada bighorn sheep as designated by the USFWS (polygon)	Manzanar	NPS	irma.nps.gov/App/Reference/DownloadDigitalFile?code=454321&file=ResourceCondition_MANZ.zip	vector	2011	N/A
	USGS Mammal Inventory of the Mojave Network	Abundance and population status of desert bighorn sheep in each Mojave Desert Network park	Mojave Desert Network	USGS	pubs.usgs.gov/of/2008/1167/of2008-1167.pdf	.doc	2008	2002–2004
Desert tortoises	California Wildlife Habitat Relationships System Desert Tortoise Distribution Map	Distribution of desert tortoise	California	CDFW	www.dfg.ca.gov/biogeodata/cwhr/cwhr_downloads.asp#CWHR_GIS_Data	vector	1998	1988–1990
	California Wildlife Habitat Relationships System Desert Tortoise Distribution Map	Critical habitat for desert tortoise in northern Arizona	Arizona strip	BLM	www.blm.gov/az/st/en/prog/maps/gis_files.html	raster	1994	N/A
	Death Valley National Park Resource Condition Geodatabase	Habitat for desert tortoise and Sierra Nevada bighorn sheep (polygon)	Death Valley	NPS	irma.nps.gov/App/Reference/DownloadDigitalFile?code=454321&file=ResourceCondition_DEVA.zip	vector	2011	N/A
	Desert Tortoise Habitat Categories	Desert tortoise habitat as defined by 1988 rangewide Desert Tortoise Plan	Arizona strip	BLM	www.blm.gov/az/st/en/prog/maps/gis_files.html	raster	1988	N/A
	Desert Tortoise Recovery Data Explorer	Spatial data explorer including threat layers, recovery action layers, and base layers for revised desert tortoise recovery plan	Mojave Desert Network	University of Redlands	www.spatial.redlands.edu/dtro/dataexplorer/	non-spatial data (map only)	2009	N/A

Table A-14.1 (continued). Sources of data relevant to effects of climate change and drought on water sources for desert bighorn sheep, desert tortoises, and kangaroo rats in the Mojave Desert Network.

Resource	Data Source	Description	Spatial extent	Originator	URL	Type	Last updated	Temporal extent of data
Desert tortoises (continued)	Desert Tortoise Recovery Model Explorer	Model explorer including threats, stresses, population effects, and recovery actions for threat layers for desert tortoise revised recovery plan	Mojave Desert Network	University of Redlands	www.spatial.redlands.edu/dtro/modelexplorer/	non-spatial data (map only)	2010	N/A
	Habimap Mojave Desert Tortoise Distribution Map	Projected distribution of Mojave desert tortoise	Arizona	AGFD	habimap.org/habimap/	non-spatial data (map only)	2011	N/A
	Joshua Tree National Park Resource Condition Geodatabase	Threatened and endangered habitat for desert tortoise as designated by USFWS (polygon)	Joshua Tree	NPS	irma.nps.gov/App/Reference/DownloadDigitalFile?code=454321&file=ResourceCondition_JOTR.zip	vector	2011	N/A
	Lake Mead National Recreation Area Resource Condition Geodatabase	Threatened and endangered habitat for desert tortoise as designated by USFWS (polygon)	Lake Mead	NPS	irma.nps.gov/App/Reference/DownloadDigitalFile?code=454321&file=ResourceCondition_LAKE.zip	vector	2011	N/A
	Mojave National Preserve Resource Condition Geodatabase	Threatened and endangered habitat for desert tortoise as designated by USFWS (polygon)	Mojave National Preserve	NPS	irma.nps.gov/App/Reference/DownloadDigitalFile?code=454321&file=ResourceCondition_MOJA.zip	vector	2011	N/A
	Parashant National Monument Resource Condition Geodatabase	Threatened and endangered habitat for desert tortoise as designated by USFWS (polygon)	Grand Canyon – Parashant	NPS	irma.nps.gov/App/Reference/DownloadDigitalFile?code=454321&file=ResourceCondition_PARA.zip	vector	2011	N/A
	Sensitive plant and animal species of western Mojave Desert	Description of desert tortoise abundance and distribution in the western Mojave Desert	western Mojave Desert Network	USGS	www.werc.usgs.gov/ProductDetails.aspx?ID=2576	.doc	2002	N/A
	Suitable Desert Tortoise Habitat (az_tortoise)	Desert tortoise habitat in Arizona. Derived from a model that was generated by the BLM and incorporated elevation, terrain, climate, and vegetation	Arizona	BLM	www.blm.gov/az/st/en/prog/maps/gis_files.html#statewide	vector	1992	N/A
	USFWS Critical Habitat for Desert Tortoise	Polygon of critical habitat for desert tortoise; based on description in the Federal Register	Mojave Desert Network	USFWS	criticalhabitat.fws.gov/crithab/	vector	2010	N/A
	USFWS Desert Tortoise Recovery Office Annual Monitoring Data 2001–2010	Sampling data including spatial and non-spatial attributes of monitoring areas, spatial and non-spatial attributes of transects, spatial and non-spatial data on live and dead desert tortoises, and characterization of focal tortoises on the basis of radio transmitters used to estimate the proportion of tortoises available for detection on transects (data must be requested)	Mojave Desert Network	USFWS Desert Tortoise Recovery Office	www.mojavedata.gov/deserttortoise_gov/recovery/data.php; data must be requested	spatial (raster, vector) and non-spatial data	2010	2001–2010
Kangaroo rats	AVHRR Normalized Difference Vegetation Index (NDVI) Composites	AVHRR 7- or 14-day composite Normalized Difference Vegetation Index, 1-km resolution	Mojave Desert Network	USGS	ivm.cr.usgs.gov/products.php	raster	2013	1989–present
	California Wildlife Habitat Relationships System	Distribution of kangaroo rats (*Dipodomys ordii*, *microps*, *panamintinus*, *deserti*, and *merriami*)	California	CDFG	www.dfg.ca.gov/biogeodata/cwhr/cwhr_downloads.asp#CWHR_GIS_Data	vector	1995	1988–1990
	Faunmap	Generalized distributions of kangaroo rats (*Dipodomys panamintinus*, *ordii*, *deserti*, and *merriami*) across the United States during the late Quaternary	Mojave Desert Network	Illinois State Museum	www.museum.state.il.us/research/faunmap/	non-spatial data (*.txt; transferrable to GIS via *.csv)	1995	N/A

Table A-14.1 (continued). Sources of data relevant to effects of climate change and drought on water sources for desert bighorn sheep, desert tortoises, and kangaroo rats in the Mojave Desert Network.

Resource	Data Source	Description	Spatial extent	Originator	URL	Type	Last updated	Temporal extent of data
Kangaroo rats (continued)	Habimap	Projected distribution of banner-tailed kangaroo rat	Arizona	AGFD	habimap.org/habimap/	non-spatial data (map only)	2011	N/A
	Habimap	Projected distribution of House Rock Valley chisel-toothed kangaroo rat	Arizona	AGFD	habimap.org/habimap/	non-spatial data (map only)	2011	N/A
Vegetation	USGS	Description of the abundance and status of mammals, including kangaroo rats, in each Mojave Desert Network park	Mojave Desert Network	USGS	pubs.usgs.gov/of/2008/1167/of2008-1167.pdf	non-spatial data (*.doc)	2008	2002–2004
	Airborne Visible/ Infrared Imaging Spectrometer (AVIRIS) images	–	portions of the Mojave Desert Network	NASA Jet Propulsion Laboratory	aviris.jpl.nasa.gov/data/get_aviris_data.html	raster	as needed	1992–2006
	Global Inventory Modeling and Mapping Studies (GIMMS)	NDVI, derived from AVHRR images	Mojave Desert Network	Global Land Cover Facility	glcf.umd.edu/data/gimms/	raster	2006	1981–2006
	Landscape Fire and Resource Management Planning Tools Project (LANDFIRE)	Potential vegetation as estimated on the basis of biophysical variables	Mojave Desert Network	LANDFIRE	www.landfire.gov/NationalProductDescriptions19.php	raster	2010	2008
	MODIS	MOD13. 16-day NDVI data derived from MODIS bands 1 and 2. 250-m resolution.	Mojave Desert Network	NASA	modis.gsfc.nasa.gov/data/dataprod/dataproducts.php?MOD_NUMBER=13	raster	2013	2001–2013
	Mojave Desert Vegetation for Desert Renewable Energy Conservation Plan	Fine-resolution vegetation map of a portion of the western Mojave Desert in California	California	CDFW	imaps.dfg.ca.gov/viewers/biospublic/app.asp?zoomtoBookmark=3396	map; other data in development	2012, data products in development	2010–2012
	Vegetation Drought Response Index	Based on data from eMODIS or AVHRR. Geospatial model of drought stress on vegetation within the conterminous United States. 1-week resolution.	Mojave Desert Network	USGS –EROS, High Plains Regional Climate Center, US Department of Agriculture	vegdri.cr.usgs.gov/viewer/viewer.htm	map, other data in development	2013	2008–2013
Water	Lake Mead National Recreation Area Resource Condition Geodatabase	500-m buffer around wetlands designated by USFWS National Wetlands Inventory, including lakes, riverine areas, and ponds	Lake Mead	NPS	irma.nps.gov/App/Reference/DownloadDigitalFile?code=454321&file=ResourceCondition_LAKE.zip	vector	2013?	N/A
	Manzanar National Historic Site Resource Condition Geodatabase	500-m buffer around wetlands designated by USFWS National Wetlands Inventory, including lakes, riverine areas, and ponds	Manzanar	NPS	irma.nps.gov/App/Reference/DownloadDigitalFile?code=454321&file=ResourceCondition_MANZ.zip	vector	2013?	N/A
	Mojave National Preserve Resource Condition Geodatabase	500-m buffer around wetlands designated by USFWS National Wetlands Inventory, including lakes, riverine areas, and ponds	Mojave Desert Network	NPS	irma.nps.gov/App/Reference/DownloadDigitalFile?code=454321&file=ResourceCondition_MOJA.zip	vector	2013?	N/A
	Nevada Water Development Atlas	Topographic maps of 1500 developed water sources (guzzlers), including coordinates in Universal Transverse Mercator units	Nevada	NDOW	www.ndow.org/Nevada_Wildlife/Guzzler_Information/	non-spatial data (map only) and spatial data (vector)	2013	N/A
	Parashant National Monument Resource Condition Geodatabase	500-m buffer around wetlands designated by USFWS National Wetlands Inventory, including lakes, riverine areas, and ponds	Grand Canyon – Parashant	NPS	irma.nps.gov/App/Reference/DownloadDigitalFile?code=454321&file=ResourceCondition_PARA.zip	vector	2013?	N/A

Table A-14.1 (continued). Sources of data relevant to effects of climate change and drought on water sources for desert bighorn sheep, desert tortoises, and kangaroo rats in the Mojave Desert Network.

Resource	Data Source	Description	Spatial extent	Originator	URL	Type	Last updated	Temporal extent of data
Water (continued)	Riparian Areas and Semi-wet Meadows (wetlands)	Wetlands classified as either riparian or semi-wet meadows	Arizona strip	BLM	www.blm.gov/az/st/en/prog/maps/gis_files.html	raster	2007	N/A
	Spring Stewardship Institute Springs Inventory Database	Springs inventory data including location (categorical and spatial) and survey information: date, time, microhabitat attributes, soils (substrate percentages, litter, wood cover), geology, solar pathfinder data and subsequent calculations, water flow, water quality, vegetation cover, invertebrates present, vertebrates (observations or other evidence, such as scat, tracks, feathers, nests), images (sketches, photographs), scores for condition of and risk to site, cultural values, quality of data collection and data entry	Arizona	Springs Stewardship Institute	springstewardship.org/inventory.html	non-spatial (*.mdb)	2013	N/A
	Spring Stewardship Institute Springs Inventory Database	Springs inventory data including location (categorical and spatial) and survey information: date, time, microhabitat attributes, soils (substrate percentages, litter, wood cover), geology, solar pathfinder data and subsequent calculations, water flow, water quality, vegetation cover, invertebrates present, vertebrates (observations or other evidence, such as scat, tracks, feathers, nests), images (sketches, photographs), scores for condition of and risk to site, cultural values, quality of data collection and data entry	Arizona	Springs Stewardship Institute	springstewardship.org/inventory.html	non-spatial (*.mdb)	2013	N/A
	Springs Inventory Data Compilation	Locations of springs in the Mojave Desert Network with categorical location (e.g. drainage, township), spatial location, water quality (e.g. water depth, dissolved oxygen, temperature), and notes on disturbance (e.g. from livestock)	Death Valley, Grand Canyon–Parashant, Joshua Tree, Mojave National Preserve, Lake Mead	NPS	access restricted to NPS	non-spatial data (*.xlsx), spatial data (vector)	2012	N/A
	USGS National Atlas, Streams	Major streams and rivers across the United States and territories (1:1,000,000 cartographic scale)	Mojave Desert Network	USGS	nationalatlas.gov/mld/1strmsl.html	vector	2012	N/A
	USGS National Atlas, Waterbodies and Wetlands	Major lakes, reservoirs, and wetlands (including swamps and marshes) across the United States and territories (1:1,000,000 cartographic scale)	Mojave Desert Network	USGS	nationalatlas.gov/mld/1lakesp.html	vector	2012	N/A

The Department of the Interior protects and manages the nation's natural resources and cultural heritage; provides scientific and other information about those resources; and honors its special responsibilities to American Indians, Alaska Natives, and affiliated Island Communities.

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