4.2.3 Vegetation Communities: Current Distribution and Status

Vegetation Communities Management Questions

1. Where are existing vegetation communities, and what is their status?
2. What change agents have affected existing vegetation communities?

Two coarse filter vegetation communities plus riparian vegetation were evaluated for the Sonoran Desert ecoregion. For the specific vegetation communities, two different sources of data were compiled (LANDFIRE EVT v1.1 and NatureServe Landcover v2.7) to depict current distribution (Figure 4-16).

Besides the differences in classes mapped, the area covered for each vegetation community type according to the two classifications differed to varying degrees (Table 4-5**)**. While a visual inspection of the two shows each vegetation community in approximately the same general locations, the actual pixel-to-pixel agreement was only fair, with percent overlaps from 40 to slightly over 50 percent. Even though there are notable differences between the two classification systems, it is more appropriate to acknowledge the differences and choose the one most meaningful for a particular purpose than to attempt to hybridize the two into a single product.

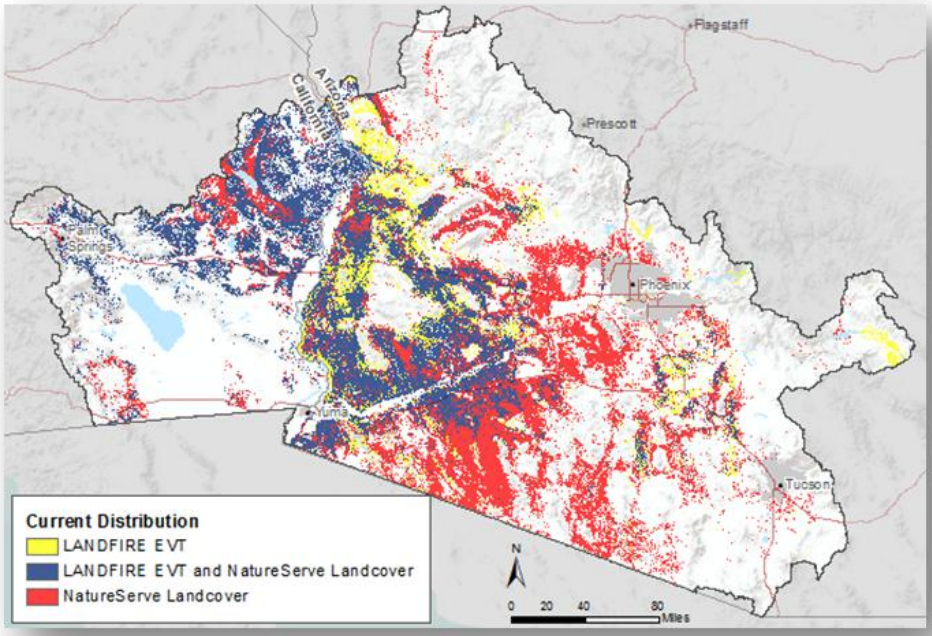
Evaluating current status for each vegetation community is challenging in several ways. Vegetation communities are dynamic over time and space, demonstrating a degree of fluidity especially along ecotonal boundaries driven by the pattern and timing of climate and natural and human disturbance. Specific plant communities are not fixed on the landscape as individual site histories dictate what community is expressed at a particular time. A fairly long history of human disturbance of the natural landscape from water management, invasive species, and grazing has had a profound impact on the native vegetation communities.

To address questions of historic change, LANDFIRE Biophysical Settings (BpS) data, modeled presettlement vegetation, was used as the reference condition. Biophysical settings provide a spatially explicit estimate of what vegetation communities would likely occur in a specific location based on physical conditions (e.g., soils, elevation, aspect, moisture, and natural fire regime). Because it is a model, any strict alignment with current distribution (i.e. LANDFIRE EVT) should not be expected. For example, the BpS and EVT maps for the creosotebush-white bursage and palo verde-mixed cacti vegetation communities show considerable overlap but also some differences (Figure 4-17). It is reasonable to assume that some of these differences represent conversion of this community type to other land uses. Overlaying current urban and agriculture land uses, roads, invasive vegetation, and uncharacteristic native vegetation against LANDFIRE BpS data highlights possible areas of change from historic reference condition for both matrix communities (Figure 4-18).

Table 4-5. Area (in thousands of acres) comparison for vegetation communities between NatureServe Landcover v2.7 and LANDFIRE EVT v1.1.

Vegetation Community	NatureServe Only	LANDFIRE Only	Both	Percent Overlap
Sonora-Mojave Creosotebush-White Bursage Desert Scrub	5,361	1,417	4,823	41.6
Sonoran Paloverde-Mixed Cacti Desert Scrub	1,797	5,332	7,373	50.8
Riparian Vegetation	1,600			

Sonora-Mojave Creosotebush-White Bursage Desert Scrub



Sonoran Paloverde-Mixed Cacti Desert Scrub

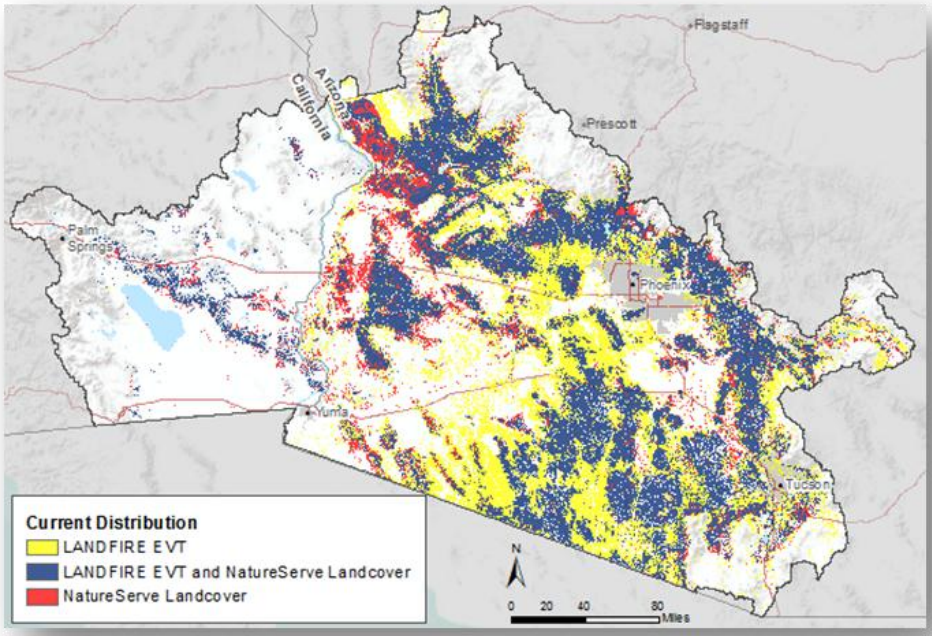


Figure 4-16. Map comparison between NatureServe Landcover v2.7 (in red) and LANDFIRE EVT v1.1 (in yellow) for the two matrix vegetation communities in the Sonoran Desert ecoregion. Areas common to both datasets are in blue.

A total of over 8.7 million acres (~37%) of the two natural vegetation communities mapped using LANDFIRE BpS were significantly changed in the ecoregion (Table 4-6). Changes due to invasive species conversion dominated the results, affecting over 5.2 million acres (Table 4-6). Conversion from urbanization and roads affected over 1.7 million acres and intensive agriculture (excluding grazing) converted over 1.1 million acres. The greatest total area changed (> 4.7 million acres or 30% of total BpS area) was in the Sonoran Paloverde-Mixed Cacti Desert Scrub class; however, the highest percent change was observed in Sonoran-Mojave Creosotebush-White Bursage Desert Scrub with 51% (>4 million acres) of its LANDFIRE BpS distribution converted by urbanization and roads, agriculture, and invasives.

More recent disturbances (approximately the last decade) such as fire, mechanical treatment, and other disturbances were analyzed in a similar fashion (Figure 4-19). A total area of about 395,000 acres (>1% of the combined area) was recently disturbed in the last decade (Table 4-7), mostly by fire (over 297,000 acres). Neither of the vegetation communities is well-adapted to fire resulting in a high probability that many of these burned areas will be later dominated by invasive grasses. Current distribution, historic change, and recent disturbance maps for each vegetation community are provided in Appendix B.

In addition to evaluating historic and recent disturbance to the matrix vegetation communities, which provides some insight into loss and recent disturbances, the status of the existing setting in which these communities currently occur was also evaluated. To do this, the current LANDFIRE EVT v 1.1 and NatureServe Landcover v 2.7 distributions for each community were overlaid against the current terrestrial landscape intactness model results. The assumption was that each natural vegetation community is affected in various ways based on the overall intactness of its immediate neighborhood.

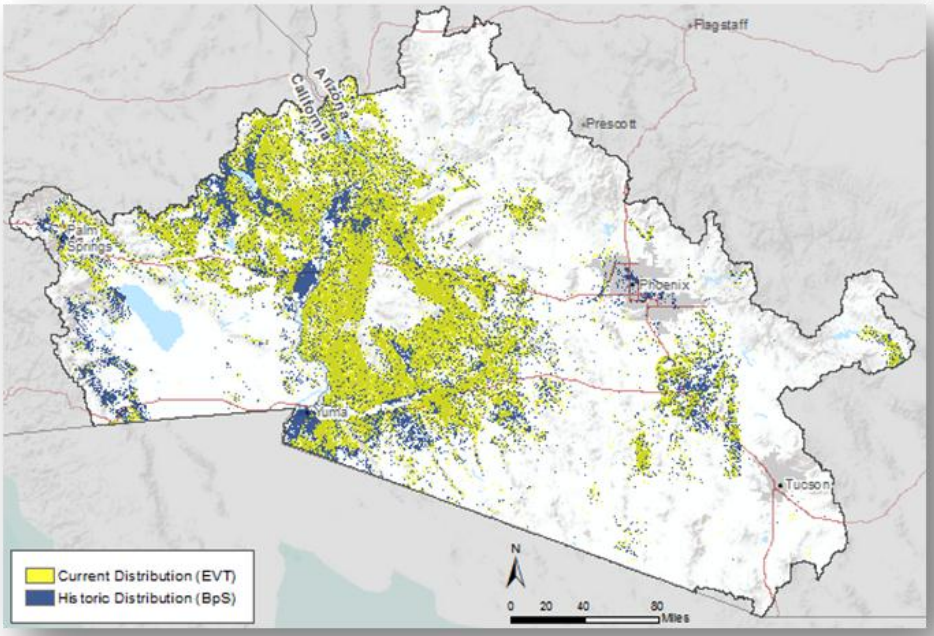
Status profiles for Sonoran-Mojave Creosotebush-White Bursage Desert Scrub for each classification were very similar and showed around 50% of the area in Moderately High or Moderately Low categories (Figure 4-20). The NatureServe version, which had more of this community mapped around Phoenix, had more of its area in the Low and Very Low categories. In either case, only around 10–14% of the area was contained in areas of Very High terrestrial landscape intactness, mostly in the northwest or south-central portion of the ecoregion. Results for Sonoran Paloverde-Mixed Cacti Desert Scrub in each classification system showed similar status profiles and better results than those for the Creosotebush-White Bursage community (Figure 4-21). Nearly 20% of this community had Very High intactness. LANDFIRE shows this community as absent in southeastern California, while it is known to be actually present there where it is a linear feature along large desert washes. These and other classification errors can be resolved at the local field office level.

4.2.3.1 Riparian Vegetation

Riparian ecological systems have undergone significant physical and biological changes throughout the ecoregion because of direct conversion to other uses; changes in natural flow regimes and suppression of fluvial processes (Stromberg 2001, Stromberg et al. 2007a); livestock grazing (Armour et al. 1994); and invasive species invasion (e.g., tamarisk, Stromberg et al. 2007b). As much as 90% of pre-settlement riparian ecosystems have been lost (LUHNA 2011).

Livestock grazing has damaged approximately 80% of stream and riparian ecosystems in the western US (Belsky et al. 1999). Grazing alters streamside morphology, increases sedimentation, degrades riparian vegetation through trampling and consumption and causes nutrient loading to the system. Invasive plants such as tamarisk often successfully out-compete native species, because tamarisk has high fecundity and it has been shown to be more tolerant to drought and flow alterations than natives (Stromberg et al. 2007a, Merritt and Poff 2010). For more details on riparian systems see the Tamarisk Case Study Insert that follows Section 4.3.

Sonora-Mojave Creosotebush-White Bursage Desert Scrub



Sonoran Paloverde-Mixed Cacti Desert Scrub

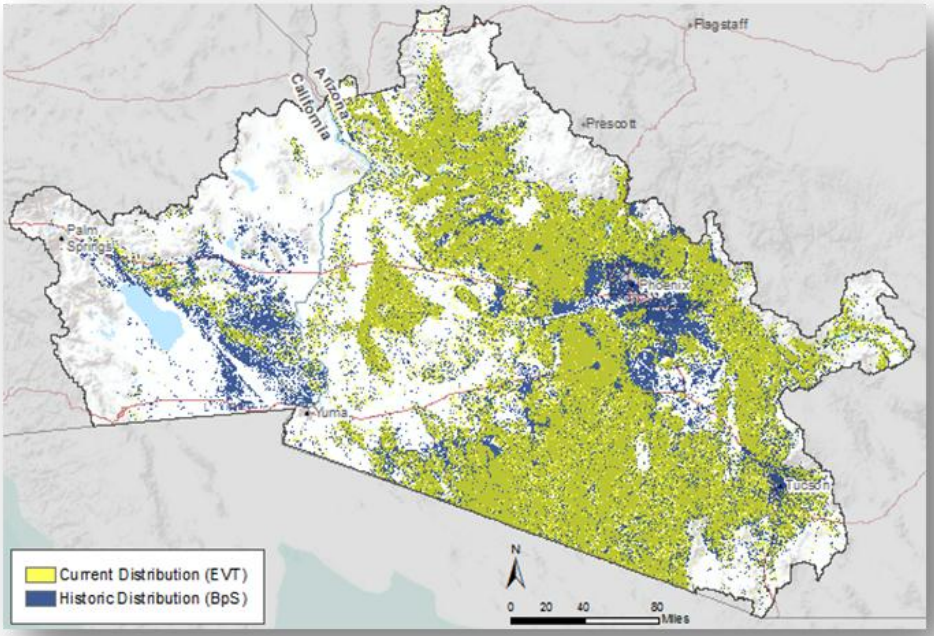
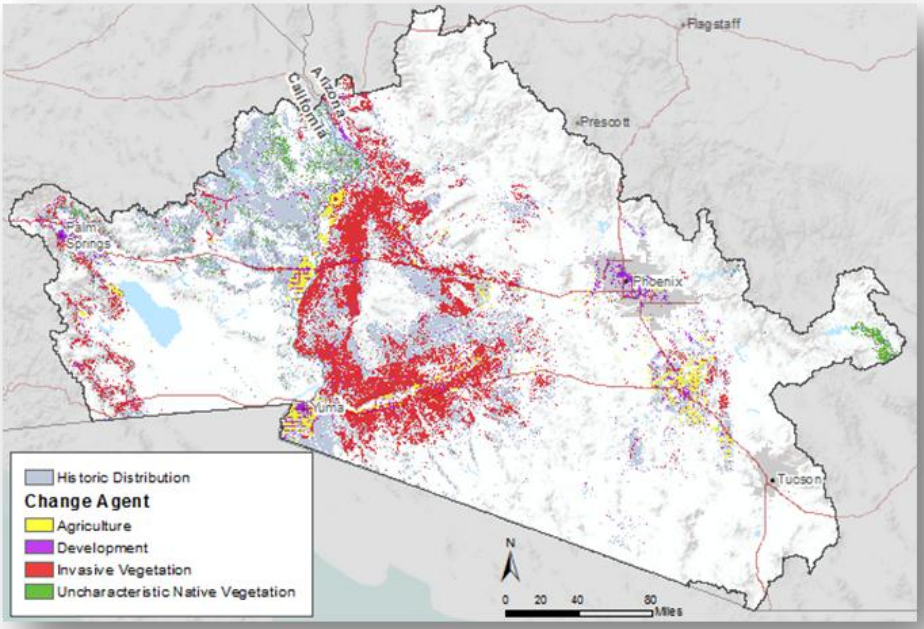


Figure 4-17. Comparison between LANDFIRE current distribution (EVT) and historic distribution (BpS) for Sonoran-Mojave creosotebush-white bursage desert scrub and Sonoran paloverde-mixed cacti desert scrub. Some of the differences between current distribution and modeled historic distribution may represent conversion of these community types to other land uses.

Sonora-Mojave Creosotebush-White Bursage Desert Scrub



Sonoran Paloverde-Mixed Cacti Desert Scrub

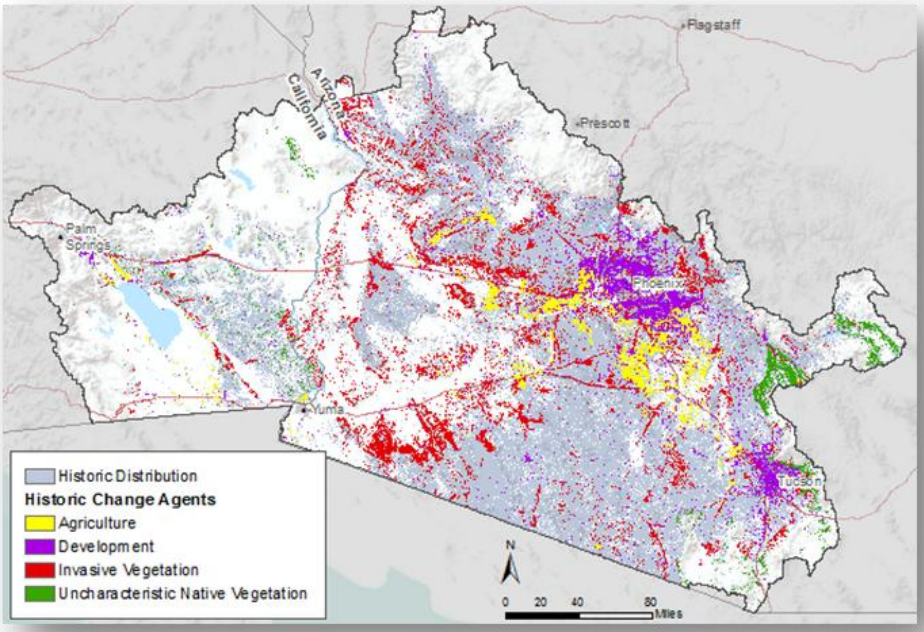
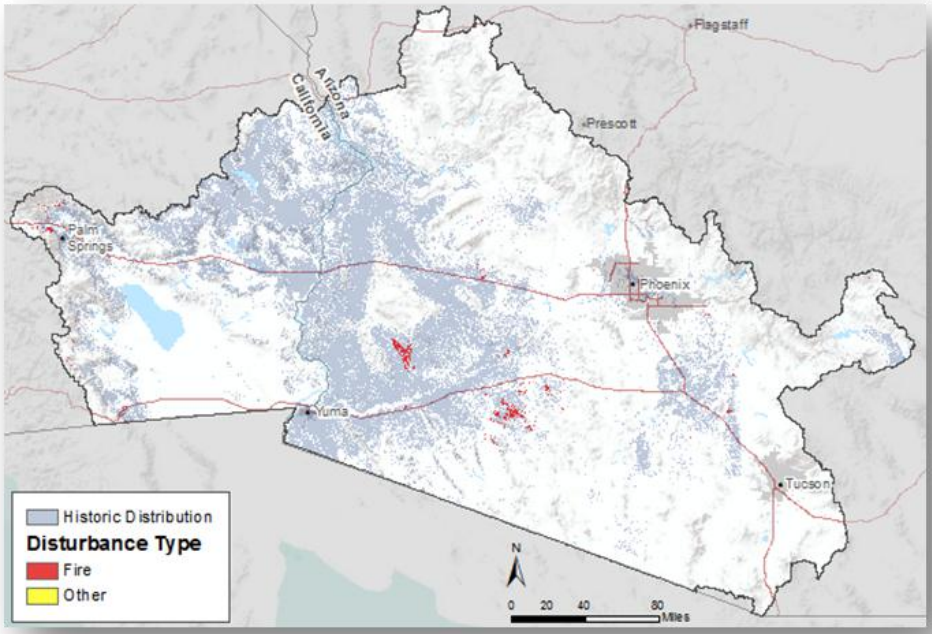


Figure 4-18. Conversion of major vegetation communities within the Sonoran Desert ecoregion. Overlaying current urban and agriculture land uses, roads, invasive vegetation, and uncharacteristic native vegetation against LANDFIRE BpS data (representing reference condition, in gray) highlights possible areas of conversion to different land cover from historic modeled reference condition for both matrix communities

Sonora-Mojave Creosotebush-White Bursage Desert Scrub



Sonoran Paloverde-Mixed Cacti Desert Scrub

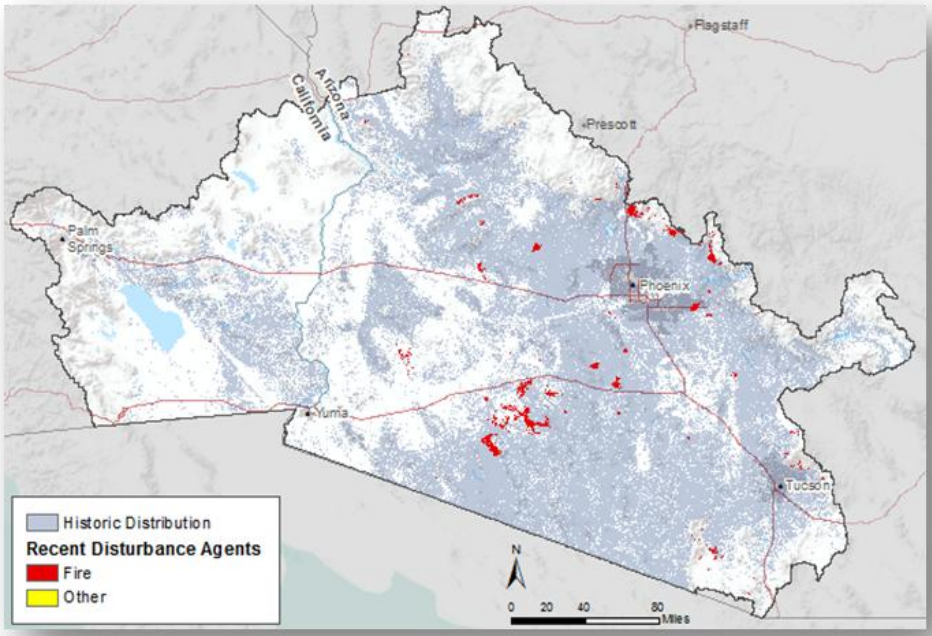


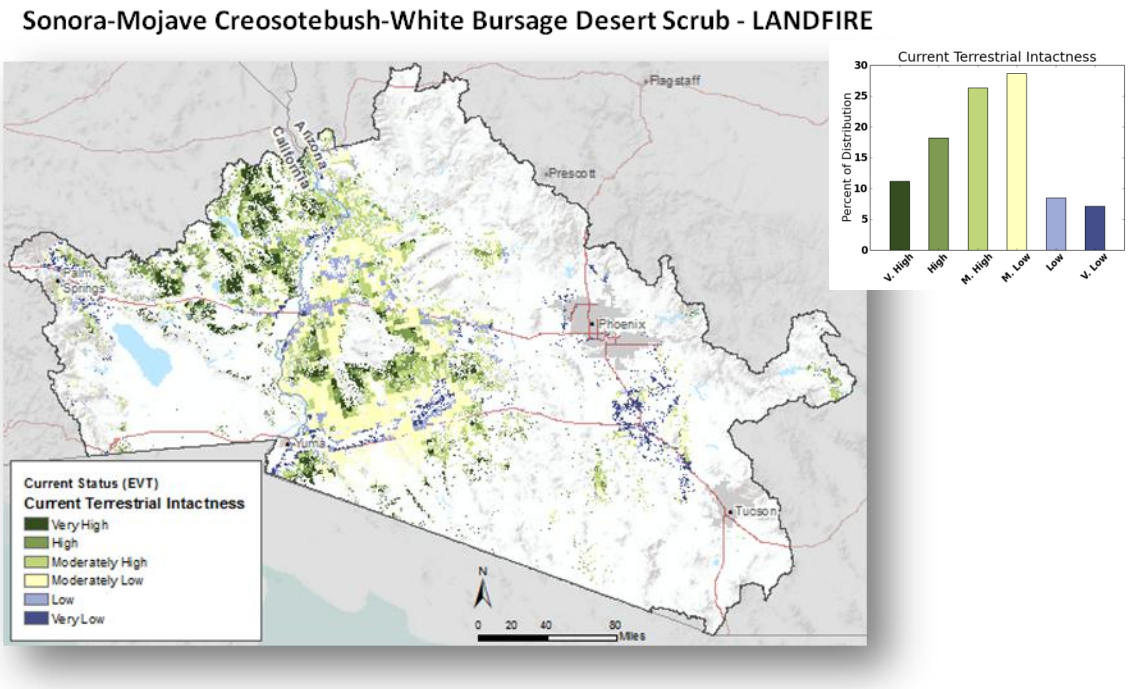
Figure 4-19. Recent disturbance (within the last decade) of major vegetation communities in the Sonoran desert ecoregion. A total area of about 298,000 acres (>1% of the combined area) was recently disturbed in the last decade in the ecoregion (Table 4-6), mostly by fire.

Table 4-6. Summary of area (in 1000s of acres) of historic change for each matrix vegetation community, comparing existing vegetation to LANDFIRE BpS data (representing reference condition). Acres represent conversion to different land cover from modeled presettlement vegetation.

Vegetation Community	Total BpS Area	Urban & Roads	Agriculture	Invasives	Unchar Native Veg	Total Changed	Percent
Sonoran-Mojave Creosotebush-White Bursage Desert Scrub	7,858	429	433	2,909	274	4,045	51.5%
Sonoran Paloverde-Mixed Cacti Desert Scrub	15,730	1,255	672	2,345	429	4,701	30%
Totals	**23,588**	**1,684**	**1,105**	**5,254**	**703**	**8,746**	

Table 4-7. Summary of area (1000s of acres) for each matrix vegetation community based on LANDFIRE BpS data that has been recently disturbed within the last decade in the Sonoran Desert ecoregion.

Vegetation Community	Total BpS Area	Fire	Mechanical	Other	Total Disturbed	Percent
Sonoran-Mojave Creosotebush-White Bursage Desert Scrub	7,858	85	0	80	165	1.1%
Sonoran Paloverde-Mixed Cacti Desert Scrub	15,730	212	0	18	230	1.4%
Totals	**23,588**	**297**	**0**	**98**	**395**	



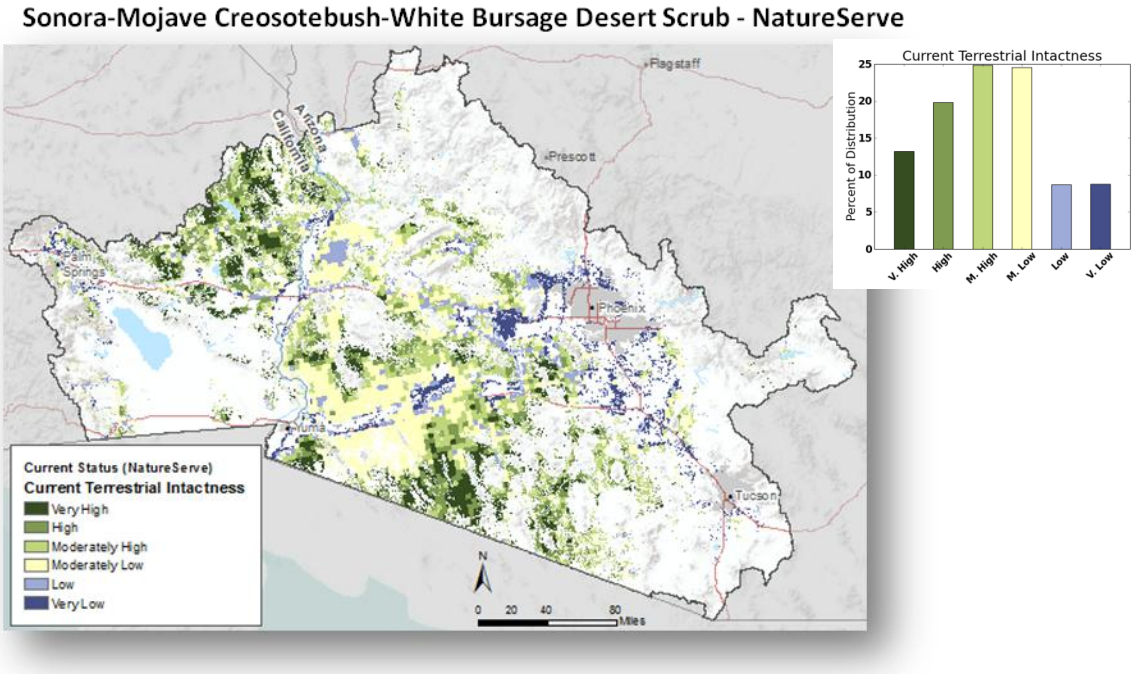
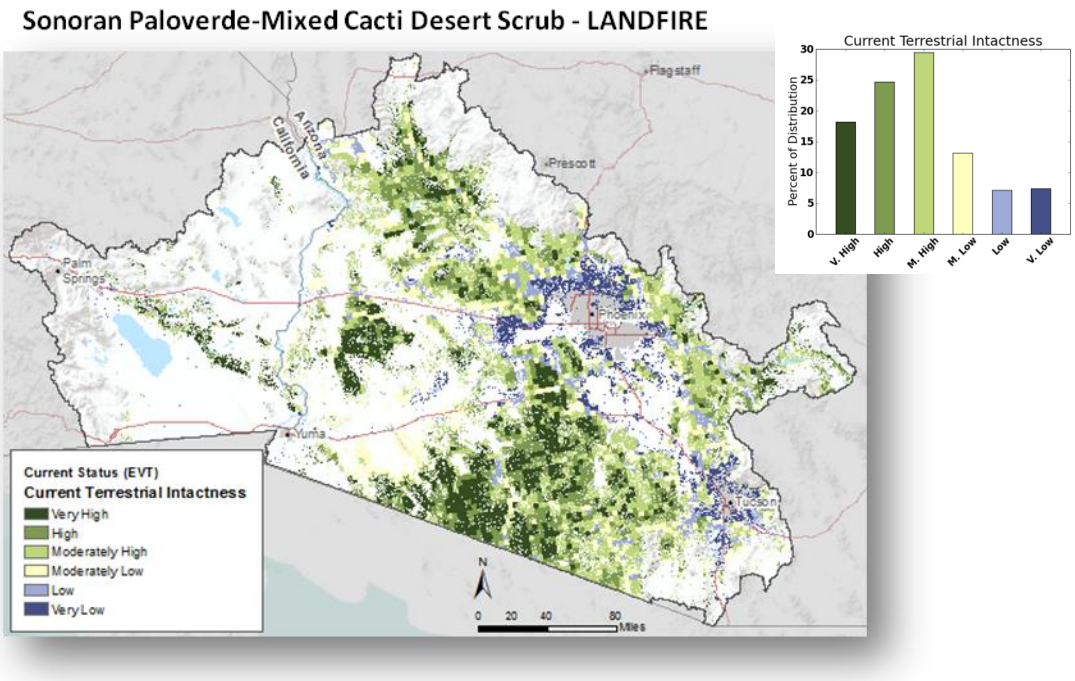


Figure 4-20. Current status for Sonoran-Mojave Creosotebush-White Bursage Desert Scrub for the Sonoran Desert ecoregion mapped by overlaying LANDFIRE existing vegetation (top) and NatureServe Landcover data (bottom) against current terrestrial intactness model results. The NatureServe version (bottom) with more of this community mapped in the Phoenix area had more of its distribution in the Low and Very Low categories.



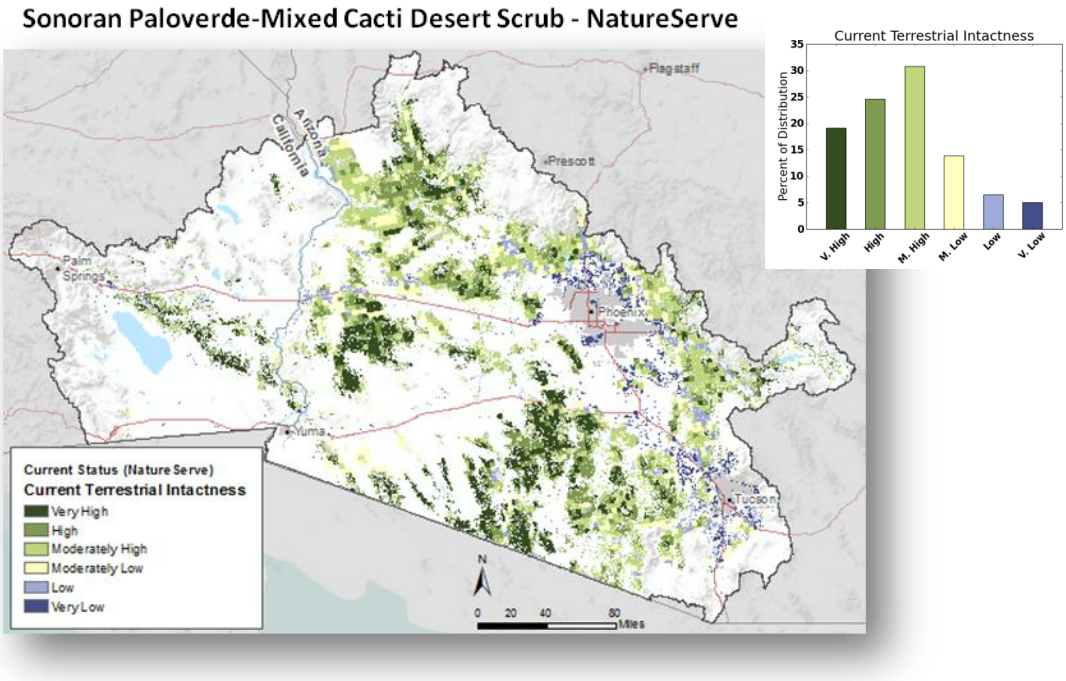


Figure 4-21. Current status for Sonoran Paloverde-Mixed Cacti Desert Scrub for the Sonoran Desert ecoregion mapped by overlaying LANDFIRE existing vegetation (top) and NatureServe Landcover data (bottom) against current terrestrial intactness model results.

Mapping riparian systems is difficult to do using satellite remote sensing. The narrow linear nature of the community makes it difficult to delineate with high levels of accuracy. For the REA assessment, NatureServe Landcover v2.7 was used to assess current distribution. Status was evaluated using the terrestrial landscape intactness results at 4km resolution. Use of the HUC as a reporting unit was considered and rejected for linear features such as riparian areas because of its lower resolution. According to the NatureServe Landcover data, 1.6 million acres of riparian vegetation currently exist in the ecoregion. Status results based on the terrestrial landscape intactness model shows that the dominant category is Moderately High with a significant number of acres at both extremes (Figure 4-22). Although a 4 km grid is an appropriate reporting unit for a region-wide assessment, it is less discriminating in characterizing linear communities. Future assessments should examine different analyses methods and reporting unit for linear features.

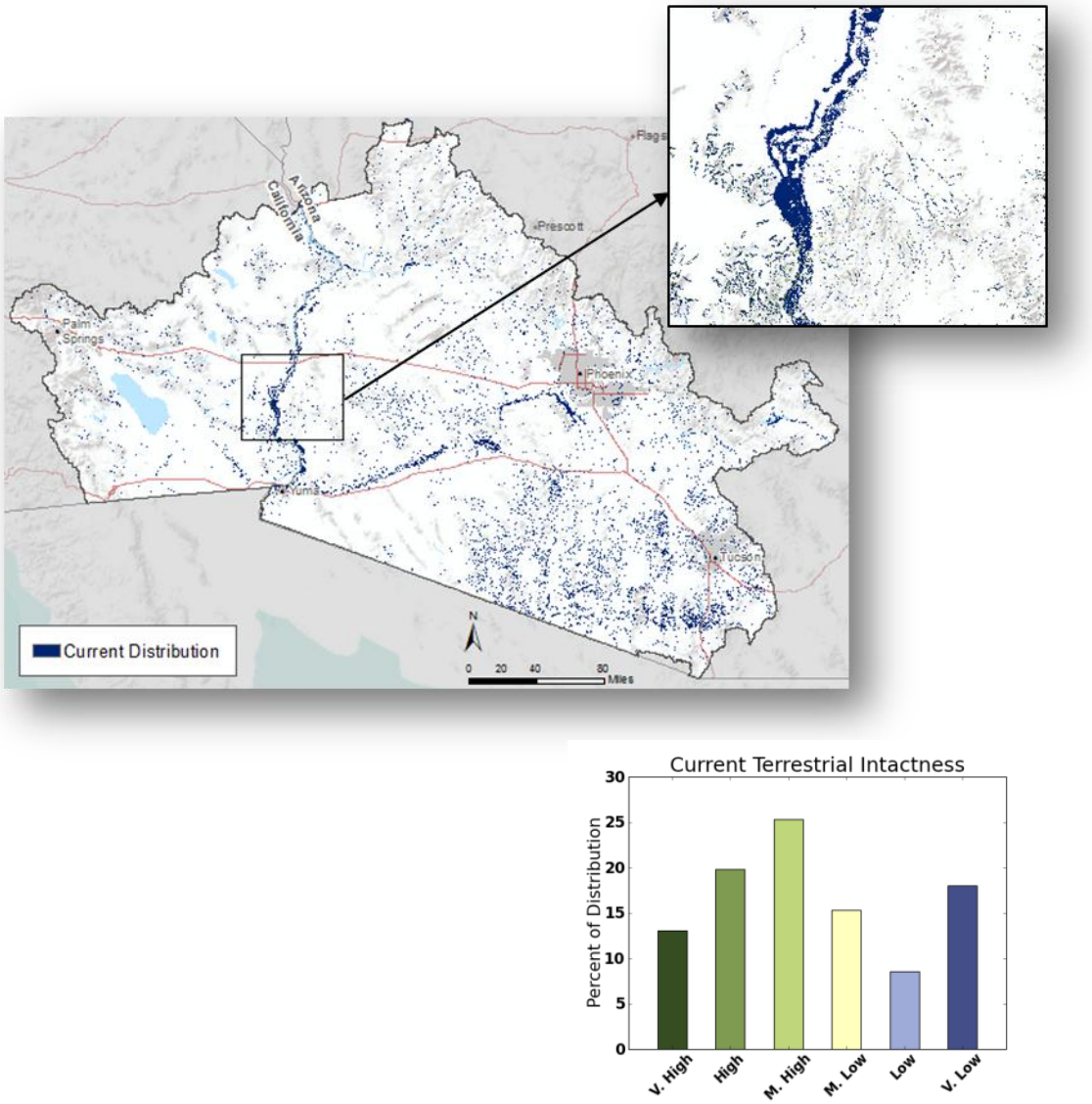


Figure 4-22. Map shows zoomed in portion of the riparian vegetation distribution (in blue) based on NatureServe Landcover v2.7 (inset) for the Sonoran Desert ecoregion. General status histogram accompanies map with percent of distribution in various classes calculated by overlaying NatureServe Landcover data against current terrestrial intactness model results.

4.2.4 Evaluating Designated Sites: Current Distribution and Status

Approximately 28% of the Sonoran Desert ecoregion (~9.2 million acres) is currently under federal, state, local government or private conservation land designation, including conservation easements (Figure 4-23). These data are limited to designated protected lands and do not include other conservation lands under current land management plans by the various agencies. In some instances, these land designations are nested, in which case the more protected designation is displayed over the top of another (e.g. wilderness area above a national park). Approximately 832 miles of wild and scenic rivers and national trails are also included in the map.

The status of these lands was evaluated by overlaying the designated land polygons over the top of terrestrial landscape intactness and summarizing the results (Figure 4-24). Wilderness Areas made up the largest proportion of the protected areas followed by Areas of Critical Environmental Concern. Other categories occupying over 500,000 acres include Other Protected Lands, State Parks, and National Monuments. Combined sites totaling between 100,000–500,000 acres include National Wildlife Refuges, National Parks, and Roadless Areas. Designations with less than 100,000 acres include Wilderness Study Areas, National Conservation Areas, and State Wildlife Management Areas. A table of total area (acres) for each status category for all designated lands in the Sonoran Desert ecoregion is located in Appendix A.

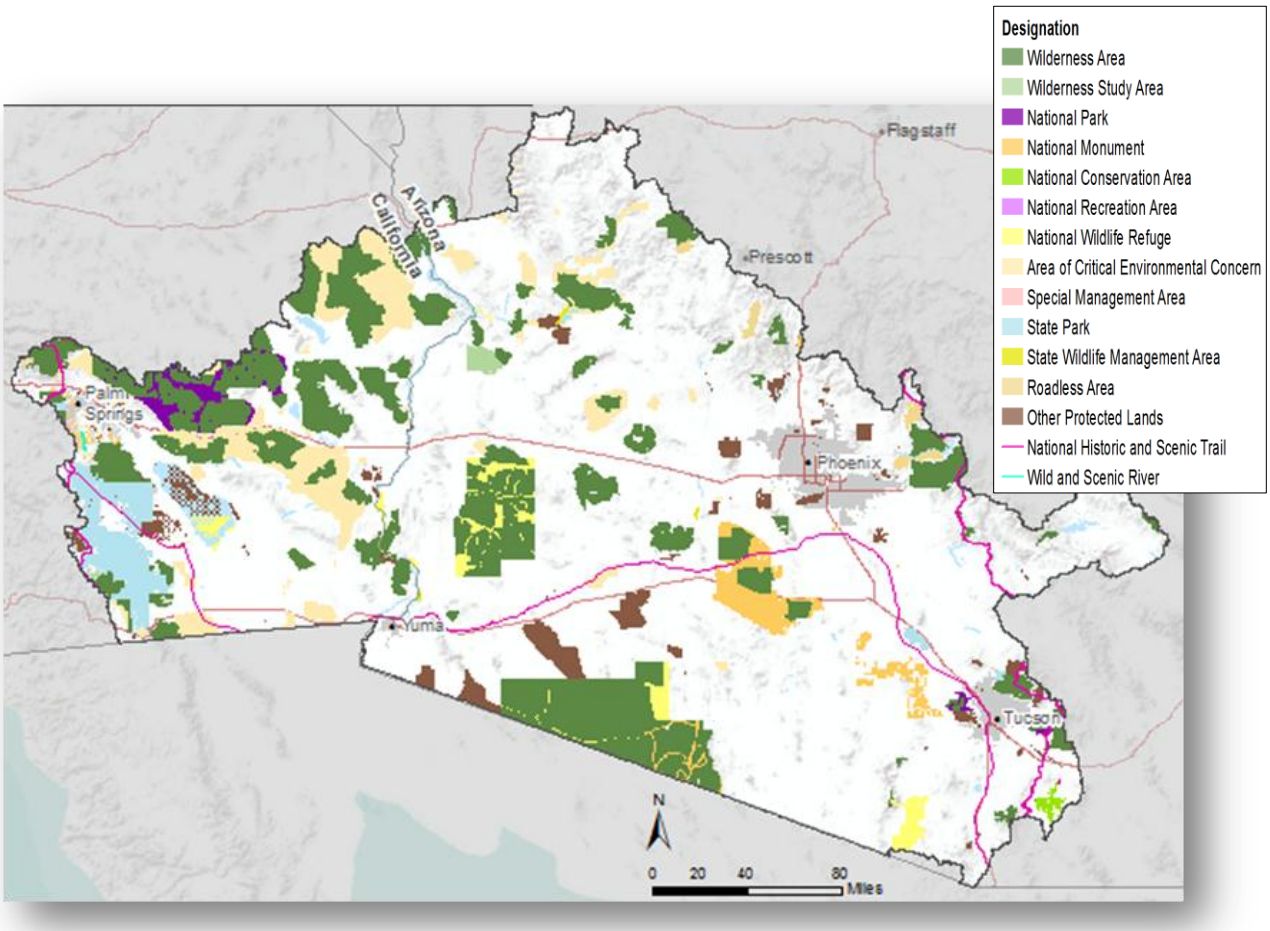
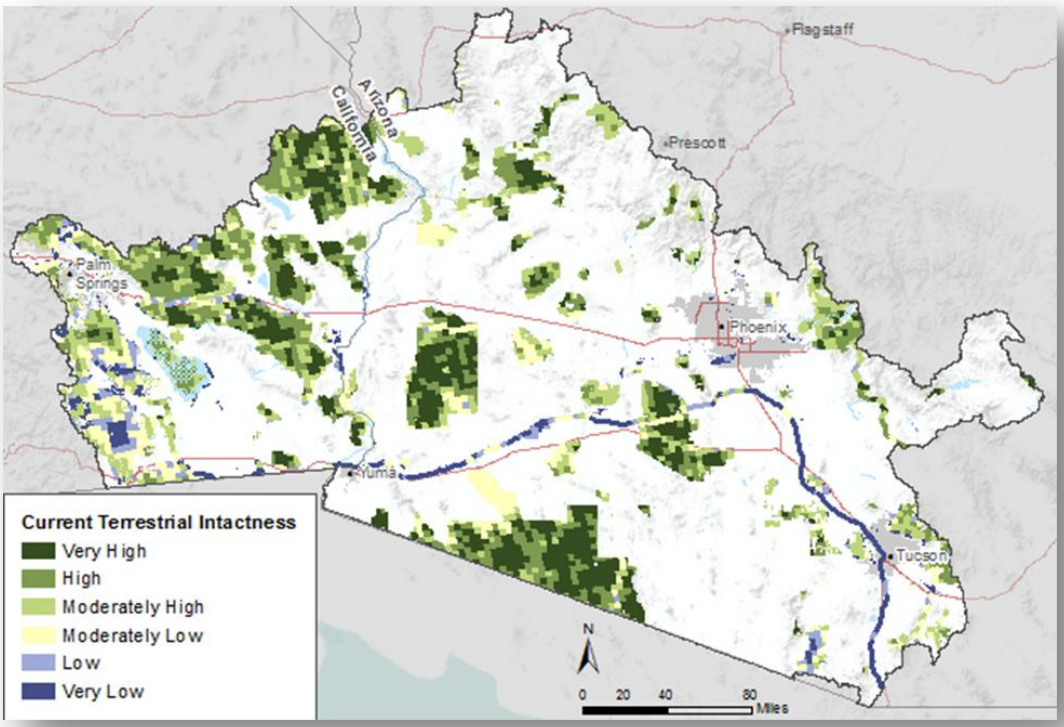


Figure 4-23. Map of designated protected lands in the Sonoran Desert ecoregion.

In general, terrestrial landscape intactness for special designated lands was heavily skewed (>75% of the area) towards more intact landscapes as one would expect; however, not all designation classes scored the same (histograms Figure 4-25). Wilderness Areas, National Wildlife Refuges, and National Monuments showed the best intactness profiles. National Parks and Areas of Critical Environmental Concern also did well. Roadless Areas, National Conservation Areas, and Wild and Scenic Rivers had similar profiles and peaked at Moderately High intactness. Wilderness Study Areas were dominated by Moderately Low intactness, which was surprising. The remaining designation types (National Historic and Scenic Trails, State Parks, State Wildlife Management Areas, and Other Protected Lands) possessed the lowest intactness profiles.



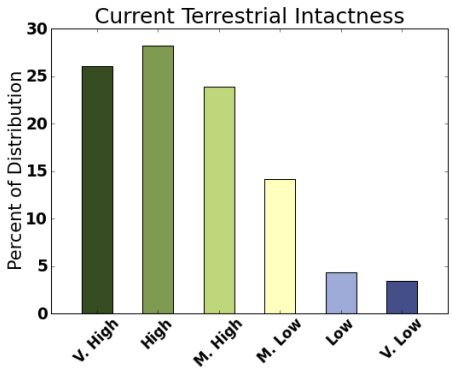


Figure 4-24. Status of designated protected lands in the Sonoran Desert ecoregion created by overlaying designated lands with current terrestrial landscape intactness.

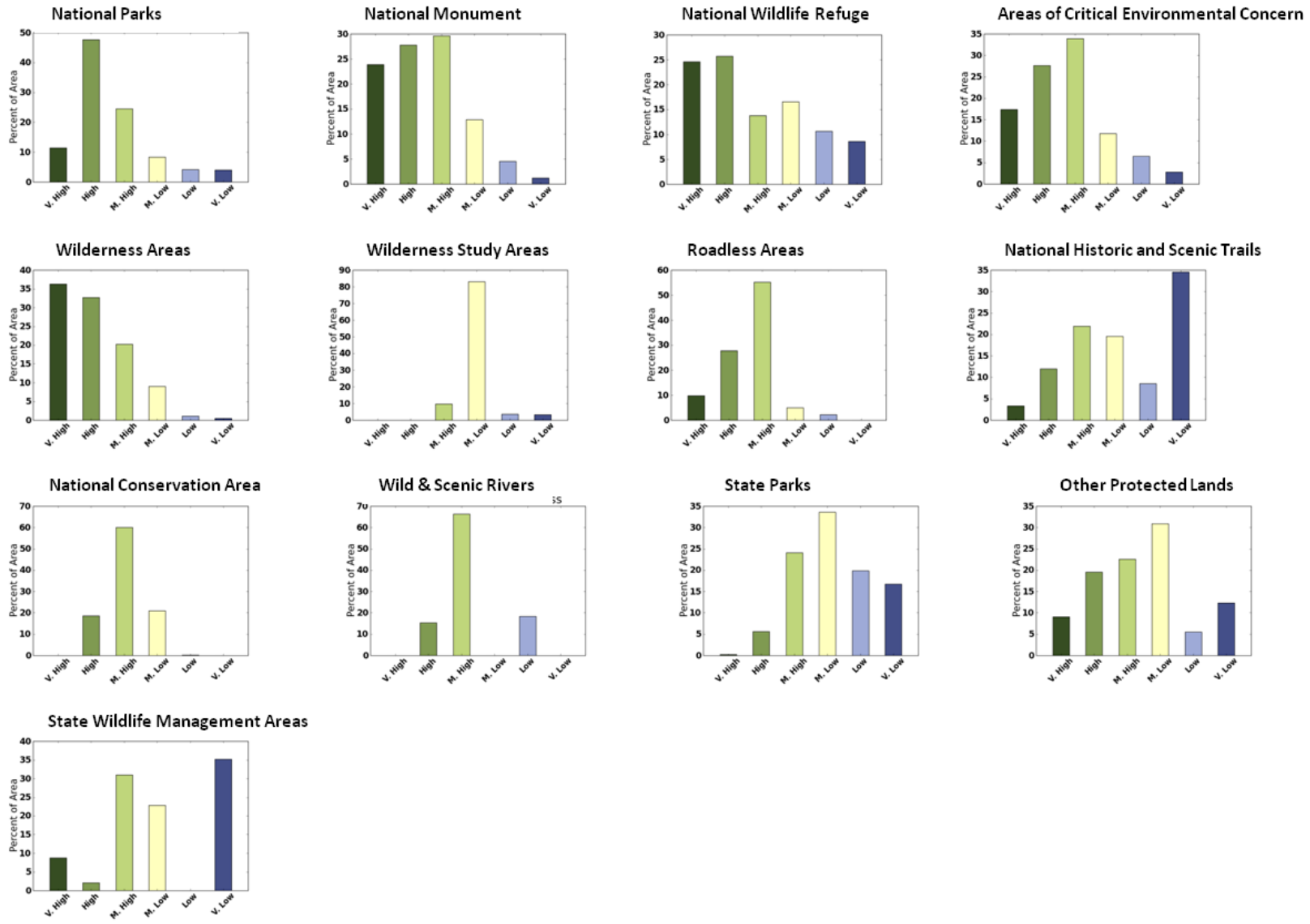


Figure 4-25. Terrestrial landscape intactness profiles for each designated land class. Note that y-axis (acres) varies for each histogram.

4.2.5 References Cited

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

AZDOT (Arizona Department of Transportation). 2006. Arizona's Wildlife Linkages Assessment. Arizona Department of Transportation and Arizona Game and Fish Department, Phoenix, Arizona.

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

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

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

Master, L.L., B.A. Stein, L.S. Kutner, and G.A. Hammerson. 2000. Vanishing assets: Conservation status of U.S. species. Pages 93–118 in Stein, B.A., L.S. Kutner, and J.S. Adams (eds.), Precious heritage: The status of biodiversity in the United States. Oxford University Press, New York.

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

Nussear, K.E., T.C. Esque, R.D. Inman, L. Gass, K.A. Thomas, C.S.A. Wallace, J.B. Blainey, D.M. Miller, and R.H. Webb. 2009. Modeling habitat of the desert tortoise (Gopherus agassizii) in the Mojave and parts of the Sonoran Deserts of California, Nevada, Utah, and Arizona. USGS Open-File Report 2009-1102, U.S. Geological Survey, Las Vegas, Nevada, 18 p.

Rorabaugh, J. 2006. Reptiles of Arizona, lowland leopard frog, *Rana yavapaiensis*. Accessed: 2/12. http://www.reptilesofaz.org/Turtle-Amphibs-Subpages/h-l-yavapaiensis.html.

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

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

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

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

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

4.3 Change Agent Distribution and Intensity

The status of conservation elements must be assessed with reference to both natural and anthropogenic disturbance factors. Although the current distribution and status of REA conservation elements were presented together in Section 4.2 to economize on presentation space, the status or condition of various conservation elements should not be discussed without examining the risks that these resources experience from a collection of regional disturbances or change agents. The primary change agents affecting the region were introduced in Section 2.4.3 (Table 2-4). This section of the report presents those change agents that are associated with current conditions—invasive vegetation, fire, and a current development footprint. Change agents associated with future conditions are presented in Chapter V Potential Future Conditions in the Sonoran Desert.

4.3.1 Invasive Vegetation

Invasive Species Management Question

MQ F1 Where are the areas dominated by tamarisk, red brome, buffelgrass, and Sahara mustard?

Invasive vegetation species are significant change agents in the Sonoran Desert ecoregion. These species alter ecosystem processes, such as fire regimes; they have the potential to expand their distribution in spite of human and natural disturbances and to adapt and shift their range in response to climate change. As these species expand in distribution and dominance on the landscape, native species and communities become increasingly marginalized, which over the long term may seriously degrade the status and function of these systems. Major invasive vegetation species in the Sonoran Desert ecoregion include red brome (*Bromus rubens*), tamarisk (e.g., *Tamarix chinensis, T. aphylla,* and *T. ramosissima*), Sahara mustard (*Brassica tournefortii*), and buffelgrass (*Cenchrus ciliaris*, syn. *Pennisetum ciliare*). Several of these species, especially the annuals, have strong potential to mediate a feedback cycle that can dramatically change the natural fire regime of ecologically significant vegetation communities, such as palo verde-mixed cacti desert scrub (Burquez-Montijo et al. 2002, Esque and Schwalbe 2002). Continued changes in fire cycle combined with projected changes from global climate change raise the possibility of widespread type conversion of desert shrublands to low-diversity nonnative communities with major effects on ecosystem function and the abundance of desert wildlife (Smith et al. 2000, Dukes and Mooney 2004).

Red brome is a nonnative, annual grass from the Mediterranean region that was introduced into the western United States in the mid-1800s (Salo 2005, Newman 2001) and that now occupies broad areas in the Arizona Upland of the Sonoran Desert (ASDM 2010, Turner and Brown 1982). Red brome typically occurs below 5,000 feet elevation on gentle to moderate slopes, often in shallow, sandy loam or clayey soils where it is tolerant of high salt and pH conditions (Wu and Jain 1978). Red brome is a prolific seed producer (Wu and Jain 1978); seeds are dispersed by wind, small mammals, and water (Drezner et al. 2001, Hulbert 1955). Red brome does not form a persistent soil seed bank. However, it germinates earlier and requires less rainfall than native annual species, and it may displace natives in wet years (Reid et al. 2008, Salo 2005, Newman 2001, Beatley 1966). On the other hand, red brome populations may be adversely affected by drought (Salo 2005). Red brome readily invades disturbed areas (Newman 2001), and it is enhanced by grazing and fire (Hulbert 1955). The species also invades undisturbed habitats (Reid et al. 2008, Burgess et al. 1991, Beatley 1966), including scrub communities and mesquite bosques in the Sonoran Desert (Simonin 2001). This ability to invade undisturbed habitat makes this species particularly problematic in the Southwest, where, by altering fire regimes, it threatens native plant communities and associated wildlife species.

Buffelgrass is a drought-tolerant, warm-season, perennial bunchgrass native to Africa, Asia, and the Middle East. It occurs primarily in disturbed sites, along roadsides, in desert washes, and on rocky hillsides (Búrquez-Montijo et al. 2002, Rutman and Dickson 2002, Burgess et al. 1991). The elevational range of this species is generally between about 66 and 2300 feet (20–701 m), although it has been found above 2950 feet (899 m) in Arizona. It spreads aggressively by seed, and it can also spread vegetatively by rhizomes (Arriaga et al. 2004, Williams and Baruch 2000). This species germinates with relatively low amounts of precipitation (Ward et al. 2006). Buffelgrass was introduced to the United States as livestock forage in the 1930s; it has also been used for erosion control and soil stabilization (SABCC 2010a). It is particularly problematic in the Sonoran Desert of southern Arizona and northern Mexico (SABCC 2010b), where it alters fire regimes, soil ecology, hydrology, and geomorphology, thereby threatening native plant communities and associated wildlife species. In Arizona, it has invaded upland desert scrub habitat and it is also considered a threat to native desert grassland, chaparral, and oak woodland (Van Devender and Dimmitt 2006). Buffelgrass is an invasive species that is currently spreading into new areas and it is considered a serious threat to key desert species including the saguaro cactus, foothill palo verde, and desert tortoise (Esque et al. 2006, Esque et al. 2004).

Sahara mustard is an annual herb native to arid and semi-arid regions of North Africa and the Middle East (Cal-IPC 2012). It occurs at low elevations in the Sonoran Desert ecoregion, often on sandy soils and stabilized dunes (J. Weigand, personal communication), where it is capable of forming dense stands; but it has also recently been found in large stands on rocky slopes (Brooks 2009). It has recently expanded its range after years of high winter rainfall (Barrows et al. 2009). This species poses several threats to native vegetation communities, including creation of continuous fuel loads in areas of discontinuous native fuels and rare fires (Brooks 2007). It also competes with native species for soil moisture and nutrients (Cal-IPC 2012).

Because of the recent expansion of Sahara mustard, which was not well captured in existing landcover classifications, and because of its potential effects on fire regime, a predictive model was developed for the species using MaxEnt (Elith et al. 2011) to identify areas of high potential for its occurrence (Figure 4-26). Occurrence data were compiled from a variety of sources, and predictive surfaces were derived from elevation, climate, distance from highways, surficial geology, and soil characteristics. Occurrence data were not uniformly collected and likely do not fully capture the range of locations on which Sahara mustard is found (Figure 3-8 in Section 3.2.5); occurrence records were particularly common along highways and notably underrepresented in sandy areas some distance from highways (J. Weigand and T. Esque, personal communication). For the occurrence locations that were available, this model performed reasonably well (Area Under Curve, AUC: 0.857). The most important factors included distance from highways, elevation, and winter precipitation. In general, large areas of higher probability occur in low elevation basins east of Yuma, north of Yuma to Parker, west of the Salton Sea, and along major highways throughout the Sonoran Desert.

Photo: Sahara mustard (*Brassica tournefortii*), Arizona-Sonora Desert Museum

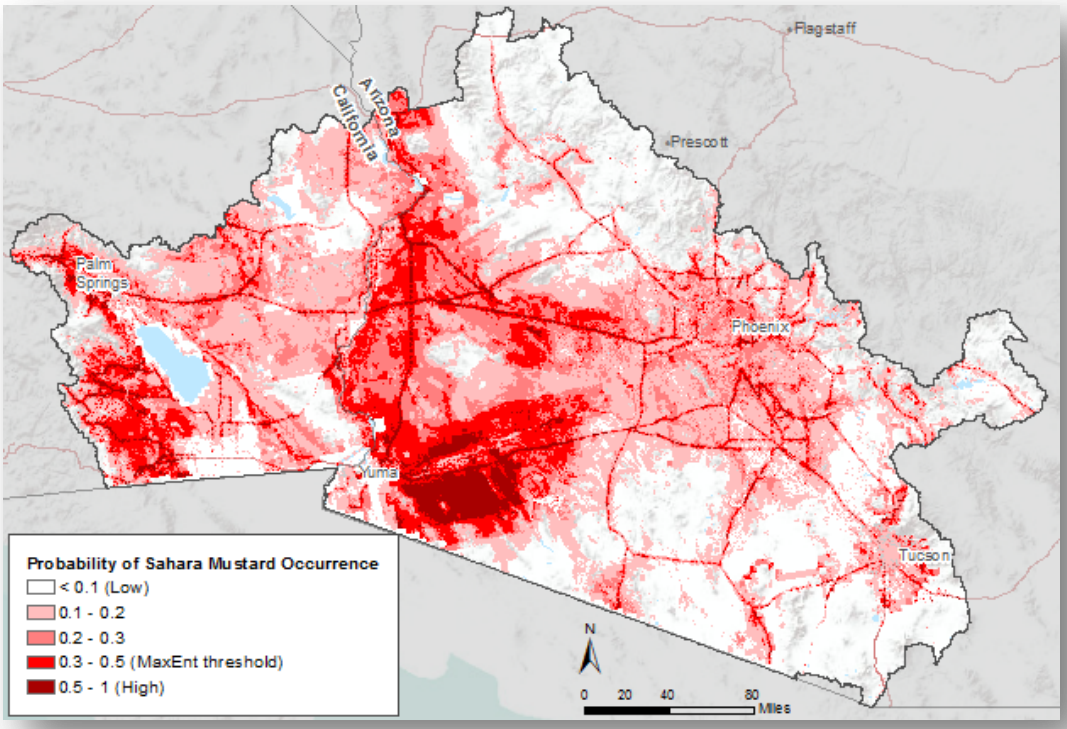


Figure 4-26. Predicted current distribution of Sahara mustard (*Brassica tournefortii*).

Another key invasive species is tamarisk (*Tamarix spp.*) with multiple species and hybrids present. Tamarisk became widely distributed in the 1800s, when it was planted as an ornamental plant; it is now found throughout nearly all western and southwestern states (Lovich 2000). Tamarisk is of particular concern because its dense and rapid growth allows it to out-compete native plant species. In addition, it is extremely drought resistant, has high fecundity, produces salts that inhibit the germination and growth of native species, and alters fire regimes (Busch and Smith 1995, Glenn et al. 1998). Tamarisk affects native wildlife by changing the composition of forage plants and the structure of native riparian systems, which is particularly important to some canopy-nesting birds. However, some native birds will use tamarisk for nesting, including the endangered southwestern willow flycatcher (van Riper et al. 2008, Brown and Trosset 1999, Sogge et al. 2005). For more discussion about riparian ecosystems and tamarisk see the Tamarisk Case Study Insert.

Accurately mapping the full distribution of major invasive vegetation species is quite difficult due to a general lack of systematically sampled occurrences, the difficulty in distinguishing low seasonal abundance within the satellite imagery often used to create land cover classifications, and the requirement of carefully calibrated satellite imagery time series to capture the particular phenology of the invasive species, such as early season green-up of invasive annual grasses. Invasives may be difficult to detect where they are co-dominants, present in the understory, or not actively growing during the season of imagery. The REA was hampered by a lack of regional invasives mapping or modeling. As a result, results from multiple mapping efforts were combined to estimate the extent of major invasive vegetation species in the Sonoran Desert (Figure 4-27). To create the map, invasive classes were extracted from LANDFIRE Existing Vegetation Type (v1.1), NatureServe Landcover (v2.7), and LANDFIRE Succession Classes, and they were combined with areas of invasive vegetation cover from the Integrated Landscape Assessment Project (ILAP 2012) Current Vegetation dataset and higher probabilities from our Sahara mustard occurrence model. Mapped areas of tamarisk and high-probability areas from a recent tamarisk probability map (Jarnevich et al. 2011) were also included. These data and models likely underestimate total distribution of invasive vegetation, because most methods used

remotely-sensed imagery and required dominance of a site by these species to be detectable. Even where these species occur as minor components of the vegetation community, they may expand and dominate quickly due to disturbance, land use and climate change. Furthermore, these species may greatly expand or contract their range and dominance during years of higher or lower available moisture during their peak growing periods; thus the current mapped distribution represents only a snapshot in time of a highly dynamic process.

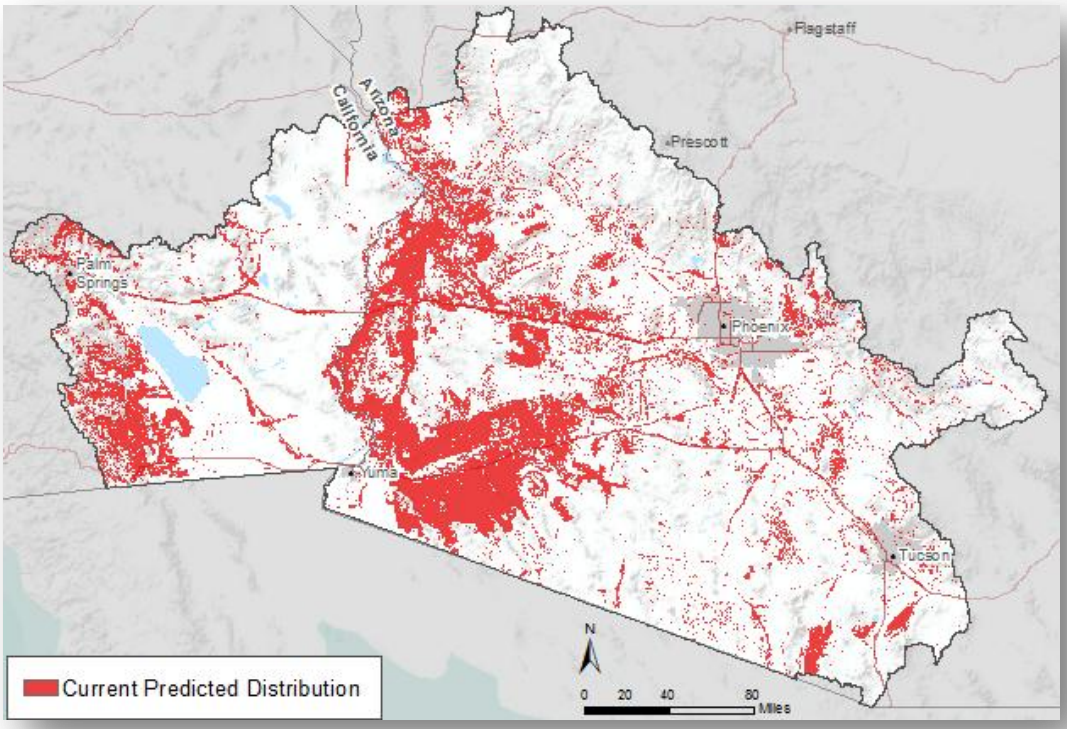


Figure 4-27. Map shows distribution of invasive riparian vegetation (tamarisk) and invasive upland vegetation (including red brome, buffelgrass, and Sahara mustard) across the Sonoran Desert ecoregion.

4.3.2 Changes in Fire Regime

Fire is a natural ecosystem process in many regions. In any given region, species are typically adapted to a particular fire regime, which can be characterized in terms of fire frequency, seasonality, severity, and size (Pausas and Keeley 2009). The degree to which fire may become an ecologically significant change agent is related to the extent to which the fire regime has been altered compared to reference conditions and the associated effects of the altered fire regime on the vegetation community. For example, certain vegetation communities adapted to frequent, low-intensity fire are threatened by the consequences of decades of effective fire suppression, which can increase the potential for large, high-severity fires (Schoennagel and Nelson 2010). In contrast, other communities adapted to very infrequent fire are now threatened by increases in fire frequency due to invasive plants and human ignitions.

Fire regimes have been altered in many Southwestern ecosystems compared to reference conditions that would have been present prior to Euro-American settlement. In recent decades, invasive species and human

activities (e.g., grazing, urbanization, fire suppression), as well as other sources of human ignitions, have altered fire regimes in many fire-adapted ecosystems and introduced fire to other ecosystems that historically rarely experienced fire. Some widely-distributed invasive species, such as red brome, increase fire frequency, size, and duration of the fire season by increasing fine fuel loads and continuity, thus allowing fires to spread into areas that were once fuel-limited (Hunter 1991, Brooks and Pyke 2001). These alterations to fire regime can promote further species invasion and thus create a tight feedback loop of increasing fire frequency (Mack and D'Antonio 1998). In the western US, the source of invasions has been linked to various anthropogenic disturbances, including but not limited to grazing, transportation (roads and trains), logging, and residential development (Kemp and Brooks 1998). Just as exotic species are likely to spread from these areas, human-caused ignitions are also likely to increase in areas with higher levels of human presence (Syphard et al. 2007, 2008).

In many ecosystems where fire historically served an important ecological function, several decades of effective fire suppression, combined with alterations to fuel load and pattern by anthropogenic land use and management practices, have led to conversions in vegetation type (e.g., shrub encroachment in semi-desert grasslands) or structure (e.g., increased canopy density as well as surface and canopy fuel loads, McPherson 1995, Van Auken 2000, Keane et al. 2002). Unless fuel loads are reduced, or unless fire occurs under non-severe weather conditions, fires in many of these communities may now become abnormally large and severe, which can result in dramatic reduction in aboveground live biomass, leading to cascading ecological impacts (DellaSala et al. 2004, Lehmkuhl et al. 2007, Hurteau and North 2009).

Fire-Related Management Questions

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

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

MQ E3 Where are fire-adapted communities?

To answer the first fire-related management question, *Where are areas with that have been recently changed by wildfire?*, estimates of areas changed by recent (1999–2010) wildfires using fire perimeters (2000–2010) were supplemented with fire disturbance data that included measures of fire severity (LANDFIRE Disturbance datasets 1999–2008, Figure 4-28). LANDFIRE estimates of fire severity should be interpreted with caution; they may have poor accuracy for predicting actual fire severity in desert systems because of methods and definitions of fire severity developed primarily for forested systems. Furthermore, fire severity in desert ecosystems is not well understood or described in the literature. In general, any area that has experienced fire has been changed by it to a degree that generally increases with increasing severity. High severity fires tend to result in early successional vegetation states followed by a recovery period during which characteristic species recolonize the site. However, areas with uncharacteristically high severity (due in part to legacy effects of fire suppression and fuel buildup) or where fire was historically rare may transition to a different vegetation state altogether, such as dominance by invasive vegetation. It is not possible given existing data to evaluate the underlying change in vegetation resulting from fire. This is due in large part to the lack of accurate region-wide maps of pre- and post-fire vegetation. While the most recent version of LANDFIRE Existing Vegetation (v1.1) has been updated in areas of disturbance, the updates are not necessarily an accurate reclassification of the post-fire vegetation, but instead appear to be the result of applying a rule set based on pre-fire vegetation type and fire severity coupled with a systematic update of the entire product to correct areas of major inaccuracy. The majority of the higher-severity fires occurred at the fringes of the ecoregion; typically in communities like chaparral that would have experienced fires under reference conditions. However, several lower-severity fires occurred in central areas of the ecoregion, including the King Valley Fire in 2005 which burned areas of the Kofa National Wildlife Refuge. Even when fires are of lower severity, they may still result in high mortality of species that are not fire-adapted and increase susceptibility of burned areas to invasion by nonnative species such as Sahara mustard (USFWS 2006, T. Esque personal communication).

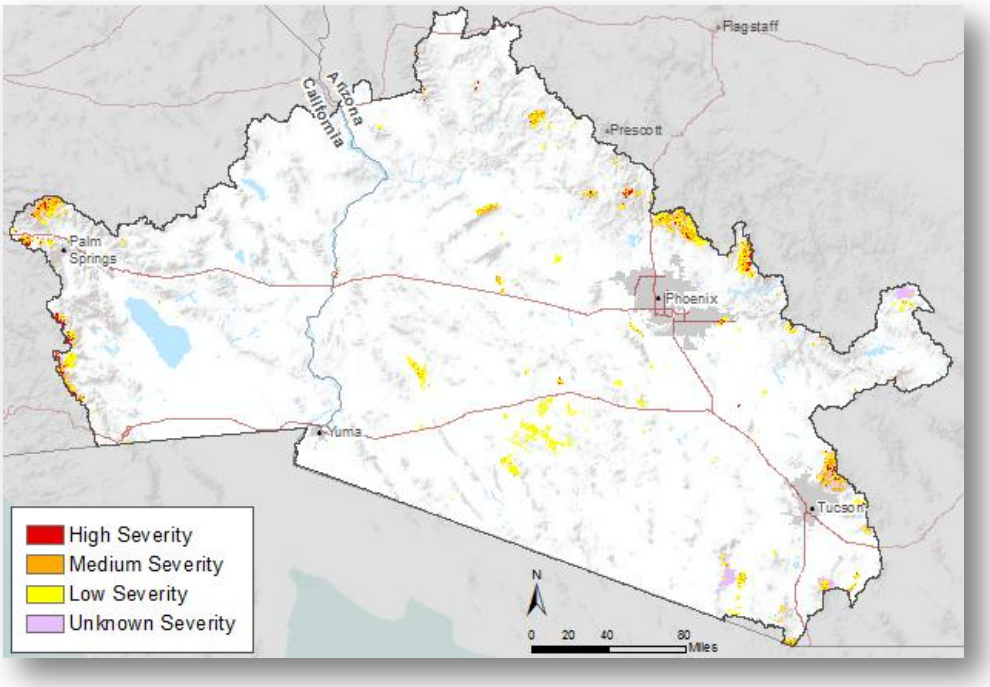


Figure 4-28**.** Fire perimeters annotated by severity (where available) for fires in the Sonoran Desert ecoregion, answering the management question MQ E1, *Where are the areas that have been changed by wildfire between 1999 and 2009*?

To answer the fire-related question (*Where are the areas with potential to change from wildfire?* MQ E2) and to estimate the distribution of these areas, MaxEnt models were developed for potential fire occurrence (Figure 4-29). In reality, fire has the potential to cause a greater or lesser magnitude of change due to fine scale fuel conditions, local fire behavior, fire weather, and pre-fire vegetation sensitivity to fire disturbance along with many other factors. It is not possible given existing data to evaluate these factors at the ecoregion scale. Instead, the focus was on predicting where fires are likely to occur on the premise that this would provide meaningful context for more detailed, local assessments of potential impacts due to fire. Thirty years of fire occurrence data were subdivided into human and naturally caused fires (21,310 human caused fires; 1,324 naturally-caused fires) and developed into separate MaxEnt models for each due to the very disparate relationship between fire cause and underlying geographic and environmental variables. Both models performed reasonably well (human-caused fire model Area Under Curve or AUC: 0.704 and natural model AUC: 0.814). The most influential factors in the human model include: distance to highways, distance to urban areas, distance to major rivers, and winter precipitation). The most influential factors in the natural model include: summer temperature, elevation, winter precipitation, and distance to major rivers. Even though the density of strong, fire-season lightning events (1990-2009) was included in the natural model, it was not a particularly important factor.

In general, the potential of naturally-caused fire occurrence increases toward the edges of the ecoregion and on a few mountain ranges, such as the Harcuvar Mountains in the north central portion of the ecoregion. Some of these areas were historically adapted to fires (chaparral); however, legacy effects of fire suppression and alteration of vegetation composition and structure may result in uncharacteristic fire behavior. Human-caused fire potential increases around Phoenix, Parker, Yuma, and Palm Springs. Some of these same areas also showed higher likelihood of invasive vegetation occurrence (Figure 4-27), indicating that increased fire occurrence due to human ignitions coupled with continuous fine fuels may result in significant impacts to the native vegetation communities and may further expand the distribution of the invasives.

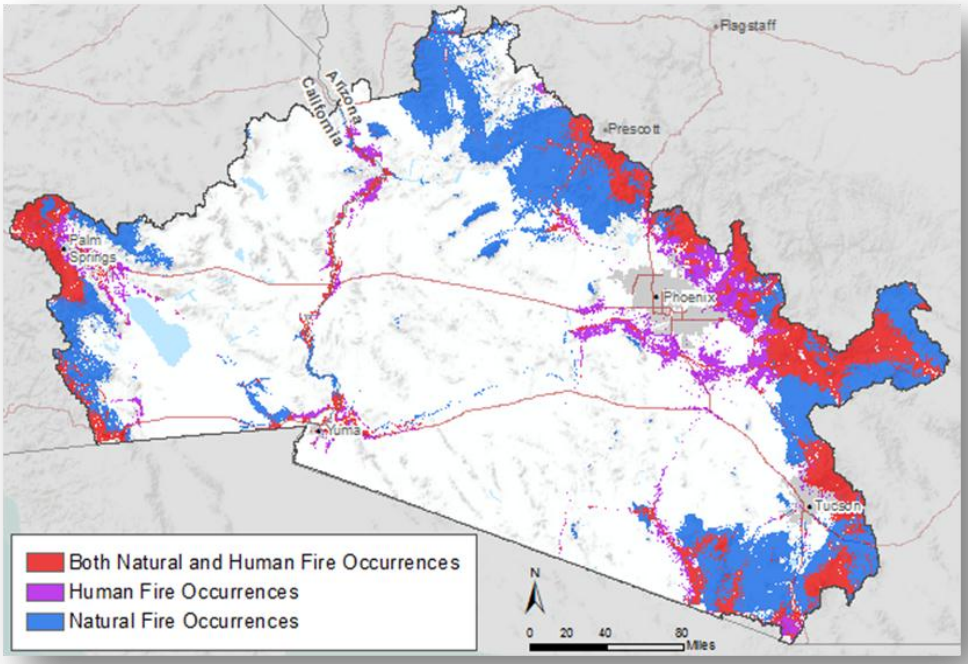


Figure 4-29. Potential fire occurrence from human and natural sources for the Sonoran Desert ecoregion, answering the management question, *Where are the areas with potential to change from wildfire?*

Fire-adapted communities were identified using the LANDFIRE Fire Regime Groups dataset (Figure 4-30, management question MQ E3). Again, these areas primarily occur at the fringes of the ecoregion, indicating that vegetation communities throughout much of the ecoregion are adapted to very rare fire occurrence. Fire occurrence in areas historically adapted to frequent fires may still produce uncharacteristic behavior, severity, or alteration of the vegetation communities due to legacy effects of fire suppression, which may lead to buildup of fuels. The degree to which fire regimes have been altered in these areas cannot be fully determined. While estimates of fire regime departure exist (e.g., LANDFIRE Fire Regime Condition Class), these estimates are based on differences in vegetation composition and structure compared to reference condition proportions of various succession classes. Such comparisons are particularly challenging in arid ecosystems, due to the difficulty in correctly detecting fine-scale differences in vegetation composition or structure using remote sensing techniques. Because these estimates of fire regime departure do not directly capture changes in fire frequency and departure, they may under-represent the degree of fire regime departure present in these communities. Existing estimates of current fire frequency and severity were not available to determine these measures during this REA.

Photo: Burn in Arizona, BLM

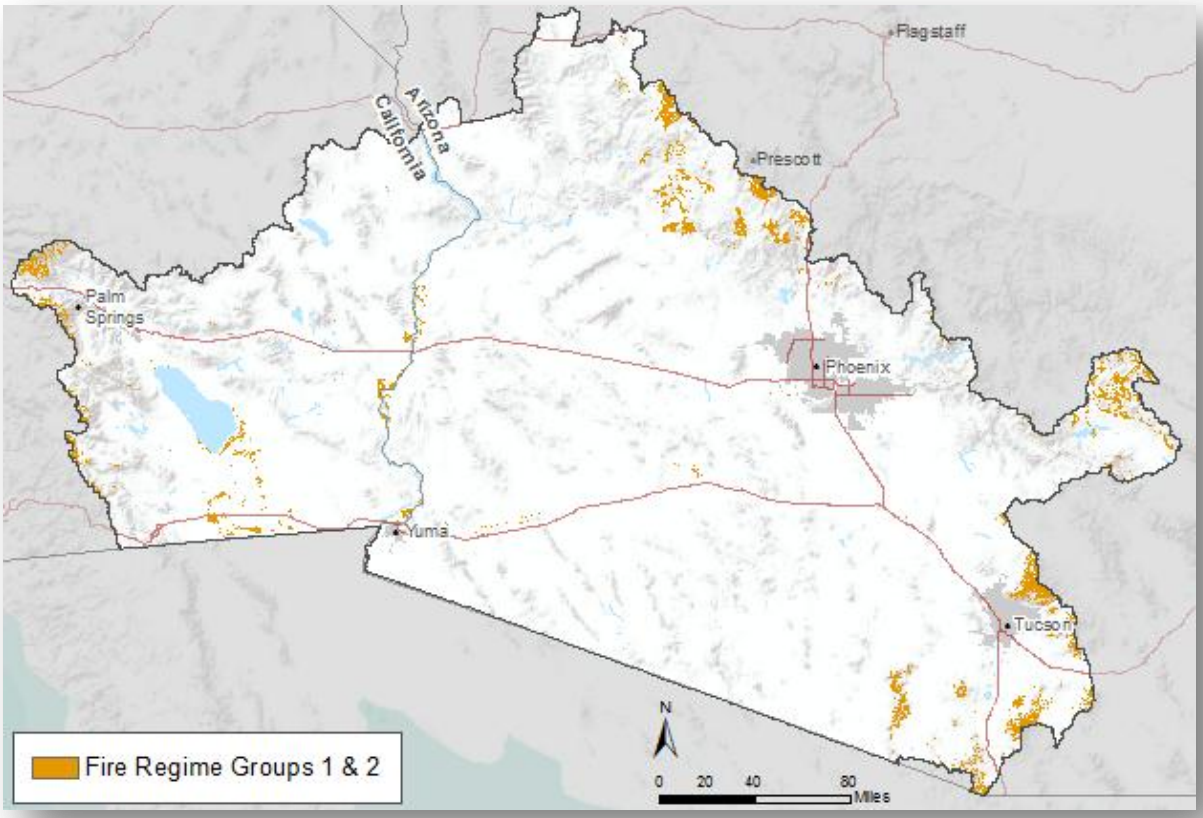


Figure 4-30**.** Map answers management question MQ E3, *Where are fire-adapted vegetation communities in the Sonoran Desert ecoregion*?

4.3.3 Current Development

Four major components of development were assessed for the ecoregion—energy, urbanization (including roads), agriculture, and recreational development. A dozen major inputs derived from multiple original datasets were compiled using a fuzzy logic model (Figure 4-31) to create a combined development footprint for the ecoregion (Figure 4-33). Reliable spatial data were available for all but recreation and existing wind energy locations (such as San Gorgonio in southeastern California). Recreation data proved to be very difficult to acquire and what was acquired was of uneven quality. The addition of these missing elements as they come available will improve the model. For the composite model, a subset of the compiled and analyzed recreation data was used to address more specific recreation management questions such as MQ H1, *Where are high-use recreation sites, developments, roads, infrastructure, or areas of intensive recreation use located (including boating)?* (See Appendix A for more details on recreation.) The recreation data used for the composite development model focused on land recreation only and included point, line, and polygon inputs (Figure 4-32D).

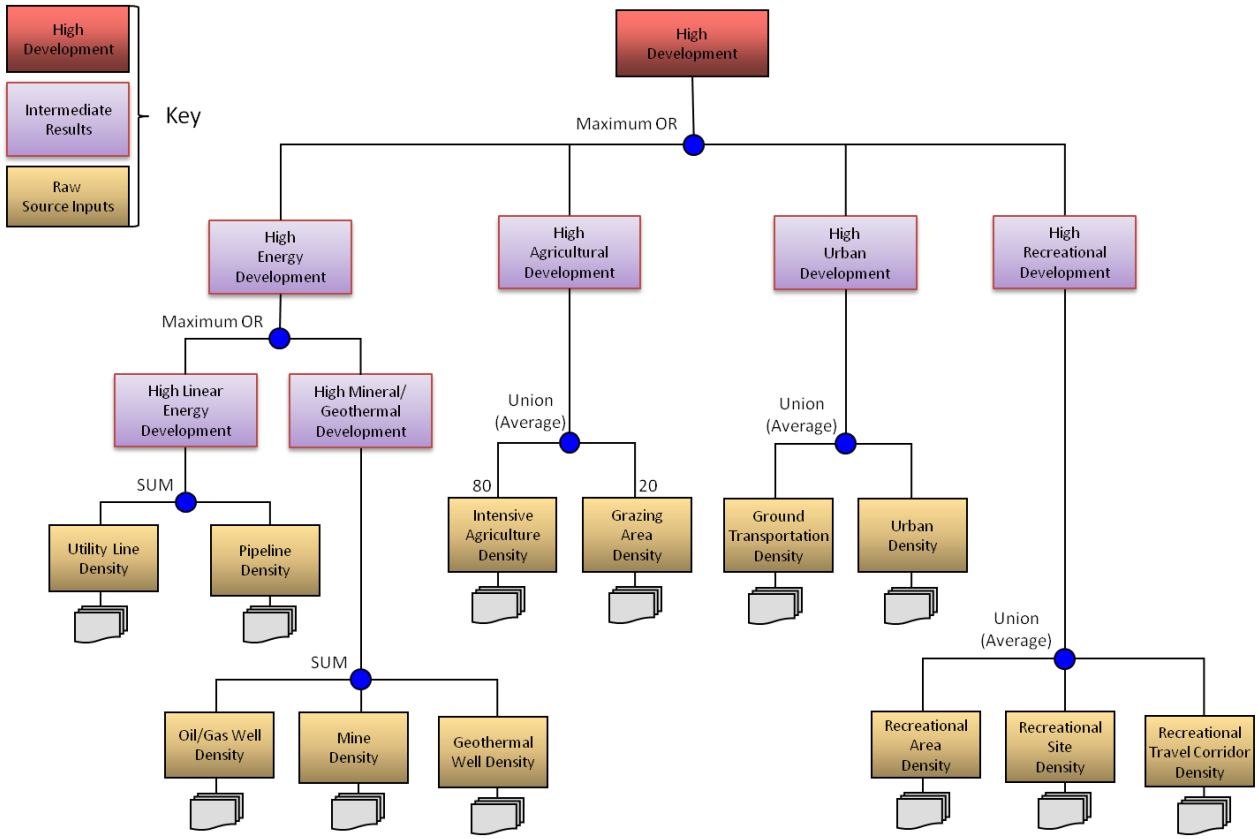


Figure 4-31. Current development fuzzy logic model for the Sonoran Desert ecoregion.

Current energy development, one of the intermediate model results (top four boxes in logic model above), was comprised of spatial data for both linear features (utility lines and pipelines) and point features (oil/gas wells, mines, and geothermal wells) that were aggregated using a *Maximum OR* logic operator (Figure 4-32A). The urban development component of the fuzzy logic model averaged urban landcover density and road density based on the transportation data files provided by BLM to create an intermediate urbanization result (Figure 4-32B). No weighting or special treatment of roads was conducted as the dataset was inconsistently attributed to allow for more detailed treatment of the road infrastructure, which ranged from OHV dirt paths to interstate highways

Agricultural development results were derived from agriculture landcover data and grazing allotment data using an *Average (or Union)* logic operator and weighting converted agricultural land vs. grazing lands by 80/20 (Figure 4-32C). Livestock grazing in the ecoregion has altered the natural landscape, important details on recent livestock density and overall range condition remains a serious data gap in the model. With more detailed and complete grazing data, the development model as well as the terrestrial and aquatic intactness models would be greatly enhanced. Recreation development data was also substandard and the model would do a better job of incorporating recreation impacts with more detailed and complete data for the wide array of recreational activities (both active and passive).

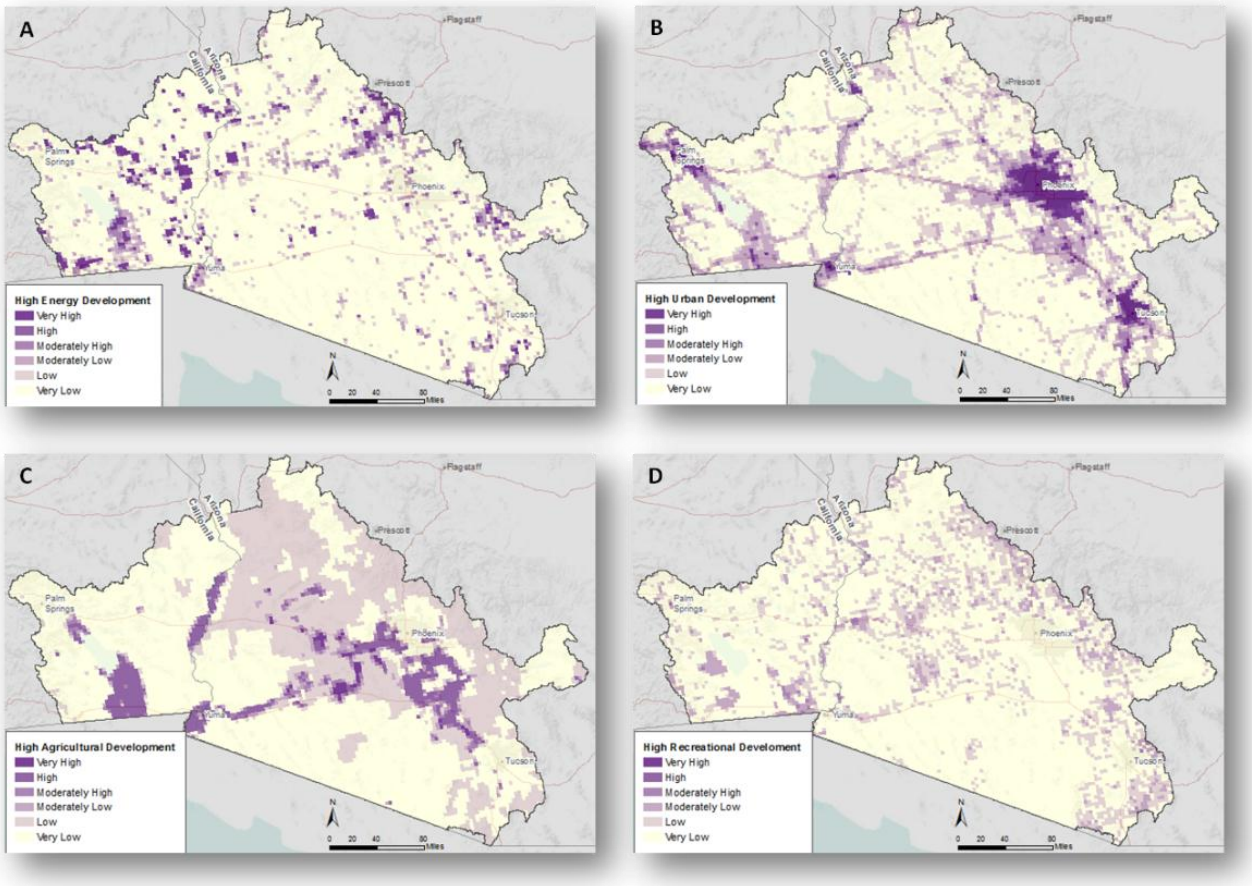


Figure 4-32. Intermediate results of the current development fuzzy logic model showing (A) current energy development, (B) urban development, (C) agriculture development, and (D) recreation development for the Sonoran Desert ecoregion.

The full development footprint for the Sonoran Desert shows a concentration of human activities in the northern and eastern portions of the ecoregion in the Phoenix-Tucson urban corridor and in the western portions of the ecoregion in the urban and agricultural areas of the Palm Springs area and the Imperial and Coachella valleys of California (Figure 4-33).

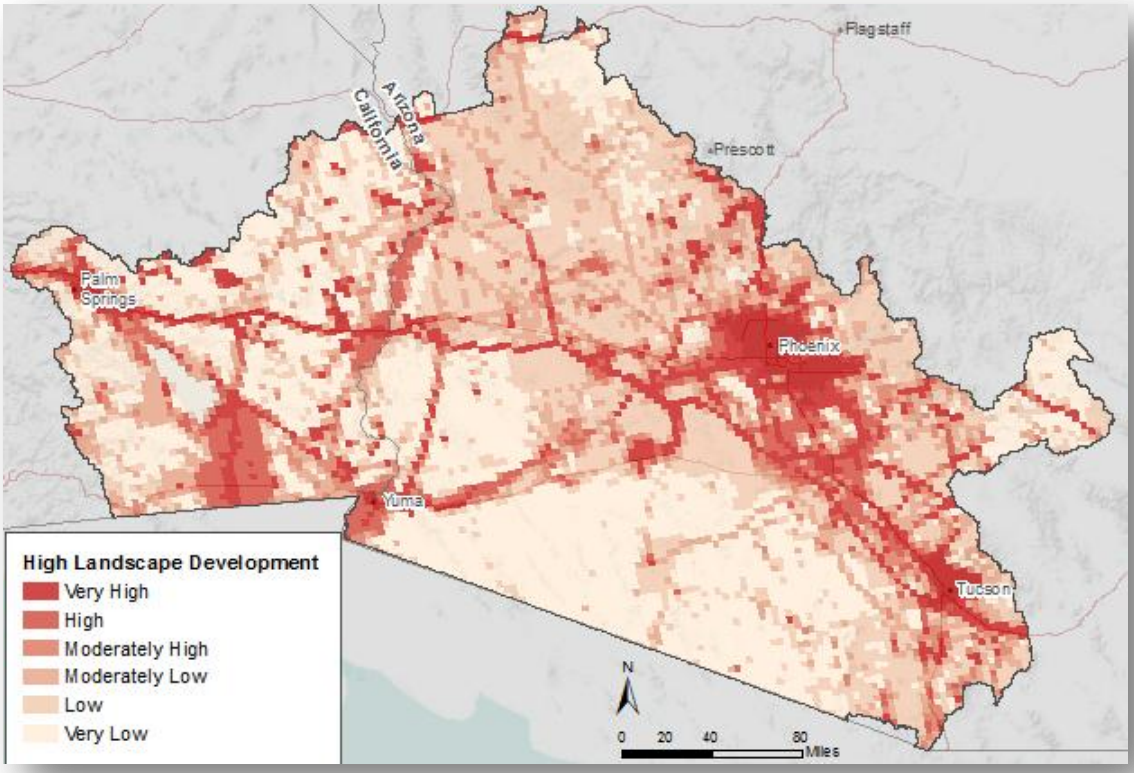


Figure 4-33. Composite of current development in the Sonoran Desert ecoregion.

Photo: Phoenix. Wikimedia Commons, gobeirne.

4.3.4 References Cited

Arriaga, L., V. Castellanos, E. Alejandro, E. Moreno, and J. Alarcon. 2004. Potential ecological distribution of alien invasive species and risk assessment: A case study of buffelgrass in arid regions of Mexico. *Conservation Biology* 18(6):1504–1514.

ASDM (Arizona-Sonora Desert Museum). 2010. Invasive species in the Sonoran Desert region. http://www.desertmuseum.org/invaders/vol_handbook/Invasive_Species_in_the_Sonoran_DesertRegion.pdf

Barrows, C.W., E.B. Allen, M.L. Brooks, and M.F. Allen. 2009. Effects of an invasive plant on a desert sand dune landscape. *Biological Invasions* 11(3):673–686.

Beatley, J.C. 1966. Ecological status of introduced brome grasses (*Bromus* spp.) in desert vegetation of southern Nevada. *Ecology* 47(4):548–554.

Brooks, M.L. 2009. Spatial and temporal distribution of nonnative plants in upland areas of the Mojave Desert. Pages 101–124 *in* Webb, R.H., L. Fenstermaker, and J. Heaton (eds), The Mojave Desert: Ecosystem processes and sustainability, University of Nevada Press, Las Vegas.

Brooks, M.L. 2007. Seed production by the non-native Sahara mustard. USGS Publication Brief for Resource Managers. U.S. Geological Survey, Western Ecological Research Center.

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

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

Burgess, T.L., J.E. Bowers, and R.M. Turner. 1991. Exotic plants at the desert laboratory, Tucson, Arizona. *Madroño* 38:96–114.

Burquez-Montijo, A., M.E. Miller, and A. Martinez-Yrizar. 2002. Mexican grasslands, thornscrub, and the transformation of the Sonoran Desert by invasive exotic buffelgrass (*Pennisetum ciliare*). Pages 126–146 *in* Tellman, B. (ed.), Invasive exotic species in the Sonoran region, University of Arizona Press and Arizona-Sonora Desert Museum, Tucson, Arizona.

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

Cal-IPC (California Invasive Plant Council). 2012. Invasive Plants of California's Wildland: *Brassica tournefortii* http://www.cal-ipc.org/ip/management/ipcw/pages/detailreport.cfm@usernumber=12&surveynumber=182.php [Accessed 2/3/2012].

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

Drezner, T.D., P.L. Fall, and J.C. Stromberg. 2001. Plant distribution and dispersal mechanisms at the Hassayampa River Preserve, Arizona, USA. *Global Ecology and Biogeography* 10(2):205–217.

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

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

Esque, T.C., and C.R. Schwalbe. 2002. Alien annual grasses and their relationships to fire and biotic change in Sonoran desertscrub. Pages 165–194 *in* Tellman, B. (ed.), Invasive exotic species in the Sonoran region, University of Arizona Press and Arizona-Sonora Desert Museum, Tucson, Arizona.

Esque, T.C., C.R. Schwalbe, D.F. Haines, and W.L. Halvorson. 2004. Saguaros under siege: Invasive species and fire. *Desert Plants* 20:49–55.

Esque, T.C., C.R. Schwalbe, J.A. Lissow, D.F. Haines, D. Foster, and M.C. Garnett. 2006. Buffelgrass fuel loads in Saguaro National Park, Arizona, increase fire danger and threaten native species. *Park Science* 24(2):33–37.

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

Hulbert, L. 1955. Ecological studies of *Bromus tectorum* and other annual brome grasses. *Ecological Monographs* 25(2):181–213.

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

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

ILAP (Integrated Landscape Assessment Project). 2012. http://oregonstate.edu/inr/ilap.

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

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

Kemp, P., and M.L. Brooks. 1998. Exotic species of California deserts. *Fremontia* 26:30–34.

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

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

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

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

Newman, D. 2001. Element stewardship abstract for *Bromus rubens*. The Nature Conservancy, Arlington, Virginia.

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

Reid, C.R., S. Goodrich, and J.E. Bowns. 2008. Cheatgrass and red brome: The history and biology of two invaders. Pages 27–32 *in* Kitchen, S.G., R.L. Pendleton, T.A. Monaco, and J. Vernon (compilers). Proceedings: Shrublands under fire: Disturbance and recovery in a changing world, June 6–8, 2006, Cedar City, Utah. Proceedings RMRS-P-52, U.S. Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado.

Rutman, S., and L. Dickson. 2002. Management of buffelgrass on Organ Pipe Cactus National Monument, Arizona. Pages 311–318 *in* B. Tellman (ed.), Invasive exotic species in the Sonoran region, Arizona-Sonora Desert Museum and University of Arizona Press, Tucson, Arizona.

SABCC (Southern Arizona Buffelgrass Coordination Center). 2010a. History and distribution. http://www.buffelgrass.org/node/11. [Accessed December 17, 2010].

SABCC (Southern Arizona Buffelgrass Coordination Center). 2010b. Why is it a threat? http://www.buffelgrass.org/node/13. [Accessed December17, 2010].

Salo, L.F. 2005. Red brome (*Bromus rubens* subsp. *madritensis*) in North America: Possible modes for early introductions, subsequent spread. *Biological Invasions* 7:165–180.

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

Simonin, K.A. 2001. *Bromus rubens, Bromus madritensis in* U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. http://www.fs.fed.us/database/feis/

Smith, S.D., T.E. Huxman, S.F. Zitzer, T.N. Charlet, D.C. Housman, J.S. Coleman, L.K. Fenstermaker, J.R. Seemann, and R.S. Nowak. 2000. Elevated CO_2 increases productivity and invasive species success in an arid ecosystem. *Nature* 408:79–82.

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

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

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

Turner, R. M., and D. E. Brown. 1982. Sonoran desertscrub *in* Brown, D.E. (ed.), Biotic communities of the American Southwest: United States and Mexico. *Desert Plants* 4: 181–221.

USFWS (U.S. Fish and Wildlife Service). 2006. King Valley Fire: Burned area rehabilitation plan [online]. U.S. Fish and Wildlife Service, Kofa National Wildlife Refuge. http://www.fws.gov/fire/downloads/ES_BAR/King_Valley_BAR_FY07_Final.pdf

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

Van Devender, T.R., and M.A. Dimmitt. 2006. Final report on conservation of Arizona upland Sonoran desert habitat: Status and threats of buffelgrass (*Pennisetum ciliare*) in Arizona and Sonora, Arizona-Sonora Desert Museum, Tucson, AZ.

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

Ward, J.P., S.E. Smith, and M.P. McClaran. 2006. Water requirements for emergence of buffelgrass (*Pennisetum ciliare*). *Weed Science* 54(4):720–725.

Williams, D.G., and Z. Baruch. 2000. African grass invasion in the Americas: Ecosystem consequences and the role of ecophysiology. *Biological Invasions* 2:123–140.

Wu, K. and K. Jain. 1978. Genetic and plastic responses in geographic differentiation of *Bromus rubens* populations. *Canadian Journal of Botany* 56:873–879.

Photo: Barrel cactus fire victim. National Park Service