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A Summary of Predation by Corvids on Threatened and Endangered Species in California and Management Recommendations to Reduce Corvid Predation

by

Joseph R. Liebezeit
and
Dr. T. Luke George
Humboldt State University
Department of Wildlife
Arcata, California 95521

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CONTRACTOR

Humboldt State University Foundation
Arcata, California 95521

PRINCIPAL INVESTIGATORS AND AUTHORS

Joseph R. Liebezeit
and
Dr. T. Luke George
Humboldt State University
Department of Wildlife
Arcata, California 95521

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Common Raven and Steller's Jay at Lake Tahoe, California

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Authors:

Joseph R. Liebezeit
Humboldt State University
Department of Wildlife
Arcata, CA 95521

Dr. T. Luke George
Humboldt State University
Department of Wildlife
Arcata, CA 95521

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SUMMARY

We document the impact of corvids on threatened and endangered species in California and make recommendations for protecting listed species from corvid predation. Breeding Bird Survey (BBS) and Christmas Bird Count data have documented substantial increases in the populations of Common Ravens (*Corvus corax*), American Crows (*Corvus brachyrhynchos*) in California over the last 30 years. While populations of Steller's Jays have generally remained stable over this period, BBS data indicates they have increased significantly in the Southern Pacific Rainforest region of northwestern California.

Corvids have been documented preying on the nests or young of the following threatened or endangered species in California: California Condors (*Gymnogyps californianus*), Greater Sandhill Cranes (*Grus canadensis tabida*), Western Snowy Plovers (*Charadris alexandrinus nivosus*), California Least Terns (*Sterna antillarum browni*), Marbled Murrelets (*Brachyramphus marmoratus*), San Clemente Island Loggerhead Shrikes (*Lanus ludovicianus mearnsi*), Least Bell's Vireo (*Vireo bellii pusillus*), and desert tortoises (*Gopherus agassizii*). American Crows and Common Ravens have been documented as the most important nest predators on Western Snowy Plovers and California Least Terns in several locations in California. In some cases, predation by crows and ravens has caused California Least Terns to abandon their nesting colonies for a season. In addition, predation by crows and ravens is the principal cause of nest failure for Western Snowy Plovers in many locations.

Techniques that have been employed to protect threatened and endangered species from corvid predation can be lumped into three general categories: lethal removal, behavioral modification, and habitat modification. Lethal removal is often used when immediate reduction in the corvid population is necessary and has been found to be very effective in reducing nest predation by corvids on colonies of California Least Terns. However, reductions are, at best, temporary with no carryover benefits one year after removal.

Behavioral modification involves changing the behavior of a species for a specific purpose and includes: Conditioned Taste Aversion (CTA), repellants, sterilants and effigies. CTA has shown some potential for reducing corvid predation on threatened and endangered species but more research is needed. Repellants and effigies generally have only very short-term effects and do not appear to be an effective means of deterring corvid predation. The use of sterilants remains largely untested with corvids.

Habitat modification includes: erecting nest exclosures, removing nesting and perch sites, restoring degraded habitat, reducing availability of anthropogenic sources of food and water, and providing subsidized food. While exclosures have been effective at reducing corvid predation on Western Snowy Plovers, it requires a large investment of time and only provides a short-term solution. Habitat restoration and reduction of anthropogenic food and water sources are attractive approaches because they have the potential for providing a long-term solution to the problem. Landfills are one of the primary sources of anthropogenic food for crows and ravens and have been implicated as a cause of their rapid increase in some locations. Research on approaches to reducing corvid access to food in landfills is currently underway.

Reducing the impacts of corvids on threatened and endangered species is a complex issue with no simple solution. Management strategies to protect particular species must be approached on a case-by-case basis. Nevertheless, some management recommendations such as limiting availability of anthropogenic food sources in locations where corvids co-occur with threatened and endangered species can, in some cases, be implemented quickly and with relatively little cost. More drastic measures, such as lethal removal, should be considered in extreme cases where an immediate decrease in corvids is necessary to save a population of threatened or endangered species.

INTRODUCTION

The family Corvidae (often referred to as corvids) is composed of over 100 species of birds including crows, ravens, jays, magpies, and nutcrackers. They are one of the most successful avian groups and occur throughout the world with species represented on all continents, except Antarctica. Corvids are conspicuous members of the avifauna of most ecosystems in North America and, in many cases, play key roles in the biotic community. For instance, Piñon Jays (*Gymnorhinus cyanocephalus*) and Clark's Nutcrackers (*Nucifraga columbiana*) are important dispersal agents for pine seeds in their respective communities. Moreover, because most corvids are omnivorous and employ many foraging strategies including predation, scavenging, and kleptoparasitism (Boarman and Heinrich 1999), they often affect many other species in their communities.

Before European colonization of North America, it is likely that corvids occurred at low densities in most communities. However, as a result of their ability to adapt and thrive in human-altered landscapes (Marzluff et al. 1994), many corvid populations are dramatically increasing in western North America, including California (Robbins et al. 1986, U.S. BLM 1990, Marzluff et al. 1994). Because corvids are effective predators on the nests and young of some threatened and endangered species, there is concern that increases in corvid populations are having a negative impact on the populations of some listed species. This report was prompted by the need to develop an ethical yet realistic means of dealing with the negative impacts of corvids on threatened and endangered species.

The purpose of this report is to review literature on the biology and management of corvids and make recommendations for protecting threatened and endangered species from corvid predation. In the following sections, we:

1. Review literature on the life histories of three corvid species, the Common Raven (*Corvus corax*), American Crow (*Corvus brachyrhynchos*), and Steller's Jay (*Cyanocitta stelleri*), that are believed to be having the most impact on threatened and endangered species in California.
2. Summarize recent population trends of the three focal corvid species in California.
3. Evaluate evidence documenting corvid predation on threatened and endangered species and the importance of corvids as predators on those species.
4. Describe management techniques that have been used to control corvids and evaluate their effectiveness.
5. Provide management recommendations to reduce corvid impacts on listed species.

Our goal is to provide state, federal, and private wildlife managers with up to date information on corvid impacts and management options. However, we leave it up to local wildlife managers to decide how to implement these actions and to tailor them to their specific situation. Likewise, we hope this work will stimulate further research to fill in gaps of knowledge that are needed in order to develop the most effective management techniques.

CORVID LIFE HISTORIES

Members of the family Corvidae are in the order Passeriformes ("perching birds"), which includes the majority of all bird species and represents the most recently evolved avian lineage. Corvids are medium to large-sized birds and are characterized by an all purpose chisel-like bill and large, strong legs. Their nostrils are usually covered with nasal bristles or plumes. Plumage coloration and pattern ranges from simple (e.g. American Crow) to highly ornate, such as the Magpie Jay (*Calocitta formosa*).

Corvids are known for their bold, aggressive demeanor and their scolding vocalizations. Despite their abrasive calls they actually have a substantial vocal repertoire and are excellent mimics of other birdcalls. They typically build a bulky nest and raise only one brood per year. Corvids are well known for storing food in "hiding places" (caches) for future consumption. Highly developed spatial memory allows them to recover caches with high accuracy (Bednekoff et al. 1997).

Many corvids possess behaviors and preferences that allow them to thrive in human dominated landscapes. Most importantly, they are omnivorous, consuming a wide variety of foods including human-produced waste. In addition, many corvids prefer fragmented habitats with a mixture of open areas and trees (Andrén 1992, Masselink 1999), thus residential areas are prime nesting habitat. Finally, corvids are highly intelligent and can quickly adapt to human disturbances.

In this section we summarize life history information on the three most important corvid predators affecting threatened and endangered species in California: Common Ravens, American Crows, and Steller's Jays. Other species that occur in California are known or suspected to be important nest predators, including the Western Scrub Jay (*Aphelocoma californica*) (Root 1969, Purcell and Verner 1999, Peterson 2001), Black-billed Magpie (*Pica pica*) (Littlefield and Thompson 1985), and the Gray Jay (*Perisoreus Canadensis*) (Darveau et al. 1997, Sieving and Willson 1998). However, the importance of these species in affecting threatened and endangered species in California is not well documented.

The following summaries are brief life history accounts. Emphasis is placed on information pertinent to managing these species and reducing their impact on threatened and endangered species. We emphasized all known life-history information specific to California (Appendix A). Information included in these accounts was obtained from many sources, although the *Birds of North America* species accounts were particularly helpful. For more thorough life history information on these species we recommend the *Birds of North America* species accounts (at this time only available for Common Raven and Steller's Jay).

AMERICAN CROW

Species description and overview

The American Crow is the most widespread corvid in North America. In appearance, it resembles the Common Raven but is distinguished by smaller size, sleeker bill, and a squared tail. The western subspecies, (*C. b. hesperis*) is smaller and more slender-billed than the nominate form, and is found throughout western North America including California.

American Crows are highly social and often breed cooperatively. Although other corvids are cooperative breeders, only American Crows cooperatively defend breeding territories and join communal roosts at night for most of the year (Caccamise et al. 1997). Crows have been highly successful in exploiting both agricultural and urban habitats (Marzluff et al. 1994) and in some cases have caused significant financial loss to agricultural crops (Simpson 1972, Salmon et al. 1986). In addition, crows are a major nest predator of other passerines and game birds (Parker 1984, Sugden and Beyersbergen 1986). In western North America, growing evidence suggests that they are important nest predators of the endangered California Least Tern (Caffrey 1993, 1994, 1995a, 1998, Keane 1999) and the threatened Western Snowy Plover (Wilson 1980, Applegate & Schultz 2000, Castelein et al. 2000a). They are also a suspected predator of the endangered Marbled Murrelet (Nelson 1997).

Distribution and seasonal movement

American Crows are widely distributed in North America, ranging from the Atlantic to Pacific coasts and from southeast Alaska and northern Canada to the Gulf of Mexico. They are absent from deserts and other treeless areas of western North America. In California, they are present throughout much of the state, although they are absent from some of the drier western portions of the San Joaquin Valley and the more arid interior foothills and valleys (Small 1994). They are also absent from San Diego County south of Oceanside (Small 1994). In winter, crows withdraw from the higher elevation areas of the north-central and northeastern portions of the state as well as from the higher foothills of the Sierra Nevada (Small 1994).

Habitat Use

Crows use a variety of natural and human-altered habitat types including rangelands, riparian woodlands (Knopf and Knopf 1983, Richards 1971), croplands, wetlands, fields, roadsides, pastures (Sullivan and Dinsmore 1992), beaches, shores of streams and lakes (Good 1952, Chamberlain-Auger et al. 1990), urban/suburban areas, and golf courses (Chamberlain-Auger et al. 1990, Caffrey 1992). In general, crows thrive in areas of mixed habitat (open areas interspersed with woods), and thus have responded well to human-altered habitats (Marzluff et al. 2001).

Diet

American Crows are omnivorous generalists and successful scavengers. Their diet includes insects, earthworms, small vertebrates (frogs, fish, baby mice), road-kills, a variety of agricultural grains and crops (corn, wheat, barely, rye, etc.), small fruits (almonds, pecans, cherries), wild fruits (blackberries, sumac, etc.) and human refuse. In urban areas, crows often feed at concentrated food sites (landfills) during the day and roost in nearby wooded areas at night (Stouffer and Caccamise 1991).

Crows are important nest predators taking both eggs and nestlings. In some areas, they specialize on the eggs of waterfowl (Klambach 1937, Sugden and Beyersbergen 1986). They are also a major threat to the eggs and young of endangered species, including the Western Snowy Plover (Castelein et al. 2000a, b) and the California Least Tern (Caffrey 1993, 1994, 1995a, 1998, Keane 1999). At some tern colonies, a few individual crows have depredated almost all the nests in a single season (Caffrey 1993).

Territoriality

Crows are typically short-distance migrants and retreat south in the winter from the northern areas of their range in Canada and southern Alaska. Some crows breeding at high-latitudes may migrate long distances (2382 km) to wintering grounds (Klambach and Aldous 1940). However, in much of their range they are resident and defend territories or "daily activity centers" (DACs) year-round (Chamberlain-Auger et al. 1990). American Crow territories tend to be smaller in urban than in rural areas (Dickinson 1998) and are highly variable in size. Territory sizes range from 0.04 km^2 in suburban New York (Dickinson 1998) to 2.6 km^2 (SD=1.4, n=10) in a waterfowl breeding area of Manitoba (Sullivan and Dinsmore 1992). Caffrey (1992) reported an extremely high breeding density of 0.8 pairs / ha on a golf course in Encino, California. This density may be explained by the abundant food and suitable nest sites (trees) available at this site, and is probably not typical (C. Caffrey, pers. comm.). Emlen (1942) also documented high densities (111 nests in 44 ha) of nesting crows in a walnut orchard in California.

In New Jersey, wintering family groups abandon their DACs in the late afternoon to feed and join communal roosts. Group members often travel independently to the roost sites (Stouffer and Caccamise 1991). Evidence suggests that members of some populations of the western subspecies (*C. b. hesperis*) do not hold "traditional" territories (McGowan 1993). Caffrey (1992) reported territories that overlapped extensively with neighbors and were not defended against conspecifics in southern California. However, in Florida, Kilham (1985) reported aggressive territorial defense during the breeding season. These observations suggest significant flexibility in territory use and defense. This complex territorial behavior is influenced by a number of factors including food availability, time of year, relatedness of individuals, and mating system.

Breeding

Unlike most corvids that breed in temperate climates, American Crows breed cooperatively (non-breeding birds assist the breeding pair in raising offspring). The degree of cooperative breeding varies from place to place, ranging from 37% (n=147) in southern California (Caffrey 2000) to 94.4% (n=54) in Massachusetts (Chamberlain-Auger et al. 1990). Breeding groups typically consist of a pair of breeding adults and the offspring of the parent birds. However, in Oklahoma, 50% of breeding groups include helpers that immigrate from other groups (C. Caffrey, pers. comm.).

In California, breeding pairs may remain together for up to 7 years or until one individual dies (C. Caffrey, pers. comm.). Helpers are mostly year-old females (Caffrey 1992) and assist the parents by helping construct the nest, performing nest sanitation duties, defending the nest against predators, and feeding the incubating female, nestlings and fledglings (Kilham 1985, Caffrey 1992). Family groups of crows contain from 2 to 15 individuals (Chamberlain-Auger et al. 1990, Dickinson 1998). Caffrey (2000) found that, although breeding pairs with helpers fledge more young than unassisted pairs, helpers were not an important cause of this effect. Moreover, helper feeding of nestlings did not contribute to breeding success, although unassisted females made more feeding trips than assisted ones (Caffrey 1999). It appears that successful pairs have more helpers as an artifact of their success from previous years. Pairs that are the most successful tend to breed earlier, avoiding nest predation, which peaks later in the breeding season (Caffrey 2000). Overall, it is unclear if cooperative breeding increases reproductive success in American Crows. This may be because cooperative breeding in crows has only been studied in highly altered habitats, thus the historic benefits of this mating system may not be detectable.

Nest construction takes place as early as January in Florida (Kilham 1984) to April in Canada (Ignatiuk et al. 1991). Length of time for nest construction varies considerably (see Good 1952). Emlen (1942) reported an average of 13 days for nest construction in California. In Florida, Kilham (1984) reported 2 nests constructed in 5 and 9 days, respectively. The primary nesting materials used are small twigs and sticks, usually less than 0.6 m in length (Good 1952). The typical nesting substrates include both native and exotic trees, represented by a variety of evergreen and deciduous species. Crows nested in gymnosperms, eucalypts (*Eucalyptus* sp.), and sycamores (*Platanus* sp.) at a golf course in southern California (Caffrey 2000). At a site in rural/suburban Massachusetts, crows nested predominantly (88%) in pitch pine (*Pinus rigida*) although white pine (*P. stobus*), spruce (*Picea* sp.) and eastern red-cedar (*Juniperus virginiana*) were also used (Chamberlain-Auger et al. 1990). Mean nest tree height at this site was 11.0 m ± 3.2 SD, while mean nest height was 9.9 m ± 3.1 SD (Chamberlain-Auger et al. 1990). At a site in rural Florida, crows nested in live oaks (*Quercus virginiana*). In Ohio, Good (1952) found that oaks were the most commonly used nesting substrate (45 of 100 nests). Other tree species used for nest sites in Ohio included: ash (*Fraxinus* sp.), elm (*Ulmus* sp.), beech (*Fagus* sp.) and 14 other, mostly deciduous, species (Good 1952). Nests are usually placed near the trunk or in the fork of a large branch (Good 1952). However, detailed micro-site habitat attributes at crow nest sites remain largely unpublished or undocumented. A new nest is typically built every year (Chamberlain-Auger et al. 1990, Dickinson 1998. Rarely, nests from previous years will be remodeled and used again (Good 1952).

Egg-laying occurs from February (Kilham 1984) to May (Ignatiuk et al. 1991). In Florida, Kilham (1984) reported a clutch initiation date of February 27 for one pair. In Saskatchewan, the mean clutch initiation date was 6 May ± 6 SD (Ignatiuk et al. 1991). Chamberlain-Auger (1990) reported the earliest date of clutch initiation as 20 March; latest date was 17 June, in Massachusetts. Typical clutch size ranges from 3-6 ($\overline{X}$ = 4.8, SD = 0.06, n=104) in Saskatchewan (Ignatiuk et al. 1991). In Ohio, Good (1952) reported a clutch size of 4-5. The incubation stage occurs from March to June. In Saskatchewan, incubation took 17-18 days ($\overline{X}$ = 17.7, SD = 0.6, n=74) (Ignatiuk et al. 1991). In Massachusetts, Chamberlain-Auger et al. (1990) reported a 14-33 day incubation period ($\overline{X}$ = 22.3, SD = 6, n=13). At a site in Los Angeles County, California, from 1986 to 1990, the average incubation initiation date was 31 March, SE = 2.0

(Caffrey 2000). Nestlings fledged in 28 to 35 days in Ohio (Good 1952) and 38, SE = 0.9, n=17 in southern California (C. Caffrey, pers. comm.).

Nesting success in crows was as low as 39% in Saskatchewan (Ignatiuk et al. 1991), and up to 43% in California (Caffrey 2000). Predation is the major cause of nest failure (Caffrey 2000) although starvation has been reported (Ignatiuk et al. 1991). Common predators of crow eggs and nestlings include raccoons (*Procyon lotor*), Great Horned Owls (*Bubo virginianus*), Cooper's Hawks (*Accipiter cooperi*), Red-tailed Hawks (*Buteo jamaicensis*), Red-shouldered Hawks (*Buteo lineatus*) and grey squirrels (*Sciurus carolinensis*) (Chamberlain-Auger et al. 1990, Dickinson 1998, Caffrey 2000).

Both parents and helpers feed fledglings for up to two months (McGowan 1993). Young birds stay with their parents as helpers for as long as 6 years. Eventually, they find a breeding opportunity, often near the natal territory, or usurp part of their parents' territory. Crows may disperse very short distances (0.075 km) from their natal nest site to breed or they may establish a breeding site some distance from their natal territory. McGowan (1996) reported a maximum dispersal distance of 65 km in New York but some birds may go further. Unlike most other cooperative breeders, it is the female crows that commonly delay dispersal and help at the nest (Caffrey 1992).

Feeding behavior

Crows spend 15-21% of the time foraging, with peak activity occurring in the morning (Stouffer and Caccamise 1991). In the afternoon, resident crows in New Jersey often stopped to feed at landfills en route to their nightly roosts (Souffer and Caccamise 1991). Foraging distances vary tremendously, ranging from an average of 382 m from the nest in southwestern Manitoba (Sullivan and Dinsmore 1992) to 18 km from a DAC in New Jersey (Stouffer and Caccamise 1991).

Crows prey on the nests of many different bird species (see Tables 5 & 6). Most evidence suggests that crows rely on visual cues to locate nests. Picozzi (1975) found that crows located marked artificial nests more frequently than unmarked nests. In addition, crows were less likely to depredate artificial duck nests that were highly concealed at a site in Saskatchewan (Sugden and Beyersbergen 1986). Buler and Hamilton (2000) reported crows "trap-lining" closely spaced artificial nests, destroying several nests sequentially within a few hours. Crows are also believed to develop search images for real nests, including those of endangered species. At Vandenberg Air Force Base in California, Persons (1995a) reported crows actively searching for Western Snowy Plover nests by flying low and flushing incubating adults and by walking from one potential nest site to another. Crows have been reported to locate nests in flight (Dwernychuk and Boag 1972), on foot (Sugden and Beyersbergen 1987), and from perches (Salathé 1987).

Roosting behavior

American crows often roost together at night throughout the year. However, peak roosting typically occurs in the winter (Gorenzel and Salmon 1995). Roosting locations may be used continually for years and contain only a few individuals or as many as 40,000 (Dickinson 1998). Individuals may fly 18 km to a roost site from their daytime territories (Stouffer and Caccamise 1991) and include "vagrant" birds as well as members of family groups (Stouffer and Caccamise 1991). However, the tight cohesion of family groups seen on territories is not exhibited at the roost (Caccamise et al. 1997).

Many tree species are used for roosting, including red maples (*Acer rubrum*), white spruce (*Picea glauca*) (Stouffer and Caccamise 1991), and cottonwoods (*Populus* sp.) (Knopf and Knopf 1983). Gorenzel and Salmon (1995) reported over 17 tree species used as roost sites in California. The principal deciduous tree species used for roosting included: ash, mulberry (*Morus* sp.), elm, alder (*Alnus* sp.), sycamore, and oaks. The principal conifers included: pines (*Pinus* sp.), deodar cedars (*Cedrus deodara*), and coast redwoods (*Sequoia sempervirens*). Gorenzel and Salmon (1995) found that deciduous trees were typically used as roost sites in the summer while conifers were used during winter. Roost trees were typically closer to roads and had greater height, crown volume, crown diameter, and diameter at breast height than non-roost trees (Gorenzel and Salmon 1995). There is some evidence that large roosts form in response to superabundant food supplies, namely, landfills (Stouffer and Caccamise 1991).

COMMON RAVEN

Species description and overview

Common Ravens are the largest of all passerines. They resemble the American Crow in appearance but are easily differentiated by larger body size, more massive bill, and a wedge-shaped tail. Although both raven and crow ranges overlap, ravens appear to be invading agricultural habitats to a greater extent than urban areas, whereas crow expansion appears to have the opposite trend (Marzluff et al. 1994).

Recent genetic studies suggest that Common Ravens in California and parts of Washington and Idaho ("California clade") are genetically distinct from all other Common Ravens worldwide ("Holarctic clade") (Omland et al. 2000). The California clade appears to be more closely related to the desert-adapted Chihuahuan Raven (*Corvus cryptoleucus*). However, morphological differences between the 2 clades are minimal. More research needs to be done to determine if classification into separate species is warranted (Omland et al. 2000).

Adult ravens form long-term pair bonds and typically defend non-overlapping nesting territories. Non-resident juvenile ravens often wander greater distances than territorial birds, and both resident and non-resident birds gather at sites with abundant food (e.g., carcasses and dumps) (Heinrich et al. 1994). However, groups of ravens typically lack the tight cohesion seen in other social birds (Heinrich et al. 1994).

Ravens are highly adaptable to a wide range of habitats and foods. Because of this, they often respond positively to human-altered habitats. In some areas, ravens have been termed a "pest" and are causing economic damage as well as harming other native wildlife.

Distribution and seasonal movement

Common Ravens are widespread throughout large regions of the Northern Hemisphere. In North America, they are found in most of Canada and Alaska, the United

States west of the continental divide, and throughout the Appalachian Mountains of the eastern United States. They occur throughout California, except for areas of the Central Valley, parts of the central coast, and cultivated valleys of the southeast (Small 1994). Ravens have recently expanded their range along the coast into San Mateo and northern Santa Cruz counties (G. Page, pers. comm.). Raven migration in California is not known. However, they are often seen at the highest elevations in the late summer (Small 1994).

Habitat Use

Ravens are found in a wide range of natural habitat types, including arctic tundra, coniferous and deciduous forests, prairies, grasslands, and deserts. They prefer areas with some vertical relief (e.g., cliffs, trees, human-made structures) to provide nesting and foraging sites (Boarman and Heinrich 1999). They thrive in many human-altered habitats, including agricultural areas (Engel and Young 1989a), roadsides and linear right-of ways, (Knight and Kawashima 1993, Sherman 1993), ranches (Roth et al. 1999), rangelands (Knight 1984), and near campgrounds and picnic areas (Wallen et al. 1998, 1999).

Diet

Like crows, ravens are generalist omnivores. The variety of food types in their diet often reflects differences within and among individuals as well as the distribution of food in a given area (Engel and Young 1989a, Stiehl and Trautwein 1991). Ravens commonly scavenge on "human-produced" foods, such as road kills (Boarman 1993), slaughterhouse wastes (U.S. BLM 1990), calf after-births (Roth et al. 1999), organic matter at landfills (Boarman 1993), grains, and fruits (Engel and Young 1989b). However, ravens are also accomplished hunters, taking a variety of small mammals, amphibians, reptiles, birds, and insects (Sherman 1993, Boarman and Heinrich 1999). Unlike other corvids, Common Ravens sometimes specialize on carrion (Heinrich 1988). Ravens are documented predators of both eggs and nestlings and may even become specialized nest robbers (Stiehl 1978, Gaston et al. 1985, Andrén 1992).

In California, ravens are known or thought to be important predators on the eggs and young of several threatened and endangered species, including the Western Snowy Plover California Least Tern, California Condor , San Clemente Island Loggerhead Shrike, Greater Sandhill Crane, Marbled Murrelet and desert tortoise (See Table 5 for citations).

Territoriality

Raven pairs often occupy a "home range" in which they forage and nest. Breeding pairs establish year-round "territories" *within* the home range. Typically, territories are non-overlapping and defended year-round (most vigorously when nesting) (Boarman and Heinrich 1999). Unlike territories, home ranges may overlap with that of neighboring raven pairs, especially when near a concentrated food source (J. Roth, pers. comm.). In addition, the entire home range is *not* defended from conspecifics. In the following description of raven territoriality, most of the papers do not distinguish between home range and territory.

Territory size varies considerably depending on availability of nest sites and food. Territories range from 1.2 km^2 at Camp Pendleton, California (Linz et al. 1992) to 40.5 km^2 in Minnesota (Bruggers 1988). The smaller territory size at Camp Pendleton may be due to the higher nesting density at this site (Linz et al. 1992).

In Utah, Smith and Murphy (1973) reported an average home range size of 6.6 km^2 (n=4) during the breeding season. Craighead and Craighead (1956) reported an average home range size of 9.4 km^2 for three raven pairs in Wyoming. At the Point Reyes peninsula, California, preliminary results suggest that non-breeding ravens have larger ranges than breeding birds (Roth et al. 1999).

Breeding

Little is know about pair formation and nest-site selection in the Common Raven. Pairs are thought to be monogamous throughout the year, although extra-pair copulations have been observed (Boarman and Heinrich 1999). Typically, ravens do not breed until 2-4 years of age (Jollie 1976).

Nesting substrates are highly variable, ranging from cliffs and trees to human-made structures, including power-line towers, telephone poles, abandoned buildings, railroad trestles, billboards, oil derricks, and highway overpasses (Boarman and Heinrich 1999). In the California Desert Conservation Area, ravens have been observed nesting in tamarisk trees (*Tamarix ramosissima*), Joshua trees (*Yucca brevifolia*), transmission towers, and rock outcrops (U.S. BLM 1990). In west Marin County, California, ravens often nest in patches of introduced trees, including Monterey cypress (*Cupressus macrocarpa*), Monterey pine (*Pinus radiata*), and eucalyptus (J. Roth, pers. comm.). Many ravens are thought to return to the same nest year to year.

Nest construction begins in early to late winter. Sticks are the main building material used (Dorn 1972, Stiehl 1978). Nest construction takes from 1-4 weeks (Goodwin 1976). Egg-laying typically begins early March to mid-April. In North America, clutch sizes range from 3-7 ($\overline{X}$ = 5.4, SD = 0.42, n = 7 study areas) (Boarman and Heinrich 1999). Incubation lasts 20-25 days (Harlow 1922, Dorn 1972, Stiehl 1978). Linz et al. (1992) reported and average of 2.9 (SD = 1.0, n = 9) nestlings per nest at Camp Pendleton, California. The nestling stage lasts 5-7 weeks, but juvenile birds will stay near the nest for up to 4 weeks following fledging (Knight and Call 1980). Most young fledge by mid-June. The female performs most of nest construction and incubating, however, both parents feed the young.

If clutches are lost early in the season, they are usually replaced within 2-3 weeks (Harlow 1922, Stiehl 1978). Large raptors, other ravens, and martens (*Martes americana*) are thought to be responsible for depredating raven nestlings (Dorn 1972). Predation on Common Raven eggs is unrecorded.

Feeding behavior

Breeding ravens have been reported to forage within a defended area during the nesting season (Sherman 1993), although this may vary with region, landscape, and food supply. Sherman (1993) found that ravens in the Mojave Desert spend an equal amount of time scavenging and live hunting and that 75% of feeding activity takes place within 400m of the nest during the breeding season. Furthermore, ravens foraged within 1.7 km of linear-right-of-ways (roads, railways, transmission power lines, telephone lines), and spent 49% of the time foraging directly on the linear-right-of-ways (Sherman 1993). However, at Malheur National Wildlife Refuge in Oregon, ravens usually do not prey on Sandhill Crane nests close (< 0.4 km) to their nest (C. Littlefield, pers. comm.). When human-subsidized food is present, ravens often concentrate their feeding on these food sources and travel distances may be significantly shorter (Engel and Young 1992b).

Ravens typically concentrate their feeding activity in the morning and late afternoon (Engel and Young 1992a, Sherman 1993), although this varies with location and time of year. On Digges Island, Canada, ravens typically depredated eggs and young of Thick-billed Murres (*Uria lomvia*) before midday (Gaston et al. 1985).

Non-breeders, usually juvenile vagrants, often form “crowds” when feeding at concentrated food sources (Heinrich 1988). Crowds lack the cohesiveness in membership that most “flocking” birds exhibit (Heinrich et al. 1994). In addition, most members in a crowd are not closely related (Parker et al. 1994). However, non-resident birds rely on crowd formation to gain access to concentrated food sources within the territories of adult birds (Heinrich 1999). Ravens often cache food for later use (Heinrich 1988) and are thought to rely mostly on visual cues to detect prey (Littlefield 1995a).

Roosting behavior

Non-breeder ravens typically roost together at night when a concentrated food source is nearby. Ravens generally roost in trees, telephone poles, or power lines. Roost size varies with the size of the food source. A single deer carcass can support a roost of 50-100 individuals for up to one week (Heinrich 1988). A large supply of grain can support a roost of >2000 birds several months or longer (Engel and Young 1989b, Littlefield and Ivey 1994). In southwestern Idaho, the average distance travelled by ravens from roosts to feeding areas was 6.9 km, although one bird was observed 62.5 km from its roost (Engle and Young 1992a). Roosts may serve as information centers for food by enabling new birds in a roost to quickly find a previously located food source (Heinrich 1988). Adults usually do not join communal roosts and often roost at the nest site, even when not breeding (Engel et al. 1992).

STELLER’S JAY

Species Description and Overview

The Steller’s Jay is common in coniferous forests of western North America. The erectile crest, black wing and tail bars distinguish this jay from all others. The Steller’s Jay exhibits considerable variation in plumage throughout its range, and sixteen races are currently recognized (Browning 1993). The Blue Jay is considered a rare visitor to California and is easily distinguished from the Steller’s Jays by its blue crest, contrasting black “necklace”, and white wing and tail spots (Goodwin 1976, Small 1994). Hybrids with intermediate plumages have been reported in parts of the U.S. where the two species meet (Williams and Wheat 1971, Wilde 1993).

Steller's Jays are usually conspicuous members of the avifauna within their range. They may be encountered individually or in pairs and can be quite boisterous. Small groups commonly come together for social display, to mob a predator, or to visit a food source (Bent 1946, Brown 1963, Goodwin 1976, Ficken 1989). Larger numbers may congregate where food is abundant (Bent 1946, Brown 1963, Goodwin 1976, Salata 1982). Periodic fall/winter invasions of, primarily, young birds have been documented on Vancouver Island, British Columbia (Stewart and Shepard 1994) and southeastern Arizona (Westcott 1969). However, adult Steller's Jays are generally nonmigratory (Bent 1946, Brown 1963, Salata 1982). Mated pairs appear to form monogamous, long-term bonds and maintain year-round areas of dominance through complex social interactions (Brown 1963, 1964, Hope 1980).

Steller's Jays are opportunistic omnivores whose diet can include the eggs and nestlings of other birds (Singer et al. 1991, S. Elliott in George 2000, Craig 1997, Sieving and Willson 1998, 1999, Liebezeit 2001) as well as "human-produced" foods (Bent 1946, Brown 1963, 1964, Bekoff et al. 1999). They are highly adaptable to anthropogenic habitat changes and can be relatively tame or even bold in locations where they have grown accustomed to people (Abbott 1929, Bent 1946, Brown 1963). Steller's Jays have been characterized as forest-edge associates, and their densities tend to increase near human-created forest edges (Bent 1946, Craig 1997, Sieving and Willson 1998, 1999, Masselink 1999). Recent evidence indicates that Steller's Jays may be major nest predators of the threatened Marbled Murrelet (Nelson and Hamer 1995a, Luginbuhl et al. in press).

Distribution and Seasonal Movement

Steller's Jays are restricted to the northern latitudes of the western hemisphere, with races known from as far south as Nicaragua (Greene et al. 1998). In the United States and Canada, they are found in forested habitats along the Pacific coast from southeastern Alaska, south to central California, and inland to an eastern boundary that roughly parallels the Continental Divide. In California, Steller's Jays inhabit the Klamath Mountains and northern Coast Ranges from the Oregon border south to Morro Bay, San Luis Obispo County; the Warner Mountains of northeastern California; and the northern Cascades south through the Sierra Nevada and Greenhorn Mountains, Kern and Tulare Counties (Small 1994). They are patchily distributed on the mountain ranges of southern California from the Tahachapis south to the Santa Rosa Mountains, San Diego County (Garrett and Dunn 1981, Weathers 1983). Rare vagrants have been reported in the Central Valley, eastern interior valleys, and southern coastal and near-coastal areas (Small 1994). They are absent from the southeastern deserts and have not been reported from any of California's offshore islands (Small 1994).

Steller's Jays are considered resident where breeding populations occur, although seasonal movements have been recorded. In the U.S. and Canada, Grinnell and Miller (1944), Phillips et al. (1964), and Small (1994) document vertical movements of high-altitude populations. Large, irruptive post-breeding movements have been reported on Vancouver Island, B.C. (Munro 1923, Bent 1946, Guiguet 1954, Stewart and Shepard

1994). Stewart and Shepard (1994) found that during the period from 1958 to 1993 intervals between invasion years were not regular, but they do indicate that "a marked decline" of Steller's Jays occurred in years immediately following major invasions. They estimated that 25% of Steller's Jays that moved through Vancouver Island during 1992/93 were adult birds (>1 yrs. old). This differs from work done by Brown (1963) in northern California who found that only young birds dispersed in fall.

Some authors have suggested that Steller's Jays exhibit southward movement during some years (Swarth 1922, McCabe and McCabe 1928, Munro and Cowan 1947, Cannings et al. 1987). However, Morrison and Yoder-Williams (1984) found no evidence of latitudinal migration from their analysis of banding and recovery records. They conclude that most Steller's Jays are sedentary and characterize movements as wanderings or dispersals. Greene et al. (1998) point out that seasonal movements might be common throughout the jay's range and cite the scarcity of banding data required to detect local trends.

Brown (1963) and Salata (1982) report Steller's Jay populations to be year-round residents near Berkeley, California, as does Harris (1996) for suitable habitats in Del Norte, Siskiyou, Trinity, Humboldt and northern Mendocino Counties. Small (1994:173) reports "minor late summer and fall" down slope movements in populations inhabiting the "Cascades-Sierra axis and southern California ranges". Hile (1993) gives no indication of seasonal movements in the Mt. Pinos population.

Habitat Use

The habitats used by Steller's Jays can vary throughout the year. During the breeding season, Steller's Jays commonly use closed-canopy forests with a conifer component, including pines, true firs (*Abies* sp.), Douglas fir (*Pseudotsuga menziesii*), spruce, western red cedar (*Thuja plicata*), western hemlock (*Tsuga heterophylla*) and redwood (Bent 1946, Cannings et al. 1987, Andrews and Righter 1992, McEneaney 1993, Hile 1993, Stewart and Shepard 1994, Small 1994, Sieving and Willson 1998, 1999). They are also reported to use coast live oak (*Quercus agrifolia*) and mixed scrub oak (*Q. berberidifolia*) woodlands (Salata 1982, Hile 1993, Small 1994), eucalyptus groves (Brown 1964), pure and mixed stands of California bay (*Umbellularia californica*) (Salata 1982), and deciduous stands of cottonwood, willow (*Salix spp.*), and alder (Sieving and Willson 1998, 1999).

Steller's Jays use coniferous and mixed coniferous-deciduous forests year-round in mid- to low elevation areas. Orchards and suburban gardens are often used during the fall and winter, and individuals may be found in these habitats at other times of the year (Bent 1946, Stewart and Shepard 1994). The edges of pasture, grain, and nut-producing agricultural fields have been utilized in late summer and early fall (Swarth 1912 and Dicks 1938, both cited in Bent 1946). During irruptive movements, flocks may move through habitats not normally occupied, including Sonoran Desert (Monson and Phillips 1981, in Greene et al. 1998) and coastal shorelines (Stewart and Shepard 1994).

Several authors report higher Steller's Jay densities at forest edges versus forest interior, especially near anthropogenic environments (Salata 1982, Hile 1993, Craig 1997, Brand 1998, Sieving and Willson 1998, 1999, Masselink 1999, Brand and George in press). At Redwood National and State parks in California, post-Memorial Day

Steller's Jay abundance was significantly greater in areas of high human use (e.g., picnic and camping areas) compared to areas with medium (e.g., backcountry trails) and low use (i.e., <0.5 km from development) (Wallen et al. 1998, 1999). These results suggest that many jays at Redwood National and State parks may obtain food subsidies in areas of high human use, particularly during the busy tourist season.

Diet

Steller's Jays are opportunistic omnivores. They consume a wide variety of plant and animal materials, including nuts, seeds, berries, fruits, arthropods, and small vertebrates, depending largely on what is available in a given area and season (Goodwin 1976, Salata 1982). Pine seeds, acorns, and the fruits of the California bay are important foods when available and are frequently cached by Steller's Jays (Bent 1946, Goodwin 1976, Vander Wall and Balda 1981, Salata 1982, Hile 1993). Foods made available by humans are readily consumed by Steller's Jays at picnic areas, campgrounds and bird feeders and are also cached (Abbott 1929, Bent 1946, Brown 1963, 1964, Bekoff et al. 1999).

Steller's Jays commonly consume the eggs and nestlings of other birds, and several recent studies implicate Steller's Jays as important nest predators (Craig 1997, Sieving and Willson 1998, 1999, Brand and George 2000). Higher Steller's Jay densities at forest edges appear to be correlated with higher nest predation rates (Craig 1997, Sieving and Willson 1998, 1999, Brand and George in press). Craig (1997) found a positive correlation between Steller's Jay density and nest predation rates of both artifical open-cup nests and natural nests of American Robins (*Turdus migratorius*). He further concludes that "commensal food resources" (e.g., bird feeders) are believed to be responsible for this increase.

In California, nest predations by Steller's Jays have been documented for the Dusky Flycatcher (*Empidonax oberholseri*) (Liebezeit 2001), Varied Thrush (*Ixoreus naevius*) (George 2000), and Marbled Murrelet (Singer et al. 1991). Other authors implicate Steller's Jays as potentially important nest predators of Marbled Murrelets outside of California (Luginbuhl et al. in press, Masselink 1999).

Feeding Behavior

Steller's Jays forage both on the ground and in trees, exhibiting a wide range of feeding behaviors depending on the resource being exploited (Salata 1982, Greene et al. 1998). Foliage gleaning, hawking, and using the bill to pry under bark, flick aside leaf litter, dig into soil, and pull/dislodge food items are all frequently employed foraging strategies (Abbott 1929, Brown 1964, Salata 1982, Hile 1993). A large food item is generally carried to an elevated perch and held with the feet while breaking it into bite-size portions with the bill (Abbott 1929, Brown 1964, Carothers and Sharber 1972). Steller's Jays commonly cache seeds and items taken from feeders and are known to steal the caches of other birds (Tomback 1978, Vander Wall and Balda 1981, Burnell and Tomback 1985). In a study on Steller's Jay use of feeders, Bekoff et al. (1999) found that they prefer sunflower seeds over other seed types, and prefer to use unoccupied bird feeders compared to feeders occupied by conspecifics or squirrels.

In Berkeley, California, Steller's Jays spent 71% of all foraging time in trees consuming fruit and insects, with peaks occurring in spring and fall (Salata 1982). Ground-foraging occurred mostly at sites with greater than 95% tree-canopy coverage and peaked in winter, coinciding with the recovery of cached food items.

Like other corvids, Steller's Jays are thought to be visual predators on the eggs and nestlings of other birds (Øuellet 1970, Ehrlich and McLaughlin 1988, Andrén 1992, Sieving and Willson 1998). Observations of jays preying on nests indicate that they will do so individually and in pairs (Singer et al. 1991, George 2000, Liebezeit 2001, D. Craig, pers. comm.). They will prey on nests that are encountered opportunistically, however, they are believed to develop a nest "search-image" (D. Craig, pers. comm.). In southeastern Alaska, Sieving and Willson (1998, 1999) found that Steller's Jays may specialize on nests when feeding nestlings and switch to other resources when their young fledge.

Breeding

Steller's Jays appear to form long-term pair bonds and are believed to be monogamous (Brown 1963, 1964). Behaviors associated with courtship and pair-bond renewal include sexual sidling, circling, wing-spreading, and mate-feeding by the male and may begin up to four months prior to the onset of nest-building (Brown 1964, Greene et al. 1998). Both members of a pair select the nest-site during early pair-bond formation (Brown 1964). Both the male and female gather nest materials and participate in nest construction, which may begin as early as late March (Bent 1946, Brown 1964, Greene et al. 1998). Nests are typically placed from 3 to 5 m above the ground on horizontal tree branches close to the trunk (Bent 1946, Harrison 1979, R. Campbell in Greene et al. 1998). Nests are constructed from plant fibers, leaves, moss, sticks, and mud.

The most commonly used nesting substrates are conifers and other evergreens early in the season, while deciduous trees and shrubs may be used after they leaf out (Bent 1946, Salata 1982, Sieving and Willson 1999). Steller's Jays show a high degree of flexibility with regard to nest-site selection and have been known to place nests in bushes very close to the ground, over 30 m up in the canopy, and on human-made structures (Harrison 1979, R. Campbell in Greene et al. 1998). In British Columbia, 66% of 70 nests were found in "human-influenced coniferous or mixed forests", with the remaining 33% in "undisturbed forests" (R. W. Campbell cited in Greene et al. 1998). Trees utilized for nesting near Berkeley, California include Coast live oak (*Quercus agrifolia*), California Bay (*Umbellularia californica*), Monterey pine (*Pinus radiata*), and Madrone (*Arbutus menziesii*) (Salata 1982). In California, suburban foothills and canyons vegetated with exotic pines, cedars, and eucalyptus, have been invaded by breeding Steller's Jays in recent years (Small 1994).

In California, Steller's Jays breed in suitable habitat from sea-level to tree line (approx. 2,600 m or 8,500 ft) (Small 1994). The peak nesting period is late April through late May (Greene et al. 1998). Steller's Jays are thought to have only one brood per season. Some birds may attempt to nest a second time after failure of the first nest (Greene et al. 1998). Clutch size can vary from 2 to 6 eggs, with 4 or 5 being typical (Harrison 1979). Mean clutch size is 3.06 (SD = 0.82, n = 33) across western North America (Cornell Laboratory Ornithology nest records, cited in Greene et al. 1998).

Incubation lasts approximately 16 days and is normally performed only by the female (Brown 1964, Goodwin 1976). Nestlings remain in the nest for about 16 days. Both parents feed the young. Fledglings may remain in a family group with their parents into the fall and winter (Bent 1946, Brown 1964). At a site in California, only 11% of first-year birds showed signs of attempted breeding (Brown 1964).

Territoriality

The social system of Steller's Jays is characterized as "site-related dominance", a social hierarchy intermediate between colonial living and territoriality (Brown 1964, 1963, Oberski and Wilson 1991). Brown (1964) found that the area of dominance for a mated pair is centered on the nest. The resident male dominates all other Steller's Jays within this area, and the resident female dominates other females. A jay's area of dominance decreases with increasing distance from the nest and is relatively small compared to the area in which it may forage. Brown (1963) estimated the area of dominance for each mated pair to be about 120 m across. The complex dominance hierarchy that results is maintained by frequent displays, vocalizations, and interactions. According to Brown (1974), this system allows a resident Steller's Jay to travel long distances to feed on temporarily rich food supplies, but still maintain a core area for nesting and foraging. Areas of dominance are maintained by a mated-pair year-round, and the social status of an individual shows some consistency from year to year (Brown 1963).

Two Steller's Jays fitted with radio transmitters were reported to have home range sizes between 29 – 65 ha in Redwood National Park, California (Wallen et al. 1999). This home range size indicates a probable maximum travel distance of approximately 1 km (radius = 0.47 km), assuming a circular home range area.

Roosting behavior

Accounts of roosting behavior in Steller's Jays are rare. Wallen et al. (1999) reported use of nocturnal roosts by 2 radio-tagged jays in Redwood National Park, California. Both birds returned to feeding sites (near human use areas) by 10:00.

CORVID POPULATION TRENDS IN CALIFORNIA

METHODS

We report population trends for the American Crow, Common Raven, and Steller's Jay from previous studies. In addition, we acquired recent population trend information from the Breeding Bird Survey (BBS) over the period of 1966 to 1999 for these species in California and the 12 physiographic ecoregions of California (Figure 1) (Sauer et al. 1999). We also summarize population trend information for these species from the Christmas Bird Count (CBC) database (Sauer et al. 1999) over the period 1959 to 1999 in California. Trends on each route were estimated using linear route regression and represent the percent change in number of individuals observed per year (Geissler and Sauer 1990). Regional trends were estimated as a weighted average of trends on individual survey routes (Sauer et al. 1999).

Both BBS and CBC data provide a large-scale perspective on bird population trends across North America. However, because all surveys are conducted from roadsides, there is a strong likelihood of overestimating corvid numbers. Corvids (particularly ravens and crows) are often found at higher densities along roadsides than other less disturbed habitats (Knight and Kawashima 1993). However, we are confident that these data provide a reliable index of corvid population trends in California because: 1. Most other biases associated with the BBS and CBC survey techniques are minimal regarding corvids and 2. Roadside habitat is prevalent across the state. In the Mojave desert alone, >57,600 km of roads cross the landscape (Sherman 1993).

DOCUMENTATION OF CORVID INCREASES IN CALIFORNIA

American Crow

Crows were relatively uncommon throughout most of the west in the early 1900s (Richards 1971). Populations increased with the arrival of Europeans and large-scale development of agriculture and irrigation (see Marzluff et al. 1994). Breeding Bird Survey and CBC data from 1965 to 1979 indicate a steady, but slight rise in breeding and wintering populations (Robbins et al. 1986). Both data sets indicate a positive correlation between human density and number of crows. In the San Francisco Bay area, CBC data from 1960 to 1997 indicates stable American Crow numbers until 1990, then a dramatic increase (Coston 1997). The most recent analysis (1959 to 1999) of CBC data from California indicates a significant increase ($P < 0.05$) in the crow population across the entire state (Table 1).

Consistent with the CBC data, recent BBS analyses show a significant increase ($P < 0.01$) in crow numbers throughout California during the period 1966 to 1999 (Table 2, Figure 2). At a regional scale, crows are increasing significantly on the Columbia Plateau, in the Central Valley, and in the California foothills ecoregions (Table 2, Figure 3). There are no significant declines in any of the ecoregions (Table 2).

Common Raven

Overall, raven populations appear to have increased in the past 50 years in most parts of the west. Prior to this, ravens were reported as becoming scarcer in settled parts of California because of human persecution (Grinnell and Miller 1944). However, as early as the 1950s ravens showed signs of increasing numbers in some areas of western North

America (Houston 1977). More recently, analysis of BBS data from 1968 to 1979 indicated an increase in raven populations throughout the west, with major increases noted in California (Robbins et al. 1986). Marzluff et al. (1994), using BBS data from 1966 to 1990, also documented an increase in raven populations in the west (Marzluff et al. 1994). They found this increase to be correlated with agricultural habitats and reported a negative correlation with human density.

The most dramatic increases in western raven populations have occurred in the southwestern deserts of California. In the Mojave Desert of California (including southern Nevada and extreme southwestern Utah), BBS data from 1968 to 1990 indicates an increase in the raven population of 1528% (Boarman and Berry 1995). In the same time period, raven numbers increased 474% in the Colorado and Sonoran desert regions of California and Arizona (Boarman and Berry 1995). In the Great Basin Desert of California and Nevada, raven populations have increased 168% in 20 years. Finally, in the southern California basin, ravens have experienced increases of 328% in twenty years (Boarman and Berry 1995).

The most recent BBS information shows a continued significant increase ($P < 0.01$) in raven populations throughout California (Table 3, Figure 4). At a regional scale, significant increases in raven abundance have occurred in the Pitt-Klamath Plateau, California Foothills, South Pacific Rainforests, and the Los Angeles Ranges (Table 3, Figure 5). The dramatic increase documented in the Mojave population (U.S. BLM 1990) has stabilized since 1990, although the population is more than three times higher than in the 1960’s (Figure 6). There have been no significant declines in the raven population in any ecoregion (Table 3).

The most recent CBC data corroborates the trends documented in the BBS data. CBC data from 1959 to 1999 indicates a significant increase ($P < 0.01$) in the raven population throughout California (Table 1).

Steller’s Jay

Steller’s Jay populations were stable for the period of 1966-1996 (BBS survey data cited by Greene et al. 1998) over their entire range. However, significant population changes are indicated for some survey regions. Steller’s Jays have increased significantly in Washington, the central Rocky Mountains region, and the central New Mexico-Arizona region, and declined by 1.7% per year in the Sierra Nevada region (Sauer et al. 1997). The Northern Pacific Rainforest region (coastal British Columbia, Washington, Oregon, and northern-coastal California) reported an annual increase of 1.3% (Sauer et al. 1997). Raphael et al. (1988) hypothesize that Steller’s Jays have decreased by about 5% since pre-settlement times in Douglas-fir forests.

Recent BBS analyses show that Steller’s Jay populations have remained stable at the state-wide level from 1966 to 1999 (Table 4, Figure 7). However, they are declining in the Sierra Nevada ecoregion (Table 4, Figure 8) and increasing in the Southern Pacific Rainforests (Table 4). Unlike the BBS data, recent CBC analyses indicate a significant ($P < 0.10$) increasing trend in the Steller’s Jay population throughout California. The disparity between the BBS and CBC may be due to the differences in the timing, techniques, or areas sampled by the two survey techniques.

CAUSES OF CORVID INCREASES

The underlying cause of corvid increases throughout California (and the world) is inextricably linked to the activities of humans. Most corvid species are “human commensals” and thrive in highly disturbed habitats including agricultural, suburban, and urban areas (Marzluff et al. 1994). A major reason why corvids are successful in these areas is because they are generalist foragers, readily eating human-produced wastes. Thus, a key factor in corvid population increases is thought to be the availability of anthropogenic food sources that “subsidize” corvid populations (Boarman 1993, Marzluff et al. 2001). Food subsidies of corvid species include garbage at landfills, dumpsters, and at the curbside (Boarman 2000), agricultural grains (Stiehl 1978, Engel and Young 1989b), fruits (Simpson 1972), cattle and sheep ranching by-products (Larsen and Dietrich 1970), feed at dairy farms (Roth et al. 1999), and road kills (Boarman and Heinrich 1999). In addition, water subsidies are thought to be a particularly important factor contributing to raven increases in desert areas of California and other arid regions (W. Boarman, pers. comm.). Sources of subsidized water for corvid species include cattle watering troughs, irrigation canals (and associated structures), reservoirs, sewage treatment areas, and irrigated agricultural areas (Boarman 1993).

Habitat fragmentation due to logging and urban/agricultural development has contributed to increases in “habitat generalist” predators, including some corvid species (Andrén 1992). Most corvid species, including the three focal species in this review, thrive in fragmented habitats. Suitable breeding habitat for corvids has also been expanded through availability of human-made nesting and perching sites including telephone poles, electrical towers, bridges, buildings and the creation of urban parks and golf courses (Marzluff et al. 1994). American Crows have responded particularly well to suburban sprawl. Suburban habitats provide the perfect combination of foraging areas (lawns, roads, and landfills) immediately adjacent to excellent breeding habitat (patches of trees) (Marzluff et al. 2001).

The social nature of corvids has probably enabled efficient exploitation of human food resources through flocking behavior (Marzluff et al. 1994) and use of communal roost sites (Stouffer and Caccamise 1991, Heinrich 1988). In addition, human persecution of corvids has been much reduced (particularly in urban areas) because of the implementation of the Migratory Bird Treaty Act in 1918, which prohibited indiscriminate killing of migratory birds (including corvids). However, farmers and ranchers are still able to obtain depredation permits to kill corvids that are feeding on crops or cattle feed. In rural areas, some corvids have altered their nesting behavior in response to human persecution (Knight 1984, Knight et al. 1987). The combined effect of these factors is believed to be responsible for the increase in the abundance and range expansion of many corvids well beyond their historic levels.

TABLE 1. Population trends (%/yr) of Common Raven (*Corvus corax*), American Crow (C*orvus brachyrynchus*), and Steller's Jay (*Cyanocitta stelleri*) in California between 1959 and 1988 from Christmas Bird Counts (Sauer et al. 1996). Lower and upper 95% confidence intervals (95% CI LB, 95% CI UB, respectively) and number of routes (N) for the linear route regression are also included.

SPECIES	TREND	95% CI LB	95% CI UB	N
Common Raven	4.6***	2.7	6.4	122
American Crow	5.9**	0.3	11.6	124
Steller's Jay	1.3*	0.0	2.6	102

* $P < 0.1$, ** $P < 0.05$, *** $P < 0.01$

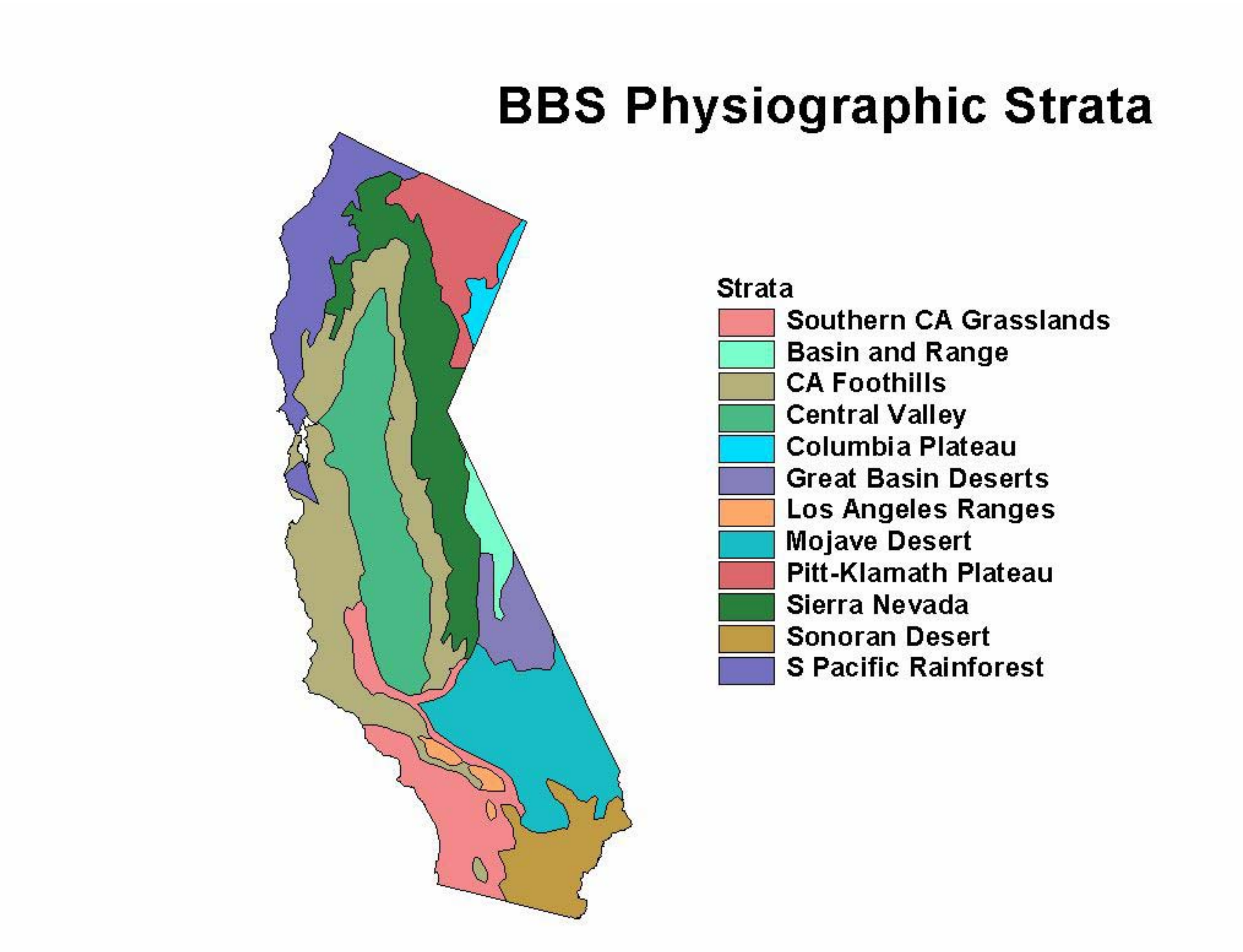

FIGURE 1. Physiographic ecoregions of California developed by USFWS for Breeding Bird Survey analysis.

TABLE 2. Population trends (% change/ yr) of the American Crow (*Corvus brachyrynchus*) in California (and eight physiographic regions) between 1966 and 1999 from Breeding Bird Surveys (from Sauer et al. 1999). Lower and upper 95% confidence intervals (95% CI LB, 95% CI UB, respectively) and number of routes (N) for the linear route regression are also included. Trend estimates with fewer than 15 routes may be unreliable.

REGION	TREND	95% CI LB	95% CI UB	N
California	2.4***	0.9	4.0	109
Columbia Plateau	5.7***	2.6	8.8	45
Pitt-Klamath Plateau	6.3	-8.2	20.9	15
S. California Grasslands	1.7	-6.7	10.2	9
Central Valley	4.1**	0.2	8.1	24
California Foothills	1.8**	0.1	3.5	44
Great Basin Deserts	-2.9	-12.9	7.2	4
Basin and Range	-1.6	-17.8	14.6	17
S. Pacific Rainforest	3.0	-1.3	7.3	65

* $P < 0.1$, ** $P < 0.05$, *** $P < 0.01$

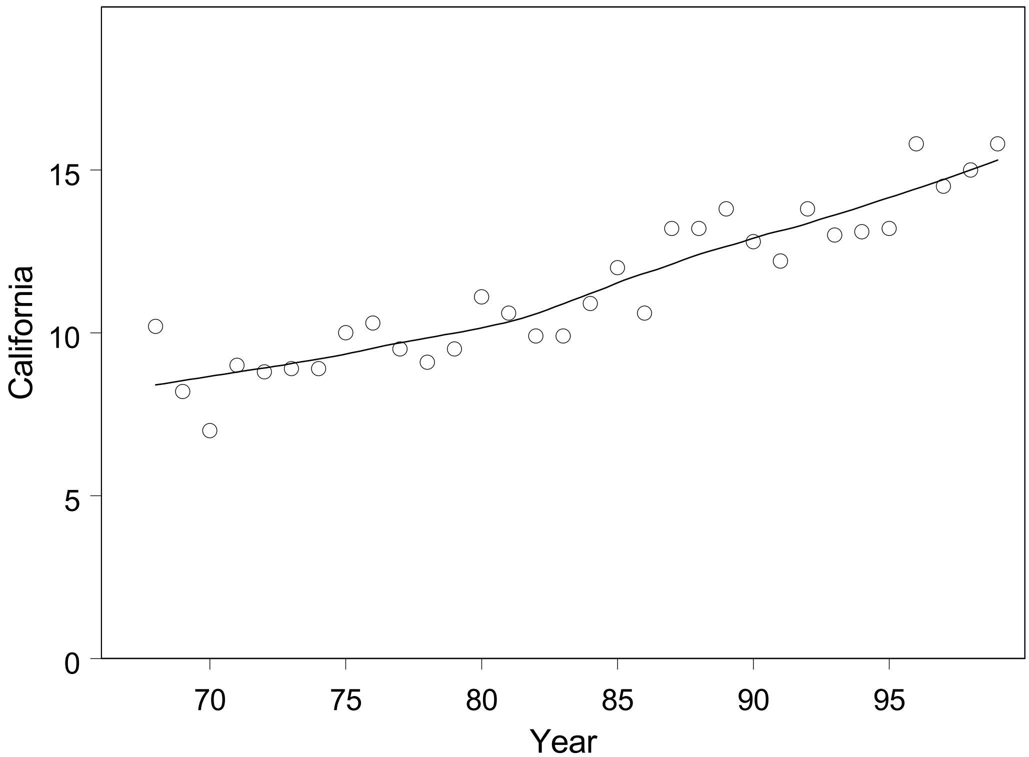

FIGURE 2. Weighted mean number (and loess trend) of American Crows (*Corvus brachyrynchus*) detected along California Breeding Bird Surveys between 1966 and 1999 (from Sauer et al. 1999).

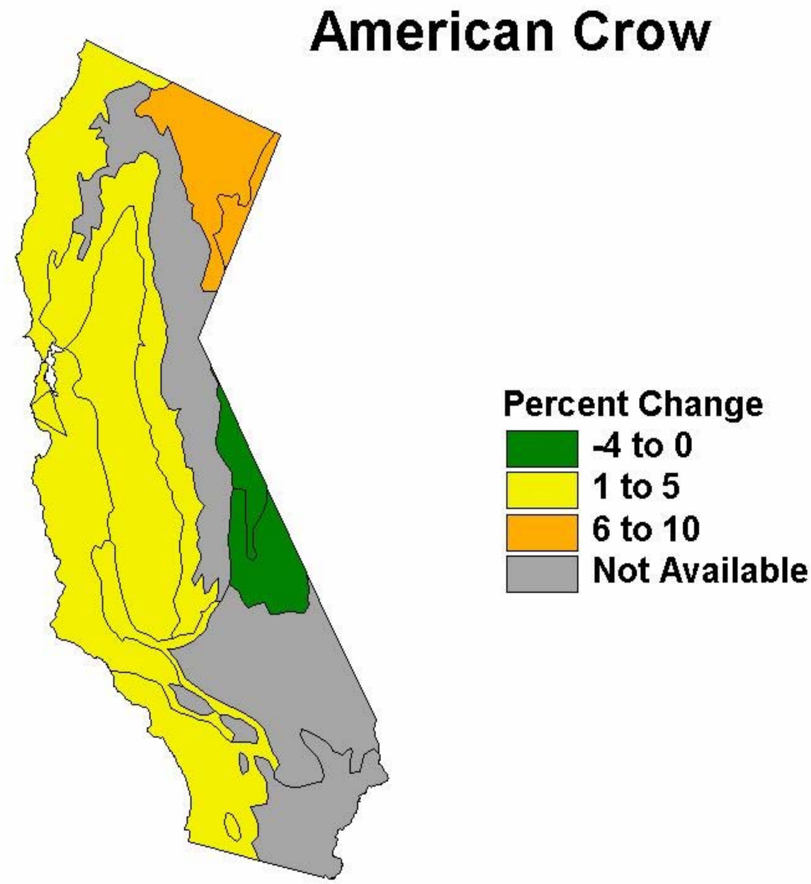

FIGURE 3. American Crow (*Corvus brachyrynchus*) population trend estimates per physiographic ecoregion in California using Breeding Bird Survey data between 1966 and 1999 (from Sauer et al. 1999).

TABLE 3. Population trends (% change/yr) of the Common Raven (*Corvus corax*) in California (and twelve physiographic regions) between 1966 and 1999 from Breeding Bird Surveys (from Sauer et al. 1999). Lower and upper 95% confidence intervals (95% CI LB, 95% CI UB, respectively) and number of routes (N) for the linear route regression are also included.

REGION	TREND	95% CI LB	95% CI UB	N
California	4.1***	2.9	5.4	165
Sierra Nevada	7.9	-3.4	19.2	15
Basin and Range	1.6	-1.7	4.9	43
Mojave Desert	5.4	-1.6	12.4	21
Pitt-Klamath Plateau	13.3***	6.0	25.5	35
S. California Grasslands	2.0	-2.7	6.8	16
Central Valley	4.8*[1]	-0.3	9.9	13
California Foothills	6.0***	2.4	9.7	40
S. Pacific Rainforests	2.2**	0.0	4.4	68
Columbia Plateau	0.4	-5.4	6.2	71
Great Basin Desert	3.8	-2.4	10.1	26
Sonoran Desert	7.1	-2.2	16.3	19
Los Angeles Ranges	6.5**[1]	2.2	10.9	8

* $P < 0.1$, ** $P < 0.05$, *** $P < 0.01$

[1] caution, trends are based on small number of count sites.

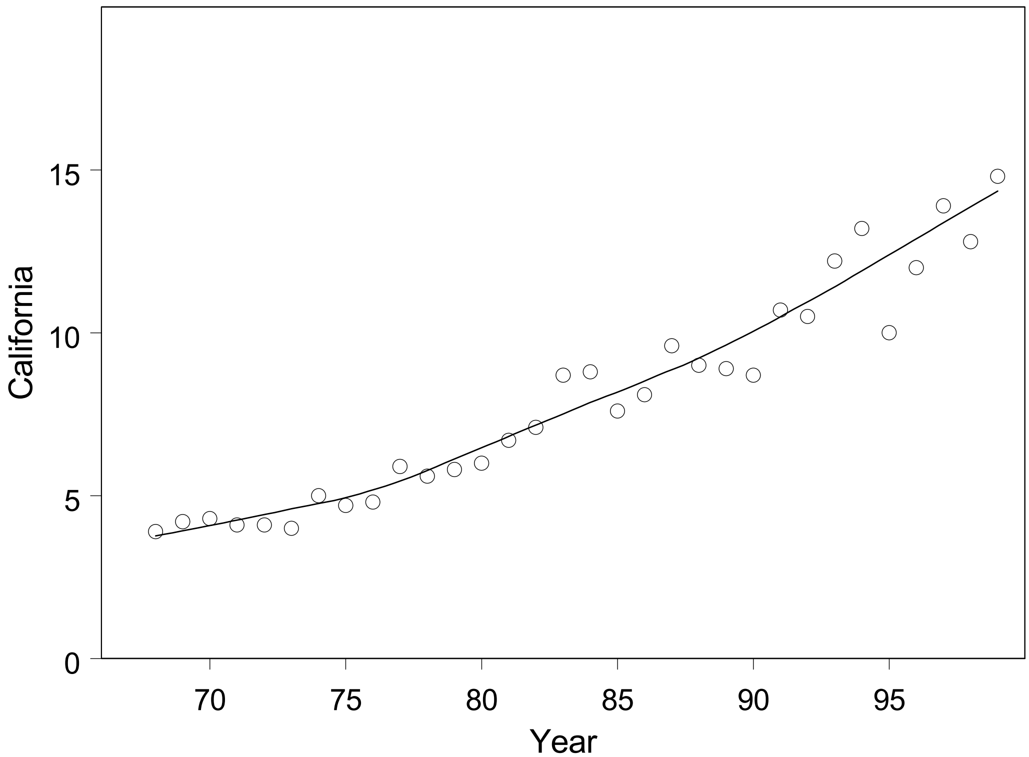

FIGURE 4. Weighted mean number (and loess trend) of Common Ravens (*Corvus corax*) detected along California Breeding Bird Surveys between 1966 and 1999 (from Sauer et al. 1999).

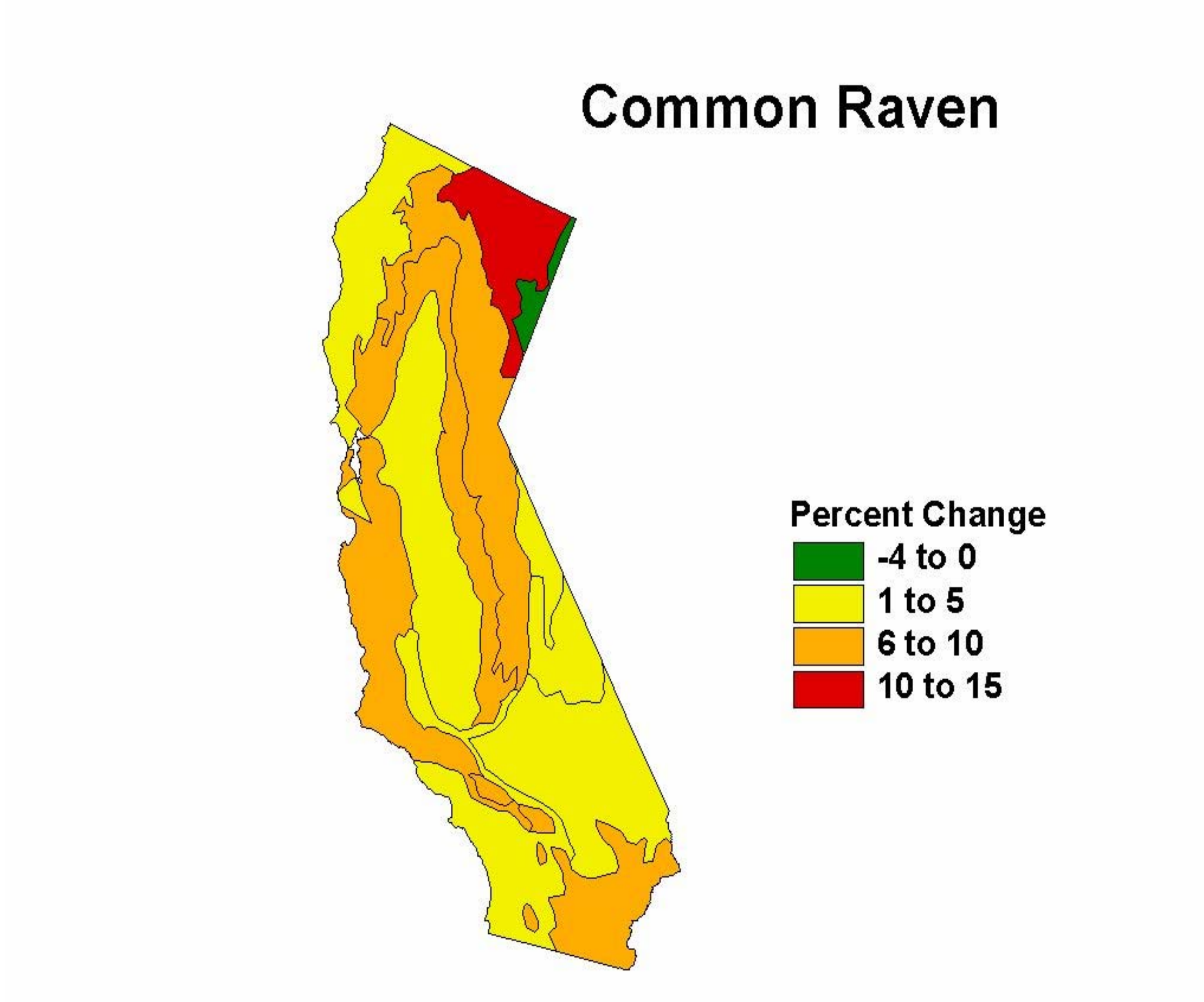

FIGURE 5. Common Raven (*Corvus corax*) population trend estimates per physiographic ecoregion in California using Breeding Bird Survey data between 1966 and 1999 (from Sauer et al. 1999).

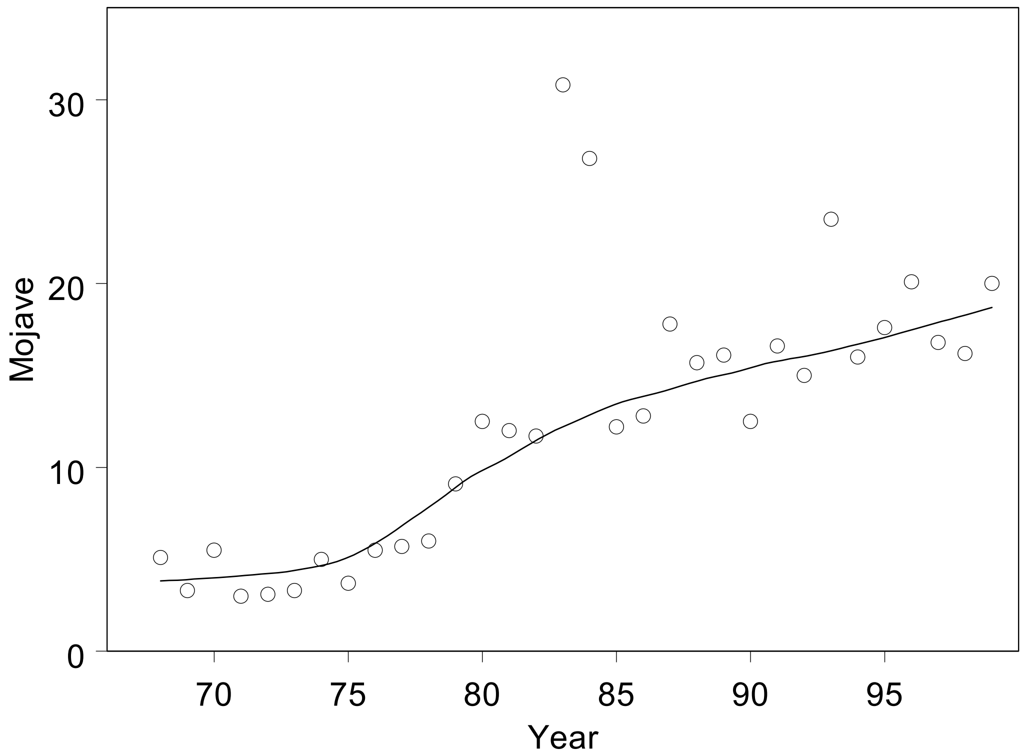

FIGURE 6. Weighted mean number (and loess trend) of Common Ravens (*Corvus corax*) detected along 21 Breeding Bird Surveys in the Mojave desert region of California between 1966 and 1999 (from Sauer et al. 1999).

TABLE 4. Population trends (% change/yr) of the Steller's Jay (*Cyanocitta stelleri*) in California (and seven physiographic regions) between 1966 and 1999 from Breeding Bird Surveys (from Sauer et al. 1999). Lower and upper 95% confidence intervals (95% CI LB, 95% CI UB, respectively) and number of routes (N) for the linear route regression are also included.

REGION	TREND	95% CI LB	95% CI UB	N
California	-0.1	-1.0	0.8	113
Sierra Nevada	-1.9***	-2.8	-0.9	23
Pitt-Klamath Plateau	-0.4	-2.1	1.3	33
S. California Grasslands	-10.3[1]	-47.0	26.4	2
Basin and Range	3.6	-4.9	12.1	8
California Foothills	0.9	-0.2	2.1	36
S. Pacific Rainforests	1.4**	0.3	2.5	69
Los Angeles Ranges	-0.1[1]	-4.9	4.7	9

* $P < 0.1$, ** $P < 0.05$, *** $P < 0.01$

[1] caution, trends are based on small number of count sites.

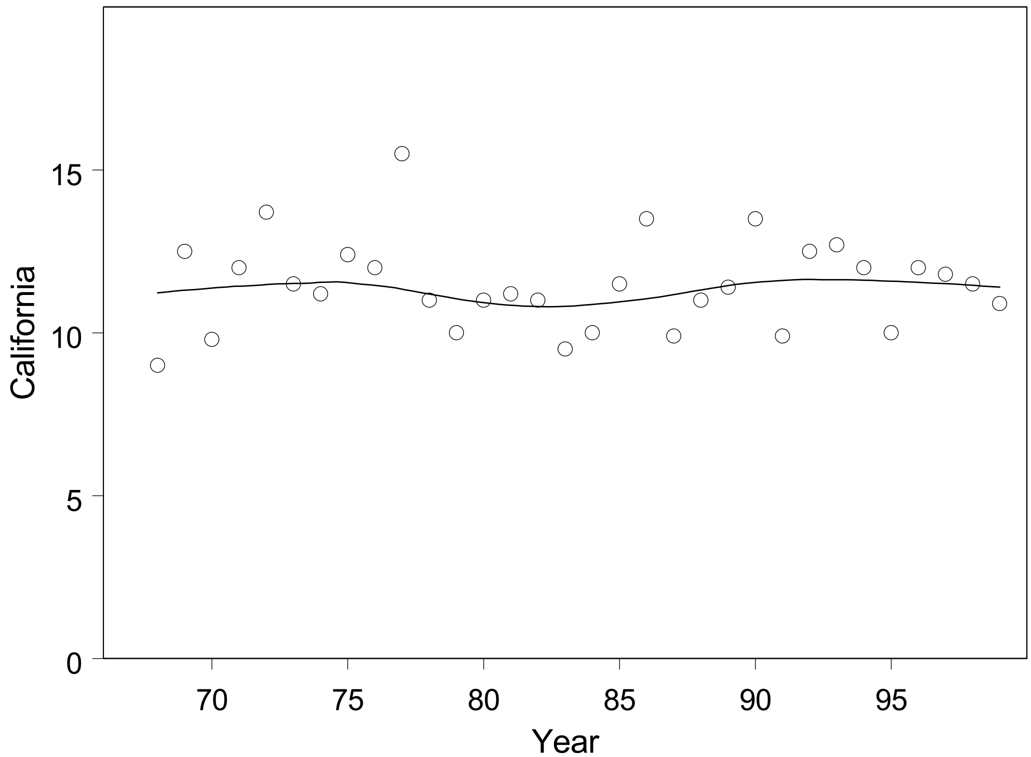

FIGURE 7. Weighted mean number (and loess trend) of Steller's Jays (*Cyanocitta stelleri*) detected along California Breeding Bird Surveys between 1966 and 1999 (from Sauer et al. 1999).

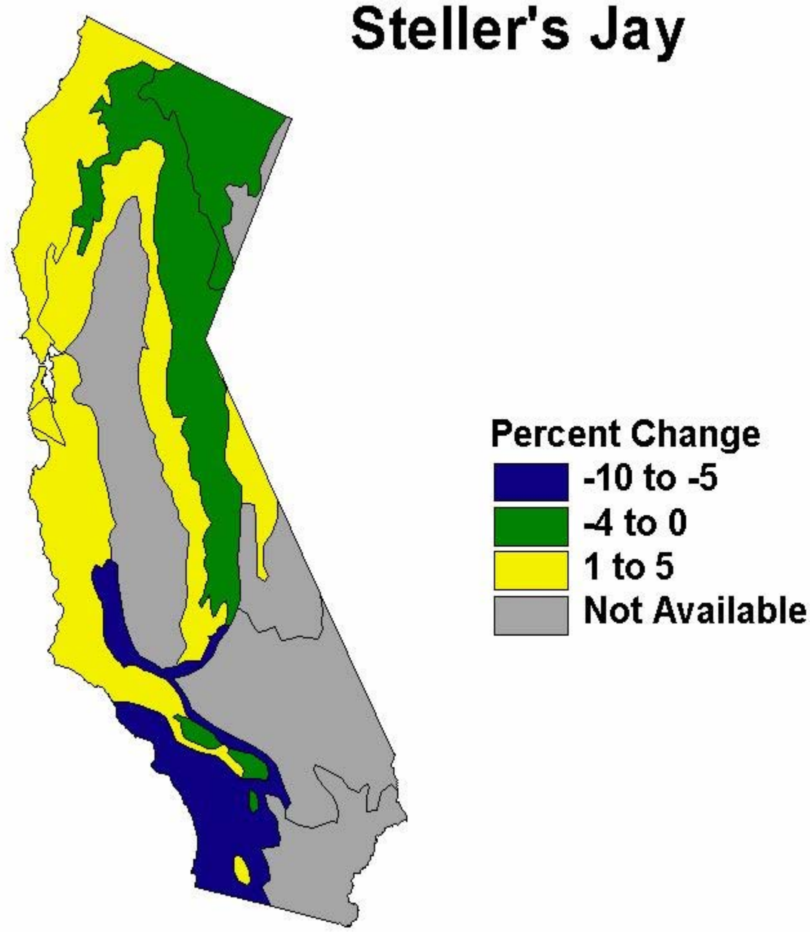

FIGURE 8. Steller's Jay (*Cyanocitta stelleri*) population trend estimates per physiographic ecoregion in California using Breeding Bird Survey data between 1966 and 1999 (from Sauer et al. 1999).

IMPORTANCE OF CORVIDS AS PREDATORS

OVERVIEW

Corvids have been documented preying on the eggs and young of a large number of bird species, from small passerines to large wading birds throughout North America and Europe (Table 5,6). In addition to avian nest predation, corvids kill and eat defenseless young of other taxonomic groups, including the threatened desert tortoise. Corvids prey on nests in open and forested locations, although predation rates are often highest along habitat edges and fragments (Andrén 1992). In a recent review of nest predation in fragmented habitats, 22 of 47 published studies from North America and Europe implicated corvids as important nest predators (Marzluff and Restani 1999).

Although nest predation by corvids is well documented in some cases, in others, corvids have been implicated as important predators with little evidence. Identification of corvids preying on nests or young ranges from conclusive (direct observation), to circumstantial, to purely conjectural evidence. Circumstantial evidence may be reliable in some cases (tracks leading to a recently depredated nest). However, identifying predators based on nest disturbance or egg/nestling remains is often unreliable (Larivière 1999).

Many studies that implicate corvids as important nest predators have been conducted with artificial nests. Artificial nests may not elicit the same cues to potential predators as real nests and, thus, may bias results toward a certain type of predator (Willebrand and Marcström 1988). However, artificial nests are advantageous in that large numbers can be monitored simultaneously, allowing comparison of predation rates between sites (e.g., edge vs. interior). Artificial nests also provide the only means of investigating nest predation in species whose real nests are rarely found (e.g., Marbled Murrelet). Thus, artificial nests are an important tool in nest predation research.

The goal of this section is to provide a comprehensive, detailed account of the importance of corvids as nest predators and as predators of threatened and endangered species in California. In this review, special attention is paid to the accuracy of predator identification and to the importance of corvid predation at specific sites.

We rate "predator identification reliability" on a scale of 1-4: 1 = Direct evidence - photograph or observations of corvids preying on eggs or young, 2 = Strong circumstantial evidence - evidence based on track plates, beak (or puncture) marks on artificial eggs, and prey remains that implicate corvids, prey remains at corvid nest or roost sites, tracks near depredated nests within 2 days of the event, 3 = Weak circumstantial evidence - tracks in nesting area, damaged (real) eggs, depredated nest characteristics, predator seen in vicinity of nest within same time period as the predation event, 4 = Conjecture - based on predator monitoring at site or hunting behavior by corvids in nesting area.

We rate "predator importance" on a scale of 1-4: 1 = Primary predator - >50% of predation events or only known predator at site, 2 = Secondary predator - 25-49% of predation events or second most important predator at site, 3 = Occasional predator - <25% of predation events, 4 = Unknown - not reported in the study.

We provide two summaries of nest predation by corvids. In both summaries we focus our analyses on the three most important corvid nest predators in western North America

(American Crow, Common Raven, and Steller's Jay). First, we summarize the importance of corvids as nest predators of species that are not listed as threatened or endangered (Table 5). Next we focus on corvids as nest predators (and predators of juveniles in the case of the desert tortoise) for the five threatened and endangered species in California and neighboring areas that are most affected by corvid predation (Table 6).

DOCUMENTATION OF THE COMMON RAVEN, AMERICAN CROW, AND STELLER'S JAY AS NEST PREDATORS

We reviewed recent published literature (1980 to the present) that implicated corvids as nest predators. We also reviewed unpublished literature specific to California. We only included studies that implicated corvids as nest predators with substantial evidence (i.e. we did not include studies where corvids were listed as "potential predators") and that identified corvids to species. Our review included studies of both artificial and real nests.

We found 33 sources that implicate the Common Raven, American Crow, or Steller's Jay as nest predators. Most documentation exists for the Common Raven and American Crow. Nine studies provide direct evidence of corvid predation at real nests (including all 3 species) (Table 5). Eleven studies document direct evidence of corvid predation at artificial nests (including all 3 species) (Table 5). Less than half the reported studies (15 of 33) relied on circumstantial or conjectural evidence to document corvid nest predator identity. Fourteen of the 33 (42%) studies reported corvids as the most important nest predator at the site. Predator importance in the remaining sources was not reported or corvids were secondary or occasional predators.

This review clearly shows that crows, ravens, and jays have been conclusively identified as nest predators and that they are often the most important nest predators at specific sites

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DOCUMENTATION OF CORVIDS AS PREDATORS OF THREATENED AND ENDANGERED SPECIES IN WESTERN NORTH AMERICA

We reviewed both the primary and the gray literature for studies that document corvids as predators of threatened and endangered species in California and neighboring states. As in the previous review, we only included studies that implicated corvids as nest predators with substantial proof (i.e. we did not include a study if corvids were listed as "potential predators") and that identified corvids to species.

We found 55 published and unpublished sources that provide evidence for corvids as predators of eight listed species in California or neighboring states. The bulk of the studies implicate the Common Raven and the American Crow; very few studies have identified the Steller's Jay as a predator on threatened and endangered species. All three species prey on both eggs and nestlings of the listed bird species. Common Ravens also prey on juvenile desert tortoise. Fourteen of 55 sources (25%) provide direct evidence of predation by corvids. Corvids were the most important predators (in at least one site) in 23 of the 55 (42%) studies (Table 6).

Although in this review we only focus on the 5 listed species with copious evidence of corvid impact, other listed species in California are negatively impacted by corvids.

Snyder and Snyder (2000) report that ravens were probably the most important threat to condor nesting success in the 1980's when they were still nesting in the wild in California. Ravens were involved in at least 5 cases of egg breakage to condor eggs (Snyder and Snyder 2000). Common Ravens have been observed preying on the eggs and nestlings of the San Clemente Loggerhead Shrike (Scott and Morrison 1990). In addition, Petersen (2001) provides direct evidence that Western Scrub Jays are the most important nest predator of the Least Bell's Vireo at a site in San Diego County. Western Scrub Jays have also been identified as important egg predators (12 of 39 events) at artificial nests at the San Joaquin Experimental Range, near Fresno, California (Purcell and Verner 1999). However, we found no other published studies that provide direct evidence of scrub jays as important nest predators.

TABLE 5. Summary of published studies (1980-2000) documenting corvid predation on both artificial and real nests.

Predator	Location	Prey species	Prey Item	source[1]	Pred. ID reliab.[2]	Pred. Import.[3]
Common Raven	E. Digges Is., Canada	Brunnich's Guillemot	Eggs / nestlings	1	1	1
Common Raven	Islands in Lake Huron	Herring/Ring-B. gulls	Eggs	3	1	1
Common Raven	East Digges Is., Canada	Thick-billed Murre	Eggs / nestlings	4	1	2
Common Raven	South-central Sweden	Artificial nest	Chicken eggs	5	2	1
Common Raven	Camp Pendleton, CA	Artificial nest	*Coturnix* spp. eggs	6	1	4
Common Raven	Washington State	Artificial nest*	Live pigeon nestling	7	1	3
Common Raven	Sweden	Artificial nest	Chicken egg	28	4	1
Common Raven	Point Reyes, CA	Common Murre	Eggs / nestlings	14	1	1
American Crow	S-C Saskatchewan	Wilson's Phalarope	Eggs	23	1	4
American Crow	Ithaca, New York	Artificial nest	Clay eggs	24	4	4
American Crow	Sandy Pt., CT	Least Tern	Eggs	8	2	2
American Crow	Front Royal, VA	Artificial nest	Northern Bobwhite eggs	26	1	3
American Crow	Centre County, PA	Artificial nest	Chicken eggs	10	3,4	4
American Crow	Illinois and Iowa	Artificial nest	Chicken eggs	9	3	4
American Crow	Can. Prarie Pothole	Ducks (>8spp.)	Eggs	11	4	2,3
American Crow	SW Manitoba, Canada	Artificial nest	Chicken eggs	15	2,3	1
American Crow	Saskatchewan, Canada	Artificial nest	Chicken eggs	16	1,2	4
American Crow	Saskatchewan, Canada	Artificial nest	Chicken eggs	17	1,2	4
American Crow	Centre County, PA	Artificial nest	Ceramic eggs	21	2,3	1
American Crow	Ipswich, MA	Piping Plover	Eggs	22	3	2
American Crow	Carter County, OK	passerine birds	Nestlings	12	1	4
American Crow	Centre County, PA	Artificial nest	Chicken eggs	18	3,4	1
American Crow	Bossier City, LA	Artificial nest	*Coturnix* spp. eggs	31	1	1
American Crow	Smoky Mtn. Natl. Pk., TN	Wood Thrush	Eggs / nestlings	32	1	3
American Crow	Ontario, Canada	Artificial nest	*Coturnix* spp. eggs	33	1	4
Steller's Jay	SE Alaska / BC Canada	Artificial nest	*Coturnix* spp. eggs	27	1	2
Steller's Jay	Colorado Front Range	Artificial nest	Plasticine eggs	29	2	1
Steller's Jay	Siskiyou County, CA	Dusky Flycatcher	Budgerigar/D. dove eggs	19	1	4
Steller's Jay	Humboldt, County, CA	Artificial nest	*Coturnix* spp. eggs	30	1	4
American Crow/Steller's Jay	Olympic peninsula, WA	Artificial nest*	Artificial egg / nestlings	25	1	3C ,1S
Common Raven/American crow/Steller's Jay	Flagstaff, AZ	Pinyon Jay	Eggs / nestlings	2	3	1C,R, 3S
Common Raven/American crow	Country Is., Nova Scotia	Roseate Tern	Eggs	20	1	1
Common Raven/American crow/Steller's Jay	Olympic Peninsula, WA	Artificial nest*	Artificial eggs / nestlings	13	1	1**R, 4C,S

* simulated MAMU nest, ** incubation stage only, C American Crow, R Common Raven, S Steller's Jay

[1] Source: 1=Gaston & Elliot (1996), 2=Marzluff (1988), 3=Ewins (1991), 4=Gaston et al. (1985), 5=Andrén (1992), 6=Avery et al. (1995), 7=Bradley and Marzluff (in press), 8=Brunton (1997), 9=Dimmick and Nicolaus (1990), 10=Yahner & Cypher (1987), 11=Johnson et al. (1989), 12=Freeman (1993), 13=Marzluff et al. (1996), 14=Roth et al. (1999), 15=Sullivan & Dinsmore (1990), 16=Sugden & Beyersbergen (1986), 17=Sugden & Beyersbergen (1987), 18=Yahner & Wright (1985), 19=Liebezeit (2001), 20=Whittam and Leonard (1999), 21=Yahner & DeLong (1992), 22=Rimmer & Deblinger (1990), 23=Colwell & Oring (1988), 24=Haskell (1995), 25=Luginbuhl et al. (in press), 26=Leimgruber et al. (1994), 27=Sieving & Willson (1998), 28=Andrén et al. (1985), 29=Craig (1997), 30=Brand & George (in press), 31=Buler & Hamilton (2000), 32=Farnsworth & Simons (2000), 33=Picman (1987)

[2] Predator identification reliability rating:

1 = Direct evidence – photograph or observed

Table 5. Summary of published studies (1980-2000) documenting corvid predation on both artificial and real nests (continued).

2 = Strong circumstantial evidence - track plates, beak (or puncture) marks on artificial eggs
3 = Weak circumstantial evidence - tracks in nesting area, damaged (real) eggs, nest remains and appearance
4 = Conjecture - based on predator monitoring or "indices of predator activity" at a site, hunting behavior by corvids in nesting area

[3] Predator importance rating:
1 = Primary predator - >50% of predation events or only known predator at site
2 = Secondary predator - 25-49% of predatoion events or secondmost out of a group.
3 = Occasional predator
4 = Unknown or not documented in the study

TABLE 6. Summary of published and unpublished papers documenting corvid predation of threatened and endangered species (nest contents or young) that occur in California.

Prey species	Predator	Location	Prey Item	source[1]	Pred. ID reliab.[2]	Pred. Import.[3]
CALIFORNIA						
Snowy Plover	Common Raven/American Crow	Batiquitos Lagoon, San Diego Co.	Eggs, nestlings	2	2R,3R,4C	1R, 3C
Snowy Plover	Common Raven	Mono Lake	Eggs	3	2,3	3,2
Snowy Plover	Common Raven	Humboldt Co.	Eggs	4	2,3	4
Snowy Plover	Common Raven	Point Reyes National Seashore	Eggs	5	2	1
Snowy Plover	Common Raven	Humboldt Co.	Eggs	6	2	4
Snowy Plover	Common Raven	Mono Lake	Eggs	18	2	2
Snowy Plover	Common Raven	Point Reyes National Seashore	Eggs	31	2,3	1
Snowy Plover	Common Raven/American Crow	Central California Coast	Eggs	31	2,3	1R,8, 3C,9
Snowy Plover	Common Raven/American Crow	Central California Coast	Eggs	32	2,3	1R,8, 1R,10, 2C,9, 3C,10
Snowy Plover	Common Raven	Camp Pendleton, San Diego Co.	Eggs	44	2,3	4
Snowy Plover	Common Raven	San Diego Co.	Eggs	45	2,3	1
Snowy Plover	Common Raven/American Crow	San Diego Co.	Eggs	46	2,3	1
Snowy Plover	Common Raven	Camp Pendleton, San Diego Co.	Eggs	47	2,3	2
Snowy Plover	American Crow	Vandenberg AFB, Santa Barbara Co.	Eggs	48	2,3	2
Snowy Plover	American Crow	Vandenberg AFB, Santa Barbara Co.	Eggs	49	2,3	2
Snowy Plover	American Crow	Vandenberg AFB, Santa Barbara Co.	Eggs	50	2,3	2
Snowy Plover	American Crow	Vandenberg AFB, Santa Barbara Co.	Eggs	51	2,3	2
Snowy Plover	American Crow	Vandenberg AFB, Santa Barbara Co.	Eggs	52	2,3	2
Snowy Plover	American Crow	Vandenberg AFB, Santa Barbara Co.	Eggs	53	2,3	2
Snowy Plover	American Crow	Vandenberg AFB, Santa Barbara Co.	Eggs	54	2,3	2
Snowy Plover	Common Raven	Point Reyes National Seashore	Eggs	55	2,3	1
California Least Tern	Common Raven	California coast**	Eggs, nestlings	22	1,2,3,4	3
California Least Tern	Common Raven/American Crow	California coast**	Eggs, nestlings	23	1,2	3, 1C,3
California Least Tern	Common Raven/American Crow	California coast**	Eggs, nestlings	24	1,2,3,4	3, 1C,3
California Least Tern	Common Raven/American Crow	California coast**	Eggs, nestlings	25	1,2,3,4	3
California Least Tern	Common Raven/American Crow	California coast**	Eggs, nestlings	26	1,2,3,4	2R,4,5,2C,5,6,3C
California Least Tern	Common Raven	California coast**	Eggs, nestlings	27	1,2,3,4	3
California Least Tern	Common Raven/American Crow	California coast**	Eggs, nestlings	28	1,2,3,4	3R, 2C,7,3C
California Least Tern	Common Raven/American Crow	California coast**	Eggs, nestlings	29	1,2,3,4	3R, 2C
California Least Tern	Common Raven/American Crow	California coast**	Eggs, nestlings	30	1,2,3,4	3
Desert Tortoise	Common Raven	Mojave desert (DTNA)	Juveniles	10	2	4
Desert Tortoise	Common Raven	Mojave desert	Juveniles	11	2	4
Desert Tortoise	Common Raven	Mojave desert (Kramer study plot)	Juveniles	12	2	4
Desert Tortoise	Common Raven	East Mojave desert	Juveniles	16	2	4
Desert Tortoise	Common Raven	E. Mojave and N. Colorado deserts	Juveniles	17	2	4
Marbled Murrelet	Steller's Jay	Santa Cruz Co.	Nestlings	8	1	4
Marbled Murrelet	Common Raven	Santa Cruz Co.	Eggs	8	1	4
Least Bell's Vireo	Western Scrub Jay	San Luis Rey River, San Diego,Co.	Eggs or nestlings?	19	1	1
San Clemente Loggerhead Shrike	Common Raven	San Clemente Island	Eggs, nestlings	43	1	2 or 3

Table 6. Summary of published and unpublished papers documenting corvid predation of threatened and endangered species (nest contents or young) (continued).

Sandhill Crane	Common Raven	Northeastern California	Eggs	7	2	2
California Condor	Common Raven	California	Eggs	1	1,2	1
OREGON						
Snowy Plover	Common Raven/American Crow	Oregon coast	Eggs	15	4	1
Snowy Plover	American Crow	Southern Oregon coast	Eggs	33	2,3	2
Snowy Plover	Common Raven	Southern Oregon coast	Eggs	34	2,3	1
Snowy Plover	Common Raven	Southern Oregon coast	Eggs	35	2,3	1
Snowy Plover	Common Raven	Southern Oregon coast	Eggs	36	2,3	1
Snowy Plover	Common Raven/American Crow	Southern Oregon coast	Eggs	37	2,3	1
Snowy Plover	Common Raven/American Crow	Southern Oregon coast	Eggs	38	2,3	1
Snowy Plover	Common Raven/American Crow	Southern Oregon coast	Eggs	39	2,3	1
Snowy Plover	Common Raven/American Crow	Southern Oregon coast	Eggs	40	2,3	4
Snowy Plover	Common Raven/American Crow	Southern Oregon coast	Eggs	41	2,3	1
Snowy Plover	Common Raven/American Crow	Southern Oregon coast	Eggs	42	2,3	1
Marbled Murrelet	Common Raven	Valley of Giants, Cape Creek	Eggs	9	3	1
Marbled Murrelet	Steller's Jay or Gray Jay	Siuslaw #2	Nestling	9	2,3	4
Sandhill Crane	Common Raven	Malheur NWR, Harney Co.	Eggs	13	2	1
Sandhill Crane	Common Raven	Malheur NWR, Harney Co.	Eggs	14	2	2
Sandhill Crane	Common Raven	Eastern Oregon	Eggs	20	2	2
ALASKA						
Marbled Murrelet	Steller's Jay	Naked Island, Alaska	Egg	21	3	4

* only during nestling stage, C American Crow, R Common Raven, S Steller's Jay, ** see Appendix B for full list of study sites

3 Venice Beach, 4 NAS Alameda, 5 Terminal Island, 6 VAFB Purisima Point, 7 Seal Beach, 8 Pt. Reyes Natl. Seashore, 9 Atascadero, 10 Wilder R = Common Raven, C = American Crow

[1] Source: 1=Snyder & Snyder (2000), 2=Powell & Collier (2000), 3=Page et al. (1983), 4=Transou & LeValley (2000), 5=White & Hickey (1997), 6=LeValley(1999), 7=Littlefield (1995b), 8=Singer et al. (1991), 9=Nelson & Hamer (1995a), 10=Campbell (1983), 11=Berry (1985), 12= Woodman & Juarez (1988), 13=Littlefield & Thompson (1985), 14=Littlefield (1995a), 15=Wilson-Jacobs & Meslow (1984), 16=Camp et al. (1993),17=Farrel (1991), 18=Page et al. (1985), 19=Peterson (2001), 20=Littlefield (1999), 21=Naslund et al. (1995), 22=Obst & Johnston (1992), 23=Johnston & Obst (1992), 24=Caffrey (1993), 25=Caffrey (1994), 26=Caffrey (1995a), 27=Caffrey (1997), 28=Caffrey (1998), 29=Keane (1998), 30=Keane (1999), 31=Page (1988), 32=Page (1990), 33=Stern et al. (1990), 34=Stern et al. (1991), 35=Craig et al. (1992), 36=Casler et al. (1993), 37=Hallett et al. (1995), 38=Estelle et al. (1997), 39=Castelein et al. (1997), 40=Castelein et al. (1998), 41=Castelein et al. (2000a), 42=Castelein et al. (2000b) 43=Scott & Morrison (1990), 44=Powell & Collier (1995), 45=Powell et al. (1996), 46=Powell et al. (1997), 47=Collier & Powell (2000), 48=Persons (1995a), 49=Persons (1995b), 50=Persons & Applegate (1996), 51=Persons & Applegate (1997), 52=Applegate & Schultz (1999), 53=Applegate & Schultz (2000), 54=Applegate & Schultz (2001) 55=Abbott & Peterlein (2001)

[2] Predator identification reliability rating:
1=Direct evidence - photograph or observed
2=Strong Circumstantial evidence - corvid evidence at prey (nest) remains, prey remains at corvid nest or under roost, tracks at depredated nest within 1 or 2 days of event, in pellet
3=Weak Circumstantial evidence - predator seen in nesting area within same time period as predation event, nest remains, tracks in nesting area
4=Conjecture - potential predator seen at study site

[3] Predator Importance rating - (see table 5)

MANAGEMENT TECHNIQUES USED TO PROTECT THREATENED AND ENDANGERED SPECIES FROM CORVID PREDATION

Corvid management techniques that have been employed to protect threatened and endangered species can be lumped into three general categories: lethal removal, behavioral modification, and habitat modification (Boarman and Heinrich 1999).

In the following section, we describe each approach and summarize management actions used to reduce corvid predation on five threatened or endangered species. We searched both the primary and gray literature for specific management techniques and their effectiveness. At the end, we summarize the advantages and disadvantages of each control technique and suggest research directions that may lead to new and improved corvid management techniques.

LETHAL REMOVAL

Lethal removal involves shooting or poisoning of the suspected predator. In some cases, predators may be trapped and subsequently euthanized. Lethal control may involve large-scale eradication of many individuals or the removal of specific "problem" individuals that are thought to be responsible for most predation at a particular site. Lethal removal is usually performed near the nest sites or colonies of the protected species. In some cases, attempts have been made to remove large numbers of corvids at specific locations (Littlefield and Thompson 1985, Boarman 2000).

Shooting is the simplest, and in some cases, most expedient means of removing predators. This method is advantageous in that specific individuals can be targeted. However, after initial removal of an individual it is often very difficult to remove subsequent individuals (J. Turman, W. Boarman, pers. comm.). In some cases, shooting efficiency can be improved by employing a stuffed corvid predator (e.g. owl or fox) as an "attractant" (with or without bait) (Slagsvold 1978) or with playbacks (Chesness et al. 1968). Slagsvold (1978) killed approximately 12 of 40 (30% success rate) Hooded Crows (*Corvus corone cornix*) that were attracted to a stuffed Eagle Owl (*Bubo bubo*). Although shooting can be effective and relatively inexpensive in the short-term, it can be costly, time-consuming, less effective over the long term. Shooting is also unpopular with the general public in some areas and may be prohibited in suburban or urban areas.

The most commonly used chemical to lethally remove corvids is DRC-1339 (3-chloro-4-methylbenzenamine HCL) (Schafer 1984, Rado 1993), developed by the U. S. Fish and Wildlife Service (USFWS 1985). This poison is injected into hard-boiled eggs (chicken or *Coturnix* sp.) or applied to meat baits where corvids have been observed feeding (Knittle and Orr 1988). The toxicity of DRC-1339 varies greatly among species and species groups, and is reported to be most acute in starlings, doves, game birds, and corvids (LD50's < 10mg/kg) (Schafer 1984, M. Avery pers. comm.). In contrast, House Sparrows (*Passer domesticus*) and all raptors that have been tested are relatively insensitive (LD50's in the hundreds of mg/kg) (M. Avery, pers. comm.). Non-violent death caused by kidney failure or central nervous system depression results in 1-2 days (DeCino et al. 1966) or up to 4 days depending on the dosage (M. Avery, pers. comm.). Potential for poisoning of secondary consumers is reported to be very low (Schafer 1984). Use of DRC-1339 is governed by Environmental Protection Agency (EPA) registration No. 56228-29. Under this label, DRC-1339 is a restricted use pesticide that can be used only by the USDA Wildlife Services (WS) personnel or their designees.

Trapping is also used for lethal, as well as non-lethal corvid management. Corvids have been successfully trapped using "drop in" traps (modified Australian crow traps), often with a live or stuffed corvid decoy inside (Stiehl 1978, Linz et al. 1990, Avery et al. 1993), rocket/cannon nets (Mahringer 1970, Dorn 1972, Stiehl 1978, Linz et al. 1990), single-end Havahart© traps (Schwan and Williams 1978), net gun, dho gaza, or bow nets (Linz et al. 1990), padded leg-hold traps (Engel and Young 1989a, Linz et al. 1990), and box traps (9"x 9"x 27" Tomahawk© traps) (D. Garcelon, pers comm.).

The most successful trap type varies from site to site. Engel and Young (1989a) reported the best success using leg-hold traps placed next to a carcass. Out of 6 trap types, Linz et al. (1990) found modified Australian crow traps were the most efficient. In most cases, effective trapping is only moderately successful because it is difficult to capture large numbers of birds at one time.

Lethal removal is often used when immediate reduction in the corvid population is necessary. Nevertheless, large-scale reductions are, at best, temporary with no carryover benefits one year after removal (Chesness et al. 1968, Slagsvold 1978, Rado 1993). Although corvid removal may initially result in decreased nest predation of the protected species, compensatory predation by non-corvids sometimes nullifies this benefit (Parr 1993, Parker 1984, Broyer et al. 1995). Selectively eliminating "problem birds" can be effective (Caffrey 1993, T. Applegate, pers. comm.) although it may require constant effort if the removed individuals are repeatedly replaced by others. In some cases, the only quick and effective means of stopping rampant nest predation at specific sites is selective removal (see Caffrey 1993).

When territorial birds are lethally removed, non-breeding birds often quickly re-occupy the vacant area. Shooting and trapping may also become more difficult as the corvids become wary. For these reason, successful lethal removal may require constant effort for an indefinite period of time with the initial cost-effectiveness lost over the long-term. Lethal removal is not acceptable to many people for ethical reasons, especially when removal occurs on a large scale. Therefore, agencies that employ lethal removal may face resistance from the public (Boarman 1993).

BEHAVIORAL MODIFICATION

Conditioned Taste Aversion

Conditioned taste aversion (CTA) (also known as taste aversion conditioning, conditioned flavor aversion, and "conditioned food aversion based on deception" – CFABD) involves the distribution of baits that mimic the taste, scent, and appearance of live prey with the intention of preventing depredation. If these baits are laced with an illness-causing substance in a way that the taste and other characteristics of the bait remain the same as live prey, predators associate illness from consuming baits with these characteristics and avoid both baits and target prey. Successful application of CTA to wildlife management problems includes distributing baits that closely mimic the taste, scent, and appearance of target prey and that cause a high intensity of illness after suitable delay (Revusky 1968).

A number of different chemicals have been applied to baits to establish CTA on mammalian (Gustavson et al. 1974, Nicolaus et al. 1989b, Semel and Nicolaus 1992) and avian predators (Dimmick and Nicolaus 1990, Nicolaus et al. 1983, Nicolaus 1987, Nicolaus et al. 1989c, Avery et al. 1995). The best substances should produce severe short-term illness, be undetectable to the predator, and the effective dose should be much less than the lethal dose (Nicolaus et. al. 1989a).

CTA has been established on free-ranging corvids in a number of experimental studies. Carbachol (carbamylcholine chloride) produced CTA in both captive and free-ranging American Crows (Nicolaus et al. 1983, Nicolaus et al. 1989c). At Malheur National Wildlife Refuge (MNWR) in Oregon, Nicolaus (1987) found that breeding Common Ravens acquired CTA after consuming as little as a single treated egg (treated with Landrin) and that conditioned territorial ravens not only avoided preying on surrogate Sandhill Crane eggs (i.e., turkey eggs dyed to resemble crane eggs) within their territories, but effectively excluded non-resident ravens from entering crane nesting areas because they aggressively defended their own breeding territory. Although ravens clearly avoided surrogate crane eggs, avoidance of real crane eggs was not studied.

Methiocarb was successfully used to condition territorial ravens (within 4-5 days) to avoid preying on Least Tern eggs within their territories at Camp Pendleton, California (Avery et al. 1995). The conditioned ravens also excluded non-breeding ravens from their territories as had been previously documented at MNWR (Nicolaus 1987). In most studies of CTA, conditioned corvids avoided baits and target prey wherever they were presented, suggesting that location was not important in the acquisition or expression of CTA (Nicolaus et al. 1983, Nicolaus 1987, Dimmick and Nicolaus 1990). However, Avery et al. (1995) found that conditioning was site-related (not transferable to eggs encountered in different locations). It is possible that the ravens were able to detect methiocarb in treated eggs on subsequent encounters (M. Avery, pers. comm.), leading to site-dependent conditioning. The Environmental Protection Agency (EPA) has registered methiocarb (registration No. 56228-33) for aversive conditioning and kits can be purchased from the USDA Wildlife Services suppy depot in Pocatello, ID (M. Avery, pers. comm.). No other substances have been officially approved for use in aversive conditioning.

Although CTA has been shown to work with a variety of both captive and free-ranging vertebrate predators on an experimental basis, this method of predator control has not been performed on a large scale or over a long period for the management of any listed species. This inaction is due to probable limitations in the application of CTA and difficulties and costs involved in registering CTA agents. A major limitation in the application of CTA is that CTA in corvids has only been successfully established using eggs as baits. Therefore, protection from the conditioned predator is restricted to the laying and incubation stages. It is possible to establish CTA on fledglings/juveniles, although, it would be unethical to spare the young of endangered species for this purpose. Deceased young found in breeding areas (or possibly the young of similar unprotected species) could be used as baits to attempt to establish CTA to nestlings.

Initially, CTA may be more labor-intensive than other types of predator control. It requires finding suitable mimic baits, setting out treated baits, monitoring them during the conditioning phase, and knowledge of the local corvid population distribution (location of nests, territory size and boundaries, and movements of non-territorial birds) (Avery et al.1993). However, once a suitable strategy for CTA treatment is established, conditioning should occur relatively quickly (Nicolaus et al. 1983, Nicolaus 1987, Avery et al. 1995), may be retained for long periods (over one year) (Dimmick and Nicolaus 1990, Semel and Nicolaus 1992), and can be established on a guild of nest predators (Nicolaus 1987). If territorial corvids are conditioned, protection may be increased further since the conditioned birds exclude conspecifics (Nicolaus 1987, Avery et al. 1995).

The most effective and practical application of CTA to protect listed species is in situations where a relatively concentrated group of breeders is also within the territory of one or a few predators. Of the five listed species covered in this report, the California Least Tern, Sandhill Crane, and in some cases the Snowy Plover, are prime candidates for protection with CTA. These species usually breed colonially (terns) or within relatively close proximity to one another (crane and plovers) and are often found within one or a few territories of corvids and mammalian predators. In some areas of California, Snowy Plovers nest within Least Tern colonies (see Collier and Powell 2000). CTA may be less practical in protecting the Marbled Murrelet because their nests are usually dispersed across a large area (not to mention difficult to find) and would require a large effort to protect few nests. It may be possible to protect young desert tortoises by establishing CTA on raven pairs that are known to specialize on desert tortoise young. However, some type of tortoise surrogate (road kill tortoises or another similar species) would have to be used to establish CTA.

The efficacy of CTA may be compromised in situations where the conditioned predators are not territorial, permitting an influx of unconditioned conspecifics. Territorial behavior of corvid predators varies from species to species and may vary by location. Although ravens are territorial around their immediate nesting area, they usually tolerate the presence of conspecifics in their larger home range (J. Roth, W. Boarman, pers. comm.). In at least one American Crow population (near a Least Tern colony) in southern California, nesting crow territories overlap extensively with neighbors and were not defended against conspecifics (Caffrey 1992). However, in other areas of North America, crows may aggressively defend territories (Kilham 1985). In order to evaluate the effectiveness of using CTA as a management tool at a particular site, a thorough understanding of the territorial behavior of local predators is necessary.

Repellents

A repellent is a substance or object that is used to eliminate or reduce the presence of unwanted animals in particular areas. Visual repellents used to deter birds include: balloons (Shirota et al. 1983), kites (Fazlul Haque and Broom 1985), flagging and streamers (Bruggers et al. 1986), and effigies (discussed in separate section). Visual repellents act by startling the intended bird species, but are often expensive and only effective for short time periods (Mason and Clark 1995).

Auditory repellents act by startling birds with noises such as propane cannons, synthetic birdcalls, pyrotechnics, and other sonic and ultrasonic devices. Cannons can be effective in protecting crops if they are moved every few days (Mason and Clark 1995). Hazing (a type of auditory repellent) involves the use of loud noises and sometimes accompanying visual stimuli to frighten animals from an area. A common hazing technique involves shooting a blank shotgun cartridge (filled with rice). Hazing has been used successfully to deter some raptor species (M. Elliott, pers. comm.) but it is not well tested with corvids.

Taste repellents act by making food distasteful (and sometimes causing illness) to the consumer (e.g., copper oxalate, methiocarb). "Behavioral repellents" such as Avitrol (4-aminopyridine) produce erratic behavior and distressing cries after ingestion. These actions then scare the rest of the flock from the area. Unfortunately, Avitrol can cause minimal to substantial mortality depending on the strength of application and should be referred to as a toxicant rather than a repellent (Mason and Clark 1995).

Methiocarb has been extensively tested as a food repellent. Methiocarb combines a noxious taste with an illness producing response (Conover 1984). Stickley and Guarino (1972) found that sprouting corn treated with methiocarb was depredated significantly less than untreated corn (0.3% versus 44%). Methiocarb application to cherries and to highbush blueberries resulted in a significant decrease in consumption of these fruits on treated versus control plots for up to 14 days after application (Guarino et al. 1974, Stone et al. 1974). However, although birds will avoid methiocarb treated foods, they will readily consume the same untreated foods at other locations or at the same location after methiocarb use is terminated (Conover 1984). Avery et al. (1995) found methiocarb to be effective in conditioning ravens to avoid Least Tern eggs (see previous section).

In many cases, wildlife soon discover that repellents are not actually harmful, and the animals may soon become accustomed to the smell, taste or sound of these deterrents. However, it is possible that food repellents may be of use in managing corvids at landfills and other areas where anthropogenic wastes are exposed, although it would require repeated application of the repellent for the best results. Repellents have been suggested as a means of limiting corvid use of anthropogenic food and water sources. Experiments are planned to test chemical repellents (e.g., methylanthranilate) to deter ravens from eating garbage or drinking water at specific locations in the Mojave Desert (Boarman 2000).

Behavioral repellency has also been proposed to counteract raven predation of the San Clemente Island Loggerhead Shrike (*Lanius ludovicicanus mearnsi*). Using this approach, an inanimate object is presented to a captured raven while it is simultaneously exposed to a "negative experience" (i.e, a spray in the face with a chemical irritant). The same inanimate object is then placed near active shrike nests. After release, the properly conditioned raven will presumably avoid nests near the "conditioning" object (Garcelon 1999). As of yet, this method remains untested and there is the possibility that unconditioned ravens might actually be attracted to the inanimate objects at nest sites.

Sterilants

A sterilant is a substance that, when injected or ingested, causes temporary or long-term infertility. Sterilants have been used to control populations of feral horses (*Equus caballus*), canids, cervids, rodents, felids, and birds (Kirkpatrick and Turner 1985). Vandenbergh and Davis (1962) treated cracked corn with a gametocide (triethylenemelamine) and found that it reduced nesting success of Red-winged Blackbirds (*Agelaius phoeniceus*) in treated marshes relative to those in untreated marshes. Other sterilants have been tested on avian "pests" (Potvin et al. 1982, Cyr and LaCombe 1992), however, none have been used to control corvid populations.

The use of sterilants remains largely untested because it is usually expensive, animals tend to develop aversions to bait treated with drugs, only short-term control has been achieved, and because some biologists are reluctant to attempt novel methods (Kirkpatrick and Turner 1985). However, sterilants are potentially more advantageous than removal methods because treated animals remain in the ecosystem, continue to consume resources, and interact with other individuals. When animals are removed from a population, reduction in the density may cause a compensatory increase in reproduction quickly nullifying the removal.

Effigies

There is little empirical support for the effectiveness of effigies as methods to deter corvid activity. Typical effigies include scarecrows and raptor models (Marsh et al. 1992). Conover (1985) tested the effectiveness of a Great-horned Owl model in protecting vegetable crops from crows. He found that unanimated models were ineffective, while two animated models reduced damage by 81% compared to control plots. Both Caffrey (1993, 1994, 1995a, 1998) and T. Applegate (pers. comm.) found that placing crow carcasses or raven heads near California Least Tern colonies was effective in keeping other crows and ravens away at a number of sites in California. At sites in southern California, crows only avoided tern colonies when the effigies were placed on the ground rather than when hung on perimeter fencing (C. Caffrey, pers. comm.). However, carcasses placed at the top of fiberglass poles worked effectively at Vandenburg AFB (T. Applegate, pers. comm.). Crows appear to react most strongly if the carcasses used were members of their family group (Tom Applegate, pers. comm.). Continued success with effigies at these same sites has been inconsistent (K. Keane, pers. comm.).

Effigies are most effective when they are life-like, have motion, and are used in combination with startling sounds (Marsh et al. 1992). However, the effectiveness of even the most realistic models usually diminishes over time (Marsh et al. 1992). Although this method warrants further investigation, successful use with corvids may be difficult. For example, ravens have been observed feeding on corvid carcasses (Knittle 1992). In addition, captive ravens did not react at all to a stuffed (in live position) raven (B. Heinrich pers. comm. to G. Schmidt).

HABITAT MODIFICATION

Nest exclosures and tortoise enclosures

Nest exclosures have been used to prevent egg predation in a number of shorebird species including the Pectoral Sandpiper (*Calidris melanotos*) (Estelle et al. 1996), Piping Plover (*Charadrius melodus*) (Rimmer and Deblinger 1990, Deblinger et al. 1992, Vaske et al. 1994, Mabee and Estelle 2000), Killdeer (*Charadrius vociferous*) (Mabee and Estelle 2000), and the Snowy Plover (White and Hickey 1997, LeValley 1999, Mabee and Estelle 2000, Castelein et al. 2000a,b). Exclosure designs vary greatly in shape (cubic, triangular, or circular), volume (30,000->60,000 cm), and the presence or absence of a top. Common features shared by most exclosures include some type of mesh fencing (welded or woven wire) with a mesh size large enough for the protected species to pass through, and fence posts (metal or wood) for support. Recommended design features include: metal mesh fencing (5 x 5 or 5 x 10 cm) supported by at least 4 posts. Posts should not extend above the exclosure (to act as predator perches), and fencing should be buried at least 20 cm below ground (Deblinger et al. 1992). Ideally, the top of the exclosure should have openings that permit the adult bird to fly out unhindered, while making it difficult for avian predators to enter. This is usually accomplished by placing pieces of twine across the exclosure top, 6 to 8 inches apart (Castelein et al. 1997, 1998a, 1998b, 2000a, 2000b).

Exclosures are usually set up around nests after the full clutch is laid, however, predation during the laying stage may necessitate earlier set-up. Exclosure set-up has not been reported to cause significant nest abandonment for Piping Plovers in coastal northeastern North America (Vaske et al. 1994), or for Snowy Plovers in Humboldt County (LeValley 1999), and Point Reyes, California (Page 1991, White and Hickey 1997).

However, nest abandonment as a result of setting up exclosures has been reported in some cases. A marked pair of Snowy Plovers deserted more than one nest after an exclosure was placed around each nest at Limantour Estero, California (G. Page, pers. comm.). Exclosures with blueberry netting for tops caused desertion by Snowy Plovers in coastal Oregon (Castelein et al. 1997).

Most published studies have documented increased hatching success in nests with exclosures compared to nests without them. Melvin et al. (1992) documented a 90% (n=29) hatching rate in exclosed compared to 17% (n=24) at unprotected Piping Plover nests in Massachusetts. Similarly, Rimmer and Deblinger (1990) documented 92% (n=26) hatching success in exclosed compared to 25% (n=24) at unprotected Piping Plover nests in Massachusetts. Estelle et al. (1996) documented a 77% (n=13) hatching success in exclosed compared to 23% (n=210) at unprotected Pectoral Sandpiper nests in Alaska. However, the experimental design and statistical rigor in some of these studies has been questioned (Mabee and Estelle 2000). In at least one study, exclosures did not appear to offer protection to Piping Plovers, Killdeer, and Snowy Plovers from predators at a site in Colorado (Mabee and Estelle 2000). At this site, the main nest predators (small mammals) were able to enter the exclosures unimpeded. Small skunks (*Mephitis mephitis*) have been observed entering exclosures at sites in coastal California as well (G. Page, pers. comm.). Thus, identification of potential predators at specific sites is essential in assessing the feasibility of using exclosures as a management option.

Although the efficacy of nest exclosures is well documented, protection is only offered during the egg-laying and incubation stages. Chicks are subject to predation once they leave the exclosure. In some situations, predators may target exclosures and wait until an adult or chick emerges (J. Watkins, pers. comm.). There also is evidence that corvids can learn to get in and out of exclosures (M. Marriot, pers. comm.) and are doing so consistently at Scott's Creek Beach, Santa Cruz County, California (G. Page, pers. comm.). Although nest exclosures have been effective in protecting Snowy Plover nests in southern Oregon in the past, American Crows have been preying on exclosed nests more frequently in recent years (Castelein et al. 2000a, 2000b). In 1999, corvids depredated 7 exclosed nests in a 3-day period. Placing additional rows of twine (5 cm increments) or blueberry netting over the top apparently reduced corvid trespass into the exclosures (Castelein et al. 2000a). In 2000, nine exclosures were fitted with electrified wiring along the top perimeter to reduce corvid perching and subsequent predation on exclosed nests. The electrified exclosures worked well (8 of 9 nests hatched), however, set-up time averaged 2 hours and the additional cost of electrification is a disadvantage (Castelein et al. 2000b). Other predators (particularly small raptors) appear to target exclosures and may enter and kill the nesting adult (Abbott and Peterlein 2001).

Material costs for exclosures range from $15.00 (Melvin et al. 1992) to $100.00 (G. Page, pers. comm.) per unit. The total cost of erecting and monitoring *one* exclosure during the incubation period is estimated at $1,000 for Snowy Plovers in Humboldt County, California (R. LeValley, pers. comm.). The cost of erecting many exclosures can quickly become prohibitive, although they can usually be reused.

Setting up exclosures can be time-consuming and labor intensive and may not be feasible in remote areas (T. Applegate, pers. comm.) or at nest sites that are difficult to access (e.g. gravel bars) (LeValley 1999). Exclosures are bulky and may be difficult or impossible to place on nests that have surrounding obstructions (e.g., woody debris).

A predator-proof field exclosure was used to enhance hatching success and survivorship of juvenile desert tortoises at Fort Irwin, California (Morafka et al. 1997). The exclosure covered a 60 X 60 m area and was constructed using 2.6 m poles and covered with chicken wire. Eight to ten gravid adult female tortoises were collected from surrounding areas, allowed to nest within the exclosure undisturbed, then released. Sixty-eight percent of neonates and juveniles survived 5 years (Morafka et al. 1997), which is lower than that seen for some unprotected populations (Turner et al. 1987). Use of exclosures may only provide significantly increased survival in situations of abnormally high predation rates. The cost of setting up such an exclosure is high, and requires regular monitoring. There is some indication that exclosed tortoises may suffer from overcrowding and poor nutrition (Morafka et al. 1997).

Nest/perch site removal and nest destruction

Attempts have been made to prevent corvids from utilizing perch and nesting sites in the vicinity of protected species. In addition to removing potential nest sites, destruction of pre-existing nests has also been used as a management tool. The main rationale for eliminating nests/nest sites is to reduce foraging by corvids during the nesting season. In the case of the desert tortoise, this would be most beneficial since raven predation on tortoises peaks in the spring when both tortoise activity is high and ravens are feeding their young (Boarman and Heinrich 1999). The best time to eliminate nests is after egg-laying; removal prior to this may result in renesting attempts within the same territory (Stiehl 1978). However, ravens have been reported to renest in the same area even when nests containing eggs were destroyed (D. Garcelon, pers. comm.). Although nest destruction has often been used to control corvids (usually in tandem with shooting or poisoning), the success of this method in increasing the nesting success or survival of protected species is not well documented or has been inconclusive because of other confounding factors (Slagsvold 1978, Clark et al. 1995).

Ravens use perches to hunt and they may facilitate nesting. Consequently, removal of perch sites and (or) establishment of anti-perching devices are methods used to discourage such behavior. Anti-perching devices lowered, but did not prohibit continued perching by ravens in Idaho (Boarman 2000). Perch site removal is probably only effective at a very localized scale because ravens will most likely switch to an alternate perch site in the same vicinity (L.Young pers. comm. to B. Boarman 1993a). At some locations where the topography is flat and the only perches are man-made (e.g. Owens Lake, California), it may be a valuable management method (G. Page, pers. comm.).

Habitat restoration

Habitat restoration, as it applies to wildlife management, includes a variety of techniques used to increase the quality and quantity of habitat for a particular species. Management of predators (including corvids) can sometimes be accomplished through effective habitat restoration. Usually, the key strategy involves establishing natural or artificial cover (i.e. vegetative, physical, or human-made structures) to provide a refuge and concealment from predators. However, too much cover may reduce the ability of prey species to detect approaching predators. A thorough knowledge of the life history requirements of both the predator and prey is necessary for habitat restoration to be of greatest effectiveness in controlling predation.

Enhancement of cover can be accomplished by planting native species, reducing grazing (Littlefield and Paullin 1990, Littlefield 1995a), and using artificial "chick shelters" constructed of various materials. Greater vegetative cover led to increased nesting success in Sandhill Cranes (Littlefield and Paullin 1990). This was apparently due to the inability of ravens to find well-concealed nests (Littlefield 1995a).

Various artificial structures can be used as chick shelters. Such items include terra cotta tiles (Keane 1998), clay pipes, driftwood, artificial plants (Massey and Atwood 1984), and fencing (Jenkins-Jay 1982). Chick shelters have been used for the protection of California Least Tern hatchlings (Massey and Atwood 1984, Keane 1998). Typically, they are provided at sites where vegetation growth is lacking, but chick use of shelters has been observed at sites where sufficient vegetation appears to be present (Keane 1998). Success with these structures is not well documented, although Jenkins-Jay (1982) reported that chick shelters decreased avian predation in a Least Tern colony in Massachusetts. However, some predators (coyotes) appear to be attracted to shelters and may methodically check them for hidden chicks (T. Applegate, pers. comm.). In addition, some chick shelters have been known to house black widow spiders (*Latrodectus mactans*) that have killed at least two Least Tern chicks (Keane 1998). Shelters may have to be cleaned periodically to avoid losses to spiders and possibly other invertebrates.

Unlike the Sandhill Crane, Snowy Plovers and Least Terns select open nesting sites. In order to attract returning terns to historical nesting areas, vegetation is often removed from the site (Caffrey 1993, 1994, 1995a, 1998; Keane 1998, 1999). Theoretically, open nest sites allow adults to detect approaching predators. However, the presence of some cover near the nest site may provide important cover for tern chicks. Therefore, it may be best to allow some vegetation to remain near a nesting colony.

Although habitat restoration is often implemented for reasons other than corvid control, it can be an important component in reducing predation rates. However, large-scale habitat restoration can be very labor-intensive and the benefits may not be realized for years.

Modify anthropogenic sources of food and water

Availability of subsidized food and water may be the most important underlying cause for the increase in corvid populations throughout the west (Boarman and Heinrich 1999). Not surprisingly, controlling these sources may be the most effective means of limiting corvid population growth for the long-term (Boarman and Heinrich 1999). However, the task is daunting. Significant reductions in organic waste availability will take the concerted effort of many governmental and private agencies throughout the state. Moreover, as of yet, little research has been done and no published studies have documented the effect of reductions in subsidized food and water sources on local corvid populations.

Food sources may be limited by changing waste management practices at landfills, by reducing crop and livestock by-products in agricultural areas, dairies, and ranches, and by reducing organic waste access in residential areas. Water sources may be limited by restricting access to sewage containment ponds, cattle troughs, irrigated areas, and other anthropogenic sources of water.

Specific actions have been proposed to reduce access of anthropogenic food and water sources to corvids (particularly ravens) in southern California (U.S. BLM 1990, Boarman 1992, Boarman 2000) (Table 8). These actions should be accompanied by corvid monitoring at landfills and other areas to determine if they are effectively reducing corvid use of these resources (Boarman 1992).

Providing subsidized food

"Raven feeding stations" were used to deter ravens from depredating San Clemente Island Loggerhead Shrike nests (Garcelon 1999). Territorial ravens defended the food subsidy from non-breeding ravens and appeared to shift their foraging activities away from areas of nesting shrikes (pers. comm. D. Garcelon). This method seems contrary to the ultimate goal of reducing corvid numbers. However, in this situation, it is warranted since this population of ravens is restricted to an island. This technique can be combined with other methods (destroying raven nests) to prevent increasing the local raven population.

OTHER METHODS

Translocation

The Bureau of Land Management (U.S. BLM 1990) has suggested translocating ravens from areas of high desert tortoise activity to reduce predation on juvenile tortoises. Stiehl (1978) recommended live-trapped birds should be moved a minimum of 200 km from the initial point of capture before release. Live-trapping using cannon netting has been successful in capturing ravens in concentrated feeding areas (U.S. BLM 1990). As of yet, no studies have been conducted on the response of corvids to relocation. Although translocation may benefit protected species at the removal site, wildlife at or near the release sites may be negatively affected by the influx of corvids.

Holding

This method involves trapping and keeping the predatory species captive until a critical life history phase (e.g. nesting) of the prey species has passed. Although this technique is advantageous in that it avoids lethal removal, the costs of maintaining captured predators may be high. In addition, another non-territorial raven may quickly replace the removed territory holder.

RESEARCH

Research on reducing the impacts of corvids populations on threatened and endangered species can be lumped into 4 categories: 1. Ecology, behavior, and life histories, 2. Monitoring, 3. Demographic modeling, 4. Developing new control methods and improving existing ones. In the following sections, we summarize each area of research.

Ecology, behavior, and life histories.

Knowledge of the ecology and behavior of the predator is essential for an effective management program. Although general life-history information is available for many corvid species, this information is sparse and may not be applicable at a local level (see Appendix A). Studies of the life histories of corvid populations affecting listed species in California are underway in some areas. The U. S. Geological Survey is currently performing studies on the territoriality, dispersal, and daily movement of ravens in the Mojave Desert (Boarman 1997, W. Boarman, pers. comm.). Raven foraging behavior, home range size and visitation to Western Snowy Plover nesting areas is currently being studied on the Point Reyes peninsula (Roth et al. 1999, Roth 2001).

Monitoring

Most monitoring efforts have, naturally, focused on the threatened or endangered species of concern. However, the most successful monitoring programs have also incorporated identifying potential predators and documenting predation events at each site. This has enabled faster response in “crisis” times when a “predator problem” arises and can enable effective proactive predator management. However, monitoring corvid populations within the vicinity of affected listed species is not typically performed. Monitoring of *both* the prey and predator populations is needed to help us understand how each population is affecting the other and would enable a more dynamic and effective management scheme. For example, monitoring both the prey and predator would help determine if there is a relationship between predator control and prey survivorship.

The practicality and effectiveness of monitoring depends on the species of concern. For the threatened and endangered species addressed in this report, monitoring has been variable. Tern populations are relatively easy to monitor because they are colonial nesters, their activities are obvious, and they live in habitats accessible to humans. Much time and money has been spent on monitoring Least Tern populations in California with positive results. However, desert tortoise populations, are difficult to monitor because individuals are highly dispersed, spend much of their time in burrows, and often live in rugged, inaccessible areas.

Monitoring of corvid populations that are preying on listed species should be directed at estimating population size, growth rate, foraging habits, and habitat requirements for survival. Point counts have been used to monitor corvid populations, however, different methods must be used to effectively monitor different species (Marzluff et al. 2001). Estimates of recruitment and survival can be obtained by finding and monitoring nests, and following marked birds over a number of years. However, this approach takes considerable time and effort. Presently, there are marked populations of some corvid species. At least three American Crow populations have a large number of marked individuals, including one in California (C. Caffrey, pers. comm.).

Corvid foraging habits have been surmised by examining prey remains around nests (Berry 1985, Farrell 1991, Woodman and Juarez 1988, Camp et al. 1993). However, corvids obtain food through predation, kleptoparasitism, and scavenging and therefore caution must be used when inferring impacts on threatened and endangered species from prey remains (Boarman 2000). Radio telemetry can provide more direct information on the foraging behavior, foraging, and habitat use of individuals and, therefore, has many advantages over other methods.

Demographic models

Development of demographic models is beneficial in allowing assessment of a range of management alternatives. Currently, demographic models are used to project population growth for some of the listed species that are impacted by corvids (Ray et al. 1993). Development of demographic models for the predatory corvid species is also needed so we can identify the life history stages that contribute most to population growth. Likewise, population modeling of the listed species may assist in determining how important predators are and what levels of predation are "acceptable" (see Beissinger and Westphal 1998). Different models may give conflicting results, nevertheless, they are a valuable tool for evaluating management options.

Developing new control methods and improving existing ones

The emphasis of corvid management has been on lethal removal. However, the success of lethal actions has not been thoroughly studied. Non-lethal means of corvid management have usually been experimental and have been implemented for short time periods (Nicolaus 1987, Avery et al. 1995). Other plans for non-lethal corvid control have been stalled due to lack of funding (pers. comm. W. Boarman). More research needs to be done on improving the methodology of current non-lethal methods as well as developing new approaches. Improving existing techniques may be accomplished by conducting experimental research on prey populations that are not endangered but have similar life histories as listed species of concern.

TABLE 7. Corvid Management Techniques – Advantages and Disadvantages.

Control Method	**Advantages**	**Disadvantages**
Lethal Removal		
Shooting	• Immediate reduction in predator population • Possibly cost-effective • Able to target specific individuals	• Not acceptable for ethical reasons • Removing territorial birds may allow non-resident bird occupation of control area • May be labor intensive • May be difficult to remove more than one individual at a time • Short-term solution
Poisoning	• Immediate reduction in predator population • Possibly cost-effective	• Not acceptable for ethical reasons • Removing territorial birds may allow non-resident bird occupation of control area • May be labor intensive • Non-target animals may be inadvertently poisoned • Short-term solution
Trapping	• Immediate reduction in predator population • Able to target specific individuals	• Not acceptable for ethical reasons • Removing territorial birds may allow non-resident bird occupation of control area • May be labor intensive • May be difficult to entice birds into trap • Short term solution

TABLE 7. Corvid Management Techniques – Advantages and Disadvantages (continued).

Behavioral Modification	**Advantages**	**Disadvantages**
Conditioned Taste Aversion	• Non-lethal • Takes advantage of territorial defense by resident birds eliminating need to control nonresident ravens (*data supporting this is still minimal*) • Conditioning may last for over 1 year • Birds can be conditioned after one treatment in some cases • May work for small populations preying on concentrated food sources	• May be labor intensive • May be difficult to find reliable egg or young mimics • May not be effective on large populations or those preying on dispersed food sources
Repellents	• Non-lethal • May be cost effective	• Predators may not develop long-term aversion to repellents • May be labor intensive
Sterilants	• Non-lethal • Prevents a compensatory increase in reproduction	• May be expensive • May be hard to administer • Only short-term control has been achieved
Effigies	• Non-lethal • Cost-effective • Immediately effective	• Only effective on very local scale • Effectiveness may diminish with time • Largely untested

TABLE 7. Corvid Management Techniques – Advantages and Disadvantages (continued).

Habitat Modification	**Advantages**	**Disadvantages**
Nest Exclosures and Turtle Enclosures	• Non-lethal • Has been shown to be effective in previous studies • Immediately effective	• Labor intensive • May lead to nest abandonment (birds) • Only protects bird eggs (not fledglings) • Can only be set-up on nests with no obstructing debris. • Not effective against all predators • Identifies nests for vandals and for aerial predators of adults
Habitat restoration	• Non-lethal • Long-term solution • May benefit entire ecosystem	• Very labor intensive • May not see results for years
Perch / nest site removal	• Non-lethal • Cost-effective • Immediately effective	• Only effective on a local scale • Birds may renest nearby • Hard to eliminate **all** perch sites • Largely untested
Modify anthropogenic sources of food and water	• Non- lethal • Principal cause of increases in corvid populations • Long-term solution • May benefit entire ecosystem	• Very labor intensive • No empirical data that indicates effectiveness • May be impractical in some areas • Difficult to ensure strict compliance • Largely untested

TABLE 7. Corvid Management Techniques – Advantages and Disadvantages (continued).

Habitat Modification (continued)	**Advantages**	**Disadvantages**
Providing subsidized food	• Non-lethal	• May alter natural foraging behavior of predator • May increase corvid population • Untested
Other methods		
Translocation	• Non-lethal	• Labor intensive • Potentially detrimental to wildlife at or near release sites • Birds may return to trap site • Untested
Holding	• Non-lethal	• Labor-intensive • Unrealistic for large, widely dispersed predator and prey populations • Removed corvid may quickly be replaced
Public education	• Non-lethal • Can be focused at problem areas • Potential for long-term reduction in corvid numbers	• Public reaction not predictable • Informing all individuals is time intensive

TABLE 8. Proposed actions[1] to limit corvid (particularly Common Raven) access to anthropogenic food and water sources.

Anthropogenic food / water source	Management action
Farms, ranches, dairies	• Reduce availability of livestock carcasses and afterbirths • Use chemical repellents (e.g., methyl anthranilate) to prevent consumption of waste grain.
Residential and public areas	• Install self-closing trash bins and dumpsters • Eliminate plastic bag use for street-side pickup • Limit corvid access to municipal compost piles • Erect fencing to prevent road kills
Landfills	• Ensure effective cover of wastes with at least 15 cm of soil or using synthetic covers • Erecting coyote-proof fencing[2] • Install overhead wiring • Use chemical repellents (e.g., methyl anthranilate) to deter corvids from eating garbage
Sewage containment ponds, irrigation facilities, other water sources[3]	• Alter edge of ponds (with vertical walls) • Place monofilament line or screen over entire pond • Install drains at truck washing facilities • Reduce water availability at stock tanks, faucets, golf course ponds, irrigation lines

[1] Proposed actions cited from: Stiehl 1978, BLM 1990, Boarman 1992, Boarman 2000.
[2] Coyotes benefit corvids by exposing food in otherwise inaccessible food containers.
[3] In desert areas, emphasis should be placed on limiting water in the spring and summer when demands are high.

HISTORY OF CORVID MANAGEMENT IN REGARDS TO THREATENED AND ENDANGERED SPECIES

Corvids are important predators of a number of listed species in California. Active predator management to protect these species ranges from intensive control to virtually none at all. In this section, we document the impact of corvid predation on five listed species in California, the Marbled Murrelet, Greater Sandhill Crane, California Least Tern, Western Snowy Plover, and the desert tortoise. Corvid predation is reported to be a contributing factor hampering the recovery of these listed species. In addition, we provide a brief historical account of corvid management for each protected species and discuss the success of these actions.

Corvids are only one factor affecting the survival of these species, and in some cases, the importance of corvids as a significant factor is highly disputed. It is important to emphasize that many other factors have contributed to the decline of these threatened and endangered species. Corvid predation is only part of the problem.

MARBLED MURRELET

The Marbled Murrelet was listed as threatened in the United States in 1992 and endangered by the state of California in the same year. In North America, this species breeds predominantly in trees of coastal old-growth and mature forests from the Aleutian islands south to Monterey Bay, California (Nelson 1997). The decline of murrelets has been attributed to loss and fragmentation of breeding habitat (e.g., from logging activities and coastal development), gill-net fishing, and oil spills (Nelson 1997).

Corvids are suspected to have caused the majority of known murrelet nest failures (Nelson and Hamer 1995a, Miller et al. 1997). Common Ravens have been observed preying on murrelet eggs and nestlings, while Steller's Jays have been observed preying on murrelet nestlings and are strongly suspected of taking eggs at active murrelet nests (Table 6) (Singer et al. 1991, Nelson and Hamer 1995a). There is also an unpublished account of a Common Raven preying on a murrelet egg in Hidden Gulch, California (K. Nelson, pers. comm.). Although observations of corvid predation at real nests are rare (Table 6), a preponderance of both direct and strong circumstantial evidence suggests that corvids are major nest predators at simulated murrelet nests (Table 5). All three focal corvids and Gray Jays have been observed preying on artificial eggs, chicks, or live pigeon nestlings at simulated murrelet nests (Table 5) (Marzluff et al. 1996, Bradley and Marzluff in press, Luginbuhl et al. in press).

The importance of corvid nest predators at real murrelet nests is not documented because of the limited number of active murrelet nests that have actually been found. However, in two studies using simulated murrelet nests (Table 5) (Marzluff et al. 1996, Luginbuhl et al. in press), corvids were the most important predator during the incubation stage.

Fragmentation of old-growth forests is reported to be a major factor in the decline of Marbled Murrelets (Miller et al. 1997). Because many corvids, particularly Steller's Jays, thrive in edge habitats created by fragmentation (Masselink 1999, Brand and George 2001), corvid predation on murrelet nests may be high along edges. In support of this, Nelson and Hamer (1995b) found that successful murrelet nests were significantly further from forest edges than unsuccessful nests. Moreover, point counts of corvid abundance had the strongest correlation with predation on artificial murrelet nests (Luginbuhl et al. in press). However, Raphael et al. (in press) did not find a clear relationship between fragmentation and nesting success of simulated murrelet nests. They attributed this to the diverse predator assemblage recorded at artificial nests. Bradley and Marzluff (in press) found sciurid mammals to be the most important nestling predators at simulated murrelet nests. However, mammalian predation has not been observed or suspected at real murrelet nests (K. Nelson, pers. comm.).

Although documentation of corvid predation at real Marbled Murrelet nests is limited, all direct documentation (2 of 2) and almost all events based on circumstantial evidence (6 of 7) involved predation of Marbled Murrelet chicks or eggs by Common Ravens or Steller's Jays (Nelson and Hamer 1995a). In addition, evidence at simulated murrelet nests indicates that corvids are a major nest predator and can greatly affect nesting success. However, a broad suite of avian nest predators are known or suspected to depredate real murrelet nests, including Steller's Jays, Gray Jays, Great Horned Owls (*Bubo virginianus*), Common Ravens (Nelson and Hamer 1995a), Sharp-shinned Hawks (*Accipiter striatus*) (Marks and Naslund 1994), and Red-shouldered Hawks (K. Nelson, pers. comm.). At simulated murrelet nests, the predator assemblage is also varied, including Barred owls (*Strix varia*), Common Ravens, Keen's mice (*Peromyscus keeni*), Townsend's chipmunk (*Eutamias townsendii*), northern flying squirrels (*Glaucomys sabrinus*), and Douglas squirrels (*Tamiasciurus douglasii*) (Bradley and Marzluff in press).

As of yet, no corvid control measures have been implemented to specifically protect the Marbled Murrelet. However, the California Department of Transportation has installed predator-proof garbage cans at state parks as mitigation for removal of old-growth trees along highways. No estimate of the effectiveness of these measures has been attempted (R. LeValley, pers. comm.). The most recent Marbled Murrelet recovery plan (Miller et al. 1997) recommends the maintenance and development of large contiguous blocks of forest. Such measures would probably reduce corvid predation rate, but it may take hundreds of years for young trees to develop the structure that murrelets need for nesting. Therefore, this approach alone is not a viable short-term solution.

SANDHILL CRANE

The Central Valley population of Greater Sandhill Cranes was listed as threatened in the state of California in 1983. This population winters in California but breeds at various locations in northern California, Oregon, Washington, and southern British Columbia. The decline of this population is thought to be due to loss of nesting habitat (primarily through grazing) and increased nest predation rates (Littlefield 1995a). Active corvid management to protect this population has occurred most intensively at Malheur National Wildlife Refuge (MNWR) in southeastern Oregon.

Ravens are the only corvid known to consistently prey on crane eggs, although Black-billed Magpies (*Pica pica*) are suspected to be occasional nest predators (Littlefield and Thompson 1985). Documentation of raven predation in the Central Valley population of the Greater Sandhill Crane has been reported almost exclusively at the MNWR in Oregon, the most important nesting area for this population (Littlefield and Thompson

1985, Littlefield 1995a). Ravens have also been documented as nest predators at other sites in eastern Oregon (Littlefield 1999) and in northeastern California (Littlefield 1995b).

Ravens are the most important avian nest predator on Greater Sandhill Cranes at MNWR. From 1966 to 1989, coyotes destroyed 214 (20%), raccoons 100 (9%), and ravens 162 (15%) of 1096 total crane clutches at the refuge (Littlefield 1995a). The importance of ravens as nest predators at this site may be, in part, due to the close proximity of raven and crane breeding areas. Typically, over 30 pairs of ravens nest on the exposed basaltic rimrock areas surrounding the prime crane nesting areas in the lower wetlands (Herziger 1997).

Evidence of nest predation by ravens is based on egg remains. Ravens are unable to carry an entire crane egg between their mandibles. A raven must first puncture a hole in the egg in which it can insert its bill before flying away with the egg. This has been corroborated by direct observation (C. Littlefield, pers. comm.). Ravens do not challenge adult cranes, therefore they only have access to eggs when a nest is left unattended.

In northeastern California, of thirty monitored nests destroyed by predators in 1988, coyotes were responsible for 17, ravens 6, and raccoons 5 (Littlefield 1995b). At sites in privately owned areas of eastern Oregon, coyotes destroyed slightly more (12.7%) clutches than ravens (11.1%) in two seasons (Littlefield 1999). Ravens are clearly a major nest predator of the Central Valley population of Greater Sandhill Cranes. However, crane nesting success is most negatively affected by the combination of both raven and coyote nest predation.

Prior to the 1940s, the crane population at MNWR was low. In 1912, an estimated 25 pairs nested in the vicinity of Malheur Lake, and only 15-20 cranes were seen in the Blitzen Valley in 1932. Between 1938 and 1972, lethal control of nest predators was implemented at the refuge in an effort to reduce predation on waterfowl and Sandhill crane eggs and nestlings. Shooting , trapping, and using bait poisoned with compound 1080 or Thallium were used to reduce the abundance of potential predators. In two years of the most intense management (1946-47), at least 2,343 corvids were killed at the refuge and surrounding areas (Littlefield and Thompson 1985). In addition, hundreds of coyotes and raccoons were also killed during this period. Sandhill crane nesting success was 89% (n=11) in 1940 (G. Sooter unpub. data) and continued to be high for the succeeding three decades of intense predator management (Littlefield and Thompson 1985).

In 1972, use of chemical toxicants on federal lands was prohibited (Presidential Executive Order 11643) and predator removal was discontinued. Within one year of terminating raven removal, they were classified as “abundant” at the refuge. In 1975, they were considered a major factor limiting bird reproduction (Littlefield and Thompson 1985).

From 1976 to 1980, winter grazing of wetlands (i.e., crane nesting areas) was reduced in selected areas at MNWR. Littlefield and Paullin (1990) documented significantly higher nest predation at grazed compared to idle (no grazing) nesting sites. Lower predation rates in idle habitat were attributed to increased vegetative cover, which provided more concealment, reduced access by terrestrial predators (due to lack of cattle pathways), and an increased prey base for potential nest predators (e.g., small mammals). Ravens took significantly more clutches from poorly-concealed nests than well-concealed nests (Littlefield 1995a), suggesting that increased vegetative cover was an important factor in limiting raven predation. Although nesting success increased at idle sites compared to success under more intensive grazing, the average proportion of chicks fledged per nesting pair in the autumn population averaged only 6% during the years of reduced grazing.

In 1982, a number of experiments were conducted at MNWR to determine if conditioned taste aversion (CTA) could be established in nest predators (particularly ravens) (Nicolaus 1987). Consumption of surrogate crane eggs (i.e., turkey eggs) treated with Landrin (a CTA agent) decreased at treated sites compared to control sites, indicating successful establishment of CTA. In addition, survival of surrogate eggs was higher within raven breeding territories, suggesting avoidance of these areas by the larger non-breeding raven population. This is supported by counts indicating significantly more non-breeding ravens outside of raven territories (59.5 ravens per km^2) than inside (8.9 ravens per km^2) (Nicolaus 1987). Conditioned taste aversion was never adopted as a management option after this trial season even though the preliminary results were promising.

In the 1980s, lethal removal of the three main nest predators (coyote, raven, and raccoons) was resumed. In 1982-83, an experiment was conducted in three areas of the refuge to compare crane nesting success at a raven removal site, a coyote/raven removal site and a control area. Crane nesting success was higher only at the site where both predators were removed (80%, n=16) compared to the control site (65%, n=13) in one season. However, none of the plots had fledging success levels that would sustain a stable crane population (Littlefield and Cornely 1997).

Because of continued declines in the number of crane breeding pairs, an intense period of predator removal was initiated in 1986 and continued through 1993 (Littlefield 1995c). At the start of the intensive predator control, 168 breeding pairs of cranes nested at MNWR; by 1997 the number of breeding pairs was up to 251 (C. Littlefield, pers. comm.).

Overall effectiveness

The greater Sandhill Crane population at MNWR has increased from lows of less than 20 pairs in the earlier part of this century (Littlefield and Thompson 1985) to approximately 250 breeding pairs most recently. It appears that a combination of predator control, including control of corvids, and improvement of vegetative cover at crane nesting sites has been important in increasing the crane population at MNWR.

Removal of ravens alone did not contribute to an increase in the crane population. It appears that removal of *all* predators was the most effective method in increasing nesting success. However, lethal raven control at this site effectively reduced the number of ravens only for short durations and constant effort was needed to keep raven numbers low. Evidence suggests that only a few “problem ravens” prey on crane nests (Stiehl 1978, G. Ivey, pers. comm.).

The raven population at the MNWR was probably much smaller prior to the introduction of livestock into the region in the late 1860s (Littlefield and Thompson 1985). Livestock carrion and waste grain are important sources of food for ravens in this area and are likely partly responsible for their increase (Stiehl 1978). A reduction in the supply of anthropogenic food sources could help reduce the corvid population at this site.

Conditions at MNWR are conducive to further attempts at using CTA, since breeding ravens have territories that encompass all Sandhill Crane breeding areas. In addition, Nicolaus (1987) showed that CTA could be established on a guild of nest predators using the same baits. Opinions differ on how practical and effective it would be to use CTA as the major management tool on the entire refuge. However, the potential will remain unknown until this is actually attempted.

CALIFORNIA LEAST TERN

The California Least Tern was federally listed as endangered in 1970 and was placed on the California Endangered Species List the following year. This species breeds along the Pacific Coast from San Francisco Bay to the Mexican border (Small 1994). The decline of the California Least Tern has been attributed to loss of nesting habitat through coastal development, detrimental irrigation practices, human disturbance, and increased predation (Thompson et al. 1997). Statewide monitoring began in 1973 and still continues. Principal monitoring activities include estimating population size and reproductive success, documenting sources of nest failure (including predation), and identifying nest predators (Caffrey 1995b). Nest predation is typically the most important cause of tern nest failure.

Because of intensive monitoring of breeding California Least Terns for over 20 years, a detailed record of tern nest predators exists (Obst and Johnston 1992, Johnston and Obst 1992, Caffrey 1993, 1995a, 1997, 1998, Keane 1998, 1999). Over 25 different nest predators have been identified at tern colonies during this time period. Documented non-corvid avian predators include gulls (*Larus* sp.), American Kestrels (*Falco sparverius*), Northern Harriers (*Circus cyaneus*), Red-tailed Hawks, Great Blue Herons (*Ardea herodias*), Burrowing Owls (*Speotyto cunicularia*), Loggerhead Shrikes, and others. Important documented mammalian predators include red foxes (*Vulpes vulpes*), coyotes, striped skunks, rats (*Rattus* sp.), raccoons, domestic cats (*Felis catus*), domestic dogs (*Canis familiaris*) and other less important predators. Since the 1970s, corvids have increasingly become a more important nest predator at California Least Tern colonies (Fancher 1992). Both crows and ravens depredate least tern eggs and chicks (Table 6). Ravens have most often been documented as occasional predators (Table 6), although at Camp Pendleton they are considered the most important nest predator (Knittle 1992). American Crows were the most important nest predators at three colonies (Venice Beach, Terminal Island, Pursima Point)(Table 6) and were believed to have caused colony abandonment at Terminal Island in 1988 (Massey 1988) and 1994 (Caffrey 1994). In 1994, kestrels, crows, and ravens were responsible for depredating 94-97% of all potentially fledged young at Terminal Island. In 1992, two crows methodically destroyed the first 39 tern nests at Venice Beach; predation stopped after one of the problem crows (an adult) was killed in front of the other one (a yearling) (Caffrey 1992). In this case, corvid control is believed to have saved the colony from abandonment. In 1993, crows caused abandonment at the Huntington Beach tern colony (Caffrey 1993).

The overall effect of corvid nest predation on tern population growth is difficult to estimate because of intensive management (including predator control) to help increase tern nesting success. In addition, the diverse number of species that potentially prey on tern nests confounds the importance of corvids as predators. Corvids have been predators at less than half of all active tern colonies, although they have been a more important nest predator in recent years (Appendix B). Typically, corvid predation is intense only at a small number of sites and is usually perpetrated by a few "problem" individuals. However, loss of only a few tern colonies can have disastrous repercussions on recruitment and population growth.

Predator management around tern nesting colonies has involved erecting fences to exclude terrestrial predators, removing vegetation from colonies prior to arrival of the terns, and providing chick shelters. Intense corvid management began in 1988, with the initiation of a pilot study to remove corvids at a tern colony in Camp Pendleton using eggs injected with poison (DRC-1339). Least Tern fledging rate was higher than it had been in previous years (246 pairs fledged 365-409 birds) (Butchko and Small 1992). In four subsequent nesting seasons (1989-1991) at the same site, all predators were controlled using the same protocol with some improvements to the method. Poisoned eggs were placed on elevated platforms to reduce the removal of eggs by non-target species. In addition, eggs were tethered to the platforms so that they could not be cached and consumed at a later time. Treated eggs were placed at the tern colony as well as in a "buffer zone" surrounding the colony. In all 4 years (except 1989), nesting success of the terns increased, indicating successful predator control (Butchko and Small 1992). However, the rate of eggs lost to ravens during 1988-1991 (8 per year), when active predator management was in place, was virtually identical to that during 1983-1987 (7.6 per year), the period just prior to intense predator management at Camp Pendleton (Avery et al. 1993). Overall, from 1983 to 1992, ravens were responsible for depredating 2% of 5,700 tern eggs produced (Avery et al. 1993). Thus, the removal of ravens during this time was probably not necessary, and it is likely that ravens were not the most important predators at Camp Pendleton.

In 1992, territorial ravens were conditioned to avoid Least Tern eggs at Camp Pendleton, California (Avery et al. 1995). The conditioned ravens actively defended their territories (and the tern colony) from non-breeding ravens, and no tern eggs were depredated. Follow-up trials at Camp Pendleton and San Diego tern colonies in 1995-1996 demonstrated that it is feasible to incorporate methiocarb-treated eggs into a regular predator management program (Avery 1997).

Lethal removal has been the most commonly used method to control corvids at tern colonies. Generally, tern monitors notified Wildlife Services (WS) (formerly Animal Damage Control) when "problem" animals were recognized. WS personnel would then selectively remove them.

Overall effectiveness

California Least Terns have made a remarkable come back. The population has increased from a low of 664 breeding pairs in 1976 to over 4,000 in 1998 (Keane 1998). The most important management factors contributing to the tern recovery include habitat improvement at tern colonies, limiting human access to tern colonies, and active control of problem nest predators. It is difficult, if not impossible, to determine the effect of corvid control alone in aiding the terns' recovery.

Predator management was effective at many sites and is thought to be largely responsible for the increase in fledgling to pair ratio from the late 1970s to 1990 (Fancher 1992). However, predator management has historically been relegated to “crisis control”. Predators are removed only after the damage has been done. Non-lethal means of controlling predation at tern colonies shows promise (Avery et al. 1995, Caffrey 1993), but these methods have not been well tested or attempted on a large scale. The USFWS previously initiated the development of a Least Tern predator management plan (Keane 1998). However, this plan was never completed. A plan that explores alternative corvid control methods is needed.

DESERT TORTOISE

In 1989 and 1990, state and federal governments listed the western Mojave population of the desert tortoise as threatened. Within the United States, desert tortoises live in the Mojave, Colorado, and Sonoran deserts of southwestern California, southern Nevada, southwestern Utah, and western Arizona. A number of factors are thought to be contributing to the decline of the desert tortoise, including illegal collecting, vandalism, upper respiratory tract disease, predation by ravens, road kill, and trampling by livestock (Berry and Medica 1995). The recent dramatic increase in raven populations in the Mojave Desert raised concern among some biologists that ravens may be a significant factor contributing to the decline of tortoises (Berry 1985). Although ravens prey on juvenile tortoises (Berry 1985, Boarman 1993), the importance of ravens as contributors to the decline in the desert tortoise population is disputed (Biosystems Analysis, Inc. 1991).

Common Ravens are known to depredate desert tortoises throughout all major regions of California deserts (Boarman 1993). Most losses appear to be centered in the western Mojave. Direct evidence of raven predation on juvenile tortoises is based entirely on numerous anecdotal observations by field personnel. Most reports in the literature are based on strong circumstantial evidence, including large numbers of tortoise shells (sometimes >200) below active raven nests or perches (Table 6) (Berry 1985, Farrell 1991, Woodman and Juarez 1988) and tortoise remains in raven pellets (Camp et al. 1993). Raven consumption of juvenile tortoises can be reliably assessed based on distinctive features of discarded tortoise shells. Young tortoises up to seven years in age (carapace length <110mm) are susceptible to raven predation (Boarman 2000).

Ravens are accomplished scavengers and kleptoparasites. Thus, it is possible that tortoise shell remains attributed to raven predation were actually scavenged or stolen from other predators. However, the fact that most shells found below raven nests were pried open while the shell was still soft indicates that they were probably not scavenged (Boarman 2000).

Although raven predation on juvenile tortoises is conclusively documented, the link between raven predation and tortoise decline is weak. Surveys conducted at five permanent study plots in the western Mojave Desert during the 1980s documented a substantial decline in juvenile tortoises relative to surveys at the same sites 5-6 years earlier (U.S. BLM 1990). Corvid predation was cited as a possible reason for the decline (Berry 1985, U.S. BLM 1990). However, the correlation between raven numbers and tortoise decline does not necessarily represent causation because a number of confounding factors may be responsible for tortoise declines. Some other factors include: loss to other types of predators, disease, habitat loss and degredation, vandalism, and starvation (Morafka 2001). At the same sites, reduced recruitment of hatchling to 8-year classes into the larger and older size classes was attributed to predation (U.S. BLM 1990). However, even though larger tortoises are easier to locate during surveys, no adjustments were made for differential capture probabilities between the different size groups (Biosystems Analysis, Inc. 1991). Moreover, observer effort, skill of observer, and habitat were not standardized among years (Biosystems Analysis, Inc. 1991).

Demographic models provide conflicting predictions about the effect of juvenile survival on tortoise populations. A model developed by Ray et al. (1993) suggests that juvenile desert tortoise mortality can be high without adverse affect to healthy tortoise populations. However, a model developed for Blanding's turtle (*Emydoidea blandingii*) by Congdon et al. (1993) suggests that juveniles must have a 75% survivorship rate per year in order to maintain a stable population. Desert tortoises may respond similarly to Blanding's turtles because they have similar life-histories (Boarman 2000).

In summary, raven predation on juvenile tortoises is well documented. In addition, territorial pairs of ravens and non-breeding individuals may specialize on juvenile tortoises (Camp et al. 1993). Thus, raven predation may be a significant contributor to juvenile tortoise mortality in some locations. However, the importance of ravens on tortoise population growth is unknown due to a myriad of potential contributing factors and a lack of solid data.

In 1989, the first program was initiated to remove ravens from selected areas in California deserts to protect desert tortoise populations. Three sites with presumed high predation rates on juvenile tortoises were targeted. Eggs poisoned with DRC-1339 and firearms were used to kill ravens. Removal was stopped after 7 days due to a temporary restraining order issued by the Humane Society. Between 106 and 120 ravens were killed (U.S. BLM 1989). Although raven numbers appeared to have been reduced, within three months, they increased to their former abundance (Rado 1993). No information was obtained on the response of the local desert tortoise population to this raven reduction.

In response to the restraining order, a "Raven Management Plan for the California Desert Conservation Area (CDCA)" was drafted in 1990. The main goal of this plan was to "restore a balanced predator/prey relationship between the desert tortoise and the raven". Specific management actions in this plan called for gathering basic life history data on desert ravens and utilizing both non-lethal and lethal control methods. This plan was never implemented because it became too controversial and had little empirical support (W. Boarman, pers. comm.).

In 1993 and 1994, an experimental raven removal program was implemented to determine if shooting ravens could effectively reduce predation on tortoises. Both selective removal (of suspected tortoise-eaters) and full-scale removal from the Desert Tortoise Natural Area (DTNA) was attempted. A total of 49 ravens were shot. Identifying and targeting offending pairs was time consuming and it was often difficult to kill both members of a nesting pair. Removing all ravens from the DTNA was even less successful (Boarman 2000). No follow-up research was performed to determine if the tortoise population responded positively to the raven reduction.

Since then, no removal of ravens has occurred, and the focus of tortoise management has shifted from predator removal to other means of halting the tortoise decline. Some of the research is directly applicable to raven control. Boarman et al. (1995a) found tortoise mortality along a 24 km section of fenced highway to be considerably lower than along a 24 km section of unfenced highway (1 vs. 34 individuals). Tortoise fencing not only reduces direct mortality along highways but also may limit food availability (road kills) for ravens, thereby decreasing raven numbers. The impact of tortoise fencing on road kills and raven numbers, however, has not been assessed.

Research has been initiated to examine raven use of anthropogenic resources in the Mojave Desert. Boarman et al. (1995b) found significantly more ravens at landfills than at sewage ponds, golf courses, city streets, and a desert reference site in the Mojave, Boron, and Kern Counties, California. These results indicate that landfills may be the principal factor causing an increase in raven populations in the Mojave. In 1992, landfill operators were asked to cover solid waste with a minimum of 15 cm of soil every day. In addition, other measures were recommended to limit raven access to landfills. These measures included: erecting coyote-proof fencing around landfills, enclosing sorting areas, installing drains at truck washing facilities, and placing coverings over septage and leachate ponds. Compaction of garbage followed by cover with 15 cm of soil effectively reduced the number of ravens using the landfill (Boarman and Heinrich 1999). Temporarily covering garbage with large tarps may also be successful if used and maintained properly (W. Boarman, pers. comm.).

A raven management plan is currently being written to be included as part of a larger "Coordinated Regional Management Plan" (W. Boarman, pers. comm.). Actions called for in this plan include: lethally removing ravens from specific areas; studying the behavior and ecology of raven predation on desert tortoise; surveying raven nest and roost locations; examining the effectiveness of and need for raven removal; experimenting with chemical aversion agents to deter raven use of anthropogenic food and water sources; studying whether removal of raven perch and nest sites will limit tortoise predation; testing the effectiveness of relocating ravens; and developing a demographic model of raven populations (Boarman 2000).

Overall effectiveness

At this point, raven control has been short-term and sporadic. Although raven numbers declined temporarily in one case, nothing is known about the effects of these programs on the desert tortoise population. The key to effectively reducing raven numbers over the long-term is through limiting their access to anthropogenic food and water sources. However, the feasibility and effectiveness of implementing this on a large scale is still unknown.

SNOWY PLOVER

The Pacific Coast population of the Western Snowy Plover was listed by the Federal government as threatened in 1993. In the United States, this species breeds along the coast from southern Washington to the Mexican border. They also breed at inland sites in south central Oregon, eastern California, western and central Nevada, northwestern

Utah, and southern Arizona. The primary cause of decline is attributed to habitat degradation and expanding recreational beach use (Page et al. 1995). The use of beach grass (*Ammophila arenaria*) to stabilize dunes and beaches has also resulted in less available breeding habitat (Page et al. 1995). Most recently, increases in nest predators have been considered to be an important threat to plover nesting success.

Both Common Ravens and American Crows are known to prey on the eggs of Western Snowy Plovers (Table 6). In published studies, direct observations of corvid predation on plover eggs are not documented (Table 6), however, eyewitness accounts have been reported (G. Page, N. Read, pers. comm.). Most documentation of corvid predators at plover nests is based on strong circumstantial evidence (Table 6) that is hard to refute (e.g., tracks leading to a recently depredated nest, sometimes with egg remains left at the nest). In San Diego County, California, both Common Ravens and American Crows have been important nest predators of Snowy Plovers in multiple seasons accounting for over 70% of all predation events (Powell et al. 1996, 1997, Collier and Powell 2000) (Table 6). In situations where both plovers and Least Terns nested in the same area, corvid predation was higher on plover nests prior to the arrival of the terns (Powell et al. 1996, 1997). Once the terns arrive, the plover nests located within the tern colony are less likely to be detected by predators. Common ravens have consistently been the most important nest predator at the Point Reyes National Seashore, California, accounting for 69% (127 of 183) of all predation events in five seasons (Hickey et al. 1995). At Vandenberg Air Force Base, California, American Crows have consistently been the second-most important predator of plover nests (coyotes are the primary predator) since 1995 (Persons 1995a, 1995b, Persons and Applegate 1996, 1997, Applegate and Schultz 1999, 2000, 2001) (Table 6). There is also anecdotal evidence that American Crows may specialize on robbing plover nests at Vandenburg Air Force Base (Applegate and Schultz 1999). In Humboldt County, California, ravens were reported to be an important nest predator (n=2) (LeValley 1999), although currently little data are available to corroborate this. Ravens were considered to be the second-most important plover nest predator (gulls were most important) at Mono Lake, California (Page et al. 1985).

Corvids are also important nest predators at a number of Snowy Plover breeding areas in Oregon. At four study sites on the Oregon coast, nest loss was attributed to movement of sand (n = 11,17%) and predation by crows and ravens (n = 19, 30%). Cause of failure for the remaining nests could not be reliably determined, but corvids were believed to be responsible (Table 6) (Wilson-Jacobs and Meslow 1984). From the start of plover monitoring on the southern Oregon coast in 1990, both American Crows and Common Ravens have consistently been the most important nest predators (Table 6) (Stern et al. 1990, 1991, Craig et al. 1992, Casler et al. 1993, Hallett et al. 1995, Estelle et al. 1997, Castelein et al. 1997, 1998, 2000a, 2000b).

Comparison of exclosed versus non-exclosed nests indicates that hatching rates are severely limited by egg predation (White and Hickey 1997, Transou and LeValley 2000, Stern et al.1991, Craig et al. 1992, Casler et al. 1993, Hallett et al. 1994, 1995, Estelle et al. 1997, Castelein et al. 1997, 1998, 2000a, 2000b). However, the overall impact of corvids on the population size of plovers is not known. Corvid predation is an important contributor to nest loss, but other predators, habitat loss and human disturbance are also important factors affecting plover populations (Page et al. 1995).

Erecting predator exclosures around plover nests has been the most frequent approach for predator management. In 1996, the Point Reyes National Seashore (PRNS) initiated a Snowy Plover Recovery Project. One of the key components of this project was placing exclosures around Snowy Plover nests. After one year of exclosure use, White and Hickey (1997) documented a 2-fold increase in plover breeding pairs (although it is not known if exclosure use was the reason for the increase). Exclosures appeared to provide benefits for plovers at this site (White and Ruhlen 1998, Ruhlen and White 1999) until the 2001 breeding season. Exclosure use was limited in 2001 at PRNS because an unidentified raptor entered at least one exclosure and killed an adult (Abbott and Peterlein 2001). In addition, ravens at Scotts Creek learned to get into exclosures (G. Page, pers. comm.). Workers at Point Reyes Bird Observatory are presently experimenting with new exclosure tops to prevent ravens from entering exclosures (G. Page, pers. comm.).

In Humboldt County, Mad River Biologists, with support from the U. S. Fish and Wildlife Service, began using nest exclosures to protect plover nests from predators. Protected nests on beaches had a hatching success almost twice that of nests without exclosures in 1999-2000 (Transou and LeValley 2000). Nests on gravel bars were *not* exclosed, yet they had high hatching success (78%, n=18) in 2000 (Transou and LeValley 2000). Nests on gravel bars are believed to suffer lower predation because adult plovers do not leave tracks and eggs are difficult to detect on the cobble substrate (R. LeValley, pers. comm.).

In Oregon, a Snowy Plover monitoring program was initiated in 1990 that included exclosure use as part of the management strategy. Nests with exclosures have consistently had higher hatching rates than unexclosed nests (Stern et al.1991, Craig et al. 1992, Casler et al. 1993, Hallett et al. 1994, 1995, Estelle et al. 1997, Castelein et al. 1997, 1998, 2000a, 2000b). Despite this success, American Crows have been depredating a greater number of exclosed nests in recent years. Alterations in exclosure design (see section on nest exclosures) have been largely successful in thwarting continued corvid intrusion into exclosed nests, although the effort in adapting these design changes has been substantial.

Creation of nesting habitat for California Least Terns using dredge spoils at Batiquitos Lagoon in San Diego, County, inadvertently provided Snowy Plovers with prime nesting sites (Powell and Collier 2000). The fledging rate in 1995 was higher (>1 fledglings/nest) at the newly created area than at the older-dredged material and natural beach areas (0.3-0.7 fledglings/nest). Nests on the newly created islands were surrounded by less vegetative cover, less debris, and shorter vegetation than nests at the other habitat types. Predation rates were probably low because of low nest density, good visibility for adult snowy plovers, and lag time in predator “discovery” of the new food source (Powell and Collier 2000). Habitat management approaches such as this may be an effective way to limit corvid predation of plover nests in some areas.

Overall effectiveness

The use of exclosures can be an effective method in limiting plover nest predation and is apparently helping to increase hatching rates in some areas. However, recent reports indicate that exclosures may attract vandals and, in some instances, may make the presence of nests known to avian predators of adult plovers. In some areas, corvids are learning how to get into exclosures. In addition, the success of hatchling survival after leaving exclosures is not known, although this is currently under study in Humboldt County (M. Colwell, pers. comm.). Presently, exclosures are a useful management tool in emergency situations but are not a viable long-term strategy because of the expense of finding nests and erecting exclosures around them (G. Page, pers. comm.). In some areas, corvid numbers are heavily subsidized by food sources at nearby ranches and agricultural areas (Roth et al. 1999). In these areas, corvid management may be best accomplished by limiting access to these anthropogenic food sources (G. Page, pers. comm.).

RECOMMENDATIONS

It is clear that reducing the impacts of corvids on threatened and endangered species is a complex issue with no simple solution. As the history of corvid management indicates, management strategies to protect particular species must be approached on a case-by-case basis. Nevertheless, some management recommendations such as limiting availability of anthropogenic food sources in locations where corvids co-occur with threatened and endangered species can, in some cases, be implemented quickly and with relatively little cost. More drastic measures, such as lethal removal, should be considered in cases where an immediate decrease in corvids is necessary to save a population of threatened or endangered species.

Management approaches at the local level will depend on a number of factors including the behavior and ecology of the protected species of concern, the importance of corvids as predators (in relation to other predators at a site), and the foraging and nesting behavior of local corvid populations. Local management will, in many cases, be handled quickly and in the short-term. However, short-term management approaches often do not eliminate the ultimate sources of the problem.

For the long-term, broad scale management measures must be implemented that will address the *source* of the "corvid problem." Implementing broad-scale measures will take considerable planning and cooperation between different agencies, and the benefits may not be realized for years. Broad-scale measures must focus on two main issues: educating the public and reducing sources of anthropogenic food and water.

In order to effectively carry out both local and broad-scale management actions to reduce corvid predation of threatened and endangered species, three main groups must be involved: the pubic, land managers, and the scientific (research) community. Increasing public awareness and participation in developing solutions to this problem will greatly enhance the success of management actions. It is also essential to include private landowners in the development of management actions and to provide incentives for landowners whose livelihood may be compromised by such actions. Research and monitoring conducted by land managers and the research community is vital to the development and implementation of specific management actions. Only continued monitoring and analysis will make management actions effective. It is important to stress that these actions are not mutually exclusive and that effective management will require close cooperation between all interested parties.

We recommend an approach to corvid management that addresses issues of scale (local/broad) and calls on land managers and the research community to develop an efficient, yet ethical, means of reducing corvid predation. At the same time, we stress public education and participation as an important component of reducing the impacts of corvids on listed species.

PUBLIC INVOLVEMENT

1. Educate the public about corvid impacts on wildlife through popular articles, pamphlets, and signs posted at public recreation areas. Literature should include:
   - The fact that many corvid populations are increasing

- The cause of this increase is due to human supply of food, water, and the creation of suitable nesting habitat.
- High corvid numbers can cause problems for their prey which may include threatened and endangered species.
-

Justification: Large sectors of the public are probably unaware of the negative affects of corvids on listed species. Public education should stress that the negative impact of corvids is due, in large part, to the activities of humans and that corvids are a natural part of the ecosystem.

Educating the public will potentially have an important impact on corvid management. This action should be implemented throughout the state at all public human-use areas, particularly ones that have listed species and corvids present.

2. Encourage public participation in programs to protect listed species from corvid predation. This may include volunteer fieldwork, educational programs, participation on planning teams, and reviewing project plans. Statewide monitoring programs (such as that for the California Least Tern) could be initiated to monitor corvid populations that are affecting listed species.

Justification: Direct participation of the public can greatly enhance the efforts of land managers as well as increase public awareness of this issue. Monitor programs have been successful in the past and have been a major contributor to the recovery of the California Least Tern (Caffrey 1995a). The development and implementation of a corvid monitoring program would, ideally, be statewide, but would rely on local level organization at each site.

MANAGEMENT RECOMMENDATIONS

Non-lethal management actions

1. Reduce the influence of corvids in areas where they are negatively affecting (or have the potential to affect) listed species by reducing availability of anthropogenic food and water sources. This may include the following actions:

- Limit corvid access to landfills, sewage treatment plants, dairy farms, ranches, and road kills
- Deploy self-closing garbage cans in public-use and residential areas

Justification: The principal reason for the increase in corvid populations over the last few decades is access to anthropogenic food and water sources. Therefore, reduction in the availability of this subsidy will help reduce corvid numbers. This management action should be performed at different spatial scales depending on the type of corvid species. For example, ravens, have been observed traveling 65.2 km to forage in Idaho (Engel and Young 1992), while Steller's Jays in Redwood National Park, California had maximum travel distances of roughly 1 km (based on 69 ha home range size, n = 2) (Wallen et al. 1999). In New Jersey, American Crows traveled up to 18 km daily to forage (Stouffer and Caccamise 1991).

Reduction of food sources adjacent to areas of listed species activity may be one of the most important and cost effective means of immediately curtailing corvid activity at specific sites (C. Caffrey, pers. comm.). Spillage from conventional garbage cans was often responsible for attracting gulls and corvids to areas near Least Tern colonies (C. Caffrey, pers. comm.). The installation of self-closing (corvid proof) garbage cans near listed species breeding areas cannot be overstressed.

2. Reduce the influence of corvids in areas where they are negatively affecting listed species by reducing availability of perch, roost, and nest sites. This includes removing perch and roost sites and destroying nests (during the incubation stage) or addling eggs.

Justification: Availability of suitable habitat features that enhance foraging and breeding may allow corvids to breed in otherwise unsuitable locations. Because corvids may do most of their foraging within a relatively limited area surrounding their nest (Sherman 1993), a reduction in the availability of these habitat features may prevent corvids from using an area.

3. Monitor local corvid populations that are known to be negatively impacting listed species at specific sites. Monitoring should be implemented to: provide reliable estimates of both local and regional population sizes, to track the response of corvid populations to management actions (monitoring should occur before *and* after management is implemented), and to bolster local life history information (specifically information relative to foraging behavior).

Justification: Ideally, a marked corvid population would provide the most reliable data and is necessary to obtain information for demographic models. We strongly encourage agencies to provide funding to study marked corvid populations. However, monitoring an unmarked corvid population can still provide meaningful information and may be the only option for land managers in many locations.

The specific corvid monitoring protocol will vary from site to site. However, we recommend constant-effort surveys (e.g. point counts) of corvids at sites used by the species of concern and at anthropogenic food sources. Over time, such monitoring would provide information on the effectiveness of management actions implemented to reduce corvid use of such sites. We also recommend locating and monitoring corvid nests within the vicinity of breeding listed species. Monitoring corvid nests will provide an estimate of their local productivity. In addition, identification of prey delivered to the nestlings may provide strong evidence (it must be confirmed that corvids are not obtaining prey by scavenging or kleptoparasitism) that corvids are preying on the listed species. Moreover, monitoring nests will enable identification of the specific corvids that are responsible for depredations on listed species. Refer to the life-history section of this paper for information on corvid nesting substrates.

4. Identify all nest predators preying on the species of concern.

Justification: The importance of corvids as nest predators varies from site-to-site; therefore management actions will be dependent on the predator assemblage at a given location. For example, the effectiveness of nest exclosures at Snowy Plover nests depends on the species that are preying on the nests (Mabee and Estelle 2000). The control of more than one predator population may be necessary to achieve the desired effect.

5. Develop recreational public-use areas that are designed to limit potential corvid impacts on species of concern.

Justification: Campgrounds, parks, and other outdoor public-use sites are often placed in remote areas or in the remnant fragments of pristine habitat (e.g. old-growth forest) that are often inhabited by threatened and endangered species. Because corvids are often attracted to areas of human use, maintaining campgrounds in these locations may increase the potential impacts of corvids on the species of concern. To limit the impact of corvids, campgrounds and other recreational sites should be placed in locations away from areas used by species of concern (see Raphael et al. in press). This management action will take cooperation between research biologists, land managers, and park planners. This action should be considered throughout the state at all public areas that have listed species and corvids present. In addition, seasonal closures of certain areas (e.g. beaches for plovers) may be necessary.

6. Implement non-lethal management methods including conditioned taste aversion (CTA), corvid hazing at colonies, effigies, trapping and relocation.

Justification: Aversive conditioning has reduced corvid predation at Sandhill Crane nesting areas in Malheur National Wildlife Refuge, Oregon (Nicolaus 1987) and at a Least Tern colony at Camp Pendleton (Avery et al. 1995). However, it has never been used consistently, or attempted at other sites to protect listed species. We encourage managers to use aversive conditioning techniques (particularly CTA) and work with scientists to further refine the application of these techniques in the field.

The use of crow carcasses (as effigies) worked well in reducing corvid presence at Least Tern colonies in southern California (C. Caffrey, pers. comm.). There is enough anecdotal evidence to warrant testing the use of such effigies on a larger scale and using strict experimental procedures to confirm or rule out their effectiveness and feasibility as a management tool. Initial testing should evaluate the use of stuffed corvid mounts (rather than carcasses). In addition, experiments testing the most ideal placement of effigies should be performed.

<u>Lethal management actions</u>

1. Use selective lethal removal in situations where the survivorship of an unacceptable number of listed animals is threatened. Ideally, this would be implemented in conjunction with other non-lethal actions. The effectiveness of such actions should be closely monitored.

Justification: In situations where corvids are threatening the persistence of listed species at a location, removal of the problem birds may be necessary.

RESEARCH RECOMMENDATIONS

1. Study life-histories of local corvid populations that are believed to be negatively impacting listed species at specific sites. Data collection should focus on behaviors and life-history characteristics of corvids that are directly influencing the predator/prey relationship, including:

- Nesting ecology – nest locations, nest-site fidelity, and nesting density
- Roosting and perching sites
- Territoriality – Do breeders exclude other breeders and non-breeders?, Are vacant territories quickly filled?, Do birds use the same breeding territory year-after-year?
- Foraging behavior – preferred food, foraging locations, use of anthropogenic food sources, methods corvids use to locate and kill the species of concern.
- Habitat use – identify habitat used for nesting, foraging, and roosting.

Justification: Local-scale knowledge of corvid life histories remains undocumented at many sites where corvids are negatively impacting listed species. A better knowledge of site-specific life histories is necessary to create effective management actions at a local level. The goal is to obtain detailed, site-specific behavioral and ecological information. However, important trends across a larger area may be necessary to understand population dynamics of corvids at the local level. Some of the goals of this research action can be incorporated into the corvid monitoring plan mentioned previously.

2. Develop new management techniques of controlling corvids and improve methods already in use employing valid experimental design and statistical analyses.

Justification: Refinement of existing corvid management techniques may improve their effectiveness. For example, the benefits of CTA may be greatly increased if CTA can be established for both eggs and nestlings. Further development of sterilants and repellents may also lead to better non-lethal corvid control methods.

3. Develop demographic models for corvid populations (in addition to the listed species).

Justification: Corvid demographic models would enable the development of more efficient management strategies by 1. Enabling simulated tests of proposed management options, 2. Identify which life stages have the greatest impact on corvid population growth. Depending on the data available, the models may be applicable to both local and regional corvid populations. Marked corvid populations are necessary for the development of demographic models. We strongly recommend that data be collected from existing marked corvid populations to be used to establish demographic databases. We also recommend studying corvid populations that are known to affect listed species.

ADMINISTRATIVE RECOMMENDATIONS

1. Establish an interagency workgroup to oversee management direction

Justification: In order for effective corvid management at a state-wide level, the actions of all agencies involved must be efficiently orchestrated. Participating agencies (Appendix C) should establish a committee or workgroup that oversees the management initiatives. We recommend establishing a state-wide corvid management plan that would have *one* main delegating workgroup. Within this larger body, we recommend establishing two committees. One would act as an interagency task force coordinating the implementation of the program. The other would oversee the technical and political aspects of the plan. A Regional Raven Management Plan for the Common Raven has been proposed that would cover almost one-third of California (Boarman 2000). We recommend incorporating this plan into the recommended state-wide plan.

The main responsibilities of the workgroup and sub-committees would be to coordinate between agencies to procure funding for implementing specific management actions, coordinate project management by developing plans with appropriate experimental design, implementing, and monitoring actions. Finally, the workgroup would be responsible for preparing and publishing results.

Using data from cooperating agencies, we recommend the establishment of a GIS database that would contain information pertinent to the management of corvids. For example, the location of all major sources of anthropogenic food and water (landfills, sewage treatment plants, etc.) should be included. In addition, the breeding areas for all listed species that are known to be preyed on by corvids should be included in the database.

It is essential to involve state or national representatives of at least one animal welfare group (e.g. Defenders of Wildlife or the Humane Society) as well as representatives from at least one national conservation organization (e.g. Audubon or Natural Resources Defense Council). These groups should be included in all planning stages of management.

2. Develop an adaptive management approach

Justification: Successful corvid management will depend on continually improving existing methods based on monitoring projects and new information. This can only be accomplished if an adaptive management approach is adopted.

CONCLUSIONS

Effective corvid management will depend on the cooperation of many agencies, with in some cases, conflicting philosophies on how to approach the problem. It is important that the decisions be based on sound science; however, scientists must understand the position of land managers (Boarman 1992). Some management decisions will have to be made quickly without all the information and may preclude the scientific rigor deemed necessary to develop the best management options. We urge more funding spent on preventative, non-lethal corvid control.

These management recommendations are nothing new. Similar actions have been proposed in other plans (U.S. BLM 1989, Boarman 2000). However, previous actions have not been implemented to their full potential, mostly because of a lack of funding or scientific justification (as in the case of raven impacts on desert tortoise). We hope that the information contained in this plan will stimulate agencies to start taking a proactive approach to corvid management. Based on the trends we have reported, corvid numbers will continue to rise, increasing their impact on listed species. Therefore, the time for concerted action is here.

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Appendix A. Life history information for corvids specific to California.

Species	Attribute	N	Estimate[1] / Description	Location	Reference
Common Raven	Nest site	36 ?	Tree, cliff, man-made Tamarisk, Joshua trees, rock outcrops, transmission lines Monterey cypress and pine, eucalypts	Camp Pendleton (San Diego Co.) California Desert Conservation Area (southern California) West Marin Co.	Avery et al. (1993) U.S. BLM (1990) J. Roth, pers. comm.
	Clutch initiation	36	Late march (peak)	Camp Pendleton (San Diego Co.)	Avery et al. (1993)
	Clutch size	36	5.4	Camp Pendleton (San Diego Co.)	Avery et al. (1993)
	No. nestlings	36	3.65	Camp Pendleton (San Diego Co.)	Avery et al. (1993)
	No. fledglings	36	2.8	Camp Pendleton (San Diego Co.)	Avery et al. (1993)
	Diet	?	Pistachios	Central Valley	Salmon et al. (1986)
	Diet	10-30	Trout (hatchery)	Fish Springs hatchery	U.S. BLM (1990)
	Diet		Desert tortoise juv.	Mojave desert	Berry (1985), U.S. BLM (1989)
	Diet	13	Small birds, rodents, and reptiles, calf carcasses and afterbirths, grain	Point Reyes Peninsula (Marin Co.)	Roth et al. (1999)
	Diet	351	Insects, grains, seeds, mammals, reptiles, bird egg, human refuse	Eastern Mojave Desert (including parts of Nevada)	Camp et al. (1993)
	Diet	13	Grain, insects, reptiles (no Desert Tortoise)	Eastern Mojave Desert	Sherman (1993)
	Foraging sites	43	Landfill	Edwards Air Force Base (Kern Co.)	W. Boarman, pers. comm.
	Foraging sites	13	Grazed grass, dunes, cattle feeding areas	Point Reyes Peninsula (Marin Co.)	Roth et al. (1999)

Common Raven	Foraging sites	?	Dumps, landfills, sewage ponds, roadside pull-offs, rest areas, campgrounds	California deserts	U.S. BLM 1990
	Foraging sites	13	All sites within 1.7 km of linear-right-of-way	Eastern Mojave Desert, CA	Sherman (1993)
	Foraging time	43	Mid-day	Edwards Air Force Base (Kern Co.)	W. Boarman, pers. comm..
	Foraging time	13	Most active in morning and in mid-afternoon	Eastern Mojave Desert, CA	Sherman (1993)
	Foraging distance	13	75% of foraging within 400m of nest	Eastern Mojave Desert, CA	Sherman (1993)
	Terr. size/density	14	1 nest / 4.7 km^2	Camp Pendleton (San Diego Co.)	Linz et al. (1990)
American Crow	Nest site	40	Gymnosperms, eucalypts, sycamores	Encino (Los Angeles Co.)	Caffrey (2000)
	Clutch initiation	225	March 31 ± SE 1.96	Encino (Los Angeles Co.)	Caffrey (2000)
	Nestling period	97	41.0 (SE = 0.9) days	Encino (Los Angeles Co.)	Caffrey (1999)
	Nest success	147	43%	Encino (Los Angeles Co.)	Caffrey (2000)
	1st year survivorship	97	68%	Encino (Los Angeles Co.)	Caffrey (1992)
	Adult Survivorship	62	93% male, 97% female	Encino (Los Angeles Co.)	Caffrey (1992)
	Family group size	35	3-4	Encino (Los Angeles Co.)	Caffrey (2000)
	Breeding density	62	0.8 pairs/ha	Encino (Los Angeles Co.)	Caffrey (1992)
	Predators	3 22	Great Horned Owl, Cooper's Hawk, Red-Shouldered Hawk	Encino (Los Angeles Co.)	Caffrey (1999) Caffrey (2000)
	Roost site	87	Ash, sycamore, mulberry, elm, evergreen and deciduous oaks, alders,	Woodland (Yolo Co.)	Gorenzel and Salmon (1995)

Steller's Jay	Nest site	? 23	High in Eucalyptus trees 11.2 m ± 2.6 SD high in Coast live oak (8), California Bay (14), Monterey Pine (5), Elderberry (1), Madrone (1)	Berkeley (Alameda Co.) 20 km SE of San Francisco Bay (Alameda & Contra Costa Co.)	Brown (1964) Salata (1982)
	Nest construction	?	Late April – late May	California	Greene et al. (1998)
	Juvenile dispersal	40	mid-September – early Oct.	Tilden Regional Park (Alameda Co.)	Brown (1963)
	Diet	93	8% beetles, 11% wasps and bees, 9% other animal, 72% plant material (mostly acorns)	California	F. Beal, cited in Bent (1946)
	Foraging location	560	71% of time in trees foraging on ground increased in summer and winter	20 km SE of San Francisco Bay (Alameda & Contra Costa Co.)	Salata (1982)
	Territory size	?	"area of dominance" = 120m wide	Berkeley (Alameda Co.)	Brown (1964)

[1] Estimates are means unless noted otherwise.

Appendix B. California Least Tern colonies with documented corvid predation[1]

Year	Documented[2]	Suspected[3]	Site
1990		R	Oakland Airport (Alameda Co.)
		R	Ormond Beach (Ventura Co.)
		R	Point Mugu (Ventura Co.)
(no AMCRs reported)	R		Sta. Margarita R. (San Diego Co.)
		R	San Elijo Lagoon (San Diego Co.)
	R		Tijuana River (San Diego Co.)
1991	R		Bolsa Chica (Orange Co.)
	C		Venice Beach (Los Angeles Co.)
	R		Sta. Margarita – north beach (San Diego Co.)
1992	C		Venice Beach (Los Angeles Co.)
		C	Newport Slough (Orange Co.)
	R		San Elijo Lagoon (San Diego Co.)
	R		Chula Vista Wldlf Res. (San Diego Co.)
		R	Saltworks (San Diego Co.)
1993		R	NAS Alameda (Alameda / S.F. Co.)
		C	VAFB Purisima Point (Santa Barbara Co.)
	C		Huntington Beach (Orange Co.)
	R		San Elijo Lagoon (San Diego Co.)
		R	Mission Bay – N. Fiesta Is. (San Diego Co.)
		R	Naval Training Center (San Diego Co.)
		R	North Island NAS (San Diego Co.)
	R		Tijuana River (San Diego Co.)
1994		R	NAS Alameda (Alameda / S.F. Co.)
		C	VAFB Purisima Point (Santa Barbara Co.)
		C	Terminal Island (Los Angeles Co.)
		C	Huntington Beach (Orange Co.)
	R		San Elijo Lagoon (San Diego Co.)
1995		C	PGE, Pittsburg (Contra Costa Co.)
	R		NAS Alameda (Alameda / S.F. Co.)
		R	Santa Clara River (mouth) (Ventura Co.)
	C		Seal Beach (Orange Co.)
	C	R	NAS North Island (San Diego Co.)
		R	D Street Fill (San Diego Co.)
		R	Saltworks (San Diego Co.)
1996	R		NAS Alameda (Alameda / S.F. Co.)
	C		Terminal Island (Los Angeles Co.)
		C	Huntington Beach (Orange Co.)
	C	R, C	Batiquitos Lagoon (San Diego Co.)
	R		Tijuana River: north (San Diego Co.)

Appendix B. California Least Tern colonies with documented corvid predation (continued).

Year	Documented[2]	Suspected[3]	Site
1997	C		Bolsa Chica (Orange Co.)
	C		Guadalupe Dunes (Santa Barbara Co.)
		C	PGE, Pittsburgh (Contra Costa Co.)
		C, R	Seal Beach (Orange Co.)
	C		Terminal Island (Los Angeles Co.)
		C	LA Harbor Terminal Island (Los Angeles Co.)
	C		Vandenburg Beach 2 (Santa Barbara Co.)
	C	C	Vandenburg Purisima Point (Santa Barbara Co.)
		C	Venice Beach (Los Angeles Co.)
		R	D Street Fill (San Diego Co.)
	R	R	Lindbergh Field (San Diego Co.)
		R	San Elijo Lagoon (San Diego Co.)
		R	Tijuana River (San Diego Co.)
		R	Mission Bay – FAA Is. (San Diego Co.)
1998	C		Venice Beach (Los Angeles Co.)
		C	PGE, Pittsburgh (Contra Costa Co.)
		R	NAS Alameda (Alameda / S.F. Co.)
	R		LA Harbor TC2 (Los Angeles Co.)
		R	Batiquitos Lagoon (San Diego Co.)
	R		D Street Fill (San Diego Co.)

[1] From Obst & Johnston (1992), Johnston & Obst (1992), Caffrey (1992, 1993, 1994, 1995, 1996), Keane (1998, 1999).

2 - Corvid (R = Common Raven, C = American Crow) observed taking an egg, chick, fledgling, or adult, or tracks led to tern remains or empty nest where eggs were not expected to hatch. for at least 3 more days. If expected hatch date was unknown, tracks led to more than 1 empty nest, any evidence left had to be consistent with that expected from the indicated predator

3 – Corvids (R = Common Raven, C = American Crow) believed to have preyed on terns or eggs, based on substantial but not conclusive evidence (e.g. tracks throughout site, tern remains, or predators observed foraging at the site).

Appendix C. Cooperating agencies and actions for a proposed state-wide corvid action plan.

Action	**Agencies***
Public Involvement	
Educate the public about corvid impacts through signage, pamphlets, and popular articles	BLM, FWS, CDFG
Developing programs the public can participate in involved in dealing with corvid impacts to listed species	BLM, FWS, CDFG
Non-lethal management	
Limit corvid access to anthropogenic food and water sources	Caltrans, FWHA, CDFG, BLM, FWS, County and city governments, NPS, CSP
Limit perch, roost, and nest sites	BLM, FWS, CDFG, WS, DOD
Monitor local corvid populations that are affecting listed species	BLM, FWS, CDFG, DOD
Identify nest predators at breeding sites	BLM, FWS, CDFG, WS
Develop public use areas designed to limit corvid impacts on listed species	BLM, FWS, CDFG, NPS
Implement non-lethal management actions	BLM, FWS, CDFG, WS, DOD
Lethal management	
Lethal removal	BLM, FWS, CDFG, WS, DOD
Research	
Obtain life-history information on local corvid populations	BLM, FWS, CDFG, NPS, Parks
Develop new corvid management techniques	BLM, FWS, CDFG
Develop demographic models of corvid populations	BLM, FWS, CDFG
Administrative	
Establish interagency workgroup	BLM, FWS, CDFG, DOD, WS, Counties, conservation and animal welfare representatives, CSP
Develop an adaptive management approach	BLM, FWS, CDFG, DOD, WS, Counties, CSP

*BLM = U.S. Bureau of Land Management
FWS = U.S. Fish and Wildlife Service
DOD = U. S. Department of Defense
WS = Wildlife Services (Formerly Animal Damage Control)
FHWA= Federal Highways Administration
CDFG = California Department of Fish and Game
NPS = National Park Service
CSP = California State Parks