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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 (5 OF 7)

June 16, 2025

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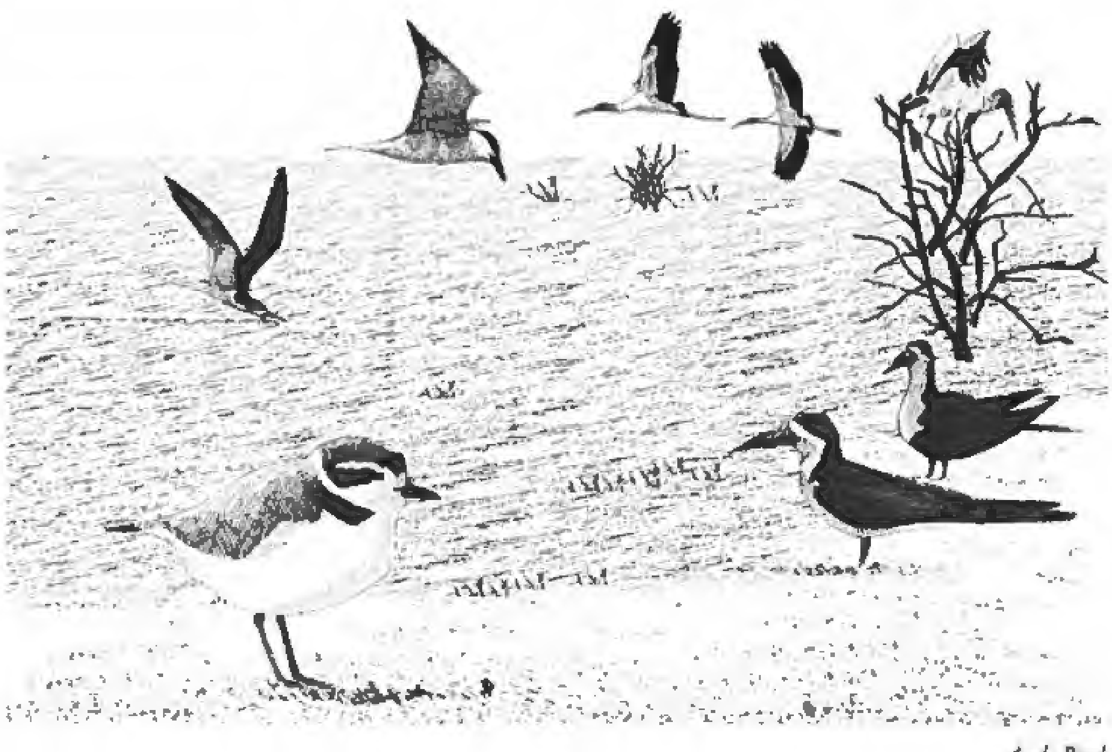
SPECIES ACCOUNTS



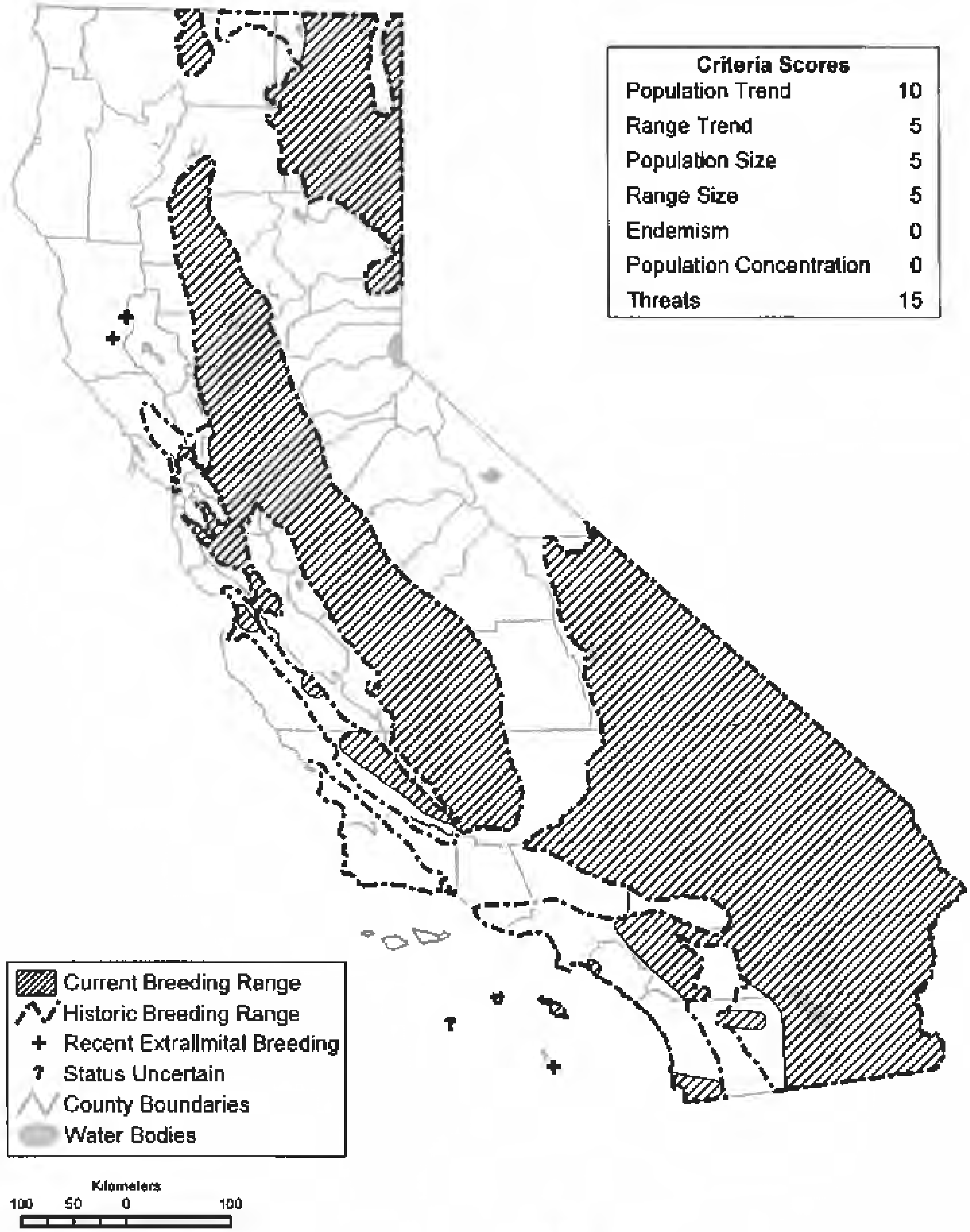
Andy Birch

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, Catlin 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 counties. Outside of this area has been observed sparingly" in Sonoma, Napa, Solano, and Marin counties (Grinnell and Wythe 1927). Willet (1933), also lacking quantitative 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, F. 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 (postbreeding 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 could 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 urban 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 agriculture 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 guidelines for maintaining Burrowing Owls and their burrows during management of agricultural water conveyance structures.
- Assess various strategies for maintaining 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.

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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

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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].

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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

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[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 habitats 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.

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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);

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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

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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 natural 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.

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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.

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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:

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THIS PROJECT WAS PREVIOUSLY: ☐ GENERAL INVESTIGATION FILE NUMBER _____ AND FILE NAME _____.

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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: ______________________

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

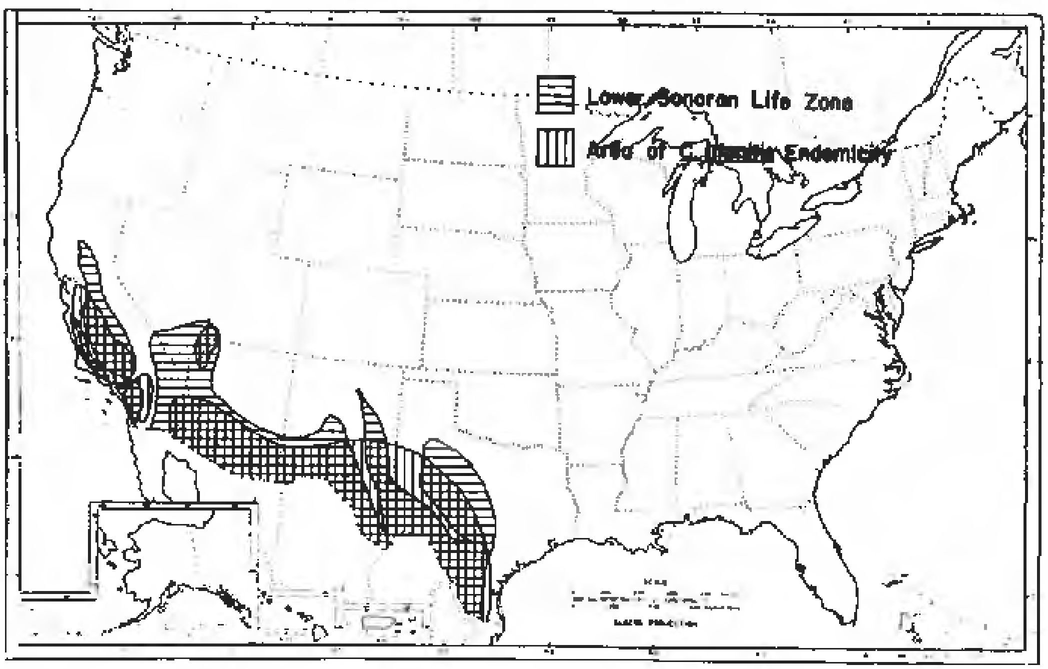

Figure 1—Lower Sonoran Life Zone and area of Coccidioides immitis endemicity in the United States [After Smith, C. E.[9]]

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 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 exposnres 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 measnres 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, 1959-1963. 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.

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Expanding Understanding of Epidemiology of Coccidioidomycosis in the Western Hemisphere

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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

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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

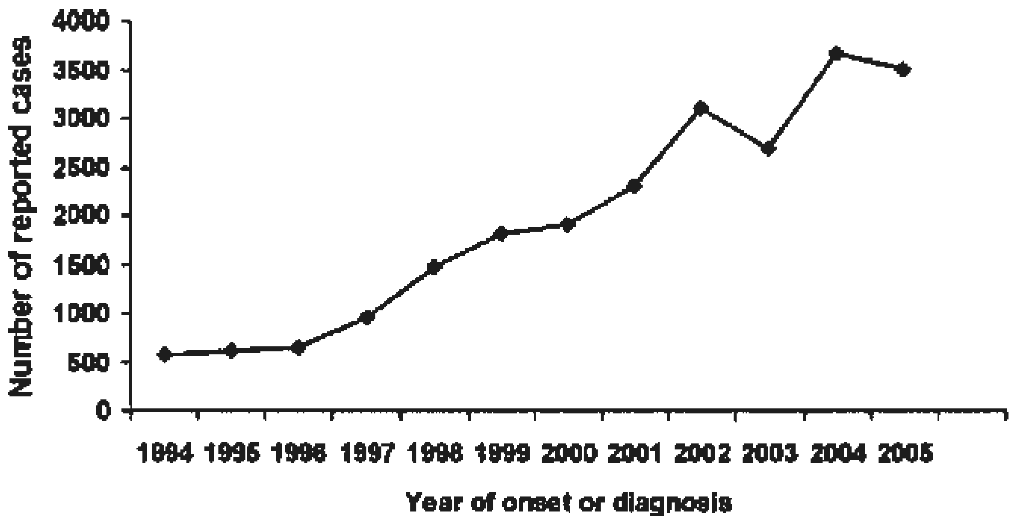

FIGURE 2. Reported cases of coccidioidomycosis, Arizona, 1994–2005. (Source: Arizona Department of Health Services, Infectious Disease Epidemiology Section.)

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 maguitude 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 being from the State of Bahia[54] and the second one from Piauí. About 15 years later, the first micro-outbreak of this mycosis in Brazil was also reported in the State of Piaui.[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.

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The costs of chronic noise exposure for terrestrial organisms

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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.

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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.

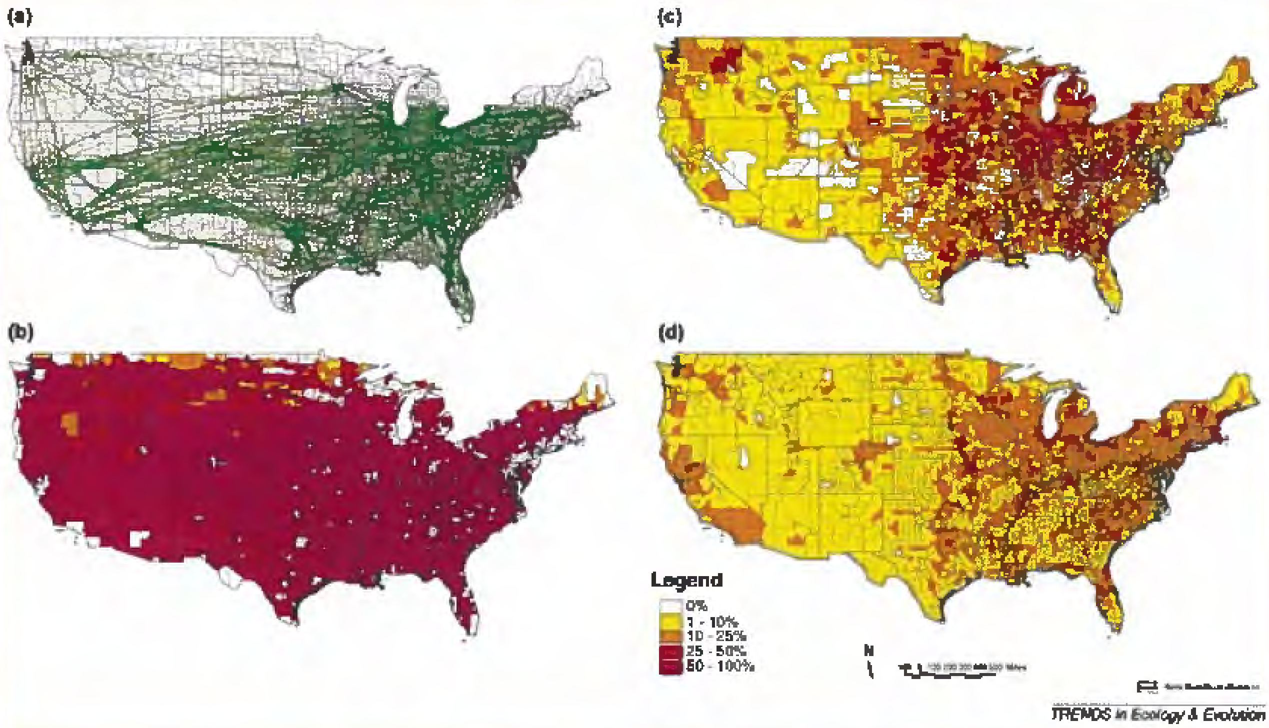

Figure I. Percent of US county areas in which transportation noise is noticeable. (a) Jet departures that occurred between 3 and 4 pm on Oct. 17, 2000, tracked to first destination. (b) Data from (a) were used to estimate the geographic extent of high altitude airway noise in the USA. The geographic 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].

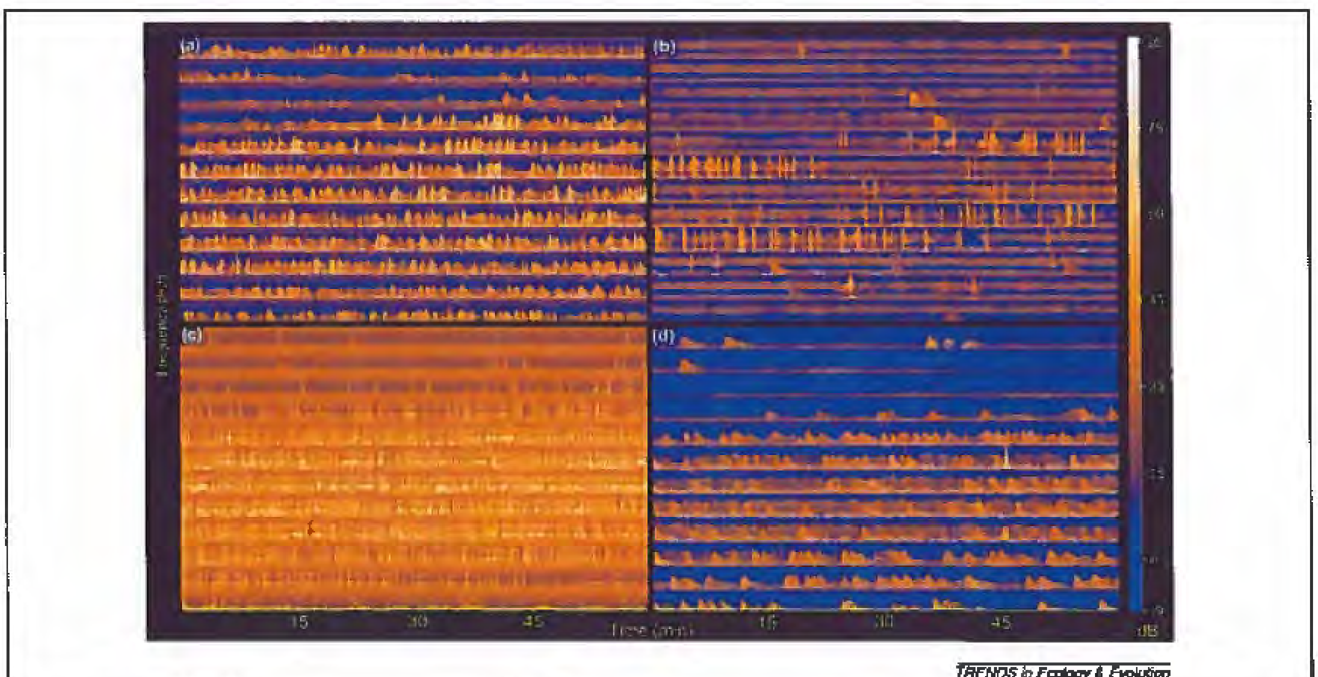

Figure 1. 24-hour spectrograms of Indian Pass in Lake Mead National Recreation Area (a), Madison Junction in Yellowstone National Park (b), Trail Ridge Road in Rocky Mountain National Park (c), and Snow Flats in Yosemite National Park (d). Each panel displays 1/3 octave spectrum sound pressure levels, 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 1:15 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.

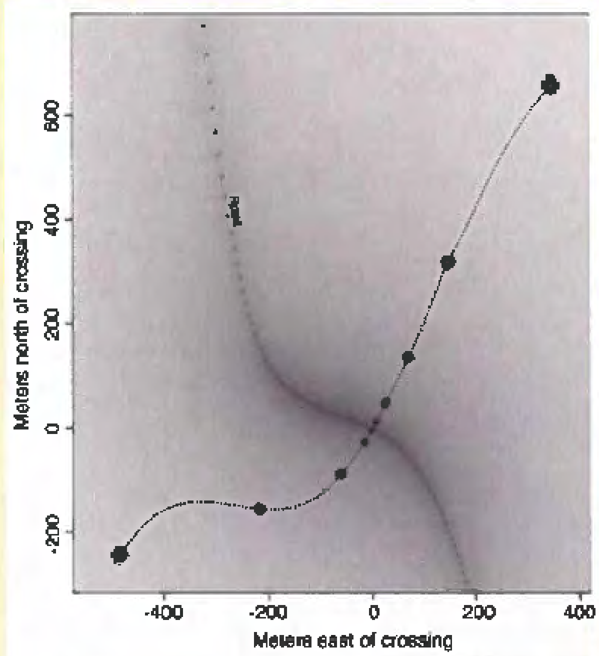

Figure I. A physical model of reduced listening area as an animal approaches a road

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.

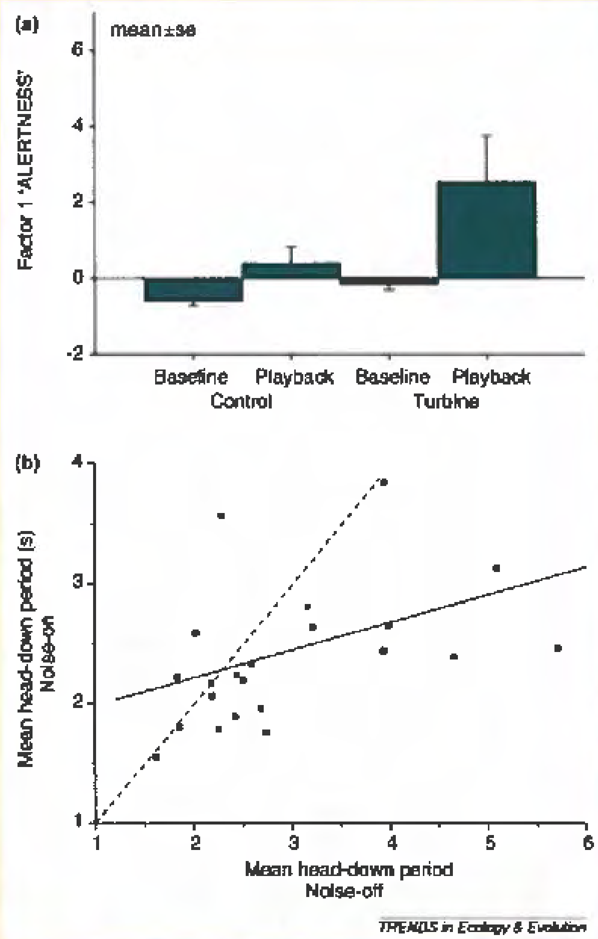

Figure I. Examples of increased vigilance behavior in noise. (a) When predator-elicited alarm calls are played back to California ground squirrels (*Spermophilus beecheyi*), adults show a greater increase in vigilance behavior at a site heavily impacted by anthropogenic noise, under power-generating 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.

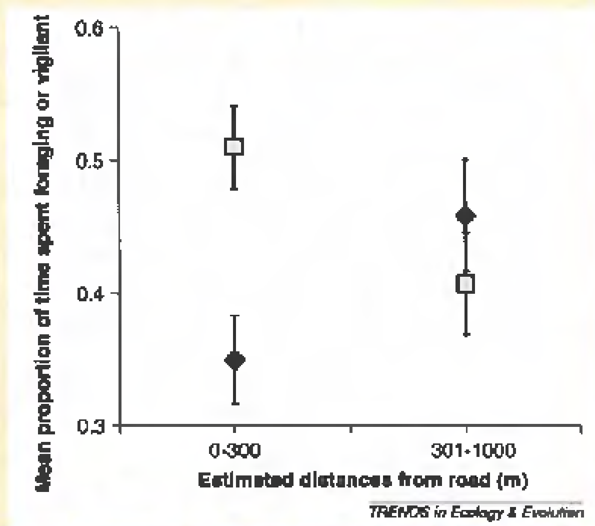

Figure II. An example of the foraging–vigilance tradeoff. Pronghorn (*Antilocapra Americana*) spend more time being vigilant (squares) and less time foraging (diamonds) within 300 meters of a road [88]. Future experiments should attempt to separate the roles of traffic as perceived threat and reduced auditory awareness on these tradeoffs. Reproduced, with permission, from Ref. [88].

[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.

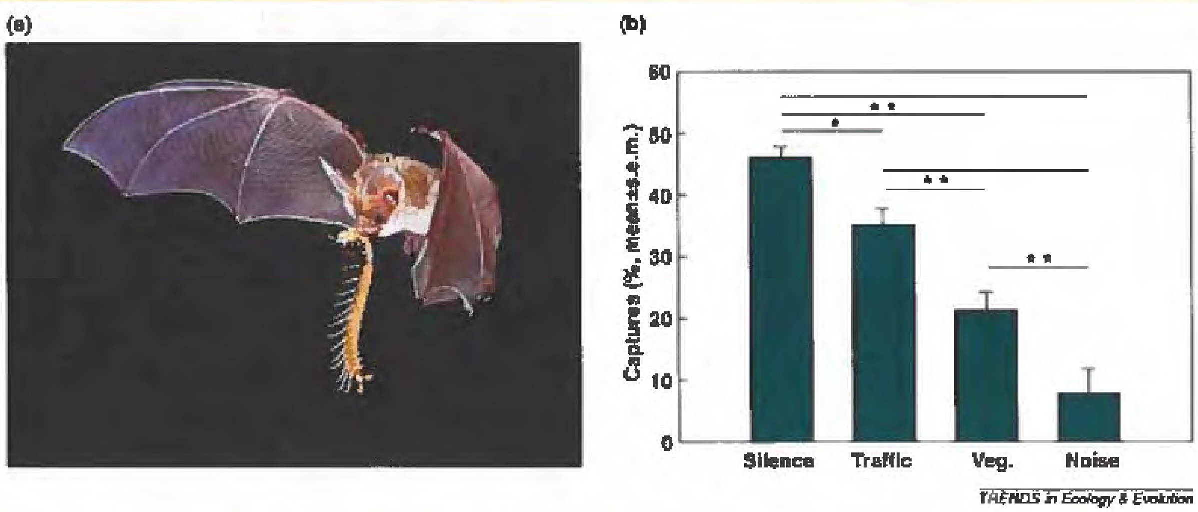

Figure I. Gleaning bats avoid hunting in noise. The pallid bat, *Antrozous pallidus* (a), relies upon prey-generated movement sounds to localize its terrestrial prey. Recent work demonstrates that another gleaning bat, the greater mouse-eared bat, *Myotis myotis*, avoids foraging in noise [74]. (b) A 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. Asterisks 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.

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Exhibit SM3

"The Effects of Noise on Wildlife"

The Effects of Noise on Wildlife

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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

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[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).

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

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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 36^{th} 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).

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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*

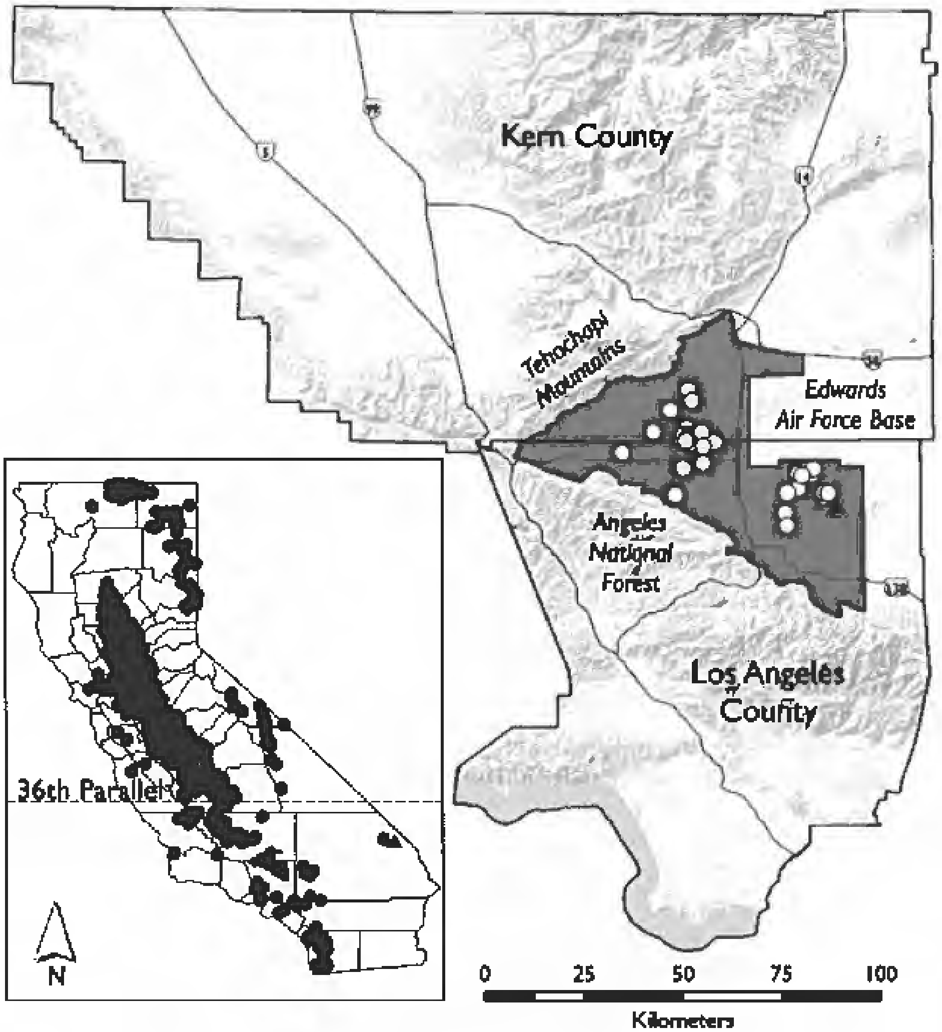

FIGURE 1. Antelope Valley Swainson's Hawk nesting locations and study area (1979–2022), Kern and Los Angeles counties, California. Gray shading, the study area; black triangles, 1978–1979 nest locations (K. Garrett pers. comm., Bloom 1980); black dots, all California nest locations outside of the study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

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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.

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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?

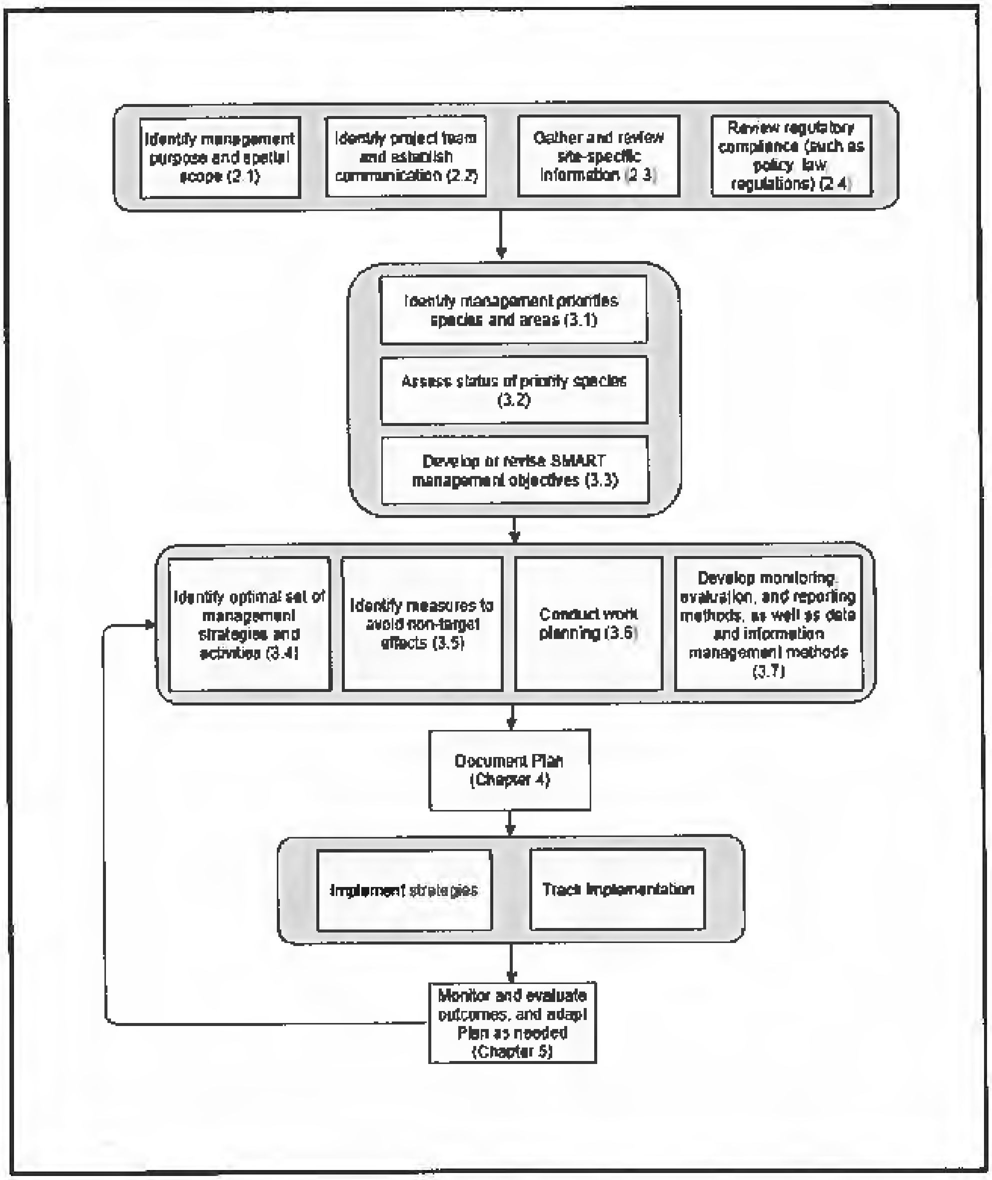

Figure 1. Strategic and adaptive invasive plant management cycle. Numbers in parentheses refer to sections of the Guide where information on that topic is located.

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 for 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.

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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 aud 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.

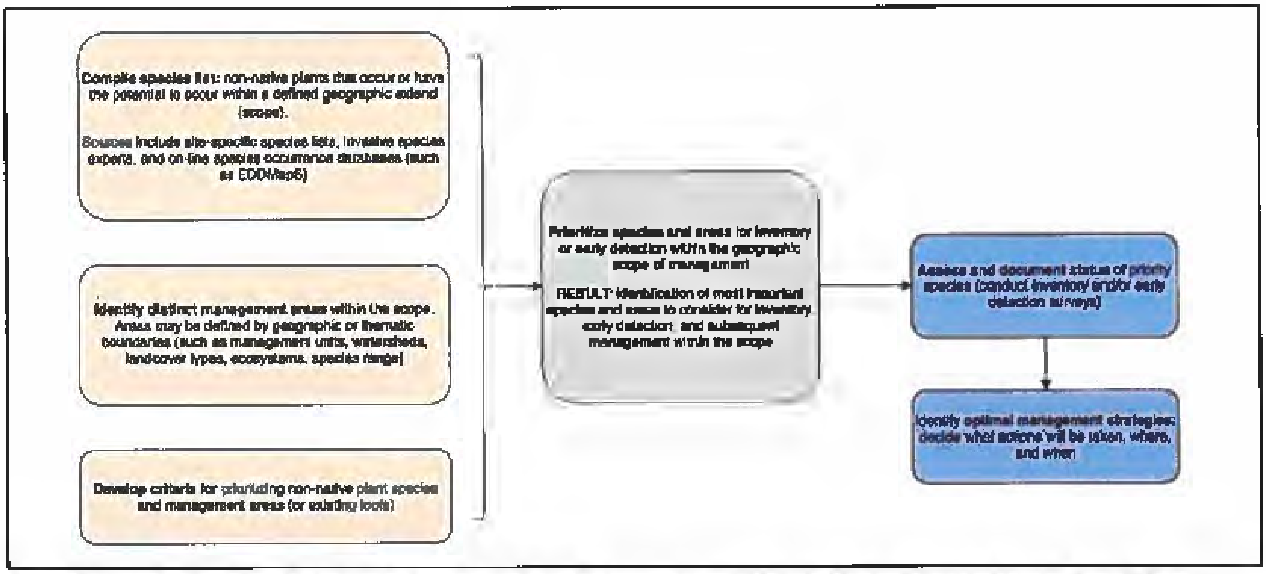

Figure 2. Generalized work flow for prioritizing species and areas for management. Initial prioritization informs what species and where inventories or early detection surveys should be focused and more generally where management efforts should focus. Subsequent inventory or early detection surveys provide 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://ecar.org/articles/cip_winter2002v2n1_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.

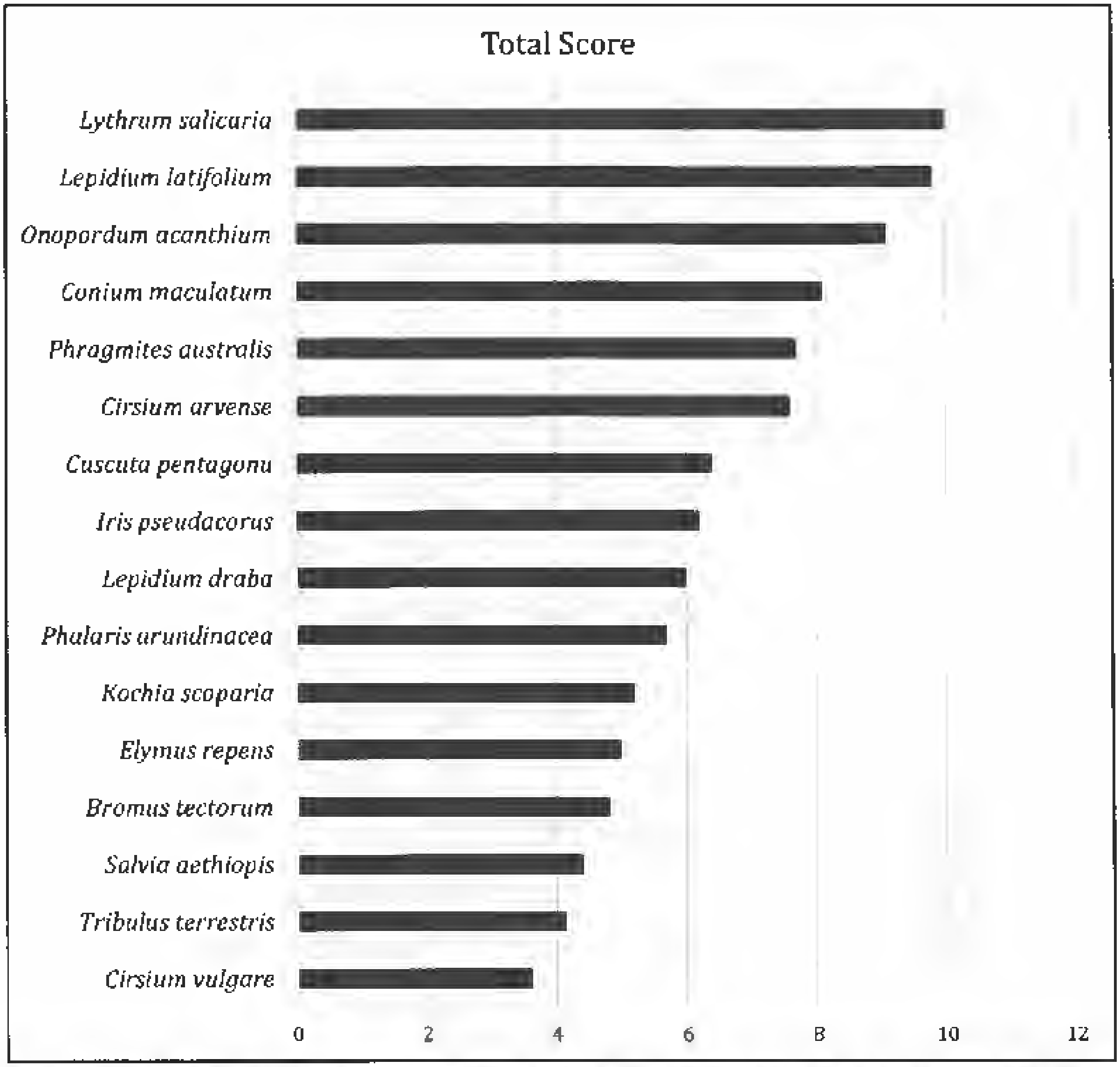

Figure 3. Invasive plant species prioritization results for Lower Klamath and Tule Lake National Wildlife Refuges: species present on-refuge (USFWS in prep.). The larger the total score, the higher the priority for management. Species prioritized using the Invasive Plant Inventory and Early Detection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

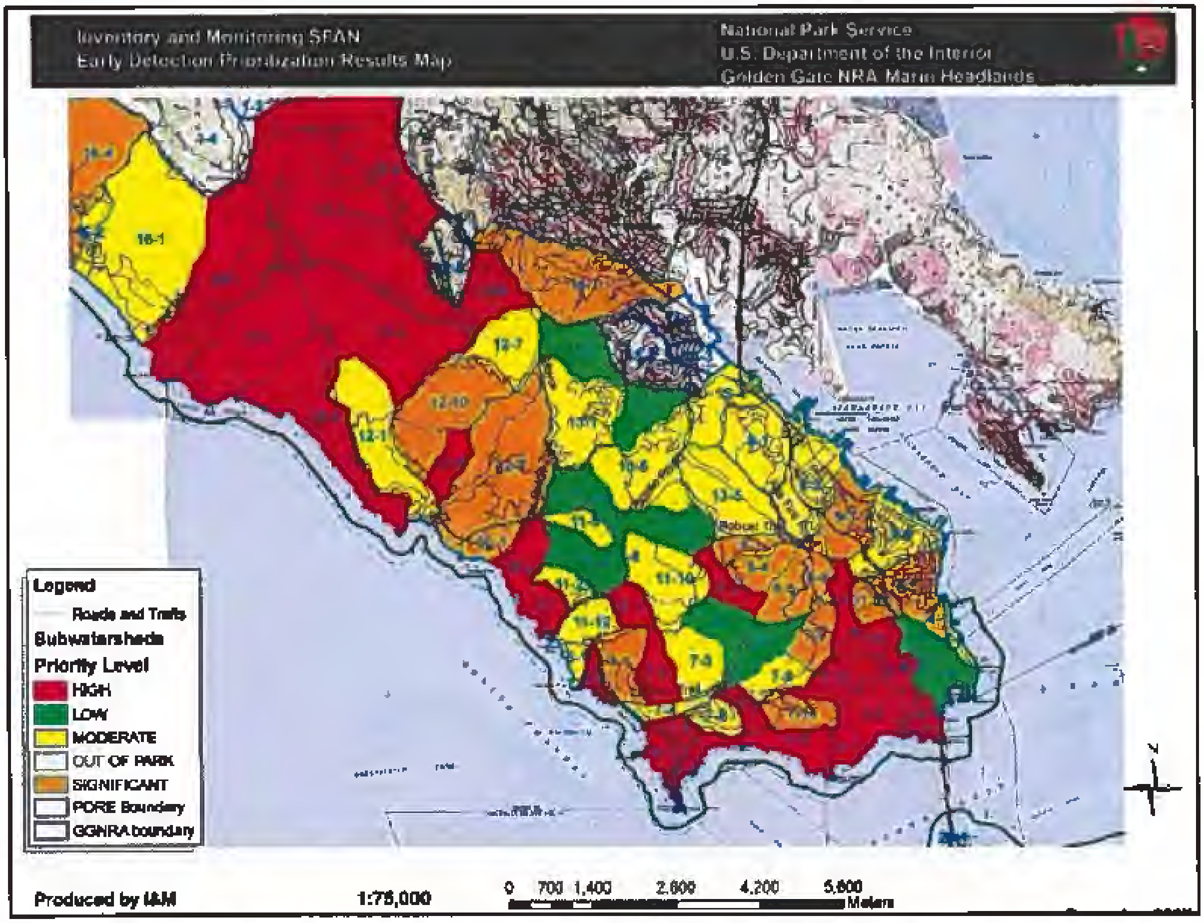

Figure 4. Map of prioritized areas (subwatersheds) for invasive plant early detection in the Golden Gate National Recreation Area, Marin Headlands, California. Hydrologic units (at a variety of scales) were used to define areas. Subwatershed prioritization criteria included abundance of rare or at-risk 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.

Aerial invasive plant survey

CREDIT: Wildlands Conservation Science, LLC.

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.).

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.

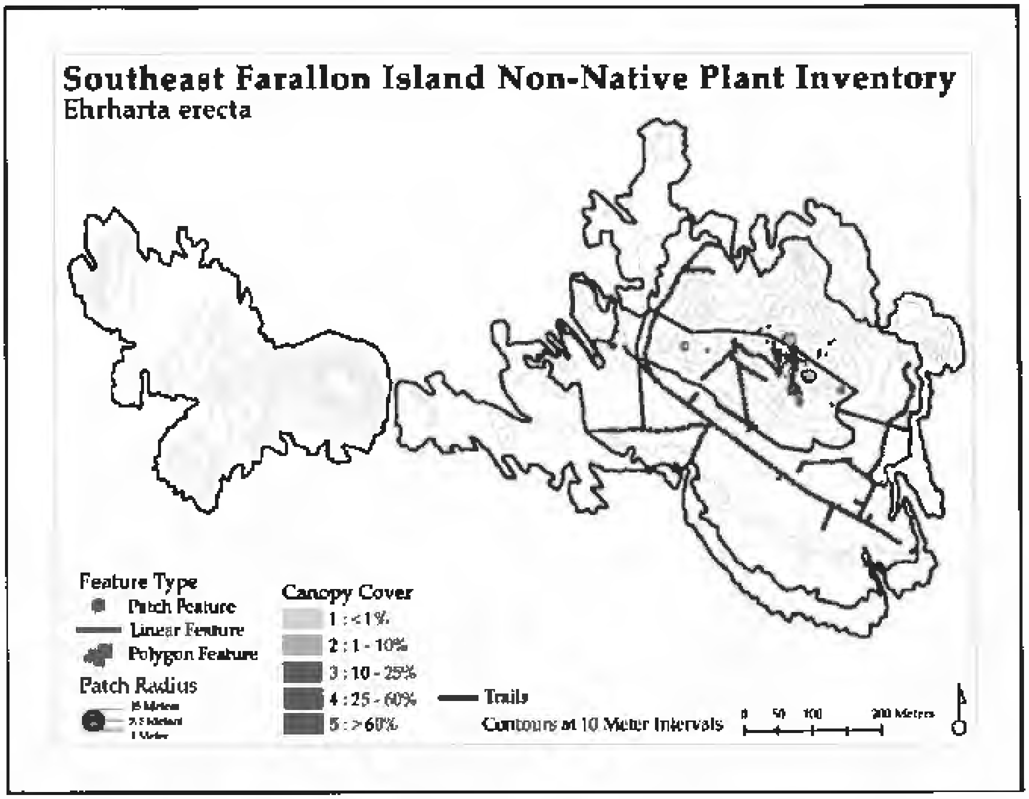

Figure 5. Farallon Island National Wildlife Refuge invasive plant inventory map: *Erharta erecta*. Results of island-wide inventory using field-based mapping methods. Source: Holzman et al. 2016.

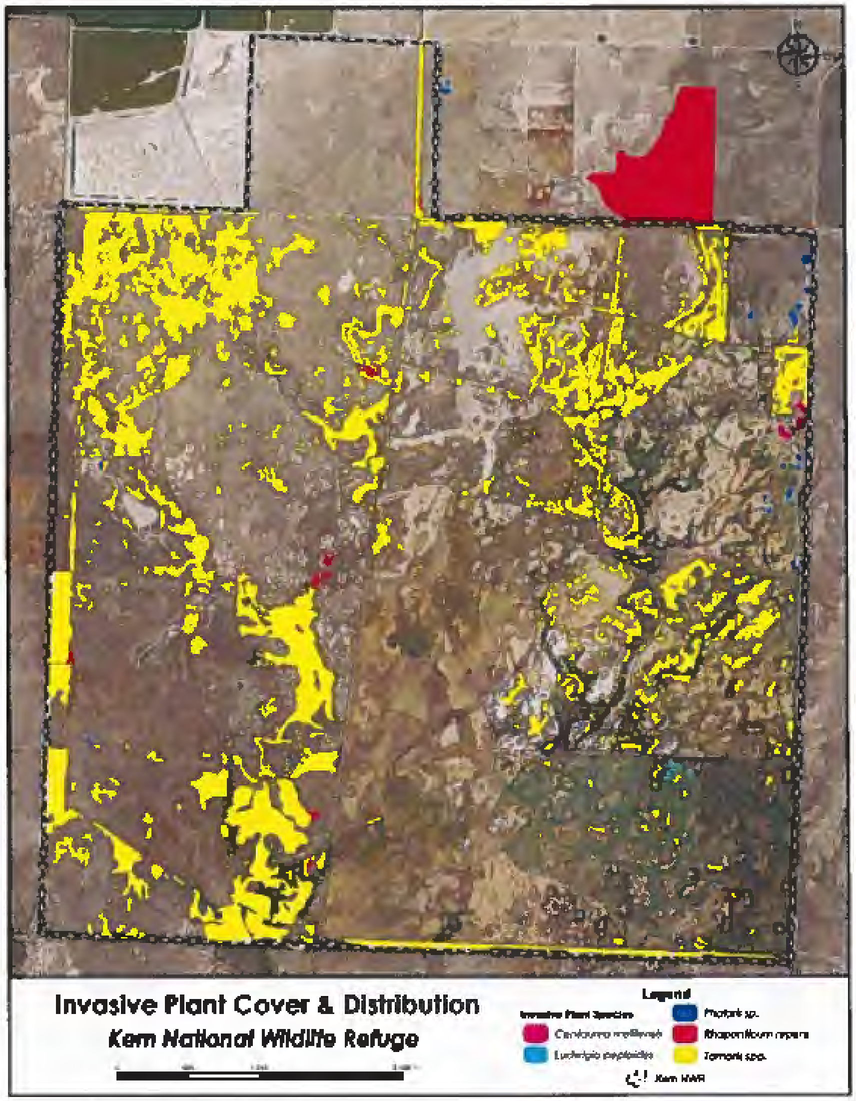

Figure 6. Kern National Wildlife Refuge invasive plant inventory map. Inventory conducted using aerial (helicopter) field-based mapping methods. Source: Ball and Olthof 2017.

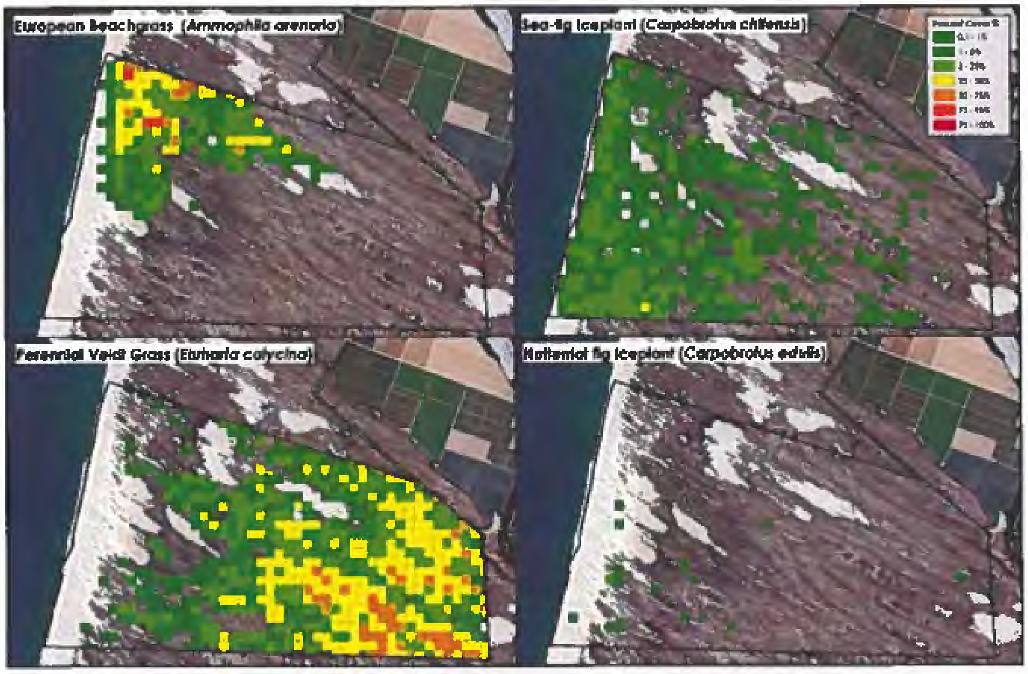

Figure 7. Guadalupe-Nipomo Dunes National Wildlife Refuge invasive plant inventory map. Inventory conducted using aerial (helicopter) grid-based mapping methods. Source: Bell and Olthof 2017.

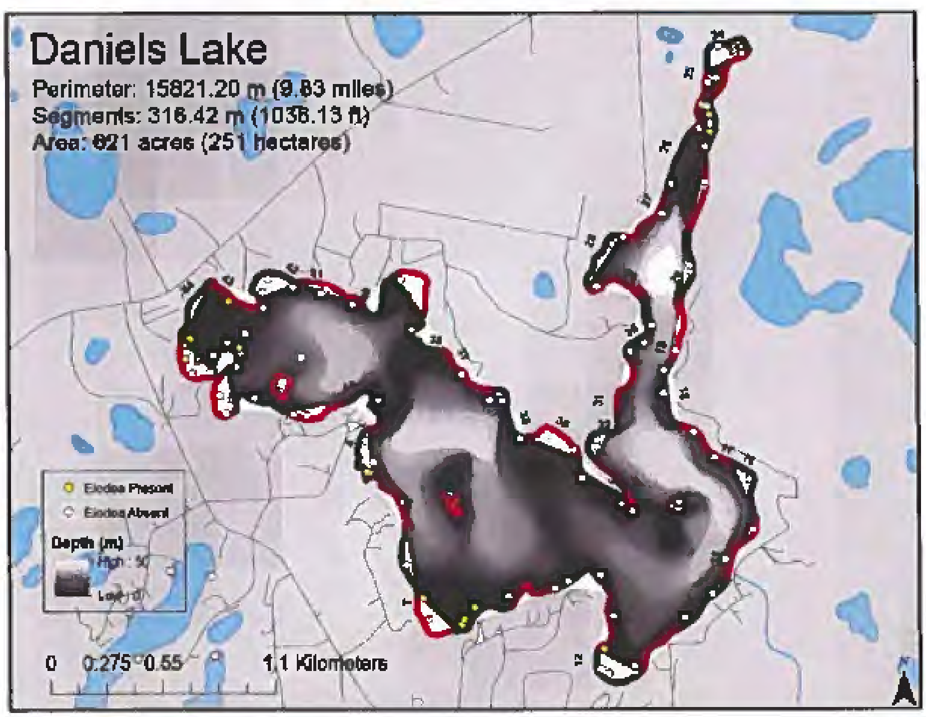

Figure 8. Results of early detection surveys for the aquatic plant Elodea (cross between *E. canadensis* and *nuttallii*) at Daniels Lake, Kenai Peninsula, Alaska. Field-based survey. Source: Bella n.d.

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 summarize 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 addition, appendix A lists other resources for understanding invasive plant ecology, imagery, and management techniques (prevention and control). Appendix B points to publicly available Plans and the types of information they contain; these Plans serve as examples of information discussed in this section.

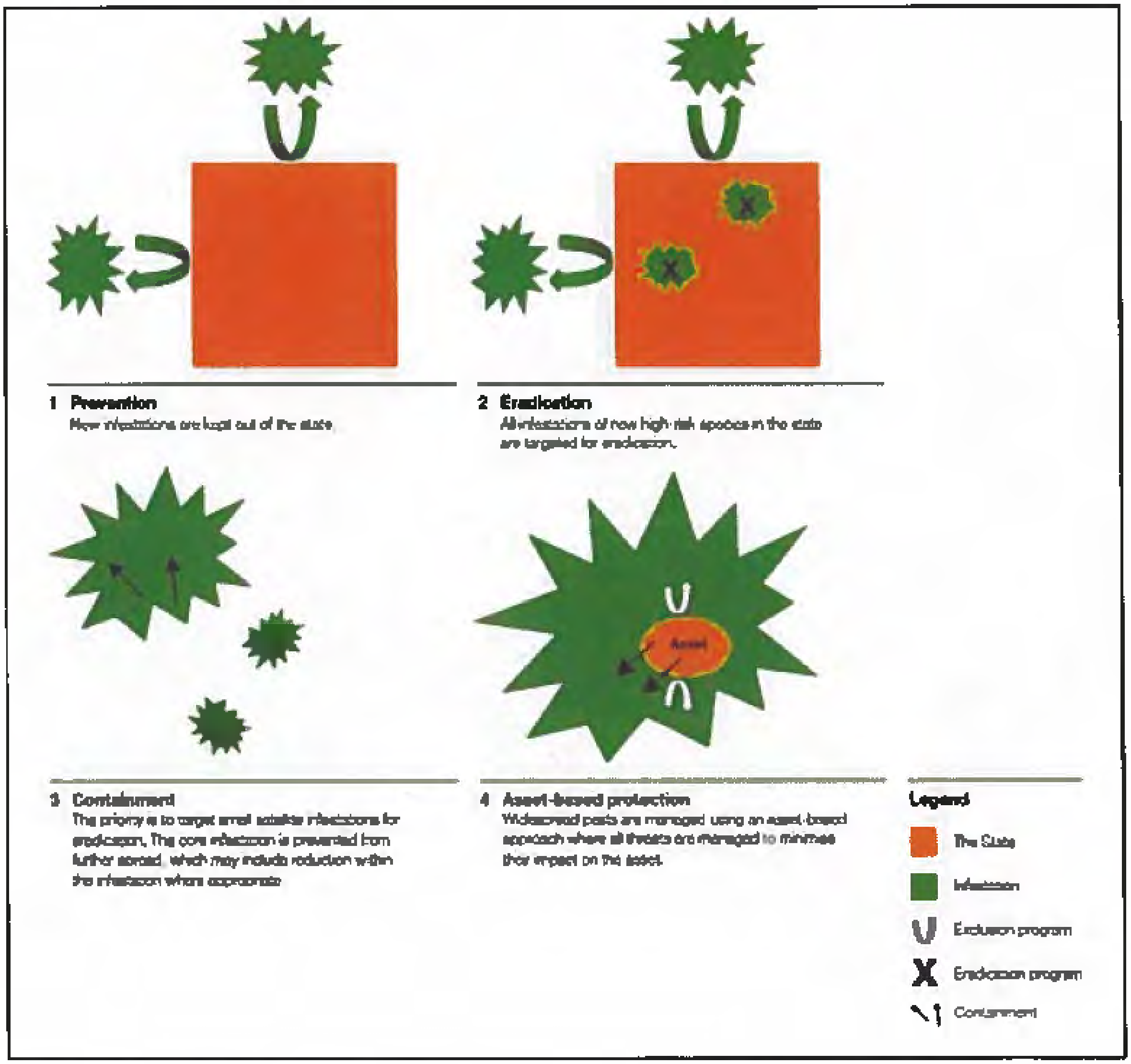

Figure 9. Approaches to invasive plant management at different stages of invasion. Source: Agriculture Victoria 2002.

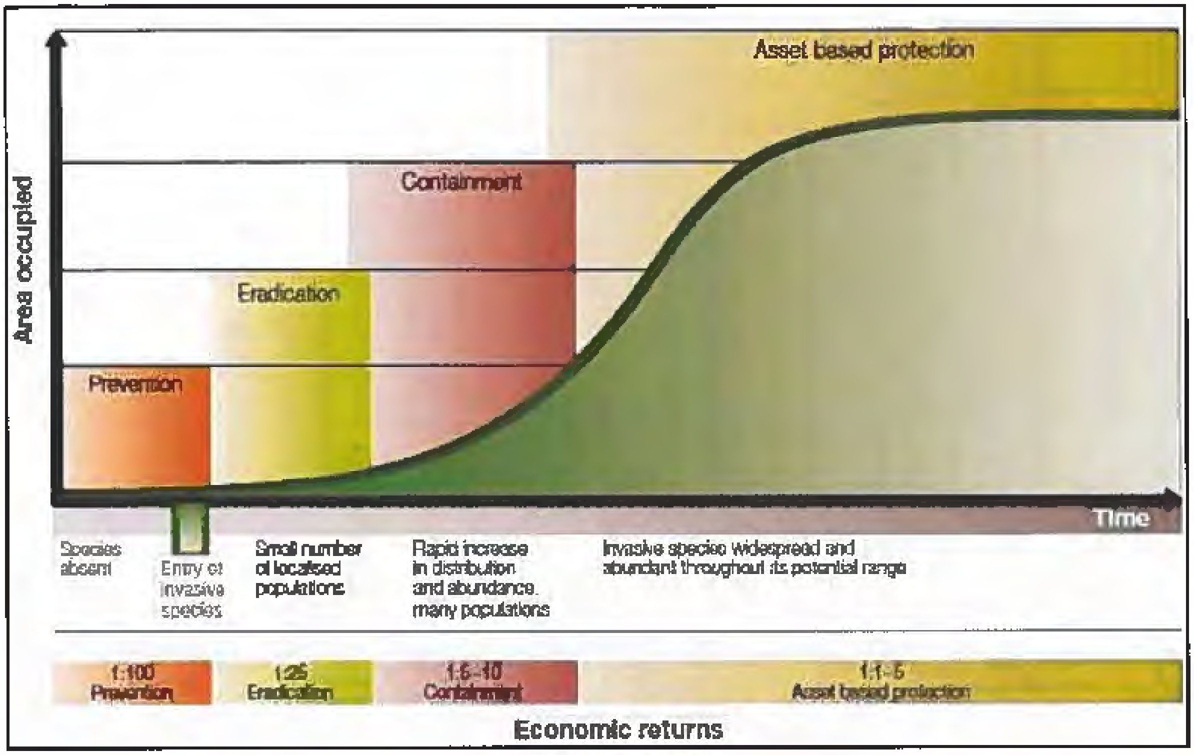

Figure 10. Generalized species invasion curve (the S-curve) and associated management approaches and cost-benefit ratios as area occupied increases. The amount of benefit for every dollar spent decreases as the area occupied increases. Source: Agriculture Victoria 2002.

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.

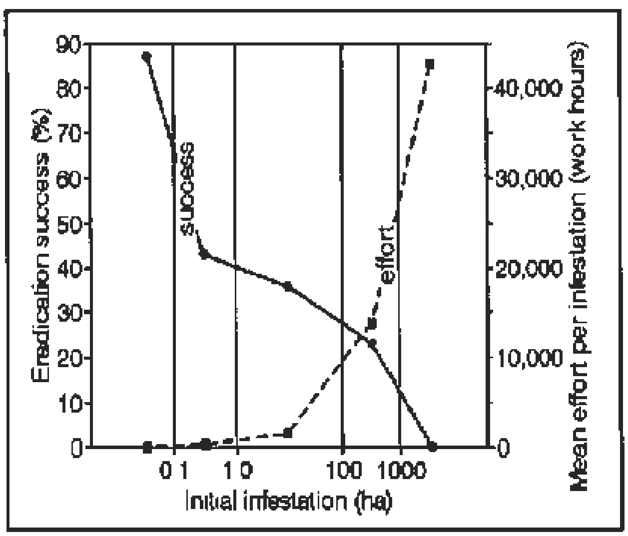

Figure 11. The dependence of the eradication success (%) and the mean eradication effort per infestation (work hours) on the initial size of infestations. Source: Rejmanek and Pitcairn 2002.

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/dd9jwo1ml8vttq9527zjhek99qr/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.

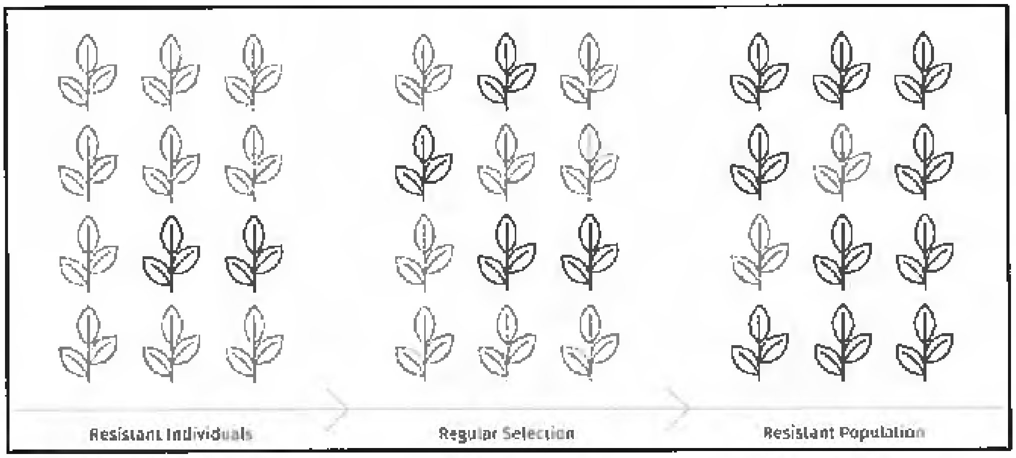

Figure 12. The process of selection for herbicide resistance. Resistance individuals (blue) increase in number over time as a result of herbicide selection pressure. Source: USA Herbicide Resistance Action Committee 2018.

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.

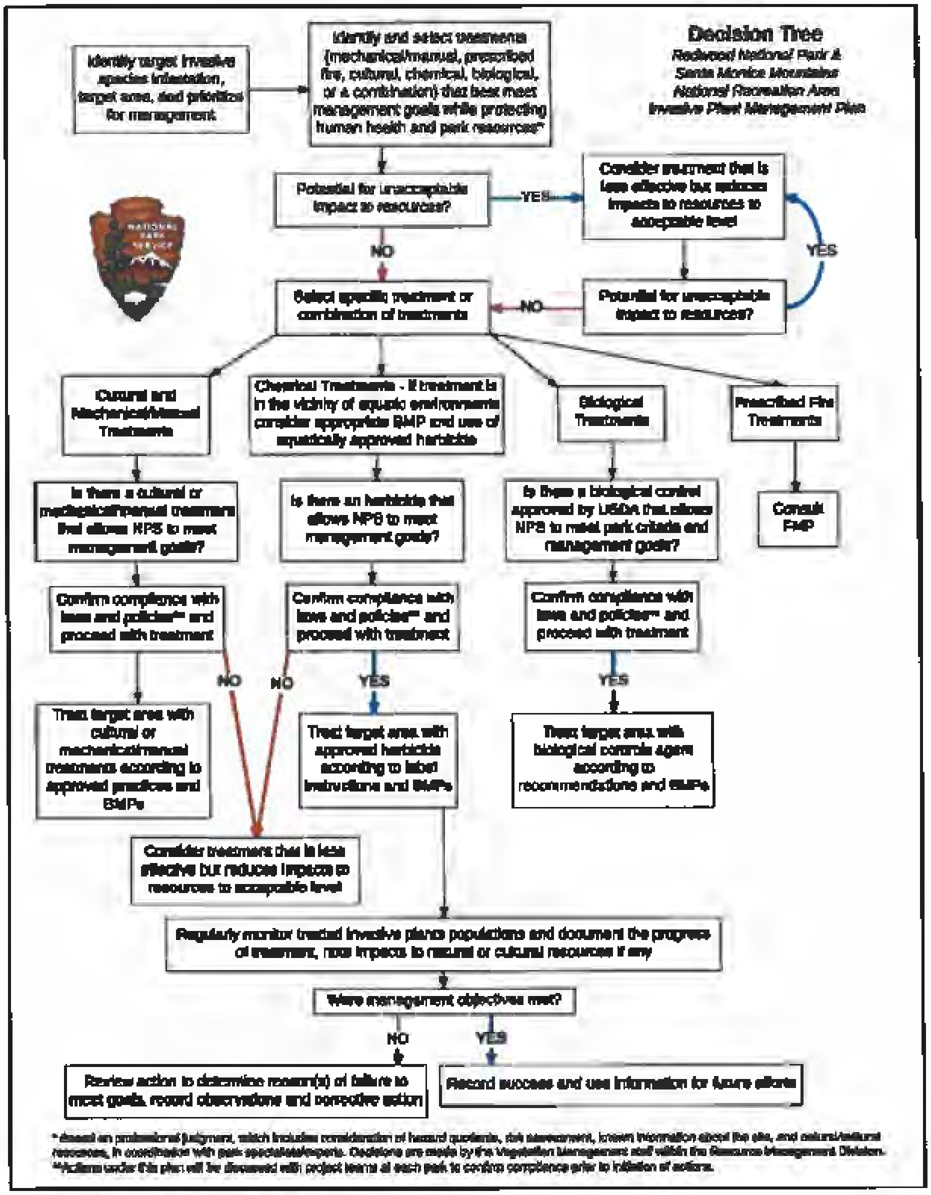

Figure 13. Invasive plant management decision tree for Redwood National Park and Santa Monica Mountains National Recreation Area. Source: National Park Service 2017.

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.

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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

Example of a spatial scope map: Kenai National Wildlife Refuge

SOURCE: USFWS

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.

4.1.3 Conservation Assets and Goals

Conservation Assets

Channel Islands fox
Urocyon littoralis

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

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.

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 evalnate 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).

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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 evaluate 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).

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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 directors 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. (2003). *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.

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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?

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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

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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 observatioos 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 unknowo. 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 <u>critical</u> 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 differences 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 impacts 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 resources 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 Torroise 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 dust (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, but the results are probably generalizable.

The finding of "hundreds of...tortoise shells" (with no indication of how long the tortoises had been 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 reduce 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 seasnn 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 nitrogeu 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 (Loftin 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 tussocks 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*, and *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: *Phacelia tanacetifolia*, *Thelypodium lasiophyllum*, and *Erodium cicutarium*.. Shrubs browsed by the sheep included *Ambrosia dumosa*, *Grayia spinosa*, *Haplopappus cooperi*, and *Acamptopappus sphoerocephalus*. 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 dumasa*), that were highly used by sheep, but provided no quantitative data. Several species eaten by sheep were also eaten by tortoises including: 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 infonnation 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 cryptoganic 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 mi² (47105 km²) 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 mi² (10663 km²): Naval Air Weapons Station ("China Lake;" 1731 mi², 4432 km²), National Training Center ("Fort Irwin;" 1016 mi², 2600 km²), Air Force Flight Training Center ("Edwards Air Force Base;" 476 mi², 1218 km²), and Marine Corp Air Ground Combat Center ("MCAGCC" or "Twentynine Palms;" 943 mi², 2413 km²).

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.

Construction, Operation, and Maintenance of Bases and Support Facilities

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 resnlts 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 explanatinn 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 doration. 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, Pauli 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 of 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 reproducing, 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.

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REVIEW OF POTENTIAL EDGE EFFECTS ON THE SAN FERNANDO VALLEY SPINEFLOWER

(*Chorizanthe parryi* var. *fernandina*)

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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.

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[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 failnre 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 promoting 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 and 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 occurreuce of some iuvasive 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 popnlations 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 polliuator 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 secoudary 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 include 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 and 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 snrface 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 assessment 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 issnes 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 bnffer 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 measures, 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 measures 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.

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Valley Fever Fact Sheet

What is Valley fever?

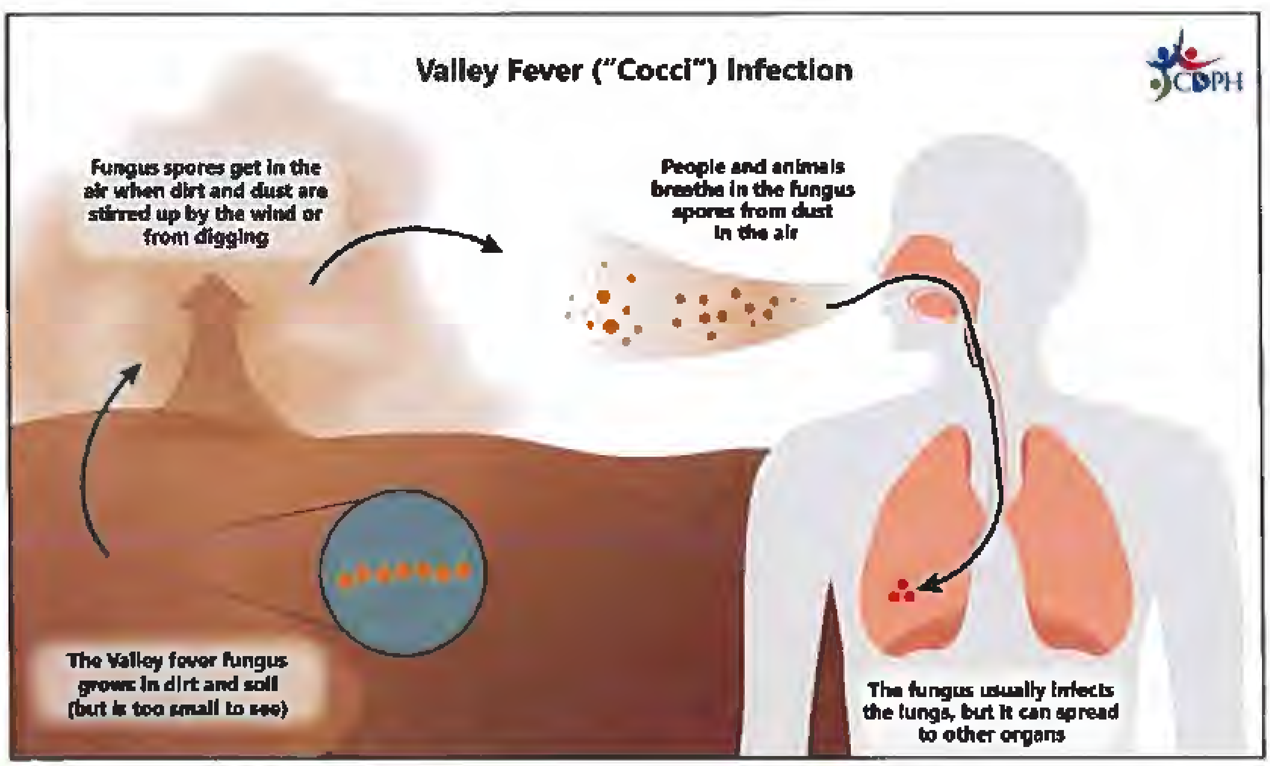
Valley fever (also called coccidioidomycosis or "cocci") is an infectious disease caused by the *Coccidioides* fungus that lives in the soil and dirt in certain areas of California and the southwestern United States. If you breathe in this fungus from dust in the air, it can infect your lungs and cause 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.

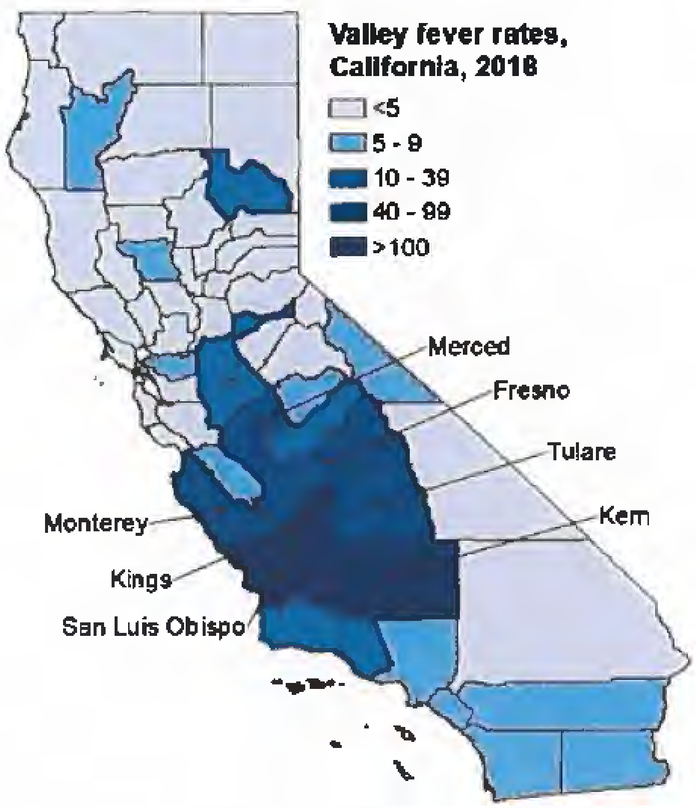
Animals, including pets, can also become infected by breathing in fungal spores. Valley fever is not contagious and cannot spread from one person or animal to another.



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 immunodeficiancy 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)

- 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

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SYMPTOMS

PREVENTION TIPS

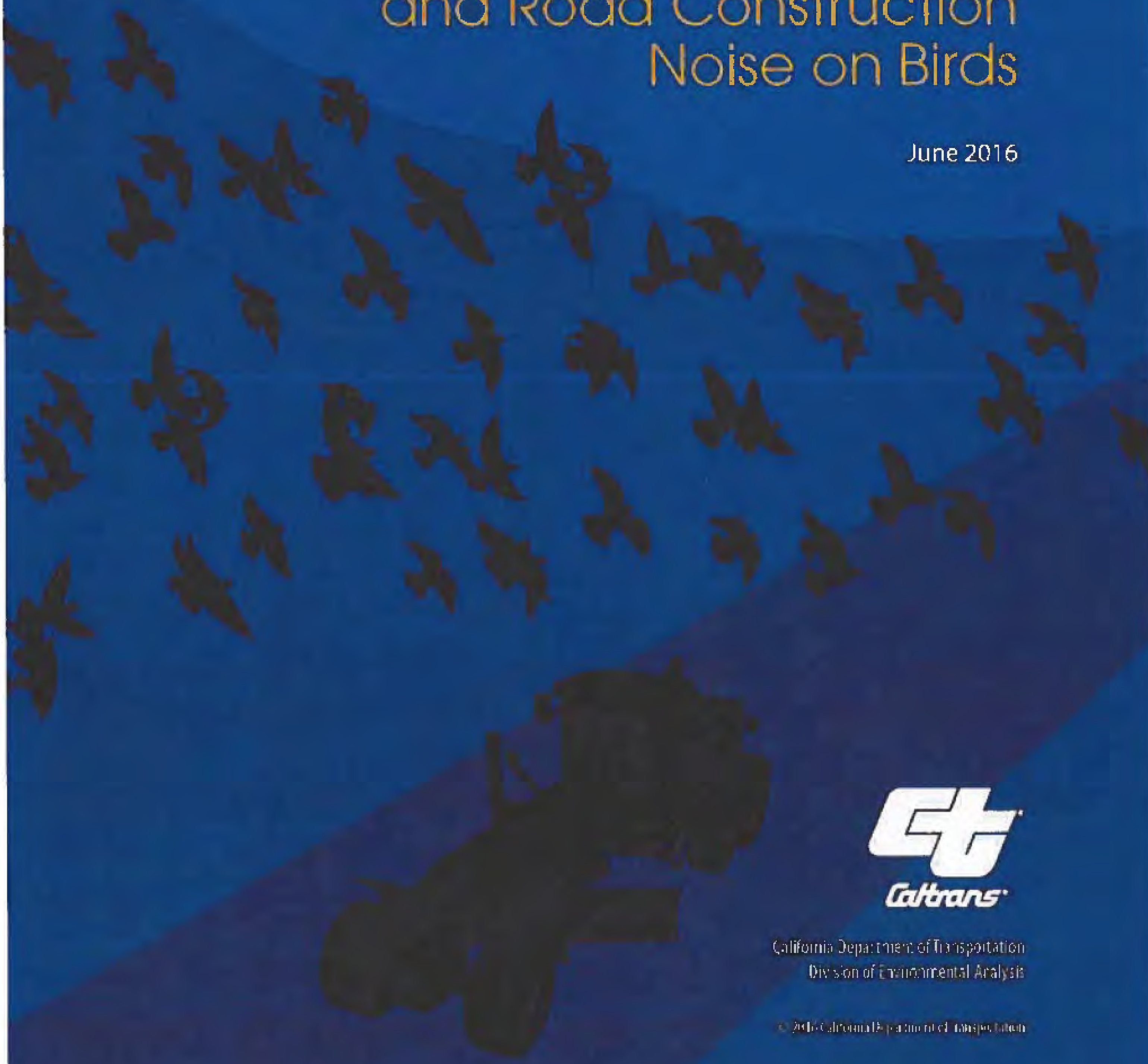
INFORMATION FOR OUTDOOR WORKERS

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



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Division of Environmental Analysis

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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

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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

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Table of Contents

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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.

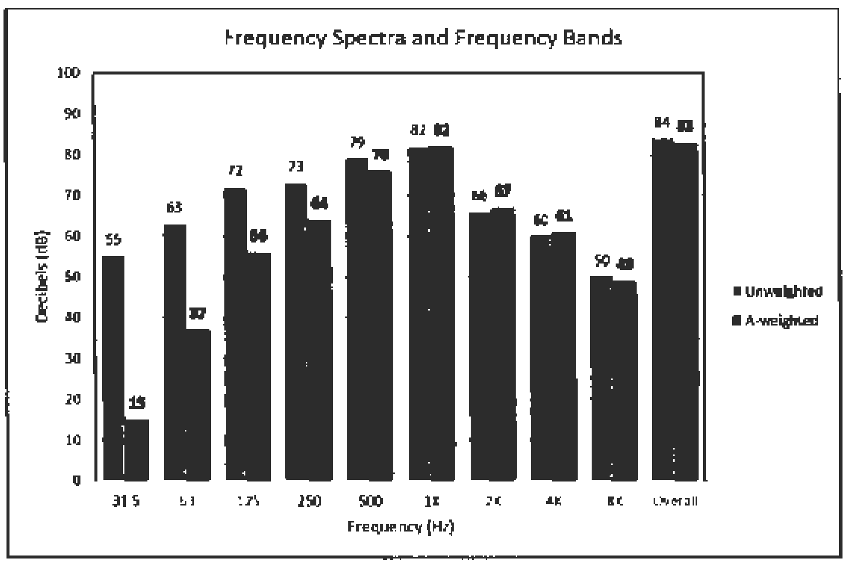

Figure ES1. Caltrans Traffic Noise Spectra Showing Differences in Unweighted and Weighted Spectra and Overall Levels[1]

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.

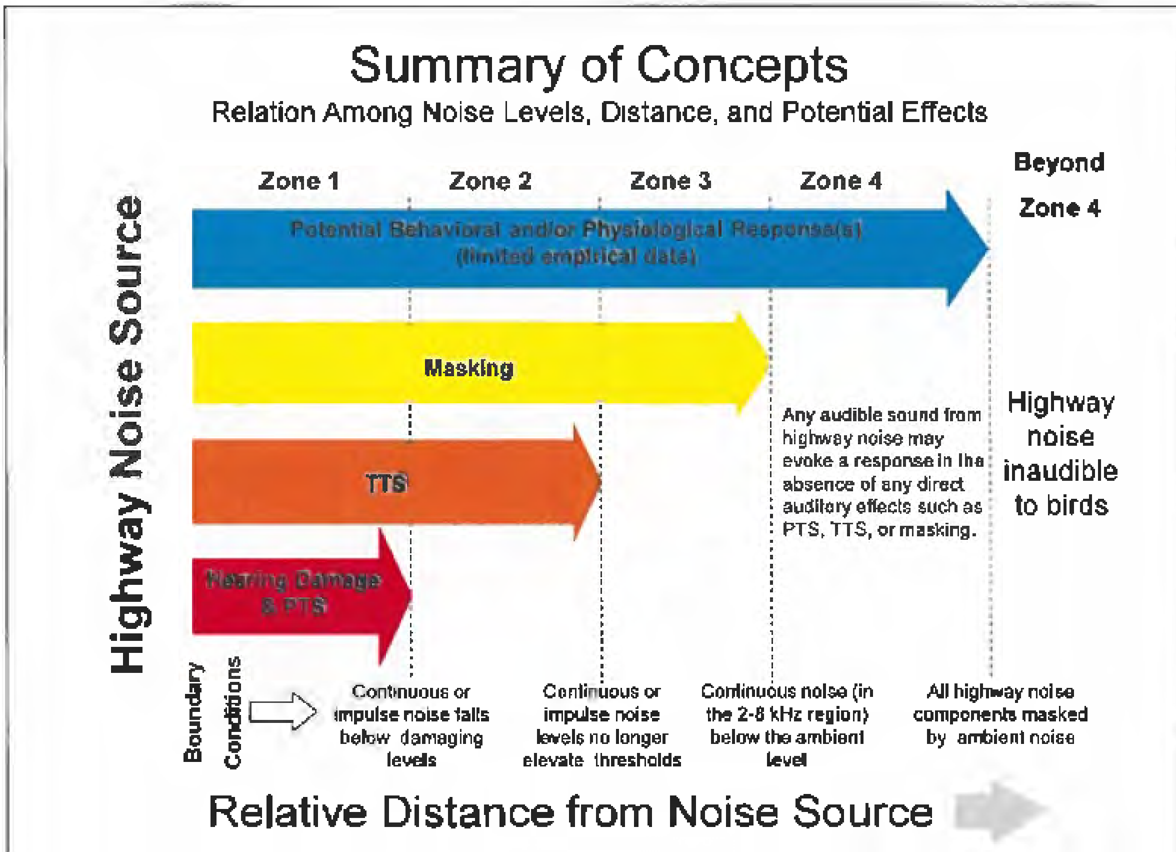

Figure ES2. Effects of Traffic and Construction Noise on Birds

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.

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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 continuons 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).

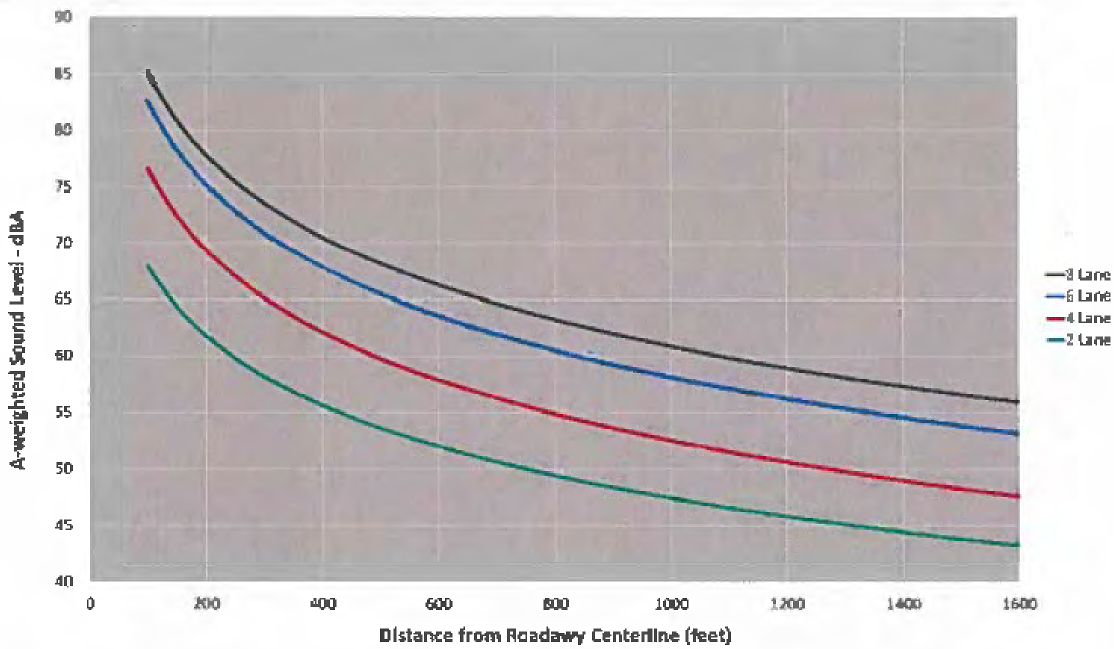

Figure 1. Typical Roadway Noise Levels as a Function of Distance
Data based on traffic conditions listed in Table 1

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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

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[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.

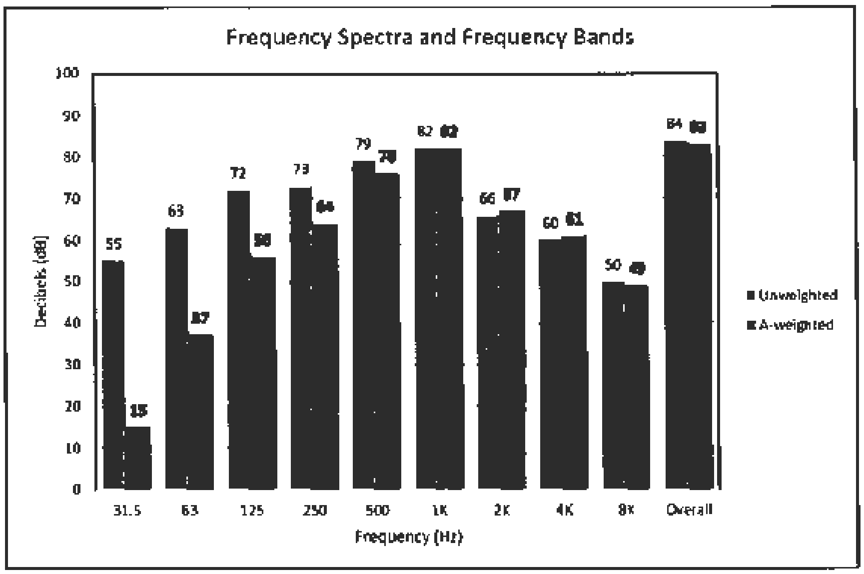

Figure 2: Caltrans Traffic Noise Spectra Showing Differences in Unweighted and Weighted Spectra and Overall Levels[11]

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 gronps, 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 freqnency, which is usually in the region of 2–4 kHz (Dooling, 1980; 1982; 1992; Dooling *et al.*, 2000b). Noctnrnal 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.

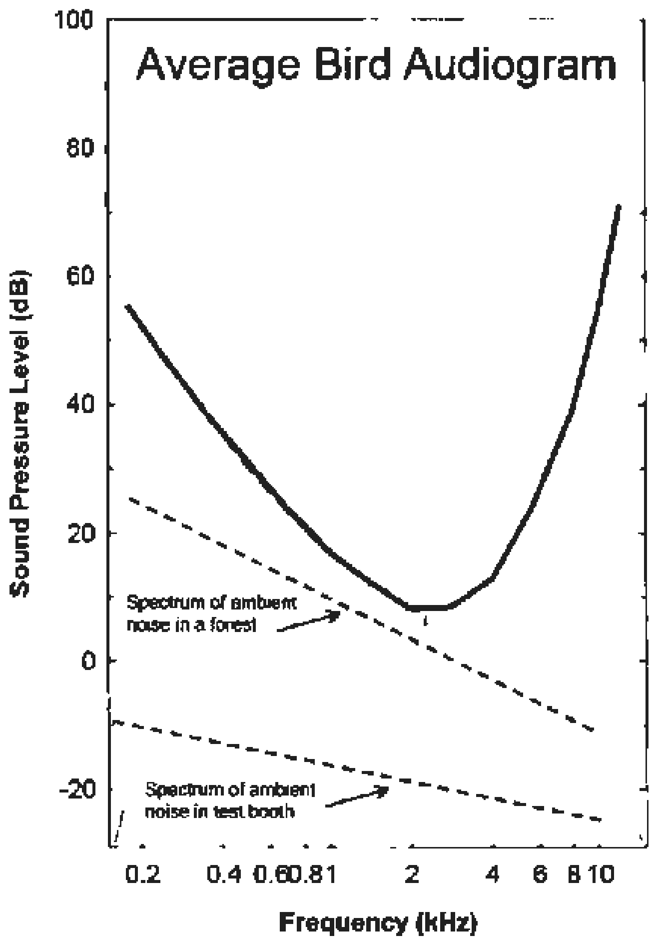

Figure 3: Bird Hearing Thresholds
Median bird hearing thresholds from 49 bird species (Appendix B measured behaviorally and physiologically in the free field in the quiet (solid line). The typical bird hears less well than humans and over a narrower bandwidth. Dotted lines show typical spectrum levels 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.

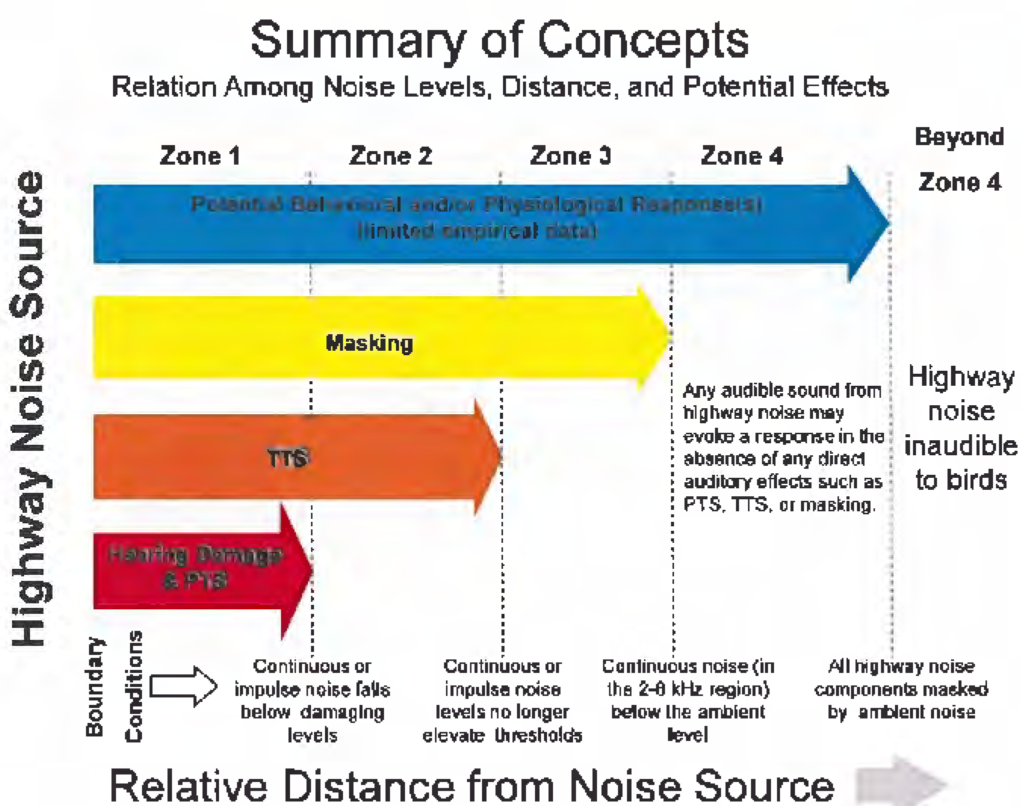

Figure 4: Potential Effects of Traffic and Construction Noise on Birds
Categories of traffic and construction noise effects on birds with distance from the source. Zone 1 is closest to the source while Zone 4 is furthest away. Sound level decreases further from the source. Note that the actual distances 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.

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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.

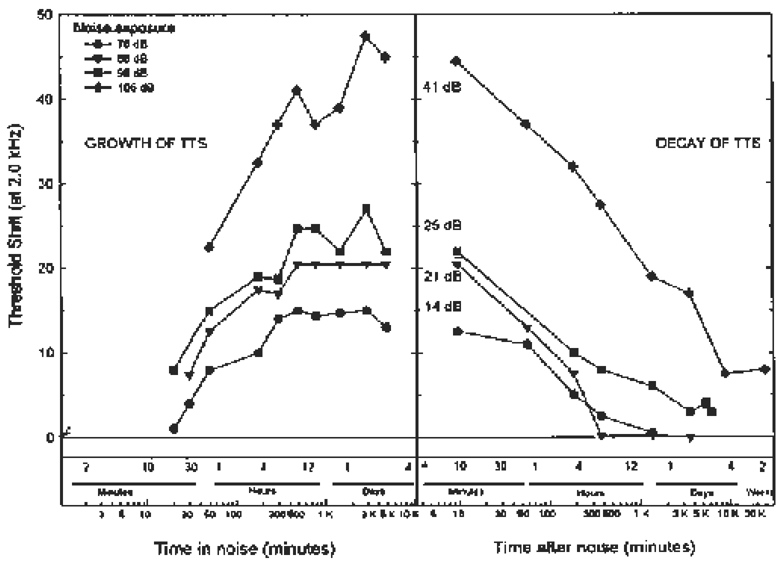

Figure 5: Threshold Shift in Birds Exposed to Noise
The growth and decay of threshold shift in four budgerigars exposed to four different levels of a one-third octave band of noise for 72 hours. Threshold shift reaches an asymptote (horizontal dashed line) after 12–24 hours regardless of the exposure 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).

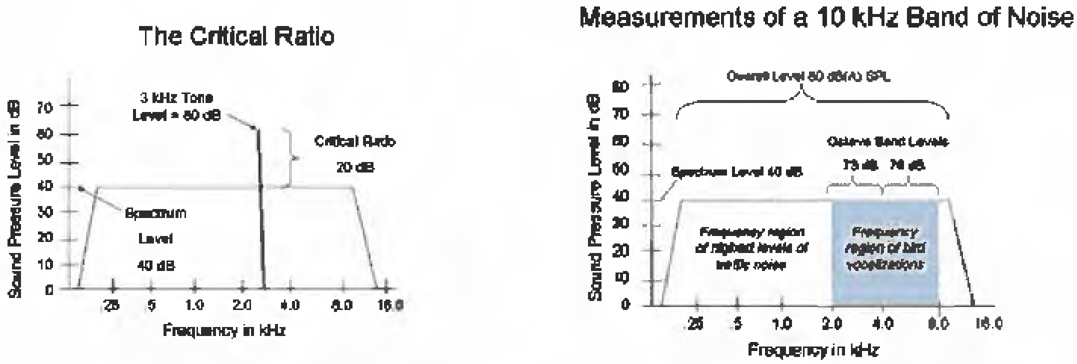

Figure 6: Avian Critical Ratios

(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.

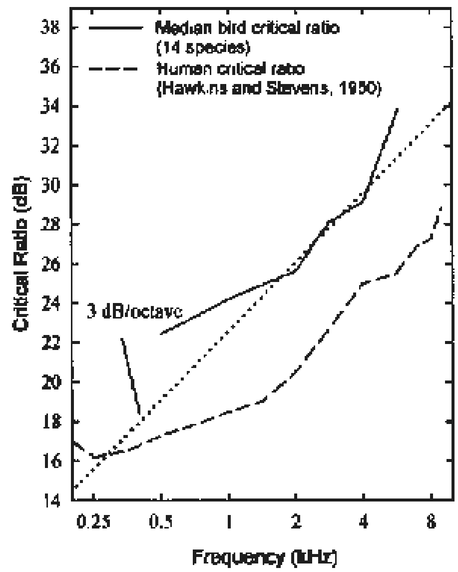

Figure 7: Critical Ratios in Birds and Humans
Median critical ratios for 14 birds (solid line) and the human (dashed line). Dotted line is a slope of 3 dB/octave. The critical ratio (s/n ratio) at threshold is about 6 dB greater in the typical bird compared to humans over the frequency range of 1–5 kHz (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.

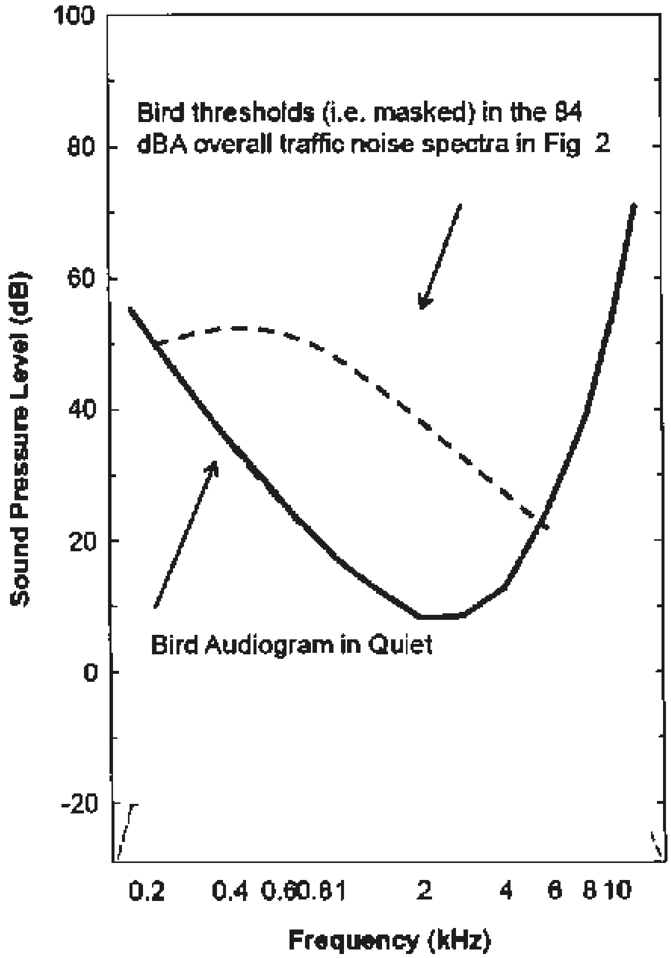

Figure 8: The Effects of Traffic Noise of 84 dBA on Hearing Thresholds of the Typical Bird
The effects of traffic noise illustrated earlier in Figure 2 raises a bird's threshold. The solid line shows the auditory thresholds (audiogram) in the quiet. The dashed line above the audiogram shows elevated 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):

$$\text{Drop} = 20 \cdot \log \quad \frac{d_{mc}}{d_o} + \frac{EA \cdot d_{mc}}{100}$$

- Drop: the amount of signal attenuation from source intensity to that at threshold;
- d_{mc}: the maximum communication distance;
- d_o: 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.

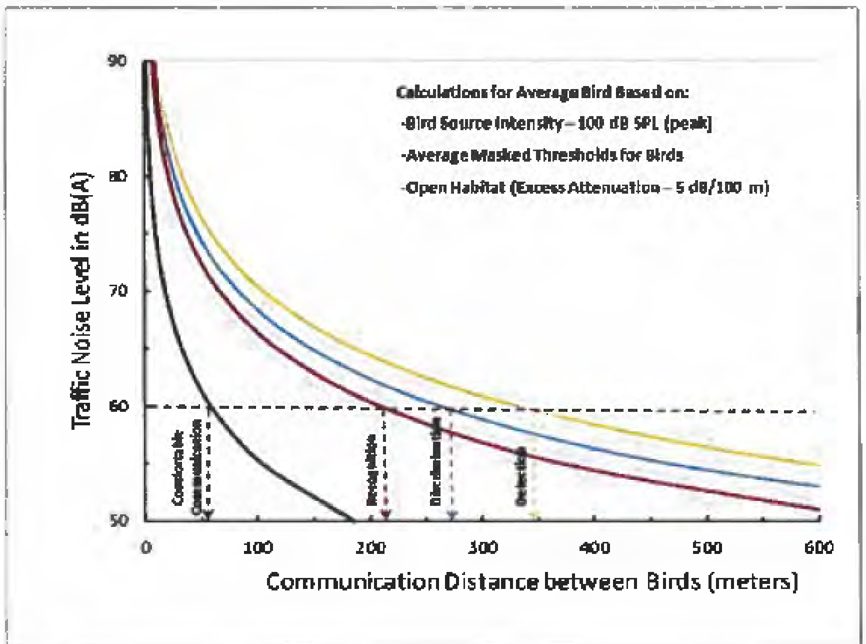

Figure 9: The Effects of Anthropogenic Traffic Noise on Four Different Behaviors Based on the Average Bird Critical Ratio Function

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.

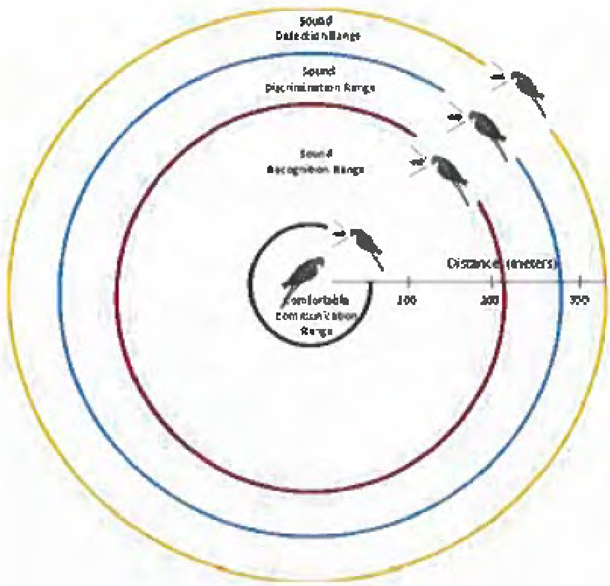

Figure 10: Diagrammatic Representation of Bird Communication

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 tn 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.

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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

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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	*Lagonosticta 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)

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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 sound. 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 intensiry (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 wher 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, playgrou public meeting rooms, public or nonprofit institutional structures, radio studios, recording studios, recreation area Section 4(f) sites, schools, television studios, trails, and tr 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 implementation of 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.

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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 informatinn 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 oo 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 aud 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.

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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.

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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.

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