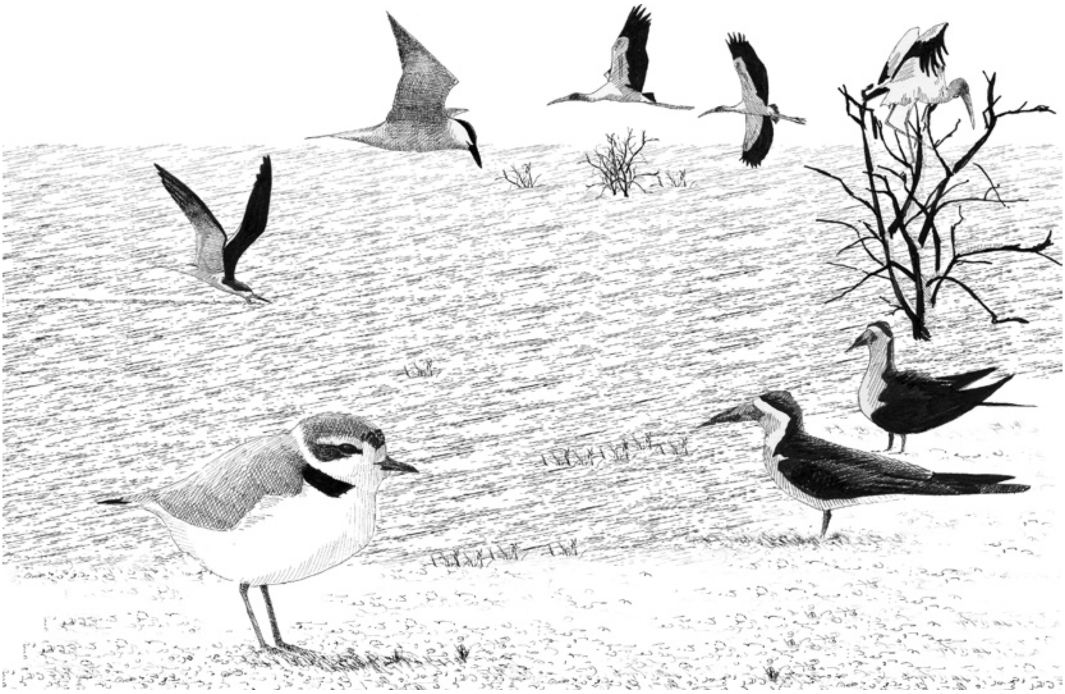
II

SPECIES ACCOUNTS



Andy Birch

PDF of Northern Harrier account from:

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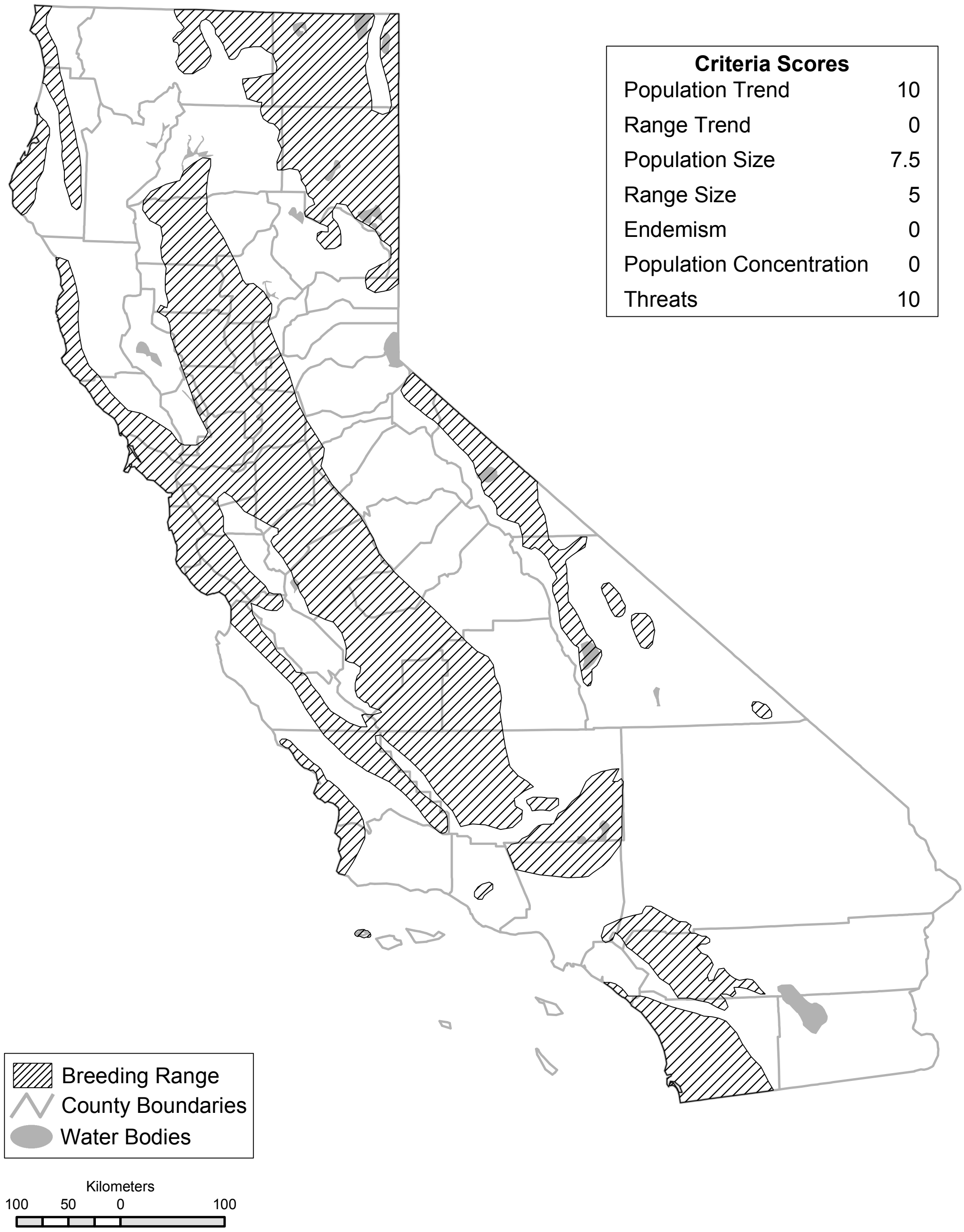
NORTHERN HARRIER (*Circus cyaneus*)

Jeff N. Davis and Chris A. Niemela



Criteria Scores	
Population Trend	10
Range Trend	0
Population Size	7.5
Range Size	5
Endemism	0
Population Concentration	0
Threats	10

Breeding range of the Northern Harrier in California; numbers have declined at least moderately since 1944. Also occupies most remaining lowland areas of the state in the nonbreeding season, when numbers are swelled greatly by out-of-state migrants.

SPECIAL CONCERN PRIORITY

Currently considered a Bird Species of Special Concern (breeding), priority 3. Included on both prior special concern lists (Remsen 1978, 2nd priority; CDFG 1992).

GENERAL RANGE AND ABUNDANCE

Two subspecies: *C. c. cyaneus* in the Old World and *C. c. hudsonius* in the New World. Breeds widely but locally in North America from northern Alaska and Canada south to mid- and lower latitudes of the United States and northern Baja California. Occurs year round in much of its breeding range in the contiguous United States and locally in southwestern and southeastern Canada. Populations in Alaska, most of Canada, and much of the midwestern and northeastern United States are migratory and winter from southern Canada (locally) to Central America (MacWhirter and Bildstein 1996). As a breeder, appears to be most numerous in the prairies and plains from southern Canada to the Dakotas and Montana (Bildstein 1988). Using Christmas Bird Count (CBC) data, Johnsgard (1990) estimated the North American wintering population in 1986 to be 111,500 birds. This is likely a minimum estimate, because an unknown proportion of breeders within this range winter south of the United States, where CBC coverage is minimal, and because it excludes breeders in Baja California.

SEASONAL STATUS IN CALIFORNIA

Occurs year round within breeding range in California. At least some breeding populations may be resident. The species occurs more broadly and in much greater numbers during migration and winter than during the breeding season, which extends from March through August (Loughman and McLandress 1994). The species appears to be nomadic, ranging widely, both within the breeding season and across years (Pavelka 1992, P. Bloom pers. comm.).

HISTORIC RANGE AND ABUNDANCE IN CALIFORNIA

The historic breeding range extended from the Modoc Plateau south to San Diego, mostly east and south of the humid northwest coast and west and north of the arid southeastern deserts (Grinnell 1915). Birds bred locally within this range, including near Mount Shasta City, Siskiyou County; at Point Reyes, Marin County; Pescadero, San Mateo County; Alviso, Santa Clara County; Modesto, Stanislaus County; near June Lake, Mono County; Los Banos, Merced County; near Salinas, Monterey County; Kings River (28 mi west of Tulare), Kings County; Dune Lakes, San Luis Obispo County; Buena Vista Lake, Kern County; Oxnard, Ventura County; Palos Verdes Hills, Los Angeles County; Corona, Riverside County; Chino, San Bernardino County; Bay City, Orange County; and San Diego, San Diego County (Grinnell and Miller 1944; CAS, MVZ, WFVZ egg set data). The known breeding range extended from about sea level near the coast to 8000 ft (2438 m) near June Lake. In the early 20th century, the species was considered a "common" breeder in California (Dawson 1923, Mailliard 1927, Willett 1912). Compared with winter, however, "relatively small numbers" remained in the state through summer to breed, and by the early 1940s the breeding population had declined substantially because of a great loss of suitable habitat (Grinnell and Miller 1944). Much of that suitable habitat was wetlands. Loss of California's wetlands began in the mid-19th century, when farmers began diking and draining them for cultivation (Dahl 1990), and accelerated in the early 20th century, so that by 1939 at least 85% of the original acreage had been modified by levees, drainage, and water-diversion projects (Hartman and Goldstein 1994). Similarly, by 1945 about 70% of the state's original native grasslands, another key habitat, had been lost to agriculture, urban development, livestock grazing, fire suppression, and exotic species invasion (Noss et al. 1995).

BREEDING BIRD SURVEY STATISTICS FOR CALIFORNIA

1968–2004					1968–1979			1980–2004			All data from Sauer et al. (2005)
Trend	*P*	*n*	(95% CI)	R.A.	Trend	*P*	*n*	Trend	*P*	*n*	Credibility
2.2	0.31	56	–2.0, 6.4	0.52	6.8	0.15	31	1.9	0.49	45	Medium

Recent Range and Abundance in California

The outline of the breeding range has changed little since the mid-1940s (see map), and harriers breed from sea level near the coast to at least 9000 ft (2743 m) in the Glass Mountain region of Mono County (Shuford and Metropulos 1996). Regardless, overall numbers have been reduced and some local populations have been extirpated. Still, breeding densities in some regions of the state (see below) currently are higher than anywhere else in North America (highest density outside California, approximately 2 nests per km^2; MacWhirter and Bildstein 1996). While local declines in breeding numbers have been documented in some regions of California, declines elsewhere in the state can only be inferred by loss or degradation of suitable breeding habitat. Notably, California lost 26% of its remaining native grasslands between 1945 and 1980 (Noss et al. 1995) and 34% of its remaining wetlands between 1954 and 1985 (Hartman and Goldstein 1994). Using CBC data from 1986, Johnsgard (1990) estimated California's wintering population at 13,200 birds. Because harriers are much more numerous in the state in winter than summer, the breeding population is surely many fewer. Actual breeding numbers vary greatly from year to year with rainfall and prey abundance, probably because the species' primary habitats, marshes and grasslands, vary annually in quality and extent (MacWhirter and Bildstein 1996).

Northwestern California. Harriers breed locally in this region, largely within the coastal lowlands, from Lake Earl in Del Norte County (Barron 2001) south to Bodega Head in Sonoma County (Burridge 1995), but also inland at Lake Berryessa in Napa County (Berner et al. 2003). The Humboldt County breeding bird atlas (BBA) found harriers in 31 blocks, mostly along the coast from Clam Beach south through the Humboldt Bay lowlands (Hunter et al. 2005). In Mendocino County, the species breeds along the coast near Fort Bragg and at MacKerricher and Manchester state beaches (≤12 pairs, R. Keiffer in litt.). Within this region, atlasers found harriers in eight blocks in Sonoma County, half at Bodega Head (Burridge 1995), and in two blocks in Napa County at Lake Berryessa (Berner et al. 2003).

Northeastern California. Northern Harriers breed widely in this region. The centers of abundance are Shasta Valley, Butte Valley, Klamath Basin around Lower Klamath and Tule Lake NWRs, Modoc NWR and Pit River Valley, Surprise Valley, Big Valley, Honey Lake Valley, Sierra Valley, valleys of Mono and Inyo counties (Antelope, Adobe, Bridgeport, Long, Deep Springs, and Owens), and the Mono Basin. At Ash Creek and Honey Lake WAs in Lassen County, Loughman and McLandress (1994) located an average of 15 nests (13–18) per year in 1987–1989, for an average density of 8.2 nests per km^2.

Central Valley. Although most of its original habitat has been destroyed or degraded, this region still supports the majority of nesting harriers in California. Harriers there breed mainly at private or public wetlands or other reserves, as well as in some types of agricultural fields and pasturelands. In the Sacramento Valley at Gray Lodge WA and Sacramento, Delevan, and Colusa NWRs, Loughman and McLandress (1994) located 13 nests in 1987 and 11 nests in 1988, for a mean density of 5.0 nests per km^2. About 7–14 pairs of harriers breed annually at the Cosumnes River Preserve, Sacramento County (J. Trochet in litt.), and that county's BBA project (unpubl. data) found them in 69 widely scattered blocks.

In Suisun Marsh at Grizzly Island WA in Solano County, Loughman and McLandress (1994) located an average of 25 nests (12–72) per year in 1987–1992, for a mean density of 8.4 nests per km^2. Atlasers found harriers in 24 blocks in the Contra Costa County portion of this region and in 7 in the Alameda County portion (unpubl. data). In the San Joaquin Valley at Mendota WA in Fresno County, Loughman and McLandress (1994) found an average of 9 harrier nests (5–15) per year in 1987–1991, for a mean density of 5.9 nests per km^2. A recent decline in harrier abundance throughout the Central Valley is inferred by a significant loss of suitable breeding habitat (see "Threats" below).

Central coast. Harriers breed widely in this region. Atlasers found them in 48 blocks throughout the coastal lowlands in Marin County (Shuford 1993), in 13 blocks within the Sonoma County portion of this region, mostly along the Petaluma River and near Tubbs Island (Burridge 1995), and in 3 blocks, near the Napa Airport and Edgerley Island, in the Napa County portion of the region (Berner et al. 2003). Harriers were widespread in San Mateo, Contra Costa, and Alameda counties, where they occurred in 19, 9, and 16 atlas blocks, respectively (Sequoia Audubon Society 2001, unpubl. atlas data). In Santa Clara County, atlasers located harriers in 14 blocks along San Francisco Bay and in Pajaro Valley (W. G. Bousman in litt.). In Santa Cruz County, harriers breed in coastal lowlands from near Swanton south to Wilder

Creek and in the foothills east of Watsonville (about 20 pairs, D. Suddjian in litt.). The only known breeding area in San Benito County is near San Felipe Lake (at least 1 pair, D. Shearwater in litt.). In Monterey County, where breeding numbers have declined in recent decades, atlasers found harriers in 20 blocks, from Elkhorn Slough and the Salinas River mouth southeast through the Salinas Valley (Roberson 1993). The San Luis Obispo County BBA found harriers in 39 blocks, mostly inland (unpubl. data). In Santa Barbara County, harriers breed at Vandenberg Air Force Base (about 12 pairs) and at San Miguel Island (1–2 pairs, D. Compton in litt.).

Southern coast. The species' range here is highly fragmented, and many local populations have been extirpated. In Ventura County, harriers are now restricted to a section of the Santa Clara River near Santa Paula, where they breed irregularly (<1 pair per year, W. Wehtje in litt.). In the Los Angles County portion of the region, harriers were documented in 5 atlas blocks (unpubl. atlas data). In Orange County, harriers were found in 9 atlas blocks, primarily near the coast, and known breeding pairs have declined over the past 35 years, from about 10 to an average of <1 successful nest per year (Gallagher and Bloom 1997, P. Bloom pers. comm.). The species no longer nests in salt marshes (Sexton and Hunt 1979), at Seal Beach (Hall 1947), or the San Joaquin Hills (Hamilton and Willick 1996). Harriers breed irregularly in small numbers in western Riverside County (J. Green in litt.). An average of 3.2 birds per year were detected on the Moreno Breeding Bird Survey route (Sauer et al. 2005), from the Badlands through the Moreno Valley, and breeding was suspected at the San Jacinto WA in 2004 (J. Green in litt.) and Tripp Flats near Anza in 2003 (S. Myers in litt.). The region's center of abundance is in San Diego County, where atlasers found harriers in 75 blocks, primarily toward the coast and including Camp Pendleton, Los Peñasquitos Canyon, and the Tijuana River estuary (Unitt 2004).

Southern deserts. Suitable habitat is extremely limited in this region. Harriers breed in the Saline and Panamint valleys (1 pair each) and the Lake Grimshaw area near Tecopa (1 pair) in Inyo County (T. & J. Heindel in litt.) and in the Fremont Valley near Cantil in eastern Kern County (Heindel 2000). Although Harper Dry Lake in western San Bernardino County had long supported harriers, breeding has not been suspected there since the mid-1990s (S. Myers in litt.). The center of abundance in this region is northern Los Angeles County, where atlasers found harriers in 8 blocks in the Antelope Valley and near Lancaster (unpubl. data).

ECOLOGICAL REQUIREMENTS

Northern Harriers breed and forage in a variety of open (treeless) habitats that provide adequate vegetative cover, an abundance of suitable prey, and scattered hunting, plucking, and lookout perches such as shrubs or fence posts. In California, such habitats include freshwater marshes, brackish and saltwater marshes, wet meadows, weedy borders of lakes, rivers and streams, annual and perennial grasslands (including those with vernal pools), weed fields, ungrazed or lightly grazed pastures, some croplands (especially alfalfa, grain, sugar beets, tomatoes, and melons), sagebrush flats, and desert sinks (MacWhirter and Bildstein 1996, J. Silveira in litt., J. Seay in litt.).

Harriers nest on the ground, mostly within patches of dense, often tall, vegetation in undisturbed areas (MacWhirter and Bildstein 1996). Plant species composition varies by site, and the average height of vegetation surrounding nests varies regionally and annually (Loughman and McLandress 1994).

Harriers feed on a broad variety of small- to medium-sized vertebrates, primarily rodents and passerines. Species taken and the proportion of mammals to birds probably vary locally and annually with abundance and availability. In many areas, voles (*Microtus* spp.) are the most common prey (Bildstein 1988). Wet habitats, including irrigated agriculture, tend to support large numbers of California Voles (*M. californicus*; Krebs 1966), a key food item in California. Waterbirds, primarily American Coots (*Fulica americana*), are common prey in the Klamath Basin (D. Mauser in litt.). Of 438 food items delivered to four nests in San Luis Obispo County, 80.6% were birds (mostly blackbirds and sparrows), 18% mammals (mostly Brush Rabbits [*Sylvilagus bachmani*] and California Voles), and 1.4% reptiles (mainly Western Fence Lizards [*Sceloporus occidentalis*]; Selleck and Glading 1943).

Although generally monogamous, Northern Harriers may also be polygynous, with harems of two to five females, especially in years of high prey abundance (MacWhirter and Bildstein 1996).

THREATS

The primary threats to breeding harriers are loss and degradation of nesting and foraging habitat and nest failure from human disturbance, predator-control projects, agricultural practices, and unnatural predation pressure. California's wetlands continue to be lost at an average rate of over 2000 ha per year (Noss et al. 1995). From 1992 to 1997, rural land in the state was lost to development at an average rate of 45,592 ha per year (American Farmland Trust 2004). Likewise, an average of 33,451 ha of agricultural land was converted to developed uses each year from 1992 to 1997. Conversion of pastureland and suitable crops, such as alfalfa, to unsuitable crops, such as vineyards and orchards, poses a substantial threat to nesting harriers in the Central Valley (e.g., Schweizer and Chesemore 1996) and has resulted in local extirpations in other areas (e.g., at Harper Dry Lake, S. Myers in litt.). In addition, overgrazing, haying, agricultural intensification, and the widespread use of rodenticides can degrade habitat by reducing numbers of small mammals on which harriers depend (MacWhirter and Bildstein 1996, Schweizer and Chesemore 1996). Decreasing water supplies may be a rising threat to harrier nesting habitats statewide. Recent water shortages in the Klamath Basin, for example, have reduced water allocations to federal wildlife refuges, resulting in diminished wetlands during the breeding season (D. Mauser in litt.).

Human disturbance is a source of nest failure throughout most of the species' range in California. People walking or recreating near nests, off-leash dogs, and off-highway vehicles are the main sources (Burridge 1995, Unitt 2004). In coastal California, another important threat is predator management aimed at protecting imperiled Light-footed Clapper Rails (*Rallus longirostris levipes*), coastal Western Snowy Plovers (*Charadrius alexandrinus nivosus*), and California Least Terns (*Sterna antillarum browni*), which annually results in the loss of adult harriers and losses in their productivity (P. Bloom pers. comm., K. Neuman pers. comm.). Such management often occurs in areas where habitat loss, human disturbance, and unnatural predation levels have already taxed harrier populations (P. Bloom pers. comm.).

Ground nests of harriers are highly vulnerable to trampling by livestock, haying, plowing, flooding, and fire associated with some agricultural operations and management activities (MacWhirter and Bildstein 1996, Hunter et al. 2005, D. Shearwater in litt.). Predation of eggs and young by non-native Red Foxes (*Vulpes vulpes*) is a growing threat, and feral cats and dogs place increasing pressure on harriers attempting to nest near urban areas (Roberson 1993, Gallagher and Bloom 1997). Contamination of the food chain by organochloride pesticides, especially DDT, was a threat in the mid-20th century, when it resulted in reproductive failure and population declines in harriers in North America (MacWhirter and Bildstein 1996). Following the regulation of DDT in the early 1970s, however, harrier populations recovered relatively quickly.

MANAGEMENT AND RESEARCH RECOMMENDATIONS

- Minimize human disturbance near nesting areas, restricting public access as necessary during the breeding season.
- Reduce livestock impacts on nesting success by limiting their access to harrier nesting areas, especially during the breeding season.
- Maintain a mosaic of large undisturbed habitats for nesting and foraging, particularly of those with an abundant prey base, for example, abandoned fields, active alfalfa fields, wet grasslands, fields with dense green and residual vegetation.
- Practice rotational grassland management, leaving some sections idle each year.
- Delay haying and plowing when possible until after nestlings have fledged (ca. mid Jul).
- Avoid raising wetland water levels during the nesting season to prevent flooding nests of harriers and other ground-nesting species.
- Conduct long-term studies on survival, reproduction, dispersal, and other factors limiting harriers in California; especially determine whether reproduction and survival differ between birds using natural habitats (wetlands and grasslands) and those using anthropogenic habitats (croplands and pasturelands).
- Study the effects of patch size and fragmentation on habitat use and nest success.
- Investigate the effects of environmental contaminants, including insecticides and rodenticides, on harriers and on their prey populations.

MONITORING NEEDS

A reliable annual statewide monitoring program for harriers is needed because present methods are inadequate. The Breeding Bird Survey is insufficient for species such as the harrier that occur in relatively low abundance, and CBC data are coarse and do not cover the California breeding population, which is the one of concern. Harriers probably are best monitored using road or foot survey routes stratified by habitat. Routes should be surveyed at least once a month from March through August to determine site occupancy and document breeding evidence. Reproductive success in high-density nesting areas (e.g., state and federal refuges) also should be monitored to gauge whether actions are needed to enhance success.

ACKNOWLEDGMENTS

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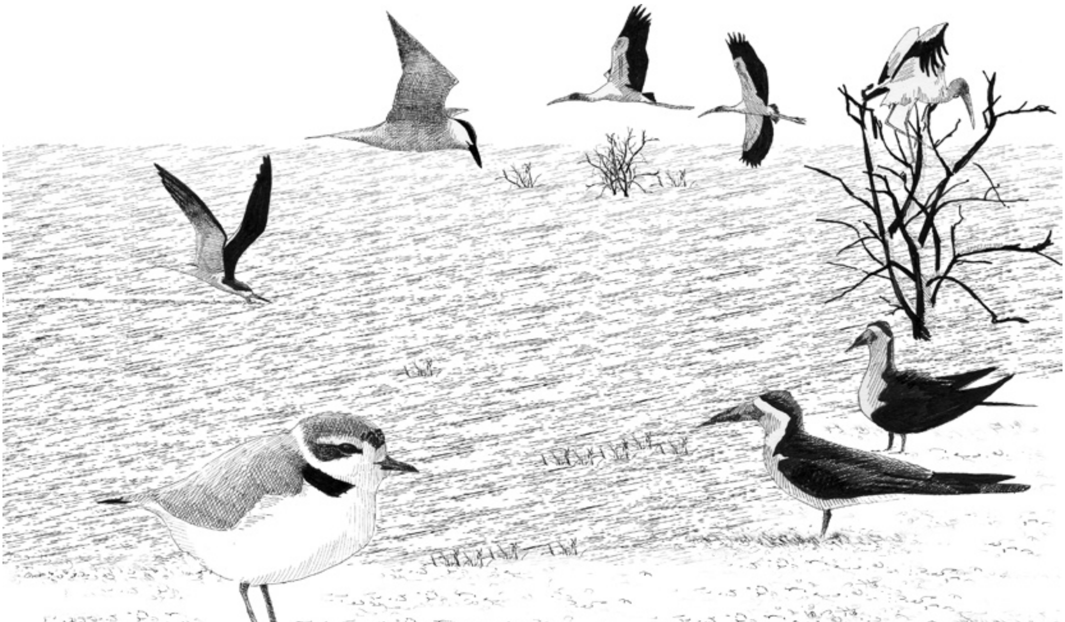
SPECIES ACCOUNTS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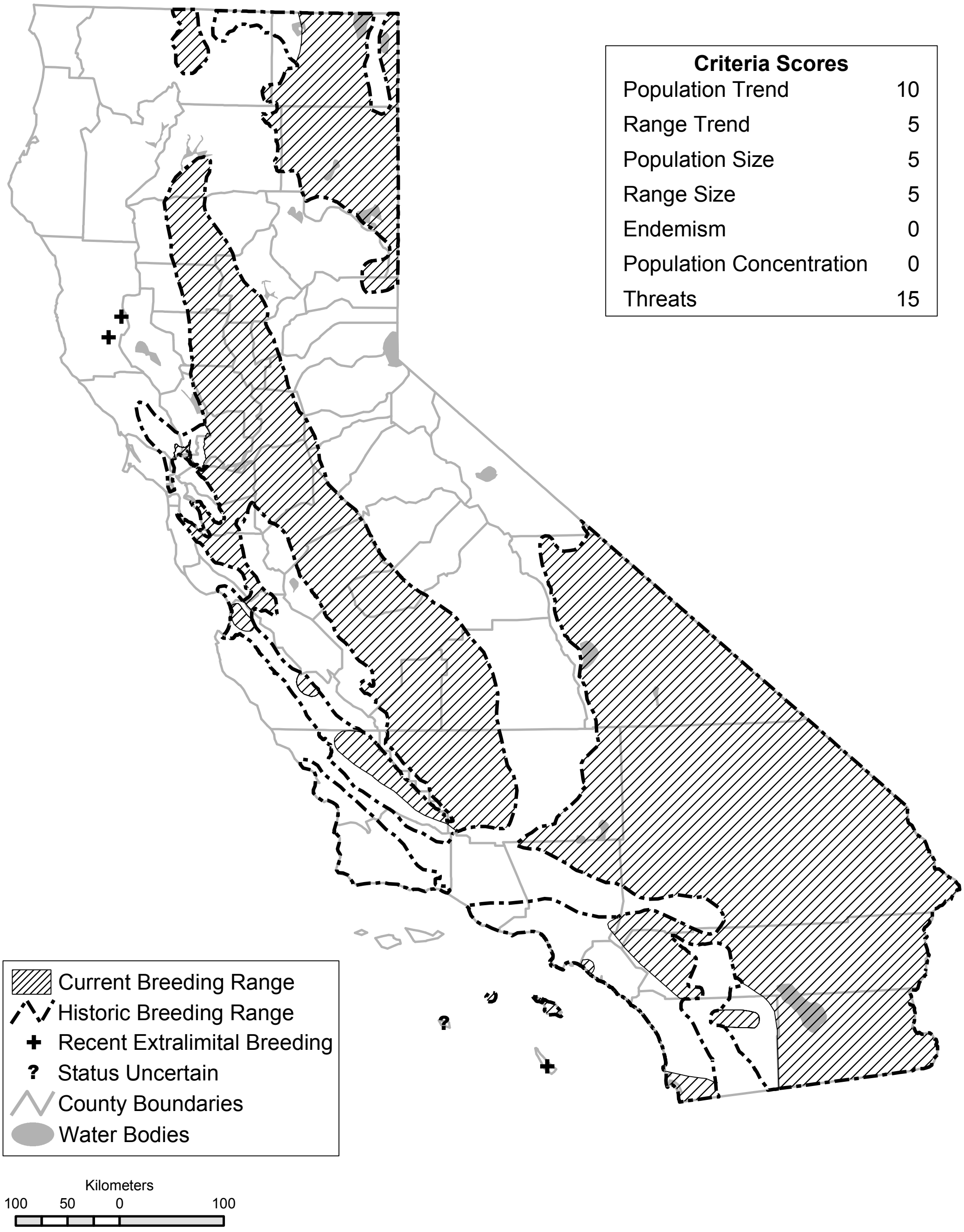
BURROWING OWL (*Athene cunicularia*)

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



Criteria Scores	
Population Trend	10
Range Trend	5
Population Size	5
Range Size	5
Endemism	0
Population Concentration	0
Threats	15

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

Special Concern Priority

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

General Range and Abundance

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

Seasonal Status in California

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

Historic Range and Abundance in California

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

Recent Range and Abundance in California

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

Breeding Bird Survey Statistics for California

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

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

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

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

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

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

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

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

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

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

Ecological Requirements

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

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

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

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

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

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

THREATS

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

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

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

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

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

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

MANAGEMENT AND RESEARCH RECOMMENDATIONS

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

MONITORING NEEDS

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

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

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HOME RANGE AND HABITAT USE OF BREEDING SWAINSON'S HAWKS IN THE SACRAMENTO VALLEY OF CALIFORNIA

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ABSTRACT.—Four adult Swainson's hawks (*Buteo swainsoni*) were radiotagged along the Sacramento River in 1992. The mean home range (minimum convex polygon) was 4038.4 ha (40.4 km^2). Core areas of intensive use (adaptive kernal) by nesting Swainson's hawks ranged from 25.9–82.2 ha. Individual hawks foraged as far as 22.5 km from the nest. In the Sacramento Valley, foraging ranges and total home range area were strongly influenced by agricultural patterns and cover types. Ruderal and fallow fields, grain crops, and safflower were the vegetative cover types that ranked highest in foraging use. The predominance of less suitable cover types within the study area may explain the relatively large home ranges exhibited by the Swainson's hawks in this study.

KEY WORDS: Buteo swainsoni; *foraging ecology*; *habitat use*; *home range*; *Swainson's hawk.*

Rango de hogar y uso del hábitat de *Buteo swainsoni* reproductivos en el Valle Sacramento de California

RESUMEN.—En 1992, cuatro individuos adultos de la especie *Buteo swainsoni* fueron radio-marcados a lo largo del Río Sacramento, California. La media de rango de hogar (polígono convexo mínimo) fue de 4038.4 ha (40.4 km^2). Areas núcleo de uso intensivo por parte de *B. swainsoni* se encontraban dentro de un rango de 25.9 a 82.2 ha. Los individuos de *B. swainsoni* se alimentaban en sitios distantes hasta 22.5 km del nido. En el Valle Sacramento, los rangos de forrajeo y el área total de rango de hogar, fueron fuertemente influenciados por patrones agrícolas y tipos de cubierta vegetacional. Campos ruderales y abandonados, cosechas de granos y cártamo fueron los tipos de cubiertas vegetativa de mayor uso como sitios alimentarios. La predominancia de cubiertas vegetales menos utilizadas en el sitio de estudio pueden explicar el rango de hogar relativamente grande exhibido por esta especie.

[Traducción de Ivan Lazo]

The Swainson's hawk (*Buteo swainsoni*) was common historically throughout most of the lowland grassland and riparian communities that once occupied the Central Valley of California (Grinnell and Miller 1944). However, an estimated 90% decline of the breeding population of this species in recent years (Bloom 1980) resulted in the listing of the Swainson's hawk in California as a threatened species. The current breeding range of the Swainson's hawk in California is generally comprised of two populations, one located in the Great Basin area in the northeastern corner of the state, and the other, larger population located primarily in the middle portion of the Central Valley (the Sacramento Valley) near Sacramento (Bloom 1980).

Very little is known about the breeding home range and foraging habitat requirements of the Swainson's hawk in the Sacramento Valley. And yet, this region is home to the highest concentration of Swainson's hawks in the state (Bloom 1980). Previous studies suggest that home-range sizes can vary significantly in response to agriculture, changes in prey availability, and various farming practices (Bechard 1982, Estep 1989, Woodbridge 1991). Using radiotelemetry, I determined home-range sizes, core-use areas, and habitat use of a small population of nesting Swainson's hawks in the Sacramento Valley.

STUDY AREA

This study was conducted in an open rural area within the city of West Sacramento, bordered on the east by the Sacramento River and the city of Sacramento. Agricultural cropland, pastureland, and areas of non-native grassland comprised the majority of the open space areas in the region. Common crop types included wheat, corn, tomatoes, alfalfa, onions, sugar beets, and safflower. Dense urban areas associated with West Sacramento and Sacramento occured to the north and east of the study area. Narrow riparian areas dominated by Frémont cottonwood (*Populus fremontii*), valley oak (*Quercus lobata*), walnut (*Juglans* sp.), willow (*Salix* sp.), and box elder (*Acer negundo*) occur along the Sacramento River to the east and along Putah Creek to the west. Isolated oak woodlands occur sporadically throughout the residential and agricultural areas.

METHODS

Swainson's hawks were trapped using dho-gazas (Hamerstrom 1963). A bal-chatri trap (Berger and Mueller 1959) and a noose carpet (Collister 1967) were used for a pair of Swainson's hawks that avoided the dho-gaza. All captured Swainson's hawks were weighed, sexed (determined by the presence or absence of a brood patch and by overall size and weight), and fitted with backpack transmitters weighing from 19.2–19.8 g. Radio signals were received using ICOM IC-03AT transceivers and three-element Yagi antennas. Each trapped hawk was also fitted with a numbered, colored plastic leg band and a standard U.S. Fish and Wildlife Service aluminum leg band.

Tracking began after each bird was fitted with a transmitter and released. In the Sacramento Valley, Swainson's hawks often congregate in large groups and begin migrating southward in September (Bloom 1980, Estep 1989). Tracking was discontinued on 31 August since home range and foraging information obtained after this period was not expected to be strongly correlated with nest territories. Each bird was followed from dawn until dusk at least 2 d/wk during the study period (1 June to 31 August).

Because of the very active and aerial nature of Swainson's hawks, these birds are regularly lost to view during periods of high-altitude soaring and straight flight. Data were recorded in 5-min intervals and only when the bird was visually observed. Behavioral information was recorded in terms of foraging or nonforaging. Foraging behavior included circling, hovering, stooping, and feeding. Nonforaging behavior included straight flight, perching (unless, because of location and habitat, it was considered foraging from a perch), incubating, and preening. Location points were plotted on aerial photographs containing field numbers for each cover type.

A geographic information system (GIS) was used to map land uses and observational points within the study area. Information associated with each observation (time, date, hawk number, vegetation type, behavior) were also incorporated into the database. Home range calculations for each radio-marked Swainson's hawk were later imported into the GIS database in order to create home-range polygons. These polygons were then overlain onto the study area map to enable analysis of hawk foraging habitat and to compare individual home ranges.

To avoid autocorrelation of data, only observations separated by at least 0.5-hr intervals were used to determine home ranges and habitat use. Lair (1987) suggested that observation points may be considered biologically independent if sufficient time has passed for the animal to have moved to a new location or, for the purposes of this study, to cross its home range. For this study it was estimated that it would take a Swainson's hawk no more than 0.5 hr to cross its home range.

Home ranges were calculated using the CALHOME program developed by J. Kie (unpubl.), and were based on field observations and locations plotted over the entire duration of the study. The home range of each hawk was determined using the minimum convex polygon (MCP) method. Because use of this method includes outlier location points (occasional or isolated movements to locations outside the normal use area) which tend to overestimate home-range sizes, a 95% contour level was used in order to exclude these points. A 50% contour level using the adaptive kernal (AK; Worton 1987) method was used for delineating core-habitat-use areas (those land areas that are used most extensively by nesting hawks as foraging habitat) within the home range. Core-use areas at the 50% MCP level were also determined for comparison.

To evaluate habitat use, information on the vegetative cover type or crop type at each Swainson's hawk observation point was also recorded. A chi-square analysis was used to compare Swainson's hawk habitat use with habitat availability.

RESULTS

Four adult Swainson's hawks, three males and one female (which was mated to one of the males), were trapped and radiotagged (Table 1). Attempts were made to trap all adults from the six pairs in the study area. The first hawk was trapped on 2 June 1992, and the last was trapped on 10 July 1992. Each radio-tagged hawk was tracked for an average of 138 hr over the duration of the study. The number of biologically independent points for each hawk ranged from 73–122.

Home Range and Core-Use Areas. Home ranges of the four radio-tagged hawks were relatively large (Table 1). At the 95% MCP contour level, home ranges varied from 723.6–7658.8 ha ($\bar{x}$ = 4038.4 ha, SD = 5348.4 ha, $N = 4$) and were linear in nature (Fig. 1). Home ranges of the three males were larger than that of the female, and averaged 5143.3 ha. The furthest any individual hawk foraged from the nest was 22.5 km.

The size (50% AK) of the core-habitat-use areas ranged from 25.9–82.2 ha ($\bar{x}$ = 48.2 ha, SD = 21.8 ha, $N = 4$) (Table 1). These core areas were generally located in the immediate vicinity of each nest. For comparison, mean core-use areas using the MCP technique was 86.5 ha (Table 1).

Habitat Use. Dominant cover types within the home ranges (100% MCP) of the radio-tagged Swainson's hawks were grain crops (17.4% of the total undeveloped land potentially available as Swainson's hawk foraging habitat), ruderal/fallow fields (16.3%), row crops (corn/milo/sudan grass; 10.9%), tomatoes (10.6%), and safflower (10.2%). When observed habitat use by the radio-tagged Swainson's hawks was compared to habitat availability, Swainson's hawks did not forage in a habitat in proportion to its availability, but were observed most often foraging over ruderal/fallow fields, alfalfa, and pastureland ($\chi^2 = 31.3$, df = 11, $P < 0.001$).

Foraging Behavior. Both sexes of the radio-tagged

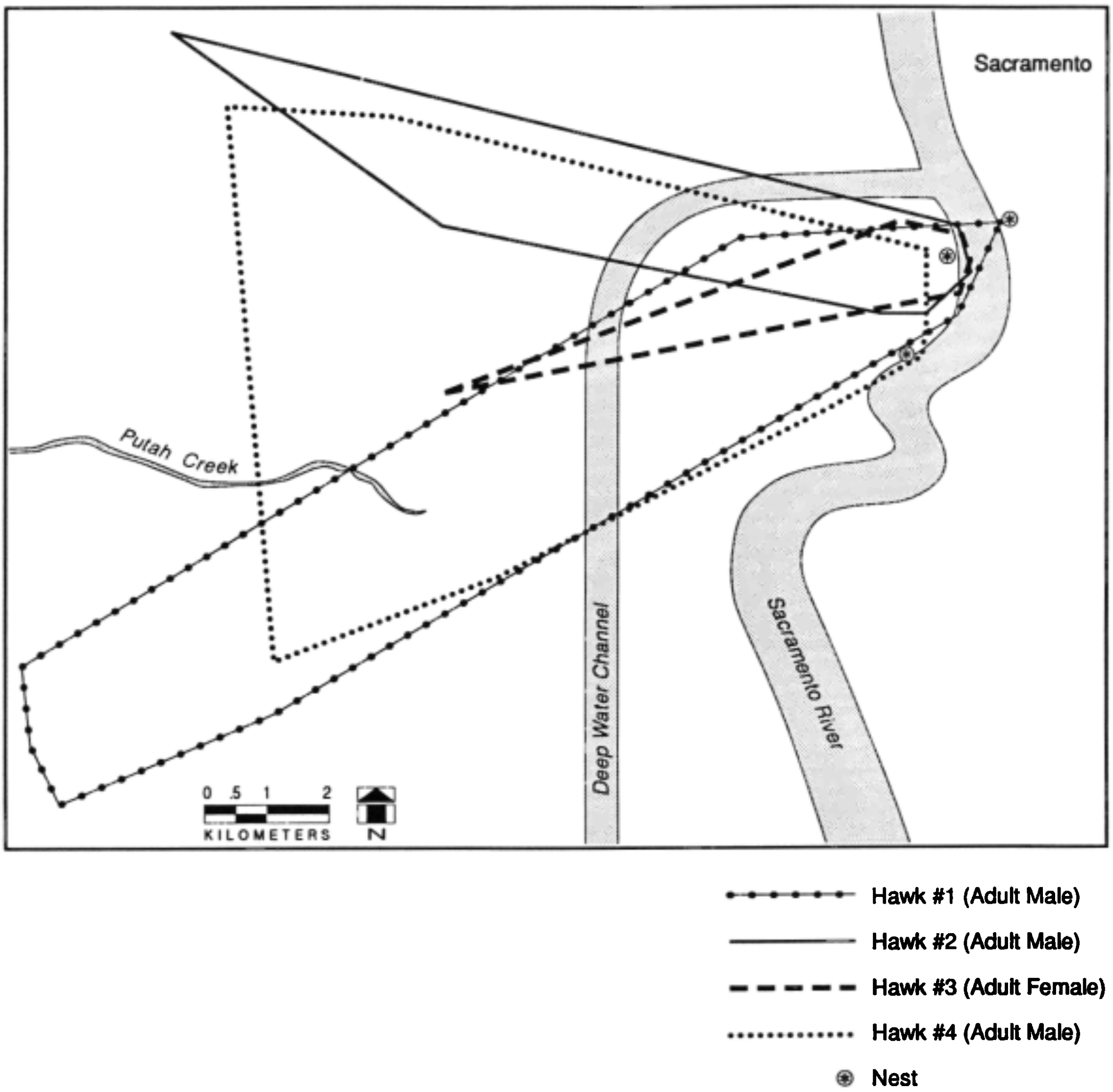

Figure 1. Nest locations and home range sizes (95% MCP) of four radio-tagged Swainson's hawks in the Sacramento Valley of California. Hawks #2 and #3 were mates.

Swainson's hawks were observed foraging almost exclusively from the air. The hawks were highly active and never spent much time over a particular field unless attracted by cutting or harvesting activities. In some instances, particularly in late July and August, large groups of Swainson's hawks, including one that contained approximately 130 individuals, were observed foraging over several adjacent fields that were undergoing some form of cutting or harvesting. Many of these birds appeared to be making shallow aerial stoops, apparently chasing and capturing flying insects.

After fields were cut, or in the case of some fields that were recently irrigated, radio-tagged hawks were often observed foraging from the ground. These birds would wait for a small rodent or insect to pass by,

Table 1. Home range information from radio-tagged Swainson's hawks in Yolo County, California, 1992.

HAWK SEX	CAPTURE DATE[a]	TOTAL HOURS TRACKED	TOTAL OBSERVATION POINTS	TOTAL FORAGING POINTS	TOTAL BIOLOGICALLY INDEPENDENT POINTS[b]	HOME RANGE (HA)[c]		
						95% MCP[d]	50% MCP	50% AK[e]
M	2 June	132	445	277	122	5339.0	21.0	32.7
M	10 July	120	380	268	80	2432.2	223.9	25.9
F	21 June	120	347	216	73	723.6	12.0	52.2
M	4 June	180	453	258	91	7658.8	88.9	82.2
Mean		138	406	255	92	4038.4	86.5	48.2

[a] Tracking ended 31 August.

[b] Total number of foraging points collected at a time interval (0.5 hr) sufficient to allow a radio-tagged Swainson's hawk to cross its home range.

[c] Based on biologically independent observation data.

[d] MCP = minimum convex polygon.

[e] AK = adaptive kernal.

and would then quickly pounce upon the particular prey item. Usually, the prey would be consumed on the ground where it was caught, especially if it was an insect (no attempt was made to identify prey items to taxonomic species). Fields containing 15–20 Swainson's hawks foraging from the ground were observed on two occasions in July and on three occasions in August.

DISCUSSION

Foraging ranges and total home range area of raptors are known to be influenced by prey abundance and prey accessibility (usually a function of vegetation cover and density), nest location, the total amount of available suitable foraging habitat within the home range, and type of vegetation (Wakeley 1978, Baker and Brooks 1981, Bechard 1982, Schmutz 1987, Estep 1989, Woodbridge 1991). Bechard (1982) reported a strong correlation between home range size of Swainson's hawks and the amount of suitable foraging habitat that was available. Preston (1990) found that red-tailed hawks (*Buteo jamaicensis*) and northern harriers (*Circus cyaneus*) responded to changes in prey abundance and cover density; patches of vegetation containing high prey populations but with dense vegetative cover were used by both species less frequently than predicted. In agricultural areas, the abundance and accessibility of prey such as small rodents and insects may change in response to growth, maturity, and harvest of certain crops. In the Sacramento Valley where agriculture is the dominant land use, Estep (1989) found that as crops matured and vegetative cover increased, Swainson's hawks enlarged their foraging ranges in order to find more accessible prey; as crops and fields nearer the nest area were cut or harvested, the foraging range was reduced, sometimes even to a single field. Although no statistical analysis was conducted in this study to determine the correlation of home-range size with agricultural activities (crop cutting or harvesting), I suspect that foraging ranges of the radio-tagged Swainson's hawks increased in size as preferred crop types matured and prey become less accessible, and decreased during periods of harvesting and mowing when prey suddenly become more available.

In the Sacramento Valley, where changing agricultural markets and the juxtaposition of agriculture areas with urban development has resulted in a wide variety of agricultural cover types dispersed over very large areas, Swainson's hawk home ranges tend to be somewhat large. Estep (1989) reported a mean home range of 2760.4 ha for Swainson's hawks in the Central Valley, which compares to the large home ranges found in this study (despite the relatively small sample size in this study). However, in areas where the land use includes a predominance of cover types with a continually available prey base and abundant prey populations, Swainson's hawks may require substantially smaller home ranges in which to breed. Woodbridge (1991) found Swainson's hawks in northeastern California that nested in areas surrounded by cover types that were high in prey density and prey accessibility and low in vegetative cover were associated with very small, circular home ranges (mean equal to 405.0 ha).

Grain crops, ruderal/fallow fields, row crops, tomatoes, and safflower were the dominant cover types in the study area. Estep (1989) found that crop patterns in the Central Valley that included a predominance of cover types with less overall vegetative cover and greater prey availability (i.e., alfalfa, fallow fields, dryland pasture) were preferred by Swainson's hawks and ranked highest in foraging use; grain crops and late-harvested row crops that had relatively small prey populations, and that were high in vegetative cover were less suitable as foraging habitat. The predominance of grain crops and row crops in my study area, combined with the large distances Swainson's hawks had to travel from nest sites to reach more compatible cover types, may explain the relatively large home ranges exhibited by the Swainson's hawks in this study. The presence of urban and residential areas to the north and east likely account for the somewhat linear nature of the home ranges in this study.

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Staff Report on Burrowing Owl Mitigation

State of California

Natural Resources Agency

Department of Fish and Game

March 7, 2012[1]

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[1] This document replaces the Department of Fish and Game 1995 Staff Report On Burrowing Owl Mitigation.

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INTRODUCTION AND PURPOSE

Maintaining California's rich biological diversity is dependent on the conservation of species and their habitats. The California Department of Fish and Game (Department) has designated certain species as "species of special concern" when their population viability and survival is adversely affected by risk factors such as precipitous declines or other vulnerability factors (Shuford and Gardali 2008). Preliminary analyses of regional patterns for breeding populations of burrowing owls (*Athene cunicularia*) have detected declines both locally in their central and southern coastal breeding areas, and statewide where the species has experienced modest breeding range retraction (Gervais et al. 2008). In California, threat factors affecting burrowing owl populations include habitat loss, degradation and modification, and eradication of ground squirrels resulting in a loss of suitable burrows required by burrowing owls for nesting, protection from predators, and shelter (See Appendix A).

The Department recognized the need for a comprehensive conservation and mitigation strategy for burrowing owls, and in 1995 directed staff to prepare a report describing mitigation and survey recommendations. This report, "1995 Staff Report on Burrowing Owl Mitigation," (Staff Report) (CDFG 1995), contained Department-recommended burrowing owl and burrow survey techniques and mitigation measures intended to offset the loss of habitat and slow or reverse further decline of this species. Notwithstanding these measures, over the past 15+ years, burrowing owls have continued to decline in portions of their range (DeSante et al. 2007, Wilkerson and Siegel, 2010). The Department has determined that reversing declining population and range trends for burrowing owls will require implementation of more effective conservation actions, and evaluating the efficacy of the Department's existing recommended avoidance, minimization and mitigation approaches for burrowing owls.

The Department has identified three main actions that together will facilitate a more viable, coordinated, and concerted approach to conservation and mitigation for burrowing owls in California. These include:

1. Incorporating burrowing owl comprehensive conservation strategies into landscape-based planning efforts such as Natural Community Conservation Plans (NCCPs) and multi-species Habitat Conservation Plans (HCPs) that specifically address burrowing owls.
2. Developing and implementing a statewide conservation strategy (Burkett and Johnson, 2007) and local or regional conservation strategies for burrowing owls, including the development and implementation of a statewide burrowing owl survey and monitoring plan.
3. Developing more rigorous burrowing owl survey methods, working to improve the adequacy of impacts assessments; developing clear and effective avoidance and minimization measures; and developing mitigation measures to ensure impacts to the species are effectively addressed at the project, local, and/or regional level (the focus of this document).

This Report sets forth the Department's recommendations for implementing the third approach identified above by revising the 1995 Staff Report, drawing from the most relevant and current knowledge and expertise, and incorporating the best scientific information available pertaining to the species. It is designed to provide a compilation of the best available science for Department staff, biologists, planners, land managers, California Environmental Quality Act (CEQA) lead agencies, and the public to consider when assessing impacts of projects or other activities on burrowing owls.

This revised Staff Report takes into account the California Burrowing Owl Consortium's Survey Protocol and Mitigation Guidelines (CBOC 1993, 1997) and supersedes the survey, avoidance, minimization and mitigation recommendations in the 1995 Staff Report. Based on experiences gained from implementing the 1995 Staff Report, the Department believes revising that report is warranted. This document also includes general conservation goals and principles for developing mitigation measures for burrowing owls.

DEPARTMENT ROLE AND LEGAL AUTHORITIES

The mission of the Department is to manage California's diverse fish, wildlife and plant resources, and the habitats upon which they depend, for their ecological values and for their use and enjoyment by the public. The Department has jurisdiction over the conservation, protection, and management of fish, wildlife, native plants, and habitats necessary to maintain biologically sustainable populations of those species (Fish and Game Code (FGC) §1802). The Department, as trustee agency pursuant to CEQA (See CEQA Guidelines, §15386), has jurisdiction by law over natural resources, including fish and wildlife, affected by a project, as that term is defined in Section 21065 of the Public Resources Code. The Department exercises this authority by reviewing and commenting on environmental documents and making recommendations to avoid, minimize, and mitigate potential negative impacts to those resources held in trust for the people of California.

Field surveys designed to detect the presence of a particular species, habitat element, or natural community are one of the tools that can assist biologists in determining whether a species or habitat may be significantly impacted by land use changes or disturbance. The Department reviews field survey data as well as site-specific and regional information to evaluate whether a project's impacts may be significant. This document compiles the best available science for conducting habitat assessments and surveys, and includes considerations for developing measures to avoid impacts or mitigate unavoidable impacts.

CEQA

CEQA requires public agencies in California to analyze and disclose potential environmental impacts associated with a project that the agency will carry out, fund, or approve. Any potentially significant impact must be mitigated to the extent feasible. Project-specific CEQA mitigation is important for burrowing owls because most populations exist on privately owned parcels that, when proposed for development or other types of modification, may be subject to the environmental review requirements of CEQA.

Take

Take of individual burrowing owls and their nests is defined by FGC section 86, and prohibited by sections 3503, 3503.5 and 3513. Take is defined in FGC Section 86 as "hunt, pursue, catch, capture or kill, or attempt to hunt, pursue, catch, capture or kill."

Migratory Bird Treaty Act

The Migratory Bird Treaty Act (MBTA) implements various treaties and conventions between the United States and Canada, Japan, Mexico, and Russia for the protection of migratory birds, including the burrowing owl (50 C.F.R. § 10). The MBTA protects migratory bird nests from possession, sale, purchase, barter, transport, import and export, and collection. The other prohibitions of the MBTA - capture, pursue, hunt, and kill - are inapplicable to nests. The regulatory definition of take, as defined in Title 50 C.F.R. part 10.12, means to pursue, hunt, shoot, wound, kill, trap, capture, or collect, or attempt to hunt, shoot, wound, kill, trap, capture, or collect. Only the verb "collect" applies to nests. It is illegal to collect, possess, and by any means transfer possession of any migratory bird nest. The MBTA prohibits the destruction of a nest when it contains birds or eggs, and no possession shall occur during the destruction (see Fish and Wildlife Service, Migratory Bird Permit Memorandum, April 15, 2003). Certain exceptions to this prohibition are included in 50 C.F.R. section 21. Pursuant to Fish & Game Code section 3513, the Department enforces the Migratory Bird Treaty Act consistent with rules and regulations adopted by the Secretary of the Interior under provisions of the Migratory Treaty Act.

Regional Conservation Plans

Regional multiple species conservation plans offer long-term assurances for conservation of covered species at a landscape scale, in exchange for biologically appropriate levels of incidental take and/or habitat loss as defined in the approved plan. California's NCCP Act (FGC §2800 et seq.) governs such plans at the state level, and was designed to conserve species, natural communities, ecosystems, and ecological processes across a jurisdiction or a collection of jurisdictions. Complementary federal HCPs are governed by the Endangered Species Act (7 U.S.C. § 136, 16 U.S.C.§ 1531 et seq.) (ESA). Regional conservation plans (and certain other landscape-level conservation and management plans), may provide conservation for unlisted as well as listed species. Because the geographic scope of NCCPs and HCPs may span many hundreds of thousands of acres, these planning tools have the potential to play a significant role in conservation of burrowing owls, and grasslands and other habitats.

Fish and Game Commission Policies

There are a number of Fish and Game Commission policies (see FGC §2008) that can be applied to burrowing owl conservation. These include policies on: Raptors, Cooperation, Endangered and Threatened Species, Land Use Planning, Management and Utilization of Fish and Wildlife on Federal Lands, Management and Utilization of Fish and Wildlife on Private Lands, and Research.

GUIDING PRINCIPLES FOR CONSERVATION

Unless otherwise provided in a statewide, local, or regional conservation strategy, surveying and evaluating impacts to burrowing owls, as well as developing and implementing avoidance, minimization, and mitigation and conservation measures incorporate the following principles. These principles are a summary of Department staff expert opinion and were used to guide the preparation of this document.

1. Use the Precautionary Principle (Noss et al.1997), by which the alternative of increased conservation is deliberately chosen in order to buffer against incomplete knowledge of burrowing owl ecology and uncertainty about the consequences to burrowing owls of potential impacts, including those that are cumulative.
2. Employ basic conservation biology tenets and population-level approaches when determining what constitutes appropriate avoidance, minimization, and mitigation for impacts. Include mitigation effectiveness monitoring and reporting, and use an adaptive management loop to modify measures based on results.
3. Protect and conserve owls in wild, semi-natural, and agricultural habitats (conserve is defined at FGC §1802).
4. Protect and conserve natural nest burrows (or burrow surrogates) previously used by burrowing owls and sufficient foraging habitat and protect auxiliary "satellite" burrows that contribute to burrowing owl survivorship and natural behavior of owls.

CONSERVATION GOALS FOR THE BURROWING OWL IN CALIFORNIA

It is Department staff expert opinion that the following goals guide and contribute to the short and long-term conservation of burrowing owls in California:

1. Maintain size and distribution of extant burrowing owl populations (allowing for natural population fluctuations).
2. Increase geographic distribution of burrowing owls into formerly occupied historical range where burrowing owl habitat still exists, or where it can be created or enhanced, and where the reason for its local disappearance is no longer of concern.
3. Increase size of existing populations where possible and appropriate (for example, considering basic ecological principles such as carrying capacity, predator-prey relationships, and inter-specific relationships with other species at risk).
4. Protect and restore self-sustaining ecosystems or natural communities which can support burrowing owls at a landscape scale, and which will require minimal long-term management.
5. Minimize or prevent unnatural causes of burrowing owl population declines (e.g., nest burrow destruction, chemical control of rodent hosts and prey).
6. Augment/restore natural dynamics of burrowing owl populations including movement and genetic exchange among populations, such that the species does not require future listing and protection under the California Endangered Species Act (CESA) and/or the federal Endangered Species Act (ESA).
7. Engage stakeholders, including ranchers; farmers; military; tribes; local, state, and federal agencies; non-governmental organizations; and scientific research and education communities involved in burrowing owl protection and habitat management.

ACTIVITIES WITH THE POTENTIAL TO TAKE OR IMPACT BURROWING OWLS

The following activities are examples of activities that have the potential to take burrowing owls, their nests or eggs, or destroy or degrade burrowing owl habitat: grading, disking, cultivation, earthmoving, burrow blockage, heavy equipment compacting and crushing burrow tunnels, levee maintenance, flooding, burning and mowing (if burrows are impacted), and operating wind turbine collisions (collectively hereafter referred to as "projects" or "activities" whether carried out pursuant to CEQA or not). In addition, the following activities may have impacts to burrowing owl populations: eradication of host burrowers; changes in vegetation management (i.e. grazing); use of pesticides and rodenticides; destruction, conversion or degradation of nesting, foraging, over-wintering or other habitats; destruction of natural burrows and burrow surrogates; and disturbance which may result in harassment of owls at occupied burrows.

PROJECT IMPACT EVALUATIONS

The following three progressive steps are effective in evaluating whether projects will result in impacts to burrowing owls. The information gained from these steps will inform any subsequent avoidance, minimization and mitigation measures. The steps for project impact evaluations are: 1) habitat assessment, 2) surveys, and 3) impact assessment. Habitat assessments are conducted to evaluate the likelihood that a site supports burrowing owl. Burrowing owl surveys provide information needed to determine the potential effects of proposed projects and activities on burrowing owls, and to avoid take in accordance with FGC sections 86, 3503, and 3503.5. Impact assessments evaluate the extent to which burrowing owls and their habitat may be impacted, directly or indirectly, on and within a reasonable distance of a proposed CEQA project activity or non-CEQA project. These three site evaluation steps are discussed in detail below.

Biologist Qualifications

The current scientific literature indicates that only individuals meeting the following minimum qualifications should perform burrowing owl habitat assessments, surveys, and impact assessments:

1. Familiarity with the species and its local ecology;
2. Experience conducting habitat assessments and non-breeding and breeding season surveys, or experience with these surveys conducted under the direction of an experienced surveyor;
3. Familiarity with the appropriate state and federal statutes related to burrowing owls, scientific research, and conservation;
4. Experience with analyzing impacts of development on burrowing owls and their habitat.

Habitat Assessment Data Collection and Reporting

A habitat assessment is the first step in the evaluation process and will assist investigators in determining whether or not occupancy surveys are needed. Refer to Appendix B for a definition of burrowing owl habitat. Compile the detailed information described in Appendix C when conducting project scoping, conducting a habitat assessment site visit and preparing a habitat assessment report.

Surveys

Burrowing owl surveys are the second step of the evaluation process and the best available scientific literature recommends that they be conducted whenever burrowing owl habitat or sign (see Appendix B) is encountered on or adjacent to (within 150 meters) a project site

(Thomsen 1971, Martin 1973). Occupancy of burrowing owl habitat is confirmed at a site when at least one burrowing owl, or its sign at or near a burrow entrance, is observed within the last three years (Rich 1984). Burrowing owls are more detectable during the breeding season with detection probabilities being highest during the nestling stage (Conway et al. 2008). In California, the burrowing owl breeding season extends from 1 February to 31 August (Haug et al. 1993, Thompsen 1971) with some variances by geographic location and climatic conditions. Several researchers suggest three or more survey visits during daylight hours (Haug and Diduik 1993, CBOC 1997, Conway and Simon 2003) and recommend each visit occur at least three weeks apart during the peak of the breeding season, commonly accepted in California as between 15 April and 15 July (CBOC 1997). Conway and Simon (2003) and Conway et al. (2008) recommended conducting surveys during the day when most burrowing owls in a local area are in the laying and incubation period (so as not to miss early breeding attempts), during the nesting period, and in the late nestling period when most owls are spending time above ground.

Non-breeding season (1 September to 31 January) surveys may provide information on burrowing owl occupancy, but do not substitute for breeding season surveys because results are typically inconclusive. Burrowing owls are more difficult to detect during the non-breeding season and their seasonal residency status is difficult to ascertain. Burrowing owls detected during non-breeding season surveys may be year-round residents, young from the previous breeding season, pre-breeding territorial adults, winter residents, dispersing juveniles, migrants, transients or new colonizers. In addition, the numbers of owls and their pattern of distribution may differ during winter and breeding seasons. However, on rare occasions, non-breeding season surveys may be warranted (i.e., if the site is believed to be a wintering site only based on negative breeding season results). Refer to Appendix D for information on breeding season and non-breeding season survey methodologies.

Survey Reports

Adequate information about burrowing owls present in and adjacent to an area that will be disturbed by a project or activity will enable the Department, reviewing agencies and the public to effectively assess potential impacts and will guide the development of avoidance, minimization, and mitigation measures. The survey report includes but is not limited to a description of the proposed project or proposed activity, including the proposed project start and end dates, as well as a description of disturbances or other activities occurring on-site or nearby. Refer to Appendix D for details included in a survey report.

Impact Assessment

The third step in the evaluation process is the impact assessment. When surveys confirm occupied burrowing owl habitat in or adjoining the project area, there are a number of ways to assess a project’s potential significant impacts to burrowing owls and their habitat. Richardson and Miller (1997) recommended monitoring raptor behavior prior to developing management recommendations and buffers to determine the extent to which individuals have been sensitized to human disturbance. Monitoring results will also provide detail necessary for developing site-specific measures. Postovit and Postovit (1987) recommended an analytical approach to mitigation planning: define the problem (impact), set goals (to guide mitigation development), evaluate and select mitigation methods, and monitor the results.

Define the problem. The impact assessment evaluates all factors that could affect burrowing owls. Postovit and Postovit (1987) recommend evaluating the following in assessing impacts to raptors and planning mitigation: type and extent of disturbance, duration and timing of disturbance, visibility of disturbance, sensitivity and ability to habituate, and influence of environmental factors. They suggest identifying and addressing all potential direct and indirect impacts to burrowing owls, regardless of whether or not the impacts will occur during the breeding season. Several examples are given for each impact category below; however, examples are not intended to be used exclusively.

Type and extent of the disturbance. The impact assessment describes the nature (source) and extent (scale) of potential project impacts on occupied, satellite and unoccupied burrows including acreage to be lost (temporary or permanent), fragmentation/edge being created, increased distance to other nesting and foraging habitat, and habitat degradation. Discuss any project activities that impact either breeding and/or non-breeding habitat which could affect owl home range size and spatial configuration, negatively affect onsite and offsite burrowing owl presence, increase energetic costs, lower reproductive success, increase vulnerability to predation, and/or decrease the chance of procuring a mate.

Duration and timing of the impact. The impact assessment describes the amount of time the burrowing owl habitat will be unavailable to burrowing owls (temporary or permanent) on the site and the effect of that loss on essential behaviors or life history requirements of burrowing owls, the overlap of project activities with breeding and/or non-breeding seasons (timing of nesting and/or non-breeding activities may vary with latitude and climatic conditions, which should be considered with the timeline of the project or activity), and any variance of the project activities in intensity, scale and proximity relative to burrowing owl occurrences.

Visibility and sensitivity. Some individual burrowing owls or pairs are more sensitive than others to specific stimuli and may habituate to ongoing visual or audible disturbance. Site-specific monitoring may provide clues to the burrowing owl's sensitivities. This type of assessment addresses the sensitivity of burrowing owls within their nesting area to humans on foot, and vehicular traffic. Other variables are whether the site is primarily in a rural versus urban setting, and whether any prior disturbance (e.g., human development or recreation) is known at the site.

Environmental factors. The impact assessment discusses any environmental factors that could be influenced or changed by the proposed activities including nest site availability, predators, prey availability, burrowing mammal presence and abundance, and threats from other extrinsic factors such as human disturbance, urban interface, feral animals, invasive species, disease or pesticides.

Significance of impacts. The impact assessment evaluates the potential loss of nesting burrows, satellite burrows, foraging habitat, dispersal and migration habitat, wintering habitat, and habitat linkages, including habitat supporting prey and host burrowers and other essential habitat attributes. This assessment determines if impacts to the species will result in significant impacts to the species locally, regionally and range-wide per CEQA Guidelines §15382 and Appendix G. The significance of the impact to habitat depends on the extent of habitat disturbed and length of time the habitat is unavailable (for example: minor – several days, medium – several weeks to months, high - breeding season affecting juvenile survival, or over winter affecting adult survival).

Cumulative effects. The cumulative effects assessment evaluates two consequences: 1) the project's proportional share of reasonably foreseeable impacts on burrowing owls and habitat caused by the project or in combination with other projects and local influences having impacts on burrowing owls and habitat, and 2) the effects on the regional owl population resulting from the project's impacts to burrowing owls and habitat.

Mitigation goals. Establishing goals will assist in planning mitigation and selecting measures that function at a desired level. Goals also provide a standard by which to measure mitigation success. Unless specifically provided for through other FGC Sections or through specific regulations, take, possession or destruction of individual burrowing owls, their nests and eggs is prohibited under FGC sections 3503, 3503.5 and 3513. Therefore, a required goal for all project activities is to avoid take of burrowing owls. Under CEQA, goals would consist of measures that would avoid, minimize and mitigate impacts to a less than significant level. For individual projects, mitigation must be roughly proportional to the level of impacts, including cumulative impacts, in accordance with the provisions of CEQA (CEQA Guidelines, §§ 15126.4(a)(4)(B), 15064, 15065, and 16355). In order for mitigation measures to be effective, they must be specific, enforceable, and feasible actions that will improve environmental conditions. As set forth in more detail in Appendix A, the current scientific literature supports the conclusion that mitigation for permanent habitat loss necessitates replacement with an equivalent or greater habitat area for breeding, foraging, wintering, dispersal, presence of burrows, burrow surrogates, presence of fossorial mammal dens, well drained soils, and abundant and available prey within close proximity to the burrow.

MITIGATION METHODS

The current scientific literature indicates that any site-specific avoidance or mitigation measures developed should incorporate the best practices presented below or other practices confirmed by experts and the Department. The Department is available to assist in the development of site-specific avoidance and mitigation measures.

Avoiding. A primary goal is to design and implement projects to seasonally and spatially avoid negative impacts and disturbances that could result in take of burrowing owls, nests, or eggs. Other avoidance measures may include but not be limited to:

- Avoid disturbing occupied burrows during the nesting period, from 1 February through 31 August.
- Avoid impacting burrows occupied during the non-breeding season by migratory or non-migratory resident burrowing owls.
- Avoid direct destruction of burrows through chaining (dragging a heavy chain over an area to remove shrubs), disking, cultivation, and urban, industrial, or agricultural development.
- Develop and implement a worker awareness program to increase the on-site worker's recognition of and commitment to burrowing owl protection.
- Place visible markers near burrows to ensure that farm equipment and other machinery does not collapse burrows.
- Do not fumigate, use treated bait or other means of poisoning nuisance animals in areas where burrowing owls are known or suspected to occur (e.g., sites observed with nesting

owls, designated use areas).

- Restrict the use of treated grain to poison mammals to the months of January and February.

Take avoidance (pre-construction) surveys. Take avoidance surveys are intended to detect the presence of burrowing owls on a project site at a fixed period in time and inform necessary take avoidance actions. Take avoidance surveys may detect changes in owl presence such as colonizing owls that have recently moved onto the site, migrating owls, resident burrowing owls changing burrow use, or young of the year that are still present and have not dispersed. Refer to Appendix D for take avoidance survey methodology.

Site surveillance. Burrowing owls may attempt to colonize or re-colonize an area that will be impacted; thus, the current scientific literature indicates a need for ongoing surveillance at the project site during project activities is recommended. The surveillance frequency/effort should be sufficient to detect burrowing owls if they return. Subsequent to their new occupancy or return to the site, take avoidance measures should assure with a high degree of certainty that take of owls will not occur.

Minimizing. If burrowing owls and their habitat can be protected in place on or adjacent to a project site, the use of buffer zones, visual screens or other measures while project activities are occurring can minimize disturbance impacts. Conduct site-specific monitoring to inform development of buffers (see Visibility and sensitivity above). The following general guidelines for implementing buffers should be adjusted to address site-specific conditions using the impact assessment approach described above. The CEQA lead agency and/or project proponent is encouraged to consult with the Department and other burrowing owl experts for assistance in developing site-specific buffer zones and visual screens.

Buffers. Holroyd et al. (2001) identified a need to standardize management and disturbance mitigation guidelines. For instance, guidelines for mitigating impacts by petroleum industries on burrowing owls and other prairie species (Scobie and Faminow, 2000) may be used as a template for future mitigation guidelines (Holroyd et al. 2001). Scobie and Faminow (2000) developed guidelines for activities around occupied burrowing owl nests recommending buffers around low, medium, and high disturbance activities, respectively (see below).

Recommended restricted activity dates and setback distances by level of disturbance for burrowing owls (Scobie and Faminow 2000).

Location	Time of Year	Level of Disturbance		
		Low	Med	High
Nesting sites	April 1-Aug 15	200 m*	500 m	500 m
Nesting sites	Aug 16-Oct 15	200 m	200 m	500 m
Nesting sites	Oct 16-Mar 31	50 m	100 m	500 m

* meters (m)

Based on existing vegetation, human development, and land uses in an area, resource managers may decide to allow human development or resource extraction closer to these area/sites than recommended above. However, if it is decided to allow activities closer than the setback distances recommended, a broad-scale, long-term, scientifically-rigorous monitoring program ensures that burrowing owls are not detrimentally affected by alternative approaches.

Other minimization measures include eliminating actions that reduce burrowing owl forage and burrowing surrogates (e.g. ground squirrel), or introduce/facilitate burrowing owl predators. Actions that could influence these factors include reducing livestock grazing rates and/or changing the timing or duration of grazing or vegetation management that could result in less suitable habitat.

Burrow exclusion and closure. Burrow exclusion is a technique of installing one-way doors in burrow openings during the non-breeding season to temporarily exclude burrowing owls, or permanently exclude burrowing owls and close burrows after verifying burrows are empty by site monitoring and scoping. Exclusion in and of itself is not a take avoidance, minimization or mitigation method. Eviction of burrowing owls is a potentially significant impact under CEQA.

The long-term demographic consequences of these techniques have not been thoroughly evaluated, and the fate of evicted or excluded burrowing owls has not been systematically studied. Because burrowing owls are dependent on burrows at all times of the year for survival and/or reproduction, evicting them from nesting, roosting, and satellite burrows may lead to indirect impacts or take. Temporary or permanent closure of burrows may result in significant loss of burrows and habitat for reproduction and other life history requirements. Depending on the proximity and availability of alternate habitat, loss of access to burrows will likely result in varying levels of increased stress on burrowing owls and could depress reproduction, increase predation, increase energetic costs, and introduce risks posed by having to find and compete for available burrows. Therefore, exclusion and burrow closure are not recommended where they can be avoided. The current scientific literature indicates consideration of all possible avoidance and minimization measures before temporary or permanent exclusion and closure of burrows is implemented, in order to avoid take.

The results of a study by Trulio (1995) in California showed that burrowing owls passively displaced from their burrows were quickly attracted to adjacent artificial burrows at five of six passive relocation sites. The successful sites were all within 75 meters (m) of the destroyed burrow, a distance generally within a pair's territory. This researcher discouraged using passive relocation to artificial burrows as a mitigation measure for lost burrows without protection of adjacent foraging habitat. The study results indicated artificial burrows were used by evicted burrowing owls when they were approximately 50-100 m from the natural burrow (Thomsen 1971, Haug and Oliphant 1990). Locating artificial or natural burrows more than 100 m from the eviction burrow may greatly reduce the chances that new burrows will be used. Ideally, exclusion and burrow closure is employed only where there are adjacent natural burrows and non-impacted, sufficient habitat for burrowing owls to occupy with permanent protection mechanisms in place. Any new burrowing owl colonizing the project site after the CEQA document has been adopted may constitute changed circumstances that should be addressed in a re-circulated CEQA document.

The current scientific literature indicates that burrow exclusion should only be conducted by qualified biologists (meeting the Biologist’s Qualifications above) during the non-breeding season, before breeding behavior is exhibited and after the burrow is confirmed empty by site surveillance and/or scoping. The literature also indicates that when temporary or permanent burrow exclusion and/or burrow closure is implemented, burrowing owls should not be excluded from burrows unless or until:

- A Burrowing Owl Exclusion Plan (see Appendix E) is developed and approved by the applicable local DFG office;
- Permanent loss of occupied burrow(s) and habitat is mitigated in accordance with the Mitigating Impacts sections below. Temporary exclusion is mitigated in accordance with the item #1 under Mitigating Impacts below.
- Site monitoring is conducted prior to, during, and after exclusion of burrowing owls from their burrows sufficient to ensure take is avoided. Conduct daily monitoring for one week to confirm young of the year have fledged if the exclusion will occur immediately after the end of the breeding season.
- Excluded burrowing owls are documented using artificial or natural burrows on an adjoining mitigation site (if able to confirm by band re-sight).

Translocation (Active relocation offsite >100 meters). At this time, there is little published information regarding the efficacy of translocating burrowing owls, and additional research is needed to determine subsequent survival and breeding success (Klute et al. 2003, Holroyd et al. 2001). Study results for translocation in Florida implied that hatching success may be decreased for populations of burrowing owls that undergo translocation (Nixon 2006). At this time, the Department is unable to authorize the capture and relocation of burrowing owls except within the context of scientific research (FGC §1002) or a NCCP conservation strategy.

Mitigating impacts. Habitat loss and degradation from rapid urbanization of farmland in the core areas of the Central and Imperial valleys is the greatest of many threats to burrowing owls in California (Shuford and Gardali, 2008). At a minimum, if burrowing owls have been documented to occupy burrows (see Definitions, Appendix B) at the project site in recent years, the current scientific literature supports the conclusion that the site should be considered occupied and mitigation should be required by the CEQA lead agency to address project-specific significant and cumulative impacts. Other site-specific and regionally significant and cumulative impacts may warrant mitigation. The current scientific literature indicates the following to be best practices. If these best practices cannot be implemented, the lead agency or lead investigator may consult with the Department to develop effective mitigation alternatives. The Department is also available to assist in the identification of suitable mitigation lands.

1. Where habitat will be temporarily disturbed, restore the disturbed area to pre-project condition including decompacting soil and revegetating. Permanent habitat protection may be warranted if there is the potential that the temporary impacts may render a nesting site (nesting burrow and satellite burrows) unsustainable or unavailable depending on the time frame, resulting in reduced survival or abandonment. For the latter potential impact, see the permanent impact measures below.
2. Mitigate for permanent impacts to nesting, occupied and satellite burrows and/or burrowing owl habitat such that the habitat acreage, number of burrows and burrowing owls impacted are replaced based on the information provided in Appendix A. Note: A

minimum habitat replacement recommendation is not provided here as it has been shown to serve as a default, replacing any site-specific analysis and discounting the wide variation in natal area, home range, foraging area, and other factors influencing burrowing owls and burrowing owl population persistence in a particular area.

3. Mitigate for permanent impacts to nesting, occupied and satellite burrows and burrowing owl habitat with (a) permanent conservation of similar vegetation communities (grassland, scrublands, desert, urban, and agriculture) to provide for burrowing owl nesting, foraging, wintering, and dispersal (i.e., during breeding and non-breeding seasons) comparable to or better than that of the impact area, and (b) sufficiently large acreage, and presence of fossorial mammals. The mitigation lands may require habitat enhancements including enhancement or expansion of burrows for breeding, shelter and dispersal opportunity, and removal or control of population stressors. If the mitigation lands are located adjacent to the impacted burrow site, ensure the nearest neighbor artificial or natural burrow clusters are at least within 210 meters (Fisher et al. 2007).
4. Permanently protect mitigation land through a conservation easement deeded to a non-profit conservation organization or public agency with a conservation mission, for the purpose of conserving burrowing owl habitat and prohibiting activities incompatible with burrowing owl use. If the project is located within the service area of a Department-approved burrowing owl conservation bank, the project proponent may purchase available burrowing owl conservation bank credits.
5. Develop and implement a mitigation land management plan to address long-term ecological sustainability and maintenance of the site for burrowing owls (see Management Plan and Artificial Burrow sections below, if applicable).
6. Fund the maintenance and management of mitigation land through the establishment of a long-term funding mechanism such as an endowment.
7. Habitat should not be altered or destroyed, and burrowing owls should not be excluded from burrows, until mitigation lands have been legally secured, are managed for the benefit of burrowing owls according to Department-approved management, monitoring and reporting plans, and the endowment or other long-term funding mechanism is in place or security is provided until these measures are completed.
8. Mitigation lands should be on, adjacent or proximate to the impact site where possible and where habitat is sufficient to support burrowing owls present.
9. Where there is insufficient habitat on, adjacent to, or near project sites where burrowing owls will be excluded, acquire mitigation lands with burrowing owl habitat away from the project site. The selection of mitigation lands should then focus on consolidating and enlarging conservation areas located outside of urban and planned growth areas, within foraging distance of other conserved lands. If mitigation lands are not available adjacent to other conserved lands, increase the mitigation land acreage requirement to ensure a selected site is of sufficient size. Offsite mitigation may not adequately offset the biological and habitat values impacted on a one to one basis. Consult with the Department when determining offsite mitigation acreages.
10. Evaluate and select suitable mitigation lands based on a comparison of the habitat attributes of the impacted and conserved lands, including but not limited to: type and structure of habitat being impacted or conserved; density of burrowing owls in impacted and conserved habitat; and significance of impacted or conserved habitat to the species range-wide. Mitigate for the highest quality burrowing owl habitat impacted first and foremost when identifying mitigation lands, even if a mitigation site is located outside of

a lead agency's jurisdictional boundary, particularly if the lead agency is a city or special district.

11. Select mitigation lands taking into account the potential human and wildlife conflicts or incompatibility, including but not limited to, human foot and vehicle traffic, and predation by cats, loose dogs and urban-adapted wildlife, and incompatible species management (i.e., snowy plover).
12. Where a burrowing owl population appears to be highly adapted to heavily altered habitats such as golf courses, airports, athletic fields, and business complexes, permanently protecting the land, augmenting the site with artificial burrows, and enhancing and maintaining those areas may enhance sustainability of the burrowing owl population onsite. Maintenance includes keeping lands grazed or mowed with weed-eaters or push mowers, free from trees and shrubs, and preventing excessive human and human-related disturbance (e.g., walking, jogging, off-road activity, dog-walking) and loose and feral pets (chasing and, presumably, preying upon owls) that make the environment uninhabitable for burrowing owls (Wesemann and Rowe 1985, Millsap and Bear 2000, Lincer and Bloom 2007). Items 4, 5 and 6 also still apply to this mitigation approach.
13. If there are no other feasible mitigation options available and a lead agency is willing to establish and oversee a Burrowing Owl Mitigation and Conservation Fund that funds on a competitive basis acquisition and permanent habitat conservation, the project proponent may participate in the lead agency's program.

Artificial burrows. Artificial burrows have been used to replace natural burrows either temporarily or long-term and their long-term success is unclear. Artificial burrows may be an effective addition to in-perpetuity habitat mitigation if they are augmenting natural burrows, the burrows are regularly maintained (i.e., no less than annual, with biennial maintenance recommended), and surrounding habitat patches are carefully maintained. There may be some circumstances, for example at airports, where squirrels will not be allowed to persist and create a dynamic burrow system, where artificial burrows may provide some support to an owl population.

Many variables may contribute to the successful use of artificial burrows by burrowing owls, including pre-existence of burrowing owls in the area, availability of food, predators, surrounding vegetation and proximity, number of natural burrows in proximity, type of materials used to build the burrow, size of the burrow and entrance, direction in which the burrow entrance is facing, slope of the entrance, number of burrow entrances per burrow, depth of the burrow, type and height of perches, and annual maintenance needs (Belthoff and King 2002, Smith et al. 2005, Barclay et al. 2011). Refer to Barclay (2008) and (2011) and to Johnson et al. 2010 (unpublished report) for guidance on installing artificial burrows including recommendations for placement, installation and maintenance.

Any long-term reliance on artificial burrows as natural burrow replacements must include semi-annual to annual cleaning and maintenance and/or replacement (Barclay et al. 2011, Smith and Conway 2005, Alexander et al. 2005) as an ongoing management practice. Alexander et al. (2005), in a study of the use of artificial burrows found that all of 20 artificial burrows needed some annual cleaning and maintenance. Burrows were either excavated by predators, blocked by soil or vegetation, or experienced substrate erosion forming a space beneath the tubing that prevented nestlings from re-entering the burrow.

Mitigation lands management plan. Develop a Mitigation Lands Management Plan for projects that require off-site or on-site mitigation habitat protection to ensure compliance with and effectiveness of identified management actions for the mitigation lands. A suggested outline and related vegetation management goals and monitoring success criteria can be found in Appendix E.

Mitigation Monitoring and Reporting

Verify the compliance with required mitigation measures, the accuracy of predictions, and ensure the effectiveness of all mitigation measures for burrowing owls by conducting follow-up monitoring, and implementing midcourse corrections, if necessary, to protect burrowing owls. Refer to CEQA Guidelines Section 15097 and the CEQA Guidelines for additional guidance on mitigation, monitoring and reporting. Monitoring is qualitatively different from site surveillance; monitoring normally has a specific purpose and its outputs and outcomes will usually allow a comparison with some baseline condition of the site before the mitigation (including avoidance and minimization) was undertaken. Ideally, monitoring should be based on the Before-After Control-Impact (BACI) principle (McDonald et al. 2000) that requires knowledge of the pre-mitigation state to provide a reference point for the state and change in state after the project and mitigation have been implemented.

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Appendix A. Burrowing Owl Natural History and Threats

Diet

Burrowing owl diet includes arthropods, small rodents, birds, amphibians, reptiles, and carrion (Haug et al. 1993).

Breeding

In California, the breeding season for the burrowing owl typically occurs between 1 February and 31 August although breeding in December has been documented (Thompson 1971, Gervais et al. 2008); breeding behavior includes nest site selection by the male, pair formation, copulation, egg laying, hatching, fledging, and post-fledging care of young by the parents. The peak of the breeding season occurs between 15 April and 15 July and is the period when most burrowing owls have active nests (eggs or young). The incubation period lasts 29 days (Coulombe 1971) and young fledge after 44 days (Haug et al. 1993). Note that the timing of nesting activities may vary with latitude and climatic conditions. Burrowing owls may change burrows several times during the breeding season, starting when nestlings are about three weeks old (Haug et al. 1993).

Dispersal

The following discussion is an excerpt from Gervais et al (2008):

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

Habitat

The burrowing owl is a small, long-legged, ground-dwelling bird species, well-adapted to open, relatively flat expanses. In California, preferred habitat is generally typified by short, sparse vegetation with few shrubs, level to gentle topography and well-drained soils (Haug et al. 1993). Grassland, shrub steppe, and desert are naturally occurring habitat types used by the species. In addition, burrowing owls may occur in some agricultural areas, ruderal grassy fields, vacant lots and pastures if the vegetation structure is suitable and there are useable burrows and foraging habitat in proximity (Gervais et al 2008). Unique amongst North

American raptors, the burrowing owl requires underground burrows or other cavities for nesting during the breeding season and for roosting and cover, year round. Burrows used by the owls are usually dug by other species termed host burrowers. In California, California ground squirrel (*Spermophilus beecheyi*) and round-tailed ground squirrel (*Citellus tereticaudus*) burrows are frequently used by burrowing owls but they may use dens or holes dug by other fossorial species including badger (*Taxidea taxus*), coyote (*Canis latrans*), and fox (e.g., San Joaquin kit fox, *Vulpes macrotis mutica*; Ronan 2002). In some instances, owls have been known to excavate their own burrows (Thompson 1971, Barclay 2007). Natural rock cavities, debris piles, culverts, and pipes also are used for nesting and roosting (Rosenberg et al. 1998). Burrowing owls have been documented using artificial burrows for nesting and cover (Smith and Belthoff, 2003).

Foraging habitat. Foraging habitat is essential to burrowing owls. The following discussion is an excerpt from Gervais et al. (2008):

> “Useful as a rough guide to evaluating project impacts and appropriate mitigation for burrowing owls, adult male burrowing owls home ranges have been documented (calculated by minimum convex polygon) to comprise anywhere from 280 acres in intensively irrigated agroecosystems in Imperial Valley (Rosenberg and Haley 2004) to 450 acres in mixed agricultural lands at Lemoore Naval Air Station, CA (Gervais et al. 2003), to 600 acres in pasture in Saskatchewan, Canada (Haug and Oliphant 1990). But owl home ranges may be much larger, perhaps by an order of magnitude, in non-irrigated grasslands such as at Carrizo Plain, California (Gervais et al. 2008), based on telemetry studies and distribution of nests. Foraging occurs primarily within 600 m of their nests (within approximately 300 acres, based on a circle with a 600 m radius) during the breeding season.”

Importance of burrows and adjacent habitat. Burrows and the associated surrounding habitat are essential ecological requisites for burrowing owls throughout the year and especially during the breeding season. During the non-breeding season, burrowing owls remain closely associated with burrows, as they continue to use them as refuge from predators, shelter from weather and roost sites. Resident populations will remain near the previous season’s nest burrow at least some of the time (Coulombe 1971, Thomsen 1971, Botelho 1996, LaFever et al. 2008).

In a study by Lutz and Plumpton (1999) adult males and females nested in formerly used sites at similar rates (75% and 63%, respectively) (Lutz and Plumpton 1999). Burrow fidelity has been reported in some areas; however, more frequently, burrowing owls reuse traditional nesting areas without necessarily using the same burrow (Haug et al. 1993, Dechant et al. 1999). Burrow and nest sites are re-used at a higher rate if the burrowing owl has reproduced successfully during the previous year (Haug et al. 1993) and if the number of burrows isn’t limiting nesting opportunity.

Burrowing owls may use “satellite” or non-nesting burrows, moving young at 10-14 days, presumably to reduce risk of predation (Desmond and Savidge 1998) and possibly to avoid nest parasites (Dechant et al. 1999). Successful nests in Nebraska had more active satellite burrows within 75 m of the nest burrow than unsuccessful nests (Desmond and Savidge

1999). Several studies have documented the number of satellite burrows used by young and adult burrowing owls during the breeding season as between one and 11 burrows with an average use of approximately five burrows (Thompsen 1984, Haug 1985, Haug and Oliphant 1990). Supporting the notion of selecting for nest sites near potential satellite burrows, Ronan (2002) found burrowing owl families would move away from a nest site if their satellite burrows were experimentally removed through blocking their entrance.

Habitat adjacent to burrows has been documented to be important to burrowing owls. Gervais et al. (2003) found that home range sizes of male burrowing owls during the nesting season were highly variable within but not between years. Their results also suggested that owls concentrate foraging efforts within 600 meters of the nest burrow, as was observed in Canada (Haug and Oliphant 1990) and southern California (Rosenberg and Haley 2004). James et al. (1997), reported habitat modification factors causing local burrowing owl declines included habitat fragmentation and loss of connectivity.

In conclusion, the best available science indicates that essential habitat for the burrowing owl in California must include suitable year-round habitat, primarily for breeding, foraging, wintering and dispersal habitat consisting of short or sparse vegetation (at least at some time of year), presence of burrows, burrow surrogates or presence of fossorial mammal dens, well-drained soils, and abundant and available prey within close proximity to the burrow.

Threats to Burrowing Owls in California

Habitat loss. Habitat loss, degradation, and fragmentation are the greatest threats to burrowing owls in California. According to DeSante et al. (2007), “the vast majority of burrowing owls [now] occur in the wide, flat lowland valleys and basins of the Imperial Valley and Great Central Valley [where] for the most part,...the highest rates of residential and commercial development in California are occurring.” Habitat loss from the State’s long history of urbanization in coastal counties has already resulted in either extirpation or drastic reduction of burrowing owl populations there (Gervais et al. 2008). Further, loss of agricultural and other open lands (such as grazed landscapes) also negatively affect owl populations. Because of their need for open habitat with low vegetation, burrowing owls are unlikely to persist in agricultural lands dominated by vineyards and orchards (Gervais et al. 2008).

Control of burrowing rodents. According to Klute et al. (2003), the elimination of burrowing rodents through control programs is a primary factor in the recent and historical decline of burrowing owl populations nationwide. In California, ground squirrel burrows are most often used by burrowing owls for nesting and cover; thus, ground squirrel control programs may affect owl numbers in local areas by eliminating a necessary resource.

Direct mortality. Burrowing owls suffer direct losses from a number of sources. Vehicle collisions are a significant source of mortality especially in the urban interface and where owls nest alongside roads (Haug et al. 1993, Gervais et al. 2008). Road and ditch maintenance, modification of water conveyance structures (Imperial Valley) and discing to control weeds in fallow fields may destroy burrows (Rosenberg and Haley 2004, Catlin and Rosenberg 2006) which may trap or crush owls. Wind turbines at Altamont Pass Wind Resource Area are known to cause direct burrowing owl mortality (Thelander et al. 2003). Exposure to pesticides may pose a threat to the species but is poorly understood (Klute et al. 2003, Gervais et al. 2008).

Appendix B. Definitions

Some key terms that appear in this document are defined below.

Adjacent habitat means burrowing owl habitat that abuts the area where habitat and burrows will be impacted and rendered non-suitable for occupancy.

Breeding (nesting) season begins as early as 1 February and continues through 31 August (Thomsen 1971, Zarn 1974). The timing of breeding activities may vary with latitude and climatic conditions. The breeding season includes pairing, egg-laying and incubation, and nestling and fledging stages.

Burrow exclusion is a technique of installing one-way doors in burrow openings during the non-breeding season to temporarily exclude burrowing owls or permanently exclude burrowing owls and excavate and close burrows after confirming burrows are empty.

Burrowing owl habitat generally includes, but is not limited to, short or sparse vegetation (at least at some time of year), presence of burrows, burrow surrogates or presence of fossorial mammal dens, well-drained soils, and abundant and available prey.

Burrow surrogates include culverts, piles of concrete rubble, piles of soil, burrows created along soft banks of ditches and canals, pipes, and similar structures.

Civil twilight - Morning civil twilight begins when the geometric center of the sun is 6 degrees below the horizon (civil dawn) and ends at sunrise. Evening civil twilight begins at sunset and ends when the geometric center of the sun reaches 6 degrees below the horizon (civil dusk). During this period there is enough light from the sun that artificial sources of light may not be needed to carry on outdoor activities. This concept is sometimes enshrined in laws, for example, when drivers of automobiles must turn on their headlights (called lighting-up time in the UK); when pilots may exercise the rights to fly aircraft. Civil twilight can also be described as the limit at which twilight illumination is sufficient, under clear weather conditions, for terrestrial objects to be clearly distinguished; at the beginning of morning civil twilight, or end of evening civil twilight, the horizon is clearly defined and the brightest stars are visible under clear atmospheric conditions.

Conservation for burrowing owls may include but may not be limited to protecting remaining breeding pairs or providing for population expansion, protecting and enhancing breeding and essential habitat, and amending or augmenting land use plans to stabilize populations and other specific actions to avoid the need to list the species pursuant to California or federal Endangered Species Acts.

Contiguous means connected together so as to form an uninterrupted expanse in space.

Essential habitat includes nesting, foraging, wintering, and dispersal habitat.

Foraging habitat is habitat within the estimated home range of an occupied burrow, supports suitable prey base, and allows for effective hunting.

Host burrowers include ground squirrels, badgers, foxes, coyotes, gophers etc.

Locally significant species is a species that is not rare from a statewide perspective but is rare or uncommon in a local context such as within a county or region (CEQA §15125 (c)) or is so designated in local or regional plans, policies, or ordinances (CEQA Guidelines, Appendix G). Examples include a species at the outer limits of its known range or occurring in a unique habitat type.

Non-breeding season is the period of time when nesting activity is not occurring, generally September 1 through January 31, but may vary with latitude and climatic conditions.

Occupied site or occupancy means a site that is assumed occupied if at least one burrowing owl has been observed occupying a burrow within the last three years (Rich 1984). Occupancy of suitable burrowing owl habitat may also be indicated by owl sign including its molted feathers, cast pellets, prey remains, eggshell fragments, or excrement at or near a burrow entrance or perch site.

Other impacting activities may include but may not be limited to agricultural practices, vegetation management and fire control, pest management, conversion of habitat from rangeland or natural lands to more intensive agricultural uses that could result in "take". These impacting activities may not meet the definition of a project under CEQA.

Passive relocation is a technique of installing one-way doors in burrow openings to temporarily or permanently evict burrowing owls and prevent burrow re-occupation.

Peak of the breeding season is between 15 April and 15 July.

Sign includes its tracks, molted feathers, cast pellets (defined as 1-2" long brown to black regurgitated pellets consisting of non-digestible portions of the owls' diet, such as fur, bones, claws, beetle elytra, or feathers), prey remains, egg shell fragments, owl white wash, nest burrow decoration materials (e.g., paper, foil, plastic items, livestock or other animal manure, etc.), possible owl perches, or other items.

Appendix C. Habitat Assessment and Reporting Details

Habitat Assessment Data Collection and Reporting

Current scientific literature indicates that it would be most effective to gather the data in the manner described below when conducting project scoping, conducting a habitat assessment site visit and preparing a habitat assessment report:

1. Conduct at least one visit covering the entire potential project/activity area including areas that will be directly or indirectly impacted by the project. Survey adjoining areas within 150 m (Thomsen 1971, Martin 1973), or more where direct or indirect effects could potentially extend offsite. If lawful access cannot be achieved to adjacent areas, surveys can be performed with a spotting scope or other methods.
2. Prior to the site visit, compile relevant biological information for the site and surrounding area to provide a local and regional context.
3. Check all available sources for burrowing owl occurrence information regionally prior to a field inspection. The CNDDB and BIOS (see References cited) may be consulted for known occurrences of burrowing owls. Other sources of information include, but are not limited to, the Proceedings of the California Burrowing Owl Symposium (Barclay et al. 2007), county bird atlas projects, Breeding Bird Survey records, eBIRD (http://ebird.org), Gervais et al. (2008), local reports or experts, museum records, and other site-specific relevant information.
4. Identify vegetation and habitat types potentially supporting burrowing owls in the project area and vicinity.
5. Record and report on the following information:
   a. A full description of the proposed project, including but not limited to, expected work periods, daily work schedules, equipment used, activities performed (such as drilling, construction, excavation, etc.) and whether the expected activities will vary in location or intensity over the project's timeline;
   b. A regional setting map, showing the general project location relative to major roads and other recognizable features;
   c. A detailed map (preferably a USGS topo 7.5' quad base map) of the site and proposed project, including the footprint of proposed land and/or vegetation-altering activities, base map source, identifying topography, landscape features, a north arrow, bar scale, and legend;
   d. A written description of the biological setting, including location (Section, Township, Range, baseline and meridian), acreage, topography, soils, geographic and hydrologic characteristics, land use and management history on and adjoining the site (i.e., whether it is urban, semi-urban or rural; whether there is any evidence of past or current livestock grazing, mowing, disking, or other vegetation management activities);
   e. An analysis of any relevant, historical information concerning burrowing owl use or occupancy (breeding, foraging, over-wintering) on site or in the assessment area;
   f. Vegetation type and structure (using Sawyer et al. 2009), vegetation height, habitat types and features in the surrounding area plus a reasonably sized (as supported with logical justification) assessment area; (Note: use caution in discounting habitat based on grass height as it can be a temporary condition variable by season and conditions (such as current grazing regime) or may be distributed as a mosaic).

g. The presence of burrowing owl individuals or pairs or sign (see Appendix B);
h. The presence of suitable burrows and/or burrow surrogates (>11 cm in diameter (height and width) and >150 cm in depth) (Johnson et al. 2010), regardless of a lack of any burrowing owl sign and/or burrow surrogates; and burrowing owls and/or their sign that have recently or historically (within the last 3 years) been identified on or adjacent to the site.

Appendix D. Breeding and Non-breeding Season Surveys and Reports

Current scientific literature indicates that it is most effective to conduct breeding and non-breeding season surveys and report in the manner that follows:

Breeding Season Surveys

Number of visits and timing. Conduct 4 survey visits: 1) at least one site visit between 15 February and 15 April, and 2) a minimum of three survey visits, at least three weeks apart, between 15 April and 15 July, with at least one visit after 15 June. Note: many burrowing owl migrants are still present in southwestern California during mid-March, therefore, exercise caution in assuming breeding occupancy early in the breeding season.

Survey method. Rosenberg et al. (2007) confirmed walking line transects were most effective in smaller habitat patches. Conduct surveys in all portions of the project site that were identified in the Habitat Assessment and fit the description of habitat in Appendix A. Conduct surveys by walking straight-line transects spaced 7 m to 20 m apart, adjusting for vegetation height and density (Rosenberg et al. 2007). At the start of each transect and, at least, every 100 m, scan the entire visible project area for burrowing owls using binoculars. During walking surveys, record all potential burrows used by burrowing owls as determined by the presence of one or more burrowing owls, pellets, prey remains, whitewash, or decoration. Some burrowing owls may be detected by their calls, so observers should also listen for burrowing owls while conducting the survey.

Care should be taken to minimize disturbance near occupied burrows during all seasons and not to “flush” burrowing owls especially if predators are present to reduce any potential for needless energy expenditure or burrowing owl mortality. Burrowing owls may flush if approached by pedestrians within 50 m (Conway et al. 2003). If raptors or other predators are present that may suppress burrowing owl activity, return at another time or later date for a follow-up survey.

Check all burrowing owls detected for bands and/or color bands and report band combinations to the Bird Banding Laboratory (BBL). Some site-specific variations to survey methods discussed below may be developed in coordination with species experts and Department staff.

Weather conditions. Poor weather may affect the surveyor’s ability to detect burrowing owls, therefore, avoid conducting surveys when wind speed is >20 km/hr, and there is precipitation or dense fog. Surveys have greater detection probability if conducted when ambient temperatures are >20° C, <12 km/hr winds, and cloud cover is <75% (Conway et al. 2008).

Time of day. Daily timing of surveys varies according to the literature, latitude, and survey method. However, surveys between morning civil twilight and 10:00 AM and two hours before sunset until evening civil twilight provide the highest detection probabilities (Barclay pers. comm. 2012, Conway et al. 2008).

Alternate methods. If the project site is large enough to warrant an alternate method, consult current literature for generally accepted survey methods and consult with the Department on the proposed survey approach.

Additional breeding season site visits. Additional breeding season site visits may be necessary, especially if non-breeding season exclusion methods are contemplated. Detailed information, such as approximate home ranges of each individual or of family units, as well as foraging areas as related to the proposed project, will be important to document for evaluating impacts, planning avoidance measure implementation and for mitigation measure performance monitoring.

Adverse conditions may prevent investigators from determining presence or occupancy. Disease, predation, drought, high rainfall or site disturbance may preclude presence of burrowing owls in any given year. Any such conditions should be identified and discussed in the survey report. Visits to the site in more than one year may increase the likelihood of detection. Also, visits to adjacent known occupied habitat may help determine appropriate survey timing.

Given the high site fidelity shown by burrowing owls (see Appendix A, Importance of burrows), conducting surveys over several years may be necessary when project activities are ongoing, occur annually, or start and stop seasonally. (See Negative surveys).

Non-breeding Season Surveys

If conducting non-breeding season surveys, follow the methods described above for breeding season surveys, but conduct at least four (4) visits, spread evenly, throughout the non-breeding season. Burrowing owl experts and local Department staff are available to assist with interpreting results.

Negative Surveys

Adverse conditions may prevent investigators from documenting presence or occupancy. Disease, predation, drought, high rainfall or site disturbance may preclude presence of burrowing owl in any given year. Discuss such conditions in the Survey Report. Visits to the site in more than one year increase the likelihood of detection and failure to locate burrowing owls during one field season does not constitute evidence that the site is no longer occupied, particularly if adverse conditions influenced the survey results. Visits to other nearby known occupied sites can affirm whether the survey timing is appropriate.

Take Avoidance Surveys

Field experience from 1995 to present supports the conclusion that it would be effective to complete an initial take avoidance survey no less than 14 days prior to initiating ground disturbance activities using the recommended methods described in the Detection Surveys section above. Implementation of avoidance and minimization measures would be triggered by positive owl presence on the site where project activities will occur. The development of avoidance and minimization approaches would be informed by monitoring the burrowing owls.

Burrowing owls may re-colonize a site after only a few days. Time lapses between project activities trigger subsequent take avoidance surveys including but not limited to a final survey conducted within 24 hours prior to ground disturbance.

Survey Reports

Report on the survey methods used and results including the information described in the Summary Report and include the reports within the CEQA documentation:

1. Date, start and end time of surveys including weather conditions (ambient temperature, wind speed, percent cloud cover, precipitation and visibility);
2. Name(s) of surveyor(s) and qualifications;
3. A discussion of how the timing of the survey affected the comprehensiveness and detection probability;
4. A description of survey methods used including transect spacing, point count dispersal and duration, and any calls used;
5. A description and justification of the area surveyed relative to the project area;
6. A description that includes: number of owls or nesting pairs at each location (by nestlings, juveniles, adults, and those of an unknown age), number of burrows being used by owls, and burrowing owl sign at burrows. Include a description of individual markers, such as bands (numbers and colors), transmitters, or unique natural identifying features. If any owls are banded, request documentation from the BBL and bander to report on the details regarding the known history of the banded burrowing owl(s) (age, sex, origins, whether it was previously relocated) and provide with the report if available;
7. A description of the behavior of burrowing owls during the surveys, including feeding, resting, courtship, alarm, territorial defense, and those indicative of parents or juveniles;
8. A list of possible burrowing owl predators present and documentation of any evidence of predation of owls;
9. A detailed map (1:24,000 or closer to show details) showing locations of all burrowing owls, potential burrows, occupied burrows, areas of concentrated burrows, and burrowing owl sign. Locations documented by use of global positioning system (GPS) coordinates must include the datum in which they were collected. The map should include a title, north arrow, bar scale and legend;
10. Signed field forms, photos, etc., as appendices to the field survey report;
11. Recent color photographs of the proposed project or activity site; and
12. Original CNDDB Field Survey Forms should be sent directly to the Department's CNDDB office, and copies should be included in the environmental document as an appendix. (http://www.dfg.ca.gov/bdb/html/cnddb.html).

Appendix E. Example Components for Burrowing Owl Artificial Burrow and Exclusion Plans

Whereas the Department does not recommend exclusion and burrow closure, current scientific literature and experience from 1995 to present, indicate that the following example components for burrowing owl artificial burrow and exclusion plans, combined with consultation with the Department to further develop these plans, would be effective.

Artificial Burrow Location

If a burrow is confirmed occupied on-site, artificial burrow locations should be appropriately located and their use should be documented taking into consideration:

1. A brief description of the project and project site pre-construction;
2. The mitigation measures that will be implemented;
3. Potential conflicting site uses or encumbrances;
4. A comparison of the occupied burrow site(s) and the artificial burrow site(s) (e.g., vegetation, habitat types, fossorial species use in the area, and other features);
5. Artificial burrow(s) proximity to the project activities, roads and drainages;
6. Artificial burrow(s) proximity to other burrows and entrance exposure;
7. Photographs of the site of the occupied burrow(s) and the artificial burrows;
8. Map of the project area that identifies the burrow(s) to be excluded as well as the proposed sites for the artificial burrows;
9. A brief description of the artificial burrow design;
10. Description of the monitoring that will take place during and after project implementation including information that will be provided in a monitoring report.
11. A description of the frequency and type of burrow maintenance.

Exclusion Plan

An Exclusion Plan addresses the following including but not limited to:

1. Confirm by site surveillance that the burrow(s) is empty of burrowing owls and other species preceding burrow scoping;
2. Type of scope and appropriate timing of scoping to avoid impacts;
3. Occupancy factors to look for and what will guide determination of vacancy and excavation timing (one-way doors should be left in place 48 hours to ensure burrowing owls have left the burrow before excavation, visited twice daily and monitored for evidence that owls are inside and can't escape i.e., look for sign immediately inside the door).
4. How the burrow(s) will be excavated. Excavation using hand tools with refilling to prevent reoccupation is preferable whenever possible (may include using piping to stabilize the burrow to prevent collapsing until the entire burrow has been excavated and it can be determined that no owls reside inside the burrow);
5. Removal of other potential owl burrow surrogates or refugia on site;
6. Photographing the excavation and closure of the burrow to demonstrate success and sufficiency;

7. Monitoring of the site to evaluate success and, if needed, to implement remedial measures to prevent subsequent owl use to avoid take;
8. How the impacted site will continually be made inhospitable to burrowing owls and fossorial mammals (e.g., by allowing vegetation to grow tall, heavy disking, or immediate and continuous grading) until development is complete.

Appendix F. Mitigation Management Plan and Vegetation Management Goals

Mitigation Management Plan

A mitigation site management plan will help ensure the appropriate implementation and maintenance for the mitigation site and persistence of the burrowing owls on the site. For an example to review, refer to Rosenberg et al. (2009). The current scientific literature and field experience from 1995 to present indicate that an effective management plan includes the following:

1. Mitigation objectives;
2. Site selection factors (including a comparison of the attributes of the impacted and conserved lands) and baseline assessment;
3. Enhancement of the conserved lands (enhancement of reproductive capacity, enhancement of breeding areas and dispersal opportunities, and removal or control of population stressors);
4. Site protection method and prohibited uses;
5. Site manager roles and responsibilities;
6. Habitat management goals and objectives:
    a. Vegetation management goals,
        i. Vegetation management tools:
            1. Grazing
            2. Mowing
            3. Burning
            4. Other
    b. Management of ground squirrels and other fossorial mammals,
    c. Semi-annual and annual artificial burrow cleaning and maintenance,
    d. Non-natives control – weeds and wildlife,
    e. Trash removal;
7. Financial assurances:
    a. Property analysis record or other financial analysis to determine long-term management funding,
    b. Funding schedule;
8. Performance standards and success criteria;
9. Monitoring, surveys and adaptive management;
10. Maps;
11. Annual reports.

Vegetation Management Goals

- Manage vegetation height and density (especially in immediate proximity to burrows). Suitable vegetation structure varies across sites and vegetation types, but should generally be at the average effective vegetation height of 4.7 cm (Green and Anthony 1989) and <13 cm average effective vegetation height (MacCracken et al. 1985*a)*.
- Employ experimental prescribed fires (controlled, at a small scale) to manage vegetation structure;

- Vegetation reduction or ground disturbance timing, extent, and configuration should avoid take. While local ordinances may require fire prevention through vegetation management, activities like disking, mowing, and grading during the breeding season can result in take of burrowing owls and collapse of burrows, causing nest destruction. Consult the take avoidance surveys section above for pre-management avoidance survey recommendations;
- Promote natural prey distribution and abundance, especially in proximity to occupied burrows; and
- Promote self-sustaining populations of host burrowers by limiting or prohibiting lethal rodent control measures and by ensuring food availability for host burrowers through vegetation management.

Refer to Rosenberg et al. (2009) for a good discussion of managing grasslands for burrowing owls.

Mitigation Site Success Criteria

In order to evaluate the success of mitigation and management strategies for burrowing owls, monitoring is required that is specific to the burrowing owl management plan. Given limited resources, Barclay et al. (2011) suggests managers focus on accurately estimating annual adult owl populations rather than devoting time to estimating reproduction, which shows high annual variation and is difficult to accurately estimate. Therefore, the key objective will be to determine accurately the number of adult burrowing owls and pairs, and if the numbers are maintained. A frequency of 5-10 years for surveys to estimate population size may suffice if there are no changes in the management of the nesting and foraging habitat of the owls.

Effective monitoring and evaluation of off-site and on-site mitigation management success for burrowing owls includes (Barclay, pers. comm.):

- Site tenacity;
- Number of adult owls present and reproducing;
- Colonization by burrowing owls from elsewhere (by band re-sight);
- Evidence and causes of mortality;
- Changes in distribution; and
- Trends in stressors.

State of California

Memorandum

: “Div. Chiefs - IFD, BDD, NED, & WMD
Reg. Mgrs. - Regions 1, 2, 3, 4, & 5

Date : October 17, 1995

From : **Department of Fish and Game**

Subject : Staff Report on Burrowing Owl Mitigation

I am hereby transmitting the Staff Report on Burrowing Owl Mitigation for your use in reviewing projects (California Environmental Quality Act [CEQA] and others) which may affect burrowing owl habitat. The Staff Report has been developed during the last several months by the Environmental Services Division (ESD) in cooperation with the Wildlife Management Division (WMD) and regions 1, 2, and 4. It has been sent out for public review and redrafted as appropriate.

Either the mitigation measures in the staff report may be used or project specific measures may be developed. Alterative project specific measures proposed by the Department divisions/regions or by project sponsors will also be considered. However, such mitigation measures must be submitted to ESD for review. The review process will focus on the consistency of the proposed measure with Department, Fish and Game Commission, and legislative policy and with laws regarding raptor species. ESD will coordinate project specific mitigation measure review with WMD.

If you have any questions regarding the report, please contact Mr. Ron Rempel, Supervising Biologist, Environmental Services Division, telephone (916) 654-9980.

COPY Original signed by
C.F Raysbrook

C. F. Raysbrook
Interim Director

Attachment

cc: Mr. Ron Rempel
Department of Fish and Game
Sacramento

STAFF REPORT ON BURROWING OWL MITIGATION

Introduction

The Legislature and the Fish and Game Commission have developed the policies, standards and regulatory mandates to protect native species of fish and wildlife. In order to determine how the Department of Fish and Game (Department) could judge the adequacy of mitigation measures designed to offset impacts to burrowing owls *(Speotyto cunicularia;* A.O.U. 1991) staff (WMD, ESD, and Regions) has prepared this report. To ensure compliance with legislative and commission policy, mitigation requirements which are consistent with this report should be incorporated into: (1) Department comments to Lead Agencies and project sponsors pursuant to the California Environmental Quality Act (CEQA); and (2) other authorizations the Department gives to project proponents for projects impacting burrowing owls.

This report is designed to provide the Department (including regional offices and divisions), CEQA Lead Agencies and project proponents the context in which the Environmental Services Division (ESD) will review proposed project specific mitigation measures. This report also includes preapproved mitigation measures which have been judged to be consistent with policies, standards and legal mandates of the Legislature,. the Fish and Game Commission and the Department's public trust responsibilities. Implementation of mitigation measures consistent with this report are intended to help achieve the conservation of burrowing owls and should compliment multi-species habitat conservation planning efforts currently underway. The *Burrowing Owl Survey Protocol and Mitigation Guidelines* developed by The California Burrowing Owl Consortium (CBOC 1993) were taken into consideration in the preparation of this staff report as were comments from other interested parties.

A range-wide conservation strategy for this species is needed. Any range-wide conservation strategy should establish criteria for avoiding the need to list the species pursuant to either the California or federal Endangered Species Acts through preservation of existing habitat, population expansion into former habitat, recruitment of young into the population, and other specific efforts.

California's burrowing owl population is clearly declining and, if declines continue, the species may qualify for listing. Because of the intense pressure for urban development within suitable burrowing owl nesting and foraging habitat (open, flat and gently rolling grasslands and grass/shrub lands) in California, conflicts between owls and development projects often occur. Owl survival can be adversely affected by disturbance and foraging habitat loss even when impacts to individual birds and nests/burrows are avoided. Adequate information about the presence of owls is often unavailable prior to project approval. Following project approval there is no legal mechanism through which to seek mitigation other than avoidance of occupied burrows or nests. The absence of standardized survey methods often impedes consistent impact assessment.

Burrowing Owl Habitat Description

Burrowing owl habitat can be found in annual and perennial grasslands, deserts, and arid scrublands characterized by low-growing vegetation (Zarn 1974). Suitable owl habitat may also include trees and shrubs if the canopy covers less than 30 percent of the ground surface. Burrows are the essential component of burrowing owl habitat. Both natural and artificial burrows provide protection, shelter, and nests for burrowing owls (Henny and Blus 1981). Burrowing owls typically use burrows made by fossorial mammals, such as ground squirrels or badgers, but also may use man-made structures such as cement culverts; cement, asphalt, or wood debris piles; or openings beneath cement or asphalt pavement.

Occupied Burrowing Owl Habitat

Burrowing owls may use a site for breeding, wintering, foraging, and/or migration stopovers. Occupancy of suitable burrowing owl habitat can be verified at a site by detecting a burrowing owl, its molted feathers, cast pellets, prey remains, eggshell fragments, or excrement at or near a burrow entrance. Burrowing owls exhibit high site fidelity, reusing burrows year after year (Rich 1984, Feeney 1992). A site should be assumed occupied if at least one burrowing owl has been observed occupying a burrow there within the last three years (Rich 1984).

CEQA Project Review

The measures included in this report are intended to provide a decision-making process that should be implemented whenever-there is potential for-an action or project to adversely affect burrowing owls. For projects subject to the California Environmental Quality Act (CEQA), the process begins by conducting surveys to determine if burrowing owls are foraging or nesting on or adjacent to the project site. If surveys confirm that the site is occupied habitat, mitigation measures to minimize impacts to burrowing owls, their burrows and foraging habitat should be incorporated into the CEQA document as enforceable conditions. The measures in this document are intended to conserve the species by protecting and maintaining viable’ populations of the species throughout their range in California. This may often result in protecting and managing habitat for the species at sites away from rapidly urbanizing/developing areas. Projects and situations vary and mitigation measures should be adapted to fit specific circumstances.

Projects not subject to CEQA review may have to be handled separately since the legal authority the Department has with respect to burrowing owls in this type of situation is often limited. The burrowing owl is protected from “take” (Section 3503.5 of the Fish and Game Code) but unoccupied habitat is likely to be lost for activities not subject to CEQA.

Legal Status

The burrowing owl is a migratory species protected by international treaty under the Migratory Bird Treaty Act (MBTA) of 1918 (16 U.S.C. 703-711). The MBTA makes it unlawful to take, possess, buy, sell, purchase, or barter any migratory bird listed in 50 C.F.R. Part 10, including feathers or other parts, nests, eggs, or products, except as allowed by implementing regulations (50 C.F.R. 21). Sections 3505, 3503.5, and 3800 of the California Department of Fish and Game Code prohibit the take, possession, or destruction of birds, their nests or eggs. To avoid violation of the take provisions of these laws generally requires that project-related disturbance at active nesting territories be reduced or eliminated during the nesting cycle (February 1 to August 31). Disturbance that causes nest abandonment and/or loss of reproductive effort (e.g., killing or abandonment of eggs or young) may be considered "take"' and is potentially punishable by fines and/or imprisonment.

The burrowing owl is a Species of Special Concern to California because of declines of suitable habitat and both localized and statewide population declines. Guidelines for the Implementation of the California Environmental Quality Act (CEQA) provide that a species be considered as endangered or "rare" regardless of appearance on a formal list for the purposes of the CEQA (Guidelines, Section 15380, subsections b and d). The CEQA requires a mandatory findings of significance if impacts to threatened or endangered species are likely to occur (Sections 21001 (c), 2103; Guidelines 15380, 15064, 15065). To be legally adequate, mitigation measures must be capable of "avoiding the impact altogether by not taking a certain action or parts of an action"; "minimizing impacts by limiting the degree or magnitude of the action and its implementation"; "rectifying the impact by repairing, rehabilitating or restoring the impacted environment"; "or reducing or eliminating the impact over time by preservation and maintenance operations during the life of the action" (Guidelines, Section 15370). Avoidance or mitigation to reduce impacts to less than significant levels must be included in a project or the CEQA lead agency must make and justify findings of overriding considerations.

Impact Assessment

Habitat Assessment

The project site and a 150 meter (approximately 500 ft.) buffer (where possible and appropriate based on habitat) should be surveyed to assess the presence of burrowing owls and their habitat (Thomsen 1971, Martin 1973). If occupied habitat is detected on or adjacent to the site, measures to avoid, minimize, or mitigate the project's impacts to the species should be incorporated into the project, including burrow preconstruction surveys to ensure avoidance of direct take. It is also recommended that preconstruction surveys be conducted if the species was not detected but is likely to occur on the project site.

Burrowing Owl and Burrow Surveys

Burrowing owl and burrow surveys should be conducted during both the wintering and nesting seasons, unless the species is detected on the first survey. If possible, the winter survey should be conducted between December 1 and January 31 (when wintering owls are most likely to be present) and the nesting season survey should be conducted between April 15 and July 15 (the peak of the breeding season). Surveys conducted from two hours before sunset to one hour after, or from one hour before to two hours after sunrise, are also preferable.

Surveys should be conducted by walking suitable habitat on the entire project site and (where possible) in areas within 150 meters (approx. 500 ft.) of the project impact zone. The 150-meter buffer zone is surveyed to identify burrows and owls outside of the project area which may be impacted by factors -such as noise and vibration (heavy equipment, etc.) during project construction. Pedestrian survey transects should be spaced to allow 100 percent visual coverage of the ground surface. The distance between transect center lines should be no more than 30 meters (approx. 100 ft.) and should be reduced to account for differences in terrain, vegetation density, and ground surface visibility. To effectively survey large projects (100 acres or larger), two or more surveyors should be used to walk adjacent transects. To avoid impacts to owls from surveyors, owls and/or occupied burrows should be avoided by a minimum of 50 meters (approx. 160 ft.) wherever practical. Disturbance to occupied burrows should be avoided during all seasons.

Definition of Impacts

The following should be considered impacts to the species:

- Disturbance within 50 meters (approx. 160 ft.) Which may result in harassment of owls at occupied burrows;
- Destruction of natural and artificial burrows (culverts, concrete slabs and debris piles that provide shelter to burrowing owls); and
- Destruction and/or degradation of foraging habitat adjacent (within 100 m) of an occupied burrow(s).

Written Report

A report for the project should be prepared for the Department and copies should be submitted to the Regional contact and to the Wildlife Management Division Bird and Mammal Conservation Program. The report should include the following information:

- Date and time of visit(s) including name of the qualified biologist conducting surveys, weather and visibility conditions, and survey methodology;
- Description of the site including location, size, topography, vegetation communities, and animals observed during visit(s);
- Assessment of habitat suitability for burrowing owls;
- Map and photographs of the site;
- Results of transect surveys including a map showing the location of all burrow(s) (natural or artificial) and owl(s), including the numbers at each burrow if present and tracks, feathers, pellets, or other items (prey remains, animal scat);
- Behavior of owls during the surveys;
- Summary of both winter and nesting season surveys including any productivity information and a map showing territorial boundaries and home ranges; and
- Any historical information (Natural Diversity Database, Department regional files? Breeding Bird Survey data, American Birds records, Audubon Society, local bird club, other biologists, etc.) regarding the presence of burrowing owls on the site.

Mitigation

The objective of these measures is to avoid and minimize impacts to burrowing owls at a project site and preserve habitat that will support viable owls populations. If burrowing owls are detected using the project area, mitigation measures to minimize and offset the potential impacts should be included as enforceable measures during the CEQA process.

Mitigation actions should be carried out from September 1 to January 31 which is prior to the nesting season (Thomsen 1971, Zam 1974). Since the timing of nesting activity may vary with latitude and climatic conditions, this time frame should be adjusted accordingly. Preconstruction surveys of suitable habitat at the project site(s) and buffer zone(s) should be conducted within the 30 days prior to construction to ensure no additional, burrowing owls have established territories since the initial surveys. If ground disturbing activities are delayed or suspended for more than 30 days after the preconstruction survey, the site should be resurveyed.

Although the mitigation measures may be included as enforceable project conditions in the CEQA process, it may also be desirable to formalize them in a Memorandum of Understanding (MOU) between the Department and the project sponsor. An MOU is needed when lands (fee title or conservation easement) are being transferred to the Department.

Specific Mitigation Measures

1. Occupied burrows should not be disturbed during the nesting season (February 1 through August 3 1) unless a qualified biologist approved by the Department verifies through non-invasive methods that either: (1) the birds have not begun egg-laying and incubation; or (2) that juveniles from the occupied burrows are foraging independently and are capable of independent survival.

2. To offset the loss of foraging and burrow habitat on the project site, a minimum of 6.5 acres of foraging habitat (calculated on a 100 m {approx. 300 ft.} foraging radius around the burrow) per pair or unpaired resident bird, should be acquired and permanently protected. The protected lands should be adjacent to occupied burrowing owl habitat and at a location acceptable to the Department. *Protection of additional habitat acreage per pair or unpaired resident bird may be applicable in some instances.* The CBOC has also developed mitigation guidelines (CBOC 1993) that can be incorporated by CEQA lead agencies and which are consistent with this staff report.

3. When destruction of occupied burrows is unavoidable, existing unsuitable burrows should be enhanced (enlarged or cleared of debris) or new burrows created (by installing artificial burrows) at a ratio of 2:1 on the protected lands site. One example of an artificial burrow design is provided in Attachment A.

4. If owls must be moved away from the disturbance area, passive relocation techniques (as described below) should be used rather than trapping. At least one or more weeks will be necessary to accomplish this and allow the owls to acclimate to alternate burrows.

5. The project sponsor should provide funding for long-term management and monitoring of the protected lands. The monitoring plan should include success criteria, remedial measures, and an annual report to the Department.

Impact Avoidance

If avoidance is the preferred method of dealing with potential project impacts, then no disturbance should occur within 50 meters (approx. 160 ft.) of occupied burrows during the nonbreeding season of September 1 through January 31 or within 75 meters (approx. 250 ft.) during the breeding season of February 1 through August 31. Avoidance also requires that a minimum of 6.5 acres of foraging habitat be *permanently* preserved contiguous with occupied burrow sites for each pair of breeding burrowing owls (with or without dependent young) or single unpaired resident bird. The configuration of the protected habitat should be approved by the Department.

Passive Relocation - With One-Way Doors

Owls should be excluded from burrows in the immediate impact zone and within a 50 meter (approx. 160 ft.) buffer zone by installing one-way doors in burrow entrances. One-way doors (e.g., modified dryer vents) should be left in place 48 hours to insure owls have left the burrow before excavation. Two natural or artificial burrows should be provided for each burrow in the project area that will be rendered biologically unsuitable. The project area should be *monitored daily for one* week to confirm owl use of burrows before excavating burrows in the immediate impact zone. Whenever possible, burrows should be excavated using hand tools and refilled to prevent reoccupation. Sections of flexible plastic pipe should be inserted into the tunnels during excavation to maintain an escape route for any animals inside the burrow.

Passive Relocation - Without One-Way Doors

Two natural or artificial burrows should be provided for each burrow in the project area that will be rendered biologically unsuitable. The project area should be *monitored daily until the owls have relocated to the new burrows.* The formerly occupied burrows may then. be excavated. Whenever possible, burrows should be excavated using hand tools and refilled to prevent reoccupation. Sections of flexible plastic pipe should be inserted into burrows during excavation to maintain an escape route for any animals inside the burrow.

Projects Not Subject to CEQA

The Department is often contacted regarding the presence of burrowing owls on construction sites, parking lots and other areas for which there is no CEQA action or for which the CEQA process has been completed. In these situations, the Department should seek to reach agreement with the project sponsor to implement the specific mitigation measures described above. If they are unwilling to do so, passive relocation without the aid of one-way doors is their only option based upon Fish and Game Code 3503.5.

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Eyas 10(1):38 Spring 1987

Reproductive Success of Burrowing Owls Using Artificial Nest Burrows in Southeastern Idaho

by Bruce Olenick

Artificial nest burrows were implanted in southeastern Idaho for burrowing owls in the spring of 1986. These artificial burrows consisted of a 12” x 12” x 8” wood nesting chamber with removable top and a 6 foot corrugated and perforated plastic drainage pipe 6 inches in diameter (Fig. 1). Earlier investigators claimed that artificial burrows must provide a natural dirt floor to allow burrowing owls to modify the nesting tunnel and chamber. Contrary to this, the artificial burrow introduced here does not allow owls to modify the entrance or tunnel. The inability to change the physical dimensions of the burrow tunnel does not seem to reflect the owls’ breeding success or deter them from using this burrow design.

In 1936, 22 artificial burrows were inhabited. Thirteen nesting attempts yielded an average clutch size of 8.3 eggs per breeding pair. Eight nests successfully hatched at least 1 nestling. In these nests, *67 of 75* eggs hatched (59.3%) and an estimated 61 nestlings (91.0%) fledged. An analysis of the egg laying and incubation periods showed that incubation commenced well after egg laying bega. Average clutch size at the start of incubation was 5.6 eggs. Most eggs tended to hatch synchronously in all successful nests.

Although the initial cost of constructing this burrow design may be slightly higher than a burrow consisting entirely of wood, the plastic pipe burrow offers the following advantages: (1) it lasts several field seasons without rotting or collapsing; (2) it may prevent or retard predation; (3) construction time is minimal; (4) it is easy to transport, especially over long distances; and (5) the flexible tunnel simplifies installation. The use of this artificial nest burrow design was highly successful and may prove to be a great resource technique for future management of this species.

For additional information on constructing this artificial nest burrow, contact Bruce Olenick, Department of Biology, Idaho State University, Pocatello, ID 83209.

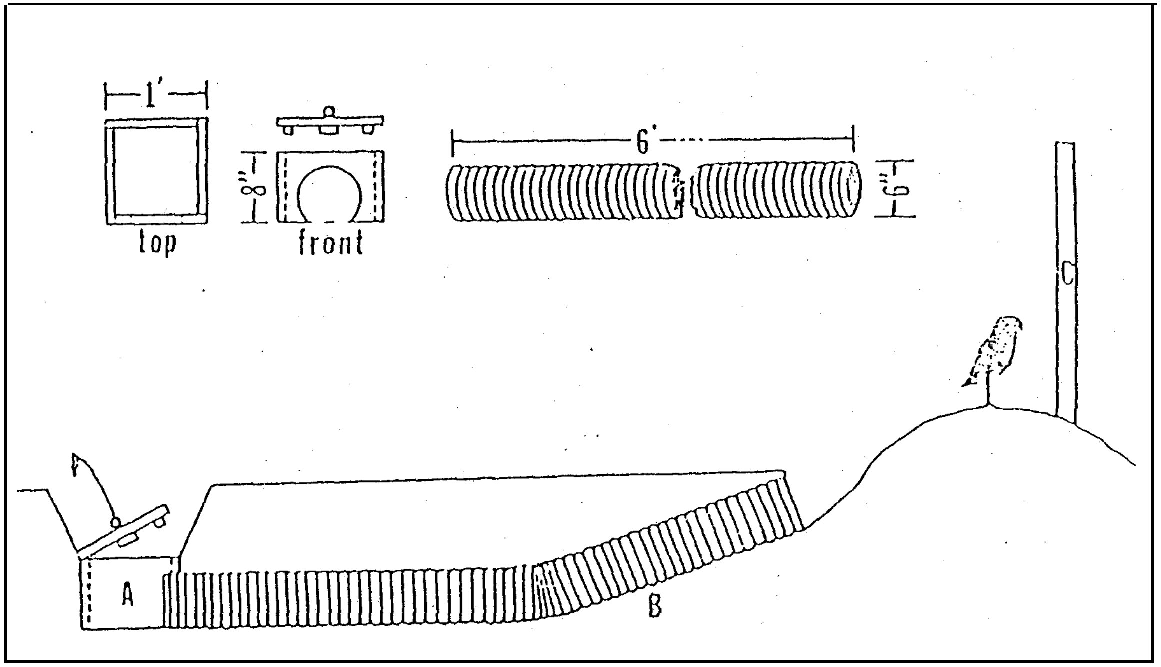

fig. 1 Artificial nest burrow design for burrowing owls Entire unit (including nest chamber) is buried 12" -- 18" below ground for maintaining thermal stability of the nest chamber. A= nest chamber, B = plastic pipe. C = perch.

ATTACHMENT A

California Fish and Wildlife JOURNAL

SPECIAL ISSUE

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California Fish and Wildlife

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California Fish and Wildlife, Recreation Special Issue; 6-10; 2020

Introduction

CAN OUR OUTDOOR ENTHUSIASM AND NATURE COEXIST?

RON UNGER, *Environmental Program Manager, Landscape Conservation Planning Program, Habitat Conservation Planning Branch, California Department of Fish and Wildlife*

[Note: As this special edition journal is published, our State, the nation, and the whole world are gripped by the corona virus pandemic. To slow its spread and not overwhelm limited healthcare resources, voluntary and mandatory directives for staying home, social distancing, and closing parks, reserves, and other public facilities have been put in place on a scale not seen for a hundred years, the time of the 1918 influenza (flu) epidemic.

Stories are emerging of more secretive wildlife seen in some park and urban areas normally filled with people, like the reports of bobcats roaming around empty Yosemite facilities, or an adult black bear roaming the nearly empty downtown Solvang. Hopefully, the pandemic and its horrible devastation will be over very soon, and we may again visit and appreciate our parks and wilderness areas. Hopefully, too, we may gain more information on wildlife's response to fewer visitors that helps us improve our management of parks and reserves in a way that protects wildlife and their habitat while also providing for great recreation experiences.]

"Everybody needs beauty as well as bread, places to play in and pray in, where nature may heal and give strength to body and soul alike" (The Yosemite, 1912). John Muir wrote so eloquently of the importance of taking time to be in, and play in, Nature to heal and nourish our spirit and help us to balance the challenges of our everyday lives. Now more than ever, people find a need to balance their work and domestic lives with the wonders, serenity, and invigorating challenges inherent in playing in Nature. In a world increasingly dominated by computers, cyberspace, and cities, people find a need to go and enjoy the Great Outdoors.

But what is the capacity of Nature to absorb the onslaught of millions of us hiking, riding, flying, boating, and otherwise tromping around the forests, fields, mountains, valleys, streams, and rivers on the other 40,000 or more species that also live in and depend on California? An increasing body of evidence is emerging that indicates non-consumptive recreational activities like hiking and biking, which don't involve harvesting of resources, can have harmful effects on species, their habitat, and efforts to protect them. As our population continues to grow and new and popular recreation technologies develop, California's natural areas are experiencing increased and changing recreation demands, such as increased numbers of hikers, nighttime group trail biking with lights, and electronic mountain bikes in wilderness areas.

Many federal, state, and local agencies' missions include non-consumptive, outdoor recreation, since it is often believed to be consistent with wildlife conservation. It is also widely believed that those who know and observe Nature are more likely to appreciate and protect her resources. Recently, however, several sites acquired primarily for conservation have experienced extreme recreation pressures such as the Disney-like crowds coming out to see "superblooms" of native flowers of the desert in the spring or mountain biking occurring in areas where it is illegal along with the creation of several miles of unauthorized trails. So, how can we continue to provide for and manage appropriate, legal recreation opportunities while also protecting California's amazing and vast diversity of plants, fish, and other wildlife species and their habitats? How and where can we acquire separate lands for recreation access and for protecting habitat instead of frequently demanding too much recreation access on lands set aside for conservation of species and habitat? And, how can we facilitate various consumptive and non-consumptive recreation groups (e.g., hikers, mountain bikers, equestrians, off-highway vehicle users, hunters, anglers) and conservation groups (e.g., environmental activists, land trusts, resource agencies) to work together to advocate for acquiring and managing separate recreation and conservation lands instead of increasingly coming into conflict with one another over the use of the same lands for both purposes?

This special edition journal seeks to tackle this and related questions. In the introductory essay, "Non-consumptive Recreation & Wildlife Conservation: Coexistence through Collaboration," Dr. Ashley D'Antonio points out the unique need and opportunity California has for addressing recreation use as a social-ecological system (SES) based on its high biodiversity and quickly increasing recreation use of protected lands. Mitrovich, Larson, Barrows, Beck, and Unger, in "Balancing Conservation and Recreation," point to a need for recreation and conservation stakeholders to work together to ensure that sufficient areas are acquired for both uses and to help plan and manage conservation lands better to reduce adverse effects on wildlife and natural resources. They summarize some of the varied research going on in the field, on wildlife behavior and physiology, habitat degradation and fragmentation, reproduction and survival, community composition and richness, and other topics. Indirect effects like the shifts in day and night activity patterns between predators and prey lead to questions on what effects that has on wildlife interactions and possible changes that may lead to in a whole ecosystem. Two case studies cover visitor perceptions and values, and the importance of having groups with different values come together and work through their differences to build trust and facilitate better management decisions and stakeholder support.

The research paper, "Increased hiking and mountain biking are associated with declines in urban mammal activity," by Larson, Reed, and Crooks provides findings on how some wildlife can respond rapidly to changes in the levels of human disturbance, which may help planners design targeted trail closures to reduce recreation impacts in important areas. Townsend, Hammerich, and Halbur conducted somewhat similar research to that of Larson, Reed, and Crooks and present their findings in "Wildlife occupancy and trail use before and after a park opens to the public." Their research provides good insights into how differently various wildlife species respond to trail use by people, including strong differences in how soon and how much species may habituate to people's presence. Baas, Dupler, Smith, and Carnes make the case in "An assessment of non-consumptive recreation effects on wildlife: current and future research, management implications, and next steps" for doing more research to help wildlife and park managers more effectively manage and respond to non-consumptive recreation impacts on wildlife species and their habitats.

Elizabeth Lucas points out deficiencies and a need to improve how recreation is sited, monitored, managed, and enforced in protected areas in her paper, "Recreation-related disturbance to wildlife in California – better planning for and management of recreation are vital to conserve wildlife in protected areas where recreation occurs." She also provides a review of several research papers in her paper, “A review of trail-related fragmentation, unauthorized trails, and other aspects of recreation ecology in protected areas.” Elizabeth points out the need for sufficient funding, science-based approaches to managing protected areas, and educating the public on recreation effects on wildlife, to achieve real protection of species and to retain the benefits of the protected lands. Elizabeth suggests several funding options including a compelling argument for establishing a recreation equipment excise fee or tax like those paid for over 80 years now by hunters and anglers to benefit habitat conservation. With so much use of outdoor areas now by “non-consumptive” recreation uses, and with declining popularity of hunting activities in the population at large, is it time to institute such a change for recreational users to pay their share of conserving and managing habitat?

Together, the articles in this special journal edition cover a broad array of research on recreation effects on wildlife. They provide interesting perspectives and offer a variety of solutions. Learning how to best manage non-consumptive recreation to provide great outdoor experiences while minimizing harmful effects on wildlife will continue to evolve as we learn more from research and experience. We hope that you find this special edition journal useful in your own exploration of this important and emerging field.

“Keep close to Nature’s heart… and break clear away, once in a while, and climb a mountain or spend a week in the woods. Wash your spirit clean.” *–John Muir*

Introduction--continued

NON-CONSUMPTIVE RECREATION AND WILDLIFE CONSERVATION: COEXISTENCE THROUGH COLLABORATION

ASHLEY D’ANTONIO, PHD, *Assistant Professor in Nature-Based Recreation Management, Gene D. Knudson Forestry Chair, Department of Forest Ecosystems and Society, Oregon State University*

The most basic principle in the field of recreation ecology—an interdisciplinary field that studies the ecological impacts of recreational activities and the management of these impacts—is that if outdoor recreation is allowed in an area, impacts to that ecosystem are inevitable. It is also established that outdoor recreation has a myriad of benefits to society that range from economic growth, improved human health and well-being, community building, and increases in an individual’s connection to nature. Moreover, outdoor recreation is one of the primary mechanisms by which humans interact with the natural world in contemporary society. As a result, many county, state, and federal park and protected area (PPA) managers around the United States (U.S.) are faced with mandates or missions that require conserving natural resources while also providing quality outdoor recreation experiences. Key challenges facing researchers, conservation practitioners, and PPA managers as they try to balance conservation goals with recreation access are: understanding the mechanism and the level and extent of these impacts; identifying what level of negative impact, if any, is acceptable; and deciding how to mitigate or manage these impacts.

Within recreation ecology, the impacts from recreation to ecosystem components such as soil and vegetation are relatively well studied. The negative impacts of recreation to environmental factors such as water, air quality, soundscapes, and wildlife are less well understood. Studying the relationships between non-consumptive recreation use and impacts to wildlife can be complex. Part of this complexity is because impacts to wildlife can be direct (e.g., harassment or feeding) and/or indirect (i.e., habitat modification) and at times can be hard to measure or observe (e.g., changes in stress hormone levels in response to recreation presence) as compared to soil or vegetation impacts. Additionally, impacts from non-consumptive recreation use can be interacting with, or compounded by, other ecosystem pressures. These added pressures include, but are not limited to, habitat loss due to development or changes in land use, pressures from consumptive recreation (hunting or fishing), and/or climate change. Moreover, impacts at the wildlife population or community level often require long-term studies, which are somewhat rare in recreation ecology but admittedly more common in the wildlife sciences.

Despite these challenges, there is a recent resurgence of interest in studying the impacts of non-consumptive recreation use on wildlife species. Meanwhile, there is a recognition that studies focusing only on the social or human aspects of a PPA system are insufficient to address current recreation and conservation issues, especially those related to wildlife. Many recreation ecologists, conservation scientists, and managers have begun to view outdoor recreation in PPAs as a complex social-ecological system (SES). As such, we must enhance our understanding of the interactions and intersections between both the ecological and social systems that make up our PPAs. Addressing wildlife conservation and recreation access in PPAs requires SES-focused thinking and collaborative problem solving.

The rich social and ecological systems comprising California make this state an excellent place to begin to address recreation use through an SES framework. California is one of the most biodiverse states in the U.S. and while 47% of the state is currently protected, 97% of these protected lands are opened to human access. Non-consumptive recreation use in PPA has increased rapidly in recent years across the U.S. but especially in Western states. California State Parks saw a 10% increase in total visitation numbers from the 2015/16 to 2016/17 fiscal year and many California national parks have seen exponential growth in visitation in recent years. As the U.S population becomes increasingly suburban and urban, PPAs that provide refugia and critical habitat for wildlife face increasing pressure from land use change and suburban expansion. Within California, this trend is evident as the state's population continues to grow while land use change, extreme droughts, and development increases pressure on California's PPAs.

Currently, PPAs and open space are limited, and wildlife species and their habitat face many ecological pressures. We are on the cusp of a resurgence and upswell of research exploring non-consumptive recreation impacts on wildlife. However, to meet conservation objectives, additional research is still needed to best inform recreation management in PPAs. Conserving and protecting wildlife species while providing quality recreation experiences to society requires interdisciplinary and transdisciplinary teams of researchers, managers, practitioners, stakeholders, and the public working together towards shared goals and objectives. Because of the social and ecological complexities and uncertainties around recreation impacts to wildlife, no individual field of science or management entity will be able to address this issue on its own. As such, this special issue is timely and important as it adds to the body of literature aimed at understanding non-consumptive recreation impacts to wildlife. Additionally, this special issue serves as a starting point for cooperatively exploring the challenge of protecting wildlife while balancing non-consumptive recreation use. If we are to meet conservation goals related to wildlife and wildlife habitat, it may not be appropriate to allow recreation use in all PPAs and at all times. However, collaborative dialogues (informed by the SES framework) around wildlife conservation are essential to guide decisions related to where, when, and how non-consumptive recreation use should be permitted in our PPAs.

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Balancing conservation and recreation

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As California's population has grown to nearly 40 million people, and as the State's beautiful natural diversity draws tourists and explorers from around the world, outdoor recreation has also grown (California Department of Parks and Recreation 2013, 2017; Monz et al. 2019). New equipment and technology enable new activities, such as night-time mountain biking, while social media brings increasing numbers of people to areas seldom visited by people only ten or twenty years ago. With increased time and more sedentary work environments, our society is understandably demanding greater access to more land for outdoor recreation. However, since several species-protection challenges already exist throughout the State due to development, fragmentation, invasive species, altered fire regimes, and climate change, consideration of opening up additional wildlands for recreation presents new challenges to conservation.

Outdoor engagement with natural areas is recognized as a necessary part of people's well-being, yet recreationists are generally attracted to the same high-value open spaces and natural areas that harbor diverse plant and animal communities (Mancini et al. 2018). Accordingly, trails, access points, and associated infrastructure need to be planned and managed appropriately to complement, rather than diminish, conservation values of lands dedicated to the protection of species and their habitats. In the absence of good planning, recreation-conservation conflicts are increasing, polarizing these two stakeholder groups and eroding their natural affinity and alliance. When conservation and recreation interest groups work together and conservation and recreation lands are planned and managed based on scientific research, a new opportunity emerges for a coordinated approach to protecting California's wildlife while also meeting the demand for high-quality recreational opportunities for diverse user groups.

Recreation and conservation interests would benefit from regular dialogue and collaboration with each other and with federal, state, and local land use authorities regarding regional and local land use planning, acquisitions, and management. A shared, basic understanding of applicable conservation objectives and regulations would provide context and perspective for recreational users and serve to help the two groups work together to ensure each of their interests are served rather than their respective needs being compromised. Without a close alliance among recreation and conservation interests, California risks having insufficient land areas set aside for the thousands of species that depend on California's natural areas, inadequate areas for recreation, and increasing conflicts between conservation and recreation needs. The necessary conversations, research, and determination to collaborate should be embraced and acted upon as soon as possible to help address these needs, reduce the potential for polarization among these stakeholders, and help ensure good land use planning and management decisions are made as development proceeds.

In this essay, we provide an overview of the mechanisms available to implement conservation in California and introduce many of the issues attributed to outdoor recreation when managing for wildlife and natural resources on conservation lands and other public open spaces. We then describe two case studies from our work in southern California that highlight the perceptions and values of outdoor recreationists when visiting conserved lands. The case studies also demonstrate what a successful balance between conservation and recreation uses can look like when moving from conflict to collaboration. We end with a discussion of what is required to achieve that balance and ways to minimize the impacts of outdoor recreation on wildlife and other natural resources.

CONSERVATION CONTEXT

As California's population grew from a few hundred thousand to nearly 40 million people in less than two hundred years, numerous species' populations have declined. Some, like the iconic grizzly bear (*Ursus arctos horribilis*), are now extinct in the state. Over 450 plant and animal species in California are now listed by the federal or state government as threatened or endangered (CDFW 2019). The cost of species recovery can be enormous, such as the tens of millions of dollars spent to save the majestic California condor (*Gymnogyps californianus*; Walters et al. 2010). To prevent further species declines, a number of laws and regulations exist to avoid, minimize, or compensate for impacts of human activities on species. In California, these include the federal Endangered Species Act (ESA), the California Endangered Species Act (CESA), the National Environmental Policy Act (NEPA), the California Environmental Quality Act (CEQA), among others. Approximately half of California is federally or state-owned lands with a variety of uses, from national forests and state parks to multi-use areas and reserves. In addition to these areas, an appreciable amount of land is conserved in California as mitigation under ESA, CESA, CEQA, and other laws and regulations.

Successful conservation leads to the protection of species and habitat and the preservation of natural landscapes. Principal types of conservation lands in California include reserves acquired and managed as part of Habitat Conservation Plans (HCPs) and Natural Community Conservation Plans (NCCPs), national parks and monuments, state ecological reserves and wildlife areas, state parks, lands owned by private entities (e.g., land trusts), lands with conservation easements, and mitigation lands. The relative importance of conservation and recreation values to the management goals of these lands vary. For example, state and national parks generally emphasize recreational uses more than mitigation lands and ecological reserves. Sixteen HCP/NCCPs have been approved in California covering part or all of seven counties. Through the new Regional Conservation Investment Strategy (RCIS) Program established in 2017, one RCIS has been approved and an additional eight Regional Conservation Investment Strategies (RCISs) are currently in development or have been submitted for review and approval by the Department of Fish and Wildlife (for more information about RCIS and NCCP programs, see Appendix I). The nine RCISs together will cover part or all of 11 counties. There are also over 130 conservation and mitigation banks in the state, privately held conservation lands, and hundreds of mitigation sites. In total, tens of thousands of acres of habitat have been conserved in California through proactive investments and mitigation. Over one and one-half million acres will be conserved in California under approved HCP/NCCPs, benefiting hundreds of species listed as endangered or threatened under federal and state species protection laws.

OUTDOOR RECREATION

Millions of Californians and visitors recreate outdoors on natural lands within the state each year (Outdoor Industry Association 2019). Examples of outdoor recreation activities include hiking, trail running, mountain biking, horseback riding, backpacking, camping, and motorized activities. The positive effects of outdoor recreation are numerous. Stewardship values are enhanced. Appreciation of nature is magnified as people are exposed to the inherent beauty, complexity, and serenity of natural systems. The next generation of land stewards and conservationists are born out of the experience of being introduced to wildlands when young. Equally important, the mental health benefits of exposure to the outdoors and participation in nature are now well-recognized (Louv 2005; Thomsen et al. 2018). For a society that is increasingly becoming more urban and digital, the restorative properties of nature and the increased social well-being of individuals and communities is ever more important.

Despite these benefits, the negative effects of recreation on wildlife can be profoundly damaging to species and their habitats and must be considered when planning for conservation areas (Hammitt et al. 2015). Trails lead to habitat degradation and fragmentation, which increase when visitors go off-trail and informal trails proliferate. Harassment of wildlife, though often unintended, occurs with increased visitation to an area. Less obvious impacts to wildlife, not easily measured, have been tied to noise, light pollution, trash, and other factors associated with recreation activities.

In general, it can be difficult to accept that recreation activities, especially quiet, non-motorized activities like hiking and mountain biking, can have harmful effects on wildlife. Many types of recreation cause little physical habitat change. Perhaps as a result, recreation was widely assumed to be a “benign use” that is compatible with conservation goals (Knight and Gutzwiller 1995) and is permitted in the vast majority of protected areas worldwide (Eagles et al. 2002; IUCN and UNEP 2014). Many HCP/NCCPs include a general provision that allows for “low-impact nature trails” without strongly defining what that means and what types and levels of use would be acceptable, given the species that are to be protected. The viewpoint that recreation is a benign use may be changing, however. In recent years, researchers have found evidence that a variety of recreation activities and intensities can have detrimental impacts on wildlife (Geffory et al. 2015; Larson et al. 2016; Samia et al. 2017).

RECREATION EFFECTS ON WILDLIFE

Behavior, activity budgets, and physiology

Behavioral reactions, such as flight, flushing, or vigilance are some of the most commonly-observed and studied wildlife responses to recreationists (Larson et al. 2016). Changes in activity budgets have also been observed, with animals typically spending less time in activities such as foraging and caring for young and more time moving or being vigilant when recreationists are present (Schummer and Eddleman 2003; Arlettaz et al. 2015). Physiological responses, such as increases in stress hormones (Arlettaz et al. 2007) or decreased body mass (McGrann et al. 2006), are less obvious to observe, and can occur even when a corresponding behavioral response does not. It is critical not to assume that an animal is tolerant of recreation simply because it does not exhibit a visible response.

Habitat degradation and fragmentation

Recreation can degrade or fragment habitat, resulting in habitat that is otherwise of high quality being used less frequently or not at all. This is particularly concerning in highly fragmented or developed landscapes where remaining habitat is scarce and there is limited opportunity for wildlife to move to alternative areas. Researchers have observed avoidance of areas used by recreationists in species as diverse as grizzly bears (Coleman et al. 2013), wolverines (*Gulo gulo*; Heinemeyer et al. 2019), caribou (*Rangifer tarandus*; Lesmerises et al. 2018), capercaillie (*Tetrao urogallus*; Coppes et al. 2017), and dolphins (*Tursiops* spp.; Lusseau 2005).

Reproduction, survival, and abundance

Assessing recreation’s impacts on wildlife population abundance or vital rates can be difficult and time-consuming, and is therefore largely unknown. In one of the few studies of population trends in relation to recreation, Garber and Burger’s long-term study (1995) observed dramatic declines in North American wood turtle populations after the area was opened to recreation. Reproductive success is one of the better-studied population vital rates; negative effects of recreation on reproductive success have been observed in several species including elk (*Cervus canadensis*; Shively et al. 2005), penguins (Giese 1996; Lynch et al. 2010), and plovers (*Charadrius* spp.; Lafferty et al. 2006; Yasué and Dearden 2006). However, other studies have found that habituation can moderate impacts of recreation on reproductive success (Baudains and Lloyd 2007).

Community composition and richness

Within an ecological community, species respond to recreation differently. This can lead to changes in community composition if more sensitive species avoid areas with recreation or decline in abundance while the habitat use or abundance of tolerant species remains constant or even increases due to reduced competition. When the sensitive species are native and the more tolerant species are non-native, this can lead to dramatic declines of native species as compared to their non-native counterparts (Reed and Merenlender 2008). Overall species richness can also decline if sensitive species disappear from local communities (Bötsch et al. 2018).

Indirect effects

Recreation can also cause other changes that indirectly affect wildlife, many of which are not well understood. Shifts in diel activity patterns could change the way that species interact with each other or with their environment, potentially leading to increased interspecific competition during nighttime hours or increased overlap between predators and their prey (Gaynor et al. 2018). Recreation can facilitate the spread of non-native species in freshwater, marine, and terrestrial environments (Anderson et al. 2015), which can have dramatic effects on native wildlife. Recreation activities also often involve infrastructure (e.g., parking lots, maintenance buildings, roads, ski lifts), which can lead to further habitat loss and fragmentation (Nellemann et al. 2010).

Examples of recreation impacts from southern California

Examples from southern California, where much of our work occurs, highlight some of the many ways recreation can impact natural resources. Results of ten years of camera-trap studies on conservation lands in Orange County indicate mule deer (*Odocoileus hemionus*) and coyotes (*Canis latrans*) are shifting the timing of activity due to the presence of humans on trails creating novel predator-prey conflicts for wildlife (Patten et al. 2017). Observed shifts toward more nocturnal activity by both species leads to greater temporal overlap in activity periods between mule deer and their principal predator, the mountain lion (*Puma concolor;* Figure 1). Greater overlap between coyotes and gray foxes (*Urocyon cinereoargenteus*) has also been observed, leading to predicted changes in predator-prey dynamics.

Bobcat (*Lynx rufus*) movement modeling using more than ten years of telemetry data in the 7,284-ha South Coast Wilderness of coastal Orange County highlights the importance of maintaining regional connectivity among isolated parcels and continued exclusion of human presence at culverts and other critical linkage points along the coast (Boydston and Tracey 2018). Within landscapes containing natural areas constrained by development, protected habitat and other high-value open space is a premium for wildlife. Providing for safe, unobstructed passage for wildlife among isolated parcels, especially at culverts and other pinch-points, is essential to enable access to high-value habitat within these otherwise constrained landscapes.

In heavily used open space areas, some wildlife appear to develop a tolerance for regular human activity on trails over time. However, patterns of wildlife habitat use can be disrupted by disturbances occurring outside this regular activity, such as large recreation events, off-trail visitor behavior, or the proliferation of new social trails, even in areas that traditionally see high levels of visitor use. At a local scale, observations of breeding bird behavior before, during, and after a mountain bike race at a wilderness park in Orange County highlights elements of both sides of this phenomenon (Hamilton et al. 2015). In this example, breeding bird behavior continued uninterrupted in areas experiencing similar amounts of activity along the racecourse during the event as to what was experienced prior. As people gathered in numbers on and off the trail at the designated start/end staging area for the event, evidence suggests behavior was disrupted as the sheer volume and continual

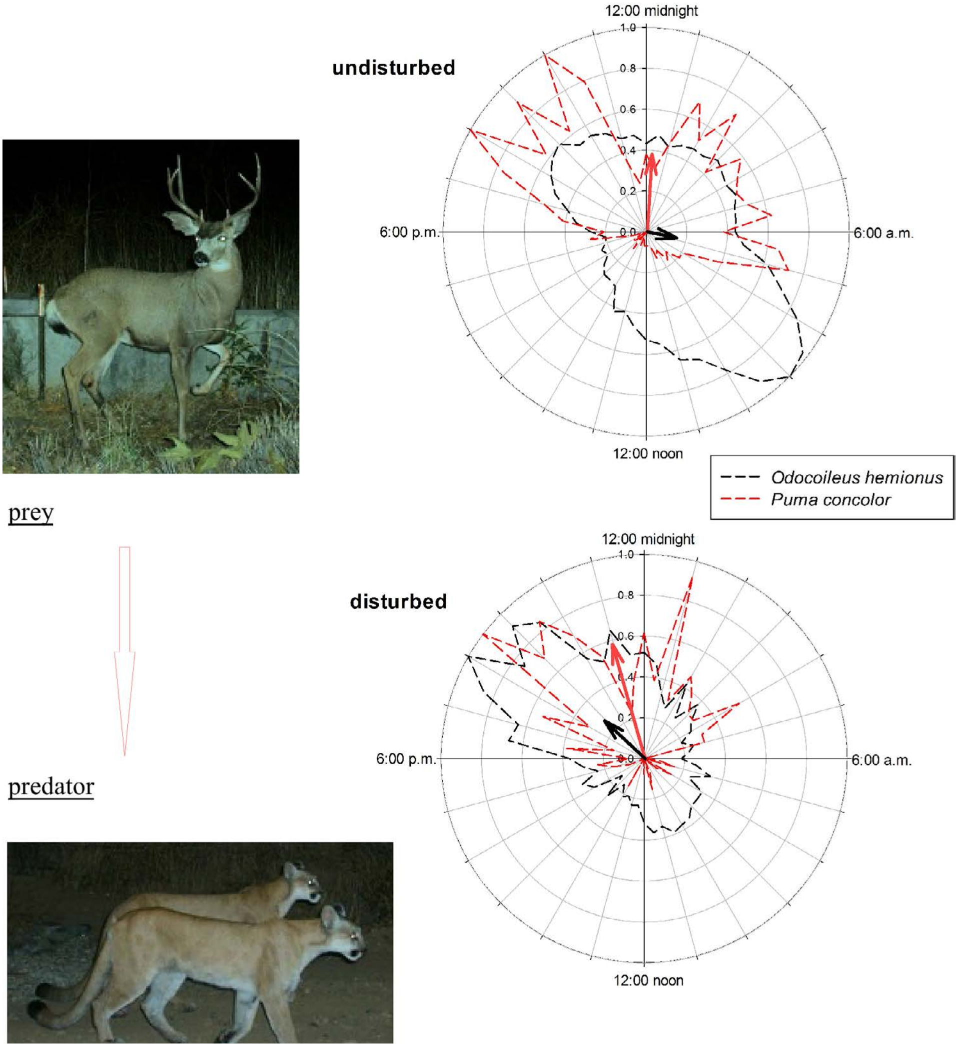

Figure 1. Diel activity of the mule deer and mountain lion with or without human disturbance. Arrows indicate time (direction) and proportional magnitude (length) of mean activity, and the "net" displays the spread of activity on a 24-h clock, binned at 30-minute intervals. Note the prey's (the deer) nocturnal shift when disturbance was present. (Figure credit: Patten et al. 2019)

presence of people gathered around the staging area was atypical for this location within the park.

CASE STUDY:

UNDERSTANDING VISITOR PERCEPTIONS AND VALUES IN ORANGE COUNTY

To successfully strike a balance, we need to know more about the human perspective of conservation. By surveying visitors to protected natural areas in southern California over the last couple of years, we learned there is potential for a shared vision of nature protection addressing the needs of both conservationists and outdoor recreationists. Clearly the issues are complex, but with good planning and communication, much can be done to support the creation of a collective vision for compatible conservation and recreation.

Natural Communities Coalition (NCC) is the non-profit management corporation overseeing implementation of the conservation strategy for the County of Orange Central and Coastal Subregion NCCP/HCP. Stretching from the Newport Coast to the Santa Ana Mountains, over 20,200 ha (50,000 acres) of conserved lands together with National Forest are embedded within the conservation plan's 84,000-ha (208,000-acre) planning area. The 75-year plan, signed in 1996, was the first landscape-scale NCCP in the state and one of the first multi-species HCPs nationally.

With 3.2 million residents in Orange County (Center for Demographic Research 2019), the demand for outdoor recreation on lands protected for conservation purposes is ever-present and increasing. Equally important is the recognized need and desire by the community to conserve the rich natural heritage of the southern California region. In Orange County, like in other high-value natural areas of the state experiencing rapid population growth, there is a strong need to strike a balance between conservation and recreation.

Recreation management is one of four main tenets of the regional landscape-level conservation strategy managed by NCC. Recognizing the increasing need to address this topic, NCC staff began focused and meaningful conversations with recreation ecologists and then followed with talking directly to park visitors to understand the human dimensions, that is, the motivations, desires, and values of visitors to the conserved lands. Partnering with Dr. Christopher Monz, Professor of Recreation Resources Management in the Department of Environment and Society at Utah State University, the organization surveyed close to 2,000 visitors in the spring and fall seasons of 2017 and in the spring of 2018 to better understand their perceptions, values, and characteristics (Sisneros-Kidd et al. 2019). In this process, the research team used a theoretical framework that allowed for the identification of internal constructs embedded within visitor questionnaires to reveal motivations and define different user groups. Through the work, two principal groups or clusters of visitors were discovered, those who are motivated most by the opportunity to experience nature immersion and those who are more focused on fitness-based recreation.

Surprisingly, given the urban-proximate setting, and in contrast to the expectations of local land managers, by almost two to one, recreationists were looking to experience nature immersion compared to those seeking fitness-based recreation. These visitors were more motivated by solitude and escape, learning about and experiencing nature, spiritual renewal, and the social experience, versus those in the fitness-based recreation group who were motivated principally by challenge and outdoor exercise. Learning that the motivation and values of most visitors are more in alignment with resource protection than expected, we had to shift our thinking. Rather than focusing on direct conflict between recreation and conservation, we had to reevaluate how the conversation about balancing recreation and conservation is framed. Knowing it is often the most vocal and well-organized user groups who receive the greatest attention, whether from rangers at a local park or elected officials at a public meeting, we recognized it was of value for decision-makers to be informed of the findings and equally consider the motivations, values, and desires of the quiet majority in these public spaces and forums.

Digging deeper into the results of the work, we found people largely recognize the value of habitat and natural resource conservation; however, they too want to be part of the story. People do not want to be left "standing on the sidelines or looking over the fence;" they want to experience the rich natural resources that make California so unique. When asked how satisfied they were in their ability to achieve a variety of experiences during their visit to a park, visitors reported they were often left wanting more when it came to learning about nature and becoming more in touch with their spiritual values.

Visitor responses indicated they experience place attachment. When asked, they recognize the lands upon which they choose to regularly recreate are not necessarily unique relative to other protected areas. However, to them these lands and parks are special, meaningful, and important. Place attachment may be reflected in the high repeat visitation rates of visitors. More than half of those surveyed visited parks more than 50 times within the same year. Furthermore, many of the visitors live within neighboring communities. For almost half of the parks included in the study, more than 25% of visitors live within 3 miles of an entrance location (Mitrovich, unpublished data). To these people, the parks are a recognized and utilized part of their local community's resources.

Recreation is multidimensional and multifaceted, and we recognize a more sophisticated approach to finding solutions is warranted when seeking to minimize recreational impacts on sensitive natural resources. Impacts and motivations vary by user group, as does the attractiveness of different topography. From the surveys, we learned mountain bikers look to avoid crowds, are most knowledgeable about "leave no trace," most interested in more trails, and most likely of all user groups to be satisfied in their ability to get away from the demands of life when out on trails. Dog walkers, on the other hand, were least knowledgeable about "leave no trace," most avoided by other recreational groups, and least satisfied in their park experiences as it relates to their ability to learn more about plants and animals. Some hikers and runners were concerned about the number of mountain bikers they encountered in particular parks and along certain trails. Different topographic features attracted different users. Steep trails that offer high speeds and technical challenges are attractive to mountain bikers but can be off-putting to other user groups. In unregulated spaces popular with the masses and advertised through social media, trails can be degraded and spider, further fragmenting and degrading available habitat. The overlap between areas used for recreation and high-value wildlife habitat may be greatest with nature-based recreationists.

One positive take-home, as we look for solutions, is that visitors in urban landscapes are much more tolerant of crowded conditions than previously recognized by land managers. Parks in Orange County have seen a dramatic increase in use over the last decade, with increases of greater than 50% not uncommon over a 4-year period (Monz et al. 2019). However, at many parks considered to be "crowded" by land managers, over 80% of respondents surveyed did not feel the presence of other people on the trail interfered with their activities or made them feel rushed or slowed them down during their visit. Equally, over 80% of respondents in 2018 did not feel the number of people at the park increased their risk of injury.

Although many folks are comfortable in a more crowded space, not everyone is comfortable with the changing dynamics and increases in observed use experienced over the last decade. Across both before-mentioned measures, there were respondents that felt the number of people at the park during their visit did increase their risk of injury at least some of the time, and other visitors and their activities interfered with their visit. Like wildlife, it appears people's tolerance of novel conditions is not fully universal and may differ across generations, by past experiences, and expectations (Shelby et al. 1983). When coupled with their understanding that off-trail activity is most impactful, the general tolerance of folks to increased visitation rates gives hope as we look for solutions to meeting increased demand while paying the necessary attention to detail to create the recreational opportunities valued by most that continue to honor the shared commitment and need for lasting conservation.

CASE STUDY:

CONFLICT TO COLLABORATION IN THE COACHELLA VALLEY

Now we turn to one example of how a region is addressing the question, what to do when trail users and sensitive species like the same habitat? Like other areas of southern California, the Coachella Valley in the desert and mountain regions of eastern Riverside County has seen a remarkable increase in the demand for outdoor recreation on trails, especially hiking and mountain biking. In this desert resort area, land of more than 100 lush golf courses, demand for golf is flat, while hiking has surged in popularity, in large part due to the influence of social media.

In 2008, the California Department of Fish and Wildlife (CDFW) and the U.S. Fish and Wildlife Service (USFWS) approved the Coachella Valley Multiple Species Habitat Conservation Plan/Natural Community Conservation Plan (CVNCCP) with a 75-year permit. Like other efforts in California and beyond, it was a visionary effort to balance conservation and development. The plan encompasses an area of almost 500,000 ha (1.1 million acres) from Palm Springs to the Salton Sea and beyond. Implementation of the plan is overseen by the Coachella Valley Conservation Commission (CVCC), made up of elected officials from participating cities, Riverside County, local water districts, and other agencies.

However, several years earlier, the conflict between trail users and agency biologists nearly derailed the CVNCCP. During development of the plan, proposals by state and federal wildlife agencies to impose seasonal closures on some trails galvanized trail users to organize and turn out in large numbers at public hearings. The proposal to close trails centered on concerns about the impacts of trail use on Peninsular bighorn sheep (*Ovis canadensis nelsoni*), a state and federally listed endangered species (Figure 2). In response, trail users read scientific literature, interviewed bighorn sheep biologists, and questioned the scientific basis of the trail restrictions. They used their newfound knowledge and spoke passionately about their concerns to elected officials, often quoting published scientists.

When the CVNCCP was approved in 2008, it did not include the trail closures that had been envisioned. Public input from trail users convinced decision-makers to avoid these measures. It also convinced conservation planners that a full trails management plan needed to be developed for the CVNCCP. Unfortunately, the process also left trail users alienated and with a lack of trust in the state and federal wildlife agencies. Wildlife agencies were suspicious of trail users' motivations. It would be years before these attitudes changed. Trail users seeking nature immersion, who could have been a natural constituency for support of the conservation proposed by the CVNCCP, continued to question the scientific basis of the trails plan. Even after the CVNCCP was completed and fully permitted, the lack of trust remained.

To provide a forum for input from trail users and local governments, the final CVNCCP called for formation of a Trails Management Subcommittee, composed of a representative from each of nine cities involved in the CVNCCP, the Agua Caliente Band of Cahuilla Indians, trail user groups (mountain bikers, hikers, equestrians), environmentalists, biologists from CDFW, USFWS, Bureau of Land Management, and other land management agencies. The group was charged with providing recommendations on trails management, annually reviewing the status of bighorn sheep, and communicating trails-related information to stakeholders. Their tasks required them to develop a shared understanding of relevant conservation objectives and regulations while they worked together to accomplish their charge.

Figure 2. In some areas of the Santa Rosa and San Jacinto Mountains National Monument, seasonal trail closures are in place to allow bighorn sheep and other wildlife access to waterholes during the hot summer months. (Photo credit: CDFW)

A dedicated group of volunteers, the subcommittee took their responsibility seriously and worked hard. Meetings were well attended and often animated. Passions flared, and sometimes sparks flew. On occasion, meetings devolved and became acrimonious and full of conflict. Trail users continued to question the scientific basis for trails management actions proposed by "the agencies." Agency biologists doubted the trail users' commitment to the protection of bighorn sheep and were reluctant to share data. Unfortunately, throughout the process, scientifically rigorous data on the effects of trail use on bighorn sheep was limited. The studies needed to understand the relationship between trail use and bighorn sheep had not been done. The CVNCCP was approved in 2008, the year the recession hit and resources for local, state, and federal agencies were further limited by lack of funding.

In 2011, the conflict between recreation and habitat ended up in the state legislature when CDFW closed the upper portion of the very popular Bump and Grind Trail to protect bighorn sheep. Though not a trail which offers the experience of solitude, the Bump and Grind provides a great cardio workout, with hikers numbering more than 1,000 some days. Questioning whether any studies to prove that hikers have an impact on the endangered bighorn had been presented, trail users went to their state legislators. Ultimately, a compromise was worked out and Governor Brown signed legislation in October 2013. The upper Bump and Grind is now closed for three months during the sensitive bighorn sheep lambing season, from February through April, and open for the remaining nine months of each year. The Coachella Valley Conservation Commission worked with CDFW to install a fence to discourage off-trail travel and educational signs about bighorn sheep.

Despite the challenges, the Trails Management Subcommittee persevered. They worked through the challenges, developed more trust, and learned to work together. They completed an update to the 2008 Trails Management Plan in 2014. The updated plan emphasizes the adaptive management approach described in the CVNCCP. It calls for research on the relationship between bighorn sheep habitat use and trail use, prior to construction of new trails. Technology has made such research more feasible, especially in the rugged and remote terrain of the Santa Rosa and San Jacinto Mountains National Monument. Since 2015, GPS collars have been placed on bighorn sheep, providing data on their movements and habitat use. The CVCC is now working on a study of bighorn sheep and trails, led by Dr. Kathleen Longshore of the U.S. Geological Survey and funded by a grant from CDFW. The trails subcommittee is actively involved with researchers in the development of the study protocol and review of all data. Field work began in fall 2019, with volunteers collecting data on recreational trail use and researchers comparing the human use data with bighorn sheep collar data.

Conflict has been replaced with collaboration. Although all of the best practices were not used initially, when they were used, they became lessons learned. If people understand why, they are more likely to go along with regulations (Marion and Reid 2007). Furthermore, when the need for regulation or constraints are understood, constraints can become a positive as they provide the basis for best practices and assure access via responsible use.

WHAT IS NEEDED TO ACHIEVE BALANCE? WHAT WORKS?

Several land management decisions are being made today with long-term implications for the state of biodiversity and human wellness within California. Without collaboration among recreation and conservation interests, California risks insufficient lands being set aside for the benefit of protecting species, insufficient lands for recreating, and poorly located lands for both purposes, with people and other species suffering the consequences. Recreation and conservation stakeholders need to talk and work with each other and with ecologists and land planners early and often in the regional visioning and land planning process to ensure both interests get what is needed in a way that strikes a balance for species and habitat protection, and people's access to the outdoors.

To achieve a better land use future for conservation and recreation outcomes, we recommend early investment in working relationships. Increased early communication among all stakeholders, land planners, and managers, together with basing decisions on the best available science, can help reduce land use conflicts, the loss of species, and lower-quality recreation experiences. Groups should accept there will be situations when they collectively agree to disagree. However, the long-term commitment to work together will increase the likelihood of achieving goals and objectives for all interests. Most land conserved through public funding sources and/or mitigation and all HCP and NCCP properties have some form of Resource Management Plan (RMP) and/or Conservation Easement attached to them. It is critical RMP's are developed with a "clean slate" to identify critical sensitive species, regional context, and wildlife linkages up front. This, in turn, identifies potential areas appropriate for trails and other recreational uses, thus reducing debate and conflict later.

We also recommend establishing appropriate monitoring programs that are used to evaluate conservation and recreation outcomes and modify management plans to better achieve the original goals and adjust to changing conditions. The wide variety of nature-based recreational activities, timing and frequency of those activities, and numbers of people that participate in them, all result in a complex array of potential effects. Adding to that is the complexity of behavioral responses and sensitivities of different species to those activities. Recognizing this complexity and planning according to research findings that are available, and the anticipated growth or other changes expected, can help planners create conservation areas and recreation areas positioned to avoid future conflicts.

Opportunities to be inclusive and reach out to stakeholders as partners in the long-term management of protected lands are numerous. By simply involving everyone up-front, community members can be engaged early in the planning process and contribute to the search for solutions. Volunteers can help to enforce site rules using peer pressure. They may also be able to help with site maintenance, monitoring, and identification of possible management actions, such as when monitoring information indicates a problem exists. An open phone line to land managers is essential and over time naturally builds relationship and trust.

How can effects be minimized?

Using good science in the decision-making process is key, as is making data transparent and remembering the importance of educating the public throughout the process. Planning efforts should search for and incorporate relevant scientific findings. Despite the variability in species responses to different types and intensities of recreation, researchers have identified some ways to minimize the effects of recreation on wildlife:

- Monitor and prevent unauthorized trail creation and off-trail use. Many animal species respond more strongly to recreationists in unexpected places, such as off-trail (Stankowich 2008; Heinemeyer et al. 2019), so increasing the predictability of human presence by constraining people to the existing trail network may help mitigate negative effects.
- Limit nighttime access to parks and trails. Since people are primarily active during the daytime, many animal species avoid interactions with people by increasing the proportion of their activity that takes place at night (Gaynor et al. 2018). While the

implications of this shift for foraging success and interspecific interactions are largely unknown, limiting activity to daytime hours may be a way for humans and wildlife to coexist in parks and natural areas. Nighttime recreation is growing in popularity but may prevent animals from temporally avoiding people, and should be limited in general, and probably all together avoided in urban-proximate wildland areas where the existence of refugia is already severely limited spatially.

- Leave areas without trails, both within individual properties and at landscape scale. For the most part, research has not yet identified 'safe' levels of human activity that result in minimal negative outcomes for wildlife. Some species appear to respond to very low levels of human activity and would benefit from blocks of trail-free habitat; in one example, mountain lions, coyotes, and bobcats increased nighttime activity and decreased daytime activity in locations with levels of use as low as two people per day (Wang et al. 2015).
- Plan access points and infrastructure carefully. Parking lots and other facilities can increase the level of use at corresponding trails (Larson et al. 2018). On the other hand, a lack of parking space at popular trails can result in public safety issues if visitors park along busy roadways. Improper parking can also impact habitat, which can cascade to impact wildlife as well.
- Use seasonal trail closures during sensitive periods. For many species, the most sensitive period is the breeding period, when disturbance can lead to reduced reproductive success (Bötsch et al. 2017), which in turn can result in population declines.
- Collect visitor use data. Without some knowledge of the intensity and distribution of recreational use, it is difficult for managers to know where and when impacts on sensitive wildlife species may be occurring. Monitoring equipment can be costly to purchase and maintain, but basic measures like periodic manual counts at parking lots or trailheads can be helpful in tracking trends, and there are promising emerging approaches using information that visitors share on social media platforms, mobile devices, and fitness applications (Fisher et al. 2018; Monz et al. 2019; Norman et al. 2019).
- Consider diverse visitor perspectives and values. Employ contemporary scientific approaches so key components in the human dimension of recreation (e.g., perceptions, characteristics, and motivations) can be understood more formally and inform a planning process for long-term sustainable use.
- Determine thresholds of acceptability of key indicators of resource and social conditions. Recognize "carrying capacities" exist for protected lands and their identification is a key component in the planning process and essential to developing a range of possible management actions, from the spatial and temporal separation of different types of recreational uses to acceptance and identification of high and low intensity use areas within the greater protected open space network.

An opportunity is emerging to expand upon local successes and encourage a new dialogue among agencies, conservationists, and recreationists, both at the local level and regionally, in support of the expanded protection of natural lands throughout California. We encourage interested parties to continue to learn more about the use of conservation planning tools and visitor use management made available through the CDFW and USFWS, and Interagency Visitor Use Management Council (Appendix I). Forming partnerships allows stakeholder groups to work together to plan ahead of growth and build regional conservation strategies for the increased protection of natural lands, addressing the long-term conservation needs of California's natural resources and the strong desire of people to experience nature.

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APPENDIX I: AVAILABLE CONSERVATION PLANNING AND VISITOR USE MANAGEMENT TOOLS

Natural Community Conservation Planning

The Natural Community Conservation Planning (NCCP) Program promotes collaborative planning efforts designed to provide for the region-wide conservation of plants, animals, and their habitats, while allowing for compatible and appropriate economic activity. https://www.wildlife.ca.gov/Conservation/Planing/NCCP

Regional Conservation Investment Strategy Program

The Regional Conservation Investment Strategy (RCIS) Program encourages a voluntary, non-regulatory regional planning process intended to result in high-quality conservation outcomes. The Program consists of three components: regional conservation assessments (RCAs), regional conservation investment strategies (RCISs), and mitigation credit agreements (MCAs). https://www.wildlife.ca.gov/Conservation/Planning/Regional-Conservation

Conservation and Mitigation Banking

Conservation and mitigation banking in California is overseen and undertaken by several Federal and State Agencies. The Banking Program coordinates with other agencies and stakeholders to develop statewide policy and guidance for the establishment and operation of conservation and mitigation banks. https://www.wildlife.ca.gov/Conservation/Planning/Banking

Biogeographic Information and Observation System (BIOS)

BIOS is a system designed to enable the management, visualization, and analysis of biogeographic data collected by the California Department of Fish and Wildlife and its Partner Organizations. https://www.wildlife.ca.gov/Data/BIOS

Areas of Conservation Emphasis (ACE)

ACE is a CDFW effort to analyze large amounts of map-based data in a targeted, strategic way, and expressed visually, so decisions can be informed around important goals like conservation of biodiversity, habitat connectivity, and climate change resiliency. https://www.wildlife.ca.gov/Data/Analysis/Ace

Visitor Use Management (VUM) Framework

VUM is a toolbox for visitor use management and addresses conservation issues. The framework also includes topic areas like capacity, indicators and thresholds, as well as the importance for monitoring recreation use. https://visitorusemanagement.nps.gov/VUM/Framework

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Recreation-related disturbance to wildlife in California – better planning for and management of recreation are vital to conserve wildlife in protected areas where recreation occurs

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Expanding levels of authorized and unauthorized non-consumptive recreation increasingly threaten sensitive biological resources in areas protected primarily or solely to conserve them. As California's human population grows, recreational use in protected areas grows commensurately. The majority of the documented effects on wildlife from non-consumptive recreation are negative; they include detrimental changes in behavior, reproduction, growth, immune system function, levels of stress hormones, and finally, to the survival of individual animals and persistence of wildlife populations and communities. This paper provides insights from the recreation ecology literature into these recreation-related disturbances to insects, amphibians, reptiles, birds, and mammals from hiking, jogging, biking, horseback riding, boating, and off-highway/all-terrain vehicles. The documented evidence of these disturbances to wildlife reveals the flaw in the prevalent assumption that recreation is compatible with biological conservation, the dual-role protected areas' core function. This assumption usually rests on the expectations of (1) allowing only ecologically sound siting of recreational areas and ecologically acceptable types, levels, and timing of recreation, and (2) providing sufficient monitoring, management, and enforcement of recreation to ensure the perpetuation of viable populations of focal sensitive species. However, it is rare that these expectations are met. The ultimate essential outcome of the information provided in this paper is the cessation of the extant recreation-related exploitation of dual-role protected areas. This calls for a societal course change involving: widespread, long-term, and continual multimedia dissemination of the science-based information about recreation-related disturbance to wildlife; application of a science-based approach to siting recreational areas and allowing only ecologically acceptable types, levels, and timing of recreation; and, perpetual personnel and funding explicitly for management at levels commensurate with recreational pressure. These measures would also improve the often cited economic, educational, and recreational/health benefits of dual-role protected areas.

Key words: dual-role protected areas, enforcement, fragmentation, management, multimedia education, non-consumptive recreation, perpetual funding, planning and siting of trail networks, recreational disturbance to wildlife, unauthorized trails

Conserving habitats is a key strategy for conserving biodiversity worldwide (Pickering 2010). In California, the core function of many areas protected for conservation is to ensure the perpetuation of sensitive species (i.e., species whose persistence is jeopardized), as is appropriate for the nation's most biologically diverse state (CDFW 2015). The level of land conservation that California enjoys is intended to ensure that the state's globally renowned biodiversity remains intact. However, of all the states in the USA, California hosts the most listed species imperiled by recreation, in part because the strongest association of outdoor recreation is with urbanization (Czech et al. 2000), which is itself an important cause of endangerment (Reed et al. 2014). The anticipated growth of the state's human population from approximately 38 million in 2013 to 50 million by mid-century with a commensurate increase in recreational demands in protected areas will likely increase the continual challenge of conserving the state's wildlife (CDFW 2015).[1, 2] The dual role of protected areas to conserve biodiversity and provide nature-based recreational and educational opportunities for millions of people rests on the assumption that non-consumptive recreation is compatible with wildlife conservation, despite documented evidence to the contrary (Reed and Merenlender 2008; Larson et al. 2016; Hennings 2017; Dertien et al. 2018; Reed et al. 2019).[3] Ecologically sound types, levels, timing, and siting of recreation, and perpetual management of recreation at or exceeding a level commensurate with the recreational pressure, are vital to ensure the perpetuation of viable populations of focal sensitive species in "dual-role" protected areas.[4, 5]

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[1] Protected areas include locally-owned lands (e.g., county and city reserves), state-owned lands (e.g., ecological reserves, wildlife areas, state parks), federally owned lands (e.g., national wildlife refuges, wilderness areas), and privately owned lands (e.g., conservation easements, conservancy lands, mitigation banks and lands). Here, the focus is on protected areas preserved primarily or solely for the perpetuation of sensitive species (e.g., ecological reserves, protected areas established pursuant to Natural Community Conservation Plans and/or Habitat Conservation Plans, mitigation banks and lands).

[2] Wildlife means all wild animals: insects, fish, amphibians, reptiles, birds, and mammals.

[3] In contrast to consumptive recreation (e.g., hunting, fishing), non-consumptive recreation is generally assumed not to directly extract a resource; it includes nature and wildlife viewing, beach-going, kayaking, hiking, biking, horseback riding, and wildlife photography (Reed and Merenlender 2008; CDFW 2016; Gutzwiller at el. 2017). From here forward, "recreation" means non-consumptive recreation, unless otherwise stated.

[4] Focal species are organisms whose requirements for survival represent factors important to maintaining ecologically healthy conditions; identified for the purpose of guiding the planning and management of protected areas in a tractable way, focal species include keystone species, umbrella species, flagship species, and indicator species (Soulé and Noss 1998; Marcot and Flather 2007). Here, the term "focal species" is intended to include those species encompassed by the guild surrogate approach of conservation; this approach entails one member or a subset of members serving as a surrogate for other members of the guild (Marcot and Flather 2007).

[5] From here forward, "management" includes monitoring, management, and enforcement with the necessary authority. The level of enforcement necessary is dependent on the level of continual management implemented; generally, the more the management, the less enforcement is necessary. In addition, monitoring and management encompass both the natural resources and human users of the protected areas. The fiscal support to be secured includes personnel and all program costs.

Insights from studies

Purpose.—The purpose of the following discussion is to provide insights to disturbances to several wildlife species from non-consumptive recreation. Accordingly, the insights are exclusively from studies that document recreation-related disturbance to wildlife. This approach reflects the evidence that the majority of documented responses of wildlife species to non-consumptive recreation are negative, as demonstrated in two systematic literature reviews (Reed et al. 2014; Larson et al. 2016) and a literature review of over 500 articles written and reviewed by the scientific community (Hennings 2017). The insights are intended to (1) illustrate that scientific studies provide clear evidence of recreation-related disturbance to wildlife, (2) elicit awareness of and concern about the disturbance, and (3) stimulate action to address it.

Sources and scope.—The 71 articles and 13 reports[6] reviewed about the recreation-related effects on wildlife generally reflect Larson et al.'s (2016) finding that studies about such effects focus on mammals (42%) and birds (37%), followed by invertebrates (12.4%), reptiles (5.5%), fish (5.1%), and amphibians (0.7%); there are no insights herein from studies of fish. Larson et al. (2016) found that some of the least-studied taxonomic groups (i.e., reptiles, amphibians, and invertebrates) had the greatest evidence for negative effects of recreation. While not all the studies selected for this paper address wildlife in California, all the studies' scenarios could occur in the state as do all species types among the studied taxa (i.e., insect, amphibian, reptile, bird, mammal).

Not all of the studies selected for this paper address sensitive species. This is primarily because current research on recreation-related effects on wildlife includes few species of conservation concern (Larson et al. 2016). However, sensitive species may experience greater levels of recreation-related disturbance than described for common species in the study insights herein. This is because many rare and isolated species are specialists, and they may be more sensitive to anthropogenic disturbance, including recreational activities, than common and widely distributed species (Bennett et al. 2013; Reilly et al. 2017). Recreation-related declines of common species warrant attention because of their functional ecological importance – local depletions of common species can have broad consequences within the food web (Säterberg et al. 2013; Baker et al. 2018; Reed et al. 2019). Recreation-related declines or disturbance in an important common prey species may affect the species in higher trophic levels (Reed et al. 2019). More than a quarter of species become functionally extinct before losing 30% of their individuals (Säterberg et al. 2013; Baker et al. 2018; Reed et al. 2019); here, functional extinction occurs when the population size of the depleted species is below the level at which another species goes extinct (Baker et al. 2018).

The scope of this paper does not include studies about snow-based recreation, though all of the 14 articles addressing snow-based recreation that Larsen et al. reviewed reveal that non-motorized and motorized snow-based activities (i.e., skiing, snowshoeing, snowmobiling) can have significant negative effects. Nor does the scope of this paper include studies exclusively about the effects of dogs on wildlife; however, a literature review on the effects of dogs on wildlife concludes that (1) people with dogs on leash, and even moreso

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[6] All the articles are published in peer-reviewed journals. Some of the reports were peer reviewed and all were written by or contributed to by professionals in the fields of biology or ecology, though none of the reports were published in peer-reviewed journals to this author's knowledge (e.g., Burger 2012; Hennings 2017; Dertien et al. 2018; Reed et al. 2019). This paper does not cite all the articles and reports this author read. And, the totals exclude documents that are not explicitly about recreation-related effects on wildlife (e.g., Tinkler et al. 2019; Taff et al. 2019; Wolf et al. 2019) and all newspaper articles.

off leash, are more alarming and detrimental to wildlife than any non-motorized recreational user group without dogs, and (2) people with dogs substantially increase the amount of wildlife habitat affected (Hennings 2016). Hennings (2016) also asserts that wildlife does not appear to habituate to the presence of dogs; effects linger after dogs are gone because the scent of dogs repels wildlife.

Management measures.—The study insights focus on the documented recreation-related disturbance to wildlife, not on management measures to prevent or minimize the disturbance. However, many of the reviewed articles and reports identify such measures, which range from full prohibition of human access, to time-of-access restrictions (e.g., seasonal or diurnal/nocturnal restrictions), to various measures based on disturbance thresholds. Disturbance thresholds are thresholds of various measurable parameters above or below (depending on the parameter) which wildlife is disturbed. Examples of disturbance thresholds are distance between trails and nesting sites, density of active trails, number of recreationists, number of recreational events per time frame, and duration of recreation. These thresholds may be used to establish management measures such as minimum widths of spatial buffers between recreational trails and wildlife.

A common theme among the management measures is that continual proactive and adaptive management is needed to protect wildlife from recreational disturbance, and that access closures should occur if the management fails.[7] Adaptive management is a cornerstone of large-scale multi-species conservation (CDFW 2014). An example of proposed management measures is Dertien et al.'s (2018) recommendation for a precautionary approach that adopts maximum values of quantitative disturbance thresholds observed for the taxa of concern, while excluding the extreme values of the thresholds.[8] This approach stems from the gaps in knowledge about quantitative disturbance thresholds of recreation; such thresholds are lacking for many species, taxonomic groups, and sources of disturbance.

Regarding spatial buffers, a general rule of minimum thresholds for distance to trails cannot be established for some species, as individual variability within species can be high and can differ among populations, types of topography, and frequencies and types of human intrusion (González et al. 2006). For example, Dertien et al. (2018) recommended a 200-m minimum buffer for ungulates; however, this would be insufficient for the circumstances of Taylor and Knight's (2003) study further cited below in which they found that mule deer (*Odocoileus hemionus*) showed a 96% probability of flushing within 100 m of recreationists located off trails, and the probability of their flushing did not drop to 70% until perpendicular distance reached 390 m. Two additional factors that influence the determination of spatial buffers are "effect zones" (i.e., areas within which wildlife is disturbed by recreational ac-

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[7] Based on section 13.5 of the California Fish and Game Code and the Natural Community Conservation Planning Act (i.e., section 2805 of Fish and Game Code), adaptive management generally means (1) improving management of biological resources over time by using new information gathered through monitoring, evaluation, and other credible sources as they become available, and (2) adjusting management strategies and practices accordingly to assist in meeting conservation and management goals (e.g., conservation of covered or focal species). Under adaptive management, program actions are viewed as tools for learning and to inform future actions.

[8] The central tenet behind the precautionary principle is that precautionary measures should be taken even if some cause and effect relationships are not fully established scientifically. Generally, the four central components of the principle are: taking preventive action in the face of uncertainty; shifting the burden of proof to the proponents of an activity; exploring a wide range of alternatives to possibly harmful actions; and increasing public participation in decision making (Kriebel et al. 2001). There are subtle differences between the precautionary principle and precautionary approach, but their consideration is beyond the scope of this paper.

tivities on trails) and the density of the trail networks. The effect zones can extend several hundred meters on either side of the trails (Reed et al. 2019). The smaller a protected area is and the denser its trail networks are, the greater the proportion of the protected area is occupied by effect zones, and the less likely it is that spatial buffers such as those Dertien et al. (2018) recommended will protect the focal species from recreational disturbance (Wilcove et al. 1986; Ballantyne et al. 2014).

There are many sources that provide information about management of recreation in protected areas, or guidance on the design or siting of trails/trail networks. These sources include management framework tools designed to address recreational use, though they vary in their attention to the needs of wildlife (Hennings 2017).

Insects

In a study of the effects of walkers, runners, and runners with dogs on the federally endangered Karner blue butterfly (*Lycaeides melissa samuelis*; Karners) at the Indiana Dunes National Lakeshore, USA, Bennett et al. (2013) found that (1) Karners flushed in the presence of recreationists as they would respond to natural agents, such as predators; (2) recreation restricted host-plant choice by reducing host-plant availability, effectively rendering the quality of habitat within 10 m of the trail unsuitable; (3) recreation had the potential to reduce oviposition rate of virtual females by 50%, and therefore population growth rates; (4) the frequency at which recreationists negatively affected the females (including their oviposition) varied substantially with habitat extent, number of recreationists, and sensitivity; and (5) habitat extent was the primary predictor variable. The authors concluded that Karners will experience less recreation-related disturbance the farther their habitat extends beyond trails.

In a study conducted near Palo Alto, California, USA focusing on 10 native oak woodland species of butterflies, Blair and Launer (1997) concluded that even small perturbations by hikers and joggers in a recreational area led to (1) a loss in the number of butterfly species (species richness) of the original oak-woodland community compared to the number of these species in a biological preserve with no recreation, and (2) a lower number of butterflies (abundance) in the recreational area compared to the biological preserve. The authors also concluded that multi-use areas may not adequately preserve butterfly species diversity.

Herpetofauna

Responses of the Iberian frog to recreational activities.—In a study involving field research in the Guadarrama Mountains in central Spain and simulation modelling to assess the effects of recreation on Iberian frogs (*Rana iberica*), an endemic species in decline, Rodríguez-Prieto and Fernández-Juricic (2005) measured frog abundance and response to human disturbance. The authors found that Iberian frog abundance (a population-level parameter): (1) was significantly affected mainly by study site location and distance to the nearest recreational area, a proxy for human disturbance; (2) was positively related to distance from recreational area (i.e., as distance decreased, abundance decreased); and (3) increased as number of humans decreased. With respect to the effects of repeated disturbances (e.g., human approaching with a steady pace) on the individual-level parameters of flight initiation distance[9] and time to resume pre-disturbance activities, the study showed that: (1) frogs' flight initiation distances were longer in areas with less vegetation cover; (2) though the flight initiation distances did not vary with repeated human approaches, the number of repeated human approaches affected the frogs' time to resume pre-disturbance activities, with second and third approaches increasing the time it took frogs to reoccupy the disturbed spot; and (3) there was an 80% decrease in the frogs' stream-bank use with a 5-fold increase in the direct disturbances per hour, and a 100% decrease in stream bank use with a 12-fold increase in human disturbances per hour. The authors concluded that direct human disturbance affects this species at the population level, and that it needs to be considered as a potential factor affecting amphibian populations with low tolerance for disturbance.

Responses of the yellow-blotched map turtle to human disturbance.—In a study along a 300-m reach of the Pascagoula River in southeastern Mississippi, USA, Moore and Siegel (2006) studied the effects from boating, fishing, jet skis, and direct anthropogenic damage to nests on the nesting and basking behavior of the yellow-blotched map turtle (*Graptemys flavimaculata*), listed as threatened under the U.S. Endangered Species Act. With respect to human disturbance of nesting turtles, the authors found that numerous turtles waited several hours near a sandbar before emerging from the water onto the beach to nest, and turtles that attempted to nest upon emerging onto the beach frequently abandoned their efforts and retreated to the water—of a total of 79 nesting attempts, only 15 successfully completed oviposition. With respect to human disturbance of basking turtles, the authors found that the number of turtles disturbed differed significantly with the type of disturbance; specifically, anglers that remained in the basking vicinity caused the most disturbance, and jet-skis caused less than an expected amount of disturbance; this was likely because of the anglers' closeness (compared to the jet-skis) to the basking logs and the long periods they remained, both of which caused turtles to bask less. Moore and Siegel (2006) concluded that: the interruption of nesting activities may have a severe impact on the viability of this population of turtles through changes in numbers of clutches; and, the interruption of basking and consequent reduction in the turtles' body temperature has the potential to negatively affect the ability of all turtles to process and digest food, and the ability of females to develop eggs during the reproductive seasons.

Responses of the common wall lizard to tourism.—In a study of common wall lizards (*Podarcis muralis*) conducted in areas with high and low levels of tourism within the same habitat in the Guadarrama Mountains in central Spain, Amo et al. (2006) examined whether the lizards differed in several parameters upon each human approach. The authors found that: (1) regardless of the level of tourism, lizards usually exhibited anti-predator behavior by fleeing to hide in refuges upon approach of a human; (2) in comparison to lizards inhabiting areas of low tourism pressure, lizards inhabiting areas with high tourism pressure, and therefore presumably escaping to hide in refuges more often, showed a poorer body condition and higher intensity of tick infection at the end of the breeding period; and (3) the intensity of tick infection was higher in male than in female lizards. The authors speculated that the higher intensity of infection probably resulted from the cumulative costs of high frequency of flight, since anti-predatory behaviors such as flight are costly in terms of losing time for other activities, including feeding—nutritional status can affect the capacity

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[9] The flight initiation distance is the distance from an approaching threat (e.g., recreationist) at which an animal initiates moving away to escape from the threat. This movement is a fitness/energy cost to the fleeing animal. For the Iberian frogs, this was the distance between an approaching human and the frog when the latter jumped into the water in response to the human's approach.

of lizards to mount an immune response to infection. Furthermore, lizards with poor body condition had low levels of immune response, which may aggravate the deleterious effects of anti-predatory behavior on body condition. Female lizards in poor body condition produced offspring of small size, and body size of infant lizards can affect their probability of survival. Additionally, females with blood parasites also showed reduced fat stores and produced smaller clutches. By these effects on infants and clutch sizes, tourism may also negatively affect the maintenance of lizards' populations.

Responses of various reptiles to recreationists.—In a study to systematically assess recreationists' direct and indirect effects on sensitive wildlife species in 14 NCCP/HCP protected areas in San Diego County, California, USA, Reed et al. (2019) integrated monitoring of both wildlife species and recreationists (e.g., hikers, mountain biker, horseback riders).[10] The authors found that recreation was associated with declines in reptilian species' richness, occupancy, habitat use, and relative activity in the NCCP/HCP protected areas. Of the three species (all lizards) for which statistical analyses were feasible, two exhibited negative relationships between occupancy and human recreation—the orange-throated whiptail (*Aspidoscelis hyperythra beldingi*, an NCCP/HCP-covered species) and common side-blotched lizard (*Uta stansburiana*).

Birds

General responses.—In Steven et al.'s (2011) review of 69 peer-reviewed articles (50 of which were research conducted in protected areas) of original research on the effects on birds from non-motorized nature-based recreation, 61 articles reported recreation as having negative effects (i.e., negative changes in physiology, behavior, abundance, and reproductive success, the latter including the number of nests, eggs laid, and/or chicks hatched or fledged). The single documented positive effect involved an increase in the abundance of corvids (e.g., crows and ravens) in campgrounds. Walking or hiking, standing or observing birds from viewing platforms or standing next to a nesting colony, dog walking, running, cycling/mountain biking, and canoeing were all reported as negatively affecting birds. A large majority (85–93 %) of the studies that examined the effects of a single person, groups of two or more people, and/or avian population-level responses, reported negative effects. The population-level responses entailed effects on density, abundance, and reproduction.

In a study using data collected in 112 urban parks throughout Melbourne, Australia, Bernard et al. (2018) tested whether birds responded differently to bikers and walkers. They found that: (1) relative to their response to walkers, four of the 12 focal species studied initiated escape from bikers at longer flight initiation distances and two escaped with greater intensity (i.e., more likely to involve flying); (2) no species responded less to bicycles than to walkers; and (3) the flight initiation distance did not differ in response to speed of bicycle travel, though the difference in the two speeds used was only 1 m/sec. In concluding that

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[10] An NCCP (Natural Community Conservation Plan) is a comprehensive, single- or multi-jurisdictional/utility plan that provides for regional habitat and species conservation at an ecosystem level while allowing local land use authorities to better manage growth and development. Upon issuing an NCCP Permit, the California Department of Fish and Wildlife (CDFW) can authorize take of selected state listed species and other species of concern, subject to the terms of coverage under the NCCP (CDFW 2015). An HCP (Habitat Conservation Plan) is the federal counterpart to an NCCP; the U.S. Fish and Wildlife Service prepares HCPs and issues HCP permits. The terms and conditions under which an NCCP/HCP's protected areas are conserved establish the types and levels of public access that are permitted (Burger 2012). The types and levels of public access vary among the NCCP/HCP protected areas from no access to guided-only access to open access. The species protected by NCCPs/HCPs are typically called covered species.

bikers can appear more or less threatening to birds than a single pedestrian, Bernard et al.'s (2018) results underscore that the responses of wildlife to recreational activities vary among species, sites, types of recreation, and exposure over time to the activities.

Songbirds.—Davis et al.'s (2010) study of the effects of mountain biking on golden-cheeked warblers (*Dendroica chrysoparia*, warblers) with nests near biking trails in the Fort Hood Military Base in Killeen, Texas, USA, and the Balcones Canyonlands Preserve in Austin, Texas, found direct and indirect effects. The direct effects included warblers flushing >20 m in response to encounters with passing mountain bikers. Indirect effects included abandonment of nests <2 m from the biking trails and a reduction in the quality of nesting habitat due to biking-related fragmentation and alteration of habitats. In comparison to the control sites, it was likely that habitat fragmentation resulting from trails in the biking sites caused the increased predation of warbler nests by rat snakes (*Elaphe obsoleta*) and other edge-adapted predators. The authors speculated that the biking sites, which were able to maintain viable populations of warblers at the time of the study, may not continue to do so with additional recreational use, fragmentation, and alteration of the habitats.

Forest birds.—Bötsch et al. (2018) examined how breeding-bird communities changed with distance to trails in four broad-leafed and mature forests in Switzerland and France; the forests were similar in size, structure, and trails, but widely different in levels of recreation (mostly walkers). The authors found that: in the forests with high levels of recreation, the density and species richness of birds decreased by 12.6% and 4.0%, respectively, at points close to trails compared to points farther away; cavity, ground, and open-cup nesters had fewer territories and species close to trails compared to farther away; and, above-ground foragers and ground foragers showed a similar pattern. None of these effects on density, species richness, nesting guild, or foraging guild occurred in the forests with low levels of recreation. Both high- and low-sensitivity species (i.e., long versus short flight initiation distances) had fewer territories and fewer species close to versus far from trails in forests with high levels of recreation; however, in forests with low levels of recreation, highly sensitive species exhibited only a slight tendency for fewer territories close to trails. The authors inferred from their findings that (1) human presence in forests disturbs avian community composition and abundance along trails in recreational areas, (2) the overall effect of recreational trails themselves depends mainly on recreational intensity and only slightly on species characteristics, and (3) the observed effects on birds in forests where recreation has occurred for decades suggest that habituation to humans has not outweighed the effects.

Raptors.—In a study along the Boise River in Idaho, USA, examining flight initiation distances of bald eagles (*Haliaeetus leucocephalus*) in response to actual and simulated walkers, joggers, anglers, bikers, and vehicles, Spahr (1990) found that the highest frequency of eagle flushing was associated with walkers, followed by anglers, bikers, joggers, and vehicles. Eagles were most likely to flush when recreationists approached slowly or stopped to observe them, and were less alarmed when bikers or vehicles passed quickly at constant speeds. However, the longest flight initiation distance was in response to bikers, followed by vehicles, walkers, anglers, and joggers. Hennings' (2017) literature review provides the following about bald eagles: pedestrians within 275 m caused a 79% eagle response rate; eagles did not resume eating for four hours after disturbance by walkers; a suggested minimum 600-m buffer around breeding eagles, beyond which response frequency dropped below 30%; an apparent threshold of about 20 daily recreational events after which eagles were slow to resume feeding, and after 40 events, feeding was uncommon; sub-adults were less tolerant of disturbance than adult eagles; and recreation-related long-term effects can include reductions in survival, particularly during winter and especially for juveniles.

With respect to the tolerance (through habitat imprinting, genetic inheritance, or habituation) of golden eagle (*Aquila chrysaetos*) for recreational disturbance, Pauli et al. (2017) used an individual-based model[11] to assess the effects of walkers and off-highway vehicles on golden eagle populations. The primary modeling results indicated that, while golden eagles can develop tolerance for recreational disturbance, tolerance for even moderate levels of disturbance may not develop within a population at a sufficient rate to offset the effects of increased recreation on breeding golden eagles, particularly because this is a long-lived species with low recruitment. Pauli et al. (2017) conclude that, taken together, the simulation results suggest that recreation-related disturbance has a substantial effect on golden eagle populations and that increased recreation activity will exacerbate such effects. Given the results and the fact that non-motorized recreation decreases the probability of egg-laying in golden eagles (Spaul and Heath 2016), the authors asserted that trail management and a reduction in recreation activity within eagle territories are necessary to maintain golden eagle populations in locations where levels of recreation are increasing.

Shorebirds.—In a controlled study conducted in Scotland of the behavioral responses of the ruddy turnstone (*Arenia interpres*) to an approaching human, Beale and Monaghan (2004) found that birds supplemented with food flushed sooner from the human and searched for predators more frequently than birds not supplemented with food. That is, birds responding most were actually the least likely to suffer any fitness consequences associated with the disturbance. This study demonstrates the possibility of misconstruing the reasons for and implications of observed responses among all wildlife species. Traditionally and intuitively, species that readily flee from or avoid human disturbance are considered to be the most in need of protection from disturbance. However, species with little suitable habitat available nearby cannot show marked avoidance of disturbance even if the costs of reduced survival or reproductive success are high, whereas species with many nearby alternative sites to move to are likely to move away from disturbance even if the costs of the disturbance are low (Gill et al. 2001). It should not be assumed that the most responsive animals are the most vulnerable (Beale and Monaghan 2004). Gill et al. (2001) asserted that the absence of an obvious behavioral response does not rule out a population-level effect. In the same vein, it may be that species occurring in protected areas that are remnant fragments within urban landscapes are forced to utilize all components of the fragments, irrespective of their land-use intensity and land cover. This may occur if animals have nowhere else to go, and may be an explanation for instances when the relative abundance of birds is greater in urban and suburban reserves than in exurban reserves (Markovchick-Nicholls et al. 2008).

Mammals

General responses within NCCP/HCP protected areas in southern California.—In series of three studies about the responses of mammals to hikers and runners, bikers, horseback riders, dog walkers, and motorized vehicles, George and Crooks (2006), Patten et al. (2017), and Patten and Burger (2018) analyzed camera-trap data captured throughout areas protected under the 1995 County of Orange Central and Coastal NCCP/HCP (Orange County NCCP/HCP). All studies analyzed bobcat (*Lynx rufus*), coyote (*Canis latrans)*, and mule

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[11] Individual-based models are simulation statistical tools that use empirical data to examine effects, such as anthropogenic population-level effects, that are difficult or impossible to study in a field setting.

deer, and Patten et al.'s (2017) analysis also considered mountain lion (*Puma concolor*), gray fox (*Urocyon cinereoargenteus)*, striped skunk (*Mephitis mephitis*), and northern raccoon (*Procyon lotor*). The authors found that: (1) mammal detections were negatively correlated with all types of recreationists; hikers and runners had the greatest negative association with wildlife, and equestrians had the least; (2) the overall trend is sharply negative: as human activity increased, mammalian activity decreased, regardless of species, type of human activity, or camera placement; (3) mammals were nearly four times as likely to be recorded on days with no human activity than on days with human activity at the same site; (4) detections of mammals decreased incrementally as the number of humans increased within a day, and fell to near zero probability at $\geq$60 humans per day; and (5) all seven species listed above exhibited short-term spatial displacement in response to events with more than 100 visitors.

Bobcats' negative associations were strongest with bikers, hikers, and domestic dogs. In areas of higher human activity, bobcat were detected less frequently along trails and appeared to show temporal displacement, becoming more nocturnal. Coyotes' overall activity was lower at the sites with the most recreation and was negatively associated with overall human, hiker, and biker visitations; and, a trend of temporal displacement in response to dogs was also evident. Generally, both bobcats and coyotes displayed a relatively wide range of activity levels at sites with low human use, but a lower and markedly restricted range of activity at those sites with the highest levels of recreation. Both coyotes and mule deer shifted their activities temporally over the long term. The mule deer's (a primary consumer) marked shift brought it into closer temporal alignment with its main predator (mountain lion) and the coyote's marked shift (secondary consumer) brought it into closer temporal alignment with a chief prey species (gray fox). These human-induced diel shifts involving animals in two trophic levels have important ramifications for predator–prey dynamics. Despite these studies' results, no evidence was found suggesting mammalian populations have declined in the Orange County NCCP/HCP protected areas between 2007 and 2016, even as human activity increased markedly across the study period. However, it is critical to consider this observation in light of: (1) the fact that, at least for the years 2007-2011, public access was controlled across most of the study area by permit-only entry, regular docent-led programs, and monthly self-guided wilderness access days—much higher levels of restrictions on public access than for most protected areas; (2) the authors' assertion that various mammalian species' avoidance behavior may yet drive mammalian populations downward upon further increase in human disturbance; and (3) the status of the Vail Colorado elk herd as recounted below—once a herd of 1,000 head diminished to 53 due to steadily increasing levels of recreation.

Overall, the results of the above three studies were similar to those of a study to assess recreationists' effects on sensitive wildlife species in 14 NCCP/HCP protected areas in San Diego County, for which Reed et al. (2019) used data from camera traps and a before-after-control-impact (BACI) experiment. Reed et al. found that bobcat, gray fox, mule deer, and northern raccoon were less active in areas with higher levels of human recreation. Bobcat habitat use was more strongly negatively associated with human recreation than urban development, which also decreased the probability of habitat use. The collective results for mule deer among the four studies suggest that mule deer may stop using some areas altogether if human recreation is too high. Reed et al. (2019) did not detect negative associations between human recreation and the habitat use or relative activity of the six following mammalian species of the 12 observed: coyote, striped skunk, ground squirrel, jackrabbit, brush rabbit

(*Sylvilagus bachmani*), and desert cottontail (*S. audubonii*). However, of special note are results from the protected area with the highest level of recreation (i.e., an average of 1,797 people per day) observed in the study, where the cameras captured only rabbits, and no other mid- to large-bodied wildlife species during 7.5 weeks of monitoring. Yet, this 2,449-ha protected area is considered a core biological area and regional wildlife corridor targeted for conservation (City of San Diego 2019). The BACI experiment conducted in another protected area showed a significant decrease in bobcat detection probability in a four-week period following a trail re-opening, suggesting that this species can modify its behavior (e.g., shift its activity patterns) rapidly after a change in human recreation. This is evidence that temporal closures have the potential to reduce disturbance during critical periods for some species. Although human recreation may not often extirpate mammalian species from urban habitat fragments, it can reduce habitat suitability and carrying capacity (Reed et al. 2019).

Responses to human voice.—Suraci et al. (2019) tested whether mammalian carnivores' responses to human voices alone can result in landscape-scale effects across wildlife communities, including cascading effects on the behavior of lower trophic level animals. The results of the study, which was conducted in the Santa Cruz Mountains of central California, USA, indicate that human voice alone does result in such effects. Where humans are absent or rare, large and medium-sized carnivores exhibit greater movement, activity, and foraging, while small mammals use less space and forage less. Where humans are present, the activity, foraging, and/or habitat use of large and medium-sized carnivores are suppressed, while small mammals increase their total space use and foraging intensity. The implications of these results are far-reaching, and include that, even in the absence of land development or habitat fragmentation, increased human presence can: (1) affect large carnivore movement, which could eventually limit carnivores' hunting and feeding behavior or force individuals to abandon high risk areas of their home range; (2) suppress activity of medium-sized carnivorous species; and (3) increase the abundance of small mammals that are prey to the large- and medium-sized predators, which could ultimately increase the abundance of small mammals in wildlife areas people visit (Suraci et al. 2019, citing other authors). Moreover, if the sublethal effects observed in the study in response to human voices alone are comparable to those effects (e.g., increased physiological stress, reduced reproductive success) that fear has been demonstrated to cause in predator-prey systems, they may amount to additional widespread but largely unmeasured effects of humans on wildlife populations (Suraci et al. 2019, citing other authors). Hennings (2017) provides additional insights about, and citations for studies on, the effects on wildlife from the human voice, concluding that conversational noise along trails can be very disturbing to wildlife.

Ungulates.— In a two-year study of elk (Cervus elaphus) in a herd near Vail in central Colorado, USA, Shively et al. (2005) found that elk reproductive success rebounded to pre-disturbance levels after the cessation of their exposure to back-country hikers during the calving season over the previous three years. Shively et al. concluded that, it seems prudent to protect elk during calving seasons, because, although the study provides evidence that elk reproduction can rebound from depressed levels when human disturbances are removed or reduced, there had been a linear decline in calf production in response to increasing levels of disturbance compared to controls without such disturbance, and it is not known if there is a threshold level of reproductive depression from which elk cannot recover. Recognizing that it is seldom easy to curb human activities that have become traditional, or to restore wildlife habitats once they have been developed, they recommended the continuation of some closures imposed on parts of both the Vail and control elk herd study areas. However, a recent article in The Guardian reported that the number of elk in this same Vail herd dropped precipitously since the early 2010s with the steady increase in human recreation; once a herd of 1,000 head of elk, it had decreased to 53 at last count in February of 2019. The article explains that, for Bill Alldredge, one of the authors of the 2005 study, there is no other explanation than the increased levels of hiking, biking, and skiing in the area that supports this elk herd (Peterson 2019). This outcome adds to the already ample evidence that pregnant animals or those with young—especially mammals—are particularly sensitive to human disturbance (Hennings 2017).

In a study subjecting 13 captive female elk in the Starkey Experimental Forest and Range in Oregon, USA, to four types of recreational disturbances (all-terrain vehicles [ATV] riding, mountain biking, hiking, and horseback riding), Naylor et al. (2009) recorded the elk's resting, feeding, and travel times in response to the disturbances. The authors found travel time (a proxy for energy expense) increased in response to all four disturbances and was highest in mornings. The authors suggest that the elk's lesser response to each disturbance in afternoons was likely due to elk moving away from the disturbances in the mornings and avoiding them for the remainder of the day. Elk travel time was highest and feeding time lowest during ATV exposure, followed by exposure to mountain biking, hiking, and horseback riding. Resting decreased with exposure to mountain biking and hiking disturbance, and elk showed no evidence of habituation to mountain biking or hiking.

In a study of how bison (*Bison bison*), mule deer, and pronghorn (*Antilocapra americana*) responded to hikers and bikers on designated recreational trails at Antelope Island State Park in Great Salt Lake, Utah, USA, Taylor and Knight (2003) found the following: with respect to alert distance, flight initiation distance, and distance moved,[12] there was little difference in how each species responded to hikers versus mountain bikers (with an exception of mule deer flight distance), though each species exhibited its own degree of response in the three parameters tested; and all three species exhibited a 70% probability of flushing from on-trail recreationists within 100 m from designated trails. Trials were also conducted with only mule deer along a randomly chosen, off-trail route to assess the response of mule deer to hikers or bikers off designated trails. From these trials, the authors found that mule deer showed a 96% probability of flushing within 100 m of recreationists located off trails, and the probability of their flushing did not drop to 70% until perpendicular distance reached 390 m. There was little evidence of habituation to recreationists among the species at the time of the study. In fact, the pronghorn at the study site did not habituate to largely predictable recreational use over a three-year period following the opening of trails at the site, and used areas that were significantly farther from trails than they had prior to the start of recreational use.

Carnivores.—In a study of mammalian carnivores in 28 protected areas located in oak woodlands in northern California, USA, Reed and Merenlender (2008) found the following about carnivores' responses to recreationists. Generally, in paired comparisons of neighboring protected areas with and without recreation, the presence of dispersed, non-motorized recreation (hiking, biking, and horseback riding) led to a five-fold decline in the

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[12] Alert distance is the distance from a stimulus at which an animal initiates vigilance behavior; more specifically in this context, it is the distance between a recreationist and an animal when the animal first becomes visibly alert to the recreationist. Flight initiation distance is defined in footnote #9. Distance moved is the distance an animal travels from its initial position until it stops (Taylor and Knight 2003).

density of native carnivores and a substantial shift in community composition from native to nonnative species. Specifically, a higher mean number of native species was detected in protected areas that did not permit recreation. By contrast, in protected areas that permitted recreation, more nonnative species were detected, domestic dogs were detected more frequently, and densities of coyotes and bobcats were more than five times lower. The authors concluded that the key variable for moderately sized protected areas (50–2000 ha) near urban development seems to be whether or not the site is open to public access.

In a study within three protected areas in Arizona, USA, Baker and Leberg (2018) found the following about how 11 mammalian carnivore species respond to varying levels of hiking, horseback riding, and border patrol activity. The study sites with the highest levels of human activity had significantly lower carnivore diversity, higher occupancy of common species (coyote, gray fox, and bobcat), and lower occupancy of all other carnivorous species. Generally, rare carnivores (e.g., mountain lion and kit fox, *Vulpes macrotis*), badgers (*Taxidea taxus*), and gray foxes avoided trails, whereas common species (except gray fox) preferred trails. Overall, edges of protected areas appeared to negatively affect occupancy of nearly all the study's species, and the presence alone of roads and trails, and not necessarily how much they are used, has a significant negative effect on the occupancy of most carnivorous species. In general, coyotes and bobcats were the carnivores least sensitive to human disturbance, gray foxes had a moderate negative association with human disturbance variables, and smaller carnivores and mountain lions seemed to be exceptionally vulnerable to human disturbance. Furthermore, the higher the level of overall disturbance in a protected area, the more sensitive carnivores were to disturbance variables.

Conclusions and Suggestions

With the expanding recreation-related disturbance to wildlife in protected areas, their dual role of conserving biological resources and providing nature-based recreational and educational opportunities for people presents a continual challenge to land managers and a continual threat to wildlife and the state's biodiversity, particularly sensitive species. The scientific literature provides clear evidence that recreation can disturb wildlife in several ways. Documented effects include detrimental changes to behavior, reproduction, growth, immune system function, levels of stress hormones, other physiological effects, and finally, the survival of individual animals and persistence of wildlife populations and communities. Having been observed on nearly every continent and in every major ecosystem on earth, recreation-related disturbance to wildlife is increasingly recognized as a threat to global biodiversity, and as having wide-ranging and, at times, profound implications for wildlife individuals, populations, and communities (Dertien et al. 2018). Yet, a prevalent assumption exists that non-consumptive recreation is compatible with wildlife conservation; sources that articulate this assumption in various ways include but are not limited to the Natural Community Conservation Plans/Habitat Conservation Plans (NCCPs/HCPs in the California Department of Fish and Wildlife's (CDFW) South Coast Region, Title 14 of the California Code of Regulations (§630(a)) about CDFW's ecological reserves, CDFW's 2016 State Wildlife Action Plan's Consumptive and Recreational Uses Companion Plan, Burger 2012, Larson et al. 2016, Dertien et al. 2018, and Reed et al. 2019. This assumption underlies the widespread acceptance of non-consumptive recreation in dual-role protected areas.

Is the assumption of compatibility flawed?—The assumption of compatibility rests on four expectations, which are often legal obligations (as with NCCPs/HCPs). First, recreation in protected areas is to occur only in ecologically sound locations. Second, only ecologically sound types, levels, and timing of recreation are acceptable. Third, monitoring is expected to regularly and reliably assess whether the types and levels of recreational activities in protected areas are disturbing the focal species to a degree that these activities should be curtailed or prohibited entirely. Fourth, changes in management are to occur promptly when monitoring determines them to be necessary (see footnote #5 for description of management). In short, the overarching expectation is that recreation would not hinder the achievement of the dual-role protected areas' primary conservation objective (i.e., perpetuation of viable populations of focal sensitive species). At least seven NCCPs/HCPs in the CDFW's South Coast Region explicitly deem recreation compatible or conditionally compatible; most articulate these expectations as conditions that recreational activities in protected areas must meet. Such activities are considered "conditionally compatible" with the protection of the covered species. However, the assumption of compatibility is flawed because: for example, designated trails and trail networks are often ecologically inappropriately planned, designed, or sited; and, even for authorized recreation, there is rarely adequate management to control the allowed types and levels of recreation such that they are compatible with conservation. While finding an appropriate balance between biodiversity conservation and recreation is complicated because recreation-related effects on wildlife vary among species and recreational activities (Larson et al. 2016), there are also societal factors at play that further complicate achieving an appropriate balance and compatibility.

Factors allowing inappropriate planning/siting and inadequate management - a societal conundrum.—The degree to which the above-listed expectations are met varies among NCCP/HCP permittees and other managers of dual-role protected areas, the primary limiting factors being fiscal constraints and each land manager's primary mission. As to the latter factor, for areas protected primarily or solely to conserve biological resources, a serious fundamental conflict with conservation arises when managers' primary mission is to provide recreational opportunities, and the protection of biological resources is a secondary or tertiary priority. As to fiscal constraints, land management budgets generally have not kept pace with the increasing levels of recreation in protected areas (CDFW 2015; Havlick et al. 2016). For example, the activities of the CDFW for resource assessment, conservation planning, and wildlife conservation at risk are "severely underfunded;" in 2005, maintenance, restoration, and management of CDFW's wildlife areas and ecological reserves were supported, on average, at the level of $13 per acre (0.40 ha) and one staff person per 10,000 acres. Many lands were operated at $1 per acre, with no dedicated staff (CDFW 2015—refer to Volume 1, Section 7.3). CDFW's fiscal shortfalls for managing its protected areas mirror the same among public agencies at the local, state, national, and international levels (CDFW 2015); these shortfalls result in continual grave shortages of management personnel and other resources.

California's State Wildlife Action Plan (CDFW 2015) and most of the literature about recreation-related ecological effects identify the economic, educational, and recreational/health benefits of protected areas. They also identify the benefits (e.g., economic) to protected areas from humans pursuing recreational activities. So, despite the documented recreation-related disturbance to wildlife, there seems to be an implicit assumption of a mutually beneficial relationship between protected areas and the humans who benefit from them. But, the severe underfunding of management for protected areas renders mutual reciprocity in this relationship infeasible; the protected areas' wildlife are heavily on the losing side. This is particularly perplexing given the evidence that lack of adequate management negatively affects not only biological resources, but also societal benefits.

Regarding the human health benefits of protected areas, visible recreation-related damage to the terrain diminishes the level of benefit people enjoy while being in nature, as illustrated by a study examining the relationship between recreational impacts in protected areas and human mental/emotional states (Taff et al. 2019). The study's results demonstrate that, as visible recreation-related ecological impacts increased, sense of wellbeing and mental state decreased, especially in response to settings with unauthorized trails. Collectively, the results show that managing tourism in protected areas in a manner that reduces such impacts is essential to providing beneficial cultural ecosystem services related to human health and wellbeing (Taff et al. 2019). As Wolf et al. (2019) put it, the more attractive a site is, the more likely it is that it will be degraded, which in turn, may diminish the quality of the human experience, and thus, visitor satisfaction. To capitalize fully on the positive aspects of tourism (including recreation) for protected areas, the degradation of resources needs to be constrained to ecologically acceptable levels, and to levels beyond visitor perception (Davies and Newsome 2009; Wolf et al. 2019); otherwise, recreationists may think it unimportant to minimize their own impacts. Also diminishing the human experience are the closures to public access as a default reaction to lack of adequate management, and the liability resulting from injuries that can occur when people use unauthorized trails (Dertien et al. 2018).

There is a two-fold irony here: despite the prevalent emphasis on the societal benefits of protected areas and the purported reciprocal relationship between protected areas and humans, most agencies responsible for managing protected areas are chronically underfunded. And, promoting the pursuit of these societal benefits without protecting the dual-role protected areas' core function (biological conservation) from that pursuit actually undermines both the human experience and biological conservation. This is a societal conundrum that stems at least in part from a societal disconnection.

The factor of a societal disconnection.—A lack of public interest in and concern about protected areas figures into the societal conundrum. Public opposition to trail closures, caps on daily visitation, or reservation systems can be strong and could damage the support for conservation agencies and organizations (Reed et al. 2019), despite the ecological need for such measures for protected areas. A disconnection pervades our society with respect to recreation-related disturbance to wildlife (Marzano and Dandy 2012): 50% of 640 backcountry trail users surveyed in 2001 did not believe that recreation negatively affects wildlife, and recreationists generally held members of other user groups responsible for stress or negative effects on wildlife rather than holding members of their own recreational user group responsible (Taylor and Knight 2003). The results of a survey conducted in 2018 for the San Diego End Extinction (SDEE) initiative to elucidate what the San Diego public know, think, feel, and do in relation to species and habitat conservation, indicate that 71% of the 600 respondents are not knowledgeable about the problems San Diego's plants and wildlife face (Tinkler et al. 2019).[13] While the passage of California Proposition 68 in 2018 reflects the voters' broad support for clean water and access to open space, which were the main elements of the Proposition that promotional efforts emphasized, it is unclear how

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[13] The respondents were San Diego County voters and were representative of the voter pool in terms of age, gender, ethnicity, and region, but voters tend to be less ethnically diverse and more educated than the San Diego County population overall (Tinkler et al. 2019).

much the biological conservation-related elements of the Proposition influenced voters.

Overall, it is probable that a large majority of the general public are unaware of or in denial about the disturbance to wildlife from non-consumptive recreation, much less the distinctions between areas protected primarily or solely for conservation and areas otherwise designated as open space (e.g., recreational fields, golf courses, small community parks). Information on these topics is not widely available, and what is in the literature, may not be reaching a broad audience even among conservation scientists and wildlife ecologists (Larson et al. 2016). What then can be done to address this unawareness as a step toward enabling dual-role protected areas to meet their conservation objectives despite the expanding recreational pressure?

Suggested plan of action.—To enable dual-role protected areas to meet their conservation objectives despite the expanding recreational pressure, the optimal approach is to: ensure that all recreational areas (e.g., trails and trail networks) are planned, designed, and sited using ecologically sound criteria; and, to continually employ sufficient proactive and adaptive management to prevent or at least minimize recreation-related disturbance to wildlife; such management would curtail the need for regular enforcement. This approach also has the potential to yield general public support for management, particularly if information provided about management challenges includes data and supporting graphics, specifically about fragmentation, to enhance the public's understanding of the challenges of poorly designed trail systems and the creation and use of unauthorized trails (Leung et al. 2011; Taff et al. 2019; Wolf et al. 2019). But this approach requires perpetual personnel and funding explicitly for management, which in turn points to the urgent need for public advocacy to secure fiscal support for management resources (i.e., fiscal support that is sustainable, perpetual, and at levels commensurate with the recreational pressure; footnote #5). How can this be achieved?

How people perceive their and others' recreation-related effects on wildlife may influence their general perspectives on such effects (Marzano and Dandy 2012). Shifting this perception-perspective nexus over time toward a common value of respecting wildlife may eventually mend some of the aforementioned societal disconnection. A shift in perspectives on the purpose of protected areas is also needed to one of understanding and acknowledging that their core function is conservation (Davies and Newsome 2009; Patten et al. 2017). The only chance there is of influencing people's perceptions is making the pertinent scientific information readily available. So, it is essential to implement a concerted campaign to disseminate science-based information about recreation-related disturbance to wildlife. Such a campaign needs to be well orchestrated, widespread, long-term, continual, and multimedia; this includes social media per Greer at al.'s (2017) conclusions about its efficacy in this context. In addition to the general public/voters (including recreationists), the following parties would be both the audience and the distributors within each of their fields and beyond: the media, environmental organizations, elected officials, policy and land-use decision makers, land management agencies and organizations, outdoor recreation merchants and associations, educational institutions, and researchers. The coverage would be framed as stories aimed to evoke appreciation for the diversity of sensitive species and the many ways they respond to our presence, and provide opportunities for what people can do to lessen the recreation-related disturbance to wildlife, which will benefit not only wildlife and other biological resources in the protected areas, but also the human experience there.

While the objectives of the campaign would be to influence people's perspectives in favor of wildlife and to modify recreational behaviors, policy, planning, and decision-making accordingly, the final goal would be to cultivate support for and harness the power of advocacy to gain the political will and action needed to secure perpetual fiscal support for management resources. Implementing such a campaign would not be easy nor fast and would take diligent oversight, as suggested by William Craven, the chief consultant for nearly 20 years of California's Senate Natural Resources and Water Committee. In an interview with the California Native Plant Society, he stated, "the best way to achieve your policy objectives is to make sure your policy objectives are funded. For example, small but important programs for the [California Department of Fish and Wildlife] are literally budget dust in the California budget, but unless someone is there to pay attention and connect the dots between the budget and the state laws, we don't get a complete resolution…[P]ositive changes in state law that everyone works so hard to accomplish are really much more effective when someone monitors the budget process to make sure those changes get as much funding as possible" (CNPS 2020). But, it seems that the choices are either to never reverse or at least halt the downward trajectory of wildlife in protected areas experiencing damaging levels and types of recreation or to ambitiously implement such a campaign toward a societal course change (Waterman 2019 for the term "course change").

Several of the results of the survey conducted for the SDEE initiative hint at a potential to mobilize a critical mass of people who learn about the recreation-related disturbance to wildlife and the associated urgent need for resources to address it, and assist in information dissemination. While the survey conducted for the SDEE initiative revealed a knowledge deficit among the respondents regarding problems plants and wildlife face, its results also indicate that, over a 12-month period, 74% of respondents voted in favor of laws to protect the environment, 31% volunteered to improve the environment, and 21% donated money to protect San Diego County's environment; in addition, approximately 70% were willing to pay additional local taxes to protect the environment, and a majority of the respondents were willing to pay up to $50 per year (Tinkler et al. 2019).

One avenue available for advocacy to secure perpetual fiscal support specifically for management of protected areas is assessing recreational fees and taxes. With respect specifically to the management of CDFW-owned protected areas, CDFW's 2005 and 2015 State Wildlife Action Plans recommended implementation of recreational fees and taxes beyond fishing and hunting licenses that would allow non-consumptive recreationists to support wildlife conservation and management of the resources they use and enjoy (CDFW 2015, 2016). To generate funds for the management of all protected areas, a long-successful model could be employed: since the 1930s, hunters have been paying federal excise taxes on the sales of sport hunting and shooting equipment to generate funding for habitat conservation (CDFW 2015). Eighty years later, these taxes plus sales of angling equipment had generated more than $10 billion towards conservation (CDFW 2015). Thus, hunters and anglers have been the primary funding sources for conservation efforts in California and North America (CDFW 2015). Considering the disturbance to wildlife from non-consumptive recreationists, it is past time for them also to pay their way for the use of protected areas through paying taxes on equipment for hiking, biking, riding, etc. to support management of these activities. A secondary benefit of such fees and taxes is that they may establish a direct connection for recreationists between their use of protected areas and the costs of protecting the protected areas, and thereby possibly diminish their disconnection from their disturbance to wildlife.

Other avenues for advocacy to secure fiscal support for management of protected areas include bond measures and voluntary contribution funds (VCF), though neither would necessarily provide a reliably perpetual source of funding. VCFs are sponsored by legislators to be enacted by the legislature; a VCF in this context would be explicitly and solely for the management of the protected areas in California, including CDFW's lands (with protected areas and management defined as described in footnotes #1 and #5, respectively). The funds must be administered such that they are made available timely. This would be similar to the VCF for California's Rare and Endangered Species Preservation Voluntary Tax Contribution Program which has funded work benefiting California's native at-risk plants, wildlife, and fish since 1983 (CDFW 2019) and now raises around $500,000 annually (FTB 2019).

Mainstream online and print media carried several articles in 2018 and 2019 about the overcrowding at and underfunding for the national parks (e.g., Simmonds et al. 2018; Waterman 2019; Wilson 2019); coverage such as this provides a good foundation of information. Articles like Yong's (2019) about the effects of the human voice alone on wildlife and Peterson's (2019) about the effects of hiking on elk represent steps in the right direction toward mainstream media honing in on specific impacts on wildlife from recreationists in protected areas. Coverage on species local to where people live is important and may make a stronger and more lasting impression with greater potential for shifting the perception-perspective nexus than species or settings remote from consumers of the media. Organizations like San Diego Zoo Global, which spearheaded the SDEE initiative (Tinkler et al. 2019), could significantly assist the campaign by engaging their media engines on behalf of local wildlife threatened by recreation.

A societal quid pro quo for protected areas?—At some point, the exploitation of protected areas resulting from recreation-related disturbance to wildlife, without commensurate reciprocity with care for the protected areas, may outweigh the benefits of public access to protected areas (Bennett et al. 2013). Many protected areas have already reached this point. Without adequate resources to combat the challenge of the obligation to conserve wildlife exposed to ecologically damaging levels and types of recreation, including unauthorized activities, the challenge will persist indefinitely at great risk of jeopardizing the protected areas' ability to meet their conservation objectives.

Regarding the pressure local, state, and federal government agencies have undergone for decades to acquire additional open space for recreation and to expand public access in existing protected areas (Wells 2000 in Reed and Merenlender 2008), elected officials and land-use decision makers need to address the demands, but not at the expense of biological conservation in protected areas. Some of the protected areas (e.g., the NCCP/HCP reserves) represent long-negotiated compromises for the sensitive species they are intended to protect in perpetuity. For some protected areas, no ecologically sound further compromise (e.g., expansion of public access) is possible; while recreation may be considered conditionally compatible in such protected areas, if open to public access at all, the extant levels of recreation may strain their ability to meet their conservation objectives. Protected areas that represent the final compromise for the species they support are particularly vulnerable to their wildlife values being compromised due to inadequate management (CDFW 2015). Ultimately, for wildlife that avoids human activity, it is unlikely that dual-role protected areas are entirely sufficient or justifiable for meeting conservation objectives; limiting or prohibiting recreation in strategic circumstances and locations within protected areas is necessary to achieve conservation objectives (Reed and Merenlender 2008; Bötsch et al. 2018; Dertien et al. 2018; Reed et al. 2019). Of course, this presumes sufficient management to maintain whatever recreational limits are set.

In summary, in the interest of wildlife in California and, more broadly, conservation within protected areas everywhere, the necessary actions with respect to non-consumptive recreation are to: (1) widely and continually disseminate science-based information about the recreation-related disturbance to wildlife; (2) apply the science to all planning for, policy- and decision-making about, and management of, recreation in dual-role protected areas; and (3) secure perpetual fiscal support for management of recreation in dual-role protected areas commensurate with the recreational pressure.

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Increased hiking and mountain biking are associated with declines in urban mammal activity

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Outdoor recreation can have negative consequences for many wildlife species (Larson et al. 2019, 2016; Monz et al. 2013; Sato et al. 2013). Increasingly, parks and preserves are embedded in a landscape of urban and suburban development (Radeloff et al. 2010), intensifying the exposure of remaining wildlife populations to human activity (Larson et al. 2018). In California, several research groups have studied wildlife responses to recreation in parks and preserves within densely populated coastal cities. Some of the resulting studies have documented negative effects, including declines in native mammal occupancy and detection rates (Patten and Burger 2018; Reed and Merenlender 2008) and reduced daytime activity (George and Crooks 2006), while others have found limited effects of recreation on wildlife occupancy and detection rates (Markovchick-Nicholls et al. 2008; Reilly et al. 2017). Managers need context-specific understanding of the nature and severity of recreation effects on wildlife to sustainably manage recreational use in protected areas, the vast majority of which are open to the public (Leung et al. 2018; UNEP-WCMC and IUCN 2019).

Experimental tests of recreation effects on wildlife can provide valuable insight into species' responses to human activity by minimizing variation in other factors that affect wildlife, such as residential development and vegetation composition. However, fewer than one-third of studies of recreation effects on wildlife include an experimental component (Larson et al. 2016), and a large proportion of experimental treatments exclusively measure immediate reactions of wildlife to an approaching human, often using flight initiation distance (e.g., Ikuta and Blumstein 2003; Jorgensen et al. 2016; Keeley and Bechard 2011). These immediate responses cause increased energy expenditure and can trigger trade-offs between foraging and flight behaviors (Duchesne et al. 2000), but it is less clear how they may translate into longer-term habitat degradation due to the regular presence of recreationists. It can be logistically difficult to experimentally alter the level of recreation on a trail segment or within a defined area, but when successfully implemented such studies have documented increased presence of nest predators (Gutzwiller et al. 2002) and reduced numbers of bird territories and bird species richness (Bötsch et al. 2017).

Conservation of mammals in densely populated and fragmented habitats such as southern California requires an understanding of the suitability of remaining habitat patches (Crooks 2002; Ordeñana et al. 2010), many of which receive high levels of recreational use (Larson et al. 2018). In this study, we assessed whether increased recreation rates were associated with reduced habitat suitability for native mammals. We conducted an opportunistic, quasi-experimental study of recreation effects on mammals using a before-after-control-impact (BACI) design, taking advantage of the closure and re-opening of an existing recreational trail in an open space park in San Diego, California. We expected that at impact locations (sampling points on the trail that was closed and re-opened), hiking and mountain biking would increase and wildlife activity would decline after the trail re-opened, while human and wildlife activity would remain similar at control locations (sampling points on trails consistently open throughout the study) within the same reserve.

The study was conducted in Black Mountain Open Space Park (32.984, -117.117) in San Diego, California, USA, which is owned and managed by the City of San Diego. The park is 951 ha, comprised primarily of coastal sage scrub and chaparral vegetation communities with some riparian and native and non-native grassland habitats. Dense suburban communities surround the park, and it contains approximately 32 kilometers of multi-use trails visited primarily by hikers and mountain bikers. The park also permits leashed dogs on the trails.

We established a total of seven sampling points on official and unofficial trails within the park in January 2017. Two points were located along the Miner's Ridge loop trail ("impact points", Figure 1), which was closed to public access from January 2017 until April 2018 for testing and remediation of elevated levels of arsenic detected in the soil. Five points were located along nearby trails not affected by the closure ("control points"; Figure 1). Point locations were selected as part of a larger project using a spatially balanced random design using the RRQRR algorithm on rasterized trail network data (Theobald et al. 2007).

To monitor human and mammal activity, we installed one motion-triggered camera (Bushnell TrophyCam HD Aggressor) at each sampling point, housed in metal security boxes and affixed to metal poles pounded into the soil facing recreational trails. We did not bait the cameras to avoid influencing animal activity (Wearn and Glover-Kapfer 2019). Cameras were programmed to take two photos per trigger with a five second delay between triggers. We began monitoring human and mammal activity at the impact points in late October 2017, leaving cameras running continuously until after the trail re-opened in April 2018. At the control points, we collected data between November 2017 and February 2018. After the trail re-opened, cameras operated at all seven sampling points for at least four weeks, ending in June 2018 (Table 1).

The seven cameras captured over 80,000 photos during the study period. Many of these were "false triggers" caused by rapidly growing vegetation, high temperatures, and wind, mostly in the mid-morning to late afternoon. Therefore, we randomly subsampled 20% of photos between 11 am and 5 pm at all sampling points to reduce time spent sorting

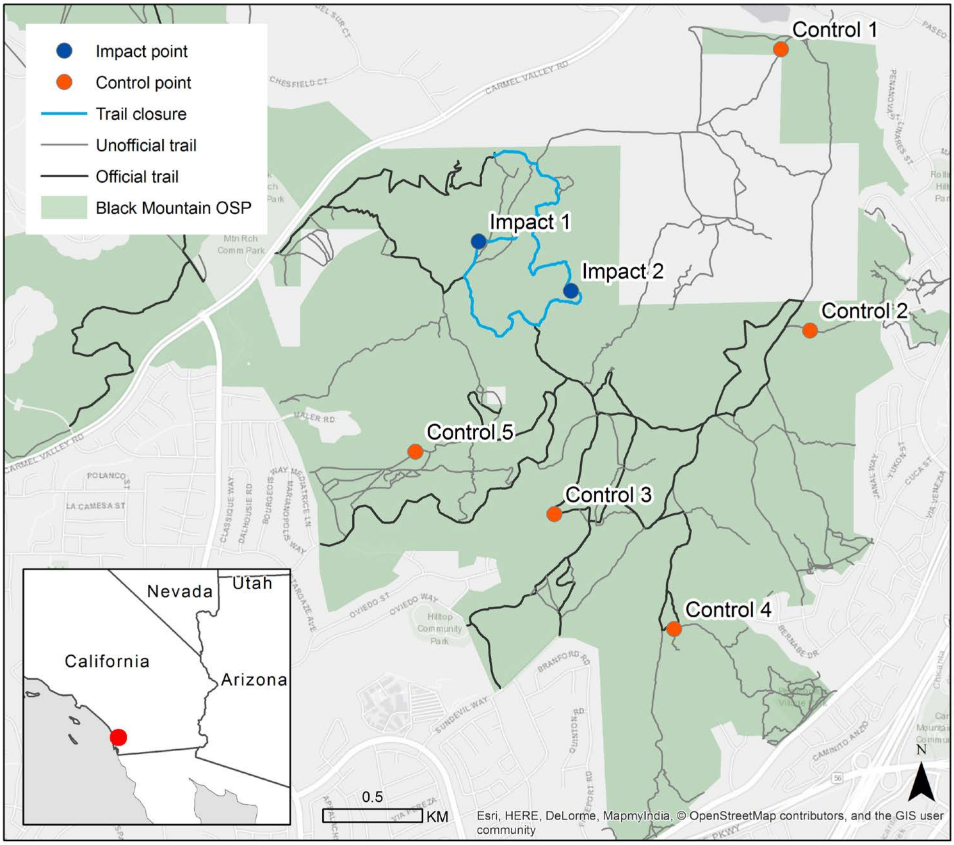

Figure 1. Location and sampling design of the before-after-control-impact (BACI) study conducted in Black Mountain Open Space Park in San Diego, CA, USA.

Table 1. Dates of camera data collection before and after the trail re-opened at impact and control sampling points at Black Mountain Open Space Park. Cameras were not installed or did not operate correctly on all days between the first and last sampling day; the "total days" columns report the number of days on which cameras were operational.

	Sampling effort before trail re-opened			Sampling effort after trail re-opened		
Point	First day	Last day	Total days	First day	Last day	Total days
Impact 1	1 Nov 2017	17 Apr 2018	134	19 Apr 2018	31 May 2018	43
Impact 2	1 Nov 2017	17 Apr 2018	168	19 Apr 2018	28 Apr 2018	27
Control 1	12 Dec 2017	1 Feb 2018	26	18 May 2018	30 May 2018	13
Control 2	12 Dec 2017	1 Feb 2018	26	4 May 2018	31 May 2018	28
Control 3	18 Nov 2017	13 Dec 2017	5	4 May 2018	30 May 2018	22
Control 4	18 Nov 2017	22 Dec 2017	26	4 May 2018	30 May 2018	28
Control 5	19 Nov 2017	22 Dec 2017	21	4 May 2018	31 May 2018	29

photos. Photos were organized in the Colorado Parks & Wildlife Photo Warehouse (Ivan and Newkirk 2016). Humans appearing in photos were categorized by activity (pedestrian, cyclist, equestrian, or vehicle) and animals were identified to species, except for brush rabbit (*Sylvilagus bachmani*) and desert cottontail (*S. audubonii*), which are difficult to distinguish in photos and were both labeled "rabbit."

To assess changes in human activity before and after the trail re-opened, we compared mean people per day at impact and control points using a non-parametric Wilcoxon-Mann-Whitney test since the data are counts. To assess changes in mammal habitat use before and after the trail re-opened, we used single-species occupancy models for each mammal species with sufficient detections using the R package *unmarked* (Fiske and Chandler 2011). Detection data were pooled into 5-day sampling occasions, resulting in ten survey occasions with five before and five after the trail re-opening. We did not include habitat covariates because minimal changes in habitat occurred between the sampling periods and because our primary goal was to investigate the interaction of treatment (control or impact sampling point) and time period (before or after the trail re-opened). Therefore, treatment and time period were the only variables included in the models, and we included the interaction (treatment*period) to test whether species showed a response to the trail re-opening. When a species was predicted to occur at all or nearly all sampling points, we assessed changes in detection probability rather than occupancy as a measure of relative activity or frequency of habitat use (Lewis et al. 2015; Wang et al. 2015).

Across all sampling points and time periods, there were an average (± 1 SD) of 12.2 ± 21.7 hikers, 7.2 ± 10.0 cyclists, 1.7 ± 3.2 dogs, and 0.01 ± 0.2 horseback riders per day at each sampling point, as well as infrequent motorized vehicles (park staff or utility personnel) at one sampling point where the trail was drivable. These recreation rates are relatively low compared to other parks and preserves in the region (Larson et al. 2018). People did not cease using the trail while it was closed, with the two impact points averaging 18.0 ± 15.8 and 20.4 ± 14.9 people per day during the closure (Figure 2). However, human activity approximately doubled at the impact points after the trail re-opened, averaging 38.2 ± 28.9 and 38.9 ± 19.6 per day (time period differences: $P < 0.001$). At the control points, human activity was similar between time periods (all $P > 0.33$) except for Control 5, which averaged 5.7 ± 8.1 people per day before and 23.2 ± 13.0 after the trail re-opened ($P < 0.001$). Control 5, located on an unofficial trail, is not part of the most obvious loop routes that could be made using the closed trail, but it could be connected with a longer loop route using unofficial trails, and therefore may have experienced depressed visitation rates during the closure period. Therefore, we ran additional occupancy models in which Control 5 was considered an impact point to ensure our results were robust to this possibility.

Mammal species we detected included rabbits (*Sylvilagus spp.*, total photos n = 537), coyotes (*Canis latrans*, n = 409), bobcats (*Lynx rufus*, n = 135), California ground squirrels (*Otospermophilus beecheyi*, n = 22), black-tailed jackrabbits (*Lepus californicus*, n = 4), raccoons (*Procyon lotor*, n = 2), and mule deer (*Odocoileus hemionus*, n = 1). However, only the bobcat, coyote, and rabbit were detected frequently enough for analysis. Bobcats were detected at six out of seven sampling points, and coyotes and rabbits were detected at all seven points; accordingly, we used detection probability rather than occupancy as our primary variable measuring changes in frequency of habitat use for all three species. At sampling points where they were detected, each species was detected at least once before and after the trail re-opening.

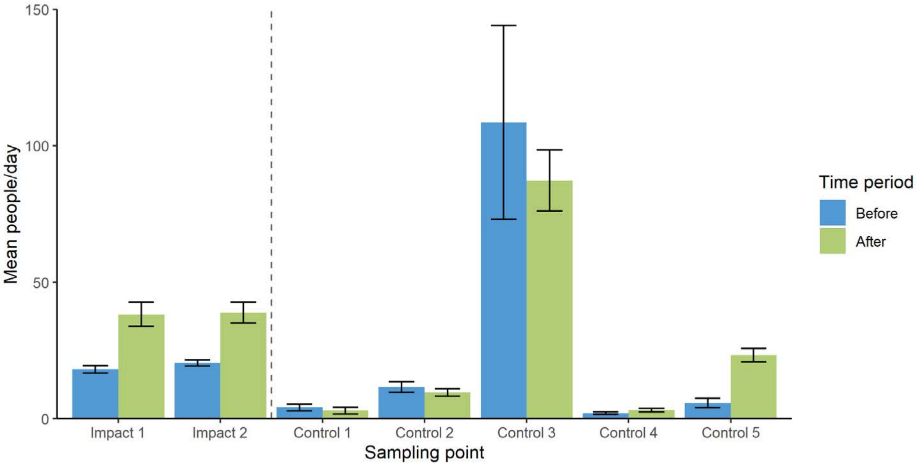

Figure 2. Human activity (mean people per day) before and after the Miners Ridge Loop trail re-opened at impact and control sampling points at Black Mountain Open Space Park. Error bars show one standard error. Differences between time periods were significant ($p < 0.05$ using a t-test) at Impact 1, Impact 2, and Control 5. The vertical dotted line divides the impact points (left) from the control points (right).

Occupancy models showed that detection probability was reduced at impact points after the trail re-opened for bobcats and coyotes, while remaining approximately the same at the control points (Figure 3). The effect was particularly strong for bobcats, with detection probability dropping from 0.90 ± 0.09 to 0.40 ± 0.15 at impact points after the trail re-opened while detection probability at control points increased slightly from 0.53 ± 0.13 to 0.65 ± 0.12. The interaction of treatment*period for bobcats was significant (z = 2.15, P = 0.03). Coyotes were detected at impact points during nearly every occasion before the trail re-opened (detection probability of 1.00 ± 0.001) but afterwards detection probability dropped to 0.70 ± 0.14, while detection probability increased slightly at control points from 0.79 ± 0.09 to 0.82 ± 0.08. However, the interaction term was not significant for coyotes (z = 0.14, P = 0.89). Rabbit detection probability did not differ significantly in relation to time period or treatment (interaction term z = 0.52, P = 0.61). Results did not change for bobcats or rabbits when Control 5 was considered an impact rather than a control point, but for coyotes patterns became less clear, with detection probability dropping more at control than impact points after the trail re-opened.

The number of sampling points was small due to the opportunistic nature of our study, limiting our ability to detect an effect of altered recreation rates on wildlife activity. Therefore, the fact that we still observed reduced activity rates by bobcats and, to a lesser extent, coyotes is particularly notable. Our findings echo those of previous studies in the region, which have found that these species and other mammals avoid human presence on short time scales (same-day occurrence; Patten and Burger 2018), and restrict their activity

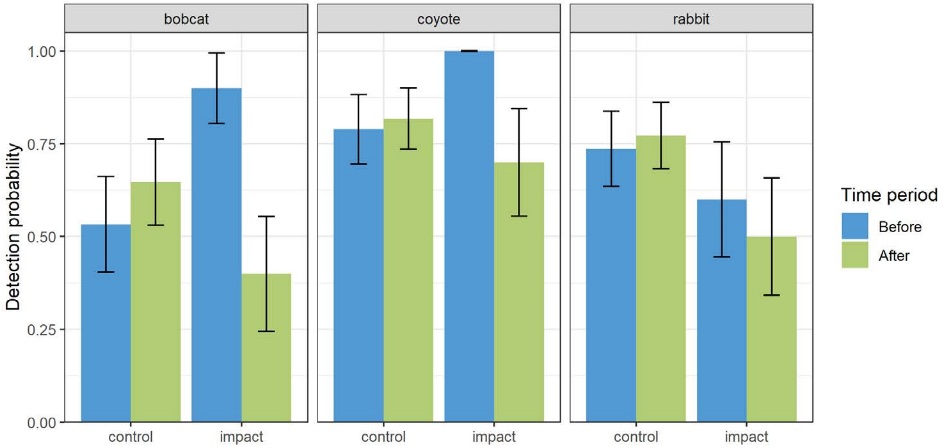

Figure 3. Predicted detection probabilities from single-species occupancy models for bobcats, coyotes, and rabbits before and after the Miners Ridge Loop trail re-opened at impact and control sampling points at Black Mountain Open Space Park. Error bars show one standard error. The interaction term for treatment*period was significant ($P < 0.05$) for bobcats.

in high human-use areas (George and Crooks 2006). We observed greater responsiveness in bobcats than in coyotes. While both carnivore species have shown sensitivity to recreation in previous studies (Patten and Burger, 2018; Reed and Merenlender 2008), coyotes can be relatively tolerant of human disturbance due to their adaptable behavior and omnivorous diet (Riley et al. 2003; Ordeñana et al. 2010). We did not observe changes in rabbit activity rates in connection with increased human activity, or by extension, reduced predator activity. Their smaller home ranges compared to bobcats and coyotes may mean that they are less able to shift their within-home range habitat use in response to short-term changes in human and predator activity.

Previous studies have also found that these species may shift their diel activity patterns to be more nocturnal in areas with higher human use (George and Crooks 2006; Reilly et al. 2017; Wang et al. 2015; Nickel et al. 2020). While shifts in diel activity patterns may have occurred in our system, overall activity levels were lower after the trail was re-opened, indicating than any temporal shift did not completely mitigate effects of human presence. However, despite changes in activity levels (as measured by detection probability), we did not observe changes in the occupancy status of the sampling points, suggesting that while the habitat may have been somewhat degraded, it was not completely unsuitable after the trail re-opened. Given the relatively small size of the park and its highly developed surroundings, reduced use of impact points by bobcats and coyotes likely indicates a partial shift in habitat use to other areas of the park. Bobcats slightly increased their use of the control points after the trail re-opened, perhaps suggesting such a shift, though this difference was negligible for coyotes.

Future experimental manipulations at larger spatial and temporal scales could help assess the consistency of our findings, increase the precision of estimated detection probability parameters, and assess responses of additional wildlife species. The opportunistic nature of our study design resulted in spatial separation of the impact and control points, which may have limited their ability to serve as true replicates due to spatial autocorrelation (Legendre 1993). A true experimental design with randomly assigned treatment and control locations would provide stronger evidence of recreation effects, such as the study by Bötsch et al. (2017) which documented reductions in bird territory establishment in response to low levels of recreation compared to areas with no recreation. Coordination with volunteer groups and docent-led programs or using recorded human voices (e.g., Suraci et al. 2019; Ware et al. 2015) could make it more feasible to experimentally apply treatments that simulate higher levels of recreation.

Though the level of human activity approximately doubled after the trail was re-opened, we speculate that the difference may not have been obvious to recreationists. Forty people per day, approximately the average level of use after the trail re-opened, is still low compared to many other San Diego-area parks and preserves (Reed et al. 2019). However, this difference appears to have been perceptible and meaningful to wildlife, and perhaps crossed a critical threshold of disturbance causing reduced rates of use of the trail. Accordingly, habitat degradation near trails due to human disturbance is likely common across parks and preserves across the region.

Our findings highlight that wildlife can respond rapidly to changes in the levels of human disturbance, even when they have experienced similar levels of disturbance previously. Data collection for the ‘after’ period started immediately after the trail was re-opened and continued for four weeks. The observed reduction in detection probabilities suggests that bobcats, and to a lesser degree coyotes, may respond to changes in the relative intensity of human activity by rapidly altering their fine-scale habitat selection. Rapid avoidance responses to recreation have been previously documented for mountain caribou (Lesmerises et al. 2018) and bottlenose dolphins (Lusseau 2004), but it is not clear how short-term behavioral avoidance may translate to fitness or population impacts (Bejder et al. 2006). Higher recreation intensity was presumably not novel to these individuals since the trail had been open to recreation for many years prior to our study, which suggests that the animals were not fully tolerant of prior levels of human disturbance. It is therefore possible that for these species, habitat degradation from recreation could be relatively quickly reversed if human activity was limited to lower levels, or spatially or temporally constrained. Land and wildlife managers often use seasonal closures to protect wildlife during periods of heightened sensitivity such as the breeding period (Burger and Niles 2013; Coleman et al. 2013; Richardson and Miller 1997), but the efficacy of these closures is rarely tested. The rapid response we observed suggests that targeted temporal closures could be a promising approach for reducing impacts of recreation.

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Author contributions:
Conceived and designed the study: CLL, SER, KRC
Collected the data: CLL
Performed the analysis of the data: CLL
Authored the manuscript: CLL
Provided critical revision of the manuscript: CLL, SER, KRC

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An assessment of non-consumptive recreation effects on wildlife: current and future research, management implications, and next steps

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Most research on the effects of non-consumptive recreation on wildlife to date has focused on birds and mammals. This research typically focuses on behavioral responses of individuals despite practical limitations in extrapolating ecological outcomes from individual behavior. Data gaps therefore present difficulties in integrating wildlife-protective policies into public access management. These gaps are exacerbated by a lack of wildlife studies that include data on public use patterns of open space areas. In a survey of park and open space managers in the San Francisco Bay Area, few of the entities surveyed restricted recreational access permanently or seasonally to address biological constraints; yet most indicated the presence of sensitive plant or animal species on their lands or stated conservation as one of their organization's purposes. To better bridge the gap between research and management practice, more research is needed on species beyond birds and mammals. This research should extend beyond noting behavioral response and should integrate investigation of outdoor recreation use patterns.

Key words: California, non-consumptive recreation, open space, parks, public access management, San Francisco Bay Area, wildlife

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Throughout the state of California, there exists a large diversity of designated open space and protected areas that allow public access and outdoor recreation. Based on data from the Survey of Public Opinions and Attitudes on Outdoor Recreation in California, the average number of days of outdoor recreation participation among adult Californians is 96 days per year (California State Parks 2012). Based on California's population of approximately 27.4 million adults in 2008, California State Parks estimated approximately 2.6 billion days of outdoor recreation by adults during that year; that figure would be higher based on current population estimates. Within regional, state, or national parks, outdoor recreation participation (i.e., adults and children) totaled an estimated 478 million days, and for non-park natural and undeveloped areas there were an estimated 368 million annual days of outdoor recreation participation (California State Parks 2011).

A large portion of outdoor recreation activity consists of frequent use in the same areas by the same visitors. Much of it is relatively close to visitors' homes, and with California's warm, Mediterranean climate, outdoor recreation use often occurs near dawn and dusk, the times of day when multiple wildlife species are most active. Many areas where outdoor recreation occurs also provide occupied or potentially suitable habitat for special status wildlife species. California includes a variety of habitats that are occupied or potentially occupied by 181 state or federally listed wildlife species (CDFW 2019).

Non-consumptive forms of outdoor recreation (defined as those activities that do not include fishing and hunting) can impact wildlife species and their habitats in a variety of ways. There may be loss of individuals along trail corridors through incidental recreational use, such as crushing burrows or destroying nests. Non-consumptive recreation may also affect habitat. For example, recreation facility development can remove habitat, and recreational use of facilities can result in water quality degradation, soil erosion, and ground cover loss (USDA 2008). Presence of humans may cause displacement or change in behavior of wildlife, both temporary and permanent, through proximity to habitat, habitual use of an area (e.g., trails), or through direct harassment (Trulio et al. 2013; Shannon et al. 2014). There may also be effects on wildlife behavior from nighttime outdoor recreation activity, including light and sound pollution, or other disturbances associated with these recreational activities. Littering can have both direct and indirect effects (Boarman 2002), and bringing pets to open space and other types of protected areas may also cause direct and indirect impacts to wildlife species (Reed and Merelender 2008; Reilly et al. 2017).

However, despite more than 40 years of research on this topic, significant information gaps exist. The purpose of this article is to: 1) summarize what is known about effects on non-consumptive recreation on wildlife, 2) summarize current management practices used by park and recreation agencies in the San Francisco Bay Area to manage public access to protect wildlife, and 3) suggest additional research that will help fish and wildlife managers as well as park and open space managers more effectively manage and respond to potential impacts of non-consumptive outdoor recreation on wildlife species and their habitats.

CURRENT STATE OF THE KNOWLEDGE

Overall state of the knowledge

To preliminarily identify potential data gaps and long-term trends in the literature, we searched Google Scholar for articles containing the keywords "non-consumptive recreation" and "wildlife" at ten-year increments from 1980 to 2019. We subsequently performed the same query substituting "plants" for "wildlife." We identified 515 results containing the keywords "non-consumptive recreation" and "wildlife" between 1980 and 2019. Of these, 26 (5%) were published in the 1980s, 82 (16%) in the 1990s, 170 (33%) in the 2000s, and

237 (46%) in the 2010s. The same search with “plants” substituted for “wildlife” yielded 298 results between 1980 and 2019—15 (5%) in the 1980s, 44 (15%) in the 1990s, 105 (35%) in the 2000s, and 134 (45%) in the 2010s.

It is clear that the number of articles related to non-consumptive recreation and plant and wildlife management has increased over time, and that wildlife is consistently more studied than plants. More granular trends in the literature are less immediately apparent. We therefore identified several comprehensive literature reviews from the last 40 years to better understand which topics in plant and wildlife management are most often studied. In particular, we sought out reviews that would elucidate long-term trends in which types of recreational activities are the most studied, whether response variables are typically quantified at the individual or population level, which taxa are the most studied, and other trends that may inform the scope of future research. Due to the higher volume of studies available on wildlife than plants, we focused our efforts on wildlife-centered articles.

Boyle and Samson (1985) conducted a comprehensive review of the state of knowledge in which they identified trends in studies containing original data on terrestrial vertebrates in North America (n = 166). These articles most often studied birds (103, 62%), followed by mammals (70, 42%), with few studies of herpetofauna (7, 4%). Boyle and Samson reported negative effects for most activities and taxa, postulating potential mechanisms such as direct disturbance and indirect effects such as habitat degradation, noting that the latter may result in simpler vegetation profiles and overall loss of habitat diversity. Positive effects on overall biodiversity were reported in a few studies, but these positive effects typically corresponded with increased abundance and diversity of common species well-adapted to frequent disturbance by humans. Based on data gaps identified through their review process, Boyle and Samson concluded that primary shortcomings in the literature included a lack of experimental, rather than observational data, and a need to move from assessment of disturbance and mortality to analysis of long-term ecological effects (Boyle and Samson 1985).

A more contemporary review conducted by Larson et al. (2016) analyzed 280 articles on the effects of non-consumptive recreation and wildlife. This review was broader in scope than that of Boyle and Samson, including a wider swath of recreational activities and all taxa globally. Although these results are not directly comparable due to differences in scope, Larson et al. identified similar trends to Boyle and Sampson 31 years earlier. The researchers found that articles remained mostly observational, with only 30% of articles containing an experimental component. Among the articles included in their review (n = 280), mammals were studied the most often (114, 42%), followed closely by birds (101, 37%). A wide gap was observed between mammals and birds and invertebrates (34, 12%), herpetofauna (17, 6.2%), and fish (14, 5.1%). Notably, the authors found that the majority of species studied with International Union for Conservation of Wildlife (IUCN) status were classified as species of least concern, and that endangered, critically endangered species, and data-deficient species were the least often studied. Similar to Boyle and Samson, most studies evaluated identified significant effects of non-consumptive recreation on wildlife, with negative effects being the most frequent. Most studies that showed unclear results as to whether effects were positive or negative had a behavior-based response variable, demonstrating the challenges associated with interpreting behavioral responses (one of which is the potential for wildlife to habituate to recurring, non-threatening recreational use), and the implications for long-term ecology and land management (Larson et al. 2016).

Most studies on the effects of non-consumptive recreation on wildlife were conducted in North America (Larson et al. 2016). In a paper on recreation impacts on wildlife submitted to the federal Interagency Visitor Use Management Council (IVUMC), Marion (2019) summarized the current state of research, with results falling into five broad categories. The categories included: 1) type of recreational activity; 2) recreationist behavior; 3) impact predictability; 4) impact frequency and magnitude; and 5) impact timing and duration. In regard to category one, Marion found mixed results on impacts from slow versus fast (e.g., walk, run, mountain bike, motorized vehicles) recreation activities. Regarding category two, he found visitors who directly approach wildlife are perceived as threatening, and wildlife are less disturbed by recreation travel that is slow, quiet, and in directions parallel to or away from them. Marion also found that wildlife are able to adapt to and tolerate consistent nonthreatening recreational activities, but unpredictable recreational activity in less visited off-trail locations can cause greater impact (category three). Repeated human interaction and disturbance of wildlife can exceed a threshold of tolerance that causes wildlife to leave a preferred habitat (category four). In regard to category five, Marion found wildlife show locational and seasonal sensitivities to recreation. Marion then describes multiple strategies to manage recreation to minimize impacts on wildlife, which are summarized later in this paper.

California-focused research

California plays an important role in this body of research due to its abundant biodiversity and large areas of protected and/or publicly-owned lands. California has been relatively well-studied, with most research focused on birds, and more recently mammalian carnivores. The discussion below is not intended to be exhaustive but rather to summarize the findings of representative research efforts with implications for recreation and wildlife management and provide context for on-the-ground practices and recommendations, with a focus on California.

In the San Francisco Bay Area, several studies on avian wildlife have emerged in recent years. A 2008 study on foraging shorebirds and trail use found no change in behavior or species diversity during trail use (Trulio and Sokale 2008). These findings indicate foraging shorebirds at regularly used trails may habituate to human activity. However, other experimental studies have found that shorebird numbers decreased with human presence on trails (Trulio et al. 2013), and that trail uses such as jogging and dog walking can increase flight distance (Lafferty 2001). Differences in shorebird response to human disturbance are likely attributable to the birds' degree of habituation to human disturbance. Studies indicate that shorebirds in areas of more frequent human disturbance display less response to human activity; although, birds tend to use these areas at lower rates than areas with less disturbance (Josselyn et al. 1989). Trulio et al. (2013) recommended keeping trail users at least 50 m from foraging habitat. They also suggested that infrequent trail use may be more disruptive to birds then frequent trail use, indicating that habitation may occur as referenced above. Similarly, Miller et al. (1998) found the composition and abundance of birds to be altered in a Colorado grassland and forest setting, with an area of influence of approximately 75 m (zone where human activity may displace wildlife from suitable habitat).

As exemplified by these studies, even the least intrusive non-consumptive recreational activities, such as hiking and picnicking, have the potential to affect wildlife. Reed and Merenlender (2008) examined this possibility in the context of mammalian carnivores in the Northern San Francisco Bay Area. They consistently found that sites where quiet, non-consumptive recreation is permitted had lower density of native mammalian carnivores than areas with no recreation. All recreational sites showed a shift in carnivore detections toward non-native carnivores such as domestic dogs and cats (Reed and Merenlender 2008). These results corroborate the relatively consistent finding that the mere presence of humans and their introduced domestic species may prove detrimental to native wildlife, regardless of the types of recreation in which they engage.

The finding that community composition shifted toward non-native species such as domestic dogs where recreation was permitted suggests a need to better understand the effects of dogs on native wildlife and the efficacy of various dog management strategies. This need is furthered by the outsized role dogs tend to play in open space management efforts. To follow up on their previous findings, Reed and Merenlender (2011) further studied the effects of different dog management policies in recreation areas. They found no significant differences in mammalian carnivore abundance or species richness between recreational sites with no dogs, sites with on-leash dogs, and sites with off-leash dogs. They did, however, identify significant differences between all three types of sites and reference sites with no recreation, suggesting that the presence of humans is a more important influence on species diversity and carnivore density than that of dogs (Reed and Merenlender 2011).

MANAGING PUBLIC ACCESS TO PROTECT WILDLIFE

To better understand whether trends identified in the literature are translated to open space management practice, we obtained information from local park, recreation, and open space area managers on how they address public access and its potential impacts on wildlife. Due to the abundance of literature focusing on the region and the richness of open space availability and biodiversity in close proximity to urban populations, we focused this effort on the San Francisco Bay Area.

Case study on San Francisco Bay Area open space management strategies

To assess current practices in addressing biological constraints in public access management and to identify how principles elucidated in the literature are applied in practice, we conducted a case study based on information obtained from ten open space management entities in the San Francisco Bay Area. Four of these were special districts, four were county agencies, and two were non-profit organizations. Each organization is identified numerically in the following discussion for the purposes of anonymity. All organizations were contacted by email in September 2019 and provided a survey with a standardized set of questions on public access management approach in areas known to contain sensitive biological resources. Each organizations' webpage was subsequently queried for supplemental information.

Five of ten organizations contacted via email responded to initial outreach efforts. Of these, three indicated that they restrict recreational access to some or all of their lands based on the presence of sensitive biological resources (County Two, Special Districts Two and Three). The other two respondents said they do not restrict access on any of their lands (Special District Four) or that they entitle open space preserves but do not hold land in the long-term or provide access opportunities (Non-Profit One).

County Two's response suggests limitations in their capacity to restrict public access for the purposes of addressing biological constraints. This County was in the process of developing a dog policy to determine where dogs are permitted and where leashes are required. In describing this policy, County Two representatives did not specify any biological factors being considered. Outside of its dog policy, the County indicated that they may restrict park access due to wet weather or public safety concerns; but that they generally do not restrict access for biological reasons apart from seasonally fencing off a small portion of one park for nesting shorebirds. In describing their shorebird protection efforts, representatives stated that they only restrict access insofar "as that is allowed."

Webpage queries of all 10 organizations demonstrated that a management approach similar to County Two's was common. There was little indication of restricted recreational access such as permit-only areas or seasonal park or trail closures to address biological constraints, with dog policies being the most common strategy to protect wildlife. Most permits were related to facility rental or special event production, with some parks containing sensitive plant species also providing scientific collection permits. Furthermore, most seasonal trail closures cited severe weather and trail washouts, and few were explicitly tied to biological concerns. Among the organizations surveyed, restricting the presence of dogs in parks was the most common strategy used by land managers to reconcile potential incompatibilities between non-consumptive recreation and sensitive species protection. Virtually all organizations had some type of dog policy in place or were in the process of establishing a dog policy. More than half of them specifically cited disturbance of wildlife or other biological constraints when describing dog access restrictions. Policies ranged from outright prohibition of dogs to requirements that dogs be kept on leashes.

Special District One was a notable exception to the patterns described above. In addition to restrictions on dogs, this organization employed a variety of methods, including permit-only access areas and seasonal trail and road closures. Special District One maintains one area that can only be accessed by permit holders. This area provides habitat for special-status avian species and other non-special status wildlife species. Recreational activities in this area are restricted to camping, hiking, horseback riding, and backpacking, and permits must be purchased in advance. Hunting is not allowed. Additionally, Special District One closes portions of one park annually for raptor nesting, and at the time of writing, one other park had trail closures for unspecified habitat protection. Special District One indicated in its response to outreach efforts that it annually and occasionally employs this technique as needed, closing trails and roads based on the presence of wildlife during sensitive windows such as nesting or mating. Moreover, correspondence with this District indicated that they purchase lands in collaboration with conservation organizations and place these lands under easement, and that when these lands become publicly accessible, permissible recreational activities are limited to those compatible with applicable habitat conservation plans. In addition to these strategies and similarly to other organizations, Special District One provides restrictions on where and how dogs may be present on their land. Biological considerations incorporated in this District's dog policy included prohibition on dogs where specified by conservation easements and in sensitive habitats such as marshes and wetlands.

The two non-profit entities included in this study had management practices that were among the most wildlife-protective. Non-Profit One indicated that opportunities for public access on their lands are very limited due to their high conservation value and the organization's emphasis on preserving biodiversity—suggesting an approach placing higher value on conservation than recreation and incidentally allocating recreational opportunities where compatible with biological constraints. Perhaps the most unique management strategy identified in our case study was employed by Non-Profit Two. This organization divided their lands into two distinctive types of preserves—with the primary purpose of one type being public outreach and education, while the other type primarily served conservation purposes. While conservation and restoration activities are held on both types of preserve, the former includes more opportunity for educational events, hiking, and community volunteer days than the latter, where public access is limited due to resource constraints.

In our outreach and website queries, we looked for permit-only access areas, seasonal trail closures, restrictions on dogs, and other management strategies. Few of the public entities included in this case study restricted recreational access permanently or seasonally to address biological constraints, with surveyed non-profit organizations doing so more holistically. Yet, most public entities indicated the presence of sensitive plant or animal species on their lands or stated conservation as one of their organization's purposes. Although this case study examines a small, non-representative sample of management entities, these findings suggest that the public land management agencies that responded to our query may be constrained by mission and purpose in their ability to limit public access relative to other organizations such as non-profits with a singularly focused purpose of resource protection.

ADDITIONAL RESEARCH NEEDS

Several implications emerge from our review: 1) research efforts need to extend beyond noting individual behavioral responses; 2) more research is needed on species beyond birds and mammals; and 3) impact studies needs to be more frequently integrated with research on outdoor recreation use patterns.

The studies we reviewed indicate that although some research has been conducted on the effects of non-consumptive recreation on wildlife, the scope is generally narrow. There is a need for additional information on other taxa, given the number of listed species that are not birds or mammals. Moreover, recreational impacts on special status plant species are consistently less studied than those on wildlife, despite the high number of listed plant species, and the fact that habitat degradation (including impacts to vegetation) is a potential mechanism for recreation's impacts on wildlife. One example of such an investigation is the Spring Mountains National Recreation Area Landscape Analysis (USDA 2008). This report included an evaluation of spatial impacts from current and future recreation facilities on habitat loss for 30 special status species, most of which were plants. Another example is the Marin County Road and Trail Management Plan (Marin County Parks and Open Space District 2014) which included an analysis of illegally constructed mountain bike trails on special status species, most of which were plants.

Our findings suggest that individual wildlife response to recreational activity is studied more often than population-level response. One exception is experimental, longitudinal research conducted by Riffell et al. (1996), who evaluated the effects of repeated intrusion by hikers to avian communities in Wyoming's Medicine Bow National Forest for 10 weeks during the breeding season over 5 years. Their study found no cumulative or yearly declines in seasonal species richness, mean richness, or mean total abundance. They did find that repeated intrusions altered the composition of the community represented by the most common species, but no widespread impacts on avian community structure were documented. Continuing this line of research will be important to evaluate recreation impacts at the population level. This is particularly crucial given the nature of Federal and State regulatory schemes for endangered species, which typically take a population-based approach to species protection. Moreover, conducting research at the population level eliminates the need to interpret individual-level responses' implications for broader conservation efforts. Extrapolating individual response to a population-level context can prove difficult (Bejder et al. 2009; Caro 2007), and eliminating the need to do so reduces uncertainty for decision-makers.

Population-based outcomes should continue to be incorporated in future studies to facilitate stronger understanding of recreation's implications for conservation. While this is a more difficult undertaking than simply investigating behavioral responses, this type of research is needed to inform policies implemented by land managers. Useful models for conducting long-term, quasi-experimental research that addresses the larger question of population viability in the context of known threats, including non-consumptive recreation, to special status species exists in previous studies and can be used to inform future research.

Additionally, the taxa studied need to be prioritized to include additional groups. Mammals and birds have been studied more often than other taxonomic groups since non-consumptive recreation became a popular topic of research in the 1980s, and continue to be the most studied today. This does not necessarily correspond with greater conservation or research needs, especially considering the high number of amphibian, reptile, and invertebrate species with special status as designated by the California Department of Fish and Wildlife or the U.S. Fish and Wildlife Service (~61% of listed species in California). If park and open space managers are to make informed, high-impact conservation decisions using the limited resources available to them, research efforts must be prioritized based on conservation need rather than focusing on the most visible species. Similar work is needed to provide frameworks for prioritizing research dollars in wildlife and open space management.

Before embarking on a new vein of research to address these above areas, it may be useful to consider comments offered by Dr. David Cole and William Hammitt, from their textbook, *Wildland Recreation: Ecology and Management*. From Hammitt and Cole (2015):

> The relationship between amount of recreational use and wildlife impacts is not well understood. Very few studies have systematically examined the effects of varying numbers of visitors on wildlife. Even fewer wildlife studies have determined an accurate population count of organisms prior to the introduction of recreation…..Previous research indicates the complexity of the relationship by stating that the number of visitors cannot be considered in isolation from species requirements and habits, setting attributes, and type of recreational use. Various aspects of use intensity are also involved, including frequency and regularity of use and number of people at one time.

Thus, the third area where additional research is needed is integrated research that links specific outdoor recreation patterns to effects on species distribution and abundance. Some of this is occurring via research by Larson, Reed, Merelender, and others. For example, Larson et al. (2018) correlated recreational use levels with habitat occupancy for seven special status species for 18 reserves in San Diego County. This is a thorough research effort that integrates a model to predict recreation use levels with whether habitats for special status species are occupied. A more comprehensive and robust effort is needed that extends this type of research to a variety of habitat types and recreational use levels throughout California. Finally, the effectiveness of the "regulatory toolkit" that park, recreation, and open space managers have to control outdoor recreation use is well-established for federal lands, but its applicability to protected areas in close proximity to urban areas is largely unknown. Marion (2019) mentions strategies on how to address recreation impacts to wildlife including: reducing use, modifying the timing and location of use, modify the type of use, visitor behavior and expectations, and maintain and/or rehabilitate the resource. In regard to modifying visitor behavior, there is an entire body of research that focuses on how well visitors comply with wilderness and other protected area regulations (Lucas 1981; Washburne 1982; Duncan and Martin 2002; Marion and Dvorak; Martin and McCurdy 2010), and a review of low impact education programs (Marion and Reid 2007), such as Leave No Trace, suggests these programs can be effective at altering visitor behaviors that can cause impacts to natural resources. However, what has not been well investigated is how widespread such programs are implemented by park, recreation, and open space managers, and their applicability to open space preserves near urbanized areas.

Furthermore, it is important for research to go beyond theory and be adopted into practice by land managers. Research findings must be placed into a conservation and management context, with actionable priorities and recommendations for park, recreation, and open space managers. Researchers should engage with park and open space managers to ensure that science-based policies are enacted. Although limited in scope, our case study indicates some potential disconnects exist between the scientific community and on-the-ground open space management entities. For example, a large portion of the San Francisco Bay Area open space management and wildlife conservation efforts focused on developing sound dog policies; yet our research on the matter suggests that the effects of dogs are secondary to those of the presence of humans. Therefore, it may be of higher impact to examine ways to limit human activity in areas with sensitive biological resources through trail routing, permanent and seasonal park closures, and other methods.

Researchers and managers should therefore work together to develop, implement, and test science-based strategies. Social science-based methods should be included when testing approaches to better understand compliance with and attitude towards various management approaches as well as park use patterns. Several studies described above (Duncan and Martin 2002; Martin and McCurdy 2009) integrated these methods into their research but were focused on compliance with wilderness regulations.

Taylor and Knight (2003) demonstrated a potential approach for researchers to integrate study of park user perceptions into their work. They used a behavior-based model to study ungulate response to hikers and mountain bikers in a state park in Utah and, importantly, analyzed visitors' perceptions of their own effects on wildlife. They found that recreationists tend to attribute adverse effects on wildlife to other recreationists' actions and not their own. These results illustrate the importance of park user education as well as collaboration between the natural and social sciences in recreation and wildlife management.

Another example may be found in research conducted by Jefferson County Open Space District in Colorado, which has documented "heat maps" of recreation use for trails that bisect their open space areas. This information can then be overlaid with known or potential occurrences of special status species. Accurately collected recreation use data such as these would help biologists and park and open space managers better understand the relationship between overall park use patterns and wildlife impacts, an area of research that we found to be notably understudied.

To move toward sound management practice that effectively accommodates demand for public access and need for species protection, methodological changes and research prioritization are needed. Through review of literature related to the effects of non-consumptive recreation on wildlife and a survey of local agencies' integration of science-based methods into open space management efforts, we found that significant data gaps exist in both science and policy. New frameworks are needed to prioritize conservation efforts, which identify sensitive resources and integrate these into management efforts. Additional research using population-based response variables is necessary to quantify effects and determine whether management strategies are effective. A holistic approach incorporating conservation status and public recreational use patterns is needed to prioritize finite research and management resources.

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Author Contributions

Conceived and designed the study: JB
Collected the data: AS
Performed analysis of the data: AS
Authored the manuscript: JB, KD, AS
Provided critical revision of the manuscript: JB, KD, RC

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Wildlife occupancy and trail use before and after a park opens to the public

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We investigated changes in wildlife trail use and occupancy from baseline conditions after a park opened to the public; we were curious if wildlife would alter either their use of the trails or the surrounding areas or both in response to the park opening. We generated single-season occupancy estimates as a site-wide occupancy metric from 23 camera traps placed at 0.5 km intervals throughout the park and wildlife and human detection rates to measure intensity of trail use from 10 camera traps placed every 500 m on the trail. We compared the findings from the four seasons before to the four seasons after the park opened to the public. Human trail use increased sharply after opening and then lessened, but was markedly higher than prior to opening. Bobcat (*Lynx rufus*), coyote (*Canis latrans*) and gray fox (*Urocyon cinereoargenteus*) did not alter trail use relative to study area occupancy. Two species, black-tailed deer (*Odocoileus hemionus*) and gray squirrel (*Sciurus griseus*) altered trail use, and puma (*Puma concolor*) and wild turkey (*Meleagris gallopavo*) altered both trail and study area use. All species, except for the raccoon (*Procyon lotor*) and wild turkey, recovered to pre-opening conditions, by the winter (that is, after approximately 9 months) following opening.

Key words: camera trapping, occupancy, open space, recreational impacts, trail use

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Protected open space is considered important for conserving wildlife and providing public recreational opportunities in the San Francisco Bay Area. Recreation is often supported by concomitant trails and infrastructure, that is, that existing trails and fire roads are used by the public and, in turn, additional infrastructure is required to facilitate access. To conserve wildlife effectively, it is important to understand how wildlife may be affected by human use of the landscape even when those uses appear benign. Wildlife often share the use of trails with humans, their dogs, cyclists, motorized vehicles, and equestrians, while also preferentially using roads and trails for movement (Whittington et al. 2005). The extent to which non-motorized recreational human uses impact wildlife that rely upon open space (for breeding, movement, foraging, etc.) is the subject of this study. Wildlife may be disturbed by human presence on trails and, as a result, vacate the surrounding landscape despite the landscape's capacity to support them. An alternate scenario may be that wildlife avoid or reduce trail use (that humans are using) but remain resident in the surrounding landscape in response to human trail use.

Wildlife can be both negatively or positively associated with human presence and zones of urbanization. Recreation has been shown to have behavioral impacts on wildlife, such as reduced feeding times (Cassirer et al. 1992), detrimental stress responses (Barja et al. 2011), reduced temporal occupancy (Wang et al. 2015), but also the reverse (Ordeñana et al. 2010; see also Reilly et al. 2016 for a review of the literature). With pressure on open space providers to accommodate human recreation and increase accessibility, understanding how access and intensity of human use affects wildlife provides essential information towards making decisions that effectively balance wildlife conservation with human interests.

We examined how public presence may affect wildlife trail use and occupancy in the surrounding landscape in the North Sonoma Mountain Regional Park and Open Space Preserve (hereafter, "Park/Preserve") in southeastern Sonoma County, California. A camera trapping array (grid) encompassed the Park/Preserve to assess changes in single season occupancy estimates (that is, we use occupancy as an index of prevalence or a surrogate of abundance in the study area; O'Brien et al. 2010; Royle and Nichols 2003; MacKenzie and Nichols 2004; MacKenzie et al. 2006; but see Burton et al. 2015 and Steenweg et al. 2018, 2019 for cautionary discussions). Additional cameras were placed on the trail to assess wildlife and human use (that is, through detection rates as a measure of intensity of use); trail construction had been completed by the time the study began.

Below we outline the key hypotheses to address the following question: How does human trail use affect wildlife trail use and occupancy in the study area?

$H_ø$: Wildlife did not change their use of trails or residency (abundance) within the Park/Preserve after it is opened to the public. Wildlife occupancy estimates (abundance) from the grid and the trail detection rates do not change after the Park/Preserve opens to the public.

H_1: Wildlife use trails less but are still resident within the study area after the Park/Preserve is open to the public. Wildlife trail detection rates decrease after human trail use increases but occupancy estimates (abundance or residency) does not change in study area after the Park/Preserve opens.

H_2: Wildlife reduce trail use and vacate the study area after the Park/Preserve is open to the public. Both wildlife trail detection rates and site-wide occupancy decrease within the Park/Preserve after it opens to the public.

H_3: Certain types of wildlife (e.g., carnivores or ungulates) may be differentially affected by the presence of humans. With regard to trail and Park/Preserve use, see H1 and H2.

H_4: Wildlife resume a similar intensity of trail use and abundance within the study area after a period of time post-opening compared to pre-opening measures (latency to habituation). Wildlife trail detection rates decrease initially after opening, but then return to the pre-opening levels after a period of time. If wildlife do leave the study area for a period of time (lower abundance), these measures (trail detection rates and occupancy estimates) will both decrease initially after Park/Preserve opening but then recover to pre-opening levels.

METHODS

Study area

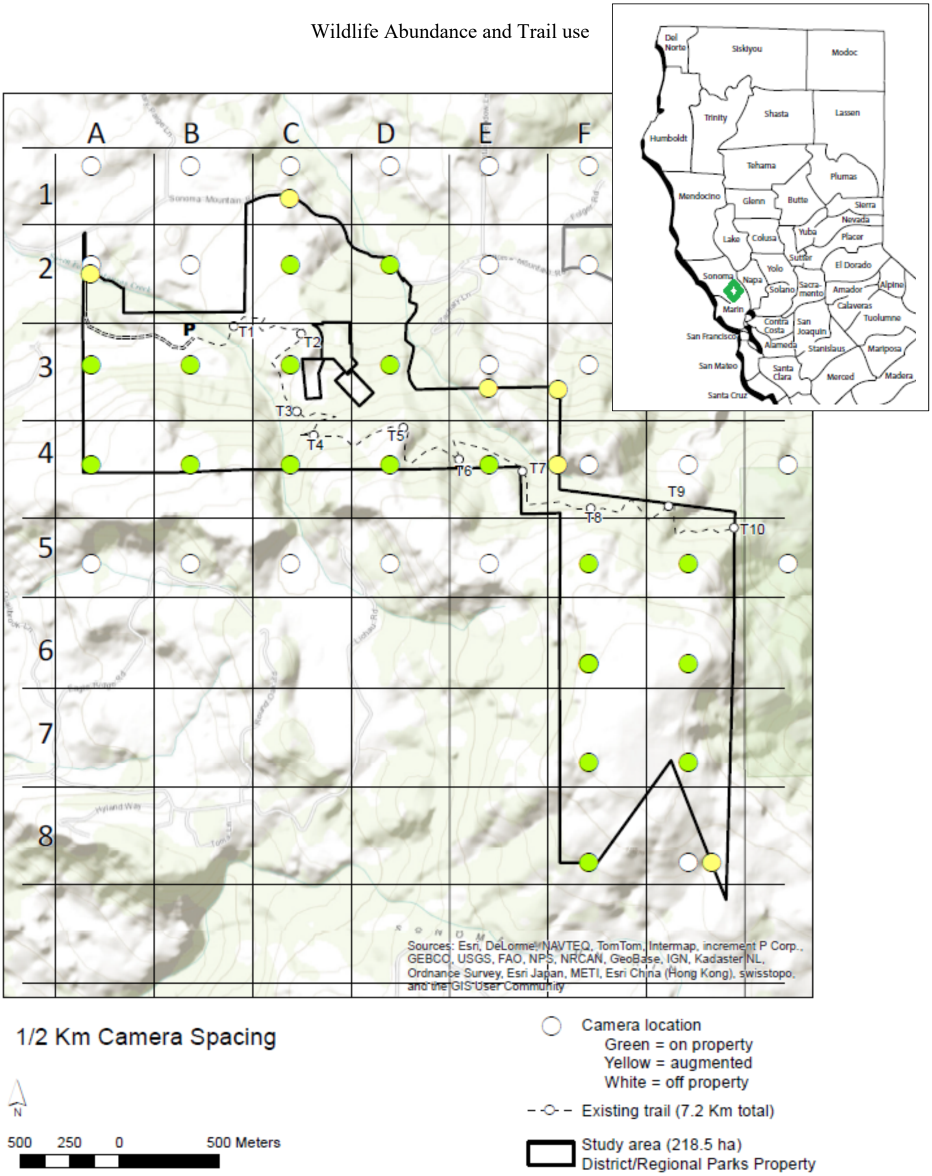
The 3.4 km^2 study area, North Sonoma Mountain Regional Park and Open Space Preserve (Park/Preserve; 38.3235 N, 122.5756 W, parks.sonomacounty.ca.gov/Visit/North-Sonoma-Mountain-Regional-Park/Park-Map/) is located in Sonoma County, California, USA (Figure 1). Sonoma County Agricultural Preservation and Open Space District (SCAPOSD) acquired the property and built the 5.95 km trail that ranges in elevation from 244 m to 750 m between June 2010 to September 2012. The Park/Preserve was then transferred to Sonoma County Parks in 2014 and opened to the public on 14 February 2015. Cattle grazing occurred before and during the study in portions of the site that supported grasslands; the site had no exclusionary fencing dividing up the site.

This area is subject to a Mediterranean climate characterized by wet, cool winters and dry, hot summers. Habitats included non-native grasslands (warm grasslands), oak-bay woodland (montane hardwood), redwood forest, mixed forest with madrone (montane hardwoods), and remnants of coast live oak forest/woodland and California bay forest (Biodiversity Portfolio Report, https://www.bayarealands.org/explorer/#, Conservation Lands Network Explorer 2016, 1 December 2016; Bay Area Open Space Council 2011). Matanzas and South Fork Matanzas creeks run through the study area. The topography is characterized by the steep hillsides of Sonoma Mountain. The surrounding land use matrix is composed of low-density rural development, protected open space, vineyards, and grazed grasslands.

Study design

A north-south grid of 23 motion and heat-differential triggered camera traps, HCO SG550V IR Scouting Cameras [and replacement Bushnell Trophy Cams (model#119636c)] were set in a randomly-generated fixed array at 0.5 km intervals covering the entire Park/Preserve (“grid cameras”). We adjusted six camera coordinates by less than 200 m to fit within the study area prior to going in the field (see yellow circles on Figure 1). Species-specific single-season occupancy estimates were generated for four seasons before and after the Park/Preserve opened to the public (see Table 1). We placed ten additional cameras at 500 m intervals along the trail (“trail cameras”; Figure 1). We calculated seasonal trail detection rates (detections per 100 trap nights) as a measure of intensity of wildlife and human use for four seasons before and after the Park/Preserve was opened to the public (see Appendix I for a list of human use categories).

Camera trapping methodology.—We followed a camera trapping and data management protocol, which is a modified version from TEAM Network 2009 and O’Brien 2010. Grid cameras were uniquely identified by line letter and number (e.g., A1, A2, A3, etc.; Figure 1). We placed camera traps within 100 m of the pre-determined coordinate during field deployment. Camera traps were attached to a wooden stake or tree with a nylon strap. Camera height was standardized to detect a mammal approximately gray fox size at a distance of 2 m at a perpendicular angle. Eight of the ten trail cameras were mounted on trees, and, after the Park/Preserve opened, were outfitted with security boxes to prevent theft. We recorded location (GPS coordinates), habitat within which the camera was placed (open, closed, or mixed), and elevation during deployment. Habitat (vegetative structure) included just three



Sources: Esri, DeLorme, NAVTEQ, TomTom, Intermap, increment P Corp., GEBCO, USGS, FAO, NPS, NRCAN, GeoBase, IGN, Kadaster NL, Ordnance Survey, Esri Japan, METI, Esri China (Hong Kong), swisstopo, and the GIS User Community

Figure 1. Camera layout for grid (yellow and green circles) and trail cameras (T1-T10) with study area location (green diamond in inset map of California counties); North Sonoma Mountain Regional Park and Open Space Preserve, California, USA, 2014–2016.

Table 1. Seasons before and after park opening, beginning and end dates for seasonal analysis, and effort (trapnights) for trail (n = 10) and grid (n = 23) camera arrays in North Sonoma Mountain Regional Park and Open Space Preserve, California, USA, 2014-2016.

Before or after opening Park/Preserve	Season	Begin and end dates	Trail trapnights	Grid trapnights
Before	Spring	1 March–30 May 2014	591	1,251
Before	Summer	1 June–31 August 2014	601	1,266
Before	Fall	1 September–31 November 2014	656	1,508
Before	Winter	1 December 2014–13 February 2015	606	1,106
Opening		*14 February 2015*		
After	Spring	1 March–30 May 2015	245	1,019
After	Summer	1 June–31 August 2015	16	701
After	Fall	1 September–31 November 2015	540	1,200
After	Winter	1 December 2015−15 January 2016	146	587

categories: closed (closed canopy), mixed (mixture of open and some overhead canopy such as oak woodland intergrading with grassland or chaparral), and open (no overhead canopy usually grassland). All cameras were set to take three images per trigger (event), a five second interval between events, 6 MP image size, high sensitivity level, and time stamp "ON." We adjusted image size and sensitivity as needed to match field conditions and improve data collection.

To verify camera station functioning during set up and maintenance, we took photographs of whiteboards with date, camera station identification, region, and subregion. We maintained camera stations regularly for proper functioning. We downloaded images from SD cards into a Windows Explorer embedded file system; EXIF image data was exported using PIE software (Picmeta v.6.75, www.picmeta.com/) into .csv files. We (authors and C. Lafayette) catalogued images to species or highest taxonomic order attainable; one of the authors (SET) vetted for accuracy during data preparation. Birds and other non-mammalian taxa were not identified to species nor included in the analysis. We categorized humans into several categories including pedestrian, cyclist, or equestrian (see full list in Appendix I). Unidentifiable images ("unknowns") and blanks were recorded as such.

Statistical analyses

We prepared a species detected list for the study area and trail compiled from before and after the Park/Preserve opened (Appendix I). We calculated single-season occupancy estimates from the camera grid and trail detection rates (detections per 100 trapnights) for terrestrial mammals (squirrel-size and larger) and wild turkeys (*Meleagris gallopavo*) from the cameras placed on trails (only). Trail cameras were not used in calculating occupancy estimates.

We calculated camera trap days ("trapnights") as the number of 24-hour periods (0000 to 2359) that the camera trap was functioning for each season [spring (March-May), summer (June-August), fall (September-November), and winter (December-February)]. We aggregated trapnights by grid and trail (Table 1) and compiled detection histories for grid cameras.

We recorded detections as the maximum number of individuals for each species in an image in a burst of three (an “event”), which are taken when the camera trap was triggered by movement and/or heat differential. For example, in a burst of three images, one image recorded two deer, in the next, three deer and in the final image, a deer; 3 deer would be recorded for that detection (maximum number of individuals in an image detected during one event).

Occupancy Analysis.—An occupancy estimate (ψ) for each species detected for the season was obtained using the program PRESENCE (v3.2, www.mbr-pwrc.usgs.gov/software/presence.html; Hines 2016). We used single-season occupancy models to estimate initial occupancy estimates (ψ) and detection probabilities (ρ) for each species (Mackenzie et al. 2003). Occupancy models account for imperfect detection and provide unbiased estimates of occupancy. To apply these models, detection histories were compiled for each species at each camera station as a series of ones (detection) and zeroes (non-detection). Each day (24-hour period commencing at 0000) the camera station was up was considered a (re)survey. Each day the camera station was “down” or not functioning was treated as a missing value.

Two pre-defined models were run, and the model with lowest delta Akaike’s Information Criterion (AIC) was used to estimate probability of detection and occupancy (Hines 2016) . The first model assumes the same occupancy probability for all camera station locations and that detection probability (ρ) was constant across both camera station location and survey occasions (i.e., two parameters). The second model assumes that all camera station locations have the same probability of occupancy (ψ), but that ρ varies between the surveys—although at each survey occasion, ρ is the same at each camera station location. The software PRESENCE uses AIC to rank models (Burnham and Anderson 2002), which relies on rules of parsimony. In this case, twice the log-likelihood values at the maximum likelihood estimates were used to calculate the AIC values in model weighting.

Comparison of seasonal occupancy estimates and detection rates.— Single-season occupancy values were compared from the season before to the season after and plotted in a seasonal time series to compare to trail detection rates relative to occupancy estimates. We added linear trend lines in several time series figures to show trend from the first season (spring 2014) to the last season of the study (winter 2015-2016).

RESULTS

We set up camera traps during February 2014 and maintained them regularly until the study ended in mid-January 2016. Camera placement elevation ranged from 252 to 737 m in closed, open, and mixed habitat. Of the 23 grid cameras, four (17%) were set in closed habitat, four (17%) in mixed, and 15 (65%) in open habitat; of the 10 trail cameras, five (50%) were in closed habitat, two (20%) in mixed, and three (30%) in open habitat. The trail was located largely within closed habitat. The Park/Preserve was open (warm grasslands, 50%) with remainder mixed and closed (41.8% montane hardwoods and 6% redwood forest; Biodiversity Report, www.bayarealands.org/explorer/#, Conservation Lands Network Explorer 2016).

The composition of the wildlife community changed little from before and after the Park/Preserve opened (Appendix I). Common and expected species including large and medium-sized carnivores were detected; a California Species of Special Concern, the American badger (*Taxidea taxus*), was detected within the study area after the Park/Preserve was opened. Several rare and data-deficient species that may occur in this region were not detected [e.g., the western spotted skunk (*Spilogale gracilis*), ringtail (*Bassariscus astutus*), porcupine (*Erethizon dorsatum*), and black bear (*Ursus americanus*)].

Seasonal analysis and effort

We generated seasonal Park/Preserve occupancy estimates and trail detection rates for eight seasons (four seasons before and after, Table 1). Trail camera trap nights averaged 425 (range = 16–656) per season. Grid trapnights averaged 1,080 (range = 587–1,508) per season. Seasonal trapping effort varied due to stolen (and replaced) camera traps, data loss due to theft of SD cards, and increased trail use filling up the SD cards with images.

Before and after seasonal comparison of occupancy estimates

Five wildlife species exhibited changes in occupancy estimates in the first season after the park opened; opossum increased (*Didelphis virginianus)* and raccoon (*Procyon lotor*), striped skunk (*Mephitis mephitis*), coyote (*Canis latrans*), and puma (*Puma concolor*) declined (Figure 2a) in the spring post-opening. Seven wildlife species exhibited changes in summer occupancy estimates; five decreased: striped skunk, gray fox (*Urocyon cinereoargenteus*), coyote, puma, and wild turkey (*Meleagris gallopavo*), and two increased [opossum and bobcat (*Lynx rufus*), Figure 2b] in the summer post-opening. Four wildlife species exhibited changes in occupancy estimates in the fall following opening; three decreased (gray fox, puma, and wild turkey) and one increased (opossum; Figure 2c). Only one wildlife species, raccoon, exhibited changes (increased) in occupancy estimates in the winter post-opening (Figure 2d).

Trail use

Even though the trail was not officially open to the public, some pre-opening trail use by “humans” (pedestrians, staff and trail crew) as well as their dogs and cyclists was observed in consistently low numbers (Figures 3a-c). The Park/Preserve did not allow dogs, and dog detection rates remained low throughout the study period (Figure 3c). Human trail detection rates increased dramatically immediately after the park opened; 4,393 detections per 100 trap nights (spring 2015) from 148 the season prior to opening (winter 2014–15, Figure 3a). Cyclists increased from an average of 53 (range 4–64) pre-opening to 228 (range 77–338) post-opening. Aggregated wildlife trail detection rates decreased after Park/Preserve opening (Figure 3d).

Comparing Wildlife Occupany in the Park/Preserve and on the Trail

We compared wildlife species’ intensity of trail use (trail detection rates) with occupancy estimates seasonally before and after park opening.

Black-tailed deer. —Black-tailed deer occupancy increased post-opening (Figure 4a) and trail use decreased for two seasons then returned to pre-opening levels (see Figure 4b).

Gray squirrel.— Gray squirrel occupancy was stable both before and after the Park/Preserve opened to the public (Figure 4a). Gray squirrels decreased trail use post-opening summer, fall and winter from pre-opening levels (Figure 4c).

Striped Skunk.— Occupancy of striped skunks decreased (slightly) post-opening

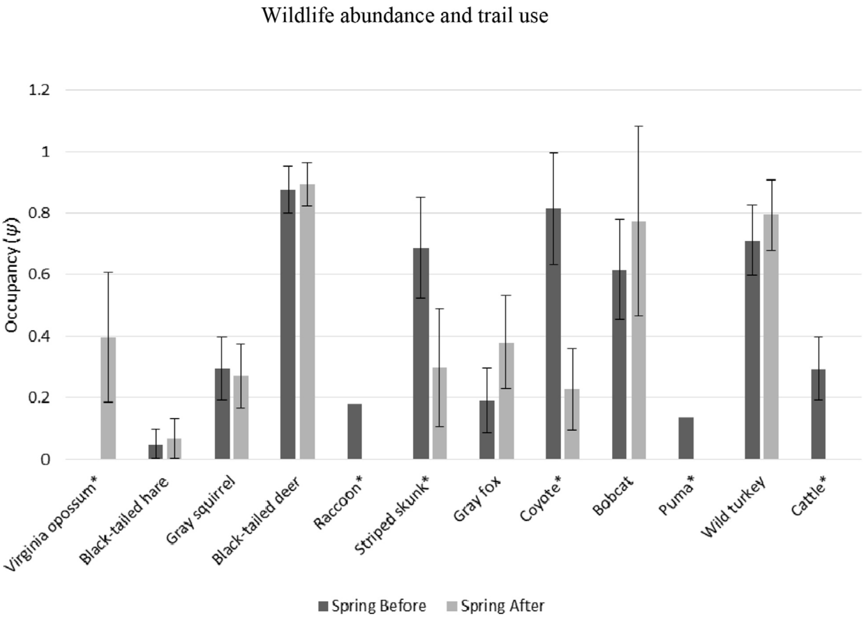


Figure 2a.

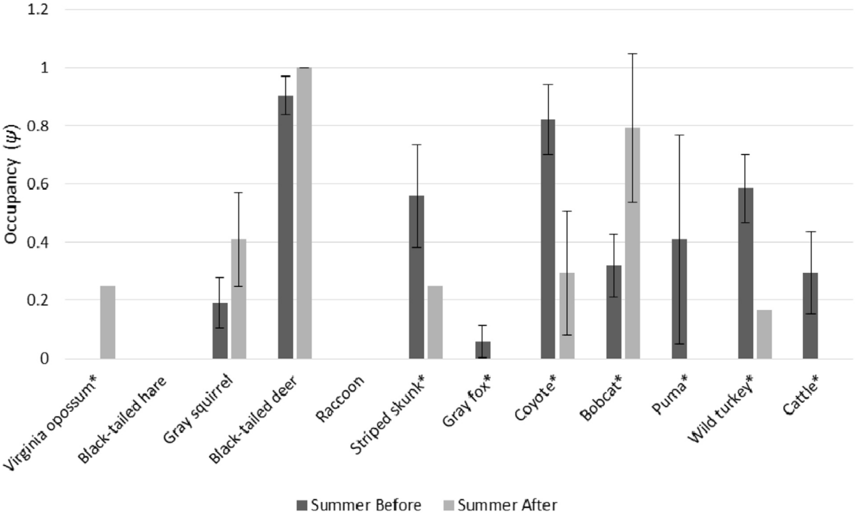
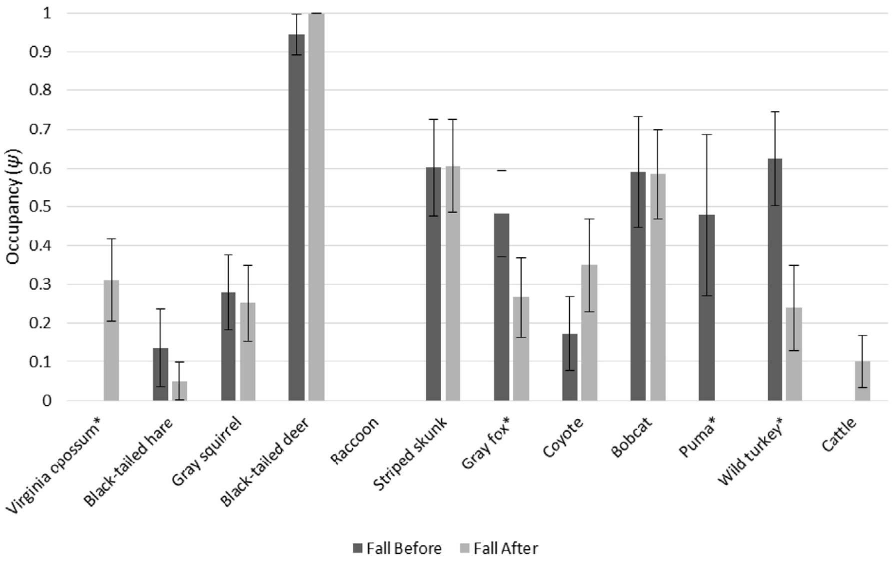


Figure 2b.

Figure 2a-d. Single-season occupancy estimates (error bars = ±SE) for wildlife species (* = difference noted between before and after occupancy estimates) in the a) spring before (2014) and after (2015), b) summer before (2014) and after (2015), c) fall before (2014) and after (2015), and d) winter before (2014_15) and after (2015_16) in North Sonoma Mountain Regional Park and Open Space Preserve, California, USA.



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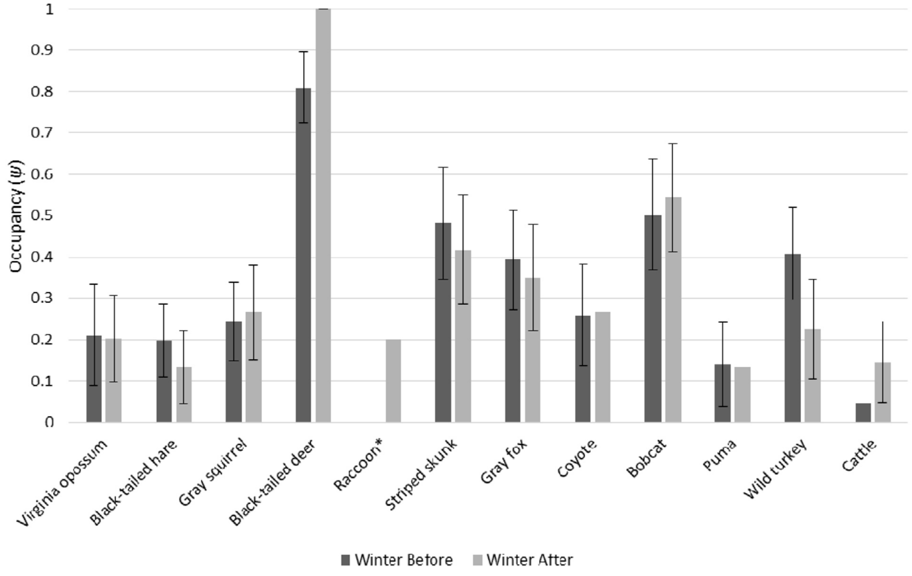

Figure 2d.

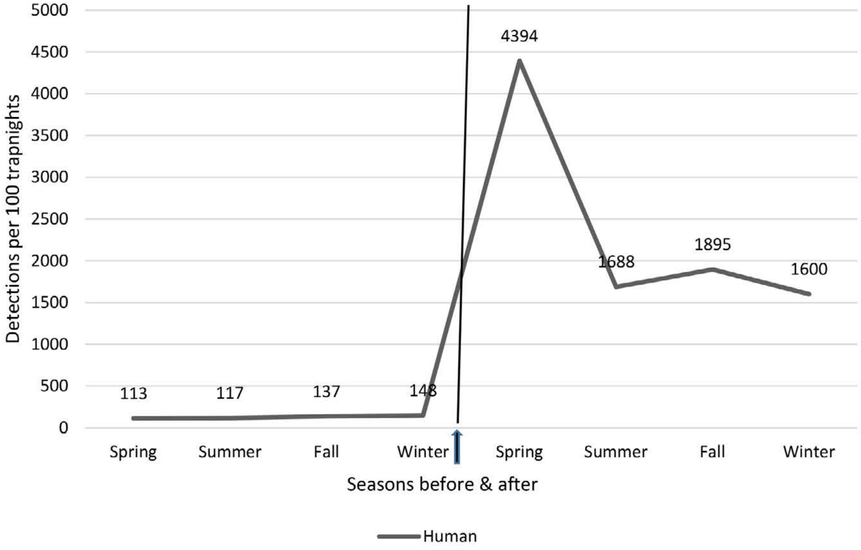

Figure 3a.

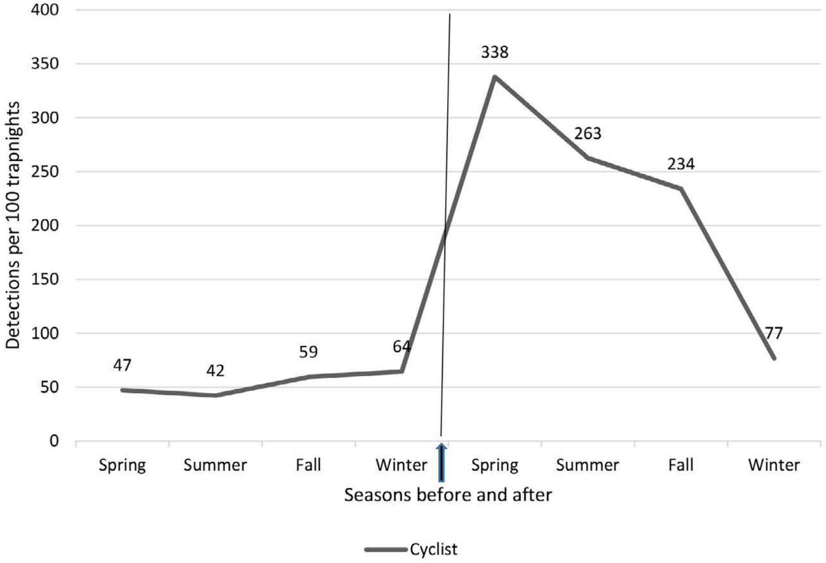

Figure 3b.

Figure 3a-d. Seasonal trail detections rates (detections per 100 trapnights) for before (spring 2014-winter 2015) and after (spring 2015-winter 2016) park opening (vertical line and arrow indicating 14 February 2015) for a) humans (non-cyclists), b) cyclists, c) domestic dog and livestock, and d) wildlife (linear = linear trend line) in North Sonoma Mountain Regional Park and Open Space Preserve, California, USA.

Wildlife abundance and trail use

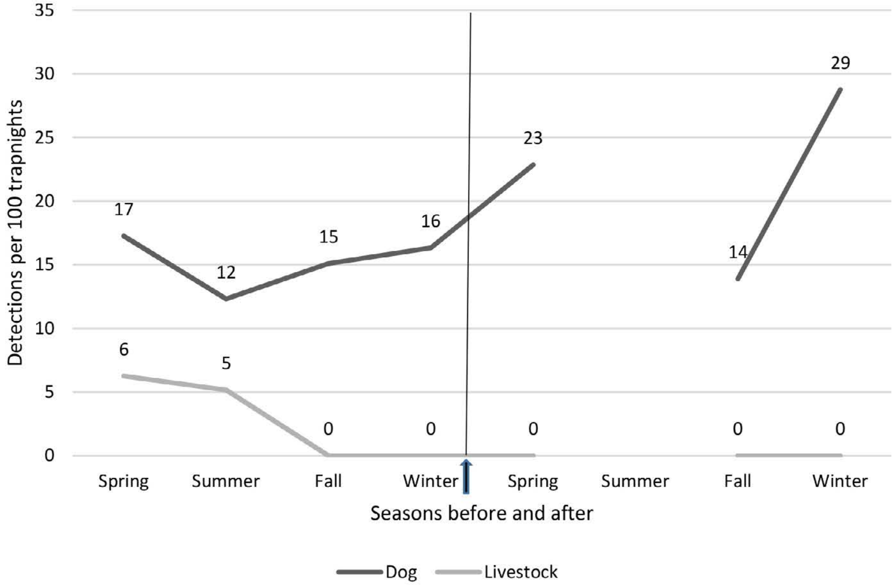

Figure 3c.

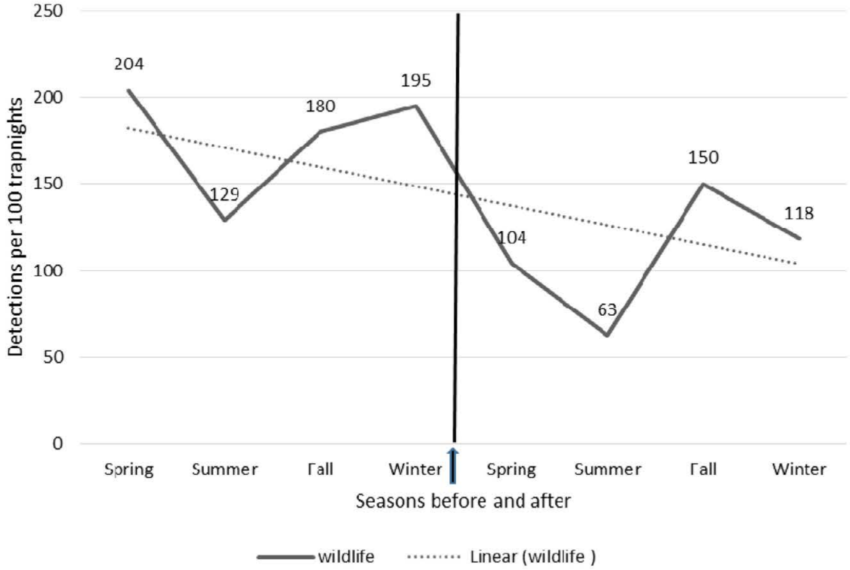

Figure 3d.

(Figure 5a). Striped skunk trail detection rates were the same post-opening for two seasons then increased to rates greater than pre-opening (Figure 5c).

Wild turkey.—Wild turkey increased in occupancy in the spring following Park/Preserve opening and decreased trail use (detection rates) post-opening (Figure 5b and 5d). Wild turkey had lower occupancy estimates and trail detection rates for post-opening summer, fall and winter.

Puma.— Puma occupancy fell to zero post-opening then increased after 3 seasons ($\psi = 0.13$, Figure 6a), potentially indicating some latency to recover. Puma decreased trail use post-opening (Figure 6c).

Bobcat.—Bobcat occupancy increased slightly in the Park/Preserve (Figure 6b) and decreased slightly in trail use (Figure 6d) post-opening.

Coyote.— Coyote occupancy decreased prior to the Park/Preserve opening and then remained relatively stable (Figure 7a). Trail use remained stable with a slight increase post-opening (Figure 7c); trail use was similar to patterns of occupancy.

Gray fox.— Gray fox occupancy was stable and similar to pre-opening occupancy (Figure 7b). Trail use was similar to patterns of occupancy (Figure 7d).

DISCUSSION

By our measures within this one study area, the wildlife that were the most affected by increased human trail use were puma and wild turkey, both decreasing in study area occupancy estimates, which we are using to detect changes in abundance and detection rates, which we are using as a measure of intensity of trail use. Additionally, the striped skunk notably increased trail use the third (fall) and fourth (winter) season after Park/Preserve opened. After two seasons post-opening, bobcat, gray fox, and coyote (three common mesocarnivores) appeared to be unaffected by public trail use both in abundance (as measured by occupancy estimates as an index of prevalence in the Park/Preserve) and trail use; these findings are consistent with a recent San Francisco Bay Area study (Reilly et al. 2016). The puma, which was present before the Park/Preserve opened, was then notably absent for three subsequent seasons post-opening. The majority of wildlife with the exception of the raccoon returned to previous occupancy levels the winter following opening (that is, after 9 months, Figure 2d).

Bobcat, coyote, and gray fox (mesocarnivores) showed little change in trail use, measured by camera detection rates on trail,and within the study area as indicated by by occupancy estimates from pre-opening measures, which support the null hypothesis, $H_{\emptyset}$ (Table 2); that is, that public trail use (at the rates we measured) did not appear to affect these species. Deer and gray squirrel showed decreased trail use despite no change in study area abundance post-opening, supporting H_1 that states that species change their trail use but not their overall use of the study area as measured by occupancy estimation. Puma and wild turkey decreased both trail use and abundance supporting H_2, which states that species will be affected by human trail use both on the trail and in the study area. Striped skunk increased trail use two seasons after opening and slightly decreased in abundance in the study area (see Table 2, Figures 5a and 5c). Deer may also have exhibited latency to habituation because their trail use resumed to pre-opening rates after two seasons (although it should be noted that human use declined; Figure 2a). Puma indicated latency to habituation for Park/Preserve abundance (Figure 6a).

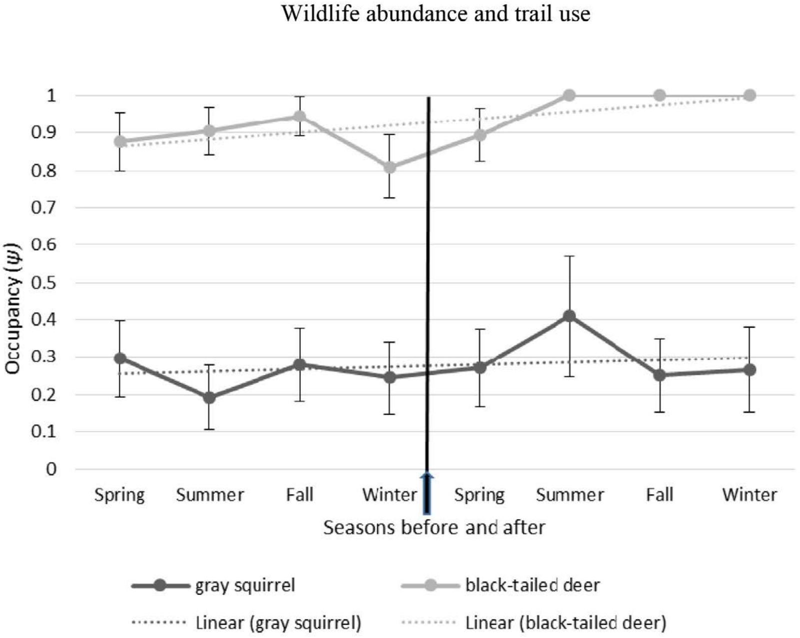

Figure 4a.

Figure 4a. Black-tailed deer and gray squirrel single-season occupancy estimates (ψ; error bar = ±SE, no error bar = no standard error) for seasons before (spring 2014–winter 2015) and after (spring 2015–winter 2016) opening (vertical line and arrow indicating 14 February 2015) in North Sonoma Mountain Regional Park and Open Space Preserve, California, USA.

In contrast to our findings, Reed and Merenlender (2008) conducted a study in the same region and found coyote and bobcat scat prevalence, as an indicator of animal presence, to be five times lower in protected areas that allowed recreation compared to sites that did not. Reilly et al. (2016), however, point out that carnivore scats are problematic as a surrogate for carnivore density because domestic dogs can consume these scats. Additionally, the human ability to visually detect scat is extremely low when compared to trained scat dogs for this purpose (i.e., humans detect only a very small fraction of scat that are present; Smith et al. 2005, Oliveira et al. 2012). Our findings were consistent with Reilly et al. (2016) that mesocarnivores appeared largely unaffected by public access and, additionally, that striped skunks increased trail use with recreational trail use.

The puma is the largest carnivore in the San Francisco Bay Area and is thought to play an important role in the ecosystem. Pumas are used as a surrogate to examine overall connectivity in the landscape due to its large body and home range size. Wang et al. (2015) examined puma behavioral responses to development and roads. According to their study, communication and denning required a four times larger buffer from human development. Findings from our study show a pattern of avoidance, at least, initially; pumas were detected very infrequently or not at all from the study area with commensurate lower trail use for three seasons post-opening; this finding was in contrast to puma adults and young consis-

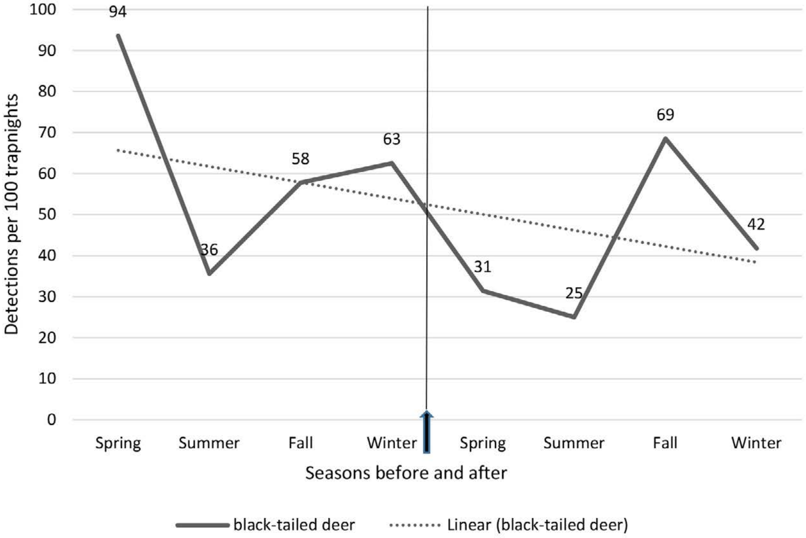

Figure 4b.

Wildlife abundance and trail use

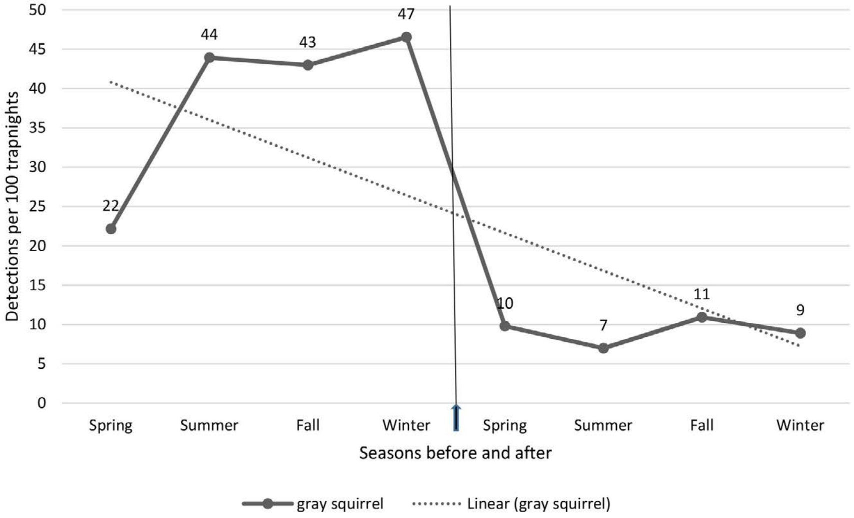

Figure 4c.

Figure 4b-c. Trail detection rates (detections per 100 trapnights) for b) black-tailed deer and c) gray squirrel for seasons before (spring 2014–winter 2015) and after (spring 2015–winter 2016) opening (vertical line and arrow indicating 14 February 2015) in North Sonoma Mountain Regional Park and Open Space Preserve, California, USA. Linear indicates linear trend line.

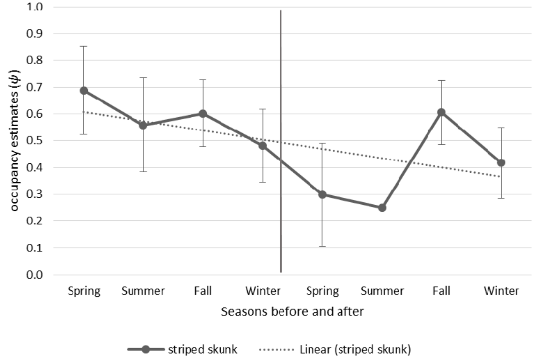

Figure 5a.

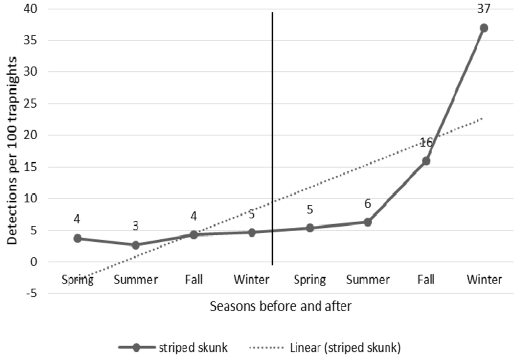

Figure 5c.

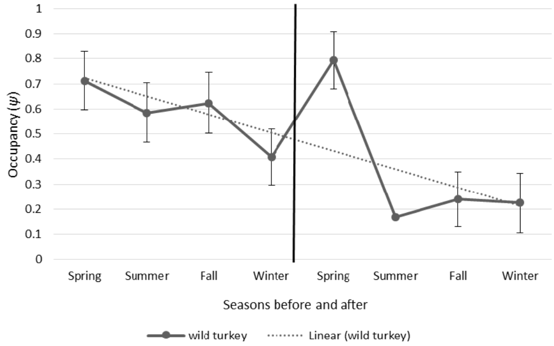

Figure 5b.

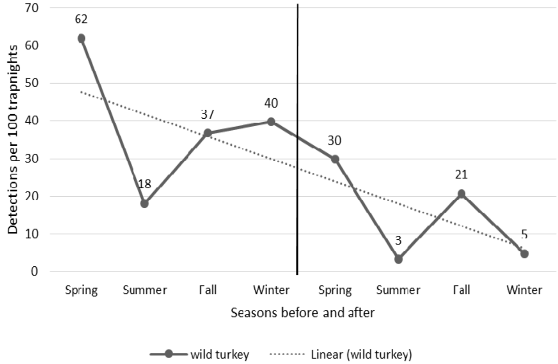

Figure 5d.

Figure 5a-b. Single-season occupancy estimates (ψ; error bar = ±SE, no error bar = observed occupancy) before (spring 2014–winter 2015) and after (spring 2015–winter 2016) opening (vertical line indicating 14 February 2015) for a) striped skunk and b) wild turkey in North Sonoma Mountain Regional Park and Open Space Preserve, California, USA. **Figure 5c-d.** Trail detection rates (detections per 100 trapnights) before (spring 2014–winter 2015) and after (spring 2015–winter 2016) opening (vertical line indicating 14 February 2015) for c) striped skunk and d) wild turkey in North Sonoma Mountain Regional Park and Open Space Preserve, California, USA. Linear indicates linear trend line.

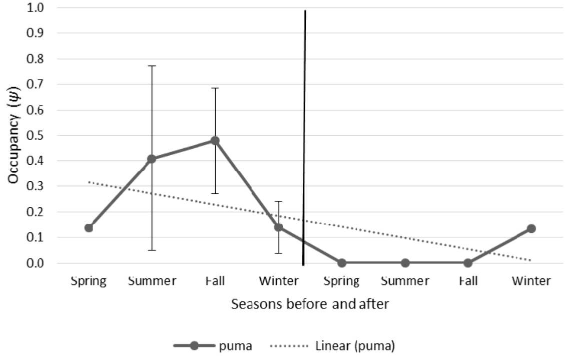

Figure 6a.

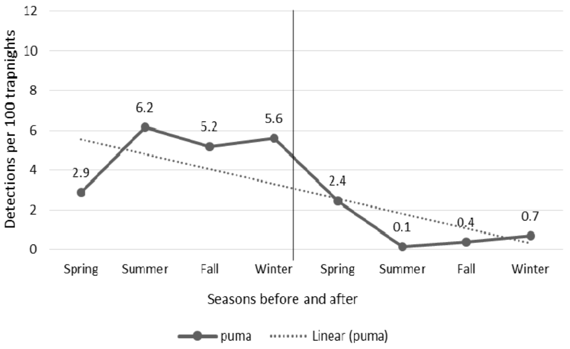

Figure 6c.

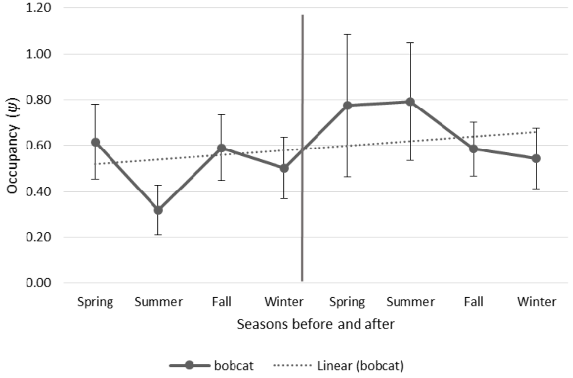

Figure 6b.

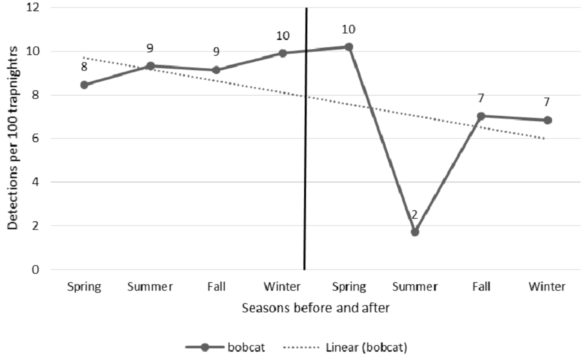

Figure 6d.

Figure 6a-b. Single-season occupancy estimates (ψ; error bar = ±SE, no error bar = observed occupancy) for seasons before (spring 2014–winter 2015) and after (spring 2015–winter 2016) opening (vertical line indicating 14 February 2015) for a) puma and b) bobcat in North Sonoma Mountain Regional Park and Open Space Preserve, California, USA. Figure 6c-d. Trail detection rates (detections per 100 trapnights) before (spring 2014–winter 2015) and after (spring 2015–winter 2016) opening (vertical line indicating 14 February 2015) for c) puma and d) bobcat in North Sonoma Mountain Regional Park and Open Space Preserve, California, USA. Linear indicates linear trend line.

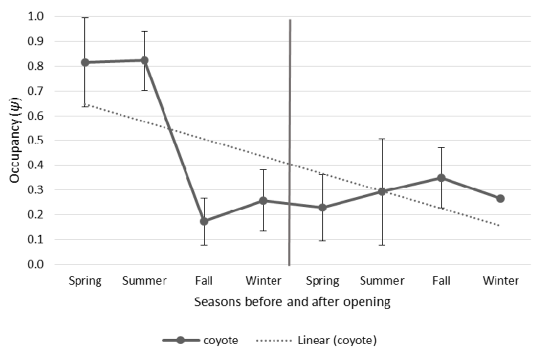

Figure 7a.

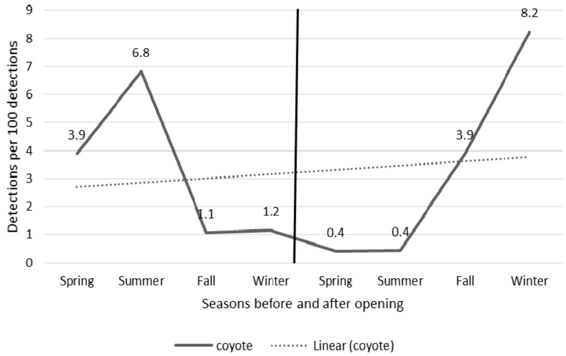

Figure 7c

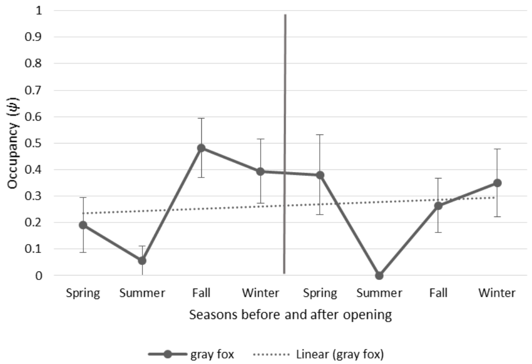

Figure 7b.

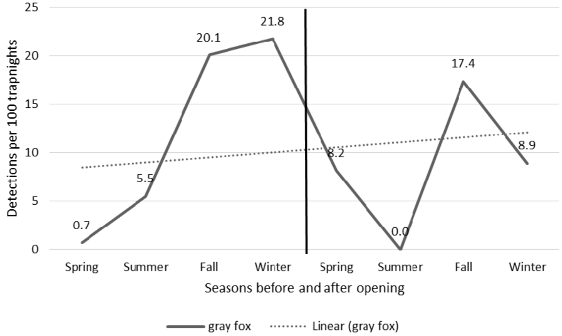

Figure 7d.

Figure 7a-b. Single-season occupancy estimates (ψ; error bar = ±SE, no error bar = observed occupancy) before (spring 2014–winter 2015) and after (spring 2015–winter 2016) opening (vertical line indicating 14 February 2015) for a) coyote and b) gray fox in North Sonoma Mountain Regional Park and Open Space Preserve, California, USA. Figure 7c-d. Trail detection rates (detections per 100 trapnights) before (spring 2014–winter 2015) and after (spring 2015–winter 2016) opening (vertical line indicating 14 February 2015) for c) coyote and d) gray fox in North Sonoma Mountain Regional Park and Open Space Preserve, California, USA. Linear indicates linear trend line.

Table 2. Which hypotheses are supported for selected wildlife species [Column headings: No change = no difference in trail use or Park/Preserve occupancy, Trail only = differences observed in trail use but not in Park/Preserve occupancy, Trail/Grid = differences observed in trail use and Park/Preserve occupancy, and Latency = recovery to pre-opening trail use and/or Park/Preserve occupancy values]. Under "Trail/Grid," minus sign indicates a decline and a plus sign indicates an increase for each respective array. An "X" indicates findings support the hypothesis. North Sonoma Mountain Regional Park and Open Space Preserve, California, USA, 2014-2016.

	Hypotheses			
Common name	No change (H_0)	Trail only (H_1)	Trail/Grid (H_2)	Latency (H_4)
Bobcat	X			
Coyote	X			
Gray fox	X			X?
Deer		X		X
Gray squirrel		X		
Puma			X-/-	X?
Striped skunk			X+/-	
Wild turkey			X-/-	

tently present in all seasons before the trail opened. Camera trap images of puma from the pre-opening year frequently had a mother with cubs or almost fully adult offspring.

Our study area represents an area with low to moderate human disturbance (both recreational and agricultural); therefore, the wildlife in our study have had exposure to humans, roads and other infrastructure. Naïve wildlife from more pristine areas (free from human influence) may behave differently to human presence on trails and may be affected for longer period of time and in a larger area; this factor (exposure to human influence) should be accounted for when planning trails and increasing recreational access. Undeveloped open space surrounding trails provides a buffer so wildlife can (initially) move away from novel human presence or disturbance even if they are able to habituate to human trail use over time. Certain species such as pumas may require large trail free "zones" near trails to habituate over time and to successfully fulfill the full suite of life history activities such as hunting, reproduction and raising young.

Finally, for this specific study area and trail, wildlife was documented using trails even with a marked increase in human use (pedestrians, cyclists and equestrians); wildlife trail use did not drop to zero with the exception of wild turkeys and puma (at least for 3 of the 4 seasons following opening). Additionally, the apparent habituation after a period of time indicated that much of the local wildlife community, but not all, may be resilient to an increased presence of humans on a trail given time to adjust; it also should be noted that the cyclist detection rates decreased to pre-opening levels of use by the 4th season after opening, so as an alternative explanation, wildlife trail use may be able to tolerate relatively high levels of human use (1600 detections per 100 trapnights) with lower levels of cyclists (77 detections per 100 trapnights compared to a high of 338 after opening)

Land acquisition and preservation can go a long way toward ensuring future open space for wildlife; however, without commensurate wildlife monitoring, particularly for things like trail building and increased human access, with concomitant changes occurring in the surrounding landscape (e.g., traffic intensity, climate change, development, fencing), the actual benefit of that land to wildlife over time will remain unknown. From a management perspective, this "unknown" is a lost opportunity. Identifying thresholds of human use beyond which wildlife or particular species are unable to adjust may differ with various disturbance regimes and for different life history needs (e.g., foraging and movement versus breeding). Determining these thresholds and for which species are important next steps in understanding the impacts of recreationalists on wildlife. Through studies that capture pre-impact conditions as well as a post-impact timeframe that is meaningful for wildlife, open space effectiveness as a conservation tool can be measured, evaluated and improved.

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Author Contributions

Conceived and designed the study: SET
Collected the data: SET, SH, MH
Performed the analysis of the data: SET
Authored the manuscript: SET
Provided critical revision of the manuscript: SH, MH, SH

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APPENDIX I. Human categories and wildlife species detected before and after park opening in each camera array for the North Sonoma Mountain Regional Park and Open Space Preserve, California, USA, 2014-2016.

Common name	Species	Grid before	Grid after	Trail before	Trail after
Human Cyclist			•	•	•
Domestic cat	*Felis sylvestris*		•		
Domestic dog	*Canis familiaris*	•	•	•	•
Equestrian		•		•	•
Hiker		•	•	•	•
Hikers with >2 dog				•	n/a
Human with dog				•	n/a
Staff				•	•
Vehicle		•	•	•	•
WPI crew		•	•	•	•
Ranger				•	n/a
Livestock					
Goats	(Goats)	•		•	•
Cattle	(Cattle)	•	•	•	
Wildlife					
Unknown	Unknown	•	•	•	•
Badger	*Taxidea taxus*		•		
Bird	(Bird)	•	•	•	•
Bat	(Bat)	•			
Black-tailed deer	*Odocoileus hemionus*	•	•	•	•
Black-tailed hare	*Lepus californicus*	•	•	•	•
Bobcat	*Lynx rufus*	•	•	•	•
Coyote	*Canis latrans*	•	•	•	•
Gray fox	*Urocyon cinereoargenteus*	•	•	•	•
Gray squirrel	*Sciurus griseus*	•	•	•	•
Opossum	*Didelphis virginiana*	•	•	•	•
Puma	*Puma concolor*	•	•	•	•
Raccoon	*Procyon lotor*	•	•	•	•
Striped skunk	*Mephitis mephitis*	•	•	•	•
Wild turkey	*Meleagris gallopavo*	•	•	•	•
Small rodent	(Small rodent)	•	•		•
Red fox	*Vulpes vulpes*			•	
Insect	(Insect)	•	•	•	
Lizard	(Lizard)	•			
Snake	(Snake)	•			

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A review of trail-related fragmentation, unauthorized trails, and other aspects of recreation ecology in protected areas

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Expanding levels of authorized and unauthorized non-consumptive recreation increasingly threaten sensitive biological resources in areas protected primarily or solely to conserve them. The majority of the documented effects on wildlife from non-consumptive recreation are negative. From a review of 84 papers in the recreation ecology literature about the effects of recreation on wildlife, the following topics emerged as warranting full consideration: trail-related internal fragmentation and expansion of the effect zone; the proliferation and use of unauthorized trails; disturbance thresholds; population-level effects; distinguishing facets of mountain biking; interpretation of observed behavioral responses by wildlife to recreation; magnitude and duration of responses; comparisons of effects among types of recreation and of results among studies; cumulative and synergistic effects; habituation; and the complexity of recreation ecology. Knowledge of these topics must inform efforts to cease the extant recreation-related exploitation of protected areas and to prevent it in the future. These efforts include: securing urgently needed perpetual monitoring, management, and enforcement commensurate with recreational pressure in dual-role protected areas to ensure the perpetuation of viable populations of focal sensitive species; preventing further use and proliferation of unauthorized trails; restoring areas damaged by inappropriate trails (i.e., unauthorized trails, unnecessarily redundant designated trails, and trails to be decommissioned); using science-based disturbance thresholds to develop management measures for recreation; using the best available science to guide all policy and decision-making about (1) the siting, design, and alignment of trails, and (2) the types, levels, and timing of recreation under consideration; and, planning separate protected areas and recreational areas in the future.

Key words: dual-role protected areas, effect zone, disturbance thresholds, internal fragmentation, mountain biking, non-consumptive recreation, perpetual monitoring/management/enforcement, recreation ecology, recreation-related disturbance to wildlife, unauthorized trails

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Conservation of habitats is a key strategy for conserving biodiversity worldwide (Pickering 2010a; Soulé and Noss 1998). The core function of many areas in California protected for conservation is to ensure that the wildlife species living in them thrive in what is the nation's most biologically diverse state (CDFW 2015).[1] Areas protected for conservation (protected areas) include locally owned lands (e.g., county and city reserves), state-owned lands (e.g., ecological reserves, wildlife areas, state parks), federally owned lands (e.g., national wildlife refuges, wilderness areas), and privately owned lands (e.g., conservation easements, conservancy lands, mitigation banks and lands). Here, the focus is on protected areas conserved primarily or solely for the perpetuation of viable populations of sensitive species (i.e., species whose persistence is jeopardized).[2] These protected areas often serve a dual role of conserving biodiversity and providing nature-based recreational and educational opportunities for millions of people, despite the evidence that even non-consumptive recreation[3] may not be compatible with protected areas' core function (Reed and Merenlender 2008; Larson et al. 2016; Dertien et al. 2018; Reed et al. 2019).

Recreation ecology is the scientific study of the ecological effects of outdoor recreation and nature-based tourism activities and their effective management in natural or semi-natural environments (Monz et al. 2013; Gutzwiller et al. 2017).[4] Studies in recreation ecology have shown that the majority of documented responses of wildlife species to recreation are negative (Steven et al. 2011; Larson et al. 2016; Hennings 2017; Patten and Burger 2018). Recreation-related disturbance to wildlife is recognized as a threat to global biodiversity, and as having wide-ranging and, at times, profound implications for wildlife individuals, populations, and communities (Dertien et al. 2018). Documented negative effects include detrimental changes to behavior, reproduction, growth, immune system function, and levels of stress hormones, and ultimately the survival of individual animals and persistence of wildlife populations and communities.

In this review, several topics about recreation ecology became apparent as warranting full consideration.[5] These topics are (1) the major issues of trail-related fragmentation and

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[1] Wildlife means all wild animals: insects, fish, amphibians, reptiles, birds, and mammals.

[2] These areas include areas protected pursuant to Natural Community Conservation Plans and/or Habitat Conservation Plans (NCCPs/HCPs). An NCCP is a comprehensive, single- or multi-jurisdictional plan that provides for regional habitat and species conservation at an ecosystem level while allowing local land use authorities to better manage growth and development. Upon issuing an NCCP Permit, the California Department of Fish and Wildlife (CDFW) can authorize take of certain state listed species and other species of concern, subject to the terms of coverage under the NCCP (CDFW 2015). An HCP is the federal counterpart to an NCCP; the U.S. Fish and Wildlife Service prepares HCPs and issues HCP permits. The terms and conditions under which an NCCP/HCP's protected areas are conserved establish the types and levels of public access that are permitted (Burger 2012). The types and levels of public access vary among the NCCP/HCP protected areas from no access to guided-only access to open access.

[3] In contrast to consumptive recreation (e.g., hunting, fishing), non-consumptive recreation is generally assumed not to directly extract a resource; it includes nature and wildlife viewing, beach-going, kayaking, hiking, biking, horseback riding, and wildlife photography (Reed and Merenlender 2008; CDFW 2016; Gutzwiller at el. 2017). From here forward, "recreation" means non-consumptive recreation, unless otherwise stated.

[4] From here forward, "management" includes monitoring, management, and enforcement. The level of enforcement necessary depends on the level of continual management implemented; generally, the more the management, the less enforcement is necessary. In addition, monitoring and management encompass both the natural resources and human users of the protected areas.

[5] The author read 71 articles and 13 reports about the recreation-related effects on wildlife; this paper does not cite all of them. All the articles are published in peer-reviewed journals. Some of the reports were peer reviewed and all were written by or contributed to by professionals in the fields of biology or ecology, though none of the reports were published in peer-reviewed journals to this author's knowledge (e.g., Burger 2012; Hennings 2017; Dertien et al. 2018; Reed et al. 2019). And, the totals exclude documents that are not explicitly about recreation-related effects on wildlife (e.g., Taff et al. 2019) and all newspaper articles.

expansion of the effect zone, unauthorized trail creation and use,[6] disturbance thresholds, population-level effects, and distinguishing facets of mountain biking, and (2) the following aspects of recreation ecology: the interpretation of observed behavioral responses by wildlife to recreation, magnitude and duration of responses, comparisons of effects among types of recreation and of results among studies, cumulative and synergistic effects, habituation, and the complexity of recreation ecology.

This paper discusses the issues identified above to inform efforts to cease the extant recreation-related exploitation of protected areas and to prevent it in the future. These efforts include: securing urgently needed perpetual management of recreation commensurate with recreational pressure to ensure the perpetuation of viable populations of focal sensitive species[7] as intended upon establishment of the protected areas; preventing further use and proliferation of unauthorized trails; restoring areas damaged by inappropriate trails (i.e., unauthorized trails, unnecessarily redundant designated trails, and trails to be decommissioned); using science-based disturbance thresholds; using the best available science to guide all policy and decision-making about the siting, design, and alignment of trails, and about the types, levels, and timing of recreation under consideration; and, planning separate protected areas and recreational areas in the future. This paper discusses the above-listed aspects of recreation ecology for consideration in designing field studies and while reviewing recreation ecology literature.

Trail-related disturbance: fragmentation, edge effects, and expansion of the effect zone

External fragmentation.—There is much peer-reviewed literature on the ecological effects of fragmentation, a process by which once-contiguous areas of habitat are physically separated by human disturbance creating a network of isolated habitat patches (Soulé et al. 1988; Ballantyne et al. 2014; Vickers et al. 2015; Cheptou et al. 2017). Most fragmentation research worldwide has concentrated on progressive losses of natural habitat through removal of vegetation as a result of development, agriculture, and resource extraction. Physical fragmentation, in conjunction with other related factors (e.g., duration of isolation of habitat fragments, low vagility of species, loss of genetic diversity), causes the isolated areas of habitat to experience a decay of species diversity over time due to local extinctions (Soulé et al. 1988). Consequently, fragmentation is a major threat to biodiversity (Cheptou et al. 2017). This fragmentation is considered external to the protected areas within a landscape, though it influences the viability of protected areas with respect to wildlife conservation.

Internal fragmentation.—Recreational trails themselves can fragment habitat, thereby causing fragmentation that is internal to the areas they traverse (Pickering 2010a; Leung et al. 2011; Burgin and Hardiman 2012; Pickering and Norman 2017). Because of their linear nature, trails can have a greater negative effect than if the affected terrain were consolidated in a more compact form (Pickering 2010a). Complex networks of trails within protected areas

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[6] The literature refers to illegally created trails and constructed trail features variously as unauthorized, informal, social, unofficial, off-trail, visitor-created, user-created, and demand trails. "Unauthorized" is the term of choice here because it is the only term among these that clearly denotes the illegality of the creation and use of such trails and features.

[7] Focal species are organisms whose requirements for survival represent factors important to maintaining ecologically healthy conditions; types of focal species include keystone species, umbrella species, flagship species, and indicator species. Focal species are identified for the purpose of guiding the planning and management of protected areas in a tractable way (Soulé and Noss 1998, Marcot and Flather 2007). Here, the term "focal species" is intended to include those species encompassed by the guild surrogate approach of conservation; this approach entails one member or a subset of members serving as a surrogate for other members of the guild (Marcot and Flather 2007).

can cumulatively affect nearly as much area as the above-mentioned external fragmentation (Ballantyne et al. 2014). Substantial evidence exists that trails may act as barriers to the movement of animals due to behavioral avoidance, the presence of a physical barrier, or development of a home range along the physical barrier (Burgin and Hardiman 2012). Trail density is a main factor influencing how wildlife respond to trail users and the ability of wildlife to disperse or reach seasonally important habitats such as breeding grounds (D'Acunto et al. 2018). Particularly when resulting from unauthorized trails or poorly sited and/or designed official trails, internal fragmentation can compound the negative effects of the external fragmentation in the surrounding landscape. The arterial spread of multiple cleared areas for trails within protected areas may cause losses of plant communities and ultimately result in long-term degradation of protected areas across large areas (Ballantyne et al. 2014).

Effects of trail presence on wildlife.—A likely consequence of internal fragmentation within protected areas is that the mere presence of trails, even in the absence of humans, can compromise protected areas' ability to sustain sensitive species (Pickering and Norman 2017; Baker and Leberg 2018). This is partly due to edge effects in the area of transition between two contrasting habitats, where resulting changes can occur in species abundance, community structure, and/or predation and parasitism (Zurita et al. 2012). Edge effects are major drivers of change in many fragmented landscapes (Laurance et al. 2007) and factor into the observations that internal fragmentation can restrict movement of some native animals and plants among habitat fragments and enhance the movement of invasive species along the trails (Barros and Pickering 2017). Baker and Leberg (2018) found that the presence alone of roads and trails, and not necessarily how often humans use them, had a significant negative effect on the occupancy of most of the 11 mammalian carnivore species they studied. Trails also potentially expose native animals to predators, including feral species such as the red fox (Vulpes vulpes), that penetrate natural areas by moving along the trails (Burgin and Hardiman 2012): a study on the effects of mountain biking on golden-cheeked warblers (Dendroica chrysoparia) found that the indirect effects from fragmentation and alteration of habitats from mountain biking trails may reduce the quality of the warblers' nesting habitat by increasing the vulnerability of warbler nests to predation by rat snakes (Elaphe obsoleta) and other edge-adapted predators (Davis et al. 2010). Edge effects associated with trails are known to affect other avian species similarly and to reduce the local abundance and nesting frequency of certain avian species, increase the incidence of nest parasitism by cowbirds, and affect avian vocalizations (Hennings 2017). The penetration of edge effects into the areas adjacent to trails is an aspect of internal fragmentation that underscores the ecological cost of unauthorized trails (Pickering and Norman 2017).

Trails expand the zone of effect.—Another notable consequence of trails is the expansion of the zone of effect of recreational disturbance to wildlife as habitats become more open, as occurs from the proliferation of unauthorized trails (Reed et al. 2019). In this context, "effect zones" are areas within which wildlife is disturbed by recreational activities on trails; effect zones encompass and extend beyond the area influenced by edge effects. The expanse of effect zones likely varies depending on the types and intensities of recreation and therefore may not be consistent across a trail network (Reed et al. 2019). Particularly in urbanized areas where protected areas are already highly confined in the surrounding urban matrix, the expansion of the effect zones further dissects and internally fragments what are already essentially habitat 'islands' (Balantyne et al. 2014; Pickering and Norman 2017).

The expansion of effect zones occurs in all protected areas with widespread trails irrespective of the sizes of the protected areas. For small protected areas (~300 ha) with dense trail networks, an effect zone of several hundred meters on either side of the trails can encompass a substantial proportion of the protected areas (Reed et al. 2019). In this way, effect zones reduce the proportion of a protected area that is suitable for various wildlife species (Reed et al. 2019), and can result in no contiguous areas across a protected area free from recreation-related disturbance to wildlife (Dertien et al. 2018).

The higher the level of recreation in protected areas, the greater the potential there is for the effects of trails and their use to extend beyond habitat loss and individual-level effects (behavioral and physiological) on wildlife into population- and community-level effects, including depletion of floral and faunal populations, alteration of trophic and community structures, and reduction of biodiversity (CDFW 2015). If habitat is available, wildlife may move to areas farther from trails, areas beyond the effect zone, to avoid recreation-related disturbance (Reed et al. 2019). However, the greater the proportion of a protected area occupied by effect zones, the fewer options there are for wildlife to move to areas outside the effect zones.

Unauthorized trails and technical trail features

General.—The implications to wildlife conservation of the disturbance to wildlife from trail-related fragmentation and expansion of effect zones are particularly grave with respect to unauthorized trails and recreational activities. The creation and use of unauthorized trails and technical trail features (TTFs) are commonplace and present concerns about the sustainability of biological resources in protected areas worldwide (Marion and Wimpey 2007; Newsome and Davies 2009; Ballantyne et al. 2014; Havlick et al. 2016; Barros and Pickering 2017).[8] Though most unauthorized trails and TTFs are readily visible and accessible, they are not officially planned or designed, approved for construction, managed, or part of a formally designated trail network (Davies and Newsome 2009; Leung et al. 2011; Hennings 2017). All user groups tend to create and use unauthorized trails, and there are several motivations for doing so, such as wanting access to trails closer to home or to engage in off-trail activities (Hennings 2017).

Though other recreationists venture off of designated trails, mountain bikers increasingly create unauthorized trails as they seek more challenging, wider-ranging, or free-riding opportunities (Havlick et al. 2016), or want a shortcut to reach specific destinations or to connect existing trails (Davies and Newsome 2009). If a trail is not sited in a place where bikers want to go, the off-trailing that results eventually forms trails (Davies and Newsome 2009).

Unauthorized trails expand the negative effects of human recreation on the flora and fauna of any protected area (Dertien et al. 2018). Similar to the above-discussed problems associated with internal fragmentation, unauthorized trails and recreational activities can negate the ecological benefits of both well-planned designated trails/trail networks and of prohibitions on access and activity (e.g., avoidance of breeding areas and seasonal access restrictions). The proliferation of unauthorized trails is often more responsible for trail-based fragmentation than formally designated trails (Ballantyne et al. 2014).

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[8] TTFs are created on mountain biking trails to increase the challenge of the ride. Examples of TTFs are jumps, ditches, mounds, bridges, ramps, ladders, drop offs, see saws, and 'skinnies' (i.e., narrow features that can be traversed) (Davies and Newsome 2009; Pickering et al. 2010c; Quinn and Chernoff 2010; Ballantyne et al. 2014; Havlick et al. 2016; Hennings 2017; Pickering and Norman 2017).

Even where unauthorized trails occupy a relatively small proportion of a landscape, they can be quite detrimental if in vital habitat; sensitive species whose territories or home ranges include the affected area(s) may be prevented via displacement or loss of habitat connectivity from accessing limited and essential resources (Gutzwiller et al. 2017). Wildlife can be more disturbed by off-trail than on-trail recreationists. For example, Taylor and Knight (2003) compared how mule deer (*Odocoileus hemionus*) respond to hikers and bikers using designated trails and one randomly chosen off-trail route. The deer exhibited a 70% probability of flushing from on-trail recreationists within 100 m from designated trails, whereas they exhibited a 96% probability of flushing within 100 m of recreationists located off trails, and their probability of flushing did not drop to 70% until the distance from the recreationists reached 390 m.

Examples.—Examples of protected areas affected by unauthorized trails include: 19 Natural Community Conservation Plan/Habitat Conservation Plan (NCCP/HCP; see footnote #2) protected areas in San Diego County, California where unauthorized trails comprise a mean of 45% (range: 8–85%) of the 1,206 km of trails mapped (Reed et al. 2014); an 829-ha area of the endangered Tall Open Blackbutt Forest in southeast Queensland, Australia, where 57% (26.5 km) of the 46.1 km of recreational trails was unauthorized when mapped in 2013 (Ballantyne et al. 2014); and, a 237-ha protected area in Argentina where 94% of the 19 km of trails found was unauthorized, resulting in landscape-level fragmentation and loss of vegetation (Barros and Pickering 2017). Another example of a protected area affected by unauthorized trails is the 191-ha Carlsbad Highlands Ecological Reserve in San Diego County. Though mountain biking is prohibited in this reserve, in addition to the 4 km of legal hiking trails in the reserve are also 27.4 km of unauthorized mountain biking trails and TTFs (E. Pert, South Coast Region, Regional Manager, California Department of Fish and Wildlife [CDFW], personal communication, 2019; Figure 1). This ecological reserve, so designated in 2000, comprises a critical component of an NCCP/HCP protected area and supports coastal sage scrub (a sensitive plant community), grasslands, thread-leaved brodiaea (*Brodiaea filifolia*, listed as threatened and endangered under the Federal and California endangered species acts, respectively), and several sensitive wildlife species: the federally threatened coastal California gnatcatcher (*Polioptila californica*), sharp-shinned hawk (*Accipiter striatus*), golden eagle (*Aquila chrysaetos*), white-tailed kite (*Elanus leucurus*), turkey vulture (*Cathartes aura*), and grasshopper sparrow (*Ammodramus savannarum*).[9]

Managing unauthorized trail creation and use.—Managing the rapid proliferation of unauthorized mountain biking trails and TTFs and their use is challenging. Even if only a small proportion of bikers is involved, the resulting vandalism can have serious ecological consequences as is well reflected in the statement, "[g]enerally when you ask people to stay out of the area no matter what the reason is, 80-90% obey you, [b]ut if you get 10% who don't obey you, you haven't done any good" (Bill Andree, retired district wildlife manager of Colorado Parks and Wildlife; Peterson 2019).

In the aforementioned Carlsbad Highlands Ecological Reserve, enforcement and education are necessary to substantially reduce the illegal riding, but the bikers monitor

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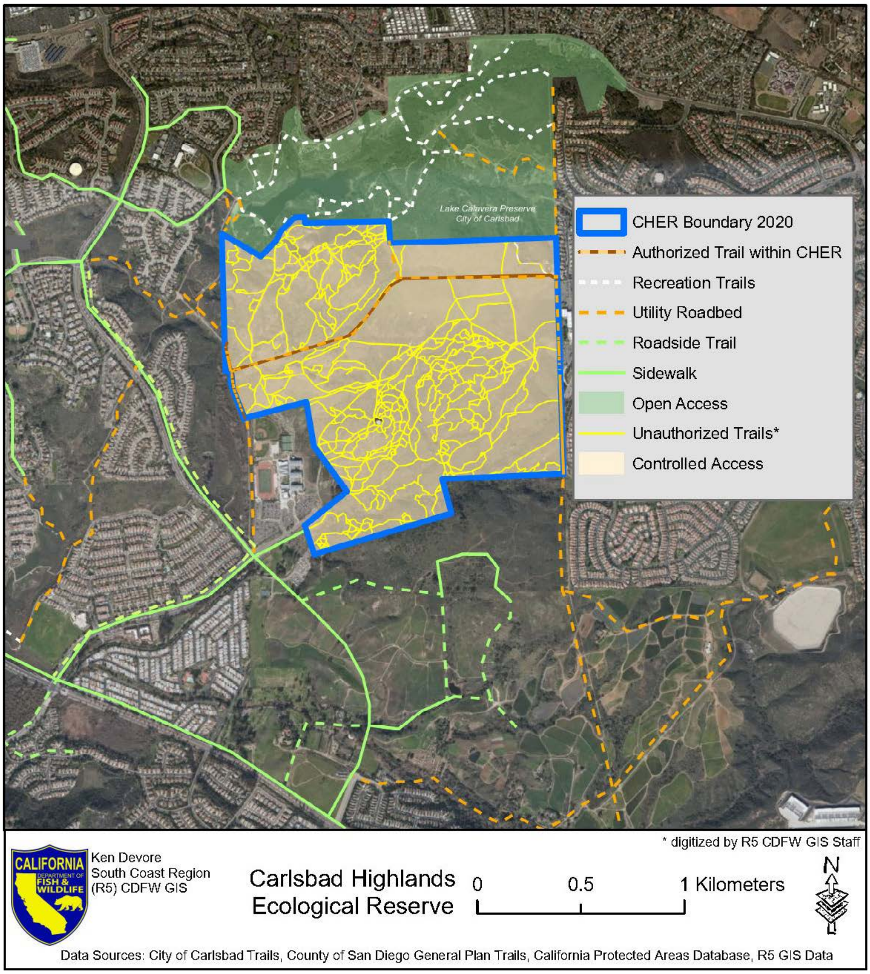
[9] Of CDFW's 136 ecological reserves (ER) statewide, biking is allowed on eight. About ERs, Title 14, California Code of Regulations §630(a) states, "All ecological reserves are maintained for the primary purpose of developing a statewide program for protection of rare, threatened, or endangered native plants, wildlife, aquatic organisms, and specialized terrestrial or aquatic habitat types. Visitor uses are dependent upon the provisions of applicable laws and upon a determination by the [Fish and Game] commission that opening an area to such visitor use is compatible with the purposes of the property."



Data Sources: City of Carlsbad Trails, County of San Diego General Plan Trails, California Protected Areas Database, R5 GIS Data

Figure 1. Carlsbad Highlands Ecological Reserve, Carlsbad, California. The yellow lines represent the unauthorized trails. Their associated effect zones occupy most, if not all of, the Ecological Reserve. (Credit: Ken Devore, South Coast Region (R5), GIS, CDFW 2017).

enforcement activity and recommence riding in the ecological reserve when enforcement officers leave (E. Pert, CDFW, personal communication, 2019). A similar protected area is the 350-ha Del Mar Mesa Preserve (Preserve) in the City of San Diego; the Preserve supports rare and endangered species such as Del Mar Manzanita (*Arctostaphylos glandulosa* ssp. *crassifolia)*, Orcutt's brodiaea (*Brodiaea orcutti)*, San Diego button celery (*Eryngium aristulatum* var. *parishii)*, San Diego mesa mint (*Pogogyne abramsii*), San Diego fairy shrimp (*Branchinecta sandiegonensis)*, and the California gnatcatcher, and was the subject of a study the City conducted to determine whether enforcement by CDFW Wildlife Officers (wardens) is an effective method to curb unauthorized trail uses (SANDAG 2015; Greer et al. 2017). Of the 32.22 km mapped trails on a 257-ha portion of this Preserve, 21.98 km are considered unauthorized (Reed et al. 2014). Prior to the study, City Park Rangers had conducted regular educational efforts in the field an average of 3–4 times monthly over a 17–month period. Despite the Rangers' efforts, non-compliance became the social norm as more users followed expanding numbers of unauthorized trails (Greer et al. 2017). The subsequent period of the CDFW Wildlife Officers' enforcement comprised 810 hours during a 12-week period with an unpredictable schedule. Prior to enforcement activities, the majority (78.7%) of the use within the study area was illegal, and over 85.5% of the illegal use was mountain biking. Illegal mountain biking decreased quickly during the enforcement period by 66.0% over the study period and stayed low during the 43-day post-enforcement period, while legal mountain biking remained the same. Other illegal use also decreased significantly, while other legal uses doubled (Greer et al. 2017). Greer et al. (2017) cite decades of research indicating that a combination of soft (i.e., education) and hard (e.g., warnings, citations, arrests, confiscation of bikes) enforcement is the most effective approach to promoting compliance. They assert that education becomes less effective in areas with chronic unauthorized trail creation and use.

Overall conclusions from Greer et al.'s (2017) study follow: (1) soft enforcement aimed at public education and redirecting social norms was not sufficient to curb unauthorized trail use in the Preserve; (2) open space enforcement by CDFW Wildlife Officers was determined to be effective in reducing unauthorized use in the Preserve; (3) the threat of sanctions (hard enforcement) has a more general utility and effectiveness in curbing non-compliant behavior than outreach to promote "awareness-of-consequence" of user actions (soft enforcement). The authors also concluded that social media has great potential to engage and educate the public on environmental issues, and that its use in combination with community policing can be a powerful tool to: redirect user attitude and subsequent behavior through peer-to-peer education about environmental impacts; answer questions regarding authorized uses; and, warn users of potential sanctions for non-compliance. They recommend the implementation of a social media component prior to and during enforcement efforts to help educate recreationists and reduce misinformation and recreationists' distrust of managers and enforcement personnel (Greer et al. 2017).

Paucity of information available.—Despite the global proliferation and use of unauthorized trails and TTFs and their far-reaching effects on wildlife in protected areas, there is a paucity of information of any depth available on such effects. The impacts of unauthorized trails and TTFs have been rarely documented (Marion and Wimpey 2007; Davies and Newsome 2009). A comprehensive literature search prior to 2010 produced only eight studies documenting the effects of unauthorized trails (Pickering et al. 2010c). Since then, additional studies have assessed the effects on vegetation from unauthorized trails, with little elucidation about their effects on wildlife. The proliferation, use, and wildlife-related effects of unauthorized trails remain understudied and insufficiently addressed. For protected areas where the creation and use of unauthorized trails and TTFs are prevalent, it is infeasible to fully assess the recreation-related effects on wildlife without including these activities and their effects. Yet, these effects have a great potential to impair the ability of protected areas to meet their conservation objectives.

Disturbance thresholds

Disturbance thresholds are predetermined levels of various measurable indicators above or below (depending on the indicator) which wildlife is disturbed (Hennings 2017).

These thresholds may be used to establish management measures such as minimum widths of spatial buffers between recreational trails and wildlife. Exceedance of a threshold may trigger the implementation of further management measures (Hennings 2017). Examples of disturbance thresholds are distance between people and wildlife or between trails and nesting sites (i.e., the distance within which wildlife species avoid people or trails), density of active trails above which wildlife alters its use of habitat, number of recreationists per day over which wildlife abundance decreases, duration of recreation, and number of recreational events per unit time (Hennings 2017; Dertien et al. 2018).

Thresholds should be set at levels equal to or more protective of predetermined levels of disturbance, and should be responsive to trends in changing conditions as identified by monitoring (Hennings 2017). Data from studies of recreational activities can be used to estimate quantitative thresholds of disturbance to wildlife (Dertien et al. 2018); however, determining these thresholds requires very specific empirical data (Rodríguez-Prieto et al. 2014).

While determining and using disturbance thresholds would be ideal for managers to optimize management decisions (Rodríguez-Prieto et al. 2014), they are difficult to determine for broad application. For example, thresholds established for distance to trail are not necessarily adequately protective of the focal species under all conditions in which they occur; a general rule of minimum thresholds for distance to trail cannot be established for some species, as individual variability within species can be high and can differ among populations, types of topography, and frequencies and types of human intrusion (González et al. 2006). As a result, the literature about recreation-related disturbance to wildlife provides limited information about quantitative thresholds for distance to trail (Dertien et al. 2018). Though their sample sizes (i.e., number of articles reviewed with such information) are accordingly small, Dertien et al. (2018) found the following examples of such thresholds: wading birds and passerines were generally affected at distances less than 100 m; larger-bodied species such as hawks and eagles had threshold effect distances greater than 400 m; small rodent species avoided areas within 50-100 m of trails or people; and some carnivores and ungulates had minimum effect distances up to 350-1000 m from trails and people.

As another example of a spatial buffer, Dertien et al. (2018) recommend a 200-m minimum buffer for ungulates; however, this would be insufficient for the circumstances of Taylor and Knight's (2003) study in which they found that mule deer showed a 96% probability of flushing within 100 m of recreationists located off trails, and the probability of their flushing did not drop to 70% until perpendicular distance reached 390 m. Two additional factors that influence the determination of spatial buffers are the density of the trail networks and the above-discussed effect zones. The smaller a protected area is and the denser its trail networks are, the greater the proportion of the protected area is occupied by effect zones, and the less likely it is that spatial buffers will protect the focal species from recreational disturbance (Wilcove et al. 1986; Ballantyne et al. 2014).

Land managers should consider both trail density and the level of human recreation before deciding on disturbance thresholds, since thresholds that work at lower levels of human activity may be ineffective when activity levels increase (D'Acunto et al. 2018). D'Acunto et al. (2018) simulated the success of trail closure strategies on reducing disturbance from Off Road Vehicles and pedestrians to nesting golden eagles during laying and incubation, focusing on eagle flushing behavior from the nest and alteration of foraging flight. They found that, for current levels of human recreation, the restrictive buffer (i.e. all trails closed within the buffer) was best at reducing flushing of incubating eagles, while closing all but the popular trails was best for foraging eagles. When the simulated human recreation was increased, trail density was the main factor influencing eagle flushing frequency.

Hennings (2017) reports the following thresholds for levels of human recreation (i.e., number of users) from four studies: for guanacos (*Lama guanicoe*), about 250 visitors per day, above which the number of birds observed declined; for sanderlings (*Calidris alba*), 20 visitors per day; for songbirds, eight out of 13 species showed thresholds ranging from 8-37 visitors per ha; and, for Mexican spotted owls (*Strix occidentalis lucida*), around 50 hikers per day. Regardless of any threshold effects, the majority of the research indicates that more visitors will generally cause more wildlife effects (Hennings 2017). However, since recreational impacts vary nonlinearly with use in a variety of ecosystems, a small number of visitors can have a disproportionate impact on sensitive species (Reed and Merenlender 2008).

Other aspects of recreation ecology to consider

Interpretation of observed behavioral responses.—It is possible to misconstrue the reasons for and implications of observed responses by wildlife to recreational activity. Traditionally and intuitively, species or individuals showing strong negative responses (e.g., readily flee or avoid) to human disturbance are those assumed to most need protection from disturbance. However, species with little suitable habitat available nearby cannot show marked avoidance of disturbance even if the fitness costs of the disturbance are high (e.g., reduction of survival or reproductive success; Gill et al. 2001). Conversely, species with many nearby alternative sites to move to are likely to move away from disturbance even if the fitness costs of the disturbance are low (Gill et al. 2001). It should not be assumed that the most responsive animals are the most vulnerable (Beale and Monaghan 2004). For example, in a controlled study of the behavioral responses of a shorebird (ruddy turnstone, *Arenia interpres*) to human disturbance (an approaching observer), Beale and Monaghan (2004) found that birds in better condition (i.e., supplemented with food) had longer flight initiation distances (i.e., flushed sooner) from the disturbance and searched for predators more frequently than control birds (i.e., not supplemented with food).[10] That is, birds responding most were actually the least likely to suffer any fitness consequences associated with the human presence; this is opposite from the response generally expected when behavior is used as an index of disturbance effects. Birds that had the most to lose by flushing, or otherwise changing their behavior in a manner that reduced feeding time, showed the least behavioral response; this could be interpreted incorrectly as meaning that these birds were not disturbed. Gill et al. (2001) assert that the absence of an obvious behavioral response does not rule out a population-level effect. In the same vein, it may be that species occurring in protected areas that are remnant fragments within urban landscapes are forced to utilize all components of the fragments, irrespective of their land-use intensity and land cover. This may occur if animals have nowhere else to go, and may be an explanation for instances when total relative abundance of birds is greater in urban and suburban reserves than in exurban reserves (Markovchick-Nicholls et al. 2008).

In addition to the reasons Gill et al. (2001) provide for an absence of detected effects, other possible reasons for finding no recreation-related effects include that there

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[10] Flight initiation distance is the distance from an approaching threat (e.g., recreationist) at which an animal begins to move away to escape from the threat.

may be a negative effect but it is not detected due to methodological issues. For example, the response variable examined (e.g., behavior versus physiology) and/or the number of replicates used compared to the amount of variation in the traits measured may not reveal the actual response of the species studied or the associated longer-term population-level effects (Steven et al. 2011). Furthermore, some studies may not include sufficiently high levels of human activity to detect responses from species that can tolerate lower levels of disturbance (Reed et al. 2019).

Threatened, endangered, and sensitive species.—Current research of recreation-related effects on wildlife does not include many species of urgent conservation concern (Larson et al. 2016). As many rare and isolated species tend to be specialists, anthropogenic activities could have a greater detrimental effect on the distribution, breeding success, and survival of individuals of these species (Beale and Monaghan 2004b; Bennett et al. 2013) than found in studies involving less sensitive species. Studies do not always reveal the strongest effects because the most disturbance-sensitive species are naturally rare in number or are already gone from disturbed sites (Hennings 2017). While recreation may not be the primary reason for the sensitive status of such species, it is a threat worth understanding for types of recreation that occur in the protected areas designated to conserve them (Larson et al. 2016).

Magnitude and duration of wildlife responses to recreation.— It is known that the nature (e.g., behavioral, physiological), magnitude, and duration of recreation-related disturbance to wildlife depend on a variety of factors, including, but not limited to, frequency and type of recreation, distribution of recreational use, season(s) of use, and environmental conditions (Marzano and Dandy 2012). Evaluating the effectiveness of measures to manage recreation can be complicated by the intensity of recreational use of a protected area because levels of use influence the magnitude of recreation-related effects on wildlife (Reed and Merenlender 2011). But studies do not always quantify the levels of recreational uses. Likewise, research seldom provides insight to the duration of wildlife species' response (e.g., nest abandonment, interruption of foraging/hunting, breeding, fleeing) to human disturbance (Marzano and Dandy 2012; Burger 2012; Larsen et al. 2016) or degree of response (e.g., how far wildlife moves away from human disturbance at a greater energetic cost and resulting in less availability of habitat). The same is true for the spatial scale at which wildlife response occurs (Burger 2012).

Generalized comparisons of effects among types of recreation.—It is clear from the literature that recreation in protected areas, particularly in more urbanized areas, can negatively affect wildlife (Larsen et al. 2016). However, it is difficult to make defensible generalized comparisons of the effects on wildlife among different types of recreation, partly because of the diversity of recreational activities, study methodologies, and observed responses (Monz et al. 2013). A comparison of results among similar studies indicates that sweeping conclusions about the effects of urbanization and human activity on wildlife need to be made with caution and are likely to be species-specific (Markovchick-Nicholls et al. 2008). For example, applying this caution to one species, the U.S. Fish and Wildlife Service (2000) concludes that attempts to ascribe relative importance, distinguish among, or generalize the effects of different human activities on bighorn sheep (*Ovis canadensis*) behavior are not supportable, given the range of potential reactions reported in the literature and the different variables impinging on given situations. Therefore, generalized comparisons of the effects on wildlife among different types of recreation are ill advised. The differences among types of recreation in their effects on wildlife are less important than the negative association for wildlife of human presence, irrespective of type of recreation (Patten and Burger 2018).

Despite the difficulty of making well-founded comparisons of the effects on wildlife among different types of recreation, comparisons are made. Among the types of recreation examined in the literature, the ecological effects of hiking and biking are most often compared. For studies done in the United States, this reflects the 22% increase to 8.3 million from 2006 to 2015 in mountain bikers, and the 24% increase to 37.2 million hikers during the same time period (Hennings 2017). And, notwithstanding the foregoing caveat about generalized comparisons, Hennings (2017) underscores that photographers, people with small children, bird watchers, and people engaging in loud conversations may be especially detrimental to bird communities because they are unpredictable and generally alarming. Photographers and wildlife watchers tend to stop, look directly at wildlife, and even follow them around, triggering stronger antipredator responses than people who simply pass by; photographers also tend to seek out rare species and look for nests. Also, curious, excited children tend to run around and shout in an unpredictable fashion (Marzano and Dandy 2012; Hennings 2017).

An absence of differences among effects.—The absence of differences among recreational activities' effects on wildlife does not equate to no effects. There can be similar levels of both benign or significant effects. For instance, in a study of bison (*Bison bison*) and pronghorn (*Antilocapra americana*), the authors found little difference in wildlife response (i.e., alert distance, flight initiation distance, or distance moved)[11] to hikers versus mountain bikers, but both species exhibited a 70% probability of flushing when within 100 m from trails with recreationists present (Taylor and Knight 2003).

Cumulative and synergistic negative effects.— The negative effects of recreation on wildlife compound, and may also act synergistically with, those from other influences (Larson et al. 2016; Reed et al. 2019). The cumulative negative effects of all anthropogenic influences on wildlife complicate efforts to minimize the effects and assess their population-level consequences (Pirotta et al. 2018). However, recreation ecology studies typically do not factor in other anthropogenic influences to which wildlife in protected areas are exposed (Pickering et al. 2010c; Erb et al. 2012; Messenger et al. 2014; Reed et al. 2019). Other anthropogenic influences include climate change and its associated effects on natural disasters; fires and other natural or human-caused disasters; consumptive recreation; non-recreational human activity such as habitat loss or alteration, the associated lack of connectivity, and the resulting loss of genetic diversity; poor air and/or water quality; invasive species; roads; vehicles; artificial light; prey declines; reverse zoonoses; drones; and noise (e.g., from vehicles, planes, ships, and boats). Recreation-related cumulative effects may be important if, for instance, the densities of different types of recreationists influence predator use of sites more than does the density of any one type of recreationist alone (Gutzwiller et al. 2017).

Wildlife habituation to human activity.—Habituation is a form of tolerance in which, as the result of a lack of negative consequences, there is a waning of response to a repeated, neutral stimulus (Whittaker and Knight 1998; Pauli et al. 2017). Habituation allows wildlife to use their energy for normal fitness-enhancing behaviors such as resting, foraging, and mating instead of fleeing when confronted with human activities that result in neutral outcomes (Whittaker and Knight 1998; George and Crooks 2006; Reilly et al. 2017). Habituation is

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[11] Alert distance is the distance from a stimulus at which an animal initiates vigilance behavior (Guay et al. 2016 in Reed et al. 2019); more specifically in this context, it is the distance between a recreationist and an animal when the animal first becomes visibly alert to the recreationist. Distance moved is the distance an animal travels from its initial position until it stops (Taylor and Knight 2003).

an apt description for crows (*Corvus* spp.) ignoring a scarecrow, or a red fox ignoring the human activity in a suburban area (Whittaker and Knight 1998). Citing several authors' work, Martínez-Abraín et al. (2008) identify level and frequency of disturbance, species, location, size and diet of species, and age of individual animals as factors that affect the degree of wildlife habituation to human disturbance.

The ability to habituate to predictable and recurrent human use of recreational trails may be an important behavioral adaptation for wildlife (González et al. 2006; Martínez-Abraín et al. 2008). However, habituated urban wildlife might be less likely to avoid contact with humans, which may increase the probability of human-wildlife conflicts and of attraction to anthropogenic food sources; both circumstances are considered problematic in many urban areas (Whittaker and Knight 1998; George and Crooks 2006). Wildlife habituation to humans may also increase wildlife aggression toward humans, or render wildlife more vulnerable to predators, hunters, poaching, or roadkill (Whittaker and Knight 1998; George and Crooks 2006; Marzano and Dandy 2012). Habituation of adult individuals may be associated with negative consequences for their offspring since habituation of adult animals does not translate to immediate habituation of juveniles (Reilly et al. 2017).

True habituation is not easily measured, and what appears to be habituation is often not (Hennings 2017). Apparent habituation is not a true measure of whether people are disturbing wildlife (Hennings 2017). Wildlife can experience significant stress without fleeing, and when this is misconstrued as habituation, disturbance effects on wildlife are underestimated (Hennings 2017). Care must be taken to avoid attributing a lack of observable response by wildlife to human presence as habituation (Beale and Monaghan 2004). Wildlife that seem not to avoid recreational disturbance may experience stress or be unable to leave a site if, for example, there is no suitable habitat nearby (Gill et al. 2001; Beale and Monaghan 2004; Markovchick-Nicholls et al. 2008).

While habituation to human disturbance could result in development of tolerance within a population (Pauli et al. 2017), Bötsch et al. (2018) infer from their findings on the recreation-related disturbance to birds in forests where recreation has occurred for decades that habituation to humans has not outweighed the effects of the disturbance. A long-lived species with low recruitment, such as the golden eagle, may be unable to experience individual learning or population-level evolutionary adaptation at a rate sufficient to compensate for a rapidly shifting anthropogenic landscape (Pauli et al. 2017).[12]

In a study subjecting captive female elk to four types of recreational disturbances (all-terrain vehicles riding, mountain biking, hiking, and horseback riding) over a two-year period, the elk showed no evidence of habituation to mountain biking. Similarly, elk travel time in response to hiking was generally above that of control periods, suggesting elk also did not habituate to hiking disturbance (Naylor et al. 2009).

In a study of how bison, mule deer, and pronghorn responded to hikers and bikers on designated recreational trails, Taylor and Knight (2003) found little evidence of habituation to recreationists among the species at the time of the study (summers of two consecutive years). In fact, the pronghorn at the study site did not habituate to largely predictable recreational use over a three-year period following the opening of trails at the site, and used areas that were significantly farther from trails than they had prior to the start of recreational use.

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[12] Evolutionary adaptation is the hereditary alteration or adjustment in structure or habits, the process by which a species or individual improves its ability to survive and pass on its genes in relationship to the environment (Ha and Campion 2019); unlike habituation, evolutionary adaptation does not result from learning during an individual's lifetime.

Hennings (2017) asserts that wildlife do not appear to habituate to the presence of dogs; impacts potentially linger after dogs are gone because the scent of dogs repels wildlife. It may be too that wildlife do not habituate to dogs (particularly off-leash dogs) because wildlife perceive dogs as predators and because they are unpredictable (Hennings 2016). Dog-specific disturbance has been studied for birds, with no evidence of habituation even with leashed dogs and even where dog-walking was frequent; the disturbance was much weaker for people without dogs (Hennings 2016).

The challenge of research.—Recreation ecology, similar to other fields of ecology, faces challenges in conducting statistically valid research (Quinn and Chernoff 2010). The degree to which and how the biotic and abiotic resources present in any one location respond directly or indirectly to recreational activities depends on many variables, some of which may be confounding (Figure 2, Table 1). Measuring the effects of human activity on wildlife is difficult because of the variability in the underlying spatial, diurnal, seasonal, and even the type of, indices being measured (Burger 2012). Recreation-related effects on wildlife vary among species (Larson et al. 2016) as different wildlife species respond differentially to visual, auditory, olfactory, and tactile stimuli (Hennings 2017). Wildlife responses to recreationists are likely influenced by a suite of variables that may differ in each field setting (Steidl and Anthony 1996; Taylor and Knight 2003), including level of human presence/activity that evokes a response as well as feedbacks and interactions with other factors (e.g., edge effects, availability of cover, exposure to disturbance, or time since fire; Patten and Burger 2018). Study methodology (i.e., design, sampling, data collection, and data analysis) itself encompasses many variables that dictate how other variables will influence the study outcomes. Even if methodology is consistent between/among two or more studies, other variables can result in different study results (Taylor and Knight 2003). Methodological issues may limit the inferences that can be made from the results (Pickering et al. 2010c).

Study design and statistical analyses can utilize methods to control for the effects of confounding variables (e.g., by using covariates). Statistical analyses can be used to examine alternative use-impact or use-response relationships between recreational activity and wildlife responses to assess the effects of recreational activity relative to other known drivers (e.g., habitat fragmentation, invasive species) of species occupancy, distribution, physiology, reproduction and survival (Monz et al. 2013; Reed et al. 2014).

Differences among study results.—Differences among studies' results can be due more to differences in variables not accounted for (e.g., space, diet, competition; Markovchick-Nicholls et al. 2008), study design, and/or analytical methodologies than to actual differences among species' responses to recreational disturbance. As to methodology, for instance, some studies may not include sufficiently high levels of human activity to detect responses from species that can tolerate lower levels of disturbance (Reed et al. 2019).

Reilly et al.'s (2017) study using camera trap data to quantify how hiking, mountain biking, horseback riding, and dog-walking affect habitat use/occupancy and diel shifts in activity patterns of ten mammalian species is illustrative for this discussion because some of its results differ markedly from those of other studies. For example, the authors found no negative association between recreation and habitat use by bobcats (*Lynx rufus*) and coyotes (*Canis latrans*), whereas Reed and Merenlender (2008) documented (in the same study area as Reilly et al.) densities of these two species more than five times lower in protected areas that permitted recreation versus those that did not. Dertien et al. (2018) identify differences in the following aspects of the two studies: field study methods, statistical analyses,

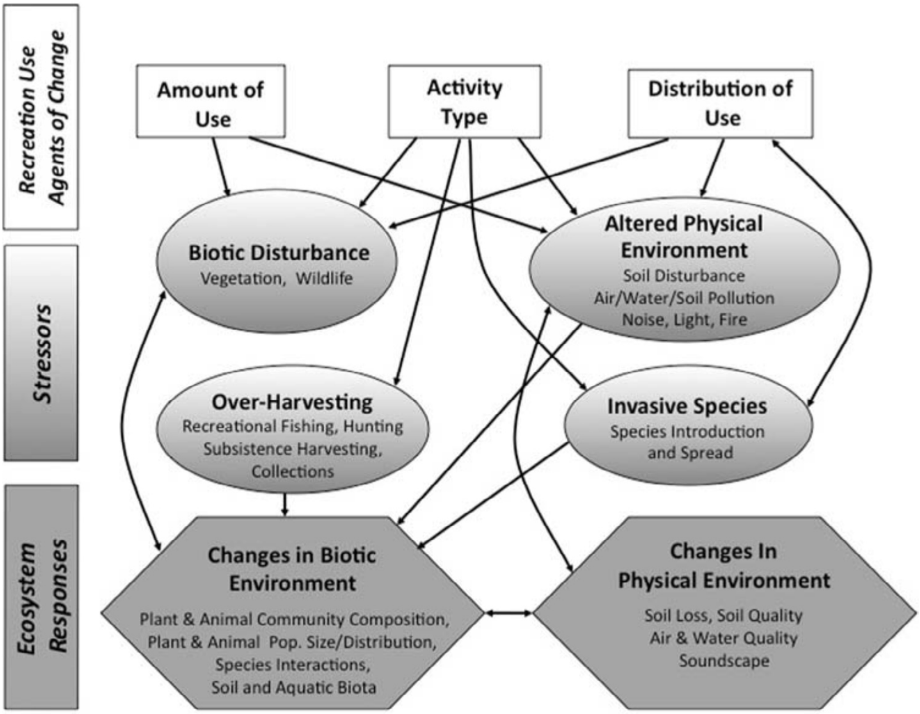

Figure 2. A conceptual model of ecological effects of outdoor recreation (Credit: Monz et al. 2010).

and research design – namely, types of study sites selected, treatment of data sources as replicates or independent of one another, and duration of data collection (one versus three years). These differences may have contributed to the greater variability observed in Reilly et al.'s (2017) study compared to Reed and Merenlender (2008).

Strong variability in other factors that are well known to influence mammalian distributions (e.g., habitat type, human development, or seasonal effects) make it difficult to conclude whether the potential effects of recreation on the target species were truly absent or simply undetected (Dertien et al. 2018). In addition, studies that use abundance, relative abundance, or species richness generally observe stronger effects of recreation than do studies such as Reilly et al.'s (2017) that use occupancy as a response variable (or occupancy interpreted as habitat use; Reed et al. 2019).

Reilly et al. (2017) acknowledge that: species vary widely in their responses to human activities; recreation-related effects on mammalian species that are rare or declining may be greater than on those that are more common or widely distributed; and birds, reptiles, amphibians, and small mammals may respond differently than the large and medium-sized mammals they studied. Finally, in contrasting their results with those of George and Crooks (2006), Reilly et al. do not acknowledge Gill et al.'s (2001) assertion that proximity to other suitable habitat influences how wildlife will respond to human disturbance; George and Crooks (2006) not only acknowledge but give credence to Gill et al.'s work.

Population-level effects

The foregoing discussion reveals many complexities of recreation ecology and provides a sense of why the population-level effects of human disturbance to wildlife are still poorly known (Burger 2012; Hennings 2017). Parameters used to measure population-level effects include population size, density, age structure, fecundity (birth rates), mortality (death rates), and sex ratio (Tarsi and Tuff 2012). Comprehensive assessments of the nonlethal effects on wildlife at the population level are rarely undertaken due to several constraints, including that robust assessment of these effects is challenging (Pirotta et al. 2018). Nonetheless, from a strictly conservation standpoint, human disturbance to wildlife is important only if

Table 1. Variables that influence the outcome of studies designed to assess the ecological effects of recreational activities. Each variable is mentioned in one or more of the cited articles (Taylor and Knight 2003; Beale and Monaghan 2004; Markovchick-Nicholls et al. 2007; Davis et al. 2010; Monz et al. 2010; Pickering 2010a; Quinn and Chernoff 2010; Burger 2012; D'Acunto 2018).

a. regional geophysical traits
b. size(s) of protected area(s) where research occurs
c. type(s) of vegetation present
d. area and density of vegetative cover
e. surrounding environment, including vegetation between the recreational activity and the target species
f. edaphic conditions (e.g., soil type, level of compaction, moisture, composition)
g. weather (temperature, precipitation, wind, shade, sun etc.)
h. timing (day / night / season)
i. time of day x location
j. design of trails (e.g., steepness of trails)
k. placement of trails (orientation to terrain - on flat, along a slope, across a slope)
l. direction of trails (ascending or descending)
m. spatial relationship between trails and target animals
n. trail density
o. wildlife present, target and non-target
p. total # of target wildlife individuals
q. spatial distribution of target wildlife
r. age classes and genders of target wildlife present (adult males/females, subadults, young of year)
s. reproductive status of target wildlife
t. fitness of target wildlife
u. predictability of recreational activity
v. degree of target animals' habituation to tested activities
w. duration of target animals' exposure
x. whether the target animals have the ability to retreat
y. type(s) of recreation
z. duration of recreational activity
aa. # of humans present (e.g., individuals or groups)
bb. # of human disturbances per day
cc. whether recreational activity is on or off an official trail
dd. recreationists' positions
ee. angle / trajectory of recreationists' approach to wildlife
ff. speed and style (e.g., 'aggressive') of recreationists' approach
gg. distance of recreational travel
hh. whether the recreationists apply best practices
ii. recreationists' behavior (e.g., talking or silent, continuous movement or stopping)
jj. encounter distance
kk. perpendicular distance
ll. encounter x perpendicular
mm. researcher bias
nn. study methodology (e.g., is recreationists' approach to wildlife direct or tangential, on or off trail; includes statistical analyses)

it affects survival or fecundity such that a population declines (Gill et al. 2001). Assessing and managing the nonlethal effects on wildlife populations has long been a goal of ecologists, land managers, and decision makers (Pirotta et al. 2018). The management of human activities that cause nonlethal effects on wildlife presents a fundamental ecological problem: how to understand the population-level consequences of changes in the behavior or physiology of individual animals that are caused by external stressors (Pirotta et al. 2018). Given the expansion of recreational activities that can disturb wildlife, quantitatively linking the effects of this disturbance to population dynamics is a major objective for modern conservation (Pirotta et al. 2018).

While behavioral responses, which are studied far more often than other types of responses (e.g., physiological; Larson et al. 2016), have the potential to affect survival or reproductive success, the actual fitness[13] costs of behavioral responses need to be quantified before the responses can be used as reliable estimates of population-level perturbations (Gill et al. 2001).

In most situations when statistical models are used to estimate or forecast the population-level effects of disturbance, selection of a model structure is likely to be driven by data availability (Pirotta et al. 2018). Collecting recreation data in conjunction with ongoing animal population monitoring efforts would be a valuable way to improve the understanding of the effects of human disturbance on demographic trends; and, studies that combine behavioral responses with physiological or demographic metrics would help calibrate the relationships between behavioral responses and population-level effects (Reed et al. 2019). Whichever models are used, uncertainty in the estimated population consequence can be reported as a distribution of potential outcomes, allowing the application of the precautionary principle if the results are used to make management decisions (Pirotta et al. 2018).[14] Application of the precautionary principle is warranted given that any simulation model simplifies reality (D'Acunto et al. 2018).

The dearth of conclusive evidence of recreation-related population-level effects in the literature does not mean that such effects are rare; logic dictates that, if the negative consequences of some observed behaviors or physiological changes in wildlife persist, negative population-level effects will eventually follow. For example, negative population-level effects on desert bighorn sheep (*Ovis canadensis nelsoni*) from recreational disturbance have been documented and are implicated in the bighorn sheep abandonment of habitat (and extirpation of the population) in the Pusch Ridge Wilderness in Arizona, USA (Longshore et al. 2013). And, recreation is one reason cited for the population of bighorn sheep in the Peninsular Ranges of California being listed in 1998 as endangered under the Federal Endangered Species Act (USFWS 2000).

The effects of hikers on elk (*Cervus elaphus*) provide another example of recreation-related population-level effects. Based on a two-year study of the response of female elk to the presence of back-country hikers during the calving season, Shively et al. (2005) recommended that some recreational closures be continued because, despite the evidence that elk reproduction can rebound from depressed levels when hikers are removed or reduced in

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[13] Fitness refers to reproductive success and reflects how well an organism is adapted to its environment (Hennings 2017).

[14] The central tenet of the precautionary principle is that precautionary measures should be taken even if some cause and effect relationships are not fully established scientifically. Generally, the four central components of the principle are: taking preventive action in the face of uncertainty; shifting the burden of proof to the proponents of an activity; exploring a wide range of alternatives to possibly harmful actions; and increasing public participation in decision making (Kriebel et al. 2001).

number, they could not determine if there is a threshold level of reproductive depression from which elk cannot recover. In fact, a 2019 article in *The Guardian* reported that the number of elk in the same herd Shively et al. (2005) studied had dropped precipitously since the early 2010s with the steady increase in recreation; what was once a herd of 1,000 head of elk, had dropped to 53 at last count in February of 2019 (Peterson 2019). The article explains that, for Bill Alldredge, one of the authors of the study, there is no other explanation than the increased levels of trail users in the area that supports this elk herd (Peterson 2019).

In a study to assess the effects of recreational activities on Iberian frogs (*Rana iberica*), an endemic species in decline and listed as vulnerable in the Spanish Red Data Book, Rodríguez-Prieto and Fernández-Juricic (2005) concluded that (1) the decrease in Iberian frog abundance with the proximity to recreational areas suggests that direct human disturbance affects this species at the population level, and (2) overall, the results suggest that direct human disturbance needs to be considered as a potential factor affecting amphibian populations with low tolerance for disturbance.

From the peer-reviewed recreation ecology literature, Steven et al. (2011) compiled 69 journal articles that describe the results of original research examining the effects of non-motorized nature-based recreation on birds. Among the articles were 33 that examined population-level avian responses (i.e., reproductive success including number of nests, number eggs laid, and number of chicks that hatched or fledged). Negative effects were reported in 85% of these 33 articles.

Patten et al.'s (2017) 10-year study of mammalian populations across the County of Orange Central and Coastal NCCP/HCP protected areas coincided with a marked increase of human activity and provides insight to potential population-level effects. Though the authors did not discern a decline in the populations studied, they did discern temporal and spatial shifts by wildlife due to human presence, and they suggested that the associated losses in prey populations are unsustainable in light of additional stressors these populations face, which range from continued loss of habitat to human disturbance in the protected areas. Furthermore, given the avoidance behavior and temporal shifts of the various mammalian species, any further increase in human disturbance may yet drive mammalian populations downward (Patten et al. 2017).

With regard to population-level effects of anthropogenic fragmentation, evolutionary adaptation to such fragmentation has received some attention. Even when adaptation to fragmentation occurs, it may not be enough to fully compensate for the environmental effects from fragmentation, and in some cases may even exacerbate them (Cheptou et al. 2017).

Distinguishing facets of mountain biking

Together with the extent of the above-discussed creation and use of unauthorized trails and TTFs by mountain bikers, the mass-marketing of the sport, and the very large numbers of mountain bikers (Burgin and Hardiman 2012), at least four facets of mountain biking distinguish it from other recreational activities such that it may be of potentially greater concern with respect to its effects on wildlife than yet accounted for in the literature. These facets are distance traveled, speed of travel, biking in the dark, and political lobbying and advocacy.

Distance traveled.—Bikers traveling faster obviously travel farther than hikers per unit time and could therefore disturb more wildlife than hikers per unit time (Taylor and Knight 2003; Burgin and Hardiman 2012); the same applies to bikers and equestrians when bikers travel faster than equestrians. Larson et al. (2016) reasoned that, since motorized activities often cover larger spatial extents than non-motorized activities, it is possible that the effects of motorized activities have been underestimated. The same logic applies to the distances traveled by bikers and hikers. For valid comparisons among recreation-related ecological effects, the comparisons must account for distances traveled and the associated levels of disturbance to wildlife along the entire route traveled.

Speed of travel.—While recreation-related effects on wildlife are generally assumed to be indirect (Dertien et al. 2018), the speed at which mountain bikers travel, combined with their relatively quiet mode of travel, can result in direct disturbance to wildlife. A relatively fast moving, quiet mountain bike may approach an animal undetected until well within the animal's normal flight response zone. The result may be a severe startle response by the animal with significant consequences to the animal and/or the mountain biker (Quinn and Chernoff 2010). The sudden encounter is the most common situation associated with grizzly bear (*Ursus arctos horribillis*) inflicted injury (Quinn and Chernoff 2010). Biking-caused wildlife fatalities likely resulting because of bikers' speed occur with amphibians and reptiles that may be attracted to trails for thermoregulation and are thus exposed to collision with bikes' wheels (Burgin and Hardiman 2012); photo-documentation provides evidence of three such fatalities in CDFW's Del Mar Mesa Ecological Reserve in San Diego where a San Diego horned lizard (*Phrynosoma coronatum blainvillii*, a species of concern under CDFW and the U.S. Fish and Wildlife Service), three western toads (*Anaxyrus boreas*), and two Baja California treefrogs (*Pseudacris hypochondriaca*) were killed by mountain bikes (J. Price, CDFW, personal communication, 2019). The treefrogs appear to have been mating when run over—the photo documentation shows eggs spilling out of the female. Biking is prohibited in this ecological reserve, and two of the run-overs occurred on unauthorized trails (J. Price, CDFW, personal communication, 2019).

Though there are methods (e.g., bells attached to bikes) for mountain bikers to give warning of their approach to other trail users, and these can be effective for this purpose, these methods themselves can introduce additional disturbance to wildlife. And, such warning sounds are ineffective for wildlife whose hearing range does not detect them or who do not hear them soon enough to avoid a collision. Moreover, when recreationists are visible on approach to wildlife, the more threatening (e.g., faster, more direct) the recreationists appear to wildlife (as potential predators), the greater the flight initiation distance from the recreationists (Stankowich 2008). Fleeing from a perceived predator represents potentially needless expenditure of valuable energy.

Biking in the dark.—Mountain biking in the dark (i.e., night riding), which is on the rise in protected areas, can disrupt the natural balance between diurnal and nocturnal wildlife. Consequently, night riding poses a dual threat to wildlife that exhibit diel shifts toward night: night riding can compound the pressure such wildlife experience from daytime recreational activities by increasing encounters with competitors and even further reducing the time available for foraging and breeding (Reilly et al. 2017). Night riding can also startle naturally nocturnal wildlife and wildlife that has become increasingly nocturnal to avoid daytime recreationists and other anthropogenic disturbances. Generally, temporal shifts by wildlife involve disruptions to both the shifting wildlife and to the wildlife naturally active during the time frame the shifting wildlife move into. In this way, such shifts set both groups of wildlife up for conflict and competition, disrupt predator/prey relationships, reduce feeding/hunting time and success, and disrupt breeding and other activities (Gaynor 2018). Temporal shifts can also result in spatial shifts and thus potentially cause further ecological disruptions. Thus, temporal shifts are disruptive not only to individuals, but also to communities, and ultimately, populations (Gaynor 2018).

Political lobbying and advocacy.—In part due to the markedly different motivation driving mountain bikers compared to other recreationists in protected areas, especially in the more extreme forms of mountain biking (Burgin and Hardiman 2012), the mountain biking community has come to wield significant lobbying and advocacy pressure throughout the United States. Networking among members if the mountain biking community has resulted in changes in land managers' decisions (Bergin and Hardiman 2012). In California, a newly formed mountain biking nonprofit aims to gain a voice at the capital with lawmakers to put trail access and trail development front and center (Formosa 2019). And, the community has much experience in planning trail networks, experience that is necessary to negotiate areas appropriate for mountain biking. In San Diego County, the local mountain biking coalition and the United States Forest Service (USFS) work in partnership to build trail networks on national forest lands; because the USFS does not have a budget for recreation, the only way trails will be built on national forest lands within the County is if the coalition pays the USFS for the agency's staff time, studies and environmental review, and project-processing needed to approve the trail networks (SDMBA 2017). While the USFS-biking coalition partnership may be similar to the accepted practice of an applicant (e.g., utility) paying a lead/permitting agency to dedicate personnel to the applicant's project(s) or a certain body of work, conflicts of interest are usually inherent in such collaborations. In addition, much of the USFS-biking coalition partnership's planning process occurs outside of public view, prior to the public knowing anything about it. It is notable that, while not all USFS lands are considered protected areas in the meaning of this paper, the wilderness areas the USFS manages are.[15]

Recommendations and conclusions

Conservation of habitats is critical to the perpetuation of viable populations of sensitive species. California is home to several types of protected areas whose primary or sole purpose is conservation of sensitive species. After conserving these protected areas, the next crucial step in biological conservation is managing how, where, and when humans use the land. However, there is rarely adequate management to control the allowed types and levels of recreation such that they are compatible with conservation, much less prevent the illegal recreation. The following discussion provides recommendations related to the major issues of recreation ecology addressed above. The implementation of most of these recommendations is considered management as the term is used in this paper (footnote #4), and land managers are familiar with most, if not all, of them. Still, it is hoped that the recommendations provide some new insights and even useful guidance for practical application in the management of dual-role protected areas, the wildlife they support, and the recreationists they serve. For simplicity, clarity, and brevity, several of the recommendations are in imperative sentences. For some of the aspects about recreation ecology discussed

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[15] The USFS manages approximately 33% of the acreage within the National Wilderness Preservation System (https://wilderness.net/learn-about-wilderness/agencies.php) and describes wilderness areas as places where nature "still calls the shots… They are final holdout refuges for a long list of rare, threatened, and endangered species, forced to the edges by modern development… They are places where law mandates above all else that *wildness* be retained for our current generation, and those who will follow" (https://www.fs.usda.gov/managing-land/wilderness).

above, there are no discrete recommendations.

Continual management is imperative.—Continual management (footnote #4) of recreation is imperative for dual-role protected areas to meet their conservation objectives. The chronic insufficiency of management resources for protected areas is of obvious concern. It is urgent that action be taken to address the chronically underfunded management of protected areas by securing perpetual fiscal support that is sufficient for the management needs in perpetuity; the perpetual fiscal support to be secured includes all costs for personnel and all program costs. The level of management must be commensurate with expanding levels of authorized and unauthorized non-consumptive recreation. Given the upward trajectory of recreational activities in protected areas, garnering broad support for securing the perpetual fiscal support requires a societal course change to a collective perspective of respecting and tending to other species in need of protection. Management that is effective for the biological resources would also improve the often cited economic, educational, and health benefits of protected areas.

Prevent further use and proliferation of unauthorized trails.—Prevent the creation and use of unauthorized trails in the first place. This approach would be far preferable to having to contend with the damage to the ecological resources and cultural ecosystem services (discussed below) from the creation and use of unauthorized trails in protected areas. Here, prevention requires continual management. Consider the lessons learned from the work Greer et al. (2017) describe, as summarized above. Where feasible, gain the trail user community's support for and involvement in proactive efforts to prevent vandalism.

Restore habitat to reverse internal fragmentation.—It is reasonable to assume that the disturbance to wildlife from internal fragmentation associated with authorized trails and from legal recreation on them, occurs at least as much from fragmentation associated with unauthorized trails and recreation on them. The internal trail-related fragmentation and expansion of the effect zone most negatively affects those species for which the fitness costs of disturbance are high but have little or no excess habitat to move to; these species are thus constrained to stay in disturbed areas and to suffer the costs in terms of reduced survival or reproductive success (Gill 2001). For these species, restoring the habitat lost to inappropriate trails (i.e., unauthorized trails, unnecessarily redundant designated trails, and trails to be decommissioned) is critical from the standpoint of the negative recreation-related population-level effects. Using restoration to minimize the effects of recreation within fragmented protected areas in urban areas might enable the fragments to better support the focal species (Reed et al. 2019).

Therefore, though the effects on wildlife from unauthorized trails and recreation, per se, have received comparatively little formal study, the precautionary principle (Kriebel et al. 2001; footnote #14) dictates that there seems no need for further study to justify prioritizing restoration of habitat lost to inappropriate trails. So, for levels or habitat loss and the associated internal fragmentation that meet some yet-to-be-established criteria, the restoration should occur. If there is competition for resources (budget/funding, personnel) between (1) research on recreation-related disturbance to wildlife and (2) restoration of habitat lost to inappropriate trails to stop the disturbance, the latter should take priority to reverse internal fragmentation.

To assess the effects of the restoration on the wildlife communities within the protected area, conduct biological surveys within a year prior to the restoration and three to five years after the completion of the groundwork and planting. For this assessment, valid pre-disturbance wildlife survey data collected prior to the loss of habitat within the footprint of the trails that will be restored and associated effect zone will help. But if there are no pre-disturbance data for the protected area or a nearby undisturbed control area, care must be taken in the interpretation of the results of the survey conducted a year prior to the restoration (i.e., the first survey). This is because the results of the first survey will likely represent wildlife communities altered from the pre-disturbed condition (Hennings 2017). It may be that the level of fragmentation, recreation, and many other factors, have caused conditions in which there are no or very few individuals of the focal species (Hennings 2017). These are reasons to be conservative in estimating the recreation-related effects on wildlife in disturbed protected areas without pre-disturbance data; if wildlife have already vacated the disturbed site before the first survey is done, the results will underestimate disturbance effects on wildlife (Hennings 2017). Here, the purpose of the survey data is to aid in determining how the restoration affects the occurrence and/or density of species (depending on the survey methodology), all other factors being equal. The assessment must account for whether the restoration involves the cessation of recreational activities on and/or in the vicinity of the trails to be restored, especially if no other recreational activities begin elsewhere within the species' effect zone throughout the restoration period. If there is funding available and a desire to monitor human activity and wildlife within the restoration areas, deploy camera traps within the areas; camera traps are the most cost-effective method currently available to monitor wildlife activity (Burger 2012).

Minimally, include the following tasks in the restoration: track the actual and in-kind costs (personnel, capital costs, volunteer hours, etc.) for the entire process; map the inappropriate trails and constructed trail features (some use of aerial imagery may work, but on-the-ground mapping validation is essential; Dertien et. al. 2018); prioritize the order of their restoration; determine the best approach for restoring each trail (e.g., passive, active, or a combination); do the restoration itself;[16] and, monitor for several years. Finally, publicize the costs of the restoration to inform the public (F. Landis, California Native Plant Society, personal communication, 2017); for this, compare the costs of the restoration with the costs of the management (footnote #4) that would have been necessary to prevent the damage requiring the restoration. Reasons for documenting the costs include being able to provide to local and state elected officials comparisons of the costs of reactive and proactive approaches to management, and to inform the public about the costs of repairing ecological vandalism.

If possible and logistically advantageous, it would be prudent and economically beneficial to collaborate with recreationists to volunteer with the restoration. For example, this would be an opportunity to mobilize well-organized volunteer contingents of the mountain biking community that are dedicated to building trails. In fact, in some areas, the mountain biking community provides well-organized volunteer assistance in the designing, building, and/or maintenance of officially designated trails in and outside of protected areas. Such volunteer dedication to the restoration of unauthorized trails is sorely needed.

In addition to the biological benefits, another motivation for this habitat restoration in protected areas is its potential to improve the human experience in protected areas open to public access. California's State Wildlife Action Plan (CDFW 2015) and much of the literature about recreation-related ecological effects point to the economic, educational, and recreational/health benefits (i.e., cultural ecosystem services) of protected areas and the species they support. Regarding the human health benefits, the visible recreation-related

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[16] Here, restoration encompasses decompacting the soil, building back and stabilizing the damaged or destroyed terrain and soil, and restoring the affected native plant communities.

damage to the terrain requires consideration beyond its ecological effects—it also affects the level of benefit people enjoy while being in nature, as illustrated by a study examining the relationship between recreational impacts in protected areas and human mental/emotional states (Taff et al. 2019). The study's results demonstrate that, as visible recreation-related ecological impacts increased, sense of wellbeing and mental state decreased, especially in response to settings with unauthorized trails. Collectively, the results show that managing tourism in protected areas in a manner that reduces such impacts is essential to optimizing beneficial cultural ecosystem services related to human health and wellbeing (Taff et al. 2019). Also diminishing the human experience is the risk of injury when using unauthorized trails and TTFs (Davies and Newsome 2009), a risk that restoration would remove. The benefits of the cultural ecosystem services from habitat restoration may increase the potential to obtain funding for such restoration.

Use science-based disturbance thresholds and the precautionary approach.—Establish and use science-based disturbance thresholds to guide management, recognizing and accounting for the notion that the imprecision of thresholds applies to all species, even those for which quantitative thresholds for known sources of disturbances under specific conditions have been identified; thresholds may not adequately protect the target focal species under all conditions in which they occur. The determination of disturbance thresholds must consider the influence of trail-related expansion of effect zones, especially with respect to reductions in the proportions of protected areas that are suitable for wildlife.

To compensate for the imprecision of thresholds when using them to guide management, (1) apply a precautionary approach that adopts maximum values of quantitative disturbance thresholds observed for the taxa of concern, while excluding the extreme values of the thresholds (Dertien et al.'s 2018),[17] (2) take into account that the default position should be a precautionary approach that assumes a priori that the functional value of species' abundance is high (Baker et al. 2018), (3) employ continual proactive and adaptive management to protect wildlife from recreational disturbance,[18] and (4) restrict access if the management fails. The need for the precautionary approach stems from the gaps in knowledge about quantitative disturbance thresholds of recreation.

In trail and trail network planning, use the best available science.—When planning new or modifying existing trails and trail networks in protected areas, the best available science ought to guide policy and decision-making about the siting, design, and alignment of the trails, and about the types, levels, and timing of recreation under consideration. To protect the sensitive species, the policy and decision-making should factor in the capacity to manage the existing and planned trails and recreation in perpetuity. No matter how high the pressure from recreationists for more recreational trails and opportunities, it must be recognized that the majority of recreation-related effects on wildlife are negative. The implications of this necessitate thorough consideration as to whether recreational accommodations that are being considered (in conjunction with all other anthropogenic effects) are compatible with

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17 The precautionary approach and the precautionary principle (footnote #14) have subtle differences between them, but consideration of the differences is beyond the scope of this paper.

18 Based on section 13.5 of the California Fish and Game Code (FGC) and the Natural Community Conservation Planning Act (i.e., section 2805 of the FGC), adaptive management generally means (1) improving management of biological resources over time by using new information gathered through monitoring, evaluation, and other credible sources as they become available, and (2) adjusting management strategies and practices accordingly to assist in meeting conservation and management goals (e.g., conservation of covered or focal species). Under adaptive management, program actions are viewed as tools for learning and to inform future actions. Adaptive management is a cornerstone of large-scale multiple species conservation (CDFW 2014).

the protected areas' conservation objectives. The planning should incorporate protective disturbance thresholds, allowing for adaptive modifications as needed. In situations where recreation has been assumed to meet the conditions of compatibility (e.g., as negotiated in NCCPs/HCPs), great care is needed to ensure the veracity of this assumption. The outcome of the planning process should be ecologically soundly designed, sited, and aligned trails and trail networks, with science-based restrictions on types, levels, and timing of recreation. In conjunction with new trail/trail network construction, restore the habitat lost to inappropriate trails within the area of the construction.

For future protected areas, plan separate recreational areas.—Planning for future protected areas and associated trail networks and recreational areas holds the greatest potential for successful collaboration among landowners, agencies, recreationists, and other stakeholders that allows for truly protective conditions for sensitive species with respect to recreation. Perhaps it is not too late for California to redirect the trajectory of the recreational juggernaut toward an inspirational conservation success story, where stakeholders come together in the planning process, and apply the prevailing science regarding recreation-related disturbance to wildlife to ensure the perpetuation of viable populations of wildlife in the very protected areas set aside primarily or solely for that purpose. Representatives of the recreation community should sit at the table when planning future protected areas and associated trail networks and recreational areas (Burgin and Hardiman 2012); if the outcome is acceptable to them, it may prevent or minimize the creation of unauthorized trails. For example, without a strong strategic approach to mountain biking that includes community engagement, the outcome will be further degradation of protected areas and, at the least, loss of individuals of wildlife, if not major threats to wildlife populations; it's likely that there will also be on-going conflict between mountain bikers and other recreationists and residents (Burgin and Hardiman 2013).

The limited availability of resources for management suggests that it may be more effective to allocate recreational uses and conservation targets among different sites, which will require a diverse suite of land conservation strategies (Reed and Merenlender 2008). At least until such time that there is management of recreation in protected areas commensurate with recreational pressure, planning for future protected areas should heed what has been commonly known for at least 60 years: if conservation of land occurs without enforcing quotas on visitors, then separate areas need to be provided to accommodate recreational activities elsewhere so that the protected land will not bear the burden of those activities (Wilson 2019). This sentiment applies far more today, principally to protected areas preserved primarily or solely for the perpetuation of sensitive species. While this approach is infeasible for many established protected areas (most protected areas in urban areas), going forward, this ought to be the paradigm of habitat and species conservation in areas of high recreational pressure.

Figure 3 depicts an idealized vision of conservation planning using this approach. For protected areas established pursuant to NCCPs/HCPs negotiated in urban settings within an already fragmented landscape, there is often limited latitude for separate areas for recreation; furthermore, sensitive species are typically distributed more evenly across the urbanized landscape than depicted in Figure 3. Nevertheless, it represents the fundamental approach of separating conservation areas from recreational areas. Even in constrained areas, if planning for recreational access occurs at the regional level, planners and land managers could ensure that protected area networks include some areas that are closed to recreation, thus

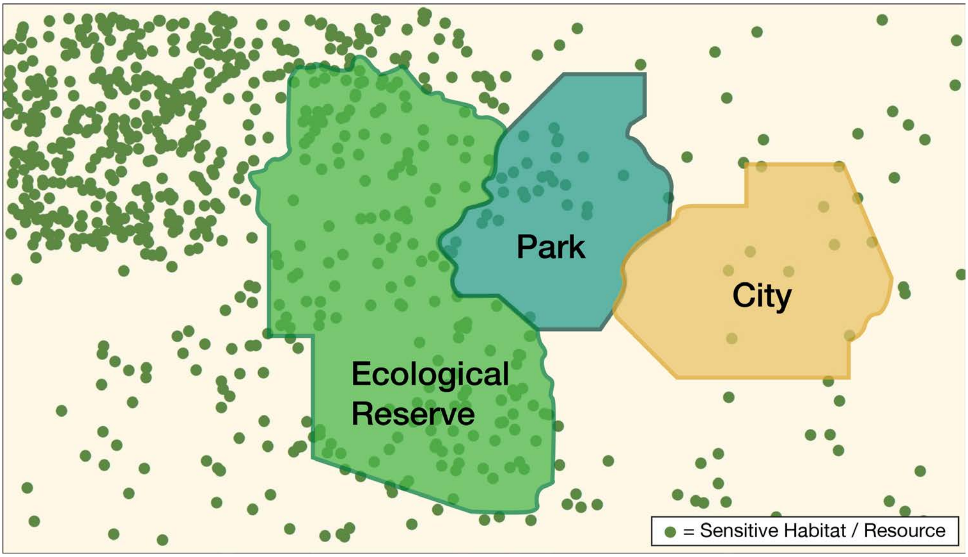

Figure 3. Effective planning for protected areas preserved primarily or solely for the perpetuation of viable populations of sensitive species: provide separate areas for conservation (e.g., ecological reserves) and recreational activities (i.e., parks). (Credit: Landscape Conservation Planning Program, CDFW 2020)

balancing the dual land uses of conservation and recreation at the scale of the protected area network instead of each individual protected area (Reed et al. 2019). Formally incorporating wildlife considerations into the trail planning process from the start is essential to reducing recreation-related disturbance to wildlife; if trail planning is well underway by the time wildlife is considered, it may be too late to gather sufficient wildlife information to inform the planning process (Hennings 2017).

A consideration often not made in conservation planning is the need to address the temporal aspect of human-wildlife interactions. For example, similar to seasonal restrictions, diurnal or nocturnal "temporal zoning" may be necessary to restrict certain human activities during times of the day when sensitive species are most active or when the likelihood of negative human-wildlife encounters is greatest (Gaynor 2018; Whittington 2019). The effectiveness of temporal closures likely depends on the amount and quality of habitat, and levels of human use and fragmentation, within the planned protected areas and in the surrounding landscape. Temporal closures may not benefit wildlife with diurnal activity patterns that differ from the timing of the temporal closures; so, full closures may be required to increase wildlife use in many situations (Whittington 2019). For situations when protected areas and recreational areas are separate but share a boundary, temporal zoning would also apply to the effect zone within the recreational area.

Conclusion.—The most sensible approach for species conservation may be to concentrate research and protection efforts on species whose populations are declining and for which human disturbance is implicated as a possible cause (Gill et al. 2001). The designation of ecological reserves and the conservation of habitat pursuant to NCCPs/HCPs are examples of processes that embody this approach. But, when recreation in such protected areas is not properly planned and adequately managed, their ecological viability and ability to meet their conservation objectives are jeopardized. Implementation of the recommendations provided herein is necessary to ensure the focal species thrive.

Ultimately, for wildlife that avoids human activity, it is unlikely that dual-role protected areas are entirely sufficient or justifiable for meeting conservation objectives; limiting or prohibiting recreation in strategic circumstances and locations within protected areas is necessary to achieve conservation objectives (Bötsch et al. 2018; Dertien et al. 2018; Reed et al. 2019). Enforced closures of inappropriate trails in all protected areas and restoration of those trails would substantially decrease the trail-related disturbance to wildlife across the landscape; waiting until after wildlife detections or estimates of habitat use decrease is too late to implement these measures (Dertien et al. 2018). These approaches require perpetual management commensurate with expanding levels of authorized and unauthorized non-consumptive recreation in protected areas. Action is urgently needed to secure perpetual fiscal support for management sufficient to ensure the perpetuation of viable populations of sensitive species in protected areas.

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INFORMATION FOR AUTHORS

The California Fish and Wildlife Journal (CFWJ) is a peer-reviewed, scientific journal focused on the biology, ecology, and conservation of the flora and fauna of California and surrounding areas, and the northeastern Pacific Ocean.

Submissions guidelines (PDF) for the Journal have been updated (July 2019).

The California Fish and Wildlife Journal accepts manuscripts in the following categories:

- Original research papers
- Research notes
- Review papers
- Book reviews
- Commentaries and Essays

Manuscripts must be submitted by e-mail following directions provided in the link: https://nrm.dfg.ca.gov/FileHandler.ashx?DocumentID=171113&inline. The journal standard for style is consistent with the Council of Science Editors (CSE) Style Manual. Instructions in the CFWJ guidelines supersede the CSE Style Manual where differences exist between formats. Please follow these formatting guidelines carefully. Manuscripts that do not conform to the guidelines will be returned for revision.

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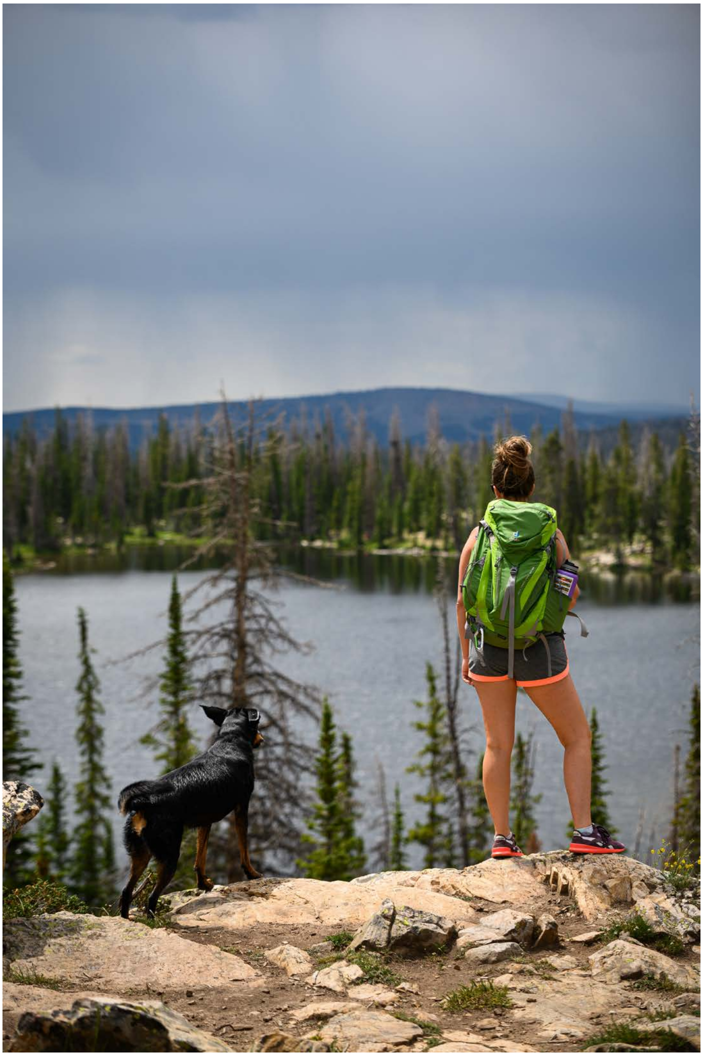

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CALIFORNIA WILDLIFE HABITAT RELATIONSHIPS SYSTEM
maintained by the
CALIFORNIA DEPARTMENT OF FISH AND GAME
and supported by the
CALIFORNIA INTERAGENCY WILDLIFE TASK GROUP
Database Version 8.1 (2005)

B111 White-tailed Kite Elanus leucurus
Family: Accipitridae Order: Falconiformes Class: Aves

Written by: C. Polite
Reviewed by: S. Bailey
Edited by: S. Bailey, P. Bloom
Updated by: CWHR Program Staff, February 2005

DISTRIBUTION, ABUNDANCE, AND SEASONALITY

Common to uncommon, yearlong resident in coastal and valley lowlands; rarely found away from agricultural areas. Inhabits herbaceous and open stages of most habitats mostly in cismontane California. Has extended range and increased numbers in recent decades.

SPECIFIC HABITAT REQUIREMENTS

Feeding: Preys mostly on voles and other small, diurnal mammals, occasionally on birds, insects, reptiles, and amphibians. Forages in undisturbed, open grasslands, meadows, farmlands and emergent wetlands. Soars, glides, and hovers less than 30 m (100 ft) above ground in search of prey. Slowly descends vertically upon prey with wings held high, and legs extended; rarely dives into tall cover (Thompson 1975).

Cover: Uses trees with dense canopies for cover. In southern California, also roosts in saltgrass and Bermudagrass.

Reproduction: Makes a nest of loosely piled sticks and twigs and lined with grass, straw, or rootlets. Nest placed near top of dense oak, willow, or other tree stand; usually 6-20 m (20-100 ft) above ground (Dixon et al. 1957). Nest located near open foraging area.

Water: Probably meets water requirements from prey.

Pattern: Uses herbaceous lowlands with variable tree growth and dense population of voles (Waian and Stendell 1970). Substantial groves of dense, broad-leafed deciduous trees used for nesting and roosting.

SPECIES LIFE HISTORY

Activity Patterns: Yearlong diurnal, and crepuscular activity.

Seasonal Movements/Migration: Apparently not migratory, but Binford (1979) found some movements in coastal California. May become nomadic in response to prey abundance (Dunk and Cooper 1994).

Home Range: Forages from a central perch over areas as large as 3 sq km (1.9 sq mi) Warner and Rudd 1975). Seldom hunts more than 0.8 km (0.5 mi) from nest when breeding (Hawbecker 1942). Henry (1983) found mean breeding home range to be 0.57 sq km (0.2 mi).

Territory: Generally not territorial, but nest site may be defended against crows, other hawks, and eagles (Pickwell 1930, Dixon et al. 1957). Defended foraging territories of about

0.10 sq km (.04 sq mi) in winter from red-tailed hawks and northern harriers (Bammann 1975). Communal roost used in nonbreeding seasons (Waian and Stendell 1970). Territory size a function of prey and competitor abundance (Dunk and Cooper 1994).

Reproduction: Monogamous; breeds from February to October, with peak from May to August. Average clutch 4-5 eggs, range 3-6. Female only incubates, for about 28 days. Young fledge in 35-40 days. During incubation and nestling period, male feeds female, and supplies her with food to feed the young. Usually single brooded; occasionally 2 broods.

Niche: Preys on rodents that may be harmful to agricultural crops. Nest may be robbed by jays, crows yellow-billed magpies, raccoons, and opossums. Great horned owls may prey on adults and young.

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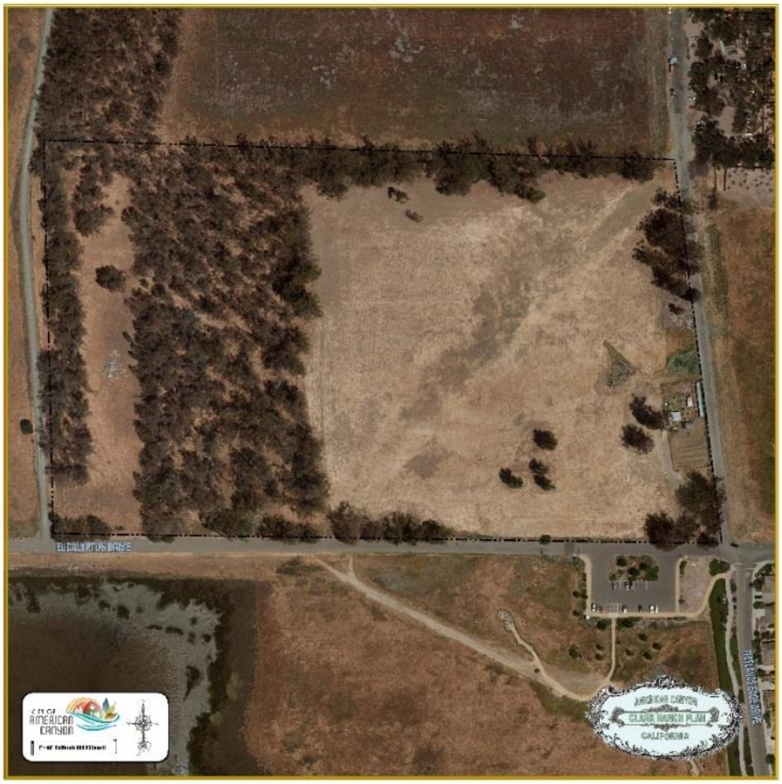
Species notes are designed to support the species-habitat relationships database models in the California Wildlife Habitat Relationships (CWHR) System and are also published within the current version of the CWHR software. Please cite as: California Department of Fish and Game. California Interagency Wildlife Task Group. 2005. California Wildlife Habitat Relationships version 8.1 personal computer program. Sacramento, California.

Exhibit A

Clarke Ranch Master Plan

November 7, 2017



1. Vision, Goals and Objectives
2. Program and Amenities
3. Final Conceptual Plan Map
4. Cost Estimate

Clarke Ranch

Vision, Goals and Objectives

Based on feedback from PCS/OSAC Joint Meeting on 9/25/17

Vision

Clarke Ranch is a community gathering place for all in American Canyon, it will serve as a hub of outdoor-based activity which will be environmentally friendly and nature based.

Synthesized Goals/Objectives:

1. Nature-based – The park shall be designed and developed in a way to support, encourage and highlight a nature-based approach to design and construction. The nature-based approach is most obvious in the selection of a nature-based playground, rather than a traditional steel and plastic, post and platform play structure.
2. Connection to the Wetlands – Clarke Ranch has a unique opportunity to provide regional hands-on educational amenities highlighting the connection to the wetlands. Through the design phase, the City shall work with environmentally oriented groups such as the Napa Solano Audubon Society, Napa County Resource Conservation District, Native Plant Society, the amateur astronomers club and others to highlight and display the wetlands connections.
3. Environmentally friendly and revenue generating – Clarke Ranch should be not only environmentally friendly, but also, fiscally efficient including revenue generating opportunities to support operational costs. We need to study the operational implications and costs through the design development phase.
4. Carbon Neutral/Sustainable Design – The City desires the park to have a sustainable design and have a net zero carbon footprint. Through the design development, we need to better study carbon impacts and what options exist to mitigate these impacts. Sustainable design components to consider include solar and wind power, composting toilets, onsite compost generation, low water/drought tolerant/native plantings, use of permeable asphalt or paving and paths, to name a few. Evaluate the benefits of pursuing a sustainable landscape program such as Bay Friendly Qualification or SITES accreditation in the design development phase.
5. Eucalyptus Trees – In the formal design, the eucalyptus groves needs studied by a qualified arborist to determine how best to manage the forest. PCS/OSAC prefers that some eucalyptus remain and some be removed in a phased approach. Any native replanting of trees (encouraging diversity) needs to start early.
6. Lease for Long-term Use – The three non-city uses must enter into a long-term lease identifying items including but not limited to: rent, development/construction standards and expectations, performance and reporting expectations, and they must identify how a regular park user will engage in their use without being a part of it (without paying).
7. Lighting – The park shall be lighted as necessary for planned programmatic uses, but sensitive to neighbors and animals on site. The lights should be designed for energy efficiency and in a way that provides minimal security when not fully in use. The lights need to be able to be shut off for astronomy events. We need to consider use of solar, and needs of plug-in vehicles.
8. Outdoor Exercise Equipment – The ACCPF has fundraised for the outdoor exercise equipment and desires it to be constructed within the next 12-18 months. The master plan calls for the outdoor exercise equipment to be constructed just north of the Napa River Bay Trail approximately 450’ from the current gate. Should this location be problematic (design or environmentally), per the direction of the PCS/OSAC Committees, City staff will find an

alternative location south of the Napa River Bay Trail in the demonstration garden area and blend it in to make it feel design-connected.

9. Equestrian Riding Area (SpiritHorse) – Having an equestrian riding area is a popular component within the park as it serves to remind us of the agricultural/ranching roots of the area. The steering committee discussed the size of the equestrian space significantly through the master plan process. The joint PCS/OSAC body directed staff to keep the space as shown on the 9/25/17 Draft Final Preferred Master Plan, approximately 2 acres in size.

Program and amenities:

Non-City Uses

1. Napa County Wildlife Rescue – The Napa Wildlife Rescue is allocated approximately 2.5 acres for their program. Their use includes two buildings (hospital and admin space), a viewing/educational area, and several outdoor cages and aviaries tucked into the flexible space under the Eucalyptus trees.
2. Equestrian Riding Area (SpiritHorse) – The equestrian riding area is allocated approximately 2.0 acres for riding and care of horses. Amenities included in the master plan are a barn, office, restroom, horse stalls, 100'x200' covered arena, and pasture space.
3. American Canyon 4H – American Canyon 4H is allocated approximately 1 acre. Their space includes a round pen, stalls for pigs, lambs, cattle and chickens. There is also two small feed barns and one larger barn included in their space.

Other Significant Amenities

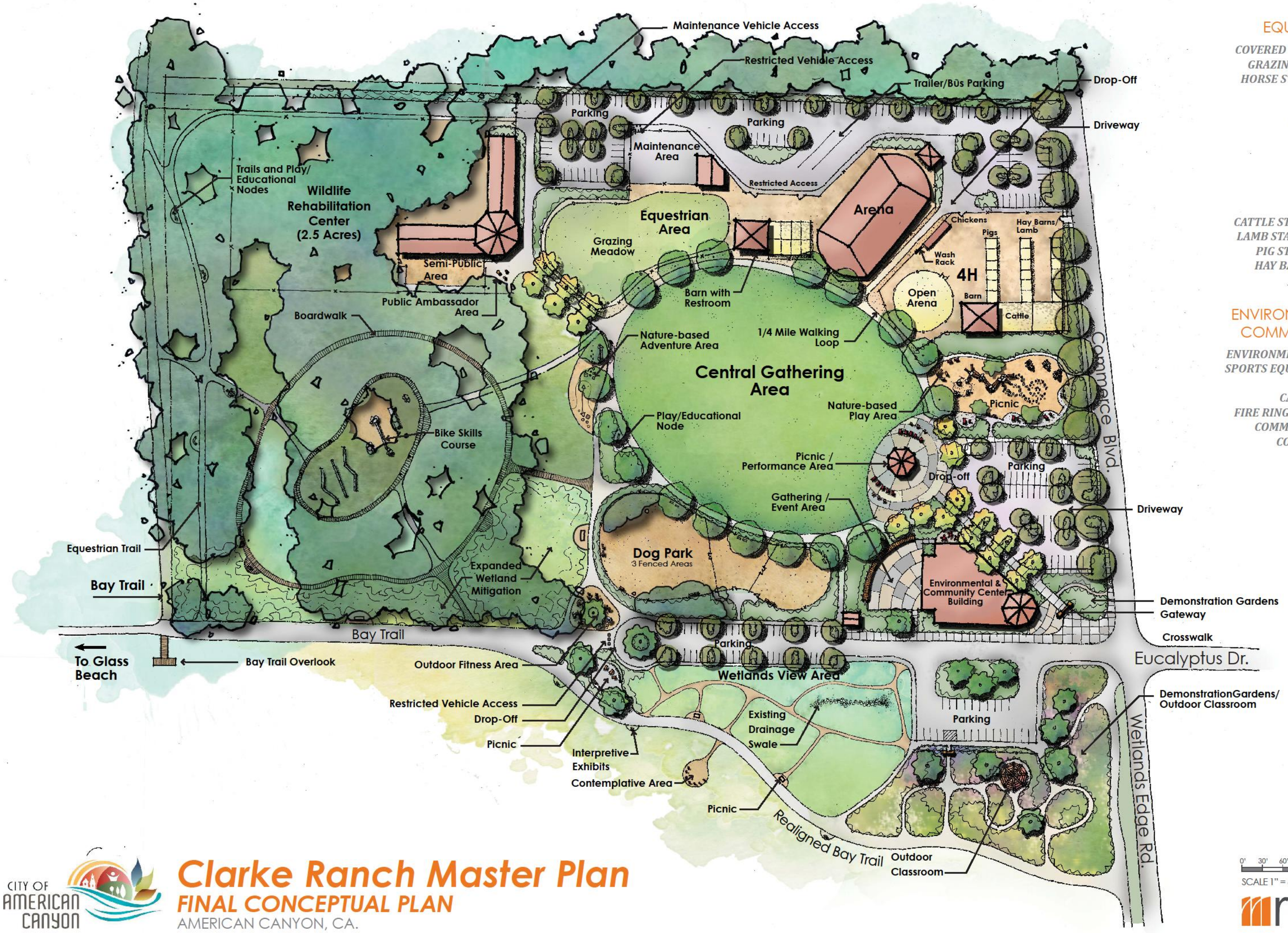
4. Regional Nature/Environmental/Community Center – The center is currently planned to be an indoor interpretative, recreation and community space with classrooms and multipurpose rooms to support programming and exhibits (watershed protection/education, wetlands habitat/biology). Possible space for vendor sales (deli/gifts) and rentals (bike/kayak). During the design development phase, the formal building program needs development and to be determined. The building is currently planned to be 15,000 square feet.
5. Central Gathering Area – A space of approximately 4 acres central to the site is planned to remain open and available for special events, passive activities like Frisbee throwing, stargazing events, and camping (by permission of the director). There is a large performance/picnic area in the southeastern corner of the central gathering area. The space is shown as turf currently.
6. Dog Park – An enclosed dog park is included alongside the Napa River Bay Trail to allow dog owners the opportunity to have their dogs socialize off leash in an enclosed area. Three dividable areas are included to allow the closure of one space for maintenance and still have two separable spaces (big vs small dog).
7. Demonstration Gardens – South of the Napa River Bay Trail are a host of garden spaces. These spaces are planned to host demonstration uses including a pollinator garden, butterfly garden, native plant area, bird garden, drought-tolerant garden, and others. There is also a small outdoor classroom space to accommodate 30 people.
8. Bike Skills Area – A bike skills area is in the existing opening in the eucalyptus grove. It is intended to be an area to teach beginning and intermediate bike riders balance, challenge, light risk, and improve their off-trail bike riding skills. An item to research further in design development is whether or not a small pump track would be a good fit for the space.
9. Nature-based Playground – One of the truly unique and hopefully iconic elements of the park will be the nature-based playground. This is envisioned to buck the trend of metal and plastic traditional play structures and instead have kids play on tree trunks, stumps, rocks, and other natural elements.

10. Wetlands – There are known seasonal wetlands on the site. The design development phase will need to complete an environmental initial study and suggest mitigation measures. The plan contemplates growing the seasonal wetlands areas in the southwest portion of the park to accommodate expected mitigation. The increased wetlands areas in the southwest corner of the site also accommodates expected sea level rise. The most recent sea level rise study of the area shows the site under water frequently by 2050.
11. Bay Trail Alignment – The proposed plan solidifies the primary Bay Trail route down the Eucalyptus Drive alignment. To eliminate an existing dead-end trail and provide a more direct route from Wetlands Linear Path to the Napa River Bay Trail, the plan suggests making a diagonal connection between the two along the southern project boundary.
12. Outdoor Exercise Equipment – See the goal above. The outdoor exercise equipment is relatively small in the grand scheme of the plan. It is envisioned to include 8-10 pieces of outdoor fitness equipment. These are growing in popularity and used throughout the country as a means to cheaply increase physical fitness, balance and mobility, specifically in older adults.
13. Play and Interpretative Activity Nodes – As this site is planned to have a strong nature/environmental/outdoor oriented theme, small play and interpretative activity nodes have been contemplated in the plan. These spaces utilize concepts of *Pathways for Play,* which incorporates play, education and learning into small tucked away activity areas.
14. Nature-based Adventure Area – This concept needs development in the next phase of design. The preliminary thought is to provide a space similar to the nature-based playground but gear it towards a slightly older age group and encourage more risk. Some preliminary concepts include a larger bouldering wall, a nature-based obstacle course, or an upper body climbing area.

Other Amenities

15. Walking Paths and Boardwalks – The top item on most needs assessments is the need and desire to walk and run on paths. The plan has several potential loops for casual walkers and runners. The path around the central gathering area creates a loop that is approximately .25 miles. It is expected that paths throughout the park will be a combination of hard and soft surface trails.
16. Standalone Restroom – A new restroom is planned along the southern edge of the park, near the Napa River Bay Trail Head. Another restroom is planned near the equestrian riding area.
17. Group gathering areas – Socialization is a major benefit and desire for parks; there are several group gathering areas scattered throughout the site including picnic areas, outdoor classroom space, and fire pit feature/teen gathering area.
18. Parking – There are approximately 150 new parking spaces scattered throughout the site.

CLARKE RANCH MASTER PLAN
FINAL CONCEPTUAL PLAN



EQUESTRIAN AREA

COVERED ARENA: *100' X 200'*
GRAZING MEADOW: *.75 AC*
HORSE STALLS: *8 @12' X 24'*
BARN: *50' X 40*
RESTROOM
TRAILER PARKING

4H AREA

HAY BARNS
CHICKEN COOP
WASH RACK
CATTLE STALLS: *2 @ 40' X 40'*
LAMB STALLS: *10 @ 12' X 16'*
PIG STALLS: *6 @ 15' X 15'*
HAY BARNS: *2 @ 12' X 16'*
ARENA: *5,600 SF*

ENVIRONMENTAL AND COMMUNITY CENTER

ENVIRONMENTAL EDUCATION
SPORTS EQUIPMENT RENTALS
EVENT SPACE
CAFE/DELI + PICNIC
FIRE RING GATHERING AREA
COMMUNITY PROGRAMS
COMMUNITY ROOMS
OFFICE SPACE

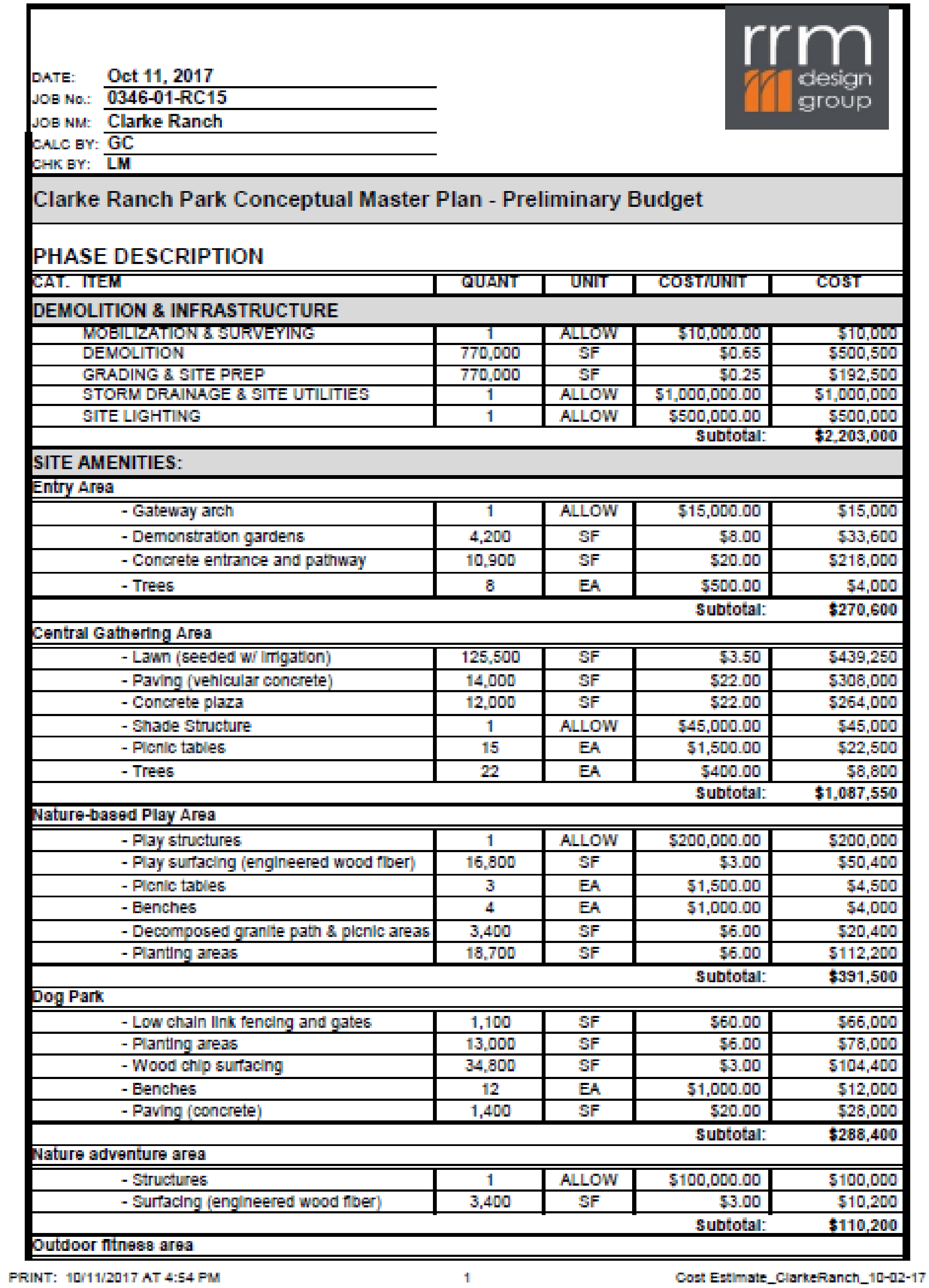
CITY OF AMERICAN CANYON

Clarke Ranch Master Plan
FINAL CONCEPTUAL PLAN
AMERICAN CANYON, CA.

SCALE 1" = 60' OCTOBER 17, 2017

rrm design group

CLARKE RANCH MASTER PLAN
PRELIMINARY BUDGET



DATE: Oct 11, 2017
JOB No.: 0346-01-RC15
JOB NM: Clarke Ranch
CALC BY: GC
CHK BY: LM

Clarke Ranch Park Conceptual Master Plan - Preliminary Budget

PHASE DESCRIPTION

CAT. ITEM	QUANT	UNIT	COST/UNIT	COST
DEMOLITION & INFRASTRUCTURE				
MOBILIZATION & SURVEYING	1	ALLOW	$10,000.00	$10,000
DEMOLITION	770,000	SF	$0.65	$500,500
GRADING & SITE PREP	770,000	SF	$0.25	$192,500
STORM DRAINAGE & SITE UTILITIES	1	ALLOW	$1,000,000.00	$1,000,000
SITE LIGHTING	1	ALLOW	$500,000.00	$500,000
			Subtotal:	**$2,203,000**
SITE AMENITIES:				
Entry Area				
- Gateway arch	1	ALLOW	$15,000.00	$15,000
- Demonstration gardens	4,200	SF	$8.00	$33,600
- Concrete entrance and pathway	10,900	SF	$20.00	$218,000
- Trees	8	EA	$500.00	$4,000
			Subtotal:	**$270,600**
Central Gathering Area				
- Lawn (seeded w/ irrigation)	125,500	SF	$3.50	$439,250
- Paving (vehicular concrete)	14,000	SF	$22.00	$308,000
- Concrete plaza	12,000	SF	$22.00	$264,000
- Shade Structure	1	ALLOW	$45,000.00	$45,000
- Picnic tables	15	EA	$1,500.00	$22,500
- Trees	22	EA	$400.00	$8,800
			Subtotal:	**$1,087,550**
Nature-based Play Area				
- Play structures	1	ALLOW	$200,000.00	$200,000
- Play surfacing (engineered wood fiber)	16,800	SF	$3.00	$50,400
- Picnic tables	3	EA	$1,500.00	$4,500
- Benches	4	EA	$1,000.00	$4,000
- Decomposed granite path & picnic areas	3,400	SF	$6.00	$20,400
- Planting areas	18,700	SF	$6.00	$112,200
			Subtotal:	**$391,500**
Dog Park				
- Low chain link fencing and gates	1,100	SF	$60.00	$66,000
- Planting areas	13,000	SF	$6.00	$78,000
- Wood chip surfacing	34,800	SF	$3.00	$104,400
- Benches	12	EA	$1,000.00	$12,000
- Paving (concrete)	1,400	SF	$20.00	$28,000
			Subtotal:	**$288,400**
Nature adventure area				
- Structures	1	ALLOW	$100,000.00	$100,000
- Surfacing (engineered wood fiber)	3,400	SF	$3.00	$10,200
			Subtotal:	**$110,200**
Outdoor fitness area				

Clarke Ranch Park Conceptual Master Plan - Preliminary Budget

PHASE DESCRIPTION

CAT. ITEM	QUANT	UNIT	COST/UNIT	COST
- Structures	7	EA	$5,000.00	$35,000
- Decomposed granite surfacing	4,000	SF	$6.00	$24,000
- Tree	1	EA	$400.00	$400
			Subtotal:	**$59,400**
Boardwalk loop				
- Composite wood decking	12,800	SF	$25.00	$320,000
- Interpretive features	4	EA	$800.00	$3,200
- Mitigated wetlands	74,300	SF	$3.00	$222,900
			Subtotal:	**$546,100**
Trail in Eucalyptus trees				
- Earth Trail	4,900	SF	$1.50	$7,350
- Educational/play features	4	EA	$2,000.00	$8,000
			Subtotal:	**$15,350**
Bike Skills Loops				
- Skills loop elements	1	ALLOW	$15,000.00	$15,000
			Subtotal:	**$15,000**
Maintenance yard				
- Concrete block wall	400	LF	$100.00	$40,000
- Building	1	ALLOW	$100,000.00	$100,000
- Paving (concrete)	7,800	SF	$20.00	$156,000
- Covered storage areas	1	ALLOW	$10,000.00	$10,000
			Subtotal:	**$306,000**
Wildlife refuge connection & fire road				
- Paving (concrete)	11,200	SF	$7.00	$78,400
- Trees	6	EA	$400.00	$2,400
- Vehicular gate	1	EA	$2,500.00	$2,500
- Fire road (asphalt)	10,600	SF	$7.00	$74,200
			Subtotal:	**$157,500**
Equestrian/4H connection				
- Paving (vehicular concrete)	7,200	SF	$22.00	$158,400
- Planting	1,000	SF	$6.00	$6,000
			Subtotal:	**$164,400**
Picnic areas in Wetlands Edge Area				
- Picnic tables	5	EA	$1,500.00	$7,500
- Benches	2	EA	$1,000.00	$2,000
- Decomposed granite surfacing	1,900	SF	$6.00	$11,400
- Planting areas	1,000	SF	$6.00	$6,000
- Trees	2	EA	$400.00	$800
			Subtotal:	**$27,700**
Demonstration garden in Wetlands Edge Area				
- Outdoor classroom shade structure	1	ALLOW	$20,000.00	$20,000
- Entry arbor	1	ALLOW	$10,000.00	$10,000
- Decomposed granite surfacing	3,400	SF	$6.00	$20,400
- Planting areas	60,000	SF	$6.00	$360,000
- Trees	6	EA	$400.00	$2,400
- Interpretive features	4	EA	$5,000.00	$20,000
- Interpretive signs	20	EA	$150.00	$3,000
- Landscape boulders	1	ALLOW	$2,500.00	$2,500
- Benches	4	EA	$1,000.00	$4,000
- Maintenance shed	1	ALLOW	$5,000.00	$5,000

Clarke Ranch Park Conceptual Master Plan - Preliminary Budget

PHASE DESCRIPTION

CAT. ITEM	QUANT	UNIT	COST/UNIT	COST
			Subtotal:	**$447,300**
Bay Trail Re-alignment				
- Paving (asphalt)	11,500	SF	$7.00	$80,500
			Subtotal:	**$80,500**
Wetland overlook				
- Composite wood decking	300	SF	$100.00	$30,000
			Subtotal:	**$30,000**
Contemplative area				
- Decomposed granite surfacing	1,500	SF	$6.00	$9,000
- Boulder seats	7	EA	$500.00	$3,500
			Subtotal:	**$12,500**
Restroom				
- Building	1	ALLOW	$225,000.00	$225,000
- Concrete paving	1,600	SF	$20.00	$32,000
			Subtotal:	**$257,000**
Commerce Drive Streetscape				
- Planting areas	20,000	SF	$6.00	$120,000
- Concrete walks	3,700	SF	$20.00	$74,000
- Trees	12	EA	$400.00	$4,800
			Subtotal:	**$198,800**
Wetlands Edge Area Parking/Dropoff Area				
- Paving (Asphalt)	20,500	SF	$7.00	$143,500
- Planting areas	1,800	SF	$6.00	$10,800
- Vehicular gate	1	EA	$2,500.00	$2,500
- Concrete walks	2,350	SF	$20.00	$47,000
- Trees	9	EA	$400.00	$3,600
- Bollards	3	EA	$1,500.00	$4,500
			Subtotal:	**$211,900**
Visitor Center Parking/Dropoff Area				
- Paving (Asphalt)	32,400	SF	$7.00	$226,800
- Planting areas	9,200	SF	$6.00	$55,200
- Concrete walks	1,600	SF	$20.00	$32,000
- Trees	11	EA	$400.00	$4,400
			Subtotal:	**$318,400**
North Parking Area				
- Paving (Asphalt)	65,500	SF	$7.00	$458,500
- Planting areas	7,500	SF	$6.00	$45,000
- Concrete walks	3,700	SF	$20.00	$74,000
- Trees	27	EA	$400.00	$10,800
			Subtotal:	**$588,300**
			IMPROVEMENTS SUBTOTAL:	**$7,787,400**
GENERAL CONDITIONS	6%	OF	$7,787,400	$467,244
BONDS	2%	OF	$7,787,400	$155,748
CONTRACTOR FEES	10%	OF	$7,787,400	$778,740
			Subtotal:	**$9,189,132**
DESIGN CONTINGENCY	15%	OF	$9,189,132	$1,378,370

Clarke Ranch Park Conceptual Master Plan - Preliminary Budget

PHASE DESCRIPTION

CAT.	ITEM	QUANT	UNIT	COST/UNIT	COST
	ESCALATION (PER YEAR)	4%	OF	$9,189,132	$367,565
				Subtotal:	**$1,745,935**
				TOTAL:	**$10,935,067**
Visitor Center					
	- Building	14,400.00	SF	$425.00	$6,120,000
	- Patio	5,400.00	SF	$22.00	$118,800
	- Patio trellis	1.00	ALLOW	$20,000.00	$20,000
	- Fire ring area	1.00	ALLOW	$20,000.00	$20,000
	- Planting areas w/ irrigation	9,100.00	SF	$6.00	$54,600
				Subtotal:	**$6,333,400**
	GENERAL CONDITIONS	6%	OF	$6,333,400	$380,004
	BONDS	2%	OF	$6,333,400	$126,668
	CONTRACTOR FEES	10%	OF	$6,333,400	$633,340
				Subtotal:	**$7,473,412**
	DESIGN CONTINGENCY	15%	OF	$7,473,412	$1,121,012
	ESCALATION (PER YEAR)	4%	OF	$7,473,412	$298,936
				Subtotal:	**$1,419,948**
				VISITOR CENTER TOTAL:	**$8,893,360**

THIS PROJECTION WAS PREPARED USING STANDARD COST AND/OR QUANTITY PROJECTION PRACTICES. IT IS UNDERSTOOD AND AGREED THAT THIS IS A PROJECTION ONLY, AND THAT THE ARCHITECT SHALL NOT BE LIABLE TO THE OWNER OR TO A THIRD PARTY FOR ANY FAILURE TO ACCURATELY PROJECT THE COST AND/OR QUANTITIES FOR THE PROJECT, OR ANY PART THEREOF.

THIS ARCHITECT'S PROJECTION IS PREPARED AS A GUIDELINE AND DOES NOT CONSTITUTE THE BASIS FOR BID. THE CONTRACTOR IS TO PERFORM HIS/HER OWN QUANTITY TAKE-OFF AND TO BID ACCORDINGLY. IN THE EVENT THAT ERRORS OR OMISSIONS ARE ENCOUNTERED THROUGH THE BIDDING PROCESS, PLEASE CONTACT THE ARCHITECT FOR CLARIFICATION.

The Auk 111(3):588-595, 1994

TERRITORY-SIZE REGULATION IN BLACK-SHOULDERED KITES

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ABSTRACT.—We studied the relationship of Black-shouldered Kite (*Elanus caeruleus*) territory size to prey abundance and competitor abundance in northwestern California. Kite territory size ranged from 1.6 to 21.5 ha ($n = 26$). The estimated mean number of California voles (*Microtus californicus*) per territory was 1,483 ± SE of 163 ($n = 25$). Competitor abundance (i.e. total raptor abundance) ranged from 4.8 to 31.0 individuals/km^2 and was strongly correlated with abundance of *M. californicus*. Both estimated prey abundance and competitor abundance were inversely correlated with kite territory size. After developing multiple-regression models using both variables, partial-correlation analysis revealed that once the effects of prey abundance were statistically controlled, competitor abundance continued to be significantly correlated with kite territory size. When the effects of competitor abundance were statistically controlled, prey abundance was no longer significantly correlated with kite territory size. In that it appears that the number of competitors was a function of number of prey, we conclude that kite territory size is proximately regulated by competitor abundance and ultimately regulated by prey abundance. *Received 3 August 1992, accepted 14 March 1993.*

THE RELATIONSHIP BETWEEN raptors and their prey has been studied extensively (e.g. Craighead and Craighead 1956, Village 1982, 1987, 1989, Korpimäki 1984, 1985a, b, 1988, Newton et al. 1986). Most raptor-prey studies have focused either on raptor abundance (Baker and Brooks 1981, Village 1982, Cully 1991) or raptor reproductive success (Hammerstrom 1979, Smith et al. 1981, Korpimäki 1986, 1988, Ridpath and Brooker 1986) in relation to prey abundance. Few studies have examined the relationship between raptor territory size and prey abundance (for exceptions, see Village 1982, 1987, Temeles 1987), probably because of the difficulty of measuring territory size of wide-ranging birds, and in estimating abundance of several types of prey (e.g. rodents, rabbits, and birds) over those relatively large areas.

Generally, theory predicts that territory size should be inversely related to food abundance (for exceptions, see Schoener 1983, 1987, Ebersole 1980). Myers et al. (1979) presented two hypotheses regarding territory-size regulation. First, individuals establish territories of a size that contains adequate resources to meet their energetic needs. Second, an individual will defend as large an area as it can, constrained by competition with other individuals. These hypotheses are not mutually exclusive, although they have generally been viewed as such (Myers et al. 1979).

We studied the relationship of the size of Black-shouldered Kite (*Elanus caeruleus*) hunting territories to prey abundance and competitor abundance. Kites are good subjects for such a study because: (1) they have distinctive hunting habits (hovering), which allows unambiguous descriptions of areas/points used for foraging; (2) they use open to semiopen habitats, facilitating long, continuous observation periods; (3) they are at the top of the food chain, so their territory size is not likely to be influenced by predation; (4) they have relatively small home ranges or territories compared with other raptors (Henry 1983); and (5) they rely almost exclusively on rodents as prey (Waian and Stendell 1970), primarily California voles (*Microtus californicus*) in California (Stendell 1972, Bammann 1975). Thus, estimating kite prey abundance is an easier task than it would be with a more generalist predator.

METHODS

The study was conducted on the Fay Slough Wildlife Area in Eureka, California. The 135-ha area was used for cattle grazing prior to 1987 when it was acquired by the California Department of Fish and Game. It is composed primarily of ungrazed pasture with small patches of alders (*Alnus* spp.) and blackberry bushes (*Rubus* spp.). Alders were planted along slough borders in 1988, providing important perching locations for raptors. A number of freshwater and

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brackish sloughs dissect the area. The topography is flat with elevations ranging from −0.5 m to 2 m. The climate in the area is maritime, with cool summers and mild winters. Summers are characterized by foggy nights and mornings. Precipitation was greatest between November and April though some rain occurred during most months of this study. The growing season is generally between April and July, when grasses develop, then drop their seeds. Vegetation composition and structure were relatively constant throughout the study, although limited (rotational) grazing occurred on approximately 35 ha beginning in December 1989. Primary plant species included Italian ryegrass (*Lolium multiflorum*), perennial ryegrass (*L. perenne*), velvet grass (*Holcus lanatus*), and buttercups (*Ranunculus* spp.).

Kites were captured with a selective pole trap (Dunk 1991), then banded with uniquely colored leg bands. Kite hunting territories were estimated from June 1989 through November 1990. Each territory was delineated by observing a kite and documenting all locations where the bird hovered, perched, or interacted with another kite or some other raptor. Locations were estimated by measuring distance with an optical range finder and direction with a compass from fixed observation spots. The accuracy of the range finder was ±1% at 100 m and ±10% at 1,000 m. For each location, distance, direction, and time (to the nearest 0.05 min) were recorded on a microcassette recorder. Each kite was observed for a minimum of 1.5 h or until the bird left the area to roost. Total observation time per territory ranged from 1.5 to 12.25 h over a one- to seven-day period. Mean number of locations per territory was 64.5 ± SD of 24.2. Adequacy of sampling hunting territories was determined graphically by plotting hunting territory size and number of locations until reaching an asymptote. Hunting territory size was estimated using the 95%-minimum-convex-polygon estimator in program HOME RANGE (Ackerman et al. 1989).

We attempted to delineate as many hunting territories as possible. We did not randomly choose individuals to observe, but selected individuals according to the following priority: (1) color-banded birds with a territory not previously measured; (2) previously unmeasured territories of unbanded individuals that were repeatedly associated with a specific geographic area prior to the sampling time; and (3) color-banded individuals whose territories had been previously estimated. Unbanded birds were identified using one or more of the following criteria: (1) unique molting patterns; (2) perches used; (3) association with (i.e. mates of) banded kites; and (4) age (i.e. whether adult or juvenile).

We sampled vegetation structure and composition within two weeks of delineating each hunting territory. After plotting territory boundaries on a 1:2,400 aerial photograph, 100 vegetation plots were established in a random systematic design for each territory in order to have complete coverage of the territory. In each plot, we measured the percentage of the plot covered by: (1) green grass (Gramineae); (2) brown grass; (3) green herbaceous vegetation; (4) brown herbaceous vegetation; (5) seed heads on grasses; (6) green rush (Cyperaceae); and (7) brown rush. Variables 6 and 7 were estimated beginning in December 1990. These seven variables were estimated by eye and placed into one of six cover classes: 0%, 1–20%, 21–40%, 41–60%, 61–80%, and 81–100%. Cover height, an estimate of the height at which there was 100% cover horizontally, also was measured in the center of each plot by pushing a meter stick through the vegetation, flush to the ground, and recording the height (cm) of the intersection between plot center and the meter stick. Plots falling on flooded areas were classified as flooded with no vegetation characteristics estimated within them.

Potential competitors of kites (other vole-eating raptors) were counted once each month within 14 days of estimating each territory. From a central location, we counted the numbers of each raptor species by scanning the entire study area (approximately 95% of the area could be viewed). Counts were terminated when three consecutive counts yielded identical results. All counts were made within 1 h of sunset because previous observations showed that this was the period of greatest raptor activity. Common Barn-Owls (*Tyto alba*) were counted by looking in the two barns on the study area.

Microtus californicus were trapped monthly within four randomly located permanent trapping grids. Grids consisted of 40 traps spaced 5 m apart and arranged in an 8 × 5 pattern. Traps were locked open for 24 to 36 h prior to being set each month. Before being set, traps were baited with bird seed and polyester pillow stuffing was added. Traps were checked just after first light and just before dark for four consecutive days. Because there was no evidence from our study to suggest that *M. californicus* activity differed between day and night, we did not distinguish between day and night captures. At each capture location, individuals were uniquely toe-clipped, weighed (g), sexed, and released (for more detailed description of small-mammal trapping, see Dunk 1992). Vole densities within grids were estimated by dividing number of individuals caught by grid area. Within one week of trapping, vegetation was sampled in each trapping grid. Vegetation plots were placed within 0.5 m of each trap, and the same parameters were estimated as in kite hunting territories.

To estimate vole abundance within each kite territory, we categorized each vegetation plot within a territory as being of high-, medium-, or low-quality habitat for voles. This was done after modelling vole-vegetation associations using stepwise discriminant analysis each month (see Dunk 1992). This procedure was effective at predicting vole abundance (mean correct classification = 70.03% ± SE of 2.66). Vegetation

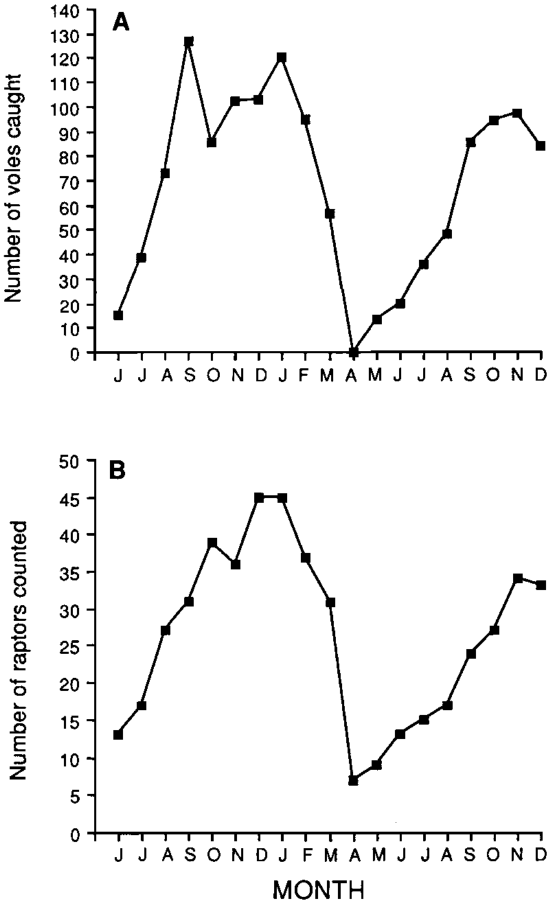

Fig. 1. (A) Number of *Microtus californicus* caught and (B) number of raptors counted each month on Fay Slough Wildlife Area (June 1989–December 1990).

plots from kite territories were classified using the discriminant model developed closest to the time the territory was delineated. Generally, vole populations increased when seeds fell from grasses and became available to them. Vole populations were larger in areas with lower cover height and larger percentage brown grass cover (Dunk 1992). High-, medium-, and low-quality vole habitat was defined relative to the largest number of individuals caught within a month. The percentage of plots in each habitat-quality category was multiplied by territory area, then by naive density (number of individuals caught/grid area) of voles for that category. These were added together to estimate vole abundance for each territory.

Three methods of relating kite territory size to vole abundance were used: (1) regressing territory size on a monthly index of vole abundance expressed as number of individuals caught per 640 trap-nights for the four grids combined; (2) regressing territory size on the estimate of the number of voles within each territory; and (3) regressing territory size on the estimate of vole density within each territory. We also examined the effects of potential competitor abundance on kite territory size by regressing territory size on the estimate of competitor abundance for the month closest to the time the territory was delineated.

We used multiple-regression and partial-correlation analyses to evaluate simultaneously the effects of food abundance and competitor abundance on territory size. Partial-correlation analysis examines the effects of each independent variable while statistically controlling for the effects of variables already entered into the multiple-regression model (Neter et al. 1989); it allows for statistical controls when experimental controls are not feasible (Siegel and Castellan 1988). Partial correlation is useful for examining relationships among a dependent and two or more independent variables when there is a strong relationship between or among independent variables.

RESULTS

In all, 26 kite hunting territories were estimated; three individual's territories were estimated more than once. We were unable to sample vegetation within the territory of one kite. Territory size ranged from 1.6 to 21.5 ha ($\bar{x}$ = 7.8 ± SE of 1.0, n = 26). Vole populations fluctuated annually (Fig. 1A) with populations ranging from 0–914 individuals/ha. Mean estimated number of voles per kite territory was 1,483 ± 163 (n = 25). Vole density within kite territories ranged from 0 to 602/ha ($\bar{x}$ = 277.8 ± 33.2). Potential competitor abundance ranged from 7 to 45 individuals ($\bar{x}$ = 28.9 ± 5.7, n = 26), or 4.8 to 31.0 individuals per km^2 (Table 1).

The correlation between kite territory size and the estimate of prey abundance within each territory was not significant ($r = 0.23$, $P = 0.261$, $n = 25$). Kite territory size was negatively correlated with both total raptor abundance and the index of prey abundance ($r = -0.78$ and -0.75, respectively; $P < 0.001$ for both; Fig. 2). Territory size was also negatively correlated with the estimated density of voles within each territory ($r = -0.71$, $P < 0.001$, $n = 25$). A significant negative correlation also was found between territory size and abundance of conspecifics ($r = -0.64$, $P < 0.001$, $n = 26$). We found a significant correlation between total raptor abundance and the index of vole abundance each month (all grids combined; $r = 0.90$, $P < 0.001$, $n = 19$).

Because separate partial correlation analyses of each year's data (June 1989 through May 1990 and June 1990 through December 1990) showed similar results (Table 2), all data were combined

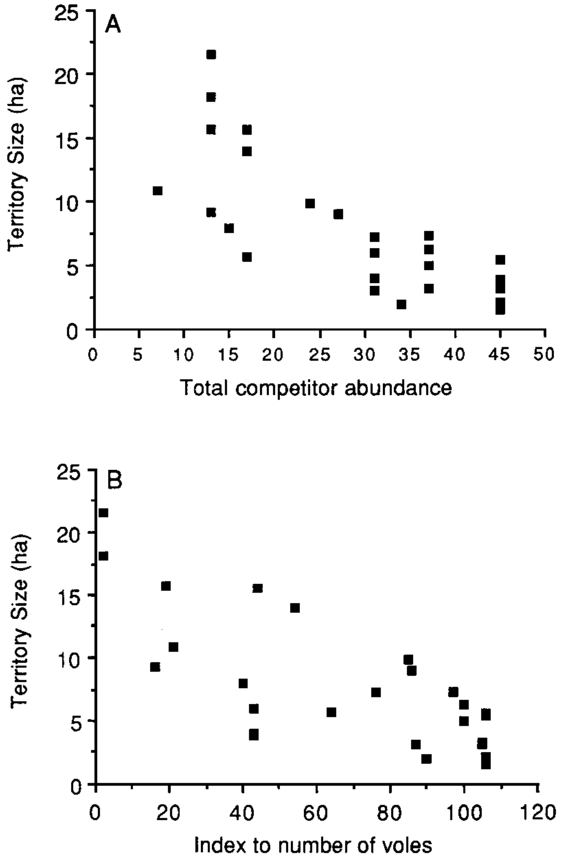

Fig. 2. Relationship of territory size (A) to competitor abundance and (B) to an index of numbers of *Microtus*.

for subsequent analyses. When regressing kite territory size on the estimate of vole density within kite territories and competitor abundance (total raptors), only competitor abundance was entered into the model ($r = -0.76$, $P < 0.001$, $n = 25$). Partial-correlation analysis revealed that once competitor abundance was controlled statistically, prey density did not explain a significant amount ($r = -0.25$, $P > 0.20$, $n = 25$) of the remaining variation in territory size (Table 3). When both variables were forced to enter the model, the partial correlation of competitor abundance remained significant ($r = -0.45$, $P < 0.05$, $n = 25$) when prey density was statistically controlled. Second, we regressed kite territory size on the estimate of prey density and kite abundance (conspecific competitors). Prey density was the only variable entered into the model ($r = -0.71$, $P < 0.001$, $n = 25$). When both variables were forced to enter the model (kite abundance was entered first), partial-correlation revealed prey density

TABLE 1. Monthly raptor counts at Fay Slough Wildlife Area, Eureka, California (June 1989–December 1990).

Species	Month																		
	J	J	A	S	O	N	D	J	F	M	A	M	J	J	A	S	O	N	D
Black-shouldered Kite (*Elanus caeruleus*)	8	11	15	17	20	17	22	19	14	14	5	4	8	7	10	15	15	12	12
Red-tailed Hawk (*Buteo jamaicensis*)	2	2	3	3	3	2	3	3	3	3	1	2	2	3	2	2	2	3	2
Red-shouldered Hawk (*B. lineatus*)	1	1	2	2	3	3	2	3	1	1	0	2	1	2	2	2	2	3	2
Rough-legged Hawk (*B. lagopus*)	0	0	0	0	2	1	1	2	0	0	0	0	0	0	0	0	0	0	0
Northern Harrier (*Circus cyaneus*)	2	3	4	5	7	7	7	7	8	5	1	1	2	3	3	3	5	6	4
American Kestrel (*Falco sparvarius*)	0	0	2	3	3	2	4	4	4	3	0	0	0	0	0	1	2	3	3
Short-eared Owl (*Asio flammeus*)	0	0	0	0	0	3	5	6	6	4	0	0	0	0	0	0	0	6	8
Common Barn-Owl (*Tyto alba*)	0	0	1	1	1	1	1	1	1	1	0	0	0	0	0	1	1	1	2
Total	13	17	27	31	39	36	45	45	37	31	7	9	13	15	17	24	27	34	33

TABLE 2. Partial-correlation analysis results, by year, using Black-shouldered Kite territory size as dependent variable, and competitor abundance and prey abundance as independent variables.

Independent variables	r	Partial correlation	P for partial correlation	Adjusted r^2	Model P	n
		Year 1				
Competitor abundance	−0.875			0.748	<0.001	
Microtus density	−0.806	−0.339	0.20 < 0.10			
Competitor abundance	−0.875			0.748	<0.001	16
Index to numbers of *Microtus*	−0.775	−0.217	<0.50			
		Year 2				
Competitor abundance	−0.522			0.169	0.50 < 0.20	9
Microtus density	−0.495	0.003	<0.50			
Competitor abundance	−0.561			0.229	0.20 < 0.10	10
Index to numbers of *Microtus*	−0.532	−0.070	<0.50			

to continue to be significantly correlated with kite territory size ($r = -0.47$, $P < 0.05$, $n = 25$; Table 3). We also regressed kite territory size on the index of vole abundance and competitor abundance. Again, competitor abundance was the only variable entered into this model ($r = -0.780$, $P < 0.001$, $n = 26$). When both variables were forced into the model, partial correlation revealed that raptor abundance continued to be significantly correlated with kite territory size ($r = -0.446$, $P < 0.02$, n = 26, Table 3).

DISCUSSION

Three other investigators used partial correlation analyses to examine the relationships among food abundance, competitor abundance, and territory size of birds. Myers et al. (1979) found that, once the interaction between prey density and intruder (competitor) density was controlled statistically, prey density had no significant effect on territory size of Sanderlings (*Caladris alba*). Those findings are identical to ours when all raptors are considered as competitors. Conversely, McFarland (1986) found that food supply, not intruder pressure, determined territory size of New Holland Honeyeaters (*Phylidonyris novaehollandiae*). During the first year of his study, Temeles (1987) found that both mouse availability and intruder pressure explained some variation in territory size of Northern Harriers, $n = 7$ territories), whereas in the second year mouse availability explained all variation in territory size ($n = 5$ territories). Myers et al. (1979) and Temeles (1987), however, acknowledged that both factors probably play a key role in territory-size regulation. Ours and the study by Temeles (1987) are the only two that have examined relationships among bird territory size, competitors, and prey in more than one year using partial-correlation analysis. Contrasting with Temeles (1987), we observed the same relationships during both years of our study (Table 2).

We suggest that kite territory size is proximately regulated by competitor abundance and

TABLE 3. Partial-correlation analysis results, for both years combined, using Black-shouldered Kite territory size as dependent variable, and competitor abundance and prey abundance as independent variables.

Independent variables[a]	r	Partial correlation	P for partial correlation	Adjusted r^2	Model P	n
Competitor abundance	−0.762	−0.448	<0.05	0.569	<0.001	25
Microtus density	−0.711	−0.246	0.20 < 0.50			
Microtus density	−0.711	−0.468	<0.02	0.380	<0.001	25
Kite abundance	−0.637	−0.246	0.20 < 0.50			
Competitor abundance	−0.780	−0.446	<0.02	0.613	<0.001	26
Index to numbers of *Microtus*	−0.746	−0.305	0.10 < 0.20			

[a] For each run, only first variable entered into model. Partial correlation for first variable comes from forcing second variable to enter model first.

ultimately regulated by prey abundance, because raptor abundance appeared to be regulated by vole abundance (Fig. 1). Myers et al. (1979) reached the same conclusion about Sanderling territory size. On the beaches inhabited by Sanderlings, food abundance changed rapidly. Myers et al. (1979) suggested that Sanderlings were most influenced by the factor that was least variable (i.e. intruder numbers). Temeles (1987) correctly stated that the strength of a correlation depends partly on the magnitude of variation in the variables examined and that his results could be explained statistically by greater variation in food abundance than intruder variables. Our system is similar in many ways to that studied by Temeles, but we measured both prey and competitor abundance differently. Although we did not examine numbers and types of interactions between territory owners and competitors, we assumed that competitor abundance was positively related to number of interactions between owners and competitors. McFarland (1986) reported a significant positive correlation between numbers of competitors and intrusion rates on his study area. It is unclear when examined over the duration of our study whether vole or competitor abundance is more variable because the two are so closely related (Fig. 1). It is possible that our measure of competitor abundance was more accurate than our measure of prey abundance within territories, which could account for competitor abundance being a better predictor of kite territory size. However, it is unlikely that we would have found such small variation (SE) in our estimate of vole abundance within kite territories if the estimates were not accurate.

We suggest that kites defend as large an area as is energetically feasible at any one time, but that increased competitor abundance reduces the defendable area. Several lines of evidence, some circumstantial, support this contention. Kites can and do adjust territory boundaries with respect to local conditions. Raptors in the Arcata-Eureka area undergo regular annual fluctuations in abundance (Table 1, Fig. 1B), numbers generally increasing in September through October, then remaining relatively constant until March or April, when they decrease dramatically. These fluctuations were closely tied to annual fluctuations of *M. californicus* ($r = 0.90$, $P < 0.001$, $n = 19$, Fig. 1). The largest number of voles estimated to be in a kite territory was 3,340 during mid-August 1990 (territory size was 15.6 ha), which was a time when vole abundance was relatively high, and when many of the winter resident raptors had not yet arrived on the study area. Also, on one occasion we were able to document the precise day that a territory was abandoned by its owner; on that same day the entire territory was incorporated into the territory of a neighboring kite. Village (1982) reported similar results with naturally caused territory abandonment of European Kestrels (*Falco tinnunculus*). Village (1990) experimentally removed European Kestrels from their territories and found that in four of seven instances neighboring birds used vacated territories, but that none of the vacated territories were permanently occupied by "new" birds. However, kestrel territories on Village's study area were much larger (most were greater than 2 km^2; Village 1990) than those of kites on our study area. Thus, dramatic expansions of neighbors into vacated territories might be less likely than on our study area where the largest of 26 territories was 0.22 km^2.

When all raptors were treated as competitors, competitor abundance but not prey abundance influenced territory size of kites. Conversely, when only kites were considered as competitors, prey abundance but not competitor abundance influenced territory size. We believe it is biologically more meaningful to treat all raptors as competitors, rather than conspecifics only. However, defense costs may depend on competitor identity (Temeles 1989, 1990a, b). Kites were commonly observed attacking Northern Harriers, Red-tailed Hawks, Rough-legged Hawks, Red-shouldered Hawks, and other kites. Also, each of the above species, plus American Kestrels and Short-eared Owls were observed to catch and eat *M. californicus* regularly. *Microtus californicus* was by far the most common item in the diet of kites (99%), Red-tailed Hawks (96%), Common Barn-Owls (84%), Short-eared Owls (70%), and Northern Harriers (50%), based on an assessment using regurgitated pellets collected throughout the Arcata/Eureka grasslands (unpubl. data).

During 19 months of small-mammal trapping only four species of small mammals were captured: *M. californicus*, *Reithrodontomys megalotis* (western harvest mice), *Mus musculus* (house mice), and *Sorex vagrans* (vagrant shrews). *Microtus californicus* consistently was the most abundant small mammal and constituted even more of the small-mammal biomass (94%). This along with our findings that the index of *M. californicus* abundance explained 81% of the variation in total raptor abundance strongly suggests that they were the major food source for almost all of the raptors on the study area. Also, kleptoparasitism by hawks on kites was regularly observed. Thus, we considered all species that were using the same food sources to be competitors, as opposed to only conspecifics.

One of our most striking findings was the temporal stability of the estimate of vole abundance within territories, suggesting that kites need about 1,500 voles within their territories. Kites should abandon territories that have much fewer than this number of voles. Apparently, prey abundance and prey availability were correlated in this instance. *Microtus* presumably need a minimum level of cover in order for large populations to survive (Birney et al. 1976). On our study area, voles were virtually absent in grazed areas with vegetation less than 10 cm in height (Cooper unpubl. data). Thus, within areas of dense and tall vegetation, kites needed about 1,500 voles (abundance) so that some were available to foraging kites. Dunk (1992) reported larger vole populations in areas with shorter vegetation (all areas were ungrazed), areas that probably simultaneously maximized their abundance and availability to raptors.

Hunting style and perch availability are also likely to influence raptor territory size. Because kites in California primarily hunt by hovering, we believe their relatively small territory size on our study area was a function of both prey abundance and the fact that they can hunt 100% of their territories. Mendelsohn (1981) reported much larger territory sizes for Black-shouldered Kites (though prey abundances were not reported) in South Africa, where the birds hunt primarily from perches.

Previous researchers of Black-shouldered Kites have reported them to be both territorial and nonterritorial (see review in Henry 1983). We only observed territorial birds, and as a result we are unable to address the role of nonterritorial kites on our study area. Mendelsohn (1981) suggested that kite territoriality would break down at some upper threshold of prey density, but found that South African kites abandoned territories most often when prey populations were low. Possibly, prey populations on our study area never reached this lower threshold. Instead, our findings suggest that territories simply become smaller as a result of increasing competitor abundance (the proximate factor regulating territory size), which is strongly correlated with food abundance (the ultimate factor regulating kite territory size).

ACKNOWLEDGMENTS

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State of California
Department of Fish and Game

Memorandum

Date: April 11, 2016

To: Michael Yaun, Acting Executive Director
Fish and Game Commission

From: Charlton H. Bonham
Director

Subject: Five Year Status Review for Swainson's Hawk (*Buteo swainsoni*)

Attached is the Five Year Status Review for Swainson's Hawk (*Buteo swainsoni*). This Status Review updates descriptions, habitat requirements, threats, research needs, etc., for this species. The Status Review recommends retaining this species as Threatened.

The Swainson's hawk was listed as a threatened species by the California Fish and Game Commission in 1983, pursuant to the CESA (Title 14, California Code of Regulations, §670.5(b)(5)(A)). According to FGC Section 2077, the Department is required to reevaluate Threatened and Endangered species every 5 years by developing a Status Review. The last status review for the Swainson's Hawk was completed in 1993. This status review was prepared to satisfy several management and reporting objectives including Section 2077.

If you have any questions regarding this item, please contact Dr. Eric Loft, Chief, Wildlife Branch, at (916) 445-3555.

Attachment

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CALIFORNIA DEPARTMENT OF
FISH AND WILDLIFE
WILDLIFE AND FISHERIES DIVISION
NONGAME WILDLIFE PROGRAM
1812 Ninth Street
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Status Review:
SWAINSON'S HAWK (*Buteo swainsoni*)

IN CALIFORNIA

Reported to:

California Fish and Game Commission

2016

FIVE-YEAR STATUS REPORT

Table of Contents

I. COMMON NAME, SCEINTIFIC NAME AND CLASSIFICATION

Common Name: Swainson's Hawk

Scientific Name: *Buteo swainsoni*

Current Classification: State Threatened

II. RECOMMENDED ACTION

The California Department of Fish and Wildlife (Department) recommends that Swainson's Hawk retain threatened status under the California Endangered Species Act.

III. SUMMARY OF REASONS FOR RECOMMENDED ACTION

The Swainson's Hawk was listed as a threatened species by the California Fish and Game Commission in 1983, pursuant to the California Endangered Species Act (CESA; Title 14, California Code of Regulations, §670.5(b)(5)(A)). The last status review was completed in 1993 (California Department of Fish and Game 1993). Timely 5-year status reviews have not been possible due to budget, staff, and workload priorities.

The primary threat to the Swainson's Hawk population in California continues to be habitat loss, especially the loss of suitable foraging habitat, but also nesting habitat in some portions of the species' breeding range due to urban development and incompatible agriculture. This impact may have been the greatest factor in reducing Swainson's Hawk range and abundance in California over the last century (California Department of Fish and Game 1993, California Department of Conservation 2011).

Urban development continues to reduce Swainson's Hawk foraging habitat in the Central Valley, particularly in the southern Sacramento Valley (California Department of Conservation 2011). Swainson's Hawk densities are the greatest in this portion of their range, particularly in Sacramento, Yolo, and San Joaquin Counties (see Figure 2). While the Swainson's Hawk is a focus of planning efforts, current General Plans within Sacramento and San Joaquin counties contain goals of converting large areas of natural and agricultural lands that contain suitable Swainson's Hawk foraging habitat to urban features that do not provide foraging habitat (Sacramento County 2011, San Joaquin County 1992). San Joaquin County, however, does have in place an approved Habitat Conservation Plan under which Swainson's Hawk preservation is a major emphasis. In Yolo County, one of the densest areas of hawk territories in the State, current policies focus on preserving both agriculture and Swainson's Hawk foraging habitat. Current efforts under the developing Yolo County Natural Heritage Program (http://www.yolohabitatconservancy.org/) are aimed at maintaining this focus into the future, thereby potentially lessening the long-term impacts to the species once the plan is approved and implemented.

Agricultural cropping patterns directly influence the distribution and abundance of the Swainson's Hawk in the Central Valley (Estep 1989). Swainson's Hawks can forage in natural grasslands, pasture, hay crops, and some irrigated crops but do not preferentially forage in other agricultural crops such as orchards and vineyards once these crops develop their typical canopy (Estep 2009, Swolgaard et al. 2008). This dependence on land use patterns poses a continuing vulnerability for a large percentage of the remaining population based on current trends toward cultivation of largely incompatible crop-types such as orchards and vineyards (California Department of Conservation Agricultural Land Mapping 2010). Compatible crop types do, however, provide a very important benefit to the species (Estep 2008). The lack of suitable nesting habitat throughout much of the San Joaquin Valley, due to conversion of riparian systems and woodland communities to agriculture, also limits the distribution and abundance of Swainson's Hawks (California Department of Fish and Game 1993). The loss of historic sage-steppe/grassland foraging habitat may also be a significant factor in a continuing decline of Swainson's Hawks in portions of the Great Basin and Mojave Desert regions of the state (California Department of Fish and Game 1993). Disturbances on the hawk's Mexican and South American wintering grounds, or during migration, may also contribute to population declines (Goldstein et al. 1996, Sarasola et al. 2005).

At this time, the Department recommends retaining the Threatened classification for this species based on the following:

- On-going cumulative loss of foraging habitats throughout California
- Significantly reduced abundance throughout much of the breeding range compared to historic estimates
- An overall reduction in the hawk's breeding range in California

IV. SPECIES DESCRIPTION AND BIOLOGY

The Swainson's Hawk is a medium-sized raptor with relatively long, pointed wings that curve up while in flight (California Department of Fish and Game 1993). There are three main plumage morphological types: light, rufous, and dark, with several intermediates (Woodbridge 1985). Light morph adults have dark heads, a light chin, and a dark breast band, set off distinctively from the lighter colored belly. In dark morph adults, however, the entire body of the bird may be a drake brown to sooty black. The cere (the fleshy region at the base of the upper bill) is bright yellow and set off distinctively from the dark head. The throat is white or partially white in dark morph adults and the wings are bicolored underneath, with the wing linings generally lighter than the dark, and with gray flight feathers. The light colored leading edge of the wing is a diagnostic feature. Juveniles have the same characteristic underwing markings; however there is more spotting and streaks on the breast and sides than adults (Bechard et al. 2010). Adults generally weigh from 550 to 1100 grams (19 to 39 oz); females, which range between 650 and 1100 grams (23 to 39 oz), are heavier than males, which range from 550 to 850 grams (19 to 30 oz) (Anderson pers. comm. 2012, Bradbury pers. comm. 2012, Estep pers. comm. 2012). Butte Valley hawks in northeastern California seem to be slightly larger than in other areas of the state, with females from 880 to 1300 grams, and males from 620 to 970 grams (Briggs pers. comm. 2012).

The Swainson’s Hawk was historically a species adapted to open grasslands and prairies, but it has become increasingly dependent on agriculture as native plant communities have been converted to agricultural lands (California Department of Fish and Game 1993). This bird also forages in large numbers in managed wetlands during the dry summer months when the vegetation in these wetlands is being mowed or disced (Feliz pers. comm. 2012). The diet of the Central Valley population is varied. The California vole (*Microtus californicus*) is the staple of the diet; however, a variety of other small mammals, birds, and insects are also taken (Estep 1989).

The Swainson’s Hawk breeds in the western United States, and Canada (California Department of Fish and Game 1993). Its winter range occurs in isolated areas of California, Mexico and Central America, through South America and as far south as Argentina (Bechard et al. 2010, Kochert et al. 2011). Generally the Swainson’s Hawk is found in wintering areas from early November through mid-March (England et al. 1997, Kochert et al. 2011, Bradbury pers. comm. 2012). In 1997, six Swainson’s Hawks from the Central Valley were fitted with satellite transmitters and tracked to determine routes of migration and the locations of wintering areas (Bechard et al. 2010). Central Valley birds were located wintering in a region north of Mexico City, Mexico, and near Bogota, Colombia (England et al. 1997), although a hawk from northeastern California was tracked to Argentina during the winter of 1996 (Feliz pers. comm. 2012). One unpublished telemetry study found that Central Valley hawks mostly winter in Central Mexico, but some also end up in central and northern South America (Anderson pers. comm. 2014). A current telemetry study on hawk in the Natomas area of California, has tracked several birds (N= 2 to 4) to Argentina, while the remaining birds went to northern South America, Central America, and Mexico (Anderson pers. comm. 2014). After their long migration north, Swainson’s Hawks arrive at their breeding sites in the Central Valley between March and April (Bechard et al. 2010).

Swainson’s Hawks are generally monogamous, with some undocumented cases of polyandry (Briggs pers. comm. 2012), and show a high degree of site fidelity by returning to the same territory year after year (England et al. 1997, Bechard et al. 2010). Breeding pairs begin to build nests soon after they arrive at their territory, and lay eggs between late-March to early-April (England et al. 1997, Bradbury pers. comm. 2012). Clutch size is between 1 and 4 eggs, but most often 2 or 3 eggs are laid (Bechard et al. 2010). The incubation period lasts 34-35 days (Bechard et al. 2010). The young typically fledge from the nest about 6 weeks after hatching, but may leave the nest as early as 5 weeks old and remain on nearby branches (Bradbury pers. comm. 2012). Craighead and Craighead (1956) reported fledging success of 0.6 young per pair. Studies conducted in the Sacramento Valley reported an average of 1.4 to 1.8 young per successful nest (Estep 2008). In the Butte Valley, Briggs (2007) found productivity to be at 2.01 fledged young per successful breeding attempt. Throughout California, most young have fledged by mid- to late-August, at which point pre-migratory groups begin to form (Bechard et al. 2010). In the Central Valley most young fledge during the first part of July (Bradbury pers. comm. 2012). Migration back to the wintering grounds begins mid-August, and by October most hawks have left California (Kochert et al. 2011).

Several studies on breeding home range have been conducted on California's Swainson's Hawk population. In the Central Valley, home range size varies from 2760 to 4038 ha, with a relatively smaller home range size of 405 ha found in the Butte Valley (Table 1). Home range size is thought to be related to quality of, and distance to foraging habitat (Estep 1989, Babcock 1995, Bechard et al. 2010).

Home Range Size (ha)	**Area**	**Reference**
2760.4	Central Valley	Estep 1989
405	Butte Valley	Woodbridge 1991
4038.4	Central Valley	Babcock 1995
3265.4	Central Valley	Sernke 1999

Table 1. Home range for the Swainson's Hawk in California.

Swainson's Hawks in the Central Valley often nest at the periphery of riparian forests or in riparian corridors where they have greater access to foraging areas, but virtually any suitable tree may be used (Estep 1989, England et al. 1995, Bechard et al. 2010). Hawks will also use lone trees in agricultural fields or pastures, and roadside trees when they are adjacent to suitable foraging habitat (Estep 1989, Anderson et al. 2007). Estep (1989) found Valley oak (*Quercus lobata*), Fremont cottonwood (*Populus fremontii*), walnut (*Juglans sp.*), and willow (*Salix* sp.) are the most commonly used nest-tree species, with an average height ranging from 12.6 to 25 m (41.3 to 82.0 ft), Similarly, Anderson et al. (2007) found Valley oak, cottonwood, willow and *Eucalyptus* spp. were more frequently used, with an average height between 14.8 to 16.2 m (48.6 to 53.1 ft).

In the Great Basin, Swainson's Hawks occupy the juniper/sagebrush community typical of the area; however, much of the lowlands have been converted to agriculture (Bloom 1980, Woodbridge et al. 1995). Junipers (*Juniperus occidentalis*), with an average height of 4.6 m (15.0 ft), are most commonly used as nest trees in the Great Basin (California Department of Fish and Game 1993). The diet of the Great Basin population consists largely of montane meadow voles (*Microtus montanus*) and Belding's ground squirrels (*Spermophilus beldingi*) (California Department of Fish and Game 1993).

Other areas in California inhabited by small populations of Swainson's Hawk include the isolated desert areas in the Mojave National Preserve regions of the western Mojave Desert, the greater Antelope Valley near Lancaster, and in the Owen's Valley along the eastern edge of the Sierra Nevada (see Figure 2). Joshua tree (*Yucca brevifolia*), ornamental trees, and lone trees along roadsides or on private property are commonly used as nest trees in these regions (Bloom 1980).

V. HABITAT REQUIREMENTS

Large open areas of suitable foraging habitat with abundant and available prey base in association with suitable nesting habitat are basic requirements for the successful reproduction of Swainson's Hawk (Estep 1989). Historically, the natural foraging habitat of the Swainson's Hawk was primarily open stands of grass-dominated vegetation and relatively sparse shrublands (Bloom 1980, Bechard et al. 2010). However, much of the original foraging habitat in California has been converted to either urban landscapes or agricultural production. Consequently, the Swainson's Hawk has shifted its foraging strategy to rely more heavily on agricultural crops (Bloom 1980, Estep 2009).

Today, suitable foraging habitat includes a variety of agriculture crops, grassland, and pasture. In the Central Valley, Swainson's Hawks forage more often in mixed agricultural lands that support irrigated hay crops (e.g. alfalfa), as well as dryland pasture, grassy ruderal lots, and some irrigated crops, due to a higher accessibility and relative abundance of prey (Bloom 1980, Estep 1989, Babcock 1995, Smallwood 1995, Swolgaard et al. 2008, Anderson et al. 2011). Alfalfa fields are more routinely used by foraging Swainson's Hawks than any other crop type (Bloom 1980, Woodbridge 1985, Estep 1989, Babcock 1995, Sernka 1999, Swolgaard et al. 2008, Anderson et al. 2011). Anderson et al. (2011) reported that 63% of observed foraging occurred in alfalfa.

The ability of the hawk to use agricultural crops for foraging is dependent on a complex interaction of crop structure and the timing of agricultural practices (Bechard 1982, Schmutz 1987, Estep1989, Woodbridge 1991, Smallwood 1995, Sernka 1999, Estep 2009). Prey species may be displaced during irrigation, burning, and harvesting activities, which often allows for ample foraging opportunities for Swainson's Hawks and other predators (Sernka 1999). The availability of prey is also largely dependent on the crop structure. Certain crops provide improved foraging opportunities for Swainson's Hawks due to high prey numbers, low vegetation structure, and favorable farming practices (e.g. mowing, irrigating; Estep 1989, Babcock 1995, Sernka 1999, Swolgaard et al. 2008, Estep 2008, Estep 2009). Some crops and managed wetlands are useful in foraging for a period after harvest, but may remain relatively unavailable in other periods of crop growth; likewise, other crops are available early in the season when a less dense vegetative structure and shorter height allows for access to prey (England pers. comm. 2012, Feliz pers. comm. 2012).

In a report to the Yolo Natural Heritage Program, Estep (2009) described the relative value (low to high) of vegetative structure and accessibility of different agricultural crop types in Yolo County to foraging Swainson's Hawk. Based on two main components, prey accessibility and prey availability, Estep (pers. comm. 2012) places high value on alfalfa, and on wheat, tomatoes, and beets during harvest; moderate value on irrigated and non-irrigated pasture, grasslands, and some other annually rotated crops; low value safflower, sunflower, corn and rice; and little to no value on orchards and vineyards. The variety of habitats used for foraging by this hawk suggests that maintenance of large heterogeneous areas of agricultural habitats and grasslands, which include a high percentage of alfalfa, should be a priority for conservation of the species (Swolgaard et al. 2008, Estep 2009, Anderson et al. 2011).

Unsuitable or low value foraging habitat includes any habitat which does not support adequate prey abundance, as well as any habitat in which prey are inaccessible to foraging hawks due to vegetation characteristics (e.g. vineyards, mature orchards, cotton fields, dense or tall vegetation). For example, orchards and vineyards in general are not suitable foraging habitat for Swainson's Hawk due to the dense woody cover making prey unavailable (Estep 1989, Babcock 1995). In a study to ascertain the extent of vineyard use by Swainson's Hawk in the Central Valley, Swolgaard et al. (2008) observed relatively low foraging levels in vineyards and stated that "large contiguous areas of vineyards are likely unsuitable for foraging by Swainson's Hawk at a population level."

Suitable nesting habitat includes trees within mature riparian forest or corridors, lone oak trees and oak groves, and mature roadside trees. It is thought that trees on the periphery of riparian habitat are preferred by Swainson's Hawk (Estep 1989, England et al. 1995, Bechard et al. 2010). The majority of documented Swainson's Hawk nest trees in the Central Valley have been found in riparian systems in Sacramento, Sutter, Yolo, and San Joaquin counties, making this habitat type critically important (Schlorff and Bloom 1983). This is likely the case for nesting hawks in the San Joaquin Valley as well; however the hawks that regularly nest here have not been extensively studied. A portion of the Swainson's Hawk population also resides in the Great Basin of Northeastern California where hawks typically nest in juniper trees (Bloom 1980). Swainson's Hawks have been observed in several studies to select nest sites in greater densities when near large tracts of agricultural lands than when adjacent to non-agricultural lands (e.g. urban, annual grassland, or even vernal pool landscapes; Bloom 1980; Estep 1989; Babcock 1995; Smallwood 1995; Swolgaard et al. 2008). Data collected during Department Swainson's Hawk nest surveys in 2002 through 2009 indicated that nests were clumped at higher densities in mixed agricultural landscapes (Gifford et al. 2012). Nest sites are generally adjacent to, or within easy flying distance to suitable foraging habitat that provides available prey resources (England et al. 1995). The Swainson's Hawk is also known to nest within urban environments, such as Davis, Stockton and Sacramento, California; however, what is known about these nesting pairs is largely anecdotal as there have been no focused studies on these hawks.

Wintering habitat in California is less critical for Swainson's Hawk because only a small number of hawks have been documented to over winter in California (Herzog 1996; Anderson pers. comm. 2012; eBird 2012). In the Central Valley Delta region, overwintering hawks have been documented to roost in numbers of 10 to 30 individuals, mostly comprised of adults and some juveniles, in large cottonwoods or eucalyptus trees (Anderson pers. comm. 2012). During the day these hawks disperse on the nearby landscape to forage either individually or in groups with red-tailed hawks, Ferruginous hawks, rough-legged hawks, corvid species, and other raptors. It is unknown where these wintering birds originated (Anderson pers. comm. 2012).

During the breeding season and just prior to their annual fall migration period, Swainson’s Hawk in California often congregate in groups from 5 up to 100+ individuals (Anderson pers. comm. 2012). Foraging often occurs during congregation, but communal roosting may also take place. Congregations during the breeding season happen nearer nesting sites and groups will sometimes form during any portion of the nesting cycle (nest building to fledgling care). Late summer-fall congregations may occur during delayed migration periods lasting up to three months starting in early August through late October. These congregation areas can occur anywhere there is food available, but are typically associated with alfalfa, other hay crops, and various row crops (excluding orchards and vineyards) that have been recently mowed, disced, harvested or irrigated (Anderson pers. comm. 2012). Support for practices that provide for these critical breeding and pre-migration congregation areas is an important conservation need.

VI. NATURE AND DEGREE OF THREAT

Foraging Habitat Conversion to Urban and Non-Suitable Habitat

Fragmentation of habitat has been observed to adversely affect long-term viability of animal populations, and can be defined as dissection of habitat into smaller portions that does not allow free movement of individuals (Fahrig 2003). Habitat fragmentation has two components, both of which contribute significantly to, and may even cause, extinctions for some species: (1) reduction in total habitat area, and (2) redistribution of the remaining area into disjunct fragments (Wilcove et al. 1986).

Significant loss of agricultural lands and foraging habitat has occurred in counties within the Sacramento and San Joaquin valleys due to urban development. According to the State of California’s 2008-2010 California Farmland Conversion Report (California Department of Conservation 2014), Southern California and San Joaquin Valley counties were included in the “top ten list” of California counties with the most acres converted from farmland to urban land. Irrigated farmland was the source of 25 percent of all new urban land statewide, with another 30 percent of new urban land derived from dryland farming and grazing uses, and 45 percent from natural vegetation or vacant lands. Direct conversion of irrigated farmland to urban land was 25 percent of total new urban growth for both the Sacramento and San Joaquin valleys. Land idling was the most prevalent in the southern San Joaquin Valley and counties in the Sacramento-San Joaquin Delta. If current trends in habitat conversion of compatible agriculture to urban development continue, the Swainson’s Hawk population will likely experience reduced foraging opportunities, which may result in a further reduction in the species’ range, distribution, and abundance.

Native foraging habitat in the lowland areas of the Great Basin also has been converted to agricultural land (Bloom 1980). The smaller Great Basin Swainson’s Hawk population, while not subject to the same urban development pressures as the Central Valley population, is becoming more dependent on the agricultural system of the region to provide suitable foraging habitat (California Department of Fish and Game 1993). As agricultural conversion continues to replace native habitat, the suitability of crop-types could determine the level of Swainson's Hawk foraging use. Ultimately the distribution of crops dictates the distribution and abundance of Swainson's Hawks in the Great Basin as it does in the Central Valley (California Department of Fish and Game 1993).

There has been a steady decline in active Swainson's Hawk territories occupying rangeland habitat in the Great Basin region of the state. Overgrazing and fire suppression have caused an increase in juniper forest and sagebrush communities (Miller and Rose 1999, Miller et al. 2001). The Swainson's Hawk decline in this area may have been a result of the increase in juniper/sage habitat at the expense of sage-steppe/grassland communities. Replacement of sage-steppe/grassland with juniper/sage habitats results in a reduction of microtine rodents and ground squirrels, the principal prey of the Swainson's Hawk in the Great Basin (California Department of Fish and Game 1993). While Swainson's Hawks have steadily declined in rangeland habitats of the Great Basin, there has been an apparent increase in breeding pairs utilizing agricultural foraging habitats such as alfalfa fields, largely due to greater prey densities and availability of prey in these areas (California Department of Fish and Game 1993).

Habitat Conversion to Vineyards and Orchards

Vineyards and orchards are considered low value foraging habitat for Swainson's Hawk because of low prey density and vegetation structure which prevents hawks from stooping on prey (Estep 1989, Smallwood 1995). Statewide, wine grape acreage has approximately doubled since 1990 (California Department of Conservation Agricultural Land Mapping 2010). Conversion of undeveloped land to vineyards involves the clearing of native upland and riparian vegetation. This type of conversion has the potential to affect Swainson's Hawk breeding and foraging habitat.

The 2008-2010 California Farmland Conversion Report (California Department of Conservation 2014) states that while urbanization is a leading component of agricultural land conversion throughout the state, economic and resource availability factors (i.e. water) also lead to conversion to more intensive agricultural uses, including orchards and vineyards. Conversion from grasslands to orchards, mainly almonds, was the most widespread form of conversion in 2010, with the Sacramento Valley having more conversions to high density olive orchards. Again, if conversion of compatible foraging habitat to non-habit continues, the Swainson's Hawk population in California will likely be impacted.

Breeding Habitat Conversion

Swainson's Hawks are not exclusively or predominately associated with nests in riparian areas, although a significant portion of the known nesting population in the Sacramento and San Joaquin Valleys occur in riparian areas (Bloom 1980, Estep 1989). Loss of suitable breeding habitat through conversion of riparian and woodland habitat to agriculture and unsuitable urban environments is a concern for breeding Swainson's Hawks across California, particularly in the San Joaquin Valley where suitable nest trees are in lower abundance. Loss of lone trees along roadsides to road maintenance and construction may also impact breeding Swainson’s Hawks as many of these trees are in proximity to suitable foraging habitat and are often used by Swainson’s Hawks.

Implementation of levee vegetation removal policies could result in significant impacts to Central Valley Swainson’s Hawk populations as a large portion of suitable nesting habitat may be removed. In April 2010, the Department’s Director and the Department of Water Resources wrote a letter to the U.S. Army Corps of Engineers (Corps; DWR and CDFW 2010) expressing concern over the Corps’ issuance and use of a new levee vegetation removal policy (USACE ETL 1110-2-571), and stating that “the proposed vegetation policy will likely have devastating environmental impacts, as the remnants of the once vast riparian forests and adjacent riverine ecosystems of the Central Valley are now concentrated on the banks and levees of its flood channels”.

Climate Change

Climate change adds unpredictability to the existing suitable breeding and foraging habitats and could cause additional stress on Swainson’s Hawk populations. These impacts, both to suitable habitats and to populations, can be generally anticipated based on current climate research. However, the level of these impacts is impossible to predict with accuracy or precision. Most climate projection studies agree that California will retain its typical Mediterranean climate (i.e. cool, wet winters and hot, dry summers), yet the degree of wetness/dryness will likely be amplified and vary by location across California (Pierce et al. 2011, Cayan et al. 2012,). Impacts may include increased winter runoff and flooding (with possible impacts to riparian nesting habitat) and sea level rise (with possible inundation of low-lying nesting or foraging habitat), more frequent extreme temperature events, and less snowpack (Pierce et al. 2011, Cayan et al. 2012).

Limited water availability in the summertime may significantly reduce the supply of water and therefore reduce prevalence of alfalfa and other high-quality foraging habitat. In addition, drought conditions associated with long-term changes in precipitation may negatively impact prey abundance (CDFW 2016), and consequently impact breeding success and survival of Swainson’s Hawks.

The 2006 Executive Order S-06-06 calls for the increased production and use of bioenergy, including ethanol and biodiesel fuels made from renewable resources, largely comprised of corn. The market price for energy crops could result in farmers shifting to those crops that do not provide high value habitat to the Swainson’s Hawk. For example, one study looking at agriculture impacts of climate change in Yolo County predicts that crops with high water utilization, such as alfalfa, are likely to become more scarce on the landscape in the future if water availability declines, and crops with a higher cash value per unit of water, such as vegetables, fruits and nuts will become more common (Jackson et al. 2009). Other potential indirect impacts may come from practices aimed at mitigating climate change. The future agricultural landscape could change from the existing mosaic of crops to grasses that can be used for carbon sequestration. Changing crop types to those less frequently irrigated and harvested, or those that would store carbon for a longer time period could still provide habitat, but research is needed to understand the potential scale of the changes and how that could affect the range and reproductive success of the Swainson’s Hawk (Bradbury 2009).

Renewable Energy Facilities

Wind energy project areas contribute to direct mortality of Swainson’s Hawk through turbine strikes, particularly where wind resource areas overlap with hawk foraging areas. Swainson’s Hawk mortality from wind turbines has been documented by Kingsley and Whittam (2001). The Solano County Wind Resource Area, which overlaps with the range of Central Valley Swainson’s Hawks, has one of the highest raptor abundances of California’s wind resource areas and initial studies show substantial numbers of bird and bat mortalities related to wind development. Birds most susceptible to this source of mortality are those that fly at or below the maximum blade height of wind turbines, particularly while hunting (Orloff and Flannery 1992), as is the case with Swainson’s Hawks.

Disease

There have been some documented cases of Swainson’s Hawk having experienced West Nile Virus (WNV) mortality. One Swainson’s Hawk has been reported to test positive for WNV in California (reported in South Lake Tahoe area, but thought to have been brought from Mono County; Center for Disease Control and Prevention database), and another was confirmed positive by the Department’s Wildlife Investigation Laboratory in 2015 from Contra Costa County (Rogers pers. comm. 2015). Eleven Swainson’s Hawks were found dead with WNV infection in the USA from 1999 to 2004 (Nemeth et al. 2006). However, the extent of vulnerability WNV presents to the Swainson’s Hawk is unknown at this time. Increased levels of WNV in California populations could exacerbate the effects of other threats on this species.

Contaminants

Insecticides are responsible for high mortality rates in hawks that migrate to Argentina. Prior to northerly migration, when flocks feed on insects in nearby harvested agriculture fields, several large-scale mortality events of Swainson’s Hawks (>1000’s found dead) were reported in Argentina due to applications of organophosphate and carbamate insecticides in agricultural fields (Goldstein et al. 1996). However, many of the birds that breed in California winter in Mexico, where the timing of pesticide applications poses less of a threat. Therefore, the importance of this factor for California’s breeding hawks is unclear.

Application of anticoagulant rodenticide (AR) is a known threat to raptors due to ingestion of poisoned prey. Numerous field monitoring studies on raptor species indicate lethal and sublethal impacts of AR exposure (Stone et al. 2003, Murray 2011, Thomas et al. 2011, Christensen et al. 2012). Pesticide use throughout the Swainson’s Hawk’s range, specifically targeting ground squirrels, may also impact Swainson’s Hawks and cause secondary poisoning. In 2015, the Department's Wildlife Investigation Laboratory confirmed two AR exposures for Swainson's Hawks, both from Contra Costa County, with the cause of death in one due to AR toxicosis (Rogers pers. comm. 2015). Although the evidence indicates raptors are negatively affected by pesticide use, further research is needed to determine what extent Swainson's Hawks also incur these same impacts.

Other Direct Mortality Agents

Swainson's Hawk mortality is reported occasionally in California. Direct mortality of birds can be due to several actions as also described elsewhere in this document, including trimming of nest trees (typically due to construction or utility maintenance activities), shooting, vehicle collisions, electrocution, or pesticides. Biologists have only occasionally found shot or electrocuted Swainson's Hawks.

Stochastic Events

A mass mortality event of wintering Swainson's Hawk was observed in Argentina during November of 2003 when 113 Swainson's Hawks were found dead as a result of a single hailstorm (Sarasola et al. 2005). In addition, 14 hawks with severe injuries were recovered alive, but only 10 of these survived. Another 45 dead birds of 11 species were collected in the area. Interviews with local landowners conducted in other areas of these wintering grounds provided further evidence of past hailstorm-related mortality involving the hawk, suggesting that such events commonly occur in the Argentine Pampas. This potential cause of mass mortality of Swainson's Hawk wintering in agricultural areas of Argentina may be significant when added to the increased mortality associated with poisoning events during the last decade. Even though California's Central Valley Swainson's Hawk population is known to largely over-winter in Mexico, the Central Valley population may experience similar events.

VII. HISTORICAL AND CURRENT DISTRIBUTION

Historical Distribution (pre-1980)

Information gathered through an extensive search of the literature and museum records allowed Bloom (1980) to estimate the historic range of the Swainson's Hawk in California (Figure 1). From this analysis, Swainson's Hawks were found throughout the state except in the Sierra Nevada, North Coast Ranges and Klamath Mountains (Bloom 1980). Historically, the species was found in large, open grassland valleys with scattered trees or groups of trees. Swainson's Hawks also established breeding territories in foothill and canyon habitat. The valleys and deserts of southern California and the coastal valleys from the Santa Rosa Valley south to the Mexican border supported significant populations of Swainson's Hawks.

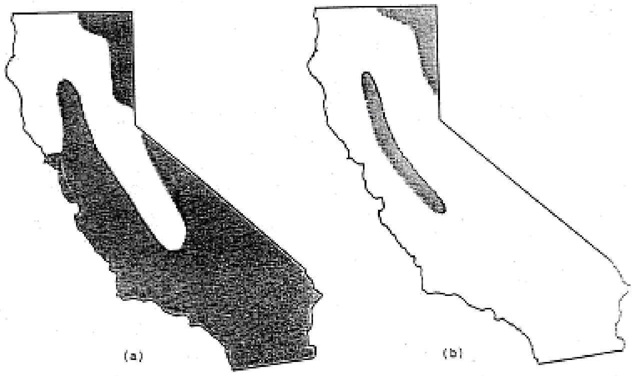

Figure 1. This figure was taken from Bloom 1980 and shows the historic (a) and current (b) range of Swainson's Hawk in California, as understood at that time.

In 1979, Bloom surveyed much of the state to determine the current distribution of Swainson's Hawks (Bloom 1980). In his report he depicted eight major geographic regions in California where Swainson's Hawk were found. The greatest number of nesting Swainson's Hawks were located in the Central Valley and also in the Great Basin of northeastern California from Butte Valley east to Nevada, south-central Modoc County and eastern Lassen County (Bloom 1980). In addition, Swainson's Hawks were also located in the Shasta and Owens valleys, and the Mojave Desert (Bloom 1980). Bloom's description of Swainson's Hawk distribution remains consistent with current knowledge and more recent data do not contradict Bloom's estimate of distribution as explained below.

Current Distribution (post-1980)

In 1988, the Department surveyed the entire Central Valley, coastal valleys, and parts of Southern California, and was provided with information from cooperators in the Great Basin region of the state. In addition, information on Swainson's Hawk activity was gathered by the Department from 1979 to 1993 throughout the state (California Department of Fish and Game 1993). These data revealed no change in the distribution of the Swainson's Hawk in California since Bloom's 1980 report (California Department of Fish and Game 1993).

In 2005 and 2006 another statewide survey of Swainson's Hawk breeding pairs was conducted using a stratified random sample design (Anderson et al. in prep). The results of these survey findings roughly duplicate Bloom's (1980) earlier findings, with the majority of Swainson's Hawk records located in the Central Valley, and with the next large population center in the Great Basin. However, this survey was only focused within the current known distribution and did not cover areas of the state where Swainson's Hawk had historically nested and the species was presumed extirpated (Anderson et al. in prep). For example, additional areas not included in the 2005 and 2006 survey include some areas in Sonoma and Napa counties. Recently, 3 to 4 Swainson's Hawk nests have been detected in upland habitat at the north end of San Francisco baylands near Highway 37 (Fish pers. comm. 2012). These nests have been monitored as part of the Golden Gate Raptor Observatory's Bay Area Raptor Nesting Survey over the last few years.

The Department's California Natural Diversity Database (CNDDB) records contain 2,394 Swainson's Hawk occurrence records, ranging from 1894 to present (California Natural Diversity Database; December 1, 2015). Eighty-five percent (2029/2394) of the CNDDB records occur within the Central Valley, and 59% (1407/2394) occur within Sacramento, Yolo, Solano, and San Joaquin counties. CNDDB records largely corroborate Bloom (1980) and Anderson et al. (in prep) results in that the majority of the records occur within the Central Valley (Figure 2). A majority of records (n=2140) are from 1990 on. Of equal importance, in areas of the state where Bloom reported that the Swainson's Hawk had been extirpated, CNDDB similarly contained no Swainson's Hawk records. There are no CNDDB records in the Sierra Nevada, North Coast Ranges, and Klamath Mountains, and with the exception of a handful of new records in Napa County, Sonoma County, and two records in San Luis Obispo County. CNDDB provides no indication that the species has reoccupied historical range in coastal valleys from Santa Rosa south.

EBird (http://ebird.org) is a citizen science database that houses bird observation data. To supplement CNDDB data, we extracted likely breeding records (e.g. observations with noted breeding activity, nest location, eggs or young) for Swainson's Hawks in California from 1995 during the breeding season (April through August). We found 716 breeding records in eBird, some of which may duplicate CNDDB occurrences (see Figure 2). Some caution should be used when interpreting eBird data for breeding activity. EBird is an observational database not meant to track breeding status of any one species, and designation of breeding status from extracted data in this case was largely gleaned from the notes a submitter entered. Therefore, some breeding observations may have been missed, while others misclassified. Although the incoming data to eBird receives some level of scrutiny via automated filters and volunteer reviewers, there is still some margin of error. Alternately, incoming records for CNDDB receive a much higher level of verification before it is added and viewable.

The data for Swainson's Hawk recorded in the CNDDB and eBird is not collected in a systematic fashion and for this reason its use as the principle measure for describing the species' distribution and range is open to criticism. Nevertheless, the accumulation of over 2,300 Swainson's Hawk observational records in CNDDB and over 700 in eBird can be used, in conjunction with other records, to form a better understanding of the species' current distribution and range.

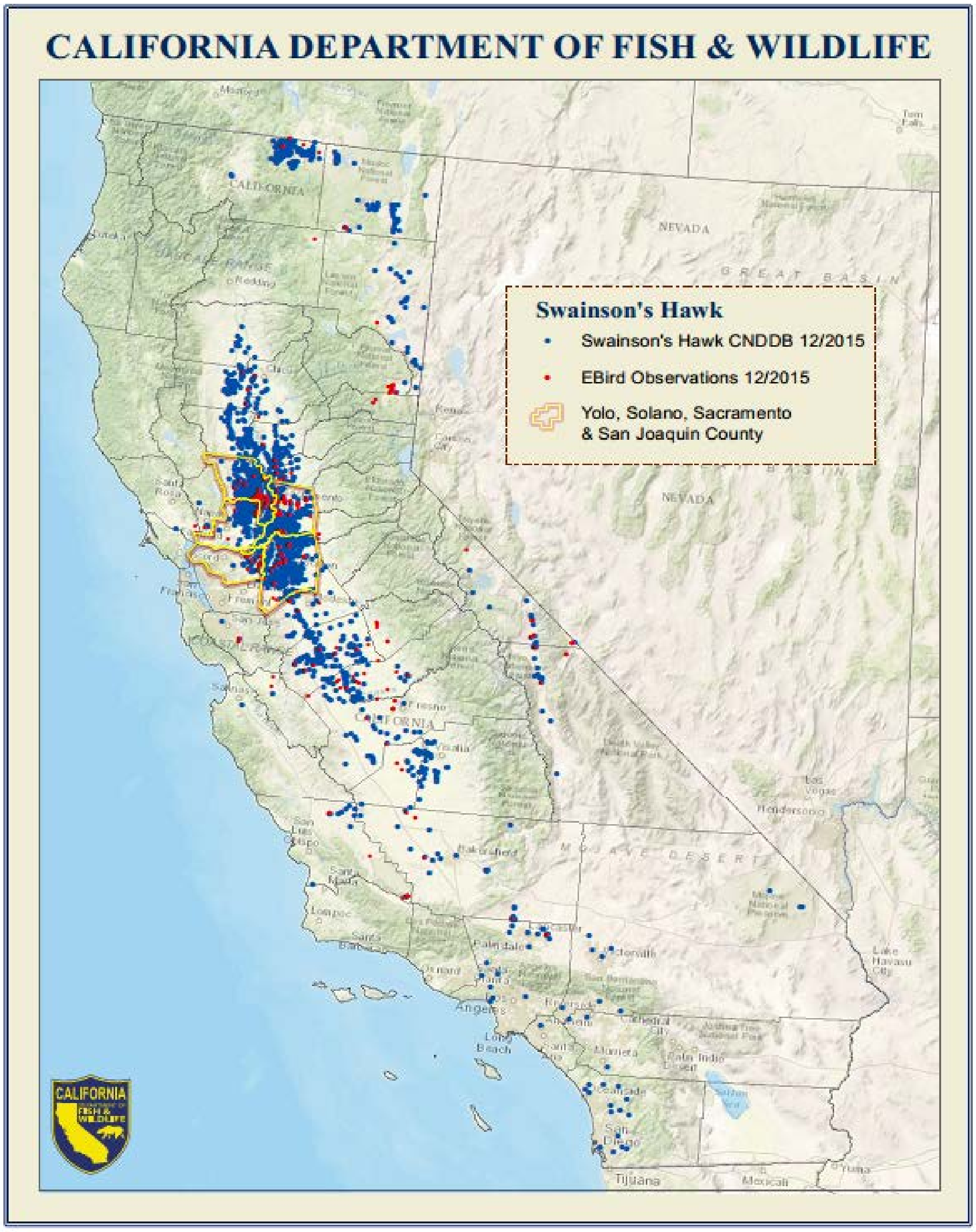

Figure 2. CNDDB and eBird data for Swainson's Hawk in California (extracted from CNDDB 12/1/2015 and eBird in 12/15/2016). The majority of the Central Valley's Swainson's Hawk population lies within an area that includes Sacramento, Yolo, Solano, and San Joaquin counties.

As previously mentioned, Bloom (1980), Gifford et al. (2012), Anderson et.al. (in prep.), CNDDB occurrence records, and eBird breeding records all indicate that the majority of Swainson's Hawk nests are located in the Central Valley and that the nesting density in the Central Valley is unevenly distributed. Approximately 70 to 80% of the Central Valley population is located in the southern Sacramento-northern San Joaquin Valley, a region composed of four counties: Yolo, Solano, Sacramento, and San Joaquin (Bloom 1980, Anderson et.al. in prep., Gifford et al. 2012). These four counties are located in the Central Valley, where suitable irrigated farmland is the primary land-use (Estep 1989). Numbers of breeding pairs decreased both to the north and south of this four county region, and no significant foothill breeding populations have been documented. Another important Swainson’s Hawk population center is in the Great Basin.

The distribution of the Swainson’s Hawk has changed little since Bloom (1980) originally described the species distribution. With few exceptions, areas within the historical range, particularly along the Central Coast and southern regions, have not been reoccupied, and the Central Valley and Great Basin continue to provide the species its core habitat in California. However, the Antelope Valley is considered reoccupied by some, probably as a result of irrigated agriculture, as well as some inner coastal valleys, portions of the Sierra foothills, and some portions of the San Joaquin Valley (Estep pers. comm. 2012).

VIII. HISTORICAL AND CURRENT ABUNDANCE

Historical Abundance

Historically, the Swainson’s Hawk was considered one of California’s most common nesting buteos (Sharp 1902), but the population declined dramatically around 1900, concurrent with a contraction of the species’ range, particularly along the central and southern coastal areas of California. Bloom (1980) estimated as many as 17,136 pairs of Swainson’s Hawks historically nested in California (includes data from 1880-1969). This estimated 90% decline in the population and the loss of a significant portion of its range prompted the hawk’s listing by the State of California as a Threatened species in 1983 by the California Fish and Game Commission pursuant to CESA. (See Cal. Code Regs., tit. 14, §670.5(b)(5)(A).

Current Abundance

In a 1979 survey, Bloom (1980) estimated that there were only 375 (±50) breeding pairs of Swainson’s Hawks remaining in California. Since this estimate was made and the hawk was listed in 1983, interest in the Swainson’s Hawk has grown considerably. Thus there has been an increased survey effort throughout the state. This increase in data collection efforts may be one reason we see higher breeding densities reported from certain areas within the state. A 1988 estimate of the Central Valley population was obtained using nest density information contained in the study by Estep (1989), where an area estimate of the habitat was multiplied by a breeding density of 0.16 pairs/sq km (0.42/sq mi) (the lowest breeding density of Estep’s four study areas in the Central Valley, totaling an area of 374.4 sq km). The results indicated an estimate of 430 pairs in the Central Valley. This estimate was further subdivided into three main regions of the Central Valley: 80 pairs were estimated south of and including the Merced River, 35 pairs north of Sutter Buttes in Sutter County, and 315 pairs between these areas. Using survey data and population estimates derived by biologists working in the Great Basin region, the population for that area was estimated to be 110 pairs (Estep 1989). In addition, five pairs were estimated for the Owens Valley area, and five for the Mojave Desert area (Estep 1989). The species was assumed to be extirpated from Southern California and coastal valleys. The individual estimates were combined to form a total statewide estimate of 550 breeding pairs in 1988 (Estep 1989). Neither Bloom 1980 nor Estep 1989 methods to estimate the population of hawks was sufficient to provide a statistically rigorous estimate.

More recently, Anderson et al. (in prep) completed a survey of the statewide breeding Swainson's Hawk population in 2005 and of the Central Valley breeding population 2006, and estimated the number of breeding pairs statewide at 1,893 (95% CI, 1462-2325) in 2005 and an estimated the number of breeding pairs in the Central Valley at 2,251 (95% CI, 1811-2690) in 2006. Another recent survey of nesting Swainson's Hawk was conducted in a portion of the Central Valley (Butte to San Joaquin counties) during the period 2002 to 2009 (Gifford et al. 2012). The latter survey yielded yearly estimates for numbers of breeding pairs of Swainson's Hawks in the Central Valley north of the Stanislaus River and south of Red Bluff: in 2002 the estimate was 593 (388-798) breeding pairs; in 2003 the estimate was 1,008 (720-1,296) breeding pairs; and in 2009 the estimate was 941 (692-1,190) breeding pairs (Gifford et al. 2012). Both Anderson et al. (in prep) and Gifford et al. (2012) methods employed to estimate the population of hawks were sufficient to provide a statistically rigorous population estimate, and are designed to be repeatable in order to accurately detect changes in the breeding population of Swainsons's Hawks within each of their study areas.

Compared to historical distribution and abundance, current surveys have indicated a smaller population occupying a restricted range that includes the core habitat areas of the Central Valley and Great Basin. Surveys subsequent to Bloom's 1979 inventory (Bloom 1980) have resulted in higher population estimates within these core areas, but it is unknown if this was due to an increase in survey effort or an actual increase in the population. Recent surveys employing repeatable survey designs hold promise for future comparative analysis.

IX. POPULATION TREND

Raptors may experience year-to-year changes or fluctuations in their population numbers due to a variety of factors including changes in prey abundance, habitat, and weather. In order to detect long-term changes over time (i.e. trends) in California's Swainson's Hawk population, it is necessary to collect data over a sufficient number of years to span any short-term population fluctuations or cycles (Hatfield et al. 1996; Newton 1998; Lewis and Gould 2000).

Historical statewide population estimates were based on a limited number of annual surveys and were not designed to be repeated (Bloom 1980, Estep 1989). Anderson et al. (in prep.) used repeatable survey efforts statewide with a repeatable survey design over two years to estimate the number of nesting hawks. Gifford's et al. (2012) also used repeatable survey efforts and covers a seven year interval; however, the study area is limited to the northern portion of the Central Valley and again, and the time period is insufficient to span population fluctuations or cycles (Hatfield et al. 1996; Newton 1998; Lewis and Gould 2000). Due to differences between the two studies in survey design, duration and scope, neither of these surveys can currently be used to accurately estimate a statewide trend for Swainson's Hawk.

The Breeding Bird Survey (BBS) is a dataset that spans a sufficient length of time to be useful in detecting trends in the Swainson's Hawk populations. The BBS is a long-term, large scale avian monitoring program initiated in 1966 (1968 in California) to track the status and trend of North American bird populations. Each year during the height of the avian breeding season, participants skilled in avian identification collect bird population data along randomly selected roadside survey routes. The raw data for survey routes in California are accessible on the BBS website, http://www.pwrc.usgs.gov/BBS/. In addition to collecting and storing raw data the website also provides tools for trend analysis.

The BBS data has been used in over 450 publications and is often the only long-term data set available for avian trend analysis. However, use of BBS data is controversial because of a number of possible sources of error. These include missing data, observer bias, alternating observers, biases due to road-only surveys, and BBS's index method for population abundance (rather than a true estimate of the population). The BBS data on Swainson's Hawk for California are marked as "data with an important deficiency" (USGS 2012). Data may be so marked because:

1. The regional abundance is less than 0.1 birds/route (very low abundance),
2. The sample is based on less than 5 routes for the long-term (very small samples), or
3. The results are so imprecise that a 5% per year would not be detected over the long-term.

Cautious of the potential for errors in interpretation, the BBS appears to be useful for analyzing population trends for Swainson's Hawk populations in California. More than 30 routes monitored over the last 40 years have recorded the occurrence of Swainson's Hawk (Sauer et al. 2011; USGS 2012). The roadside surveys are conducted in peak breeding season while Swainson's Hawk are active, visible and easily identified as they rear young. Therefore, the data collected by BBS presents a potentially valuable resource for trend analyses.

The trend analysis presented in Figure 3 for Swainson's Hawk populations is taken from the BBS website and is based on the current BBS hierarchical model for population change (Sauer and Link 2011, Sauer et al 2011). The analysis tools used were from the Species Group Summaries Results where the species group is Neotropic Migrant, the Period is 1968-2009, and the Region is California. This tool gives a Swainson's Hawk trend index of 3.6 at ($P<0.05$, $N=38$), which translates into an increasing trend of 3.6%

per year. The index value is a measure of percent change per year, and in this case is listed as “significant.” The P value is the likelihood that the result is attributable to chance alone, and in this case the P value is significant. Figure 3 suggests that a low initial value for Swainson’s Hawk detected followed by a slow rate of increase thru the 1990s, followed by a faster rate of increase in 2000’s.

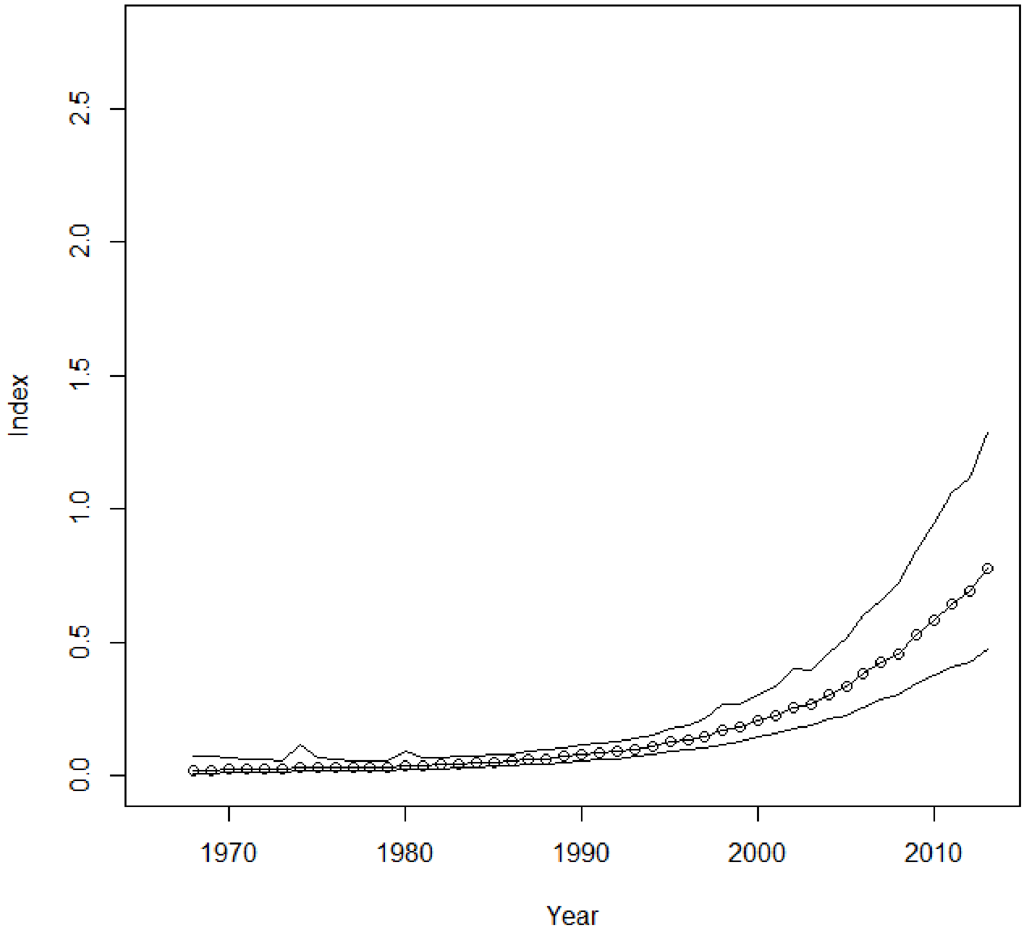

Figure 3. Breeding Bird Survey trend (with 95% confidence intervals shown) for the Swainson’s Hawk from 38 survey routes in California from 1966 to 2013. The x axis is year and the y axis is the relative abundance estimates for all years, estimated as yearly predicted abundances from the hierarchical model analysis (see Sauer and Link 2011).

As mentioned earlier there are only three statewide estimates for breeding pairs of Swainson’s Hawk ranging from 1980 to 2007 (Bloom 1980; Estep 1989; Anderson et al. in prep). The 1979 and 1988 surveys yielded comparable population estimates: 375 (±50) and 550 breeding pairs respectively (Bloom 1980; Estep 1989). The 1988 survey effort was designed to be repeatable and consisted of several years of surveys. The 2005 statewide survey yielded a higher population estimate (1,893 pairs; Anderson et al. in prep.). This more recent effort was a stratified random sample that involved numerous biologists throughout the state; a level of effort substantially greater than previous efforts which undoubtedly influenced its greater population estimate.

Based on the results of the three statewide surveys occurring in California, it is possible to conclude that the population is increasing over time. However, this perception is tempered by the differences in effort, design, technique and time frame of data collection of the three studies. The latest population estimate (Anderson et al. in prep) is still below the historical population estimate, and there is little evidence to indicate that this hawk has reoccupied much of its former range in the central and south coast valley and Southern California. Although the three statewide estimates are not sufficient to form a trend line, cautious speculation that the Swainson’s Hawk population has experienced a modest increase within the Central Valley may be warranted .

X. EXISTING MANAGEMENT EFFORTS

Regulations, Protections, and Conservation

California Endangered Species Act (CESA; Fish and G. Code, § 2050 et seq.). The Swainson’s Hawk was listed as a threatened species in 1983 by the California Fish and Game Commission pursuant to CESA, (Cal. Code Regs., tit. 14, § 670.5(b)(5)(A).)

Under CESA it is unlawful to take (Fish & G. Code, §86) a species listed as “threatened” of “endangered” (or a candidate) by the State of California unless 1) the take is incidental to an otherwise lawful activity, 2) the impacts of the lawful take are fully minimized and mitigated, 3) the take is consistent with Fish and Game Code sections 2112 and 2114, and 4) adequate funding to implement the permitted take’s mitigation and monitoring measures is ensured.

Section 2053 of the Fish and Game Code states, in part, "it is the policy of the state that state agencies should not approve projects as proposed which would jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of habitat essential to the continued existence of those species, if there are reasonable and prudent alternatives available consistent with conserving the species and or its habitat which would prevent jeopardy." Section 2054 states "The Legislature further finds and declares that, in the event specific economic, social, and or other conditions make infeasible such alternatives, individual projects may be approved if appropriate mitigation and enhancement measures are provided."

Loss or alteration of foraging habitat or nest site disturbance which results in:
(1) nest abandonment; (2) loss of young; (3) reduced health and vigor of eggs and/or nestlings (resulting in reduced survival rates), may ultimately result in the take of nestling or fledgling Swainson’s Hawks incidental to otherwise lawful activities. The taking of Swainson’s Hawks in this manner can be a violation of CESA. This interpretation of take has been judicially affirmed by the 1992 landmark appellate

court decision, *Department of Fish and Game v. Anderson-Cottonwood Irrigation District* (8 Cal.App. 4th, 1568), which emphasized that the intent and purpose of CESA applies to all activities that take or kill endangered or threatened species, even when the taking is incidental to otherwise legal activities.

California Environmental Quality Act (CEQA, Pub. Resources Code, § 21000 et seq.). CEQA requires adoption of mandatory findings of significance if a project's impacts to threatened or endangered species are likely to occur (§21001 (c), §21083, Guidelines §15380, §15064, and §15065). Impacts must be avoided or mitigated to less than significant levels unless the CEQA Lead Agency makes and supports findings of Overriding Consideration. Mitigation for impacts to Swainson's Hawk foraging habitat varies among CEQA lead agencies, but essentially does not occur at a rate greater than 1:1 habitat lost to habitat protected.

Fish and Game Code §§ 3503, 3503.5, and 3800. These Fish and Game Code sections prohibit the take, possession, or destruction of birds, their nests or eggs.

Migratory Bird Treaty Act (MBTA). Swainson's Hawks are protected under the federal MBTA of 1918 (16 U.S.C. 703 711). The MBTA makes it unlawful to take, possess, buy, sell, purchase, or barter any migratory bird listed in §50 of the Code of Federal Regulations (C.F.R.) Part 10, including feathers or other parts, nests, eggs or products, except as allowed by implementing regulations (50 C.F.R. 21).

Conservation Plans

Regional conservation planning efforts take a comprehensive approach to ecosystem conservation while allowing land use authorities the ability to manage anticipated growth and development. A few regional conservation plans currently being administered are designed to provide conservation of nesting and foraging Swainson's Hawk habitat within the bird's nesting range, including: the San Joaquin County Multi-species Habitat Conservation and Open Space Plan, the Natomas Basin Habitat Conservation Plan, the Metro Air Park Habitat Conservation Plan, and the East Contra Costa County Habitat Conservation Plan and Natural Community Conservation Plan. Each of these plans has a unique strategy for providing conservation value for the Swainson's Hawk; however none of these provide habitat at a rate greater than 1:1 habitat lost to habitat protected. In addition to the plans described above, there are several jurisdictions with conservation plans in the development stage which aim to provide good conservation value to the Swainson's Hawk, including: Butte County, Yolo County, Solano County, Sacramento County, Yuba and Sutter Counties, and Placer County.

XI. DATA GAPS

The Swainson's Hawk has been listed under the California Endangered Species Act since 1983, and yet there is still much to learn about the species. Several surveys have been conducted throughout the state, but the purposes and methodologies have been independent for each. Some long-term studies have been or are being conducted in Yolo

County and Butte Valley; however, these studies provide information at a regional scale rather than statewide (Estep pers. comm. 2012).

A long-term repeatable statewide breeding/nest survey, possibly using a stratified random sampling survey design, is needed to assess the population's trend, distribution and range, temporal variation, and abundance. Surveys outside of the known range should be included to determine if range expansions are occurring and at what level.

Additional research is needed to inform managers who are responsible for conserving the species. Research topics of need include: assessing survival, recruitment levels, breeding success, characteristics of migration, disease and parasites, and contaminant studies, specifically how contaminants may affect egg shells.

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ORIGINAL RESEARCH REPORT

Space Use by Swainson's Hawk (*Buteo swainsoni*) in the Natomas Basin, California

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We used satellite-based remote sensing to estimate home ranges for Swainson's Hawk, a species listed as threatened in California (USA), on its breeding grounds in the Natomas Basin (northern Central Valley, California) and to evaluate whether the species' space-use intensity (statistically derived density of telemetry locations) was associated with land cover, sex, reproductive success, or life stage of offspring. We differentiated seven classes of land cover—alfalfa, annually rotated irrigated crops, developed, grassland, orchard / vineyard, rice, and water. From 2011–2013, we fitted transmitters with global positioning systems to 23 adult Swainson's Hawks. We recorded a minimum of six locations per day per bird from spring through early autumn of each year. We used a fixed, bivariate-normal kernel estimator to calculate a utilization distribution at 30-m resolution for each life stage of each individual within each year. We used a linear mixed model to estimate the associations between intensity of space use and land cover, sex, and reproductive status. The majority of adult Swainson's Hawks traveled distances up to 8–10 km from the nest throughout the breeding season. Median seasonal home-range sizes in a given year ranged from 87–172 km^2. The association between intensity of space use and grassland was 50–139% stronger, and the association between intensity of space use and alfalfa 23–59% stronger, than the associations between intensity of space use and any other land-cover type. Intensity of space use did not vary as a function of sex, reproductive status, or life stage. Given our results and additional knowledge of the species' ecology, we suggest that reproductive success and, in turn, population-level recruitment may be associated equally if not more closely with availability of nesting sites than with the current distribution of land cover.

Keywords: Central Valley; home range; land cover; land use; nesting success; utilization distribution

Introduction

Spatially extensive changes in land use and land cover affect the survival and reproduction of numerous species. Human activities reduce the probability of persistence of many native species, but generalists may adapt to or benefit from some forms of agricultural and suburban or exurban development [1, 2, 3]. Relations between population dynamics and different types of development vary geographically and among species. Identifying such relations can be highly relevant to regional land-use planning and regulatory compliance, especially for species with legal protection.

Swainson's Hawk (*Buteo swainsoni*) is among the species most often invoked in assessments of the potential biological effects of contemporary land-use change in California's Central Valley. For example, conservation of Swainson's Hawk was emphasized in the Natomas Basin Habitat Conservation Plan [4] and is addressed in planning documents or administrative drafts for the Yuba–Sutter and Yolo County Habitat Conservation Plans [5, 6].

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Historical losses of the species' habitat in the state and decreases in its estimated abundance led to its listing as threatened under the California Endangered Species Act in 1983. Swainson's Hawks nest in riparian woodlands, oak woodlands, stands of trees along roads or edges of agricultural fields, and isolated trees [7]. They breed from southern Alberta, Saskatchewan, and Manitoba, Canada south through the Intermountain West and Great Plains of the United States and into northern Mexico. Isolated populations breed in interior valleys of British Columbia, Canada, California's Central Valley, and some valleys in the western Great Basin. The species winters from Mexico south through South America and occasionally in the southern United States [7, 8]. Some individuals also winter in the Sacramento–San Joaquin Delta in California [9].

The largest and densest populations of Swainson's Hawks in California are believed to occur in the Central Valley counties of Yolo, Solano, Sacramento, and San Joaquin [10, 11]. In Butte Valley (Siskyou County, California), on the western edge of the Great Basin, annual apparent survival of adults varied from 0.85 to 0.9 over 30 years [12]. At least locally, food availability may be associated with temporal variation in brood size and nestling survival [13].

Throughout their range, Swainson's Hawks forage in native and non-native grasslands and in relatively open shrublands and woodlands [7]. In the Central Valley and other agricultural areas, Swainson's Hawks also forage in irrigated fields or pastures and in fields in which alfalfa, other hay crops, and some row crops and grains are grown [7, 14]. During the breeding season, Swainson's Hawks eat rodents, rabbits, birds, insects, and reptiles. Small mammals generally comprise the majority of the biomass consumed by adults during the breeding season and delivered to nestlings, although prey composition is spatially and temporally variable [15, 16]. During the non-breeding season, the species continues to feed opportunistically. When present, insects (especially grasshoppers) are a primary food source [7].

The extent to which Swainson's Hawks use different land-cover types and crops, and the extent to which use is associated with the distribution of the species or its reproductive success, varies considerably. For example, in northern Colorado, abundance of Swainson's Hawks during the breeding season was positively correlated with cover of tallgrass prairies and hay fields and not correlated with cover of pavement, buildings, and urban vegetation or with distance to prairie dog towns (a potential source of prey) [17]. The average distance between the nest site and agriculture (primarily irrigated alfalfa), and the average area of agriculture within a 500-m radius of the nest, were negatively related to adult survival in Butte Valley, California [12]. In southeastern Arizona, the density of Swainson's Hawks was higher in agricultural areas than in grasslands (similar to density patterns in the Central Valley) and desert scrub, but breeding success did not differ among those land-cover types [18].

From 1973–2000, agriculture consistently covered about 72% of the Central Valley [19]. Although the total area of agriculture has changed little, row crops, hay, and grains have been replaced locally by orchards and vineyards, and some regional droughts resulted in widespread fallowing. Additionally, increases in California's human population are leading to gradual increases in the developed area of the Central Valley. Swainson's Hawks nesting in the Natomas Basin (Sacramento and Sutter Counties, California; roughly at the lower end of the northern third of the Central Valley; **Figure 1**) have been monitored annually by J.E. since 2001 to comply with the Natomas Basin Habitat Conservation Plan, which was adopted in 1997 and revised in 2003 [4]. Habitat Conservation Plans are required prior to issuance of a federal permit to non-federal parties for incidental take of species that are listed under the U.S. Endangered Species Act or that may become listed during the permit period. Forty-three to 65 nesting territories were reported as active each year, and the number of pairs nesting in the Natomas Basin from 2001–2013 was stable, if not gradually increasing [20]. The mean number of young fledged per active nesting territory per year during this period was 1.00 (SD 0.33, range 0.26–1.55) (J.E. unpublished data).

We capitalized on the increasing feasibility of satellite-based remote sensing to objectively track space use (the locations where animals are present) of Swainson's Hawks. Past inferences about habitat use or environmental attributes associated with presence of Swainson's Hawks during the breeding season or the winter [9] largely were based on visual surveys [9, 14, 21], point counts [17], or the locations of nests [22]. Use of radio telemetry to assess space or habitat use by Swainson's Hawks has been relatively uncommon but see [23, 24, 25] and, to the best of our knowledge, global positioning systems (GPS) have not been used previously. Space use is most informative when it can be related to a measure of survival or reproduction. With the goal of informing ongoing and future land-use planning in the Natomas Basin, we estimated home ranges for Swainson's Hawks that are present in or adjacent to the basin during the breeding season and evaluated whether space-use intensity (statistically derived density of telemetry locations) was associated with land cover, sex, reproductive success, or life stage of offspring.

Materials and Methods

We differentiated seven classes of land cover—alfalfa, annually rotated irrigated crops, developed, grassland, orchard / vineyard, rice, and water—within a 2000-km^2 area that encompassed the Natomas Basin and a 16-km buffer around the perimeter of the basin (Supporting Information). We selected these land-cover classes on the basis of the assumption that they were the most likely to be associated with space use by Swainson's Hawks (J.E. and R.L.A. unpublished data). Alfalfa was dominated by alfalfa and clover. Annually rotated irrigated crops primarily included row crops, field crops, and grains. Developed included all intensities of development (including roads), open space in developed areas (e.g., parks and golf courses), and barren areas. Grassland included non-alfalfa hay crops, grass-covered areas used as pasture, fallowed

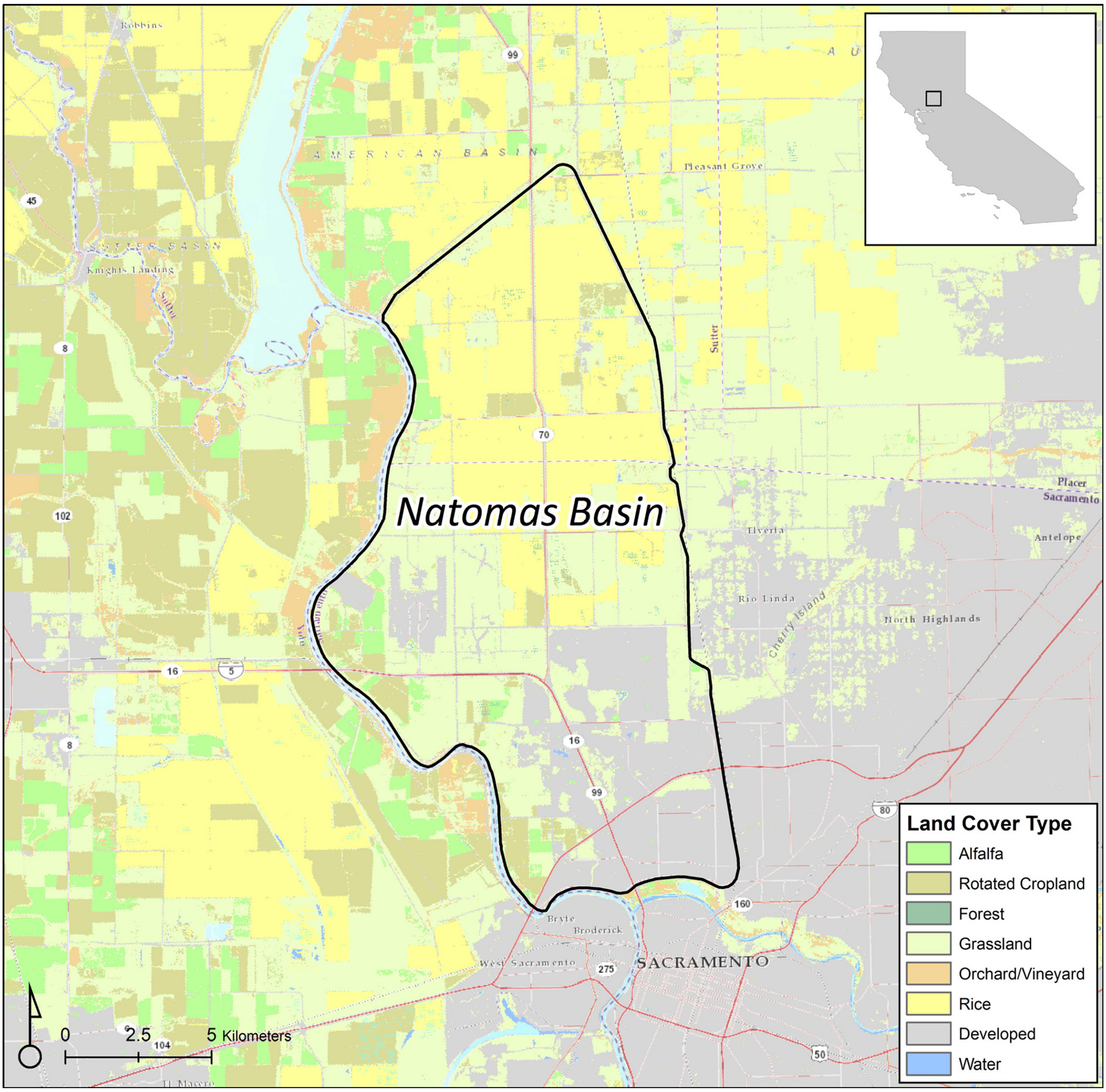

Figure 1: The Natomas Basin (Sacramento and Sutter Counties, California) and its location within California (inset).

cropland, non-native annual grasses, and a small proportion of native annual and perennial grasses. About 5% of orchard / vineyard was non-agricultural trees and shrubs; about 90% was orchard. Detailed definitions of each class are in Supporting Information. We used the National Agricultural Statistics Service's Cropland Data Layer (http://nassgeodata.gmu.edu/CropScape/) for 2012 to determine the locations and extents of each land-cover class. Alfalfa covered 9% of the analysis area, annually rotated irrigated crops 21%, grassland 26%, orchard / vineyard 11%, rice 14%, developed 18%, and water 1%. Field evaluation (described fully in Supporting Information) indicated that classification accuracy was ≥84% and that within these classes, land cover changed little from 2011 through 2013.

We selected Swainson's Hawks to be fitted with satellite transmitters from pairs on nesting territories in the Natomas Basin or within a 3-km buffer around the Natomas Basin. In this manuscript, we define a nesting territory as the area around the nest that is defended by an adult; this is a narrower definition than that of Steenhof and Newton [26]. We searched for nesting Swainson's Hawks by driving all accessible roads within the Natomas Basin, including both sides of the peripheral watercourses (Sacramento River, Natomas Cross Canal, and Steelhead Creek). Where no roads provided access to trees large enough to be used by Swainson's Hawks for nesting [27, 28], we conducted surveys on foot. We searched for Swainson's Hawks in potential nest trees with binoculars or a spotting scope.

We trapped Swainson's Hawks with the dho-gaza method [29, 30, 31]. We tethered a potential predator, a live Great Horned Owl (*Bubo virginianus*), behind a 2 m × 6 m, four-shelf mist net (210 denier / 2 ply, 100 mm mesh)

(Avinet, Inc., Freeville, New York). To elicit defensive behavior by the adult Swainson's Hawks, we arranged the trap near active nests with young. We targeted nests that were accessible by foot and that had enough open, unobstructed area nearby that a Swainson's Hawk could dive safely on the owl. The locations in which we arranged the trap also allowed us to protect the owl from contact with the hawk and had sufficient shade in which to process the captured hawk. We trapped adults at nests in which the young were 10 days of age or older to minimize the probability of adults abandoning the nest or of juveniles being unable to thermoregulate on their own. We made trapping attempts between 0500 and 1300 on a given day. Given the limited number of nests, we did not stratify nests by environmental covariates or attempt to trap a certain proportion of either sex.

From July 2011 through April 2013, we fitted transmitters with GPS (22-gram solar-powered Argos / GPS PTT 100, Microwave Telemetry, Columbia, Maryland) to 23 adult Swainson's Hawks. In 2011, we captured 12 Swainson's Hawks. In 2012, we fitted transmitters to nine Swainson's Hawks. In April 2013, we fitted transmitters to two Swainson's Hawks before nesting began. We used Teflon ribbon to attach the transmitters as backpack mounts. Processing time for each Swainson's Hawk was one to two hours and included taking measurements, fitting the backpack-transmitter to the hawk, and placing a band (Bird Banding Laboratory, Pautuxent, Maryland) on its leg. The weight of a transmitter was equivalent to 3% or less of the average body weight of an adult Swainson's Hawk [32]. All transmitters had motion-based mortality sensors.

Six of the transmitters were programmed to record six locations per day at fixed times (one every other hour from 0900 to 1900, the period during which Swainson's Hawks typically are most active) from 16 March through 10 October. Coordinates were transmitted via satellite. These six transmitters also included very high frequency (VHF) transmitters with signals that were broadcast continuously from 0900 through 2100 each day from 16 March through 15 September (data not reported or analyzed here). The 15 remaining transmitters, two of which were fitted to different birds during different years, were programmed to record 12 locations per day at fixed times (one per hour from 0900 to 2000) from 16 March through 10 October. Coordinates were transmitted via satellite. All transmitters were programmed to record two locations per day from 11 October through 15 March; data on overwintering locations will be reported elsewhere. The manufacturer estimated that the GPS sensors had a horizontal accuracy of ± 18 m. Except in the situations noted below, the analyses reported here were based on the six or 12 locations per day.

We visually monitored all nesting territories from which Swainson's Hawks were captured until the hawks migrated. Visits were spaced about a week apart, although neither the number of days on which surveys were conducted nor the interval between surveys was standardized. During each visit, we recorded whether the territory was occupied by a pair of Swainson's Hawks (regardless of nesting success), whether the nest was active (eggs or young were present), and, where relevant, the number of young fledged [23]. We defined a successful nest as one in which at least one young fledged [26] and a failed nest as one in which no young fledged.

We used a fixed, bivariate-normal kernel estimator to calculate a utilization distribution (UD [33]) at 30-m resolution for each life stage (see below) of each individual within each year. The bivariate normal is a statistical distribution of values of pairs of related, normally distributed variables (e.g., in two-dimensional space, an x-coordinate and a y-coordinate [34]). A kernel is a point or polygon around which one estimates the density of other points, lines, or polygons, including natural or human-created environmental features. This method places a circular kernel over each cell in a regular lattice, and produces a weighted count, or density estimate, of the telemetry locations that fall within the kernel; the weights decrease as the distance from the center of the kernel increases [34]. The bandwidth selector, *h*, defines the radius of the kernel and therefore the level of overall smoothing. A UD permits one to make a probabilistic estimate of the intensity of space use for each individual at a given location [35]. Thus, the intensity of space use represents the density of telemetry locations as derived from the UDs. We estimated UDs with the *cvh* bandwidth selector, which limited both identification of areas of high space-use intensity where Swainson's Hawks were not recorded (i.e., oversmoothing) [36] and fragmentation of the home range [37]. The cvh also helps minimize potential lack of independence among locations or biases in acquisition of locations [36]. We defined the home range represented by each UD as the area within its 99% isocline (the area bounded by 99% of its volume).

We excluded some data for a subset of Swainson's Hawks in one or more years because the animal died or its transmitter temporarily or permanently stopped functioning. We also excluded points from individuals for which life stages were not defined, either because the bird did not nest or because nest-observation data were insufficient. We further excluded data when the number of locations recorded for a given individual for a given life stage was fewer than 30.

Where data were sufficient or applicable, we calculated UDs for each of four life stages within each year, two of which are associated with development of offspring: arrival – day of arrival in the Natomas Basin through day before first observation at the nest, prehatching – day of first observation at nest through day before first observation of hatched young, nestling – day of first observation of hatched young through day before first observation of fledging, and premigration – day of first observation of fledging through day before seasonal departure from the Natomas Basin. We determined seasonal departure on the basis of GPS locations and directional movements away from the Natomas Basin.

We used a linear mixed model to estimate the associations between intensity of space use and land cover, sex, and reproductive status for each year. We restricted our models to points that fell within the 99% isoclines of the corresponding UDs. Our data indicated that late in the breeding season, Swainson's Hawks sometimes fly distances that are quite long relative to typical daily movements earlier in the breeding season. The 99% isocline excluded many of the points corresponding to these long-distance movements. We used orchard / vineyard, the only land-cover class over which all Swainson's Hawks in our sample were recorded in all three years, as the reference land-cover class. Female was the reference class for sex. We differentiated three classes of reproductive status: nested and at least one young fledged, nested but no young fledged, and did not nest, with did not nest as the reference class.

We included a random intercept in our models, with levels defined by individual, life stage, and year, to account for variation in space-use intensity among individuals and over time. Additionally, to account for the fact that many points were near nests, we manually defined for each individual, within each year and life stage, a distance from the nest (i.e., a radius) beyond which there was little evidence of concentrated use of space. We recorded whether each telemetry location was within or beyond this radius (referenced as the *nest zone threshold*), and included the binary (within or beyond) variable as a random-effect level. To account for potential reduction in space-use intensity as distance from the nest increased, our final model also included a random slope effect, with the same two levels of the nest zone threshold, for distance from the nest site.

We used spatial covariance functions to explore the modeling of spatial autocorrelation among model residuals [38]. We did not include a covariance model in the final model due to overestimation of function parameters, which apparently were related to concentration of use around the nest. Accordingly, we used the Huber-White sandwich estimator [39] to calculate standard errors in the presence of any remaining subject-level heterogeneity. We used Akaike's Information Criterion (AIC) to assess whether the support for our global model was substantially different (e.g., a difference ≥ 4 AIC) than that for a null model. We used a paired *t*-test ($\alpha = 0.05$) to test, for each variable, whether there was a statistically significant difference in the strength of associations between occupancy and a given class (of land cover, sex, reproductive status, life stage, or year) versus the reference class of that variable. We derived all estimates with SAS PROC MIXED (v9.3; SAS Institute, Inc., Cary, North Carolina).

Results

In 2011, six of the Swainson's Hawks that we tracked fledged young and four nested but did not fledge young. In 2012, 14 of the Swainson's Hawks fledged young, two nested but did not fledge young, and one did not nest. In 2013, four of the Swainson's Hawks fledged young, nine nested but did not fledge young, and one did not nest.

We included in our analyses a total of 3,374 locations from six Swainson's Hawks (five males) in 2011, 10,917 locations from 15 Swainson's Hawks (12 males) in 2012, and 4,880 locations from 4 Swainson's Hawks (all male) in 2013 (**Table 1**). The analyses included data for 3 males that were tracked in all three years, 2 males tracked in two of three years, and 8 males and 4 females tracked in one year.

In all years, the vast majority of adult Swainson's Hawks traveled distances up to 8–10 km from the nest throughout the breeding season. Nest zone thresholds (**Figure 2**) for individuals that fledged young generally ranged from about 1000 to 4000 m, and did not appear appreciably different during different life stages (**Table 1**). The upper bound of nest zone thresholds for individuals that did not produce offspring or whose offspring did not fledge was greater, to about 7500 m. Data were insufficient to analyze statistically whether nest zone thresholds varied as a function of year, sex, life stage, and reproductive success.

Home-range sizes varied among individuals, years, and life stages (**Table 1**). Median seasonal home-range sizes, calculated on the basis of points within the 99% isocline, were 109 km^2 (range 10–336) in 2011, 172 km^2 (range 43–1427) in 2012, and 87 km^2 (range 77–341) in 2013. Home ranges generally were smallest when adult Swainson's Hawks were caring for young. Median home-range sizes during the arrival stage were 25 km^2 (range 2–12860, or 2–47 excluding one individual's anomalously large home range) in 2012 and 323 km^2 (one individual) in 2013 (**Table 1**). During the pre-hatching stage, median home-range sizes were 33 km^2 (range 4–202) in 2012 and 110 km^2 (range 41–226) in 2013. Median home-range sizes during the nestling stage were 80 km^2 (range 4–137), 69 km^2 (range 6–350), and 30 km^2 (range 5–174) in 2011, 2012, and 2013, respectively. Home ranges of premigratory Swainson's Hawks were 95 km^2 (range 8–357) in 2011, 201 km^2 (range 6–604) in 2012, and 88 km^2 (range 4–400) in 2013, respectively.

The global model (presented here) had an AIC value 411.3 lower than the null model. Across the three years, intensity of space use was more strongly associated with five of the land-cover classes—grassland, alfalfa, developed, water, and rice—than with the reference class, orchard / vineyard (**Table 2**, **Figure 3**). The association between intensity of space use and grassland was 50–139% stronger than the positive associations between intensity of space use and any other land-cover type. The association between intensity of space use and alfalfa was 23–59% stronger than the association between space use and water, developed, or rice (**Table 2**). Strengths of association between intensity of space use and either water or developed were relatively similar. Intensity of space use did not vary as a function of sex, reproductive status, or life stage.

Discussion

Although there was considerable annual variation in reproductive success of the Swainson's Hawks we tracked, intensity of space use did not vary as a function of reproductive status. Our nest zone thresholds generally were consistent with previous estimates from elsewhere in California. For example, in Butte Valley from 1984 through 1994, nesting Swainson's Hawks moved 2.2 ± 0.23 km (mean ± SE), with a range from 0.97–6.3 km [40]. In the

Identification number	Sex	Year	Number of young fledged	Life stage	Nest zone threshold (m)	Area of home range (km²), 99% isocline	Area of home range (km²), 95% isocline
105921	male	2011	1	annual	1750	250	130
				nestling	1500	40	20
				premigration	1500	291	155
105922	male	2011	1	annual	1750	78	35
				premigration	1750	78	35
105922	male	2012	2	annual	1000	156	71
				arrival	1000	47	17
				prehatching	1000	133	60
				nestling	1000	138	69
				premigration	1000	163	94
105922	male	2013	1	annual	1200	341	208
				prehatching	1250	226	133
				nestling	1250	174	109
				premigration	1000	400	245
105923	male	2012	1	annual	2500	204	69
				prehatching	2000	202	70
				nestling	2000	63	30
105925	female	2012	1	annual	1500	43	20
				premigration	1500	43	20
105926	male	2012	2	annual	2000	249	122
				premigration	2000	249	122
105927	male	2012	2	annual	4000	1428	498
				arrival	4000	12,860	7842
				prehatching	3000	11	3
				nestling	3500	69	31
				premigration	3000	267	153
105928	male	2011	2	annual	2500	57	20
				premigration	2500	57	20
105928	male	2012	2	annual	2000	58	10
				arrival	1500	2	1
				prehatching	1750	44	17
				nestling	1750	6	2
				premigration	1750	37	15
105928	male	2013	1	annual	4000	89	15
				prehatching	3000	142	33
				nestling	4000	5	2
				premigration	4000	88	30
105930	male	2011	2	annual	1000	10	2
				nestling	1000	4	2
				premigration	1000	8	2
105930	male	2012	2	annual	1500	172	47
				prehatching	1100	4	2
				nestling	1500	32	10

Identification number	Sex	Year	Number of young fledged	Life stage	Nest zone threshold (m)	Area of home range (km^2), 99% isocline	Area of home range (km^2), 95% isocline
				premigration	1500	491	237
105930	male	2013	1	annual	1100	84	21
				arrival	900	323	171
				prehatching	1200	41	10
				nestling	1300	30	10
				premigration	1250	4	2
105931	female	2012	0	annual	1700	81	38
				premigration	1700	81	38
105933B	male	2012	1	annual	2000	69	26
				premigration	2100	70	26
105935	female	2011	1	annual	2200	336	177
				nestling	1500	137	69
				premigration	2000	357	196
105936	male	2011	1	annual	2200	141	63
				nestling	1500	121	71
				premigration	2100	112	56
105936	male	2012	1	annual	2000	65	15
				arrival	2000	3	2
				prehatching	2000	21	8
				nestling	1750	15	7
				premigration	1750	105	39
117526	male	2012	1	annual	2000	228	123
				nestling	2000	350	211
				premigration	2500	240	127
117527	male	2012	1	annual	6000	571	288
				nestling	7500	209	134
				premigration	4000	604	305
117527	male	2013	1	annual		78	46
117528	male	2012	2	annual	2000	88	27
				nestling	2000	200	115
				premigration	2000	6	3
117529	female	2012	1	annual	2500	313	162
				premigration	2500	313	162
117530	male	2012	2	annual	2200	610	338
				premigration	2200	399	231

Table 1: Swainson's Hawks tracked with satellite telemetry in 2011, 2012, and 2013, associated demographic attributes, and nest zone thresholds for individuals that nested or reproduced. Data were insufficient to calculate nest zone thresholds for all life stages for all individuals. There was no indication of concentrated use of space around the nest for individual 117527 in 2013. Arrival – day of arrival in the Natomas Basin through day before first observation at the nest, prehatching – day of first observation at nest through day before first observation of hatched young, nestling – day of first observation of hatched young through day before first observation of fledging, premigration – day of first observation of fledging through day before seasonal departure from the Natomas Basin. Nest zone thresholds are fairly discrete clusters of points beyond which there was little to no evidence of concentrated use of space. Area of home range defined on the basis of the 99% or 95% isocline.

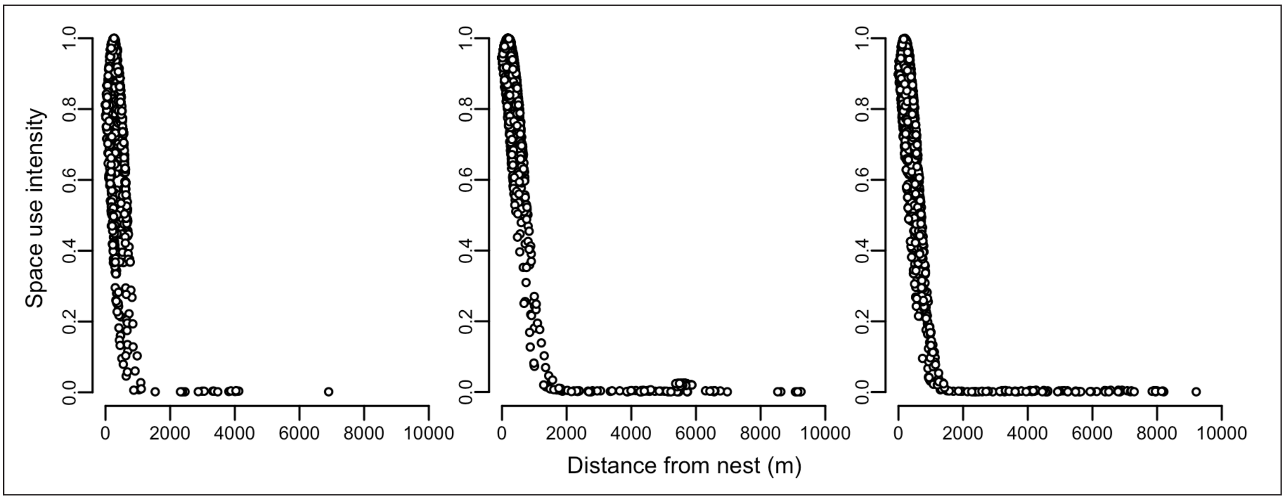

Figure 2: Space-use intensity as a function of distance from the nest for an illustrative male Swainson's Hawk (identification number 105930) throughout the 2011, 2012, and 2013 breeding seasons. Data included 1060 locations for 2011, 1651 for 2012, and 1604 for 2013. We defined the nest-zone thresholds as 1000 m for 2011, 1500 m for 2012, and 1100 m for 2013.

Variable	Estimate	SE	*t*-value	*p*-value
Land cover				
Alfalfa	0.63	0.20	3.21	0.00
Annually rotated irrigated crops	0.37	0.19	1.98	0.05
Grassland	0.81	0.17	4.82	0.00
Rice	0.42	0.21	2.02	0.04
Developed	0.44	0.19	2.26	0.02
Water	0.26	0.10	2.62	0.01
Sex				
Male	–0.07	1.78	–0.04	0.97
Reproduction	0.00			
Nested, without young	0.31	4.01	0.08	0.94
Life Stage				
Arrival	1.95	1.47	1.33	0.19
Offspring are pre-hatching	0.08	1.39	0.05	0.96
Offspring are nestlings	0.10	1.12	0.09	0.93
Year				
2011	–0.12	1.57	–0.08	0.94
2012	0.06	1.17	0.05	0.96

Table 2: Space use intensity for Swainson's Hawks that were tracked from 2011 through 2013. Estimates are the values of regression coefficients for each effect. SE, standard error.

Sacramento Valley, the longest recorded distance from the nest that a Swainson's Hawk foraged was 22.5 km [25]. The distance traveled from the nest may be negatively correlated with probability of nestling survival [12]. Our observations of Swainson's Hawks within their nesting territories suggested that among individuals, there was substantial variation in the mean distance from the nest that was associated with different behaviors (e.g., flying, circling, feeding) (R.L.A. unpublished data). These differences among individuals appeared to be considerably greater than any differences in behavior as a function of distances from nests.

Our estimates of home-range size during the breeding season were as much as two times larger than previous estimates in the same general geographic area that were based on radio telemetry and minimum convex polygons [23, 25].

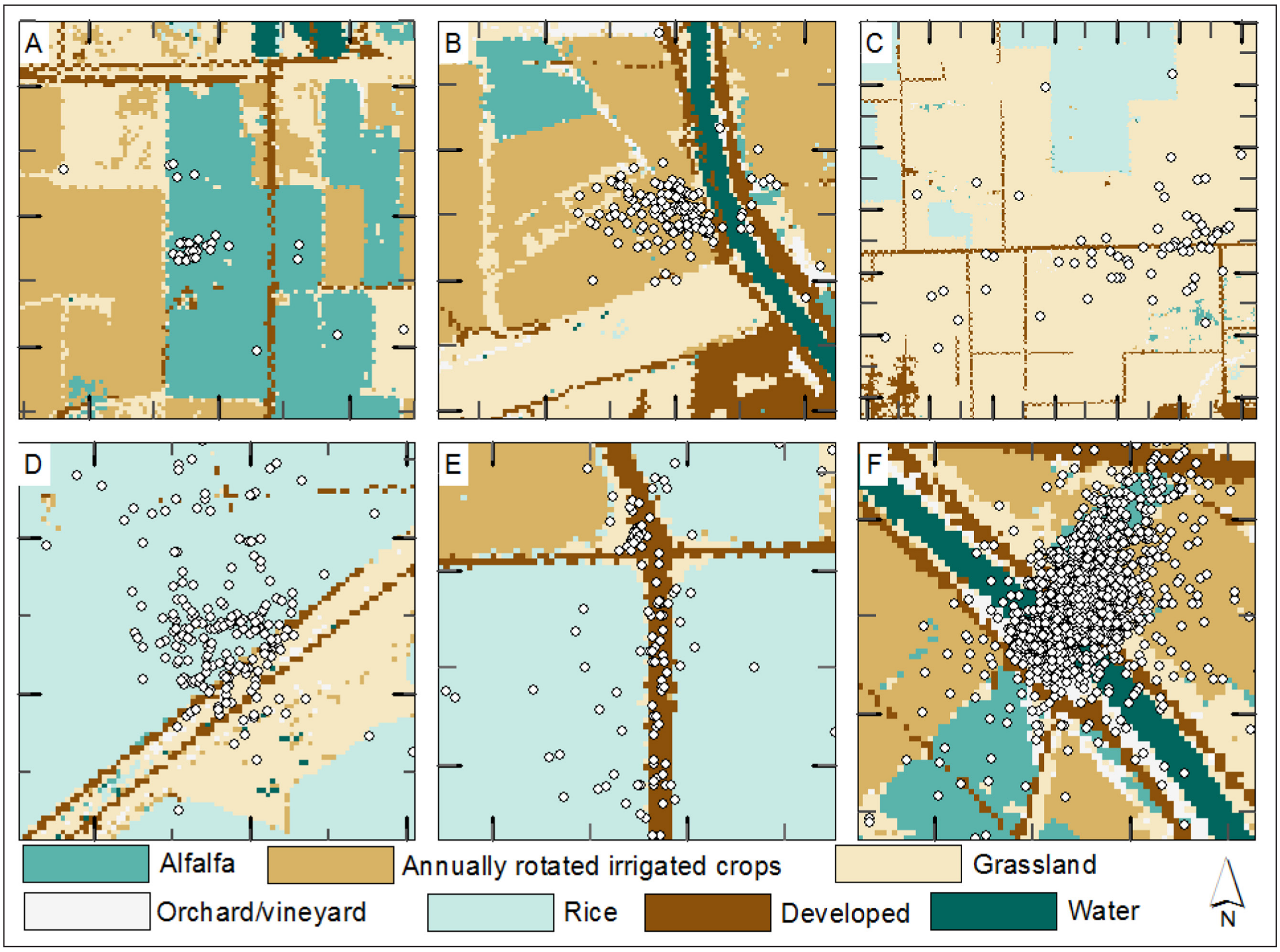

Figure 3: Examples of high relative intensity of space use by Swainson's Hawks over alfalfa (a), annually rotated irrigated crops (b), grassland (c), rice (d), developed areas (e), and water (f). Each small, white dot is one location record. Each panel includes data from a different Swainson's Hawk during one or more life stages. Large and small ticks on the axes indicate distances of 1000 m and 500 m, respectively.

Estep [23] recorded a median home-range size of 20 km^2 for 12 Swainson's Hawks (range 3–87), and Babcock estimated a median home-range size of 39 km^2 (range 7–77) for 4 Swainson's Hawks. There are at least three reasons why our estimates may have been larger. First, we recorded movements over a 10-hour period each day regardless of the birds' locations, whereas Estep [23] recorded movements of birds that could be observed visually over a 4-hour period and Babcock [25] recorded movements of birds that could be observed visually on ≥ 2 days per week. Estimates that are strictly visual may underestimate home-range size, especially during life stages in which movement distances are relatively long and birds cannot readily be followed by the human eye. Second, home ranges estimated with minimum convex polygons are not directly comparable to those estimated with UDs [41]. Third, Babcock [25] based home-range estimates on the 95% contour rather than the 99% contour. Use of the 99% contour or isocline allowed us to maximize our sample size for inferences about land-cover associations while excluding late-season movements that were quite long relative to typical daily movements earlier in the breeding season. For consistency, we used the same isocline to estimate home-range size. However, median seasonal home-range sizes that were based on the 95% isocline were considerably smaller than those based on the 99% isocline: 49 km^2 (range 2–177) in 2011, 69 km^2 (range 10–498) in 2012, and 34 km^2 (range 15–208) in 2013 (**Table 1**).

Although our remotely sensed location data did not provide information on an animal's behavior at a particular location, we believe that our use of satellite telemetry data and our analytical focus on space-use intensity offer a more objective basis for assessing land-cover associations of Swainson's Hawks than do use-availability assessments [24]. Use-availability assessments compare attributes of locations at which one or more individuals were present to a random sample of locations from the area assumed to be available to those individuals or to the estimated proportion of different land-cover types. The results and inferences from use-availability models can be biased by errors in definition of the size and configuration of the available area [42]. Integration of a use-availability assessment with a UD-based model of space-use intensity would require us to identify nest-site cutoffs on the basis of the data and then to exclude points near the nests from analyses of UDs. It also would be necessary to assign greater weight to points far from the nest than to points close to the nest, and potentially to make assumptions about resource use on the basis of distances from the nest or land-cover types. The biological rationale for selecting a given weighting algorithm is not apparent, and we do not have a strong biological basis for assuming that use of space within the nest zone is less meaningful than use of space further away. We did not omit data near the nest because Swainson's Hawks frequently use that space and because our telemetry data did not allow us to make assumptions about differences in behavior at different distances from the nest. In other words, space use may affect fitness and population viability regardless of proximity to the nest.

Our results are consistent with others' suggestions that Swainson's Hawks are generalists that have adapted to agriculture and development [7]. The manner in which we aggregated finely resolved land-use and land-cover types into a smaller number of land-cover classes (Supporting Information) may explain many of the positive associations we identified between a given land-cover class and space use. For example, as noted above, the most intensively used land-cover class, grasslands, included land uses from fallowing of crops to livestock grazing. Similarly, the developed land-cover class included roads in agricultural areas, along which Swainson's Hawks often perch on power lines or telephone poles (e.g., **Figure 3**). Additionally, Swainson's Hawks may attempt to nest in relatively old suburban neighborhoods in which planted or retained trees are relatively tall and foraging habitat is within a few km [28]. The resolution of our data does not allow us to assess whether intensity of development is associated with intensity of space use. The positive association between intensity of space use and water likely reflects that the nests of most of the birds we tracked were near watercourses. Because rice is grown near many of the nests of the Swainson's Hawks we tracked, some locations over rice fields probably were inevitable.

Data on the locations at which Swainson's Hawks are obtaining food has potential to provide information about relations among space use, resource use, and reproduction. Raw measures of space use could be correlated with food captured. However, there are two reasons why raw measures of space use may not reflect habitat with the greatest food availability or the greatest proportion of food captures. First, birds are not always foraging. There are few reliable data on activity budgets of Swainson's Hawks, especially during the nesting period but see [43]. Second, birds may not spend most of their time in the highest-quality habitat. For example, it is possible that in high-quality habitat, birds catch prey quickly and then leave to take the prey back to the nest, whereas in low-quality habitat, it takes more time to find food. It also is possible that a successful forager can obtain its food requirements for the day relatively quickly and then spend time soaring or perching. If any of these scenarios apply to Swainson's Hawks, raw measures of space use could produce misleading inferences.

Because our data do not allow us to draw inference to behavior, it is unclear whether space-use associations reflect use of resources within different land-cover types. For example, we cannot determine whether a Swainson's Hawk that was recorded over grassland was foraging, feeding, or engaged in a different behavior. The assumption that Swainson's Hawks spend the most time in locations with the greatest food availability could be tested by identifying an association between telemetry data and a direct measure of the prey base or food capture. It remains unknown whether food availability is a limiting resource in the Natomas Basin or elsewhere in the species' range. Bechard [43] suggested that vegetation type was more strongly associated than prey abundance with foraging locations of Swainson's Hawks in southeastern Washington.

Visual observations of Swainson's Hawks in alfalfa fields and other hay crops and observed attempts to capture prey [24] have led to inferences that most foraging occurs in those land-cover types, especially when harvest or mowing exposes rodents [9, 17, 24]. For example, investigators conducted road surveys in the Central Valley in which Swainson's Hawks were considered to be foraging if they were observed coursing, circling, kiting, or hovering ≤ 100 m above a field [14]. These observations were compared to the proportion of different land-cover types within the area surveyed, with the inference that Swainson's Hawks foraged in vineyards less than expected, and in dryland grain or irrigated alfalfa, ryegrass, or clover more than expected, although apparent use of different crops varied throughout the breeding season [14]. Swolgaard et al. [14] also suggested that plant height, density, and cover may affect the location or intensity of foraging by Swainson's Hawks.

Because Swainson's Hawks are listed as threatened in California, they will continue to be a focus of regional conservation plans regardless of their biological status. Given that Swainson's Hawks are highly territorial, we suggest that mating and reproductive success and, in turn, population-level recruitment may be associated equally if not more closely with availability of nesting sites [13] than with the current distribution of land-cover types.

Competing Interests

The authors declare that they have no competing interests.

Acknowledgments

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Supporting Information

Details of the Natomas Basin land-cover mapping and accuracy assessment is available at http://dx.doi.org/10.1525/collabra.35.smo

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THE NESTING OF THE WHITE-TAILED KITE IN SOUTHERN SANTA CRUZ COUNTY, CALIFORNIA

WITH TWO ILLUSTRATIONS

By ALBERT C. HAWBECKER

The White-tailed Kite (*Elanus leucurus majusculus*) is so generally conceded to be becoming increasingly rare and hard to find, that the discovery of four nesting pairs in the vicinity of Watsonville, Santa Cruz County, California, last spring was a welcome surprise. Casual observations, made to acquaint myself with the habits of the bird, revealed traits either not treated by other writers or only intimated by them. Consequently, a more intensive study was made of their behavior during the nesting season.

The pair that was most accessible and most intensively studied nested on a farm near the settlement of Corralitos which is about seven miles north of Watsonville. These birds were first seen by I. B. Andersen, of Watsonville, on April 18, 1939, and reported to me on that date. The nesting site was a dense five-acre patch of coast redwood, coast live oak, madrone, hazel, and poison oak located on a steep slope, bounded on the west by an apricot orchard and on the other three sides by pasture or grassland. An occupied farmhouse stood within approximately three hundred yards of the nesting trees and stock, dogs, chickens, and children roamed at will over the entire area. This habitat did not resemble any of the other three, nor did it conform closely to the general type as described by Pickwell (Condor, vol. 32, 1930, p. 221), as water, marshland and willows were all absent. The nesting trees used were situated near the top of the slope and both were live oaks, trees commonly used according to Pickwell (*op. cit.*). The grassland partly surrounding the woodland had a high population of meadow mice at this time.

The tenant of the farm upon which the birds nested stated that the birds had appeared about midwinter, coming in after a storm and taking up their abode in the woodland, and spending some time feeding over the adjacent grassland. During the whole period of observation the birds were not seen by me more than one-half mile away from the nesting site, although a farmer reported that once he saw them over a mile away. Within this territory, if indeed it may be called that, this pair lived a rather solitary existence, as far as other kites were concerned. Pickwell (*op. cit.*) found them nesting near others of their own kind, but this pair was the only one for several miles around, possibly due to the limited food supply, there being a less extensive hunting ground here than that occupied by the more communal pairs. Here also, no raptors were tolerated save Turkey Vultures, with Red-tailed and Cooper hawks the main objects of attack. All other birds were tolerated.

The first nest, a typical one, was found April 19, 1939, on the farm near Corralitos, in the top of a thirty-foot coast live oak, effectively concealed from below but characteristically in the open above. Halfway up the nesting tree there was another nest identical in construction with the one in use, but apparently it had never been used since it resembled closely a nest containing eggs rather than one that had been used by young. Perhaps this nest was one of the "dummy" or cock nests suggested by Pickwell. Both nests were well built.

The tree was climbed and four nestlings were found. There was one large bird and one small one with two of intermediate size; continued observation suggested that there was approximately one day between the ages of the birds. The nest, at first rather clean, soon became floored with about an inch of meadow mouse hair. This was apparently not pellet material, as the pellets were found below and around the nest, but hair pulled from the body of the mouse while it was being eaten. The act of excretion was performed over (not shot over) the edge of the nest, and the whole area surrounding and below the nest was liberally limed.

The young of this brood were banded on April 20, 1939 (band numbers 34-643001, -002, -004, and -005). This was a rather perilous and unpleasant job. Since the nest was in the very top of a live oak, it was necessary to climb into branches that were not over one inch in diameter. Luckily there were several of them, but still the added weight and the wind made them sway in an alarming manner. In addition to this, the adults took turns diving at the intruder's head which added to his discomfiture. The young showed their resentment of the intrusion by making stabs at the bander's hands with their feet, much as young barn owls do. The mouth was opened wide, but no attempt was made to use the beak in defense. Finally, when each young was lifted from the nest above the writer's head, it defecated copiously, one bird even added insult to injury by defecating twice. The writer succeeded in reaching the ground, safely but in a very ruffled and bespattered condition. The smallest youngster, band 34-643002, was later found dead beneath the nest.

The young birds first left the nest approximately May 15. They were fairly good at flying at that time, although a little awkward. The main difficulty the youngsters had was in landing. Rather than settling gracefully on a slender branch as the old birds did, they would make a stab at the tree in general and end up with outspread wings supporting them in the tree. In aviation parlance they "cracked up," but they were unharmed and gradually learned the art of flying and alighting so that they were soon as graceful and proficient as the parents.

After the first leaving of the nest, the young were still cared for at the nest by the parents. Several times a youngster was frightened from the nest, and upon examination a portion of a meadow mouse was found there. Pellets also were found there, indicating that the nest probably was used for perching as well as for a feeding place. Care of this brood of young continued, gradually decreasing, until about the time the second brood was hatched, whereupon the first brood dispersed.

A second nest was built and the eggs laid while the first brood was still being fed. Whether this nest was built by the male, or by the female, or both, is not known. On May 15, 1939, the first brood of young was found recently to have left the nest and was just able to fly, and on June 1, while the first brood was still being cared for, another nest belonging to the same pair was found a little higher on the ridge. It contained five eggs, the only ones seen by me. The nest was much the same as the first, both in structure and location in the tree. On June 25, on my return after an absence of twenty-three days, only three downy young and one infertile egg were present. This again left three young which in all probability would have reached maturity but for an unforeseen development. These young, even more clearly than those of the first brood, appeared to have been hatched on successive days, as they differed but little in size. When first seen, they were clothed with a tan down that rapidly took on a blue cast as the dark sheaths of pinfeathers developed beneath the skin surface. On June 26 this brood was in the down stage and by July 5, when they were found dead, much of the down was submerged in the well-developed juvenal plumage.

The ill-fated second brood was well started when a blind was erected for Dr. A. A. Allen, who wished to take pictures and sound recordings of the birds. This was on June 29. On June 30 the birds were still quite active, but when I returned on July 5 the birds were dead. The only explanation that could be offered for this desertion was the erection of the blind, as all my other operations around the two nests had not driven the adults away. The old birds were watched carefully in hopes that a new nest might be started, but after a few days they were no longer seen.

The first White-tailed Kite seen by me in Santa Cruz County was noticed west of Watsonville in a willow bottom on April 17, 1939. After the death of the three young near Corralitos, this area was investigated further and on August 11 a nest was found about eighteen feet up in a willow. This area was much different from the first and more nearly conformed to some of those described by Pickwell (*op. cit.*). The site consisted of a willow-lined drainageway, with not more than two or three willows abreast down the bottom, which was bordered by an alfalfa-ryegrass meadow and a field that is rotated between pasture and hay. Both of these open areas, especially the meadow, had a high concentration of meadow mice in 1939. This pair of birds did not wander out of this territory as far as is known and quite unlike the Corralitos pair, it tolerated raptors within it. Sparrow Hawks, Red-tails and Marsh Hawks were ignored even when they flew past the nest. Other kites were in this area also. A total of nine, two adults (the third pair), and seven young, being seen about one-half mile down the bottom. Food was much more abundant here, which may account for the greater concentration of kites.

This nest also contained four young, but one was pushed from the nest by the adult the first time I approached it. The youngster that was pushed out was evidently just hatched, as it was still in the tan down state and had the "egg tooth" on the upper mandible. It was kept for three days before dying, apparently from internal injuries sustained in the fall. The egg tooth had nearly disappeared by this time. The three remaining young developed much as did the two broods near Corralitos. They were banded on August 31, with the assistance of R. H. Mors (bands 39-643009, -010 and -011). They reacted to handling much as the other brood did, but the adults, after a few half-hearted dives, retired to their usual perching tree and mournfully screamed. The young birds at this stage, approximately twenty-one days old, showed the black in the shoulders; the back was brown as was the breast, and the belly was more nearly white. The eyes were dark brown and the feet were yellow. They reacted to an intruder

Fig. 29. A young White-tailed Kite at twenty-one days of age.

by backing away from him out into the surrounding branches. Later, with progressive development, they backed further out, then into branches farther from the nest, and finally attempted to fly to nearby trees, usually landing in nettles from which they were painfully extracted. On September 15 the largest bird was in another tree close by and the smaller two were in the nest, whereas on September 18 they had all left the nest and flew when approached.

Near this nest a perch was found surrounded by pellets and down. From all appearances young had perched here, and this, with the seven young seen in the area, leads me to believe that the nest contained a second brood when I visited it and that possibly a first brood had been raised there also.

The nest of the fourth pair was never found, but the hunting ground of the old birds (later seen with their young) was near Palm Beach southwest of Watsonville. The willow-lined Pajaro River flows about a mile away on the south while there is a willow-lined marsh on the northeast, either of which may have been the nesting ground. This pair scouted over quite a distance for food, following a grass and pickleweed-bordered drainage ditch for two miles or so.

The different behavior of the first and second pairs in the care of the nest, eggs and young is interesting. Both pairs placed the nests deep enough in the tree to provide at least a dappled shade when the young were left for long periods of time, but in actual nest protection they reacted differently. The second pair behaved much as Pickwell (*op. cit.*) described, but the Corralitos pair was courageous in their defense of the nest. This latter pair did not attempt to protect the eggs, but after the young had hatched interest was quickened and a few dives were made at the intruder at the nest. As the young birds developed, these dives were greater in number, more vicious, and accompanied by a hoarse cry that was echoed by the young. By the time the young were ready to leave the nest, the old birds would take turns diving to within about eighteen inches of the intruder's head, turning upward for another dive. This continued even after the young had left the nest and when the second nest contained eggs. The second pair usually left as soon as the observer arrived. A few half-hearted dives were made as climbs were made to the nest, but they in no way equalled the ferocity of the Corralitos pair.

It is hard to believe, in view of the numbers of kites in 1939, that the species did not nest during 1937 and 1938 (my first two years in the area), but no birds were seen. The following discussion of food habits may, however, cast light on the subject.

The food habits of the White-tailed Kite have not been discussed in most of the studies to date. This seems strange, as pellets are readily picked up from beneath the perching trees and in the general vicinity of the nest. The White-tailed Kites observed did not use insects as did the Mississippi Kites observed by Sutton (Condor, vol. 41, 1939, p. 50). Rather the meadow mouse (*Microtus californicus*) is the *piece de resistance* in the White-tailed Kite diet. This is shown in the following table:

Corralitos pair, 111 pellets	Individuals in pellets	Percentage of individuals
Microtus californicus	141	88.1
Reithrodontomys megalotis	15	9.5
Thomomys bottæ	1	0.6
Neotoma fuscipes (young)	1	0.6
Citellus beecheyi (very old)	1	0.6
Sylvilagus (young)	1	0.6
	160	100.0
Second pair, west of Watsonville, 52 pellets		
Microtus californicus	54	100

In addition, one whole meadow mouse and two pellets containing meadow mouse skulls were taken from under a perch of a single kite in the San Andreas district near Watsonville, and 15 pellets containing 24 meadow mice were found under a kite perch at Palm Beach.

From observation it appears that the kite population of southern Santa Cruz County is governed more by the relation of food to suitable nesting places than by the whims of gunners. Miles of kiteless willow bottom that have no adjacent meadow or well managed pasture show that it takes more than the necessary nesting trees to make a nesting site. It is believed that a convenient source of suitable food is just as necessary as the trees. From the food habits summary, one might narrow the above statement down to the apparent fact that a high population of meadow mice is necessary for successful kite nesting. As before stated, the observer saw no White-tailed Kites during the nesting seasons of 1937 and 1938, both years relatively low in meadow mouse numbers. In the winter of 1938-1939, the meadow mouse population was high, possibly due to the fact that during wet years the normal habitat of the meadow mouse in this area is flooded or nearly so during the winter, while in dry years, such as 1938-1939, it is not. The fact that the kites nested twice in at least one, and probably in two instances, may also depend upon the food supply. As the idea has been advanced that there is some correlation between the numbers of deer and mountain lions, as well as between the numbers of kit foxes and kangaroo rats, so the idea might be advanced that there is some correlation between the nesting and numbers of White-tailed Kites and the numbers of meadow mice. Of course much of the above is merely conjecture and should be accounted as such. The amount of meadow, alfalfa, willow and oak land depends on the farmer and I am sorry to say that more and more willow, oak, and meadow disappears each year before the onslaught of intensive farming. The Soil Conservation Service has been able to encourage farmers to make plantings, has suggested that certain willow-lined channels be left or improved, and in general has been able to stem somewhat the tide of clean farming, which action may assist kites as well as other species.

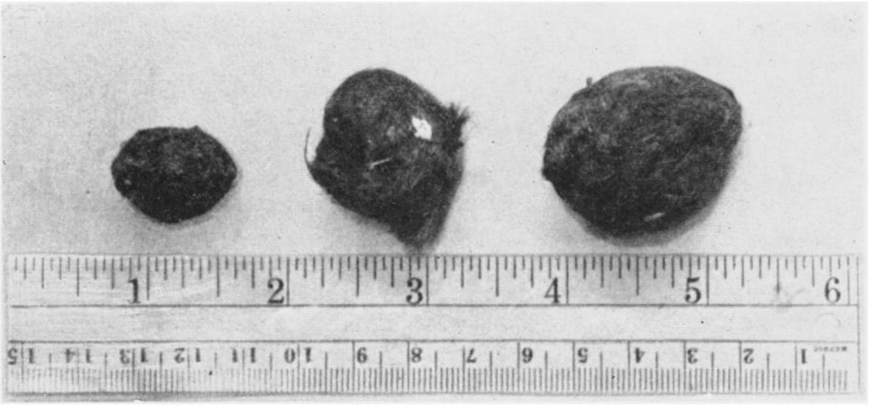

Fig. 30. Kite pellets resemble those of owls rather than those of hawks in size, compactness, and inclusion of undigested bones.

SUMMARY

Twenty-three White-tailed Kites were seen in southern Santa Cruz County in the nesting season of 1939. Eight of these were adult nesting birds.

One pair is known without any doubt to have nested twice, bringing one brood to maturity and abandoning the second through interference of man. A second pair is believed to have nested twice. The first brood is cared for as long as necessary, even after the second set of eggs is hatched.

One pair nested late (probably a second brood), very small young being found August 11, 1939, which left the nest September 15 to 18, 1939.

The young are fed on the nest even after taking wing.

The principal food is the common meadow mouse (*Microtus*), the abundance of which may be a determining factor in the nesting, nesting site, and number of broods raised per year.

Soil Conservation Service, Watsonville, California, November 8, 1939.

CRITERIA FOR GOLDEN EAGLE, FERRUGINOUS HAWK, AND PRAIRIE FALCON NEST SITE PROTECTION*

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Abstract

Establishment of buffer zones around raptor nest sites has become an important management tool in areas undergoing energy development or increasing recreational pressure. We conducted a survey of field researchers who had distributed Golden Eagle, Ferruginous Hawk, and Prairie Falcon during their research. Bases for and limitations of the use of buffer zones to protect nesting raptors are discussed.

Introduction

Energy development and other human activities can diminish raptor populations by altering habitat and by disturbing nesting activities. Disturbance of nesting raptors can result in complete desertion of nests, eggs, or young. Temporary departure by adults can cause overheating, chilling, or desiccation of eggs or young, predation on eggs or young, or missed feedings. Three studies of the Golden Eagle (*Aquila chrysaëtos*) found that 46, 71, and 85 percent of nesting failures were due to human disturbance (Boeker and Ray 1971, Camenzind 1969, D'Ostilio 1954). The effects of such disturbance range from loss of a year's reproduction to long-term loss of the nest site if the disturbance is chronic. Raptor researchers found that by disturbing birds they can jeopardize the reproductive activity being studied (Fyfe and Olendorff 1976).

Concern over disturbance has resulted in the establishment of spatial or temporal buffers (restriction of activity within an area or period of time) between some energy developments and raptor nest sites. Geothermal development proposals for sites in Utah and Idaho resulted in recommendations for buffers by federal agencies (ERDA 1977, Fisher 1978, USGS 1977). Buffer zones were established to protect raptor nest sites along the Trans-Alaska pipeline (Olendorff and Zeedyk 1978) and were recommended for the proposed Mackenzie gas pipeline (Jacobson 1974). These recommendations were based primarily on the experience of the individuals involved because of the absence of a body of literature on responses of the birds to these disturbances or any consensus of the raptor research community concerning control of disturbance. This study summarizes and expands the bases for such decisions relative to the Golden Eagle, Ferruginous Hawk (*Buteo regalis*), and Prairie Falcon (*Falco mexicanus*).

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Methods

Raptor field research usually involves some disturbance and often allows observation of the effects of other sources of disturbance. Unfortunately, these observations are not routinely reported. To get information, a survey form (table 1, shown with results) was sent to 74 appropriate raptor researchers; a second copy was sent to nonrespondents 2 months later. Questions were framed in terms of the level of disturbance that would elicit a reaction from 20 percent of nesting birds. This criterion was used to avoid the high variance associated with estimates of the reaction of a hypothetical, most sensitive bird. Because the Golden Eagle, Ferruginous Hawk, and Prairie Falcon are not classified as threatened or endangered, protection need not be absolute. These species were chosen because they are the most sensitive raptor species with which western developments will frequently conflict.

Because some survey returns indicated that the use of buffer zones is controversial, a workshop on raptor disturbance was conducted at the 1978 Raptor Research Foundation meeting. While the large attendance and short duration of this workshop prevented the formulation of a consensus, the issues were clearly defined and are discussed below.

Results

Twenty-four surveys were completed and returned with numerical information; 6 additional respondents provided only comments. Numerical results are summarized in table 1. Since the distribution of responses to each question was positively skewed, the median provides the best measure of central tendency. The median is also more useful than the mean because it represents a central or typical response rather than the average magnitude of responses. Median reaction distances were lowest for the Prairie Falcon and highest for the Ferruginous Hawk, but most of the differences between species were not statistically significant.

Factors other than distance and stage in the breeding cycle that were thought to be important in determining the response to a particular disturbance by more than one respondent were existence of a clear line of sight, security of the nest, history of disturbance to which the birds have been exposed, elevation of the disturbance relative to the nest, and whether the birds were the focus of attention. Recommended buffer zones for these species found in the literature or received in response to the surveys are presented in table 2.

Discussion

The objection to nest-site protection most frequently raised at the workshop was that the entire habitat must be protected. If this were necessary, raptors would be absent from areas supporting any human activity. The habitat factors requiring protection are those that limit the population size or that may become limiting as a result of development. Olendorff and Stoddard (1974) found that nest-site availability apparently limits raptor poulations in northeastern Colorado and southeastern Washington. Edwards (1969) found that Golden Eagle density was limited by nest-site availability in western Utah, and Boeker and Ray (1971) found the same to be true for the Southwest in general. Smith and Murphy (1978) attribute the low nesting density of Ferruginous Hawks primarily to nest-site limitations. This is likely to be the case in much of the arid and semiarid west when a sufficiently large area is considered because prey habitat is abundant relative to nesting habitat. Golden Eagles and Prairie Falcons typically require cliff

Table 2. Recommended Buffer Zones for Golden Eagle, Ferruginous Hawk, or Prairie Falcon Nest Sites.

Distance	Species	Development type	Restriction	Source
1 km (0.6 mile)	Golden Eagle Prairie Falcon	Geothermal drilling	No drilling	ERDA 1977
0.5 mile (0.8 km) all year and 1 mile (1.6 km) March 1–July 15	Ferruginous Hawk	Geothermal drilling	No surface disturbance	Fisher 1978
1 mile (1.6 km)	All eagles	Pipeline		Olendorff and Zeedyk 1978
2 miles (3.2 km) all year	Golden Eagle	Pipeline	No construction	Jacobson 1974
2 miles (3.2 km) March 1–Sept. 1	Golden Eagle	Pipeline	No ground activity	Jacobson 1974
0.25–0.5 mile (0.4–0.8 km)	Golden Eagle	General		M. R. Fuller[a]
200–500 m	All three species	General		N. Woffinden[a]
0.5 mile (0.8 km)	Grassland raptors	General		R. P. Howard[a]
1 mile (1.6 km) line of sight	Golden Eagle	General		R. P. Howard[a]

[a] Suggestions received in response to the raptor disturbance survey.

sites. Ferruginous Hawks are more versatile, but most require a tree or rock outcropping. This use of elevated nest sites contrasts sharply with the open-land hunting habit of these species. The importance of nest sites is confirmed by Fyfe and Armbruster's (1976) and Anderson and Follet's (1978) success in increasing the productivity of Prairie Falcons and Ferruginous Hawks, respectively, by nest-site creation and manipulation (see also Howard and Hilliard 1980, White 1974).

Nest-site protection is only advantageous if the prey base remains adequate following development. Many types of development such as oil, gas, and geothermal exploitation, pipeline and road construction, and development of campgrounds and interpretive facilities on public lands remove vegetation from small areas. If important prey concentrations such as ground squirrel colonies are avoided, raptors should be able to coexist with these developments provided nesting sites are undisturbed. The responses to survey question 5 indicate that development should be kept at least 400 m from such prey concentrations.

Another objection to nest-site protection was that disturbance might occur because of the establishment of buffer zones. This disturbance could be caused by irate supporters of the development that would be restricted or by nest robbers, varmint shooters, amateur naturalists, or photographers who are attracted to identified nest sites. The location of nest sites should be revealed only to those who are directly involved in facility siting. Developers should be reminded that the nest site, not the individual birds, is being protected. Shooting the birds would not eliminate the need to restrict development near the site.

General suggestions for buffer zone sizes can be made on the basis of survey responses. To avoid thermal stress to eggs or young, activities such as geological, biological, or soil surveys that are performed intermittently by a few individuals should be kept at least 500 m from active nest sites or limited to a few minutes and periods of moderate temperature. Construction and similar noisy, extended activities should be kept at least 1 km from nest sites to avoid nest abandonment. At this distance, nesting birds are also out of rifle range and are relatively inconspicuous to users of new roads or other facilities. These suggested distances lie within the range of buffer zone sizes listed in table 2. They are not absolute and should be modified by knowledgeable individuals to fit the circumstances of the project and nest site. Knowledgeable advice is also necessary to determine if buffer zones are the appropriate management tool for the circumstances.

Temporal buffers may supplement or be used in place of spatial buffers. Temporal buffers should include all nesting activities but must at least extend from the time of arrival of the adult birds in the nesting area through the first few weeks of nestling development (see Call 1978 for average dates). After this time young are increasingly able to thermoregulate, and adults are reluctant to abandon them. Activity close to the nest (within flushing distance) must wait until fledging is completed and young are independent of the nest area. The use of temporal buffers depends on the ability to schedule activities on an annual basis.

A second alternative to spatial buffers around existing nest sites is the construction of artificial nesting sites. This technique was reviewed by Olendorff and Stoddard (1974) as a method to introduce raptors into unused grassland. The disadvantages of artificial sites as a mitigation technique are that they may not always prove acceptable to the displaced species, they may attract the "wrong" species, and they are typically more conspicuous than natural sites.

Further support for raptor preservation must be provided by field research. One approach is to experimentally disturb nesting birds (White et al. 1979). This type of research is limited by the ability to realistically simulate development activities and by the small number of pairs available. The most valuable information will come from the monitoring of responses to real developments and observation of the distribution of active nests relative to ongoing human activities. These observations should appear more frequently in the literature.

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Table 1. Responses to the raptor disturbance survey (distances in meters).

Question	Golden Eagle			Ferruginous Hawk			Prairie Falcon		
	Med	Range	n	Med	Range	n	Med	Range	n
1. At what distance would an individual or small group of people approaching a nest cause 20% of sitting birds of each species listed to flush from the nest during the following periods?									
a. laying	160	15-1073	16	275	91-1609	17	166	15-1609	14
b. incubation	100	15-805	17	274	5-805	18	91	15-402	15
c. rearing young	333	15-1609	16	337	75-1609	18	250	15-610	15
2. At what distance would extended activities involving several persons and approximately 90 dBA noise, e.g., drilling or earth moving, cause abandonment of the nest by 20% of individuals of each species during the following periods?									
a. nest construction	550	250-4023	16	902	300-3218	15	366	61-1609	13
b. laying	478	250-4023	16	600	105-3218	16	400	61-1609	15
c. incubation	402	150-1609	15	451	100-1609	16	366	30-1295	15
d. rearing young	383	100-1609	16	333	30-1609	16	274	30-805	15
3. At what distance would a noisy									

species listed to flush from a nest during the following periods?									
a. laying	183	30-2414	15	402	91-3218	16	191	30-1609	14
b. incubation	187	30-2414	16	366	25-1609	15	126	25-805	14
c. rearing young	350	30-2414	15	384	75-1609	14	183	30-805	14
4. At what distance would frequent (1 per hour) noisy off-road vehicles cause abandonment of the nest by 20% of the individuals of each species during the following periods?									
a. nest construction	457	91-5632	15	500	91-3218	14	274	91-1407	11
b. laying	457	91-2414	15	451	75-1609	14	201	46-1609	13
c. incubation	402	91-2414	15	383	75-1609	14	200	46-805	13
d. rearing young	200	50-2414	15	250	25-1609	14	200	30-805	13
5. Within what distance of an activity involving several people and large equipment would at least 80% of the members of each species hunt and kill prey?	400	30-3218	15	402	30-3218	17	320	30-1609	13
6. What factors, other than distance and stage in the breeding cycle, do you believe are important in determining the probability that a particular activity will cause a bird of the above species to temporarily or permanently abandon a nest site?					(See text)				

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Recommendations for protecting raptors from human disturbance: a review

Cary T. Richardson and Clinton K. Miller

In a survey of resource managers, LeFranc and Millsap (1984) identified human-associated disturbance as a primary threat to raptor populations. Several studies have demonstrated declines in raptor populations resulting from human-associated disturbance (Voous 1977, Swenson 1979, Craighead and Mindell 1981). Resource managers can successfully use spatial and temporal buffer zones in concert to protect raptors from the effects of recreational activity (Swenson 1979, Knight and Skagen 1988, Holmes et al. 1993), human development (Ramakka and Woyewodzic 1993), and oil development (Squires et al. 1993). Spatial and temporal restrictions (buffer zones) are useful tools for resource managers to protect raptors during periods of extreme sensitivity (Knight and Skagen 1988, Knight and Temple 1995). We present information relevant to the establishment of buffer zones and the guidelines for assessing spatial and temporal buffer zones for a variety of raptors in North America. This review may serve as a general guideline for resource managers and others interested in protecting raptors.

The need for nest site protection

Human activities are known to impact raptors in at least 3 ways: (1) by physically harming or killing eggs, young, or adults; (2) by altering habitats; and (3) by disrupting normal behavior (Postovit and Postovit 1987). Due to the broad range of direct and indirect human-associated impacts and the fluctuating levels of sensitivity for individual raptors, depending on life stage and time of year, buffer zones are most effective when spatial and temporal restrictions are congruent.

The direct effects of human disturbance may seem inconsequential to uninformed or unconcerned outdoor recreationists. Activities like rock-climbing, can have severe impacts on nesting raptors, even when climbers do not have direct contact with eggs, young, or adults (Lanier and Joseph 1989, Kelly 1996). This sport often involves shouting and other noises which are disturbing enough to raptors to keep them away from their nests (Call 1979, Ratcliffe 1980). Even brief absence by parent birds can lead to missed feedings, predation on eggs or young, or to overheating, chilling, or desiccation of eggs or young (Call 1979, Suter and Joness 1981). Rock-climbing near peregrine falcon (*Falco peregrinus*) eyries during the nesting season can cause nest abandonment; some peregrine falcons are extremely sensitive and refuse to breed if humans have been in the vicinity of their eyries (Snow 1972, Olsen and Olsen 1980). Ferruginous hawks (*Buteo regalis*) tend to desert their nests if adults are exposed to human activity during incubation (White and Thurow 1985). Van Daele and Van Daele (1982) found that incubation at successful osprey (*Pandion haliaetus*) nests occurred during 99.5-100% of daylight hours. Human disturbance during the critical periods of incubation and the early nesting stages can be fatal to embryos and nestlings.

The presence of humans detected by a raptor in its nesting or hunting habitat can be a significant habitat-altering disturbance even if the human is far from an active nest. Impacts of human activities on wild animals are often reduced when animals are shielded visually from such activities (Postovit and Postovit 1987, Knight and Temple 1995). A clear line of sight is an important factor in a raptor's response to a par-

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ticular disturbance (Suter and Joness 1981). A Geographic Information System-assisted viewshed approach combined with a designated buffer zone distance was found to be an effective tool for reducing potential disturbance to golden eagles (*Aquila chrysaetos*) in Northern Colorado (R. L. Knight, Colo. State Univ., Fort Collins, pers. commun.).

Human disturbance was listed as the cause of 85% of all known nest losses occurring during Boeker and Ray's (1971) study of golden eagles. Disturbance of wintering bald eagles (*Haliaeetus leucocephalus*) resulted in both increased energy expenditures due to avoidance flights and decreased energy intake due to interference with feeding activities (Stalmaster 1983). The enforcement of spatial and temporal buffer zones can protect raptors from the effects of visual disturbances (e.g., human development or recreation), audible disturbances, and direct disturbances (e.g., shooting, recreational rock-climbing).

Determining adequate protection

Several authors have provided general recommendations for determining adequate site-specific buffer zones (Postovit and Postovit 1987, Pomerantz et al. 1988, Holmes et al. 1993). Postovit and Postovit (1987) detailed steps for mitigation planning. Pomerantz et al. (1988) gave a useful set of guidelines that could be used to determine the compatibility of recreational activities in sensitive resource areas. In designing appropriate buffer zones the most important factors are: site-specific information on the horizontal and vertical proximity of a nest to a potential disturbance, source or duration of disturbance, and disturbance history of the individual raptors (Suter and Joness 1981, Postovit and Postovit 1987, Knight and Skagen 1988, Holmes et al. 1993).

Site-specific information

Physical characteristics (i.e., topography, vegetation) are important variables to consider when establishing buffer zones based on raptors' visual-and auditory-detection distances. Horizontal spatial restrictions can be shortened or lengthened depending on the height of perching or nesting sites (Holmes et al. 1993). Given variable nesting phenology of different species and regional climatic variation, exact dates of nest-site closures should be modified according to local situations (U.S. Fish and Wildl. Serv. 1984). White and Thurow (1985) recommend that the degree to which a nest is exposed or concealed should be considered when designing buffers for ferruginous hawks. They also suggested that information on the general health and status of individual populations be considered. For example, in years of food scarcity, spatial buffers should be expanded substantially.

Source or type of disturbance

Management plans should be tailored to each species, habitat, season, and source of disturbance. For example, Holmes et al. (1993) argued that, because humans in vehicles are less disruptive to raptors than pedestrians, management plans should offer different restrictions based on disturbance type. Squires et al. (1993) suggested that prairie falcons (*Falco mexicanus*) could cope with limited development on their foraging areas if their nest sites were secure from direct human disturbance. Nonthreatening activities, such as those occurring on recreational trails, may be compatible with a nest or perch location in close proximity if that activity is visually or audially buffered by vegetation or topography (Knight and Temple 1995).

Prior disturbance history of individual raptors

Due to variation of tolerance between bald eagle populations, Stalmaster and Newman (1978) suggested monitoring adult behavior prior to the establishment of management recommendations and buffer zones to determine to what extent the individuals had been sensitized to human disturbance. They noted that although a single direct disturbance may have insignificant impacts, repeated direct disturbances may cause abandonment of a nest or perch location.

Spatial and temporal buffer recommendations

Spatial buffers

Spatial buffer-zone recommendations depend on site specific considerations, and vary considerably for species such as osprey, Cooper's hawk (*Accipiter cooperii*), northern goshawk (*Accipiter gentilis*), sharp-shinned hawk (*Accipiter striatus*), golden eagle, red-tailed hawk, (*Buteo jamaicensis*), ferruginous hawk, bald eagle, prairie falcon, peregrine falcon, and American kestrel (*Falco sparverius*; Table 1). Median distances recommended for buffer zones for nesting raptors (based on the information summarized in Table 1) are as follows: osprey = 1,000 m (range = 400-1,500 m, *n* = 3), Cooper's hawk = 525 m (range = 400-600 m, *n* = 2), northern goshawk = 450 m (*n* = 1), sharp-shinned hawk = 450 m (*n* = 1), golden eagle = 800 m (range = 200-1,600 m, *n* = 3),

Table 1. Summary of recommendations for spatial and temporal buffer-zones for nesting raptors.

Species	Spatial (m)	Temporal	Reason for closure	Source
osprey	1,500	not discussed	human activity	Van Daele and Van Daele 1982
	400	Apr 1–Aug 31	no explanation	Colo. Div. Wildl. 1995
	1,000	during incubation	recreational disturbance	Swenson 1979
Cooper's hawk	600	not specified	habitat alteration	Bosakowski et al. 1993
	400–500	not specified	unspecified disturbance	Jones 1979
northern goshawk	400–500	not specified	unspecified disturbance	Jones 1979
sharp-shinned hawk	400–500	not specified	unspecified disturbance	Jones 1979
golden eagle	200 from cliff tops; 400 from base	Mar 1–Jun 30	human disturbance	M. Ball, U.S. For. Serv., Fort Collins, Colo., pers. commun.
	800	Feb 1–Aug 1	noise	Call 1979
	200–1,600	Mar 1–Sep 1	visual, audible	Suter and Joness 1981
	800	Feb 1–Jul 15	no explanation	Colo. Div. Wildl. 1995
red-tailed hawk	800	Feb 1–Aug 1	noise	Call 1979
ferruginous hawk	200–800	arrival–post fledging	visual, audible	Suter and Joness 1981
	250	during incubation	human activity	White and Thurow 1985
	800	Feb 1–Jul 15	no explanation	Colo. Div. Wildl. 1995
bald eagle	400	Feb 1–Aug 15	human disturbance	D. Flath, Mont. Dep. Fish, Wildl. & Parks, Bozeman, pers. commun.
	800	Feb 1–Aug 1	noise	Call 1979
	500	not discussed	human disturbance	Fraser 1983
	250	prior to egg laying through incubation	human activity	Grier et al. 1983
	800	Nov 15–Jul 31	no explanation	Colo. Div. Wildl. 1995
prairie falcon	200 from cliff tops; 400 from base	Mar 1–Jun 30	human disturbance	M. Ball, U.S. For. Serv., Fort Collins, Colo., pers. commun.
	800	Feb 1–Aug 1	noise	Call 1979
	200–800	arrival-post fledging	visual, audible	Suter and Joness 1981
	800	Mar 15–Jul 31	no explanation	Colo. Div. Wildl. 1995
	50	Mar 15–post fledging	visual	Natl. Park Serv. 1995
peregrine falcon	800	Feb 1–Jul 15	climbing disturbance	S. Johnson, Natl. Park Serv., pers. commun.
	800–1,500	not discussed	recreational disturbance	Windsor 1975
	800	Feb. 1–Aug. 1	noise	Call 1979
	1,600	Feb 1–Aug 31	human activity	U.S. Fish and Wildl. Serv. 1984
	800	Mar 15–Jul 31	no explanation	Colo. Div. Wildl. 1995
American kestrel	200 from cliff tops; 400 from base	Mar 1–Jun 30	human disturbance	M. Ball, U.S. For. Serv., Fort Collins, Colo., pers. commun.
	50	Mar 15–post-fledging	visual	Natl. Park Serv. 1995

red-tailed hawk = 800 m ($n = 1$), ferruginous hawk = 500 m (range = 200–800 m, $n = 3$), bald eagle = 500 m (range = 250–800 m, $n = 5$), prairie falcon = 650 m (range = 50–800 m, $n = 4$), peregrine falcon = 800 m (range = 800–1,600 m, $n = 5$), and American kestrel = 50–200 m ($n = 2$). Several studies have recorded flushing distances for raptors resonding to disturbances from pedestrians and vehicles (Table 2).

Table 2. Flushing distances (m) for raptors in response to disturbance by pedestrians and vehicles.

Species	Pedestrian disturbance	Vehicle disturbance	Source
golden eagle	105–390	14–190	Holmes et al. 1993
ferruginous hawk	13–165	110–280	Holmes et al. 1993
	136.4 (range = 29–291)	117.2 (range = 24–316)	White and Thurow 1985
rough-legged hawk	55–900	9–170	Holmes et al. 1993
bald eagle	50–990	50–990	Holmes et al. 1993
	57–991 (91% > 200 m)	not studied	Fraser 1983
			Fraser et al. 1985
prairie falcon	24–185	18–200	Holmes et al. 1993
American kestrel	10–100	12–115	Holmes et al. 1993
merlin	17–180	44–85	Holmes et al. 1993

Temporal buffers

For temporal restrictions to be effective, they must be tailored to individual populations. In addition, temporal restrictions need only be in effect when raptors are using a critical resource such as a nest site or foraging area (Knight and Skagen 1988). Temporal buffers should encompass all nesting activities and extend at least from the arrival of the adult birds in the nesting area through the first few weeks of nestling development (Fyfe and Olendorff 1976, Suter and Joness 1981, Grier et al. 1983, White and Thurow 1985). Adult birds often sit tightly on eggs or young nestlings, and when adults flush abruptly due to disturbances, there is increased liklihood of their ejecting the contents of their nests (Grier and Fyfe 1987).

Summary

Several studies have documented flushing distance responses of raptors to a variety of activities during breeding and nonbreeding seasons (Table 2); however, except for anecdotal and incidental reports, few studies have experimentally documented disturbance distances for use in buffer-zone recommendations (White and Thurow 1985, Holmes et al. 1993). The wide range of recommendations (Table 1) probably reflects site-specific anthropogenic and environmental conditions (Suter and Joness 1981, Fraser 1983). To be effective, buffer zones should be based on empirical evidence of wildlife responses to disturbance (Knight and Skagen 1988). Several authors suggest the need for further disturbance studies to determine flushing responses among different species (White and Thurow 1985, Postovit and Postovit 1987, Knight and Temple 1995).

The City of Boulder Open Space Department and Mountain Parks Division have used spatial and temporal buffer zones successfully for a number of years to protect cliff-nesting peregrine falcons, prairie falcons, and golden eagles. Closures are in effect from February through July annually and vary in distance by 50–400 m depending on topography, nest location, and species. Extensive public education accompanies the closures, including direct mailings to outdoor recreation shops in the area, closure signs at trailheads, press releases, and access to a 24-hour telephone information line and a site on the World Wide Web. In addition, nest sites are monitored weekly by trained volunteers. With proper planning, extensive observations of target individuals and groups, and aggressive public education, spatial and temporal buffer zones provide a useful tool for protecting raptors to resource managers.

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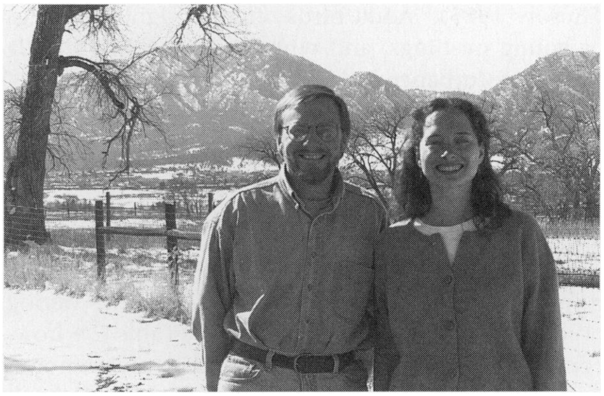

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BURROWING OWLS AND DEVELOPMENT: SHORT-DISTANCE NEST BURROW RELOCATION TO MINIMIZE CONSTRUCTION IMPACTS

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ABSTRACT.—During June–July 1998, we used a combination of active and passive relocation to move five Burrowing Owl (*Athene cunicularia*) nests in artificial burrow systems (ABS) that faced destruction by development in southwestern Idaho. Regulatory agencies agreed that relocation of the nest burrows would allow construction to proceed and provide an opportunity to determine the efficacy of moving occupied Burrowing Owl nests as a mitigation technique. Relocated nests contained one to five nestlings, ranging in age from 27–45 d. ABS (plastic chamber and tunnel), wooden perches, and dependent young were relocated (active relocation) to adjacent areas that contained natural vegetation; adults were not moved but were expected to travel the short distances to new burrow locations on their own (passive relocation). Access to natural burrows near original nest locations was restricted where possible. Relocation distances averaged 153 m and ranged from 72–258 m. Because terrain was flat, new nest locations generally were within view of original burrow locations. Relocations were successful at two of five nests. For two other nests, both adults and young returned to the vicinity of the original nest and occupied natural burrows 1 d after relocation. Owls from the fifth nest were not detected following burrow relocation and presumably vacated the immediate vicinity of the construction.

KEY WORDS: *Burrowing Owl;* Athene cunicularia; *nest relocation; artificial burrow system; active relocation; passive relocation; mitigation technique.*

Búhos Cavadores y desarrollo: redisposición de las cuevas nido a corta distancia papa minimizar los impactos de la construcción

RESUMEN.—Durante Junio–Julio 1998, usamos una combinación de reubicación activa y pasiva para mover 5 nidos de Búho Cavador (*Athene cunicularia*) a sistemas de cuevas artificiales (ABSs), estos nidos estaban a punto de ser destruidos por el desarrollo en el sudoeste de Idaho. Las agencias reguladoras estuvieron de acuerdo que la redisposición de los nidos cueva debería permitir proseguir la construcción y proveer una oportunidad para determinar la eficacia de mover nidos ocupados de Búho Cavador como una alternativa de mitigación. Los nidos reubicados contenían de uno a cinco polluelos, con edades entre 27–45 d. Los ABSs (cámara y túnel plásticos), perchas de madera, y los jóvenes nidícolas fueron reubicados (reubicación activa) a áreas adyacentes que contenían vegetación natural; los adultos no fueron movidos pero se esperaba que recorrieran por su propia cuenta las cortas distancias a los nuevos sitios de las cuevas (reubicación pasiva). El acceso a las cuevas naturales cerca de los sitios de los nidos originales fue restringido a donde quiera que fue posible. Las distancias a la reubicación promediaron 153 m en un rango de 72–258 m. Debido a que el terreno era plano, las nuevas ubicaciones de los nidos generalmente estaban a la vista desde los sitios de las cuevas originales. La reubicación fue exitosa en dos de los cinco nidos. Para los otros dos nidos, ambos adultos y el joven retornaron a la vecindad del nido original y ocuparon cuevas naturales 1 día después de la reubicación. No se detecto que los búhos del quinto nido siguieran la reubicación de la cueva y presumiblemente se dispersaron de la vecindad inmediata de la construcción.

[Traducción de Victor Vanegas y César Márquez]

Burrowing Owl (*Athene cunicularia*) populations are declining throughout much of their range in North America (De Smet 1997, James and Espie 1997, Sheffield 1997). Human disturbances, such as elimination of burrowing mammals, use of pesticides and herbicides, and conversion of grasslands to agricultural or urban areas, are factors contributing to the decline in Burrowing Owl num-

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bers (Zarn 1974, Haug et al. 1993). Anthropogenic habitat change is continually displacing owls, forcing them from previous seasons' nesting areas, reducing prey abundance and foraging areas, and potentially limiting opportunities for breeding. Although regulations protect the owls, situations where Burrowing Owls and land uses conflict continue to arise.

To minimize direct impacts resulting from habitat conversion for agriculture or development, mitigation efforts often attempt to provide Burrowing Owls with suitable habitat near areas scheduled for development. Once mitigation land is established near an impact area, owls are either evicted (i.e., passive relocation) or actively relocated (Trulio 1995, Feeney 1997). Passive relocation usually occurs in the nonbreeding season or immediately before the breeding season commences. Under this scenario, owls are excluded from available natural burrows in areas slated for development and are forced to seek alternate burrows in nearby habitat outside the areas directly affected by construction. Active relocation entails: 1) capturing owls and moving them to suitable habitat, which is generally well removed from the original site; and 2) releasing the owls at a new site, often after a period of acclimation in temporary aviaries. To replenish or reintroduce populations, Burrowing Owls also have been translocated into areas where suitable habitat remained but natural populations had declined or were extirpated (Martell 1990, Dyer 1991). Translocation projects require active capture and transport of adults and juveniles from breeding areas and then release in establishment sites.

The efficacy of these mitigation techniques (active relocation, passive relocation, and translocation) has varied. Most relocation projects resulted in fewer breeding pairs of Burrowing Owls at the mitigation site than at the original site, and translocation projects generally have failed to produce self-sustaining populations. Investigators attribute the limited success of management efforts to: 1) strong site tenacity exhibited by Burrowing Owls, and 2) potential risks associated with forcing owls to move into unfamiliar and perhaps less preferable habitats (Trulio 1995, Delevoryas 1997, Feeney 1997). Further research on methods of Burrowing Owl relocation and translocation may lead to an increase in the success of these techniques.

In this study, we examined the responses of Burrowing Owl families to short-distance nest burrow relocation. We predicted that nest-site fidelity would be overcome through parental responses to their offspring, thus eliminating the need to capture and relocate adults.

We conducted this research in response to the planned destruction of a 130-ha field, in which five pairs of Burrowing Owls nested in 1998. Each artificial burrow system (ABS) contained a pair of adults and their dependent fledglings, which were still closely associated with their nest burrow. Before young were ready to leave their natal area (i.e., flight skills improving, but still dependent on adults), the field became a borrow pit for construction of a wastewater treatment facility; ultimately, the site will function as an effluent field in which alfalfa and other cover crops are grown. To allow the project to proceed, state and federal regulatory agencies agreed that the situation offered an opportunity to examine the feasibility of relocation of Burrowing Owl nest burrows to minimize construction impacts. We decided that nest burrows would be relocated to the periphery of the construction project, into a buffer strip surrounding the field. Burrow relocations would allow construction to continue without costly delays that would result from waiting until the owls migrated from the construction area after the breeding season.

This study provides data on relocation of ABS occupied by Burrowing Owls to determine if passive adult and active fledgling relocation is a feasible mitigation technique to avoid or reduce direct impacts from construction or other anthropogenic pressures.

METHODS

Study Area. Five Burrowing Owl nests were located approximately 3 km south of Kuna, Ada County, which is 32 km southwest of Boise, Idaho and <23 km north of the Snake River Canyon. Topography was flat to rolling, and elevations ranged from 841–896 m. Rock outcrops and a few isolated buttes (e.g., Kuna Butte, elevation 896 m) exist in the region. Annual temperatures range from −20 to +45°C, and annual precipitation typically averages <20 cm (NOAA 1985).

The study area was once a typical shrub-steppe community dominated by big sagebrush (*Artemisia tridentata wyomingensis*, Hironaka et al. 1983). Range fires and other disturbances have converted much of the surrounding shrublands to exotic grasslands dominated by cheatgrass (*Bromus tectorum*) and tumble mustard (*Sisymbrium altissimum*). The area contained a few homes, several large dairy farms, paved and gravel roads, and irrigated agricultural fields that grew primarily alfalfa, mint, and sugar beets. Irrigated agricultural fields bordered the northern, eastern, and southern sides of the field that was scheduled for construction, and a two-lane highway bordered the field's western edge. Previously excavated badger (*Taxidea taxus*) burrows were abundant throughout the study area and served as nest and shelter sites for Burrowing Owls (King 1996, King and Belthoff 2001).

Fledgling Data. Before moving nest burrows, we estimated the age of juveniles based on feather growth (Landry 1979) and the estimated hatching date of the brood (±1 d, Smith 1999). For individual recognition in the field, each owl received one United States Geological Survey aluminum leg band and a unique combination of three plastic color bands (National Band and Tag Co., Newport, KY).

Nest Relocation. Each of the five nest burrows were in ABS deployed as part of another study (Smith 1999, Smith and Belthoff 2001) in 1997 (Nos. 1, 3, and 5) and in 1998 (Nos. 2 and 4). Therefore, active relocation of nests and juveniles was relatively simple when compared with moving nests from natural burrows. This project occurred during the latter part of the nesting cycle; thus, we expected adult owls to move the short distance from the original nest area to the relocation site (i.e., passive relocation). However, nest burrows and fledglings were physically moved (i.e., active relocation) to sites outside the impacted area.

All five nests were relocated to a buffer strip between 25 June–9 July 1998. The buffer strip was along the western and southern borders of the field, was approximately 25 m wide, and was the nearest habitat with natural vegetation suitable for ABS placement (Fig. 1). We selected new nest locations that were as close as possible to the original nest location in areas deemed to provide sufficient space and habitat for owls. New sites generally were no closer to neighboring nests than were original sites (except for Nos. 3 and 5; Table 1) and, in each case, new nest locations were within view of original nests. After site selection, we: 1) dug holes to place relocated ABS, 2) removed all fledglings from their nest chambers, 3) removed each ABS intact (i.e., the chamber and tunnel), 4) buried each ABS at the new location with the same orientation as the original burrows, and 5) returned juveniles to nest chambers. We also moved the wooden perches from the original sites to the new sites to lure adult owls, who used the perches for roosting. Each ABS was encircled with highly-visible flagging to reduce chances that construction personnel would inadvertently disturb the new sites. To determine the fate of each relocated nest, we monitored relocation areas (via spotting scope from a vehicle as far away as possible) each day after relocation for 2 wk, and at least three times/wk thereafter until the date that migration normally commenced.

Burrowing Owls exhibit strong site attachment behavior (Trulio 1995, Delevoryas 1997, Feeney 1997), so we were aware that some owls might return to their original nest locations after the nest burrow was removed. To minimize this possibility, we first placed Owl Exclusionary Devices (OED) at natural burrows near the original nest site. Each OED consisted of a 0.5-m section of perforated plastic drainage pipe and a piece of transparent Plexiglas® attached to a hinge at one end of the pipe. Once placed at the entrance to a natural burrow, OED allowed any owls that were underground to exit but prevented owls from taking up residence at such burrows. We also attempted to coordinate relocations such that original nest areas would be destroyed shortly after nest burrows were moved, thus reducing the likelihood that owls would return to original nest locations.

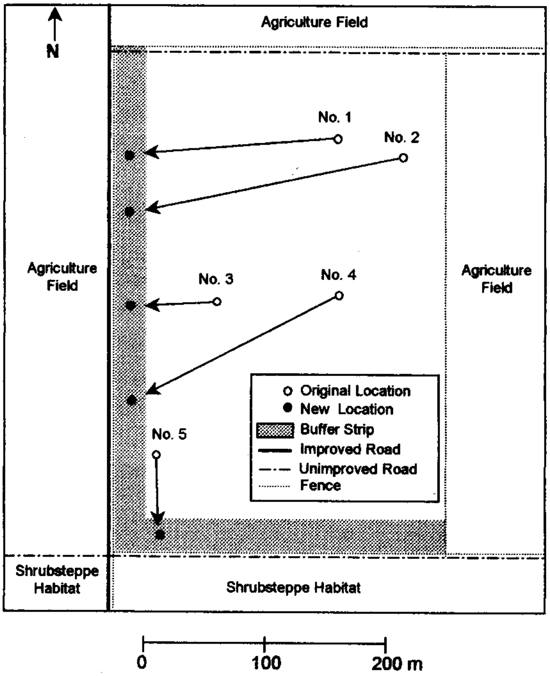

Figure 1. Original and new locations of artificial burrow systems relocated to minimize construction impacts on Burrowing Owl nests in southwestern Idaho, 1998. Numbers indicate nest burrows and their associated young that were relocated to a buffer strip along the western and southern border of the field; adults were not captured but were expected to locate the new sites on their own. The entire field (except the buffer strip) was leveled by machinery soon after all nests were relocated.

Upon relocating each ABS, we measured the distance (to nearest 0.5 m) and direction from the original nest location to its new site. We considered a relocation successful if the owl family took up residence at its new location and remained until dispersal or migration. Unsuccessful relocations occurred when owl families returned to their original nest areas or immediately disappeared from the study area; dispersal from natal areas at this young age is not characteristic of Burrowing Owls (King 1996, King and Belthoff 2001).

RESULTS

Fledgling Data. At the time of relocation, the number of juveniles at each ABS varied from one to five, ranging from 27–45 d post-hatch (Table 1). These young had developed modest to good flight capabilities, but they still depended on parental

Table 1. Information on Burrowing Owl young, relocation measurements, and apparent fates of relocated nests. Juveniles and artificial nest burrows were relocated during the 1998 breeding season to minimize construction impacts in Ada County, Idaho.

NEST	NUMBER OF YOUNG	AGE (d) OF YOUNG[a]	RELOCATION DATE	DISTANCE MOVED (m)	NEAREST NEST BEFORE (m)	NEAREST NEST AFTER (m)	FATE
No. 1	2	39–40	25 June	174	55	55	Accepted new site
No. 2	4	38–39	25 June	258	55	55	Site tenacity
No. 3	5	35–38	7 July	79	102	85	Disappeared
No. 4	1	27	9 July	183	102	85	Site tenacity
No. 5	3	44–45	7 July	72.5	290	271	Accepted new site

[a] Estimated based on morphological development and estimated hatching dates. Young >28 d are considered fledglings.

care and remained associated with natal burrows. We captured and relocated all juveniles within each ABS except at No. 5 where, upon our approach to the nest, one fledgling flew ca. 25 m away. At No. 1, both young were captured and relocated, but immediately after being relocated one juvenile flew across the two-lane highway in the opposite direction of the original site.

Nest Relocation. Relocation distances averaged 153 m, ranging from 72.5–258 m, and four of the five nests were moved in a westerly direction (Table 1; Fig. 1). Overall, two families accepted their relocation sites (40%), two families (40%) returned to the vicinity of their original nest burrows, and one family (20%) disappeared from the field (Table 1). All family members from Nos. 1 and 5 were observed at their new sites 1 d after relocation, and both adults and fledglings from each family used their new sites for several weeks until they disappeared. In contrast, two families (Nos. 2 and 4) did not remain in the relocation areas. Instead, 1 d after relocation, family groups from these nests were observed at natural burrows <20 m away from their original nest burrows. The adult male from No. 4 began using the perch, and possibly the ABS, at the new site approximately 10 d after relocation, but his young and his mate remained near the original nest. Fates of birds from these nests are unknown, except for the female from No. 2 (see below). We believe family No. 3 moved from the immediate vicinity of both the original nest and the relocated burrow, even though this nest was moved only 79 m from the original site. After moving this ABS and all five fledglings, no members of the family were observed again at the original or relocation sites, or in nearby areas that contained suitable habitat for Burrowing Owls. The fates of the members of this family were also unknown, except for the male from No. 3 (see below). Finally, within the period of our study, dates of relocation events did not appear to be related to relocation outcomes (Table 1).

In 1999, two adults returned to the area and fledged young successfully from ABS that had been relocated to the buffer strip in 1998. The adult female that nested in No. 2 in 1998 (an unsuccessful relocation) nested at the relocated No. 2 ABS in 1999. The male that nested at No. 3 in 1998 (also an unsuccessful relocation) nested at the relocated No. 5 ABS. This represented a 20% return rate (by sex, and overall) for adults affected by construction in this field in 1998. During 1999, we observed none of the 15 fledglings from 1998 nests, despite continued work in the area.

DISCUSSION

Burrowing Owls typically remain within 50–100 m of their nest or satellite burrows during daylight hours (Haug and Oliphant 1990) and exhibit strong nest-site tenacity, even after a site has been disturbed (Zarn 1974, Feeney 1997). Because Burrowing Owls commonly use burrows in close proximity to their nest burrows for roosting, escape cover, and other activities (Zarn 1974, Haug et al. 1993), relocated nests should be in close proximity to the original nest burrow (Trulio 1995). For successful relocations in our study (Nos. 1 and 5), burrows were generally closer to their original sites than were those relocations considered unsuccessful (Nos. 2–4). However, three of five relocation distances were greater than the 100-m maximum distance that Trulio (1995, 1997) recommended for passive relocation techniques. Because shorter relocations generally were more successful, distance also may have been a relevant factor in the type of relocations we employed. However, as No. 3 family members were relocated only 79 m and apparently disappeared from the study area, other factors besides distance must play a role in relocation success.

Burrowing Owls commonly return to the same or nearby nest burrows year after year (Thomsen 1971, Rich 1984, Botelho and Arrowood 1998). For the relocations that we considered to be successful (Nos. 1 and 5), banding information from our study area showed that both adult males and one adult female bred successfully in the same field during the previous (1997) breeding season. Such experience could have made these owls more familiar with relocation areas and led to their increased willingness to accept new sites. For the three relocations we considered unsuccessful (Nos. 2–4), one adult male was known to have nested in this field during 1997, and the family dispersed from the field immediately following relocation. Ages and previous breeding experiences were unknown for the two remaining pairs, as these birds were not banded before they entered the 1998 breeding season. Nonetheless, familiarity with this field may have influenced whether a family accepted their relocation site, returned to the original nest area, or dispersed from the area.

Although immediate success was realized for two relocations, long-term success of relocations and their effects on Burrowing Owls are also important. In 1999, one female and one male returned to the buffer strip to nest (both had new mates). Of the two remaining ABS, one was occupied by a pair of unmarked owls and the other was unoccupied. The fifth ABS was destroyed during the nonbreeding season. Return rates for females on the impacted area were similar to female return rates over the entire area (20% vs. 24%, respectively) for 1997–98, but were lower for males on the impacted area than over the entire area (20% vs. 44%, respectively, J. Belthoff and B. Smith unpubl. data). We failed to detect any of the juveniles from this study in the impacted field or in surrounding areas during 1999. However, this is not surprising because only 15 juveniles were associated with this field, and first-year return rates are very low (<4% of banded individuals during 1997–98) for birds in our area (J. Belthoff and B. Smith unpubl. data). Nonetheless, the subsequent return and successful nesting of two adults to the impacted site in 1999 suggested that our methods provided both immediate and longer-term success for some of the owls involved.

Other factors also may have affected the owls' willingness to accept new sites. Unfamiliar disturbances (e.g., traffic) could have caused the owls to reject the new sites (Feeney 1997). Both Nos. 2 and 4 (unsuccessful relocations) were relocated from relatively quiet portions of the field to <25 m from a busy road (Fig. 1). Given surrounding land use and destruction of the field, the placement of each relocated nest was restricted to the buffer strip because it offered the nearest "suitable" habitat. Also, we were unable to have the original nest areas destroyed immediately because of inclement weather (i.e., destruction of sections of the field did not occur on planned dates). These delays, or our inability to locate all natural burrows near original nest locations to place OED, potentially allowed two families (Nos. 2 and 4) to return to natural burrows near their original nest areas.

Finally, for the two successful relocations (Nos. 1 and 5), one juvenile from each nest either was not captured or escaped during the relocation process. At the time of relocation, juveniles from successful nests also were older than those from unsuccessful nests. It is not clear if or why these factors would affect the tendency for families to remain in the relocation area. Possibly, separation of family members led to increased rate of contact vocalizations by juveniles, which lured adults to the new site more readily, or the older individuals were more visible because of increased activity (i.e., practice flights, perching, hunting) around the relocation site.

Our results indicated that short-distance relocation of occupied nests was successful under some circumstances, although the factors associated with success remained unclear. Regardless, the relocations we performed avoided the almost certain death of many young owls that would have resulted from construction. Because this was a small study (five nests), success rates for the techniques described here should be quantified in much larger studies before such relocations are considered viable options. Additionally, whether the techniques we examined would relate also to owls nesting in natural burrows (the most likely situation faced in many areas) remains unknown. Currently, we recommend that these techniques be used only when no alternatives exist. Postponing mitigation and construction activities until the nonbreeding season (i.e., after dispersal and/or migration occurs), as well as compensating for any habitat loss or degradation, would be the preferred approach to reduce impacts on Burrowing Owls. If mitigation activities cannot be avoided, original nest areas should be destroyed immediately after moving the owls so they cannot return to the original burrow, or any other burrow, in the impacted area (Trulio 1995). Finally, it remains unknown whether actively relocating adults with their dependent young would affect success rates of short-distance relocations. If the stress of capture on owls is not severe, it seems reasonable that including adults would increase relocation success. However, it may be difficult to capture adults late in the nesting cycle, so timing of the relocation would be important. Therefore, passive relocation of adults and active relocation of fledglings may encourage adult Burrowing Owls to overcome nest-burrow tenacity and inhabit new burrows to care for young when relocations are over short distances.

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BURROWING OWL STATUS IN THE GREATER SAN FRANCISCO BAY AREA

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Abstract. We reviewed the status of the Western Burrowing Owl *(Athene cunicularia hypugaea)* in the greater San Francisco Bay Area (SFBA). Despite numerous Burrowing Owl surveys conducted in this region, estimating population size for the SFBA counties remains difficult. We found that the most abundant populations were in Alameda, Contra Costa, and Santa Clara counties and locally abundant within portions of Solano and San Mateo counties historically. For the remainder of the SFBA counties (Napa, San Francisco, Marin, and with the possible exception of Santa Cruz), Burrowing Owls were neither widespread nor abundant. Breeding Burrowing Owls are extirpated from four of 10 SFBA counties (Napa, Marin, San Francisco and Santa Cruz) and likely extirpated from two others (southwest Solano and San Mateo counties). Burrowing Owls are no longer abundant and may be disappearing entirely from western Contra Costa, western Alameda, and Santa Clara counties. Numerous breeding pairs have recently been detected in portions of eastern Contra Costa and Alameda counties; both these areas are under threat. We identified the following threats: habitat loss, current eviction methods (both passive and active), inadequate replacement acreage for mitigation, pest management, and rodent poisoning. Further research is important for conservation; for example, determining if the SFBA Burrowing Owl population is genetically unique and how that might affect population viability. There may be hope for the SFBA Burrowing Owl population; however, it has already undergone a substantial decline in abundance and is greatly diminished in extent from its former distribution.

Key Words: Athene cunicularia, Burrowing Owl, California, population status, San Francisco Bay Area.

Organizers of the California Burrowing Owl Symposium (November 2003) invited speakers to prepare summaries on the regional status of Burrowing Owls throughout most of California. Our task was to collate current information on population and trends of Burrowing Owls in the greater San Francisco Bay Area (SFBA).

We briefly review what was known historically and what has been published about the distribution and trends in abundance of Burrowing Owls, most of which was based on the information provided in the *Petition for Listing the California Population of the Western Burrowing Owl as an Endangered or Threatened Species under the California Endangered Species Act* (Center for Biological Diversity [CBD] et al. 2003; hereafter the Petition). We updated this information with our informal investigation of abundance, distribution, and population trends of Burrowing Owls within the SFBA.

Our estimate for Burrowing Owl numbers is only for areas for which we could obtain information; there remain portions of these SFBA counties that have not been surveyed and sightings of Burrowing Owls that have gone unreported. We also sought to obtain information about trends of Burrowing Owl numbers in areas where trends were identified as "unknown" from the Petition. In addition, we identify what we consider to be on-going threats to the SFBA population and make suggestions for management and conservation research.

METHODS

STUDY AREA

Our study area encompassed 10 counties or portions of counties in the SFBA: Alameda (eastern and western), Contra Costa (eastern and western), Marin, Napa, San Francisco, San Mateo, Santa Clara, Santa Cruz, Solano (southwest), and Sonoma counties.

DISTRIBUTION AND TRENDS IN POPULATION

We used the Petition to establish an estimate of population and trends. We augmented the Petition's findings with personal knowledge and various documents (Breeding Bird Atlas data for Contra Costa, Napa/Solano, Marin, and San Francisco), unpublished and published technical reports, supporting documentation for the Petition, and the California Natural Diversity Database (CNDDB 2003). Additionally, we contacted and interviewed knowledgeable biologists, environmental consultants, open space managers, and members of local Audubon chapters.

Petition authors generated trends for Burrowing Owls by county or a portion of a county (for example, Alameda and Contra Costa counties were divided into eastern and western). We updated these population estimates and trends based on our findings. We relied on our personal knowledge and local experts to fill in informational gaps about numbers of Burrowing Owls in our study area. In particular, we focused on obtaining trend and abundance information for areas in which the Petition cited as "unknown" for Burrowing Owls. We summarized this additional information, but this information is insufficient for estimating population size. We know from personal experience that some areas support undocumented breeding Burrowing Owls, in particular, the eastern grasslands of Alameda and Contra Costa counties (see Addendum).

We identified factors that we believe have contributed to the local extirpation of the Burrowing Owl in the SFBA. These findings are based on our extensive experience in the field, opinions from local biologists, information in the Petition, and one of the author's (CL) current research.

RESULTS

DISTRIBUTION

Within the SFBA, the Burrowing Owl was historically considered to be "most numerous in parts of Alameda, Contra Costa, and Santa Clara counties" (Grinnell and Wythe 1927:123). In the last century, the species was also documented as locally abundant in southwestern Solano County and San Mateo County (WFVZ 2001). In Santa Cruz County (Skirm 1884, McGregor 1901), Burrowing Owls were noted as abundant but by the mid-1900's were rare due to California ground squirrel *(Spermophilus beecheyi)* poisoning (Streator 1947, see Petition). Although there are documented records of breeding Burrowing Owls in Napa, Marin, and San Francisco counties, they do not appear to have ever been widespread or common within these three counties (see the Petition for details of historical records).

Two of the 10 counties, Napa and San Francisco, have few to no records of Burrowing Owl occurrence historically (Table 1). Three counties, Marin, Sonoma, and San Mateo, have records from a few locations indicating that Burrowing Owls were not widespread throughout any of these counties. Southwest Solano County had a locally abundant population (see Table 1, CBD et al. 2003). While Burrowing Owls in Santa Cruz were present after the mid-1900s decline, they were not widespread or abundant (CBD et al. 2003).

Currently, breeding Burrowing Owls are extirpated from four SFBA counties (Napa, Marin, San Francisco, and Santa Cruz counties) and likely extirpated from two others (Sonoma and San Mateo counties) where we were unable

TABLE 1. Historical summary of abundance of Burrowing Owls in seven of 10 San Francisco Bay Area counties.

County	Historical Summary	Findings
Napa	No records of breeding	Never abundant
San Francisco	Few records	Never abundant
Marin	Few breeding areas	Not widespread or abundant
Sonoma	Recorded from four areas	Unknown numbers, not widespread
SW Solano	Locally abundant in the City of Benicia	Not widespread; abundant in one location
Santa Cruz	Fairly common until mid-1900s	Decline in the mid-1900s; later decline
San Mateo	Present in Redwood City and record from Menlo Park	Not widespread; at least one healthy breeding population

to confirm any observations of recent breeding attempts. We could not verify any observations of breeding pairs from portions of two additional counties (southwestern Solano and western Contra Costa). Our findings supported the findings about trends for these counties in the Petition (Table 2).

Based on historical records and qualitative assessments of abundance, these six counties (above) represent areas where Burrowing Owls were neither abundant nor widespread historically with the exception perhaps of Santa Cruz; however, the decline in Santa Cruz predates loss of habitat due to development (Table 2). These counties may have been more vulnerable to extirpation due to the historically low numbers and isolated colonies (groups of breeding pairs) or individual breeding pairs; this distribution may indicate that the habitat available in these counties was (and most likely still is) marginal or patchy for Burrowing Owls (Tables 1 and 2).

TABLE 2. Estimated number of Burrowing Owl pairs and trends from the Petition and historical and recent findings for seven of the 10 San Francisco Bay Area counties.

County	Estimated pairs* (year)	Trend*	Estimated pairs (recent findings)	Trends (recent findings) and historical abundance
Napa	0 (1993)	Extirpated	No records	Appears extirpated; historically rare
San Francisco	0 (1993)	Extirpated	0?	Appears extirpated; historically rare
Marin	0 (1993)	Extirpated	Few wintering	Appears extirpated, never abundant
Sonoma	1-2 (1993)	Nearly extirpated	None recorded	Appears extirpated; not widespread
SW Solano	0? (2002)	Nearly extirpated	Known populations; no numbers: currently observed in Travis AFB, Birds Landing, Grizzly Island, Chadbourne Rd	Unknown trend; historically not widespread
Santa Cruz	0 (1993)	Extirpated	Winter only	Appears extirpated; decline mid-1900s
San Mateo	1-2 (2001)	Nearly extirpated	No records	Localized population greatly diminished or absent; historically not widespread.

*Petition (CBD et al. 2003)

According to historical records (see CBC et al. 2003 for exhaustive discussion), Burrowing Owls were abundant in Alameda, Contra Costa (particularly in the eastern portion), and Santa Clara counties. Following the Petition, we considered trends separately for eastern and western Contra Costa and Alameda counties. Three parts of our study area that supported widespread, abundant populations of Burrowing Owls have experienced extirpation of local populations: western Alameda, western Contra Costa, and Santa Clara counties. The Petition noted a declining trend for these counties or portions of them (Table 3), and our results supported those findings.

Santa Clara County supported Burrowing Owls in open and undeveloped spaces in the cities of San Jose and Santa Clara, which have experienced rapid commercial and residential development in the past 20 years. For unknown reasons, there was little information about, and little evidence to suggest, that Burrowing Owls were abundant in the more southern portion of this county. Despite loss of previously occupied habitat to development in Santa Clara County, the extant monitored populations of Burrowing Owls at NASA Ames Research Center (including Moffett Federal Airfield) and Mineta San Jose International Airport showed good survivorship and owls there were increasing in numbers (see Trulio and Chromczak and Barclay, this volume; Table 3). However, these

TABLE 3. Burrowing Owl pair estimates for the parts of the counties listed as "Unknown" in the Petition (2003).

Location	Estimates†
Eastern Alameda County	
Ohlone/Sunol	Occasional wintering birds
SFWD* San Antonio Reservoir	2 pairs
Dublin	2-3 pairs
Camp Parks, Dublin	11 pairs
North Livermore Ave.	At least 2-3 pairs
EBRPD* Brushy Peak	2 pairs
Bethany Reservoir	Several pairs
Windfarm at Altamont Pass	Unknown
Altamont Hills	Unknown, probably many
Mountain House Golf Course	At least 9 pairs
Sweet Ranch, Altamont Pass	2-3 pairs
Western Alameda County	
Oakland Hills	Extirpated
Oakland Airport	At least 1 pair
Martin Luther King Shoreline	1 pair
Hayward/Fremont	At least 4 pairs
Eastern Contra Costa County	
Concord/Pittsburgh	Unknown
EBRPD Black Diamond Mines	Unknown
South Antioch	> 8 pairs left
Brentwood	Several pairs
West Brentwood	3+ known pairs
Byron	Unknown, probably many
EBRPD Vasco Caves	3-4 pairs
Western Contra Costa County	(No information)
Santa Clara County	
San Jose Airport	29 pairs (2003)
Alviso and nearby	31 pairs (2000)
South Bay Study Area*	41 pairs

†from our findings (2003)

*Legend: SFWD- San Francisco Water District, EBRPD-East Bay Regional Park District, South Bay Study Area – L. Trulio.

populations are thought to represent a fraction of the original Burrowing Owl population for this part of the SFBA. The Petition noted declining trends for Santa Clara County; the authors agreed that the population has likely declined over the past 20 years due to development (Table 4).

Western Alameda and Contra Costa counties along the East Bay shore have also experienced a loss of occupied habitat due to build out within the last 10 to 20 years. Subsequently, locally known Burrowing Owl colonies have disappeared or greatly diminished in numbers. For example, Burrowing Owls were known to breed at the Oakland Airport (Thomsen 1971) and the Martin Luther King Shoreline Park but now each of these sites supports only one pair (Table 3). The Petition noted a "nearly extirpated" trend for western Contra Costa and declining trend in western Alameda County (Table 4). Our findings regarding trends supported the Petition's findings.

The Petition cited an unknown trend for both eastern Alameda and Contra Costa counties (Table 4). They currently support Burrowing Owls consistent with historical locations with some exceptions (Table 3). Burrowing Owls have been detected near Antioch, Brentwood, and Byron (Table 3). However, these cities are rapidly developing. We assume that Burrowing Owls are declining in eastern Contra Costa County due to the recent loss of occupied habitat to development (Tables 3 and 4). In eastern Alameda County in the Altamont Hills, a large number of breeding Burrowing Owls appeared to be extant although there are no good estimates of the total number of breeding pairs. We noted that there likely exist numerous breeding pairs, particularly in the Altamont Pass area (ST and CL, see Addendum), and it may be that the population is stable (although see section below). Because of the lack of numerical data, the trend remains unknown (Tables 3 and 4).

Based on the literature, the authors' experience, and our extensive conversations with local biologists, the threats to Burrowing Owls included habitat loss, current eviction methods, inadequate mitigation acreage, pest management, and California ground squirrel poisoning.

DISCUSSION

DISTRIBUTION

Based on historical records, Burrowing Owls do not appear to have been widespread or abundant in six of the 10 counties within the SFBA. These counties include Napa, San

TABLE 4. Estimated number of Burrowing Owl pairs and trends from the Petition and our findings for Alameda (eastern and western), Contra Costa (eastern and western) and Santa Clara counties.

County	Estimated pairs (from Petition)	Trend (from Petition)	Estimated Pairs*	Trend
Alameda (eastern)	Unknown	Unknown (historical decline)	At least 41 pairs	Unknown; Stable? threats to existing population (e.g., Altamont Hills)
Alameda (western)	Unknown	Declining	At least 6 pairs	Declining (post-decline?)
Contra Costa (eastern)	Unknown	Unknown	At least 14 pairs	Assume declining based on recent loss of occupied habitat
Contra Costa (western)	0? (2002)	Nearly extirpated	No records	Extirpated?
Santa Clara	121 – 141 (1997)	Declining	101 pairs (2000 and 2003)	Reduced from historical; 2 study areas stable and increasing; unknown for single pairs and colonies (e.g., Alviso)

* See Table 3

Francisco, Marin, Sonoma, southwest Solano, and San Mateo. Two of these counties, southwest Solano and San Mateo, may have supported a locally abundant breeding Burrowing Owl population. Santa Cruz may have had a moderate number of Burrowing Owls historically but experienced a major decline in the mid 1900s. Burrowing Owls were historically abundant in Alameda, Contra Costa, and Santa Clara counties. Portions of these counties have experienced build-out and high rates of development that have subsequently diminished owls' density and distribution in these areas.

TRENDS AND POPULATION

Trend information may be the most meaningful in understanding the overall health (viability) of the SFBA Burrowing Owl population. Based on our findings, Burrowing Owls have most likely been extirpated as a breeding species from Napa, San Francisco, Marin, Sonoma, San Mateo, and Santa Cruz counties. Southwest Solano County may still support several breeding groups; however, we were unable to obtain reliable numbers of breeding pairs (see Table 2). A few wintering individuals have recently been reported in Marin and Santa Cruz counties (CL).

Burrowing Owls are extant but appear to be greatly diminished in distribution and abundance in western Alameda and Santa Clara counties and may be extirpated from western Contra Costa County where they were historically abundant and widespread (the Petition and our findings; Table 4). We assume that this is due to build out (commercial and residential development) in areas historically and, in many cases, recently occupied. There are several explanations of what may happen to displaced birds. The remaining breeding pairs may occupy marginal habitat, or breeding habitat may just not be available; Burrowing Owls excluded from former breeding habitat may have moved into entirely new breeding areas elsewhere in California. Unfortunately, we have very little understanding of Burrowing Owl movement once they are excluded from breeding habitat. NASA Ames Research Center and San Jose Airport are exceptions that support breeding groups that recently increased in numbers and density (Trulio and Chromczak and Barclay, this volume).

Within the SFBA, eastern Contra Costa and Alameda counties may support the last healthy breeding populations in their original historical habitat (besides arguably the above San Jose Airport and NASA Ames Research Center breeding groups). These areas are not protected. Extirpation or large declines from areas that historically and even recently supported healthy Burrowing Owl populations may be more significant to the overall persistence of the Burrowing Owl in this region than extirpation from counties where Burrowing Owls were historically rare or less numerous.

The rate of loss of occupied habitat in eastern Contra Costa County should be quantified. However, under the current statutes of protection for the Burrowing Owl, there is no mechanism that guarantees that Burrowing Owl habitat will be protected in order to sustain this local population. In eastern Alameda County in the Altamont Hills, development is occurring (Mountain House golf course). However, there are other factors such as mortality due to wind turbine strikes and secondary loss of habitat that will likely result from recent increases in ground squirrel poisoning in the open spaces around these windfarms that could reasonably be expected to negatively affect breeding populations. The long-term detrimental impacts to Burrowing Owls in this particular region may result in the extirpation of one of two of the last healthy breeding populations of Burrowing Owls in the SFBA (if indeed the breeding population in eastern Contra Costa County could be considered healthy).

THREATS TO POPULATION AND HABITAT

Habitat loss caused by development is the most immediate threat to Burrowing Owls that reside in high growth areas of the SFBA. Unfortunately, loss of Burrowing Owl habitat will likely continue well into the future. Currently 7,091,700 people live in the SFBA and that number is projected to increase to 8,747,100 by 2030 (ABAG 2005). As urbanization increases and local Burrowing Owl populations decline, they become vulnerable to stochastic events (demographic, genetic, and environmental) associated with small population size, creating the potential for extinction "vortex" (Gilpin and Soulé 1986).

Current protocols for managing Burrowing Owls in developing areas may also threaten their survival and reproduction. Eviction methods (i.e., passive and active relocation methods; CDFG 1995) used to remove Burrowing Owls from project sites prior to development likely causes stress to these Burrowing Owls. When using one-way doors for passive relocation, CDFG recommends that replacement burrows are available on nearby or adjacent lands secured as long-term Burrowing Owl habitat; however, in practice, replacement burrows are usually not provided. Therefore, this "avoidance" measure only serves to accelerate habitat loss especially when alternative nearby replacement burrows and conserved lands are not available. Even when alternative nesting habitat is provided, the adequacy of these sites is often unknown. Furthermore, in our experience, post-eviction monitoring (e.g., banding and/or telemetry) to determine the fate of these owls is rarely implemented or required. This lack of monitoring is especially problematic as Burrowing Owls exhibit strong site fidelity even after a nest site is disturbed (Feeney 1997, Millsap and Bear 1997). Displaced birds, unfamiliar with new areas, are less likely to breed (Peltz 2005) and are more susceptible to mortality from predators (Dyer 1987). Local biologists expressed concern over the fate of Burrowing Owls experiencing a high rate of disturbance around developing and urban areas.

Currently, 6.5 ac (2.6 ha) off-site mitigation lands are placed into conservation easements per displaced pair or resident individual to mitigate for the loss of Burrowing Owl habitat in the SFBA both for wintering and breeding habitat. While setting aside land for Burrowing Owls is positive, off-site mitigation leads to the loss of demonstrated breeding habitat for the resident owls directly affected by development. A radial zone of 300 ft (91.44 m) around an active Burrowing Owl nest equals 6.5 ac (26,304.6 sq m). This radial distance from the burrow was originally intended to determine where disturbance might cause a significant zone of impact to a nest (CBOC 1997) and not to calculate off-site mitigation acreage for Burrowing Owls affected by development projects.

In our opinion, protecting the few remaining Burrowing Owl colonies *in situ* is extremely important. We suspect that few truly undisturbed Burrowing Owl colonies remain in the SFBA. Furthermore, data suggest that the number of pairs per breeding group has decreased in the SFBA (DeSante et al. 1997), resulting in a risk to population viability. Population models based on recent research in Florida suggest that owl colonies may disappear when the population drops below 20 adults (Peltz 2005). Burrowing Owls within the SFBA, living in highly fragmented habitat exposed to human-induced disturbance, may be highly sensitive to local extinction according to small population theory (Goodman 1987, Johnson 1997). Therefore, active Burrowing Owl colonies within the greater SFBA need much better protection and should be managed as source groups for the larger SFBA population.

Another class of threats is grassland management, primarily of open space and private lands. Pest management, grazing regime, and wind turbines influence Burrowing Owl populations in open grassland habitat in the SFBA. Ground squirrel poisoning reduces nesting opportunities for Burrowing Owls and causes breeding locations to shift across the landscape. Furthermore, poisoning kills more than ground squirrels. Commonly used second generation anticoagulant rodenticides are highly toxic to small mammals and birds, including diurnal raptors and owls that consume poisoned prey (S. Hoover, Altamont Wind Resource Area Wildlife Ecologist, personal communication, Mendenhall and Pank 1980, Peeters 1994, Stone et al. 1999).

In California, ground squirrels, like black-tailed prairie dogs *(Cynomys ludovicianus)*, play a key and often dual role as prey and/or provider of burrows (Kotliar 2001) for a variety of species including Burrowing Owls, Golden Eagles *(Aquila chrysaetos)*, badgers *(Taxidea taxus)*, coyotes *(Canis latrans)*, rattlesnakes (*Crotalus* sp.), kit fox (*Vulpes macrotis* ssp.), California tiger salamanders *(Ambystoma californiense)*, and red-legged frogs (*Rana aurora* ssp.). Throughout the SFBA, California ground squirrels are the primary provider of burrows used by Burrowing Owls. The ecological cost of reducing ground squirrel densities is high in light of the influence these squirrels have on California's special-status species and other native wildlife. Ideally, managers of California's grasslands should carefully consider the ecological cost of poisoning programs. In addition, impacts resulting from poisoning need to be adequately considered under existing environmental laws (Federal and State Endangered Species Acts [FESA and CESA], National Environmental Policy Act [NEPA] and the California Environmental Quality Act [CEQA]).

RECOMMENDATIONS

We recommend that a scientifically-defensible Burrowing Owl breeding pair survey using peer-reviewed methodologies be conducted in eastern Contra Costa and Alameda counties to collect baseline abundance data. In addition, we recommend focusing protection measures on what appear to be healthy populations of owls that occur in portions of counties where they were also formerly abundant (historical data from the Petition and our findings). Using all available conservation tools to protect and enhance Burrowing Owl habitat (e.g., mitigation, easements, park expansion) will allow the opportunity to prevent further declines in counties where Burrowing Owl presence has been greatly diminished both in extent and numbers. An issue which we did not address but should be considered for conservation purposes is to determine to what extent the SFBA plays a role in providing overwintering habitat and how this affects the stability of regional populations of Burrowing Owls.

Finally, considering the near extirpation of the Burrowing Owl from areas where they were formerly abundant, the remaining Burrowing Owls should receive more consideration for conservation. The standard for listing under CESA is that the species is "declining in a significant portion of its range"; we believe that the Burrowing Owl has declined in a significant portion of the greater SFBA. Whether the Burrowing Owl needs further protection by listing under CESA or FESA or is more carefully considered by regulatory agencies with its current status, we do not believe that the current level of protection that is being afforded the SFBA Burrowing Owls ensures their future survival.

Many Burrowing Owl observations and locations remain unrecorded. The CNDDB maintains location records of California wildlife, and we encourage those working on the Burrowing Owl to submit locations to this database (www.dfg.ca.gov/whdab/pdfs/natspec.pdf; see McGriff, this volume). This database is underutilized for Burrowing Owls, and much of the data on Burrowing Owls in the SFBA remains inaccessible in researchers' and consultants' files. In light of the plight of this species and in the interest in conserving it, we proposed developing a free and easily accessible web-based database for California Burrowing Owls. We have much to learn by carefully documenting the Burrowing Owl story.

RESEARCH FOR CONSERVATION

Research and conservation questions specific to SFBA Burrowing Owls should be pursued and any existing studies to date should be published, preferably in peer-reviewed journals. During our efforts to assess the status of SFBA Burrowing Owls, we identified several topics for further research:

1. Is this SFBA population uniquely adapted to the SFBA and its maritime and estuarine influences? Is it reproductively isolated from the greater Central Valley population? If so, how can we preserve its viability considering current threats?
2. Are SFBA Burrowing Owl breeding populations viable? It seems that several studied colonies are still doing relatively well. Barclay and Trulio and Chromczak (this volume) show that survivorship and productivity are comparably good in colonies they studied (and continue to monitor), while human-induced disturbance in other less-protected areas may be causing higher mortality and decreased nesting activity (DeSante et al. 1997 and CL).
3. Burrowing Owls in this region have high site fidelity (Feeney 1997). How does this affect their ability to shift breeding and/or overwintering locations after an eviction or disturbance especially in those areas that are ostensibly set aside for Burrowing Owl mitigation?

Despite the apparent decline and local extirpation of Burrowing Owls in the SFBA, we believe there is still reason for guarded optimism. Specific sites within Santa Clara County still sustain viable colonies (for example, Mineta San Jose International Airport, NASA Ames Research Center) and breeding pairs are present in eastern Contra Costa and Alameda counties. These examples illustrate that with some management and consideration, these highly adaptable animals can continue to survive in the SFBA.

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ADDENDUM

Surveys for nesting Burrowing Owls in 2006 on the EBRPD's Souza and Vasco Caves parcels in the Altamont Hills in eastern Contra Costa County recorded 25 nesting pairs of owls in the 604 ha area surveyed (Albion Environmental, Inc. 2006a). Thirty-one nesting pairs were recorded during similar surveys in approximately 600 ha on the nearby Department of Water Resources proposed Dyer reservoir and Brushy Creek pipeline in eastern Alameda County (Albion Environmental, Inc. 2006b). Results of these surveys suggest nesting Burrowing Owls may be relatively abundant in some areas of the Altamont Hills.

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Appendix W

State Fish and Game Staff Report Regarding Mitigation for Impacts to Swainson's Hawks in the Central Valley of California

Staff Report regarding Mitigation
for Impacts to Swainson's Hawks *(Buteo swainsoni)*
in the Central Valley of California

INTRODUCTION

The Legislature and the Fish and Game Commission have developed the policies, standards and regulatory mandates which, if implemented, are intended to help stabilize and reverse dramatic population declines of threatened and endangered species In order to determine how the Department of Fish and Game (Department) could judge the adequacy of mitigation measures designed to offset impacts to Swainson's hawks in the Central Valley, Staff (WMD, ESD and Regions) has prepared this report. To ensure compliance with legislative and Commission policy, mitigation requirements which are consistent with this report should be incorporated into: (1) Department comments to Lead Agencies and project sponsors pursuant to the California Environmental Quality Act (CEQA); (2) Fish and Game Code Section 2081 Management Authorizations (Management Authorizations); and (3) Fish and Game Code Section 2090 Consultations with State CEQA Lead Agencies.

The report is designed to provide the Department (including regional offices and divisions), CEQA Lead Agencies and project proponents the context in which the Environmental Services Division (ESD) will review proposed project specific mitigation measures. This report also includes "model" mitigation measures which have been judged to be consistent with policies, standards and legal mandates of the Legislature and Fish and Game Commission. Alternative mitigation measures, tailored to specific projects, may be developed if consistent with this report. Implementation of mitigation measures consistent with this report are intended to help achieve the conservation goals for the Swainson's hawk and should complement multi-species habitat conservation planning efforts currently underway.

The Department is preparing a recovery plan for the species and it is anticipated that this report will be revised to incorporate recovery plan goals. It is anticipated that the recovery plan will be completed by the end of 1995. The Swainson's hawk recovery plan will establish criteria for species recovery through preservation of existing habitat, population expansion into former habitat, recruitment of young into the population, and other specific recovery efforts.

During project review the Department should consider whether a proposed project will adversely affect suitable foraging habitat within a ten (10) mile radius of an active (used during one or more of the last 5 years) Swainson's hawk nest(s). Suitable Swainson's hawk foraging habitat will be those habitats and crops identified in Bechard (1983), Bloom (1980), and Estep (1989). The following vegetation types/agricultural crops are considered small mammal and insect foraging habitat for Swainson's hawks:

- alfalfa
- fallow fields
- beet, tomato, and other low-growing row or field crops
- dry-land and irrigated pasture
- rice land (when not flooded)
- cereal grain crops (including corn after harvest)

The ten mile radius standard is the flight distance between active (and successful) nest sites and suitable foraging habitats, as documented in telemetry studies (Estep 1989, Babcock 1993). Based on the ten mile radius, new development projects which adversely modify nesting and/or foraging habitat should mitigate the project's impacts to the species. The ten mile foraging radius recognizes a need to strike a balance between the biological needs of reproducing pairs (including eggs and nestlings) and the economic benefit of development(s) consistent with Fish and Game Code Section 2053.

Since over 95% of Swainson's hawk nests occur on private land, the Department's mitigation program should include incentives that preserve agricultural lands used for the production of crops, which are compatible with Swainson's hawk foraging needs, while providing an opportunity for urban development and other changes in land use adjacent to existing urban areas.

LEGAL STATUS

Federal

The Swainson's hawk is a migratory bird species protected under the Migratory Bird Treaty Act (MBTA) of 1918 (16 U.S.C. 703-711). The MBTA makes it unlawful to take, possess, buy, sell, purchase, or barter any migratory bird listed in Section 50 of the Code of Federal Regulations (C.F.R.) Part 10, including feathers or other parts, nests, eggs or products, except as allowed by implementing regulations (50 C.F.R. 21).

State

The Swainson's hawk has been listed as a threatened species by the California Fish and Game Commission pursuant to the California Endangered Species Act (CESA), see Title 14, California Code of Regulations, Section 670.5(b)(5)(A).

LEGISLATIVE AND COMMISSION POLICIES, LEGAL MANDATES AND STANDARDS

The FGC policy for threatened species is, in part, to: "Protect and preserve all native species…and their habitats ...." This policy also directs the Department to work with all interested persons to protect and preserve sensitive resources and their habitats. Consistent with this policy and direction, the Department is enjoined to implement measures that assure protection for the Swainson's hawk.

The California State Legislature, when enacting the provisions of CESA, made the following findings and declarations in Fish and Game Code Section 2051:

> a) "Certain species of fish, wildlife, and plants have been rendered extinct as a consequence of man's activities, untempered by adequate concern and conservation";
>
> b) "Other species of fish, wildlife, and plants are in danger of, or threatened with, extinction because their habitats are threatened with destruction, adverse modification, or severe curtailment because of overexploitation, disease, predation, or other factors (emphasis added)"; and
>
> c) "These species of fish, wildlife, and plants are of ecological, educational, historical, recreational, esthetic, economic, and scientific value to the people of this state, and the conservation, protection, and enhancement of these species and their habitat is of statewide concern" (emphasis added).

The Legislature also proclaimed that it "is the policy of the state to conserve, protect, restore, and enhance any endangered or threatened species and its habitat and that it is the intent of the Legislature, consistent with conserving the species, to acquire lands for habitat for these species" (emphasis added).

Section 2053 of the Fish and Game Code states, in part, "it is the policy of the state that state agencies should not approve projects as proposed which would jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of habitat essential to the continued existence of those species, if there are reasonable and prudent alternatives available consistent with conserving the species and/or its habitat which would prevent jeopardy" (emphasis added).

Section 2054 states "The Legislature further finds and declares that, in the event specific economic, social, and or other conditions make infeasible such alternatives, individual projects may be approved if appropriate mitigation and enhancement measures are provided" (emphasis added).

Loss or alteration of foraging, habitat or nest site disturbance which results in:

(1) nest abandonment; (2) loss of young; (3) reduced health and vigor of eggs and/or nestlings (resulting in reduced survival rates), may ultimately result in the take (killing) of nestling or fledgling Swainson's hawks incidental to otherwise lawful activities. The taking of Swainson's hawks in this manner can be a violation of Section 2080 of the Fish and Game Code. This interpretation of take has been judicially affirmed by the landmark appellate court decision pertaining to CESA (DFG v. ACID, 8 CA App. 4, 41554). The essence of the decision emphasized that the intent and purpose of CESA applies to all activities that take or kill endangered or threatened species, even when the taking is incidental to otherwise legal activities. To avoid potential violations of Fish and Game Code Section 2080, the Department recommends and encourages project sponsors to obtain 2081 Management Authorizations for their projects.

Although this report has been prepared to assist the Department in working with the development community, the prohibition against take (Fish and Game Code Section 2080) applies to all persons, including those engaged in agricultural activities and routine maintenance of facilities. In addition, sections 3503, 3503.5, and 3800 of the Fish and Game Code prohibit the take, possession, or destruction of birds, their nests or eggs.

To avoid potential violation of Fish and Game Code Section 2080 (i.e. killing of a listed species), project-related disturbance at active Swainson's hawk nesting sites should be reduced or eliminated during critical phases of the nesting cycle (March 1 – September 15 annually). Delineation of specific activities which could cause nest abandonment (take) of Swainson's hawk during the nesting period should be done on a case-by-case basis.

CEQA requires a mandatory findings of significance if a project's impacts to threatened or endangered species are likely to occur (Sections 21001 {c}, 21083, Guidelines Sections 15380, 15064, 15065). Impacts must be avoided or mitigated to less than significant levels unless the CEQA Lead Agency makes and supports findings of Overriding Consideration. The CEQA Lead Agency's Findings of Overriding Consideration does not eliminate the project sponsor's obligation to comply with Fish and Game Code Section 2080.

NATURAL HISTORY

The Swainson's hawk (*Buteo swainsoni*) is a large, broad winged buteo which frequents open country. They are about the same size as a red-tailed hawk (*Buteo jamaicensis*), but trimmer, weighing approximately 800-1100 grams (1.75 - 2 lbs). They have about a 125 cm. (4+foot) wingspan. The basic body plumage may be highly variable and is characterized by several color morphs - light, dark, and rufous. In dark phase birds, the entire body of the bird may be sooty black. Adult birds generally have dark backs. The ventral or underneath sections may be light with a characteristic dark, wide "bib from the lower throat down, to the upper breast, light colored wing linings and pointed wing tips. The tail is gray ventrally with a subterminal dusky band, and narrow, less conspicuous barring proximally. The sexes are similar in appearance; females however, are slightly larger and heavier than males, as is the case in most sexually dimorphic raptors, There are no recognized subspecies (Patmer 1988).

The Swainson's hawk is a long distance migrator. The nesting grounds occur in northwestern Canada, the western U.S., and Mexico and most populations migrate to wintering grounds in the open pampas and agricultural areas of South America (Argentina, Uruguay, southern Brazil). The species is included among the group of birds known as "neotropical migrants". Some individuals or small groups (20-30 birds) may winter in the U.S., including California (Delta Islands). This round trip journey may exceed 14,000 miles. The birds return to the nesting grounds and establish nesting territories in early March.

Swainson's hawks are monogamous and remain so until the loss of a mate (Palmer 1988). Nest construction and courtship continues through April. The clutch (commonly 3-4 eggs) is generally laid in early April to early May, but may occur later. Incubation lasts 34-35 days, with both parents participating in the brooding of eggs and young. The young fledge (leave the nest) approximately 42-44 days after hatching and remain with their parents until they depart in the fall. Large groups (up to 100+ birds) may congregate in holding areas in the fall and may exhibit a delayed migration depending upon forage availability. The specific purpose of these congregation areas is as yet unknown, but is likely related to: increasing energy reserves for migration; the timing of migration; aggregation into larger migratory groups (including assisting the young in learning migration routes); and providing a pairing and courtship opportunity for unattached adults.

Foraging Requirements

Swainson's hawk nests in the Central Valley of California are generally found in scattered trees or along riparian systems adjacent to agricultural fields or pastures. These open fields and pastures are the primary foraging areas. Major prey items for Central Valley birds include: California voles *(Microtus californicus),* valley pocket gophers (*Thomomys bottae),* deer mice (*Peromyscus maniculafus),* California ground squirrels *(Spermophilus beecheyi),* mourning doves *(Zenaida macroura),* ring-necked pheasants *(Phasianus colchicus),* meadowlarks (*Sturnella neglecta),* other passerines, grasshoppers *(Conocephalinae* sp.*),* crickets *(Gryllidae* sp.*),* and beetles (Estep 1989). Swainson's hawks generally search for prey by soaring in open country and agricultural fields similar to *northern hariers (Circus cyaneus)* and ferruginous hawks *(Bufeo regalis).* Often several hawks may be seen foraging together following tractors or other farm equipment capturing prey escaping from farming operations. During the breeding season, Swainson's hawks eat mainly vertebrates (small rodents and reptiles), whereas-during migration vast numbers of insects are consumed (Palmer 1988).

Department funded research has documented the importance of suitable foraging habitats (e.g., annual grasslands, pasture lands, alfalfa and other hay crops, and combinations of hay, grain and row crops) within an energetically efficient flight distance from active Swainson's hawk nests (Estep pers. comm.). Recent telemetry studies to determine foraging requirements have shown that birds may use in excess of 15,000 acres of habitat or range up to 18.0 miles from the nest in search of prey (Estep 1989, Babcock 1993). The prey base (availability and abundance) for the species is highly variable from year to year, with major prey population (small mammals and insects) fluctuations occurring based on rainfall patterns, natural cycles and agricultural cropping and harvesting patterns. Based on these variables, significant acreages of potential foraging habitat (primarily agricultural lands) should be preserved per nesting pair (or aggregation of nesting pairs) to avoid jeopardizing existing populations. Preserved foraging areas should be adequate to allow additional Swainson's hawk nesting pairs to successfully breed and use the foraging habitat during good prey production years.

Suitable foraging habitat is necessary to provide an adequate energy source for breeding adults, including support of nestlings and fledglings. Adults must achieve an energy balance between the needs of themselves and the demands of nestlings and fledglings, or the health and survival of both may be jeopardized. If prey resources are not sufficient, or if adults must hunt long distances from the nest site, the energetics of the foraging effort may result in reduced nestling vigor with an increased likelihood of disease and/or starvation. In more extreme cases, the breeding pair, in an effort to assure their own existence, may even abandon the nest and young (Woodbridge 1985).

Prey abundance and availability is determined by land and farming patterns including crop types, agricultural practices and harvesting regimes. Estep (1989) found that 73.4% of observed prey captures were in fields being harvested, disced, mowed, or irrigated. Preferred foraging habitats for Swainson’s hawks include:

- alfalfa;
- fallow fields;
- beet, tomato, and other low-growing row or field crops;
- dry-land and irrigated pasture;
- rice land (during the non-flooded period); and
- cereal grain crops (including corn after harvest).

Unsuitable foraging habitat types include crops where prey species (even if present) are not available due to vegetation characteristics (e.g. vineyards, mature orchards, and cotton fields, dense vegetation).

Nesting Requirements

Although the Swainson's hawk's current nesting habitat is fragmented and unevenly distributed, Swainson's hawks nest throughout most of the Central Valley floor. More than 85% of the known nests in the Central Valley are within riparian systems in Sacramento, Sutter, Yolo, and San Joaquin counties. Much of the potential nesting habitat remaining in this area is in riparian forests, although isolated and roadside trees are also used. Nest sites are generally adjacent to or within easy flying distance to alfalfa or hay fields or other habitats or agricultural crops which provide an abundant and available prey source. Department research has shown that valley oaks (*Quercus lobata*), Fremont's cottonwood (*Populus fremontii),* willows *(Salix* spp*.),* sycamores (*Platanus* spp.), and walnuts (*Juglans* spp.) are the preferred nest trees for Swainson's hawks (Bloom 1980, Schlorff and Bloom 1983, Estep 1989).

Fall and Winter Migration Habitats

During their annual fall and winter migration periods, Swainson's hawks may congregate in large groups (up to 100+ birds). Some of these sites may be used during delayed migration periods lasting up to three months. Such sites have been identified in Yolo, Tulare, Kern and San Joaquin counties and protection is needed for these critical foraging areas which support birds during their long migration.

Historical and Current Population Status

The Swainson's hawk was historically regarded as one of the most common and numerous raptor species in the state, so much so that they were often not given special mention in field notes. The breeding population has declined by an estimated 91% in California since the turn of the century (Bloom 1980). The historical Swainson's hawk population estimates are based on current densities and extrapolated based on the historical amount of available habitat. The historical population estimate is 4,284-17,136 pairs (Bloom 1980). In 1979, approximately 375 (± 50) breeding pairs of Swainson's hawks were estimated in California, and 280 (75%) of those pairs were estimated to be in the Central Valley (Bloom 1980). In 1988, 241 active breeding pairs were found in the Central Valley, with an additional 78 active pairs known in northeastern California. The 1989 population estimate was 430 pairs for the Central Valley and 550 pairs statewide (Estep, 1989). This difference in population estimates is probably a result of increased survey effort rather than an actual population increase.

Reasons for decline

The dramatic Swainson's hawk population decline has been attributed to loss of native nesting and foraging habitat, and more recently to the loss of suitable nesting trees and the conversion of agricultural lands. Agricultural lands have been converted to urban land uses and incompatible crops. In addition, pesticides, shooting, disturbance at the nest site, and impacts on wintering areas may have contributed to their decline. Although losses on the wintering areas in South America may occur, they are not considered significant since breeding populations outside of California are stable. The loss of nesting habitat within riparian areas has been accelerated by flood control practices and bank stabilization programs. Smith (1977) estimated that in 1850 over 770,000 acres of riparian habitat were present in the Sacramento Valley. By the mid-1980s, Warner and Hendrix (1984) estimated that there was only 120,000 acres of riparian habitat remaining in the Central Valley (Sacramento and San Joaquin Valleys combined). Based on Warner and Hendrix's estimates approximately 93% of the San Joaquin Valley and 73% of the Sacramento Valley riparian habitat has been eliminated since 1850.

MANAGEMENT STRATEGIES

Management and mitigation strategies for the Central Valley population of the Swainson's hawk should ensure that:

- suitable nesting habitat continues to be available (this can be accomplished by protecting existing nesting habitat from destruction or disturbance and by increasing the number of suitable nest trees); and

- foraging habitat is available during the period of the year when Swainson's hawks are present in the Central Valley (this should be accomplished by maintaining or creating adequate and suitable foraging habitat in areas of existing and potential nest sites and along migratory routes within the state).

A key to the ultimate success in meeting the Legislature's goal of maintaining habitat sufficient to preserve this species is the implementation of these management strategies in cooperation with project sponsors and local, state and federal agencies.

DEPARTMENT'S ROLES AND RESPONSIBILITIES IN PROJECT CONSULTATION AND ADMINISTRATION OF CEQA AND THE FISH AND GAME CODE

The Department, through its administration of the Fish and Game Code and its trust responsibilities, should continue its efforts to minimize further habitat destruction and should seek mitigation to offset unavoidable losses by (1) including the mitigation measures in this document in CEQA comment letters and/or as management conditions in Department issued Management Authorizations or (2) by developing project specific mitigation measures (consistent with the Commission's and the Legislature's mandates) and including them in CEQA comment letters and/or as management conditions in Fish and Game Code Section 2081 Management Authorizations issued by the Department and/or in Fish and Game Code Section 2090 Biological Opinions.

The Department should submit comments to CEQA Lead Agencies on all projects which adversely affect Swainson's hawks. CEQA requires a mandatory findings of significance if a project's impacts to threatened or endangered species are likely to occur (Sections 21001 {c} 21083. Guidelines 15380, 15064, 15065). Impacts must be: (1) avoided; or (2) appropriate mitigation must be provided to reduce impacts to less than significant levels; or (3) the lead agency must make and support findings of overriding consideration. If the CEQA Lead Agency makes a Finding of Overriding Consideration, it does not eliminate the project sponsor's obligation to comply with the take prohibitions of Fish and Game Code Section 2080. Activities which result in (1) nest abandonment; (2) starvation of young; and/or (3) reduced health and vigor of eggs and nestlings may result in the take (killing) of Swainson's hawks incidental to otherwise lawful activities (urban development, recreational activities, agricultural practices, levee maintenance and similar activities). The taking of Swainson's hawk in this manner may be a violation of Section 2080 of the Fish and Game Code. To avoid potential violations of Fish and Game Code Section 2080, the Department should recommend and encourage project sponsors to obtain 2081 Management Authorizations.

In aggregate, the mitigation measures incorporated into CEQA comment letters and/or 2081 Management Authorizations for a project should be consistent with Section 2053 and 2054 of the Fish and Game Code. Section 2053 states, in part, "it is the policy of the state that state agencies should not approve projects as proposed which would jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of habitat essential to the continued existence of those species, if there are reasonable and prudent alternatives available consistent with conserving the species and or its habitat which would prevent jeopardy". Section 2054 states: "The Legislature further finds and declares that, in the event specific economic, social, and/or other conditions make infeasible such alternatives, individual projects may be approved if appropriate mitigation and enhancement measures are provided."

State lead agencies are required to consult with the Department pursuant to Fish and Game Code Section 2090 to ensure that any action authorized, funded, or carried out by that state agency will not jeopardize the continued existence of any threatened or endangered species. Comment letters to State Lead Agencies should also include a reminder that the State Lead Agency has the responsibility to consult with the Department pursuant to Fish and Game Code Section 2090 and obtain a written findings (Biological Opinion). Mitigation measures included in Biological Opinions issued to State Lead Agencies must be consistent with Fish and Game Code Sections 2051-2054 and 2091-2092.

NEST SITE AND HABITAT LOCATION INFORMATION SOURCES

The Department's Natural Diversity Data Base (NDDB) is a continually updated, computerized inventory of location information on the State's rarest plants, animals, and natural communities. Department personnel should encourage project proponents and CEQA Lead Agencies, either directly or through CEQA comment letters, to purchase NDDB products for information on the locations of Swainson's hawk nesting areas as well as other sensitive species. The Department's Nongame Bird and Mammal Program also maintains information on Swainson's hawk nesting areas and may be contacted for additional information on the species.

Project applicants and CEQA Lead Agencies may also need to conduct site specific surveys (conducted by qualified biologists at the appropriate time of the year using approved protocols) to determine the status (location of nest sites, foraging areas, etc.) of listed species as part of the CEQA and 2081 Management Authorization process. Since these studies may require multiple years to complete, the Department shall identify any needed studies at the earliest possible time in the project review process. To facilitate project review and reduce the potential for costly project delays, the Department should make it a standard practice to advise developers or others planning projects that may impact one or more Swainson's hawk nesting or foraging areas to initiate communication with the Department as early as possible .

MANAGEMENT CONDITIONS

Staff believes the following mitigation measures (nos. 1-4) are adequate to meet the Commission's and Legislature's policy regarding listed species and are considered as preapproved for incorporation into any Management Authorizations for the Swainson's hawk issued by the Department. The incorporation of measures 1-4 into a CEQA document should reduce a project's impact to a Swainson's hawk(s) to less than significant levels. Since these measures are Staff recommendations, a project sponsor or CEQA Lead agency may choose to negotiate project specific mitigation measures which differ. In such cases, the negotiated Management Conditions must be consistent with Commission and Legislative policy and be submitted to the ESD for review and approval prior to reaching agreement with the project sponsor or CEQA Lead Agency.

Staff recommended Management Conditions are:

1. No intensive new disturbances (e.g. heavy equipment operation associated with construction, use of cranes or draglines, new rock crushing

activities) or other project related activities which may cause nest abandonment or forced fledging, should be initiated within 1/4 mile (buffer zone) of an active nest between March 1 - September 15 or until August 15 if a Management Authorization or Biological Opinion is obtained for the project. The buffer zone should be increased to 1/2 mile in nesting areas away from urban development (i.e. in areas where disturbance [e.g. heavy equipment operation associated with construction, use of cranes or draglines, new rock crushing activities] is not a normal occurrence during the nesting season). Nest trees should not be removed unless there is no feasible way of avoiding it. If a nest tree must be removed, a Management Authorization (including conditions to off-set the loss of the nest tree) must be obtained with the tree removal period specified in the Management Authorization, generally between October 1- February 1. If construction or other project related activities which may cause nest abandonment or forced fledging are necessary within the buffer zone, monitoring of the nest site (funded by the project sponsor) by a qualified biologist (to determine if the nest is abandoned) should be required. If it is abandoned and if the nestlings are still alive, the project sponsor shall fund the recovery and hacking (controlled release of captive reared young) of the nestling(s). Routine disturbances such as agricultural activities, commuter traffic, and routine facility maintenance activities within 1 /4 mile of an active nest should not be prohibited.

2. Hacking as a substitute for avoidance of impacts during the nesting period may be used in unusual circumstances after review and approval of a hacking plan by ESD and WMD. Proponents who propose using hacking will be required to fund the full costs of the effort, including any telemetry work specified by the Department.

3. To mitigate for the loss of foraging habitat (as specified in this document), the Management Authorization holder/project sponsor shall provide Habitat Management (HM) lands to the Department based on the following ratios:

(a) Projects within 1 mile of an active nest tree shall provide:

- <u>one acre of HM land</u> (at least 10% of the HM land requirements shall be met by fee title acquisition or a conservation easement allowing for the active management of the habitat, with the remaining 90% of the HM lands protected by a conservation easement [acceptable to the Department] on agricultural lands or other suitable habitats which provide foraging habitat for Swainson's hawk) <u>for each acre of development authorized (1:1 ratio);</u> or

- one-half acre of HM land (all of the HM land requirements shall be met by fee title acquisition or a conservation easement [acceptable to the Department] which allows for the active management of the habitat for prey production on the HM lands) for each acre of development authorized (0.5:1 ratio).

(b) Projects within 5 miles of an active nest tree but greater than 1 mile from the nest tree shall provide 0.75 acres of HM land for each acre of urban development authorized (0.75:1 ratio). All HM lands protected under this requirement may be protected through fee title acquisition or conservation easement (acceptable to the Department) on agricultural lands or other suitable habitats which provide foraging habitat for Swainson's hawk.

(c) Projects within 10 miles of an active nest tree but greater than 5 miles from an active nest tree shall provide 0.5 acres of HM land for each acre of urban development authorized (0.5:1 ratio). All HM lands protected under this requirement may be protected through fee title acquisition or a conservation easement (acceptable to the Department) on agricultural lands or other suitable habitats which provide foraging habitat for Swainson's hawk.

4. Management Authorization holders/project sponsors shall provide for the long-term management of the HM lands by funding a management endowment (the interest on which shall be used for managing the HM lands) at the rate of $400 per HM land acre (adjusted annually for inflation and varying interest rates).

Some project sponsors may desire to provide funds to the Department for HM land protection. This option is acceptable to the extent the proposal is consistent with Department policy regarding acceptance of funds for land acquisition. All HM lands should be located in areas which are consistent with a multi-species habitat conservation focus. Management Authorization holders/project sponsors who are willing to establish a significant mitigation bank (> 900 acres) should be given special consideration such as 1.1 acres of mitigation credit for each acre preserved.

PROJECT SPECIFIC MITIGATION MEASURES

Although this report includes recommended Management Measures, the Department should encourage project proponents to propose alternative mitigation strategies that provide equal or greater protection of the species and which also expedite project environmental review or issuance of a CESA Management Authorization. The Department and sponsor may choose to conduct cooperative, mufti-year field studies to assess the site's habitat value and determine its use by nesting and foraging

Swainson's hawk. Study plans should include clearly defined criteria for judging the 'project's -impacts on Swainson's hawks and the methodologies (days of monitoring, foraging effort/ efficiency, etc.) that will be used.

The study plans should be submitted to the Wildlife Management Division and ESD for review. Mitigation measures developed as a result of the study must be reviewed by ESD (for consistency with the policies of the Legislature and Fish and Game Commission) and approved by the Director.

EXCEPTIONS

Cities, counties and project sponsors should be encouraged to focus development on open lands within already urbanized areas. Since small disjunct parcels of habitat seldom provide foraging habitat needed to sustain the reproductive effort of a Swainson's hawk pair, Staff does not recommend requiring mitigation pursuant to CEQA nor a Management Authorization by the Department for infill (within an already urbanized area) projects in areas which have less than 5 acres of foraging habitat and are surrounded by existing urban development, unless the project area is within 1/4 mile of an active nest tree.

REVIEW

Staff should revise this report at least annually to determine if the proposed mitigation strategies should be retained, modified or if additional mitigation strategies should be included as a result of new scientific information.

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ASSESSING CHANGES IN THE DISTRIBUTION AND ABUNDANCE OF BURROWING OWLS IN CALIFORNIA, 1993-2007[1]

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Abstract. The Western Burrowing Owl (*Athene cunicularia hypugaea*) has declined in recent decades across much of its range, including California, where it is classified as a Species of Special Concern. During 2006-2007, we surveyed the entire breeding range of the species in California, except the Channel Islands. Relying largely on volunteers, we surveyed 860 5km x 5km blocks, and documented exact locations of 1,758 pairs. Using data from randomly-selected blocks, we extrapolated a statewide, breeding-season population of 9,187 (SE = 2,346) pairs. For all of the species' California range, except the Modoc Plateau and the Mojave and Sonoran deserts, we compared results with those of DeSante et al. (2007) using identical methods and study area boundaries during 1991-1993. Our 2006-2007 estimate of 8,128 (SE = 2,391) pairs was 10.9% lower than the previous estimate, but the difference was not statistically significant. The major patterns of Burrowing Owl occurrence across California appeared to be relatively unchanged since 1993, although non-significant declines were apparent in numerous regions. Burrowing Owls appear to have declined particularly sharply in two urban areas: the San Francisco Bay Area and Bakersfield. Our surveys of previously unsurveyed portions of the species' California range yielded few or no owls in the Modoc Plateau/Great Basin, Northern Mojave/eastern Sierra Nevada, eastern Mojave, and Sonoran Desert regions (excluding the Palo Verde Valley) but detected large aggregations in the Palo Verde Valley and the western Mojave Desert region.

Key words: Burrowing Owl, California, Athene cunicularia, *citizen science*

EVALUAR LOS CAMBIOS EN LA DISTRIBUCIÓN Y ABUNDANCIA DEL BÚHO LLANERO EN CALIFORNIA, 1993-2007

Resumen. El Búho Llanero (*Athene cunicularia hypugaea*) ha disminuido en las últimas décadas en gran parte de su distribución, incluyendo a California, donde está clasificado como una especie de interés especial. Durante el periodo 2006-2007, encuestamos a todo el rango reproductivo de la especie en California, con la excepción de las Channel Islands.

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Dependimos en gran medida de voluntarios para contar búhos en 860 bloques de 5 km x 5 km, y para documentar la ubicación exacta de 1,758 parejas. Usando datos de los bloques seleccionados al azar, extrapolamos una población para la temporada de reproducción en todo el estado de 9,187 (SE = 2,346) pares. En toda la distribución de la especie de California, con excepción de la Modoc Plateau y los Mojave y Sonoran Deserts, se compararon los resultados con los de DeSante et al. (2007) utilizando métodos idénticos y los límites del área de estudio durante 1991-1993. Nuestra 2006-2007 estimación de 8,128 (SE = 2,391) pares fue 10.9% inferior a la estimación anterior, pero la diferencia no fue estadísticamente significativa. Los principales patrones de ocurrencia del Búho Llanero a través de California parece no haberse cambiado desde 1993. Aunque hubo disminuciones evidentes en numerosas regiones, éstas no fueron estadisticamente significativas. La disminución del Búho Llanero fue especialmente marcada en dos áreas urbanas: el San Francisco Bay Area y Bakersfield. Nuestro estudio de las regiones préviamente no investigadas de distribución de la especie de California dio pocos o ningunos búhos en la Modoc Plateau/Great Basin, el norte de Mojave/este de Sierra Nevada, el este de Mojave, y regiones del Sonoran Desert (excluyendo el Palo Verde Valley), pero detectó grande agregaciones en el Palo Verde Valley y la región occidental del Mojave Desert.

Palabras clave: Búho Llanero, California, *Athene cunicularia*, ciencia ciudadana

INTRODUCTION

Burrowing Owls (*Athene cunicularia hypugaea*) in California breed in natural grasslands and a variety of human-modified habitats, including areas of intense agriculture (Cuolombe 1971, DeSante et al. 2004), as well as airports (Thomsen 1971, Barclay 2007b) and other open areas in urban environments (Trulio 1997, Trulio and Chromczak 2007). Once considered "abundant" and "common" throughout California (Baird 1870, Keeler 1891, Grinnell 1915, Dawson 1923), the species has been declining since at least the 1940s (Grinnell and Miller 1944, Remsen 1978, James and Ethier 1989, DeSante et al. 2007) and is now classified as a Species of Special Concern (Gervais et al. 2008, Shuford and Gardali 2008). The species has declined throughout much of its range (Wedgwood 1978, James and Ethier 1989, Sheffield 1997a, Holroyd et al. 2001, Wellicome and Holroyd 2001) with suggested causes including conversion of grassland habitats to urbanization and inhospitable forms of agriculture (DeSante et al. 2007, Gervais et al. 2008), eradication of fossorial mammals (Zarn 1974, Remsen 1978, Holroyd et al. 2001) and perhaps exposure to pesticides and other contaminants (James and Fox 1987, Haug et al. 1993, Sheffield 1997b; but see also Gervais and Anthony 2003).

In the early 1990s, DeSante et al. (2007) coordinated a survey of the species' entire California breeding range, except for the Modoc Plateau/Great Basin region and the Mojave and Sonoran deserts. At that time Burrowing Owl populations in the southern San Francisco Bay region and in the northern and central portions of the Central Valley appeared to have been declining rapidly, and populations elsewhere in the census area, including the coastal slope of central and southern California, had virtually disappeared. DeSante et al. (2007) estimated that the entire survey area contained >9,000 pairs, with 71% of the estimated population occupying the Imperial Valley south of the Salton Sea (an area comprising just 2.5% of the state) and 24% occupying the Central Valley, primarily in the southern portion of the San Joaquin Valley. Prior to the present study, adequate information to assess Burrowing Owl population trends since 1993 was not available, and in the Great Basin and Mojave and Sonoran desert regions no systematic assessment of population size had ever been made.

Local-scale demographic studies of four focal populations (Imperial Valley, Carrizo Plain, Naval Air Station Lemoore, and the San Jose area) suggest highly variable demographic rates (Gervais 2002, Ronan 2002, Gervais and Anthony 2003, Rosenberg and Haley 2004). Breeding Bird Survey (Sauer et al. 2008) trend results for California exist but are difficult to interpret, because the great majority of detections are clustered on a small number of routes in the Imperial Valley, home to one of the largest concentrations of the species anywhere (DeSante et al. 2007).

We undertook the present study to assess how Burrowing Owl distribution and abundance in California may have changed since 1993, and to determine the status of Burrowing Owl populations in the previously unsurveyed Modoc Plateau and desert regions of the state.

METHODS

STUDY AREA

For their 1991-1993 study, DeSante et al. (2007) defined and surveyed 11 distinct geographic regions, comprising the entire California breeding range, except for the Sonoran and Mojave deserts and the Modoc Plateau. To maximize comparability, we retained all of the region boundaries established by DeSante et al. (2007; Fig. 1).

We re-surveyed 8 of the 11 regions defined for the early 1990s survey (Table 1); because populations in the San Francisco Bay Area Coast, Central-western Coast, and Southwestern Coast regions were well studied and known to be very small or extirpated entirely, we opted not to devote volunteer resources to surveying those, but instead to rely on published literature and/or local experts for population estimates.

In addition to resurveying most of the DeSante et al. (2007) regions, we also targeted the state's Modoc/Great Basin and desert, in order to assess the species' heretofore largely unknown abundance and distribution within these areas, and to better understand their relative importance to the state's overall Burrowing Owl population. DeSante et al. (2007) omitted these areas from the 1991-1993 survey because adequate numbers of local volunteer surveyors were not available. We were able to include these regions in the 2006-2007 effort by surveying them with a crew of full-time field technicians, rather than relying on local volunteers.

We divided the previously unsurveyed portions of the California breeding range into five new regions, four of which are described in greater detail in Wilkerson and Siegel (*in press;* Fig. 1): Northern Mojave Desert/Eastern Sierra Nevada, Western Mojave Desert, Eastern Mojave Desert, Sonoran Desert. The fifth, the Modoc Plateau/Great Basin region (Fig. 1), matches the geographic boundaries of the "Jepson area" mapped as "Modoc Plateau" by Hickman (1993) and the California Gap Analysis Project (1998). The region lies entirely above the 610m elevation contour, which was used as the upper limit for high elevation subregions in ten of the 12 regions defined by DeSante et al. (2007). We therefore did not stratify our sampling within this region by elevation. Rather, we classified the entire region as "upland". Because of the presence of large tracts of forested areas that are not suitable Burrowing Owl habitat, we used the Forest Multi-source Landcover Data (California Department of Forestry and Fire Protection 2002) in conjunction with Burrowing Owl habitat characterizations produced for the region by Cull and Hall (2007) to assess the extent of potential habitat within each survey block. All land area above 1,830m was excluded from the sample frame because it consists of mountainous and forested habitat. We classified the remaining survey blocks as having either greater than or less than 50% suitable Burrowing Owl habitat, and then drew our random sample of blocks such that 2/3 had >50% suitable habitat cover and 1/3 had <50% suitable habitat cover. Survey blocks with <5% suitable habitat cover were not included.

SURVEY DESIGN

Within each region previously surveyed by DeSante et al. (2007), we used the grid defined for their 1991-1993 survey, which divides all the land in the study area into 5-km by 5-km blocks, oriented and referenced according to the Universal Transverse Mercator (UTM) system. Each block was classified as belonging primarily to the lowland subregion or the upland subregion, using a set of classification rules that varied slightly by region (see details in DeSante et al. 2007). Survey effort was stratified by elevational subregion because Burrowing Owl densities are generally much higher in lowland areas throughout California than in upland areas (DeSante et al. 2007). For logistical reasons, we discarded the small number of blocks that could not be accessed anywhere by roads, and then stratified sampling effort among the remaining blocks by region and subregion, randomly selecting as many blocks as we

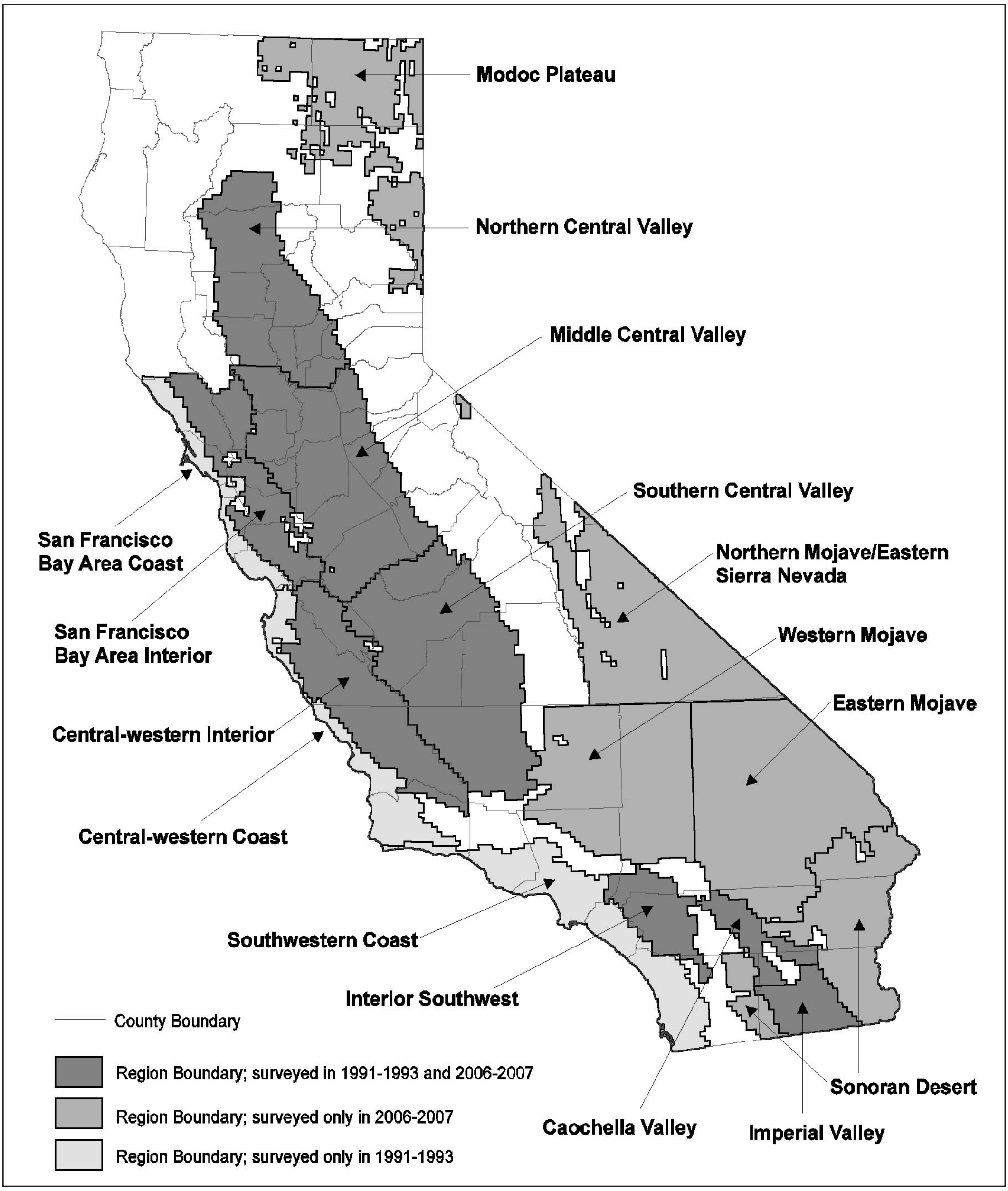

FIGURE 1. Burrowing Owl regions delineated and surveyed for The Institute for Bird Populations' 1991-1993 and/or 2006-2007 statewide Burrowing Owl surveys.

thought we would have the manpower to survey. Blocks in each region were then assigned to be surveyed in a randomly determined order to avoid bias if our volunteers and field crew were unable to survey all of the selected blocks.

We used Geographic Information System (GIS) software to define grids of 5-km by 5-km blocks covering each of the four new regions in a manner consistent with the previously established grid. The 1991-1993 survey drew from a sampling frame of 5,990 blocks (DeSante et al. 2007). The five new survey regions

TABLE 1. Regions of California defined and surveyed for The Institute for Bird Populations' California Burrowing Owl surveys during 1991-1993 and/or 2006-2007.

Region	Status during 2006-2007 survey
Regions surveyed during the 1991-1993 survey	
Northern Central Valley	Resurveyed
Middle Central Valley	Resurveyed
Southern Central Valley	Resurveyed
San Francisco Bay Area Interior	Resurveyed
San Francisco Bay Area Coast	Not resurveyed – population extirpated
Central-western Interior	Resurveyed
Central-western Coast	Not resurveyed – population likely extirpated
Southwestern Coast	Not resurveyed – small, well-known population
Southwestern Interior	Resurveyed
Coachella Valley	Resurveyed
Imperial Valley	Resurveyed
Regions not previously surveyed	
Modoc Plateau/Great Basin	Surveyed for the first time
Northern Mojave Desert/Eastern Sierra Nevada	Surveyed for the first time
Western Mojave Desert	Surveyed for the first time
Eastern Mojave Desert	Surveyed for the first time
Sonoran Desert	Surveyed for the first time

contained an additional 4,991 blocks. After removing those regions from the 1991-1993 survey we decided not to survey, our sampling frame contained a total of 9,823 blocks.

Random sample blocks were selected separately by region and elevation stratum. The selected number of blocks to be visited in each subregion was proportional to its size and amount of estimated surveyor effort available over the two-year survey period. Because Burrowing Owls are known to be more abundant in the lower elevation strata throughout our sample area (DeSante et al. 2007), low elevation blocks comprised 2/3 of the random sample selected to be visited while high elevation substrata blocks comprised 1/3 of selected random sample blocks in all survey regions.

We also identified additional blocks (hereafter, "historic breeding blocks") where Burrowing Owls were known to have been detected during the breeding season in any year since 1981. Historic breeding blocks were identified by querying or consulting the following sources for historical detections: the database compiled by DeSante et al. (2007), which includes Burrowing Owls detected during the 1991-1993 survey as well as historical detections gathered from multiple sources from the decade prior to that survey; the California Natural Diversity Database (CNDDB; California Dept. Fish and Game 2006); and knowledgeable researchers and birders with local expertise throughout the state.

Based on previous knowledge from the 1991-1993 survey, we estimated that it was feasible to visit approximately 670 blocks in the eight regions being resurveyed, and 230 blocks in the five new survey regions, for a total of 900 blocks. Prior to the 2006 field season, we identified 500 historic breeding blocks (459 historic breeding blocks in the eight regions surveyed in 1991-1993 and 41 historic breeding blocks in the five new survey regions); a few additional historic blocks were identified during the course of our two-year survey. We also selected 520 random blocks to be surveyed: 340 in regions scheduled to be resurveyed and 180 in the new regions, of which 47 also happened to be historic breeding blocks in which Burrowing Owls had been detected during the 1991-1993 survey. The total number of blocks drawn for surveying during 2006-2007 was 973 (slightly more than we thought we could survey, in case some selected blocks proved to be inaccessible or we were able to sample more blocks than we anticipated).

All selected blocks were assigned to a randomly generated order. In each subregion, half of all blocks in each elevation stratum and each category (random or historic) were

TABLE 2. Number of blocks surveyed, Burrowing Owl pairs found, and population estimate for each geographic region surveyed during 2006-2007 that was also surveyed in 1991-1993 by DeSante et al. (2007). For each region and elevational subregion, we considered our "best estimate" of the number of pairs in 2006-2007 to be the larger of a) the extrapolated estimate of pairs, based only on results from randomly-selected blocks, or b) the actual number of pairs counted, pooling data from randomly-selected blocks and historic breeding blocks.

		All blocks			Random blocks only				
Region	Total area of region (km2)	Random & historic breeding blocks surveyed	Square km surveyed	No of pairs found	Random blocks surveyed	Square km surveyed	No. of pairs found	Estimated no. of pairs (SE)	"Best estimate" of no. of pairs (SE)[a]
Northern Central Valley									
Lowland	10,900	37	822	12	22	497	0	0	12
Upland	8,975	11	252	0	11	252	0	0	0
All	19,875	48	1,074	12	33	749	0	0	12
Middle Central Valley									
Lowland	16,400	174	3,903	339	59	1,265	34	502 (209)	502 (209)
Upland	10,858	25	433	43	12	236	0	0	43
All	27,258	199	4,336	382	71	1,501	34	502 (209)	545 (209)
Southern Central Valley									
Lowland	18,650	121	2,902	204	63	1,544	72	968 (342)	968 (342)
Upland	13,025	43	714	32	18	323	3	145 (118)	145 (118)
All	31,675	164	3,616	236	81	1,867	75	1,113 (460)	1,113 (460)
San Francisco Bay Area Interior									
Lowland	4,903	69	1,592	98	20	447	0	0	98
Upland	6,275	21	515	14	12	290	1	21 (21)	21 (21)
All	11,178	90	2,107	112	32	737	1	21 (21)	119
Central-western Interior									
Lowland	5,325	20	308	8	17	276	0	0	8
Upland	11,225	24	477	13	13	233	2	76 (51)	76 (51)
All	16,550	44	785	21	30	509	2	76 (51)	84
Southwestern Interior									
Lowland	1,250	12	301	37	3	75	1	17 (17)	37
Upland	5,050	56	1,265	113	8	173	0	0	113
All	6,300	68	1,566	150	11	248	1	17 (17)	150
Coachella Valley									
Lowland	1,615	10	245	12	4	100	1	16 (16)	16 (16)
Upland	2,350	10	251	37	4	100	0	0	37
All	3,965	20	496	49	8	200	1	16 (16)	53

TABLE 2. Continued.

Region	Total area of region (km2)	All blocks			Random blocks only				"Best estimate" of no. of pairs (SE)[a]
		Random & historic breeding blocks surveyed	Square km surveyed	No of pairs found	Random blocks surveyed	Square km surveyed	No. of pairs found	Estimated no. of pairs (SE)	
Imperial Valley									
Lowland	2,810	12	301	499	5	126	254	5,701 (2,244)	5,701 (2,244)
Upland	1,780	3	53	22	2	50	17	707 (140)	707 (140)
All	4,590	15	354	521	7	176	271	6,408 (2,384)	6,408 (2,384)

[a] For each subregion we considered our "best estimate" of the number of Burrowing Owl pairs to be the larger of a) the extrapolated estimate of pairs, based only on results from randomly-selected blocks, or b) the actual number of pairs counted, pooling data from randomly-selected blocks and historic breeding blocks. We then summed the "best estimate" for each subregion to obtain "best estimates" for each region.

assigned to observers for sampling in year one (2006) based on the firing order. All random and historic blocks not sampled in 2006 were assigned to be surveyed in 2007. Most blocks were sampled during one of the two years in our survey period. In the few instances that a block was surveyed during both years (generally because volunteer observers became interested in "their" blocks during 2006 and independently chose to resurvey them in 2007), we used data from the first survey year (2006) in our analysis.

DATA COLLECTION

Adhering to the strategy developed by DeSante et al. (2007), we relied largely upon volunteer observers, many associated with local California Audubon Society chapters, to collect our field data in the regions that were surveyed during 1991-1993. We also deployed a crew of full-time field biologist technicians to a) survey some of the blocks in regions where the number of volunteer observers was inadequate to reasonably survey all the selected blocks, and b) survey all of the selected blocks in the Sonoran, Mojave, and Great Basin regions, where potential volunteers were very scarce.

Volunteer surveyors and IBP field crews surveyed blocks using the field methodology developed for the 1991-1993 survey (DeSante et al. 2007). For most regions, surveyors were instructed to visually scan all of the area in their blocks at least once during morning (dawn to 10:00 AM) or late-afternoon (4:00 PM to dusk) during the two-month period between May 15 and July 15, when breeding Burrowing Owls are likely to be feeding nestlings or recently-fledged young. The survey season was shifted two weeks earlier in the Western and Eastern Mojave Desert, Sonoran Desert, Southwestern Interior, and the Coachella and Imperial Valley regions to account for phenological differences among areas.

We provided surveyors with 1:24,000 scale topographic maps with survey block boundaries and clearly marked locations of any owls known or suspected to have bred anytime since 1981. Surveyors delineated the extent of appropriate habitat in their block, visually scanned all areas of appropriate habitat for owls, and plotted the locations of any detections on their maps. For each detection location, observers provided a count of all owls seen (identified to age and sex, if possible), an estimate of the number of breeding pairs present, and standardized habitat information. The latter included information on vegetation community type and structure, land use, distance to irrigation canals, local abundance of ground squirrels, and other variables. Finally, observers provided a detailed assessment of how much of their block they were actually able to survey adequately. In some cases this was <100%, due to private property restrictions or physiographic barriers.

For each region (except for the Modoc Plateau and desert regions where we relied strictly on IBP field crews) we recruited one or more local area coordinators, who helped recruit volunteers and coordinate their efforts. Prior to the start of the first field season, we developed a training presentation to explain the rationale and goals of the survey, provide tips for identifying Burrowing Owls and determining their age and sex, and teach volunteers how to conduct the survey and complete data forms in a standardized manner. We gave the presentation at eight live meetings and workshops, and also posted it as an online presentation on our website so that it was available to volunteers who could not attend a local training session. We also prepared a detailed data collection protocol which was provided to all observers prior to data collection.

STATISTICAL ANALYSES

We estimated the number of breeding pairs in each subregion and region surveyed. First we calculated the density of breeding pairs on each randomly-selected surveyed block, as the quotient of the number of pairs observed divided by the area of the block that was adequately surveyed. Densities were then averaged across all randomly-selected blocks surveyed in each subregion. Estimates are presented with standard errors, except in cases where the estimate was zero pairs and the SE could not be calculated.

For each subregion and region, we also totaled the actual number of pairs detected, as the sum of all pairs found on randomly-selected blocks plus all pairs found on historic breeding blocks. We present these totals without standard errors, since they are minimum counts rather than statistical estimates.

For each subregion, we considered our “best estimate” of the number of Burrowing Owl pairs to be the larger of a) the extrapolated estimate of pairs, based only on results from randomly-selected blocks, or b) the minimum number of pairs counted, pooling data from randomly-selected blocks and historic breeding blocks (in other words, we only used the minimum count as our “best estimate” if it was larger than the extrapolated estimate). We then summed the “best estimate” for each subregion to obtain “best estimates” of the number of pairs in each region, and across the state.

For subregions and regions surveyed in the 1990s, we compared the 2006-2007 population estimate (no. of pairs) with the estimate obtained for 1991-1993 by a) using Levene’s Test to determine whether variances for the two estimates were similar, and then b) using F-tests to assess statistical significance of differences between the estimates (Zar 1984). Such comparisons were only possible when our best estimate for the number of pairs in a region was derived from randomly-selected sample blocks; in cases where our “best estimate” was the actual number of pairs counted (aggregating results from randomly-selected blocks and historic breeding blocks) there was no variance associated with the estimate, so we provide only qualitative, rather than statistical, assessments of population change since the early 1990s.

We used ArcMap to determine land ownership (public versus private) or land managing agency (various federal agencies, state government, local government, tribal areas) at all occupied sites, based on the California Department of Fish and Game Region 6 Spatial Data Framework’s Public and Conservation Lands shapefile (“govconfee_1”).

We used a paired t-test to assess whether owl abundance changed between the 1991-1993 and 2006-2007 surveys for historic breeding blocks where owls were detected during 1991-1993. We used logistic regression to assess whether the probability of detecting owls on these blocks during 2006-2007 was related to the number of owl pairs detected on them during 1991-1993.

RESULTS

With the help of 21 local coordinators, we recruited 394 volunteers to participate in surveying one or more blocks during the 2006 or 2007 field season. These volunteers spent over 6,400 hr surveying blocks and completing data forms. Their efforts were augmented by our full-time crews of field biologist technicians, who largely focused their efforts in the new survey regions, where few volunteers were available, and in the southern Central Valley, where the large number of historic breeding blocks surpassed the survey capacity of the local pool of volunteers.

During our 2006-2007 efforts we were able to complete surveys at 453 of the 500 historic breeding owl blocks identified prior to the start of the 2006 field season; 47 historic breeding blocks thus went unsurveyed. However, 24 of those unsurveyed historic breeding blocks were surveyed but yielded no owl detections during the 1991-1993 survey. In other words, the occupancy records were from before 1991-1993, and occupancy could not be confirmed during the 1991-1993 survey. Thus, only 26 historic blocks known to have owls during the 1991-1993 survey went unsurveyed during 2006-2007.

We completed surveys of 860 blocks during 2006-2007. Of these, 444 were randomly selected, and 453 were historic breeding blocks (37 of which were also randomly selected and were treated as random blocks in our analysis). During the course of this survey, we documented the exact locations of 1,758 Burrowing Owl pairs, and have provided this information to the California Department of Fish and Game for their conservation planning purposes.

INDIVIDUAL REGIONS

NORTHERN CENTRAL VALLEY

We surveyed 33 randomly-selected and 15 historic breeding blocks in this region (Fig. 2). We detected no burrowing owls in the random blocks and 10 pairs in the historic breeding blocks; 2 pairs were incidentally detected outside our targeted blocks. All pairs were detected on lowland blocks in Tehama and Yuba counties.

Our random-sample based population estimate for this region is zero. Using our criteria stated earlier, the "best estimate" is 12 pairs for the lowland subregion and zero for upland subregion. The number of Burrowing Owl pairs detected in the region declined only moderately, from 18 pairs to 12 pairs between the 1991-1993 and 2006-2007 surveys, but because 11 pairs were found on randomly-selected blocks during 1991-1993 (compared to no owls detected on randomly-selected blocks during 2006-2007), DeSante et al. (2007) extrapolated their early 1990s findings to estimate that 231 pairs were present in the region, a number greatly in excess of both our estimate of zero pairs extrapolated from random blocks only ($\chi^2_1 = 4.274$, $P = 0.039$; Table 3) and our "best estimate" of 12 pairs, reflecting the actual number of pairs we detected on all blocks surveyed (Table 4).

MIDDLE CENTRAL VALLEY

We surveyed 71 randomly-selected blocks and 128 historic breeding blocks in this region (Fig. 3). Surveys of random blocks yielded 34 Burrowing Owl pairs, and surveys of historic breeding blocks yielded 348 pairs, for a total of 382 pairs. Substantial concentrations of owls were located in lowland areas of Yolo, Solano, Sacramento, Contra Costa, and San Joaquin counties. However we found only two pairs in all of Stanislaus County, and detected only one pair incidentally in Merced County. We found no Burrowing Owls in the upland (foothill) blocks of western El Dorado, Amador, Calaveras, Tuolumne, and Merced counties.

In the 59 randomly-selected lowland blocks surveyed, we found 34 owl pairs, yielding a random-sample based estimate of 502 ± 209 pairs. This estimate was greater than the total number of pairs we actually found in the lowland subregion (34 pairs on randomly-selected blocks plus 305 pairs on historic breeding blocks), so it serves as our "best estimate" for the upland subregion. No owls were detected on randomly-selected upland blocks anywhere in the region, so our random-sample based estimate for the upland subregion is zero pairs. However, we found 43 pairs on upland historic breeding blocks, so our "best estimate" for the upland subregion is the actual number of pairs we found in upland blocks: 43 pairs. Summing our estimate of 502 ± 209 pairs in the lowland subregion and our count of 43 pairs on the upland blocks surveyed, our estimate for the Middle Central Valley region is 545 pairs, 8.2% fewer than the 594 pairs DeSante et al. (2007) estimated to be present in the early 1990s (Table 4).

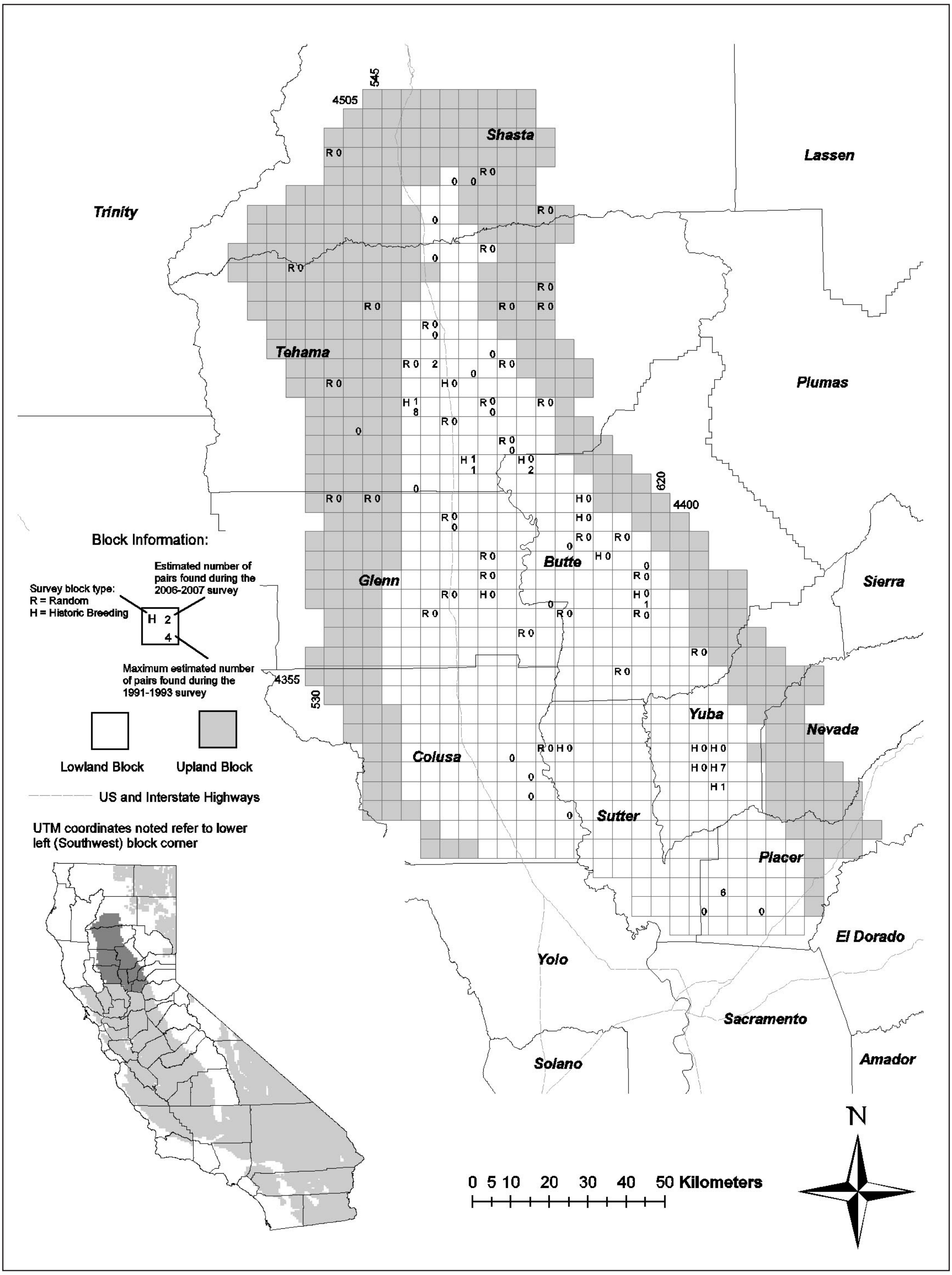

FIGURE 2. Results from the Northern Central Valley region, including numbers of Burrowing Owl pairs detected during 1991-1993 and 2006-2007. Shown are all 5-km x 5-km lowland blocks (white) and upland blocks (gray) assigned to the region. The entire 2006-2007 survey area and the location of the Northern Central Valley region are shown in the inset.

TABLE 3. Comparison of Burrowing Owl regional population estimates extrapolated from randomly-selected blocks for regions that were surveyed during both the 1991-1993 and 2006-2007 surveys. Although we present extrapolated population estimates for all regions here, in many cases the extrapolated number of pairs based on random blocks only was not judged to be the "best estimate" of the regional population.

	1991-1993 survey		2006-2007 survey			
Region	No. of random blocks surveyed	Extrapolated no. of pairs (SE)	No. of random blocks surveyed	Extrapolated no. of pairs (SE)	Change in estimated no. of pairs	Percent change in estimated no. of pairs
Northern Central Valley						
Lowland	22	231 (153)	22	0	-231	-100%
Upland	2	0	11	0	0	n/a
All	24	231 (153)	33	0	-231	-100%
Middle Central Valley						
Lowland	163	577 (122)	59	502 (209)	-75	-13.0%
Upland	28	17 (17)	12	0	-17	-100%
All	191	594 (139)	71	502 (209)	-92	-15.5%
Southern Central Valley						
Lowland	41	1,000 (410)	63	968 (342)	-32	-3.2%
Upland	11	396 (182)	18	145 (118)	-251	-61.4%
All	52	1,396 (592)	81	1,113 (460)	-283	-20.3%
Entire Central Valley	267	2,221 (884)	185	1,615 (669)	-606	-27.3%
San Francisco Bay Area Interior						
Lowland	86	41 (20)	20	0	-41	-100%
Upland	25	0	12	21 (21)	+21	n/a
All	111	41 (20)	32	21 (21)	-20	-51.2%
Central-western Interior						
Lowland	14	0	17	0	0	n/a
Upland	16	31 (27)	13	76 (51)	+45	+145.2%
All	30	31 (27)	30	76 (51)	+45	+145.2%
Southwestern Interior						
Lowland	4	100 (100)	3	17 (17)	-83	-83%
Upland	10	127 (81)	8	0	-127	-100%
All	14	227 (181)	11	17 (17)	-210	-95.2%
Coachella Valley						
Lowland	5	0	4	16 (16)	+16	n/a
Upland	6	0	4	0	0	n/a
All	11	0	8	16 (16)	+16	n/a
Imperial Valley						
Lowland	15	6,429 (1,135)	5	5,701 (2,244)	-728	-11.32%
Upland	1	142	2	707 (140)	+565	+397.9%
All	16	6,577	7	6,408 (2,384)	-163	-2.6%

SOUTHERN CENTRAL VALLEY

We surveyed 81 randomly-selected blocks and 83 historic breeding blocks in this region (Fig. 4). Surveys of random blocks yielded 75 Burrowing Owl pairs, and surveys of historic breeding blocks yielded 161 pairs, for a total of 236 pairs.

Owls were considerably more abundant in the southern portion of this region than in the northern portion. We found Burrowing Owls in only one survey block in Madera County (though it had 12 pairs), and our detections were nearly as sparsely distributed in Fresno and Kings counties. We found substantial concentrations only in Tulare and Kern counties. As in the other Central Valley regions, the great majority of owls we found were in lowland blocks; in the upland blocks covering the Sierra foothills we found owls in just one block in each

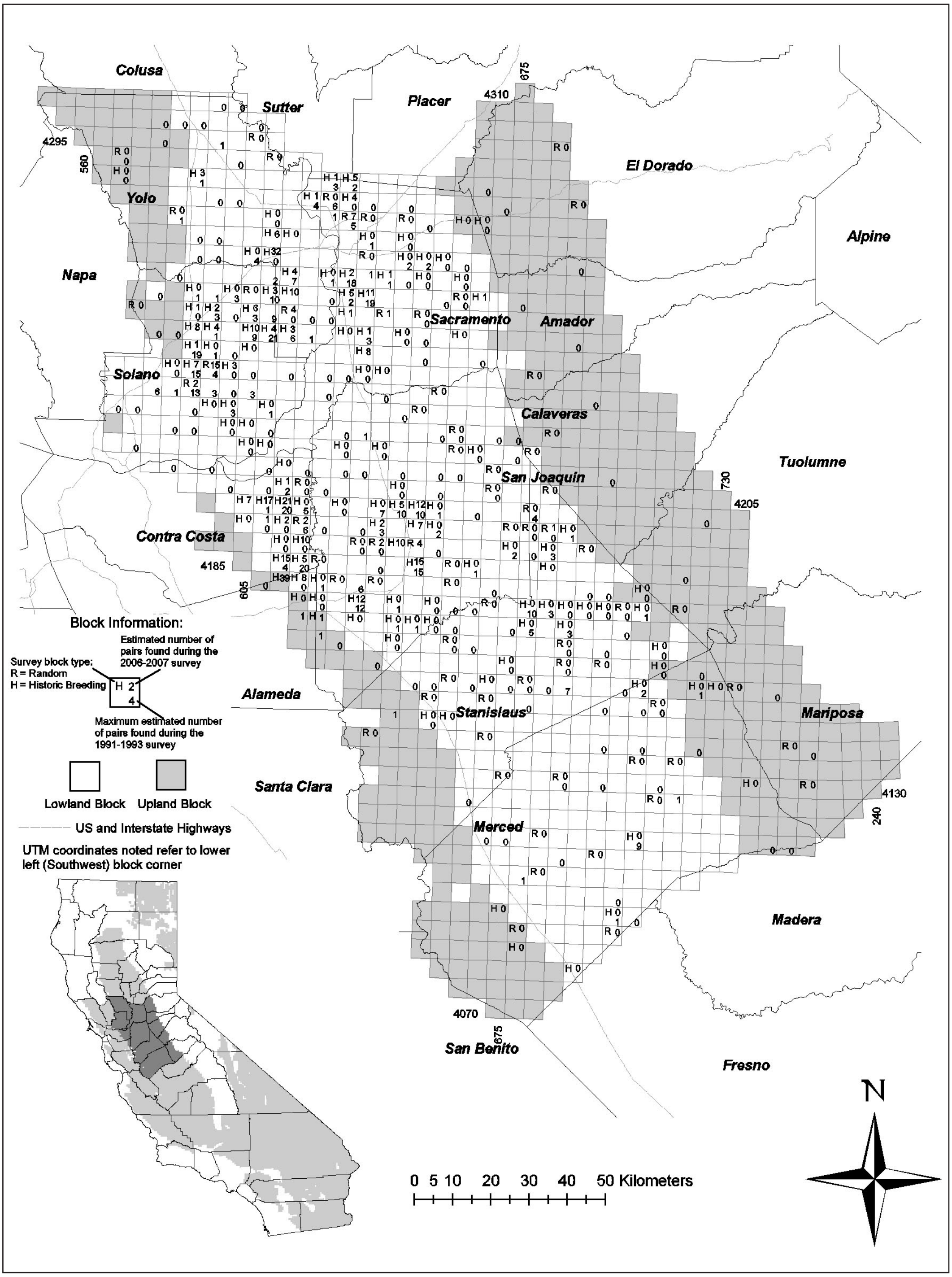

FIGURE 3. Results from the Middle Central Valley region, including numbers of Burrowing Owl pairs detected during 1991-1993 and 2006-2007. Shown are all 5-km x 5-km lowland blocks (white) and upland blocks (gray) assigned to the region. The entire 2006-2007 survey area and the location of the Middle Central Valley region are shown in the inset.

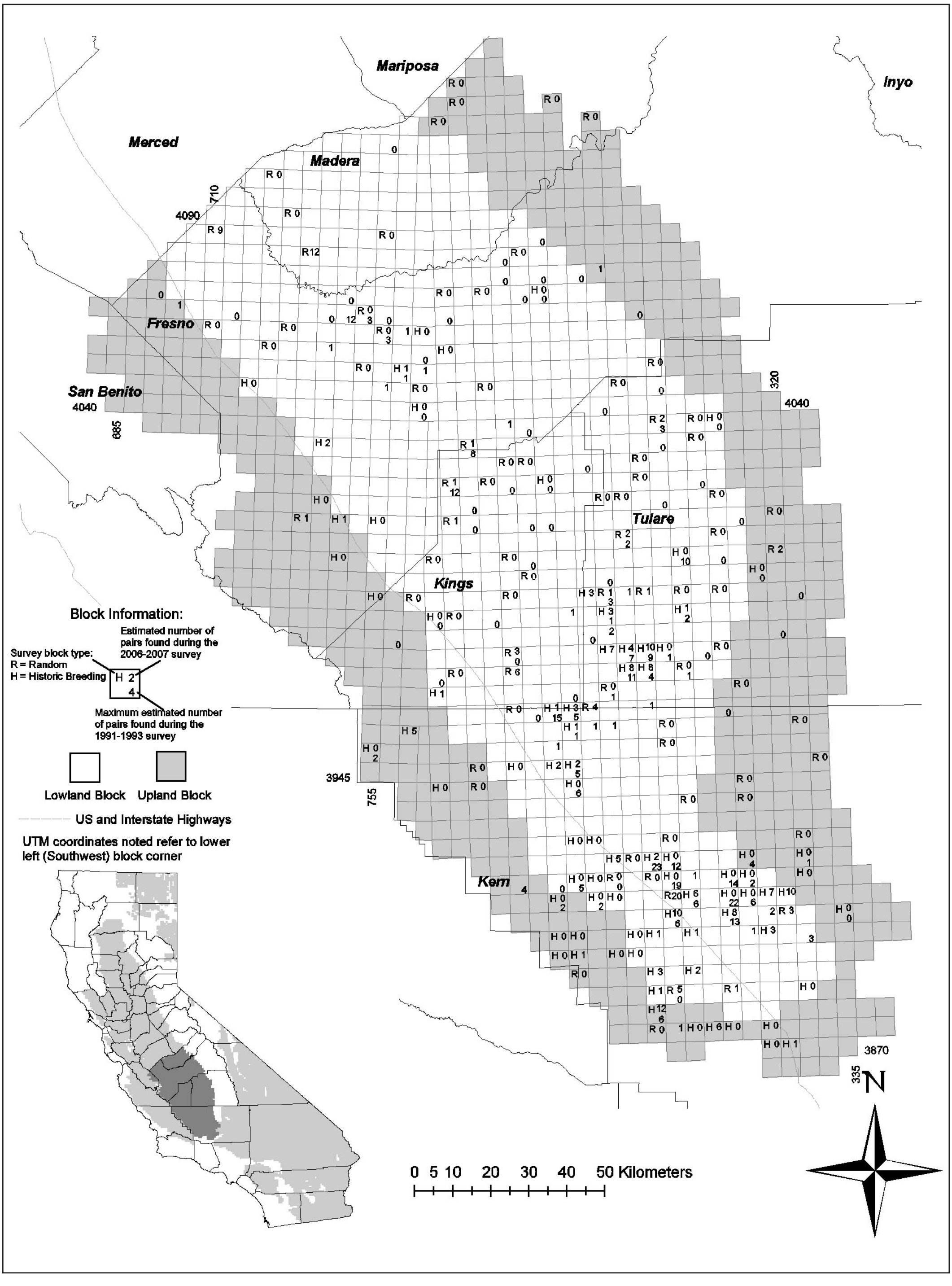

FIGURE 4. Results from the Southern Central Valley region, including numbers of Burrowing Owl pairs detected during 1991-1993 and 2006-2007. Shown are all 5-km x 5-km lowland blocks (white) and upland blocks (gray) assigned to the region. The entire 2006-2007 survey area and the location of the Southern Central Valley region are shown in the inset.

TABLE 4. Comparison of regional and statewide "best estimates" of population size from the 1991-1993 and 2006-2007 Burrowing Owl surveys.

	1991-1993 survey		2006-2007 survey			
Region	No. of pairs found	"Best estimate" of pairs in region[a]	No. of pairs found	"Best estimate" of pairs in region[a]	Change in estimated no. of pairs	Percent change in estimated no. of pairs
Northern Central Valley						
Lowland	18	231 (153)	12	12	-219	-94.8%
Upland	0	0	0	0	0	n/a
All	18	231 (153)	12	12	-219	-94.8%
Middle Central Valley						
Lowland	404	577 (112)	339	502 (209)	-75	-13.0%
Upland	1	17 (17)	43	43	+26	+152.9%
All	405	594 (129)	382	545	-49	-8.2%
Southern Central Valley						
Lowland	259	1,000 (410)	204	968 (342)	-32	-3.2%
Upland	19	396 (182)	32	145 (118)	-251	-63.4%
All	278	1,396 (592)	236	1,113 (460)	-283	-20.3%
San Francisco Bay Area Interior						
Lowland	154	154	98	98	-56	-36.4%
Upland	11	11	21	21	+10	+90.9%
All	165	165	119	119	-46	-27.9%
San Francisco Bay Area Coast[b]						
Lowland	0	0	0	0	0	n/a
Upland	0	0	0	0	0	n/a
All	0	0	0	0	0	n/a
Central-western Interior						
Lowland	7	7	8	8	+1	+14.3%
Upland	3	31 (27)	13	76 (51)	+45	+145.2%
All	10	38	21	84	+46	+121.1%
Central-western Coast[c]						
Lowland	8	8	0	0	-8	-100%
Upland	0	0	0	0	0	n/a
All	8	8	0	0	-8	-100%
Southwestern Coast[d]						
Lowland	8	36 (36)	16	16	-20	-55.6%
Upland	0	0	26	26	+26	n/a
All	8	36 (36)	42	42	+6	+16.7%
Southwestern Interior						
Lowland	12	100 (100)	37	37	-63	-63%
Upland	6	127 (81)	113	113	-14	-11.0%
All	18	227 (181)	150	150	-77	-33.9%
Coachella Valley						
Lowland	0	0	12	16 (16)	+16	n/a
Upland	0	0	37	37	+37	n/a
All	0	0	49	53	+53	n/a
Imperial Valley						
Lowland	1,041	6,429 (1,135)	499	5,701 (2,244)	-728	-11.3%
Upland	4	142	22	707 (140)	+565	+397.9%
All	1,045	6,571	521	6,408 (2,384)	-163	-2.5%
Modoc Plateau/Great Basin						
All	Not surveyed		0	0	n/a	n/a
Northern Mojave/Eastern Sierra Nevada[e]						
Lowland	Not surveyed		1	1	n/a	n/a
Upland	Not surveyed		0	0	n/a	n/a
All	Not surveyed		1	1	n/a	n/a

TABLE 4. Continued.

	1991-1993 survey		2006-2007 survey			
Region	No. of pairs found	"Best estimate" of pairs in region[a]	No. of pairs found	"Best estimate" of pairs in region[a]	Change in estimated no. of pairs	Percent change in estimated no. of pairs
Western Mojave Desert[e]						
Lowland	Not surveyed		94	560 (268)	n/a	n/a
Upland	Not surveyed		0	0	n/a	n/a
All	Not surveyed		94	560 (268)	n/a	n/a
Eastern Mojave Desert[e]						
Lowland	Not surveyed		1	32 (32)	n/a	n/a
Upland	Not surveyed		0	0	n/a	n/a
All	Not surveyed		1	32 (32)	n/a	n/a
Sonoran Desert[e]						
All	Not surveyed		179	179	n/a	n/a
Statewide, excluding "new" regions						
Number of pairs found	1,955		1,532		-423	-21.6%
Extrapolated no. of pairs		9,127 (1,243)		8,128 (2,391)	-999	-10.9%
"Best estimate" of no. of pairs		9,266		8,526	-740	-8.0%
Statewide, including "new" regions						
Number of pairs found			1,758			
Extrapolated no. of pairs				9,187 (2,346)		
"Best estimate"				9,298		

[a] Numbers in parenthesis indicate the standard error of the estimate. Estimates lacking a standard error indicate the actual count of breeding pairs detected in a subregion and are presented as the "best estimate" in cases where the count is higher than the region's calculated population estimate, which is based on randomly-selected blocks only and excludes data from historic breeding blocks that were not randomly selected

[b] The San Francisco Bay Area Coast region was not surveyed as part of this study in 2006-2007. Our "best estimate" of zero pairs in both the lowland and upland subregions is based on local knowledge (D. DeSante, *pers. comm.*) and information in Townsend and Lenihan (2007).

[c] The Central-western Coast region was not surveyed as part of this study in 2006-2007. Our "best estimate" of zero pairs in both the lowland and upland subregions is based on local knowledge (D. Roberson, *pers. comm.*).

[d] The Southwestern Coast region was not surveyed as part of this study in 2006-2007. Our "best estimates" of 16 pairs in the lowland subregion and 26 pairs in the upland subregion are based on information in Lincer and Bloom (2007) and Kidd et al. (2007).

[e] Reported in Wilkerson and Siegel (*in press*).

of Fresno, Tulare, and Kern counties.

In the 63 randomly-selected lowland blocks surveyed, we found 72 pairs of owls, yielding a random-sample based estimate of 968 ± 342 pairs. This estimate was greater than the total number of pairs found in the lowland subregion (72 pairs on randomly-selected blocks plus 132 pairs on historic breeding blocks), so it serves as our "best estimate" for the lowland subregion. In the 18 randomly-selected upland blocks surveyed, we found three pairs of owls, yielding a random-sample based estimate of 145 ± 118 pairs in the upland subregion. This estimate is greater than the total number of pairs found in the upland subregion (three pairs on randomly-selected blocks plus 32 pairs on historic breeding blocks), so it serves as our "best estimate" for the upland subregion. Summing our extrapolated estimates for the lowland and upland subregions, our estimate for the Southern Central Valley region is 1,113 ± 460 pairs (Table 3), 20.3 % fewer than the 1,396 pairs DeSante et al. (2007) estimated in the early 1990s (Table 4), but not a statistically significant difference ($F_{1,131}$ = 0419, $P = 0.838$).

Examining blocks that contained Burrowing Owls in the early 1990s and were resurveyed during 2006-2007 indicates two areas in the region where substantial, concentrated losses appear to have occurred: six blocks in western

Bakersfield lost a total of 53 breeding pairs, and further west, in agricultural land located west of Rosedale and south of Shafter, 42 fewer pairs were detected on three survey blocks (Fig. 5). Concentrated losses of Burrowing Owls on the western edge of Bakersfield occurred in blocks where substantial urban land conversion occurred between 1992 and 2001 (Multi-resolution Land Characteristics Consortium 2001).

SAN FRANCISCO BAY AREA INTERIOR

We surveyed 32 randomly-selected blocks and 58 historic breeding blocks in this region (Fig. 6). The relatively large proportion of historic breeding blocks reflects the excellent pre-survey information available about the region's Burrowing Owls. Surveys of random blocks yielded only a single pair, located on an upland block in northeastern Alameda County, north of Livermore. Pooling data from random and historic breeding blocks, we found 119 pairs.

All of the Burrowing Owls detected in the region were in Alameda or Santa Clara counties. During the 1990s survey small numbers of Burrowing Owl pairs were also detected in San Mateo County (one pair) and Sonoma County (two pairs), but our surveyors were unable to find owls in these or other locations throughout those counties.

In Alameda County, we detected no Burrowing Owls in the western, lowland portion adjacent to San Francisco Bay, where 34 pairs were found distributed across nine blocks in the early 1990s (Fig. 7). In contrast, we found 14 pairs of owls in the upland blocks of the eastern half of the county (compared with 11 pairs found in the early 1990s) along the Highway 580 corridor between Dublin and Livermore and in the Altamont Hills northeast of Livermore, an area where relatively large numbers of breeding Burrowing Owls have recently been observed (Barclay and Harman 2007). The richest area in Alameda County was the south-central lowland portion; we observed 25 pairs on a single block at Don Edwards San Francisco Bay National Wildlife Refuge. Two pairs were detected on the same block during the 1991-1993 survey. Nine additional pairs were distributed across two historic breeding blocks to the north of this area, apparently in urban park or industrial yard settings in the cities of Fremont and Newark.

In Santa Clara County, detections were restricted to the lowland area in the northwestern corner, as they were during the early 1990s. We detected 56 pairs on two blocks in San Jose and two blocks in Mountain View (Fig. 7), reduced from 97 pairs in the early 1990s.

We detected no pairs on randomly-selected lowland blocks anywhere in the San Francisco Bay Area Interior region, resulting in a zero population estimate for the lowland subregion. We detected 98 pairs on lowland historic breeding blocks, resulting in our "best estimate" of 98 pairs for the lowland subregion. On the 12 randomly-selected upland blocks surveyed, we found one owl pair, yielding a random-sample based estimate of 21 ± 21 pairs throughout the upland subregion. This estimate was greater than the total number of pairs we found on surveyed blocks in the upland subregion (one pair on randomly-selected blocks plus 13 pairs on historic breeding blocks), so it serves as our best estimate for owl pairs in the upland subregion. Summing our count from the lowland blocks and our estimate in the upland subregion, our "best estimate" for the number of Burrowing Owl pairs in the San Francisco Bay Area Interior region is 119. This estimate represents a nearly 28% reduction from the 165 pairs estimated from the 1991-1993 survey (Table 4). Like our estimate, the early 1990s estimate was also an actual count of all pairs found, rather than an extrapolated estimate based on randomly-selected blocks only, so the statistical significance of the apparent decline cannot be tested. However, because the region is very well known by the local birding community (which helped us identify historical breeding blocks), it seems unlikely that there could be more than a few pairs that went undetected during either survey.

SAN FRANCISCO AREA COAST

DeSante et al. (2007) were unable to find any Burrowing Owls in this region during the 1990s survey (Table 4). This relatively small region is well-monitored and well-known by the local birding community. We did not resurvey the region for the 2006-2007 study, as consultation with local experts as well as information in Townsend and Lenihan (2007) strongly indicates that the species remains extirpated from the region.

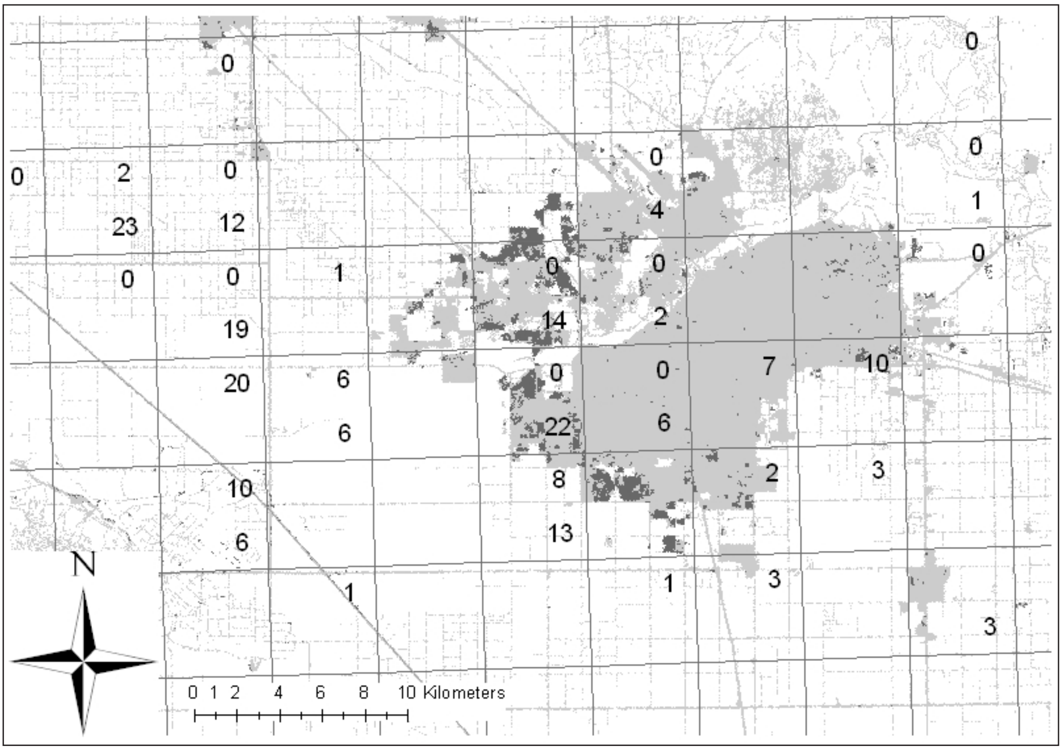

FIGURE 5. The number of Burrowing Owl pairs detected in the Bakersfield area during IBP's 1991-1993 survey (indicated in lower right corner of each block) and 2006-2007 survey (indicated in upper right corner of survey block). The large shaded area represents metropolitan Bakersfield; light gray shading indicates urban land cover as of 1992; dark gray shading indicates areas that were not mapped as urban in 1992, but were converted to urban use between 1992 and 2001 (Multi-resolution Land Characteristics Consortium 2001). Note the concentrated losses of Burrowing Owls in blocks on the western edge of Bakersfield, where substantial urban land conversion occurred between 1992 and 2001.

CENTRAL-WESTERN INTERIOR

We surveyed 30 randomly-selected blocks and 14 historic breeding blocks in this region (Fig. 8). Surveys of random blocks yielded just two Burrowing Owl pairs, both located on upland blocks of San Luis Obispo County. Pooling data from random and historic breeding blocks, we found 21 pairs in the region. Small clusters of owls were found in four areas: Bolsa Valley northwest of Hollister, San Benito County; low foothills of the Coast Range east of King City, Monterey County; northeast corner of San Luis Obispo County; and the Carrizo Plain, southeastern San Luis Obispo County (Fig. 8).

Since no Burrowing owls were detected on randomly-selected lowland blocks anywhere in this region, our random-sample based population estimate for the lowland subregion is zero pairs. However, we found 8 pairs on lowland historic breeding blocks, so our "best estimate" for the lowland subregion is the actual number of pairs we found: eight. On the 13 randomly-selected upland blocks we surveyed, we found two pairs, yielding a random-sample based estimate of 76 ± 51 pairs throughout the upland subregion. This estimate was greater than the total number of pairs we found in the upland subregion (two pairs on randomly-selected blocks plus 11 pairs on historic breeding blocks), so it serves as our best estimate for owl pairs in the upland subregion. Summing our count on the lowland blocks and our estimate in the upland subregion, our estimate for the Central-western Interior region is 84 pairs, a 121% increase from the estimate of 38 pairs during the 1991-1993 survey (Table 4).

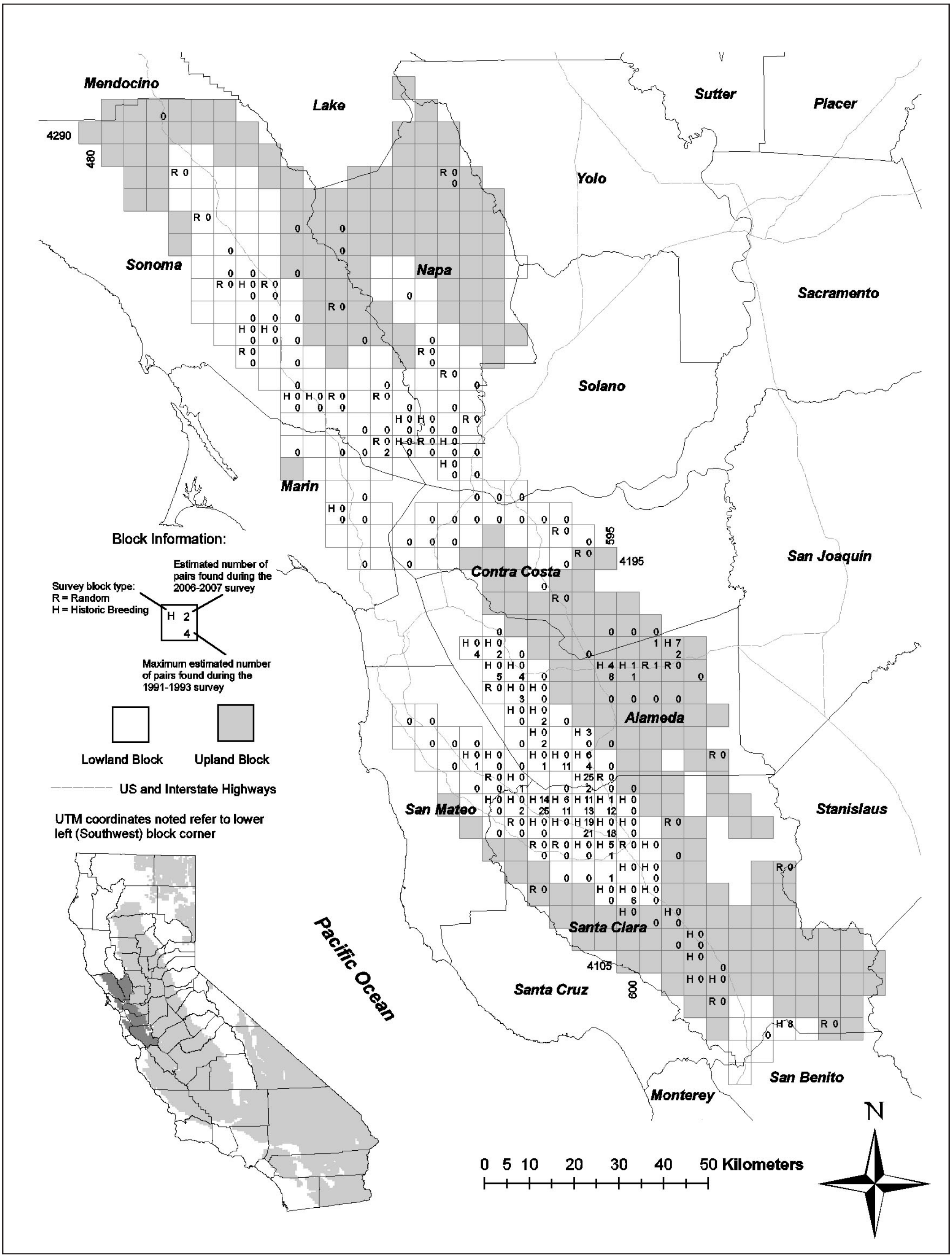

FIGURE 6. Results from the San Francisco Bay Area Interior region, including numbers of Burrowing Owl pairs detected during 1991-1993 and 2006-2007. Shown are all 5-km x 5-km lowland blocks (white) and upland blocks (gray) assigned to the region. The entire 2006-2007 survey area and the location of the San Francisco Bay Area Interior region are shown in the inset.

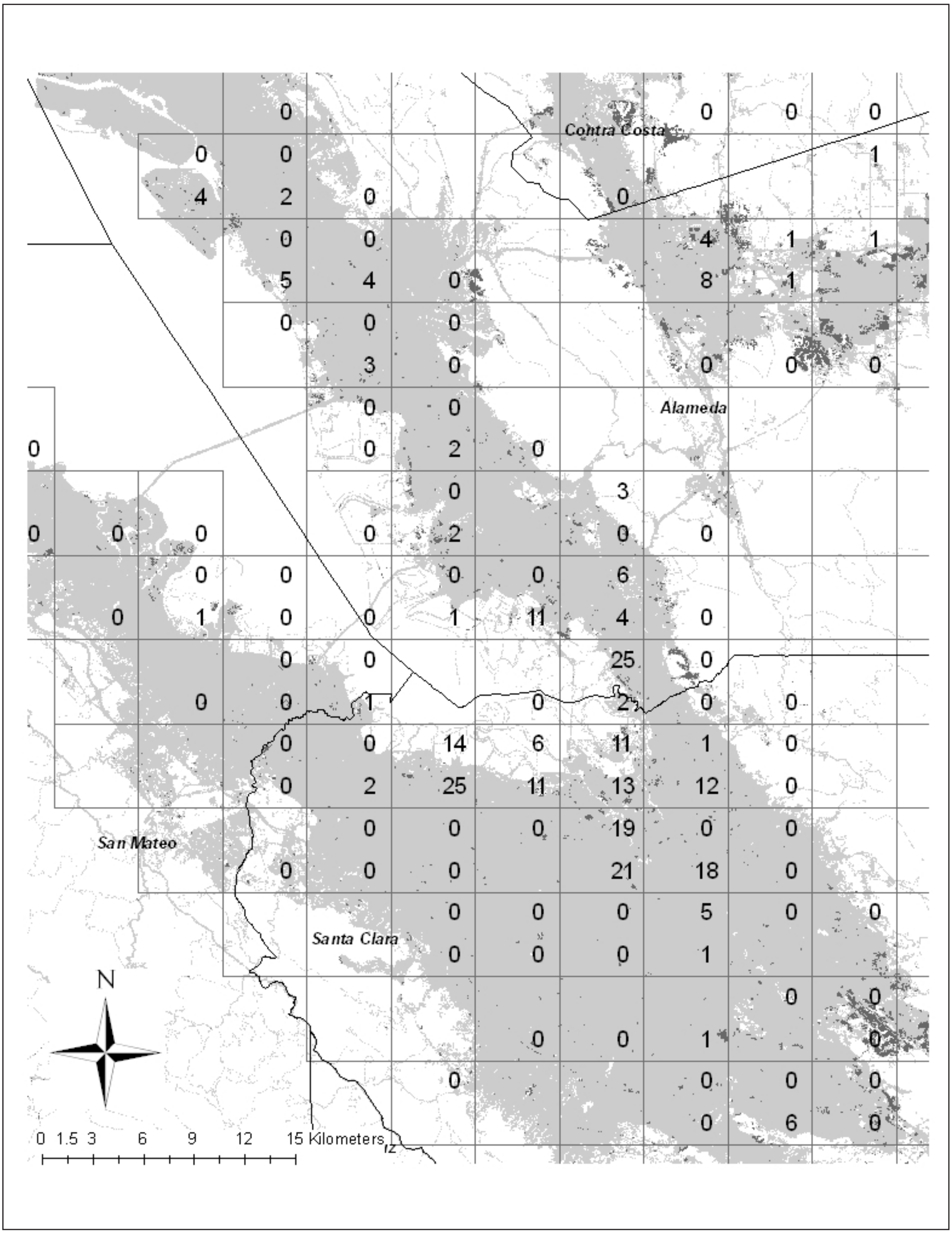

FIGURE 7. The number of Burrowing Owl pairs detected on survey blocks in the southern and eastern San Francisco Bay Area during IBP's 1991-1993 survey (indicated in lower right corner of each block) and 2006-2007 survey (indicated in upper right corner of survey block). Light gray shading indicates urban land cover as of 1992; dark gray shading indicates areas that were not mapped as urban in 1992, but were converted to urban use between 1992 and 2001 (Multi-resolution Land Characteristics Consortium 2001).

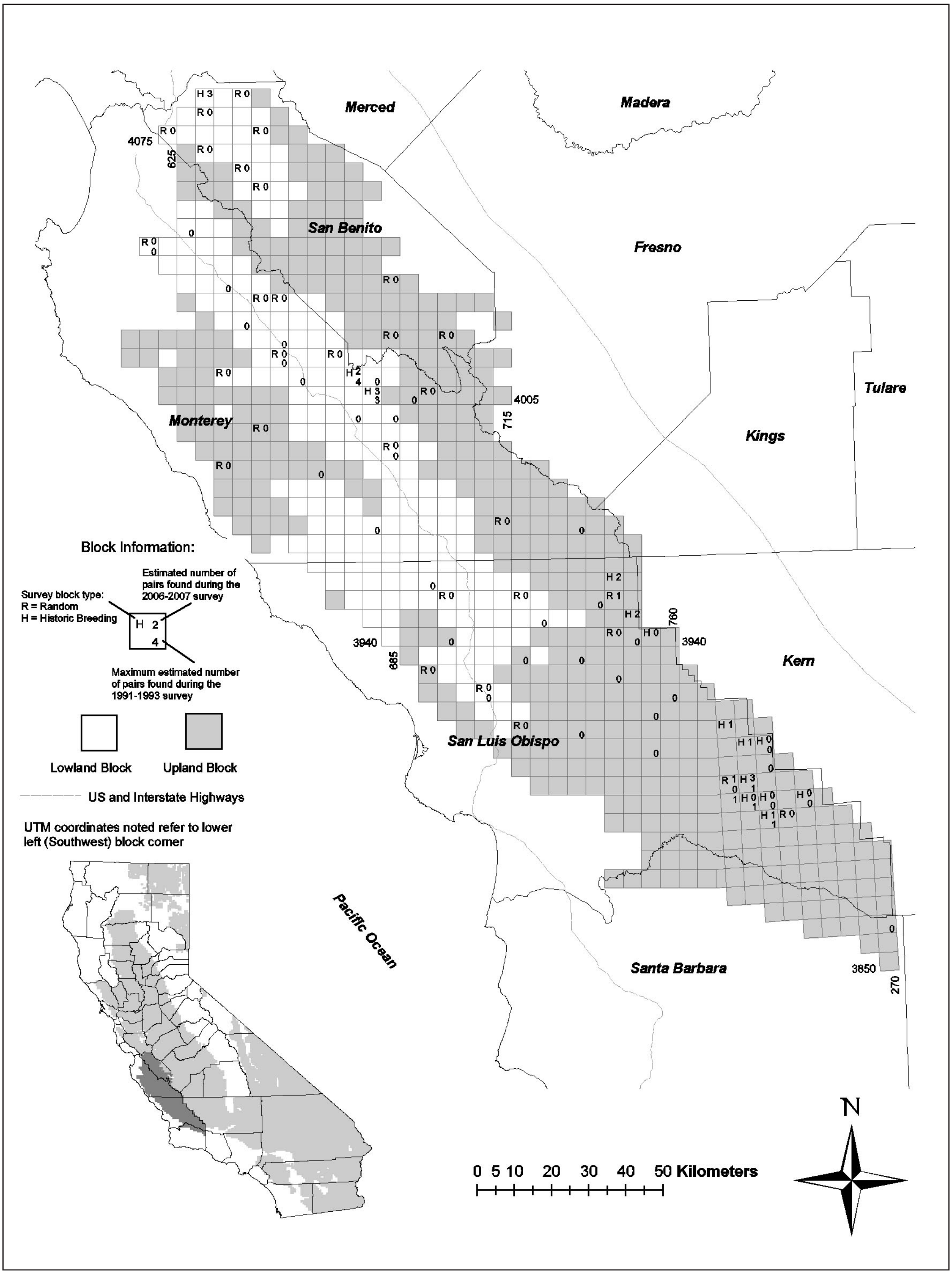

FIGURE 8. Results from the Central-western Interior region, including numbers of Burrowing Owl pairs detected during 1991-1993 and 2006-2007. Shown are all 5-km x 5-km lowland blocks (white) and upland blocks (gray) assigned to the region. The entire 2006-2007 survey area and the location of the Central-western Interior region are shown in the inset

CENTRAL-WESTERN COAST

This is one of the three coastal regions we did not survey during 2006-2007. In the 1991-1993 survey, eight pairs of Burrowing Owls were detected in the region; seven pairs were near Salinas, Monterey County, and a single pair was in northern Santa Barbara County (DeSante et al. 2007). The Salinas owls were distributed between two areas; five pairs were at the Salinas Airport and two pairs were near the town of Boronda. Visits to both of those sites by local birders in the last decade have yielded no detections, and foraging habitat adjacent to the airport colony has been developed (D. Roberson, *pers. comm*.). The single pair from northern Santa Barbara County was present in 1992, but could not be relocated when the same survey block was revisited in 1993 (DeSante et al. 2007). Consequently Burrowing Owls are likely extirpated from the region (Table 4).

SOUTHWESTERN COAST

Because the few breeding owls present in this region are already well monitored, we did not survey the region. Kidd et al. (2007) determined that Burrowing Owl populations in western Santa Barbara, Ventura, and Los Angeles counties had been extirpated; however, they documented three breeding pairs in Orange County as recently as 2005. In a thorough assessment of the species' status in San Diego County, Lincer and Bloom (2007) determined there were between 41 and 46 pairs present; all but two were within our region boundaries, allowing for a count of between 39 and 44 pairs. The lower count of 39 plus the three pairs from Orange County yields an estimate of 42 pairs for the Southwestern Coast region (Table 4). The 1991-1993 "best estimate" for this region was 36 pairs, although only eight pairs were actually detected (DeSante et al. 2007). The apparent increase could be from the more thorough coverage provided by Lincer and Bloom (2007) or a slight but real increase in the region's owl population.

SOUTHWESTERN INTERIOR

We surveyed 11 randomly-selected blocks and 57 historic breeding blocks in this interior region (Fig. 9). The relatively large proportion of historic breeding blocks reflects the excellent pre-survey information we received from a UC Riverside graduate student studying the local Burrowing Owl population (Ginny Short, *pers. comm.*). Our surveys of random blocks yielded only a single pair, located in a lowland block at Ontario International Airport, San Bernardino County. However, we found 149 pairs utilizing diverse habitats on historic breeding blocks, yielding a total of 150 pairs of owls detected in the region.

The one pair of owls found on the three randomly-selected lowland blocks yielded a random-sample based estimate of 17 ±17 pairs throughout the lowland subregion. Since this estimate was lower than the total number of pairs found in the lowland subregion (pooling data detections from random and historic breeding blocks) our "best estimate" for the number of owl pairs in the lowland subregion is the actual number of pairs counted: 37. Since no Burrowing Owls were detected on any of the eight randomly-selected upland blocks, our random-sample based estimate for the upland subregion is zero pairs. However, we found 113 pairs on upland historic breeding blocks, so our best estimate for the upland subregion is the actual number of pairs found: 113. Summing our counts from lowland and upland blocks, our estimate for the Southwestern Interior region is 150 pairs, 33.9% fewer than were estimated to be present during the 1991-1993 survey (Table 4). We note, however, that the 1990s estimate was extrapolated from surveys of random blocks while our estimate is our actual count of all owls on random and historic breeding blocks, and was based on more extensive pre-survey information. Thus, comparing these "best estimates" may be somewhat problematic.

COACHELLA VALLEY

We surveyed eight randomly-selected blocks and 12 historic breeding blocks in this region (Fig. 10). Surveys of random blocks yielded just one pair of Burrowing Owls, while surveys of historic breeding blocks yielded 48 pairs, for a total of 49 pairs detected in the region. The highest densities of detections were clustered at the northern end of the region around the town of Desert Hot Springs and south to Interstate 10. Smaller numbers of owls (1-4 pairs per block) were detected along the Interstate 10 corridor as far south as the town of Mecca. A single pair was located on a randomly-selected block along

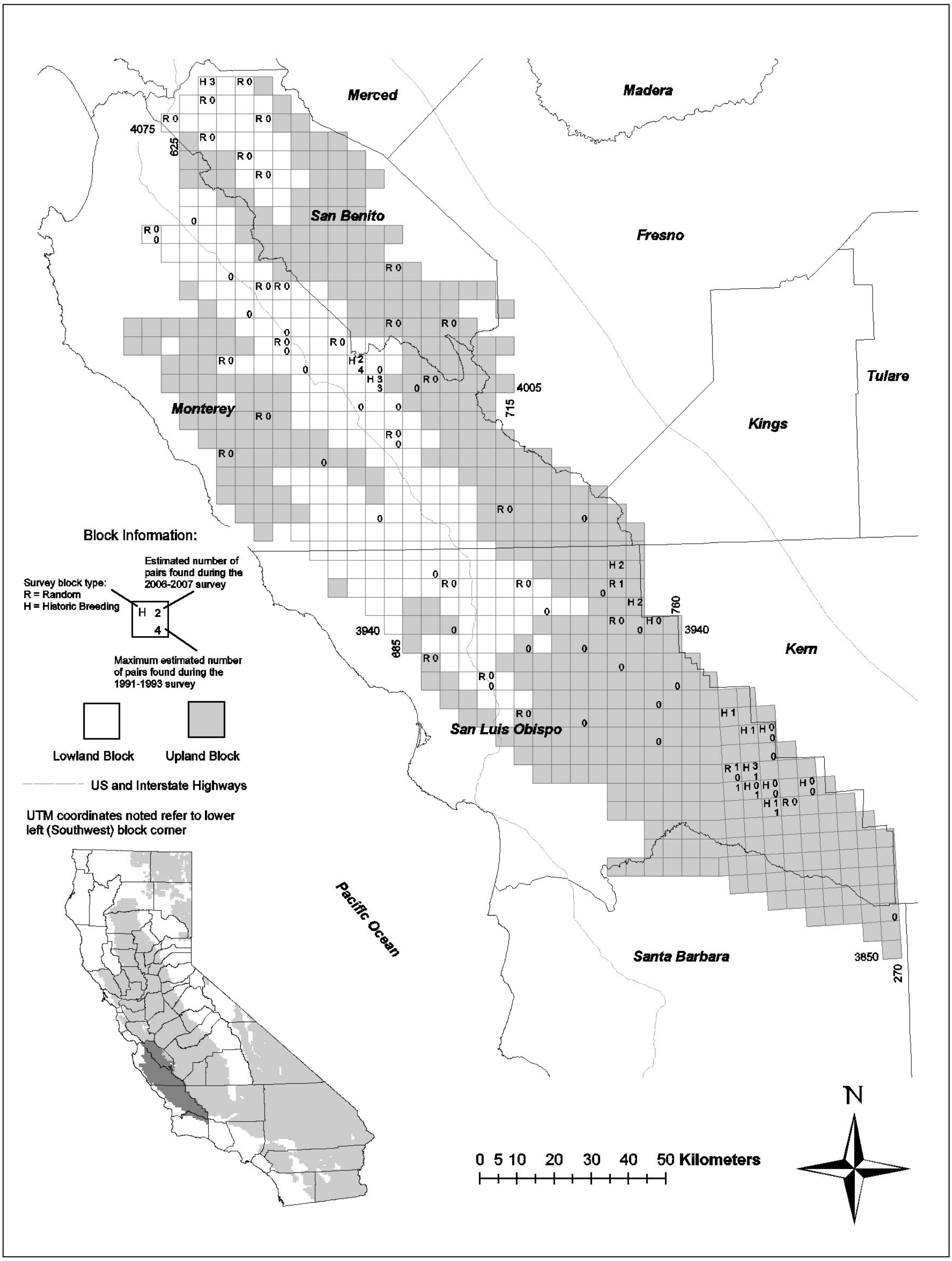

FIGURE 9. Results from the Southwestern Interior region, including numbers of Burrowing Owl pairs detected during 1991-1993 and 2006-2007. Shown are all 5-km x 5-km lowland blocks (white) and upland blocks (gray) assigned to the region. The entire 2006-2007 survey area and the location of the Southwestern Interior region are shown in the inset.

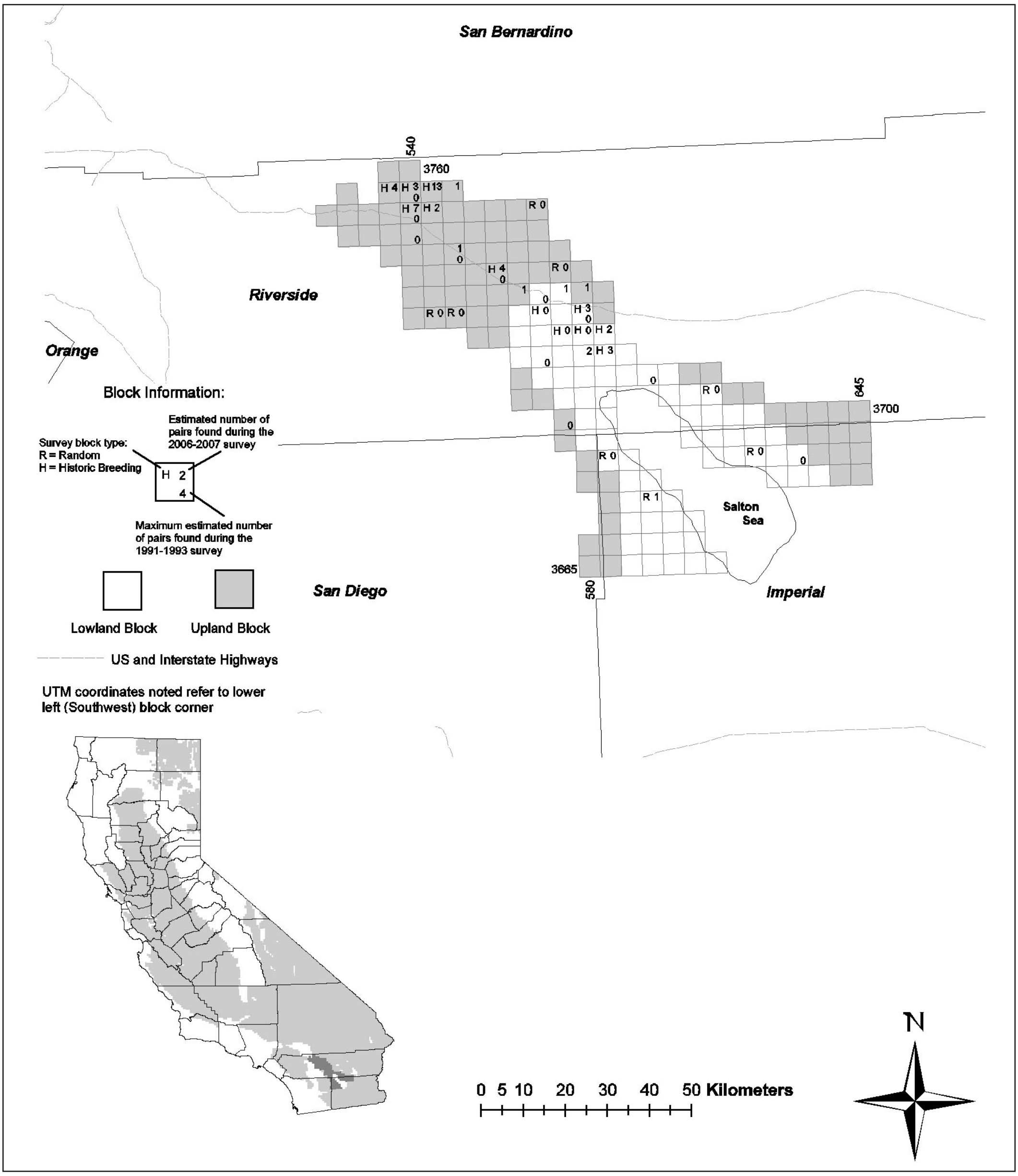

FIGURE 10. Results from the Coachella Valley region, including numbers of Burrowing Owl pairs detected during 1991-1993 and 2006-2007. Shown are all 5-km x 5-km lowland blocks (white) and upland blocks (gray) assigned to the region. The entire 2006-2007 survey area and the location of the Coachella Valley region are shown in the inset.

the west side of the Salton Sea, at the southern end of Salton City.

In the four randomly-selected lowland blocks surveyed, we found one pair of owls, yielding a random-sample based estimate of 16 ± 16 pairs throughout the lowland subregion. This estimate was slightly greater than the total number of pairs found in the lowland subregion (one pair on randomly-selected blocks plus 11 pairs on historic breeding blocks), so it serves as our "best estimate" for owl pairs in the lowland subregion. No Burrowing Owls were detected on the four randomly-selected upland blocks, so our random-sample based population estimate for the upland subregion is zero pairs. However, we found 37 pairs on upland historic breeding blocks, so our "best estimate" for the upland subregion is the actual number of pairs we found: 37. Summing our estimate from the lowland subregion and our count on the upland blocks surveyed, our estimate for the Coachella Valley region is 53 pairs, a remarkable change from the 1991-1993 estimate of zero pairs (Table 4). Four historic breeding blocks (two upland blocks at the northern end of the region plus an additional upland and lowland block further south), in which we found multiple pairs, were also surveyed in the early 1990s (then also selected as random blocks), when no owls were detected. These results suggest the blocks may have been colonized since the 1991-1993 survey.

IMPERIAL VALLEY

We surveyed seven randomly-selected blocks and eight historic breeding blocks in this region (Fig. 11). Surveys of random blocks yielded 271 Burrowing Owl pairs, and surveys of historic breeding blocks yielded 250 pairs, for a total of 521 pairs detected.

In the five randomly-selected lowland blocks surveyed, we found 254 pairs, yielding a random-sample based estimate of 5,701 ± 2,244 pairs throughout the lowland subregion. This estimate was greater than the total number of pairs found in the lowland subregion (254 pairs on randomly-selected blocks plus 245 pairs on historic breeding blocks), so it serves as our "best estimate" for pairs in the lowland subregion. In the two randomly-selected upland blocks surveyed, we found 17 pairs of owls, yielding a random-sample based estimate of 707 ± 140 pairs throughout the upland subregion. This estimate was greater than the number of pairs we found in the upland subregion (17 pairs in randomly-selected blocks plus five pairs in historic breeding blocks), so it serves as our "best estimate" in the upland subregion. Summing our estimates for the lowland and upland subregions, our estimate for the Imperial Valley region is 6,408 ± 2,384 pairs, 2.5% fewer than the 6,571 pairs estimated during the 1991-1993 survey (Table 4), a statistically insignificant decline ($F_{1,12} = 0.3163$, $P = 0.584$).

MODOC PLATEAU/GREAT BASIN

We surveyed 13 randomly-selected blocks, and two historic breeding blocks in this region (Fig. 12). All blocks surveyed were classified as upland blocks, because the entire bioregion lies well above the upper bound of the lower elevation zones for all of our other survey regions.

We detected no Burrowing Owls on random blocks or historic breeding blocks, so our "best estimate" for the number of pairs in the region is zero pairs. Subsequent to our survey, breeding has been observed in Sierra Valley as recently as 2009 (Richard Carlson, *pers. comm.*), although information is lacking to determine whether this breeding location was active during 2006-2007 when we conducted our field work.

NORTHERN MOJAVE DESERT/EASTERN SIERRA NEVADA

We surveyed 36 randomly-selected blocks and two historic breeding blocks in this region; none of them yielded Burrowing Owl detections. However, one pair was detected incidentally on an otherwise unsurveyed block (see Wilkerson and Siegel, *in press*, for additional details).

WESTERN MOJAVE DESERT

We surveyed 48 randomly-selected blocks and 19 historic breeding blocks in this region. Our "best estimate", based on 25 pairs of owls detected on 42 pairs of owls detected on the random blocks, is 560 ± 268 pairs (see Wilkerson and Siegel, *in press*, for additional details).

EASTERN MOJAVE DESERT

We surveyed 43 randomly-selected blocks and two historic breeding blocks in the Eastern Mojave Desert region. Our "best estimate" for the region, based on one pair of owls detected on the randomly-selected blocks, is 32 ± 32 pairs (see Wilkerson and Siegel, *in press*, for additional details).

SONORAN DESERT

We surveyed 31 randomly-selected blocks, and 16 historic breeding blocks in the Sonoran Desert region. Our "best estimate" for the region, based on 179 pairs of owls detected

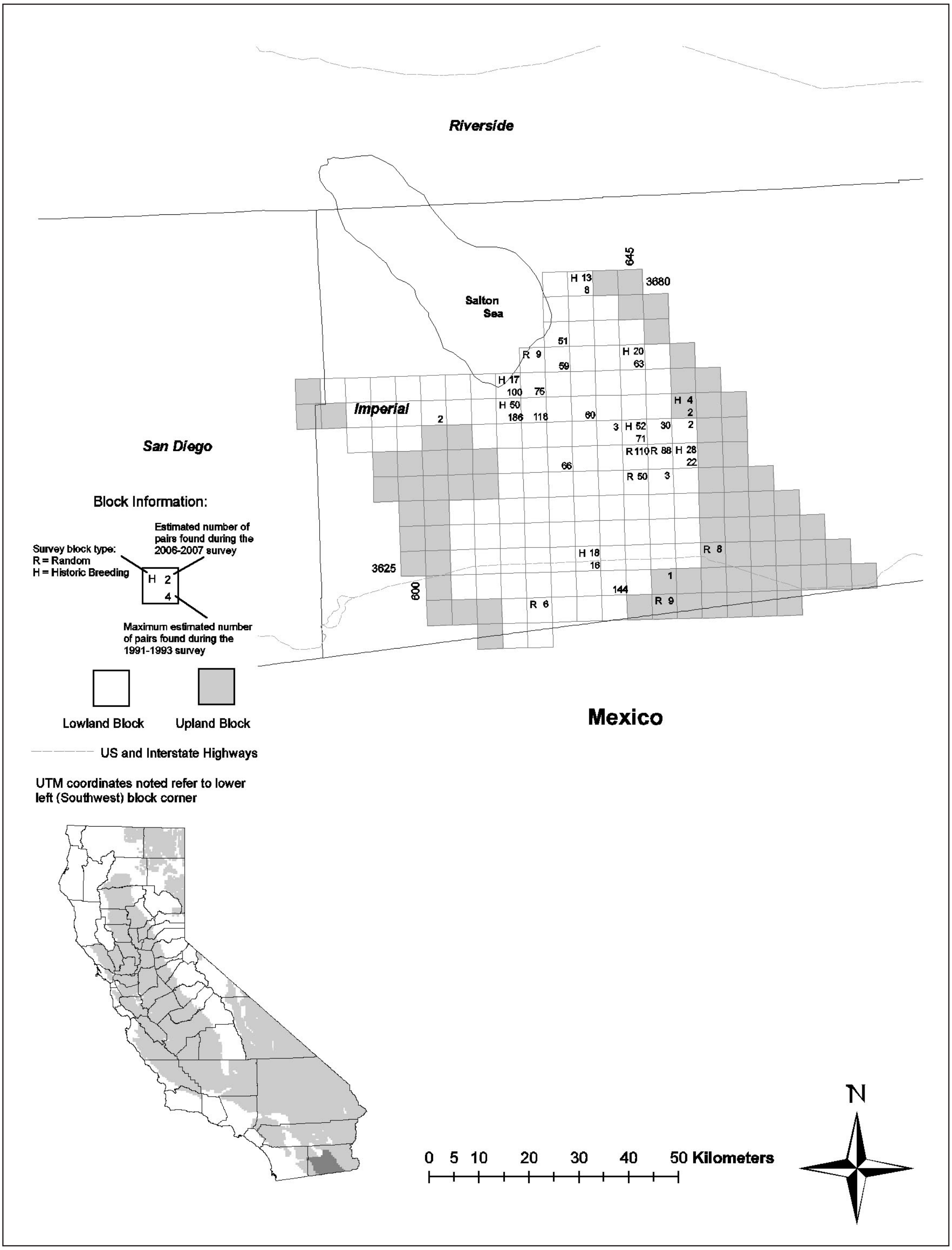

FIGURE 11. Results from the Imperial Valley region, including numbers of Burrowing Owl pairs detected during 1991-1993 and 2006-2007. Shown are all 5-km x 5-km lowland blocks (white) and upland blocks (gray) assigned to the region. The entire 2006-2007 survey area and the location of the Imperial Valley region are shown in the inset.

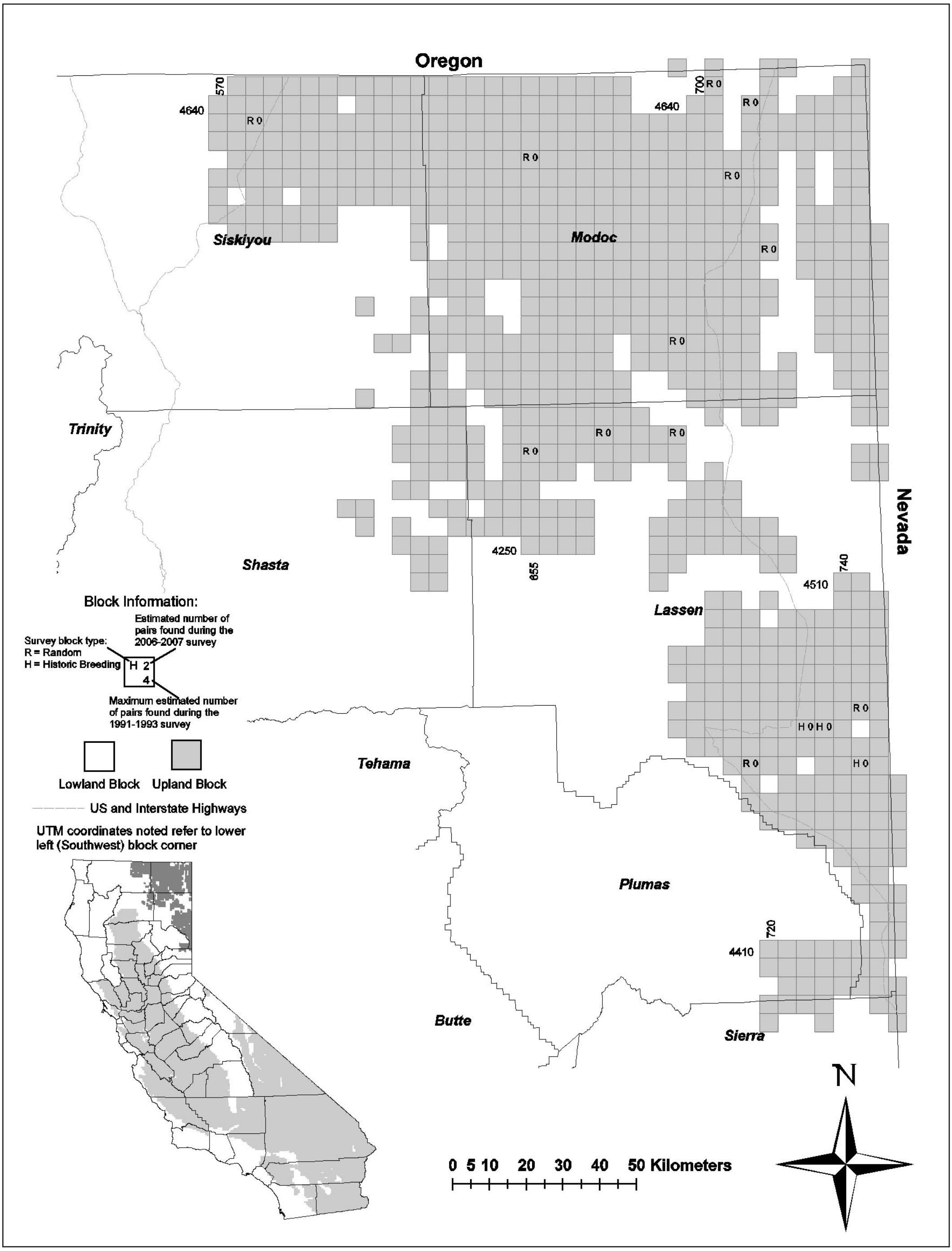

FIGURE 12. Results from the Modoc Plateau/Great Basin region of the 2006-2007 California Burrowing Owl survey. Shown are all 5-km x 5-km assigned to the region; in the case of this region, all blocks were classified as upland. The entire 2006-2007 survey area and the location of the Modoc Plateau/Great Basin region are shown in the inset.

exclusively within the Palo Verde Valley, and no owls detected elsewhere in the region, is our actual pair count in the Palo Verde Valley: 179 pairs (see Wilkerson and Siegel, *in press*, for additional details).

AGGREGATED STATEWIDE RESULTS

Aggregating results across all 2006-2007 survey regions yields a "best estimate" of 9,298 pairs of Burrowing Owls (Table 4). The population is highly concentrated in the Imperial Valley (68.9% of the California population) and to a lesser extent, the Southern Central Valley (12.0% of the statewide population) (Fig. 13). DeSante et al. (2007) reported very similar proportions of the estimated statewide population in 1991-1993 in these two regions.

Omitting the "new" survey regions (Modoc Plateau/Great Basin , Northern Mojave/Eastern Sierra Nevada, Western Mojave, Eastern Mojave, and Sonoran Desert), the aggregated "best estimate" for all regions that were previously surveyed in 1991-1993 is 8,526 pairs, 8% lower than the corresponding estimate generated from 1991-1993 (Table 4). Much of the apparent decline appears to be concentrated in two regions: the Northern Central Valley (231 pairs in 1991-1993 to 12 pairs in 2006-2007), and the Southern Central Valley (1,396 pairs in 1991-1993 to 1,113 pairs in 2006-2007). Other regions with reduced "best estimates" between 1991-1993 and 2006-2007 include the Middle Central Valley (-49 pairs), San Francisco Bay Interior (-46 pairs), Southwestern Interior (-77 pairs), and the Imperial Valley (-163 pairs, but the relatively high absolute numbers make this unlikely to be a meaningful change). In contrast to the overall pattern of declines, our 2006-2007 "best estimates" were higher than the corresponding 1991-1993 estimates for three regions: Central-western Interior (+46 pairs), Southwestern Coast (+6 pairs), and Coachella Valley (+53 pairs).

Because the statewide "best estimate" of the number of pairs is an aggregate of regional extrapolated population estimates and regional minimum counts there is no way to test the statistical significance of the apparent decline between 1991-1993 and 2006-2007. However, we can test for statistically significant change in our population estimates extrapolated only from surveys of randomly-selected blocks. DeSante et al. (2007) provided an extrapolated estimate of 9,127 ± 1,243 pairs for their entire study area; our 2006-2007 estimate extrapolated from randomly-selected blocks across the same survey regions is 8,128 ± 2,391 pairs (Table 4), a non-significant ($F_{1,710} = 0.0533$, $P = 0.817$) reduction of 10.9%.

Including the "new" survey regions, our 2006-2007 estimate extrapolated from randomly-selected blocks is 9,187 ± 2,346 pairs (Table 4). Our "best estimate" for the same comprehensive area is a very similar 9,298 pairs (Table 4).

LAND OWNERSHIP AND HABITATS.

Similar to the findings reported by DeSante et al. (2007), we found that the vast majority of California's breeding Burrowing Owls occur on private lands (Table 5). Small numbers were also found on lands managed by four federal agencies, California state government, and local municipalities (Table 5).

The Burrowing Owls detected during our survey occupied a wide range of habitats, including natural grasslands, agricultural lands, and other human-modified areas (Table 6). Nearly one third of breeding sites were located on the banks of irrigation canals or other concrete or earthen water conveyance structures (Table 6).

DeSante et al. (2007) reported a strong association between Burrowing Owl breeding sites and the presence of ground squirrels. Our results corroborated this finding, but also revealed that association to be far weaker for owls nesting along irrigation canals and other water conveyance structures (Table 6). This weaker association presumably stems from owls not having to depend on ground squirrels for burrow excavation along canal banks, where earthen banks may be particularly easy to excavate, and concrete-lined banks often provide attractive nesting spaces between the concrete lining and the underlying soil.

OWL PERSISTENCE ON SURVEY BLOCKS OCCUPIED DURING THE 1991-1993 SURVEY

Considering blocks surveyed during both 1991-1993 and 2006-2007, in which owls were detected during the first (1991-1993) survey ($N = 149$), we found that abundance significantly declined (mean difference = -2.68 ± 0.50; $t = -5.37$; df = 148; $P < 0.0001$). The probability of detecting

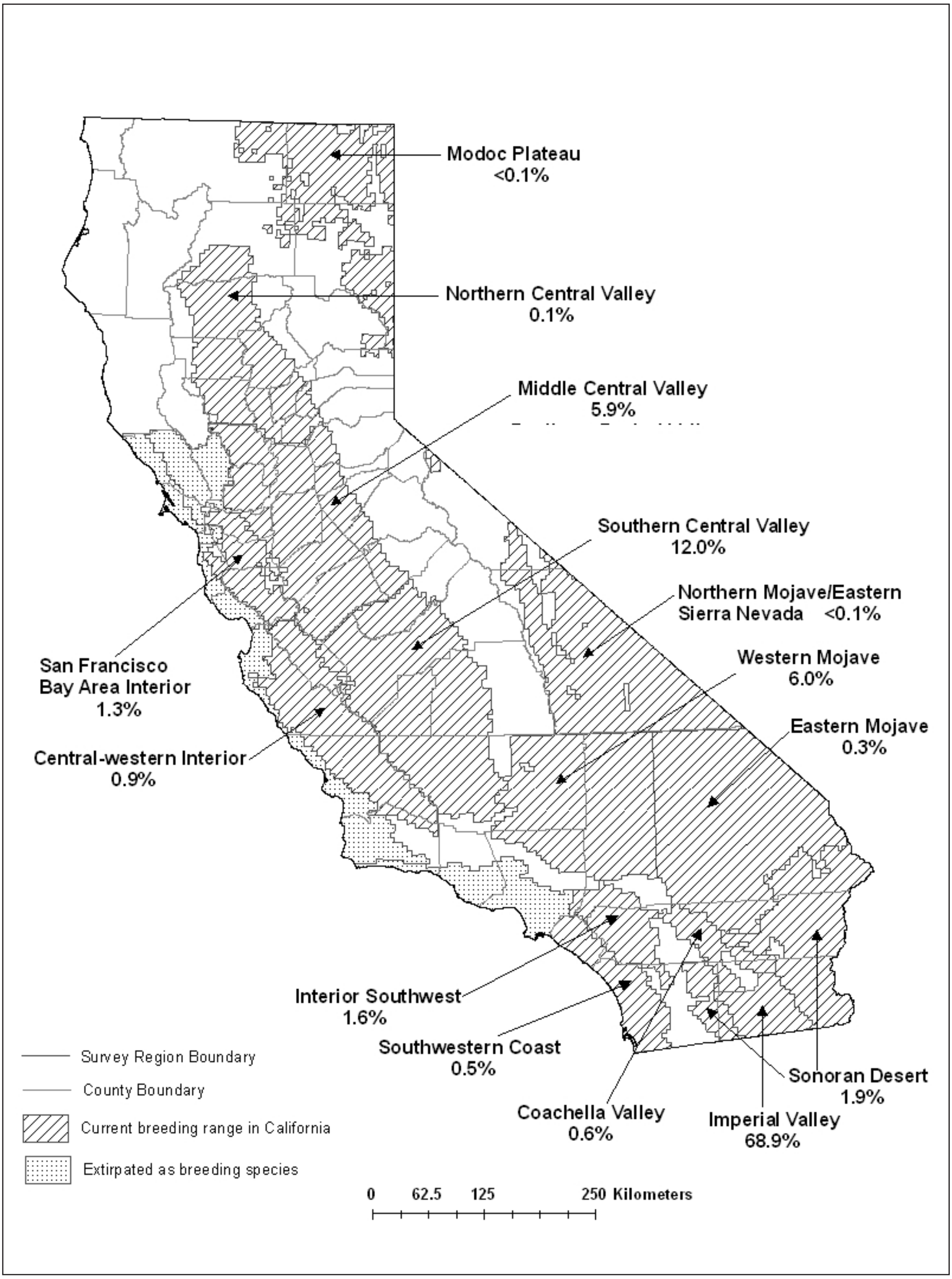

FIGURE 13. Current and former breeding range of Burrowing Owl in California, and percent of the 2006-2007 statewide breeding population estimated to occur in each region based on "best' estimates" (see Methods for explanation of "best" estimates) during the 2006-2007 survey.

TABLE 5. Number and percent of owl pairs detected during the 2006 and 2007 statewide Burrowing Owl survey, classified by land ownership or jurisdiction.

	Number of Burrowing Owl pairs detected	
Land ownership or jurisdiction	Randomly-selected blocks only	Randomly-selected blocks and historic breeding blocks
Private	415 (96.7%)	1,592 (90.6%)
Federal		
Bureau of Land Management	2 (0.5%)	18 (1.0%)
Department of Defense	12 (2.7%)	50 (2.8%)
NASA	0	11 (0.6%)
National Wildlife Refuge System	0	38 (2.2%)
Local government	0	26 (1.5%)
State government	0	22 (1.3%)
Tribal	0	1 (0.1%)
Total	429	1,758

TABLE 6. Primary habitats indicated by field observers at sites where Burrowing Owl pairs were found, and prevalence of ground squirrels at those sites.

Primary habitat	No. of breeding sites[a]	No. of sites where ground squirrel presence was assessed	Percentage of assessed sites with ground squirrels present
Irrigation canal[b]	383	285	19
Natural grassland	211	211	92
Idle or fallow field	121	103	76
Field crop	114	10	60
Pasture	100	100	87
Brushland	75	75	67
Airport	45	45	91
Golf course	30	30	100
Levee	27	26	92
Railroad	26	26	85
Grain or hayfield	25	21	57
Row crop	14	6	43
Other	116	107	48
Total	1,287	1,045	64

[a] In many cases breeding sites encompassed multiple Burrowing Owl pairs.

[b] Here the term "irrigation canal" is used broadly to indicate any man-made concrete or earthen water conveyance structure.

owls on those blocks during the 2006-2007 survey increased as a function of the number of owls detected during the 1991-1993 survey (Fig. 14; $\chi^2_1 = 12.41$; $P = 0.0004$). For example, the predicted probability of detecting owls during the 2006-2007 survey in blocks where just one pair was detected during the 1991-1993 survey was about 0.36, compared to 0.93 in blocks where 25 pairs of owls were detected during the 1991-1993 survey.

DISCUSSION

Our survey method likely contains some systematic sources of error. As DeSante et al. (2007) pointed out, the inability of observers to

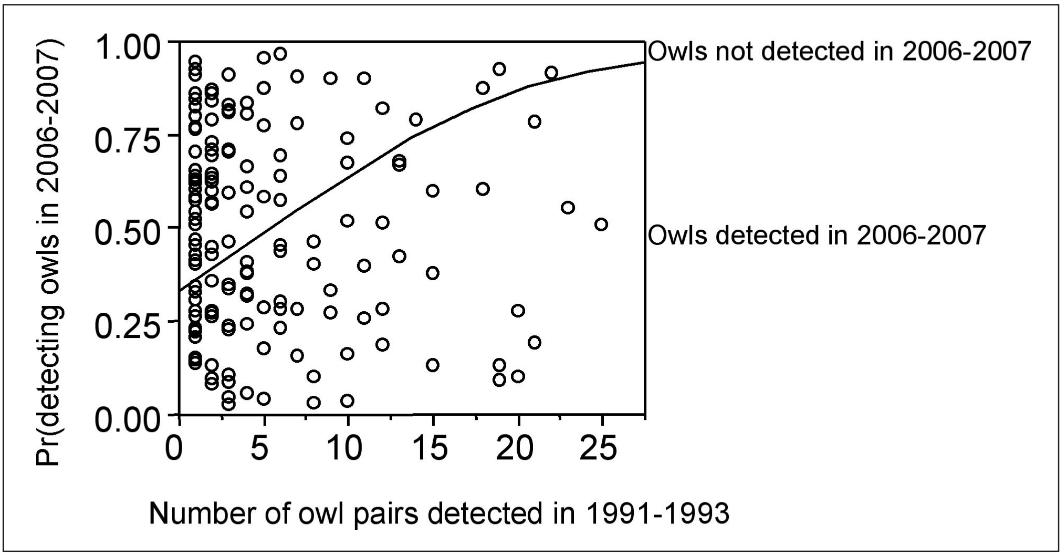

FIGURE 14. Probability of detecting owls during the 2006-2007 survey in blocks where owls were detected in 1991-1993 as a function of the number of owl pairs detected on the block in 1991-1993. The predicted probability of detection in 2006-2007 is shown by the curve. Data points below the curve are blocks on which owls were detected during both surveys; data points above the curve represent blocks where owls were detected in 1991-1993 but not detected in 2006-2007. Data points are plotted at their 1991-1993 owl pair (x-axis) values and randomly jittered in the probability (y-axis) space (below or above the curve, depending on whether owls were present in 2006-2007) to show the distribution of the data.

reliably detect all owls in sampled areas (Conway and Simon 2003, Conway et al. 2008), particularly in areas with limited or no road access may tend to bias our estimates low. Additionally, observers generally assumed that whenever they detected a single adult Burrowing Owl, it represented a breeding pair. To the extent that unmated adult birds may have been detected, this could result in an upward bias in our estimate of breeding pairs. Another potentially confounding factor was that surveyors were unable to gain access to some military installations and private landholdings; if such areas were more or less likely to be occupied by owls than other areas, bias in one direction or the other could have been introduced into our estimates. Finally, our survey methodology incorporated no means for assessing detection probability, which in some environments (such as desert areas with very low road density) may have been quite low. Perhaps of even greater concern than detection probability being low is that it could have varied substantially across survey blocks or survey regions with different physiographic characteristics.

Nevertheless, we believe the sheer volume of data collected counterbalances some of the methodological limitations described above, and ensures that the broader patterns in distribution and abundance are meaningful. Additionally, because our methods adhered to those established by DeSante et al. (2007), any biases affecting our results likely affected the 1991-1993 study, too, so that comparisons between the two surveys are appropriate. Finally, our survey documented the exact locations of 1,758 Burrowing Owl pairs (18.9% of the estimated total) across California, information that should be of great use for ongoing and future conservation efforts.

The generally large variances associated with our regional and statewide population estimates extrapolated from randomly-selected blocks indicate that our statistical power to detect changes in abundance was rather weak. Indeed, the Northern Central Valley was the only region for which our 2006-2007 population estimate differed significantly from the 1991-1993 estimate of DeSante et al. (2007). Moreover, many of our regional "best estimates" were not obtained by extrapolating data from the randomly-selected blocks, but rather by simply counting all of the owl pairs that could be found in either randomly-selected or historical breeding blocks. We had no means for assessing statistical significance of such estimates from the corresponding 1991-1993 "best estimates", many of which were generated in the same manner. Nevertheless, inspection of our results, and qualitative comparisons with results from the 1991-1993 survey, still yield some important conclusions.

The major patterns in Burrowing Owl distribution and abundance across California described by DeSante et al. (2007) have not changed dramatically since 1991-1993, when the species was already extirpated or nearly extirpated from the San Francisco Bay Area Coast, Central-western Coast, and Southwestern Coast regions. The Imperial Valley still accounts for slightly more than two-thirds of the estimated statewide population, and the Southern Central Valley remains the second largest Burrowing Owl population center. Populations in other regions of the state that were surveyed in 1991-1933 all remain much smaller than those in the two most heavily populated regions.

While not statistically significant, we observed apparent declines in two urban areas: San Francisco Bay Area Interior Region and the Bakersfield area in the Southern Central Valley region. The San Francisco Bay Area Interior region's breeding owl population is both small and well-known by local birders and researchers, making it very likely that the "best estimates" from both the 1991-1993 and 2006-2007 surveys reflect very nearly all the owl pairs actually present. Consequently, the apparent loss of 27.9% of the population, from 165 to 119 pairs since the early 1990s survey, is somewhat alarming. This loss includes the last known pairs of owls in both Sonoma and San Mateo counties, and suggests that Burrowing Owls have now been extirpated as a breeding species in the entire San Francisco Bay Area, except for Alameda and Santa Clara counties, where populations have also declined. It should be noted that Burrowing Owl populations can fluctuate annually, so our lower count of owls in the region does not necessarily indicate a deterministic decline. However, the increasingly restricted distribution of the species throughout the region would seem to indicate that such a trend is real.

In the greater Bakersfield area, heavy losses (nine blocks lost a total of 96 pairs) appear to be associated with recent land conversion from agriculture to urban, though a finer resolution spatial assessment would be helpful to determine whether such land conversion really has driven the losses. In any case, it seems that like the San Francisco Bay Area, the greater Bakersfield area is in danger of losing most if not all of its once substantial Burrowing Owl population. This is particularly unfortunate because the species exhibits a remarkable degree of tolerance for human alteration of natural habitats (Klute et al. 2003, Chipman et al. 2008), often nesting within landfills, golf courses, airports, and vacant lots within urban areas (Haug et al. 1993, Trulio 1997). This tolerance of humans and their activities would seem to provide ample opportunity for successful conservation efforts, even in the context of urban areas with growing human populations. One result, showing that the likelihood of Burrowing Owls persisting through 2006-2007 on survey blocks where they were present in 1991-1993 was strongly and positively related to the number of owls that were present on the blocks in 1991-1993, underscores the precariousness of dwindling urban-area populations, and the need for rapid action to prevent local extirpation.

In contrast to areas where we noted declines, we also noted areas where Burrowing Owls may have increased since the 1991-1993 survey: the Central-Western Interior region and the Coachella Valley. However, we surveyed a much greater number of upland blocks in contrast to the earlier survey in the Central-Western Interior region, so the apparent increase could be an artifact due to increased surveys effort. In contrast, the apparent increase (from zero to 53 owl pairs) in the Coachella Valley seems more likely to indicate a real increase in owl presence, especially because we found multiple Burrowing Owl pairs on four blocks in the region that were also surveyed in the early 1990s, but yielded no detections at that time. Interestingly, none of the pairs we found in Coachella Valley appeared to be associated with agriculture or water conveyance structures;

rather they occupied a variety of relatively arid habitats including brushland, desert scrub, and natural grasslands, and appear to be clustered on the outskirts of urban development.

Large confidence intervals make comparing our statewide population estimate with that of DeSante et al. (2007) during 1991-1993 difficult, especially since the difference in the estimates is relatively small. Three quarters of owl pairs in our aggregated population estimate reside in the densely occupied Imperial Valley, where the standard error associated with our regional estimate is well over 2,000 pairs. Thus, the lack of precision in this single regional estimate could easily mask a real statewide decline, or for that matter, potentially even obscure a statewide increase. Future survey efforts could perhaps minimize the problem of low statistical power by focusing monitoring efforts on smaller areas selected for high owl population density or other factors, and sustaining those efforts for multiple successive breeding seasons.

Our survey of the "new" survey regions covering the Modoc Plateau/Great Basin, Mojave Desert, and Sonoran Deserts represents the first systematic survey of Burrowing Owls across vast portions of California. We found Burrowing Owls to be distributed heterogeneously among these regions, with few or no owls in the Modoc Plateau/Great Basin, Northern Mojave/Eastern Sierra Nevada, Eastern Mojave, or Sonoran Desert regions (excluding the Palo Verde Valley). However, we found much larger aggregations of burrowing Owls in the Western Mojave region, and in one small area of the Sonoran Desert—the Palo Verde Valley.

CONSERVATION IMPLICATIONS

A comprehensive conservation strategy for Burrowing Owl in California is under development by California Department of Fish and Game and its partners (Burkett and Johnson, 2008). Here we provide a few conservation-related conclusions and recommendations that stem directly from our results:

1) Despite the apparent robustness of the population in the Imperial Valley, smaller populations elsewhere in the state, particularly in and near urban areas, appear to have continued to decline since the 1991-1993 survey.

2) The vast majority of the state's breeding Burrowing Owls continue to nest on private lands; any meaningful conservation efforts must therefore engage private stakeholders.

3) Across much of California, Burrowing Owl nesting remains closely associated with the presence of ground squirrels, another factor that must be considered in developing successful conservation measures.

4) In a few key areas, particularly the Imperial Valley and the Palo Verde Valley, Burrowing Owls are not closely associated with ground squirrels, and instead rely heavily on the banks of concrete and earthen water conveyance structures for nesting sites. Comprehensive conservation planning for Burrowing Owl in California must take into consideration the importance of these artificial structures.

5) Although Burrowing Owl detections were scarce across most of the land area of the newly surveyed Modoc Plateau/Great Basin and southern California desert regions, substantial populations persist in the Sonoran Desert (Palo Verde Valley) and the western Mojave Desert regions (particularly in and around the Antelope, Apple, and Lucerne valleys). We estimate the western Mojave Desert region to contain ~6% of California's breeding Burrowing Owls, superseded in numerical importance to the statewide population only by the Imperial Valley and the Southern Central Valley regions. Successful conservation planning for this species must address the particular needs of these substantial desert populations (Wilkerson and Siegel, *in press*).

6) A statewide conservation strategy will likely need to incorporate a statewide monitoring program to assess the effectiveness of conservation measures. Our study demonstrates the potential value of citizen-science participation in single-species studies, particularly of raptors or other highly charismatic species like Burrowing Owls that are relatively easy to find and identify. While many of our volunteer observers were highly skilled birders, and in some cases, even wildlife professionals, others had little or no birding experience. With a fairly modest investment of time and money for recruiting, training, and supporting volunteer surveyors, we were able to extend our survey across a vast area. Engaging citizen-scientists in monitoring could reduce the cost and extend the scope of any owl monitoring project, and may also yield less tangible benefits — participants in

citizen science monitoring programs can reap an increased awareness and appreciation of study organisms and their habitats, which may then translate into tangible actions on their behalf (Evans et al. 2005).

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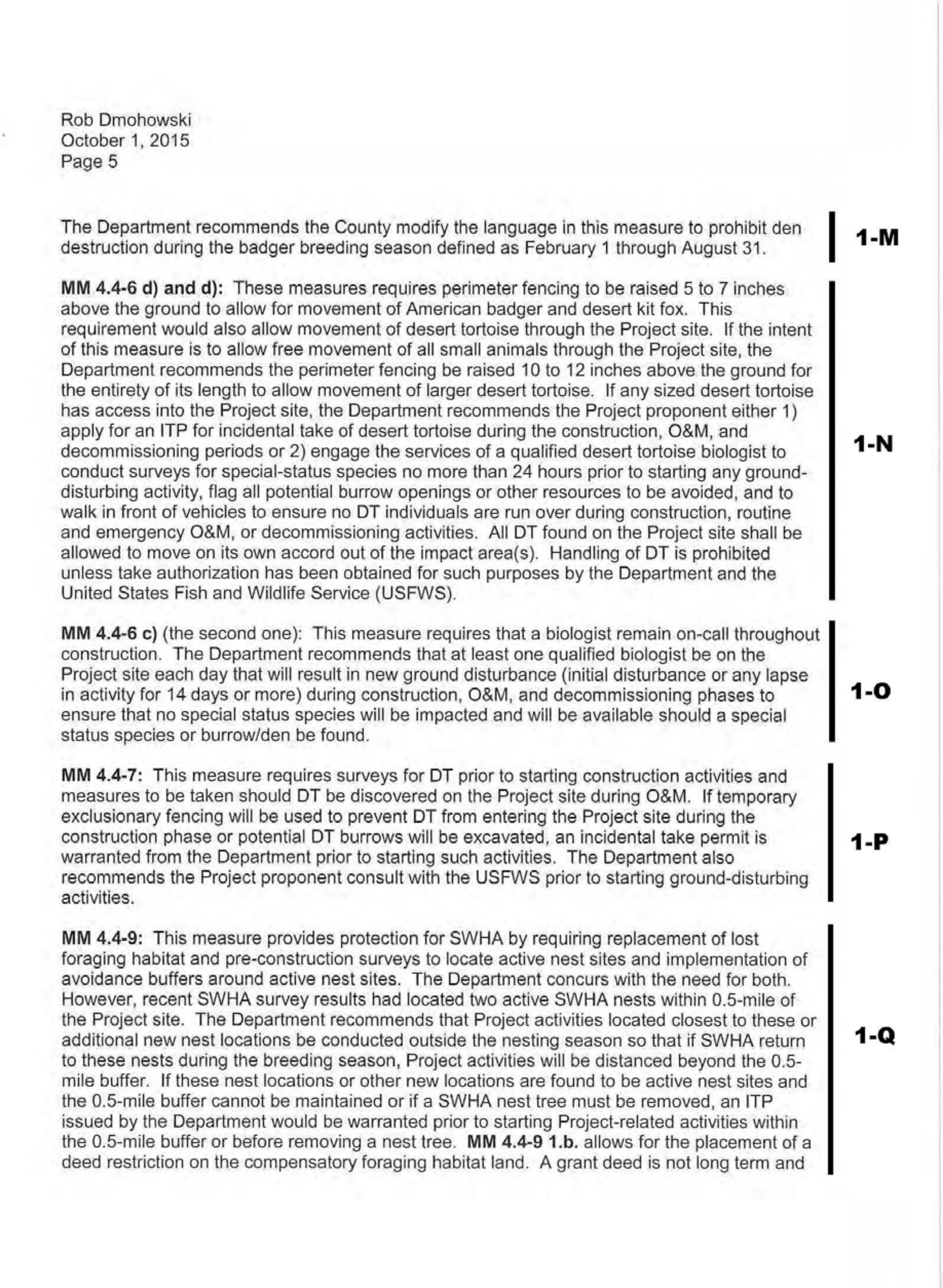

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The Department recommends the County modify the language in this measure to prohibit den destruction during the badger breeding season defined as February 1 through August 31. **1-M**

MM 4.4-6 d) and d): These measures requires perimeter fencing to be raised 5 to 7 inches above the ground to allow for movement of American badger and desert kit fox. This requirement would also allow movement of desert tortoise through the Project site. If the intent of this measure is to allow free movement of all small animals through the Project site, the Department recommends the perimeter fencing be raised 10 to 12 inches above the ground for the entirety of its length to allow movement of larger desert tortoise. If any sized desert tortoise has access into the Project site, the Department recommends the Project proponent either 1) apply for an ITP for incidental take of desert tortoise during the construction, O&M, and decommissioning periods or 2) engage the services of a qualified desert tortoise biologist to conduct surveys for special-status species no more than 24 hours prior to starting any ground-disturbing activity, flag all potential burrow openings or other resources to be avoided, and to walk in front of vehicles to ensure no DT individuals are run over during construction, routine and emergency O&M, or decommissioning activities. All DT found on the Project site shall be allowed to move on its own accord out of the impact area(s). Handling of DT is prohibited unless take authorization has been obtained for such purposes by the Department and the United States Fish and Wildlife Service (USFWS). **1-N**

MM 4.4-6 c) (the second one): This measure requires that a biologist remain on-call throughout construction. The Department recommends that at least one qualified biologist be on the Project site each day that will result in new ground disturbance (initial disturbance or any lapse in activity for 14 days or more) during construction, O&M, and decommissioning phases to ensure that no special status species will be impacted and will be available should a special status species or burrow/den be found. **1-O**

MM 4.4-7: This measure requires surveys for DT prior to starting construction activities and measures to be taken should DT be discovered on the Project site during O&M. If temporary exclusionary fencing will be used to prevent DT from entering the Project site during the construction phase or potential DT burrows will be excavated, an incidental take permit is warranted from the Department prior to starting such activities. The Department also recommends the Project proponent consult with the USFWS prior to starting ground-disturbing activities. **1-P**

MM 4.4-9: This measure provides protection for SWHA by requiring replacement of lost foraging habitat and pre-construction surveys to locate active nest sites and implementation of avoidance buffers around active nest sites. The Department concurs with the need for both. However, recent SWHA survey results had located two active SWHA nests within 0.5-mile of the Project site. The Department recommends that Project activities located closest to these or additional new nest locations be conducted outside the nesting season so that if SWHA return to these nests during the breeding season, Project activities will be distanced beyond the 0.5-mile buffer. If these nest locations or other new locations are found to be active nest sites and the 0.5-mile buffer cannot be maintained or if a SWHA nest tree must be removed, an ITP issued by the Department would be warranted prior to starting Project-related activities within the 0.5-mile buffer or before removing a nest tree. **MM 4.4-9 1.b.** allows for the placement of a deed restriction on the compensatory foraging habitat land. A grant deed is not long term and **1-Q**

can be changed or eliminated with ease. The Department recommends that the County remove this as a possible option with regard to the long-term protection of compensatory SWHA foraging habitat. The Department also recommends the County require that the SWHA foraging habitat mitigation be completed prior to starting Project activities unless an irrevocable letter of credit of other acceptable form of security is provided. **1-Q**

MM 4.4-13: This measure requires development of a Raven Management Plan and contribution to the Raven Management Fund. The Department recommends that the County modify this measure to require the Plan and funding occur prior to the start of Project activities. **1-R**

The Project site is in the range of MGS. MGS surveys were conducted in 2010/2011. However, these MGS surveys were not conducted to acceptable CDFW protocol and as a result, it is not clear that the survey effort would have detected MGS on site, even if present. Further, these survey results are no longer valid given they were conducted over 4 years ago. Portions of the Project (including the gen-tie routes) contain appropriate habitat for MGS; therefore, the Department recommends that the Project proponent take one of the following actions: 1) assume presence of MGS and obtain an ITP from the Department prior to starting construction; 2) identify all small mammal burrows that could potentially be occupied by MGS using flagging and avoid by at least 50 feet; or 3) conduct MGS protocol level surveys in the season prior to starting construction to determine presence or absence of MGS. Because the Project site is greater than 160 acres and the gen-tie routes are longer than 1 mile, the Project proponent will need to modify the CDFW-approved MGS survey protocol to accommodate the larger site size and submit it for review and approval by the Department prior to implementation. **1-S**

Thank you for providing the opportunity to comment on this renewable energy project. If you have any questions regarding these comments, please contact Lisa Gymer, Senior Environmental Scientist (Specialist), at the address provided on this letterhead, by telephone at (559) 243-4014, extension 238, or by electronic mail at Lisa.Gymer@wildlife.ca.gov. **1-T**

Sincerely,

Julie Vance
Regional Manager

cc: See Page Seven