DOCKETED	
Docket Number:	21-AFC-02
Project Title:	Willow Rock Energy Storage Center
TN #:	264484
Document Title:	COMMENTS OF CALIFORNIA UNIONS FOR RELIABLE ENERGY ON THE PRELIMINARY STAFF ASSESSMENT ATTACHMENT D (3 OF 7)
Description:	N/A
Filer:	Alisha Pember
Organization:	California Unions for Reliable Energy
Submitter Role:	Intervenor
Submission Date:	6/27/2025 1:14:09 PM
Docketed Date:	6/27/2025

STATE OF CALIFORNIA

STATE ENERGY RESOURCES CONSERVATION AND DEVELOPMENT COMMISSION

IN THE MATTER OF: WILLOW ROCK ENERGY STORAGE CENTER	Docket No. 21-AFC-02

COMMENTS OF CALIFORNIA UNIONS FOR RELIABLE ENERGY ON THE PRELIMINARY STAFF ASSESSMENT

ATTACHMENT D (3 OF 7)

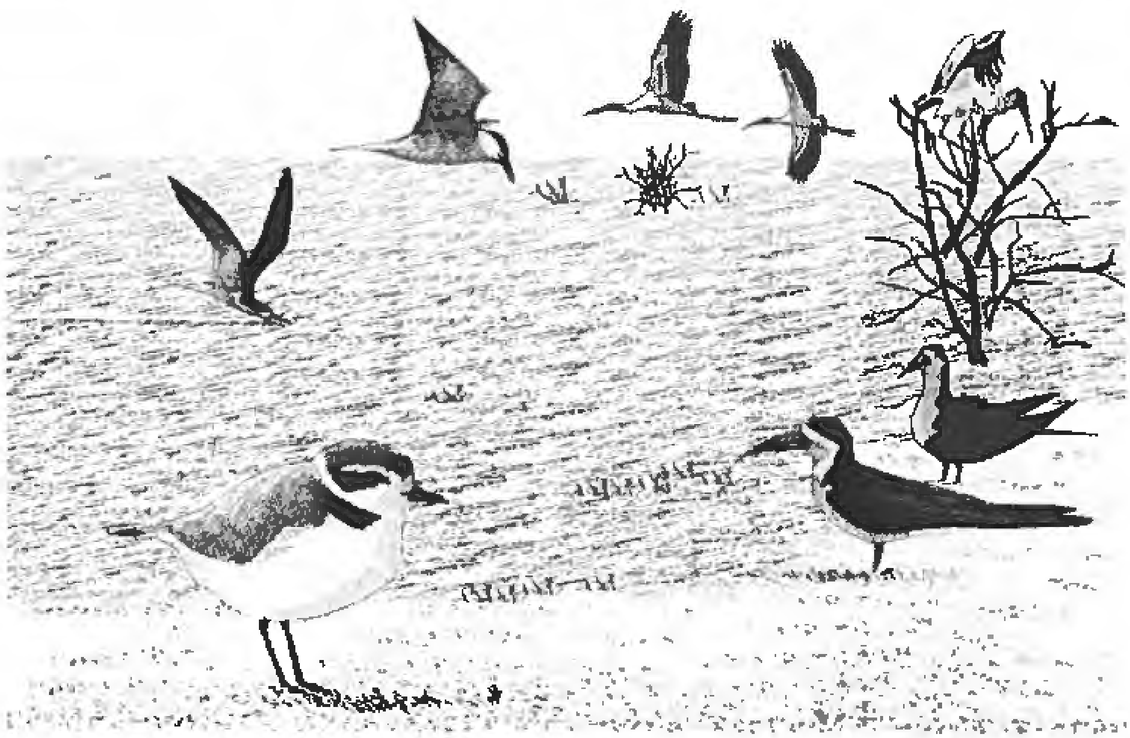
June 16, 2025

Richard M. Franco
Adams Broadwell Joseph & Cardozo
601 Gateway Blvd., Suite 1000
South San Francisco, CA 94080
(650) 589-1660
rfranco@adamsbroadwell.com

Attorneys for California Unions for Reliable Energy

II

SPECIES ACCOUNTS



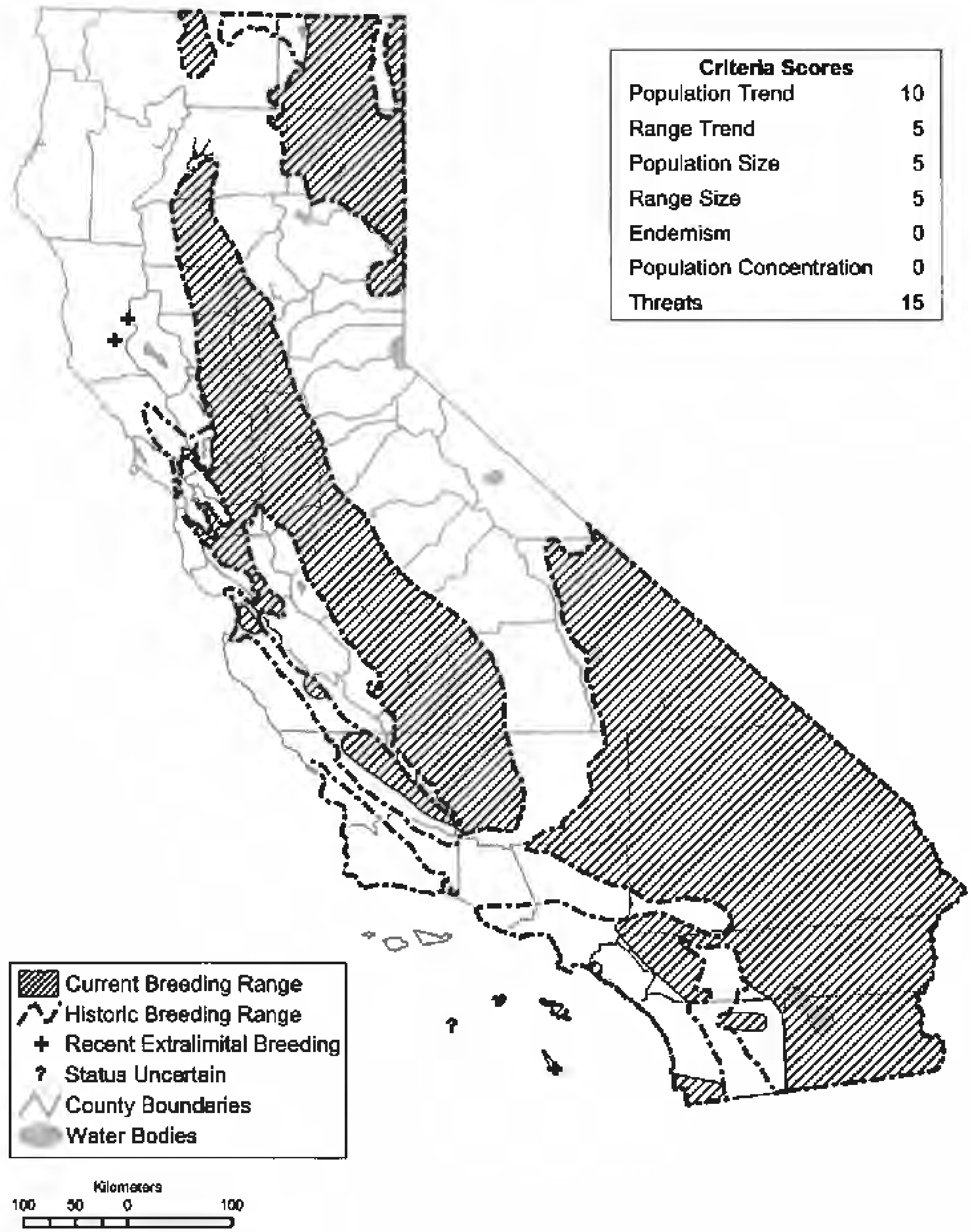
Andy Birch

PDF of Burrowing Owl account from:

Shuford, W. D., and Gardali, T., editors. 2008. California Bird Species of Special Concern: A ranked assessment of species, subspecies, and distinct populations of birds of immediate conservation concern in California. Studies of Western Birds 1. Western Field Ornithologists, Camarillo, California, and California Department of Fish and Game, Sacramento.

BURROWING OWL (*Athene cunicularia*)

Jennifer A. Gervais, Daniel K. Rosenberg, and Lyann A. Comrack



Current and historic (ca. 1944) breeding range of the Burrowing Owl in California. Numbers have declined at least moderately overall, though they are greatly augmented in the Imperial Valley, and the range has retracted in northeastern California and along the coast. During migration and winter, more widespread in lowland areas of the state and reaches more offshore islands.

Special Concern Priority

Currently considered a Bird Species of Special Concern (breeding), priority 2. Included on both prior special concern lists (Remsen 1978, 2nd priority; CDFG 1992).

General Range and Abundance

Broadly distributed in western North America; also occurs in Florida, Central and South America, Hispaniola, Cuba, the northern Lesser Antilles, and the Bahamas (Haug et al. 1993). Two recognized subspecies in North America: *A. c. hypugaea* in the West, *A. c. floridana* in Florida and the Bahamas (Haug et al. 1993, Desmond et al. 2001). Owls in Florida and the southern portion of the western range generally are year-round residents (Haug et al. 1993), but elsewhere in North America they appear to migrate south in a leap-frog fashion (James 1992). Scant data on migration suggest that most Burrowing Owls that breed in North America winter in Mexico (G. Holroyd pers. comm.), Arizona, New Mexico, Texas, Louisiana, and California, which is considered one of the most important wintering grounds for migrants (James and Ethier 1989). A lack of genetic differentiation among migratory and resident owl populations in western North America suggests that these populations interbreed (Korfanta et al. 2005). These results are supported by recent stable isotope analyses (Duxbury 2004).

Seasonal Status in California

Year-round resident throughout much of the state. Seasonal status varies regionally, with birds retreating from higher elevations such as the Modoc Plateau in winter (Grinnell and Miller 1944). Observations of color-banded and/or radio-tagged owls demonstrate year-round residency in the Central Valley, San Francisco Bay region, Carrizo Plain, and Imperial Valley (Brenckle 1936, Coulombe 1971, Thomsen 1971, Carlin 2004, Johnson 1997b, L. Trulio et al. and D. K. Rosenberg et al. unpubl. data). Migrants from other parts of western North America may augment resident lowland populations in winter. The breeding season in California is March to August, but can begin as early as February and extend into December (Rosenberg and Haley 2004, J. A. Gervais unpubl. data).

Historic Range and Abundance in California

Grinnell and Miller (1944) described the historic range of this owl as throughout most of California and most of its islands, except the coastal counties north of Marin and mountainous areas. Noting that the species was originally common or even "abundant" in the state, they reported "large" numbers of owls still occurred in "favorable localities" but that owls were in decline in areas of human settlement. Grinnell and Wythe (1927) reported that Burrowing Owls were "fairly common in the drier, unsettled, interior parts of [the San Francisco Bay] region; most numerous in parts of Alameda, Contra Costa, and Santa Clara connties. Outside of this area has been observed sparingly" in Sonoma, Napa, Solano, and Marin counties (Grinnell and Wythe 1927). Willet (1933), also lacking qnantitative information, described the species on the southern coast as a "common resident from coast to base of mountains." In San Diego County, at least, historical descriptions suggest that the populations may have been quite extensive (Unitt 2004). The increase in abundance of owls in some agricultural environments, such as the Imperial Valley, from presettlement times likely began prior to the late 1920s, when desert country was converted to irrigated agriculture (DeSante et al. 2004, Molina and Shuford 2004). The draining of wetlands associated with European settlement in the Central Valley may also have increased owl distribution and abundance.

Recent Range and Abundance in California

The Burrowing Owl's overall breeding range in California has changed only modestly since 1945 (see map), but the local distribution of owls across the state has changed considerably. There are three primary patterns in the current distribution. First, declines and local extirpations have been mainly

Breeding Bird Survey Statistics for California

1968–2004					1968–1979			1980–2004			All data from Sauer et al. (2005)
Trend	*P*	*n*	(95% CI)	R.A.	Trend	*P*	*n*	Trend	*P*	*n*	Credibility
5.6	0.02	32	1.1, 10.1	1.76	–0.9	0.92	19	7.1	0.11	25	High

along the central and southern coast (DeSante et al. 1997a, b; 2007), regions that are undergoing rapid urbanization. Second, sizable to very large breeding populations remain in agricultural areas in the Central and Imperial valleys, where Burrowing Owls have adapted to highly modified habitats (Coulombe 1971, Rosenberg and Haley 2004). Third, it appears that the vast majority of owls occur on private lands (DeSante et al. 1997a, 2004), largely because of the high densities in agricultural areas. These patterns will present distinct challenges and unique opportunities in the conservation of this species.

Numbers of Burrowing Owls on Breeding Bird Survey (BBS) routes in California increased significantly from 1968 to 2004 (Sauer et al. 2005). Conversely, Christmas Bird Count data, 1959–1988, show declines in midwinter numbers of Burrowing Owls in California (Sauer et al. 1996). Other recent evaluations conclude that declines have occurred in the Central Valley, San Francisco Bay region, and southern coast (DeSante et al. 1997a, 2007; Trulio 1997; Comrack and Mayer 2003). However, preliminary BBS analyses of regional patterns within California detected declines in some regions of California, but increases in the Imperial Valley (DeSante et al. 2007, C. Conway pers. comm.). Understanding the details of spatial patterns of changes in BBS data, and their limitations due to insufficient data, would help resolve the apparent inconsistencies.

Concern over declines on the coast and in urbanized areas of the Central Valley led to surveys of selected 5 x 5 km survey blocks within core areas of the state in 1992 and 1993 (DeSante et al. 1997a, b; 2007). Surveys failed to locate breeding owls in the coastal counties of Napa, Marin, San Francisco, Santa Cruz, and Ventura, and very few were located in Sonoma, San Mateo, Santa Barbara, and Orange counties. These surveys in selected blocks were not intended as a census of all owls. Many of these areas may never have supported sizable breeding populations (e.g., Grinnell and Wythe 1927), although data are generally lacking. There also appeared to be substantial reductions in numbers of breeding owls in other counties around San Francisco, San Pablo, and Suisun bays (DeSante et al. 1997a, 1997b, 2007; Klute et al. 2003). The south San Francisco Bay population, estimated at 103 breeding pairs, was considered to be declining sharply (DeSante et al. 1997a, 2007; Trulio 1997). Finally, the survey concluded that Burrowing Owls were in decline throughout the Central Valley, but this conclusion was based on mostly anecdotal data and not the actual survey (DeSante et al. 1997a). Several large populations (e.g., Naval Air Station Lemoore and Carrizo Plain National Monument) were severely underestimated or missed altogether, and previously undetected populations were also found (DeSante et al. 2007, D. K. Rosenberg et al. unpubl. data), largely due to the survey methods that often had low, but unestimated, detection probabilities (DeSante et al. 2004). In contrast, Burrowing Owls remain abundant in the Imperial Valley, where current densities in that agricultural system apparently far exceed those found in the native desert prior to agricultural conversion (DeSante et al. 2004, Rosenberg and Haley 2004).

Additional information from anecdotal sightings or multispecies surveys offer further insight into status and declines in other regions of the state as outlined below.

Northeastern California. Although its status in this region is poorly known, the species appears to be scarce and may have been so historically. To the west, a few owls are currently known from Shasta Valley, Siskiyou County, but they may have been extirpated as breeders from the Klamath Basin since the early 1990s (Summers 1993, Cull and Hall 2007, R. Ekstrom and K. Spencer fide W. D. Shuford). Burrowing Owls currently nest in small numbers in the Honey Lake basin of Lassen County and in the Plumas County portion of Sierra Valley, and they have been reported from most other large valleys in the region, including Big Valley, Lassen and Modoc counties, and at Modoc NWR and Surprise Valley in Modoc County (Cull and Hall 2007, E. Hall in litt.).

Central and southern coast. The Burrowing Owl has declined in Monterey County, with small populations remaining near Salinas and King City (Roberson 2002). It has been nearly extirpated as a breeding species from coastal San Luis Obispo, Santa Barbara, Ventura, Los Angeles, and Orange counties (Comrack and Mayer 2003); historic population sizes are not known. The San Diego region has apparently seen steady declines of owls, down from possibly sizable populations less than a century ago (Willett 1933, Unitt 2004). Elsewhere on the coastal slope, small numbers persist at scattered sites, many of which are threatened by further development. The largest numbers remaining in this region appear to be the minimum of 350 pairs known to be breeding in Riverside and San Bernardino counties, collectively (G. Short pers. comm.), followed by a lesser number in San Diego County (Unitt 2004). Sites occupied include the vicinity of San Bernardino, Chino, and Ontario, San Bernardino County; near Perris, Lakeview

(San Jacinto WA), Winchester, French Valley, Temecula, and the Pechanga Indian Reservation, Riverside County; and two military bases in San Diego, Otay Mesa, and Warner Valley, San Diego County (Unitt 2004, Calif. Nat. Diversity Database unpubl. data). Both the historic and recent status are unclear on the Channel Islands, but breeding has been documented in recent years only on Santa Barbara and Santa Catalina islands (Collins and Jones in press).

Southern deserts. Burrowing Owls occur across most of the Mojave and Colorado deserts of Inyo, eastern Kern, northern Los Angeles, San Bernardino, eastern Riverside, eastern San Diego, and Imperial counties (Miller 2003, references therein). Garrett and Dunn (1981) described the species as "quite scarce" from Inyo County south through the eastern Mojave Desert. Overall, regional numbers are low and occupied areas are widely scattered, which is likely typical for this species in desert systems.

By contrast, numbers have increased greatly with the expansion of agriculture, particularly in the Imperial Valley and apparently along the lower Colorado River, where the species was not reported prior to the advent of large-scale agriculture early in the 20th century (Rosenberg et al. 1991). An estimated 5600 pairs (95% confidence interval: 3405–7795) nested in the Imperial Valley during 1992 and 1993 (DeSante et al. 2004), and approximately 250 pairs nested in the Palo Verde Valley near the Colorado River in Riverside County during 2001–2002 (J. Kidd in litt.).

Ecological Requirements

The Burrowing Owl is primarily a grassland species, but it persists and even thrives in some landscapes highly altered by human activity (Thomsen 1971, Haug et al. 1993, Millsap 2002, Gervais et al. 2003, Rosenberg and Haley 2004). The overriding characteristics of suitable habitat appear to be burrows for roosting and nesting and relatively short vegetation with only sparse shrubs and taller vegetation (Green and Anthony 1989, Haug et al. 1993). Owls in agricultural environments nest along roadsides and water conveyance structures (open canals, ditches, drains) surrounded by crops (DeSante et al. 2004, Rosenberg and Haley 2004). Burrowing Owls often nest near and under runways and associated structures (Thomsen 1971, Gervais et al. 2003). In urban areas such as much of Santa Clara County, Burrowing Owls persist in low numbers in highly developed parcels, such as Moffett Federal Airfield, in busy urban parks, and adjacent to roads with heavy traffic (Trulio 1997, D. K. Rosenberg pers. obs.).

Nest and roost burrows of the Burrowing Owl in California are most commonly dug by ground squirrels (e.g., *Spermophilus beecheyi*; Trulio 1997, D. K. Rosenberg et al. unpubl. data), but they may use Badger (*Taxidea taxus*), Coyote (*Canis latrans*), and fox (e.g., San Joaquin Kit Fox, *Vulpes macrotis mutica*) dens or holes (Ronan 2002). Because the owls may excavate their own burrows in the soft earthen banks of the ditches and canals in the Imperial Valley (D. K. Rosenberg et al. unpubl. data), availability of burrows may not limit population size in that region. Owls in the Imperial Valley also use the small holes of Round-tailed Ground Squirrels (*Citellus tereticaudus*) and Botta's Pocket Gophers (*Thomomys bottae*) as "starts" (Coulombe 1971, Rosenberg and Haley 2004). Structures such as culverts, piles of concrete rubble, and pipes also are used as nest sites (Rosenberg et al. 1998). Nest boxes are often used by owls, and their installation may be an important management tool in California (e.g., Trulio 1995, Rosenberg et al. 1998).

The diet of Burrowing Owls in California includes a broad array of arthropods (centipedes, spiders, beetles, crickets, and grasshoppers), small rodents, birds, amphibians, reptiles, and carrion, similar to their diet rangewide (Thompson and Anderson 1988, Green et al. 1993, Plumpton and Lutz 1993, Gervais et al. 2000, York et al. 2002). Although insects dominate the diet numerically, vertebrates account for the majority of biomass in some regions (Green et al. 1993). In California, there is evidence that rodent populations, particularly those of California Voles (*Microtus californicus*), may greatly influence survival and reproductive success (Gervais and Anthony 2003, Gervais et al. 2006). Food limits the number of fledged young in some years and at some sites (Haley 2002). This is not surprising given the large clutch size (up to 14 eggs; Haug et al. 1993, Todd and Skilnick 2002).

During the breeding season, owls forage close to their burrows but have been recorded hunting up to 2.7 km away (Haug and Oliphant 1990, Gervais et al. 2003). Over 80% of foraging observations in agricultural areas of the southern San Joaquin and Imperial valleys occurred within 600 m of the nest burrow (Gervais et al. 2003, Rosenberg and Haley 2004). Home-range size is likely related to food abundance (Newton 1979), but this relationship is unclear for Burrowing Owls. Owls in Saskatchewan appeared to avoid cropland in a mixed landscape in two instances, and one owl avoided fallow land in the same study (Sissons et al. 2001); in the same region, owls avoided cropland in favor of grass-forb habitat (Haug and Oliphant 1990; but see Gervais et al. 2003 for methodological issues). Foraging owls in agricultural areas of California exhibited little or no selection for cover types; instead, foraging locations were best predicted by distance to nest (Gervais et al. 2003, Rosenberg and Haley 2004).

The Burrowing Owl is often considered a sedentary species (e.g., Thomsen 1971). A large proportion of adults show strong fidelity to their nest site from year to year, especially where resident, as in Florida (74% for females, 83% for males; Millsap and Bear 1997). In California, nest-site fidelity rates were 32%–50% in a large grassland and 57% in an agricultural environment (Ronan 2002, Catlin 2004, Catlin et al. 2005). Differences in these rates among sites may reflect differences in nest predation rates (Catlin 2004, Catlin et al. 2005). Despite the high nest fidelity rates, dispersal distances may be considerable for both juveniles (natal dispersal) and adults (post-breeding dispersal), but this also varied with location (Catlin 2004, Rosier et al. 2006). Distances of 53 km to roughly 150 km have been observed in California for adult and natal dispersal, respectively (D. K. Rosenberg and J. A. Gervais unpubl. data), despite the difficulty in detecting movements beyond the immediate study area (Koenig et al. 1996).

These large dispersal patterns likely were responsible for the lack of genetic differences among the three California populations that were analyzed for genetic structure (Korfanta et al. 2005). Although even Burrowing Owls from resident populations may disperse widely, inbreeding does occur (Johnson 1997a, Millsap and Bear 1997, D. K. Rosenberg et al. unpubl. data).

Threats

Habitat loss and degradation from rapid urbanization of farmland in the core areas of the Central and Imperial valleys is the greatest threat to Burrowing Owls in California. Ongoing urbanization in coastal regions, changes in agricultural practices, and continuing eradication of ground squirrels are also serious threats.

The importance of habitat loss is emphasized by the fact that most owl populations suffering either extirpation or drastic reduction have been in coastal counties that experienced tremendous urbanization in recent decades. The human population of the Central Valley alone is projected to reach well over 10 million by 2040; this valley is considered among the most threatened of all U.S. farmland regions (American Farmland Trust, www.farmland.org/programs/states/ca/default.asp). Loss of agricultural and other open lands will negatively affect owls. Because of their need for open habitat with low vegetation, Burrowing Owls also are unlikely to persist in agricultural lands dominated by vineyards and orchards. They nest in some of California's urban environments, but in Florida, areas with higher densities of development supported fewer owls and were correlated with lower rates of nest success (Millsap and Bear 2000). However, urban development at moderate levels appeared to benefit owls by increasing prey availability (arthropods and lizards) near homes and reducing mortality from natural causes (Millsap and Bear 2000, Millsap 2002). This pattern may hold for California, but presently this is not known.

In addition to loss of nesting burrows from extermination of ground squirrels, developed environments pose a substantial risk to Burrowing Owls from mortality caused by traffic (Klute et al. 2003, D. K. Rosenberg et al. unpubl. data). Owls nesting along roadsides or parking lots are at greatest risk, although owls foraged along roads over 1 km from the nest burrow (Gervais et al. 2003). Wind turbines are a potential population-level threat to Burrowing Owls at Altamont Pass (Thelander et al. 2003), but sites appropriate for wind development will not be located in the lowland habitats where most Burrowing Owls occur. Migrating owls may be at risk, but this must be evaluated on a case-by-case basis, as many factors influence risk (e.g., Drewitt and Langston 2006). Burrowing Owl migration routes and patterns are still poorly understood. High-voltage electrical fences around prisons have caused mortality locally in the Imperial Valley (D. K. Rosenberg et al. unpubl. data), but the implications for populations are unknown.

Pesticides may affect Burrowing Owl populations in croplands and rangelands (James and Fox 1987, James et al. 1990). In the southern San Joaquin Valley, however, there was no indication that foraging owls either selected or avoided fields recently treated for pesticides, although owls did use crops extensively for foraging (Gervais et al. 2003). Although some individuals may be affected by persistent pesticides (Gervais et al. 2000, Gervais and Catlin 2004), the owls' high densities and strong demographic rates provide evidence that pesticide impacts overall are not sufficient to offset the benefits of nesting in agricultural regions (Gervais and Anthony 2003, Rosenberg and Haley 2004, D. K. Rosenberg et al. unpubl. data). Pesticide impacts may be mediated by environmental conditions, however. Gervais and Anthony (2003) found that body burdens of DDE were associated with declines in productivity only during a year of prey scarcity. Although the proportion of the population affected was small, changes in prey abundance in the future or other stresses conld modify the impact of DDE (Gervais et al. 2006).

Farming practices are likely a greater threat to Burrowing Owls in agricultural environments. Discing to control weeds in fallow fields may destroy burrows (Rosenberg and Haley 2004). Road and ditch maintenance in agricultural areas poses a threat to both owls and their nests, but these impacts can be minimized through management actions (Catlin and Rosenberg 2006). Burrowing Owls in the Imperial Valley may be affected by proposed plans to line ditches and fallow fields to increase water supplies to nrban areas, and by efforts to alleviate increasing salinity in the Salton Sea (Molina and Shuford 2004).

Emerging diseases such as West Nile virus may be significant threats to Burrowing Owl populations, but few data currently exist. Given that West Nile virus is known to be particularly virulent in raptors, concern seems warranted as West Nile virus expands in California.

Management and Research Recommendations

- Develop a conservation strategy with specific population goals, desired densities, and distribution that can be modified as more information is gained. Use risk-assessment modeling to identify populations critical for regional persistence.
- Place sizable tracts of grassland under conservation easements or agreements with agricultural (grazing) operations to maintain populations through best management practices, such as the elimination or restriction of small mammal poisoning.
- Also seek conservation agreements with landowners of row-crop agricnlture to encourage appropriate management of water conveyance structures, roadsides, and field margins. It will be necessary to work closely with landowners to alleviate concerns that maintaining owls on their property is a liability in terms of flexibility in land management practices necessary to maintain economic viability.
- Maintain suitable vegetation structure through mowing, revegetation with low-growing and less dense native plants, or controlled grazing, as appropriate.
- Where nesting burrows are lacking, enhance habitat by using artificial burrows or encouraging the presence of ground squirrels.
- Control off-road vehicles and unleashed pets within occupied Burrowing Owl habitat.
- Develop prescriptions that mimic natural processes and that preferably do not require ongoing management for maintaining Burrowing Owls.
- Develop gnidelines for maintaining Burrowing Owls and their burrows during management of agricultural water conveyance structures.
- Assess various strategies for maintaiuing owl populations in urbanizing areas.
- Determine owl distribution and abundance in publicly owned grasslands and other sites of known or likely occurrence that have not yet been well characterized.
- Assess the risk Burrowing Owls pose to aircraft operations safety, and develop management guidelines for owls at airports where they occur.
- Conduct research examining the factors that attract owls, and maintain them in locations from which populations were previously extirpated. In particular, rigorously evaluate translocation to determine when, if ever, it is an effective management tool.
- Determine patterns of long-distance dispersal.
- Identify the magnitude and source of wintering populations.

Monitoring Needs

Monitoring of changes in the abundance or demographic rates of Burrowing Owls should be linked with efforts both to identify the causes of any declines and to assess the response of the population to management actions (Noon 2003). Management strategies, and thus monitoring efforts, should be region-specific to account for the varied threats each region faces. Areas of the state with declining populations for which potential causes have been identified (such as urbanization) should have priority in the design and implementation of conservation strategies, whose effectiveness should be evaluated with subsequent monitoring. Monitoring itself can be effective only when population goals have been identified and the monitoring strategy evaluated to ensure that it is sufficiently sensitive to detect population changes considered noteworthy for management.

Effective methods for estimating actual or relative abundance of this species are clearly habitat specific. For example, call surveys have been effective in extensive grasslands (Haug and Didiuk 1993, Ronan 2002, Conway and Simon 2003), whereas counts of owls along edges of farm fields from vehicles are very effective in intensive agricultural areas (Rosenberg and Haley 2004). Methods that use counts need to account for the variable probability of detection among habitats if patterns of distribution and change are to be inferred from surveys. Data from large-scale surveys such as the BBS should be critically evaluated to identify regional patterns within California and to assess the effectiveness of this monitoring approach given the often small numbers of owls detected and the inconsistent observer effort.

Acknowledgments

T. Heindel, K. Hunting, B. Johnson, and D. Roberson reviewed earlier drafts of this manuscript. G. Short (Univ. Calif. Riverside, Dept. Biol., Ctr. for Conserv. Biol.) generously shared unpublished data.

Literature Cited

Brenckle, J. F. 1936. The migration of the Western Burrowing Owl (*Speotyto cunicularia hypogaea*). Bird-Banding 7:166–168.

California Department of Fish and Game (CDFG). 1992. Bird species of special concern. Unpublished list, July 1992, Calif. Dept. Fish & Game, 1416 Ninth St., Sacramento, CA 95814.

Catlin, D. H. 2004. Factors affecting within-season and between-season breeding dispersal of Burrowing Owls in California. M.S. thesis, Oregon State Univ., Corvallis.

Catlin, D. H., and Rosenberg, D. K. 2006. Nest destruction increases mortality and dispersal of Burrowing Owls in the Imperial Valley, California. Southwest Nat. 51:406–409.

Catlin, D. H., Rosenberg, D. K., and Haley, K. L. 2005. The effects of nesting success and mate fidelity on breeding dispersal in Burrowing Owls. Can. J. Zool. 83:1574–1580.

Collins, P. W., and Jones, H. L. In press. Birds of California's Channel Islands: Their Status and Distribution. Santa Barbara Mus. Nat. Hist., Santa Barbara, CA.

Comrack, L., and Mayer, D. 2003. An evaluation of the petition to list the California population of the Western Burrowing Owl (*Athene cunicularia hypugaea*) as an endangered or threatened species. Unpublished report, Calif. Dept. Fish & Game, 4949 Viewridge Ave., San Diego, CA 92123.

Conway, C. J., and Simon, J. C. 2003. Comparison of detection probability associated with Burrowing Owl survey methods. J. Wildl. Mgmt. 67:501–511.

Coulombe, H. N. 1971. Behavior and population ecology of the Burrowing Owl, *Speotyto cunicularia*, in the Imperial Valley of California. Condor 73:162–176.

Cull, R. L., and Hall, F. 2007. Status of Burrowing Owls in northeastern California, in Proceedings of the California Burrowing Owl Symposium, November 2003 (J. H. Barclay, K. W. Hunting, J. L. Lincer, J. Linthicum, and T. A. Roberts, eds.), pp. 42–51. Bird Populations Monogr. 1. The Institute for Bird Populations and Albion Environmental, Inc.

DeSante, D. F., Ruhlen, E. D., Adamany, S. L., Burton, K. M., and Amin, S. 1997a. A census of Burrowing Owls in central California in 1991. Raptor Res. Rep. 9:38–48.

DeSante, D. F., Ruhlen, E. D., and Rosenberg, D. K. 1997b. The distribution and relative abundance of Burrowing Owls in California: Evidence for a declining population. Institute for Bird Populations (Contr. 58), P.O. Box 1346, Pt. Reyes Station, CA 94956.

DeSante, D. F., Ruhlen, E. D., and Rosenberg, D. K. 2004. Density and abundance of Burrowing Owls in the agricultural matrix of the Imperial Valley, California. Studies Avian Biol. 27:116–119.

DeSante, D. F., Ruhlen, E. D., and Scalf, R. 2007. The distribution and relative abundance of Burrowing Owls in California during 1991–1993: Evidence for a declining population and thoughts on its conservation, in Proceedings of the California Burrowing Owl Symposium, November 2003 (J. H. Barclay, K. W. Hunting, J. L. Lincer, J. Linthicum, and T. A. Roberts, eds.), pp. 1–41. Bird Populations Monogr. 1. The Institute for Bird Populations and Albion Environmental, Inc.

Desmond, M. J., Parsons, T. J., Powers, T. O, and Savidge, J. A. 2001. An initial examination of mitochondrial DNA structure in Burrowing Owl populations. J. Raptor Res. 35:274–281.

Drewitt, A. L, and Langston R. H. W. 2006. Assessing the impacts of wind farms on birds. Ibis 148:29–42.

Duxbury, J. M. 2004. Stable isotope analysis and the investigation of the migrations and dispersal of Peregrine Falcons (*Falco peregrinus*) and Burrowing Owls (*Athene cunicularia hypugaea*). Ph.D. dissertation, Univ. Alberta, Edmonton.

Garrett, K., and Dunn, J. 1981. The Birds of Southern California: Status and Distribution. Los Angeles Audubon Soc., Los Angeles.

Gervais, J. A., and Anthony, R. G. 2003. Chronic organochlorine contaminants, environmental variability, and demographics of a Burrowing Owl population. Ecol. Applications 13:1250–1262.

Gervais, J. A., and Catlin, D. H. 2004. Temporal patterns of DDE in Burrowing Owl eggs from the Imperial Valley, California. Southwest Nat. 49:509–512.

Gervais, J. A., Hunter, C. M., and Anthony, R. G. 2006. Interactive effects of prey and *p,p'*DDE on Burrowing Owl population dynamics. Ecol. Applications 16:666–677.

Gervais, J. A., Rosenberg, D. K., and Anthony, R. G. 2003. Space use and pesticide exposure risk of male Burrowing Owls in an agricultural landscape. J. Wildl. Mgmt. 67:156–165.

Gervais, J. A., Rosenberg, D. K., Fry, D. M., Trulio, L., and Sturm, K. K. 2000. Burrowing Owls and agricultural pesticides: Evaluation of residues and risks for three populations in California. Environ. Toxicol. and Chem. 19:337–343.

Green, G. A., and Anthony, R. G. 1989. Nesting success and habitat relationships of Burrowing Owls in the Columbia basin, Oregon. Condor 91:347–354.

Green, G. A., Fitzner, R. E., Anthony, R. G., and Rogers, L. E. 1993. Comparative diets of Burrowing Owls in Oregon and Washington. Northwest Sci. 67:88–93.

Grinnell, J., and Miller, A. H. 1944. The distribution of the birds of California. Pac. Coast Avifauna 27.

Grinnell, J., and Wythe, M. W. 1927. Directory to the bird-life of the San Francisco Bay region. Pac. Coast Avifauna 18.

Haley, K. L. 2002. The role of food limitation and predation on reproductive success of Burrowing Owls in southern California. M.S. thesis, Oregon State Univ., Corvallis.

Haug, E. A., and Didiuk, A. B. 1993. Use of recorded calls to detect Burrowing Owls. J. Field Ornithol. 64:188–194.

Haug, E. A., Millsap, B. A., and Martell, M. S. 1993. Burrowing Owl (*Speotyto cunicularia*), in The Birds of North America (A. Poole and F. Gill, eds.), no. 61. Acad. Nat. Sci., Philadelphia.

Haug, E. A., and Oliphant, L. W. 1990. Movements, activity patterns, and habitat use of Burrowing Owls in Saskatchewan. J. Wildl. Mgmt. 54:27–35.

James, P. C. 1992. Where do Canadian Burrowing Owls spend the winter? Blue Jay 50:93–95.

James, P. C., and Ethier, T. J. 1989. Trends in the winter distribution and abundance of Burrowing Owls in North America. Am. Birds 43:1224–1225.

James, P. C., and Fox, G. A. 1987. Effects of some insecticides on productivity of Burrowing Owls. Blue Jay 45:65–71.

James, P. C., Fox, G. A., and Ethier, T. J. 1990. Is the operational use of strychnine to control ground squirrels detrimental to Burrowing Owls? J. Raptor Res. 24:120–123.

Johnson, B. S. 1997a. Characterization of population and family genetics of the Burrowing Owl by DNA fingerprinting with pV47 2. Raptor Res. Rep. 9:58–63.

Johnson, B. S. 1997b. Demography and population dynamics of the Burrowing Owl. Raptor Res. Rep. 9:28–33.

Klute, D. S., Ayers, A. W., Green, M. T., Howe, W. H., Jones, S. L., Shaffer, J. A., Sheffield, S. R., and Zimmerman, T. S. 2003. Status assessment and conservation plan for the Western Burrowing Owl in the United States. Biological Tech. Publ. FWS/BTP-R6001-2003, U.S. Fish & Wildl. Serv., Washington, DC. Available at http://library.fws.gov/BTP/westernburrowingowl03.pdf.

Koenig, W. D., Van Vuren, D. D., and Hooge, P. N. 1996. Detectability, philopatry, and the distribution of dispersal distances in vertebrates. Trends Ecol. and Evol. 11:514–517.

Korfanta, N. M., McDonald, D. B., Glenn, T. C., and Handel, C. M. 2005. Burrowing Owl (*Athene cunicularia*) population genetics: A comparison of North American forms and migratory habits. Auk 122:464–478.

Miller, J. 2003. Petition to the State of California Fish and Game Commission and supporting information for listing the California population of the Western Burrowing Owl (*Athene cunicularia hypugaea*) as an endangered or threatened species under the California Endangered Species Act. Available from Ctr. Biol. Diversity, 1095 Market St., Suite 511, San Francisco, CA 94103 or at www.biologicaldiversity.org/swcbd/species/b-owl/index.html.

Millsap, B. A. 2002. Survival of Florida Burrowing Owls along an urban-development gradient. J. Raptor Res. 36:3 10.

Millsap, B. A., and Bear, C. 1997. Territory fidelity, mate fidelity, and dispersal in an urban-nesting population of Florida Burrowing Owls. Raptor Res. Rep. 9:91 98.

Millsap, B. A., and Bear, C. 2000. Density and reproduction of Burrowing Owls along an urban development gradient. J. Wildl. Mgmt. 64:33–41.

Molina, K. C., and Shuford, W. D. 2004. Introduction. Studies Avian Biol. 27:1–11.

Newton, I. 1979. Population Ecology of Raptors. T&AD Poyser Ltd., London.

Noon, B. R. 2003. Conceptual issues in monitoring ecological resources, in Monitoring Ecosystems: Interdisciplinary Approaches for Evaluating Ecoregional Initiatives (D. E. Busch and J. C. Trexler, eds.), pp. 27–72. Island Press, Washington, DC.

Plumpton, D. L., and Lutz, R. S. 1993. Prey selection and food habits of Burrowing Owls in Colorado. Great Basin Nat. 53:299–304.

Remsen, J. V., Jr. 1978. Bird species of special concern in California: An annotated list of declining or vulnerable bird species. Nongame Wildl. Invest., Wildl. Mgmt. Branch Admin. Rep. 78-1, Calif. Dept. Fish & Game, 1416 Ninth St., Sacramento, CA 95814.

Roberson, D. 2002. Monterey Birds, 2nd ed. Monterey Peninsula Audubon Soc., Carmel, CA.

Ronan, N. A. 2002. Habitat selection, reproductive success, and site fidelity of Burrowing Owls in a grassland ecosystem. M.S. thesis, Oregon State Univ., Corvallis.

Rosenberg, D. K., and Haley, K. L. 2004. The ecology of Burrowing Owls in the agroecosystem of the Imperial Valley, California. Studies Avian Biol. 27:120–135.

Rosenberg, D. K., Gervais, J. A., Ober, H., and DeSante, D. S. 1998. An adaptive management plan for the Burrowing Owl population at Naval Air Station Lemoore, California. Publication 95, Institute for Bird Populations, P.O. Box 1346, Pt. Reyes Station, CA 94956.

Rosenberg, K. V., Ohmart, R. D., Hunter, W. C., and Anderson, B. W. 1991. Birds of the Lower Colorado River Valley. Univ. Ariz. Press, Tucson.

Rosier, J. R., Ronan, N. A., and Rosenberg, D. K. 2006. Post-breeding dispersal of Burrowing Owls in an extensive California grassland. Am. Midland Nat. 155:162–167.

Sauer, J. R., Hines, J. E., and Fallon, J. 2005. The North American Breeding Bird Survey, results and analysis 1966–2004, version 2005.2. USGS Patuxent Wildl. Res. Ctr., Laurel, MD. Available at www.mbr-pwrc.usgs.gov/bbs/bbs.html.

Sauer, J. R., Schwartz, S., and Hoover, B. 1996. The Christmas Bird Count Home Page, version 95.1. Patuxent Wildl. Res. Ctr., Laurel, MD. Available at www.mbr-pwrc.usgs.gov/bbs/cbc.html.

Sissons, R. A., Scalise, K. L., and Wellicome, T. I. 2001. Nocturnal foraging and habitat use by male Burrowing Owls in a heavily cultivated region of southern Saskatchewan. J. Raptor Res. 35:304–309.

Summers, S. D. 1993. A birder's guide to the Klamath Basin. Klamath Basin Audubon Soc., Klamath Falls, OR.

Thelander, C. G., Smallwood, K. S., and Rugge, L. 2003. Bird risk behaviors and fatalities at the Altamont Pass Wind Resource Area. Report NREL/SR 500-33829, National Renewable Energy Laboratory, 1617 Cole Blvd., Golden, CO 80401-3393. Available at www.nrel.gov/docs/fy04osti/33829.pdf.

Thomsen, L. 1971. Behavior and ecology of Burrowing Owls on the Oakland Municipal Airport. Condor 73:177–192.

Thompson, C. D., and Anderson, S. H. 1988. Foraging behavior and food habits of Burrowing Owls in Wyoming. Prairie Nat. 20:23–28.

Todd, L. D., and Skilnick, J. 2002. Large clutch size of a Burrowing Owl, *Athene cunicularia*, found in Saskatchewan. Can. Field-Nat. 116:307–308.

Trulio, L. 1995. Passive relocation: A method to preserve Burrowing Owls on disturbed sites. J. Field Ornithol. 66:99–106.

Trulio, L. 1997. Burrowing owl demography and habitat use at two urban sites in Santa Clara County, California. Raptor Res. Rep. 9:84–89.

Unitt, P. 2004. San Diego County bird atlas. Proc. San Diego Soc. Nat. Hist. 39.

Willett, G. 1933. A revised list of the birds of southwestern California. Pac. Coast Avifauna 21.

York, M., Rosenberg, D. K., and Sturm, K. K. 2002. Diet and food-niche breadth of Burrowing Owls (*Athene cunicularia*) in the Imperial Valley, California. W. North Am. Nat. 62:280–287.

Protocols for Surveying and Evaluating Impacts to Special Status Native Plant Populations and Sensitive Natural Communities

STATE OF CALIFORNIA
CALIFORNIA NATURAL RESOURCES AGENCY
DEPARTMENT OF FISH AND WILDLIFE

DATE: March 20, 2018

TABLE OF CONTENTS

1. INTRODUCTION AND PURPOSE

The conservation of special status native plants and their habitats, as well as sensitive natural communities, is integral to maintaining biological diversity. The purpose of these protocols is to facilitate a consistent and systematic approach to botanical field surveys and assessments of special status plants and sensitive natural communities so that reliable information is produced and the potential for locating special status plants and sensitive natural communities is maximized. These protocols may also help those who prepare and review environmental documents determine when botanical field surveys are needed, how botanical field surveys may be conducted, what information to include in a botanical survey report, and what qualifications to consider for botanical field surveyors. These protocols are meant to help people meet California Environmental Quality Act (CEQA)[1] requirements for adequate disclosure of potential impacts to plants and sensitive natural communities. These protocols may be used in conjunction with protocols formulated by other agencies, for example, those developed by the U.S. Army Corps of Engineers to delineate jurisdictional wetlands[2] or by the U.S. Fish and Wildlife Service to survey for the presence of special status plants[3].

---

1 Available at: http://resources.ca.gov/ceqa

2 Available at: http://www.usace.army.mil/Missions/CivilWorks/RegulatoryProgramandPermits/techbio.aspx

3 U.S. Fish and Wildlife Service Survey Guidelines: https://www.fws.gov/sacramento/es/Survey-Protocols-Guidelines/

Department of Fish and Wildlife Trustee and Responsible Agency Mission

The mission of the California Department of Fish and Wildlife (CDFW) is to manage California's diverse wildlife and native plant resources, and the habitats upon which they depend, for their ecological values and for their use and enjoyment by the public. CDFW has jurisdiction over the conservation, protection, and management of wildlife, native plants, and habitat necessary to maintain biologically sustainable populations (Fish & G. Code, § 1802). CDFW, as trustee agency under CEQA Guidelines section 15386, provides expertise in reviewing and commenting on environmental documents and provides protocols regarding potential negative impacts to those resources held in trust for the people of California.

Certain species are in danger of extinction because their habitats have been severely reduced in acreage, are threatened with destruction or adverse modification, or because of a combination of these and other factors. The California Endangered Species Act (CESA) and Native Plant Protection Act (NPPA) provide additional protections for such species, including take prohibitions (Fish & G. Code, § 2050 *et seq.*; Fish & G. Code, § 1908). As a responsible agency, CDFW has the authority to issue permits for the take of species listed under CESA and NPPA if the take is incidental to an otherwise lawful activity; CDFW has determined that the impacts of the take have been minimized and fully mitigated; and the take would not jeopardize the continued existence of the species (Fish & G. Code, § 2081, subd. (b); Cal. Code Regs., tit. 14 § 786.9, subd. (b)). Botanical field surveys are one of the preliminary steps to detect special status plant species and sensitive natural communities that may be impacted by a project.

Definitions

Botanical field surveys provide information used to determine the potential environmental effects of proposed projects on special status plants and sensitive natural communities as required by law (e.g., CEQA, CESA, and federal Endangered Species Act (ESA)).

Special status plants, for the purposes of this document, include all plants that meet one or more of the following criteria:

- Listed or proposed for listing as threatened or endangered under the ESA or candidates for possible future listing as threatened or endangered under the ESA (50 C.F.R., § 17.12).
- Listed or candidates for listing by the State of California as threatened or endangered under CESA (Fish & G. Code, § 2050 et seq.)[4]. In CESA, "endangered species" means a native species or subspecies of plant which is in serious danger of becoming extinct throughout all, or a significant portion, of its range due to one or more causes, including loss of habitat, change in habitat, overexploitation, predation, competition, or disease (Fish & G. Code, § 2062). "Threatened species" means a native species or subspecies of plant that,

[4] Refer to current online published lists available at: https://nrm.dfg.ca.gov/FileHandler.ashx?DocumentID=109390&inline

although not presently threatened with extinction, is likely to become an endangered species in the foreseeable future in the absence of the special protection and management efforts required by CESA (Fish & G. Code, § 2067). "Candidate species" means a native species or subspecies of plant that the California Fish and Game Commission has formally noticed as being under review by CDFW for addition to either the list of endangered species or the list of threatened species, or a species for which the California Fish and Game Commission has published a notice of proposed regulation to add the species to either list (Fish & G. Code, § 2068).

- Listed as rare under the California Native Plant Protection Act (Fish & G. Code, § 1900 et seq.). A plant is rare when, although not presently threatened with extinction, the species, subspecies, or variety is found in such small numbers throughout its range that it may be endangered if its environment worsens (Fish & G. Code, § 1901).
- Meet the definition of rare or endangered under CEQA Guidelines section 15380, subdivisions (b) and (d), including:
    - Plants considered by CDFW to be "rare, threatened or endangered in California." This includes plants tracked by the California Natural Diversity Database (CNDDB) and the California Native Plant Society (CNPS) as California Rare Plant Rank (CRPR) 1 or 2[5];
    - Plants that may warrant consideration on the basis of declining trends, recent taxonomic information, or other factors. This may include plants tracked by the CNDDB and CNPS as CRPR 3 or 4[6].
- Considered locally significant plants, that is, plants that are not rare from a statewide perspective but are rare or uncommon in a local context such as within a county or region (CEQA Guidelines, § 15125, subd. (c)), or as designated in local or regional plans, policies, or ordinances (CEQA Guidelines, Appendix G). Examples include plants that are at the outer limits of their known geographic range or plants occurring on an atypical soil type.

Sensitive natural communities are communities that are of limited distribution statewide or within a county or region and are often vulnerable to environmental effects of projects. These communities may or may not contain special status plants or their

[5] See CNDDB's Special Vascular Plants, Bryophytes, and Lichens List for plant taxa with a CRPR of 1 or 2: https://nrm.dfg.ca.gov/FileHandler.ashx?DocumentID=109383&inline

[6] CRPR 3 plants (plants about which more information is needed) and CRPR 4 plants (plants of limited distribution) may warrant consideration under CEQA Guidelines section 15380. Impacts to CRPR 3 plants may warrant consideration under CEQA if sufficient information is available to assess potential impacts to such plants. Impacts to CRPR 4 plants may warrant consideration under CEQA if cumulative impacts to such plants are significant enough to affect their overall rarity. Data on CRPR 3 and 4 plants should be submitted to CNDDB. Such data aids in determining and revising the CRPR of plants. See CNDDB's Special Vascular Plants, Bryophytes, and Lichens List for plant taxa with a CRPR of 3 or 4: https://nrm.dfg.ca.gov/FileHandler.ashx?DocumentID=109383&inline

habitat. CDFW's *List of California Terrestrial Natural Communities*[7] is based on the best available information, and indicates which natural communities are considered sensitive at the current stage of the California vegetation classification effort. See the Vegetation Classification and Mapping Program (VegCAMP) website for additional information on natural communities and vegetation classification[8].

2. BOTANICAL FIELD SURVEYS

Evaluate the need for botanical field surveys prior to the commencement of any activities that may modify vegetation, such as clearing, mowing, or ground-breaking activities. It is appropriate to conduct a botanical field survey when:

- Natural (or naturalized) vegetation occurs in an area that may be directly or indirectly affected by a project (project area), and it is unknown whether or not special status plants or sensitive natural communities occur in the project area;
- Special status plants or sensitive natural communities have historically been identified in a project area; or
- Special status plants or sensitive natural communities occur in areas with similar physical and biological properties as a project area.

Survey Objectives

Conduct botanical field surveys in a manner which maximizes the likelihood of locating special status plants and sensitive natural communities that may be present. Botanical field surveys should be floristic in nature, meaning that every plant taxon that occurs in the project area is identified to the taxonomic level necessary to determine rarity and listing status. "Focused surveys" that are limited to habitats known to support special status plants or that are restricted to lists of likely potential special status plants are not considered floristic in nature and are not adequate to identify all plants in a project area to the level necessary to determine if they are special status plants.

For each botanical field survey conducted, include a list of all plants and natural communities detected in the project area. More than one field visit is usually necessary to adequately capture the floristic diversity of a project area. An indication of the prevalence (estimated total numbers, percent cover, density, etc.) of the special status plants and sensitive natural communities in the project area is also useful to assess the significance of a particular plant population or natural community.

Survey Preparation

Before botanical field surveys are conducted, the botanical field surveyors should compile relevant botanical information in the general project area to provide a regional

7 Available at: https://www.wildlife.ca.gov/Data/VegCAMP/Natural-Communities#natural%20communities%20lists

8 Available at: https://www.wildlife.ca.gov/Data/VegCAMP

context. Consult the CNDDB[9] and BIOS[10] for known occurrences of special status plants and sensitive natural communities in the project area prior to botanical field surveys. Generally, identify vegetation and habitat types potentially occurring in the project area based on biological and physical properties (e.g. soils) of the project area and surrounding ecoregion[11]. Then, develop a list of special status plants and sensitive natural communities with the potential to occur within the vegetation and habitat types identified. The list of special status plants with the potential to occur in the project area can be created with the help of the CNDDB QuickView Tool[12] which allows the user to generate lists of CNDDB-tracked elements that occur within a particular U.S. Geological Survey 7.5' topographic quad, surrounding quads, and counties within California. Resulting lists should only be used as a tool to facilitate the use of reference sites, with the understanding that special status plants and sensitive natural communities in a project area may not be limited to those on the list. Botanical field surveys and subsequent reporting should be comprehensive and floristic in nature and not restricted to or focused only on a list. Include in the botanical survey report the list of potential special status plants and sensitive natural communities that was created, and the list of references used to compile the background botanical information for the project area.

Survey Extent

Botanical field surveys should be comprehensive over the entire project area, including areas that will be directly or indirectly impacted by the project. Adjoining properties should also be surveyed where direct or indirect project effects could occur, such as those from fuel modification, herbicide application, invasive species, and altered hydrology. Surveys restricted to known locations of special status plants may not identify all special status plants and sensitive natural communities present, and therefore do not provide a sufficient level of information to determine potential impacts.

Field Survey Method

Conduct botanical field surveys using systematic field techniques in all habitets of the project area to ensure thorough coverage. The level of effort required per given area and habitat is dependent upon the vegetation and its overall diversity and structural complexity, which determines the distance at which plants can be identified. Conduct botanical field surveys by traversing the entire project area to ensure thorough coverage, documenting all plant taxa observed. Parallel survey transects may be necessary to ensure thorough survey coverage in some habitats. The level of effort should be sufficient to provide comprehensive reporting. Additional time should be allocated for plant identification in the field.

---

9 Available at: https://www.wildlife.ca.gov/Data/CNDDB

10 Available at: https://www.wildlife.ca.gov/Data/BIOS

11 Ecological Subregions of the United States, available at: http://www.fs.fed.us/land/pubs/ecoregions/toc.html

12 Available at: https://www.wildlife.ca.gov/Data/CNDDB/Maps-and-Data. When creating a list of special status plants with the potential to occur in a project area, special care should be taken to search all quads with similar geology, habitats, and vegetation to those found in the project area.

Timing and Number of Visits

Conduct botanical field surveys in the field at the times of year when plants will be both evident and identifiable. Usually this is during flowering or fruiting. Space botanical field survey visits throughout the growing season to accurately determine what plants exist in the project area. This usually involves multiple visits to the project area (e.g. in early, mid, and late-season) to capture the floristic diversity at a level necessary to determine if special status plants are present[13]. The timing and number of visits necessary to determine if special status plants are present is determined by geographic location, the natural communities present, and the weather patterns of the year(s) in which botanical field surveys are conducted.

Reference Sites

When special status plants are known to occur in the type(s) of habitat present in a project area, observe reference sites (nearby accessible occurrences of the plants) to determine whether those special status plants are identifiable at the times of year the botanical field surveys take place and to obtain a visual image of the special status plants, associated habitat, and associated natural communities.

Use of Existing Surveys

For some project areas, floristic inventories or botanical survey reports may already exist. Additional botanical field surveys may be necessary for one or more of the following reasons:

- Botanical field surveys are not current[14];
- Botanical field surveys were conducted in natural systems that commonly experience year to year fluctuations such as periods of drought or flooding (e.g. vernal pool habitats or riverine systems);
- Botanical field surveys did not cover the entire project area;
- Botanical field surveys did not occur at the appropriate times of year;
- Botanical field surveys were not conducted for a sufficient number of years to detect plants that are not evident and identifiable every year (e.g. geophytes, annuals and some short-lived plants);

[13] U.S. Fish and Wildlife Service Guidelines for Conducting and Reporting Botanical Inventories for Federally Listed, Proposed and Candidate Plants available at: https://www.fws.gov/sacramento/es/Survey-Protocols-Guidelines/

[14] Habitats, such as grasslands or desert plant communities that have annual and short-lived perennial plants as major floristic components may require yearly surveys to accurately document baseline conditions for purposes of impact assessment. In forested areas, however, surveys at intervals of five years may adequately represent current conditions. For forested areas, refer to "Guidelines for Conservation of Sensitive Plant Resources Within the Timber Harvest Review Process and During Timber Harvesting Operations", available at: https://nrm.dfg.ca.gov/FileHandler.ashx?DocumentID=116396&inline

- Botanical field surveys did not identify all plants in the project area to the taxonomic level necessary to determine rarity and listing status;
- Fire history, land use, or the physical or climatic conditions of the project area have changed since the last botanical field survey was conducted;
- Changes in vegetation or plant distribution have occurred since the last botanical field surveys were conducted, such as those related to habitat alteration, fluctuations in abundance, invasive species, seed bank dynamics, or other factors; or
- Recent taxonomic studies, status reviews or other scientific information has resulted in a revised understanding of the special status plants with potential to occur in the project area.

Negative Surveys

Adverse conditions from yearly weather patterns may prevent botanical field surveyor from determining the presence of, or accurately identifying, some special status plants in the project area. Disease, drought, predation, fire, herbivory or other disturbance may also preclude the presence or identification of special status plants in any given year. Discuss all adverse conditions in the botanical survey report[15].

The failure to locate a known special status plant occurrence during one field season does not constitute evidence that the plant occurrence no longer exists at a location, particularly if adverse conditions are present. For example, botanical field surveys over a number of years may be necessary if the special status plant is an annual or short-lived plant having a persistent, long-lived seed bank and populations of the plant are known to not germinate every year. Visiting the project area in more than one year increases the likelihood of detecting special status plants, particularly if conditions change. To further substantiate negative findings for a known occurrence, a visit to a nearby reference site may help ensure that the timing of botanical field surveys was appropriate.

3. REPORTING AND DATA COLLECTION

Adequate information about special status plants and sensitive natural communities present in a project area will enable reviewing agencies and the public to effectively assess potential impacts to special status plants and sensitive natural communities and will guide the development of avoidance, minimization, and mitigation measures. The information necessary to assess impacts to special status plants and sensitive natural communities is described below. For comprehensive, systematic botanical field surveys where no special status plants or sensitive natural communities were found, reporting

and data collection responsibilities for botanical field surveyor remain as described

[15] U.S. Fish and Wildlife Service Guidelines for Conducting and Reporting Botanical Inventories for Federally Listed, Proposed and Candidate Plants available at: https://www.fws.gov/sacramento/es/Survey-Protocols-Guidelines/

below, excluding specific occurrence information.

Special Status Plant and Sensitive Natural Community Observations

Record the following information for locations of each special status plant and sensitive natural community detected during a botanical field survey of a project area.

- The specific geographic locations where the special status plants and sensitive natural communities were found. Preferably this will be done by use of global positioning system (GPS) and include the datum[16] in which the spatial data was collected and any uncertainty or error associated with the data. If GPS is not available, a detailed map (1:24,000 or larger) showing locations and boundaries of each special status plant population and sensitive natural community in relation to the project area is acceptable. Mark occurrences and boundaries as accurately as possible;
- The site-specific characteristics of occurrences, such as associated species, habitat and microhabitat, structure of vegetation, topographic features, soil type, texture, and soil parent material. If a special status plant is associated with a wetland, provide a description of the direction of flow and integrity of surface or subsurface hydrology and adjacent off-site hydrological influences as appropriate;
- The number of individuals in each special status plant population as counted (if population is small) or estimated (if population is large);
- If applicable, information about the percentage of each special status plant in each life stage such as seedling, vegetative, flowering and fruiting;
- The density of special status plants, identifying areas of relatively high, medium and low density of each special status plant in the project area; and
- Digital images of special status plants and sensitive natural communities in the project area, with diagnostic features.

Special Status Plant and Sensitive Natural Community Documentation

When a special status plant is located, data must be submitted to the CNDDB. Data may be submitted in a variety of formats depending on the amount and type of data that is collected[17]. The most common way to submit data is the Online CNDDB Field Survey Form[18], or equivalent written report, accompanied by geographic locality information (GPS coordinates, GIS shapefiles, KML files, topographic map, etc.). Data submitted in digital form must include the datum[19] in which it was collected.

If a sensitive natural community is found in a project area, document it with a Combined

---

[16] NAD83, NAD27 or WGS84

[17] See https://www.wildlife.ca.gov/Data/CNDDB/Submitting-Data for information on acceptable data submission formats.

[18] Available at: https://www.wildlife.ca.gov/Data/CNDDB/Submitting-Data

[19] NAD83, NAD27 or WGS84

Vegetation Rapid Assessment and Relevé Field Form[20] and submit the form to VegCAMP[21].

Voucher Collection

Voucher specimens provide verifiable documentation of special status plant presence and identification and a scientific record. This information is vital to conservation efforts and valuable for scientific research. Collection of voucher specimens should be conducted in a manner that is consistent with conservation ethics, and in accordance with applicable state and federal permit requirements (e.g. scientific, educational, or management permits pursuant to Fish & G. Code, § 2081, subd. (a)). Voucher collections of special status plants (or possible special status plants) should only be made when such actions would not jeopardize the continued existence of the population. A plant voucher collecting permit[22] is required from CDFW prior to the take or possession of a state-listed plant for voucher collection purposes, and the permittee must comply with all permit conditions.

Voucher specimens should be deposited in herbaria that are members of the Consortium of California Herbaria[23] no later than 120 days after the collections have been made. Digital imagery can be used to supplement plant identification and document habitat. Record all relevant collector names and permit numbers on specimen labels (if applicable).

Botanical Survey Reports

Botanical survey reports provide an important record of botanical field survey results and project area conditions. Botanical survey reports containing the following information should be prepared whenever botanical field surveys take place, and should also be submitted with project environmental documents:

Project and location description

- A description of the proposed project;
- A detailed map of the project area that identifies topographic and landscape features and includes a north arrow and bar scale;
- A vegetation map of the project area using Survey of California Vegetation Classification and Mapping Standards[24] at a thematic and spatial scale that allows the display of all sensitive natural communities;
- A soil map of the project area; and

---

20 Available at: https://www.wildlife.ca.gov/Data/VegCAMP/Natural-Communities/Submit

21 Combined Vegetation Rapid Assessment and Releve Field Forms can be emailed to VegCAMP staff. Contact information available at: https://www.wildlife.ca.gov/Data/VegCAMP/Natural-Communities/Other-Info

22 Applications available at: https://www.wildlife.ca.gov/Conservation/Plants/Permits

23 A list of Consortium of California Herbaria participants is available at: http://ucjeps.berkeley.edu/consortium/participants.html

24 Available at: https://www.wildlife.ca.gov/data/vegcamp/publications-and-protocols

- A written description of the biological setting, including all natural communities; geological and hydrological characteristics; and land use or management history.

Detailed description of survey methodology and results

- Names and qualifications of botanical field surveyor(s);
- Dates of botanical field surveys (indicating the botanical field surveyor(s) that surveyed each area on each survey date), and total person-hours spent;
- A discussion of the survey preparation methodology;
- A list of special status plants and sensitive natural communities with potential to occur in the region;
- Description(s) of reference site(s), if visited, and the phenological development of special status plant(s) at those reference sites;
- A description and map of the area surveyed relative to the project area;
- A list of all plant taxa occurring in the project area, with all taxa identified to the taxonomic level necessary to determine whether or not they are a special status plant;
- Detailed data and maps for all special status plants and sensitive netural communities detected. Information specified above under the headings "Special Status Plant and Sensitive Natural Community Observations," and "Special Status Plant and Sensitive Natural Community Documentation," should be provided for the locations of each special status plant and sensitive natural community detected. Copies of all California Native Species Field Survey Forms and Combined Vegetation Rapid Assessment and Relevé Field Forms should be sent to the CNDDB and VegCAMP, respectively, and included in the project environmental document as an Appendix[25];
- A discussion of the potential for a false negative botanical field survey;
- A discussion of how climatic conditions may have affected the botanical field survey results;
- A discussion of how the timing of botanical field surveys may affect the comprehensiveness of botanical field surveys;
- Any use of existing botanical field surveys and a discussion of their applicability to the project;
- The deposition locations of voucher specimens, if collected; and
- A list of references used, including persons contacted and herbaria visited.

---

[25] It is not necessary to submit entire environmental documents to the CNDDB

Assessment of potential project impacts

- A discussion of the significance of special status plant populations in the project area considering nearby populations and total range and distribution;
- A discussion of the significance of sensitive natural communities in the project area considering nearby occurrences and natural community distribution;
- A discussion of project related direct, indirect, and cumulative impacts to special status plants and sensitive natural communities;
- A discussion of the degree and immediacy of all threats to special status plants and sensitive natural communities, including those from invasive species;
- A discussion of the degree of impact, if any, of the project on unoccupied, potential habitat for special status plants; and
- Recommended measures to avoid, minimize, or mitigate impacts to special status plants and sensitive natural communities.

4. BOTANICAL FIELD SURVEYOR QUALIFICATIONS

Botanical field surveyors should possess the following qualifications:

- Knowledge of plant taxonomy and natural community ecology;
- Familiarity with plants of the region, including special status plants;
- Familiarity with natural communities of the region, including sensitive natural communities;
- Experience with the CNDDB, BIOS, and Survey of California Vegetation Classification and Mapping Standards;
- Experience conducting floristic botanical field surveys as described in this document, or experience conducting such botanical field surveys under the direction of an experienced botanical field surveyor;
- Familiarity with federal, state, and local statutes and regulations related to plants and plant collecting; and
- Experience analyzing the impacts of projects on native plant species and sensitive natural communities.

5. SUGGESTED REFERENCES

Bonham, C.D. 1988. Measurements for terrestrial vegetation. John Wiley and Sons, Inc., New York, NY.

California Native Plant Society, Rare Plant Program. Most recent version. Inventory of rare and endangered plants (online edition). California Native Plant Society. Sacramento, CA. Available at: http://www.rareplants.cnps.org/.

California Native Plant Society. Most recent version. A manual of California vegetation. California Native Plant Society. Sacramento, CA. Available at: http://www.cnps.org/cnps/vegetation/manual.php.

California Department of Fish and Wildlife, California Natural Diversity Database. Most recent version. Special vascular plants, bryophytes and lichens list. Updated quarterly. Available at: https://nrm.dfg.ca.gov/FileHandler.ashx?DocumentID=109383&inline.

Elzinga, C.L., D.W. Salzer, and J. Willoughby. 1998. Measuring and monitoring plant populations. BLM Technical Reference 1730-1. U.S. Dept. of the Interior, Bureau of Land Management. Denver, Colorado. Available at: https://www.blm.gov/nstc/library/pdf/MeasAndMon.pdf.

Jepson Flora Project (eds.) Most recent version. Jepson eFlora. Available at: http://ucjeps.berkeley.edu/eflora/.

Leppig, G. and J.W. White. 2006. Conservation of peripheral plant populations in California. Madroño. 53:264-274.

Mueller-Dombois, D. and H. Ellenberg. 1974. Aims and methods of vegetation ecology. John Wiley and Sons, Inc. New York, NY.

U.S. Fish and Wildlife Service. 1996. Guidelines for conducting and reporting botanical inventories for federally listed plants on the Santa Rosa Plain. Sacramento, CA.

U.S. Fish and Wildlife Service. 1996. Guidelines for conducting and reporting botanical inventories for federally listed, proposed and candidate plants. Sacramento, CA.

Van der Maarel, E. 2005. Vegetation Ecology. Blackwell Science Ltd. Malden, MA.

INTAKE FORM AFTER APPROVAL, SEND COPIES TO: ☐ ACCESS ☐ ACP: ☐ LJL ☐ ACC ☐ JML

TYPE: REOPEN DATE OF CHANGE: 6/4/25 (For changes, fill in only the appropriate fields.)

CLIENT INFORMATION		
	FILE NAME First Solar Hearland Solar Project	PRIMARY FILE NUMBER 4602
	RESPONSIBLE PARTNER TAG ATTORNEY PARALEGAL RLL	LOCATION: South San Francisco FOLDER: Select one:
	CLASSIFICATION: GOV Government (env'l permit participation) IF OTHER, DESCRIBE: CURE CATEGORY: Solar PV	BILLING RATE Select one: If Other: $
	NAME OF PRIMARY CLIENT CURE NAME OF SECONDARY CLIENTS (INDICATE PERCENTAGE SPLIT AND MATTER ID)	CASE OPEN DATE
	ARE PRIMARY/SECONDARY CLIENTS NEW? No IF YES, ATTACH ENGAGEMENT LETTER.	DATE OF CALL/EMAIL
	BUSINESS ADDRESS (CITY, STATE, ZIP)	BUSINESS PHONE
	OFFICER	FAX PHONE
	CALLER	E-MAIL

BILLING	
	WILL CLIENT BE BILL PAYER? Yes IF NO, WHO WILL PAY THE BILL? (WRITTEN CONSENT MUST BE OBTAINED.) COPIES OF BILLS TO:

TRANSFER	
	THIS PROJECT WAS PREVIOUSLY: ☐ GENERAL INVESTIGATION FILE NUMBER AND FILE NAME _____. ☐ 00788 ☐ 00986 OR OTHER (SPECIFY) _____ ALL SUBFILES RELATED TO THIS PROJECT SHOULD BE DELETED AND THE CONTENTS TRANSFERRED TO THE PRIMARY FILE NUMBER AND FILE NAME SHOWN ABOVE THE FILE OPEN DATE IS THE SAME AS THE DATE THAT THE ORIGINAL FILE WAS OPENED ATTACH A COPY OF THE ORIGINAL INTAKE FORM.

LOBBYING		
	WILL THERE BE LOBBYING ACTIVITY? No IF YES, COMPLETE THIS SECTION.	
	CLIENT S INDUSTRY TRADE OR PROFESSIONAL ASSOCIATION	DESCRIPTION OF ASSOCIATION
	AGENCIES TO BE LOBBIED	EFFECTIVE DATE OF LOBBYING: PERIOD OF CONTRACT:
	LOBBYING PROCEEDINGS RELATED TO (74 character limit)	

MONITORING	
	WILL THERE BE MONITORING OF THE PROJECT PERMITTING STATUS? Yes IF YES, INSERT AGENCIES TO BE MONITORED AND MONITORING SCHEDULE: Per TAG (1/29/25) Check County website and CEQANet for project updates. Submit PRA request to County every 90 days.https://www.fresnocountyca.gov/Departments/Public-Works-and-Planning/divisions-of-public-works-and-planning/development-services-division/planning-and-land-use/environmental-impact-reports/eir-7564-heartland-hydrogen https://ceqanet.opr.ca.gov/2022100609
STAT	DESCRIPTION OF PROJECT

	CURRENT STATUS OF PROJECT Notice of project refinement released 12/16/24- Project is only the solar componment (hydrogen element case 5698 dropped) https://www.fresnocountyca.gov/files/sharedassets/county/v/1/public-works-and-planning/development services/environmental-impact-reports/heartland_informational_2024_1216.pdf NOP of DEIR released 10/27/22, Fresno County is lead agency https://ceqanet.opr.ca.gov/2022100609 As of 11/24/21: Per planner, they have received two CUPs and a Variance application - Unclassified Conditional Use Permit Nos. 3630 and 3631, Variance Application No. 4122 - "The agreements for the preparation of the EIR for the subject land-use permit applications are still being drafted by County staff. Currently, review of the draft agreements are occurring with senior staff and anticipate further review/revision by County Counsel. After County draft and review is complete, these agreements would be sent to the Consultant and Applicant for their review. As we are still in the early stages of the EIR process, staff does not have a timeline set for the completion and release of the EIR."
APPROVAL	FORM PREPARED BY Rachel L Levine BY: [signature] TANYA A GULESSERIAN

The importance of coccidioidomycosis as an occupational disease has increased in the southwestern United States. This report discusses the aspects of the disease in terms of its geography, the agent, occupation, dust conditions, and various other factors. A control program is outlined.

EXPOSURE FACTORS IN OCCUPATIONAL COCCIDIOIDOMYCOSIS

Lawrence L. Schmelzer, M.P.H., and Irving R. Tabershaw, M.D., F.A.P.H.A.

THE rapid and increasing influx of industry and agriculture into the southwestern United States has heightened the importance of coccidioidomycosis as an occupational disease. Before 1938, this disease was of little interest because relatively few clinical cases were recognized and the morbidity caused by primary infection was not appreciated. In that year, Dickson and Gifford,[1] reporting on several years of study, clearly established that the benign, primary form of the disease was an important cause of illness in the endemic areas, and that the disease is caused by inhalation of spores of Coccidioides immitis. During World War II, coccidioidomycosis was shown to be the cause of significant illness among soldiers in training at camps in the endemic areas. Studies by Smith, et al.,[2] showed that preventive measures, notably dust control, were effective in reducing the rate of infection and the seriousness of epidemics.

Epidemics have also been reported in susceptible groups of university personnel that entered endemic areas. In 1942, Davis, et al.,[3] reported infection in seven of 14 students and staff from Stanford University who made a field trip to the San Joaquin Valley. In 1954, four students from the University of California at Los Angeles contracted the disease in similar circumstances, and one student, not participating in the field trip, developed disease through the handling of contaminated specimens in the laboratory.[4] In 1962, 100 per cent infection was reported in a group of 16 persons from UCLA who participated in an archaeological field study near Los Banos, Calif.[4] Again in 1965 three students from UC Berkeley developed clinical disease after a field trip in the same general area.

Coccidioidomycosis ranks high among the infectious occupational diseases[5] as shown in Table 1. Further, the case fatality rate closely parallels that of tuberculosis as shown in Table 2.[6] These rates are based on reported clinically recognized cases. In both diseases, primary infection usually goes unnoticed. Fatality rates for both diseases are considerably less when based on total number of infections.

In spite of the fact that coccidioidomycosis is in most instances inapparent or mild, the disease causes significant dis-

Table 1—Number of disability cases of selected occupational diseases in California by fiscal year of report*

Disease	Number of disability cases 1962-1963	1963-1964	1964-1965	3-year total
Coccidioidomycosis	21	34	27	82
Tuberculosis	28	29	24	81
Anthrax, brucellosis, Q fever	11	13	13	37
Psittacosis	1	1	1	3
Tetanus	1	2	1	4

* From: Work Injuries in California, Quarterly Statistical Summary. State of California Department of Industrial Welfare, Division of Labor Statistics and Research.

ability in California workers. Although the 106 cases reported in six years[7] may not appear an unduly large number, the degree of disability in these cases is noteworthy (Table 3).

A large proportion required hospitalization and absence from work lasting weeks or months was not unusual. As late as 1957, coccidioidomycosis caused more disability at Williams Air Force Base in Arizona than any other disease including the upper respiratory infections.[8] While the average incidence of both infections was the same, the average disability of 34.6 days caused by coccidioidomycosis was seven times higher than that caused by upper respiratory infections.

Since it is not now possible to provide artificial immunity to those entering an endemic area and since susceptibility to coccidioidomycosis is essentially universal, the introduction of industrial or agricultural workers into endemic areas carries with it the responsibility of assessing the hazard of the disease to such populations. None of the exposure factors in the production of coccidioidomycosis is susceptible to control to the degree necessary to prevent infection entirely. Sufficient knowledge of the direct and predisposing causes of the disease, however, does exist so that it may be possible to reduce both the incidence of infection and its severity.

Geography

Coccidioides immitis has been reported only in the arid and semiarid regions of southwestern United States, in Mexico, Central America, Venezuela, and in the Chaco region of Argentina. The areas of endemicity roughly parallel the boundaries of the lower Sonoran Life Zone, which is characterized by scant rainfall, hot dry summers, alkaline soil, mild winters, sparce flora and fauna and, until recently, few human inhabitants (Figure 1).[9] The creosote bush, Larrea tridentata, is often considered a specific indicator of this life zone.

Evaluation of geography and ecology as exposure factors is complicated by the fact that areas within the lower Sonoran Life Zone may be free of C. immitis, and conversely small endemic areas may occur outside the zone. However, the potential of serious sequelae to infection is sufficient justification to consider any entry into suspected endemic areas as leading to exposure to the disease.

Infectious Agent

Spores of C. immitis are found in the first few inches of the soil and in larger numbers in the vicinity of rodent bur-

Table 2—Case fatality rates for coccidioidomycosis and tuberculosis in California 1960-1963*

	Case fatality rates† 1960	1961	1962	1963
Coccidioidomycosis	8.6	12.8	12.3	11.1
Tuberculosis	15.7	12.7	13.1	12.1

* From: California Public Health Statistical Report 1963, Part II Communicable Diseases. California State Department of Public Health.
† Case fatality rates are per 100 cases reported.

Table 3—Number of cases of occupational coccidioidomycosis reported in California during the period January, 1959, to March, 1965, by industry*

Industry		Cases reported
Agriculture		32
Animal husbandry	16	
Field crops	11	
Gardening	3	
Other	2	
Construction		39
Equipment operator	19	
Truck driver-mechanic	6	
Building trades	14	
Professional		22
Engineer	9	
Scientist	8	
Geologist	5	
Other and unknown		13
Total		106

* From: Summary of Reports of Occupationally Contracted Coccidioidomycosis 1959-1965. California State Department of Public Health, Bureau of Occupational Health.

rows.[10] These spores produce mycelial growth during the winter rains and, as the soil dries in the spring, arthrospores are again produced. Tests have shown that the concentration of arthrospores in the soil is highest at the end of the wet season and becomes lower as the dry season progresses. Season and rainfall patterns must therefore be considered in the evaluation of exposure potential for persons entering endemic zones. Importance of this has been shown by Smith, et al.,[2] in the San Joaquin Valley, and by Hugenholtz in a study of 13 years' experience at Williams Air Force Base in Arizona.[11] The average number of infections of base personnel was found to decrease during rainy months and to increase during the dry periods.

The highly infectious nature of C. immitis is illustrated by the fact that from seven to 15 arthrospores insufflated intranasally into mice causes infection and dissemination to the liver and spleen in 35 per cent to 40 per cent of susceptible

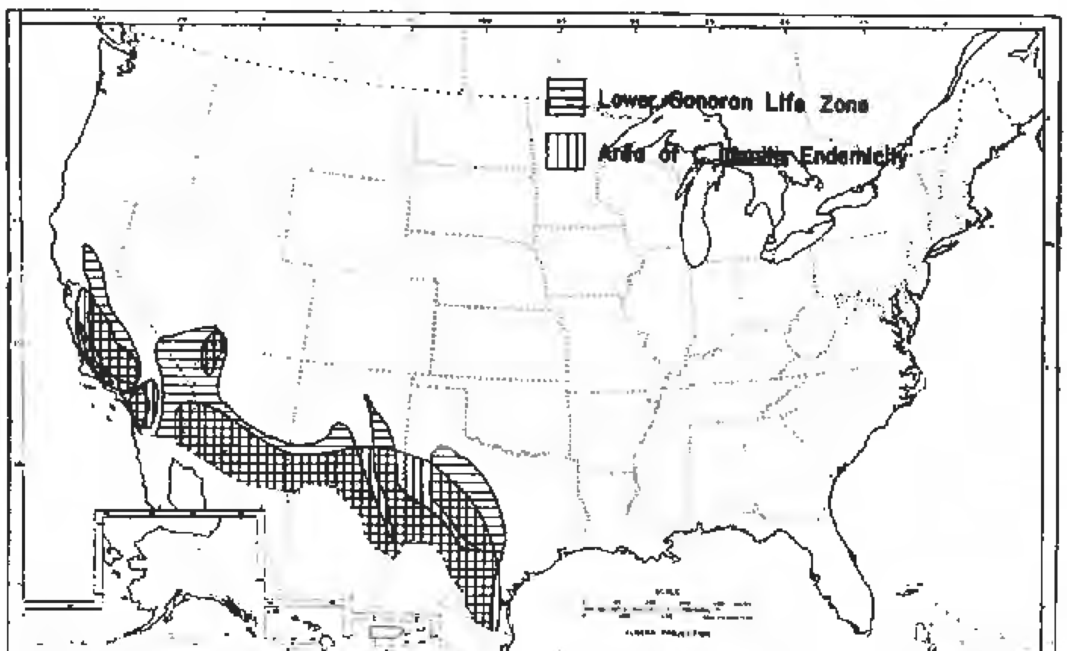


Figure 1—Lower Sonoran Life Zone and area of Coccidioides immitis endemicity in the United States [After Smith, C. E.[9]]

animals.[12] The organism has very simple nutritional requirements for growth, grows on practically any medium, and has been shown to prefer a saline environment[13] including body fluids.

Physical Properties

Typical mature hyphae of C. immitis yield barrel-shaped arthrospores, approximately 2.5 microns in diameter and 4 microns long, alternating with smaller sterile cells. The empty cells rupture easily to free the spores, leaving on the latter cell wall fragments which add to the length of the spore and also decrease the apparent specific gravity. Particle dynamics help to explain the highly infectious nature of the C. immitis and its wide distribution by winds. The important factors are terminal settling velocity and impingement forces, both of which are proportional to the particle size and specific gravity. Although actual spore dimensions vary and the specific gravity is not accurately known, it can be postulated that effective spore diameter is about 5 microns and its specific gravity is about 0.75. Terminal settling velocity for the spores is 0.01 centimeters per second when computed on the basis of these figures. In comparison, a quartz particle having this terminal settling velocity would have a diameter of 1.4 microns. From this it is clear that spores of C. immitis are easily air-borne, settle slowly, can penetrate into the smallest bronchioles and alveoli, and that a significant percentage of retention in the lung can be expected.

Dust Conditions

In the heat of early summer, what little ground cover that exists in the endemic areas withers and dies, winds disturb the surface dust and lift the spores into the air. The slow terminal settling velocity permits the spores to become essentially a permenent atmospheric contaminant under turbulent wind conditions. Such conditions are not unusual in arid regions where thermal phenomena generate severe atmospheric disturbances. Very small, intense, local whirlwinds, known as "dust devils," can raise dust containing large numbers of spores if they pass over pockets of high concentration in the soil. Large, rapidly moving air masses are also common, such as the "Santa Ana Winds" which blow from the Mojave Desert south into the San Fernando Valley. These winds will carry spores into nonendemic areas but the concentration will be low because of the nonselective raising of dust. Soil tests, therefore, cannot assure that an area within or close to an endemic zone is free of the organism and surface travel through or near endemic areas has resulted in exposure and infection.

Occupation

Varying racial and sexual susceptibility influences the severity and disability from coccidioidomycosis. However, since it results from inhalation of air-borne arthrospores, occupational factors must be considered in relation to the magnitude of probable dust exposure. It has been shown that a susceptible population entering an endemic area can experience an annual infection rate of about 20 per cent.[2] No overt dust exposure is necessary; infection can result from wind-borne spores traveling long distances in turbulent air conditions. Labor groups where occupation involves close contact with the soil are at greater risk, especially if the work involves dusty digging operations. The period of disability in cases of occupational coccidioidomycosis reported in California is classified by industry in Table 4.[7] The significant differences in the periods of disability can be ascribed to the variations in exposure resulting from occupation.

Agricultural workers suffered less disability because their exposure is probably to a few spores at a time. In field crop operations, burrowing rodents are not tolerated and the focus of endemicity associated with them is not present. Tilling of the soil will tend to disperse pockets of high spore concentration so that the dust raised can be expected to contain a relatively low concentration of spores. Similarly, a sheepherder would not be expected to receive a heavy, concentrated dose of arthrospores. This would tend to produce milder disease and a large proportion of inapparent and mild infections.

In the construction trades, exposures may be very different depending on the specific operations. Pipeline, highway, and utility construction often involves work in remote areas where the soil has not been disturbed and where foci of endemicity are usual. When these foci are disturbed, the dust raised can have a high concentration of spores. Digging of foundation and pipe trenches in residential or commercial buildings can lead to similar massive exposure. Similarly, engineers involved in highway or other heavy construction may be subjected to heavy doses if they are working with the construction crews, but may suffer exposure comparable to an agricultural worker if they are only surveying.

The exposures of professionals are highly variable and difficult to predict. Groups of paleontologists and archaeologists have suffered 100 per cent infection when their pursuits led them to dig in or around rodent burrows. Other groups digging in endemic areas have completely escaped infection.

Discussion

Prevention of coccidioidomycosis is complicated by the fact that the organism is a natural and persistent inhabitant of the environment. Determination of concentration of spores in specific locations is not feasible because the selection of appropriate sampling sites and identification of C. immitis is difficult and time-consuming. Furthermore, as previously mentioned, spores can be air-borne for long periods of time and travel great distances. Consequently, the importation of any susceptible labor force into endemic areas carries with it the responsibility for reducing the rate and severity of infection through whatever dust control measures are possible and for providing a vigorous program of medical surveillance.

Control of dust for the prevention of coccidioidomycosis is not a simple matter because of the wide variations in exposures. General dust control measures can afford some degree of protection to all persons working and living in an en-

Table 4—Number of disability cases of occupational coccidioidomycosis in California by length of disability and industry for the period January, 1959, to June, 1962

Industry	Period of disability in days					Total
	0	1–14	15–29	30–50	>60	
Agriculture	6	0	4	4	4	18
Construction	2	1	0	5	13	21
Professions	5	1	2	6	8	22

From: Summary of Reports of Occupationally Contracted Coccidioidomycosis, 1959-1965. California State Department of Public Health, Bureau of Occupational Health.

demic area. As shown by Smith,[2] oiling of parade grounds and barracks areas in military establishments reduced the rate of infection. Similarly, planting of trees and lawns around residences and industrial plants can reduce the rate of infection by about half.[14] Further protection can be provided by filtering and conditioning of air supplied to plants and offices, but this is not complete since it does not control infection resulting from exposure outside the working hours. Protection of agricultural workers and animal husbandmen to any realistic degree is exceedingly difficult. Their exposure to dust is an inseparable part of their employment and working conditions preclude the effective use of respiratory protection.

Operators of heavy earth moving equipment can be effectively protected during working hours by providing air conditioned cabs. This not only protects from coccidioidomycosis but also controls exposure to other dust, noise, and engine exhaust fumes. Efficient and comfortable hoods for individual use are now available with powered blowers for providing filtered air. These are useful on smaller earth moving equipment and for semistationary operations such as oil well drilling. Exposures resulting from manual digging are less easily controlled. Continued use of respirators is very uncomfortable in the usually high ambient temperatures, and workers resist use of this kind of protection. The wearing of respirators can, however, be enforced during recognized periods of high exposure. For instance, building tradesmen should wear respirators when digging foundation excavations or pipeline trenches. Similarly, highway engineers can wear respirators when working around earth moving machinery but could dispense with this when surveying ahead of or behind construction crews. Scientists should be protected during actual digging operations but not necessarily during exploration.

Skin testing for previous infection by C. immitis is easy to perform and defines the immune population. All persons hired for work in endemic areas (or whose assignments take them there) should be tested. Assigning immune workers to operations involving known heavy exposures can effectively reduce the incidence of infection. Hiring lifelong residents of the endemic areas can also reduce the incidence of infection since the level of immunity in these people can be expected to be high. This should not, however, be substituted for a program of skin testing and medical surveillance. Negroes and Filipinos have been shown to be more susceptible to developing the highly fatal disseminating form of the disease.[15] Unless such individuals are shown to have developed immunity, they should whenever possible be assigned to work in areas or at jobs where exposure to high concentrations of spores will be minimal.

Periodic medical examinations or interviews are useful to discover a history of low grade or subclinical infection and to evaluate the level of health of the individual. This examination must include repeated skin testing of susceptibles until the patient shows conversion to a positive reaction signifying immunity. Such an individual can then be dropped from medical surveillance for coccidioidomycosis. The medical management of any respiratory ailment suffered by persons at risk who are not immune to coccidioidomycosis should include a skin test.

Research is presently being pursued to develop an effective antigen for producing artificial active immunity to coccidioidomycosis. If successful, this vaccine will make possible the total protection of populations entering endemic areas. However, since man is not the reservoir of the disease, but only an accidental host, eradication will not be possible. Consequently the efforts to prevent disability from coccidioidomycosis must be continued so long as susceptible populations enter endemic areas.

Control Program

A program for limiting the incidence of occupational coccidioidomycosis and reducing the severity of disease in those who become infected would entail the following:

1. Determine if the work location is within the endemic area.
2. Hire resident labor whenever available, particularly if dust exposures may be heavy.
3. Establish a medical program including:
   a. Skin tests on all new employees. If positive they can be assigned to any job; if negative, especially Negroes and Filipinos, job exposure must be carefully evaluated. If heavy concentration of dust cannot be avoided, those with negative skin tests should not be employed at that job.
   b. Retest of susceptibles. This should be continued every three to six months until immunity is demonstrated by conversion to a positive reaction.
   c. Prompt treatment of respiratory illness in susceptibles. Coccidioidomycosis is a suspect in such illnesses (and if such is the case early chemotherapy can reduce the severity).
4. Educate the exposed population.
   a. New employees should be informed of the potential of infection and its consequences.
   b. All employees should be advised to seek prompt medical treatment for any respiratory illness and to inform the attending physician of their possible exposure to the fungus, particularly if the physician practices outside the endemic area.
5. Control dust exposure by:
   a. Oiling or planting of areas around plants, offices, and residences.
   b. Filtering and conditioning of air supplies to plants and offices; providing air conditioned cabs on heavy equipment.
   c. Providing respirators, air supplied helmets, and the like, as indicated.
   d. Preventing transport of C. immitis outside endemic area by thoroughly cleaning equipment and specimens before shipment to other work locations.

REFERENCES

1. Dickson, E. C., and Gifford, M. A. Coccidioides Infection (Coccidioidomycosis); the Primary Type of Infection. Arch. Int. Med. 62:852–871, 1938.
2. Smith, C. E.; Beard, R. R.; Rosenberg, H. G.; and Whiting, E. G. Effect of Season and Dust Control on Coccidioidomycosis. J.A.M.A. 132:833–838, 1946.
3. Davis, B. L.; Smith, R. T.; and Smith, C. E. An Epidemic of Coccidioidal Infection (Coccidioidomycosis). Ibid. 118:1182–1186, 1942.
4. Huberty, G. T. An Epidemic of Coccidioidomycosis. J. Am. College Health A. 12:131 (abstract), 1963.
5. State of California Department of Industrial Welfare. Work Injuries in California. Quart. Statist. Summary. State of California Department of Industrial Welfare. P. O. Box 965, San Francisco, Calif.
6. California State Department of Public Health. California Public Health Statistical Report, 1963, Part II, Communicable Diseases. California State Department of Public Health, 2151 Berkeley Way, Berkeley, Calif.
7. California State Department of Public Health. Summary of Reports of Occupationally Contracted Coccidioidomycosis 1959-1965. California State Department of Public Health, 2151 Berkeley Way, Berkeley, Calif.
8. Hugenholtz, P. G. Skin Test Survey at Williams Air Force Base, Arizona. Proceedings of Symposium on Coccidioidomycosis. PHS Publ. No. 575. Atlanta, Ga.: U. S. Public Health Service, 1957.
9. Smith, C. E. Diagnosis of Pulmonary Coccidioidal Infections. California Med. 75:385–391, 1951.
10. Elconin, A. F.; Egeberg, R. O.; and Lubarsky, R. Growth Patterns of Coccidioides immitis in the soil of endemic areas. Proceedings of Symposium of Coccidioidomycosis. PHS Publ. No. 575. Atlanta, Ga.: U. S. Public Health Service, 1957.
11. Hugenholtz, P. G. Climate and Coccidioidomycosis. Proceedings of Symposium on Coccidioidomycosis. PHS Publ. No. 575. Atlanta, Ga.: U. S. Public Health Service, 1957.
12. Kong, Y. M.; Levine, H. B.; Madin, S. H.; and Smith, C. E. Fungal Multiplication and Histopathological Changes in Vaccinated Mice Infected with Coccidioides immitis. J. Immunol. 92:779–790, 1964.
13. Elconin, A. F.; Egeberg, R. O.; and Egeberg, M. C. Significance of Soil Salinity on the Ecology of Coccidioides immitis. J. Bact. 87:500–503, 1964.
14. Maddy, K. T.; Doto, I. L.; Furcolow, M. L.; Lehan, P. H.; Rubin, H.; Greene, J. C.; Casper, E.; and Coleman, P. J. Coccidioidin, Histoplasmin, and Tuberculin Sensitivity of Students in Selected High Schools and Colleges in Arizona. Proceedings of Symposium on Coccidioidomycosis. PHS Publ. No. 575. Atlanta, Ga.: U. S. Public Health Service, 1957.
15. Gifford, M. A.; Buss, W. C.; and Douds, R. J. Data on Coccidioides Fungus Infection, Kern County, 1900-1936. Kern County Health Dept. Annual Rep. 1936-1937. Bakersfield, Calif.: Kern County Health Department.

Mr. Schmelzer is environmental health and safety officer, and Dr. Tabershaw is professor of occupational medicine and head, Environmental Health and Safety, Office of Environmental Health, University of California, Berkeley, Calif.

This paper was presented before the Occupational Health Section of the American Public Health Association at the Ninety-Fourth Annual Meeting in San Francisco, Calif., October 31, 1966.

Expanding Understanding of Epidemiology of Coccidioidomycosis in the Western Hemisphere

RAFAEL LANIADO-LABORIN[a,b]

[a]*Facultad de Medicina Tijuana, Universidad Autónoma de Baja, California, Mexico*

[b]*Hospital General de Tijuana, ISESALUD de Baja California, Tijuana, Mexico*

ABSTRACT: Coccidioidomycosis is a disease of both national and worldwide importance that is most often diagnosed in nonendemic regions. The endemic region for *Coccidioides* spp. lies exclusively in the Western Hemisphere. *Coccidioides* spp. has long been identified in semiarid areas of the United States and Mexico, and endemic foci have been described in areas of Central and South America. Infection is usually the result of activities that cause the fungus to become airborne and inhaled by a susceptible host. Underlying medical diseases that affect T cell function are known to increase the risk of disseminated disease and include human immunodeficiency virus, cancer, and disease processes requiring transplantation and its subsequent immunosuppressive agents. In recent years the incidence of the coccidioidomycosis has increased in California and Arizona, which may be partially due to the massive migration of Americans to the Sunbelt states. To date the highest number of cases reported in Arizona was in 2004, when a total of 3,665 cases of coccidioidomycosis was reported, representing a 281% increase since 1997. Statistics on the prevalence and incidence of coccidioidomycosis in Latin America either are fragmentary or simply are not available.

KEYWORDS: coccidioidomycosis; epidemiology; Western hemisphere

INTRODUCTION

Coccidioidomycosis is the oldest of the major mycoses.[1] The disease was described in 1892 and was first thought to be parasitic in nature.[2] It is caused by two nearly identical species, *Coccidioides (C.) immitis* and *C. posadasii*, generally referred as the "Californian" and non-Californian" species, respectively.[3] These two organisms are genetically different, but at this time they

Address for correspondence: Rafael Laniado-Laborin, M.D., M.P.H, F.C.C.P., Facultad de Medicina Tijuana, Universidad Autónoma de Baja, P.O. Box 436338, San Ysidro, CA 92143-6338. Voice/fax: 011-52-664-686-5626.
rafaellaniado@gmail.com

**Ann. N.Y. Acad. Sci. 1111: 19–34 (2007). © 2007 New York Academy of Sciences.
doi: 10.1196/annals.1406.004**

cannot be distinguished phenotypically nor is the disease or immune response to the organisms distinguishable.[4] This article discusses up to date issues in the epidemiology of coccidioidomycosis, as presented at the Sixth International Symposium on Coccidioidomycosis.

ECOLOGY

The endemic region for *Coccidioides* spp. lies exclusively in the Western Hemisphere, nearly all of it between the 40 latitudes north and south. This life zone corresponds with the hot deserts of the southwestern United States and northwestern Mexico (the Mojave, Sonoran, and Chihuahuan deserts). This region is situated below 4,500 feet where creosote (*Larrea tridentata*), jojoba, paloverde, mesquite, bursage, and cacti abound. The climate is arid with a yearly rainfall ranging from 10 to 50 cm, with extremely hot summers, winters with few freezes and alkaline, sandy soil.[5,6]

In the United States this semiarid zone encompasses the southern parts of Texas, Arizona, New Mexico, and much of central and southern California. Endemic regions have long been identified also in semiarid areas of Mexico and endemic foci have been described in areas of Central and South America (FIG. 1).[3]

Cases of coccidioidomycosis may also arise outside endemic areas. Such cases also occur because of a recent visit to an endemic area or infection through exposure to fomites from such an area.[7] In this setting the diagnosis is often delayed because the infection is not considered initially.[6]

RISK FACTORS FOR INFECTION AND DISEASE

Infection is usually the result of activities that cause the arthroconidia to become airborne and inhaled by a susceptible host. Coccidioidomycosis is not spread from person to person except in extraordinary circumstances. The main risk factors for acquiring the infection or developing active disease are discussed in the following subsections.

Exposure to Dust

Environmental conditions appear to have an important impact on coccidioidomycosis incidence. Some studies have identified associations linking climate and other factors to seasonal patterns of coccidioidomycosis and to interannual variability and trends in the disease. Significant variables included drought indices, precipitation, temperature, wind speed, and dust during the preceding one or more years.[8,9] Infection usually occurs during the dry season.

FIGURE 1. Geographic distribution of coccidioidomycosis. (From Hector and Laniado-Laborin.[80] Reproduced by permission.)

Because *Coccidioides* infects humans by the respiratory route, exposure to dust is one critical factor determining the risk of infection. The main risk factors for acquiring infection from *Coccidioides* spp. are activities that bring one into contact with dust from undisturbed soil in the endemic areas.[4] *Coccidioides* spp. are distributed unevenly in the soil and a majority of positive sites seem to be concentrated around animal burrows and ancient Indian burial sites. It is usually found 10 to 30 cm below the surface of the soil.[10,11]

Existing *Coccidioides* mycelia present in dry soil need increased soil moisture to grow, followed by a dry period during which fungal hyphae desiccate, mature, and form arthroconidia. Wind or other disturbance is required to fragment the hyphae and disperse the spores for inhalation by a host. On average, peaks in exposure to the fungal spores occur during the drier and dustier months of the year. Fewer exposures occur during the wetter and less dusty months.[12,13]

From 1991 through 1992 there was a dramatic increase in the number of cases of coccidioidomycosis reported from Kern County in the San Joaquin Valley, California, with 995 cases reported in 1991 and 3,027 cases in 1992.[14] After a 5-year drought in this region heavy rains fell in March 1991 and in February and March 1992. This increased precipitation may have brought on the germination of arthroconidia from mycelia accumulated over 5 years.

Dust storms in the endemic area are often followed by outbreaks of coccidioidomycosis. One particularly severe dust storm in 1977 carried dust from the San Joaquin Valley up to the San Francisco Bay area and resulted in hundreds of cases of nonendemic coccidioidomycosis in areas north of the San Joaquin Valley.[15]

Above this ambient risk occupational and recreational dust exposure as well as natural phenomena has occasionally caused outbreaks. Outbreaks of coccidioidomycosis have been described under several different circumstances: military maneuvers, construction work,[16] earthquakes,[17] model airplane competitions, and hunting (armadillo) expeditions.[18]

Coccidioidomycosis has long been and continues to be a threat to military personnel who reside or train in areas where *Coccidioides* spp. is endemic as the Army, Navy, Marines, and Air Force have traditionally deployed large numbers of personnel to endemic areas.[19] During World War II, when several training airfields were built in the San Joaquin Valley, California, coccidioidomycosis was the most common cause of hospitalization at many airbases in the southwest.[20] More recently, there was an outbreak of coccidioidomycosis among Navy SEALs during training exercises in Coalinga, California. Ten (45%) of 22 men had serologic evidence of acute coccidioidomycosis, the highest attack rate ever reported for a military unit. All patients were symptomatic, and 50% had abnormal chest radiographs.[19] Coccidioidomycosis must be considered an occupational disease that occurs with increased frequency among personnel exposed to the soil in endemic areas during military training.[21]

A coccidioidomycosis outbreak occurred in Ventura County, and was directly linked to dust clouds that emanated from landslides in the Santa Susanna Mountains caused by the Northridge earthquake in January 1994. In all, 170 cases were reported in a 7-week period following the earthquake. This outbreak is unusual in that Ventura County is not typically considered a hyperendemic area of coccidioidomycosis.[17]

Gender

Males are more often infected, which is likely related to occupational dust exposures; however, males also appear to be at a higher risk for dissemination, suggesting a hormonal or genetic component.[21] Drutz *et al*. studied the direct effect of human sex hormones and related compounds on the growth and maturation of *C. immitis in vitro*. 17β-estradiol, progesterone, and testosterone were highly stimulatory for the parasitic phase of *Coccidioides* spp. growth, whereas cholesterol, ergosterol, and 17α-estradiol (a physiologically inactive stereoisomer of 17β-estradiol), lacked such effects. Rates of spherule maturation and endospore release were accelerated, in a dose-dependent fashion, with the most striking effects seen at levels encountered in advanced pregnancy. A stimulatory effect of 17β-estradiol on the saprobic phase of fungal growth was also detected. This suggests that direct stimulation of *Coccidioides* spp. by human sex hormones may help to account for sex- and pregnancy-related predisposition to dissemination of coccidioidomycosis.[22]

Race

There is no known racial predilection for the acquisition of disease; however, disseminated disease occurs 10–175 times more often among Filipinos and African Americans. Whether Native Americans, Hispanics, or Asians have a higher risk is debated.[23] The 1977 dust storm in California provided a natural means of confirming this increased risk. The incidence of disseminated coccidioidomycosis in the non-Caucasian population was disproportionate to its overall representation.[1] During this wind-borne outbreak of coccidioidomycosis in the nonendemic disease region of Sacramento County, California, the rate per 100,000 of disseminated coccidioidomycosis among African American men compared with Caucasian men was 23.8 versus 2.5 (ratio 9.1:1). This difference could not be explained by differential exposure.[15] More recently, in the endemic area of Kern County, California, African American men had an adjusted odds ratio for disseminated coccidioidomycosis 28 times higher than that of any other ethnic group. The apparent variation in susceptibility among ethnic groups suggests that genetic factors influence the development of disseminated coccidioidomycosis.[1]

Although little is known about the role of T cells in eliminating *Coccidioides* spp., activated T cells elicit a delayed-type hypersensitivity (DTH) inflammatory response, indicating a Th1-type response. While DTH reactivity is regulated by class II HLA interactions with T cells, the host immune response to intracellular pathogens is primarily regulated by class I HLA molecules. Deresinski *et al.* found a significant association of blood group B and disseminated coccidioidomycosis. HLA-A9 and blood group B are both more common in persons of black and Filipino ancestry.[24]

Louie *et al.*[25] examined host genetic influences on coccidioidomycosis severity among class II HLA loci and the ABO blood group. Participants included African American, Caucasian, and Hispanic persons with mild or severe disseminated coccidioidomycosis. Among Hispanics, predisposition to symptomatic disease and severe disseminated disease is associated with blood types A and B, respectively. The HLA class II DRB1*1301 allele marks a predisposition to severe disseminated disease in each of the three groups. Reduced risk for severe disease is associated with DRB1 * 0301-DQB1 * 0201 among Caucasians and Hispanics and with DRB1 * 1501-DQB1 * 0602 among African Americans. These data support the hypothesis that host genes, in particular HLA class II and the ABO blood group, influence susceptibility to severe coccidioidomycosis.

Immunosuppression

Underlying medical diseases that affect T cell function are known to increase the risk of disseminated disease including human immunodeficiency virus (HIV), cancer (particularly Hodgkin's disease), and disease processes requiring transplantation and subsequent immunosuppressive agents.

Dissemination among patients with cancer appears to be related to the immunosuppressive effect of the chemotherapy rather than radiation therapy or the nature of the disease itself.[23]

Coccidioidomycosis is a recognized opportunistic infection among persons infected with HIV. The first reports of coccidioidomycosis associated with acquired immunodeficiency syndrome (AIDS) occurred just a few years after the initial reports of AIDS.[26]

A prospective study in the late 1980s revealed that almost 25% of a cohort of HIV-infected individuals living in coccidioidal-endemic region developed symptomatic coccidioidomycosis within 3.5 years of follow-up.[27] Two predictive variables for the development of coccidioidomycosis were a peripheral blood CD4 lymphocyte count of <250 cells/μL and a diagnosis of AIDS.[27]

Although nearly 50% of the cases of coccidioidomycosis occurring in persons with AIDS were found to be from the coccidioidal endemic area (>90% from Arizona or California), the rest were from all other regions in the United States.[28] Therefore, the diagnosis of coccidioidomycosis should be considered in any immunosuppressed HIV-infected patient presenting with a compatible clinical syndrome.[26]

Early in the HIV epidemic, most cases presented as overwhelming diffuse pulmonary disease with a high mortality rate.[29] The incidence of severe symptomatic coccidioidomycosis has declined dramatically since the advent of potent antiretroviral therapy. Although these cases are still seen, they are typically in patients with previously undiagnosed HIV infection and extremely low peripheral blood CD4 cell counts.[26]

Pregnancy

Pregnant women have long been considered to be at increased risk of developing severe or disseminated coccidioidomycosis, presumably because of a general depression in cell-mediated immunity or because of changes in the levels of hormones that stimulate the growth of the fungus.[22]

A recent review of the literature identified 81 cases of coccidioidomycosis in pregnancy. Disseminated disease was strongly associated with the trimester of pregnancy: 50% of the cases diagnosed in the first trimester, 62% of the cases diagnosed in the second trimester, and 96% of the cases diagnosed in the third trimester had dissemination. In addition, African American women had a 13-fold increased risk of dissemination compared to that of white women.[30]

However, another viewpoint suggests that higher dissemination and mortality rates in pregnancy are contrary to the experience of practitioners and academic physicians in endemic areas and further, that maternal death is rare. It has been hypothesized that reports of increased maternal morbidity and mortality rates might be artifacts of reporting bias, which have led to an inaccurate portrayal of the natural history of coccidioidomycosis in pregnancy.[31]

Age

Coccidioidomycosis occurs in all age groups. In general, the incidence rate increases with age; the extremes of age carry a higher risk for complicated disease, including chronic pulmonary infection and dissemination.[23]

Solid-Organ Transplantation

Coccidioidomycosis is the most common endemic mycosis to cause disease in solid-organ transplant patients in North America.[32] Underlying renal and liver disease, T lymphocyte suppression from antirejection medication, and activation of immunomodulating viruses, such as cytomegalovirus, all increase the risk for coccidioidomycosis among these patients. About one-half of all cases are the result of reactivation of previously acquired coccidioidal infection and occur during the first year after transplantation. Although disseminated disease is common, most of these patients manifest with pulmonary symptoms.[32] Coccidioidomycosis has been reported in patients who receive organs from donors infected with the fungus.[33]

Hemodyalisis for Chronic Renal Failure

Dialysis patients are at increased risk for fungal infections compared to the general population, which substantially decreases patient survival. In a study by Abbott *et al.* dialysis patients had an age-adjusted incidence ratio for fungal infections of 9.8 compared to the general population, with candidiasis accounting for 79% of all fungal infections, followed by cryptococcosis (6.0%) and coccidioidomycosis (4.1%).[34]

RECENT TRENDS OF COCCIDIOIDOMYCOSIS IN THE UNITED STATES

An estimated 150,000 new infections occur annually in areas of the southwestern United States. However, since coccidioidomycosis is not a nationally reportable disease (reportable only in Arizona and California), the exact incidence is unknown.

In recent years the incidence of the disease has increased in California and Arizona, which may be partially due to the massive migration of Americans to the Sunbelt states and, in particular, to Arizona, one of the fastest-growing states in the United States. The regions in Arizona in which *C. immitis* is most intensely endemic were previously sparsely populated and now contain major population centers, filled primarily with persons who have moved from areas where *C. immitis* was not endemic.[35] For example, Maricopa County (Phoenix) in 1950 had a population of 0.1 million;[35] in 2005 the estimated population had reached 3.6 million;[36] for Pima county (Tucson) population in 1950 was 0.1 million;[35] in 2005 it was estimated at 924,000.[36] Similar population expansion has also occurred in central California and west Texas.[35] As these populations have expanded in endemic areas, a growing segment of persons unusually susceptible to the most serious consequences of infection has also emerged.

In 1997 laboratory reporting of coccidioidomycosis became mandatory in Arizona. This was followed by a marked increase in the number of reported cases. To date the highest number of cases reported in Arizona was in 2004, when a total of 3,665 cases of coccidioidomycosis was reported (62.7 cases per 100,000 population), which represents a 281% increase since 1997 (958 cases). [37] From January to July 2006 the Arizona Department of Health Services reported 3,510 cases of coccidioidomycosis (compared to 1,425 cases during the same period in 2005; FIG. 2). [37]

Cases have recently been discovered outside areas previously identified as endemic, suggesting the endemic region may be wider than originally described.[23] In 2001 an outbreak of acute respiratory disease occurred among persons working at a Native American archeological site at Dinosaur National Monument in northeastern Utah. Ten workers met the clinical case definition; 9 had serologic confirmation of coccidioidomycosis, and 8 were hospitalized. All 10 were present during sifting of dirt through screens. This outbreak documents a new endemic focus of coccidioidomycosis, which extends northward its known geographic distribution in Utah by approximately 200 miles.[38]

COCCIDIOIDOMYCOSIS OUTSIDE THE ENDEMIC AREAS

Coccidioidomycosis is a disease of both national and worldwide importance that is often diagnosed in nonendemic regions, typically related to travel.[39] It

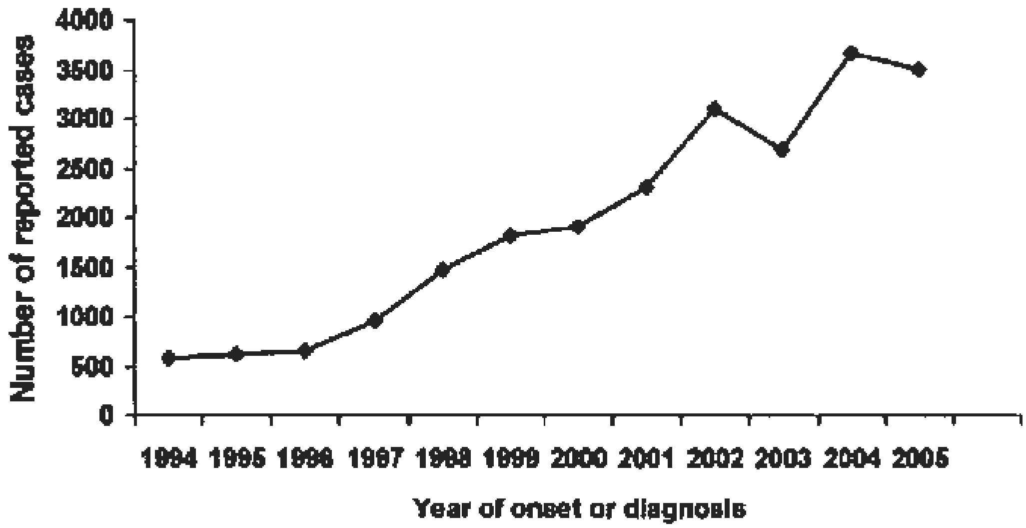


FIGURE 2. Reported cases of coccidioidomycosis, Arizona, 1994–2005. (Source: Arizona Department of Health Services, Infectious Disease Epidemiology Section.)

is usually diagnosed when individuals who live in a nonendemic region return home from visiting an endemic area.

For example, in July 1996 the Washington State Department of Health in Seattle was notified of a cluster of a flu-like, rash-associated illness in a 126-member church group. The group had recently returned from Tecate, Mexico, where members had assisted with construction projects at an orphanage. Eventually there were 21 serologically confirmed cases of coccidioidomycosis (attack rate, 17%) among this group.[16]

Chaturvedi *et al.*[39] reported that during a 5-year period (1992 1997), 161 persons in New York State had hospital discharge diagnoses of coccidioidomycosis, and from 1989 to 1997, 49 cultures from patients were confirmed as *C. immitis*; 26 of these patients had traveled to disease-endemic areas. Sixteen patient isolates were available for multilocus genotyping; all these patients had a history of travel to the Southwest, with 12 of 16 traveling to Arizona. Furthermore, while information on travel history was limited, all 16 patients from whom information was obtained had traveled to disease-endemic areas before becoming ill.

Coccidioidomycosis can create a clinical dilemma even in countries far away from the endemic areas. A 60-year-old Israeli resident traveled to Arizona, developed influenza-like infection, and returned to Israel with an airspace-occupying lesion in the lung. Since the patient was a heavy smoker, lung cancer was suspected and he was operated on. A granuloma with spherules was reported on stain preparations and *C. immitis* was isolated by culture.[40]

Diagnosis is often delayed because the infection is not considered initially.[6] Travelers visiting regions where *Coccidioides* spp. is endemic should be made aware of the risk of acquiring coccidioidomycosis, and health care providers should be familiar with the presenting signs and symptoms of this disease.

COCCIDIOIDOMYCOSIS IN LATIN AMERICA

Statistics on the prevalence and incidence of coccidioidomycosis in Latin America are either fragmentary or simply not available.

Mexico

Skin test surveys carried out in Mexico indicate that *Coccidioides* spp. infections are as prevalent there as in the endemic areas of the United States.[41] The studies by González-Ochoa (*Encuesta Nacional 1961 1965*) on skin testing with coccidioidin defined the epidemiologic distribution of coccidioidomycosis infection in three endemic zones in the country: the Northern zone, the Pacific Coast zone, and the Central zone, with variable rates of infection in the states of Baja California, Chihuahua, Colima, Coahuila, Durango, Guanajuato, Guerrero, Jalisco, Michoacán, Nayarit, Nuevo León, San Luis Potosí, Sinaloa, Tamaulipas, and Zacatecas.[42] More recently, coccidioidin skin test regional surveys for prevalence of infection have shown rates of 10% (Tijuana, Baja California, 1991[43]), 40% (Torreón, Coahuila, 1999[44]), and 93% (12 communities in the state of Coahuila, 2005[45]).

As mentioned, coccidioidomycosis is caused by two nearly identical species. To determine the prevalent species in northern Mexico, Bialek *et al.*,[46] through conventional nested PCR and real-time PCR assay, tested 120 clinical strains isolated within 10 years in Monterrey, Nuevo Leon, Mexico. All the strains corresponded to the Silveira strain (now known to be *C. posadasii*), as expected from the previous geographical studies by Fisher *et al.*[47]

In Mexico most clinical case reports originate in the northern region of the country. Since coccidioidomycosis is not a reportable disease, its true incidence is unknown.[48]

Tuberculosis and coccidioidomycosis share epidemiological, clinical, radiographic, and even histopathological features. Since tuberculosis is also endemic in Mexico, coccidioidomycosis and tuberculosis can coexist, making the correct diagnosis of both entities extremely difficult in such cases.[49]

Central America

In Central America coccidioidin surveys conducted more than 40 years ago, showed that 21% of children tested at the Motagua River Valley in Guatemala, gave positive reactions, and in the Comayagua Valley of Honduras, skin test surveys revealed an overall prevalence of 25% positivity among the subjects tested.[41] The first human case of coccidioidomycosis in Nicaragua was reported in 1979[50]; there are no published reports of prevalence of infection in that country.

South America

Argentina

Historically, Argentina is of the greatest interest because the first known case was reported by Posadas from that country in 1892.[2] Coccidioidomycosis is one of the three endemic systemic mycoses in Argentina (histoplasmosis and paracoccidioidomycosis being the other two). The endemic area includes the semidesert regions from Puna to Patagonia.[51,52]

Few coccidioidin skin test surveys have been carried out in Argentina, and thus the magnitude of infections in the endemic areas is unknown. In Santiago del Estero, a skin test survey by Negroni *et al.* revealed a prevalence of 19% positive reactions among 2,213 children aged 6 to 16 years.[41] In a more recent skin test survey in the Catamarca province, another skin test survey conducted by Negroni *et al.* in 827 children 6 to 15 years of age revealed a prevalence of infection of 16%. In 1979 in the province of San Luis, which included 1,609 school children and adults, Bonardello *et al.* reported a prevalence of 14.8%.[53]

Brazil

The first autochthonous cases of coccidioidomycosis in Brazil were reported in 1978 and 1979, the first case beiug from the State of Bahia[54] and the second one from Piaui. About 15 years later, the first micro-outbreak of this mycosis in Brazil was also reported in the State of Piauí.[55] Since then, the number of published cases has increased considerably. The association between this infection and the digging of armadillo (*Dasypus novemcinctus*) burrows has been described.[56] The fungus has already been isolated from tissues of this animal, from dogs, and from soil samples collected in armadillo burrows.[56] Currently, this systemic mycosis is considered endemic in the Northeast Brazilian States of Bahia, Ceará, Piauí, and Maranhão.[57]

Other Countries in South America

Little is known about areas of coccidioidomycosis in Paraguay and Bolivia. The probable endemic areas are in the Gran Chaco region, which both countries share with Argentina.[41] In Paraguay, Gomez[58] reported that coccidioidomycosis was endemic in the departments of Boqueron and Olimpo. He found that 44% of a group of Guazurangue Indians living in the department of Boqueron had positive reactions to coccidioidin.

There have been case reports of coccidioidomycosis from Colombia.[41,59] There are no reports of coccidioidin skin test surveys, and therefore the extension of the endemic area and the prevalence of infection remain unknown.

The states of Falcon, Lara, and Zulia in Venezuela have long been considered an endemic area for coccidioidomycosis on the basis of case reports and skin test surveys.[41,60]

COCCIDIOIDOMYCOSIS IN NONHUMAN HOSTS

The organism has been described in a wide spectrum of mammalian hosts and a few captive reptiles,[61] but no report of coccidioidomycosis in an avian species exists to date. Animals of virtually any age may be susceptible. It is not understood why some animals have no clinical signs of coccidioidal infection, whereas others develop disease that progresses even in the face of antifungal treatment.[62]

Coccidioidomycosis has been reported in armadillos,[56] cattle, sheep, dogs,[63] swine, horses,[64] burros, rodents, chinchillas,[65] coyotes,[66] cats,[67] and mountain lions (*Felis concolor*).[68] In addition, the disease has been reported in the following captive free-living wild animals: llamas (*Lama* spp.),[69] Bengal tigers (*Leo tigris*) maintained in a Davis, California, compound,[70] in a giant red kangaroo (*Macropus rufus*) shipped from Australia to the El Paso Zoo, Texas,[71] a tapir (*Tapirtis terrestris*),[72] a mountain gorilla (*Gorilla beringeri*) exhibited at the San Diego Zoo, California,[73] a sooty mangabey (*Cercocebus atys*) transported to Davis, California, from Sierra Leone,[74] and a gelada baboon (*Theropithecus gelada*) imported to Canada from Southern California.[75] Even marine species can acquire the infection and develop disease, including the Pacific bottlenose dolphin,[76] the California sea lion,[77,78] and the southern sea otter.[79]

CONCLUSIONS

Because of its apparent regional confinement, coccidioidomycosis is not perceived to have a substantial impact outside the areas classically considered as endemic. This view should be reconsidered, however. An estimated 150,000 new infections of coccidioidomycosis occur annually in areas of the southwestern United States. In recent years the incidence of clinically apparent disease has increased in California and Arizona, which may be partially due to the massive migration of Americans to the Sunbelt states; cases have recently been discovered outside the traditional areas, suggesting the endemic area may be wider than originally described.

Coccidioidomycosis is often diagnosed in nonendemic regions; diagnosis in that case is often delayed because the infection is not considered initially. Travelers visiting regions where *Coccidioides* spp. are endemic should be made aware of the risk of acquiring coccidioidomycosis, and health care providers should be familiar with the presenting signs and symptoms of this disease.

REFERENCES

1. EINSTEIN, H.E. & R.H. JOHNSON. 1993. Coccidioidomycosis: new aspects of epidemiology and therapy. Clin. Infect. Dis. **16:** 349–356.
2. POSADA, A. 1892. Un nuevo caso de micosis fungoidea con psorospermias. Ann. Circulo Medico Argentino **15:** 585–597.
3. HECTOR, R.F. & R. LANIADO-LABORIN. 2005. Coccidioidomycosis: A fungal disease of the Americas. PLoS Med. **2:** 2e.
4. CATANZARO, A. 2004. Coccidioidomycosis. Sem. Respir. Care Med. **25:** 123–128.
5. GALGIANI, J.N. 1993. Coccidioidomycosis. West. J. Med. **159:** 153–171.
6. PAPPAGIANIS, D.1988. Epidemiology of coccidioidomycosis. Curr. Top. Med. Mycol. **2:** 199–238.
7. DESAI, S.A., O.A. MINAI, S.M. GORDON, *et al.* Coccidioidomycosis in non endemic areas: a case series. Respir. Med. **95:** 305–309.
8. KOLIVRAS, K.N., P. JOHNSON, A.C. COMRIE & S.R. YOOL. 2001. Environmental variability and coccidioidomycosis (valley fever). Aerobiologia **17:** 31–42.
9. COMRIE, A.C. 2005. Climate factors influencing coccidioidomycosis seasonality and outbreaks. Environ. Health Perspect. **113:** 688–692.
10. MADDY, K.T. 1958. The geographic distribution of *Coccidioides immitis* and possible ecologic implications. Ariz. Med. **15:** 178–188.
11. KIRKLAND, T.N. & J. FIERER. 1996. Coccidioidomycosis: a reemerging infectious disease. Emerg. Infect. Dis. **2:** 192–199.
12. KOLIVRAS, K.N. & A.C. COMRIE. 2003. Modeling valley fever (coccidioidomycosis) incidence on the basis of climate conditions. Int. J. Biometeorol. **47:** 87–101.
13. PAPPAGIANIS, D. 1994. Marked increase in cases of coccidioidomycosis in California: 1991, 1992, and 1993. Clin. Infect. Dis. **19:** S14–S18.
14. CENTERS FOR DISEASE CONTROL AND PREVENTION. Update. 1994. Coccidioidomycosis—California, 1991–1993. MMWR Morb. Mortal. Wkly. Rep. **43:** 421–423.
15. PAPPAGIANIS, D. & H. EINSTEIN. 1978. Tempest from Tehachapi takes toll or *Coccidioides immitis* conveyed aloft and afar. West J. Med. **129:** 527–530.
16. CAIRNS, L., D. BLYTHE, A. KAO, *et al.* 2000. Ontbreak of coccidioidomycosis in Washington State residents returning from Mexico. Clin. Infect. Dis. **30:** 61–64.
17. SCHNEIDER, E., R.A. HAJJEH, R.A. SPIEGEL, *et al.* 1997. A coccidioidomycosis outbreak following the Northridge, Calif, earthquake. JAMA **277:** 904–908.
18. WANKE, B., M. LAZERA, P.C. MONTEIRO, *et al.* 1999. Investigation of an outbreak of endemic coccidioidomycosis in Brazil's northeastern state of Piaui with a review of the occurrence and distribution of *Coccidioides immitis* in three other Brazilian states. Mycopathologia **148:** 57–67.

19. CRUM, N., C. LAMB, G. UTZ, *et al.* 2002. Coccidioidomycosis outbreak among United States Navy SEALs training in a *Coccidioides immitis*-endemic area-Coalinga, California. J. Infect. Dis. **186:** 865–868.
20. SMITH, C.E., R.R. BEARD, H.G. ROSENBERG & E.G. WHITING. 1946. Effect of season and dust control on coccidioidomycosis. JAMA **132:** 833–888.
21. AMPEL, N.M., M.A. WIEDEN & J.N. GALGIANI. 1989. Coccidioidomycosis: clinical update. Rev. Infectious Dis. **11:** 897–911.
22. CRUM, N.F., E.R. LEDERMAN, C.M. STAFFORD, *et al.* 2004. Coccidioidomycosis. A descriptive survey of a reemerging disease. Clinical characteristics and current controversies. Medicine **83:** 149–175.
23. DRUTZ, D.J., M. HUPPERT, S.H. SUN & W.I. MCGUIRE. 1981. Human sex hormones stimulate the growth and maturation of *Coccidioides immitis*. Infect. Immun. **32:** 897–907.
24. DERESINSKI, S.C., D. PAPPAGIANIS & D.A. STEVENS. 1979. Association of ABO blood group and outcome of coccidioidal infection. Sabouradia **17:** 261–264.
25. LOUIE, L., S. NG, R. HAJJEH, *et al.* 1999. Influence of host genetics on the severity of coccidioidomycosis. Emerg. Infect. Dis. **5:** 672–680.
26. AMPEL, N.M. 2005. Coccidioidomycosis in persons infected with HIV Type 1. Clin. Infect. Dis. **41:** 1174–1178.
27. AMPEL, N.M., C.L. DOLS & J.N. GALGIANI. 1993. Coccidioidomycosis during human immunodeficiency virus infection: results of a prospective study in a coccidioidal endemic area. Am. J. Med. **94:** 235–240.
28. JONES, J.L., P.L. FLEMING, C.A. CIESIELSKI, *et al.* 1995. Coccidioidomycosis among persons with AIDS in the United States. J. Infect. Dis. **171:** 961–966.
29. AMPEL, N.M., K.J. RYAN, P.J. CARRY, *et al.* 1986. Fungemia due to *Coccidioides immitis*: an analysis of 16 episodes in 15 patients and a review of the literature. Medicine. **65:** 312–321.
30. CRUM, N.F. & G. LANDA-BALLON. 2006. Coccidioidomycosis in pregnancy: case report and review of the literature. Am. J. Med. (online). **119:** 993.e11–e17.
31. CALDWELL, J.W., E.I. ARSURA, W.B. KILGORE, *et al.* 2000. Coccidioidomycosis in pregnancy during an epidemic in California. Obst. Gynecol. **95:** 236–239.
32. LOGAN, J.L., J.E. BLAIR & J.N. GALGIANI. 2001. Coccidioidomycosis complicating solid organ transplantation. Semin. Respir. Infect. **16:** 251–256.
33. BLAIR, J.E. 2006. Coccidioidomycosis in liver transplantation. Liver. Transpl. **12:** 31–39.
34. ABBOTT, K.C., I. HYPOLITE, D.J. TVEIT, *et al.* 2001. Hospitalizations for fungal infections after initiation of chronic dialysis in the United States. Nephron **89:** 426–432.
35. GALGIANI, J.N. 1999. Coccidioidomycosis: A regional disease of national importance. Rethinking approaches for control. Ann. Intern. Med. **130:** 293–300.
36. http://quickfacts.census.gov/qfd/states(accessed on 4 January 2007).
37. Arizona Department of Health Services Website: http://www.azdhs.gov/.
38. 2001.Coccidioidomycosis in workers at an archeologic site: Dinosaur National Monument, Utah, June–July 2001. MMWR **50:** 1005–1008.
39. CHATURVEDI, V., R. RAMANI, S. GROMADZKI, *et al.* 2000. Coccidioidomycosis in New York State. Emerg. Infect. Dis. **6:** 25–29.
40. LEFLER, E., D. WEILER-RAVELL, D. MERZBACH, *et al.* 1992. Traveller's coccidioidomycosis: case report of pulmonary infection diagnosed in Israel. J. Clin. Microbiol. **30:** 1304–1306.

41. AJELLO, L. 1967. Comparative ecology of respiratory mycotic disease agents. Bacteriol. Rev. **31:** 6–24.
42. GONZÁLEZ-OCHOA, A. 1966. La coccidioidomicosis en México Rev. Invest. Salud Publ. (Mex). **26:** 245–262.
43. LANIADO-LABORÍN, R., R.P. CÁRDENAS-MORENO & M. ÁLVAREZ-CERRO. 1991. Tijuana: zona endémica de infección por *Coccidioides immitis*. Salud Pública Mex. **33:** 235–239.
44. PAPUA, A., V. MARTÍNEZ-ORDAZ, V.M. VELASCO-RODRIGUEZ, *et al.* 1999. Prevalence of skin reactivity to coccidioidin and associated risk factors in snbjects living in a northern city of Mexico. Arch. Med. Res. **30:** 388–392.
45. MONDRAGÓN-GONZÁLEZ, R., L.J. MÉNDEZ-TOVAR, E. BERNAL-VÁZQUEZ, *et al.* 2005. Detección de infección por *Coccidioides immitis* en zonas del estado de Coahuila, México. Rev. Argentina Microbiol. **37:** 135–138.
46. BIALEK, R., J. KERN, T. HERRMANN, *et al.* 2004. PCR assays for identification of *Coccidioides posadasii* based on the nucleotide sequence of the antigen 2/proline-rich antigen. J. Clin. Microbiol. **42:** 778–783.
47. FISHER, M.C., G.L. KOENIG, T.J. WHITE & J. W. TAYLOR. 2002. Molecular and phenotype description of *Coccidioides posadasii* spp. nov., previously recognized as the non-Californian population of *Coccidioides immitis*. Mycologia **94:** 73–84.
48. CASTAÑÓN-OLIVARES, L.R., A. AROCH-CALDERÓN, E. BAZÁN-MORA & E. CÓRDOVA-MARTÍNEZ. 2004. Coccidioidomicosis y su escaso conocimiento en nuestro país. Rev. Fac. Med. UNAM **47:** 145–148.
49. CASTAÑEDA-GODOY, R. & R. LANIADO-LABORÍN. 2002. Coexistencia de tuberculosis y coccidioidomicosis. Presentación de dos casos clínicos. Rev. Inst. Nal. Enf. Resp. Mex. **15:** 98–101.
50. RIOS-OLIVARES, E.O. 1979. 1st human case of coccidioidomycosis in Nicaragua. Rev. Latinoam. Microbiol. **21:** 215–218.
51. NEGRONI, P., C.R. BRAVO, R. NEGRONI, *et al.* 1978. Estudios sobre el *Coccidoides immitis*. Encuesta epidemiológica efectuada en la Provincia de Catamarca. Bol. Acad. Nal. Med. **56:** 327–339.
52. MASIH, D.T., B.E. MARTICORENA, N. BORLETTO, *et al.* 1987. Epidemiologic study of bronchopulmonary mycosis in the Province of Cordova, Argentina. Rev. Inst. Med. Trop. São Paulo. **29:** 59–62.
53. BONARDELLO, N.M. & C.G. DE GAGLIARDI. 1979. Intradermal reactions with coccidioidins in different towns of San Luis Province. Sabouraudia **17:** 371–376.
54. GOMES, O.M., R.R.P. SERRANO, H.O.V. PRADO, *et al.* 1978. Coccidioidomicose pulmonar: primeiro caso nacional. Rev. Assoc. Med. Bras. **24:** 167–168.
55. BEZERRA, C., R, DE LIMA, M. LAZERA,et al. 2006. Viability and molecular authentication of *Coccidioides immitis* strains from Culture Collection of the Instituto Oswaldo Cruz, Rio de Janeiro, Brazil. Rev. Soc. Brasileira Med. Trop. **39:** 241–244.
56. EULALIO, K.D., R.L. MACEDO, M.A.S. CAVALCANTI, *et al.* 2001. *Coccidioides immitis* isolated from armadillos (*Dasypus novemcinctus*) in the state of Piaui, northeast Brazil. Mycopathologia **149:** 57–61.
57. NOBRE-VERAS, K., B.C. DE SOUZA-FIGUEIRÊDO, L.M. SOARES-MARTINS, *et al.* 2003. Coccidioidomycosis: an unusual cause of acute respiratory distress syndrome. J. Pneumol. **29:** 45–48.
58. GOMEZ, R. F. 1950. Endemism of coccidioidomycosis in the Paraguayan Chaco. California Med. **73:** 35–38.

59. ROBLEDO, M.V. 1965. Coccidioidomicosis. Antioquia Med. **15:** 361–362.
60. CAMPINS, H. 1961. Coccidioidomicosis. Comentarios sobre la casuistica Venezolana. Mycopathol. Mycol. Appli. **15:** 306–316.
61. TIMM, K.I., R.J. SONN & B.D. HULTGREN. 1988. Coccidioidomycosis in a Sonoran gopher snake, *Pituophis melanoleucus affinis*. J. Med. Vet. Mycol. **26:** 101–104.
62. SHUBITZ, L.F. & S.M. DIAL. 2005. Coccidioidomycosis: a diagnostic challenge. Clin. Tech. Small Anim. Pract. **20:** 220–226.
63. SHUBITZ, L.F., C.D. BUTKIEWICZ, S.M. DIAL & C.P. LINDAN. 2005. Incidence of *Coccidioides* spp. infection among dogs residing in an endemic region. J. Am. Vet. Med. Assoc. **226:** 1846–1850,
64. ZIEMER, E.L., D. PAPPAGIANIS, J.E. MADIGAN, *et al.* 1992. Coccidioidomycosis in horses: 15 cases (1975–1984). J. Am. Vet. Med. Assoc. **201:** 910–916.
65. ASHBURN, L.L. & C.W. EMMONS. 1942. Spontaneous coccidioidal granuloma in the lungs of wild rodents. Archs. Path. **34:** 791–800.
66. STRAUB, M., R.J. TRAUTMAN & J.W. GREENE. 1961. Coccidioidomycosis in 3 coyotes. Am. J. Vet. Res. **22:** 811–812.
67. GREENE, R.T. & G.C. TROY. 1995. Coccidioidomycoisis in 48 cats: a retrospective study (1984–1993). J. Vet. Int. Med. **9:** 86–91.
68. ADASKA, J.M. 1999. Peritoneal coccidioidomycosis in a mountain lion in California. J. Wildlife Dis. **35:** 75–77.
69. FOWLER, M.E., D. PAPPAGIANIS & I. INGRAM. 1992. Coccidioidmycosis in llamas in the United States: 19 cases (1981–1989). J. Am. Vet. Med. Assoc. **201:** 1609–1614.
70. HENRIKSON, R.V. & E.L. BIBERSTEIN. 1972. Coccidioidomycosis accompanying hepatic disease in two Bengal tigers. J. Am. Vet. Med. Ass. **161:** 674–677.
71. HUTCHINSON, L.R., F. DURAN, C.D. LANE, *et al.* 1973. Coccidioidomycosis in a giant red kangaroo (*Macropus rufus*). J. Zoo Anim. Med. **4:** 22–24.
72. DILLEHAY, D.L., T.R. BOOSINGER & S. MACKENZIE. 1985. Coccidioidomycosis in a tapir. J. Am. Vet. Med. Assoc. **187:** 1233–1234.
73. MCKENNEY, F.D., J. TRAUM & A.E. BONESTELL. 1944. Acute coccidioidomycosis in a mountain gorilla (*Gorilla beringeri*) with anatomical notes. J. Am. Vet. Med. Assoc. **104:** 136–140.
74. PAPPAGIANIS, D., J. VANDERLIP & B. MAY. 1973. Coccidioidomycosis naturally acquired by a monkey, *Cercocebus atys*, in Davis, California. Sabouraudia **11:** 52–55.
75. RAPLEY, W.A. & J.R. LONG. 1974. Coccidioidomycosis in a baboon recently imported from California. Can. Vet. J. **15:** 39–41.
76. REIDARSON, T.H., L.A. GRINER, D. PAPPAGIANIS & J. MCBAIN. 1988. Coccidioidomycosis in a bottlenose dolphin. J. Wildlife Dis. **34:** 629–631.
77. REED, R.E., G. MIGAKI & J.A. CUMMINGS. 1976. Coccidioidomycosis in a California sea lion. J. Wildlife Dis. **12:** 372–375.
78. FAUQUIER, D.A., F.M.D. GULLAND, J.G. TRUPKIEWICZ, *et al.* 1996. Coccidioidomycosis in free-living California sea lions (*Zalophus californianus*) in central California. J. Wildl. Dis. **32:** 707–710.
79. CORNELL, L.H., K.G. OSBORN, J.E. ANTRIM JR. & J.G. SIMPSON. 1979. Coccidioidomycosis in California sea otter (*Enhydra lutris*). J. Wildl. Dis. **15:** 373–378.
80. HECTOR, R.F. & R. LANIADO-LABORÍN. 2005. Coccidioidomycosis: a fungal dease of the Americas. PLoS Med. **2(1):** e2.

The costs of chronic noise exposure for terrestrial organisms

Jesse R. Barber[1], Kevin R. Crooks[1] and Kurt M. Fristrup[2]

[1] Department of Fish, Wildlife and Conservation Biology, Colorado State University, Fort Collins, CO 80523, USA
[2] National Park Service, Natural Sounds Program, Fort Collins, CO 80525, USA

Growth in transportation networks, resource extraction, motorized recreation and urban development is responsible for chronic noise exposure in most terrestrial areas, including remote wilderness sites. Increased noise levels reduce the distance and area over which acoustic signals can be perceived by animals. Here, we review a broad range of findings that indicate the potential severity of this threat to diverse taxa, and recent studies that document substantial changes in foraging and anti-predator behavior, reproductive success, density and community structure in response to noise. Effective management of protected areas must include noise assessment, and research is needed to further quantify the ecological consequences of chronic noise exposure in terrestrial environments.

Anthropogenic noise and acoustic masking

Habitat destruction and fragmentation are collectively the major cause of species extinctions [1,2]. Many current threats to ecological integrity and biodiversity transcend political and land management boundaries; climate change, altered atmospheric and hydrologic regimes and invasive species are prominent examples. Noise also knows no boundaries, and terrestrial environments are subject to substantial and largely uncontrolled degradation of opportunities to perceive natural sounds. Noise management is an emergent issue for protected lands, and a potential opportunity to improve the resilience of these areas to climate change and other forces less susceptible to immediate remediation.

Why is chronic noise exposure a significant threat to the integrity of terrestrial ecosystems? Noise inhibits perception of sounds, an effect called masking (see Glossary) [3]. Birds, primates, cetaceans and a sciurid rodent have been observed to shift their vocalizations to reduce the masking effects of noise [4–7]. However, compromised hearing affects more than acoustical communication. Comparative evolutionary patterns attest to the alerting function of hearing: (i) auditory organs evolved before the capacity to produce sounds intentionally [8], (ii) species commonly hear a broader range of sounds than they are capable of producing [9], (iii) vocal activity does not predict hearing performance across taxa [9,10], (iv) hearing continues to function in sleeping [11] and hibernating [12] animals; and (v) secondary loss of vision is more common than is loss of hearing [13].

Masking is a significant problem for the perception of adventitious sounds, such as footfalls and other byproducts of motion. These sounds are not intentionally produced and natural selection will typically favor individuals that minimize their production. The prevalence and characteristics of adventitious sounds have not been widely studied [14–16], although their role in interactions

Glossary

Alerting distance: the maximum distance at which a signal can be perceived. Alerting distance is pertinent in biological contexts where sounds are monitored to detect potential threats.
Atmospheric absorption: the part of transmission loss caused by conversion of acoustic energy into other forms of energy. Absorption coefficients increase with increasing frequency, and range from a few dB to hundreds of dB per kilometer within the spectrum of human audibility.
Audible: a signal that is perceptible to an attentive listener.
A-weighting: A method of summing sound energy across the frequency spectrum of sounds audible to humans. A-weighting approximates the inverse of a curve representing sound intensities that are perceived as equally loud (the 40 phon contour). It is a broadband index of loudness in humans in units of dB(A) or dBA. A-weighting also approximates the shapes of hearing threshold curves in birds [20].
Decibel (dB): a logarithmic measure of acoustic intensity, calculated by 10 $\log_{10}$(sound intensity/reference sound intensity). 0 dB approximates the lowest threshold of healthy human hearing, corresponding to an intensity of 10^{-12} Wm^{-2}. Example sound intensities: −20 dB, sound just audible to a bat, owl or fox; 10 dB, leaves rustling, quiet respiration; 60 dB, average human speaking voice; 80 dB, motorcycle at 15 m.
Frequency (Hz and kHz): for a periodic signal, the maximum number of times per second that a segment of the signal is duplicated. For a sinusoidal signal, the number of cycles (the number of pressure peaks) in one second (Hz). Frequency equals the speed of sound (~340 ms^{-1}) divided by wavelength.
Ground attenuation: the part of transmission loss caused by interaction of the propagating sound with the ground.
Listening area: the area of a circle whose radius is the alerting distance. Listening area is the same as the 'active space' of a vocalization, with a listener replacing the signaler as the focus, and is pertinent for organisms that are searching for sounds.
Masking: the amount or the process by which the threshold of detection for a sound is increased by the presence of the aggregate of other sounds.
Noticeable: a signal that attracts the attention of an organism whose focus is elsewhere.
Scattering loss: the part of transmission loss resulting from irregular reflection, diffraction and refraction of sound caused by physical inhomogeneities along the signal path.
Spectrum, power spectrum and spectral profile: the distribution of acoustic energy in relation to frequency. In graphical presentations, the spectrum is often plotted as sound intensity against sound frequency (Figure 1, main text).
1/3 octave spectrum: acoustic intensity measurements in a sequence of spectral bands that span 1/3 octave. The International Standards Organization defines 1/3rd octave bands used by most sound level meters (ISO 266, 1975). 1/3rd octave frequency bands approximate the auditory filter widths of the human peripheral auditory system.
Spreading loss: more rigorously termed divergence loss. The portion of transmission loss attributed to the divergence of sound energy, in accordance with the geometry of environmental sound propagation. Spherical spreading losses in dB equal $20*\log10(R/R_0)$, and result when the surface of the acoustic wavefront increases with the square of distance from the source.
White noise: noise with equal energy across the frequency spectrum.

Corresponding author: Barber, J.R. (barber.jesse@gmail.com).

0169-5347/$ – see front matter doi:10.1016/j.tree.2009.08.002 Available online 15 September 2009

Box 1. Geographic extent of transportation noise in the USA

Transportation noise is a near ubiquitous component of the modern acoustical landscape. The method used here to estimate the geographic extent of airway (Figure Ia,b), railway (Figure Ic) and roadway (Figure Id) noise in the continental USA is calculated using the average human 'noticeability' of noise. Noise was deemed noticeable when the modeled noise intensity from transportation [in dB(A)] exceeded the expected noise intensity as predicted from population density [also dB(A)]. Although noticeability is a conservative metric of the geographic extent of transportation noise, this analysis only indicates the potential scope of the problem. How anthropogenic noise changes the temporal and spectral properties of naturally-occurring noise (Figure 1, main text) and the life histories of individual species will be crucial components of a more thorough analysis.

The maps in Figure I reflect the following calculations: (i) noise calculations are county-by-county for a typical daytime hour; (ii) county population density is transformed into background sound level using an EPA empirical formula (see Ref. [84]); higher density implies higher background sound levels; (iii) the geographic extent of transportation noise is determined by calculating the distance from the vehicle track at which the transportation noise falls below the background sound level, multiplying twice that distance by the length of the transportation corridor in the county (giving a noticeability area), and comparing that area with the total area in the county to compute the percentage land area affected. A low percentage noticeability can result if either the population density is high or the number of transportation segments is low in the county. This analysis indicates that transportation noise is audible above the background of other anthropogenic noise created by local communities in most counties in continental USA. See Ref. [84] for more details.

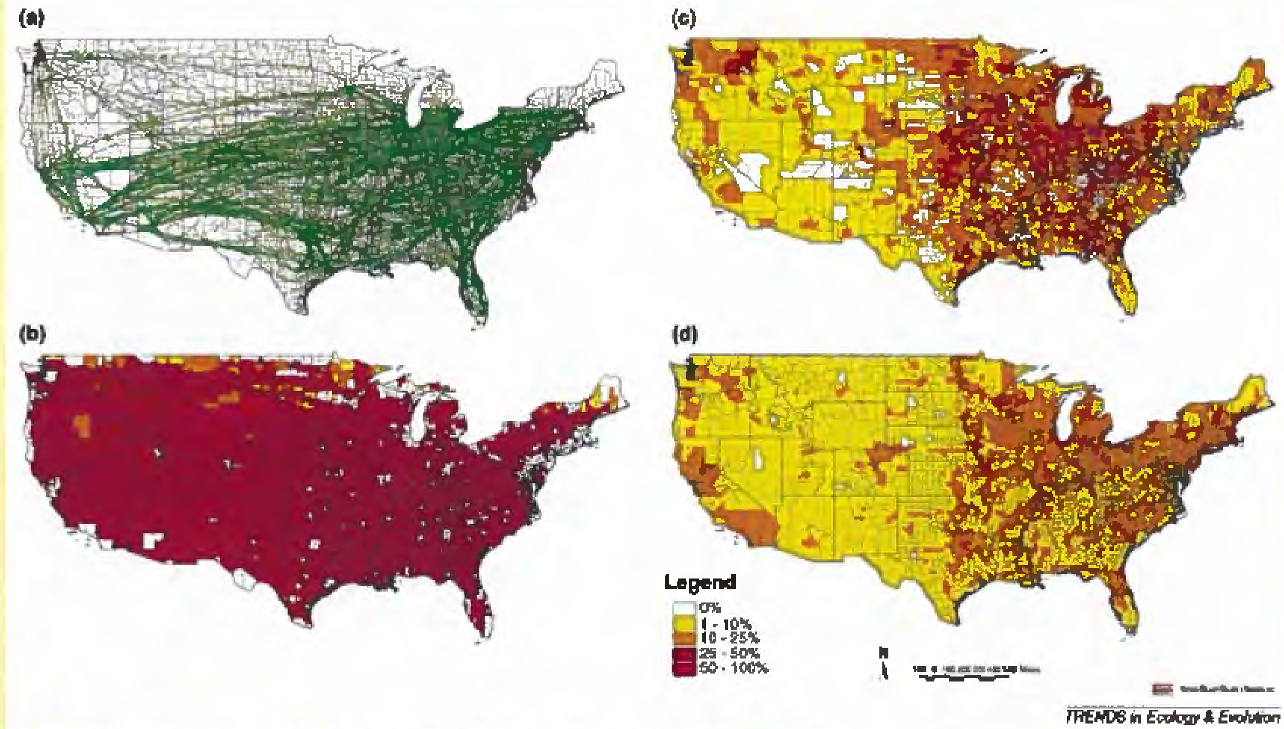


Figure I. Percent of US county areas in which transportation noise is noticeable. (a) Jet departures that occurred between 3 and 4 pm on Oct. 17, 2000, tracked to first destination. (b) Data from (a) were used to estimate the geographic extent of high altitude airway noise in the USA. The geographic extent of noise from railway and highway networks is depicted in (c) and (d), respectively. The color-coded divisions (see legend; divisions increase in size as the percent increases) were chosen assuming that, as noticeability increases, so do estimate errors due to noticeability area overlap from different transportation segments. Adapted with permission from Ref. [84].

among predators and prey is unquestionable. In animal communication systems, both the sender and receiver can adapt to noise masking, but for adventitious sounds the burden falls on listeners.

Anthropogenic disturbance is known to alter animal behavioral patterns and lead to population declines [17,18]. However, animal responses probably depend upon the intensity of perceived threats rather than on the intensity of noise [19]. Deleterious physiological responses to noise exposure in humans and other animals include hearing loss [20], elevated stress hormone levels [21] and hypertension [22]. These responses begin to appear at exposure levels of 55–60 dB(A), levels that are restricted to relatively small areas close to noise sources [20].

The scale of potential impact

The most spatially extensive source of anthropogenic noise is transportation networks. Growth in transportation is increasing faster than is the human population. Between 1970 and 2007, the US population increased by approximately one third (http://www.census.gov/compendia/statab). Traffic on US roads nearly tripled, to almost 5 trillion vehicle kilometers per year (http://www.fhwa.dot.gov/ohim/tvtw/tvtpage.cfm). Several measures of aircraft traffic grew by a factor of three or more between 1981 and 2007 (http://www.bts.gov/programs/airline_information/air_carrier_traffic_statistics/airtraffic/annual/1981_present.html). Recent reviews of the effects of noise on marine mammals have identified similar trends in shipping noise (e.g. Refs [23,24]). In addition to transportation, resource extraction and motorized recreation are spatially extensive sources of noise on public lands.

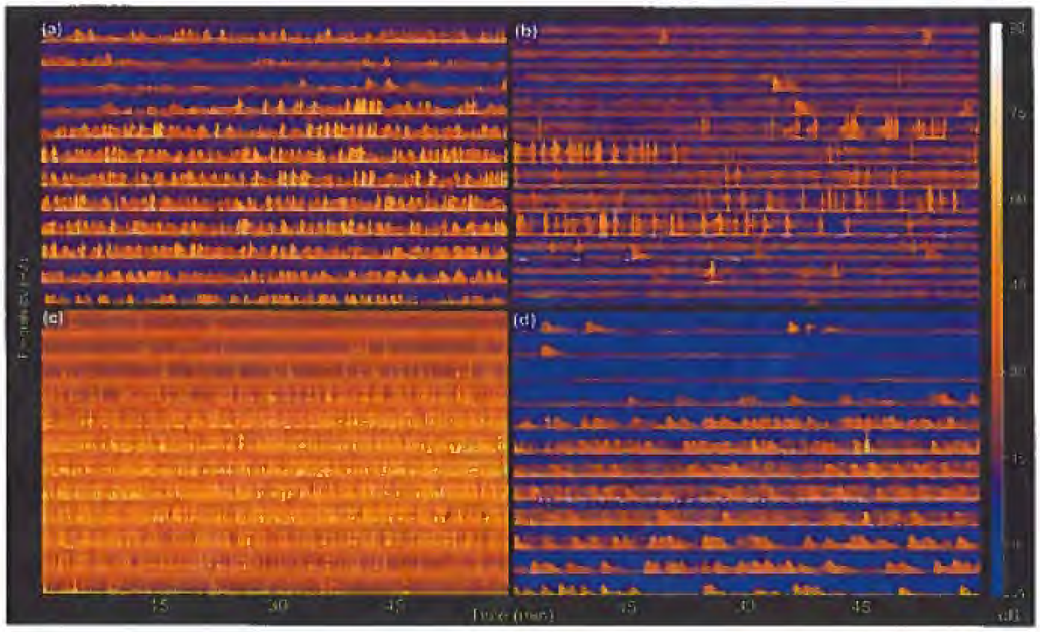
Systematic monitoring by the Natural Sounds Program of the US National Park Service (http://www.nature.nps.gov/naturalsounds) confirms the extent of noise intrusions. Noise is audible more than 25% of the hours between 7am and 10pm at more than half of the 55 sites in 14 National Parks that have been studied to date; more than a dozen sites have hourly noise audibility percentages exceeding 50% (NPS, unpublished). Remote wilderness areas are not immune, because air transportation noise is widespread, and high traffic corridors generate substantial noise increases on the ground (Box 1). For example, anthropogenic sound is audible at the Snow Flats site in Yosemite National Park nearly 70% of the time during peak traffic hours. Figure 1 shows that typical noise levels exceed natural ambient sound levels by an order of magnitude or more.

Roads are another pervasive source of noise: 83% of the land area of the continental US is within 1061 m of a road [25]. At this distance an average automobile (having a noise source level of 68 dB(A) measured at 15 m) will project a noise level of 20 dB(A). This exceeds the median natural levels of low frequency sound in most environments. Trucks and motorcycles will project substantially more noise: up to 40 dB(A) at 1 km. Box 2 provides a physical model of the reduced listening area that can be imposed by these louder background sound levels.

Acoustical ecology

Intentional communication, such as song, is the best studied component of the acoustical world, and these signals are often processed by multiple receivers. These communication networks enable female and male songbirds, for example, to assess multiple individuals simultaneously for mate choice, extra-pair copulations and rival assessment [26]. Acoustic masking resulting from increasing background sound levels will reduce the number of individuals that comprise these communication networks and have unknown consequences for reproductive processes [27].

Reproductive and territorial messages are not the only forms of acoustical communication that operate in a network. Social groups benefit by producing alarm calls to warn of approaching predators [28] and contact calls to maintain group cohesion [29]. A reduction in signal transmission distance created by anthropogenic noise might decrease the effectiveness of these social networks. The inability to hear just one of the alarm calling individuals can result in animals underestimating the urgency of their response [30].



TRENDS in Ecology & Evolution

Figure 1. 24-hour spectrograms of Indian Pass in Lake Mead National Recreation Area (a), Madison Junction in Yellowstone National Park (b), Trail Ridge Road in Rocky Mountain National Park (c), and Snow Flats in Yosemite National Park (d). Each panel displays 1/3 octave spectrum sound pressure levels, with two hours represented horizontally in each of 12 rows. The first three rows in each panel represent the quietest hours of each day, from midnight to 6 am. Frequency is shown on the y axis as a logarithmic scale extending from 12.5 Hz to 20 kHz, with the vertical midpoint in each row corresponding to 500 Hz. The z axis (color) describes sound pressure levels in dB (unweighted); the color scaling used for all four panels is indicated by the color bar on the right hand edge. The lowest 1/3 octave levels are below 0 dB, the nominal threshold of human hearing. White dots at the upper edge of some rows in the panels on the right side denote missing seconds of data. Low-frequency, broadband signatures from high altitude jets are present in all four panels. Distinct examples are present just before 6 am in (a), near 12:45 am in (b) and (c), and between midnight and 12:30 am in (d). Fixed wing aircraft signatures (tonal contours with descending pitch) are present in (a) and (d), with a good example at 1:16 am in (d). Broadband signatures with very low frequency tonal components in (a) are due to low-altitude helicopters, that are prominent from ~7 am until 8 pm. Another prominent helicopter signature is at 11:30 am in (d). (b) illustrates snowmobile and snowcoach sounds recorded ~30 m from the West Entrance Road in Yellowstone. (c) illustrates traffic noise recorded 15 m from Trail Ridge Road in Rocky Mountain National Park, during a weekend event featuring high levels of motorcycle traffic. Background sound levels at the Rocky Mountain site were elevated by sounds from the nearby river.

Box 2. Physical model of reduced listening area in noise

The maximum detection distance of a signal decreases when noise elevates the masked hearing threshold. The masked detection distance: original detection distance ratio will be the same for all signals in the affected frequency band whose detection range is primarily limited by spreading losses. For an increase of N dB in background sound level, the detection distance ratio is: $k = 10^{-N/20}$. The corresponding fraction of original listening area is: $k = 10^{-N/10}$. A 1-dB increase in background sound level results in 89% of the original detection distance, and 79% of the original listening area. These formulae will overestimate the effects of masking on alerting distance and listening area for signals that travel far enough to incur significant absorptive and scattering losses. More detailed formulae would include terms that depend upon the original maximum range of detection.

Figure I illustrates the expected noise field of a road treated as a line source (equal energy generated per 10 m segment). An animal track is marked by ten circular features, that depict the listening area of a signal whose received level (expressed as a grey-scaled value for each possible source location) decreases with the inverse square of distance from the listener. The apparent shrinkage of the circles is due to masking by the increasingly dark background of sound projected from the road, just as noise would shrink the listening area. The circles span 9 dB in road noise level, in 1-dB steps from the quietest location (upper right) to the noisiest (at the crossing).

Masking effects are reduced with increasing spectral separation between noise and signal. The model presumes that the original conditions imposed masked hearing thresholds, so organisms that are limited by their hearing thresholds will not be as affected by masking. A diffuse noise source is illustrated, but the same results would be obtained if some spatial release from masking were possible, so long as the original conditions implied masked hearing thresholds (see Ref. [85] for a review of release strategies).

These measures of lost listening opportunity are most pertinent for chronic exposures. They imply substantial losses in auditory awareness for seemingly modest increases in noise exposure. Analyses of transportation noise impacts based on perceived loudness often assert that increases of up to three dB have negligible effects; this corresponds to a 50% loss of listening area.

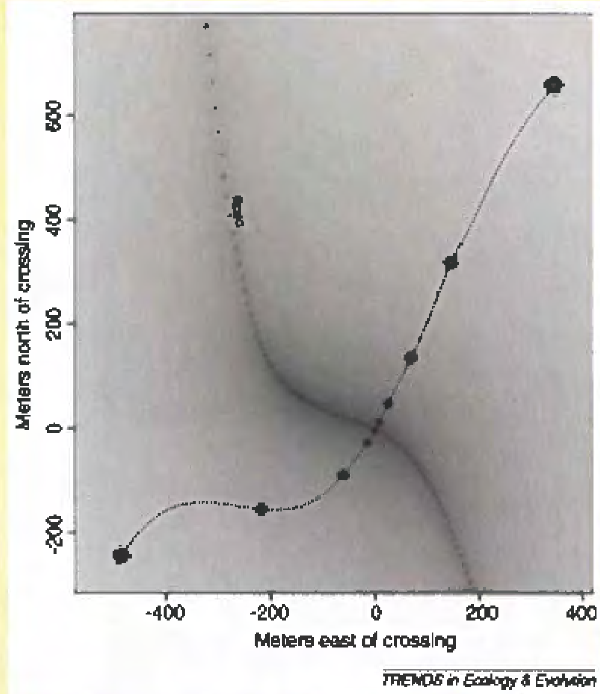


Figure I. A physical model of reduced listening area as an animal approaches a road.

Many vertebrate and invertebrate species are known to listen across species' boundaries to one another's sexual (e.g. Ref. [31]), alarm (e.g. Ref. [32]) and other vocalizations. Recent examples include gray squirrels, *Sciurus carolinensis*, listening in on the communication calls of blue jays, *Cyanocitta cristata*, to assess site-specific risks of cache pilfering [33]; and nocturnally migrating songbirds [34] and newts (Ref. [35] and Refs therein) using heterospecific calls to make habitat decisions. Reduced listening area imposed by increased sound levels is perhaps more likely to affect acoustical eavesdropping than to interfere with deliberate communication. The signaler is under no selective pressure to ensure successful communication to eavesdroppers and any masking compensation behaviors will be directed at the auditory system and position of the intended receiver rather than of the eavesdropper.

Acoustical communication and eavesdropping comprise most of the work in bioacoustics, but the parsimonious scenario for the evolution of hearing involves selection for auditory surveillance of the acoustical environment, with intentional communication evolving later [8]. Adventitious sounds are inadequately studied, in spite of their documented role in ecological interactions. Robins can use sound as the only cue to find buried worms [36]; a functional group of bats that capture prey off surfaces, gleaners, relies on prey-generated noises to localize their next meal [37]; barn owls (*Tyto alba*; [38]), marsh hawks (*Circus cyaneus*; [39]), and grey mouse lemurs (*Microcebus murinus*; [15] have been shown to use prey rustling sounds to detect and localize prey; big brown bats, *Eptesicus fuscus*, have the ability to use low-frequency insect flight sounds to identify insects and avoid protected prey [40]. In addition to prey localization, spectrally unstructured movement sounds are also used to detect predators. White-browed scrubwren (*Sericornis frontalis*) nestlings become silent when they hear the playback of footsteps of pied currawong, *Strepera graculina*, their major predator [41]; and tungara frogs, *Physalaemus pustulosus* avoid the wingbeat sounds of an approaching frog-eating bat, *Trachops cirrhosus* [42]. We are aware of only one study that has examined the role of adventitious sounds other than movement noises; African reed frogs, *Hyperolius nitidulus* flee from the sound of fire [43]. It is likely that other ecological sounds are functionally important to animals.

It is clear that the acoustical environment is not a collection of private conversations between signaler and receiver but an interconnected landscape of information networks and adventitious sounds; a landscape that we see as more connected with each year of investigation. It is for these reasons that the masking imposed by anthropogenic noise could have volatile and unpredictable consequences.

Separating anthropogenic disturbance from noise impacts

Recent research has reinforced decades of work [44,45] showing that human activities associated with high levels of anthropogenic noise modify animal ecology: for example, the species richness of nocturnal primates, small ungulates and carnivores is significantly reduced within ~ 30 m of roads in Africa [46]; anuran species richness in Ottawa, Canada is negatively correlated with traffic density [47]; aircraft overflights disturb behavior and alter time budgets in harlequin ducks (*Histrionicus histrionicus*; [48]) and mountain goats (*Oreamnos americanus*; [49]); snowmobiles and off-road vehicles change ungulate vigilance behavior and space use, although no evidence yet links these responses to population consequences [50,51]; songbirds show greater nest desertion and abandonment, but reduced predation, within 100 m of off-road vehicle trails [52]; and both greater sage-grouse (*Centrocercus urophasianus*; [53]) and mule deer (*Odocoileus hemionus*; [54]) are significantly more likely to select habitat away from noise-producing oil and gas developments. Thus, based on these studies alone, it seems clear that activities associated with high levels of anthropogenic noise can re-structure animal communities; but, because none of these studies, nor the disturbance literature in general, isolates noise from other possible forces, the independent contribution of anthropogenic noise to these effects is ambiguous.

Other evidence also implicates quiet, human-powered activities, such as hiking and skiing, in habitat degradation. For example, a paired comparison of 28 land preserves in northern California that varied substantially in the number of non-motorized recreationists showed a five-fold decline in the density of native carnivores in heavily used sites [55]. Further evidence from the Alps indicates that outdoor winter sports reduce alpine black grouse, *Tetrao tetrix* populations [17] and data from the UK link primarily quiet, non-motorized recreation to reduced woodlark, *Lullula arborea* populations [18]. A recent meta-analysis of ungulate flight responses to human disturbance showed that humans on foot produced stronger behavioral reactions than did motorized disturbance [45]. These studies strengthen a detailed foundational literature suggesting that anthropogenic disturbance events are perceived by animals as predation risk, regardless of the associated noise levels. Disturbance evokes anti-predator behaviors, interferes with other activities that enhance fitness and, as the studies above illustrate, can lead to population decline [44]. Although increased levels of noise associated with the same disturbance type appear to accentuate some animal responses (e.g. Refs [44,48]), it is difficult to distinguish reactions that reflect increasingly compromised sensory awareness from reactions that treat greater noise intensity as an indicator of greater risk.

To understand the functional importance of intact acoustical environments for animals, experimental and statistical designs must control for the influence of other stimuli. Numerous studies implicating noise as a problem for animals have reported reduced bird densities near roadways (reviewed in Ref. [56]). An extensive study conducted in the Netherlands found that 26 of 43 (60%) woodland bird species showed reduced numbers near roads [57]. This research, similar to most road ecology work, could not isolate noise from other possible factors associated with transportation corridors (e.g. road mortality, visual disturbance, chemical pollution, habitat fragmentation, increased predation and invasive species along edges). However, these effects extended for over a mile into the forest, implicating noise as one of the most potent forces driving road effects [58]. Later work, with a smaller sample size, confirmed these results and contributed a significant finding: birds with higher frequency calls were less likely to avoid roadways than birds with lower frequency calls [59]. Coupled with the mounting evidence that several animals shift their call frequencies in anthropogenic noise [4–7], these data are suggestive of a masking mechanism.

A good first step towards disentangling disturbance from noise effects is exemplified by small mammal translocation work performed across roadways that varied greatly in traffic amount. The densities of white-footed mice, *Peromyscus leucopus* and eastern chipmunks *Tamias striatus* were not lower near roads and both species were significantly less likely to cross a road than cover the same distance away from roads, but traffic volume (and noise level) had no influence on this finding [60]. Thus, for these species, the influence of the road surface itself appears to outweigh the independent contributions of direct mortality and noise.

Recent findings on the effects of anthropogenic noise

Two research groups have used oil and gas fields as 'natural experiments' to isolate the effects of noise from other confounding variables. Researchers in Canada's boreal forest studied songbirds near noisy compressor stations [75–90 dB(A) at the source, 24 hrs a day, 365 days a year] and nearly identical (and much quieter) well pads. Both of these installations were situated in two to four ha clearings with dirt access roads that were rarely used. This design allowed for control of edge effects and other confounding factors that hinder interpretation of road impact studies. The findings from this system include reduced pairing success and significantly more first time breeders near loud compressor stations in ovenbirds (*Seiurus aurocapilla*; [61]), and a one-third reduction in overall passerine bird density [62]. Low territory quality in loud sites might explain the age structuring of this ovenbird population and, if so, implicates background sound level as an important habitat characteristic. In addition to the field data above, weakened avian pair preference in high levels of noise has been shown experimentally in the lab [63]. These data suggest masking of communication calls as a possible underlying mechanism; however the reduced effectiveness of territorial defense songs, reduced auditory awareness of approaching predators (see Box 3 for a discussion of the foraging/vigilance tradeoff in noise), or reduced capacity to detect acoustic cues in foraging, cannot be excluded as explanations of the results.

A second research group, working within natural gas fields in north-west New Mexico, US, used pinyon, *Pinus edulis*-juniper, *Juniperus osteosperma* woodlands adjacent to compressor stations as treatment sites and woodlands adjacent to gas wells lacking noise-producing compressors as quiet control sites [64]. The researchers were able to turn off the loud compressor stations to perform bird counts, relieving the need to adjust for detection differences in noise [62]. This group found reduced nesting species richness but in contrast to Ref.

Box 3. Do rising background sound levels alter vigilance behavior?

Predation risk and human disturbance increase vigilance behaviors (e.g. Refs [50,86]), at a cost to foraging efficiency [87,88]. Habitat features that influence predator detection, such as vegetation height, predict predation risk [88]. If background sound level interferes with the ability of an animal to detect predators, risk can increase. Do animals perceive background sound level as a habitat characteristic that predicts predation risk? Two recent studies document increased vigilance behaviors in high levels of noise (Figure I). It seems probable that these increased anti-predator behaviors are the result of attempted visual compensation for lost auditory awareness. Evidence from ungulates near roads suggests this is the case (Figure II); however, the distinct contributions of traffic as perceived threat and traffic noise as a sensory obstacle are confounded in road studies. Experimental research with birds and mammals suggests that lost visual awareness owing to habitat obstruction reduces food-searching bouts and increases vigilance (reviewed in Ref. [89]). Although no evidence exists (but see Ref. [64]), if noise shifts the spatial distribution of foraging effort, then plant growth and seed dispersal could also be altered.

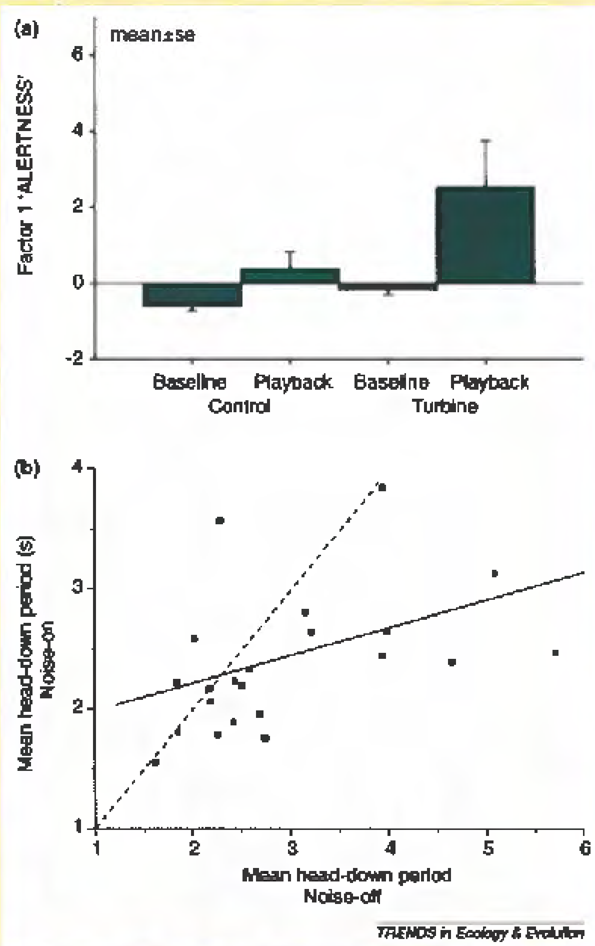


Figure I. Examples of increased vigilance behavior in noise. (a) When predator-elicited alarm calls are played back to California ground squirrels (*Spermophilus beecheyi*), adults show a greater increase in vigilance behavior at a site heavily impacted by anthropogenic noise, under power-generating wind turbines, than in a quiet control site [67]. (b) Further work on vigilance behaviors in noise comes from controlled, laboratory work with foraging chaffinches (*Fringilla coelebs*). In noise these birds decrease the interval between head-up scanning bouts, which results in fewer pecks and, thus, reduced food intake [90]. Dots depict the mean head-down period for each individual with and without white noise playback. Points below the dashed line (slope = 1) document individuals who increased scanning effort in noise. The solid regression line shows that the general trend was a more dramatic response from individuals with the lowest scanning effort. (a) adapted and (b) reproduced, with permission from Refs [67] and [90], respectively.

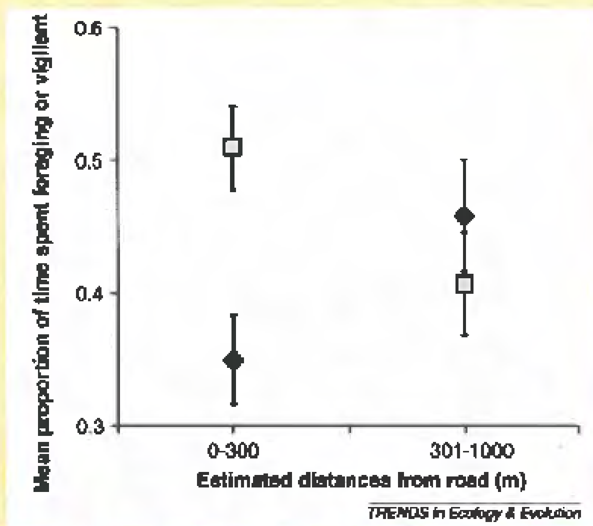


Figure II. An example of the foraging-vigilance tradeoff. Pronghorn (*Antilocapra Americana*) spend more time being vigilant (squares) and less time foraging (diamonds) within 300 meters of a road [86]. Future experiments should attempt to separate the roles of traffic as perceived threat and reduced auditory awareness on these tradeoffs. Reproduced, with permission, from Ref. [86].

[62], no reduction in overall nesting density. Unexpectedly, nest success was higher and predation levels lower in loud sites (also see Ref. [52]). The change in bird communities between loud and quiet sites appears to be driven by site preference; the response to noise ranged from positive to negative, with most responses being negative (e.g. three species nested only in loud sites and 14 species nested only in quiet, control sites). However, given the change in community structure, habitat selection based on background sound level is not the only interpretation of these data, as birds might be using cues of reduced competition pressure or predation risk to make habitat decisions [64]. The major nest predator in the study area, the western scrub jay, *Aphelocoma californica*, was significantly more likely to occupy quiet sites, which might explain the nest predation data [64]. It is probable that nest predators rely heavily on acoustic cues to find their prey. The study also found that the two bird species most strongly associated with control sites produce low-frequency communication calls. These observations suggest masking as an explanatory factor for these observed patterns. This work highlights the potential complexity of the relationship between noise exposure and the structure and function of ecological systems.

Adjusting temporal, spectral, intensity and redundancy characteristics of acoustic signals to reduce masking by noise has been demonstrated in six vertebrate orders [4–7,65]. These shifts have been documented in a variety of signal types: begging calls of bird chicks [66], alarm signals in ground squirrels [67], contact calls of primates [68], echolocation cries of bats [65] and sexual communication signals in birds, cetaceans and anurans [4–7,69]. Vocal adjustment probably comes at a cost to both energy balance and information transfer; however, no study has addressed receivers.

Masking also affects the ability of animals to use sound for spatial orientation. When traffic noise is played back to grey treefrog, *Hyla chrysoscelis* females as they attempt to localize male calls, they take longer to do so and are significantly less successful in correctly orienting to the male signal [70]. Similar studies with the European tree frog, *Hyla arborea* show decreased calling activity in played back traffic noise [71]. *H. arborea* individuals appear to be unable to adjust the frequency or duration of their calls to increase signal transmission, even at very high noise intensities (88 dB(A), [71]); although other frogs have been shown to slightly shift call frequencies upward in response to anthropogenic noise [69]. These are particularly salient points. It is likely that some species are unable to adjust the structure of their sounds to cope with noise even within the same group of organisms. These differences in vocal adaptability could partially explain why some species do well in loud environments and others do poorly [5,7,72].

Under many conditions, animals will minimize their movement sounds. For example, mice preferentially select quieter substrates on which to move [73]. Adventitious sounds of insects walking contain appreciable energy at higher frequencies (main energy ~3–30 kHz [16]) and are thus unlikely to be fully masked by most anthropogenic noise (<2 kHz [4–7]) but the spectral profile near many noise sources contains significant energy at higher frequencies (e.g. Ref [74]). Foundational work with owls and bats has shown that frequencies between approximately three and eight kHz are crucial for passive sound localization accuracy [38,75]. In fact, a recent laboratory study demonstrated that gleaning bats avoided hunting in areas with played back road noise that contained energy within this spectral band ([74]; Box 4).

Adapting to a louder world

Animals have been under constant selective pressure to distinguish pertinent sounds from background noise. Two

Box 4. Effects of acoustic masking on acoustically specialized predators

Laboratory work has demonstrated that gleaning bats (who use prey-generated sounds to capture terrestrial prey; Figure Ia) avoid noise when foraging (Figure Ib). Interestingly, treefrogs, a favorite prey of some neotropical gleaning bats, tend to call from sites with high ambient noise levels (primarily from waterfalls) and bats prefer frog calls played back in quieter locations [91]. Extinction risk in bats correlates with low wing aspect ratios (a high cost and low wing-loading morphology), a trait that all gleaning bats share [92]. A recent analysis indicates that urbanization most strongly impacts bats with these wing shapes [93]. However, low wing aspect ratio is also correlated with habitat specialization, edge intolerance and low mobility [92,93], obscuring the links between a gleaning lifestyle, louder background sound levels and extinction risk as urbanization reduces available habitat, fragments landscapes and generates noise concomitantly.

A radio-tag study showed that a gleaning bat, *Myotis bechsteinii*, was less likely to cross a roadway (three of 34 individuals) than was a sympatric open-space foraging bat, *Barbastella barbastellus* (five out of six individuals; [94]), implicating noise as a fragmenting agent for some bats. The latter species hunts flying insects using echolocation (an auditory behavior that uses ultrasonic signals above the spectrum of anthropogenic noise) [94]. Similar findings suggest acoustically mediated foragers are at risk: terrestrial insectivores were the only avian ecological guild to avoid road construction in the Amazon [95] and human-altered landscapes limited provisioning rates of saw-whet owls [96]. That these animals plausibly rely on sound for hunting might not be coincidental.

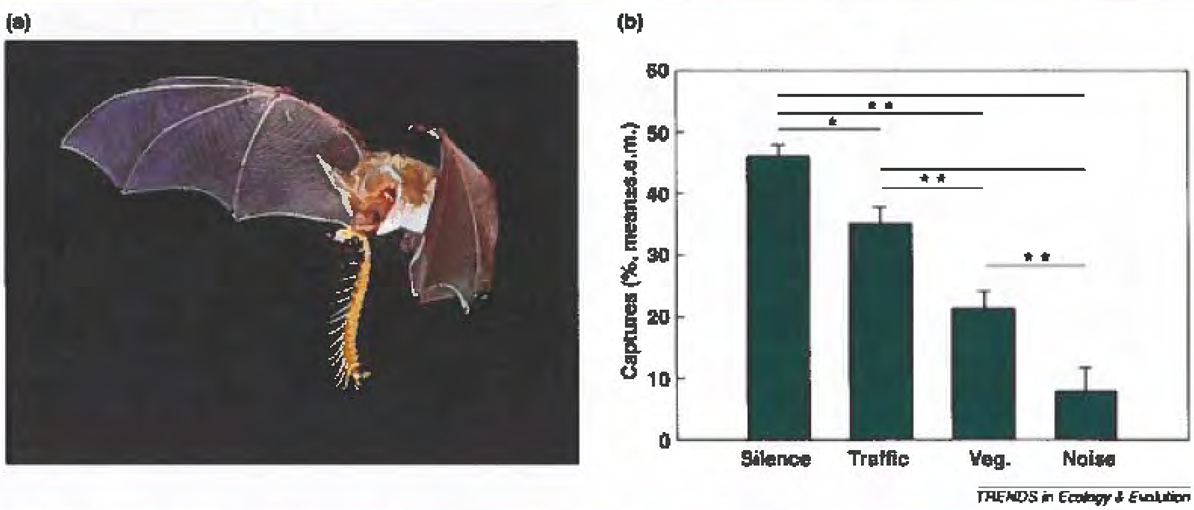


Figure I. Gleaning bats avoid hunting in noise. The pallid bat, *Antrozous pallidus* (a), relies upon prey-generated movement sounds to localize its terrestrial prey. Recent work demonstrates that another gleaning bat, the greater mouse-eared bat, *Myotis myotis*, avoids foraging in noise [74]. (b) A laboratory two-compartment choice experiment showed that this bat preferred to forage in the compartment with played-back silence versus the compartment with played-back traffic, wind-blown vegetation or white noise. This pattern held true whether the percentage of flight time, compartment entering events, the first 25 captures per session or overall capture percentage were compared across silent and noise playback compartments. Asteriks indicate the results of post repeated-measure ANOVA, paired t-tests (**$P<0.01$, *$P<0.05$, $N=7$ bats). The differences between noise types (traffic, vegetation and white noise) probably reflect increased spectral overlap between prey-generated movement sounds and the spectral profile of the noise. Reproduced with permission from Scott Altenbach (a) and Ref. [74] (b).

Box 5. Outstanding questions

- Multiple studies with birds have demonstrated signal shifts in anthropogenic noise that does not substantially overlap in frequency with the birds' song [4–7,72]. To what extent does low-frequency anthropogenic noise inhibit perception of higher frequency signals? Mammals appear more prone to the 'upward spread' of masking than do birds [85,97]. Noise commonly elevates low frequency ambient sound levels by 40 dB or more, so small amounts of spectral 'leakage' can be significant. Laboratory studies should be complimented by field studies that can identify the potential for informational or attentional effects [98]. This work should use anthropogenic noise profiles and not rely on artificial white noise as a surrogate. Furthermore, we suggest that future studies measure or model sound levels (both signal and background) at the position of the animal receiver (*sensu* Ref. [23]).
- What roles do behavioral and cognitive masking release mechanisms [85] have in modifying the capacity of free-ranging animals to detect and identify significant sounds? Only one study has examined the masked hearing thresholds of natural vocal signals in anthropogenic noise [97]. This work found that thresholds for discrimination between calls of the same bird species were consistently higher than were detection thresholds for the same calls [97]. This highlights the lack of knowledge concerning top-down cognitive constraints on signal processing in noise. Can noise divide attention and reduce task accuracy by forcing the processing of multiple streams of auditory information simultaneously [99]?
- Do animals exploit the temporal patterning of anthropogenic noise pollution (see Ref. [4])? Alternatively, what constitutes a chronic exposure and how does this vary in relation to diel activity schedules?
- Does noise amplify the barrier effects of fragmenting agents, such as roads [94,100]?
- What routes (exaptation, behavioral compensation, phenotypic plasticity and/or contemporary evolution) lead to successful tolerance of loud environments?
- What role does audition have in vigilance behaviors? Are visually mediated predators at an advantage in loud environments when prey animals rely upon acoustical predator detection?
- Do animals directly perceive background sound level as a habitat characteristic related to predation risk? A noise increase of 3 dB(A) is often identified as 'just perceptible' for humans, and an increase of 10 dB(A) as a doubling of perceived loudness. These correspond to 30% and 90% reductions in alerting distance, respectively. Do organisms assess reduced alerting distance by monitoring other acoustical signals?

examples include penguin communication systems being shaped by wind and colony noise [76] and frog systems driven to ultrasonic frequencies by stream noise [77]. A meta-analysis of the acoustic adaptation hypothesis for birdsong (the idea that signals are adapted to maximize propagation through the local habitat) found only weak evidence for this claim [78]. Physiological constraints and selective forces from eavesdropping could explain this weak relationship [78], in addition to variation of noise profiles across nominally similar habitat types (e.g. insect noise, [79]).

Phenotypic plasticity enables one adaptation to anthropogenic noise. The open-ended song learning documented in great tits, *Parus major* helps explain the consistent song shifts observed in all ten comparisons between urban and rural populations [72]. Contemporary evolution (fewer than a few hundred generations) has now been quantified in several systems [80] and we might anticipate similar microevolutionary changes in many species with rapid generation times that consistently experience acoustical environments dominated by noise, particularly in increasingly fragmented landscapes.

Perhaps the greatest predictors of the ability of a given species to succeed in a louder world will be the degree of temporal and spectral overlap of biologically crucial signals with anthropogenic noise (Figure 1), and their flexibility to compensate with other sensory modalities (e.g. vision) when auditory cues are masked. Given known sensory biases in learning [81], many animals will be constrained in their ability to shift from acoustical inputs to other sensory cues for dynamic control of complex behavioral sequences.

Conclusions and recommendations

The constraints on signal reception imposed by background sound level have a long history of being researched in bioacoustics, and it is increasingly clear that these constraints underlie crucial issues for conservation biology. Questions have been raised about the value of behavioral studies for conservation practice (for a review see Ref [82]), but ethological studies of auditory awareness and the consequences of degraded listening opportunities are essential to understanding the mechanisms underlying ecological responses to anthropogenic noise (Box 5). These studies are more challenging to execute than observation of salient behavioral responses to acute noise events, but they offer opportunities to explore fundamental questions regarding auditory perception in natural and disturbed contexts.

Chronic noise exposure is widespread. Taken individually, many of the papers cited here offer suggestive but inconclusive evidence that masking is substantially altering many ecosystems. Taken collectively, the preponderance of evidence argues for immediate action to manage noise in protected natural areas. Advances in instrumentation and methods are needed to expand research and monitoring capabilities. Explicit experimental manipulations should become an integral part of future adaptive management plans to decisively identify the most effective and efficient methods that reconcile human activities with resource management objectives [83].

The costs of noise must be understood in relation to other anthropogenic forces, to ensure effective mitigation and efficient realization of environmental goals. Noise pollution exacerbates the problems posed by habitat fragmentation and wildlife responses to human presence; therefore, highly fragmented or heavily visited locations are priority candidates for noise management. Noise management might also offer a relatively rapid tool to improve the resilience of protected lands to some of the stresses imposed by climate change. Shuttle buses and other specialized mass transit systems, such as those used at Zion and Denali National Parks, offer promising alternatives for visitor access that enable resource managers to exert better control over the timing, spatial distribution, and intensity of both noise and human disturbance. Quieting protected areas is a prudent precaution in the face of sweeping environmental changes, and a powerful affirmation of the wilderness values that inspired their creation.

Acknowledgements
We thank colleagues in the Natural Sounds Program and the NPS personnel who support monitoring projects and foster the development of acoustical resource management. We thank Lisa Angeloni, Casey Brown, Amanda Hardy, Chris Burdett and Olivier Devineau for productive discussion; Charlotte Formichella, Damon Joyce and Emma Lynch for assistance in creating figures; Shan Burson for Yellowstone monitoring data and Jessie Rinella for Lake Mead monitoring data, the researchers who shared their data and three anonymous reviewers who improved this review. The preparation of this manuscript was supported by a Postdoctoral Fellowship to J.R.B. from the National Park Service/Natural Sounds Program and Colorado State University.

References

1 Wilcove, D.S. *et al.* (1998) Quantifying threats to imperiled species in the United States. *Bioscience* 48, 607–615
2 Crooks, K.R. and Sanjayan, M., eds (2006) *Connectivity Conservation*, Cambridge University Press
3 Klump, G.M. (1996) Bird communication in a noisy world. In *Ecology and Evolution of Acoustic Communication in Birds* (Kroodsma, D.E. and Miller, E.H., eds), pp. 321–338, Cornell University Press
4 Brumm, H. and Slabbekoorn, H. (2005) Acoustic communication in noise. *Adv. Stud. Behav.* 35, 151–209
5 Patricelli, G.L. and Blickley, J.L. (2006) Avian communication in urban noise: causes and consequences of vocal adjustment. *Auk* 123, 639–649
6 Warren, P.S. *et al.* (2006) Urban bioacoustics: it's not just noise. *Anim. Behav.* 71, 491–502
7 Slabbekoorn, H. and Ripmeester, E.A. (2008) Birdsong and anthropogenic noise: implications and applications for conservation. *Mol. Ecol.* 17, 72–83
8 Fay, R.R. and Popper, A.N. (2000) Evolution of hearing in vertebrates: inner ears and processing. *Hearing Res.* 149, 1–10
9 Fay, R.R. (1988) *Hearing in Vertebrates. A Psychophysics Databook*, Hill-Fay Associates
10 Bradbury, J.W. and Vehrencamp, S.L. (1998) *Principles of Acoustic Communication*, Sinauer
11 Rabat, A. (2007) Extra auditory effects of noise in laboratory animals: the relationship between noise and sleep. *J. Am. Assoc. Lab. Anim. Sci.* 46, 35–41
12 Lyman, C.P. and Chatfield, P.O. (1955) Physiology of hibernation in mammals. *Physiol. Rev.* 35, 403–425
13 Fong, D.W. *et al.* (1995) Vestigialization and loss of nonfunctional characters. *Annu. Rev. Ecol. Syst.* 26, 249–268
14 Siemers, B.M. and Güttinger, R. (2006) Prey conspicuousness can explain apparent prey selectivity. *Curr. Biol.* 16, R157–R159
15 Goerlitz, H.R. and Siemers, B.M. (2007) Sensory ecology of prey rustling sounds: acoustical features and their classification by wild grey mouse lemurs. *Funct. Ecol.* 21, 143–153
16 Goerlitz, H.R. *et al.* (2008) Cues for acoustic detection of prey: insect rustling sounds and the influence of walking substrate. *J. Exp. Biol.* 211, 2799–2806
17 Patthey, P. *et al.* (2008) Impact of outdoor winter sports on the abundance of a key indicator species of alpine ecosystems. *J. Appl. Ecol.* 10, 2–8
18 Mallord, J.W. *et al.* (2007) Linking recreational disturbance to population size in a ground-nesting passerine. *J Appl. Ecol.* 44, 185–195
19 Bowles, A.E. (1995) Responses of wildlife to noise. In *Wildlife and Recreationists: Coexistence through Management and Research* (Knight, R.L. and Gutzwiller, J., eds), pp. 109–156, Island Press
20 Dooling, R.J. and Popper, A.N. (2007) The effects of highway noise on birds. Report to California Department of Transportation, contract 43A0139. (http://www.dot.ca.gov/hq/env/bio/avian_bioacoustics.htm)
21 Babisch, W. (2003) Stress hormones in the research on cardiovascular effects of noise. *Noise Health* 5, 1–11
22 Jarup, L. *et al.* (2008) Hypertension and exposure to noise near airports: the HYENA study. *Environ. Health Persp.* 116, 329–333
23 Nowacek, D.P. *et al.* (2007) Responses of cetaceans to anthropogenic noise. *Mammal Rev.* 37, 81–115
24 Weilgart, L.S. (2007) The impacts of anthropogenic noise on cetaceans and implications for management. *Can. J. Zool.* 85, 1091–1116
25 Riitters, K.H. and Wickham, J.D. (2003) How far to the nearest road? *Front. Ecol. Environ.* 1, 125–129
26 McGregor, P. (ed.) (2005) *Animal Communication Networks*, Cambridge University Press
27 Hansen, I.J.K. *et al.* (2005) Communication breakdown? Habitat influences on black-capped chickadee dawn choruses. *Acta Ethol.* 8, 111–120
28 Caro, T. (2005) *Antipredator Defenses in Birds and Mammals*, The University of Chicago Press
29 Marler, P. (2004) Bird calls: a cornucopia for communication. In *Nature's Music: The Science of Birdsong* (Marler, P. and Slabbekoorn, H., eds), pp. 132–176, Elsevier
30 Sloan, J.L. and Hare, J.F. (2008) The more the scarier: adult Richardson's ground squirrels (*Spermophilus richardsonii*) assess response urgency via the number of alarm signalers. *Ethology* 114, 436–443
31 Zuk, M. and Kolluru, G.R. (1998) Exploitation of sexual signals by predators and parasitoids. *Q. Rev. Biol.* 73, 415–438
32 Ridley, A.R. *et al.* (2007) Interspecific audience effects on the alarm-calling behavior of a kleptoparasitic bird. *Biol. Lett.* 3, 589–591
33 Schmidt, K.A. and Ostefeld, R.S. (2008) Eavesdropping squirrels reduce their future value of food under the perceived presence of cache robbers. *Am. Nat.* 171, 388–393
34 Mukhin, A. *et al.* (2008) Acoustic information as a distant cue for habitat recognition by nocturnally migrating passerines during landfall. *Behav. Ecol.* 19, 716–723
35 Slabbekoorn, H. and Bouton, N. (2008) Soundscape orientation: a new field in need of a sound investigation. *Anim. Behav.* 76, e5–e8
36 Montgomerie, R. and Weatherhead, P.J. (1997) How robins find worms. *Anim. Behav.* 54, 143–151
37 Neuweiler, G. (1989) Foraging ecology and audition in echolocating bats. *Trends Ecol. Evo.* 4, 160–166
38 Knudsen, E.I. and Konishi, M. (1979) Mechanisms of sound localization in the barn owl (*Tyto alba*). *J. Comp. Physiol. A* 133, 13–21
39 Rice, W.R. (1982) Acoustical location of prey by the marsh hawk: adaptation to concealed prey. *Auk* 99, 403–413
40 Hamr, J. and Bailey, E.D. (1985) Detection and discrimination of insect flight sounds by big brown bats (*Eptesicus fuscus*). *Biol. Behav.* 10, 105–121
41 Magrath, R.D. *et al.* (2007) How to be fed but not eaten: nestling response to parental food calls and the sound of a predator's footsteps. *Anim. Behav.* 74, 1117–1129
42 Bernal, X.E. *et al.* (2007) Sexual differences in the behavioral response of túngara frogs, *Physalaemus pustulosus*, to cues associated with increased predation risk. *Ethology* 755–763
43 Grafe, T.U. *et al.* (2002) Frogs flee from the sound of fire. *Proc. R. Soc. Lond. B* 269, 999–1003
44 Frid, A. and Dill, L. (2002) Human-caused disturbance stimuli as a form of predation risk. *Conserv. Ecol.* 6 In: http://www.ecologyandsociety.org/vol6/iss1/art11/main.html
45 Stankowich, T. (2008) Ungulate flight responses to human disturbance: a review and meta-analysis. *Biol. Conserv.* 141, 2159–2173
46 Laurance, W.F. *et al.* (2008) Impacts of roads, hunting, and habitat alteration of nocturnal mammals in African rainforests. *Conserv. Biol.* 22, 721–732
47 Eigenbrod, F. *et al.* (2008) The relative effects of road traffic and forest cover on anuran populations. *Biol. Conserv.* 141, 35–46
48 Goudie, R.I. (2006) Multivariate behavioural response of harlequin ducks to aircraft disturbance in Labrador. *Environ. Conserv.* 33, 28–35
49 Goldstein, M.I. *et al.* (2005) Mountain goat response to helicopter overflights in Alaska. *Wildlife Soc. B.* 33, 688–699
50 Borkowski, J.J. *et al.* (2006) Behavioral responses of bison and elk in Yellowstone to snowmobiles and snow coaches. *Ecol. Appl.* 16, 1911–1925
51 Preisler, H.K. *et al.* (2006) Statistical methods for analyzing responses of wildlife to human disturbance. *J. Appl. Ecol.* 43, 164–172
52 Barton, D.C. and Holmes, A.L. (2007) Off-highway vehicle trail impacts on breeding songbirds in northeastern California. *J. Wildl. Manage.* 71, 1617–1620
53 Doherty, K.E. *et al.* (2008) Greater sage-grouse winter habitat selection and energy development. *J. Wildl. Manage.* 72, 187–195

54 Sawyer, H. *et al.* (2006) Winter habitat selection of mule deer before and during development of a natural gas field. *J. Wildl. Manage.* 70, 396–403
55 Reed, S.E. and Merenlender, A.M. (2008) Quiet, nonconsumptive recreation reduces protected area effectiveness. *Conserv. Lett.* 1, 146–154
56 Reijnen, R. and Foppen, R. (2006) Impact of road traffic on breeding bird populations. In *The Ecology of Transportation: Managing Mobility for the Environment* (Davenport, J. and Davenport, J.L., eds), pp. 255–274, Springer
57 Reijnen, R. *et al.* (1995) The effects of car traffic on breeding bird populations in Woodland. III. Reduction of density in relation to the proximity of main roads. *J. Appl. Ecol* 32, 187–202
58 Forman, R.T.T. and Alexander, L.E. (1998) Roads and their major ecological effects. *Ann. Rev. Ecol. Syst.* 29, 207–231
59 Rheindt, F.E. (2003) The impacts of roads on birds: Does song frequency play a role in determining susceptibility to noise pollution? *J. Ornithol.* 144, 295–306
60 McGregor, R.L. *et al.* (2007) Do small mammals avoid roads because of the traffic? *J. Appl. Ecol.* 45, 117–123
61 Habib, L. *et al.* (2007) Chronic industrial noise affects pairing success and age structure of ovenbirds *Seiurus aurocapilla*. *J. Appl. Ecol.* 44, 176–184
62 Bayne, E.M. *et al.* (2008) Impacts of chronic anthropogenic noise from energy-sector activity on abundance of songbirds in the boreal forest. *Conserv. Biol.* 22, 1186–1193
63 Swaddle, J.P. and Page, L.C. (2007) High levels of environmental noise erode pair preferences in zebra finches: implications for noise pollution. *Anim. Behav.* 74, 363–368
64 Francis, C.D. *et al.* (2009) Noise pollution changes avian communities and species interactions. *Curr. Biol.* 19, 1415–1419
65 Gillam, E.H. and McCracken, G.F. (2007) Variability in the echolocation of *Tadarida brasiliensis*: effects of geography and local acoustic environment. *Anim. Behav.* 74, 277–286
66 Leonard, M.L. and Horn, A.G. (2008) Does ambient noise affect growth and begging call structure in nestling birds? *Behav. Ecol.* 19, 502–507
67 Rabin, L.A. *et al.* (2006) The effects of wind turbines on antipredator behavior in California ground squirrels (*Spermophilus beecheyi*). *Biol. Conserv.* 131, 410–420
68 Egnor, S.E.R. *et al.* (2007) Tracking silence: adjusting vocal production to avoid acoustic interference. *J. Comp. Physiol. A* 193, 477–483
69 Parris, K.M. *et al.* (2009) Frogs call at a higher pitch in traffic noise. *Ecol. Soc.* 14, 25 In: http://www.ecologyandsociety.org/vol14/iss1/art25
70 Bee, M.A. and Swanson, E.M. (2007) Auditory masking of anuran advertisement calls by road traffic noise. *Anim. Behav.* 74, 1765–1776
71 Lengagne, T. (2008) Traffic noise affects communication behaviour in a breeding anuran, *Hyla arborea*. *Biol. Conserv* 141, 2023–2031
72 Slabbekoorn, H. and den Boer-Visser, A. (2006) Cities change the songs of birds. *Curr. Biol.* 16, 2326–2331
73 Roche, B.E. *et al.* (1998) Route choice by deer mice (*Peromyscus maniculatus*): reducing the risk of auditory detection by predators. *Am. Midl. Nat.* 142, 194–197
74 Schaub, A. *et al.* (2008) Foraging bats avoid noise. *J. Exp. Biol.* 211, 3174–3180
75 Fuzessery, Z.M. *et al.* (1993) Passive sound localization of prey by the pallid bat (*Antrozous p. pallidus*). *J. Comp. Physiol.* A 171, 7667–7777
76 Aubin, T. and Jouventin, P. (2002) How to vocally identify kin in a crowd: the penguin model. *Adv. Stud. Behav.* 31, 243–277
77 Arch, V.S. and Narins, P.M. (2008) 'Silent' signals: selective forces acting on ultrasonic communication systems in terrestrial vertebrates. *Anim. Behav.* 76, 1423–1428
78 Boncoraglio, G. and Saino, N. (2007) Habitat structure and the evolution of bird song: a meta-analysis of the evidence for the acoustic adaptation hypothesis. *Funct. Ecol.* 21, 134–142
79 Kirschel, A.N.G. *et al.* (2009) Birdsong tuned to the environment green hylia song varies with elevation, tree cover, and noise. Behav. Ecol. Published online July 14, 2009
80 Stockwell, C.A. *et al.* (2003) Contemporary evolution meets conservation biology. *Trends Ecol. Evol.* 18, 94–100
81 Garcia, J. *et al.* (1974) Behavioral regulation of the milieu interne in man and rat. *Science* 185, 824–831
82 Angeloni, L. *et al.* (2008) A reassessment of the interface between conservation and behavior. *Anim. Behav.* 75, 731–737
83 Blickley J.L. and Patricelli, G.L., Impacts of anthropogenic noise on wildlife: Research priorities for the development of standards and mitigation. *J Int. Wildlife Law Policy.* (in press)
84 Miller, N.P. (2003) Transportation Noise and Recreational Lands *Noise/News International* 11(1), 1-20, International Institute of Noise Control Engineering, (http://www.hmmh.com/presentations-papers.html)
85 Moore, B.C.J. (2003) *An Introduction to the Psychology of Hearing*, (5th edn), Academic Press
86 Gavin, S.D. and Komers, P.E. (2006) Do pronghorn (*Antilocapra Americana*) perceive roads as predation risk? *Can. J. Zool* 84, 1775–1780
87 Fortin, D. *et al.* (2004) Foraging costs of vigilance in large mammalian herbivores. *Oikos* 107, 172–180
88 Verdolin, J.L. (2006) Meta-analysis of foraging and predation risk trade-offs in terrestrial systems. *Behav. Ecol. Sociobiol.* 60, 457–464
89 Bednekoff, P.A. and Blumstein, D.T. (2009) Peripheral obstructions influence marmot vigilance: integrating observational and experimental results. Behav. Ecol. Published online July 16, 2009
90 Quinn, J.L. *et al.* (2006) Noise, predation risk compensation and vigilance in the chaffinch *Fringilla coelebs*. *J. Avian Biol.* 37, 601–608
91 Tuttle, M.D. and Ryan, M.J. (1982) The role of synchronized calling, ambient light, and ambient noise, in anti-bat-predator behavior of a treefrog. *Behav. Ecol. Sociobiol.* 11, 125–131
92 Jones, K.E. *et al.* (2003) Biological correlates of extinction risk in bats. *Am. Nat.* 161, 601–614
93 Duchamp, J.E. and Swihart, R.K. (2008) Shifts in bat community structure related to evolved traits and features of human-altered landscapes. *Landsc. Ecol.* 23, 849–860
94 Kerth, G. and Melber, M. (2009) Species-specific barrier effects of a motorway on the habitat use of two threatened forest-living species. *Biol. Conserv.* 142, 270–279
95 Canaday, C. and Rivadeneyra, J. (2001) Initial effects of petroleum operation on Amazonian birds: terrestrial insectivores retreat. *Biodiv. Conserv.* 10, 567–595
96 Hinam, H.L. and St. Clair, C.C. (2008) High levels of habitat loss and fragmentation limit reproductive success by reducing home range size and provisioning rates of Northern saw-whet owls. *Biol. Conserv.* 141, 524–535
97 Lohr, B. *et al.* (2003) Detection and discrimination of natural calls in masking noise by birds: estimation of the active space of a signal. *Anim. Behav.* 65, 763–777
98 Gutschalk, A. *et al.* (2008) Neural correlates of auditory perceptual awareness under informational masking. *PLoS Biol* 6, 1156–1165
99 Barber, J.R. *et al.* (2003) Can two streams of auditory information be processed simultaneously? Evidence from the gleaning bat *Antrozous pallidus*. *J. Comp. Physiol. A* 189, 843–855
100 St. Clair, C.C. (2003) Comparative permeability of roads, rivers, and meadows to songbirds in Banff National Park. *Conserv. Biol.* 17, 1151–1160

Exhibit SM3

"The Effects of Noise on Wildlife"

The Effects of Noise on Wildlife

Research prepared by Meghan C. Sadlowski, Environmental Scientist,
Division of Migratory Bird Management, US Fish & Wildlife Service

Noise standards for wind turbines developed by countries such as Sweden and New Zealand and some specific site level standards implemented in the U.S. focus primarily on sleep disturbance and annoyance to humans. However noise standards do not generally exist for wildlife, except in a few instances where federally listed species may be impacted. Findings from recent research clearly indicate the need to better address noise-wildlife issues. As such, noise impacts to wildlife should clearly be included as a factor in wind turbine siting, construction and operation. Some of the key issues include 1) how wind facilities affect background noise levels; 2) how and what fragmentation, including acoustical fragmentation, occurs especially to species sensitive to habitat fragmentation; 3) comparison of turbine noise levels at lower valley sites – where it may be quieter – to turbines placed on ridge lines above rolling terrain where significant topographic sound shadowing can occur having the potential to significantly elevate sound levels above ambient conditions; and 4) correction and accounting of a 15 decibel (dB) underestimate from daytime wind turbine noise readings used to estimate nighttime turbine noise levels (e.g. van den Berg 2004, J. Barber Colorado State Univ. and National Park Service pers. comm., K. Fristrap National Park Service pers. comm.).

Turbine blades at normal operating speeds can generate significant levels of noise. Based on a propagation model of an industrial-scale 1.5 MW wind turbine at 263 ft hub height, positioned approximately 1,000 ft apart from neighboring turbines, the following decibel levels were determined for peak sound production. At a distance 300 ft from the blades, 45-50 dBA were detected; at 2,000 ft, 40 dBA; and at 1 mi, 30-35 dBA (Kaliski 2009). Declines in densities of woodland and grassland bird species have been shown to occur at noise thresholds between 45 and 48 dB, respectively; while the most sensitive woodland and grassland species showed declines between 35 and 43 dB, respectively. Songbirds specifically appear to be sensitive to very low sound levels equivalent to those in a library reading room (~30 dBA)[1] (Foreman and Alexander 1998). Given this knowledge, it is possible that effects to sensitive species may be occurring at ≥ 1 mile from the center of a wind facility at periods of peak sound production.

Noise does not have to be loud to have negative effects. Very low frequency sounds including infrasound are also being investigated for their possible effects on both humans and wildlife. Wind turbine noise results in a high infrasound component (Salt and Hullar 2010). Infrasound is inaudible to the human ear but this unheard sound can cause human annoyance, sensitivity, disturbance, and disorientation (Renewable Energy World 2010). For birds, bats, and other wildlife, the effects may be more profound. Noise from traffic, wind and operating turbine blades produce low frequency sounds (< 1-2 kHz; Dooling 2002, Lohr et al. 2003). Bird vocalizations are generally within the 2-5 kHz frequency range (Dooling and Popper 2007) and birds hear best between 1-5 kHz (Dooling 2002). Although traffic noise generally falls below the frequency of bird communication and hearing, several studies have documented that traffic noise can

[1] CA Department of Transportation 1998

have significant negative impacts on bird behavior, communication, and ultimately on avian health and survival (e.g., Lohr et al. 2003, Lengagne 2008, Barber et al. 2010). Whether these effects are attributable to infrasound effects or to a combination of other noise factors is not yet fully understood. However, given that wind-generated noise including blade turbine noise produces a fairly persistent, low frequency sound similar to that generated by traffic noise (Lohr et al. 2003: Dooling 2002), it is plausible that wildlife effects from these two sound sources could be similar.

A bird's inability to detect turbine noise at close range may also be problematic. For the average bird in a signal frequency of 1-4 kHz, noise must be 24-30 dB above the ambient noise level in order for a bird to detect it. As noted above, turbine blade and wind noise frequencies generally fall below the optimal hearing frequency of birds. Additionally, by the inverse square law the sound pressure level decreases by 6 dB with every doubling of distance. Therefore, although the sound level of the blade may be significantly above the ambient wind noise level and detectable by birds at the source, as the distance from the source increases and the blade noise level decreases toward the ambient wind noise level, a bird may lose its ability to detect the blade and risk colliding with the moving blade. A bird approaching a moving blade under high wind conditions may be unable to see the blade due to motion smear, and may not hear the blade until it is very close – if it is able to hear it at all (Dooling 2002). Another concern involves the effect of ambient noise on communication distance and an animal's ability to detect calls. For effects to birds, this can mean 1) behavioral and/or physiological effects, 2) damage to hearing from acoustic over-exposure, and 3) masking of communication signals and other biologically relevant sounds (Dooling and Popper 2007). Of the 49 bird species whose behavioral audibility curves and/or physiological recordings have been determined, Dooling and Popper (2007) developed a conceptual model for estimating the masking effects of noise on birds. Based on the distance between birds and the spectrum level, bird communication was predicted to be "at risk" (e.g., at ~ 755 ft distance where noise was 20 dB), "difficult" (.e.g., at ~755 ft where noise was 25 dB) and "impossible" (e.g., at ~755 ft where noise was 30 dB). While clearly there is variation between species and there is no single noise level where one- size-fits-all, this masking effect of turbine blades is of concern and should be considered as part of the cumulative impacts analysis of a wind facility on wildlife. It must be recognized that noise in the frequency region of avian vocalizations will be most effective in masking these vocalizations (Dooling 2007).

Barber et al. (2010) assessed the threats of chronic noise exposure, focusing on grouse communication calls, urban bird calls, and other songbird communications. They determined that while some birds were able to shift their vocalizations to reduce the masking effects of noise, when shifts did not occur or were insignificant, masking could prove detrimental to the health and survival of wildlife (Barber et al. 2010). Although much is still unknown in the real world about the masking effects of noise on wildlife, the results of a physical model analyzing the impacts of transportation noise on the listening area[2] of animals resulted in some significant findings. With a noise increase of

[2] The listening area is the active space of vocalization in which animals search for sounds (Barber et al. 2010).

just 3 dB – a noise level indentified as "just perceptible to humans" – this increase corresponded to a 50% loss of listening area for wildlife (Barber et al. 2010). Other data suggest noise increases of 3 dB to 10 dB correspond to 30% to 90% reductions in alerting distances[3] for wildlife, respectively (Barber et al. 2010). Impacts of noise could thus be putting species at risk by impairing signaling and listening capabilities necessary for successful communication and survival.

Swaddle and Page (2007) tested the effects of environmental noise on pair preference selection of Zebra Finches. They noted a significant decrease in females' preference for their pair-bonded males under high environmental noise conditions. Bayne et al. (2008) found that areas near noiseless energy facilities had a total passerine density 1.5 times greater than areas near noise-producing energy facilities. Specifically, White-throated Sparrows, Yellow-rumped Warblers, and Red-eyed Vireos were less dense in noisy areas. Habib et al. (2007) found a significant reduction in Ovenbird pairing success at compressor sites (averaging 77% success) compared to noiseless well pads (92%). Quinn et al. (2006) found that noise increases perceived predation risk in Chaffinches, leading to increased vigilance and reduced food intake rates, a behavior which could over time result in reduced fitness. Francis et al. (2009) showed that noise alone reduced nesting species richness and led to a different composition of avian communities. While they found that noise disturbance ranged from positive to negative, responses were predominately negative.

Schaub et al. (2008) investigated the influence of background noise on the foraging efficiency and foraging success of the greater mouse-eared bat, a model selected because it represents an especially vulnerable group of gleaning bats that rely on their capability to listen for prey rustling sounds to locate food. Their study clearly found that traffic noise, and other sources of intense, broadband noise deterred bats from foraging in areas where these noise were present presumably because these sounds masked relevant sounds or echos the bats use to locate food.

Although there are few studies specifically focused on the noise effects of wind energy facilities on birds, bats and other wildlife, scientific evidence regarding the effects of other noise sources is widely documented. The results show, as documented in various examples above, that varying sources and levels noise can affect both the sending and receiving of important acoustic signaling and sounds. This also can cause behavioral modifications in certain species of birds and bats such as decreased foraging and mating success and overall avoidance of noisy areas. The inaudible frequencies of sound may also have negative impacts to wildlife. Given the mounting evidence regarding the negative impacts of noise – specifically low frequency levels of noise such as those created by wind turbines on birds, bats and other wildlife, it is important to take precautionary measures to ensure that noise impacts at wind facilities are thoroughly investigated prior to development. Noise impacts to wildlife must be considered during the landscape site evaluation and construction processes. As research specific to noise effects from wind turbines further evolves these findings should be utilized to develop technologies and measures to further minimize noise impacts to wildlife.

[3] The alerting distance is the maximum distance at which a signal can be heard by an animal and is particularly important for detecting threats (Barber et al. 2010).

REFERENCES:

Barber, J.R., K.R. Crooks, and K. Fristrup. 2010. The costs of chronic noise exposure for terrestrial organisms. Trends Ecology and Evolution 25(3): 180–189. Available at: http://www.sciencedirect.com/

Bayne, E.M., L. Habib and S. Boutin. 2008. Impacts of Chronic Anthropogenic Noise from Energy-Sector Activity on Abundance of Songbirds in the Boreal Forest. Conservation Biology 22(5) 1186-1193. Available at: http://oz.biology.ualberta.ca/faculty/stan_boutin/uploads/pdfs/Bayne%20etal%202008 %20ConBio.pdf

California Department of Transportation, Technical Noise Supplemental, A Technical Supplement to the Traffic Noise Analysis Protocol, October 1998. Available at: http://www.dot.ca.gov/hq/env/noise/pub/Technical%20Noise%20Supplement.pdf

Dooling, R. 2002. *Avian Hearing and the Avoidance of Wind Turbines.* National Renewable Energy Laboratory, NREL/TP-500-30844. 83 p. Available at: http://www.nrel.gov/wind/pdfs/30844.pdf

Dooling R. J., and A. N. Popper. 2007. The effects of highway noise on birds. Report to the California. Department of Transportation, contract 43AO139. California Department of Transportation, Division of Environmental Analysis, Sacramento, California, USA. Available at: http://www.dot.ca.gov/hq/env/bio/files/caltrans_birds_10-7-2007b.pdf

Foreman, Richard T.T. and L.E. Alexander. 1998. Roads and their major ecological effects. Annual Review of Ecological Systems 29: 207-231. Available at: http://pracownia.org.pl/pliki/roads_and_their_major_ecological_effects.pdf

Francis, C.D., C.P. Ortega and A. Cruz. 2009. Noise Pollution Changes Avian Communities and Species Interactions. Current Biology, in press, doi: 10.1016/j.cub.2009.06.052. Available at: http://www.sciencedirect.com/

Habib, L, E.M. Bayne and S. Boutin. 2007. Chronic industrial noise affects pairing success and age structure of ovenbirds *Seiurus aurocapilla.* Journal of Applied Ecology 44: 176-184. Available at: http://oz.biology.ualberta.ca/faculty/stan_boutin/ilm/uploads/pdfs/Habib%20etal%202 007%20JAE.pdf

Kaliski, K. 2009. Calibrating Sound Propagation Models for Wind Power Projects, State of the Art in Wind Siting Seminar, October. National Wind Coordinating Collaborative.

SWAINSON'S HAWK NESTING POPULATION IN THE ANTELOPE VALLEY OF THE WESTERN MOJAVE DESERT, CALIFORNIA

PETER H. BLOOM, RAINEY G. BARTON, and MICHAEL J. KUEHN, Bloom Research Inc.,13611 Hewes Avenue, Santa Ana, California 92705; petebloom@bloomresearch.org

ABSTRACT: The Swainson's Hawk (*Buteo swainsoni*) has a long history of breeding in California, but a severe decline in the statewide breeding population was identified in 1979, when in all of southern California only two pairs were found, one in the Antelope Valley of the western Mojave Desert. That area was little studied until we began banding Swainson's Hawks there in 1997. Over 20 breeding seasons between 1979 and 2022, we documented in the Antelope Valley 124 attempts to nest, in which the mean clutch and brood sizes were 2.49 and 2.37, respectively. From 2004 through 2006, we observed two to four breeding pairs annually; from 2009 through 2022, three to 14 breeding pairs. The rate of success of the 91 nests revisited to determine if any young fledged was 64%. Nest trees consisted of 81.5% non-native species, 13.7% native species, including Joshua trees (*Yucca brevifolia*), and 4.8% unidentified deciduous trees. Between 1997 and 2022, in 50 nests, we recorded 170 vertebrate prey items, of which 90 were gophers (*Thomomys bottae*). Though the Antelope Valley population has grown since 1980, its nesting and foraging habitat now face multiple threats. To conserve occupied nesting territories, we recommend creation of nesting and foraging habitat reserves that include both native desert and cultivated alfalfa close to existing conserved land.

Swainson's Hawk (*Buteo swainsoni*) breeds throughout the wide-open spaces of western North America, spending six months on the breeding grounds and six months migrating or wintering, mostly in Argentina (Brown and Amadon 1968, Bechard et al. 2020), though recently it has begun wintering (short stopping) in western Mexico (Airola et al. 2019). The occurrence of Swainson's Hawk in southern California, and specifically Los Angeles County, dates back to the Pleistocene (Stock 1930). There is considerable historical evidence of a large coastal southern California breeding population that extended south into northern Baja California, potentially as far as Ensenada de Todos Santos, Baja California (Bent 1937). From museum records, between 1880 and 1933, 132 Swainson's Hawk egg sets were collected in California, 20 of them in cismontane Los Angeles County (Bloom 1980). In their overview of California birds, Grinnell and Miller (1944) were the first to report the statewide reduction in the number and breeding distribution of Swainson's Hawk. From about 1940 to 1979, California experienced an estimated 91% decline in its breeding population with nearly complete extirpation of breeding pairs below the 36th parallel, essentially all southern California (Bloom 1980).

Prior to the 1979 survey, the last known attempts of Swainson's Hawk to nest in southern California were in 1933, when Ed N. Harrison collected three sets of eggs in northwestern San Diego County (WFVZ, Western Foundation of Vertebrate Zoology; https://collections.wfvz.org/; EN-173347, EN-173348, EN-173349), 1939, when James B. Dixon took a set of eggs near Adelanto, San Bernardino County (WFVZ EN 29168), and 1946, when Sidney B. Peyton collected a set of eggs near Adelanto (WFVZ EN-82220; Bloom 1980).

The Antelope Valley of northern Los Angeles and southern Kern counties has a long history of documentation of Swainson's Hawks, beginning with 28 specimens, mostly adults, collected during the breeding season between 7 July 1904 and 1 April 1931 near Neenach, Lancaster, or Palmdale (MCZ, Museum of Comparative Zoology, Harvard University, http://digir.mcz.harvard.edu/ipt/resource?r=mczbase; WFVZ, https://collections.wfvz.org/; MVZ, Museum of Vertebrate Zoology, https://arctos.database.museum/SpecimenSearch.cfm?guid_prefix=MVZ%3ABird). Of the two nests from which eggs were collected near Palmdale, one was in a "yucca palm," presumably a Joshua tree (*Yucca brevifolia*). More evidence of nesting in the Mojave Desert just to the east in San Bernardino County includes five nests in the vicinity of Victorville between 1916 and 1946, four of which were in Joshua trees (Bloom 1980). No observations of nesting Swainson's Hawks were confirmed in the Antelope Valley between 1931 and 1978 (K. Garrett pers. comm., Bloom 1980).

Concern over Swainson's Hawk's statewide decline prompted surveys in 1978 and 1979. These revealed only two nesting territories remaining in southern California, one northeast of Lancaster in the eastern portion of the Antelope Valley (K. Garrett pers. comm.) and one near Cima in eastern San Bernardino County (E. A. Cardiff pers. comm., Bloom 1980; Figure 1). In both years, the attempts in the Antelope Valley failed during nest building or incubation, and no active nests were found in 1980 (K. Garrett pers. comm.). The territory near Cima was inactive whenever visited over multiple years from 1981 to 2022. Although there were museum specimens and observations from the Antelope Valley over the 17 years following 1979, Swainson's Hawk was not confirmed nesting there again until 1997, when we began our efforts at banding.

Since at least 1979 (Bloom 1980), the Antelope Valley Swainson's Hawk population has been relatively isolated from other breeding pairs but has become less isolated over the last 10 years (Bloom unpubl. data). The closest known active nesting territories found in the last 10 years include 11 from 40 to 70 km north of the western end of the Antelope Valley in the vicinity of Bakersfield, Bealville, and Caliente, Kern County (observed in 2016, 2018, and 2020; Bloom unpubl. data), one 5.5 km south of Owens Dry Lake near Olancha, Inyo County (observed in 2015, 2016, and 2021; Bloom unpubl. data), and one isolated nesting territory approximately 130 km south of the Antelope Valley at Naval Weapons Station Seal Beach, Orange County (observed in 2019, 2020, 2021, and 2022; R.S. Winkleman pers. comm; Figure 1).

Here we detail the history, ecology, productivity, and diet of Swainson's Hawks nesting in the Antelope Valley, on the basis of intermittent surveys and banding of nestlings and adults from 1979 to 2022 (Figure 1).

STUDY AREA

If the surrounding native desert habitats and fragments of native habitat remaining on the valley floor are representative of what occurred historically, prior to the advent of agriculture, the Antelope Valley was dominated by Joshua tree woodland, creosote bush (*Larrea tridentata*), burrow-weed (*Ambrosia dumosa*), rabbitbrush (*Ericameria* spp.), and saltbrush (*Atriplex*

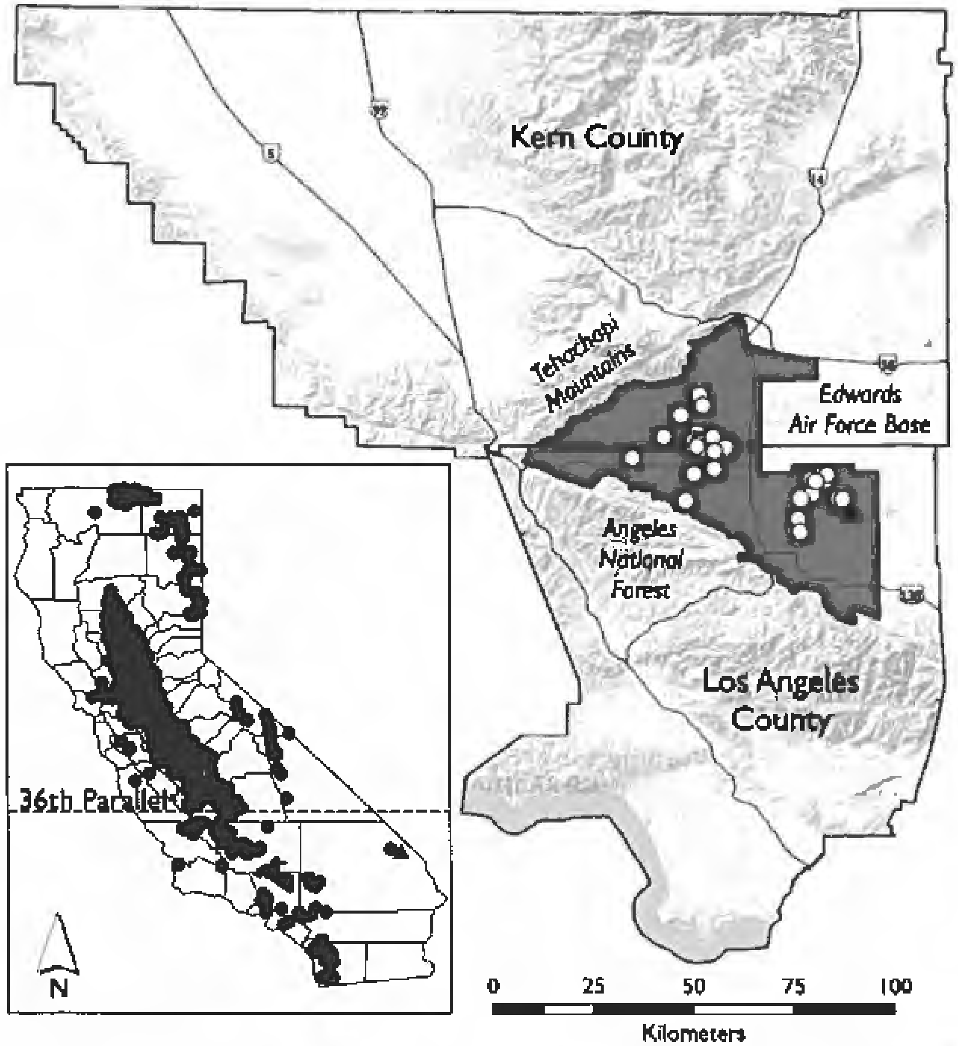


FIGURE 1. Antelope Valley Swainson's Hawk nesting locations and study area (1979–2022), Kern and Los Angeles counties, California. Gray shading, the study area; black triangles, 1978–1979 nest locations (K. Garrett pers. comm., Bloom 1980); black dots, all California nest locations outside of the study area (Calif. Dept. Fish and Wildlife data); white dots, Antelope Valley nest locations, 1997–2022.

spp.). Joshua trees were likely the principal sites of Swainson's Hawk nests in the Antelope Valley in the early 20th century before thousands of hectares of native desert habitats were removed and replaced with alfalfa, a crop that supports abundant vertebrate prey and in which Swainson's Hawk forages regularly throughout California (Bloom 1980, unpubl. data, Woodbridge 1991, Babcock 1995, Briggs et al. 2011). As the local groundwater basin has been depleted (*Los Angeles County Waterworks District No. 40 v. Diamond Farming Co.*; Antelope Valley Groundwater Cases 2020), fallow alfalfa fields have become the dominant habitat. While the maximum number of hectares under alfalfa cultivation in the early to mid-1900s is unknown, from 1948 to 1988 the number of hectares of alfalfa in the Antelope Valley fell from ap proximately 25,000 to 4000 (Templin et al. 1995). By 2021 we estimate active alfalfa to have decreased further, to 2000 hectares.

As of 2022, solar-energy facilities, residential development, and wind farms have expanded over much of the Antelope Valley, now occupying former native desert and agricultural land.

METHODS

The 1979 Swainson's Hawk survey of the Antelope Valley was part of a California statewide effort focused on the species' nesting habitat. These "windshield" surveys entailed driving at 40 to 48 km/hour with periodic stops to survey potential nesting habitat (Bloom 1980). We followed the same procedures in more recent years. The objective was to locate all nesting ter ritories to identify and locate any population decline. Binoculars and a 25- to 60-power spotting scope were used to search for hawks and confirm active nests. Five days in May 1979 were dedicated to the Antelope Valley floor from 300th Street West east to 170th Street East, and from Willow Springs in the north to Palmdale in the south, excluding Edwards Air Force Base. The same area was surveyed again in 1980, 1997, 1998, 2016, 2018, and 2020, when we attempted censuses of the population. Targeted surveys of known nesting territories, with limited searching for new territories, were conducted from 2004 through 2006, 2009 through 2015, and in 2017, 2019, 2021, and 2022.

We define an active nest as one newly built or recently added to, with adults present on the nest and or defending it, or with eggs or young present. Upon finding an active Swainson's Hawk nest, we climbed the nest tree to band young, collect unhatched eggs, and identify prey remains. Bloom identified prey from whole and partial carcasses, feathers, tails, claws, skulls, mandibles, and other skeletal remains and teeth in the nest. No prey remains were brought into the lab for identification; all were left at the nest. Trees were climbed only once to reduce disturbance and potential predation by the bobcat (*Lynx rufus*; Bloom 1974). We recorded each nest's location (Figure 1), date, number of young, success, supporting tree, and, if the nest was entered, prey species and clutch size. We considered a nest successful if at least one young fledged. We considered a chick to have fledged if it was at least three-quarters grown (5.5 weeks old) during the final observation (Steenhof 1987). Chicks older than 2.5 weeks were banded with U.S. Geological Survey aluminum bands and beginning in 2011 were also banded with alphanumeric color bands. Addled eggs were deposited at the WFVZ.

RESULTS

Current Population Status

After 1980, the population of Swainson's Hawk in the Antelope Valley began to increase and spread, in both native desert and agricultural areas. From 1995 to 1999 breeding was probable or confirmed in the Antelope Valley in four blocks defined for the Los Angeles County breeding bird atlas (Allen et al. 2016). Five pairs may have been present in the Antelope Valley in 2005. Our highest count of active territories was 14 in 2021, of which three successfully fledged young, two failed with chicks in the nest, and nine failed prior to confirmation of hatching. In five other years from 2015 to 2022 we located 9 to 11 active nests (Table 1).

Nesting Ecology

From 1979 to 2022 we documented 124 nest attempts. Of these, 22 failed prior to egg laying or without egg laying being confirmed. In the remaining 102 nests, which contained at least one egg, if not also young, the average clutch size was 2.49 eggs (SE = 0.08). In the 99 nests that contained at least one young the average brood size was 2.37 chicks (SE = 0.08; Table 1). For the 91 nests revisited to determine if any young fledged, the rate of success ranged from 0% in 1979 and 2017 to 100% in 1997, 1998, 2004 through 2006, and in 2009 and 2010, averaging 64.4% across all years. A minimum of 126 young were fledged from 54 nests over the 20-year study period (Table 1). However, we did not revisit all nests to determine if young had fledged. Between 1991 and 2022, 198 young were banded, of which 124 received auxiliary bands with an alphanumeric code.

TABLE 1 Annual Nest Success and Size of Clutch and Brood for Swainson's Hawk Nests in the Antelope Valley, California, 1979–2022

Year	Nests[a]	Chicks banded	Percent successful[b]	Mean clutch size[c]	Mean brood size
1979	1	0	0		
1997	2	6	100	3	3
1998	3	6	100	3.33	3.33
2004	2	3	100	1.5	1.5
2005	4	11	100	2.75	2.75
2006	4	7	100	3.25	2.75
2009	6	16	100	3	2.83
2010	5	13	100	3.2	2.6
2011	8	20	75	3.29	3
2012	6	7	50	1.75	1.75
2013	6	3	50	2	1.6
2014	5	8	50	2.2	2
2015	10	21	80	2.89	2.78
2016	6	9	40	2	1.83
2017	9	17	0	2.5	2.25
2018	11	16	50	2.25	2.25
2019	3	1	33.3	2	2
2020	9	21	88.9	2.67	2.33
2021	14	5	21.4	1.4	1.4
2022	10	8	50	1.89	1.44
Mean		198	64.4	2.49	2.37
SE				0.08	0.08

[a]Number of active nests observed.

[b]Nests which had young ≥5.5 weeks old at the time of the last observation.

[c]Not all nests which failed prior to young being observed were examined for the presence of eggs.

Trees supporting Swainson's Hawk nests were largely non-native species (81.5%, *n* = 101), including elm (*Ulmus* spp., 43 nesting attempts), Aleppo pine (*Pinus halepensis*, 32), black locust (*Robinia pseudoacacia*, 10), tamarisk (*Tamarix* spp., 11), and Arizona cypress (*Cupressus arizonica*, 5). Native trees constituted 13.7% (*n* = 17) of the 124 observed nesting attempts, which included eight attempts in Joshua trees, six in Fremont cottonwoods (*Populus fremontii*), two in willows (*Salix* spp.), and one in a California juniper (*Juniperus californica*). An additional six nest trees (4.8%), most of which have either died or have been removed, were documented as deciduous without being identified to species. Figures 2 and 3 show examples of non-native and native trees in which Swainson's Hawks nested and the surrounding habitat.

Diet and Feeding Ecology

Between 1979 and 2022 we recorded the remains of 170 prey observed in 50 Swainson's Hawk nests, identified to the lowest taxonomic level allowable (Table 2). Observed prey consisted entirely of vertebrates and almost entirely of rodents (82.9%, *n* = 141). Birds represented 7.1% (*n* = 12) of the total prey items, followed by reptiles (6.5%, *n* = 11) and amphibians (3.5%, *n* = 6). Of the 170 items observed, Botta's pocket gopher (*Thomomys bottae*) was the dominant prey, found in 23 nests. The California vole (*Microtus californicus*) and California ground squirrel (*Otospermophilus beecheyi*) were the next most common prey, observed in 10 and five nests, respectively.

FIGURE 2. Swainson's Hawk nest in elm tree and adjacent alfalfa fields in the Antelope Valley, Los Angeles County, California, July 2020.

Photo by Peter H. Bloom

FIGURE 3. Swainson's Hawk nest in Joshua tree in native desert in the Antelope Valley, Kern County, California, November 2021.

Photo by Kerry G. Ross

DISCUSSION

Population and Distribution

Except for the isolated territory near Olancha, Inyo County, the substantial increase in the Antelope Valley population is the only change in the nesting distribution of Swainson's Hawk in the Mojave Desert since the 1979 statewide survey that led to the species being listed as threatened. While California's entire Mojave Desert has not been systematically surveyed for nesting Swain son's Hawks, much of the known nesting habitat (Joshua tree woodland) has been examined repeatedly over the decades, and hundreds of Red-tailed Hawk (*Buteo jamaicensis*), Great Horned Owl (*Bubo virginianus*), and Common Raven (*Corvus corax*) nests have been identified and their young banded (Bloom unpubl. data). In addition, no nests of Swainson's Hawk in areas of the Mojave Desert outside of the Antelope Valley have been reported to us or through https.eBird.org, www.iNaturalist.org, or the California Natural Diversity Data Base (https://map.dfg.ca.gov/rarefind/view/RareFind.aspx, accessed 3 Apr 2021).

Population increases documented in other portions of the species' range during this same period (Battistone et al. 2019, Furnas et al. 2022) were not taking place in the Mojave Desert outside of the Antelope Valley. As a nesting species, Swainson's Hawk may always have been less abundant in the Mojave Desert (Grinnell and Miller 1944) than west of the Tehachapi Mountains and Sierra Nevada. Given the extensive distribution of the Joshua tree (Munz 1974), nesting is plausible anywhere the tree is found in California. Dawson

Table 2 Prey Observed in Swainson's Hawk Nests in the Antelope Valley, California, 1979–2022.

Common name	Scientific name	Quantity	Percent
Botta's pocket gopher	*Thomomys bottae*	90	52.9
California vole	*Microtus californicus*	14	8.2
California ground squirrel	*Otospermophilus beecheyi*	12	7.1
Merriam's kangaroo rat	*Dipodomys merriami*	9	5.3
Desert cottontail	*Sylvilagus audubonii*	8	4.7
Unidentified snake		7	4.1
Black tailed jackrabbit	*Lepus californicus*	5	2.9
Unidentified frog or toad		5	2.9
Horned Lark	*Eremophila alpestris*	4	2.4
Unidentified rodent		3	1.8
Eurasian Collared Dove	*Streptopelia decaocto*	2	1.2
Western whiptail	*Aspidoscelis tigris*	2	1.2
Unidentified passerine		2	1.2
Western Meadowlark	*Sturnella neglecta*	1	0.6
Brewer's Blackbird	*Euphagus cyanocephalus*	1	0.6
Barn Owl	*Tyto alba*	1	0.6
Domestic poultry	*Gallus* spp.	1	0.6
Desert horned lizard	*Phrynosoma platyrhinos*	1	0.6
Western toad	*Anaxyrus boreas*	1	0.6
Unidentified lizard		1	0.6
Total		170	100

(1923) referred to Swainson's Hawk as "less common on the south-eastern deserts" (probably referring to the Colorado Desert) and included a photo of a nest in a Joshua tree in the Mojave Desert, as did Bent (1937). Grinnell and Miller (1944) referred to it as "apparently scarce in summer on Colorado and Mojave Deserts; but known to nest near Cima, San Bernardino County." While the historic distribution of nesting Swainson's Hawks in the Mojave Desert was widely dispersed around the Antelope Valley, Olancha, Adelanto, Victorville, and Cima, the core nesting population is now found in the Antelope Valley.

Four decades have elapsed since the 1979 survey of the Antelope Valley, and the population has experienced a 14 fold increase. The ultimate cause of the increase in the Antelope Valley and Central Valley (Battistone et al. 2019) is unknown since no active management was undertaken in the preceding decades. However, the Central Valley population was recognized as the core state population and reproductively healthy. Whether the increase in Antelope Valley nesting pairs was a product of local reproduction or immigration is unknown. Equally puzzling is why so little breeding has extended to southern California counties except in the Antelope Valley. Given the species' propensity for short-distance dispersal (Woodbridge et al. 1995), all or the majority of adults breeding in the Antelope Valley likely fledged from nests in the Antelope Valley. However, occasional long-distance dispersers are known and could have originated from the core population in the Central Valley or a state other than California (Bloom unpubl.). The species' typically short distance of natal dispersal may have contributed to the growing population's failure to spread widely in southern California.

Nesting Ecology

Although our sample size was relatively small (124 nests), the combined annual reproductive performance of Swainson's Hawks in the Antelope Valley was like that reported elsewhere in California and western North America (Bloom 1980, Woodbridge et al. 1995, England et al. 1995). The species' clutches typically range from one to four eggs; average clutch sizes have been reported as 2.66 in Washington, 2.34 in Colorado, and 2.48 in New Mexico (Bechard et al. 2020). Similarly, Bloom (1980) reported an average clutch size of 2.58 and an average brood size of 2.27 for California nesting pairs. Our observed mean clutch size of 2.49 and mean brood size of 2.37 are consistent with the results of other studies. Various studies throughout the western U.S. found that pairs were successful in 54.6% (Olendorff 1978), 65% (Woodbridge et al. 1995), 64.7% to 82.1% (England et al. 1995), 81.3% (Fitzner 1978), and 89.5% (Bloom 1980) of reproductive attempts. The success rate of 64.4% in the Antelope Valley is consistent with these findings. However, this rate may be biased toward lower success, in part because of our not returning to all nest sites to confirm if young fledged and the assumption that young less than 5.5 weeks old at the time of the last observation did not successfully fledge. Thus our findings support the conclusion of Risebrough et al. (1989) that in California organochlorine pesticides were not a significant factor in the decline in the state's breeding population.

Five mostly drought-tolerant species of exotic trees currently provide the majority of nest sites for Swainson's Hawks in the Mojave Desert (81.5%), while native Fremont cottonwoods, Joshua trees, California junipers, and willows provide nest substrates only occasionally (13.7%).

Diet and Feeding Ecology

In California (Bloom 1980, Woodbridge 1991) and throughout the western United States (Andersen 1995, Bechard 1983, Gilmer and Stewart 1984), Swainson's Hawks have a strong preference for ground squirrels, gophers, and voles. We found the same to be true in the Antelope Valley where gophers, voles, and ground squirrels accounted for 68% ($n = 116$) of the prey observed in Swainson's Hawk nests. These rodents occur in both disturbed and native desert habitats in the Antelope Valley, but active alfalfa fields support the highest abundance of gophers and voles (pers. obs.). This observation is consistent with the high densities of gophers, mice, and voles in pastureland (including alfalfa) found in Washington by Bechard (1982) as well as Wood bridge's (1991) finding of a high abundance of voles, ground squirrels, and gophers in alfalfa fields in northern California. One nest that we examined adjacent to alfalfa fields contained the remains of 59 gophers, four ground squirrels, one cottontail, and two jackrabbits. This nest successfully fledged four offspring and contained 38.8% (n 66) of all prey items observed ($n =$ 170) and 65.6% of all gophers observed. We found Swainson's Hawk nests in native desert rarely to yield more than three prey items. In the Antelope Valley, reptiles such as the desert horned lizard ($n = 1$) and western whiptail ($n = 2$) are found almost exclusively in pristine native desert habitats, and our finding their remains in Swainson's Hawk nests suggest that despite the species' predilection for foraging in alfalfa fields when nesting in agricultural areas (Bloom 1980, Woodbridge 1991), adults also hunt in native habitats.

Conservation and Future Studies

While the population has grown in recent years, it is under increasing pressure from the conversion of nesting and foraging habitat to solar-energy facilities, residential housing, wind farms, and other development. These landscape-level changes in the Antelope Valley are incompatible with continued Swainson's Hawk nesting and foraging. Along with the diminishing availability of water in the Mojave basin and climate change, these factors have cumulative and compounding effects, potentially setting the stage for a significant and rapid population decline. Creation of reserves dedicated to the conservation of both foraging and nesting habitat for nesting and migrating Swainson's Hawks in the Antelope Valley should include both native desert and alfalfa components and be located as close to nesting territories and existing reserves as possible.

The Antelope Valley's population of Swainson's Hawk has been understudied in comparison to breeding populations elsewhere in California. A telemetry study involving adults equipped with GPS/GSM transmitters would provide considerable insight into how Mojave Desert Swainson's Hawks use their territories, which may include native desert, agricultural crops, solar fields, and windfarms, among other areas. Further, a telemetry study may allow for the customization of conservation areas, based upon movements of specific nesting pairs and known territory configurations. The population is small and may have been reestablished by the single pair found in 1978 and

1979. Therefore the species' strong philopatry and territory fidelity suggest that individuals may be closely related, a hypothesis to be tested with genetic studies. If conservation efforts are successful and this population continues to expand, it may serve as a source for recolonization of other regions of southern California in which Swainson's Hawks once nested.

ACKNOWLEDGMENTS

We thank the many wildlife biologists and volunteers who contributed to this project over the years, including Hal Batzloff, Breanna Bartels, Jaqueline Catino-Davenport, Alexandra Gresham, Dave Hacker, Scott Harris, Jeff Kidd, Bev McIntosh, Karly Moore, Chris Niemela, Nicollet Overby, Joe Papp, Daniel Potter, Katie Quint, Steve Ritt, Kerry Ross, Karyn Sernka, Cheryl Thomas, Scott Thomas, Eric Weiss, and Erinn Wilson-Olgin. The comments of Carie Battistone, Kimball Garrett, Lloyd Kiff, Chris Vennum, and Chris Swarth significantly improved the manuscript. Funding and support was provided by Bloom Research, Inc., EDF Renewables, Defenders of Wildlife, the Swainson's Hawk Technical Advisory Committee, Sea and Sage Audubon Society, and the National Audubon Society. We acknowledge Tip Top Arborists for their assistance and professionalism.

LITERATURE CITED

Airola, D. A., Estep, J. A., Krolick, D. R., Anderson, R. L., and Peters, J. R. 2019. Wintering areas and migration characteristics of Swainson's Hawks that breed in the Central Valley of California. J. Raptor Res. 53:237–252.

Allen, L. W., Garrett, K. L., and Wimer, M. C. 2016. Los Angeles County Breeding Bird Atlas. Los Angeles Audubon Soc., Los Angeles.

Andersen, D. E. 1995. Productivity, food habits, and behavior of Swainson's Hawks breeding in southeast Colorado. J. Raptor Res. 29:158–165.

Antelope Valley Groundwater Cases. 2020. Calif. Appeals Court, 5th Dist. 58:343.

Babcock, K. W. 1995. Home-range and habitat use of breeding Swainson's Hawks in the Sacramento Valley of California. J. Raptor Res. 29:193–197.

Battistone, C. L., Furnas, B. J., Anderson, R. L., Dinsdale, J. L., Cripe, K. M., Estep, J. A., Chun, C. S. Y., and Torres, S. G. 2019. Population and distribution of Swainson's Hawks (*Buteo swainsoni*) in California's Great Valley: A framework for long-term monitoring. J. Raptor Res. 53:253–265; doi.org/10.3356/JRR-18-34.

Bechard, M. J. 1982. Vegetative cover on foraging site selection by Swainson's Hawk. Condor 84:153–159; doi.org/10.2307/1367658.

Bechard, M. J. 1983. Food supply and the occurrence of brood reduction in Swainson's Hawk. Wilson Bull. 95:233–242.

Bechard, M. J., Houston, C. S., Saransola, J. H., and England, A. S. 2020. Swainson's Hawk (*Buteo swainsoni*), *in* The Birds of the World (A. F. Poole, ed.). Cornell Lab Ornithol., Ithaca, NY; doi.org/10.2173/bow.swahaw.01.

Bent, A. C. 1937. Life histories of North American birds of prey. U.S. Natl. Mus. Bull. 167; doi.org/10.5479/si.03629236.167.i.

Bloom, P. H. 1974. Some precautions to be used in banding studies of nestling raptors. W. Bird Bander 49:3–5.

Bloom, P. H. 1980. The status of the Swainson's Hawk in California, 1979. Final Report II-8.0, U.S. Bureau of Land Management and Federal Aid in Wildlife Restoration, Proj. W-54-R-12, Calif. Dept. Fish and Game, Sacramento, CA.

Briggs, C., Woodbridge, B., and Collopy, M. 2011. Correlates of survival in Swainson's Hawks breeding in northern California. J. Wildlife Mgmt. 75:1307–1314; doi.org/10.1002/jwmg.167.

Brown, L. H., and Amadon, D. 1968. Eagles, Hawks and Falcons of the World. McGraw-Hill, New York.

Dawson, W. L. 1923. The Birds of California. South Moulton Co., Los Angeles, CA.

England, A. S., Estep, J. A., and Holt, W. R. 1995. Nest-site selection and reproductive performance of urban-nesting Swainson's Hawks in the Central Valley of California. J. Raptor Res. 29:179–186.

Fitzner, R. E. 1978. Behavior ecology of Swainson's Hawk (*Buteo swainsoni*) in southeastern Washington. Ph.D. dissertation, Wash. State Univ., Pullman.

Furnas, B. J., Wright, D. H., Tennant, E. N., O'Leary, R. M., Kuehn, M. J., Bloom, P. H., and Battistone, C. L. 2022. Rapid growth of the Swainson's Hawk population in California since 2005. Ornithol. Appl. 124:1–12; doi.org/10.1093/ornithapp/duac006.

Gilmer, D. S., and Stewart, R. E. 1984. Swainson's Hawk nesting ecology in North Dakota. Condor 86:12 18; doi.org/10.2307/1367335.

Grinnell, J., and A. H. Miller. 1944. The distribution of the birds of California. Pac. Coast Avifauna 27.

Munz, P. A. 1974. A Flora of Southern California. Univ. Calif. Press, Berkeley; doi.org/10.1525/9780520338654.

Olendorff, R. R. 1978. Population status of large raptors in northeastern Colorado, 1970–1972. Raptor Res. Rep. 3:185–205.

Risebrough, R. W., Schlorff, R. W., Bloom, P. H., and Littrell, E. E. 1989. Investigations of the decline of Swainson's Hawk populations in California. J. Raptor Res. 23:63–71.

Steenhof, K. 1987. Assessing raptor reproductive success and productivity, *in* Raptor Management Techniques Manual (B. A. Giron Pendleton, B. A. Millsap, K. W. Cline, and D. M. Bird, eds.), pp. 157 170. Natl. Wildlife Fed., Washington, D.C..

Stock, C. 1930. Rancho La Brea: A record of Pleistocene life in California. Los Angeles County Mus. Nat. Hist. Sci. Ser. 1.

Templin, W. E., Phillips, S. P., Cherry, D. E., DeBortoli, M. L., et al. 1995. Land use and water use in the Antelope Valley, California. Water-Resources Investigations Report 94-4208. U.S. Geol. Surv.; https://pubs.usgs.gov/wri/1994/4208/report.pdf.

Woodbridge, B. 1991. Habitat selection by nesting Swainson's Hawks: A heirarchical approach. Master's thesis, Ore. State Univ., Corvallis.

Woodbridge, B., Finley, K. K., and Bloom, P. H. 1995. Reproductive performance, age structure, and natal dispersal of Swainson's Hawks in the Butte Valley, California. J. Raptor Res. 29:187–192.

Accepted 13 October 2022
Associate editor: Christopher W. Swarth

Land Manager's Guide to Developing an Invasive Plant Management Plan

December 2018

CAL-IPC
CALIFORNIA INVASIVE PLANT COUNCIL

ON THE COVER
Arctic National Wildlife Refuge Alaska
Credit: USFWS

SUGGESTED CITATION:

U.S. Fish and Wildlife Service and California Invasive Plant Council. 2018. *Land Manager's Guide to Developing an Invasive Plant Management Plan.* Cal-IPC Publication 2018-01. National Wildlife Refuge System, Pacific Southwest Region, Inventory and Monitoring Initiative, Sacramento, CA. California Invasive Plant Council, Berkeley, CA. Available at www.cal-ipc.org and data.gov.

This guidance document is the product of a technical advisory committee of invasive plant management experts convened by the U.S. Fish and Wildlife Service and the California Invasive Plant Council.

Name	*Title*	*Organization*
Giselle Block (Lead)	Inventory and Monitoring Specialist, Inventory and Monitoring Program	U.S. Fish and Wildlife Service, National Wildlife Refuge System Pacific Southwest Region
Doug Johnson (Co-Lead)	Executive Director	California Invasive Plant Council
Ramona Robison (Co-Lead)	(Former) Science Program Manager	California Invasive Plant Council
David Bakke	Pesticide Use Specialist / Invasive Plants Program Manager	U.S. Forest Service—Northeast Region
Elizabeth Brusati	Senior Environmental Scientist	California Department of Fish and Wildlife
Jim Dempsey	Environmental Scientist	California State Parks
Pete Frye	Vegetation Specialist	Marin County Open Space
Cindy Hall	National Integrated Pest Management Coordinator	U.S. Fish and Wildlife Service, National Wildlife Refuge System
Jon Hall	Stewardship Director	The Land Conservancy of San Luis Obispo
Tim Hyland	Environmental Scientist	California State Parks
Irina Irvine	Ocean and Coastal Resources Program Manager	National Park Service—Pacific West Region
Shannon Johnson	Senior Terrestrial Biologist	Pacific Gas and Electric
Dean Kelch	Primary Botanist	California Department of Food and Agriculture
Drew Kerr	Treatment Program Manager	Invasive Spartina Project
Rachel Kesel	Conservation Management Specialist	One Tam
James Law	Habitat Restoration Services Manager	Santa Ana Watershed Association
Allyssa Overbay	Biological Sciences Technician	U.S. Fish and Wildlife Service, National Wildlife Refuge System Pacific Southwest Region
Cindy Roessler	Senior Resource Management Specialist (Retired)	Midpeninsula Regional Open Space District
Coty Sifuentes-Winter	Senior Resource Management Specialist	Midpeninsula Regional Open Space District
Bobbi Simpson	Liaison, California Exotic Plant Management Team	National Park Service
Dustin Taylor	Integrated Pest Management Coordinator	U.S. Fish and Wildlife Service, National Wildlife Refuge System Pacific Southwest Region
Don Thomas	Integrated Pest Management Specialist	San Francisco Public Utilities Commission
Joel Trumbo	Senior Environmental Scientist	California Department of Fish and Wildlife
Valerie Vartanian	Natural Resources Manager	Naval Base Ventura County (Point Mugu), U.S. Department of Defense
Lisa Wayne	Director	San Francisco Natural Areas Program
Cheryl Wilen	Area Integrated Pest Management Advisor / U.C. Statewide Integrated Pest Management Program & Natural Resources Extension Coordinator	U.C. Cooperative Extension
Andrea Williams	Vegetation Ecologist	Marin Municipal Water District

Acknowledgements

Developing this Guide would not have been possible without the work of invasive species innovators and practitioners who continually strive to increase our success in managing invasive species and who share what they learn. In addition to the technical advisory committee listed above, several other invasive species management experts within the U.S. Fish and Wildlife Service provided important feedback during development of this Guide including Laura Eaton (Region 5), Geoffrey Grisdale (Region 8), John Klavitter (Region 9), William G. Thomas (Region 4), Karen Viste-Sparkman (Region 3), and Heather Whitlaw (Region 4). Funding for this work was provided by the U.S. Fish and Wildlife Service, National Wildlife Refuge System, Pacific Southwest Region, Inventory and Monitoring Initiative, Sacramento, California.

Contents

FIGURES

TABLES

Abbreviations

BMP	best management practice
Cal-IPC	California Invasive Plant Council
EDRR	early detection and rapid response
Guide	*Land Managers Guide to Developing an Invasive Plant Management Plan*
IPM	integrated pest management
Plan	invasive plant management plan
USDA	U.S. Department of Agriculture
USFWS	U.S. Fish and Wildlife Service

Chapter 1
Introduction

European beachgrass
Ammophila arenaria
CREDIT: USFWS

1.1 Purpose

The *Land Manager's Guide to Developing an Invasive Plant Management Plan* (Guide) is intended to help natural resource managers develop a strategic, integrative, and adaptive invasive plant management plan (Plan) (figure 1). More importantly, this guide covers the *process* of invasive plant management planning, whether you are developing a stand-alone Plan or integrating invasive plant management into other land management planning efforts such as vegetation management, fire management, species/ecosystem recovery planning, or climate change adaptation. The Guide is applicable at any scale, wherever invasive plants (terrestrial or aquatic) are a conservation concern and where resources will be expended to prevent, reduce, or eliminate them.

The Guide addresses topics common to many land management situations but also recognizes that each situation is unique given the diversity of environmental, legal, political, and other factors that can influence a site. Common constraints—such as limited staff or funds, site accessibility, spatial scale, sensitive resource concerns, and political or cultural issues—can impact where, when, and how we manage invasive plants and are addressed throughout the Guide, as applicable. This Guide is not intended to prescribe specific methods or techniques for invasive plant prevention, control, or inventory/monitoring. Furthermore, it does not address specific policies or regulations, as these can differ according to the agencies or organizations involved. Rather, it guides the process of decision-making to meet site-specific needs and conditions.

This Guide describes a step-wise process for developing and documenting an approach to managing invasive plants, and points to a wealth of freely available resources and examples. The intent is to help land managers develop effective Plans, even when management resources are limited and variable. Information in this Guide integrates and builds upon the best available information, including published and unpublished literature, decision-support tools, expert opinion, and past invasive plant management or integrated pest management (IPM) planning guides (such as Olkowski and Olkowski 1983; Tu and Meyers-Rice 2002; U.S. Fish and Wildlife Service [USFWS] 2004; IUCN 2018).

This Guide helps land managers address these key questions:

- Why is invasive plant management needed?
- What are the desired outcomes—management objectives?
- Which invasive plant species should be a management focus and where?
- What is the status (distribution, abundance) of invasive plants?
- What management strategies should be implemented? Who will implement? Where and when will they be implemented? Cost?
- How will the effectiveness of strategies be evaluated?
- What is the process for learning and adapting management strategies over time?

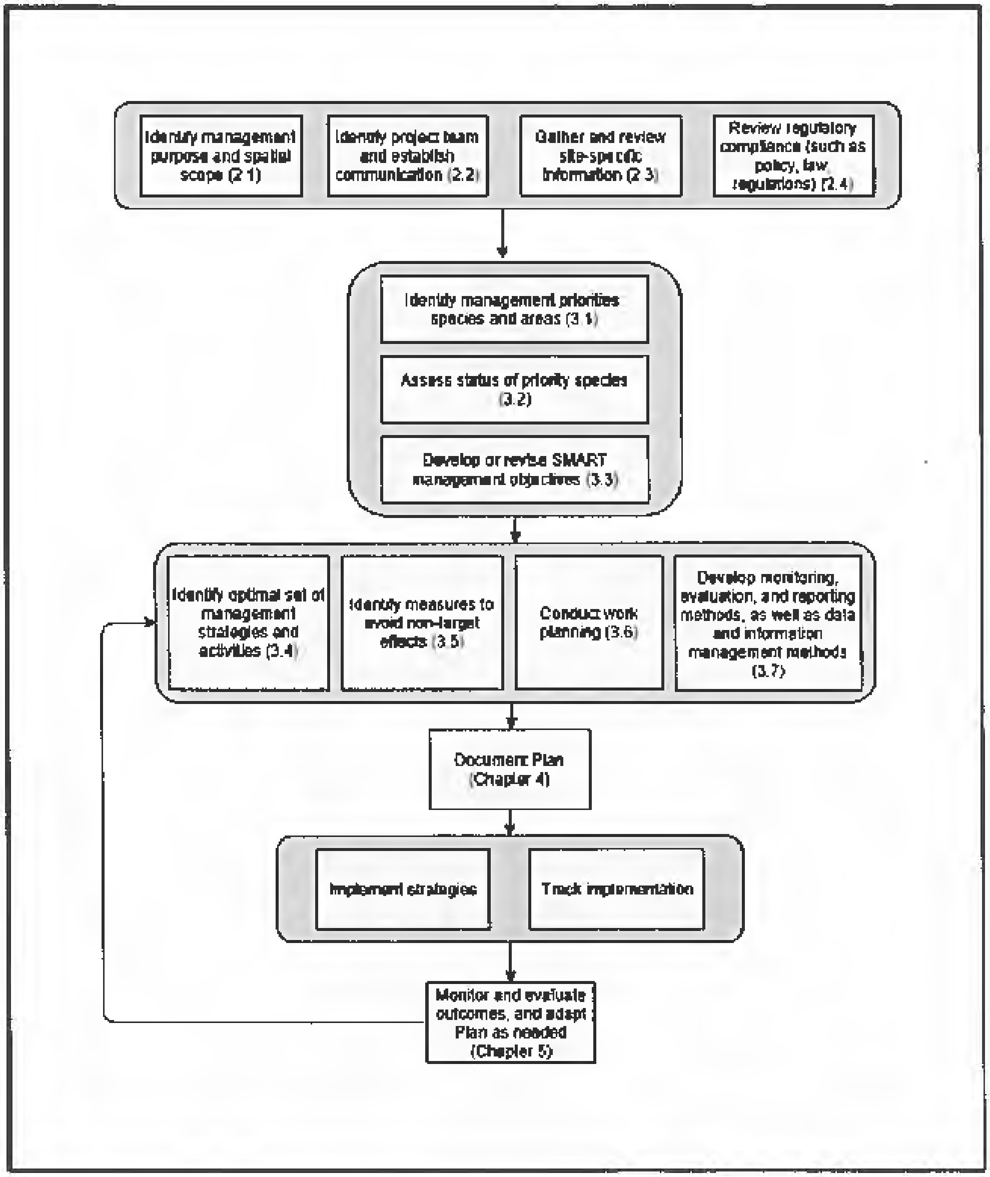


Figure 1. Strategic and adaptive invasive plant management cycle. Numbers in parentheses refer to sections of the Guide where information on that topic is located.

1.2 How to Use This Guide

This Guide is designed to take you through the major phases of developing a Plan (table 1): preparing to write the Plan (chapter 2), analyzing the situation and designing a management strategy (chapter 3), writing the Plan (chapter 4), and evaluating outcomes and adapting management strategies (chapter 5). A glossary follows chapter 5. Appendix A provides a list of useful online resources, appendix B provides plan examples, and appendix C provides a structured checklist of questions which serve as a Plan template.

Table 1. Steps for developing a strategic, integrative, and adaptive invasive plant management plan.

Step	*Description*	*Guide location*
Identify management purpose and spatial scope	The management purpose identifies the reasons why a Plan is needed, its intended audience(s), and how it will be used. The spatial scope identifies the geographic area where management activities prescribed by the Plan will occur and sets the stage for what types of information should be gathered to inform the Plan.	Section 2.1
Identify project team and establish communications	The project team is the larger group of people involved in your invasive plant management program, including land managers, stakeholders, researchers, governing boards, and other key players. The project team often includes a smaller core team who coordinates the planning effort and is ultimately responsible for developing and implementing the Plan. Identify the means for communication during the planning process, both within and outside your organization.	Section 2.2
Gather site-specific information	Gather basic information (plans, reports, data) for your sites, including organizational vision, conservation priorities, management goals and objectives, invasive plant issues, and management history. Identify gaps in information that need to be filled.	Section 2.3
Review regulatory compliance	Gather and review organizational policies and legislation that apply to invasive plant management planning or actions within your scope.	Section 2.4
Identify management priorities: species and areas	Select and document plant species that will be the focus of the Plan. A Plan may focus on a single species or address multiple species. If multiple species are being considered, prioritize which species are most critical to address. Define management areas within the Plan scope and prioritize where to focus management efforts.	Section 3.1
Evaluate the status of priority invasive plants in priority areas	Assess invasive plant abundance, distribution, pattern of spread, and spatial relationships with abiotic and biotic features in the environment.	Section 3.2
Develop SMART (specific, measurable, achievable, results-oriented, time-bound) management objectives	Develop statements that detail what success would look like as a result of your invasive plant management program.	Section 3.3
Develop optimal set of management strategies	Develop a suite of strategies to meet your SMART invasive plant management objectives using the best available information.	Section 3.4
Identify measures to avoid non-target effects	Use the best available information to develop measures to prevent, avoid, or mitigate any potential negative effects on humans, natural or cultural resources, or infrastructure as a result of invasive plant management activities.	Section 3.5
Work Planning	Describe who, what, where, and when invasive plant management activities will occur; this step guides on-the-ground implementation.	Section 3.6
Develop inventory, monitoring, and evaluation methods	Identify methods to track implementation of management activities, monitor plant community status and trends, and assess and report on progress in attaining invasive plant management objectives (or thresholds for management action).	Section 3.7
Develop data and information management methods	Develop data standards and structures for ensuring the data are easily accessed, understood, and utilized to their fullest potential.	Section 3.7
Write your Plan	Summarize your planning process and results of your analysis.	Chapter 4
Adapt your Plan (as needed)	After implementation, monitoring, and evaluation, revise your Plan at a regular interval to incorporate new information and other changes in approach.	Chapter 5

Terminology Matters

The language and terminology used to describe invasive species varies among countries, agencies, organizations, professionals, and members of the public. Terms like ***alien, non-native, invasive, pest*** and ***weed*** are often used interchangeably in scientific literature, confusing readers and even muddling the science (Lockwood et al. 2013). In this Guide, ***non-native species*** are defined as species found outside of their natural range, and ***invasive species*** are non-native organisms whose introduction causes or is likely to cause economic or environmental harm or harm to human, animal, or plant health (Executive Order No. 13751, 2016). It is important to emphasize that not all non-native species are invasive. Likewise, there may be native species that cause harm to ecosystems or human health (often referred to as ***native nuisance species***). Throughout this Guide we use the term ***invasive*** but recognize different terms may be preferred by different users and that planning efforts may also include native nuisance species.

alien: with respect to a particular ecosystem, an organism—including its seeds, eggs, spores, or other biological material capable of propagating that species—that occurs outside of its natural range (Executive Order 13751, 2016). Synonymous with ***non-native, nonindigenous*** and ***exotic***.

aquatic nuisance species: a nonindigenous species that threatens the diversity or abundance of native species or the ecological stability of infested waters or commercial, agricultural, aquacultural, or recreational activities dependent on such waters (Nonindigenous Aquatic Nuisance Prevention and Control Act 1990).

noxious weed: any plant or plant product that can directly or indirectly injure or cause damage to crops (including nursery stock or plant products), livestock, poultry, or other interests of agriculture, irrigation, navigation, the natural resources of the United States, the public health, or the environment (Public Law 106-224).

pest: organisms that damage or interfere with desirable plants in our fields and orchards, landscapes, or wildlands, or that damage homes or other structures. Pests also include organisms that impact human or animal health (University of California Statewide IPM Program 2018).

weed: a plant that causes economic losses or ecological damage, creates health problems for humans or animals, or is undesirable where it is growing (Weed Society Science of America 2016).

1.3 Invasive Plant Management: An Overview

There are many reasons to manage invasive plants in natural areas. Most often cited are the threat invasive plants pose to native biodiversity and the alterations to natural processes. Many studies have demonstrated how invasive plants can alter ecosystem processes, structure, and composition, as well as the genetic makeup of native species populations through hybridization (Bossard et al. 2000; DiTomaso et al. 2013; Foxcroft et al. 2017; Hobbs and Humphries 1995; Lockwood et al. 2013). Invasive plants can also negatively impact infrastructure or other parts of the built environment (such as damaging irrigation systems) or pose harm to humans (such as increasing wildfire intensity or frequency). Finally, invasive plant encroachment may alter aesthetics or interfere with a recreational or cultural value of a place or property.

An important aspect of developing an invasive plant management plan is to make clear connections between the rationale(s) for managing invasive plants and your organization's mission, resources of conservation concern, and management goals. Such connections help land managers focus management efforts (set priorities), help stakeholders and others understand the motivation and need fur management, and can ultimately increase management support. After addressing *why* your organization must manage invasive plants, the bulk of the planning process is focused on *how* your organization will manage those plants. The foundational principles for how to manage invasive plants is based on IPM, which is a decision-making process that integrates management goals, consensus building, pest biology, monitoring, environmental factors, and best-available technologies to achieve desired outcomes while minimizing unwanted effects.

Why Develop a Plan?

Successful invasive plant management is a lot more complicated than simply killing weeds—it requires a strategic and adaptive approach that is well-documented (figure 1). As Ben Franklin said, "if you fail to plan you are planning to fail." The planning process itself provides the opportunity for focused analysis, prioritization, and being clear about what you hope to achieve – your objectives. A well-crafted Plan provides guidance for a consistent management approach over time with parameters for adapting actions as environmental conditions or available resources change. It documents where you are now, where you would like to be, and how best to get there.

Almost all land managers can point to shortages of funding and resources as barriers to successful invasive plant management. A well-crafted Plan can help address these problems by identifying and documenting priorities for action in the face of limited and variable resources. A Plan can also help address other common barriers to successful invasive plant management, such as:

- **Lack of understanding about the impact of invasive plants.** The degree to which invasive plants harm priority conservation targets and impede the attainment of site goals may not be well-understood. This lack of understanding—especially among leadership within an organization or by important stakeholders—can lead to a lack of support and resources. The planning process itself provides a platform for building collective understanding, support, and consensus among management staff, leadership, partners, landowners, and local communities. Without consensus and support, a Plan simply becomes irrelevant.
- **Lack of prevention and early detection and rapid response (EDRR).** Despite the higher economic and ecological returns per unit effort they provide, prevention and EDRR are often overshadowed by already abundant and widespread invasive plant issues. Although there may exist a need to manage existing invasive plant infestations, placing little or no emphasis on preventing new invasions or further spreading can lead to economic and ecological harm (Cusack et al. 2009). The challenge is to balance managing well-established invasive plant infestations, preventing new infestations, and responding to new infestations before they become widespread. Plans should highlight the need for prevention and EDRR and detail exactly how these activities will actually be carried out.

- **Lack of inventory and monitoring of invasive plants.** Inventory and monitoring are essential to successful invasive plant management (DiTomaso 2000; Olkowski and Olkowski 1983; Stohlgren and Schnase 2006), but in the face of limited resources, managers often plan and implement their management strategies with little to no data about the status of the infestations they intend to manage or whether their strategies are actually working. This paradoxical dilemma is difficult to overcome, as many land managers feel the need to use limited resources on controlling invasive plants rather than on conducting inventory and monitoring. Without inventory and monitoring, we lack evidence that our strategies are creating the desired result, have no basis for learning and adapting, and leave no legacy of knowledge for those who come after us (or for communicating with the public), and therefore risk repeating failures.
- **Lack of an integrative approach.** A single-strategy approach, such as only using a chemical control method for long periods, can lead to species resistance, unintended non-target effects, and ultimately failure over the long term. Ideally, employing multiple management strategies that work together is more successful over the long-term than any one single strategy.
- **Lack of SMART (i.e., specific, measurable, achievable, results-oriented, time-bound) invasive plant management objectives and/a built-in process for evaluation and feedback.** Without SMART objectives describing the expected result(s) of invasive plant management and a process for evaluation and feedback, managers lack a basis for evaluating progress, testing assumptions, learning, and adapting. We risk repeating practices of the past without regard to whether implemented strategies are working (or not) at different spatial and temporal scales.
- **Action is more reactive than proactive.** Ideally, the establishment of highly invasive species is wholly prevented, detected, or eradicated in the early phases of invasion. An introduced species can remain at low levels for a long period of time (such as years) before rapidly expanding. This is known as the *lag phase*. Whether or when a species leaves the lag phase and rapidly expands can depend on several factors including (1) development of genotypes that allow the species to spread, (2) changed environmental conditions that promote rapid population spread, or (3) continuous expansion of the species population that goes unnoticed until it becomes widespread (Hobbs and Humphries 1995). It is more cost-effective to remove or prevent establishment of invasive species before they become widespread and abundant—in other words, taking a more proactive than reactive approach.

Principles of Integrated Pest Management

The concept of IPM was first articulated by University of California entomologists in the 1950s, and in 1972, the concept of IPM became part of national policy with the establishment of an interagency IPM Coordinating Committee. While historically focused on insects and disease-causing organisms affecting agriculture, IPM now applies to all pest taxa and non-crop situations such as invasive plants in natural resource conservation areas.

The term *integrated* means to apply a combination of management techniques that work better together than separately. Using an integrated management approach increases the likelihood of success and reduces the likelihood that a pest will become immune (i.e., develop resistance) to a management technique, particularly in the case of herbicides.

Integrated Pest Management (IPM)

"A science-based decision-making process that incorporates management goals, consensus building, pest biology, monitoring, environmental factors, and selection of the best available technology to achieve desired outcomes while minimizing effects to non-target species and the environment and preventing unacceptable levels of pest damage" (USFWS 2010).

While the concept and policies surrounding IPM have evolved over time and vary across organizations and agencies, contemporary descriptions have common elements (for example, USFWS 2004; DiSalvo and Parson 2011; Flint and Gouveia 2014; UC-IPM 2018) such as:

- Know your resource (site description: ecosystems and landcover, infrastructure, conservation goals, etc.).
- Know your pest; identify priority pest species and understand their ecology and harm (or potential harm).
- Assess the status of pest populations.
- Prevent pest problems.
- Use a combination of techniques to control pest populations.
- Develop guidelines or thresholds for management action.
- Describe your expected management outcomes or results (objectives).
- Build consensus and regularly communicate with those who may be affected by your pest management program or who can contribute expertise.
- Monitor management outcomes, learn, and adapt management.

This Guide is designed to help you consider each of these elements as you develop your Plan.

This page intentionally left blank.

Chapter 2

Preparing to Write a Plan

This chapter is focused on laying the foundation of your Plan—its spatial scope, who should be involved in its development, understanding which invasive plant species occur (or could occur in the future), conservation focus, invasive plant management history, and regulatory considerations.

Purple loosestrife
Lythrum salicaria
CREDIT: ©2009 Barry Rice

2.1 Identify Plan Purpose and Spatial Scope

An essential first step in the planning process is to identify the Plan's purpose and its spatial scope. The Plan should present a compelling case for why invasive plant management is needed and how it is impeding your ability to achieve your organization's mission and conservation goals. The spatial scope identifies the broad geographic area where invasive plant management activities will occur and sets the stage for what types of information should be gathered to inform the plan (section 2.3), what laws or policies will govern invasive plant management activities (section 2.4), and who should be involved in strategic analysis for the Plan and the types of communication needed (section 2.2). The Plan may focus on a single, geographically distinct site such as a park, refuge, watershed, or forest, or a collection of sites within a large landscape. The scope could also be more thematic in nature, such as a particular ecosystem within a landscape.

2.2 Identify Project Team and Establish Communication

The project team is the group of people who are involved in developing a Plan. The project team can be a small group of people who do most of the work (core team), decision-maker(s), stakeholders (such as the public or adjacent landowners), invasive species experts, and others who will implement the Plan or who have a vested interest in conservation activities or outcomes at your site. It's worth carefully considering your project team's composition and, if needed, pushing your organization to recognize the importance of this step. The ultimate utility of a Plan can depend heavily on who is involved in its development. Project team members will likely include representatives from the implementing organization but may include others outside the organization. Being outside the organization might mean these individuals play different roles on the team, but they may still be essential for successfully implementing your invasive plant management program.

The core planning team—those who will be closely involved with moving the process forward—should form at the start of Plan development and then promptly identify everyone who should be involved within the broader project team and revisit the Plan scope. The composition of the project team may change as you move through Plan implementation, although it is usually helpful to maintain continuity. Once you have identified the project team, identify and communicate roles. Begin communicating with your team early in the planning process to help everyone understand the planning process, their roles, and how information will be shared.

It is critical that communication continues throughout the planning process to help build consensus, ensure the time and resources you spend on planning are not wasted, and the team is connected and supportive of the final product. While some Plans will be primarily internal, for others the external use will be just as important. Near urban areas, or in high-use areas, land management decisions may be politically charged, and a great deal of public review and participation may be needed to develop a Plan that reflects the interests of all stakeholders. Political leaders may need help in understanding the factors that go into developing a Plan, and a communication strategy for outreach to the broader community may be needed. Beyond their perspective as stakeholders, community members can also be a great resource for ideas and assistance. General tips for improving communication during the planning process are listed below:

- Design the Plan to suit the needs of the target audience(s).
- Make the Plan readable; minimize jargon and technical details that are not explained.
- Communicate early and often with all levels of management in your organization on the need for the Plan.
- Anticipate potential internal and external concerns; develop a communications approach to address these concerns.
- Design an ongoing process for building consensus between technical experts, decision-makers, and stakeholders.

Who Should Be on the Project Team?

- People who will develop the Plan
- People who will implement the Plan
- Key decision-makers
- Partners or other important stakeholders
- Technical advisors

2.3 Gather and Review Site-Specific Information

Gathering and reviewing information relevant to the Plan scope will provide a foundation for developing your Plan and increase how efficiently it is developed. Information should be gathered to answer questions such as:

- What is the focus of conservation at the site, and what are the associated conservation goals?
- What are current and potential invasive plant species that prevent attainment of conservation goals, and how do they prevent attainment of goals?
- What is the current distribution and trend of each invasive plant species?
- What strategies have been employed to manage species currently and previously, and how effective have they been?
- From whom is support needed for Plan development and implementation? Where might obstacles and resistance to invasive plant management support be likely to materialize?

Table 2 lists information that would typically be gathered and used to inform development of a Plan.

Table 2. Common types of information to support invasive plant management planning.

Item	*Source*	*Rationale*
Personal knowledge or expertise	Interviews with leadership, invasive plant program staff, adjacent land owners, and local (or regional) invasive species experts.	Increases understanding about current invasive plant issues, future potential invasive plant issues (early detection), management history, management effectiveness, and potential barriers to successful management.
Site surveys	Tours of management areas with staff familiar with the areas and history of invasive plant management efforts.	Increases understanding about conservation targets, sensitive species issues, invasive plant threats, stress, status, and trends; informs invasive plant management strategies.
Management plans and records	Site-specific or surrounding landscape conservation plans; past invasive plant management plans, reports, or management records; and stakeholder lists.	Identifies conservation targets, goals, or existing invasive plant management objectives within the spatial scope or in the surrounding landscape. Increases understanding about the status and trends of invasive plant threats and the harm they cause as well as understanding of potential management strategies. May identify restrictions on management methods.
Spatially referenced information	Maps or spatial data: site boundaries, management units, landcover, vegetation communities, hydrology, roads/trails, infrastructure, cultural resources, sensitive species locations, and invasive species distribution.	Increases understanding about the status and trends of invasive plants, relationships with other environmental features (biotic and abiotic). Informs priorities for invasive plant management (what species and where) and strategy development.
Invasive plant lists	Site-specific invasive plant lists, management plans, natural resource reports, and outside databases (from state invasive species councils, natural heritage programs, NatureServe Explorer, EDDMapS, herbaria, etc.).	Informs what species should be the focus of management. If there are multiple plant lists for a single site, compile into one list and standardize taxonomy (such as to the International Integrated Taxonomic Information System standard, available at *www.itis.gov*).
Early detection plant lists	Web-based species occurrence databases like EDDMapS and CalWeedMapper and information from early detection networks, county agricultural extension agents, and weed management areas.	Informs what species should be the focus of early detection efforts.
Non-native plant invasiveness rankings and legal status	Invasive species risk assessments conducted by larger landscape agencies or organizations, such as invasive plant councils; includes federal and state noxious weed lists.	Informs prioritization of non-native plants species for management.

2.4 Review Regulatory Compliance

Compliance with regulations (acts, laws, policies, regulations, permits, certifications, etc.) is always a component of developing and implementing invasive plant management programs and may ultimately influence the types, location, and timing of invasive plant management activities at your site. While regulatory compliance is an important component of planning, it is not a focus of this Guide, as requirements can vary geographically (such as by state) and across private and public organizations.

We recommend consulting within your organization to gain a clear understanding of the policies, laws, permits, required training, and other regulatory compliance applicable to invasive plant management activities within the Plan's scope. If your organization has limited knowledge or experience with regulatory compliance issues, reach out to similar organizations in your area who may have more expertise. In the case of federal or state agencies or for Plans that encompass public lands, be sure to review your agency's regulatory framework. It is always useful to reach out to invasive species experts, within or outside your organization, to better understand the regulatory framework that will influence invasive plant management planning and implementation.

Chapter 3

Analyzing the Situation and Designing a Management Strategy

This chapter guides you through analysis of information gathered (chapter 2) to identify your priorities, define what you want to achieve (objectives), and design a management strategy. *Strategies* here refers to a collection of activities that work together to achieve a particular outcome—the objective(s). Ultimately, the level of detail provided about strategies and associated activities should be tailored to the situation and intended users of the Plan. For example, if the Plan is intended to direct on-the-ground management activities, then a high level of detail is warranted.

Section 3.1 covers identifying priority species and areas, and section 3.2 covers evaluating the status (abundance and distribution) of priority species in priority areas. Setting invasive plant management objectives and establishing strategies are discussed in sections 3.3 and 3.4, respectively. Section 3.5 covers how to avoid non-target effects, or the unintended impacts of carrying out invasive species management. The final two sections—section 3.6 and 3.7—discuss how you will implement your plan. Section 3.6 addresses work planning, a critical step in which you will document what needs to get done, where, and when, as well as how much it will likely cost; work planning is an essential step in ensuring that your Plan is implemented effectively and consistently over time. Section 3.7 discusses establishing inventory, monitoring, and evaluation procedures.

Water hyacinth
Eichhornia crassipes
CREDIT: USFWS

3.1 Identify Management Priorities: Species and Areas

One key aspect of any invasive plant management planning process is prioritization: selecting which species to work on, where, and when. Ideally, prioritization is conducted before significant resources are invested in invasive plant inventories, early detection, or management actions. Managing for all non-native species everywhere within a site is impractical. Natural resource managers are often constrained by funding, available resources, time, and personnel, and several have developed credible ways to make decisions about which invasive plants to focus on and where (such as Hiebert and Stubbendieck 1993; Randall 2000; Skurka Darin et al. 2011; USFWS and Utah State University 2018).

The prioritization process (shown schematically in figure 2 below) is an opportunity to develop or refine the focus of invasive plant management activities, ensuring resources are dedicated where they are most needed. Ideally, decisions about what invasive species to focus on and where should be transparent, repeatable, and defensible (Hiebert and Stubbendieck 1993; Randall et al. 2008; Warner et al. 2003). This approach helps build consensus and support, fosters continuity in management over time as people or conditions change, and builds in management flexibility as funding and staff levels change. Prioritization does not mean that a species or area identified as "low priority" should never be addressed; even low priority species and areas may be addressed at some point in the future. Alternatively, it is worth evaluating if there are invasive plant species currently under management that shouldn't be. It's important to remember prioritization is intended to inform decision-making rather than to make decisions directly. Prioritization results should be discussed among your project management team to make final decisions.

While most teams find both species and area prioritizations useful, there may be cases where there are few (such as fewer than five) invasive plants of concern within or adjacent to the Plan's spatial scope, negating the need for species prioritization. Here, the decision process may shift to where invasive plant management should be focused, especially when the scope encompasses thousands or millions of acres.

The following sections describe the general process and tools for prioritization. Also, see appendix A for tools and resources for prioritization and appendix B for links to reports or plans that contain invasive plant prioritization examples.

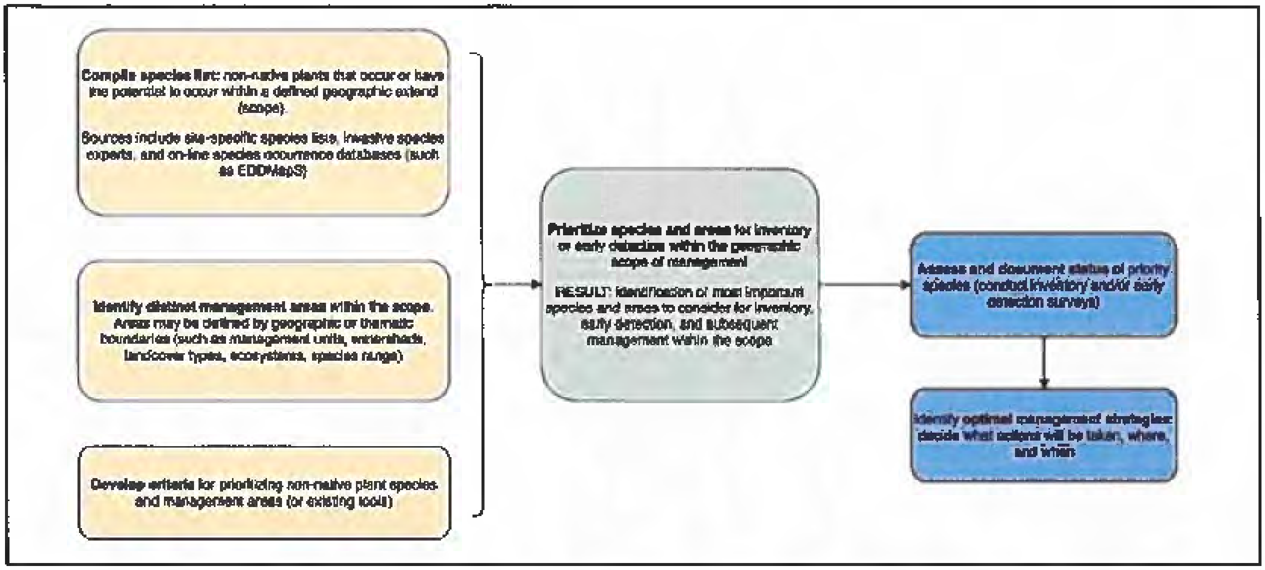


Figure 2. Generalized work flow for prioritizing species and areas for management. Initial prioritization informs what species and where inventories or early detection surveys should be focused and more generally where management efforts should focus. Subsequent inventory or early detection surveys provide details to inform and direct on-the-ground management action (what to do, where/what populations, and when). Survey data also provide a basis for evaluating progress over time.

3.1.1 Identify and Prioritize Plant Species

The first step in species prioritization is to compile a list of non-native plant species known to occur within the Plan spatial scope as well as species with the potential to occur in the future. Ideally, a list of current and potential species is compiled from available sources and scientific names are standardized to your preferred taxonomic standard (such as the International Taxonomic Information System). Once compiled, the lists can then be prioritized by the project team using one or more criteria (table 3).

Many larger landscape organizations such as the U.S. Department of Agriculture (USDA) and state invasive plant councils have assessed invasiveness or "noxiousness" of non-native plant species to wildlands across large landscapes of the United States (see table 4 for examples). These assessments are based on risk assessment criteria such as the NatureServe *Invasive Species Assessment Protocol* (Morse et al. 2004), and they often rely on scientific literature and expert knowledge to provide a comprehensive review of species ecology, biology, distribution, and impacts on the environment. While these larger landscape lists can be a useful tool in identifying management priorities, when used alone, they may not provide enough information to identify local scale priorities. For example, when many of the species on your list are found on one of these larger landscape lists, management priorities may be less apparent. In such cases, it may be useful to apply additional criteria (table 3) or use a tool (table 5) to help identify site-specific priorities. A more structured approach can help teams come to consensus on which species should be a focus of management as well as provide a legacy of information about how decisions were made. An example of a species prioritization exercise from the Klamath National Wildlife Refuge Complex is provided in figure 3.

3.1.2 Identify and Prioritize Management Areas

A first step in prioritizing areas for management is to define the areas of your Plan's spatial scope that are under management consideration. Over the long term, the intent may be to manage invasive plants across all areas within the Plan's spatial scope, but when resources are limited, area priorities help inform where to use those resources. Areas should have clear boundaries defined by one or a combination of features such as jurisdictional management boundaries, ecosystem types, vegetation communities, sensitive species populations/habitat, watersheds/hydrology, soils, or topography. Several criteria can be used to help decide which areas within the Plan's spatial scope are a priority for managing invasive plants. These include the current level of infestation, risk of invasion, and importance to high value conservation resources; table 6 provides a list of criteria often used to prioritize areas, and table 5 provides a list of prioritization tools. An example of an area prioritization from the National Park Service Golden Gate Recreation Area is provided in figure 4.

Table 3. Criteria commonly used to prioritize species for invasive plant management.

Category	*Criteria*
Larger Landscape Invasiveness	The degree to which a species is likely to cause harm to wildlands or overall biodiversity. Invasiveness rankings have been developed for larger landscapes and are based on expert opinion and comprehensive review of the scientific literature (see table 5).
Status and Habitat Suitability	Characteristics of the species within the Plan's spatial scope. Includes criteria such as presence or proximity, abundance, distribution, and habitat availability/potential to spread.
Ecological Impacts	The severity of current or potential impacts the plant causes (or could cause) on conservation targets within the Plan's spatial scope.
Difficulty of Control	The difficulty of managing the species within the Plan's spatial scope. Includes criteria such as cost, time, and technical difficulty.
Larger Landscape Importance	The degree to which the species is a priority for management on adjacent lands or in the larger landscape.
Other	The degree to which a species is important for management because of political, public, cultural, or other reasons (defined by the user).

Table 4. Examples of invasive plant ranking systems.

System title	*Species ranking criteria*	*Web link*
Alaska Invasiveness Ranking System	Preliminary climate screening to identify species that could invade environments found in Alaska or areas with similar climate; includes ecological impact, biology, management difficulty, and distribution.	*http://accs.uaa.alaska.edu/invasive-species/non-native-plant-species-list*
California Invasive Plant Inventory	Species ecological impact, ecosystems or communities invaded, invasive potential, documentation level, and distribution.	*http://www.cal-ipc.org/plants/inventory/*
Federal and State Noxious Weed Lists	Criteria vary across states.	*https://plants.usda.gov/java/noxComposite*
Invasive Non-Native Plants That Threaten Wildlands in Arizona	Species ecological impacts, invasiveness, ecological amplitude, and distribution.	*http://www.swvma.org/invasive-non-native-plants-that-threaten-wildlands-in-arizona/*
Hawaii Weed Risk Assessment	Species ecological impact, ecosystems or communities invaded, invasive potential, documentation level, and distribution.	*https://sites.google.com/site/weedriskassessment/home*
NatureServe I-ranks	Species ecological impact, biology, abundance, management difficulty, non-target management impacts, diversity of habitats or ecological systems invaded, and distribution.	*http://explorer.natureserve.org/servlet/NatureServe?init=Species*
New York State Ranking System for Evaluating Non-Native Plant Species for Invasiveness	Species ecological impact, biology, abundance, management difficulty, and distribution.	*https://www.conservationgateway.org/Documents/New-York-State-Invasive-Plant-Ranking-System.doc*
Virginia Invasive Plant Ranking System	Species ecological impact, abundance, biology, management difficulty, and distribution.	*www.dcr.virginia.gov/natural-heritage/document/nh-invasive-plant-list-2014.pdf*

Table 5. Examples of tools for prioritizing invasive plant species and areas for management, organized from low to high levels of technical expertise required.

Tool	*Prioritization focus*	*Level of expertise required*	*Description*
Invasive Plant Inventory and Early Detection Prioritization Tool (IPIEDPT)	Species and Areas	Low	The IPIEDPT is a Microsoft Access tool that integrates larger landscape invasive plant rankings and local knowledge to generate a prioritized list of species and areas for inventory and early detection, and ultimately management. Species criteria include larger landscape invasiveness rankings, impacts (known or probable), proximity, potential for spread, and abundance/distribution. Area criteria include ecological integrity (health), level of infestation, density of vector pathways, frequency and intensity of vector events, and disturbance. Source: USFWS and Utah State University (2018). Web link: *https://catalog.data.gov/dataset/an-invasive-plant-inventory-and-early-detection-prioritization-tool*
Spreadsheet	Species and Areas	Low	Prioritization of species or areas can be done with an Excel spreadsheet. User defines criteria and scoring for species or area rankings.
CalWeedMapper*	Species and Areas	Low	Provides statewide (California) distribution data (via *calflora.org*) for invasive plants and generates a management opportunities report for user-defined areas (e.g., a National Forest, National Wildlife Refuge, ecoregion, or a county). Results from CalWeedMapper should be combined with local knowledge to set site-specific priorities. Web link: *https://calweedmapper.cal-ipc.org/*
Weed Heuristics: Invasive Population Prioritization for Eradication Tool (WHIPPET)*	Species and Areas	Moderate	WHIPPET prioritizes spatially referenced (mapped) invasive plant populations for eradication based on potential impact, potential spread, feasibility of control, and location (outlier status, proximity to vector pathways, and accessibility). Source: Darin 2008; Skurka Darin et al. 2011. Web link: *https://whippet.cal-ipc.org/pages/view/guide*
ArcGIS	Species and Areas	High	Spatial data such as invasive plant locations and environmental features (such as roads, trails, hydrology, soils, topography, ecosystem/communities, and sensitive resource locations) are overlaid and analyzed (user defined attributes) to identify priority areas and/or species (if spatial data are available) for management. Example area prioritization: National Park Service's early detection protocol (Williams et al. 2009), available at *www.sfnps.org/download_product/1256/0*

Tool	*Prioritization focus*	*Level of expertise required*	*Description*
NatureServe Invasive Species Assessment Protocol	Species	High	The protocol is a multi-criteria tool for assessing, categorizing, and listing non-native invasive vascular plants according to their impact on native species and natural biodiversity in a large geographical area such as a nation, state, province, or ecological region. The tool has typically been used to develop larger landscape invasive plant rankings but can be adapted and used at a local scale. Requires in-depth knowledge about plant ecology and impacts or an in-depth literature search. Web link: *http://explorer.natureserve.org/servlet/NatureServe?init=Species*
Alien Plant Ranking System	Species	High	The system guides users through 25 questions in three sections relating to individual species: (1) current level of impact, (2) potential of a species to become a problem, and (3) feasibility of control. The sections include questions about the distribution and abundance of species, the number of seeds they produce, and their dispersal capabilities. There are also questions about whether a species is known to seriously impact other sites. The tool has typically been used to develop larger landscape invasive plant rankings (such as for Alaska) but can be adapted and used at a local scale. Requires in-depth knowledge about plant ecology and impacts or an in-depth literature search. Source: Hiebert and Stubbendieck (1993). Web link: *http://hear.org/articles/cip_winter2002n01_prioritizing_weeds.pdf*

*Note: WHIPPET and CalWeedMapper are specific to California, but their algorithms may be useful for others. Both were designed and built with funding from the USDA Forest Service, State & Private Forestry.

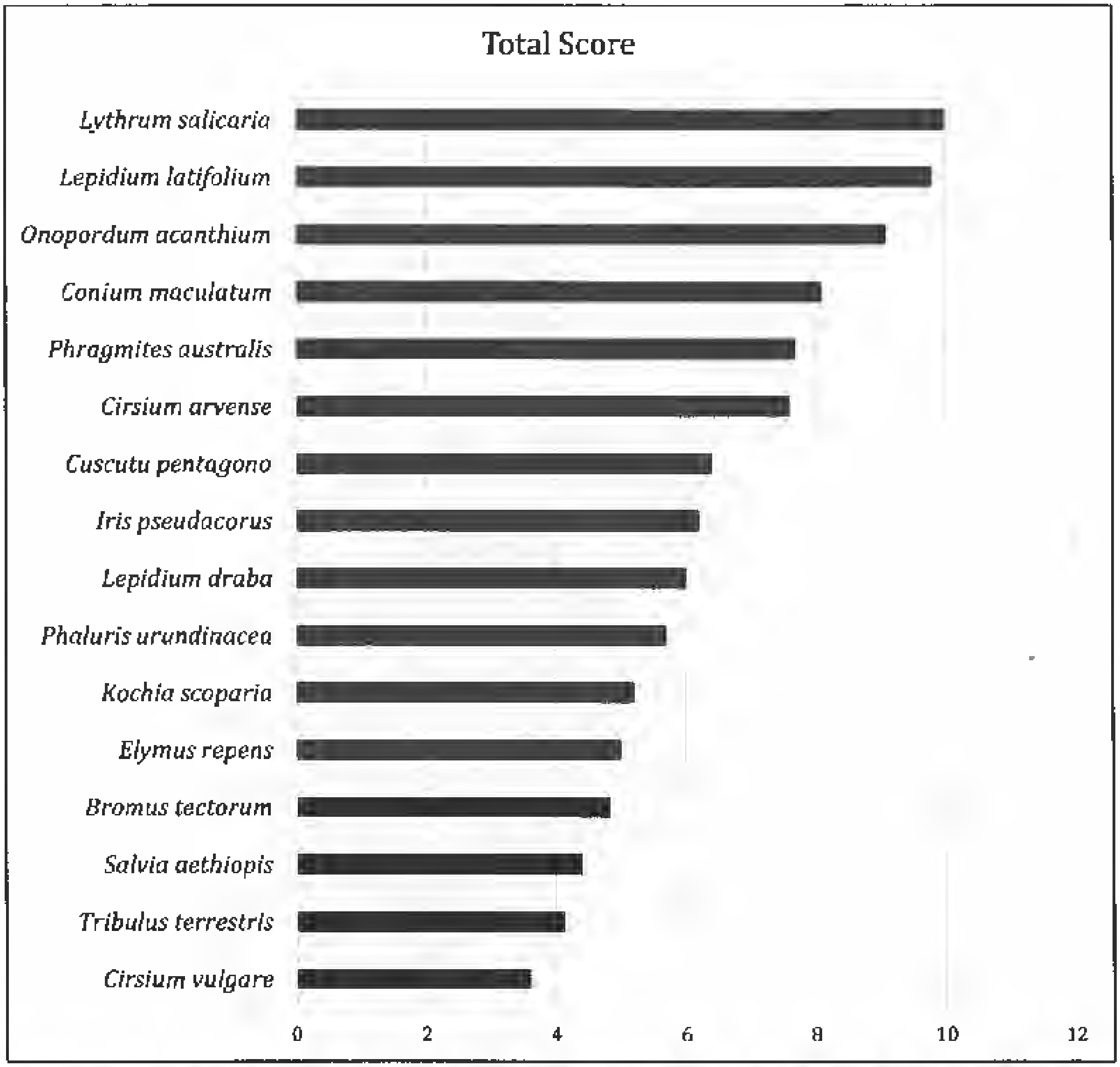


Figure 3. Invasive plant species prioritization results for Lower Klamath and Tule Lake National Wildlife Refuges: species present on-refuge (USFWS in prep.). The larger the total score, the higher the priority for management. Species prioritized using the Invasive Plant Inventory and Early Detection Prioritization Tool (IPIEDPT) (USFWS and Utah State University 2018).

Table 6. Criteria commonly used to prioritize areas for invasive plant management.

Category	*Criteria*
Importance to Conservation Targets	The importance of the area to natural resources of priority conservation concern (conservation targets) as it relates to the presence or proximity of a natural, cultural, or other important resource. Areas important to resources of conservation concern are often a high priority for detecting and removing invasive plants. These are often species, species alliances/guilds/communities, or ecosystems but can include other resources of concern, such as cultural resources.
Integrity or "Intactness" of Resources	The degree to which an area is believed to be healthy, intact, or unimpaired, with major ecological (or cultural) attributes functioning within the bounds of natural disturbance regimes. For example, ecosystem structure and processes are intact and function within their natural ranges of variation. Areas with relatively high integrity often have high conservation value and are a priority for preventing or reducing anthropogenic threats such as introduction of invasive plants.
Innate Resistance to Invasion	The innate capacity of an ecosystem (or other system) to resist establishment and spread of invasive plant species. Environmental factors that can influence innate resistance include resident native plant diversity, density of native vegetative cover, abiotic conditions such as nutrient levels, soil or water quality, and natural disturbance regimes such as flooding and wildfire.
Risk of Invasion: Invasion Pathways and Vectors	Invasion pathways and vectors provide the means for invasive plant transport from one location to another. Here, *pathways* are transportation pathways such as roads, trails, levees, waterways, etc. *Vectors* are the vehicles for transmitting or carrying invasive plant propagules along pathways, specifically human based vectors such as hikers, cars, boats, or machinery. Criteria for assessing risk of spread from pathways and vectors include assessing the density of vector pathways (both terrestrial and aquatic) and the types, frequency, and intensity of vector events—opportunities for vectors to transmit invasive plants (such as from high recreation use or frequent management activity). Areas where terrestrial pathways are widely distributed and occur at high densities are at greater risk for invasion. Areas that experience freqnent vector events (such as recreational areas) are also at risk.
Risk of Invasion: Anthropogenic Disturbance	*Disturbance* facilitates invasive plant invasions and can be described as a "relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate availability, or the physical environment" (Lockwood et al. 2013; White and Pickett 1985). Here, we are focused on anthropogenic disturbances such as restoration/enhancement activities, regular maintenance activities, resource extraction, and toxic spills. Consider the intensity, duration, and frequency of human-caused disturbance events. Areas that are exposed to intense, frequent, or long-duration disturbance events are at high risk for invasion.
Infestation Level	This category considers the richness and abundance of invasive plant species within an area. Areas considered "clean" of invasive plants are often a higher priority than areas already heavily infested.
Investments	Degree of previous investment in invasive plant removal efforts.

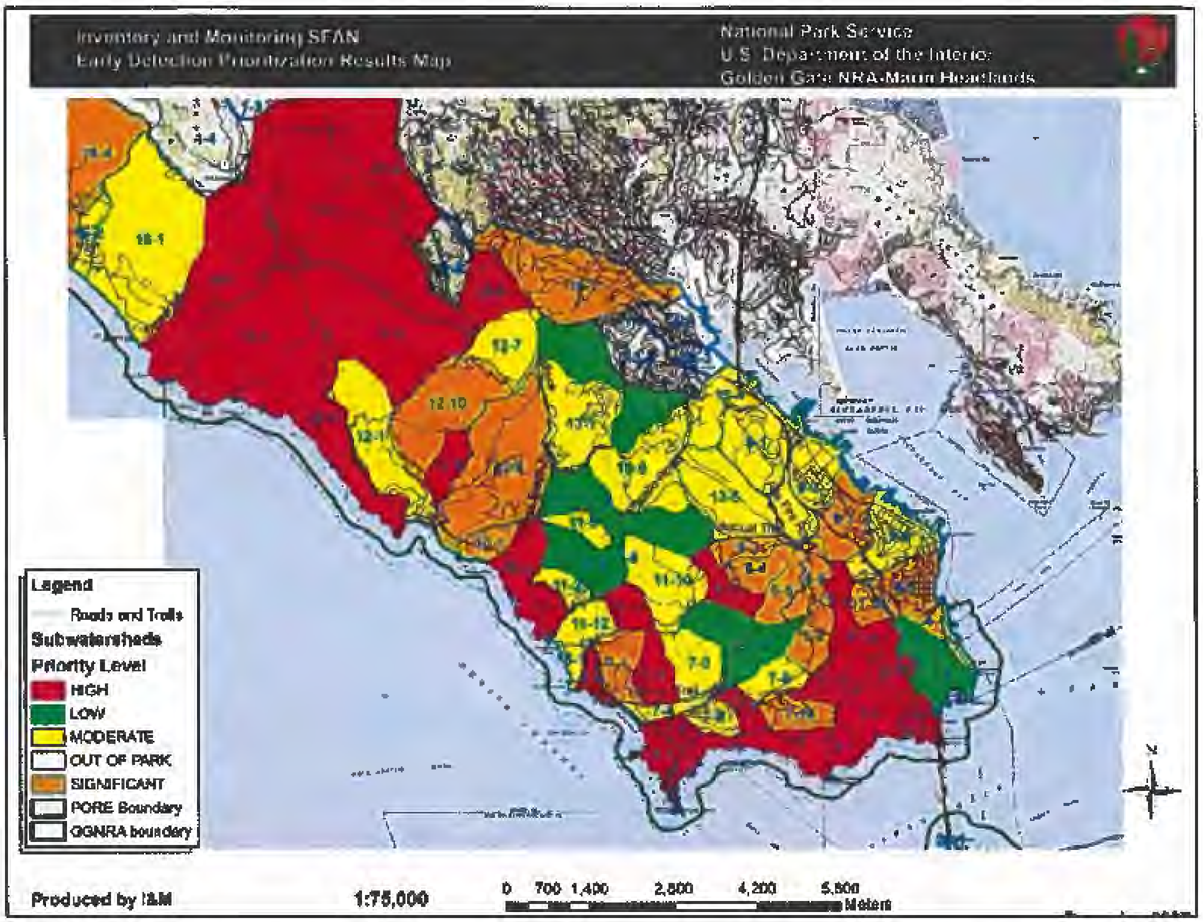


Figure 4. Map of prioritized areas (subwatersheds) for invasive plant early detection in the Golden Gate National Recreation Area, Marin Headlands, California. Hydrologic units (at a variety of scales) were used to define areas. Subwatershed prioritization criteria included abundance of rare or at-risk native species or species alliances, current level of invasive plant species richness and abundance, risk of invasion, and level of previous investment. Source: Williams et al. 2009.

3.2 Evaluate the Status of Priority Species and Areas

Some of the most important pieces of information that help managers develop an effective and efficient Plan is an understanding of the ecology as well as the status of the species they intend to manage. *Status* here refers to the location, distribution, and abundance of invasive plant species obtained through invasive plant inventories or early detection surveys. Data obtained from these surveys are used to:

- Develop specific and measurable objectives (section 3.3)—in order to ask, *where are we now?*, there must be a clear and definitive answer to the question, *where did we start*?
- Understand patterns of invasive plant introduction and spread.
- Inform the development and prioritization of management strategies.
- Guide on-the-ground management activities.
- Evaluate management effectiveness, learn, and adapt (section 3.7).

In addition, data and visualizations of invasive species status (such as species occurrence maps) can increase understanding of the invasive plant problem and may, as a result, lead to increased support. Decision-makers, the private sector, and the general public often have limited understanding of the threats posed by invasive species to the environment, economies, human health, and cultural values. Invasive species management competes for funding with many other interests. Lack of awareness, support, and funding often constrain adequate invasive species management.

If quantitative data concerning the status of priority invasive plants within the Plan's spatial scope are lacking, we recommend these surveys are conducted before developing management objectives and strategies. If this is not possible, consider the following: conducting interviews with field staff or local invasive species experts, mining online species occurrence databases, or reviewing reports or papers that contain information about vegetation within the Plan's spatial scope. Ideally, inventory and monitoring of invasive plants (or vegetation as a whole) becomes an integral component of your Plan (see section 3.7).

3.2.1 Inventories

An inventory is a type of survey that is used to determine the location or condition of a resource at a specific time. In this Guide, *inventory* refers to a catalogue of invasive species that includes information on their location, abundance, and distribution in a defined location (see the examples in figures 5–8). Inventories provide a snapshot of the distribution and abundance of invasive plants across a landscape and are critical for understanding the invasion problem, patterns of spread, and impacts (economic and ecological) and ultimately building a strategic and adaptive Plan (Rew and Pokorny 2006). When resources are limiting, consider inventorying the highest priority areas first and phasing inventory of lower priority areas over time.

"An inventory serves to diagnose the weed problems within a landscape, and not until the diagnosis is complete can comprehensive and complete management actions be taken. In a sense, weed inventories (or early detection) are as critical to land health as medical exams are to human health, and a tangible weed map is just as vital to a land manager as an x-ray would be to a medical professional."

Andersen and Dewey 2007

3.2.2 Early Detection

Early detection monitoring consists of systematic and repeated surveys of areas deemed high-risk for becoming infested with new invaders and is typically focused along likely routes of invasion and in areas believed to be un-infested ("clean" areas). Early detection surveys are focused on detecting the location of invasive species that are not yet established within a defined area, but the potential for establishment exists (Olsen et al. 2015). Early detection is critical for documenting new and highly invasive species for eradication before they become established, widespread, and abundant and cause both economic and ecological harm.

3.2.3 Inventory and Early Detection Methods

There is not a prescriptive, "one-size-fits-all" method or approach for invasive plant inventories or early detection. The methods will vary depending on survey objectives, species detectability (influenced by abundance, phenology, color, or size), spatial scale, ecosystem type, budget, and available expertise.

In the broadest sense, there are two basic approaches to inventory and early detection surveys: (1) ground-based and (2) remote. Below we provide a summary of these two approaches adapted from the USFWS's *Invasive Plant Inventory and Early Detection Guide* (USFWS in prep.).

Aerial invasive plant survey
CREDIT: Wildlands Conservation Science, LLC.

As the name implies, *ground-based inventory methods* are those in which the surveyor is observing and recording the location of invasive plant infestations from the ground. Depending on the terrain and accessibility of the site, many of these ground-based methods can be carried out on foot or with the aid of vehicles such as trucks, ATVs, boats, etc., that can enhance the efficiency of the survey. Ground-based methods include corridor surveys, grid-based surveys, full coverage swaths, opportunistic sampling, line transects, belt transects, permanent plot monitoring, and photo points.

As compared to ground-based methods, *remote methods* are generally accomplished by sensors deployed on planes, helicopters, and drones from which visual data are collected (collectively referred to as *remote sensing*). Remote methods also include aerial mapping of invasive plant populations by human observers from a helicopter.

The ability to detect weeds remotely depends on the unique properties of the weed of interest, the size or extent of the infestation, and the spectral and spatial resolution of the sensors employed (Bradley 2014). In some cases, the spatial extent or size of the images available is in direct conflict with image resolution. For example, flying at a lower altitude to capture more detail will require more passes to cover a given area. An integral part of remote sensing is performing a field-based accuracy assessment to ground-truth results.

There are many remote sensing methods that have been used to survey invasive plants. Excellent descriptions of different techniques as well as examples of how those techniques have been used have been published by several authors (Bradley 2014; Huang and Asner 2009; Lass et al. 2005; Madden 2004) and should be read by those considering remote sensing approaches to invasive plant inventory; many of these reviews are summarized in table 2 of USFWS (2018). The U.S. Forest Service Remote Sensing Applications Center website (*https://www.fs.fed.us/eng/rsac/*) also provides excellent guidelines on plant characteristics needed to employ remote sensing techniques as well as criteria for selecting the best approach for a given survey objective.

The USFWS's *Invasive Plant Inventory and Early Detection Guide* (2018) summarizes factors to consider when planning these surveys and points to existing survey methods, protocols, and mapping guides.

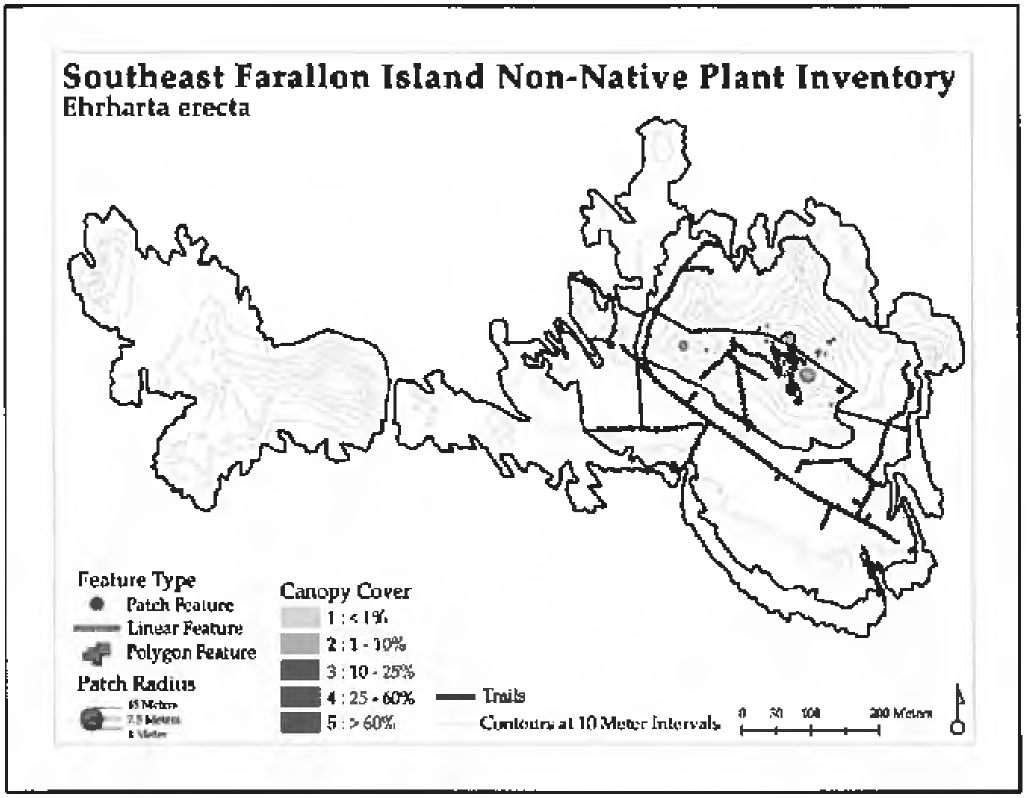


Figure 5. Farallon Island National Wildlife Refuge invasive plant inventory map: *Erharta erecta*. Results of island-wide inventory using field-based mapping methods. Source: Holzman et al. 2016.

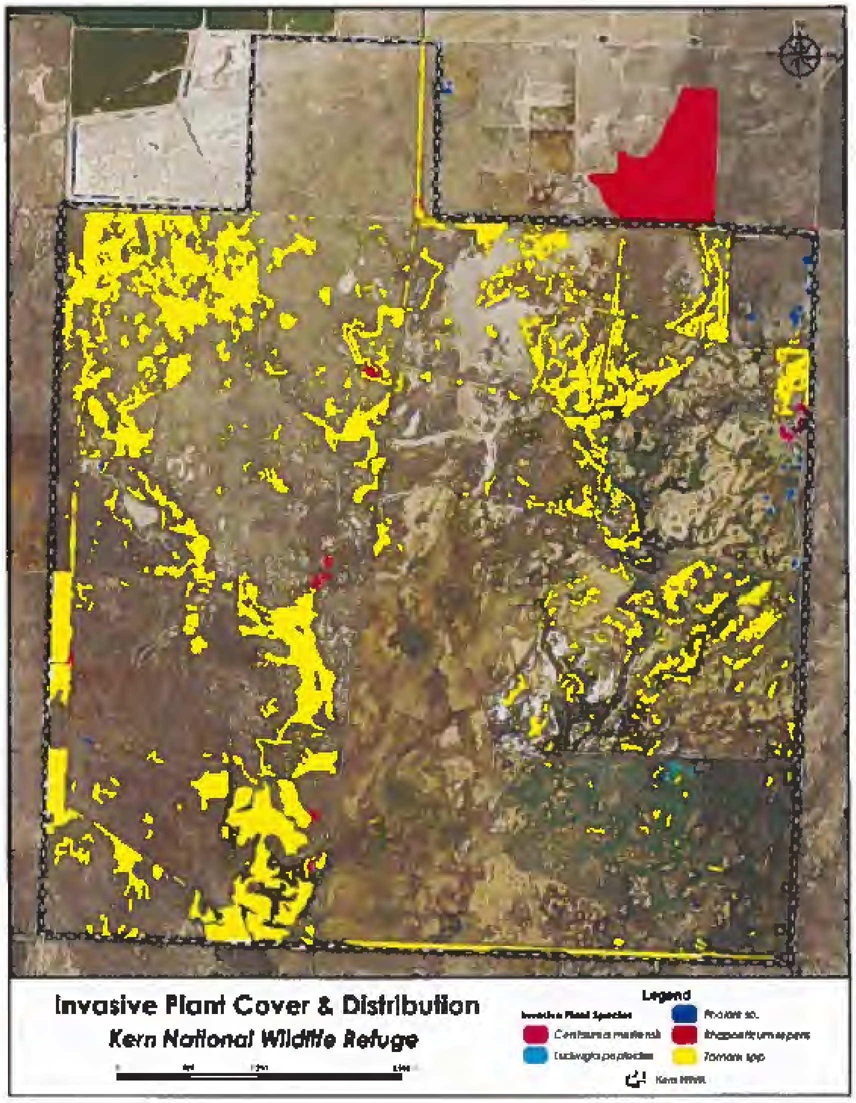


Figure 6. Kern National Wildlife Refuge invasive plant inventory map. Inventory conducted using aerial (helicopter) field-based mapping methods. Source: Ball and Olthof 2017.

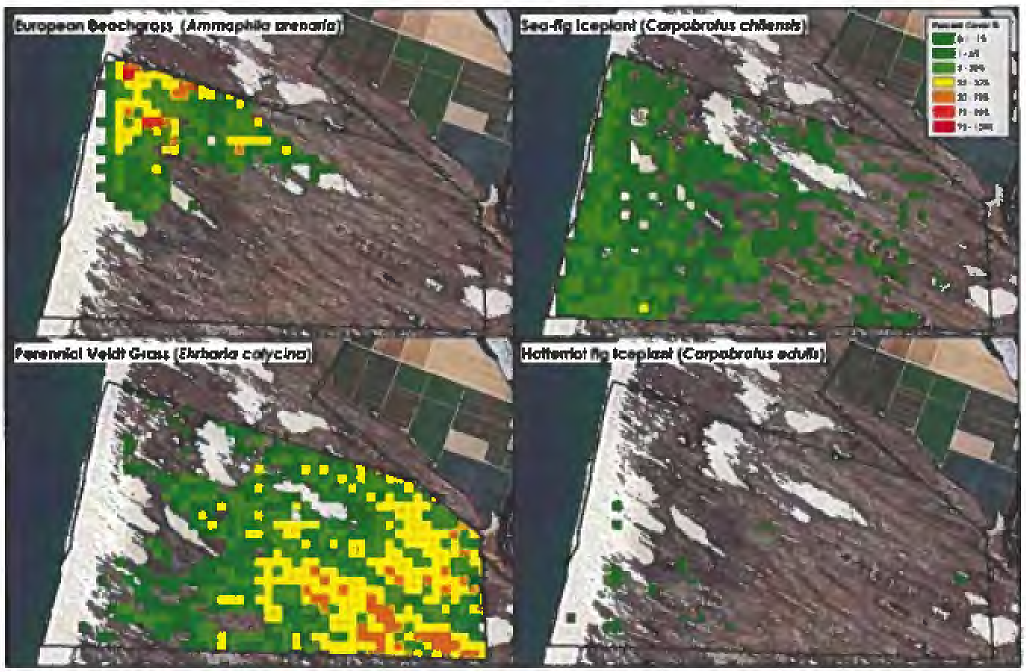


Figure 7. Guadalupe-Nipomo Dunes National Wildlife Refuge invasive plant inventory map. Inventory conducted using aerial (helicopter) grid-based mapping methods. Source: Ball and Olthof 2017.

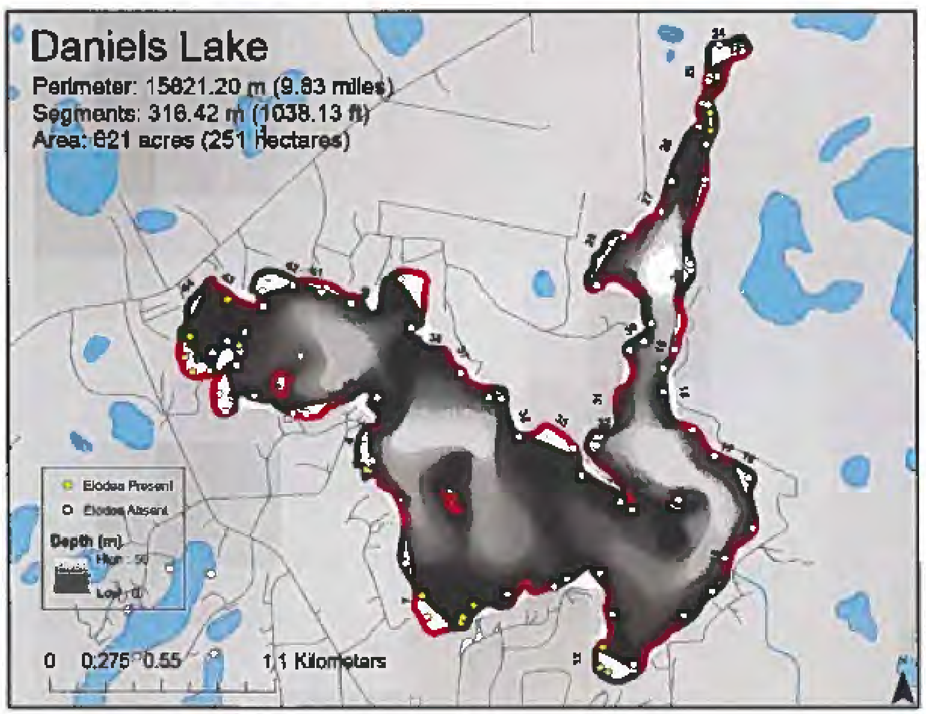


Figure 8. Results of early detection surveys for the aquatic plant Elodea **(cross between *E. canadensis* and *nuttallii*) at Daniels Lake, Kenai Peninsula, Alaska. Field-based survey. Source: Bella n.d.**

3.3 Develop Invasive Plant Management Objectives

Put simply, an *objective* is a statement detailing the desired outcome or result of management—what success looks like. Depending on the invasive plant management situation, expected outcomes likely fall into one or more of the categories below:

- Preventing introduction of new and highly invasive species
- Containing the extent/preventing further spread of existing infestations
- Reducing the cover of existing infestations
- Eradicating species

Four questions an objective should answer:

1. What is the expected change and where?
2. How much change do you want to see, and in what direction?
3. What needs to be measured to evaluate change?
4. Over what time period is change expected to occur?

Although mentioned previously, it's worth repeating here the importance of well-crafted objectives; they provide the foundation for evaluation, learning, and adaption of management to ultimately improve outcomes. They help us answer the following questions: *is our management program working?* and *if not, why not?*

To answer these questions, we first need to know what the desired impact is—the objective. Second, we need to track whether strategies were implemented or not. Third, we need to monitor attribute(s) of invasive plants (spelled-out in the objective).

A well-crafted objective meets the following SMART criteria (Foundations of Success 2009).

- Specific—*what is expected* and *where* are clearly defined so that all people involved in the project have the same understanding of what the terms in the objective mean.
- Measurable—definable in relation to some standard scale (numbers, percentage, fractions, or all/nothing states).
- Achievable—achievable and appropriate within the context of the project site and available resources. Considerations: people, technical capacity, funding, and political, economic, and other constraints.
- Results-oriented—focuses on the result of management actions, not the actions themselves.
- Time-bound—specifies when results are expected.

Avoid ambiguity by wording objectives clearly. A clearly worded objective is easy to understand and difficult to misinterpret. Avoid or minimize using words and terms that are subject to interpretation without numeric/measurable values attached, such as *high quality*, *reduce*, *enhance*, and *restore*. Objectives should contain a measurable element that can be monitored to evaluate progress; it should be clear from the objective what needs to be measured.

Objectives—no matter how measurable or clearly written—must be achievable. Avoid setting your program up for failure. If you cannot resolve constraints on achieving an objective, then consider discarding or rewriting it. Consider both short- and long-term objectives. Be realistic about what is required to successfully achieve an objective, and use sound professional judgment to develop reasonable expectations of time, staff, and funds available to pursue the objective. Objectives should specify an end result rather than state the action(s) that will be taken; when reading a results-oriented objective, it should be clear what success looks like in terms of the result, not the actions taken (such as how many gallons of herbicide sprayed in a given year). Examples of objectives and how well they pass the "SMART test" are provided in table 7.

Table 7. Examples of invasive plant management objectives and the degree to which they are SMART (specific, measurable, achievable, results-oriented, and time-bound). Generic area names are provided in cases where objectives are drawn from existing plans.

Objective	*S*	*M*	*A**	*R*	*T*	*Notes*
Broom-free by 2003!	N	Y	Y	Y	Y	Lacks specificity: which broom species? Where?
Decrease the abundance and extent of target invasive species in management areas A and B.	N	N	Y	N	N	What are the target invasive species? By when?
Eradicate high-priority species from high-quality habitats.	N	Y	Y	Y	N	Lacks specificity about what species, where eradication will occur, and by when.
Reduce cover of non-native species in Area C by 10% by 2020.	N	Y	Y	Y	Y	Which non-native species are being referred to? Plants?
Conduct EDRR surveys on an annual basis for yellow starthistle along all road within District X.	Y	Y	Y	N	N	Statement about actions that will be taken rather than the result.
Annually spray all known populations of Elodea in Refuge X.	N	Y	Y	N	Y	What species of Elodea? Specifies the management action that will be taken rather than the result of the management action.
Eradicate barbed goatgrass from Area D by 2020, defined as finding no evidence of plants for a period of five growing seasons.	Y	Y	Y	Y	Y	SMART objective
Reduce cover of French broom in Area E to 5% by 2019.	Y	Y	Y	Y	Y	SMART objective
Populations of Spotted Knapweed at Areas B and C will decrease at a rate of 25% per year until eradicated by 2010.	Y	Y	Y	Y	Y	SMART objective

*Note: We assume objectives were written to be achievable.

Developing objectives that are achievable over the life of your Plan requires examining several key pieces of information including:

- What species and areas are a focus of management?
- What is the status of priority species within the Plan scope?
- What are the major constraints: accessibility, spatial scale, availability of people or funding, technical capacity, regulations, politics, etc.?

3.4 Develop Invasive Plant Management Strategies

An *invasive plant management strategy* is a collection of activities or projects aimed at preventing, eradicating, containing, and/or suppressing (asset-based protection) targeted invasive plant species. Deciding which activities to employ and where can be a complex process because there are many factors to consider, such as species abundance and ecology; site characteristics such as scale, sensitive resources, and accessibility; capacity to implement (people, funding, and technical expertise); and socio-political issues. If you have completed the initial planning steps—gathering site specific information, prioritizing, assessing status, and developing SMART objectives (chapter 2 and sections 3.1–3.3)—you are well-positioned to design an effective and achievable strategy.

In this section, we summarizu the four basic approaches to invasive plant management (prevention, eradication, containment, and control) (figures 9 and 10) and point to techniques and resources to help you design an optimal invasive plant management strategy. In additiun, appendix A lists other resources for understanding invasive plant ecology, imagery, and management techniques (prevention and control). Appendix B points tu publicly available Plans and the types of information they contain; these Plans serve as examples of information discussed in this section.

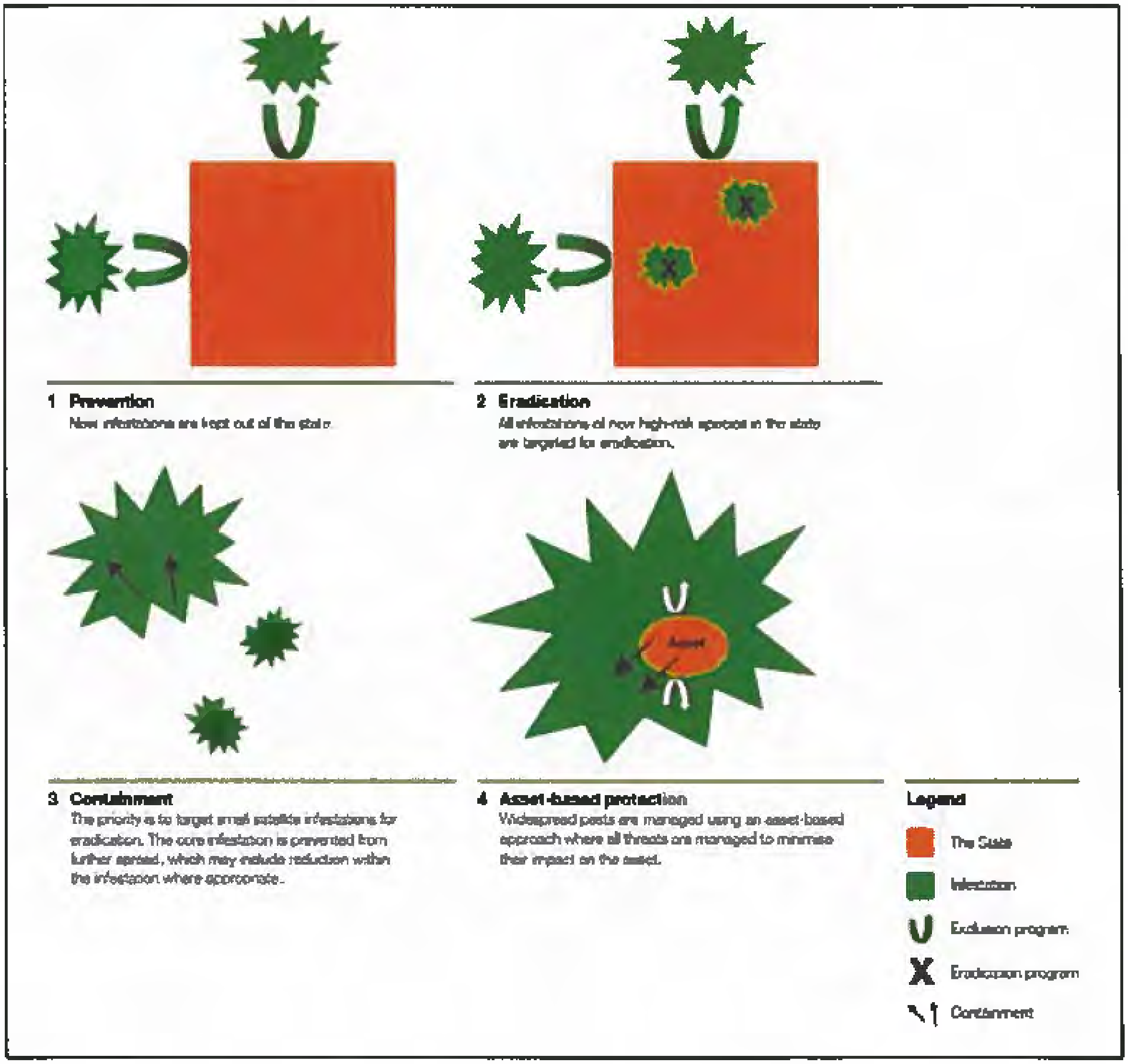


Figure 9. Approaches to invasive plant management at different stages of invasion. Source: Agriculture Victoria 2002.

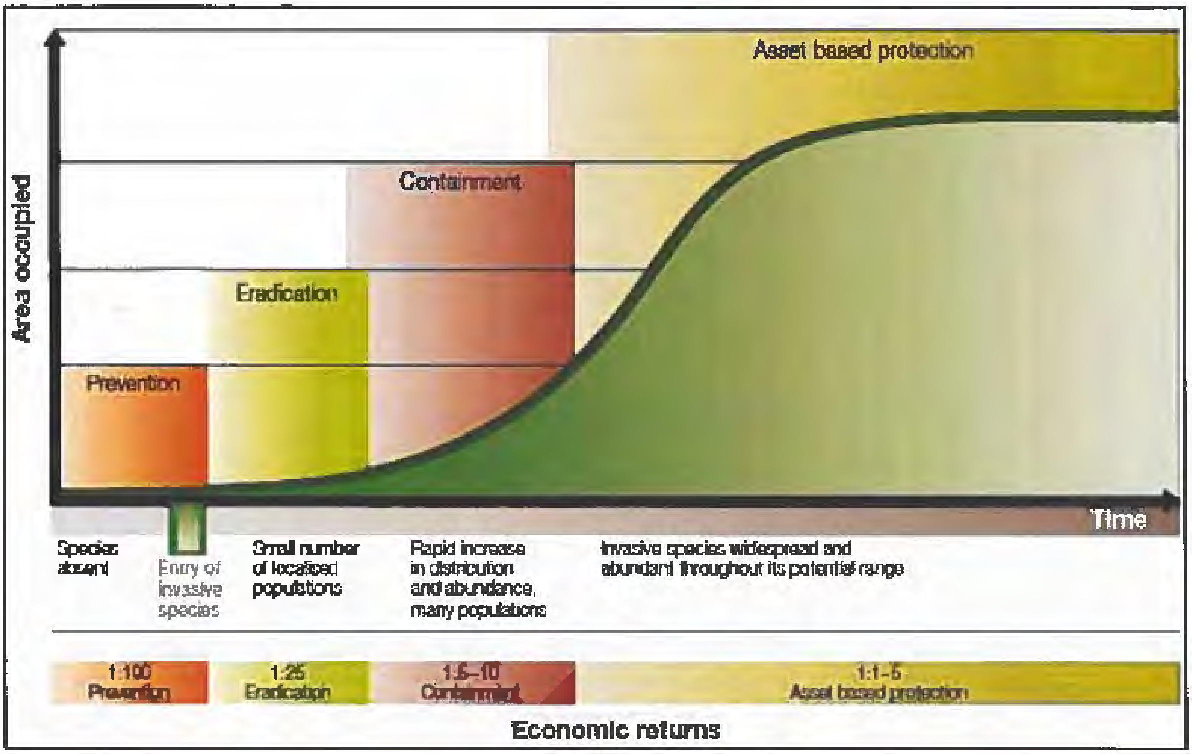


Figure 10. Generalized species invasion curve (the S-curve) and associated management approaches and cost-benefit ratios as area occupied increases. The amount of benefit for every dollar spent decreases as the area occupied increases. Source: Agriculture Victoria 2002.

3.4.1 The Four Basic Approaches to Invasive Plant Management

Prevention (or Biosecurity)

Preventing the introduction of invasive plant species is the first line of defense against invasive species (figures 9 and 10). Together, prevention with EDRR is the most cost-efficient way of reducing the economic and ecological costs of invasive species. Once established, invasive species can be extremely difficult and costly to remove. Even after successful removal, damage to food web dynamics, nutrient flow mechanisms, and other intricacies of the original ecosystem may persist.

Invasive plants are introduced (and spread) by vectors. A vector is the conveyance that moves a non-native propagule to its novel location (Lockwood et al. 2013). Invasive plants can be transported by natural means such as wildlife, wind, and water. Transport also occurs by anthropogenic means; human activities that can inadvertently lead to invasive plant introductions include:

- Importation of contaminated materials such as plants, mulch, wood, soil, gravel, or animal feed.
- Recreational activities such as hiking, biking, boating, and camping.
- Land management activities (carried out by staff, volunteers, partners, and contractors) that involve movement of people, vehicles, or tools. Examples include inventory and monitoring, routine maintenance activities (such as mowing), restoration activities, fire management activities, and invasive plant management activities.
- Other human activities that lead to disturbance or disruption of ecological processes, thereby creating novel situations and opportunities for invasion.

Examples of locations that are vulnerable to invasion include:

- **Vector pathways.** A *vector pathway* is the route between the non-native propagule source and release location (Lockwood et al. 2013). Common vector pathways include roadsides, trails, waterways, and utility corridors.
- **Areas where humans and their vehicles/tools frequent or congregate** such as buildings, boat launch sites, campsites, and vehicle or tool storage areas.
- **Areas of high intensity or frequent disturbance (natural and anthropogenic).** Disturbance facilitates invasion and can be described as a "relatively discrete events in time that [disrupt] ecosystem, community, or population structure and [change] resources, substrate availability, or the physical environment" (Lockwood et al. 2013; White and Pickett 1985). Examples of anthropogenic disturbances include restoration or enhancement activities, regular maintenance activities (such as mowing), resource extraction, and toxic spills. Examples of natural disturbance events include floods, tides, fire, and erosion.

Understanding the likely means of introduction and transport of invasive plants at your site is key to developing prevention or biosecurity strategies.

Eradication

Eradication is the complete removal of an invasive plant species (including reproductive propagules) from a defined area (figures 9 and 10). Eradication is most feasible when an infestation is small.

To understand how the size of an infestation affects whether eradication is an achievable objective, Rejmanek and Pitcairn (2002) analyzed decades of eradication efforts by the California Department of Food and Agriculture and found that eradication of infestations smaller than 1 hectare (2.5 acres) was usually successful, while only a third of infestations between 1 and 100 hectares (2.5 and 250 acres) and a quarter of all infestations between 101 and 1,000 hectares (250 and 2,500 acres) were eradicated (figure 11). Costs associated with eradication increase dramatically with size of infestation.

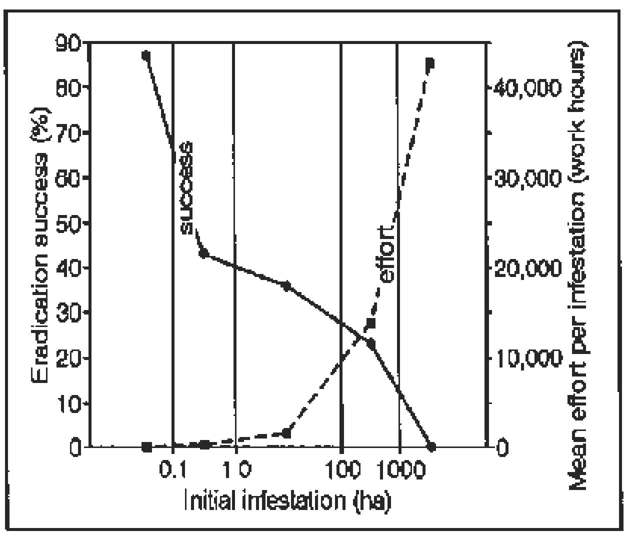


Figure 11. The dependence of the eradication success (%) and the mean eradication effort per infestation (work hours) on the initial size of infestations. Source: Rejmanek and Pitcairn 2002.

Successful eradication projects require (1) having adequate resources and commitment to see the project through to completion; (2) having an entity with authority to implement eradication; (3) fully understanding the biology of the species; (4) having the ability to detect the target species at low densities; and (5) having capacity for subsequent restoration of the system (Simberloff 2003).

Eradicating invasive plant populations when they are small requires that we first detect them and then eradicate them quickly before they become widespread and abundant — a concept commonly referred to as EDRR. Early detection involves systematic and repeated surveys for new species. Early detection surveys are commonly focused in areas at high risk of invasion such as vector pathways, areas where vectors congregate or frequent, and disturbance areas (see *Prevention* section above for more detail on this topic). USFWS's *Invasive Plant Inventory and Early Detection Guide* (2018) summarizes factors to consider when planning early detection surveys and points to existing survey methods and protocols. For example, the National Park Service has developed invasive plant early detection protocols for several of its park networks (*https://www.nps.gov/im/networks.htm*).

Along with formal early detection surveys, an organization should also have a structure in place for reporting incidental observations of potentially new and harmful invasive plant species. Observations should be confirmed by an expert, and then the priority of the species for eradication should be evaluated.

Containment

Containment is defined as any action taken to prevent establishment or to control a plant species beyond a predefined area known as the *containment unit*. *Control* is defined as the act of reducing the occurrence or abundance of invasive plants using one or more IPM chemical, biological, cultural, or mechanical removal techniques.

The containment unit comprises the area where the species currently exists (occupied zone) plus a surrounding buffer zone that is free from plants but can receive propagules (such as seeds) (Fletcher et al. 2015). Containment is typically undertaken when eradication fails or is infeasible (figures 9 and 10). Containment involves repeated searching and removal of individuals (EDRR) that arise within the buffer zone, but it can also encompass prevention activities to slow the rate of spread into the buffer zone as well as suppression of populations within the occupied zone. Containment must continue indefinitely unless the means to suppress and ultimately eradicate the core infestation become available. Given this reality, it is worth examining the cost of eradication versus long-term containment.

Containment may be a viable option (over eradication) wherever a species occupies a large area, has small dispersal distances, and has long-lived seed banks (Fletcher et al. 2015). In addition, the longer an infestation has been established and the further it has spread, the more likely containment will be cheaper than eradication (Fletcher et al. 2015). Containment may also be a viable option in the short term when resources are extremely limited. As additional resources become available, reducing—and ultimately eradicating—the extent and abundance of plants in the occupied zone may become more feasible. If containment of a species is the desired approach, your Plan should clearly define the containment strategy, including what species will be contained, how, under what conditions, and where (defining the containment unit or area). See Fletcher et al. (2015) for more information on how to assess whether containment can outperform eradication, and under what conditions it is a valid management approach.

Asset-Based Protection

Asset-based protection means limiting invasive plant control activities to portions of an infestation that directly threaten high-value conservation targets (such as areas supporting a high-valued species, community, ecosystem, or culturally significant asset) (figure 9, 10). Asset-based protection is commonly practiced when an invasive species is widespread and abundant and there is little hope of eradication. As with eradication and containment, a variety of techniques can be used to control invasive plants (see section 3.4.2).

3.4.2 Prevention and Control Techniques

Prevention Techniques

Identifying the most appropriate techniques for preventing the introduction or spread of invasive plants requires:

1. Clear objectives—knowing what you want to prevent and where.
2. Site-specific knowledge about risk—areas within your spatial scope at high risk of invasion and human activities that are likely to lead to invasion.

This information will directly inform the types of techniques and best management practices (BMPs) to reduce risk of invasive plant introduction and spread. Useful references for conducting invasive species risk assessments and identifying prevention techniques suited to your situation are listed below:

- *Preventing the Spread of Invasive Plants: Best Management Practices for Land Managers* (California Invasive Plant Council [Cal-IPC] 2012). This resource includes helpful BMPs for a range of activities. Web link: *https://www.cal-ipc.org/docs/bmps/dd9jwo1ml8vitq9527zjhek99qr/BMPLandManager.pdf*
- *Compendium of Recommended Procedures and Best Management Practices Relevant to Minimizing the Introduction of Invasive Species by Service Activities* (USFWS 2016). Covering all taxa, this resource points to a wealth of information about risk assessment methods, prevention techniques and practices, and outreach and communication materials. Web link: *https://ecos.fws.gov/ServCat/Reference/Profile/105555* (Appendix 2)
- *Guide to Noxious Weed Prevention Practices* (USDA Forest Service 2001). This guide includes helpful BMPs for a range of activities. Web link: *https://www.fs.fed.us/invasivespecies/documents/FS_WeedBMP_2001.pdf*

Ideally, a formal invasive plant risk assessment is conducted as part of the planning process. If time does not allow for an assessment, it should be called out as an activity so that prevention measures are focused on the highest-risk areas and activities.

Control Techniques

As noted above, invasive plant control is the act of reducing the occurrence or abundance of invasive plants using one or more techniques (such as chemical, biological, mechanical, or cultural removal). Several factors should be considered when selecting control techniques, including:

- Management objectives—what you are trying to achieve (see section 3.3)
- Target species ecology, distribution, and abundance
- Capacity to implement—people, cost, and technical capacity
- Site characteristics such as scale, accessibility, and politics
- Potential non-target effects
- Likelihood of success

Ideally, multiple techniques are employed for a given species or species group to avoid development of resistance (figure 12). *Resistance* is a decline in effectiveness of a particular control technique over time. Reliance on any single technique to control weeds results in selection for species or populations that can survive that practice (Coble and Schroeder 2016). A clear sign that resistance is occurring is a decline in effectiveness over time. Invasive plants can develop resistance to any type of control technique (such as mechanical, chemical, or biological), but it is more commonly associated with herbicide use. The International Survey of Herbicide Resistant Weeds (2018) reports there are currently 496 unique cases (species x site of action) of herbicide-resistant weeds globally, with 255 species (148 dicots and 107 monocots). Further, weeds have evolved resistance to 23 of the 26 known herbicide sites of action and to 163 different herbicides.

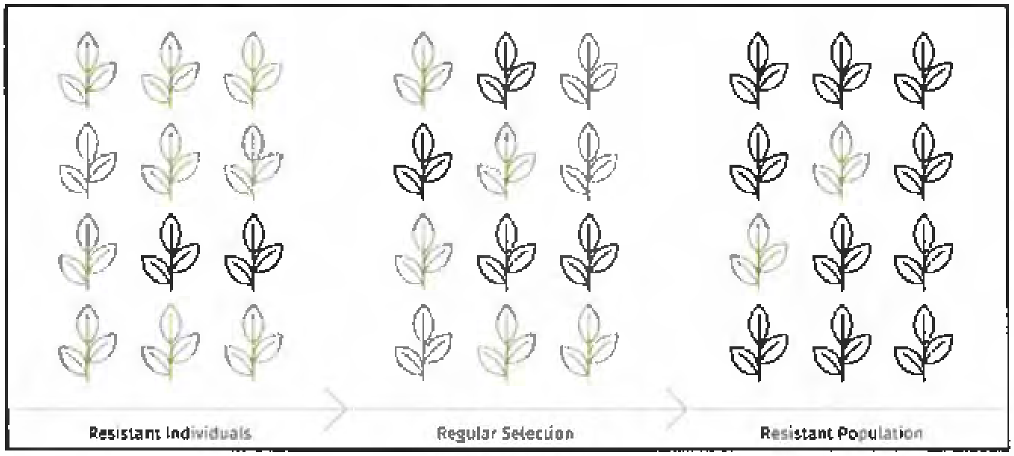


Figure 12. The process of selection for herbicide resistance. Resistance individuals (blue) increase in number over time as a result of herbicide selection pressure. Source: USA Herbicide Resistance Action Committee 2018.

Table 8 below summarizes invasive plant control techniques and related advantages and disadvantages in their use. Each technique can be carried out using a variety of methods. A review of species-specific control information (published literature, books, invasive species websites, local experts) is a necessary step in developing your overall strategy (section 3.4.3). There is no single resource for invasive plant control techniques. Appendix A provides a wealth of online resources for invasive plant management, many of which lead to species-specific control information.

Table 8. Summary of invasive plant control techniques (adapted from Tu and Robinson 2013).

Technique	*Advantages*	*Disadvantages*
Manual: physical removal of invasive plants using non-mechanical tools such as hands, shovels, picks, axes, hand-saws, or machetes	Little training is needed for safe use of many tools, and they can be used in a variety of situations; hand tools are relatively low cost and can provide very specific and targeted control. Ideal for smaller infestations.	May be time- and labor-intensive for moderate to large infestations. Some manual tools may be dangerous to use. Potential non-target effects: inadvertent disturbance to or removal of non-target species.
Mechanical: physical removal of invasive plants using mechanized tools such as mowers, brush-cutters, chainsaws, or earth-moving equipment	Many tools/equipment can be used in a variety of situations and have low implementation costs. Can provide very specific and targeted control. Ideal for small infestations.	May be time- and labor-intensive for moderate to large infestations. May require qualified individuals or training to operate some mechanized tools or equipment. Potential non-target effects: inadvertent disturbance to or removal of non-target species.
Cultural: land management practices such as grazing, prescribed fire, or irrigation/flooding	Control of moderate to large infestations may be possible. Can be low effort and cost per unit acre relative to other techniques. In some cases, may lead to positive response by native plants.	In some cases, may lead to an increase in invasive plants if not used appropriately. Often will not completely eliminate the target species from an area. Potential non-target effects: inadvertently disturbs or removes non-target species and promotes invasive plant spread.
Biological: introduction of novel predators, parasites, and pathogens such as insects, fungi, or microbes, to attack an invasive plant species	Relatively low cost per unit acre. May keep invasive plants at a low level across large landscapes. Long-term effectiveness is limited; must repeatedly treat invasive plant infestations once biocontrol agents are established.	May be expensive to develop. Often does not lead to eradication of the target invasive species. High risk of unintended consequences to native species and communities.
Chemical: application of herbicides to kill invasive plants	May be a cost-effective approach for larger infestations and lead to effective control when used appropriately. Often a variety of application mechanisms available (ground and aerial).	High risk of unintended consequences to native species and communities. Unintended consequences may include contamination of soil or water, harm to or removal of non-target species, human exposure, and health issues for applicators. May be expensive to obtain and/or apply chemicals. Often more regulatory requirements to apply. May be controversial in some areas.
Restoration of ecosystem processes or composition	Works to bring the project site to a desired and/or native state that is more resistant to invasion over the long term.	High cost. There may be a time lag to realized benefits. May not lead to elimination of the target invasive species.

3.4.3 Selecting an Optimal Set of Strategies

An invasive plant management strategy encompasses species or area-specific activities to achieve your objectives and avoid unintended harm to natural or cultural resources (non-target effects). Developing an optimal strategy requires evaluating the impact and feasibility of different combinations of approaches, techniques, and methodologies (we refer to these combinations collectively as *activities*). We suggest brainstorming potential activities with your objectives in mind, and then selecting a portfolio of feasible activities that is most likely to help you attain your objectives. It's worth emphasizing here that objectives should be the major factor driving the brainstorming process. Other factors used to evaluate the value of different invasive plant management activities are presented in table 9.

Tools and approaches for selecting an optimal set of strategies range from simple to complex, but most involve answering questions about the performance of a project or activity relative to your objectives, the feasibility of carrying it out, and the likelihood of non-target effects. Regardless of the method, involving your project team to build consensus around decisions is important. Tables 10 and 11 provide simple examples of evaluating alternative activities. Decision trees (figure 13) can also be a useful approach. The *Invasive Plant Management Decision Analysis Tool* (*https://ipmdat.org/ipmdat.html*) is an online decision support tool for evaluating different approaches to managing particular species. This tool does not tell you what to do; rather, it helps you evaluate various alternatives you have brainstormed.

Whether your approach is simple or complex, brainstorming and evaluating impacts and feasibility lead to more objective and transparent decisions, help teams reach consensus, provide a record of how decisions were made, and increase the likelihood that the strategy is implemented and successful.

Table 9. Factors to consider when developing an invasive plant management strategy.

Factor	*Description*
Management objective(s)	The degree to which an activity will lead to achieving a management objective
Species or species group characteristics	The degree to which the activity is well-suited to the species ecology, distribution, and abundance within the management scope
Non-target effects	The likelihood and degree to which the activity will result in unintended negative impacts on the environment or humans
Likelihood of success	The level of certainty that the activity can be successfully implemented and will work as expected
Feasibility of implementing	Cost and duration; technical expertise required and available; sociopolitical concerns; training or certifications required. Your organization may have sophisticated cost-estimating software, but in many cases a simple spreadsheet will do. Inventory data, if available, can be used to estimate costs. Cost per unit area can be derived from past management onsite or from interviews with others who have implemented similar activities.

Table 10. Simplified example of evaluating alternative invasive plant management activities for objectives focused on preventing establishment of new invasive plant populations (Objective 1), eradicating Species A from the entire site (Objective 2), containment of Species B to current extent (Objective 3), and suppressing Species C (Objective 4). Objectives drove the development of activities.

Activity	*Objective(s) addressed category*	*Impact*	*Feasibility*	*Non target effects*
Develop and provide staff and contractor training for preventing the spread of invasive plants (BMPs); implement BMPs	1–4	High	High	Low
Develop and implement early detection protocol focused on priority early detection species	1	High	Low	Low
Eradicate all early detection species, if found, using non-chemical methods	1	High	Medium	Low
Eradicate Species A from all management areas using herbicides (alternating Herbicides X and Y)	2	Medium	High	Medium
Eradicate Species A from all management areas using mechanical (mowing) and chemical methods (Herbicide X)	2	High	High	Medium
Contain current extent of Species B using chemical control (alternating Herbicides X and Y)	3	High	High	Low
Contain current extent of Species B using manual or mechanical methods (hand pulling and mowing)	3	Medium	Low	Low
Flood areas infested with Species C, followed by active native plant restoration	4	High	Medium	Medium
Use fire to suppress abundance of Species C within areas containing rare plants	4	Medium	Medium	Low
Use grazing and Herbicide Z to suppress abundance of Species C within areas containing rare plants	4	High	Medium	Low

Notes: impact = the degree to which the action will help meet one or more invasive plant management objectives; feasibility = degree to which activity is financially, technically, and politically feasible; non-target effects = potential for harm to natural or cultural resources as a result of invasive plant management activities.

Table 11. Simplified example of evaluating alternative invasive plant management activities for objectives focused on preventing establishment of new invasive plant populations (Objective 1), keeping clean areas clean from priority invasive plants (Objective 2), eradicating Species A (Objective 3), preventing spread and reducing extent of cover of current infestations of Species B (Objective 4), and understanding distribution of priority invasive plants and using this information to refine objectives (Objective 5). Objectives drove the types of activities proposed.

Activity	*Objective(s) addressed*	*Impact*	*Feasibility*	*Non-target effects*
Conduct invasive plant risk assessment (identify high risk areas and activities)	1, 2, 3, 4	High	High	Low
Develop and provide staff and contractor training for prevention and avoiding the spread BMPs; implement BMPs	1, 2, 3, 4	High	Medium	Low
Develop and implement ED protocol focused on priority early detection species. Surveys conducted annually in high priority areas (clean areas, wetlands, areas containing rare species) and every 2–3 years in lower priority areas.	1, 2	High	Medium	Low
Eradicate Species A using a combination of non-chemical methods (hand pulling, mowing)	3	Medium	Medium	Medium
Eradicate Species A from all management areas using manual (hand pulling), mechanical (mowing), and chemical methods (Herbicide X)	3	High	High	Medium
Contain current extent of Species B and reduce abundance of infestations in high priority areas using Herbicides X and Y.	4	High	Low	Medium
Contain current extent of Species B and reduce abundance of infestations in high priority areas using goats or other herbivores.	4	High	Medium	Medium
Use fire to suppress abundance of Species B within areas containing rare plants	4	Medium	Medium	High
Conduct inventory of priority invasive plants	5	High	Medium	Low

Notes: Impact = the degree to which the action will help meet one or more invasive plant management objectives; feasibility = degree to which activity is financially, technically, and politically feasible; non-target effects = potential for harm to natural or cultural resources as a result of invasive plant management activities.

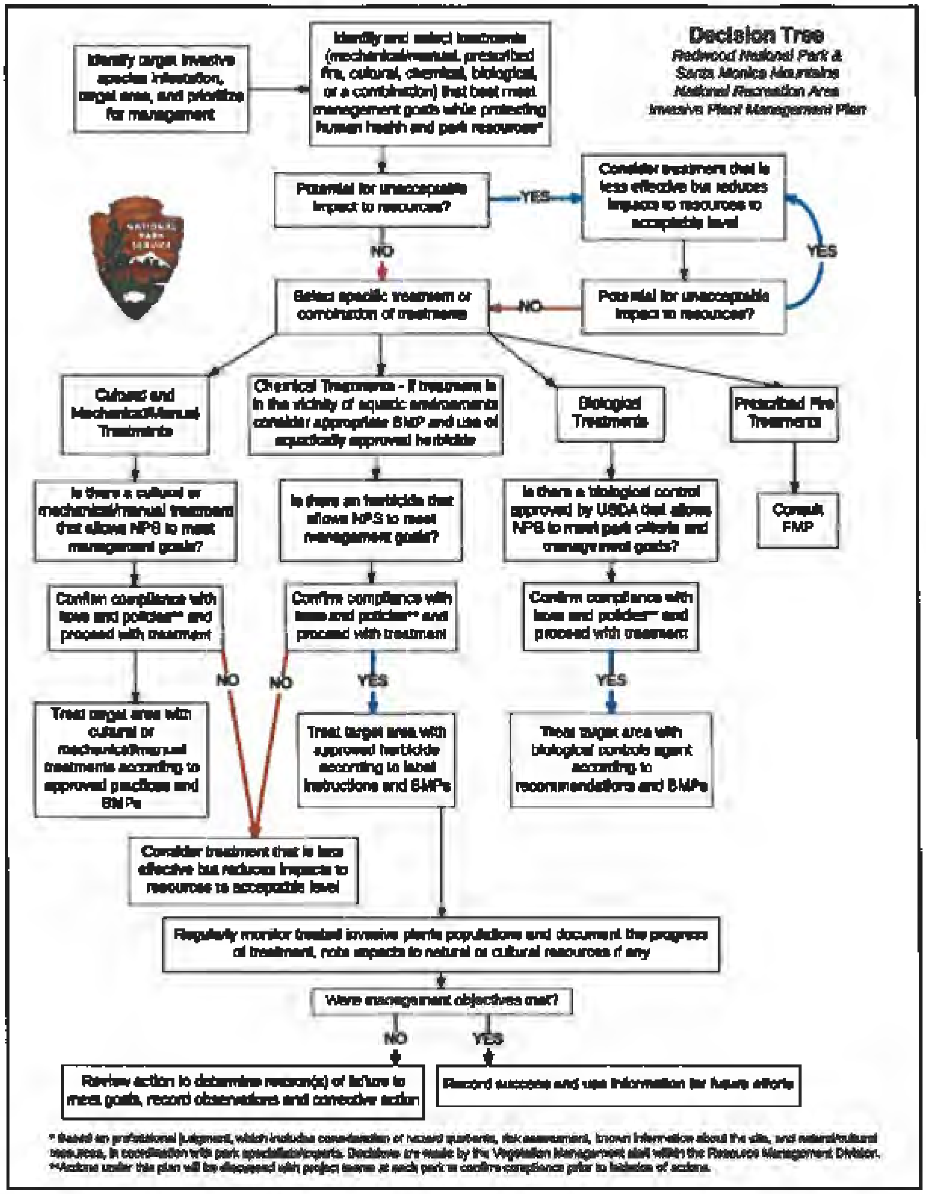


Figure 13. Invasive plant management decision tree for Redwood National Park and Santa Monica Mountains National Recreation Area. Source: National Park Service 2017.

3.5 Avoid Unintended Impacts of Invasive Plant Management

Although the purpose of invasive plant management is to prevent and reduce harm to important natural and/or cultural resources, unintended negative consequences (non-target effects) can result such as soil erosion, loss of native species or species habitat, reinvasion, secondary invasions, or further spread of invasive plants (table 12) (Zarnetske et al. 2010; Cal-IPC 2015; Pearson et al. 2016).

Table 12. List of commonly cited unintended consequences of invasive plant management activities. Many of the consequences listed here are possible with any invasive plant management activity.

Unintended consequence	*Description*
Soil disturbance, compaction, or erosion	Equipment use results in soil disturbance or compaction. Removal of plants and creation of bare ground can lead to erosion.
Water quality impacts	Chemicals or other introduced materials (such as sediment) can impair water quality.
Harm to non-target plants	People, equipment, or materials result in impairment or mortality of native plants.
Direct harm to wildlife	People, equipment, or materials result in wildlife displacement, impairment, or mortality.
Indirect harm to wildlife	People, equipment, or materials result in alteration of wildlife habitat.
Direct or indirect harm to cultural resources	People, equipment, or materials result in cultural resource damage or loss.
Further spread of invasive plants	People and/or equipment become vectors of invasive plant spread.
Create conditions for reinvasion	Activity results in soil disturbance or creation of open areas that are re-infested.
Human safety risk	Activity poses a risk to human safety.

Steps to reduce the likelihood of non-target effects include:

1. Assess the types and magnitude of non-target effects from proposed invasive plant management activities.
2. To the extent feasible, choose a portfolio of invasive plant management activities with the lowest likelihood of non-target effects.
3. Integrate BMPs into your invasive plant management program to avoid non-target effects.
4. In cases where non-target effects cannot be avoided, develop measures to help mitigate the non-target effect. This is a typical requirement of environmental permitting, which may contain specific restrictions based on the invasive plant management work in relation to high-value resources such as special-status species, sensitive species habitats, or wetlands.

A useful resource for developing BMPs to avoid non-target effects from invasive plant management activities is:

- *Best Management Practices (BMPs) for Wildland Stewardship: Protecting Wildlife When Using Herbicides for Invasive Plant Management* (Cal-IPC 2015). Among other information, the manual contains risk charts for potential impacts on wildlife for commonly used herbicides. Many of the BMPs in this document are applicable for other invasive plant management activities other than herbicide use.

Also see section 3.4.2 for a list of resources that include BMPs for preventing the spread of invasive plants and appendix B for examples of BMPs in existing Plans.

3.6 Conduct Work Planning

Up to this point, you have identified priorities, developed objectives, and devised a set of strategies to achieve your objectives. The information generated so far does not provide the specificity for implementation—this is the job of *work planning* (often referred to as *implementation planning* or *operational planning*). The purpose of an operational plan is to provide those responsible for implementing your strategy (and associated activities) with a clear picture of what needs to get done, where, and when, as well as how much it will likely cost over a specified period of time. Commonly, organizations develop 2- to 5-year operational plans that guide annual work. Without an operational plan, it is highly likely your invasive plant management strategy will not be implemented.

The level of detail needed in an operational plan depends on the intended purpose and audience. In general, a multi-year operational plan should be developed that specifies:

- Tasks and locations associated with Plan activities
- Who is responsible for carrying out activities
- Costs associated with activities
- Performance measures or indicators—in other words, a means for assessing the degree to which an activity or task was carried out.

Because conditions change over time, such as fluctuations in funding and/or staff, the operational plan will change and should be revisited frequently (such as annually). This information is critical to informing your organization's work on an annual basis. See appendix B for examples of Plans with work planning.

3.7 Monitor and Evaluate

Following implementation of invasive plant management strategies, managers should be able to answer these key questions:

1. Were activities implemented as planned? If not, why not?
2. Are we achieving our management objectives (or moving towards achievement)?

Answering these questions requires monitoring. *Monitoring* is the periodic process of gathering data to assess outcomes relative to your actions *and* your objectives. If you intend to practice adaptive management, monitoring should be conducted so that your organization can understand whether your program is on track and identify adjustments to improve outcomes. Other important benefits include:

- Enhancing accountability, credibility, and transparency with external donors, policymakers, and the public.
- Strengthening ownership of the work by partners and stakeholders, thereby improving the sustainability of the work.
- Capturing lessons to share with the broader conservation community, thereby improving learning beyond your organization.

"Monitoring should be done for learning, adapting, and improving. As such, it is important to collect the right information that will help you learn the most about your project site and the effectiveness of your interventions."
Foundations of Success 2009

3.7.1 Protocol Development

Regardless of the survey purpose, any natural resource survey effort, such as monitoring invasive plants, requires a set of instructions or a protocol. A protocol should include enough detail so that someone unfamiliar with the survey understands what, why, where, by whom, when, and how a survey is conducted (USFWS 2013). This includes identification of the management objective the survey will inform, what will be measured, how measures will be taken, considerations and costs for data collection, data management, analysis, and reporting of results.

Before investing in protocol development, determine if an existing protocol could be adapted to meet your needs by searching online databases (such as the National Park Service Data Store [*https://irma.nps.gov/DataStore/*] or the USFWS Service Catalog [*https://ecos.fws.gov/ServCat/*]) or talking with local organizations involved in vegetation and invasive plant management. More detailed information about developing monitoring protocols can be found in *How to Develop Survey Protocols: A Handbook* (USFWS 2013), *Guidelines for Long-term Monitoring Protocols* (Oakley et al. 2003), and *Guidance for Designing an Integrated Monitoring Program* (National Park Service 2012). A good resource for developing survey designs is *Measuring and Monitoring Plant Populations* (Elzinga et al. 1998). In addition, USFWS recently completed an *Invasive Plant Inventory and Early Detection Guide* (USFWS in prep.). Lastly, examples of invasive plant inventory and monitoring protocols and reports are provided in appendix B.

3.7.2 Data Management

Invasive plant management involves the collection and management of data about (1) management actions (when, where, what, by whom) and (2) the status and trends of plants. Good management of data, whether they be spatial or non-spatial data, makes the data easier to access, understand, use, and share, but is one of the most commonly overlooked aspects of invasive plant management. Over time, poor data management can result in wasted time and money because the data cannot be found or understood. Ideally, a Plan should emphasize the importance of data management and describe basic data management practices that should be followed, such as:

- Metadata standards that should be used, such as the Federal Geographic Data Committee geospatial metadata standards or the North American Invasive Species Management Association standards (for invasive plant surveys).
- Describing how data will be organized and stored (such access databases, geodatabases, or established data management systems).
- Describing naming standards for species, such as the International Taxonomic Information System.
- Establishing file naming conventions.

Well-developed data management systems and workflows can save an organization significant amounts of time and money, provide continuity of work despite staff turnover, and provide a strong legacy of information to guide future decisions. Examples of Plans with data management elements are identified in appendix B.

3.7.3 Evaluation

Evaluation here refers to the regular assessment of outcomes. Such information is used to adjust your management strategies, as needed, to achieve your management objectives. Organizations should identify a mechanism for regularly checking in to assess outcomes. Evaluation should be conducted by people who are implementing the Plan as well as those who direct or planned the work. This may include annual evaluation and work planning as well as longer-term-interval (such as 5-year) Plan updates.

This page intentionally left blank.

Chapter 4

Writing Your Plan

Sahara mustard
Brassica tournefortii
CREDIT: ©Ryan O'Dell

This chapter describes the suggested elements and associated content of a Plan. It parallels the content generated in chapter 3 and follows the Plan template in appendix C. Appendix B also points to publicly available Plan examples. The level of detail a Plan contains depends on its audience and intended use. For example, if the Plan's purpose is to guide on-the-ground invasive plant management activities, then a high level of detail may be needed to increase the likelihood that the Plan is carried out as intended, especially as staff change over time.

4.1 Plan Introduction

The introductory sections of a Plan state its purpose and need and provide an overview of the management context. Further topics include the spatial scope, environmental and/or cultural setting, conservation targets, existing management goals and objectives, history of invasive plant issues and management, and regulatory context. These topics are summarized below and appear in the Plan template (appendix C).

4.1.1 Plan Purpose and Need

Your Plan should identify the purpose and need for an invasive plant management program, clearly articulating why the organization must take action. Plans often start by describing how invasive plants currently (or have the potential to) decrease biodiversity, degrade habitat, decrease water availability, or threaten recreational uses or infrastructure. Some also detail how invasive plant management is important for meeting the organization's conservation vision and goals. The more links you can draw between site conservation goals and how invasive plants impede those goals, the better. Doing so increases the likelihood that the need for invasive plant management is understood by leadership and other stakeholders and is ultimately supported. You may also want to consider linking invasive plant management at your site to other local, regional, or national efforts aimed at reducing harm from invasive plants.

Ideally, this section of the Plan also describes the intended audience and how the Plan should be used (and adapted) over time.

4.1.2 Spatial Scope and Setting

A clear spatial scope shows the rough geographic boundaries where invasive plant management will occur. To orient readers, it is useful to include in your Plan a text description of the spatial scope as well as maps showing boundaries, management units, and place names. Other relevant spatially referenced information may include topography, watersheds, hydrology, soils, ecosystems, vegetation communities, roads, trails, and/or infrastructure.

The setting should provide a brief background on site establishment and governance. It should also provide an overview of major environmental features such as ecosystems, landcover (such as hydrology, soils, or vegetation communities), important ecological features or functions, sensitive biological resources such as federal or state-listed endangered species, important cultural resources, and any other defining characteristics of the site that should be considered in the context of invasive plant management. This information helps to ground invasive plant management in the larger context of your organization's work. It may also point to particular challenges that should be considered when developing or implementing invasive plant management strategies.

Example of a spatial scope map: Kenai National Wildlife Refuge

SOURCE: USFWS

4.1.3 Conservation Assets and Goals

Conservation Assets

The term *conservation assets* here refers to species, communities, or ecosystems that are the focus of conservation efforts within the Plan's spatial scope. Conservation assets may also include important physical, cultural, or paleontological resources. Although you may want to conserve all biodiversity or other important features of a site, focusing explicitly on protecting all high-valued assets of a site from invasive plants is usually infeasible because of constraints on time, funding, and staff.

Your Plan should identify and describe the most valued or representative conservation assets because that effort informs (1) the species and locations on which invasive plant management should be focused, (2) the types of strategies to implement, and (3) the assessment of whether invasive plant management efforts are achieving the desired effect on assets over the long term.

Channel Islands fox
Urocyon littoralis

SOURCE: https://www.nps.gov/chis/learn/nature/island-fox.htm

It is also useful to describe how invasive plants will harm conservation assets if they were to spread and how they may cause harm in the future if invasive plant management does not occur. Specific examples will help readers understand the consequences of not adequately addressing invasive plant threats and will reinforce the need for management. Examples include how an invasive plant may outcompete native plant communities, increase fire frequency, lead to vegetation type conversions, or alter wildlife diversity.

Conservation Goals

It is important to identify and review existing conservation goals and objectives of the Plan scope (and consider including them in the Plan introduction) because they provide context, rationale, and focus for invasive plant management efforts and will help inform what species are a priority for management, where management should be focused, and the types of strategies that may be appropriate. This information is often found in conservation plans developed for the site and may be very broad or quite specific.

Existing site-specific management or conservation plans ideally contain goals or objectives that describe the desired state of resources (such as species, natural resource communities, ecosystems, or cultural resources). They may also contain specific objectives related to invasive plants, such as prevention or eradication of a particular species or a decrease in the overall extent or abundance of invasive plants. In many cases, invasive plant objectives may not yet exist or, even if they do, they may need refinement and should be re-examined as part of the planning process. Sections 3.3 and 4.4 address development and refinement of invasive plant management objectives.

Below is an example of a conservation target and related conservation goal and invasive plant management objective.

- **Conservation target:** tidal marsh ecosystem
- **Conservation goal:** By FY 2025, extent of high quality tidal marsh within Refuge X increases to 14,500 acres. High quality = unimpaired hydrology, dominated by native tidal-marsh associated plant species.
- **Invasive plant management objective:** By 2022, eradicate Algerian sea lavender at Refuge X.

4.1.4 Invasive Plant Management History

In cases where invasive plant management has occurred or is ongoing within the Plan scope, it is useful to describe management history, including focal species and locations, strategies employed, and successes and failures. This overview helps readers understand what has come before and what can be and was learned. This may include efforts to prevent, eradicate, control, study, inventory, or monitor invasive plants. When possible, cite sources of information, such as personal communications, pesticide use reports, maps, or reports.

4.1.5 Relevant Invasive Species Laws and Policies

Most Plans include a description of the legal (and sometimes political) context of invasive plant management at the site, including laws and policies governing invasive plant management planning and implementation. The level of detail here depends on the organization. Often times, relevant laws, policies, and regulations are summarized.

4.2 Methods

The methods chapter identifies who was involved in developing the Plan; information resources and processes used to inform its design; the people (public, leadership, others) or organizations who were informed of its development or engaged in the planning process; and how decisions were made. Use of a Plan by its intended audience will depend in large part on the readers' confidence that (1) the right people were involved in designing the Plan and (2) that its contents were developed using the best available information and processes. The methods chapter should describe any tools or processes that were used or developed to make decisions such as which species to focus on, which areas to focus on, and what strategies and activities to employ. This may be as simple as citing existing tools or describing new processes that were developed as part of the planning process. Lastly, it's useful to describe how the public, stakeholders, or others were informed about or engaged in the planning process. This helps readers understand how much others already know about what has been planned, whether or not they support those actions, and any considerations that need to be kept in mind as Plan implementation begins.

4.3 Invasive Plant Priority Species and Areas

A Plan should identify and describe the species and areas that are the focus of invasive plant management efforts within the spatial scope.

4.3.1 Species Descriptions

Describe the species, or species groups, that are the focus of the Plan. These can include current invasive plant species or species that have the potential to occur in the future (early detection). A *species description* (also known as a *species account*) is basically a written summary of a species, or group of similar species, and includes the following information:

- Plant ecology
    - Plant life cycle: annual, perennial, biennial
    - Growth form: herb, shrub, tree, vine, aquatic
    - Reproduction
    - Seed longevity, dispersal distance
    - Phenology such as blooming time and best time for detection
    - Habitat
    - Dispersal mode(s)
    - Spread rates
- History of management
- Current status within the scope and/or the larger landscape, including data and maps if available
- Impacts on natural resources, ecological processes, or human infrastructure: current or potential future
- Visuals such as photos

There is a wealth of information available online to help describe invasive plant species ecology, known impacts on wildlands or agriculture, and management. A few freely available online resources are highlighted below and others can be found in appendix A:

- Global Invasive Species Database (*http://www.iucngisd.org/gisd/*)
- Invasive.org (*www.invasive.org*)
- National Association of Invasive Plant Councils (*www.na-ipc.org*). This site provides links to invasive plant councils and weed management areas throughout the United States, each of which can provide useful species specific information. Example: Cal-IPC maintains a detailed database of the state's top invasive plant species (*http://www.cal-ipc.org/plants/inventory/*)
- USDA National Agricultural Library (*https://www.invasivespeciesinfo.gov/plants/main.shtml*)
- USDA PLANTS Database (*https://plants.usda.gov/java/*).
- Invasive Plant Atlas of New England (*https://www.eddmaps.org/ipane/*)
- Weed Research and Information Center (*http://wric.ucdavis.edu*)

It is always a good idea to consult with local weed experts, weed management areas, or invasive species councils to identify local or region-specific resources (such as books and scientific papers). Appendix A points to several other resources, and appendix B provides a list of Plans with examples.

4.3.2 Area Descriptions

If distinct management areas have been defined for the Plan scope, provide a map showing these areas with a brief description. Types of information to consider include:

- Plant communities or ecosystems
- Sensitive resources
- Abiotic features such as hydrology, soils, or topography
- Size
- Invasive plant status: the degree to which the area is invaded by one or more invasive plant species
- Vectors or vector pathways, roadway locations and types
- Level of anthropogenic disturbance
- Maps showing area boundaries and other environmental features of importance

4.4 Objectives, Strategies, and Activities

This section of a Plan is where (1) SMART invasive plant management objectives or overall vegetation management objectives are presented and (2) strategies and associated activities to help achieve them are described in enough detail to be useful for the intended audience. Appendix B presents several Plans from a variety of agencies, providing ideas on how to craft this element of your Plan that meets your needs. Below is a list of the types of information to consider including.

- Strategy description—each strategy should be described in enough detail so that people who are expected to implement understand what needs to happen. This can include descriptions of the following:
  - Objective(s) it supports
  - The approach(es) it involves: prevention, containment, control
  - Techniques/tactics it involves such as education, research, assessments, chemical/physical/biological/cultural control
  - Where it will be implemented
  - When (years, seasonality) or how frequently it will be implemented
  - Specific activities to be implemented
  - Who will be involved with implementation
  - Training or certifications required
  - Equipment and supplies needed
  - Expected costs

Strategies can be presented in table form by species and then areas or by distinct areas.

4.5 Measures to Avoid Non-Target Effects

Most invasive plant management programs employ BMPs internally to minimize the non-target effects of their activities, but these may not be formally documented. This section provides a place to summarize the potential non-target effects of your invasive plant management activities and measures or BMPs to avoid or mitigate them. BMPs may be presented as a checklist for specific management strategies or activities and included as an appendix to your Plan to be used in the field. This section may also cite laws or policies applicable to your situation.

4.6 Work Planning and Reporting

This section of your Plan should provide enough detail for the people or organizations who must carry out the Plan. Information to include is listed below:

- A multi-year timeline for activities and surveys
- Expected annual costs
- Timing of management activities (relative to phenology of target plants and other applicable factors)
- Roles: generally who is involved in carrying out activities and surveys
- How annual evaluation and work planning will happen
- Reporting (if needed): content, format, frequency, storage, and sharing

Because annual work planning is dynamic, it can be helpful to use spreadsheets or some other data system to handle changes through time following development of the initial Plan.

4.7 Monitoring and Evaluation Methods

The monitoring and evaluation portion of your Plan should contain information about what types of surveys are needed to inform you work, links to Plan objectives or activities they support, expected frequency, and information on how they will be carried out (protocols). If a protocol exists, they can be included as an appendix or cited. If protocol development is needed, specify when and how a protocol will be developed.

This section can also include information about software or data system(s) that will be used to manage invasive plant data (spatial and non-spatial) as well as how information (files) will be organized and stored.

Chapter 5
Adapting Your Plan

New Zealand spinach
Tetragonia tetragonioides
CREDIT: ©Jean Pawek

It is important to remember that a Plan is not static—it should set the stage for a dynamic and flexible process of doing, evaluating, learning, and adapting. To be successful, any conservation program or project must evaluate progress and adjust to improve outcomes. This adaptive management process should ideally be built into your Plan. For instance, your Plan may specify that every 5 years your organization will revisit objectives, strategies, and other key provisions of the Plan. Additional revisions may be dictated by external forces. And the development of annual workplans will necessarily incorporate lessons learned from the previous year's experiences. The key is to provide a mechanism to periodically re-examine assumptions as well as implementation effectiveness.

A successful plan must be based on both sound project assumptions and good implementation. An adaptive management approach helps teams plan their projects such that they will be able to trace their failures back to poor assumptions, poor implementation, or a combination of the two (Salafsky et al. 2001). Otherwise, when projects do not produce desired results, the conclusion is often that strategies were not implemented as planned or the project team did not do a good job with implementation. In some cases, the same strategy may be implemented year after year without anyone really questioning whether it is achieving the intended result.

The intention of this Guide is to promote a more adaptive approach to invasive plant management, regardless of the organization or agency involved, scale, environment, or socio-political environment. We expect that new information on how to improve the practice of invasive plant management will continue to grow. We encourage you to continue to explore new and improved invasive plant management techniques and practices and to share what you learn with the larger conservation community.

Adaptive management is a structured process that promotes flexible, informed decisions that allow us to make adjustments as we better understand outcomes from management actions and other events. Careful monitoring of these outcomes both advances scientific understanding and helps adjust policies or operations as part of an iterative learning process (USFWS 2013).

This page intentionally left blank.

Glossary

action: an activity designed to apply a particular strategy to a specific situation in order to help achieve an objective. Also called a *tactic*.

adaptive management: a structured process that promotes flexible, informed decisions that allow us to make adjustments as we better understand outcomes from management actions and other events. Careful monitoring of these outcomes both advances scientific understanding and helps adjust policies or operations as part of an iterative learning process (USFWS 2013).

alien: with respect to a particular ecosystem, an organism, including its seeds, eggs, spores, or other biological material capable of propagating that species, that occurs outside of its natural range (Executive Order 13751 [2016]). Considered synonymous with *exotic* and *non-native*, the latter of which is used in this Guide.

aquatic nuisance species: a nonindigenous species that threatens the diversity or abundance of native species or the ecological stability of infested waters, or commercial, agricultural, aquacultural, or recreational activities dependent on such waters (Nonindigenous Aquatic Nuisance Prevention and Control Act [1990]).

asset-based protection: a strategy in which control activities for a widespread invasive species is focused on those areas where the control protects high-priority conservation assets.

best management practices (BMPs): methods or techniques found to be the most effective and practical in achieving an objective, such as preventing or reducing invasive plant spread, while making optimal use of resources (Cal-IPC 2012).

conservation target: the focus of conservation within a specified area. Conservation targets may be biological in nature (species, communities, or ecosystems) or reflect human well-being (such as culture, recreation, infrastructure, or safety). Often, a limited number of conservation targets are identified to collectively represent the full suite of biodiversity or values within a specified area (Foundations of Success 2009).

containment: actions taken to prevent establishment and reproduction of an invasive plant species beyond a predefined area or the *containment unit*. The containment unit comprises the area where the species currently exists (occupied zone) plus a surrounding buffer zone that is free from plants but can receive propagules (such as seeds) (Panetta and Cacho 2014).

control: the act of reducing the occurrence or abundance of invasive plants using one or more integrated pest management techniques (such as chemical, biological, mechanical removal techniques).

drone: An aerial machine that can be used for remote mapping. Also known as *unmanned aerial vehicle (UAV)* or *unmanned aerial system (UAS)*.

early detection: a type of survey focused on detecting the location and abundance of highly invasive species that are not yet established within a defined area (but the potential for establishment exists) or occur in small isolated populations within a defined spatial scope (Olsen et al. 2015). A process of surveying for, reporting, and verifying the presence of a non-native species before the founding population becomes established or spreads so widely that eradication is no longer feasible (U.S. Department of the Interior 2016).

eradication: the complete removal of an invasive plant species (including reproductive propagules) from a defined area.

integrated pest management (IPM): a science-based decision-making process that incorporates management goals, consensus building, pest biology, monitoring, environmental factors, and selection of the best available technology to achieve desired outcomes while minimizing effects on non-target species and the environment and preventing unacceptable levels of pest damage (USFWS 2010).

indigenous: see *native species*.

introduced: see *alien*.

invasive species: a non-native organism whose introduction causes or is likely to cause economic or environmental harm, or harm to human, animal, or plant health (Executive Order 13751 [2016]).

inventory: a type of survey that is used to determine the location or condition of a resource (e.g., presence, abundance, distribution, status) at a specific time. Inventories may also establish a beginning time-step (baseline) or reference information for subsequent monitoring (USFWS 2013). In this Guide, an *inventory* refers to a catalogue of invasive species that can include information on their location, abundance, and distribution in a defined region.

monitoring: consists of repeated survey efforts and is more complex than inventories because it is conducted to understand how resources vary over time (e.g., months to years) and space. *Baseline monitoring* can be used to produce a time series of indicators such as water salinity or fish survival. Results from this type of monitoring can be used to assess changes in a system or to develop models of system function. *Monitoring to inform management* is the other type of monitoring for which a survey protocol is developed and has the additional purpose of directly influencing a management decision. This form of monitoring may be used to evalute model values and performance in adaptive management projects or used to identify effects on trends in attributes produced by quasi-experiments (USFWS 2013).

native nuisance species: a native species that causes harm to the environment or human health.

native species: with respect to a particular ecosystem, a species that, other than as a result of an introduction, historically occurred or currently occurs in that ecosystem (Executive Order 13112 [1999]).

non-native species: see *alien*.

noxious weed: any plant or plant product that can directly or indirectly injure or cause damage to crops (including nursery stock or plant products), livestock, poultry, or other interests of agriculture, irrigation, navigation, the natural resources of the United States, the public health, or the environment (Public Law 106 – 224 [2000]).

objective: a concise statement of desired outcomes that specifies what we want to achieve, how much we want to achieve, when and where we want to achieve it, and who is responsible for achieving it. A meaningful objective will be SMART specific, measurable, achievable, results-oriented, and time-bound (USFWS 2013).

prevention: the act of preventing the introduction and spread (transmission) of invasive species. Also referred to as *biosecurity*.

pest: organisms that damage or interfere with desirable plants in our fields and orchards, landscapes, or wildlands, or damage homes or other structures. Pests also include organisms that impact human or animal health (UC-IPM 2018).

protocol: detailed instructions for conducting a survey. This includes information on sampling procedures, data collection, management and analysis, and reporting of results (USFWS 2013).

strategy: a group of actions with a common focus that work together to reduce threats, capitalize on opportunities, or restore conservation targets. Strategies include one or more activities and are designed to achieve specific objectives and goals (Foundations of Success 2009).

survey: a specific data-collection effort to complete an inventory or conduct monitoring of biotic or abiotic resources (USFWS 2013).

vector (or transport vector): the conveyance (e.g., wind, water, animal, human, mechanical, etc.) that moves a non-native propagule to its novel location (Lockwood et al. 2013).

vector pathway (or transport pathway): the route between the non-native propagule source and release location (Lockwood et al. 2013).

weed: a plant that causes economic losses or ecological damage, creates health problems for humans or animals, or is undesirable where it is growing (Weed Society Science of America 2016).

References

Agriculture Victoria. 2002. *Invasive Plants and Animals Policy Framework.* Available <http://agriculture.vic.gov.au/agriculture/pests-diseases-and-weeds/protecting-victoria-from-pest-animals-and-weeds/invasive-plants-and-animals/invasive-plants-and-animals-policy-framework>; accessed October 8, 2018.

Andersen, K.A., and S.A. Dewey. 2007. *USU Wildland Weed Mapping Methods Training Supplement.* Master's Thesis. Utah State University. Logan, UT. 107 pp.

Ball, M., and Olthof, K. 2017. *Invasive Plant Inventory Report: Kern National Wildlife Refuge.* Unpublished Report. Wildlands Conservation Science. Lompoc, CA. 42 pp.

Bella, E. n.d. *Elodea in Southcentral Alaska: Early Detection and Rapid Response in Action.* U.S. Fish and Wildlife Service, Kenai National Wildlife Refuge. Available: <https://www.fws.gov/uploadedFiles/Bella_E_2015b.pdf>; accessed October 8, 2018.

Bossard, C.C., J.M. Randall, and M.C. Hoshovsky (eds.). 2000. *Invasive Plants of California's Wildlands.* First edition. University of California Press. Berkeley, CA.

Bradley, B.A. 2014. Remote detection of invasive plants: a review of spectral, textural and phenological approaches. *Biological Invasions* 16, 1411–1425. DOI: https://doi.org/10.1007/s10530-013-0578-9.

California Invasive Plant Council [Cal-IPC]. 2015. *Best Management Practices for Wildland Stewardship: Protecting Wildlife When Using Herbicides for Invasive Plant Management.* No. Cal-IPC Publication 2015-1. Berkeley, CA.

Cal-IPC. 2012. *Preventing the Spread of Invasive Plants: Best Management Practices for Land Managers.* No. Cal-IPC Publication 2012-03. California Invasive Plant Council. Berkeley, CA.

Coble, Harold D., and J. Schroeder. 2016. Call to action on herbicide resistance management. *Weed Science* 64(1):661–666.

Cusack, C., M. Harte, and S. Chan. 2009. *The Economics of Invasive Species.* Sea Grant Oregon.

Darin, G.S. 2008. *Prioritizing Weed Populations for Eradication at a Regional Level: The California Department of Food and Agriculture's A-rated Weeds.* Master's Thesis. University of California Davis. Davis, CA.

DiSalvo, C., and T. Parson. 2011. Integrated pest management in the National Park Service and the U.S. Fish and Wildlife Service. *Outlooks on Pest Management* 22:171–176.

DiTomaso, J. 2000. Invasive weeds in rangelands: species, impacts, and management. *Weed Science* 48:255–265.

DiTomaso, J.M., G.B. Kyser, et al. 2013. *Weed Control in Natural Areas in the Western United States.* University of California Weed Research and Information Center. Davis, CA.

Elzinga, C.L., D.W. Salzer, and J.W. Willoughby. 1998. *Measuring and Monitoring Plant Populations.* BLM Technical Reference 1730-1; BLM/RS/ST-98/005+1730.

Fletcher, C.S., D.A. Westcott, H.T. Murphy, A.C. Grice, and J.R. Clarkson. 2015. Managing breaches of containment and eradication of invasive plant populations. *Journal of Applied Ecology* 52:59–68.

Flint, M.L., and P. Gouveia. 2014. *IPM in Practice: Principles and Methods of Integrated Pest Management.* Second edition. Agriculture and Natural Resources Communication Services, University of California.

Foundations of Success. 2009. *Conceptualizing and Planning Conservation Projects and Programs: A Training Manual.* Bethesda, MD.

Foxcroft, L.C., P. Pysek, D.M. Richardson, P. Genovesi, and S. MacFadyen. 2017. Plant invasion science in protected areas: progress and priorities. *Biological Invasions* 19:1,353–1,378. DOI: https://doi.org/10.1007/s10530-016-1367-z

Hiebert, R.D., and J. Stubbendieck. 1993. *Handbook for Ranking Exotic Plants for Management and Control*. No. Natural Resources Report NPS/NRMWRO/NRR-93/08. National Park Service. Denver, CO.

Hobbs, R.J., and S.E. Humphries. 1995. An integrated approach to the ecology and management of plant invasions. *Conservation Biology* 9:761–770. DOI: https://doi.org/10.1046/j.1523-1739.1995.09040761.x

Holzman, B.A., Q.J. Clark, G.J. McChesney, and G. Block. 2016. *Farallon Islands 2016 Invasive Plant Inventory*. Unpublished report, San Francisco State University, San Francisco, CA, and U.S. Fish and Wildlife Service, Fremont, CA.

Huang, C., and G. Asner. 2009. *Applications of Remote Sensing to Alien Invasive Plant Studies*. Sensors 9:4,869–4,889. DOI: https://doi.org/10.3390/s90604869

International Survey of Herbicide Resistant Weeds. 2018. [Website.] Available: <http://www.weedscience.org/>; accessed December 7, 2018.

IUCN. 2018. *Guidelines for Invasive Species Planning and Management on Islands*. Cambridge, UK and Gland, Switzerland. 40 pp.

Lass, L.W., T.S. Prather, N.F. Glenn, K.T. Weber, J.T. Mundt, and J. Pettingill. 2005. A review of remote sensing of invasive weeds and example of the early detection of spotted knapweed (*Centaurea maculosa*) and babysbreath (*Gypsophila paniculata*) with a hyperspectral sensor. *Weed Science* 53:242–251. DOI: https://doi.org/10.1614/WS-04-044R2

Lockwood, J.L., M.F. Hoopes, and M.P. Marchetti. 2013. *Invasion Ecology*. Second edition. Wiley-Blackwell, Chichester. West Sussex, UK.

Madden, M. 2004. Remote sensing and geographic information system operations for vegetation mapping of invasive exotics. *Weed Technology* 18:1457–1463. DOI: https://doi.org/10.1614/0890-037X(2004)018[1457:RSAGIS]2.0.CO;2

Morse, L., J.M. Randall, N. Benton, R.D. Hiebert, and S. Lu. 2004. *An Invasive Species Assessment Protocol: Evaluating Non-Native Plants for Their Impact on Biodiversity*. Version 1. NatureServe. Arlington, VA.

National Park Service. 2017. *Invasive Plant Management Plan and Environmental Assessment for Redwood National Park and Santa Monica Mountains National Recreation Area*. Available: <https://parkplanning.nps.gov/document.cfm?parkID=341&projectID=44351&documentID=83505>; accessed October 8, 2018.

National Park Service. 2012. *Guidance for Designing an Integrated Monitoring Program*. No. Natural Resource Report NPS/NRSS/NRR—545. Fort Collins, CO.

Oakley, K., L. Thomas, and S. Fancy. 2003. Guidelines for long-term monitoring protocols. *Wildlife Society Bulletin* 31:1000–1003.

Olkowski, W., and H. Olkowski. 1983. *Integrated Pest Management for Park Managers: A Training Manual*. National Park Service. Napa, CA.

Olsen, H.E., G. Block, and C. Ransom. 2015. *Invasive Plant Inventory and Early Detection Prioritization Tool*. Version 1.0. U.S. Fish and Wildlife Service and Utah State University.

Panetta, F.D., and O.J. Cacho. 2014. Designing weed containment strategies: An approach based on feasibilities of eradication and containment. *Diversity and Distributions* 20: 555–566.

Pearson, D.E., Y.K. Ortega, J.B. Runyon, and J.L. Butler. 2016. Secondary invasion: The bane of weed management. *Biological Conservation* 197:8–17. DOI: https://doi.org/10.1016/j.biocon.2016.02.029

Randall, J.M. 2000. Improving management of nonnative invasive plants in wilderness and other natural areas. USDA Forest Service Proceedings RMRS-P-15-VOL-5. 2000. pp. 64–73.

Randall, J.M., L.E. Morse, N. Benton, R. Hiebert, S. Lu, and T. Killeffer. 2008. The invasive species assessment protocol: A tool for creating regional and national lists of invasive nonnative plants that negatively impact biodiversity. *Invasive Plant Science and Management* 1:36–49. https://doi.org/10.1614/IPSM-07-020.1

Rejmanek, M., and M. Pitcairn. 2002. When is eradication of exotic pest plants a realistic goal? In: *Turning the Tide: The Eradication of Invasive Species., Occasional Paper of the IUCN Species Survival Commission No. 27*. Presented at the International Conference on Eradication of Island Invasives, IUCN Switzerland, p. 414.

Rew, L.J., and M.L. Pokorny. 2006. *Inventory and Survey Methods for Nonindigenous Plant Species*. Montana State University Extension. Bozeman, MT.
Salafsky, N., R. Margoulis, and K. Redford. 2001. *Adaptive Management: A Tool for Conservation Practitioners*. Biodiversity Support Program. Washington, D.C.
Simberloff, D. 2003. Eradication: Preventing invasions at the outset. *Weed Science* 51:247–253.
Skurka Darin, G.M., S. Schoenig, J.N. Barney, F.D. Panetta, and J.M. DiTomaso. 2011. WHIPPET: A novel tool for prioritizing invasive plant populations for regional eradication. *Journal of Environmental Management* 92:131–139. DOI: https://doi.org/10.1016/j.jenvman.2010.08.013
Stohlgren, T.J., and J.L. Schnase. 2006. Risk Analysis for biological hazards: What we need to know about invasive species. *Risk Analysis* 26:163–173. https://doi.org/10.1111/j.1539-6924.2006.00707.x
Tu, M., and B. Meyers-Rice. 2002. *Site Weed Management Plan Template*. The Nature Conservancy.
University of California Statewide IPM Program [UC-IPM]. 2018. *What is Integrated Pest Management (IPM)?* Available: <http://www2.ipm.ucanr.edu/WhatIsIPM/>; accessed October 8, 2018.
USA Herbicide Resistance Action Committee. 2018. Best Management Practices. Available: <http://hracglobal.com/usa/prevention-management/best-management-practices>; accessed October 10, 2018.
U.S. Department of the Interior. 2016. *Safeguarding America's Lands and Waters from Invasive Species: A National Framework for Early Detection and Rapid Response*. U.S. Department of the Interior. Washington, D.C.
USDA Forest Service. 2001. *Guide to Noxious Weed Prevention Practices*. Washington, D.C.
U.S. Fish and Wildlife Service [USFWS]. In prep. *Invasive Plant Inventory and Early Detection Guide*. Portland, OR.
USFWS. 2018. *Invasive Plant Workshop: Lower Klamath and Tule Lake National Wildlife Refuges*. U.S. Fish and Wildlife Service, Pacific Southwest Region, National Wildlife Refuge System, Sacramento, CA.
USFWS. 2016. *U.S. Fish and Wildlife Service Pacific Region Policy on Minimizing the Introduction of Invasive Species by Service Activities*. Portland, OR.
USFWS. 2013. *How to Develop Survey Protocols: A Handbook*. Version 1.0. National Wildlife Refuge System, Natural Resource Program Center. Fort Collins, CO.
USFWS. 2010. *Integrated Pest Management Policy*. No. U.S. Fish and Wildlife Service Policy 569 FW 1.
USFWS. 2004. *Integrated Pest Management: Guidance for Preparing and Implementing Integrated Pest Management Plans*. 12 pp.
USFWS and Utah State University. 2018. *Invasive Plant Inventory and Early Detection Prioritization Tool: A Users Guide*. Version 4.0. February. USFWS, National Wildlife Refuge System, Pacific Southwest Region, Inventory and Monitoring Program, Sacramento, CA. 136 pp.
Warner, P., C. Bossard, C., et al. 2003. *Criteria for Categorizing Invasive Non-Native Plants that Threaten Wildlands*. California Invasive Plant Council and Southwest Vegetation Management Association. Berkeley, CA.
Weed Science Society of America. 2016. Weed definition. WSSA Fact Sheet. Available: <http://wssa.net/wp-content/uploads/WSSA-Weed-Science-Definitions.pdf>; accessed December 11, 2018.
Pickett, S.T., and P.S. White. 1985. *The Ecology of Natural Disturbance and Patch Dynamics*. Academic Press. Orlando, FL. 472 pp.
Williams, A., S. O'Neil, E. Speith, and J. Rodgers. 2009. *Early Detection of Invasive Plant Species in the San Francisco Bay Area Network: A Volunteer-Based Approach*. No. Natural Resource Report NPS/SFAN/NRR—2009/136. National Park Service. Fort Collins, CO.
Zarnetske, P.L., E.W. Seabloom, and S.D. Hacker. 2010. Non-target effects of invasive species management: beachgrass, birds, and bulldozers in coastal dunes. *Ecosphere* 1(5):Article 13.

This page intentionally left blank.

Appendix A

Invasive Plant Information: Online Resources

Below is an alphabetized list of online invasive species information resources. There are many more resources than we could ever list here. We chose to highlight a few of the most resources—many of them point to species or location-specific resources. The U.S. Department of Agriculture (USDA) National Invasive Species Information Center maintains a list of invasive species resources by state (*https://www.invasivespeciesinfo.gov/resources/orgstate.shtml*) as well as resources by species (*https://www.invasivespeciesinfo.gov/plants/main.shtml*). We encourage users to seek out additional local or regional resources.

Center for Invasive Plant Management (CIPM) (*www.weedcenter.org*). Though no longer funded, the CIPM remains a useful resource for information about invasive plant biology, management, and education and outreach. The site provides numerous links to other web-based sources of invasive plant-related information across the United States.

Center for Invasive Species and Ecosystem Health (CISEH) (*https://www.bugwood.org/*). The mission of the CISEH is to serve a lead role in development, consolidation, and dissemination of information and programs focused on invasive species, forest health, natural resource, and agricultural management through technology development, program implementation, training, applied research, and public awareness at the state, regional, national and international levels. The site hosts a database of imagery, provides links to publications on invasive species management, and lists websites related to invasive plant management across the United States.

Invasive.org (*www.invasive.org*). Run by the Center for Invasive Species and Ecosystem Health at the University of Georgia, this site provides a wealth of information including an easily accessible archive of high quality images of invasive and exotic species of North America with identifications, taxonomy, and descriptions for use in educational applications and species-specific control information.

National Association of Invasive Plant Councils (NAIPC) (*www.na-ipc.org*). NAIPC comprises state and multi-state organizations that coordinate invasive plant managers and information. Each entity typically maintains an invasive plant list and holds an annual conference. The site provides links to state invasive plant councils.

National Invasive Species Council (NISC) (*www.invasivespecies.gov*). The NISC was established to ensure that federal programs and activities to prevent and control invasive species are coordinated, effective, and efficient. The national invasive species management plan can be found on this site.

New York Invasive Species Research Institute (*http://www.nyisri.org/*). To improve the scientific basis of invasive species management, the New York Invasive Species Research Institute serves the scientific research community, natural resource and land managers, and state offices and sponsored organizations by promoting information-sharing and developing recommendations and implementation protocols for research, funding, and management.

North American Invasive Species Management Association (NAISMA) (*https://www.naisma.org/*). NAISMA is a network of professionals—land managers, water resource managers, state, regional, and federal agency directers and staff, and nonprofit organizations—challenged by invasive species. This website lists standards (weed-free forage and gravel, mapping), invasive plant management online training, and a variety of other resources useful to managers.

USDA Forest Service Invasive Species Program (*www.fs.fed.us/invasivespecies*). This site links to the agency's policy framework for invasive species as well as its management activities, with information on research, management planning, and pest-specific control techniques.

USDA National Invasive Species Information Center (*https://www.invasivespeciesinfo.gov/index.shtml*). This is a gateway to invasive species information covering federal, state, local, and international sources. The resource library provides links to many of the sites listed in this appendix plus many more resources for managers.

USDA PLANTS Database (*https://plants.usda.gov/java/*). The PLANTS Database provides standardized information about the vascular plants, mosses, liverworts, hornworts, and lichens of the United States and its territories. It includes names, plant symbols, checklists, distributional data, species abstracts, characteristics, images, crop information, automated tools, onward web links, and references. It also includes links to federal and state noxious weed lists.

U.S. Fish and Wildlife Service: Invasive Species (*www.fws.gov/invasives*). The website provides background on a range of invasive species topics and points to a variety of resources for land managers.

Weed Research and Information Center (*http://wric.ucdavis.edu*). The Weed Research and Information Center is an interdisciplinary collaboration that fosters research in weed management and facilitates distribution of associated knowledge for the benefit of agriculture and for the preservation of natural resources. This is an excellent resource for control techniques by weed species.

Weed Science Society of America (WSSA) (*http://wssa.net*). The WSSA is a non-profit professional society that promotes research, education, and extension outreach activities related to weeds; provides science-based information to the public and policy-makers; and fosters awareness of weeds and their impacts on managed and natural ecosystems. WSSA publishes three professional journals: *Weed Science*, *Weed Technology*, and *Invasive Plant Science and Management*. The website provides a variety of resources—including invasive plant images, identification resources, and a list of resources for biological control—and covers the topic of weed resistance.

Appendix B

Examples: Plans, Reports, and Protocols

The tables below list invasive plant management plans, inventory or monitoring protocols, and other related guidance documents and the topical areas they address (designated by an "X"). Full citations and web links are provided at the end of this appendix.

Invasive Plant Management Planning Documents

Author, date, and title	*Species prioritization*	*Area prioritization*	*Area or species descriptions*	*SMART objectives or thresholds for action*	*Species or area specific strategies*	*Prevention*	*Inventory or monitoring*	*Work planning*	*BMPs to avoid non-target effects*
Dendra (2012). *Management Priorities for Invasive Non-native Plants: A Strategy for Regional Implementation, San Diego County, California.*	X			X	X				
Evans et al. (2008). *Invasive Plant Species Inventory and Management Plan for the Hanford Reach National Monument.*	X	X	X		X		X	X	
Hall (2015). *Integrated Vegetation Management Plan for Open Space Lands of the City of San Luis Obispo.*	X		X	X	X		X		X
Hogle et al. (2007). *San Pablo Bay National Wildlife Refuge* Lepidium latifolium *Control Plan.*		X	X		X		X		X
Marriott et al. (2013). *South San Francisco Bay Weed Management Plan.*	X		X		X	X			X
May and Associates (2015). *Vegetation and Biodiversity Management Plan: Marin County Parks and Open Space District.*	X	X	X		X	X	X	X	X

Author, date, and title	*Species prioritization*	*Area prioritization*	*Area or species descriptions*	*SMART objectives or thresholds for action*	*Species or area specific strategies*	*Prevention*	*Inventory or monitoring*	*Work planning*	*BMPs to avoid non-target effects*
Midpeninsula Regional Open Space District (2014). *Midpeninsula Region Open Space District Integrated Pest Management Program Guidance Manual.*					X	X	X		
National Park Service (2003). *Rocky Mountain National Park Invasive Exotic Plant Management Plan and Environmental Assessment.*	X	X	X	X			X		
National Park Service (2008). *Lassen Volcanic National Park Weed Management Plan and Environmental Assessment.*	X		X	X	X		X		X
National Park Service (2010). *Yosemite National Park Invasive Plant Management Plan Update Environmental Assessment.*	X		X	X	X	X			X
National Park Service (2018). *Yosemite Invasive Plant Management Program 2018 Work Plan.*	X				X	X	X	X	
National Park Service (2017). *Invasive Plant Management Plan and Environmental Assessment for Redwood National Park and Santa Monica Mountains National Recreation Area.*	X	X	X			X			X

Author, date, and title	*Species prioritization*	*Area prioritization*	*Area or species descriptions*	*SMART objectives or thresholds for action*	*Species or area specific strategies*	*Prevention*	*Inventory or monitoring*	*Work planning*	*BMPs to avoid non-target effects*
Shelterbelt Builders and MIG/TRA Environmental Sciences (2016). *Integrated pest management plan for the Bear Creek Redwoods Open Space Preserve.*	X				X			X	
U.S. Fish and Wildlife Service (2012). *Integrated Pest Management Plan for Chesapeake Marshlands National Wildlife Refuge Complex.*			X		X		X		

Notes: species or area prioritization = reference uses multiple criteria used to prioritize species or areas; SMART objectives = reference contains objectives that are focused on vegetation and are specific, measurable, achievable, results-oriented, and time-bound; prevention = reference identifies specific prevention practices or activities; inventory or monitoring = reference has an inventory or monitoring element; work planning: reference contains one or more elements that will inform implementation, such as specific tasks and when they will be carried out, costs, how new activities or projects will be evaluated, and who will implement the work.

Examples of invasive plant prioritization reports and survey protocols.

Author, date, and title	*Species prioritization*	*Area prioritization*	*SMART objectives*
Ball and Olthof (2017). *Aerial Invasive Plant Survey: Guadalupe-Nipomo Dunes National Wildlife Refuge.*	X	X	
Holzman et al. (2016). *Farallon National Wildlife Refuge Southeast and West End Islands 2016 Invasive Plant Inventory.*	X	X	X
Keefer et al. (2014). Early Detection of Invasive Species Surveillance, Monitoring, and Rapid Response: Version 2.0.	X		X
Rew and Pokorny (2006). *Inventory and Survey Methods for Nonindigenous Plant Species.*	X		X
Williams et al. (2009). *Early Detection of Invasive Plant Species in the San Francisco Bay Area Network: A Volunteer-Based Approach.*	X	X	X

Notes: species or area prioritization = reference uses multiple criteria used to prioritize species or areas; SMART objectives = reference contains objectives that are focused on vegetation and are specific, measurable, achievable, results-oriented, and time-bound; prevention = reference identifies specific prevention practices or activities.

Literature Cited

Ball, M., and K. Olthof. 2017. *Aerial Invasive Plant Survey: Guadalupe-Nipomo Dunes National Wildlife Refuge.* Unpublished report. Wildlands Conservation Science, Lompoc, CA. Available: <https://catalog.data.gov/dataset/aerial-invasive-plant-survey-guadalupe-nipomo-dunes-national-wildlife-refuge>; accessed October 8, 2018.

Dendra. 2012. *Management Priorities for Invasive Non-native Plants: A Strategy for Regional Implementation, San Diego County, California.* Available: <https://www.sandiegocounty.gov/content/dam/sdc/awm/docs/CBI_Strategic%20Plan9-10-12s.pdf>; accessed December 11, 2018.

Evans, J., J. Nugent, J. Meisel. 2003. *Invasive Plant Species Inventory and Management Plant for the Hanford Reach National Monument.* The Nature Conservancy. Seattle, WA. Available: <https://www.fws.gov/uploadedFiles/Region 1/NWRS/Zone 2/Mid-Columbia_River Complex/Hanford Reach National Monument/Documents/weed-plan.pdf>; accessed October 8, 2018.

Hall, J. 2015. *Integrated Vegetation Management Plan for Open Space Lands of the City of San Luis Obispo.* Land Conservancy of San Luis Obispo, San Luis Obispo, CA. Available: <https://www.slocity.org/home/showdocument?id=8611>; accessed December 11, 2018.

Hogle, I., R. Spenst, S. Leininger, and G. Block. 2007. *San Pablo Bay National Wildlife Refuge* Lepidium latifolium *Control Plan.* U.S. Fish and Wildlife Service, San Pablo Bay National Wildlife Refuge. Petaluma, CA. Available <https://www.fws.gov/invasives/staffTrainingModule/pdfs/planning/SPBNWR_Control_Plan_061807.pdf>; accessed October 8, 2018.

Holzman, B., G. Block, and G. McChesney. 2016. *Farallon National Wildlife Refuge Southeast and West End Islands 2016 Invasive Plant Inventory (Invasive plant inventory).* U.S. Fish and Wildlife Service. San Francisco, CA. Available: <https://catalog.data.gov/dataset/farallon-national-wildlife-refuge-southeast-and-west-end-islands-2016-invasive-plant-inven>; accessed October 8, 2018.

Keefer, J.S., J.S. Wheeler, D.R. Manning, M.R. Marshall, B.R. Mitchell, and F. Dieffenbach. 2014. *Early Detection of Invasive Species Surveillance, Monitoring, and Rapid Response.* Version 2.0. No. Natural Resource Report NPS/ERMN/NRR 2014/837. National Park Service, Fort Collins, CO. Available: <https://irma.nps.gov/DataStore/DownloadFile/376529>; accessed December 11, 2018.

Marriott, M., R. Tertes, and C. Strong. 2013. *South San Francisco Bay Weed Management Plan. 1st Edition. Unpublished report of the U.S. Fish and Wildlife Service, Fremont, CA.* 82 pp. Available: <https://www.fws.gov/uploadedFiles/South%20Bay%20Weed%20Management%20Plan %201st_edition_11_20_13.pdf>; accessed December 11, 2018.

May and Associates. 2015. *Vegetation and Biodiversity Management Plan. Marin County Parks and Marin County Open Space District, San Rafael, CA.* Available: <https://www.marincounty.org/-/media/files/departments/pk/projects/open-space/vmbp/2015_05mcpvmbpv9lowresweb.pdf?la=en>; accessed October 8, 2018.

Midpeninsula Regional Open Space District. 2014. *Midpeninsula Open Space District Integrated Pest Management Program Guidance Manual.* Accessed: <https://www.openspace.org/sites/default/files/IPM_Guidance_Manual.pdf>; accessed October 8, 2018.

National Park Service. 2003. *Rocky Mountain National Park Invasive Exotic Plant Management Plan and Environmental Assessment.* National Park Service, Rocky Mountain National Park, CO. Available: <https://www.nps.gov/romo/learn/management/upload/exotic plant ea final.pdf>; accessed December 11, 2018.

National Park Service. 2008. *Lassen Volcanic National Park Weed Management Plan and Environmental Assessment.* National Park Service, Lassen Volcanic National Park. Available: <https://www.nps.gov/lavo/learn/management/upload/LAVO_Weed_Management_Plan_EA.pdf>; accessed December 11, 2018.

National Park Service. 2010. Yosemite *National Park Invasive Plant Management Plan Update Environmental Assessment (EA).* National Park Service, El Portal, CA. Available: <https://www.nps.gov/yose/learn/management/upload/IPMP Update-_Print_Version_12_13_2010_reduced.pdf>; accessed December 11, 2018.

National Park Service. 2017. *Invasive Plant Management Plan and Environmental Assessment for Redwood National Park and Santa Monica Mountains National Recreation Area.* National Park Service. Available: <https://parkplanning.nps.gov/document.cfm?parkID=341&projectID=44351&documentID=83505>; accessed December 11, 2018.

National Park Service. 2018. *Yosemite Invasive Plant Management Program 2018 Work Plan.* National Park Service, El Portal, CA. Available: <https://www.nps.gov/yose/learn/nature/upload/2018-IPM-Work-Plan.pdf>; accessed December 11, 2018.

Rew, L.J., and M.L. Pokorny. 2006. *Inventory and Survey Methods for Nonindigenous Plant Species.* Montana State University Extension. Bozeman, MT. Available: <http://www.weedcenter.org/store/images/books/INVENTORYBOOK.pdf>; accessed October 8, 2018.

Shelterbelt Builders and MIG/TRA Environmental Sciences. 2016. *Integrated Pest Management Plan for the Bear Creek Redwoods Open Space Preserve.* Midpeninsula Regional Open Space District. Available: <https://www.openspace.org/our-work/projects/integrated-pest-managment>; accessed October 8, 2018.

U.S. Fish and Wildlife Service. 2012. *Integrated Pest management Plan for Chesapeake Marshlands National Wildlife Refuge Complex.* Available: <https://catalog.data.gov/dataset/integrated-pest-management-plan-for-chesapeake-marshlands-national-wildlife-refuge-complex>; accessed October 8, 2018.

White, R., C. Nordman, L. Smart, T. Leibfreid, B. Moore, R. Smyth, and T. Govus. 2011. *Vegetation Monitoring Protocol for the Cumberland Piedmont Network.* Version 1. Natural Resource Report NPS/CUPN/NRR—2011/XXX. National Park Service. Fort Collins, CO. Available: <https://irma.nps.gov/DataStore/DownloadFile/515620>; accessed December 11, 2018.

Williams, A., S. O'Neil, E. Speith, and J. Rodgers. 2009. *Early Detection of Invasive Plant Species in the San Francisco Bay Area Network: A Volunteer-Based Approach.* No. Natural Resource Report NPS/SFAN/NRR 2009/136. National Park Service. Fort Collins, CO. Available: <http://www.sfnps.org/download_product/1260/0>; accessed October 8, 2018.

Appendix C

Plan Template

This template provides an outline of the contents that should be considered for inclusion in your Plan and hyperlinks to sections of the Guide where information on that topic is located.

Chapter 1: Introduction (Chapter 2 and Section 4.1)

- Plan Purpose and Need

  Why is this Plan needed?

  Why are invasive plants a concern?

  Who is the intended audience?

- Spatial Scope and Setting

  What is the geographic scope where management activities are prescribed?

- Conservation Assets and Goals

  What are the ecological/environmental characteristics of the scope and associated conservation goals?

- History of Invasive Plant Management

  What is the history of invasive plant management within the scope?

- Regulatory Context

  What are the relevant organizational policies and legislation that apply to invasive plant management within the Plan scope?

Chapter 2: Methods (Chapter 2 and Section 4.2)

- Project Team

  Who coordinated the planning effort and wrote the Plan?

  Who else was involved in the planning process (internal and external)

- Internal and External Communication, Outreach, and Engagement

  What were the methods of communication and engagement during the planning process?

- Information Gathering

  What information was gathered and used to inform the planning process?

- Prioritization of Species and Management Areas

  What methods were used to identify priority species and areas?

- Identifying Management Strategies

  What methods were used to identify and rank alternative management strategies?

Chapter 3: Species and Area Priorities (Sections 3.1, 3.2, and 4.3)

- Species Priorities

  What plant species (one or multiple) are a priority to manage? Include ranked list of species if a prioritization process was conducted

Priority species characteristics? Such as ecology, status within the scope and surrounding areas (abundance/distribution), history of invasion, maps, imagery. Use existing species profiles for basic characteristics (if available)

- Area Priorities

  What areas are a priority to manage? Include ranked list of areas if a prioritization process was conducted

  Priority area characteristics? Such as ecological characteristics, invasion status, history of invasion, maps, imagery.

Chapter 4: Work Plan (Sections 3.3–3.6 and 4.6)

- SMART Invasive Plant Management Objectives

  What would success look like as a result of your invasive plant management program?

- Management Strategies and Activities

  What are the invasive plant strategies and associated activity (or activities)?

  - *When should they be implemented?*
  - *Thresholds for implementation?*
  - *Where will they be implemented?*
  - *Who is responsible for implementation?*
  - *Budget and operational requirements?*
  - *Required training, certification, or permits*

- Best Management Practices for Avoiding Non-Target Effects

  Are there any potential negative effects on humans, natural/cultural resources, or infrastructure because of invasive plant management activities?

  What measures will be implemented to prevent, avoid, or mitigate potential negative impacts?

Chapter 5: Monitoring and Evaluation (Sections 3.7 and 4.7)

- Monitoring and Evaluation

  What methods will be used to evaluate progress in implementing strategies and achieving SMART objectives?

  When and how should progress on implementing strategies and achieving objectives be evaluated?

- Adaptation

  How will monitoring and evaluation used to revise the work plan?

  How often should the work plan be evaluated and by whom?

- Data Management

  What standards or systems will be used to manage invasive plant program data?

USGS
science for a changing world

Threats to Desert Tortoise Populations: A Critical Review of the Literature

Prepared for:

West Mojave Planning Team, Bureau of Land Management

U.S. DEPARTMENT OF THE INTERIOR
U.S. GEOLOGICAL SURVEY
WESTERN ECOLOGICAL RESEARCH CENTER

Threats to Desert Tortoise Populations: A Critical Review of the Literature

By William I. Boarman, Ph.D.[1]

U.S. GEOLOGICAL SURVEY
WESTERN ECOLOGICAL RESEARCH CENTER

Prepared for:

West Mojave Planning Team
Bureau of Land management

[1]San Diego Field Station
USGS Western Ecological Research Center
5745 Kearny Villa Road, Suite M
San Diego, CA 92123

Sacramento, California
Aug 9, 2002

U.S. DEPARTMENT OF THE INTERIOR
GALE A. NORTON, SECRETARY

U.S. GEOLOGICAL SURVEY
Charles G. Groat, Director

For additional information, contact:

Center Director
Western Ecological Research Center
U.S. Geological Survey
7801 Folsom Blvd., Suite 101
Sacramento, CA 95826

TABLE OF CONTENTS

INTRODUCTION

Decisions in resource management are generally based on a combination of sociopolitical, economic, and environmental factors, and may be biased by personal values. These three components often contradict each other resulting in controversy. Controversies can usually be reduced when solid scientific evidence is used to support or refute a decision. However, it is important to recognize that data often do little to alter antagonists' positions when differences in values are the basis of the dispute. But, supporting data can make the decision more defensible, both legally and ethically, especially if the data supporting all opposing viewpoints are included in the decision-making process.

Resource management decisions must be made using the best scientific information currently available. However, scientific data vary in two important measures of quality: reliability and validity. The reliability of the data is a measure of the degree to which the observations or conclusions can be repeated. Validity of the data is a measure of the degree to which the observation or conclusion reflects what actually occurs in nature. How the data are collected strongly affects the reliability and validity of ecological conclusions that can be made. Research data potentially relevant to management come from different sources, and the source often provides clues to the reliability and, to a certain extent, validity of data. Understanding the quality of data being used to make management decisions helps to separate the philosophical or value-based aspects of arguments from the objective ones, thus helping to clarify the decisions and judgements that need to be made.

The West Mojave Plan is a multispecies, bioregional plan for the management of natural resources within a 9.4 million-acre area of the Mojave Desert in California. The plan addresses the legal requirements for the recovery of the desert tortoise (*Gopherus agassizii*), a threatened species, but also covers an additional approximately 80 species of plants and animals assigned special status by the Bureau of Land Management, U. S. Fish and Wildlife Service, and California Department of Fish and Game. Within the planning area, 28 separate jurisdictions (counties, cities, towns, military installations, etc.) seek programmatic prescriptions that will facilitate stream-lined environmental review, result in expedited authorization for development projects, and protect listed and unlisted species into the foreseeable future to avoid or minimize conflicts between proposed development and species' conservation and recovery. All of the scientific data available concerning the biology and management of these approximately 80 species and their habitats must be evaluated to develop a scientifically credible plan.

This document provides an overview and evaluation of the knowledge of the major threats to the persistence and recovery of desert tortoise populations. I was specifically asked to evaluate the scientific veracity of the data and reports available. I summarize the data presently available with particular focus on the West Mojave Desert, evaluate the scientific integrity of those data, and identify major gaps in the available knowledge. I do not attempt to provide in-depth details on each study or threat; for more details I encourage the reader to consult the individual papers or reports cited throughout this report (many of which are available at most university libraries and at the West

Mojave Plan office in Riverside, California). I also do not attempt to characterize or evaluate the past or present management actions, except where they have direct bearing on evaluation of threats, nor do I attempt, for the most part, to acquire, generate, or evaluate new or existing, but uninterpreted data.

Two Important Caveats

Lack of scientific evidence supporting a purported impact should not be confused with automatically supporting the alternative, that there is no impact, and vice versa. Or as it is sometimes said: "absence of evidence is not evidence of absence." It may just mean that credible or definitive studies testing the hypothesized effects have either not been conducted or not been reported adequately.

Additionally, when I critique a particular study I am neither criticizing the scientist's ability or intent. Often, studies have inherent weaknesses that are completely or largely out of the control of the researcher. For example, as discussed below, it is often very difficult to have a proper control for a study in nature and it is often too expensive or impossible to adequately replicate a natural study. Rather than abandoning the questions altogether, scientists forge ahead with the study in spite of its limitations and collect data that hopefully are useful for managers. I point out the weaknesses here so managers will understand the limitations of such data, not to criticize the researchers not to render the studies useless. Virtually all studies have some inherent value, but their utility falls at different points on the continuum of risk to managers depending in part on how they were conducted and reported.

USE OF DATA TO MAKE MANAGEMENT DECISIONS

Scientific investigations follow an orderly, repeatable process. Many such investigations begin with anecdotes from ranchers, recreationists, or casual observers of nature. These might include issues of concern to managers, such as "I'm seeing fewer tortoises these days" or "tortoises and cattle can coexist." Anecdotes are useful for pointing out to researchers what critical problems may need to be solved through scientific investigation. Most scientific research follows up anecdotes that seem plausible with more craftily constructed hypotheses and direct observation by experienced observers. If such observations warrant further investigation, scientifically based observational studies are initiated. Most studies pertaining to desert tortoises fall into this category. However, observational studies may have problems, such as lack of adequate controls, insufficient sample sizes, or researcher bias in study design or interpretation. In a few cases, experiments are used to objectively test hypotheses that were developed from anecdotal or observational data. Experiments or carefully designed observational studies may lead to development of conceptual or mathematical theories that can then be used to predict responses of valued resources to management actions. Theory can then be tested with further experimentation or well-designed observations. Very little theory has been applied to problems related to land-management practices in the Mojave Desert.

Types of Data

The quality of data depends on how the questions were formulated and how the data were collected. Research questions in tortoise biology and management rarely employ a standard scientific method called "strong inference" (Platt 1964). For strong inference, progress is generally made by devising clear, falsifiable alternative hypotheses and conducting experiments designed to test competing predictions of these hypotheses. The strongest support for one alternative comes from experimental results that exclude other alternatives. Studies that test only one hypothesis are weak because they fail to show that the same results cannot be explained by other hypotheses. In tortoise research we generally see studies that are designed to support a pre-determined "ruling theory" or "working hypothesis" (Chamberlin 1965) or to simply describe nature. Such studies do little to explicate the phenomenon and to truly advance the management objectives supported by the research.

There are several types of studies that vary by how the data were collected. These categories are listed below in descending order from those generally providing the strongest, most valid conclusions to those providing the weakest, least reliable information. Value specifically refers to the level of risk a manager is taking when making a decision based on the data. The lower the value, the higher the risk. The actual conclusion may be right on target, but if it is from a risky type of data collection, the manager runs a higher risk of making an unsound decision.

Experiment

The strongest scientific data, those demonstrating cause and effect relationships, are generated via well-controlled and replicated experiments (Hairston 1989, Lubchenco and Real 1991). Such experiments involve manipulating one variable (treatment, such as presence of cattle) while holding all other variables constant (such as tortoise density or soil type). Such a design must have a control (or reference site) wherein ideally the only difference is the lack of the treatment. Any resultant change in the treatment area is likely to be caused by the particular treatment. However, one of many uncontrollable factors may occur that could result in a change independent of the treatment. These uncontrollable features, called random error, can fatally compromise the results. To reduce the effects of random errors (or chance), a properly designed study must have replicates - two or more sites that serve as control and two or more sites that serve as the treatment sites (Hurlbert 1984). The more replicates there are, the lower the chance that differences observed between treatment or control sites can be caused by random error. Another source of error that is mitigated by replication is uncontrollable (or unrecognized) differences among study sites (e.g., soil type, grazing history, and slope).

Any experiment that fails to have an adequate number of replicate treatment and control sites fails to satisfy an essential requisite for strong inference. Admittedly, it is often difficult or even impossible in natural settings to establish true control sites where the only difference is the lack of a treatment, not to mention have multiple replicates of the treatment and control. But having a proper control is an important feature and conclusions drawn from studies that lack a control suffer as a result.

Furthermore, the strength of any experiment, its ability to be broadly applicable, is bolstered by sample size. However, when comparing a given treatment with a given control, the sample size is the number of replicate study sites, not the number of measurements taken within each site. It is all too common for studies, particularly non-peer reviewed ones, to artificially inflate their sample sizes thus often reporting a significant effect (i.e., difference between treatment and control caused by the treatment factor) when in fact one did not occur or when the study was inadequately designed or carried out to discern a difference if one indeed existed. For example, when studying the effect of a factor like off-road vehicle (ORV) activity on desert habitat, it is common to measure number of plants and plant species within an ORV area versus outside of the area. If the researcher measured number of plants and plant species along ten transects within a single plot inside and ten transects within a single plot outside, the sample size is not 10 (nor 20) rather it is 1, because there is only one pair of plots being compared. Any differences observed may actually be caused by other factors such as different elevation or vegetation type. To avoid the random error of non-replication, multiple plots should be studied and these should be inside and outside of several ORV areas.

Correlation

Many studies in natural environments measure how a given factor (e.g., animal density) varies at different levels of some treatment (e.g., intensity of cattle grazing). This type of experiment can only show a correlation between the two factors. It provides no evidence that one factor causes a change in the other. Any correlation may just as well be from some unmeasured feature of the environment that affects both factors measured or it may be caused by chance. A cause and effect relationship can only be demonstrated if it can be shown that varying one factor (the independent variable) causes a predictable and consistent change in the other factor (dependent variable). Unfortunately, this is often the only means we have to study phenomena in the natural environment.

Description/Observation

Many studies simply describe a particular physical state or phenomenon (e.g. amount of trash or number of tortoises in a study area). The description can be simply qualitative (e.g., "a lot" or "many") or may be quantitative involving complex statistics (e.g., means, standard deviations, confidence intervals). Such studies may provide excellent descriptions, but cannot test for cause and effect relationships.

Anecdote

Generally, a non-quantitative description limited in scope (usually a single observation of the given phenomenon) and depth of detail is considered an anecdote. An example of an anecdote is: "in 1978 I saw a tortoise eat a balloon." Anecdotes usually lack any formal documentation and are most often made by untrained, casual observers, but professionals often report anecdotal observations. Sample sizes are extremely limited. Anecdotes are highly risky for basing management decisions because of their lack of rigor, repeatability, and objectivity.

Anecdotes need to be properly evaluated using sound scientific methodology. They can often form the basis for more formal observations, hypothesis development, or experimentation. Occasionally, there are attempts to legitimize anecdotes by compiling many into a single report and attempting a quantified or statistical treatment. These are misguided attempts because the extreme weakness and subjectivity of the basic data limit entire analyses: the anecdote. An appropriate expression is "the plural of anecdote is not data" (Green 1995).

Speculation

People will often make guesses about possibilities for which there are no hard data. When those guesses are based on clearly stated and well-founded assumptions, the guesses are called hypotheses and can help to direct future conceptual and experimental pursuits (Resnik 1991). When assumptions are weak or unstated the guesses are speculations. An example of a speculation is that fallout from nuclear tests in Nevada in the 1950s is responsible for the prevalence of disease in tortoises today. There is no evidence that fallout from nuclear testing can cause the diseases harming tortoises and no reports detailing the amount of fallout that occurred in tortoise habitat. There are no attempts to correlate probable fallout amounts with incidence of disease. The assertion is strictly a speculation because, on the face of it, it makes some sense.

Speculations may be seductive; often they present a series of progressively dependent statements that have an internal logic of their own. The logic may appear compelling and is often bolstered by attempts to provide "proof" through analogies. Such argumentation often collapses when primary assumptions are nullified or when they are tested against real data, but too often the test is never made. Although they may sometimes form the basis for hypotheses and experiments, speculations are risky to base management decisions on because there is essentially no way to evaluate them and their predictive value is low.

Source of Data

Data sources fall into several categories with varying probabilities of adequate reliability and validity. The source of data provides some indication of its quality. However, it is possible that a particular conclusion based on data from a less reliable source is more true or accurate than one from a more reliable source, but the likelihood of this being the case is low. Thus it is less risky to base judgements on data obtained from more reliable sources. The basic sources of data follow, in order of increasing risk to management (i.e., decreasing reliability):

Peer Reviewed Open Literature

Open literature refers to articles readily available in university and public libraries and published in professional, publicly available outlets. Easy availability allows anyone to obtain and evaluate the data on which decisions are made.

Peer review is a cornerstone of the scientific process. Rigorous peer review has two essential components: 1) thorough review by two or more scientists (generally anonymous) knowledgeable on the topic and 2) the possibility of rejection if the report does not meet generally accepted scientific standards. The latter component is an important feature that is lacking in less reliable data sources. The review process helps to ensure (but does not guarantee) that: 1) only reliable data with valid conclusions are published because the reviewers make certain that data are presented in sufficient detail to allow adequate evaluation of the conclusions; 2) the collection and analysis methods followed modern scientific standards and were appropriate for making the tests reported, 3) were reported in sufficient detail to allow someone to adequately evaluate and repeat the study; 4) the conclusions follow logically from the data; and 5) relevant related data (e.g., peer-reviewed publications), whether supporting or contradicting the study's conclusions, are cited. Most professional scientific journals (e.g., Ecology, Range Management, Journal of Wildlife Management, Herpetologica, Bulletin of the Wildlife Society) are peer reviewed. The Desert Tortoise Council is now implementing an external review process for its annual symposium proceedings.

Technical Books, Theses, and Dissertations

Most technical books are peer reviewed, but often without the true possibility of rejection. They are often reviewed by an in house editor or panel of editors who may or may not be experts in the particular field. Opinions differ on whether master's theses and doctoral dissertations should be considered peer reviewed. They do not undergo the same blind review that papers in scientific journals do, but they probably receive a much higher level of scrutiny than most papers. Furthermore, there is much more at risk if the thesis or dissertation fails review: the student is not awarded the Masters or Ph.D. In this report, they are treated as technical books being reviewed by a panel (i.e., the student's graduate committee).

Non-peer Reviewed Open Literature

Articles from this source are often used to support decisions or recommendations probably because there are many of them available, the sources are widely available, and the fact that they have been published adds a perception of respectability. However, there are often risks of using this type of data source. The authors and editors may not be specialists in the field they are writing about or are not scientists. Additionally, there is often no attempt at a logical, unbiased, rationally supported presentation. Occasionally, special interest groups that are pushing a specific interest and land ethic (e.g., Audubon Society, Rangelands, Desert Tortoise Council) publish outlets cited.

By definition, non-peer reviewed sources do not follow the established methods of peer review: there is usually no independent, objective evaluation of the data presentation and no guarantee that articles will be rejected if they fail to meet accepted scientific standards. Often missing is information necessary to allow the reader to evaluate the reliability of data collection and analysis. Statements such as "many tortoises were killed by vehicles" or "tortoises depend on cow dung for nutritional needs" are made without details about how the author determined if a vehicle killed a tortoise, how often tortoises actually eat cow pies, or what are the nutritional needs of tortoises.

Most proceedings of meetings (e.g., past issues of the Proceedings of the Desert Tortoise Council Symposium -) as well as abstracts from meetings are incompletely or not peer reviewed, and contents are usually printed verbatim with little or no editing and no possibility of rejection. Proceedings papers and abstracts often contain preliminary analyses of data and conclusions may change following the final complete analysis and rigorous peer review. The same criticisms holds for many official bulletins and newsletters of professional societies (e.g., Bulletin of the Ecological Society of America, Rangelands).

Technical Reports

Technical reports are generally written by agency and contract scientists and biologists and sometimes individuals untrained in the practices of science and biology. Technical reports are probably the most commonly used source of data for basing management decisions. Many agency biologists do not have the time, opportunity, encouragement, need, or training to publish their data. Sometimes reports are generated for the purpose of providing a quick analysis for management decisions that cannot wait for the one to two years often necessary to become published in a peer reviewed outlet. Such reports may not be subjected to review by competent scientists and are rarely rejected. "Draft" reports may never be finalized and become widely used even though they may be incomplete or fatally flawed. Because they do not appear in the open literature, refutations or critiques of the reports are rarely available. Finally, they may be difficult to locate, which prevents independent evaluation of their findings.

Reports by government biologists and biological consultants are variable in quality. Many are well designed, researched, and written and draw adequately on the existing body of scientific knowledge. Others demonstrate a lack of knowledge of tortoise biology and common management practices; fail to properly cite previous studies, particularly when contrary to the conclusions or recommendations being made in the report; make recommendations that are untested or unwarranted; and have not been peer reviewed. Such reports form the basis of many management decisions that have or are being made and may result in implementation of non-standard mitigation measures and speculative conclusions that were not tested for their efficacy.

Unpublished Data

There are many data sets (e.g., raw data, tables of compiled data, GIS maps, etc.) that are cited and used even though they may not have been checked for errors, analyzed, or adequately documented (e.g., data collection methods may be unknown). Reliance on such data for making decisions is risky particularly when there is no documentation (e.g., metadata) of how the data were collected and limitations of the data are not discussed.

Professional Judgement

When the proper research has not been conducted or completed, or time or expertise is not readily available, managers often rely on the professional judgement of staff biologists or other scientists. Reliance on professional judgement requires managers to use data that are unreliable if only because they cannot necessarily be independently evaluated or examined. The judgement may involve unsupported speculation, data that have been improperly or incompletely analyzed, or may involve faulty recall of the facts. On the other hand, professional judgements may be very sound, reliable, and based on an objective evaluation of the information available. The manager may not be able to separate good from poor judgements because there is generally too little information to evaluate. Judgements solicited from several competent professionals is advisable when possible. Also, the professionals chosen to provide input should provide citations and critical analyses of the data they are using to make the judgement. They should clearly state where the strengths and weaknesses in their judgements lie. Following steps like these can help to ensure the value of professional judgement.

Science Lore

Science lore, best defined as being the collective knowledge of the scientific, resource professional, or layperson community, is often based more on observation, assumption, and speculation than on scientifically-collected and analyzed data. Facts entrenched in science lore are not necessarily incorrect. They are unreliable because the connection between the hard data and the interpretation may be unknown. Common sources of Science Lore include Television programs, hobbyist journals, newsgroups, and casual conversations with professionals and laypersons.

A common example of Science Lore is the statement that "tortoises live to be 100 years old or more." This may be true, but in fact the oldest tortoises for which any documentation exists were two captive animals; one was at least 67 years old and maybe in its mid seventies and the other was probably at least 74 and maybe older (the former was adult-sized when first captured 52 years earlier, Jennings 1981; and the later was adult-sized when captured and grew little in the 59 years before it died, Glenn 1986). No one has followed marked animals in the field long enough to know the average or maximum longevity. In the pair of studies usually cited as evidence for long life, six marked tortoises, recorded as adults by Woodbury and Hardy (1948) in the early 1940's, were refound still living in the 1960's (Hardy 1976). They may have been over 100 or perhaps as young as 30 - 50 years when refound. Since they were of unknown (or unreported) age at the time of capture, we do not know their true age. Using scute annuli (age rings), Germano (1992) estimated that most desert tortoises live 25-35 years, but some live more than 40 years. The cohort of tortoises reported on in Turner et al. (1987a) is still being followed; these known-aged animals are now 40-41 years old (Medica pers. comm.).

The onus is on the scientific community to identify statements that fall into this category. Researches should then investigate the underlying assumptions, find or collect supporting or refuting data and publish the results. Then, fact-based science lore can be elevated to known facts, and unsound lore can be modified or dropped from our lexicon of apparent facts.

This report identifies the quality of the data available on the major threats confronting desert tortoise populations in the hope that the scientific-based components of the final decisions can be clearly separated from the value-based components.

Two Final Caveats

The citation of draft reports or completed but unpublished ones is not normal scientific practice. Because this is a critique of all data that may be relevant to decision making for the West Mojave Plan, draft and incomplete reports are cited. This was done because such documents are often relied upon heavily for making management decisions.

Second, this report includes some papers and observations that are highly speculative or made by laymen, sometimes only in casual conversation. These were included here because they are often pervasive parts of the lore of the tortoise or desert communities and deserve some evaluation even if they were not made in scientific literature.

DESERT TORTOISE BIOLOGY

Knowledge of many characteristics of the basic biology of an organism is essential for making informed decisions concerning the management of that organism. Many aspects of tortoise biology are well known. The reader is referred to the following papers for general summaries of what is known: Berry (1978), Hohman and Ohmart (1980), Bury (1982), Bury and Germano (1994), USFWS (1994), Ernst et al. (1994), Grover and DeFalco (1995), and Boarman (2002). No comprehensive critical summary of tortoise biology exists and is sorely needed. A recent summary of anthropogenic impacts to desert habitat is Lovich and Bainbridge (1999).

SPECIFIC THREATS TO TORTOISE POPULATIONS

Threats occur under two major categories, direct and indirect, although they are not necessarily mutually exclusive. Direct threats are those that affect the survival or reproduction of tortoises (e.g., road mortality, illegal collecting, disease, predation). Indirect threats affect tortoise populations through their effect on other factors, primarily habitat (e.g., drought, habitat alterations from livestock grazing, recreational activities, global warming, etc.). Direct threats are usually more easily measured and therefore more easily evaluated than indirect effects.

To determine the impact of a specific threat on tortoise populations, it is insufficient to measure the threat solely (e.g., number of cars or density of mines in an area.) One must determine the effect the threat has on some aspect of tortoise reproduction or survival. Many parameters of tortoise biology can be measured when attempting to determine impacts of threats. Sometimes, the easiest and most intuitive response is mortality. It is difficult to deny that a motorized vehicle killed a fresh, smashed tortoise found on a paved highway. When tortoises die they leave behind a shell that can last for four years or more (Woodman and Berry 1984). Often that shell bears evidence of the cause of death (e.g., tooth marks, conchoidal fractures, fracture from blunt trauma, etc.). However, interpreting these signs is subjective and little scientific work that can aid interpretation has been conducted (but see, Berry 1985, 1986a) and most assumptions made in interpreting the evidence are not reported. Reproduction is more problematical, but at least clutch size and frequency can be measured with x-rays or sonograms or by locating nests and monitoring hatching success (Gibbons and Greene 1979; Turner et al. 1986, 1987b; Rostal et al. 1994). Survival of the young is an essential component to understanding the effect of threats on tortoise populations, but is very difficult to measure (e.g., Turner et al 1987b, Morafka 1994). Growth (Medica et al. 1975, Germano 1988, Turner et al. 1981, Patterson and Brattstrom 1972), behavior (Ruby and Niblick 1994, Ruby et al. 1994), and physiology (Nagy and Medica 1986, O'Connor et al. 1994a, Christopher et al. 1994) vary with environmental conditions and may be useful parameters for measuring the effect of impacts, but their efficacy at doing so has yet to be demonstrated. Modeling population demography (i.e., age-specific survival and reproduction), when using accurate measures from the population, can be an excellent way of evaluating the effects of threats and management actions on population growth (Congdon et al. 1993, Heppell 1998).

Relative Importance of Threats

The rating of relative importance of different threat factors is a challenging undertaking for several reasons. First, it is very hard to determine the cause of death of animals and it is even harder to determine how much decline is really attributable to the various indirect causes of mortality (e.g., habitat alteration). Educated guesses can be made about causes of death (Berry 1984, 1985, 1986a, 1990 as amended), but most of the methods used have not been described or subjected to experimentation, independent evaluation, or peer review. Second, not enough is known about several potential threats to evaluate their absolute or relative impact. For example, it has been suggested that toxic chemicals may be responsible for a disease of the shell affecting some populations. However, it is not known if chemicals are the causative agent, which chemicals are the problem, or the source of chemicals. Also, little is known about neither the epidemiology of the disease nor how much mortality is actually caused by it. Third, which mortality factors are functioning is very site specific. Highway mortality is an important factor for populations along highways; it may drain populations two miles or more away (von Seckendorff Hoff and Marlow 1997). On the other hand, for populations away from highways, this may be a very low or non-existent threat. Regional differenees occur, also. Urbanization and development are major factors in portions of the west Mojave, but are probably relatively unimportant in much of the east Mojave (outside of the Las Vegas and St. George areas). Finally, as discussed above, factors that caused the declines (e.g., disease) may not be the same factors that are preventing recovery (e.g., genetic or demographic consequences of small populations, fragmentation, and raven predation). For all of these reasons the controversial and subjective task of ranking impacts was avoided here.

Specific threats are easy to discuss and identify, but more pervasive problems often exist when multiple threats interact to make for larger environmental problems. The three largest of these broader impacts affecting tortoise populations are habitat loss, degradation, and fragmentation; urbanization and development; and access by humans to tortoise habitat. I will first focus on specific threats then discuss three broader, more cumulative types of threats. There are virtually no published studies looking specifically at the effect of these general factors on tortoise populations.

Agriculture

Probably the greatest affect agriculture has on tortoise populations is through loss of habitat: when tortoise habitat is converted for agricultural use it becomes mostly unusable by tortoises for foraging or burrowing. Indirect impaets could include facilitation of increases in raven population, drawdown of water table, production of fugitive dust, possible introduction of toxic chemicals, and introduction of invasive plants along corridors and when the fields go fallow.

I found no substantiated references in the literature indicating that desert tortoises use agricultural fields, although alfalfa, with its high nitrogen content, could be a healthy source of food for tortoises (Bailey, 1928, provides an anecdotal account from untrained observers of "tortoises eagerly eating alfalfa."). Berry and Nicholson (1984a) cited one anecdotal report from an individual with unreported credentials as evidence that "tortoises are known to enter...alfalfa fields" (p. 3-21). Disking, plowing, mowing, and baling would destroy burrows and kill tortoises (as they do the marginated tortoise, *T. marginata*, in the Mediterranean region; Stubbs 1989). There are no reports of desert tortoise burrows in agricultural fields.

The Common Raven, a predator on juvenile desert tortoises, makes considerable use of agricultural fields in the west Mojave Desert (Knight et al. 1993, 1999, Knowles et al. 1989). Agricultural fields probably are important sources of food (i.e., insects, rodents, and seeds) and water for ravens during times of the year when those resonrces are generally in low abundance elsewhere, thus resulting in more ravens surviving the summers and winters (Boarman 1993, unpubl. data). See "Predation," below, for more discussion.

Pumping of ground water for irrigation can result in a major change in vegetation or habitat type. Koehler (1977) reported that the drawing of water for irrigation from Koehn Dry Lake, near Cantil in the Western Mojave, lowered the water table by 240 ft between 1958 and 1976. Berry and Nicholson (1984a) state that this lowering of the water table has approached the Desert Tortoise Natural Area (DTNA) and imply that it may affect tortoise habitat, although no data were presented to support the implication. Closer inspection of the maps provided in Koehler (1977) show that the water-level decline is lower (30 - 180 ft) near tortoise habitat south and southeast of Koehn Dry Lake. There are no data to indicate what effect this lowering of the water table has on mesquite, other vegetation, or tortoise habitat in the area, but there are data on the effect water table lowering has on mesquite in other arid regions (Nilsen et al. 1984).

Agricultural fields cause dust storms, called fugitive dnst (Wilshire 1980). Fugitive dust coats plants, which in turn may reduce photosynthesis and water-use efficiency (Sharifi et al. 1997). The end result is lower productivity of forage plants. Their study did not specifically look at agricultural dust, hut the results are probably generalizable.

The finding of "hundreds of...tortoise shells" (with no indication of how long the tortoises had heen dead) was reported anecdotally and second hand by Berry and Nicholson (1984a) and was correlated with application of an unspecified pesticide to kill jackrabbits in a nearby (distance unspecified) alfalfa field. Aside from this single unsupported speculation, there are no references to possible toxic effects on tortoises of pesticides, herbicides, and other chemicals used in agriculture. Pesticide use, particularly aerial applications apparently are now very limited in the desert.

Collecting by Humans

Humans collect turtles and tortoises for several reasons, and these activities are responsible for population declines in several of the threatened and endangered species throughout the world (Stubbs 1991). Collecting desert tortoises for pets was probably a major activity in the recent past (Berry and Nicholson 1984a), although most evidence is anecdotal in nature. Since 1961, it has been illegal under State law to collect tortoises in California and since 1989 collecting has been a Federal offense (USFWS 1994). The Desert Tortoise Recovery Plan (USFWS 1994) cites several documented instances of illegal collecting more recent than those in Berry and Nicholson (1984a), including the unauthorized removal of marked study animals from known study areas. It must be cautioned that some of the examples cited in the Recovery Plan are circumstantial or speculative. For instance, Stewart (1993) reported one strongly supported (tortoise found in a car in Idaho) and one speculative (transmitter and human footprints found on ground and tortoise was missing) example of poaching. Berry (1990 as amended) gives purely speculative and circumstantial evidence for poaching (namely, marked drop in estimated density on a study plot over a 5-year period with relatively few carcasses being found coupled with observations of possibly human-excavated burrows nearby and other evidence for poaching several miles away). The available evidence suggests that collecting for pets is still occurring, but perhaps at a level lower than previously, although this statement is speculative at present. Evaluating the extent of the problem is very difficult because of the cryptic nature of the activity.

A newly documented problem is the collection of wild tortoises by recent immigrants for cultural observances (USFWS 1994, Berry et al. 1996). Berry et al. (1996) reported that 7.7% of tortoise burrows found showed evidence of being excavated by humans and that the number of such burrows is greater near versus far from dirt roads. Their study suggests that poaching tends to occur near roads, even lightly maintained ones, thus the presence of roads may help to facilitate poaching. However, there was no statistically significant difference in distance from roads for disturbed versus undisturbed burrows and the method for determining if a burrow was excavated was circumstantial and subjective.

The bottom line is that there is little evidence to suggest that illegal collecting is currently a widespread problem, but there is also little evidence to the contrary.

Construction Activities

Construction activities here refer specifically to the generally short-term effects of actual construction (clearing land, movement of heavy equipment, presence of construction crews, etc.). The lasting effects of the constructed facility, once in place, are discussed in "Urbanization and Development," "Energy and Mineral Development," "Utility Corridors," and "Habitat Loss, Degradation, and Fragmentation" sections below. In many ways, most construction projects have similar impacts on tortoises and their habitat, regardless of what is being constructed. Those impacts may include: loss of habitat by the project footprint; incidental destruction of habitat in a buffer area around the footprint; damage to soil and cryptogams on the periphery; incidental death of unseen tortoises along roads, beneath crushed vegetation, or in undetected burrows; destruction of burrows; handling of tortoises; entrapment of tortoises in pits or trenches dug for transmission or fiber optic lines, water, and gas pipelines and other utilities; attraction of ravens and facilitation of their survival by augmenting food or water; and fugitive dust

(Olson et al. 1992, EG&G 1993, Olson 1996). There are little data on the extent of these potential impacts. But, Olson (1996) reported that a construction of a natural gas pipeline had the greatest impact on tortoises and habitat, construction of a transmission line had intermediate impacts, and a fiber optic line was the most benign. The differences are largely related to the scale of the project, ability of crews to avoid disturbing burrows, and timing of construction to avoid peak activity periods of tortoises (e.g., spring). In an analysis of 171 Biological Opinions issued by the USFWS in California and Nevada, Circle Mountain Biological Consultants (1996, see also LaRue and Dougherty 1999) found that the majority of tortoise mortality occurred along linear construction projects (e.g., pipeline, fiber optic, and transmission lines) with the extensive Mojave-Kern Pipeline causing the greater number of deaths (38). Tortoise mortality also occurred on mining, landfill, and military projects. The total number of deaths reported on the projects was well below the level authorized by the USFWS (59/1096 5.4%). This study was strictly an evaluation of known tortoise mortalities occurring during projects authorized by the USFWS under Section 7 of the Endangered Species Act. It therefore likely underestimates actual tortoise mortality (e.g., tortoises buried during construction or otherwise not found, accidentally killed but not reported, etc.) that occurred.

Disease

Disease in general is a normal and natural phenomenon within wild animal populations. Diseases can weaken individuals, reduce reproductive output, and cause mortality. Epidemic outbreaks of some diseases can become catastrophic, particularly in small or declining populations (Dobson and Meagher 1996, Biggins et al. 1997, Daszek et al. 2000). Sometimes disease can be controlled by wildlife managers by attacking the pathogen; isolating diseased from non-diseased individuals, populations, or species; immunizing healthy individuals; or facilitating habitat conditions that increase individual's immune systems. Other times there may simply be nothing a manager can do. It is important to understand disease etiology and epidemiology before effective management actions, if any, can be determined.

Two diseases have been identified as possibly affecting the stability of some desert tortoise populations: Upper Respiratory Tract Disease (URTD; Jacobson et al. 1991) and cutaneous dyskeratosis affecting the shell (Jacobson et al. 1994). A third disease, a herpesvirus, was recently identified and may have population-level consequences, but very little is known about it (Berry et al. 2002, Origgi et al. 2002). URTD has been found in several populations that have experienced high mortality rates, including some in the west Mojave (Jacobson et al. 1996, Berry 1997). Much is published in peer reviewed journals about the etiology of this disease, which has been found in captive turtles of this and several other species (Jacobson et al. 1991) and in wild populations of the gopher tortoise (*Gopherus polyphemus*; Jacobson 1994). Brown et al. (1994a) showed definitively that URTD can be caused by a bacterium, *Mycoplasma agassizii*. It is likely transmitted by contact with a diseased individual or through aerosols infected with *M. agassizii*. The organism attacks the upper respiratory tract causing lesions in the nasal cavity, excessive nasal discharge, swollen eyelids, sunken eyes, and in its advanced stage, lethargy and probably death (Jacobson et al. 1991, Schumacher et al. 1997, Homer et al. 1998, Berry and Christopher 2001). It must be noted, however, that some of these clinical signs may also be characteristic of other health condition such as dehydration, allergy, or infection with herpesvirus or the bacteria *Chlamydia* or *Pasteurella* (e.g., Pettan-Brewer et al. 1996, Schumacher et al. 1997).

Malnutrition is known to result in immunosuppression in humans and turtles (Borysenko and Lewis 1979) and is associated with many disease breakouts. It is possible that nutritional deficiency in tortoises caused by human-mediated habitat change and degradation may be partly responsible for the apparent spread of URTD and its perceived impact on tortoise populations (Jacobson et al. 1991, Brown et al. 1994a). Short-term droughts may temporarily reduce immune reactions and increase susceptibility to URTD (Jacobson et al. 1991), although this is speculative. Whereas animals may become debilitated by chronic immune stimulation, no biochemical indicators of stress have been identified in diseased compared to non-diseased turtles (Borysenko 1975, Grumbles 1993, Christopher et al 1993, 1997).

Although evidence indicates a correlation between high rates of mortality and incidence of URTD within populations (Berry 1997), there is little direct evidence that URTD is the cause of the high rates of loss. In two preliminary analyses (Avery and Berry 1993, Weinstein 1993), animals exhibiting clinical signs of (both studies) or testing positively for (latter study) URTD were no more likely to die over a one year period in the west Mojave than were those not exhibiting signs or testing positive. This may be because factors other than disease caused much of the mortality or many animals not showing clinical signs of disease in the field were still infected. A serological test for presence of antibodies against *M. agassizii* has been developed and is now being used to document presence and spread of the disease (Schumacher et al. 1993). But, the test, an enzyme-linked immunosorbent assay (ELISA) does not indicate present infection, only a probability of past exposure. A polymerase chain reaction (PCR) test, which has been developed for *M. agassizii* is more effective for determining active infection (Brown et al. 1995). Lance et al. (1996) reported that infected tortoises had significantly lower testosterone and estradiol levels and that diseased females tended to lay eggs less often. Finally, there is some evidence that animals at the DTNA, where URTD breakout has been particularly intense, may recover from infection (Brown et al. 1994a, b). Interestingly, Berry (2002) reported that none of 119 wild tortoises tested at 9 locations throughout the California deserts in 2000 and 2001 tested positive for URTD. No discussion of this result was provided. A thorough epidemiological study is badly needed to identify the factors involved in the incidence, spread, and virility of the disease in wild populations (D. Brown pers. comm.).

A shell disease, cutaneous dyskeratosis (CD), has been identified in desert tortoise populations (Jacobson et al. 1994). CD consists of lesions along scute sutures of the plastron and to a lesser extent on the carapace. Over time, the lesions spread out onto the scutes. This disease may be caused by the toxic effect of chemicals in the environment, but evidence is lacking to test this hypothesis. Naturally-occurring or human-introduced toxins such as selenium, chlorinated hydrocarbons, organophosphates, nitrogenous compounds, and alkaloids have all been implicated (Homer et al. 1998), but there are no data showing a direct link. The disease may also be caused by a nutritional deficiency (Jacobson et al. 1994). It is not known whether or not CD is caused by an infectious pathogen or if secondary pathogens act to enhance the lesions (Homer et al. 1998, Homer pers. comm.). It is unclear if the disease is actually lethal or responsible for declines in infected tortoise populations (Homer et al. 1998). Only one documented case of CD from the West Mojave Desert was found in the literature (Homer et al. 1998).

If the shell diseases are toxicoses, toxic responses to environmental toxins (e.g., heavy metals, chlorinated hydrocarbons, organophosphates, and selenium), then there may be a direct link between these diseases and human activities unless the toxin is a natural component of the physical environment. Chaffee et al. (1999) found no significant correlation between elevated levels of metals in organs of ill tortoises and in the soil where the tortoises came from. If there is a link to human activities, then we can consider solutions that would rednce levels of input of the toxic chemical. However, this link is currently highly speculative.

There is some recent, albeit weak, preliminary evidence linking heavy metals to disease in tortoises. In necropsies of 31 mostly ill tortoises, Homer et al. (1994, 1996) found elevated levels of potentially toxic metals and minerals in the liver or kidney of one or more of the animals. Since most of the animals were ill to begin with, an association was made between the presence of the toxicants and presence of the disease. However, that study is strictly correlative, and fails to demonstrate a cause and effect relationship. Berry (1997) claims that "the salvaged tortoises with cutaneous dyskeratosis had elevated concentrations of toxicants in the liver, kidney, or plasma...and/or nutritional deficiencies." However, closer examination of the data presented in Homer et al. (1994, 1996) and cited in Berry (1997) reveals a remarkably low association with only 1 out of 12 tortoises with CD having at least one toxicant concentration greater than two standard deviations above the mean. Four other animals also had unusually high levels of at least one toxicant, but did not suffer from CD. Furthermore, Homer et al. (1994, 1996) identified abnormally high levels as being those concentrations that are greater than two standard deviations from the average concentration found in the 31 tortoises. In a normally distributed set of 20 randomly selected values, 1 will, by definition, fall outside of 2 standard deviations from the mean, because 2 standard deviations is defined as including only 95% of the samples. So if 100 comparisons are made, then 5 levels will be considered abnormally high or low just by chance. In the study, 689 values would be reported, thus 34 (or 95%) would be expected to be greater than twice the standard deviation from the mean just by chance. In fact, 32 were identified as falling outside this range of two standard deviations. These data are in need of a thorough statistical analysis. Homer (pers. comm.) has found significantly higher levels of iron (in liver) and cadmium (in kidneys and liver) of tortoises with URTD compared to those in a control group. It is not known if the levels identified by Homer et al. (1994, 1996, pers. comm.) as being abnormally high are biologically significant. Homer (pers. comm.) has found significantly reduced levels of calcium in the livers of tortoises with CD, which suggests a nutritional deficiency may be involved in the disease.

Several other diseases and infections have been identified in desert tortoises (Homer et al. 1998). These include a poorly known shell necrosis, which can result in sloughing of entire scutes; bacterial and fungal infections; and urolithiasis, a solid ball-like deposition of urate crystals in the bladder (i.e., bladder stones; Homer et al. 1998). There is no evidence to suggest that any of these diseases are at this time widespread, threatening population stability, or hindering population recovery.

Beyond taking precautions to avoid spreading the disease when handling many animals (Rosskopf 1991, Berry and Christopher 2001), educate the public against releasing potentially-diseased captive animals (Berry 1997), include only healthy individuals in translocation efforts (Brown 1994a), the practical management implications of the disease data are unclear. Tully (1998) states, without explanation, that URTD infections are not likely to be controlled by immunizations. Improving habitat conditions may help reduce stress-induced immunosuppression (Brown 1994a), but the link between stress from poor habitat quality and susceptibility to URTD is only speculative.

Drought

A drought is an extended period of abnormally low precipitation. Unlike kangaroo rats and some other desert vertebrates, tortoises acquire much of their water, and maintain and overall positive energy balance, from standing sources (Peterson 1996). O'Connor et al. (1994a) showed that water deprivation in a group of semi-wild tortoises caused higher levels of physiological stress (using several blood assay profiles) compared to a group of semi-wild tortoises with water supplements and a group of free-ranging tortoises. Peterson (1994a) recorded abnormally high levels of mortality in two tortoise populations (west and east Mojave) during a three-year period of an extended drought. The deaths in one population (Ivanpah Valley) were attributed to drought-induced starvation and dehydration and occurred in the third year of study. Ken Nagy (pers. comm.) has stated that tortoises can probably survive 1-2 years without drinking water but will start dying of dehydration after that. The primary source of mortality, which occurred throughout the three-year study, at the DTNA was coyote predation. The coyotes may have switched to the less desirable tortoises following hypothesized drought-induced reduction in coyotes' normal prey (black-tailed jackrabbits; see also Jarchow 1989). Alternatively, tortoises may have been in a weakened condition due to URTD, but Peterson (1994a) found little evidence of disease in his study animals. Low rainfall can also reduce reproductive output with tortoises producing fewer eggs or suspending egg-laying altogether in low-rainfall years (Turner et al. 1984, Lovich et al. 1999). Avery et al. (2002) documented higher survival and reproduction among females at higher elevation site that received more rain than a lower one in Ivanpah valley. Tortoises may survive drought periods by eating less nutritious cacti and shrubs (Turner et al. 1984, Avery 1998).

Much of the desert experienced short-term drought conditions in the late 1980s (Corn 1994a, Hereford 2002), a period when rapid declines and high mortality were reported in some tortoise populations (Berry 1990 as amended, Corn 1994a, Peterson

1994a). However, Corn (1994a) reported that, between 1977-1989 there was no correlation between winter precipitation and relative abundance of large (< 180 mm median carapace length [MCL]) or small (<180 mm MCL) tortoises, but there was a significant correlation between summer precipitation and relative abundance of small tortoises. Some reports exist of dehydrated and emaciated tortoises being found (Berry 1990 as amended, Peterson 1994a, Homer et al. 1996).

Drought is a normal phenomenon in the Mojave Desert (Peterson 1994a, Hereford 2002). Desert tortoises have lived in the Mojave Desert for over 10,000 years and probably have evolved under similar boom-bust conditions (Peterson 1994 a,b, 1996; Henen 1997; Nagy and Medica 1986). It is possible that drought can cause episodic mortalities punctuated by periods of low mortality during years with more abundant rainfall. It is reasonable to speculate that drought-induced stress in concert with other threats (e.g., disease, predation) resulted in significant mortality (Peterson 1994a), but there are little data to test this hypothesis. An epidemiological study is needed to evaluate the effect drought has on tortoise populations.

Energy and Mineral Developments

Energy and mineral development includes: presence of utility lines, transmission lines, and gas pipelines; development of land for oil and gas leases; geothermal and solar energy generation; and digging exploratory pits for and extraction of minerals. Impacts from energy and mining developments can include habitat destruction and direct mortality from off-road travel to explore and access sites; habitat loss to road and development construction, leachate ponds, tailings, rubbish, etc.; introduction of toxins; fugitive dust and soil erosion; and urban-type developments to support large mining operations. The extent of area directly affected by energy and mining is difficult to assess because the data are not readily available. According to Luke et al. (1991), as of 1984, 41% of high density tortoise habitat rangewide was leased or partially leased for oil or gas and 2% was directly impacted by mining operations or leased for geothermal development. However, no indication was given for how these figures were obtained. Most mining operations are point sources of disturbance with potentially little effect beyond the immediate site of development. The greatest effect may come from the cumulative impact of many relatively small mining-related disturbances combined with facilitation of rural or urban development (e.g., Randsburg) to support the mining operations in a given area. However, large-scale operations that depend on frequent haul trucks to transport excavated minerals may also present vehicle-related impacts such as increased road kills and air pollution.

There are few data on the effects of energy and mineral development on tortoise populations. Mortalities have occurred in association with mining activities (LaRue and Dougherty 1999). Hard rock mining, particularly pit mining and operations in dry lakebeds, can be a major source of fugitive dust (Wilshire 1980). Loss of habitat and soil and vegetation disturbance can be substantial and major, depending on the size of the area. Although illegal, cross-country travel to drill and access test pits, stake claims, and evaluate mineral potentials still occur (pers. obs.) and needs to be properly documented and evaluated.

Energy development has similar impacts, particularly direct and indirect loss of habitat, fragmentation of habitat and population, and effects of access roads, which are likely to be relatively light once construction has ended (Brum et al. 1983). Construction of transmission lines requires grading of new roads for construction of towers and maintenance of the lines, and clearing or terracing of habitat for tower placement. Not only is habitat lost (0.16 to 0.24 mi^2 per mile of transmission line; Robinette 1973, cited in Luke et al. 1991), but the new road may help to fragment the population and provide access to areas for other human-related impacts (see "Utility Corridors" section, below). The access roads are also an important source of windblown dust and attendant erosion (Wilshire 1980). The presence of new utility lines, necessary to distribute the electricity, may help facilitate nesting by ravens in specific areas they did not nest in before, if those areas did not have adequate nesting substrates before the new towers were erected (Boarman 1993, Knight and Kawashima 1993). For more discussion, see "Utility Corridors" section, below.

Aside from loss of habitat and other consequences associated with access roads and transmission lines, there is little evidence that energy generation negatively impacts tortoise populations. If designed and managed properly, wind generation may be compatible with tortoise populations (Lovich and Daniels 2000). Tortoises made extensive use of wind turbine pads for burrow cover and, by restricting access, the wind park served as a de facto reserve that minimized several other harmful human activities such as ORV travel, vandalism, and illegal collections. The only study found on solar energy impacts showed that here were only very small changes in air temperature, wind speed, and evaporation rates downwind from a solar power plant in the western Mojave Desert (Rundel and Gibson 1996). They did not study impacts to tortoise populations.

Fire

Fire, once considered a rare event in the Mojave Desert (Humphrey 1974), now occurs with ever-increasing frequency causing a greater threat to tortoises and their habitat (USFWS 1994, Brooks 1998). Fire frequency has increased with the proliferation of introduced plants, particularly the grasses, red brome (*Bromus rubens)* and split grass (*Schismus barbatus* and *S. arabicus*), which provide fuel for fires (Brown and Minnich 1986, Brooks 1999b). These plants help to spread fire because they are often common, tend to grow in large relatively dense mats, and fill the intershrub spaces, which are largely devoid of native vegetation (Brown and Minnich 1986, Rundel and Gibson 1996, Brooks 1999b). Fires cause direct mortality when tortoises are burned or inhale lethal amounts of smoke, which can happen both in and out of burrows. Documented cases of tortoises being burned by fires are uncommon, but do occur (e.g., Woodbury and Hardy 1948 - circumstantial, secondhand account of 14; Homer et al. 1998, reports 1; Esque et al. in press, reports 5, which is 4-13% of the study population; Lovich, pers. comm., found 1). Fires are probably most hazardous to tortoises when they occur during the active season for tortoises (e.g., spring in the West Mojave). Previously rare, frequency of spring fires are now on the increase (Brooks 1998).

There are several possible indirect impacts of fires. Fires remove dry and some living forage plants. They facilitate proliferation of non-native grasses (Brown and Minnich 1986, Brooks and Berry 1999). The effect this has on tortoises is as yet unresolved. There is some evidence that tortoises may selectively avoid exotic grasses (Jennings 1993, Avery 1998), but Esque (1994) showed that tortoises may choose to eat a majority of non-native plants, particularly in drier years. The physiological consequences of foraging on non-native grasses is also not entirely known, but, in a manipulative study with semi-captive tortoises, Nagy et al. (1998) showed that grasses, native and non-native) provided tortoises with much less nitrogen than did forbs and tortoises tended to loose water when eating them. Avery (1998) also showed that tortoises eating only split grass lost weight, assimilated less protein, and were in a negative nitrogen balance, whereas those that were fed a native forb (*Camissonia boothii*) maintained their weight and experienced a positive nitrogen balance. Those tortoises that fed on both plat types maintained their weight but experienced a net loss of protein. By removing vegetation, fires may alter the thermal environment by increasing temperature extremes experienced by seeds, plants, and burrowing tortoises (Esque and Schwalbe 2002). Soil erosion is enhanced by the loss of stabilizing vegetation, roots, and cryptogamic crusts (Ahlgren and Ahlgren 1966). Fires fragment tortoise habitat by creating patches of unusable habitat, at least over the short term. There is some evidence of an increase in availability of nitrogen and other nutrients for a short while following fires (Loflin 1987), but none demonstrating that plant growth is stimulated by this nutrient flush. Overall effects on vegetation are variable, and may depend in large part on the intensity of the fire, characteristics of the plants, and post-fire precipitation (Esque and Schwalbe 2002). Brown and Minnich (1986) found an increase in annual vegetation following a fire during an unusually rainy period. On the other hand, O'Leary and Minnich (1981) found no difference during a drier year.

The structural characteristics of vegetation in years following fires has been studied. Following burns in creosote scrub community in the Colorado Desert, Brown and Minnich (1986) found 23% higher cover by annual forbs, most of which were exotics. Cover by some native forbs, including ones preferred by tortoises, were also higher in burned vs. unburned areas. They also found that perennial plants, particularly creosote bush, were damaged and exhibited low levels of stump sprouting and germination following more intense fires. A change in dominant shrub type resulted, but the study only reported on 3-5 years post-burn; no data were presented on possible long-term successional changes or recovery. Dense cover by annuals, particularly introduced grasses, provides higher fuel loads, which results in more fires that are also hotter (Brown and Minnich 1986, USFWS 1994, Brooks 1999b).

The amount of tortoise habitat burned by recent fires is relatively low, but increasing. For example, between 1980 and 1990, 243,317 acres burned in the Mojave Desert in California, which is an average of 38 mi^2 per year (USFWS 1994). The increase in number of fires per year over the ten-year period was statistically significant. Tracy (1995) reports that fires occur much more frequently near roads and towns, but no data were presented in this abstract. Duck et al. (1995) reported that tortoises may be killed by fire-fighting activities, including by large fire trucks driving off of roads in tortoise habitat, and recommended training and fire management techniques to reduce the problem.

Through its destructive effect on woody shrubs, fire has been used to manage (i.e., improve for cattle foraging) desert grasslands. In desert grassland of southern Arizona, fire removed 9-90% of targeted shrubs (i.e., mesquite, *Prosopsis juliflora*; burro-weed, *Aplopappus tenuisectus*; prickly pear cactus, *Opuntia occidentalis*; and cholla, *Opuntia* sp.; Reynolds and Bohning 1956). This work was not conducted in tortoise habitat and the efficacy of using fire in similar ways has not been tested in the Mojave Desert nor has its effectiveness at improving habitat for tortoises been tested.

Garbage and Litter

Garbage illegally dumped in the desert is unsightly, may cause local habitat alteration, and may affect individual tortoises. Indeed, in a popular article, Burge (1989) cited an instant of a tortoise losing its leg after getting it caught in the string of a disposed balloon. She also reports finding foil and glass chips in tortoise scat. No details were provided. There are no data to suggest that litter is a widespread or major problem for tortoise populations. The relationship between organic litter and raven predation on tortoises is covered under "Predation," below.

Illegal dumping of hazardous wastes is an increasing problem in the California deserts (John Key, pers. comm.) Toxins are known to cause a myriad of problems for wildlife (Jacobson et al. 1994), and presumably elevated levels (see "Disease" section, above) of certain metals (e.g., cadmium, copper, molybdenum, mercury, lead) have been found in the tissues of desert tortoises (Homer et al. 1994, 1996, 1998). The distribution and limited size of illegal dumps and hazardous spills suggests that this is a minor problem for tortoise populations as a whole, but they may be of concern on a localized basis. Metals and other pollutants may enter the environment from other sources including mining and air pollution, but their effects on tortoise populations remain speculative.

Handling and Deliberate Manipulation of Tortoises

Handling and deliberate manipulation of tortoises includes curious members of the public picking them up and sometimes removing them from the wild, biologists relocating and translocating them to new sites, pet owners releasing captive tortoises into the wild, and researchers manipulating tortoises for scientific experimentation. The effects can be manifold, depend on the type of handling, and remain largely unstudied.

Members of the public will sometimes pick up tortoises when they find them on roads or alongside trails. They do so out of curiosity or to remove the animal from harm's way (Ginn 1990; picking up a tortoise to cause harm is covered in the

"Vandalism" section, below). Any such handling or even disturbance of a tortoise is illegal under the Endangered Species Act, although it is unlikely that USFWS would prosecute a person who moves a tortoise out of harm's way (pers. obs.).

There are several possible effects of this type of well-meaning handling, but most of them fit into the realm of speculation or science lore. First, when tortoises are handled they sometimes void the contents of their bladder, which may represent loss of important fluids and it is thought this loss could be fatal (Averill-Murray 1999). Averill-Murray (1999) provided some evidence that handling-induced voiding may jeopardize survivability, although usually relatively small amounts of fluid are discharged. Smaller animals were more likely to void, but, if the animal was recaptured at a later date, its growth was not inhibited as a result of voiding previously. The statistical significance of his results may be compromised by his decision not to adjust the level of significance to account for making multiple tests (a problem similar to that noted about Homer 1994, 1996, in the "Disease" section above). Nonetheless, the results suggest there may indeed be a trend towards voiding affecting tortoise survival, particularly in drought years, and this should be followed up with more experimentation.

Other problems with handling tortoises can occur. Diseases might be transferred between tortoises if people handle more than one tortoise without sterilizing their hands or using different clean or sterilized gloves for each handling (Rosskopf 1991, Berry and Christopher 2001). It is claimed that turning over a tortoise to look at its underside will harm its internal organs, break eggs, or cause shock (Rosskopf 1991), but there is no evidence to support this contention. It may be detrimental to a handled tortoise if it is released outside of its home range, far from known burrows, or away from shade (e.g., Stewart 1993). This could be particularly hazardous during hot, dry weather or late in the afternoon, but again no data exist to support this likely speculation. Finally, the disruption of behavior by handling or just approaching the tortoise could be harmful if the disruption causes the animal to withdraw into its shell long enough to prevent it from being able to eat, drink, or retreat to a safe cover site (e.g., burrow, pallet, or shrub) for the night, thus leaving it exposed to predators or harsh environmental conditions. The probability of this disruption being hazardous to the tortoise is likely low, unless disruptions occur extremely frequently. Tortoises can go many months without eating or drinking (Peterson 1996), so a few minutes of disruption is not likely to alter their nitrogen, energy, or water balance. All of these claims need further study to substantiate their validity.

Relocation of animals to a new area is frequently recommended, and is occasionally implemented to save tortoises from construction and other ground disturbing activities. Possible problems with translocation efforts include increased risk of mortality, spread of disease, and reduced reproductive success. There have been a few studies of the effectiveness of relocation efforts, and most of the relocations generally have been marginal to unsuccessful. A study summarized in Berry (1986b) found that 22% (13/43) of the animals translated 16 to 88 km from their capture sites stayed at their relocation sites for more than several days, but only five remained for 15 months to 6 years. Few mortalities were observed, but many disappearances from unknown causes occurred; these animals may have died or wandered away. In another relocation effort, 91% (10/11) stayed within the relocation area, which was only about 450 m from where they were moved, for at least 3 months and at least 36% (4/11) were present after 16 months (Stewart and Baxter 1987). In a third effort, 56% (9/16) of relocated tortoises stayed in the area (5.6 km from their original home ranges) for at least 1.5 years (Stewart 1993). At least 25% (4/16) died within about 2.5 years. A fourth relocation effort was conducted in Nevada. Several tortoises were moved to an area immediately adjacent to a development site (Corn, 1994b, 1997). These 13 animals were moved to areas 2 km away, which was still within or very close to their pre-translocation home ranges. There was no difference in survival, but displaced animals had larger home ranges than did the residents. A preliminary analysis of a fifth study showed that mortality was significantly greater among guests (tortoises moved to a pen immediately adjacent to their capture sites) than hosts (resident tortoises; Weinstein 1993). All of these relocation studies covered short time periods and only measured movements and survival. None of them looked at reproductive success or long-term survival, two of the most important measures of success.

An ongoing project translocating tortoises many miles from their capture site apparently is showing success, but no reports or publications (other than abstracts) are available. Apparently, survivorship and reproduction are equivalent between relocated tortoises and resident tortoises (Nussear et al. 2000). Relocated tortoises did move more during their first year in the new site, but after that their movements were not significantly different than those of resident tortoises. Tortoises released in Utah also moved more than did resident tortoises there (Wilson et al. 2000). Both of these studies need further analyses and complete presentations before their results can be adequately evaluated. The success of desert tortoise relocations probably depends on distance of relocations, habitat quality, density of host population, rainfall, and health condition of the relocated and host animals.

Probably tens of thousands of desert tortoises are held in captivity throughout southern California, Nevada, and elsewhere, some were taken from the wild, others were reared in captivity. There are several documented cases of captive tortoises being released into the wild (Howland 1989, Ginn 1990), an activity that is now illegal. Release of captives may be detrimental to both captives and resident tortoises. Released captive tortoises may die (Berry et al. 1990) because they do not know how to fend for themselves in the wild; will not initially know where to find cover sites, good forage, sources of water, or essential minerals; and may not have genetic adaptations necessary to survive in the particular area. However, 25 formerly-captive tortoise were released in Nevada (Field et al. 2000). The animals were equipped with radio transmitters and followed for 14 months. The unpublished results indicate that movements and weights did not differ between released and resident tortoises. No adults died (released or resident) and 2 (out of 8) released juveniles died compared to neither of the two residents studied.

Of greater concern for the stability or recovery of tortoise populations is the possible impact of the released captives on resident (host) tortoises. The greatest likely effect is the introduction of disease to the wild population. URTD, the disease presently believed by many to have detrimental effects on several wild tortoise populations (see

"Disease" section, above), is commonly found in captive tortoises (Berry et al. 2002, Johnson 2002). Releasing into the wild tortoises that are infected with URTD may introduce the disease-causing bacterium, *Mycoplasma agassizii*, to previously uninfected individuals and populations. There is some evidence that the incidence of disease is greater in areas of known releases of captives and around urban areas where release or escape of captives is likely to be relatively frequent (Jacobson 1993, Berry pers. comm.). However, data on the rangewide incidence of disease have not been peer reviewed and are not generally available, so it is not possible to evaluate this hypothesis.

Desert tortoises have been manipulated in many ways as part of scientific studies. They have been probed, stuck with needles, affixed with transmitters, implanted with transponders, weighed, measured, pulled and sometimes dug out of burrows, tom name a few. All manipulative research involving desert tortoises must be permitted by USFWS to ensure that risk of harm to the tortoises is minimized. USFWS closely evaluates methods and qualifications of researchers before issuing a permit. There is very little written on the effects of research manipulation. In a preliminary analysis from one study, Weinstein (1993) reported that significantly fewer animals whose blood was sampled on a regular basis subsequently died compared to those whose blood was not sampled. In an evaluation of the possible effects of one research tool, Boarman et al. (1998) summarized from the literature on possible impacts to turtles of different ways of attaching radio transmitters. They concluded that there is little evidence of negative impacts of transmitters on turtles and particularly tortoises. Their concluded this partly because of paucity of published accounts of problems experienced. There are a few undocumented reports of individual animals dying from excessive bleeding following blood extraction and possible excessive mortality of animals that had blood extracted 3-4 times per year for several years, but none of this is reported in the literature and thus remains anecdotal. Kuchling (1998) hypothesized that X-rays, used to measure reproductive success, are hazardous to turtles. Using empirical data, Hinton et al. (1997) argued that x-rays are safe when extremely low dosages of radiation are employed, which can be accomplished with use of rare earth screens.

Invasive Plants

The introduction and proliferation of invasive plants is a continuing and increasing problem in the desert. The most common invasive plants found in tortoise habitat in the west Mojave Desert are cheatgrass (*Bromus tectorum*), red brome (foxtail chess, *Bromus madritensis rubens*), split grass (*Schismus barbatus*, and *S. arabicus*), redstem filaree (*Erodium cicutarium*), Russian thistle (tumbleweed, *Salsola tragus*), Sahara mustard (*Brassica tournefortii*), and fiddleneck (*Amsinckia tessellata*; Kemp and Brooks 1998). Fiddleneck is a native species to the U. S., but others are natives to Eurasia, Africa, or South America (Kemp and Brooks 1998, Esque et al. in press). By one estimate, alien annuals comprised 9-13% of all annual plant species but 3 species (red brome, split grass, and redstem filaree) comprised 66% of all annual plant biomass in one wet year (Brooks 1998, 2000). Other less common weedy species are listed in USFWS (1994, p. D21) and Kemp and Brooks (1998).

Invasive grass species (e.g., split grass) tend to have thin, filamentous roots that spread quickly and easily through shallow compacted soil where the surface crust has been broken (Adams et al. 1982a, b). The root structure allows plants with filamentous roots to quickly take advantage of small amounts of water in the soil following light rains and may allow them to outcompete native, non-weeds, which often grow slower, have thicker tap roots that are less efficient at pushing through dense, compacted soil (Adams et al. 1982a, b). There is some empirical evidence that split grass and red brome inhibit or prevent the growth of native plants, including fiddleneck (Brooks 2000), indicating that competition may be occurring and that the native plants are less available to foraging tortoises. However, in Nevada, Hunter (1989, cited in USFWS 1994, p. D22) found no correlation between native plant density and density of red brome.

In general, invasive plants tend to proliferate in areas of disturbance (Hobbs 1989), but the effect of disturbance may be weak compared to that of rainfall and soil nutrient levels. Density or biomass of weedy plants in the Mojave Desert may be higher in areas disturbed by ORVs (Davidson and Fox 1974), livestock (Webb and Stielstra 1979, Durfee 1988), paved roads (Frenkel 1970, Johnson et al. 1975), and dirt roads (Brooks 1998, 1999a). In a strictly correlative study, Brooks (1999a) found that the biomass of two annual exotic plants was weakly associated with levels of disturbance (disturbance was from ORVs and sheep grazing). Biomass of the introduced plants was also positively associated with soil nutrient levels and the proportion of total biomass and species richness (number of species in a given area) comprising exotic species was negatively associated with annual rainfall (i.e., relative proportion of exotic annuals was greater in years with low annual rainfall).

An additional factor that may facilitate proliferation of alien plants is increased nitrogen deposition from airborne pollutants (Allen et al 1998). Nitrogen, in the form of nitric acid and nitrate from automobile exhaust, deposits on plants and soil downwind from urban areas (Fenn et al. 1998) and perhaps from roads. Brooks (1998) has shown experimentally that the addition of nitrogen to west Mojave soil increases the biomass of brome and split grass thereby potentially increasing their competitive advantage over native plants (Eliason and Allen 1997). The effect ORV-based exhaust has on desert vegetation has not been established.

It is often stated that non-native plants are of lower nutritional quality than native species preferred as forage by tortoises, but this is not always the case. The difference in nutritional quality may have more to do with the type of plant (e.g., grass versus forb, Nagy et al. 1998) or annual differences in nutritional quality related to precipitation (Oftedal 2001). For example, the non-native split grass, which is often eaten and sometimes preferred by tortoises (Esque 1994), has been shown empirically to deplete tortoises of nitrogen and phosphorus and water and cause weight losses (Avery 1998, Nagy et al. 1998, Hazard et al. 2001), but so does the native Indian rice grass (*Achnatherum hymenoides*, Nagy et al. 1998). Avery (1998) also demonstrated that split grass was lower in overall quality, crude protein, essential amino acids, water, and vitamin concentrations and higher in fiber and heavy metal concentrations than three non-grass species measured (one introduced and two native forbs). The introduced forb, redstem filaree, had higher aluminum and iron concentrations, but was otherwise similar to native forbs. Where lower-quality weedy grasses can outcompete preferred higher-quality forbs (Brooks 2000), forbs may be less available to tortoises, tortoises would have to eat the lower quality invasives, and they would then suffer from a nitrogen and phosphorus (or other nutrient) deficiencies (Hazard et al. 2001). This speculation requires further testing.

Mechanical injury from invasive grasses has been observed with instances of the sharp awn of *Bromus rubens* being stuck in the nares of tortoises as well as impacting the food in the upper jaws of the tortoises (Medica, pers. comm.). The interactive effect that invasives and fires have on tortoises was discussed in the "Fire" section, above.

Landfills

There are approximately 27 authorized sanitary landfills and an unknown number of unauthorized, regularly used dumpsites in the California deserts. In the West Mojave Desert, there are 11 authorized landfills. The potential impacts landfills have on tortoise populations include: loss of habitat, spread of garbage, introduction of toxic chemicals, increased road kills from vehicles driving to or from the landfill, proliferation of predatory raven populations, and possible facilitation of increases in coyote and feral dog populations. Other than for raven predation, there are virtually no data to evaluate most of these possible threats.

Loss of habitat to landfills is relatively minor except when viewed in the context of habitat degradation and fragmentation caused by the myriad of human developments that are proliferating in the desert. Spread of garbage probably poses a very small problem for tortoise populations (see "Garbage and Litter" section, above), but there are no data available to evaluate this. The possible effect of toxic chemicals in general is treated in the "Disease" section, above, but toxins from sanitary landfills are likely to have very little effect on tortoise populations. Modern sanitary landfills are designed to prevent the seepage of toxic chemicals and present a very low level (or probability) of risk, and any seepage from these or less optimally operated landfills would probably affect a very small proportion of tortoises. Landfills do generate methane gas, but because desert landfills are so dry, the generation of methane is extremely low and not likely to affect tortoises. Fugitive dust is probably a localized problem and generally minimized through frequent sprinkling of the dirt. Increase in road kills is probably proportional to the level of traffic, speed of vehicles, density of tortoises, and length of road. For most landfills, these factors are relatively low, so the impact of road kills on tortoise populations from vehicles going to landfills is probably relatively minor, but they do happen (LaRue and Dougherty 1999). However, several landfills are slated to be closed and converted to transfer or community collection stations. The garbage would be deposited into dumpsters or large compactors at these stations, then transported to a small number of larger regional landfills. This activity could increase the amount of traffic at these fewer landfills thereby increasing the number of road kills.

The greatest potential impact landfills have on tortoise populations is through their probable role in facilitating increased predation by ravens, and perhaps coyotes.

Ravens make heavy use of landfills for food (Engel and Young 1992, Boarman et al. 1995, Kristan and Boarman 2001). The food eaten probably helps ravens to survive the summer and winter, when natural sources of food are in low abundance (Boarman 1993, in prep.). As a result, more ravens are present at the beginning of their breeding season (February - June) to move into tortoise habitat, nest, raise young, and feed on tortoises. Healthier ravens are more likely to raise chicks successfully, who in turn will move to the landfills and experience higher than normal levels of survival, and the cycle continues. Predation by ravens is probably relatively low immediately around landfills where tortoise populations are relatively low, but increase as ravens disperse to distant nest sites (Kristan and Boarman 2001). See the "Predation" section, below, for more details.

Livestock Grazing

Grazing by livestock (cattle and sheep) is hypothesized to have direct and indirect effects on tortoise populations including: mortality from crushing of animals or their burrows, destruction of vegetation, alteration of soil, augmentation of forage (e.g., presence of livestock droppings, and stimulation of vegetative growth or nutritive value of forage plants), and competition for food.

Reduce Tortoise Density

There are very few data available to determine if grazing has caused declines in tortoise populations. The Beaver Dam Slope, Utah, was grazed heavily by sheep until 1950's and cattle are still grazing there today (Oldemeyer 1994). Tortoise populations on the Beaver Dam Slope were estimated at 150 tortoises/mi^2 (Woodbury and Hardy 1948), but, using very different methods, the population apparently dropped to 34-47/mi^2 in 1986 (Coffeen and Welker 1987, cited in Bury et al. 1994). The reductions have been attributed to grazing, but another cause may include the potential spread of disease from captive tortoises released in the area (Luke et al. 1991). High mortalities and population declines in Piute Valley, Nevada, have also been attributed to grazing (Mortimer and Schneider 1983, and Luke et al. 1991), but 1981 was a drought year and a high level of recent mortalities may have occurred. Such was the case in Ivanpah Valley where 18.4% of radio-transmittered tortoises died (Turner et al. 1984). It is interesting to note that there appeared to be more tortoise mortalities in the section of the Piute Valley study area that experienced lower levels of recent cattle grazing (Mortimer and Schneider 1983), but the data are insufficient to make a definitive judgement. No population trends in California have been attributed with hard data to livestock grazing.

An alternative hypothesis, proposed by Bostick (1990), is that tortoise population declines paralleled declines in cattle grazing throughout the West that began in 1934 with the implementation of the Taylor Grazing Act. Unfortunately, there are no reliable data to test this hypothesis. But its underlying assumption, that tortoises depend on cattle dung for protein, has no empirical support (see "Cow Dung as a Food Source" section, below).

Direct Impacts

CRUSHING TORTOISES

Some observations of tortoises being crushed by livestock exist in the literature, but often with little or no data to allow in-depth evaluation. Berry (1978, p. 28) stated that "smaller tortoises can be crushed easily by cattle or sheep," but provided no data to support the statement. Berry (1978, pp. 19-21) also reported that "a small two-to-three-year old tortoise with a hole through its shell was found near a temporary watering trough near the DTNA. It appeared to have been killed by sheep within the last few days; the hole in the shell was about the size and shape of a sheep's hoof." Ravens also peck holes in the shells of young tortoises; insufficient information was provided to know if the hole was inconsistent with raven predation. Ron Marlow (pers. comm., cited in Berry 1978) described the disappearance of a marked juvenile tortoise and its small burrow by the trampling by sheep. Apparently the marked tortoise was never observed again, so Marlow determined the sheep killed it. The tortoise may have been killed when sheep trampled the burrow. However, marked juveniles are often never seen again, so the tortoise either survived or died from one of many causes. Any one of these anecdotes may be a true indicator of the nature of tortoise-cattle interactions, but the information provided is inadequate to allow for rigorous evaluation and are very susceptible to alternative explanations.

Sheep and cattle may not step on tortoises because they are very cautious of stepping on uneven ground (rocks, bushes, etc.) for fear of losing their footing. This view is supported by the paucity of documentation of tortoises being crushed by cattle and sheep. One published paper (Balph and Malecheck 1985) reported a test of a related hypothesis: cattle will avoid stepping on clumps of bunchgrass because the clumps form an uneven surface that may cause the cow to trip. Cattle significantly avoided crested wheatgrass (*Agropyron cristatum*) tussocks, avoidance was independent of cattle density, and taller tossocks were less apt to be trampled than short ones. Out of 288 hoofprints recorded, 15 (5%) were on tussocks. This well designed study lends support to the contention that cattle will try to avoid stepping on tortoises, at least large tortoises, but clearly tortoises are not grass tussocks. However, this speculation can be countered by the equally plausible contention that the study's results only shows that cattle will avoid stepping on food; they have no bearing on the propensity for sheep to step on non-food items (e.g., juvenile tortoises).

Sheep, on the other hand, may step on many juvenile tortoises, but appear to avoid stepping on subadult and adult tortoises. Tracy (1996) provides an analysis of data from an aborted BLM study. Without providing details of methods, Tracy (1996) reported that 20% of the Styrofoam model juvenile tortoises placed in natural habitat were trampled by sheep, 87% of those trampled models were crushed. Sheep damaged only about 3% of the subadult models and about 2% of the adult models.

CRUSHING BURROWS

No one has rigorously evaluated whether livestock crush a significant proportion of tortoise burrows. Few cases in the literature document livestock trampling actual burrows and a small number of studies shows increased number of collapsed burrows following grazing. Nicholson and Humphreys (1981) measured impacts of sheep grazing immediately after a band of 1000 sheep passed through their West Mojave study site for 12 days. Sheep trampled and partly collapsed a burrow with an adult female inside; apparently the tortoise was unharmed. Sheep completely destroyed the burrow of a juvenile tortoise while the animal was inside; the field workers extracted the unharmed tortoise. The burrow of an adult male was damaged probably with no tortoise inside. On re-examination of burrows found prior to grazing, 4.3% (7/164) were totally destroyed and 10% were damaged after sheep grazed in the area. Most damaged burrows (86%) were in moderate to heavily grazed areas and were relatively exposed. Most burrows placed beneath shrubs escaped damage (Nicholson and Humphreys 1981). This was an observational study. Webb and Stielstra (1979) reported observing crushed tortoise burrows on the south slope of the Rand Mountains in the western Mojave, but gave no data or additional details. In a report on grazing near the DTNA, Berry (1978) reported that sheep trampled most shallow burrows and pallets that were in the open (no numbers were given), and they also crushed and caved in those near the edges of or within shrubs. Berry (1978) also reported that "cattle and sheep frequently trample shallow tortoise burrows," but provided no data. She further speculated that damage to burrows might be deadly to a tortoise that reaches it on a hot morning only to find it unusable. This is a reasonable expectation based on tortoise behavior and thermal ecology, but no supporting data are available. Avery (1997) found significantly more damaged burrows outside of a cattle exclosure versus inside and also found that tortoises outside the exclosure spent more nights in the open, presumably because many of their burrows were collapsed. There is one account of a tortoise burrow being collapsed by a cow in Utah (Esque pers. comm.). A tortoise was found crushed inside.

Tracy (1996) provided an analysis of data from 2 unpublished BLM studies on the effects of sheep grazing on tortoise burrows: the Tortoise and Burrow Study (TABS study) and Styrofoam model tortoise study (Goodlett unpubl.). The TABS study (cited in Tracy 1996) evaluated the condition of tortoise burrows before and after grazing inside and outside of areas grazed by domestic sheep in the Mojave Desert. They found that 2.5% (8/315) of the tortoise burrows were completely destroyed, which was significantly more than before grazing and more than were destroyed outside the grazing area. In the Goodlett study (unpubl.; cited in Tracy 1996), 3.7% (36/969) of the artificial burrows dug to look like desert tortoise burrows were destroyed after grazing. Significantly more juvenile and immature burrows were destroyed compared to adult burrows and destruction was greatest in the open spaces between shrubs. The proportion of burrows destroyed in these two studies and Nicholson and Humphreys (1981) were not significantly different (Tracy 1996).

Indirect Effects

A commonly held assertion is that the Mojave desert plant species and communities evolved in the presence of, and are probably adapted to, a rich fauna of Pleistocene herbivores (Edwards 1992a, 1992b). Therefore, the argument continues, livestock grazing is compatible with present day plant assemblages, in part because Mojave plants respond to grazing by producing more vegetative material, thus becoming more vigorous in the presence of grazing. This argument has several flaws. First, most large herbivores that coexisted in the Mojave desert region 10,000-20,000 years ago likely primarily browsed leaves from woody shrubs, they did less grazing of grasses and herbaceous annual vegetation, like cattle, sheep, and tortoises primarily do (Edwards 1992a). Second, the mammals of the Late Pleistocene and Early Holocene Mojave existed under considerably different vegetative and climatic conditions ago (Van Devender et al. 1987). A major climatic and vegetative transition occurred between 11,000 and 8,000 years ago. It was more mesic and the area was not a desert. The present vegetation assembly, dominated by creosote shrub, did not arrive in the Mojave Desert region until approximately 8000-10,000 years ago (Van Devender et al. 1987). Third, no one has any idea what density the Pleistocene grazers existed at, so grazing intensity is completely unknown. Thus, there is little justification for arguing that tortoises evolved in the presence of grazers and their survival is thus dependent on cattle, as a surrogate for their coevolved grazing species.

Soil Compaction

Grazing can affect soils by increasing soil compaction and decreasing infiltration rate, the capacity of the soil to absorb water. A lower infiltration rate means less water will be available for plants and more surface erosion may occur. In a review of studies investigating the hydrologic effect of grazing on rangelands, Gifford and Hawkins (1978) concluded that grazing at any intensity reduces the infiltration rate of the soil. Heavy grazing reduced infiltration rate by 50% and light to moderate intensities reduced infiltration by 25% over ungrazed; the differences are statistically significant. Contrarily, Avery (1998) found significantly greater compaction at a livestock water source, but no difference between protected and grazed areas away from the water source.

Soil compaction affects vegetation by reducing water absorption (thereby availability to plants) and making it more difficult for plants to spread their roots, particularly tap roots (Adams et al. 1982a, b). Growth and perhaps spread of split grass (*Schismus barbatus* and *S. arabicus*) is facilitated by compaction because of root structure. This may lead to a conversion in the vegetation community type and increased fire hazard. Although, fire spreads slowly and discontinuously with split grass compared to *Bromus* grasses (Brooks 1999b).

Empirical evidence shows that infiltration is higher in grazed areas. , Rauzi and Smith (1973) conducted a comparative experiment in the central plains of Colorado. They demonstrated that infiltration rate was significantly reduced by heavy grazing (vs. moderate and light grazing). Infiltration rate was significantly correlated with total plant material on the surface (standing crop) in two of the three soil types tested. Species composition was different. Experimental water run-off tests showed moderate grazing areas had 7 times the runoff of light grazing areas and heavily grazed areas had 10 times the runoff as lightly grazed areas. In the Mojave Desert of Nevada and Arizona, signs of increased soil compaction were evident in grazed areas compared to ungrazed areas between highway and highway right-of-way fences (Durfee 1988). Avery (1998) measured soil type, bulk density, and infiltration in an exclosure that cattle were excluded from for approximately 12 years and compared them to grazed areas outside the exclosure. He demonstrated that soil in heavily trampled areas near water tanks was coarser, had higher bulk density, greater penetration resistance, and lower infiltration rates (all are measures of compaction) than in the protected area.

Although they did not measure compaction or infiltration, Nicholson and Humphreys (1981) quantified the proportion of soil disturbed after a band of 1000 sheep spent 12 days foraging and bedding within a 1.6 km^2 study plot. They estimated that 80% of the soil in bedding areas was disturbed, 67% in watering areas, 37% in grazing areas, and 5% in areas not used by sheep. Soil was considered disturbed if the surface crust was broken or missing and was independent of cause. This non-replicated observational study had a control, did not document what effect the measured disturbance had on vegetation or soil parameters, but did suggest the extent of surface disturbance caused by the grazing.

In a comparison of soil conditions following sheep grazing in the Western Mojave, Webb and Stielstra (1979) noted disruption of soil crusts in intershrub spaces and on the coppice mounds of creosote bushes. Surface strength (a measure of compaction) was significantly greater in grazed vs. ungrazed areas, particularly in the upper 10-cm of the soil. Bulk density and moisture content did not differ, perhaps because of the high gravel content of the soil or compaction in both areas from grazing activity in previous years.

CHANGES IN SOIL TEMPERATURE

Another potential indirect effect of livestock grazing on tortoise habitat is alteration of soil temperature due to change in vegetation structure or soil compaction. Steiger (1930 cited in Luke et al. 1991) measured a significant increase in soil temperature at depths of 2.5, 7.5, and 15 cm in clipped versus unclipped plots. Browsing of shrubs may also alter soil temperature, but in unexpected ways. Using models that accurately duplicated the thermal profiles of desert tortoises, Hillard and Tracy (1997), a graduate student from University of Nevada, Reno, found that soils were cooler beneath shrubs with sparse and open undercanopies and hotter when the undercanopy was entirely closed. Apparently, the open undercanopy allowed cooling by both shade and wind, whereas closed undercanopies trapped hot air. Hence, if livestock browse, graze or otherwise reduce density of the undergrowth of a shrub while leaving the canopy with intact shading properties, then soil temperatures may be reduced. Alternatively, if grazing also reduces the shrub's canopy, then soil temperatures may increase. It is unknown what effect grazing-induced changes in soil temperature might have on tortoises. The temperature during incubation (Spotila et al. 1994) determines sex of tortoises: incubation temperatures above 89.3°F result in females, and below result in males. Although this has not been tested in the field, it is possible that significant increases in soil temperature resulting from grazing-induced vegetation changes may significantly skew the sex ratio of the tortoise population in favor of females and vice versa. Also, Spotila et al. (1994) found that hatching success was highest for eggs incubated between 78.8°F and 95.5°F.

CHANGES IN VEGETATION

Grazing by cattle can alter vegetation in several ways: damage from trampling, change in species composition perhaps resulting in type conversion (change in plant community type), and introduction of invasive plants.

TRAMPLING OF VEGETATION AND SEEDS

Livestock may cause direct damage to vegetation when they step on or push into shrubs and herbaceous annuals, and this impact was measured in a few studies. In the west Mojave Desert, none of the perennials on plant transects where sheep grazed were trampled, whereas 17% found in the bedding area were trampled (Nicholson and Humphreys 1981). Webb and Stielstra (1979) reported that sheep trample creosote bush when seeking shade to bed in. Annuals, which are prevalent on coppice mounds beneath creosote, were also trampled or eaten. As noted above, Balph and Malechick (1985) provided empirical evidence that cattle usually avoided stepping on clumps of crested wheatgrass, but still stepped on them 5% of the time.

Trampling by livestock may help to bury seeds and improve germination through their trampling action. In sagebrush scrub of northern Nevada, Eckert et al. (1986) found that light trampling increased germination of perennial grasses, but not perennial forbs, and heavy trampling decreased emergence of perennial grasses while increasing emergence of sagebrush and perennial forbs. Cattle grazing in Chihuahua Desert grassland enhanced revegetation by non-native grasses, but rain may have confounded the results (Winkel and Roundy 1991). Unfortunately, no similar studies from the Mojave Desert are available. However, biomass of seeds in the soil seed bank was significantly higher inside compared to immediately outside the DTNA, a 38 mi^2 fence enclosed preserve, where sheep grazing and ORVs had been excluded for 15 years (Brooks 1995); this in spite of there being more seed-eating rodents inside the DTNA. The biomass of annual vegetation, including the introduced species, was also greater inside the DTNA, but the total biomass of natives was proportionally higher inside than outside. Several other uses occurring outside the DTNA were absent from inside the preserve, thus the differences cannot be attributed solely to grazing. However, the changes noted are the expected effect of removal of surface disturbance from the reserve.

Near the DTNA, sheep trampled and uprooted perennial shrubs, such as burrobush (*Ambrosia dumosa*), goldenhead (*Acamptopappus sphaerocephalus*), and

Anderson thornbush (*Lycium andersoni*). "Even large creosote bushes (*Larrea tridentata*) were uprooted" (Berry 1978, p 512). "In many areas near stock tanks [in Lanfair Valley, California] the ground is devoid of vegetation for hundreds of meters. Trailing is heavy and damage extensive within 4.6 to 6.4 km of the tanks" (Berry 1978, p. 512). These reports are anecdotal; no data or additional details were provided.

PLANT COMMUNITY CHANGES

As early as 1898, range scientists observed that cattle ranges in the southwest were becoming overgrazed and urged that restorative actions were necessary (Bentley 1898). Since then, several studies have documented vegetation changes over the past century by comparing photographs or field notes taken in both centuries (Humphrey 1958, Humphrey 1987). The dominant change was a conversion from grass- to shrub-dominated communities (type conversion). Whereas livestock grazing has been implicated as an important cause for these changes, separation of the effect of grazing from the effects of fire suppression, rodents and other herbivores, competition, and climate changes is difficult (Humphrey 1958, 1987). Several studies compared grazed areas to nearby ungrazed areas particularly in southeast Arizona. They generally show a similar reduction in grass species in the grazed areas. Unfortunately, none of these studies occurred in the Mojave Desert and, because the grass-dominated ecosystem of southeast Arizona is very different from the non-grass deserts of California, there is little value in extrapolating from one to the other.

In 1980, the BLM created a 672-hectare cattle exclosure in Ivanpah Valley, eastern Mojave Desert of California, to determine the effects of cattle grazing on desert tortoises and their habitat. In the study establishing baseline data for a long-term comparison, Turner et al. (1981) found no significant differences between plots in biomass of annuals, weight or length of tortoises, proportion of reproductively active females, and tortoise home range sizes. Sex ratios and size classes of tortoises were comparable between the two plots. The lack of differences could be attributed to: (1) low use by cattle of the non-excluded area in both years of the study; 2) tortoise and vegetation recovery, if they are to happen, are likely to take much longer to be observable; and (3) sample size (n=1) too small to detect differences. Changes in tortoise weight with time, estimated clutch sizes, and concentrations of some nutrients in some plant species differed between plots, indicating that some differences existed between control and treatment at the start of the study. Over so short a time frame, differences are likely due to prior spatial differences in habitat or populations rather than grazing treatment. There was a similar level of differences between control and treatment plots one year later (Medica et al. 1982).

Avery (1998) conducted a follow up study at the Ivanpah study plot in the early 1990's. Avery (1998) compared vegetation inside and outside the exclosure. Compared to the ungrazed exclosure, the grazed area had significantly larger creosote bushes, more dormant or dead burrobush, *Ambrosia dumosa* (a perennial shrub), fewer and smaller, galleta grass, *Pleuraphis [Hilaria] rigida* (a native, perennial grass) representing less biomass, more of the disturbance-loving shrub, *Hymenoclea salsola*, and lower diversity of winter annuals. They found significantly more desert dandelions (*Malacothrix glabrata*), a plant preferred by both cattle and tortoises, and a greater increase in basal area but not density of the native perennial galleta grass, *P. rigida*, in the protected area. *P. rigida* did increase in basal area over a 12 year period in the grazed area, indicating that level of grazing (0.31 - 2.60 animal unit months) does not cause mortality in *P rigida*. Biomass, cover, density, and species richness of annuals did not differ. Recovery of Mojave Desert vegetation following alteration by cattle grazing could be very slow (Oldemeyer 1994), so 12 years of exclusion may be insufficient to detect a more significant effect.

A recent study compared soil characteristics, vegetation, and tortoise density within and around three exclosures in the Mojave Desert, including 2 in the west Mojave (Larsen et al. 1997). They reported finding few differences between "grazed" and "ungrazed" plots in percent canopy cover, and the differences found were relatively minor. Grazing reduced native forb density and increased soil compaction. Numbers of live tortoises, tortoise carcasses, and tortoise burrows were no different between grazed and ungrazed areas. Details provided were insufficient to adequately evaluate the methods or results and virtually no statistical analyses were provided.

Durfee (1988) compared structural features of the plant community between ungrazed areas along fenced highways and grazed areas outside of the right-of-way fences. A greater proportion of introduced plants, more bare ground, fewer perennial grasses, and lower spatial heterogeneity in species composition occurred in the grazed areas (see also Waller and Micucci 1997).

As cited above, Brooks (1995) found significantly higher annual plant and seed biomass in the DTNA, an area protected from sheep grazing, compared to an area outside the preserve. Berry (1978) characterized the qualitative effect of sheep grazing near the DTNA: "sheep removed almost all traces of annual forbs and grasses; the desert floor appeared more devoid of herbaceous growth than in drought years." No further data were provided in the latter report.

In all of these studies, spatial differences obtained in soil, weather, and vegetation may be independent of cattle grazing. Furthermore, the size of exclosures may be insufficient to allow the ecosystem to function independent of grazing activities outside the exclosure (which is probably not a big problem at the DTNA, studied by Brooks 1992). Furthermore, many of the above studies, particularly the older and observational ones, were reporting on the effects of long-term heavy grazing, whereas grazing regimes being implemented today are generally much lighter (Oldemeyer 1994).

Water for cattle is usually provided at specific points, at either springs or troughs. Because they will only wander a certain distance from the water source, affect of cattle on the environment will be greatest immediately around the water source and will decrease with distance (e.g. Avery 1998). Fusco (1993), Fusco et al. (1995), Bleecker (1988), and Soltero et al. (1989) recorded significant increases in biomass and density of grasses and other species with distance from water sources. Changing the location of water sources would have the effect of reducing the intensity of impact around each water source, but may increase the impacts at other sites. It is unknown if impacts would be below the (unknown) threshold for significant effect on the environment.

The impact of sheep grazing has been studied only once. In an observational study, Nicholson and Humphreys (1981) noted that areas not grazed by sheep had 2.3 times more cover and 1.6 times higher frequency of annual plants than in sheep bedding areas and 1.8 times more cover and 1.3 times higher frequency than grazed areas. Annual plant cover decreased by 70% in a heavy-use area compared to 50% in a light-use and 40% in a non-use area before grazing versus after grazing one month later. They also found a 96-99% reduction in annual plant cover between April and June in areas receiving heavy and light grazing by sheep. None of the perennials on plant transects where sheep did not graze showed damage after sheep left the area; 18% in the grazed area were damaged and 91 to 99% in the bedding areas were damaged. Apparently, trampling caused most of the damage in the bedding areas whereas most in the light-use area was from browsing. However, differences may be caused by other factors such as soil that may have differed between the sites independent of grazing pressure. Rather than using exclosures, the sheep and herder were allowed to select the areas they grazed. Hence, the sheep avoided ungrazed treatments for this study. This may have biased the results since there may be inherent differences in these areas that caused the sheep to avoid them.

An often cited benefit of grazing is "compensatory growth," growth of plant tissue following clipping, removal, or damage to plants resulting in increased growth or vigor (e.g., Bostick 1990, McNaughton 1985, Savory 1989). The concept is controversial, has gained little empirical support in semi-arid grasslands and ranges (Detling 1988, Bartolome 1989, Weltz et al. 1989, Wilms et al. 1990), may only be viable in wet, fertile, monocultural environments (Painter and Belsky 1993), and has not been tested in the Mojave Desert (e.g., Painter and Belsky 1993). What little evidence exists from the Mojave Desert fails to support the compensatory growth hypothesis. Avery (1998) found that *Pleuraphis [-Hilaria] rigida*, a native grass consumed by both cattle and desert tortoises, was significantly smaller in grazed versus ungrazed areas. More *Ambrosia dumosa*, which is sometimes eaten by cattle in drought years (Medica pers. comm.), was found dead or dormant in the grazed compared to ungrazed plots. Creosote (*L. tridentata*) was larger in grazed areas, but is consumed by neither cattle nor tortoises (Avery 1998).

Invasive Plants

Grazing has been implicated in the proliferation of invasive plants in the Mojave Desert (Mack 1981, Jackson 1985, Brooks 1995). Webb and Stielstra (1979) noted that *Schismus* and *Erodium* densities remained unchanged between a grazed and ungrazed area probably because they have an adaptive tolerance to environmental disruption such as soil compaction thus giving them a competitive edge over many native annuals. Berry (1978) reported that the heavily grazed Lanfair Valley "now contains a high percentage of weedy, invader, perennial species typical of overgrazed desert lands," but provided no data. Bostick (1990) argued that cattle grazing helped tortoise populations by aiding the spread of cacti. Some evidence from outside the Mojave suggests that grazing does aid in the spread of cacti, but the evidence is equivocal. Also, tortoises do eat cacti, which may be an important source of water and nutrition during drought periods (Turner et al. 1984, Avery 1998). But, the evidence in support of Bostick's hypothesis is weak.

COMPETITION

An important effect livestock grazing may have on tortoise populations is competition for food. Because of the enormous differences in size and energy requirements of the two species, the competition, if it occurs, is likely to be heavily asymmetric, with cattle affecting the tortoise populations, but probably not the converse. Three conditions must be met for asymmetric competition to occur: overlap in use of some resource (e.g., food), the resource must somehow limit or constrain one or both species in question, and use of the resource by one species must negatively affect the other species (Begon et al. 1990). Some data exist to help determine if competition for forage exists between cattle and tortoises, but less exist for sheep.

Many studies provide qualitative insights into forage species of tortoises (Woodbury and Hardy 1948, Burge and Bradley 1976, Hansen et al. 1976, Hohman and Ohmart 1980, Luckenback 1982, Nagy and Medica 1986) and three major studies quantified diet and forage selection in desert tortoises (Jennings 1993, Esque 1994, and Avery 1998). Tortoises primarily eat annual herbs in the spring and switch to grasses, perennial succulents (cacti), and dried annuals later in spring and early summer (Avery 1998). Tortoises are active again in the late spring and early fall as temperatures cool. As a result of localized late summer rains, sporadic green up of the vegetation can occur. At this time annuals germinate and bunch grasses (e.g., *Hilaria rigida*) green up and set seed. Cattle then eat the bunch grasses (Medica et al. 1992). In a drought year, tortoises in Ivanpah Valley consumed little food other than cacti during the latter part of the season (Turner et al. 1984). Thus, cacti may serve as a reserve supply of energy, more importantly as a potential source of water.

Four studies quantified plant foods eaten by cattle in the Mojave Desert (Coombs 1979, Burkhardt and Chamberlain 1982, Avery and Neibergs 1997). Avery and Neibergs (1997) followed cattle on horseback in the eastern Mojave Desert. By recording the species of plant and number of bites taken by the free-ranging cattle they found that foods chosen by cattle varied with season. In winter cattle primarily ate the perennial grass, big galleta grass (*Pleuraphis* [=*Hilaria*] *rigida*) and dried annuals from the previous spring (Medica et al. 1982, documented that cattle and tortoises eat perennial grasses in fall). Contrarily, Burkhardt and Chamberlain (1982) found perennial shrubs to predominate the diet of cattle in winter, annual grasses and green forbs did so in spring. Coombs (1979) found that cattle in the eastern Mojave of Utah particularly ate *Bromus* sp., *Ephedranevadensis*, and *Eurotia lanata* and ate perennial grasses considerably more often than expected based on their relatively uncommon presence. All of these studies illustrated that cattle in the desert eat diverse foods and that the foods eaten vary with season, locality, and availability.

Several studies provided evidence that tortoise and cattle diets overlap (Coombs 1979, Sheppard 1981, Medica et al. 1982, Avery and Neibergs 1997, Avery 1998), three of which did so quantitatively. Coombs (1979) and Sheppard (1981) used fecal samples, which are biased because they overestimate food items that contain large undigestible parts (e.g., silica-containing stems of grasses) and underestimate items that are highly digestible (e.g., moist forbs). Sheppard (1981) showed that plaintain (*Plantago insularis*), filaree, and *Schismus* experienced the highest levels of overlap , but overlap varied considerably between months and years. Coombs (1979) found that overlap existed, but neither study provided a species-by-species comparison or an explanation of how overlap was calculated. *Camassonia boothii, Malacothrix glabrata, Rafinesquia neomexicana, Schismus barbatus*, aud *Stephanomeria exigua* were major forage items of both cattle and tortoises in Ivanpah Valley (Avery and Neibergs 1997, Avery 1998). Diet overlap between the two herbivores was greatest in early spring (38% Vs 16% in late spring, Avery and Neibergs 1997, Avery 1998).

Three studies provide data on forage overlap between sheep and tortoises. Webb and Stielstra (1979) reported that in the western Mojave Desert, sheep primarily ate herbaceous vegetation from the coppice mounds around the base of perennial shrubs. By comparing biomass of plants in a grazed area versus a nearby ungrazed area, they determined that three species were primarily removed: *Phocelia tanacetifolia, Thelypodium lasiophyllum*, and *Erodium cicutarium*.. Shrubs browsed by the sheep included *Ambrosia dumosa*, *Grayia spinosa*, *Haplopappus cooperi*, and *Acamptopappus sphaerocephalus*. Cover, volume, and biomass of these shrubs were significantly lower in grazed vs. ungrazed areas. However, because measurements were not taken before grazing it is possible that some differences may have existed before grazing commenced. Hansen et al. (1976) estimated that 15% of sheep diet in the western Mojave was composed of grasses and 52% of desert tortoise diets was composed of grasses. Nicholson and Humphreys (1981) reported several species of plants, particularly flowering annuals and burrobush (*Ambrosia dumosa*), that were highly used by sheep, but provided no quantitative data. Several species eaten by sheep were also eaten by tortoises iucluding: split grass (*Schismus arabicus*), checker fiddleneck (*Amsinckia tessellata*), desert dandelion (*Malacothrix glabrata*), filaree (*Erodium cicutarium*), Fremont pincushion (*Chaenactis fremontii*), Parry rock pink (*Stephanomeria parryi*), chickory (*(Rafinesquia neomexicana*), snake's head (*Malacothrix coulteri*), red brome (*Bromus rubens*).

Only two studies directly tested for competition between tortoises and livestock. In an extensive study, Avery (1998) showed that cattle and tortoise diets overlap (38% in early spring, 16% in late spring). He also demonstrated that tortoise foraging was altered in the area where both species co-occurred. In late spring in the absence of cattle, tortoises primarily ate herbaceous perennials (91% of diet), whereas in the grazed areas, tortoises primarily ate annual grasses (59%) followed by herbaceous perennials (21%). The species of herbs also differed: in the exclosure tortoises preferred desert dandelion (*Malacothrix glabrata*), whereas in the grazed areas they ate primarily the exotic grass, splitgrass (*Schismus barbatus*). The availability of desert dandelion was significantly higher in the ungrazed area, which indicates a response to grazing, and of splitgrass was equivalent in the two areas. In one dry year, tortoises spent significantly more time (approximately three times more) foraging in the grazed than in the protected areas, presumably in search of nutritionally-adequate food to fill up on. Thus, two of the three conditions necessary to confirm that cattle compete with tortoises for food were clearly supported empirically. The final condition, that one species must negatively impact the other, was also demonstrated, but more indirectly. In a separate, independent study, tortoises eating primarily *Schismus barbatus* have been shown to be put in a negative water and nitrogen balance (Nagy et al. 1998), which could increase mortality particularly during periods of extended drought (Peterson 1994a, Avery 1998). Furthermore, Henen (1997) demonstrated that lower nitrogen intake reduces reproductive output in female tortoises. A long-term comparison of differential survival and reproductive success of tortoises within and outside an exclosure would be an excellent empirical test of the effect cattle grazing has on tortoise populations.

Tracy (1996) found that in years of very low annual productivity, tortoises lay fewer eggs. They also found that cattle foraging reduced tortoise forage abundance enough to cause tortoises to lay fewer eggs than normal. The conclusion is that, in low rain years, cattle may remove enough forage to reduce tortoise reproductive output, thus competition occurs in those years. The authors did not track hatchling success to determine if the fewer eggs still resulted in the same number of successful hatchlings.

COW DUNG AS A FOOD SOURCE

Bostick (1990) argued that declines in tortoise populations is caused by a reduction in the availability of cow dung which has declined with the reduction in numbers of cattle grazing in the southwest. He argued that cow dung is an important source of food for tortoises. However, Avery (1998) studied tortoise foraging behavior where tortoises coexisted with cattle. He observed over 30,000 bites of items and observed only 231 bites of cow dung. Esque (1994) also observed over 30,000 bites on food objects. He reported that 107 of them were of feces, but none were from livestock. Furthermore, Allen (1999) evaluated the nutritional quality of cow dung and found it to be deficient for tortoises. In fact, even when cow pies were their only choice of food for one month, most tortoises (71%) refused to eat. Those that did eat, assimilated virtually none of the nitrogen. Thus, whereas Bostick (1990) presented an intriguing alternative hypothesis for tortoise population declines, there is no empirical support for its basic assumptions.

Summary

Surprisingly little information is available on the effects of grazing on the Mojave Desert ecosystem (Oldemeyer 1994, Rundel and Gibson 1996, Lovich and Bainbridge 1999). Differences in rainfall patterns, nutrient cycling, and foraging behavior of herbivores and how these three factors interact make applications of research from other areas of limited value in understanding the range ecology of the Mojave Desert. The paucity of information is surprising given the controversy surrounding grazing in the

Mojave and the importance of scientific information for making resource management decisions affecting grazing. Studies mostly from other arid and semi-arid regions tells us that grazing can alter community structure, compact soil, disturb cryptogamic soils, increase fugitive dust and erosion. Some impacts to tortoises or their habitat have been demonstrated, but the evidence is not overwhelming.

Military Operations

The California deserts were used for military exercises as far back as 1859 when Fort Mojave was first built (Krzysik 1998). The most extensive use was for World War II training when 18400 mi^2 (47105 km^2) in California and Arizona were designated as the Desert Training Center and used extensively for training with tank and armored vehicles. Today, four major, active military installations occur within the West Mojave and comprise a total of 4165 mi^2 (10663 km^2): Naval Air Weapons Station ("China Lake;" 1731 mi^2, 4432 km^2), National Training Center ("Fort Irwin;" 1016 mi^2, 2600 km^2), Air Force Flight Training Center ("Edwards Air Force Base;" 476 mi^2, 1218 km^2), and Marine Corp Air Ground Combat Center ("MCAGCC" or "Twentynine Palms;" 943 mi^2, 2413 km^2).

As outlined in the Recovery Plan (USFWS 1994), impacts to tortoise populations come from four basic types of military activities:

> "(1) construction, operation, and maintenance of bases and support facilities (air strips, roads, etc.); (2) development of local support communities, including urban, industrial, and commercial facilities; (3) field maneuvers; including tank traffic, air to ground bombing, static testing of explosives, littering with unexploded ordinance, shell casings, and ration cans; and (4) distribution of chemicals." (USFWS 1994, p. D14)

A fifth potential impact is above ground nuclear weapons testing, which took place in Nevada in the 1950s and 1960s.

<u>Construction, Operation, and Maintenance of Bases and Support Facilities</u>

All four major military bases in the west Mojave Desert each have facilitated the growth or development of large internal support communities. The development of these communities destroyed tortoise habitat and likely brought with them all of the other impacts generally associated with large human settlements (fragmentation, ORVs, release of disease, facilitation of raven population growth, domestic predators, etc.), each of which are discussed elsewhere in this report. There is some evidence that the tortoise population around China Lake declined within four decades following development of the base at China Lake (Berry and Nicholson 1984a). However likely this conclusion probably is, the data used were based solely on anecdotal observations (Bury and Corn 1995); and the data only show a correlation, not a cause and effect. Removal (translocation) of tortoises from construction sites, runways, and other heavy use areas to other parts of the desert occurs and may affect the tortoises moved (Berry and Nicholson 1984a; see "Handling and Deliberate Manipulation" section, above). Another impact is the fragmentation of the habitat by the apparent haphazard placement of facilities throughout major portions of habitat (pers. obs.).

Development of Local Support Communities

The four major military bases in the west Mojave Desert have facilitated the growth or development of large external support communities: Ridgecrest, Barstow, Lancaster, Palmdale, and Twentynine Palms, which each have problems for tortoises typical of large suburban areas in the desert (see "Urbanization and Development" section, below).

Field Maneuvers

Tank maneuvers cause some of the most drastic and long-lasting impacts to the Mojave Desert habitats. Extensive tank training operations were conducted in the 1940's and in 1964 over 17,500 mi^2 of desert (Lathrop 1983, Prose and Metzger 1985, Krzysik 1998) and even more intensive maneuvers are currently taking place within an 819 mi^2 area on Fort Irwin (Krzysik 1998) and on MCAGCC (Baxter and Stewart 1990). Direct mortality to tortoises is relatively rare or not often reported, but does occur (Stewart and Baxter 1987, Quillman pers. comm.). Tanks damage vegetation, compact soil, cause fugitive dust, and run over tortoise burrows and tortoises. The results are largely denuded habitat, and altered vegetation composition, abundance, and distribution (Wilshire and Nakata 1976, Lathrop 1983, Baxter and Stewart 1990, Prose et al. 1987, Krzysik 1998). Natural recovery can take a long time; 55 year old tank tracks can still be seen throughout many parts of the desert (Wilshire and Nakata 1976, Krzysik 1998). Krzysik (1998) reported a significant reduction in tortoise densities (62-81% over six years) in active training areas of Fort Irwin and no change or increases in densities in areas with light and no activity. The effect of tank maneuvers was highest in valley bottoms and progressively less in high bajadas, talus slopes, and rugged mountain ranges where training activities were considerably lower.

Bombing and other explosive ordinance cause impacts in some areas, but no documentation was found of their effect on tortoise populations or habitat.

Distribution of Chemicals

It has been suggested that diseases affecting tortoise shells may be caused by residual chemical remains left over from military operations, but the evidence is highly speculative (See "Disease" section, above).

Nuclear Weapons Testing

Between 1951 and early 1963, the U. S. Atomic Energy Commission detonated 100 atomic devices above ground at the Nevada Test Site, Nevada (U. S. Department of Energy 1994). From mid 1960s to early 1990s only underground tests were conducted. Resource Concepts Inc. (1996) argued that radiation released into the atmosphere during these tests might explain tortoise declines. They cited two anecdotal accounts, one of many sheep getting sick near Cedar City, Utah, and another of high Geiger counter levels around the mouth of a cow in the same area. They suggested that nuclear fallout might explain the presence of disease in tortoise populations. Beatley (1967) found only very low levels of radiation at a plant study plot 8 km east of a below-ground test blast and attributed vegetative defoliation to dust from heavy vehicular traffic on a nearby dirt road.

The University of California, Laboratory of Nuclear Medicine and Radiation Biology conducted experimental radioecology research studies in Rock Valley located along the southern boundary of the Nevada Test Site. These irradiation studies involved the chronic exposure of plants and animals from a centrally located 137 cesium source located atop of a 50-ft tower within a 21-ac fenced plot. Rundel and Gibson (1996) provided a brief summary of the results of the Rock Valley irradiation experiment. Beyond direct mortality from the test blasts, there was very little persistent effect of radiation on the surrounding lizard populations. Little long-term effect on the pocket mouse, *Perognathus formosus*, was found (Turner 1975). On the other hand, female lizards at Rock Valley were found to be sterile several years after the experiment began (Turner 1975, Turner and Medica 1977). There were five adult tortoises present throughout most of the study and four still remained in 2001 (Medica pers. comm.).

I could find no data that bear directly on the potential effects of nuclear weapons testing on tortoise populations. The map in Gallagher (1993) suggests that fallout was nearly nonexistent in the west Mojave (which is consistent with predominant wind patterns), where URTD is rampant (Berry 1997). Therefore, if there is an effect from testing, it probably cannot be a universal explanation for rangewide declines nor can it explain the markedly high losses and levels of disease documented in the west Mojave.

Noise and Vibration

The following is largely paraphrased from my contribution to the Desert Tortoise Recovery Plan (USFWS 1994). Anthropogenic noise and vibrations may impact tortoises in several ways including: disruption of communication, and damage to the auditory system. A body of peer reviewed scientific literature exists demonstrating how background noise may mask important vocal signals in insects and amphibians (e.g., Bushcrickets, *Conocephalus brevipennis*, Bailey and Morris, 1986; Green Treefrogs, *Hyla cinerea*, Ehret and Gerhardt, 1980). Hierarchical social interactions, hearing, and vocal communication have all been identified in desert tortoises (Adrian et al. 1938, Campbell and Evans 1967, Patterson 1971, 1976, and Brattstrom 1974, Bowles et al. 1999). Patterson (1976) identified eleven different classes of vocal signals used by desert tortoises in various of social interactions, but he did not demonstrate that animals who hear the signals react or change their behavior in any way, a necessary component in identifying communication. The signals are relatively low amplitude, have fundamental frequencies 200 Hz or lower, and harmonics that reach as high as 4500 Hz (Patterson, 1976).

The portions in the following excerpt from USFWS (1994) pertaining to desert tortoises is purely speculative with no direct empirical support for desert tortoises:

> " Many anthropogenic noises, such as automobile, jet, and train noises, cover a wide frequency bandwidth. When such sounds propagate through the environment, the high frequencies rapidly attenuate, but the low frequencies may travel great distances (Lyon, 1973). The dominant frequencies that remain after propagation correspond closely to the frequency bandwidth characteristic of desert tortoise vocalizations. Therefore, masking of these signals may significantly alter an animal's ability to effectively communicate or respond in appropriate ways. The same holds true for incidental sounds made by approaching predators; masking of these sounds may reduce a tortoise's ability to avoid capture by the predator. The degree to which masking by noise affects tortoise survival and reproduction depends on the physical characteristics (i.e., frequency, amplitude, and short- and long-term timing) of the noise and the animal signal, propagation characteristics of the sounds in the particular environment, auditory acuities of the tortoises, and importance of the signal in mediating social or predator interactions. There are no studies to test the masking effect of noise on tortoise behavior, but the effect is likely to be relatively low given that vocal communication is probably not extremely important in mediating social interactions and that noises loud enough to mask sounds important to tortoises are generally uncommon and short in duration. The only place the noise would be continuous enough may be alongside heavily traveled roads, where tortoise abundance is generally quite low.
>
> "Loud noises (and associated vibrations) may damage the hearing apparatus of tortoises. Little research has been performed on tortoise ears, but it is clear that tortoises are able to hear, and the relatively complex vocal repertoires demonstrated by tortoises suggests that their hearing acuity is similarly complex. Brattstrom and Bondello (1983) experimentally demonstrated that off-highway vehicle noise can reduce the hearing thresholds of Mojave Fringe-toed Lizards (*Uma scoparia*). Relatively short, single bursts (500 sec) of loud sounds (95 dBA at 5 meters) caused hearing damage to seven test lizards (Brattstrom and Bondello, 1983). Comparable results were obtained when desert iguanas (*Dipsosaurus dorsalis*) were exposed to one to ten hours of motorcycle noise (Bondello, 1976). It is likely that repeated or continuous exposure to damaging noises will cause a greater reduction in auditory response of these lizards. It is not unreasonable to expect loud noises to similarly impact the auditory performance of desert tortoises."

A study conducted by Bowles et al. (1999) showed very little behavioral or physiological effect on tortoises of loud noises that simulated jet over flights and sonic booms. They also demonstrated that tortoise hearing is fairly sensitive (mean 34 dB SPL) and was most sensitive to sounds between 125 and 750 Hz, well within the range of the fundamental frequency of most of their vocalizations. The authors concluded that tortoises probably could tolerate occasional exposure to sonic boom level sounds (140 dB SPL), but some may suffer permanent hearing loss from repeated long-term exposure to loud sounds such as from ORVs and construction blasts.

ORV Activities

Like most other threats, off road vehicle (ORV) activities may affect tortoise populations in multiple ways: direct mortality by crushing tortoises on the surface or in burrows, or indirect mortality through habitat alteration from soil compaction, vegetation destruction (direct or indirect via dust), or toxins from exhaust. However, different types of ORV activities will likely have different effects on tortoise populations. There are basically four categories of activity that may have very different impacts: free play where vehicles are not restricted to designated routes and cross travel or off-road and off-trail activity probably occurs regularly; non-competitive recreational uses outside of free play areas are limited to designated roads and trails with any driving off of those routes being illegal; competitive events are organized races that are restricted to designated open areas; and unauthorized cross-country travel for recreational or commercial (e.g., mining exploration) purposes. Hence in this report, ORV refers to motorized vehicle travel off of paved and graded dirt roads whether they are on ungraded dirt roads, trails, or cross country driving. ORVs can include dirt bikes, sport utility vehicles, all-terrain vehicles, sand rails, and any other type of motorized vehicle that travels such roads.

<u>Reduce Tortoise Density</u>

A number of reports document ORVs may directly kill tortoises (see below), however the data are insufficient to evaluate the extent of its overall impact on tortoise populations. We must rely more on other measures such as differences in tortoise densities between areas used by ORVs and those free from such activity. For example, Bury and Luckenback (1986) compared tortoise densities inside and outside of an ORV free-play area. They found 3.8 times more tortoises in a control area lacking ORV activity compared to a nearby open area and the animals were significantly heavier ($p<0.01$) in the control area. They also found 2.8 times the number of burrows, more of which were active, in the control area. Most of the burrows in the ORV area were in the section most lightly used by vehicles. The denser vegetation in the control area made searching much slower, hence 3.6 times more effort was spent searching the control area. The differences in number of tortoises are not likely to be a consequence of differences in search time because identical and consistent methods were used to sample each area (Bury and Luckenbach 1977). As this study was unreplicated (only one control, and one treatment area were surveyed), it is conceivable that the differences detected are due to causes other than ORV activity (e.g., soil or habitat differences or natural patchiness of tortoise populations).

Berry et al. (1986) compared tortoise populations inside of the DTNA and immediately outside where heavy ORV activity occurs. Using methods that are of questionable validity (Corn 1994a), they noted that significant declines occurred over a six-year period among juveniles and immatures in both areas, but that the declines were significantly greater in the adjacent area with more ORV activity.

Berry et al. (1994; for published abstract see Berry et al. 1996), compared evidence of human activity and tortoise sign (i. e., number of tracks, scat, and burrows, which is positively correlated to tortoise density; Turner et al. 1985) along 100 transects conducted in 1977-79 and 150 in 1990. They found that vehicle trails in 1990 were positively associated with areas classified as having low to medium densities of tortoises, but that numbers of vehicle trails and tracks were not directly correlated to actual number of tortoise sign. In one area, ORV activity had been stopped by BLM one year prior to the study, so vehicle tracks had been obliterated or were aged and did not accurately reflect the level of ORV activity the tortoise population had experienced over the past several years. Furthermore, the study lacked an adequate control site, but it is difficult to have good controls in a broad field study like this.

An indirect piece of evidence that ORVs reduce tortoise population density comes from Nicholson (1978). She reports on the findings of sets of transects walked at varying distances from the edges of several paved roads and highways in the Mojave desert. The study was designed to measure the effects of paved roads, not dirt roads or ORV travel on tortoise populations, thus is of little relevance to evaluating ORV impacts. She found that counts of tortoise sign increased with distance from paved roads. However, along Shadow Mountain Road, she found a reduction in tortoise sign 880 meters from the road edge, in an area with "excessive ORV use." She provided no statistical analysis of this observation, nor did she comment on the presence or absence of ORV activity along any of the 39 other transects she walked.

Direct Effects

CRUSHING TORTOISES AND BURROWS

Several accounts occur in the non-scientific literature of tortoises being crushed by ORVs, but most of these are anecdotal or unique incidents. In a popular account of ORV impacts to the desert environment, Luckenbach (1975) states: "I have personally found horned lizards, whiptails, zebra-tails, sand lizards, and tortoises crushed by ORVs;" no documentation or quantification was provided. Similar anecdotal statements were made in Berry and Nicholson (1984a) and Bury and Marlow (1973).

Berry and Nicholson (1984a) observed dead tortoises that were crushed in burrows that were apparently collapsed by ORVs, but no data or details were provided. Bury and Marlow's (1973) popular article about general impacts of ORVs on tortoises also makes the claim that burrows are crushed by ORVs, but provide no data. Fifteen burrows found in 1976 and 1977 in an ORV-use area were collapsed in 1985, their collapse being "related to ORV activity from trails through the area" (Bury and Luckenback 1986), although they gave no further indication of how they determined the cause of collapse. Woodman (1986) and Burge (1986) found no crushed burrows following the Parker 400 and Frontier 500 races, respectively.

Four studies quantified vehicle-related mortality on study sites with frequent ORV traffic. In her preliminary analysis of 1357 tortoise carcasses found on 14 permanent study plots for studying tortoise populations, Berry (1990 as amended) attributed approximately 57 (4%) to vehicles (some of the data were presented in Berry et al. 1986). It must be noted that 787 (58%) of the shells were not evaluated or were unclassifiable either because they bore no diagnostic characteristics or were too fragmented to analyze. Campbell (1985) found 2 vehicle-killed tortoises, one apparently killed by a 4-wheel vehicle on a dirt road inside the preserve and another killed outside the preserve by a sheep watering truck. In their comparative study of ORV impacts, Bury and Luckenback (1986) indicated that one immature tortoise was found crushed in a motorcycle trail. In a review of tortoise population dynamics, Marlow (1974) states that "nine recently crushed tortoises were observed in an area supposedly closed to ORVs. From tracks surrounding most of the carcasses there was little question as to the cause of their deaths."

It is the correspondence between tortoise and ORV enthusiasts' habitat preference that is likely responsible for some of the conflicts between the two. Jennings (1997) showed that tortoises spent significantly more time in washes, washlets, and on small hills. This is because their preferred food plants occurred in these habitats and they tend to burrow and travel more in washes and washlets than in other habitats. Jennings (1997) claims these habitats are also preferred disproportionately by ORV recreationists, but presented no supporting data.

Indirect Effects

Compaction of Soil

Soil becomes compacted, at least temporarily, when a motorized vehicle passes over it, and that compaction changes with the weight of the vehicle, soil type, and moisture content of the soil (Webb 1983). But, the affect this compaction has on tortoise populations depends on the lasting effect of compaction, its effect on vegetation and burrow digging abilities, how widespread the compaction is, and the respective effects on tortoise survival and reproduction.

Davidson and Fox (1974) investigated the effect a motorcycle dual sport race had on Mojave vegetation and soil. The soil, which was of similar type at both sites, was significantly denser and less porous at a pit area and alongside a trail than at a control site several hundred meters away. Significantly fewer plant species, fewer individuals, and less cover were found in impacted areas compared to the control site. However, the study was unreplicated. An increase in bulk density of the soil was measured in an evaluation of the impacts of the 1974 Barstow to Vegas Race (BLM 1975). However, many of the measurements were taken one week after a rain, so, because compaction is intensified on wet and moist soil (Webb 1983), the results may be unreliable.

Babcock and Sons (1973) found 10% or more increase in bulk density in disturbed versus undisturbed sites in alluvial wash, alluvial fan, and desert flat areas, but only a 3% increase in compaction in disturbed sand. Similarly, Wilshire and Nakata (1976) found sand dunes to be more resistant to compaction than playas or alluvial fans. Compaction was relatively light in heavily used dry washes and heavy in well used alluvial fans. Dry playas, which dry out fast after rains, resist compaction more than do wet playas (Wilshire and Nakata 1976), which are moist on or near the surface. Compaction on wet playas was measurable down to 15 cm or more.

In their manipulative experiment on the effect of vehicle type, number of passes, soil type, and soil moisture, Adams et al (1982a, b) measured soil compaction with a penetrometer. They found that compaction by a SUV was greater than that of a motorcycle. The SUV compacted wet soil significantly after only one pass on wet soil and after five passes on dry soil. The motorcycle compacted wet soil after 20 passes. Single passes by motorcycles on wet soil and SUVs on dry soils did not differ significant from the controls. The great variability in environmental conditions makes it difficult to make unambiguous generalizations.

Greater temperature extremes occurred in more compacted soils in heavy ORV use areas, probably from removal of vegetation and changes in soil characteristics from compaction (Willis and Raney 1971, Webb et al. 1978). This possible effect on soil temperature not only affects plant germination and growth, but may have interesting, if unexplored, implications for tortoise growth, development, and morphology. A further likely, but untested potential impact of soil compaction may be to make it difficult for tortoises to burrow, which would not only affect tortoises directly but would also reduce tortoises' role in reducing compaction through soil turnover (Prose et al. 1987).

Infiltration rate is a measure of the soil's ability to absorb moisture. More compacted soils have a lower infiltration rates so less water is available for plants (Webb 1983). Babcock and Sons (1973) found much lower infiltration rates on disturbed versus undisturbed desert sites, except in very sandy areas (dunes and washes). Webb (1983) measured 73% lower infiltration rate compared to a control site after 200 vehicle passes over wet sandy loam. The greatest decrease occurred after the first few passes. Infiltration rates of sands and clays are least affected by compaction, whereas loamy sands and gravelly soils are with a mixture of particle sizes are most affected.

Destruction of Cryptogamic Soils

Cryptogamic soils are important for reducing soil erosion, controlling water infiltration, regulating soil temperatures, fixing (catching and converting) atmospheric nitrogen, and accumulating organic matter (Cline and Rickard 1973, Paul i 1964, Rogers et al. 1966). Cryptogamic soils are collections of mostly symbiotic bacteria, algae, fungi, and lichen that live on or slightly below the soil surface and create a semi-permeable soil surface. They often occur in the open spaces between desert shrubs and help to facilitate seedling establishment and plant growth (St. Clair et al. 1984, DeFalco 1995).

ORVs, livestock, and other surface disturbances easily damage cryptogamic soils (Belnap 1996). Damage from compaction, even minor, can greatly reduce nitrogen fixation by the crust, an effect that sometimes increases rather than decreases with time since compaction (Belnap 1996). It is not certain how tortoises are affected by damage to cryptogamic soils and a 1980 review of the effects of ORVs on desert soils was inconclusive (Rowlands 1980). DeFalco (1995) found that, in the one season studied, tortoises selectively avoided foraging on plants growing on crusts. Although crusts fix nitrogen and the nitrogen can then be transferred to plants growing in close proximity to the crusts (Maryland and McIntosh 1966), concentration of nitrogen in tortoise forage plants were generally lower on cryptogamic soils (DeFalco 1995). However, many other nutrients are important to tortoises, and it is unknown if their concentrations are augmentated by cryptogams in associated tortoise forage plants. In non-tortoise habitat in southwest Utah, Belnap and Harper (1995) showed that nitrogen, phosphorus, potassium, calcium, magnesium, and iron concentrations were higher in some plant species growing on encrusted soils compared to those growing where there were no crusts. The primary importance of cryptogamic soils to tortoise populations could be in stabilizing the soils against wind and water erosion (Belnap and Gardner 1993, DeFalco 1995), but more research is clearly needed.

CHANGES IN VEGETATION

Several studies measured the effect ORVs have on vegetation; most of them evaluated damage from competitive events. Burge (1986) described how many perennial shrubs were damaged along the edge of the Frontier 500 competitive race. She counted 1170 uprooted or crushed shrubs (no species identified) after the race. Davidson and Fox (1974) measured plant diversity, number of individuals, and amount of cover in a pit area (where vehicles were parked), alongside a dual sport race trail, and "several hundred yards away" (i.e., control area). They found significantly lower values for all three parameters in the pit area, moderate values alongside the trail, and the highest values at the control site. Woodman (1986) recorded the destruction of several creosote and burrobushes around the periphery of the pit area for the 1981 Parker 400 race. A BLM report detailing damage to vegetation caused by the 1974 Barstow to Vegas Motorcycle Race (BLM 1975) showed that 0 to 76% of the plants, particularly seedlings and small shrubs, were damaged in each of 26 sites.

Berry et al. (1990) measured habitat changes over a six-year period inside and outside of the DTNA where ORV non-race activity occurred. They found a 23% increase in habitat loss around a staging/pit area and that ORV trails increased in width by 130% and 157% in area.

Vegetation is clearly degraded by heavy ORV activity. Bury and Luckenback (1986) compared vegetation inside (treatment) and outside (control) an ORV use area south of Barstow. There were 1.7 times the number uf live perennials on control, and 2.4

times number of dead ones (mostly *Ambrosia dumosa*) on the treatment area. Plant cover was 3.9% higher in the treatment area. This study suffers from a lack of replication. Comparing aerial photographs taken at the same points 19 to 25 years apart in six different locations in the Mojave and Colorado Deserts, Lathrop (1983) measured an average of 49% reduction in shrub density in ORV areas. Ground-based transects in control and treatment (disturbed) sites yielded 48-97% reductions in perennial plant cover in the ORV use areas. Thirty-four to 46% reductions in density resulted from single race events at two separate locations (Lathrop 1983). Luckenbach (1975) reports, that "in one Hounds-and-Hare race, an estimated 140,000 creosote bushes (*Larrea tridentata*), 64,000 burro-weed *(Franseria dumosa)*, and 15,000 Mojave yuccas (*Yucca schidigera*) were destroyed or severely damaged over a stretch of 100 miles." No additional details were provided.

Rowlands et al. (1980) and Adams et al. (1982b) conducted one of the only manipulative experiments on ORV effects on Mojave desert vegetation. They studied the effect that different numbers of passes over the same area by a motorcycle and a 4-wheel drive sports utility vehicle (SUV) had on plant growth. They also looked at the interactive effects of soil moisture and soil type. Plant density, biomass, and cover generally were reduced following any level of disturbance with motorcycles requiring a greater number of passes to equal the reduction caused by the SUV. Grama grass (*Bouteloua barbata*), appeared to respond positively to light disturbance, but less so to heavy disturbance. The introduced weed, split grass *(Schismus barbatus)*, was significantly more abundant within tracks than in control areas, probably because the fibrous nature of their roots allowed them to become better established than more tap-rooted natives in compacted soil.

Vollmer et al. (1976) found annual plant density to be significantly lower within experimentally created tracks from two 4-wheel drive vehicles compared to the hump between the tracks and in an area randomly covered by the same vehicles. No difference in density occurred between the randomly driven area compared to the control site. Shrubs in the regularly driven area (42 passes by vehicles) suffered twice as much damage as those in the randomly driven area. This study lacked replication and proper controls, but data collection and analysis were well executed.

Kuhn (1974, cited in Lathrop 1983) reported a reduction in plant density of 24% and plant cover of 85% in ORV-disturbed plots compared to undisturbed controls in foredunes at Kelso Dunes. Similarly, comparing aerial photographs taken 21 years apart, Lathrop (1983) measured a 50% reduction in shrub density in the same foredunes.

Erosion and Loss of Soil

ORV activity can increase erosion, which removes soil nutrients and soil that is penetrable to roots (Adams and Endo 1980a, Wilshire 1980). ORVs modify various features that help to stabilize the soil against erosion including surface crusts, coarse particles, desert pavements, and vegetation (Hinckley 1983). They also alter the configuration of the ground surface thus affecting water runoff patterns (Hinckley 1983).

The net loss of soil at specific ORV-use areas has been documented. Wilshire and Nakata (1976) estimated 150 metric tons of dirt were lost to erosion from one 68-meter long western Mojave hillside trail with a 44-58% slope. Total estimated loss for the portion of hill used for an unspecified number of years was 11,000 metric tons. Snyder et al. (1976) estimated that 150-230 mm of soil was lost per year along transects in an ORV use area over two to five years at Dove Canyon. That amount is compared to estimates of natural erosion rates of 1.0 to 4.6 mm per year in arid areas (reported in Hinckley et al. 1983). No control or low-impact reference sites were established in this study. Webb et al. (1978) reported a loss of 0.3 to 3.0 metric tons per m^2 from an ORV trail in arid land at a heavily used ORV park in central California. They further reported that erosion was greatest on sand loam and gravelly sandy loam and least on clay and clay loam.

In artificial rain trials, Iverson (1979) found greater sediment yield (soil runoff) in vehicle-disturbed versus undisturbed slopes from loosening of soil and alteration of flow patterns. The difference was thought to be from increased water flow velocity and more channeling of the flow, not from reduced filtration. Consequently the effect would be more pronounced during intense thunderstorms than during more mild winter frontal-type storms. Also using artificial rain, Eckert et al. (1977) looked at infiltration and sedimentation rates at two Mojave desert sites in Nevada following single and multiple passes of truck and motorcycle. Single passes made no measurable difference. Multiple passes increased rates of infiltration and sedimentation, particularly in interplant spaces versus beneath plants. However, the artificial rainfall rates were similar to rare very heavy thunderstorms; they were unlike the winter cyclonic rainfall that is more typical of the western Mojave desert. Furthermore, Reicosky (1979) suggested that movement of water towards vehicle tracks compensates for decreased infiltration rates. Hinckley et al. (1983) suggested that water erosion would be the least in areas that are relatively flat, experience short, low-intensity storms, and have a coarse (gravelly) surface.

Fugitive dust, dust blown from the ground by wind and vehicle activity, can potentially be a problem for desert tortoises. Fugitive dust is related to vehicle speed, surface texture, surface moisture, and probably vehicle type (with heavy four-wheel drive vehicles causing the most dust followed by light four-wheel drive vehicles followed by motorcycles; Adams and Endo 1980b). The threshold velocity for wind erosion (TV), the lowest wind speed necessary to create dust, is highest for desert pavement and areas with hard surface crusts. Soils with a large proportion of fine particles will be more susceptible to wind erosion. Disturbances that lower the TV will increase the incidence of dust storms. Disturbance of sand dunes and sandy washes does not alter their TV. Areas protected by cryptogamic soils and desert pavement had greatest reduction in TV following disturbance, and more so with siltier versus sandy soils (Adams and Endo 1980b, Gillette and Adams 1983). Winds of 20-30 mph at 6 ft above ground caused fugitive dust in these areas. Erodibility also varies with width of disturbed area up to about five meters (Wilshire pers comm., cited in Adams and Endo 1980a)

Satellite images taken on January 1, 1973, captured dust storms from Santa Ana wind conditions (Bowden et al. 1974, Wilshire 1980). Many of the dust plumes, which were 10 to 30-km long and covered 300 km^2, originated in areas of intensive ORV activity in the western Mojave. BLM (1975) measured three to five times more suspended particulate density for fugitive dust during the 1974 Barstow to Vegas race site compared to before the race.

The main effect of wind erosion on productivity is removal and redistribution of surface nutrients, not reduction in soil depth. Loss of soil nutrients found in the top 5 to 10 cm of soil significantly reduced perennial cover in a similar arid environment in Australia (Charley and Cowling 1968). Sharifi et al. (1997, 1999) showed that photosynthesis and plant productivity are hampered by dust on the leaves of desert shrubs, but that the effect may be ameliorated by heavy summer rainfall.

LIGHT ORV USE

Most of the foregoing discussion relates specifically to competitive events and heavy use like what now occurs within open use or freeplay areas. They are of limited applicability to understanding the effect of lighter travel in areas where traffic is legally restricted to designated routes (i.e., dirt roads). Indeed, very little data are available to evaluate these impacts primarily because the focus of most research has been on the effects of heavier ORV use. There are a few studies that demonstrated that occasional vehicles riding off of roads (including for parking or camping within 100 ft of roads, which is currently permitted, Bureau of land Management 1980), can damage the soil and vegetation, the amount of damage being less than heavier off road travel. Webb (1983) found that the greatest increase in compaction occurred the first few time a motorcycle crossed an area and compaction increased with more crossings, but at a lower rate. Similarly, Adams and Endo (1980a) discovered that just a few passes by an SUV were sufficient to significantly increase compaction and a single pass did so in some wet soils. Vollmer et al. (1976) found that there was damage to plants in an area subjected to random four-wheel drive activity, but that damage was higher in areas that were repeatedly driven over. Bury and Luckenbach (1977) reported little difference in the number of creosote shrubs in moderate use versus undisturbed plots, but did find that half were broken or damaged in the moderate use area. Likewise, a "sparsely" used ORV area within the Jawbone Canyon Open Area showed 35% less perennial plant cover than an unused control area (Lathrop 1978). Finally, just stepping on cryptogamic crusts can damage and decrease nitrogen fixing activities of the crusts (Belnap 1996).

All of these studies indicate that some damage is likely to occur when vehicles stray off of established roads. Goodlett and Goodlett (1993) demonstrated that ORV enthusiasts will not always obey signs indicating routes are closed, nor do they always stay on designated routes. However, their study was conducted in an area that had recently changed from an open free play area to a limited use one. Although it is likely that number of tracks will be highest in close proximity to roads (e.g., LaRue, pers obs.), no studies have tested for this pattern. Many of the problems associated with light ORV use likely relate to increased human access the roads and trails afford (see "Human Access to Tortoise Habitat" section, below).

Summary

Although each study comparing tortoise densities inside and outside of ORV areas has limitations, they all lend evidence to reductions in tortoise population densities in heavy ORV use areas. The causes for these declines are less certain. Tortoises and their burrows are crushed by ORVs, although it is difficult to evaluate the full impact this activity currently has on tortoise populations, partly because there are probably relatively few tortoises in most open use areas. ORVs damage and destroy vegetation. Density, cover, and biomass are all reduced inside versus outside of ORV use areas, particularly following multiple passes by vehicles. Split grass (*Schismus barbatus)*, a weedy introduced grass, in particular appears to benefit from ORV activity. Very light, basically non-repeated, vehicle use probably has relatively little long-term impact. Soil becomes compacted by vehicles. The compaction increases with moisture content of the soil, weight of vehicle (particularly high weight to tire surface area ratio), and soil type. Cohesionless sand, such as in sand dunes and washes, are largely immune to compaction while moist soils are much more susceptible than dry ones. Compaction, lower infiltration rates, loss of plants and cryptogamic soils all contribute to increased wind and water erosion and fugitive dust, particularly when such areas are several meters in width. More research is needed to understand the effect light ORV use has on tortoise populations and habitat.

Predation/Raven Predation/Subsidized Predators

Desert tortoises have several natural predators including: coyotes, kit foxes, feral dogs, bobcats, skunks, badgers, common ravens, and golden eagles. The dominant predator probably varies temporally, spatially, and with size of the tortoise (Berry 1990 as amended). Few studies have attempted to quantify or estimate the relative proportion of mortality attributable to the various predators at specific sites, and none attempt to characterize it regionally.

One of the earliest publications reporting that ravens are potentially important predators on desert tortoises was Campbell (1983). He found 140 shells of juvenile tortoises (36 to 103 mm MCL) at the base of fence posts along the 30.5 miles of fencing surrounding the DTNA. He attributed 136 to raven predation, but gave no indication why. Berry (1985) evaluated 403 juvenile tortoise shells found on 27 desert tortoise study plots throughout the Mojave Desert. She determined that ravens killed 35%. Her evaluation was based on circumstantial evidence because the reference collection was shells found beneath perch sites that may have been used by other predators or scavengers. Although the patterns of shell damage she used are consistent with the patterns Boarman and Hamilton (in prep.) obtained from 266 shells collected from beneath raven nests. Also, ravens are scavengers as well as predators, so some of the shells attributable to raven predation may actually have been found and eaten after death (Boarman 1993).

During the first 5 to 7 years of life, the tortoise shell is incompletely ossified; it is soft and easy to puncture and rip open. When pecked open by a raven, the soft shell will bend then dry in place leaving parts of the shell pushed in or pulled out. Carcasses found in this condition were likely pried open when the tortoise was alive or shortly after death. The shell soon dries after death. Once this happens the shell will fracture when pecked open, giving a different appearance. Although based on sound knowledge of the biology of tortoises, this scenario has not been subjected to quantification or controlled experimentation.

Woodman and Juarez (1988) reported finding 250 shells, probably killed over a four year period, dead beneath one raven nest near the Kramer Hills. Some of the carcasses found were of young animals found alive and individually marked by the same researchers several weeks earlier and apparently in healthy condition. This provided the first hard evidence that ravens almost certainly were killing some tortoises, not just scavenging them. Since that time, several observations have been made of ravens carrying away live juvenile tortoises (Boarman 1993). One researcher reported finding a tortoise eviscerated, but still alive, beneath a raven nest (R. Knight pers. comm.). These reports all remain anecdotal, but, because observing the act of predation by a predatory bird is notoriously difficult, it is unlikely we will ever be able to acquire an adequate number of good hard data on the phenomenon. One published account evaluated food of ravens in the Mojave desert by looking at pellets, indigestible portions of food that were coughed up at their nests (Camp et al 1993). They found tortoise remains in only 1.3% of the pellets. However, they did not report the 19 shells they found at several of those nests because they only reported on pellet contents (Camp pers. comm., Boarman pers. obs.); shell fragments usually are not found in pellets. They also did not establish whether all nests studied were in tortoise habitat.

The fact that ravens do kill some tortoises does not alone indicate that the losses are serious enough to warrant management action. We must understand the extent of predation and if it is having an impact on tortoise populations. Evaluating raven predation is perplexing because of the difficulties in finding small carcasses over such a large area of desert and in monitoring small, hard to find young tortoises (Berry and Turner 1986, Shields 1994). The extent of predation can be estimated by evaluating juvenile tortoise carcasses found throughout the desert. Berry (1985) and Boarman and Hamilton (in prep) analyzed the characteristics of 150 and 266, respectively, juvenile tortoise shells found in the deserts of California. Their reports indicate that primarily animals less than 100 mm MCL (less than approximately 5-7 years old) are taken throughout most portions of the desert in California. Beneath 23 transmission towers in Nevada, McCullough Ecological Systems (1995) found the remains of 78 juvenile tortoises, many showing signs consistent with raven predation.

A common argument made against raven predation being of management concern is that we must concentrate on protecting adult female tortoises (Doak et al. 1994). This is partly because adult females are the ones actually reprodueing, thus contributing most to the persistence of the population and partly because juvenile animals typically experience high mortality, so losses to ravens are natural and the population can sustain the losses. This is a correct prediction from life history theory for many animal species, but not for long-lived ones that first reproduce later in life (approaching 20 years), like the desert tortoise (Congdon et al. 1993, 2002). Life history theory predicts that stable populations of such animals can sustain annual mortality of juveniles of 25%. However, when adult populations are declining, juvenile mortality must be reduced to approximately 5% to ensure recruitment of new individuals into the breeding population (Congdon et al. 1993). This finding is based on well developed life history theory. Therefore, in tortoise populations that are experiencing overall declines, additional losses of juveniles to ravens may decrease the stability or at least prevent recovery.

A survey of tortoise remains found beneath raven nests was recently completed (Boarman and Hamilton in prep.). It showed that ravens prey on tortoises throughout the Mojave Desert in California, but probably not all ravens nesting in tortoise habitat ate tortoises. The most shells found at one nest in one year between 1991 and 1997 was 28, which were found beneath each of two nests in the eastern Mojave Desert. The results are preliminary and conservative because they pertain only to remains dropped beneath or near the raven nests. Many shells are found at locations well away from nests. During the raven breeding season, however, most foraging is probably done near the nest (Sherman 1993) and most food is likely brought back to or near the nest, so the results are probably relatively accurate if conservative.

There are little data available to determine the effect other predators might have on desert tortoise populations. For example, finding shells chewed by mammals, probably canids, and tortoise remains in coyote scat, Berry (1990 as amended) reported evidence of canid or felid predation at four out of twelve study plots in California. Proportion of deaths attributable to mammalian predators over all 12 plots was 53.% (ranged – 1.8% to 45.3% among the 4 plots where mammal-related mortality determined). Turner et al. (1997b) determined that most tortoise nests that failed were dug up by coyotes or kit foxes, but no data were presented. In 1998 and 1999, 47% and 12%, respectively, of nests studied at Twentynine Palms (MCAGCC) were dug up, probably by kit foxes (Bjurlin and Bissonette 2001). Bjurlin and Bissonette (2001) also believed that feral dogs cause a significant amount of mortality among adult tortoises in the area, but presented evidence for only one such death. They did report a high incidence of canid-like shell damage to live tortoises and the presence of feral dogs and dog packs within their study site. The effect that feral dog predation has on tortoise populations appears to be an emerging problem that warrants further documentation.

Non-ORV Recreation

Non-ORV recreation in the Mojave Desert includes camping, nature study, rock collecting, sight-seeing, hunting, horseback riding, mountain biking, and target practice. There are no studies concerning their impacts on tortoise populations: hence, there may or may not be impacts. Likely impacts include handling and disturbance of tortoises; loss of habitat to campgrounds, picnic areas, scenic pull outs, vandalism, and other support facilities; increase in road kills; and support of ravens when organic garbage is left behind. There could also be soil compaction and damage of vegetation and cryptogamic crusts from off-trail travel by mountain bikes, horses, and hikers. All of these impacts are related to the problems with increased access to tortoise habitat (discussed in "Human Access to Tortoise Habitat" section, below). Given the increased interest in non-motorized recreation in the deserts, this is an important area for future research. There are no studies that directly measured the impacts of non-motorized recreation on tortoise populations or their habitats and only one that showed that hiking off of trails can significantly damage cryptogamic crusts (Belnap 1996).

Hunting and target practicing are two additional recreational activities that may impact tortoises. One of the primary anthropogenic causes for wildfire in the desert is from bullets striking rocks (R. Franklin, BLM Fire Management Officer, pers. comm.), which can occur while hunting or target practicing. The California Department of Fish and Game has constructed an array of small- and big-game guzzlers to help facilitate growth of game species populations. Not only can ravens sometimes access water at the big game guzzlers, but tortoises can get caught and die in some types of small game guzzlers. Hoover (1996) found the remains of 26 tortoises in 89 of the upland game watering devices in California. Finally, people target practicing, which is a very different activity than hunting, might also illegally use tortoises as targets (Berry 1986a, see "Vandalism," below).

Roads, Highways, and Railroads

Roads, highways, and railroads have several impacts on desert tortoises and their habitat. Direct impacts may include mortality through road and train kills and destruction of habitat (including burrows). Possible indirect effects include degradation of habitat because they serve as corridors of dispersal for invasive plants, predators, development, recreation, and other anthropogenic sources of impact. Roads, highways, and railroads also serve to fragment the habitat and populations (see "Habitat Degradation, Fragmentation, and Destruction," below).

Many tortoises fall victim to road kills. For instance, Boarman and Sazaki (1996) reported finding 115 tortoise carcasses along 28.8 km of highway in the west Mojave. This represents a conservative estimate of 1 tortoise killed per 3.3 km of road surveyed per year. This source of mortality primarily affects subadults and adults, although the results are partially skewed by the difficulty of finding smaller carcasses and their quicker loss to scavengers and decay. The figures cannot be extrapolated to all roads and highways to estimate total losses to road kills in the desert because mortality rate likely depends on traffic speed and volume, density and demography of surrounding tortoise population, and perhaps width and age of road. The results also cannot be applied to lightly traveled paved or dirt roads because of a four-way relationship between tortoise density, road conditions, traffic volume, and road kill rate. A tortoise depression zone exists along highway edges and extends to 0.4 km or further (Nicholson 1978, Berry and Turner 1987, Berry et al. 1990, LaRue 1993, Boarman and Sazaki 1996, von Seckendorff Hoff and Marlow 1997, cf. Baepler et al. 1994). The cause is probably primarily road kills, but illegal collections, noise, and other factors may also contribute although there are no data to evaluate their likely or relative effects.

A common mitigation for the impacts of roads and highways is a barrier fence, which has been shown to be highly effective at reducing mortality in tortoises and other vertebrates in the west Mojave (Boarman and Sazaki 1996). However, fences only increase the fragmenting effects of roads. Preliminary results of an eight-year long study indicate that culverts are used by tortoises to cross highways (Boarman et al. 1998), but it is unknown whether their use is sufficient to ameliorate the fragmenting effects of fenced highways (Boarman and Sazaki 1996).

Roads are also major attractants for common ravens, which are predators on juvenile tortoises (Knight and Kawashima 1993, Boarman 1993). Ravens, being partly scavengers, are known for cruising road edges in search of road kills (Boarman and Heinrich 1999), but risk of predation is not increased near roads (Kristan and Boarman 2001).

The flush of vegetation that grows alongside roads (Frenkel 1970, Johnson et al. 1975) as a result of rainwater runoff and collection may benefit tortoises by providing a more consistent source of food over a more extended period of time, even in relatively dry years (Boarman et al. 1997). Alternatively, the abundance of food may bring them into harms way if (1) they wander onto the road, (2) vehicles pull onto the vegetated shoulder of the road, (3) grading or mowing activities occur during times of tortoise activity, (4) herbicides are applied to control growth of weeds along the road shoulder, or (5) they are seen and caught by passers-by. Brooks (1998) found a significant positive correlation between number of alien annual plant species near roads and density of dirt roads., and the species richness and biomass of alien annuals is higher near roads than away from them (Brooks pers. comm.).

Railroads may also impact tortoise populations through train kills and perhaps by tortoises getting caught between the rails (Mount 1986). No published studies were found that looked for train-killed tortoises along extensive sections of railroad tracks. However, Ron Marlow (pers. comm.) found eight carcasses between the rails along approximately 100 km of railroad tracks in the eastern Mojave. Noise or vibration may also affect tortoises that live alongside railroads, but has not been studied (see "Noise and Vibration," above). Railroads provide a positive benefit: tortoises regularly build burrows in railroad berms that are not covered with gravel. It is not known if train noise negatively affects the behavior, audition, or reproductive success of these tortoises.

Utility Corridors

Corridors formed by utility and energy rights-of-way cause linear impacts to populations and may have levels of impacts well beyond those of many point sources of impacts. In a retrospective evaluation of results of 234 Biological Opinions issued by USFWS in California and Nevada (LaRue and Dougherty 1999), 80% (47/59) of the tortoises reportedly killed in California and Nevada were killed along utility corridors. Most of those were along the Kern-Mojave Pipeline (Olson et al. 1993, Olson 1996). Considerable habitat destruction or alteration occurs when pipelines and transmission lines are constructed and the impacts are repeated as maintenance operations or new pipelines or power lines are placed along existing corridors. Trenches opened for laying or maintaining pipes may serve as traps for tortoises and other animals (Olson et al.

1993). Dirt roads used for maintenance-related access create dust (Wilshire 1980) and provide access to less disturbed habitat (Brum et al. 1983). The habitat conversions during early stages of post-construction succession along pipeline corridors (Vasek et al. 1975) not only may suppress regular use by tortoises, but may function to reduce dispersal across the corridor thus effectively fragmenting a previously intact population (this view is speculative).

The presence of transmission towers in areas otherwise devoid of other raven nesting substrates (e.g., Joshua trees, palo verdes, cliffs), may introduce heavy predation to an area previously immune to such predation (Boarman 1993). Most raven predation on tortoises appears to occur during the raven breeding season (April - May, pers. obs.). By one estimate, ravens probably do most (75%) of their foraging within 400 m of their nest (Sherman 1993) and raven predation pressure is notably intense near their nests (Kristan and Boarman 2001). Therefore, ravens nesting on transmission towers, where no other nesting substrate exists within about 800 m, may significantly reduce juvenile tortoise populations within 400 m of the corridor, but this effect is quite localized. However, recent unpublished data on the distribution of raven depredated juvenile tortoises suggests that not all ravens nesting within tortoise habitat actually eat tortoises (at least they do not bring the shells back to the nest; Boarman and Hamilton in press).

Data collected along paved highways indicate that road kills can substantially reduce tortoise populations within at least 0.4-0.8 km of such roads (see "Roads, Highways, and Railroads" section, above), and their impact is likely lower along newer and more lightly traveled roads (Nicholson 1978). But, there are no data on the impact of lightly traveled dirt roads (e.g., utility maintenance/access roads) on tortoise population densities.

Vandalism

Vandalism is the "purposeful killing or maiming of tortoises" (Luke et al. 1991, p. 4-61). Reports of tortoises being vandalized include shooting, crushing, running over, chopping off heads, and turning them over (Berry and Nicholson 1984a, Berry 1986a, Bury and Marlow 73). Most reports of specific incidents are anecdotal, but sometimes substantial. The most quantitative accounts are for gunshot deaths (Berry 1986a, 1990 as amended), but are mostly based on postmortem forensic analysis. Berry (1986a) found 91 tortoises carcasses (14.3% of those collected at 11 sites) showing evidence of being shot. The proportion of carcasses showing evidence of gunshots was significantly higher from west Mojave sites (20.7%) than from east Mojave (1.5%) and Colorado (2%) desert sites. Eleven of the 58 (19%) tortoise found dead on the Beaver Dam Slope, Utah, showed signs of traumatic injury. This category included individuals exhibiting gunshot wounds. These ranged from pellet wounds through .22 caliber holes to one individual exhibiting a .44 caliber bullet wound.

Wild Horses and Burros

Wild burro and tortoise ranges overlap in some places, but the overlap is quite low in the West Mojave. No published studies were found that investigated the impact burros or horses (neither of which are native to North America) have on tortoise populations. The primary effect is likely to be habitat alteration through soil compaction and vegetation change. Burro populations are probably not extensive enough in most areas to pose a major threat to tortoise populations, but this is speculative.

CUMULATIVE THREATS TO TORTOISE POPULATIONS

Human Access to Tortoise Habitat

Perhaps the most important general threat to tortoise populations relates to actual human presence in tortoise habitat and thus refers primarily to access. Many of the individual threats discussed above relate to the level of access to tortoise habitat afforded to people. For instance, law enforcement officials have documented illegal collecting of tortoises for food or cultural ceremonies on a few occasions (USFWS 1994). One study supported the intuitive impression that poaching occurs close to roads (Berry et al. 1996), but the methods employed were not very precise (counting burrows that appeared to have been dug up with shovels) making the results weak at best. Since roads likely provide access to poachers, a logical conclusion of their study is that a larger proportion of the tortoise population will be under the risk of being poached where more roads intrude on tortoise habitat.

The presence of a road poses potential harm to tortoises and their habitat and the more roads there are the greater is the proportion of the tortoise population that is under the threat of illegal off-road activity. Boarman and Sazaki (1996) demonstrated that tortoises regularly die from collisions with automobiles and Nicholson (1978) showed that the rate of mortality probably increases with traffic volume. So, road kill is probably proportionally lower on lightly traveled dirt roads, but may still exist. However, because tortoise populations are probably less depressed alongside lightly traveled roads (Nicholson 1978) and if tortoises are less inhibited from crossing narrower, dirt-covered roads (for which there are no data), we may speculate that proportionally more tortoises may cross lightly traveled roads. The possibility does exist that ORVs may crush tortoises or their burrows on or off of roads (Marlow 1974, Bury and Luckenbach 1986, Berry 1990 as amended).

Mortality on roads is not the only type of vehicle-related impact; ORVs sometimes drive off of established routes, including within 100 ft to camp and park (Bureau of Land Management 1980). One study has supported the hypothesis that off-road activity is high near dirt roads even in an area that was heavily signed (Goodlett and Goodlett 1993). For example, they counted an average of one track every 31 feet along transects walked perpendicular to authorized routes. As expected, the density of tracks decreased with distance from the road from an average of 2.1 per 20 ft near the road to 0.5 per 20 feet 250 to 300 feet away. No statistical analyses were made. Goodlett and

Goodlett (1993) also demonstrated that ORV recreationists ignored BLM signs indicating trails and roads were closed to vehicles in the Rand Mountains. An average of 11.5 new tracks was counted along 17 trails 6 to 7 days after the trails were raked. An average of 10.0 tracks was found along 20 unmarked routes (again, no statistical analyses were provided), which suggests that the signs were essentially ineffective at preventing people from riding on closed trails. The motorcycle activity occurred over Thanksgiving weekend, 1991.

Furthermore, there is ample evidence that occasional driving off of roads compacts soil and damages vegetation (Vollmer et al. 1976, Webb 1983, Adams et al. 1982a, b, see also "ORV" section, above). The greatest increase in compaction can occur after a single or very few passes by a vehicle over unimpacted soil (Webb 1983), or at least soil strength (a measure of compaction) is significantly increased after a very few passes by an SUV (Adams et al. 1982a, b). Any driving or even walking over cryptogamic crusts damages the crust (Belnap 1996). As discussed in the "ORV Activities" section, above, there are very little data to indicate how these habitat alterations might affect tortoise populations.).

Other potentially harmful activities that likely occur in greater numbers near roads include: mineral exploration, illegal dumping of garbage and toxic wastes, release of ill tortoises, vandalism, anthropogenic fire, handling and harassing of tortoises, and trailing of sheep (Berry and Nicholson 1984a). Invasive plants also proliferate near roads and where road densities are higher (Brooks 1995, 1999a). The threat posed to tortoise populations by all of these factors likely increases with increased access afforded by the proliferation of roads, even very lightly traveled ones. Furthermore, some of these individual threats may be relatively low, but their cumulative impact may be great. Berry (1990 as amended, 1992), presents data that suggests a correlation between tortoise population declines and density of roads, trails, and tracks on tortoise study plots, but the results have not been treated to statistical analysis. This important association between access and tortoise wellbeing needs further study.

Habitat Loss, Degradation, and Fragmentation

One of the most pervasive problems for desert tortoise populations is also among the most difficult to evaluate: habitat loss, degradation, and fragmentation from the myriad activities that take place in the desert. This is the cumulative result of several of the individual threats discussed above.

Habitat loss is generally quite apparent (e.g., loss of useable habitat when paved for a parking lot or plowed for agriculture), but is sometimes less than obvious (e.g., a given area may be rendered unusable by tortoises after soil is heavily compressed and vegetation is destroyed after many vehicles drive over the area). Previously useful habitat may be rendered unusable, but may appear superficially similar to useable habitat.

Habitat degradation consists of human-mediated changes in habitat characteristics that render an area less valuable to, but still potentially usable by, tortoises. The degradation may be manifested in altered soil structure, increased exotic plants, lower abundance of preferred forage plants, reduced availability of effective cover sites, or a combination of these traits. The degradation may not directly cause increased mortality in tortoise populations, but may reduce reproductive output or cause some animals to leave the area in search of less degraded habitat. Although these responses have been hypothesized, there have been no studies on tortoise habitat choice or preference patterns changing as a result of habitat changes.

Many of the impacts discussed above fit easily into the category of habitat degradation that may significantly reduce habitat quality for tortoises. A single vehicle driving over a section of ground may have little impact by itself (Adams et al. 1980a, b), but when that is added to a pile of trash nearby, compaction from grazing (Avery 1998), and reduced primary productivity of plants because of dust from a nearby dirt road (Sharifi et al. 1997), the cumulative habitat degradation may significantly reduce quantity or quality of forage for tortoises. The cumulative effects of factors leading to habitat loss and habitat degradation have been implicated as causes in the extirpation and drastic reductions in tortoise populations from the Antelope, Searles, and Indian Wells valleys, and in the vicinity of several other communities in the West Mojave (e.g., Barstow, Mojave, and Victorville; Berry and Nicholson 1984a, Feldmeth and Clements 1990, Tierra Madre Consultants 1991, USFWS 1994).

Fragmentation is the process by which solid blocks of habitat and populations depending on the habitat are broken up into smaller subunits with limited dispersal between habitat blocks (Meffe and Carroll 1997). Rivers, mountain ranges, major changes in soil or habitat type all represent natural causes of fragmentation. Highways, railroad tracks, towns, and other developments, isolated and conglomerated, are examples of anthropogenic factors that fragment desert tortoise habitat in the West Mojave Desert. Smaller populations are more susceptible to local extinctions as a result of both genetic and demographic (population) processes. A smaller population has fewer individuals available for interbreeding, which may result in genetic deterioration: inbreeding depression and loss of genetic diversity within the population (Frankham 1995). Genetic deterioration can result in the inability to adapt to short- or long-term environmental changes, which makes the population more vulnerable to extinction. Small populations are also susceptible to extinctions from random fluctuations in birth rate, death rate, age distributions, and sex ratios (Opdam 1988). Small populations suffer from the Allee Effect, the fact that it is harder to find a mate when there are fewer individuals in a population (Allee et al. 1949). Finally, smaller populations are more vulnerable to catastrophic events (e.g., disease epidemics, earthquakes, and floods) and random environmental fluctuations in such things as food resources. These processes (genetic deterioration and demographic consequences of small populations) are theoretical possibilities, but have not been documented empirically in desert tortoises populations (see USFWS 1994 for a theoretical analysis).

An additional problem associated with fragmentation is that the negative effects of habitat edges are increased considerably (Murcia 1995, Meffe and Carroll 1997). Edges, or boundaries, are problems for ecosystems because the microenvironment in the edge is different than in the interior: temperature, humidity, light, chemical inputs, etc., may all differ in edge regions. The distribution and persistence of many plant and animal species are often strongly affected by these microenvironmental conditions, so the communities are usually different along edges. Furthermore, edge conditions often facilitate the introduction, establishment, and spread of exotic species that may become predators or competitors with plants or animals in the interior (Janzen 1986, Wilcove et al. 1986). For desert tortoises, the edge effect is a theoretical possibility, but it has not been well documented in tortoise populations. Furthermore, some edge effects may only function over relatively short distances (e.g., tens of yards) or not at all (Ratti and Reese 1988, Murcia 1995).

There are little data that directly test this hypothesized cumulative effect of multiple impacts on tortoise populations. Berry and Nicholson (1984a) do cite anecdotal evidence of the loss of previously-existing populations in now heavily-populated areas of Antelope, Lucerne, and Yucca valleys. Berry et al. (1994) present correlative data showing that declines in tortoise populations in the Rand Mountains and Fremont Valleys correlate with increases in a suite of human impacts. The Desert Tortoise Recovery Plan (USFWS 1994) provides data that show significant declines occurred in populations exhibiting high rates of human-caused mortality.

Urbanization and Development

Whereas construction activity (treated as an individual threat, above) has impacts specific to the activities of building new structures (e.g., temporary compaction of vegetation and soil, fugitive dust, disturbance and possible death of tortoises), these impacts largely cease once construction has been completed (although for some impacts, such as soil compaction, there is a residual effect caused by delayed recovery, Lovich and Bainbridge 1999). The result of the construction activity is the presence of new structures, which are called here "developments," and which have its attendant impacts. These impacts include long-term or permanent loss or alteration of habitat, impacts from maintenance activities, disruption of tortoise behavior, and road kills (Berry and Nicholson 1984a, Luke et al. 1991).

Developments may be relatively isolated from each other, but "Urbanization" refers to cumulative effects of multiple and nearly contiguous developments including construction of permanent residences that cover large areas. Urbanization has several impacts associated with the presence of many people in the area, not, all of which are well documented. Urbanization results in considerable fragmentation, loss of habitat, and habitat alteration to the point of being largely useless to tortoise populations (Berry and Nicholson 1984a, Feldmeth and Clements 1990, Tierra Madre Associates 1991, section titled "Habitat Loss, Degradation, and Fragmentation"). Some recreational activities may emanate directly from urban areas. Wild dogs may be more prevalent (e.g., Bjurlin and Bissonette 2001) and collecting, handling and vandalism of tortoises could increase where there are more people. Captive tortoises, potentially infected URTD (see "Disease" section, above), are more likely to escape and help spread disease to the native population (Jacobson 1993, Berry pers. comm.). Illegal dumping is prevalent (pers. obs.), raven populations are larger (Knight et al. 1993), and exotic plants predominate

(Humphrey 1987, Brooks 1998) around urban developments. Urban areas and associated flood control channels in the desert are often the source of much fugitive dust (Wilshire 1980). Many of these impacts may be relatively minor by themselves, but their cumulative effects on nearby tortoise populations may be great.

There is some evidence that tortoise populations can persist in the presence of light industrial developments. In the 1980s 460 wind turbines and 51 electrical transformers were erected in tortoise habitat at Mesa, California. Approximately 10-20 years later, there were still tortoises living and reproducing in the same area; some burrow beneath and rest upon concrete support pads for the turbines (Lovich and Daniels 2000). Reproductive output is higher than at any other site studied to date (Lovich et al. 1999). However, there are no data available to determine if the population has increased, decreased, or remained stable since construction. Tortoises may persist in this area because of the relatively low level of actual human activity in the wind park and the high productivity in the area, which is in the ecotone between creosote scrub and coastal sage scrub habitat.

ACKNOWLEDGMENTS

Harold Avery, Bill Haigh, Ed LaRue, Jeffrey Lovich, Phil Medica, and Dave Morafka reviewed earlier drafts of this report in their entirety. Individual sections were reviewed by an army of experts including: David Bainbridge Jayne Belnap, Matt Brooks, Dan Brown, Steve Corn, David Crohn, Lesley DeFalco, Todd Esque, Gilbert Goodlett, Bruce Homer, Elliott Jacobson, Ron Marlow, Phil Medica, Kathie Meyer, Richard Minnich, Ken Nagy, Ted Owens, Phil Rundel, Sean Swezey, and Bob Webb. Emily Cohen provided editorial assistance. I apologize to those scientists whose works I may have inadvertently misrepresented or failed to include in the report.

LITERATURE CITED

Adams, J. A. and A. S. Endo. 1980a. Controlled experiments on soil compaction produced by off-road vehicles in the Mojave Desert, California. Pp. 121-134, Chapt. V. *In*: The effects of disturbance on desert soils, vegetation and community process with emphasis on off road vehicles: a critical review (Rowlands, P. G., Ed.). Desert Plan Staff, Bureau of Land Management, Riverside, CA.

Adams, J. A. and A. S. Endo. 1980b. Soil impacts from off-road vehicles. Pp. 1-45, Chapt. I. *In*: The effects of disturbance on desert soils, vegetation and community process with emphasis on off road vehicles: a critical review (Rowlands, P. G., Ed.). Desert Plan Staff, Bureau of Land Management, Riverside, CA.

Adams, J.A., E.S. Endo, L.H. Stolvy, P.G. Rolands, and H.B. Johnson. 1982a. Controlled experiments on soil compaction produced by off-road vehicles in the Mojave Desert, California. Journal of Applied Ecology 19:167-175.

Adams, J. A., L. H. Stolzy, A.S. Endo, P.G. Rowlands, H.B. Hyrum. 1982b. Desert soil compaction reduces annual plant cover. California Agriculture (September-October): 6-7.

Adrian, E. D., K. J. W. Craik, & R. S. Sturdy. 1938. The electrical response of the ear: vertebrates. Proc. Royal Soc. Lond. 125 : 435-455.

Ahlgren, I. F. and C. E. Ahlgren. 1966. Ecological effects of forest fires. Botan. Rev. 26:483-535.

Allee, W. C., A. E. Emerson, O. Park, T. Park, and K. P Schmidt. 1949. Principles of animal ecology. Saunders, Philadelphia.

Allen, E. B., P. E. Padgett, A. Bytnerowicz, and R. A. Minnich. 1998. Nitrogen deposition effects on coastal sage vegetation of southern California. Pages 131-140. *In* Proceedings of the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems, Riverside, CA, February 5-9, 1996. USDA Forest Service, Pacific Southwest Research Station, PSW-GTR-166.

Allen, M. E. 1999. Cattle, dung and tortoises: symbiosis? Proc. 1997-1998 Desert Tort. Counc. Symp. 1999:99. Abstract.

Averill-Murray, R. C. 1999. Effects on growth and survival of tortoises voiding their bladders during handling. Proc. 1997-1998 Desert Tort. Counc. Symp. 1999:99-100. Abstract.

Avery, H.W. 1997. Effects of cattle grazing on the desert tortoise, *Gopherus agassizii*: nutritional and behavioral interactions. Proceedings: Conservation, Restoration, and Management of Tortoises and Turtles - An International Conference, pp. 13-20.

Avery, H.W. 1998. Nutritional ecology of the desert tortoise (*Gopherus agassizii,*) in relation to cattle grazing in the Mojave Desert. Ph.D. dissertation, University of California, Los Angeles.

Avery, H.W. and K.H. Berry. 1993. Upper respiratory tract disease and high adult death rates in western Mojave tortoise populations, 1989-1990. Proceedings of the 1987-1991 Symposia of the Desert Tortoise Council.

Avery, H.W. and A.G. Neibergs. 1997. Effects of cattle grazing on the desert tortoise, *Gopherus agassizii* : nutritional and behavioral Interactions. Proceedings: Conservation, Restoration, and Management of Tortoises and Turtles - An International Conference, pp. 13-20.

Avery, H. W., J. D. Congdon, and J. R. Spotila. 2002. Life history and demographic analysis of the desert tortoise at Fort Irwin and reference sites: study design and early findings. Desert Tortoise Council Symposium, March 22-25, 2002, Palm Springs, CA. Abstract.

Babcock and Sons, and Gallaher and Bovey, Geotechnical Consultants. 1973. A study of southeastern Mojave desert. California Division of Mines and Geology. Special Report 83.

Baepler, D. H., A. Heindl, A. K. Singh, and A. Pandley. 1994. A study of the impacts of highways on desert tortoise populations. Report to Nevada Department Transportation. Harry Reid Center for Environmental Studies, Las Vegas, NV.

Bailey, V. 1928. The desert tortoise: an example of an unusual adaptation. Nature Magazine 12:372-374.

Bailey, W. J. & G. K. Morris. 1986. Confusion of phonotaxis by masking sounds in the Bushcricket *Conocephalus brevipennis* (Tettigoniidae: Conocephalinae). Ethology 73: 9-28.

Balph, D. F. and J. C. Malechek. 1985. Cattle trampling of crested wheatgrass under short-duration grazing. J. Range Manage. 38:226-227.

Bartolome, J. 1989. Review of Holistic Resource Management by Allan Savory. Journal of Soil and Water Conservation 44:591-592.

Baxter, G. R. And G. R. Stewart. 1990. Report of the continuing fieldwork on the desert tortoise (*Gopherus agassizii*) at the Twentynine Palms Marine Corps Base. Proc. 1986 Desert Tort. Counc. Symp. 1990:128-140.

Beatley, J. C. 1967. Effects of radioactive and non-radioactive dust upon *Larrea divaricata* Cav., Nevada Test Site. Health Physics 11:1671-1625.

Begon, M., J. L. Harper, and C. R. Townsend. 1990. Ecology: individuals, populations and communities. Blackwell Scientific Publ., Boston.

Belnap, J. 1996. Soil surface disturbances in cold deserts: effects on nitrogenase activity in cyanobacterial-lichen soil crusts. Biol. Fertil. Soils 23(4): 362-367.

Belnap, J., and J. S. Gardner. 1993. Soil microstructure in soils of the Colorado Plateau: the role of the cyanobacterium *Microcoleus vaginatus*. Great Basin Natur. 53:40-47.

Belnap, J. and K. T. Harper. 1995. Influence of cryptobiotic soil crusts on elemental content of tissue of two desert seed plants.

Bentley, H. L. 1898. Cattle ranges of the southwest: a history of the exhaustion of the pasturage and suggestions for its restoration. Farmers' Bulletin No. 72: 1-32.

Berry, K.H. 1978. Livestock grazing and the desert tortoise. 43rd North American Wildlife and Natural Resources Conference, Phoenix, Arizona.

Berry, K.H. 1985. Avian predation of the desert tortoise (*Gopherus agassizii*) in California. U.S. Department of the Interior, Bureau of Land Management. Riverside, CA.

Berry, K.H. 1986a. Incidence of gunshot deaths in desert tortoise populations in California. Wildlife Society Bulletin 14: 127-132.

Berry, K. H. 1986b. Desert tortoise (*Gopherus agassizii*) relocation: implications of social behavior and movements. Herpetologica 42:113-125.

Berry, K.H. 1990 (as amended). The status of the desert tortoise in California in 1989. U.S. Bureau of Land Management, Riverside, California; amended to include data from 1990, 1991, and 1992.

Berry, K.H. 1997. Demographic consequences of disease in two desert tortoise populations in California, USA. Proceedings: Conservation, Restoration, and Management of Tortoises and Turtles - An International Conference, pp. 91-99.

Berry, K. H. and M. M. Christopher. 2001. Guidelines for the field evaluation of desert tortoise health and disease. J. Wildl. Disease 37:427-450.

Berry, K. H., F. G. Hoover, and M. Walker. 1996. The effects of poaching desert tortoises in the western Mojave Desert: Evaluation of landscape and local impacts. Proc. 1996 Desert Tort. Council Symp. 1996:45.

Berry, K.H. and L.L. Nicholson. 1984a. A summary of human activities and their impacts on desert tortoise populations and habitat in California. In Berry, K.H., ed. The status of the desert tortoise (*Gopherus agassizii*) in the United States. U.S. Department of the Interior, Bureau of Land Management. Riverside, California.

Berry, K.H. and L.L. Nicholson. 1984b. Attributes of populations at twenty-seven sites in California. In Berry, K.H., ed. The status of the desert tortoise (Gopherus agassizii) in the United States. U.S. Department of the Interior, Bureau of Land Management. Riverside, California.

Berry, K.H. and L.L. Nicholson. 1984c. The distribution and density of desert tortoise populations in California in the 1970's. In Berry, K.H., ed. The status of the desert tortoise (Gopherus agassizii) in the United States. U.S. Department of the Interior, Bureau of Land Management. Riverside, California.

Berry, K. H., L. L. Nicholson, S. Juarez, A.P. Woodman. 1990. Changes in desert tortoise populations at four study sites in California. Proceedings of the 1986 Symposium of the Desert Tortoise Council. 1990:60-80.

Berry, K. H., T. Okamoto, K. Anderson, M. B. Brown, L. Wendland, and F. Origgi. 2002. Health assessments of captive and wild desert tortoises at 17 sites in the Mojave and Colorado Deserts, California. Desert Tortoise Council Symposium, March 22-25, 2002, Palm Springs, CA. Abstract.

Berry, K. H., T. Shields, A. P. Woodman, T. Campbell, J. Roberson, K. Bohuski, and A. Karl. 1986. Changes in desert tortoise populations at the Desert Tortoise Research Natural Area between 1979 and 1985. Proc. Symp. Desert Tort. Counc. 1986:100-123.

Berry, K. H. and F B. Turner. 1986. Spring activities and habitats of juvenile desert tortoises, *Gopherus agassizii*, in California. Copeia 1986:1010-1012.

Berry, K. H. and F. B. Turner. 1987. Notes on the behavior and habitat preferences of juvenile desert tortoises (*Gopherus agassizii*) in California. Proc. 1984 Desert Tortoise Council Symposium 1987:111-130.

Berry K. B., M. Weinstein, G. O. Goodlett, A. P. Woodman, and G. C. Goodlett. 1994. The distribution and abundance of desert tortoises and human uses in 1990 in the Rand Mountains, Fremont Valley, and Spangler Hills (Western Mojave Desert), California. Draft report. Bureau of Land Management, Riverside, CA.

Biggins, D. E., B. J. Miller, and T. W. Clark. 1997. Management of an endangered species: The black-footed ferret. Pp. 420-426 *In*: Principles of conservation biology (G. K. Meffe and C. R. Carroll, eds.). 2nd Ed. Sinauer Assoc., Inc. Publ., Sunderland, MA.

Bjurlin, C. D. and J. A. Bissonette. 2001. The impact of predator communities on early life history stage survival of the desert tortoise at the Marine Corps Air Ground Combat Center, Twentynine Palms, California. U. S. Dept. of the Navy Contract N68711-97-LT-70023. UCFWRU Pub. # 00-4: 1-81.

Bleeker, M. 1988. An inventory, analysis and monitoring of grazing in the east Mojave desert of California: a geographic information system approach. M.S. Thesis, University of California, Riverside. 103 pp.

Boarman, W. I. 2002. The desert tortoise. *In* Sensitive animals and plants in the western Mojave Desert (W. I. Boarman and K. Beaman, eds.).

Boarman, W. I. 1993. When a native predator becomes a pest: a case study. For: Conservation and resource management (S.K. Majumdar, et al., eds.), pp. 186-201. Penn. Acad. Sci. Easton, PA.

Boarman, W. I. In prep. Managing subsidized predator populations: reducing predation on desert tortoises by common ravens in the Mojave and Colorado Deserts.

Boarman, W. I., M. L. Beigel, Glenn C. Goodlett, and M. Sazaki. 1998. A passive integrated transponder system for tracking animal movements. Wildlife Society Bulletin 26:886-891.

Boarman, W.I., R. J. Camp, M. Hagan, W. Deal. 1995. Raven abundance at anthropogenic resources in the western Mojave Desert, California. Report to Edwards Air Force Base, CA.

Boarman, W. I., Goodlett, T., G. C. Goodlett. 1998. Review of radio transmitter attachment techniques for chelonian research and recommendations for improvement. Herpet. Rev. 29:26-33.

Boarman, W. I. and P. Hamilton. In prep. Mortality in juvenile desert tortoises caused by avian predators. U. S. Geological Survey, San Diego, CA.

Boarman, W. I. and B. Heinrich. 1999. The Common Raven. *In* A. Poole and F. Gill (eds.), The Birds of North America, No. 476. The Birds of North America, Philadelphia, PA.

Boarman, W. I. and M. Sazaki. 1996. Highway mortality in desert tortoises and small vertebrates: success of barrier fences and culverts. Pages 169 - 173 in Transportation and wildlife: reducing wildlife mortality and improving wildlife passageways across transportation corridors. G. Evink, D. Zeigler, P. Garrett, and J. Berry, editors. U.S. Department of Transportation, Federal Highway Administration, Washington, DC.

Boarman, W.I., M. Sazaki, and W. B. Jennings. 1997. The effect of roads, barrier fences, and culverts on desert tortoise populations in California, USA. *In* J. Van Abbema (ed.), Proceedings: Conservation, Restoration, and Management of Tortoises and Turtles-An International Conference, pp. 54-58. New York Turtle and Tortoise Society, New York.

Bondello, M. C. 1976. The effects of high-intensity motorcycle sounds on the acoustical sensitivity of the Desert Iguana, *Dipsosaurus dorsalis*. M.A. thesis, California State Univ., Fullerton.

Borysenko, M. 1975. Cellular aspects of humoral immune responsiveness in Chelydra. Adv. Exp. Biol. Med. 64:277.

Borysenko, M. and S. Lewis. 1979. The effect of malnutrition on immunocompetence and whole body resistance to infection in *Chelydra serpentina*. Developmental and Comparative Immunology 3:89-100.

Bostick, V. 1990. The desert tortoise in relation to cattle grazing. Rangelands 12:149-151.

Bowden, L. W., J. R. Hutchinson, C.F. Johnson, C.F. 1974. Satellite photograph presents first comprehensive view of local wind: the Santa Ana. Science 134: 1077-1078.

Bowles, A. E., S. Eckert, L. Starke, E. Berg, L. Wolski, J. Matesic, Jr. 1999. Effects of flight and sonic booms on hearing, behavior, heart rate, and oxygen consumption of desert tortoises (*Gopherus agassizii*). Sea World Research Institute, San Diego, CA.

Brattstrom, B. H. 1974. The evolution of reptilian social behavior. Amer. Zool. 14 : 35-49.

Brattstrom, B. H., & M. C. Bondello. 1983. The experimental effects of off-road vehicle sounds on three species of desert vertebrates. Pp. 167-206 in Environmental effects of off-road vehicles: impacts and management in arid regions (Webb, R. H., & H. G. Wilshire, eds.). Springer-Verlag, New York.

Brooks, M. L. 1992. Ecological impact of human disturbance on the desert tortoise natural area, Kern County, California, 1978-1992. Master's thesis. Fresno, California State University: 51 pp.

Brooks, M. 1995. Benefits of Protective Fencing to Plant and Rodent Communities of the Western Mojave Desert. Environmental Management 19(1): 65-74.

Brooks, M. L. 1998. Ecology of a biological invasion: alien annual plants in the Mojave Desert. Ph.D. dessert. U. Calif. Riverside.

Brooks, M. L. 1999a. Habitat invasibility and dominance by alien annual plants in the western Mojave Desert. Biol. Invasions 1:325-337.

Brooks, M. L. 1999b. Alien annual grasses and fire in the Mojave Desert. Madrono 46:13-19.

Brooks, M.L. 2000. Competition between alien annual grasses and native annual plants in the Mojave Desert. American Midland Naturalist 144:92-108.

Brooks, M. L. and K. H. Berry. 1999. Ecology and management of alien annual plants in the California deserts. Calif. Exotic Pest Plant Newsl. 7(3/4):4-6.

Brown, D.E. and R.A. Minnich. 1986. Fire and changes in creosote bush scrub of the western Sonoran desert, California. American Naturalist 116(2):411-422.

Brown, M.B., I.M. Schumacher, P.A. Klein, K. Harris, T. Correll, E.R. Jacobson, 1994a. *Mycoplasma agassizii* causes upper respiratory tract disease in the desert tortoises. Infection and immunity 62(10): 4580-4586.

Brown, M., P.A. Klein, I.M. Schumacher, and K.H. Berry. 1994b. Health profiles of free ranging desert tortoises in California: results of a two year serological testing for antibody to *Mycoplasma agassizii*. University of Florida, Gainesville. Burean of Land Management Contract No. B950-C2-0.

Brown, D. R., B. C. Crenshaw, G. S. McLaughlin, I. M. Schumacher, C. E. McKenna, P. A. Klein, E. R. Jacobson, and M. B. Brown. 1995. Taxonomic analysis of the tortoise mycoplasmas *Mycoplasma agassizii* and *Mycoplasma testudinis* by 16S rRNA gene sequence comparison. International J. Syst. Bacteriol. 45:348-350.

Brum, G. D., R. S. Boyd, and S. M. Carter. 1983. Recovery rates and rehabilitation of powerline corridors. Pp. 303-314 in Environmental effects of off-road vehicles: impacts and management in arid regions (Webb, R. H., & H. G. Wilshire, eds.). Springer-Verlag, New York.

Bureau of Land Management. 1975. 1974 Barstow-Las Vegas Motorcycle Race: Evaluation Report. Riverside, CA.

Bureau of Land Management. 1980. The California Desert Conservation Area plan. Bureau of Land management, California Desert District, CA.

Burge, B.L. and W.G. Bradley. 1976. Population density, structure and feeding habits of the desert tortoise, Gopherus agassizii, in a low desert study area in southern Nevada. Proceedings of the Desert Tortoise Council 1976 Symposium 1976:57-74.

Burge, B. L. 1986. Impact of frontier 500 off-road vehicle race on desert tortoise habitat. Proceedings of the 1983 Desert Tortoise Council Symposium 1986:27-38.

Burge, B.L. 1989. What goes up must come down. Massive balloon releases are a potential threat to tortoises and other wildlife. Tortoise Tracks 10(3):4.

Burkhardt, J.W. and D. Chamberlain. 1982. Range cattle food habits of the Mojave Desert. Report for Projects 617 to Nevada Agricultural Expt. Station. 8 pp.

Bury, R. B. (editor). 1982. North American tortoises: conservation and ecology. U. S. Dept. Interior, Fish and Wildl. Serv., Wildlife Research Rpt. 12, Washingtun, DC.

Bury, R.B., and P.S. Corn. 1995. Have desert tortoises undergone a long-term decline in abundance? Wildl. Soc. Bull. 18 (1): 1-7.

Bury, R. B. and D. J. Germano (editors). 1994. Biology of North American tortoises. U. S. Dept. Interior, National Biol. Surv., Fish and Wildlife Research 13, Washington, DC.

Bury, R. B. and R. A. Luckenbach. 1977. Censusing desert tortoise populations using a quadrat and grid location system. Proc. 1977 Desert Tortoise Council Symp. 1977: 169-178.

Bury, R. B. and R. A. Luckenbach. 1986. Abundance of desert tortoises (*Gopherus agassizii*) in natural and disturbed habitats. Fort Collins, CO. United States Fish and Wildlife Service.

Bury, R. B. and R. W. Marlow. 1973. The desert tortoise: Will it survive? The Environmental Journal. June: 9-12.

Camp, R.J., R.L. Knight, H.A.L. Knight, M.W. Sherman, and J.Y. Kawashima. 1993. Food habits of nesting common ravens in the eastern Mojave desert. Southwest. Natur. 38:163 165.

Campbell, T. 1983. Some natural history observations of desert tortoises and other species on and near the Desert Tortoise Natural Area, Kern County, California. Proc. 1983 Desert Tortoise Council Symp. 1983: 80-88.

Campbell, T. 1985. Hunting and other activities on and near the Desert Tortoise Natural Area, California. Proceedings of the Desert Tortoise Council 1982 Symposium 1985:90-98.

Campbell, H. W., and W. E. Evans. 1967. Sound production in two species of tortoise, *Gopherus agassizii* and *Geochelone carbonaria*. Herpetologica 23: 204-209.

Chamberlin, T. C. 1965. The method of multiple working hypotheses. Science 148:754-759 (reprinted from Science 15 [old series]:92; 1890).

Charley, J.L. and S.W. Cowling. 1968. Changes in soil nutrient status resulting from overgrazing and their consequences in plant communities of semi-arid areas. Proc. Ecol. Soc. Austral. 3:28-38.

Chaffee, M. A., K. H. Berry, and B. B. Houser. 1999. The relation between the geochemistry of surficial materials and desert tortoise mortality in selected study sites, southeastern California--a progress report. Proceedings of the 1997-1998 Desert Tortoise Council Symposia. Abstract.

Christopher, M. M. I. Wallis, K. A. Nagy, B. T. Henen, C. C. Peterson, B. Wilson, C. Meienberger, and I. Girard. 1993Laboratory health profiles of free-ranging desert tortoises in California: interpretation of physiologic and pathologic alterations. Report to Bureau of Land management, Riverside, Ca.

Christopher, M. M., R. Brigmon, and E. Jacobson. 1994. Seasonal alterations in plasma ß-hydroxybutyrate and related biochemical parameters in the desert tortoise (*Gopherus agassizii*). Comp. Biochem. Physiol. 108A:303-310.

Christopher, M. M., K. A. Nagy, I. Wallis, J. K. Klaassen, and K. H. Berry. 1997. Laboratory health profiles of desert tortoises in the Mojave Desert: a model for health status evaluation of chelonian populations. Proceedings: Conservation, Restoration, and Management of Tortoises and Turtles - An International Conference, pp. 76-82.

Circle Mountain Biological Consultants. 1996. Federal Biological Opinion analysis for the pruposed Eagle Mountain Landfill Project. Rpt. to CH2M Hill, Santa Ana, CA. Circle Mountain Biological Consultants, Wrightwood, CA.

Cline, J. F., and W. H. Rickard. 1973. Herbage yields in relation to soil water and assimilate nitrogen. J. Range Manag. 26:296-298.

Congdon, J. D., A. E. Dunham, R.C. Van Loben Sels. 1993. Delayed sexual maturity and demographics of Blanding's turtles (Emydoidea blandingii): Implications for conservation and management of long-lived organisms. Conservation Biology 7(4): 826-833.

Congdon, J. D., H. W. Avery, and J. R. Spotila. 2002. Feasible demography analyses: consequences of additional adult and juvenile mortality on a stable population of desert tortoises. Desert Tortoise Council Symposium, March 22-25, 2002, Palm Springs, CA. Abstract.

Coombs, E.M. 1979. Food habits and livestock competition with the desert tortoise on the Beaver Dam slope, Utah. Proceedings of the Desert Tortoise Council Symposium 1979:132-147.

Corn, P.S. 1994a. Recent trends of desert tortoise populations in the Mojave Desert. In Bury, R.B. and D.J. Germano. The biology of North American tortoises. Washington, D.C.: United States Department of the Interior, National Biological Service.

Corn, P.S. 1994b. Displacement of desert tortoises: Overview of a study at the Apex Heavy Industrial Zone, Clark County, Nevada. Proc. 1991 Desert Tortoise Council Symposium. 1995: 295-301.

Corn, P. S. 1997. Effects of short-distance translocations on desert tortoises. Abstract. 1997 meeting of Am. Soc. of Ichthyol. and Herpet., Seattle.

Davidson, E. and M. Fox. 1974. Effects of off-road motorcycle activity on Mojave Desert vegetation and soil. Madrono 22:381-412.

DeFalco, L. A. 1995. The influence of cryptobiotic crusts on winter annuals and foraging movements of the desert tortoise. M. S. Thesis, Colorado State University, Fort Collins, CO.

Detling, J.K. 1988. Grasslands and savannas: regulation of energy flow and nutrient cycling by herbivores. Pp. 131-148 In L. R. Pomeroy and J. J. Alberts, (eds.), Concepts in ecosystem ecology: a comparative view. Springer-Verlag, New York.

Doak, D., P. Kareiva, and B. Klepetka. 1994. Modeling population viability for the desert tortoise in the western Mojave Desert. Ecol. Applic. 4:446-460.

Dobson, A. and M. Meagher. 1996. The population dynamics of brucellosis in the Yellowstone National Park. Ecology 77:1026-1036.

Duck, T. A., T. C. Esque, and T. J. Hughes. 1995. Fighting wildfires in desert tortoise habitat: considerations for land managers. Proc. 1994 Desert Tort. Counc. Symp. 1995:58-67.

Durfee, J. A. 1988. Response of Mohave Desert communities to release from grazing pressure. M.S. Thesis. Brigham Young Univ., Orono, UT.

Eckert, R. E., Jr., F. F. Peterson, M. S. Meurisse, and J. L. Stephens. 1986. Effects of soil-surface morphology on emergence and survival of seedlings in big sagebrush communities. J. Range Manage. 39:414-420.

Eckert, R.E., F.F. Peterson, M.K. Wood, and W.H. Blackburn. 1977. Properties, Occurrence and Management of Soils with Vesicular Surface Horizons. Special Study by USDI, BLM and Nevada Agricultural Experiment Station. Contract No. 52500-CT5(N).

Edwards, SW. 1992a. Observations on the prehistory and ecology of grazing in California. Fremontia 20:3-11.

Edwards, SW. 1992b. Observations on the prehistory and ecology of grazing in California. Fremontia 20:34-35.

EG&G Energy Measurements. 1993. Yucca Mountain Biological Resources Monitoring Program: Annual Report FY92. Rpt. to U.S. Dept. of Energy. EG&G Energy Measurements, Santa Barbara Operations, Goleta, CA.

Ehret, G., & H. C. Gerhardt. 1980. Auditory masking and effects of noise on responses of the Green Treefrog (Hyla cinerea) to synthetic mating calls. J. Comp. Physiol. 141:13-18.

Eliason, S. A. and E. B. Allen. 1997. Exotic grass competition in suppressing native shrubland re-establishment. Restoration Ecology 5:245-255.

Engel, K. A., and L. S. Young. 1992. Movements and habitat use by Common Ravens from roost sites in southwestern Idaho. J. Wildl. Manage. 56: 596-602.

Ernst, C. H. J. E. Lovich, and R. Barbour. 1994. Turtles of the United State and Canada. Smithsonian Institution Press, Washington, D.C. 578 pp.

Esque, T. C. 1994. Diet and diet selection of the desert tortoise (Gopherus agassizii) in the northeast Mojave Desert. M.Sc. Thesis, Colorado State Univ. Fort Collins, CO.

Esque, T. C. and C. R. Schwalbe. In press. Alien annual plants and their relationships to fire and vegetation change in Sonoran Desertscrub. *In* Invasive organisms in the Sonoran Desert. Tellman, B. and T. R. Van Devender, eds. Arizona-Sonoran Desert Museum and University of Arizona Press, Tucson.

Esque, T. C., A. Burquez, C. R. Schwalbe, T. R. Van Devender, M. J. Nijhuis, and P. J. Anning. 2002. Effects of fire on desert tortoises and their habitats. Chapt. 10, *in* Sonoran desert tortoise: natural history, biology and conservation. Van Devender, T. R., ed. . Arizona-Sonoran Desert Museum and University of Arizona Press, Tucson.

Feldmeth, R. and R. F. Clements. 1990. City-wide survey of desert tortoise and Mojave ground squirrel: Final report. Rpt. for City of Palmdale. Ecological Research Services, Claremont, CA.

Fenn, M. E., M. A. Poth, J. D. Aber, J. S. Baron, B. T. Bormann, D. W. Johnson, A. D. Lemly, S. G. Mculyy, D. F. Ryan, and R. Stottlemyer. 1998. Nitrogen excess in North American ecosystems: predisposing factors, ecosystem responses, and management strategies. Ecol. Applic. 8:706-733.

Field, K., C. R. Tracy, P. A. Medica, R. M. Marlow, and P. S. Corn. 2000. Translocation as a tool for conservation of the desert tortoise: Can pet tortoises be repatriated? Desert Tortoise Council Symposium, April 20-23, 2000, Las Vegas, NV. Abstract.

Frankham, R. 1995. Inbreeding and extinction: a threshold effect. Conserv. Biol. 9, 792 799.

Frenkel, R. E. 1970. Ruderal vegetation along some California roadsides. Univ. Calif. Press, Berkeley.

Fusco, M.J. 1993. Influence of range condition and watering points on forage production and composition in southcentral New Mexico. Master's Thesis, New Mexico State University, Las Cruces, New Mexico.

Fusco, M.J., J. Holecheck, A. Tembo, A. Danico and M. Cardenas. 1995. Grazing influences on watering point vegetation in the Chihuahauan desert. Journal of Range Management 48:32-38.

Gallagher, C. 1993. American ground zero: the secret nuclear war. Random House Publ.

Germano, D. J. 1988. Age and growth histories of desert tortoises using scute annuli. Copeia 1988:914-920.

Germano, D. J. 1992. Longevity and age-size relationships of populations of desert tortoises. Copeia 1992:367-374.

Gifford, G. F. and R. H. Hawkins 1978. Hydrologic impact of grazing on infiltration: a critical review. Water Resources Research 14: 303-313.

Gibbons, J. W., and J. L. Greene. 1979. X-ray photography: a technique to determine reproductive patterns of freshwater turtles. Herpetologica 35:86-89.

Gillette, D. A. and J. Adams. 1983. Accelerated wind erosion and prediction of rates. Pp. 97-109 in Environmental effects of off-road vehicles: impacts and management in arid regions (Webb, R. H., & H. G. Wilshire, eds.). Springer-Verlag, New York.

Ginn, S. E. 1990. Observations and activities of the naturalist for the Desert Tortoise Natural Area, Kern County, California, March 18-June 2, 1990., Desert Tortoise Preserve Committee, Inc. (Ridgecrest, CA) and the U.S. Bureau of Land Management, Riverside, CA.

Glenn, J. L. 1986. A note on the longevity of a captive desert tortoise (*Gopherus agassizii*). Proc. 1983 Desert Tort. Counc. Symp 1986:131-132.

Goodlett, G. O. and G. C. Goodlett. 1993. Studies of unauthorized off-highway vehicle activity in the Rand Mountains and Fremont Valley, Kern County, California. Proc. 1992 Desert Tort. Counc. Symp. 1993:163-187.

Grover, M. C. and L. A. DeFalco. 1995. Desert tortoise (*Gopherus agassizii*): status-of-knowledge outline with references. Intermountain Research Station, U.S. Forest Service, General Tech. Rpt. INT-GTR-316.

Grumbles, J. S. 1993. Proceedings of the 1993 Desert Tortoise Council Symposium. Abstract.

Hairston, N. G. 1989. Ecological experiments: purpose, design, and execution. Cambridge Univ. Press. Cambridge.

Hansen, R. M., M. K. Johnson, and T. R. Van Devender. 1976. Foods of the desert tortoise *Gopherus agassizii*, in Arizona and Utah. Herpetologica 32:247-251.

Hardy, R. 1976. The Utah population - a look in the 1970's. Proc. Desert Tort. Counc. Symp. 1976:84-88.

Hazard, L. C., D. R. Shemanski, and K. A. Nagy. 2001. Calcium and phosphorus availability in native and exotic food plants. Desert Tortoise Council Symposium, March 2001, Las Vegas, NV. Abstract.

Henen, B.T. 1997. Seasonal and annual energy budget of female desert tortoise (*Gopherus agassizii*). Ecology 76: 283-296.

Heppell, Selina S. 1998. Application of life-history theory and population model analysis to turtle conservation. Copeia 1998: 367-375.

Hereford, R. 2002. Climate variation and geomorphic processes since 1900 in the central Mojave Desert. Desert Tortoise Council Symposium, March 22-25, 2002, Palm Springs, CA. Abstract.

Hillard, S., and C. R. Tracy, C. 1997. Annual activity for juvenile desert tortoises: constraints imposed by the thermal environment. Abstracts of the Third World Congress of Herpetology, 2-10 August 1997, Prague, Czech Republic. Abstract.

Hinckley, B. S., R. M. Iverson, and B. Hallet. 1983. Accelerated water erosion in ORV-use areas. Pp. 81-96 in Environmental effects of off-road vehicles: impacts and management in arid regions (Webb, R. H., & H. G. Wilshire, eds.). Springer-Verlag, New York.

Hinton, T. G., P. D. Fledderman, J. E. Lovich, J. D. Congdon, and J. W. Gibbons. 1997, Radiographic determination of fecundity: is the technique safe for developing turtle embryos? Chelonian Cons. Biol. 2:409-414.

Hobbs, R. J. 1989. The nature and effects of disturbance relative to invasions. Pp. 389-405 In J. A. Drake et al. (eds.). Biological invasions: a global perspective. John Wiley & Sons Ltd., New York.

Hohman, J. and R.D. Ohmart. 1980. Ecology of the desert tortoise on the Beaver Dam Slope, Arizona. Arizona State University. Report for the Bureau of Land management, Arizona Strip Office, St. George, UT.

Homer, B. L., K. H. Berry, M. B. Brown, G. Ellis, E. R. Jacobson. 1998. Pathology of diseases in wild desert tortoises from California. J. Wildl. Diseases 34(3):508-523

Homer, B.L., K.H. Berry, M.M. Christopher, M.B. Brown, E.R. Jacobson. 1994. Necropsies of desert tortoises from the Mojave and Colorado Deserts of California and the Sonoran Desert of Arizona. University of Florida, Gainesville.

Homer, B.L., K.H. Berry, and E.R. Jacobson. 1996. Necropsies of eighteen desert tortoises from the Mojave aud Colorado deserts of California. Final Report to the United States Department of the Interior, National Biological Service, Research Work Order No. 131, Riverside, California, 120 pp.

Hoover, F. G. 1996. An investigation of desert tortoise mortality in upland game guzzlers in the deserts of southern California. Proc. 1995 Desert Tort. Counc. Symp. 1996:36-43.

Howland, J. M. 1989. Observations and activities of the naturalist for the Desert Tortoise Natural Area, Kern County, California, March 18-June 2, 1989. Contract CA 950-CT9-44, Desert Tortoise Preserve Committee, Inc. (Ridgecrest, CA) and the U.S. Bureau of Land Management, Riverside, CA.

Humphrey, R. R. 1958. The desert grassland: a history of vegetational change and an analysis of eauses. Botanical Review 24(4): 193-252.

Humphrey, R.R. 1974. Fire in the deserts and desert grassland of North America. Pp. 365-400 *In* Fire and Ecosystems (Kozlowski, T.T. and C.E. Ahlgrens, eds.). Academic Press, New York.

Humphrey, R. R. 1987. 90 years and 535 miles: vegetation changes along the Mexican border. Albuquerque, New Mexico: University of New Mexico Press.

Hurlbert, S. H. 1984. Pseudoreplication and the design of ecological field experiments. Ecol. Monogr. 54:187-211.

Iverson, R.M. 1979. Processes of accelerated erosion on desert hill-slopes modified by vehicular traffic. Submitted to Earth Surface Processes.

Jackson, L. 1985. Ecological origins of California's Mediterranean grasses. J. Biogeogr. 12:349-361.

Jacobson, E. R. 1993. Implications of infectious diseases for captive propagation and introduction programs of threatened/endangered reptiles. J. Zoo Wildl. Med. 24:245-255.

Jacobson, E. R. 1994. Causes of mortality and diseases in tortoises: a review. J. Zoo and Wildl. Med. 25:2-17.

Jacobson, E. R., M. B. Brown, P. A. Klein, I. Schumacher, D. Morafka, and R. A. Yates. 1996. Serologic survey of desert tortoises, *Gopherus agassizii*, in and around the National Training Center, Fort Irwin, California, for exposure to *Mycoplasma agassizii*, the causative agent of Upper Respiratory Tract Disease. Proc. 1996 Desert Tort. Counc. Symp. 1996:53-54. Abstract.

Jacobson, E.R., J.M. Gaskin, M.B. Brown, R.K. Harris, C.H. Gardiner, J.L. LaPointe, H.P. Adams, C. Reggiardo. 1991. Chronic upper respiratory tract disease of free-ranging desert tortoises (*Xerobates agassizii*). Journal of Wildlife Diseases 27(2):296-316.

Jacobson, E.R., J. Schumacher, and K.H. Berry. 1994. Cutaneous dyskeratosis in free-ranging desert tortoises, *Gopherus agassizii*, in the Colorado desert if southern California. Journal of Zoo Wildlife Medicine 25(1):68-81.

Janzen, D. H. 1986. The eternal external threat. *In* M. E. Soule (ed.) Conservation biology: the science of scarcity and diversity, pp. 286-303. Sinauer Assoc., Sunderland, MA.

Jarchow, J. L. 1989. Report on investigation of desert tortoise mortality on the Beaver Dam Slope, Arizona and Utah. Report to Arizona Game & Fish Dept. Neglected Fauna International, Tucson, AZ.

Jennings, M. R. 1981. *Gopherus agassizii* (desert tortoise). Longevity. Herp. Review 12:81-82.

Jennings, W. B. 1993. Foraging ecology of the desert tortoise (*Gopherus agassizii*) in the western Mojave desert. Master's thesis. Arlington, University of Texas: 101 pp.

Jennings, W. B. 1997. Habitat use and food preferences of the desert tortoise, *Gopherus agassizii*, in the western Mojave Desert and impacts of off-road vehicles. *In* J. Van Abbema (ed.), Proceedings: Conservation, Restoration, and Management of Tortoises and Turtles-An International Conference, pp. 42-45. New York Turtle and Tortoise Society, New York.

Johnson, A. E. Jacobson, D. J. Morafka, F. Origgi, and L. Wendland. 2002. Prevalence of URTD in captive desert tortoises on and adjacent to Fort Irwin: potential impacts to wild populations. Desert Tortoise Council Symposium, March 22-25, 2002, Palm Springs, CA. Abstract.

Johnson, H. B., F. C. Vasek, and T. Yonkers. 1975. Productivity, diversity, and stability relationships in Mojave Desert roadside vegetation. Bull. Torrey Bot. Club 102:106-115.

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

Knight, R. L., and J. Kawashima. 1993. Responses of raven and Red-tailed Hawk populations to linear right-of-ways. J. Wildl. Manage. 57: 266-271.

Knight, R. L., H. L. Knight, and R. J. Camp. 1993. Raven populations and land-use patterns in the Mojave Desert, California. Wildl. Soc. Bull. 21: 469-471.

Knight, R. L., R. J. Camp, W. I. Boarman, and H. A. L. Knight. 1999. Predatory bird populations in the east Mojave Desert, California. Great Basin Natur. 59:331-338.

Knowles, C., C. Guntow, P. Knowles, P. Houghton.1989. Relative abundance and distribution of the common raven in the desert of southern California. Boulder, MT, FaunaWest Wildlife Consultants.

Koehler, J.H. 1977. Ground water in the Koehn Lake area, Kern County, California. U.S. Geological Survey, Water Resources Investigations 77-66.

Kristan, W.B. III, and W. I. Boarman. 2001. The spatial distribution of common ravens (*Corvus corax*) and raven depredation. *In:* W. B. Kristan, III, ed., Effects of habitat selection on avian population ecology in urbanizing landscapes. Ph.D. Dissertation, University of California, Riverside. Riverside, CA 92521.

Krzysik, A. J. 1998. Desert tortoise populations in the Mojave Desert and a half-century of military training activities. Pp. 61-73 in (J. Van Abbema, ed.) Proceedings: conservation, restoration, and management of tortoises and turtles-an international conference. NY Turtle and Tortoise Soc., NY.

Kuchling, G. 1998. How to minimize risk and optimize information gain in assessing reproductive condition and fecundity of live female Chelonians. Chelonian Cons. Biol. 3:118-123.

Lance, V. A., D. C. Rostal, J. S. Grumbles, and I. Schumacher. 1996. Effects of Upper Respiratory Tract Disease on reproduction and steroid hormone levels in male and female desert tortoises. Proc. 1995 Desert Tort. Counc. Symp. 1996:104.

Larsen, R. E., P. E. Padgett, S.N. Kee, E.B. Allen, H.W. Avery. 1997. Accomplishment Report. FY 1996-1997 DANR Competitive Grant Project #15. San Bernardino, UC Cooperative Extension: 17.

LaRue, E. L., Jr. 1993. Distribution of desert tortoise sign adjacent to Highway 395, San Bernardino County, California. Proceedings of the 1992 Desert Tortoise Council Symposium 1993: 190-204.

LaRue, E. and S. Dougherty. 1999. Federal Biological Opinion analysis for the Eagle Mountain Landfill project. Proc. 1997-1998 Desert Tort. Counc. Symp. 1999:52-58.

Lathrop, E. W. 1978. Plant response parameters to recreational vehicles in the California Desert Conservation Area. Final Report, Contract CA-060-CT7-2824, Desert Plan Program, Bureau of Land management, Riverside, CA.

Lathrop, E. W. 1983. Recovery of perennial vegetation in military maneuver areas. Pp. 265-277 in (R. H. Webb and H. G. Wilshire, eds) Environmental effects of off-road vehicles: impacts and management in arid regions. Springer-Verlag, NY.

Loftin, S. R. 1987. Postfire dynamics of a Sonoran Desert ecosystem. Master's Thesis. University of Arizona, Tempe. 89 pp.

Lovich, J. E. and D. Bainbridge. 1999. Anthropogenic degradation of the Southern California desert ecosystem and prospects for natural recovery and restoration. Environ. Mgmt. 24:309-326.

Lovich, J. E. and R. Daniels. 2000. Environmental characteristics of desert tortoise (*Gopherus agassizii*) burrow locations in an altered industrial landscape. Chelon. Conserv. Biol. 3:714-721.

Lovich, J., P. Medica, H. Avery, K. Meyer, G. Bowser, and A. Brown. 1999. Studies of reproductive output of the desert tortoise at Joshua Tree National Park, the Mojave National Preserve, and comparative sites. Park Science 19:22-24.

Lubchenco, J. and L. A. Real. 1991. Manipulative experiments as tests of ecological theory. Pp. 715-733 *In* Foundations of ecology: Classic papers with commentaries (L. A. Real and J. H. Brown, eds.). Univ. of Chicago Press, Chicago, IL.

Luckenbach, R. A. 1975. What the ORVs are doing to the desert? Fremontia 2(4): 3-11.

Luckenbach, R. 1982. Ecology and management of the desert tortoise (*Gopherus agassizii*) in California. *In* North American Tortoises: Conservation and Ecology. R. B. Bury. Washington, D.C., United States Department of the Interior Fish and Wildlife Service. Wildlife Research Report 12.

Luke, C., A. Karl, P. Garcia. 1991. A status review of the desert tortoise. Tiburon, California: Biosystems Analysis, Inc.

Lyon, R. H. 1973. Propagation of environmental noise. Science 179 : 1083-1090.

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

Marlow, R. W. 1974. Preliminary Report on Population Dynamics in the Desert Tortoise (*Gopherus agassizii*) in the Western Mojave Desert, California. Berkeley, CA, University of California Museum of Vertebrate Zoology: 31.

Maryland, H. F. and T. H. McIntosh. 1966. Availability of biologically fixed atmospheric nitrogen-15 to higher plants. Nature 209:421-422.

McCullough Ecological Systems. 1995. Avian predation of juvenile desert tortoises along transmission line corridors in the Piute-Eldorado Critical Habitat Unit. McCullough Ecological Systems, Las Vegas.

McNaughton, S. J. 1985. Ecology of a grazing system: the Serengeti. Ecol. Monogr. 55:259-294.

Medica, P.A., B.R. Bury, and F.B. Turner. 1975. Growth of the desert tortoise (*Gopherus agassizii*) in Nevada. Copeia 4:639-643.

Medica, P. A., C. L. Lyons, F.B. Turner. 1982. A Comparison of 1981 Populations of Desert Tortoises (*Gopherus agossizii*) in Grazed and Ungrazed Areas in Ivanpah Valley, California. Desert Tortoise Council 1982 Symposium, Las Vegas, NV, Desert Tortoise Council.

Meffe, G. K. and C. R. Carroll. 1997. Principles of Conservation Biology. 2nd Ed. Sinauer Assoc., Inc., Publ., Sunderland, MA.

Morafka, D. J. 1994. Neonates: Missing links in the life histories of North American tortoises. Pp. 161-173 *In* Biology of North American tortoises (R. B. Bury and D. J. Germano, eds.). Fish and Wildlife Research No. 13. U. S. Dept. Interior, National Biological Survey, Wash., DC.

Mortimer, C. and P. Schneider. 1983. Population studies of the desert tortoise (*Gopherus agassizii*) in the Piute Valley study plot of southern Nevada. Report to Nevada Dept. of Wildlife.

Mount, R.H. 1986. Special Concern: Box turtle *Terrapene carolina* ssp. In Mount, R.H., ed. Vertebrate animals of Alabama in need of special attention. Alabama Agriculture Experimental Station, Auburn University.

Murcia, C. 1995. Edge effects in fragmented forests: implications for conservation. Trends Ecol Evol. 10:58-62.

Nagy, K.A., B.T. Henen, and D.B. Vyas. 1998. Nutritional quality of native and introduced food plants of wild desert tortoises. J. Herpet. 32:260-267.

Nagy, K.A. and P.A. Medica. 1986. Physiological ecology of desert tortoises in southern Nevada. Herpetologica 42(1):73-92.

Nicholson, L. 1978. The effects of roads on desert tortoise populations. Proceedings of 1978 Desert Tortoise Council Symposium 1978:127-129.

Nicholson, L. and K. Humphreys 1981. Sheep grazing at the Kramer study plot, San Bernardino County, California. Proceedings of the Desert Tortoise Council 1981 Symposium

Nilsen, E. T., P. W. Rundel, and M. R. Sharifi. 1984. Productivity in native stands of *Prosopsis glandulosa*, mesquite, in the Sonoran Desert of southern California and some management implications. Pages 722-727 *in* R. E. Warner and K. M. Hendrix (eds.). California riparian systems: ecology, conservation, and productive management. Univ. Calif. Press, Berkeley.

Nussear, K. E., C. R. Tracy, P. A. Medica, R. A. Marlow, M. B. Saethre, and P. S. Corn. 2000. Translocation as a tool for conservation of the desert tortoise: Nevada studies. Desert Tortoise Council Symposium, April 20-23, 2000, Las Vegas, NV. Abstract.

O'Connor, M. P., J. S. Grumbles, R. H. George, L. C. Zimmerman, and J. R. Spotila. 1994a. Potential hematological and biochemical indicators of stress in free-ranging desert tortoises and captive tortoises exposed to a hydric stress gradient. Herpetol. Monogr. 8:5-26.

O'Connor, M. P., L. C. Zimmerman, D.E. Ruby, S.J. Bulova, J.R. Spotila. 1994b. Home range size and movements by desert tortoises, *Gopherus agassizii*, in the eastern Mojave Desert. Herpetological Monographs 8: 60-71.

Oftedal, O. 2001. Low rainfall affects the nutritive quality as well as the total quantity of food available to the desert tortoise. Desert Tortoise Council Symposium, March 16-18, 2001, Las Vegas, NV. Abstract.

O'Leary, J. F. and R. A. Minnich.1981. Postfire Recovery of Creosote Bush Scrub Vegetation in the Western Colorado Desert. Madrono 28(2): 61-66.

Oldemeyer, J. 1994. Livestock grazing and the desert tortoise in the Mojave Desert. *In* The biology of North American tortoises. R. B. Bury and D. J. Germano. Washington, D. C., United States Department of the Interior, National Biological Service: pp. 95-104.

Olson, T. E. 1996. Comparison of impacts and mitigation measures along three multi-state linear construction projects. Proceedings of the Desert Tortoise Council 1996 Symposium. 1996:1-9.

Olson, T. E., K. Jones, D. McCullough, and M. Tuegel. 1993. Effectiveness of mitigation for reducing impacts to desert tortoise along an interstate pipeline route. Proceedings of the 1992 Desert Tortoise Council Symposium 1993:209-219.

Olson, T. E., C. Luke, K. W. MacDonald, and H. D. Hiatt. 1992. Monitoring for desert tortoises during construction of two interstate pipelines. 1992 Transactions of the Western Section of the Wildlife Society 28:15-21.

Opdam, P. 1988. Populations in fragmented habitat. In: Schreiber, K. F. (Ed.), Connectivity in Landscape Ecology. Proceedings of the Second International Seminar of the International Association for Landscape Ecology, Munster, Germany, 1987. Munstersche Geographische Arbeiten 29, pp. 75-78.

Origgi, F., C. H. Romero, P. Klein, K. Berry, and E. Jacobson. 2002. Serological and molecular evidences of herpesvirus exposure in desert tortoises from the Mojave Desert of California. Desert Tortoise Council Symposium, March 22-25, 2002, Palm Springs, CA. Abstract.

Painter, E. L. and A. J. Belsky. 1993. Application of herbivore optimization theory to rangelands of the western United States. Ecol. Monogr. 3:2-9.

Pauli, F. 1964. Soil fertility problems in arid and semi-arid lands. Nature 204:1286-1288.

Patterson, R. G. 1971. Vocalization in the Desert Tortoise, *Gopherus agassizii*. M.A. Thesis, California State Univ., Fullerton.

Patterson, R. G. 1976. Vocalization in the desert tortoise. Proc. Desert Tortoise Council Symp. 1976: 77-83.

Patterson, R. and B. Brattstrom. 1972. Growth in captive *Gopherus agassizii*. Herpetologica 28:169-171.

Peterson, C. C. 1994a. Different rates and causes of high mortality in two populations of the threatened desert tortoise *Gopherus agassizii*. Biol. Conserv. 70:101-108.

Peterson, C. C. 1994b. Physiological ecology of the desert tortoise, *Xerobates agassizii* . pp. 213-224 in: P. R. Brown and J. W. Wright, eds., Herpetology of the North American Deserts. Proc. of a Symp.

Peterson, C.C. 1996. Ecological energetics of the desert tortoise (*Gopherus agassizii*): effects of rainfall and drought. Ecology 77:1831-1844.

Pettan-Brewer, K. C. B., M. L. Drew, E Ramsay, F. C. Mohr, J. J. Lowenstine. 1996. Herpesvirus particles associated with oral and respiratory lesions in a California desert tortoise (Gopherus agassizii). Journal of Wildlife Diseases 32: 521-526.

Platt, J. R. 1964. Strong inference. Science 146:347-353.

Prose, D.V., and S. K. Metzger. 1985. Recovery of soils and vegetation in World War II military base camps, Mojave Desert. U.S. Geological Survey Open-File Report 85-234, Reston.

Prose, D.V., S. K. Metzger, and H. G. Wilshire. 1987. Effects of substrate disturbance on secondary plant succession: Mojave Desert, California. J. Appl. Ecol. 24:305-313.

Ratti, J. T. and K. P Reese. 1988. Preliminary tests of the ecological trap hypothesis. J. Wildl. Manag. 52:484-491.

Rauzi, F. and F. M. Smith. 1973. Infiltration rates: three soils with three grazing levels in northeastern Colorado. Journal of Range Management 26(2): 126-129.

Reicosky, D.C., W.B. Voorhess, and J.K. Radke. 1979. Unsaturated water flow through a simulated wheel track. Abstract of talk presented at 1979 annual meetings of American Society of Agronomy, Fort Collins, CO.

Resnik, D. B. 1991. How-possibly explanations in Biology. Acta Biotheoretica 39:141-149.

Resource Concepts. 1996. Desert tortoise situation review. Report to Public Works Group, County of San Bernardino, CA. Resource Concepts, Inc., Carson City, NV.

Reynolds, H.G. and J.W. Bohning. 1956. Effects of burning on a desert grass-scrub range in southern Arizona. Ecology 37:769-778.

Rogers, RF. W., T. T. Lange, and D. J. D. Nichols. 1966. Nitrogen fixation by lichens of arid soil crusts. Nature 209:96-97.

Rosskopf, W. J., Jr. 1991. Protocols for handling live tortoises. Chapt. III *In* Interim techniques handbook for collecting and analyzing data on desert tortoise populations and habitats. Publ. by Arizona Game and Fish Dept.

Rostal, D. C., V. A. Lance, J. S. Grumbles, and A. C. Alberts. 1994. Seasonal reproductive cycle of the desert tortoise (*Gopherus agassizii*) in the eastern Mojave Desert. Herp. Monogr. 8:72-82.

Rowlands, P.G. 1980. Soil crusts. Pp. 46-62, Chapt. II. *In*: The effects of disturbance on desert soils, vegetation and community process with emphasis on off road vehicles: a critical review (Rowlands, P. G., Ed.). Desert Plan Staff, Bureau of Land Management, Riverside, CA.

Rowlands, P.G., J.A. Adams, H.B. Johnson, and A.S. Endo. 1980. Experiments on the effects of soil compaction on establishment, cover, and pattern of winter and summer annuals in the Mojave desert. Pp. 75-120, Chapt. VI. *In*: The effects of disturbance on desert soils, vegetation and community process with emphasis on off road vehicles: a critical review (Rowlands, P. G., Ed.). Desert Plan Staff, Bureau of Land Management, Riverside, CA.

Ruby, D. E., and H. A. Niblick. 1994. A behavioral inventory of the desert tortoise: development of an ethnogram. Herpetol. Monogr. 8:88-102.

Ruby, D. E., L. C. Zimmerman, S. J. Bulova, C. J. Salice, M. P. O'Connor, and J. R. Spotila. 1994. Behavioral responses and time allocation differences in desert tortoises exposed to environmental stress in semi-natural enclosures. Herpetological Monographs 8:27-44.

Rundel, P. W. and A. C. Gibson. 1996. Ecological communities and processes in a Mojave Desert ecosystem: Rock Valley, Nevada. Cambridge Univ. Press, Cambridge.

St. Clair, L. L., B. L. Webb, J. R. Johansen, and G. T. Nebecker. 1984. Cryptogamic soil crusts: enhancement of seedling establishment in disturbed and undisturbed areas. Reclam. Reveg. Res. 3:129-136.

Savory, A. 1989. Holistic resource management. Island Press, Washington, DC.

Schumacher, I. M., M. B. Brown, E. R. Jacobsun, B. R. Collins, P. A. Klein. 1993. Detection of antibodies to a pathogenic mycoplasma in desert tortoises (*Gopherus agassizii*) with upper respiratory tract disease. Journal of Clinical Microbiology 31(6):1454-1460.

Schumacher, I. M., D. B. Hardenbrook, M. B. Brown, E. R. Jacobson, and P. A. Klein. 1997. Relationship between clinical signs of Upper Respiratory Tract Disease and antibodies to *Mycoplasma agassizii* in desert tortoises from Nevada. J. Wildl. Diseases 33:261-266.

Sharifi, M. R., A. C. Gibson, and P. W. Rundel. 1997. Surface dust impacts on gas exchange in Mojave Desert shrubs. J. Appl. Ecol. 34:837-846.

Sharifi, M. R., A. C. Gibson, and P. W. Rundel. 1999. Phenological and physiological responses of heavily dusted creosote bush (*Larrea tridentata*) to summer irrigation in the Mojave Desert. Flora 194:369-378.

Sheppard, G. P. 1981. Desert tortoise population of the Beaver Dam Slope in northwestern Arizona. Desert Tortoise Council Proceedings of the 1981 Symposium.

Sherman, M. W. 1993. Activity patterns and foraging ecology of nesting Common Ravens in the Mojave Desert, California. M. S. thesis, Colorado State Univ., Fort Collins.

Shields, T. 1994. Field sampling of small tortoises: three experiments. Proc. 1987-1991 Desert Tortoise Council Symposium. 1994:374.

Snyder, C.T., D.G. Frickel, R.F. Hadley, and R.F. Miller. 1976. Effects of off-road vehicle use on the hydrology and landscape of arid environments in central and southern California. U.S. Geological Survey Water Resources Investigations 76-99, 27 pp.

Soltero, S., F. C. Bryant, and A. Melgoza. 1989. Standing crop patterns under short duration grazing in northern Mexico. J. Range Manag. 42:20-22.

Spotila, J. R., L. C. Zimmerman, C. A. Binckley, J. S. Grumbles, D. C. Rostal, A. List, Jr., E. C. Beyer, K. M. Philips, and S. J. Kemp. 1994. Effects of incubation conditions on sex determination, hatching success, and growth of hatchling desert tortoises, *Gopherus agassizii*. Herptol. Monogr. 8:103-116.

Steiger, T.L. 1930. Structure of prairie vegetation. Ecology 11:170-217.

Stewart, G. A. 1993. Movements and survival of desert tortoises (*Gopherus agassizii*) following relocation from the Luz Solar Electric Plant at Kramer Junction. Proc. 1992 Desert Tort. Council Symp. 1992:234-261.

Stewart, G. R. and R. J. Baxter. 1987. Final report and habitat management plan for the desert tortoise (*Gopherus agassizii*) in the West and Sand Hill Training Areas of the Twentynine Palms MCAGCC. Rpt. to U.S. Dept. of Navy. Calif. State Polytech. Univ., Pomona.

Stubbs, D. 1989. *Testudo kleinmanni*. Egyptian tortoise. In Swingland, I. R. and M.W. Klemens, eds. The conservation biology of tortoises. Occasional papers of the IUCN Species Survival Commission No. 5 and the Durrell Institute of Conservation Ecology. IUCN.

Stubbs, D. 1991. Tortoise and freshwater turtles: an action plan for their conservation. IUCN/SSC Tortoise and Freshwater Turtle Specialist Group. 2nd Ed. IUCN-The World Conservation Union, Gland, Switzerland.

Tierra Madre Consultants. 1991. Biological assessment for Lancaster City and Planning Area: Relative density surveys for desert tortoises and cumulative human impact evaluations for Mohave ground squirrel habitat. Rpt. for City of Lancaster. Tierra Madre Consultants, Riverside, CA.

Tracy, C. R. 1995. Patterns of fire incidence and implications for management of Desert Wildlife Management Areas. Proc. 1994 Desert Tort. Counc. Symp. 1995:179.

Tracy, C. R. 1996. Nutritional ecology of the desert tortoises: preliminary assessment of some impacts dne to sheep grazing in the California Desert District. Report to National Biological Service. Univ. Nevada, Reno.

Tully, J. G. 1998. The important and continuing role of mycoplasmas in respiratory diseases of various animal hosts. Desert Tortoise Couneil Symposium, April 3-5, 1998, Tncson, AZ. Abstract.

Turner, F. B. 1975. Effects of continuous irradiation on animal populations. Pp. 83-144 in: J. T. Lett and H. Adler, eds., Advances in radiation biology, vol. 5. Academic Press, NY.

Turner, F.B., K.H. Berry, D.C. Randall, and G.C. White. 1987b. Population ecology of the desert tortoise at Goffs, California 1983-1986. Report to South. Calif. Edison. Laboratory of Biomed. and Environ. Sci., Univ. Calif, Los Angeles.

Turner, F. B., P. Hayden, B. L. Burge, and J. B. Roberson. 1986. Egg production by the desert tortoise (Gopherus agassizii) in California. Herpetologica 42:93-104.

Turner, F. B. and P. A. Medica. 1977. Sterility among female lizards (*Uta stansburiana*) exposed to continuous Y irradiation. Radiation Research 70:154-163.

Turner, F.B., P.A. Medica, and C.L. Lyons. 1981. A comparison of populations of the desert tortoise (*Gopherus agassizii*) in grazed and ungrazed areas in Ivanpah Valley, California. Proceedings of the 1981 Desert Tortoise Council Symposium.

Turner, F.B., P.A. Medica, and C.L. Lyons. 1984. Reproduction and survival of the desert tortoise (*Scaptochelys agassizii*) in Ivanpah Valley, California. Copeia 1984:811-820.

Turner, F.B., P.A. Medica, and R. B. Bury. 1987a. Age-size relationships of desert tortoises (*Gopherus agassizii*) in southern Nevada. Copeia 1987:974-979.

Turner, F. B., C. G. Thelander, D. C. Pearson, and B. L. Burge. 1985. An evaluation of the transect technique for estimating desert tortoise density at a prospective power plant site in Ivanpah Valley, California. Proceedings of the 1982 Desert Tortoise Council Symposium 1985:134-153.

U. S. Department of Energy. 1994. United States nuclear tests July 1945 through 1992. DOE/NV 209 (Rev. 14) Dec. 1994. USDOE Nevada Operations Office.

U. S. Fish and Wildlife Service. 1994. Desert tortoise (Mojave population) Recovery Plan. U.S. Fish and Wildlife Service, Portland OR.

Van Devender, T. R., R. S. Thompson, J. L. Betancourt. 1987. Vegetation history of the deserts of southwestern North America; the nature and timing of the late Wisconsin-Holocene transition. Pages 323-352 *In* W. F. Ruddiman and H. E. Wright (eds.), North America and adjacent oceans during the deglaciation. The Geological Society of America, Boulder, CO.

Vasek, F. C., H. B. Johnson, and D. H. Eslinger. 1975. Effects of pipeline construction on creosote bush scrub vegetation on the Mojave Desert. Madrono 23:1-13.

Vollmer, A.T., B.G. Maza, P.A. Medica, F.B. Turner and S.A. Bamberg. 1976. The impact of off-road vehicles on a desert ecosystem. Environmental Management. 1(2):115-129.

von Seckendorff Hoff, K., and R. Marlow. 1997. Highways and roads are population sinks for desert tortoises. Proceedings: Conservation, Restoration, and Management of Tortoises and Turtles - An International Conference, p. 482.

Waller, T., and P. A. Micucci. 1997. Land use and grazing in relation to the genus *Geochelone* in Argentina. Proceedings: Conservation, Restoration, and Management of Tortoises and Turtles - An International Conference, pp. 2-9.

Webb, R. H. 1983. Compaction of desert soils by off-road vehicles. Pp. 51-79 in Environmental effects of off-road vehicles: impacts and management in arid regions (Webb, R. H., & H. G. Wilshire, eds.). Springer-Verlag, New York.

Webb, R. H., H. C. Ragland, W. H. Godwin, and D. Jenkins. 1978. Environmental effects of soil property changes with off-road vehicle use. Environ. Management 2(3):219-233.

Webb, R. H. and S. S. Stielstra. 1979. Sheep grazing effects on Mojave Desert vegetation and soils. Environ. Manage. 3:517-529.

Weinstein, M. 1993. Health profile results from the Honda desert tortoise relocation project. Proc. 1992 Desert Tort. Counc. Symp.:58.

Weltz, M., M.K. Wood, and E.E. Parker. 1989. Flash grazing and trampling: effects on infiltration rates and sediment yield in a selected New Mexico range site. J. Arid Environ. 16:95-100.

Wilcove, D. S., C. H. McLellan, and A. P. Dobson. 1986. Habitat fragmentation in the temperate zone. . *In* M. E. Soule (ed.) Conservation biology: the science of scarcity and diversity, pp. 237-256. Sinauer Assoc., Sunderland, MA.

Willis, W.O. and W.A. Raney. 1971. Effects of compaction on content and transmission of heat in soils, pp. 165-177. *In* Compaction of Agricultural Soils. American Society of Agricultural Engineers Monograph.

Wilms, W.D., S. Smoliak, and J.F. Dormarr. 1990. Vegetation response to time-controlled grazing on mixed and fescue prairies. J. Range Manage. 43:513-517.

Wilshire, H.G. 1980. Human causes of accelerated wind erosion in California's desert. In Coates, D.R. and J.D. Vitek. Thresholds in geomorphology. London: Allen and Unwin, Limited.

Wilshire, H. G. and J. K. Nakata 1976. Off-road vehicle effects on California's Mojave desert. California Geology (June 1976): 123-132.

Wilson, D. S., C. R. Tracy, K. E. Nussear, E. T. Simandle, R. M. Marlow, P. A. Medica, and P. S. Corn. 2000. Translocation as a tool for conservation of the desert tortoise: Utah studies. Desert Tortoise Council Symposium, April 20-23, 2000, Las Vegas, NV. Abstract.

Winkel, V.K. and B.A. Roundy . 1991. Effects of cattle trampling and mechanical seedbed preparation on grass seedling emergence. J. Range Mgmt. 44:176-183.

Woodbury, A. M. and R. Hardy. 1948. Studies of the desert tortoise, Gopherus agassizii. Ecological Monographs 18: 145-200.

Woodman, A. P. 1986. Effects of Parker 400 off-road race on desert tortoise habitat in Chemehuevi Valley, CA. Proc. Desert Tortoise Council 1983: 69-79.

Woodman, A.P. and K.H. Berry. 1984. A description of carcass deterioration of the desert tortoise and a preliminary analysis of disintegration rates at two sites in the Mojave Desert, California. In Berry, K., Ed. The status of the desert tortoise (*Gopherus agassizii*) in the United States. Riverside, California: United State Department of the Interior, Bureau of Land Management.

Woodman, A.P. and S.M. Juarez. 1988. Juvenile desert tortoises utilized as primary prey of nesting common ravens near Kramer, California. Paper presented at the 13th Annual Meeting and Symposium of the Desert Tortoise Council held March 26-27, 1988, Laughlin, Nevada.

REVIEW OF POTENTIAL EDGE EFFECTS ON THE SAN FERNANDO VALLEY SPINEFLOWER

(*Chorizanthe parryi* var. *fernandina*)

Prepared for:

Ahmanson Land Company
and
Beveridge & Diamond, LLP

Prepared by:

Conservation Biology Institute

January 2000

INTRODUCTION

The recent discovery of the San Fernando Valley spineflower (*Chorizanthe parryi* var. *fernandina*) on the Ahmanson Ranch project site in Ventura County, California prompted preliminary investigations into the biology of that taxon. The purpose of these studies is to develop a conservation strategy to protect, maintain, manage, and, possibly, reintroduce the spineflower into appropriate habitat. While the proposed development would remove a portion of the spineflower population, the majority of the known population is proposed to be conserved onsite. Residential development is planned adjacent to the proposed spineflower preserve area.

An effective conservation strategy should emphasize preserve design and habitat and species management. Accepted principles of preserve design include maximizing the width of the buffer between development and sensitive resources, minimizing habitat loss, fragmentation, and edge effects, maximizing genetic diversity and connectivity with other habitat patches, maintaining adequate habitat to allow for spatial and temporal population fluctuations, and maintaining a sustainable population size,[1] among others. Habitat and species management may be necessary to mitigate impacts from adjacent development and to maintain the functions and values of the population being conserved.

This paper assesses potential impacts to the conserved spineflower population from adjacent development based on a review of the scientific literature on edge effects (adverse effects of land uses on adjacent biological resource areas, such as weed invasions or changes in hydrology). A thorough literature search on edge effects has not been conducted for this paper due to time limitations. The summary presented herein is intended to (1) focus on potential impacts to sensitive plant species, and (2) address those risk factors associated with edge effects most likely to affect the spineflower, based on current knowledge of the species' biology. All identified risk factors have the potential to negatively impact some aspect of the species' biology or habitat; however, information is not yet available to definitively determine which factors pose the most serious threat to the species' persistence. This paper analyzes identified risk factors in relation to preserve design and proposes management actions and alternative scenarios to minimize or reduce the potential impacts of these risk factors.

SPINEFLOWER BIOLOGY

The biology of a species holds implications for preserve design and habitat management. Additional research is needed to assess the long-term viability of the spineflower population on Ahmanson Ranch and to identify specific management measures to ensure its persistence. This section summarizes our current knowledge of spineflower biology and limitations to our knowledge.

---

1 Note that a sustainable population is not measured by species presence alone, but by the *effective* size of a population in contributing to future generations relative to an *ideal* population. The effective population size may be smaller than the census population number. Estimates of effective population size may be determined through demographic monitoring or genetic studies (Barrett and Kohn 1991).

San Fernando Valley spineflower is a small annual plant in the buckwheat family (Polygonaceae). This low-growing species is characterized by prostrate to ascending stems, small white flowers, and straight involucral awns (Hickman 1993). Historical habitat for the San Fernando Valley spineflower was apparently deep, low nutrient soils of sand benches, or soils with similar characteristics that occurred as mosaics within coastal sage scrub and, possibly, valley grassland (GLA 1999). Although soils on the property are generally well drained, acidic, and low in nitrogen and organics (GLA 1999), preliminary studies indicate that the spineflower population on Ahmanson Ranch occurs in open areas on compacted or recently disturbed soils that support few other plant species. It is unclear whether this association indicates that the spineflower prefers compacted soils, or if it is restricted to compacted soils by competition from other plant species that avoid the compacted soils. Also, it is not clear whether the density of spineflower plants differs between compacted and non-compacted soils. If spineflower densities are lower than normal on compacted soils, this may have long-term genetic consequences if spineflower populations are restricted to compacted areas in the future, either by preserve design or lack of effective management to reduce competition from other species. It has been suggested that lowered plant density has the same effect on reproductive success as small population size (Lamont et al. 1993; van Treuren et al. 1993; Groom 1998) for some insect-pollinated plants. Theoretical models that have included population density or size as input factors indicate that extinction rates increase dramatically as density declines, and extinction becomes almost inevitable below certain density thresholds (Dennis 1989 in Groom 1998; Kunin and Iwasa 1996 in Groom 1998). Groom (1998) documented that small patches of an annual herb suffered reproductive failure due to lack of effective pollination when critical thresholds of isolation were exceeded. In contrast, large patches attracted pollinators regardless of the level of isolation.

San Fernando Valley spineflower most likely forms a persistent seed bank in the soil, with seeds germinating under specific climatic conditions (e.g., appropriate temperature and amount and timing of rainfall). Seed banks typically contain multiple genotypes from various years, but years yielding large seed crops contribute disproportionately to the bank (Templeton and Levin 1979). In this respect, seed banks contain the "evolutionary memory" of a species (Del Castillo 1994). Seed banks buffer changes in population size, and help maintain genetic diversity and genetic spatial distribution (Del Castillo 1994). Seedling survival may depend on adequate rainfall, as well as light and nutrient conditions. These factors influence the degree of competition between the spineflower and other plant species.

Little is known about the reproductive biology of the spineflower (including whether the species is strictly outcrossing or can also self-pollinate). A wide range of insect visitors was observed on spineflower flowers during the 1999 field surveys (GLA 1999), but it has not yet been determined if any of these are effective pollinators. Insects observed on the spineflower included ants (mostly of the *Dorymyrex insanus* complex), ant-like spiders (possibly *Micaria* spp.), European honeybee (*Apis mellifera*), bee-flies (Bombyliidae), a small bumblebee (*Bombus* sp.), and tachnid flies (possibly *Archytas* spp.) (GLA 1999). Of these species, the ants appeared to be the most frequent flower visitors. Determination of the reproductive strategy is necessary to assess whether pollinators are important in maintaining the spineflower population. Determining the specific pollinator(s) is important in identifying the range and type of habitat(s) required for maintaining an effective pollinator population(s). Based on his work with a related taxon (*Eriogonum*) and on the spineflowers' floral morphology, Dr. James Reveal (pers. comm.) suggests that the San Fernando Valley spineflower may be capable of both cross-pollination and self-pollination. Outcrossing would likely be the primary means of reproduction, because of (1) the presumed differential timing between pollen release and stigma receptivity and (2) the spatial separation between anthers and stigma. Late in the pollination cycle, however, the still-receptive stigma may roll back and pick up any remaining pollen, thereby resulting in self-pollination. Although self-pollination may result in production of viable seed and ensure short-term persistence, it may also lead to reduced genetic diversity over time (Reveal pers. comm.). Dr. Eugene Jones (pers. comm.) is in the process of determining some of these reproductive characteristics for the San Fernando Valley spineflower (e.g., whether the flowers are protandrous versus protogynous, whether self-pollination is autogamous versus geitonogamous, etc.).[2] Dr. Jones notes that related taxa having similar floral structures may function differently from one another.

Reveal (pers. comm.) has observed other spineflower species being effectively pollinated by ants, but indicates that, in those cases, ants are incidental (secondary) rather than primary pollinators. Jones (pers. comm.) observed high densities of ants in and out of spineflower corollas in the field, and suggests that ants may play an important role in pollination of this species. Hickman (1974) demonstrated ant pollination as a specialized mutualistic system in another annual species within the buckwheat family, *Polygonum cascadense*. *Polygonum cascadense* shares several similarities with the spineflower, including habit (e.g., low, erect annual), habitat (e.g., open, dry slopes), and possibly, reproductive characteristics (e.g., stamens maturing before the stigma).

The spineflower involucre (whorl of modified leaves adjoining each flower) is characterized by straight spines, which may be an adaptation for animal dispersal of seeds and may help anchor seeds to suitable substrate (GLA 1999). Seeds apparently remain in the involucre even after the plant disarticulates. Small mammals or even ants may play a role in seed dispersal; however, studies have not yet been conducted to determine whether any animals onsite effectively disperse spineflower seed. Reveal (pers. comm.) notes that gallinaceous birds that peck and scratch at the soil surface can be effective in planting seeds of chorizanthoid species as non-incidental dispersal agents, and that localized dispersal may also be accomplished by small mammals. The one season of data indicates relatively high seed production for the spineflower. It is not known whether seed predation by animals significantly affects the seed bank.

---

[2] Protandry refers to the condition in which flowers shed their pollen before the stigma becomes receptive. Protogyny refers to the opposite condition, i.e., the stigma matures and becomes receptive before the anthers dehisce and shed pollen. Autogamy refers to self-pollination that occurs when a flower is pollinated by its own pollen, whereas geitonogamy is the condition in which a flower is pollinated by pollen from another flower on the same plant. The latter is, in effect, self-pollination because the results are genetically identical to pollination by autogamy (Proctor et al. 1996).

Fire has been suggested as a possible management tool for maintaining or enhancing spineflower habitat. The effects of fire on germination of the San Fernando Valley spineflower have not yet been established. Studies on a closely related taxon (*Chorizanthe parryi* var. *parryi*) that occurs in similar habitat indicate that fire has at least a short-term inhibitory effect on seed germination (Ellstrand 1994; Ogden 1999).

BACKGROUND ON EDGE EFFECTS

In the context of conservation biology and preserve design, edge effects are defined as adverse changes to natural communities as a result of their proximity to human-modified areas (Lovejoy et al. 1986; Yahner 1988; Sauvajot and Buechner 1993) or, more simply, the adverse effects of development on adjacent biological resources. Examples of edge effects include increases in invasive, weedy species, increased trampling and soil compaction from human recreation, or increases in nonnative animal species. Edge effects have been documented within specified distances of developed lands, although the impacts may be species- or resource-specific and tempered by a host of site-specific factors, including microtopography (McEvoy and Cox 1987; Andersen 1991), distribution and size of gaps (Bergelson et al. 1993), and intactness of the natural community (Sauvajot and Buechner 1993). A number of empirical studies have concluded that detrimental effects to biological resources can occur at distances ranging from 150 to 600 feet from the edge of the urban-wildland interface (e.g., Gates and Gysel 1978; Brittingham and Temple 1983; Andren et al. 1985; Wilcove 1985; Angelstam 1986; Wilcove et al. 1986; Temple 1987; Andren and Angelstam 1988; Santos and Telleria 1992; Alberts et al. 1993; Scott 1993; Vissman 1993). The majority of these studies focus on impacts to wildlife habitat. Few studies that we reviewed focus specifically on edge effects to plant species.

Buffer Considerations

Kelly and Rotenberry (1993) provide guidelines for effective buffers around urban reserves that are useful in recommending buffer widths and assessing potential edge effects on the San Fernando Valley spineflower resulting from the proposed preserve design. Kelly and Rotenberry (1993) note that the effective size of an ecological preserve is almost always smaller than the area within the preserve boundary, or the total preserve size. The effective size is generally referred to as the *core area*. The preserve boundary or *edge* surrounds the core area. The width of the edge is a function of the permeability of the boundary to negative external influences or risk factors. Edge effects can be particularly significant for small reserves because of their relatively large perimeter to core ratios (Soulé et al. 1988; Bolger et al. 1991; Saunders et al. 1991). An effective buffer width can be determined on a site-specific basis by (1) identifying risk factors and potential impacts to the species of concern within the preserve and (2) determining the permeability of the urban-wildland boundary to vectors of those risk factors. Altering the boundary permeability through habitat management is a potential method for mitigating identified impacts (Kelly and Rotenberry 1993). However, this method may not be effective for all types of risk factors (e.g., wind-blown seed of invasive plant species). Incorporating appropriate site design measures and land use restrictions into the development abutting the preserve is an alternative method of avoiding and minimizing impacts to the preserve (i.e., designating a land use buffer outside the preserve).

RISK FACTORS AND POTENTIAL IMPACTS

Preliminary studies on the biology and ecology of the San Fernando Valley spineflower (GLA 1999) indicate that the following parameters may play a role in the persistence of this taxon on the Ahmanson Ranch and may be negatively influenced at the urban-wildland interface:

- gaps in vegetation cover (i.e., areas of bare soil)
- low nutrient soils
- pollinators
- seed dispersal agents
- extant seed bank

Risk factors at the urban-wildland boundary that may affect these parameters include the following:

- nonnative, invasive plant and animal species
- vegetation clearing for fuel management or creation of trails
- trampling
- increased water supply due to suburban irrigation and runoff
- chemicals (e.g., herbicides, pesticides, fertilizers)
- increased fire frequency

Some of these risk factors could affect more than one of the parameters. Potential effects of these risk factors on the spineflower population are discussed below.

Invasive Plant Species

San Fernando Valley spineflower appears to prefer open patches of bare ground, which are often invaded by exotic plant species, as well (Amor and Stevens 1976; Forcella and Harvey 1983; Bazzaz 1986; Alberts et al. 1993). Although the spineflower on the Ahmanson Ranch was noted on thin, compacted soils lacking nonnative grasses, it is not clear whether spineflower density is significantly lower on these soils versus on deeper soils or whether nonnative grasses may be more abundant in these areas in years with average or above-average rainfall. Brooks (1995) noted that with increased rainfall, annual grasses gradually gain dominance once they have colonized an area, regardless of management or other protective measures. Gordon-Reedy (pers. obs.) has also observed large fluctuations in nonnative grass density in open coastal sage scrub in Riverside County in years with variable rainfall amounts.

Direct competition between native and exotic plant species is well documented (Alberts et al. 1993). Furthermore, the successful invasion of exotic species may alter habitats and lead to displacement or extinction of native species over time. For example, exotic invasions have been shown to alter hydrological and biochemical cycles and disrupt natural fire regimes (MacDonald et al. 1988; Usher 1988; Vitousek 1990; D'Antonio and Vitousek 1992; Alberts et al. 1993). Vitousek and Walker (1989) noted that aggressive nonnative species might displace native species by altering soil fertility.

MacDonald et al. (1988) reported that reserves surrounded by development areas supporting populations of exotic species are most subject to invasion. However, in studies on the effects of urban encroachment into natural areas in the Santa Monica Mountains, Sauvajot and Buechner (1993) found that direct habitat alteration or disturbance within natural areas is a more significant factor in the extension of edge effects into those areas than proximity to urban development alone. Several other studies have also correlated invasions by alien plants into nature reserves with elevated levels of disturbance, high light conditions, and, in some cases, increased water availability (McConnaughay and Bazzaz 1987; Laurance 1991; Tyser and Worley 1992; Brothers and Spingarn 1992; Matlack 1993).

In a review of biological invasions of 24 nature reserves, Usher (1988) reported a positive correlation between the number of human visitors and the number of introduced species. Further, he cited circumstantial evidence that invasive plant species are most common near paths through the reserves. Tyser and Worley (1992) provided data indicating that alien plant species extend up to about 325 feet into natural habitat from primary roads, secondary roads, and backcountry trails. They found a gradual decline in species richness with distance from the edge, and effects along trails were less prominent (but still evident) than along roads. Ghersa and Roush (1993) noted that the number of propagules available rarely limits the abundance of weeds in a given setting; rather, one needs to consider both the dispersal strategies of the invading species and potential vehicles for dispersal. Well-known dispersal agents include humans (Usher 1988; Ghersa and Roush 1993), vehicles, and road construction (Amor and Stevens 1976; Amor and Piggin 1977; Lonsdale and Lane 1991 in Hobbs and Humphries 1995; Hobbs and Humphries 1995). In addition to promotiug biological invasions by acting as dispersal vectors, humans can impact spineflower habitat by disturbing the soil surface, trampling individual plants, and increasing the fire frequency within or adjacent to reserves.

Factors that affect the success of invasions include dispersal ability of the invasive species, in conjunction with size and distribution of gaps in the vegetation (McConnaughay and Bazzaz 1987; Bergelson et al. 1993) and the timing of seed dispersal relative to environmental conditions or "invasion windows'" (Johnstone 1986). Bergelson et al. (1993) documented an *average* dispersal distance for the ruderal, wind-dispersed annual plant, *Senecio vulgaris*, of 1.1 feet; however, they also noted dispersal events for this same species of over 50 feet. McEvoy and Cox (1987) reported that 89% of seeds of another wind-dispersed species (*Senecio jacobaea*) traveled 16 feet or less, while no seeds were observed >45 feet from the source in a mark-recapture study. They noted, however, that secondary dispersal and animal dispersal may increase initial dispersal distances under some conditions. For example, in dry, open habitats, seeds may be moved along the ground or swept into the air by wind (McEvoy aud Cox 1987).

Laurance (1991), in a study of edge effects in tropical forest fragments, found a striking abundance of invasive plants within 650 feet of forest edges, and lower (but still elevated) levels of invasive plants 1,640 feet from the edges. Tyser and Worley (1992), in a study in the intermountain region of western North America, found invasive plants extending over 325 feet from road and trail edges, although there was a gradual decline in invasive species richness beyond about 80 feet. Amor and Stevens (1976) also found a general decline in invasive plants with increasing distances from roads into sclerophyll forests in Australia. They reported that at 100 feet from a road edge, the majority of invasive species either dropped out altogether or occurred in lower percentages than at the road shoulder, particularly in drier plant communities. In the presence of artificial sources of water, however, the occurrence of some invasive species remained high regardless of distance from the edge (Amor and Stevens 1976). Where there is a large perimeter between the preserve and urban interface, larger numbers of colonizing propagules can be expected to enter the preserve (Alberts et al. 1993). In general, Alberts et al. (1993) found that ruderals tend to invade reserves quickly, given appropriate site conditions, whereas ornamental species invade reserves over a longer period of time, and their presence is correlated with increased sources of water.

Invasive Animal Species

The effect of nonnative animal species on biological resources within reserves has been well documented (e.g., Gates and Gysel 1978; Brittingham and Temple 1983; Wilcove 1985; Andren and Angelstam 1988; Langen et al. 1991; Donovan et al. 1997); however, most of this literature pertains to effects on wildlife species. For example, both domestic dogs and cats are known to adversely impact native wildlife, with effects ranging from harassment to disturbance of breeding activities to predation (Kelly and Rotenberry 1993; Spencer and Goldsmith 1994). Domestic dogs have been observed within reserves at a distance of greater than 325 feet from the edge, while cats have been observed within reserves more than 1 mile from human dwellings in Riverside County (Kelly and Rotenberry 1993). An increase in nonnative predators as a result of development adjacent to the spineflower preserve could potentially affect populations of rodents (e.g., kangaroo rats, pocket mice, pocket gophers) that may act as seed dispersal agents or play a role in bioturbation.[3] In a study of two populations of house cats on a suburban-desert interface near Tucson, Arizona, Spencer and Goldsmith (1994) found that most prey were diurnal species of rodents, birds, and reptiles. Radio-tracking studies indicated that the cats spent over 90% of their time within 100 feet of houses, although this may have been related to an abundant coyote population. Spencer and Goldsmith (1994) suggested that impacts of cats on native wildlife are concentrated within 100-200 feet of the urban-wildland interface in the presence of predators (e.g., coyotes), but may extend further in their absence.

If rodents consume spineflower seeds, then a reduction in the rodent population may reduce seed dispersal into sites suitable for germination. Perry and Gonzalez-Andujar (1993) developed a model to assess the role of seed dispersal on metapopulation growth

[3] Bioturbation is the aeration and mixing of soil by organisms.

and persistence of an annual plant, like the spineflower, that forms a seed bank and occurs in drought-like and disturbed environmental conditions. This model predicts that a strongly dispersing metapopulation is hardly affected by temporal environmental heterogeneity, while metapopulations with moderate or no dispersal capabilities suffered extinction in every replication. However, granivorous rodents tend to selectively harvest large seeds (Brown and Lieberman 1973; Brown et al. 1979; Samson et al. 1992; Brown and Harney 1993). Spineflower seeds are relatively small (ca. 2 mm), and may only be used by smaller rodents (e.g., pocket mice) that clip clusters of involucres. Even if rodents do not play a significant role in spineflower seed dispersal through seed predation, they may still effect some localized dispersal when the awn-tipped involucres (and seeds) become temporarily attached to their bodies. In addition, rodents may indirectly benefit the spineflower by suppressing populations of larger-seeded annual plants that compete with the spineflower (Davidson et al. 1984; Samson et al. 1992; Brown and Harney 1993).

Decreases in the rodent population may also reduce the amount of potentially high quality habitat for spineflower establishment. Rodent activities that result in bioturbation and bare soil patches have been associated with spineflower plants on Ahmanson Ranch (GLA 1999). Long-term studies in the Southwest have demonstrated that selective removal of kangaroo rats, for example, resulted in much less disruption of the soil surface, higher densities of tall perennial and annual grasses, increased accumulation of litter, decreased foraging by granivorous birds, and differential colonization by rodents typical of grassland habitats (Brown and Heske 1990; Thompson et al. 1991; Brown and Harney 1993).

Conversely, Mills (1996) demonstrated that edges could have higher populations of certain mammalian seed predators (e.g., deer mice [*Peromyscus* spp.]) than core areas, which may result in reduced plant recruitment. Deer mice are good edge specialists, and can reach high densities under appropriate conditions. Because they are generalists that can switch among food resources, they often exert a heavier toll on a certain food resource (like seeds) than specialists whose populations track the specific resource more closely. Jules and Rathcke (1999) found reduced recruitment of a native herbaceous perennial plant species (*Trillium ovatum*) within about 200 feet of a forest/clearcut edge, and demonstrated that this was significantly correlated, in part, with seed predation by rodents (species unspecified). To date, no studies have been conducted that define the role of rodent populations (if any) in spineflower seed dispersal or predation. In light of these uncertainties, it therefore seems important to maintain as natural a mix of native seed dispersers/predators as possible, and to minimize ecological imbalances due to abundant nonnative species.

One invasive species that has been documented on the Ahmanson Ranch and may potentially increase in dominance over time is the Argentine ant. Ant surveys indicated that the Argentine ant is abundant in some areas of the project site, but currently occurs in very low numbers in or near spineflower habitat, presumably due to xeric conditions (Hovore pers. comm.).

Disturbed habitats are often considered vulnerable to Argentine ant invasions. There is evidence that this exotic species rapidly invades disturbed areas within stands of native habitat (Erickson 1971; Ducote 1977 in Suarez et al. 1998; Ward 1987; DeKock and Giliomee 1989; Knight and Rust 1990; Suarez et al. 1998). Suarez et al. (1998) found Argentine ants most abundant along the edge of urban preserve areas, with densities of ants in the preserve decreasing with distance from the edge. They found that ant activity was highest within about 325 feet of the nearest urban edge, whereas areas sampled beyond 650 feet contained few or no Argentine ants. However, Argentine ants have also been found at distances of approximately 1,300 feet and 3,280 feet from the edge, respectively, in other urban reserves in southern California (Suarez et al. 1998). DeKock and Giliomee (1989) documented extensive penetration of this species into natural areas in South Africa along roads. Recent studies indicate that the Argentine ant may be capable of invading undisturbed habitat, as well (Cole et al. 1992; Human and Gordon 1996).

Argentine ants appear to be confined to low elevation areas with permanent soil moisture (Erickson 1971; Tremper 1976 in Suarez et al. 1998; Ward 1987; Knight and Rust 1990; Holway 1995, 1998). Tremper (1976) reported that Argentine ants desiccate more easily and are less tolerant of high temperatures than native ants. Suarez et al. (1998) indicated that the presence of the Argentine ants in urban reserves might be dependent on water runoff from developed areas. Holway (1998) found that the rate of Argentine ant invasion is primarily dependent on abiotic conditions (e.g., soil moisture), rather than on disturbance. He suggested that disturbed areas are often a point of introduction, but encourage invasions only if they increase the availability of a limiting resource such as water. Blachly and Forschler (1996) found Argentine ants thriving in areas disturbed by human activity, but indicated that their presence is also related to added ground cover, permanent water supplies, and a simplified native ant fauna.

Although the reproductive strategy of the San Fernando Valley spineflower is not yet known, field studies indicate that flowers are visited by a number of invertebrate species. Presumably, one or more of these species function as effective pollinators of the spineflower. Invasive faunal species (e.g., Argentine ants, parasites) have the potential to negatively impact pollinator populations. Loss or limitation of pollinators may adversely affect the long-term survivability of the spineflower by reducing seed output (e.g., reproductive failure) if there is no selfing (Jennersten 1988; Bawa 1990) or decreasing the effective population size through reduced gene flow (Bawa 1990; Menges 1991; Aizen and Feinsinger 1994). Some studies have shown that pollinator limitation can reduce seed output by 50-60% (Jennersten 1988; Pavlik et al. 1993; Bond 1995). Jules and Rathcke (1999) demonstrated that pollinator limitation was significantly related to reduced recruitment of a native plant species within 200 feet of a forest/clearcut edge.

It has been hypothesized that native ants may be a primary or secondary pollinator of the San Fernando Valley spineflower (GLA 1999). The Argentine ant is known to displace native ant species (Erickson 1971; Tremper 1976 in Suarez et al. 1998; Ward 1987; Holway 1995; Human and Gordon 1996; Suarez et al. 1998), although this apparently has not yet occurred in spineflower habitat on the Ahmanson Ranch. Nonetheless, potential negative interactions between native ant species or other insect pollinators and the Argentine ant would be a concern if the spineflower were insect-pollinated.

Ant pollination is considered relatively uncommon in plants (Proctor et al. 1996), although Jones (pers. comm.) indicates that ants may be a major pollinator of cushion plants in desert areas and Hickman (1974) has demonstrated effective ant pollination in a taxon related to the spineflower. Ant-pollinated plants tend to occur in hot, dry habitats and are further characterized by a prostrate or low-growing habit, small, inconspicuous flowers close to the stem, intertwining plants within a population, few seeds per flower, and small pollen volume and nectar quantity (Hickman 1974). The San Fernando Valley spineflower possesses many of these characteristics. In a study conducted in the South African fynbos,[4] Paton (1986 in Visser et al. 1996) correlated high densities of ants (species undetermined) in inflorescences of *Protea eximia* with lower numbers of other insects. Visser et al. (1996) investigated whether Argentine ants influenced the number of insect species and individuals present in the inflorescences of *Protea nitida*, and found that 10 of 11 insect taxa showed reduced numbers where Argentine ants were present and, in 5 cases, these reductions were highly significant. In addition, the total number of insects was significantly suppressed in inflorescences with high numbers of Argentine ants. Visser et al. (1996) speculated that a reduction in the diversity and abundance of insect visitors could result in reduced pollination and ultimately affect the reproductive capacity of the plant. In the species they studied, ants were not considered effective pollinators, and an increase in ant abundance was not expected to promote pollination.

Ants may also function as primary or secondary dispersers of seeds (Roberts and Heithaus 1986; Louda 1989). They have been reported to contribute to the spatial heterogeneity of seed distribution (Reichman 1984, 1979) and they decrease seed abundance of some numerically dominant ruderal species in relation to less dominant native annual species (Inouye et al. 1980). Displacement of native ant species by the Argentine ant could negatively affect spineflower persistence by reducing spineflower seed number and distribution. Bond and Slingsby (1984) investigated the effects of displacement of native ant species by the Argentine ant on a myrmecochorous plant[5] in South Africa, and found that the Argentine ant negatively affected seed dispersal and plant regeneration. Native ant species typically carry seeds to their nests, where they remain or are later discarded in nearby middens. While the ants derive nutritional benefits from the seeds, this process also increases seedling recruitment by minimizing competition near the parental plant, reducing seed predation at the soil surface, and enhancing plant growth in the nutrient-enriched soils of the nests or middens (Marshall et al. 1979; Heithaus et al. 1980; O'Dowd and Hay 1980; Bond and Slingsby 1984). In contrast, Argentine ants are slower to discover seeds, move them a shorter distance, and fail to store them in below-ground nests, thus resulting in decreased dispersal and increased seed predation (Bond and Slingsby 1984; Holway 1999). Bond and Slingsby (1984) reported significant decreases in seed germination and establishment in areas infested with Argentine ants compared with uninfested areas, and ascribed these differences primarily to increased seed

[4] Fynbos is a chaparral-like vegetation community found in mediterranean climate regions of South Africa and Australia. It is dominated by evergreen shrubs with sclerophyllous (hard) leaves (Dallman 1998).

[5] A myrmecochorous plant is dependent on ants for seed dispersal.

predation. They further suggested that the negative effects of Argentine ants on myrmecochorous species with a persistent seed bank will only become apparent over relatively long time periods (e.g., decades) as the seed bank becomes depleted.

DeKock (1990) found that the first native ant species to be driven off by Argentine ants are those that are most effective in seed dispersal. She suggested that the effects of Argentine ant invasions on native plants would be indirect and related to a depleted seed bank. It should be noted that many ant-dispersed seeds have structural adaptations such as oily seed coats or fat-bearing appendages (elaiosomes) that provide nutritional rewards for the dispersing ants (Stebbins 1974; Marshall et al. 1979). Hughes and Westoby (1992) demonstrated that seed dispersal by ants was, in general, significantly higher for seeds with elaiosomes, although this effect was ant species-specific, and some dispersal did occur in the absence of these structures. It is not known whether spineflower seeds have any adaptations that would predispose them to ant-dispersal.

Vegetation Clearing

Disturbance of native vegetation communities can produce appropriate site conditions for germination of weedy species (Bazzaz 1986; Westman 1990; Alberts et al. 1993; Hobbs and Humphries 1995). In general, ruderal weedy species possess a number of characteristics that allow them to rapidly colonize gaps or bare areas. These inclnde the production of abundant, typically wind-dispersed seeds that are quick to germinate, establish, and grow (Frenkel 1970; Amor and Piggin 1977; Bazzaz 1986). Thus, weedy exotics often out-compete native species that utilize similar habitats. Clearing of vegetation along the urban-wildland interface (e.g., firebreaks, roads) or within a preserve system (roads, trails) may provide opportunities for such weedy species to gain a foothold in the preserve (Amor and Stevens 1976; Amor and Piggin 1977; Lonsdale and Lane 1991 in Hobbs and Humphries 1995).

Trampling

Trampling can affect the spineflower either by damaging individual plants or altering the ecosystem. Maschinski et al. (1997) demonstrated that the combination of trampling and poor climatic conditions resulted in an accelerated extinction probability for a native plant species. In this case, trampling directly affected plant fitness, resulting in significantly lower fruit production. Trampling can also create gaps in vegetation that provide opportunities for exotic plant establishment (Hobbs and Huenneke 1992). Cole (1987) reported that even low levels of trampling caused a substantial loss of vegetation cover aud species diversity, and resulted in an increase in soil compaction, whereas soil erosion occurred with higher levels of trampling. In other studies (see Dale and Weaver 1974; Bright 1986), species diversity increased in areas subject to trampling, but species composition shifted to those plants that are resistant to trampling. In general, plants with tough, wiry leaves or thick leaves and a tufted growth form (e.g., grasses) are more resistant to trampling than herbaceous plants, such as the spineflower, whose branches or stems could be easily crushed or broken (Cole 1987; Hall and Kuss 1989). Refer to the literature cited above (plant invasions, vegetation clearing) for discussions on invasion of gaps or vegetation disturbances by weedy versus native species.

Harrison (1981) found that the season or timing of trampling influences the effects on native species and their recovery. The ability to recover from trampling is also dependent on environmental conditions (temperature, moisture) and growth form characteristics (Cole 1987). Some adverse effects of trampling (soil compaction, erosion) are less easily reversed than others. For these factors, recovery may be difficult after only a few years of trampling at relatively high intensities (Cole 1987).

Increased Water Supply

Changes in surface and subsurface hydrological conditions at or near the urban-wildland boundary could occur as a result of removal of native vegetation, increased runoff from roads or other paved surfaces, and residential or commercial irrigation. Increased surface water flows may result in increased erosion and transport of particulate matter (Saunders et al. 1991). Altered patterns of erosion may deposit new substrates for plant colonization, although such areas are often quickly colonized by weedy species that require both disturbance and nutrient-rich substrates for establishment (Hobbs and Atkins 1988). Increased surface flows may also be a conduit for introducing invasive species into the preserve. Holway (1998) indicated that Argentine ant colonies are often dispersed into new areas by jump-dispersal events such as floods, and that these types of dispersal events are an important component of the large-scale dynamics of Argentine ant invasions.

Increased surface moisture or underground seepage that results in increased soil moisture levels may also promote the establishment of exotic plant species (Alberts et al. 1993; McIntyre and Lavorel 1994; Amor and Stevens 1976) or wetland-dependent native plant species, facilitate invasion by Argentine ants (Suarez et al. 1998), alter seed bank characteristics, and modify habitat for ground-dwelling fauna (Saunders et al. 1991). Seepage is expected to be minimal in most areas along the urban-wildland interface due to the underlying substrate. However, the current project design includes a few hundred feet of man-made slopes between two stands of the spineflower, and there is the potential for some seepage on these fill soils (Barker pers. comm.).

Chemicals (Herbicides, Insecticides, Fertilizers)

Chemical pollutants can adversely affect biological resource areas in many ways, including decreases in pollinators, increases in weedy exotic species, or damage to or direct killing of native plants. The use of herbicides to maintain open areas within or adjacent to the preserve can result in chemical habitat fragmentation and consequent reductions in pollinator populations (Buchmann and Nabhan 1996). Insecticide spraying in adjacent residential areas can result in pollution drift that kills pollinators in reserve areas (Kelly and Rotenberry 1993; Allen-Wardell et al. 1998). Boutin and Jobin (1998) reported that chemical pesticide drift using ground equipment has been estimated at 1-10% of the application rate within about 30 feet of the target. In a study on the effects of various herbicides on native plant species in a nature reserve, Marrs et al. (1989) demonstrated that the maximum safe distance (i.e., no lethal effects) was about 20 feet from the spray source, although the average safe distance was 6.5 feet or less. They also found that adverse but non-lethal effects of spraying (e.g., plant damage, flower suppression) occurred at slightly greater distances than lethal effects, and showed seasonal variability. For example, no damage was detected beyond about 8 feet for most of the species they tested in fall. A few species, however, appeared to be particularly sensitive to herbicides during this time period, and showed damage between 33 and 65 feet from the spray source. In spring, the maximum distance at which damage effects were apparent was about 25 feet from the spray source. However, most damaged plants recovered completely by the end of the growing season. Based on these results, Marrs et al. (1989) advocated the use of a 16 to 33-foot buffer zone to minimize lethal effects to herbaceous plants from herbicide drift, and noted that wider buffers (e.g., 50 feet) would reduce risks even further.

Other chemicals, such as are included in fertilizers, may enhance growth of weedy species and, thus, should not be used adjacent to the preserve. For example, nitrogen is a limiting factor in plant growth, and the addition of nitrogen fertilizers enhances the growth of many plant species. Many native plant species, however, are adapted to low-nitrogen systems (Vitousek et al. 1997; Zink and Allen 1998). Vitousek et al. (1997) stated that the addition of nitrogen to such systems, through direct fertilization or runoff from adjacent areas, could cause shifts in species dominance and reduce overall species diversity. Furthermore, nitrogen-rich systems may promote exotic weedy species to the detriment of native species (Zink and Allen 1998).

Aerial fallout of nitrogenous compounds from automobiles may also contribute to increased nitrogen in the soil. Allen (1996) has observed high mortality of coastal sage scrub shrubs in areas with high soil nitrogen levels, and hypothesizes that nitrogen deposition from air pollution may be responsible for this mortality (Allen et al. 1996). Vegetation and soils are known to be important sinks for other atmospheric pollutants from automobiles, as well, although a number of biological and environmental factors may affect the actual absorption or accumulation of such compounds. The level of pollutants in roadside plants has been positively correlated with traffic density. Singh et al. (1995) reported the most significant effects where traffic volume was high (e.g., >4,000 vehicles per 2 hours).

Increased Fire Frequency

The effects of fire on the San Fernando Valley spineflower are not yet known. Seed germination of a closely related taxon, Parry's spineflower (*Chorizanthe parryi* var. *parryi*) appears to be inhibited by fire in both greenhouse and natural settings (Ellstrand 1994; Ogden 1999). Despite the inhibitory effect of direct scorching, fire may also prove beneficial to the spineflower by creating openings and temporarily reducing competition.

San Fernando Valley spineflower occurs primarily in openings in coastal sage scrub, although much of its habitat on the Ahmanson Ranch appears to have been invaded by nonnative grasses. The coastal sage scrub community is adapted to fire, but not completely dependent on it for continued viability. In general, it is considered a relatively stable vegetation community over a broad range of fire frequencies, particularly if detrimental factors such as fragmentation and exotic weed species invasions are minimized. However, excessively long or short fire intervals may result in (1) shifts in the composition of the dominant species of this community (Westman 1987, 1981; Keeley 1991) or (2) displacement of native species by nonnative species, such as annual grasses. Nonnative grasses exert a number of undesirable effects on native plant communities, including altering fire regimes. Colonization of an area by nonnative grasses provides the fine fuel needed to start and maintain fires. This can lead to increased fire frequency, extent, and intensity. Nonnative grasses typically recover more quickly than native species following grass-fueled fires, thereby initiating a cycle of increasing fire susceptibility (D'Antonio and Vitousek 1992; Hobbs and Huenneke 1992). Changes in fire regimes due to invasive species can result in a wide range of ecosystem changes, including nutrient loss, altered local microclimate, and prevention of succession (D'Antonio and Vitousek 1992).

The use of fire has been suggested as one method for controlling nonnative grasses. Controlled burns have been used with some success to control nonnative grasses, particularly in grassland communities (Zavon 1982 in Pollack and Kan 1998; Ahmed 1983 in Pollack and Kan 1998; Keeley 1990; George et al. 1992; Pollack and Kan 1998). Pollack and Kan (1998) and others (see Menke 1992) found that late-spring fires were an effective method of controlling annual species that do not have well-developed seed banks, or of reducing the size of the seed bank in those alien species that do form a seed bank. Pollack and Kan (1998) suggested that knowledge of the target species' phenology is critical in effective timing of burns. In their study, late-spring burns were associated with more intense fire behavior and the need for fire suppression equipment (Pollack and Kan 1998). Controlled or prescribed burns are often suggested as a management tool to improve habitat characteristics, and a recent report of the Wildland/Urban Interface Task Force (1994) included a wildland fire management-planning model designed to facilitate prescribed burning and post-fire management. However, recent attempts to incorporate burns (or even "let-burn" policies) into habitat management plans in southern California have met with resistance from local fire control agencies, particularly near urban areas.

In addition to the fire-inducing effects of nonnative grasses, fire frequency near urban-wildland boundaries may increase due to other human-related activities (e.g., construction or utility maintenance activities, children playing with matches).

ANALYSIS OF RISK FACTORS

The objectives of this analysis are to (1) determine how risk factors can be reduced through buffers and management actions; (2) provide a relative ranking of risk factors that pose the greatest threat to spineflower persistence, based on boundary permeability; and (3) recommend buffer/management scenarios that effectively address risk factors. This analysis utilizes a step-wise approach by first considering buffer widths alone as a means of reducing risk factors, then overlaying buffers with proposed management actions[6] to reduce potential negative effects from risk factors. Ranking of risk factors is based on the literature review, field observations, and professional judgment. Risk factors that can be least controlled by management are considered to present the highest risk to spineflower persistence.

Buffer Widths

Buffers are an important component of preserve design. Here, the buffer is defined as the distance between the edge of the current spineflower population within the preserve and the edge of the preserve. Various buffer widths were assessed to determine their effectiveness in minimizing identified risk factors. The five buffer widths included in this analysis range from a minimum width (15 feet) to greater widths shown to be effective in the edge effect literature for specific risk factors. Table 1 presents the relative assessmeut of varying buffer widths in minimizing risk factors.

Table 1
ESTIMATED BUFFER EFFECTIVENESS FOR MINIMIZING EDGE EFFECTS OF SELECTED RISK FACTORS ON THE SPINEFLOWER

RISK FACTORS	BUFFER WIDTHS (FEET) [1]				
	15	30-50	80-100	200	300
Invasive Animals	L	L	L	M	M
Increased Fire Frequency	L	L	L	M	M
Invasive Plants	L	L	M	H	H
Vegetation Clearing	L	L	M	H	H
Increased Water Supply	L	L	M	H	H
Trampling	L	L	M	H	H
Chemicals	L	M	H	H	H

[1] Estimated effectiveness rankings: Low (L) = Unlikely to be effective; Moderate (M) = moderately effective; High (H) = highly likely to be effective.

Table 1 indicates that ranking of risk factors (i.e., from highest risk to the spineflower to lowest risk), based on buffer widths, can be grouped as follows:

- **Invasive Animals and Increased Fire Frequency** -- Literature on invasive animals indicates that most impacts that could affect the spineflower are concentrated within about 100-325 feet of the edge. Nonetheless, both cats and dogs have the ability to disperse much further into preserve areas. Argentine ants also have the ability to disperse further into preserve areas, but apparently only in

[6] For the purpose of this analysis, other preserve design elements, land use restrictions, and engineering designs are included under management actions.

the presence of adequate water supplies. Buffer width alone is not expected to be highly effective in reducing fire frequency.

- **Invasive Plants, Vegetation Clearing, Increased Water Supply, and Trampling** --Invasive plant species and vegetation clearing are closely related risk factors. Literature reviewed on invasive plants in temperate systems indicates that they may extend up to 325 feet into preserve areas, with a gradual decline in invasive species beyond about 80-100 feet. Further, the effectiveness of invasions is related to suitable substrates (e.g., gaps or disturbances, which may be created by vegetation clearing) and dispersal ability of the invasive species, among other factors.

  Surface runoff on the project site will be controlled through engineering designs. There is the potential for underground seepage, however, which may have a zone of influence that extends up to about 200 feet, depending on the substrate. The effects of trampling are primarily direct and limited to the area of impact, although associated trespass by humans can be an effective means of introducing nonnative species into the preserve.

- **Chemicals** -- Literature indicates that the majority of pesticide drift from chemicals will extend less than 35 feet from the source. Although the effects of fertilizers are typically localized, these compounds may be more widely dispersed through surface runoff or seepage. Atmospheric pollutants from cars can adversely affect plants, particularly where traffic density is very high; however, this may not be a factor in a residential development.

Management Actions

Management actions are expected to have varying degrees of effectiveness in reducing negative effects of identified spineflower risk factors. For example, the project proposes to control alterations in surface and subsurface hydrology through engineering designs. Restrictions on landscaping palettes, irrigation, and habitat disturbance adjacent to the preserve will reduce the potential for ornamental, invasive species in the preserve by limiting both the source material and appropriate site conditions for colonization. However, these restrictions do not address nonnative, weedy species that are already present in the area, and which have also been identified as major risk factors to spineflower persistence.

Table 2 overlays various management measures and buffer widths for each risk factor to assess their combined effectiveness in controlling edge effects. This analysis considers a wide range of management measures, not just those considered to be the most effective in controlling edge effects. These recommendations may not be comprehensive, and their effectiveness can only be roughly estimated at this time, based on the known biology of the species and conditions on the Ahmanson Ranch. Ranking of these measures also does not consider implementation or enforcement feasibility for each measure.

Table 2
ESTIMATED MANAGEMENT AND BUFFER EFFECTIVENESS FOR REDUCING EDGE EFFECTS

RISK FACTORS/MANAGEMENT MEASURES	BUFFER WIDTHS (FEET)[1]				
	15	30-50	80-100	200	300
Invasive Animals					
• No Specific Management Measures[2]	L	L	L	M	M
• Restrict landscaping palettes adjacent to the preserve to exclude use of invasive exotic species	L	L	M	H	H
• Restrict irrigation in and adjacent to the preserve	L	L	M	H	H
• Maintain current surface and subsurface hydrological conditions within the preserve through engineering design of adjacent areas	M	M	M	M	H
• Utilize french drains to minimize seepage on fill slopes, as determined necessary	H	H	H	H	H
• Inspect plants used in revegetation efforts in or adjacent to the preserve for pest species (e.g., Argentine ants)	L	L	M	H	H
• Avoid use of barriers (e.g., walls) with subsurface footings within or adjacent to the preserve	H	H	H	H	H
• Implement a bait control program for Argentine ants, as determined necessary through monitoring	L	L	M	M	M
• Bell cats in residential areas adjacent to the preserve and educate homeowners on the danger of coyotes to free-roaming cats	L	L	M	M	M
• Maintain habitat connectivity between preserve areas to encourage native predators in the preserve (thereby reducing populations of nonnative predators) and allow for recolonization of edge areas by native mammals	L	M	M	H	H
• Minimize internal fragmentation (e.g., roads, trails) and close unnecessary existing dirt roads	M	H	H	H	H
• Construct barriers to exclude nonnative animals (e.g., dogs)	M	M	M	M	M
Increased Fire Frequency					
• No Specific Management Measures[2]	L	L	L	M	M
• Implement a weed control program to reduce fine fuel capacity in fire-susceptible habitats	L	L	M	M	M

Table 2 (continued)
ESTIMATED MANAGEMENT AND BUFFER EFFECTIVENESS FOR REDUCING EDGE EFFECTS

RISK FACTORS/MANAGEMENT MEASURES	BUFFER WIDTHS (FEET)[1]				
	15	30-50	80-100	200	300
Increased Fire Frequency (continued)					
• Implement prescribed burning if shown to be advantageous to spineflower persistence and if allowed within the preserve by fire control agencies	M	M	H	H	H
• Restrict the use of construction or utility maintenance equipment in or adjacent to the preserve to avoid or minimize potential fires due to sparking (e.g., metal blades from bulldozers or other construction equipment striking rocks) or downed electrical lines	M	M	M	M	M
Invasive Plants					
• No Specific Management Measures[2]	L	L	M	H	H
• Restrict landscaping palettes adjacent to the preserve to exclude use on invasive exotic species	L	L	M	H	H
• Restrict irrigation adjacent to the preserve	L	L	M	H	H
• Maintain fuel breaks outside preserve boundary	L	L/M	M	H	H
• Minimize or prohibit vegetation clearing within the preserve (e.g., roads, trails)	H	H	H	H	H
• Restrict vegetation clearing immediately adjacent to the preserve	L	L	M	H	H
• Restore cleared areas with native species as soon as possible, subject to other conservation objectives	M	M	H	H	H
• Maintain current surface and subsurface hydrological conditions within the preserve through engineering design of adjacent developed areas	M	M	M	H	H
• Utilize french drains to minimize seepage on fill slopes, as determined necessary	H	H	H	H	H
• Control invasive weeds within the preserve and adjacent to the preserve (most appropriate method[s] to be determined)	L	L	M	H	H
• Reduce potential for invasion by weedy species by restoring selected disturbed areas within the preserve and adjacent to the urban boundary to reduce disturbance gaps	M	M	H	H	H

Table 2 (continued)
ESTIMATED MANAGEMENT AND BUFFER EFFECTIVENESS FOR REDUCING EDGE EFFECTS

RISK FACTORS/MANAGEMENT MEASURES	BUFFER WIDTHS (FEET) [1]				
	15	30-50	80-100	200	300
Invasive Plants (continued)					
• Reduce potential for invasion by weedy species by selecting sites for habitat enhancement or species reintroduction that minimize the potential for weed invasion	M	M	M	H	H
Vegetation Clearing					
• No Specific Management Measures[2]	L	L	M	H	H
• Site fire or fuel breaks outside preserve boundaries	L	L	M	H	H
• Minimize or prohibit vegetation clearing within the preserve (e.g., roads, trails)	H	H	H	H	H
• Restore cleared areas with native species as soon as possible, subject to other conservation objectives	M	M	H	H	H
Increased Water Supply					
• No Specific Management Measures[2]	L	L	M	H	H
• Maintain current surface and subsurface hydrological conditions within the preserve through engineering design of adjacent developed areas	M	M	M	H	H
• Utilize french drains to minimize seepage on fill slopes, as determined necessary	H	H	H	H	H
• Divert runoff from roads away from the preserve	M	M	M	H	H
• Restrict irrigation adjacent to the preserve	L	L	M	H	H
Trampling					
• No Specific Management Measures[2]	L	L	M	H	H
• Construct solid barriers to exclude or restrict pedestrian traffic	H	H	H	H	H
• Prohibit motorized vehicles, bicycles, and equestrian uses within the preserve	H	H	H	H	H
• Eliminate or reroute trails through the preserve to avoid sensitive biological resources	M	M	H	H	H
• Erect signs denoting boundary of the preserve and permitted uses	M	M	H	H	H
• Initiate an educational program (kiosks, information brochures, school programs, docent program)	M	M	H	H	H

Table 2 (continued)
ESTIMATED MANAGEMENT AND BUFFER EFFECTIVENESS FOR REDUCING EDGE EFFECTS

RISK FACTORS/MANAGEMENT MEASURES	BUFFER WIDTHS (FEET) [1]				
	15	30-50	80-100	200	300
Chemicals					
• No Specific Management Measures[2]	L	M	H	H	H
• Restrict use of herbicides within the preserve, and avoid use of pesticides within and adjacent to the preserve; herbicides must have no toxic effects on invertebrates	M	H	H	H	H
• Avoid use of herbicides and pesticides under conditions that would promote pollution drift (e.g., windy conditions)	L	M	H	H	H
• Avoid use of fertilizers within and adjacent to the preserve	M	M	H	H	H

1 Estimated effectiveness rankings: Low (L) – Unlikely to be effective; Moderate (M) = moderately effective; High (H) = highly likely to be effective.

2 Rankings indicate buffer effectiveness only (see Table 1), and are provided for comparison purposes.

Depending on buffer width and proposed land uses adjacent to the preserve, many of the recommended land use restrictions will require cooperation from homeowners. In addition, management measures in Table 2 are not weighted. It may be that some measures ranked as low are highly effective when combined with other measures. Conversely, some measures ranked high may be less important in minimizing risk factors than other measures with lower rankings (e.g., inspecting plants used in revegetation efforts versus restricting irrigation adjacent to the preserve). In some cases, there may be conflicts between various management measures. For example, a solid barrier would be highly effective in restricting human access and associated trampling effects. However, if the barrier includes subsurface footings, it may encourage nesting of Argentine ants. Some of the measures presented below may conflict with other objectives of spineflower protection, as well (e.g., habitat restoration). It is presumed that these measures will be refined during development of a detailed conservation strategy and management program for the spineflower. Finally, rankings in Table 2 consider individual effects only, and do not address the potential benefits of cumulative management measures. Combinations of certain management actions may have an enhanced capacity to address certain risk factors, as discussed in a later section of this document.

Table 2 indicates that individual management measures do, in fact, vary in their effectiveness for a specific risk factor. This makes it difficult to easily discern which buffer width would be expected to reduce a given risk factor to an adequate or acceptable level. Using a lowest common denominator approach (i.e., grouping risk factors according to the *least* effective management measure) results in the following ranking of risk factors, based on both management actions and buffer widths:

- **Invasive Animals and Increased Fire Frequency** -- Based on this analysis, invasive animals and fire frequency are considered the highest risk factors to the spineflower because they require the largest buffer width (>300 feet) in order for *all* management measures to be highly effective. Management measures for both risk factors are considered moderately effective at 80-100 feet.

- **Invasive Plants, Vegetation Clearing, and Increased Water Supply** -- Management measures for these three factors are all considered moderately effective at a buffer width of 80-100 feet and highly effective at widths of 200 feet or greater. Because control of these factors can presumably be achieved at narrower buffer widths than the factors above, they are given a lower ranking in terms of risk to the spineflower than either invasive animals or fire frequency.

- **Chemicals and Trampling** -- All management measures for these risk factors are considered moderately effective at buffer widths of 30-50 feet and highly effective at buffer widths of 80 feet or greater. Therefore, these factors are given the lowest ranking in terms of risk to the spineflower, assuming management measures are implemented.

DISCUSSION

The analyses above assume that (1) risk factors are equivalent in their potential detrimental effects on spineflower persistence and (2) management measures are equally effective in ameliorating edge effects to the spineflower. Neither of these assumptions is likely to be valid, although the information needed to verify this is not available. Ranking of risk factors as a result of the combined effect of buffer width and management actions focused on individual management measures, and did not consider the interaction between different measures. For example, different levels of effectiveness may be achieved when management measures are combined. Even though some measures may be ranked low in effectiveness, they could increase in value when combined with other measures. For this reason, measures with low rankings are generally still considered important. Some management measures may not be as effective as others. They could override the positive effects of more effective measures or at least result in situations where management measures are effective for one component of a risk factor and less effective for others. Finally, it should be noted that there is no descriptive model for the spineflower or related taxa to demonstrate how this species may respond to either the risk factors or management measures. Risk factors are discussed below with respect to expected management effectiveness as a result of either management measure interactions or shortcomings.

1. *Invasive Animals.* Eleven management actions have been recommended to reduce edge effects due to invasive animal species. Invasive animals have a high potential to adversely affect the spineflower, although no such effects have yet been documented.

Of particular concern are (a) changes in soil moisture conditions that could alter habitat for rodents (potential seed dispersers) or encourage invasion of spineflower habitat by Argentine ants; (b) introduction of nonnative animal species (e.g., Argentine ants) on plant materials or along roads; and (c) habitat fragmentation that could lead to reduced levels of native predators (e.g., coyotes) and concomitant increases in nonnative predators (e.g., cats) that could affect rodent populations. Controlling irrigation and maintaining habitat connectivity between the spineflower preserve and other open space areas in order to encourage native predators in the preserve will be key issues in management effectiveness for this risk factor. Despite the potential seriousness of invasive animals on spineflower persistence, it appears that management measures are available to control the most detrimental aspects of animal invasions, given adequate buffer widths and appropriate preserve design.

2. *Increased Fire Frequency*. None of the buffer widths considered in this analysis would be effective in stopping the spread of fire into the preserve from adjacent areas, but three management measures have been recommended to reduce the frequency and intensity of fires within the preserve. At this time, the effect of fire on the spineflower is not known. It can be assumed, however, that frequent or intense fires would be detrimental to individual spineflowers and spineflower habitat. Changes in natural fire cycles are related, in part, to the presence of fine fuels (especially nonnative grasses) within the preserve. While complete removal of grasses within the preserve is highly unlikely, a weed control program can potentially reduce nonnative grass cover and inhibit the spread of grasses into currently unoccupied areas of the preserve. Despite weed control measures within the preserve, reinvasions may occur from sources outside the preserve, and the probability of such reinvasions increases with narrow buffer widths (<80 feet).

3. *Invasive Plants*. Eleven management actions have been recommended to reduce edge effects due to invasive plant species. While some of these measures were ranked as having low effectiveness at narrow buffer widths, they are still important in reducing overall invasiveness, particularly in combination with other measures. For example, restrictions on landscaping and irrigation adjacent to the preserve, in conjunction with revegetation of disturbed areas, are expected to reduce opportunities for invasion of nonnative ornamental plant species. The same combination of measures is not expected to be as effective in reducing either the invasion or increasing dominance of nonnative weedy species already present in the area. Field studies have indicated that competition with these weedy species may already play a major role in limiting spineflower distribution. Because of the uncertainty of controlling additional weed invasions into the preserve, invasive plants may pose the highest risk factor to the spineflower.

4. *Vegetation Clearing*. Three management actions have been recommended to reduce edge effects from this risk factor, and two of these are expected to be moderately to highly effective even at relatively narrow buffer widths. Vegetation clearing is of concern because it provides gaps that facilitate invasions by nonnative plant species. This risk factor is considered relatively high because of its relationship to invasive

plants and the uncertainty of controlling this factor outside the preserve. For example, vegetation clearing will occur adjacent to the preserve during the development process, and may be a long-term condition, depending on fuel break requirements. While weed control will likely occur within the preserve, there is a lesser chance of effective controls outside the preserve; thus, cleared areas outside the preserve may provide a constant source of propagules (seeds) for invasions into the preserve. At narrower buffer widths (<80 feet), the potential for dispersal of invasive species into the preserve is relatively high.

5. *Increased Water Supply.* This risk factor plays a key role in the success of nonnative plant and animal species invasions. Control of surface and soil moisture alone may be adequate to reduce invasions of nonnative ornamental plant species and the Argentine ant into the spineflower preserve. The ranking of this risk factor assumes that all recommended management measures (including irrigation restrictions) would be implemented.

6. *Chemicals.* As with vegetation clearing, the greatest uncertainty in controlling this risk factor is expected to be the use of chemicals adjacent to the preserve. Edge effects from chemicals do not appear to have as wide a zone of influence as other risk factors, as evidenced by a high level of management/buffer effectiveness at 80-100 feet, and at least moderate levels at 30-50 feet. The effects of chemicals on the spineflower are not known; however, they may affect both vegetation and pollinator populations. Any application of herbicides within the preserve (e.g., for weed control purposes) should be experimental in nature to determine the effects on both vegetation and pollinator populations. Placement of heavily traveled roads adjacent to the preserve should be evaluated relative to contribution to increased nitrogen levels in the soil or atmospheric pollutants that could be detrimental to native plant species or enhance growth of weedy species.

7. *Trampling.* Trampling has the potential to directly damage spineflower plants, resulting in lowered reproductive success. Other potential trampling effects include the loss of vegetation cover and species diversity, and an increase in soil compaction or erosion. Some of these potential effects (loss of vegetation cover, soil compaction) might appear beneficial to the spineflower. However, they may also promote invasion of spineflower habitat by trampling-resistant plant species that may outcompete the spineflower and further alter site conditions. There is a high potential for effective control of this risk factor, however, with all recommended management measures having a moderate or high effectiveness at a buffer width of 30-50 feet. This effectiveness ranking assumes a solid barrier to inhibit trespass into the preserve. The use of subsurface footings for such a barrier should be discouraged, however, since they may provide suitable nesting habitat for Argentine ants.

CONCLUSIONS

In designing and managing effective buffers for preserves, it is useful to consider both potential risk factors to biological resources from urban areas and the permeability of the urban-wildland boundary to those factors (Stamps et al. 1987; Kelly and Rotenberry 1993). The analysis and discussion above focused on (1) identifying potential risk factors and the ways they may negatively influence the spineflower population, (2) assessing the permeability of the boundary to those risk factors, and (3) identifying methods of changing or managing the boundary permeability to reduce potential impacts. In cases where boundary permeability cannot be managed effectively, an increased setback or buffer between sensitive biological resources and the development boundary, coupled with intensive management efforts and land use restrictions near the preserve, may be required to conserve the spineflower population.

Table 3 summarizes the overall effectiveness of management measures for each risk factor (based on the lowest common denominator) at each buffer width. Ranking of risk factors in Table 3 reflects the increased effectiveness in controlling risk factors when all management measures are combined for a given factor. For example, it appears that management measures, if implemented, may be more effective in controlling invasive animals than invasive plants.

Table 3
SUMMARY OF COMBINED BUFFER WIDTH AND MANAGEMENT EFFECTIVENESS[1] FOR REDUCING RISK FACTORS FOR THE SPINEFLOWER ON THE AHMANSON RANCH PROJECT

RISK FACTORS[2]	BUFFER WIDTHS (FEET)[3]				
	15	30-50	80-100	200	300
Invasive Plants	L	L	M	H	H
Vegetation Clearing	L	L	M	H	H
Increased Fire Frequency	L	L	M	M	M
Invasive Animals	L	L	M	M	M
Increased Water Supply	L	L	M	H	H
Chemicals	L	M	H	H	H
Trampling	M	M	H	H	H

1 Effectiveness rankings in Table 3 reflect the lowest common denominator for each risk factor, or the least effective management measure.
2 Risk factors are listed according to the level of threat they present to the spineflower (i.e., highest threat to lowest threat), assuming all management measures in Table 2 are implemented.
3 Estimated effectiveness rankings: Low (L) = Unlikely to be effective; Moderate (M) = moderately effective; High (H) = highly likely to be effective.

Based on this analysis, it is estimated that a buffer width of 15 feet, in combination with specific management measnres, would be moderately effectively in controlling 1 risk factor (trampling) and unlikely to be effective in controlling the remaining 6 factors. A buffer width of 30-50 feet, in combination with management, would be moderately effective in controlling 2 risk factors (trampling and chemicals) and unlikely to control 5 factors. A buffer width of 80-100 feet, in combination with management measures, would be moderately effective in reducing the 5 greatest risk factors to the spineflower and highly effective in reducing the remaining risk factors. There appear to be no detectable differences in buffer effectiveness between 200 and 300 feet based on the literature reviewed. At both distances, management measnres would be highly effective for 5 risk factors and moderately effective for the remaining 2 risk factors. Selection of an appropriate buffer/management package should focus on achieving an acceptable level of effectiveness in reducing the highest risk factors.

LITERATURE CITED

Ahmed, E.O. 1983. Fire ecology of *Stipa pulchra* in California annual grassland. Ph.D. dissertation, University of California, Davis, CA. 71 pp.

Aizen, M.A. and P. Feinsinger. 1994. Habitat fragmentation, pollination, and plant reproduction in a Chaco Dry Forest, Argentina. Ecology 75:330-351.

Alberts, A.C., A.D. Richman, D. Tran, R. Sauvajot, C. McCalvin, and D.T. Bolger. 1993. Effects of habitat fragmentation on native and exotic plants in southern California coastal scrub. Pages 103-110 *in* Proceedings of the symposium: interface between ecology and land development in California, Keeley, J.E., editor. May 1-2, 1992; Occidental College, Los Angeles.

Allen, E.B. 1996. Vegetation change, nitrogen deposition, and restoration of coastal sage scrub. Abstract from Research and land management conference: exploring ways to apply biological research to the management of southern California's coastal sage scrub ecosystems. September 12-13, 1996, San Diego Zoo.

Allen, E.B., P.E. Padgett, A. Bytnerowicz, and R.A. Minnich. 1996. Nitrogen deposition effects on coastal sage vegetation of southern California. *In* Proceedings Air pollution and climate change effects on forest ecosystems, Bytnerowicz, A, editor. February 5-9, 1996, Riverside, CA.

Allen-Wardell, G., P. Bernhardt, R. Bitner, A. Burquez, S. Buchmann, J. Cane, P.A. Cox, V. Dalton, P. Feinsinger, M. Ingram, D. Inouye, C.E. Jones, K. Kennedy, P. Kevan, H. Koopowitz, R. Medellin, S. Medellin-Morales, G.P. Nabhan, B. Pavlik, V. Tepedino, P. Torchio, and S. Walker. 1998. The potential consequences of pollinator declines on the conservation of biodiversity and stability of food crop yields. Conservation Biology 12(1):8-17.

Amor, R.L. and C.M. Piggin. 1977. Factors influencing the establishment and success of exotic plants in Australia. Ecological Society of Australia 10:15-26.

Amor, R.L. and P.L. Stevens. 1976. Spread of weeds from a roadside into sclerophyll forests at Dartmouth, Australia. Weed Research 16:111-118.

Andersen, M. 1991. Mechanistic models for the seed shadows of wind-dispersed plants. American Naturalist 137:476-497.

Andren, H. and P. Angelstam. 1988. Elevated predation rates as an edge effect in habitat islands: experimental evidence. Ecology 69:544-547.

Andren, H., P. Angelstam, E. Lindstrom, and P. Widen. 1985. Differences in predation pressure in relation to habitat fragmentation: an experiment. Oikos 45:273-277.

Angelstam, P. 1986. Predation in ground-nesting birds' nests in relation to predator densities and habitat edge. Oikos 47:367-373.

Barker, R. 1999. Engineer, Psomas. Personal communication with P. Gordon-Reedy. November 16.

Barrett, C.H. and J.R. Kohn. 1991. Genetic and evolutionary consequences of small population size in plants: implications for conservation. Pages 3-30 *in* Genetics and conservation of rare plants, Falk, D.A. and K.E. Holsinger, editors. New York, NY: Oxford University Press.

Bawa, K.S. 1990. Plant-pollinator interactions in tropical rain forests. Annual Review of Ecology and Systematics 21:399-422.

Bazzaz, F.A. 1986. Life history of colonizing plants: some demographic, genetic, and physiological features. Pages 96-110 *in* Ecology of biological invasions of North America and Hawaii, Mooney, H.A. and J.A. Drake, editors. New York, NY: Springer-Verlag.

Bergelson, J., J.A. Newman, and E.M. Floresroux. 1993. Rates of weed spread in spatially heterogeneous environments. Ecology 74(4):999-1,011.

Blachly, J.S. and B.T. Forschler. 1996. Suppression of late-season Argentine ant (Hymenoptera: Formicidae) field populations using a perimeter treatment with containerized baits. Journal of Economic Entomology 89(6):1,497-1,500.

Bolger, D.T., A.C. Alberts, and M.E. Soulé. 1991. Occurrence patterns of bird species in habitat fragments: sampling, extinction, and nested species subsets. American Naturalist 137:155-166.

Bond, W.J. 1995. Assessing the risk of plant extinction due to pollinator and disperser failure. Pages 122-128 *in* Extinction rates, Lawton, J.G. and R.M. May, editors. Oxford, UK: Oxford University Press.

Bond, W. and P. Slingsby. 1984. Collapse of an ant-plant mutualism: the Argentine ant (*Iridomyrmex humilis*) and myrmecochorous Proteaceae. Ecology 65(4):1,031-1,037.

Boutin, C. and B. Jobin. 1998. Intensity of agricultural practices and effects on adjacent habitats. Ecological Applications 8(2):544-557.

Bright, J.A. 1986. Hiker impact on herbaceous vegetation along trails in an evergreen woodland of central Texas. Biological Conservation 36:53-69.

Brittingham, M.C. and S.A. Temple. 1983. Have cowbirds caused forest songbirds to decline? BioScience 33:31-35.

Brooks, M.L. 1995. Benefits of protective fencing to plant and rodent communities of the western Mojave Desert, California. Environmental Management 19(1):65-74.

Brothers, T.S. and A. Spingarn. 1992. Forest fragmentation and alien plant invasion of central Indiana old-growth forests. Conservation Biology 6(1):91-100.

Brown, J.H. and B.A. Harney. 1993. Population and community ecology of heteromyid rodents in temperate habitats. Pages 618-651 *in* Biology of the heteromyidae, Genoways, H.H. and J.H. Brown, editors. Special publication no. 10. American Society of Mammalogists.

Brown, J.H. and E.J. Heske. 1990. Control of a desert-grassland transition by a keystone rodent guild. Science 250:1,705-1,707.

Brown, J.H. and G.L. Lieberman. 1973. Resource utilization and coexistence of seed-eating desert rodents in sand dune habitats. Ecology 54:788-797.

Brown, J.H., O.J. Reichman, and D.W. Davidson. 1979. Granivory in desert ecosystems. Annual Review of Ecology and Systematics 10:201-227.

Buchmann, S.L. and G.P. Nabhan. 1996. The forgotten pollinators. Washington, DC: Island Press.

Cole, D.N. 1987. Effects of three seasons of experimental trampling on five montane forest communities and a grassland in western Montana, USA. Biological Conservation 40:219-244.

Cole, E.R., A.C. Medeiros, L.L. Loope, and W.W. Zuchlke. 1992. Effects of the Argentine ant on arthropod fauna of Hawaiian high-elevation shrubland. Ecology 73:1,313-1,322.

Dale, D. and T. Weaver. 1974. Trampling effects on vegetation of the trail corridors of north Rocky Mountain forests. Journal of Applied Ecology 11:767-772.

Dallman, P.R. 1998. Plant life in the world's Mediterranean climates: California, Chile, South Africa, Australia, and the Mediterranean basin. Sacramento, CA: California Native Plant Society and Berkeley, CA: University of California Press. 257 pp.

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

Davidson, D.W., R.S. Inouye, and J.H. Brown. 1984. Granivory in a desert ecosystem: experimental evidence for indirect facilitation of ants by rodents. Ecology 65(6):1,780-1,786.

DeKock, A.E. 1990. Interactions between the introduced Argentine ant, *Iridomyrmex humilis* Mayr, and two indigenous fynbos ant species. Journal of the Entomological Society of South Africa 53(1):107-108.

DeKock, A.E. and J.H. Giliomee. 1989. A survey of the Argentine ant, *Iridomyrmex humilis* (Mayr) (Hymenoptera, Formicidae) in south African fynbos. Journal of the Entomological Society of Southern Africa 52:157-164.

Del Castillo, R.F. 1994. Factors influencing the genetic structure of *Phacelia dubia*, a species with a seed bank and large fluctuations in population size. Heredity 72:446-458.

Dennis, B. 1989. Allee effects: population growth, critical density, and the chance of extinction. Natural Resource Modeling 3:481-538.

Donovan, T.M., P.W. Jones, E.M. Annand, and F.R. Thompson, III. 1997. Variation in local-scale edge effects: mechanisms and landscape context. Ecology 78(7):2,064-2,075.

Ducote, K.A. 1977. A microgeographic analysis of an introduced species: the Argentine ant in the Santa Monica Mountains. Dissertation, University of California, Los Angeles.

Ellstrand, N.C. 1994. Conservation biology of five rare plant species at the Shipley-Skinner Reserve: report on 1993-94 research. 19 pp + maps.

Erickson, J.M. 1971. The displacement of native ant species by the introduced Argentine ant *Iridomyrmex humilis* (Mayr). Psyche 78:257-266.

Forcella, F. and S.J. Harvey. 1983. Eurasian weed infestation in western Montana in relation to vegetation and disturbance. Madroño 30(2):102-109.

Frenkel, R.E. 1970. Ruderal vegetation along some California roadsides. University of California Publications in Geography 20:1-163.

Gates, J.E. and L.W. Gysel. 1978. Avian nest dispersion and fledgling outcome in field-forest edges. Ecology 59:871-883.

George, M.R., J.R. Brown, and W.J. Clawson. 1992. Application of nonequilibrium ecology to management of Mediterranean grasslands. Journal of Range Management 45:436-440.

Ghersa, C.M. and M.L. Ronsh. 1993. Searching for solutions to weed problems: do we study competition or dispersion? BioScience 43(2):104-135.

Glenn Lukos Associates, Inc. (GLA). 1999. Report: biology of the San Fernando Valley spineflower, Ahmanson Ranch, Ventura County, California. Prepared for the Ahmanson Land Company.

Groom, M.J. 1998. Allee effects limit population viability of an annual plant. The American Naturalist 151(6):487-496.

Hall, C. and F.R. Kuss. 1989. Vegetation alteration along trails in Shenandoah National Park, Virginia. Biological Conservation 48:211-227.

Harrison, C. 1981. Recovery of lowland grassland and heathland in southern England from disturbance by seasonal trampling. Biological Conservation 19:119-130.

Heithaus, E.R., D.C. Culver, and A.J. Beattie. 1980. Models of some ant-plant mutualisms. The American Naturalist 116(3):347-361.

Hickman, J.C. 1974. Pollination by ants: a low-energy system. Science 184(4,143):1,290-1,292.

Hickman, J.C., editor. 1993. The Jepson manual: higher plants of California. Berkeley, CA: University of California Press. 1,400 pp.

Hobbs, R.J. and L. Atkins. 1988. The effect of disturbance and nutrient addition on native and introduced annuals in the western Australian wheatbelt. Australian Journal of Ecology 13:171-179.

Hobbs, R.J. and L.F. Huenneke. 1992. Disturbance, diversity, and invasion: implications for conservation. Conservation Biology 6:324-337.

Hobbs, R.J. and S.E. Humphries. 1995. An integrated approach to the ecology and management of plant invasions. Conservation Biology 9(4):761-770.

Holway, D.A. 1995. The distribution of the Argentine ant (*Linepithema humile*) in central California: a twenty year record of invasion. Conservation Biology 9:1,634-1,637.

Holway, D.A. 1998. Factors governing rate of invasion: a natural experiment using Argentine ants. Oecologia 115:206-212.

Holway, D.A. 1999. Competitive mechanisms underlying the displacement of native ants by the invasive Argentine ant. Ecology 80(1):238-251.

Hovore, F. 1999. Biological consultant, Hovore Associates. Personal communication with P. Gordon-Reedy. November 23.

Hughes, L. and M. Westoby. 1992. Effect of diaspore characteristics on removal of seeds adapted for dispersal by ants. Ecology 73(4):1,300-1,312.

Human, K.G. and D.M. Gordon. 1996. Exploitation and interference competition between the invasive Argentine ant, *Linepithema humile*, and native ant species. Oecologia 105:405-412.

Inouye, R.S., G.S. Byers, and J.H. Brown. 1980. Effects of predation and competition on survivorship, fecundity, and community structure of desert annuals. Ecology 61:1,344-1,351.

Jennersten, O. 1988. Pollination in *Dianthus deltoides* Caryophyllaceae: effects of habitat fragmentation on visitation rate and seed set. Conservation Biology 2:359-366.

Johnstone, I.M. 1986. Plant invasion windows: a time-based classification of invasion potential. Biological Review 61:369-394.

Jones, E. 1999. Professor of Botany, California State University, Fullerton. Personal communication with P. Gordon-Reedy. November 23.

Jules, E.S. and B.J. Rathcke. 1999. Mechanisms of reduced *Trillium* recruitment along edges of old-growth forest fragments. Conservation Biology 13(4):784-793.

Keeley, J.E. 1990. The California valley grassland. Pages 3-23 *in* Endangered plant communities of southern California, Schoenherr, A.A., editor. Claremont, CA: Southern California Botanists, special publication no. 3.

Keeley, J.E. 1991. Seed germination and life history syndromes in the California chaparral. The Botanical Review 57:81-116.

Kelly, P.A. and J.T. Rotenberry. 1993. Buffer zones for ecological reserves in California: replacing guesswork with science. Pages 85-92 *in* Proceedings of the symposium: interface between ecology and land development in California, Keeley, J.E., editor. May 1-2, 1992; Occidental College, Los Angeles.

Knight, R.L. and M.K. Rust. 1990. The urban ants of California with distributional notes of imported species. Southwestern Entomologist 15:167-178.

Kunin, W. and Y. Iwasa. 1996. Pollinator foraging strategies in mixed floral arrays: density effects and floral constancy. Theoretical Population Biology 49:232-263.

Lamont, B.B., P.G.L. Klinkhamer, and E.T.F. Witkowski. 1993. Population fragmentation may reduce fertility to zero in *Banksia goodii* – a demonstration of the Allee effect. Oecologia 94:446-450.

Langen, T.A., D.T. Bolger, and T.J. Case. 1991. Predation on artificial bird nests in chaparral fragments. Oecologia 86:395-401.

Laurance, W.F. 1991. Edge effects in tropical forest fragments: application of a model for the design of nature reserves. Biological Conservation 57:205-219.

Lonsdale, W.M. and A.M. Lane. 1991. Vehicles as vectors of weed seeds in Kakadu National Park. Kowari 2:167-169.

Louda, S.M. 1989. Predation in the dynamics of seed regeneration. Pages 25-52 *in* Ecology of soil seed banks, Leck, M.A., V.T. Parker, and R.L. Simpson, editors. San Diego, CA: Academic Press, Inc. 462 pp.

Lovejoy, T.E., R.O. Bierregaard, Jr., and A.B. Rylands. 1986. Edge and other effects of isolation on Amazon forest fragments. Pages 257-285 *in* Conservation biology: the science of scarcity and diversity, Soulé, M.E., editor. Sunderland, MA: Sinauer Associates.

MacDonald, I.A.W., D.M. Graber, S. DeBenedetti, R.H. Groves, and E.R. Fuentes. 1988. Introduced species in nature reserves in Mediterranean-type climatic regions of the world. Biological Conservation 44:37-66.

Marrs, R.H., C.T. Williams, A.J. Frost, and R.A. Plant. 1989. Assessment of the effects of herbicide spray drift on a range of plant species of conservation interest. Environmental Pollution 59:71-86.

Marshall, D.L., A.J. Beattie, and W.E. Bollenbacher. 1979. Evidence for diglycerides as attractants in an ant-seed interaction. Chemical Ecology 5:335-344.

Maschinski, J. R. Frye, and S. Rutman. 1997. Demography and population viability of an endangered plant species before and after protection from trampling. Conservation Biology 11(4):990-999.

Matlack, G.R. 1993. Microenvironment variation within and among forest edge sites in the eastern United States. Biological Conservation 66:185-194.

Matlack, G.R. 1993. Sociological edge effects – spatial distribution of human impact in suburban forest fragments. Environmental Management 17(6):829-835.

McConnaughay, K.D.M. and F.A. Bazzaz. 1987. The relationship between gap size and performance of several colonizing annuals. Ecology 68(2):411-416.

McEvoy, P.B. and C.S. Cox. 1987. Wind dispersal distances in dimorphic achenes of ragwort *Senecio jacobaea*. Ecology 68(6):2,006-2,015.

McIntyre, S. and S. Lavorel. 1994. Predicting richness of native, rare, and exotic plants in response to habitat and disturbance variables across a variegated landscape. Conservation Biology 8(2):521-531.

Menges, E.S. 1991. Seed germination percentage increases with population size in a fragmented prairie species. Conservation Biology 5(2):158-163.

Menke, J.W. 1992. Grazing and fire management for native perennial grass restoration in California grasslands. Fremontia 20(2):22-25.

Mills, L.S. 1996. Fragmentation of a natural area: dynamics of isolation for small mammals on forest remnants. Pages 199-219 *in* Natural parks and protected areas, Wright, R.G., editor. Cambridge, MA: Blackwell Science.

O'Dowd, D.J. and M.E. Hay. 1980. Mutualism between harvester ants and a desert ephemeral: seed escape from rodents. Ecology 61:531-540.

Ogden Environmental and Energy Services Co., Inc. (Ogden). 1997. Disturbance history of vegetation on reserve land. Southwestern Riverside Connty Multi-Species Reserve. Prepared for Metropolitan Water District of Southern California. February. 50 pp + appendices.

Ogden Environmental and Energy Services Co., Inc. (Ogden). 1999. Final report: Eastside Reservoir project seed testing program. Prepared for the Metropolitan Water District of Southern California. 30 pp.

Paton, J. 1986. The role that insects play in the pollination of *Protea neriifolia* and *Protea eximia* (Proteaceae). B.Sc. (Hons) project, University of Cape Town.

Pavlik, B.M., N. Ferguson, and M. Nelson. 1993. Assessing limitations on the growth of endangered plant populations. II. Seed production and seed bank dynamics of *Erysimum capitatum* ssp. *angustatum* and *Oenothera deltoides* ssp. *howellii*. Biological Conservation 65:267-278.

Perry, J.N. and J.L. Gonzalez-Andujar. 1993. Dispersal in a metapopulation neighbourhood model of an annual plant with a seedbank. Journal of Ecology 81:453-463.

Pollak, O. and T. Kan. 1998. The use of prescribed fire to control invasive exotic weeds at Jepson Prairie Preserve. Pages 241-249 *in* Ecology, conservation, and management of vernal pool ecosystems, Witham, C.W., E.T. Bauder, D. Belk, W.R. Ferren, Jr., and R. Ornduff, editors. Sacramento, CA: California Native Plant Society.

Proctor, M., P. Yeo, and A. Lack. 1996. The natural history of pollination. Portland, OR: Timber Press. 479 pp.

Reichman, O.J. 1979. Desert granivore foraging and its impact on seed densities and distributions. Ecology 60:1,085-1,092.

Reichman, O.J. 1984. Spatial and temporal variation of seed distributions in Sonoran Desert soils. Journal of Biogeography 11:1-11.

Reveal, J. 1999. Professor of Botany, University of Maryland (retired). Personal communication with P. Gordon-Reedy and I. Mendez. November 18.

Roberts, J.T. and E.R. Heithaus. 1986. Ants rearrange the vertebrate-generated seed shadow of a neotropical fig tree. Ecology 67:1,046-1,051.

Samson, D.A., T.E. Philippi, and D.W. Davidson. 1992. Granivory and competition as determinants of annual plant diversity in the Chihuahuan desert. Oikos 65:61-80.

Santos, T. and J.L. Tellería. 1992. Edge effects on nest predation in Mediterranean fragmented forests. Biological Conservation 60:1-5.

Saunders, D.A., R.J. Hobbs, and C.R. Margules. 1991. Biological consequences of ecosystem fragmentation: a review. Conservation Biology 5:18-32.

Sauvajot, R.M. and M. Buechner. 1993. Effects of urban encroachment on wildlife in the Santa Monica Mountains. Pages 171-180 *in* Interface between ecology and land development in California, Keeley, J.E., editor. Los Angeles, CA: Southern California Academy of Sciences, Los Angeles.

Scott, T.A. 1993. Initial effects of housing construction on woodland birds along the wildland urban interface. Pages 181-188 in Interface between ecology and land development in California, Keeley, J.E., editor. Los Angeles, CA: Southern California Academy of Sciences, Los Angeles.

Singh, N., M. Yunus, K. Srivastava, S.N. Singh, V. Pandey, J. Misra, and K.J. Ahmad. 1995. Monitoring of auto exhaust pollution by roadside plants. Environmental Monitoring and Assessment 34:13-25.

Soulé, M.E., D.T. Bolger, A.C. Alberts, J. Wright, M. Sorice, and S. Hill. 1988. Reconstructed dynamics of chaparral requiring birds in urban habitats. Conservation Biology 2:75-92.

Spencer, W.D. and A. Goldsmith. 1994. Impacts of free-ranging house cats on wildlife at a suburban-desert interface. Abstract of paper presented at the eighth annual meeting of the Society for Conservation Biology. Guadalajara, Jalisco, Mexico. June 1994.

Stamps, J.A., M. Buechner, and V.V. Krishnan. 1987. The effects of edge permeability and habitat geometry on emigration from patches of habitat. American Naturalist 129(4):533-552.

Stebbins, G.L. 1974. Flowering plants: evolution above the species level. Cambridge, MA: The Belknap Press of Harvard University Press. 397 pp.

Suarez, A.V., D.T. Bolger, and T.J. Case. 1998. Effects of fragmentation and invasion on native ant communities in coastal southern California. Ecology 79(6):2,041-2,056.

Temple, S.A. 1987. Predation of turtle nests increases near ecological edges. Copeia 1987:250-252.

Templeton, A.R. and D.A. Levin. 1979. Evolutionary consequences of seed pools. American Naturalist 114:232-249.

Thompson, D.B., J.H. Brown, and W.D. Spencer. 1991. Indirect facilitation of granivorous birds by desert rodents: experimental evidence from foraging patterns. Ecology 72(3):852-863.

Tremper, B.D. 1976. Distribution of the Argentine ant, *Iridomyrmex humilis* Mayr. in relation to certain native ants of California: ecological, physiological, and behavioral aspects. Dissertation, University of California, Berkeley, CA.

Tyser, R.W. and C.A. Worley. 1992. Alien flora in grasslands adjacent to road and trail corridors in Glacier National Park, Montana (U.S.A.). Conservation Biology 6(2):253-262.

Usher, M.B. 1988. Biological invasions of nature reserves: a search for generalizations. Biological Conservation 44:119-135.

van Treuren, R., R. Bijlsma, N.J. Ouborg, and W. van Delden. 1993. The effects of population size and plant density on outcrossing rates in locally endangered *Salvia pratensis*. Evolution 47(4):1,094-1,104.

Visser, D., M.G. Wright, and J.H. Giliomee. 1996. The effect of the Argentine ant, *Linepithema humile* (Mayr) (Hymenoptera: Formicidae), on flower-visiting insects of *Protea nitida* Mill. (Proteaceae). African Entomology 4(2):285-287.

Vissman, S. 1993. Predation on avian nests in coastal sage: an ecological study on nest height and edge effects. Masters thesis, San Diego State University.

Vitousek, P.M. 1990. Biological invasions and ecosystem processes: towards an integration of population biology and ecosystem studies. Oikos 57:7-13.

Vitousek, P., J. Aber, R.W. Howarth, G.E. Likens, P.A. Matson, D.W. Schindler, W.H. Schlesinger, and G.D. Tilman. 1997. Human alteration of the global nitrogen cycle: causes and consequences. Issues in Ecology 1:1-15.

Vitousek, P.M. and L.R. Walker. 1989. Biological invasion by *Myrica faya* in Hawai'i: plant demography, nitrogen fixation, ecosystem effects. Ecological Monographs 59:247-265.

Ward, P.S. 1987. Distribution of the introduced Argentine ant (*Iridomyrmex humilis)* in natural habitat of the lower Sacramento Valley and its effects on the indigenous ant fauna. Hilgardia 55:1-16.

Westman, W.E. 1981. Factors influencing the distribution of species of Californian coastal sage scrub. Ecology 62(2):439-455.

Westman, W.E. 1987. Implications of ecological theory for rare plant conservation in coastal sage scrub. Pages 133-140 *in* Conservation and management of rare and endangered plants, Elias, T.S., editor. Proceedings of a California conference on the conservation and management of rare and endangered plants. California Native Plant Society, Sacramento. November 5-8, 1986.

Westman, W.E. 1990. Park management of exotic plant species: problems and issues. Conservation Biology 4(3):251-260.

Wilcove, D.S. 1985. Nest predation in forest tracts and the decline of migratory songbirds. Ecology 66:1,211-1,214.

Wilcove, D.S., C.H. McLellan, and A.P. Dobson. 1986. Habitat fragmentation in the temperate zone. Pages 237-256 in Conservation biology: the science of scarcity and diversity, Soulé, M.E., editor. Sunderland, MA: Sinauer Associates.

Wildland/Urban Interface Task Force. 1994. Report of the Wildland Urban/Interface Task Force. Submitted to Larry J. Holms, Director of Fire Services, Orange County Fire Department, Orange County, CA.

Yahner, R.H. 1988. Changes in wildlife communities near edges. Conservation Biology 2:33-339.

Zavon, J.A. 1982. Grazing and fire effect on annual grassland composition and sheep diet selectivity. M.S. thesis, University of California, Davis, CA. 41 pp.

Zink, T.A. and M.F. Allen. 1998. The effects of organic amendments on the restoration of a disturbed coastal sage scrub habitat. Restoration Ecology 6(1):52-58.

Valley Fever Fact Sheet

What is Valley fever?

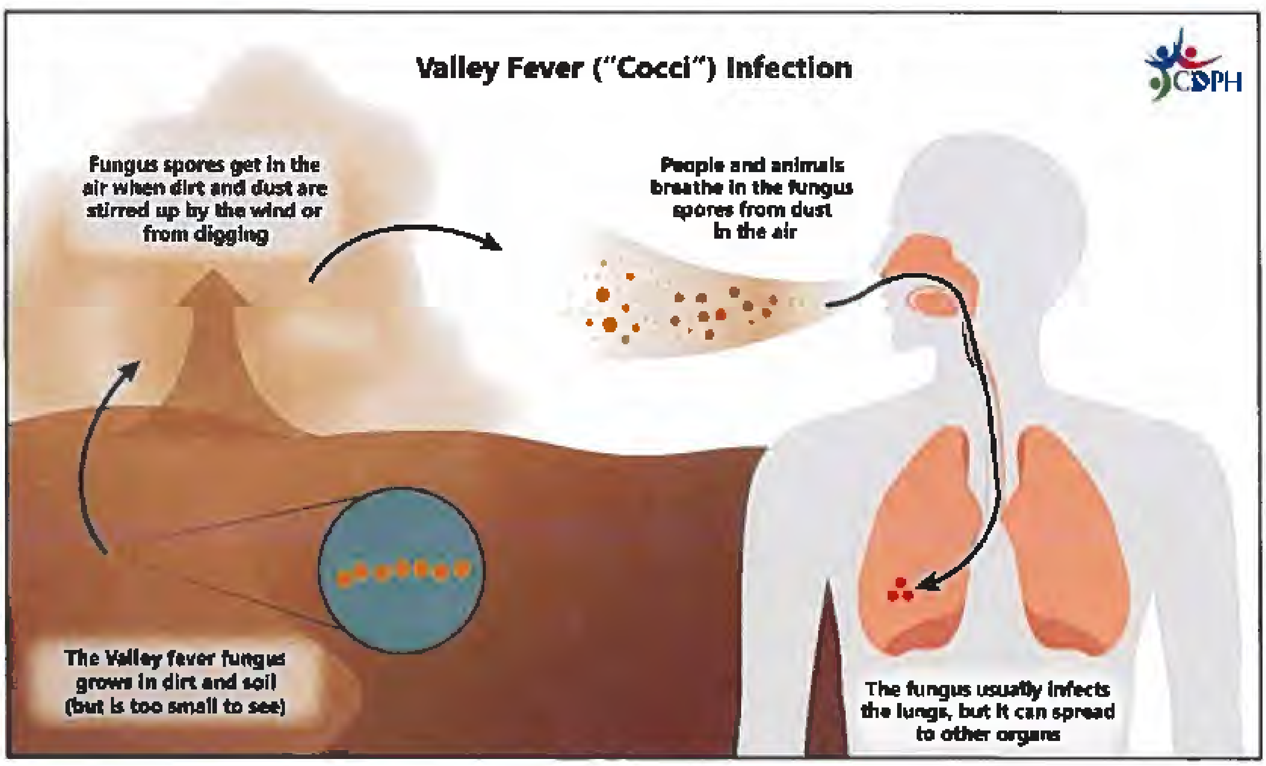
Valley fever (also called coccidioidomycosis or "cocci") is an infectious disease caused by the *Coccidioides* fungus that lives in the soil and dirt in certain areas of California and the southwestern United States. If you breathe in this fungus from dust in the air, it can infect your lungs and cause symptoms such as cough, fever, chest pain, or tiredness. Some people with Valley fever may develop severe disease, which may require hospitalization. In rare cases, the infection can spread beyond the lungs to other parts of the body (this is called disseminated Valley fever).

In California, the number of reported Valley fever cases has greatly increased in recent years. Since 2000, the number of cases has increased from less than 1,000 cases to more than 9,000 cases in 2019.

How do people get Valley fever?

People can get Valley fever by breathing in dust that contains spores of the *Coccidioides* fungus. Like seeds from plants, a fungus grows and spreads from tiny spores that are too small to see. When soil or dirt are stirred up by strong winds or while digging, dust containing these fungal spores can get into the air. Anyone who lives, works, or visits in an area where the Valley fever fungus grows can breathe in these fungal spores without knowing it and become infected.

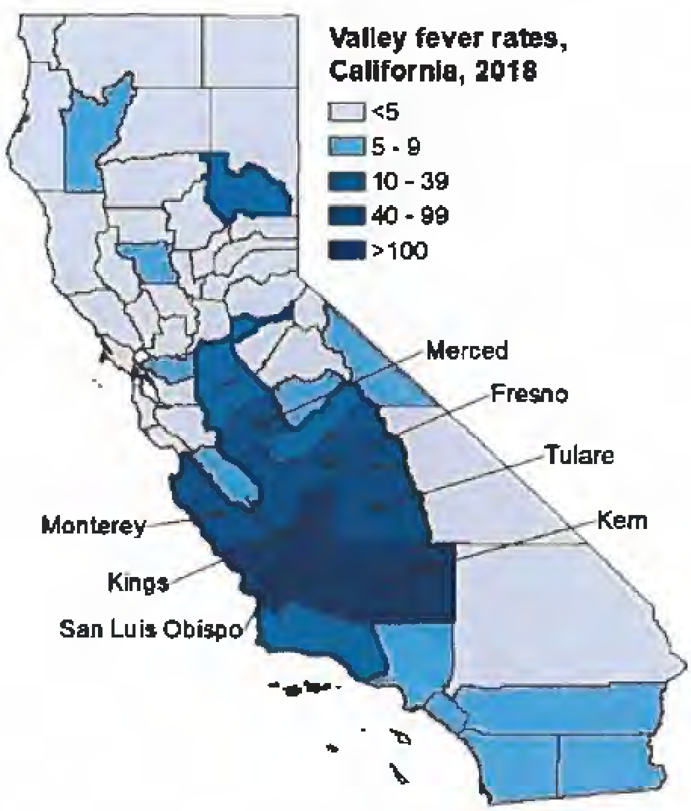
Animals, including pets, can also become infected by breathing in fungal spores. Valley fever is not contagious and cannot spread from one person or animal to another.



When and where do people get Valley fever?

People can get Valley fever any time of the year, but more people are likely to be infected with the fungus that causes Valley fever in the late summer and fall than at other times of the year.

People are more likely to get Valley fever if they live, work, or visit in areas where the fungus grows in the soil or is in dust in the air. There is no test available to see if the Valley fever fungus is growing in the soil in certain areas, but we do know that Valley fever has been diagnosed in people living in counties throughout California. Most cases of Valley fever in California (over 65%) are reported in people who live in the Central Valley and Central Coast regions. The map below shows the rates of reported Valley fever cases by county in California, with darker shaded counties having higher rates than lighter shaded counties.



Rates of reported Valley fever cases per 100,000 population.
Darkest colored counties had the highest rates of Valley fever.

Outside of California, Valley fever occurs in Arizona, and some areas of Nevada, New Mexico, Utah, and Texas, and parts of Mexico and Central and South America.

What are the signs and symptoms of Valley fever?

Most people (about 6 in 10) infected with Valley fever have no symptoms, and their bodies will fight off the infection naturally. People who do get sick usually develop symptoms 1–3 weeks after breathing in the fungus.

Valley fever usually infects the lungs, and some people can develop respiratory symptoms or pneumonia (a lung infection). People who get sick may have some of the following symptoms:

- Fatigue (tiredness)
- Cough
- Chest pain
- Fever
- Rash on upper body or legs
- Headaches
- Muscle or joint aches
- Night sweats
- Unexplained weight loss

Some of these symptoms are similar to those of other common illnesses (including COVID-19 and the flu), but Valley fever symptoms can last a month or more.

Most people fully recover from Valley fever. In rare cases, Valley fever can spread to other parts of the body and infect the brain, joints, bone, skin, or other organs. This form of Valley fever can be very serious and fatal.

How is Valley fever diagnosed and treated?

If you have Valley fever symptoms that last more than a week, talk to a healthcare provider. Since Valley fever symptoms are similar to those of other common illnesses, your provider may order a blood test or other tests (such as a chest x-ray) to help diagnose Valley fever.

Treatment may not be needed for mild infections, which can sometimes get better on their own. However, all people with symptoms should see a healthcare provider who can determine if treatment is needed. There are no over-the-counter medications to treat Valley fever.

If you are diagnosed with Valley fever, it is very important to follow the instructions given by your healthcare provider about treatment, follow-up testing, and appointments.

If a person has had Valley fever before, can they get it again?

If a person has already had Valley fever, their immune system will most likely protect them from getting it again. Although it is rare, some people who have already had Valley fever could get sick again if their immune system weakens because of certain medical conditions (such as cancer) or by taking certain medications, like those for cancer, organ transplant, or autoimmune disease.

Are certain people at greater risk for Valley fever?

Anyone can get Valley fever, including healthy adults and children. Certain groups may be at higher risk of getting Valley fever, and other groups may be at higher risk of having severe or disseminated disease if infected.

People at higher risk of getting Valley fever:

People who live, work, or travel in areas with high rates of Valley fever (see map above) may be at higher risk of getting infected than others, especially if they:

- Participate in outdoor activities that involve close contact to dirt or dust, including yard work, gardening, and digging
- Live or work near areas where dirt and soil are stirred up, such as construction or excavation sites
- Work in jobs where dirt and soil are stirred up or disturbed, including construction, farming, military work, and archaeology
  - If you work in a job where dirt or soil is disturbed in a place where Valley fever is common, you and your employer may want to review the CDPH website for preventing work-related Valley fever.

More cases of Valley fever have been reported among men than among women, and among adults than among children. Work and outdoor exposure among adult men may explain the higher rates of Valley fever in this group.

People at higher risk of having severe or disseminated Valley fever if infected:

- Older adults (60+ years old)
- People who are Black or Filipino
- Pregnant women, especially in the later stages of pregnancy
- People with diabetes
- People with health conditions that weaken their immune system such as:
  - Cancer
  - Human immunodeficiency virus (HIV) infection
  - Treatment with chemotherapy, steroids, or other medications that affect the immune system
  - Organ transplant

How can I help reduce my risk of getting Valley fever?

It is very difficult to avoid breathing in the Valley fever fungus in areas where it is common in the environment. People who live, work, or travel in these areas can try to avoid spending time in dusty areas as much as possible to reduce the risk of breathing in the Valley fever fungus from dust in the air. There is no vaccine to prevent Valley fever.

Some practical tips may help reduce the risk of getting Valley fever. It is important to know that these steps have not been proven to prevent Valley fever.

Avoid dust in places where Valley fever is common (where Valley fever rates are high):

- Stay inside and keep windows and doors closed when it is windy outside and the air is dusty, especially during dust storms.
- Consider avoiding outdoor activities that involve close contact to dirt or dust, including yard work, gardening, and digging, especially if you are in one of the groups at higher risk for severe or disseminated Valley fever.
- Cover open dirt areas around your home with grass, plants, or other ground cover to help reduce dusty, open areas.
- While driving in these areas, keep car windows closed and use recirculating air, if available.
- Try to avoid dusty areas, like construction or excavation sites.
- If you cannot avoid these areas, or if you must be outdoors in dusty air, consider wearing an N95 respirator (a type of face mask) to help protect against breathing in dust that can cause Valley fever.
    - N95 respirators are available at drugstores and hardware supply stores.
    - To be effective, N95 masks must be fitted properly. Instructions can be found on several websites, including the U.S. Centers for Disease Control and Prevention instruction video for using disposable respirators.

When digging in dirt or stirring up dust in areas where Valley fever is common:

- Stay upwind of the area where dirt is being disturbed.
- Wet down soil before digging or disturbing dirt to reduce dust.
- Consider wearing an N95 respirator (mask).
- After returning indoors, change out of clothes if covered with dirt.
    - Be careful not to shake out clothing and breathe in the dust before washing. If someone else is washing your clothes, warn the person before they handle the clothes.

What is being done about Valley fever in California?

The California Department of Public Health (CDPH) and local health departments track cases of Valley fever and monitor the number of people who get sick with Valley fever in California.

CDPH also reviews data and investigates outbreaks of Valley fever to better understand:

- Where Valley fever is most common
- Who is most affected by Valley fever
- If disease trends of Valley fever are changing
- How people can reduce their risk of getting Valley fever

CDPH also works to raise awareness of Valley fever among healthcare providers and the public and provides information to employers to help prevent Valley fever in the workplace.

Where can I get more information about Valley fever?

Contact your local health department or visit CDPH's Valley fever website for more information about Valley fever. You can also visit the CDC's Valley fever website.

Updated June 2021

VALLEY FEVER

en Español sa Tagalog

Anyone, even healthy adults and children, can get Valley fever after breathing in the Valley fever fungus from dust in outdoor air, especially in the Central Valley or Central Coast areas of California. Certain people have a higher risk of getting Valley fever, especially those who spend more time outdoors and are exposed to dirt and dust. Other groups have a higher risk of getting very sick from Valley fever and being hospitalized if they are infected.

Groups at Risk for Valley Fever

People at higher risk of *getting Valley fever* include:

People who live, work, or travel in areas with high rates of Valley fever, especially if they:

- Participate in outdoor activities that involve close contact with dirt or dust, including digging projects or landscaping
- Live or work near areas where dirt and soil are stirred up, such as construction or excavation sites
- Work in jobs where dirt and soil are stirred up or disturbed, including construction, field work, military work, and archaeology
  - If you work in a job where dirt or soil is disturbed in a place where Valley fever is common, you and your employer should review the **CDPH website for preventing work-related Valley fever.**

People at higher risk of *severe Valley fever or getting very sick* if they are infected include:

- Older adults (60+ years old)

- People who are Black or Filipino
- Pregnant women, especially in the later stages of pregnancy
- People with diabetes
- People with health conditions that weaken the immune system, such as:
  - Cancer
  - Human immunodeficiency virus (HIV) infection
  - Autoimmune illnesses
  - Treatment with chemotherapy, steroids, or other medications that affect the immune system
  - Organ transplant

Learn More

SYMPTOMS

PREVENTION TIPS

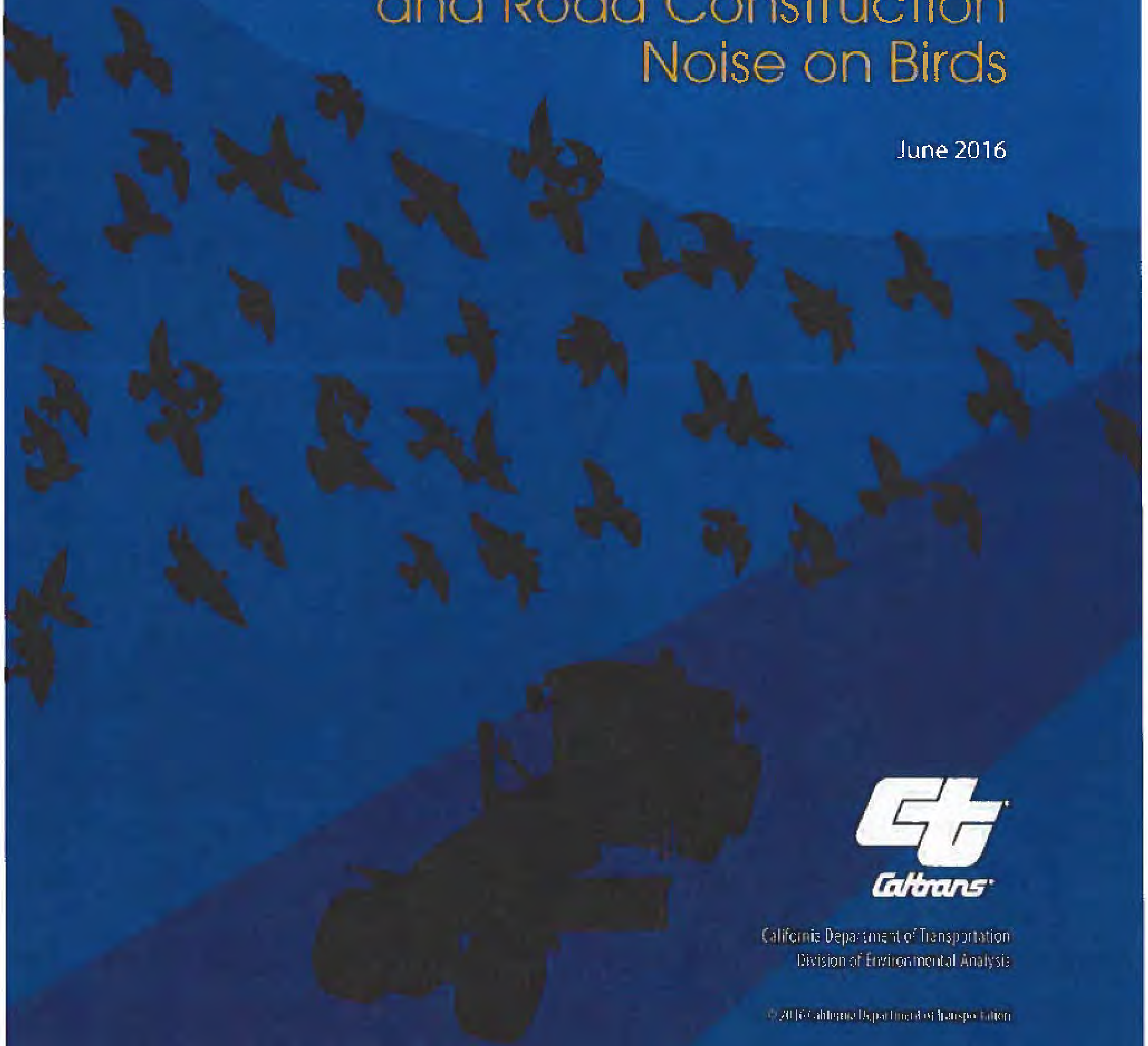
INFORMATION FOR OUTDOOR WORKERS

Page Last Updated : August 12, 2022

Technical Guidance for
Assessment and Mitigation of the

Effects of Traffic Noise and Road Construction Noise on Birds

June 2016



Caltrans

California Department of Transportation
Division of Environmental Analysis

This page left intentionally blank

1. Report No.	2. Type of Report	3. Report Phase and Edition
CTHWANP-RT-15-306.04.2	Guidance Manual	Final

4. Title and Subtitle	5. Report Date
Technical Guidance for Assessment and Mitigation of the Effects of Highway and Road Construction Noise on Birds	June 2016
6. Author(s) Robert J. Dooling, Arthur N. Popper (Contract manager, David Buehler, PE)	7. Caltrans Project Coordinators: Bruce Rymer PE, Amy Bailey, James Henke, Amy Golden,
8. Performing Organization Names and Addresses California Department of Transportation Division of Environmental Analysis 1120 N Street, MS-27 Sacramento CA 95814 www.dot.ca.gov/hq/env/	9. Task Order No. 4 10. Contract No. 43A0306 - ICF International Robert J. Dooling, Phd., and Arthur N. Popper, Phd.
11. Sponsoring Agency Name and Address California Department of Transportation Division of Environmental Analysis 1120 N Street, MS-27 Sacramento, CA 95814	12. Caltrans Functional Reviewers: *Division of Environmental Analysis* Amy Bailey, James Henke, Amy Golden, Bruce Rymer
13. Supplementary Notes This is an update to the Sept. 30, 2007 document, *The Effects of Highway Noise on Birds.*	14. External Reviewers

15. Abstract

The purpose of this report is to provide Department engineers, biologists, and consultants with guidance related to the effects of traffic noise and road construction noise on birds. This manual covers bird hearing and communication within the highway operational and construction sound environment. The following topics are discussed: stress and physiological effects, acoustic over-exposure, masking, dynamic behavioral and population effects, extrapolation of data from humans and birds to other species. Interim guidelines for estimating the effects of these noise sources on birds is provided.

16. Key Words	17. Distribution Statement	18. No. of pages
Highway traffic noise, construction noise, bioacoustics, bird ear anatomy, bird vocalization, avian hearing, threshold shift, critical masking ratio, acoustic over-exposure	Available to the general public	96

Technical Guidance for Assessment and Mitigation of the Effects of Highway and Road Construction Noise on Birds

California Department of Transportation
Division of Environmental Analysis
1120 N Street, Room 4301 MS27
Sacramento, CA 95814

June 2016

For individuals with sensory disabilities, this document is available in alternate formats. For alternate format information, contact Caltrans Division of Environmental Analysis at (916) 653-6557, TTY 711, or write to Caltrans at the above address.

The California Department of Transportation. 2016. *Technical Guidance for Assessment and Mitigation of the Effects of Highway and Road Construction Noise on Birds.* June. (Contract 43A0306.) Sacramento, CA. Prepared by ICF International, Sacramento, CA, Robert Dooling, Gaithersburg, MD, and Arthur Popper, Silver Spring, MD.

Table of Contents

This page left intentionally blank

Executive Summary

Recent literature on the effects of noise in the environment has shown that the world is becoming a noisier place and that the effects of chronic noise exposure on terrestrial animals, including birds, could be significant. Furthermore, with population increases and urbanization, traffic and road construction are major and increasing sources of environmental noise.

A. Overview of this Guidance Document

There is a long-standing concern that roadway construction noise and subsequent traffic noise may be detrimental to wildlife, and especially birds, which relies heavily on acoustic communication. The Endangered Species Act provides additional, compelling, motivation for understanding the effects of traffic and construction noise on federally listed bird species that are in danger of extinction. Effects of construction and/or traffic noise may be nonexistent in certain circumstances, such as when the level of these noises is below natural ambient noise levels, and insignificant in other circumstances, such as when the noise adds very little to existing ambient noise levels.

In contrast, construction or traffic noise that adds significantly to natural ambient noise has the possibility of producing a suite of significant short- and long-term behavioral and physiological changes in birds. These may include changes in foraging location and behavior; interference with acoustic communicate between conspecifics; failure to recognize other important biological signals, such as sounds of predators and/or prey; decreasing hearing sensitivity temporarily or permanently; and/or increasing stress and altering steroid hormone levels. Any of these effects could have long-term consequences and enduring impacts that include interference with breeding by individuals and populations, thereby threatening the survival of individuals or species.

This Guidance Document is an updated version of the 2007 report entitled *The Effects of Highway Noise on Birds* prepared by the authors (Dooling & Popper, 2007).

B. Definitions

Several terms are used in this report. Some of these terms have multiple meanings and are defined herein. Other terms are defined in the glossary.

- **Construction Noise**: Noise produced during the construction of a roadway.
- **Effects**: Any response by birds to traffic and construction noise. This simple definition does not invoke or imply regulatory definitions of "effect" as found in any law or regulation affecting birds.
- **Roadway**: Any paved road on which there is vehicular traffic.
- **Traffic Noise**: Noise produced by vehicles on any paved roadway, ranging from highways to single-lane streets.

C. Findings

A review of relevant literature provided insight on several important issues regarding the effects of traffic and construction noise on birds.

1) Stress and physiological effects:
   a) There are no studies definitively identifying traffic noise as the critical variable affecting bird behavior near roadways and highways.
   b) There are well-documented adverse effects of sustained traffic noise on humans, including stress, physiological and sleep disturbances, and changes in feelings of well-being that may be applicable to birds.
   c) Traffic and construction noise below a bird's masked threshold has no effect.

2) Acoustic overexposure:
   a) Birds are more resistant to both temporary and permanent hearing loss or to hearing damage from acoustic overexposure than are humans and other animals that have been tested.
   b) Birds can regenerate the sensory hair cells of the inner ear, thereby providing a mechanism for recovering from intense acoustic overexposure, a capability not found in mammals.
   c) The studies of acoustic overexposure in birds have considerable relevance for estimating hearing damage effects of traffic noise, non-continuous construction noise, and for impulsive-type construction noise, such as that from pile driving.

3) Masking:
   a) Continuous noise of sufficient intensity in the frequency region of bird hearing can have a detrimental effect on a bird's ability to detect and discriminate between the vocal signals of other birds.
   b) Noise in the spectral region of the vocalizations has a greater masking effect than noises outside this range. Thus, traffic noise will cause less masking than other environmental noises of equal overall level but that contain energy in a higher spectral region (around 2–4 kilohertz [kHz]) (e.g., insects, vocalizations of other birds).
   c) Generally, human auditory thresholds in quiet and in noise are better than that of the typical bird; therefore:
      (1) The typical human can hear a single vehicle, traffic noise, and construction noise at a much greater distance from the roadway than can the typical bird. This fact provides a valuable, common sense, easy-to-apply risk criterion.
      (2) However, the typical human is also able to hear a bird vocalizing in a noisy environment at twice the distance that a typical bird, which suggests, in this case, that relying on human hearing as the primary criterion seriously underestimates the effects of noise on bird communication.
   d) From knowledge of: (i) bird hearing capabilities in quiet and noise, (ii) the Inverse Square Law, (iii) excess attenuation in a particular environment, and (iv) species-specific acoustic characteristics of vocalizations, reasonable predictions can be made about possible maximum communication distances between two birds in continuous noise.

e) The amount of masking of vocalizations can be predicted from the peak in the total power spectrum of the vocalization and the bird's critical ratio (i.e., signal-to-noise ratio) at that frequency of peak energy.
f) Birds, like humans and other animals, employ a range of short-term behavioral strategies, or adaptations, for communicating in noise resulting in a doubling to quadrupling of the efficiency of hearing in noise.

4) Dynamic behavioral and population effects:
    a) Any components of traffic noise that are audible to birds may have effects independent of and beyond the effects listed above. At distances from the roadway where traffic noise levels fall below ambient noise levels in the spectral region for vocal communication (i.e., 2–8 kHz) (Figure ES1), low-level but audible sound in non-communication frequencies (e.g., the rumbling of a truck) can potentially cause may cause physiological or behavioral responses). Because the more recent literature points to noise as possibly having wide-ranging effects on birds, the additive effects of traffic noise and environmental noise must be considered beyond solely the effects due specifically to traffic noise.

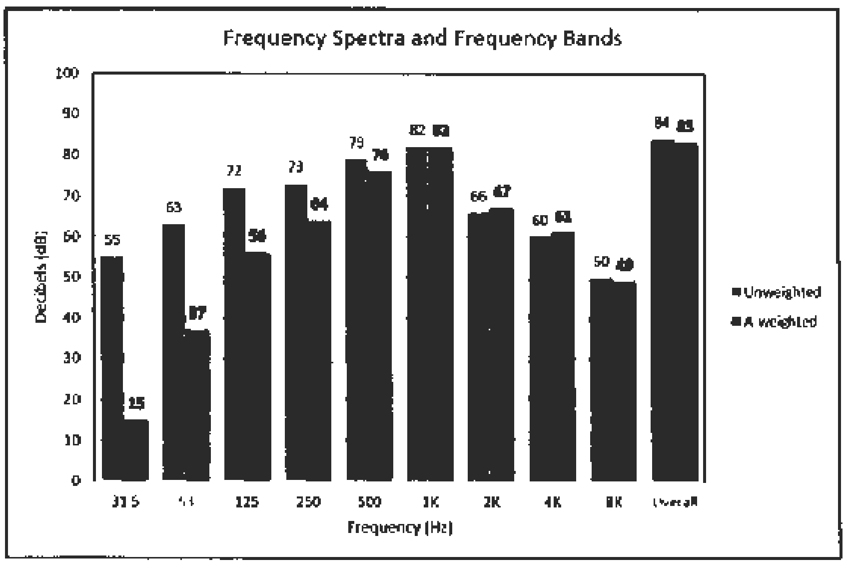


Figure ES1. Caltrans Traffic Noise Spectra Showing Differences in Unweighted and Weighted Spectra and Overall Levels[1]

5) Extrapolation of data from humans and birds to other species:
    a) Since there is substantial variation in bird hearing and behavior, considerable care must be taken when trying to extrapolate data between species, particularly when the species have different hearing capabilities and acoustic behaviors.
    b) Data on human hearing has some relevance to understanding effects of sound on birds. In particular, data on physiological effects in humans may have general implications for birds, but applications to specific situations will require additional study.

6) Much more data are needed on:

[1] Figure from: http://www.dot.ca.gov/hq/env/noise/online_training_module1/slides/slide50.htm

a) Physiological effects of sound on birds.
b) How responses vary between species with regard to masking, hearing loss, and hearing recovery.
c) Hearing in young animals and how it compares to adult hearing.
d) Additional, carefully selected species so there is a large enough database from which to allow extrapolation between species and enable broader generalizations regarding the effects of noise on birds.
e) A broader range of studies, as discussed in detail in Appendix F.

The authors suggest the *interim* compliance guidelines in Figure ES2 and Table ES1 and a science-based approach, using human and avian data from both the laboratory and the field, to address potential impacts of noise on bird species.

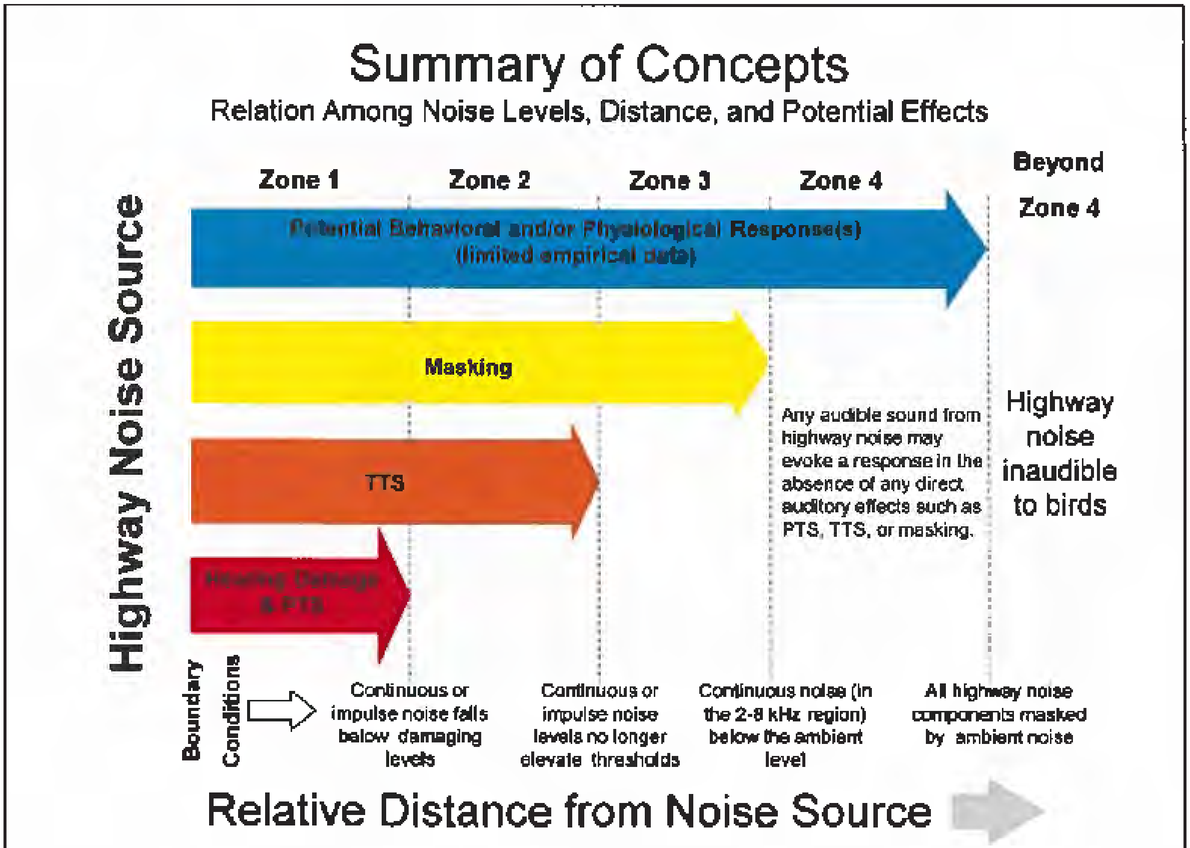


Figure ES2. Effects of Traffic and Construction Noise on Birds

Categories of traffic and construction noise effects on birds with distance from the source. Zone 1 is closest to the source while Zone 4 is farthest away. Sound level decreases farther from the source. See text for discussion.

This Guidance Document reviews four classes of potential effects of traffic noise on birds, as discussed below. The basis for the guidelines for each of the classes differs. Table ES1 provides specific interim criteria.

1. *Behavioral and/or physiological effects*: There are no definitive studies showing that traffic noise exclusively (as opposed to correlated variables) has an adverse effect on birds. While a wealth of human data and experience suggest traffic noise could have a number of adverse effects, there are several studies (e.g., Awbrey *et al.*, 1995) showing that birds (as well as other animals) adapt quite well, and may even appear to sometimes prefer, environments that include high levels of traffic noise. Given the lack of empirical data on this point, it is recommended that subjective human experience with the noise in question be used as an interim guideline to estimate acceptable noise levels for avoiding stress and physiological effects. Noise types and levels that appear to increase stress and adverse physiological reactions in humans may also have similar consequences in birds.

2. *Damage to hearing from acoustic overexposure*: While many behavioral and physiological studies lack specificity, there are many definitive studies showing precise effects of intense noise on bird hearing and auditory structures. These extensive data show that birds are much more resistant to hearing loss and auditory damage from acoustic overexposure than are humans and other mammals. Traffic and construction noise, even at extreme levels, is unlikely to cause threshold shift, hearing loss, auditory damage, or damage to other organ systems in birds and, therefore, interim guidelines for hearing damage in birds from traffic and construction noise are probably not needed. Nevertheless, in rare instances where birds may be in close proximity to construction noise sources, such as impulse noise from pile driving, such noises may reach high enough levels to cause damage to auditory structures in birds.

3. *Masking of communication signals and other biologically relevant sounds*: Many laboratory masking studies precisely show the effects of continuous noise (including traffic noise) on sound detection in over a dozen species of birds. In a sense, these studies describe a "worst case" scenario because the noise is continuous and the myriad of short-term adaptive behavioral responses for mitigating the effects of noise are not available to the bird in a laboratory test situation. These masking studies led to an overall noise level guideline of around 60 A-weighted decibels (dBA) for continuous noise. A number of things have changed since this 60-dBA criterion was first suggested. Controlled laboratory and field studies have now shown that there are differences among bird species in signal-to-noise ratios at masked threshold. It is also now quite clear that probably all species of birds can use various short-term, adaptive behavioral responses in their natural environments to improve their signal-to-noise ratio. In other words, critical ratios vary across bird species by as much as 10 dB, strongly suggesting that acoustic communication in some species might be affected by an overall traffic and construction noise level of even less than 60 dBA. For some other bird species, communication between individuals, especially if they can employ short-term behavioral strategies for hearing in noise, might be unaffected at even higher levels of noise, perhaps approaching 70 dBA. These short-term behavioral adaptations include scanning (head turning), raising vocal output, and changing singing location. Each of these strategies alone can result in a significant gain in signal level or signal-to-noise ratio (under masking conditions) of about 10 dB, and birds can employ all three strategies simultaneously.

4. *Practical guidelines arising from masking studies*: The following are common sense, practical guidelines that emerge from basic hearing knowledge of birds and humans—specifically, the 6-decibel (dB) difference in masking (critical ratio) functions between typical bird and human listeners with normal hearing. 1) Humans can hear traffic noise, in a natural environment, at twice the distance from the roadway than can birds. In other words, if, in a natural environment, distant traffic noise is barely audible to humans, it is certainly inaudible to birds and will have no effect on any aspect of their acoustic behavior. 2) Humans can hear a bird singing against a background of noise at twice the distance than can the typical bird. This provides an informal estimate of maximum communication distance between two birds vocalizing against a background of continuous traffic noise. This works not only for the typical bird, but it is probably also valid for most species.

Table ES1. Recommended Interim Guidelines for Potential Effects from Different Noise Sources

Noise Source Type	Hearing Damage	TTS	Masking	Potential Behavioral/ Physiological Effects
Single Impulse (e.g., starter's pistol 6" from the ear	140 dBA[1]	NA[3]	NA[5]	Any audible component of traffic and construction noise has the potential of causing behavioral and/or physiological effects independent of any direct effects on the auditory system of PTS, TTS, or masking
Multiple Impulse (e.g., jack hammer, pile driver)	125 dBA[1]	NA[3]	Ambient dBA[6]	
Non-Strike Continuos (e.g., construction noise)	None[2]	93 dBA[4]	Ambient dBA[6]	
Traffic and Construction	None[2]	93 dBA[4]	Ambient dBA[6]	
Alarms (97 dB/100 ft)	None[2]	NA[2]	NA[7]	

TTS temporary threshold shift
dBA A-weighted decibel
PTS – permanent threshold shift
[1] Estimates based on bird data from Hashino et al. (1988) and other impulse noise exposure studies in small mammals.
[2] Noise levels from these sources do not reach levels capable of causing auditory damage and/or permanent threshold shift based on empirical data on hearing loss in birds from the laboratory.
[3] No data available on TTS in birds caused by impulsive sounds.
[4] Estimates based on study of TTS by continuous noise in the budgerigar and similar studies in small mammals.
[5] Cannot have masking to a single impulse.
[6] Conservative estimate based on addition of two uncorrelated noises. Above ambient noise levels, critical ratio data from 14 bird species, well-documented short-term behavioral adaptation strategies, and a background of ambient noise typical of a quiet suburban area would suggest noise guidelines in the range of 50–60 dBA.
[7] Alarms are non-continuous and, therefore, unlikely to cause masking effects.

These recommended guidelines for estimating the effects that traffic noise has on masking in birds are interim guidelines for the following reasons.

1. The interim guidelines are based on median data taken from masking studies done for a limited number of bird species. Thus, they represent the typical bird, based on the species studied. However, it is important to recall that different bird species can differ considerably in how they hear in the presence of noise; some have masked thresholds that approach those of humans, while others have masked thresholds that are 3–4 dB worse than thresholds for the typical bird presented here. Therefore, final noise guidelines will

require testing more species with appropriate experimental adjustment for the species in question.

2. Traffic noise characteristics are influenced by transmission through the environment as are the spectral, temporal, and intensive aspects of bird vocalizations through differences in excess attenuation. In other words, there is inherent variability in estimating the signal-to-noise ratio at the bird's ear in a natural environment. Traffic or construction noise varies from moment to moment. And the level of the signal reaching the receiver's i.e., the bird) ears will vary depending on the location of both the sender and the receiver. Final guidelines will require more data to quantify this variation.

This page left intentionally blank

The Effects of Traffic Noise and Road Construction Noise on Birds

1. Introduction, Overview, Direction

Recent literature on the effects of noise in the environment has shown that the world is becoming a noisier place and that the effects of chronic noise exposure on terrestrial animals, including birds, could be significant (e.g., Barber *et al.*, 2010; Pijanowski *et al.*, 2011a; Pijanowski *et al.*, 2011b; Luther and Magnotti, 2014; Merchant *et al.*, 2015). Furthermore, with population increases and urbanization, traffic and road construction are increasing sources of environmental noise. However, because environmental noise is an inherently complex topic, it is important to define and isolate the sources of variation in determining when noise produced during the construction and operation of roadways has an impact on bird behavior and physiology.

The Endangered Species Act provides additional compelling motivation for understanding the effects of traffic and roadway construction noise on federally listed species. Effects of such noise may be nonexistent in certain circumstances, such as when the sound level of traffic and construction noise is below natural ambient noise levels, and effects may be insignificant in other circumstances, such as when such noise adds very little to existing ambient noise levels. In contrast, construction or traffic noise that adds substantially to natural ambient noise has the potential to produce a suite of significant short- and long-term behavioral and physiological changes in birds. These may include the following changes.

- Changes in the selection of foraging locations.
- Interference with acoustic communications between conspecifics.
- Failure to recognize other important biological signals such as sounds of predators and/or prey.
- Loss of hearing sensitivity temporarily or permanently.
- Increased stress and/or altered steroid hormone levels or other physiological effects.

Any of these effects could have long-term consequences and enduring impacts by interfering with breeding by individuals and populations, thereby threatening the survival of individuals or species.

This Guidance Document represent an updated version of the report entitled *The Effects of Highway Noise on Birds* (Dooling and Popper, 2007) prepared by the current authors. It should be noted that the vast majority of the research literature discussed in this document focuses on effects of traffic noise on birds, and there have been few, if any, studies on effects of roadway construction on birds. This is likely because roadway noise is far more prevalent and continuous than construction noise. Consequently, the models and analysis presented in this document focus on traffic noise.

A. Definitions

Several terms are used in this report. Some of these terms have multiple meanings and are defined herein. Other terms are defined in the glossary.

- **Construction Noise**: Noise produced during the construction of a roadway.
- **Effects**: any response by birds to traffic and construction noise. This definition does not invoke or imply regulatory definitions of "effect" as found in any law or regulation affecting birds.
- **Roadway**: Any paved road on which there is vehicular traffic.
- **Traffic Noise**: Noise produced by vehicles on any paved roadway, ranging from highways to single-lane streets.

B. Organization and Purpose of This Guidance Document

Sections 2 and 3 of this Guidance Document discuss bird audition, including how and what birds hear and how environmental noise can generally affect the auditory system and hearing. This is followed by Section 4, which discusses the effects of traffic and construction noise on birds, the challenges in surveying what is known about the effects of traffic and construction noise on birds, and the scientific literature on the topic. Section 5 summarizes the different classes of effects of noise on birds. Finally, Section 6 poses a first set of *interim* criteria to protect birds from traffic and construction noise. For readers interested in additional information, Appendix D discusses fundamentals of traffic noise (prepared by ICF Jones and Stokes), Appendix E presents a review of the older literature from the 2007 report, and Appendix F describes recommendations for critical future research that the authors suggest would enhance overall understanding of effects of traffic noise on birds.

The purpose of this Guidance Document is two-fold. First, it critically discusses what is known about the effects of highway construction and traffic noise on birds, with emphasis on the best available science. Generally, the reviewed literature has been directed at assessing and mitigating the impacts of noise produced by highway construction and operation on birds. This Guidance Document shows that there are still major gaps in this body of literature and very few firm conclusions, although there has been a substantial increase in knowledge since the first report (Dooling and Popper, 2007). As a Guidance Document should always reflect recent changes in the science, Appendix F points to areas for future research that would substantially enhance our future understanding of traffic noise on birds.

Second, this Guidance Document suggests *interim* compliance guidelines and a science-based approach, using human and avian data from both the laboratory and the field, to address potential impacts of noise on bird species. In areas such as hearing and masking of sounds as a result of noise, rigorous data are available from a wide range of species so that it is reasonable to extrapolate the effects on federally listed species. Such guidelines are done in coordination and consultation with compliance protocols for the federal Endangered Species Act.

C. Analysis of United States Fish and Wildlife Service (2006) Report

On July 26, 2006, the Arcata Fish and Wildlife Service Office (AFWO) of the U. S. Fish and Wildlife Service (FWS) issued guidance for estimating the effects of auditory and visual disturbance to northern spotted owls (*Strix occidentalis caurina*) and marbled murrelets (*Brachyramphu marmoratus*) in Northwestern California (AFWO, 2006).[2] These two species live

[2] http://goo.gl/3FLFCA

a rather solitary lifestyle and are expected to be particularly sensitive to noise disturbance. The purpose of the FWS guidance was to promote consistent and reasonable determinations of potential effects on either species that could result from elevated human-generated sounds or human activities in close proximity to nests during the breeding season. FWS acknowledged that its report is to be viewed as a living document subject to continued, ongoing revision, and improvement as additional data and experience are acquired.

The FWS document provides excellent guidance as to how a person in the field should make determinations with regard to the potential effects of construction and traffic noise on these two avian species, especially with regard to harassment.[3] This guidance is particularly valuable because it takes into consideration critical variables and tries to integrate them into a simple practical model. These variables include those listed below.

- Types of sound sources.
- Distances from the sound sources to the birds.
- Level of ambient noise in the environment.
- Levels of anthropogenic (human-generated) noise in the environment.
- Sound-modifying features in the environment.
- Visual cues correlated with the noise.
- The hearing sensitivity of the bird.

The FWS report provides a worthwhile potential strategy for estimating particular kinds of noise effects on these birds; however, the report has several limitations in terms of its applicability to other species. First, it is based on two relatively non-social species and does not address the kinds of effects that may be relevant for more gregarious species that flock and engage in continuous vocal communication with conspecifics.

Second, as discussed below, there are substantial differences between species in the ability to hear in noisy environments. As a consequence, one noise level is not likely to affect all species in the same way since some species will hear a particular level of sound and others will not due to their overall hearing sensitivity.

Third, how a bird responds to and integrates acoustic and visual stimuli in different contexts (e.g., breeding season or brooding) is likely to have a profound effect on whether harassment occurs. For example, very low level sounds bearing some resemblance to the sounds of a natural predator are likely to be far more important to the bird than other sounds of equal sound level but with no history of signaling danger. Such experiential factors will undoubtedly vary significantly by species.

Finally, the noise levels discussed in the FWS guidance are geared toward those that result in harassment or flushing from the roost or nest. There are other effects, such as masking of communication signals, that are also very important for species that must learn their vocalizations

[3] The Act's implementing regulations further define harass as "... an intentional or negligent act or omission which creates the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavioral patterns which include, but are not limited to, breeding, feeding or sheltering" [50 CFR §17.3]. (Taken verbatim from p.4 of FWS (2006) report.)

and are engaged in continuous vocal communication with conspecifics throughout their lifetime, that are not considered in the FWS document.

Despite these caveats, the FWS report, together with information reviewed in this Guidance Document, may have value in helping reach a decision metric on possible effects of traffic and construction noise on birds. Moreover, the specific recommendations made in the FWS guidance report, while not fully applicable to situations involving continuous traffic and construction noise, represent a thoughtful approach to identifying and quantifying some of major variables for consideration.

D. Literature Surveyed in this Guidance Document

The material presented in this Guidance Document is based on a careful evaluation of technical reports and peer-reviewed articles, much of which is discussed in Section 4. The scientific approach and analysis used in each study differs, and so extrapolation between the studies, and especially those done in different locations or by different groups of investigators, is difficult and must be done with considerable caution.

In addition to primary peer-reviewed literature, this Guidance Document also cites a number of reviews covering various aspects of the issues considered here. These reviews, even if they have gone through appropriate peer review, often reflect the opinions and biases of the authors based on their analysis of the original material from peer-reviewed research articles.

Finally, wherever possible, this Guidance Document incorporates new material that has been produced since the authors' original review (Dooling and Popper, 2007). Taken together, the previously reviewed literature (see Appendix E) and the more recent literature significantly inform the conclusions and recommendations in this Guidance Document.

E. Metrics and Terminology

This Guidance Document contains a number of acoustic and biological terms. To facilitate understanding of terminology, most of the terms are defined in the glossary in Appendix A. Appendix D discusses fundamentals of traffic noise.[4] Those unfamiliar with fundamental concepts relating to traffic noise are advised to review information published by the California Department of Transportation (Caltrans) on the topic of highway traffic noise. This includes the Caltrans Traffic Noise Analysis Protocol (Protocol) (Caltrans, 2011),[5] the Technical Noise Supplement to Protocol (Caltrans 2013), and Caltrans online noise training.[6]

It is also important to define what is meant by "behavior" in this Guidance Document because the word is used for a wide range of activities, and usage also varies between different authors. For example, the term may be used to refer to the complex interaction of signals and rituals that animals use during mating or may also be used to refer to the movements of animals from one feeding

[4] Material in Appendix D was prepared by Caltrans and not by the authors of this report.
[5] http://www.dot.ca.gov/hq/env/noise/pub/ca_tnap_may2011.pdf
[6] http://www.dot.ca.gov/hq/env/noise/training_license.htm.

ground to another. In the context of this Guidance Document, "behavior" is used in its broadest possible sense unless otherwise qualified

F. Typical Roadway Operational and Construction Noise Levels

Traffic noise produced by vehicles traveling on a highway is a function of the traffic volume, vehicle mix, vehicle speed, and pavement type. For example, Table 1 summarizes typical traffic conditions for several typical highway configurations.

Table 1. Typical Highway Conditions

Number of Lanes	Highway Type	Worst Hour Traffic Volume	Speed	Heavy Truck %[1]
2	Highway	3,000	55 mph	2%
4	Highway	6,000	65 mph	2%
6	Freeway	12,000	65 mph	6%
8	Freeway	16,000	65 mph	8%

[1] Truck percentages can vary widely depending on the proximity of a roadway to commercial uses and truck routes. The truck percentages shown here are generally conservative for the roadway construction shown.

A considerable amount of work has enabled traffic engineers to model noise levels expected under various traffic conditions, road types, and vehicle speeds. Figure 1 shows traffic noise levels at various distances (in feet) from the roadway as predicted by the Federal Highway Administration (FHWA) Traffic Noise Model[7] (TNM) version 2.5 for each traffic condition in Table 1. Neutral atmospheric conditions (no inversion, moderate temperature, and wind speed less than 11 miles per hour [mph]) and soft ground surface (lawn) assumptions as recommended by FHWA were used. Additional assumptions included that the roadway was undivided, had no median lanes, was the typical 12 foot (3.6 meters) wide, and had average pavement, dry conditions, and moderate temperatures, with wind speed below 11 mph (17.7 kilometers per hour [km/h]).

With multiple lanes and a large number of vehicles, free-flowing traffic on a roadway acts like a line source. Geometric attenuation for a line source is 3 dB per doubling of distance. Additional attenuation resulting from ground absorption can add attenuation of about 1.5 dB per doubling of distance. Excess attenuation from ground effects, atmospheric absorption, wind, and temperature gradient effects, etc., are highly complex and can add attenuation over 5–10 dB per 100 m depending on the environment (e.g., Marten and Marler, 1977).

In contrast to the continuous noise produced by large volumes of traffic, noise produced by construction equipment is likely to be intermittent and impulsive (with very short rise-times), such as impact noise from a pile driver. Noise produced by construction equipment is a function of the type of equipment. Table 2 summarizes typical maximum noise levels at 50 feet (15.2 m) produced by typical construction equipment (see FHWA, 2006)[8]). In contrast to traffic noise, equipment used in roadway construction acts like a point source and will typically off at a rate of 6 dB per doubling of distance, although there is also likely to be additional attenuation that varies with the environment. Moreover, these are maximum noise levels which are not typically sustained over

[7] http://www.fhwa.dot.gov/environment/noise/construction_noise/rcnm/rcnm.pdf
[8] http://www.fhwa.dot.gov/environment/noise/construction_noise/rcnm/rcnm.pdf

long periods of time. Energy average sound levels can be developed based on utilization factors (FHWA, 2006).

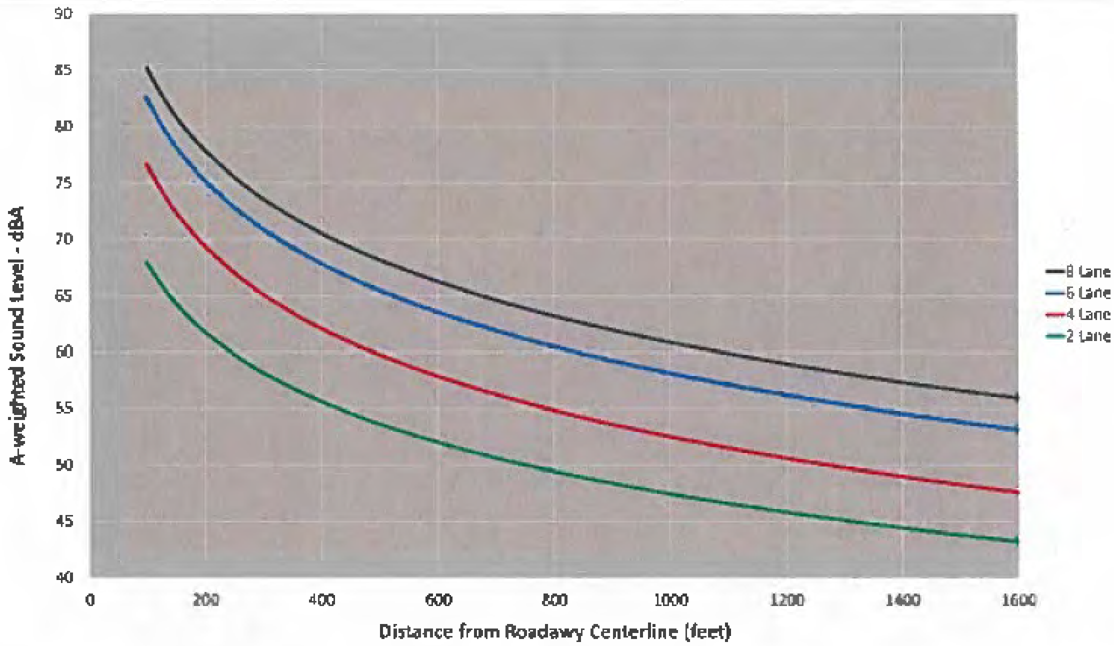


Figure 1. Typical Roadway Noise Levels as a Function of Distance
Data based on traffic conditions listed in Table 1

.

G. Relation between A-Weighted Sound Level in and Spectrum Level[9]

The noise levels described in Section 1.F for both traffic noise and construction noise are given in dBA[10] (see Appendix G for discussion of history of dBA for bird studies). The dBA scale for measuring sound levels takes into account the equal loudness contours of human hearing—that sounds at low frequencies and high frequencies presented at the same sound pressure level as intermediate frequencies are judged as softer than the sounds at intermediate frequencies. This scale is incorporated in most sound level meters and is thus convenient for the person doing the measurements. It may not always the most accurate measure for determining the effects of noise on bird hearing, however, because birds are even less sensitive to sound below 1 kHz than are humans, and birds have extremely poor hearing at frequencies about 10 kHz. Thus, the most relevant measure of noise for estimating the masking effects of noise on bird hearing is the spectrum level (the intensity level of a sound within a 1 hertz (Hz) band) in the frequency region where birds vocalize most and hear best—typically around 2–5 kHz.

Traffic noise and non-impact construction noise often show a sloping spectrum (Figure 2) with less energy in the region of 2–4 kHz than at lower frequencies. Thus, estimating the spectrum level

[9] Note that this Guidance Document does not include a direct discussion of the idea of 60 dBA that has been found in much of the earlier literature. A history of the use of 60 dBA is found in Appendix G.
[10] For a detailed discussion of dBA see: https://en.wikipedia.org/wiki/A-weighting

in the region of 2–4 kHz from an overall dBA level could overestimate the energy in the region of 2–4 kHz. On the other hand, traffic noise still has a considerable amount of energy around 1 kHz, and this band of energy contributes significantly to the overall dBA level actually resulting in a significant underestimate of the noise level actually in the 2–4 kHz bands that contain most bird vocalizations. Thus, in many cases, the overall level of the noise measured as dBA does not provide an accurate estimate of the noise level in the frequency region where birds communicate. Depending on the overall spectrum of the noise, it could underestimate, or more often overestimate, the masking effects of traffic noise on hearing and vocal communication in birds. In Figure 2, for instance, the overall level of noise is 84 dB (83 dB measured on the A scale) and this value is almost entirely accounted for by the energy in the octave band around 1 kHz. The level of noise in the frequency region that birds use for acoustic communication is much less, at around 60–65 dB.

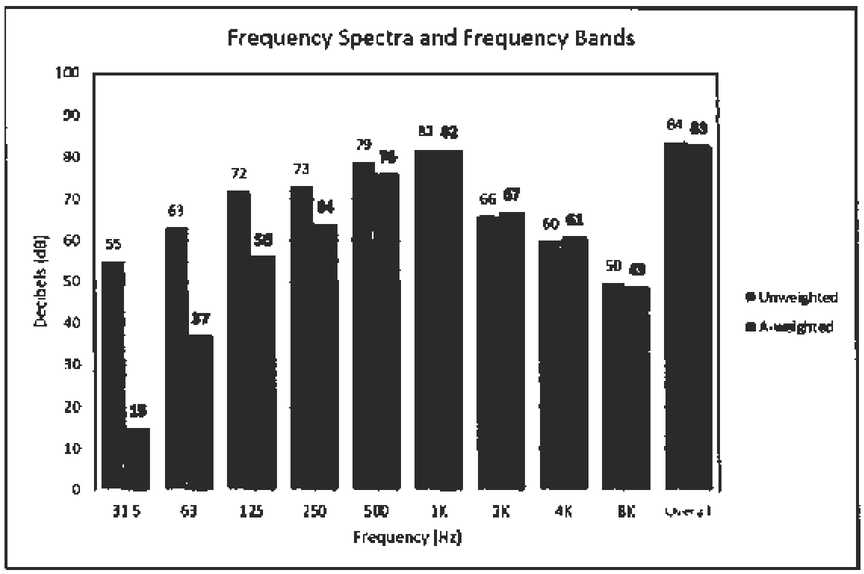


Figure 2: Caltrans Traffic Noise Spectra Showing Differences in Unweighted and Weighted Spectra and Overall Levels[11]

For traffic and construction noises, measuring overall sound levels in in dBA is likely to overestimate the effects of traffic and construction noise on communication in birds. A more accurate estimate would be obtained with measures of the sound pressure level in the octave bands at 2 kHz and 4 kHz From these two measurements, given the characteristics of traffic and construction noise, reasonably accurate estimates of spectrum levels can be obtained for the critical frequency range in which birds communicate and from these spectrum levels, decisions can be made about whether the noise will interfere with vocal communication. At 2.0 kHz, the spectrum level is roughly 33 dB less than the octave band level; at 4.0 kHz, the spectrum level is about 36 dB less than the octave band level.

2. The Bird Ear and Hearing

[11] Figure from: http://www.dot.ca.gov/hq/env/noise/online_training_module1/slides/slide50.htm

In order to appreciate the potential effects of traffic and construction noise on bird hearing, it is important to have some understanding of the bird ear and the basic hearing capabilities of birds both in quiet and in high noise settings (Dooling *et al.*, 2000a). It is also worthwhile to appreciate why birds, or any animals (including humans) hear, and why hearing may have evolved. In the case of many animals, especially birds and humans, hearing is closely related to acoustic communication (Dooling, 1982; Dooling *et al.*, 1992). Indeed, birds, more than most any vertebrate group other than primates, make use of a rich array of sounds for communicating, finding mates, expressing territorial occupation, and numerous other social behaviors.

Table 2: Construction Equipment Noise Emission Levels (greatest-to-least)[12]

Equipment	Typical L_{max} at 50 feet (15.2 m) from Source (dBA, Slow)
Pile Driver (Impact)	95
Vibratory Pile Driver	95
Rock Drill	85
Paver	85
Scraper	85
Crane	85
Jack Hammer	85
Concrete Mixer Truck	85
Dozer	85
Grader	85
Jackhammer	85
Pneumatic Tool	85
Crane	85
Chain Saw	85
Roller	85
Tractor	84
Concrete Pump Truck	82
Generator	82
Compactor (ground)	80
Compressor (Air)	80
Backhoe	80
Vibratory Concrete Mixer	80
Pumps	77

Source: Federal Highway Administration 2006. Table 1. http://goo.gl/PXltyy

Birds, as with humans and other animals, also use hearing to learn about their overall environments. Bregman (1990) refers to this as the "acoustic scene." This acoustic scene is the array of sounds in the environment, not just vocalizations, which may arise from biological or non-biological sources, such as predators moving through the environment or the wind moving through trees. This acoustic scene covers an area all around an animal, and it is just as rich at night as

[12] http://www.fhwa.dot.gov/environment/noise/construction_noise/rcnm/rcnm00.cfm

during the day when animals can use vision. The acoustic scene tells an animal a great deal about its extended environment. So, while this Guidance Document focus on the effect of noise on communication signals, it is important to also realize that other aspects of the animal's acoustic scene are also affected.

The bird ear and bird hearing has been well described over the years (e.g., Dooling *et al.*, 2000a; Gleich and Manley, 2000; Saunders *et al.*, 2000; Saunders and Henry, 2014). It consists of an external membrane (tympanic membrane), a middle ear (Saunders *et al.*, 2000; Saunders and Henry, 2014), and an inner ear (Gleich and Manley, 2000; Saunders and Henry, 2014). There is no external structure that resembles the mammalian outer ear flap, or pinna (except in owls). Instead, the tympanic membrane is the outermost covering of the middle ear.

The avian inner ear is similar to that of most vertebrates in that is has three semicircular canals to determine angular acceleration of the head and three otolith organs to detect motions of the head relative to gravity. In addition, birds have a cochlear duct that contains a basilar papilla upon which sit the sensitive sensory hair cells used for hearing. However, the basilar papilla is shorter and rather different in structure than that found in mammals (Tanaka and Smith, 1978; Smith, 1985; Gleich and Manley, 2000; Manley, 2000) and the differences may, to a degree, account for the much narrower range of frequencies detected by birds as compared to mammals.

Another factor that probably limits the frequency range over which birds hear is the presence of a single-bone middle ear rather than the three-bone middle ears (malleus, incus, stapes) that are characteristic of mammals (Manley, 2010). It has been suggested that the single columella in place of the three ear bones found in mammals is what limits hearing in most avian species to not much more than 10 kHz (Saunders *et al.*, 2000; Manley, 2010).

A. Behavioral Measures of Avian Hearing the Audiogram

The minimum sound pressure that can be detected at frequencies throughout an animal's range of hearing defines the audiogram, or audibility curve.[13] This is the most basic measure of hearing and one most people are familiar with from having their own hearing tested. Over the past 50 years, behavioral audibility curves have been collected for about 39 species of birds, and this database can be extended by another 10 species of birds by including data from physiological recordings (Appendix B, also see Fay, 1988). These data are fit with a polynomial function to provide a continuous curve describing the minimum audible sound pressure over the range of hearing for a particular species.

Figure 3 shows the median audiogram based on the species in Appendix B. For animals, and sometimes for humans, the audiogram is measured in a sound attenuated room (an audiometric test chamber) so that the background noise is minimized and there is no interference by other sounds (i.e., masking). Thus the audiogram represents an ideal detection threshold that is rarely, if ever, attained in the real world, which always has some measurable amount of background noise.

[13] This is a measure of hearing "threshold." It should be noted that the threshold (the lowest sound detectable at a given frequency) is not a fixed value. There are slight variations from animal to animal and larger differences across species. Testing conditions and context can also play a role. Typically, the "threshold" is a statistical measure indicating the lowest sound pressure level that an animal can detect 50% of the time.

Audiograms are often described and compared on several features, such as the softest sound that can be heard (often referred to as best sensitivity or lowest intensity), the frequency at which hearing is best (best frequency—the frequency at which the subject can hear the softest sound), the bandwidth (the width of the audiogram to the point where it is raised by 30 dB on either side of the best frequency), lowest intensity (at the best frequency), and the low and high frequency limits of hearing (the frequencies at which thresholds are 30 dB above the best intensity) for both birds and humans. Interestingly, compared to species in other vertebrate groups, there is not wide variation in hearing sensitivity between different bird species. This suggests that the recommendations in this Guidance Document apply to most birds.

Generally, birds hear best at frequencies between about 1 and 5 kHz (Figure 3), with absolute (best) sensitivity often approaching 0–10 dB SPL[14] at the most sensitive frequency, which is usually in the region of 2–4 kHz (Dooling, 1980; 1982; 1992; Dooling *et al.*, 2000b). Nocturnal predators, such as most owls, can generally detect much softer sounds than can either Passeriformes (e.g., songbirds, such as sparrows, canaries, starlings, finches) or other non-Passeriformes (e.g., chickens, turkeys, pigeons, parrots, owls) over their entire range of hearing, sometimes with levels as low as -10 to -15 dB SPL. Passeriformes also tend to have better hearing at high frequencies than non-Passeriformes, while non-Passeriformes can detect softer signals at low frequencies than do Passeriformes. This difference is usually on the order of 5 to 10 dB. A recent correlative study of hearing characteristics (using the database in Appendix B) with several biological parameters confirms significant correlations among body weight, inner ear anatomy, and low- and high-frequency hearing in birds, with the exception of owls (Gleich *et al.*, 2005). Simply put, large birds hear better at low frequencies and small birds hear better at high frequencies. On average, however, the frequency range available to the typical bird for long distance vocal communication extends, at best, from about 1 to 4 kHz, the region of best sensitivity.

B. The Hearing Range and Vocalization Spectrum of Birds

Almost all avian species rely heavily on acoustic communication for species and individual recognition, mate selection, territorial defense, and other social activities. Studies of bird hearing have long shown a strong correlation between the range of hearing in birds and the frequency spectrum of bird vocalizations (Konishi, 1969; Dooling, 1980; 1982). That is, with the exception of some nocturnal predators such as barn owls, birds typically hear best in the spectral region of their species-specific vocalizations. Barn owls hear better at higher frequencies than do most other bird species because they have evolved to use high frequency cues to localize their prey in darkness. The importance of the general observation of a close match between hearing thresholds and vocalizations is that concerns over the effects of masking or hearing damage from noise should focus attention on the critical frequency region of about 1–6 kHz—the spectral region used for acoustic communication in birds (Dooling, 1982).

[14] SPL, or sound pressure level, is a widely used expression of the sound pressure using the decibel (dB) scale and the standard reference pressures 20 µPa for air.

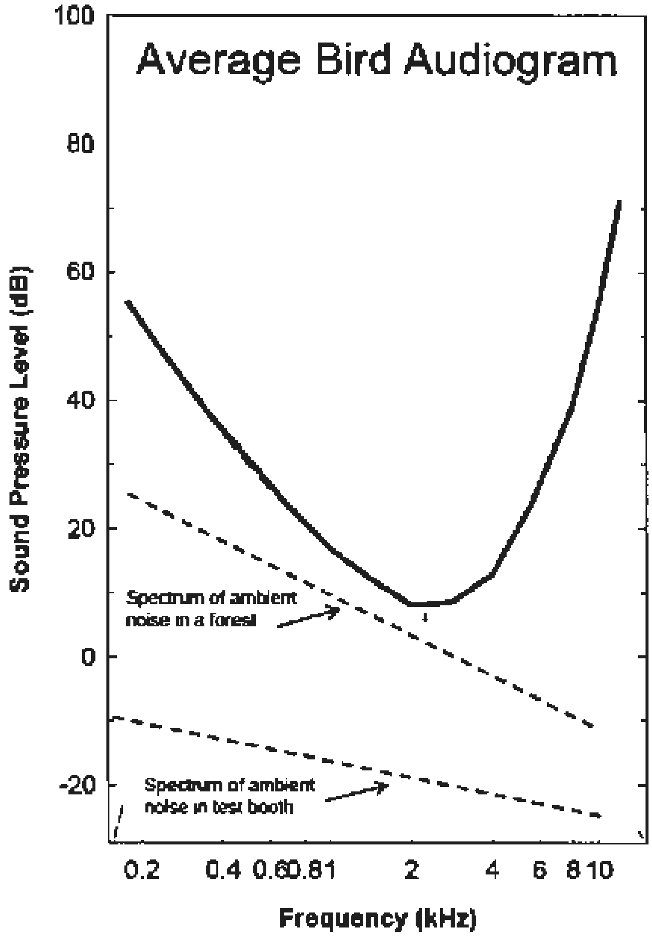


Figure 3: Bird Hearing Thresholds

Median bird hearing thresholds from 49 bird species (Appendix B measured behaviorally and physiologically in the free field in the quiet (solid line). The typical bird hears less well than humans and over a narrower bandwidth. Dotted lines show typical spectrum levels of the background noise in a double-walled acoustic isolation testing chamber and the spectrum level of ambient noise that a bird might encounter in a typical forest environment. An ambient noise spectrum level at least 20 dB below the audiogram will have no effect on hearing thresholds (i.e., no masking). An ambient noise level less than 20 dB below the audiogram thresholds, which is the case in almost all natural environments, will raise the animal's thresholds (i.e., cause masking).

C. The Hearing Capabilities of Nestlings

Less is known about hearing in nestlings and young birds as compared to sexually mature birds. However, a limited amount of data from young songbirds and parrots suggest that the auditory system of altricial birds (i.e., birds that are in an undeveloped stage at hatching in the nest and require care and feeding from parents[15]) does not function well at hatching. Auditory Brainstem Response (ABR, a type of physiological recording) studies of budgerigars[16] (*Melopsittacus undulatus*) and canaries (*Serinus canaria domestica*) indicate that hearing thresholds during the

[15] Altricial birds include all Passeriformes (songbirds). Altricial birds hatch with their eyes closed and with few, if any, feathers. In contract, precocial birds hatch with eyes open and are generally ready to leave the nest within two days of hatching—see: http://www.stanford.edu/group/stanfordbirds/text/essays/Precocial_and_Altricial.html

[16] Also known as a parakeet.

first two weeks after hatching of altricial birds are 30–40 dB higher than hearing thresholds of adults By the time nestlings are 20–30 days old and just getting ready to leave the nest; however, hearing thresholds as measured by the ABR approach adult levels of sensitivity (Brittan-Powell and Dooling, 2004).

Hearing thresholds in young birds and nestlings in the presence of noise have not yet been measured. While it is unlikely that nestlings can hear better in noise than adults, the fact that this is a critical stage in vocal development means that any additional noise, as from construction or traffic, may affect a bird's ability to acquire and develop its species-typical vocalizations. Recent laboratory work in zebra finches has now confirmed this suspicion (Potvin and MacDougall-Shackleton, 2015).

3. General Principles of the Effects of Noise on Birds

There are four general overlapping categories of construction and traffic noise effects on birds: permanent threshold shift (PTS permanent hearing loss), temporary threshold shift (TTS—temporary hearing loss which recovers over a period of minutes to days from the end of noise exposure), masking, and other physiological and behavioral responses. The actual auditory effect that is encountered depends upon the level of noise arriving at the bird's ear, which is highly correlated with the proximity of the bird(s) to the noise source (Figure 4, Table 3).The existing scientific literature provides a considerable amount of data that can be used to define the boundaries between these categories of effects e.g., Dooling *et al.*, 2008; Salvi *et al.*, 2008; Saunders and Salvi, 2008).
Based on Figure 4, it is possible to generalize on the potential effects of highway and construction noise on birds, depending on their distance from the source. The distance of each zone is arbitrary and depends on the level of the source. Thus, if the level of the source is very high, each zone will be large, whereas if the sound level at the source is low, the distances between the zones will be smaller. Regardless, as is shown, these zones no doubt overlap with regard to potential effects.

a. Zone 1: If a bird is in this region, it is close to the noise source such that traffic and construction noise can *potentially* result in all four effects—permanent threshold shift, temporary threshold shift, masking, and other behavioral and/or physiological effects. Laboratory evidence shows that continuous noise levels above 110 dBA SPL lasting over 12–24 hours, or a single impulsive noise over 140 dB SPL (125 dB SPL for multiple blasts), can cause damage and loss of inner ear sensory hair cells resulting in a large initial threshold shift, followed by a small (~10–15 dB) lingering threshold shift even after all hair cells have been regenerated (Saunders and Dooling, 1974; Dooling and Saunders, 1975; Dooling *et al.*, 2008).

b. Zone 2: At greater distances from the roadway, starting where the received noise levels fall below 110 dBA continuous exposure, hearing loss and permanent threshold shift are unlikely to occur. However, continuous traffic and construction noise above 93 dBA SPL might still temporarily elevate a bird's threshold, mask important communication signals, and possibly lead to other behavioral and/or physiological effects.

c. Zone 3: At even greater distances from the roadway, where the spectrum level of the noise is still at or above the natural ambient noise level, masking of communication signals from this added noise may occur. This, in turn, may also result in other behavioral and/or physiological effects.

d. Zone 4: Once the level of traffic and construction noise falls below ambient noise levels in the critical frequencies for communication, masking of communication signals is no longer an issue. However, faintly heard sounds, such as the low rumble of a truck, or an alarm from a construction site, may still lead to a chronic state of increased arousal and, thus, lead to other behavioral and/or physiological effects.

e. Beyond Zone 4: At this boundary, the energy in traffic noise and construction noise at all frequencies is completely inaudible (i.e., falls below the level of the ambient noise). The bird cannot hear this noise and, thus, the noise has no effects of any kind on the bird.

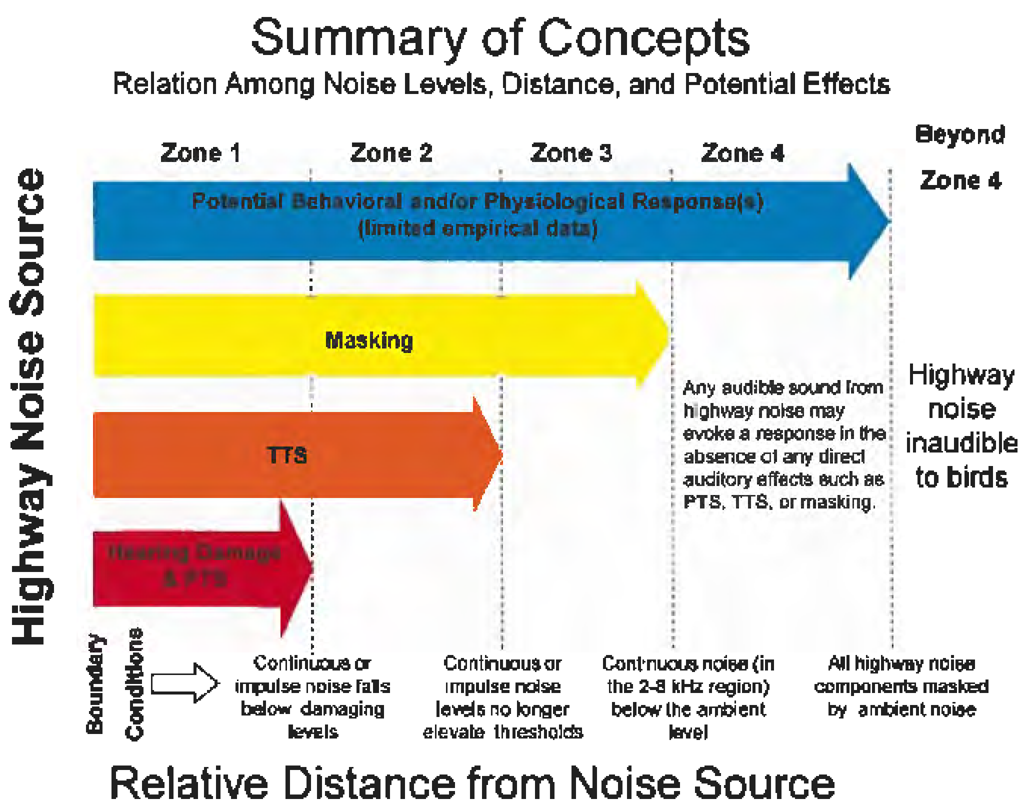


Figure 4: Potential Effects of Traffic and Construction Noise on Birds
Categories of traffic and construction noise effects on birds with distance from the source. Zone 1 is closest to the source while Zone 4 is furthest away. Sound level decreases further from the source. Note that the actual distances for the Zones ae not given since that would depend on the source sound level, hearing sensitivity of the receiver, and the propagation distance from the source to the receiver. See text for detailed discussion.

Based on Figure 4, it is possible to generalize on the potential effects of highway and construction noise on birds, depending on their distance from the source. The distance of each zone is arbitrary and depends on the level of the source. Thus, if the level of the source is very high, each zone will be large, whereas if the sound level at the source is low, the distances between the zones will be smaller. Regardless, as is shown, these zones no doubt overlap with regard to potential effects.

a. Zone 1: If a bird is in this region, it is close to the noise source such that traffic and construction noise can *potentially* result in all four effects permanent threshold shift, temporary threshold shift, masking, and other behavioral and/or physiological effects. Laboratory evidence shows that continuous noise levels above 110 dBA SPL lasting over 12–24 hours, or a single impulsive noise over 140 dB SPL (125 dB SPL for multiple blasts), can cause damage and loss of inner ear sensory hair cells resulting in a large initial threshold shift, followed by a small (~10–15 dB) lingering threshold shift even after all hair cells have been regenerated (Saunders and Dooling, 1974; Dooling and Saunders, 1975; Dooling *et al.*, 2008).

b. Zone 2: At greater distances from the roadway, starting where the received noise levels fall below 110 dBA continuous exposure, hearing loss and permanent threshold shift are unlikely to occur. However, continuous traffic and construction noise above 93 dBA SPL might still temporarily elevate a bird's threshold, mask important communication signals, and possibly lead to other behavioral and/or physiological effects.

c. Zone 3: At even greater distances from the roadway, where the spectrum level of the noise is still at or above the natural ambient noise level, masking of communication signals from this added noise may occur. This, in turn, may also result in other behavioral and/or physiological effects.

d. Zone 4: Once the level of traffic and construction noise falls below ambient noise levels in the critical frequencies for communication, masking of communication signals is no longer an issue. However, faintly heard sounds, such as the low rumble of a truck, or an alarm from a construction site, may still lead to a chronic state of increased arousal and, thus, lead to other behavioral and/or physiological effects.

e. Beyond Zone 4: At this boundary, the energy in traffic noise and construction noise at all frequencies is completely inaudible (i.e., falls below the level of the ambient noise). The bird cannot hear this noise and, thus, the noise has no effects of any kind on the bird.

Before considering the effects on the auditory system of birds from traffic and construction noise, it is important to understand three facts about potential behavioral and physiological effects of traffic and construction noise. One is that these effects can occur alone or in combination with effects on the auditory system of birds. Second, behavioral and physiological effects may be less dependent on noise level and more dependent on environmental context and the salience of the traffic and construction noise component(s) to the bird. Third, in contrast to the effects of noise on the bird auditory system, there are fewer empirical data available on behavioral and physiological effects, and especially for those effects that occur alone, as in Zone

Table 3: Recommended Interim Guidelines for Potential Effects from Different Noise Sources

Noise Source Type	Hearing Damage	TTS	Masking	Potential Behavioral/ Physiological Effects
Single Impulse (e.g., starter's pistol 6" from the ear	140 dBA[1]	NA[3]	NA[5]	Any audible component of traffic and construction noise has the potential of causing behavioral and/or physiological effects independent of any direct effects on the auditory system of PTS, TTS, or masking
Multiple Impulse (e.g., jack hammer, pile driver)	125 dBA[1]	NA[3]	ambient dBA[6]	
Non-Strike Continuos (e.g., construction noise)	None[2]	93 dBA[4]	ambient dBA[6]	
Traffic and Construction Noise	None[2]	93 dBA[4]	ambient dBA[6]	
Alarms (97 dB/100 ft)	None[2]	NA[2]	NA[6]	

[1] Estimates based on bird data from Hashino et al. (1988) and other impulse noise exposure studies in small mammals.

[2] Noise levels from these sources do not reach levels capable of causing auditory damage and/or permanent threshold shift based on empirical data on hearing loss in birds from the laboratory.

[3] No data available on TTS in birds caused by impulsive sounds.

[4] Estimates based on study of TTS by continuous noise in the budgerigar and similar studies in small mammals.

[5] Cannot have masking to a single impulse.

[6] Conservative estimate based on addition of two uncorrelated noises. Above ambient noise levels, critical ratio data from 14 bird species, well documented short term behavioral adaptation strategies, and a background of ambient noise typical of a quiet suburban area would suggest noise guidelines in the range of 50–60 dBA.

[7] Alarms are non-continuous and therefore unlikely to cause masking effects.

A. Effects of Noise on Hearing in Birds Threshold Shift

Birds (as well as humans and other animals) show a shift in hearing sensitivity in response to sounds that are sufficiently long and/or intense. There are several recent reviews of the effects of trauma to the auditory system of birds (Dooling *et al.*, 2008; Salvi *et al.*, 2008; Saunders and Salvi, 2008). Taken together, the data show that birds can tolerate continuous (i.e., up to 72 hours) exposure to noises of up to received levels of 110 dBA without experiencing hearing damage or a significant permanent threshold shift.

Permanent Threshold Shift: A PTS occurs if the intensity and duration of the noise is sufficient to damage or kill the inner ear sensory hair cells or other structures in the inner ear. In birds, thee specific damage to sensory hair cells depends on the type, intensity, and duration of the acoustic trauma (reviewed in Cotanche, 1998). Since hearing depends on the function of these hair cells, their permanent loss in mammals, including humans, results in permanent hearing loss. However, since birds can regenerate damaged or destroyed sensory hair cells usually within a month, there can be substantial recovery of hearing, although there is often still a small, insignificant 10 dB threshold shift that remains permanent (Dooling and Saunders, 1974; Saunders and Dooling, 1974).

A number of comparative studies on hearing loss in birds are instructive in understanding important sources of variation on the effects of sound exposure on birds. For example, Japanese quail (*Coturnix coturnix japonica*) exposed to a 1.5 kHz octave band noise at 116 dB SPL for four hours showed hearing loss of up to 50 dB immediately following exposure (Niemiec *et al.*, 1994). Hearing loss was most severe at frequencies at and above 1.0 kHz, although there was considerable variation between subjects. Hearing loss was accompanied by a significant loss of sensory hair cells in the basilar papilla. Nevertheless, hearing improved rapidly within the first week following exposure, and recovered to pre-exposure levels within 8–10 days. Damaged hair cells were observed up to 2 weeks post exposure, but there was little evidence of damage to hair cells at 5 weeks post-exposure. Similar patterns of threshold shifts and recoveries were seen after repeated exposures to noise, although recovery times increased with increasing exposure duration. The authors found there can be a return to normal sensitivity prior to complete regeneration of the sensory hair cells (Bennett *et al.*, 1994) suggesting birds do not need a full complement of hair cells for normal hearing.

Ryals and colleagues (1999) found that the amount of hearing loss and the time course of recovery varied considerably among different bird species, even with identical exposure and test conditions. In one study, Japanese quail and budgerigars were exposed to pure tones of 112–118 dB SPL for 12 hours, with the frequency of the sounds centered in the region of best hearing of each species. Quail showed much greater susceptibility to acoustic trauma than did budgerigars, and showed significantly larger threshold shifts and hair cell loss. Quail showed a threshold shift of 70 dB at 2.86 kHz at one day following over-exposure, and this hearing loss remained virtually unchanged for 8–9 days after exposure. Hearing began to improve by about 1 dB/day until recovery at day 50, at which time recovery reached asymptote. This left the quail with a permanent threshold shift of approximately 20 dB, which remained even 1 year following exposure. In contrast, budgerigars showed a threshold shift of about 35–40 dB and a much faster recovery than the quail. By thee days after exposure, budgerigars' thresholds had improved to within 10 dB of normal. In human hearing, elevated thresholds of 10 dB are still considered within the normal range.

In another experiment, budgerigars, canaries, and zebra finches were exposed to the same band pass noise (2–6 kHz) at 120 dBA SPL for 24 hours. Thresholds at 1.0 kHz were initially elevated by 10–30 dB but returned to within normal limits by about 10 days after exposure in all three species. Moreover, at 2.86 kHz, the center of the exposure band, all three species showed a 50 dB threshold shift. Recovery began immediately after the noise was terminated for canaries, while zebra finches recovered to within 10 dB of normal by about 30 days after exposure. However, thresholds remained elevated for 10 days before recovery begin to occur in budgerigars. By 50 days after exposure, thresholds for budgerigars still only recovered to about 20 dB above normal. Thus, in this experiment, there was significantly more rapid recovery in canaries and zebra finches than in budgerigars.

These comparative studies, and especially those by Ryals and her colleagues (Ryals and Rubel, 1985a, b; Ryals *et al.*, 1999), are important for understanding the effects of intense noise on hearing in birds. The Ryals et al. (1999) study showed that different species, tested under identical noise exposure and test conditions, all showed resistance to hearing damage from noise. In addition, these studies show that there is considerable variation among species in the amount of damage and the time-course of loss and recovery from acoustic trauma. Thus, concern over the effects of loud sounds on the ear and hearing is quite reasonable (McFadden and Saunders, 1989; Saunders *et al.*, 1991; Adler *et al.*, 1992; Adler *et al.*, 1993; Pugliano *et al.*, 1993; Saunders and Salvi, 1993). These studies suggest that, for birds, permanent hearing loss from traffic noise or construction noise is probably not a significant concern.

Temporary Threshold Shift: At continuous noise levels below 110 dBA down to about 93 dBA, birds may experience a temporary threshold shift (TTS) which lasts from seconds to days, depending on the intensity and duration of the noise to which the animal was exposed. In contrast to a PTS, hearing recovers completely from TTS to the level that it was before the exposure. Nevertheless, during this period of TTS the bird's hearing is temporarily impaired and this could affect a variety of auditory and vocal communication behaviors, including detection of predators, communication with young, auditory feedback, etc. There have been a number of studies quantifying the relation between noise exposure and temporary threshold shift in birds. Several of the most relevant studies are described below.

Budgerigars exposed to a narrow band of noise centered at 2 kHz for 72 hours at levels of 76–106 dB SPL showed maximum hearing losses at 2 kHz with a TTS ranging from 10–40 dB depending on the level of the noise to which the birds were exposed (Saunders and Dooling, 1974; Dooling, 1980) (Figure 5). Importantly, a PTS of 7–10 dB was observed only with the 106 dB exposure (Dooling, 1980). A 72-hour continuous exposure to a narrowband of noise at 106 dB would result in severe and permanent hearing loss in humans due to irrevocable damage to the sensory cells of the inner ear. TTSs in these birds also lasted less time than typically seen in mammals and were also restricted to a narrower range of frequencies (e.g., Luz and Hodge, 1971; Dooling, 1980; Henderson and Hamernik, 1986). The maximum threshold shift in budgerigars occurred at the exposure frequency (rather than at higher frequencies in mammals) and showed much less spread of threshold shift to other frequencies.

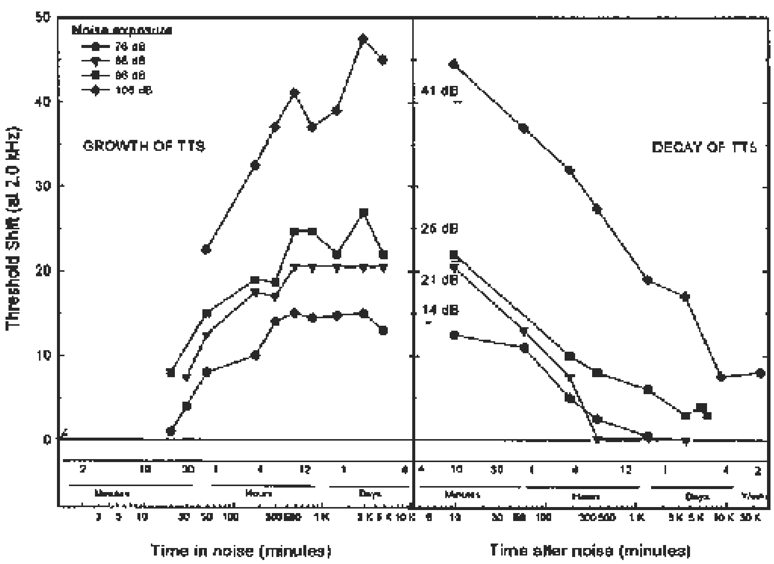


Figure 5: Threshold Shift in Birds Exposed to Noise
The growth and decay of threshold shift in four budgerigars exposed to four different levels of a one-third octave band of noise for 72 hours. Threshold shift reaches an asymptote (horizontal dashed line) after 12 24 hours regardless of the exposure level. Exposure to a 76 dB noise results in a threshold shift of 14 dB which recovers within a few hours following the termination of the noise. Exposure to a 106 dB noise, however, leads to longer recovery time and a permanent threshold due to damage to the inner ear (Dooling, 1982).

Finally, all the experiments described above were conducted with continuous noise, much as would be expected with dense traffic or continuous construction noise (Table 1, Figure 1). Impulse noises, such as those produced by single pieces of construction equipment, are short, intermittent, high intensity, and have very fast rise times (Table 2).

Much less is known about the effects on avian hearing resulting from high-level impulse sounds as might be experienced in close proximity to construction equipment as compared to lower level, continuous noise as from traffic. There is a single report in the literature that exposed budgerigars to four 169 dB SPL blast impulses produced by starter pistol shots in close proximity (20 cm) to the bird. In contrast to results from a continuous noise exposure, this impulsive exposure initially caused more low frequency (~60 dB) than high frequency (~40 dB) hearing loss (Hashino *et al.*, 1988). Even from this extremely intense exposure, however, thresholds at 1 and 4 kHz (the frequencies at which budgerigars sing and hear best) returned to almost normal within 20 days following the exposure. At 500 Hz, there remained a permanent threshold shift of about 20 dB even 40 days after exposure. These results confirm that birds are resistant to permanent auditory damage and hearing loss from noise exposure, even following extraordinarily exposure to intense impulse noise.

B. Masking and the Characteristics of Noise

Masking is the interference of the detection of one sound by another. For example, two people in a room talking at a comfortable level can easily hear one another because the level of the speech signal arriving at the ear is sufficiently greater than the background noise. If the people are having the same conversation in a noisy restaurant, it may be much harder for them to hear one another because the level of the background noise approaches the level of the speech signal from their companion. This is an example of the masking of speech by speech. Moreover, masking can also occur from other kinds of noises that also have energy in the spectral region of speech (e.g., noisy fans, air conditioners, traffic noise).

The simplest kind of masking experiment is to measure the sound detection thresholds for pure tones (the signal) in the presence of a broadband noise (see Appendix A). The noise in such an experiment is usually described in terms of a spectrum level (i.e., sound energy per Hz) rather than the overall sound pressure level. The signal level in the case of a pure tone is, of course, simply the level of the tone in dB. Experiments on masking in birds (and other animals) show that at low- to mid-levels, it is the noise in the frequency region of a signal that is most important in masking the signal—not noise at more distant frequency regions (Dooling *et al.*, 2000b). It could be the case that if the masker energy is at a low to moderate level in a frequency range that does not overlap with that of the pure tone, there may be no change in threshold for the pure tone.[17]

Masking of signals by noises in the same frequency range is an important phenomenon to keep in mind when estimating the effects of different kinds of noises on hearing. Common experience shows that acoustic communication can be severely constrained if background noise is of a sufficient level.[18] Such noise decreases signal-to-noise-ratios and thereby restricts the range over which a signal produced by a bird can be heard by another bird. In simple terms, background noise

[17] The amount of masking depends primarily on the amount of energy in the masker in the frequency region surrounding the pure tone. This band of frequencies around the pure tone in which masking will still occur is called the "critical band."

[18] The exact level depends on many factors, including masker level and the hearing sensitivity of the species of concern.

makes it harder for an animal (including humans) to hear sounds of conspecifics or other sounds that may be biologically relevant. Otherwise said, it limits the organism's active acoustic space.

The masking case described above with a pure tone and broad band noise is very simple. In a natural setting, the situation is usually much more complex. The signal is rarely a pure tone, and the masker is rarely flat, broadband noise. Moreover, human work shows that it has been difficult to come up with a broadly acceptable definition of noise because of extreme variations in both the physical properties of noise and the perceptual preferences of listeners.[19] For humans, perhaps the broadest, most universally accepted definition is that noise is simply unwanted sound. This definition, however, is not useful in trying to predict the effects of masking on animal communication.

To make matters even more complex, noises can be continuous or intermittent, broadband or narrowband, or predictable or unpredictable in time or space. These noise characteristics determine the strategies that birds might employ to minimize the effects of noise on acoustic communication. Most laboratory studies measuring the effects of noise on signal detection (as described above) use continuous noises with precisely defined bandwidths, intensities, and spectral shapes. Because traffic noise on heavily traveled roads can approximate some of these features (e.g., relatively continuous, relatively constant spectrum and intensity), it increases the validity of using laboratory results to make predictions about how far away two birds can be in a natural setting and still hear one another in a background of traffic noise. In fact, for this purpose, laboratory masking studies define the worst case estimate of communication distance in the natural setting. This is because the animal being tested in the laboratory is in a fixed location with respect to the loudspeaker that is producing both the noise and the signal and head movement is restricted. Whenever these two conditions are not met, as is usually the case in a natural setting, the amount of masking from traffic noise is likely to be less, and sometimes considerably less, than predicted from signal-to-noise ratios measured in the laboratory.

C. Comparative Masking Effects in Birds   Critical Ratio

The ratio between the power in a pure tone at threshold and the power per Hz (the spectrum level) of the background noise is called the *critical ratio* (Fletcher. 1940). The masking principles discussed above that govern the critical ratio are shown schematically in Figure 6 (see also Figure 7). The critical ratio (left panel of Figure 6) is defined as the sound pressure level of a tone (when it is just masked) minus the spectrum level of the noise. In this case, the spectrum level of the noise is 40 dB SPL, and the level of a 3 kHz pure tone that can just be heard is 60 dB SPL, resulting in a critical ratio of 20 dB. Since it is noise in the spectral region of the tone that contributes most to the masking of the tone, measuring overall noise level over a very wide band of frequencies is not very useful unless the noise is flat and one can accurately estimate the level of noise around the signal. For a flat noise with an overall noise level of about 80 dBA, when measured across the whole band of noise, would have a spectrum level of 40 dB across the whole spectrum and in the region of the pure tone. When the noise is not flat, it is hard to calculate the spectrum level in the frequency region around 2–6 kHz   the frequency region that contains most of the energy in bird vocalizations.

[19] What is "noise" to one listener may be music to another, and vice versa.

Critical ratio data have now been obtained behaviorally for 14 species of birds, including songbirds (e.g., canary, sparrows, etc.), non-songbirds (e.g., budgerigars, pigeons), and some nocturnal predators (e.g., barn owl) (Dooling *et al.*, 2000b). Figure 7 shows the median critical ratio functions for the 14 species of birds (see Appendix C for these data) with corresponding values from the literature on tone masking by noise in human. There is species variation in bird critical ratios, with some birds approaching human levels of sensitivity and others being much worse than the median curve. However, the median function shows the typical pattern of approximately a 2–3 dB/octave increase in signal-to-noise ratio that has come to be characteristic of these functions in mammals, including humans (roughly a 3 dB/octave slope). The correlation between the increase in masking effectiveness and frequency is thought to be related to the mechanics of the peripheral auditory system (von Békésy, 1960; Greenwood, 1961a; b; Klump *et al.*, 1995).

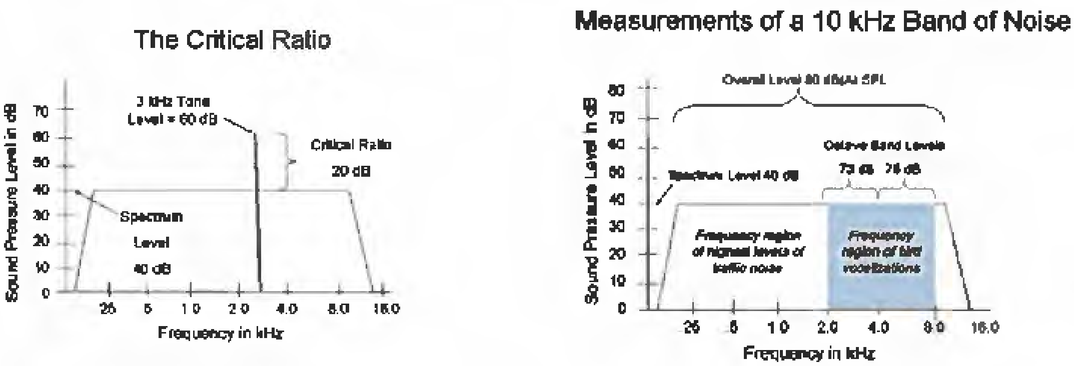


Figure 6: Avian Critical Ratios

(Left) Schematic representation of the critical ratio. A 60 dB tone at 3 kHz is just masked by a broad band noise with a spectrum level of 40 dB. The critical ratio is defined as the level of the tone minus the spectrum level of the noise. (Right) The relationship for overall sound pressure level, spectrum level, and octave band levels between 2 and 8 kHz for a flat broad band noise. The overall level of noise of 80 dBA is greater than the amount of noise falling in the octave band of 2–4 kHz (73 dB) and 4–8 kHz (76 dB). Much of the energy in traffic and construction noise falls in lower frequencies, while bird vocalizations fall in mid- to higher frequencies. Measuring noise that is in the spectral region of bird vocalizations is critical to understanding whether masking occurs because it is predominantly the noise in this spectral region that contributes to the masking.

In practical terms, this critical ratio curve describes the level in decibels above the spectrum level of the background noise that a sound (usually a pure tone or other narrow band sound) must be in order to be heard. For the typical bird, a pure tone (or tonal vocalization) in the region of 3 kHz must be at about 27 dB (± 3dB) above the spectrum level of noise in order to be detected. In fact, birds vary in their critical ratios from about 21 dB (budgerigar) to about 32 dB (canary) at 3 kHz. For the human, the same pure tone need only be about 21 dB above the spectrum level of noise to be heard—a difference of about 6 dB from the typical bird (Dooling and Popper, 2000).

These data raise two important issues. First, there is little variation in how humans with normal hearing are able to detect signals in noise. The same is true of animals within a species. However, there is considerable variation across species in how well organisms can hear in noise, including among different species of birds. As is the case with susceptibility to auditory damage from noise exposure, there is no way to tell from a bird's vocalizations, physical appearance, or behavior, whether it hears well or less well in noise. Thus any complete model for predicting masking for a given species should use the species' critical ratio. The next best solution is to use the average or median values of all bird critical ratios.

Second, the difference in masked thresholds of 6 dB between humans and a "representative" bird with median masking thresholds for the 14 avian species studied has important implications for the detection of a point source of sound (e.g., a single vehicle, a piece of construction equipment, a bird singing, etc.) in a natural setting. Recall that sound pressure level decreases about 6 dB for a point source with every doubling of distance (by the inverse square law). What this means is that if a human listener can barely hear the sound of an automobile or a piece of construction equipment at 100 meters from the highway because of background ambient noise, the typical bird could not hear it at all. The bird would have to move twice as close to the highway (i.e., 50 meters) to barely hear the sound of an automobile. For a line source (e.g., a stream of traffic) which deceases at 3 dB/doubling of distance, this difference between birds and humans is a factor of 4.

Generally, since human auditory thresholds in quiet and in noise are about 6 dB better than that of the typical bird, this leads to the following two facts when conclusion on assessing the effect of noise on birds:

(1) When estimating whether a bird might be disturbed by hearing traffic or construction noise from a distant site, this 6 dB difference in masked thresholds means that if a human can barely hear traffic or construction noise from a distant site, a bird certainly cannot hear the noise and therefore can't be disturbed by it. The rule that "if a human can't hear it, a bird can't either" thus proves a handy rule of thumb for estimate whether a distant noise from construction equipment might be disturbing.

(2) However, when trying to estimate whether two birds can acoustically communicate against a background of traffic or construction noise, this 6 dB difference also means that the typical bird must be much closer to a singing bird to be able to hear it than does a human. So, if a human can barely hear a singing bird in the distance, the typical bird would not be able to hear it. In fact the bird would have to be even closer (i.e., half the distance) in order to hear the singing bird. In this case, human perceptual experience provides a dangerously poor estimate of whether two birds can hear one another against a background of traffic noise. It underestimates the effect of noise on communicating birds by over estimating the distance over which birds can communicate.

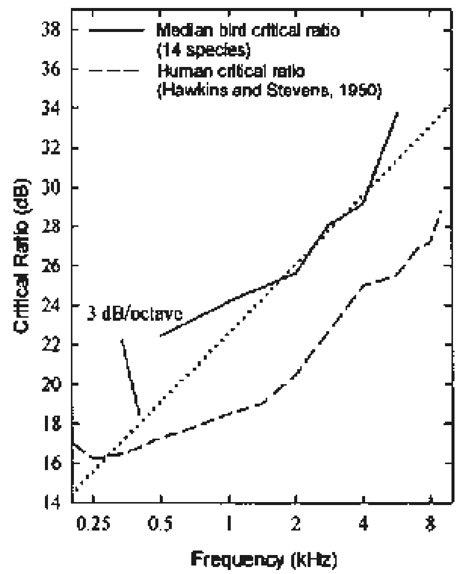


Figure 7: Critical Ratios in Birds and Humans

Median critical ratios for 14 birds (solid line) and the human (dashed line). Dotted line is a slope of 3 dB/octave. The critical ratio (s/n ratio) at threshold is about 6 dB greater in the typical bird compared to humans over the frequency range of 1–5 kHz (Dooling *et al.*, 2000b). These median critical ratios for birds represent the best available science of how birds hear in noise and can be used to predict how well birds can communicate in noise.

D. Understanding the Implications of Masking and Hearing in Noise

As discussed earlier, the audiogram represents the lowest sound pressure level (in dB) of pure tones throughout the range of hearing that can be detected in the quiet background of a test booth (see Figure 2). But since all hearing in natural settings is against a background of noise, the pure tone audiogram is not very useful for estimating what a bird can hear in a natural setting. In other words, in all environments, other than a quiet background of a test booth, ambient noise in the background has a large effect on what can be heard (i.e., the critical ratio). Therefore, the critical ratio (Figure 6) provides the metric for estimating the effects of noise on the audiogram because it shows the level (in dB) that a pure tone must be above the spectrum level of noise in order to be heard.

The realization that all hearing in natural settings are masked thresholds and that a signal, in order to be heard, must be a certain level above the noise, provides a way to estimate the effect a particular continuous noise on the hearing of the typical bird. In the case of the 84 dBA traffic noise illustrated in Figure 8, there is a large masking effect from traffic noise at low and mid frequencies of the bird audiogram but less at high frequencies. Birds living in city environments tend to have higher pitched vocalizations than their rural counterparts because there is less masking from traffic noise at higher frequencies in rural environments.

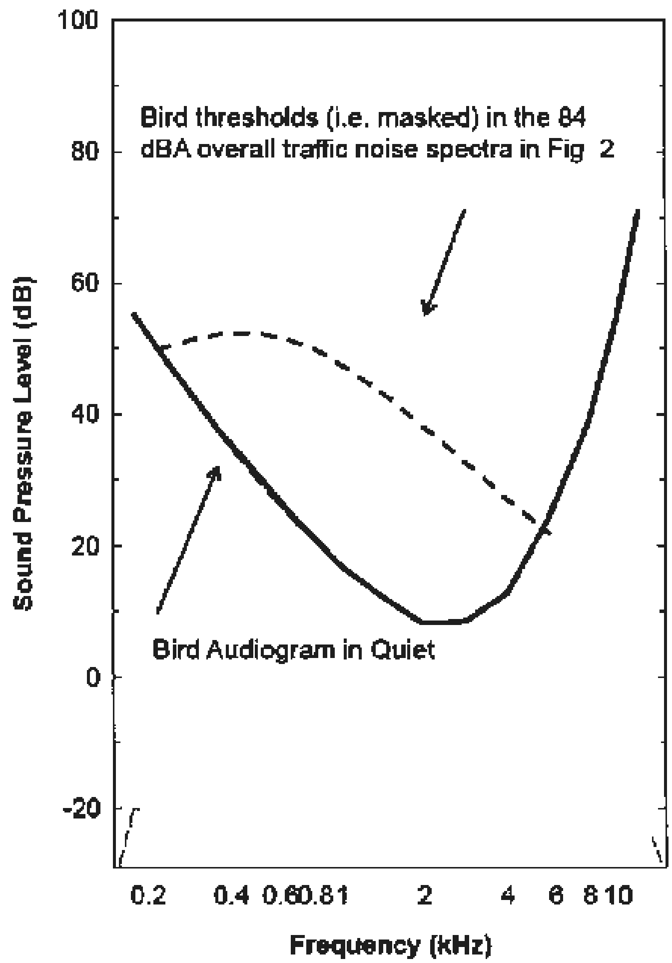


Figure 8: The Effects of Traffic Noise of 84 dBA on Hearing Thresholds of the Typical Bird
The effects of traffic noise illustrated earlier in Figure 2 raises a bird's threshold. The solid line shows the auditory thresholds (audiogram) in the quiet. The dashed line above the audiogram shows elevated thresholds due to masking by traffic noise at a level of 84 dBA. Thresholds are considerably elevated at low- to mid-frequencies.

4. Effects of Traffic and Construction Noise on Birds—A Review of Relevant Literature

A. Overview

Reviewing effects of traffic noise on birds has been challenging in several ways as; it is difficult to find an effective way to evaluate information from very diverse perspectives to arrive at a useful predictive tool. One challenge is separating the effects of noise on birds from the effects of other variables (usually visual, but possibly vibratory or olfactory) that may occur along with the noise. Another challenge is in is applying findings from well-controlled laboratory studies involving a few species to the effects of noise exposure on birds in their natural environments. Under controlled circumstances in the laboratory, hearing capabilities can be measured to a precise degree. As mentioned above, these measures, when taken to the field, represent a worst case in terms of predicting the effects of noise on birds. This is because in laboratory studies, the noise is presented continuously, the signal and the noise are coming from the same location, and any other environmental cues ordinarily associated with the signal (e.g., visual cues) or the noise that might aid auditory perception of important biological signals in a natural setting are not present. Wild animals use an array of short term and long term strategies for counteracting the effects of noise in more natural environments, as described later. These are similar to the behaviors that humans employ in trying to hear and communicate in a noisy environment such as turning the head, raising the voice, moving closer to the source, etc.

Studies and reviews of the effects of traffic and construction noise on birds are often included in a broader literature on the effects on birds of other noise sources, most notably those produced by aircraft (airplane or helicopter) over-flight (e.g., Brown, 1990). Such studies sometimes provide insight into the effects of noise on breeding biology (e.g., Bunnell *et al.*, 1981), survival of eggs and young birds (Burger, 1983; Leonard and Horn, 2008), and non-auditory physiological effects. A number of these papers might also serve as more controlled experimental studies where the effects of noise on birds could be isolated and understood, and such studies may provide guidance for the type(s) of studies that are needed in order to better understand the effects of traffic and construction noise on birds.

At the same time, the characteristics of noise from aircraft is sufficiently different from that produced by traffic that extrapolation from one set of response data to the other is very difficult (Stansfeld *et al.*, 2005; Murphy and King, 2014) and perhaps should not be done at all. These differences include sound level and temporal distribution. Generally, at similar distances from the source, aircraft noise is far more intense than noise from roadways. Moreover, exposure to aircraft noise is almost always intermittent, whereas traffic noise can often be characterized and modeled as a continuous, lower level noise source. Birds respond to such differences in sounds in different ways; therefore, it becomes questionable whether it is possible to extrapolate between sound sources in trying to assess the effects of traffic noise on birds.

There is considerable evidence that road noise can contribute to stress and alter human physiology in many ways (Miller, 1974; Öhrström and Rylander, 1982; Öhrström and Björkman, 1983; Ouis, 2001; Le Prell *et al.*, 2012; Murphy and King, 2014). While caution should rule in the extrapolation of data from humans to birds or other animals, the many similarities in physiology between humans and birds, and the reliance of both on sound for communication, suggests the possibility that stress and physiological effects on humans may be paralleled in birds (and other terrestrial vertebrates).

B. Birds and Traffic and Construction Noise

As pointed out at the beginning of this Guidance Document, the world is becoming a noisier place and the cost of chronic noise exposure for terrestrial organisms could become significant (Barber *et al.*, 2010; Pijanowski *et al.*, 2011a, b; Luther and Magnotti, 2014; Merchant *et al.*, 2015). When the original 2007 report (Dooling and Popper, 2007) was written, there were relative few well-controlled studies on the effects of traffic noise on birds and a considerable amount of grey literature consisting of uncontrolled studies and anecdotal observations studies all suggesting the possibility of negative effects of traffic noise on birds. For instance, at that time there were reports from several investigators, later confirmed and published, suggesting that there may be differences in vocalizations between city birds and country birds, with city birds generally singing at a higher pitch presumably due to greater amounts of low frequency noise from urbanization, including traffic noise (Nemeth and Brumm, 2009; Nemeth and Brumm, 2010a; Slabbekoorn *et al.*, 2012). However, these studies in aggregate also led to two other inescapable conclusions: there were likely to be large species differences in susceptibility to increased noise, and there is an enormous challenge ahead in pinpointing the precise effects of traffic and construction noise on birds.

However, in the past eight years, there has been a number of more refined laboratory and experimental field research and observations published in peer-reviewed journals that has clarified some of the outstanding issues that were identified in earlier work. There is now a body of scientific literature which allows much stronger statements regarding the effects of noise on birds and the strategies birds use to adapt to increasing noise levels. While there are still numerous questions, especially with regard to species differences, it is overwhelmingly clear that many species of birds do respond to traffic noise (though no studies have focused on construction noise). However, it is also becoming apparent, as also discussed below, that many bird species successfully use the same kinds of strategies that humans and other animals use to hear and communicate in a noisy environment such as that created by traffic noise.

Results up to 2007: Many of the key issues involving the effects of traffic noise on birds were raised in the earlier literature, as were suggestions for future research. More recent findings have relied on this earlier work, and there is now a growing body of data that resolve some of the earlier issues. This Guidance Document focuses a review on this more recent data. For a complete review of the earlier work, please refer to the original report (attached as Appendix

Many of these earlier studies were in a very real sense pioneering. They also in many cases revealed considerable species variation and often did not have sufficient control of critical variables; therefore, these studies could not isolate the potential effects of highway noise on birds or provide general guidance (Clark and Karr, 1979; Ferris, 1979; Van der Zande *et al.*, 1980; Reijnen and Foppen, 1994; 1995; Reijnen *et al.*, 1995; Lee and Fleming, 1996; Llacuna *et al.*, 1996; Kuitunen *et al.*, 1998; Reijnen *et al.*, 1998; Clench-Aas *et al.*, 2000; Stone, 2000; Fernández-juricic, 2001; Forman *et al.*, 2002; Peris and Pescador, 2004). This literature has been reviewed several times in recent years (e.g., Sarigul-Klijn *et al.*, 1997; Kaseloo, 2005; Warren *et al.*, 2006; van der Ree *et al.*, 2011; Ortega, 2012; Slabbekoorn *et al.*, 2012; Merchant *et al.*, 2015); therefore, it will not be re-reviewed here. Instead, issues arising from this earlier work are listed below as a framework in which to understand the more recent, and generally more scientifically rigorous, work that has followed.

1) What evidence is there to suggest that results from one species or set of conditions can be generalized to all bird species?
2) Which aspects of a bird's behavior are likely to be affected by traffic noise?
3) How can one be sure that the effects of traffic noise on a bird is due to noise and not to other accompanying visual (i.e., moving vehicles) or olfactory (i.e., exhaust emissions, or tactile (i.e., vibration) stimuli?
4) Most studies are of adult birds. What are the effects of traffic noise on birds that must learn their vocalizations from auditory information?
5) Laboratory masking studies typically use white noise. Do the general masking principles emerging likely to hold for other anthropogenic noises?

Studies Since 2007: Many of the more recent studies discussed below add more high-quality information to the growing body of literature on this topic. Other studies are aimed specifically at some of the lingering questions from the last review and now allow conclusions on these questions, leading to an overall better understanding of how construction and traffic noise could impact birds.

Regarding the prevalence of noise effects on birds, a within-genera comparison of singing in 529 bird species within 109 genera has recently showed that species occurring in urban environments generally vocalize at higher frequencies than non-urban congeneric species without differing in body size or the vegetation density of their natural habitats (Hu and Cardoso, 2009, 2010). For example, white-crowned sparrow (*Zonotrichia leucophrys*) song increased in minimum frequency from 1969 to 2005 in San Francisco, and male birds responded more strongly to current songs than to earlier songs indicating current songs are most effective in the noisier environment (Luther and Baptista, 2010a; Luther and Derryberry, 2012; Luther and Magnotti, 2014).

For some species, it is clear that the whole communication process is affected and not just by the level of noise but by the actual signal-to-noise ratio. European robins (*Erithacus rubecula*) were presented with two playback songs, one with noise, one without; the male birds responded to the song in noise with increased minimum frequency and decreased song complexity and song duration (McMullen *et al.*, 2014).

In another study, low frequency traffic noise reduced female canary responsiveness to low-frequency, more attractive songs but did not affect responsiveness to high-frequency songs (Huet des Aunay *et al.*, 2014). In the great tit (*Parus major*), low frequency songs by males are related to female fertility and sexual fidelity. Urban noise impairs male-female communications shifting communication to higher frequency songs (Halfwerk *et al.*, 2011). Interestingly, artificial noise in nest boxes shows that female great tits can steer male singing behavior under noisy conditions, making males sing closer to the nest boxes even though males were not themselves exposed to noise (Halfwerk *et al.*, 2012). In another study, great tits were 6 dB better at detecting high frequency songs than low frequency songs in urban noise, but not in woodland noise. Moreover, discrimination between low frequency variants of song was less efficient than discriminating high frequency variants. High frequency elements were used by birds in urban noise, while all song elements were used in discriminating between songs in woodland noise (Pohl *et al.*, 2012).

A great deal of research has also examined the relation between the increase in vocal intensity and the increase in vocalization frequency and whether there is a cause-effect relationship between these changes or if they occur independently (reviewed in (Zollinger *et al.*, 2012). Some birds adjust both loudness and peak frequency in their songs to compensate for traffic noise rather than simply adjusting loudness with a correlated frequency shift (Cardoso and Atwell, 2011). Other species vary multiple parameters. With increasing noise levels, plumbeous vireos (*Vireo plumbeus*) sang shorter songs with higher minimum frequencies while grey vireos (*Vireo vicinior*) sang longer songs with higher maximum frequencies suggesting that vocal plasticity may help some species occupy noisy areas (Francis *et al.*, 2011a, b). But the results are likely environmentally determined. The common blackbird (*Turdus merula*) preferentially sang higher frequency songs elements that can be produced at higher intensities and, at the same time, are less masked by low frequency traffic noise (Nemeth *et al.*, 2013b).

But it was also shown that for the common blackbird and the great tit, increasing frequency (song pitch) was less effective at increasing communication distance in noisy environments than was increasing vocal amplitude (Nemeth and Brumm, 2010a). Silvereyes (*Zosterops lateralis*) exposed to low and high frequency noise lowered the minimum frequency of their calls, and this shift was independent of amplitude which increased in all noises. Thus, silvereyes are clearly capable of flexible adjustments of call frequency, amplitude, and duration to maximize signal-to-noise ratio in noisy environments (Potvin and Mulder, 2013).

The variation noted in the earlier literature is still a leading finding. There are substantial species differences in which song features are adjusted. In the house wren (*Troglodytes aedon*), anthropogenic noise reduced bandwidth, increased trill rate, and increased minimum frequency (Redondo *et al.*, 2013). On the other hand, both northern cardinals (*Cardinalis cardinalis*) and American robins (*Turdus migratorius*) increased frequency range as noise increased but did not change song length or singing rate (Seger-Fullam *et al.*, 2011). A study in house sparrows (*Passer domesticus*) revealed that chronic noise exposure reduced fitness by masking parent-offspring communication rather than male-female communication (Schroeder *et al.*, 2012). Moreover, black-capped chickadees (*Poecile atricapillus*) use shorter, higher frequency vocalizations when traffic noise is high, and longer, lower frequency songs when noise abates (Proppe *et al.*, 2011). The same species sing at higher pitches with elevated anthropogenic noise but not with decreasing canopy cover, suggesting noise is the main factor, and not vegetation, that leads to increased song pitch (Proppe *et al.*, 2012). Finally, a pattern seen among seven songbird species is that noise contributes to declines in urban diversity by reducing the abundance of select species in noisy areas, especially species with low frequency songs (Proppe *et al.*, 2013).

Noise effects are complex, usually related to level, and can be both short- and long term. Serins (*Serinus serinus*), a small European songbird related to canaries, responded to increasing levels of anthropogenic noise by increasing song activity up to noise levels of about 70 dBA, after which singing activity decreased with further increases in noise level (Díaz *et al.*, 2011). Male cardinals gave stronger responses to songs of average frequency than to songs with shifted frequency at low levels of background noise, but the difference disappeared at high noise levels, suggesting that frequency shifted songs were not advantageous in terms of communication at higher noise levels (Luther and Magnotti, 2014). Red-winged blackbirds (*Agelaius phoeniceus*) increased song tonality when temporarily exposed to low frequency white noise, and birds living in noisier environments showed increased tonality when singing in quiet, suggesting both short-term and long-term effects (Hanna *et al.*, 2011). On the other hand, male red buntings (*Emberiza bruniceps*) adjusted their songs immediately in response to noise singing at higher frequency and a lower rate when noise level were high, suggesting short-term, rather than long-term, adaptations (Kane *et al.*, 2010).

The effects of noise on bird songs are usually, but not always, negative. The female American kestrel (*Falco sparverius*) had higher cortisol levels and abandoned nests more frequently near busy roads and developed areas (Strasser and Heath, 2013). In a study of a number of bird species in northwestern New Mexico, noise alone decreased nesting species richness and this led to different communities of birds with less interaction with one another. But, unexpectedly, this same noise indirectly facilitated reproductive success of individuals nesting in noisy areas as a result of disruption of predator–prey relationships (Francis *et al.*, 2009). Experimental noise exposure data in six European songbird species revealed a noise-related earlier start of dawn singing for two out of six species but revealed no impact on four species with more variable starting times for dawn singing (Arroyo-Solís *et al.*, 2013).

Another study of six different American songbird species also found that the effects of urban noise on song were mixed. Minimum song frequency increased with noise level for two species, with those species singing in lower frequencies being most affected. On the other hand, maximum frequency and frequency range decreased for two species, with increasing urban noise at quiet sites (Dowling *et al.*, 2011). A recent paper examined the effects of noise on a bird's ability to discriminate between various levels of song degradation—a cue used by birds to gauge the distance from other singing birds. The great tit's overall responses in a noisy dawn chorus were, unexpectedly, very similar to their performance in silence.

Finally, Ware et al. (2015) conducted a well-controlled and designed study that separated the effects of traffic noise from the other sensory effects that accompany traffic noise such as exhaust (i.e., olfactory) and vehicular traffic (i.e., visual) by creating a "phantom road." Results across species were decidedly mixed. Some species avoided the noisy area, and some lost weight, while others did not. It's possible that presenting traffic noise without the attendant visual (e.g., moving vehicles), olfactory (i.e., exhaust emissions), and tactile (i.e., vibration) cues is itself stressful to some birds because these cues all normally occur together. Results from these recent studies confirm that the effects of traffic noise remain complicated and are highly likely to vary by species and other conditions (see also Merchant *et al.*, 2015).

Recent studies with young birds and nestlings, add even more complexity to the mixed effects described above. Young birds would not be expected to have had experience with noisy objects, such as vehicles, in their environment and, thus, the effects of noise alone might be easier to gauge. Crino et al (2013) showed nestling white crowned sparrows (*Zonotrichia leucophrys*) exposed to traffic noise had lower glucocorticoid levels and improved condition relative to control nests. Nestling Eastern bluebirds, young enough to be constrained to the nestling box were recorded in their natural habitat at various locations from quiet to near highways, parking lots, and other noisy environments. Birds did not increase the amplitude or structural characteristics of the begging calls in response to increasing noise levels (Swaddle *et al.*, 2012). On the other hand, a recent study on zebra finches by Potvin and MacCougall-Shackleton (2015) showed that chronic, long-term exposure to traffic noise in an experimental setting had both immediate and long-term effects on song but not in a way that would reduce masking. Moreover, the noise exposure resulted in a decrease in corticosterone suggesting reduced stress.

Finally, a recent study examined the effects of traffic noise played to juvenile free-living house sparrows (*Passer domesticus*) and showed that exposed birds had shorter telomeres (chromosome ends) than birds not exposed, although the experimental and control birds were identical in all other ways, including health (Meillère *et al.*, 2015). Telomeres decrease in size with aging, and it is generally accepted that there is a correlation between telomere length and longevity. Thus, these results, though the first of their kind and only for single species, suggest a new mechanism by which traffic noise might affect birds.

The emerging picture from the latest research on the effects of noise on birds is one of more careful data collection and focused research designs but with complex outcomes still occurring and large species differences still the rule. Finally, extreme noise events may also have more extreme effects.

Using weather radar technology, it was documented that thousands of birds take flight following evening fireworks displays lasting 45 min. The peak densities of fleeing birds extended to altitudes of at least 500 feet (Shamoun-Baranes *et al.*, 2011). While this is the only report of its kind, it may have implications for the effects of short-term, high-level construction noise, especially when it occurs at night.

Summary of Recent Studies on Effects of Traffic Noise on Birds: The overall picture that emerges from the research since 2007 is still one of considerable complexity and variation. It is now abundantly clear that noise has a widespread effect on many species of birds. However, this is not to say that it is any easier to predict the specific effects of traffic noise on any particular species in its natural habitat. The recent literature also shows that the same noise can affect different species sometimes in the same way but often in different ways. And it is still the case that there are clear examples where traffic noise actually benefits a species rather than causing harm.

Nevertheless, it is difficult to argue with the notions that the world is an increasingly noisy place and noise affects birds and interferes with their acoustic communication. It follows that there should be an effort made to monitor anthropogenic noise and decrease noise levels where possible. The challenge in pinpointing specific effects of noise or finding invariant noise levels that cause harm across conditions should not be surprising. The same lack of specificity is true of humans living and communicating in noisy environments. Personal experiences (e.g., conversing in a noisy restaurant) make it clear that humans can and do employ a plethora of both short-term and long-term adaptive strategies for communicating effectively in noise, which makes it impossible to determine that a particular type or level of noise is accurately predictive. It is evermore clear from field studies and well-controlled laboratory studies that birds can and do use human-like strategies, described below, for counteracting the effects of an increasingly noisy environment. And, as with humans, it is possible from laboratory studies on birds to define a level of noise that would represent a "worst case" scenario in terms of interfering with acoustic communication. In other words, there is a precise signal-to-noise ratio at the ears below which communication is impossible without employing short term adaptation strategies (i.e., those typically available to freely moving birds in their natural habitat). That signal-to-noise value comes from laboratory studies and is the critical ratio.

C. Short-Term Adaptations to Noise Masking

A critical question is how birds, or any animal, including humans, adapt to noise (traffic) masking in the short term. Based on both highly controlled laboratory and field studies, it is apparent that in natural settings, birds can use many strategies to maximize their hearing in noise. For one, birds are able to adjust the characteristics of their vocalizations in response to temporary changes in the background noise. There is now a considerable amount of literature demonstrating that birds can adjust the amplitude of their vocalizations in response to increased noise by a phenomenon first referred to in humans as the Lombard effect. A number of species of birds have been shown to raise the level of their vocal output by as much as 10 dB in the presence of moderate background noise that is loud enough affect the bird's perception of its own vocalizations (Potash, 1972; Cynx *et al.*, 1998; Manabe *et al.*, 1998; Brumm and Todt, 2002; 2003; Hu and Cardoso, 2010; Nemeth *et al.*, 2013a).

The ability of birds to adjust vocalization in the presence of noise has now been demonstrated by studying behaving birds trained to wear headphones while vocalizing (Osmanski and Dooling, 2006). In these experiments, presenting noise through headphones caused the bird to raise the amplitude of vocal output by as much as 10 dB. These highly controlled laboratory studies are now complemented by a variety of field studies such as a study showing that males of the common nightingale (*Luscinia megarhynchos*) sing louder in noisier territories, and birds in urban areas sing louder on working days than on weekend days when noise levels are reduced (Brumm, 2004).

Paralleling what is known from humans communicating in noise, there is limited evidence that at least some birds use repetition rate or increases in call duration to increase the efficiency of signal transmission. Japanese quail increase the number of call syllables per call series in noise (Potash, 1972) and king penguins (*Aptenodytes patagonicus*) respond to increasing levels of background noise due to wind by increasing the number of syllables in their calls (Lengagne *et al.*, 1999).

Birds are also capable of making short term alterations in the spectrum of their vocalizations (Hultsch and Todt, 1996; Manabe, 1997). The basic mechanisms for this was more recently examined in budgerigars trained to produce vocalizations while wearing headphones. Such birds can be induced to pitch-shift their vocalizations in real time. Artificially shifting the pitch of auditory feedback of the bird's own vocalizations resulted in the bird compensating by shifting the pitch of its vocalization in the opposite direction (Osmanski and Dooling, 2009). These experiments demonstrate that birds have some short-term control over the pitch of their vocalizations and may use this ability to maximize information transfer in a noisy environment.

Clearly, humans can choose to communicate when noise levels are low and limit communication when noise levels are so high as to make communication impossible. It is also well known that birds can adjust the timing of their vocalizations to avoid competition for acoustic space with other species or to coincide with low noise periods to prevent auditory masking (Cody and Brown, 1969; Wasserman, 1977; Ficken *et al.*, 1985; Popp *et al.*, 1985; Popp and Ficken, 1987; Evans, 1991; Luther and Baptista, 2010b; Nemeth and Brumm, 2010b).

Birds (both senders and receivers) can also behaviorally counteract the effects of masking noise on acoustic communication by changing their location. One strategy that can improve signal-to-noise ratio is to move to a position in the habitat in which the transmission pathway is better for the signal than the noise (Brumm and Slabbekoorn, 2005). Thus, moving higher up into the canopy of the vegetation is another response that will improve the signal-to-noise ratio (Mathevon *et al.*, 1996; Holland *et al.*, 1998). With European blackbirds *(Turdus merula)*, it is estimated that moving up from the ground to a perch at about 9 meters (29.5 feet) high would result in an increase in audibility that is comparable to the receiver moving 90 meters (295 feet) closer to the sender horizontally (Dabelsteen *et al.*, 1993).

Birds (like humans and other binaural animals) enjoy a "spatial release" from masking when the noise source is spatially separated from the signal source. That is, when the signal to be detected comes from a different location in space than the noise, having two ears leads to an improvement in signal detection (Popper and Fay, 2005). In human hearing, this can represent a large effect, but there were some questions whether birds, with their closely spaced ears, would enjoy a similar benefit (Dent *et al.*, 1997). A Laboratory study with budgerigars under controlled conditions has shown that the amount of this masking release is can be as much as 10–15 dB when the noise and the signal arrive at the bird's ears from 90 degrees apart (Dent *et al.*, 1997) paralleling the advantage gained by humans when they scan the environment using head movements to hear a weak acoustic signal. Recalling that sound pressure decreases roughly 6 dB with each doubling of distance, this could translate into a quadrupling of distance over which two birds could communicate if they position themselves optimally with regards the noise source (i.e., at 90 degrees).

D. Long-Term Adaptations to Noise Masking

Even without human-generated noise, natural habitats have particular patterns of ambient noise (the acoustic scene) resulting from, among other things, wind, animal and insect sounds, and other noise-producing environmental factors such as a streams, waterfalls, etc. Biologists have long suspected that such noise has exerted a selection pressure on the evolution of acoustic signals, especially in birds (e.g., Morton, 1975; Brenowitz, 1982; Wiley and Richards, 1982; Ryan and Brenowitz, 1985; Slabbekoorn, 2004; Smith *et al.*, 2008, 2013). Brumm and Slabbekoorn (2005) reported that the large-billed leaf-warbler (*Phylloscopus magnirostris*), which lives close to river torrents in the Himalayas, evade masking of their territorial songs by producing high-pitched notes in narrow frequency bands around 6 kHz (Dubois and Martens, 1984). In fact, differences in song or call structure based on differences in habitat have been reported, or suspected, in a number of avian species (Douglas and Conner, 1999; Slabbekoorn and Smith, 2002; Slabbekoorn and Peet, 2003), such as for the songs of little greenbuls *(Andropadus virens)*. It remains an intense area of study as to whether a given vocalization is adapted to environmental noise by evolutionary or ontogenetic changes or both.

E. Estimating Maximum Communication Distance between Two Birds Using Laboratory Masking Data

The question of whether noise affects vocalization structure raises a parallel question of how much noise is too much. In other words, how loud does a noise have to be before the bird must begin to alter the structure of its vocalizations in order to communicate? To address this question with quantitative rigor, Lohr *et al.* (2003) examined the effects of masking on the detection and discrimination of species-specific vocalizations in zebra finch and the budgerigar using two different types of continuous noise—one a flat, broadband noise and the other shaped like traffic noise with more energy at low frequencies and less at high frequencies.

Lohr and his colleagues used both budgerigar vocalizations (narrow band and tonal) and zebra finch vocalizations (broadband and harmonic) and measured both detection and discrimination because being able to detect a sound is not the same as being able to discriminate effectively between sounds or to recognize a particular sound. Results show exactly this for —it requires slightly better signal-to-noise ratio for birds to discriminate between two sounds in noise than to detect the sounds in noise at equivalent levels of performance. This is much like the case of perceiving speech in human listeners where hearing or detecting speech is not the same as actually hearing it well enough to understand what is being said.

These results enabled the investigators to estimate the theoretical maximum communication distance (d_{mc}) by solving the following equation adopted from Marten and Marler (Marten and Marler, 1977) and Dooling (Dooling, 1982):

$$\mathrm{Drop} = 20 \cdot \log \frac{d_{mc}}{d_{o}} + \frac{EA \cdot d_{mc}}{100}$$

Drop: the amount of signal attenuation from source intensity to that at threshold;

d_{mc}: the maximum communication distance;

d_0: the distance at which source intensity is measured; and

EA: the amount of excess attenuation (linear attenuation, not due to spherical spreading).

Solving the above equation for both detection and discrimination of each species calls in both types of noise, and it is possible to generate a series of curves to describe maximum effective communication distances for a given level of background noise (Lohr *et al.*, 2003). In this analysis, a source intensity level of 95 dB SPL at 1 meter was assumed, as was an excess attenuation of 5 dB/100 meters (appropriate for an open area) (Lohr *et al.*, 2003). These values fall within the range of those measured in the field but are near the high end for source intensity (Brackenbury, 1979a, b) and the low end for excess attenuation (Marten and Marler, 1977; Brenowitz, 1982).

Such an approach provides a way to estimate maximum communication distance under fairly good conditions from the perspective of a receiver and revealed both species differences and vocalization differences. The results demonstrate that it is easier for birds to hear vocalizations in traffic noise than flat noise. A bird can detect and discriminate budgerigar calls at longer distances than it can zebra finch calls. Budgerigars do better than zebra finches. And the distances over which signals may be discriminated are shorter than distances at which those same signals may be detected. These predictive distances from the laboratory masking data do not take into account any gains from short term adaptation strategies animals are able to use in their natural habitats. So, the distances obtained from this model represent the worst case scenario.

F. Putting It All Together Predicting the Effects of Noise on Bird Acoustic Communication

It is clear that acoustic communication can be constrained if background noise is of a sufficient level, and can become impossible in very high noise levels. These effects occur because the noise decreases signal-to-noise ratios, thereby limiting the acoustic space of a sound. Noises can be continuous or intermittent, broadband or narrowband, and predictable or unpredictable in time or space. Background noise makes it harder for an animal (including humans) to detect sounds that may be biologically relevant, to discriminate among these sounds, to recognize these sounds, and to communicate easily.

Since the early studies by Lohr *et al.* (2003), more recent work (Dooling and Blumenrath, 2014) has elaborated on predicting communication distance in noise by considering not just detection and discrimination, but other meanings of hearing, including recognition and comfortable communication. It is now clear that signal discrimination requires a higher signal-to-noise ratio than detection; that recognition in both humans and birds requires an even higher signal-to-noise ratio than discrimination; and comfortable communication requires an even higher signal-to-noise ratio (Lohr *et al.*, 2003; Freyaldenhoven *et al.*, 2006). Interestingly, there is about a 3 dB difference in signal to noise ratio required between detection (i.e., the critical ratio) and discrimination, and between discrimination and recognition for both birds and humans. It is not possible to measure comfortable communication in a bird, but in humans a signal-to-noise ratio of about 15 dB is required. The similarity between birds and humans on the different signal-to-noise ratios required for detection, discrimination, and recognition strongly suggest that the 15 dB signal-to-noise ratio required for comfortable communication can probably also be applied to birds.

The approach developed from the above discussion integrates the spectrum level of the masking noise, how well the bird hears in noise (i.e., the critical ratio), the level at which the bird sings (Brackenbury, 1979b), as well as some simple acoustic characteristics of the environment. The model is based on the spectrum and the level of both the noise and the signaler's vocalization at the receiver's ear. These values for spectrum and level of noise and signal can either be measured directly or they can be estimated by applying signal attenuation algorithms to both the noise source and the signal source. The model is particularly relevant because it incorporates the notion that different auditory behaviors from detection (i.e., the critical ratio) to communicating comfortably (i.e., 15 dB greater signal-to-noise ratio than the detection threshold). For the listening bird, the model provides distances corresponding to the human perceptual experience of communicating comfortably versus just being able to detect that something was said.

Figure 9 shows the effects of anthropogenic traffic noise on four different auditory behaviors based on the median bird critical ratio function (see Figure 7 and discussion of masking). The specific case illustrated is for a background noise level at the listening bird of 60 dBA—a level that is typical of traffic noise measured roughly 300 meters (984 feet) from a busy 6 lane roadway. This example assumes the calling bird is vocalizing at a peak SPL of 100 dB (as measured 1 meter (3.3 feet) from the bird) through an open area and that the vocalization is affected by excess attenuation, in addition to the loss due to spherical spreading, of 5 dB/100 meters (328 feet).

In this noise, a comfortable level of communication between two birds requires a distance between them of less than 60 meters (197 feet). Recognition of a bird vocalization by the receiver can still occur at greater inter-bird distances up to about 220 meters (722 feet). Discrimination between two vocalizations is possible at inter-bird distances up to 270 meters (886 feet). And finally, simple detection of another bird's vocalization can occur at distances up to 345 meters (1,132 feet) in this noise. These findings can be plotted in terms of a bird's active auditory space as in shown in Figure 10 as a set of concentric circles with a listening bird in the center and a calling bird located at various distances from the listener representing the kind of auditory behavior that is possible at that distance.

G. Defining Guidelines for Effects

The model described above (Lohr *et al.*, 2003; Dooling *et al.*, 2009; Dooling and Blumenrath,

2014) incorporates many factors that should be considered when establishing guidelines for the effects of traffic and construction noise on birds. Based on psychophysical thresholds measured in a laboratory setting, it shows maximum communication distance for a typical bird in a natural setting based on the intensity with which the bird vocalizes and the transmission loss from the environment due to the excess attenuation. The threshold for effect would also have to take into account what is known about the spectral characteristics of vocalizations, the distance over which conspecific acoustic communication (e.g., the territory size) normally occurs, and the existing levels of ambient noise. Noise levels that limit the maximum communication distances to a distance that is less than the diameter of the bird's territory size (or known communication distances in ambient noise) may have serious biological consequences. The level of natural ambient noise already present in the bird's environment is a key factor in determining whether additional noise from traffic and construction would have any effect. Traffic or construction noise below ambient noise levels would not affect communication.

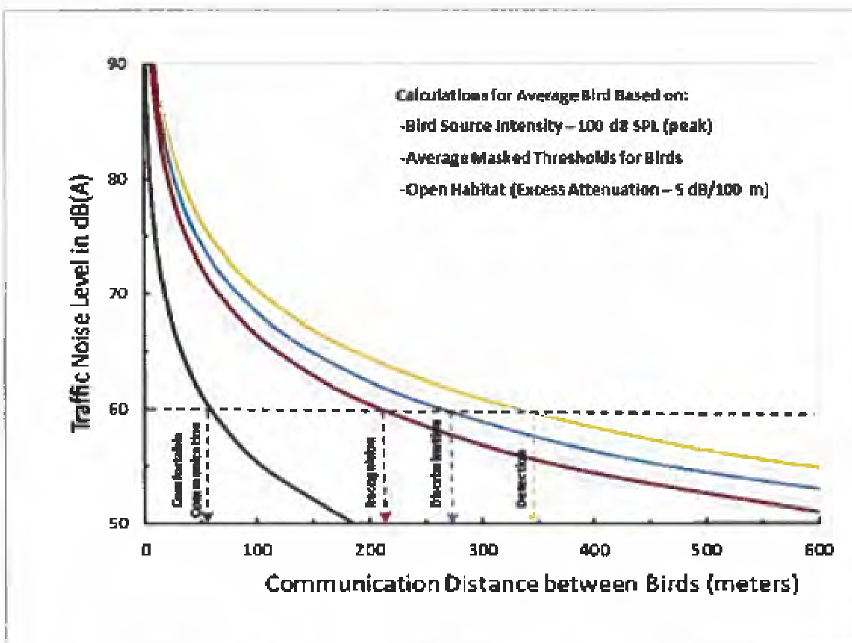


Figure 9: The Effects of Anthropogenic Traffic Noise on Four Different Behaviors Based on the Average Bird Critical Ratio Function

Based on the traffic noise spectrum shown in Figure 2 at a level of 60 dBA, comfortable communication occurs up to 60 meters; recognition of a vocalization can occur up to about 110 meters; discrimination between two vocalizations at about 270 meters, and detection at about 340 meters. Beyond this distance, a bird is not likely to detect the signal. This is based on laboratory critical ratio data and, thus, defines a worst case scenario. In a natural setting, birds would be expected to use their demonstrated short-term adaptation strategies for communicating in noise.

Clearly, variation in territory size, the size of the critical ratio among birds, and natural ambient noise levels are key variables that make it impossible to use a single noise level as a one-level-fits-all level in terms of estimating whether traffic and/or construction noise is limiting communication distance by causing additional masking. In fact, species differences and habitat differences can make rather large differences in the distance. There are species differences in critical ratios and therefore these plots would look different for different species. Because budgerigars hear better in noise (smaller critical ratios) than, for instance, canaries, under the same conditions of an open habitat canaries would a much smaller active vocal space than do budgerigars in the same amount of noise. The model used here is successful in predicting communication distance in a variety of environments and a variety of species. When this model is combined with commercial software (e.g., SoundPlan[20]) for predicting noise characteristics at different distances from a highway, a map can be made describing the bird's communication difficulty at any location from the highway.

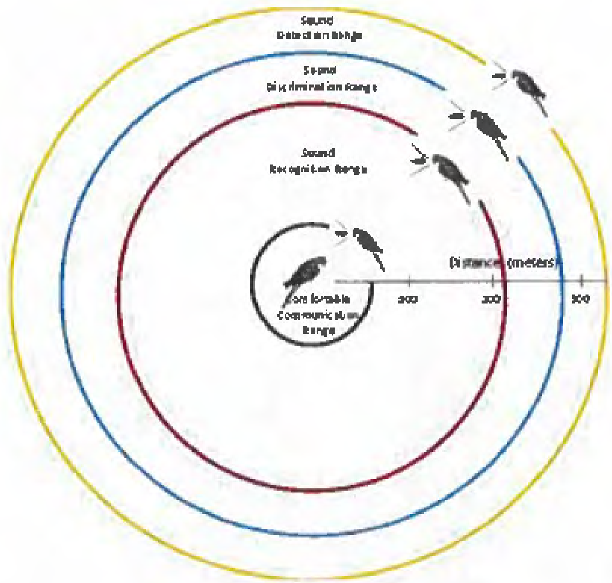


Figure 10: Diagrammatic Representation of Bird Communication

A diagrammatic representation of data in Figure 9 showing the quality of hearing for a bird in noise located at different distances from a sound-emitting bird. A bird can just hear a vocalizations (i.e., detect it) at a much greater distance than is required for comfortable communication. This represents the worst case scenario based on critical ratio data from the laboratory and does not include short-term adaptation strategies described earlier, which would improve communication.

Based on laboratory data, this Guidance Document recommends several guidelines—two dealing with hearing damage and threshold shift, one dealing with masking, and a fourth dealing with stress and annoyance. As illustrated in Figure 3, these guidelines are as follows.

(1) Received noise levels less than 110 dBA SPL continuous are extremely unlikely to cause hearing damage or permanent threshold shift in birds.

(2) Received continuous noise levels below 93 dBA SPL are unlikely to cause even temporary threshold shifts in birds. This value, based solely on bird studies, is in harmony with much of the literature on human hearing. Consider, for example, that OSHA standards require hearing conservation procedures only when noise levels in the workplace reach continuous levels of 85 dBA for 8 hours.

(3) At further distances from the highway, once the received level of traffic and construction noise falls below the ambient noise level (particularly in the region of 2-4 kHz), there is

[20] http://www.soundplan.eu/english/soundplan-acoustics/

little or no additional masking of communication signals beyond what already occurs from natural ambient noise.

(4) In the absence of empirical data from birds, received levels of traffic and construction noise known to annoy humans provide a useful interim guideline for the potential to cause physiological stress and behavioral disturbance in birds. Generally, construction noise, because it is both short term and more intermittent, is likely to have less of an effect that traffic noise. This is expected except in rare in cases where birds may remain in close proximity to very high level impulsive noise as from pile driving.

Two common sense guidelines also arise from review of the data on masking. First, the typical human listener can hear traffic and construction noise at distances 2–4 times greater than can the typical bird. It follows that traffic and construction noise from either traffic or construction activity that is just barely audible to humans at any given distance, almost certainly cannot be heard by birds at the same distance. Second, the converse is also true, if a human listener can barely hear a bird singing against a background of traffic and construction noise, masking data suggest that another bird would have to half again as close to singing bird in order to hear it. In this case, using human hearing as a guide underestimates the effects of noise on bird communication.

5. Summary and Overview of the Effects of Traffic Noise on Birds

1) Stress and physiological effects:
   a) There are no studies definitively identifying traffic noise as the critical variable affect bird behavior near roadways and highways.
   b) There are well-documented adverse effects of sustained traffic noise on humans, including stress, physiological and sleep disturbances, and changes in feelings of well-being that may be applicable, when viewed with care, to birds.
   c) Traffic/construction noise below the bird's masked threshold has no effect.

2) Acoustic over-exposure:
   a) Birds are more resistant to both temporary and permanent hearing loss or to hearing damage from acoustic overexposure than are humans and other animals that have been tested.
   b) Birds can regenerate the sensory hair cells of the inner ear, thereby providing a mechanism for recovering from intense acoustic over-exposure, a capability not found in mammals.
   c) The studies of acoustic over-exposure in birds have considerable relevance for estimating hearing damage effects of traffic noise, non-continuous construction noise, and for impulsive-type construction noise such as pile drivers.

3) Masking:
   a) Continuous noise of sufficient intensity in the frequency region of bird hearing can have a detrimental effect on the detection and discrimination of vocal signals by birds.
   b) Noise in the spectral region of the vocalizations has a greater masking affect than noises outside this range. Thus, traffic noise will cause less masking than other environmental noises of equal overall level but that contain energy in a higher spectral region around 2–4 kHz (e.g., insects, vocalizations of other birds).

c) Generally, human auditory thresholds in quiet and in noise are better than that of the typical bird, which leads to the following conclusions:
   (1) The typical human will be able to hear single vehicle, traffic noise, and construction noise at a much greater distance from the roadway than will the typical bird, thereby providing a valuable, common sense, easy-to-apply, risk criterion.
   (2) However, the typical human will also be able to hear a bird vocalizing in a noisy environment at twice the distance that a typical bird, meaning that relying on human hearing underestimates the effects of noise on bird communication.

d) From knowledge of: (i) bird hearing in quiet and noise, (ii) the Inverse Square Law, (iii) Excess Attenuation in a particular environment, and (iv) species-specific acoustic characteristics of vocalizations, reasonable predictions can be made about possible maximum communication distances between two birds in continuous noise.

e) The amount of masking of vocalizations can be predicted from the peak in the total power spectrum of the vocalization and the bird's critical ratio (i.e., signal-to-noise ratio) at that frequency of peak energy.

f) Birds, like humans and other animals, employ a range of short-term behavioral strategies, or adaptations, for communicating in noise, resulting in a doubling to quadrupling of the efficiency of hearing in noise.

4) Dynamic behavioral and population effects:
   a) Any components of traffic noise that are audible to birds may have effects independent of and beyond the effects listed above. At distances from the roadway where traffic noise levels fall below ambient noise levels in the spectral region for vocal communication (i.e., 2–8 kHz), low level but audible sound in non-communication frequencies (e.g., the rumbling of a truck) can potentially cause may cause physiological or behavioral responses). Beyond effects due specifically to traffic noise, since the more recent literature points to noise as possibly having wide ranging effects on birds, consideration must be given to the additive effects of traffic noise and environmental noise.

5) Extrapolation of data from humans and birds to other species:
   a) Since there is substantial variation in bird hearing and behavior, considerable care must be taken when trying to extrapolate data between species, and particularly when the species have different hearing capabilities and acoustic behaviors.
   b) Data from humans has relevance to understanding effects of sound in birds. In particular, data on physiological effects in humans may have implications for birds, but additional study is needed.

6) Much more data are needed on:
   a) Physiological effects of sound on birds.
   b) How responses vary between species with regard to masking, hearing loss, and hearing recovery.
   c) Hearing in young animals and how this compares to that in adults.
   d) Additional, and carefully selected, species so there is a large enough database from which to allow extrapolation between species, and broader generalizations on effects of noise on birds.
   e) A broader range of studies, as discussed in detail in Appendix F.

6. Estimating Effects of Traffic Noise on Birds, Rationale, and *Interim* Guidelines

This Guidance Document has reviewed three classes of potential effects of traffic noise on birds. The basis of the guidelines for each class of effects differs. Table 3 and Figure 3 provide specific *interim* criteria.

1. *Behavioral and/or physiological effects*: There are no definitive studies showing that traffic noise exclusively (as opposed to correlated variables) has an adverse effect on birds. While a wealth of human data and experience suggest traffic noise could have a number of adverse effects, there are several studies (e.g., Awbrey *et al.*, 1995) showing that birds (as well as other animals) adapt quite well, and even appear sometimes to prefer, environments that include high levels of traffic noise. Given the lack of empirical data on this point, it is recommended using subjective human experience with the noise in question as an *interim* guideline to estimate acceptable noise levels for avoiding stress and physiological effects. Noise types and levels that appear to increase stress and adverse physiological reactions in humans may also have similar consequences in birds.

2. *Damage to hearing from acoustic overexposure*: In contrast to the above, there are many definitive studies showing the effects of intense noise on bird hearing and auditory structures. These extensive data show that birds are much more resistant to hearing loss and auditory damage from acoustic overexposure than are humans and other mammals. Traffic and construction noise, even at extreme levels, is unlikely to cause threshold shift, hearing loss, auditory damage, or damage to other organ systems in birds and, therefore, *interim* guidelines for hearing damage from traffic and construction noise are probably not needed. Construction noise, such as impulse noise from pile driving, does reach high levels and may be capable of causing damage to auditory structures in birds.

3. *Masking of communication signals and other biologically relevant sounds*: Many laboratory masking studies show precisely the effects of continuous noise (including traffic noise) on sound detection in over a dozen species of birds. These studies describe a sort of worst case scenario because the noise is continuous and the myriad of short-term adaptive behavioral responses for mitigating the effects of noise are not available to the bird in a laboratory test situation. These masking studies led to an overall noise level guideline of around 60 dBA for continuous noise. Since this 60 dBA criterion was developed, however, controlled laboratory and field studies have extended the range of species differences in signal-to-noise ratios as well as the gain in signal-to-noise ratio that occurs with various short-term, adaptive behavioral responses that birds might use in natural environments. Critical ratios vary across species as much as 10 dB, strongly suggesting that acoustic communication in some species might be affected by an overall traffic and construction noise level even less than 60 dBA, while others would not. For some other species, communication between individuals, especially if they can employ short-term behavioral strategies for hearing in noise, might be unaffected at even higher levels of noise perhaps approaching 70 dBA. These short term behavioral adaptations include scanning (head turning), raising vocal output, and changing singing location. Each of these strategies alone can result in a significant gain in signal level or signal-to-noise ratio of

about 10 dB (under masking conditions), and birds can employ all three strategies simultaneously.

4. *Practical guidelines arising from masking studies*: There is a common sense, extremely practical guideline that emerges from basic hearing knowledge of birds and humans. Specifically, the 6 dB difference in masking (critical ratio) functions between the typical bird and human listeners with normal hearing provide two common sense guidelines: (1) Humans can hear traffic noise, in a natural environment, at twice the distance from the roadway/highway than can birds. In other words, if in a natural environment, distant traffic noise is barely audible to humans, it is certainly inaudible to birds, and will have no effect on any aspect of their acoustic behavior. (2) Humans can hear a bird singing against a background of noise at twice the distance than can the typical bird. This provides an informal estimate of maximum communication distance between two birds vocalizing against a background of continuous traffic noise. This works not only for the typical bird, but it is probably also valid for most species.

These recommended guidelines for estimating effects that traffic noise has on masking in birds are *interim* guidelines for several reasons.

1. The *interim* guidelines are based on median data from masking studies from a limited number of the thousands of bird species. Thus, they represent the typical bird, based on the species studied. However, it is important to recall that bird species can vary considerably in how they hear in the presence of noise; some have masked thresholds that approach those of humans, while others have masked thresholds that are 3–4 dB worse than thresholds for the typical bird presented here. Therefore, final noise guidelines will require testing more species with appropriate experimental adjustment for the species in question.

2. Traffic noise characteristics are influenced by transmission through the environment, as are the spectral, temporal, and intensive aspects of bird vocalizations through differences in excess attenuation.

This page left intentionally blank

References Cited

Adler, H. J., Kenealy, J., Dedio, R. M., and Saunders, J. C. (1992). "Threshold shift, hair cell loss, and hair bundle stiffness following exposure to 120 and 125 dB pure tones in the neonatal chick," Acta oto-laryngologica 112, 444-454.

Adler, H. J., Poje, C. P., and Saunders, J. C. (1993). "Recovery of auditory function and structure in the chick after two intense pure tone exposures," Hearing research 71, 214-224.

AFWO (2006). "Estimating the effects of auditory and visual disturbance to northern spotted owls and marbelled murelетts in northewestern California," (U. S. Fish and Wildlife Service, Arcata, CA).

Aleksandrov, L., and Dmitrieva, L. (1992). "Development of auditory sensitivity of altricial birds: absolute thresholds of the generation of evoked potentials," Neuroscience and behavioral physiology 22, 132-137.

Arroyo-Solís, A., Castillo, J. M., Figueroa, E., López-Sánchez, J. L., and Slabbekoorn, H. (2013). "Experimental evidence for an impact of anthropogenic noise on dawn chorus timing in urban birds," Journal of Avian Biology 44, 288-296.

Awbrey, F. T., Hunsacker, D., and Church, R. (1995). "Acoustical responses of California gnatcatchers to traffic noise," Internoise, 971-974.

Barber, J. R., Crooks, K. R., and Fristrup, K. M. (2010). "The costs of chronic noise exposure for terrestrial organisms," Trends in ecology & evolution 25, 180-189.

Barrett, D. E. (1996). "Traffic-noise impact study for Least Bell's vireo habitat along California State Route 83," Transportation Research Record: Journal of the Transportation Research Board 1559, 3-7.

Barton, L., Bailey, E., and Gatehouse, R. W. (1984). "Audibility curve of bobwhite quail (Colinus virginianus)," Journal of Auditory Research.

Bennett, W. M., Falter, C. M., Chipps, S. R., Niemela, K., and Kinney, J. (1994). "Effects of underwater sound stimulating the intermediate scale measurement system on fish and zooplankton of Lake Pend Oreille, Idaho," (Arlington, Virginia).

Brackenbury, J. (1979a). "Corrections to the Hazelhoff model of airflow in the avian lung," Respiration physiology 36, 143-154.

Brackenbury, J. (1979b). "Power capabilities of the avian sound-producing system," The Journal of experimental biology 78, 163-166.

Bregman, A. S. (1990). *Auditory scene analysis: The perceptual organization of sound* (MIT Press, Cambridge, Massachusetts).

Brenowitz, E. A. (1982). "The active space of red-winged blackbird song," Journal of Comparative Physiology 147, 511-522.

Brittan-Powell, E. F., and Dooling, R. J. (2004). "Development of auditory sensitivity in budgerigars (Melopsittacus undulatus)," The Journal of the Acoustical Society of America 115, 3092-3102.

Brown, A. (1990). "Measuring the effect of aircraft noise on sea birds," Environment International 16, 587-592.

Brumm, H. (2004). "The impact of environmental noise on song amplitude in a territorial bird," Journal of Animal Ecology 73, 434-440.

Brumm, H., and Slabbekoorn, H. (2005). "Acoustic communication in noise," Advances in Behavior, 151-209.

Brumm, H., and Todt, D. (2002). "Noise-dependent song amplitude regulation in a territorial songbird," Animal behaviour 63, 891-897.
Brumm, H., and Todt, D. (2003). "Facing the rival: directional singing behaviour in nightingales," Behaviour 140, 43-53.
Bunnell, F. L., Dunbar, D., Koza, L., and Ryder, G. (1981). "Effects of disturbance on the productivity and numbers of white pelicans in British Columbia: observations and models," Colonial Waterbirds, 2-11.
Burger, J. (1983). "Bird control at airports," Environmental Conservation 10, 115-124.
Caltrans (2011). "Traffic Noise Analysis Protocol for New Highway Construction, Reconstruction, and Retrofit Barrier Projects."
Caltans (2013). *Technical Noise Supplement: TeNS: a Technical Supplement to the Traffic Noise Analysis Protocol* (California Department of Transportation, Environmental Program, Environmental Engineering-Noise, Air Quality, and Hazardous Waste Management Office).
Cardoso, G. C., and Atwell, J. W. (2011). "On the relation between loudness and the increased song frequency of urban birds," Animal behaviour 82, 831-836.
Clark, W. D., and Karr, J. R. (1979). "Effects of highways on red-winged blackbird and horned lark populations," The Wilson Bulletin, 143-145.
Clench-Aas, J., Bartonova, A., Klæboe, R., and Kolbenstvedt, M. (2000). "Oslo traffic study–part 2:: quantifying effects of traffic measures using individual exposure modeling," Atmospheric environment 34, 4737-4744.
Cody, M. L., and Brown, J. H. (1969). "Song asynchrony in neighbouring bird species," Nature 222, 778-780.
Cohen, S. M., Stebbins, W. C., and Moody, D. B. (1978). "Audibility thresholds of the blue jay," The Auk, 563-568.
Coles, R. B., KONISHI, M., and Pettigrew, J. D. (1987). "Hearing and echolocation in the Australian grey swiftlet, Collocalia spodiopygia," Journal of experimental biology 129, 365-371.
Cotanche, D. A. (1998). "Structural recovery from sound and aminoglycoside damage in the avian cochlea," Audiology & neuro-otology 4, 271-285.
Crino, O. L., Johnson, E. E., Blickley, J. L., Patricelli, G. L., and Breuner, C. W. (2013). "Effects of experimentally elevated traffic noise on nestling white-crowned sparrow stress physiology, immune function and life history," The Journal of experimental biology 216, 2055-2062.
Cynx, J., Lewis, R., Tavel, B., and Tse, H. (1998). "Amplitude regulation of vocalizations in noise by a songbird, Taeniopygia guttata," Animal behaviour 56, 107-113.
Dabelsteen, T., Larsen, O. N., and Pedersen, S. B. (1993). "Habitat-induced degradation of sound signals: Quantifying the effects of communication sounds and bird location on blur ratio, excess attenuation, and signal-to-noise ratio in blackbird song," The Journal of the Acoustical Society of America 93, 2206-2220.
Dent, M. L., Larsen, O. N., and Dooling, R. J. (1997). "Free-field binaural unmasking in budgerigars (Melopsittacus undulatus)," Behavioral Neuroscience 111, 590.
Díaz, M., Parra, A., and Gallardo, C. (2011). "Serins respond to anthropogenic noise by increasing vocal activity," Behavioral Ecology 22, 332-336.
Dooling, R. J. (1980). "Behavior and psychophysics of hearing in birds," in *Comparative studies of hearing in vertebrates* (Springer), pp. 261-288.

Dooling, R. J. (1982). "Auditory perception in birds," in *Acoustic communication in birds*, edited by D. E. Kroodsma, and E. H. Miller (Academic Press, New York), pp. 95-130.

Dooling, R. J. (1992). "Hearing in birds," in *The evolutionary biology of hearing* (Springer), pp. 545-559.

Dooling, R. J., and Blumenrath, S. H. (2014). "Masking Experiments in Humans and Birds Using Anthropogenic Noises " in *The Effects of Noise on Aquatic Life, II*, edited by A. N. Popper, and A. D. Hawkins (Springer Science+Business Media, New York), p. in press.

Dooling, R. J., Brown, S. D., Klump, G. M., and Okanoya, K. (1992). "Auditory perception of conspecific and heterospecific vocalizations in birds: Evidence for special processes," Journal of comparative psychology 106, 20.

Dooling, R. J., Dent, M. L., Lauer, A. M., and Ryals, B. M. (2008). "Functional recovery after hair cell regeneration in birds," in *Hair cell regeneration, repair, and protection* (Springer), pp. 117-140.

Dooling, R. J., Fay, R. R., and Popper, A. N. (2000a). *Comparative hearing. Birds and reptiles* (Springer, New York).

Dooling, R. J., Lohr, B., and Dent, M. L. (2000b). "Hearing in birds and reptiles," in *Comparative hearing: Birds and reptiles* (Springer), pp. 308-359.

Dooling, R. J., Okanoya, K., Downing, J., and Hulse, S. (1986). "Hearing in the starling (Sturnus vulgaris): Absolute thresholds and critical ratios," Bulletin of the psychonomic Society 24, 462-464.

Dooling, R. J., Peters, S. S., and Searcy, M. H. (1979). "Auditory sensitivity and vocalizations of the field sparrow (Spizella pusilla)," Bulletin of the Psychonomic Society 14, 106-108.

Dooling, R. J., and Popper, A. N. (2000). "Hearing in birds and reptiles: An overview," in *Comparative Hearing: Birds and Reptiles* (Springer), pp. 1-12.

Dooling, R. J., and Popper, A. N. (2007). "The effects of highway noise on birds," in *See http://www.dot.ca.gov/hq/env/bio/files/caltrans_birds_10-7-2007b.pdf.*

Dooling, R. J., and Saunders, J. C. (1974). "Threshold shift produced by continuous noise exposure in the parakeet (Melopsittacus undulatus)," The Journal of the Acoustical Society of America 55, S77-S77.

Dooling, R. J., and Saunders, J. C. (1975). "Hearing in the parakeet (Melopsittacus undulatus): absolute thresholds, critical ratios, frequency difference limens, and vocalizations," Journal of comparative and physiological psychology 88, 1-20.

Dooling, R. J., West, E. W., and Leek, M. R. (2009). "Conceptual and computation models of the effects of anthropogenic sound on birds," in *Proceedings of the Institute of Acoustics*.

Dooling, R. J., Zoloth, S. R., and Baylis, J. R. (1978). "Auditory sensitivity, equal loudness, temporal resolving power, and vocalizations in the house finch (Carpodacus mexicanus)," Journal of comparative and physiological psychology 92, 867.

Douglas, H. D. I., and Conner, W. E. (1999). "Is there a sound reception window in coastal environments? Evidence from shorebird communication systems," Naturwissenschaften 86, 228-230.

Dowling, J. L., Luther, D. A., and Marra, P. P. (2011). "Comparative effects of urban development and anthropogenic noise on bird songs," Behavioral Ecology.

Dubois, A., and Martens, J. (1984). "A case of possible vocal convergence between frogs and a bird in Himalayan torrents," Journal für Ornithologie 125, 455-463.

Dyson, M., Klump, G., and Gauger, B. (1998). "Absolute hearing thresholds and critical masking ratios in the European barn owl: a comparison with other owls," Journal of Comparative Physiology A 182, 695-702.
Evans, C. S. (1991). "Of ducklings and Turing machines: interactive playbacks enhance subsequent responsiveness to conspecific calls," Ethology 89, 125-134.
Fay, R. R. (1988). *Hearing in vertebrates: A psychophysics databook* (Hill-Fay Associates, Winnetka, IL).
Fernández-juricic, E. (2001). "Avian spatial segregation at edges and interiors of urban parks in Madrid, Spain," Biodiversity & Conservation 10, 1303-1316.
Ferris, C. R. (1979). "Effects of Interstate 95 on breeding birds in northern Maine," The Journal of Wildlife Management, 421-427.
FHWA (2006). "Roadway Construction Noise Model User's Guide ".
Ficken, R. W., Popp, J. W., and Matthiae, P. E. (1985). "Avoidance of acoustic interference by ovenbirds," The Wilson Bulletin, 569-571.
Fletcher, H. (1940). "Auditory patterns," Reviews of modern physics 12, 47.
Foppen, R., and Reijnen, R. (1994). "The effects of car traffic on breeding bird populations in woodland. II. Breeding dispersal of male willow warblers (Phylloscopus trochilus) in relation to the proximity of a highway," Journal of Applied Ecology, 95-101.
Forman, R. T., Reineking, B., and Hersperger, A. M. (2002). "Road traffic and nearby grassland bird patterns in a suburbanizing landscape," Environmental management 29, 782-800.
Francis, C. D., Ortega, C. P., and Cruz, A. (2009). "Noise pollution changes avian communities and species interactions," Current biology : CB 19, 1415-1419.
Francis, C. D., Ortega, C. P., and Cruz, A. (2011a). "Different behavioural responses to anthropogenic noise by two closely related passerine birds," Biology letters 7, 850-852.
Francis, C. D., Ortega, C. P., and Cruz, A. (2011b). "Vocal frequency change reflects different responses to anthropogenic noise in two suboscine tyrant flycatchers," Proceedings. Biological sciences / The Royal Society 278, 2025-2031.
Freyaldenhoven, M. C., Fisher Smiley, D., Muenchen, R. A., and Konrad, T. N. (2006). "Acceptable noise level: Reliability measures and comparison to preference for background sounds," Journal of the American Academy of Audiology 17, 640-648.
Gleich, O., Dooling, R. J., and Manley, G. A. (2005). "Audiogram, body mass, and basilar papilla length: correlations in birds and predictions for extinct archosaurs," Naturwissenschaften 92, 595-598.
Gleich, O., and Manley, G. A. (2000). "The hearing organ of birds and crocodilia," in *Comparative hearing: Birds and reptiles* (Springer), pp. 70-138.
Gray, L., and Rubel, E. W. (1985). "Development of absolute thresholds in chickens," The Journal of the Acoustical Society of America 77, 1162-1172.
Greenwood, D. D. (1961a). "Auditory masking and the critical band," The Journal of the Acoustical Society of America 33, 484-502.
Greenwood, D. D. (1961b). "Critical bandwidth and the frequency coordinates of the basilar membrane," The Journal of the Acoustical Society of America 33, 1344-1356.
Halfwerk, W., Bot, S., Buikx, J., van der Velde, M., Komdeur, J., ten Cate, C., and Slabbekoorn, H. (2011). "Low-frequency songs lose their potency in noisy urban conditions," Proceedings of the National Academy of Sciences of the United States of America 108, 14549-14554.

Halfwerk, W., Bot, S., and Slabbekoorn, H. (2012). "Male great tit song perch selection in response to noise-dependent female feedback," Functional Ecology 26, 1339-1347.
Hanna, D., Blouin-Demers, G., Wilson, D. R., and Mennill, D. J. (2011). "Anthropogenic noise affects song structure in red-winged blackbirds (Agelaius phoeniceus)," The Journal of experimental biology 214, 3549-3556.
Hashino, E., and Okanoya, K. (1989). "Auditory sensitivity in the zebra finch (Poephila guttata castanotis)," Journal of the Acoustical Society of Japan (E) 10, 51-52.
Hashino, E., Sokabe, M., and Miyamoto, K. (1988). "Frequency specific susceptibility to acoustic trauma in the budgerigar (Melopsittacusundulatus)," The Journal of the Acoustical Society of America 83, 2450-2453.
Heise, G. A. (1953). "Auditory thresholds in the pigeon," The American journal of psychology, 1-19.
Henderson, D., and Hamernik, R. (1986). "Impulse noise: critical review," The Journal of the Acoustical Society of America 80, 569-584.
Hienz, R. D., and Sachs, M. B. (1987). "Effects of noise on pure-tone thresholds in blackbirds (Agelaius phoeniceus and Molothrus ater) and pigeons (Columba livia)," Journal of comparative psychology 101, 16-24.
Hienz, R. D., Sinnott, J., and Sachs, M. (1977). "Auditory sensitivity of the redwing Blackbird (Agelaius phoeniceus) and brown-headed cowbird (Molothrus ater)," Journal of comparative and physiological psychology 91, 1365.
Holland, J., Dabelsteen, T., Pedersen, S. B., and Larsen, O. N. (1998). "Degradation of wren Troglodytes troglodytes song: implications for information transfer and ranging," The Journal of the Acoustical Society of America 103, 2154-2166.
Hu, Y., and Cardoso, G. C. (2009). "Are bird species that vocalize at higher frequencies preadapted to inhabit noisy urban areas?," Behavioral Ecology 20, 1268-1273.
Hu, Y., and Cardoso, G. C. (2010). "Which birds adjust the frequency of vocalizations in urban noise?," Animal behaviour 79, 863-867.
Huet des Aunay, G., Slabbekoorn, H., Nagle, L., Passas, F., Nicolas, P., and Draganoiu, T. I. (2014). "Urban noise undermines female sexual preferences for low-frequency songs in domestic canaries," Animal behaviour 87, 67-75.
Hultsch, H., and Todt, D. (1996). "Rules of parameter variation in homotype series of birdsong can indicate a 'sollwert'significance," Behavioural processes 38, 175-182.
Kane, A. S., Song, J., Halvorsen, M. B., Miller, D. L., Salierno, J. D., Wysocki, L. E., Zeddies, D., and Popper, A. N. (2010). "Exposure of fish to high-intensity sonar does not induce acute pathology," Journal of fish biology 76, 1825-1840.
Kaseloo, P. A. (2005). "Synthesis of noise effects on wildlife populations," Road Ecology Center.
Klump, G., Kretzschmar, E., and Curio, E. (1986). "The hearing of an avian predator and its avian prey," Behavioral ecology and sociobiology 18, 317-323.
Klump, G. M., Gleich, O., and Langemann, U. (1995). "An excitation-pattern model for the starling (Sturnusvulgaris)," The Journal of the Acoustical Society of America 98, 112-124.
Konishi, M. (1969). "Time resolution by single auditory neurones in birds."
Konishi, M. (1970). "Comparative neurophysiological studies of hearing and vocalizations in songbirds," Journal of Comparative Physiology A: Neuroethology, Sensory, Neural, and Behavioral Physiology 66, 257-272.

Konishi, M. (1973). "Locatable and nonlocatable acoustic signals for barn owls," American Naturalist, 775-785.
Konishi, M., and Knudsen, E. I. (1979). "The oilbird: hearing and echolocation," Science, 425-427.
Kuhn, A., Müller, C., Leppelsack, H.-J., and Schwartzkopff, J. (1982). "Heart-rate conditioning used for determination of auditory threshold in the starling," Naturwissenschaften 69, 245-246.
Kuitunen, M., Rossi, E., and Stenroos, A. (1998). "Do highways influence density of land birds?," Environmental management 22, 297-302.
Langemann, U., Gauger, B., and Klump, G. M. (1998). "Auditory sensitivity in the great tit: perception of signals in the presence and absence of noise," Animal behaviour 56, 763-769.
Le Prell, C. G., Henderson, D., Fay, R. R., and Popper, A. N. (2012). "Noise-induced hearing loss: Scientific advances," edited by C. G. Le Prell, D. Henderson, R. R. Fay, and A. N. Popper (Springer Science+Business Media, LLC, New York).
Lee, C. S., and Fleming, G. G. (1996). "Measurement of highway-related noise," (Citeseer).
Lengagne, T., Aubin, T., Lauga, J., and Jouventin, P. (1999). "How do king penguins (Aptenodytes patagonicus apply the mathematical theory of information to communicate in windy conditions?," Proceedings of the Royal Society of London. Series B: Biological Sciences 266, 1623-1628.
Leonard, M. L., and Horn, A. G. (2008). "Does ambient noise affect growth and begging call structure in nestling birds?," Behavioral Ecology 19, 502-507.
Llacuna, S., Gorriz, A., Riera, M., and Nadal, J. (1996). "Effects of air pollution on hematological parameters in passerine birds," Archives of environmental contamination and toxicology 31, 148-152.
Lohr, B., Lauer, A., Newman, M. R., and Dooling, R. J. (2004). "Hearing in the red-billed firefinch Lagonosticta senegala and the Spanish timbrado canary Serinus canaria: the influence of natural and artificial selection on auditory abilities and vocal structure," Bioacoustics 14, 83-98.
Lohr, B., Wright, T. F., and Dooling, R. J. (2003). "Detection and discrimination of natural calls in masking noise by birds: estimating the active space of a signal," Animal behaviour 65, 763-777.
Luther, D., and Baptista, L. (2010a). "Urban noise and the cultural evolution of bird songs," Proceedings of the Royal Society B: Biological Sciences 277, 469-473.
Luther, D., and Baptista, L. (2010b). "Urban noise and the cultural evolution of bird songs," Proceedings. Biological sciences / The Royal Society 277, 469-473.
Luther, D., and Magnotti, J. (2014). "Can animals detect differences in vocalizations adjusted for anthropogenic noise?," Animal behaviour 92, 111-116.
Luther, D. A., and Derryberry, E. P. (2012). "Birdsongs keep pace with city life: changes in song over time in an urban songbird affects communication," Animal behaviour 83, 1059-1066.
Luz, G. A., and Hodge, D. C. (1971). "Recovery from Impulse-Noise Induced TTS in Monkeys and Men: A Descriptive Model," The Journal of the Acoustical Society of America 49, 1770-1777.
Maiorana, V., and Schleidt, W. (1972). "The auditory sensitivity of the turkey," J. Aud. Res 12, 203-207.

Manabe, K. (1997). "Vocal plasticity in budgerigars: Various modifications of vocalization by operant conditioning," BIOMEDICAL RESEARCH-TOKYO 18, 125-132.
Manabe, K., Sadr, E. I., and Dooling, R. J. (1998). "Control of vocal intensity in budgerigars (Melopsittacus undulatus): differential reinforcement of vocal intensity and the Lombard effect," The Journal of the Acoustical Society of America 103, 1190-1198.
Manley, G. A. (2000). "The hearing organs of lizards," in *Comparative hearing: Birds and reptiles* (Springer), pp. 139-196.
Manley, G. A. (2010). "An evolutionary perspective on middle ears," Hearing research 263, 3-8.
Manley, G. A., Köppl, C., and Yates, G. K. (1997). "Activity of primary auditory neurons in the cochlear ganglion of the emu Dromaius novaehollandiae: spontaneous discharge, frequency tuning, and phase locking," The Journal of the Acoustical Society of America 101, 1560-1573.
Marten, K., and Marler, P. (1977). "Sound transmission and its significance for animal vocalization," Behavioral ecology and sociobiology 2, 271-290.
Mathevon, N., Aubin, T., and Dabelsteen, T. (1996). "Song degradation during propagation: importance of song post for the wren Troglodytes troglodytes," Ethology 102, 397-412.
McFadden, E. A., and Saunders, J. C. (1989). "Recovery of auditory function following intense sound exposure in the neonatal chick," Hearing research 41, 205-215.
McMullen, H., Schmidt, R., and Kunc, H. P. (2014). "Anthropogenic noise affects vocal interactions," Behavioural processes 103, 125-128.
Meillère, A., Brischoux, F., Ribout, C., and Angelier, F. (2015). "Traffic noise exposure affects telomere length in nestling house sparrows," Biology letters 11.
Merchant, N. D., Fristrup, K. M., Johnson, M. P., Tyack, P. L., Witt, M. J., Blondel, P., and Parks, S. E. (2015). "Measuring acoustic habitats," Methods in Ecology and Evolution 6, 257-265.
Miller, J. D. (1974). "Effects of noise on people," The Journal of the Acoustical Society of America 56, 729-764.
Morton, E. S. (1975). "Ecological sources of selection on avian sounds," American Naturalist, 17-34.
Murphy, E., and King, E. (2014). *Environmental Noise Pollution: Noise Mapping, Public Health, and Policy* (Elsevier, Burlington, MA).
Nemeth, E., and Brumm, H. (2009). "Blackbirds sing higher-pitched songs in cities: adaptation to habitat acoustics or side-effect of urbanization?," Animal behaviour 78, 637-641.
Nemeth, E., and Brumm, H. (2010a). "Birds and anthropogenic noise: are urban songs adaptive?," The American naturalist 176, 465-475.
Nemeth, E., and Brumm, H. (2010b). "Birds and Anthropogenic Noise: Are Urban Songs Adaptive?," The American naturalist 176, 465-475.
Nemeth, E., Pieretti, N., Zollinger, S. A., Geberzahn, N., Partecke, J., Miranda, A. C., and Brumm, H. (2013a). *Bird song and anthropogenic noise: vocal constraints may explain why birds sing higher-frequency songs in cities*.
Nemeth, E., Pieretti, N., Zollinger, S. A., Geberzahn, N., Partecke, J., Miranda, A. C., and Brumm, H. (2013b). "Bird song and anthropogenic noise: vocal constraints may explain why birds sing higher-frequency songs in cities," Proceedings. Biological sciences / The Royal Society 280, 20122798.
Nieboer, E., and Van der Paardt, M. (1976). "Hearing of the African woodowl, Strix woodfordii," Netherlands Journal of Zoology 27, 227-229.

Niemiec, A. J., Raphael, Y., and Moody, D. B. (1994). "Return of auditory function following structural regeneration after acoustic trauma: behavioral measures from quail," Hearing research 79, 1-16.
Öhrström, E., and Björkman, M. (1983). "Sleep disturbance before and after traffic noise attenuation in an apartment building," The Journal of the Acoustical Society of America 73, 877-879.
Öhrström, E., and Rylander, R. (1982). "Sleep disturbance effects of traffic noise—a laboratory study on after effects," Journal of Sound and Vibration 84, 87-103.
Okanoya, K., and Dooling, R. J. (1987). "Hearing in passerine and psittacine birds: a comparative study of absolute and masked auditory thresholds," Journal of comparative psychology 101, 7.
Okanoya, K., and Dooling, R. J. (1988). "Hearing in the swamp sparrow, Melospiza georgiana, and the song sparrow, Melospiza melodia," Animal behaviour 36, 726-732.
Ortega, C. P. (2012). "Effects of noise pollution on birds: a brief review of our knowledge," Ornithological Monographs 2012, 6-22.
Osmanski, M., and Dooling, R. (2006). "Auditory feedback of vocal production in budgerigars using earphones," The Journal of the Acoustical Society of America 119, 3350-3350.
Osmanski, M. S., and Dooling, R. J. (2009). "The effect of altered auditory feedback on control of vocal production in budgerigars (Melopsittacus undulatus)," The Journal of the Acoustical Society of America 126, 911-919.
Ouis, D. (2001). "Annoyance from road traffic noise: a review," Journal of environmental psychology 21, 101-120.
Peris, S., and Pescador, M. (2004). "Effects of traffic noise on paserine populations in Mediterranean wooded pastures," Applied Acoustics 65, 357-366.
Pettigrew, J., Larsen, O., Rowe, M., and Aitkin, L. (1990). "Directional hearing in the Plains-wanderer, Pedionomus torquatus," Information Processing in Mammalian Auditory and Tactile Systems, 179-190.
Pijanowski, B. C., Farina, A., Gage, S. H., Dumyahn, S. L., and Krause, B. L. (2011a). "What is soundscape ecology? An introduction and overview of an emerging new science," Landscape ecology 26, 1213-1232.
Pijanowski, B. C., Villanueva-Rivera, L. J., Dumyahn, S. L., Farina, A., Krause, B. L., Napoletano, B. M., Gage, S. H., and Pieretti, N. (2011b). "Soundscape ecology: the science of sound in the landscape," BioScience 61, 203-216.
Pohl, N. U., Leadbeater, E., Slabbekoorn, H., Klump, G. M., and Langemann, U. (2012). "Great tits in urban noise benefit from high frequencies in song detection and discrimination," Animal behaviour 83, 711-721.
Popp, J. W., and Ficken, R. W. (1987). "Effects of non-specific singing on the song of the ovenbird," Bird Behavior 7, 22-26.
Popp, J. W., Ficken, R. W., and Reinartz, J. A. (1985). "Short-term temporal avoidance of interspecific acoustic interference among forest birds," The Auk, 744-748.
Popper, A. N., and Fay, R. R. (2005). *Sound source localization* (Springer, New York).
Potash, L. (1972). "Noise-induced changes in calls of the Japanese quail," Psychonomic Science 26, 252-254.
Potvin, D. A., and MacDougall-Shackleton, S. A. (2015). "Experimental chronic noise exposure affects adult song in zebra finches," Animal behaviour 107, 201-207.

Potvin, D. A., and Mulder, R. A. (2013). "Immediate, independent adjustment of call pitch and amplitude in response to varying background noise by silvereyes (Zosterops lateralis)," Behavioral Ecology.

Proppe, D. S., Avey, M. T., Hoeschele, M., Moscicki, M. K., Farrell, T., St Clair, C. C., and Sturdy, C. B. (2012). "Black-capped chickadees Poecile atricapillus sing at higher pitches with elevated anthropogenic noise, but not with decreasing canopy cover," Journal of Avian Biology 43, 325-332.

Proppe, D. S., Sturdy, C. B., and St. Clair, C. C. (2011). "Flexibility in Animal Signals Facilitates Adaptation to Rapidly Changing Environments," PloS one 6, e25413.

Proppe, D. S., Sturdy, C. B., and St. Clair, C. C. (2013). "Anthropogenic noise decreases urban songbird diversity and may contribute to homogenization," Global change biology 19, 1075-1084.

Pugliano, F. A., Pribitikin, E., Adler, H. J., and Saunders, J. C. (1993). "Growth of evoked potential amplitude in neonatal chicks exposed to intense sound," Acta oto-laryngologica 113, 18-25.

Redondo, P., Barrantes, G., and Sandoval, L. (2013). "Urban noise influences vocalization structure in the House Wren Troglodytes aedon," Ibis 155, 621-625.

Reijnen, M. J. S. M., Veenbaas, G., and Foppen, R. P. B. (1998). "Predicting the effects of motorway traffic on breeding bird populations."

Reijnen, R., and Foppen, R. (1994). "The effects of car traffic on breeding bird populations in woodland. I. Evidence of reduced habitat quality for willow warblers (Phylloscopus trochilus) breeding close to a highway," Journal of Applied Ecology, 85-94.

Reijnen, R., and Foppen, R. (1995). "The effects of car traffic on breeding bird populations in woodland. IV. Influence of population size on the reduction of density close to a highway," Journal of Applied Ecology, 481-491.

Reijnen, R., Foppen, R., Braak, C. T., and Thissen, J. (1995). "The effects of car traffic on breeding bird populations in woodland. III. Reduction of density in relation to the proximity of main roads," Journal of Applied ecology, 187-202.

Ryals, B. M., Dooling, R. J., Westbrook, E., Dent, M. L., MacKenzie, A., and Larsen, O. N. (1999). "Avian species differences in susceptibility to noise exposure," Hearing research 131, 71-88.

Ryals, B. M., and Rubel, E. W. (1985a). "Differential susceptibility of avian hair cells to acoustic trauma," Hearing research 19, 73-84.

Ryals, B. M., and Rubel, E. W. (1985b). "Ontogenetic changes in the position of hair cell loss after acoustic overstimulation in avian basilar papilla," Hearing research 19, 135-142.

Ryan, M. J., and Brenowitz, E. A. (1985). "The role of body size, phylogeny, and ambient noise in the evolution of bird song," American Naturalist, 87-100.

Salvi, R., Popper, A. N., and Fay, R. R. (2008). *Hair cell regeneration, repair, and protection* (Springer, New York, N.Y.).

Sarigul-Klijn, N., Karnopp, D., and Team, B. R. (1997). *Environmental Effects of Transportation Noise: A Case Study: Noise Criteria for the Protection of Endangered Passerine Birds* (Transportation Noise Control Center, Department of Mechanical and Aeronautical Engineering, University of California, Davis).

Saunders, J., and Dooling, R. (1974). "Noise-induced threshold shift in the parakeet (Melopsittacus undulatus)," Proceedings of the National Academy of Sciences 71, 1962-1965.

Saunders, J. C., Cohen, Y. E., and Szymko, Y. M. (1991). "The structural and functional consequences of acoustic injury in the cochlea and peripheral auditory system: a five year update," The Journal of the Acoustical Society of America 90, 136-146.
Saunders, J. C., Duncan, R. K., Doan, D. E., and Werner, Y. L. (2000). "The middle ear of reptiles and birds," in *Comparative Hearing: Birds and Reptiles*, edited by R. J. Dooling, R. R. Fay, and A. N. Popper (Springer-Verlag, New York), pp. 13-69.
Saunders, J. C., and Henry, W. J. (2014). "The Peripheral Auditory System in Birds: Structural," in *The Comparative Psychology of Audition: Perceiving Complex Sounds*, edited by R. J. Dooling, and S. Hulse (Psychology Press, New York), pp. 35-66.
Saunders, J. C., and Pallone, R. L. (1980). "Frequency selectivity in the parakeet studied by isointensity masking contours," The Journal of experimental biology 87, 331-342.
Saunders, J. C., Rintelmann, W. F., and Bock, G. R. (1979). "Frequency selectivity in bird and man: A comparison among critical ratios, critical bands and psychophysical tuning curves," Hearing research 1, 303-323.
Saunders, J. C., and Salvi, R. J. (2008). "Recovery of function in the avian auditory system after ototrauma," in *Hair Cell Regeneration, Repair, and Protection* (Springer), pp. 77-116.
Saunders, S. S., and Salvi, R. J. (1993). "Psychoacoustics of normal adult chickens: thresholds and temporal integration," The Journal of the Acoustical Society of America 94, 83-90.
Schroeder, J., Nakagawa, S., Cleasby, I. R., and Burke, T. (2012). "Passerine Birds Breeding under Chronic Noise Experience Reduced Fitness," PloS one 7, e39200.
Schwartzkopff, J. (1949). "Über Sitz und Leistung von Gehör und Vibrationssinn bei Vögeln," Zeitschrift für vergleichende Physiologie 31, 527-608.
Seger-Fullam, K. D., Rodewald, A. D., and Soha, J. A. (2011). "Urban noise predicts song frequency in northern cardinals and American robins," Bioacoustics 20, 267-276.
Shamoun-Baranes, J., Dokter, A. M., van Gasteren, H., van Loon, E. E., Leijnse, H., and Bouten, W. (2011). "Birds flee en mass from New Year's Eve fireworks," Behavioral Ecology 22, 1173-1177.
Slabbekoorn, H. (2004). "Habitat-dependent ambient noise: consistent spectral profiles in two African forest types," The Journal of the Acoustical Society of America 116, 3727-3733.
Slabbekoorn, H., and Peet, M. (2003). "Ecology: Birds sing at a higher pitch in urban noise," Nature 424, 267.
Slabbekoorn, H., and Ripmeester, E. A. P. (2008). "Birdsong and anthropogenic noise: implications and applications for conservation," Molecular ecology 17, 72-83.
Slabbekoorn, H., and Smith, T. B. (2002). "Habitat-dependent song divergence in the little greenbul: an analysis of environmental selection pressures on acoustic signals," Evolution; international journal of organic evolution 56, 1849-1858.
Slabbekoorn, H., Yang, X. J., and Halfwerk, W. (2012). "Birds and anthropogenic noise: singing higher may matter," The American naturalist 180, 142-145; author reply 146-152.
Smith, C. A. (1985). "Inner ear," Form and function in birds 3, 273-310.
Smith, T. B., Harrigan, R. J., Kirschel, A. N., Buermann, W., Saatchi, S., Blumstein, D. T., de Kort, S. R., and Slabbekoorn, H. (2013). "Predicting bird song from space," Evolutionary applications 6, 865-874.
Smith, T. B., Mila, B., Grether, G. F., Slabbekoorn, H., Sepil, I., Buermann, W., Saatchi, S., and Pollinger, J. P. (2008). "Evolutionary consequences of human disturbance in a rainforest bird species from Central Africa," Molecular ecology 17, 58-71.

Stansfeld, S. A., Berglund, B., Clark, C., Lopez-Barrio, I., Fischer, P., Öhrström, E., Haines, M. M., Head, J., Hygge, S., and Van Kamp, I. (2005). "Aircraft and road traffic noise and children's cognition and health: a cross-national study," The Lancet 365, 1942-1949.
Stone, E. (2000). "Separating the noise from the noise: a finding in support of the "niche hypothesis," that birds are influenced by human-induced noise in natural habitats," Anthrozoos: A Multidisciplinary Journal of The Interactions of People & Animals 13, 225-231.
Strasser, E. H., and Heath, J. A. (2013). "Reproductive failure of a human-tolerant species, the American kestrel, is associated with stress and human disturbance," Journal of Applied Ecology 50, 912-919.
Swaddle, J. P., Kight, C. R., Perera, S., Davila-Reyes, E., and Sikora, S. (2012). "Constraints on acoustic signaling among birds breeding in secondary cavities: The effects of weather, cavity material, and noise on sound propagation," Ornithological Monographs 74, 63-77.
Tanaka, K., and Smith, C. A. (1978). "Structure of the chicken's inner ear: SEM and TEM study," American Journal of Anatomy 153, 251-271.
Trainer, J. E. (1946). "Auditory Acuity of Certain Birds," (Graduate School of Cornell University).
van der Ree, R., Jaeger, J. A., van der Grift, E. A., and Clevenger, A. P. (2011). "Effects of roads and traffic on wildlife populations and landscape function: road ecology is moving toward larger scales," Ecology and society 16, 48-48.
Van der Zande, A., Ter Keurs, W., and Van der Weijden, W. (1980). "The impact of roads on the densities of four bird species in an open field habitat—evidence of a long-distance effect," Biological conservation 18, 299-321.
Van Dijk, T. (1972). "A comparative study of hearing in owls of the family Strigidae," Netherlands Journal of Zoology 23, 131-167.
von Békésy, G. (1960). *Experiments in hearing* (McGraw-Hill New York).
Ware, H. E., McClure, C. J. W., Carlisle, J. D., and Barber, J. R. (2015). "A phantom road experiment reveals traffic noise is an invisible source of habitat degradation," Proceedings of the National Academy of Sciences.
Warren, P. S., Katti, M., Ermann, M., and Brazel, A. (2006). "Urban bioacoustics: it's not just noise," Animal behaviour 71, 491-502.
Wasserman, F. (1977). "Intraspecific acoustical interference in the white-throated sparrow (Zonotrichia albicollis)," Animal behaviour 25, 949-952.
Wiley, R. H., and Richards, D. G. (1982). "Adaptations for acoustic communication in birds: sound transmission and signal detection," Acoustic communication in birds 1, 131-181.
Wright, T. F., Cortopassi, K. A., Bradbury, J. W., and Dooling, R. J. (2003). "Hearing and vocalizations in the orange-fronted conure (Aratinga canicularis)," Journal of comparative psychology 117, 87.
Zollinger, S. A., Podos, J., Nemeth, E., Goller, F., and Brumm, H. (2012). "On the relationship between, and measurement of, amplitude and frequency in birdsong," Animal behaviour 84, e1-e9.

This page left intentionally blank

Appendix A: Glossary

Altricial: Species that are in an undeveloped state at hatching or birth and require care and feeding from parents.

Audiogram: A measure of hearing sensitivity, or threshold, at each frequency in the hearing range of an animal or human.

Auditory brainstem response (ABR): A physiological method to determine hearing bandwidth and sensitivity of animals without training. Electrodes (wires) are placed on the head of the animal just outside of the base of the brain (brainstem) to record electrical signals (emitted by the brain) in response to sounds that are detected by the ear. These signals are averaged and used to determine if the animal has detected the sound. It is possible to determine auditory thresholds for fishes using this method. The same method is used for numerous other species, including measurement of hearing capabilities of newborn human babies.

Auditory threshold: The lowest detectable sound, generally at a specific frequency. Most often, thresholds are the level at which a signal is detected some per cent of the time—often 50% or 70%. Absolute thresholds are the lowest level of signal that is detectable when there is no background (masking) noise.

Bandwidth: The range of frequencies over which a sound is produced or received.

Basilar papilla: The auditory region of the inner ear of birds. The basilar papilla referred to as the avian cochlea since it may be evolutionarily related to the mammalian hearing organ, the cochlea.

Broadband: Defined as noise that covers a wide range of frequencies relative to which the ear is sensitive. In contrast, narrowband noise covers only a limited number of (contiguous) frequencies. In relation to bird or human hearing, for instance, a broadband noise might contain sound energy from 100 to 10,000 Hz, whereas a narrowband noise may contain sound energy from 500 to 550 Hz.

Critical ratio: Defined as the ratio of the intensity of a pure tone to the intensity per hertz of a noise (i.e., the spectrum level) at a listener's threshold. For example, if a listener can just hear a 60 dB pure tone against a background of noise whose spectrum level is 40 dB, the listener's critical ratio is said to be 20 dB. In fact, the human critical ratio at 2 kHz is approximately 20 dB.

Conspecific: A member of the same species.

Decibel (dB): A customary scale most commonly used (in various ways) for reporting levels of sound. A difference of 10 dB corresponds to a factor of 10 in sound power. The actual sound measurement is compared to a fixed reference level and the decibel value is defined to be $10 \log_{10}$ (actual/reference), where (actual/reference) is a power ratio. Because sound power is usually proportional to sound pressure squared, the decibel value for sound pressure is $20\log_{10}$ (actual pressure/reference pressure). As noted above, the standard reference for underwater sound pressure is 1 micro Pascal (µPa). The dB symbol is followed by a second symbol identifying the specific reference value (i.e., re 1 µPa).

Effects: In this document, we have defined *effect* to mean any response by birds to traffic and construction noise. Our definition does not invoke or imply regulatory definitions of *effect*, as found in any law or regulation affecting birds.

Frequency spectrum: See *Spectrum*.

Hertz (Hz): The units of frequency where 1 hertz = 1 cycle per second.

Impulse sound: Transient sound produced by a rapid release of energy, usually electrical or chemical such as circuit breakers or explosives. Impulse sound has extremely short duration and extremely high peak sound pressure.

KiloHertz (kHz): A unit of frequency representing 1,000 Hz.

Noise: Generally an unwanted sound. Noise is often in the "ear of the beholder" in that a signal may be an important sound to one listener and unwanted "noise" to another.

Noise level: The noise power, usually relative to a reference level. Noise level is usually measured in decibels (dB) for relative power or picowatts for absolute power. Levels are represented in dB to denote specific aspects of the measurement and to also indicate the reference base or specific aspects of the measurement. Most frequently, sound levels for birds are referenced in terms of dB or weighted as dBA.

Octave: An octave is any band where the highest included frequency is exactly two times the lowest included frequency. For example, the frequency band that covers all frequencies between 707 Hz and 1,414 Hz is an octave band. The next octave band would be 1,414 to 2,828.

Ontogenetic: Development of an organism, usually from time of fertilization until it reaches its mature form.

Otolithic organs: The end organs in the vertebrate ear (saccule, utricle, lagena) associated with determination of head position relative to gravity. Along with the semicircular canals, these make up the vertebrate vestibular system.

Passeriformes: Song birds.

Permanent threshold shift (PTS): A permanent loss of hearing caused by some kind of acoustic or drug trauma. PTS results in irreversible damage to the sensory hair cells of the ear, and thus a permanent loss of hearing.

Power spectrum: "For a given signal, the power spectrum gives a plot of the portion of a signal's power (energy per unit time) falling within given frequency bins. The most common way of generating a power spectrum is by using a discrete Fourier transform, but other techniques such as the maximum entropy method can also be used."[21]

Semicircular canals: Three canals in the vertebrate ear that are mutually perpendicular to one another. They are involved in the detection of angular acceleration of the head, and provide the brain with information about movement of the head (and body). They are critically important to help maintain fixed gaze of the eyes on an object, even as the head moves. The semicircular canals and the otolithic organs make up the vestibular part of the ear.

Sensory hair cells: The cells in the basilar papilla and other end organs of the ear that are responsible for converting (transducing) mechanical energy of sound to signals that can stimulate the nerve from the ear to the brain (eighth cranial nerve).

Sound pressure level (SPL): The sound pressure level or SPL is an expression of the sound pressure using the decibel (dB) scale and the standard reference pressures 20 µPa for air and other gases.

Spectrum level: The intensity level of a sound within a 1 Hz band.

Spectrum (Spectra): A graphical display of the contribution of each frequency component contained in a sound.

Temporary threshold shift (TTS): Temporary loss of hearing as a result of exposure to sound over time. Exposure to high levels of sound over relatively short time periods will cause the same amount of TTS as exposure to lower levels of sound over longer time periods. The mechanisms underlying TTS are not well understood, but there may be some temporary damage to the sensory hair cells. The duration of TTS varies depending on the nature of the stimulus, but there is generally recovery of full hearing over time.

Threshold: The threshold generally represents the lowest signal level an animal will detect in some statistically predetermined percent of presentations of a signal. Most often, the threshold is the level at which an animal will indicate detection 50% of the time. Auditory thresholds are the lowest sound levels detected by an animal at the 50% level.

Weighting: An electronic filter which has a frequency response corresponding approximately to that of human hearing. Human hearing is most sensitive to sounds from about 500 Hz to 4000 Hz, and less sensitive at lower and higher frequencies. The overall level of a sound is usually expressed in terms of dBA and this is generally measured using a sound level meter with an "A-weighting" filter. The level of a sound in dBA is a good measure of the loudness of that sound. Different sources having the same dBA level generally sound about equally loud.

[21] From: http://mathworld.wolfram.com/PowerSpectrum.html

This page left intentionally blank

Appendix B: Complete Table of all Behavioral Studies of Hearing in Birds

Order	Common Name	Genus and Species	References
Anseriformes	mallard duck	*Anas platyrhynchos*	(Trainer, 1946)
Apodiformes	Australian grey swiftlet	*Collocalia Spodiopygia*	(Coles *et al*., 1987)
Caprimulgiformes	oilbird	*Steatornis caripensis*	(Konishi and Knudsen, 1979)
Casuariiformes	emu	*Dromaius novaehollandiae*	(Manley *et al*., 1997)
Charadriiformes	plains wanderer	*Pedionomus torquatus*	(Pettigrew *et al*., 1990)
Columbiformes	pigeon	*Columbia livia*	(Trainer, 1946; Heise, 1953; Hienz *et al*., 1977)
Falconiformes	American kestrel	*Falco sparverius*	(Trainer, 1946)
Falconiformes	European sparrowhawk	*Accipiter nisus*	(Trainer, 1946; Klump *et al*., 1986)
Galliformes	bobwhite quail	*Colinus virgianus*	(Barton *et al*., 1984)
Galliformes	chicken	*Gallus*	(Gray and Rubel, 1985; Saunders and Salvi, 1993)
Galliformes	Japanese quail	*Coturnix japonica*	(Niemiec *et al*., 1994)
Galliformes	turkey	*Meleagris gallopavo*	(Maiorana and Schleidt, 1972)
Passeriformes	American robin	*Turdus migratorius*	(Konishi, 1970)
	blue jay	*Cyanocitta cristata*	(Cohen *et al*., 1978)
	brown-headed cowbird	*Molothrus ater*	(Hienz *et al*., 1977)
	bullfinch	*Pyrrhula*	(Schwartzkopff, 1949)
	chipping sparrow	*Spizella passerina*	(Konishi, 1970)
	common canary	*Serinus canarius*	(Okanoya and Dooling, 1987)
	common crow	*Corvus brachyrhynchos*	(Trainer, 1946)
	European starling	*Sturnus vulgaris*	(Trainer, 1946; Konishi, 1970; Kuhn *et al*., 1982; Dooling *et al*., 1986)
	field sparrow	*Spizella pusilla*	(Dooling *et al*., 1979)
	fire finch	*Lagonosticta senegala*	(Dooling *et al*., 2000b)
	great tit	*Parus major*	(Klump *et al*., 1986; Langemann *et al*., 1998)
	house finch	*Carpodacus mexicanus*	(Dooling *et al*., 1978)
	house sparrow	*Passer domesticus*	(Konishi, 1970; Aleksandrov and Dmitrieva, 1992)
	pied flycatcher	*Ficedula hypolueca*	(Aleksandrov and Dmitrieva, 1992)
	red-winged blackbird	*Agelaius phoeniceus*	(Hienz *et al*., 1977)
	slate-colored junco	*Junco hyemalis*	(Konishi, 1970)
	song sparrow	*Melospiza melodia*	(Okanoya and Dooling, 1987; 1988)
	swamp sparrow	*Melospiza georgiana*	(Okanoya and Dooling, 1987; 1988)
	western meadowlark	*Sturnella neglecta*	(Konishi, 1970)
	zebra finch	*Taeniopygia guttata*	(Okanoya and Dooling, 1987; Hashino and Okanoya, 1989)
Psittaciformes	Bourke's parrot	*Neophema bourkii*	Dooling et al. Unpublished Data
	budgerigar	*Melopsittacus undulatus*	(Dooling and Saunders, 1974; 1975; Saunders *et al*., 1979; Saunders and Pallone, 1980; Okanoya and Dooling, 1987; Hashino *et al*., 1988)
	cockatiel	*Nymphicus hollandicus*	(Okanoya and Dooling, 1987)
	orange-fronted conureq	Aratinga canicularis	(Wright *et al*., 2003)
Strigiformes	African wood owl	*Strix woodfordii*	(Nieboer and Van der Paardt, 1976)
	barn owl	*Tyto alba*	(Konishi, 1970; 1973; Dyson *et al*., 1998)
	brown fish owl	*Ketupa zeylonensis*	(Van Dijk, 1972)
	eagle owl	*Bubo*	

Order	Common Name	Genus and Species	References
	forest eagle owl	*Bubo nipalensis*	
	great horned owl	*Bubo virginianus*	(Trainer, 1946)
	long eared owl	*Asio otus*	(Van Dijk, 1972)
	mottled owl	*Strix virgata*	
	scops owl	*Otus scops*	
	snowy owl	*Nyctea scandiaca*	
	spotted wood owl	*Strix seloputo*	
	tawny owl	*Strix aluco*	
	white-faced scops owl	*Otus leucotis*	

Appendix C: Complete Table of all Behavioral Studies of Critical Ratios in Birds

Order	Common Name	Genus and Species	References
Columbiformes	pigeon	*Columbia livia*	(Hienz and Sachs, 1987)
Passeriformes	brown-headed cowbird	*Molothrus ater*	(Hienz and Sachs, 1987)
	common canary	*Serinus canarius*	(Okanoya and Dooling, 1987)
	European starling	*Sturnus vulgaris*	(Okanoya and Dooling, 1987)
	fire finch	*Laganosticta senegala*	(Lohr *et al* . 2004)
	great tit	*Parus major*	(Langemann *et al.*, 1998)
	red-winged blackbird	*Agelaius phoeniceus*	(Hienz and Sachs, 1987)
	song sparrow	*Melospiza melodia*	(Okanoya and Dooling, 1987)
	swamp sparrow	*Melospiza georgiana*	
	zebra finch	*Taeniopygia guttata*	
Psittaciformes	budgerigar	*Melopsittacus undulatus*	(Dooling and Saunders, 1975; Dooling *et al.*, 1979; Saunders *et al.*, 1979; Okanoya and Dooling, 1987; Hashino *et al.*, 1988; Hashino and Okanoya, 1989)
	cockatiel	*Nymphicus hollandicus*	(Okanoya and Dooling, 1987)
	orange-fronted conure	*Aratinga canicularis*	(Wright *et al*, 2003)
Strigiformes	barn owl	*Tyto alba*	(Konishi, 1973; Dyson *et al*, 1998)

This page left intentionally blank

Appendix D: Fundamentals of Highway Traffic Noise

(Provided by Caltrans)

Fundamentals of Traffic Noise

The following is a brief discussion of fundamental traffic-noise concepts. For a detailed discussion, please refer to the *Technical Noise Supplement* (Caltrans 2013) available on the Caltrans Web site (http://www.dot.ca.gov/hq/env/noise).[22]

Sound, Noise, and Acoustics

Sound is a disturbance that is created by a moving or vibrating source in a gaseous or liquid medium or the elastic stage of a solid and that is capable of being detected by the hearing organs. Sound can be described as the mechanical energy of a vibrating object transmitted by pressure waves through a medium to a hearing organ, such as a human ear. For traffic sound, the medium of concern is air. *Noise* is defined as loud, unpleasant, unexpected, or undesired sound.

Sound is actually a process that consists of three components: the sound source, the sound path, and the sound receiver. All three components must be present for sound to exist. Without a source to produce sound or a medium to transmit sound-pressure waves, there is no sonnd. Sound must also be received; a hearing organ, sensor, or object must be present to perceive, register, or be affected by sound or noise. In most situations, there are many different sound sources, paths, and receivers, not only one of each. *Acoustics* is the field of science that deals with the production, propagation, reception, effects, and control of sound.

Frequency and Hertz

A continuous sound can be described by its *frequency* (pitch) and its *amplitude* (loudness). Frequency relates to the number of pressure oscillations per second. Low-frequency sounds are low in pitch, like the low notes on a piano, whereas high-frequency sounds are high in pitch, like the high notes on a piano. Frequency is expressed in terms of oscillations, or cycles, per second. Cycles per second are commonly referred to as Hertz (Hz) (e.g., a frequency of 250 cycles per second is referred to as 250 Hz). High frequencies are sometimes more conveniently expressed in kilo-Hertz (kHz), or thousands of Hertz. The extreme range of frequencies that can be heard by the healthiest human ears spans from 16–20 Hz on the low end to about 20,000 Hz (20 kHz) on the high end.

Sound-Pressure Levels and Decibels

The *amplitude* of a sound determines its loudness. Loudness of sound increases and decreases with increasing and decreasing amplitude. Sound-pressure amplitude is measured in units of micro-Newtons per square meter (N/m^2) , also called micro-Pascals (µPa). One µPa is approximately one-hundred billionth (0.00000000001) of normal atmospheric pressure. The pressure of a very loud sound may be 200 million ΦPa, or 10 million times the pressure of the weakest audible sound (20 µPa). Because expressing sound levels in terms of ΦPa would be cumbersome, *sound-pressure level* (SPL) is used to describe in logarithmic units the ratio of actual sound pressures to a reference pressure squared. These units are called bels, named after Alexander Graham Bell. To provide finer resolution, a bel is divided into 10 decibels (dB).

[22] http://www.dot.ca.gov/hq/env/noise/pub/TeNS_Sept_2013B.pdf

Addition of Decibels

Because decibels are logarithmic units, SPL cannot be added or subtracted by ordinary arithmetic means. For example, if 1 automobile produces an SPL of 70 dB when it passes an observer, 2 cars passing simultaneously would not produce 140 dB; rather, they would combine to produce 73 dB. When two sounds of equal SPL are combined, they produce a combined SPL 3 dB greater than the original individual SPL. In other words, sound energy must be doubled to produce a 3-dB increase. If two sound levels differ by 10 dB or more, the combined SPL is equal to the higher SPL; the lower sound level would not increase the higher sound level.

A-Weighted Decibels

SPL alone is not a reliable indicator of loudness. The frequency of a sound also has a substantial effect on how humans respond. Although the intensity (energy per unit area) of the sound is a purely physical quantity, the loudness or human response is determined by the characteristics of the human ear.

Human hearing is limited in the range of audible frequencies as well as in the way it perceives the SPL in that range. In general, the healthy human ear is most sensitive to sounds from 1,000–5,000 Hz and perceives a sound within that range as being more intense than a sound of higher or lower frequency with the same magnitude. To approximate the frequency response of the human ear, a series of SPL adjustments is usually applied to the sound measured by a sound level meter. The adjustments, referred to as a *weighting network*, are frequency-dependent.

The A-scale weighting network approximates the frequency response of the average young ear when listening to most ordinary sounds. When people make judgments of the relative loudness or annoyance of a sound, their judgments correlate well with the A-scale sound levels of those sounds. Other weighting networks have been devised to address high noise levels or other special problems (e.g., B-, C-, and D-scales), but these scales are rarely used in conjunction with highway-traffic noise. Noise levels for traffic-noise reports are typically reported in terms of A-weighted decibels (dBA). In environmental noise studies, A-weighted SPLs are commonly referred to as noise levels. Table D1 shows typical A-weighted noise levels.

Human Response to Changes in Noise Levels

Under controlled conditions in an acoustics laboratory, the trained, healthy human ear is able to discern 1-dB changes in sound levels when exposed to steady, single-frequency ("pure-tone") signals in the mid-frequency range. Outside such controlled conditions, the trained ear can detect 2-dB changes in normal environmental noise. However, it is widely accepted that the average healthy ear can barely perceive 3-dB noise level changes A 5-dB change is readily perceptible, and a 10-dB change is perceived as being twice or half as loud. As discussed above, doubling sound energy results in a 3-dB increase in sound; therefore, doubling sound energy (e.g., doubling the volume of traffic on a highway) would result in a barely perceptible change in sound level.

Table D1. Typical Noise Levels

Common Outdoor Activities	Noise Level (dBA)	Common Indoor Activities
	— 110 —	Rock band concert
Jet flyover at 300 meters (1,000 feet)		
	— 100 —	
Gas lawn mower at 1 meter (3 feet)		
	— 90 —	
Diesel truck at 15 meters (50 feet) at 80 kilometers per hour (50 miles per hour)		Food blender at 1 meter (3 feet)
	— 80 —	Garbage disposal at 1 meter (3 feet)
Noisy urban area, daytime		
Gas lawn mower, 30 meters (100 feet)	— 70 —	Vacuum cleaner at 3 meters (10 feet)
Commercial area		Normal speech at 1 meter (3 feet)
Heavy traffic at 90 meters (300 feet)	— 60 —	
		Large business office
Quiet urban daytime	— 50 —	Dishwasher next room
Quiet urban nighttime	— 40 —	Theater, large conference room (background)
Quiet suburban nighttime		
	— 30 —	Library
Quiet rural nighttime		Bedroom at night
	— 20 —	
		Broadcast/recording studio
	— 10 —	
Lowest threshold of human hearing	— 0 —	Lowest threshold of human hearing

Source: Caltrans 2013.

Noise Descriptors

Noise in our daily environment fluctuates over time. Some fluctuations are minor, but some are substantial. Some noise levels occur in regular patterns, but others are random. Some noise levels fluctuate rapidly, but others slowly. Some noise levels vary widely, but others are relatively constant. Various noise descriptors have been developed to describe time-varying noise levels. The following are the noise descriptors most commonly used in traffic-noise analysis.

- *Equivalent Sound Level (L_{eq})*: L_{eq} represents an average of the sound energy occurring over a specified period. In effect, L_{eq} is the steady-state sound level that in a stated period would contain the same acoustical energy as the time-varying sound that actually occurs during the same period. The 1-hour A-weighted equivalent sound level (L_{eq}[h]), is the energy average of the A-weighted sound levels occurring during a 1-hour period and is the basis for noise-abatement criteria (NAC) used by Caltrans and the FHWA.

- *Percentile-Exceeded Sound Level (L_x)*: L_x represents the sound level exceeded for a given percentage of a specified period (e.g., L_{10} is the sound level exceeded 10% of the time, L_{90} is the sound level exceeded 90% of the time).

- *Maximum Sound Level (L_{max})*: L_{max} is the highest instantaneous sound level measured during a specified period.

- *Day-Night Level (L_{dn})*: L_{dn} is the energy average of the A-weighted sound levels occurring during a 24-hour period with 10 dB added to the A-weighted sound levels occurring between 10 p.m. and 7 a.m.

- *Community Noise Equivalent Level (CNEL)*: CNEL is the energy average of the A-weighted sound levels occurring during a 24-hour period with 10 dB added to the A-weighted sound levels occurring between 10 p.m. and 7 a.m. and 5 dB added to the A-weighted sound levels occurring between 7 p.m. and 10 p.m.

Sound Propagation

When sound propagates over a distance, it changes in level and frequency content. The manner in which noise reduces with distance depends on the following factors.

- *Geometric spreading:* Sound from a small, localized source (i.e., a point source) radiates uniformly outward as it travels away from the source in a spherical pattern. The sound level attenuates (or drops off) at a rate of 6 dBA for each doubling of distance. Traffic and construction noise is not a single, stationary point source of sound. The movement of the vehicles on a highway makes the source of the sound appear to emanate from a line (i.e., a line source) rather than a point. This line source results in cylindrical spreading rather than the spherical spreading that results from a point source. The change in sound level from a line source is 3 dBA per doubling of distance.

- *Ground absorption*: The noise path between the highway and the observer is usually very close to the ground. Noise attenuation from ground absorption and reflective-wave canceling adds to the attenuation associated with geometric spreading. Traditionally, the excess attenuation has also been expressed in terms of attenuation per doubling of distance. This approximation is done for simplification only because prediction results based on this scheme are sufficiently accurate for distances of less than 60 meters (200 feet). For acoustically hard sites (i.e., those sites with a reflective surface, such as a parking lot or a smooth body of water, between the source and the receiver), no excess ground attenuation is assumed. For acoustically absorptive or soft sites (i.e., those sites with an absorptive ground surface, such as soft dirt, grass, or scattered bushes and trees, between the source and the receiver), an excess ground-attenuation value of 1.5 dBA per doubling of distance is normally assumed. When added to the geometric spreading, the excess ground attenuation results in an overall drop-off rate of 4.5 dBA per doubling of distance for a line source and 7.5 dBA per doubling of distance for a point source.

- *Atmospheric effects*: Research by Caltrans and others has shown that atmospheric conditions can have a significant effect on noise levels within 60 meters (200 feet) of a highway. Wind has been shown to be the most important meteorological factor within approximately 150 meters (500 feet) of the source, whereas vertical air-temperature gradients are more important for greater distances. Other factors such as air temperature, humidity, and turbulence also have significant effects. Receptors located downwind from a source can be exposed to increased noise levels relative to calm conditions, whereas locations upwind can have lower noise levels. Increased sound levels can also occur as a result of temperature inversion conditions (i.e., increasing temperature with elevation).

Shielding by natural or human-made features: A large object or barrier in the path between a noise source and a receiver can substantially attenuate noise levels at the receiver. The amount of attenuation provided by this shielding depends on the size of the object and the frequency content of the noise source. Natural terrain features (e.g., hills and dense woods) and human-made features (e.g., buildings and walls) can substantially reduce noise levels. Walls are often constructed between a source and a receiver specifically to reduce noise. A barrier that breaks the line of sight between a source and a receiver will typically result in at least 5 dB of noise reduction. A taller barrier may provide as much as 20 dB of noise reduction.

D. Federal and State Regulations, Standards, and Policies

Federal and state regulations, standards, and policies relating to traffic noise are discussed in detail in the Protocol. A transportation project affected by the Protocol is referred to as type 1 project, which is defined in 23 CFR 772 as a proposed federal or federal-aid highway project for construction of a highway on a new location or the physical alteration of an existing highway that significantly changes the horizontal or vertical alignment or increases the number of through traffic lanes. The FHWA has clarified its interpretation of type 1 projects by stating that a type 1 project is any project that has the potential to increase noise levels at adjacent receivers. This includes projects to add interchange, ramp, auxiliary, or truck-climbing lanes to an existing highway. A project to widen an existing ramp by a full lane width is also considered to be a type 1 project. Caltrans extends this definition to include state-funded highway projects. The project alternatives evaluated in this report are considered to be a Type 1 project because they involve federal funding and adding lanes to the existing mainline highway.

Applicable federal and state regulations, standards, and policies are discussed below.

National Environmental Policy Act

NEPA is a federal law that establishes environmental policy for the nation, provides an interdisciplinary framework for federal agencies to prevent environmental damage, and contains action-forcing procedures to ensure that federal agency decision-makers take environmental factors into account. Under NEPA, impacts and measures to mitigate adverse impacts must be identified, including impacts for which no mitigation or only partial mitigation is available. The FHWA regulations discussed below constitute the federal noise standard. Projects complying with this standard are also in compliance with the requirements stemming from NEPA.

Federal Highway Administration Regulations

23 CFR 772 provides procedures for conducting highway-project noise studies and implementing noise-abatement measures to help protect the public health and welfare, supply NAC, and establish requirements for information to be given to local officials for use in planning and designing highways. Under this regulation, noise abatement must be considered for a type 1 project if the project is predicted to result in a traffic-noise impact. A traffic-noise impact is considered to occur when the project results in a *substantial noise increase* or when the predicted noise levels *approach or exceed* NAC specified in the regulation. 23 CFR 772 does not specifically define what constitutes a substantial increase or the term approach; rather, it leaves interpretation of these terms to the states.

Noise-abatement measures that are *reasonable* and *feasible* and likely to be incorporated into the project, as well as noise impacts for which no apparent solution is available, must be identified before adoption of the final environmental document for the project. Table D2 summarizes the FHWA's NAC.

Table D2. Activity Categories and Noise Abatement Criteria

Activity Category	Activity $L_{eq}[h]$[1]	Evaluation Location	Description of Activities
A	57	Exterior	Lands on which serenity and quiet are of extraordinary significance and serve an important public need and where preservation of those qualities is essential if the area is to continue to serve its intended purpose.
B[2]	67	Exterior	Residential.
C[2]	67	Exterior	Active sport areas, amphitheaters, auditoriums, campgrou cemeteries, day care centers, hospitals, libraries, medical facilities, parks, picnic areas, places of worship, playgrour public meeting rooms, public or nonprofit institutional structures, radio studios, recording studios, recreation area Section 4(f) sites, schools, television studios, trails, and tra crossings.
D	52	Interior	Auditoriums, day care centers, hospitals, libraries, medica facilities, places of worship, public meeting rooms, public nonprofit institutional structures, radio studios, recording studios, schools, and television studios.
E	72	Exterior	Hotels, motels, offices, restaurants/bars, and other develop lands, properties, or activities not included in A- D or F.
F			Agriculture, airports, bus yards, emergency services, industrial, logging, maintenance facilities, manufacturing, mining, rail yards, retail facilities, shipyards, utilities (wat resources, water treatment, electrical), and warehousing.
G			Undeveloped lands that are not permitted.

[1] The $L_{eq}(h)$ activity criteria values are for impact determination only and are not design standards for noise abatement measures. All values are A-weighted decibels (dBA).

[2] Includes undeveloped lands permitted for this activity category.

Primary consideration is given to exterior areas. In situations where no exterior activities are affected by traffic noise the interior criterion (activity category E) is used as the basis for noise abatement consideration.

California Environmental Quality Act

CEQA is the foundation of environmental law and policy in California. The main objectives of CEQA are to disclose to decision-makers and the public the significant environmental effects of proposed activities and to identify ways to avoid or reduce those effects by requiring implemeotation nf feasible alternatives or mitigation measures. Under CEQA, a substantial noise increase may result in a significant adverse environmental effect; if so, the noise increase must be mitigated or identified as a noise impact for which it is likely that only partial (or no) mitigation measures are available. Specific economic, social, environmental, legal, and technological conditions can make mitigation measures for noise infeasible.

Traffic-Noise Analysis Protocol for New Highway Construction and Reconstruction Projects

The Protocol specifies the policies, procedures, and practices to be used by agencies that sponsor new construction or reconstruction projects. NAC specified in the Protocol are the same as those specified in 23 CFR 772. This report defines a noise increase as substantial when the predicted noise levels with project implementation exceed existing noise levels by 12 dBA -L_{eq} (h). The Protocol also states that a sound level is considered to approach an NAC level when the sound level is within 1 dB of the NAC identified in 23 CFR 772. For example, a sound level of 66 dBA is considered to approach the NAC of 67 dBA, but 65 dBA is not.

This page left intentionally blank

Appendix E: Review of Pre-2007 Literature on Effects of Traffic Noise on Birds

From (Dooling and Popper, 2007)

The literature on the actual effects of traffic noise on birds is limited and the methodology is often insufficient to provide a clear correlation between traffic noise and any effects on bird physiology and/or behavior. One particular concern is that whereas there is indirect evidence that traffic noise may affect birds (e.g., Reijnen and Foppen, 1994; 1995; Reijnen *et al.*, 1995; Forman *et al.*, 2002), there are also correlated variables that could have impact such as visual stimuli, air pollution produced by autos and trucks (e.g., Llacuna *et al.*, 1996; Clench-Aas *et al.*, 2000), and changes in the physical environment around the roadways (e.g., Ferris, 1979). Differentiating among these and other variables is often difficult or impossible. While there is statistical evidence (debated by some, see below) to suggest that noise may affect birds in some way (e.g., Reijnen and Foppen, 1994; 1995; Reijnen *et al.*, 1995), there have yet to be definitive experiments that clearly isolate noise as an exclusive source of disturbance. Even when noise is implicated as a contributing factor, there are still are many variables which are poorly understood, such as noise levels at the birds (received levels), effects of frequency of disturbances (e.g., how many cars/trucks come by a bird in some time interval – (Forman *et al.*, 2002), and species. Complicating this picture even further are substantial species differences in the way that birds respond to noise and how readily they may acclimate or habituate to various disturbances (e.g., Ferris, 1979; Kuitunen *et al.*, 1998; Fernández-juricic, 2001; Slabbekoorn and Ripmeester, 2008; Slabbekoorn *et al.*, 2012).

The overall literature has been critically reviewed several times in recent years (e.g., Sarigul-Klijn *et al.*, 1997; Kaseloo, 2005; Warren *et al.*, 2006; van der Ree *et al.*, 2011; Ortega, 2012). These reviews suggest that a good portion of the literature is not relevant to the issues at hand since the literature often does not take into consideration all appropriate variables (e.g., variables other than sound) or that the publications have problems with data analysis and/or interpretation.

In one analyses, Warren *et al.* (2006) evaluated data suggesting that noise could affect bird behavior. However, the authors pointed out that while the data could be interpreted as indicating that noise may affect birds, none of the earlier work can clearly be used to reach any firm conclusions about any one species, or all species. Indeed, Warren *et al.* (2006) point out the need for very specific and highly controlled laboratory and field studies to assess how highway (or any other) noise will affect birds. Such experiments are very difficult (and expensive) to design and execute, and all other variables must be taken into consideration in design of these experiments.

The four major sets of studies considered by Warren *et al.* (2006) are helpful to understanding the issues. In one series of papers, Reijnen and colleagues (Foppen and Reijnen, 1994; Reijnen and Foppen, 1994; 1995; Reijnen *et al.*, 1995) reviewed in (Reijnen *et al.*, 1998) examined the effects of motorway traffic on breeding bird populations in the Netherlands. The investigators concluded that traffic noise has an impact on birds within several hundred meters of the road and that roadway noise lowers the extent of bird breeding near highways. The study by Reijnen and colleagues showed that when traffic noise level was constant, there was no discernable effect from visual disturbance. But when visual disturbance was kept constant, bird distribution patterns were statistically correlated with traffic noise. Furthermore the authors noted that visual disturbance and vehicular pollutants extended outward only a short distance from the roadway, whereas both traffic noise and reduced bird densities extended outward much further. This differential effect distance approach suggests that if it is appropriately integrated into the experimental designs of future studies, it could provide more tractable means for isolating the effects of the confounding variables and better extracting focused information on noise-specific impacts.

While the data from Reijnen et al. are interesting and possibly instructive, the work has been severely criticized for poor statistical analysis and poor controls, and for lack of analysis of individual bird species (Sarigul-Klijn *et al.*, 1997) which concluded that the number of birds studied was too low for reliable statistical measures and that levels of significance used varied between study years. Sarigul-Klijn *et al.* (1997) also concluded that Reijnen et al., in reaching their conclusions, also did not consider construction as another potential point of impact on birds.

Most importantly, the Transportation Noise Control Center study (Sarigul-Klijn *et al.*, 1997) points out that Reijnen and colleagues pooled all of their data so that they presented a possible effect on all species, rather than determine whether there are species-specific effects. The importance of the species variability in response to noise (and other factors) has been emphasized in several other studies which have shown variability in whether different species respond to noise or not (e.g., Clark and Karr, 1979; Ferris, 1979; Van der Zande *et al.*, 1980; Kuitunen *et al.*, 1998; Fernández-juricic, 2001; Peris and Pescador, 2004). Indeed, lack of consideration of species variability in life style is also the basis for the poor generality of the FWS (2006) recommended procedures for analysis of the effects of sounds on spotted owls and marbled murrelets.

In another study, Stone (2000) did transects to determine bird populations over a wide range of land use types. The results led to the suggestion that there is a marked decrease in bird populations in noisier areas, despite the specific land use. However, Warren *et al.*, (2006) criticized the Stone (2000) study and pointed out that while noise was one variable that could have affected bird populations in some types of land use and not in others, Stone (Stone, 2000) did not do a multi-factor analysis to determine if other habitat issues, such as whether there were also differences ground surface, vegetative type, or other variables that could have altered a bird's behavior.

A more convincing case that traffic noise may affect birds is a study by Forman et al. (Forman *et al.*, 2002) which looked at the presence of five species of grassland bird populations at different distances from roadways in and around Boston. The authors argue that there is an effect on density of species studied by roadway noise, but that the extent of the effect, in terms of decreased populations at different distances, varied depending upon the level of traffic on the road. They found that when traffic was less than 8,000 vehicles/day there was no effect on grassland bird populations. In areas with from 8,000-15,000 vehicles per day, there was no effect on population levels per se, but there were fewer breeding birds up to 400 m from the road. Bird presence and breeding was decreased at up to 700 m from the roadway when there were from 15,000-30,000 vehicles per day, whereas this distance increased to 1,200 m for more than 30,000 vehicles per day (a multilane highway). While the authors conclude that noise may be the major factor affecting these grassland species, but that other environmental variables such as visual signals, air pollutants, and lack of prey near the roadways may help explain the decline in bird populations. Clearly, direct experimental evidence of effects of increased chronic noise of different levels and sound spectra (Lee and Fleming, 1996) is needed to confirm this hypothesis (also see Warren *et al.*, 2006).

Still, it is important to recognize that the results from Forman *et al.* (2002) may not be applicable to all species, or in all situations. For example, Peris and Pescador (2004) examined the effects of low, medium, and high traffic volumes on bird populations of 20 passerine species in pasture-woodland environments near several roads in western central Spain. While it is hard to specifically compare results between the two studies since Peris and Pescador (2004) did not define road density in terms of actual number of vehicles/day, the different results are instructive. In contrast to Forman et al. (Forman *et al.*, 2002), Peris and Pescador (2004) provided sound level measures at distances of 50-100 m from the roadways. They reported that the high traffic volume area had sound levels of 69±5 dB, medium density 46±3 dB, and low density at 36±2 dB (it was not indicated if this was dB SPL or dBA). Peris and Pescador (2004) showed that there were differences between the number of birds and the extent of breeding populations in each of the three areas, but the differences varied by species. In effect, no one pattern of bird presence was appropriate for all of the species studied over the two year period.

For example, corn bunting (*Miliaria calandra*), rock sparrow (*Petronia petronia*), and house sparrow (*Passer domesticus*) had a higher breeding density in the high traffic (noisier) environment than they did in the low traffic volume areas. In contrast, breeding density was higher for wheatear (*Oenanthe* sp.) in low and moderate traffic areas (quieter) than in high traffic areas. The authors concluded that 55% of the species did not show any difference in breeding density between the three noise level sites, whereas other birds did show statistically significant differences. The authors suggest that the differences in responses of the various species may depend on hearing sensitivity of the species, with birds that have more sensitive hearing showing greater avoidance of road noise than birds with poorer hearing.

This page left intentionally blank

Appendix F: Recommendations for Research to Refine Future Guidance

The three classes of potential effects of traffic noise on birds: (1) behavioral and/or physiological effects; (2) damage to hearing from acoustic over-exposure; and (3) masking of communication signals. All of these can cause dynamic behavioral, and population effects. These three classes of potential effects lead to separate, *but overlapping*, recommendations for future work (see Table F1 and Table F2). Some of this work is at high priority while other work is of lower priority depending on the criteria for making decisions. High priority could be to go for those issues that can be tackled by efficiency of data collection and the precision of the results (e.g., noise exposure studies in the laboratory), or, at by taking on the problem that extends the furthest from the roadway (e.g., field studies of stress and disturbance effects at distances far beyond those at which hearing damage and masking from traffic noise might occur). Or highest priority could be assigned to some combination of studies which give the greatest potential value for moving us forward to better and more useful *interim* guidelines. Experiments that can quickly improve the *interim* guidelines are given a higher priority than longer-term (and often more difficult) experiments that may not refine the *interim* guidelines efficiently. It should be noted that while not always stated explicitly, all studies should be done on several species.

7) Stress and physiological effects:[23]
   a) Obtain a definitive answer to the question of whether traffic noise alone can cause stress, physiological reactions, and disturbances in social behavior in birds by using artificial traffic noises broadcast in large areas while birds (preferably captive) are monitored for stress indices (low priority).
   b) Conduct studies comparatively to determine if stress effects are species specific (low priority).
   c) Conduct studies on birds of different ages and with different degrees of experience with loud noises to determine if experience is a factor in stress-related impacts (low priority).

8) Acoustic over-exposure effects:
   a) Conduct lab experiments to definitively rule out the possibility that continuous loud traffic noise can damage avian hearing (low priority).
   b) Examine effects of different levels of continuous noise on temporary and permanent hearing loss in different bird species (high priority).
   c) Examine effects of impulsive noise such as that produced by construction equipment and pile driving on hearing loss in different bird species. Consider a range of variables including: the intensity of the noise, the number of impulses, inter-pulse interval, and effects of different "rest periods" between pulses on hearing loss. Also include combinations of continuous traffic noise and impulse noises since some mammalian data suggest a synergistic effect (high priority).

9) Masking effects:
   a) Extend what is known about masking effectiveness of traffic noise on the vocalizations of birds by conducting behavioral tests with a wider range of individual and species

[23] It should be noted that precise definition of the questions and issues of the effects of traffic noise on birds should be developed with the guidance of individuals who are expert on avian endocrinology and the literature on this topic.

vocalizations, different types and levels of traffic noise, traffic noises filtered through various habitats, and recorded at various distances from the roadway (high priority)

b) Assemble current data or generate new data on vocalizations of endangered species including types, levels, preferred singing location preferences, habitat characteristics, territory size, effects of habitat characteristics on vocalization and noise transmission. This will allow precise modeling of the masking effects of traffic noise acoustic communication (high priority).

c) Obtain ABR measures of hearing (audiogram) and masking (critical ratios) in endangered species to determine how well they conform to the emerging model of masking of vocalizations by noise which, to date, is based primarily on laboratory species of birds (high priority).

d) Develop a generalized quantitative model for estimating communication distance based on masking data, habitat characteristics, territory size, the bird's singing position preferences, and different traffic noise profiles (high priority).

10) Dynamic behavioral effects[24]

a) Evaluate population dynamic shifts (i.e., population range, predator prey relationships, etc.) based on increases in ambient traffic noise and construction related activities.

b) Evaluate any secondary effects of implementing adaptations in order to avoid masking. How does this interact with other life-cycle activities such as mate attraction, prey identification, territory size, etc.

c) Understand behavioral indicators of harassment or stress such as flushing from a nest, territorial behaviors, etc. associated with noise.

The recommendations are summarized in Tables F1 and F2. Table F1 presents the data in terms of examining the effects in terms of specific sound types.

Table F1: Research recommendations based on interim guidelines

Noise Source Type	Hearing Damage	Masking	Behavioral/ Physiological
Single Impulse (e.g., Blast)	Expose multiple species to impulsive noises (at different levels/distances) and measure hearing loss & recovery.	Not applicable	Examine animals post exposure for signs of stress (e.g., droppings, etc.)
Multiple Impulse (e.g., jackhammer, pile driver)	Expose multiple species to multiple strikes (at different levels/distances/intervals) and measure hearing loss and recovery.	In multiple species, examine masking by low level noises from multiple strikes to compare with results from continuous noise masking(Lab study)	Examine animals post exposure for signs of stress (e.g., droppings, etc.)
Non-Strike Continuous (e.g., construction noise)	Not applicable	In multiple species, examine masking by low level noises from multiple strikes to compare with results from continuous noise masking(Lab study)	Examine animals post exposure for signs of stress (e.g., droppings, etc.)
Traffic and Construction Noise	Not applicable	In multiple species, examine masking by low level traffic and construction noises to compare with results from continuous noise masking(Lab study)	Examine animals post exposure for signs of stress (e.g., droppings, etc.)
Alarms (97 dB/100 ft)	NA	NA	Future research

[24] Get input from experts in behavioral ecology on the types of population effects that might be expected.

Table F2: Additions to basic science data to inform decisions on interim guidelines and future analyses	
Topic	**Method**
Audiograms in Birds	Measure hearing thresholds in a variety of species using the ABR(lab & field)
Masked Thresholds in Birds	Measure masked thresholds and critical ratios in a variety of (endangered) species using the ABR(lab & field)
Vocalization & Communication Distance	Review literature for description of vocalizations, territory size, and communication range, young learning songs, female choice in breeding
Acoustic Communication Model	Develop a model that combines habitat characteristics (e.g., sound transmission), vocalization characteristics (e.g., spectrum, intensity, etc.) and masked thresholds to refine estimates of the effects of masking by noise on communication.
Attenuation/Avoidance/Minimization/Mitigation Methods	Evaluate ways which may inform decisions regarding equipment use, attenuation methods, avoidance, minimization/mitigation methods.

This page left intentionally blank

Appendix G: A History of the 60 dBA Criterion

In 1987, a biologist, John Rieger, developed a criterion for a California highway project by measuring noise levels at the nests of birds along a highway. On average, these levels approximated 60 dBA (Barrett, 1996). According to Barrett, Rieger assumed that if birds were successfully breeding, then this noise level is, by definition, not detrimental to the birds. Unaware of this work, and completely independently, Dooling also provided the California Fish and Wildlife Service with a noise level of 60 dBA for traffic noise that would begin to raise concerns about potential masking of communication sounds between birds by traffic noise. Barret's number came from actual observations of birds nesting in noisy areas near a highway. Dooling's number came from an auditory model that calculated whether noise levels from traffic rose above ambient noise levels enough to affect acoustic communication between two birds. In neither case was this number intended to set a precedent or become a standard for noise-impact mitigation. The level of 60 dBA for traffic noise only applies, at best, under a narrow range of specific conditions having to with the sound-affecting aspects of the habitat, the species life style and dependence on acoustic communication, the level of ambient noise without any traffic noise, as well as whether the species' predators use acoustic signals to locate their prey. The use of one number like 60 dBA provides only a crude and probably conservative estimate. A precise answer would require the information just discussed as well as information about the level and spectrum of the ambient noise, of the traffic noise, and of the bird's vocalizations.

Nevertheless, it appears that the 60 dBA criterion has been inappropriately used in many reports over the past 25 years as a hard and fast rule regarding the effects of highway and other anthropogenic noise on birds. The evidence today clearly shows that the application of this criterion to construction noise is likely to be far too conservative and unnecessarily restrictive. There are several reasons for this conclusion: (1) birds do not hear as well as humans at low frequencies which contain the bulk of energy in traffic noise; (2) bird vocalizations are at higher frequencies than traffic noise; (3) the use of the A scale on the sound level meter which mirrors human hearing, as opposed to bird hearing, overestimates the effects of traffic noise on bird hearing because traffic and construction noises are predominantly low frequency; and (4) birds, like humans, can and do employ a number of short term behavioral strategies for hearing in noise such as turning their heads, changing height or location, raising their voice, and timing their communication to coincide with periods of low noise.

This page left intentionally blank

This page left intentionally blank

Draft
Environmental Impact Report
SCH# 2023050214

Volume 1
Chapters 1 through 10

ENTERPRISE SOLAR STORAGE PROJECT
by Enterprise Solar Storage, LLC *(PP23401)*

Site 1
Zone Change Case No. 60, Map No. 196
Conditional Use Permit No. 62, Map No. 196
Nonsummary Vacations Map No. 196

Site 2
Specific Plan Amendment No. 34, Map No. 196
Zone Change Case No. 61, Map No. 196
Conditional Use Permit No. 63, Map No. 196
Nonsummary Vacations Map No. 196

Site 3
Zone Change Case No. 3, Map No. 195
Zone Change Case No. 62, Map No. 196
Conditional Use Permit No. 2, Map No. 195
Conditional Use Permit No. 64, Map No. 196
Conditional Use Permit No. 65, Map No. 196
Specific Plan Amendment No. 35, Map No. 196
Nonsummary Vacations Map No. 195
Nonsummary Vacations Map No. 196

Site 4
Specific Plan Amendment No. 4, Map No. 212
Zone Change Case No. 4, Map No. 195
Zone Change Case No. 3, Map No. 212
Conditional Use Permit No. 3, Map No. 195
Conditional Use Permit No. 6, Map No. 212
General Plan Amendment No. 3, Map No. 212
Nonsummary Vacations Map No. 212

Site 5 (Substation)
Conditional Use Permit No. 20, Map No. 197

Kern County
Planning and Natural Resources Department
Bakersfield, California
November 2023

Level of Significance after Mitigation

With implementation of Mitigation Measures MM 4.1-4 through MM 4.1-7, impacts would be less than significant.

SCE Transmission Line Relocation and Interconnection Work

Southern California Edison (SCE) will relocate an existing 66 kilovolt (kV) transmission line to remove it from the project footprint. An approximately 20-foot-wide access road will be graded within SCE's right-of-way to facilitate construction and future maintenance of the transmission line. After the relocated transmission line is installed and energized, the old transmission line will be dismantled using equipment used to install and tension the relocated transmission line. Additionally, SCE will interconnect the project's 500 kV transmission line into the SCE-owned Windhub Substation 500 kV banks located on the south side of the facility. It is anticipated that SCE's interconnection work will involve upgrades within the substation footprint and potential relocation of existing high-voltage transmission poles and conductors, and/or installation of new poles and conductors adjacent to or within the footprint of the Windhub Substation. The relocated transmission line and interconnection work would result in less than significant aesthetic impacts and would result in minimal changes to the appearance of existing SCE facilities, producing no noticeable change to existing views. The SCE work would not increase the amount of lighting on-site or result in glare during construction and operation. SCE would implement best management practices during construction and operation, and impacts would be less than significant.

4.1.5 Cumulative Setting Impacts and Mitigation Measures

As presented in Chapter 3, *Project Description*, of this Draft EIR, there are 18 cumulative projects. These have the potential to result in cumulative impacts to aesthetics when considered together with the project. The "scarcity" rating criterion is likely to be impacted by widespread development in the area, as unobstructed views of regional topographical features and undeveloped lands would be less available as acreage is developed with PV panels, wind energy projects, and new transmission lines.

As discussed above, the project would have less than significant impacts as it relates to scenic vistas as project distance, topography, and intervening development would reduce the visual prominence of the proposed solar development. Due to the developed nature of the landscape visible from the PCT (i.e. existing solar facilities, wind energy facilities and transmission line infrastructure) in the general project area, project development would not have a substantial adverse effect on existing views from the PCT. As such, cumulative impacts would be less than significant and not cumulatively considerable.

With regard to impacts related to damaging scenic resources within a scenic highway, the project would not be visible from any Officially Designated State or County Scenic Highway as there are no Officially Designated State or County Scenic Highways in the vicinity of the project site. SR-58 and SR-14, which are in proximity to the project site, are eligible state scenic highways. With the exception of Site 2 (Western) and the gen-tie line, most of the project site would not be clearly visible from the eligible scenic segment of SR-14. Views to the project are available from SR-58, particularly Site 3 (Eastern) located immediately to the south. Components of the project would not substantially alter existing long-distance views of the mountain and valley landscape or other natural features visible from the designated scenic segments of SR-

The importance of coccidioidomycosis as an occupational disease has increased in the southwestern United States. This report discusses the aspects of the disease in terms of its geography, the agent, occupation, dust conditions, and various other factors. A control program is outlined.

EXPOSURE FACTORS IN OCCUPATIONAL COCCIDIOIDOMYCOSIS

Lawrence L. Schmelzer, M.P.H., and Irving R. Tabershaw, M.D., F.A.P.H.A.

THE rapid and increasing influx of industry and agriculture into the southwestern United States has heightened the importance of coccidioidomycosis as an occupational disease. Before 1938, this disease was of little interest because relatively few clinical cases were recognized and the morbidity caused by primary infection was not appreciated. In that year, Dickson and Gifford,[1] reporting on several years of study, clearly established that the benign, primary form of the disease was an important cause of illness in the endemic areas, and that the disease is caused by inhalation of spores of Coccidioides immitis. During World War II, coccidioidomycosis was shown to be the cause of significant illness among soldiers in training at camps in the endemic areas. Studies by Smith, et al.,[2] showed that preventive measures, notably dust control, were effective in reducing the rate of infection and the seriousness of epidemics.

Epidemics have also been reported in susceptible groups of university personnel that entered endemic areas. In 1942, Davis, et al.,[3] reported infection in seven of 14 students and staff from Stanford University who made a field trip to the San Joaquin Valley. In 1954, four students from the University of California at Los Angeles contracted the disease in similar circumstances, and one student, not participating in the field trip, developed disease through the handling of contaminated specimens in the laboratory.[4] In 1962, 100 per cent infection was reported in a group of 16 persons from UCLA who participated in an archaeological field study near Los Banos, Calif.[4] Again in 1965 three students from UC Berkeley developed clinical disease after a field trip in the same general area.

Coccidioidomycosis ranks high among the infectious occupational diseases[5] as shown in Table 1. Further, the case fatality rate closely parallels that of tuberculosis as shown in Table 2.[6] These rates are based on reported clinically recognized cases. In both diseases, primary infection usually goes unnoticed. Fatality rates for both diseases are considerably less when based on total number of infections.

In spite of the fact that coccidioidomycosis is in most instances inapparent or mild, the disease causes significant dis-

Table 1—Number of disability cases of selected occupational diseases in California by fiscal year of report*

Disease	Number of disability cases 1962-1963	1963-1964	1964-1965	3-year total
Coccidioidomycosis	21	34	27	82
Tuberculosis	28	29	24	81
Anthrax, brucellosis, Q fever	11	13	13	37
Psittacosis	1	1	1	3
Tetanus	1	2	1	4

* From: Work Injuries in California, Quarterly Statistical Summary. State of California Department of Industrial Welfare, Division of Labor Statistics and Research.

ability in California workers. Although the 106 cases reported in six years[7] may not appear an unduly large number, the degree of disability in these cases is noteworthy (Table 3).

A large proportion required hospitalization and absence from work lasting weeks or months was not unusual. As late as 1957, coccidioidomycosis caused more disability at Williams Air Force Base in Arizona than any other disease including the upper respiratory infections.[8] While the average incidence of both infections was the same, the average disability of 34.6 days caused by coccidioidomycosis was seven times higher than that caused by upper respiratory infections.

Since it is not now possible to provide artificial immunity to those entering an endemic area and since susceptibility to coccidioidomycosis is essentially universal, the introduction of industrial or agricultural workers into endemic areas carries with it the responsibility of assessing the hazard of the disease to such populations. None of the exposure factors in the production of coccidioidomycosis is susceptible to control to the degree necessary to prevent infection entirely. Sufficient knowledge of the direct and predisposing causes of the disease, however, does exist so that it may be possible to reduce both the incidence of infection and its severity.

Geography

Coccidioides immitis has been reported only in the arid and semiarid regions of southwestern United States, in Mexico, Central America, Venezuela, and in the Chaco region of Argentina. The areas of endemicity roughly parallel the boundaries of the lower Sonoran Life Zone, which is characterized by scant rainfall, hot dry summers, alkaline soil, mild winters, sparce flora and fauna and, until recently, few human inhabitants (Figure 1).[9] The creosote bush, Larrea tridentata, is often considered a specific indicator of this life zone.

Evaluation of geography and ecology as exposure factors is complicated by the fact that areas within the lower Sonoran Life Zone may be free of C. immitis, and conversely small endemic areas may occur outside the zone. However, the potential of serious sequelae to infection is sufficient justification to consider any entry into suspected endemic areas as leading to exposure to the disease.

Infectious Agent

Spores of C. immitis are found in the first few inches of the soil and in larger numbers in the vicinity of rodent bur-

Table 2—Case fatality rates for coccidioidomycosis and tuberculosis in California 1960-1963*

	Case fatality rates† 1960	1961	1962	1963
Coccidioidomycosis	8.6	12.8	12.3	11.1
Tuberculosis	15.7	12.7	13.1	12.1

* From: California Public Health Statistical Report 1963. Part II Communicable Diseases. California State Department of Public Health.
† Case fatality rates are per 100 cases reported.

Table 3—Number of cases of occupational coccidioidomycosis reported in California during the period January, 1959, to March, 1965, by industry*

Industry		Cases reported
Agriculture		32
Animal husbandry	16	
Field crops	11	
Gardening	3	
Other	2	
Construction		39
Equipment operator	19	
Truck driver-mechanic	6	
Building trades	14	
Professional		22
Engineer	9	
Scientist	8	
Geologist	5	
Other and unknown		13
Total		106

* From: Summary of Reports of Occupationally Contracted Coccidioidomycosis 1959-1965. California State Department of Public Health, Bureau of Occupational Health.

rows.[10] These spores produce mycelial growth during the winter rains and, as the soil dries in the spring, arthrospores are again produced. Tests have shown that the concentration of arthrospores in the soil is highest at the end of the wet season and becomes lower as the dry season progresses. Season and rainfall patterns must therefore be considered in the evaluation of exposure potential for persons entering endemic zones. Importance of this has been shown by Smith, et al.,[2] in the San Joaquin Valley, and by Hugenholtz in a study of 13 years' experience at Williams Air Force Base in Arizona.[11] The average number of infections of base personnel was found to decrease during rainy months and to increase during the dry periods.

The highly infectious nature of C. immitis is illustrated by the fact that from seven to 15 arthrospores insufflated intranasally into mice causes infection and dissemination to the liver and spleen in 35 per cent to 40 per cent of susceptible

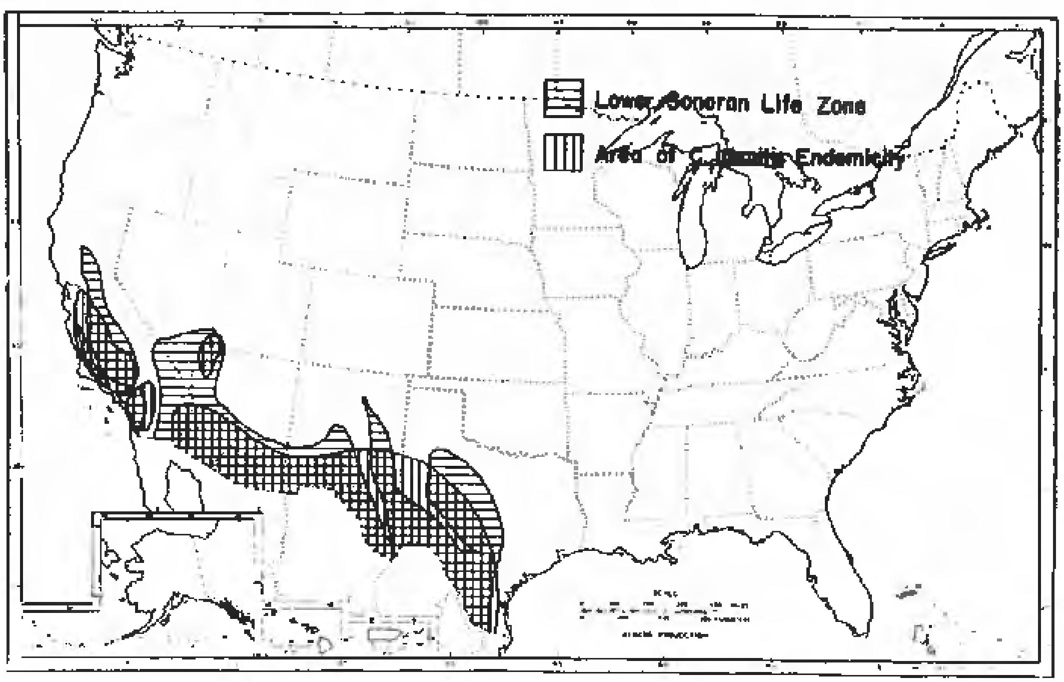


Figure 1—Lower Sonoran Life Zone and area of Coccidioides immitis endemicity in the United States [After Smith, C. E.[9]]

animals.[12] The organism has very simple nutritional requirements for growth, grows on practically any medium, and has been shown to prefer a saline environment[13] including body fluids.

Physical Properties

Typical mature hyphae of C. immitis yield barrel-shaped arthrospores, approximately 2.5 microns in diameter and 4 microns long, alternating with smaller sterile cells. The empty cells rupture easily to free the spores, leaving on the latter cell wall fragments which add to the length of the spore and also decrease the apparent specific gravity. Particle dynamics help to explain the highly infectious nature of the C. immitis and its wide distribution by winds. The important factors are terminal settling velocity and impingement forces, both of which are proportional to the particle size and specific gravity. Although actual spore dimensions vary and the specific gravity is not accurately known, it can be postulated that effective spore diameter is about 5 microns and its specific gravity is about 0.75. Terminal settling velocity for the spores is 0.01 centimeters per second when computed on the basis of these figures. In comparison, a quartz particle having this terminal settling velocity would have a diameter of 1.4 microns. From this it is clear that spores of C. immitis are easily air-borne, settle slowly, can penetrate into the smallest bronchioles and alveoli, and that a significant percentage of retention in the lung can be expected.

Dust Conditions

In the heat of early summer, what little ground cover that exists in the endemic areas withers and dies, winds disturb the surface dust and lift the spores into the air. The slow terminal settling velocity permits the spores to become essentially a permanent atmospheric contaminant under turbulent wind conditions. Such conditions are not unusual in arid regions where thermal phenomena generate severe atmospheric disturbances. Very small, intense, local whirlwinds, known as "dust devils," can raise dust containing large numbers of spores if they pass over pockets of high concentration in the soil. Large, rapidly moving air masses are also common, such as the "Santa Ana Winds" which blow from the Mojave Desert south into the San Fernando Valley. These winds will carry spores into nonendemic areas but the concentration will be low because of the nonselective raising of dust. Soil tests, therefore, cannot assure that an area within or close to an endemic zone is free of the organism and surface travel through or near endemic areas has resulted in exposure and infection.

Occupation

Varying racial and sexual susceptibility influences the severity and disability from coccidioidomycosis. However, since it results from inhalation of air-borne arthrospores, occupational factors must be considered in relation to the magnitude of probable dust exposure. It has been shown that a susceptible population entering an endemic area can experience an annual infection rate of about 20 per cent.[2] No overt dust exposure is necessary; infection can result from wind-borne spores traveling long distances in turbulent air conditions. Labor groups where occupation involves close contact with the soil are at greater risk, especially if the work involves dusty digging operations. The period of disability in cases of occupational coccidioidomycosis reported in California is classified by industry in Table 4.[7] The significant differences in the periods of disability can be ascribed to the variations in exposure resulting from occupation.

Agricultural workers suffered less disability because their exposure is probably to a few spores at a time. In field crop operations, burrowing rodents are not tolerated and the focus of endemicity associated with them is not present. Tilling of the soil will tend to disperse pockets of high spore concentration so that the dust raised can be expected to contain a relatively low concentration of spores. Similarly, a sheepherder would not be expected to receive a heavy, concentrated dose of arthrospores. This would tend to produce milder disease and a large proportion of inapparent and mild infections.

In the construction trades, exposures may be very different depending on the specific operations. Pipeline, highway, and utility construction often involves work in remote areas where the soil has not been disturbed and where foci of endemicity are usual. When these foci are disturbed, the dust raised can have a high concentration of spores. Digging of foundation and pipe trenches in residential or commercial buildings can lead to similar massive exposure. Similarly, engineers involved in highway or other heavy construction may be subjected to heavy doses if they are working with the construction crews, but may suffer exposure comparable to an agricultural worker if they are only surveying.

The exposures of professionals are highly variable and difficult to predict. Groups of paleontologists and archaeologists have suffered 100 per cent infection when their pursuits led them to dig in or around rodent burrows. Other groups digging in endemic areas have completely escaped infection.

Discussion

Prevention of coccidioidomycosis is complicated by the fact that the organism is a natural and persistent inhabitant of the environment. Determination of concentration of spores in specific locations is not feasible because the selection of appropriate sampling sites and identification of C. immitis is difficult and time-consuming. Furthermore, as previously mentioned, spores can be air-borne for long periods of time and travel great distances. Consequently, the importation of any susceptible labor force into endemic areas carries with it the responsibility for reducing the rate and severity of infection through whatever dust control measures are possible and for providing a vigorous program of medical surveillance.

Control of dust for the prevention of coccidioidomycosis is not a simple matter because of the wide variations in exposures. General dust control measures can afford some degree of protection to all persons working and living in an en-

Table 4—Number of disability cases of occupational coccidioidomycosis in California by length of disability and industry for the period January, 1959, to June, 1962

Industry	Period of disability in days					Total
	0	1–14	15–29	30–50	>60	
Agriculture	6	0	4	4	4	18
Construction	2	1	0	5	13	21
Professions	5	1	2	6	8	22

From: Summary of Reports of Occupationally Contracted Coccidioidomycosis, 1959-1965. California State Department of Public Health, Bureau of Occupational Health.

demic area. As shown by Smith,[2] oiling of parade grounds and barracks areas in military establishments reduced the rate of infection. Similarly, planting of trees and lawns around residences and industrial plants can reduce the rate of infection by about half.[14] Further protection can be provided by filtering and conditioning of air supplied to plants and offices, but this is not complete since it does not control infection resulting from exposure outside the working hours. Protection of agricultural workers and animal husbandmen to any realistic degree is exceedingly difficult. Their exposure to dust is an inseparable part of their employment and working conditions preclude the effective use of respiratory protection.

Operators of heavy earth moving equipment can be effectively protected during working hours by providing air conditioned cabs. This not only protects from coccidioidomycosis but also controls exposure to other dust, noise, and engine exhaust fumes. Efficient and comfortable hoods for individual use are now available with powered blowers for providing filtered air. These are useful on smaller earth moving equipment and for semistationary operations such as oil well drilling. Exposures resulting from manual digging are less easily controlled. Continued use of respirators is very uncomfortable in the usually high ambient temperatures and workers resist use of this kind of protection. The wearing of respirators can, however, be enforced during recognized periods of high exposure. For instance, building tradesmen should wear respirators when digging fouudation excavations or pipeline trenches. Similarly, highway engineers can wear respirators when working around earth moving machinery but could dispense with this when surveying ahead of or behind construction crews. Scientists should be protected during actual digging operations but not necessarily during exploration.

Skin testing for previous infection by C. immitis is easy to perform and defines the immune population. All persons hired for work in endemic areas (or whose assignments take them there) should be tested. Assigning immune workers to operations involving known heavy exposures can effectively reduce the incidence of infection. Hiring lifelong residents of the endemic areas can also reduce the incidence of infection since the level of immunity in these people can be expected to be high. This should not, however, be substituted for a program of skin testing and medical surveillance. Negroes and Filipinos have been shown to be more susceptible to developing the highly fatal disseminating form of the disease.[15] Unless such individuals are shown to have developed immunity, they should whenever possible be assigned to work in areas or at jobs where exposure to high concentrations of spores will be minimal.

Periodic medical examinations or interviews are useful to discover a history of low grade or subclinical infection and to evaluate the level of health of the individual. This examination must include repeated skin testing of susceptibles until the patient shows conversion to a positive reaction signifying immunity. Such an individual can then be dropped from medical surveillance for coccidioidomycosis. The medical management of any respiratory ailment suffered by persons at risk who are not immune to coccidioidomycosis should include a skin test.

Research is presently being pursued to develop an effective antigen for producing artificial active immunity to coccidioidomycosis. If successful, this vaccine will make possible the total protection of populations entering endemic areas. However, since man is not the reservoir of the disease, but only an accidental host, eradication will not be possible. Consequently the efforts to prevent disability from coccidioidomycosis must be continued so long as susceptible populations enter endemic areas.

Control Program

A program for limiting the incidence of occupational coccidioidomycosis and reducing the severity of disease in those who become infected would entail the following:

1. Determine if the work location is within the endemic area.
2. Hire resident labor whenever available, particularly if dust exposures may be heavy.
3. Establish a medical program including:
   a. Skin tests on all new employees. If positive they can be assigned to any job; if negative, especially Negroes and Filipinos, job exposure must be carefully evaluated. If heavy concentration of dust cannot be avoided, those with negative skin tests should not be employed at that job.
   b. Retest of susceptibles. This should be continued every three to six months until immunity is demonstrated by conversion to a positive reaction.
   c. Prompt treatment of respiratory illness in susceptibles. Coccidioidomycosis is a suspect in such illnesses (and if such is the case early chemotherapy can reduce the severity).
4. Educate the exposed population.
   a. New employees should be informed of the potential of infection and its consequences.
   b. All employees should be advised to seek prompt medical treatment for any respiratory illness and to inform the attending physician of their possible exposure to the fungus, particularly if the physician practices outside the endemic area.
5. Control dust exposure by:
   a. Oiling or planting of areas around plants, offices, and residences.
   b. Filtering and conditioning of air supplies to plants and offices; providing air conditioned cabs on heavy equipment.
   c. Providing respirators, air supplied helmets, and the like, as indicated.
   d. Preventing transport of C. immitis outside endemic area by thoroughly cleaning equipment and specimens before shipment to other work locations.

REFERENCES

1. Dickson, E. C., and Gifford, M. A. Coccidioides Infection (Coccidioidomycosis); the Primary Type of Infection. Arch. Int. Med. 62:852–871, 1938.
2. Smith, C. E.; Beard, R. R.; Rosenberg, H. G.; and Whiting, E. G. Effect of Season and Dust Control on Coccidioidomycosis. J.A.M.A. 132:833–838, 1946.
3. Davis, B. L.; Smith, R. T.; and Smith, C. E. An Epidemic of Coccidioidal Infection (Coccidioidomycosis). Ibid. 118:1182–1186, 1942.
4. Huberty, G. T. An Epidemic of Coccidioidomycosis. J. Am. College Health A. 12:131 (abstract), 1963.
5. State of California Department of Industrial Welfare. Work Injuries in California. Quart. Statist. Summary. State of California Department of Industrial Welfare. P. O. Box 965, San Francisco, Calif.
6. California State Department of Public Health. California Public Health Statistical Report, 1963, Part II, Communicable Diseases. California State Department of Public Health, 2151 Berkeley Way, Berkeley, Calif.
7. California State Department of Public Health. Summary of Reports of Occupationally Contracted Coccidioidomycosis 1959-1965. California State Department of Public Health, 2151 Berkeley Way, Berkeley, Calif.
8. Hugenholtz, P. G. Skin Test Survey at Williams Air Force Base, Arizona. Proceedings of Symposium on Coccidioidomycosis. PHS Publ. No. 575. Atlanta, Ga.: U. S. Public Health Service, 1957.
9. Smith, C. E. Diagnosis of Pulmonary Coccidioidal Infections. California Med. 75:385–391, 1951.
10. Elconin, A. F.; Egeberg, R. O.; and Lubarsky, R. Growth Patterns of Coccidioides immitis in the soil of endemic areas. Proceedings of Symposium of Coccidioidomycosis. PHS Publ. No. 575. Atlanta, Ga.: U. S. Public Health Service, 1957.
11. Hugenholtz, P. G. Climate and Coccidioidomycosis. Proceedings of Symposium on Coccidioidomycosis. PHS Publ. No. 575. Atlanta, Ga.: U. S. Public Health Service, 1957.
12. Kong, Y. M.; Levine, H. B.; Madin, S. H.; and Smith, C. E. Fungal Multiplication and Histopathological Changes in Vaccinated Mice Infected with Coccidioides immitis. J. Immunol. 92:779–790, 1964.
13. Elconin, A. F.; Egeberg, R. O.; and Egeberg, M. C. Significance of Soil Salinity on the Ecology of Coccidioides immitis. J. Bact. 87:500–503, 1964.
14. Maddy, K. T.; Doto, I. L.; Furcolow, M. L.; Lehan, P. H.; Rubin, H.; Greene, J. C.; Casper, E.; and Coleman, P. J. Coccidioidin, Histoplasmin, and Tuberculin Sensitivity of Students in Selected High Schools and Colleges in Arizona. Proceedings of Symposium on Coccidioidomycosis. PHS Publ. No. 575. Atlanta, Ga.: U. S. Public Health Service, 1957.
15. Gifford, M. A.; Buss, W. C.; and Douds, R. J. Data on Coccidioides Fungus Infection, Kern County, 1900-1936. Kern County Health Dept. Annual Rep. 1936-1937. Bakersfield, Calif.: Kern County Health Department.

Mr. Schmelzer is environmental health and safety officer, and Dr. Tabershaw is professor of occupational medicine and head, Environmental Health and Safety, Office of Environmental Health, University of California, Berkeley, Calif.

This paper was presented before the Occupational Health Section of the American Public Health Association at the Ninety-Fourth Annual Meeting in San Francisco, Calif., October 31, 1966.

Message from the Director

Valley Fever affects thousands of residents in Kern County every year, and its impact can be devastating. As someone who has personally faced the challenges of this illness, I can speak to the gravity of its effects. In May of 2024, I was diagnosed with a rare and severe form of Valley Fever called cocci meningitis. Cocci meningitis is a permanent Valley Fever infection in the brain and spine. This experience has given me an intimate understanding of the long-term consequences Valley Fever can have on our health, as well as the challenges—both positive and negative—that accompany antifungal treatment.

While it is easy to become complacent or desensitized to the warnings about Valley Fever, it is crucial that we remain vigilant. The symptoms of Valley Fever can often be mistaken for other illnesses, but the earlier it is diagnosed, the better our chances of preventing severe complications. I encourage everyone to familiarize themselves with the symptoms of Valley Fever and take them seriously. If you are experiencing symptoms, ask your doctor for a Valley Fever test.

Early diagnosis and treatment can make a significant difference in your health outcome. Let's all work together to raise awareness, promote early detection, and reduce the impact that Valley Fever has on our community. Thank you for visiting our website, and for taking the time to learn more about Valley Fever. Your health matters, and by staying informed, you can protect yourself and your loved ones in our community.

— Brynn Carrigan, Director of Public Health Services

Valley Fever, also called coccidioidomycosis or "cocci," is a disease caused by a fungus that grows in soil and dirt in parts of California and the southwestern U.S. When the soil is disturbed by wind or activity, tiny fungal spores can become airborne and be breathed in, potentially making people and animals sick. The infection usually affects the lungs, causing symptoms like cough, fever, chest pain, and fatigue. While some may experience mild or no symptoms, others can develop severe illness. Early detection is key—if you've had a lingering cough, fever, or painful breathing for more than two weeks, ask your doctor about Valley Fever. It's in the air. Be aware!

‹

Quon Louey
Executive Director,
Telehealthdocs

Learn More

Bryı
Direc
Healt

Learr

›

Valley Fever: Many Faces, Many Stories

Valley Fever doesn't discriminate—it can affect anyone. The tiny fungal spores that cause it are carried in the air, stirred up by wind or dust, and breathed in without anyone realizing it. Some people may only experience mild symptoms, like a lingering cough or fatigue, while others face severe illness that can last for months or even years. No two experiences are the same.

Behind every case of Valley Fever is a real person with a story to tell—a mother, a teacher, a director, a neighbor. Each journey is different. Their voices bring awareness. Their stories bring hope. These are the faces of Valley Fever.

Have a story you'd like to share?

Email Us

Microsoft Power BI   Pages

Do you have a question about Valley Fever? Email: valdovinosli@kerncounty.com

Message from the Director

Valley Fever affects thousands of residents in Kern County every year, and its impact can be devastating. As someone who has personally faced the challenges of this illness, I can speak to the gravity of its effects. In May of 2024, I was diagnosed with a rare and severe form of Valley Fever called cocci meningitis. Cocci meningitis is a permanent Valley Fever infection in the brain and spine. This experience has given me an intimate understanding of the long-term consequences Valley Fever can have on our health, as well as the challenges—both positive and negative—that accompany antifungal treatment.

While it is easy to become complacent or desensitized to the warnings about Valley Fever, it is crucial that we remain vigilant. The symptoms of Valley Fever can often be mistaken for other illnesses, but the earlier it is diagnosed, the better our chances of preventing severe complications. I encourage everyone to familiarize themselves with the symptoms of Valley Fever and take them seriously. If you are experiencing symptoms, ask your doctor for a Valley Fever test.

Early diagnosis and treatment can make a significant difference in your health outcome. Let's all work together to raise awareness, promote early detection, and reduce the impact that Valley Fever has on our community. Thank you for visiting our website, and for taking the time to learn more about Valley Fever. Your health matters, and by staying informed, you can protect yourself and your loved ones in our community.

— Brynn Carrigan, Director of Public Health Services

Valley Fever, also called coccidioidomycosis or "cocci," is a disease caused by a fungus that grows in soil and dirt in parts of California and the southwestern U.S. When the soil is disturbed by wind or activity, tiny fungal spores can become airborne and be breathed in, potentially making people and animals sick. The infection usually affects the lungs, causing symptoms like cough, fever, chest pain, and fatigue. While some may experience mild or no symptoms, others can develop severe illness. Early detection is key—if you've had a lingering cough, fever, or painful breathing for more than two weeks, ask your doctor about Valley Fever. It's in the air. Be aware!

Quon Louey
Executive Director, Telehealthdocs

Learn More

Bryı
Direc
Healt

Learı

Valley Fever: Many Faces, Many Stories

Valley Fever doesn't discriminate—it can affect anyone. The tiny fungal spores that cause it are carried in the air, stirred up by wind or dust, and breathed in without anyone realizing it. Some people may only experience mild symptoms, like a lingering cough or fatigue, while others face severe illness that can last for months or even years. No two experiences are the same.

Behind every case of Valley Fever is a real person with a story to tell—a mother, a teacher, a director, a neighbor. Each journey is different. Their voices bring awareness. Their stories bring hope. These are the faces of Valley Fever.

Have a story you'd like to share?

Email Us

Microsoft Power BI Pages

Do you have a question about Valley Fever? Email: valdovinosli@kerncounty.com

Mycopathologia
DOI 10.1007/s11046-016-0105-5

Large-Scale Land Development, Fugitive Dust, and Increased Coccidioidomycosis Incidence in the Antelope Valley of California, 1999–2014

Aaron J. Colson · Larry Vredenburgh · Ramon E. Guevara · Natalia P. Rangel · Carl T. Kloock · Antje Lauer

Received: 27 May 2016 / Accepted: 16 December 2016

Abstract Ongoing large-scale land development for renewable energy projects in the Antelope Valley, located in the Western Mojave Desert, has been blamed for increased fugitive dust emissions and coccidioidomycosis incidence among the general public in recent years. Soil samples were collected at six sites that were destined for solar farm construction and were analyzed for the presence of the soil-borne fungal pathogen *Coccidioides immitis* which is endemic to many areas of central and southern California. We used a modified culture-independent nested PCR approach to identify the pathogen in all soil samples and also compared the sampling sites in regard to soil physical and chemical parameters, degree of disturbance, and vegetation. Our results indicated the presence of *C. immitis* at four of the six sites, predominantly in non-disturbed soils of the Pond-Oban complex, which are characterized by an elevated pH and salt bush communities, but also in grassland characterized by different soil parameters and covered with native and non-native annuals. Overall, we were able to detect the pathogen in 40% of the soil samples ($n = 42$). Incidence of coccidioidomycosis in the Antelope Valley was positively correlated with land use and particulate matter in the air (PM10) (Pearson correlation coefficient >0.5). With the predicted population growth and ongoing large-scale disturbance of soil in the Antelope Valley in coming years, incidence of coccidioidomycosis will likely further increase if policy makers and land developers continue to ignore the risk of grading land without implementing long-term dust mitigation plans in Environmental Impact Reports.

Keywords Fugitive dust · Renewable energy · Coccidioidomycosis · Mojave Desert · Soil

Electronic supplementary material The online version of this article (doi:10.1007/s11046-016-0105-5) contains supplementary material, which is available to authorized users.

A. J. Colson · N. P. Rangel · C. T. Kloock · A. Lauer (✉)
Department of Biology, California State University Bakersfield (CSUB), 9001 Stockdale Highway, Bakersfield, CA 93311-1022, USA
e-mail: alauer@csub.edu

R. E. Guevara
County of Los Angeles Department of Public Health, 600 S. Commonwealth Ave, Suite 700, Los Angeles, CA 90005, USA

L. Vredenburgh
Bureau of Land Management, Bakersfield Field Office, 3801 Pegasus Drive, Bakersfield, CA 93308, USA

Introduction

Large-scale land development in the Antelope Valley, located in northern Los Angeles County in California, provides new residences for expanding populations, facilities for businesses, fields for agriculture, and more recently provided opportunities for renewable energy production. However, arid and semiarid areas in the Southwestern US may require better care in managing soil disturbance from such projects because of greater risk of fugitive dust emissions and coccidioidomycosis, caused by the soil-born fungal pathogen *Coccidioides* spp. Fugitive dust is the suspension of particulate matter in the air by wind or human activities usually indicated as particulate matter up to 10 μm (PM10). The particulate matter is primarily soil but can contain crystalline silica, asbestos fibers, heavy metals, and airborne spores and conidia from microorganisms. Fugitive dust in general can cause breathing difficulties, low acute and chronic respiratory illnesses, increased risk of death from aggravated heart or lung disease [2, 12, 25, 27], increased risk of traffic accidents from poor road visibility [4], and reduced agricultural crop yield and desertification [75]. Fugitive dust emissions observed in the Antelope Valley frequently exceed California standards of 50 μg/m^3 for PM10 (24 h averages) and 30 μg/m^3 (annual arithmetic mean), respectively, which are stricter than federal standards (see http://www.arb.ca.gov/research/aaqs/caaqs/caaqs.htm for current California Ambient Air Quality Standards [CAAQS]). The increase in air pollution with coarse particulate matter (PM10) has raised the concern of public health officials and the general public [59], because of increased incidence of coccidioidomycosis among residents of the Antelope Valley (County of Los Angeles Department of Public Health, Annual Morbidity and Specials Studies Reports 2000–2014). Incidence of coccidioidomycosis in the Antelope Valley increased about 13-fold between 2000 and 2014 (supplementary figure S1). Strong Santa Ana winds can deliver dust from the desert to the LA Basin and deliver conidia of the pathogen to an area that is thought to be non-endemic for the pathogen [58].

The Antelope Valley is located in the Western Mojave Desert within the endemic zone of *Coccidioides* spp. which is comprised of certain areas in Arizona, California, Nevada, New Mexico, Texas, Utah, Washington, and Central and South America (see map in [55]). Fugitive dust that carries arthroconidia of *Coccidioides immitis* or *C. posadasii* can cause coccidioidomycosis in humans and animals primarily through inhalation of these dormant forms of the pathogen. Coccidioidomycosis primarily affects the pulmonary system in people and animals [16, 23], but dissemination of the disease to other organs can occur [28, 57]. Although about 60% of infected people develop mild to no symptoms, the other 40% experience weeks to months of debilitating disease that can include fatigue, shortness of breath, cough, fever, night sweats, loss of appetite or weight, chest pain, headache, body aches, skin rash, and pneumonia [63]. Less than 5% of these patients develop disseminated coccidioidomycosis, which increases the risk of lifelong complications and death [23, 43, 71]. Despite considerable efforts, no vaccine to protect humans from coccidioidomycosis currently exists [74].

The issue of fugitive dust carrying *Coccidioides* spp. arthroconidia is important not just for workers involved in land development projects, but also for residents of nearby communities, residents of newly built neighborhoods, and visitors working, studying, or travelling through the area. Furthermore, strong winds can transport conidia far distances, sometimes hundreds of miles, which can cause disease in humans and animals in non-endemic areas [21, 34, 35, 56].

The Antelope Valley of California provides an opportunity to examine how changes in the environment due to large-scale land development effect incidence of coccidioidomycosis in humans. Consisting of over 1800 mi^2 of fertile lands, the Antelope Valley is located approximately 2500 ft above sea level and is part of the "Lower Sonoran Lifezone" [53], sometimes referred to as the "High Desert," a common name for a subregion located mostly in northwestern San Bernardino County, northeastern Los Angeles County, and far eastern Kern County in areas above 2000 ft in altitude [39, 77]. The valley experiences an annual precipitation of 6–9 inches per year, a mean annual high temperature of 98 °F in the summer and 59 °F in the winter, with temperatures commonly above 100 °F in July and August [51]. Mountains along the Southern and Western border of the Mojave Desert block most of the moisture-bearing westerly winds from the coast, limiting precipitation and air humidity, and strong prevailing winds can result in severe dust storms [62].

The Antelope Valley has the greatest potential for land development in Los Angeles County, and its land use increased notably between 2001 and 2011 (Fig. 1). Guevara et al. [33] showed that disturbance of soil during the "housing boom" that peaked between 2004 and 2005 was positively correlated with a spike in coccidioidomycosis incidence at the same time. In recent years, the Antelope Valley has become the focus of renewable energy projects to provide solar- and wind-generated energy for Southern California [11, 38]. Solar farms constructed by multiple companies will ultimately cover more than 30,000 acres in the valley (e.g. [17, 19, 31], for an overview of all planned renewable energy projects). Overall, the DRECP affects approximately 22,858,000 acres of semiarid and arid soils in the counties of Los Angeles, Kern, Inyo, San Bernardino, Riverside, Imperial, and San Diego.

Purpose and Scope

This project aimed to determine whether *C. immitis* is established in soils destined for photovoltaic system construction in the Antelope Valley, characterize the ecologic features of *C. immitis* positive sites, and correlate field findings with existing epidemiologic, geologic, and geographic data. Soil samples collected at six photovoltaic system sites either completed or destined for construction by 2014 or 2015 (Bureau of Land Management [BLM], CA, DRECP) were tested for the presence of *Coccidioides* spp. with a culture-independent polymerase chain reaction (PCR)-based approach. The sampled sites included non-disturbed locations covered with natural vegetation, predominantly *Atriplex polycarpa*; disturbed grassland with native and non-native annuals; fallow agricultural fields; and land impacted by sheep grazing. With this study, we hope to raise awareness of an increasing environmental health hazard that has been neglected in the past. Policy makers and others involved with large-scale land development projects could use the results from this study to implement better dust control approaches with more stringent requirements to reduce fugitive dust emissions and incidence of coccidioidomycosis and other dust-related illnesses among construction workers and the general public.

Materials and Methods

Soil Sampling Area

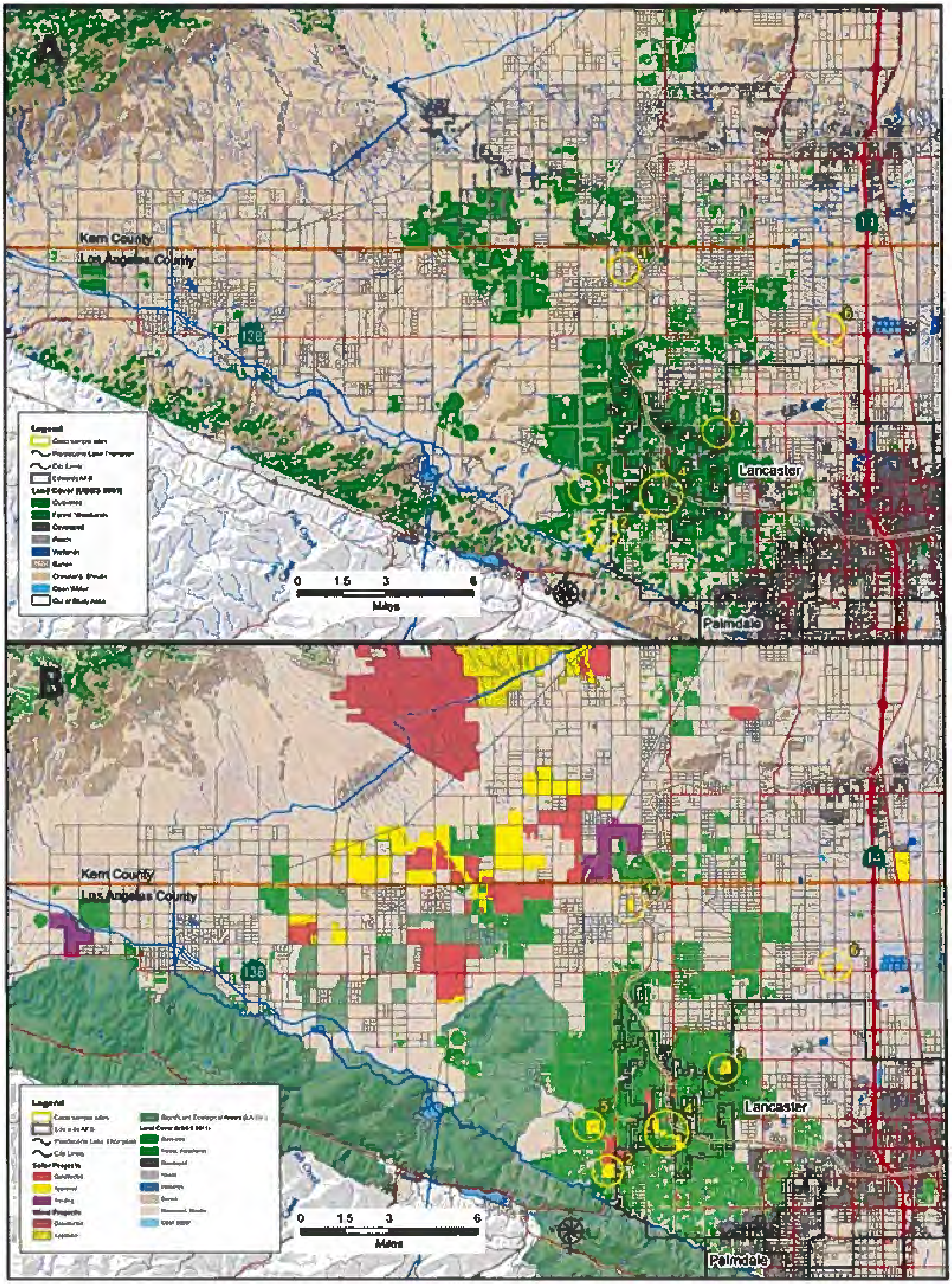
All soil sampling sites were located in the Antelope Valley subsection of the Western Mojave Desert in northern Los Angeles County west of the city of Lancaster and south of the rural town Antelope Acres (Fig. 2). The Antelope Valley watershed is a large topographic depression with no hydrologic outlet to the ocean. The runoff into the basin from surrounding creeks is conveyed via broad ephemeral washes toward several dry lakes. Two large dry basins, or playas, the Rosamond and Roger's dry lake beds (Kern County) form dominant natural landscape features within the Antelope Valley and are located east of the sampling area.

Ecological Landscape Characterization of Sampling Sites

Soil samples were collected at six sites destined for solar panel construction. These locations included site 1: North Lancaster Silverado Project, site 2: West Antelope Silverado Project, site 3: American Silverado Project, site 4A and B: Antelope Silverado Project, site 5: Silver Sun Silverado Project, and site 6: Lancaster WAD Project. Soil parameter information for all sites was obtained from the United States Department of Agriculture (USDA) websoilsurvey database. Coordinates of all sampling spots were documented, and the appearance of soils, as well as the vegetation cover (plant species and degree of coverage and disturbance), was documented. Plant species were identified using the Jepson Desert Manual [5] and other literature [49, 54]. Rodent activity was observed at all sites in form of pellets, burrows or both. Soil samples were collected from soil types that were dominant in the locations destined for solar panel constructions and were collected from 5 to 7 cm depth. The pH of all soil samples was analyzed as well (two replicates). Pictures of all sampling sites can be seen in Fig. 3. Detailed site descriptions can be found in supplementary table S1.

Soil Samples Collection

Thirty-one samples were collected at six sites on May 14 and 16 2014. Three to six individual soil samples (~25 g) were collected aseptically at several individual sampling spots at each of the six locations, using a small garden shovel and 50-ml Falcon tubes. After evaluation of all results from the 2014 sampling set, additional 11 soil samples were collected in May 2016 from site 6 only. All samples were transported to the laboratory on ice to prevent changes in the microbial communities and were stored at −20 °C before being


A
Kern County
Los Angeles County
138
14
Lancaster
Palmdale
0 1.5 3 6
Miles
B
Kern County
Los Angeles County
138
14
Lancaster
Palmdale
0 1.5 3 6
Miles
Legend

◀ **Fig. 1** Overview of land use in the Antelope Valley in 2001 (**a**) compared to 2011 with indication of renewable energy projects (**b**). The six sampling sites investigated in this study are indicated as *yellow circles*. (Color figure online)

processed the following week. The sampling sites were documented photographically, coordinates were determined, and vegetation cover and visual appearance of all soils in regard to disturbance, erosion, rodent activity, soil moisture, and soil color was described.

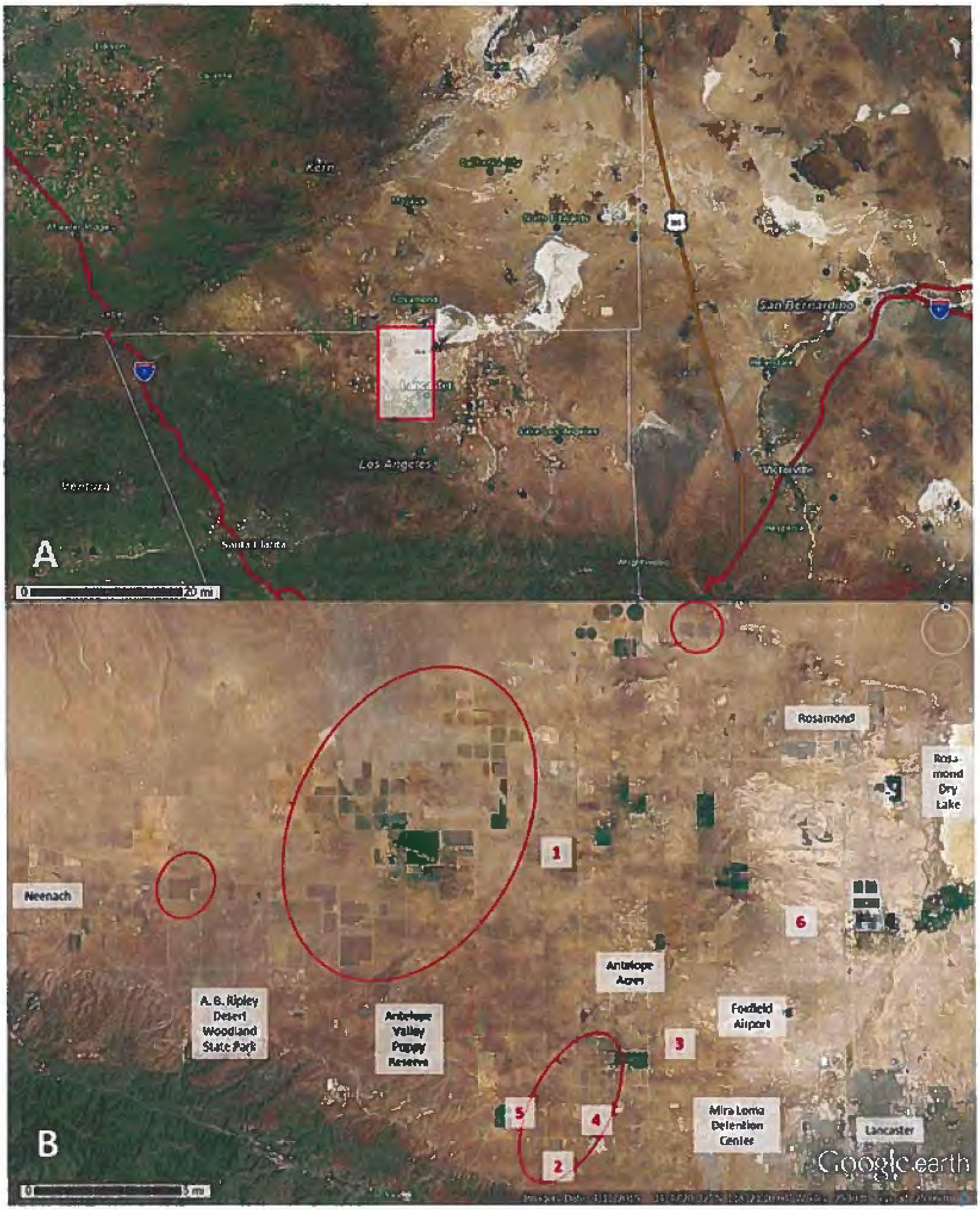
DNA Extraction and PCR

Soil samples were first mixed thoroughly by vortexing until homogenized. Prior to DNA extraction using the Powersoil DNA extraction kit (MoBio, Carlsbad, CA), 0.25 g of each soil sample was transferred into buffer-containing MoBio Powerbead tubes and incubated at 70 °C for 30 min, followed by an incubation step with 100 µl proteinase K (10 mg/ml) at 56 °C for additional 30 min [79] to enhance lysis of microbial spores and conidia. DNA extraction was performed according to the manufacturer's protocol (MoBio, Carlsbad, CA) using a MoBio vortex adapter for the bead-beating process. Two replicates were analyzed for each sample. The amount of DNA was quantified using the Qubit™ 3.0 Fluorometer (Invitrogen Life Technologies, Carlsbad, CA).

To determine the presence of *C. immitis* in all soil samples, a nested PCR approach based on the method published by Baptista-Rosas et al. [7] was used with modifications. A nested PCR can be superior to a one-step PCR method in that it excludes non-target DNA, therefore reducing possibilities of non-specific amplification. As the final diagnostic PCR step, we used 3 different primer pairs: (1) We replaced the originally suggested diagnostic primer pair with the ITSC1Af/ITSC2r primer pair (~220 bp, ITS 2 region) published by Greene et al. [32], which we found superior in specificity for *Coccidioides* spp. than the diagnostic primer pair used in Baptista-Rosas et al. [7] (data not shown), which was originally published by Binnicker et al. [9]. (2) We also used the EC3f/EC100r diagnostic primer set [36, 37] to detect *C. immitis*, which amplifies a ~500-bp amplicon, large enough to distinguish the 2 species within the genus *Coccidioides* and which covers both ITS regions of the ribosomal gene. (3) We also used the diagnostic primer pair ITS1Cf/ITS1Cr which amplifies a ~130 bp region of the ITS1 region of the ribosomal gene, published by Vargas-Gastélum et al. [76]. Overall, three sets of primers were used for each nested PCR approach. Aliquots of all PCR amplicons were analyzed using 2% (wt/vol) agarose gel electrophoresis to determine the correct size of the amplicons using a PCR marker (Promega G3161) (Promega Madison, WI) and ethidium bromide staining (0.5 mg/l). The first primer combination NSA3/NLC2 targets the ribosomal gene (18S and 5.8S DNA and both ITS regions) of all fungi and results in a ~1,100-bp amplicon. Amplicons from the NSA3/NLC2 combination were then used as a template in a nested PCR approach using primer combination NSI1/NLB4 which results in a ~910-bp fragment targeting a fragment of the ribosomal gene of Basidiomycetes and Ascomycetes only (see [7] for details). The final PCR step was the diagnostic PCR using a 1:25 dilution of the amplicons obtained with primer pair NSI1/NLB4 as a template and one of the diagnostic primer sets mentioned earlier in a final PCR. All PCR reactions were performed in duplicate, and the PCR cycling conditions as described in the original protocols were used (see Table 1 for details). PCR reactions contained 12.5 µl of GoTaq Green Mastermix (Promega, Madison, WI), 1.5 µl of each primer (10 pmol/µl), 2 µl of DNA extract or 1.5 µl of the product of a previous PCR reaction for the nested PCRs, as well as sterile water to a final volume of 25 µl. Negative control reactions, which contained all reactants with the exception of template DNA, were also included in all amplifications. These controls were carried through the entire nested PCR process along with the environmental products. Leftover PCR amplicons obtained via diagnostic PCR of approximately correct size (~220, ~500, ~130 bp) were subsequently treated with exoSAP-IT (Affymetrix, Santa Clara, CA), sequenced at the Center for Bioinformatics at the University of Florida, and subsequently compared to entries in the GenBank nucleotide database available at the National Center of Biotechnology Information (NCBI) (http://blast.ncbi.nlm.nih.gov/Blast.cgi) [1]. The sequencing step was necessary because occasionally false-positive amplicons were obtained.

Analysis of pH

Soil pH was determined on a 1:1 (w/v) soil/water mixture composed of 5 g of soil and 5 mL deionized



water. Samples were stirred before and after an equilibration period of 1 h and were then measured with an Oakton-510 bench-top pH meter (Oakton Instruments, Vernon Hills, IL) after calibration to pH buffers 4, 7 and 10. Two replicates were performed for each soil sample and the average was determined.

◀**Fig. 2** **a** Aerial view of the Western Mojave Desert with indication of our sampling area (*red rectangle*) in the Antelope Valley, west of the city of Lancaster (Los Angeles County). **b** Aerial photo of the Antelope Valley as of April 2015 (landsat 8). *Red numbers* indicate all sampling sites (*site 1* North Lancaster Silverado project, *site 2* West Antelope Silverado Project, *site 3* American Silverado Project, *site 4A and B* Antelope Silverado Project, *site 5* Silver Sun Silverado Project, *site 6* Lancaster WAD project). The *red circles* indicate areas where photovoltaic stations were constructed between 2009 and 2015. Construction sites outside these circles were not completed when this study was undertaken. The city of Lancaster is indicated in the *lower right corner* of the photo, south of the Rosamond dry lake bed. Also indicated are the Antelope Valley Poppy Reserve, the Arthur B. Ripley Desert Woodland State Park and the Mira Loma Detention Center. The settlement Antelope Acres is situated between the construction sites west of Foxfield Airport. (Color figure online)

Results

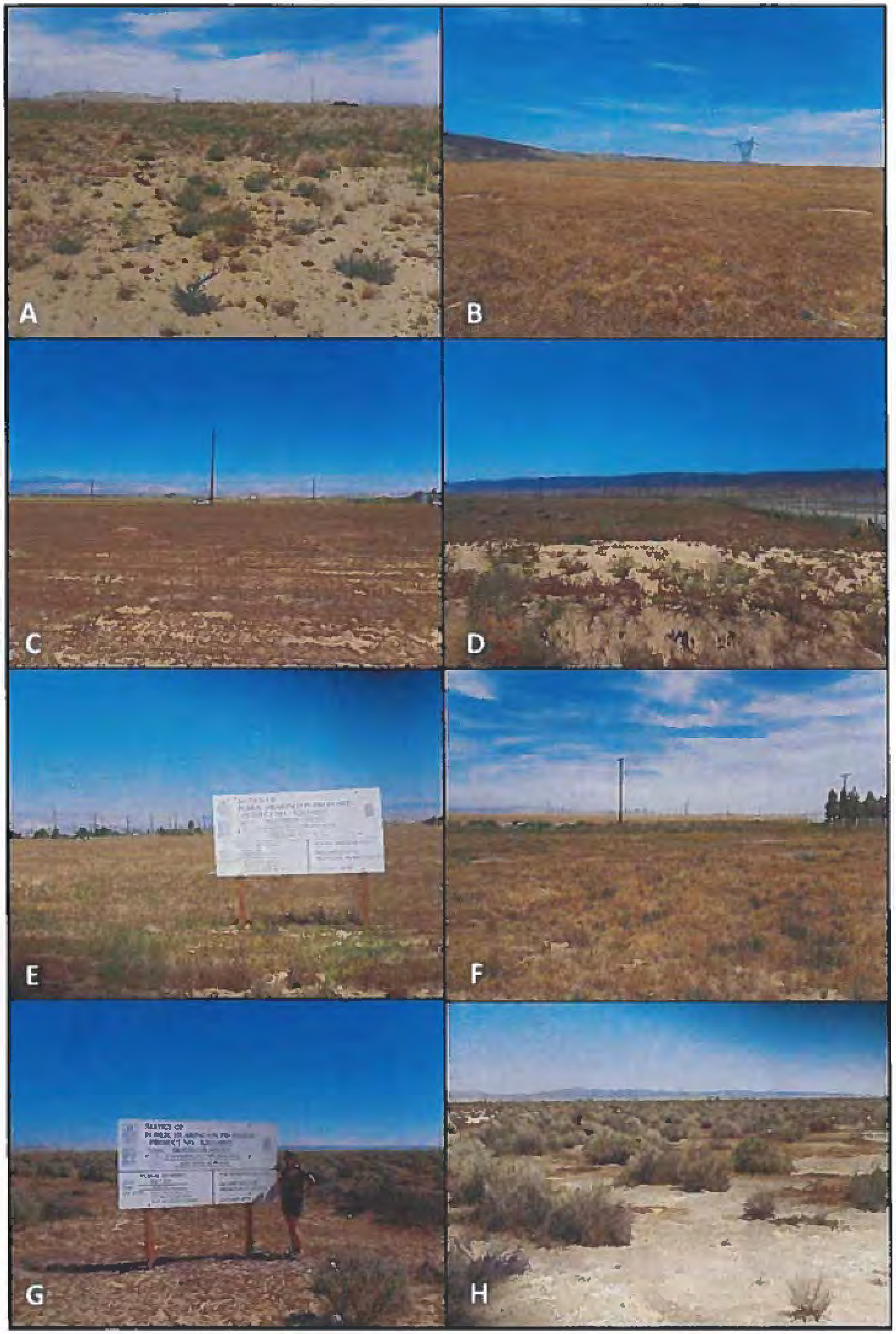
DNA Extraction and PCR

DNA of high quality was successfully extracted from all samples as confirmed by 2% agarose gel electrophoresis and subsequent ethidium bromide staining which resulted in distinct bands of non-sheared DNA. The amount of DNA extracted from 0.25 g of soil varied between soil samples and ranged between 29.2 and 9780 ng/ml. Site 6 had the smallest amount of DNA extracted (29.2–2420 ng/ml), whereas DNA extractions from samples collected at site 4 resulted in the highest amount of extracted DNA (3840–9780 ng/ml) (Table 2).

The nested PCR approach to detect *Coccidioides* spp. confirmed DNA of fungal origin in all soil samples and also confirmed DNA of Ascomycetes and/or Basidiomycetes in 90% of the samples. An example of nested PCR results including all three individual PCR steps with all diagnostic primer pairs is shown in Fig. 4 for a subset of samples. Table 2 summarizes the results of all PCRs and includes the closest matches in the GenBank nucleotide database for all sequenced amplicons. After comparing all sequences to entries in the Genbank nucleotide database, 17 soil samples (40.48%) were found positive for the pathogen with PCR amplicons of 99 or 100% similarity to a *C. immitis* entry in the GenBank nucleotide database (6.7, collected in 2016, showed a faint band of correct size with primer pair ITS1Cf/ITS1Cr which could not be confirmed by sequencing). Two additional soil samples resulted in amplicons that were 89% related to *C. posadasii* (sites 4B3 and 5.4 collected in 2014). Most of the false-positive PCR products were related to fungi in the order Capnodiales (*Cladosporium* spp.). In some occasions multiple species contributed to an amplicon, resulting in a "noisy" sequence that could not be identified. PCR products obtained with diagnostic primer pair EC3f/EC100r resulted more often in false-positive results than PCR products obtained with primer pair ITSC1Af/ITSC2r. Diagnostic primer pair ITS1Cf/ITS1Cr was the most specific of all three primer pairs tested, resulting in no false-positive amplicons. This primer pair was also the most sensitive one, because it detected the pathogen in 23.81% of the samples (28.57% if the two unconfirmed samples are considered as well). Primer pair EC3f/EC100r detected the pathogen in 11.9% of the samples, while primer pair ITSC1Af/ITSC2r detected *C. immitis* in 19.05% of the samples. Samples 6.2 collected in 2016 was the only sample that tested positive for the pathogen with all three diagnostic primer pairs. Five samples collected in 2014 and one sample collected in 2016 were indicated positive with two out of the three diagnostic primer pairs. Individual sampling spots where the pathogen was detected are shown in supplementary figure S2. Examples of high-quality sequences obtained with all 3 diagnostic primer pairs were deposited in the GenBank nucleotide database available at the National Center for Bioinformatics and Information (NCBI) (Accession No. KY306689-KY306699).

Characterization of Soil Samples

Variation in soil characteristics was observed for all sampling sites (USGS Soil Survey Antelope Valley, www.usdawebsoilsurveydatabase; Table 3; and supplementary figure S3). Soils in the sampling area varied in soil parent material, and in regard to chemical and physical parameters, as indicated by different USDA soil map units. Overall, the soil types that were the most common in our sampling area were characterized as Hesperia fine sandy loam (~10% of the sampling area), Greenfield sandy loam (~18%), Cajon loamy sand (~5.5%), Pond loam (5%), Rosamond fine sandy loam (~4%), Sunrise sandy loam (~6.5%); several others each covered <4% of the study area. Soils belonging to the Pond-Oban complex covered a large area of the valley around Rosamond and Roger's

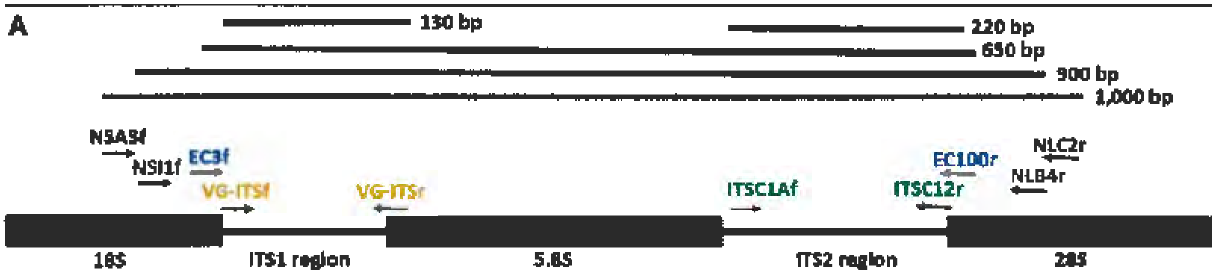

A
B
C
D
E
F
G
H

◂**Fig. 3** Landscape overview of all sampling sites at the time of sampling (May 2014). **a** Site 1, a disturbed site with scattered native and non native vegetation. **b** Site 2, grassland with native and non-native annuals. **c** Site 3, disturbed land with scattered native and non-native vegetation. Surrounding areas grew rabbit brush (*Ericameria nauseosa*) (as can be seen in the *background*). **d** Site 4A, a disturbed area with scattered native and non-native vegetation. **e** Site 4B, grassland with native and non native annuals. **f** Site 5, disturbed site with grasses and other non-native and native species. **g** and **h** Site 6, dominated by scattered salt bushes and occasional rabbit brush. Dried *Lastenia californica* can be seen in between the salt bushes (*Atriplex* spp.)

dry lake bed and comprised ~15% of the eastern study area where sampling site 6 was located. Soil pH generally increased with proximity to the Rosamond dry lake bed and ranged between pH 5 and 9.4. The pH varied considerably for subsamples from sites 2, 4A and 6, but were more uniform among samples from sites 1, 3, 4B and 5. Furthermore, average soil pH results observed in our laboratory differed from the averaged values indicated in the USDA websoilsurvey database. For example, fine sandy loam samples from sites 1 and 6 appeared less alkaline when analyzed in our laboratory. Samples positive for *C. immitis* also varied in pH, but the majority of the positive soil samples showed a pH higher than 7 and less DNA could be retrieved compared to samples with a lower pH (Fig. 5; supplementary figure S4). Soil parameters that were indicative of the presence of *C. immitis* in previous research in the Southern San Joaquin Valley (Kern County) [44, 45] predicted site 6 in the Antelope Valley as a potential growth site of the pathogen (Table 3). However, sites 2, 3 and 5 where the pathogen was detected as well were not indicated as potential growth sites based on soil parameters. Other soil types near our sampling sites, such as Rosamond loam and Tray loam, share some of the parameters that were indicative of the presence of the pathogen in the San Joaquin Valley, but these soils were not

Table 1 Position of primers on ribosomal gene (A), all primer pairs used for nested PCR reactions with PCR amplification conditions and references (B)

A



B

Primer	Sequence	Tm	Cycling conditions[a]: Annealing temperature (°C)	Temperature (°C)	Time (s)	Number of cycles	Expected product size	Reference
NSA3f	5′-AAACTCTGTCGTGCTGGGGATA-3′	58.1	55	95, 55, 72	30, 40, 40	35	~1100	Martin [50]
NLC2r	5′-GAGCTGCATTCCCAAACAACTC-3′	56.8	55				~1100	Martin [50]
NSI1f	5′-GATTGAATGGCTTAGTGAGG-3′	50.8	60	95, 60, 72	30, 40, 40	35	~910	Martin [50]
NLB4r	5′-GGATTCTCACCCTCTATGAC-3′	51.7	60				~910	Martin [50]
ITSC1Af	5′-CATCATAGCAAAAATCAAA-3′	45.3	53	94, 53, 72	60, 60, 60	45	~220	Greene et al. [32]
ITSC2r	5′-AGGCCCGTCCACACAAG-3′	58	53				~220	Greene et al. [32]
EC3f	5′-ATTAAAGTGGCGTCCGGCTG-3′	58.1	57	95, 57, 72	30, 40, 40	45	~500	Johnson et al. [36,37]
EC100r	5′-CGATGAAGTGATTTCCCATACA-3′	52.3	57				~500	Johnson et al. [36,37]
VG-ITSf	5′-GTGGCGTCCGGCTGCGCACCTCCCCCGCGG-3′	78.1	70	95, 70, 72	30, 40, 60	45	~130	Vargas-Gastélum et al. [76]
VG-ITSr	5′-GCGCAAGGCGGGCGATCCCCGGCAGCC-3′	76.2	70				~130	Vargas-Gastélum et al. [76]

f Forward primer, *r* reverse primer

[a] All samples were subjected to an initial melting step of 94 or 95 °C for 10 min and a final extension step of 72 °C for 10 min

Table 2 Results of nested PCRs with indication of closest matches in the GenBank nucleotide database for amplicons obtained with the diagnostic primer pairs ITSC1Af/ITSC2r, EC3f/EC100r and ITS1Cf/ITS1Cr

Sample ID	NSA3/ NLC2	NSI1/NLB4	ITSC1Af/ITSC2r		EC3f/EC100r		ITS1Cf/ITS1Cr		DNA extracted	
2014	All fungi	*Ascomycetes/ Basidiomycetes*	*Coccidioides* spp.	% Similarity to closest match in GenBank	*Coccidioides* spp.	% Similarity to closest match in GenBank	*Coccidioides* spp.	% Similarity to closest match in GenBank	(ng/ml)	pH
1.1	Positive	Positive	Negative		False positive	unc. *Capnodiales*, JF691038, 99%	Negative		2800	6.90
1.2	Positive	Positive	Negative		False positive	unc. *Capnodiales*, JF691038, 94%	Negative		2640	7.13
1.3	Positive	Positive	Negative		False positive	unc. *Capnodiales*, JF691038, 95%	Negative		1960	7.13
2A.1	Positive	Positive	**Positive**	***Coccidioides immitis*, HG380500, 100%**	False positive	unc. *Capnodiales*, JF691038, 93%	**Positive**	***Coccidioides immitis*, KM679413, 100%**	3320	5.95
2.2	Positive	Positive	Negative		False positive	unc. *Capnodiales*, JF691038, 94%	Negative		4360	7.28
2.3	Negative	Negative	Negative		Negative		Negative		7280	6.90
2.4	Positive	Positive	Negative		False positive	unc. *Capnodiales*, JF691038, 90%	Negative		1406	5.24
2.5	Positive	Positive	Negative		False positive	unc. *Capnodiales*, JF691038, 93%	Negative		1858	6.29
2.6	Positive	Positive	**Positive**	***Coccidioides immitis*, HG380500, 100%**	Negative		**Positive**	***Coccidioides immitis*, KM679413, 100%**	4700	7.12
3.1	Positive	Positive	**Positive**	***Coccidioides immitis*, HG380500, 99%**	False positive	unc. *Capnodiales*, JF691038, 95%	**Positive**	***Coccidioides immitis*, KM679413, 100%**	3520	6.73
3.2	Positive	Negative	Negative		?*	Multiple sequences***	Negative		3480	6.76
3.3	Positive	Positive	Negative		?	Multiple sequences	Negative		2500	6.03
3.4	Positive	Positive	False positive	unc. *Eurotiales*, HQ389458, 95%	False positive	unc. *Capnodiales*, JF691038, 99%	Negative		6620	6.79
3.5	Positive	Positive	**Positive**	***Coccidioides immitis*, KJ783449, 100%**	**Positive**	***Coccidioides immitis*, KJ783449, 100%**	Negative		4400	7.11
4A.1	Positive	Positive	False positive	unc. *Eurotiales*, HQ389458, 96%	False positive	unc. *Capnodiales*, JF691038, 94%	Negative		3840	6.81
4A.2	Positive	Positive	Smear**		False positive	unc. *Capnodiales*, JF691038, 96%	Negative		9780	6.50
4A.3	Positive	Positive	Negative		False positive	*Ascochyta* sp., KC959210, 85%	Negative		6980	5.59
4B.1	Positive	Negative	Smear		False positive	unc. *Capnodiales*, JF691038, 95%	Negative		4520	7.38
4B.2	Positive	Positive	False positive		False positive		Negative		5900	7.55

Table 2 continued

Sample ID	NSA3/NLC2	NSI1/NLB4	ITSC1Af/ITSC2r		EC3f/EC100r		ITS1Cf/ITS1Cr		DNA extracted	
2014	All fungi	*Ascomycetes/ Basidiomycetes*	*Coccidioides* spp.	% Similarity to closest match in GenBank	*Coccidioides* spp.	% Similarity to closest match in GenBank	*Coccidioides* spp.	% Similarity to closest match in GenBank	(ng/ml)	pH
				Trichophyton terrestre, LN714614, 96%		unc. *Capnodiales*, JF691038, 94%				
4B.3	Positive	Positive	False positive	*Coccidioides posadasii*, KF386150, 89%	False positive	unc. *Capnodiales*, JF691038, 94%	Negative		8020	7.36
5.1	Positive	Negative	Negative		False positive	unc. *Capnodiales*, JF691038, 94%	Negative		7040	6.67
5.2	Positive	Positive	False positive	*Cladosporium macrocarpum*, KC311478, 95%	False positive	unc. *Capnodiales*, JF691038, 92%	**Positive**	***Coccidioides immitis*, KM679413, 100%**	3760	6.44
5.4	Positive	Positive	False positive	no similarity found	False positive	*Coccidioides posadasii*, JX051631, 89%	Negative		8040	6.69
5.5	Positive	Positive	Negative		False positive	Multiple sequences	Negative		3280	6.75
5.6	Positive	Positive	Negative		False positive	unc. *Capnodiales*, JF691038, 93%	Negative		7120	6.50
6.1	Positive	Positive	?	multiple sequences	**Positive**	***Coccidioides immitis*, KJ783449, 99%**	Negative		1796	7.44
6.2	Positive	Positive	**Positive**	***Coccidioides immitis*, KJ783449, 99%**	Negative		Negative		1270	8.15
6.3	Positive	Positive	**Positive**	***Coccidioides immitis*, KJ783449, 99%**	Negative		Negative		2420	7.04
6.4	Positive	Positive	Smear		Negative		Negative		2080	7.00
6.5	Positive	Positive	**Positive**	***Coccidioides immitis*, KJ783449, 99%**	**Positive**	***Coccidioides immitis*, KJ783449, 99%**	Negative		174	9.45
6.6	Positive	Positive	Negative		Negative		**Positive**	***Coccidioides immitis*, KM679413, 100%**	29.2	8.79
2016										
6.1	Positive	Positive	Negative		**Positive**	No signal (faint PCR product)	**Positive**	***Coccidioides immitis*, KM679413, 98%**	120	8.12
6.2	Positive	Positive	**Positive**	***Coccidioides immitis*, KJ783449, 99%**	**Positive**	***Coccidioides immitis*, KJ783449, 94%**	**Positive**	***Coccidioides immitis*, KM679413, 100%**	85.8	7.38
6.3	Positive	Positive	Negative		**Positive**	***Coccidioides immitis*, KJ783449, 99%**	**Positive**	No signal, faint PCR product	53	6.91
6.4	Positive	Positive	Negative		False positive	Fungal endophyte, KT291114, 94%	**Positive**	***Coccidioides immitis*, KM679413, 100%**	1014	7.94
6.5	Positive	Positive	Negative		Negative		Negative		324	8.25
6.6	Positive	Positive	Negative		Negative		Negative		55.6	9.53

Table 2 continued

Sample ID	NSA3/NLC2	NSI1/NLB4	ITSC1Af/ITSC2r		EC3f/EC100r		ITS1Cf/ITS1Cr		DNA extracted	
2014	All fungi	*Ascomycetes/ Basidiomycetes*	*Coccidioides* spp.	% Similarity to closest match in GenBank	*Coccidioides* spp.	% Similarity to closest match in GenBank	*Coccidioides* spp.	% Similarity to closest match in GenBank	(ng/ml)	pH
6.7	Positive	Positive	Negative		False positive	Fungal endophyte, KT291114, 96%	Positive	No signal, faint PCR product	964	7.85
6.8	Positive	Positive	Negative		Negative		Negative		210	6.84
6.9	Positive	Positive	Negative		False positive	No similarity found	Negative		458	7.55
6.1	Positive	Positive	Negative		False positive	No similarity found	Positive	*Coccidioides immitis*, KM679413, 100%	244	9.21
6.11	Positive	Positive	Negative		Negative		Positive	*Coccidioides immitis*, KM679413, 100%	766	7.31

Positive samples and their closest match in the GenBank database are presented in bold

investigated in this study (soils where the pathogen was detected are indicated as positive [bold]).

Environmental Parameters and Incidence of Coccidioidomycosis

Environmental parameters, such as fugitive dust emission (PM10), total annual precipitation (inches), and wind speed (gust max.), were obtained for the Mojave Air Basin for the years 2000–2015. In addition, we obtained land-use data (acres) [15] and coccidioidomycosis incidence data [14] for the same time period and area (Fig. 6). An increase in incidence of coccidioidomycosis over time can be seen with highest incidence in the Antelope Valley in 2005, 2011 and 2014, spiking shortly after years with increased soil disturbance due to the "housing boom" between 2003 and 2007 [33] and the renewable energy boom described in this study. Between 2005 and 2014, the number of approved permits for solar farms and wind parks increased with additional large-scale and small-scale projects pending permission. So far, more than 20,000 acres of land have been disturbed as of 2014 for renewable energy projects in the Antelope Valley and the surrounding foothills of the Tehachapi and San Bernardino Mountains [30, 31]. The acreage of field crops increased by 48% compared to the year 2000 and then steadily declined by 2014 reaching values close to those documented before 2008 (County of Los Angeles Crop and Livestock Report 2014). The correlation between incidence of coccidioidomycosis in the Antelope Valley and the amount of acres of land disturbed for renewable energy projects and amount of acres under agricultural management (field crops) between 2000 and 2014 was strong, as revealed by a correlation coefficient of $r^2 = 0.623$ (Pearson product-moment correlation coefficient) and $r^2 = 0.388$. The correlation between PM10 (Mojave Air Basin) and disease incidence was at best weak with a Pearson coefficient of 0.283 and an r^2 value of 0.0664 (see Fig. 6 for all correlation values). However, the correlation between PM10 and incidence of the disease was strong when only the years between 2009 and 2014 were considered (renewable energy boom), with a Pearson coefficient of 0.641 and an r^2 value of 0.411. To investigate these relationships in more detail, a multiple regression analysis was conducted (Table 4). This analysis shows that neither PM10 nor levels of precipitation appear to have had a

Fig. 4 Displayed are 2% agarose gels after ethidium bromide staining, showing examples of diagnostic PCR results for samples from some locations. Sequences from PCR amplicons circled in black were 99% related to a GenBank database entries of *C. immitis*. *White arrows* point toward amplicons of correct size including some that were revealed as false positives. **a** Results of amplification with primer pair ITSC1Af/ITSC2r showing amplicons of ~220. **b** Results of amplification with primer pair EC3f/EC100r showing amplicons of ~500 bp. **c** Results of amplification with diagnostic primer pair ITS1Cf/ITS1Cr showing positive results for samples 5.2 and 6.6 (2014). (*C.i. Coccidioides immitis* used as positive control; *NC* negative control)

significant relationship with coccidioidomycosis incidence (2000–2014). However, the total acres of land under the three land-use types considered (wind, solar and agricultural) did have a significant, positive relationship with coccidioidomycosis incidence ($p < 0.01$) over the same time period. We investigated the relationship with land use further by conducting a second multiple regression, removing the non-significant factors and disaggregating the three land-use types, to determine whether effects could be attributed to specific type of lands use. This analysis revealed no significant differences between the effects of the different land-use categories (Table 5).

Discussion

A correlation between soil disturbances due to large-scale renewable energy construction projects, agricultural management practices and PM10 fugitive dust emission with increased incidence of coccidioidomycosis was clearly indicated by results of this study. The increasing incidence of coccidioidomycosis in the Antelope Valley of California, which has reached epidemic character, is concerning and shown in supplementary figure S1. The *C. immitis* positive sites detected in this study are located west of the cities of Lancaster and Palmdale and south of the community of Antelope Acres which are part of what is known as the Greater Antelope Valley Economic Alliance (GAVEA) which has experienced a population increase of 24% between 2000 and 2010 (US Census Bureau). It has been predicted that the population will continue to grow another ~46% by 2035, to 758,881 residents [31]. The predicted population growth will result in continued urbanization as of yet unknown proportions, but certainly of significant size. Therefore, it is expected that fugitive dust emissions from ongoing construction sites will continue or even increase. This environmental health hazard will put humans and animals at an increased risk for contracting coccidioidomycosis, especially if dust mitigation continues to be inefficient or absent. In addition to increased urbanization and renewable energy development in this area, an ongoing drought with decreasing precipitation and sinking ground water tables has been blamed for soil erosion and fugitive dust development in the Antelope Valley. The ongoing drought has also resulted in a significant reduction in farming activities over the last years, resulting in large areas of abandoned fields. For example, the farmed acreage of orchards decreased from 2013 to 2014 by 53.06%, and the farmed acreage for grapes decreased by 22.6% during the same time in the County [48].

It has been difficult in the past to determine a clear correlation between incidence of coccidioidomycosis and certain environmental parameters, because of combined immediate or delayed positive or negative effects on the growth of the pathogen in the soil. Previous work by Talamantes et al. [72] determined a weak correlation between precipitation and wind speed and coccidioidomycosis incidence in Kern County. Smith et al. [69] and Kirkland and Fierer [40] had already pointed out that a rainy winter can cause an increase in coccidioidomycosis incidence in the following dry season, especially after a prolonged drought that might have caused a shift in the microbial community toward spore and conidia formers, among

Table 3 Averaged soil physical and chemical parameters for dominant soil types found in our sampling area with indication of soil map unit symbols, as obtained from the USDA websoilsurvey database (pH was also analyzed at CSUB). (Color figure online)

Soil parameters (averaged data)	Sampling sites				
Sites	1*	1, 3*	2*, 4A*, 4B*, 5	2, 4, 5*	6*
Map unit name	Rosamond fine Sandy loam	Hesperia fine Sandy loam	Greenfield Sandy loam	Ramona coarse Sandy loam	Pond Oban complex
Landform	Alluvial fans	Alluvial fans	terraces, alluvial fans	terraces	Basin floors
Parent material	Alluvium derived from granite	Alluvium derived from granite	Alluvium derived from granite	Alluvium derived from granite	Alluvium derived from granite
Map unit symbols	Ro	HkA	GsA/GsC	RcB/RcC	Px
Physical parameters					
Surface texture	Fine sandy loam	Sandy loam	Sandy loam	Coarse sandy loam	Fine sandy loam
Clay (%)	18.8	13	11	7.5	25.7
Sand (%)	51.9	70.5	66	69.8	46.5
Silt (%)	29.3	16.5	23	22.9	27.8
Available water capacity (cm/cm)	0.14	0.13	0.13	0.1	0.08
Available water supply (0–25 cm)	3.45	2.5	3.25	2.5	2.04
Organic matter (%)	0.17	0.08	0.75	0.75	0.25
Water content (15 bar) (%)	11.1	8.1	7.4	6	15.8
Water content (1/3 bar) (%))	23.3	17.2	16.4	14.7	27.5
Sat. hydraulic conductivity (Ksat) (µm/s)	21.67	28	28	28	4.8
Chemical parameters					
pH (websoilsurvey database)	8	7.4	6.7	6.7	8.9
pH (determined at CSUB)	7.1 (site 1)	6.7 (site 3)	6.4 (site 2), 6.1/6.6 (site 4A/4B)	6.6	8
$CaCO_3$	5	2	0	0	8
Cation exchange capacity (CEC7) (milliequivalents/100 g)	10	7.5	7.5	7.5	15.8
Gypsum	0	0	0	0	0
Sodium adsorption ratio (SAR)	0	0	0	0	12.7
Electrical conductivity (EC) (decisiemens/m)	1.7	0.7	0	0	14
Wind erodibility index (tons per acre per year)	86	86	86	86	86
Detection of *C. immitis*	Negative	Positive	Positive	Positive	Positive

Soil parameters	Other dominant soils in the sampling area					
Sites	1	3	3	6	2	4
Map unit name	Rosamond loam	Cajon loamy sand	Tray sandy loam, Saline-alkali	Tray Loam	Hanford Coarse sandy loam	Sunrise sandy loam
Landform	Alluvial fans	Alluvial fans	Basin floors	Basin floors	Alluvial fans	basin floors
parent material	Alluvium derived from granite	Alluvium derived from granite	Alluvium derived from granite	Alluvium derived from granite	Alluvium derived from granite	alluvium derived from granite
Map unit symbols	Rp	CaA	Tv	Tw	HbC	Sv
Physical parameters						
Surface texture	Loam	Loamy sand	Sandy loam	Loam	Sandy loam	Sandy loam
Clay (%)	20.5	3.7	12.7	20	12.5	15
Sand (%)	34.7	83.1	65.8	42.1	68.2	65.9
Silt (%)	44.8	13.2	21.6	37.9	19.3	19.1
Available water capacity (cm/cm)	0.16	0.08	0.1	0.15	0.13	0.12
Available water supply (0–25 cm)	3.85	1.98	2.5	3.5	3.25	3
Organic matter (%)	0.17	0.57	0.58	0.75	0.58	0.25
Water content (15 bar) (%)	12.3	3.3	8.4	12.5	8.9	9.2
Water content (1/3 bar) (%))	27.7	10.9	17.7	27.1	17.8	18.3
Sat. hydraulic conductivity (Ksat) (µm/s)	9	92	21.7	9	28	28
Chemical parameters						
pH (websoilsurvey database)	8.2	7.2	9.1	9	6.7	7.9
pH (determined at CSUB)	Not determined					
$CaCO_3$	5	1	3	3	0	8
Cation Exchange Capacity (CEC7) (milliequivalents/100 g)	10	3	7.5	7.5	7.5	7.5
Gypsum	0	0	0	0	0	0
Sodium adsorption ratio (SAR)	0	0	3	3	0	0
Electrical conductivity (EC) (decisiemens/m)	1.3	0.2	5	5	0	1
Wind erodibility index (tons per acre per year)	56	134	86	48	86	86
Detection of *C. immitis*	Not investigated					

Soil parameters that were indicative of the presence of the pathogen in the Southern San Joaquin valley [44, 45] are indicated in red. Additionally, results from our PCR-based approach to detect *C. immitis* are included (at some sampling sites, more than one soil type was detected; therefore, the soil type of the soil samples analyzed is indicated with an *; soil types where the pathogen was detected are indicated with a red rectangle)

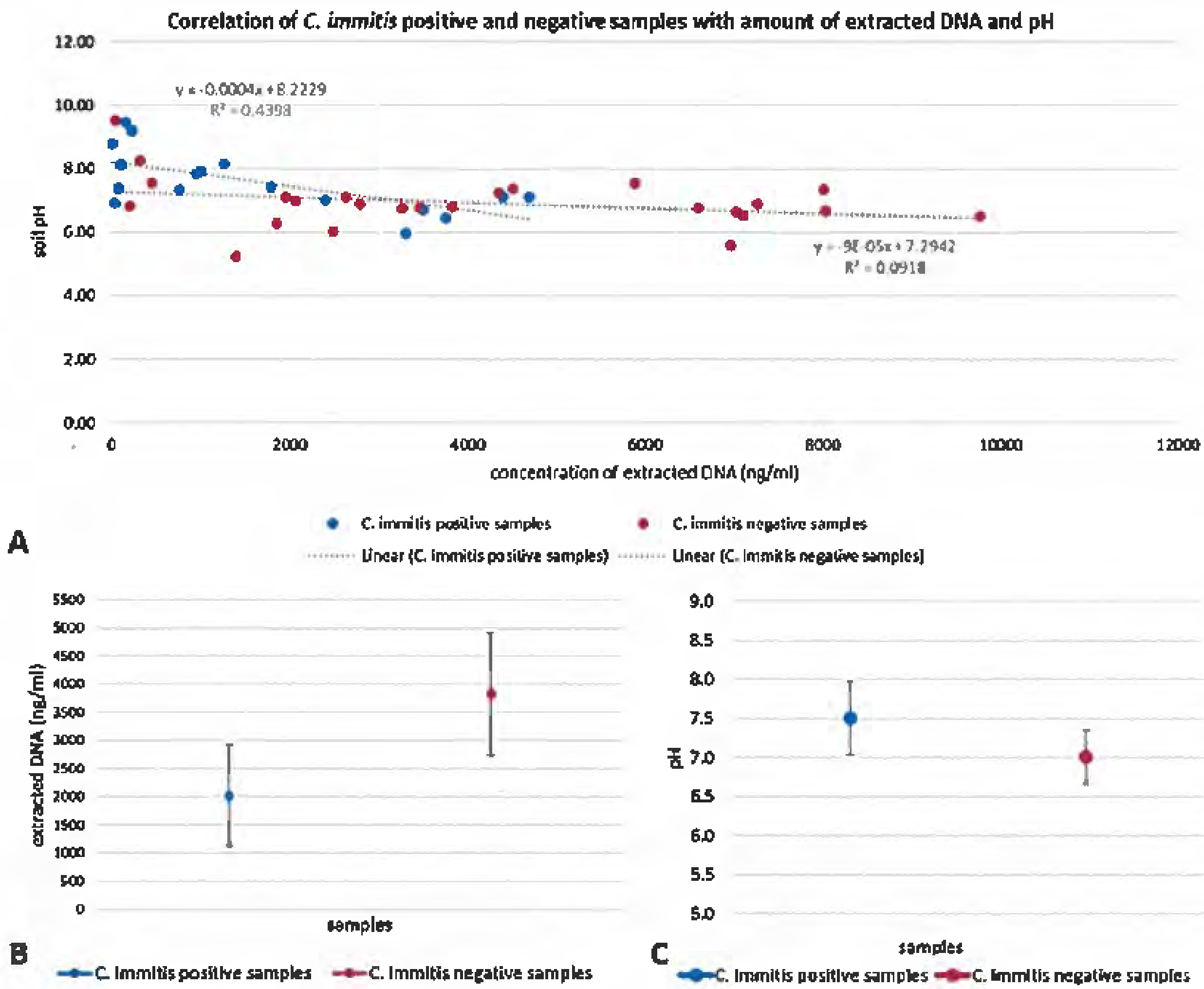


Fig. 5 **a** Correlation between soil pH and amount of extracted DNA. **b** The amount of extracted DNA from *C. immitis* positive soil samples was significantly lower than the amount of DNA extracted from *C. immitis* negative soil samples. **c** The pH of soils in which the pathogen was detected was higher than the pH of soils that were negative for the pathogen. However, the difference was not significant (data were normally distributed based on the Shapiro-Wilkes test for normality of the residuals). (Color figure online)

them *Coccidioides* spp. In our study, we were able to clearly link land use and soil disturbance to valley fever incidence, but also found a positive correlation between PM10 and wind speed; however, the correlation was rather weak. The continued increase in coccidioidomycosis incidence in 2012 and 2013 when construction of new renewable energy projects slowed down was likely due to the long-term effect of large areas of graded soils, which continue to be a major source of fugitive dust emission in the Antelope Valley and beyond. In the past, California had been plagued with long-term and short-term droughts, for example the prolonged drought from 1985 to 1992 which resulted in increased fugitive dust emissions that reached a 24-h record PM10 concentration of 780 μg/m^3 in downtown Lancaster in 1991 (Antelope Valley Air Quality Monitoring District).

We were able to detect the pathogen *C. immitis* predominantly in undisturbed alkaline soils of the Pond-Oban complex, located closest to the Rosamond dry lake bed, a location commonly referred to as "barren land" with different species of salt bushes, that indicate a saline and alkaline environment. Site 6 was the only sampling site that was suspected to harbor *C. immitis* based on averaged soil parameter information (USDA websoilsurvey database) that

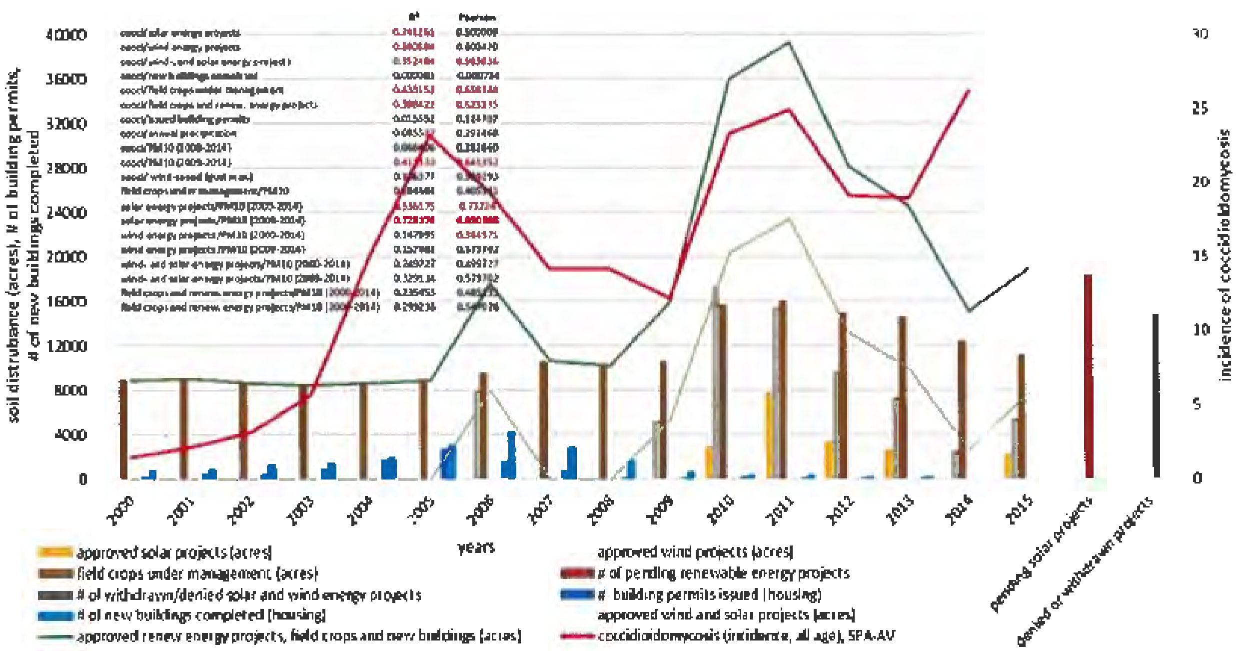


Fig. 6 Increase in soil disturbance over time in the Antelope Valley due to large-scale renewable energy project construction, changes in agricultural management correlated with incidence of coccidioidomycosis. Displayed are approved solar and wind projects (acres), agricultural fields under management (acres), as well as the number of issued building permits and completed buildings (housing developments) between 2000 and 2014. Included in the calculations (but not included in the graph) is also PM10 data (High National 24 Hour Average), precipitation (inches), and wind-speed data (gust max.) for the city of Lancaster measured at Foxfield Airport (KWJF). Solar and wind farms were graphed one year after the permit approval date because the begin of construction generally began in the year after permits were issued (data sources: www.arb.ca.gov/adam, http://publichealth.lacounty.gov/acd/Publications.htm, http://planning.lacounty.gov/assets/upl/project/energy_projects.pdf, http://pcd.kerndsa.com/planning/renewable-energy, http://lacfb.org/crop-reports-2/)

Table 4 Results of the initial multiple regression model: the model was coccidioidomycosis incidence = PM10 + precipitation +acres of land use (summed across Solar projects, wind projects, and active agricultural use

	Estimate	SE	t	p
(Intercept)	4.90	3.06	1.61	0.13
PM10	-6.14×10^{-3}	1.06×10^{-2}	−0.580	0.57
Annual Precipitation (in)	1.34	1.08	1.24	0.24
Acres of land use	4.04×10^{-4}	1.20×10^{-4}	3.36	0.0057**

Final model is: coccidiomycosis incidence = 4.9 − 0.0062 × PM10 + 1.33 precipitation + 0.00040 land use. $F_{3,12} = 4.8$, $p < 0.05$, multiple $r^2 = 0.54$

were indicative of the presence of the pathogen in the Southern San Joaquin Valley [44, 45]. The Southern San Joaquin Valley Desert is geologically somewhat related to the Western Mojave Desert where the Antelope Valley is located, but differs in elevation and climate [24]. Soils of both locations developed from quaternary alluvium and similar underlying parent material and have been described as alluvial fans or fan remnants and basin floors, with high concentrations of fine particulate matter that accumulated since the late Pleistocene and earlier. However, the pathogen was also detected in grassland from soils characterized as Greenfield sandy loam, Hesperia fine sandy loam, and Ramona fine sandy loam (sites 2, 3 and 5). The grassland appeared similar to a strong growth site of the pathogen, Sharktooth Hill near Oildale, east of

Table 5 Results of the 2nd multiple regression model: the model was coccidioidomycosis incidence = acres of solar projects + acres of wind projects + acres active agricultural use

	Estimate	SE	t	p
(Intercept)	−6.02	15.8	−0.38	0.711
Solar	8.86×10^{05}	1.72×10^{03}	0.052	0.96
Wind	5.16×10^{05}	7.34×10^{04}	0.070	0.945
Crop	1.78×10^{03}	1.66×10^{03}	1.07	0.306

$F_{3,12} = 2.3$, $p > 0.1$, multiple $r^2 = 0.36$

Bakersfield, not far away from a severely disturbed area, the Kern River oilfields, but the physical and chemical parameters of soils from Sharktooth Hill (high clay content) were more similar to those determined at site 6. Fossil diggers repeatedly became infected with *C. immitis* at Sharktooth Hill, where the presence of the pathogen has been confirmed repeatedly [20, 45, 64, 70]. Overall, soils from all *C. immitis* positive sites in the Antelope Valley and the Bakersfield area can be characterized as fine particulate sandy loam. The investigation of other soil types should be included in future studies to refine the set of environmental parameters that are indicative of the presence of the pathogen and to deepen our understanding of the ecology of *C. immitis* in California. The diversity of habitats that *C. immitis* can apparently grow in indicates that the pathogen is able to adapt to somewhat different soil environments or that different ecotypes of the pathogen exist which might explain its "spotty distribution" [6, 20]. Furthermore, it should be noted that site 6 where the pathogen was detected repeatedly had the lowest amounts of extracted DNA. A fungal species such as *Coccidioides* spp. which is missing some key enzymes needed to grow successfully as a saprophyte in soil might benefit from a low diversity of overall soil microbes that could include potential competitors and antagonists [65].

It has been difficult and expensive to detect *Coccidioides* spp. in soil and dust samples in the past [8, 22], but modern culture-independent molecular methods became available in recent years which allow for successful screening of environmental samples for the presence of *C. immitis* and *C. posadasii* [7, 36, 37, 42, 44–46, 66, 76]. However, soil analyses for the detection of soil-borne pathogens, such as *Coccidioides* spp., have not been included in Environmental Impact Reports (EIRs) for any construction project planned in the Antelope Valley or in other endemic areas of the pathogen in the Southwestern US. The scarcity of experts who are familiar with the procedures to detect the pathogen in its natural environment, additional costs of soil analyses, and a general underestimation of the risk of otherwise healthy people of contracting coccidioidomycosis from dust exposure might explain this potentially risky situation.

Mitigation and regulatory efforts are likely to be inadequate if the current trends in development and associated fugitive dust emissions continue. During spring 2014, fugitive dust emissions in the Antelope Valley negatively impacted air quality to an extent never documented before, reaching values up to and above 1000 μg/m^3, which reminded people of the Great Dust Bowl of the 1930's in Oklahoma [47], or the extreme dust storms documented in Owens Valley after the 110 mi^2 Owens Lake had been dried to support the water thirsty city of Los Angeles for a little more than a decade (1913 1926, feeding the Los Angeles aqueduct, see [60, 67]). Wilken et al. [78] indicated the inability of current dust mitigation strategies to protect construction workers from infections with *Coccidioides* spp. Lack of regulatory expertise and unrealistic expectations have resulted in costly failures of dust mitigation methods in the Western Mojave Desert in the past as described in McRae [52]. Environmental Impact Reports (EIRs) have been particularly criticized for not describing how dust mitigation measures are implemented and supervised, and no long-term dust control mitigation measures are included in the reports [73].

Mitigation and regulations are important considerations because some of the construction projects are in the immediate neighborhood of schools or close to human settlements. For example, the Del Sur Solar Project [Conditional Use Permit (Nos. 14-15 and 14-16)] is located adjacent to and upwind of Del Sur Elementary School. As of October 2012, the enrollment consisted of approximately 750 students in grades K-8 who would be directly and constantly affected by fugitive dust emissions because of daily westerly winds.

Although rarely implemented, potential mitigation procedures have been developed. Re-vegetation of disturbed land as a long-term strategy of dust control has been suggested by the Antelope Valley Dustbusters Taskforce, a group which consists of private entities, as well as federal, city, and county government representatives [3, 29, 41], but the implementation of their recommendations into Dust Control Plans (DCPs) rarely occurred [10]. Based on 20 years of dust mitigation experience in the Antelope Valley, The Dustbusters Task Force of 1991 developed handbooks for farmer and homeowners in the Antelope Valley which are publicly available at (http://www.kernair.org/Main_Pages/information.html#; see also [18, 29, 61, 68]). Based on the findings of this study, we recommend that EIRs include soil analyses for *Coccidioides* spp. on land destined for construction of any type in endemic areas of the pathogen.

Conclusion

Although the change from non-renewable to renewable energy is generally welcomed in California, disturbing soils in endemic areas of a soil-borne pathogen that already causes disease incidence of epidemic character should only be considered if successful long-term dust mitigation measures are implemented, supervised, and controlled. The increasing demand for renewable energy shows promise for our planet in the future and will reduce some airborne emissions. However, there are hazards when sourcing new locations. One such danger is *Coccidioides* spp. arthroconidia becoming airborne when soil is disturbed and dust mitigation measures are inefficient or absent.

Acknowledgements The project was supported by the California State University Research Council for Undergraduates Program (RCU) (D10720). The authors also like to thank Paulina Le for analyzing the soil pH.

Compliance with Ethical Standards

Conflict of interest The authors declare that they have no conflict of interest.

Informed consent Informed consent was obtained from all individual participants included in the study.

References

1. Altschul SF, Gish W, Miller W, Myers EW, Lipman DG. Basic local alignment search tool. J Mol Biol. 1990;215:403–10.
2. Anderson JO, Thundiyil JG, Stolbach A. Clearing the air: a review of the effects of particulate matter air pollution on human health. J Med Toxicol. 2012;8:166–75.
3. Antelope Valley Dustbusters. Large area land managers guide to controlling windblown sand and dust. Antelope Valley Dustbusters Publication, pp. 1–29. http://www.kernair.org/Documents/Dust_Buster/Dustbusters%20Agricultural%20Guide%2010-25-10.pdf (2010). Accessed 2 Oct 2015.
4. Baddock MC, Strong CL, Murray PS, McTainsh GH. Aeolian dust as a transport hazard. Atmos Environ. 2013;71:7–14.
5. Baldwin BG. The Jepson desert manual: vascular plants of southeastern California. California: University of California Press; 2002.
6. Baptista-Rosas RC, Hinojosa A, Riquelme M. Ecological niche modeling of *Coccidioides* spp. in western North American deserts. Ann NY Acad Sci. 2007;1111:35–46.
7. Baptista-Rosas RC, Catalan-Dibene J, Romero-Olivares AL, Hinojosa A, Cavazos T, Riquelme M. Molecular detection of *Coccidioides* spp. from environmental samples in Baja California: linking Valley Fever to soil and climate conditions. Fung Ecol. 2012;5:177–90.
8. Barker BM, Tabor JA, Shubitz LF, Perrill R, Orbach MJ. Detection and phylogenetic analysis of *Coccidioides posadasii* in Arizona soil samples. Fung Ecol. 2012;5:163–76.
9. Binnicker MJ, Buckwalter SP, Eisberner JJ, Stewart RA, McCullough AE, Wohlfiel SL, Wengenack NL. Detection of *Coccidioides* species in clinical specimens by real-time PCR. J Clin Microbiol. 2007;45:173–8.
10. Bruckner RJ. Renewable energy projects and fugitive dust control, agenda item no. 79-B, board motion of May 14th, 2013, Los Angeles County, Dept. of Regional Planning 2013, Dust Control. http://file.lacounty.gov/bc/q1_2014/cms1_208946.pdf (2013). Accessed 28 Nov 2015.
11. Carlisle JE, Kane SL, Solan D, Joe JC. Support for solar energy: examining sense of place and utility scale development in California. Energy Res Soc Sci. 2014;3:124–30.
12. Correia AW, Pope CA III, Dockery DW, Wang Y, Ezzati M, Dominici F. The effect of air pollution control on life expectancy in the United States: an analysis of 545 US counties for the period 2000 to 2007. Epidemiology. 2013;24:23.
13. County of Los Angeles Crop and Lifestock report 2014. http://file.lacounty.gov/acwm/cms1_232575.pdf. Accessed 8 Jan 2016.
14. County of Los Angeles Department of Public Health. Public Health Reports 1999–2014. http://publichealth.lacounty.gov/phcommon/public/reports/rptspubdisplay.cfm?unit=all&ou=ph&prog=ph. Accessed 10 Oct 2015.
15. County of Los Angeles Department of Regional Planning. Renewable energy. http://planning.lacounty.gov/energy. Accessed 9 Oct 2015.
16. Crum NF, Lederman ER, Stafford CM, Parrish JS, Wallace MR. Coccidioidomycosis: a descriptive survey of a reemerging disease. Clinical characteristics and current controversies. Medicine. 2004;83:149–75.
17. Desert Renewable Energy Conservation Plan (DRECP) http://www.drecp.org/ (2012). Accessed 9 Oct 2015.
18. Farber RJ, Cowherd C, Grelinger MA, Zink T, Cloud Hughes M, Roberts E et al. Stifling the dust in the Antelope

Valley. Paper 2010 A 612 AWMA, Antelope Valley Air Monitoring District. http://www.avaqmd.ca.gov/Modules/ShowDocument.aspx?documentid=2709 (2010). Accessed 17 Oct 2015.
19. First Solar. AV Solar Ranch One Solar Power Plant Achieves 100 MW Milestone, First Solar. http://investor.firstsolar.com/releasedetail.cfm?ReleaseID=741974 (2013). Accessed 1 Oct 2015.
20. Fisher FS, Bultman MW, Johnson SM, Pappagianis D, Zaborsky E. *Coccidioides* niches and habitat parameters in the southwestern United States. Ann NY Acad Sci. 2007;1111:47–72.
21. Flynn NM, Hoeprich PD, Kawachi MM, Lee KK, Lawrence RM, Goldstein EJ, et al. An unusual outbreak of windborne coccidioidomycosis. N Engl J Med. 1979;301:358–61.
22. Galgiani JN. Coccidioidomycosis: a regional disease of national importance: rethinking approaches for control. Ann Intern Med. 1999;130:293–300.
23. Galgiani JN, Ampel NM, Blair JE, Catanzaro A, Johnson RH, Stevens DA, Williams PL. Coccidioidomycosis. Clin Infect Dis. 2005;41:1217–23.
24. Germano DJ, Rathbun GB, Saslaw LR, Cypher BL, Cypher EA, Vredenburgh LM. The San Joaquin Desert of California: ecologically misunderstood and overlooked. Nat Areas J. 2011;1:138–47.
25. Gibson JM, Brammer AS, Davidson CA, Folley T, Launay FJ, Thomsen JT. Burden of disease from outdoor air pollution. In: Environmental burden of disease assessment, pp. 73–108. Springer; 2013
26. Glikson M, Rutherford S, Simpson RW, Mitchell CA, Yago A. Microscopic and submicron components of atmospheric particulate matter during high asthma periods in Brisbane, Queensland, Australia. Atmos Environ. 1995;29:549–62.
27. Gold DR, Mittleman MA. New insights into pollution and the cardiovascular system 2010 to 2012. Circulation. 2013;127:1903–13.
28. Goldstein EJ, Johnson RH, Einstein HE. Coccidioidal meningitis. Clin Infect Dis. 2006;42:103–7.
29. Grantz DA, Vaughn DL, Farber RJ, Bong K, Vancuren R, Campbell R, Bainbridge D, Zink T. DustBusters reduce pollution, wind erosion: though difficult to achieve, revegetation is best way to stabilize soil. 1998. doi:10.3733/ca.v052n04p8 **(Preprint)**.
30. Greater Antelope Valley Economic Alliance (GAVEA). Economic Round Table Report. http://socalleadingedge.org/wp-content/uploads/2014/12/2014gavea report.pdf (2014). Accessed 28 Sept 2015.
31. Greater Antelope Valley Economic Alliance (GAVEA). Economic Round Table Report. http://kedc.com/wp-content/uploads/2013/11/Greater-Antelope-Valley-Economic-Alliance-Round-Table-Report-2015.pdf. Accessed 30 April 2016.
32. Greene DR, Koenig G, Fischer MC, Taylor JW. Soil isolation and molecular identification of *Coccidioides immitis*. Mycologia. 2000;92:406–10.
33. Guevara RE, Motala T, Terashita D. The changing epidemiology of coccidioidomycosis in Los Angeles (LA) County, California, 1973–2011. PLoS ONE 2015;10:e0136753. doi:10.1371/journal.pone.0136753.
34. Huckabone SE, Gulland FM, Johnson SM, Colegrove KM, Dodd EM, Pappagianis D, et al. Coccidioidomycosis and other systemic mycoses of marine mammals stranding along the Central California, USA coast: 1998–2012. J Wildl Dis. 2015;51:295–308.
35. Jibson RW, Harp EL, Schneider E, Hajjeh RA, Spiegel RA. An outbreak of coccidioidomycosis (valley fever) caused by landslides triggered by the 1994 Northridge, California, earthquake. Rev Eng Geol. 1998;2:53–62.
36. Johnson L, Gaab EM, Sanchez J, Bui PQ, Nobile CJ, Hoyer KK, et al. Valley fever: danger lurking in a dust cloud. Microbes Infect. 2014;16:591–600.
37. Johnson SM, Carlson EL, Fisher FS, Pappagianis D. Demonstration of *Coccidioides immitis* and *Coccidioides posadasii* DNA in soil samples collected from Dinosaur National Monument, Utah. Med Mycol. 2014;52:610–7.
38. Kern County Department of Regional Planning. Renewable Energy. http://pcd.kerndsa.com/planning/renewable-energy. Accessed 4 Oct 2015.
39. Kerns V. Scenes from the high desert: Julian Steward's life and theory. Illinois: University of Illinois Press; 2008.
40. Kirkland TN, Fierer J. Coccidioidomycosis: a reemerging infectious disease. Emerg Infect Dis. 1996;2:192.
41. Lancaster N, Gillies JA. Design, Implementation & measurement of stage I pilot projects to reduce fugitive particulate emissions. 2011. Desert Research Institute (DRI), Reno, NV. Report submitted to the San Luis County Air Pollution Control District (SLOAPCD), California Department of Parks and Recreation and the County of San Luis Obispo. http://slocleanair.org/images/cms/upload/files/air/pdf/2011/PMStudy/OceanoDunesPilotProjectsReportFinal%203-15-11.pdf. Accessed 9 Jan 2016.
42. Laniado-Laborín R. Expanding understanding of epidemiology of coccidioidomycosis in the Western hemisphere. Ann NY Acad Sci. 2007;1111:19–34.
43. Laniado-Laborín R, Cabrales-Vargas MN. Amphotericin B: side effects and toxicity. Rev Iberoam Micol. 2009;26:223–7.
44. Lauer A, Baal JDH, Baal JCH, Verma M, Chen JM. Detection of *Coccidioides immitis* in Kern County, California, by multiplex PCR. Mycologia. 2012;104:62–9
45. Lauer A, Talamantes J, Castañon Olivares LR, Medina LJ, Baal JDH, Casimiro K, et al. Combining forces—the use of landsat TM satellite imagery, soil parameter information, and multiplex PCR to detect *Coccidioides immitis* growth sites in Kern County, California. PLoS ONE. 2014;. doi:10.1371/journal.pone.0111921.
46. Litvintseva AP, Marsden-Haug N, Hurst S, Hill H, Gade L, Driebe EM, et al. Valley Fever: Finding new places for an old disease: *Coccidioides immitis* found in Washington State soil associated with recent human infection. Clin Infect Dis. 2015;60:e1–3.
47. Lockeretz W. The lessons of the dust bowl: several decades before the current concern with environmental problems, dust storms ravaged the Great Plains, and the threat of more dust storms still hangs over us. Am Sci. 1978;66:560–9.
48. Los Angeles County, Report INT GTR 315. Los Angeles County Crop and Livestock Report, pp. 113-5. http://file.lacounty.gov/acwm/cms1_232575.pdf (2014). Accessed 6 Nov 2015.
49. MacKay P. Mojave desert wildflowers, 2nd: a field guide to wildflowers, trees, and shrubs of the Mojave desert, including the Mojave National Preserve, Death Valley

National Park, and Joshua Tree National Park. Globe Pequot; 2013.
50. Martin KJ, Rygiewicz PT. Fungal-specific PCR primers developed for analysis of the ITS region of environmental DNA extracts. BMC Microbiol. 2005;5:28.
51. Miles SR, Goudey CB. Ecological subregions of California: section and subsection descriptions. PSW RP-005. USDA Forest Service, Redding, California. 1997.
52. McRae P. The Mojave dust bowl of 2014—causes and solutions, quattro environmental. http://www.quattroenvironmental.com/the-mojave-dust-bowl-of-2014-causes-and-solutions/ (2014). Accessed 2 Oct 2015.
53. Merriam CH. Geographic distribution of animals and plants in North America. U.S. Department of Agriculture. 1895.
54. Morhardt S, Morhardt E. California desert flowers: an introduction to families, genera, and species. California: University of California Press; 2004.
55. Ochoa AG. Coccidioidomycosis in Mexico. In: Aiello L, editor. Proceedings of the symposium on coccidioidomycosis. Phoenix: University of Arizona Press; 1967.
56. Pappagianis D, Einstein H. Tempest from Tehachapi takes toll or *Coccidioides immitis* conveyed aloft and afar. West J Med. 1978;29:527–30.
57. Pappagianis D. Epidemiology of coccidioidomycosis. In: Coccidioidomycosis. Springer; 1980. pp. 63–85.
58. Raphael MN. The Santa Ana winds of California. Earth Interact. 2003;7:1–13.
59. Rawls A. The ugly battle between rural residents and alternative energy mandates in California. Whats up with that? (WUWT). http://wattsupwiththat.com/2012/04/06/the-ugly-battle-between-rural-residents-and-alternative-energy-mandates-in-california/ (2012). Accessed 8 Nov 2015.
60. Reheis MC. Owens (Dry) Lake, California: A Human-Induced Dust Problem. Impacts of Climate Change and Land Use in the Southwestern United States. U.S. Geological Survey. Retrieved 6 March 2010. http://geochange.er.usgs.gov/sw/impacts/geology/owens/ (2006). Accessed 28 Oct 2015.
61. Renewable Energy Action Team (REAT). Best management practices and guidance manual, desert renewable energy projects. California Energy Commission. 2010;265.
62. Reynolds RL, Yount JC, Reheis M, Goldstein H, Chavez P, Fulton R, Whitney J, Fuller C, Forester RM. Dust emission from wet and dry playas in the Mojave Desert, USA. Earth Surface Processes and Landforms. 2007;32:1811–27. http://www.energy.ca.gov/33by2020/documents/index.htm. Accessed 28 Oct 2015.
63. Saubolle MA, McKellar PP, Sussland D. Epidemiologic, clinical, and diagnostic aspects of coccidioidomycosis. J Clin Microbiol. 2007;45:26–30.
64. Schmidt RT, Howard DH. Possibility of *C. immitis* infection of museum personnel. Publ. Health Rep. 1968;83:882.
65. Sharpton TJ, Stajich JE, Rounsley SD, Gardner MJ, Wortman JR, Jordar VS, Maiti R, Kodira CD, Neafsey DE, Zeng Q, Hung CY. Comparative genomic analyses of the human fungal pathogens *Coccidioides* and their relatives. Genome Res. 2009;19:1722–31.
66. Sheff KW, York ER, Driebe EM, Barker BM, Rounsley SD, Waddell VG, et al. Development of a rapid, cost-effective TaqMan Real Time PCR Assay for identification and differentiation of *Coccidioides immitis* and *Coccidioides posadasii*. Med Mycol. 2010;8:466–9.
67. Siegler K. Owens Valley salty as Los Angeles water battle flows into Court. National Public Radio (NPR), 3 Nov 2011. http://www.npr.org/2013/03/11/173463688/owens-valley-salty-as-los-angeles-water-battle-flows-into-court (2011). Accessed 10 Nov 2015.
68. Slayback RD, Bunter WA, Dean LR. Restoring Mojave Desert farmland with native shrubs. In: Wild land shrub and arid land restoration symposium: proceedings, p. 113. DIANE Publishing; 1996.
69. Smith CE, Beard RR, Rosenberger HG, Whiting EG. Effect of season and dust control on coccidioidomycosis. JAMA. 1946;132:833–8.
70. Swatek FE. Ecology of *Coccidioides immitis*. Mycopath Mycol Appl. 1970;41:3–12.
71. Takacs FJ, Tomkiewicz ZM, Merrill JP. Amphotericin B nephrotoxicity with irreversible renal failure. Ann Int Med. 1963;59:716–24.
72. Talamantes J, Behseta S, Zender CS. Statistical modeling of valley fever data in Kern County, California. Int J Biometeorol. 2007;51:307–13.
73. Team, Renewable Energy Action (The DRECP Independent Science Panel). Independent science review for the California desert renewable energy conservation plan (DRECP). http://www.drecp.org/documents/docs/independent_science_2012/2012-09-10_DRECP_Independent_Science_Panel_2012_Draft_Report.pdf (2012). Accessed 18 Sept 2015.
74. Thompson GR III, Stevens DA, Clemons KV, Fierer J, Johnson RH, Sykes J, Rutherford G, Peterson M, Taylor JW, Chaturvedi V. Call for a California coccidioidomycosis consortium to face the top ten challenges posed by a recalcitrant regional disease. Mycopathologia. 2015; 179:1–9.
75. Tolba MK. A harvest of dust? Environ Conserv. 1984;11:1–2.
76. Vargas-Gastélum L, Romero-Olivares AL, Escalante AE, Rocha-Olivares A, Brizuela C, Riquelme M. Impact of seasonal changes on fungal diversity of a semi-arid ecosystem revealed by 454 pyrosequencing. FEMS Microbiol Ecol. 2015;91:5–44.
77. Welsh HH Jr. Bioregions: an ecological and evolutionary perspective and a proposal for California. Calif Fish Game. 1994;80:97–124.
78. Wilken JA, Sondermeyer G, Shusterman D, McNary J, Vugia DJ, McDowell A, Borenstein P, Gilliss D, Ancock B, Prudhomme J, Gold D. Coccidioidomycosis among Workers Constructing Solar Power Farms, California, USA, 2011–2014. Emerg Infect Dis. 2015;21:1997.
79. Zhou J, Bruns MA, Tiedje JM. DNA recovery from soils of diverse composition. Appl Environ Microbiol. 1996;62:316–22.

Anthropogenic Degradation of the Southern California Desert Ecosystem and Prospects for Natural Recovery and Restoration

JEFFREY E. LOVICH
US Geological Survey
Western Ecological Research Center
Department of Biology, University of California
Riverside, California 92521-0427 USA

DAVID BAINBRIDGE
United States International University
Environmental Studies Program
10455 Pomerado Road
San Diego, California 92131-1799 USA

ABSTRACT / Large areas of the southern California desert ecosystem have been negatively affected by off-highway vehicle use, overgrazing by domestic livestock, agriculture, urbanization, construction of roads and utility corridors, air pollution, military training exercises, and other activities. Secondary contributions to degradation include the proliferation of exotic plant species and a higher frequency of anthropogenic fire. Effects of these impacts include alteration or destruction of macro- and micro-vegetation elements, establishment of annual plant communities dominated by exotic species, destruction of soil stabilizers, soil compaction, and increased erosion. Published estimates of recovery time are based on return to predisturbance levels of biomass, cover, density, community structure, or soil characteristics. Natural recovery rates depend on the nature and severity of the impact but are generally very slow. Recovery to predisturbance plant cover and biomass may take 50–300 years, while complete ecosystem recovery may require over 3000 years. Restorative intervention can be used to enhance the success and rate of recovery, but the costs are high and the probability for long-term success is low to moderate. Given the sensitivity of desert habitats to disturbance and the slow rate of natural recovery, the best management option is to limit the extent and intensity of impacts as much as possible.

We've mined it, dammed it, irrigated it, developed it, and subjected it to nuclear assault, yet the desert, somehow both fragile and tough, manages to endure, a rugged old touchstone for us to measure ourselves against.

Malcolm Jones, Jr. 1996

The landscape and native vegetation of the southern California deserts have been significantly altered during the last century by a variety of factors including: livestock grazing (Bentley 1898, Humphrey 1958), introduction of exotic species (Mooney and others 1986, Rejmánek and Randall 1994), off-road vehicle use (see reviews in Webb and Wilshire 1983), urbanization and its attendant effects (Reible and others 1982, Walsh and Hoffer 1991), and military activities (Lathrop 1983a, Prose and others 1987). Extreme temperatures, intense sun, high winds, limited moisture and the low fertility of desert soils make natural recovery of the desert very slow after disturbance (Bainbridge and Virginia 1990). Conditions suitable for plant establishment occur only infrequently and irregularly, and it may take hundreds of years for full recovery to take place without active intervention. Many of the actions of desert development and utilization have profound effects on ecosystem stability, diversity, and productivity (Rundel and Gibson 1996).

The literature on human impacts to the biotic and physical components of the Mojave Desert is large and diffuse. In this paper we review the major human-induced impacts on the California desert, and the prospects for natural recovery and restoration, by characterizing the effects of past actions on the Mojave Desert ecosystem and other arid lands. In addition, we briefly suggest practical strategies and methods for planning and implementing desert restoration projects and improving recovery of these areas by soil management, transplanting, direct seeding, and other techniques.

Area of Study

Our review focuses on the Mojave and Colorado Deserts of southern California, an area of approximately 10 million ha. The Mojave Desert occupies portions of Inyo, Kern, Los Angeles, Riverside, and San Bernardino counties in California. The geographical and ecological boundaries of the Mojave Desert are

KEY WORDS: Mojave Desert; Colorado Desert; California; Human impacts; Recovery; Restoration

discussed in detail by Vasek and Barbour (1977) and Hickman (1993). The modern plant community of the Mojave has been characterized as "desert scrub" (Turner 1982, Hickman 1993), even though it is composed of several recognizable community types including: creosote bush scrub, saltbush scrub, shadscale scrub, blackbush scrub, and Joshua tree woodland (Vasek and Barbour 1977). Perennial plant diversity is low compared to the Colorado Desert: areas dominated by *Larrea tridentata* and *Ambrosia dumosa* occupy about 70% of the Mojave (Lathrop and Rowlands 1983). More than 250 species of annual plants are found in the Mojave, including 80–90 species that are endemic (Turner 1982). In Death Valley and the Salton Sink, annuals account for 42% and 47% of the local flora, respectively (Johnson and others 1978). Overall plant diversity is low below 1000 m, but increases to levels approaching more temperate habitats at higher elevations (Cody 1986).

The Colorado Desert is that part of the Sonoran Desert found mostly in Imperial and Riverside counties, California (Burk 1977). The Colorado Desert is generally separated from the Mojave Desert to the north by the Little San Bernardino, Cottonwood, and Eagle Mountains. The boundary between the two desert ecosystems is poorly defined to the east of these mountain ranges (Vasek and Barbour 1977). A bimodal rainfall pattern composed of winter frontal systems and summer convectional storms distinguishes the Colorado Desert from the western Mojave Desert (Burk 1977), where most precipitation comes from winter rains. In addition, the region is generally lower, flatter, hotter in the summer and warmer in the winter, and hosts a slightly different flora than the Mojave Desert (Hickman 1993). Dominant vegetation in the Colorado Desert is "Sonoran creosote–bush scrub" (Hickman 1993). Plant communities recognized by Burk (1977) include creosote bush scrub, cactus scrub, wash woodland, palm oasis, saltbush scrub, and alkali scrub. There is broad overlap of plant species between the Mojave and Colorado Deserts, but there are a significant number of freeze-sensitive arboreal species that are found only in the Colorado Desert.

Both deserts are characterized by dominant perennial plant species that are long-lived (Bowers and others 1995), some exceptionally so (Vasek 1980). Density and cover of long-lived species increases with age of the site surface (Webb and others 1987, 1988, Bowers and others 1997).

While our focus is specifically directed to the problems of desert lands in California (most of our experience is in the Colorado Desert), we believe our review will prove useful for desert management in other parts of the Southwest, northern Mexico, and in other drylands around the world.

Factors Contributing to Habitat Degradation

The following sections summarize major anthropogenic degradation factors in the southern California desert ecosystem other than agricultural development and urbanization. An understanding of the nature and the effect of disturbances is useful in estimating recovery times or determining what course of action may be required to restore a habitat. Table 1 summarizes the estimated time intervals required for affected plant communities to fully or partially recover from human-induced disturbances.

Impacts on the desert can be loosely divided into historic and current impacts. There is rarely a complete distinction between the two but, in general, the historic impacts include such things as overgrazing, aqueduct building, and the operation of the Desert Training Center in World War II. Grazing still continues, but the major impacts from grazing occurred in the mid to late 1800s. A very rough estimate of the magnitude and extent of these different activities is shown in Table 2. The following factors are not presented in order of importance.

Livestock and Grazing

Cattle and sheep have grazed almost continuously through large areas of the region from the mid-1800s to the present, although the numbers have dropped off in recent years. The establishment of ranching fostered the development of a major industry in the western United States that prospered until droughts, harsh winters, and overgrazing caused a series of dramatic herd declines in the late 1800s. Populations of sheep (60,000) and cattle (67,000) peaked in Imperial County in 1920. In 1968 there were 25,000 cattle and 138,000 sheep grazing on Bureau of Land Management (BLM) and National Monument desert lands in California, predominantly in the Mojave (Ruch 1968). In 1979, 1.8 million ha of public lands administered by the BLM in the California desert were grazed by 75,000 sheep and 14,000 cattle (Bureau of Land Management 1980). Excellent histories of grazing in the desert southwest are provided by Humphrey (1958, 1987).

No published studies have yet fully documented the impact of grazing by livestock in the California desert or estimated the time required for heavily grazed areas to recover to pregrazing levels of plant diversity, density, and cover (Oldemeyer 1994). The rarity of undisturbed reference sites and long-term studies makes it difficult to quantify the effects of grazing, but it is possible to

Table 1. Estimated natural recovery times in years for California desert plant communities subjected to various anthropogenic impacts

Impact	Location	$T_{recovery}$	Reference
Tank tracks (military)	eastern Mojave	65,[a] 76[b]	Lathrop (1983a)
Tent areas (military)	eastern Mojave	45,[a] 58[b]	Lathrop (1983a)
Dirt roadways (military)	eastern Mojave	112,[a] 212[b]	Lathrop (1983a)
Tent sites (military)	eastern Mojave	8–112[c]	Prose and Metzger (1985)
Tent roads (military)	eastern Mojave	57–440[f]	Prose and Metzger (1985)
Parking lots (military)	eastern Mojave	35–440[f]	Prose and Metzger (1985)
Main roads (military)	eastern Mojave	100–infinity[e]	Prose and Metzger (1985)
Military	eastern Mojave	1500–3000[d]	Prose and Metzger (1985)
Townsites	northern Mojave	80–110,[a] 20–50,[b] 1000+[f]	Webb and Newman (1982)
Pipeline	southern Mojave	centuries[g]	Vasek et al. (1975a)
Powerline	southern Mojave	33[h]	Vasek et al. (1975b)
Fire	western Colorado Desert	5[h,i]	O'Leary and Minnich (1981)
Off-road vehicle use	western Mojave	probably centuries	Webb et al. (1983)
Pipeline (berm and trench)	Mojave Desert	100[j]	Lathrop and Archbold (1980b)
Pipeline (road edge)	Mojave Desert	98	Lathrop and Archbold (1980b)
Powerline pylons and road edges	Mojave Desert	100[j]	Lathrop and Archbold (1980b)
Under powerline wires	Mojave Desert	20[j]	Lathrop and Archbold (1980b)

[a]Recovery time to control density

[b]Recovery time to control cover

[c]Estimated recovery time for *Larrea tridentata* to reach control densities

[d]Estimated recovery time ("if at all") for recovery to original vegetative structure assuming establishment of control densities

[e]Compaction recovery time.

[f]Total estimated recovery time.

[g]30–40 years assuming linear rates of succession; 3000 years until formation of large creosote clonal rings

[h]Incomplete recovery time in areas of high impact.

[i]Time for appearance of perennial seedlings. See Brown and Minnich (1986) in section on fire

[j]Biomass recovery assuming that successional vegetative growth is approximated by a straight line. Recovery of long-lived species is estimated to take at least three times longer than indicated.

Table 2. Adverse impacts on California desert, their relative intensity and historical occurrence

Impact	Intensity	Current/historic
Grazing	moderate	primarily historic
Removal of native people	moderate	historic
Invasive plants	moderate/severe	historic/current
Highways	severe	current
Urbanization	severe	current
Off road vehicles	severe	current
Agriculture	severe	both
Military operations	severe	both
Mining	locally severe	both
Linear corridors	locally severe	current

describe the nature of these impacts and their probable extent. Consequently, conclusions about the effects of grazing on arid ecosystems have been contradictory and controversial (Anonymous 1991, Borman and Johnson 1990, Coe 1990, Field 1990, General Accounting Office 1992, Gillis 1991, Poling 1991). Some argue that grazing is beneficial to rangelands, suggesting that the act of grazing stimulates new plant growth (Savory 1988). Other putative positive benefits include the dispersal of seeds, production of fertilizer in the form of excrement, and churning of soil generated by moving hooves (but see Balph and Malecheck 1985). Others point to negative impacts of grazing including soil compaction and increased erosion, trampling of plants, and overcropping. Grazing effects on arid ecosystems are reviewed in detail by Archer and Smeins (1991)

The effects of overgrazing are far less controversial. As early as the late 1800s there was recognition of dramatic range deterioration in the United States as a result of overstocking of cattle (Bentley 1898). In his report, Bentley concluded that "The ranges have been almost ruined, and if not renewed will soon be past all hope of permanent improvement." In spite of early recognition of a problem, solutions have still not been satisfactorily implemented (General Accounting Office 1992).

The impacts of grazing, whether positive or negative, may be extensive. In a recent biological assessment in the western Mojave Desert of California, 100% of a 234-square-km area was impacted to some extent by sheep grazing (Tierra Madre Consultants 1991). In a detailed analysis of the effects of sheep grazing on 2.6 square km of desert tortoise habitat, Nicholson and Humphreys (1981) observed soil disturbances in 80% of the area used by sheep. Thirty three percent of the plot was heavily used by sheep.

Livestock grazing, by its very nature, causes a decrease in plant cover and biomass, at least initially. Decreases in cover have been shown to be associated with a decrease in the diversity and abundance of lizards and other wildlife species in arid ecosystems (Busack and Bury 1974, Germano and Hungerford 1981, Germano and others 1983, Germano and Lawhead 1986). In the Mojave Desert Nicholson and Humphreys (1981) observed large decreases in plant cover in areas grazed by sheep. Similar results were reported by Webb and Stielstra (1979) in the Mojave. In addition, they observed a 60% reduction in above-ground biomass on plots grazed by sheep. Other studies, in American deserts outside of the Mojave Desert, have not detected appreciable differences between grazed and ungrazed plots (Heske and Campbell 1991, Rice and Westoby 1978), but most sites had been grazed before the studies were initiated. An important point to make is that the response of plants to grazing varies according to species, season, plant phenology (Genin and Badan-Dangon 1991), local conditions (drought, edaphic factors, etc.), and past historical use.

Direct effects of grazing on desert animals such as the desert tortoise (*Gopherus agassizii*) are not well documented. Grazing sheep can damage tortoise burrows. Nicholson and Humphreys (1981) reported that of 164 tortoise burrows on a 2.6-square-km study site, 10% were damaged and 4% were destroyed. Most burrows were well protected since they were generally located under shrub cover. Damage was considered to be insignificant since tortoises were often observed digging new burrows in late spring regardless of the availability of existing burrows. Others have gone so far as to suggest that cattle dung actually serves as an important food supply for desert tortoises (Bostick 1990), although this has never been rigorously substantiated (Hal Avery personal communication).

Webb and Stielstra (1979) observed that soils in the Mojave Desert exhibited greater surface strength in areas where sheep bedded and grazed relative to control areas. The greatest compaction occurred in the upper 10 cm but compaction was also observed at lower depths. At the surface, soils are trampled by grazing, often obliterating cryptobiotic soil crusts leading to increased erosional potential. Erosion is of special concern for desert soils because the nutrient capital is often concentrated in the surface soil. Gross disorganization of community structure is possible with the loss of only a few centimeters of soil (Charley and Cowling 1968).

Even limited grazing can cause significant shifts in vegetation and damage to soil crusts. Kleiner and Harper (1977) found that seven plant species that were common in the ungrazed area were absent or insignificant in a comparable grazed section of Canyonlands National Park. They attributed this in part to changes in cryptobiotic soil crust, which decreased from 38% cover in the ungrazed area to 5% in the lightly grazed area. Grazing also increases the spatial and temporal heterogeneity of water, nitrogen, and other soil resources, fostering increased desertification of productive arid lands (Schlesinger and others 1990).

As stated above, the rate of natural recovery of habitats exposed to grazing depends on the intensity of past grazing and local conditions. In a blackbrush (*Coleogyne ramosissima*) association in Utah and Arizona, shrub cover is greater in areas that have never been grazed than in grazed areas. In the same area, plots protected from grazing for ten years showed no difference from heavily grazed areas indicating slow rates of recovery (Jeffries and Klopatek 1987). Exclusion of grazing for 14–19 years did not allow recovery of native perennial grasses in southeastern Arizona (Roundy and Jordan 1988). In the deserts of Kuwait land degradation does not necessarily stop following protection from grazing (Omar 1991). Drought, erosion, and sand encroachment continue to degrade land in the absence of grazing. Human activities and grazing may hasten degradation, but in concert with drought the three can be devastating.

In a recent review of the effects of grazing on public land in the hot deserts (Chihuahuan, Mojave, and Sonoran) of the American Southwest, the General Accounting Office (1992) concluded that a high environmental cost has been exacted on these fragile ecosystems and that land degradation due to grazing is continuing. The report concluded by noting that the high environmental risks, budgetary costs, low economic benefits, and management problems associated with livestock grazing on hot desert public lands merits Congressional consideration. Recommended options included raising grazing fees or appropriating additional funds to offset costs of administration and monitoring, and discontinuing livestock grazing altogether in hot desert areas.

Different plant communities respond to grazing in a variety of ways related to a complexity of factors. Results for the Mojave Desert suggest that livestock grazing can have locally significant effects on the plants (Figure 1) and ultimately on desert wildlife. Efforts to restore

Figure 1. Cattle grazing can have locally significant effects on vegetation and soils, as shown in this photo of a cattle watering area and corral in what is now the Mojave National Preserve, California. Note the almost total destruction of perennial plants in the immediate area. The visual effect is greatly diminished as distance from the watering area increases. Photo by Jeff Lovich.

degraded rangeland in the Mojave should start by considering the effects of grazing and the potential impacts of soil compaction, erosion, and plant community alteration.

Linear Corridors

Roads, railways, powerlines, and pipelines, some of the most conspicuous elements of the modern Mojave Desert landscape, are all characterized by long and relatively narrow corridors of disturbance. The fact that most linear corridors are narrow does not necessarily imply that their impacts are minimal. According to Brum and others (1983), over 8000 km of overhead power transmission lines were present in the California desert in 1980, impacting more than 28,000 ha of land. An additional 50,000 ha of land will be impacted by the year 2000 if the projected threefold increase in power demand is accurate. Information summarized in the California Desert Conservation Area Plan (Bureau of Land Management 1980) suggests that an additional 2000 km of energy production and utility corridors are needed to meet the needs of southern California to the year 2000.

The immediate effect of linear corridor construction on soil conditions and plant cover is one of nearly complete destruction (Vasek and others 1975a). In some cases recovery is retarded due to operation and maintenance of corridors (Artz 1989). Other negative secondary effects of corridors include mortality of animals along roadways (Rosen and Lowe 1994, Boarman and Sazaki 1996), habitat fragmentation and restriction of movements and gene flow, increased access to remote areas for illegal collection and vandalism of plants and animals (Nicholson 1978, Garland and Bradley 1984, Boarman and Sazaki 1996, Jennings 1991), and increased erosion (Wilshire and Prose 1987). The steel towers associated with many electrical energy transmission corridors provide nest sites and hunting perches for ravens (*Corvus corax*), a native predator that has increased dramatically in recent years due to human subsidy. The towers may allow ravens to hunt more effectively for the federally threatened desert tortoise (*Gopherus agassizii*) and other desert wildlife (Boarman 1993). Corridors can also serve as a source of exotic invasive plants brought in on construction equipment (Zink and others 1995). Invasive plants prosper in the disturbed conditions and contribute to an increased likelihood of fire. The construction of pipelines for gas, oil, and water and much more destructive than overhead lines because extensive trenching is usually required. This traditionally has led to severe soil impacts (leaving subsoil on the surface), disturbing stabilized crusts and rock surfaces, and concentrating runoff and erosion. More recent pipelines have incorporated some environmental protection and some rehabilitation but the low value of the desert land, the high cost of revegetation, and the lack of money for enforcement and supervision has often led to neglect and minimal treatment.

The impacts of linear structures can extend far beyond the boundaries of the immediate disturbance. Schlesinger and others (1989) studied the effects of diversion structures (earthen dikes) along the Colorado River Aqueduct on plants and soil. The structures were constructed to prevent runoff due to precipitation from washing sediments into open portions of the canal. Large areas downslope of the diversion structures received only incident precipitation, with essentially no runoff from the extensive drainages in the uplands above the diversion structures. As a result, large areas of desert habitat on the downslope side of the diversion structures had a lower biomass of perennial and annual plants in comparison to adjacent areas with no diversion structures.

Garland and Bradley (1984) observed that some species of rodents in the Mojave of Nevada are more abundant near highways, while others are not. However, reduced abundance may have been an artifact of natural habitat heterogeneity since no mortality was observed during the 11-month study. Another effect of roads is edge enhancement in which perennial shrubs along roadsides are denser, larger, more vigorous, and support greater numbers of foliage arthropods than those away from roadsides (Vasek and others 1975b, Lightfoot and Whitford 1991). Johnson and others (1975) noted that primary productivity, as measured by standing crop, at study sites in the Mojave Desert of California increased about 17 times on the basis of vegetated area alone and 6 times when the area of the bare road surface was included as part of the productive unit. Unpaved roads showed increases of 6 and 3 times, respectively, in each category. Increased water availability from pavement runoff and increased retention of moisture under the pavement are probably responsible for the observed increase in plant vigor, although removal of competing plants that formerly occupied the roadway may confer an advantage to plants along the berm (Vasek and others 1975a). The increase in vigor attracts herbivorous insects (Lightfoot and Whitford 1991).

The effects and recovery of linear corridor construction in deserts have been studied by several researchers. The process of natural recovery, following powerline construction in the Sonoran Desert starts immediately with invasion by pioneering annual species, but perennial species may not return for over five years. The density and diversity of annual species may increase in comparison with undisturbed sites, perhaps due to the removal of large woody species (Hessing and Johnson 1982). An effect that is apparently linked to changes in plant abundance and composition is a reduction in the density, but not the community composition, of arthropods following establishment of access roads for powerline construction (Johnson and others 1983).

In the Mojave Desert, plant cover also increases following powerline construction. The rate of increase and composition of colonizing species varies considerably, confounding the ability to predict succession relative to adjacent undisturbed areas. Ground cover of short-lived perennial species increases in areas of severe disturbance, under the central wires, and along the edge of maintenance roads. After 33 years there was a noticeable, but not complete, recovery of predisturbance vegetation (Vasek and others 1975b). Natural revegetation (0–41% ground cover) by long-lived perennials has been observed 12 years after construction of a pipeline by trenching, piling, and refilling (Vasek and others 1975a). Disturbed and control areas appear to have similar cover, biomass, and densities of vegetation following partial recovery, but similarities disappear when the proportions of long-lived and dominant species are compared (Lathrop and Archbold 1980a,b). Species with these characteristics are not well represented on disturbed sites.

Management strategies for minimizing the effects of linear corridor construction include: placement of power poles closer to existing access roads, modifying construction techniques for buried pipelines, less frequent road grading, and limiting the width of motorcycle race corridors along powerlines (Artz 1989). Lathrop and Archbold (1980b) proposed several recommendations for routing corridors to minimize environmental impacts including: (1) routing them through gently sloping areas to minimize erosion, (2) routing them through areas occupied by colonizing species such as cheesebush (*Hymenoclea salsola*), (3) avoiding areas dominated by high nitrogen fixation communities such as cat's claw acacia (*Acacia greggii*), and (4) avoidance of undue soil compaction with implementation of soil loosening efforts to aid natural revegetation. Revegetation of linear corridors was evaluated by Kay (1979, 1988), Graves and others (1978), and Brum and others (1983).

The slow recovery of the desert to linear corridor impacts is perhaps best demonstrated by the visibility of many of the old Native American trade routes. Long-term use by foot traffic alone was sufficient to compact the soil and recovery after several hundred years has not been enough to hide these trails (personal observation).

Mining

Mining has been an important activity in the California desert since the late 1880s. Mining communities such as Kokoweef, Hart Mountain, Boron, Johannesburg, and many others have had mostly localized impacts on the desert. The most obvious forms of degradation are pits, ore dumps, and tailings, but the once-great demand for fuel and timber, grazing, and road building associated with mines was unquestionably more important in the past. Fugitive dust and toxic tailings are a more recent concern from some of these mining areas.

The Bureau of Land Management (1980) estimated that 12,545 ha in the California Desert Conservation

Area had been affected by major mining operations. If the many small prospects and adits are included, the area affected by mining would certainly be larger. The brine evaporation and dry lake mine operations are extensive and lead to substantial wind erosion (Wilshire 1983). Another problem is animal mortality at poorly managed cyanide extraction gold mines in the Mojave Desert (Clark and Hothem 1991, Henny and others 1994).

Military Training Operations

Large areas of the California desert have been impacted by temporary and ongoing military activities. Major training exercises included activities by General Patton in the early 1940s, the Desert Strike operation in 1964, and Bold Eagle in 1976. Between 1942 and 1944 more than a million soldiers passed through these training facilities, which covered more than 46,800 square km (Bureau of Land Management, 1990). The camps were effectively small cities, up to 2800 ha in size (e.g., Camp Granite) (Prose and Metzger 1985). Continuing impacts are generated by active military bases including the National Training Center (at Fort Irwin, the Marine Corps Air Ground Combat Center at Twentynine Palms, China Lake Naval Air Weapons Station, and the Chocolate Mountain Aerial Gunnery Range (Lathrop 1983a). Military operations cause intensive damage in many areas but also provide protection of thousands of hectares from other sources of disturbance by prohibiting public access. At Fort Irwin alone, the area in need of remediation is estimated to exceed 50,000 ha.

The recovery of large areas of the eastern Mojave Desert subjected to military training exercises almost 36 years earlier was studied by Lathrop (1983b). Impacted areas included tent sites, roads, and tank tracks. All impacted areas exhibited significant reductions in plant density and cover relative to control areas. Reductions of cover and density were greatest in tank tracks and least in tent areas. Recovery to predisturbance levels of cover and density varied according to disturbance type. Tent areas showed the greatest recovery, and roadways showed the least, reflecting the intensity of disturbance. Recovery in tank tracks was intermediate. Diversity of dominant perennials also varied between disturbed and nondisturbed areas but results were clouded by low species richness at the study sites and small sample sizes of the subdominants. However, diversity in disturbed transects at the Camp Ibis study site was low relative to control sites. Species similarity decreased between control and disturbed transects with increased disturbance and use intensity.

Similar observations and conclusions were reached by Prose and Metzger (1985) and Prose and others (1987) at abandoned military camps in the eastern Mojave. Long-lived species such as *Larrea tridentata* were dominant in all control areas but percentage cover and density were reduced in impacted areas. Dominant plants in disturbed areas included pioneer species such as *Ambrosia dumosa* and *Hymenoclea salsola*. Percentage cover values for pioneer species in disturbed areas were equal to or greater than control values.

Differences in vegetative structure between control and impacted plots were due to soil compaction, changes in soil texture, removal of the top layer of soil, and alteration of drainage channel density (Prose and others 1987). Penetrometer measurements show that a single pass by a "medium" tank can increase average soil resistance values by 50% relative to adjacent untracked soil in the upper 20 cm, but values of up to 73% were recorded. Dirt roadways could not be penetrated with a penetrometer below 5–10 cm due to extreme compaction. Physical modifications to the soil beneath tank tracks extended vertically to a depth of 25 cm and outward from the track edge to 50 cm (Prose 1985).

Recovery times to predisturbance levels of density and cover were estimated by Lathrop (1983b) assuming linear rates (Table 1). Recovery to predisturbance species composition would require much longer, if it were to occur at all. Areas receiving the greatest amount of soil compaction, such as roadways, require the longest recovery times. Tank tracks and tent areas recover in a shorter amount of time. Overall, recovery in plant density is slow relative to increases in cover. In other words, the number of individuals changes little following recovery from disturbance, but surviving individuals cover larger areas. A major conclusion from Lathrop's study was that recovery to some original level of community composition and stability may not occur in the foreseeable future. However, recovery of comparable disturbed areas has been excellent on restoration test plots at the Marine Corps Air Ground Combat Center near Twentynine Palms, California (Zink personal communication).

Off-Road Vehicles

Off-road vehicle (OHV) use is one of the major recreational activities in the deserts of California. The Motorcycle Industry Council estimated that 4.7 million motorcycles were used by 11.7 million people in 1978 for off-highway recreation in the United States, a figure that does not include dune buggies and four-wheel drive vehicles (Kockelman 1983).

The impacts of OHVs have been well documented (Webb and Wilshire 1983) and include destruction of soil stabilizers (see section on biotic components of soil), soil compaction, reduced rates of water infiltration, increased wind and water erosion, noise, decreased abundance of lizard populations (Busack and Bury 1974), and destruction of vegetation (Vollmer and others 1976). Compaction of a desert soil reduces the root growth of desert plants and makes it much harder for seedlings to survive (Bainbridge and Virginia 1990, Bainbridge and others 1995a). An excellent review of the effects of OHVs in the Mojave and other deserts is contained in Webb and Wilshire (1983) and the reader is referred to that document for information beyond that presented herein.

Soil compaction is a common effect of any compressive action on most soils. Compaction results from a variety of factors other than OHV use, including trampling by grazers, human trampling (Liddle 1991, 1997), and even raindrops (see review in Webb 1982). In the case of OHVs, compaction occurs at shallow depths related to the geometry of the contact surface between the tire and the soil interface. In one study the greatest increase in soil density occurred at a depth of 30–60 cm after being compacted by a motorcycle (Webb 1983). Soil density increases as a function of the number of vehicle passes, while soil infiltration rate decreases. Soils that are most susceptible to compaction are loamy sands and coarse gravelly soils with variable particle sizes. Wet soils are more susceptible to compaction than dry soil. Soils that are least affected include sands and clays.

Another by-product of heavy OHV use is increased wind and water erosion. The degree of erosion experienced in an area exposed to OHV use is affected by two main factors. First, increased water erosion is partially attributable to decreased infiltration rates due to compaction. Second, OHVs destroy surface stabilizers (see section on biotic components of soil), making soils more susceptible to erosion (Hinkley and others 1983). The enormity of the problem in the Mojave Desert is underscored by the fact that satellite photos revealed six dust plumes covering over 1700 square km of the western Mojave on 1 January 1973 that were attributed to surface destabilization primarily by OHVs (Nakata and others 1976, Gill 1996).

As shown in numerous photographs in Webb and Wilshire (1983), the effects of erosion can have indirect effects, since debris flows (Nakata 1983) can bury plants at some distance from the impacted area. Areas that are least susceptible to water and wind erosion following OHV use are dunes, playas, and areas with abundant coarse surface material (Gillette and Adams 1983, Hinckley and others 1983). Restoration of OHV areas affected by erosion requires actions to not only stop continuing erosion (Harding 1990, Heede 1983, Middleton 1990), but also action to restore past damage.

Desert soils vary in their susceptibility to OHV damage. Susceptibility is generally high in all areas except barren sand dunes (but see Bury and Luckenbach 1983), and the clay flats of playas. Soil damage caused by OHVs is environmentally significant due to the fact that desert soils may take 10,000 years to develop (Dregne 1983). From this estimate, Dregne concluded that it was futile to speak of disturbed soil recovery in time frames related to human occupancy.

Another major effect of OHV use is the destruction of plants. Lathrop (1983a) examined aerial photographs of nine disturbed and undisturbed areas in the Mojave Desert to assess the effects of OHV usage. Perennial plant density and cover were dramatically reduced in OHV areas. The percentage of cover and/or density in OHV-impacted areas relative to control areas was less than 15% in three of the sites examined. Destruction of plants resulted not only from crushing stems and foliage, the extensive root systems that fill the intershrub spaces, and germinating seeds, but also from the superstructure of the vehicle. The latter factor is important since it is responsible for plant destruction in an area wider than the track width of the vehicle. The wheel tracks of a full-size off-road vehicle operating in an undisturbed area can damage almost 0.5 ha of land with every 6.44 km traveled. Support vehicles, including very large and heavy motor homes, are very destructive, and camping areas are especially hard hit.

An easily detected but poorly understood effect of OHVs is noise. Noise from certain types of OHVs can reach 110 decibels, which is near the threshold of human pain. Brattstrom and Bondello (1983) demonstrated that OHV use in the Mojave Desert caused noise levels that caused hearing loss in animals such as kangaroo rats, desert iguanas, and fringe-toed lizards; interfered with the ability of kangaroo rats to detect predators such as rattlesnakes; and caused unnatural emergence of spadefoot toads that were estivating until the arrival of rain for breeding, a situation that could result in death. The authors noted that although OHVs are not the loudest source of human-generated sound in the Mojave, they occur more frequently than any other high-intensity sound source. In their report, Brattstrom and Bondello recommended that OHV areas be located away from the ranges of "all undisturbed desert habitats, critical habitats, and all ranges of threatened, endangered, or otherwise protected desert species."

The impact of OHV use on desert tortoises in the Mojave Desert of California was examined by Bury and Luckenbach (1986) in an unpublished report. Significantly more tortoises and active burrows were found on a 25-ha control plot than on a similar plot exposed to OHV use. In addition, subadult and adult tortoises on the control plot exhibited larger body mass than those on the OHV plot.

Impacts related to OHV use present a serious challenge to desert restoration projects for three reasons: (1) the potentially severe impact of OHV use in desert ecosystems, (2) the widespread nature of the OHV impacts in the California desert, and (3) the fact that OHV areas are often located in or near environmentally sensitive habitats. Areas targeted for restoration should be closed to OHV use prior to initiating procedures to ameliorate past damages.

Invasive Plants

Invasive exotic plants have had a significant impact on the natural communities of California (Mooney and others 1986, Rejmánek and Randall 1994), including the southern California desert ecosystem. Invasion has been facilitated by habitat disturbances that allow exotic species to colonize habitats once dominated by native species (Hunter and others 1987). Once established, exotic plants may diminish the abundance of native species due to competitive interactions or by disruption of natural processes such as fire frequency and intensity.

Some of the more important exotic plants in the southern California desert are saltcedar (*Tamarix ramosissima*), also known as tamarisk (Lovich and de Gouvenain 1998), Russian thistle (*Salsola iberica*) (Young 1991), filaree (*Erodium cicutarium*), and several grass species including split grass (*Schismus* spp.) and bromes (*Bromus* spp.) (Brown and Minnich 1986, Hunter 1991). Immense areas of desert are colonized by these species. Although other exotic plants are present in the Mojave Desert, these are important because of their ubiquity.

Exotic plants present two major problems to the integrity of the desert ecosystem. First exotic annuals increase the fuel load and frequency of fire in a community that is poorly adapted to fire. Second, some exotic plants exhibit allelopathic effects that negatively affect native species, especially annuals. Negative interactions have been demonstrated between Russian thistle and other species in the laboratory (Allen 1982a, Lodhi 1979). In addition, competition of Russian thistle with native perennial grasses increases under drought conditions (Allen 1982b), furthering establishment of the exotic. Fortunately, Russian thistle competes poorly with established vegetation and rarely supplants well-established native populations. Unfortunately, once the soil is disturbed and native plants are eliminated, Russian thistle gains a strong foothold (Young 1991). General reviews of the threats posed by exotic species invasions in native ecosystems are summarized by Cheater (1992) and D'Antonio and Dudley (1993).

Air Pollution

One of southern California's most famous exports is smog. While most noticeable in the inland valleys of the state, smog is often transported via atmospheric processes into the Mojave Desert (Pryor and Hoffer 1991). Anthropogenic pollutants include ozone, sulfur dioxide, and various particulates. Atmospheric tracer experiments have shown that pollutants released in the San Fernando Valley impact the southern Mojave Desert towns of Adelanto and Palmdale, while those released in the southern San Joaquin Valley impact the northern Mojave Desert towns of Mojave and China Lake (Reible and others 1982). Experimental tracers used in atmospheric transport studies are diluted by factors of only 2–3 during passage between source and receptor areas. Impacts are maximized during evening and nighttime hours, independent of the time of release in the San Joaquin Valley, because of the diurnal mountain–valley wind cycle. Ozone levels in the Mojave Desert can exceed 100 parts per billion (ppb) or more when offshore wind transports atmospheric pollutants from the Los Angeles Basin (Thompson and others 1984a). By comparison, ozone levels in remote areas range from 20 to 40 ppb.

The most obvious effect of smog in the Mojave Desert has been visibility degradation in an area historically distinguished by extraordinary visibility (Walsh and Hoffer 1991). Median visibility is 48–88 km in large urban areas and 104–128 km in nonurban locations. Visibility has decreased 10%–30% from the middle of the 1950s to the early 1970s at many recording stations (Trijonis 1979).

Much of the visibility loss is related to particulates, including nitrogen-rich compounds. Dryfall of these compounds from air pollution can be a major source of supplemental N for plants. This favors many exotic plant species over native annuals and perennials. Wedin and Tilman (1996) found that half the native plant species in a Minnesota grassland were lost from the community at supplemental N levels mimicking dryfall deposition rates.

A less obvious effect is damage to plants. Stolte (1991) observed injurious effects to desert plants exposed to ozone and sulfur dioxide in laboratory experiments. Annual plant species of the genera *Camissonia* and *Cryptantha* exhibit high sensitivity to both gases. The grass *Oryzopsis hymenoides* exhibits high sensitivity to sulfur dioxide, as do some types of cryptogamic soils. Responses of cryptogamic soils include increased electrolyte leakage, chlorophyll degradation, and reduced nitrogen fixation (Belnap 1991).

Studies of plants from the Mojave and Colorado Deserts show that perennial species vary in their re-

Figure 2. The effects of fire in the desert are obvious in this photo taken near Palm Springs, California, about five years after the blaze. Note the almost complete elimination of perennial shrubs in the burned area to the left. Perennial plant species in the Mojave and Colorado Deserts are long-lived and very sensitive to fire, traits that collectively contribute to the long recovery times typical of many desert plant communities after fire. Photo by Jeff Lovich.

sponse to SO_2 and NO_2. *Larrea tridentata* is sensitive to fumigation by these pollutants under experimental conditions, displaying extensive leaf injury and reduced growth or dry weight. *Encelia farinosa* and *Ambrosia dumosa* show intermediate responses, while *Atriplex canescens* appears to be resistant (Thompson and others 1980). Sensitivity also varies among native annual plants, with *Camissonia claviformis, C. hirtella,* and *Cryptantha nevadensis* exhibiting leaf injury at low concentrations of SO_2 and O_3 (Thompson and others 1984b).

Fisher (1978) suggested that high rates of mortality in desert holly (*Atriplex hymenelytra*) in the northern Mojave Desert (Death Valley) were related to elevated ozone levels. During the summer months he recorded ozone levels that were twice the national standard of 0.08 ppm. Photosynthesis and water use was significantly reduced in greenhouse experiments where seedlings were exposed to 0.15–0.18 ppm ozone for 3 h. Ozone-induced reduction in water-use efficiency was postulated to be the cause of declining *Atriplex* populations in Death Valley.

Additional summaries of the impacts of air pollution in the Mojave and Colorado Deserts are provided by Mangis and others (1991), Thompson (1995), and VanCuren (1995).

Anthropogenic Fire

Fire was not an important factor in shaping the prehistoric structure and dynamics of plant communities in the California desert. The infrequency of fire in the prehuman landscape of the desert was due to limited biomass, large intershrub spacing, low combustibility of some native plants, sparse groundcover to support and propagate combustion, and the absence of human-mediated fire suppression activities (Humphrey 1974, O'Leary and Minnich 1981, Minnich 1983, Brown and Minnich 1986). Such is not the case in other desert and semidesert areas of the American Southwest, including parts of the Sonoran and Chihuahuan deserts, where fire was an important prehistoric agent in maintaining grassland seral stages (Humphrey 1958, 1963, 1987, Reynolds and Bohning 1956).

The proliferation of exotic annual plant species such as *Bromus, Schismus,* and *Salsola* has dramatically increased the fuel load and frequency of fires in many ecosystems around the world (D'Antonio and Vitousek 1992), including parts of the California desert (O'Leary and Minnich 1981, Brown and Minnich 1986), in recent years. The frequency of fires in the Colorado Desert of California is further enhanced by the proximity of previously burned areas (Chou and others 1990). Native perennial shrubs are poorly adapted to relatively low-intensity fires as evidenced by low rates of recovery (Figure 2). In the upper Coachella Valley on the east scarp of the San Jacinto Mountains near Palm Springs, California, burned creosote bush scrub is replaced by open stands of *Encelia farinosa*, native ephemerals, and exotic species such as *Schismus* and *Bromus* (Brown and Minnich 1986).

Postfire vegetational recovery along a chaparral–desert ecotone including parts of Anza-Borrego Desert State Park in San Diego County, California was examined by Tratz and Vogl (1977). They observed high recovery (as measured by speed of resprouting) in chaparral shrubs and desert-wash plants, but low recovery in cacti. Herbivorous mammals present before the burn were also present afterwards, since rapid recovery of shrubs provided adequate food supplies for wildlife, even in the first months after the fire. If California desert perennial plant communities are not well adapted to fires, animals that coevolved in the ecosystem should not be expected to respond favorably to fire either.

According to fire personnel at the California Desert District (CDD) Office of the Bureau of Land Management (BLM), the CDD (including the Mojave and Colorado Deserts) had a ten-year average of 175 fires per year prior to 1992 (range 100–475) that affect an average of 10,927 ha annually (range 607–34,400 ha). The CDD estimates include a very small amount of BLM land outside the desert.

Impacts on Biotic Components of Soil: The Invisible Component of Biodiversity

Although emphasis is often placed on the physical and chemical properties of various soils, they contain important biotic components as well including: soil surface stabilizers such as algae and lichens, nematodes and other metazoans, various bacteria, and mycorrhizae. Odum (1994) referred to these organisms as the invisible component of biodiversity. While not as conspicuous as macrofloral elements, biotic components of soil are important symbionts that are easily destroyed by certain human activities.

Undisturbed desert areas are characterized by the presence of soil stabilizers, including lichen, fungal, bacterial, and algal crusts; desert pavement; mechanical crusts; and chemical crusts. The biotic components of these stabilizers are collectively referred to as cryptobiotic soil. Mineral-derived crusts form under a variety of physical and chemical conditions that may actually be facilitated by biotic components (Elvidge and Iverson 1983, Taylor-George and others 1983). Soil stabilizers are important agents in preventing erosion but are easily disturbed since they occur at the surface. Stabilization mechanisms include binding soil particles with thallial filaments in the case of biotic stabilizers, armoring the surface, and increasing surface roughness. Crusts also provide germination sites for vascular plants (but see Wood and others 1982), and conserve water (see review in Cole 1990). The susceptibility of crusts to damage varies according to the composition of the underlying soil. In soils subjected to large shear stresses, a single pass by a vehicle is capable of destroying well-developed crust. When the forces are mainly compressive, crusts can survive a single pass in a slightly modified form; however, OHV use is capable of quickly eliminating crusts in an impact area (Wilshire 1983).

Considerable research has been conducted on the impacts of grazing and other agents of trampling on cryptobiotic soil crusts. These crusts are very important not only because of the soil-stabilization functions mentioned above, but because they facilitate the accumulation of organic material and soil nutrients, particularly nitrogen in the upper layers of soil (Kleiner and Harper 1977, Johansen 1993), and enhance soil moisture retention (Belnap and Gardner 1993). Research in desert and semidesert areas in Utah and Arizona has consistently shown that cryptobiotic soil is heavily impacted by grazing, even light winter grazing (Kleiner and Harper 1977, Anderson and others 1982, Brotherson and others 1983). Impacts include the destruction of surface pinnacles associated with development of cryptogamic soils (Anderson and others 1982) and the virtual obliteration of biotic elements (Cole 1990). Lichens and mosses are most sensitive to disturbance, with algal components being more resilient (Brotherson and others 1983).

Cole (1990) conducted an interesting experiment at Grand Canyon National Park to examine the effect of trampling by hikers wearing lug-soled boots. Only 15 passes were required to destroy crusts. Visual evidence of biotic components was reduced to near zero after 50 passes. The results of Cole's experiment clearly illustrate the fragility of crusts to trampling.

Cryptobiotic soil recovery may require long time intervals without intervention. Following exclusion of grazing in a Utah semidesert study site, cryptobiotic cover increased from 4%–15% in 14–18 years, but only 1% per year for the next 20 years (Anderson and others 1982). Cole (1990) observed partial recovery from human trampling in one to three years and extensive recovery after five years. However, surface irregularities associated with well-developed cryptogamic cover remained low even after five years, suggesting that recovery was incomplete. Belnap (1993) noted that over 250 years may be required for full recovery on the Colorado Plateau. Recovery was improved but was still very slow when scalped experimental plots were inoculated with crusts from surrounding areas. In the northern Mojave Desert, lichen crusts may not reoccupy heavily disturbed areas even after 63 years (Wilshire 1983). Details of the formation and recovery of chemical and mechanical crusts are discussed in detail by Wilshire (1983). The nitrogen-fixation capabilities of damaged soil may take over 50 years to recover (Belnap 1995).

Important symbiotic relationships have developed between certain species of vascular plants and vesicular arbuscular mycorrhizal (VAM) fungi and rhizobia. The small-diameter hyphae of symbiotic fungi serve as energy efficient root hairs, enabling the host plant to better absorb nutrients, particularly phosphorus (Bloss 1985) and water (Bethlenfalvay and others 1984). Rhizobia are bacteria capable of fixing atmospheric nitrogen for use by plants. The importance of VAM fungi in desert plant communities is underscored by the fact that in a recent survey of 38 plant species (19 families) in Anza-Borrego Desert State Park in the Colorado Desert of California all were colonized by VAM species (Bethlenfalvay and others 1984). Plants naturally associated with VAM that are also found in the western Mojave Desert include *Hymenoclea*, *Ambrosia*, *Opuntia*, and *Larrea*. Bloss (1985) reported numerous plant associations in the Sonoran Desert of Arizona as well.

Previous studies have demonstrated the importance of maintaining and enhancing soil microbes in restoration projects (St. John 1984, Bainbridge 1990). Establishing plants in disturbed areas with marginal soils may be difficult or impossible without the presence of a vigorous population of microbial symbionts. These symbionts are adversely affected by soil compaction. Studies have shown 1–2 m of hyphae per gram of soil in Mojave and Sonoran soils, yet virtually none in disturbed areas (Zink personal communication). Restoration is complicated by the fact that fertilizers can inhibit mycorrhizae growth.

Can the Desert Be Restored?

Plant growth and establishment are naturally slow under the extreme conditions of the desert, and disturbance makes these conditions even more severe (Bainbridge 1990). Disturbance typically reduces both the infiltration of water into the soil and the moisture-holding capacity of the soil (Bainbridge and Virginia 1990). This increases the value of rapid deep root growth, which is made more difficult by increases in soil strength from compaction and reduced soil moisture. These synergistic effects make plant establishment much more difficult after disturbance. Revegetation and restoration work can help mitigate many of these impacts and speed recovery, but the severe conditions and unpredictable rainfall still make restoration of these sites very challenging.

A brief history of revegetation studies in the deserts of California was provided by Kay and Graves (1983). Studies in the Mojave Desert are few and relatively recent. One of the earliest studies evaluated the success of revegetation efforts along the second Los Angeles Aqueduct (Kay 1979, 1988). Construction involved stripping the vegetation from an area 200 km long × 60 m between 1968 and 1970. The seeds of seven species of native plants were distributed at six 2- to 15-ha sites on the aqueduct. The seeds of all but one species, *Atriplex polycarpa*, were from local stock. Surface preparation involved ripping the soil to 25 cm on 60-cm centers to relieve compaction. A rangeland drill was used to set the seeds at a depth of about 1 cm. Success varied among plant species. *Ambrosia dumosa* exhibited good establishment on three of six sites, but only one site had numbers approaching that of adjacent undisturbed areas. *Larrea tridentata* exhibited similar results. The other species, including *Atriplex polycarpa*, *Ephedra nevadensis*, *Hymenoclea salsola*, and *Lepidospartum squamatum*, were totally unsuccessful. *Atriplex canescens* suffered as a result of heavy grazing. The most abundant shrub along the aqueduct, *Chrysothamnus nauseosus*, established itself naturally, although it was uncommon in adjacent undisturbed areas. Kay (1988) concluded that natural revegetation is good in many years and poor in others, while artificial seeding did not consistently hasten or improve plant recovery.

In another experiment along the aqueduct, Graves and others (1978) tested the effects of a single irrigation and the success of direct seeding versus transplanting. The two methods of establishment exhibited widely variable success rates from site to site and according to species, but were not enhanced by irrigation. Substrate characteristics may influence the success of irrigation as measured by the appearance of native winter annuals (Johnson and others 1978).

The overall success of the revegetation attempt along the aqueduct was low. The vast majority of the aqueduct was still a highly visible scar in the early 1980s (Kay and Graves 1983), but recovery was inhibited by grazing and OHV use. Conclusions from the study were that more attention should be focused on establishment of visually dominant species such as *Larrea tridentata*, seeding should take place as soon after disturbance as possible, areas should be protected from grazing and OHV use, and local seed stock should be utilized for all species.

Highway revegetation studies were also reviewed by Kay and Graves (1983). Survival of container-grown shrubs planted in October 1973 and February 1974 at a site in Mojave, California, was 90% in May 1974. The roots of the transplants were exposed after a heavy rain in December 1974, and all plants were dead by October 1975. *Atriplex* spp., *Chrysothamnus* spp., and *Ephedra* spp. exhibited the greatest survival. Success was limited by rabbit overgrazing and competition from Russian thistle (*Salsola*). Container plantings were more successful when planted in the late winter or early spring. Application of fertilizer encouraged both the invasion of native woody shrubs and the nonnative annual grass *Schismus arabicus*.

Others have experienced similar success in revegetation. Brum and others (1983) observed low, long-term seedling establishment for a variety of species under several irrigation treatments along a powerline transmission corridor. The overall germination–establishment rate for seedling and postseeding irrigation success was 0.3%, and 26% for transplanted seedlings. *Larrea* exhibited poor germination under field conditions and responded poorly to all revegetation attempts.

More successful revegetation has been achieved at the Nevada Test Site in the northern Mojave Desert (Romney and others 1990). Greater than 80% survival of transplanted native shrubs and grasses was achieved when plants were protected from jackrabbits and irrigation was provided periodically.

Restoration efforts in the Colorado Desert of California were reviewed by Bainbridge and Virginia (1990). Although the plant communities differ somewhat between the Colorado and Mojave Deserts, both ecosystems pose similar challenges to restoration attempts: high temperatures, intense sunlight, limited moisture availability, high levels of herbivory by rodents and rabbits, and low soil fertility. Much of the success in revegetation experiments in the Colorado Desert is due to efforts to protect plants from herbivores and the use of buried water reservoirs for irrigation. Direct seeding attempts have generally been unsuccessful relative to transplants. *Larrea tridentata*, in particular, responded well to transplanting, especially if pruned prior to planting to increase the root-to-shoot ratio.

Assessing the nature and magnitude of human-induced disturbances makes restoration planning more efficient by enabling limited resources to be directed at critical problems. Ongoing studies (Bainbridge and others 1995a,b) of the effectiveness of desert restoration techniques are steadily advancing our ability to rehabilitate degraded arid lands in the southwestern United States, and the reader is referred to these references for details beyond the brief overview given in this section.

Plant recovery usually requires container-planting activities as well as site improvement. The most common method of direct seeding is simple hand seeding, which allows species to be matched to specific site conditions, appropriate planting depths, and results in a more natural appearance than machine planting. However, limited rainfall and removal of seeds by rodents and harvester ants may severely limit seedling establishment during typical years.

Transplanting is increasingly being used to provide nurse and seed plants for the disturbed areas (Bainbridge and others 1995b). The dominant shrubs and trees of the Colorado Desert are relatively easy to grow in a nursery or maintained landscape setting, and they are well adapted to transplanting with after-care. They are more challenging to establish in the field in a low- or no-maintenance situation, although once established, growth rates can be high. Reestablishment of annuals has been more difficult. New containers and soil mixes have improved plant survival. Deep pipe and buried pot irrigation and hand watering have also been effective. Tree shelters to limit herbivory and wind damage are also important.

A full appreciation of the ecological setting and adaptation of desert plants can make establishment less costly and more successful, but it is still expensive. The cost of restoring road edge areas in Joshua Tree National Park is fairly well established (after almost 10 years of work) and runs up to $15,000 per ha to establish large potted perennials in areas that are easily accessed. The cost of duplicating this type of work at remote sites would be much higher. Research conducted by colleagues at San Diego State University has emphasized lower-cost, less-intensive restoration, but the costs (excluding research) are still on the order of $12,000–25,000/ha. Even these high project costs provide no guarantee of success.

Conclusions

Desert areas disturbed by human activities may take centuries to recover without active intervention. Undisturbed desert soils are often in a relatively stable equilibrium developed over hundreds or thousands of years. Removal of vegetation and disturbance of soil crusts or soil structure can destroy this equilibrium, leading to wind and water erosion that are very difficult or impossible to control without very high investments in material and labor.

One of the key lessons of our research in the Mojave and Sonoran deserts is the critical importance of minimizing the intensity, frequency, and area of disturbance. Past research summarized in this paper has identified the wide range of effects from human disturbance and the difficulty and the high cost of mitigating damage. While recovery rates can be increased with modest expenditures, a major restoration program to improve recovery for just the OHV-damaged areas in the California desert region could exceed one billion dollars. Available funding will permit only a limited restoration for selected sites, even with continuing generous contributions of volunteer labor. Fences, signs, and enforcement to prevent further damage may often be a better investment than intensive restoration.

Recent research in the Mojave Desert demonstrates the benefits that protection can impart, even to previously disturbed areas. Brooks (1995) conducted a comparison between the Desert Tortoise Research Natural Area (DTNA) and unprotected land immediately adjacent. The DTNA was fenced to prohibit both OHV use and sheep grazing between 1978 and 1979. By the time of his study in 1990–1992, Brooks demonstrated that aboveground live annual biomass was generally greater inside than outside the fenced area, with the exception that the exotic annual grass *Schismus barbatus* produced more biomass outside the fenced area. Percent cover of perennial shrubs, seed biomass, and rodent density and diversity were also greater inside the fenced area.

To be successful, revegetation and restoration require careful attention to ecological relationships, both above and below ground, herbivory, soil characteristics, microclimate, and patterns of moisture availability (Bainbridge 1990, Bainbridge and others 1995a). Undoing the damage done to the soil system by disturbance is a critical step toward recovery and restoration. In general, strategies that recreate or mimic natural conditions are most likely to speed recovery of the entire ecosystem.

Research conducted in the Mojave and Colorado desert ecosystem has important applications for the American Southwest and throughout the world's arid zones. These areas have deteriorated rapidly under pressure from overgrazing, poor farming, and removal of trees and shrubs for fuelwood. The lessons learned in the desert ecosystem of southern California may help people living in these areas to protect or restore the productivity of their lands, and improve their lives.

Acknowledgments

We gratefully acknowledge the following people for their assistance and comments during the preparation of this manuscript Hal Avery, Kristin Berry, Bill Boarman, David Cole, Richard Franklin, June Latting, Michael Liddle, Mike Mitchell, and Howard Wilshire. Research was supported by a grant from the California Department of Parks and Recreation, Off Highway Motor Vehicle Recreation Division. This paper is dedicated to the memory of June Latting.

Literature Cited

Allen, E. B. 1982a. Germination and competition of *Salsola kali* with native C3 and C4 species under three temperature regimes. *Bulletin of the Torrey Botanical Club* 109:39–46.

Allen, E. B. 1982b. Water and nutrient competition between *Salsola kali* and two native grass species (*Agropyron smithii* and *Bouteloua gracilis*). *Ecology* 63:732–741.

Anderson, D. C., K. T. Harper, and S. R. Rushforth. 1982. Recovery of cryptogamic soil crusts from grazing on Utah winter ranges. *Journal of Range Management* 35:355–359.

Anonymous. 1991. Society for Range Management: desert tortoise report. *Renewable Resources Journal* 9:30.

Archer, S., and F. E. Smeins. 1991. Ecosystem-level processes. Pages 109–139 *in* R. K. Heitschmidt and J. W. Stuth (eds.), Grazing Management an Ecological Perspective. Timber Press, Portland, Oregon.

Artz, M. C. 1989. Impacts of linear corridors on perennial vegetation in the East Mojave Desert: implications for environmental management and planning. *Natural Areas Journal* 9:117–129.

Bainbridge, D. A. 1990. The restoration of agricultural lands and drylands. Pages 4–13 *in* J. J. Berger (ed.), Environmental restoration: science and strategies for restoring the earth. Island Press, Washington, DC.

Bainbridge, D. A., and R. A. Virginia. 1990. Restoration in the Sonoran desert of California. *Restoration and Management Notes* 8:3–13.

Bainbridge, D. A., R. McAller, M. Fidelibus, R. Franson, A. C. Williams, and L. Lippitt. 1995a. A beginners guide to desert restoration. National Park Service, Denver Service Center, 34 pp.

Bainbridge, D. A., M. Fidelibus, and R. McAller. 1995b. Techniques for plant establishment in arid ecosystems. *Restoration and Management Notes* 13:190–197.

Balph, D. F., and J. C. Malechek. 1985. Cattle trampling of crested wheatgrass under short-duration grazing. *Journal of Range Management* 38:226–227.

Belnap, J. 1991. Sensitivity of desert cryptogams to air pollutants: soil crusts and rock lichens. Pages 112–119 *in* D. Mangis, J. Baron and K. Stolte (eds.), Acid rain and air pollution in desert park areas. National Park Service, United States Department of the Interior.

Belnap, J. 1993. Recovery potential of cryptobiotic crusts: inoculant use and assessment methods. *Great Basin Naturalist* 53:89–95.

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

Belnap, J., and J. S. Gardner. 1993. Soil microstructure in soils of the Colorado Plateau: the role of the cyanobacterium *Microcoleus vaginatus*. *Great Basin Naturalist* 53:40–47.

Bentley, H. L. 1898. Cattle ranges of the southwest: a history of the exhaustion of the pasturage and suggestions for its restoration. US Department of Agriculture, Farmers Bulletin 72, pp. 1–31.

Bethlenfalvay, G. J., S. Dakessian, and R. S. Pacovsky. 1984. Mycorrhizae in a southern California desert: ecological implications. *Canadian Journal of Botany* 62:519–524.

Bloss, H. E. 1985. Studies of symbiotic microflora and their role in the ecology of desert plants. *Desert Plants* 7:119–127.

Boarman, W. I. 1993. When a native predator becomes a pest: A case study. Pages 191–206 *in*: S. K. Majumdar, E. W. Miller, D. E. Baker, E. K. Brown, J. R. Pratt, and R. F. Schmalz (eds.), Conservation and Resource Management. The Pennsylvania Academy of Science, Easton, Pennsylvania.

Boarman, W. I., and M. Sazaki. 1996. Highway mortality in desert tortoises and small vertebrates: success of barrier fences and culverts. Pages 169–173 *in* G. J. Evink, P. Garrett, D. Zeigler, and J. Berry (eds.), Trends in addressing transportation related wildlife mortality. Proceedings of the transportation related wildlife mortality seminar. Environmental Management Office, Department of Transportation, Tallahassee, Florida.

Borman, M. M., and D. E. Johnson. 1990. Evolution of grazing and land tenure policies on public lands. *Rangelands* 12:203–206.

Bostick, V. 1990. The desert tortoise in relation to cattle grazing. *Rangelands* 12:149–151.

Bowers, J. E., R. H. Webb, and R. J. Rondeau. 1995. Longevity, recruitment and mortality of desert plants in Grand Canyon, Arizona, USA. *Journal of Vegetation Science* 6:551–564.

Bowers, J. E., R. H. Webb, and E. A. Pierson. 1997. Succession of desert plants on debris flow terraces, Grand Canyon, Arizona, USA. *Journal of Arid Environments* 36:67–86.

Brattstrom, B. H., and M. C. Bondello. 1983. Effects of off-road vehicle noise on desert vertebrates. Pages 167–206 *in* R. H. Webb and H. G. Wilshire (eds.), Environmental effects of off-road vehicles: impacts and management in arid regions. Springer-Verlag, New York.

Brooks, M. L. 1995. Benefits of protective fencing to plant and rodent communities of the western Mojave Desert, California. *Environmental Management* 19:65–74.

Brotherson, J. D., S. R. Rushforth, and J. R. Johansen. 1983. Effects of long-term grazing on cryptogam crust cover in Navajo National Monument, Ariz. *Journal of Range Management* 36:579–581.

Brown, D. E., and R. A. Minnich. 1986. Fire and changes in creosote bush scrub of the western Sonoran Desert, California. *American Midland Naturalist* 116:411–422.

Brum, G. D., R. S. Boyd, and S. M. Carter. 1983. Recovery rates and rehabilitation of powerline corridors. Pages 303–314 *in* R. H. Webb and H. G. Wilshire (eds.), Environmental effects of off-road vehicles: impacts and management in arid regions. Springer-Verlag, New York.

Bureau of Land Management. 1980. The California desert conservation area plan. US Department of the Interior, Riverside, California. 173 pp.

Burk, J. H. 1977. Sonoran desert. Pages 869–889 *in* M. G. Barbour and J. Major (eds.), Terrestrial Vegetation of California. John Wiley & Sons, New York.

Bury, R. B., and R. A. Luckenbach. 1983. Vehicular recreation in arid land dunes: biotic responses and management alternatives. Pages 207–221 *in* R. H. Webb, and H. G. Wilshire (eds.), Environmental effects of off-road vehicles: impacts and management in arid regions. Springer-Verlag, New York.

Bury, R. B., and R. A. Luckenbach. 1986. Abundance of desert tortoises (*Gopherus agassizii*) in natural and disturbed habitats. Unpublished draft report. US Fish and Wildlife Service.

Busack, S. D., and R. B. Bury. 1974. Some effects of off-road vehicles and sheep grazing on lizard populations in the Mojave Desert. *Biological Conservation* 6:179–183.

Charley, J. L., and S. W. Cowling. 1968. Changes in soil nutrient status resulting from overgrazing and their consequences in plant communites of semi-arid areas. *Proceedings of the Ecological Society of Australia* 3:28–38.

Cheater, M. 1992. Alien invasion. *Nature Conservancy* September/October:24–29.

Chou, Y.-H., R. A. Minnich, L. A. Salazar, J. D. Power, and R. J. Dezzani. 1990. Spatial autocorrelation of wildfire distribution in the Idyllwild quadrangle, San Jacinto Mountain, California. *Photogrammetric Engineering and Remote Sensing* 56:1507–1513.

Clark, D. R., Jr., and R. L. Hothem. 1991. Mammal mortality at Arizona, California, and Nevada gold mines using cyanide extraction. *California Fish and Game* 77:61–69.

Cody, M. L. 1986. Spacing patterns in Mojave Desert plant communities: Near-neighbor analyses. *Journal of Arid Environments* 11:199–217.

Coe, M. 1990. The conservation and management of semi-arid rangelands and their animal resources. Pages 219–249 *in* A. S. Goudie (ed.), Techniques for desert reclamation. John Wiley & Sons, Chichester.

Cole, D. N. 1990. Trampling disturbance and recovery of cryptogamic soil crusts in Grand Canyon National Park. *Great Basin Naturalist* 50:321–325.

D'Antonio, C., and T. L. Dudley. 1993. Alien species. *Pacific Discovery* Summer:9–11.

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

Dregne, H. E. 1983. Soil and soil formation in arid regions. Pages 15–30 *in* R. H. Webb and H. G. Wilshire (eds.), Environmental effects of off-road vehicles: impacts and management in arid regions. Springer-Verlag, New York.

Elvidge, C. D., and R. M. Iverson. 1983. Regeneration of desert pavement and varnish. Pages 225–243 *in* R. H. Webb and H. G. Wilshire (eds.) Environmental effects of off-road vehicles: impacts and management in arid regions. Springer-Verlag, New York.

Field, T. G. 1990. The future of grazing on public lands. *Rangelands* 12:217–219.

Fisher, J. C., Jr. 1978. Studies relating to the accelerated mortality of *Atriplex hymenelytra* in Death Valley National Monument. Unpublished Masters thesis. University of California, Riverside.

Garland, T. J., and W. G. Bradley. 1984. Effects of a highway on Mojave Desert rodent populations. *American Midland Naturalist* 111:47–56.

General Accounting Office. 1992. Rangeland management: BLM's hot desert grazing program merits reconsideration. United States General Accounting Office. GAO/RCED-92-12, Washington, DC.

Genin, D., and A. Badan-Dangon. 1991. Goat herbivory and plant phenology in a Mediterranean shrubland of northern Baja California. *Journal of Arid Environments* 21:113–121.

Germano, D. J., and C. R. Hungerford. 1981. Reptile population changes with manipulation of Sonoran Desert shrub. *Great Basin Naturalist* 41:129–137.

Germano, D. J., and D. N. Lawhead. 1986. Species diversity and habitat complexity: does vegetation organize vertebrate communities in the Great Basin? *Great Basin Naturalist* 46:711–719.

Germano, D. J., R. Hungerford, and Martin S. Clark. 1983. Responses of selected wildlife species to the removal of mesquite from desert grassland. *Journal of Range Management* 36:309–311.

Gill, T. E. 1996. Eolian sediments generated by anthropogenic disturbance of playas: Human impacts on the geomorphic system and geomorphic impacts on the human system. *Geomorphology* 17:207–228.

Gillette, D. A., and J. Adams. 1983. Accelerated wind erosion and prediction of rates. Pages 97–109 *in* R. H. Webb and H. G. Wilshire (eds.), Environmental effects of off-road vehicles: impacts and management in arid regions. Springer-Verlag, New York.

Gillis, A. M. 1991. Should cows chew cheatgrass on commonlands? *Bioscience* 41:668–675.

Graves, W. L., B. L. Kay, and W. A. Williams. 1978. Revegetation of disturbed sites in the Mojave Desert with native shrubs. *California Agriculture* March:4–5.

Harding, M. V. 1990. Erosion control effectiveness: Comparative studies of alternative mulching techniques. Pages 149–156 *in* J. J. Berger (ed.), Environmental restoration: Science and strategies for restoring the earth. Island Press, Washington, DC.

Heede, B. H. 1983. Control of rills and gullies in off-road vehicle traffic areas. Pages 245–264 *in* R. H. Webb and H. G. Wilshire (eds.), Environmental effects of off-road vehicles: impacts and management in arid regions. Springer-Verlag, New York.

Henny, C. J., R. J. Hallock, and E. F. Hill. 1994. Cyanide and migratory birds at gold mines in Nevada, USA. *Ecotoxicology* 3:45–58.

Heske, E. J., and M. Campbell. 1991. Effects of an 11-year livestock exclosure on rodent and ant numbers in the Chihuahuan desert, southeastern Arizona. *Southwestern Naturalist* 36:89–93.

Hessing, M. B., and C. D. Johnson. 1982. Disturbance and revegetation of Sonoran desert vegetation in an Arizona powerline corridor. *Journal of Range Management* 35:254–258.

Hickman, J. C. (ed.). 1993. The Jepson manual: Higher plants of California. University of California Press, Berkeley. 1400 pp.

Hinckley, B. S., R. M. Iverson, and B. Hallet. 1983. Accelerated water erosion in ORV use areas. Pages 81–96 *in* R. H. Webb and H. G. Wilshire (eds.), Environmental effects of off-road vehicles: impacts and management in arid regions. Springer-Verlag, New York.

Humphrey, R. R. 1958. The desert grassland: A history of vegetational change and an analysis of causes. *Botanical Review* 24:193–252.

Humphrey, R. R. 1963. The role of fire in the desert and desert grassland areas of Arizona. Pages 45–62 *in* Proceedings, 2nd annual Tall Timbers fire ecology conference. Tall Timbers Research Station, Tallahasee, Florida.

Humphrey, R. R. 1974. Fire in the deserts and desert grassland of North America. Pages 365–400 *in* T. T. Kozlowski and C. E. Ahlgren (eds.), Fire and ecosystems. Academic Press, New York.

Humphrey, R. R. 1987. 90 years and 535 miles: vegetation changes along the Mexican border. University of New Mexico Press, Albuquerque. 448 pp.

Hunter, R. 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.

Hunter, R., F. B. Turner, R. G. Lindberg, and K.-B. Hunter. 1987. Effects of land clearing on bordering winter annual populations in the Mohave Desert. *Great Basin Naturalist* 47:234–238.

Jeffries, D. L., and J. M. Klopatek. 1987. Effects of grazing on the vegetation of the blackbrush association. *Journal of Range Management* 40:390–392.

Jennings, B. W. 1991. Desert tortoise carcass surveys along State Highways 58 and 395 San Bernardino Co., California. Prepared for the Bureau of Land Management, Riverside, California.

Johansen, J. R. 1993. Cryptogamic crusts of semiarid and arid lands of North America. *Journal of Phycology* 29:140–147.

Johnson, H. B., F. C. Vasek, and T. Yonkers. 1975. Productivity, diversity and stability relationships in Mojave Desert roadside vegetation. *Bulletin of the Torrey Botanical Club* 102:106–115.

Johnson, H. B., F. C. Vasek, and T. Yonkers. 1978. Residual effects of summer irrigation on Mojave Desert annuals. *Bulletin of the Southern California Academy of Science* 77:95–108.

Johnson, J. R., J. R. Beley, T. M. Ditsworth, and S. M. Butt. 1983. Secondary succession of arthropods and plants in the Arizona Sonoran desert in response to transmission line construction. *Journal of Environmental Management* 16:125–137.

Kay, B. L. 1979. Summary of revegetation attempts on the second Los Angeles aqueduct. *Mojave Revegetation Notes* 22:1–23.

Kay, B. L. 1988. Artificial and natural revegetation of the second Los Angeles aqueduct. *Mojave Revegetation Notes* 24:1–32.

Kay, B. L., and W. L. Graves. 1983. History of revegetation studies in the California deserts. Pages 315–324 *in* R. H. Webb, and H. G. Wilshire (eds.), Environmental effects of off-road vehicles: impacts and management in arid regions. Springer-Verlag, New York.

Kleiner, E. F., and K. T. Harper. 1977. Soil properties in relation to cryptogamic groundcover in Canyonlands National Park. *Journal of Range Management* 30:202–205.

Kockelman, W. J. 1983. Introduction. Pages 1–11 *in* R. H. Webb and H. G. Wilshire (eds.), Environmental effects of off-road vehicles: impacts and management in arid regions. Springer-Verlag, New York.

Lathrop, E. W. 1983a. Recovery of perennial vegetation in military maneuver areas. Pages 265–277 *in* R. H. Webb and H. G. Wilshire (eds.), Environmental effects of off-road vehicles: impacts and management in arid regions. Springer-Verlag, New York.

Lathrop, E. W. 1983b. The effect of vehicle use on desert vegetation. Pages 154–166 *in* R. H. Webb and H. G. Wilshire (eds.), Environmental effects of off-road vehicles: impacts and management in arid regions. Springer-Verlag, New York.

Lathrop, E. W., and E. F. Archbold. 1980a. Plant response to Los Angeles aqueduct construction in the Mojave Desert. *Environmental Management* 4:137–148.

Lathrop, E. W., and E. F. Archbold. 1980b. Plant responses to utility right of way construction in the Mojave Desert. *Environmental Management* 4:215–226.

Lathrop, E. W., and P. G. Rowlands. 1983. Plant ecology in deserts: an overview. Pages 113–152 *in* R. H. Webb and H. G. Wilshire (eds.), Environmental effects of off-road vehicles:

impacts and management in arid regions. Springer-Verlag, New York.

Liddle, M. J. 1991. Recreation ecology: effects of trampling on plants and coral. *Trends in Evolution and Ecology* 6:13–17.

Liddle, M. J. 1997. Recreation ecology. Chapman and Hall, London, 639 pp.

Lightfoot, D. C., and W. G. Whitford. 1991. Productivity of creosotebush foliage and associated canopy arthropods along a desert roadside. *American Midland Naturalist* 125:310–322.

Lodhi, M. A. K. 1979. Allelopathic potential of *Salsola kali* L. and its possible role in rapid disappearance of weedy stage during revegetation. *Journal of Chemical Ecology* 5:429–437.

Lovich, J. E., and R. G. de Gouvenain. 1998. Saltcedar invasion in desert wetlands of the southwestern United States: ecological and political implications. Pages 447–467 *in* S. K. Majumdar, E. W. Miller, and F. J. Brenner (eds.), Ecology of wetlands and associated systems. Pennsylvania Academy of Science, Easton, Pennsylvania.

Mangis, D., J. Baron, and K. Stolte. 1991. Acid rain and air pollution in desert park areas. National Park Service, United States Department of the Interior. 129 pp.

Middleton, N. J. 1990. Wind erosion and dust-storm control. Pages 87–108 *in* A. S. Goudie (ed.), Techniques for desert reclamation. John Wiley & Sons, Chichester.

Minnich, R. A. 1983. Fire mosaics in southern California and northern Baja California. *Science* 219:1287–1294.

Mooney, H. A., S. P. Hamburg, and J. A. Drake. 1986. The invasions of plants and animals into California. Pages 250–272 *in* H. A. Mooney and J. A. Drake (eds.), Ecology of biological invasions of North America and Hawaii. Springer-Verlag, New York.

Nakata, J. K. 1983. Off-road vehicular destabilization of hill slopes: the major contributing factor of destructive debris flows in Ogden, Utah, 1979. Pages 343–353 *in* R. H. Webb and H. G. Wilshire (eds.), Environmental effects of off-road vehicles: impacts and management in arid regions. Springer-Verlag, New York.

Nakata, J. K., H. G. Wilshire, and G. C. Barnes. 1976. Origin of Mojave Desert dust plumes photographed from space. Geology 4:644–648.

Nicholson, L. 1978. The effects of roads on desert tortoise populations. Pages 127–129 *in* M. Trotter (ed.), Proceedings of the 1978 symposium of the Desert Tortoise Council.

Nicholson, L., and K. Humphreys. 1981. Sheep grazing at the Kramer study plot, San Bernardino County, California. Pages 163–194 *in* K. A. Hashagen (ed.), Proceedings of the 1981 symposium of the Desert Tortoise Council.

Odum, E. P. 1994. Conservation of biodiversity. Page 18–25 *in* S. K. Majumdar, F. J. Brenner, J. E. Lovich, E. W. Miller, and J. F. Schalles (eds.), Biological diversity: problems and challenges. Pennsylvania Academy of Science, Easton, Pennsylvania.

Oldemeyer, J. L. 1994. Livestock grazing and the desert tortoise in the Mojave Desert. Pages 95–103 *in* R. B. Bury and D. J. Germano (eds.), Biology of North American tortoises. National Biological Survey, Washington, DC.

O'Leary, J. F., and R. A. Minnich. 1981. Postfire recovery of creosote bush scrub vegetation in the western Colorado Desert. *Madroño* 23:61–66.

Omar, S. A. S. 1991. Dynamics of range plants following 10 years of protection in arid rangelands of Kuwait. *Journal of Arid Environments* 21:99–111.

Poling, M. A. 1991. Legal milestones in range management. *Renewable Resources Journal* 9:7–10.

Prose, D. V. 1985. Persisting effects of armored military maneuvers on some soils of the Mojave Desert. *Environmental Geology and Water Sciences* 7:163–170.

Prose, D. V., and S. K. Metzger. 1985. Recovery of soils and vegetation in World War II military base camps, Mojave Desert. US Geological Survey, Open File Report 85-234. 114 pp.

Prose, D. V., S. K. Metzger, and H. G. Wilshire. 1987. Effects of substrate disturbance on secondary plant succession; Mojave Desert, California. *Journal of Applied Ecology* 24:305–313.

Pryor, S. C., and T. E. Hoffer. 1991. A case study of pollutant transport from Los Angeles to the desert south-west. *Atmospheric Environment* 26A:243–250.

Reible, D. D., J. R. Ouimette, and F. H. Shair. 1982. Atmospheric transport of visibility degrading pollutants into the California Mojave Desert. *Atmospheric Environment* 16:599–613.

Rejmanek, M. and J. M. Randall. 1994. Invasive alien plants in California: 1993 summary and comparison with other areas in North America. *Madroño* 41:161–177.

Reynolds, H. G., and J. W. Bohning. 1956. Effects of burning on a desert grass-shrub range in southern Arizona. *Ecology* 37:769–777.

Rice, B., and M. Westoby. 1978. Vegetative responses of some Great Basin shrub communities protected against jack rabbits or domestic stock. *Journal of Range Management* 31:28–34.

Romney, E. M., R. B. Hunter, and Wallace, A. 1990. Field trip report: natural and managed recovery of vegetation on disturbed areas at the Nevada Test Site. Pages 344–349 *in* E. D. McArthur and others (eds.), Proceedings, symposium on cheatgrass invasion, shrub die-off, and other aspects of shrub biology and management. US Forest Service, Intermountain Research Station, Ogden, Utah.

Rosen, P. C., and C. H. Lowe. 1994. Highway mortality of snakes in the Sonoran desert of southern Arizona. *Biological Conservation* 68:143–148.

Roundy, B. A., and G. L. Jordan. 1988. Vegetation changes in relation to livestock exclusion and rootplowing in southeastern Arizona. *Southwestern Naturalist* 33:425–436.

Ruch, J. B. 1968. The California Desert. California State Office BLM and Western Regional Office National Park Service, Sacramento, California, 377 pp.

Rundel, P. W., and A. C. Gibson. 1996. Ecological communities and processes in the Mojave Desert ecosystem: Rock Valley, Nevada. Cambridge University Press, Cambridge, UK 369 pp.

Savory, A. 1988. Holistic resource management. Island Press, Washington, DC, 564 pp.

Schlesinger, W. H., P. J. Fonteyn, and W. A. Reiner. 1989. Effects of overland flow on plant water relations, erosion, and soil water percolation on a Mojave Desert landscape. *Soil Science Society of America Journal* 53:1567–1572.

Schlesinger, W. H., J. F. Reynolds, G. L. Cunningham, L. F. Huenneke, W. M. Jarrell, R. A. Virginia, and W. G. Whitford.

1990. Biological feedbacks in global desertification. *Science* 247:1043–1048.

St. John, T. V. 1984. Mycorrhizal fungi and revegetation. Pages 87–93 *in* J. P. Reiger and B. A. Steele (eds.). Proceedings of the native plant revegetation symposium. Southern California Chapter and San Diego Chapter of the California Native Plant Society.

Stolte, K. W. 1991. Sensitivity of plant ecosystems in desert areas to gaseous pollutants. Pages 42–51 *in* D. Mangis, J. Baron, and K. Stolte (eds.). Acid rain and air pollution in desert park areas. National Park Service, United States Department of the Interior.

Taylor-George, S., F. Palmer, J. T. Staley, D. J. Borns, B. Curtiss, and J. B. Adams. 1983. Fungi and bacteria involved in desert varnish formation. *Microbial Ecology* 9:227–245.

Thompson, C. R. 1995. Air pollution effects on desert plants. Pages 481–488 *in* J. Latting, and P. G. Rowlands (eds.). The California desert: An introduction to natural resources and man's impact, Vol. II. June Latting Books, Riverside, California.

Thompson, C. R., G. Kats, and R. W. Lennox. 1980. Effects of SO_2 and/or NO_2 on native plants of the Mojave Desert and eastern Mojave-Colorado Desert. *Journal of the Air Pollution Control Association* 30:1304–1309.

Thompson, C. R., G. K. Olszyk, A. Bytnerowicz, P. J. Dawson, and J. Wolf. 1984a. Air pollutant injury on plants of the Mojave Desert. Southern California Edison Company, Rosemead, California. 31 pp.

Thompson, C. R., D. M. Olszyk, G. Kats, A. Bytnerowicz, P. J. Dawson, and J. W. Wolf. 1984b. Effects of ozone and sulfur dioxide on annual plants of the Mojave Desert. *Journal of the Air Pollution Control Association* 34:1017–1022.

Tierra Madre Consultants. 1991. Biological assessment for Lancaster City and planning area: Relative density surveys for desert tortoises and cumulative human impact evaluations for Mohave ground squirrel habitat. Riverside, California.

Tratz, W. M., and R. J. Vogl. 1977. Postfire vegetational recovery, productivity, and herbivore utilization of a chaparral-desert ecotone. Pages 426–430 *in* H. A. Mooney and C. E. Conrad (eds.). Proceedings of the symposium on the environmental consequences of fire and fuel management in Mediterranean ecosystems. Forest Service, US Department of Agriculture, Washington, DC.

Trijonis, J. 1979. Visibility in the Southwest: An exploration of the historical data base. *Atmospheric Environment* 13:833–843.

Turner, R. M. 1982. Mojave desertscrub. *Desert Plants* 4:157–168.

VanCuren, T. 1995. Air pollution in the California desert: A planning perspective. Pages 489–510 *in* J. Latting and P. G. Rowlands (eds.). The California desert: An introduction to natural resources and man's impact, Vol. II. June Latting Books, Riverside, California.

Vasek, F. C. 1980. Creosote bush: long-lived clones in the Mojave Desert. *American Journal of Botany* 67(2):246–255.

Vasek, F. C., and M. G. Barbour. 1977. Mojave Desert scrub vegetation. Pages 835–867 *in* M. G. Barbour and J. Major (eds.). Terrestrial vegetation of California. John Wiley & Sons, New York.

Vasek, F. C., H. B. Johnson, and D. H. Eslinger. 1975a. Effects of pipeline construction on creosote bush scrub vegetation of the Mojave Desert. *Madroño* 23(1):1–13.

Vasek, F. C., H. B. Johnson, and G. D. Brum. 1975b. Effects of power transmission lines on vegetation of the Mojave Desert. *Madroño* 23:114–131.

Vollmer, A. T., B. G. Maza, P. A. Medica, F. B. Turner, and S. A. Bamberg. 1976. The impacts of off-road vehicles on a desert ecosystem. *Environmental Management* 1:115–129.

Walsh, P. A., and T. E. Hoffer. 1991. The changing environment of a desert boomtown. *The Science of the Total Environment* 105:233–258.

Webb, R. H. 1982. Off-road motorcycle effects on a desert soil. *Environmental Conservation* 9:197–208.

Webb, R. H. 1983. Compaction of desert soils by off-road vehicles. Pages 51–79 *in* R. H. Webb and H. G. Wilshire (eds.). Environmental effects of off-road vehicles: Impacts and management in arid regions. Springer-Verlag, New York.

Webb, R. H., and E. B. Newman. 1982. Recovery of soil and vegetation in ghost-towns in the Mojave Desert, southwestern United States. *Environmental Conservation* 9:245–248.

Webb, R. H., and S. S. Stielstra. 1979. Sheep grazing effects on Mojave Desert vegetation and soils. *Environmental Management* 3:517–529.

Webb, R. H., and H. G. Wilshire. 1983. Environmental effects of off-road vehicles: Impacts and management in arid regions. Springer-Verlag, New York. 534 pp.

Webb, R. H., H. G. Wilshire, and M. A. Henry. 1983. Natural recovery of soils and vegetation following human disturbance. Pages 279–302 *in* R. H. Webb and H. G. Wilshire (eds.) Environmental effects of off-road vehicles: Impacts and management in arid regions. Springer-Verlag, New York.

Webb, R. H., J. W. Steiger, and R. M. Turner. 1987. Dynamics of Mojave Desert shrub assemblages in the Panamint Mountains, California. *Ecology* 68:478–490.

Webb, R. H., J. W. Steiger, and E. B. Newman. 1988. The response of vegetation to disturbance in Death Valley National Monument, California. *US Geological Survey Bulletin* 1793, 103 pp.

Wedin, D. A., and D. Tilman. 1996. Influence of nitrogen loading and species composition on the carbon balance of grasslands. *Science* 274:1720–1723.

Wilshire, H. G. 1983. The impact of vehicles on desert soil stabilizers. Pages 31–50 *in* R. H. Webb and H. G. Wilshire (eds.). Environmental effects of off-road vehicles: Impacts and management in arid regions. Springer-Verlag, New York.

Wilshire, H., and D. Prose. 1987. Wind energy development in California, USA. *Environmental Management* 11:13.

Wood, M. K., R. E. Eckert, Jr., W. H. Blackburn, and F. F. Peterson. 1982. Influence of crusting soil surfaces on emergence and establishment of crested wheatgrass, squirreltail, Thurber needlegrass, and fourwing saltbush. *Journal of Range Management* 35:282–287.

Young, J. A. 1991. Tumbleweed. *Scientific American* 264:82–87.

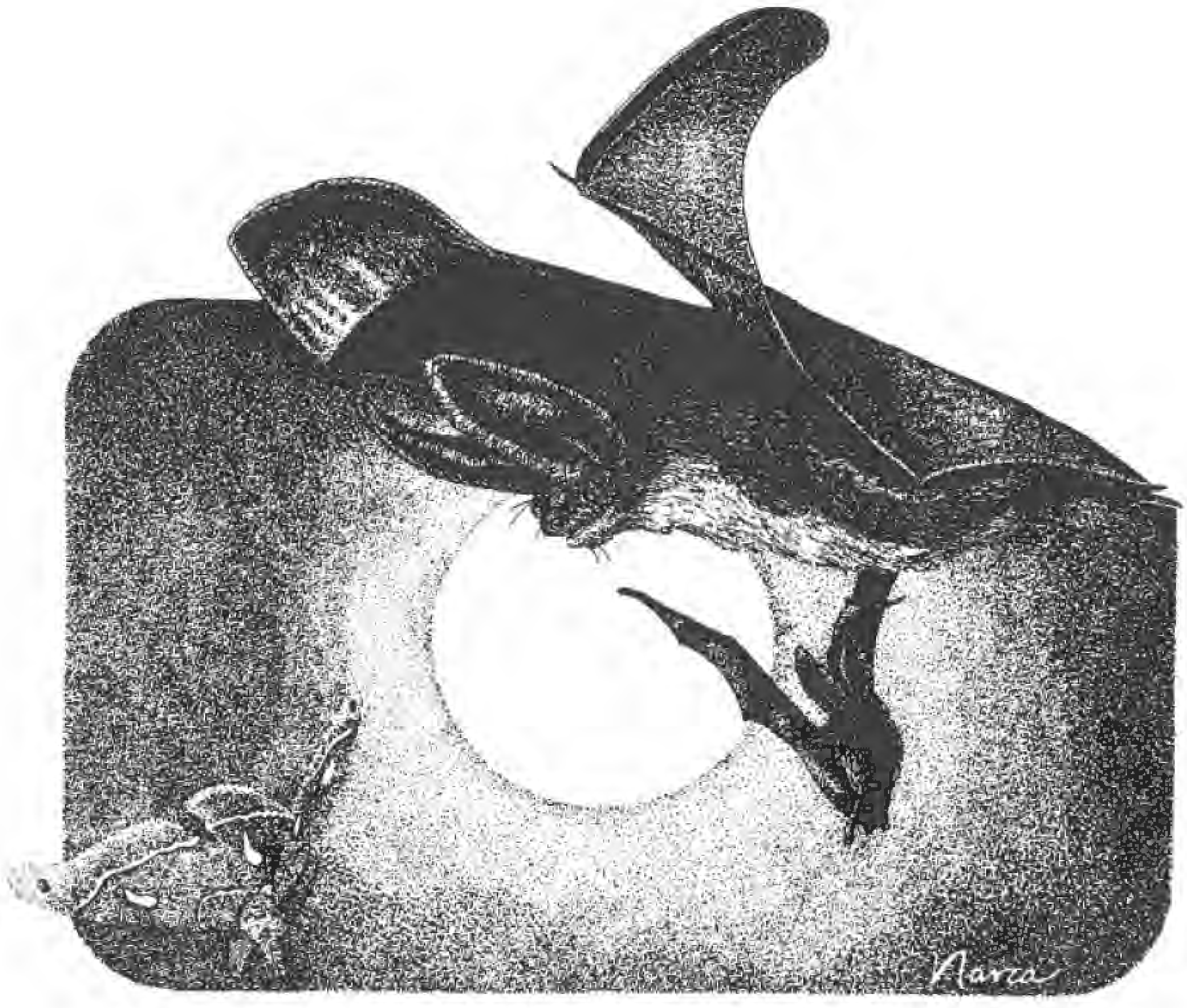
Zink, T. A., M. F. Allen, B. Heindl-Tenburen, and E. B. Allen. 1995. The effect of a disturbance corridor on an ecological reserve. *Restoration Ecology* 3:304–310.

STATE OF CALIFORNIA
THE RESOURCES AGENCY
DEPARTMENT OF FISH AND GAME

MAMMALIAN SPECIES OF SPECIAL CONCERN IN CALIFORNIA

by

Daniel F. Williams
Department of Biological Sciences
California State University, Stanislaus
Turlock, California 95380



Cover: Townsend's Big-eared Bat (Plecotus townsendii)

Page 11: Suisun Shrew (Sorex ornatus sinuosus)

Page 49: White-eared Pocket Mouse (Perognathus alticola)

Page 72: Mountain Beaver (Aplodontia rufa)
Fisher (Martes pennanti)

Artwork by Narca Moore-Craig

STATE OF CALIFORNIA
THE RESOURCES AGENCY
DEPARTMENT OF FISH AND GAME

MAMMALIAN SPECIES OF SPECIAL CONCERN IN CALIFORNIA[1/]

by

Daniel F. Williams
Department of Biological Sciences
California State University, Stanislaus
Turlock, California 95380

ABSTRACT

The native species of land mammals of California which currently do not have state or federal Threatened or Endangered Species status were investigated in order to identify those potentially threatened with extinction. Investigations concentrated on determining historic and current distributions, habitat associations, population status, and the nature and proximity of threats of extinction. Information was developed primarily from the literature, museum records, and field notes, and from contacts with biologists with knowledge of current developments in the field. Detailed studies were conducted in some areas, but only cursory field work was undertaken in other areas of concern. Populations of 36 species and subspecies were considered to be potentially jeopardized. These are placed in three priority categories. The 13 taxa in the Highest Priority face a high probability of extinction if current trends continue; the 11 taxa in the Second Priority are definitely declining in population size and appear jeopardized, but the threats are less immediate; the 12 taxa in the Third Priority appear not to face extinction soon, but their populations are declining seriously or they are otherwise highly vulnerable to human developments. Information on distribution, population status, habitat, and taxonomy, and recommendations for management actions are presented for each species on the List of Concern. Brief remarks are included for 56 other taxa considered in developing the final List of Concern.

Species limited to or primarily dependent upon riparian and wetland communities have been affected most severely by human developments. Five geographic areas of critical concern are: the Colorado River riparian corridor; the San Joaquin Valley lowlands, including grassland, riparian and wetland communities; the tidal marshes of the Los Angeles Basin; the tidal marshes of San Francisco and San Pablo bays; and the grasslands of the southern California coastal basins. Loss and fragmentation of mature and old-growth forests, lack of data on population structure of some game and fur-bearing species, and human disturbances of sensitive species are other important factors generating concerns for several species.

1/ Wildlife Management Division Administrative Report 86-1 (June 1986). Supported by federal Endangered Species Act grant-in-aid funds for Nongame Bird and Mammal Section project E-W-4, IV-14.1; and by internal Nongame Bird and Mammal Section research funds.

RECOMMENDATIONS

In addition to recommendations contained in the species accounts for the preservation of California's mammals, the California Department of Fish and Game and I recommend the following:

1. Give high priority to the preservation and/or restoration of plant communities essential to wildlife:

   a. Restore and protect riparian forests and wetlands in California, with special attention to those of the Colorado River and the San Joaquin Valley.

   b. Restore and protect tidal wetlands, especially those in San Francisco, San Pablo, and Suisun bays and those along the southern California coast in Ventura, Los Angeles, and Orange counties.

   c. Preserve and protect native grasslands and desert shrub communities in the San Joaquin Valley, Salinas Valley, and the southern California coastal basins in Los Angeles, Orange, Riverside, and San Diego counties.

   d. Preserve and protect mature and old-growth conifer forests in blocks large enough to support species such as Fishers.

2. Propose species on the List of Special Concern that meet the criteria of Threatened or Endangered Species to the California Fish and Game Commission and the U.S. Fish and Wildlife Service for addition to the lists of Threatened and Endangered Wildlife.

3. Initiate programs to determine the effects of hunting and trapping on game and furbearing species on the List of Special Concern and modify regulations as appropriate.

4. Encourage the protection of all species of bats in California and initiate an educational program to inform the public of the role of bats in control of insects and the sensitivity of bats to disturbance in maternity roosts and hibernacula. Support and assist the development of regulations prohibiting the poisoning or killing of bats as control measures in human structures.

5. Encourage governmental, educational, and conservation agencies and institutions involved in wildlife, land, and resource management to give high priority to Species of Special Concern in research programs and land and resource management decisions.

6. Encourage persons with information on Species of Special Concern or other species that may be threatened to bring the information to the attention of the Department of Fish and Game. Revise the List of Special concern every two years to reflect current information on distribution, population status, and management recommendations.

PREFACE

The primary objectives in preparing this document were to identify taxa of mammals in California that had no status as Endangered, Threatened, or Fully-protected, but which appeared to be vulnerable to extinction, and to develop a set of priorities for determining their status and ensuring their survival. As originally conceived and implemented, the project provided no resources for field investigations, although most areas of the state were visited and limited field work was conducted. In the ensuing five years, however, opportunities to conduct more extensive field work in several areas have arisen and the investigations have resulted in removing several species from the draft List of Concern, moving others to lower categories, and elevating others to higher categories. Three species included on the final list were nor investigated in the same detail as others, because in the early stages of the project I had decided there were no indications that they were in jeopardy. Subsequent to preparation of the draft final report, however, reconsideration of their status has resulted in their inclusion. I thought it better to include them with only partial data available rather than to delay the preparation of the final report.

A rough draft of the accounts of 52 species and subspecies to be included in this report was prepared and submitted for comment to the California Department of Fish and Game in 1981. A completed draft of the report was submitted later in 1981. The Department of Fish and Game finished its review and returned the draft to me for final revisions in June, 1984. By the time it was returned, considerable new information had been gathered for several species, and substantial revisions were envisioned; in addition, I had incurred a number of commitments that precluded work on the document until fall of 1985. In the interest of making the information that was gathered for the original report available, I have decided reluctantly to forego major revisions. Most sections of the report have been reorganized, 18 species have been deleted from the List of Concern (they are discussed in the section entitled "other candidate species"), three species have been added to the list of concern, and some new information, gathered during subsequent field work by me and others, has been incorporated.

23 February 1986
Turlock, California

TABLE OF CONTENTS

INTRODUCTION

During the past 150 years, 515 species and subspecies of native land mammals have been recorded as occurring in California (Hall, 1981; Williams, 1984), a number much greater than in any comparable political unit north of the American tropics. Seven species, including the Mexican Jaguar (Felis onca arizonensis), Plains Bison (Bison bison bison), Gray Wolf (Canis lupus fuscus and C. l. youngi), California Grizzly Bear (Ursus arctos californicus; taxonomy of Hall, 1984), and the Western White-tailed Deer (Odocoileus virginianus ochrourus), are now extinct or extirpated within California. Coues' White-tailed Deer (O. v. cousei) probably also occurred within California, occupying the riparian forests of the Colorado River floodplain. Specimens are unknown from California, although Hall (1981) listed a record from Ehrenburg, Arizona, on the Colorado River opposite California. This species was probably already rare along the Colorado River by the time rapid immigration triggered by the the gold rush, and has been extinct long since. The Long-eared Kit Fox (Vulpes macrotis macrotis) was driven to extinction by humans in the southern coastal basins of California. Pronghorns (Antilocapra americana americana) have been extirpated from most of their historic range in California, in west-central and southern parts of the state; and Tule Elk (Cervus elaphus nannodes), while no longer threatened with extinction, exist only under full protection. The decline or extinction of most of these species was primarily due to persecution and/or overharvesting by humans.

Twenty-three mammalian species (29 subspecies) in California were designated as Threatened or Endangered species in 1985, including nine marine mammals. The terrestrial species are listed in Table 1, together with those known to be extinct or extirpated from the state. The principal factor jeopardizing the rodents and Kit Fox (Table 1) has been loss of habitat. Human disturbance and intrusion into their habitats have been significant factors in the decline of the Sierra Nevada Red Fox, Southern Wolverine, and the California and Peninsular Bighorn Sheep.

Twenty native game and fur-bearing species of mammals are partially protected by hunting and trapping regulations established by the California Fish and Game Commission and administered by the California Department of Fish and Game. Additionally, the Ringtail (Bassariscus astutus), Fisher (Martes pennanti), and Northern Flying Squirrel (Glaucomys sabrinus) are protected through legislative action or by the California Fish and Game Commission. The White-eared Pocket Mouse (Perognathus alticola), Point Arena Mountain Beaver (Aplodontia rufa nigra), Point Reyes Mountain Beaver (A. r. phaea), and all species of bats are partly protected from taking through restrictions on scientific collection by the California Department of Fish and Game. The Mountain Lion (Felis concolor) is designated as a game species but presently there is no open hunting season.

All other terrestrial species of mammals in California have little or no protection, and for those and many game, fur-bearing, fully-protected, and state Endangered and Threatened species, no management plans are in operation to ensure the preservation of unique populations. Yet, the

California Endangered Species Act of 1970 and subsequent amendments mandate the preservation of all species of native mammals. Rapid agricultural development and urbanization of vast areas in California have profoundly diminished the extent of several biotic communities and jeopardized their unique biotas. For example, the floor of the San Joaquin Valley measures approximately 3.44 million hectares (8.5 million acres). Major plant communities once included valley grassland (dominated by perennial species such as Stipa pulchra and S. cernua), San Joaquin saltbush (dominated by several xeric-adapted and halophytic shrubs, including Atriplex spp., Suaeda fruticosa, Allenrolfea occidentalis, and Ephedra californica), tule marsh (dominated by species of Scirpus and Typha), and riparian forests and savannas (Kuchler, 1977). By 1979, nearly all of this land was developed, either as large urban areas or as cultivated crop land. Less than 60,700 hectares (150,000 acres) were uncultivated, and significant portions of this latter amount were developed for petroleum extraction, strip-mined for gypsum and clay, or occupied by roads, canals, air strips, cities, oil-storage facilities, pipelines, and evaporation and percolation basins. The vast tule marshes were nearly gone; most of those that remained were degraded by seasonal flooding with waters laden with metallic salts drained from irrigated fields. The grassland communities were reduced to small remnants, mostly fringing the valley floor, and their perennial species were largely replaced by exotic species of weedy annuals such as Bromus spp., Avena spp., and Erodium spp. Of the original 404,700 hectares (about 1 million acres) or more of riparian communities in the Central Valley, less than 10% existed in 1979 (Warner, 1979); much of that was significantly degraded in quality.

Populations of mammals native to the arid coastal basins of southern California have been reduced drastically in size as a result of habitat loss. Today, nearly all of the populations of species native to the plant communities on the floor of the southern coastal basins face proximate threats of extinction. The Long-eared Kit Fox succumbed to the pressure of increasing numbers of people in southern California several decades ago (Waithman and Roest, 1977). Other species extirpated there and/or throughout southern California include the Mexican Jaguar (Merriam, 1919; Strong, 1926), Great Basin Gray Wolf (Grinnell et al., 1937), Pronghorn (McLean, 1944), and Coues' White-tailed Deer.

Populations of mammalian species dependent upon freshwater and tidewater riparian and wetland communities have declined markedly in nearly every region within California as a result of loss and degradation of their habitats. White-tailed Deer, dependent upon riparian communities in California, have already been extirpated (see section on Other Candidates in Results section) and several other species are seriously jeopardized.

The rapid, widespread loss of old-growth forests by logging activities raises concerns that cannot be resolved with existing information. Although no mammalian species in California is known to be limited to old-growth forests, a few require extensive tracts of relatively undisturbed forest. Fragmentation and isolation of relatively small blocks of mature and old-growth stands of Yellow Pine, Douglas Fir, Mixed Conifer, and Red Fir by recent forest-management practices has been extensive and could prove to be disastrous for wide-ranging species such as the Fisher and Marten, and for species with limited vagility such as

Table 1. Extinct, Threatened, and Endangered species of land mammals in California as of January, 1986. CE = California Endangered; CT - California Threatened; FE - Federal Endangered; FT - Federal Threatened.

Species	Status
San Joaquin Antelope Squirrel (Ammospermophilus nelsoni)	CT
Mohave Ground Squirrel (Spermophilus mohavensis)	CT
Morro Bay Kangaroo rat (Dipodomys heermanni morroensis)	CE, FE
Giant Kangaroo Rat (Dipodomys ingens)	CE
Fresno Kangaroo Rat (Dipodomys nitratoides exilis)	CE, FE
Stephens' Kangaroo Rat (Dipodomys stephensi)	CT
Salt Marsh Harvest Mouse (Reithrodontomys raviventris raviventris and R. r. halicoetes)	CE, FE
Amargosa Vole (Microtus californicus scirpensis)	CE, FE
Cascades Gray Wolf (Canis lupus fuscus)	Extinct
Great Basin Gray Wolf (Canis lupus youngi)	Extinct
Island Fox (Vulpes littoralis catalinae, V. l. clementae, V. l. dickeyi, V. l. littoralis, V. l. santacruzae, and V. l. santarosae)	CT
Long-eared Kit Fox (Vulpes macrotis macrotis)	Extinct
San Joaquin Kit Fox (Vulpes macrotis mutica)	CT, FE
Sierra Nevada Red Fox (Vulpes vulpes necator)	CT
California Grizzly Bear (Ursus arctos californicus)	Extinct
Southern Wolverine (Gulo gulo luteus)	CT
Mexican Jaguar (Felis onca arizonensis)	Extirpated
Western White-tailed deer (Odocoileus virginianus ochrourus)	Extirpated
California Bighorn Sheep (Ovis canadensis californiana	CT
Peninsular Bighorn Sheep (Ovis canadensis cremnobates)	CT
Bison (Bison bison bison)	Extirpated

Red Tree Mice. Likewise, some species of bats may depend on roosting sites under loose bark and in hollow cavities of standing trees and foraging sites within the forest, and their populations may be significantly impacted by loss of old-growth forests and current forest-management activities

Concern over the rapid loss of major biotic communities within California and the increasing threats to several species of mammals prompted the Nongame Bird and Mammal Section of the California Department of Fish and Game to initiate this study of potentially threatened species of mammals. The principal objectives were to identify potentially jeopardized populations of mammals within California, and to group them into priority categories for management actions, including proposals for protected status, expenditure of research funds, and preservation of essential habitat.

METHODS

Investigations of potentially threatened mammals were limited to taxa without Threatened or Endangered status at the state or federal level. Taxa concurrently being investigated through other projects of the Department of Fish and Game also were excluded from detailed studies (i.e., some subspecies of the Bobcat, Felis [Lynx] rufus, and the River Otter, Lutra canadensis).

Priorities for the investigations were:

1. species endemic to California;
2. subspecies endemic to California;
3. species and subspecies whose geographic ranges extend beyond California and which appear to be jeopardized throughout their geographic range;
4. populations of taxa threatened within California but which also occur elsewhere and appear not to be threatened throughout their geographic range (e.g., Myotis velifer velifer, the Cave Myotis).

These priorities had some influence on which taxa received greatest consideration for inclusion in detailed investigations. Priority rankings, however, were based solely on the apparent proximity and nature of threats to populations. Locally depleted populations of wider-ranging taxa within California were not studied in detail, with the exception of the American Badger (Taxidea taxus).

A working list of 86 candidate species was assembled first and then information for each candidate was gathered. Blair Csuti (The Nature Conservancy) nominated some of the candidates. Data on life histories, habitat associations, historic and current distributions, systematic status, population status, and nature of potential threats were sought for each candidate. Initially, information was gathered without benefit of field work, being derived instead from published sources, other biologists, and previous field experiences in California. The intent was to develop information needed to direct limited funds for field investigations to studies of biotic communities and taxa with the greatest perceived threats to their existence. During the course of the original investigation, however, most areas in the state where loss of habitat posed a threat to one or more species of mammals were visited. Only cursory field studies were carried out, however, as time and funding did not permit detailed field work. Subsequently, more detailed studies have been conducted in the San Joaquin Valley (Williams, 1985; Glenn Basey, unpubl. ms.; Cheryl Johnson, unpubl. ms.); San Bernardino Mountains (Sulentich, 1983; Williams, 1983);) San Gabriel mountains (Williams, 1983); Tehachapi Mountains (Sulentich, 1983); east-central slope of the Sierra Nevada (Williams, 1984); western slope of the central Sierra Nevada (unpubl. data); San Pablo and Suisun bay marshes (K. Ford, unpubl. data; Williams, 1983); Point Arena area, Mendocino Co. (Dale Steele, unpubl. data); San Francisco Bay (Ford, 1986); San Pablo Creek Marsh (by Foreman; T. Rado, pers. comm.); and on Santa Catalina Island (Collins and Martin, 1985; Williams, 1983). For the majority of the species included here, however, there have been no recent, detailed, field studies undertaken.

Letters and questionnaires seeking information on potentially threatened taxa of mammals were sent to all members of the American Society of

Mammalogists with mailing addresses in California, and to other, selected persons in state and federal agencies, colleges and universities, and natural history museums. Approximately 500 persons were contacted in this manner. Sixteen persons responded to the request for information, and collectively listed 16 species they believed should be investigated. Of the 16, two were species of marine mammals which were not included in this project, and four were species already with state Rare or Endangered status. All of the remainder were already on the working list. Subsequent to development of the draft report, questions regarding the status of 11 other taxa were raised by Ronald Nowak (Endangered Species Office, U.S. Fish and Wildlife Service). Little additional research was conducted on most of the latter species; reasons for excluding them and other candidate species are given in brief accounts (see section on Other Candidate Species).

Information on the distributions of species on the working lists was developed from the literature and from museum records. No attempt was made to locate all records, although literature and museum surveys were as complete as time and resources permitted. Data on historic distributions were assembled from these sources, but for most taxa no detailed field surveys had been conducted prior to wide spread human developments; thus, the historical distribution records of most species are scanty. Because the investigations included a large number of taxa, and because many museums do not catalog specimens by subspecies, it would have been unreasonable to ask personnel at most museums to furnish the required information. Instead, I visited and obtained distributional data from 13 major natural history museums and acquired information from nine others through correspondence (listed in the Appendix). Additional data from other mammal collections were obtained from the card files at the National Museum of Natural History, where for several decades early in the century the staff recorded specimens known to be housed in other museums. I also contacted several persons directly to inquire about specific taxa and human developments affecting plant communities in selected areas of California. This approach was more productive than broadcasting appeals for input. Unfortunately, however, identifying persons with the information needed was often impractical.

The species accounts in the Results section are arranged in groups by priority category and, within groups, systematically by genera, following Hall (1981). Species and subspecies are arranged alphabetically within genera. Records of distribution are listed alphabetically by county and, within counties, by locality with respect to reference point (e.g., city or physiographic feature). Localities using the same reference point are arranged by compass direction, starting with north and proceeding clockwise. Localities are listed as recorded on specimen tags and in museum catalogs or in the literature. The number of specimens, when known, and the museum of deposition or literature citation conclude each locality record. Cases where localities or identification of specimens seemed erroneous are discussed in the Remarks section.

Statements about habitat associations and population status are necessarily brief for most taxa because of lack of detailed information. Whenever possible, statements were developed primarily from information based on studies of the populations of concern rather than from studies conducted elsewhere. To extrapolate extensively from studies based on populations outside the area of concern carries a high risk of incorrectly characterizing habitat use and population biology. Comments in the Recommendation sections are suggestions for priority actions and are neither detailed nor complete. Remarks on systematic status are intended to flag potential problems which should be considered in designing studies to elucidate the status of populations. Although neither state nor federal regulations require that a jeopardized population have unique taxonomic identity to be designated Rare, Threatened, or Endangered, the political conflict resulting from unresolved taxonomic controversies involving endangered species is best avoided whenever possible.

In interpreting and compiling the information in this report, and in assigning taxa to priority categories, my philosophy has been to give the benefit of doubt to the species under consideration. I have attempted to convey the essence of available information, but time and resources did not permit detailed field investigations or development of lengthy species accounts. A principal purpose of this report is to stimulate others with information to come forward. I have not cited all of the relevant literature for most taxa, especially reports that added no new information on the limited set of topics included in the accounts.

Scientific names for species are those listed by Jones et al. (1982) unless noted otherwise in the species accounts. Common names are from Grinnell (1933) and Williams (1979), or are ones commonly used by the Department of Fish and Game. Common names for subspecies are generally unnecessary, but, because subspecies can be accorded Threatened or Endangered status under both state and federal regulations, use of common names for subspecies is necessary in this report. Names for subspecies are from the list of Grinnell (1933), but often amended to reflect current taxonomy or to shorten needlessly long names. I coined common names or adopted those of other authors for subspecies not listed by Grinnell (1933). Scientific names for subspecies are from Hall (1981) except in cases where I disagree with his taxonomy or when more recent publications have altered his taxonomic arrangements. Departures from Hall's (1981) subspecies taxonomy are noted in the species accounts.

RESULTS

Of the species and subspecies investigated for this report, 36 warrant greatest concerns. These are listed in Table 2 in three categories: Highest, Second, and Third priorities. The definitions for these categories are based on the perceived proximity of threats of extinction. Species listed in the Highest Priority category appear to face a high probability of extinction or extirpation from their entire geographic range in California if current trends continue. Populations of species in the Second Priority category are definitely jeopardized and declining, but the threats of extinction or extirpation appear less imminent. Populations of species listed in the Third Priority category appear not to face extinction in the near future, but they are declining seriously or are otherwise highly vulnerable to extirpation because of human developments, and require special attention in land- and resource-management decisions. Some species listed in the Second and Third Priority categories are relatively rare and virtually no current data on their distributions and population status are available; when investigated in detail, some of these may be found to face greater or lesser threats. Accounts of species included in the three priority categories are presented in the following section.

Other species investigated, but considered to warrant lesser concerns are listed in the fourth group, entitled Other Candidates, beginning on page 73. Brief remarks on the probable status of these species, deleted from the final List of Concern, are included. Lack of information for some species listed is cause for concern about their status; their deletion from the final List of Concern here should not be construed as a definitive indication of their population status.

Major contributing factors in jeopardizing most species on the List of Concern are the diminishment and degradation of natural communities (Table 3 and Species Accounts). Loss of native plant communities in four regions of California present the most acute problems threatening unique (i.e., taxonomically recognized) populations of mammals: the Colorado River riparian communities; the southern California coastal basins from the San Fernando Valley southward to the Mexican boundary, including lowland grassland and desert communities and tidal marshes along the south coast; the San Joaquin Valley desert, grassland, and riparian and wetland communities; and the tidal marsh communities of San Francisco and San Pablo bays. Loss of riparian and wetland communities is no less serious elsewhere, but fewer unique taxa of mammals are threatened. In addition to loss of habitat, disturbances by humans, especially in hibernacula and maternity roosts is perceived as a serious threat to most of the species of bats. Habitat degradation and lack of information on hunter take and population dynamics for use in management are viewed as potential threats to the five species of rabbits and hares, currently designated as game species, on the List of Concern. Cutting and fragmentation and isolation of blocks of mature and old-growth conifer forests pose potential threats to two species on the List of Concern.

Potential occurrences of Species of Special Concern on lands administered by several state and federal agencies are listed in Table 4. Information in Table 4 was developed from a "Federal Public Lands Responsibility Map" of California (1:7500,000 scale; 1978) and from other sources. There is

Table 2. Mammalian species of special concern in California, listed by priority categories. See text for definitions of priority categories. Page refers to starting page of species accounts.

Species	Page
HIGHEST PRIORITY	
Buena Vista Lake Shrew (Sorex ornatus relictus)	13
Suisun Shrew (Sorex ornatus sinuosus)	14
Santa Catalina Shrew (Sorex ornatus willetti)	16
Salt-marsh Wandering Shrew (Sorex vagrans halicoetes)	17
Arizona Myotis (Myotis lucifugus occultus)	19
Arizona Cave Myotis (Myotis velifer velifer)	21
Riparian Brush Rabbit (Sylvilagus bachmani riparius)	23
Point Arena Mountain Beaver (Aplodontia rufa nigra)	24
Los Angeles Pocket Mouse (Perognathus longimembris brevinasus)	25
Pacific Pocket Mouse (Perognathus longimembris pacificus	27
Tipton Kangaroo Rat (Dipodomys nitratoides nitratoides)	28
Colorado River Cotton Rat (Sigmodon arizonae plenus)	30
Yuma Mountain Lion (Felis concolor browni)	31
SECOND PRIORITY	
So. California Salt-marsh Shrew (Sorex ornatus salicornicus)	34
California Leaf-nosed Bat (Macrotus californicus)	35
Townsend's Big-eared Bat (Plecotus townsendii)	37
Pocketed Free-tailed Bat (Tadarida femorosacca)	39
California Mastiff Bat (Eumops perotis californicus)	39
Salinas Pocket Mouse (Perognathus inornatus psammophilus)	42
White-eared Pocket Mouse (Perognathus alticola alticola)	43
So. Marsh Harvest Mouse (Reithrodontomys megalotis limicola)	44
Riparian Woodrat (Neotoma fuscipes riparia)	45
White-footed Vole (Arborimus albipes)	46
Point Reyes Jumping Mouse (Zapus trinotatus orarius)	48
THIRD PRIORITY	
Big Free-tailed Bat (Tadarida macrotis)	50
Pygmy Rabbit (Brachylagus idahoensis)	51
Oregon Snowshoe Hare (Lepus americanus klamathensis)	52
Sierra Nevada Snowshoe Hare (Lepus americanus tahoensis)	54
Western White-tailed Hare (Lepus townsendii townsendii)	55
Point Reyes Mountain Beaver (Aplodontia rufa phaea	57
Tehachapi Pocket Mouse (Perognathus alticola inexpectatus)	58
Short-nosed Kangaroo Rat (Dipodomys nitratoides brevinasus)	59
Red Tree Vole (Arborimus longicaudus)	61
Pacific Fisher (Martes pennanti pacificus)	64
American Badger (Taxidea taxus)	66
Channel Islands Spotted Skunk (Spilogale gracilis amphiala)	70

Table 3. Distribution, habitat, and major causes for concern for mammalian species of special concern in California. Refer to species accounts for details. CA - California; CI - Santa Cruz and Santa Rosa Islands; CR - Colorado River; des. - desert; disturb. - human disturbances; GB - Great Basin steppes; Mtn. - mountains; N - north; NW - Northwest; Pt. A. - Point Arena area; Pt. R. - Point Reyes area; SB - San Bernardino; SCB - southern California coastal basins; SCM - southern California coastal marshes; SCI - Santa Catalina Island; SFB - San Francisco Bay; SJV - San Joaquin Valley; SN - Sierra Nevada; So - southern; SPB - San Pablo Bay; SV - Salinas Valley; Tr. R.- Transverse Ranges; W - west.

Species	Range	Habitat	Cause for Concern
	HIGHEST PRIORITY		
Buena Vista Lake Shrew	SJV	Wetland	Habitat loss
Suisun Shrew	SPB	Salt marsh	Habitat loss
Santa Catalina Shrew	SCI	Riparian	Habitat loss?
Salt-marsh Wandering Shrew	SFB	Salt marsh	Habitat loss
Arizona Myotis	CR	Riparian	Habitat loss, disturb.
Arizona Cave Myotis	CR	Riparian	Habitat loss, disturb.
Riparian Brush Rabbit	SJV	Riparian	Habitat loss
Point Arena Mountain Beaver	Pt. A.	Wetland	Habitat loss
Los Angeles Pocket Mouse	SCB	Grass/Des.	Habitat loss
Pacific Pocket Mouse	SCB	Grass/Des.	Habitat loss
Tipton Kangaroo Rat	SJV	Grass/Des.	Habitat loss
Colorado River Cotton Rat	CR	Riparian	Habitat loss
Yuma Mountain Lion	CR	Riparian	Habitat loss
	SECOND PRIORITY		
So. California Salt-marsh Shrew	SCM	Salt Marsh	Habitat loss
California Leaf-nosed Bat	So CA	Grass/Des.	Habitat loss, disturb.
Townsend's Big-eared Bat	CA	Various	Disturb.
Pocketed Free-tailed Bat	So CA	?	Disturb.?
California Mastiff Bat	W CA	Widespread	Disturb., pesticides?
Salinas Pocket Mouse	SV	Grassland	Habitat loss
White-eared Pocket Mouse	SB Mtn.	?	Habitat loss?
So. Marsh Harvest Mouse	SCM	Salt marsh	Habitat loss
Riparian Woodrat	SJV	Riparian	Habitat loss
White-footed Vole	NW CA	Riparian?	Habitat loss?
Point Reyes Jumping Mouse	Pt. R.	Wetland	Habitat loss
	THIRD PRIORITY		
Big Free-tailed Bat	So. CA	?	Disturb.?
Pygmy Rabbit	GB	Sagebrush	Habitat loss
Oregon Snowshoe Hare	NE CA	Thickets	Habitat loss
Sierra Nevada Snowshoe Hare	SN	Thickets	Habitat loss
Western White-tailed Hare	GB SN	Grassland	Habitat loss
Point Reyes Mountain Beaver	Pt. R.	Wetland	Habitat loss
Tehachapi Pocket Mouse	Tr. Mt.	Grass/Des.	Habitat loss
Short-nosed Kangaroo Rat	SJV	Grass/Des.	Habitat loss
Red Tree Vole	NW CA	Fir	Habitat loss
Pacific Fisher	N CA	Conifer	Habitat loss, disturb.
American Badger	W CA	All	Habitat loss, taking
Channel Islands Spotted Skunk	CI	?	?

no similar, pictorial summary of state-managed lands nor a central source of information on state-owned lands. Thus, only lands managed by the California Departments of Fish and Game and Parks and Recreation are included. Information displayed in Table 4 should not be construed as indicating non-occurrence on lands administered by other governmental agencies or that other Species of Concern do not occur on lands administered by these agencies.

Table 4. Occurrence of Mammalian Species of Special Concern in California on lands administered by selected state and federal agencies. BLM - U.S. Bureau of Land Management (Dept. of Interior); CPR - Calif. Dept. of Parks and Recreation; DFG - Calif. Dept. of Fish and Game; DD - U.S. Department of Defense; DE - U.S. Department of Energy; FS - U.S. Forest Service (Dept. of Agriculture); FWS - U.S. Fish and Wildlife Service (Dept. of Interior); NPS - U.S. National Park Service (Dept. of Interior). See Table 2 for scientific names.

Species	CPR	DFG	BLM	DD	DE	FS	FWS	NPS
	HIGHEST PRIORITY							
Buena Vista Lake Shrew	X		X		X		X	
Suisun Shrew	X	X		X			X	
Santa Catalina Shrew								
Salt-marsh Wandering Shrew	X	X					X	
Arizona Myotis	X		X				X	
Arizona Cave Myotis	X		X				X	
Riparian Brush Rabbit	X			X				
Point Arena Mountain Beaver	X		X	X				
Los Angeles Pocket Mouse		X	X	X				
Pacific Pocket Mouse	X			X				
Tipton Kangaroo Rat	X	X	X	X			X	
Colorado River Cotton Rat	?		X				X	
Yuma Mountain Lion	X		X	X		X	X	X
	SECOND PRIORITY							
So. California Salt-marsh Shrew	X	X		X			X	
California Leaf-nosed Bat	X	X	X	X			X	X
Townsend's Big-eared Bat		X	X	X	X	X	X	X X
Pocketed Free-tailed Bat	X	X	X			X	X	X
California Mastiff Bat	X	X	X	X	X	X	X	X
Salinas Pocket Mouse	?		X	X				
White-eared Pocket Mouse			X			X		
So. Marsh Harvest Mouse	X	X		X			X	
Riparian Woodrat	X			X				
White-footed Vole	X	X				X		X
Point Reyes Jumping Mouse	X							X
	THIRD PRIORITY							
Big Free-tailed Bat	X	X	X	X		X		
Pygmy Rabbit	X	X	X	X		X		
Oregon Snowshoe Hare	X		X			X	X	
Sierra Nevada Snowshoe Hare	X					X		X
Western White-tailed Hare	X	X	X	X		X	X	X
Pt. Reyes Mountain Beaver	X							X
Tehachapi Pocket Mouse	X		X			X		
Short-nosed Kangaroo Rat		X	X		X			
Red Tree Vole	X	?	X			X		X
Pacific Fisher	X	?	X			X		X
American Badger	X	X	X	X	X	X	X	X
Channel Islands Spotted Skunk								X

HIGHEST PRIORITY LIST

Buena Vista Lake Shrew
Sorex ornatus relictus

1932. *Sorex ornatus relictus* Grinnell, Univ. California Publ. Zool., 38:389. *Type Locality:* evacuated slough just outside of east side levee, Buena Vista Lake, 290 ft, Kern Co., California.

Distribution: Grinnell (1933) speculated that Buena Vista Lake Shrews once occupied the marshlands of the San Joaquin Valley floor throughout most of the Tulare Basin. He noted that by 1933, their range was much restricted because of the disappearance of lakes and sloughs.

Populations Status: Nothing is known about the current population status of the Buena Vista Lake Shrew. Nearly all of the valley floor in the Tulare Basin is now cultivated. Most of the lakes and marshes have been drained and are also cultivated. All of the Buena Vista Lake bed is cultivated, and most of the canals in the area are steep-sided and kept free of vegetation by use of herbicides. Ornate shrews (*Sorex ornatus*) may be extant in places such as the Kern National Wildlife Refuge, in wetlands of the Kern River percolation area, and along sloughs and canals on the valley floor leading into Goose Lake, although there are no records of shrews from any of these areas. Any extant populations found within the Tulare Basin may or may not be representative of *S. o. relictus*. Clark et al. (1982) used pitfall traps in an unsuccessful attempt to capture Buena Vista Lake Shrews on The Nature Conservancy Paine Wildflower Preserve and the Voice of America transmitter site west of Delano.

Habitat: Buena Vista Lake Shrews occupied marshes on the perimeter of Lake Buena Vista (Grinnell, 1932). Farther north, in the San Joaquin Basin, Ornate Shrews live in dense vegetation along streams and sloughs and around the perimeter of tule marshes (unpub. data). Presumably, Buena Vista Lake Shrews occurred in similar wetlands in the Tulare Basin.

Recommendations: A survey for Buena Vista Lake Shrews should be undertaken to establish its distribution and population status and to identify potential threats to remaining populations. Initial efforts should be concentrated around the Buena Vista Lake Aquatic Recreation Area (Lake Webb and Lake Evans), on the Kern National Wildlife Refuge, the Tule Elk Reserve, and along Buena Vista and Goose Lake sloughs. Pitfall traps are most effective for shrews (Williams and Braun, 1983), and can be left in place for extended times offsetting the extra effort required to set them.

Remarks: Nothing has been recorded on this taxon since the original description. The more upland subspecies, *S. o. ornatus*, occupies the area surrounding the range of *S. o. relictus*. Intergradation probably occurred along the lower courses of the streams entering the floor of the Tulare Basin.

Distribution Records: *KERN CO.*: N side Buena Vista Lake, 1 (USNM); east side levee, 298 ft, Buena Vista Lake, 290 ft, 3 (MVZ); Buttonwillow, 1 (CAS).

Suisun Shrew
Sorex ornatus sinuosus

1913. Sorex sinuosus Grinnell, Univ. California Publ. Zool., 10:181.
Type Locality: Grizzly Island, Suisun Bay, Solano Co., California.
1979. Sorex ornatus sinuosus , Williams, Ann. Carnegie Mus., 48:426.

Distribution: According to Rudd (1955a), Suisun Shrews occur in the tidal marshes of the northern shores of San Pablo and Suisun bays, as far east as Grizzly Island and as far west as the mouth of Petaluma Creek. Brown and Rudd (1981), however, redefined the western boundary of the range as Sonoma Creek and Tubbs Island. Shrews living in the marshes as far east as Collinsville represent S. o. californicus (Williams, 1983).

Populations Status: At one time, San Pablo and Suisun bays were lined with salt and brackish water marshes, but today marshes are broken into several small, isolated units. The marshes of Suisun Bay are the most extensive, but Suisun Shrew populations there may be threatened in part by management of the marshes to favor growth of Scirpus. Present habitat is much less extensive in San Pablo Bay. Very few of the extant tidal marshes have a full profile of marshland vegetation, and few border on significant upland areas where marshland species can seek refuge from flooding. Suisun Shrews inhabit a smaller area and are more restricted in the habirats they occupy than are Salt Marsh Harvest Mice, an Endangered Species.

The population status of S. o. sinuosus was investigated during spring and summer, 1983, following a winter of record flooding in the marshes of San Pablo and Suisun bays (Williams, 1983). No live Suisun Shrews were captured on any of 34 transects in marshes throughout the geographic range of S. o. sinuosus. One dead shrew was found, however. Subsequent surveys with Sherman live traps found one Suisun shrew in a marsh on the northern perimeter of Suisun Bay in 1985 (K. Ford, pers. comm.). The principal problem in both areas, but most acute in San Pablo Bay, is the lack of upland areas continuous with the marshes where Suisun Shrews and other terrestrial animals can refuge during times of flooding (Williams, 1983). Management of the marshes has not included refuges from flooding as a required element.

Habitat: Suisun Shrews typically inhabit tidal marshes characterized in order of decreasing tolerance to inundation, by Spartina foliosa, Saliconria ambigua, and Grindelia cuneifolia, and brackish marshes dominated by Scirpus californicus and Typha latifolia (Rudd, 1955a). They appear to require dense, low-lying cover where invertebrates are abundant. Rudd stated that structure ("growth form") of the plant community, not species composition, was the determining factor in shrew occupancy. Driftwood and other litter above the mean high-tide line is probably essential for nesting and foraging sites. Upland habitats, continuous with the marshlands, offering sufficient cover and sources of food to sustain shrews during prolonged flooding of marshes and dikes, such as occurred during the winter of 1982-83, are also probably essential (Williams, 1983).

Recommendations: When the results of additional population surveys for Suisun Shrews are completed (U.S. Fish and Wildlife Service, in litt.), marshland management practices and plans should be reviewed to determine their impact on populations of Suisun Shrews. Acquisition of upland areas for refuge from flooding, or creation of refuge sites on dikes above flood level will probably be required to provide suitable habitat for Suisun Shrews and other small mammals inhabiting the tidal marshes.

Remarks: Rudd (1955a) analyzed structural characters of Sorex ornatus-group shrews from the San Pablo and Suisun bays area and determined that shrews of the sinuosus type were recognizably distinct from S. o. californicus, but that spatially intermediate populations exhibited some intermediate characteristics. He thought that because the character gradients were rather abrupt, recognition of sinuosus as a species was justified. Brown (1971, and unpubl. dissert., 1970) found that S. ornatus and S. sinuosus had identical karyotypes, which differed from S. vagrans of the San Francisco Bay area. Karyotypes of shrews from populations previously thought to be S. v. vagrans or hybrids of vagrans, ornatus and sinuosus were shown to be identical to the ornatus-sinuosus karyotype. Brown and Rudd (1981) investigated the relationships of these shrews further, determining that sinuosus was best treated as a subspecies of S. ornatus, and restricting the distribution of sinuosus to an area approximately east of Sonoma Creek, but including Tubbs Island within the range of sinuosus.

The disposition of specimens referred to by Rudd (1955a) is not clear. Some of these are in the Museum of Vertebrate Zoology and others are in the collections at the University of California Davis. Differences in the locality descriptions on the specimen labels and in Rudd's (1955a) paper and differences in total numbers of specimens made it difficult to decide if these referred to the same specimens. Several specimens in museums, collected in marshes in San Pablo Bay in Sonoma County are designated as S. sinuosus, but are now considered to be S. o. californicus. Two specimens in the Museum of Vertebrate Zoology from "Solano County; Lake Chabot (seepage area below dam)" were assigned to the subspecies S. o. sinuosus. Lake Chabot is located on a golf course, west of the Solano Co. Fairgrounds. These specmiens may be S. o. californicus or S. o. sinuosus; they are not listed below.

Distribution Records: SOLANO CO.: 0.5 mi NE Cordelia Salt Marsh, 5 (MVZ); Gray Goose Duck Club, 1.5 mi SW Suisun, 1 (CAS); Grizzly Island, Suisun Bay, 1 (LACM), 20 (MVZ), 5 (UDAV), 2 (USNM), 19 (Rudd, 1955a); Grizzly Island, State Dept. of Fish and Game headquarters, 1 (CSCS); Honker Gun Club, near Dutton, Van Sickle Island, 1 (CAS); 3 mi E Mare Island Bridge, adjacent to White Slough and Hwy. 37, 1 (MVZ); 2.7 mi W jct. Napa Road and Hwy. 37 (on 37), 1 (MVZ); 8 mi N Rio Vista, 80 ft, 1 (CSUF); Sears Point Road, 6 mi NW Vallejo, 65 (MVZ); Sears Point Road, 8 mi NW Vallejo, 7 (MVZ); Sears Point Road, San Pablo Bay, 47 (Rudd, 1955a); South Hampton Bay, near Solano Co. public dump, 15 (MVZ); South Hampton Bay, 13 (Rudd, 1955a); Suisun City, salt marsh adjacent to Cordelia St., 4 (MVZ); Suisun marshes, periphery of Grizzly Island, 13 (Rudd, 1955a); 3 mi NW Vallejo, 1 (UDAV); Van Sickle Island, 1 mi S Dutton, 1 (CAS); Van Sickle Island, 1 (CAS), 3 (UDAV).

Santa Catalina Shrew
Sorex ornatus willetti

1941. *Sorex willetti* von Bloeker, Bull. So. California Acad. Sci., 40:163. Type Locality: Avalon Canyon, Santa Catalina Island, Los Angeles Co., California.

1967. *Sorex ornatus willetti*, von Bloeker, Proc. Symposium Biol. California Islands, p. 246.

Distribution: Santa Catalina Shrews are known only from Santa Catalina Island, Los Angeles California (Williams, 1983). The localities for the specimens and three sight records collectively span the length of the island (Collins and Martin, 1985; Williams, 1983).

Population Status: Only a single Santa Catalina Shrew was captured in a survey during January, 1983 employing pitfalls on 22 10-trap transects distributed in all major plant communities on the island (Williams, 1983). A subsequent survey, during summer, 1985, employing pitfalls on 25 10-trap transects failed to capture any additional shrews (Collins and Martin, 1985). Williams believed that the Santa Catalina Shrew might have been on the verge of extinction. Degradation of woodland, riparian, and wetland communities by introduced ungulates, especially sheep, cattle, feral goats, and wild pigs, and further degradation of these communities by diversion of water from springs and streams to the urbanized areas on the island probably were factors contributing to the scarcity of shrews. Another important factor was probably the marginal suitability of the island communities for shrews. Santa Catalina Island contains only a few small, degraded riparian communities and wetlands. Another important element threatening shrews was the high density of feral cats on the island.

Although most of the island is being managed by the Santa Catalina Island Conservancy with the objective of preserving the native biota, the Conservancy presently derives significant income from commercial hunting of wild pigs, goats, and deer, and sale of some bison. Free roaming ungulates on the island are the major threat to its native biota and a potential source of irreconcilable conflict of management objectives. These ungulates and introduced blackbuck (*Antilope carvicapra*) are noticeably degrading some natural plant communities and preventing others from recovering from damage caused when densities of domestic and feral ungulates were greater. Wild pigs are causing the most apparent damage in wetlands and streamside communities and on wooded slopes with deep layers of leaf litter: areas likely to be of major importance to shrews. Additional serious threats stem from the growing human resident and transient populations on the island where potable water supplies are already overtaxed. Diversion of increasing amounts of fresh water for human use will probably result in additional degradation of the already damaged riparian systems on the island.

Habitat: Although on the mainland, Ornate Shrews are often captured in upland plant associations such as coastal sagebrush and chaparral, their precise habitat requirements are unknown, and it has not been shown that communities such as these are important to breeding populations. Over most of their range in California, Ornate Shrews are most abundant in riparian and wetland communities (Collins and Martin, 1985; Owen and

Hoffmann, 1983; Williams, 1983). The only specimen of S. o. willetti trapped was taken in a tangle of driftwood and woody roots a few feet above a creek bed; riparian vegetation was poorly developed along that portion of the creek (Williams, 1983).

Recommendations: Two recent surveys for Santa Catalina Shrews have shown the extreme rarity of this insular population, but have failed to delineate critical elements of the biotic communities that must be protected in order to ensure survival of Santa Catalina Shrews. Thus, any activities taken to help protect the population possibly could have the opposite effect. Specific actions to enhance habitat for shrews on the island should only be made after obtaining results from detailed studies of Ornate Shrews on the mainland in similar communities with a similar climate. Studies should be designed to provide information needed in determining the best course of action to enhance the chances of survival of Santa Catalina Shrews. Regardless of these studies, however, preserving wetland and riparian systems should be given high priority in management of the island. Elimination of all ungulates, starting with wild pigs, feral goats, and bison would remove the greatest sources of damage to plant communities on the island. An intensive campaign to reduce the size of the population of feral cats would alleviate some pressure on shrews, but both the practicality of such a campaign and its importance in protecting shrews is in doubt.

The possibility that insular populations of Ornate Shrews occur on Santa Cruz, Santa Rosa, San Miguel, and/or San Clemente islands should not be overlooked (von Bloeker, 1967; Walker, 1980).

Remarks: The Santa Catalina Shrew was described as a species, but later was relegated to subspecific status under Sorex ornatus (von Bloeker, 1967). It differs from mainland shrews of the ornatus group in being larger and somewhat darker in color dorsally, with lighter-colored underparts. Present data are insufficient to determine the degree of relationship between Ornate and Santa Catalina Shrews, although available evidence supports treating it as a subspecies of S. ornatus (unpubl. data). A single, subfossil cranium of Sorex was found among midden remains in a rock fissure on the coast of San Miguel Island, Santa Barbara Co. (Walker, 1980). This specimen appeared to be structurally similar to S. o. willetti, according to Walker (1980).

Anecdotal accounts of sight records of Santa Catalina Shrews by residents of the island are: Middle Canyon W of Thompson Dam (Williams, 1983); Bunk House at Middle Ranch, Middle Canyon; and Road to west end, 0.2 mi W Isthmus Dump (Collins and Martin, 1985).

Distribution Records: LOS ANGELES CO.: Avalon Canyon, Santa Catalina Island, 1 (LACM); 0.1 mi W Cottonwood Reservoir, Cottonwood Canyon, 76 m, 1 (CM).

Salt Marsh Wandering Shrew
Sorex vagrans halicoetes

1913. Sorex halicoetes Grinnell, Univ. California Publ. Zool., 10:183.
Type Locality: salt marsh near Palo Alto, Santa Clara Co., California.

1928. Sorex vagrans halicoetes , Jackson, N. Amer. Fauna, 51:108.

Distribution: Sorex v. halicoetes is limited to the salt marshes of the south arm of San Francisco Bay (Findley, 1955).

Populations Status: Johnston and Rudd (1957) reported fluctuations in relative numbers between 1951 and 1955, based upon numbers of active nests found. They determined that Salt Marsh Wandering Shrews represented about 10% of the small mammal community of the marshes and were less abundant than Mus musculus, Reithrodontomys raviventris, Rattus norvegicus, and Microtus californicus.

Most of the once-extensive salt marshes of San Francisco Bay have been lost by human developments. The extent of remaining habitat for these shrews is small. Because they use only a limited area within the marshes, there is less habitat for them than for Salt Marsh Harvest Mice. Therefore, S. v. halicoetes may be under greater threats of extinction than the Endangered Salt Marsh Harvest Mice. Ford (1986) found 16 live Salt Marsh Wandering Shrews in a popualtion survey of marshes in San Francisco Bay during 1985. He recommended federal Endangered Species status for S. v. halicoetes.

Habitat: Johnston and Rudd (1957) provided a detailed sketch of the habitat of Salt Marsh Wandering Shrews. The shrews frequented areas in tidal marshes that provide dense cover, abundant food (primarily invertebrates), suitable nesting sites, and fairly continuous ground moisture. Their center of activity was in the "medium high marsh," about 6 to 8 ft above sea level, and in lower-lying marsh not regularly inundated. Suitable sites were characterized by abundant driftwood and other debris scattered among Salicornia. The Salicornia was usually 1 to 2 ft tall. The detritus preserved moisture and offered refuge in dry periods to amphipods, isopods, and other invertebrates, and resting sites for shrews. Nesting material consisted of plant parts, primarily Spartina duff. The higher-lying marsh, 8 to 9 ft in elevation, was too dry and offered only minimal cover — few or no shrews occupied this zone. The lower zone, dominated by Spartina, was subjected to daily tidal floods and had cover too sparse for shrews.

Recommendations: Any planned developments or activities that would modify marsh vegetation and degree of inundation with the range of the Salt Marsh Wandering Shrew should be reviewed to determine impact on this species. The feasibility of modification of extant marshes to enhance suitability for Salt Marsh Wandering Shrews, Salt Marsh Harvest Mice, and other jeopardized species should be investigated.

Remarks: The taxonomy of the Sorex vagrans species group has had a complex and confusing history and additional changes in scientific names will probably be required when relationships among populations are finally resolved (Findley, 1955; Hennings and Hoffmann, 1977; Rudd, 1955a). There appears to be no controversy about the taxonomic status of the halicoetes population, however. Brown (1974) found that the karyotype of one Salt Marsh Wandering Shrew was similar to those of three specimens of S. v. vagrans from the northern part of the San Francisco Bay region. These karyotypes were most similar to S. vagrans from areas farther north in California and Oregon. Eight specimens identified as S. v. halicoetes in the collection of the California Academy of Sciences, captured from Inverness, Marin Co., are, in my opinion, S. v. sonomae.

Rudd (1955b) compared ages, sexes, and weights of S. v. vagrans, S. v. paludivagus, S. ornatus californicus, and S. o. sinuosus. Methods of distinguishing Salt Marsh Wandering Shrews from Ornate Shrews were given in Hennings and Hoffmann (1977) and Junge and Hoffmann (1981). The report by Ford (1986) was completed too late to include details here, but is the most complete account of the distribution and current population status of S. v. halicoetes.

Distribution Records: ALAMEDA CO.: 1 mi N bay Farm Island, Melrose Marsh, 1 (MVZ); Berkeley, 1 (USNM); Dumbarton Point, 1 (KU); Elmhurst, 4 (MVZ); Hayward, 1 (MVZ); Hayward Landing, end of Russell City Road, 6 (MVZ); Melrose, 1 ((MVZ); 1 mi NW Newark, 1 (MVZ); Oakland airport, 1 (MVZ); S side Oakland Airport, 12 (MVZ); West Berkeley, 1 (USNM). CONTRA COSTA CO.: Giant (Atlas Powder Co. salt marsh), 1 (MVZ); 3 mi NE Oakley, E side Grizzly Island, 1 (MVZ); mouth San Pablo Creek, 1 (MVZ); San Pablo Creek Salt Marsh, 29 (MVZ); San Pablo Marsh, Richmond (Johnston and Rudd, 1957); salt marsh, 3 mi N Richmond, 1 (CM); salt marsh, 4 mi N Richmond, 1 (CM). SAN FRANCISCO CO.: Lake Merced, 1 (CAS); San Francisco, 4 (CAS), 1 (MVZ), Presidio, San Francisco, 1 (CAS). SAN MATEO CO.: no specific locality, 1 (MVZ); Belmont, 1 (MVZ), 2 (USNM); 0.9 mi NE Coloma, 17 (CAS); 0.5 mi S Chinese Cemetery, Coloma, 1 (CAS); Coloma, 1 (CAS); W approach Dumbarton Bridge, 3 (MVZ); Juncitas, 1 (SDSNH); Menlo Park, 1 (SDSNH); Pacifica, 0.25 mi E Westview, 1 (CAS); Palo Alto, 2 (LACM); adjacent to Palo alto Yacht Harbor, 1 (UDAV); San Mateo, 2 (USNM); Redwood City, 1 (LACM), 3 (MVZ), 23 (SDSNH); Woodside, 1 (SDSNH). SANTA CLARA CO.: 1.75 mi NE Alviso, 79 (MVZ); Los Esteros Road, 0.5 mi NE Alviso (salt marsh), 20 (MVZ); 1 mi SSW Alviso (salt marsh), 3 (MVZ); county line between Santa Clara and San Mateo counties, on bay between Palo Alto and Redwood City, 2 (CM); Palo Alto, 7 (MVZ), 3 (USNM).

Arizona Myotis
Myotis lucifugus occultus

1909. Myotis occultus Hollister, Proc. Biol. Soc. Washington, 22:43.
Type Locality: west side Colorado River, 10 mi above Needles, San Bernardino Co., California.
1967. Myotis lucifugus occultus, Findley and Jones, J. Mamm., 48:443.

Distribution: Arizona Myotis occur from southeastern California and Sonora, Mexico, to western Chihuahua, Mexico, and northward in Arizona and western New Mexico (Fenton and Barclay, 1980; Hall, 1981). In California, the Arizona Myotis occurs only along the Colorado River lowlands and in the adjacent desert mountain ranges.

Populations Status: Populations of M. lucifugus have drastically declined in numbers in many parts of its range. Fenton and Barclay (1980) attributed declines, in part, to the use of pesticides, control measures in nursery colonies, collecting of bats by researchers, and disturbance of hibernating individuals. They underscored the importance in disturbances of hibernating bats, noting that it causes bats to lose weight, thus decreasing their chances of survival. Judging from incidental accounts, Arizona Myotis were formerly common along the Colorado River. More recent observations suggest that this population has declined drastically (K. Stager, P. Leitner, pers. comm.). Arizona Myotis, like many other bats, may be sensitive to disturbances in their maternity roosts too. Many females fail to return to maternity colonies in years subsequent to disturbances by humans. They may abandon a colony prematurely, just after the young have learned to fly, but before than have learned to capture insects. This could lower recruitment into the population. Stream channelization and loss of the riparian vegetation may also be factors in the decline of M. l. occultus in California. Pesticide use in agricultural areas along the lower Colorado River and elsewhere could be a contributing factor in the decline (Fenton and Barclay, 1980; Geluso et al., 1976).

Habitat: Both Hollister (1909) and Grinnell (1914) shot Arizona Myotis flying among Cottonwood trees on the floodplain of the Colorado River. Others were collected by Grinnell "over water in a back eddy of the river." Stager (1943) found one Arizona Myotis about 100 ft in from the entrance of a large copper mine in the Riverside Mountains, in the northeastern corner of Riverside County. He also collected individuals from a maternity colony of about 800 bats, located on the underside of a bridge near Blythe. Although M. l. occultus is known only from the low desert along the Colorado River in California, it is most commonly associated with pine forests at elevations from 6000 to 9000 ft in other parts of its range (Barbour and Davis, 1969).

In most areas, roosts of Arizona Myotis have been found beneath bridges and in attics of buildings (Barbour and Davis, 1969). Arizona Myotis probably also use hollows in trees and protected crevices in rocks for roosts. Stager's (1943) record is the only known occurrence in a mine or cave.

Arizona Myotis may migrate out of California to spend the autumn and winter elsewhere, or they may make more local migrations to suitable hibernacula. The earliest and latest records of occurrence in California are 30 April and 16 August, respectively (Stager, 1943).

Recommendations: Highest priority should be given to locating populations, especially maternity colonies, and obtaining estimates of colony size. The most critical need is to establish a data base on population size so that future trends can be more reliably monitored. Where Arizona Myotis may pose a public health problem or a nuisance, exclusion or non-lethal aversion devices should be the only control methods allowed (Barclay et al., 1980; Constantine, 1979).

Remarks: Earlier, I (Williams, 1979) recognized occultus as a species despite evidence presented by Findley and Jones (1967) and Barbour and Davis (1970) indicating interbreeding in an area in New Mexico and southern Colorado between M. lucifugus and occultus. My decision was based on the results of a phenetic analysis of the genus Myotis (Findley, 1972) that suggested that occultus and M. l. carissima (the taxon with which occultus was thought to interbreed) were best place in different species groups. Clearly, genetic studies are needed to resolve these conflicting findings. In the meantime, however, it seems best to follow

Findley and Jones (1967) in treating occultus as a subspecies of M. lucifugus.

Distribution Records: IMPERIAL CO.: Potholes, 1 (SDSNH); 4 mi S Potholes, Colorado River, 1 (MVZ); 5 mi NE Yuma, 1 (KU), 4 (MVZ). RIVERSIDE CO.: Blythe, 6 (CM), 59 (LACM), 1 (MVZ); Ft. Yuma, 3 (CAS); Ripley, 5 mi S Blythe, 3 (LACM); Riverside Mountains, 1 (LACM). SAN BERNARDINO CO.: 10 mi N Needles, 2 (USNM).

Arizona Cave Myotis
Myotis velifer velifer

1890. Vespertilio velifer J. A. Allen, Bull. Amer. Mus. Nat. Hist., 3:177. Type Locality: Santa Cruz del Valle, Guadalajara, Jalisco, Mexico.
1897. Myotis velifer, Miller, N. Amer. Fauna, 13:56.

Distribution: M. velifer velifer is found from extreme southeastern California eastward to western New Mexico and southward to Guatemala. In California, it is known only from the lowlands of the Colorado River and adjacent desert mountain ranges (Vaughan, 1959).

Population Status: Vaughan (1959) found large colonies, each containing approximately 1000 individuals, in several mine tunnels in the Riverside Mountains of Riverside and San Bernardino counties. More recent observations in the area (P. Brown, in litt.), suggest a significant decline in population size. Mines previously occupied now have few bats. The extent of decline and its causes are speculative. Loss of riparian habitat for foraging could be a principal factor, together with human disturbances of colonies. Renewed mining activities and casual exploration of caves and mines by rock collectors and sight-seers may be major sources of disturbances. Use of pesticides might have contributed to the decline by reducing abundance of insects and by poisoning bats.

Habitat: Cave Myotis are habitual cave dwellers and are highly colonial. They inhabit arid zones in the southwestern United States. During the season of reproduction, in spring and summer, they form large colonies in warm caves and mines and less often in buildings and other structures (Barbour and Davis, 1969). In California, they have been found in an old storehouse (Grinnell, 1914) and in mine tunnels (Stager, 1939; Vaughan, 1959). In large portions of their range they are typically associated with Brazilian Free-tailed Bats (Tadarida brasiliensis). Vaughan (1959) noted that Cave Myotis used a variety of temporary roosts: buildings, caves, and mine tunnels.

Vaughan (1959) found that in the vicinity of the Riverside Mountains, Cave Myotis foraged primarily over the floodplain of the Colorado River. He found that they maintained regular foraging paths "over low vegetation, along the files of dense vegetation that line the oxbows and main channel of the river, between the scattered thick patches of vegetation that dot the floodplain, or above bodies of water." He noted that the dense, linear stands of Mesquite, Tamarisk, and Catclaw Acacia bordering still water of oxbow ponds seemed to constitute optimal foraging habitat. Most foraging bats were observed between about 6 and 15 ft above the ground, primarily close to vegetation. Kunz (1974) found that most individuals from large colonies in south-central Kansas dispersed nightly for considerable distances to feed. Insects of a variety of orders were consumed; beetles (Coleoptera) comprised the single largest class (Kunz, 1974).

In parts of their geographic range, especially in Kansas, Oklahoma, and Texas, Cave Myotis hibernate in winter (Twente, 1955). M. v. incautus may be quite different in its life history from M. v. velifer, however (Hayward, 1970). Stager (1939) did not find Cave Myotis in California during winter and early spring. He believed that they were only in California from May to October. They probably migrate into Mexico, but this has not been established.

Recommendations: Information pertaining to location and size of extant colonies is needed. Most colonies will probably be located in caves and mines, but buildings may contain some colonies. Data on population size are essential for assessing future trends and making management decisions. Colonies of Cave Myotis located in mines that are to be reactivated will likely abandon the sites as disturbances increase in frequency. Because the bats will most likely leave a roost in autumn, the best time to bat-proof structures is during the winter months, November to March (Constantine, 1979).

Agency biologists working in the southeastern California desert area must be able to identify Myotis velifer. They should be familiar with the situation outlined here, and be aware of the sensitivity of colonies to disturbances. They should be instructed in methods of estimating population size and techniques of non-lethal control of bats in human-made structures (Barclay et al., 1980; Constantine, 1979). Public health, law enforcement, and agricultural officials should refer all reports of bats and all requests for information on eradication of bats to the Department of Health Services, Veterinary Unit in Berkeley.

Remarks: Dr. Patricia Brown (in litt.) has monitored a population of M. velifer in a mine in the Whipple Mountains since 1968. This population declined in size by over 50% between about 1960 and 1980. The decline was at least partly the result of vandals entering the mine and killing several bats. A gradual decrease in numbers has continued since that incident, however.

Hayward (1970) determined that M. v. brevis Vaughan, 1954, was not recognizably distinct from M. v. velifer. Hall (1981) did not cite Hayward's (1970) paper.

Distribution Records: IMPERIAL CO.: 4 mi S Potholes, 1 (MVZ); 5 mi NE Yuma, 5 (MVZ). RIVERSIDE CO.: Alice Mine, Riverside Mountains, 56 (LACM); Dollar Mine, 5.5 mi S, 0.6 mi E Vidal, 1 (CSLB); Mountaineer Mine, 5.5 mi S, 0.6 mi E Vidal, 750 ft, 1 (MVZ); Mule Mountains (P. Brown, in litt.); Riverside Mountains, 35 mi N Blythe, 81 (Vaughan, 1959); Riverside Mountains, NE corner of county, 1 (MVZ); Riverside Mountains, 6 mi S Vidal, 8 (MVZ); Riverside Mountains, 7 mi S Vidal, 10 (MVZ). SAN BERNARDINO CO.: Needles, 3 (MVZ); W side river, above Needles, 2 (USNM); Whipple Mountains (P. Brown, in litt.).

Riparian Brush Rabbit
Sylvilagus bachmani riparius

1935. *Sylvilagus bachmani riparius* Orr, Proc. Biol Soc. Washington, 48:29.
Type Locality: 2 mi NE Vernalis, Stanislaus Co., California.

Distribution: Orr (1940) collected specimens from a single locality along the west side of the San Joaquin River in northern Stanislaus County. He also observed, but did not collect, Riparian Brush Rabbits in adjacent San Joaquin County. Orr (1940) believed that the range of *S. b. riparius* extended along the San Joaquin River from Stanislaus County to the Delta region. The only presently known population is found on the lower Stanislaus River in Caswell State Park; there are probably other, tiny colonies between Caswell State Park and the confluence of the Stanislaus and San Joaquin rivers (Glenn Basey, in litt.).

Population Status: Only one moderate-sized population is known to be extant within Caswell State Park, San Joaquin County (G. Basey, in litt.). This population, which numbers perhaps less than 100 individuals at the start of the breeding season, was adversely affected by clearing and burning of brush in the Park in winter of 1984-85. Further development of recreation facilities in the Park and continuing brush clearing for fuel control pose significant threats to this population. Elsewhere between Caswell State Park and the San Joaquin River, Basey (in litt.) found only inconclusive evidence of tiny, scattered populations. Use of most of the property in the floodplain of the lower Stanislaus River is restricted by wildlife easements obtained as mitigation for loss of wildlife habitat as a result of construction of New Melones Dam. The Army Corps of Engineers is the administering agency for the easements. In his surveys, Basey (in litt.) found these wildlife areas were being used for rifle ranges, livestock grazing, off-road vehicle recreation, and other activities. Although Basey's studies are still in progress, activities such as these probably compound the threats to remaining populations of Riparian Brush Rabbits.

The Army Corps of Engineers is preparing to clear much of the brush and trees from within the lower San Joaquin River floodplain, between an area below Friant Dam, Fresno Co., and the delta in San Joaquin Co. (U.S. Fish and Wildlife Service, in litt.). Although no extant populations of Riparian Brush Rabbits have been found yet along the San Joaquin River, these activities pose a substantial threat to any remaining populations there and will virtually preclude reestablishment of populations in the future.

The major threat to remaining populations may stem from annual flooding. All land bordering the diked floodplains of the lower San Joaquin and Stanislaus rivers is cultivated or otherwise developed. In most areas, much or all of the woody vegetation has been removed for control of stream flow and to enhance growth of herbaceous plants for livestock grazing. In times of severe flooding, such as in the winters of 1979-80, 1982-83, and 1985-86, nearly all habitat available to Riparian Brush Rabbits is inundated. Animals living in the floodplain are forced to shelter on dikes or in areas lacking adequate cover, exposing them to increased predation and cold, wet weather. A single, severe episode of flooding, only slightly worse than was witnessed during March, 1986, could possibly cause the extinction of remaining populations.

The Riparian Brush Rabbit is currently designated a Resident Small Game species and may be hunted. Although the effect of hunting on populations of Brush Rabbits, in general, are probably insignificant, hunting might easily extirpate most of the small, scattered populations of Riparian Brush Rabbits outside of Caswell State Park.

Habitat: In general, brush rabbits are associated with chaparral or other types of dense brush (Chapman, 1974). Riparian Brush Rabbits occupy dense thickets of Wild Rose (*Rosa* sp.), Willows (*Salix* spp.), and Blackberries (*Rubus* sp.) growing along the banks of the river. Orr (1940) never observed Riparian Brush Rabbits in loose brush or in open fields. Studies by Glenn Basey (in litt.) indicated that, in summer when herbaceous plants were largest and provided densest cover, Riparian Brush Rabbits used weedy fields and clearings adjacent to patches of shrubs. These areas of herbaceous vegetation were not entered in autumn and winter when the plants were dead and cover was minimal or absent. Basey (in litt.) found that patches of herbaceous growth at the immediate edge (within less than a meter) of shrubs were important sources of food.

Distribution Records: SAN JOAQUIN CO.: Caswell State Park, 2 (CSCS); San Joaquin River, extreme southern part of county (Orr, 1935). STANISLAUS CO.: Kincaid's Ranch, 2 mi NE Vernalis, W side San Joaquin River, 2 (CAS), 3 (MVZ).

Point Arena Mountain Beaver
Aplodontia rufa nigra

1914. *Aplodontia nigra* Taylor, Univ. California Publ. Zool., 12:297.
Type Locality: Point Arena, Mendocino Co., California.

1918. *Aplodontia rufa nigra*, Taylor, Univ. California Publ. Zool., 17:479.

Distribution: *Aplodontia rufa nigra* is known only from the vicinity of Point Arena, Mendocino County. According to Taylor (1918), it originally occupied a total area of about 24 square miles. Camp (1918) stated that known colonies extended from the town of Point Arena to Alder Creek, 7.5 miles north of town.

Population Status: An investigation of the status of the Point Arena Mountain Beaver by Dale Steele is presently in progress (U.S. Fish and Wildlife Serv., in litt.). Some populations are threatened by housing developments and others are small and apparently vulnerable to any event that would alter the character of their habitat. The restricted distribution of this population and the fact that Mountain Beavers are often heavily persecuted by humans gives additional cause for concern. Logging may benefit Mountain Beavers in that logging opens up the forest and stimulates growth of deciduous trees and herbaceous plants. Urbanization generally results in the decline or local extirpation of Mountain Beavers. Home developments are likely in the range of the Point Arena Mountain Beaver.

Habitat: Specific information on the Point Arena Mountain Beaver was given by Camp (1918). Only north-facing slopes of ridges and gullies were found to be inhabited. In general, Mountain Beavers inhabit wooded regions along the Pacific Coast with abundant moisture and herbaceous plants for food. Soils must be sufficiently soft for burrowing. Mountain Beavers are generally most numerous on the slopes of ridges and gullies. They are usually associated with deciduous thickets with a tangle of undergrowth of both herbaceous and woody vegetation (Scheffer, 1929). The brushy successional stages of most coniferous communities (excepting the more arid, pine forests) provide the best habitat. Often, Mountain Beavers locate their burrows in areas of seepage springs. Some running water within their burrow systems may be required. Point Arena Mountain Beavers mostly occupy thickets of thimbleberries on north-facing slopes.

Mountain Beavers eat a great variety of herbaceous and woody plants (Scheffer, 1929). Although Mountain Beavers prefer succulent stems of herbaceous plants, little vegetation within their reach seems unacceptable for food or use in construction of nests. Generally, only the bark of woody plants is eaten, and most woody material is consumed in winter when herbaceous plants are unavailable. Mountain beavers cut a great deal of herbaceous material and leave it scattered around to dry. Much of this material spoils and is never eaten.

Recommendations: When available, the results of the field survey by Dale Steele, for the U.S. Fish and Wildlife Service, should be reviewed to determine if additional protection for this population is needed.

Remarks: This nearly black population of Mountain Beavers is apparently isolated from all others by unsuitable terrain (Hall, 1981; Taylor, 1918).

Distribution Records: MENDOCINO CO.: Point Arena, 3 (CM), 6 (MVZ); Point Arena, Christensen Ranch, 2 (MVZ); 5 mi N Point Arena, 1 (UDAV).

Los Angeles Pocket Mouse
Perognathus longimembris brevinasus

1900. *Perognathus panamintinus brevinasus* Osgood, N. Amer. Fauna, 18:30.
Type Locality: San Bernardino, San Bernardino Co., California.
1928. *Perognathus longimembris brevinasus*, Huey, Trans. San Diego Soc. Nat. Hist., 5:88.

Distribution: The geographic range of Los Angeles Pocket Mice is restricted to lower elevation grasslands and Coastal Sage associations in the Los Angeles Basin, from approximately Burbank and San Fernando on the northwest to San Bernardino on the northeast, and Cabazon, Hemet, and Aguanga on the east and southeast. Their geographic limits on the southwest are not clear, but probably lie somewhere near the Hollywood Hills.

Population Status: Data on the population status of *P. l. brevinasus* are not available. Urbanization and cultivation of the majority of the land within the interior valleys of the Los Angeles Basin have made a large percentage of its historic range uninhabitable. Stephens (1906) noted that Los Angeles Pocket Mice were usually rare, but were infrequently abundant, probably following years of much greater than average precipitation.

Small populations probably still exist in isolated parts of the historic range of P. l. brevinasus. The extent of available habitat cannot be estimated without a better understanding of its habitat requirements.

Habitat: Nothing specific has been recorded about the habitat requirements of P. l. brevinasus. Grinnell (1933) stated that it inhabited open ground with soils composed of fine sands. Stephens (1906) remarked that he thought that Los Angeles Pocket Mice did not often dig burrows. He said that they hid under weeds and dead leaves instead. This would be unusual for any species of Perognathus should it prove to be true.

Recommendations: Field notes of persons who surveyed the available habitat of Stephens' Kangaroo Rat may include some data on recent occurrence and abundance of P. l. brevinasus; information should be sought from them. Efforts should be made to determine the habitat associations and current distribution of this subspecies.

Remarks: Huey (1939) stated that specimens from Cabazon were not typical of P. l. brevinasus. The specimens from White Water Ranch may represent P. l. bangsi. I believe that specimens from near Ranchita (2 mi E, Culp Valley) and Warner Pass, San Diego Co., stated to be P. l. brevinasus by Bond (1977), are P. l. internationalis. Huey (1939) assigned the specimens from "Warner's Pass" to P. l. internationalis, but did not mention those from near Ranchita.

The Perognathus longimembris species group is in need of taxonomic revision. The characteristics diagnostic of P. l. brevinasus, especially the short rostrum, are nearly unique among the subspecies of P. longimembris. Osgood (1900) remarked that this taxon possibly represented a distinct species. If this is the case, P. l. brevinasus is probably conspecific with P. longimembris pacificus, which has a contiguous range along the coast and which also has a short rostrum.

Distribution Records: LOS ANGELES CO.: Burbank, 1 (USNM); Garney, San Fernando Valley, 1 (MVZ; see also Grinnell and Swarth, 1913); Hyperion [= El Segundo], 1 (LACM); San Fernando, 1 (MVZ). SAN BERNARDINO CO.: Cajon Wash, 9 (LACM); N of Etiwanda, 1 (LACM); Ferndale (Osgood, 1900); Reche Canyon, 1250 ft, 4 mi SE Colton, 1 (MVZ); mouth Reche Canyon, near Colton, 1 (MVZ), 44 (USNM); 4.75 mi N San Bernardino, 1600 ft, 3 (MVZ); 5 mi NW San Bernardino, 32 (SDSNH); Slover Mountain, near Colton, 1 (MVZ). SAN DIEGO CO. : 2.5 mi N Oak Grove, 7 (SDSNH). RIVERSIDE CO. : Aguanga, 3 (SDSNH); 0.25 mi ENE Aguanga, 2050 ft, 1 (MVZ); 1 mi SE Aguanga, 2150 ft, 3 (MVZ); Banning, base San Jacinto Mountains, 2 (MVZ), 2 (USNM); 2 mi E, 0.5 mi S Banning, 2100 ft, 1 (MVZ); 2 mi E, 3 mi N Beaumont, 3000 ft, 1 (MVZ); Cabazon, 70 (SDSNH); near Cabazon, base San Jacinto Mountains, 19 (MVZ); 2 mi NE Cabazon, 2 (CSLB); 0.25 mi E Cabazon, 1800 ft, 2 (MVZ); 1 mi S Cabazon, 3 (LACM); 2 mi S Cabazon, 7 (LACM); 3 mi S Cabazon, 2 (SDSNH); 1 mi N, 2 mi W Cabazon, 2200 ft (MVZ); Dos Palmas Spring, Santa Rosa Mountains, 3500 ft, 4 (MVZ); Eden Hot Springs, 1500 ft, 1 (MVZ); Hemet, 1 (USNM); Menifie, 27 (LACM); Vallevista, San Jacinto Valley, 1800 ft, 8 (MVZ); White Water Ranch, 2 (SDSNH); Winchester, 3 (SDSNH).

Pacific Pocket Mouse
Perognathus longimembris pacificus

1898. Perognathus pacificus Mearns, Bull. Amer. Mus. Nat. Hist., 10:299. Type Locality: Mexican boundary monument No. 258, shore of Pacific Ocean, San Diego Co., California.

1932. Perognathus longimembris pacificus, von Bloeker, Proc. Biol. Soc. Washington, 45:128.

Distribution: P. l. pacificus inhabits the narrow coastal plains from the vicinity of the Mexican border, northward to El Segundo, Los Angeles Co., California.

Population Status: The present status of P. l. pacificus is unknown. Field work on the Irvine Ranch, San Joaquin Hills, Orange County (M'Closkey, 1972; Meserve, 1976) showed these mice to be scarce. This area is now urbanized. This subspecies may not be endangered, but habitat loss due to highways, urbanization, and off-road vehicle activities has likely been extensive. Probably all populations north of the San Joaquin Hills are extinct. This may also be the case from San Diego south, although some small areas near the Tiajuana River may still be inhabited.

Habitat: The types of soil and plant communities required by Pacific Pocket Mice are not known. Several authors noted that these Pocket Mice were found only on soils of fine, alluvial sands near the ocean (Bailey, 1939; Grinnell, 1933; Mearns, 1898; von Bloeker, 1931). M'Closkey (1972) and Meserve (1976) found Pacific Pocket Mice to be rare in a Coastal Sage-scrub community in a dry, rocky and gravelly site in the San Joaquin Hills of Orange Co. M'Closkey (1972) remarked that P. l. pacificus had an unpredictable pattern of residence there. Only open spaces in an otherwise dense, weedy area were occupied near San Diego (von Bloeker, 1931).

Principal plant species at several sites inhabited by P. l. pacificus were Telephone Weed, Foxtail, Saltgrass, Ice Plant, Star Thistle, and Arrow Weed (von Bloeker, 1930). Pacific Pocket Mice apparently eat mostly seeds of grasses and forbs (Meserve, 1976; von Bloeker, 1931), although other plant materials are consumed in smaller quantities (Meserve, 1976).

Recommendations: Information on areas of present occurrence and estimates of population densities are needed. Sites with favorable habitat may be rare, and these may serve as the only reservoirs from which individuals disperse to areas of lower quality habitat. A live-trapping program designed to provide data on distribution and abundance should be carried out. All persons undertaking field work on small mammals in the area should be encouraged to gather needed information on Pacific Pocket Mice. Public agencies administering land within the range of P. l. pacificus should be informed of the need for information and of the sensitivity of Pacific Pocket Mice to further loss of habitat. The Camp Pendleton Marine Corps Base may be the only area where habitat for P. l. pacificus can be protected relatively easily.

Remarks: Some of the localities, listed below, in San Diego County probably refer to the same place (e.g., 4 mi N Oceanside; sand dunes near Oceanside; mouth of Santa Margarita River). Specimens listed from

Tiajuana Valley by Huey (1939) are probably the same ones that von Bloeker reported on from "near the type locality."

The systematic relationships of the P. longimembris group needs to be reviewed, especially the relationships among P. l. pacificus, brevinasus, bangsi, internationalis and bombycinus of southern California. These taxa, and others in Baja California, possibly represent a species distinct from P. longimembris from the Mojave Desert and the Great Basin. P. l. cantwelli is a synonym of P. l. pacificus (Huey, 1939).

Distribution Records: LOS ANGELES CO.: Cliffton, 100 ft, 1 (MVZ); Del Rey, 6 (LACM); Del Rey Hills, near Loyola, 2 (LACM); El Segundo, 8 (SDSNH); 1 mi N El Segundo, 8 (SDSNH); 0.5 mi NW El Segundo, 1 (MVZ); Hyperion [= El Segundo], 50 ft, 6 (MVZ); Palisades del Rey (von Bloeker, 1932); Paya Del Rey, 6 (LACM); Wilmington, 3 (MVZ). ORANGE CO.: Dana Point, 5 mi W Capistrano Beach, 10 (LACM); Irvine Ranch, between Buck Gully and San Joaquin Reservoir, San Joaquin Hills (M'Closkey, 1972; Meserve, 1976); near San Juan Capistrano Point (Grinnell, 1933); Star Ranch, N of San Juan Capistrano Beach (V. Bleich, pers. comm.). SAN DIEGO CO.: Oceanside, 3 (LACM), 6 (MVZ), 38 (SDSNH); 4 mi N Oceanside, 8 (LACM), 7 (SDSNH); sand dunes near Oceanside (Bailey, 1939); shore of Pacific Ocean, near U.S.-Mexican boundary monument #258, 11 (MVZ), 3 (USNM); San Onofre, 1 (SDSNH); 2 mi E San Onofre (von Bloeker, 1930); mouth of Santa Margarita River, near Oceanside, 1 (USNM); near mouth Tia Juana River, 4 (SDSNH); Tia Juana Valley, 79 (SDSNH); 2 mi N U.S.-Mexican boundary monument #258, 17 (LACM).

Tipton Kangaroo Rat
Dipodomys nitratoides nitratoides

1894. Dipodomys merriami nitratoides Merriam, Proc. Biol. Soc. Washington, 9:113. Type Locality: Tipton, San Joaquin Valley, Tulare Co., California.

1921. Dipodomys nitratoides nitratoides, Grinnell, J. Mamm., 2:96.

Distribution: The historic distribution of Tipton Kangaroo Rats included most of the San Joaquin Valley floor in the Tulare Basin, from approximately Lemoore and Hanford in Kings Co. on the north, to Visalia, Tipton, Delano and Bakersfield on the east, and to the edge of the alkaline-sink plant communities on the west. Present distribution is limited to a few remaining areas of uncultivated ground, mostly with alkaline soils (Williams, 1985). The western boundary was defined by Williams (1985) as being approximately coincident with the route of the California Aqueduct. D. n. nitratoides apparently interbred with the Short-nosed Kangaroo Rat (D. n. brevinasus) along the southern and western edges of the valley floor, but the geographic ranges of the two subspecies are no longer in contact (Hafner, 1979; Williams, 1985).

Population Status: Hafner (1979) determined that populations of Tipton Kangaroo Rats were extant on parts of the Kern and Pixley National Wildlife Refuges and in several small areas still uncultivated in 1978. The only large block of habitat was located in the area east and north of Buttonwillow, Kern County. Williams (1985) did not find Tipton Kangaroo Rats on the Kern National Wildlife Refuge in 1985, and speculated that they may have been extirpated by floods that inundated even the dikes on the refuge during Winter 1982-83. Williams (1985) estimated that the historical geographic range of the Tipton Kangaroo Rat encompassed approximately 695,174 hectares (1,716,480 ac.). By July, 1985, the area inhabited had been reduced, primarily by cultivation, to about 25,665 ha. (63,367 ac.), only about 3.7% of the historical acreage. The loss of habitat was expected to continue until there is none privately owned, perhaps within the next four or five years. Only 2486 ha. (6137 ac.) of federally administered land was found to have historically supported populations; Tipton kangaroo rats may have been extirpated from 1053 ha. of this total between 1982 and 1985. Of all local, state and federal government-administered lands, only about 2606 ha. (6434), divided among five separate parcels, had populations of low- to moderate-densities of Tipton kangaroo rats that were relatively secure from loss of habitat to cultivation or extirpation due to flooding. These areas probably were not large enough to support populations sufficient in numbers to prevent continuing loss of genetic diversity and subsequent extinction. Other potential threats to small, isolated, remnant populations included diseases, predation and poisoning by rodenticides (Williams, 1985).

Habitat: Tipton Kangaroo Rats are limited to arid-land communities occupying the valley floor of the Tulare Basin in level or nearly level terrain. They occupy alluvial fan and floodplain soils ranging from fine sands to clay-sized particles (because of the high alkalinity of these soils, some of the finer-textured soils tend to be powdery when dry rather than hard-packed). Generally, woody shrubs of one or more species are sparsely scattered over occupied terrain with scant-to-moderate ground cover of grasses and forbs. Woody shrubs commonly associated with Tipton Kangaroo Rats are: *Atriplex spinifera*, *A. polycarpa*, *A. phyllostegia*, *A. lentiformis*, *Allenrolfea occidentalis*, *Haplopappus acradenius*, and *Prosopis juliflora*. A conspicuous semiwoody species is *Suaeda fruticosa* (Williams, 1985).

In areas with vernal pools and alkaline playas, Tipton Kangaroo Rats place their burrows in any elevated terrain available, such as where wind-blown soil particles have accumulated around some obstruction or on slight ridges, usually no more than 0.5 to 1.0 meter above the playa beds. Tipton Kangaroo Rats sometimes colonize areas that are flooded in winter and spring. Favored areas include iodine bush shrublands which are flooded seasonally and where alkaline water lies close to the surface of the soil, year around. Presumably, these individuals are either drowned or escape to higher ground when the floods return (Williams, 1985).

Recommendations: The most important need is for preservation of relatively large blocks of habitat on the valley floor in the Tulare Basin where this species still lives. Most remaining uncultivated parcels of habitat may be to small to support populations of Tipton Kangaroo Rats indefinitely (Williams, 1985). Williams (1985) recommended that federal Endangered Species status be sought for *D. n. nitratoides*. Review of the status of the Tipton Kangaroo Rat for state Endangered Species designation is called for.

Remarks: Records of distribution from along the western edge of the San Joaquin Valley listed by Hafner (1979) and in the draft version of this report have been excluded below. These specimens came from areas with habitat typical of D. n. brevinasus; it was not clear from Hafner's report if the specimens were D. n. nitratoides or D. n. brevinasus. Williams (1985) reviewed the status of these specimens and determined they were best assigned to brevinasus. The specimens listed below from Tulare Lake (USNM) had no precise locality information. They could have come from either Kings or Tulare counties. They are listed under Kings County because others in the U.S. National Museum were collected in that county. Williams (1985) reported that extant populations of Tipton Kangaroo Rats probably occurred on at least 154 separate parcels in 54 separate blocks of uncultivated land in June, 1985; these localities are not listed below.

The taxonomy of the subspecies of D. nitratoides was reviewed by Hoffmann (1975) and Hafner (1979).

Distribution Records: KERN CO.: Adobe Station, 4 (USNM); Bakersfield, 1 (MVZ); 8 mi NE Bakersfield, 15 (MVZ); 12 mi SW Bakersfield, 1 (MVZ); 5 mi W Bakersfield, 1 (MVZ); 19 mi SE Bakersfield, 4 (CSUF); 20 mi S, 8 mi W Bakersfield, 1 (MVZ); 3 mi N Buena Vista Lake, 5 (MVZ); Buena Vista dry Lake bottom, 15 mi NE Taft, 4 (CSLB); east side levee, 298 ft, Buena Vista Lake, 4 (MVZ); S of Buena Vista Lake Reservoir, near Taft, 1 (USNM); Buttonwillow, 15 (USNM); 4.6 mi E Buttonwillow (M. Hafner); 9.6 mi E Buttonwillow (M. Hafner); 15 mi S Corcoran, 19 (MVZ); 0.5 mi N, 1 mi E Corners [Conners ?], 6 (LACM); Delano, 16 (USNM); 1 mi S, 5.4 mi W Delano (M. S. Hafner); 1.4 mi S, 8.3 mi E Edison (M. S. Hafner); Famoso, 1 (USNM); 2.5 mi S, 6.3 mi E Famoso (M. S. Hafner); Kern National Wildlife Refuge, 18 mi W Delano, 2 (LACM); 2.5 mi N, 6.2 mi E Lost Hills (M. S. Hafner); 7 mi N, 5.3 mi E Lost Hills (M. S. Hafner); 7 mi N, 6.3 mi E Lost Hills (M. S. Hafner); 5.2 mi N, 1.5 mi W Mettler (M. S. Hafner); 7 mi W Old River (M. S. Hafner); 1 mi N Pond, 1 (MVZ); 4. mi S, 0.5 mi E Shafter (M. S. Hafner); 1.5 mi W Tupman, 2 (LACM); 1.2 mi S, 1.5 mi E Tupman (M. S. Hafner). KINGS CO.: 15 mi S Corcoran (Grinnell, 1933); 3.5 mi S, 2 mi W Lemoore (M. S. Hafner); Tulare Lake, 13 (USNM); W side Tulare Lake, 1 (USNM); Lemoore Naval Air Station (T. O'Farrell, pers. comm.). TULARE CO.: Alila [= Earlimart], 16 (USNM); 1.5 mi NE Angiola, 3 (CSUF); 2 mi NE Angiola, 39 (CSUF); 1.3 mi N, 5 mi W Delano (M. S. Hafner); Earlimart, 7 (LACM); 1 mi N Earlimart, 2 (MVZ); 2 mi W Earlimart, 5 (MVZ); 3.5 mi S, 5 mi W Pixley (M. S. Hafner); Tipton, 266 ft, 43 (MVZ); 2 mi NE Tipton, 1 (CSUF); 7 mi NE Tipton, 1 (MVZ); 3.5 mi S, 8 mi W Tipton (M. S. Hafner); 10 mi SW Tipton, 3 (CSUF); 12 mi SW Tipton, 2 (CSUF).

Colorado River Cotton Rat
Sigmodon arizonae plenus

1928. Sigmodon hispidus plenus Goldman, Proc. Biol. Soc. Washington, 41:205. Type Locality: Parker, 3500 ft, Yuma Co., Arizona.
1970. Sigmodon arizonae plenus, Zimmerman, Publ. Mus., Michigan State Univ., Biol. Ser., 4:435.

Distribution: Sigmodon arizonae plenus is found in California in marshy and dense, grassy areas along the Colorado River floodplain from the Nevada border to about the level of Bard. Goldman (1928) listed only three localities for S. arizonae in California: Needles; Colorado River opposite Parker, Arizona; and 15 mi SW of Ehrenberg, Arizona. The distribution was apparently spotty, rather than continuous.

Population Status: This population is known to be extant only in one area in California (Brad Blood, in litt.). In 1965, Bradley (1966) collected along the Colorado River in Nevada, but caught no Cotton Rats. The marsh where he and Hall (1946) had caught Cotton Rats had dried up and was largely devoid of Cattails. Bradley (1966) surmised that changes in vegetation resulted from absence of flooding due to channeling of the river and restriction of river flow while filling Lake Powell. He suggested that S. arizonae was extinct In Nevada.

Habitat: Goldman (1928) noted that this subspecies appeared to be restricted to "isolated sections of alluvial bottom along the Colorado River." The climate is too hot and arid to support Cotton Rats anywhere except immediately adjacent to the river. Within this habitat, Cotton Rats inhabit areas supporting Sedges, Rushes, Cane, and other grass-like plants. Hall (1946) collected Cotton Rats from a small marsh in Nevada supporting Cattails and Bermuda Grass and ringed by Mesquite. The marsh was 0.5 mi N of the California border, along the Colorado River. Bradley (1966) collected in the same area in 1961 and found no Cotton Rats, and none were found in nearby areas with "riparian vegetation."

Recommendations: Extant populations in California should be investigated further to determine their status. Efforts to preserve existing habitat for this species should be initiated. The feasibility of securing additional habitat along the Colorado River should be investigated.

Remarks: Brad Blood and David Huckaby of California State University, Long Beach, investigated the status of S. arizonae in California. Localities listed below without numbers of specimens refer to specimens that Brad Blood (in litt.) identified as S. arizonae.

Zimmerman (1970) showed that Sigmodon hispidus and Sigmodon arizonae differed greatly in chromosome number and structure, and were distinguishable in skeletal structure. S. arizonae plenus has two more chromosomes than S. a. arizonae. Severinghaus and Hoffmeister (1978) reconfirmed the structural distinctness of the two species and discussed methods for identification of skulls.

Distribution Records: IMPERIAL CO.: near Bard, 7 mi NE Ft. Yuma, 1 (MVZ); 2 mi above Laguna Dam, 3 (MVZ); Palo Verde, 3 (MVZ); 0.5 mi S Palo Verde (B. Blood - CSLB). RIVERSIDE CO.: Blythe, 1.5 mi W Colorado River, 8 (MVZ); 3 mi NE Blythe (B. Blood - CSLB); 15 mi SW Ehrenberg, 3 (USNM); Neighbors Road (B. Blood - LACM); opposite Riverside Mountains, Colorado River (B. Blood - LACM). SAN BERNARDINO CO.: 2 mi E Earp, 4 (CSLB); Colorado River, opposite Parker, Arizona, 25 (MVZ), 1 (USNM); Colorado River, 33 mi N Blythe, off Highway 62, via Earp, 2 (CSLB); Needles, 1 (USNM); 4 mi NE Parker, Arizona, on Colorado River, 2 (CSLB).

Yuma Mountain Lion
Felis concolor browni

1903. Felis aztecus browni Merriam, Proc. Biol. Soc. Washington, 16:73.
Type Locality: Colorado River, 12 mi below Yuma, Arizona.
1929. Felis concolor browni, Nelson and Goldman, J. Mamm., 10:347.

Distribution: Yuma Mountain Lions have been recorded as occurring at low elevations in the Colorado River Valley of southeastern California, southwestern Arizona, and adjacent areas in Sonora and Baja California (Currier, 1983; Young and Goldman, 1946). Halloran (1946) reported on a specimen which was trapped on the Kofa Game Range, 50 mi NE Yuma, Yuma Co., Arizona. The area was described as rough, desert terrain with an elevation of about 1000 ft. F. c. browni probably occupies the desert areas of southeastern California, perhaps ranging as far north as the Amargosa Mountains of California and the McCullough Range of southern Nevada. Its range extended westward into the Imperial Valley. Grinnell (1933) listed an occurrence at Calexico. Weaver noted occurrence of Mountain Lions from a few areas of the desert ranges of southeastern California: New York Mountains (Weaver et al., 1969); Little San Bernardino Mountains (Weaver and Hall, 1971); near Picacho State Recreation Area (Weaver and Mensch, 1969). Leslie and Douglas (1979) reported sight records of Mountain Lions on the east side of the River Mountains, near Lake Mead, Nevada. Halloran (1946) mentioned a few reports of sightings from the Imperial National Wildlife Refuge during the 1940's.

Population Status: No current information is available. Bounties on only two Mountain Lions from Imperial County were collected between 1907 and 1950 (McClean, 1954), suggesting that at least in that county, mountain lions were very rare. Because these bounty records did not specify area within counties, it is not known if the animals from Riverside and San Bernardino counties were Yuma Mountain Lions. Most of the riparian floodplain forest of the lower Colorado River and the woody growth along desert washes have diminished or disappeared in this century.

Habitat: Grinnell (1933) stated that Yuma Mountain Lions lived mostly in the dense vegetation of the bottomland along the Colorado River, and also noted their presence in adjacent, rocky uplands. The habitat requirements of F. c. browni are essentially unknown. Mountain Lions in adjacent regions prefer brush and timber in rocky terrain (McLean, 1954). Most of the desert ranges of southeastern California are too arid to support more than sparse brushlands or Juniper woodlands. Adequate numbers of deer for food are another usual requirement (Currier, 1983). Weaver (in litt.) thought that Burro Deer (Odocoileus hemionus eremicus), principally inhabitants of bottomlands and arroyos vegetated with Willows, Mesquite, Acacia, Ironwood, and Palo Verde, were the staple food of Yuma Mountain Lions. Merriam (1903) remarked that the unusually small teeth and "weak" rostrum of F. c. browni indicated smaller species of prey than the deer typically preyed upon by adjacent populations of Mountain Lions. Burrow Deer are larger in size than adjacent populations of Mule Deer (Cowan, 1936). The tiny-sized Coue's White-tailed Deer, which was perhaps the most abundant deer species along the Colorado River in prehistoric times, may have been the major prey species for F. c. browni. Yuma Mounrain Lions probably occasionally prey upon Mountain Sheep, Burros, rabbits and hares, various rodents, and calves, in addition to Burro Deer.

Recommendations: Field surveys for Mountain Lions in the eastern halves of Imperial, San Bernardino, and Riverside counties should be conducted to determine their current status. Information on the principal food of Yuma Mountain Lions is needed to determine current and future prospects for survival and to develop a plan to ensure the recovery of F. c. browni.

The geographic range of F. c. browni in California should be more clearly defined. The racial characters of this subspecies are apparent externally (Young and Goldman, 1946; Grinnell, 1914; Merriam, 1903); thus, examination of live-captured individuals should be sufficient to confirm racial identity. Systematic review of extant specimens from the California desert area should be carried out.

Mountain Lions in the California desert area should continue to be completely protected. Any individuals preying on livestock should be captured alive, if possible, and relocated, rather than killed.

Protection of riparian communities along the Colorado River should be undertaken as soon as possible. The Bureau of Land Management, U.S. Fish and Wildlife Service, and U.S. Department of Defense should be informed of the need for information about Yuma Mountain Lions on lands they administer within the potential range of F. c. browni.

Remarks: According to Goldman (Young and Goldman, 1946), F. concolor browni is medium-sized, very pale-colored, and has short, sparse pelage. Goldman noted that this population intergraded "along the mountain barrier west of the California desert area."

See the Distribution section for circumstantial records of occurrence.

Distribution Records: IMPERIAL CO.: Colorado River, 20 mi N Picacho, 1 (MVZ); Calexico (Grinnell et al., 1937).

SECOND PRIORITY LIST

Southern California Salt Marsh Shrew
Sorex ornatus salicornicus

1932. *Sorex ornatus salicornicus* von Bloeker, Proc. Biol. Soc. Washington, 45:131. *Type Locality*: Playa del Rey, Los Angeles County, California.

Distribution: Southern California Salt Marsh Shrews are confined to the coastal marshes in Los Angeles, Orange, and Ventura counties. Known occurrence extends from Point Mugo, Ventura County on the north to the salt marshes around Anaheim Bay and Newport Beach in Orange County, on the south.

Population Status: Nothing is known about the status of the Southern California Salt Marsh Shrew. Loss of habitat to human developments along the coast has been significant, and populations probably have declined as a result. Lack of refuge sites above the marshes to escape from flooding during seasonal high tides and periodic storms may be the most crucial factor threatening extant populations.

Habitat: Nothing has been recorded concerning habitat requirements of the Southern California Salt Marsh Shrew, other than that individuals were captured in coastal marshes (von Bloeker, 1932). In general, Ornate Shrews are insectivorous; the populations living in salt marshes probably subsist primarily on amphipods, isopods, insects, and other invertebrates. They probably require fairly dense ground cover, nesting sites above mean high tide and free from inundation, and fairly moist surroundings, based on studies of other shrews in similar habitats elsewhere (Johnston and Rudd, 1957).

Recommendarions: Information is needed on the current status, habitat requirements, and extent of available habitat. Threats to these populations need to be identified and steps taken to ensure the survival of Southern California Salt Marsh Shrews. Pitfalls used as livetraps are effective for capturing shrews, but pitfalls are nearly impossible to keep in the ground in tidelands. Other effective methods of surveying populations are searching for nests (Johnston and Rudd, 1957) and shrews sheltering under debris (Ford, 1986), and, to a lesser extent, use of Sherman live traps. Surveys could be carried out in conjunction with studies of *Reithrodontomys megalotis limicola*, another Species of Special Concern sharing these saltmarsh communities.

Remarks: These communities are also used by the Light-footed Clapper Rail (*Rallus longirostris levipes*) and some potential habitat for Southern California Salt Marsh Shrews is currently in Department of Fish and Game "ecological reserves" (Upper Newport Bay and Bolsa Chica). Shrews have not been reported from marshes in Bolsa Bay, but they should be expected to occur there. Specimens in the San Diego Society of Natural History collection from Big Bear Lake and Lytle Creek, San Bernardino Co., identified as *Sorex ornatus salicornicus*, are *S. o. ornatus*.

Distirbution Records: LOS ANGELES CO.: Nigger Slough, 1 (von Bloeker, 1932); Playa del Rey, 2 (LACM), 1 (UCLA), 1 (SDSNH), 5 (von Bloeker, 1932). ORANGE CO.: S side Newport Bay, 1 mi above Coast Highway, 2 (MVZ). VENTURA CO.: Point Mugu, 1 (LACM).

California Leaf-nosed Bat
Macrotus californicus

1859. *Macrotus californicus* Baird, Proc. Acad. Nat. Sci.. Philadelphia, 10:116. Type Locality: Old Fort Yuma, Imperial Co., California.

Distribution: California Leaf-nosed Bats occur from southern Nevada, southern California and Western Arizona southward through Baja California Sur and Sonora, Mexico (Hall, 1981). In California, they occupy the low-lying desert areas of southern California, including the coastal basins.

Population Status: *Macrotus californicus* has disappeared from most areas of the coastal basins from Los Angeles to San Diego. Elsewhere in southern California, populations have declined, but appear to have stabilized (P. Brown, in litt.).

Loss of foraging habitat in the coastal basins is probably a principal factor in the decline of populations. Elsewhere, disturbances of roosts may be the primary cause of decline. Grinnell (1918) discussed an episode which suggests that maternity colonies are sensitive to disturbance; one colony apparently abandoned a cave after biologists had visited and collected bats. Judging from the description of depth of guano, the cave had been used for a long time. Wintering bats are probably even more sensitive to disturbances, as sites suitable for overwintering colonies are probably few. Disturbances cause extra activity and elevated metabolism, resulting in extra expenditure of energy at a time when energy economy may be critical to successful overwintering.

Habitat: In California, Leaf-nosed Bats are found in lowland desert associations. They appear to be limited to areas with suitable day-roosts, which must provide shelter from excessive heat and aridity. Howell (1920a) stated that California Leaf-nosed Bats were common in the desert only in caves and old mines, especially along the old shore line of the Salton Sea bed, near sea level. He remarked that the sexes roosted together only during the mating season. Barbour and Davis (1969) indicated that copulation occurred from September to November.

Vaughan (1959) studied *M. californicus* in the Riverside Mountains near the Colorado River. There, Leaf-nosed Bats occupied day-roosts only in caves and deserted mine tunnels. He found a day-roost in a cabin near the Colorado River, however. Leaf-nosed Bats roosted in groups of several to 100 or more. Most groups were encountered within 30 to 80 ft from the entrances of tunnels, and seemed not to require darkness. Vaughan believed that day-roosts of *M. californicus* must be enclosed and have overhead protection from the weather. Favored sites were large enough for bats to enter by flying because these bats cannot crawl. They also provided considerable ceiling surface and flying space (individuals are not in contact while hanging).

Temperatures in day-roosts are warm, but considerably cooler than ambient day-time temperatures outside the roost (84 vs 110 deg. F, respectively, Vaughan, 1959). California Leaf-nosed Bats occupy a great variety of night roosts, where they rest and consume captured prey. The only common feature Vaughan (1959) could detect of night-time roosts was adequate overhead protection -- old adobes, deserted wooden buildings, cellars, porches and a wide variety of caves and mine tunnels were used.

Leaf-nosed Bats apparently are entirely insectivorous in California. Food includes a diversity of flightless and flying arthropods (Barbour and Davis, 1969; Vaughan, 1959). Vaughan noted that Leaf-nosed Bats hovered frequently and flew low and slowly over the ground while foraging. Generally, they flew within 3 feet of the ground over dry washes and nearby flats. They also foraged close to vegetation. Vaughan surmised that they picked insects off of the vegetation and the substrate while hovering. California Leaf-nosed Bats may travel up to a mile from their day-roosts while foraging, but Vaughan suggested that their foraging range was small, perhaps within a few hundred yards of the roost on frequent occasions.

Some California Leaf-nosed Bats may migrate out of the state (probably to Mexico) during winter; P. Brown (in litt.) stated that many are resident. They do not hibernate. Stephens (1906) failed to find bats of this species in winter, but could always locate them in spring and summer. Grinnell (1918) reported that March and September were the earliest and latest dates, respectively, for M. californicus in California. She noted, however, that one specimen in the Museum of Vertebrate Zoology was collected on 31 December in a mine near Palo Verde, and recounted information on presence of Leaf-nosed Bats on 20 February.

Recommendations: Reliable quantitative data are greatly needed on population size, distribution, and currently used maternity and overwintering roosts, if the latter are found in California. Trends cannot be unequivocally established until these baseline data are gathered. United States Geological Survey maps are a good source of information for locating abandoned mines. The old shore line of the Gulf of California around the perimeter of the Salton Sink contains many shallow grottos and caves, as does that along the Colorado River lowlands. Careful search, while avoiding excessive disturbances, should reveal colonies. Colony size can be estimated by counting emerging bats (usually, Leaf-nosed Bats emerge about 1 to 1.5 hours after sunset).

Protection of nursery colonies from disturbances by humans is extremely important. The means of accomplishment would depend upon the situation. The entrances to tunnels could probably be sealed off to prevent access by humans but permit entry by bats (Leaf-nosed Bats must be able to enter by flying). Some important roosting areas might more simply be closed to access by vehicles. This method, however, is unlikely to reduce sufficiently the level of disturbances in many areas, and is impractical in many others.

A carefully designed campaign to inform the public of the usefulness of bats and their sensitivity to disturbances may be helpful. Department of Fish and Game personnel in Southern California should be informed about the need for preservation of this and other species of bats. Agency personnel in field positions should be trained to recognize M. californicus, as well as all other species, and should receive instructions on methods of estimating colony size and control of bats in man-made structures (see Barclay et al., 1980; Constantine, 1979). They should be informed of the sensitivity of colonies to disturbances, and use only approved methods of observation and control, when necessary. Persons in agencies such as public health, law enforcement, and agriculture should be similarly informed.

Remarks: Grinnell (1918) listed the localities, Santa Margarita Ranch and River, as being in San Diego County, whereas Riverside County was listed on the specimen tags. She also listed the localities, Indian Wells and Mecca, as being in Imperial County, but they are actually located in Riverside County.

The systematic status of Macrotus californicus has been a subject of disagreement among researchers. Anderson and Nelson (1965) believed that the evidence they examined demonstrated that californicus was a subspecies of M. waterhousii. Davis and Baker (1974) interpreted additional morphometric and karyotypic evidence as showing specific distinctness of these taxa. Hall (1981) treated californicus as a subspecies of waterhousii because of "inconsistencies" in the literature. I believe the evidence favors specific recognition.

Distribution Records: IMPERIAL CO.: Clapp Mine, 15 mi NE Yuma, 5 (MVZ); Ft. Yuma, 18 (CAS), 28 (KU), 14 (MVZ), 2 (PM), 1 (UDAV), 1 (USNM), 1 (TCWC); 6 mi SW Ft. Yuma, 10 (KU); Palo Verde, 4 (MVZ), 1 (SDSNH); 24 mi S Palo Verde, 1 (MVZ); Picacho Mine, 16.5 mi N Ft. Yuma, 1 (MVZ); Potholes, 25 (CM), 3 (SDSNH); 1 mi N Potholes, 1 (SDSNH); 2 mi N Potholes, 200 ft, 34 (MVZ); 3 mi N Potholes, 4 (KU); Senator Mine, 10 mi N Potholes, 11 (SDSNH); Tumco Mine, 5 mi N, 2 mi E Ogilby, 9 (UDAV), 6 (MVZ). LOS ANGELES CO.: Iverson Ranch, Santa Susanna Pass, Chatsworth, 1 (MVZ); 2 mi N Owensmouth (Howell, 1920). RIVERSIDE CO.: Alice Mine, 7 mi S Vidal, 2 (MVZ); Indian Wells, 2 (MVZ), 4 (USNM); Mecca, 2 (MVZ); Mountaineer Mine, 5.5 mi S, 0.6 mi E Vidal, 2 (CSLB); Mountaineer Mine, 5.6 mi S, 0.3 mi E Vidal, 14 (MVZ); Mountaineer Mine, T2S, R24E, 5 (MVZ); Neighbors, 11 (LACM); Riverside Mountains, 35 mi N Blythe, 22 (KU); Riverside Mountains, 7 mi S Vidal, 86 (MVZ); Santa Margarita River, 17 (CAS); W of State Highway 95, 9 (UDAV); Torres [Toro], 61 (MVZ); cave near Torres [Toro], 2 (KU); 5 mi E Vidal, 1 (UDAV); 5.3 mi S Vidal, 7 (CAS). SAN BERNARDINO CO.: 2 mi N Colorado River, 0.2 mi W Nevada boundary, 1000 ft, 2 (MVZ). SAN DIEGO CO.: Anza Borrego State Park, 2 (SDSNH); Artery Mine, Dulzura, 1 (SDSNH); De Luz, 8 (CAS); 2 (USNM); Donahue Mines, near Dulzura, 2 (LACM); Pauma Valley, 2 (SDSNH); 10 mi NW Ripley, 2 (KU); Vallecito, 38 (MVZ), 4 (SDSNH), 20 (USNM).

Townsend's Big-eared Bat
Plecotus townsendii

1837. Plecotus townsendii Cooper, Ann. Llyc. Nat. Hist. New York, 4:73. Type Locality: Columbia River [Fort Vancouver, Clark Co., Washington].

Distribution: Townsend's Big-eared Bats occur throughout California, although details of their distribution are scanty. There are two subspecies within California: P. t. townsendii occupies the humid, coastal regions of northern and central California and P. t. pallescens occurs in the remainder of the state (Hall, 1981).

Populations Status: Little specific information is available on populations trends, although a marked decline in numbers appears to have occurred over the last 40 years. Pearson et al. (1952) postulated that there was an increase in numbers prior to the 1950's due to increased roosting sites available in human-made structures. In recent years, populations of Townsend's Big-eared Bats seem to have declined in numbers in most areas of the United States (Barbour and Davis, 1969; Humphrey and Kunz, 1976). Partricia Brown (in litt.) noted that populations of P. t. pallescens in the desert area of southeastern California were declining. She stated that they no longer occupied any of the roosts used 30 years previously (about 1950). Graham (1966) found no extant colonies in California's limestone caves, and speculated that all had been abandoned due to human activities. My impression, based on numbers found and turned in to the Biology Department at California State University, Stanislaus, by local residents, is that P. t. pallescens was common in central California into the 1960's, but by the early 1970's was rarely seen. I have captured only one individual during 14 years of netting bats in central California, and none have been turned in to me by others.

Habitat: Townsend's Big-eared Bats live in a variety of communities, including coastal conifer and broad-leaf forests, oak and conifer woodlands, arid grasslands and deserts, and high-elevation forests and meadows. Throughout most of its geographic range, it is most common in mesic sites (Kunz and Martin, 1982). Known roosting sites in California include limestone caves, lava tubes, mine tunnels, buildings, and other human-made structures (Dalquest, 1947; Graham, 1966; Pearson et al., 1952). Habitat for Townsend's Big-eared Bats must include appropriate roosting, maternity, and hibernacula sites free from disturbances by humans. A single visit by humans can cause the bats to abandon a roost. Females typically roost in large maternity colonies which are highly susceptible to disturbances by humans (Barbour and Davis, 1969). Males usually roost singly or in small groups and are probably not affected as much as females by disturbances. Both sexes hibernate in buildings, caves, and mine tunnels, either singly (males) or in small groups (P. Brown, in litt.; Pearson et al., 1952).

Recommendations: Data documenting population status and size and location of maternity colonies and hibernacula are needed most. Great care must be observed in gathering such information; roosts should not be entered. Public land-administering agencies should give special consideration to protecting roosts located on public lands from human disturbances.

Remarks: Early in the investigations, I decided information was insufficient to determine a priority listing for Plecotus townsendii, but that it was probably jeopardized to some degree. Unfortunately, this decision also meant that I quit collecting detailed information on distribution records. Since development of the draft List of Concern, I have decided that this species warranted higher priority than the "sensitive" category to which it was assigned. Thus, no documentation of distribution records are included.

Pocketed Free-tailed Bat
Tadarida femorosacca

1889. Nyctinomus femorosaccus Merriam, N. Amer. Fauna, 2:23. Type Locality: Agua Caliente [Palm Springs], Riverside Co., California.
1924. Tadarida femorosacca, Miller, Bull. U.S. Natl. Mus., 128:86.

Distribution: Pocketed Free-tailed Bats have a spotty distribution, ranging from southern California through the Baja California Penninsula, and southward through southwestern Arizona to at least Michoacon and Tamaulipas, Mexico (Hall and Kelson, 1959). In California, they are known only from a few records in the arid lowland of the southern part of the state. Pocketed Free-tailed Bats are probably resident within the state.

Population Status: There is little current information on the status of T. femorosacca in California. According to P. Brown (in litt.), the roost described by Krutzsch (1944a) is no longer occupied. The paucity of records suggests that T. femorosacca has always been rare, or at least rarely encountered, in California. Because T. femorosacca can easily be confused with the widespread and common T. brasiliensis, it may be more common than records suggest. I listed it here in an attempt to stimulate gathering of information about its status in California.

Habitat: Pocketed Free-tailed Bats are characteristically associated with rocky, desert areas with relatively high cliffs. They generally use crevices in rocks as day-roosts, although they sometimes are found in man-made structures. Krutzsch (1944a) found a colony of 50 to 60 individuals in a crevice in the side of a cliff on a southwestern-facing slope in San Diego Co. T. femorosacca drops from the roost in order to gain speed for flight.

Recommendations: The principal need at this time is for reliable data on the distribution and population status of T. femorosacca. Biologists and public health personnel who routinely work with bats should be alerted to the need for information on T. femorosacca and briefed about methods of distinguishing species of Tadarida.

Remarks: There is no controversy about the status of this species. It is similar only to T. brasiliensis in the United States (see Barbour and Davis, 1969).

Distribution Records: IMPERIAL CO: Mouth of Colorado River (Grinnell, 1933). RIVERSIDE CO.: Agua Caliente [Palm Springs], 1 (USNM); Barker Dam Reservoir, Joshua Tree National Monument (P. Brown, in litt.); Palm Canyon, near Palm Springs, 1 (FMNH), 1 (MVZ). SAN DIEGO CO.: Palm Canyon, Borrego Valley, 1 (MVZ); 2 mi SE Suncrest store, Suncrest (Krutzsch, 1944b).

California Mastiff Bat
Eumops perotis californicus

1890. Molossus californicus Merriam N. Amer. Fauna, 4:31. Type Locality: Alhambra, Los Angeles County, California.
1932. Eumops perotis californicus, Sanborn, J. Mamm., 13:351.

Distribution: Mastiff Bats occur from central California, southward to central Mexico. In California, they have been recorded from Butte County southward in the western lowlands through the southern California coastal basins and the western portions of the southeastern desert region (Sanborn, 1932). Vaughan (1959) observed mastiff bats at Hetch Hetchy Reservoir in the central Sierra Nevada and there are also records of this species from Yosemite Valley. All others are from lower-lying regions. E. perotis is resident within the state, although some bats probably migrate from the colder areas to winter in the southern lowlands (Krutzsch, 1955).

Population Status: Available records indicate that Mastiff Bats were widespread in the San Joaquin Valley, Salinas Valley, and Coastal lowlands from the San Francisco Bay area southward to San Diego. Incidental information suggests that populations of E. perotis have undergone significant declines in recent years (P. Brown, in litt.; P. Leitner, pers. comm.). Historically known roosts in central California are no longer occupied (unpubl. data). Reasons for their decline are only conjecture. Extensive loss of habitat due to urbanization of coastal basins, marsh drainage, and cultivation of major foraging areas are likely factors in the decline. Widespread use of insecticides may have also reduced insect abundance and also poisoned some bats.

Habitat: Mastiff Bats are resident at low elevations in the coastal basins of southern California. They appear to favor rugged, rocky areas where suitable crevices are available for day-roosts. Characteristically, day-roosts are located in large cracks in exfoliating slabs of granite or sandstone. The crevices must open downward, be at least 5 cm wide and 30 cm deep, and narrow to at least 2.5 cm at their upper end (Vaughan, 1959). The crevices typically open high on a cliff and are at least 2 m above the substrate (Krutzsch, 1955; Vaughan, 1959). Mastiff Bats have great difficulty taking flight, and must drop at least 2 to 3 m for launching. Mastiff Bats also frequently roost in buildings, provided these have sheltering spaces with conditions similar to those described above. Howell (1920b) speculated that prior to construction of buildings, Mastiff Bats roosted in cracks in rocks and in hollow trees.

Vaughan (1959) observed Mastiff Bats and estimated that they foraged as much as 2000 ft above the ground. He noted that in some places they regnlarly foraged at 100 to 200 ft over the substrate. They probably forage for considerable distances from their roosting sites. For example, colonies roosting in suitable sites in the Diablo and Temblor ranges, flanking the San Joaquin Valley, likely foraged over the valley floor where insecrs were more abundant.

Mastiff Bats become torpid on a circadian cycle during the winter montha (Leitner, 1966).

Recommendations: Data on the nature end extent of changes in population densities are needed. Historic rooating sites, especially those in areas unlikely to be disturbed by humans, should be surveyed to determine current numbers. Efforts to locate occupied roosts and to estimate numbers should be undertaken. Pesticide loads in captured bats, those found dead, and layered accumulations of guano should be determined.

Colonies which must be controlled because of health hazards or for other reasons should not be poisoned. Efforts to exclude such colonies from structures should be made only at times least likely to result in increased mortality (i.e., not in winter or when young, non-volant bats are present; Constantine, 1979).

Persons in public agencies who handle questions or problems dealing with bats should be informed of the need for information on E. perotis and of its sensitivity to disturbances. Sources of information on methods of non-lethal control of bats should be brought to the attention of such agencies (e.g., Barclay et al., 1980; Constantine, 1979).

Remarks: Eumops perotis is structurally distinct from all other species of Eumops. Eumops perotis californicus is the only subspecies occurring in North America. A second, disjunct population, E. p. perotis, is found in South America, as far south as Argentina (Eger, 1977).

Distribution Records: California (no specific locality), 1 (USNM). ALAMEDA CO.: Hayward, 1 (USNM). BUTTE CO.: Oroville (Eger, 1977). FRESNO CO.: Fresno, 2 (CAS), 1(MVZ); Little Table Mountain, 1 (CSUF); Mendota (von Bloeker, 1943); Trimmer, 1 (MVZ). IMPERIAL CO.: 24 mi S Palo Verde, 1 (MVZ). KERN CO.: near Bakersfield (Krutzsch, 1955); Buena Vista Lake, 5 mi NE Taft, 1 (CSLB); Buttonwillow, 1 (CAS); 1 mi SW Democrat Springs, 1950 ft, Kern River, 19 (MVZ); 10 mi N McKittrick, 1 (CAS); Summer, 1 (MVZ); 10 mi NW Taft, 1 (CSLB), 1 (LSU), 1 (ROM); 10 mi W Taft, 4 (CSLB); Twisselman Ranch, near McKittrick (Krutzsch, 1955). LAKE CO.; Arabella Lake (Storer, 1926); Middleton (Storer, 1926). LOS ANGELES CO.: Alhambra, 1 (AMNH), 1 (LACM), 1 (UI), 2 (USNM); Azusa, 1 (AMNH), 2 (CAS), 17 (LACM), 2 (LSU), 1 (MCZ), 1 (MVZ), 1 (SDSNH), 4 (USNM); Citrus College, Azusa, 1 (CSLB); Claremont (Vaughan, 1959); Covina, 1 (AMNH), 5 (CSLB), 4 (FMNH), 6 (LACM), 5 (MVZ), 4 (UI), 12 (USNM); Cardena (von Bloeker, 1932b); Glendora, 2 (CSLB); Los Angeles, 1 (AMNH), 16 (LACM), 1 (LSU), 1 (MVZ), 16 (USNM); Palms, 2 (SDSNH); Pasadena, 6 (MVZ); Pomona, 1 (KU), 1 (SDSNH), 7 (USNM); Santa Monica, 3 (SDSNH); Santa Monica Mountains, crest at east end (Vaughan, 1959); Sierra Madra, 9 (MVZ). MARIPOSA CO: Mariposa, 1 (ANSP); Yosemite Valley (Vaughan, 1959). MONTEREY CO: Camphora, 2 mi N Soledad, 1 (MVZ). ORANGE CO.: Santa Ana, 1 (ROM). RIVERSIDE CO.: 4 mi SW Lakeview, 2 (KU); Mecca, Colorado Desert, 1 (MVZ); near Perris (Vaughan, 1959); 6 mi W Riverside (Vaughan, 1959). SAN BENITO CO.: Silver Creek, 7 mi ESE Panoche, 900 ft, 18 (MVZ; Dalquest, 1946). SAN BERNARDINO CO.: Colton, 1 (KU), 7 (MVZ), 1 (PM), 1 (TCWC), 2 (UDAV), 1 (UI); Slover Mountain, Colton, 1 (MVZ); 1 mi N, 6 mi W Lucerne Valley (Vaughan, 1959). SAN DIEGO CO.: 1.5 mi N Barrett Junction, 7 (MVZ); Barrett Dam (Krutzsch, 1943); Bow Willow Ranger Station, Anza Borrego Desert, 1 (SDSNH); Dos Cabasas, 1 (USNM); Dulzura, 4 (FMNH), 19 (SDSNH); El Cajon (Vaughan, 1959); Lake Hodges, near Escondido, 1 (MVZ); Otay, 1 (CAS); San Diego, 2 (SDSNH); southeast of Suncrest (Krutzsch, 1945); Yaqui Well, 1 (SDSNH). TULARE CO.: Porterville (Constantine and Humphrey, 1979); Traver, 1 (CAS). TUOLUMNE CO.: Hetch Hetchy Reservoir (Vaughan, 1959).

Salinas Pocket Mouse
Perognathus inornatus psammophilus

1937. Perognathus longimembris psammophilus von Bloeker, Proc. Biol. Soc. Washington, 50:153. Type Locality: west side Arroyo Seco Wash, 150 ft, 4 mi S Soledad, Monterey Co., California.

Distribution: As defined herein (see Remarks), the known distribution of P. i. psammophilus extends from near Soledad southward to Hog Canyon in the Salinas Valley, Monterey Co. The relationships of populations on the Carrizo Plains (von Bloeker, 1937), Cuyama Valley, and upper Salinas River watershed are uncertain (see Remarks).

Populations Status: Most natural communities in the Salinas Valley have been cultivated or otherwise developed. Extant populations of Salinas Pocket Mice have not been located in that area in recent years. Areas where they may still be found are privately owned and access to search for Salinas Pocket Mice is difficult to acquire.

Habitat: According to von Bloeker (1937), Salinas Pocket Mice occur on fine-textured, sandy soils. They may also occur on a variety of other substrates in annual grassland and desert shrub communities, especially where plant cover is not dense and soils are friable.

Recommendations: The most pressing need is for information on the location and status of extant populations. Efforts to locate populations and to preserve suitable habitat in the Salinas Valley should be given relatively high priority. The systematic relationships of P. i. psammophilus, P. i. inornatus, and P. i. neglectus should be resolved prior to any proposals for protecting the Salinas Pocket Mouse.

Remarks: All of the specimens available to von Bloeker (1937) when he described P. longimembris psammophilus were subadults. He also described P. inornatus sillimani from the same locality (Arroyo Seco) at the same time (1937). All of the specimens of P. i. sillimani that von Bloeker (1937) reported on and used in his diagnosis were adults. Although von Bloeker (1937) compared both subspecies with other subspecies of their respective species, he did not compare the two new subspecies with each other. I examined all of the extant specimens of both taxa and found no evidence suggesting that there was more than one species represented. The two subspecies are based upon different age groups drawn from the same population. Neither group of specimens closely resembles P. longimembris, which, in my opinion, does not occur anywhere in the central California lowlands, despite indications to the contrary in Hall (1981) and other publications (see below). I consider psammophilus to be a subspecies of P. inornatus, and P. i. sillimani to be a junior synonym of P. i. psammophilus. The latter decision is based upon the superior page position of the description of psammophilus (von Bloeker, 1937).

Available data show that populations of P. inornatus living on the Carrizo Plain, San Luis Obispo Co., are closely similar in structure and karyotype to those of P. i. neglectus from along the western margin of the San Joaquin Valley (Williams, 1978, and unpubl. data). On the bases of different karyotypes (50 vs 56 chromosomes) and body proportions, P. i. neglectus and P. i. inornatus are probably different species. P.

longimembris does not occur in central California. Mice from within the San Joaquin, Sacramento, and Salinas valleys, identified by others as P. longimembris, are all referable to one or the other group of P. inornatus (unpubl. data).

Specimens listed here are from within the geographic range of P. i. sillimani as defined by von Bloeker (1937), but which I consider to be indistinguishable from P. i. neglectus. Their relationship with P. i. psammophilus is unresolved; it is possible that psammophilus will be found to be a synonym of neglectus. SAN LUIS OBISPO CO.: 5.5 mi SW Caneros Spring, 1 (CAS); Carrizo Plains, W end Salt Lake, 1 (USNM); Carrizo Plains, 8 mi E [SE] Simmler, 3 (USNM); 8.75 mi SE Cholame, 2200 ft, 1 (CAS); Cuyama Valley, 3 mi E Cuyama Ranch, 3 (USNM); Elk Horn Plain, 1 (CM); Indian Creek, 1500 ft, 13 mi S Shandon, 1 (MVZ); Salinas Valley, 2 mi S San Miguel, 7 (MVZ); San Diego Joe's, 2700 ft, 6 (MVZ); San Diego Joe's, 1 mi S, 7 mi E Simmler, 2700 ft, 7 (MVZ); Santiago Springs, 1 (MVZ), 4 (USNM); Santiago Springs, 1.5 mi S, 8 mi E Simmler, 10 (MVZ); 4 mi S, 5 mi E Shandon, 1175 ft, 4 (MVZ); 7 mi SE Simmler, 5 (MVZ); Soda Lake, 1 (CSLB). VENTURA CO.: Quatal Canyon, 6 mi E Venucopa, 1 ((MVZ).

Localities (above) listed as San Diego Joe's and Santiago Springs refer to the same place. This has led to considerable confusion because Santiago Creek is in Kern County on the San Joaquin Valley side of the Temblor Range and San Diego Joe's was located along San Diego Creek on the western slope of the Temblor Range. Also, the locality referred to as San Diego Joe's or Santiago Springs is around 2500 ft.

Distribution Records: MONTEREY CO.: Arroyo Seco, W side, 4 mi S Soledad, 150 ft, 8 (LACM), 30 (MVZ); Arroyo Seco River, 10 mi S Paraiso Springs, 1 (USNM); Hog Canyon, 1 (LACM); Metz, Salinas Valley, 200 ft, 1 (MVZ); mouth Wild Horse Canyon, 500 ft (von Bloeker, 1937).

White-eared Pocket Mouse
Perognathus alticola alticola

1894. Perognathus alticolus Rhoads, Proc. Acad. Nat. Sci. Philadelphia, 45:412. Type Locality: Squirrel Inn, 5500 ft near Little Bear Valley, San Bernardino Mountains, San Bernardino Co., California.
1900. Perognathus alticola, Osgood, N. Amer. Fauna, 18:39.

Distribution: Perognathus alticola alticola is known only from the western portion of the San Bernardino Mountains in the vicinity of Strawberry Peak, at altitudes extending from about 5400 ft to 5800 ft. Grinnell (1908) reported that an immature Perognathus, partly eaten, was found in a trap set in Sagebrush at the north base of Sugarloaf Peak, at 7500 ft. This was likely a White-eared Pocket Mouse.

Population Status: Stephens (1906) remarked that White-eared Pocket Mice were uncommon, and that considerable trapping had yielded only six specimens. A few additional specimens were collected in the years prior to the 1940's. I am not aware of any specimens collected later than 1934. Jim Sulentich (1983) searched extensively for White-eared Pocket Mice, starting in June of 1979 and using pitfalls and livetraps. To date, no White-eared Pocket Mice have been taken.

Habitat: Only scant information has been recorded. Grinnell (1933) stated that members of this population inhabited the Transition life-zone, living on the "dry floor of open Pine forest where Bracken Fern grows." White-eared Pocket Mice are likely to be found among Sagebrush and other shrubs in open, Ponderosa Pine forests and Pinyon-Juniper woodlands and in Sagebrush covered areas on the northern slopes and Big Bear Basin of the San Bernardino Mountains.

Recommendations: The U.S. Forest Service should be informed of the need for information on White-eared Pocket Mice, as it is the principal land-administering agency within the area. Efforts to locate extant populations should be intensified, including searches in the San Gabriel Mountains. If and when such populations are located, action designed to ensure their security should be taken.

Remarks: Jim Sulentich and David Huckaby at California State University, Long Beach are principal authorities on the current population status of this taxon. White-eared Pocket Mice will probably be found to be most common on the northern slopes of the San Bernardino and San Gabriel Mountains.

Rhoad's (1894) specific epithet, alticolus, apparently referred to the mountain home of this species (altus, high; cola, inhabiting). However, the root, col, is a prefix to be used before r and meaning with or together. Osgood (1900) amended the spelling without comment. As Rhoads (1894) made no reference to what the name was supposed to mean or how it was derived, one can only guess about his intent. I suspect, therefore, that Osgood's amendment was inappropriate, but as this spelling (alticola) is well-entrenched in the literature, its continued use will best preserve nomenclatural stability. I raise this issue here because two prominent works on California's mammals (Grinnell, 1933; Ingles, 1965) used Rhoads' spelling. Rhoads' paper was dated 1893, but was published in 1894.
The White-eared Pocket Mouse is an apparently isolated allospecies of the *Perognathus parvus* group (Williams, 1978).

Distribution Records: SAN BERNARDINO CO.: San Bernardino Mountains, 3 (USNM), 1 (ANSP), 1 (SDSNH); Squirrel Inn, San Bernardino Mountains, 5500 ft, 2 (LACM), 3 (MVZ), 6 (SDSNH); 0.5 mi E Strawberry Peak, San Bernardino Mountains, 2 (SDSNH); 1 mi E Strawberry Peak, San Bernardino Mountains, 5750 ft, 5 (MVZ), 3 (SDSNH); 2 mi E Strawberry Peak, 5750, ft, 13 (MVZ); Strawberry Peak, 5000 to 5700 ft, 21 (LACM).

Southern Marsh Harvest Mouse
Reithrodontomys megalotis limicola

1932. *Reithrodontomys megalotis limicola* von Bloeker, Proc. Biol. Soc. Washington, 45:133.

Distribution: Southern Marsh Harvest Mice are known to occur only in coastal salt marshes in Ventura, Los Angeles, Orange, and Santa Barbara counties, California.

Population Status: No current data are available. The urbanization of the areas within this mouse's range has been extensive. Stream channelization and development of coastal areas have resulted in habitat loss. These developments and urbanization could have fragmented remaining habitat and isolated populations too small to endure. Some of the marshes are currently protected as ecological reserves, however.

Habitat: Southern Marsh Harvest Mice are strictly confined to marshy areas, generally coastal salt marshes dominated by *Salicornia*. Adjacent weedy areas and marshes in brackish sites may also be inhabited (von Bloeker, 1932a).

Recommendations: The immediate need is for data on distribution and population status. A live-trapping program should be undertaken.

Remarks: Specimens identified as *R. m. limicola* from Dominguez Hills and Palo Verde Hills, Los Angeles County (CSLB), are probably referable to *R. megalotis longicaudus*. According to von Bloeker's (1932a) description, *R. m. limicola* is easily distinguished from adjacent subspecies of Harvest Mice.

Distribution Records: LOS ANGELES CO.: Del Rey Marsh, 3 (LACM), 1 (USNM); 0.5 mi NW El Segundo, 1 (MVZ); Hyperion, 2 (LACM); Marina Del Rey, salt marsh N of Culvor Blvd., 1 (CSLB); Nigger Slough, Gardena, 5 (LACM); Playa del Rey, salt marsh, 46 (LACM); 1 mi N Playa del Rey, salt marsh, 1 (LACM); Torrence, Avalon Blvd. and Dominquez Channel, 1 (CSLB). ORANGE CO.: Anaheim Bay, 8 (LACM); Los Altos, E of Ammo. Dump, 1 (CSLB); Newport Beach, 2 mi NE Highway 101, 2 (CSLB); San Gabriel River Blvd., 1 (CSLB). SANTA BARBARA CO.: Carpinteria, salt marsh, 3 (LACM). VENTURA CO.: Point Mugu, 16 (LACM).

Riparian Woodrat
Neotoma fuscipes riparia

1938. *Neotoma fuscipes riparia* Hooper, Univ. California Publ. Zool., 42:223. Type Locality: Kincaid's Ranch, 2 mi NE Vernalis, Stanislaus Co., California.

Distribution: Riparian Woodrats are known only from an area along the San Joaquin, Stanislaus, and Tuolumne rivers in Stanislaus and San Joaquin counties (Hooper, 1938; unpubl. data). Hall and Kelson (1959) assigned a specimen from El Nido, Merced Co., to this subspecies on geographical grounds (see Remarks).

Population Status: Trapping programs along the Tuolumne (C. Johnson, unpubl. ms.) and Stanislaus (G. Basey, unpubl. ms.) rivers have resulted in a few captures only in Caswell State Park. The current status of this population is unknown. Regulation of stream flow, stream channelization, cultivation of the floodplains, and brush and tree removal have diminished available riparian habitat. Removal of undergrowth and large trees with dead and hollow limbs in parks has reduced habitat further. If current trends continue, additional habitat loss can be expected. During floods, there are few or no refuges for Woodrats, as nearly all land bordering the rivers is cultivated. A population is still extant in the general area of the type locality (unpubl. data), but its demographic features are unknown.

Habitat: Dusky-footed Woodrats are found only in areas supporting brush. Generally, they occur in areas with mixtures of trees and brush. They require suitable nesting sites such as cavities in trees, snags, or logs; spaces in talus; or lodges built of downed woody material. These houses or lodges are usually a conspicuous feature of areas inhabited by N. fuscipes.

Nothing specific has been recorded about the habitat of Riparian Woodrats. Basey (unpubl. ms.) captured a few Riparian Woodrats in traps set in runways of Riparian Brush Rabbits. Students at California State Univ., Stanislaus, have noted that these Woodrats occasionally use nest boxes placed in trees for Wood Ducks along the lower San Joaquin and Tuolumne rivers. Apparently, Woodrats also eat the eggs of the Wood Ducks. The San Joaquin Valley is generally devoid of suitable habitat for Woodrats, except along rivers where trees and brush were found. Thus, this population is confined to riparian habitat.

Recommendations: A survey of riparian areas along all of the principal tributaries of the San Joaquin River should be undertaken to determine distribution and abundance of N. f. riparia. The status and relationships of the Woodrats in Corral Hollow should be clarified. Appropriate wildlife organizations and county fish and wildlife committees involved in placing and monitoring Wood Duck nesting boxes should be alerted to the need for information about Riparian Woodrats.

Remarks: There is no habitat for N. fuscipes in the area around El Nido today, because of agricultural developments. The specimen probably came from somewhere along the San Joaquin River west or south of El Nido. A specimen from 10 mi SW Los Banos (Merced Co.; CSUF) is either N. f. perplexa or N. lepida (I do not recall looking at the specimen when the data were obtained from the museum records).

Distribution Records: MERCED CO.: El Nido (Hall and Kelson, 1959). SAN JOAQUIN CO.: Corral Hollow, 1 (MVZ); Corral Hollow Creek, 4 mi E Alameda Co. line, 1 (MVZ); San Joaquin River, 3 mi NE Vernalis, 3 (MVZ). STANISLAUS CO.: Kincaid's Ranch, 2 mi NE Vernalis, 2 (MVZ).

White-footed Vole
Arborimus albipes

1901. Phenacomys albipes Merriam, Proc. Biol. Soc. Washington, 14:125. Type Locality: Redwoods, near Arcata, Humboldt Bay, Humboldt Co., California.

1979. Arborimus albipes, Williams, Ann. Carnegie Mus., 48:426.

Distribution: White-footed Voles occur in the humid coastal forest region from the Columbia River, Oregon, southward to Humboldt County, California. They are known from near sea level to as high as 3500 ft (Maser and Johnson, 1967). The localities in California are all from low-lying areas.

Population Status: No data are available to allow determination of the population status of A. albipes. White-footed Voles have been captured infrequently, and must be considered rare. Because nearly all known localities of capture have been associated with small streams in forested areas, White-Footed Voles may be sensitive to logging and other alterations of riparian habitats.

Habitat: Maser and Johnson (1967) reviewed habitat notes associated with each specimen known to them. White-footed Voles have been taken in a variety of situations, but generally seemed to be associated with small streams in forested areas. Available evidence indicates that these voles are terrestrial, unlike A. longicaudus. Their structural features, however, suggest that they are more scansorial than typical voles (Johnson and Maser, 1982). They probably climb shrubs and downed woody material in the understory, and perhaps even trees.

All records of capture are from areas in the humid coastal forest region. Captures were mostly in forested sites, but one was taken in a logged area (two years after logging), and one was captured in a grassy spot on a partly logged ridge top (Maser and Johnson, 1967). Very small clearings, created by fallen timber and supporting herbaceous growth, may be important habitat for this species. Thickets of Alders and other riparian communities along small streams (Maser et al., 1980) may also be essential habitat.

Howell (1928) found only roots of herbaceous plants in three stomachs of White-footed Voles, whereas Maser and Johnson (1967) found green plant material in several stomachs.

Recommendations: Arborimus albipes should be given special consideration in forest management plans within its range. There is little public land within its known range in California. White-footed Voles occur in the Redwood National Park's northern section in Del Norte and Humboldt counties; special effort should be made to determine their distribution and habitat needs there.

White-footed Voles will probably be found farther inland at higher elevations on State and U.S. National Forest lands. Intensive searches should be conducted to determine the limits of their distribution in California, habitat needs, and population status. Pitfall trapping may prove more successful for capture of White-footed Voles than conventional snap-traps.

Remarks: Taylor (1915) described Arborimus as a subgenus of Phenacomys. Johnson (1973) presented and discussed evidence which indicated that Tree Voles (Arborimus) were divergent from Heather Voles (Phenacomys); he elevated Arborimus to generic rank. Unfortunately, Johnson (1973) did not state whether albipes was an Arborimus or a Phenacomys. Based upon the obviously close similarity of albipes to A. longicaudus (Howell, 1920c, 1926; Taylor, 1915), Williams (1979) treated albipes as a member of the genus Arborimus. Johnson and Maser (1982) discussed characteristics diagnostic for Arborimus and Phenacomys and formally transfered albipes to the genus Arborimus.

Distribution Records: DEL NORTE CO.: Wilson Creek, N of Klamath [city], 1 (CM). HUMBOLDT CO.: Arcata, Humboldt Bay, 1 (USNM); 5.5 mi E Bayside, 1 (MVZ); 3 mi N Orick, 1 (UCLA); Trinidad, 1 (HSU), 1 (MVZ), 2 (SDSNH); N of Trinidad, 1 (MVZ), 3 (USNM).

Point Reyes Jumping Mouse
Zapus trinotatus orarius

1899. Zapus orarius Preble, N. Amer. Fauna, 15:29. Type Locality: Point Reyes, Marin County, California.
1944. Zapus trinotatus orarius, Hooper, Misc. Publ. Mus. Zool., Univ. Michigan, 59:67.

Distribution: Zapus trinotatus orarius is apparently confined to a small area on the Point Reyes Peninsula, Marin County, California (Krutzsch, 1954).

Population Status: No data are available on the current status of Point Reyes Jumping Mice. The habitat for this population is not extensive. Post-Pleistocene changes in climate may continue to modify habitat unfavorably for this species in the Point Reyes area. Grazing by Cattle may have been a factor contributing to adverse conditions for Jumping Mice at Point Reyes, but this problem has been resolved. The impact of introduced Axis and Fallow deer is unknown. The principal causes for concern are the small range and relictual nature of the population, and the fact that these mice are seemingly scarce.

Habitat: Howell (1920d) listed the habitat of this population as bunch grass marshes on the uplands of Point Reyes. Krutzsch (1954) stated that Point Reyes Jumping Mice occur in moist areas that are safe from continuous inundation. Farther north, Pacific Jumping Mice are characteristic of the Pacific coastal coniferous forests. They seem to prefer riparian Alder communities and treeless openings with tall, dense herbaceous growth of grasses and forbs (Maser et al., 1981). Wet, marshy sites are often said to be preferred, although Pacific Jumping Mice also occur occasionally in closed forests with little understory, in dry meadows, and more often in thickets of deciduous, woody vegetation along streams and seepage areas.

Zapus trinotatus eats seeds of herbaceous plants, especially grasses and grass-like monocots. Stems of plants are frequently cut to obtain ripening seed-heads. Fruits and insects are also eaten (Krutzsch, 1954; Jones, et al., 1978).

Recommendations: A live-trapping program employing pitfalls is the best method of sampling for Jumping Mice. Conventional livetraps rarely capture Jumping Mice in comparison to pitfalls (Williams and Braun, 1983). The Point Reyes area should be thoroughly inventoried for Jumping Mice, and their habitat associations, specific requirements, and population status established.

Remarks: Point Reyes Jumping Mice are the southernmost population of Pacific Jumping Mice. The structural characters of this population are distinctive, and no evidence exists of intergradation with Jumping Mouse populations farther north along the California coast (Howell, 1920d). Hooper (1944) arranged orarius as a subspecies of trinotatus, suggesting that intergrades could be expected from geographically intermediate areas. Krutzsch (1954) concurred with Hooper's (1944) arrangement, but noted that "Interbreeding in the wild between Z. t. orarius and Z. t. eureka probably does not take place, because these subspecies are separated by terrain not suited to Jumping Mice."

Distribution Records: MARIN CO.: Elk, Tennessee Valley (barn on Lewis Dairy — from owl pellet), 1 (USNM); W end Elk Valley, 10 ft, 1 (MVZ); Fort Berry, 1 (CAS); Fort Cronkhite, 1 (CAS); 3 mi W Inverness, 300 ft, 16 (MVZ); Point Reyes, 1 (USNM); 5 mi NNE point Reyes Lighthouse, 12 (MVZ); 6 mi NE Point Reyes, 1 (MVZ); West Portal, Fort Barry, 2 (MVZ); 6 mi SSE Tomales Bay, 1 (CAS).

THIRD PRIORITY LIST

Big Free-tailed Bat
Tadarida macrotis

1839. Nyctinomus macrotis Gray, Ann. Mag. Nat. Hist., 4:5. Type Locality: Cuba.
1962. Tadarida macrotis, Miller, Bull. U.S. Natl. Mus., 28:86.

Distribution: Big Free-tailed Bats are found from northern South America and the Caribbean Islands northward to the western United States. Populations in the southwestern U.S. appear to be widely scattered. Known breeding localities in the U.S. are found in parts of Arizona, New Mexico, and Texas. In California, T. macrotis is known only from a few low-lying arid areas of southern California and a single specimen from Berkeley, Alameda Co. (see Remarks). According to Barbour and Davis (1969), records of accidental occurrence (stray, migratory individuals) are known from widely scattered localities in western North America, including British Columbia and Iowa.

Population Status: Very little is known of Tadarida macrotis in California. Possibly it does not occur regularly within the state. The nature of the records, however, suggests that Big Free-tailed Bats breed somewhere within the state — most likely in the rugged, wooded, mountainous areas of southern California. At least one of the specimens from California was a young bat (Hall, 1946), collected on 11 August. Huey (1932, 1954a) reported on two Big Free-tailed Bats from San Diego, one collected on 1 February; the second, when found, was mummified. The specimen from Berkeley was taken on 18 December (unpubl. data).

Tadarida macrotis may not be under any threats within California. Its rarity, however, gives cause for concern about its present status. Listing it here may stimulate efforts by others to clarify its status.

Habitat: Big Free-tailed Bats inhabit relatively rocky areas in the arid southwestern U.S. They apparently roost primarily in crevices in cliffs (Barbour and Davis, 1969). Findley et al. (1975) described maternity colonies situated in a crevice in the roof of a large sandstone rock shelter and under slabs of lava on a perpendicular cliff. They found Big Free-tailed Bats up to 8000 ft, but they were most common in communities below 6000 ft in New Mexico. In New Mexico, at least, females apparently inhabit forested areas during the reproductive season in the summer. Big Free-tailed Bats appear to be migratory in most of the U.S. Winter records, however, include specimens from San Diego, California, and Yuma, Arizona (Barbour and Davis, 1969).

Recommendations: An attempt to gather current information on the distribution and populations status of the Big Free-tailed Bat should be undertaken. Biologists and public health personnel who routinely receive information from the public about bats or who conduct research on bats should be alerted to the need for information on this species. They should be provided with a description to aid identification of this species. Students and faculty members at colleges and universities in southern California should be encouraged to undertake studies of the occurrence and habitat associations of bats in various parts of southern California.

Remarks: *Tadarida macrotis* has been listed as *T. molossa* in much of the literature. A note in the card catalog of the Museum of Vertebrate Zoology, Univ. California, Berkeley, questioned the specimen from Berkeley on the basis that the tag was attached to the jar rather than the specimen. Everything else appeared to be in order and the jar seemed not to have been opened or otherwise disturbed, according to the notation. All things considered, the record should be accepted as accurate.

Distribution Records: California (no precise locality, but presumably along the Colorado River near Nevada), 1 (USNM). ALAMEDA CO.: Berkeley, Univ. California Campus, 1 (MVZ). SAN DIEGO CO.: San Diego, 2 (SDSNH); Mission Beach, San Diego (August and Dingman, 1973).

Pygmy Rabbit
Brachylagus idahoensis

1891. *Lepus idahoensis* Merriam, N. Amer. Fauna, 5:76. Type Locality: Pahsimeroi Valley, near Goldburg, Custer Co., Idaho.
1904. *Brachylagus idahoensis*, Lyon, Smithsonian Misc. Coll., 45:411.

Distribution: Pygmy rabbits are found only in the Great Basin and contiguous areas in Sagebrush-dominated communities. They occur from southeastern Washington southward to about Inyo Co., California, and eastward to western Utah and southeastern Idaho. In California, they occur in eastern Modoc, Lassen, and Mono counties and probably in northern Inyo County. Their occurrence is very spotty throughout their geographic range (Green and Flinders, 1980).

Population Status: Pygmy Rabbits have a limited and spotty distribution, being strictly confined to suitable stands of Sagebrush (primarily *Artemisia tridentata*) and Rabbitbrush (*Chrysothamnus* spp.). Although Pygmy Rabbits may seem common where found, they are not numerous over their geographic range.

The current population status of Pygmy Rabbits in California is unknown, but their numbers have probably declined in the past several years. Loss of habitat to cultivation is less of a factor than loss of habitat by overgrazing. Even though overgrazing favors growth of woody shrubs such as Sagebrush over perennial grasses, cattle often congregate in tall stands of Sagebrush, seeking shade in summer, protection from wind, and relief from insects. Frequently, cattle trample and otherwise open up the understory from ground level up to 1 to 1.5 m, reducing food and shelter for Pygmy Rabbits. Brush clearing on rangelands and range fires also reduce available habitat for Pygmy Rabbits. In Oregon, Weiss and Verts (1984) found evidence of a marked decrease in occupancy of sites between 1981 and 1983. They suggested that Pygmy Rabbit populations are susceptible to rapid declines and local extirpations. The fragmentation of Sagebrush communities poses a potential threat to extant populations because of this susceptibility.

Habitat: Pygmy Rabbits are associated only with dense stands of tall Sagebrush and rabbitbrush. Soils must be friable for burrowing. Weiss and Verts (1984) found that Pygmy Rabbits were associated with areas with greater shrub cover, shrub height, soil strength, and soil depth in comparison to adjacent, unoccupied sites. There was no significant association with percent basal area of perennial grasses, density of annual grasses, density of forbs, or cryptogam cover. Burrows of Pygmy Rabbits are usually placed on slopes (Green and Flinders, 1980). Food consists principally of Big Sagebrush (A. tridentata) in winter (up to 99% of the diet), whereas grasses form a significant portion of the diet in summer. White et al. (1982) found differential use of populations of Big Sagebrush, but were unable to associate this with known variations in the chemical composition of A. tridentata populations. Captives, however, showed no significant preference as food for one type of Sagebrush subspecies over others. Weiss and Verts (1984) determined that occupancy of sites by Pygmy Rabbits in Oregon was not dependent upon the presence of particular subspecies of Sagebrush.

Recommendations: Detailed information on distribution, habitat associations, abundance, and numbers of Pygmy Rabbits killed by hunters in California are needed. Management of habitat for Pygmy Rabbits is probably the most important element in ensuring their survival. Land and resource management agencies in California should be encouraged to develop needed information on Pygmy Rabbit populations and to consider habitat for Pygmy Rabbits in land and resource management

Remarks: The Pygmy Rabbit is designated as a Resident Small Game species in California. There is an open season for hunting from 1 July through January, with a daily bag limit of five. Although the effect of hunting on Pygmy Rabbit populations is not known, hunters probably do not kill many because Pygmy Rabbits are quite secretive and rarely venture from dense brush; they usually retreat to their burrows when threatened.

Brachylagus is structurally and behaviorally distinct from all species of Sylvilagus and apparently of an ancient lineage separate from that of Sylvilagus. It was first treated as a species of Sylvilagus by Grinnell et al. (1930) for reasons that have little or no bearing on its phyletic relationships. Since then, most authors have treated it as a species of Sylvilagus (e.g., Hall, 1981). Green and Flinders (1980) reviewed the taxonomic history of B. idahoensis.

Distribution Records: LASSEN CO.: 7 mi E Ravendale, 5000 ft, 19 (MVZ); 3 mi S Ravendale, 5300 ft, 1 (MVZ); vicinity of Red Rock Post Office, 5300 ft (Grinnell et al., 1930. MODOC CO.: Goose Lake, 2 (USNM). MONO CO.: Bodie, 8374 ft, 2 (CAS), 8 (MVZ); north of Crowley Lake (Jones, 1957); Mono Dunes (Harris, 1982); Mono Flats, 7000 ft, 3 (CAS); Mono Lake, near Mono Mills, 4 (CSUF); Mono Lake, Black Point (Harris, 1982); Mono Lake County Park (Harris, 1982).

Oregon Snowshoe Hare
Lepus americanus klamathensis

1899. Lepus klamathensis Merriam, N. Amer. Fauna, 16:100. Type Locality: head of Wood River, near Fort Klamath, Klamath County,

Oregon.
1936. Lepus americanus klamathensis, Bailey, N. Amer. Fauna, 55:95.

Distribution: Lepus americanus klamathensis is found in the Cascade Mountains from Mt. Hood, Oregon, southward to Mt. Shasta and the Trinity Mountains of California (Bailey, 1936; Orr, 1940). In California, it is known from the vicinity of Mt. Shasta, the Trinity Mountains, and from the Warner Mountains. The Warner Mountain population is probably isolated from all others by expanses of unsuitable habitat.

Population Status: Apparently, Oregon Snowshoe Hares never have been common in California within historic times, judging from early accounts (e.g., Merriam, 1899) and from the small number of specimens in museums (Orr, 1940). Mossman (1979) considered determination of their present abundance within California to be important. Mailliard (1927) failed to find L. americanus in the Warner Mountains; the species is apparently known from a single specimen from that mountain range. I saw no evidence of L. americanus in the Warner Mountains during 3 days of field work in 1983.

Populations of Snowshoe Hares undergo periodic fluctuations in numbers, so field researchers may have simply searched when populations were low. The evidence, however, favors an interpretation of rarity.

The Snowshoe Hare is a game species in California (Resident Small Game), and may be hunted from 1 July through January, with a bag limit of five per day. Many hunters probably fail, however, to distinguish Snowshoe Hares from other hares (Black-Tailed and White-Tailed) which have no season or bag limits in California. The effect of hunting on populations of L. americanus in California is unknown.

No definite decline in numbers has been established. The apparent rarity of these hares and the potential for habitat loss due to logging and usurpation of riparian habitat by humans require that the population status of L. a. klamathensis be determined.

Habitat: Snowshoe Hares are primarily found in riparian areas with thickets of deciduous trees such as Alders and Willows, and in dense thickets of young conifers, particularly firs. Dense patches of Ceanothus and Arctostaphylos are sometimes inhabited. In California, Snowshoe Hares are found mainly above the Yellow Pine zone, in Canadian and Hudsonian associations (Orr, 1940). Merriam (1899) reported that Snowshoe Hares formerly were common in marshy places along streams near Fort Klamath, Oregon. He did not obtain specimens, however, during field work at Mt. Shasta. Mossman (1979) indicated that Snowshoe Hares only inhabit areas in northern California which have snow in the winter.

Recommendations: The highest priority should be given to determining the status of the Warner Mountain population, an apparently isolated and unique population. Detailed assessments of population numbers in all areas of the range of L. a. klamathensis should be made. Habitat needs of L. americanus klamathensis should be considered in managing State and U.S. National Forest lands within its range.

Remarks: Orr (1940) regarded the Washington Snowshoe Hare as specifically distinct from L. americanus, and treated klamathensis as a subspecies of L. washingtoni (but see Dalquest, 1942).

Distribution Records: MODOC CO.: vicinity of Fort Bidwell, 6000 ft, 1 (MVZ). SHASTA CO.: McCloud River, 12 mi E McCloud Post Office, 1 (MVZ); south fork Salmon River (Grinnell, 1933). SISKIYOU CO: T. H. Benson Estate, Butte Creek, 5000 ft, 1 (MVZ); Gottville, 1 (USNM); McCloud, 1 (MVZ); N of Mt. Shasta (city), 1 (UDAV). TRINITY CO.: head of Bear Creek, 2 (MVZ); Rush Creek, 12 mi from Weaverville, about 3000 ft, 1 (MVZ).

Sierra Nevada Snowshoe Hare
Lepus americanus tahoensis

1933. Lepus washingtoni tahoensis Orr, J. Mamm., 14:45. Type Locality: 0.5 mi S Tahoe Tavern, Placer Co., California.
1942. [Lepus americanus] tahoensis, Dalquest, J. Mamm., 23:176.

Distribution: Sierra Nevada Snowshoe Hares range along the mid-elevations of the Sierra Nevada from the vicinity of Mt. Lassen southward to Mono and Tulare counties (unpubl.). They are known from Nevada only in the vicinity of Lake Tahoe (Hall and Kelson, 1959). They occupy altitudes above 4800 ft in the north of their range and 5000 ft in the south. Upper elevational limits are unknown; these hares generally occur below 8000 ft.

Population Status: Judging from circumstantial evidence, Sierra Nevada Snowshoe Hares are more abundant than Oregon Snowshoe Hares, but still are relatively uncommon. There is no evidence of a population decline, but L. americanus is vulnerable to widespread habitat alterations due to some logging activities and use of meadows for agriculture, grazing, and other activities. Furthermore, this is a Resident Small Game species which is hunted from 1 July through January. Many hunters likely fail to distinguish L. americanus from white-tailed hares (Lepus townsendii), which have no season or bag limits in California. The effect of hunting on L. a. tahoensis populations is unknown, but is unlikely to be much of a factor.

Habitat: The habitat of Sierra Nevada Snowshoe Hares is similar to that of L. a. klamathensis. They live only in boreal zones, typically inhabiting riparian communities with thickets of deciduous trees and shrubs such as Willows and Alders. They also frequent dense thickets of young conifers and chaparral composed of Ceanothus and Arctostaphylos (Orr, 1940). In the Lake Tahoe region in summer, Orr, (1940) noted signs of Snowshoe Hares only near brush adjacent to both meadows and riparian deciduous vegetation. He found no evidence of Snowshoe Hares on ridgetops or brush-covered upper slopes. In the Mt. Lassen region, Grinnell et al. (1930) found that Snowshoe Hares were uncommon, being infrequently encountered among thickets of Snow-brush and young Firs, and in or near thickets of Alders and Willows situated in meadows. In winter, Grinnell et al. (1930) found tracks of Snowshoe Hares to be more numerous in and near thickets of Snow-brush, Willows and small conifers than on more open ridges.

Recommendations: The principal need is for reliable data on distribution and abundance of L. a. tahoensis. Special consideration of habitat needs of L. americanus in managing public lands in California is needed.

Remarks: Lepus americanus tahoensis differs from L. a. klamathensis in its darker coloration, other, minor details of coloration, and in skull proportions. There is no controversy about its status.

Distribution Records: ALPINE CO.: Pacific Valley, half way between Woodfords and Big Trees, 1 (MVZ). EL DORADO CO: no precise locality, 1 (MCV); Echo Lake, 1 (MVZ). LASSEN CO.: Warner Creek (Grinnell, 1933). MONO CO.: 4 mi N, 1 mi E Mammoth, 1 (HSU). NEVADA CO.: Sagehen Creek, 4 mi NW Hobart Mine, 1 (MVZ); Spruce, 1 (MVZ); 10 mi W Truckee, 1 (MVZ); Truckee, 1 (MVZ). PLACER CO.: Cisco, 1 (MVZ); Donner, 1 (USNM); near S shore Donner Lake, 6000 ft, 1 (MVZ); Donner Summit, 7000 ft, 1 (MVZ); Lake Tahoe, Rubicon subdivision between Emerald Bay and Meeks Bay, 3 (CAS); near Tahoe City, 19 (MVZ); 0.5 mi S Tahoe Tavern, 1 (MVZ). PLUMAS CO.: Willow Lake, 5600 ft, 1 (MVZ). SIERRA CO.: east side Yuba Pass, 6000 ft, 1 (MVZ). TEHEMA CO.: 7 mi E junction highways 32 and 36, 1 (CSUC); Mineral, 4800 ft, 1 (MVZ); Summit Creek, 5200 ft, 2 mi E Mineral, 2 (MVZ). TUOLUMNE CO.: Niagra Creek, 5600 ft (Orr, 1940); between Pine Crest and Belle Meadows, 1 (MVZ); 1 mi NW Strawberry, 6000 ft, 1 (CAS).

Western White-tailed Hare
Lepus townsendii townsendii

1839. Lepus townsendii Bachman, J. Acad. Nat. Sci. Philadelphia, 8(1):90.
Type Locality: Fort Walla Walla, Walla Walla Co., Washington.

Distribution: Western White-tailed Hares are found from the Okanogan Valley of southern British Columbia southward east of the Cascade Crest to the southern Sierra Nevada of California, and eastward to extreme western Montana and western Colorado (Hall, 1981). In California, White-tailed Hares principally occupy open forests and sagebrush-grassland associations in the Great Basin Province. They occur also at high elevations along the main crest of the Sierra Nevada and rarely to as low as 6000 ft on the western slope. I observed white-tailed hares near Markwood Meadow at about 6000 ft, and on Tamarack Ridge at about 7500 ft in Fresno Co., approximately 30 mi west of the Sierra Nevada crest. The southernmost published locality known is Mt. Silliman, Tulare County in Sequoia National Park (Fry, 1924).

Population Status: Little current information is available for California. Airola (1980) stated that this species had not been seen in Lassen County in the last 20 years. Evidence suggests that populations of L. townsendii have declined drastically, especially at lower elevations in the sagebrush-grassland associations of the Great Basin province (Grayson, 1977). Competition with domestic livestock is probably a principal factor in the drastic population declines throughout its range, including California (Dalquest, 1948; Moseman, 1979). Loss of habitat to cultivation and other developments are probably also important factors in the decline. Hunting could be a factor, but is probably less important than habitat loss.

Dalquest (1948) noted that early explorers and settlers found White-tailed

Hares in abundance in eastern Washington. At the same time, Black-tailed Hares apparently were absent from the state prior to 1870 (Couch, 1927). With grazing by livestock the bunchgrass habitat was rapidly reduced, the Black-tailed Hares expanded their range explosively northward, and the White-tailed Hares survived only at the upper elevational limits of their habitat in the yellow pine/sagebrush-grassland ecotone. Today, White-tailed Hares are extinct over the greater part of their historic range on the Columbia Plateau. In California, they are nowhere common. Grinnell et al. (1930) found scant evidence of White-tailed Hares in the area from east of Lassen Peak to the Nevada border. Grinnell and Storer (1924) remarked that even in the center of their range in the Yosemite region they were much less common than the Black-tailed Hares of lower country. Howell (1924) failed to find White-tailed Hares in the Mammoth region, although he saw evidence of their presence. Orr (1940) declared that White-tailed Hares were one of the rarer members of the genus Lepus in California, noting that they were by no means abundant anywhere.

Currently, this is a Resident Small Game species in California with no closed season or daily bag limits.

Habitat: White-tailed Hares occupy a variety of habitats, including Sagebrush-covered slopes on the eastern Sierra Nevada, grasslands and meadows to timberline or above, and forests of Lodgepole Pine, Yellow Pine, Western Juniper, Dwarf Juniper, Red Fir, and mixed conifers. In the Sierra Nevada, White-tailed Hares apparently migrate to higher areas in summer and descend to lower regions in winter, particularly the Sagebrush-covered eastern slopes of the Sierra Nevada (Merriam, 1904; Orr, 1940).

Orr (1940) found signs of White-tailed Hares to be most abundant on exposed, flat-topped ridges above 8500 ft in El Dorado Co. Farther north in Oregon and Washington, White-tailed Hares were formerly most numerous on the bunchgrass prairies and the Sagebrush-bunchgrass associations near the Yellow-Pine zone (Bailey, 1936; Dalquest, 1948). Grinnell and Storer (1924) stated that in the Yosemite region, White-tailed Hares "generally keep to open places where they can see unobstructedly for long distances. Around Tuolumne Meadows, flat-topped hills bearing moderately open stands of trees together with some brush were often occupied."

Lepus townsendii apparently requires thickets of young or stunted conifers, or deciduous, woody plants for day-time cover. It feeds primarily at night in meadows during summer. Winter staples probably include bark of Willows, Alders and other woody plants

Recommendations: Population surveys throughout the range of L. townsendii in California should be carried out. Habitat needs of White-tailed Hares should be considered in management decisions on state and federal lands within their geographic range.

Remarks: Orr (1940) regarded the population in California formerly known as L. t. sierrae, as inseparable from L. t. townsendii.

Distribution Records: ALPINE CO.: Carson Valley, 5270 ft, 1 (MVZ); Hope Valley, 7800 ft, 1 (SDSNH), 1 (USNM); Woodfords, 1 (MVZ). EL DORADO CO.: 1 mi NW Dick's Peak, 9000 ft (Orr, 1940); Velma Lakes region (Orr, 1940).

FRESNO CO.: Badger Flat, E of Huntington Lake, 1 (CSUF); near Dinky Creek Ranger Station, about 6000 ft (DFW); Markwood Meadow, 1 (CSUF); Tamarack Ridge at Highway 168, about 7500 ft (DFW). INYO CO.: above Bishop, 1 (LACM); W slope Mt. Langley, 13800 ft (Grinnell). LASSEN CO.: Red Rock, 3 (MVZ), 2 (SDSNH); 7 mi W Termo (Grinnell, 1933). MARIPOSA CO.: E base Half Dome, 7500 ft (Grinnell and Storer, 1924). MODOC CO.: Eagleville, 1 (CAS); Goose Lake (Mailliard, 1927); Parker Creek, Warner Mountains, 6300 ft, 1 (MVZ); 3 mi S Pine Creek, 1 (MVZ); 2 mi E Pitt River Ranger Station, 1 (CAS); Steele Meadows, 4700 ft, 1 (MVZ); Warner Mountains, 2 (CAS). MONO CO.: 9 mi W Benton, 8300 ft, 1 (MVZ); Bodie, 8374 ft, 1 (MVZ); Bridgeport, 6473 ft, on U.S. Highway 395, 1 (MVZ); 8 mi SW Bridgeport, 7000 ft, 1 (CSLB); Cabin Creek, 10500 ft, White Mountains, 1 (MVZ); 10 mi N Crowley Lake, 7000 ft on U. S. Highway 395, 1 (MVZ); Farrington, 1 (MVZ); Glass Mtn., 11000 ft, 1 (MVZ); Lake Mary, 9000 ft, 1 (CAS); Log Cabin Rd. (Harris, 1982); Long Valley, Mammoth region (Howell, 1924); slopes of Mono Craters (Grinnell and Storer, 1924); Mono Lake, 1 (MVZ), 8 (USNM); Mono Mills (Grinnell and Storer, 1924); Mono Pass (Harris, 1982); Mt. Dana (Harris, 1982); 2.5 mi N Mt. Patterson, Sweetwater Mountains, 10000 ft, 1 (MVZ); 1.75 mi W Mt. Patterson, Sweetwater Mountains, 10500 ft, 1 (MVZ); 1 mi WNW Mt. Patterson, Sweetwater Mountains, 10800 ft, 1 (MVZ); Saddlebag Lake (Harris, 1982); Slate Creek Valley (Harris, 1982; 2 mi W mouth Sweetwater Canyon, Sweetwater Mountains, 10000 ft, 1 (MVZ); 4 mi SE Lower Twin Lake, 8000 ft (Summer Meadows, A. Farrington's Ranch), 2 (MVZ); Walker Lake (Harris 1982); Mt. Warren (Harris, 1982); near Williams Butte, 1 (MVZ); White Mountains, 12700 ft, 1 (CAS); White Mountain Research Station, White Mountains, 12000 ft, 1 (MVZ). PLACER CO.: Tahoe City, Lake Tahoe, 1 (MVZ). PLUMAS CO.: Willow Lake, 5600 ft (Grinnell, 1933). SHASTA CO.: vicinity of Fort Cook, near Burgettville, 1 (USNM). TULARE CO.: Big Cottonweel Meadows, Mt. Whitney, 1 (FMNH); SW end Lake South America, 11900 ft (Orr, 1940); Mineral King (Fry, 1924; Monache Meadows (Grinnell, 1933); N spur Mt. Silliman, 10400 ft (Fry, 1924). TUOLUMNE CO.: near Dana Creek, 2 mi W Mono Pass (Merriam, 1904); east base Half Dome, 7500 ft (Grinnell and Storer, 1924); Tuolumne Meadows, 8600 ft, Yosemite National Park, 1 (MVZ).

Point Reyes Mountain Beaver
Aplodontia rufa phaea

1899. *Aplodontia phaea* Merriam, Proc. Biol. Soc. Washington, 13:19.
1918. *Aplodontia rufa phaea*, Taylor, Univ. California Publ. Zool., 17:480.

Distribution: According to Taylor (1918), *A. r. phaea* inhabited an area of approximately 110 square miles in the Point Reyes area of Marin County. Apparently, most colonies were located on north facing slopes of hills and gullies (Camp, 1918).

Population Status: The status of *A. r. phaea* is unknown. Much of the region within its general range is unsuited to Mountain Beavers. The actual inhabited area was only a fraction of the 110 square miles encompassed by the margins of its range (Camp, 1918). The restricted distribution of *A. R. phaea* gives cause for concern. Much of the area within its range, however, is included in the National Park system, giving cause for optimism about the fate of this population.

Habitat: The general remarks about the habitat of *Aplodontia rufa* in the account of *A. r. nigra* applies also to this subspecies. Grinnell (1933) noted that *A. r. phaea* was found on hillsides in seepage areas overgrown with Sword Ferns and Thimbleberries, and that all localities of known occurrence were below 1000 ft in elevation. Camp (1918) found colonies on a north facing bluff among treeless hills west of Inverness.

Recommendations: Surveys of the distribution and habitat associations of Point Reyes Mountain Beavers should be undertaken. The National Park Service and other public agencies responsible for land management within the range of Point Reyes Mountain Beavers should be informed of the need for information on this species. The impact on its population of all potential developments within its range should be determined. Also, the effects of introduced cervids (Axis and Fallow deer) on habitat for Mountain Beavers should be assessed.

Remarks: *Aplodontia rufa phaea* is isolated from all other populations of Mountain Beavers by uninhabited terrain (Hall and Kelson, 1959). They are smaller in body sizes than all other populations, and are the lightest colored of those found along the Pacific Coast.

Distribution Records: MARIN CO.: Bear Valley Ranch, 2 (MVZ); Heim's Ranch, 5 mi W Inverness, 48 (MVZ); 3 mi W Inverness, 1 (MVZ); near highway, 4 mi W Inverness, 2 (UDAV); 6 mi W Inverness, 7 (MVZ); Lagunitas, 1 (CAS), 2 (MVZ); Limantour Bay, 1 (CAS); Marshall Ranch, 8 (MVZ); Murphy Ranch, about 4 mi WSW Inverness, 10 (MVZ); 4 mi S Olema, 4 (MVZ); 2 mi W Olema, 1000 ft, Mt. Wittenberg, 1 (MVZ); Point Reyes, 1 (CM), 15 (USNM); Point Reyes, 4 mi W Inverness, 2 (CM); 3 mi NE Point Reyes (city), 1 (CM); Wildcat Canyon, Tevis Ranch, 9 mi W Olema, 2 (MVZ).

Tehachapi Pocket Mouse
Perognathus alticola inexpectatus

1926. *Perognathus alticola inexpectatus* Huey, Proc. Biol. Soc. Washington, 39:121. Type Locality: 14 mi W Lebec, 6000 ft, Kern Co., California.

Distribution: Tehachapi Pocket Mice are known from a few scattered localities from Tehachapi Pass on the northeast to the area of Mt. Pinos on the southwest, and around Elizabeth, Hughes, and Quail Lakes on the southeast. Known localities are between about 3500 and 6000 ft in elevation.

Populations Status: Trapping for the Tehachapi Pocket Mouse by Sulentich (1983) produced 11 specimens from two localities on the eastern side of Tehachapi Pass in an area where a single specimens was caught in 1975 (Williams, 1978). In view of the scarcity of specimens and the general inability to find Tehachapi Pocket Mice, their populations must be small, scattered, and vulnerable to changes in habitat quality.

Habitat: The habitat of the Tehachapi Pocket Mouse is not well defined. Four specimens were captured in a fallow grain field at about 4000 ft, several years after it had been planted to grain (Sulentich, 1983; Williams, 1978). The predominate plant in the area when I collected there was Russian Thistle (Salsola sp.). Desert shrubs and grasses dominated nearby hills. Cantwell (in Huey, 1926) collected Tehachapi Pocket Mice on a grassy flat among scattered Yellow Pine trees. Other areas where Tehachapi Pocket Mice have been collected support arid annual grassland and desert shrub communities.

Recommendations: Additional studies of habitat requirements and distribution are needed. Areas likely to support populations are the mountain slopes of the Tehachapi and San Gabriel ranges that front on the Mojave Desert.

Remarks: Perognathus alticola inexpectatus appears to be allied with P. a. alticola structurally. Sulentich (1983) believed that the Tehachapi Pocket Mouse was a distinct species from the White-eared Pocket Mouse. Populations of P. a. inexpectatus may be in contact with and intergrade with populations of P. parvus xanthonotus, which is known from an area of the Tehachapi Mountainsl in the vicinity of Walker Pass and adjacent canyons. Habitat for Pocket Mice of this species group appears to be nearly continuous along the desert slopes of the southern Sierra Nevada, Tehachapi Mountains, San Gabriel Mountains, and San Bernardino Mountains. The allopatric distributions of these taxa (Hall, 1981) probably is more apparent than real. Eight Tehachapi Pocket Mice, in addition to the specimens listed below, were captured in the area of Tehachapi Pass by Sulentich (1983, and unpubl.).

Distribution Records: LOS ANGELES CO.: Elizabeth Lake, 4 (LACM); Gorman, 1 (LACM); 2 mi E Gorman, 3500 ft, 3 (MVZ); 2.5 mi SE Gorman, 1 (MVZ); Lake Hughes, 1 (LACM); 2 mi W Quail Lake, 3500 ft, 1 (LACM). KERN CO.: 0.5 mi SW Cameron, 3900 ft, 4 (MVZ); 14 mi W Lebec, 6000 ft, 3 (SDSNH); Cuddy Valley, Mt. Pinos, 1 (LACM); Sand Canyon, about 8 mi E (by rd.) Tehachapi, sec. 28, T32S, R34E, 4080 ft, 3 (CSLB), 1 (MSB); Tehachapi, 2 (USNM); Tehachapi Peak, 2 (USNM).

Short-nosed Kangaroo Rat
Dipodomys nitratoides brevinasus

1920. Dipodomys merriami brevinasus Grinnell, J. Mamm., 1:179. Type Locality: Hays Station, 19 mi SW Mendota, Fresno Co., California.
1921. Dipodomys nitratoides brevinasus, Grinnell, J. Mamm., 2:96.

Distribution: Short-nosed kangaroo Rats are found on the western side of the San Joaquin Valley, from near Los Banos, Merced Co., southward west of the San Joaquin River and a line approximately coincident with the Kettleman Hills, Lost Hills, and Elk Hills to the southern end of the valley. They also occur in the Panoche Valley, San Benito Co., the Sunflower Valley, Kings Co., the Antelope Plain in Kern Co., the Carrizo Plain in San Luis Obispo Co., the Cuyama Valley in San Luis Obispo and Santa Barbara counties, and at the edge of the valley floor around the south end of the San Joaquin Valley from the vicinity of Maricopa on the West to east of Bakersfield on the east (Hall, 1981; Williams, 1985, and unpubl. data). Within this area, they are found mostly on flat and gently sloping terrain and on hill tops in desert-shrub associations, primarily Atriplex.

Population Status: Loss of habitat has been extensive throughout the range of the Short-nosed Kangaroo Rat, particularly on flatter lands in the Cuyama, San Joaquin and Panoche valleys, and on the Antelope and Carrizo plains. Cultivation of native communities has been the principal reason for loss of habitat. Relatively small, isolated populations are found on the southern Antelope Plain west of Buttonwillow and in the vicinity of Taft and Maricopa, Kern Co.; on uncultivated sites in the Kettleman Hills, and in and around oil fields near Coalinga, Fresno Co.; west of Interstate Highway 5 on unirrigated lands at the edge of the valley in Fresno Co.; and around Soda Lake on the Carrizo Plain, San Luis Obispo Co. Where they are found, they are typically common. A major cause for concern is that virtually all of the areas where Short-nosed Kangaroo Rats still are found are privately owned and have moderate to good potentials for cultivation. Only the lack of irrigation water has prevented cultivation of most areas until now. Of greater concern is the lack of lands dedicated to preserving natural communities within its range, except for the area immediately around Soda Lake on the Carrizo Plain (few Short-nosed Kangaroo Rats occur within the confines of the present preserve). Public-owned lands within the geographic range of Short-nosed Kangaroo Rats, such as those on the Carrizo and Elk Horn plains, recently have been subject to intense grazing pressures with virtually no plant litter remaining (some areas are being grazed year around), and providing little opportunity for herbaceous plants to fruit (unpubl. observ.). Preliminary field work in some of these areas during the past year found no Short-nosed Kangaroo Rats in areas where they were fairly common as recently as 1982 (Kerry Kilburn, pers. comm.; unpubl. data).

The Short-nosed Kangaroo Rat was not included on the priority List of Concern when the draft copy was assembled, primarily because it appeared to warrant lesser concern than many other species. In the ensuing five years, however, additional field work within its geographic range, focused mostly on Giant Kangaroo Rats, found that some populations once considered secure were declining or had disappeared. Some other areas where Short-nosed Kangaroo Rats were presumed to occur, based on the type of plant communities, proved not to have extant colonies. Thus, in reassessing its status, it seemed best to elevate it to the Third Priority Category to emphasize potential threats to remaining populations and to expedite the gathering of information needed to determine its status.

Habitat: *Dipodomys nitratoides brevinasus* occupies grassland and desert-shrub associations on friable soils. They inhabit highly alkaline soils around Soda Lake, on the Carrizo Plains, and less saline soils elsewhere. On the valley floor, around Los Banos, Merced Co., small populations live on dikes secure from winter flooding, then move into seasonally flooded iodine bush shrublands during the summer months, where at least some individuals reproduce (unpubl. data). Hawbecker (1951) reported that, in the Panoche Valley, San Benito Co., Short-nosed Kangaroo Rats were found on gentle slope and rolling, low hill-tops where some shrubs were present. He suggested that they did not "venture far out of this habitat." Light to moderate grazing by livestock probably enhances habitat for Short-nosed Kangaroo Rats.

Recommendations: The primary needs are for detailed information on distribution and populations status of *Dipodomys nitratoides brevinasus*, and land use projections within areas still inhabited. The U.S. Bureau of Land Management should pay particular attention to this species in its management plans for lands in western Kern Co., and on the Carrizo and Elk Horn plains in San Luis Obispo Co. The impact on Short-nosed Kangaroo Rats and other native species by the present pattern of livestock grazing should be determined and grazing leases reviewed accordingly. The California Department of Fish and Game should review its opportunities to purchase and develop lands as a preserve for this and other jeopardized species of the San Joaquin Valley lowlands, especially in northwestern Fresno and eastern San Benito counties in the Panoche Valley area.

Remarks: Because the Short-nosed Kangaroo Rat was not on the draft List of Concern, distribution records were not obtained from most museums visited. The basic distributional area is documented in Grinnell (1922) and Hoffmann (1975). Williams (1985) discussed, in detail, the boundary between the subspecies *D. n. nitratoides* and *brevinasus*. Ten specimens identified as *D. nitratoides* from 3 mi E Cuyama Ranch, 2200 ft, Cuyama Valley (probably San Luis Obispo Co.) collected in 1916 (USNM collection) were not mentioned by Grinnell (1922) or Hall (1981). Hoffmann (1975) examined specimens from the Cuyama Valley, but did not identify the number or museum where those specimens were housed (I presume that they were the specimens in the U.S. National Museum). He compared them with samples of *nitratoides* from other localities and determined that they exhibited characteristics more diagnostic of *Dipodomys merriami* (fide Lidicker, 1960), but equivocated as to their identity. Any surveys for *Dipodomys nitratoides brevinasus* should include the Cuyama Valley and should be designed to obtain information useful in resolving the specific identity of that population, presuming it is extant.

Red Tree Vole
Arborimus longicaudus

1890. *Phenacomys longicaudus* True, Proc. U.S. Natl. Mus., 13:303. Type Locality: Marshfield, Coos Co., Oregon.
1973. *Arborimus longicaudus*, Johnson, J. Mamm., 54:239.

Distribution: Red Tree Voles are found along the Pacific coastal lowlands in Oregon and Northern California (Howell, 1926; Hall, 1981; but see Remarks). In California, they range from the Oregon border southward to Sonoma County along the coast, and in the coastal mountain ranges southward to about Mt. Sanhedrin, Mendocino Co. Marginal records on the east are Bridgeville, Humboldt Co.; South Fork Mountain, Trinity Co.; Mt. Sanhedrin, Mendocino Co.; and Occidental, Sonoma Co. (Hall, 1981; Howell, 1926). Records of elevation extend from near sea level to about 3100 ft in California (Grinnell, 1933; unpubl. data).

Population Status: Red Tree Voles have always been considered rare, although they are common locally (Taylor, 1915). Records of distribution within California suggest a spotty pattern of dispersion. Their reproductive potential is lower than typical voles (Hamilton, 1962), and presumably, populations could be jeopardized by greatly increased mortality associated with extensive logging. Clearing trees for agriculture and homesites, and logging, especially clear-cutting, within the range of Red Tree Voles have significantly reduced available habitat and fragmented populations (Maser et al., 1981). Construction of roads and powerlines also have contributed to loss of habitat and fragmentation and isolation of populations.

The Red Tree Vole was not listed in any of the three priority categories on the draft List of Concern, but was considered a sensitive species. Information on its population status was unavailable, but it did not seem to be jeopardized. Since development of the draft list, I have reconsidered its status in light of a trip through portions of its geographic range in search of populations, made in April, 1984. Although Red Tree Voles are still locally common in the foothills of mountains on the east edge of the coastal plain in Humboldt Co., loss and fragmentation of habitat has been extensive everywhere within its range. Furthermore, these trends are likely to continue at an accelerated pace in the future. The Red Tree Vole has been elevated to the Third Priority Category to emphasize that current trends in logging may jeopardize the species and to emphasize the need for definitive information on distribution and population status.

Habitat: Red Tree Voles live only in coastal coniferous forests consisting of Douglas Fir, Grand Fir, Western Hemlock, and/or Sitka Spruce. Taylor (1915) remarked that most Red Tree Voles were found in Douglas Fir and Grand Fir trees, and that only Todd (1891) found these voles in Sitka Spruce. Maser and Storm (1970) stated that Red Tree Voles also inhabit Western Hemlock trees. They live, nest, and feed within the forest canopy. Males are partly terrestrial, but females rarely are found on the ground. Red Tree Voles feed on the needles, buds, and tender bark of twigs of Douglas Fir, Western Hemlock, and Grand Fir (Howell, 1926; Maser and Storm, 1970; Maser et al., 1981; and Taylor, 1915). They do not eat Redwood needles; hence pure stands of Redwood are not occupied (Taylor, 1915). Redwood is the principal forest type in preserves within the range of the Red Tree Vole. Generally, large trees are preferred as homes, although small trees are also inhabited, especially where larger ones are unavailable (Benson and Borrell, 1931; Howell, 1926). Condensation of water on leaves from dew and fog or frequent rains may be required as a source of drinking water (Taylor, 1915). Although many of the factors determining the occurrence of Red Tree Voles are not known, these animals probably require fairly dense, mature stands of conifer forest composed of at least some Douglas Fir or Grand Fir. Clear-cuts, forest fires, and other factors that create openings in the forest and isolate blocks of trees are detrimental to Red Tree Voles.

Large nests are constructed in trees, generally around the trunk, typically from 20 to 60 ft above ground (Benson and Borrell, 1931). Nests have been found as low as 4 ft to as high as 100 ft or more. The nests consist primarily of the resin ducts of fir needles, the only part not eaten by the voles, and fecal material. The bulk of the nest increases over the years as a series of residents occupy the site and add material.

Recommendations: The most pressing need is for detailed information on the present distribution and population status of Red Tree Voles. Known and projected developments within its present range should be assessed to determine their impact on the species. State agencies, the U.S. Forest Service, U.S. Bureau of Land Management, and the U.S. National Park Service should conduct detailed surveys for Red Tree Voles on lands they manage. Clear-cutting in areas inhabited by Arborimus longicaudus should be avoided in preference to selective logging whenever possible.

Remarks: Until recently, the Red Tree Vole was considered to be a member of the genus Phenacomys. Johnson (1973) reviewed the structural, ecological, and behavioral characters of Arborimus and Phenacomys. Additional studies of the generic differences of Arborimus and Phenacomys were presented by Johnson and Maser (1982). According to Maser et al. (1981) and Sara George (pers. comm.), the Red Tree Vole in California is a different species than that found in most of Oregon; Murray Johnson is currently investigating the relationships of these sibling populations and their geographic distributions.

During the research for this report, I decided Arborimus longicaudus was not to be included in a priority category on the List of Concern; thus, I stopped recording distributional records during visits to museums. Therefore, the following list does not include records of specimens from all of the museums visited. Noteworthy are the collections at the California Academy of Sciences and Humboldt State University.

Distribution Records: HUMBOLDT CO.: 5 mi E Arcata, 1 (CSLB); Big Bend, Mad River, 2 (MVZ); Bridgeville, 1 ((MVZ); 4.3 mi N Bridgeville, 1 ((MVZ); 4.6 mi N Bridgeville, 2 ((MVZ); 2 mi E Bridgeville (Benson and Borrell, 1931), 1 (MVZ), 1 (PM); 4.6 mi by rd. (Hwy 36) E Bridgeville, 1 (MVZ); 1 mi W Bridgeville (Benson and Borrell, 1931); 1.5 mi W Bridgeville, 1 (MVZ); 2 mi W Bridgeville, 2 (UCLA); 3 mi N Capetown, Oil Creek, 2 (MVZ); 5.3 mi NE Capetown, Morrow Ranch, 1450 ft, 5 (MVZ); Carlotta, 5 (MCZ), 13 (MVZ), 2 (ROM), 4 (SDSNH), 3 (UCLA), 18 (USNM); Chaparral Mt., 2 (MVZ); Coyote Peak, 1 (MVZ); Cuddeback, 1 (MVZ); French Camp, 3100 ft, 1 (MVZ); 2 mi along Johnson Rd., off Bald Hill Rd., 1 (CSLB); 8.2 mi (by rd.) E Korbel, 1 (MVZ); Maple Creek, 1 mi N jct. Mad River, 4 (MVZ); 0.5 mi S (by rd.) Maple Creek, 1 (MVZ); 3.2 mi NW Petrolia, 1 (MVZ); 5.2 mi SE Petrolia, 1 (MVZ). MENDOCINO CO.: Anchor Bay, 3 (CM); 10 mi E Point Arena, Garcia River, 3 (CM); 8 mi SW Laytonville, Clark Ranch, 1 (MVZ); Mendocino City, 16 (MVZ); Mt. Sanhedrin, 1 (ANSP); 4 mi S Mt. Sanhedrin, Leerly's Ranch, 2340 ft, 1 (MVZ); 0.8 mi S Slick Rock Creek on State Hwy #1, 2 (MVZ); 3 mi W Whiskey Spring, Graveyard Rd., 800 ft, 1 (MVZ). SONOMA CO.: 1 mi N, 0.2 mi E Bridgehaven, 50 ft, 1 (MVZ); 1 mi N Camp Meeker, 1 (MVZ); 3.5 mi N Camp Meeker, 1 (MVZ); Duncan Mills, 1 (LACM); 3 mi E Duncan Mills, 1 (MVZ); 2 mi W Duncan Mills, 4 (MVZ); 3 mi W Duncan Mills, 1 (MVZ); 3.5 mi W Duncan Mills, 1 (MVZ); 7 mi N (by rd.) Fort Ross, 1 (MVZ); 1 mi E Jenner, 3 (MVZ); near Monte Rio, Bohemian Grove on Russian River, 2 (MVZ); 2 mi SE Monte Rio, 1 (CM); 2 mi S Monte Rio, 1 (MVZ); 2.3 mi S Monte Rio, 11 (MVZ); 3 mi S Monte Rio, 4 (MVZ); 4 mi S Monte Rio, 1 (MVZ); 1 mi W Monte Rio, 1 (CM); Occidental, 2 (LACM); 2 mi N Occidental, 3 (MVZ); 0.5 mi S Occidental, 2 (MVZ); 1 mi S Occidental, 1 (MVZ); 0.25 1 mi W jct. Sheephouse Creek and Russian River, 1 (MVZ); 0.5 mi E Stewart's Point, 1 (MVZ); 10 mi SSE Stewart's Point, 1 (MVZ). TRINITY CO.: Mad River, 2500 ft, South Fork Mtn., 1 (MVZ); Reilley's Ranch, 3000 ft, South Fork Mtn., 4 (MVZ).

Pacific Fisher
Martes pennanti pacifica

1898. Mustela canadensis pacifica Rhoads, Trans. Amer. Philos. Soc., new ser., 19:435. Type Locality: Lake Keechelus, Kittitas Co., Washington.
1912. Martes pennanti pacifica, Miller, Bull. U.S. Mus., 79:94.

Distribution: Martes pennanti was formerly widely distributed in boreal forests across Canada and the northern United States, extending south in the mountains to California and Utah in the west and Tennesee and North Carolina in the east (Powell, 1981; Strickland et al., 1982a).

Pacific Fishers are found along the Pacific Coast from north-central coastal British Columbia to northern California, in the Cascade Mountains, and the Sierra Nevada of California. They range eastward to the Blue Mountains of northeastern Oregon (Hall, 1981). In California, they occur from the Oregon Border in the northwestern part of the state south to about Clear Lake, Lake County, in the Coastal Mountains and to Sonoma County along the coast. They range eastward in northern California to around Eagle Lake, Lassen County, and southward in the Sierra Nevada to at least Greenhorn Mountain, Kern County (Schempf and White, 1977). Altitude of occurrence varies from near sea level to over 11000 ft.

Population Status: Schempf and White (1977) reviewed historic and recent records of Fishers in California, including sightings on file in U.S. Forest Service offices and from California Department of Fish and Game records. Data they gathered suggested that Fishers were relatively common in the North Coast region, but were rare or uncommon in the Sierra Nevada and appeared to be decreasing. These data were unsupported by field surveys, so recent trends were unknown. Particular attention was drawn to the apparent decline of populations in the southern Sierra Nevada and possible declines in the northern Sierra Nevada. Buck et al. (1983) studied a population of Fishers In Trinity Co. to determine habitat use and possible effects of timber harvest on Fishers. They found that clearcutting and the general trend toward reduction in size and isolation of mature stands were probably detrimental to Fisher populations. They also determined that selective cutting of trees decreased the quality of habitat for Fishers.

Habitat: Fishers prefer heavy stands of mixed species of mature timber, but they range widely in forested regions (Buck et al., 1983). They are frequently encountered in second-growth forests, and sometimes in forest openings. A few records of Fishers have been found in communities such as scrub woodlands at lower elevations (Schempf and White, 1977). This does not mean, however, that Fishers can live in these habitats permanently. In California, Fishers primarily inhabit mixed conifer forests composed of Douglas Fir and associated conifers, although they also are encountered frequently in higher elevation, fir and pine forests such as Red Fir and Lodgepole Pine, and mixed evergreen/broad leaf forest.

Fishers prey on a variety of small- and medium-sized mammals, especially Chipmunks, Squirrels, Marmots, Woodrats, Porcupines, Rabbits, Hares, Martens, Foxes and other small carnivores (Powell, 1981; Strickland et al., 1982a; Yocum and McCollum, 1973). Grinnell et al. (1937) speculated that Fishers also feed on birds such as Grouse.

Fishers are known to den in cavities near the tops of large trees, in hollow logs on the ground, in talus, and crevices in rock outcrops (Grinnell et al., 1937).

Recommendations: Field surveys are needed to establish a data base against which future trends can be assessed, and to define more accurately current status. Attention should focus on the Sierra Nevada, as evidence suggests declining populations there (Schempf and White, 1977).

Effects of various forest harvesting practices on Fisher populations should be determined over a broader area (Buck et al., 1983); and specific habitat needs such as stand density and tree size, food sources, and den sites should be determined. Snags, damaged and senescent trees with large cavities, and hollow logs are probably important for Fishers, especially where talus and rock crevices are unavailable.

Remarks: Schempf and White (1977) presented several records (mostly sightings) in addition to those listed below and referred to an unpublished paper where these records were documented. Yocum and McCollum (1973) mapped several localities in Del Norte, Humboldt and Trinity counties, but did not present descriptions of the localities. Bruce and Weick (1973) and Gould (1978) also presented data on Fishers in northwestern California and the northern Sierra Nevada, respectively.

Grinnell et al. (1937) considered *M. p. pacifica* to be inseparable from *M. p. pennanti*. Goldman (1935), however, found that the two taxa were separable. Hall and Kelson (1959) believed that evidence presented by Goldman was sufficient for subspecific status.

Distribution Records: DEL NORTE CO.: 3 localities unspecified (Yocum and McCollum, 1973). HUMBOLDT CO.: Waterman Ridge, Hawkins Bar Road (Yocum and McCollum, 1973). LAKE CO.: near Lakeport, 1 (CAS). LASSEN CO.: Eagle Lake (Grinnell et al., 1930). MARIPOSA CO.: Big Creek, near Wawona, 2 (USNM); Big Meadows, Coulterville Road, 4000 ft, 1 (MVZ); Bridalveil Creek (Cunningham, 1959); Chinquapin, 1 (MVZ); near Crane Flat, Yosemite National Park, 1 (MVZ); Fort Monroe, 1 (MVZ); Grouse Creek, 1(MVZ); Yosemite National Park, 1 (MVZ); Yosemite Valley, 3 (MVZ), 1 (USNM); Wawona, 1 (USNM). MENDOCINO CO.: Cahto, 1 (USNM); near Covelo, 1 (USNM); Eden Valley, 1 (MVZ). SHASTA CO.: Cassel, Burney Mountain, 1 (USNM); Cassel, Rock Creek Mountains, 1 (USNM); Fort Crook, 1 (USNM). SIERRA CO.: Webber Lake Area (Gould, 1978). SISKIYOU CO.: near Cecilville, 4 (MVZ); Mount Shasta, 1 (USNM). TRINITY CO.: near Helena, 2 (MVZ); Wells Creek, E fork, 15 mi E Hay Fork, 1 (USNM); S fork Trinity River, sec. 29, T1N, R7E, 2000 ft, 1 (MVZ). TULARE CO.: Atwell's Mill, 1 (USNM); Kern River, 2 mi downstream from junction with Little Kern River, 1 (MVZ). TUOLUMNE CO.: near Hetch Hetchy Valley, 3 (MVZ); Hog Ranch Ranger Station, Yosemite National Park, 1 (MVZ); head Lyell Canyon (Grinnell and Storer, 1924); Tuolumne Big Trees, 2 (MVZ).

American Badger
Taxidea taxus

1778. Ursus taxus Schreber, Die Saugthiere. . ., 3:520. Type Locality: Labrador and Hudson Bay, Canada.
1894. Taxidea taxus, Rhoads, Amer. Nat., 28:524.

Distribution: American Badgers occur from northern Alberta southward to central Mexico. They range from the Pacific Coast eastward through Ohio. They are absent from the humid coastal forests and from other regions with dense forests. In California, Badgers ranged throughout the state except for the humid coastal forests of northwestern California in Del Norte Co. and the northwestern portion of Humboldt Co. (Long 1973; unpubl. data).

Population Status: Badger populations have declined drastically in California within the last century (Grinnell et al., 1937; Longhurst, 1940). Grinnell et al. (1937) noted that Badgers were reduced in numbers over almost all of their range in California by 1937. At that time they were still numerous in the Central Valley, but now they survive only in low numbers in peripheral parts of the valley and adjacent lowlands to the west in eastern Monterey, San Benito and San Luis Obispo counties. In the coastal areas from Mendocino county south they have been drastically reduced in numbers. They have been extirpated from many areas in southern California. Long and Killingby (1983) regarded the status of Badgers in California as poor. Deliberate killing probably has been a major factor in the decline of Badger populations. Most people regard Badgers as detrimental to their interests and attempt to kill them. Cultivation is adverse to Badgers, as they do not survive on cultivated land. Agricultural and urban developments have been the primary causes of decline and extirpation of populations of Badgers in California. Rodent and predator poisoning pose double threats through direct and secondary poisoning of Badgers and elimination of the food Badgers are dependent upon. Shooting and trapping of Badgers for animal "control" is another source of mortality. The U.S. Fish and Wildlife Service took 4086 Badgers in California from 1966 to 1976 (Lee, 1977). Trapping of Badgers for the fur trade probably has had little impact on populations in many areas because the fur was of low economic value. In the late 1920's to at least the late 1930's, Badger fur was in high demand and trapping increased to levels that may have decimated local populations (Grinnell et al., 1937). Again, subsequent to 1975, demand for Badger pelts has increased and increased efforts are being expended to trap Badgers.

No current data exist on the status of Badger populations in California, but they have obviously declined or disappeared in large sections of the state, particularly areas west of the Cascade-Sierra Nevada mountain axis and in coastal basins of southern California. Badgers were common in mountainous areas only in large, treeless meadows and expanses near timberline. Longhurst (1940) noted that they had nearly disappeared from Napa County by 1940.

Despite the probable continuing decrease in numbers of Badgers statewide, reports of numbers trapped for the fur market indicate substantial increases in captures in recent years (e.g. 107 Badgers reported trapped in 1975-76 and 299 in 1976-77; California Dept. of Fish and Game, unpubl. report). Most of these Badgers were taken in the northern and eastern counties, although Fresno and San Benito counties produced 45 and 20 respectively, in 1976-77. The increase in numbers trapped most likely reflects the increased prices paid for pelts and the consequently greater effort expended in trapping Badgers. For example, 931 trapping licenses were sold in 1975-76 and 1692 in 1976-77. Less than one-half of the licencees filed reports of their captures both years (207 and 751, respectively).

Habitat: In California, Badgers occupy a diversity of habitats. The principal requirements seem to be sufficient food, friable soils, and relatively open, uncultivated ground. Grasslands, savannas, and mountain meadows near timberline are preferred. Badgers prey primarily on burrowing rodents such as Gophers (*Thomomys*), Ground Squirrels (*Spermophilus*, *Ammospermophilus*), Marmots (*Marmota*), and Kangaroo Rats (*Dipodomys*). They are predatory specialists on these rodents, although they will eat a variety of other animals, including mice, Woodrats, reptiles, birds and their eggs, bees and other insects, etc. Grinnell et al. (1937) recounted reports of Badgers breaking open bee hives to eat both the brood and honey. They regularly dig out nests of Bumble Bees.

One report of densities of Badgers reviewed by Long gave an estimated density of one Badger per square mile. Bailey (1905) noted that one Badger spent a summer in a 20-acre field. Sargent and Warner (1972) found that a radio-collared female had a home range of 850 hectares (2091 acres): 725 hectares (1783 acres) in summer, 53 hectares (130 acres) in fall, and 2 hectares (5 acres) in winter. Messick and Hornocker (1981) found that home ranges averaged 2.4 and 1.6 sq. km for adult males and females, respectively, in Idaho.

Recommendations: Current data on Badger populations are needed throughout the state, especially from the lowlands of western California. The effects of continuing habitat loss, rodent poisoning, and trapping for the fur trade should be assessed. Mandatory reporting of take (including animals discarded) by trappers and hunters should be required. Information on home range size and density of prey required by Badgers is needed for effective management. The impact on Badgers of the use of rodenticides and trapping for the fur trade should be assessed.

Remarks: Long (1972) revised *Taxidea taxus* primarily on the basis of specimens in the U.S. National Museum. While his paper has contributed much to an understanding of geographic variation in this species, it has also confused the taxonomy of Badger populations in California. *T. t. jeffersonii* (Harlan) generally ranges in the better-watered areas of California, including coastal areas, most of the Sierra Nevada, and most of the Great Basin Province. *T. t. berlandieri* Baird ranges through the hotter, drier desert and grassland associations of southeastern California and the Central Valley. This makes sense from an ecogeographic point of view: larger, darker-colored Badgers from cooler, moister areas and smaller, lighter-colored Badgers from hotter, drier areas. The problem occurred with the assignment of specimens to particular subspecies. A specimen from Alila (Earlimart), Tulare County, on the floor of the southern San Joaquin Valley, was assigned to *jeffersonii*, while another specimen from Alila was assigned to *berlandieri*.

Specimens from geographically adjacent and ecologically continuous areas (hot and arid) of the San Joaquin Valley were assigned to jeffersonii (e.g. Tulare Lake, Huron, Stanley, Alcalde, and Taft), while others from Tracy, San Joaquin County, were assigned to berlandieri. These and other taxonomic assigments by Long (1972) make no sense from a geographic or environmental perspectives. Long (1972) noted that some of these specimens exhibit intermediate characters, suggesting intergradation. This is probably the case. I can see no compelling reason, however, to assign the Central Valley population to berlandieri. In fact, Long assigned most of the specimens from this region to jeffersonii, although his range map and statements lead to the opposite conclusion. A better arrangement would be to include all specimens from the Central California lowlands in T. t. jeffersonii. For this account, however, I do not use trinomials for Badger populations. The principal concern is for populations in the lowlands of western California, west of the main Cascade-Sierra Nevada mass and the southern California coastal region. This would include some populations that Long (1972) assigned to T. t. jeffersonii and some he called T. t. berlandieri.

Distribution Records: ALAMEDA CO.: Oakland, 2 mi NE Mills College, 1 (MVZ). BUTTE CO.: 18 mi W Oroville, 1 (CSUC). CONTRA COSTA CO.: Rattlesnake Canyon, near Orinda, 1 (MVZ). EL DORADO CO.: Echo, 7500 ft, 2 (MCZ). FRESNO CO.: Alcalde, 1 (USNM); 7 mi SW Coalinga, 1 (CAS); Huron, 374 ft, 1 (USNM); 0.6 mi NE Marion Lake, 10500 ft, Kings Canyon National Park, 1 (MVZ); Panoche Creek, 550 ft, 1 (MVZ). HUMBOLDT CO.: 27 mapped localities without locality descriptions, based on sight records (C. F. Yocum, in litt.). INYO CO.: no specific locality, 1 (LACM); Furnace Creek Ranch, 1 (MVZ); 3 mi NE Jackass Spring, 1 (MVZ); 7 mi E Laws, at Silver Creek, 1 (MVZ); Wild Rose Canyon, 1 (MVZ). IMPERIAL CO.: Alamo Duck Preserve, 8 mi NW Calipatria, 1 (MVZ); Bard, 1 (UCLA); 3 mi N Bard, 2 (SDSNH); 6 mi W Bard, 1 (SDSNH); 5 mi N Laguna Dam, 1 (MVZ); Manganese Wells, Lower Colorado River, 1 (MVZ); Palo Verde, 1 (LACM), 1 (MVZ); 0.75 mi N Palo Verde, 1 (MVZ); 13 mi N Palo Verde, 1 (MVZ); 18 mi WNW Palo Verde, 1 (MVZ); 20 mi N Picacho, Colorado River, 1 (MVZ); Silsbee, 1 (MVZ). KINGS CO.: Stanley, 1 (USNM). KERN CO.: Antelope Valley, near Neenach, 1 (FMNH); Bakersfield, 1 (MVZ); Buttonwillow, 1 (CAS); 3 mi S Cantil P. O., 1 (LACM); 3 mi SE Cantil, M & R Ranch, 1 (LACM); 3 mi ENE Hart's Place, 1 (LACM); 4 mi S Inyokern, 1 (LACM); 4 mi SW Inyokern, 1 (LACM); S Fork Kern River, 25 mi from Kernville, 1 (USNM); 5 mi NW Mojave, 3350 ft, 1 (MVZ); Tulare Lake, mouth of Kern River, 2 (USNM); Taft (Long, 1972); E side Walker Pass, 5000 ft, 1 (LACM); Willow Springs, 1 (AMNH). LAKE CO.: Lakeport, 1 (CAS); several miles N Upper Lake, 1 (WFBM). LASSEN CO.: Amedee, 1 (USNM); Calneva, 1 (MVZ); Hayden Hill, 1 (USNM); Karlo, 2 (MVZ); 2 mi S Madeline, 1 (HSU); Merrillville, 1 (USNM); 7 mi N Observation Peak, 5300 ft, 1 (MVZ); Poison Lake, 1 (USNM); 20 mi E Susanville, 1 (CSUC); 10 mi E Ravendale, 5400 ft, 1 (CAS); Susanville, 1 (USNM); Termo, 1 (MVZ); Willow Creek, Barran Ranch, 1 (CSUC). LOS ANGELES CO.: Covina, 1 (UCLA); Fairmont, Antelope Valley, 1 (LACM), 1 (MVZ); Los Angeles, 1 (LACM); near Lovejoy Buttes, 1 (MVZ); Tejunga Wash, 1 (MVZ). MADERA CO.: San Joaquin Experimental Range (Newman and Duncan, 1973); head San Joaquin River, 2 (USNM). MARIN CO.: Bear Valley Ranch, Olema, 1 (MVZ); Bolinas, 1 (MVZ); 0.75 mi from beach, 1.25 mi NW Bolinas, 1 (MVZ); Fort Barry, 1 (MVZ); 3 mi W Inverness, 2 (MVZ); Millerton Gulch, 2.25 mi NE Inverness, 1 (MVZ); 7 mi N Novato, 1 (MVZ); Tomales Point, 1 (MVZ). MARIPOSA CO.: no locality specified, 1 (USNM); Wawona, 1 (USNM). MENDOCINO CO.: Clarke Ranch, 8 mi SW

Earlimart], 283 ft, 6 (USNM); Otosi, 4 (USNM); 4 mi SW Porterville, 1 (AMNH); White River, 1 (CAS); Whitney Meadows, 9800 ft, 1 (MVZ). TUOLUMNE CO.: Tuolumne Meadows, 4 (MVZ), 1 (USNM). VENTURA CO.: Mount Pinos, 1 (MVZ); Mount Pinos, 5500 ft, 1 (LACM); Saticoy, 1 (MVZ). YOLO CO.: Davis, 1 (UDAV); Woodland, 1 (UDAV).

Channel Islands Spotted Skunk
Spilogale gracilis amphiala

1929. *Spilogale phenax amphialus* Dickey, Proc. Biol. Soc. Washington, 42:158. Type Locality: 2.5 mi N ranch house near coast, Santa Rosa Island, Santa Barbara Co., California.
1933. *Spilogale gracilis amphialus*, Grinnell, Univ. California Publ. Zool., 40:105.

Distribution: Channel Islands Spotted Skunks are known to occur only on the islands of Santa Cruz, Santa Rosa and San Miquel. They are probably extinct on San Miquel Island, however (Walker, 1980).

Population Status: Nothing specific is known about the status of Spotted Skunks on the Channel Islands. Grinnell et al. (1937) noted that "quite a few" skins of these Skunks were received from Santa Cruz Island by Colburn's taxidermy shop in Los Angeles in 1918. Laughrin (1973) noted that Spotted Skunks were quite rare when he surveyed Santa Cruz Island in 1973. According to von Bloeker (1967), Spotted Skunks were once very common on Santa Cruz and Santa Rosa Islands, but by 1967 they were rarely found on either island, at least near human dwellings.

Remarks by these authors were subjective impressions; there have been no studies of population size on either island. The seeming rarity of Spotted Skunks may indicate normal population fluctuations, or reflect a real decline in numbers.

Santa Rosa and Santa Cruz are the two largest of the Channel Islands. Both are privately owned, and both have had less habitat alteration and fewer introductions of exotic mammals than most of the other islands. According to Laughrin (1973) Wapiti (*Cervus elaphus*), Mule Deer (*Odocoileus hemionus*), Wild Pigs (*Sus scrofa*), Cattle and Horses, occupied Santa Rosa Island in 1973 in addition to native mammals. Sheep formerly were present, but apparently have been completely removed. A list of currently extant, introduced species on Santa Cruz Island is unavailable. Von Bloeker (1967) mentioned Horses, Wild Pigs, Cattle, and Roe Deer (*Capreolus capreolus*) as being present, and implied that feral cats were established on both islands. Laughrin (1973) noted that Sheep also occurred on Santa Cruz Island, but fences were erected to restrict them to the north side. A hunting program to reduce their numbers was in effect at that time.

The principal reason for concern about the Channel Islands Spotted Skunk is the scanty information available suggesting a significant decline in populations. Because island biota are more prone to extinction than those of mainlands, concern is heightened. Human disturbances on the islands are probably not sufficient to cause this decline. Domestic cats and/or dogs have possibly introduced diseases to which the Skunks are

Laytonville, 1 (MVZ); Eden Valley Ranch, 18 mi above Willits, 1 (MVZ). MERCED CO.: Los Banos, 1 (MVZ); Los Banos Game Refuge, 2 (MVZ); 10 mi S Merced, 1 (CSCS); 2 mi SW Stevinson, 1 (CSCS). MODOC CO.: 3 mi E Alturas, 1 (MVZ); 16 mi W Alturas, 1 (USNM); Cedarville, 1 (USNM); Dry Creek, 3 (MVZ); Eagleville, 1 (CAS); 5 mi NE Fort Bidwell, 1 (MVZ); 7 mi E Fort Bidwell, 1 (MVZ); Lake City, 1 (MVZ); 2 mi N Lake City, 1 (MVZ); 3 mi E Likeley, 1 (MVZ); Modoc National Forest, Bump Head Mtns., near Clear Lake, 1 (WFBM); Parker Creek, Warner Mountains, 3 (MVZ); 12 mi E Steele Meadow, 5200 ft, 1 (MVZ); Warren Peak, 1 (MVZ). MONO CO.: Sonora Pass, 9600 ft, 1 (MVZ). MONTEREY CO.: Arroyo Seco Wash, 2 mi S Soledad, 150 ft, 1 (MVZ); 10 mi S Carmel, by Road # 1, 1 (MVZ); Highway G16, at Cachagua Creek, 0.3 mi W divide, 1 (MVZ); Jamesburg, 1 (USNM); King City, 1 (UDAV); 2.5 mi E Monterey Municipal Airport, Highway 117, 1 (MVZ); 9 mi E Parkfield, 1 (MVZ); 7 mi E San Lucas, 1 (MVZ); Seaside, 1 (MVZ). NAPA CO.: Napa (Grinnell et al., 1937); 3 mi SW Napa, 1 (MVZ). PLUMAS CO.: Quincy, 1 (USNM). RIVERSIDE CO.: Banning, San Jacinto foothills, 2200 ft, 1 (MVZ); Cahuilla (=Anza), San Jacinto Mountains, 1 (MVZ); Indio, 1 (UCLA); 18 mi NW Palo Verde, 1 (MVZ); Pinto Basin, 1750 ft, 1 (MVZ); San Jacinto, 1 (CAS); Temecula, 2 (MVZ). SACRAMENTO CO.: Indian Mound, N of Hood, 1 (MVZ); Polk, 1 (USNM). SAN BENITO CO.: Hollister, 1 (CAS). SAN BERNARDINO CO.: Cedar Canyon, 5000 ft, Providence Mountains, 2 (MVZ); near Cimia, 3 (MVZ); Indian Cove, 3000 ft, 2 mi S, 6 mi W Twenty Nine Palms, 1 (MVZ); 2.5 mi SW Kelso, 2100 ft, 1 (MVZ); near Lake Arrowhead, San Bernardino Mountains, 1 (LACM); Reche Canyon, near Colton, 1 (MVZ); 2 mi ESE Rock Spring, 4700 ft, Lanfair Valley, 1 (MVZ); San Bernardino, 1 (SDSNH); Vidal, 1 (LACM). SAN DIEGO CO.: California Border, 1 (AMNH); Colorado Desert, Laguna Station, New River, 1 (USNM); near El Cajon, 2 (UCLA); El Cajon Valley, 1 (USNM); Escondido, 1 (SDSNH); Hillsdale, 1 (SDSNH); Lakeside (Bond, 1977); 1 mi N La Jolla, 1 (SDSNH); La Puerta Valley, 2 (CAS), 1 (MVZ), 4 (SDSNH), 1 (USNM); Mexican Boundary at monument 258, 1 (USNM); Ramona, 1 (SDSNH); W of San Marcos, 1 (SDSNH); Santa Ysabel, 1 (USNM); Sequan District, 1 (USNM); Sweetwater Reservoir, 1 (SDSNH); Twin Oaks, 1616 ft, 2 (USNM); Witch Creek, 4 (SDSNH). SAN FRANCISCO CO.: 1666 46th Ave, San Francisco, 1 (CAS); Golden Gate Park, San Francisco, 1 (CAS). SAN JOAQUIN CO.: Corral Hollow, 1 (MVZ); Tracy, 14 (USNM). SAN LUIS OBISPO CO.: 3 mi W Red Hills, near Shandon, 1 (MVZ); 12 mi S Shandon, 2 (MVZ); 14.5 mi S Shandon, 1 (MVZ); 15 mi S Shandon, 1 (MVZ); 6 mi SE Shandon, 1 (MVZ); 7 mi SE Shandon, 1 (MVZ); 8 mi SE Shandon, 1 (MVZ); 9 mi SE Shandon, 2 (MVZ); 10 mi SE Shandon, 2 (MVZ); 17 mi SE Shandon, 1 (MVZ); 18 mi SE Shandon, 2 (MVZ); 18.5 mi SE Shandon, 1 (MVZ); 19 mi SE Shandon, 1 (MVZ); 1.5 mi W Yeguas Mountain, 1 (MVZ). SAN MATEO CO.: Alpine, 1 (SDSNH); Menlo Park, 1 (CAS); near Peak Mountain, 1 (CAS); Pescadero, 1 (USNM); San Francisco Game Refuge, 2 (MVZ). SANTA BARBARA CO.: 4 mi E Cuyama Ranch, 2200 ft, 1 (USNM); 10 mi E Gaviota Pass, 1 (MVZ); Santa Anita Ranch, 7 mi W Gaviota, 2 (MVZ). SANTA CLARA CO.: San Antonio Creek, vicinity of Mountain View, 1 (CAS). SANTA CRUZ CO.: Aptos, 1 (USNM). SHASTA CO.: Burney, 1 (USNM); Dickey Ridge, near Crystal Creek, 4000 ft, 20 mi W Redding, 1 (MVZ); Fort Crook, 1 (USNM); Warner Creek, 6600 ft, 1 (MVZ). SISKIYOU CO.: Beswick, 1 (USNM); Butte Creek, 1 (CAS); 5 mi N Edgewood, 2800 ft, 1 (MVZ); Teenor, 1 (USNM). SONOMA CO.: no locality designated, 1 (CAS); Freestone, 1 (MVZ). STANISLAUS CO.: 2 mi S La Grange, 1 (CSCS). TEHEMA CO.: 10 mi NW Bluff, 1 (CSUC); South Yolla Bolly Mt., 1 (USNM); 2 mi S South Yolla Bolly Mt., 1 (MVZ). TRINITY CO.: on Highway 299, near Big Bar, T5N, R8E, sec. 22, 1 (HSU). TULARE CO.: no locality designated, 1 (USNM); Alila [=

susceptible.

Habitat: Nothing specific has been recorded of the habitat of Channel Islands Spotted Skunks. According to von Bloeker (1967), they were formerly common around human dwellings, particularly on Santa Cruz Island, but by 1967 they were rarely found near any man-made structures on either island.

In much of their range, Western Spotted Skunks (Spilogale gracilis) are common only in areas of rock outcrops such as hillsides and rocky canyons. Spotted Skunks often take up residence in or under buildings, where these occur. They seem generally to avoid flat expanses, cultivated fields, and grasslands where rocks or brush are unavailable for cover and dens. They sometimes use burrows of other animals for dens (Grinnell et al., 1937).

Spotted Skunks appear to be more tolerant of xeric environments than Striped Skunks, and may be found far from water. Food consists of a great variety of small animals -- arachnids, insects, mice and rats, lizards, snakes, eggs, and birds -- as well as fruits and seeds. Insects appear to be the staple, at least during months when they are abundant.

Recommendations: A survey of the islands to determine present numbers of spotted skunks and provide a data base for future monitoring is encouraged. If populations are found to be low, attempts to determine cause of the declines should be made.

Remarks: Von Bloeker (1967) did not indicate the disposition of the nine specimens mentioned in his paper. Some or all of these are probably included in the museum records listed below.

Channel Island Spotted Skunks are larger and with more black coloration (less white) than their mainland relatives. The status of this subspecies is not disputed in the literature. Walker (1980) reported on a subfossil specimen from a midden on San Miguel Island. He referred to unpublished field notes by C. D. Voy (1893), in which Voy related the capture of a small Skunk in a trap on San Miguel Island some years prior to his visit.

Distribution Records: SANTA BARBARA CO.: Santa Cruz Island, 9 (von Bloeker, 1967); Santa Cruz Island, Prisoners Harbor, 2 (MVZ); Santa Cruz Island, Stanton Ranch headquarters, 1 (MVZ); San Miguel Island (Walker, 1980); Santa Rosa Island, Becker's Bay, 16 (LACM); Santa Rosa Island, 2.5 mi N ranch house near coast (Dickey, 1929); Santa Rosa Island, Skunk Point, 1 (LACM).

OTHER CANDIDATES

The species listed in Table 5 were on the original working list or were suggested as candidate Species of Special Concern by others, species listed by the International Union for the Conservation of Nature, or designated as Sensitive by the U.S. Forest Service or the U.S. Bureau of Land Management. None are thought to face significant threats at this time, although lack of information is sufficient reason to consider some as sensitive species. Researchers and land and resource managers should endeavor to gather needed information on distribution and population status for those species indicated as sensitive.

Table 5. List of other candidate species not included on the List of Concern. Page number refers to starting page of species account. Table continues on next page.

Species	Page
Mt. Lyell Shrew (*Sorex lyelli*)	74
San Bernardino Dusky Shrew (*Sorex monticolus parvidens*)	74
Inyo Shrew (*Sorex tenellus*)	75
Monterey Vagrant Shrew (*Sorex vagrans paludivagus*)	75
Salinas Ornate Shrew (*Sorex ornatus salarius*)	75
Angel Island Mole (*Scapanus latimanus insularis*)	75
Alameda Island Mole (*Scapanus latimanus parvus*)	76
Long-tongued Bat (*Choeronycteris mexicana*)	76
Mammoth Little Brown Myotis (*Myotis lucifugus relictus*)	76
San Joaquin Myotis (*Myotis yumanensis oxalis*)	76
Spotted Bat (*Euderma maculatum*)	77
Sierra Nevada Mountain Beaver (*Aplodontia rufa californica*)	77
Kingston Mountain Chipmunk (*Tamias panamintinus acrus*)	77
Mt. Pinos Chipmunk (*Tamias speciosus callipeplus*)	78
Santa Catalina Ground Squirrel (*Spermophilus beecheyi nesioticus*)	78
San Bernardino Golden-mantled Ground Squirrel (*Spermophilus lateralis bernardinus*)	78
Palm Springs Round-tailed Ground Squirrel (*Spermophilus tereticaudus chlorus*)	79
Townsend Soft-haired Ground Squirrel (*Spermophilus townsendii mollis*)	79
Rock Squirrel (*Spermophilus variegatus grammurus*)	79
San Bernardino Flying Squirrel (*Glaucomys sabrinus californicus*)	80
Buena Vista Lake Pocket Gopher (*Thomomys bottae ingens*)	80
Honey Lake Pocket Gopher (*Thomomys townsendii relictus*)	80
San Joaquin Pocket Mouse (*Perognathus inornatus inornatus*)	81
McKittrick Pocket Mouse (*Perognathus inornatus neglectus*)	81
Arroyo Seco Pocket Mouse (*Perognathus inornatus sillimani*)	81
Yellow-eared Pocket Mouse (*Perognathus parvus xanthonotus*)	82
Sierra Valley Kangaroo Mouse (*Microdipodops megacephalus californicus*)	82
Pale Kangaroo Mouse (*Microdipodops pallidus pallidus*)	82
Point Conception Kangaroo Rat (*Dipodomys agilis fuscus*)	82
Lesser California Kangaroo rat (*Dipodomys californicus eximius*)	83
Big-eared Kangaroo Rat (*Dipodomys elephantinus*)	83
Berkeley Kangaroo Rat (*Dipodomys heermanni berkeleyensis*)	83

Table 5 (continued). List of other candidate species not included on the List of Concern. Page number refers to starting page of species account.

Species	Page #
Merced Kangaroo Rat (Dipodomys heermanni dixoni)	84
Argus Mountain Kangaroo Rat (Dipodomys panamintinus argusensis)	84
Panamint Kangaroo Rat (Dipodomys panamintinus panamintinus)	84
Santa Cruz Kangaroo Rat (Dipodomys venustus venustus)	85
Sonora Beaver (Castor canadensis repentinus)	85
Golden Beaver (Castor canadensis subauratus)	86
Santa Catalina Harvest Mouse (Reithrodontomys megalotis catalinae)	87
Salinas Harvest Mouse (Reithrodontomys megalotis distichlis)	87
Santa Cruz Harvest Mouse (Reithrodontomys megalotis santacruzae)	87
Anacapa Island Deer Mouse (Peromyscus maniculatus anacapae)	87
Santa Catalina Deer Mouse (Peromyscus maniculatus catalinae)	88
Western Cotton Rat (Sigmodon hispidus eremicus)	88
Colorado Valley Woodrat (Neotoma albigula venusta)	88
Monterey Vole (Microtus californicus halophilus)	88
Mohave River Vole (Microtus californicus mohavensis)	89
San Pablo Vole (Microtus californicus sanpabloensis)	89
South Coast Marsh Vole (Microtus californicus stephensi)	89
Owens Valley Vole (Microtus californicus vallicola)	89
San Bernardino Vole (Microtus longicaudus bernardinus)	90
Ringtail (Bassariscus astutus)	90
Humboldt Marten (Martes americana humboldtensis)	90
Inyo Long-tailed Weasel (Mustela frenata inyoensis)	91
Pallid Bobcat (Felis rufus pallescens)	91
Northwestern White-tailed Deer (Odocoileus virginianus ochrourus)	92

Mt. Lyell Shrew

Sorex lyelli was generally thought to live only in montane communities at high elevations in the central Sierra Nevada (Ingles, 1965). Review of accounts of capture of four of the five known specimens (Grinnell and Storer, 1924; Howell, 1924), together with information from recent captures, suggests that S. lyelli is widespread in high montane and cold steppe communities of the central and eastern slopes of the Sierra Nevada (Williams, 1984). Furthermore, S. lyelli is probably conspecific with S. preblei, ranging in similar habitats northward to Washington and eastward at least to northern Utah, Wyoming, and Montana (Williams, 1984, and unpubl. data). They face no threat of extinction.

San Bernardino Dusky Shrew

Published information on the distribution of Sorex monticolus parvidens included records of specimens from a desert spring at about 4200 ft, upward through mixed-conifer forest at about 7500 ft in elevation in the San Bernardino and San Gabriel mountains (Findley, 1955; Hennings and Hoffmann, 1977; Jackson, 1928). Grinnell (1908) captured only Ornate Shrews in the San Bernardino Mountains, including some from the type locality of S. m. parvidens (described by Jackson in 1921). Steve Clifton and I conducted a field study to determine the population status of the San Bernardino Dusky Shrew, trapping at 51 sites in the San Gabriel and

San Bernardino mountains (Williams, 1983). Although several shrews were captured from localities between 6000 and 9000 ft, no Dusky Shrews were found. Comparison of the shrews captured by us and San Bernardino Dusky Shrews identified by Hennings and Hoffmann (1977) with reference series of S. monticolus and S. ornatus showed that S. m. parvidens was indistinguishable from S. ornatus ornatus. Sorex monticolus parvidens is, therefore, considered a junior synonym of S. o. ornatus; these populations are not threatened (Williams, 1983).

Inyo Shrew

At the time this project was initiated, only three specimens of Sorex tenellus were known from two localities in California (Hoffmann and Owen, 1980). Their probable distribution, on the eastern slopes of the southern Sierra Nevada and in the White Mountains, suggested no cause for concern. Subsequently, discovery of Inyo Shrews in the Sweetwater Mountains (Williams, 1984) and the Mono Basin (Harris, 1982), and at a site on the western slope of the Sierra Nevada in Red Fir forest (Williams, 1984), shows that they are fairly widespread and under no threats.

Monterey Vagrant Shrew

Sorex vagrans paludivagus lives in riparian and tidal and freshwater wetlands of the San Francisco Peninsula, the Salinas River Delta, and adjacent lowlands in the Monterey Bay area (Findley, 1955; Hennings and Hoffmann, 1977). Only scanty information is available to document distribution and populations status. Although the area within the geographic range of Monrerey Vagrant Shrews is under intense pressure ftom human developments, several wetland communities, including the type locality, Elkhorn Slough, are protected from development. S. v. paludivagus may be more vulnerable than many wetland species, particularly birds and Long-toed Salamanders which have received special management considerations in different wetlands within the region. Therefore, it should be considered sensitive.

Salinas Ornate Shrew

Sorex ornatus salarius occupies a variety of riparian, wetland, and upland terrestrial communities in the vicinity of the Salinas River Delta (Owen and Hoffmann, 1983; unpubl. data). Although the region is undergoing intense development, I could not find information documenting its current status. The relatively wide range of communities providing habitat for S. o. salarius suggests that it is probably not jeopardized.

Angel Island Mole

Scapanus latimanus insularis is confined to Angel Island, Marin Co., in the northern portion of San Francisco Bay (Hall, 1981). The island is a state park and under no threat of intensive development. Introduced Mule Deer (Odocoileus hemionus) serious damaged vegetation on the island, resulting in considerable soil erosion, which was probably the major threat to moles. The California Department of Parks and Recreation has instituted measures to control the size of the deer population. Determining the current population status of Angel Island Moles should receive high priority in management of the island.

Alameda Island Mole

Scapanus latimanus parvus is known only from Alameda Island, Alameda, Alameda Co. (Hall, 1981). The island is intensely developed with the Alameda Naval Air Station (Nimitz Field) occupying nearly half; most of the remaining area is occupied by commercial and residential developments. Alameda Memorial State Beach Park is the most extensive state-managed property on the island. I have no current information on the population status of S. l. parvus, but development of Alameda Island warrants treating it as a sensitive species.

Long-tongued Bat

Choeronycteris mexicana seems to be a sporadic visitor to California from Mexico, appearing in San Diego County in autumn (Huey, 1954b; Olson, 1947). Records include bats taken in San Diego in September, 1946 (Olson, 1947), October and December, 1947 (Huey, 1954b), and October, 1963 (Banks and Parrish, 1965). Long-tongued Bats feed mostly on nectar and pollen of night-blooming succulents such as agave. They roost in relatively well-lighted caves and in buildings, and under shallow, cave-like overhangs and portals on buildings. No protection or management of Long-tongued Bats is required in California. Additional information on their occurrence within the state is needed, however, and this need should be brought to the attention of agency biologists and others likely to receive reports of bats.

Mammoth Little Brown Myotis

Myotis lucifugus relictus is known from the Long Valley and Owens Valley (Harris, 1974). In general, M. lucifugus is associated with large, permanent bodies of water in western North America. They often roost and form maternity colonies in the attics of buildings, under bridges, and other structures (Fenton and Barclay, 1980). M. l. relictus was suggested as a candidate species by the U.S. Fish and Wildlife Service (in litt.). I have not found evidence that it is jeopardized, although information on current population status is lacking. Major developments in the region (impoundment and diversion of rivers and streams) probably enhance habitat for the species.

San Joaquin Myotis

Myotis yumanensis oxalis occupies areas on the valley floor and foothills of the San Joaquin Basin and Delta, westward to at least Berkeley, Alameda Co. (Hall, 1981). This species was on the original working list. Within their range, San Joaquin Myotis are among the commonest bats, forming relatively large maternity colonies in attics of buildings, barns, and other structures (unpubl. observations). Populations of all bats appear to have diminished greatly in the last two decades in central California, and this trend will probably continue as more land is developed, more insecticides are used, and as barns and other structures used as roosts and sites for maternity colonies become less numerous. I do not believe the situation warrants high priority consideration now, but treatment as a sensitive species is advisable.

Spotted Bat

Euderma maculatum was listed as a rare species in the U.S. Bureau of Sports Fisheries and Wildlife Rare and Endangered Species List of 1968 (Watkins, 1977). Listing did not confer legal, protected status for the species. E. maculatum was also listed in the 1969 edition of the International Union for Conservation of Nature Red Data Book, and as a Category 2 species by the U.S. Fish and Wildlife Service (1982). Spotted Bats range widely in western North America from southern British Columbia (Woodsworth et al., 1981) to Mexico (Watkins, 1977). In California, records of Spotted Bats include the eastern and southern portions of California, the central Sierra Nevada and the foothills of the Sierra Nevada in the San Joaquin Valley. They probably occur throughout the state in suitable habitat. Most often, Spotted Bats have been found in arid desert and open pine forests in rough, rocky terrain (Easterla, 1971; Findley and Jones, 1965; Leonard and Fenton, 1983; Watkins, 1977; Woodsworth et al., 1981). Spotted bats appear to roost mainly in rock crevices (Leonard and Fenton, 1983; Watkins, 1977), and females appear to favor Ponderosa Pine forest as habitat during the reproductive season (Findley and Jones, 1965). I found no evidence that the Spotted Bat is threatened in California. A review of its distribution and population status by Fenton et al. (1983) found no significant threats that warranted seeking federal Threatened or Endangered Species status.

Sierra Nevada Mountain Beaver

Unpublished observations (Dale Steele) on a population of Aplodontia rufa californica, living near a freshwater seep along historic Lee Vining Creek, about 1 km from Mono Lake, provided the basis for inclusion of this population among the species on the draft List of Concern. It was the only local population of a more widely distributed taxon to be so included. Other populations of Sierra Nevada Mountain Beavers are found at numerous sites, particularly where there is a dense growth of small deciduous trees and shrubs, wet soil, and/or nearby water, and an abundance of herbaceous forbs, ferns, berry vines, etc. (Camp, 1918; Scheffer, 1929). Because of the unique habitat of the Mono Basin population in semidesert surroundings and because water-flow in Lee Vining Creek is usually diverted by the Los Angeles Metropolitan Water District, this population could be jeopardized. On this basis, the population was listed as a Category 2 species by the U.S. Fish and Wildlife Service (1982). Subsequently, additional information on the status of Mountain Beavers in the Mono Basin was published by Harris (1982). John Harris (pers. comm.) believes that Mountain Beavers living near Mono Lake are immigrants from higher elevations along the eastern slope of the Sierra Nevada, and that, periodically, some establish homes in suitable sites. In reconsidering its status in light of the observations by Harris (1982, and pers. comm.) and the recent litigation that has returned water to Lee Vining Creek (at least temporarily), there seems to be little justification for including this population on the List of Concern.

Kingston Mountain Chipmunk

Tamias panamintinus acrus occurs only in the Kingston Mountains in the desert area of northeastern San Bernardino Co. (Hall, 1981). This population is apparently isolated from all other populations of Tamias by hot, low-lying desert communities (Johnson, 1943). Within the Kingston Mountains, the chipmunks live in arid pinyon-juniper woodlands, occupying nests among rocks in fissured cliffs and ledges. The U.S. Bureau of Land Management listed T. p. acrus as a Sensitive Species in its California Desert Conservation Area Plan (unpubl. documents). Johnson (1943) noted that Kingston Mountain Chipmunks were few in number and occupied a potential geographic range of less than 40 square miles. Major uses of land in the area include cattle grazing and mining (U.S. Bureau of Land Management, in litt.); neither activity poses serious threats to the population. No evidence known to me supports considering it as sensitive.

Mt. Pinos Chipmunk

Tamias speciosus callipeplus is apparently restricted in distribution to the upper slopes and summit of Mt. Abel and Mt. Frazier of the Transverse Ranges, (southwestern Tehachapi Range). Howell (1929) recorded a single specimen from Canyon de Uvas (Grapevine Canyon). Records of elevation extend from 6000 ft on north-facing slopes to 8800 ft in open, coniferous forest (A. I. Roest, in litt.; Grinnell, 1933). Lodgepole Chipmunks (T. speciosus) are generally found in open forests with a mix of shrubs and trees. They are nearly always associated with Lodgepole Pine, although Red Fir, Jeffrey Pine, and Chinquapin are also common plant associates (Johnson, 1943); Red Fir does not occur, and White Fir is probably an important associated species within the range of T. s. callipeplus. Lodgepole Chipmunks frequently take refuge in trees and rarely venture far from tree cover. No information is available to document the status of this isolated population. Dr. Aryan Roest expressed concern about its status and stated that a high level of recreational activities by humans in that area, including extensive vacation home developments might be responsible for a perceived decline in population density. The relatively wide-spread occurrenne of Mt. Pinos Chipmunks on public lands and their preference for relatively open forests suggests that they are not currently threatened, but this should be established by field studies. Treatment as a sensitive species is recommended.

Santa Catalina Ground Squirrel

Spermophilus beecheyi nesioticus occurs only on Santa Catalina Island, Los Angeles Co. Although this species was on the original working list, primarily because of its insular distribution, it was shortly dropped from consideration. California Ground Squirrels are widely distributed, occurring from below sea level to high elevations in temperate, mixed conifer forests (Grinnell, 1933). They are most numerous in disturbed communities undergoing secondary succession. During field work on Santa Catalina Island in January, 1983 (Williams, 1983), I observed several Santa Catalina Ground Squirrels. The general degradation of native plant communities by introduced Bison, Feral Goats, Wild Pigs, and other introduced ungulates favor this species. There is no reason to treat it as sensitive.

San Bernardino Golden-mantled Ground Squirrel

Spermophilus lateralis bernardinus is a relictual population of the Golden-mantled Ground Squirrel and is distributed from about 6500 to 11500 ft in the elevation in the San Bernardino Mountains (Grinnell, 1933). In the western Sierra Nevada, Golden-mantled Ground Squirrels chiefly inhabit forest floors, especially in open forests with a mix of tall trees, brush, and open ground supporting herbaceous plants (White et al., 1980). This species was included on the original List of Concern primarily on the basis of my experiences with *S. lateralis* on the western slope of the central Sierra Nevada. During field work in the San Bernardino Mtns. (Williams, 1983, and unpubl.), Golden-mantled Ground Squirrels were observed in open pine and mixed conifer forests and in pinyon-juniper woodlands on the northern, desert slopes of the mountains. Their abundance in a broader range of plant communities in the San Bernardino Mountains lessens the importance of threats to the taxon from home and recreational developments.

Palm Springs Round-tailed Ground Squirrel

Spermophilus tereticaudus chlorus was on the draft List of Concern as a Third Priority species. Populations are restricted to areas below about 1200 ft in the Coachella Valley (Grinnell and Dixon, 1918) in Riverside Co. Here, Round-tailed Ground Squirrels occupy dry, level, sandy areas, apparently being most numerous in areas with fine sandy soils supporting abundant herbaceous growth; Mesquite (*Prosopis* sp.), *Suaeda*/*Distichlis*, and Creosote (*Larrea*) associations are also commonly used. They do not inhabit cultivated fields and lawns but they feed in such areas when the distance from their home burrows is not too great. The habitat for *S. t. chlorus* has been reduced substantially by urbanization, cultivation, and construction of roads, railroads, airports, and golf courses. Little public land is found in the Coachella Valley, and not all that is publicly owned supports this species. The reason for removing it from the List of Concern is because of the Coachella Valley Macropreserve agreement between the U.S. Department of the Interior, The Nature Conservancy and other organizations. *S. t. chlorus* should be treated as a sensitive species until information on its status is available.

Townsend Soft-haired Ground Squirrel

Spermophilus townsendii mollis occurs in two disjunct areas in eastern California: the region from the upper Owens Valley to the Mono Lake area in Mono Co.; and in the Honey Lake area in Lassen Co. (Hall, 1981). Townsend Ground Squirrels live in communities with sandy soils supporting Big Sagebrush and Rabbitbrush. Typically, they are found in loosely organized colonies. In California, they are dormant from about July to January or February (Harris, 1982). They do not live in cultivated fields. *S. t. mollis* was on the working list, but definitive information on its status has not been developed. Much of the land around Honey Lake is under cultivation and the species could be threatened there. There are fewer developments that would threaten the species in the southern sector of its range where extant colonies are known from the Mono Lake Basin (Harris, 1982; and unpubl. data). I found no evidence to support listing this species.

Rock Squirrel

In California, *Spermophilus variegatus grammurus* only occurs near the eastern border in desert mountains ranges (Clark and Providence ranges) of San Bernardino Co. (Hall, 1981). There, it lives in pinyon-juniper associations among rocks (Ingles, 1965). I found no information to indicate that the species is threatened in California.

San Bernardino Flying Squirrel

Glaucomys sabrinus californicus is known from the San Bernardino and San Jacinto ranges, living between about 5200 and 8500 ft. There probably is a population of G. sabrinus in the San Gabriel Mountains also (Vaughan, 1954). Summer (1927) conducted an extensive trapping survey south of Big Bear Lake in the San Bernardino Range in summer, 1926. He took 22 specimens over the course of several months, all in White Fir trees. Grinnell (1933) remarked that these flying squirrels inhabited woodlands where Black Oak or White Fir trees were common. Grinnell and Swarth (1913) captured a single animal in the San Jacinto Mountains. It was taken in a Black Oak at about 6000 ft. Records of 38 specimens from the two mountain ranges, most of which were taken prior to the 1960's, were located. The San Bernardino Flying Squirrel was listed as a Third Priority species on the draft List of Concern because of high densities of homes and recreation sites in mid-elevation forests in both ranges and killing of trees by air pollution, and because authorities had remarked on the seeming scarcity of these animals (Grinnell, 1908; Grinnell and Swarth, 1913; Summer, 1927). Field work in the San Gabriel and San Bernardino mountains (Williams, 1983) convinced me that the threats from developments were of much lesser magnitude than feared earlier. G. s. californicus was deleted from the final List of Concern for these reasons. All flying squirrels are protected from taking in California, therefore having some protection from direct depredation by humans.

Buena Vista Lake Pocket Gopher

Thomomys bottae ingens is limited to areas in the vicinity of historic Buena Vista Lake, in the southern San Joaquin Valley, Kern Co. (Hall, 1981). Buena Vista Lake has been drained and is under irrigated cultivation; cotton is grown on the historic Lake Bed and on most surrounding land. Periodic field work, especially in 1985 (Williams, 1985) produced evidence of extant colonies in the South Coles Levee Oil Field and the Buena Vista Lake Recreation Area north of the lake bed, and elsewhere within its limited geographic range. Given the ability of pocket gophers to adapt to pastures and fields in perennial crops such as alfalfa and to golf courses, T. b. ingens was dropped from active consideration.

Honey Lake Pocket Gopher

Thomomys townsendii relictus occurs only in a small area in the vicinity of Honey Lake in the western part of Honey Lake Valley, Lassen Co. (Hall, 1981). Its geographic range extends from near Susanville in the northwest to near Amedee in the northeast and Doyle in the south (Thaeler, 1968). Within this area, Honey Lake Pocket Gophers are confined to deep soils of lacustrine origin. They are found in moist meadows along the western edge of the Honey Lake Valley and range into the more arid grasslands in the eastern part of the valley, where their range appears to be coincident with the distribution of saltgrass (Distichlis; Thaeler (1968). Soil type may be the primary limiting factor for this gopher. Although much of the Honey Lake Valley is under cultivation or in irrigated pasture, Honey Lake

Pocket Gophers appear to be common and not threatened by developments in the region (J. L. Patton, pers. comm.).

San Joaquin Pocket Mouse

Perognathus inornatus inornatus occurs along the eastern side of the San Joaquin Valley and possibly in the Sacramento Valley (unpubl. data). The boundaries between the geographic ranges of P. i. inornatus, with 50 chromosomes, P. i. neglectus, with 56 chromosomes (Williams, 1978), and the population from the Sacramento Valley (a single specimen from Lake Co. had 60 chromosomes; unpubl. data) are not well documented. Taxonomic problems in this species complex have complicated the resolution of its populations status further. Typical San Joaquin Pocket Mice from the eastern edge of the San Joaquin valley have been found in areas with friable soils in grasslands and Blue Oak savannas, from near sea level to about 1500 ft in elevation. Some of the oldest and largest collections, made near the beginning of the century (unpubl. data) were from areas of wind-drifted sand, a habitat that has essentially disappeared due to cultivation and urbanization. This species was listed in the Third Priority on the draft List of Concern, based primarily on the extensive loss of habitat in the San Joaquin Valley. Reflection has resulted in removing it from the List of Concern, although it should be treated as a sensitive species until information is available to resolve its population and systematic status.

McKittrick Pocket Mouse

Perognathus inornatus neglectus occurs along the western side of the San Joaquin Valley, in the Mojave Desert, on the Carrizo Plain, and in the upper Salinas Valley (unpubl. data). Its distributional limits and relationships to P. i. inornatus, P. i. psammophilus, and the inornatus group pocket mice from the Sacramento Valley are unclear. McKittrick pocket mice have been taken on a variety of soil types in desert shrub associations supporting Atriplex spp., Ephedra californica, and annual herbaceous plants such as Bromus spp. and Erodium. Most of the available habitat for this pocket mouse is located on the sloping, western margin of the San Joaquin Valley and the adjacent, rugged hills. McKittrick pocket mice seem to be uncommon, or at least only locally common, but probably are not under serious threats now.

Arroyo Seco Pocket Mouse

Perognathus inornatus sillimani was described by von Bloeker (1937); he also described the Salinas pocket mouse (Perognathus longimembris psammophilus) at the same time. The type series of both taxa were collected from the same locality. I have examined all of the available specimens of both species. The specimens of P. i. psammophilus were all subadults and the specimens von Bloeker (1937) called P. i. sillimani were all adults. The two taxa are based upon different age groups of the same population, which is allied with the inornatus species complex, not P. longimembris. The Arroyo Seco Pocket Mouse is, therefore, a junior synonym of Perognathus inornatus psammophilus. The latter name has priority because of page precedence in the publication describing both taxa (see account of P. i. psammophilus).

Yellow-eared Pocket Mouse

Perognathus parvus xanthonotus is known from the southeastern slopes of the Tehachapi Mountains in the vicinities of Freeman (Walker Pass), Horse, Indian Wells, and Sage canyons in Kern Co. (unpubl. data). Yellow-eared Pocket Mice are associated primarily with arid desert shrub and Joshua Tree communities and in ecotonal areas supporting a sparse cover of Pinyon Pine trees. Altitudes recorded for specimens range from about 4000 to 5300 ft. The Yellow-eared pocket mouse was treated as a separate species from *P. parvus* by Hall (1981). Its karyotype and aspects of its structure are indistinguishable from most members of the *parvus* species group (Williams, 1978), and I see no reason to treat it as a separate species. The U.S. Bureau of Land Management designated the Yellow-eared Pocket Mouse as a Sensitive Species in the California Desert Conservation Area Plan. There are no known or projected developments that would jeopardize this taxon, despite its apparent restricted, geographic range and narrow altitudinal distribution.

Sierra Valley Kangaroo Mouse

In California, *Microdipodops megacephalus californicus* is known only from the type locality in the Sierra Valley, Plumas Co. (Hall, 1981). Although found on substrates of gravelly texture, they are most often associated with sandy soils. The restricted distribution of *M. m. californicus* was the primary reason for including it on the working list. There is no information on its current status.

Pale Kangaroo Mouse

Microdipodops pallidus pallidus occurs in California as two disjunct populations: one in the Fish Lake Valley, near Oasis, Mono Co., and one in the Deep Springs Valley, Inyo Co. Inhabited areas in California are steppe-desert associations on fine, sandy soils from about 5000 to 5500 ft in elevation (Grinnell, 1933). I found no information on the current population status of *M. p. pallidus*. Most of the land supporting both population is either privately owned or is public land outside specified multiple-use classes of the U.S. Bureau of Land Management California Desert Conservation Area Plan. Future developments in these areas would pose threats to the only known populations of *M. pallidus* in California, although I am not aware of any planned developments. During a one-day visit to the area in 1982, I found no evidence that the populations are under serious threats.

Point Conception Kangaroo Rat

Dipodomys agilis fuscus was placed on the working list because of its apparently restricted distribution. It is known only from the vicinity of Point Conception, Santa Barbara Co., in coastal chaparral communities (Boulware, 1943). Best (1983) could find no characteristics separating the more widely distributed subspecies, *D. a. perplexus*, from *fuscus*. I believe detailed studies will show the populations are in contact and do not warrant taxonomic recognition as separate subspecies. This is the primary reason for not ranking this kangaroo rat in a priority category.

Lesser California Kangaroo Rat

Dipodomys californicus eximius is found only in the vicinity of Marysville Buttes in Sutter Co. According to Grinnell and Linsdale (1929), *eximius* differs from the wider-ranging subspecies, *D. c. saxitilis*, only in its paler coloration. Neither subspecies was numerous at the time Grinnell and Linsdale (1929) investigated their status. *D. c. saxitilis* apparently was limited to foothill communities just above the edge of the valley floor, in sites on slopes with soils that remained "well-drained through the wet winter months" (Grinnell and Linsdale, 1929). Presumably, the Lesser California Kangaroo Rat has similar requirements. According to Blair Csuti (pers. comm.), *eximius* had not been collected for several years. It was included on the working list, but was dropped after review of information on its taxonomic status suggested that there was little justification for recognizing the wider-ranging *D. c. saxitilis* as a subspecies separate from *eximius*. Current information on its population status and a review of the taxonomy of *D. californicus* is needed. The Lesser California Kangaroo Rat should be considered vulnerable and treated as a sensitive species until its status is determined.

Big-eared Kangaroo Rat

Published information on the distribution of *Dipodomys elephantinus* suggests that its range is limited to the southern portion of the Gabilan Mountains in San Benito Co., in the vicinity of Pinnacles National Monument (Grinnell, 1922; Hall, 1981). The Big-eared Kangaroo Rat was included on the working list because of its small geographic range and because it was listed by the International Union for Conservation of Nature *Red Data Book*. Also, it was listed as a Category 2 species by the U.S. Department of the Interior (1982). I collected kangaroo rats with diagnostic characteristics of *D. elephantinus* in chaparral communities in Del Puerto Canyon in western Stanislaus Co., and from Clear Creek in the southern Diablo Range, San Benito Co. An additional specimen in the Collections at Fresno State University, from near the border of Fresno and San Benito counties is of this species. I found no justification for recognizing *elephantinus* as a species separate from *D. venustus* (Narrow-faced Kangaroo Rat) or to be concerned about its population status. Troy Best (pers. comm.) is presently studying the relationships between *D. venustus* and *elephantinus*.

Berkeley Kangaroo Rat

Published records of occurrence of *Dipodomys heermanni berkeleyensis* are from the Berkeley Hills, Mount Diablo, and the Livermore Valley (Hall, 1981). Berkeley Kangaroo Rats probably occupy suitable habitat throughout the mountain ranges east of San Francisco Bay and west of the San Joaquin Valley. Habitat for Berkeley Kangaroo Rats, in so far as is known, consists of open, grassy hilltops (Grinnell, 1933) and open spaces in chaparral and Blue Oak/Digger Pine woodlands (unpubl. data). Habitat for Berkeley Kangaroo Rats has been reduced significantly in the hills and valleys north of Livermore, and this gives some cause for concern about its status. However, *D. heermanni* with characteristics attributed to *D. h. berkeleyensis* (Grinnell, 1922) are fairly common on the east flank of the Diablo Range in Del Puerto Canyon, Stanislaus Co., and Corral Hollow, Alameda Co. (unpubl. data). The limits of the geographic range of *D. h.*

berkeleyensis and its relationships to D. h. tularensis (Tulare Kangaroo Rat) need to be reviewed in order to clarify their distributional and population status.

Merced Kangaroo Rat

Dipodomys heermanni dixoni is known from eastern Merced and Stanislaus counties (Hall, 1981) in grassland and savanna communities. It was included on the working list because of its limited distribution and the rapid rate of conversion of native plant communities to irrigated fields in this portion of the San Joaquin Valley. Thousands of acres of land in eastern Stanislaus and Merced counties, considered safe from cultivation six years ago has since been converted to orchards and vineyards. No additional information on its population status has been located, but threats to its population from continuing loss of habitat seem great enough to warrant treating it as sensitive until its status is clarified.

Argus Mountain Kangaroo Rat

Dipodomys panamintinus argusensis occurs only in the Argus Mountain Range in Inyo County (Hall, 1981). Its population is isolated from others of D. panamintinus. Nothing specific is recorded of its habitat associations, but, in general, Panamint Kangaroo Rats occur in arid, mountain steppe communities vegetated with Yucca, Pinyon Pine, Juniper, and Big Sagebrush. They appear to be most common on coarse-textured soils on sloping ground (Johnson et al., 1948). This taxon was included on the working list because it was designated as a Sensitive Species in the U.S. Bureau of Land Management California Desert Conservation Area Plan. A portion of the Argus Mountains lies within the China Lake Naval Weapons Center. Nearly all of the rest of the Argus Range is public land administered by the U.S. Bureau of Land Management This land is designated as Class L (limited use), which means use is limited to livestock grazing, mining, and other activities that are unlikely to jeopardize the population of kangaroo rats. There is no known reason for granting special consideration to this population.

Panamint Kangaroo Rat

Dipodomys panamintinus panamintinus occupies the Panamint Range in Inyo Co., between about 4600 and 7000 ft. Distributional records include several specimens from around Jackass Spring, the head of Willow Creek, 1 mi s. of Lee Pump, and South Park Meadow (Johnson et al., 1948; Hall, 1981). Habitat for D. p. panamintinus is probably similar to other populations of this species (see above). It was included on the working list because it was designated as a Sensitive Species in the U.S. Bureau of Land Management California Desert Conservation Area Plan. Panamint Kangaroo Rats were fairly common in Pinyon-Juniper communities around Jackass Spring during field work in 1974 (unpubl. data). Most of the Panamint Mountains lie within Death Valley National Monument and are inaccessible to off-road vehicles and protected from most forms of development. Other portions of the geographic range of the Panamint Kangaroo Rat are public lands designated for limited use (Class L) by the U.S. Bureau of Land Management. There is no known evidence of threats to the Panamint Kangaroo Rat.

Santa Cruz Kangaroo Rat

The geographic range of Dipodomys venustus venustus extends from the Santa Cruz Mountains eastward to Mount Hamilton in the Diablo Range, and southward to Freemont Peak in the northern end of the Gabilan Range (Grinnell, 1922; Hall, 1981). Kangaroo rats of this group are also known from the Diablo Range in western Stanislaus Co., and from the southern Diablo Range in San Benito and Fresno counties (unpubl. data; see account of the Big-eared Kangaroo Rat). Some populations of Dipodomys venustus are found on slopes with a heavy cover of chaparral (Grinnell, 1933). Hawbecker (1940) found burrows of Santa Cruz Kangaroo Rats on sand, sandy loam, and loam soils, but never in soils of "heavy" texture. He trapped Santa Cruz Kangaroo Rats in two abandoned agricultural fields which had supported chaparral originally (Chamise, Black Sage, Manzanita, Buck Brush, and Coyote Brush) and conifer forest (Coast Redwood, Douglas Fir, Madrone, and Tanoak). He speculated that they also inhabited the areas with chaparral, as he trapped Santa Cruz Kangaroo Rats at the edge of chaparral. Animals I have captured in the Diablo Range were taken in chaparral (unpubl. data). In the Santa Cruz Mountains, known collecting localities have sandy soils. Available evidence suggests a severe decline in populations of Santa Cruz Kangaroo Rats in the Santa Cruz Mountains. Michael Marangio (in litt.) reported that his searches at several sites failed to uncover evidence of extant colonies. Increasing density of residential housing and the domestic and feral house cats may be factors in extirpating some populations. Quarrying of sand has probably led to extirpation of others (M. Marangio, in litt.). I have not included the Santa Cruz Kangaroo Rat on the List of Concern because I could find no threats to populations in most portions of the Diablo Range; however, measures to locate and protect populations in the Santa Cruz Mountains should be considered.

Sonora Beaver

Castor canadensis repentinus was originally found along the Colorado River in California. In 1911, upon completion of construction of irrigation and drainage canals between the Colorado River and the Imperial Valley, Sonora Beavers colonized portions of the Imperial Valley (Grinnell et al., 1937). They inhabit slow- to moderate-flowing warers of the main channels of the Colorado River and canals, sloughs, and oxbow lakes. Where the water is shallow and the banks are of rock, lodges are constructed. Elsewhere, however, beavers place nests in burrows in banks and levees. Dams are constructed where waters are shallow, but are uncommon along the Colorado River (Tappe, 1942). Sonora Beavers eat bark and twigs of Willows, Mesquite, Cottonwood, and other woody plants, and Cattails, grasses, Arrow Weed, and other herbaceous plants. The Sonora Beaver was included on the draft List of Concern in a Fourth Priority category for sensitive species, and is currently designated as a furbearer and may be trapped in California. Dixon (1922) estimated that 100 Sonora Beavers lived in the Imperial Valley in 1921. Trapping and water shortage caused a subsequent decrease in the population, and by 1940, Tappe (1942) estimated that there were only 32 left in the Imperial Valley, mostly living along the lower portion of the Alamo River. Tappe found a total of 13 Sonora Beavers in canals and sloughs in the Palo Verde Valley and 272 along the Colorado River (129 on the California side). The present status of the Sonora Beaver population is unknown. Loss of riparian habitat along the Colorado River has been extensive since the 1940's. River channelization, phreatophyte control, and lining of canals and drains with concrete are probably the most troublesome factors affecting Sonora Beavers. No Sonora Beavers were taken by animal damage-control agents of the U.S. Fish and Wildlife Service in Imperial, Riverside, or San Bernardino counties between 1969 and 1976 (Lee, 1977). A single beaver was reported as taken by a fur trapper in Imperial Co. during the 1975-76 season (Lee, 1977). Periodic contamination of the Alamo River in the Imperial Valley by raw sewage from Mexicali, Mexico, may threaten any remaining Sonora Bevers living there. The Sonora Beaver should be consider as a sensitive species and its status determined.

Golden Beaver

Castor canadensis subauratus was originally found throughout the lower courses of the San Joaquin and Sacramento river systems, including the streams draining into Tulare and Buena Vista lakes. Altitudinal range was from sea level up to about 1000 ft. Golden Beavers probably did not occupy the upper segments of streams of the western Sierra Nevada (Grinnell, 1933), although they have been introduced there at numerous sites (in the 1940's alone, there were 172 different transplants of Golden Beavers). Golden Beavers inhabit slow- to moderate-flowing streams, ponds, and lakes. The main requirement seems to be sufficient food, which consists of roots, bulbs, grasses, Cattails, and other herbaceous plants, and bark and twigs of Willows, Cottonwoods, Alders, and other woody plants (Grinnell et al., 1937). Golden Beavers usually dig burrows in banks of streams and levees. and less frequently construct dams and lodges. The Golden Beaver was included on the draft List of Concern as a sensitive species (Fourth Priority group). Golden Beavers were nearly exterminated by unregulated harvest prior to 1911, when legislation was passed in California to protect Beavers fully. The populations responded to protection by increasing to densities to where trapping was allowed again in 1925. Controls were ineffective against illegal trapping and full protection was extended again in 1933. In 1957, beavers were classified as a furbearer and a season for harvest was designated. No bag limit on the number taken has been in effect since 1957 (Tappe, 1942; Lee, 1977). The history of the harvest of beavers in California suggests that most populations have been fairly stable over the 25 to 30 years prior to 1977 (Lee, 1977). Also apparent from fur trappers' reports is an upward trend in numbers of persons purchasing trapping licenses in the past decade and the increasing prices being paid for furs (California Dept. Fish and Game, unpubl. Licensed Fur Trappers Report, 1976-77). Pelts of beavers did not increase in average price paid, however. In any case, beavers are highly vulnerable to trapping and any market trends or socio-economic factors causing increased trapping pressure could result in a rapid population decline. Alteration of aquatic habitats, including decreased stream flow, increased pollution, channelization of streams, stream-side brush clearing, and regulation of stream flow, also could effect beaver populations adversely. The Golden Beaver should be treated as a sensitive species. Close monitoring of populations of Golden Beavers, including regular field surveys, is recommended. Decisions on trapping regulations should not be made solely on data from reports of licensed trappers, as many trappers fail to report their take (44% of the licensees in 1976-77 did not file reports), and there is no assurance that data based upon such reports are accurate.

Santa Catalina Harvest Mouse

Reithrodontomys megalotis catalinae is found only on Santa Catalina Island, Los Angeles Co. It was included on the original working List of Concern in the Third Priority category because of its insular distribution and because no information was available on its status or habitat requirements (Howell, 1914; von Bloeker, 1967). During field work on Santa Catalina Island during January, 1983 (Williams, 1983), Santa Catalina Harvest Mice were found in all plant communities and were fairly common. There are no known threats to R. m. catalinae.

Salinas Harvest Mouse

Reithrodontomys megalotis distichlis occurs in the region of Monterey Bay (Hall, 1981), in fresh and brackish water wetlands and probably in adjacent upland grasslands. It was included on the working list because of its restricted distribution and the high rate of urbanization in that area. Most of the wetland communities where these mice live are under protection. There is no evidence that they are threatened.

Santa Cruz Harvest Mouse

Santa Cruz Island, Santa Barbara Co., is home to Reithrodontomys megalotis santacruzae (Pearson, 1951). Pearson (1951) noted that Santa Cruz Harvest Mice were found only in a small, grassy area adjacent to a small, fresh-water marsh of Scirpus, Typha, and Salix. He remarked that this habitat was rare, perhaps not found elsewhere on the island, and that Santa Cruz Harvest Mice were probably limited to that one spot on the island. R. m. santacruzae was included on the draft List of Concern in the Highest Priority because of its apparent restricted habitat, Pearson's (1951) belief that the total insular population was very small, and the damage to plant communities on the island by introduced ungulates. Laughrin (1973) Pearson (1951), and von Bloeker (1967) collectively listed Cattle, Pigs, Sheep, Wapiti (Cervus elaphus), Mule Deer, and Roe Deer (Capreolus pyargus = C. capreolus) as introduced mammals to the island. R. m. santacruzae was removed from the final List of Concern because field studies by Bills (1969) found that Santa Cruz Island Harvest Mice were widely distributed on the island, and because the island has recently come under the management of The Nature Conservancy. The current population status of the Santa Cruz Harvest mouse is unknown, but the balance of available information suggests no reason for including it on the final List of Concern. Studies to determine its population status and the nature of any existing threats should be undertaken.

Anacapa Island Deer Mouse

Peromyscus maniculatus anacapae is found on East, Middle, and West Anacapa islands, Ventura Co. (von Bloeker, 1942). Anacapa deer mice live in all terrestrial habitats on the islands and also use the intertidal zone for foraging (Grinnell, 1933; P. W. Collins, in litt.). P. m. anacapae was included on the draft List of Concern as a sensitive species (Priority Four) on the basis of evidence furnished by Paul W. Collins (in litt.). According to Collins (in litt.; unpubl. U.S. Natl. Park Serv. Rept.), Anacapa Deer Mice were restricted to grassland habitats on the islands because of competition with black rats (Rattus rattus). He estimated that populations were eliminated from 75% of the available habitat. The population on East Anacapa Island was affected most severely. The U.S. National Park Service (Dept. of the Interior) has administrative authority for the Anacapa Islands, which are part of the Channel Islands National Monument. The Anacapa Deer Mouse was not included in a higher-priority category because the National Park Service is aware of the situation and has presumably taken measures to reduce or eliminate black rats from the islands. P. m. anacapae should be treated as a sensitive species and its population status should be monitored.

Santa Catalina Deer Mouse

Peromyscus maniculatus catalinae is confined to Santa Catalina Island, Los Angeles Co., where it inhabits most terrestrial plant communities (von Bloeker, 1967). It was placed on the working list because of its insular distribution, the general degradation of the island by introduced ungulates, possible predation by feral cats which occur in high density on the island (Laughrin, 1973), and possible competition with introduced black rats and house mice (Mus musculus; von Bloeker, 1967). It was not assigned to a priority category on the final List of Concern because during field work on the island in January, 1983, Santa Catalina Deer Mice were found to be fairly common and under no threats (Williams, 1983).

Western Cotton Rat

According to Hall (1981), Sigmodon hispidus eremicus is confined to the region along the lower Colorado River in California. Western Cotton Rats also occur in the Imperial Valley, having immigrated there shortly after completion of the canal from the Colorado River to the Imperial Valley (Dixon, 1922). Western Cotton Rats inhabit wetlands and upland habitats with dense grass and other herbaceous plants. In times of population irruptions, they may be found far from wetlands in a variety of plant communities atypical of their usual habitat (unpubl. data). S. h. eremicus was placed on the working list because of its restricted distribution in an area known to have been significantly altered by human developments. Brad Blood (in litt.) found Colorado River Cotton Rats to be fairly common during field studies in 1979 and 1980. Although not widespread, they are probably not jeopardized.

Colorado Valley Woodrat

In California, Neotoma albigula venusta is found in the low-lying desert areas of Imperial, San Diego, and Riverside counties (Grinnell, 1933; Hall, 1981). It is closely associated with patches of Beaver-tail Cacti (Opuntia spp.) and Mesquite. N. a. venusta was included on the working list, but no evidence indicating that it was threatened was found.

Monterey Vole

According to von Bloeker (1937), who named Microtus californicus halophilus, it is confined to saltwater wetlands around Moss Landing and Seaside, Monterey Co. Although not placed on the original working list, the status of this vole has been questioned since. My impression is that California Voles (including the Monterey Vole) are common in both marshes and in adjoining upland grasslands, vineyards, and weedy fields. They appear to range widely, especially in the winter months. Although Monterey Voles are recorded from only a restricted area, They are probably more widely distributed. Most wetlands within its range are protected.

Mohave River Vole

Microtus californicus mohavensis is known to occur only along the Mohave River in the vicinity of Victorville and Oro Grande, San Bernardino Co. (Grinnell, 1933). Although on the working list, no information on recent occurrence was found. The desert region in the vicinity of Victorville is undergoing a human population explosion that could threaten many native species. A brief visit to the area in 1983 did not provide time to search for evidence of voles, but habitat suitable for this species (weedy herbaceous growth in wet areas along the river) was noted at several sites between Hesperia and Victorville. Also, some irrigated land in pasture and alfalfa could provide additional habitat for the Mojave River Vole. M. c. mohavensis should be treated as a sensitive species until its population status is determined.

San Pablo Vole

Microtus californicus sanpabloensis is known from the salt marshes of San Pablo Creek, Contra Costa Co., on the south shore of San Pablo Bay (Hall, 1981). M. c. sanpabloensis was not included on the original working list but is included here because of subsequent questions about its status. The decision not to give this and other populations of California Voles with restricted distributions greater consideration was based upon the fact that California Voles live in a wide variety of grassland associations, especially in the wet months when there is an abundance of green, herbaceous plants. Extirpation of a small population at one site is likely to soon be offset by immigration from populations occupying other sites nearby. Field work in San Pablo and Suisun Bays on the opposite shore from where the San Pablo Vole lives, found California Voles to have withstood episodes of record flooding during the winter of 1982-83 (Williams, 1983). Foreman (fide T. Rado, pers. comm.) determined the distribution and status of M. c. sanpabloensis during 1985. On the basis of his findings, priority listing is not warranted.

South Coast Marsh Vole

Populations of Microtus californicus stephensi are recorded from tidal marshes at Point Mugo, Orange Co., and Playa del Rey and Sunset Beach, Los Angeles Co. (Hall, 1981). M. c. stephensi was not included on the original working list for the reasons stated for M. c. sanpabloensis. No information on the current status of M. c. stephensi is available. Human developments in the region may have more severely restricted voles to the extant marshes, and catastrophic episodes of flooding or epidemics may pose a greater threat of extinction to this subspecies than to California Voles elsewhere. M. c. stephensi should be treated as a sensitive species until its status is determined and threats to its population identified.

Owens Valley Vole

Microtus californicus vallicola has been recorded from several sites in the Owens Valley, Inyo Co., from near the head of Willow Creek in the

Panamint Mountains, Inyo Co., and from near Benton, Mono Co. According to Grinnell (1933), Owens Valley Voles inhabit wetlands and "lushly grassy ground." M. c. vallicola was placed on the original working list, but soon was dropped from further consideration. M. c. vallicola was listed as a Category 2 species by the U.S. Dept. of Interior (1982). Even though water diverted from the Owens Valley to the Los Angeles Basin has had significant environmental impact within the range of this subspecies, considerable habitat, including irrigated pastures and alfalfa fields still remain. There is no evidence to support listing this species in a priority category or as sensitive.

San Bernardino Vole

Microtus longicaudus bernardinus has been recorded from the higher elevations, above about 7500 ft, in the San Bernardino Mountains (Hall, 1981). According to Grinnell (1933), it inhabits streamsides and wet mountain meadows. M. l. bernardinus was included on the original working list but was dropped later. Subsequently, in summer of 1982, extensive trapping for shrews, conducted by Steve Clifton and me, produced a number of voles from the San Bernardino Mountains. None were found that could be distinguished from M. californicus, a wide-spread species at lower elevations in southern California. In my opinion, M. l. bernardinus is probably a subspecies of californicus, but the taxonomic studies are incomplete. Regardless, these voles are not threatened.

Ringtail

Bassariscus astutus is found throughout most of California, from below sea level to at least 8000 ft in the Sierra Nevada (Hall, 1981). Its principal habitat requirements seem to be den sites among boulders or in hollows in trees and sufficient food in the form of rodents and other small animals. I included two of the four subspecies occurring in California (B. a. octavus, B. a. raptor) on the working list. The status of a third subspecies (B. a. willetti) was also questioned since development of the draft report. Certainly, urbanization in the Southern California coastal basins and in the San Francisco Bay area, and loss and degradation of riparian communities throughout California have depleted and extirpated some populations, but there is no evidence of threats to any of the subspecies over a broad area. Ringtails are protected from taking by state regulations.

Humboldt Marten

Martes americana humboldtensis occurs in California in the coastal conifer forests from Sonoma Co., northward into Oregon. The altitudinal range is from near sea level to over 4000 ft (Grinnell et al., 1937; unpubl. data). Grinnell et al. (1937) remarked that Humboldt Martens were sparsely distributed within this region, although the evidence they reviewed suggested that it was more widely distributed and numerous at an earlier time. According to their report, Humboldt Martens mainly used ridgetops in spruce forests and were rare in redwood forest. Trappers they interviewed believed that squirrels (Tamiasciurus douglasii), mice, and birds were the major prey of Humboldt Martens. Information on the status of the Humboldt Marten was provided by Schempf and White (1977). Although there is no evidence that it is currently threatened, a number of studies summarized by Strickland et al. (1982) have demonstrated that clear-cutting of forests is incompatible with Martens, whereas low-intensity fires and selective logging have less impact. Mature forest is essential habitat for Humboldt Martens, providing nesting sites in snags, hunting sites, overhead cover from predators, and preferred prey species. Martens are easily trapped and overharvesting is always a potential problem where trapping pressures are high. Even a short trapping period, early in the season, may not solve the problem of overharvest. Public agencies administering lands and resources within the range of the Humboldt Marten should treat it as a sensitive species requiring special consideration in land and resource management decisions.

Inyo Long-tailed Weasel

Mustela frenata inyoensis has been recorded from a few localities in the Owens Valley, in the vicinity of Lone Pine, Independence, and Alvord, Inyo Co. (Hall, 1981). Long-tailed Weasels occur throughout all of California except the southeastern deserts, south and east of Owens Valley. They are common wherever their favored prey, pocket gophers, voles and other rodents are plentiful, even in suburban areas where sufficient open space is available (unpubl. data). The Inyo Long-tailed Weasel was not included on the original working list, but is included here because questions about its status have been raised since. There are no known developments within the range of this subspecies that would pose serious threats to its population.

Pallid Bobcat

Felis rufus pallescens is found on the Modoc Plateau and adjacent areas of the Great Basin province of northeastern California in Lassen, Modoc, Shasta, and Siskiyou counties (Gould, 1977a; Hall, 1981). Bobcats live in most terrestrial communities, but are seldom found on open, flat grasslands and deserts or in areas densely populated by humans. Areas with broken, rocky terrain and/or dense brush are associated with high densities of Bobcats, and also support high densities of small mammals and lagomorphs, their favored prey (Grinnell et al., 1937). The Pallid Bobcat was included on the draft List of Concern as a sensitive species (Priority Four), primarily because of the high price offered for its pelt and evidence reviewed by Gould (1977a). Some of the following data apply to all Bobcats in California, but they illustrate recent trends affecting Pallid Bobcats. The reported annual take of Bobcats in California by fur trappers increased from 241 in 1966-67 to 3618 in 1976-77, a 15-fold increase. Average prices paid for Bobcat pelts increased from $18.00 in 1966-67 to $133.50 in 1975-76 (Lee, 1978). Prices paid for prime pelts of Pallid Bobcats were much higher than this average, the highest recorded by 1977 was $405 (Gould, 1977a). Hunters also take substantial numbers of Bobcats each year. Gould (1977b) estimated that between 9600 and 11400 Bobcats were killed by hunters statewide during 1976. The U.S. Fish and Wildlife Service killed an additional 205 Bobcats in California for predator control. Gould (1977a) documented a probable decline in numbers of Pallid Bobcats between 1971 and 1976, listing increased fur trapping as the likely cause. A 4.5-fold increase in numbers taken by trappers in Lassen, Modoc, Shasta, and Siskiyou counties occurred between 1960-61 and 1975-76 (91 and 406). This probably represents only a fraction of the total take, because, according to Gould (1977a), considerable illegal trapping and hunting occurs, both within and outside of season. Also, many trappers do not file reports, and hunters not marketing pelts are not required to report on taking Bobcats. Zezulak (1980) studied Pallid bobcats between 1976 and 1979, radio-collaring and determining movements and size of home range for 5 adult males, 5 adult females, and 5 juveniles. He also determined litter size for females over a 2-year period. He reported that trappers filing reports took 162 Pallid bobcats near his study area between 1976 and 1979. Zezulak (1980) concluded that because 75% of the animals trapped were less than 2-years old, the population was expanding in response to "moderately heavy" human-caused mortality. Another interpretation of these data are that increased mortality due to trappers and hunters caused the reduction in the average age of bobcats in the population, without speculating as to whether the population was stable, declining, or increasing in size. In any case, evidence available to Zezulak (1980) suggested that the population was not jeopardized by the level of pressure exerted by hunters and trappers during the period of his study. Present regulations and management practices should be changed to facilitate collection of reliable estimates on take by hunter, illegal trapping, and licensed trapping by persons who fail to submit reports on captures. Treating the Pallid Bobcat as a sensitive species, requiring periodic monitoring of its population status and special consideration in land management decisions, is recommended.

Northwestern White-tailed Deer

Odocoileus virginianus ochrourus were reported from localities in Lassen, Modoc, Mono, Shasta, and Siskiyou counties (Grinnell, 1933; Hall and Kelson, 1959). Little is known of the habitat of White-tailed Deer in California. Walsingham (1873) remarked that they occurred on the plains around the Klamath marshes in Oregon and in the upper Pit River drainage of Shasta Co., California. He stated that in east-central Oregon, near Bend, they frequented thick clumps of willows and other woody shrubs bordering streams and marshes. Walsingham (1873) noted that Mule Deer were abundant on hillsides and that White-tailed Deer occupied the floodplains. Cowan (1936) wrote that over much of their former range, this subspecies of White-tailed Deer occupied riparian and floodplain communities. Bailey (1936) reviewed field notes of early explorers and other sources and concluded that members of this subspecies lived mainly in thickets of willows and other woody plants along streams. He noted that sometimes they were found in the hills and attributed this to being crowded out of more favorable areas by settlers in the valleys. Fisher (in Grinnell et al., 1937) believed that White-tailed Deer in eastern Lassen County occupied a geographic range approximately corresponding to the average winter range of Mule Deer. O. v. ochrourus was included on the working list because of its disputed status. Adams (1963) reviewed evidence of White-tailed Deer in California and disputed its occurrence within the state in historic times. Adams' (1963) conclusion was based on two factors: the habitat in California was not like the habitat for this species in Montana; and a general lack of undisputed specimens known to him. He stated that the habitat for White-tailed Deer in northeastern California was "strikingly different from that of the whitetail habitat that . . . [he] had known in northwestern Montana." To reach his conclusion he had to discount one specimen of a White-tailed Deer from California that he could not dispute and ignore the account by Walsingham (1873), an international authority on the family Cervidae, of his travels through northern California and his observations of White-tailed deer. He also discredited the writings of a number of other prominent naturalists (e.g., Bailey, 1936; Cowan, 1936; Dixon, 1927; and N. Hollister and C. H. Merriam in Adams, 1963) supporting the occurrence of White-tailed Deer in California. That few specimens from California were located by him is true, although there is no indication that Adams (1963) attempted to locate and examine a complete skin and partial skeleton (for a whole mount) donated to the Oakland Museum (Dixon, 1927). The lack of scientific specimens is not surprising in view of the lack or scarcity of specimens from California of other large mammals, such as the Bison, California Grizzly Bear, Gray Wolf, and Mexican Jaguar (Grinnell et al., 1937; Merriam, 1919, 1926). Geographic variation in habitat requirements was apparently unfamiliar to Adams, or at least that White-tailed Deer in California might differ in their use of habitat from those in Montana was a possibility that he did not accept. Arguing against his viewpoint is an extant population of Columbian White-tailed deer, in the vicinity of Roseburg, Oregon, less than 100 miles from the northern border of California. Its habitat is quite different from the riparian habitat along the Columbia River where the other population is found (Fisher et al., 1977; Suring and Vohs, 1979). In summary, the evidence strongly supports the occurrence of Northwestern White-tailed Deer in California. They were probably extirpated from California sometime between the 1930's and 1950's, mainly beccause of loss of habitat.

DISCUSSION

The major developments threatening mammalian species are virtually the same as identified in jeopardizing several species of birds in California (Remsen, 1978), with the loss and degradation of riparian woodlands and wetlands at the top of the list. Of the 36 species and subspecies on the List of Concern (Table 2), 15 are limited to or depend upon riparian and wetland communities, or are thought to be so dependent (Table 3). Four of these require habitat in riparian communities along the Colorado River: the Arizona Myotis, Arizona Cave Myotis, Colorado River Cotton Rat, and Yuma Mountain Lion. Three others require habitats in riparian and wetland communities on the floor of the San Joaquin Valley: the Buena Vista Lake Shrew, Riparian Brush Rabbit, and Riparian Woodrat. The Salt-marsh Wandering Shrew and Suisun Shrew occupy tidal marshes in San Francisco and San Pablo bays, respectively. The Southern California Salt-marsh Shrew and Southern Marsh Harvest Mouse live only in the tidal marshes along the southern California Coast. The Point Reyes Mountain Beaver and Point Reyes Jumping Mouse require wetland communities in the Point Reyes region, north of San Francisco Bay, and the Point Arena Mountain Beaver occupies wetlands in the vicinity of Point Arena, Mendocino Co. Additionally, Snowshoe Hares use riparian communities extensively, and the two on the List of Concern (Oregon Snowshoe Hare, Sierra Nevada Snowshoe Hare) require habitat in these communities.

Species of concern confined to lowland desert, grassland, and savanna communities of the southern California coastal basins include the Los Angeles Pocket Mouse and Pacific Pocket Mouse (Table 3). Also of concern in that region and elsewhere in the southern lowlands of California are the American Badger, California Leaf-nosed Bat, Big Free-tailed Bat, Pocketed Free-tailed Bat, and California Mastiff Bat. Loss of habitat probably is the principal factor jeopardizing these species.

Species jeopardized by loss of grassland and desert communities in the San Joaquin Valley and contiguous areas such as the Carrizo Plain are the Tipton Kangaroo Rat, Short-Nosed Kangaroo Rat, and populations of the American Badger. Loss and degradation of native communities in the San Joaquin Valley lowlands are especially serious problems. The importance of the San Joaquin Valley to populations of the California Mastiff Bat is unestablished, but it was probably a major area for foraging Mastiff Bats. No recent records of California Mastiff Bats are known from the San Joaquin Valley, although scanty data suggest they were more common into the 1950's or 1960's. In addition to the four species mentioned here and the three species requiring riparian and wetland communities listed above, four species of mammals with Rare and Endangered Status are confined to the region (San Joaquin Antelope Squirrel, Fresno Kangaroo Rat, Giant Kangaroo Rat, and San Joaquin Kit Fox; Table 1). There are also other endemic species and subspecies of mammals of the San Joaquin Valley, all of whose populations have been seriously reduced in size. Some, including the San Joaquin Pocket Mouse (*Perognathus inornatus inornatus* and *P. i. neglectus*) and Tulare Grasshopper Mouse (*Onychomys torridus tularensis*) also may be in danger, but available data suggest lesser threats to these species than those listed.

Loss and fragmentation of old-growth conifer forests pose threats to two species on the list, the Red Tree Mouse and the Pacific Fisher. Too little is known about the distribution and habitat of the White Footed Vole to state with certainty, but it too may be jeopardized by current forest harvest practices. Insufficient information is available to identify species of bats that might be similarly jeopardized, although a number of species are distributed mainly in conifer forests and may be jeopardized by widespread loss of mature and old-growth forests.

Nine species of concern are presently classified as game or furbearing species: the Riparian Brush Rabbit and Yuma Mountain Lion in the highest priority; and the Pygmy Rabbit, Oregon Snowshoe Hare, Sierra Nevada Snowshoe Hare, Western White-tailed Hare, Pacific Fisher, American Badger, and Channel Islands Spotted Skunk in the third priority. Only the Pacific Fisher is currently protected from taking. The Yuma Mountain Lion is classified as a game species, although at present there is no open hunting season on Mountain Lions in California. Regardless of whether or not other populations of Mountain Lions in California are allowed to be hunted, those of the southeastern desert areas should continue to be fully protected. Killing and trapping are likely to be major threats only for the populations of the American Badger of concern. For the American Badger, the Riparian Brush Rabbit, and species of hares on the List of Concern, data sufficient to justify the seasons and bag limits must be gathered. Determining the status of, and the impacts of hunting on, these species should be high priorities of the Department of Fish and Game.

Emphasis in the species accounts has been given to determining the population status and potential threats to individual species. State and federal legal protection should be sought for those species for which information is adequate to estabish their threatened or endangered status. While these activities will focus concern on individual species and provide a measure of protection for those found to require it, they are probably not the most efficient and cost-effective wey to ensure their survival.

Preservation of the mammalian species of concern listed here, species already with state or federal threatened and endangered status, and species of birds, reptiles, plants, and other taxonomic groups can be best accomplished by concentrating conservation efforts on their biotic communities rather than emphasizing single-species management. This also would provide more security to members of their communities that are not normally accorded protected status, but which may be essential to the perpetuation of their communities (e.g., lower plants, fungi, invertebrates).

An integrated, intergovernmental development/conservation approach which focuses upon preserving representative segments of each unique biotic community while other resource- and land-use goals are being formulated is needed. This would lessen the need for listing of most of these species as threatened or endangered and probably save much of the money and duplication of efforts expended on managment of threatened and endangered species on a one-by-one basis.

ACKNOWLEDGMENTS

Special acknowledgment goes to John R. Gustafson for initiating the project and providing advice and support during its execution. Wendy L. Ball, Kerry S. Kilburn, Karen A. Rogers, Gerard G. Rogers, and Susan D. Williams assisted with the literature searches, clerical tasks, and typing of drafts of the report. Kerry Kilburn edited the draft report and provided suggestions for improving it. Susan D. Williams assisted with production and editing of the final draft. The persons listed below contributed information and suggestions during the investigations; I am grateful for their assistance and the congenial hospitality several of them extended during my travels for this project: Carl T. Benz, Vernon C. Bleich, Brad R. Blood, Michael A. Bogan, Patricia E. Brown, Michael T. Chapel, Dana R. Chapman, David Chesemore, Ronald E. Cole, Paul W. Collins, Denny G. Constantine, Blair Csuti, David E. Davis, Hugh H. Genoways, Cynthia Gray, Randall L. Gray, Walt M. Hoffmann, David G. Huckaby, Monty D. Knudsen, Gail C. Kobetich, Karen A. Koos, Timothy E. Lawlor, Philip Leitner, William Z. Lidicker, Jr., Michael S. Marangio, Ronald M. Nowak, Robert D. Ohmart, James L. Patton, Ted Rado, Aryan I. Roest, Robert L. Rudd, Larry Sitton, Howard S. Shellhammer, James D. Smith, Kenneth E. Stager, Dale T. Steele, James M. Sulentich, Richard A. Weaver, and Don E Wilson.

LITERATURE CITED

Adams, L. 1963. Do white-tailed deer occur in northeastern California? J. Mamm., 44:518-522.

Airola, D. A., ed. California Wildlife Habitat Relationships Program: Northeast Interior Zone. U.S.D.A. Forest Service, San Francisco, California, IV:1-255.

Anderson, S., and C. E. Nelson. 1965. A systematic revision of Macrotus (Chiroptera). Amer. Mus. Novitates, 2212:1-39.

August, P. V., and R. E. Dingman. 1973. Two additional records of Tadarida macrotus and Euderma maculatum from San Diego, California. Bull. So. California Acad. Sci., 72:165-166.

Bailey, V. 1905. Biological survey of Texas. N. Amer. Fauna, 25:1-222.

_____. 1936. The mammals and life zones of Oregon. N. Amer. Fauna, 55:1-416.

_____. 1939. The solitary lives of two little pocket mice. J. Mamm., 20:325-328.

Banks, R. C., and C. Parrish. 1965. Additional records of Choeronycteris mexicana from southern California and Baja California. Bull. So. California Acad. Sci., 64:163-164.

Barbour, R. W., and W. H. Davis. 1969. Bats of America. Univ. Press of Kentucky, Lexington, 286 pp.

_____. 1970. The status of Myotis occultus. J. Mamm., 51:150-151.

Barclay, R. M. R., D. W. Thomas, and M. B. Fenton. 1980. Comparison of methods used for controlling bats in buildings. J. Wildl. Manage., 44:502-506.

Benson, S. B., and A. E. Borrell. 1931. Notes on the life history of the red tree mouse, Phenacomys longicaudus. J. Mamm., 12:226-233.

Best, T. L. 1983. Intraspecific variaiton in the agile kangaroo rat (Dipodomys agilis). J. Mamm., 426-436.

Bills, A. R. 1969. A study of the distribution and morphology of the mice of Santa Cruz Island. An example of divergence. Unpubl. M.A. thesis, Univ. California, Santa Barbara, 91 pp.

Bond, S. I. 1977. An annotated list of the mammals of San Diego County, California. Trans. San Diego Soc. Nat. Hist., 18:229-247.

Boulware, J. T. 1943. Two new subspecies of kangaroo rats (genus Dipodomys) from southern California. Univ. California Publ. Zool., 46:391-396.

Bradley, W. G. 1966. Status of the cotton rat in Nevada. J. Mamm., 47:349-350.

Brown, R. J. 1971. A comparative study of the chromosomes of some Pacific Coast shrews (genus Sorex). Dissert. Abst., 31:7684-B.

_____. 1974. A comparative study of the chromosomes of three species of shrews, Sorex bendirii, Sorex trowbridgii, and Sorex vagrans. Wassmann J. Biol., 32:303-326.

Brown, R. J., and R. L. Rudd. 1981. Chronosomal comparisons within the Sorex ornatus-S. Vagrans complex. Wasmann J. Biol., 39:30-35.

Bruce, P., and S. Weick. 1973. Wolverine, fisher, and marten occurrence and winter movements in northwestern California. California Dept. Fish and Game, Spec. Wildl. Invest., Project W-54-R, 9 pp.

Buck, S., C. Mullis, and A. Mossman. 1983. Corral Bottom-Hayfork Bally Fisher study. U.S.D.A., Forest Service, San Francisco, California, Unpubl. Rept., 136 pp.

Camp, C. L. 1918. Excavations of burrows of the rodent Aplondontia, with observations on the habits of the animal. Univ. California Publ. Zool., 18:517-536.

Chapman, J. A. 1974. Sylvilagus bachmani. Mamm. Species, 34:1-4.

Choate, J. R., and H. H. Genoways. 1975. Collections of Recent mammals in North America. J. Mamm., 56:452-502.

Clark, W. A., S. M. Juarez, and D. L. Chesemore. 1982. Nature Conservancy small mammal inventory of Paine Wildflower Preserve and the Voice of America in Kern County, California. The Nature Conservancey, San Francisco, California, unpubl. rept., 47 pp.

Clough, G. C., and R. E. Thorne. 1976. Ecological problems and management practices: Santa Catalina Island. Summary Rept., Center for Natural Areas, Washington, D.C., 27 pp.

Collins, P. W., and T. D. Martin. 1985. A review of the population status of the Santa Catalina Island shrew, Sorex ornatus willetti. U.S.D.I., Fish and Wildlife Serv., Endangered Species Office, Sacramento, California, Final Rept., 10181-5693 (BW) '85, SE-0018-85, 92 pp.

Constantine, D. G. 1979. Bat rabies and bat management. Bull. Soc. Vector Ecol., 4:1-9.

Constantine, D. G., and G. L. Humphrey. 1979. Rabies in Myotis thysanodes, Lasiurus ega, Euderma maculatum and Eumops perotis in California. J. Wildl. Dis., 15:343-345.

Couch, L. K. 1927. Migration of the Washington black-tailed jack rabbit. J. Mamm., 8:313-314.

Coues, E. 1875. A critical review of the North American Saccomyidae.

Proc. Acad. Nat. Sci. Philadelphia, pp. 272-327.

Cowan, I. McT. 1936. Distribution and variation in deer (genus Odocoileus of the Pacific Coastal Region of North America. California Fish and Game, 22:155-246.

Cunningham, J. D. 1959. The wolverine and fisher in the Yosemite region. J. Mamm., 40:614-615.

Currier, M. J. P. 1983. Felis concolor. Mamm. Species, 200:1-7.

Dalquest, W. W. 1942. Geographic variation in the northwestern snowshoe hares. J. Mamm., 23:166-183.

_____. 1946. The daytime retreat of a California mastiff bat. J. Mamm., 27:86-88.

_____. 1947. Notes on the natural history of bats Corynorhinus rafinesquii in California. J. Mamm., 28:17-30.

_____. 1948. Mammals of Washington. Univ. Kansas Publ., Mus. Nat. Hist., 2:1-444.

Davis, B. L., amd R. J. Baker. 1974. Morphometrics, evolution, and cytotaxonomy of mainland bats of the genus Macrotus (Chiroptera: Phyllostomatidae). Syst. Zool., 23:26-39.

Dickey, D. R. 1929. The spotted skunk of the Channel Islands of southern California. Proc. Biol. Soc. Washington, 42:157-160.

Dixon, J. S. 1922. Rodents and reclamation in the Imperial Valley. J. Mamm., 3:136-146.

_____. J. S. 1927. The white-tailed deer in California. J. Mamm., 8:308.

Easterla, D. A. 1971. Notes on young and adults of the spotted bat, Euderma maculatum. J. Mamm., 52:475-476.

Eger, J. L. 1977. Systematics of the genus Eumops (Chiroptera: Molossidae). Life Sci. Contrib., Royal Ontario Mus., 110:1-69.

Fenton, M. B., and R. M. R. Barclay. 1980. Myotis lucifugus. Mamm. Species, 142:1-8.

Fenton M. B., D. C. Tennant, and J. Wyszecki. 1983. A survey of the distribution of Euderma maculatum (Chiroptera: Vespertilionidae) throughout its known range in the United States and Canada by monitoring its audible echolocation calls. U.S.D.I. Fish and Wildl. Serv., Endangered Species Office, Albuquerque, New Mexico, 14-16-0002-82-210, 44 pp.

Findley, J. S. 1955. Speciation of the wandering shrew. Univ. Kansas Publ., Mus. Nat. Hist., 9:1-68.

_____. 1972. Phenetic relationships among bats of the genus Myotis. Syst. Zool., 21:31-52.

Findley, J. S., and C. Jones. 1965. Comments on spotted bats. J. Mamm., 46:679-680.

_____. 1967. Taxonomic relationships of bats of the species Myotis fortidens, M. lucifugus, and M. occultus. J. Mamm., 48:429-444.

Findley, J. S., A. H. Harris, D. E. Wilson, and C. Jones. 1975. Mammals of New Mexico. Univ. New Mexico Press, Albuquerque, 360 pp.

Fisher, D. A., et al. 1977. Revised Columbian white-tailed deer recovery plan. U.S.D.I. Fish and Wildl. Serv., Endangered Species Office, Washington, D.C., 65 pp.

Ford, R. K. A review of the population status of the salt marsh wandering shrew, Sorex vagrans halicoetes. U.S.D.I. Fish and Wildl. Serv., Endangered Species Office, Sacramento, California, Final Rept., FWS 8502, 48 pp.

Fry, W. 1924. The Sierra white-tailed jack rabbit. California Fish and Game, 10:172-175.

Geluso, K. N., J. S. Altenbach, and D. E. Wilson. Bat mortality: pesticide poisoning and migratory stress. Science, 194:184-186.

Goldman, E. A. 1928. Three new rodents from western Arizona. Proc. Biol. Soc. Washington, 41:203-206.

_____. 1935. New American mustelids of the genera Martes, Gulo, and Lutra. Proc. Biol. Soc. Washington, 48:175-186.

Gould, G. I., Jr. 1977a. Bobcat distribution in northeastern California. California Dept. Fish and Game, Nongame Wildl. Invest., Job IV-1.2, Progress Rept., 10 pp.

_____. 1977b. Estimated hunter take of bobcat during 1976. California Dept. Fish and Game, Nongame Wildl. Invest., Project W-54-R9, Job IV-1.0, Progress Rept., 9 pp.

_____. 1978. Furbearer study, Sierra County, California. California Dept. Fish and Game, Nongame Wildl. Invest., Project W-54-10, 9 pp.

Graham, R. E. 1966. Observations on the roosting habits of the big-eared bat, Plecotus townsendii in California limestone caves. Cave Notes, 8:17-22.

Grayson, D. K. 1977. On the Holocene history of some northern Great Basin lagomorphs. J. Mamm., 58:507-513.

Green, J. S., and J. T. Flinders. 1980. Brachylagus idahoensis. Mamm. Species, 125:1-4.

Grinnell, H. W. 1918. A synopsis of bats of California. Univ.

California Publ. Zool., 117:223-404.

Grinnell, J. 1908. The biota of the San Bernardino Mountains. Univ. Calif. Publ. Zool., 5:1-170.

_____. 1914. An account of the mammals and birds of the lower Colorado Valley with especial reference to the distributional problems presented. Univ. California Publ. Zool., 12:51-294.

_____. 1921. Revised list of the species in the genus Dipodomys. J. Mamm., 2:94-97.

_____. 1922. A geographical study of the kangaroo rats of California. Univ. California Publ. Zool., 24:1-124.

_____. 1932. A relic shrew from central California. Univ. California Publ. Zool., 38:389-390.

_____. 1933. Review of the Recent mammal fauna of California. Univ. California Publ. Zool., 40:71-284.

Grinnell, J., and J. Dixon. 1918. Natural history of the ground squirrels of California. Bull. California State Comm. Horticulture, 7:597-708.

Grinnell, J., and J. M. Linsdale. 1929. A new kangaroo rat from the upper Sacramento Valley, California. Univ. California Publ. Zool., 30:453-459.

Grinnell, J., and T. I. Storer. 1924. Animal life in the Yosemite. An Account of the mammals, birds, reptiles, and amphibians in a cross section of the Sierra Nevada. Univ. California Press, Berkeley, 752 pp.

Grinnell, J., and H. S. Swarth. 1913. An account of the birds and mammals of the San Jacinto area of southern California. Univ. California Publ. Zool., 10:197-406.

Grinnell, J., J. S. Dixon, and J. M. Linsdale. 1930. Vertebrate natural history of a section of northern California through the Lassen Peak Region. Univ. California Publ. Zool., 35:1-594.

_____. 1937. Fur-bearing mammals of California. Their natural history, systematic status, and relations to man. Univ. California Press, Berkeley, 1:1-375, 2:376-777.

Hafner, M. S. 1979. Density, distribution, and taxonomic status of Dipodomys nitratoides nitratoides Merriam, 1894 (Rodentia - Heteromyidae). California Dept. Fish and Game, Nongame Wildl. Investigations, Draft Final Rept., 17 pp.

Hall, E. R. 1946. Mammals of Nevada. Univ. California Press, Berkeley, 710 pp.

_____. 1981. The mammals of North America, 2nd. ed. John Wiley & Sons, New York, 1:1-600 + 90, 2:601-1181 + 90.

_____. 1984. Geographic variation among brown and grizzly bears (Ursus arctos) in North America. Spec. Publ. Mus. Nat. Hist., Univ. Kansas, 13:1-16.

Hall, E. R., and K. R. Kelson. 1959. Mammals of North America. The Ronald Press, New York, 1:1-546 + 79, 2:547-1083 + 79.

Halloran, A. F. 1946. A recent record of the Yuma mountain lion in Arizona. J. Mamm., 27:89-90.

Hamilton, W. J., III. 1962. Reproductive adaptations of the red tree mouse. J. Mamm., 43:486-504.

Harris, A. H. 1974. Myotis yumanensis in interior southwestern North America, with comments on Myotis lucifugus. J. Mamm., 55:589-607.

Harris, J. H. 1982. Mammals of the Mono Lake - Tioga Pass region. David Gaines/Kutsavi Books, Lee Vining, California, 55 pp.

Hawbecker, A. C. 1940. The burrowing and feeding habits of Dipodomys venustus. J. Mamm., 21:388-396.

_____. 1951. Small mammal relationships in an ephedra community. J. Mamm., 32:50-60.

Hayward, B. J. 1970. The natural history of the cave bat, Myotis velifer. Western New Mexico Univ. Res. Sci., 1:1-74.

Hennings, D., and R. S. Hoffmann. 1977. A review of the taxonomy of the Sorex vagrans species complex from western North America. Occas. Papers Mus. Nat. Hist., Univ. Kansas, 68:1-35.

Hoffmann, R. S., and J. G. Owen. Sorex tenellus and Sorex nanus. Mamm. Species, 131:1-4.

Hoffmann, W. M. 1975. Geographic variation and taxonomy of Dipodomys nitratoides from the California San Joaquin Valley. Unpubl. M.A. Thesis, California State Univ., Fresno, 75 pp.

Hollister, N. 1909. Two new bats from the southwestern United States. Proc. Biol. Soc. Washington, 22:43-44.

Hooper, E. T. 1944. San Francisco Bay as a factor influencing speciation in rodents. Misc. Publ. Mus. Zool., Univ. Michigan, 59:1-89.

Howell, A. B. 1920a. Some Californian experiences with bat roosts. J. Mamm., 1:169-177.

_____. 1920b. Contribution to the life history of the California mastiff bat. J. Mamm., 1:111-117.

_____. 1920c. A second record of Phenacomys albipes in California, with a discussion of the species. J. Mamm., 1:242-243.

_____. 1920d. A study of the California jumping mice of the genus Zapus. Univ. California Publ. Zool., 21:225-238.

_____. 1924. The mammals of Mammoth, Mono County, California. J. Mamm., 5:25-36.

_____. 1926. Voles of the genus Phenacomys. I. Revision of the genus Phenacomys. II. Life history of the red tree mouse (Phenacomys longicaudus). N. Amer. Fauna, 18:1-66.

_____. 1928. The food and habitat preferences of Phenacomys albipes. J. Mamm., 9:153-154.

Howell, A. H. 1914. Revision of the American harvest mice (genus Reithrodontomys). N. Amer. Fauna, 36:1-97.

_____. 1918. Revision of the American flying squirrels. N. Amer. Fauna, 44:1-64.

_____. 1929. Revision of the American chipmunks (genera Tamias and Eutamias). N. Amer. Fauna, 52:1-157.

Huey, L. M. 1926. A new Perognathus from the vicinity of Mount Pinos, Kern County, California. Proc. Biol. Soc. Washington, 39:121-122.

_____. 1932. The Large-eared free-tailed bat in San Diego. J. Mamm., 13:160.

_____. 1939. The silky pocket mice of southern California and northern Lower California, Mexico, with the description of a new race. Trans. San Diego Soc. Nat. Hist., 11:47-54.

_____. 1954a. Second record of Tadarida molossa from San Diego, California. J. Mamm., 35:435-436.

_____. 1954b. Choeronycteris mexicana from southern California and Baja California, Mexico. J. Mamm., 35:436-437.

Humphrey, S. R., and T. H. Kunz. 1976. Ecology of a Pleistocene relict, the western big-eared bat (Plecotus townsendii) in the southern Great Plains. J. Mamm., 57:470-494.

Ingles, L. G. 1965. Mammals of the Pacific states. Stanford Univ. Press, Stanford, California, 506 pp.

Jackson, H. H. T. 1921. Two unrecognized shrews from California. J. Mamm., 2:161-162.

_____. 1928. A taxonomic review of the American longtailed shrews (genera Sorex and Microsorex). N. Amer. Fauna, 51:1-238.

Johnson, D. H. 1943. Systematic review of the chipmunks (genus Eutamias) of California. Univ. California Publ. Zool., 48:63-148.

Johnson, D. H., M. D. Bryant, and A. H. Miller. 1948. Vertebrate animals of the Providence Mountains area of California. Univ. California Publ. Zool., 48:221-376.

Johnson, M. L. 1973. Characters of the heather vole, Phenacomys, and the red tree vole, Arborimus. J. Mamm., 54:239-244.

Johnson, M. L., and C. Maser. 1982. Generic relationships of Phenacomys albipes. Northwest Sci., 56:17-19.

Johnston, R. F., and R. L. Rudd. 1957. Breeding of the salt marsh shrew. J. Mamm., 38:157-163.

Jones, F. L. 1957. Southern extension of the range of the pygmy rabbit in California. J. Mamm., 38:274.

Jones, G. S., J. O. Whitaker, Jr., and C. Maser. 1978. Food habits of jumping mice (Zapus trinotatus and Z. princeps) in western North America. Northwest Sci., 52:57-60.

Jones, J. K., Jr., D. C. Carter, H. H. Genoways, R. S. Hoffmann, and D. W. Rice. 1982. Revised checklist of North American mammals north of Mexico. Occas. Papers Mus. Texas Tech Univ., 80:1-22.

Junge, J. A., and R. S. Hoffmann. 1981. An annotated key to the long-tailed shrews (genus Sorex) of the United States and Canada, with notes on the Middle American Sorex. Occas. Papers Mus. Nat. Hist., Univ. Kansas, 94:1-48.

Krutzsch, P. H. 1943. Zonal distribution of the California mastiff bat. J. Mamm., 24:269.

_____. 1944a. Notes on the little-known pocketed bat. J. Mamm., 25:201.

_____. 1944b. Fourth record of the pocketed bat in California. J. Mamm., 25:413-414.

_____. 1945. Observations on a colony of molossids. J. Mamm., 26:196.

_____. 1954. North American jumping mice (genus Zapus). Univ. Kansas publ., Mus. Nat. Hist., 7:349-472.

_____. 1955. Observations on the California mastiff bat. J. Mamm., 36:407-414.

Kuchler, A. W. 1977. The map of natural vegetation in California. Pp. 909-938 + supplement, in The terrestrial vegetation in California (M. B. Barbour and J. Major, eds.). John Wiley and Sons, New York, 1002 pp.

Kunz, T. H. 1974. Feeding ecology of a temperate insectivorous bat (Myotis velifer). Ecology, 55:693-711.

Kunz, T. H., R. A. Martin. 1982. Plecotus townsendii. Mamm. Species, 175:1-6.

Laughrin, L. L. 1973. California island fox survey. California Dept. Fish and Game, Wildl. Mgmt. Branch Admin. Rept., 73-3, 39 pp.

Lee, R. C. 1977. The status of harvested furbearers in California – badger, beaver, gray fox, mink, muskrat and raccoon. California Dept. Fish and Game, Nongame Wildl. Invest., Progress Rept., Project W-54-R-9, Job II-1.0, 29 pp.

_____. 1978. Status of the bobcat in California. California Dept. Fish and Game, Nongame Wildl. Invest., 6 pp.

Leitner, P. 1966. Body temperature, oxygen consumption, heart rate and shivering in the California mastiff bat, Eumops perotis. Comp. Biochem. Physiol., 19:431-443.

Leonard, M. L., and M. B. Fenton. 1983. Habitat use by spotted bats (Euderma maculatum, Chiroptera: Vespertilionidae): roosting and foraging behaviour. Canadian J. Zool., 61:1487-1491.

Leslie, D. M., Jr. and C. L. Douglas. 1979. Desert bighorn sheep of the River Mountains, Nevada. Wildl. Monogr., 66:1-56.

Lidicker, W. Z., Jr. 1960. An analysis of intraspecific variation in the kangaroo rat Dipodomys merriami. Univ. California Publ. Zool., 67:125-218.

Long, C. A. 1972. Taxonomic revision of the North American badger, Taxidea taxus. J. Mamm., 53:725-759.

_____. 1973. Taxidea taxus. Mamm. Species, 26:1-4.

Long, C. A., and C. A. Killingby. 1983. The badgers of the world. C. C. Thomas Publ., Springfield Illinois, 406 pp.

Longhurst, W. M. 1940. Mammals of Napa County, California. California Fish and Game, 26:240-270.

Mailliard, J. 1927. The birds and mammals of Modoc County, California. Proc. California Acad. Sci., 16:261-359.

Maser, C., and M. L. Johnson. 1967. Notes on the white-footed vole (Phenacomys albipes). Murrelet, 48:24-27.

Maser, C., and R. M. Storm. 1970. A key to the Microtinae of the Pacific Northwest. Oregon State Univ. Bookstore, Corvallis, 162 pp.

Maser, C., B. R. Mate, J. F. Franklin, and C. T. Dyrness. 1981. Natural history of Oregon Coast mammals. Pacific Northwest Forest and Range Experiment Station, U.S.D.A. Forest Serv., Gen. Tech. Rept., PNW-133:1-496.

McLean, D. D. 1931. White-tailed deer in California. California Fish and Game, 17:342.

_____. 1944. The prong-horned antelope in California. California Fish and Game, 30:221-241.

_____. 1954. Mountain lions in California. California Fish and Game, 40:147-166.

M'Closkey, R. T. 1972. Temporal Changes in populations and species diversiry in a California rodent community. J. Mamm., 53:657-676.

Mearns, E. A. 1898. Descriptions of three new forms of pocket mice from the Mexican border of the United States. Bull. Amer. Mus. Nat. Hist., 10:299-320.

Merriam, C. H. 1889. Revision of the North American Pocket mice. N. Amer. Fauna, 1:1-29.

_____. 1899. Results of a biological survey of Mount Shasta, California. N. Amer. Fauna, 16:1-179.

_____. 1903. Eight new mammals from the United States. Proc. Biol. Soc. Washington, 16:73-78.

_____. 1904. Jack rabbits of the Lepus campestris group. Proc. Biol. Soc. Washington, 18:131-134.

_____. 1919. Is the jaguar entitled to a place in the California fauna? J. Mamm., 1:38-40.

_____. 1926. The buffalo in northeastern California. J. Mamm., 7:211-214.

Meserve, P. L. 1976. Food relationships of a rodent fauna in a California coastal sage communiry. J. Mamm., 57:300-319.

Messick, J. P., and M. G. Hornocker. 1981. Ecology of the badger in southwestern Idaho. Wildl. Monog., 76:1-53.

Mossman, A. 1979. Mammal narratives, in California Wildlife/Habitat Relationships Program. North Coast/Cascade Zone. USDA - Forest Serv., Pacific Southwest Forest and Range Exp. Sta., Berkeley, vol. 3, 189 unnumbered pp. + 67.

Olson, A. C., Jr. 1947. First record of Choeronycteris mexicana in California. J. Mamm., 28:183-184.

Orr, R. T. 1935. Description of three new races of brush rabbit from California. Proc. Biol. Soc. Washington, 48:27-30.

_____. 1940. The rabbits of California. Occas. Papers California Acad. Sci., 19:1-227.

Osgood, W. H. 1900. Revision of the pocket mice of the genus Perognathus. N. Amer. Fauna, 18:1-65.

Owen, J. G., and R. S. Hoffmann. 1983. Sorex ornatus. Mamm. Species, 212:1-5.

Pearson, O. P. 1951. Additions to the fauna of Santa Cruz Island, California, with description of a new subspecies of Reithrodontomys megalotis. J. Mamm., 32:366-368.

Pearson, O. P., M. R. Koford, and A. K. Pearson. 1952. Reproduction of the lump-nosed bat (Corynorhinus rafinesquii) in California. J. Mamm., 33:273-320.

Powell, R. A. 1981. Martes pennanti. Mamm. Species, 156:1-6.

Remsen, I. V. Jr. 1978 [1982]. Bird species of special concern in California: An annotated list of declining or vulnerable bird species. California Dept. of Fish and Game, Wildl. Manage. Branch Admin. Rept., 78-1:1-54.

Rhoads, S. N. 1894. Descriptions of new species of North American mammals with remarks on species of the genus Perognathus. Proc. Acad. Sci. Philadelphia, 1893:404-412.

Rudd, R. L. 1955a. Population variation and hybridization in some California shrews. Syst. Zool., 4:21-34.

_____. 1955b. Age, sex, and weight comparisons in three species of shrews. J. Mamm., 36:323-339.

Sanborn, C. C. 1932. The bats of the genus Eumops. J. Mamm., 13:347-357.

Sargent, A. B., and D. W. Warner. 1972. Movements and denning habits of a badger. J. Mamm., 53:207-210.

Scheffer, T. H. 1929. Mountain beavers in the Pacific Northwest: their habits, economic status, and control. Farmers Bull., U.S. Dept. Agric., 1598:1-18.

Schempf, P. F., and M. White. 1977. Status of six furbearer populations in the mountains of northern California. USDA - Forest Serv., California Region, San Francisco, 51 pp.

Severaid, J. 1950. The pygmy rabbit (Sylvilagus idahoensis) in Mono County, California. J. Mamm., 31:1-4.

Severinghaus, W. D., and D. F. Hoffmeister. 1978. Qualitative cranial characters distinguishing Sigmodon hispidus and Sigmodon arizonae and the distribution of these two species in northern Mexico. J. Mamm., 59:868-870.

Stager, K. E. 1939. Status of Myotis velifer in California with notes on its life history. J. Mamm., 20:225-228.

_____. 1943. Remarks on Myotis occultus in California. J. Mamm., 24:197-199.

Stephens, F. 1906. California mammals. West Coast Publ. Co., San Diego, 351 pp.

Storer, T. I. 1926. What is the northward limit of range for the mastiff bat? J. Mamm., 7:131.

Strickland, M. A., C. W. Douglas, M. Novak, and N. P. Hunziger. 1982a. Fisher. Pp. 586-598, in Wild mammals of North America: Biology, Managment, Economics (J. A. Chapman and G. A. Feldhamer, eds.). The Johns Hopkins Univ. Press, Baltimore, 1147 pp.

______. 1982b. Marten. Pp. 599-612, in Wild mammals of North America: Biology, Managment, Economics (J. A. Chapman and G. A. Feldhamer, eds.). The Johns Hopkins Univ. Press, Baltimore, 1147 pp.

Strong, W. D. 1926. Indian records of California carnivores. J. Mamm., 7:59-60.

Sulentich, J. M. 1983. The systematics and evolution of the *Perognathus parvus* species group in southern California (Rodentia: Heteromyidae). Unpubl. M.S. thesis, California State Univ., Long Beach, 97 pp.

Summer, E. L., Jr. 1927. Notes on the San Bernardino flying squirrel. J. Mamm., 8:314-316.

Suring, L. H., and P. A. Vohs, Jr. 1979. Habitat use by Columbian white-tailed deer. J. Wildl. Manage., 43:610-619.

Tappe, D. T. 1942. The status of beaver in California. Game Bull., California Dept. Fish and Game, 3:1-59.

Taylor, W. P. 1915. Descriptions of a new subgenus (*Arborimus*) of *Phenacomys*, with a contribution to knowledge of the habits and distribution of *Phenacomys longicaudus* True. Proc. California Acad. Sci., 4th ser., 5:111-161.

_____. 1918. Revision of the rodent genus *Aplodontia*. Univ. California Publ. Zool., 17:435-504.

Thaeler, C. S., Jr. 1968. An analysis of the distribution of pocket gopher species in northeastern California (genus *Thomomys*). Univ. California Publ. Zool., 86:1-46.

Todd, A. 1891. An Oregon mouse. West. Amer. Sci., 7:240-242.

Twente, J. W. 1955. Some aspects of habitat selection and other behavior of cavern-dwelling bats. Ecology, 36:706-732.

U.S. Fish and Wildlife Service. 1982. Notice of review of endangered and threatened wildlife and plants: review of vertebrate wildlife for listing as endangered or threatened species. Federal Register, 47(251):58454-58460.

Vaughan, T. A. 1954. A new subspecies of bat (*Myotis velifer*) from southeastern California and Arizona. Univ. Kansas Publ., Mus. Nat. Hist., 7:507-512.

_____. 1959. Functional morphology of the three bats: Eumops, Myotis, Macrotus. Univ. Kansas Publ., Mus. Nat. Hist., 12:1-153.

von Bloeker, J. C., Jr. 1930. Extension of range of Perognathus pacificus. J. Mamm., 11:431-432.

_____. 1931. Perognathus pacificus from the type locality. J. Mamm., 12:369-372.

_____. 1932a. Three new mammals from salt marsh areas in southern California. Proc. Biol. Soc. Washington, 45:131-138.

_____. 1932b. Summer records of Eumops californicus. J. Mamm., 13:274.

_____. 1937. Four new rodents from Monterey County, California. Proc. Biol. Soc. Washington, 50:153-158.

_____. 1942. Contributions from the Los Angeles Museum - Channel Islands biological survey no. 22--A new subspecies of white-footed mouse from the Anacapa Islands, California. Bull. So. California Acad. Sci., 40:161-162.

_____. 1943. Extensions of range for three bats in California. J. Mamm., 24:403-404.

_____. 1967. Land mammals of the southern California Islands. Pp. 245-263, in Proceedings of the symposium on the biology of the California islands (R. N. Philbrick, ed.). Santa Barbara Botanical Gardens, Santa Barbara, California, 363 pp.

Waithman, J., and A. I. Roest. 1977. A taxonomic study of the kit fox, Vulpes macrotis. J. Mamm., 58:157-164.

Walker, P. L. 1980. Archaeological evidence for the Recent extinction of three terrestrial mammals on San Miguel Island. Pp. 703-717, in The California Islands: Proceedings of a multidisciplinary symposium (D. M. Power, ed.). Santa Barbara Nat. Hist. Museum, Santa Barbara, California, 787 pp.

Walsingham, T. L. 1873. On the distribution of the different species of deer and other ruminants in northern California and Oregon. Proc. Zool. Soc. London, pp. 561-563.

Warner, R. E. 1979. The California riparian study program. Phase I: background studies and program design for phase II. California Dept. Fish and Game, Planning Branch, Sacramento, 179 pp.

Watkins, L. C. 1977. Euderma maculatum. Mamm. Species, 77:1-4.

Weaver, R. A., and J. Hall. 1971. Bighorn sheep in Joshua Tree National Monument and adjacent areas. California Dept. Fish and Game, Wildl. Mgmt. Admin. Rept., 71-7:1-48.

Weaver, R. A., and J. L. Mensch. 1969. A report on desert bighorn sheep in eastern Imperial County, November 1968--January 1969. California

Dept. Fish and Game, Project W-51-R-14, Appendix 3, 19 pp.

Weaver, R. A., J. L. Mensch, and R. D. Thomas. 1969. A report on desert bighorn sheep in northeastern San Bernardino County. January 1969—July 1969. California Dept. Fish and Game, Project W-51R-14, Appendix 2, 26 pp.

Weiss, N. T., and B. J. Verts. 1984. Habitat and distribution of pygmy rabbits (Sylvilagus idahoensis) in Oregon. Great Basin Nat., 44:563-571.

White, M., R. H. Barrett, A. S. Boss, T. F. Newman, T. J. Rahn, and D. F. Williams. 1980. Mammals. Pp. 321-433, in California wildlife and their habitats: western Sierra Nevada (J. Verner and A. S. Boss, eds.). Pacific Southwest Forest and Range Exp. Sta., Forest Serv., U.S. Dept. Agric., Gen. Tech. Rept., PSW-37:1-439.

White, S. M., J. T. Flinders, and B. L. Welch. 1982. Preference of pygmy rabbits (Brachylagus idahoensis) for various populations of big Sagebrush (Artemisia tridentata). J. Range Manage., 35:724-726.

Williams, D. F. 1978. Karyological affinities of the species groups of silky pocket mice (Rodentia, Heteromyidae). J. Mamm., 59:599-612.

_____. 1979. Checklist of California mammals. Ann. Carnegie Mus., 48:425-433.

_____. 1983. Populations surveys of the Santa Catalina, San Bernardino and Suisun shrews. U.S.D.I., Fish and Wildlife Serv., Endangered Species Office, Sacramento, California, 11310-0242-2, 69 pp.

_____. 1984. Habitat associations of some rare shrews (Sorex) from California. J. Mamm., 65:325-328.

_____. 1985. A review of the population status of the Tipton kangaroo rat, Dipodomys nitratoides nitratoides. U.S.D.I., Fish and Wildlife Serv., Endangered Species Office, Sacramento, California, Final Report, 10181-4861 (rs) '85, SE-0020-4, 44 pp.

Williams, D. F., and S. E. Braun. 1983. Comparison of pitfalls and conventional traps for sampling small mammal populations. J. Wildl. Mgmt.

Woodsworth, G. C., G. P. Bell, and M. B. Fenton. 1981. Observations of the echolocation, feeding behaviour, and habitat use of Euderma maculatum (Chiroptera: Vespertilionidae) in southcentral British Columbia. Canadian J. Zool., 59:1099-1102.

Yocum, C. F., and M. T. McCollum. 1973. Status of the fisher in northern California, Oregon, and Washington. California Fish and Game, 59:305-309.

Young, S. P., and E. A. Goldman. 1946. The puma. Mysterious American cat. Dover Publ. Inc., New York, 1964 reprint, 358 pp.

Zezulak, D. S. 1980. Northeastern California bobcat study. California Dept. Fish and Game, Nongame Wildl. Investigations, Project E-W-2, Job IV-1.6, 34 pp.

Zimmerman, E. G. 1970. Karyology, systematics and chromosomal evolution in the rodent genus, Sigmodon. Publ. Mus., Michigan State Univ., Biol. Ser., 4:389-454.

APPENDIX

Acronyms for museums where research collections of specimens and other documentation of localities referred to in the lists of distribution records are listed below. Museum acronymns generally follow Choate and Genoways (1975).

AMNH	American Museum of Natural History, New York
ANSP	Philadelphia Academy of Natural Sciences
CAS	California Academy of Sciences, San Francisco
CM	Carnegie Museum of Natural History, Pittsburgh
CSCS	California State University, Stanislaus
CSLB	California State University, Long Beach
CSUC	California State University, Chico
CSUF	California State University, Fresno
CSUS	California State University, Sacramento
DFW	Sight records and uncataloged specimens, Daniel F. Williams
FMHN	Field Museum of Natural History, Chicago
HSU	Humboldt State University
KU	University of Kansas, Museum of Natural History
LACM	Natural History Museum of Los Angeles County
LSU	Louisiana State University
MCZ	Harvard University, Museum of Comparative Zoology
MSB	University of New Mexico, Museum of Southwestern Biology
MVZ	University of California, Berkeley, Museum of Vertebrate Zool.
PM	Paris Museum
ROM	Royal Ontario Museum, Toronto
SDSNH	San Diego Society of Natural History Museum
SFSU	San Francisco State University
TCWC	Texas A & M University, Texas Cooperative Wildlife Collections
UCLA	University of California, Los Angeles
UDAV	University of California, Davis, Museum of Zoology
UI	University of Illinois, Museum of Natural History
UMMZ	University of Michigan, Museum of Zoology
USNM	National Museum of Natural History, Washington, D.C.
WFBM	University of California Davis, Museum of Wildlife and Fisheries Biology

BEFORE THE CALIFORNIA ENERGY COMMISSION
STATE OF CALIFORNIA

In the Matter of:)
Application for Certification for the) **Docket No. 07-AFC-5**
Ivanpah Solar Electric)
Generating System)

STAFF RESPONSE TO APPLICANT'S PROPOSED SCHEDULE AND REQUEST FOR REVISED SCHEDULING ORDER

INTRODUCTION

On October 1, 2008, the applicant for the Ivanpah Solar Electric Generating System Project (ISEGS) filed a proposed schedule and a request for a revised scheduling order, or (alternatively) for a scheduling conference. The California Energy Commission committee responsible for this project subsequently issued an order for such a conference on October 15, 2008, and requested that parties respond to the applicant's proposed schedule by October 10, 2008.

During the past two weeks, Energy Commission staff has conferred with the staff of the U.S. Bureau of Land Management (BLM) to determine the feasibility of applicant's proposed schedule. Such collaboration is necessary because the Preliminary Staff Assessment (PSA) for this project is a "joint" document that will also serve as the Draft Environmental Impact Statement (DEIS) for BLM's right-of-way permit, the federal land permit required for ISEGS. The DEIS satisfies federal National Environmental Policy Act (NEPA) requirements for BLM's issuance of right-of-way permits. After conferring with BLM, Commission staff subsequently met with the applicant to discuss the proposed schedule.

APPLICANT'S PROPOSED SCHEDULE IS IMPRACTICAL

Prior to the outset of the ISEGS proceeding, BLM staff and Commission staff entered into a Memorandum of Understanding (MOU) regarding the joint production of the PSA/DEIS document and the Final Staff Assessment/DEIS document.[1] The goal of the MOU is to provide simultaneous and consistent state and federal environmental review.

[1] The MOU is titled "Memorandum of Understanding between the U.S. Department of the Interior, Bureau of Land Management California Desert District and the California Energy Commission Staff Concerning Joint Environmental Review for Solar Thermal Power Plant Projects." The MOU was signed by the Commission's Executive Director on August 8, 2007.

Proof of Service (Revised 7/14/08) filed with original.
Mailed from Sacramento on 10/10/08

Among other things, the MOU provides that the two agencies will share in the preparation and review of the environmental analyses for solar thermal projects, and indicates that the environmental documents will serve the dual purpose of satisfying both state California Environmental Quality Act (CEQA) and federal NEPA requirements. (MOU, p. 4.) According to the MOU attachment titled "BLM & CEC Combined Processing Plan," the PSA/DEIS is to be issued during the same approximate time frame as the circulation (for federal NEPA purposes) of the DEIS. In the federal NEPA process, BLM must submit the DEIS to a "Notice of Availability" (NOA) review process by the Department of Interior before the DEIS can be noticed in the Federal Register and publicly released. The NOA review period can take several weeks before the DEIS is publicly issued and BLM's 90-day comment period on the document can begin. During the NOA review period the PSA/DEIS will be virtually complete, but not yet publicly released. Thus, there can be little further schedule progress during this period.

Applicant's proposal is to try to avoid the delay caused by the NOA review period by having Commission staff publish the PSA first (apparently captioned solely as the PSA) and initiating Staff workshops on the document during the NOA review period. However, BLM staff believe that such a shortcut is inconsistent with the MOU and with the agencies' agreement to release the PSA/DEIS as a joint document that meets federal NEPA requirements, including NOA review and Federal Register publication before its release. BLM believes that participation in workshops on what is in essence the DEIS, but before the DEIS is officially reviewed and published for federal purposes, violates federal requirements. BLM and Commission staff believe that any departure from the joint document, including during the NOA review period, is contrary to the MOU, and BLM staff further believes that such departure would require BLM to produce its own separate DEIS.

If the applicant's proposal is accepted, and BLM proceeds to divorce itself from the Commission process and produce its own DEIS without the Commission's PSA, this breakdown of the joint document collaboration will add many months to the schedule for federal approval. Thus, in an effort to gain some weeks, applicant's proposed schedule risks significant project delay from the breakdown of the state/federal agency collaboration. For these reasons, Commission staff opposes applicant's proposed schedule.

Dated: October 10, 2008

Respectfully submitted,

RICHARD C. RATLIFF
Staff Counsel IV

BEFORE THE ENERGY RESOURCES CONSERVATION AND DEVELOPMENT COMMISSION OF THE STATE OF CALIFORNIA

APPLICATION FOR CERTIFICATION FOR THE ***IVANPAH SOLAR ELECTRIC GENERATING SYSTEM***	DOCKET NO. 07-AFC-5 **PROOF OF SERVICE** **(Revised 7/14/08)**

<u>INSTRUCTIONS:</u> All parties shall 1) send an original signed document plus 12 copies <u>OR</u> 2) mail one original signed copy AND e-mail the document to the web address below, AND 3) all parties shall also send a printed <u>OR</u> electronic copy of the documents that <u>shall include a proof of service declaration</u> to each of the individuals on the proof of service:

CALIFORNIA ENERGY COMMISSION
Attn: Docket No 07-AFC-5
1516 Ninth Street, MS-15
Sacramento, CA 95814-5512
docket@energy.state.ca.us

APPLICANT

Solar Partners, LLC
John Woolard, Chief Executive Officer
Alicia Torre, Project Manager
1999 Harrison Street, Suite #500
Oakland, CA 94612
ATorre@BrightSourceEnergy.com

***Steve De Young**
Ivanpah Solar Electric Generating System
Director, Environmental, Safety and Health
1999 Harrison Street, Ste. 2150
Oakland, CA 94612
sdeyoung@brightsourceenergy.com

APPLICANT'S CONSULTANTS

John L. Carrier, J. D.
2485 Natomas Park Dr. #600
Sacramento, CA 95833-2937
jcarrier@ch2m.com

COUNSEL FOR APPLICANT

Jeffrey Harris
Ellison, Schneider & Harris L.L.P.
Attorneys at Law
2015 H Street
Sacramento, CA 95814-3109
jdh@eslawfirm.com

INTERESTED AGENCIES

***California ISO**
P.O. Box 639014
Folsom, CA 95763-9014
e-recipient@caiso.com

Tom Hurshman, Project Manager
Bureau of Land Management
2465 South Townsend Ave.
Montrose, CO 81401
tom_hurshman@blm.gov

Sterling White, Field Manager
Bureau of Land Management
Needles Field Office
1303 South Highway 95
Needles, CA 92363
Sterling_White@blm.gov

Becky Jones
California Department of Fish & Game
36431 41st Street East
Palmdale, CA 93552
dfgpalm@adelphia.net

INTERVENORS

California Unions for Reliable Energy ("CURE")
Tanya A. Gulesserian
Marc D. Joseph
Adams Broadwell Joseph & Cardozo
601 Gateway Boulevard, Suite 1000
South San Francisco, CA 94080
tgulesserian@adamsbroadwell.com

ENERGY COMMISSION

JEFFREY D. BYRON
Commissioner and Presiding Member
jbyron@energy.state.ca.us

JAMES D. BOYD
Commissioner and Associate Member
jboyd@energy.state.ca.us

Paul Kramer
Hearing Officer
pkramer@energy.state.ca.us

Che McFarlin
Project Manager
Cmcfarli@energy.state.ca.us

Dick Ratliff
Staff Counsel
dratliff@energy.state.ca.us

Public Advisor
pao@energy.state.ca.us

DECLARATION OF SERVICE

I, Janet Preis, declare that on October 10, 2008, I deposited copies of the attached Staff Response to Applicant's Proposed Schedule and Request for Revised Scheduling Order in the United States mail at Sacramento with first-class postage thereon fully prepaid and addressed to those identified on the Proof of Service list above.

AND/OR

Transmission via electronic mail was consistent with the requirements of California Code of Regulations, title 20, sections 1209, 1209.5, and 1210. All electronic copies were sent to all those identified on the Proof of Service list above.

I declare under penalty of perjury that the foregoing is true and correct.

Janet Preis
Legal Secretary

MYCARE
MYCology Advocacy, Research, & Education

HOME FUNGAL DISEASE EDUCATION RESEARCH/SCHOLARSHIP CLINICAL TRIALS ABOUT NEWS/PRESS PATI

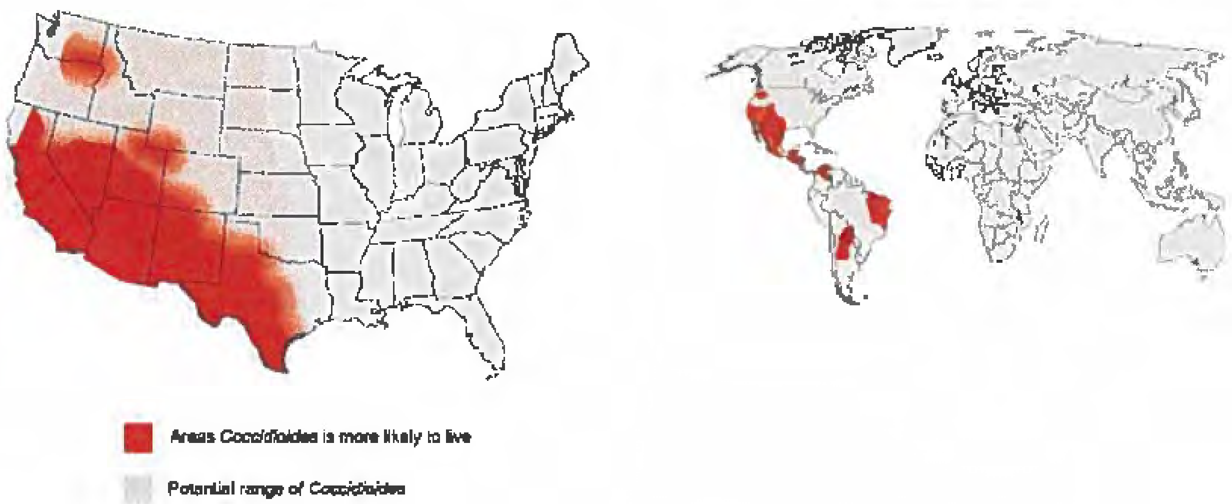
Cocci is a fungus present in certain areas of the Western Hemisphere. You can get infected with Cocci if you live or travel to these areas. Read on to learn
infection or the fungus that causes it.

WHAT IS IT?

Overview

Coccidioidomycosis, (pronounced kok-SID-ee-oy-doh-my-KOH-sis), commonly called Valley Fever or Cocci, is a fungal disease caused by the *Coccidioides* (p *oy-deez*) fungus. The Cocci fungus is **endemic** in the Southwest United States and the arid areas of Eastern Washington State, as shown in Figure 1. Outsid this fungus is present in parts of Mexico and South and Central America. The areas where Cocci can be found seem to be expanding, potentially due to ch temperature associated with climate change. Periods of drought can also increase the number of Cocci infections in the endemic region.

Figure 1. Areas where *Coccidioides* fungi can be found. Reproduced with permission from CDC and MSGERC.



Cocci is a growing problem. In 2019, there were 20,003 confirmed cases of Valley Fever reported in the United States. Most cases are diagnosed in people Arizona or California. The number of cases is believed to be drastically underestimated because many people affected by the disease aren't diagnosed. Mc experience Cocci will recover, but unlike symptoms associated with more common respiratory illnesses, the symptoms associated with Cocci may last six r average, approximately 200 deaths a year are attributed to Cocci in the United States. This is also likely underreported. The economic impact of the diseas more than 1.5 billion US dollars. For these reasons, improving awareness, diagnosis, and the management of Cocci will have a large public health impact.

Key Term:

Endemic means regularly occurring within an area or community.

What's in a Name?

You may hear Cocci referred to in several ways. For simplicity, on first mention we will use the technical term, then through the rest of the document we w Cocci. Here are some of the other terms you may encounter that describe the disease or the fungus, along with a brief explanation of how those terms ar

- **"Valley Fever"**—this is the term typically used to describe the fungal lung infection caused by Cocci. Sometimes it's used to describe other forms of th name because the first cases described in the United States occurred in in the San Joaquin Valley of California
- **"San Joaquin Valley Fever"**—another term that shows a geographical association
- **"Desert rheumatism"**—this name came about because Cocci is often present in the desert region. Rheumatism is added to "desert" because people joints from the fungus because of their body's reaction to the fungus (inflammation)
- **"Cocci"**—this term is just a lot shorter and easier to say than *Coccidioides* or coccidioidomycosis
- **"Disseminated cocci"** or **"Disseminated Valley Fever"**—this term means that the disease has spread from the primary location (usually the lung) to
- **"Cocci Meningitis," "Valley Fever Meningitis,"** or **"CNS Cocci"** - these terms refer to disease that has invaded the brain or central nervous system

Science Sidebar:

Why is coccidioidomycosis such a long, difficult name? There is a reason. The people who first described Cocci saw a large structure in the tissue and th **protozoan** parasite similar to *Coccidia*. That's where the confusion arose and how the term *Coccidioides* (resembling *Coccidia*) arose. Later on, research were looking at the spherule phase of a fungal organism (more on that in the other Science Sidebar). In your reading, you may come across the term cc the gastrointestinal illness caused by *Coccidia*. It typically occurs in livestock. Don't get confused—these are different diseases. People with Valley Fever around chickens or other livestock.

Key Term:

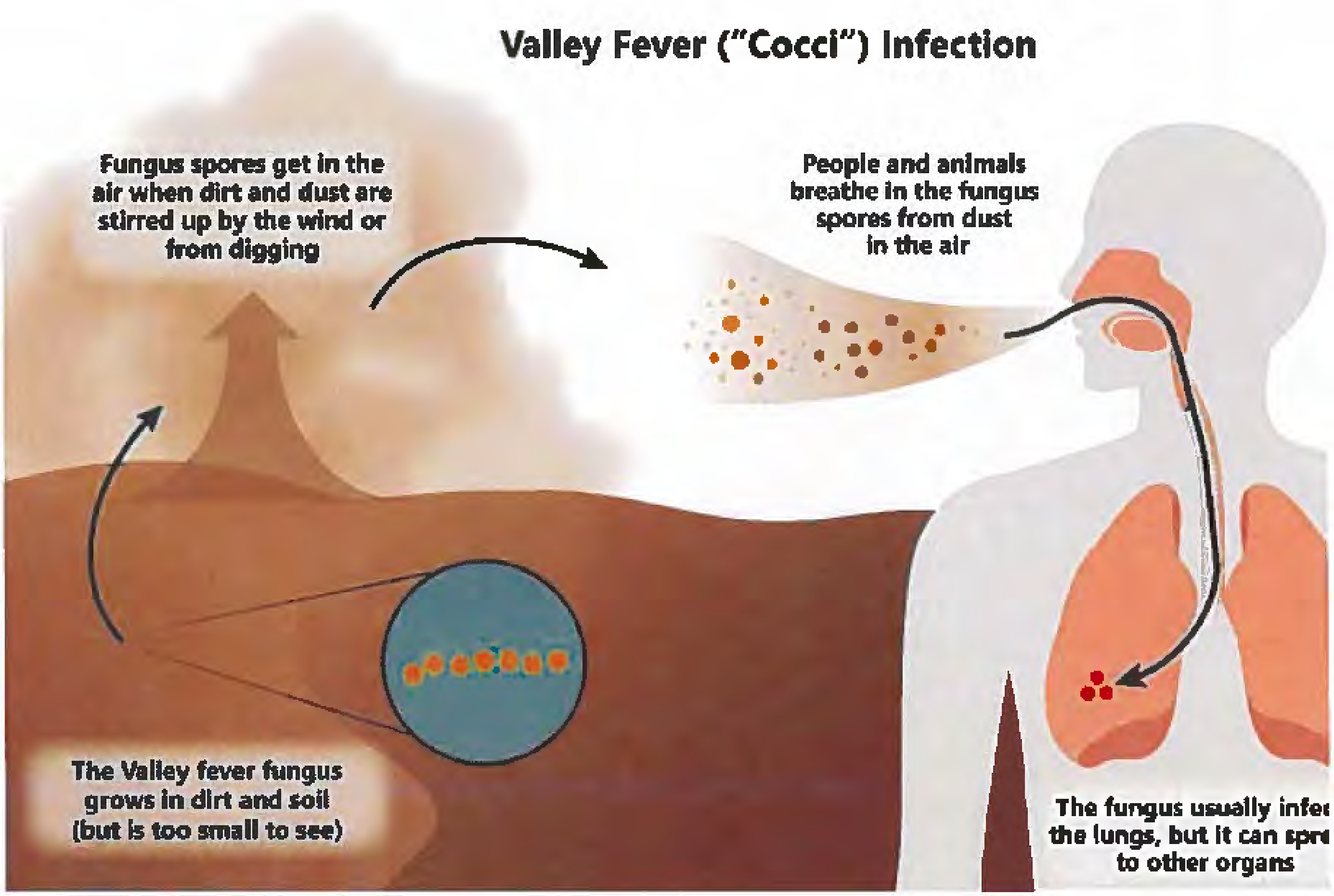
Protozoan is a single-cell animal of the family *Protista*, which includes the amoeba.

How Do You Get Infected?

From airborne spores

Coccidioides fungi live in the soil in what is called the mycelial phase (see SCIENCE SIDEBAR for detail) in environments that are dry (arid or semiarid). As sh the soil is disturbed—by human activities like excavation, animal activities, or weather events like earthquakes or fires—fungal arthrospores (we'll call ther are released into the air. These spores are not visible to the naked eye, and once in the air can travel 75 miles or more from where they became airborne. can inhale these spores. Once inside the body, the fungus changes to a different form and grows into spherules, which eventually release new fungal cells process continues. This is the lifecycle of the fungus and the typical method of disease transmission.

Figure 2. Lifecycle of Cocci, showing the phases in the soil and in the mammalian host. Reproduced with permission Valley Fever Fact Sheet.



Through your skin

Less commonly, fungal spores can enter your skin through a cut, wound, or splinter and cause a skin infection. This is known as an implantation mycoses. transmission is rare but has been documented in fieldworkers or people with traumatic injuries.

Frequently Asked Questions

Can Cocci be spread from person to person?

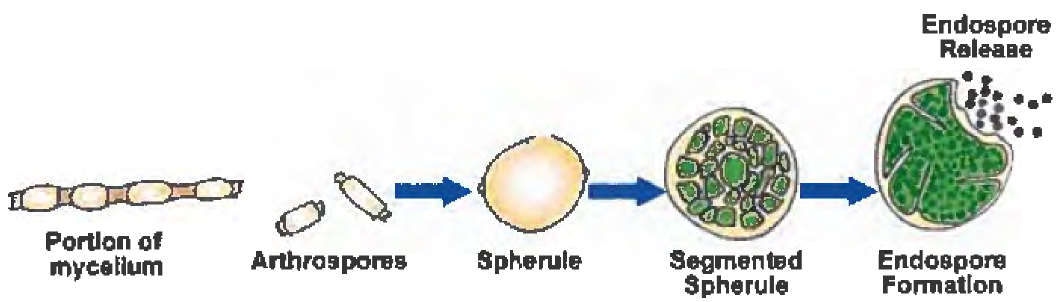
Key Terms:

Genus is a category of an organism that is "above" a species and "below" a family in the organism taxonomic (organizational) system.

Mycelium is a root-like fungal structure with branching, thread-like filaments called hyphae.

Arthrospores are yeast-like individual fungal cells formed by the breaking down of hyphae.

Figure 3: Forms of Cocci during the different lifecycle stages. Modified from image provided by Saurabh Patil, via Wikimedia Commons.



Science Sidebar:

Coccidioides – the Shape Shifter

Coccidioides is a fungal **genus**. A genus is usually italicized and written with an initial capital letter. It is often written in connection with a specific specie lowercase and is also italicized. Sometimes the genus is abbreviated to a 1 letter (eg. *C.* for *Coccidioides*). Can you recognize this genus/species: *Homo sa* The species of *Coccidioides* that cause disease are *Coccidioides immitis* and *Coccidioides posadasii*.

Coccidioides is described as a dimorphic fungus, which means it exists in two distinct forms as shown in Figure 3. It can exist as a **mycelium** (mold) cont typically in the soil. When the soil dries out, hyphae develop into **arthrospores**. These arthrospores (spores) are small enough to become airborne and inhaled. Once a spore enters the lung, it undergoes a change to become a spherule, which is a big, sturdy sack-like structure. Inside the spherule, the f become endospores, and fill up the sack. When the spherules rupture in a few weeks, the endospores are released and spread out (disseminate). They spherules, and the cycle continues

MyCARE Foundation is a nonprofit, tax-exempt charitable organization (tax ID number 99-0942343) under Section 501(c)(3) of the Internal Revenue Code. I deductible as allowed by law.

MyCARE has received stakeholder and industry support from Associates of Cape Cod, Inc., Astellas Pharma US; F2G; IMMY; Melinta Therapeutics, Inc; Mira Mundipharma; and the Mycoses Study Group Education & Research Consortium.

About MyCare Contact Do

kvpr | npr Network Donate

New Valley Fever Skin Test Shows Promise, But Obstacles Remain

KVPR | By Kerry Klein

Published November 21, 2016 at 12:51 PM PST

LISTEN • 5:55

Kerry Klein/KVPR

Eleven-year-old Faith Herrod, recovering from valley fever, plays with Moses, Ninja and her other four pets when she has the energy for it.

We continue our reporting this week on the fungal disease known as valley fever with a

KVPR
1A

and a new skin test could be used to screen for that immunity—but that's only if the test overcomes some major hurdles.

Faith Herrod wants to be a veterinarian when she grows up. The 11-year-old lives in the small Central California town of Lemoore with her family, three dogs and three cats. Someday, she'll get a rabbit, too—as soon as her mother lets her.

In her free time, Faith should be out playing with her dogs. But for almost a year, she was not able to do so. She would come home from school at 4 p.m. and go right to bed. That's because last October the sixth-grader was diagnosed with valley fever, a fungal infection that kept her out of school for months and left her with regular headaches and chronic pain.

"Sometimes, when you get super tired, it'll feel like your ribs will go in," she said, wrapping her arms around her stomach. "It'll feel like your ribs go in and hurt really bad."

Faith contracted valley fever by breathing in fungal spores carried by the wind. That's how the disease is contracted, and it can happen at any time. Faith's mother, Caren Herrod, isn't entirely sure when it happened, but her best guess is while they were doing yard work one day—something they had done dozens of times before. Herrod never imaged that, after so much time, Faith would not have built up natural immunity and that she could still be at risk.

"If I had known that she was susceptible, it would've been different," Herrod said. "We would've done things differently."

As it turns out, Faith and her mother could have known. A new skin test called Spherusol can detect whether a person has developed natural immunity, meaning they've overcome valley fever before. Because most valley fever cases are asymptomatic, many people whose immune systems have battled the disease may never know it.

Advocates are excited about the test. So are doctors — like Dr. John Galgiani, director of the University of Arizona's Valley Fever Center for Excellence. He dreams of seeing Spherusol being used as a tool to screen for past infections.

"I think that Spherusol's best use will be in primary care doctors' offices, to test their

Galgiani said. If patients knew they had never conquered valley fever, they could better prepare themselves against it; and doctors might be more likely to diagnose the disease if patients showed unusual symptoms.

Experts call for a change in FDA rules on promising test

But despite its promise, the test isn't in wide use.

"Frankly, I don't use it very often myself," Galgiani said. "Even specialists don't use it very often."

That's because the Food and Drug Administration hasn't approved Spherusol for testing immunity. Instead, the test is supposed to be used by clinicians only after a person has been diagnosed with the disease. Galgiani and others would like to see the FDA change the rules to allow its use whenever a clinician thinks a test is warranted.

"If the labeling is changed to allow the test to be used to test for prior infection, then it opens up a whole different value of the test to the clinical community," Galgiani said.

Valley fever lurks in dirt and dust in the desert throughout the Southwest. Most people who inhale the spores fight off the disease without ever knowing they had it. Some develop flu- or pneumonia-like symptoms. In rare cases, it can cause severe lung infection or disseminate throughout the body, requiring lifelong treatment or leading to fatal meningitis.

But there is some good news.

"Once you've had valley fever and gotten over it, you are for all practical purposes immune from a second infection," Galgiani said.

Credit Nielsen Biosciences

Spherusol comes in vials of 10 doses. A positive result indicates a previous bout with valley fever and an acquired immunity to the disease.

Revealing a disease exposure with a skin test

That's why Spherusol could have such an impact: It could reveal a person's history of exposure. Before, that was something most people could only wonder about.

Spherusol works similarly to a tuberculosis skin test. A clinician injects a small amount of spores under the skin, and the reaction indicates whether immunity has developed. Similar skin tests for valley fever were first developed in the 1930s, but the most recent iteration was discontinued in the 1990s after the company that produced it was unable to turn a profit.

Spherusol was released in 2014 and costs about $62. No studies have directly compared its efficacy to previous tests.

Right now, Spherusol is only indicated for use after a patient has already been diagnosed with the disease. So, instead of being used as a way to determine valley fever exposure, it's considered a sort of immune system checkup for those recovering from the disease.

"It's indicated for understanding how their body is reacting to the disease," said Tom Carpenter, president and CEO of Nielsen Biosciences, the San Diego-based company that developed Spherusol. "Is their immune system engaged? Or is their immune system overwhelmed? Or are they potentially immune-compromised and not even able to respond to the infection?"

Carpenter says that screening patients could be a great way to use the test, but getting the FDA to approve a change in the labeling could take years. It would involve new clinical trials and potentially millions of dollars of investment. He says his company is looking into it.

In the meantime, however, he points out that doctors are already allowed to use Spherusol for off-label uses.

"Health care providers right now have the ability to make a medical judgment on how best to use the skin test," Carpenter said. "So, it's certainly not preventing them from making that use, but we can't speak to that use."

But using Spherusol off-label has challenges —like its price tag. A single test may not break a budget, but regular valley fever screening would mean periodic trips to the doctor to pay $62 for a prick under the skin. Most health insurers in California, including Medi-Cal, say they cover it — but only for its prescribed use. Even then, some insurers could reimburse as little as $4. By contrast, a patient getting a flu shot would likely pay nothing, and the clinic would be reimbursed up to around $35.

And then there's the fact that the test is packaged in bulk. Spherusol can only be bought in vials of 10 doses; once the vial is open, the countdown to expiration begins. Dr. Royce Johnson, chief of infectious disease at Kern Medical in Bakersfield, says the test is "extraordinarily stable." Even so, he said, many pharmacies have policies against storing products more than 30 days after they've been opened. "If you don't use it all, it costs a lot of money," Johnson said. "So there's some resistance to stocking it."

Another problem? Spherusol is only approved for 18- to 64-year-olds, even though children can be hit hard by valley fever and the CDC says people over 60 are most at risk.

Prisoners benefit from new test

Despite all of its obstacles, the test has been used to screen one very large patient group: California state prisoners, where it appears to be helping to prevent the disease immunity to valley fever, are not sent to serve their terms in the two Central Valley prisons hit hardest by the disease.

Caren Herrod may wish that her daughter Faith had access to Spherusol a year ago, but she admits that still wouldn't have solved the root of the problem: that she didn't take valley fever seriously enough because so little information about it is available.

"It's so, so limited," Herrod said. "With a disease that is so prevalent, for there to be so little information, it's sad. It's very sad."

Meanwhile, Faith is back at school full-time, and hopes she can soon take her dogs Moses and Jasmine out for a walk.

This project results from a new venture – the Center for Health Journalism Collaborative – which currently involves the Bakersfield Californian, Radio Bilingüe in Fresno, Valley Public Radio in Fresno and Bakersfield, Vida en el Valle in Fresno, the Voice of OC in Santa Ana, the Arizona Daily Star in Tucson, La Estrella de Tucsón and CenterforHealthJournalism.org. The collaborative is an initiative of the Center for Health Journalism at the University of Southern California's Annenberg School for Communication and Journalism.

Next in this series: Accurate valley fever counts elude health officials

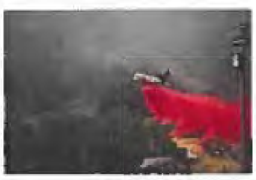
Tags Health | disease | valley fever | lemoore | health | centers for disease control and prevention | valley fever center for excellence

Kerry Klein

Kerry Klein is an award-winning reporter whose coverage of public health, air pollution, drinking water access and wildfires in the San Joaquin Valley has been featured on NPR, KQED, Science Friday and Kaiser Health News. Her work has earned numerous regional Edward R. Murrow and Golden Mike Awards and has been recognized by the Association of Health Care Journalists and Society of Environmental Journalists. Her podcast Escape From Mammoth Pool was named a podcast "listeners couldn't get enough of in 2021" by the radio aggregator NPR One.

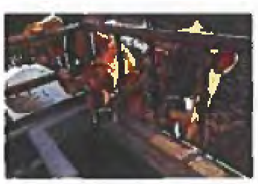
See stories by Kerry Klein

Latest News



Carbon emissions from forest fires are surging globally, UC Merced professors find

Kern County greenlights plan to build the state's first carbon storage and removal facility



Merced County confirms first human case of bird flu this year

Soria, Rose clash in election forum for key Central Valley seat in state assembly

KVPR
1A

KVPR
1A

KVPR
1A

Related Content

Health

New Legislation Would Allocate $1 Million For Valley Fever Research

Diana Aguilera, May 7, 2015

The fight against valley fever may reach a new milestone. A bill in the state legislature would fund research for this disease in hopes of finding a cure....

LISTEN • 1:17

KVPR
1A

Health

Valley Fever Clinical Trial To Begin At Kern Medical

October 14, 2016

Bakersfield's Kern Medical has selected to work with the National Institutes of Health and the Centers for Disease Control on a new clinical trial for...

KVPR
1A

Health

Forecasting An Epidemic: Does Weather Hold The Key To Predicting Valley Fever Outbreaks?

November 13, 2016

When a punishing drought besieged California in the late 1980s, relief came with 30 days of rain in 1991 — dubbed the March Miracle because of how it...

KVPR
1A

Health

Collaborative Launches New Reporting Series On Valley Fever

Joe Moore, November 15, 2016

Valley fever has long been a major health concern for people who live in the San Joaquin Valley. A fungus that grows in the soil can become airborne. If...

LISTEN • 13:06

KVPR
1A

Health

CDC Technology Advances Promise Better Valley Fever Detection

November 21, 2016

New technology could reveal the microscopic, sometimes deadly spores that cause valley fever that currently float in the air undetected.The U.S. Centers...

KVPR
1A

Health

My Valley, My Story: Valley Fever After Decades Of Dust

Kerry Klein, November 22, 2016

The fungal disease can afflict individuals of any age and ethnic group—even those who have lived and worked in the valley for decades. As part of our...

LISTEN • 2:21

Stay Connected

KVPR
1A

Programs

Schedule

Privacy

Contact

Staff

Support

Business Sponsorships

FCC Applications

KPRX Public File

KVPR Public File

Contest Rules

KVPR
1A

State of California – Natural Resources Agency
DEPARTMENT OF FISH AND WILDLIFE
Central Region
1234 East Shaw Avenue
Fresno, California 93710
(559) 243-4005
www.wildlife.ca.gov

GAVIN NEWSOM, Governor
CHARLTON H. BONHAM, Director

May 7, 2021

Terrance Smalls
Kern County Planning and Natural Resources Department
2700 "M" Street Suite 100
Bakersfield CA, 93301

Subject: Raceway 2.0 Solar, by sPower Development Corporation, LLC (Project) Draft Environmental Impact Report (DEIR) SCH No.: 2020079007

Dear Mr. Smalls:

The California Department of Fish and Wildlife (CDFW) received a DEIR from the Kern County Planning and Natural Resources Department for the above-referenced Project pursuant to the California Environmental Quality Act (CEQA) and CEQA Guidelines.[1]

Thank you for the opportunity to provide comments and recommendations regarding those activities involved in the Project that may affect California fish and wildlife. Likewise, CDFW appreciates the opportunity to provide comments regarding those aspects of the Project that CDFW, by law, may be required to carry out or approve through the exercise of its own regulatory authority under Fish and Game Code. 2-A

CDFW ROLE

CDFW is California's **Trustee Agency** for fish and wildlife resources and holds those resources in trust by statue for all the people of the State (Fish & G. Code, §§ 711.7, subd. (a) & 1802; Pub. Resources Code, § 21070; CEQA Guidelines § 15386, subd. (a)). CDFW, in its trustee capacity, has jurisdiction over the conservation, protection, and management of fish, wildlife, native plants, and habitat necessary for biologically sustainable populations of those species (*Id.*, § 1802). Similarly, for purposes of CEQA, CDFW is charged by law to provide, as available, biological expertise during public agency environmental review efforts, focusing specifically on projects and related activities that have the potential to adversely affect fish and wildlife resources. 2-B

[1] CEQA is codified in the California Public Resources Code in section 21000 et seq. The "CEQA Guidelines" are found in Title 14 of the California Code of Regulations, commencing with section 15000.

CDFW is also submitting comments as a **Responsible Agency** under CEQA (Pub. Resources Code, § 21069; CEQA Guidelines, § 15381). CDFW expects that it may need to exercise regulatory authority as provided by the Fish and Game Code. As proposed, for example, the Project may be subject to CDFW's lake and streambed alteration regulatory authority (Fish & G. Code, § 1600 et seq.). v Likewise, to the extent implementation of the Project as proposed may result in "take" as defined by State law of any species protected under the California Endangered Species Act (CESA) (Fish & G. Code, § 2050 et seq.), related authorization as provided by the Fish and Game Code may be required. 2-C

Nesting Birds: CDFW has jurisdiction over actions with potential to result in the disturbance or destruction of active nest sites or the unauthorized take of birds. Fish and Game Code sections that protect birds, their eggs and nests include, sections 3503 (regarding unlawful take, possession or needless destruction of the nest or eggs of any bird), 3503.5 (regarding the take, possession or destruction of any birds-of-prey or their nests or eggs), and 3513 (regarding unlawful take of any migratory nongame bird).

PROJECT DESCRIPTION SUMMARY

Proponent: sPower Development Company, LLC

Project Description: The proposed project would involve construction and operation of two solar photovoltaic (PV) power-generating facilities, on six discontinuous sites, which would produce a combine total of approximately 291 megawatts (MW) of renewable electricity with up to 291 megawatt hours (MWh) energy storage on 1,330 acres of land in unincorporated Kern County. 2-D

Location: The proposed project is in the western extent of the Mojave Desert near Rosamond, California between Rosamond Boulevard and Avenue A, and between 70th Street West and 90th Street West in Sections: 20, 21, 28, 29 and 32, T9N/R13W in the eastern portion of unincorporated Kern County, California.

COMMENTS AND RECOMMENDATIONS

CDFW is concerned regarding potential impacts to special-status species including, but not limited to, the State and Federally threatened desert tortoise (*Gopherus agassizii*); the State threatened Swainson's Hawk (*Buteo swainsonii*); the State candidate for listing under CESA western Joshua tree (*Yucca brevifolia*); and the State species of special concern burrowing owl (*Athene cunicularia*). Mitigation Measures for these species, as proposed in the DEIR, may not reduce impacts to less than significant or result in unauthorized take. Our specific comments follow. 2-E

COMMENT 1: Western Joshua Tree (Joshua tree)

Table 4.4-3 states that no Joshua trees were observed on the Project site, but the species does occur within the gen-tie route. In addition, Table 4.4-1 lists 2.01 acres of Joshua tree woodland, a CDFW sensitive plant community, occurring within the gen-tie route. The DEIR cites measures from the Willow Springs Specific Plan that are applicable to Joshua tree. These measures include:

- Measure 15: Where possible, project development within the Specific Plan Update area shall be designed to avoid displacement of destruction of Joshua tree habitat, to the satisfaction of the Kern County Agricultural Commissioner's Office. Areas adjacent to the woodland shall have a 50-foot setback from the Joshua tree plants. Within that setback, a native plant cover should be restored to natural habitat values to serve as a buffer, if such plant cover is not present.

- Measure 16: A Joshua Tree Preservation and Transportation Plan shall be developed by the applicants for each parcel where Joshua trees are located on site. The plan shall be submitted to the Kern County Agricultural Commissioner's office for review and approval prior to grading permit issuance.

- Measure 23: A Joshua Tree Preservation and/or Transplantation Plan shall be developed by applicants of discretionary projects for each parcel where Joshua trees are located on site. The plan shall be submitted to the Kern County Agricultural Commissioner for review and approval prior to grading permit issuance.

2-E (con.)

As noted above, Joshua tree is currently a candidate for listing pursuant to CESA. Candidate species are protected as if they were listed as a threatened or endangered species under CESA. Measures 16 and 23 would require take authorization from CDFW to relocate individual Joshua trees in order to comply with CESA. CESA applies to every life stage of a listed species, and for Joshua tree, this would include the seed bank. The 50-foot no-disturbance buffer from individual Joshua trees required in Measure 15 of the Willow Springs Specific Plan, as well as the 25-foot buffer listed in DEIR Mitigation Measure 4.4.-12 for special status plants, is likely insufficient to avoid impacts to the seedbank. Vander Wall et. al. 2006 documented 290 feet as maximum distance of seeds dispersed carried by rodents. A 290-foot buffer is warranted to not only avoid impacts to individual trees, but potential impacts to the seed bank as well. CDFW recommends the following edits to the DEIR.

Recommended Mitigation Measure 1: Western Joshua Tree Avoidance

CDFW recommends a no-disturbance buffer for individual western Joshua trees of 290 feet. If buffers cannot be maintained, then consultation with CDFW is warranted to determine appropriate minimization and mitigation measures for impacts to special-status plant species. 2-F

Recommended Mitigation Measure 2: Western Joshua Tree Take Authorization

If a 290-foot buffer for Joshua tree is not feasible, consultation with CDFW is warranted to determine if the Project can avoid take of that species. If take cannot be avoided, take 2-G

Terrence Smalls
Kern County Planning and Natural Resources Department
May 7, 2021
Page 4

authorization would need to occur through acquisition of an Incidental Take Permit (ITP) from CDFW to comply with CESA and/or Fish and Game Code section 1900 and California Code of Regulations, title 14, section 786.9, subdivision (b). If Joshua trees will be translocated to comply with the Willow Springs Specific Plan, acquisition of an ITP is necessary to comply with CESA. 2-G (con.)

COMMENT 2: Desert Tortoise

The DEIR (e.g., Table 4.4-3) states that the potential for desert tortoise to occur on-site is low because there is a lack of suitable habitat. The DEIR defines desert tortoise habitat as alluvial fans, washes, canyon bottoms, and hillsides. Desert tortoise have been observed in other habitat type and the lack of their presence does not preclude desert tortoise from occurring with the Project site. Based on the information provided, CDFW cannot conclude that desert tortoise is absent from the Project site. CDFW recommends the following edits to the DEIR. 2-H

Recommended Mitigation Measure 3: Desert Tortoise Protocol Surveys

CDFW recommends surveys for desert tortoise be conducted by a qualified wildlife biologist who understands and will follow the pre-project survey protocol as outlined in "Preparing for any action that may occur within the range of the Mojave Desert tortoise (*Gopherus agassizii*)" (USFWS, 2010) and has previous experience surveying for desert tortoise. Survey results are advised to be submitted to both CDFW and the USFWS. 2-I

Recommended Mitigation Measure 4: Desert Tortoise Take Authorization

If desert tortoise is found within the Project area during surveys advised in Recommended Mitigation 3 above, DEIR Mitigation Measure 4.4-4: Preconstruction Clearance Surveys, or construction activities, consultation with CDFW is advised to discuss how to implement the Project and avoid take; or if avoidance is not feasible, to acquire an ITP pursuant to Fish and Game Code section 2081 subdivision (b) prior to any vegetation- or ground-disturbing activities. Any take of desert tortoise without take authorization would be a violation of Fish and Game Code section 2080. 2-J

COMMENT 3: Swainson's Hawk (SWHA)

Table 4.4.-3 of the DEIR states that one active SWHA nest was observed on the Project site during 2020 burrowing owl surveys. The DEIR states that additional active nests occur within 5 miles of the Project site and that suitable foraging habitat occurs throughout the Project site. Several measures are provided as part of DEIR Mitigation Measure 4.4.-8: Swainson's Hawk Mitigation and Monitoring Plan. DEIR Mitigation Measure 4.4-8b requires "no new disturbances, habitat conversions, or other project-related activities that may cause nest abandonment or forced fledgling occur within 0.5 miles of an active nest between March 1 and September 15. Buffer zones 2-K

may be adjusted in consultation with CDFW and the County." However, these Project activities are not defined and CDFW advises that the 0.5-mile buffer apply to all Project-related activities to avoid unauthorized take. We acknowledge that not all Project-related activities may require a 0.5-mile buffer, but the type of activity should be discussed as part of the consultation with CDFW and the County for a reduced buffer.

DEIR Mitigation Measure 4.4-8e requires habitat management (HM) lands to mitigate the loss of Swainson's hawk foraging habitat by providing "HM lands within the Antelope Valley Swainson's hawk breeding range at a minimum 1:1 ratio for such habitat impacted within a 5-mile radius of active Swainson's hawk nest(s)." The Swainson's Hawk Survey Protocols, Impact Avoidance, and Minimization Measures for Renewable Energy Projects in the Antelope Valley of Los Angeles and Kern Counties, California (CEC and CDFG 2010) recommends "mitigating loss of Swainson's hawk foraging habitat by providing HM lands within the Antelope Valley Swainson's hawk breeding range at a minimum 2:1 ratio for such habitat impacted within a five-mile radius of active Swainson's hawk nest(s)." Based on the information provided in the DEIR, CDFW cannot conclude that a ratio lower than 2:1 will reduce impacts to Swainson's hawk foraging habitat to less than significant. CDFW recommends the following edits to the DEIR. 2-K (con.)

Recommended Mitigation Measure 5: SWHA No Disturbance Buffer

CDFW recommends that the type of activities for the 0.5-mile buffer in DEIR Mitigation Measure 4.4-8b are changed from "project-related activities that may cause nest abandonment or forced fledgling" to all "all Project activities." CDFW recommends the nest buffer remain in place until the breeding season has ended or until a qualified biologist has determined that the birds have fledged and are no longer reliant upon the nest or parental care for survival. 2-L

Recommended Mitigation Measure 6: SWHA Foraging Habitat

CDFW recommends that the amount of foraging habitat mitigation required by DEIR Mitigation Measure 4.4-8e is increased from a minimum 1:1 ratio to a minimum 2:1 ration to reduce impacts to SWHA foraging habitat to less than significant based on Swainson's Hawk Survey Protocols, Impact Avoidance, and Minimization Measures for Renewable Energy Projects in the Antelope Valley of Los Angeles and Kern Counties, California (2010). 2-M

Recommended Mitigation Measure 7: SWHA Take Authorization

If an active SWHA nest is detected and a 0.5-mile no-disturbance buffer around the nest cannot feasibly be implemented, consultation with CDFW is warranted to discuss how to implement the project and avoid take. If take cannot be avoided, take authorization through the acquisition of an ITP, pursuant to Fish and Game Code section 2081 subdivision (b) is necessary to comply with CESA. 2-N

COMMENT 4: Burrowing Owl (BUOW)

DEIR Mitigation Measure 4.4-6: Preconstruction Burrowing Owl Surveys requires a buffer of no fewer than 100 meters (330 feet) from an active BUOW burrow during the breeding season (i.e., February 1 to August 31) and buffer of no fewer than 50 meters (165 feet) from a BUOW burrow during the non-breeding season. CDFW typically recommends greater no-disturbance buffers based on the "Staff Report on Burrowing Owl Mitigation" (CDFG 2012). No explanation was provided as why these reduced buffers are sufficient to avoid take of BUOW or nest failure. Therefore, CDFW recommends extending the BUOW no-disturbance buffers. CDFW recommends the following edits to the DEIR. 2-O

Recommended Mitigation Measure 8: BUOW Avoidance

CDFW recommends the no-disturbance buffers listed in DEIR Mitigation Measure 4.4-6 be expanded to the buffers recommended in the "Staff Report on Burrowing Owl Mitigation" and listed in the table below. Specifically, CDFW's Staff Report recommends that impacts to occupied burrows be avoided in accordance with the following table unless a qualified biologist approved by CDFW verifies through non-invasive methods that either: 1) the birds have not begun egg laying and incubation; or 2) that juveniles from the occupied burrows are foraging independently and are capable of independent survival. 2-P

Location	Time of Year	Level of Disturbance		
		Low	Med	High
Nesting sites	April 1-Aug 15	200 m*	500 m	500 m
Nesting sites	Aug 16-Oct 15	200 m	200 m	500 m
Nesting sites	Oct 16-Mar 31	50 m	100 m	500 m

* meters (m)

Editorial Comments and/or Suggestions

Comment 5: Nesting birds

CDFW generally encourages Project implementation at individual Project sites occur during the bird non-nesting season if suitable nesting bird habitat is present. However, if ground-disturbing activities must occur during the breeding season (February through mid-September), the Project's applicant is responsible for ensuring that implementation of the Project does not result in violation of the Migratory Bird Treaty Act or relevant Fish and Game Codes as referenced above. 2-Q

To evaluate Project-related impacts on nesting birds if suitable habitat is present, CDFW recommends that a qualified wildlife biologist conduct pre-activity surveys for active nests no more than 10 days prior to the start of ground disturbance to maximize the probability that nests that could potentially be impacted are detected. CDFW also recommends that surveys cover a sufficient area around the work site to identify nests and determine their status. A sufficient area means any area potentially affected by the Project. In addition to direct impacts (i.e. nest destruction), noise, vibration, and movement of workers or equipment could also affect nests. Prior to initiation of construction activities, CDFW recommends a qualified biologist conduct a survey to establish a behavioral baseline of all identified nests. Once construction begins, CDFW recommends a qualified biologist continuously monitor nests to detect behavioral changes resulting from the Project. If behavioral changes occur, CDFW recommends the work causing that change cease and CDFW consulted for additional avoidance and minimization measures.

If continuous monitoring of identified nests by a qualified wildlife biologist is not feasible, CDFW recommends a minimum no-disturbance buffer of 250 feet around active nests of non-listed bird species and a 500-foot no-disturbance buffer around active nests of non-listad raptors. These buffers are advised to remain in place until the breeding season has ended or until a qualified biologist has determined that the birds have fledged and are no longer reliant upon the nest or parental care for survival. Variance from these no disturbance buffers is possible when there is compelling biological or ecological reason to do so, such as when the construction area would be concealed from a nest site by topography. CDFW recommends that a qualified wildlife biologist advise and support any variance from these buffers and notify CDFW in advance of implementing a variance. 2-Q

ENVIRONMENTAL DATA

CEQA requires that information developed in environmental impact reports and negative declarations be incorporated into a database which may be used to make subsequent or supplemental environmental determinations (Pub. Resources Code, § 21003, subd. (e)). Accordingly, please report any special-status species and natural communities detected during Project surveys to the California Natural Diversity Database (CNDDB). The CNDDB field survey form can be found at the following link: https://www.wildlife.ca.gov/Data/CNDDB/Submitting-Data. The completed form can be mailed electronically to CNDDB at the following email address: CNDDB@wildlife.ca.gov. The types of information reported to CNDDB can be found at the following link: https://www.wildlife.ca.gov/Data/CNDDB/Plants-and-Animals. 2-R

FILING FEES

If it is determined that the Project has the potential to impact biological resources, an assessment of filing fees will be necessary. Fees are payable upon filing of the Notice of Determination by the Lead Agency and serve to help defray the cost of environmental review by CDFW. Payment of the fee is required for the underlying project approval to be operative, vested, and final (Cal. Code Regs, tit. 14, § 753.5; Fish & G. Code, § 711.4; Pub. Resources Code, § 21089). 2-S

CDFW appreciates the opportunity to comment on the Project to assist the Kern County Planning and Natural Resources Department in identifying and mitigating the Project's impacts on biological resources. 2-S (con.)

More information on survey and monitoring protocols for sensitive species can be found at CDFW's website (https://www.wildlife.ca.gov/Conservation/Survey-Protocols). If you have any questions, please contact Jaime Marquez, Environmental Scientist, at the address provided on this letterhead, by telephone at (559) 243-4014, extension 291, or by electronic mail at Jaime.Marquez@wildlife.ca.gov. 2-T

Sincerely,

DocuSigned by:
FA83F09FED8945A
Julie A. Vance
Regional Manager

cc: Office of Planning and Research, State Clearinghouse, Sacramento

Final
Environmental Impact Report

SCH# 2020079007

Volume 5

Chapter 7 Response to Comments

Raceway 2.0 Solar Project
By sPower Development Company

SPA 33, ZM 231; ZCC 154, ZM 231; CUP 116 ZM 231; SPA 34, ZM 231; SPA 35, ZM 231; ZCC 155, ZM 231; CUP 117, ZM 231; SPA 36, ZM 231; SPA 37, ZM 231; ZCC 156, ZM 231; CUP 118, ZM 231; SPA 38, ZM 231; CUP 119, ZM 231; CUP 4, ZM 231-20; SPA 39, ZM 231; SPA 3, ZM 231-20; SPA 5, ZM 231-21; SPA 5, ZM 231-28; ZCC 3, ZM 231-21; ZCC 3, ZM 231-28; CUP 3, ZM 231-21; CUP 7, ZM 231-28; SPA 6, ZM 231-21; SPA 6, ZM 231-28; SPA 7, ZM 231-21; ZCC 4, ZM 231-21; CUP 4, ZM 231-21;
Cancellation of a Williamson Act Contract 20-06;
Kern County Franchise Agreement

Kern County Planning and Natural Resources Department
2700 M Street, Suite 100
Bakersfield, CA 93301-2370
(661) 862-8600

Technical Assistance by:

Environmental Science Associates
626 Wilshire Boulevard, Suite 2200
Los Angeles, CA 90017
(213) 599-4300

June 2021

Comment Letter 1: California Department of Fish and Wildlife (September 11, 2014)

State of California – Natural Resources Agency EDMUND G. BROWN JR., Governor
DEPARTMENT OF FISH AND WILDLIFE CHARLTON H. BONHAM, Director
Central Region
1234 East Shaw Avenue
Fresno, California 93710
(559) 243-4005
www.wildlife.ca.gov

September 11, 2014

Matthew Hall
Kern County Planning Department
2700 M Street, Suite 100
Bakersfield, California 93301-2323

Subject: Revised Draft Environmental Impact Report (RDEIR) for the Rosamond Solar Array Project by First Solar, Inc. (SCH No. 2010031030)

Dear Mr. Hall:

The California Department of Fish and Wildlife (CDFW) has reviewed the RDEIR for the Rosamond Solar Array Project by First Solar, Inc. (Project). The Project consists of constructing and operating a 150 megawatt (MW) photovoltaic solar power plant on approximately 1,175 acres in the Antelope Valley, at the western edge of the Mojave Desert. The power plant would consist of solar panels, an electrical collection system, a substation, an operations and maintenance building, security gates, fencing, temporary dust control ponds during construction a temporary concrete batch plant during construction, and a three to five mile-long transmission line (gen-tie) to connect to the existing Whirlwind substation, which is operated by Southern California Edison. 1-A

The revisions made in the RDEIR did not address most of the comments in CDFW's draft EIR comment letter, dated January 28, 2014. Therefore, many of the comments in this letter are the same as in the January letter.

The California Environmental Quality Act (CEQA) Mandatory Findings of Significance require that the lead agency find that the Project may have a significant effect on the environment if the Project has the potential to "...cause a fish or wildlife population to drop below self-sustaining levels..." The Swainson's hawk (*Buteo swainsoni*) population in the Antelope Valley is small and likely to drop below self-sustaining levels if renewable energy development continues without adequate habitat conservation within the Antelope Valley. As written, the RDEIR does not require the developer of this Project to conserve any habitat in the Antelope Valley or implement any other measures that would effectively conserve Swainson's hawk and burrowing owl (*Athene cunicularia*) populations in the Antelope Valley. 1-B

Mitigation measures 4.4-4 and 4.4-6 through 4.4-8 are not based on the baseline survey results which were that the area supports abundant burrowing owls and Swainson's hawks, and instead defer the development of habitat compensation requirements to consultation with other agencies, based on less-comprehensive, future, pre-construction survey results. CDFW recommends that the County instead require quantifiable habitat compensation in the Environmental Impact Report (EIR) based on the known baseline survey results and Project 1-C

Conserving California's Wildlife Since 1870

Matthew Hall
September 11, 2014
Page 2

impacts, and require the pre-construction surveys primarily for avoiding direct "take" of wildlife during Project construction and to determine whether additional mitigation may be warranted. 1-C

In this letter, we recommend specific, quantifiable measures to minimize and offset the impacts to the Antelope Valley Swainson's hawk and burrowing owl populations based on the Project's baseline data in the RDEIR. Since there is potential that CDFW will be only a Trustee Agency on this Project, CDFW does not necessarily have a mechanism to require and enforce these measures; thus, we find that the RDEIR's method of deferring the development of habitat compensation measures to later discussions with CDFW is unlikely to mitigate the Project's biological impacts to less than significant levels as the RDEIR concludes. 1-D

Here is a summary of our comments and recommendations in this letter:

- Require 60 acres of desert tortoise (*Gopherus agassizii*) habitat compensation in perpetuity. 1-E
- Require 2:1 habitat compensation within the Antelope Valley for Swainson's hawk and burrowing owl for all permanent impacts, not just if Swainson's hawks are found within 0.5 mile during pre-construction surveys (the baseline surveys already found them nesting within 0.5 mile, and the foraging habitat that supports nesting extends greater than 0.5 mile from the nest sites). 1-F
- Require a 0.5-mile construction buffer from Swainson's hawk nests, and avoid converting suitable habitat within that buffer. 1-G
- Require transmission lines to be underground to minimize Swainson's hawk, burrowing owl, and other avian fatalities. 1-H
- Assume that all beavertail cacti within the Project limits are the State and federally endangered Bakersfield cactus (*Opuntia basilaris* var. *treleasei*) unless further genetic studies conclude otherwise. 1-I
- Prohibit the use of chain link fence or require that chain link fences are modified to prevent desert kit foxes (*Vulpes macrotis arsipus*) from placing their heads through the mesh openings and becoming ensnared. 1-J
- If dust palliatives are to be used, require the applicant to use only substances that are organic, non-petroleum, and non-salts, such as many of the plant-based products that are commercially available for fugitive dust control. 1-K

A discussion of CDFW's jurisdiction for this Project and specific comments on the RDEIR, follows.

CDFW Jurisdiction

CEQA and California Endangered Species Act (CESA) Authority: CDFW is a Trustee Agency for fish and wildlife resources with the responsibility under CEQA for commenting on projects that could impact fish and wildlife resources. In this role, CDFW is responsible for providing, as available, biological expertise to review and comment on environmental documents and impacts arising from project activities. Pursuant to Fish and Game Code §1802, 1-L

Matthew Hall
September 11, 2014
Page 3

CDFW has jurisdiction over the conservation, protection, and management of fish, wildlife, native plants, and habitat necessary for biologically sustainable populations of those species.

CDFW is a Responsible Agency when a subsequent permit or other type of discretionary approval is required from CDFW, such as an Incidental Take Permit (ITP), pursuant to CESA, or a Lake or Streambed Alteration Agreement (LSAA) issued under Fish and Game Code §§1600 *et seq.* If direct take cannot be avoided, then the Project would warrant an ITP for take of Bakersfield cactus, which is listed as endangered pursuant to both the federal Endangered Species Act (ESA) and CESA, the federal endangered and State threatened species desert tortoise, and the State threatened species Swainson's hawk. 1-L

CDFW's issuance of an ITP and/or an LSAA is also considered a "project" subject to CEQA (CEQA Guidelines §15378). CDFW typically relies on the Lead Agency's CEQA compliance to make findings pursuant to CEQA Guidelines §15091. For the Lead Agency's CEQA document to suffice for CESA ITP issuance, it must fully describe the potential Project-related impacts to State-listed species, analyze potential impacts of the entire Project including private and public land components, and commit to measures to avoid or minimize, and fully mitigate impacts to these resources. CDFW may not be able to issue an ITP by relying on an EIR containing a Statement of Overriding Considerations when impacts to a State-listed species are not mitigated to less than significant.

If the EIR completed for this Project does not include mitigation measures to reduce impacts to State-listed species to less than significant, or if it does not describe and analyze all components of the Project (including those proposed on federal land), CDFW may as a result need to act as a CEQA Lead Agency and complete a subsequent EIR to support issuance of an ITP. This could significantly delay ITP issuance and consequently Project implementation. 1-M

Specific Comments on the RDEIR

Mitigation Measure 4.4-4: This measure regarding pre-construction wildlife surveys begins with "Prior to construction, the Applicant may contact the United States Fish and Wildlife Service and California Department of Fish and Wildlife to determine if pre-construction surveys and exclusionary fencing are warranted given the low likelihood of occurrence of sensitive species." The pre-construction surveys are warranted. The baseline surveys for the Project documented burrowing owl sign (whitewash and pellets) at burrows within the proposed solar arrays and elsewhere surrounding the site. Thus, CDFW does not concur that there is a "low likelihood of occurrence of sensitive species" on this Project site. The already-documented presence is also sufficient evidence that burrowing owls are likely to be on and/or near the construction area and might be taken during construction. CDFW recommends the RDEIR require pre-construction surveys for burrowing owl to facilitate take avoidance measures. 1-N

Desert tortoises are known to occur in the native desert scrub communities that are contiguous with the 80 acres of desert scrub habitat in the northern solar array and along the gen-tie route. Because this Project would convert those 80 acres of creosote bush scrub to a solar array, and disturb ground in desert tortoise habitat along the gen-tie route, CDFW recommends pre-construction surveys for desert tortoise are included in the RDEIR as a mitigation measure to determine if desert tortoise occur in the Project area. In the event that desert tortoise is 1-O

Matthew Hall
September 11, 2014
Page 4

detected, consultation with CDFW would be warranted to discuss how to implement the Project and avoid take. If take can't be avoided, an ITP would be required to comply with CESA. Alternatively, the Project applicant can assume presence and apply for an ITP. 1-O

Swainson's hawks are also known to nest approximately 0.5 mile from the proposed solar arrays, as documented in the Project's baseline biological report. Given this proximity, the trees bordering the Project site have a relatively high potential to host nesting Swainson's hawks. CDFW recommends pre-construction surveys for Swainson's hawks to determine if the species is present within 0.5 mile during construction activities. Depending on the results of these surveys, consultation with CDFW may be warranted to determine if the Project can avoid take. If take can't be avoided, an ITP is necessary to comply with CESA. 1-P

Mitigation Measure 4.4-6: This measure regarding pre-construction surveys and exclusion fencing for desert tortoise also contains the word "may," and gives the Applicant an option to not perform pre-construction surveys or install exclusion fencing for desert tortoises. As discussed above, the proposed northern solar array areas contains 80 acres of native desert scrub that adjoins known, occupied desert tortoise habitat. We recommend that the County require the Applicant to complete pre-construction surveys to avoid take of tortoises within these 80 acres and to install tortoise exclusionary fence along the solar array construction area boundaries north of Rosamond Boulevard and along the gen-tie route. CDFW recommends exclusion fencing be installed only after pre-construction surveys are completed. If desert tortoise are observed within the exclusion fencing, CDFW may consider the individuals captured. The capture of any species listed under CESA would require an ITP, as capture (or attempt to do so) is defined as take under Fish and Game Code Section 86. 1-Q

Desert Tortoise: The 80 acres of the desert scrub that the Project would displace are part of a larger area of occupied desert tortoise habitat. CDFW recommends that the County require the applicant to conserve 80 acres of desert tortoise habitat in perpetuity to offset this habitat loss 1-R

Burrowing Owl: The baseline surveys found nine burrows within 150 meters (m) of the gen-tie routes and a 150 m buffer of the eastern solar array area. They found that all of the Project site was suitable habitat. It is important to note that the owls using burrows in the non-tilled areas are foraging in the tilled areas, and that burrowing owls do breed in active agricultural fields in the Antelope Valley (Wilkerson and Siegel, 2011).

This Project area is of relatively high value for conservation of burrowing owls. The baseline survey findings corroborate the finding of Wilkerson and Siegel (2011) that burrowing owls in the Mojave Desert are clustered in the agricultural and urban interface areas of the Antelope, Apple, and Lucerne valleys. 1-S

Measure 4.4-7, regarding burrowing owl, requires that "Should burrowing owls be found on-site, compensatory mitigation for lost breeding and/or wintering habitat shall be implemented..." The context of that statement is following a pre-construction survey. This measure ignores the baseline data that confirmed that the Project site is indeed burrowing owl habitat.
Since the site is already known to be burrowing owl habitat, we recommend clarifying that the determination of whether habitat compensation is required does not depend on the future pre-construction survey results. We recommend stating that the Applicant shall compensate for 1-T

Matthew Hall
September 11, 2014
Page 5

the already documented burrowing owl habitat loss and specifying a minimum acreage. That would also help to clarify that pre-construction surveys are for avoiding direct take of owls during construction and determining whether more owls may be present than were found in the partial surveys of 2010 or 2011. If pre-construction surveys find that the site supports more owls than have already been found, then additional habitat compensation should be required. 1-T

Measure 4.4-7 also does not prescribe any amount or location of habitat mitigation. The initial surveys were completed in 2010 and 2011; these baseline surveys provided enough information to form a mitigation plan in the RDEIR. We recommend including the burrowing owl habitat mitigation plan in the EIR and requiring that it be implemented prior to the habitat loss to minimize the temporal habitat impacts. For reference, burrowing owl kernel home ranges in desert agricultural settings have been estimated at 112 ± 45 acres (Rosenberg and Haley, 2004). Rosenberg and Haley also found that owls foraged mostly within 600 m of their burrows, which puts most of the eastern solar arrays well within core foraging range of the burrows that were documented in the 150 m buffer survey there. The number of owls found along the gen-tie routes and in the eastern array buffer suggests that additional burrowing owl burrows occur within 600 m of the Project. Burrowing owl habitat compensation can be accomplished within the area to be used for Swainson's hawk mitigation in the Antelope Valley. 1-U

Mitigation measure 4.4-7 also allows the applicant to combine or "stack" burrowing owl habitat compensation with agricultural mitigation that may occur through in-lieu fee programs (none of which are identified in the RDEIR) and/or agricultural lands conservation in the Central Valley. Further, there is no requirement in the measure that the agricultural lands support burrowing owls. Neither of these options would mitigate impacts to the Antelope Valley population of burrowing owls, and may not provide any benefit to burrowing owls since the measure does not require actually conserving habitat that supports burrowing owls 1-V

CDFW recommends requiring the burrowing owl habitat compensation to occur within the Antelope Valley, in areas that support burrowing owls and can be enhanced to support more burrowing owls.

Swainson's Hawk: Mitigation measure 4.4-8 requires Swainson's hawk habitat compensation only if Swainson's hawks are found within 0.5 mile of the Project during pre-construction surveys. CDFW does not concur that this approach would mitigate Swainson's hawk habitat impacts to a less than significant level. The baseline surveys for the Project already established that Swainson's hawks nest within 0.5 mile and that the project is within the core nesting and foraging area for the Antelope Valley Swainson's hawk population. Requiring habitat compensation only if they are found during the pre-construction survey ignores the baseline conditions. CDFW recommends pre-construction surveys to determine whether additional mitigation may be warranted based on changing circumstances and to detect new nest sites that need to be buffered from construction activities to avoid take, including nest failure. The measure as currently drafted is inconsistent with this approach and with CDFW's "Swainson's Hawk Survey Protocols, Impact Avoidance, and Minimization Measures for Renewable Energy Projects in the Antelope Valley of Los Angeles and Kern Counties, California" which requires habitat compensation for foraging habitat losses that occur within 5 miles of nest sites. 1-W
Also inconsistent with the existing guidance is the RDEIR's requirement to mitigate only those habitat impacts within 0.5 mile of nest sites, and to allow habitat compensation to occur anywhere within 10 miles of a nest. Habitat 10 miles from a nest is much less valuable to

Matthew Hall
September 11, 2014
Page 6

nesting Swainson's hawks, and may be used very little by the species, compared to habitat within 0.5 mile of a nest. The proposed measure would not provide habitat of equal or greater value. 1-W

Further, the measure allows in-lieu fees and/or habitat compensation within the Central Valley, which would not mitigate the impacts to the Antelope Valley population, and would likely contribute to the loss of this population. As discussed above for burrowing owl, this measure does not guarantee that any Swainson's hawk habitat would actually be conserved.

We could concur that Swainson's hawk habitat losses would be mitigated to less than significant levels if the County, as lead agency, required habitat mitigation based on the already-known habitat loss impacts. 1-X

Requiring that the mitigation occur within the Antelope Valley is important. The CEQA Mandatory Findings of Significance require that a lead agency find that the Project may have a significant effect on the environment if the Project has the potential to "...cause a fish or wildlife population to drop below self-sustaining levels..." The Swainson's hawk population in the Antelope Valley is small and likely to drop below self-sustaining levels if renewable energy development continues without adequate habitat conservation within the Antelope Valley. As written, the RDEIR does not require the developer of this Project to conserve any habitat in the Antelope Valley or implement any other measures that would effectively conserve Swainson's hawk and burrowing owl populations in the Antelope Valley.

Lastly, the RDEIR does not specify a construction buffer distance from Swainson's hawk nests. CDFW recommends maintaining a 0.5-mile buffer from any active Swainson's hawk nest during construction. CDFW also recommends avoiding development on suitable habitat within 0.5 mile of known nests, and maintaining those areas as suitable foraging habitat. 1-Y

These recommendations are consistent with the California Energy Commission's and CDFW's June 2, 2010, recommendations titled "Swainson's Hawk Survey Protocols, Impact Avoidance, and Minimization measures for Renewable Energy Projects in the Antelope Valley of Los Angeles and Kern Counties, California." We encourage the County to adopt the habitat compensation measures already outlined in that document. 1-Z

Bakersfield Cactus: Recent genetic analysis on *Opuntia basilaris* varieties suggests that all of the *O. basilaris* found on the Project site have the potential to be *O. b.* var. *treleasei*, which is the State- and federally listed variety. The morphological characteristics used to attempt to differentiate the two varieties here are no longer believed to be reliable. Thus, absent genetic analysis of the cacti on the Project sites, CDFW recommends assuming that all *O. basilaris* found on site belong to the listed population. We recommend disclosing in the RDEIR a potential for the Project to take Bakersfield cactus so that, if take cannot be avoided, the final EIR may be used as a basis for CDFW issuance of an ITP. 1-A2

Desert Kit Fox: The Project is within the known range of desert kit fox. CDFW recommends that the County require the applicant to prevent kit foxes from becoming ensnared in chain-link fence. A kit fox was found dead, ensnared in a chain link fence, at a facility on the nearby Alta Oak Creek Mojave project. This is a known issue for San Joaquin kit foxes (*Vulpes macrotis mutica*) within their range as well. The foxes attempt to go through a chain-link fence headfirst, 1-B2

Matthew Hall
September 11, 2014
Page 7

find that the mesh openings are too small, and then cannot pull their heads back through the fence. Individuals may learn to go through chain link when they are pups only to become ensnared when they grow larger. We recommend either not constructing chain link fence or modifying the chain link (e.g., installing plastic privacy slats) to prevent foxes from putting their heads through the mesh openings. Leaving a 3.5-inch to 5-inch opening between two crossbars near ground level to allow free movement for foxes is one solution to minimize the risk of fatalities from chain link fence and also the direct habitat loss that would result from an impermeable fence. Another solution used on other constructed solar projects is to install inverted deer fence (larger openings at the bottom) instead of chain link. Both of these designs have met safety and security requirements on utility-scale solar projects even larger in size than the proposed Project. 1-B2

Dust Palliatives: It has come to CDFW's attention that salts, surfactants, and other toxic compounds are being used to control dust on projects in the Project vicinity. Many are toxic to both plants and animals, can migrate from the site with runoff, and can degrade into other toxins. This Project is within endangered species habitat and habitat for other wildlife, and application of such materials may have unintended, adverse impacts to these species. CDFW recommends that if dust palliatives are to be used, the County require the applicant to use only substances that are organic, non-petroleum, and non-salts, such as many of the plant-based products that are commercially available for dust control. 1-C2

Overhead Transmission Lines: Collisions with overhead power lines are one of the most common causes of California condor mortality and injury. Power lines also kill high numbers of other bird species, particularly raptors, even when diverters are installed. CDFW recommends that all power lines to be constructed for this Project be placed underground to avoid take of burrowing owls, Swainson's hawks, and other birds. 1-D2

Thank you for the opportunity to provide input on the RDEIR for this renewable energy project. If you have any questions regarding these comments, please contact Dave Hacker, Senior Environmental Scientist (Specialist), at 3196 Higuera Street, Suite A, San Luis Obispo, California 93401, by telephone at (805) 594-6152, or by e-mail at David.Hacker@wildlife.ca.gov

Sincerely,

Dean Marston for

Jeffrey R. Single, Ph.D.
Regional Manager

cc: Jessica Rempel
United States Fish and Wildlife Service
2493 Portola Road, Suite B
Ventura, California 93003

ec: California Department of Fish and Wildlife
Julie Vance, Central Region
Dave Hacker, Central Region

Matthew Hall
September 9, 2014
Page 8

Literature Cited

Rosenberg, D. K., and K. L. Haley. 2004. The ecology of burrowing owls in the agroecosystem of the Imperial Valley, California. Studies in Avian Biology 27:120-135.

Wilkerson, R.L., and R. Siegel. Distribution and abundance of western burrowing owls (*Athene Cunicularia hypugaea*) in southeastern California. The Southwestern Naturalist 56(3):378-384

Volume III

Chapter 7
Response to Comments

SCH# 2010031030

Rosamond Solar Array Project
by First Solar, Inc.

Conditional Use Permit 25, Map 232
Specific Plan Amendment 14, Map 232
ZCC 31, Map 232

Kern County Planning and
Community Development Department
Bakersfield, California

September 2014

SFGATE Newsletters Sign in

NEWS

Lawsuits Threatened Over Shoreline Noise

By Carolyne Zinko, *Style Reporter*
Nov 28, 1996

Frustrated by Mountain View's refusal to respond to its concerns about noise from Shoreline Amphitheater, a group of Palo Alto residents is threatening to sue to muffle the roar.

Watch More

SFGATE

The Mountain View City Council rebuffed a request by Palo Alto to turn down the volume at the amphitheater, the source of noise that has led to more than 10,000 complaints from homeowners during the decade the theater has operated.

Council members on Tuesday said they did not want to lower the volume because they weren't convinced it would work, arguing that the inversion layer phenomenon along the bay front is what causes sound to bounce into the atmosphere and carry miles from its source.

They also said they didn't want to hurt the amphitheater's viability. It pays the city about $1.6 million annually in rent.

"It's terrible," Councilwoman Dena Bonnell said of the noise problem, made clear to her this summer when a Palo Alto resident phoned and made her listen to the applause and music audible in the caller's back yard. "But I don't want to try something that won't work and give residents false hope."

In response, a Palo Alto coalition called Abate Shoreline Amphitheater Noise is drawing up battle plans, vowing to take its complaints en masse to small claims court. Both the city of Mountain View and Bill Graham Presents, which operates Shoreline, could be named in the suits.

Palo Alto might assist with the small claims action, said City Attorney Ariel Calonne.

"I talked to some residents who were just coming unglued over the issue, and my sense is they're not about to take last night's decision without a fight," said the coalition's founder, Jim Lewis.

The coalition also plans to ask Santa Clara County District Attorney George Kennedy to file charges against Shoreline for disturbing the peace, and it hopes to persuade Palo Alto to make its noise ordinance more stringent so it will have more ammunition to fight Mountain View.

Citizens have used small claims actions to eradicate nuisances like crack houses in their neighborhoods and airport noise, among other problems. A judge could impose an injunction preventing Shoreline from operating or award damages to residents if the amphitheater were found to be a nuisance, Calonne said.

He expressed surprise at the Mountain View council's reaction to Palo Alto's proposal.

"Saying they shouldn't do anything for noise reduction unless it will completely solve the problem is like saying you're not going to research a cure for cancer because there's always going to be cancer," Calonne said.

Complaints about Shoreline are not unique: Such problems affect other outdoor amphitheaters across the state and the nation.

Sacramento recently filed suit to block construction of a new 23,000-seat arena next to the Cal Expo amphitheater, charging that its potential impact on air quality, traffic and noise has not been adequately reviewed. Noise from concerts at Cal Expo has plagued nearby residents for years.

Near Chicago, the New World Music Theatre in Tinley Park was fined $13,000 in 1993 for violating state noise pollution laws more than two dozen times.

And in Columbus, Ohio, the Polaris Amphitheater has been forced to hire a noise consultant to monitor sound during rock concerts to appease the complaints of citizens from neighboring Westerville, where the sound drifts. Weeknight concerts there have been limited to 10 p.m.

Nov 28, 1996

Carolyne Zinko

STYLE REPORTER

Carolyne Zinko, a native of Wisconsin, joined The San Francisco Chronicle in 1993 as a news reporter covering Peninsula crime, city government and political races. She worked as the paper's society columnist from 2000 to 2004, when she wrote about the lifestyles of the rich but not necessarily famous. Since then, she has worked for the Sunday Style and Datebook sections, covering gala night openings and writing trend pieces. Her profiles of personalities have included fashion designer Diane von Furstenberg, Twitter co-founder Biz Stone and Emanuel Ungaro fashion house owner Asim Abdullah, to name a few. In a six-month project with The Chronicle's investigative team, she recently revealed the misleading practices of a San Francisco fashion charity that took donations from wealthy philanthropists but donated little to the stated cause of helping the developmentally disabled. On the lifestyle front, her duties also including writing about cannabis culture for The Chronicle and its cannabis website, www.GreenState.com website.

Let's Play

Cross|word Pile-Up Poker Typeshift Really Ba

SFGATE

Top

About

Contact

Services

Quick Links

Biol. Rev. (2016), 91, pp. 982–1005.
doi: 10.1111/brv.12207

A synthesis of two decades of research documenting the effects of noise on wildlife

Graeme Shannon[1,†*], Megan F. McKenna[2,†], Lisa M. Angeloni[3], Kevin R. Crooks[1], Kurt M. Fristrup[2], Emma Brown[2], Katy A. Warner[1], Misty D. Nelson[1], Cecilia White[1], Jessica Briggs[1], Scott McFarland[1] and George Wittemyer[1]

[1]*Department of Fish, Wildlife, and Conservation Biology, Colorado State University, Fort Collins, CO 80523, U.S.A.*
[2]*Natural Sounds and Night Skies Division, National Park Service, Fort Collins, CO 80525, U.S.A.*
[3]*Department of Biology, Colorado State University, Fort Collins, CO 80523, U.S.A.*

ABSTRACT

Global increases in environmental noise levels – arising from expansion of human populations, transportation networks, and resource extraction – have catalysed a recent surge of research into the effects of noise on wildlife. Synthesising a coherent understanding of the biological consequences of noise from this literature is challenging. Taxonomic groups vary in auditory capabilities. A wide range of noise sources and exposure levels occur, and many kinds of biological responses have been observed, ranging from individual behaviours to changes in ecological communities. Also, noise is one of several environmental effects generated by human activities, so researchers must contend with potentially confounding explanations for biological responses. Nonetheless, it is clear that noise presents diverse threats to species and ecosystems and salient patterns are emerging to help inform future natural resource-management decisions. We conducted a systematic and standardised review of the scientific literature published from 1990 to 2013 on the effects of anthropogenic noise on wildlife, including both terrestrial and aquatic studies. Research to date has concentrated predominantly on European and North American species that rely on vocal communication, with approximately two-thirds of the data set focussing on songbirds and marine mammals. The majority of studies documented effects from noise, including altered vocal behaviour to mitigate masking, reduced abundance in noisy habitats, changes in vigilance and foraging behaviour, and impacts on individual fitness and the structure of ecological communities. This literature survey shows that terrestrial wildlife responses begin at noise levels of approximately 40 dBA, and 20% of papers documented impacts below 50 dBA. Our analysis highlights the utility of existing scientific information concerning the effects of anthropogenic noise on wildlife for predicting potential outcomes of noise exposure and implementing meaningful mitigation measures. Future research directions that would support more comprehensive predictions regarding the magnitude and severity of noise impacts include: broadening taxonomic and geographical scope, exploring interacting stressors, conducting larger-scale studies, testing mitigation approaches, standardising reporting of acoustic metrics, and assessing the biological response to noise-source removal or mitigation. The broad volume of existing information concerning the effects of anthropogenic noise on wildlife offers a valuable resource to assist scientists, industry, and natural-resource managers in predicting potential outcomes of noise exposure.

Key words: acoustics, noise pollution, human disturbance, vocal communication, acoustic metrics, masking, physiology, behaviour, mitigation, fitness, conservation.

CONTENTS

*Address for correspondence (Tel: +1970 4915020; E-mail: graeme.shannon@colostate.edu).

† Authors contributed equally to the study.

I. INTRODUCTION

Noise generated by human activities has increased dramatically over recent decades as a result of population growth, urbanisation, globalisation of transportation networks, and expansion of resource extraction. Road traffic in the USA, for example, has outstripped population growth over the past 40 years by a factor of ten, and the number of domestic passenger flights has more than tripled since the early 1980s (Barber, Crooks & Fristrup, 2010). In marine environments, the distribution and effects of human activity (e.g. offshore oil extraction, commercial ship traffic) are extensive (Halpern *et al.*, 2008), and shipping alone is estimated to have increased low-frequency background sound levels by 12 dB over the past few decades (Hildebrand, 2009). With the rapid escalation of noise pollution, there is growing concern regarding its impacts on human health and the functioning of natural systems (Chepesiuk, 2005; McGregor *et al.*, 2013).

Anthropogenic changes to the acoustic environment include increases in the number of high-intensity noise events and chronically elevated and homogenised background sound levels. The impact of these changes has been most thoroughly assessed in humans, with profound physiological and psychological consequences, including increased risk of cardiovascular disease (Babisch *et al.*, 2005; Hansell *et al.*, 2013), sleep deprivation (Fyhri & Aasvang, 2010), and cognitive impairment (Szalma & Hancock, 2011). These impacts are estimated to cost at least one million healthy life years per annum in Western Europe (Fritschi *et al.*, 2011). Protective legislation for human communities was implemented four decades ago in the USA (Noise Control Act of 1972, Quiet Communities Act of 1978) and more recently in the European Union (Environmental Noise Directive 2002/49/EC).

Quantifying the effects of anthropogenic noise on wildlife is challenging. Sensitivity to noise varies widely across taxa (Kaseloo & Tyson, 2004; Brumm & Slabbekoorn, 2005; Morley, Jones & Radford, 2013; Slabbekoorn, 2013), and may also vary depending upon context, sex, and life history (Ellison *et al.*, 2012; Francis & Barber, 2013). Noise can induce compound biological responses (e.g. shifts in vocalisation and movement; McLaughlin & Kunc, 2013), and is rarely isolated from other forms of environmental disturbance, such as habitat alteration and visual disturbance, confounding interpretation of biological responses to noisy environments (Summers, Cunnington & Fahrig, 2011). Furthermore, determining the scale and extent of disturbance involves carefully measuring characteristics of the sound source, such as duration (chronic, intermittent), frequency content, and intensity (Nowacek *et al.*, 2007; Southall *et al.*, 2007; Francis & Barber, 2013; Gill *et al.*, 2015).

Despite these challenges, a coherent research focus on noise impacts has recently emerged. Review papers have either focussed broadly on wildlife (Brumm & Slabbekoorn, 2005; Barber *et al.*, 2010; Kight & Swaddle, 2011), or targeted specific taxonomic groups such as birds (Patricelli & Blickley, 2006; Slabbekoorn & Ripmeester, 2008; Ortega, 2012; Slabbekoorn, 2013), fish (Slabbekoorn *et al.*, 2010; Radford, Kerridge & Simpson, 2014), and invertebrates (Morley *et al.*, 2013). The Marine Mammal Protection Act stimulated noise regulation for marine mammals, and there have been several reviews of the effects of noise on these species (Richardson *et al.*, 1995; Boyd *et al.*, 2008; Tyack, 2008; Southall *et al.*, 2009; Ellison *et al.*, 2012). Some reviews have focused on specific behaviours (Luther & Gentry, 2013) or responses to noise (Wright *et al.*, 2007; Hotchkin & Parks, 2013), while conceptual frameworks for evaluating noise impacts to wildlife have also recently been published (Moore *et al.*, 2012; Francis & Barber, 2013).

This review provides a systematic and standardised synthesis of the peer-reviewed literature published from 1990 to 2013 reporting responses of wildlife to anthropogenic noise in terrestrial and aquatic habitats. It documents prominent trends in research topics and methods, the kinds of noise sources that have been studied and the measurements used to characterise them, and gaps in research coverage that merit attention in future research. Ultimately, we highlight the utility of existing scientific information concerning the effects of anthropogenic noise on wildlife for predicting potential outcomes of noise exposure and implementing meaningful mitigation measures.

II. LITERATURE REVIEW METHODOLOGY

We conducted a detailed literature search using Thompson's *ISI Web of Science* within the following subject areas 'Acoustics', 'Zoology', 'Ecology', 'Environmental Sciences', 'Ornithology', 'Biodiversity Conservation', 'Evolutionary Biology', and 'Marine Freshwater Biology' from 1990 to 2013. The specific search terms were [WILDLIFE or ANIMAL or MAMMAL or REPTILE or AMPHIBIAN or BIRD or FISH or INVERTEBRATE] and [NOISE or SONAR]), which returned a total of 2205 scientific peer-reviewed articles. These papers were filtered so only empirical studies focussed on documenting the effects of anthropogenic noise on wildlife were included in the final data set ($N = 242$). Reviews, syntheses, method papers ($N = 32$), and studies dealing solely with natural acoustic sources ($N = 22$) were excluded.

We reviewed the remaining publications to systematically characterise each study using 21 attributes, including details on the publication (journal, discipline, and year published), study design (playback or natural experiment, field or laboratory-based), and biological information (geographic region, general taxonomic grouping, and whether the study occurred in aquatic or terrestrial habitats). Journal titles were used to classify each of the papers using the following disciplinary categories: acoustics, behaviour, captive animals and welfare, conservation and management, ecology, environment, general biology, taxon-specific, physiology, and other. In addition, studies were classified based on the type of anthropogenic noise source, the acoustic metrics reported to describe the noise source and the biological responses measured in the study (see online Supporting Information, Appendix S1 for full details of extracted information).

Prior to commencing the full literature review process, we characterised ten randomly selected publications as a group to ensure accuracy and consistency of reporting across individual reviewers. Each of the authors then characterised a subset of the publications (five studies) across all 21 attributes to ensure that definitions of categories were clear and assignments were unambiguous. To improve the consistency of the data-collection process further, each paper was reviewed independently by at least two authors with G.S. and M.F.M. resolving any inconsistencies.

(1) Noise-source categories

We considered all anthropogenic sound sources as noise, regardless of whether the noise was intentionally produced, such as seismic exploration, sonars, acoustic deterrents, or an unintended by-product of human activity such as maritime shipping, traffic corridors, and construction. Furthermore, we categorised noise sources based on anthropogenic activity, not necessarily the characteristics of the noise stimulus, although we also recorded and present this information (see online Appendix S1). Six noise-source categories were used: environmental, transportation, industrial, military, recreation, and other.

Studies were assigned to the environmental noise category when the noise investigated was not attributed to a specific source, but rather included all the acoustic energy generated by human activity at a given location and time, also known as urban noise or background noise. In many cases, these acoustic environments include sources from the other defined noise categories that were not identified in the experimental design. Noise sources in the transportation category comprised both commercial and private vehicles, including road traffic (motorcycles, automobiles, buses), waterway traffic (boats, ferries, commercial ships), and non-military aerial traffic (commercial jets, helicopters). Studies that investigated specific recreational activities, such as whale-watching boats and air tour helicopters, were separated from the transportation studies. The industrial noise source category included studies that examined the effects of energy exploration (e.g. seismic surveys), construction (e.g. pile driving), and the operations associated with different energy sectors. Military sources included gunfire, explosions, aircraft, naval sonar, and in some cases, entire military training operations. We categorised the remainder of the studies as 'other', with most studies in this category using a simulated noise source, such as white noise, and not representing a specific human activity.

(2) Acoustic measurements

We evaluated if complete and accurate characterisation of acoustic environments, signals or stimuli, was provided. Information was collected on the acoustic metrics reported, where the reported level was measured (i.e. on site, on animal, not reported, estimated), and if background sound levels were measured. In addition, we recorded whether details on spectral characterisation (e.g. bandwidth and frequency weighting) and analysis (e.g. duration of measurement, sampling frequency, reference pressure) were reported. If details on the analysis of the acoustic data were not presented, we noted whether the study referenced an established standard or included details on the settings of a commercially available instrument.

(3) Biological responses

A categorical framework was developed to summarise the biological responses measured in each study. The biological responses were classified into nine distinct categories to help assess the distribution of studies across types of responses. These included: (*i*) physiology (stress, hearing loss/damage, immune function, gene expression); (*ii*) direct fitness metrics (survival, fecundity, clutch size); (*iii*) mating behaviour (attraction, mating success, territorial behaviour, pair bonding); (*iv*) foraging behaviour (foraging rate, predation rate, hunting/foraging success); (*v*) movement (spatial distribution, fleeing rate, avoidance, dive pattern); (*vi*) vigilance; (*vii*) vocal behaviour (call rate, intensity/amplitude, frequency shift, song length, call type, signal timing); (*viii*) population metrics (abundance, occupancy, settlement, density); and (*ix*) community-level metrics (species composition, predator–prey interactions). If studies measured multiple biological responses, a second category was noted.

III. STATE OF THE KNOWLEDGE

Research on the impacts of anthropogenic noise on wildlife has steadily risen over the past two decades (1990–2013), with a rapid increase in the volume of published, peer-reviewed articles since 2010 (Fig. 1). The 242 studies that we reviewed have been published in 97 scientific journals, covering a broad range of scientific disciplines from general biology to conservation to physiology (Table 1). Documented responses to a variety of anthropogenic noise sources (Table 2) have included shifts in physiology (e.g. impaired hearing, elevated stress hormone levels), alteration of key behaviours (e.g. foraging, vigilance, movement), and interference with ability to detect important natural sounds (e.g. vocalisations of conspecifics) (Table 3). In the following sections, we explore topics that emerged from our analysis of the existing literature and provide supporting examples.

(1) The taxonomic and geographical diversity of noise research

Many animals have specialised auditory organs and utilise sound for a variety of ecological functions from navigation and detection of resources to alerting conspecifics to the presence of predators. It is not surprising that noise impacts have been investigated in many taxonomic groups of animals, including vertebrates and invertebrates, and across a diverse range of terrestrial and aquatic habitats (Table 1). This broad taxonomic and geographic sampling is crucial to understanding how animals respond to noise across a range of auditory capabilities, behavioural contexts, levels of prior exposure, and noise sources. Further, investigating the effects of noise on a diversity of taxa within a study system enables detailed exploration of the complex and potentially differential responses to the same noise source. For example, in the woodlands of north-western New Mexico, USA, species richness of nesting birds was reduced as a function of anthropogenic noise, but birds that were able to tolerate noisier habitats had higher reproductive success due to reduced predation (Francis, Ortega & Cruz, 2009).

Although the published literature includes broad taxonomic sampling, birds and marine mammals are by far the most studied groups (Table 1; Fig. 1). Terrestrial research has focused mainly on effects on vocal communication, while aquatic research has also explored noise effects on movement, foraging, and physiology (Table 3). Underrepresented taxa in the published literature include reptiles, amphibians, and invertebrates (Table 1). Invertebrate studies, for instance, contributed only 4% of the total data set, yet this group contains 97% of the world's documented animal species, fulfilling varied and important ecological roles, such as prey species, pollinators, and serving as sensitive indicators of environmental change (de Soto *et al.*, 2013). Invertebrate species also provide excellent model species for studying the complex effects of noise given their size, rapid generation time, and the ease of maintaining laboratory populations (reviewed by Morley *et al.*, 2013).

Similar to its taxonomic focus, research on the effects of anthropogenic noise on terrestrial systems has been geographically biased, with 81% of the research conducted in either North America or Europe (this includes all laboratory and theoretical studies), while South America, Asia, and Africa remain underrepresented (Table 1). Yet developing nations are likely to experience the greatest level of population and economic growth over coming decades (Bloom, 2011). This situation provides important opportunities and motivation to study the effects of noise in less-disturbed habitats and to introduce known mitigation strategies to avoid negative consequences, particularly given that South America, Asia, and Africa are also home to some of the most biodiverse regions on the planet (Jenkins, Pimm & Joppa, 2013). Individual-, population-, and community-level reactions to a novel noise stimulus will likely differ between areas previously exposed to anthropogenic noise over extended periods of time and areas where anthropogenic noise exposure is lower and the source was recently introduced.

(2) Isolating the effects of noise

Anthropogenic noise is commonly associated with human activities that produce multiple types of disturbances (e.g. visual, habitat fragmentation). A number of experimental approaches have been developed to isolate noise from these other confounding variables, these include natural experiments contrasting noisy and quiet areas while holding other variables constant (e.g. natural gas compressor studies; Habib, Bayne & Boutin, 2006; Bayne, Habib & Boutin, 2008; Francis *et al.*, 2009), and controlled playback experiments where noise is introduced in isolation to the other forms of disturbance (e.g. for free-ranging populations of marine

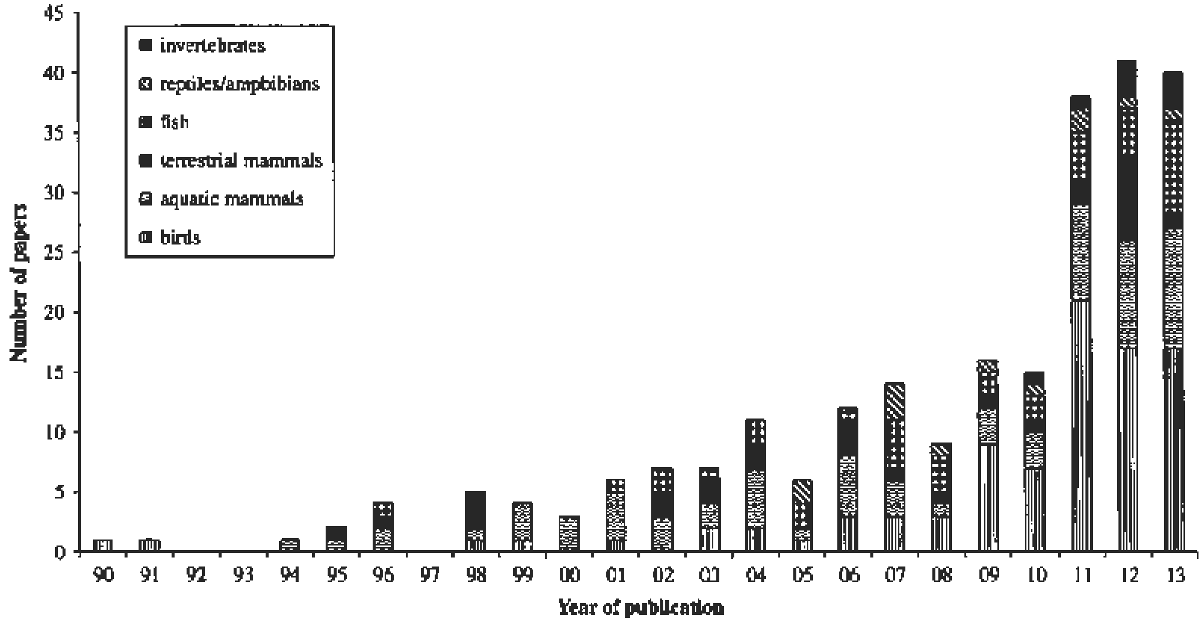


Fig. 1. Number of peer-reviewed publications reporting the effects of anthropogenic noise on wildlife from 1990 to 2013. Publications are divided into broad taxonomic categories: birds, aquatic mammals, terrestrial mammals, fish, reptiles/amphibians and invertebrates.

mammals and birds: Blickley, Blackwood & Patricelli, 2012*a*; Blickley *et al.*, 2012*b*; Goldbogen *et al.*, 2013; McClure *et al.*, 2013).

Studies that have isolated noise from potentially confounding variables have provided crucial evidence that noise alone can directly alter behaviour (Karp & Root, 2009; DeRuiter *et al.*, 2013*b*), reduce habitat quality (Blickley *et al.*, 2012*b*), and cause physiological impacts (Mooney, Nachtigall & Vlachos, 2009) across a range of species. For example, a recent playback study created a 0.5 km 'phantom acoustical road' to compare migratory bird habitat utilisation during 'on' and 'off' conditions (McClure *et al.*, 2013). The results from this sequence of trials, combined with concurrent observations of nearby control habitat (similar vegetation, no noise playback), provide decisive evidence that noise alone causes rapid changes in habitat use.

A combination of research approaches has proved important in identifying the consequences of noise disturbance. Natural experiments utilising existing acoustical gradients over time or space (48% of reviewed studies) have the potential to confound the effects of noise with other disturbances (see Summers *et al.*, 2011), but can be complimentary to controlled playback experiments conducted on free-ranging populations (15% of reviewed studies). Furthermore, biologically relevant responses at the individual, population, and community level can be identified in the field, whereas noise and the specific mechanisms driving changes in behaviour and physiology can be isolated with greater ease under laboratory conditions (Kight & Swaddle, 2011).

(3) Relationship between the perception of noise and response

Biological responses to noise are varied (Table 3), in part because responses depend upon the perception of noise (reviewed by Francis & Barber, 2013). Noise can be perceived as a threat, as observed when animals respond similarly to playbacks of anthropogenic noise and predator calls (e.g. Tyack *et al.*, 2011). In other cases, noise causes sensory degradation or the inability to detect acoustic cues from conspecifics, predators, prey or the environment, which can alter predator–prey interactions (Siemers & Schaub, 2011), reduce reproductive success (Halfwerk *et al.*, 2011*b*), and change settlement dynamics (Holles *et al.*, 2013). Additionally, noise can distract animals from attending to more crucial stimuli in the environment (Chan *et al.*, 2010), it can be a direct stressor causing pain or elevated stress hormone levels (Blickley *et al.*, 2012*b*; Rolland *et al.*, 2012), or in some instances, noise may provide a shelter from disturbance-sensitive predators (Francis *et al.*, 2009; Brown *et al.*, 2012).

The mechanisms by which animals respond to noise are not necessarily mutually exclusive. For example, animals that remain in a 'noisy' habitat because it provides a shelter from predators will likely have to contend with sensory degradation, either through changes in vocalisations (Mockford & Marshall, 2009; Mockford, Marshall & Dabelsteen, 2011) or vigilance patterns (Quinn *et al.*, 2006; Rabin, Coss & Owings, 2006). Noise can also induce the same response *via* compound mechanisms; for instance, reductions in foraging activity may be driven by a combination of increased perceived predatory threat,

Table 1. Summary of peer-reviewed literature reporting the effects of anthropogenic noise on wildlife published from 1990 to 2013 ($N = 242$

Journal type									
General biology	Taxon-specific	Environment	Conservation and management	Behaviour	Acoustics	Ecology	Physiology	Captive animals & welfare	Other
22%	13%	13%	12%	12%	11%	8%	3%	2%	2%
Taxonomic diversity									
Birds	Aquatic mammals	Fish	Terrestrial mammals	Reptiles/ amphibians	Invertebrates	Multiple species			
37%	28%	15%	11%	4%	4%	1%			
Terrestrial geographic distribution ($N = 128$, two studies that occurred in both aquatic and terrestrial habitats were excluded)									
North America	Lab/theoretical	Europe	South America	Australia	Asia	Global	Africa	Antarctica	
36%	24%	21%	7%	7%	4%	1%	0%	0%	
Aquatic geographic distribution ($N = 110$, two studies that occurred in both aquatic and terrestrial habitats were excluded)									
Lab/theoretical	Atlantic Ocean	Pacific Ocean	Mediterranean Sea	North/ Norwegian Sea	Artic	Australia	Africa estuary	Antarctica	Indian Ocean
44%	25%	12%	7%	5%	3%	2%	1%	1%	1%
Noise sources									
	Transportation	Environmental	Industrial	Military	Other	Recreation			
Aquatic	28%	5%	30%	22%	12%	3%			
Terrestrial	30%	35%	13%	8%	12%	2%			
Biological responses									
	Vocalisation	Movement	Physiological	Population	Vigilance	Foraging	Mating	Direct fitness	Community
Aquatic	21%	37%	32%	4%	0%	4%	1%	2%	0%
Terrestrial	44%	15%	10%	16%	6%	2%	4%	2%	2%

Table 2. Proportion of studies in different noise-source categories

Noise-source category	Examples	Per cent of terrestrial studies	Per cent of aquatic studies
Environmental	General background noise (urban and developed areas, no specific source identified)	5	35
Transportation	Commercial (maritime shipping, commercial aircraft, train, bus) and private (general traffic, automobile, motorcycle, small boat) transport noise	30	13
Industrial	General construction, machinery, energy (wind, oil and gas, development and operation, pile driving, seismic survey (air-guns), echo sounder, and underwater communication network noise	23	8
Military	Gun fire, explosion, naval sonar, and aircraft noise	12	12
Recreation	Hunting, whale-watching, air tour, snowmobile, and race-track noise	3	2
Other	Simulated (white, pink, tones), human voice, alarm, aquarium, and chainsaw noise	27	31

distraction, stress-induced loss of appetite, and masking of prey cues (Bracciali *et al.*, 2012; Wale, Simpson & Radford, 2013).

Evidence suggests that the characteristics of the acoustic signal (e.g. frequency, duration, onset, intensity) and the biology of the species in question (e.g. hearing range, behavioural state, habitat, vocal behaviours) are important for predicting how noise is likely to affect a particular organism (reviewed by Francis & Barber, 2013; Parris & McCarthy, 2013). Chronic noise sources are likely to degrade auditory cues important for predator/prey detection (Siemers & Schaub, 2011), communication (Hatch *et al.*, 2012) and orientation (Ellison *et al.*, 2012), especially if the noise source is high intensity and overlaps in frequency with an organism's hearing capabilities or the sound of interest (e.g. footfalls, leaves rustling; see Goerlitz, Greif & Siemers, 2008). Shifts in vocal rate, call intensity, call type, call frequency (as reviewed by Slabbekoorn, 2013), the timing of singing (Fuller, Warren & Gaston, 2007), and duration of calling (Diaz, Parra & Gallardo, 2011) have been studied extensively among birds (and marine mammals) to explore how vocal communication is affected by anthropogenic noise (see Tables 1 and 3), and to examine possible behavioural adaptations that are employed to overcome masking. The link between vocal flexibility and persistence in noisy environments has been demonstrated in a number of species (Francis *et al.*, 2011*d*; Proppe, Sturdy & St Clair, 2013*b*) and vocal behaviour and ability to learn can influence a vocal response to noise (Hu & Cardoso, 2010; Ríos-Chelén *et al.*, 2012). Recent theoretical work predicted the reduction in active space of vocal signals for birds moving from rural to urban habitat and identified the communication benefits of raising vocal frequency in noisy environments, particularly for species with calls in the lower frequency range (reviewed by Parris & McCarthy, 2013). Nevertheless, a change in vocalisation may come with significant consequences, including altered energy budgets and loss of vital information (Read, Jones & Radford, 2014).

Although explored to a lesser extent, responses to reduced cue detection, such as movement away from the noise (e.g. Miksis-Olds & Wagner, 2011; McLaughlin & Kunc, 2013) and a reduction in foraging efficiency (Schaub, Ostwald & Siemers, 2008; Siemers & Schaub, 2011), have also been demonstrated in the presence of chronic noise.

Noise sources that are novel, unpredictable, or are acoustically similar to biologically relevant sounds are predicted to elicit responses similar to those associated with predation risk (flee, hide, startle responses; reviewed by Francis & Barber, 2013). Although the sound must be detected, the noise does not need to overlap with peak hearing capabilities or be received at a high intensity to elicit antipredator behaviour. For example, beaked whales (*Ziphius cavirostris*) responded similarly to playbacks of military sonar and calls of killer whales (their main predator) (Tyack *et al.*, 2011). In this case sonar overlapped with the peak hearing range of the study species, but sonar also elicited antipredator responses in blue whales (*Balaenoptera musculus*) with hearing sensitivities in much lower frequencies (Goldbogen *et al.*, 2013), and failed to elicit responses in Atlantic herring (*Clupea harengus*), despite overlap with their most sensitive hearing range (Doksæter *et al.*, 2009). Thus, the frequency and intensity of noise are just a few of the factors driving responses, with temporal and spatial context of the disturbance, prior experience and similarity to relevant biological sounds also playing key roles (reviewed by Ellison *et al.*, 2012).

Current research is furthering our understanding of the specific mechanisms driving the observed biological responses to noise and the contextual factors that shape them. For example, the presence of young (Maier *et al.*, 1998), social status (Bruintjes & Radford, 2013), and spatial orientation relative to a noise source (Delaney *et al.*, 1999; Ellison *et al.*, 2012) can all drive differential responses. The duration and timing of noise stimuli are also important, as extended exposure to a chronic noise source may ultimately lead to tolerance or habituation, particularly if it provides an indirect benefit (e.g. a predator shelter; Francis *et al.*, 2009;

Table 3. Distribution of studies by biological response and noise source

	Biological response	Noise source: Environmental	Transportation	Industrial	Military	Other
All Studies (N=212)	*vocal behaviour*	20.3%	9.9%	1.4%	1.9%	2.8%
	movement	1.9%	4.2%	5.7%	6.1%	4.2%
	physiology	–	4.2%	5.2%	2.4%	7.5%
	population metrics	1.4%	4.2%	4.7%	0.5%	–
	vigilance	–	0.9%	0.5%	0.9%	0.5%
	mating behaviour	–	1.4%	0.9%	–	0.5%
	foraging behaviour	–	2.4%	–	0.5%	–
	direct fitness metrics	0.5%	0.5%	0.9%	–	–
	community-levelmetrics	0.5%	–	0.5%	–	–
Terrestrial Studies (N=120)	*vocal behaviour*	[illegible]	11.7%	–	–	4.2%
	movement	2.5%	2.5%	–	3.3%	3.3%
	physiology	–	0.8%	2.5%	0.8%	5.8%
	population metrics	2.5%	5.8%	7.5%	0.8%	
	vigilance	–	1.7%	0.8%	1.7%	0.8%
	mating behaviour	–	1.7%	1.7%	–	0.8%
	foraging behaviour	–	1.7%	–	–	–
	direct fitness metrics	0.8%	-	0.8%	–	–
	community-levelmetrics	0.8%	-	0.8%	–	–
Aquatic Studies (N=92)	*vocal behaviour*	5.4%	[illegible]	3.3%	4.3%	1.1%
	movement	1.1%	6.5%	[illegible]	[illegible]	5.4%
	physiology	–	[illegible]	[illegible]	4.3%	[illegible]
	population metrics	–	2.2%	1.1%	–	–
	vigilance	–	–	–	–	–
	mating behaviour	–	1.1%	–	–	–
	foraging behaviour	–	3.3%	–	1.1%	–
	direct fitness metrics	–	1.1%	1.1%	–	–
	community-level metrics	–	–	–	–	–

Only studies that reported a statistically measured response were included. Colour shading indicates the relative number of studies in each category.

Brown *et al.*, 2012). Studies combining different metrics of response, such as spatial distribution and vocal activity, may offer further insight into the varied consequences and trade-offs for species and communities exposed to noise (McLaughlin & Kunc, 2013). Ultimately, predicting how noise characteristics, behavioural contexts, and animal biology interact will be central in identifying habitats that are of conservation concern and implementing effective mitigation strategies.

(4) Ecological consequences of noise

A diverse range of biological responses to noise, from altered hearing thresholds of captive fish to changes in movement and foraging behaviour of large marine mammals in the open ocean, have been measured. Of the 242 studies included in this review, 88% reported a statistically measured biological response to noise exposure (see Table 3 & online Appendix 1 for further details). A small number of these studies have begun examining the impacts of noise using metrics associated with population persistence (survival, reproductive fitness), community interactions (predator–prey interactions), and ecosystem services (pollination) to understand the biological costs of anthropogenic noise. For example, studies on the impacts of noise to population persistence measured declines in productivity of breeding (Kight, Saha & Swaddle, 2012), reduction in fitness (Schroeder *et al.*, 2012), and change in timing of settlement (Pine, Jeffs & Radford, 2012).

Investigating the effects of noise on multiple taxa within a study system enables detailed exploration of the complex and interactive nature of noise impacts. Noise was found to impact key ecological services, enhancing pollination *via* reduced predation in noisy areas for hummingbirds, while decreasing seed dispersal for dominant plants because key dispersers avoided noisy areas (Francis *et al.*, 2012). Investigating the effects of noise on lower trophic levels can also reveal community-level impacts of noise. For instance, exposure to continuous turbine noise interfered with natural settlement cues for two species of abundant estuarine crabs, likely disrupting food-web interactions (Pine *et al.*, 2012). Noise altered species interactions, including predator–prey interactions in terrestrial (Schaub *et al.*, 2008; Siemers & Schaub, 2011) and marine (Kuningas *et al.*, 2013; Wale *et al.*, 2013) communities, while social interactions of cichlid fish shifted in the presence of boat noise (Bruintjes & Radford, 2013). Although these studies did not directly test the consequences for community structure and function, changes in species interaction may ultimately translate into community-level effects.

The majority of noise research has used comparatively short-term natural or controlled experiments that commonly focus on behavioural change in single species and are spatially discrete. While this approach has proved pivotal in revealing the widespread impacts associated with noise, evidence for long-term effects on populations and communities is generally only suggestive. Long-term experiments conducted over broad spatial scales may offer a more complete understanding of the population-level and interacting effects of noise on wildlife.

(5) Application of research to develop and implement noise mitigation

The global increase in anthropogenic noise levels across both human-dominated and natural habitats presents a significant conservation challenge, especially when considered in conjunction with other threats to wildlife and ecosystem integrity. There is a real need for research on the impacts of noise on wildlife to translate into management actions or recommendations (Tabarelli & Gascon, 2005). While a variety of noise-mitigation methods exist, only 9% of the studies reviewed provided specific recommendations. Recommendations included the use of physical barriers to noise, geographical and temporal restrictions to human activity, and quiet technology (Table 4), yet few studies directly tested the effectiveness of these methods. The majority of studies with mitigation statements were published in conservation, ecology, and environmental journals and focussed primarily on terrestrial ecosystems.

Physical barriers are a commonly suggested mitigation tool that have been used along roadways to reduce noise levels for human populations (Murphy & King, 2011) and wildlife (Slabbekoorn & Ripmeester, 2008). However, the benefits of these barriers extend a relatively short distance. Barriers also can compound fragmentation effects by restricting animal movement, and their costs may well exceed other mitigation approaches (Summers *et al.*, 2011). Collectively, these considerations suggest that noise barriers are most suitable for roadside habitat of especially high conservation value, or to enhance the effectiveness of road-crossing structures or tunnels. Alternative options to reduce road noise include restrictions to traffic flows during sensitive life-history periods, speed reductions, improved road surface substrates, and new tyre technology. Controlled studies documenting changes in wildlife behaviour and habitat utilisation in response to reducing roadway noise would extend the findings of recent noise broadcast studies, and document the conservation value of quiet-road investment.

The noise-barrier approach can be effective for industrial activities such as resource extraction and construction, where machinery generates a point noise source that is spatially compact (Table 4). Specific methods have included the use of bubble curtains to reduce pile-driving noise in marine environments (Würsig, Greene & Jefferson, 2000) and the erection of sound barriers to minimise noise from terrestrial gas compressor stations (Francis *et al.*, 2011*a*). Implementing barrier mitigation measures may prove expensive (e.g. \$175–200 million to reduce oil and gas extraction noise by 4 dB; Bayne *et al.*, 2008), making it unlikely that industry will adopt these measures without specific regulations in place (Ortega, 2012).

(6) Characterising complex acoustic stimuli

Anthropogenic noise is a complex and challenging source of pollution to quantify, varying in duration, amplitude, and frequency content, while being modified by the medium through which it travels. The detailed reporting of acoustic measurements is necessary to repeat experiments, provide insight on the kinds of noise stimuli that induce a response, and synthesise results across studies. We were surprised that acoustic metrics and measurement methods were not always documented in these papers. Although the majority of studies used common acoustic metrics such as root-mean square sound pressure level (SPL), sound exposure level (SEL), or equivalent noise level (Leq) (see online Appendices S1 and S2 for descriptions of these metrics), 30% provided no details on the received sound levels of the noise stimulus and 10% simply reported a dB level without information on how the value was measured or calculated (Fig. 2). A notable proportion of studies (38%) lacked a record of the spectral analysis, such as duration of the measurement, frequency range, and weighting function (Fig. 2). Measurements of the background acoustic environment prior to exposure to a noise source (excluding the environmental noise category) were reported in only 53% of the literature (Fig. 2). Given the cross-disciplinary nature of terrestrial and aquatic bioacoustic research and the difference in reference pressure between air and water, it is surprising that the majority of studies (51%) did not report the reference pressure used when reporting a dB value (see Rossing, 2007 for further details). Ninety per cent of these studies were conducted in terrestrial environments, implying the use of the standard reference pressure in air, but this is a potential source of confusion (reviewed by Chapman & Ellis, 1998).

Table 4. Examples of reported mitigation methods for reducing the negative effects of noise on wildlife

	Environmental	Transportation	Industrial	Military
Birds	Urban planning (e.g. maintaining green spaces and reducing noise levels) to maintain biological communities [3]	Engineering solutions (e.g. road surfaces, tyres and vehicle engines) that reduce noise [13]	Use of sound barriers around compressors to reduce affected area by 70% and maintain occupancy and nest success rates [4]	
	Reduction of aircraft noise exposure to <80 dBA of river habitats used by harlequin ducks [5]	Closing key roads during breeding season; reducing traffic speed and volume [10]		
	Placement of new acoustically dominant features (roads, machinery) further from nesting areas; limits to production during sensitive periods of breeding; abatement of current noise by altering structures (e.g. sound walls, dense vegetation, removing highly reflective surfaces, rerouting traffic) [6]	Use of 105 m hemispherical protection to eliminate owl flush response to overflights; minimising flights 3 h following sunset and preceding dawn; separating overflights by at least 7 days [2]		
		Restricting traffic flow and heavy truck use [14]		
		Wise planning along transportation corridors and mitigation of noise along their paths to enhance habitat for the highest number of bird species [16]		
Terrestrial mammals		Setting criteria for height and density of road bordering vegetation, filling in gaps in tree lines and encouraging canopy growth [15]	Noise barriers; construction scheduling to avoid noise-sensitive experiments [12]	Limiting military training exercises during calving and post-calving season [8]
Aquatic fish/mammmals		Ship design and construction that includes methods to reduce underwater noise and limited navigation permitted within fish spawning grounds during the spawning season [17]	Air bubble curtains and 'Hydro Sound Dampers' [18]	Use of dose–function methods in predicting the harassment of marine mammals [20]
		Definition of noise-free areas or seasonal restriction of noise activities during sensitive biological periods [11]	Avoiding pile-driving during calving and when animals are in 500 m exclusion zone; soft start or alarm sound before operations; restricting operations to low tide; decoupling equipment from hull of piling vessel; use of bubble curtain within 25 m radius of the pile [1]	Consider the likely contexts of exposure and the foraging ecology of baleen whales in planning military sonar operations [19]

Table 4. Continued

	Environmental	Transportation	Industrial	Military
Reptiles & amphibians	Use of noise barriers on road network; construction of new roads at distances away from protected areas; technological advances; laws with standard noise emission for vehicles, speed and driver behaviour [7]	Dense vegetation along roadsides (as a less costly alternative to solid barriers) to attenuate traffic noise [9]		
Invertebrates			Applying the precautionary principle when planning high-intensity activities such as explosions, construction or seismic exploration, in spawning areas of marine invertebrates with high natural and economic value [21]	

[1] David (2006); [2] Delaney *et al.* (1999); [3] Fontana, Burger, & Magnusson (2011); [4] Francis *et al.* (2011*d*); [5] Goudie & Jones (2004); [6] Kight *et al.* (2012); [7] Lengagne (2008); [8] Maier *et al.* (1998); [9] Parris & Schneider (2009); [10] Parris *et al.* (2009); [11] Picciulin *et al.* (2010); [12] Rasmussen *et al.* (2009); [13] Summers *et al.* (2011); [14] Zhang *et al.* (2012); [15] Zurcher, Sparks, & Bennett (2010); [16] Proppe *et al.* (2013*b*); [17] Liu *et al.* (2013); [18] Dähne *et al.* (2013); [19] Goldbogen *et al.* (2013); [20] Houser, Martin & Finneran (2013*b*); [21] de Soto *et al.* (2013).

IV. IDENTIFYING NOISE LEVELS THAT ELICIT A BIOLOGICAL RESPONSE

Our compilation and synthesis of research on the effects of anthropogenic noise on wildlife offers an opportunity to identify the noise levels that elicit biological responses. To integrate information on wildlife responses to noise into a common framework, we identified a subset of studies (69 terrestrial and 62 aquatic) that documented a significant response to a noise stimulus and also reported an acoustic value and metric at which a response occurred. Our classification of a 'significant response' was based upon the study reporting a statistical change in the particular biological metric as a function of noise exposure. A variety of metrics with different frequency weighting and bandwidths were reported in this subset of studies (see online Appendix S2). It was not possible to adjust all values to a common acoustic metric to compare across studies. Instead, we reported the metrics used in each study and the specific sound level (see Fig. 3); this provided graphical indications of the different metrics to reveal potential artefacts or differences in interpretation (Madsen, 2005).

Extracted noise levels were sorted to produce a cumulative weight-of-evidence curve as a function of the noise level at which a biological response was documented, thereby summarising the number of studies reporting a response at or below a given noise level. We compiled the results for terrestrial and aquatic studies separately because they used different reference pressures to derive noise levels. These cumulative curves span a wide range of species and biological responses, in addition to different acoustic metrics. This framework was modelled after a dose–response relationship, but each increment in the weight-of-evidence function does not represent an increasing number of responsive species. Rather, these curves depict an increasing number of studies documenting a response at a given noise level. Therefore, the curves suggest accumulation of evidence, not accumulation of response.

The cumulative weight-of-evidence curves provide support for natural resource managers seeking to establish management objectives for anthropogenic impacts or developing policy on noise (Fig. 3). For example, a limit on allowable noise levels can be supported by citing the percentage or number of studies that have documented biological impacts at or below that level. Lower noise thresholds are more protective, but they are supported by a smaller number of studies. Note that responses have been documented in terrestrial environments at noise exposure levels as low as 40 dB SPL, and 14 studies documented responses below 50 dB (Fig. 3A). Predictions of natural sound levels for the coterminous USA range from 24 to 40 dB (LAeq, 1 s, median summer daytime level; Mennitt, Fristrup & Nelson, 2013). The terrestrial weight-of-evidence curve includes all noise-source categories and species groups, although representation is unbalanced (Table 5). Multiple bird studies documented changes in song characteristics,

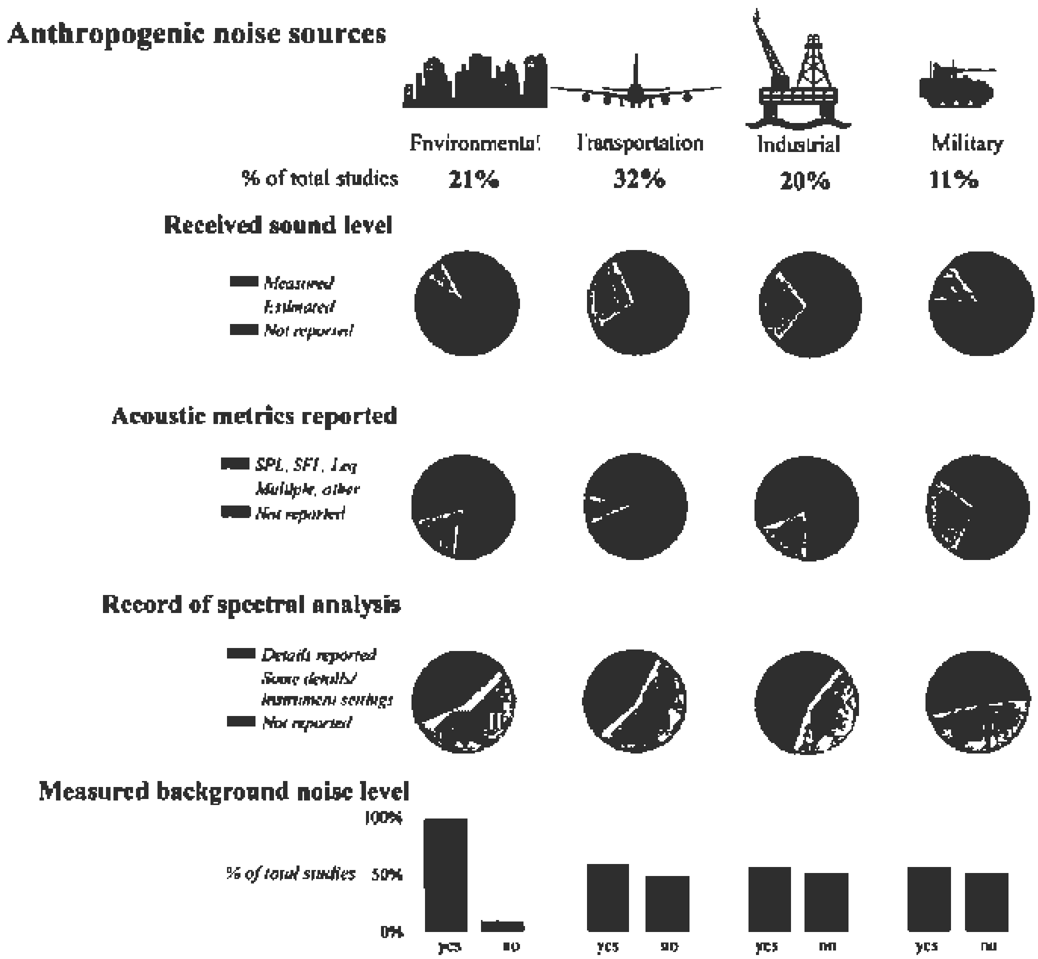


Fig. 2. Reporting of acoustic noise-source measurements. The chart divides all of the studies into noise source categories and highlights different components of acoustic analysis, including whether the received sound level was measured, the types of acoustic metric reported (see online Appendices S1 and S2 for acoustic metric definitions), whether details of the spectral analysis were provided and whether background noise was measured (note that background noise provided the noise source for most studies in the environmental category). Black-filled graphics indicate the proportion of studies in which details were not reported

reproduction, abundance, stress hormone levels, and species richness at levels ≥45 dBA SPL (re 20 μPa). Terrestrial mammals exhibited increased stress levels and decreased reproductive efficiency at noise levels between 52 and 68 dBA SPL (re 20 μPa). Traffic noise exceeding 60 dBA SPL (re 20 μPa) impacted the vocal behaviour of male anurans and traffic noise exceeding 80 dBA SPL (re 20 μPa) reduced the foraging efficiency of gleaning bats.

The diversity of responses and metrics creates opportunities for misinterpretation. For example, it might seem reasonable to utilise the median of this cumulative distribution as a noise-impact criterion that is robustly supported by this body of literature. This would yield a value of 60 dB. This level would be cause for concern in a community setting: it causes conversational speech interference. The EPA (1974) recommended a 55 dB criterion to protect the health and welfare of the American public. The inflated character of this median can be explained by examining the metrics associated with the points in Fig. 3A. Many of the studies that fall above the median utilised metrics that typically exceed LAeq (SPL max, SEL), or the studies did not specify the metric and measurement procedure. Accordingly, the most useful portion of this curve lies to the left of the median.

To provide insight into the relative effects of noise on humans and wildlife, the cumulative curve for the terrestrial wildlife studies was compared against human responses to noise derived from a meta-analysis of human survey data on annoyance at different noise levels (ANSI, 2005). The human response curve represents the predicted percentage of residents in quiet rural communities predicted to be highly annoyed by a new or unfamiliar noise source. Despite the heterogeneity in the wildlife noise metrics and responses, the range of noise levels documented to induce annoyance in humans and responses in terrestrial wildlife are similar (40–100 dB SPL re 20 μPa). Evidence

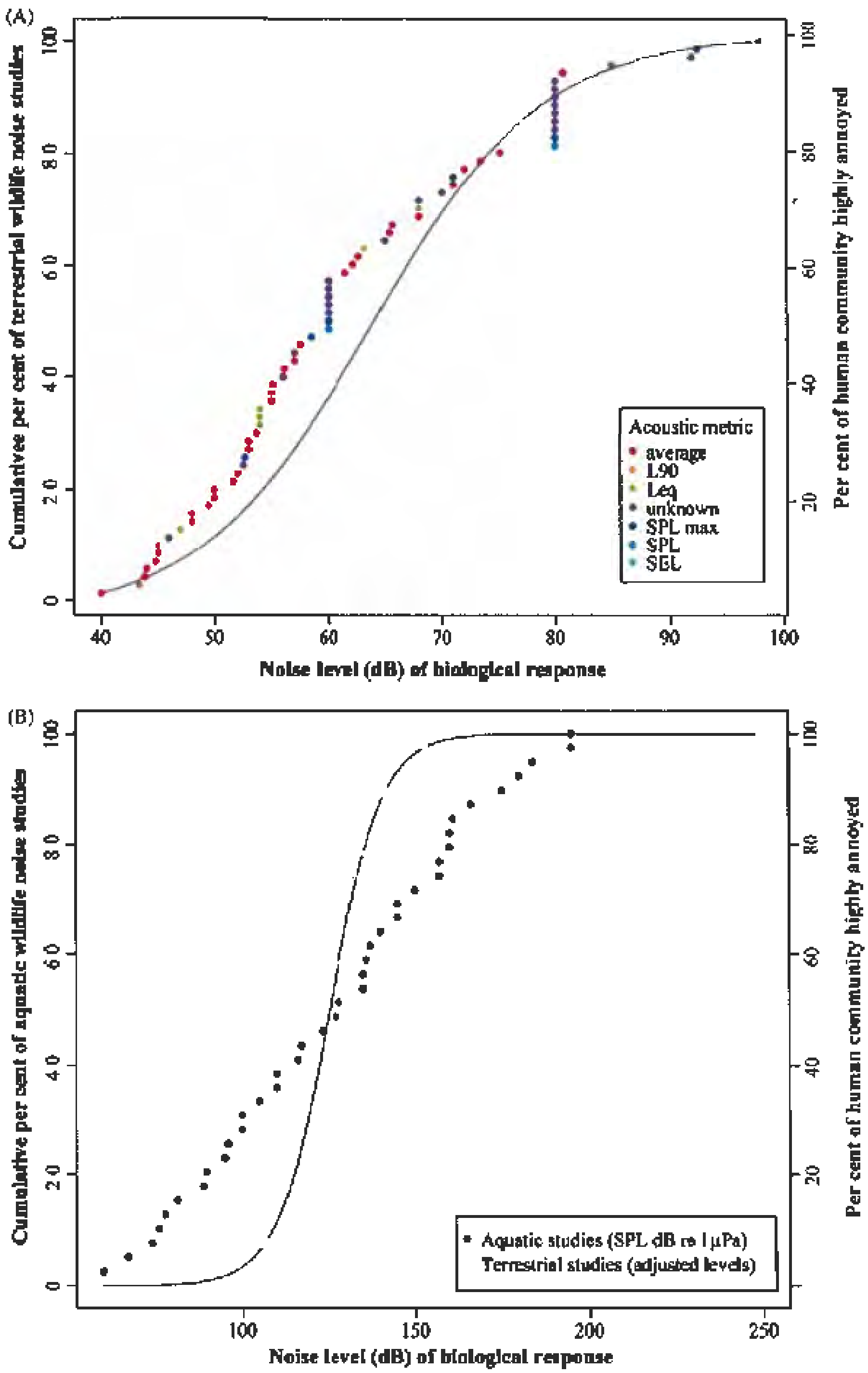


Fig. 3. Cumulative per cent of studies reporting biological responses by wildlife for a given noise level. Only studies that reported acoustic measurements are included ($N = 131$). See Appendices S1 and S2 for additional details on the noise levels, acoustic metric definitions, frequency weighting and bandwidths for each study used to generate these curves. (A) Results from terrestrial studies. Coloured symbols are used to reveal the potential influence of different metrics on the shape of the terrestrial curve. The human response curve (solid line) represents the predicted percentage of residents in quiet rural communities predicted to be highly annoyed by a new or unfamiliar noise source. (B) Results from the aquatic studies. Only SPL dB values were used to generate the cumulative curve. For comparison, received levels from the terrestrial wildlife studies and the human response curves (right y axis) are also plotted. The noise levels in the terrestrial wildlife and human studies were adjusted to the same scale as the aquatic studies. This was done by adding 61.5 dB to the sound level values to account for the difference in reference pressure and impedance.

for wildlife responses to noise accumulates at lower exposure levels than the rural human annoyance curve, although the slopes are similar. Another connection between human and wildlife noise studies is the onset of health effects. Epidemiological studies suggest that humans may experience elevated risk of hypertension when LAeq is greater than 55 dB (Stansfeld & Matheson, 2003). Physiological and fitness effects were documented by five papers included in this review at noise exposure levels of 52, 52, 58, 60, and 68 dBA SPL.

The aquatic studies generally provided better descriptions of their measurements, although in this literature variation in the bandwidth of the noise stimulus varies and presents a challenge for interpretation of the cumulative evidence curves. Fifty per cent of the aquatic studies measured a biological response at or below 125 dB (re 1 μPa) (Fig. 3B). The different reference pressure and acoustic impedances between air and water account for 61.5 dB of the differences in levels between terrestrial and aquatic studies (Leighton, 2012). The terrestrial data and human annoyance curves are included in Fig. 3B after accounting for this correction factor.

The studies contributing to the aquatic weight-of-evidence curve include all noise source categories and species groups (Table 6). Manatees shifted their foraging and movement behaviour when one-third octave band levels (4 kHz) exceeded 60 dB SPL, a notably low level. Otherwise, fishes, mammals, and invertebrates responded to noise across a wide range of noise levels (67–195 dB SPL re 1 μPa) (see online Appendix S2). Industrial noise, particularly high-intensity sound sources such as seismic air guns, impacted the physiology, vocal communication, and activity budgets of aquatic species, with reduced abundance and catch rates of fishes during relatively high levels of industrial noise (248 dB SPL re 1 μPa). Marine mammals responded to industrial noise by altering spatial movement patterns (107 dB Leq re 1 μPa), hearing thresholds (226 dB peak–peak re 1 μPa), and calling behaviour (82 dB SPL re 1 μPa) (Table 6). Naval sonar was the main source of concern in the military category (92% of aquatic studies with military sources). Sonar caused active avoidance, disrupted foraging, and temporary hearing loss among marine mammals in close proximity to the source (67 dB SPL re 1 μPa), yet showed limited effects on fish with all documented responses occurring at higher noise levels (195 dB SPL re 1 μPa) (Table 6).

V. RESEARCH RECOMMENDATIONS

Our review has highlighted the substantial body of information concerning the effects of anthropogenic noise on wildlife. Such research can assist scientists, natural resource managers, industry, and policy makers in both predicting potential outcomes of noise exposure as well as implementing meaningful thresholds and mitigation measures. Refinement and focus on several key research areas will further strengthen the conclusions and inferences that can be drawn regarding the impacts of noise on wildlife.

(1) Expand geographic and taxonomic sampling

Research on the effects of anthropogenic noise on terrestrial systems has been taxonomically and geographically biased, with 65% of studies conducted on birds and marine mammals and 81% of research carried out in either North America or Europe (includes all theoretical and laboratory-based studies). Investigating the effects of noise across a broader array of species and habitats is crucial for developing theories that explain variations in response to noise in terms of unique auditory capabilities, social structure, life history, ecological role, and evolutionary adaptation. Greater knowledge of taxon-level responses to noise will also be useful in predicting the likely responses of species that are too rare or elusive to study directly and may reveal responses in species previously thought unaffected because they occupy noisy areas (Shannon *et al.*, 2014) or have peak hearing sensitivities outside of a particular noise source (Goldbogen *et al.*, 2013).

(2) Explore interacting effects

In most cases, it remains unclear whether responses to noise will be further compounded by the introduction of potentially heterotypic stressors such as artificial light and habitat fragmentation. Designing studies that explore and quantify how the addition of other stressors influences observed biological responses to noise will facilitate evaluation of the benefit of reducing noise in environments facing multiple threats.

(3) Remove or reduce noise

Documenting biological responses in environments that have experienced a reduction in noise, such as closure of a road, closure of an energy facility, or a change in ship traffic routes, may reveal how systems recover from chronic noise exposure. Successful design requires knowledge and coordination with proposed changes in order to capture conditions prior to the reduction in noise levels.

(4) Invest in large-scale studies

To date there are very few studies that have attempted to explore the effects of noise at the landscape scale and/or over long temporal periods (e.g. seasonal, yearly), likely due to the logistical and experimental challenges that it presents, particularly in isolating the effects of noise from other sources of disturbance (e.g. habitat fragmentation, human presence). Nonetheless, in contrast to single-exposure, single-species research, larger-scale approaches can provide direct insight into the cumulative effects of noise exposure related to population persistence, ecological integrity, and evolutionary processes. Developing a systematic approach to sampling of multiple species within a community and multiple metrics of biological responses will therefore require coordination across scientific disciplines and organisations.

Biological Reviews 91 (2016) 982–1005 © 2015 Cambridge Philosophical Society

Table 5. Biological responses to different noise-source categories by terrestrial taxa. Studies in this table are included in Fig. 3A

	Environmental		Transportation		Industrial		Military	
Birds	Changes in frequency components of vocalisations [1–22]	44–73 dBA# 54 dBA** 60 dBA* 59 dBA† 53–80 dBA?	Changes in frequency components of vocalisations [33–35]	60–65 dBA? 50 dBA#	Changes in song frequency and length [53]	45 dBA**	Increase in vigilance and alert behaviour [64, 65]	63 dBA** 80 dBA†
	Changes in call rate and duration [14–18]	60 dBA* 48–66 dBA# 57 dBA?	Increase in amplitude of vocalisations [36]	57 dBA#	Increase in physiological stress levels [54]	52 dBA#		
	Increase in amplitude of vocalisations [19–23]	54 dB$,** 53-62 dBA# 80 dB$?	Shifts in timing of vocalisations [37]	80 dBA?	Reduced breeding success [55]	68 dBA#		
	Shifts in timing of vocalisations [24]	63 dBA#	Preference for roosting in quieter areas [38]	47 dBA**	Decline in occupancy and abundance [56–58]	48 dBA# 55 dBA# 45 dBA#		
	Decreases in acoustic complexity of songbird community [25]	53 dBA†	Reduction in reproductive success in presence of road noise [39]	68 dBA?	Changes in community and species interactions [59, 60]	60 dBC? 50 dBC#		
	Decline in species diversity [26, 27]	40 dB$# 45 dBA*	Effects on physiology and development [40]	60 dBA*				
	Avoidance of noisy environments [28]	70 dBA?	Changes in abundance, species richness, distribution and occupancy [41–43]	46 dBA? 45 dBA# 55 dBA**				
	Decline in reproductive success [29, 30]	58 dBA* 43 dBA***						
Mammals	Shifts in call frequency and amplitude for echolocating bats [31]	80 dB$	Disruption of foraging in gleaning bats [44, 45]	80 dBA*	Increase in physiological stress from construction noise [61, 62]	52 dBL# 92 dBA?	Short-term increase in heart rates and shifts in resting and movement behaviours of ungulates [66–68]	85 dB$* 98 dBA‡ 92 dB$†
					Reduced reproductive efficiency of laboratory mammals exposed to construction noise [63]	68 dBA**		

Table 5. Continued

	Environmental		Transportation		Industrial	Military
Reptiles and amphibians	Shifts in energy distribution of cicadas songs towards higher frequency [32]	62 dBC#	Reduction in chorus tenure and duration by male anurans exposed to traffic noise [46–48]	72 dBA# 60–80 dBC?		
			Difficulty locating mates [49]	75 dBC#		
			Increased minimum frequency of vocalisations [50]	60 dBA?		
			Change in calling time [51]	71 dBC†		
Inverte–brates			Higher frequency components in courtship signal of grasshoppers [52]	81 dBA?		

Symbols: * sound pressure level (SPL); ** equivalent continuous sound level (Leq or LAeq); † SPL max; ‡ sound exposure level (SEL); # average; ? unknown.
See Appendices S1 and S2 for acoustic metric definitions. ***A-weighting***, like the human ear, cuts off the lower and higher frequencies that the average person cannot hear. At higher sound levels (100 dB and above), the ear's response is flatter, as shown in the **C-weighting**. L (linear) or **unweighted** (also known as Z-weighting) is a flat frequency response of 10–20 kHz ±1.5 dB.

$ indicates studies where it was unclear if a frequency weighting function was applied.

[1] Pohl *et al.* (2012); [2] Seger-Fullam, Rodewald & Soha (2011); [3] Bermúdez-Cuamatzin *et al.* (2011); [4] Dowling, Luther & Marra (2012); [5] Bermúdez-Cuamatzin *et al.* (2009); [6] Mendes, Colino-Rabanal & Peris (2011); [7] Nemeth & Brumm (2010); [8] Hu & Cardoso (2010); [9] Proppe *et al.* (2012); [10] Slabbekoorn & Peet (2003); [11] Goodwin & Podos (2013); [12] Montague, Danck-Gontard & Kunc (2013); [13] Nemeth & Brumm (2009); [14] Mockford & Marshall (2009); [15] Halfwerk & Slabbekoorn (2009); [16] Ríos-Chelén *et al.* (2013); [17] Potvin, Parris & Mulder (2011); [18] Gross, Pasinelli & Kunc, 2010); [19] Redondo, Barrantes & Sandoval (2013); [20] Pohl *et al.* (2009); [21] Nemeth *et al.* (2013); [22] Wood & Yezerinac (2006); [23] Lowry, Lill & Wong (2012); [24] Fuller *et al.* (2007); [25] Pieretti & Farina (2013); [26] Patón *et al.* (2012); [27] Proppe *et al.* (2013*b*); [28] McLaughlin & Kunc (2013); [29] Kight *et al.* (2012); [30] González-Oreja *et al.* (2012); [31] Hage *et al.* (2013); [32] Shieh *et al.* (2012); [33] Halfwerk *et al.* (2011*b*); [34] Potvin & Mulder (2013); [35] Verzijden *et al.* (2010); [36] Brumm (2004); [37] Arroyo-Solís *et al.* (2013); [38] Zhang *et al.* (2012); [39] Halfwerk *et al.* (2011*a*); [40] Grino *et al.* (2013); [41] Arévalo & Newhard (2011); [42] Goodwin & Shriver (2011); [43] McClure *et al.* (2013); [44] Schaub *et al.* (2008); [45] Siemers & Schaub (2011); [46] Lengagne (2008); [47] Sun & Narins (2005); [48] Kaiser *et al.* (2011); [49] Bee & Swanson (2007); [50] Cunnington & Fahrig (2010); [51] Lampe *et al.* (2012); [52] Vargas-Salinas & Amézquita (2013); [53] Francis, Ortega, & Cruz (2011*a*); [54] Blickley *et al.* (2012*b*); [55] Schroeder *et al.* (2012); [56] Bayne *et al.* (2008); [57] Blickley *et al.* (2012*a*); [58] Francis, Ortega, & Cruz (2011*c*); [59] Francis *et al.* (2009); [60] Francis *et al.* (2012); [61] Powell *et al.* (2006); [62] Westlund *et al.* (2012); [63] Rasmussen *et al.* (2009); [64] Conomy *et al.* (1998); [65] Goudie & Jones (2004); [66] Krausman *et al.* (1998); [67] Maier *et al.* (1998); [68] Weisenberger *et al.* (1996).

Table 6. Biological responses to different noise-source categories by aquatic taxa. Studies in this table are included in Fig. 3B

	Environmental		Transportation		Industrial		Military	
Fishes			Decrease in detection of communication signals and increase in stress hormones [5–7]	135 dBL** 111 dBL** 153 dB**	Change in the movement behaviour of fishes [23, 24]	195 dB* 147 dB‡	Change in movement behaviour in the presence of sonar signals [39]	137 dB*
			Changes in spatial movement and orientation [8–10]	142 dB** 135 dB* 90 dB*	Reduction in local abundance and catch rate [25]	248 dB†	Auditory threshold shift in fish exposed to low frequency active (LFA) sonar [40, 41]	193 dB† 195 dB*
			Changes in territorial and social behaviour [11, 12]	161 dBL** 127 dB*	Damage to fish ears [26, 27]	180 dB* 174 dB‡		
			Temporary loss in hearing [13]	142 dBL**				
Mammals	Adjustments to vocalisation and singing behaviour [1–3]	110 dB# 105 dB* 117 dB#	Loss of communication space [14]	166 dB*	Changes in movement behaviour [28–32]	184 dB* 170 dB? 116 dB** 107 dB** 139 dB‡	Change in auditory response [42–45]	196 dB+ 67 dB* 75 dB* 210 dB‡
	Changes in the proportion of time spent feeding and milling [4]	60 dB*	Adjustment to vocalisation and singing behaviour [15, 16]	110 dB* 135 dB* 95 dB*	Shifts in hearing thresholds after exposure to seismic airguns [33]	226 dB+	Disruption in trained behaviour [46, 47]	150 dB* 175 dB*
			Increase in stress hormones [17]		Changes in vocalisations [34, 35]	82 dB# 116 dB#		
			Change in spatial movement patterns [18–20]	78 dB* 161 dB* 74 dB*			Change in spatial distribution and behaviour [48–51]	128 dB# 140 dB* 89 dB* 116 dB+
Invertebrates			Increase in larvae settlement [21]	100 dB*	Damage to sensory systems in cephalopod [36]	157 dB*		
			Disruption of foraging and anti-predator behaviour [22]	145 dB*	Development delays and body malformations [37, 38]	136 dB* 145 dB*		

Symbols: * sound pressure level (SPL); ** equivalent continuous sound level (Leq); † SPL max (Lmax, peak SPL); ‡ sound exposure level (SEL); + SPL peak–peak.

See Appendices S1 and S2 for acoustic metric definitions.

[1] Holt, Noren, & Emmons (2011); [2] Parks *et al.* (2011); [3] Scheifele *et al.* (2005); [4] Miksis-Olds & Wagner (2011); [5] Codarin *et al.* (2009); [6] Vasconcelos, Amorim, & Ladich (2007); [7] Wysocki, Dittami, & Ladich (2006); [8] Picciulin *et al.* (2010); [9] Sarà *et al.* (2007); [10] Holles *et al.* (2013); [11] Sebastianutto *et al.* (2011); [12] Bruintjes & Radford (2013); [13] Liu *et al.* (2013); [14] Hatch *et al.* (2012); [15] Holt *et al.* (2009); [16] Melcón *et al.* (2012); [17] Rolland *et al.* (2012); [18] Lemon *et al.* (2006); [19] de Soto *et al.* (2006); [20] Tripovich *et al.* (2012); [21] Wilkens, Stanley, & Jeffs (2012); [22] Wale *et al.* (2013); [23] Wardle *et al.* (2001); [24] Fewtrell & McCauley (2012); [25] Engås *et al.* (1996); [26] McCauley, Fewtrell, & Popper (2003); [27] Casper *et al.* (2013); [28] Brandt *et al.* (2011); [29] Goold (1996); [30] Kastelein *et al.* (2005); [31] Kastelein *et al.* (2006); [32] Dähne *et al.* (2013); [33] Finneran *et al.* (2002); [34] Risch *et al.* (2012); [35] Blackwell *et al.* (2013); [36] Sole *et al.* (2013); [37] de Soto *et al.* (2013); [38] Pine *et al.* (2012); [39] Doksæter *et al.* (2012); [40] Popper *et al.* (2007); [41] Halvorsen *et al.* (2013); [42] Finneran *et al.* (2000); [43] Kastelein, Hoek, & de Jong (2011*a*); [44] Kastelein, Hoek, & de Jong (2011*b*); [45] Mooney *et al.* (2009); [46] Houser, Martin, & Finneran (2013*a*); [47] Houser *et al.* (2013*b*); [48] McCarthy *et al.* (2011); [49] Tyack *et al.* (2011); [50] Deruiter *et al.* (2013*a*); [51] Deruiter *et al.* (2013*b*).

(5) Measure responses over a gradient of noise levels

Additional studies are needed that investigate a gradient of noise exposure rather than quiet *versus* loud treatments. A gradient design provides insight on the levels of noise at which a response is initiated and how the response changes with increasing noise levels. This design can also reveal how animals recover from exposure to noise, while exploring the relationship between levels and duration of noise exposure and habituation or sensitisation by different species.

(6) Evaluate mitigation measures

There is a need to evaluate the ecological benefit of mitigation measures in both terrestrial and aquatic environments. Technological innovations (such as quieter ship propellers, car and aeroplane engines, tyres, and asphalt), modifications to standard operations (e.g. slower ship and vehicle speeds, traffic flow control, road closures), and sound barriers can significantly reduce noise levels in a particular habitat; however the benefits to wildlife are not fully understood. For example, how long does a road need to be closed for the biological community to recover from traffic noise? Do the unintended consequences of sound barriers (e.g. fragmentation or acoustically reflective surfaces) outweigh the benefits (Parris & Schneider, 2009)? Further, design and implementation of mitigation methods should match the timing and locations of biological activity, particularly during biologically sensitive periods, such as breeding (e.g. lekking behaviour in sage grouse *Centrocercus urophasianus*; Blickley *et al.*, 2012*a*,*b*) or seasonal movement (e.g. spring migration in cetaceans; Patenaude *et al.*, 2002).

(7) Improve reporting of acoustic metrics

Identifying the conditions that elicit biological responses is impossible without exposure information. Relevant details should include specification of acoustic metrics, temporal characteristics of the measurement (duration of recordings), frequency range measured, weighting filters applied, and the reference pressure used. Additionally, recording equipment and measurement procedures (distances and duration) should be documented for the source and received levels. Spectral descriptions or graphics provide important detail on the dominant frequencies of the noise source and can be compared to the hearing sensitivities of different species. The current state of the literature limits proper meta-analytical approaches that would allow compilation, comparison, and projection.

VI. CONCLUSIONS

(1) The substantial body of scientific research reviewed here provides considerable evidence that anthropogenic noise is detrimental to wildlife and natural ecosystems.

(2) Expertise from a diverse range of disciplines is required to improve understanding of the impacts associated with noise, especially considering that the effects may be expressed from the cellular to the ecosystem level.

(3) It is essential that research on the effects of anthropogenic noise evolves to report acoustic metrics accurately, test gradients of noise exposure, measure long-term consequences of responses to noise, assess cumulative effects of disturbance, investigate effectiveness of mitigation measures and recovery from chronic noise exposure, and fill in gaps with more diverse taxonomic groups and noise sources.

(4) We provide a cumulate weight-of-evidence summary of the recent literature, an initial step in providing guidance for natural resource managers when evaluating anthropogenic impacts or developing conservation policy.

(5) The interface between marine mammal research, regulation, and mitigation regarding noise provides an exemplar for controlling impacts for other taxa and ecosystems (Southall *et al.*, 2007; Stokstad, 2014). While the strides taken in the past decades have been impressive and provide a solid basis for shaping this critically important field of research, future activities should attempt to manage these impacts on temporal and spatial scales relevant to wildlife.

VII. ACKNOWLEDGEMENTS

The National Park Service, Natural Sounds and Night Skies Division funded this study. We thank two anonymous referees for their helpful comments and A. Cooper and W. Foster for their editorial assistance.

VIII. REFERENCES

**References marked with asterisk have been cited within the supporting information*

*American National Standards Institute-ANSI (2004). Acoustical terminology. ANSI S1.1-1994 (R2004). Available at http://www.ansi.org/ Accessed 15.08.2014.

American National Standards Institute-ANSI (2005). Quantities and procedures for description and measurement of environmental sound, noise assessment and prediction of long-term community response). ANSI S12.9-4. Available at http://www.ansi.org/ Accessed 15.08.2014.

*Amoser, S., Wysocki, L. E. & Ladich, F. (2004). Noise emission during the first powerboat race in an Alpine lake and potential impact on fish communities. *The Journal of the Acoustical Society of America* 116, 3789–3797.

*Anderson, P. A., Berzins, I. K., Fogarty, F., Hamlin, H. J. & Guillette, L. J. (2011). Sound, stress, and seahorses: the consequences of a noisy environment to animal health. *Aquaculture* 311, 129–138.

*André, M., Solé, M., Lenoir, M., Durfort, M., Quero, C., Mas, A., Lombarte, A., van der Schaar, M., López-Bejar, M., Morell, M., Zaugg, S. & Houégnigan, L. (2011). Low-frequency sounds induce acoustic trauma in cephalopods. *Frontiers in Ecology and the Environment* 9, 489–493.

Arévalo, J. E. & Newhard, K. (2011). Traffic noise affects forest bird species in a protected tropical forest. *Revista de Biologia Tropical* 59, 969–980.

Arroyo-Solís, A., Castillo, J. M., Figueroa, E., López-Sánchez, J. L. & Slabbekoorn, H. (2013). Experimental evidence for an impact of anthropogenic noise on dawn chorus timing in urban birds. *Journal of Avian Biology* 44, 288–296.

Babisch, W., Beule, B., Schust, M., Kersten, N. & Ising, H. (2005). Traffic noise and risk of myocardial infarction. *Epidemiology* 16, 33–40.

*Baldwin, A. L., Primeau, R. L. & Johnson, W. E. (2006). Effect of noise on the morphology of the intestinal mucosa in laboratory rats. *Journal of the American Association for Laboratory Animal Science* 45, 74–82.

Barber, J. R., Crooks, K. R. & Fristrup, K. M. (2010). The costs of chronic noise exposure for terrestrial organisms. *Trends in Ecology & Evolution* 25, 180–189.

Bayne, E. M., Habib, L. & Boutin, S. (2008). Impacts of chronic anthropogenic noise from energy-sector activity on abundance of songbirds in the boreal forest. *Conservation Biology* 22, 1186–1193.

*Becker, A., Whitfield, A. K., Cowley, P. D., Järnegren, J. & Næsje, T. F. (2013). Does boat traffic cause displacement of fish in estuaries? *Marine Pollution Bulletin* 75, 168–173.

Bee, M. A. & Swanson, E. M. (2007). Auditory masking of anuran advertisement calls by road traffic noise. *Animal Behaviour* 74, 1765–1776.

Bermúdez-Cuamatzin, E., Ríos-Chelén, A. A., Gil, D. & Garcia, C. M. (2009). Strategies of song adaptation to urban noise in the house finch: syllable pitch plasticity or differential syllable use? *Behaviour* 146, 1269–1286.

Bermúdez-Cuamatzin, E., Ríos-Chelén, A. A., Gil, D. & Garcia, C. M. (2011). Experimental evidence for real-time song frequency shift in response to urban noise in a passerine bird. *Biology Letters* 7, 36–38.

*Birke, L. (2002). Effects of browse, human visitors and noise on the behaviour of captive orang utans. *Animal Welfare* 11, 189–202.

Blackwell, S. B., Nations, C. S., McDonald, T. L., Greene, C. R., Thode, A. M., Guerra, M. & Michael Macrander, A. (2013). Effects of airgun sounds on bowhead whale calling rates in the Alaskan Beaufort Sea. *Marine Mammal Science* 29, 342–365.

Blickley, J. L., Blackwood, D. & Patricelli, G. L. (2012*a*). Experimental evidence for the effects of chronic anthropogenic noise on abundance of greater sage-grouse at leks. *Conservation Biology* 26, 461–471.

Blickley, J. L., Word, K. R., Krakauer, A. H., Phillips, J. L., Sells, S. N., Taff, C. C., Wingfield, J. C. & Patricelli, G. L. (2012*b*). Experimental chronic noise is related to elevated fecal corticosteroid metabolites in lekking male greater sage-grouse (*Centrocercus urophasianus*). *PLoS ONE* 7, e50462.

Bloom, D. E. (2011). 7 billion and counting. *Science* 333, 562–569.

*Bowles, A. E., Smultea, M., Würsig, B., DeMaster, D. P. & Palka, D. (1994). Relative abundance and behavior of marine mammals exposed to transmissions from the Heard Island Feasibility Test. *The Journal of the Acoustical Society of America* 96, 2469–2484.

Boyd, I., Brownell, B., Cato, D., Clarke, C., Costa, D., Evans, P. G. H., Gedamke, J., Gentry, R., Gisiner, B., Gordon, J., Jepson, P., Miller, P., Rendell, L., Tasker, M., Tyack, P., Vos, E., Whitehead, H., Wartzok, D. & Zimmer, W. (2008). *The Effects of Anthropogenic Sound on Marine Mammals – A Draft Research Strategy*. European Science Foundation and Marine Board, Oxford.

Bracciali, C., Campobello, D., Giacoma, C. & Sarà, G. (2012). Effects of nautical traffic and noise on foraging patterns of Mediterranean damselfish (*Chromis chromis*). *PLoS ONE* 7, e40582.

Brandt, M. J., Diederichs, A., Betke, K. & Nehls, G. (2011). Responses of harbour porpoises to pile driving at the Horns Rev II offshore wind farm in the Danish North Sea. *Marine Ecology Progress Series* 421, 205–216.

*Branstetter, B. K., Trickey, J. S., Bakhtiari, K., Black, A., Aihara, H. & Finneran, J. J. (2013). Auditory masking patterns in bottlenose dolphins (*Tursiops truncatus*) with natural, anthropogenic, and synthesized noise. *The Journal of the Acoustical Society of America* 133, 1811–1818.

*Bredeh, J. B., Hernández, F., Bingham, R. L., Silvy, N. J. & Waggerman, G. L. (2008). Effects of traffic noise on auditory surveys of urban White-winged Doves. *The Wilson Journal of Ornithology* 120, 384–389.

*Brown, A. L. (1990). Measuring the effect of aircraft noise on sea birds. *Environment International* 16, 587–592.

Brown, C. L., Hardy, A. R., Barber, J. R., Fristrup, K. M., Crooks, K. R. & Angeloni, L. M. (2012). The effect of human activities and their associated noise on ungulate behavior. *PLoS ONE* 7, e40505.

Bruintjes, R. & Radford, A. N. (2013). Context-dependent impacts of anthropogenic noise on individual and social behaviour in a cooperatively breeding fish. *Animal Behaviour* 85, 1343–1349.

Brumm, H. (2004). The impact of environmental noise on song amplitude in a territorial bird. *Journal of Animal Ecology* 73, 434–440.

Brumm, H. & Slabbekoorn, H. (2005). Acoustic communication in noise. *Advances in the Study of Behavior* 35, 151–209.

*Buckstaff, K. C. (2004). Effects of watercraft noise on the acoustic behavior of bottlenose dolphins, Tursiops truncatus, in Sarasota Bay, Florida. *Marine Mammal Science* 20, 709–725.

*Byrnes, P., Goosem, M. & Turton, S. M. (2012). Are less vocal rainforest mammals susceptible to impacts from traffic noise? *Wildlife Research* 39, 355–365.

Casper, B. M., Smith, M. E., Halvorsen, M. B., Sun, H., Carlson, T. J. & Popper, A. N. (2013). Effects of exposure to pile driving sounds on fish inner ear tissues. *Comparative Biochemistry and Physiology Part A: Molecular & Integrative Physiology* 166, 352–360.

*Castellote, M., Clark, C. W. & Lammers, M. O. (2012). Acoustic and behavioural changes by fin whales (*Balaenoptera physalus*) in response to shipping and airgun noise. *Biological Conservation* 147, 115–122.

Chan, A. A. Y. H., Giraldo-Perez, P., Smith, S. & Blumstein, D. T. (2010). Anthropogenic noise affects risk assessment and attention: the distracted prey hypothesis. *Biology Letters* 6, 458–461.

Chapman, D. M. & Ellis, D. D. (1998). The elusive decibel: thoughts on sonars and marine mammals. *Canadian Acoustics* 26, 29–31.

Chepesiuk, R. (2005). Decibel hell. *Environmental Health Perspectives* 113, 35–41.

Codarin, A., Wysocki, L. E., Ladich, F. & Picciulin, M. (2009). Effects of ambient and boat noise on hearing and communication in three fish species living in a marine protected area (Miramare, Italy). *Marine Pollution Bulletin* 58, 1880–1887.

Conomy, J. T., Dubovsky, J. A., Collazo, J. A. & Fleming, W. J. (1998). Do black ducks and wood ducks habituate to aircraft disturbance? *The Journal of Wildlife Management* 62, 1135–1142.

Crino, O. L., Johnson, E. E., Blickley, J. L., Patricelli, G. L. & Breuner, C. W. (2013). Effects of experimentally elevated traffic noise on nestling white-crowned sparrow stress physiology, immune function and life history. *The Journal of Experimental Biology* 216, 2055–2062.

*Croll, D. A., Clark, C. W., Calambokidis, J., Ellison, W. T. & Tershy, B. R. (2001). Effect of anthropogenic low-frequency noise on the foraging ecology of Balaenoptera whales. *Animal Conservation* 4, 13–27.

*Culik, B. M., Koschinski, S., Tregenza, N. & Ellis, G. M. (2001). Reactions of harbor porpoises *Phocoena phocoena* and herring *Clupea harengus* to acoustic alarms. *Marine Ecology Progress Series* 211, 255–260.

Cunnington, G. M. & Fahrig, L. (2010). Plasticity in the vocalizations of anurans in response to traffic noise. *Acta Oecologica* 36, 463–470.

Dähne, M., Gilles, A., Lucke, K., Peschko, V., Adler, S., Krügel, K., Sundermeyer, J. & Siebert, U. (2013). Effects of pile-driving on harbour porpoises (*Phocoena phocoena*) at the first offshore wind farm in Germany. *Environmental Research Letters* 8, 025002.

David, J. A. (2006). Likely sensitivity of bottlenose dolphins to pile-driving noise. *Water and Environment Journal* 20, 48–54.

*Delaney, D. K., Grubb, T. G., Beier, P., Pater, L. L. & Reiser, M. H. (1999). Effects of helicopter noise on Mexican spotted owls. *The Journal of Wildlife Management* 63, 60–76.

Delaney, D. K., Pater, L. L., Carlile, L. D., Spadgenske, E. W., Beaty, T. A. & Melton, R. H. (2011). Response of red-cockaded woodpeckers to military training operations. *Wildlife Monographs* 177, 1–38.

*De Robertis, A., Hjellvik, V., Williamson, N. J. & Wilson, C. D. (2008). Silent ships do not always encounter more fish: comparison of acoustic backscatter recorded by a noise-reduced and a conventional research vessel. *ICES Journal of Marine Science* 65, 623–635.

DeRuiter, S. L., Boyd, I. L., Claridge, D. E., Clark, C. W., Gagnon, C., Southall, B. L. & Tyack, P. L. (2013*a*). Delphinid whistle production and call matching during playback of simulated military sonar. *Marine Mammal Science* 29, 46–59.

DeRuiter, S. L., Southall, B. L., Calambokidis, J., Zimmer, W. M. X., Sadykova, D., Falcone, E. A., Friedlaender, A. S., Joseph, J. E., Moretti, D., Schorr, G. S., Thomas, L. & Tyack, P. L. (2013*b*). First direct measurements of behavioural responses by Cuvier's beaked whales to mid-frequency active sonar. *Biology Letters* 9, 20130223.

Díaz, M., Parra, A. & Gallardo, C. (2011). Serins respond to anthropogenic noise by increasing vocal activity. *Behavioral Ecology* 22, 332–336.

Di Iorio, L. & Clark, C. W. (2010). Exposure to seismic survey alters blue whale acoustic communication. *Biology Letters* 6, 51–54.

Doksæter, L., Handegard, N. O., Godø, O. R., Kvadsheim, P. H. & Nordlund, N. (2012). Behavior of captive herring exposed to naval sonar transmissions (1.0–1.6 kHz) throughout a yearly cycle. *The Journal of the Acoustical Society of America* 131, 1632–1642.

Doksæter, L., Rune Godø, O., Olav Handegard, N., Kvadsheim, P. H., Lam, F.-P. A., Donovan, C. & Miller, P. J. O. (2009). Behavioral responses of herring (*Clupea harengus*) to 1–2 and 6–7 kHz sonar signals and killer whale feeding sounds. *The Journal of the Acoustical Society of America* 125, 554–564.

*Doresky, J., Morgan, K., Ragsdale, L., Townsend, H., Barron, M. & West, M. (2001). Effects of military activity on reproductive success of red-cockaded woodpeckers. *Journal of Field Ornithology* 72, 305–311.

Dowling, J. L., Luther, D. A. & Marra, P. P. (2012). Comparative effects of urban development and anthropogenic noise on bird songs. *Behavioral Ecology* 23, 201–209.

*Duarte, M. H., Vecci, M. A., Hirsch, A. & Young, R. J. (2011). Noisy human neighbours affect where urban monkeys live. *Biology Letters* 7, 840–842.

*Egnor, S. R., Wickelgren, J. G. & Hauser, M. D. (2007). Tracking silence: adjusting vocal production to avoid acoustic interference. *Journal of Comparative Physiology A* 193, 477–483.

*Ellis, D. H., Ellis, C. H. & Mindell, D. P. (1991). Raptor responses to low-level jet aircraft and sonic booms. *Environmental Pollution* 74, 53–83.

Ellison, W. T., Southall, B. L., Clark, C. W. & Frankel, A. S. (2012). A new context-based approach to assess marine mammal behavioral responses to anthropogenic sounds. *Conservation Biology* 26, 21–28.

Engås, A., Løkkeborg, S., Ona, E. & Soldal, A. V. (1996). Effects of seismic shooting on local abundance and catch rates of cod (*Gadus morhua*) and haddock) (*Melanogrammus aeglefinus*). *Canadian Journal of Fisheries and Aquatic Sciences* 53, 2238–2249.

EPA (1974). Information on levels of environmental noise requisite to protect public health and welfare with an adequate margin of safety. EPA/ONAC 550/9-74-004. United States Environmental Protection Agency, Washington, DC.

Fewtrell, J. L. & McCauley, R. D. (2012). Impact of air gun noise on the behaviour of marine fish and squid. *Marine Pollution Bulletin* 64, 984–993.

*Filiciotto, F., Giacalone, V. M., Fazio, F., Buffa, G., Piccione, G., Maccarrone, V., Di Stefano, V., Mazzola, S. & Buscaino, G. (2013). Effect of acoustic environment on gilthead sea bream (*Sparus aurata*): sea and onshore aquaculture background noise. *Aquaculture* 414, 36–45.

Finneran, J. J., Schlundt, C. E., Carder, D. A., Clark, J. A., Young, J. A., Gaspin, J. B. & Ridgway, S. H. (2000). Auditory and behavioral responses of bottlenose dolphins (*Tursiops truncatus*) and a beluga whale (*Delphinapterus leucas*) to impulsive sounds resembling distant signatures of underwater explosions. *The Journal of the Acoustical Society of America* 108, 417–431.

Finneran, J. J., Schlundt, C. E., Dear, R., Carder, D. A. & Ridgway, S. H. (2002). Temporary shift in masked hearing thresholds in odontocetes after exposure to single underwater impulses from a seismic watergun. *The Journal of the Acoustical Society of America* 111, 2929–2940.

Fontana, C. S., Burger, M. I. & Magnusson, W. E. (2011). Bird diversity in a subtropical South-American City: effects of noise levels, arborisation and human population density. *Urban Ecosystems* 14, 341–360.

*Foote, A. D., Osborne, R. W. & Hoelzel, A. R. (2004). Environment: whale-call response to masking boat noise. *Nature* 428, 910.

Francis, C. D. & Barber, J. R. (2013). A framework for understanding noise impacts on wildlife: an urgent conservation priority. *Frontiers in Ecology and the Environment* 11, 305–313.

Francis, C. D., Kleist, N. J., Ortega, C. P. & Cruz, A. (2012). Noise pollution alters ecological services: enhanced pollination and disrupted seed dispersal. *Proceedings of the Royal Society of London B: Biological Sciences* 279, 2727–2735.

Francis, C. D., Ortega, C. P. & Cruz, A. (2009). Noise pollution changes avian communities and species interactions. *Current Biology* 19, 1415–1419.

*Francis, C. D., Ortega, C. P. & Cruz, A. (2011*a*). Different behavioural responses to anthropogenic noise by two closely related passerine birds. *Biology Letters* 7, 850–852.

*Francis, C. D., Ortega, C. P. & Cruz, A. (2011*b*). Noise pollution filters bird communities based on vocal frequency. *PLoS ONE* 6, e27052.

*Francis, C. D., Ortega, C. P. & Cruz, A. (2011*c*). Vocal frequency change reflects different responses to anthropogenic noise in two suboscine tyrant flycatchers. *Proceedings of the Royal Society B: Biological Sciences* 278, 2025–2031.

*Francis, C. D., Paritsis, J., Ortega, C. P. & Cruz, A. (2011*d*). Landscape patterns of avian habitat use and nest success are affected by chronic gas well compressor noise. *Landscape Ecology* 26, 1269–1280.

*Frantzis, A. (1998). Does acoustic testing strand whales? *Nature* 392, 29.

*Fristrup, K. M., Hatch, L. T. & Clark, C. W. (2003). Variation in humpback whale (*Megaptera novaeangliae*) song length in relation to low-frequency sound broadcasts. *The Journal of the Acoustical Society of America* 113, 3411–3424.

Fritschi, L., Brown, A. L., Kim, R., Schwela, D. H. & Kephalopoulos, S. (2011). *Burden of Disease from Environmental Noise. Quantification of Healthy Life Years Lost in Europe*. WHO European Centre for Environment and Health, Bonn Office, Copenhagen, Denmark. Available at http://www.who.int/quantifying_ehimpacts/publications/e94888/en/ Accessed 15.08.2014.

Fuller, R. A., Warren, P. H. & Gaston, K. J. (2007). Daytime noise predicts nocturnal singing in urban robins. *Biology Letters* 3, 368–370.

Fyhri, A. & Aasvang, G. M. (2010). Noise, sleep and poor health: modeling the relationship between road traffic noise and cardiovascular problems. *Science of the Total Environment* 408, 4935–4942.

*Gailey, G., Würsig, B. & McDonald, T. L. (2007). Abundance, behavior, and movement patterns of western gray whales in relation to a 3-D seismic survey, Northeast Sakhalin Island, Russia. *Environmental Monitoring and Assessment* 134, 75–91.

Gill, S. A., Job, J. R., Myers, K., Naghshineh, K. & Vonhof, M. J. (2015). Toward a broader characterization of anthropogenic noise and its effects on wildlife. *Behavioral Ecology* 26, 328–333.

Goerlitz, H. R., Greif, S. & Siemers, B. M. (2008). Cues for acoustic detection of prey: insect rustling sounds and the influence of walking substrate. *Journal of Experimental Biology* 211, 2799–2806.

Goldbogen, J. A., Southall, B. L., DeRuiter, S. L., Calambokidis, J., Friedlaender, A. S., Hazen, E. L., Falcone, E. A., Schorr, G. S., Douglas, A., Moretti, D. J., Kyburg, C., McKenna, M. F. & Tyack, P. L. (2013). Blue whales respond to simulated mid-frequency military sonar. *Proceedings of the Royal Society of London B: Biological Sciences* 280, 20130657.

González-Oreja, J. A., De La Fuente-Díaz, A. A., Hernández-Santín, L., Bonache-Regidor, C. & Buzo-Franco, D. (2012). Can human disturbance promote nestedness? Songbirds and noise in urban parks as a case study. *Landscape and Urban Planning* 104, 9–18.

Goodwin, S. E. & Podos, J. (2013). Shift of song frequencies in response to masking tones. *Animal Behaviour* 85, 435–440.

Goodwin, S. E. & Shriver, W. G. (2011). Effects of traffic noise on occupancy patterns of forest birds. *Conservation Biology* 25, 406–411.

Goold, J. C. (1996). Acoustic assessment of populations of common dolphin Delphinus delphis in conjunction with seismic surveying. *Journal of the Marine Biological Association of the United Kingdom* 76, 811–820.

Goudie, R. & Jones, I. L. (2004). Dose-response relationships of harlequin duck behaviour to noise from low-level military jet over-flights in central Labrador. *Environmental Conservation* 31, 289–298.

*Graham, A. L. & Cooke, S. J. (2008). The effects of noise disturbance from various recreational boating activities common to inland waters on the cardiac physiology of a freshwater fish, the largemouth bass (*Micropterus salmoides*). *Aquatic Conservation: Marine and Freshwater Ecosystems* 18, 1315–1324.

Gross, K., Pasinelli, G. & Kunc, H. P. (2010). Behavioral plasticity allows short-term adjustment to a novel environment. *The American Naturalist* 176, 456–464.

*Grubb, T. G., Pater, L. L., Gatto, A. E. & Delaney, D. K. (2013). Response of nesting northern goshawks to logging truck noise in northern Arizona. *The Journal of Wildlife Management* 77, 1618–1625.

*Gutscher, M., Wysocki, L. E. & Ladich, F. (2011). Effects of aquarium and pond noise on hearing sensitivity in an otophysine fish. *Bioacoustics* 20, 117–136.

Habib, L., Bayne, E. M. & Boutin, S. (2006). Chronic industrial noise affects pairing success and age structure of ovenbirds *Seiurus aurocapilla*. *Journal of Applied Ecology* 44, 176–184.

Hage, S. R., Jiang, T., Berquist, S. W., Feng, J. & Metzner, W. (2013). Ambient noise induces independent shifts in call frequency and amplitude within the Lombard effect in echolocating bats. *Proceedings of the National Academy of Sciences* 110, 4063–4068.

Halfwerk, W., Bot, S., Buikx, J., van der Velde, M., Komdeur, J., ten Cate, C. & Slabbekoorn, H. (2011*a*). Low-frequency songs lose their potency in noisy urban conditions. *Proceedings of the National Academy of Sciences* 108, 14549–14554.

Halfwerk, W., Holleman, L. J., Lessells, C. K. & Slabbekoorn, H. (2011*b*). Negative impact of traffic noise on avian reproductive success. *Journal of Applied Ecology* 48, 210–219.

Halfwerk, W. & Slabbekoorn, H. (2009). A behavioural mechanism explaining noise-dependent frequency use in urban birdsong. *Animal Behaviour* 78, 1301–1307.

Halpern, B. S., Walbridge, S., Selkoe, K. A., Kappel, C. V., Micheli, F., D'Agrosa, C., Bruno, J. F., Casey, K. S., Ebert, C., Fox, H. E., Fujita, R., Heinemann, D., Lenihan, H. S., Madin, E. M. P., et al. (2008). A global map of human impact on marine ecosystems. *Science* 319, 948–952.

Halvorsen, M. B., Zeddies, D. G., Chicoine, D. & Popper, A. N. (2013). Effects of low-frequency naval sonar exposure on three species of fish. *The Journal of the Acoustical Society of America* 134, 205–210.

Hansell, A. L., Blangiardo, M., Fortunato, L., Floud, S., de Hoogh, K., Fecht, D., Ghosh, R. E., Laszlo, H. E., Pearson, C., Beale, L., Beevers, S., Gulliver, J., Best, N., Richardson, S. & Elliott, P. (2013). Aircraft noise and cardiovascular disease near Heathrow airport in London: small area study. *British Medical Journal* 347, f5432.

*Harris, R. E., Miller, G. W. & Richardson, W. J. (2001). Seal responses to airgun sounds during summer seismic surveys in the Alaskan Beaufort Sea. *Marine Mammal Science* 17, 795–812.

Hatch, L. T., Clark, C. W., Van Parijs, S. M., Frankel, A. S. & Ponirakis, D. W. (2012). Quantifying loss of acoustic communication space for right whales in and around a U.S. National Marine Sanctuary. *Conservation Biology* 26, 983–994.

*Herrera-Montes, M. I. & Aide, T. M. (2011). Impacts of traffic noise on anuran and bird communities. *Urban Ecosystems* 14, 415–427.

Hildebrand, J. A. (2009). Anthropogenic and natural sources of ambient noise in the ocean. *Marine Ecology Progress Series* 395, 5–20.

Holles, S., Simpson, S. D., Radford, A. N., Berten, L. & Lecchini, D. (2013). Boat noise disrupts orientation behaviour in a coral reef fish. *Marine Ecology Progress Series* 485, 295–300.

Holt, M. M., Noren, D. P. & Emmons, C. K. (2011). Effects of noise levels and call types on the source levels of killer whale calls. *The Journal of the Acoustical Society of America* 130, 3100–3106.

Holt, M. M., Noren, D. P., Veirs, V., Emmons, C. K. & Veirs, S. (2009). Speaking up: killer whales (*Orcinus orca*) increase their call amplitude in response to vessel noise. *The Journal of the Acoustical Society of America* 125, 27–32.

Hotchkin, C. & Parks, S. (2013). The Lombard effect and other noise-induced vocal modifications: insight from mammalian communication systems. *Biological Reviews* 4, 809–824.

Houser, D. S., Martin, S. W. & Finneran, J. J. (2013a). Behavioral responses of California sea lions to mid-frequency (3250–3450 Hz) sonar signals. *Marine Environmental Research* **92**, 268–278.

Houser, D. S., Martin, S. W. & Finneran, J. J. (2013b). Exposure amplitude and repetition affect bottlenose dolphin behavioral responses to simulated mid-frequency sonar signals. *Journal of Experimental Marine Biology and Ecology* **443**, 123–133.

*Hovem, J. M., Tronstad, T. V., Karlsen, H. E. & Lokkeborg, S. (2012). Modeling propagation of seismic airgun sounds and the effects on fish behavior. *IEEE Journal of Oceanic Engineering* **37**, 576–588.

*Hu, Y. & Cardoso, G. C. (2009). Are bird species that vocalize at higher frequencies preadapted to inhabit noisy urban areas? *Behavioral Ecology* **20**, 1268–1273.

Hu, Y. & Cardoso, G. C. (2010). Which birds adjust the frequency of vocalizations in urban noise? *Animal Behaviour* **79**, 863–867.

*Hulfeldt, N. P. & Dabelsteen, T. (2013). Impact of a noise-polluted urban environment on the song frequencies of a cosmopolitan songbird, the Great Tit (*Parus major*), in Denmark. *Ornis Fennica* **90**, 94–102.

*Iglesias, C., Mata, C. & Malo, J. E. (2012). The influence of traffic noise on vertebrate road crossing through underpasses. *Ambio* **41**, 193–201.

Jenkins, C. N., Pimm, S. L. & Joppa, L. N. (2013). Global patterns of terrestrial vertebrate diversity and conservation. *Proceedings of the National Academy of Sciences of the United States of America* **110**, 2602–2610.

*Jung, C. A. & Swearer, S. E. (2011). Reactions of temperate reef fish larvae to boat sound. *Aquatic Conservation: Marine and Freshwater Ecosystems* **21**, 389–396.

Kaiser, K., Scofield, D. G., Alloush, M., Jones, R. M., Marczak, S., Martineau, K., Oliva, M. A. & Narins, P. M. (2011). When sounds collide: the effect of anthropogenic noise on a breeding assemblage of frogs in Belize, Central America. *Behaviour* **148**, 215–232.

*Kane, A. S., Song, J., Halvorsen, M. B., Miller, D. L., Salierno, J. D., Wysocki, L. E., Zeddies, D. & Popper, A. N. (2010). Exposure of fish to high-intensity sonar does not induce acute pathology. *Journal of Fish Biology* **76**, 1825–1840.

*Karp, D. S. & Guevara, R. (2011). Conversational noise reduction as a win–win for ecotourists and rain forest birds in Peru. *Biotropica* **43**, 122–130.

Karp, D. S. & Root, T. L. (2009). Sound the stressor: how Hoatzins (*Opisthocomus hoazin*) react to ecotourist conversation. *Biodiversity and Conservation* **18**, 3733–3742.

Kaseloo, P. A. & Tyson, K. O. (2004). *Synthesis of Noise Effects on Wildlife Populations*. US Department of Transportation, Federal Highway Administration, Washington, DC.

*Kastak, D., Schusterman, R. J., Southall, B. L. & Reichmuth, C. J. (1999). Underwater temporary threshold shift induced by octave-band noise in three species of pinniped. *The Journal of the Acoustical Society of America* **106**, 1142–1148.

*Kastelein, R. A., De Haan, D., Vaughan, N., Staal, C. & Schooneman, N. M. (2001). The influence of three acoustic alarms on the behaviour of harbour porpoises (*Phocoena phocoena*) in a floating pen. *Marine Environmental Research* **52**, 351–371.

Kastelein, R. A., Hoek, L. & de Jong, C. A. (2011a). Hearing thresholds of a harbor porpoise (*Phocoena phocoena*) for helicopter dipping sonar signals (1.43–1.33 kHz) (L). *The Journal of the Acoustical Society of America* **130**, 679–682.

Kastelein, R. A., Hoek, L. & de Jong, C. A. (2011b). Hearing thresholds of a harbor porpoise (*Phocoena phocoena*) for sweeps (1–2 kHz and 6–7 kHz bands) mimicking naval sonar signals. *The Journal of the Acoustical Society of America* **129**, 3393–3399.

Kastelein, R. A., van der Heul, S., van der Veen, J., Verboom, W. C., Jennings, N., de Haan, D. & Reijnders, P. J. (2007). Effects of acoustic alarms, designed to reduce small cetacean bycatch in gillnet fisheries, on the behaviour of North Sea fish species in a large tank. *Marine Environmental Research* **64**, 160–180.

*Kastelein, R. A., van der Heul, S., Verboom, W. C., Jennings, N., van der Veen, J. & de Haan, D. (2008). Startle response of captive North Sea fish species to underwater tones between 0.1 and 64 kHz. *Marine Environmental Research* **65**, 369–377.

Kastelein, R. A., van der Heul, S., Verboom, W. C., Triesscheijn, R. J. & Jennings, N. V. (2006). The influence of underwater data transmission sounds on the displacement behaviour of captive harbour seals (*Phoca vitulina*). *Marine Environmental Research* **61**, 19–39.

*Kastelein, R. A., Verboom, W. C., Muijsers, M., Jennings, N. V. & Van der Heul, S. (2005). The influence of acoustic emissions for underwater data transmission on the behaviour of harbour porpoises (*Phocoena phocoena*) in a floating pen. *Marine Environmental Research* **59**, 287–307.

Kight, C. R., Saha, M. S. & Swaddle, J. P. (2012). Anthropogenic noise is associated with reductions in the productivity of breeding Eastern bluebirds (*Sialia sialis*). *Ecological Applications* **22**, 1989–1996.

Kight, C. R. & Swaddle, J. P. (2011). How and why environmental noise impacts animals: an integrative, mechanistic review. *Ecology Letters* **14**, 1052–1061.

*Klinck, H., Nieukirk, S. L., Mellinger, D. K., Klinck, K., Matsumoto, H. & Dziak, R. P. (2012). Seasonal presence of cetaceans and ambient noise levels in polar waters of the North Atlantic. *The Journal of the Acoustical Society of America* **132**, 176–181.

*Koschinski, S., Culik, B. M., Henriksen, O. D., Tregenza, N., Ellis, G., Jansen, C. & Kathe, G. (2003). Behavioural reactions of free-ranging porpoises and seals to the noise of a simulated 2 MW windpower generator. *Marine Ecology Progress Series* **265**, 263–273.

*Krausman, P. R., Harris, L. K., Blasch, C. L., Koenen, K. K. & Francine, J. (2004). Effects of military operations on behavior and hearing of endangered Sonoran pronghorn. *Wildlife Monographs* **157**, 1–41.

Krausman, P. R., Wallace, M. C., Hayes, C. L. & DeYoung, D. W. (1998). Effects of jet aircraft on mountain sheep. *The Journal of Wildlife Management* **62**, 1246–1254.

Kuningas, S., Kvadsheim, P. H., Lam, F. P. A. & Miller, P. J. O. (2013). Killer whale presence in relation to naval sonar activity and prey abundance in northern Norway. *ICES Journal of Marine Science* **70**, 1287–1293.

*Kvadsheim, P. H., Sevaldsen, E. M., Folkow, L. P. & Blix, A. S. (2010). Behavioural and physiological responses of hooded seals (*Cystophora cristata*) to 1 to 7 kHz sonar signals. *Aquatic Mammals* **36**, 239–247.

*Ladich, F. & Schulz-Mirbach, T. (2013). Hearing in cichlid fishes under noise conditions. *PLoS ONE* **8**, e57588.

*La Manna, G., Manghi, M., Pavan, G., Lo Mascolo, F. & Sarà, G. (2013). Behavioural strategy of common bottlenose dolphins (*Tursiops truncatus*) in response to different kinds of boats in the waters of Lampedusa Island (Italy). *Aquatic Conservation: Marine and Freshwater Ecosystems* **23**, 745–757.

Lampe, U., Schmoll, T., Franzke, A. & Reinhold, K. (2012). Staying tuned: grasshoppers from noisy roadside habitats produce courtship signals with elevated frequency components. *Functional Ecology* **26**, 1348–1354.

*Leader, N., Wright, J. & Yom-Tov, Y. (2005). Acoustic properties of two urban song dialects in the orange-tufted sunbird (*Nectarinia osea*). *The Auk* **122**, 231–245.

Leighton, T. G. (2012). How can humans, in air, hear sound generated underwater (and can goldfish hear their owners talking)? *The Journal of the Acoustical Society of America* **131**, 2539–2542.

Lemon, M., Lynch, T. P., Cato, D. H. & Harcourt, R. G. (2006). Response of travelling bottlenose dolphins (*Tursiops aduncus*) to experimental approaches by a powerboat in Jervis Bay, New South Wales, Australia. *Biological Conservation* **127**, 363–372.

Lengagne, T. (2008). Traffic noise affects communication behaviour in a breeding anuran, *Hyla arborea*. *Biological Conservation* **141**, 2023–2031.

*Leonard, M. L. & Horn, A. G. (2008). Does ambient noise affect growth and begging call structure in nestling birds? *Behavioral Ecology* **19**, 502–507.

*Leonard, M. L. & Horn, A. G. (2012). Ambient noise increases missed detections in nestling birds. *Biology Letters* **8**, 530–532.

*Le Roux, D. S. L. & Waas, J. R. (2012). Do long-tailed bats alter their evening activity in response to aircraft noise? *Acta Chiropterologica* **14**, 111–120.

*Lesage, V., Barrette, C., Kingsley, M. & Sjare, B. (1999). The effect of vessel noise on the vocal behavior of belugas in the St. Lawrence River estuary, Canada. *Marine Mammal Science* **15**, 65–84.

Liu, M., Wei, Q. W., Du, H., Fu, Z. Y. & Chen, Q. C. (2013). Ship noise-induced temporary hearing threshold shift in the Chinese sucker *Myxocyprinus asiaticus* (Bleeker, 1864). *Journal of Applied Ichthyology* **29**, 1416–1422.

*Lowry, H., Lill, A. & Wong, B. (2011). Tolerance of auditory disturbance by an avian urban adapter, the noisy miner. *Ethology* **117**, 490–497.

Lowry, H., Lill, A. & Wong, B. B. (2012). How noisy does a noisy miner have to be? Amplitude adjustments of alarm calls in an avian urban 'adapter'. *PLoS ONE* **7**, e29960.

Luther, D. & Baptista, L. (2010). Urban noise and the cultural evolution of bird songs. *Proceedings of the Royal Society B: Biological Sciences* **277**, 469–473.

*Luther, D. A. & Derryberry, E. P. (2012). Birdsongs keep pace with city life: changes in song over time in an urban songbird affects communication. *Animal Behaviour* **83**, 1059–1066.

Luther, D. & Gentry, K. (2013). Sources of background noise and their influence on vertebrate acoustic communication. *Behaviour* **150**, 1045–1068.

Madsen, P. T. (2005). Marine mammals and noise: problems with root mean square sound pressure levels for transients. *The Journal of the Acoustical Society of America* **117**, 3952–3957.

*Madsen, P. T. & Møhl, B. (2000). Sperm whales (*Physeter catodon* L. 1758) do not react to sounds from detonators. *The Journal of the Acoustical Society of America* **107**, 668–671.

*Madsen, P. T., Møhl, B., Nielsen, B. K. & Wahlberg, M. (2002). Male sperm whale behaviour during exposures to distant seismic survey pulses. *Aquatic Mammals* **28**, 231–240.

*Maes, J. H. R. & de Groot, G. (2003). Effects of noise on the performance of rats in an operant discrimination task. *Behavioural Processes* **61**, 57–68.

Maier, J. A., Murphy, S. M., White, R. G. & Smith, M. D. (1998). Responses of caribou to overflights by low-altitude jet aircraft. *The Journal of Wildlife Management* 62, 752–766.

McCarthy, E., Moretti, D., Thomas, L., DiMarzio, N., Morrissey, R., Jarvis, S., Ward, J., Izzi, A. & Dilley, A. (2011). Changes in spatial and temporal distribution and vocal behavior of Blainville's beaked whales (*Mesoplodon densirostris*) during multiship exercises with mid-frequency sonar. *Marine Mammal Science* 27, 206–226.

McCauley, R. D., Fewtrell, J. & Popper, A. N. (2003). High intensity anthropogenic sound damages fish ears. *The Journal of the Acoustical Society of America* 113, 638–642.

McClure, C. J. W., Ware, H. E., Carlisle, J., Kaltenecker, G. & Barber, J. R. (2013). An experimental investigation into the effects of traffic noise on distributions of birds: avoiding the phantom road. *Proceedings of the Royal Society of London B: Biological Sciences* 280, 20132290.

*McDonald, M. A., Hildebrand, J. A. & Webb, S. C. (1995). Blue and fin whales observed on a seafloor array in the Northeast Pacific. *The Journal of the Acoustical Society of America* 98, 712–721.

McGregor, P. K., Horn, A. G., Leonard, M. L. & Thomsen, F. (2013). Anthropogenic noise and conservation. In *Animal Signals and Communication*, pp. 409–444. Springer, Berlin, Heidelberg.

McLaughlin, K. E. & Kunc, H. P. (2013). Experimentally increased noise levels change spatial and singing behaviour. *Biology Letters* 9, 20120771.

Melcón, M. L., Cummins, A. J., Kerosky, S. M., Roche, L. K., Wiggins, S. M. & Hildebrand, J. A. (2012). Blue whales respond to anthropogenic noise. *PLoS ONE* 7, e32681.

Mendes, S., Colino-Rabanal, V. J. & Peris, S. J. (2011). Bird song variations along an urban gradient: the case of the European blackbird (*Turdus merula*). *Landscape and Urban Planning* 99, 51–57.

Mennitt, D., Fristrup, K. M. & Nelson, L. (2013). Mapping the extent of noise on a national scale. http://acoustics.org/pressroom/httpdocs/166th/4aNS5-Mennitt.html Accessed 15.08.2014.

*Merrill, S. B. & Erickson, C. R. (2003). A GPS-based method to examine wolf response to loud noise. *Wildlife Society Bulletin* 31, 769–773.

*Miksis-Olds, J. L., Donaghay, P. L., Miller, J. H., Tyack, P. L. & Reynolds, J. E. (2007). Simulated vessel approaches elicit differential responses from manatees. *Marine Mammal Science* 23, 629–649.

Miksis-Olds, J. L. & Wagner, T. (2011). Behavioral response of manatees to variations in environmental sound levels. *Marine Mammal Science* 27, 130–148.

*Miller, P. J., Biassoni, N., Samuels, A. & Tyack, P. L. (2000). Whale songs lengthen in response to sonar. *Nature* 405, 903.

Mockford, E. J. & Marshall, R. C. (2009). Effects of urban noise on song and response behaviour in great tits. *Proceedings of the Royal Society of London B: Biological Sciences* 276, 2979–2985.

Mockford, E. J., Marshall, R. C. & Dabelsteen, T. (2011). Degradation of rural and urban great tit song: testing transmission efficiency. *PLoS ONE* 6, e28242.

Montague, M. J., Danek-Gontard, M. & Kunc, H. P. (2013). Phenotypic plasticity affects the response of a sexually selected trait to anthropogenic noise. *Behavioral Ecology* 24(2), 343–348.

Mooney, T. A., Nachtigall, P. E. & Vlachos, S. (2009). Sonar-induced temporary hearing loss in dolphins. *Biology Letters* 5, 565–567.

Moore, S. E., Reeves, R. R., Southall, B. L., Ragen, T. J., Suydam, R. S. & Clark, C. W. (2012). A new framework for assessing the effects of anthropogenic sound on marine mammals in a rapidly changing arctic. *BioScience* 62, 289–295.

Morley, E. L., Jones, G. & Radford, A. N. (2013). The importance of invertebrates when considering the impacts of anthropogenic noise. *Proceedings of the Royal Society of London B: Biological Sciences* 281, 20132683.

*Morton, A. B. & Symonds, H. K. (2002). Displacement of Orcinus orca (L.) by high amplitude sound in British Columbia, Canada. *ICES Journal of Marine Science* 59, 71–80.

Murphy, E. & King, E. A. (2011). Scenario analysis and noise action planning: modelling the impact of mitigation measures on population exposure. *Applied Acoustics* 72, 487–494.

*Nachtigall, P. E., Supin, A. Y., Pawloski, J. & Au, W. W. (2004). Temporary threshold shifts after noise exposure in the bottlenose dolphin (*Tursiops truncatus*) measured using evoked auditory potentials. *Marine Mammal Science* 20, 673–687.

*Naguib, M., van Oers, K., Braakhuis, A., Griffioen, M., de Goede, P. & Waas, J. R. (2013). Noise annoys: effects of noise on breeding great tits depend on personality but not on noise characteristics. *Animal Behaviour* 85, 949–956.

Nemeth, E. & Brumm, H. (2009). Blackbirds sing higher-pitched songs in cities: adaptation to habitat acoustics or side-effect of urbanization? *Animal Behaviour* 78, 637–641.

Nemeth, E. & Brumm, H. (2010). Birds and anthropogenic noise: are urban songs adaptive? *The American Naturalist* 176, 465–475.

Nemeth, E., Pieretti, N., Zollinger, S. A., Geberzahn, N., Partecke, J., Miranda, A. C. & Brumm, H. (2013). Bird song and anthropogenic noise: vocal constraints may explain why birds sing higher-frequency songs in cities. *Proceedings of the Royal Society of London B: Biological Sciences* 280, 20122798.

Nowacek, D. P., Thorne, L. H., Johnston, D. W. & Tyack, P. L. (2007). Responses of cetaceans to anthropogenic noise. *Mammal Review* 37, 81–115.

Ortega, C. P. (2012). Effects of noise pollution on birds: a brief review of our knowledge. *Ornithological Monographs* 74, 6–22.

*Owen, M. A., Swaisgood, R. R., Czekala, N. M., Steinman, K. & Lindburg, D. G. (2004). Monitoring stress in captive giant pandas (*Ailuropoda melanoleuca*): behavioral and hormonal responses to ambient noise. *Zoo Biology* 23, 147–164.

*Owens, J. L., Stec, C. L. & O'Hatnick, A. (2012). The effects of extended exposure to traffic noise on parid social and risk-taking behavior. *Behavioural Processes* 91, 61–69.

*Parks, S. E., Clark, C. W. & Tyack, P. L. (2007). Short- and long-term changes in right whale calling behavior: the potential effects of noise on acoustic communication. *The Journal of the Acoustical Society of America* 122, 3725–3731.

Parks, S. E., Johnson, M., Nowacek, D. & Tyack, P. L. (2011). Individual right whales call louder in increased environmental noise. *Biology Letters* 7, 33–35.

*Parks, S. E., Urazghildiiev, I. & Clark, C. W. (2009). Variability in ambient noise levels and call parameters of North Atlantic right whales in three habitat areas. *The Journal of the Acoustical Society of America* 125, 1230–1239.

Parris, K. M. & McCarthy, M. A. (2013). Predicting the effect of urban noise on the active space of avian vocal signals. *The American Naturalist* 182, 452–464.

Parris, K. M. & Schneider, A. (2009). Impacts of traffic noise and traffic volume on birds of roadside habitats. *Ecology and Society* 14, 29.

Parris, K. M., Velik-Lord, M., North, J. M. & Function, L. (2009). Frogs call at a higher pitch in traffic noise. *Ecology and Society* 14, 25.

Patenaude, N. J., Richardson, W. J., Smultea, M. A., Koski, W. R., Miller, G. W., Würsig, B. & Greene, C. R. (2002). Aircraft sound and disturbance to bowhead and beluga whales during spring migration in the Alaskan Beaufort Sea. *Marine Mammal Science* 18(2), 309–335.

Patón, D., Romero, F., Cuenca, J. & Escudero, J. C. (2012). Tolerance to noise in 91 bird species from 27 urban gardens of Iberian Peninsula. *Landscape and Urban Planning* 104, 1–8.

Patricelli, G. L. & Blickley, J. L. (2006). Avian communication in urban noise: causes and consequences of vocal adjustment. *The Auk* 123, 639–649.

*Payne, C. J., Jessop, T. S., Guay, P. J., Johnstone, M., Feore, M. & Mulder, R. A. (2012). Population, behavioural and physiological responses of an urban population of black swans to an intense annual noise event. *PLoS ONE* 7, e45014.

*Penna, M. & Hamilton-West, C. (2007). Susceptibility of evoked vocal responses to noise exposure in a frog of the temperate austral forest. *Animal Behaviour* 74, 45–56.

*Penna, M., Pottstock, H. & Velasquez, N. (2005). Effect of natural and synthetic noise on evoked vocal responses in a frog of the temperate austral forest. *Animal Behaviour* 70, 639–651.

*Perry, E. A., Boness, D. J. & Insley, S. J. (2002). Effects of sonic booms on breeding gray seals and harbor seals on Sable Island, Canada. *The Journal of the Acoustical Society of America* 111, 599–609.

Picciulin, M., Sebastianutto, L., Codarin, A., Farina, A. & Ferrero, E. A. (2010). In situ behavioural responses to boat noise exposure of *Gobius cruentatus* (Gmelin, 1789; fam. *Gobiidae*) and *Chromis chromis* (Linnaeus, 1758; fam. *Pomacentridae*) living in a Marine Protected Area. *Journal of Experimental Marine Biology and Ecology* 386, 125–132.

Pieretti, N. & Farina, A. (2013). Application of a recently introduced index for acoustic complexity to an avian soundscape with traffic noise. *The Journal of the Acoustical Society of America* 134, 891–900.

Pine, M. K., Jeffs, A. G. & Radford, C. A. (2012). Turbine sound may influence the metamorphosis behaviour of estuarine crab megalopae. *PLoS ONE* 7, e51790.

*Pirotta, E., Milor, R., Quick, N., Moretti, D., Di Marzio, N., Tyack, P., Boyd, I. & Hastie, G. (2012). Vessel noise affects beaked whale behavior: results of a dedicated acoustic response study. *PLoS ONE* 7, e42535.

Pohl, N. U., Leadbeater, E., Slabbekoorn, H., Klump, G. M. & Langemann, U. (2012). Great tits in urban noise benefit from high frequencies in song detection and discrimination. *Animal Behaviour* 83, 711–721.

Pohl, N. U., Slabbekoorn, H., Klump, G. M. & Langemann, U. (2009). Effects of signal features and environmental noise on signal detection in the great tit, *Parus major*. *Animal Behaviour* 78, 1293–1300.

van *Polanen Petel, T. D., Terhune, J. M., Hindell, M. A. & Giese, M. A. (2006). An assessment of the audibility of sound from human transport by breeding Weddell seals (*Leptonychotes weddellii*). *Wildlife Research* 33, 275–291.

Popper, A. N., Halvorsen, M. B., Kane, A., Miller, D. L., Smith, M. E., Song, J., Stein, P. & Wysocki, L. E. (2007). The effects of high-intensity, low-frequency active sonar on rainbow trout. *The Journal of the Acoustical Society of America* 122, 623–635.

*Popper, A. N., Smith, M. E., Cott, P. A., Hanna, B. W., MacGillivray, A. O., Austin, M. E. & Mann, D. A. (2005). Effects of exposure to seismic airgun use on hearing of three fish species. *The Journal of the Acoustical Society of America* 117, 3958–3971.

Potvin, D. A. & Mulder, R. A. (2013). Immediate, independent adjustment of call pitch and amplitude in response to varying background noise by silvereyes (*Zosterops lateralis*). *Behavioral Ecology* 24, 1363–1368.

Potvin, D. A., Parris, K. M. & Mulder, R. A. (2011). Geographically pervasive effects of urban noise on frequency and syllable rate of songs and calls in silvereyes (*Zosterops lateralis*). *Proceedings of the Royal Society of London B: Biological Sciences* 278, 2464–2469.

Powell, D. M., Carlstead, K., Tarou, L. R., Brown, J. L. & Monfort, S. L. (2006). Effects of construction noise on behavior and cortisol levels in a pair of captive giant pandas (*Ailuropoda melanoleuca*). *Zoo Biology* 25, 391–408.

Proppe, D. S., Avey, M. T., Hoeschele, M., Moscicki, M. K., Farrell, T., St Clair, C. C. & Sturdy, C. B. (2012). Black-capped chickadees *Poecile atricapillus* sing at higher pitches with elevated anthropogenic noise, but not with decreasing canopy cover. *Journal of Avian Biology* 43, 325–332.

Proppe, D. S., Byers, K. A., Sturdy, C. B. & St Clair, C. C. (2013*a*). Physical condition of Black-capped Chickadees (*Poecile atricapillus*) in relation to road disturbance. *Canadian Journal of Zoology* 91, 842–845.

Proppe, D. S., Sturdy, C. B. & St Clair, C. C. (2013*b*). Anthropogenic noise decreases urban songbird diversity and may contribute to homogenization. *Global Change Biology* 19, 1075–1084.

Quinn, J. L., Whittingham, M. J., Butler, S. J. & Cresswell, W. (2006). Noise, predation risk compensation and vigilance in the chaffinch *Fringilla coelebs*. *Journal of Avian Biology* 37, 601–608.

Rabin, L. A., Coss, R. G. & Owings, D. H. (2006). The effects of wind turbines on antipredator behavior in California ground squirrels (*Spermophilus beecheyi*). *Biological Conservation* 131, 410–420.

Radford, A. N., Kerridge, E. & Simpson, S. D. (2014). Acoustic communication in a noisy world: can fish compete with anthropogenic noise? *Behavioral Ecology* 25, 1022–1030.

Rasmussen, S., Glickman, G., Norinsky, R., Quimby, F. W. & Tolwani, R. J. (2009). Construction noise decreases reproductive efficiency in mice. *Journal of the American Association for Laboratory Animal Science* 48, 363–370.

Read, J., Jones, G. & Radford, A. N. (2014). Fitness costs as well as benefits are important when considering responses to anthropogenic noise. *Behavioral Ecology* 25, 4–7.

Redondo, P., Barrantes, G. & Sandoval, L. (2013). Urban noise influences vocalization structure in the house wren *Troglodytes aedon*. *Ibis* 155, 621–625.

*Rendell, L. E. & Gordon, J. C. D. (1999). Vocal responses of long-finned pilot whales (*Globicephala melas*) to military sonar in the Ligurian sea. *Marine Mammal Science* 15, 198–204.

*Rheindt, F. E. (2003). The impact of roads on birds: does song frequency play a role in determining susceptibility to noise pollution? *Journal für Ornithologie* 144, 295–306.

Richardson, W. J., Greene, C. R. Jr., Malme, C. I. & Thomson, D. H. (1995). *Marine Mammals and Noise*. Academic Press, San Diego.

Ríos-Chelén, A. A., Quirós-Guerrero, E., Gil, D. & Garcia, C. M. (2013). Dealing with urban noise: vermilion flycatchers sing longer songs in noisier territories. *Behavioral Ecology and Sociobiology* 67, 145–152.

Ríos-Chelén, A. A., Salaberria, C., Barbosa, I., Macías Garcia, C. & Gil, D. (2012). The learning advantage: bird species that learn their song show a tighter adjustment of song to noisy environments than those that do not learn. *Journal of Evolutionary Biology* 25, 2171–2180.

*Ripmeester, E. A., Kok, J. S., van Rijssel, J. C. & Slabbekoorn, H. (2010). Habitat-related birdsong divergence: a multi-level study on the influence of territory density and ambient noise in European blackbirds. *Behavioral Ecology and Sociobiology* 64, 409–418.

Risch, D., Corkeron, P. J., Ellison, W. T. & Van Parijs, S. M. (2012). Changes in humpback whale song occurrence in response to an acoustic source 200 km away. *PLoS ONE* 7, e29741.

Rolland, R. M., Parks, S. E., Hunt, K. E., Castellote, M., Corkeron, P. J., Nowacek, D. P., Wasser, S. K. & Kraus, S. D. (2012). Evidence that ship noise increases stress in right whales. *Proceedings of the Royal Society of London B: Biological Sciences* 279, 2363–2368.

*Romano, T. A., Keogh, M. J., Kelly, C., Feng, P., Berk, L., Schlundt, C. E., Carder, D. A. & Finneran, J. J. (2004). Anthropogenic sound and marine mammal health: measures of the nervous and immune systems before and after intense sound exposure. *Canadian Journal of Fisheries and Aquatic Sciences* 61, 1124–1134.

Rossing, T. D. (2007). *Springer Handbook of Acoustics*. Springer, New York.

*Saha, D. C. & Padhy, P. K. (2011). Effect of air and noise pollution on species diversity and population density of forest birds at Lalpahari, West Bengal, India. *Science of the Total Environment* 409, 5328–5336.

*Salaberria, S. & Gill, D. (2010). Increase in song frequency in response to urban noise in the great tit Parus major as shown by data from the Madrid (Spain) city noise map. *Ardeola* 57, 3–11.

Sarà, G., Dean, J. M., D'Amato, D., Buscaino, G., Oliveri, A., Genovese, S., Ferro, S., Buffa, G., Lo Martire, M. & Mazzola, S. (2007). Effect of boat noise on the behaviour of bluefin tuna *Thunnus thynnus* in the Mediterranean Sea. *Marine Ecology Progress Series* 331, 243–253.

Schaub, A., Ostwald, J. & Siemers, B. M. (2008). Foraging bats avoid noise. *Journal of Experimental Biology* 211, 3174–3180.

Scheifele, P. M., Andrew, S., Cooper, R. A., Darre, M., Musiek, F. E. & Max, L. (2005). Indication of a Lombard vocal response in the St. Lawrence River beluga. *The Journal of the Acoustical Society of America* 117, 1486–1492.

*Scholik, A. R. & Yan, H. Y. (2002*a*). Effects of boat engine noise on the auditory sensitivity of the fathead minnow, *Pimephales promelas*. *Environmental Biology of Fishes* 63, 203–209.

*Scholik, A. R. & Yan, H. Y. (2002*b*). The effects of noise on the auditory sensitivity of the bluegill sunfish, Lepomis macrochirus. *Comparative Biochemistry and Physiology Part A: Molecular & Integrative Physiology* 133, 43–52.

Schroeder, J., Nakagawa, S., Cleasby, I. R. & Burke, T. (2012). Passerine birds breeding under chronic noise experience reduced fitness. *PLoS ONE* 7, e39200.

Sebastianutto, L., Picciulin, M., Costantini, M. & Ferrero, E. A. (2011). How boat noise affects an ecologically crucial behaviour: the case of territoriality in *Gobius cruentatus* (Gobiidae). *Environmental Biology of Fishes* 92, 207–215.

Seger-Fullam, K. D., Rodewald, A. D. & Soha, J. A. (2011). Urban noise predicts song frequency in Northern Cardinals and American Robins. *Bioacoustics* 20, 267–276.

*Sembulingam, K., Sembulingam, P. & Namasivayam, A. (1998). Effect of chronic noise stress on some selected stress indices in albino rats. *Journal of Environmental Biology* 19, 63–66.

*Shamoun-Baranes, J., Dokter, A. M., van Gasteren, H., van Loon, E. E., Leijnse, H. & Bouten, W. (2011). Birds flee en mass from New Year's Eve fireworks. *Behavioral Ecology* 22, 1173–1177.

Shannon, G., Angeloni, L. M., Wittemyer, G., Fristrup, K. M. & Crooks, K. R. (2014). Road traffic noise modifies behaviour of a keystone species. *Animal Behaviour* 94, 135–141.

Shieh, B. S., Liang, S. H., Chen, C. C., Loa, H. H. & Liao, C. Y. (2012). Acoustic adaptations to anthropogenic noise in the cicada Cryptotympana takasagona Kato (*Hemiptera: Cicadidae*). *Acta Ethologica* 15, 33–38.

*Shier, D. M., Lea, A. J. & Owen, M. A. (2012). Beyond masking: endangered Stephen's kangaroo rats respond to traffic noise with footdrumming. *Biological Conservation* 150, 53–58.

Siemers, B. M. & Schaub, A. (2011). Hunting at the highway: traffic noise reduces foraging efficiency in acoustic predators. *Proceedings of the Royal Society of London B: Biological Sciences* 278, 1646–1652.

*Sivle, L. D., Kvadsheim, P. H., Fahlman, A., Lam, F. P. A., Tyack, P. L. & Miller, P. J. (2012). Changes in dive behavior during naval sonar exposure in killer whales, long-finned pilot whales, and sperm whales. *Frontiers in Physiology* 3, 1–11.

Slabbekoorn, H. (2013). Songs of the city: noise-dependent spectral plasticity in the acoustic phenotype of urban birds. *Animal Behaviour* 85, 1089–1099.

*Slabbekoorn, H. & den Boer-Visser, A. (2006). Cities change the songs of birds. *Current Biology* 16, 2326–2331.

Slabbekoorn, H., Bouton, N., van Opzeeland, I., Coers, A., ten Cate, C. & Popper, A. N. (2010). A noisy spring: the impact of globally rising underwater sound levels on fish. *Trends in Ecology & Evolution* 25, 419–427.

Slabbekoorn, H. & Peet, M. (2003). Ecology: birds sing at a higher pitch in urban noise. *Nature* 424, 267.

Slabbekoorn, H. & Ripmeester, E. A. P. (2008). Birdsong and anthropogenic noise: implications and applications for conservation. *Molecular Ecology* 17, 72–83.

*Smith, M. E., Kane, A. S. & Popper, A. N. (2004). Noise-induced stress response and hearing loss in goldfish (*Carassius auratus*). *Journal of Experimental Biology* 207, 427–435.

Solé, M., Lenoir, M., Durfort, M., López-Bejar, M., Lombarte, A., Van der Schaar, M. & André, M. (2013). Does exposure to noise from human activities compromise sensory information from cephalopod statocysts? *Deep Sea Research Part II: Topical Studies in Oceanography* 95, 160–181.

de Soto, N. A., Delorme, N., Atkins, J., Howard, S., Williams, J. & Johnson, M. (2013). Anthropogenic noise causes body malformations and delays development in marine larvae. *Scientific Reports* 3, 2831.

de Soto, N. A., Johnson, M., Madsen, P. T., Tyack, P. L., Bocconcelli, A. & Fabrizio Borsani, J. (2006). Does intense ship noise disrupt foraging in deep-diving Cuvier's beaked whales (*Ziphius cavirostris*)? *Marine Mammal Science* 22, 690–699.

Southall, B. L., Berkson, J., Bowen, D., Brake, R., Eckman, J., Field, J., Gisiner, R., Gregerson, S., Lang, W. & Lewandowski, J. (2009). *Addressing the Effects of Human-Generated Sound on Marine Life: An Integrated Research Plan for US Federal Agencies*. Interagency Task Force on Anthropogenic Sound and the Marine Environment of the Joint Subcommittee on Ocean Science and Technology, Washington.

Southall, B. L., Bowles, A. E., Ellison, W. T., Finneran, J. J., Gentry, R. L., Greene, C. R. Jr., Kastak, D., Ketten, D. R., Miller, J. H. & Nachtigall, P. E.

2007). Marine mammal noise exposure criteria: initial scientific recommendations. *Aquatic Mammals* 33, 411–509.

Stansfeld, S. A. & Matheson, M. P. (2003). Noise pollution: non-auditory effects on health. *British Medical Bulletin* 68, 243–257.

Stokstad, E. (2014). US regulators unveil new ocean noise rules for marine mammals. *Science* 343, 128.

Summers, P. D., Cunnington, G. M. & Fahrig, L. (2011). Are the negative effects of roads on breeding birds caused by traffic noise? *Journal of Applied Ecology* 48, 1527–1534.

Sun, J. W. & Narins, P. M. (2005). Anthropogenic sounds differentially affect amphibian call rate. *Biological Conservation* 121, 419–427.

*Swaddle, J. P. & Page, L. C. (2007). High levels of environmental noise erode pair preferences in zebra finches: implications for noise pollution. *Animal Behaviour* 74, 363–368.

Szalma, J. L. & Hancock, P. A. (2011). Noise effects on human performance: a meta-analytic synthesis. *Psychological Bulletin* 137, 682–707.

Tabarelli, M. & Gascon, C. (2005). Lessons from fragmentation research: improving management and policy guidelines for biodiversity conservation. *Conservation Biology* 19, 734–739.

*Thompson, P. M., Lusseau, D., Barton, T., Simmons, D., Rusin, J. & Bailey, H. (2010). Assessing the responses of coastal cetaceans to the construction of offshore wind turbines. *Marine Pollution Bulletin* 60, 1200–1208.

*Todd, S., Lien, J., Marques, F., Stevick, P. & Ketten, D. (1996). Behavioural effects of exposure to underwater explosions in humpback whales (*Megaptera novaeangliae*). *Canadian Journal of Zoology* 74, 1661–1672.

Tripovich, J. S., Hall-Aspland, S., Charrier, I. & Arnould, J. P. (2012). The behavioural response of Australian fur seals to motor boat noise. *PLoS ONE* 7, e37228.

Tyack, P. L. (2008). Implications for marine mammals of large-scale changes in the marine acoustic environment. *Journal of Mammalogy* 89, 549–558.

Tyack, P. L., Zimmer, W. M. X., Moretti, D., Southall, B. L., Claridge, D. E., Durban, J. W., Clark, C. W., D'Amico, A., DiMarzio, N., Jarvis, S., McCarthy, E., Morrissey, R., Ward, J. & Boyd, I. L. (2011). Beaked whales respond to simulated and actual navy sonar. *PLoS ONE* 6, e17009.

*Uran, S. L., Aon-Bertolino, M. L., Caceres, L. G., Capani, F. & Guelman, L. R. (2012). Rat hippocampal alterations could underlie behavioral abnormalities induced by exposure to moderate noise levels. *Brain Research* 1471, 1–12.

Vargas-Salinas, F. & Amézquita, A. (2013). Traffic noise correlates with calling time but not spatial distribution in the threatened poison frog *Andinobates bombetes*. *Behaviour* 150, 569–584.

Vasconcelos, R. O., Amorim, M. C. P. & Ladich, F. (2007). Effects of ship noise on the detectability of communication signals in the Lusitanian toadfish. *Journal of Experimental Biology* 210, 2104–2112.

Verzijden, M. N., Ripmeester, E. A. P., Ohms, V. R., Snelderwaard, P. & Slabbekoorn, H. (2010). Immediate spectral flexibility in singing chiffchaffs during experimental exposure to highway noise. *The Journal of Experimental Biology* 213, 2575–2581.

*Wada, Y., Mogi, T., Inoue, H. & Koizumi, A. (1995). A mouse model of a sudden death induced by noise exposure is useful to investigate human responses to physical stress. *Industrial Health* 33, 29–33.

*Wahlberg, M. & Westerberg, H. (2005). Hearing in fish and their reactions to sounds from offshore wind farms. *Marine Ecology Progress Series* 288, 295–309.

Wale, M. A., Simpson, S. D. & Radford, A. N. (2013). Noise negatively affects foraging and antipredator behaviour in shore crabs. *Animal Behaviour* 86, 111–118.

Wardle, C. S., Carter, T. J., Urquhart, G. G., Johnstone, A. D. F., Ziolkowski, A. M., Hampson, G. & Mackie, D. (2001). Effects of seismic air guns on marine fish. *Continental Shelf Research* 21, 1005–1027.

*Weir, C. R. (2007). Observations of marine turtles in relation to seismic airgun sound off Angola. *Marine Turtle Newsletter* 116, 17–20.

*Weir, C. R. (2008). Overt responses of humpback whales (*Megaptera novaeangliae*), sperm whales (*Physeter macrocephalus*), and Atlantic spotted dolphins (*Stenella frontalis*) to seismic exploration off Angola. *Aquatic Mammals* 34, 71–83.

Weisenberger, M. E., Krausman, P. R., Wallace, M. C., De Young, D. W. & Maughan, O. E. (1996). Effects of simulated jet aircraft noise on heart rate and behavior of desert ungulates. *The Journal of Wildlife Management* 60, 52–61.

Westlund, K., Fernström, A. L., Wergård, E. M., Fredlund, H., Hau, J. & Spångberg, M. (2012). Physiological and behavioural stress responses in cynomolgus macaques (*Macaca fascicularis*) to noise associated with construction work. *Laboratory Animals* 46, 51–58.

Wilkens, S. L., Stanley, J. A. & Jeffs, A. G. (2012). Induction of settlement in mussel (*Perna canaliculus*) larvae by vessel noise. *Biofouling* 28, 65–72.

Wood, W. E. & Yezerinac, S. M. (2006). Song sparrow (*Melospiza melodia*) song varies with urban noise. *The Auk* 123, 650–659.

Wright, A. J., Aguilar Soto, N., Baldwin, A. L., Bateson, M., Beale, C. Clark, C. Deak, T., Edwards, E. F., Fernández, A., Godinho, A., Hatch, L., Kakuschke, A., Lusseau, D., Martineau, D., Romero, L. M., et al. (2007). Anthropogenic noise as a stressor in animals: a multidisciplinary perspective. *International Journal of Comparative Psychology* 20, 250–273.

Würsig, B. B., Greene, C. R. C. & Jefferson, T. A. T. (2000). Development of an air bubble curtain to reduce underwater noise of percussive piling. *Marine Environmental Research* 49, 79–93.

Wysocki, L. E., Dittami, J. P. & Ladich, F. (2006). Ship noise and cortisol secretion in European freshwater fishes. *Biological Conservation* 128, 501–508.

Zhang, M., Cheong, K., Leong, K. & Zou, F. (2012). Effect of traffic noise on black-faced spoonbills in the Taipa–Coloane Wetland Reserve, Macao. *Wildlife Research* 39, 603–610.

Zurcher, A. A., Sparks, D. W. & Bennett, V. J. (2010). Why the bat did not cross the road? *Acta Chiropterologica* 12, 337–340.

IX. SUPPORTING INFORMATION

Additional supporting information may be found in the online version of this article.

Appendix S1. Summary of information gathered from all studies used in this review.

Appendix S2. Summary of information gathered from studies used to generate Fig. 3.

(Received 1 October 2014; revised 28 May 2015; accepted 5 June 2015; published online 26 June 2015)

Swainson's Hawk
Survey Protocols, Impact Avoidance, and Minimization Measures for Renewable Energy Projects in the Antelope Valley of Los Angeles and Kern Counties, California

State of California
California Energy Commission and Department of Fish and Game
June 2, 2010

Swainson's Hawk Background Information

The Swainson's hawk (*Buteo swainsoni*) is listed as a California state threatened species under the California Endangered Species Act (CESA). The species is not listed as threatened or endangered under the federal Endangered Species Act. To comply with state wildlife protection requirements and receive project approvals, renewable energy project developers proposing projects in the Desert Renewable Energy Conservation Plan (DRECP) area may be required to conduct surveys and avoid or minimize impacts to Swainson's hawks and related nesting and foraging habitat. The survey protocols and mitigation and monitoring plan recommendations provided below suggest approaches and measures for complying with protection requirements.

Antelope Valley Swainson's hawks are known to have historically nested in Joshua tree woodlands and foraged in grasslands and native desert scrub communities. Currently, they nest in Joshua tree woodlands, ornamental roadside trees, and windrow or perimeter trees in active and historical agricultural areas. Foraging habitat includes dry land and irrigated pasture, alfalfa, fallow fields, low-growing row or field crops, new orchards, and cereal grain crops. Swainson's hawks may also forage in grasslands, Joshua tree woodlands, and other desert scrub habitats that support a suitable prey base. Gophers dominate the prey base of agriculturally based pairs while Swainson's hawks nesting in natural desert habitats consume a wider variety of prey species. While California's Central Valley Swainson's hawk population winters in Mexico, Central America South America, and a small percentage in the Central Valley, the migration habits of the Antelope Valley population are unknown. Recent observations suggest that they may arrive in nesting territories generally later than the Central Valley Population (Pete Bloom, raptor biologist, personal communication).

Environmental Review Considerations

The California Environmental Quality Act (CEQA), Warren-Alquist Act and implementing regulations, and CESA require consideration of direct, indirect, temporary, permanent, individual project, and cumulative impacts. CEQA allows approval of projects with significant effects when measures have been included to avoid or mitigate those effects, or specific considerations make such measures infeasible and specific benefits outweigh the significant effects. (CEQA Guidelines §21081). CESA regulates the taking of state-listed species. "Take" is defined as to "hunt, pursue, catch, capture, or kill, or to attempt to hunt, pursue, catch, capture, or kill." (Fish and Game Code §86). Incidental take authorization requires that all impacts to the species are minimized and fully mitigated and that mitigation is roughly proportional to the extent of the impacts of the taking. (14CCR § 783.4). This "full mitigation" standard is intended to ensure that the status of the species is the same or better after project and mitigation implementation as it was prior to project implementation.

Renewable energy project development could cause direct, indirect, individual, and cumulative adverse impacts to Swainson's hawks when facility construction and operation areas (such as wind turbines, power plants, solar panels and tower sites, access roads, staging areas, and pulling/splicing locations) occur in areas where hawks are present. Potential impacts include loss of foraging habitat and disruption of breeding activities due to increased dust, noise, and human presence. Direct mortality from vehicle strikes and collisions with wind turbines is also known to occur. Construction disturbance during the breeding season and habitat loss could cause incidental loss of fertile eggs or nestlings, or otherwise lead to nest abandonment.

The current land uses in the Antelope Valley area support approximately 10 breeding pairs. This area comprises the southernmost edge of the known breeding range for this species in California. The small number of breeding Swainson's hawks in the Antelope Valley and the potential isolation from other Swainson's hawk populations makes the Antelope Valley population particularly susceptible to extirpation. Swainson's hawks have high nest site fidelity, meaning they return to the same site year after year (Estep 1989, Woodbridge et al. 1995) This may limit exchange of individual birds between distant breeding groups (Hull et al. 2007). Hull et al. (2007) found evidence suggesting that the Central Valley population has had little recent genetic exchange with other populations east of the Sierra Nevada. Due to the geographical isolation of the Antelope Valley Swainson's hawk population from other breeding populations, together with the species' high site fidelity, it is reasonable to infer that rapid re-colonization of the Antelope Valley would be unlikely if nesting pairs were lost. Given these facts, the California Department of Fish and Game (Department) would consider impacts to breeding pairs to be potentially significant because they may cause the population to become less than self-sustaining.

A substantial reduction in numbers or habitat of a rare, threatened, or endangered species would be considered a significant impact under CEQA. Potentially significant impacts may result from activities that cause nest abandonment, loss of nest trees, loss of foraging habitat that would reduce nesting success (loss or reduced health or vigor of eggs or young), or direct mortality. Due to the Swainson's hawk's known preference for areas of low vegetation that support abundant prey, such as grasslands or alfalfa fields (Bechard 1982, Babcock 1995), the Department considers conversion of foraging areas to renewable energy power plant facility sites to be habitat loss. For example, solar panel arrays are expected to eliminate most or all foraging potential. Significant habitat loss may result from individual projects and cumulatively, from multiple projects. Each project which contributes to a significant cumulative effect must offset its contribution to that effect in order to determine that the cumulative impacts have been avoided.

The Department considers a nest site to be active if it was used at least once during the past 5 years. Impacts to suitable habitat or individual birds within a five-mile radius of an active nest will be considered significant and to have the potential to "take" Swainson's hawks as that term is defined in §86 of the Fish and Game Code. Please consult with the Department when determining whether "take" authorization is warranted for a specific project.

Special Considerations for Wind Energy Development

Wind turbines present an additional, continuous, long-term risk of Swainson's hawk take throughout the life of a project. This continuous risk is not always considered in the environmental analyses for other types of projects that may have limited short-term impacts (e.g. construction related impacts). It has been documented elsewhere in California that Swainson's hawks are killed by wind turbines. Turbine strikes could occur during migration or during the nesting season. Swainson's hawk surveys for wind energy development should follow the same methods as for solar energy projects, described below, but the impacts analysis and corresponding mitigation should consider the additional continuous long-term risk of turbine-related fatalities. Habitat impact analysis should consider both the ground surface area and the air space that is used by Swainson's hawks. The mitigation methods described below are specific to ground surface impacts. Wind energy development project proponents should consult with the Department to develop avoidance measures and mitigation specific for the loss of air space and the potential for on-going take of Swainson's hawk during project operations." For additional avian considerations that are applicable to Swainson's hawk, please refer to the "California Guidelines for Reducing Impacts to Birds and Bats from Wind Energy Development" (California Energy Commission and California Department of Fish and Game 2007). The guidelines can be found at http://www.energy.ca.gov/windguidelines/index.html.

Survey Protocol

The following survey protocols and monitoring/mitigation recommendations suggest surveys and acquisition of mitigation lands prior to construction of the project if nests are found within five miles of a project site. Before conducting surveys for a particular project, project developers are encouraged to contact the Department and the appropriate lead agencies for up-to-date, site-specific issues and possible refinement of the following survey protocols and monitoring/mitigation recommendations. Survey methods may be flexible depending on surveyor experience and/or already-known nesting status for a given site. Please contact the Department (Region 4 for Kern County and Region 5 for Los Angeles County) to use an alternate survey plan from that suggested within this document.

A qualified raptor biologist with Swainson's hawk survey experience, approved by the Department and the appropriate lead agency, should conduct surveys in a manner that maximizes the potential to observe the adult Swainson's hawks and the nest/chicks via visual and audible cues within a five-mile radius of the project. All potential nest trees within the five-mile radius shall be surveyed for presence of nests. Surveys should be conducted prior to environmental analysis. Surveys should be repeated within the 5-mile radius if a survey season ensues or elapses before the onset of project related activities. If construction begins mid-survey season the year after the initial surveys, then the surveys should continue for that part of the season before construction.

Examples of suitable habitats are Joshua tree woodlands, grasslands, desert scrub communities, and agricultural lands (such as alfalfa, fallow fields, beet, tomato, onions, and other low-growing row or field crops, dry-land and irrigated pasture, cereal grain crops [including corn after harvest], and new orchards). Consult with the Department when determining whether the project site is within five miles of already-known nest sites. If hawks or known nest sites are found within the five-mile radius, consult with the Department and the appropriate lead agency for follow-up to the surveys.

Minimum Equipment
Minimum survey equipment includes a high-quality pair of binoculars and a high quality spotting scope. Surveying even the smallest project area will take hours, and poor optics often result in eye-strain and difficulty distinguishing details in vegetation and subject birds. Other equipment includes good maps, GPS units, flagging, and notebooks.

Walking vs Driving
Driving or "windshield surveys" are usually preferred to walking if an adequate roadway is available through or around the project site. While driving, the observer can typically make a closer approach to a hawk without causing the bird to fly. Although it might appear that a flying bird is more visible, they often fly away from the observer using trees as screens; and it is difficult to determine from where a flying bird originated. Walking surveys are useful in locating a nest after a nest territory is identified, or when driving is not an option.

Angle and Distance to the Tree
Surveying subject trees from multiple angles will greatly increase the observer's chance of detecting a nest or hawk, especially after trees are fully leafed and when surveying multiple trees in close proximity. When surveying from an access road, survey in both directions. Maintaining a distance of 50 meters to 200 meters from subject trees is optimal for observing perched and flying hawks without greatly reducing the chance of detecting a nest/young. Once a nesting territory is identified, a closer inspection may be required to locate the nest.

Speed
Travel at a speed that allows for a thorough inspection of a potential nest site. Survey speeds should not exceed 5 miles per hour to the greatest extent possible. Stop frequently to scan subject trees with binoculars and a spotting scope.

Visual and Audible Cues
Focus surveys on both observations and vocalizations. Observations of nests, perched adults, displaying adults, and chicks during the nesting season are all indicators of nesting Swainson's hawks. In addition, vocalizations are extremely helpful in locating nesting territories. Vocal communication between hawks is frequent (1) during territorial displays, (2) during courtship and mating, (3) through the nesting period as mates notify each other that food is available or that a threat exists, (4) and as older chicks and fledglings beg for food.

Distractions
Minimize distractions while surveying. Although two pairs of eyes may be better than one pair at times, conversation may limit focus. Radios should be off, not only are they distracting, they may cover a hawk's call.

Notes and Species Observed
Take thorough field notes. Detailed notes and maps of the location of observed Swainson's hawk nests are essential for filling gaps in the California Natural Diversity Data Base; please note all observed nest sites, including date and time of observation, location name, UTM coordinates, number of young, and any behavioral observations. Also document the occurrence of nesting great horned owls, red-tailed hawks, red-shouldered hawks and other potentially competitive species. These species will infrequently nest within 100 yards of each other, so the presence of one species will not necessarily exclude another.

Timing
To meet *the minimum level* of protection for the species, surveys should be completed for *at least* the two survey periods immediately prior to a project's initiation. For example, if a project is scheduled to begin on June 1, you should complete three surveys in Period II and three surveys in Period III. However, it is always recommended that surveys be completed in Periods II, III, and IV prior to environmental review.

Survey Period I
Survey dates: January-March 31 (optional but recommended; pre-arrival)
Survey Time: All day
Number of Surveys: 1
Justification and search image: Prior to Swainson's hawks arrival from wintering grounds, it is very helpful to survey the project area to determine potential nest locations. Most nests are easily observed from relatively long distances, giving the surveyor the opportunity to identify potential nest sites, as well as becoming familiar with the project area. It also gives the surveyor the opportunity to locate and map competing species nest sites such as great horned owls from February on, and red-tailed hawks

from March on. After March 1, surveyors may observe Swainson's hawks staging in traditional nest territories.

Survey Period II
Survey dates: April 1 – April 30 (arrival; nest building)
Survey Time: All day
Number of Surveys: 3
Justification and search image: Most Antelope Valley Swainson's hawks return by April 1, and immediately begin occupying their traditional nest territories. For those few that do not return by April 1, there are often hawks ("floaters") that act as place-holders in traditional nest sites; they are birds that do not have mates, but temporarily attach themselves to traditional territories and/or one of the site's "owners." Floaters are usually displaced by the territories' owner(s) if the owner returns. Most trees are leafless and are relatively transparent; it is easy to observe old nests, staging birds, and competing species. The hawks are usually in their territories during the survey hours, but typically soaring and foraging in the mid-day hours. Swainson's hawks may often be observed involved in territorial and courtship displays, and circling the nest territory. Potential nest sites identified by the observation of staging Swainson's hawks will usually be active territories during that season, although the pair may not successfully nest/reproduce that year. Both males and females are actively nest building, visiting their selected site frequently. Later in this survey period, territorial and courtship displays are increased, as is copulation. The birds tend to vocalize often, and nest locations are most easily identified. This period may require a great deal of "sit and watch" surveying.

Survey Period III
Survey dates: May 1 – May 30 (egg laying; incubation)
Survey Time: daylight hours, as needed to monitor known nest sites only
Number of Surveys: 3
Justification and search image: Nests are extremely difficult to locate this time of year, and even the most experienced surveyor may miss them, especially if the previous surveys have not been done. During this phase of nesting, the female Swainson's hawk is in brood position, very low in the nest, laying eggs, incubating, or protecting the newly hatched and vulnerable chicks; her head may or may not be visible. Nests are often well-hidden, built into heavily vegetated sections of trees or in clumps of mistletoe, making them all but invisible. Trees are usually not viewable from all angles, which may make nest observation impossible. Following the male to the nest may be the only method to locate it, and the male will spend hours away from the nest foraging, soaring, and will generally avoid drawing attention to the nest site. Even if the observer is fortunate enough to see a male returning with food for the female, if the female determines it is not safe she will not call the male in, and he will not approach the nest; this may happen if the observer, or others, are too close to the nest or if other threats, such as rival hawks, are apparent to the female or male.

Survey Period IV
Survey dates: June 1 – July 15 (fledging)
Survey Time: Sunrise to 1200, 1600 to sunset
Number of Surveys: 3
Justification and search image: Young are active and visible, and relatively safe without parental protection. Both adults make numerous trips to the nest and are often soaring above, or perched near or on the nest tree. The location and construction of the nest may still limit visibility of the nest, young, and adults.

Reporting
Provide the Department and the appropriate lead agency with pre-construction survey results in a written report, within 30 days prior to commencement of construction activities. Report should include date of the report, authors and affiliations, contact information, introduction, methods, study location (include map), results, discussion, and literature cited. For surveys intended to support environmental impact analyses prior to project approval, provide the Department and the lead agency with written survey reports within 30 days of survey completion. Submit California Natural Diversity Database (CNDDB) forms for any listed, fully protected, or species of special concerncountered and positively identified. CNDDB forms may be found at the following link: http://www.dfg.ca.gov/biogeodata/cnddb/pdfs/CNDDB_FieldSurveyForm.pdf.

Monitoring and Mitigation Plan Recommendations

1. If surveys locate a nest site, prepare a Swainson's hawk Monitoring and Mitigation Plan in consultation with the Department and the appropriate lead agency. Plans should be prepared by a qualified biologist approved by the Department and the appropriate lead agency. Include in the plans detailed measures to avoid and minimize impacts to Swainson's hawks in and near the construction areas. For example:

    a. If a nest site is found, design the project to allow sufficient foraging and fledging area to maintain the nest site.

    b. During the nesting season, ensure no new disturbances, habitat conversions, or other project-related activities that may cause nest abandonment or forced fledging occur within 1/2 mile of an active nest between March 1 and September 15. Buffer zones may be adjusted in consultation with the Department and the lead agency.

    c. Do not remove Swainson's hawk nest trees unless avoidance measures are determined to be infeasible. Removal of such trees should occur only during the timeframe of October 1 and the last day in February.

2. Monitoring plans should include measures for injured Swainson's hawks:

   a. For hawks found injured during project-related activities on the project site, plans should call for immediate relocation to a raptor recovery center approved by a Department regional representative.
   b. A system should be set-up so that costs associated with the care or treatment of such injured Swainson's hawks will be borne by the project developer.

   c. Include appropriate contact information for immediate notification of the Department and the appropriate lead agency of a hawk injury incident. Have approved procedures in place to notify the Department and the lead agency outside normal business hours. Notify the appropriate personnel via telephone or email, followed by a written incident report. Include the date, time, location, and circumstances of the incident in the reports.

3. Mitigation plans should focus on providing habitat management (HM) lands. Lands which are currently in urban use or lands that have no existing or potential value for foraging Swainson's hawks will not require mitigation nor would they be suitable for mitigation. The plans should call for mitigating loss of Swainson's hawk foraging habitat by providing HM lands within the Antelope Valley Swainson's hawk breeding range at a minimum 2:1 ratio for such habitat impacted within a five-mile radius of active Swainson's hawk nest(s). The Department considers a nest active if it was used one or more times within the last 5 years.

   Project developers may consider delegating responsibilities for acquisition and management of the HM lands to the Department or a third party, such as a non-governmental organization dedicated to Mojave Desert habitat conservation. Seek approval of such delegations from the Department and the appropriate lead agency.

   Approaches for acquisition and management of HM lands:

   a. <u>HM Land Selection Criteria.</u> Identify the region within which lands would be acquired, and the type/quality of habitat to be acquired. Foraging habitat should be moderate to good with a capacity to improve in quality and value to Swainson's hawks, and must be within the Antelope Valley Swainson's hawk breeding range. Foraging habitat with suitable nest trees is preferred.

   b. <u>Review and Approval of HM Lands Prior to Acquisition</u>. Provide an acquisition proposal to the Department and the appropriate lead agency for their approval at least 3 months before acquiring the property. The proposal should discuss the suitability of the property by comparing it to the selection criteria.

   c. <u>Land Acquisition Schedule and Financial Assurances.</u> Complete acquisition of proposed HM lands before initiating ground-disturbing project activities. If an irrevocable letter of credit or other form of security is provided, complete land acquisition within 12 months prior to beginning ground-disturbing project activities. Provide financial assurances for dedicating adequate funding for impact avoidance, minimization and compensation measures required for project approval (see 3. d. below).

d. HM Lands Acquisition. Be prepared to provide a preliminary title report, initial hazardous materials survey report, biological analysis, at a minimum to the Department and the appropriate lead agency. The information will likely also be reviewed by the California Department of General Services, Fish and Game Commission and/or Wildlife Conservation Board.

   Fee title or conservation easement will likely be transferred to a Department of Fish and Game-approved non-profit third party and the Department, or solely to the Department. Be prepared to support enhancement and endowment funds for protection and enhancement of acquired lands. The Department will approve establishment and management of the funds, ensuring that qualified non-profit organizations or the Department will manage the funds in an appropriate manner. Contributed funds and any related interest generated from the initial capital endowment would support long-term operetion, management, and protection of the approved HM lands, including reasonable administrative overhead, biological monitoring, improvements to carrying capacity, law enforcement measures, and any other action designed to protect or improve the habitat values of the HM lands. Be prepared to reimburse the Department or other entities for all land acquisition costs.

References

Babcock, K.W. 1995. Home range and habitat use of breeding Swainson's hawks in the Central Valley of California. Journal of Raptor Research 29:193-197.

Bechard, Marc J. Effect of vegetative cover on foraging site selection by Swainson's Hawk. The Condor 84: 153-159.

California Energy Commission and California Department of Fish and Game 2007. California guidelines for reducing impacts to birds and bats from wind energy development. Commission final report. California Energy Commission, Renewables Committee, and Energy Facilities Siting Division and California Department of Fish and Game, Resources Management and Policy Division. CEC-700-2007-008-CMF.

Estep, J.A. 1989. Biology, movements, and habitat relationships of the Swainson's Hawk in the Central Valley of California, 1986–87. California Department of Fish and Game, Nongame Bird and Mammal Section Report.

Woodbridge B, Finley KK, Bloom PH. 1995. Reproductive performance, age structure, and natal dispersal of Swainson's Hawks in the Butte Valley, California. Journal of Raptor Research 29:187–192.

CALIFORNIA AMPHIBIAN *and* REPTILE SPECIES *of* SPECIAL CONCERN

Robert C. Thomson,
Amber N. Wright, and H. Bradley Shaffer

Betsy Bolster, Project Manager
Kristi Cripe, Cartographer and GIS Specialist
Sean J. Barry, Robert N. Fisher, Hartwell H. Welsh Jr., Technical Advisory Committee

CALIFORNIA DEPARTMENT OF FISH AND WILDLIFE

UNIVERSITY OF CALIFORNIA PRESS

CALIFORNIA AMPHIBIAN *and* REPTILE SPECIES *of* SPECIAL CONCERN

This page intentionally left blank

CALIFORNIA AMPHIBIAN *and* REPTILE SPECIES *of* SPECIAL CONCERN

Robert C. Thomson,
Amber N. Wright, and H. Bradley Shaffer

Betsy Bolster, Project Manager
Kristi Cripe, Cartographer and GIS Specialist
Sean J. Barry, Robert N. Fisher, Hartwell H. Welsh Jr., Technical Advisory Committee

CALIFORNIA DEPARTMENT OF FISH AND WILDLIFE

UNIVERSITY OF CALIFORNIA PRESS

University of California Press, one of the most distinguished university presses in the United States, enriches lives around the world by advancing scholarship in the humanities, social sciences, and natural sciences. Its activities are supported by the UC Press Foundation and by philanthropic contributions from individuals and institutions. For more information, visit www.ucpress.edu.

University of California Press
Oakland, California

Copublished with the California Department of Fish and Wildlife

Library of Congress Cataloging-in-Publication Data

Names: Thomson, Robert C., 1982– author. | Wright, Amber N., 1978– author. | Shaffer, H. Bradley (Howard Bradley), 1953– author.
Title: California amphibian and reptile species of special concern / Robert C. Thomson, Amber N. Wright, H. Bradley Shaffer.
Description: Oakland, California : University of California Press, [2016] | ©2016 | Includes bibliographical references and index.
Identifiers: LCCN 2015048152 (print) | LCCN 2015050096 (ebook) | ISBN 9780520290907 (pbk. : alk. paper) | ISBN 9780520964839 (epub)
Subjects: LCSH: Rare amphibians—California. | Rare reptiles—California.
Classification: LCC QL644.73.U5 T46 2016 (print) | LCC QL644.73.U5 (ebook)

Manufactured in China
25 24 23 22 21 20 19 18 17 16
10 9 8 7 6 5 4 3 2 1

The paper used in this publication meets the minimum requirements of ANSI/NISO Z39.48-1992 (R 2002) (*Permanence of Paper*).

CONTENTS

This page intentionally left blank

FOREWORD

California boasts one of the most biologically diverse faunas in the United States, as well as one of the most threatened. One of the key elements of the state's efforts to protect its vertebrate fauna is through its Species of Special Concern program. The current volume, *California Amphibian and Reptile Species of Special Concern*, is an essential foundation upon which both biologists and state and federal agencies can manage the biological resources of the state. California has exceedingly sensitive species and ecosystems, many of which are at risk of extirpation or extinction as the state's environment changes at rates greater than at any time in history.

This book builds upon the shoulders of its predecessor from two decades ago (Jennings and Hayes 1994a), but it is not just a simple update. Jennings and Hayes surveyed an enormous number of experts to create a comprehensive publication on California's special concern amphibians and reptiles, and their volume was a key management tool for a generation of biologists. However, this new book goes several steps further, making it a necessary reference for wildlife and land managers, biologists, and nature lovers interested in amphibians and reptiles.

First, the maps generated for this book are stunning. They are literally beautiful enough to be framed, and detailed enough to guide resource managers. Second, there are color images of every taxon, generally taken in the field and highlighting the key features of each species. Third, the authors rely on the published literature to the maximum extent possible, pulling in the gray literature only when it is needed (which is often because many of these species are poorly known). But perhaps most importantly, the authors used multifactorial risk metrics that bring several measures of potential and actual threat into a single numeric score that captures the sensitivity of the species. The result is a tool that provides an important first pass at the difficult task of identifying those taxa that should be candidate Special Concern species.

Of course, there will always be important biological considerations that may argue against a strict interpretation of the metric scores, as the authors fully realize. For example, there are species on the Special Concern list that are so narrowly precinctive that the narrowness of their geographic range alone signals reason to be extra cautious about the species. The sandstone night lizard is one such taxon; its geographic range is much smaller than listed species such as black toad (*Bufo exsul*), and we know much less about the night lizard than we do about black toads. Regardless of the risk model score, this is a scary situation, and the narrowness of geographic range alone signals reason to be extremely cautious. Herpetologists are well aware of extinctions of entire species that were so narrowly precinctive that very subtle (sometimes unknown) environmental changes have caused those extinctions (e.g., the golden toad of Costa Rica, which had a geographic range the same size as that of the sandstone night lizard).

There are other species covered in this volume that will be challenging to manage for their protection in California. For example, the Gila monster (*Heloderma suspectum*) can be found in the extreme eastern part of the Mojave Desert in California (east of 116° longitude), where it has been recorded fewer than 30 times in the last 150 years. Within the distribution of Gila monsters in California, the pattern of rainfall includes winter rains and summer (monsoonal) rains; this biphasic pattern is typical in Utah, Nevada, and Arizona where Gila monsters are relatively more common. Throughout their geographic range, Gila monsters depend upon climate conditions conducive for reproduction by small mammals because neonatal small mammals are the principal prey for this species. However, climate is demonstrably changing in California to be warmer (especially in summers) and with increased frequencies of drought. These changes may not be mitigable at a local level, and this creates conservation challenges. Nevertheless, knowledge of both changes of climate and the biology of Gila monsters is meager, and this signals both that the Gila monster is clearly a reasonable candidate for SSC status and a need for additional research.

In keeping with this example, this volume calls for significantly increasing research and monitoring of these species. This is a recommendation that must be taken very seriously. Change to California wildlife is accelerating at a more rapid rate than ever before in history, and the best chance to protect California's Species of Special Concern from extirpation or extinction is increasing our knowledge of these poorly studied animals. Long-term monitoring of the status of populations is key, and contemporary methods such as population genomics can provide insights into population status and viability that were not possible just a few years ago.

As complete as it is, this volume should be considered a beginning, rather than a final set of definitive answers, for understanding ecologically sensitive amphibians and reptiles in California. It constitutes an enormously valuable benchmark, and also provides solid information about the biology and ecology of amphibian and reptile species in California. Now we need to pursue its recommendations so that we can facilitate the needed science that will help us protect California's biological resources. California needs to expand science and management of the state's precious biological resources so that our children and grandchildren, hopefully, will be able to experience no fewer species than are present in California today. This book is an important step in that direction.

C. RICHARD TRACY
Professor, Department of Biology
University of Nevada, Reno
Reno, NV 89557

PREFACE

California's amphibians and reptiles are unique in the United States for the tremendous amount of evolutionary and ecological diversity that they represent. California is second only to Texas in terms of the number of native amphibians and reptiles found within a state and contains endemic species of all major groups except turtles and tortoises. The state is home to what might be the best-known example of ring speciation (in *Ensatina* salamanders), which provides a unique view into the process of species diversification. California is home to the tailed frog (*Ascaphus truei*), a species that is among the last surviving members of an ancient lineage that is the sister group to all other frogs on earth. It houses reptile and amphibian species with genetic- and temperature-dependent sex determination; species that lay eggs in the water, on land, or that are live-bearing; and species with a two-staged life cycle that undergo a profound metamorphosis, switching between distinctly different habitats in the process.

The California Department of Fish and Wildlife (formerly, California Department of Fish and Game) is the trustee agency for California's fish and wildlife resources. The challenges associated with effective management and conservation of these resources are formidable in California, where a large human population, diverse stakeholder interests, and extremely high biotic diversity must be jointly managed. Despite the challenge of implementing effective conservation in the state, doing so is an important and worthy goal given the vast diversity that the state supports. We have attempted to evaluate conservation status for the state's amphibians and reptiles openly and transparently, relying on both the best available science and the breadth of expert opinions relating to amphibian and reptile conservation in California. We have sought (and received) broad feedback from a wide range of interested parties including agency representatives, academic scientists, and avocational herpetologists and used this combined input to make informed recommendations about conservation risk and management needs for California's amphibians and reptiles. We have also highlighted where data are lacking and discussed how the community might fill these gaps in our knowledge. Our goal is for this volume to serve as both a summary of where we stand and a launching point for what we can achieve in the management and restoration of healthy amphibian and reptile populations in California.

ROBERT C. THOMSON
Honolulu, Hawaii

AMBER N. WRIGHT
Honolulu, Hawaii

H. BRADLEY SHAFFER
Los Angeles, California
May 2015

This page intentionally left blank

ACKNOWLEDGMENTS

This project was a collaborative effort between the California Department of Fish and Wildlife, the Amphibian and Reptile Species of Special Concern Technical Advisory Committee, and contributors from government agencies, universities, nongovernmental organizations, and private citizens drawn from across California. Without the guidance, assistance, and advice from many individuals, this project would not have been possible. In particular, we thank Betsy Bolster for her help and support through all phases of this project. Betsy's deep experience has been indispensable in guiding and focusing the scope and aims for our efforts, and her advice has been an indispensable part of the success of the project. We also thank Kristi Cripe for her efforts and expertise in developing the maps for this document and her general advice on cartography. Sean Barry, Robert Fisher, and Hart Welsh, together comprising our Technical Advisory Committee, constituted a key resource at several phases of this project. Their advice directly shaped the development of the scoring metrics, nominee list, and the composition of the final Species of Special Concern list. They also provided extensive unpublished data and knowledge about California's amphibians and reptiles, answered countless questions, and served as a sounding board throughout the project.

Many people contributed at various points in the project by providing feedback on the metric scoring system, suggesting additional taxa for review, contributing their unpublished data, answering questions, and providing advice: Allison Alberts, Jack Allen, Bertin Anderson, Don Ashton, Curt Babcock, Michael Bank, Anthony Barley, Kent Beaman, Beth Behm, Mike Benard, Stephen Bennett, Carol Berry, Jamie Betasso, Ryan Bourque, David Bradford, Cathy Brown, Jeff Brown, Tracey Brown, Gary Bucciarelli, Allen Calvert, Becky Chong, David Cook, Joe Croteau, Robert Douglas, Erik Dugan, Jeremiah Easter, Tag Engstrom, Bobby Espinoza, Edward Ervin, Robert Fisher, Marina Gerson, Levi Gray, Luke Groff, Ken Halama, Bob Hansen, Marc Hayes, Brian Hinds, Garth Hodgson, Brian Hubbs, Elizabeth Jockusch, Jarrett Johnson, Christina Jones, Mark Jorgensen, Roland Knapp, Linda Kuhnz, Adam Leaché, Dean Leavitt, Jeff Lemm, Karen Leyse, Amy Lind, Jeff Lovich, Clark Marhdt, Mike Oldham, Ted Papenfuss, Jim Parham, Greg Pauly, Ryan Peek, Michael Peters, Jonah Piovia-Scott, Karen Pope, Sean Reilly, Jonathan Richmond, Jim Rorabaugh, Sean Rovito, Wesley Savage, Kirk Setser, Jackson Shedd, Christina Sousa, Lara Sparks, Phillip Spinks, Mike Sredl, Rick Staub, Sam Sweet, John Swett, Dick Tracy, Michael van Hattem, Kent

Vansooy, David Wake, Ian Wang, Nick Waters, Clara Wheeler, Dustin Wood, and Brenda Zaun.

Several museum collections and other sources of locality data allowed the use of their data in constructing the species range maps. These data greatly improved the quality of the maps, and we appreciate the generosity of these collections: American Museum of Natural History, Arizona State University, Brigham Young University, California Academy of Sciences, California Academy of Sciences (Stanford University Collection), California State University at Chico, Carnegie Museum of Natural History, Cincinnati Museum Center, Cornell University Museum of Vertebrates, Humboldt State University, Los Angeles County Museum, Museum of Comparative Zoology at Harvard University, Museum of Vertebrate Zoology at University of California, Berkeley, National Museum of Natural History, Royal Ontario Museum, San Diego Natural History Museum, Santa Barbara Museum of Natural History, Slater Museum of Natural History, Sternberg Museum of Natural History, University of Alberta Museum of Zoology, University of Arizona Mnseum of Natural History, University of California, Davis – Zoology Collection, University of California, Santa Barbara, University of Colorado Museum of Natural History, University of Michigan Museum of Zoology, University of Nevada, Reno, University of Texas at El Paso, Yale Peabody Museum, and the Zoological Institute of the Russian Academy of Sciences. In addition to museum collections, we also appreciate the locality information that we obtained from the following sources: CalPhotos (UC Berkeley), California Biogeographic Information and Observation System, California Natural Diversity Database, Mendocino Redwood Company, North American Field Herping Association, United States Forest Service, and the United States Geological Survey. Finally, we would like to thank several individuals that contributed the photographs for use in this book: John Andermann, Beth Behm, Adam Clause, William Flaxington, Brian Freiermuth, Rob Grasso, Robert Hansen, Nicholas Hess, Robert Hess, Troy Hibbitts, Jeff Lemm, Rob Lovich, Luke Mahler, Rob Schell, Barrett Scurlock, and Jackson Shedd.

The review process for this document was extensive, and we appreciate the effort that so many people made toward reading, editing, and improving the document. In particular, Betsy Bolster, Lyanne Comrack, Scott Osborn, Laura Patterson, and Dale Steele provided an extensive review of the document and provided guidance on policy-related matters. We also thank the large group of dedicated individuals who provided peer reviews of one or more species accounts or the document as a whole: Bernard Aguilar, Adam Backlin, Amy Bailey, Cam Barrows, Sean Barry, Kent Beaman, Mike Benard, Bob Bezy, Jeff Bouudy, David Bradford, Jennifer Bull, Bruce Bury, Becky Chong, Jack Crayon, Catherine Darst, Carlos Davidson, Lowell Diller, Paul Divine, Dawne Emery, Tag Engstrom, James Erdman, Ed Ervin, Bobby Espinoza, Gary Fellers, Mark Fisher, Robert Fisher, Terra Fuller, Justin Garwood, Dave Germano, Levi Gray, Luke Groff, Bob Hansen, Marc Hayes, Brian Hinds, Garth Hodgson, Jef Jaeger, Elizabeth Jockusch, Sharon Keeney, Roland Knapp, Shawn Kuchta, Sarah Kupferberg, Kris Lappin, Dean Leavitt, Christina Liang, Amy Lind, Mitch Lockhart, Jeff Lovich, Rob Lovich, Winsor Lowe, Clark Mahrdt, Steve Morey, Peter Moyle, Bob Murphy, Al Muth, Ted Papenfuss, Jim Parham, Greg Pauly, Chris Pearl, Jonah Piovia-Scott, Karen Pope, Sean Reilly, Carlton Rochester, Wesley Savage, Arthur Shapiro, Jackson Shedd, Phillip Spinks, Glenn Stewart, Paul Stone, Sam Sweet, Michael van Hattem, David Wake, Ian Wang, Nick Waters, Jim Weigand, and Hart Welsh.

Many current and former members of the Department of Biology at the University of Hawai'i at Mānoa and the Department of Evolution and Ecology at the University of California at Davis also gave a substantial amount of time to this project. In particular, we would like to thank Anthony Barley, Gideon Bradburd, Adam Clause, Levi Gray, Ben Johnson, Jarrett Johnson, Max Lambert, Genevieve Mount, Cathy Newman, Greg Pauly, Maureen Ryan, Robyn Screen, Chris Searcy, Phillip Spinks, and Ian

Wang for reading and providing feedback on several different parts of the project, as well as serving as a source of ideas and feedback. Adam Clause also generously donated the use of several photos for the project's website.

Primary financial support for the Amphibian and Reptile Species of Special Concern project was provided by the California Department of Fish and Wildlife's Nongame Wildlife Program via a State 2006-07 One-Time General Fund Augmentation for Nongame Fish and Wildlife Trust Resources and State Wildlife Grant T-28-R-1 from the United States Fish and Wildlife Service.

This page intentionally left blank

OVERVIEW

This page intentionally left blank

ABSTRACT

We provide a synthesis of the conservation risk faced by amphibians and reptiles in California that qualify as Species of Special Concern. After assembling a full list of the native amphibian and reptile taxa that are known to occur in the state, we developed a potential set of 73 nominee taxa that might qualify as Species of Special Concern. We developed eight metrics that capture key elements of declining and at-risk species, scored them for all 73 nominee taxa based on an extensive literature review, examined them on a case-by-case basis, and developed a final set of 45 Species of Special Concern. We then developed species accounts for each Species of Special Concern, documenting available information on their basic biology, known or hypothesized reasons for decline, and proposed management and future research needs. Overall, we sought to produce a clear, transparent document that explicitly states why decisions were made and supported with a summary of the best available science. We relied on peer-reviewed literature whenever possible to support those decisions.

Our evaluation resulted in 16 Species of Special Concern categorized as Priority 1 (those of greatest concern), 14 as Priority 2, 12 as Priority 3, and 3 which we could not prioritize based on available data. Our comparative analyses demonstrated that there were certain sets of organisms, geographic areas, and groups of ecological specialists in which species of greatest concern tended to be concentrated. Taxonomically, frogs, salamanders, and turtles all had higher average metric scores than lizards or snakes, mirroring the fraction of those taxa listed at the state and federal levels, and suggesting that these lineages are often of greatest conservation concern. There was also a strong trend for aquatic taxa to experience a greater conservation risk than terrestrial species. Geographically, southern California harbored more Species of Special Concern than central or northern California. This pattern was driven primarily by reptiles, which have a preponderance of at-risk species in the Southern California Coast, Southern California Mountains and Valleys, and the Mojave Desert ecoregions. Amphibian Species of Special Concern tended

to be more evenly distributed across northern and southern California ecoregions.

In a troublingly large number of cases, we found a striking lack of critical data for many aspects of the basic biology of amphibian and reptile species, and this lack of field ecology, natural history, and genetic data hindered our ability to make strong management recommendations. The solution to this lack of data is clear: California needs to launch a program that funds strong, peer-review quality analyses of basic ecology, combined with long-term monitoring studies to evaluate demographic trends at a set of sites for each species. Such studies need not be expensive and would make an enormous difference in our ability to manage many Species of Special Concern, hopefully precluding the need for future state and/or federal listing. Meaningful collaboration between the California Department of Fish and Wildlife and other research groups (be they other agencies, universities, nongovernmental organizations, or avocational groups) has helped to fill some of these gaps, particularly for federally listed species, and such collaborations for Species of Special Concern are the key to developing management plans into the future. We also found that in many cases population genetic approaches can help to fill critical gaps in our knowledge regarding species and subspecies boundaries, effective population sizes, corridors of likely habitat use, migration frequencies and pathways, and levels of hybridization with native and introduced species. These genetic measures should complement, rather than replace field studies, and they offer the opportunity to conduct relatively fast analyses that can and should provide critical early guidance for management decisions.

As critical basic biodiversity work in California continues, we are increasingly recognizing that the complex geology and changing environmental conditions in the state have led to the evolution of an amazing array of endemic taxa, many of which are extreme habitat specialists. To our knowledge, none of these sensitive species have been lost to extinction yet, although several are dangerously close. However, at least four taxa whose range limits historically entered the margins of the state may already be gone from California's boundaries, and some of the endemic species may be next. The identification of Species of Special Concern and the compilation of information, research needs, and management recommendations represents an important step to help California land managers prevent further declines, stabilize key populations, and potentially initiate recovery programs before formal listing is necessary.

INTRODUCTION

From a biodiversity perspective, California resides at one of the most important crossroads in the United States. The California Floristic Province is the only globally recognized biodiversity hot spot in North America north of Mexico, and one of three recognized in the north-temperate region (Myers et al. 2000). With a 2010 population of more than 37 million people, California accounts for roughly one-eighth of the human population of the United States (US Census Bureau 2013), has the largest agricultural production of any state in the country (USDA 2007), and has one of the highest average land values in the nation (Davis and Heathcote 2007). Conserving biodiversity in California is therefore both enormously important and extremely difficult from an economic and political standpoint and requires strong scientific guidance and the collective will of multiple stakeholder groups.

Formal species protection in California is accomplished via the California Endangered Species Act and/or the Federal Endangered Species Act. The California Department of Fish and Wildlife (CDFW) is responsible for implementing the latter. As of January 2014, over 150 animals in our state were listed as threatened or endangered under either one or both acts. To help preclude the need to list additional species, the CDFW administratively designates Species of Special Concern. The intent of designating Species of Special Concern is to (1) focus attention on animals at conservation risk by the CDFW, other state, local, and federal governmental entities, regulators, land managers, planners, consulting biologists, and others; (2) stimulate needed research on poorly known species; and (3) achieve conservation and recovery of these animals before they meet California Endangered Species Act criteria for listing as threatened or endangered. Species of Special Concern carry no formal legal status but are widely viewed as one of the important front lines in species conservation planning and management. Regardless of the stakeholder group involved, whether members of the conservation, agricultural, or urban development communities, it is in everyone's best interest to maintain stable populations of Species of Special Concern to avoid the need for formal listing.

The Species of Special Concern designation is used to promote conservation in various ways by the CDFW, land managers, and others to promote conservation. For example, Species of Special Concern are considered "Species of Greatest Conservation Need" in California's Wildlife Action Plan (Bunn et al. 2007, http://www.wildlife.ca.gov/SWAP). State Wildlife Action Plans outline the steps needed to conserve these taxa before they become rarer and more costly to protect and provide access to funds for this purpose. Species of Special Concern are also considered when evaluating environmental impacts under the California Environmental Quality Act (California Public Resources Code Sections 21000-21177). The California Environmental Quality Act requires state agencies, local governments, and special districts to evaluate and disclose impacts to wildlife and habitat from proposed projects. Specifically, Species of Special Concern may meet the definitions of endangered, rare, and/or threatened in Section 15380 of the California Environmental Quality Act guidelines. Also, Section 15065 relates to the standards under which the lead agency determines if impacts to biological resources should be considered significant. Impacts to Species of Special Concern are generally considered significant if they are based on factors such as population-level effects, proportion of the taxon's range affected by the project, and effects on habitat. Environmental impact reports that analyze and evaluate the potential impacts on Species of Special Concern caused by the proposed project must be prepared before planned projects can move forward. Large-scale planning efforts, such as Habitat Conservation Plans and Natural Community Conservation Plans, also may include conservation measures for non-listed, at-risk species including Species of Special Concern. In addition, Species of Special Concern are tracked by the California Natural Diversity Database (http://www.dfg.ca.gov/biogeodata/cnddb), an important source of information on species distribution. Federal land management agencies like the Bureau of Land Management and US Forest Service often add Species of Special Concern to their sensitive species lists to focus attention on these taxa. In all, the Species of Special Concern designation results in a greater depth of knowledge about species as well as proactive conservation aimed at maintaining or restoring populations to avoid the need for future, formal listing.

In this volume, we update and evaluate the original Species of Special Concern document for amphibians and reptiles (Jennings and Hayes 1994a). The first Species of Special Concern document compiled was for birds (Remsen 1978). Over the following three decades, documents have been published or updated for birds (Shuford and Gardali 2008), mammals (Williams 1986; Bolster 1998), and fishes (Moyle et al. 1989, Moyle et al. 1995). As these documents have matured and been revised, so too have the methods by which Species of Special Concern have been identified from the potential pool of candidate taxa. With the exception of the 2008 bird publication, previous iterations of these assessments were largely based on expert opinion. A list of native California taxa was assembled, screened for risk potential, and evaluated by a small team of experts (usually in consultation with many additional experts throughout the state). The most at-risk taxa not already listed under the California Endangered Species Act were then selected as Species of Special Concern.

The Species of Special Concern assessment process changed profoundly with the 2008 bird publication (Shuford and Gardali 2008). A key change, and one that we also follow here, was to formalize the criteria by which species receive this designation. Following Shuford and Gardali (2008) and current CDFW standards (http://www.dfg.ca.gov/wildlife/nongame/ssc/index.html), we created a set of eight metrics that capture the extent to which an amphibian or reptile species is at risk of extinction in California. We used this system to increase transparency, facilitate clear feedback from a broad group of individuals on our scoring, and enhance the ability of the CDFW and other agencies to replicate this process in the future. We then ranked all species by their summed metric scores, presented that ranking to a wide-ranging group of experts, and determined inclusion or exclusion from the special concern list. This approach provided a clear connection between data and ranking, and an explicit description of the most important factors contributing to ongoing declines. It also provided a strong connection between the evaluation process for different taxonomic groups and therefore greater uniformity in the methodology used among all CDFW Species of Special Concern publications.

The current volume is divided into two sections. In Part I (this section), we provide a detailed description of our methods, including the metrics and their scoring, outreach strategies for public input, locality mapping, and the roles of different contributors in producing the set of Species of Special Concern taxa. Following this is an overview of the results of our review and several quantitative descriptions of geographical, ecological, and taxonomic patterns of Species of Special Concern. We end with a discussion of the results and present recommendations for the conservation of amphibian and reptile Species of Special Concern in California. Throughout, we emphasize immediate research needs, both for particular species and for broader assemblages and landscapes within the state. Part II consists of a series of species accounts that provide a synopsis of information for each Species of Special Concern. Each account also includes a map documenting localities where the species has been collected or observed along with a depiction of its current range.

Throughout this document, we have used the peer-reviewed literature as our primary source of information and have included unpublished reports, web sites, and data from the field notes of professional and avocational herpetologists to fill in gaps in the primary literature. We rely primarily on the peer-reviewed literature because it has been evaluated by independent experts and deemed admissible into the scientific literature. However, we also recognize that the published literature for many species is sparse, and in those cases we also evaluated and included a large amount of unpublished information. Finally, we particularly emphasized the more recent, post-1990 literature, given the extensive review by Jennings and Hayes (1994a) of the earlier literature.

METHODS

Overview of Project Design and Process

The process of developing this document involved cooperation among several groups. The initial study design was developed collaboratively between the CDFW and the authors (Thomson, Wright, and Shaffer). We then assembled a Technical Advisory Committee comprising members with broad geographical and taxonomic expertise in California's amphibian and reptile fauna. This group developed the set of metrics used in evaluating potential Species of Special Concern, as well as a standardized format for species accounts. We then reached out to all segments of the herpetology community, including academics, land

and resource managers, avocational herpetologists, and the interested public for further information, feedback, and review at various points in the process. Our goal throughout was to keep our actions and decisions transparent and accessible to anyone with an interest in herpetological conservation in California.

We began by developing a current list of all native amphibian and reptile species and subspecies known to occur in the state (Appendix 1). Based on the broad knowledge of field herpetology represented by the authors and the Technical Advisory Committee, we used this list to develop a set of Special Concern nominees. Our goal was to include in this nominee list all taxa that anyone felt were declining or in need of protection in the state. The authors conducted preliminary reviews of each of these taxa, searching the literature and interviewing experts, and used these data to produce a set of preliminary scores for each of the nominees using the risk metrics. These scores were reviewed and refined by the Technical Advisory Committee and then further reviewed and refined based on input from the herpetological community at large. The authors and Technical Advisory Committee used the metric scores, as explained later in this document, to construct a set of taxa for inclusion as Species of Special Concern. After the list was finalized, we produced species accounts for each of the Species of Special Concern.

During this evaluative process, we compiled locality information for each taxon, which we then combined with data from the California Natural Diversity Database and Biogeographic Observation and Information System to produce distribution maps for each nominee species. The Technical Advisory Committee, the CDFW, and other experts reviewed these range maps, resulting in the maps in this document.

Species List, Taxonomy, and Units of Conservation

We developed our species list by compiling information from existing taxonomic lists and recent taxonomic literature. We included all recognized or proposed species, subspecies, and distinct population lineages that have been identified. We generally used the most recent revisionary studies, although we sometimes made decisions based on the degree to which the scientific community had accepted proposed changes and the quality and strength of data informing proposed revisions. Little consensus exists on taxonomy for certain groups (e.g., California mountain kingsnake, *Lampropeltis zonata*), and we tried to strike a balance between incorporating the most current, reliable information while also maintaining taxonomic stability in the face of current uncertainty. For example, Frost et al. (2006a) proposed a large number of taxonomic changes for California amphibians, often shifting species into new generic name combinations (e.g., the western toad, *Bufo boreas*, changes to *Anaxyrus boreas* under this scheme). These changes have been vigorously debated (Crother et al. 2009, Frost et al. 2009a, Pauly et al. 2009), and we have taken the conservative approach of retaining the traditional nomenclature.

We focused our evaluation primarily at the species level, although we also considered subspecies and (rarely) parts of an otherwise stable species range that appeared to be in decline. This follows most similar efforts to date in recognizing species as the fundamental units of conservation, while still acknowledging that significant diversity exists and should be maintained within species. This also allowed us to limit the extent to which taxonomic controversy might negatively impact important conservation efforts. For example, if we were to consider only species (or formally described subspecies), we would fail to consider currently unnamed populations in need of conservation action. The southern populations of the common garter snake (*Thamnophis sirtalis*) are an example of such a population, as are the southern populations of the Coast Range newt (*Taricha torosa*). Throughout this document we use the term "taxa" to refer to species, subspecies, or distinct populations.

Development of the Nominee List

The first stage in the process was to develop a list of nominee Species of Special Concern from the comprehensive list of taxa that occur in the state. We included all taxa from the previous amphibian and reptile Species of Special Concern document (Jennings and Hayes 1994a), those that were recently extirpated or possibly extirpated from the state, and all taxa currently listed under the Federal Endangered Species Act. We excluded any taxa that were already legally designated by the state (i.e., Endangered or Threatened under the California Endangered Species Act), because Species of Special Concern status would provide no further state-level protections. Although federally listed taxa also experience a higher level of protection than Species of Special Concern, we still considered them in the evaluation process because federal status could potentially be the result of conservation needs from parts of the species' range outside of California. Because of this, an assessment of each species focusing on its California range provides information about its status within the state.

We included additional nominee taxa that members of the Technical Advisory Committee identified as potentially at risk based on their experience with that taxon in the field. If at least one member of the committee suspected that a taxon might qualify as a Species of Special Concern, we included it for evaluation. Additional taxa were added through consultation with experts on specific species or larger taxonomic groups and by suggestion during the public comment phase of the project (see below). We then evaluated these taxa with the risk metrics and used the resulting scores as our primary basis for Species of Special Concern determination (see below).

Definition of Species of Special Concern

We define a Species of Special Concern as any native species, subspecies, or distinct population of amphibian or reptile occurring in the state that currently meets one or more of the following criteria (see also Comrack et al. 2008):

- Is extirpated from the state within the recent past;
- Is listed as federally, but not state, Threatened or Endangered and/or meets the state definition of Threatened or Endangered but has not formally been listed;
- Is experiencing, or formerly experienced, serious, noncyclical, population declines or range retractions that, if continued or resumed, could qualify it for state Threatened or Endangered status;
- Has naturally small populations and/or range size and exhibits high susceptibility to risk from any factor(s) that, if realized, could lead to declines that would qualify it for state Threatened or Endangered status.

We developed a set of risk metrics to address the latter two criteria. Taxa scoring high on these risk metrics were then judged to be prime candidates for inclusion on the list. Taxa meeting the first two criteria were included automatically. All taxa were scored for the risk metrics and included in our quantitative analyses.

Risk Metrics

Working with the Technical Advisory Committee and using CDFW criteria (http://www.dfg.ca.gov/wildlife/nongame/ssc/index.html), we developed a set of conservation risk metrics to quantify the level of threat to California's at-risk amphibians and reptiles. Although quantification of conservation risk is necessarily approximate, the metric approach allows for improved repeatability between Species of Special Concern updates and a framework for discussion and revision. Earlier Species of Special Concern documents were based largely on expert opinion and the use of risk metrics does not completely eliminate this important element of the assessment process. Rather, the risk metrics place expert opinion, as well as data, within a standardized framework that makes decisions more transparent. For example, our ecological tolerance metric provides a clear definition of how we quantified the ecological specialization of each taxon and how it relates to conservation risk. If, at a later time, additional data become available or other workers disagree with our interpretation of the existing data, there now exists a clear way in which this new information can be incorporated into the overall score for any species.

The possible score for each metric ranged from 0 (little or no risk) up to a maximum of 25 (high risk), reflecting the relative importance of the risk quantified by that metric. We weighted metrics that measure *documented* conservation concerns, such as declines in abundance, more highly than other metrics that focused on *potential* conservation concerns, such as life history factors that contribute to sensitivity. We did this for two reasons. First, our weighting reflects the emphasis on these factors in the definition of Species of Special Concern. Second, documented conservation concerns usually require more immediate management action and are likely more serious threats to survival than potential conservation concerns. The result of this decision is that some metrics, such as those measuring declines in distribution or abundance, affected the overall risk metric score more than, for example, a naturally small range size. The eight risk metrics are as follows.

i. Range Size

The range size metric estimates the percentage of California that each taxon occupies. Though this measure could be treated as continuous, we have approximated it with discrete categories for two reasons. First, we have little biological reason to believe that a taxon that occupies, for example, 35% of California is under any greater conservation risk than a taxon that occupies 42%. Both of these hypothetical taxa occupy moderate portions of the state and probably experience similar risk arising from the size of their range. Second, there is inherent uncertainty in many amphibian and reptile range predictions as portrayed in range maps, and we felt that it was more appropriate to broadly categorize ranges rather than attempt to precisely estimate them. We therefore categorize range size as *small*, which includes those taxa that are at immediate risk from relatively small scale disturbances; *medium*, which includes taxa that occupy a portion of the state that is big enough so that a single large catastrophic event would be unlikely to affect the entire range; and *large*, which includes those taxa that occupy such a large portion of the state that range size itself is unlikely to have any significant impact on threat. Patchiness and ecological specialization of species that limit range on a local scale are quantified in other metrics. Our aim for this metric is only to estimate the actual size of the species range. In the few cases where the known range is strictly limited by habitat specialization or limitation (e.g., desert populations of the regal ring-necked snake, *Diadophis punctatus regalis*, or the Gila monster, *Heloderma suspectum*) and the taxon almost certainly does not occur between isolated habitat patches, we treated the known populations as individual polygons in scoring this metric.

(I) RANGE SIZE (% OF CALIFORNIA OCCUPIED)	SCORE
Small (<10%)	10
Medium (10–50%)	5
Large (>50%)	0

ii. Distribution Trend

The distribution trend metric aims to quantify documented decreases in the overall range of each taxon based on extirpation of previously known localities. The total score for this metric comes from two sources. First, we attempted to quantify the extent of known range reductions, scoring them using the categories below. We classified the extent of range reduction into discrete categories for similar reasons as range size. We then added an additional 5 points if the documented reduction in range appears to have been ongoing since the last Species of Special Concern document was published (Jennings and Hayes 1994a) and has not yet stabilized or reversed. We did this to increase the weight of declines that are continuing at present, and which therefore are likely to continue in the immediate future. As a result, a species might attain a particular score through either a documented reduction or a less severe reduction that is ongoing. In scoring this metric, we used peer-reviewed published data whenever possible. The best data for this metric came from repeated field surveys of habitat through time, and we used them whenever they were available. However, datasets of this type are, at present, uncommonly available for amphibian and reptiles of California.

(II) DISTRIBUTION TREND	SCORE
Severely (>80%) reduced	20
Greatly (>40–80%) reduced	15
Moderately (>20–40%) reduced	10
Slightly (<20%) reduced or suspected of having been reduced but trend unknown	5
Stable (~0% reduced) or increasing	0

Add 5 additional points if negative trend is ongoing for a total of 25 points possible for this metric.

III. Population Concentration/Migration

This metric focuses on whether features of the life history of individual taxa, such as migration events or aggregations, make them naturally vulnerable to decline or extirpation. For instance, taxa that migrate to breed in ponds are exposed to additional risk during the migration itself (e.g., road crossings) as well as increased risk while concentrated in the breeding habitat. This latter risk could come about if a catastrophic event occurs during the breeding concentration (e.g., if a toxic spill or group of predators killed the breeding animals) or because the actual breeding site is destroyed (e.g., draining of the aquatic breeding habitat). We score this trait either *present* or *absent* based on the available life history data for each taxon.

(III) POPULATION CONCENTRATION/MIGRATION	SCORE
Vulnerable life stages present	10
No vulnerable life stages	0

IV. Endemism

The endemism metric captures the percentage of a species' entire range that occurs in California. Endemism determines the extent to which conservation actions in California are likely to impact the taxon's persistence range-wide. From another perspective, this is a way of measuring California's responsibility to conserve individual species. Taxa whose range is completely, or nearly completely, contained within California's borders are in need of greater conservation consideration from our state than taxa whose range only extends peripherally into California. We recognize that this presumes appropriate conservation measures are also being implemented in other areas of North America (including Mexico and Canada). We again made this measure discrete in recognition of the inherent uncertainty in our knowledge of range limits.

(IV) ENDEMISM (% OF ENTIRE RANGE IN CALIFORNIA)	SCORE
100% (endemic)	10
>66–99%	7
33–66%	3
<33%	0

V. Ecological Tolerance

This metric measures ecological specialization. Species that are narrow specialists on specific ecological resources (such as habitat, prey, temperature regimes) are inherently more sensitive to ecological disturbance than species that can tolerate a wider range of ecological conditions. In addition to the degree of specialization, we also considered the extent to which the resource that each taxon specializes on is common or rare. For instance, several saxicolous (rock loving) lizard species (e.g., the leaf-toed gecko, *Phyllodactylus nocticolus*) use rocky habitats that occur throughout extensive areas of the species' total range. We scored cases like this as specialists on a common resource. Conversely, vernal pool breeding amphibians (e.g., Couch's spadefoot, *Scaphiopus couchii*) require temporary aquatic pools that are rare throughout their range for successful breeding. We scored these taxa as specialists on a rare resource. We adjusted the rareness of the resource with respect to its availability within the species' range, rather than its availability within the state.

(V) ECOLOGICAL TOLERANCE	SCORE
Narrow ecological specialist on a rare resource	10
Narrow ecological specialist on a common resource	7
Moderate ecological specialist	3
Broad ecological tolerance	0

VI. *Population Trend*

The population trend metric captures changes in abundance at localized, population-level sites. This is distinct from the distribution trend, which measures extirpation of localities; population trend captures declining abundances at localities that are not extirpated. In many cases, distributional declines as measured by distribution trend will be associated with earlier declines as measured by population trend. This raises the potential of scoring taxa twice for the same decline. To avoid this, we scored population declines that have led to extirpation under the distribution trend metric. We gave those same taxa high scores for the population trend metric only if additional population declines have been documented at currently extant sites. We scored population trend in the same way as distribution trend, first scoring the extent of the decline and then adding an additional 5 points if evidence suggests that the trend is ongoing. As a result, a species might attain a particular score through either a documented reduction or a less severe reduction that is ongoing.

(VI) POPULATION TREND	SCORE
Severe declines (>80% reduced)	20
Great declines (>40–80% reduced)	15
Moderate declines (20–40%)	10
Slight (<20%) or suspected declines	5
Stable (~0% reduced) or increasing	0
Add 5 additional points if declines are ongoing.	

VII. *Vulnerability to Climate Change*

The climate change metric measures a taxon's sensitivity to the projected effects of climate change. We scored this metric using the projected impacts on California landscapes based on the California Climate Action Team assessments (Cayan et al. 2008a), followed by our interpretations of how these impacts are likely to affect each taxon based on life history and habitat requirements. For example, climate projections suggest that snowpack in the Sierra Nevada is likely to decrease by 30–90% (depending on carbon emissions and the climate model used) over the next 100 years, leading to a narrower window of time over which the spring snowmelt will occur (Maurer and Duffy 2005, Cayan et al. 2006, Maurer 2007). This is likely to have an impact on the snowmelt-dependent aquatic habitats that many Sierran amphibians use for one or more life stages, and may also reduce the time period over which moist microhabitats will occur in forest ecosystems. Other impacts that we considered for this metric included changing hydrology (amount and variation of precipitation), temperature, wildfire frequency and intensity, and changes in the extent of habitat and vegetation types. Given our imprecise knowledge of both future climate change effects and their impacts on species, we discretized this impact into four broad categories.

(VII) VULNERABILITY TO CLIMATE CHANGE	SCORE
Highly sensitive	10
Moderately sensitive	7
Slightly sensitive	3
Unlikely to be sensitive	0

viii. Projected Impacts

The projected impacts metric estimates the effect that future threats may have on each species over the near term (20 years). It does not incorporate threats arising from changing climate, because these are captured in a separate metric. This includes impacts stemming from known threats, such as planned or projected habitat loss and, to a lesser extent, impacts from irregularly occurring threats, such as disease outbreaks. Given the potential for these risks to be reduced by management, plus the inherent uncertainty associated with complex projections, we considered potential threats to be of relatively less importance than documented threats such as population declines.

(VIII) PROJECTED IMPACTS (OF THREATS OVER THE NEXT 20 YEARS)	SCORE
Serious	10
Moderate	7
Slight	3
No substantial impact	0

Scoring Nominee Taxa

We scored all of the nominee taxa for each of the eight metrics based on the best available evidence. To begin with, the primary authors produced a brief summary of the state of conservation knowledge for each nominee taxon and used these summaries to perform a preliminary scoring assessment. In making these assessments, we included the peer-reviewed literature, unpublished reports, survey data, field notes, and the opinions of knowledgeable biologists. In several cases, few data were available to make assessments for a given metric. In these cases, if the data appeared to be strong enough to clearly indicate that a threat was present, we scored that taxon using the most precise estimate that we were able to make. In cases where no data were available or the limited data were ambiguous, we scored taxa as "data deficient" for that metric. Following these preliminary assessments, we circulated all of the scores and taxon summaries to the Technical Advisory Committee for review and further input. In the rare cases of substantial disagreement, we discussed the issue and evaluated the data as a group, and reached a consensus on the most reasonable score for a given taxon.

After this preliminary scoring process was complete, we created an overall score for each taxon by summing its metric scores and dividing by the total score possible for that taxon (Total Score/Total Possible). Using the ratio of total score to total possible score allowed us to normalize the scores across varying levels of data deficiencies. For example, in cases where a taxon was scored as data deficient for one or more metrics, the total possible score was lower than would be the case if all metrics had been scored. This would result in a lower risk assessment due to uncertainty as opposed to data, and we used standardization by the Total Possible score in order to focus on documented risks.

Public Comment

After the scoring assessments were complete, we opened a 60-day public comment period by posting all of our initial findings on the project's website and sought input widely on herpetological and conservation-oriented email lists and websites (Appendix 2). We requested comments and feedback on the initial set of scores, additional data that could inform the scoring (particularly for the metrics that had been scored as data deficient), and feedback on the process to date. When individuals suggested changes to the metric scores, we asked for a short explanation of what should be changed and why, along with any data and/or field notes that were available to support the proposed change. At the close of the public comment period, we compiled and evaluated all of the information that we received (see Results, Public comment). We evaluated each proposed change on a case-by-case basis, usually making the change if it was reasonable, supported by information (in the form of unpublished reports, data, or field notes), and not in strong conflict with other existing data. In cases where a suggested change was in strong conflict with other data, we asked that the contributor supply additional data justifying their viewpoint and made a decision on the final resolution of any conflicting information.

We also asked that contributors send additional data that could be incorporated into the locality maps (see below). To facilitate this process, we supplied a standardized data sheet similar to that used for data submission to the California Natural Diversity Database. These localities were added to the California Natural Diversity Database and to our set of existing localities, and they were used in developing range maps.

Ranking and Determination of Species of Special Concern Status

After incorporating the information received during the public comment period, we worked with the Technical Advisory Committee to develop the set of Species of Special Concern taxa. Taxa with the highest scores were included on the list, while those with intermediate scores were evaluated on a case-by-case basis; this combined approach was similar to that used in the Bird Species of Special Concern (Shuford and Gardali 2008). Specifically, taxa that had intermediate scores but had a combination of exceedingly small range size, extreme ecological specialization, and high projected impacts were included as Species of Special Concern. In essence, this approach weights the combination of these factors more heavily in order to meet the last of the four criteria for inclusion as a Species of Special Concern, "small populations and/or range size and exhibits high susceptibility to risk from any factor(s), that if realized, could lead to declines that would qualify it for state Threatened or Endangered status" (Comrack et al. 2008).

We further ranked Species of Special Concern into three priority categories based on the severity and immediacy of threats affecting each taxon. Priority 1 Species of Special Concern are those taxa that are likely to experience severe future declines and/or extirpation without immediate conservation actions. Priority 2 Species of Special Concern require substantial conservation and management actions, although the threats facing them are less immediate and severe than those in Priority 1. Finally, Priority 3 Species of Special Concern are clearly at risk but likely are not experiencing a substantial and immediate threat of extirpation, although the potential for this threat to develop exists if no management actions are undertaken. One of the primary goals of the Species of Special Concern designation is to identify taxa for which managers can undertake relatively small scale and achievable conservation actions that will negate the need for more costly and serious listings at a later date. Priority 3 taxa are prime candidates for such efforts.

Watch List and Additional Taxa in Need of Research and Monitoring

Taxa that were previously considered Species of Special Concern but are no longer included comprise a Watch List (Appendix 3). Appendix 3 includes an explanation for each taxon's change in status and discusses future conservation concerns regarding Watch List taxa. In Appendix 4, we discuss several other taxa in need of research and monitoring that did not warrant inclusion as Species of Special Concern. Some of these were taxa that had scores indicating a lower, but still substantial, amount of risk. Although we decided that they were at a lower priority than the Priority 3 Species of Special Concern and therefore should not be so designated, they formed a group of species to reevaluate in the future. We were also missing important information for some taxa that would have allowed us to make more informed judgments about conservation status. We devote a paragraph to each of these additional taxa in need of research and monitoring in Appendix 4, briefly describing the threats facing each and outlining research and management needs.

Species Accounts

We prepared a species account for each Species of Special Concern that summarized our findings and the relevant aspects of the taxon's biology. We also provided management and research recommendations for each taxon. These accounts follow a standardized format containing each of the following sections.

Status summary. The status summary is a short explanation of each animal's current and former status as a California Species of Special Concern, including its priority level. In the first version of the Amphibian and Reptile Species of Special Concern monograph, Jennings and Hayes (1994a) categorized each taxon according to whether they felt it was a Species of Special Concern or met the criteria for listing as Threatened or Endangered under the California Endangered Species Act. However, this strategy led to some potential confusion because the Jennings and Hayes (1994a) Threatened and Endangered categories did not correspond to actual state listing categories, nor had taxa they described as Threatened or Endangered undergone the rigorous status evaluation required to assess status under the California Endangered Species Act. To avoid this confusion, we used Priority categories (1, 2, or 3) to convey similar information on relative severity of threat as represented in the ranking of Species of Special Concern. This section also contains the overall metric score.

Identification. The identification section summarizes and explains the diagnostic characters for each animal, providing a guide for identifying it in the field. This section also explains how to differentiate each taxon from similar species with which it may be confused. Several taxa within the state are members of morphologically similar species complexes that have been identified primarily based on molecular data. In some of these cases, accurate identifications using morphological characters alone are difficult or impossible, and we generally recommend that biologists rely on geographic range. We also provide references to the taxonomic literature to guide the reader to the more thorough and technical descriptions of morphology that are beyond the scope of this document.

Taxonomic relationships. In addition to identification information, we provide a summary of the taxonomic status of each animal. This section contains information on current controversies over scientific names, at either the species or higher taxonomic levels. It also summarizes our current understanding of phylogenetic relationships, intraspecific variation, and species boundaries among closely related taxa.

Life history. This section summarizes the current state of knowledge for each taxon's life history, which broadly includes ecology, natural history, and breeding biology. As an exhaustive review of life history information would be enormous for some taxa, we focused on information that is most relevant to current and future management actions and to the risk metrics. Specifically, we concentrated on information that relates to timing and duration of reproductive activity, daily and seasonal activity, and dietary information. Because management efforts for many taxa could be greatly enhanced by a better understanding of life history, we attempted to point out the areas that require further study rather than speculating about the details of life history where the data are weak. We emphasized data from California populations, but used data from other areas of the range or similar species when those were the best available data. We note when we used data from non-California populations and why we believed that the data could be accurately applied.

Habitat requirements. This section focused on the current state of knowledge concerning habitat use, preferences, and requirements. We attempted to distinguish between habitat *preferences*, the habitats in which the taxon is most frequently found, and habitat *requirements*, which are the characteristics of the habitat that the taxon requires for survival over long timescales.

Distribution. This section describes each animal's current distribution and makes an assessment of changes that have occurred throughout its documented history in the state. We focused primarily on the known distribution within the state, although we also discussed the distribution outside of California if applicable. Finally, to stimulate additional fieldwork, we point out areas where the distribution is poorly known or there is a high probability of significant new localities being discovered.

Trends in abundance. This section reviews information relating to changes in abundance throughout each taxon's documented history. For current population status, we used quantitative population-level analyses where available. However, these kinds of data are rare. Historical data tend to be spotty and incomplete for amphibians and reptiles, and much of the historical information comes from nonquantitative sources, including field reports and personal communications from experienced field biologists.

Nature and degree of threat. This section contains a detailed description of the principal threats that each taxon faces. We highlighted both the nature and severity of different threat sources, while discussing any uncertainty and conflicting data in the literature associated with these threats. We evaluated the weight of evidence and discussed what threats might be playing the largest role(s) in causing declines.

Status determination. This section connects the information on different sources of threat to the metric scores and Species of Special Concern priority categories. We explained the rationale for our determination and the seriousness of the different major threats facing each taxon.

Management recommendations. This section makes recommendations aimed at achieving sound, biologically based management and status improvement for each Species of Special Concern. Wherever possible, we made these recommendations both taxon-specific and action-oriented to allow conservation resources to be put directly into management efforts, rather than into further development of management strategies. We did, however, recommend further research and strategy development as a prerequisite to effective management for taxa that lacked necessary data.

Monitoring, research, and survey needs. This section outlines the additional information necessary to achieve effective management and status improvement. In general, information needed to inform management actions falls into the general areas of monitoring, research, or surveys, and we discuss each as appropriate.

Maps. We developed locality maps to complement the distribution information in the text for each taxon by compiling data from museum collections, state agency databases (e.g., California Natural Diversity Database), and other online databases (e.g., North American Field Herping Association) (Table 1). Data from the CDFW's California Natural Diversity Database and the Biogeographic Information and Observation System were assessed up through April 2012. Museum locality data from HerpNet and the Global Biodiversity Information Facility were assessed through February 2012. Our goal was to develop a set of annotated and geo-referenced localities that accurately describe each taxon's range. Records that appeared to be possibly erroneous (i.e., those that occurred in unexpected areas) were checked individually and excluded in those instances where no supporting information could be found or where the specimens were misidentified (see individual species accounts). We attempted to verify all records coming from online databases and the public by requesting, minimally, photo vouchers or detailed field notes to substantiate the record. The California Natural Diversity Database contains localities that lack this information, so we followed up on questionable records by attempting to contact the individual(s) that initially reported the record. We submitted most new localities that we gathered to the California Natural Diversity Database to make them available for future workers. In a few cases, we could not obtain permission to include localities in the database, so these were included in the maps in this volume, but

TABLE 1

List of museum collections and other data sources that were queried for locality records

Museum Collections	Other Sources
American Museum of Natural History	Cal Photos
Arizona State University	California Biogeographic Information and Observation System
Brigham Young University	California Natural Diversity Database
California Academy of Sciences	Field Notes
California Academy of Sciences, Stanford University Collection	Literature Records
California State University, Chico	Mendocino Redwood Company
Carnegie Museum of Natural History	North American Field Herping Association
Cincinnati Museum Center	Our Own Surveys
Cornell University Museum of Vertebrates	Public Input/Personal Communications
Humboldt State University	US Forest Service
Los Angeles County Museum	US Geological Survey
Museum of Comparative Zoology, Harvard University	
Museum of Vertebrate Zoology, University of California, Berkeley	
National Museum of Natural History	
Royal Ontario Museum	
San Diego Natural History Museum	
Santa Barbara Museum of Natural History	
Slater Museum of Natural History	
Sternberg Museum of Natural History	
University of Alberta Museum of Zoology	
University of Arizona Museum of Natural History	
University of California, Davis – Zoology Collection	
University of California, Santa Barbara	
University of Colorado Museum of Natural History	
University of Michigan Museum of Zoology	
University of Nevada Reno	
University of Texas at El Paso	
Yale Peabody Museum	
Zoological Institute of the Russian Academy of Sciences	

excluded from the database. The complete geospatial dataset and associated metadata from this project are accessioned in the CDFW's Biogeographic Information and Observation System (BIOSds644).

After removing erroneous and questionable records from the data, we developed point locality maps with our CDFW Geographic Information System specialist by projecting all localities for each taxon to the California (Teale) Albers projection (figure 1). We used the California Wildlife Habitat Relationships (http://www.dfg.ca.gov/biogeodata/cwhr) mapping protocol to develop range maps for each taxon using these localities. California Wildlife Habitat Relationships is a comprehensive information system for the state's terrestrial vertebrates that seeks to integrate data on species life history, habitat needs, and ranges.

To develop species range estimates, we selected the full set of US Department of Agriculture ecoregion subsections that contained at least one locality and used these as a starting point for range maps (figure 2). We then overlaid existing range maps from California Wildlife Habitat Relationships, as well as data layers for habitat types, watersheds, elevation, land use, and urbanization. Using these draft maps,

FIGURE 1 Development of range maps for each species. We began by plotting localities on a base map in a geographic information system (A). We then selected the intersection of these localities with an objective geographic object such as US Department of Agriculture (USDA) Ecoregion subsection boundaries, elevational boundaries, or watershed boundaries (in this example, watershed boundaries were used). The particular geographic object that we used varied according to the biology of the taxon (e.g., watershed boundaries for stream-dwelling amphibians, elevation for high-elevation taxa) (B). We then interpolated between the geographic objects that had known localities using expert opinion to develop an approximate range boundary (C). The approximate range boundary and known localities were then drawn together to produce a map for this document (D).

we restricted range boundaries based on ecoregion subsection, watersheds, and other data layers to a more biologically realistic species range. In accordance with the California Wildlife Habitat Relationships guidelines, our goal was to define the current maximum geographic extent of the species within the state, where maximum geographic extent is defined as the area within the range boundary where the species can potentially be expected to occur given suitable habitat conditions. We delineated the range boundaries to minimize errors of omission, even to the extent of allowing some commission error. For certain species, significant fractions of the range are potentially extirpated (see the species accounts for additional detail). No range shading is included for the species that are presumed extirpated in California (see individual species accounts).

In most cases, we defined the edges of species ranges by selecting meaningful landscape characteristics to set a boundary, such as elevation, rivers, or watershed boundaries. Our goal was to identify specific places on the landscape

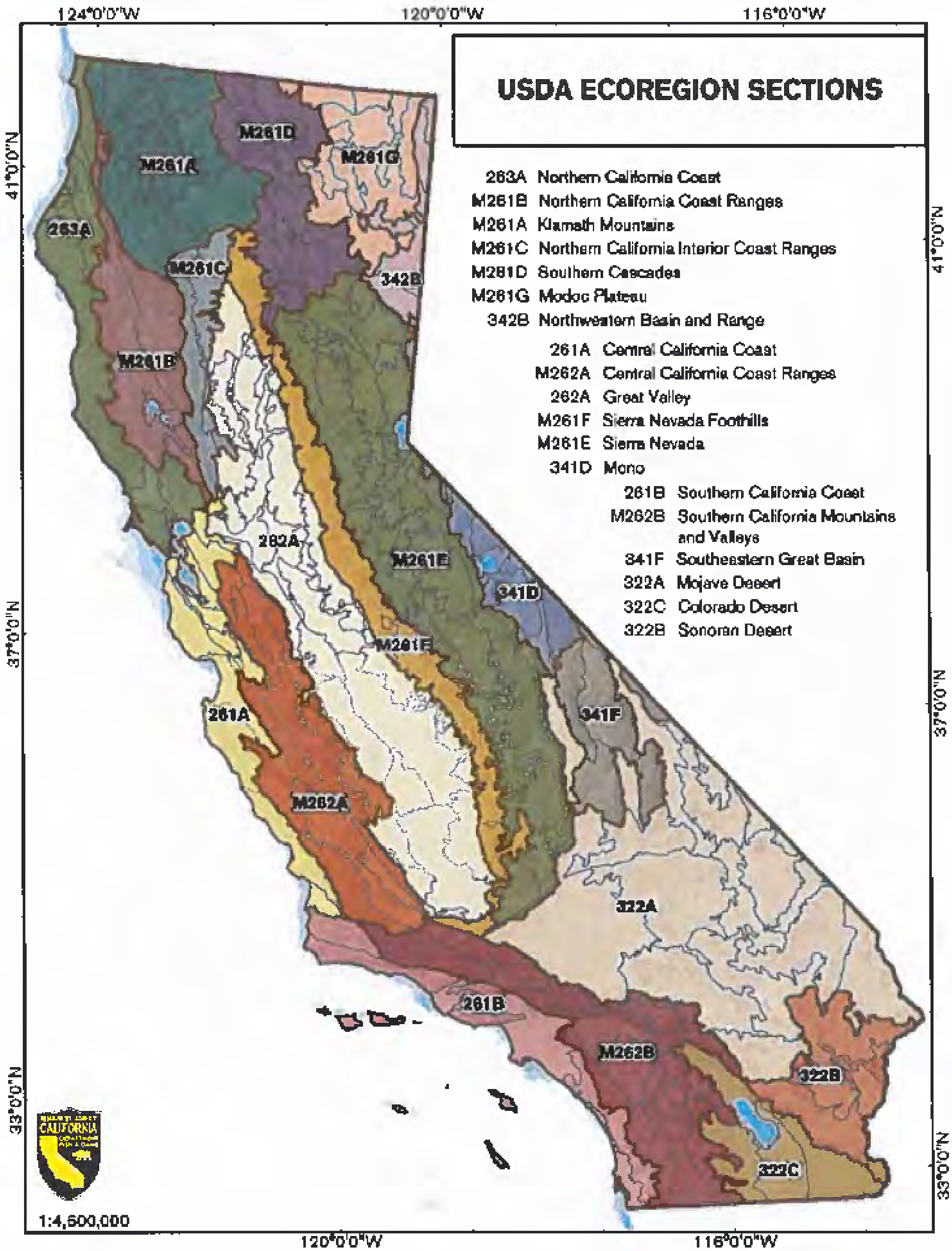


FIGURE 2 United States Department of Agriculture (USDA) Ecoregion subsections which were used in developing range maps.

where future surveys could be conducted to further characterize the species' range. Range maps that lack specific and objective boundaries provide only generalized starting points for such surveys. In total, our range maps present comprehensive estimates based on currently available species locality data and represent our best effort to use these data to approximate a species range, fully recognizing that such ranges are hypotheses to be tested rather than fixed entities.

Review Process

All phases of this project were reviewed by the three authors, the Technical Advisory Committee, and the CDFW. Most parts of the project were also subject to a wider review from members of the herpetological conservation community. For each taxon, we asked at least two experts to review the species account, including the maps and any appendix information. Finally, the Technical Advisory Committee, the CDFW, biologists from state and federal land management agencies, and other interested parties reviewed the finished manuscript as a whole.

RESULTS

Status Lists

We identified 217 native species, subspecies, and distinct population segments that are, or are suspected to be, present in California (Appendix 1). Seventy-three of these taxa were considered nominee Species of Special Concern and underwent evaluations using the risk metrics. Four additional taxa were initially considered for evaluation but were subsequently state listed and removed from further consideration (see Watch List). Of the 73 candidates, we determined that 28 did not merit special status at this time and 45 met our criteria for Species of Special Concern status (figures 3 and 4 and Table 2). Three of these species qualified for Species of Special Concern status by definition because they were listed under the Federal, but not the California, Endangered Species Act (the arroyo toad, *Bufo californicus*; the California red-legged frog, *Rana draytonii*; and the Yosemite toad, *B. canorus*). We conducted the scoring separately for the two subspecies of the western pond turtle (*Emys marmorata marmorata* and *E. m. pallida*) because the severity of threats facing one population appeared to be larger than those facing the other. However, both populations merited inclusion as Species of Special Concern, resulting in a single species account where threats to each population are discussed separately.

We ranked the Species of Special Concern taxa according to the magnitude of risks that they face, with the two pond turtle populations receiving separate Priority scores. This resulted in 16 taxa categorized as Priority 1, 14 as Priority 2, and 12 as Priority 3. Three additional species clearly qualify as Species of Special Concern, although the scarcity of field records precludes their accurate prioritization at this time: the regal ring-necked snake (*Diadophis punctatus regalis*), Cope's leopard lizard (*Gambelia copeii*), and the Gila monster (*Heloderma suspectum*). In these three cases, we have not assigned a priority score pending additional fieldwork.

Performance of Metrics

Spearman's rank correlations among the eight risk metrics indicated that approximately two-thirds (18/28) of the possible pairwise correlations among metrics were significant (Table 3). Some metrics were not highly correlated with other metrics (e.g., endemism was not correlated with any other metrics), while other metrics were correlated with four or five other metrics (e.g., distribution trend, population concentration/migration, and population trend). Some pairs of correlations indicated that there was considerable overlap in the scores received across taxa. The strongest correlation among metric scores was between distribution trend and population trend ($\rho = 0.66$, $p < 0.001$), indicating that animals that have

TABLE 2

List of California amphibian and reptile Species of Special Concern and priority designations

Three species qualify as Species of Special Concern, although the scarcity of data precludes their accurate prioritization at this time (see text for further discussion)

Scientific Name	Common Name	Priority
Ambystoma macrodactylum sigillatum	Southern long-toed salamander	2
Aneides flavipunctatus niger	Santa Cruz black salamander	3
Anniella pulchra	California legless lizard	2
Arizona elegans occidentalis	California glossy snake	1
Ascaphus truei	Coastal tailed frog	2
Aspidoscelis tigris stejnegeri	Coastal whiptail	2
Batrachoseps campi	Inyo Mountains salamander	3
Batrachoseps minor	Lesser slender salamander	1
Batrachoseps relictus	Relictual slender salamander	1
Bufo alvarius	Sonoran Desert toad	1
Bufo californicus	Arroyo toad	1
Bufo canorus	Yosemite toad	1
Coleonyx variegatus abbotti	San Diego banded gecko	3
Crotalus ruber	Red diamond rattlesnake	3
Diadophis punctatus regalis	Regal ring-necked snake	Undefined
Dicamptodon ensatus	California giant salamander	3
Elgaria panamintina	Panamint alligator lizard	3
Emys marmorata marmorata	Northern western pond turtle	3
Emys marmorata pallida	Southern western pond turtle	1
Gambelia copeii	Cope's leopard lizard	Undefined
Heloderma suspectum	Gila monster	Undefined
Kinosternon sonoriense	Sonora mud turtle	1
Masticophis flagellum ruddocki	San Joaquin coachwhip	2
Masticophis fuliginosus	Baja California coachwhip	3
Phrynosoma blainvillii	Coast horned lizard	2
Phrynosoma mcallii	Flat-tailed horned lizard	2
Rana aurora	Northern red-legged frog	2
Rana boylii	Foothill yellow-legged frog	1
Rana cascadae	Cascades frog	2
Rana draytonii	California red-legged frog	1
Rana pipiens	Northern leopard frog	1
Rana pretiosa	Oregon spotted frog	1
Rana yavapaiensis	Lowland leopard frog	1
Rhyacotriton variegatus	Southern torrent salamander	1
Salvadora hexalepis virgultea	Coast patch-nosed snake	2
Scaphiopus couchii	Couch's spadefoot	3
Spea hammondii	Western spadefoot	1
Taricha rivularis	Red-bellied newt	2

Taricha torosa, Southern populations	Coast range newt	2
Thamnophis hammondii	Two-striped garter snake	2
Thamnophis sirtalis, Southern populations	Common garter snake	1
Uma notata	Colorado Desert fringe-toed lizard	2
Uma scoparia	Mojave fringe-toed lizard	3
Xantusia gracilis	Sandstone night lizard	3
Xantusia vigilis sierrae	Sierra night lizard	3

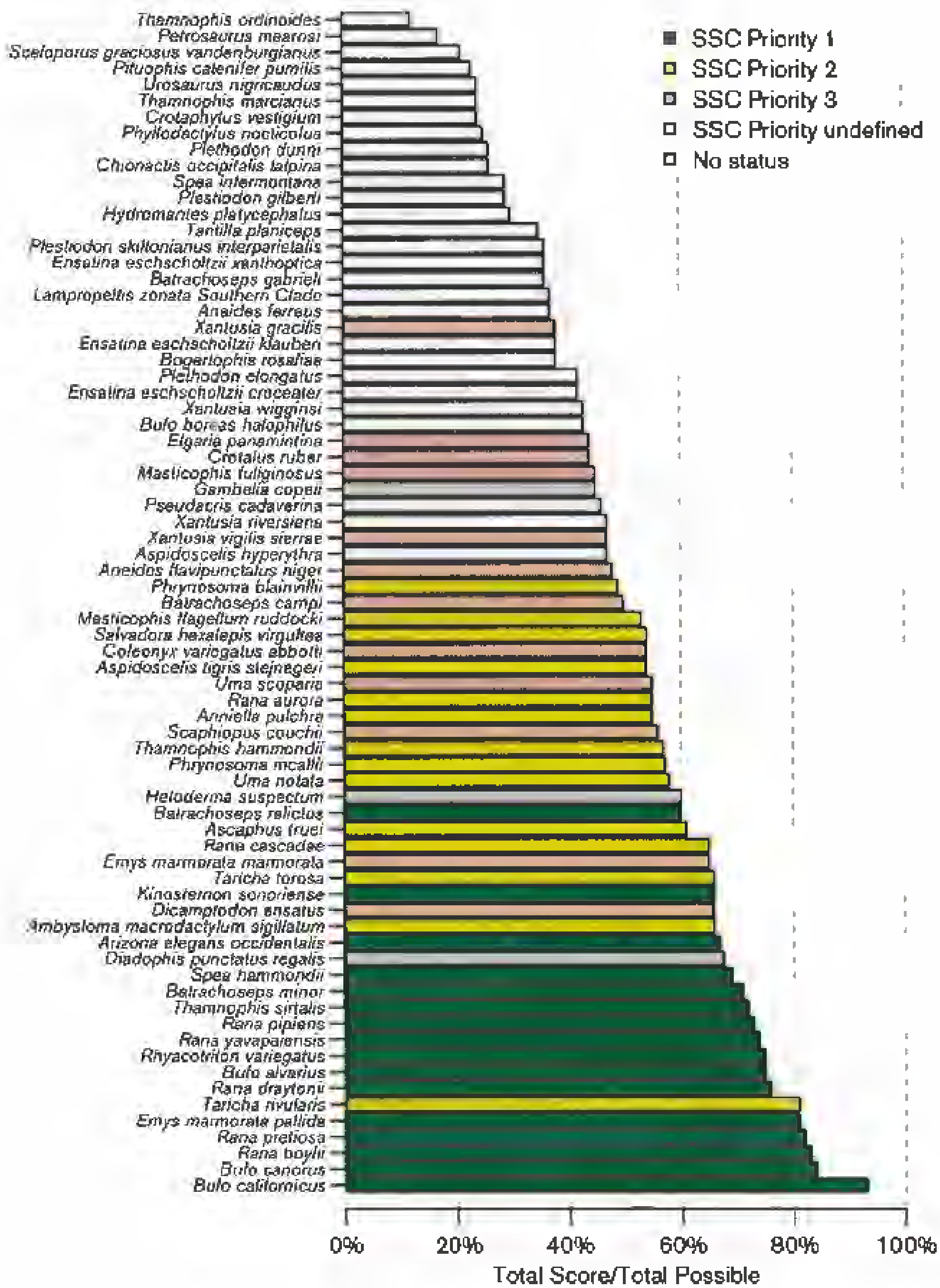


FIGURE 3 Total Score/Total Possible for 73 taxa evaluated for Species of Special Concern status.

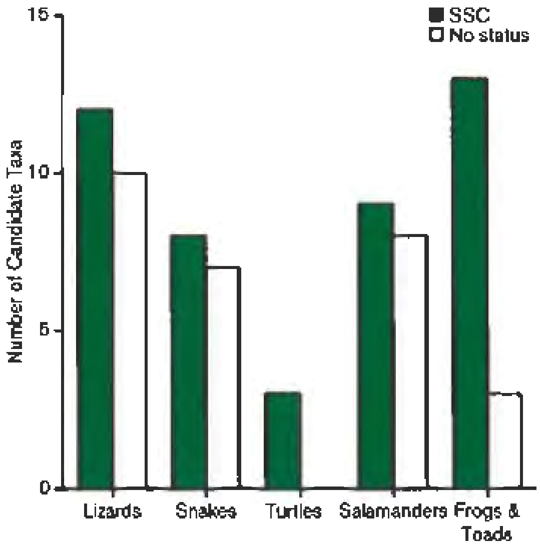


FIGURE 4 Number of taxa in each status category among the 73 nominee taxa by taxonomic group. Species of Special Concern (SSC) are represented by filled bars. Open bars are nominee taxa that did not receive SSC status.

been extirpated from historic localities tended to also be undergoing declines in abundance in currently occupied sites. Taxa experiencing high levels of extirpation also tended to have vulnerable life stages (correlation between distribution trend and population concentration/migration) and be more at risk from future threats (correlation between distribution trend and projected impacts). Those with vulnerable life stages also tended to be declining in abundance (correlation between population concentration/migration and population trend) and were more vulnerable to climate change (correlation between population concentration/migration and vulnerability to climate change).

All but two metrics (range size and endemism) were significantly positively correlated with Total Score/Total Possible (Table 3). Distribution trend and population trend were a priori given the greatest weight (each had a maximum score of 25 vs. a maximum score of 10 for all other metrics), and they were also the most highly correlated with Total Score/Total Possible ($\rho = 0.77$ and 0.87, respectively). Projected impacts, population concentration/migration, and vulnerability to climate change also stood out as contributing to risk, although the relationships were not as strong ($\rho = 0.57$–0.68).

Principal components analysis of the metric scores for the 73 evaluated taxa showed that the first two principal component axes accounted for about half (54%) of the total variation. Distribution trend, population trend, and projected impact of threats loaded most strongly on the first principal component axis, and Species of Special Concern taxa tended to have positive values for this axis (80% of Species of Special Concern taxa positive; figures 5 and 6). Ecological tolerance and range size loaded most strongly on the second PC axis. However, there is little correlation with special concern status along this axis (figure 6).

Patterns in the Metric Scores

The Total Score/Total Possible ratios for the Species of Special Concern taxa were normally distributed with a mean of 63%, ranging from 38% to 93% (Shapiro–Wilk test for normality, $W = 0.98$, $p = 0.58$). Three of the Species of Special Concern taxa are also federally listed as endangered or threatened, and all of these taxa (California red-legged frog, *Rana draytonii*; arroyo toad, *Bufo californicus*; Yosemite toad, *B. canorus*) had a Total Score/Total Possible greater than 75%, occurring in roughly the top 20% of Species of Special Concern (figure 3). The top 20% of taxa were amphibians, with the exception of the western pond turtle (*Emys marmorata pallida*) (figure 3). In contrast, the lowest scoring 20% of Species of Special Concern taxa were all reptiles with the exception of the Santa Cruz black salamander (*Aneides flavipunctatus niger*) (figure 3). On average, turtles and frogs and toads had the highest scores among the five major taxonomic groups (frogs and toads, salamanders, lizards, snakes, and turtles; figure 7).

We were unable to score certain metrics due to a lack of data. Population trend had the largest number of deficiencies with 26% (19/73). Distribution trend was data deficient for 8% of

TABLE 3

Spearman's rank correlations (ρ) among the eight ranking criteria scores

Values below the diagonal are for the 73 candidate taxa. Values above the diagonal are for the 45 Species of Special Concern taxa

	RS	DT	PCM	EN	ET	PT	CC	PI	TS/TP
Range Size (RS)	—	−0.31*	−0.06	−0.16	0.29	−0.24	0.02	−0.07	−0.04
Distribution Trend (DT)	−0.27*	—	0.30	−0.29	−0.41**	0.46**	0.05	0.28	0.56***
Population Concentration/Migration (PCM)	−0.27*	0.41***	—	−0.12	−0.08	0.49**	0.39*	0.00	0.73***
Endemism (EN)	−0.13	−0.10	−0.04	—	0.26	−0.14	−0.15	−0.40**	0.02
Ecological Tolerance (ET)	0.21	0.00	0.09	0.15	—	−0.41*	0.12	−0.25	0.03
Population Trend (PT)	−0.31*	0.66***	0.57***	−0.01	0.02	—	0.39*	0.33*	0.79***
Vulnerability to Climate Change (CC)	−0.09	0.22	0.50***	0.01	0.30*	0.40**	—	0.08	0.47**
Projected Impact of Threats (PI)	−0.07	0.61***	0.23	−0.15	0.26*	0.65***	0.25*	—	0.25
Total Score/Total Possible (TS/TP)	−0.12	0.77***	0.66***	0.14	0.39***	0.87***	0.57***	0.68***	—

*$p < 0.05$; **$p < 0.01$; ***$p < 0.001$.

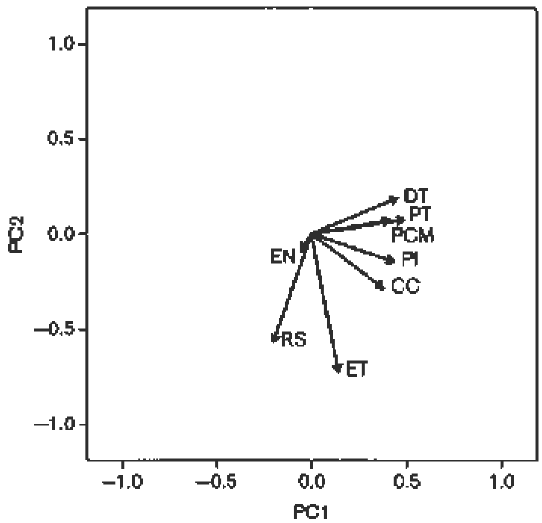


FIGURE 5 Vectors of PCA loading coefficients on first two PC axes. These two axes explain approximately half of the variation in metric score among the 73 nominee taxa. Distribution trend (DT), population trend (PT), population concentration/migration (PCM) loaded strongly onto PC1. Range size (RS) and ecological tolerance (ET) loaded strongly onto PC2. Climate change (CC) loaded equally and moderately on both axes, and endemism (EN) did not load strongly onto either axis.

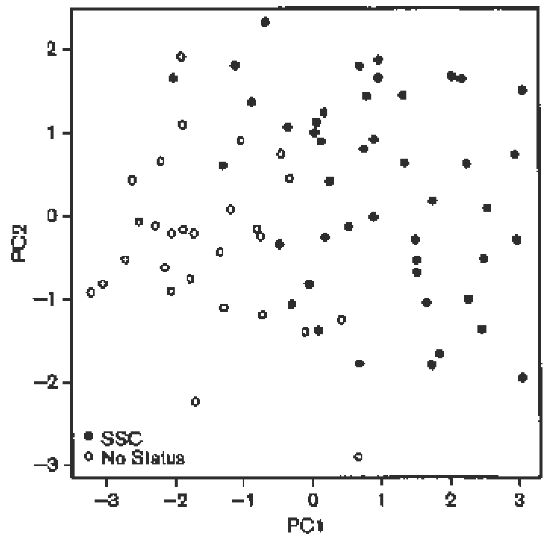


FIGURE 6 Distribution of all 73 taxa evaluated for Species of Special Concern (SSC) status along PCA axes 1 and 2. Most SSC taxa are positive for PC1 and most taxa with "No Status" are negative for PC1. There is little separation among taxa along PC2. SSC are represented by filled symbols. Open symbols are nominee taxa that did not receive SSC status.

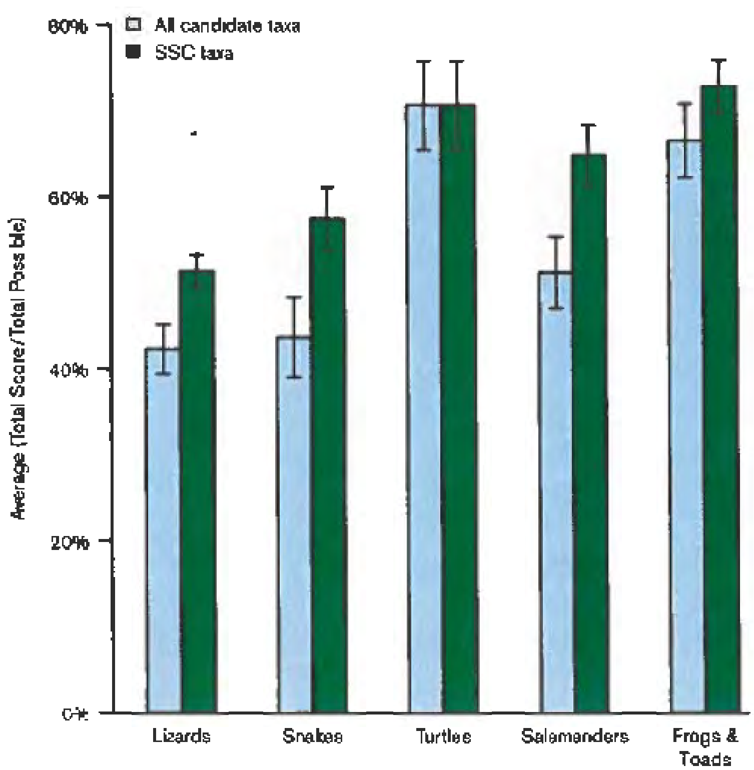


FIGURE 7 Average Total Score/Total Possible by taxonomic group. Filled bars are averages across the Species of Special Concern (SSC) taxa. Open bars are averages across all 73 nominee taxa. Error bars are standard errors.

taxa (6/73), and only a few taxa lacked data on vulnerability to climate change (2/73), projected impacts (3/73), and population concentration/ migration (2/73). Among the Species of Special Concern, nine species were data deficient for the critically important population trend metric: Cope's leopard lizard (*Gambelia copeii*), coast patch-nosed snake (*Salvadora hexalepis virgultea*), regal ring-necked snake (*Diadophis punctatus regalis*), California giant salamander (*Dicamptodon ensatus*), Gila monster (*Heloderma suspectum*), Sonora mud turtle (*Kinosternon sonoriense*), lowland leopard frog (*R. yavapaiensis*), Sonoran Desert toad (*B. alvarius*), and red-bellied newt (*Taricha rivularis*). Southern populations of the common garter snake (*Thamnophis sirtalis*) were data deficient for population concentration/migration, and the Oregon spotted frog (*R. pretiosa*) was data deficient for vulnerability to climate change. The Gila monster (*H. suspectum*) was data deficient for three metrics (distribution trend, population trend, and projected impacts), and the regal ring-necked snake was data deficient for the same three metrics plus population concentration/migration.

Certain geographic areas of the state emerged as experiencing a high degree of conservation risk, measured by the number of Species of Special Concern contained within them. At least two important geographic trends emerged from our analysis (figure 8). First, California ecoregions north of San Francisco Bay tended to have far fewer at-risk taxa than those from southern California (figure 8). In particular, the Southern California Coast, Southern California Mountains and Valleys, and the Mojave Desert ecoregions all contained a large number of Species of Special Concern (figures 2 and 8). Second, the geographic pattern of risk varied between amphibians and reptiles. Overall, reptiles experienced the highest risk in the three previously mentioned ecoregions as well as the Colorado Desert, while the northern ecoregions generally had only a single reptile Species of Special Concern (western pond turtle, *E. m. marmorata*). However, amphibian Species of Special Concern taxa were more evenly distributed among ecoregions across the state, with a slight peak in the mountains surrounding the Central Valley and in northern coastal California (generally 7–8 species) and a slight drop-off in the southern ecoregions (generally 5–6 species; figure 8).

To assess possible correlations between habitat type and conservation risk, we scored all 73 nominee taxa as predominantly terrestrial or aquatic, based largely on where reproduction takes place. Our categorization of aquatic versus terrestrial was not identical to that used in Jennings and Hayes (1994a), although it is broadly similar. We categorized amphibians based on their breeding biology—those that lay aquatic eggs and have free-living aquatic larvae were considered aquatic, whereas those with terrestrial eggs and direct development were considered to be terrestrial. Under these criteria, all frogs and toads were scored as aquatic, as well as the salamander genera *Ambystoma*, *Dicamptodon*, *Rhyacotriton*, and *Taricha*. Terrestrial salamander genera were all from the family Plethodontidae, and included *Aneides*, *Batrachoseps*, *Ensatina*, *Hydromantes*, and *Plethodon*. All lizards and snakes, including the semiaquatic garter snakes (*Thamnophis*) were considered terrestrial, since all either lay terrestrial shelled eggs or are live-bearing, and all spend the majority of their time on land. All of the turtles were considered to be aquatic since they spend the vast proportion of their lives, including all feeding and mating activities, in freshwater aquatic habitats. Categorizing taxa in this manner shows that there is an overall effect of habitat on Total Score/ Total Possible (One-way Anova, $p < 0.0001$; figure 9). The same pattern was true for aquatic versus terrestrial salamanders (figure 9).

Public Comment

The formal public comment period lasted for 60 days over the summer of 2009, although we continued to solicit and incorporate feedback after this period closed. During the public comment phase of the project, the website was visited 886 times by visitors from 17 countries. The majority

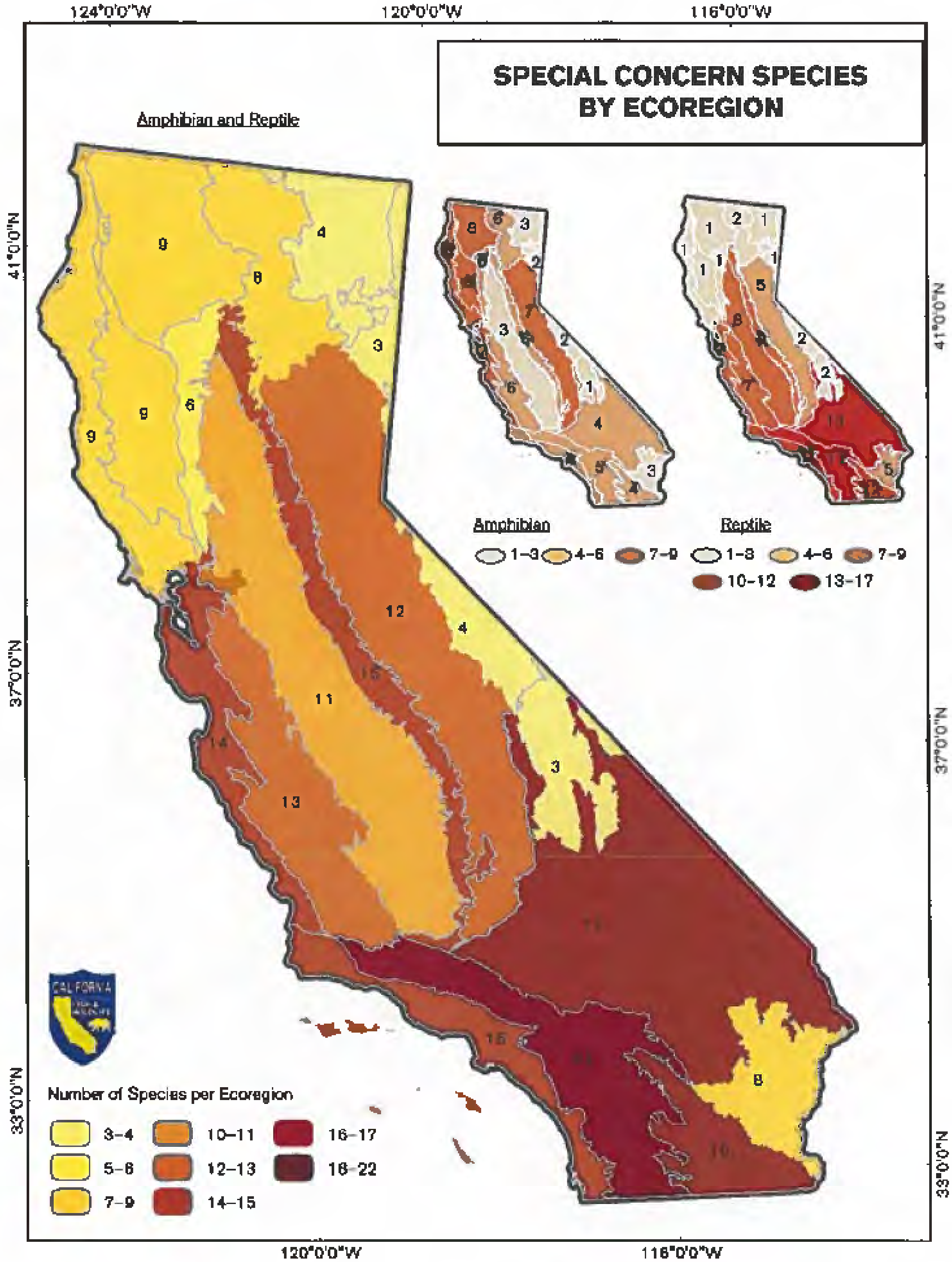


FIGURE 8 The number of Species of Special Concern that occurs within each US Department of Agriculture (USDA) Ecoregion section.

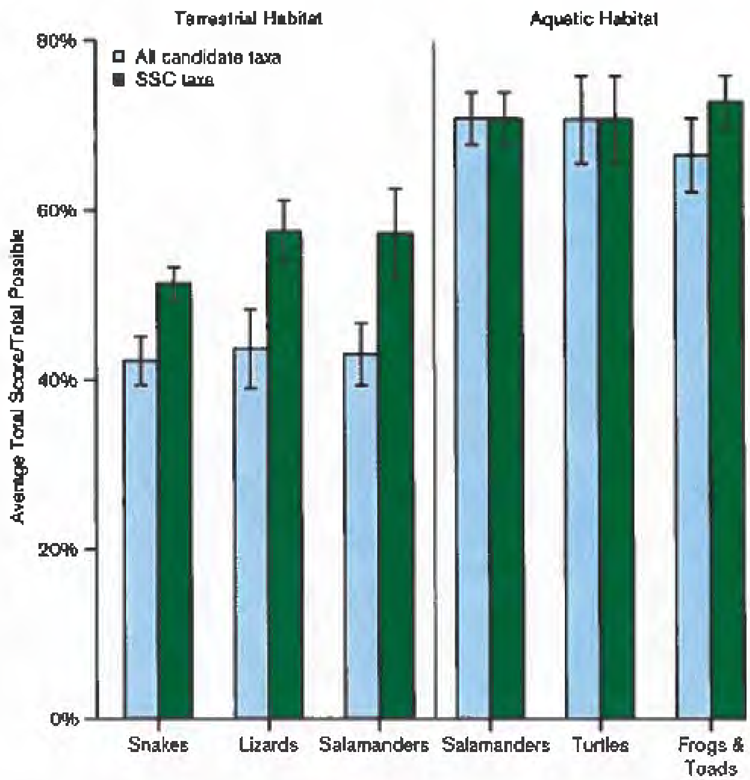


FIGURE 9 Average Total Score/Total Possible by aquatic or terrestrial habitat type. Filled bars are averages across the Species of Special Concern (SSC) taxa. Open bars are averages across all 73 nominee taxa. Error bars are standard errors.

of visitors (575) were from California, followed by visitors from neighboring states (Washington: 32; Oregon: 28; Arizona: 26). We received feedback from a wide variety of conservation professionals, academics, and enthusiasts. Because much of this feedback came from informal conversations on the telephone or at workshops, meetings, and conferences, we cannot precisely quantify the number of data contributors to this project. However, we received substantial contributions in the form of unpublished data, reprints, field notes, and/or localities during the public comment period from approximately 45 individuals (see Acknowledgments).

DISCUSSION

Risk Metrics

Overall, the metrics performed well, successfully identifying taxa that herpetologists generally consider to be at risk across the state, such as ranid frogs. Similarly, scores for the Species of Special Concern that are federally listed suggested that the metrics were performing well. Evaluating all taxa within a metric framework also facilitated identification of patterns among the metric scores that revealed insights into the geographic and ecological factors associated with declines. As emphasized by Shuford and Gardali (2008) for birds, no single set of metrics can capture the intricacies of the natural world fully. The strengths of our approach were that the eight metrics covered a wide range of factors that indicate declines and established a repeatable and transparent baseline for the evaluation of Species of Special Concern. During the initial public input phase of the project, we observed firsthand how a metric-based framework facilitated incorporation of feedback into conservation decisions, regardless of disagreements over the particular metrics used.

That is, when disagreements arose, the metrics allowed us to discuss conflicting scores for individual taxa, focusing discussions on specific issues and questions.

Our metrics covered four basic categories that spanned the diversity of conservation issues faced by any species: geography of declines, changes in population biology over time, key ecological attributes associated with risk, and estimates of future impacts. Metric scores within these categories were often correlated, capturing real patterns in how declines occur. For example, the high correlation between distribution trend and population trend reflects the fact that populations tend to become smaller and smaller as they become isolated and fragmented over time. This general shrinking of populations for many taxa with naturally extensive metapopulations will lead to a high score for population trend. However, as this trend continues over time, those isolated, declining populations experience much greater demographic stochasticity (Lande 1988), leading to more frequent extirpations of local populations and thus high scores for distribution trends. Thus, although these two metrics could be decoupled in principle, our assessments indicate that they tend to be associated in natural systems, and the metrics reflect this association rather than a redundancy in the approach. They also highlight the importance of measuring population connectivity as a research goal and of maintaining or reestablishing it as a management objective.

The correlation among metric scores may help explain why the rankings were robust to data deficiencies. This feature of the rankings is critical when evaluating reptile and amphibian taxa that can be cryptic, rare, and for which survey data are often lacking. We ranked taxa using the ratio of the total score to the total possible, rather than just the total score, to account for the different possible total scores for each species arising from data deficiencies. An implication of this approach is that each species' score is based on the data available and that the metrics differentially influenced scores depending on data availability. For example, population size is difficult to estimate with precision and generally requires extensive multiyear field studies. As a result, we could not score population trend for eight Species of Special Concern. If such data deficiencies were biasing our results, then this would be reflected in a different distribution of Priority 1, 2, 3 and Undefined scores for data-deficient taxa compared to the overall set of Species of Special Concern, but this was not the case ($\chi^2 = 5.4$, df $= 3$, $p = 0.14$). We acknowledge that data deficiencies in key metrics, such as distribution and/or population trend, could allow for taxa to achieve high Total Score/Total Possible ratios based on having only moderate scores for the remaining metrics. Although this was rarely an issue in our analyses, we also believe that this captures a realistic axis of risk. Taxa that have life histories indicating some amount of risk, particularly small range size and high ecological specialization, but for which we have no data on trends in abundance or distribution, are prime candidates both for unnoticed declines and for further research or monitoring. By scoring them as data deficient and basing their overall score only on available data, we explicitly upweight the importance of those metrics for which we do have information, appropriately bringing them to the attention of biologists and resource managers.

The metric scores were informative for broadly categorizing risk, with generally accepted high-risk taxa receiving the highest scores (e.g., arroyo toad, *Bufo californicus*) and clearly low-risk taxa receiving the lowest scores (e.g., northwestern garter snake, *Thamnophis ordinoides*). If a few strongly correlated risk metrics were uniformly high for at-risk taxa, this could have produced a sharp break point in overall score for Special Concern taxa, but this was not the case. Instead, the risk metric scores formed a smooth continuum from very low to extremely high Total Score/Possible Score values, indicating that a wide variety of combinations of metric scores characterized different taxa (figure 3). This smooth continuum in scores made it difficult to use metric scores alone to decide on special concern status, particularly for the lower-ranking taxa. It also forced us to focus on the specific biology of taxa with lower metric scores in evaluating whether they should or should not be Species of Special Concern. For example, the yellow-blotched ensatina (*Ensatina eschscholtzii croceater*) has much of its small range on private land, and concerns regarding the management and development of that land was a primary motivation for its previous designation as a Species of Special Concern (Jennings and Hayes 1994a). However, more recent planning efforts have emphasized the importance of retaining much of the yellow-blotched ensatina's habitat as unfragmented space (e.g., Tejon Ranch Conservancy 2008). This shift to regional conservation planning addressed the concerns about habitat loss for this species as described in the previous amphibian and reptile Species of Special Concern document (Jennings and Hayes 1994a), so we placed it on the Watch List. However, we identified the sandstone night lizard (*Xantusia gracilis*), which has a lower metric score, as a Species of Special Concern because of its tiny range size and associated potential for extinction.

The same was generally true for assigning priority rankings to individual taxa. Once again, there are no clear cut-offs in ranking scores among Species of Special Concern taxa in figure 3, making the identification of unambiguous criteria for priority score difficult. If the correlation between ranking and priority were perfect (or if we defined priority based solely on ranking), then all Priority 1 (green) taxa would be at the bottom of figure 3, Priority 2 (yellow) would be next, Priority 3 (peach) next, followed by taxa with No Status (white) at the top of the figure. This is close to, but not identical with, our priority ranking scheme.

We could have simply imposed priority-level cut-offs using the metric scores themselves rather than trying to add information that goes beyond a ranking based entirely on metrics. We did not do so because we felt that this would amount to a statement that all relevant biological information for each species was captured in the metric data. For example, the red-bellied newt (*Taricha rivularis*) ranked in the top 20% of taxa but is considered a Priority 2 Species of Special Concern. This decision was made because the ecological and population size data for this taxon are limited in scope, such that it was not possible to conclude that severe future declines and/or extirpation are likely without immediate conservation actions. Overall, we view the metrics as a useful but necessarily approximate guide for informing conservation decisions, not a complete replacement for careful consideration of the biology of each taxon on a case-by-case basis.

Taxonomic Patterns in Metric Scores

Taxonomic patterns among the Species of Special Concern can be measured as the total number of taxa, the fraction of the total number of species in the state that are Species of Special Concern, or as the average numerical metric score (Total Score/Total Possible) for different taxonomic groups. Each is informative, and together they provide a more complete overall picture of the status of the amphibian and reptile fauna of California than does any single measure.

When viewed in the context of all 217 taxa that are known to naturally occur in California (Appendix 1), turtles and amphibians are the most at-risk taxonomic groups. Among the candidate taxa, turtles and frogs had similar average metric scores (71% and 67%, respectively; figure 7), and many of these taxa are Species of Special Concern. All of California's nonmarine turtles are at risk at the Species of Special Concern or State Threatened level (figure 10). This pattern mimics the situation for turtles and tortoises globally; according to the IUCN, turtles have the highest fraction of Red List taxa among any major group (39% of all species and 62% of the currently evaluated species; Rhodin et al. 2010). While very few turtle species occur in the state, half of California's frogs and toads

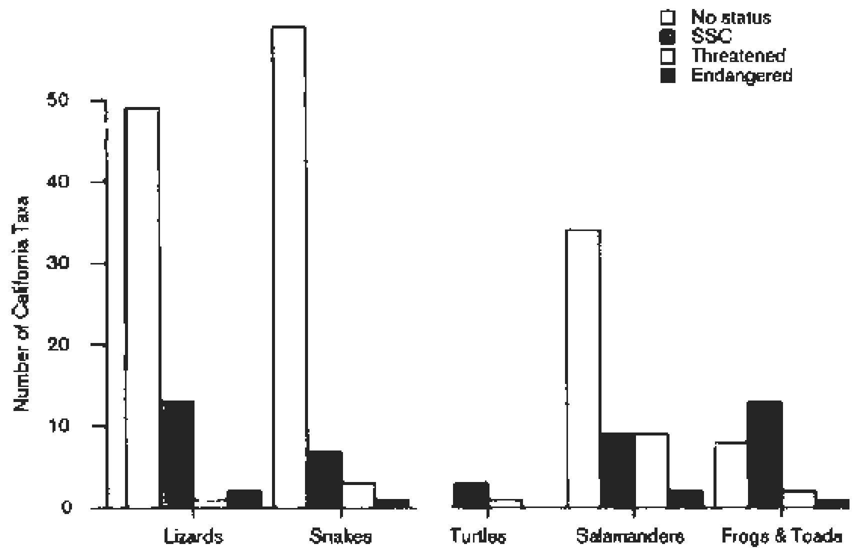


FIGURE 10 Percent of California reptile and amphibian taxa ($n = 217$ by state protected status: Endangered, Threatened, Species of Special Concern [SSC], No Status).

are included as Species of Special Concern. The state's other amphibian group, salamanders, has the next highest fraction of included taxa, with squamates (lizards and snakes) being least at risk at the state level (figure 10). These patterns are consistent with global concerns about amphibian declines in recent decades (Lannoo 2005). No frogs or toads were included in the additional taxa in need of research and monitoring category (Appendix 4), which confirms that a disproportionately large research effort has focused on this globally imperiled group compared to other taxa (Stuart et al. 2004).

Ecological Patterns in Metric Scores

Although taxa can be categorized along a variety of ecological axes, one clear distinction is between aquatic and terrestrial primary habitat requirements. The most striking overall pattern is the higher Total Score/Total Possible scores for aquatic (all frogs and toads, aquatic salamanders, turtles) compared to terrestrial (terrestrial salamanders, lizards and snakes) taxa. Jennings and Hayes (1994a) suggested that taxa having aquatic life stages were more extinction prone than terrestrial taxa, and our analysis supports this conclusion. However, phylogenetic and ecological patterns are confounded in this analysis because all frogs and turtles that we scored are also aquatic and all of the lizards and snakes were terrestrial. Thus, it is not clear whether frogs, toads, and turtles as taxonomic groups are at risk or whether obligatorily aquatic taxa are at risk. Salamanders provide some insight into this issue, as both aquatic and terrestrial taxa occur in California. The Total Score/Total Possible metric scores for Species of Special Conceru in these two groups are strikingly different (terrestrial salamanders 57%, aquatic salamanders 71%) and consistent with the interpretation that aquatic taxa are, on average, at greater risk than terrestrial ones. Even within salamanders, however, phylogeny is still a confounding variable because all salamanders in the family Plethodontidae are terrestrial, whereas all of the other California salamanders are aquatic. While the overall pattern of higher scores for aquatic taxa is clear, it is not possible to infer causality from this analysis.

Concluding Thoughts on Metric Score Patterns

Two general conclusions emerge from our analyses of metric scores across taxa. First, regardless of whether the pattern is driven by evolutionary relatedness or some intrinsic feature of aquatic ecosystems, aquatic species are at greater risk than terrestrial ones. Second, amphibians overall are at greater risk than reptiles. Both of these conclusions may stem from the ecology of aquatic and terrestrial taxa, particularly in the relatively arid landscape that dominates much of California. Although amphibians have been characterized in the past as harbingers of habitat deterioration due to their permeable skin and sensitivity to environmental chemicals, recent work suggests that this may be less of a general conclusion than was previously thought (Kerby et al. 2010). However, what is clear is that water is a limiting resource over most of California, and climate change predictions for the next 50–100 years indicate that this limitation will only increase in the future. Aquatic habitats in California have also been particularly negatively impacted by nonnative fish, amphibian, and invertebrate introductions (see discussion below), and managing and preventing future introductions is a major challenge to conserving aquatic habitats. Aquatic invasive predators, combined with water modification and overutilization, have led taxa that rely on water, be it a mountain stream or vernal pool, to more precipitous declines than purely terrestrial taxa.

The fact that aquatic taxa are more at risk does not, however, indicate that terrestrial taxa are uniformly secure, now or in the future. The greatest biodiversity hot spot for terrestrial lizards and snakes in the state is in southern California (Parisi 2003; figure 8). Much of this region has experienced heavy development which has lead to major conservation concerns. Coastal taxa that are diurnally active and highly mobile (e.g., coast patch-nosed snake, *Salvadora hexalepis virgultea*; coastal whiptail, *Aspidoscelis tigris stejnegeri*) are particularly at risk, in part because habitat fragmentation and heavy road traffic, interactions with humans, their commensals (e.g., raccoons, skunks, rats, crows), and pets (dogs and cats), as well as general problems with fragmented habitat and a loss of metapopulation dynamics. In addition, some of the greatest areas of urban growth in California are in the relatively sparsely populated inland xeric regions, where remote conditions and lack of easily developed water and infrastructure have thus far protected many species. As these regions become more heavily populated and more fragmented by roads and urban centers, we predict a shift in endangerment patterns over the next several decades.

To help avoid future population declines, listings, and extinctions, amphibian and reptile Species of Special Concern are sometimes considered in both urban and large-scale planning efforts. Large-scale efforts originate at both the state (Natural Community Conservation Plan [NCCP]) and federal (Habitat Conservation Plan [HCP]) levels and involve cooperation between the two jurisdictions and other public and private partners. For example, five amphibian or reptile Species of Special Concern are included in the heavily populated planning area covered by the San Diego Multiple Species Habitat Conservation Plan (http://www.wildlife.ca.gov/Conservation/Planning/NCCP/Plans/San-Diego-MSCP). As of December 2013, nine approved NCCPs were being implemented, some of which include amphibian and reptile taxa, and 16 NCCPs were in the planning phase. Of the nine plans undergoing implementation, 1.5 million acres (0.6 million hectares) have been committed to reserve lands. The total planning area for the 25 NCCPs covers over 33 million acres (13.3 million hectares) (http://www.dfg.ca.gov/habcon/nccp/). As of 25 June 2014, there are 147 approved Federal HCPs in California (http://ecos.fws.gov/conserv_plans/). HCPs are primarily focused on federally listed species, so any benefit to ARSSC taxa is typically incidental to the plan.

Other large-scale wildlife planning efforts include a statewide assessment of essential habitat connectivity sponsored by the CDFW and the California Department of Transportation. The effort identified large remaining blocks of intact habitat or natural landscape and linkages between them that need to be maintained, particularly as corridors for wildlife (http://www.dfg.ca.gov/habcon/connectivity/).

Peripheral Populations and Endemic Taxa

At least 10 of the 45 Species of Special Concern are best considered peripheral in California. For these species, the bulk of their range occurs outside of the state, where they may be abundant and in little danger (e.g., Couch's spadefoot, *Scaphiopus couchii*), of relatively uncertain status (e.g., regal ring-necked snake, *Diadophis punctatus regalis*), or declining and protected (e.g., Oregon spotted frog, *Rana pretiosa*). Particularly for those taxa that are common rangewide, a reasonable question to ask is whether they should be protected in California, where they may occur in marginal habitat at the edge of their ranges. From a biological perspective, conditions beyond the state's borders are clearly relevant to range-wide conservation risk. However, from a political and jurisdictional perspective, managing populations outside of California is not the state's responsibility. We consider peripheral taxa as valid Species of Special Concern because the CDFW's mission is to "maintain native fish, wildlife, plant species and natural communities for their intrinsic and ecological value and their benefits to people [...] include[ing] habitat protection and maintenance in a sufficient amount and quality to ensure the survival of all species and natural communities" that naturally occur in California (http://www.dfg.ca.gov/about). Therefore, peripheral populations are similar to taxa whose entire range occurs within the state in that they are established, natural components of the biodiversity of California; whether they require special conservation measures should be based on their current status in the state. Two of our metrics, range size and endemism, take the peripheral nature of populations into account, at least indirectly. Range size generally upweights these populations, since they have small ranges within the state. Countering this, endemism measures the fraction of the species' overall range that occurs in California, which tends to downweight such taxa. Each had a maximum score of 10, so they had equal impacts in the total score for each taxon.

Endemic taxa, by contrast, are clearly one of the state's most important conservation responsibilities (Table 4). Because they occur nowhere else, these taxa make up a critical component of California's unique amphibian and reptile fauna, so conservation successes or failures within the state are likely to have much larger impacts on these species than taxa that range more widely.

Geographic Patterns in Species of Special Concern

Range maps are an important resource in delimiting changes in the distribution of taxa. However, range is also difficult to determine precisely for many reptiles and amphibians due to their naturally low population densities, cryptic natural history, and the paucity of survey data. In constructing these range maps, we included, rather than excluded, regions where the likelihood of occurrence was high but no specimens have been documented to date. Our reasons for doing so were twofold. First, by setting boundaries that may be too large, we hope to encourage field researchers to expand their geographical horizons when searching for new localities. Second, since the taxa are at-risk, we want to err on the side of potential habitat inclusion for conservation purposes. We used previously established units (watershed boundaries, ecoregions, etc.) rather than arbitrary polygons around localities to provide objective boundaries from which future surveys can work. For instance, where we drew a species as being present in one watershed but absent in the next, this provides a very straightforward way to focus additional surveys. Surveyors can ask the question, "Is the taxon present in the adjacent watershed?," and focused efforts can answer that question, refining range boundaries in an organized, efficient manner.

These maps also highlight an important, frequently overlooked point: we need a mecha-

TABLE 4
Endemic and Near Endemic Species of Special Concern

Endemic	
Aneides flavipunctatus niger	Santa Cruz black salamander
Batrachoseps campi	Inyo Mountains salamander
Batrachoseps minor	Lesser slender salamander
Batrachoseps relictus	Relictual slender salamander
Bufo canorus	Yosemite toad
Dicamptodon ensatus	California giant salamander
Elgaria panamintina	Panamint alligator lizard
Masticophis flagellum ruddocki	San Joaquin coachwhip
Taricha rivularis	Red-bellied newt
Taricha torosa, Southern populations	Coast Range newt
Thamnophis sirtalis, Southern populations	Common garter snake
Xantusia gracilis	Sandstone night lizard
Xantusia vigilis sierrae	Sierra night lizard
Near endemic	
Aniella pulchra	California legless lizard
Bufo californicus	Arroyo toad
Emys marmorata marmorata	Northern western pond turtle
Emys marmorata pallida	Southern western pond turtle
Phrynosoma blainvillii	Coast horned lizard
Rana boylii	Foothill yellow-legged frog
Rana draytonii	California red-legged frog
Spea hammondii	Western spadefoot
Uma scoparia	Mojave fringe-toed lizard

nism, including a curated database, that tracks docnmented absence as well as documented presence data. Documenting, and even defining, absence is often a very difficult problem, but these efforts can be helped by collating survey results (including both positive and negative occurrence data) into a publically available and easily accessible format. Locality data from the past couple of decades tend to come from sight records, survey data, and other field research that does not result in the collection of museum specimens (figure 11). While museums are increasingly making their data accessible through online databases, there is currently no centralized way to collate locality data from other sources across all California reptiles and amphibians. The California Natural Diversity Database is an important means by which the state collates status and location information for Species of Special Concern and those listed under the federal and California Endangered Species Acts. Currently, this resource does not document absence data for sites where only negative surveys have occurred and focuses solely on those taxa on California's Special Animals list. Expanding the scope of this

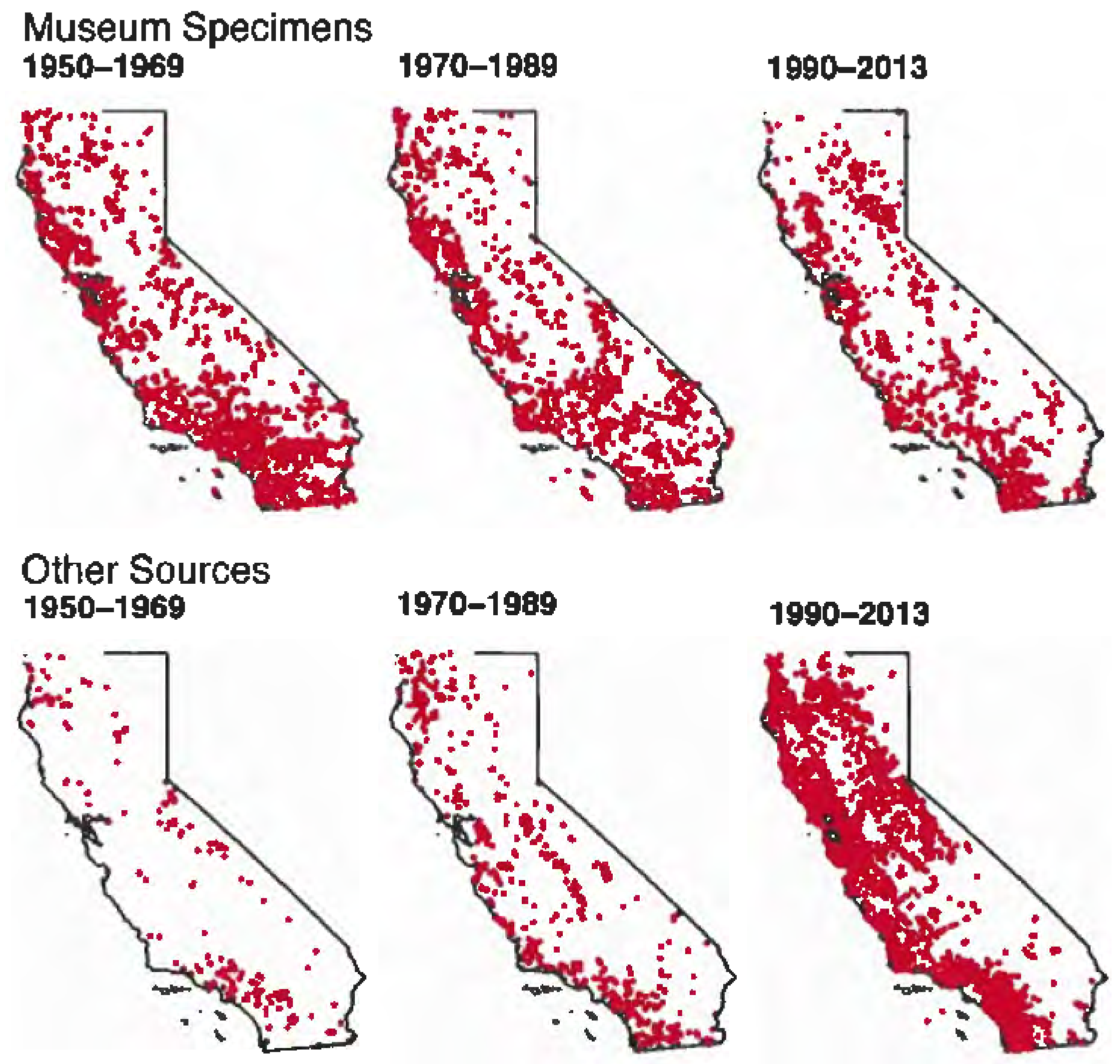


FIGURE 11 Distribution of Species of Special Concern locality records over time. Data from other sources include records from the California Natural Diversity Database and the Biogeographic Information and Observation System, both of which contain some museum records, though the majority of records plotted are from survey data.

database or adding an additional database to capture negative occurrence data, as well as survey data from other taxa, would help the state's efforts to improve estimates of species ranges.

When we plot the number of at-risk species contained within each ecoregion, geographic patterns in conservation risk emerge (figure 8). The southern California coast and mountains and the Mojave Desert have the largest number of at-risk species overall, although this pattern is due largely to trends among reptiles. This important area of conservation risk is driven along the south coast by habitat loss and degradation arising from the massive land use changes that this area has experienced over the last century. The Mojave Desert, conversely, is often viewed as being less disturbed and protected by reserves, parks, and military reservations. Our analysis highlights that this is not entirely true. The Mojave Desert has experienced some degree of habitat degradation and loss, although, to date, not as strongly as that which has occurred along the coast where extensive urbanization has effectively removed large areas of habitat. However, the Mojave

Desert, as well as the Great Basin, Colorado, and Sonoran Deserts, and some of the southern Sierra Nevada and associated foothills constitute the 22.5 million acre planning area for future renewable energy development (wind, solar) in southern California. In addition, many of the at-risk species in the Mojave Desert use specialized and rare resources that have experienced a disproportionate amount of habitat degradation relative to other areas of the desert. For example, the fringe-toed lizards of the genus *Uma* exclusively use sand dune habitats, which also disproportionately attract off-highway vehicle use even in some protected areas (see species accounts for additional details). The Mojave Desert is also home to a large number of narrowly distributed or rare taxa that may exist at the edge of their physiological tolerance and persist in small, often isolated areas (e.g., Gila monster, *Heloderma suspectum*). These species may be at particular risk of further declines as climate change occurs. Importantly, it is not the case that all desert species are declining equally, since the Great Basin and Sonoran ecoregions have relatively few at-risk reptiles, while an intermediate number occur in the Colorado Desert.

For amphibians, the areas of largest conservation risk are the mountainous areas surrounding the Central Valley and the forested regions of central and northern California (figure 8). These areas have not experienced massive land use change per se, although they have experienced considerable habitat fragmentation and modification stemming from water diversions, timber harvest, and nonnative species (Bunn et al. 2007, http://www.wildlife.ca.gov/SWAP). Some studies indicate that agriculture in the Central Valley has had an impact on some species in the Sierra Nevada and Cascades Range via increased exposure to pesticide drift from the Central Valley (e.g., Davidson et al. 2002, Davidson 2004, Lind 2005). In addition, many of these regions are heavily exploited for timber harvest, and this has also had an impact on both stream-dwelling and terrestrial amphibians (e.g., Olson et al. 2007, Welsh and Hodgson 2008). An emerging threat in northern California is marijuana cultivation, which can degrade both terrestrial and aquatic amphibian habitat (CDFW 2013). Increased sedimentation, dewatering of headwater streams, and application of agricultural chemicals are all potential negative effects of marijuana growing, and these effects should be monitored and potentially regulated. High elevation mountainous areas are expected to experience large impacts from climate change through the altered timing and amount of snowmelt (Cayan et al. 2008b), and this future risk probably affects amphibians to a greater extent than co-occurring reptiles (figure 8). Increasing temperatures associated with climate change may also lead to phenological shifts in several species, which could interact with several of the existing threats (Todd et al. 2011). This pattern in both amphibians and reptiles is driven to some extent by species richness of the respective groups. Southern California and the deserts have the highest richness of reptile diversity, whereas the Sierra Nevada and northern Coast Ranges are home to greater amphibian species richness (Parisi 2003, Stebbins 2003).

Finally, for all taxa we note that the distribution of locality data is uneven and patchy across the state (figure 12). At first glance, it appears that the areas with the greatest human impacts and populations (southern coastal California, the Bay Area) are also the areas with the greatest number of locality records, and it may be that these are simply the areas that have received the greatest efforts from field biologists. Unfortunately, we cannot unambiguously say whether the sparse locality records, for example, from the Mojave Desert reflect sparse fieldwork, underreporting of data, or a genuine low density of animals in the region. Our sense is that all of these factors are contributing to the distribution of locality records. That is, it is almost certainly the case that there has been much more intensive sampling effort, and consequently a larger number of records, in San Diego County than in the eastern Mojave

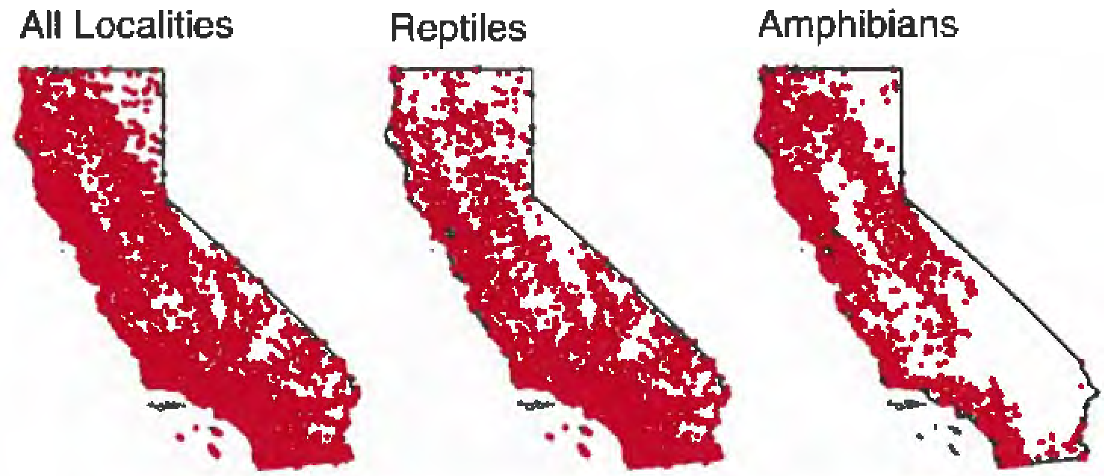


FIGURE 12 Occurrence of Species of Special Concern taxa locality records throughout the state. Regions with few occurrence records may represent areas with few SSC taxa, low sampling effort, or both.

(figure 11). However, it is also true that both reptiles and amphibians are sparsely represented in the eastern Mojave (compare reptile and amphibian maps in figure 12), even though this is a region of high abundance and species richness for reptiles. However, the large number of sensitive species (figure 8) and the recent, intensive development in San Diego County cause many environmental impact assessments to be undertaken under the California Environmental Quality Act, and this has likely contributed to the larger number of records compared to the deserts of southern California.

Differences between This Document and Jennings and Hayes (1994a)

Species priority assessments for conservation purposes are subject to revision over time as factors that affect risk, including habitat protection, invasive species, and scientific knowledge change. Although the number of species identified as being of concern was similar (49 vs. 45), a number of differences exist between the current and previous assessments. Jennings and Hayes (1994a) based their assessment on a combination of their own knowledge and that of a large group of leading experts on individual species; we follow a similar procedure here but summarize the available data using a metric-based approach. Jennings and Hayes (1994a, p.183) felt that for no species of amphibian or reptile was there compelling evidence to "downgrade" status from more threatened to less threatened, whereas we removed several taxa from the Species of Special Concern designation. In total, 34 taxa occur on both lists; we added 11 taxa that were not included by Jennings and Hayes (1994a) and excluded 15 taxa that were previously included (Table 5, Appendix 3).

The status of 43% (26/60) of Species of Special Concern taxa has changed between 1994 and the present. Of the 26 species that changed status, approximately half were upgraded and half were downgraded: 58% (15/26) of the taxa were on the previous list but not the current one, and 42% (11/26) were upgraded from having no formal status to Species of Special Concern (Table 5). These changes reflect differences in approach between these two compilations, insights gained from an additional 20 years of field and systematic research, and real changes that have occurred in the abundance of species. However, on face value, it appears that the past two decades have not been a completely negative period for amphibian and reptile biodiversity in California.

Several factors contribute to these changes. In Table 5, we broadly categorized reasons for changes into three categories. "Listing status"

TABLE 5

Comparison of Species of Special Concern between this publication and Jennings and Hayes (1994a)

Gray cells denote species designated by both publications (see text for additional details)

Taxon	Jennings and Hayes	Thomson et al.	Reason
Ambystoma californiense	X		Listing status
Ambystoma macrodactylum sigillatum		X	New data
Aneides flavipunctatus niger		X	New data
Aniella pulchra	X	X	
Arizona elegans occidentalis		X	New data
Ascaphus truei	X	X	
Aspidoscelis hyperythra[1]	X		New data
Aspidoscelis tigris stejnegeri		X	New data
Batrachoseps sp. "*Breckenridge*"[2]	X	X	
Batrachoseps campi	X	X	
Batrachoseps minor		X	Taxonomy
Batrachoseps relictus	X	X	
Bogertophis rosaliae[3]	X		New data
Bufo alvarius	X	X	
Bufo californicus[4]	X	X	
Bufo canorus	X	X	
Coleonyx variegatus abbotti		X	New data
Crotalus ruber	X	X	
Diadophis punctatus regalis		X	New data
Dicamptodon ensatus		X	New data
Elgaria panamintina	X	X	
Emys marmorata marmorata[5]	X	X	
Emys marmorata pallida[6]	X	X	
Ensatina eschscholtzii croceater	X		New data
Ensatina eschscholtzii klauberi	X		New data
Gambelia copeii		X	New data
Heloderma suspectum	X	X	
Hydromantes platycephalus	X		New data
Hydromantes sp. "Owens Valley"	X		Taxonomy
Kinosternon sonoriense	X	X	
Lampropeltis zonata parvirubra	X		New data
Lampropeltis zonata pulchra	X		New data
Masticophis flagellum ruddocki	X	X	
Masticophis fuliginosus		X	Taxonomy
Phrynosoma blainvillii[7]	X	X	
Phrynosoma mcallii	X	X	

(continued)

TABLE 5 *(continued)*

Taxon	Jennings and Hayes	Thomson et al.	Reason
Pituophis catenifer pumilis[8]	X		New data
Plestiodon skiltonianus interparietalis[9]	X		New data
Plethodon elongatus	X		New data
Rana aurora	X	X	
Rana boylii	X	X	
Rana cascadae	X	X	
Rana draytonii[10]	X	X	
Rana muscosa	X		Listing status
Rana pipiens	X	X	
Rana pretiosa	X	X	
Rana sierrae[11]	X		Listing status
Rana yavapaiensis	X	X	
Rhyacotriton variegatus	X	X	
Salvadora hexalepis virgultea	X	X	
Scaphiopus couchii	X	X	
Spea hammondii[12]	X	X	
Taricha rivularis		X	New data
Taricha torosa (Southern populations)	X	X	
Thamnophis hammondii	X	X	
Thamnophis sirtalis ssp.	X	X	
Uma notata	X	X	
Uma scoparia	X	X	
Xantusia gracilis	X	X	
Xantusia vigilis sierrae	X	X	

1. Evaluated under the name *Cnemidophorus hyperythrus beldingi* in Jennings and Hayes (1994a).
2. Now included within *Batrachoseps relictus*.
3. Evaluated under the name *Elaphe rosaliae* in Jennings and Hayes (1994a).
4. Evaluated under the name *Bufo microscaphus californicus* in Jennings and Hayes (1994a).
5. Evaluated as a single species, *Clemmys marmorata*, in Jennings and Hayes (1994a).
6. Evaluated as a single species, *Clemmys marmorata*, in Jennings and Hayes (1994a).
7. Evaluated as two subspecies, *Phrynosoma coronatum blainvillii* and *Phrynosoma coronatum frontale* in Jennings and Hayes (1994a).
8. Evaluated under the name *Pituophis melanoleucus pumilis* in Jennings and Hayes (1994a).
9. Evaluated under the name *Eumeces skiltonianus interparietalis* in Jennings and Hayes (1994a).
10. Evaluated under the name *Rana aurora draytonii* in Jennings and Hayes (1994a).
11. Evaluated as part of *Rana muscosa* in Jennings and Hayes (1994a).
12. Evaluated under the name *Scaphiopus hammondii* in Jennings and Hayes (1994a).

applies to a few taxa, like the California tiger salamander (*Ambystoma californiense*), that are no longer considered Species of Special Concern because they were listed under the California Endangered Species Act between 1994 and 2014. These taxa are still considered to be at risk, but their state listing precludes inclusion as a Species of Special Concern. "Taxonomy" is more difficult to categorize because many taxa have had name changes between the two lists.

However, in Table 5 we highlight taxonomic changes that led to either the recognition of a new at-risk taxon or the elimination of a previously recognized taxon that is no longer considered valid. An example of the former is the Baja California coachwhip (*Masticophis fuliginosus*), which was considered a part of the widespread and relatively common coachwhip (*M. flagellum*) in 1994, but has since become more widely recognized as a distinct species (Grismer 2002). We note taxonomic changes in Table 5 that did not impact special concern status, like the elevation of the arroyo southwestern toad (*Bufo microscaphus californicus*) (Jennings and Hayes 1994a) to the arroyo toad (*B. californicus*) (current document) as footnotes. The remaining taxa changed special concern status because of new data. This category covers a variety of factors, ranging from better and more extensive field survey data which has revised our understanding of the severity of threats (e.g., the Mount Lyell salamander, *Hydromantes platycephalus*) to new threats that have been identified since 1994 (e.g., predation by introduced fishes for the southern long-toed salamander, *Ambystoma macrodactylum sigillatum*). Some of the difference in threat evaluation stems from our choice of metrics. For example, climate change is currently a particularly important aspect of conservation risk that was not previously considered. In some cases, the availability of suitable habitat has changed, either positively or negatively. Habitat may be set aside for conservation (e.g., Tejon Ranch appears to be setting aside considerable land that will benefit the yellow-blotched ensatina, *Ensatina eschscholtzii croceater*) but is usually lost (e.g., coastal scrub habitat for the California glossy snake, *Arizona elegans occidentalis*). Finally, we note that the factors listed in Table 5 are an over-simplification of the reasons behind our decisions. An explanation for each of the 15 taxa that appeared on the previous list but not on the new list is also included in Appendix 3.

Ultimately, the comparison of the two Species of Special Concern documents emphasizes what can be learned by periodically updating and evaluating the conservation status of taxa on a regular basis. For the 34 taxa that have remained Species of Special Concern, we can and should ask what more can be done to improve their status. Some of the taxa that are no longer Species of Special Concern may inform the kinds of positive changes that can be brought about by management, research, or both. For example, additional surveys and taxonomic research on the Mount Lyell salamander (*H. platycephalus*) have shown that the species is more widespread than previously thought and clarified the taxonomic status of populations in Owens Valley, which were previously suspected of being distinct and of conservation concern. Finally, the challenges of incompletely known taxonomy that were emphasized by Jennings and Hayes (1994a) still pose a major challenge to effective management; if we do not have a complete catalogue of the taxa that occur in California, we cannot even enumerate what may need protection to maintain biodiversity.

Management Recommendations for California Amphibians and Reptiles

While effective management of the Species of Special Concern will generally require development of specific management strategies tailored to the biology of individual taxa, several general recommendations have emerged from this document.

1. Protect aquatic habitats. The metric scores indicate that aquatic species are at greater risk than terrestrial ones, suggesting that remaining aquatic habitats with native amphibian and turtle populations should be high conservation priorities. California's aquatic habitats have been highly modified from a faunal perspective. As of 2002, there were 51 nonnative freshwater fishes in California, the majority of which were deliberately introduced to enhance recreational fisheries (Moyle 2002). Nonnative fishes now predominate in many California waterways, raising concerns about increased competition, predation, habitat interference, disease, and hybridization with native species (CDFG 2008). A large body of ecological research has demonstrated a negative effect of introduced fishes and bullfrogs (*Rana catesbeiana*) on California's native anurans (e.g., Hayes and Jennings 1986, Tyler et al. 1998, Knapp and Matthews 2000, Vredenburg 2004, Knapp 2005, Leyse 2005, Welsh et al. 2006, Pope et al. 2008). As a result, predatory salmonids, centrarchids, catfishes, and other nonnative species should be eradicated wherever feasible and should not be introduced into remaining native amphibian or reptile habitat. Maintaining appropriate water flow regimes for stream-dwelling taxa is also critical, as are broad riparian buffers to maintain lotic habitats and reduce siltation (e.g., Lind et al. 1996, Yarnell 2005, Hancock 2009).

Specific management recommendations include the following:

- Control, or eliminate where possible, invasive aquatic species, particularly predatory fishes, crayfish, and bullfrogs. For widespread, established invasives, plans should be developed with actions that reflect those identified in the California Aquatic Invasive Species Management Plan (CDFG 2008). For bullfrogs in particular, plan Objectives 5 and 6 apply: Education and Outreach and Long-Term Control and Management. Invasive species in the early stages of colonization (e.g., *Nerodia fasciata*, *N. sipedon* and *N. rhombifer*) should be eradicated as soon as possible to prevent further spread. Known to be present in California since the 1990s, coordinated efforts have yet to effectively coalesce to make significant progress toward eradicating *Nerodia*, though educational (http://biology.unm.edu/mmfuller/WebDocs/HTMLfiles/nerodia.html) and occasional agency efforts occur.
- Eliminate, limit, or mitigate effects of dams, water diversions, and other hydrological disturbances to breeding streams whenever possible, and particularly during breeding seasons.
- When biologically appropriate, enhance connectivity and continuity of streams to allow free movement of aquatic species. Conversely, the potential for increasing connectivity to facilitate the spread of invasive species or disease should be considered on a species-by-species basis.
- Maintain riparian vegetation buffers and adjacent upland habitat.
- Eliminate roads within buffer zones and mitigate their effects in high-use amphibian migration areas whenever possible to avoid siltation and road mortality.
- Restrict use of heavy equipment on dirt roads and upland habitats, particularly during the breeding season when eggs and small larvae may be most affected by siltation.
- Maintain culverts under roads adjacent to breeding streams to reduce siltation.

2. *Protect integrity and connectivity of large terrestrial habitat patches.* The size of habitat patches necessary to support healthy populations of most species may be larger than previously recognized (Prugh et al. 2008). The amount and configuration of habitat clearly has a strong impact on the overall extirpation and recolonization dynamics of adjacent populations, and ultimately, of entire species. Besides the general conclusion that more intact habitat is always desirable, specific requirements will always need some level of study on a species-by-species basis. For example, ongoing work on the state and federally endangered California tiger salamander (*Ambystoma californiense*) suggests that this species routinely moves long distances (up to 2 km) away from breeding ponds, suggesting that the extent and quality of upland habitat is likely to have a strong impact on the species' long-term persistence (Trenham and Shaffer 2005, Searcy and Shaffer 2008, Searcy and Shaffer 2011). Several diurnally active and wide-ranging reptile species in southern California appear to be sensitive to habitat fragmentation and disappear from patches of small suit-

able habitat (e.g., coastal whiptail, *Aspidoscelis tigris stejnegeri*; coast patch-nosed snake, *Salvadora hexalepis virgultea*). Habitat fragmentation is a strong driver of declines for many species, and we recommend that land managers pay particular attention to preserving extensive habitat blocks where possible (see Mitrovich et al. 2009, for a well-worked example).

Although the individual conservation needs of species vary, formal conservation planning occurs on a broader scale that considers large areas of habitat for many species simultaneously. Because of many aspects of their shared biology, amphibians and reptiles are often considered as a group, and some excellent, general guidelines for their management have been developed (see, e.g., the Partners in Amphibian and Reptile Conservation habitat management guidelines http://www.parcplace.org/parcplace/publications/habitat-management-guidelines.html). In addition, the biology of amphibian and reptile species needs to be jointly considered within the framework of larger conservation initiatives. The California Natural Community Conservation Planning program is one such initiative that takes an area-wide approach to conservation planning, simultaneously considering conservation of many plant and animal species as well as potential land use activities (see Fish and Game Code Section 2800-2840). These broadscale, integrative approaches to conservation planning promise to be among the more effective strategies for achieving habitat protection and should become an increasingly central mechanism for conservation planning in California. Preserving linkages between adjacent habitat patches is also a key priority in these landscape-level conservation initiatives. Biologically, these linkages maintain metapopulation connectivity and habitat corridors that are often essential for long-term conservation. The California Essential Habitat Connectivity Project seeks to identify corridors between large remaining blocks of intact habitat and is one step in this direction (Spencer et al. 2010). Projects such as these are critically important for maintaining gene flow and migration among localized populations and should continue to be considered as landscape-level conservation initiatives move forward in the state.

Specific management recommendations include the following:

- All Species of Special Concern and the taxa discussed in Appendices 3 and 4 should be considered in Habitat Conservation Plans, Natural Community Conservation Plans, and other local and regional habitat management planning efforts.
- Develop species-specific ecological and landscape genetic datasets to determine the most important habitat corridors for protection and management of amphibian and reptile Species of Special Concern on specific landscapes.
- Identify and either eliminate or mitigate land uses that interrupt connectivity across habitat blocks that have been set aside for conservation. These might include roads, grazing, mining, timber harvest, and many other land uses and activities.

3. *Mitigate the effects of roads as a source of mortality and habitat fragmentation.* Roads have two primary effects: mortality and fragmentation (Fahrig et al. 1995, Gibbs and Shriver 2002, Mazerolle 2004, Gibbs and Shriver 2005; see also review in Andrews et al. 2008). The overall impact of road mortality on amphibian and reptile populations varies across road types, from species to species, geographically, temporally, and seasonally, and road-associated mortality levels interact with the movement patterns and seasonal migrations of individual taxa. In other parts of the country, roads have been documented to significantly contribute to fragmentation and reduced gene flow, interrupting normal metapopulation dynamics (Fahrig et al. 1995, Hels and Buchwald 2001, Langen et al. 2009, Clark et al. 2010, Sutherland et al. 2010), and the same presumably occurs in California. For example, surveys of 21 roads for migratiug, federally endangered

California tiger salamander (*A. californiense*) in Sonoma County suggest widespread mortality that has increased over time as traffic volume has increased. For surveys of one 1200-ft section of Stony Point Road conducted from 2001 to 2010, 160 of 262 salamanders (61%) found were road mortalities, suggesting that vehicular traffic is a substantial form of death in this extremely endangered species (D. Cook, unpublished data). Langen et al. (2009) identified predictors of hot spots of amphibian and reptile road mortality for use when planning roads or when conducting surveys on existing roads to locate priority areas for mitigation.

Although they have been employed infrequently in California, tunnels that assist amphibian and reptile movements can be an effective management tool that should be more actively investigated (for a comprehensive summary of published and unpublished literature, see Caltrans 2012). Two important aspects of migration tunnels are that they must have some capacity to funnel individuals into the tunnels (drift fences, concrete walls, or other similar structures), and they must be actively maintained. Without regular, scheduled maintenance, tunnels fill with debris, drift fences become covered with leaves, runoff soil, trash, and woody debris, and the tunnel quickly ceases to function. Tunnels may also play a role in the deserts of southern and eastern California, particularly as vehicular traffic increases, and roads fragment previously contiguous habitat. For additional recommendations regarding herpetofauna and roads, see Schmidt and Zumbach (2008).

Specific management recommendations include the following:

- Limit traffic, and consider road closures, during amphibian breeding migrations on sensitive public lands.
- Use signage (e.g., "Newt Crossing" warning signs) to warn vehicular traffic that they are in key migration areas.
- Develop standards for and install, maintain, and monitor usage of tunnels, underpasses or other passage mechanisms to reduce road-related mortality.
- Use various media resources for public education campaigns.

4. ***Translocate animals only when biologically appropriate.*** A general management strategy, variously referred to as relocation, repatriation, or translocation (Germano and Bishop 2009), is the practice of moving animals across landscapes, often from a site destined for development to a protected site. These efforts have become increasingly common as partial or complete mitigation for development projects that affect amphibians and reptiles. Several key biological issues need to be considered before animals are translocated. Disease transmission is an important problem that has had devastating consequences for several species (Jacobson 1993). The well-known upper respiratory tract infection that has decimated desert tortoise (*Gopherus agassizii*) populations is thought to be derived from released captive animals (Jacobson 1993). Genetic consequences of relocation programs should also be considered. Increasingly, genetic data are allowing researchers to elucidate fine-scaled genetic structure among populations, and the insights gained from nonlethal genetic sampling allow insight into biological parameters that are relevant for conservation including population subdivision, gene flow, migration corridors, and population sizes. However, the overall extent and functional consequence of this variation is still poorly understood for most organisms.

Moving individuals around the landscape has the potential for deleterious effects, either by diluting or eliminating unique historical lineages or by disrupting genetic variation that may be an important component of local adaptation. As emphasized in a recent review (Germano and Bishop 2009), homing and poor habitat quality are two of the primary reasons why translocation efforts may fail, and they should be carefully studied on a case-by-case basis. A recent document providing guidelines for translocations for the California tiger salamander (Shaffer et al. 2009) may serve as a model for some other taxa as well. It emphasizes that translocations should only be attempted into unoccupied habitat, and only after the threats that caused the initial declines have been effectively removed. It also emphasizes that sufficient research must have been conducted to provide compelling evidence that the potential damages that can be done to existing conspecific and heterospecific taxa do not outweigh the potential gains to the animals and populations being relocated.

In some cases, headstarting programs may represent a suitable alternative to repatriation or translocation, particularly if the headstarting is done under seminatural conditions. Many species experience the most severe mortality during early life stages. Raising individuals in captivity from a given site to the size or age where they are past this initial peak of mortality and then releasing them at the site where they were initially collected may avoid many of the potential issues associated with translocations while also providing a temporary boost to populations that are in decline. Headstarting is only appropriate, however, where suitable unoccupied habitat exists, or where introduction of individuals will not create problems for existing species at the introduction site.

Specific management recommendations include the following:

- Only translocate animals when other alternatives do not exist.
- Only translocate animals into situations where other animals at the translocation site will not be adversely affected by the introduced animals.
- Only translocate animals when the ecological requirements of the species exist in the new habitat.
- Utilize methods to increase the likelihood that translocations will be successful. These potentially include "soft" translocations (i.e., moving young animals rather than adults with established home ranges) and moving a sufficiently large number of individuals to ensure that a successful breeding population can establish (Germano and Bishop 2009).

Research, Survey, and Monitoring Needs

Both new research and continuing, long-term monitoring are integral parts of the science-driven protection and recovery of sensitive species. For California amphibians and reptiles, our level of basic knowledge on natural history is frequently so fragmentary that even rudimentary information is lacking, and increasing our understanding of these animals is critical for effective management. Many of the particular research needs are discussed in individual species accounts under the "Monitoring, research, and survey needs" section; here, we highlight several basic research and monitoring needs that are common to virtually all taxa.

Distribution

A statewide survey for all amphibians and reptiles is essential to establish baseline data for ongoing status determination and monitoring. Survey efforts are particularly needed for those Special Concern taxa whose population status or range size are a high priority for clarification. These surveys should employ standardized and repeatable methods, with the data emerging from these efforts made widely and easily accessible (Heyer et al. 1994). The Partners in Amphibian and Reptile Conservation Inventory and Monitoring guide (Graeter et al. 2013) serves as an important resource in the detailed design of these distributional surveys. Greatest need taxa include (1) those that may be recently extirpated, but for which comprehensive surveys have yet to be conducted (e.g., the Sonora mud turtle, *Kinosternon sonoriense*); (2) recently discovered taxa that are currently known from relatively small ranges, which may also be tied to specific narrow habitat types, that have yet to be thoroughly surveyed (e.g., the regal ring-necked snake, *Diadophis punctatus regalis*); (3) at-risk taxa that are difficult to detect or that have ranges that are poorly understood because they occur in remote, difficult-to-survey areas (e.g., the Gila monster, *Heloderma suspectum*); and (4) taxa that may occur only on private land where gaining access can be challenging (e.g., the Oregon spotted frog, *Rana pretiosa* or the western spadefoot, *Spea hammondii*). In addition, surveys of virtually all Species of Special Concern, particularly at their hypothesized range edges, would greatly enhance our knowledge of range boundaries for most taxa.

Natural History

Basic natural history and ecology information is the foundation for effective management, and for most amphibian and reptile Species of Special Concern, it is either fragmentary or completely lacking. Home range sizes, habitat suitability analyses, food habits, the effects of invasive plants and animals, compatibility with grazing and agriculture, the effects of human activities including forestry, recreation, and water diversions are unknown for many of the taxa considered here. For some questions and species, this probably is not a pressing problem calling the southern long-toed salamander (*Ambystoma macrodactylum sigillatum*) a "generalist predator" is, to the best of our knowledge, correct, and filling in the precise details of which invertebrates are the most important prey in specific situations may not be an urgent management issue. However, in other cases, filling in at least some of this basic ecology is absolutely critical. For example, of the 19 species of pond/stream breeding Species of Special Concern amphibians, we do not have a well-tested, clearly understood model of terrestrial habitat use for a single taxon. For example, we have little idea of whether the southern long-toed salamander (*A. m. sigillatum*) requires 10, 1000, or 10,000 m radius habitat patches around breeding ponds. Filling in these fundamental information gaps, hopefully across a range of habitat types, constitutes the highest priority conservation-related research need for Species of Special Concern.

Climate Change

Climate change is likely to have a number of effects on the California landscape that are relevant to amphibian and reptile conservation. While the impact that climate change has on California's landscape is undergoing extensive study (reviewed by Cayan et al. 2008a) and is a CDFW focus (http://www.wildlife.ca.gov/Conservation/Climate-Science), the impact that these effects will have on amphibian and reptile species requires additional study. The Association of Fish and Wildlife Agencies has initiated work on this problem in the southeastern United States, and the CDFW, in collaboration with the Southwest Climate Science Center, initiated a detailed investigation of future climate impacts on amphibians and reptiles across California (Wright et al. 2013). A major focus of these projects, and one that requires additional research effort, is to model a full range of future climate change predictions and their impacts on both common and rare amphibian and reptile taxa.

Importantly, the interplay between conservation risks that climate change presents and competing factors that will arise needs careful examination. For example, many climate projections forecast a decrease in the snowpack in the Sierra Nevada, as well as a shift in the speed and timing of snowmelt to be both more rapid and earlier in the year (Maurer and Duffy 2005, California Climate Action Team 2006, Maurer 2007). Even for the lowest carbon emissions scenarios and relatively conservative estimates of increasing temperatures, current models predict a 30–60% decrease in Sierra Nevada snowpack (Cayan et al. 2006). This is likely to have important, direct impacts on amphibians that rely on snowmelt-fed streams and lakes for their breeding habitat. In addition, it is likely to further stress California's already overburdened water resources, setting the stage for further conflicts between the ecological needs of at-risk species and municipal and agricultural demands for increasingly limited water.

The combined impacts of changes in climate on biological diversity are likely to be strong. Several studies have documented ongoing (Walther et al. 2002, Parmesan and Yohe 2003, Root et al. 2003, Root et al. 2005, Parmesan 2006, Pounds et al. 2006) and expected (Hughes 2000) implications of climate change, with some estimates predicting 35% or more (Harte et al. 2004, Thomas et al. 2004) of species being "committed to extinction" under mid-range warming scenarios. These effects will likely be especially pronounced for amphibians, which generally exhibit limited dispersal and are already experiencing severe declines (Stuart et al. 2004, Lawler et al. 2010). The uncertainties involved with estimating specific effects that will occur on landscapes, species' responses to these changes, and the interplay of factors that will result from climate change (e.g., agricultural and municipal water needs vs. amphibian breeding habitat needs, alternative energy development in the desert vs. reptile habitat needs) clearly indicate that this topic requires further study. An important step in this direction is a recent initiative by the US Fish and Wildlife Service (USFWS) to fund the California Landscape Conservation Cooperative, an interdisciplinary program to facilitate research and planning across scientific and management agencies in the state (http://californialcc.org/about-us). Results of the CDFW and Southwest Climate Science Center collaboration mentioned above should be integrated into the California Landscape Conservation Cooperative process.

Threats from Disease

Diseases in amphibian and reptile populations have become an issue of global significance. In particular, the pathogenic chytrid fungus, *Batrachochytrium dendrobatidis* (*Bd*), has been linked to precipitous declines in several amphibian species in the state (e.g., the Sierra Nevada yellow-legged frog, *Rana sierrae*) and globally (Stuart et al. 2004). At the present time, no broadly effective management strategies for controlling or mitigating the effects of this pathogen are known, and this is a critical, active research area. Proposed management strategies that would benefit from further study include altering population dynamics to minimize disease outbreaks, treating individual amphibians and habitats to control the prevalence or spread of disease, biological control of *Bd* using the zooplankter *Daphnia magna*, and in the most dire cases, maintenance of captive assurance colonies followed by repatriation with assisted selection (Buck et al. 2011, reviewed by Woodhams et al. 2011). Efforts to develop management strategies should not focus exclusively on strategies for the short term, such as direct control of *Bd* in the wild or captive breeding. Rather, management strategies that allow susceptible amphibians to persist in the wild in the presence of *Bd* are needed for long-term conservation of sensitive species (Woodhams et al. 2011).

The extent and type of interactions that *Bd* may have with other threats, such as climate change, pesticide exposure, or other pathogens, are also key research needs. A growing body of work on *Bd* indicates that it has negative consequences on at-risk species of amphibians in California (Davidson et al. 2007, Morgan et al. 2007, Andre et al. 2008, Lacan et al. 2008, Padgett-Flohr 2008, Briggs et al. 2010), that synergistic interactions with pesticides may have strong biological effects (Davidson et al. 2007), and that terrestrial amphibians may serve as vectors for the disease (Schloegel et al. 2009, Weinstein 2009). Other emerging diseases, particularly those that have their origins in human pets or are a result of human-mediated movements and relocations of animals are also high-priority research targets. Important examples include ranaviruses and iridoviruses, both of which have also been linked to amphibian declines (e.g., Picco et al. 2007, Schloegel et al. 2009).

Phylogeography and Landscape Genetics

Another important research need, and one that may be easier to fill than comprehensive ecological studies, is genetic analyses for most species. Some limited phylogeographic and landscape genetic work has been completed for a few California amphibians and reptiles (or their close relatives), and these analyses have provided key insights into the importance of drainages on stream- and pool-breeding amphibians and reptiles (Shaffer et al. 2000, Spinks and Shaffer 2005, Dever 2007, Wang 2009b, Lind et al. 2011), corridors of land use (Wang et al. 2009), the importance of environmental variables in structuring populations (Savage et al. 2010), and a variety of other problems (e.g., the provenance of introduced populations; Johnson et al. 2010). At a broader, regional-to-range-wide scale, phylogeographic studies have been conducted for several Species of Special Concern, in many cases indicating either that previous subspecies (which often serve as proxies for genetic lineages) are nondiagnosable and correspond poorly to genetic patterns (Rodríguez-Robles et al. 1999b) or that unappreciated lineage diversity is stronger than previously suspected (Shaffer et al. 2004, Leavitt et al. 2007, Parham and Papenfuss 2009). We are aware of phylogeographic work for roughly half of the Species of Special Concern (although many of those studies rely on a single mitochondrial gene and need data from additional nuclear gene analyses), and we strongly encourage the research community to gather these data for the remaining taxa.

Monitoring

To establish that a species or population is declining or recovering requires long-term monitoring. Such efforts can take many forms, each with strengths and weaknesses. Ideally, monitoring data would be generated by intensive, multiyear mark–recapture-based studies that follow the fate of individuals through time, leading to a detailed inventory of population increases and decreases (Heyer et al. 1994). Such monitoring is not difficult conceptually, but it requires time, effort, and often substantial financial resources. However, this is also an area that is undergoing renewed methodological development. Monitoring methods now exist that require less recapture effort and that can incorporate detection probabilities in a rigorous manner, both of which can help to effectively monitor rare and/or cryptic taxa (reviewed by Mazerolle et al. 2007). One such example is the emerging techniques to monitor rare or cryptic taxa via detection of persistent DNA in environmental samples (Ficetola et al. 2008, Dejean et al. 2011).

Techniques to survey amphibians and reptiles vary, depending on the taxon, habitat, and life stage involved. Although standardized survey protocols are essential to proper inventory and monitoring, relatively few have been developed, representing an ongoing research need, particularly for rare taxa or taxa that are difficult to detect. Some of this standardization is beginning to take place and a few excellent resources are available or forthcoming (Heyer et al. 1994; the ongoing Amphibian Research and Monitoring Initiative being undertaken by the US Geological Survey, and the Partners in Amphibian and Reptile Conservation Inventory and Monitoring guides are such examples). In the absence of detailed, multiyear monitoring, we advocate at least two potential approaches that have received relatively little attention to date for amphibian and reptile taxa. The first is single-pass monitoring via population surveys conducted on public lands. Such surveys can be incredibly informative, yet only require a few field days per year to monitor a large number of species and sites (e.g., Thomson et al. 2010). A recent example for 75 ponds from the East Bay Regional Park District provided multiyear data for five species of pond-breeding amphibians and two species of semiaquatic garter snakes, and demonstrates the kind of data that can be collected even with very cursory efforts for each site (S. Bobzien, unpublished data; M. Ryan, unpublished data). A critical goal of such monitoring efforts should be to publish the results in the peer-reviewed literature and/or deposit in a publically available, curated dataset. Our sense is that a great deal of valuable monitoring data exists, but is not easily accessible because it has never been published or made publically accessible. Another type of single-pass "monitoring" can be genetic monitoring. By collecting non destructive, but vouchered, tissue samples, reasonable estimates of the effective population size (Wang 2009a, Wang et al. 2011), historical population increases or decreases (Piry et al. 1999), and ongoing movement between existing populations (Wilson and Rannala, 2003) can be applied to many populations and species. Although each of these genetic approaches has its own set of assumptions and caveats, together they form a powerful addition to traditional field-based studies of population monitoring.

A second approach to monitoring falls under the more general category of "citizen science" (Bonney et al. 2009, Dickinson et al. 2010). Although often less rigorous and more error prone than more formal monitoring, the interested public comprises a large network of knowledgeable, committed individuals who will often willingly contribute to overall monitoring efforts. These efforts can help identify general patterns of population increases and decreases, as has been amply demonstrated by the very successful Breeding Bird Surveys (Sauer et al. 2011) and Christmas Bird Counts (National Audubon Society 2011) conducted for North American birds. Several programs for citizen-science-based frog and toad monitoring programs are in place in other parts of the United States (e.g., the FrogWatch USA program, http://www.aza.org/frogwatch), and they have provided valuable data on breeding time, duration, and population sizes for frogs and toads based on their audible calls at breeding sites. Road surveys (Coleman et al. 2008) can also provide valuable data on population sizes, although the confounding effects of mortality induced by vehicular traffic is always a concern in such studies. That said, documentation of road mortality, particularly during key migration seasons, is an ideal topic of additional citizen science initiatives. California has recently initiated at least two citizen-science web-based projects focusing on southern California reptiles and amphibians (RASCals; see http://www.nhm.org/site/activities-programs/citizen-science/rascals, and the California chapter of the Field Herp Forum http://www.fieldherpforum.com), both of which seek to increase communication and the dissemination of distributional information on California amphibians and reptiles.

Finally, because monitoring provides the basic information upon which much of conservation rests, a temptation naturally arises to "over-monitor." By this, we mean that additional monitoring becomes favored over the implementation of management actions. Monitoring efforts constitute the most important strategy for measuring the effectiveness of conservation actions. However, monitoring also carries a cost, because these efforts require valuable conservation resources that otherwise might be spent on direct management efforts. Monitoring efforts should have clearly defined goals and well-characterized statistical power, including an assessment of the added benefit to be gained from future monitoring efforts. Monitoring efforts should be clearly documented, and results should be readily accessible. In some cases, the optimal strategy may be limited, but consistent, monitoring combined with direct conservation actions, rather than ever-more detailed monitoring with fewer actions. The implementation of effective management in the face of imperfect knowledge about the status of populations is one of the greatest challenges facing the conservation of many amphibian and reptile species.

Species of Special Concern Conservation Recommendations

To promote the conservation of amphibian and reptile Species of Special Concern in California, we make the following recommendations:

- Maintain a Species of Special Concern Technical Advisory Committee with explicit expertise covering the taxonomic and geographic scope of taxa in California. We recommend that membership on this committee be of relatively limited term (e.g., 10 years) to ensure that new voices and fresh problem-solving strategies are available. We especially encourage that committee composition include some early career scientists, particularly those with strong statistical and technical skills. This group should meet periodically in order to update and revise the status information on the Species of Special Concern.

- Develop and implement a web-based mechanism whereby the Species of Special Concern document can be more easily updated and improved, creating a "living document" that is responsive to changing conditions and new data.
- In conjunction with efforts to facilitate future revisions of this document, support the development of a database that collates species occurrence data. This database should house information on both positive and negative occurrence data and not be limited to species that are already designated as Species of Special Concern.
- Increase wildlife agency capacities to address management needs of California's amphibians and reptiles, as funding and staffing allow.
- Establish both a priority list and a funding stream for critical research needs for Species of Special Concern.
- Continue to promote strong collaborations between wildlife agencies and the university/research communities throughout California to ensure that the strongest possible science is brought to bear on important management needs and that the state's research priorities are being pursued.
- Use forthcoming analyses of predicted road usage and construction as a management guide for conservation planning for Species of Special Concern. Included in this analysis should be ways to use tunnels or other constructs to minimize the effects of new and existing roads on Species of Special Concern.
- Create a coordination network for localities, voucher specimens, and tissue samples for amphibian and reptiles from throughout California. Roadkill specimens are a particularly valuable source of information, since they represent vouchered specimens and, in some cases, sources of DNA for genetic research and life history data (diet, body condition, etc.) for ecological studies.
- Create a mechanism by which both professional biologists and concerned citizens can contribute locality, natural history, and other data types that might help detect or quantify conservation risk for Species of Special Concern. Improve data sharing and communication among wildlife agencies, amphibian and reptile conservation groups, and organizations in the avocational herpetological community.
- To facilitate data collection, streamline the process for appropriate permitting for research by professionals, and in the case of citizen science projects, the public.
- Encourage publication of data arising from these efforts in the peer-reviewed literature to increase access to management-relevant findings, particularly for government reports and studies conducted by private consultants.
- Integrate information from this document, as appropriate, with that of an upcoming analysis of the existing regulatory situation for all of California's amphibians and reptiles and their general conservation needs.

SPECIES ACCOUNTS


COUNTIES AND ISLANDS OF CALIFORNIA
124°0'0"W
120°0'0"W
116°0'0"W
41°0'0"N
37°0'0"N
33°0'0"N
Del Norte
Siskiyou
Modoc
Trinity
Shasta
Humboldt
Lassen
Tehama
Plumas
Mendocino
Glenn
Butte
Sierra
Nevada
Lake
Colusa
Yuba
Sutter
Placer
Yolo
Sonoma
Napa
El Dorado
Sacramento
Amador
Alpine
Solano
Marin
Calaveras
Contra Costa
San Francisco
San Joaquin
Tuolumne
Mono
Alameda
San Mateo
Stanislaus
Mariposa
Santa Clara
Merced
Santa Cruz
Madera
San Benito
Fresno
Inyo
Monterey
Tulare
Kings
San Luis Obispo
Kern
Santa Barbara
San Bernardino
Ventura
Los Angeles
San Miguel
Santa Rosa
Santa Cruz
Santa Barbara
Orange
Riverside
San Nicolas
Santa Catalina
San Clemente
San Diego
Imperial

COASTAL TAILED FROG

Ascaphus truei Stejneger 1899

Status Summary

Ascaphus truei is a Priority 2 Species of Special Concern, receiving a Total Score/Total Possible of 61% (67/110). During the previous evaluation, it was also considered a Species of Special Concern (Jennings and Hayes 1994a).

Identification

Ascaphus truei is a small (2.5–5.0 cm SVL) dark frog with an olive, brown, gray, or reddish dorsum and lighter colored ventral surface. Other color characters include a pale triangular blotch on the snout and a dark eye stripe. This species has rough, granular skin, and the outermost toes on the hind feet are broad. Males have a unique tail-like copulatory organ that is unmistakable. This frog is nocturnal and adults have vertical pupils (Stebbins 2003).

Larvae grow up to 6.0 cm in TL and are adapted to life in fast-flowing streams. They have dorsoventrally flattened bodies and large sucking mouthparts that extend nearly halfway down their head-body on the ventral surface. These morphological traits allow larvae to attach to rock substrates (Altig and Brodie 1972, Nussbaum et al. 1983, Welsh and Hodgson 2011). Larvae often have a light-colored tail tip with a proximal dark band (Stebbins 2003).

Coastal Tailed Frog: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	10
ii. Distribution trend (25)	15
iii. Population concentration/ migration (10)	0
iv. Endemism (10)	0
v. Ecological tolerance (10)	10
vi. Population trend (25)	15
vii. Vulnerability to climate change (10)	10
viii. Projected impacts (10)	7
Total Score	67
Total Possible	110
Total Score/Total Possible	0.61

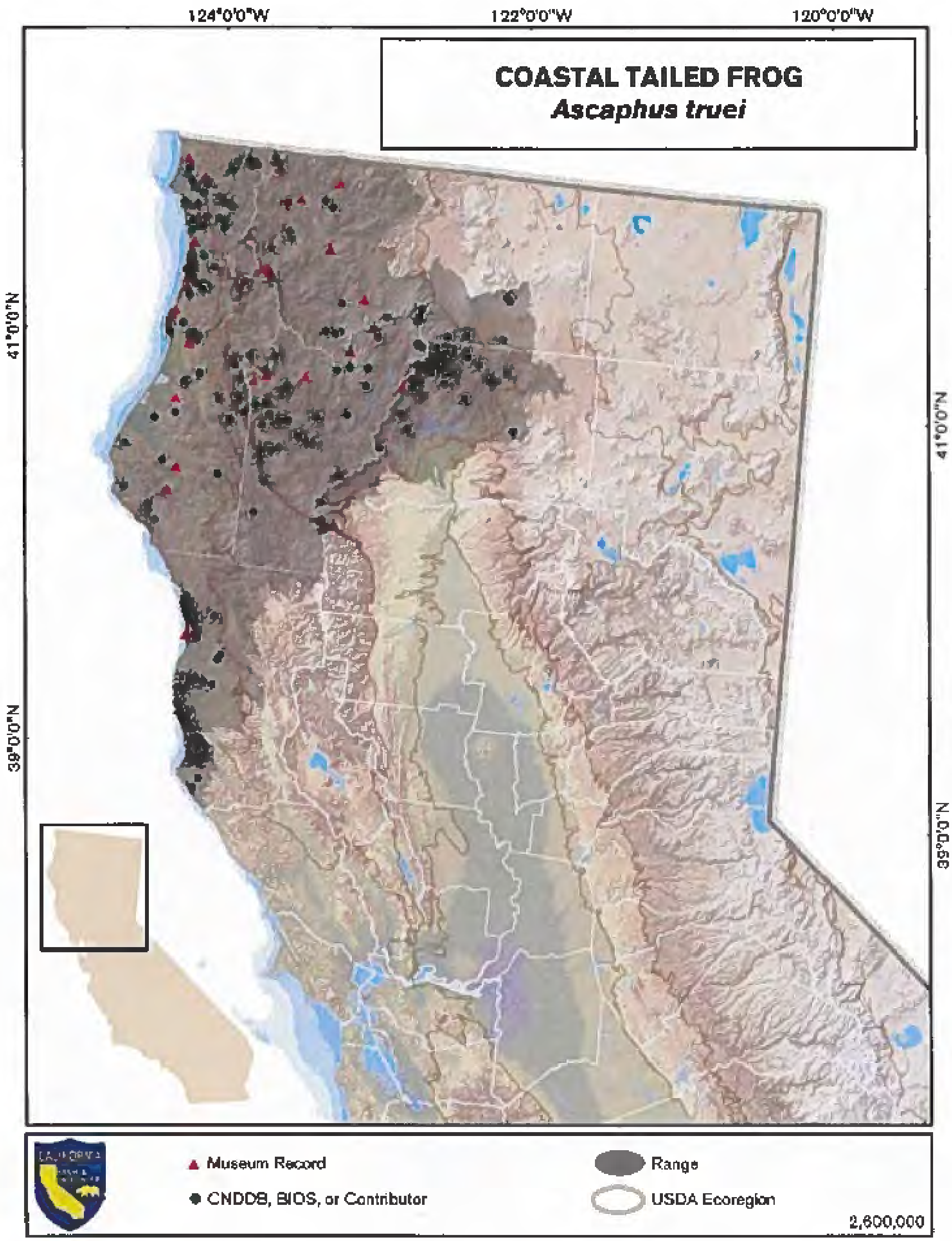


PHOTO ON PREVIOUS PAGE: Coastal tailed frog, Del Norte County, California. Courtesy of Rob Schell Photography.

In California, metamorphosed *A. truei* may be confused with co-occurring foothill yellow-legged frogs (*Rana boylii*). *Rana boylii* have horizontal pupils, more robust hind legs, and males lack enlarged toes and "tails" (Stebbins 2003). In addition, the enlarged mouthparts of *A. truei* tadpoles are distinctive.

Taxonomic Relationships

The formerly monotypic genus *Ascaphus* was recently split into a coastal (*A. truei*) and an inland species (*A. montanus*), but California populations remain *A. truei* (Nielson et al. 2001, Nielson et al. 2006). The two species of *Ascaphus* comprise the family Ascaphidae. This family forms the sister group to all other anurans either alone or in combination with the New Zealand endemic Leiopelmatidae (Roelants et al. 2007). In either case, it is from one of the oldest and most phylogenetically distinctive extant anuran lineages.

Life History

Ascaphus truei exhibits substantial geographic variation in life history. Here, we focus on data from California populations where possible. Breeding occurs primarily in the spring and summer in coastal populations (Sever et al. 2001, Burkholder and Diller 2007), but there are reports from Trinity County of animals found in breeding condition in the fall (J. Garwood, pers. comm., in Burkholder and Diller 2007). Females likely breed in alternate years (Burkholder and Diller 2007) and can store viable sperm for up to a year (Nussbaum et al. 1983, Sever et al. 2001). Eggs begin developing in the fall, and oviposition occurs the following summer between July and September in California populations (Sever et al. 2001, Karraker et al. 2006). Egg diameter is 4 mm on average (Brown 1977), and clutch size averages around 40 for the species with a range of 28–89 eggs per clutch documented in California populations (Karraker et al. 2006). Egg masses can be difficult to find in the field (Karraker et al. 2006). Recent surveys in coastal California have found single and multiple clutches, with the timing of the surveys (late August–early September) likely the most important factor for detecting eggs (R. Bourque, pers. comm.). Clutches are pearl-like strings of eggs and have been found attached to the underside of cobble or boulder substrates in riffles and pools (Karraker et al. 2006).

Time to metamorphosis in lowland coastal California populations (elevation <200 m) is 1–2 years (Wallace and Diller 1998, Bury and Adams 1999). Longer developmental times have been observed in montane populations (e.g., 4 years to metamorphose in a Washington population at ~1500 m elevation; Brown 1990). In a population in Humboldt County, California, females reached sexual maturity 2.5–3 years after metamorphosis, while males were sexually mature 1.5–2 years after metamorphosis (Burkholder and Diller 2007). Post-metamorphic frogs grow year-round, with growth rates fastest in the summer (Burkholder and Diller 2007).

Adults and post-metamorphic juveniles are generalist invertebrate predators (Bury 1970b). Larvae are generalist grazers and scrapers, consuming diatoms and other periphyton (observations from *A. montanus*; Metter 1964).

Landscape genetic studies have detected different patterns of connectivity among populations in California and Washington. In four watersheds in Mendocino County at the southern range limit of the species, high population structure among watersheds suggested limited long-distance gene flow, and movements within watersheds were inferred to occur along waterways (Aguilar et al. 2013). By contrast, a study in Washington concluded that some animals engage in long-distance dispersal through terrestrial habitats, and these movements do not rely on stream connectivity (Spear and Storfer 2008). These differences may be due to regional variation in climate and forest type, though additional studies are needed.

Habitat Requirements

Ascaphus truei requires cold, permanent, swift-flowing streams with coarse (e.g., cobble, boulder, bedrock) substrates. Some populations may persist in streams that occasionally dry depending on the length of the larval period (Wallace and Diller 1998). *Ascaphus truei* tends to be more common in mature and old-growth forest relative to younger stands, in terms of both presence and abundance (Bury and Corn 1988, Corn and Bury 1989, Welsh 1990, Gomez and Anthony 1996, Welsh and Lind 2002, Welsh et al. 2005, Ashton et al. 2006).

Several studies have examined the relationship between *A. truei* presence and abundance and environmental variables at different scales. Larvae are positively associated with low stream temperatures, high water velocity, steep gradients, and the presence of riffles, waterfalls, and cobble and boulder substrates (Hawkins et al. 1988, Bury et al. 1991, Welsh and Ollivier 1998, Diller and Wallace 1999, Adams and Bury 2002, Welsh and Lind 2002, Wahbe and Bunnell 2003). Larvae are negatively associated with fine sediment load (i.e., embeddedness), pools, and slow-flowing stream habitat (Hawkins et al. 1988, Corn and Bury 1989, Welsh and Ollivier 1998, Diller and Wallace 1999, Welsh and Hodgson 2008). Steep gradients allow for flushing of fine sediments, although gradient effects may be more pronounced in harvested compared to primary forest habitat (Corn and Bury 1989). Adults are positively associated with high rainfall, moist forest habitats, and pool habitat, and negatively associated with fine sediment loads (Welsh and Lind 2002, Ashton et al. 2006). Adults and larvae in the Mattole Watershed were restricted to headwater channels, and canopy closure was the best single predictor of *A. truei* presence (Welsh and Hodgson 2011). *Ascaphus truei* were never detected in streams where canopy closure was less than 83% (Welsh and Hodgson 2011).

Some researchers have suggested a positive association between *A. truei* and the presence of harder, more consolidated parent geologies because they produce less sediment (Diller and Wallace 1999, Dupuis et al. 2000, Wilkins aud Peterson 2000). However, *A. truei* does occur in streams with unconsolidated geologies, such as those derived from marine sediments, particularly in areas uot subjected to recent or historical anthropogenic disturbance (e.g., Adams and Bury 2002, Welsh and Lind 2002, Ashton et al. 2006). The absence of *A. truei* from some streams with unconsolidated geologies may be because the presence of easily erodable substrates exacerbates the impacts of habitat disturbance, which can have long-lasting effects (Adams and Bury 2002, Welsh and Lind 2002, Ashton et al. 2006).

Ascaphus truei is extremely sensitive to warm temperatures at all life stages. Eggs have a temperature tolerance range from 5°C to 18.5°C (Brown 1975a). The critical thermal maximum range for larvae is 28.9–30.1°C, and larvae avoided temperatures above 22°C in laboratory trials (de Vlaming and Bury 1970). First-year larvae collected from Del Norte County selected temperatures below 10°C along a thermal gradient in the laboratory, while second-year larvae selected temperatures closer to 15°C (de Vlaming and Bury 1970). The critical thermal maxima for adults ranged on average from 27.6°C to 29.6°C (data from *A. montanus*; Claussen 1973). Field temperatures at occupied sites are usually well below these limits, with larvae occurring in streams with a mean of 11.6°C (range 5.7–15.8°C; Welsh and Hodgson 2008).

In addition to narrow thermal tolerances, *A. truei* is also extremely sensitive to desiccation (Brattstrom 1963), which may limit adult use of upland habitat to periods of wet weather conditions (Nussbaum et al. 1983). One mark-recapture study in Humboldt County documented movements of only 0–30 m along the stream channel over a two-year period (Burkholder and Diller 2007). However, recapture probabilities were low, and some auimals may have moved beyond the study area. Longer distance movements have been documented from populations outside of California, from tens of meters up to 400 m into upland habitat (McComb et al. 1993, Gomez and Anthony 1996, Vesely 1996, Wahbe et al. 2004, Matsuda and Richardson 2005). Seasonal variation in adult location in managed forests in Washington was hypothesized to be a localized breeding migration, with downstream movements for oviposition and a return upstream in late summer (Hayes et al. 2006). It is unknown whether similar movements also occur in older, less disturbed forests in the area. In an *A. montanus* population in Montana, seasonal movements may be due to behavioral thermoregulation (Adams and Frissell 2001).

Distribution (Past and Present)

Ascaphus truei ranges from British Columbia to northern California, mostly west of the Cascades Mountains (Stebbins 2003). California is the southern limit of the range, with *A. truei* occurring south from the Oregon border along the coast to Mendocino County and east to Shasta County (Grinnell and Camp 1917, Mittleman and Myers 1949, Salt 1952, Bury et al. 1969, Welsh 1985). *Ascaphus truei* ranges from near sea level in Humboldt County up to 2150 m in the Trinity Alps (J. Garwood, pers. comm.).

Random sampling of streams has documented higher occupancy rates for *A. truei* in unmanaged or older forests compared to managed or younger stands (Welsh 1990). We therefore assume that some historically occupied localities are no longer occupied due to disturbance. In one study in the Mattole Watershed in Mendocino and Humboldt counties, *A. truei* was present in 71% of streams in old and mature forests, but was not found in second growth forests (Welsh et al. 2005). Further studies in the Mattole Watershed have found *A. truei* in 67% (14/21) of streams in unmanaged forests, but only in 4% (1/28) of streams in managed stands (H. Welsh and G. Hodgson, unpublished data). Streams with mixed harvest histories in the South Fork of the Trinity River had an intermediate level of occupancy, with 28% of streams occupied (17/60; Welsh et al. 2010). Studies from outside of California also indicate that *A. truei* is present in a greater proportion of streams in unmanaged forests (Bury and Corn 1988, Corn and Bury 1989, Hayes et al. 2006). A survey of streams in private, managed timber lands all less than 80 years old along the northern California coast found stream occupancy rates of 37% (18/49) at the level of 30 m sampling reaches and 76% (54/72) at the level of entire stream reaches (Diller and Wallace 1999). The relatively high occupancy rates in these young forests are thought to be due to the ameliorating effect of maritime climate, as most sites were within 30 km of the coast (Bury 1968, Diller and Wallace 1999).

Trends in Abundance

Ascaphus truei tends to be lower in abundance in managed compared to unmanaged forest stands (Bury and Corn 1988, Corn and Bury 1989, Welsh 1990, Gomez and Anthony 1996, Welsh and Lind 2002, Ashton et al. 2006). Clear-cuts can have immediate effects on abundance. Larval densities were higher in late-succession and old-growth forests compared to adjacent clear-cuts lacking streamside buffers in Oregon and British Columbia (Dupuis and Steventon 1999, Biek et al. 2002). Upland pitfall trapping in clear-cuts and mature forests in British Columbia found similar total numbers of *A. truei* in both forest types, but very few adults in clear-cuts, suggesting that immature frogs in clear-cuts are transients or incur high mortality rates (Matsuda and Richardson 2005). Several researchers have predicted declines or continuing declines if anthropogenic disturbances continue (e.g., Corn and Bury 1989, Dupuis and Steventon 1999, Welsh and Lind 2002, Ashton et al. 2006, Olson et al. 2007).

Nature and Degree of Threat

Declines and local extirpations to date are largely due to land management including timber harvesting and road construction (Welsh and Ollivier 1998, Welsh et al. 2005). Marijuana cultivation and climate change are also emerging as potential threats to this taxon.

The mechanisms underlying declines and extirpations due to timber harvesting and road construction are primarily increased sedimentation, increased stream temperatures, and fragmentation. While the initial impacts of road construction may be relatively short-lived, longer-term impacts are caused by sedimentation due to runoff from poorly maintained dirt and gravel roads (L. Diller, pers. comm.). Reduced canopy cover does not seem to increase temperatures as much at high-elevation sites, and *Ascaphus truei* may be more resilient to timber harvesting in areas where stream temperature is cooler due to overall climate (e.g., Diller and Wallace 1999, Wahbe and Bunnell 2003). Reductions in canopy or riparian vegetation that result in increased light levels may cause shifts in the algal community (i.e., from diatoms to filamentous green algae) that negatively affect the quality and abundance of larval food (L. Diller, pers. comm.). Landscape genetic studies in Washington suggest that significant overland dispersal occurs through terrestrial habitat, with gene flow detected between populations on a scale of 25–30 km (Spear and Storfer 2008). While timber harvests have some initial effect on gene flow, it may take multiple generations before the effects of fragmentation on population genetic structure can be detected.

An emerging threat to *A. truei* is large-scale marijuana cultivation, though little data is currently available due to limited accessibility of private lands. Similar to timber harvesting, marijuana cultivation requires clearing land and building roads which can increase sedimentation. Contamination from pesticides used on marijuana grows has been documented to negatively affect mammals in the field (Thompson et al. 2014), and amphibians are likely to be susceptible as well because of their permeable skin. Of particular concern for headwater amphibians like *A. truei* is the dewatering of waterways that are diverted for irrigation (CDFG 2013).

Climate change poses potential risks to *A. truei* through increased temperatures, changes in hydrology, changes in fire regime, and vegetation shifts. Mean annual temperatures are expected to increase throughout northwestern California (reviewed in PRBO 2011). The frequency of extremely hot days is projected to increase, with roughly nine additional days over 32.2°C (Bell et al. 2004). Such temperatures exceed the critical thermal maxima for all life stages of *A. truei*, though water temperatures, microhabitat structure, and behavioral thermoregulation may ameliorate these effects. For coastal populations, upwelling is expected to intensify, which may increase fog development and contribute to cooler, moister conditions (Snyder et al. 2003, Lebassi et al. 2009). Coastal areas may therefore continue to provide more favorable climatic conditions than areas farther inland. Potential changes in precipitation are less clear, with some models predicting modest increases, some modest decreases, and some reductions in rainfall of up to 28% (reviewed in PRBO 2011). Warmer temperatures will result in less precipitation stored as snow, and reductions of 30–80% are predicted for snowpack accumulation in northwestern California (Snyder et al. 2004, Cayan et al. 2008b). The timing of spring snowmelt has shifted later in the spring in this region over the last 50 years (Stewart et al. 2005), though the timing of future shifts is unknown. Reductions in water availability due to reduced snowpack and possibly reduced precipitation will affect the timing and magnitude of stream flows and may lead to a mismatch between the timing of breeding and appropriate stream conditions. How fire regime will be affected by climate change in northwestern California is not well understood. Some models predict little change in fire regime or even decreases in area burned along the northern coast (Fried et al. 2004, Lenihan et al. 2008). Increases in area burned have been predicted for the southern coast of northwestern California (Lenihan et al. 2008). Westerling et al. (2011) projected a 100% increase in area burned in northwestern California under some scenarios. Direct mortality of adults and larvae due to fire has been documented in *A. montanus* populations (P. Van Eimeren, pers. comm., in Pilliod et al. 2003, Hossack et al. 2006). Short-term impacts of fire may be due to warmer temperatures and/or increased ammonia levels or other changes to water chemistry (Pilliod et al. 2003), but long-term impacts are understudied. Vegetation communities are expected to shift from moist conifer to drier mixed evergreen forest, with reductions in Douglas fir and redwood forest in particular (Lenihan et al. 2008, PRBO 2011). It is unclear what effect these shifts may have on *A. truei* because stream conditions and forest age seem to be more important indicators of habitat quality than forest type.

Status Determination

Ascaphus truei is a specialist of cold, headwater stream habitats in old and mature forests, a habitat type that incurs substantial disturbance from land management activities. Declines in distribution and abundance have been documented in response to anthropogenic disturbances, and climate change has the potential to further negatively impact this species. These factors all contribute to a Priority 2 designation for this species.

Management Recommendations

Remaining old and mature forest habitats should be protected, with a focus on managing the entire stream network (Olson et al. 2007, Welsh 2011). Retaining streamside buffers on managed lands can help mitigate the effects of logging and roadbuilding, but more research is needed to determine buffer prescriptions, particularly how to preserve stream network processes (Olson et al. 2007). One model recommends riparian management zones 40–150 m wide and patch reserves along headwater streams to accommodate upland habitat use and promote connectivity among drainages (Olson et al. 2007). The ecological effects of buffer protections may vary across habitat types, and narrower buffers may be effective in more mesic coastal habitat compared to more xeric inland sites in the California range of *Ascaphus truei*.

Construction of new roads should be minimized or avoided in areas where protecting *A. truei* is a high conservation priority. To reduce the sedimentation impacts of runoff from roads, forest roads should be disconnected from stream systems (e.g., through the use of ditch-relief culverts). Use of heavy equipment should be avoided or restricted on forest roads when larvae are present in nearby aquatic habitat. Road management strategies should be applied to all forest roads, not just those used for timber harvest.

Ascaphus truei management would benefit from greater legal clarity regarding state and federal law on marijuana cultivation in California. Currently, some cultivation is legal under state law but prohibited under federal law, which may be hampering regulation of cultivation sites. Greater enforcement of existing environmental and land use laws is needed, and development of additional regulations should consider environmental impacts on *A. truei*.

Monitoring, Research, and Survey Needs

The presence of uncut streamside buffers on the entire channel network can ameliorate the impacts of land management on *Ascaphus truei* populations, but more research is needed into optimum buffer widths as they relate to different life history requirements and different portions of the catchment network. Studies from *A. truei* populations in British Columbia and Oregon have found positive effects of buffers 5–60 m wide (Bull and Carter 1996, Dupuis and Steventon 1999, Stoddard and Hayes 2005, Pollett et al. 2010). Experiments to determine optimal buffer widths in California habitats are needed. We recommend, at a minimum, that comparative data from coastal Mendocino County (the southern limit of the species range), coastal Humboldt/Del Norte Counties (the northern limit of the species range in California), and inland Trinity County are needed to assess the minimum forest buffer on industrial timber lands to retain key temperature and stream clarity conditions for *A. truei*.

Much of the research on *A. truei* has focused on stream-breeding habitat and presence/absence studies. While more difficult, monitoring efforts to document abundance and population dynamics are needed to gain insight into declines that cannot be inferred from presence/

absence surveys (Welsh 2011). Such studies could also determine which life history stages limit population growth in this species. When possible, population estimates in managed forests should be compared to *A. truei* abundance in nearby undisturbed mature forest stands (i.e., reference populations) to assess the impacts of disturbance (Welsh 2011).

More studies are needed on use of upland habitats by adults and dispersing animals. Such studies should be targeted at identifying terrestrial habitat corridors, if present, which can then be protected to maintain connectivity among populations (Olson et al. 2007, Olson and Burnett 2009). Landscape genetic analyses from replicate California populations may be particularly informative, given that recent studies from different parts of the range reach different conclusions about population connectivity (Spear and Storfer 2008, Spear and Storfer 2010, Aguilar et al. 2013).

Field research on impacts of marijuana cultivation on amphibian populations would contribute to development of environmental regulations for this growing industry and inform management strategies in cultivated areas.

SONORAN DESERT TOAD

Bufo alvarius Girard 1859

Status Summary

Bufo alvarius is a Priority 1 Species of Special Concern, receiving a Total Score/Total Possible of 75% (64/85). During the previous evaluation, it was also designated as a Species of Special Concern (Jennings and Hayes 1994a). The species has not been found in California since 1955 (but see the "Distribution" section).

Identification

Bufo alvarius is a large (10.1–19.0 cm SVL) olive, brown, or gray toad with prominent cranial crests and large elongate paratoid glands (Stebbins 2003). The skin is smoother than in other North American toads, with few warts along the dorsum. *Bufo alvarius* has large warts on the hind limbs and prominent white warts at the corners of the mouth (Stebbins 2003). The call is a low-pitched bleat or screech (Elliott et al. 2009).

This species is unlikely to be confused with any other anuran within its California range. All other true toads (family Bufonidae) in the region have extensive warts over the entire dorsal surface and lack large warts on the hind legs. The spadefoots (*Scaphiopus* and *Spea*, family Scaphiopodidae) are much smaller as adults

Sonoran Desert Toad: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	10
ii. Distribution trend (25)	20
iii. Population concentration/ migration (10)	10
iv. Endemism (10)	0
v. Ecological tolerance (10)	7
vi. Population trend (25)	Data deficient
vii. Vulnerability to climate change (10)	7
viii. Projected impacts (10)	10
Total Score	64
Total Possible	85
Total Score/Total Possible	0.75