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California Gnatcatchers and Coastal Sage Scrub: The Biological Basis for Endangered Species Listing

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Abstract. The California gnatcatcher's northernmost, nominate subspecies (*Polioptila Californica californica*) is currently proposed for addition to the U.S. List of Endangered and Threatened Wildlife, primarily as a result of past and continuing habitat destruction. Restricted in the U.S. to the arid coastal lowlands between Los Angeles and San Diego, California gnatcatchers occur on some of the most expensive undeveloped real estate in the nation. Consequently, efforts to provide legal protection for this species have engendered substantial controversy between conservationists and the southern California building industry. This paper summarizes the biological data that formed the basis of the U.S. Fish and Wildlife Service's decision to propose listing the gnatcatcher as endangered. Although more complete information concerning the ecology and distribution of the species is now being collected, none of these studies have yet been published in peer-reviewed, technical journals. These additional data will no doubt eventually prove valuable in developing conservation plans for gnatcatchers and coastal sage scrub. However, there is no indication that pending results will negate currently available evidence which supports the contention that *Polioptila c. californica* should be afforded protection under federal and state endangered species laws.

Keywords: California gnatcatcher; coastal sage scrub; endangered species; habitat fragmentation; habitat loss; *Polioptila californica*; urban development.

Introduction

During fall 1990 a small gray songbird, the California gnatcatcher (*Polioptila californica*), became the focal point of a major debate that is still continuing between conservationists and the southern California building industry. At issue in this confrontation is whether or not the gnatcatcher's northernmost, nominate subspecies should receive formal protection under federal or state endangered species laws. Some conservationists have considered listing of the gnatcatcher as an endangered species to be their best hope for slowing rampant urban growth between Los Angeles and San Diego. Representatives of the southern California building industry, on the other hand, have described the gnatcatcher as their "worst enemy," and have predicted "economic meltdown" should endangered species protection be granted under state or federal laws.

Such rhetoric from both sides of this argument has frequently obscured the fact that effective conservation of the California gnatcatcher will ultimately depend on management policies developed through objective analysis of basic scientific questions - rather than emotional charges and counter-charges. The gnatcatcher should not be viewed as a tool for stopping urban growth or blocking particular development projects. Neither is it reasonable to claim that protection of this species would pose a serious threat to the economic stability of southern California. Instead, solution of "the gnatcatcher problem" should emphasize application of technical information concerning the species' distribution, ecology, and behavior to regional land use planning efforts aimed at maintaining stable populations. Whether or not such planning can occur without the legal imperatives provided through formal listing of the gnatcatcher as endangered remains to be seen.

In this paper I summarize the main biological data that formed the basis of endangered species petitions submitted by the Natural Resources Defense Council (NRDC) and Manomet Bird Observatory (MBO) to the U.S Fish and Wildlife Service and the State of California Fish and Game Commission. Much of this information was contained in Atwood (1990), an unpublished report that, although widely circulated among the southern California building industry, has not been readily accessible to the scientific community. Data contained in this document were central to the U.S. Fish and Wildlife Service's decision to propose addition of the California

gnatcatcher's nominate subspecies to the U.S. List of Endangered and Threatened Wildlife (Salata 1991b).

Following submission of the NRDC - MBO petitions, new information concerning gnatcatcher distribution, behavior, and ecology began to be collected by various consulting firms employed by the southern California building industry. Although the stated goal of these studies was to ensure that "good science" would form the basis of the Fish and Wildlife Service's final rule-making, many conservation groups instead believed that this was "research with an agenda" - namely, to discredit the findings of Atwood (1990) and thereby defeat any effort to list the gnatcatcher as endangered. As of October 1992, none of these "proprietary" data have been published in peer-reviewed, technical journals. Consequently, it remains difficult to assess the scientific validity of many of these studies, or to include their conclusions in any review of the California gnatcatcher's status.

Therefore, the information presented here is, by necessity, admitedly incomplete. It is hoped that eventually the substantial body of new results that have been collected since 1990 will be published and made available to public review, thereby augmenting the data upon which the NRDC - MBO petitions were based.

Methods

Distributional records and information regarding gnatcatcher ecology and behavior were collected from literature accounts, job reports prepared by various environmental consulting firms, notes of currently active field ornithologists, the Natural Diversity Data Base of the California Department of Fish and Game, data on file at the southern California field office of the U.S. Fish and Wildlife Service, and museum collections. Unpublished reports produced since January 1991, describing preliminary results of work currently in progress, are not included in this review.

Taxonomy and General Distribution

Species Limits

The genus *Polioptila* is a distinctive, easily recognized group of small (6-7 gram), long-tailed, insectivorous birds. Distributed throughout the Neotropics, the genus shows little phenetic divergence among its component species; all gnatcatchers are predominately gray, with white outer tail feathers and varying amounts of black on the head. Because the California gnatcatcher (*P. californica*) is morphologically similar to the black-tailed gnatcatcher (*P. melanura*) found in the desert regions of the southwestern United States and Mexico, only recently have the two species been shown to be specifically distinct (Atwood 1988).

Polioptila californica was originally described as a distinct species (Brewster 1881). Later, and despite recognition that the vocalizations of *P. californica* and *P. melanura* are easily distinguished (Grinnell 1904), Grinnell (1926) proposed that the two forms be considered conspecific based on the morphological similarity of birds from the Cape region of Baja California to those from the Sonoran desert regions of Arizona and southeastern California. The Fourth and Fifth Editions of the A.O.U. Check-List (American Ornithologists' Union 1931, 1957) followed Grinnell's (1926) treatment, and considered gnatcatchers occurring in coastal southern California to be a subspecies of *P. melanura* (*Polioptila m. californica*); this population was often referred to by the common names "Coastal black-tailed gnatcatcher" or "California black-tailed gnatcatcher" to distinguish it from *P. melanura lucida* of the Sonoran desert (McCaskie and Pugh 1964; Atwood 1980). In 1982 the U.S. Fish and Wildlife Service designated *Polioptila melanura californica* as a Category 2 candidate for addition to the List of Endangered and Threatened Wildlife (Federal Register 47: 58454), and in subsequent Notices of Review this taxon was retained in this Category (Federal Register 50: 37958, Federal Register 54: 554).

My taxonomic review of the gnatcatchers breeding in North America's arid regions showed that *P. californica* was specifically distinct from *P. melanura* (Atwood 1988). This conclusion was based on vocal differences, the results of playback experiments, and the existence of assortative mating in a limited area of Baja California where both vocal types occurred sympatrically. Subtle morphological differences also existed between the two song types. This revision was subsequently followed by the A.O.U. Check-List Committee (American Ornithologists' Union 1989), and represents the presently accepted scientific opinion regarding the taxonomic status of the California gnatcatcher at the species level.

Although *P. californica* and *P. melanura* occur sympatrically in limited areas of northeastern Baja California , the ranges of the two species are quite distinct (Atwood 1988). California gnatcatchers are distributed from coastal southern California south in the lowland areas of Baja California to Cabo San Lucas, and black-tailed gnatcatchers occur in the Sonoran and Chihuahuan deserts of the southwestern United States, northeastern Baja California, and mainland Mexico (Atwood 1988; American Ornithologists' Union 1989).

Subspecies Limits

The most recent actions of the A.O.U. Check-List Committee have not formally addressed questions regarding subspecific limits in North American birds (American Ornithologists' Union 1983). The Fifth Edition of the A.O.U. Check-List (American Ornithologists' Union 1957) recognized three subspecies within what is now considered representative of the California gnatcatcher: *P. c. californica* (occurring from coastal southern California south in northwestern Baja California to 30° N latitude), *P. c. pontilis* (central Baja California, specifically from 28° 30' N south to 27° N latitude), and *P. c. margaritae* (Cape region of Baja California south of 27° N latitude). Atwood (1988) suggested that only two subspecies of California gnatcatcher should be recognized: *P. c. californica*, distributed from southern California south to the Cape region of Baja California, and *P. c. margaritae*, distributed south of approximately 28° N latitude.

Later, however, I retracted this conclusion regarding subspecific limits within *Polioptila californica* (Atwood 1991). Based on a reanalysis that was prompted by statistical questions raised by Johnson (1989) and Banks (1989), I found that California gnatcatchers distributed north of 30° N are, as has been consistently maintained since Grinnell's (1926) revision, adequately distinct morphologically from populations occurring farther south to warrant recognition of a subspecific boundary near El Rosario (Atwood 1991). Patterns of variation shown by specimens included in my 1988 analysis indicate that the southern distributional limit of *P. c. californica* does not extend south of 30° N latitude (Atwood 1991).

The estimated historic range of *P. c. californica* in the U.S. thus represents approximately 40 percent of the subspecies' total latitudinal range (34° 15' N south to 30° 00' N latitude), and approximately 50 percent of the linear extent of the subspecies' range when crudely measured along the coastlines of southern California and Baja California. Comparison of the actual historic extent of *P. c. californica*'s habitat in southern California vs. Baja California can only be approximate, because the species' distribution has always been somewhat discontinuous (Grinnell 1898; Grinnell and Miller 1944; Atwood 1980). Based on elevation contours that approximately delineate those areas that might potentially have supported California gnatcatchers prior to human development activities, Atwood and Bolsinger (1992) concluded that approximately 65-70 percent of the historic range of *P. c. californica* may have occurred in the U.S. as opposed to Mexico. Regardless of whatever patterns of gnatcatcher abundance might currently exist in the two countries, there is strong evidence that most of the range of the nominate subspecies was historically located in southern California rather than Baja California.

Ecology and Behavior

Habitat characteristics

In the U.S., California gnatcatchers are ecologically restricted to a plant community known as coastal sage scrub (Atwood 1980, 1988). Although gnatcatchers occassionally occur in riparian or chaparral habitats adjacent to coastal sage scrub, especially during the nonbreeding season, nearly all nesting records come from coastal sage scrub vegetation (Atwood 1980, 1988). Woods (1928) early noted that "probably no other California bird is so strictly confined to the brushlands as [the California gnatcatcher, which] . . . almost invariably turns back when it reaches the limits of the natural vegetation."

The floristic composition of coastal sage scrub shows substantial geographic variation, although in general it is dominated by aromatic, waist-high, drought-deciduous species such as *Artemisia californica*, *Salvia mellifera*, *S. leucophylla*, *S. apiana*, *Encelia californica*, and *Eriogonum fasciculatum* (O'Leary 1990). Four major associations have been recognized within the vegetation type: Diablan (San Francisco region south to northern Santa Barbara County), Venturan (northern Santa Barbara County south through coastal Los Angeles County), Riversidian (inland Los Angeles County, western Riverside County, and inland San Diego County), and Diegan (Orange County and coastal San Diego County south into northwestern Baja California to El Rosario) (O'Leary 1990). Even within these broad categories coastal sage scrub is often highly diverse; DeSimone (1989) identified 5 distinct subassociations within a limited area of Diegan coastal sage scrub in Orange County. Although California gnatcatchers occur only in the Venturan, Riversidian, and Diegan associations of coastal sage scrub, not all areas within each of these broad associations support gnatcatchers.

California gnatcatchers do not appear to be obligately dependent on any particular plant species found in coastal sage scrub. However, certain subassociations may represent habitats that are marginal or unsuitable for gnatcatchers. For instance, Bontrager (1991) found that areas of coastal sage scrub dominated by *Salvia mellifera* were not occupied, and that gnatcatchers were similarly absent if "a major component of the area was large woody shrubs such as *Rhus integrifolia* and/or *Malosma laurina*."

Home range requirements

California gnatcatchers are permanent residents and generally appear to remain in the same approximate home range from year-to-year. Home ranges of banded, breeding individuals varied from 4.0-11.5 acres ($\bar{X}$=6.5, *S.D.*=2.2, *n*=12) near Mission Viejo (Bontrager 1991), and from 13.3-39.2 acres ($\bar{X}$=23.8, *S.D.*=7.7, *n*=7) near El Cajon (ERCE 1990a). This variability probably reflects differences in habitat quality. Seasonal differences also exist in home range size. Bontrager (1991) found that of 11 pairs whose movements were mapped during breeding and non-breeding seasons, all showed larger home ranges during the non-breeding season, with increases varying from 2-163 percent.

Few quantitative studies of the specific habitat requirements of California gnatcatchers have been been made available, and all have been of narrow geographic scope. Limited data from vegetation transects conducted in known breeding home ranges are summarized in Table 1. At most of these sites *Artemisia californica* was a major component of the plant community. However, in the single inland study, located in Riverside County, *Artemisia* was poorly represented. In a more geographically extensive survey of habitat characteristics in which vegetation sampling was not based on precise home range boundaries, Anderson (1991) found that *Artemisia californica* was dominant in most coastal areas, with *Erigonum fasciculatum* and *Malosma laurnia* becoming more dominant in inland regions.

Table 1. Percent dominance of selected plant species in California gnatcatcher home ranges at six sites.

SPECIES	CP[a]	SD[b]	EC[c]	PE[d]	PV[e]	MV[f]
Artemisia californica	69	47	23	1	12	49
Eriogonum fasciculatum	-	27	53	2	-	37
Baccharis sp.	23	-	3	-	-	-
Malosma laurina	3	8	10	-	-	-
Viguiera laciniata	-	13	10	-	-	-
Encelia sp.	-	-	-	35	50	-
Salvia mellifera	-	-	-	11	-	-
Opuntia sp.	-	-	-	-	32	-
Other	6	4	2	51	6	14

[a] Camp Pendleton, San Diego Co.; Atwood (unpublished data).
[b] Rancho San Diego, San Diego Co.; Roach (1989).
[c] El Cajon, San Diego Co.; (ERCE 1990a).
[d] Perris, Riverside Co.; Tattersall (personal communication).
[e] Palos Verdes Peninsula, Los Angeles Co.; Atwood (unpublished data).
[f] Mission Viejo, Orange Co.; Bontrager (1991).

Dominant plant species recorded in mapped gnatcatcher home ranges are generally 1 m high or less (Table 2). In a study of seven breeding home ranges located near El Cajon, ERCE (1990a) found an average percentage gap in the shrub canopy of approximately 38 percent; Bontrager (1991) obtained a mean value of 34 percent based on 12 home ranges located near Mission Viejo.

Table 2. Height of dominant plant species in California Gnatcatcher home ranges.

	Mean Height (m)		
SPECIES	CP[a]	EC[b]	MV[c]
Artemisia californica	1.04	0.57	1.09
Eriogonum fasciculatum	-	1.02	0.89
Baccharis sp.	1.45	1.14	-
Malosma laurina	1.66	2.36	-
Viguiera laciniata	-	0.79	-

[a] Camp Pendleton, San Diego Co.; Atwood (unpublished data).
[b] El Cajon, San Diego Co.; (ERCE 1990a).
[c] Mission Viejo, Orange Co.; Bontrager (1991).

Coastal sage scrub is frequently subject to fire, especially in areas where the habitat occurs in close proximity to human population centers (Westman 1982; Keeley 1982). Data from Riverside County indicated that California gnatcatcher territories were located in areas of coastal sage scrub that had burned 8 or 9 years previously; more recently burned areas, as well as older, unburned habitat, were unoccupied (E. Tattersall personal communication). WESTEC Services, Inc. (1987) stated that California gnatcatchers were "absent from areas burned within the last few years." Westman and O'Leary (1986) and Malanson and O'Leary (1982) showed that some coastal sage scrub species are more resilient to fire than others; and Anderson (1991) suggested that "high fire frequency may permanently alter the floristic composition and structure of a site including the extirpation of weak resprouters such as *Artemisia californica*." Anderson (1991) further noted that two sites which had burned 10 and 12 years previously were characterized by "sparse canopy cover" (35 and 24 percent); nonetheless, these areas were occupied by gnatcatchers.

Foraging Ecology

Several recent studies have provided information regarding details of California gnatcatcher foraging ecology. Roach (1989) found that both sexes fed primarily on arthropods gleaned from two plant species, *Eriogonum fasciculatum* and *Artemisia californica*. "Small" (Arachnida, Cicadellidae, Fulgaridae) and "medium" (Arachnida, Cicadellidae, Fulgaridae, Chrysomelidae) arthropods were significantly more abundant on *Artemisia* than on *Eriogonum*; numbers of "small" arthropods present on *Artemisia* declined from March to June, while numbers of medium arthropods on this plant species remained constant throughout the gnatcatcher breeding season (Roach 1989). No temporal changes were observed in the abundance of small or medium-sized arthropods sampled on *Eriogonum* (Roach 1989).

ERCE (1990a) similarly observed that California gnatcatchers foraged most frequently on *Artemisia* and

Eriogonum. *Artemisia* was used significantly more often than was *Eriogonum* in comparison to the two species' relative abundances on the study site. This study also found that *Viguiera laciniata* and *Baccharis sarothroides* were underutilized by California gnatcatchers relative to their occurrence on the study site. Mock et al. (1990) reported that *Salvia mellifera* was avoided by foraging gnatcatchers. While these data suggest that *Artemisia* is of major importance to foraging California gnatcatchers, and that some coastal sage scrub species may be avoided, interpretation of the results suffers from the narrow geographic limits and small sample sizes of these studies.

Nesting Behavior

In the U.S., the breeding season of California gnatcatchers extends from mid-February through July, with May 5 being the mean date for nest initiation based on analysis of museum egg collection data (Atwood 1988). Because renesting after initial failures is common, this calculated mean date for nest initiations probably exceeds the actual peak for the initial surge of breeding activity. Most initial nesting attempts by California gnatcatchers probably occur during mid-late March and early April (Roach 1989; Atwood unpublished data).

The outer dimensions of California gnatcatcher nests measure approximately 5 cm wide and 6 cm deep; the internal depth of the nest cup is approximately 4 cm (Roach 1989). Nests are composed of grasses, bark strips, small leaves, spider webs, down, and other miscellaneous materials (Woods 1949). Nest height is probably somewhat determined by the structure of vegetation available within a given breeding territory. Using egg collection data from a variety of southern California localities, I calculated a mean nest height of 1.04 m (Atwood 1988). Studies conducted in more geographically restricted areas have found mean nest heights of 0.66 m (Roach 1989, near Rancho San Diego), 0.87 m (E. Tattersall, personal communication, near Perris), 0.81 m (ERCE 1990a, near El Cajon), 0.61 m (J. Atwood unpublished data, Palos Verdes Peninsula), and 0.70 m (Bontrager 1991, near Mission Viejo).

Possible preferences for particular plant species as sites for nest placement have not been adequately studied. Egg collection records obtained throughout southern California indicate that nests were most frequently found in *Artemisia californica* , followed, in order of frequency, by *Salvia apiana*, *Salvia mellifera*, *Adenostoma fasciculatum*, and *Opuntia* (Atwood 1980). Roach (1989) found 33 percent of nests (n=30) to be located in *Eriogonum fasciculatum*, 23 percent in *Artemisia californica*, and 17 percent in *Viguiera laciniata*; *Eriogonum* and *Artemisia* are generally the most abundant plant species in the coastal sage scrub community found in the area of Roach's study (P. Mock personal communication). Bontrager (1991) found 60 percent of 52 nests near Mission Viejo to be located in *Artemisia californica*, 21 percent in *Eriogonum fasciculatum*, and 12 percent in *Salvia apiana*; the relative dominance of these three plant species on the study plot was 49 percent, 37 percent, and 4 percent, respectively.

Based on egg collection data, I calculated the mean clutch size of *P. c. californica* to be 3.84 (*S.D.* = 0.57, range = 2-5, mode = 4, n = 61) (Atwood 1988). Roach (1989) obtained a similar mean clutch size of 3.67 (*S.D.* = 0.61, n = 27) near Rancho San Diego. Near Mission Viejo, Bontrager (1991) calculated a mean clutch size of 3.88 (*S.D.* = 0.23, range = 3-5, n = 33).

Both pair members contribute to nest construction, which early in the breeding season may be protracted over an approximately 10 day period, but which may occur as rapidly as 4-5 days late in the season (Atwood unpublished data). Nests may be constructed and then abandoned, prior to laying, for no apparent cause (Woods 1949; Atwood unpublished data). Materials from these abandoned nests, or from nests that have been disturbed by predators, are frequently recycled in construction of a subsequent nest (Woods 1949; Roach 1989).

Both pair members incubate the eggs, which measure approximately 14 x 11 mm and weigh approximately 1 gram (Hanna 1934); incubation begins with the laying of the ultimate or penultimate egg (Bontrager 1991). During incubation under moderate temperature regimes, only the head and tail of the parent bird protrude above the rim of the deep nest cup; under high ambient temperatures, adults may stand over the eggs, sometimes with their wings spread, to provide shade (Woods 1928; E. Tattersall personal communication). Defense of the nest and young is conducted by both sexes (Woods 1949). The mean incubation period of California gnatcatcher eggs is approximately 14 days, followed by an approximately 16 day period when both parents care for the nestlings (Woods 1949; E. Tattersall personal communication; Roach 1989).

Published, quantitative studies of reproductive success in California gnatcatchers are few; however, rates of nest failure appear to be relatively high. Bontrager (1990) stated that the species has "very low nesting success rates. In 1990, only 5 of 12 intensively followed pairs (42 percent) successfully fledged young and for the most part this was only after several nesting attempts. The average number of nests built per pair was 3.6. One pair had seven nest failures and never had a successful nesting."

Woods (1949) stated that fledglings are tended by the parents for a period of about three weeks. However, ERCE (1990b) described color-banded juveniles fledged

in early May that remained with their parents through mid-September, and Bontrager (1991) found that "young birds accompanied their parents for 4 to 5 weeks."

No data have yet been published regarding California gnatcatcher dispersal behavior. In general, it is believed that the species' obligate association with coastal sage scrub renders it unlikely to disperse across barriers of non-coastal sage scrub habitat, especially non-native vegetation types such as characterize most housing development projects. Some native habitats, such as grasslands, may also act as effective barriers to dispersal. A real possibility exists that major roads and freeway systems may act to isolate some populations from one another, but no focused studies have yet been conducted.

Given the highly fragmented condition of coastal sage scrub in southern California, the lack of detailed data concerning gnatcatcher dispersal behavior may be the most important information gap that will need to be filled in order to develop effective conservation strategies for the California gnatcatcher. Several studies are currently in progress that will attempt to provide these crucial results.

Survivorship and sources of mortality

Data have not yet been published concerning annual survivorship of California gnatcatchers. However, the small body size of gnatcatchers may render them more vulnerable to short-term perturbations in weather conditions or resource availability than larger species. P. Mock (personal communication) observed substantial disappearances of color-banded California gnatcatchers following a several week period of cold, rainy weather, and attributed these disappearances to weather-related mortality. Walsberg (1990) found that the slightly smaller black-tailed gnatcatcher occasionally forms communal roosts at night, apparently as an adaptation to cold winter temperatures. These preliminary data suggest that short term weather fluctuations may occasionally be an important factor affecting the population dynamics of the California gnatcatcher.

Various native species are possible predators of gnatcatcher eggs or nestlings, including greater roadrunner (*Geococcyx californianus*), scrub jay (*Aphelocoma coerulescens*), common crow (*Corvus brachyrhynchos*), common raven (*Corvus corax*), cactus wren (*Campylorhynchus brunneicapillus*), opossum (*Didelphis marsupialis*), raccoon (*Procyon lotor*), gray fox (*Urocyon cinereoargenteus*), coyote (*Canis latrans*), bobcat (*Lynx rufus*), coachwhip (*Masticophis flagellum*), striped racer (*Masticophis lateralis*), gopher snake (*Pituophis melanoleucus*), rosy boa (*Lichanura trivirgata*), common kingsnake (*Lampropeltis getulus*), and southern alligator lizard (*Gerrhonotus multicarinatus*). In areas located near human development, nests or juveniles are probably taken by domestic or feral cats (*Felis domestica*). Rodents (wood rats, *Neotoma*; field mice, *Peromyscus*; Norway rats, *Rattus rattus*; house mice, *Mus musculus*) may also disrupt nesting attempts, either as predators or by usurping the nest cup itself (Atwood, unpublished data).

Population status and threats

Section 4 of the U.S. Endangered Species Act identifies five factors that may threaten the continued survival of a species, any one of which may justify formal protection of that species through its addition to the U.S. List of Endangered and Threatened Wildlife and Plants. These factors include: "(A) the present or threatened destruction, modification, or curtailment of its habitat or range; (B) overutilization for commercial, recreational, scientific, or educational puposes, (C) disease or predation, (D) the inadequacy of existing regulatory mechanisms, and (E) other natural or man-made factors affecting its continued existence." Of these, factors (A) and (D) clearly exemplify threats to the California gnatcatcher's continued existence in the U.S., and factor (C) represents a potential, although presently undocumented, threat to the survival of some populations. These three factors are described below in greater detail.

Threat due to habitat loss

Various plant ecologists have noted the decline of coastal sage scrub during recent times. Klopatek et al. (1979) concluded that "coastal sagebrush" present in 1967 showed a 37 percent decrease relative to its "potential" area. Hanes (1976) stated that "the Coastal Sage Scrub community is the most endangered vegetative type in Southern California due to the pressures of urbanization, flood control projects and rock quarries." Kirkpatrick and Hutchinson (1977) described coastal sage scrub as "one of the least known and fastest disappearing types of vegetation in California," and Axelrod (1978) observed that the community is "rapidly disappearing under spreading urbanization." Mooney (1977) noted that coastal sage scrub "often occupies choice development sites and is being destroyed over large areas of the state." Westman (1981a) calculated that coastal sage scrub in California had been "reduced to 10-15 percent of its former extent;" because this calculation presumably included the Diablan association that occurs in the comparatively undeveloped portion of coastal California north of Ventura County, the relative degree of coastal sage scrub loss in the southern California range of the California gnatcatcher may be even higher.

Westman (1987) believed coastal sage scrub to be "one of the most endangered habitat types in the nation." O'Leary (1990) concluded that "the present decade likely represents an 'eleventh-hour' period" for the "imperiled" plant community.

Coastal sage scrub has significantly declined even in the relatively unpopulated areas of northern Baja California. Bowler (1990) stated that habitat in this region is being "grazed, burned to increase grass production, converted to row crop agriculture, and graded for urban development and construction of beach houses." Rea and Weaver (1990) similarly noted that coastal sage scrub near Valle de las Palmas, Baja California, "has been seriously degraded by burning, grazing, and conversion to vineyards during the past two decades (Marcos Camacho, personal communication)." The San Quintin kangaroo rat, an endemic Baja California species that historically occurred in coastal lowlands from San Telmo to El Rosario, is nearly extinct as a result of habitat destruction caused by agricultural conversion (Best 1983).

Because of this widespread habitat loss in the U.S. and Mexico, many components of the coastal sage scrub community are known or suspected to be showing serious population declines. In addition to the California gnatcatcher, at least 97 species or subspecies that frequently occur in U.S. portions of coastal sage scrub have been listed as threatened or endangered, are considered candidates for listing by the U.S. Fish and Wildlife Service or the State of California Department of Fish and Game or are listed as species of special concern (Table 3).

The distribution of California gnatcatchers in the U.S. was somewhat localized even prior to the agricultural and urban development of southern California. Grinnell (1898) found the species to be "numerous in [the] San Fernando Valley and about Pomona and Claremont, but around Pasadena, which is between the two localities and apparently offers similar attractions, I have never seen but one specimen." Dawson (1923) stated that the species was "rather scarce and irregularly distributed." Woods (1949) observed that "although frequenting the environs of some of the most populous districts of the West, the [California gnatcatcher] remains one of the least familiar North American birds. For some reason this species is localized and comparatively few in numbers . . . even in the most favorable situations [it] is not abundant."

This patchy distributional pattern was very early accentuated by the agricultural and urban development of southern California. In reference to the similarly distributed coastal populations of cactus wren, Dawson (1923) observed that "all proper desert areas west of San Gorgonio Pass are being threatened sharply by the human invasion. Those joyous bits of desert 'washes' which the canyons of the San Gabriel Mountains shoot like arrows into the heart of the plains, have become . . . cluttered up with bungalows and chicken coops . . . The cactus wren has receded from many parts of the San Diego-Ventura section already, and is in danger of being altogether cut off." Grinnell (1923) described *P. c. californica* as a "common resident locally," but, only twenty years later, Grinnell and Miller (1944) noted that suitable habitat for the subspecies had been "somewhat reduced." Pyle and Small (1961) commented that "the California subspecies is very rare, and lack of recent records of this race compared with older records may indicate a drastic reduction in population." McCaskie and Pugh (1964) believed that the California gnatcatcher had "been driven from most of its former range along the coast of the region." Atwood's (1980) preliminary survey of the species' status in California concluded that "continued reduction of [*P. c. californica*'s] already limited habitat is almost certain, and warrants immediate concern for the survival" of the subspecies in the U.S.

California gnatcatcher declines have been most dramatic in Ventura, Los Angeles, San Bernardino and Riverside counties. Populations have also been diminished in Orange and San Diego counties, but vague locality descriptions of historic sites make it difficult to confidently identify areas where local extinctions have occurred. To facilitate discussion of the specific factors contributing to past and projected population declines, the following data are presented on a county-by-county basis. Specific localities of occurrence are listed in Appendix A.

Ventura County. Few records of California gnatcatchers exist from Ventura County. During the late 1800's and early 1900's, egg sets and specimens were collected along the Santa Clara River drainage near Santa Paula, Ventura, and Fillmore, as well as in the Simi Valley. As early as 1933, J. S. Appleton observed that cactus wrens, a "formerly common resident [of coastal sage scrub in] Simi Valley, southern Ventura County," had disappeared as a result of habitat loss caused by agricultural clearing (Rea and Weaver 1990).

There are no certain records of California gnatcatchers in Ventura County since 1924, although the species has been reported twice on recent Sespe Wildlife Area Christmas Bird Counts (Audubon Field Notes 23: 428, 1969; American Birds 30: 614, 1976). I questioned the validity of these records (Atwood 1980), and no subsequent observations have been made during counts conducted from 1980 - 1988. However, the Sespe Wildlife Area count circle is positioned such that uncertain localities of historic gnatcatcher occurrence near Fillmore are probably included within its boundaries. A slight possibility exists that very small numbers of California gnatcatchers still persist in the Santa Clara River drainage of Ventura County.

Table 3. Sensitive, threatened, and endangered species associated with the coastal sage scrub community of southern California.[a]

COMMON NAME	SCIENTIFIC NAME	U.S.[b]	CALIF[c]
Invertebrates			
Wright's checkerspot butterfly	*Euphydryas editha quino*		S
Hermes copper butterfly	*Lycaena hermes*	C2	
Palos Verdes blue butterfly	*Glaucopsyche lygdamus palosverdesensis*	E	
Dun skipper	*Euphyes vestris harbisoni*	C2	
Wandering skipper	*Pseudocopaeodes eunus eunus*	C2	
Coastal giant skipper	*Megathymus yuccae* ssp.		S
Yucca moth	*Tegeticula yuccasella*		S
Amphibians			
Arroyo southwestern toad	*Bufo microscaphus californicus*	C2	
Reptiles			
San Diego banded gecko	*Coleonyx variegatus abbotti*	C2	
San Diego horned lizard	*Phrynosoma coronatum blainvillei*	C2	S
Coronado skink	*Eumeces skiltonianus interparietalis*	C2	
Orange-throated whiptail	*Cnemidophorus hyperythrus*	C2	S
Coastal western whiptail	*Cnemidophorus tigris multiscutatus*	C2	
Coastal rosy boa	*Lichanura trivirgata rosafusca*	C2	
Coast patch-nosed snake	*Salvadora hexalepis virgultea*	C2	
Red-diamond rattlesnake	*Crotalus ruber*		S
Birds			
California horned lark	*Eremophila altestris actia*	C2	
Coastal cactus wren	*Campylorhynchus brunnecapillus couesi*	P	S
Coastal Claifornia gnatcatcher	*Polioptila californica californica*	PE	
Southern California rufous-crowned sparrow	*Aimophila ruficeps canescens*	C2	
Bell's sage sparrow	*Amphispiza belli belli*	C2	
Mammals			
Pacific little pocket mouse	*Perognathus longimembris pacificus*	C2	S
Los Angeles little pocket mouse	*Perognathus lonimembris brevinasus*	C2	S
Northwestern San Diego pocket mouse	*Chaetodipus fallax fallax*	C2	
Pallid San Diego pocket mouse	*Chaetodipus fallax pallidus*	C2	
Dulzura California pocket mouse	*Chaetodipus californicus femoralis*	C2	
Stephens kangaroo rat	*Dipodomys stephensi*	E	T
San Bernardino Merriam's kangaroo rat	*Dipodomys merriami parvus*	C2	
Southern grasshopper mouse	*Onychomys torridus ramona*	C2	
San Diego black-tailed jackrabbit	*Lepus californicus benettii*	C2	
Plants			
Nevin's barberry	*Mahonia nevinii*	C1	E
Davidson's bush mallow	*Malacothamnus davidsonii*		S
Cliff spurge	*Euphorbia miseria*		S
Coulter's matilija poppy	*Romneya coulteri*		S
Payson's jewelflower	*Caulanthus simulans*	C2	S
Succulent wallflower	*Erysimum suffrutescens* ssp. *suffrutescens*		S
Snake cholla	*Opuntia parryi* var. *serpentina*	C2	S
Golden-spined cereus	*Bergerocactus emoryi*		S
San Diego barrel cactus	*Ferocactus viridescens*	C2	S
Orcutt's spineflower	*Chorizanthe orcuttiana*	C1	E
San Fernando Valley spineflower	*Chorizanthe parryi* var. *fernandina*	C1	S
Parry's spineflower	*Chorizanthe parryi* var. parryi	C2	S

Table 3. Sensitive, threatened, and endangered species associated with the coastal sage scrub community of southern California (continued).[a]

COMMON NAME	SCIENTIFIC NAME	U.S.[b]	CALIF[c]
Plants (cont'd)			
Slender-horned spineflower	*Dodecahema leptoceras*	E	E
Conejo buckwheat	*Eriogonum crocatum*	C2	T
Aphanisma	*Aphanisma blitoides*	C2	S
Del Mar manzanita	*Arctostaphylos glandulosa* var. *crassifolia*	C2	
Western dichondra	*Dichondra occidentalis*		S
Peirson's morning-glory	*Calystegia peirsonii*	C2	
Santa Ana River woolystar	*Eriastrum densifolium* ssp. *sanctorum*	E	E
Palmer's grapplinghook	*Harpagonella palmeri*		S
Greene's ground-cherry	*Physalis greenei*		S
Crown beard	*Verbesina dissita*		T
Orcutt's bird's-beak	*Cordylanthus orcuttianus*	C2	S
Short-lobed broomrape	*Orobanche parishii* spp. *brachyloba*		S
San Diego thornmint	*Acanthomintha ilicifolia*	C1	E
Munz's sage	*Salvia munzii*		S
Willowy monardella	*Monardella linoides* ssp. *viminea*	C2	E
Pringle's manardella	*Monardella pringlei*		S
Laguna Beach dudleya	*Dudleya stolonifera*	C1	T
Santa Monica Mountains dudleya	*Dudleya cymosa* ssp. *ovatifolia*		S
Conejo dudleya	*Dudleya parva*	C1	S
San Gabriel Mountains dudleya	*Dudleya densiflora*	C1	
Orcutt's dudleya	*Dudleya attenuata* ssp. *Orcuttii*		S
Bright green dudleya	*Dudleya virens*		S
Sticky dudleya	*Dudleya viscida*	C1	S
Blochman's dudleya	*Dudleya blochmaniae* ssp. *blochmaniae*		S
Short-leaved dudleya	*Dudleya brevifolia*	C1	
Variegated dudleya	*Dudleya variegata*	C2	S
Many-stemmed dudleya	*Dudleya multicaulis*	C2	S
Verity's dudleya	*Dudleya verityi*	C2	S
Small-leaved rose	*Rosa minutifolia*		E
Braunton's milk vetch	*Astragalus brauntonii*	C2	S
Dean's milk vetch	*Astragalus deanei*	C2	S
California adolphia	*Adolphia californica*		S
Pringle's yampah	*Perideridia pringlei*		S
Mission Canyon bluecup	*Githopsis diffusa* ssp. *filicaulis*	C2	
San Diego County viguiera	*Viguiera laciniata*		S
Sea dahlia	*Coreopsis maritima*		S
San Diego bur-sage	*Ambrosia chenopodiifolia*		S
San Diego ambrosia	*Ambrosia pumila*	C2	S
Santa Susana tarweed	*Hemizonia minthornii*	C2	
Otay tarweed	*Hemizonia conjugens*	C2	E
Tecate tarweed	*Hemizonia floribunda*	C2	
Palmer's ericameria	*Ericameria palmeri* ssp. *palmeri*	C2	
Rush-like bristleweed	*Haplopappus junceus*		S
Orcutt's hazardia	*Hazardia orcuttii*	C2	S
Palmer's haplopappus	*Haplopappus palmeri* ssp. *palmeri*	C2	S
San Diego sand aster	*Corethrogyne filaginifolia* var. *incana*		S
Del Mar sand aster	*Corethrogyne filaginifolia* var. *linifolia*	C2	S
Plummer's baccharis	*Baccharis plummerae*		S
San Diego sagewort	*Artemisia palmeri*		S
Nevin's brickellia	*Brickellia nevinii*		S
Catalina mariposa lily	*Calochortus catalinae*		S
Shaw's agave	*Agave shawii*		S
Dehesa bear-grass	*Nolina interrata*	C1	E
Munz's onion	*Allium frimbriatum* var. *munzii*	C1	T
San Diego goldenstar	*Muilla clevelandii*	C2	S

[a] Based on lists included in Salata (1991a), California Department of Fish and Game (1991), and U.S. Fish and Wildlife Service (1991a,b).
[b] Federal categories: E = Endangered, PE = Proposed Endangered, P = Petitioned, C1 = Candidate, Category 1 (substantial information exists to support biological appropriateness of listing as threatened or endangered, but proposed rules not issued because of having been precluded by other listing activity), C2 = Candidate, Category 2 (inconclusive data exists that suggest listing as threatened or endangered may be appropriate).
[c] State of California categories: E = Endangered, T = Threatened, S = Sensitive species of concern.

Egg sets collected in 1902 and 1904 from the Simi Valley are the only records of California gnatcatchers from this portion of Ventura County. Suitable habitat in the Simi Valley may have been continuous in the early 1900's with coastal sage scrub located to the north along the Santa Clara River drainage. Alternatively, gnatcatchers from the abundant San Fernando Valley population may have occasionally colonized the Simi Valley by dispersing west over the relatively low (approx. 450 m) Santa Susanna Pass. If this later scenario is correct, the Santa Clara River population may have been largely isolated, even prior to extensive habitat loss, from the species' nearest occurrence in Los Angeles County. Certainly now, with the complete or near complete extirpation of California gnatcatchers from inland Los Angeles County, any unknown pairs that still exist in Ventura County are fully isolated from other southern California populations.

Los Angeles County. Records of California gnatcatchers in Los Angeles County indicate that the species was once common and widespread from the San Fernando Valley east along the base of the San Gabriel Mountains to Claremont, and at the lower elevations of the San Jose, Los Coyotes, and Palos Verdes Hills. No records are known from the Santa Monica Mountains, despite the fact that coastal sage scrub plant communities occur throughout this region. A single 1936 nest record from Mint Canyon, located near Solemint, suggests that the species may historically have straggled into this area when source populations existed in the San Fernando Valley or near Fillmore.

Actual numbers of California gnatcatchers present in Los Angeles County prior to the region's urban development are unknown. However, the magnitude of the species' decline in the county is probably reflected by the fact that over 96 percent of the total low elevation (less than 250 m) acreage in Los Angeles County that might historically have supported suitable coastal sage scrub has been largely or entirely developed. Based on relatively large numbers of specimens and egg sets collected in the early 1900's, California gnatcatchers must have been historically common near San Fernando, Azusa, and Claremont. There are no confirmed, post-1985 records of the species from any of these sites, although limited tracts of degraded coastal sage scrub still persist in these areas. The only "major" California gnatcatcher population known to remain in Los Angeles County is located on the Palos Verdes Peninsula, where fewer than 50 pairs were estimated to persist in 1990 (K. Garrett personal communication).

All California gnatcatchers known to remain in Los Angeles County are fully isolated from other portions of the species' range in southern California. Even on a local level, the Palos Verdes Peninsula population is highly fragmented, with most remnant patches of coastal sage scrub in this area being completely surrounded by housing developments. Although a detailed assessment of the population status and dynamics of gnatcatchers on the Palos Verdes Peninsula remains to be conducted, most fragments of habitat in this area probably support only 1-4 pairs (J. Atwood unpublished data). There is no possibility of recruitment to the Palos Verdes Peninsula population from other gnatcatcher source populations in southern California. Soule et al. (1988) postulated that for resident bird species occurring in chaparral and coastal sage scrub habitats, "immigration is virtually negligible if the development barrier is wider than 50 - 100 m, at least in the time scale of a century or so." Many pairs of gnatcatchers in southern California now occur on remnant patches of habitat that are isolated from neighboring sites by barriers of unsuitable habitat that are frequently many kilometers in width.

Orange County. Historic (pre-1960) records of California gnatcatchers in Orange County are few, being limited to the West Coyote Hills and Laguna. However, widespread observations made since since 1960 suggest that the species was historically distributed throughout much of Orange County at elevations less than 250 m. Recent observations are concentrated near Irvine, Mission Viejo, Dana Point, and Orange; however, gnatcatchers probably occur throughout much of the remaining undeveloped lowland areas of Orange County. Most of these areas are owned by large land development companies that have been generally uncooperative in providing details regarding the distribution of gnatcatchers on their properties. However, there appears to be a high degree of overlap between sites of known occurrence and those regions of the county expected to show extensive human population growth during the next 20 years (Fig. 1). Projected expansion of Orange County's transportation network will also impact remaining gnatcatcher populations through direct loss of habitat, growth inducement, and further fragmentation of remaining large tracts of coastal sage (County of Orange Environmental Management Agency 1989; U.S. Department of Transportation and San Joaquin Hills Transportation Corridor Agency 1990; P&D Technologies 1990).

Riverside County. Known sites of historic occurrence in Riverside County include Corona, Riverside, Menifee Valley, Mockingbird Canyon, Pedley, Cabazon, Valle Vista, Banning, the Jurupa and Box Springs Mountains, and (extralimitally) Palm Springs. Grinnell and Swarth (1913) described California gnatcatchers as "one of the most abundant species" near Valle Vista, but described their habitat as "an extensive area covered with the [coastal sage scrub] once so characteristic of the southern California valleys, and now so nearly disappeared through the clearing and cultivation of the land."

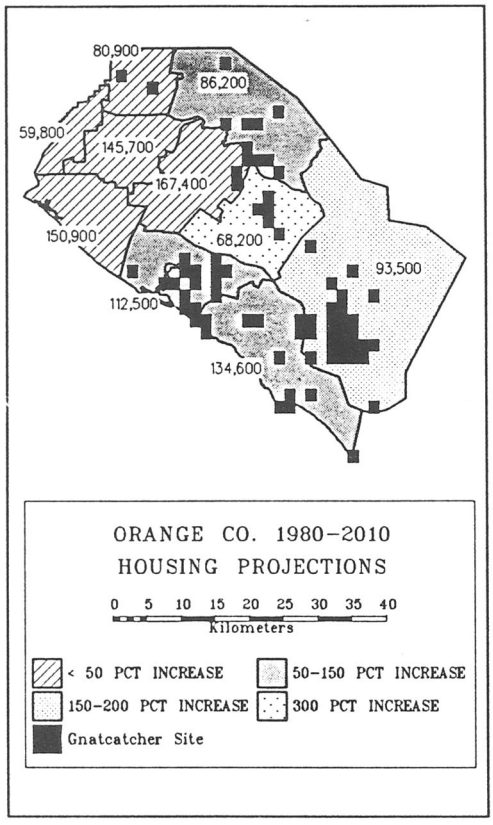

Figure 1. Projected housing growth in Orange County, relative to known sites of California gnatcatcher occurrence. Shading patterns indicate anticipated percent increases of housing units from 1980 - 2010. Numbers indicate total estimated housing units projected for each area by the year 2010 (County of Orange Environmental Management Agency 1985). Documented sites of gnatcatcher occurrence, indicated as solid squares, are located to the nearest 1-minute block of latitude-longitude.

Gnatcatchers have apparently been extirpated from the vicinities of Riverside, Banning, Cabazon, and Valle Vista (Atwood 1980). Recent records are concentrated south and west of Lake Matthews and northwest of Perris.

Several specimen records of California gnatcatchers were obtained in the early 1900's from Palm Springs, suggesting that the species may have regularly straggled east across the San Gorgonio Pass and into this area, where *Polioptila melanura* is common (Grinnell 1904; Atwood 1988). There is no evidence that Palm Springs was ever truly a part of the California gnatcatcher's regular geographic range. However, the occasional appearance of *P. c. californica* in the Palm Springs region may reflect historically higher population levels near Banning, Cabazon, and the San Jacinto Valley.

In comparison with other southern California counties, relatively extensive portions of western Riverside County still remain undeveloped. However, the Riverside County Planning Department (1990a) reported that as of January 1, 1990 [Riverside County] is the fastest growing county in California on a percentage basis." As a result of this increasing development pressure, A. Sanders (personal communication) estimated that less than 10 percent of the original coastal sage scrub community in Riverside County is still present. Furthermore, most presently known sites where California gnatcatchers occur are located in areas of Riverside County showing explosive rates of human population growth (Fig. 2). For example, in the Lake Matthews region, projected housing increases over the next 20 years exceed 315 percent (Riverside County Planning Department 1990a,b). Other areas are threatened by agricultural and mining pressures (L. LaPre personal communication).

San Bernardino County. Historic records of California gnatcatchers in San Bernardino County are known from the vicinities of Colton, Reche Canyon, and San Bernardino near Lytle Creek Wash. Confirmed recent observations of in San Bernardino County are limited to

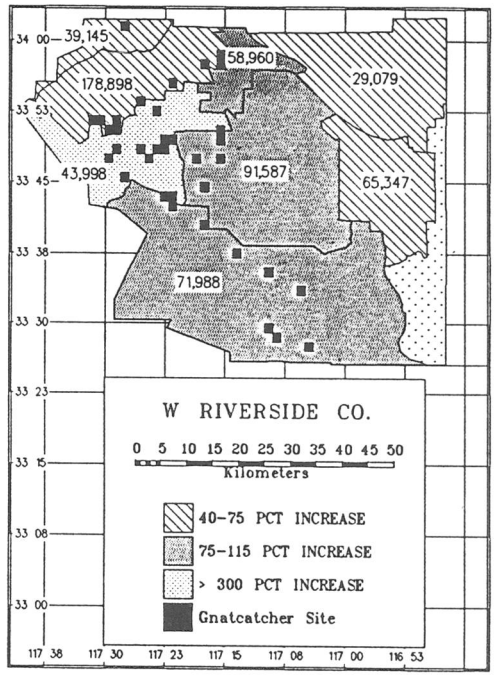

Figure 2. Projected housing growth in western Riverside County, relative to known sites of California gnatcatcher occurrence. Shading patterns indicate anticipated percent increases of housing units from 1990 - 2010. Numbers indicate total estimated housing units projected for each area by the year 2010 (Riverside County Planning Department 1990a,b). Documented sites of gnatcatcher occurrence, indicated as solid squares, are located to the nearest 1-minute block of latitude-longitude.

a single bird seen in 1990 near the confluence of Cajon Wash and Lytle Creek Wash (D. Willick personal communication). One or two California gnatcatchers were reported on four San Bernardino Valley Christmas Bird Counts during the late 1960's and early 1970's (Audubon Field Notes 23: 423-424, 1969; 24: 454, 1970; American Birds 25: 504-505, 1971; 27: 526-527, 1973). However, because of difficulties in field identification posed by the genus *Polioptila*, I consider these reports hypothetical. California gnatcatchers were not reported from the San Bernardino Valley Christmas Bird Count during the period 1973 - 1988.

These data indicate that California gnatcatchers have been largely or entirely extirpated from their areas of former occurrence in San Bernardino County. A few individuals may persist in the Lytle Creek Wash area and, possibly, near the Jurupa Mountains on the border between San Bernardino and Riverside counties. Like any unknown gnatcatcher populations that may persist in Ventura County or inland Los Angeles County, remnant populations in San Bernardino County, if they exist at all, must be extremely vulnerable to extinction due to their small size and complete isolation from larger, source populations.

San Diego County. California gnatcatchers were historically distributed throughout most of the lowland areas of San Diego County west of the Laguna Mountains. Early records were concentrated near San Diego, National City, the San Pasqual Valley, La Mesa, El Cajon, Bonita, and Escondido. Most of these regions still support some gnatcatchers, but extant populations are undoubtedly much reduced and fragmented in comparison to historic levels.

As in Orange and Riverside counties, most sites of recent gnatcatcher occurrence in San Diego County are concentrated in those areas projected to show large increases in human population by the year 2010 (Fig. 3). These include Fallbrook, Vista, San Marcos and Escondido (92.4 percent human population growth projected from 1986 - 2010), the North County West region, including Oceanside, Carlsbad, and Encinitas (66.5 percent projected growth), and the South Suburban region, including Chula Vista and Otay (63.2 percent projected growth) (San Diego Association of Governments 1989). Approximately 78 percent of the remaining coastal sage scrub in San Diego County, exclusive of Camp Pendleton, is privately owned (RECON unpublished data), and therefore subject to development pressures.

In contrast to other southern California counties, where virtually all important California gnatcatcher populations occur on private lands, several Federal properties in San Diego County support the species. These areas include Camp Pendleton Marine Corps Base, Fallbrook Naval Weapons Station, and Miramar Naval Air Station. Although protected from most of the housing development pressures faced by populations located on private lands, gnatcatcher populations on these military bases are still at risk. In 1989 and 1990 alone, approximately 15,000-20,000 acres of natural vegetation were burned on Camp Pendleton Marine Corps Base in fires started incidental to military training activities (D. Lawson personal communication). Although the location of these fires was such that only approximately 7 percent of the California gnatcatcher pairs present on this site were affected (L. Salata personal communication), the extent of the burns documents the vulnerability of habitat located on these military bases.

Baja California. There is no indication that the status of the California gnatcatcher's nominate subspecies in Baja California is significantly less threatened than in the U.S. In fact, land use changes along the coastal strip of Baja California between Tijuana and San Quintin closely mirror the threats to coastal sage scrub habitat that characterized southern California during the

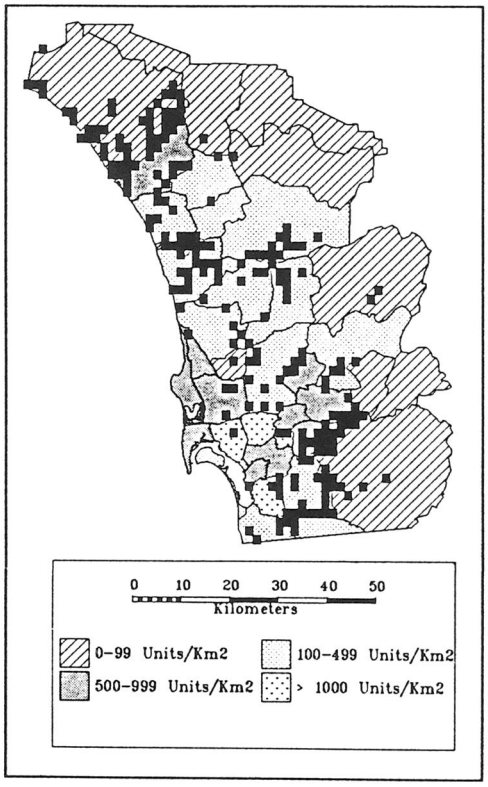

Figure 3. Projected housing growth in western San Diego County, relative to known sites of California gnatcatcher occurrence. Shading patterns indicate projected density of housing units for the year 2010 (San Diego Association of Governments 1988, 1989). Documented sites of gnatcatcher occurrence, indicated as solid squares, are located to the nearest 1-minute block of latitude-longitude.

1920's-1940's. Many areas of northwestern Baja California below 250 m have been, or are being, converted to grazing or agricultural uses (B. Massey personal communication). Urban areas near Tijuana and Ensenada are rapidly expanding into regions that had been cleared for agriculture in the last 25 years; 1990 census figures indicate that the population of Baja California (Norte) exceeds that of San Diego County, California's second most populous county. Furthermore, as U.S. citizens are faced with high land and building costs in southern California, increasing numbers of vacation homes are being built in Mexico by people whose primary residence is north of the border, thus extending the southern California housing boom into Mexico.

Population estimates. Atwood (1980) speculated that the number of California gnatcatchers remaining in the United States was "no more than 1,000 to 1,500 pairs," with specific estimates of 30 pairs in Ventura County, 130 pairs in Los Angeles County, 50 pairs in San Bernardino County, 325 pairs in Orange County, 400 pairs in Riverside County, and 400 pairs in San Diego County. These values, derived from reports of various observers, limited field work, and visual estimates of habitat availability coupled with the assumption that gnatcatchers were distributed throughout all subassociations of coastal sage scrub vegetation, were clearly identified as "preliminary" and "speculative" (Atwood 1980).

Atwood and Bolsinger (1992) showed that California gnatcatcher distribution may be defined by substantially lower elevational limits than previously thought. In coastal areas of southern California (Los Angeles, Orange, and San Diego counties), most records occurred at or below 250 m elevation; in inland regions (Riverside County), most sites of known occurrence were located between 250 - 500 m elevation (Atwood and Bolsinger 1992). Based on these results and the extent of undeveloped land present in the late 1970's, Atwood (1992) liberally estimated that approximately 1811 - 2291 pairs of California gnatcatchers might currently remain in southern California. This value agrees favorably with estimates, based on direct surveys of lands by biologists employed by the southern California building industry, of 1645 - 1880 pairs (Michael Brandman Associates, Inc. 1991).

Threat due to predation

Rates of passerine nest predation generally have been shown to increase as habitat fragments decrease in size, although none of these studies have specifically examined gnatcatchers or the coastal sage scrub ecosystem (Wilcove 1985; Small and Hunter 1989). Soule et al. (1988) noted that coyotes were absent from small, isolated patches of chaparral (including coastal sage scrub), and speculated that the disappearance of such large predators may allow population levels of smaller "bird predators," such as foxes, opossums or domestic cats, to increase. These authors suggested that increased predation pressures resulting from the absence of coyotes may contribute to local extinctions of bird species that occur in small, fragmented patches of native vegetation.

Brood parasitism of California gnatcatchers by brown-headed cowbirds (*Molothrus ater*) has been noted for many years (Woods 1930; Friedmann 1934; Hanna 1934), and has now been recorded in most of the counties where gnatcatchers historically occurred: San Diego County (Friedmann 1934; B. Jones personal communication; R. Patton, fide P. Mock; Atwood 1984), Orange County (J. Weintraub personal communication; Bontrager 1991), Riverside County (E. Tattersall personal communication), San Bernardino County (Hanna 1909) and Los Angeles County (Woods 1930). Such impacts may exacerbate population declines associated with habitat loss. Work in eastern deciduous forests has demonstrated that rates of cowbird parasitism are greater in small fragments of forest than in large, continuous tracts of habitat (Temple and Cary 1988; Robinson 1988); whether such a relationship also exists in different-sized fragments of coastal sage scrub is unknown.

Gnatcatchers are able to rapidly renest following the loss of eggs or juveniles to predators (Woods 1928; Bontrager 1990). However, based on duration of the incubation, nestling, and fledgling periods of brown-headed cowbirds (Bent 1958; Harrison 1978), a single instance of brood parasitism that proceeds to fledging of the juvenile cowbird may occupy at least 40 days. Assuming a potential breeding season of March 20 - June 15 (87 days), nearly 50 percent of the total nesting season available to a pair of gnatcatchers could potentially be occupied in raising a juvenile cowbird; depending on the date of nest initiation, subsequent nesting efforts during that season might be precluded. Assuming that California gnatcatchers are, like most species of small, temperate zone passerines, relatively short-lived, the cumulative effects of annual breeding failures caused by nest predation and cowbird parasitism may seriously impact the lifetime reproductive output of a specific pair. Especially in areas where remaining coastal sage scrub is highly fragmented, resulting in small gnatcatcher subpopulations being isolated from one another by barriers of unsuitable habitat, the effects of predation and brood parasitism may have serious consequences on long-term population viability.

Threat due to inadequacy of existing regulatory mechanisms

No regulatory mechanisms currently exist in the U.S. or Mexico that provide adequate protection for the California gnatcatcher's nominate subspecies. Under State law, the species has not, as of July 1992, been granted protection under the California Endangered Species Act. Although environmental impact documents for projects located on private lands are reviewed by the State Department of Fish and Game, compliance with Department recommendations is not required by the California Environmental Quality Act, and local jurisdictions often allow projects with adverse impacts to proceed on the basis of "overriding social and economic considerations."

As a result, regional land-use planning is frequently lacking, and there is little integration of the mitigation activities required for various projects. Local coordination of open space planning, without regulatory control by State or Federal wildlife agencies, usually results in incremental losses of habitat with serious cumulative effects. For example, the City of San Diego examined development impacts on native plant communities within its jurisdiction during the period 1985-1990. This study revealed a 97 percent loss of coastal sage scrub (384 of 395 acres) in conjunction with 15 projects; an additional eight projects, in which environmental reviews failed to distinguish between coastal sage scrub and chaparral, docmented a 95 percent loss (1,308 of 1,371 acres) of these two vegetation types (Salata 1991b).

Even when land-use planning attempts to address the needs of coastal sage scrub conservation, these efforts often fail to consider the specific habitat requirements of California gnatcatchers. For example, many tracts of coastal sage scrub that are included in proposed open space areas in Orange County are located above the 250 m elevation contour which crudely defines the distributional limit of gnatcatchers in this region (Fig. 4). Consequently, although many of these proposed parks and open space areas do protect coastal sage scrub as a general vegetation type, much of this habitat consists of subassociations that are marginal or unsuitable for gnatcatchers.

Typical of this problem are designs for the Foothill Ranch Planned Community which include dedication of approximately 40 percent of the property as Whiting Ranch Regional Park (Michael Brandman Associates, Inc. 1988). Approximately 30 percent of this proposed park has been identified as coastal sage scrub; however, most of this habitat occurs in proximity to chaparral rather than grassland areas, suggesting coastal sage scrub subassociations that are probably unsuitable or marginal for California gnatcatchers. Even with its sizable mitigation effort, construction of the Foothill Ranch Planned Community may result in elimination of prime gnatcatcher habitat and protection of areas that are only marginally suitable for the species.

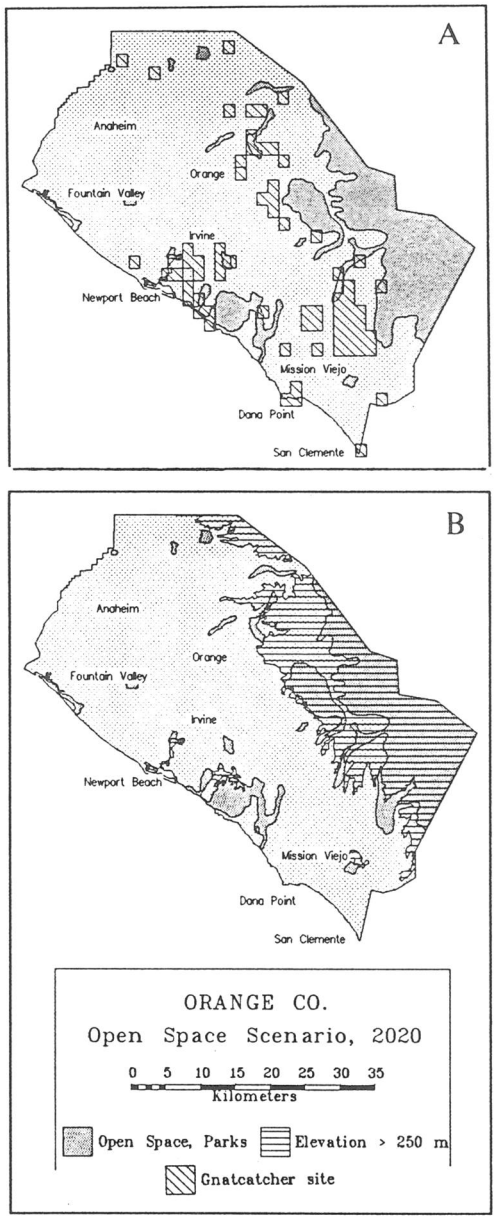

Figure 4. Approximate location of projected open space areas in Orange County relative to (a) known distribution of California gnatcatchers, and (b) 250 m elevation contour. Documented sites of gnatcatcher occurrence are located to the nearest 1-minute block of latitude-longitude.

Similar examples of inadequate or misdirected conservation efforts can be cited from many recent development proposals in southern California. Unfortunately, in some cases land developers have not made even token gestures to protect coastal sage scrub habitat, but instead have intentionally destroyed areas known to

support California gnatcatchers in apparent attempts to circumvent existing environmental review processes or avoid anticipated protection of the species under Federal or State endangered species laws (Salata 1991b).

Such habitat destruction is frequently conducted under the guise of agricultural clearing or weed abatement activities. Agricultural grading permits are not required in Orange or Riverside counties. The San Diego County Board of Supervisors, in adopting an ordinance imposing interim regulations on grading and clearing, stated that "clearing and illegal grading have been used to destroy environmental resources prior to application for a land development permit, during the permit process, after project approval but prior to the application of protecting open space easements, and after dedication of open space . . . Grading violations, when reported, result in relatively minimal fines and, because of the difficulty in obtaining convictions, are not a serious threat to illegal grading. A fine often will not prevent a violation of this ordinance because a fine may be considered simply as an additional development cost . . . Clearing for legitimate reasons (geotechnical exploration and access for percolation tests and wells, and clearing for fire protection) is frequently done well in excess of the minimum necessary to accomplish the purpose" (Salata 1991b).

Conclusions

There is little doubt that the California gnatcatcher's nominate subspecies is threatened by extinction throughout most of its range, and especially within the U.S. Extensive habitat loss has already occurred, and projected patterns of continuing urban development will result in increasingly fragmented and isolated populations. Such small, isolated populations are generally considered vulnerable to local extinction caused by a variety of stochastic events. Because of the species' specialized habitat requirements, dispersal among isolated populations is probably already being hindered.

Existing land-use regulations in southern California have failed to provide effective conservation of gnatcatcher habitat. Without a dramatic change in current planning trends, the California gnatcatcher will be eliminated from much of its U.S. range in the immediate future. Consequently, based on the criteria of the U.S. Endangered Species Act, the California gnatcatcher should be added to the U.S. List of Threatened and Endangered Wildlife and Plants.

In response to public pressure and potential legal constraints associated the possibility of a formal listing of the gnatcatcher as endangered, a variety of land-use planning efforts have been initiated at the State, county, and local levels that did not exist at the time that the NRDC - MBO endangered species petition was filed (Salata 1991b). These attempts to conserve viable gnatcatcher populations remain in their early, formative stages; it is beyond the scope of this paper to review their details. Nonetheless, it is apparent that in all cases the ultimate success of these efforts is closely linked to the possibility that the U.S. Fish and Wildlife Service will ultimately list the gnatcatcher as endangered. Without the political and legal impetus provided by a formal addition of the gnatcatcher to the U.S. List of Endangered and Threatened Wildlife, continued progress toward scientifically-driven, regional land-use planning may be significantly hindered.

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Appendix A

Summary of localities used in analysis of California gnatcatcher distribution. Sources of information provided in Atwood (1990). HA = historic approximate (pre-1960, approximate location); RA = recent approximate (1960-1984, approximate location); RE = recent exact (1960-1984, exact location); CA = current approximate (1985-1990, approximate location); CE = current exact (1985-1990, exact location).

VENTURA CO. — HA: Fillmore; Santa Paula; Santa Paula, 2.5 Mi W; Simi; Ventura.

LOS ANGELES CO. — HA: Arcadia; Arcadia, Big Santa Anita Wash; Azusa, San Gabriel Wash; Burbank, Rosco; Claremont; Claremont, Indian Hill; Claremont, 0.25 mi SW of Claremont; Claremont, 1.5 mi N of Claremont; Los Angeles; Mint Canyon; North Hollywood, Taluca; Pasadena; Pomona; Redondo; San Fernando; San Fernando, San Fernando Valley; San Fernando Valley San Fernando Valley, Lankershim; San Fernando Valley, Tujunga Wash; Whittier; RA: Azusa, San Gabriel Wash; Pasadena, Arroyo Seco; Big Tujunga Wash; RE: Claremont, Rancho Santa Ana Botanic Gardens; Palos Verdes Estates, Agua Amarga Canyon, E of Paseo Lunado and Via Rivera; Palos Verdes Estates, Coronell Canyon, E of Zunita and Via Landeta; Rancho Palos Verdes, Portugese Bend Club; Rancho Palos Verdes, Portugese Canyon, 1 mi NW of Forrestal Canyon Rd.; Rancho Palos Verdes, 1.5 km NW of White Point; Rancho Palos Verdes, 6 km NNE of Long Point; Rancho Palos Verdes, 7 km NNW of White Point; Rolling Hills, 6 km SSW of jct. Palos Verdes Dr. North and Crenshaw Blvd.; CE: Palos Verdes Estates, Agua Amarga Canyon, E of Paseo Lunado and Via Rivera; Palos Verdes Estates, Agua Amarga Canyon, E of Paseo Lunado and Via Rivera; Rancho Palos Verdes, Portugese Bend, Paintbrush Canyon; Rancho Palos Verdes, Portugese Canyon, 1 mi NW of Forrestal Canyon Rd.; Rancho Palos Verdes, Portugese Canyon, 1 mi NW of Forrestal Canyon Rd.; Rancho Palos Verdes, 0.1 mi N of jct. Pirate Dr. and Sea Raven Dr.; Rancho Palos Verdes, 1.5 km NW of White Point; Rancho Palos Verdes, 1.5 km SE of jct. Crest Rd. and Hawthorne Blvd.; Rancho Palos Verdes, 3 km SE of jct. Crest Rd. and Hawthorne Blvd.; Rancho Palos Verdes, 6 km NNE of Long Point.

ORANGE CO. — HA: La Habra, West Coyote Hills, Murphy Oil Lease; Laguna; RA: Laguna Canyon; San Juan Capistrano, Ronald W. Caspers Wilderness Park; RE: Corona Del Mar, Buck Gully; Corona Del Mar, N of PCH, 0.4 mi S of jct. PCH and Cameo Shores Rd.; Corona Del Mar, northern end of Crystal Cove State Park; Dana Point, bluff N of Doheny State Beach; Dana Point, 0.2 mi NW of jct. Green Lantern Dr. and Scenic Dr.; Dana Point, 0.2 mi SW of jct. Del Obispo and Stonehill Dr.; Dana Point, 0.4 mi N of jct. Del Obispo and Pacific Coast Hwy.; Irvine, Bonita Reservoir; Irvine, N of Bonita Canyon Rd., E of MacArthur Blvd, W of UCI; Irvine, N of jct. Sierra Oro Rd. and Turtle Rock Rd.; Irvine, San Joaquin Reservoir, SSW of Coyote Canyon Rd.; Irvine, W of Ridgeline Rd., between Turtle Rock Dr. and University Dr.; Irvine, W of Sand Canyon Reservoir, E of Ridgeline Dr.; Laguna Niguel, near jct. Cabot Rd. and Crown Valley Parkway; Laguna Niguel, 0.3 mi E of jct. Crown Valley Parkway and Alicia Parkway; Laguna Niguel, 0.4 mi W of jct. I-5 and Oso Parkway; Lemon Heights, 0.3 km E of Lower Peters Canyon Reservoir; Mission Viejo, jct. El Retiro and Felipe; Mission Viejo, 0.2 mi SW of jct. I-5 and Oso Parkway; Mission Viejo, 0.4 mi WNW of jct. Crown Valley Parkway and Marguerite Parkway; San Clemente, W of Christianitos Rd, near jct. with I-5; San Juan Capistrano, NW of jct. Malaspina and Paseo Cardero; CA: Irvine, Signal Peak; Mission Viejo, Canada Gobernadora; San Juan Capistrano; CE: Carlton, Soquel Canyon; Corona Del Mar, Abalone Point; Corona Del Mar, Buck Gully; Corona Del Mar, Crystal Cove; Corona Del Mar, Muddy Canyon; Corona Del Mar, northern end of Crystal Cove State Park; Corona del Mar, Pelican Hill above Los Trancos Canyon; Costa Mesa, Fairview Regional Park; Costa Mesa, 2 km S of Fairview Regional Park; Dana Point, jct. Quail Run Rd. and Del Obispo; Dana Point, 0.2 mi NW of jct. Green Lantern Dr. and Scenic Dr.; El Toro, between Rattlesnake Reservoir and Hicks Canyon; El Toro, immediately N of Rattlesnake Reservoir; El Toro, Serrano Creek, 0.8 mi NNW jct. Sta Margarita Pkwy and El Toro Rd; El Toro, Siphon Reservoir; El Toro, 1.3 mi N of Rattlesnake Reservoir; El Toro Marine Corps Station, Agua Chiron Creek at NE base boundary; Fullerton, 0.4 mi SSE of jct. Brea Blvd. and Bastanchury Rd.; Fullerton, 0.4 mi SSE of jct. Brea Blvd. and Bastanchury Rd.; Irvine, Bommer Canyon, 4 mi NW of jct. El Toro Rd. and Laguna Canyon Rd.; Irvine, Bonita Reservoir; Irvine, immediately S of jct. Hwy 73 and Jamboree Blvd.; Irvine, N of Bonita Canyon Rd., E of MacArthur Blvd, W of UCI; Irvine, N of jct. Sierra Oro Rd. and Turtle Rock Rd.; Irvine, San Joaquin Reservoir, SSW of Coyote Canyon Rd.; Irvine, Sand Canyon Res., 0.7 mi E jct. Ridgline Dr. and Turtle Rock Dr.; Irvine, UCI, 0.2 mi N of jct. Bonita Canyon Dr. and Coyote Canyon Rd.; Irvine, W of Ridgeline Rd., between Turtle Rock Dr. and University Dr.; Irvine, W of Sand Canyon Reservoir, E of Ridgeline Dr.; Irvine, 0.1 mi S of jct. Ridgeline Dr. and University Dr.; La Habra, 1.3 mi SE jct. Imperial Hwy. and Beach Blvd.; La Habra, 1.3 mi SE jct. Imperial Hwy. and Beach Blvd.; Laguna Hills, Upper Wood Canyon; Lemon Heights; Lemon Heights, 0.3 km E of Lower Peters Canyon Reservoir; Lemon Heights, 1 mi SE of jct. Crawford Canyon Rd. and Newport Blvd.; Mission Viejo, near Canada Chiquita and Canada Gobernadora; Mission Viejo, NE of jct. Trabuco Rd. and Lake Forest Dr.; Mission Viejo, Starr Ranch Audubon Sanctuary, N of Crow Canyon; Mission Viejo, Wagon Wheel Canyon; Naciente Mission Viejo Planning Unit, W of Plano Trabaco; Newport Beach, 0.1 mi WSW of jct. San Joaquin Hills Rd. and Jamboree

Rd.; Newport Beach, 0.5 mi W of jct. Ford Rd. and Jamboree Rd.; Orange, Irvine Regional Park; Orange, Peters Canyon Reservoir, 0.5 mi WSW jct. Peters Canyon Rd and Santiago Canyon Rd; Orange, Villa Park Dam Regional Park; Panorama Heights, Santiago Reservoir, 4.5 km E of Peters Canyon Reservoir; Placentia, between Walnut Creek Reservoir and Anaheim Hills Golf Course; Placentia, Oak Canyon Nature Center; Rancho Santa Margarita, Upper Portion of Tijeras Cr., Approx. 5 mi E El Toro; San Clemente, TRW Capistrano Test Site; San Clemente, W of Christianitos Rd, near jct. with I-5; Santiago Reservoir, 0.5 km N of N end of Santiago Reservoir; Siphon Reservoir, 0.2 mi S of Siphon Reservoir.

RIVERSIDE CO. — HA: Banning, 2 mi SE of Banning; Cabazon, 1.5 mi S of Cabazon; Corona; Corona, 5 Mi N; Menifee; Mockingbird Canyon; Norco; Palm Springs; Pedley; Riverside; Valle Vista, 1 mi SW ; RA: Jurupa Mountains; Lake Matthews, Monument Peak; Perris, University of California Motte Rimrock Reserve; Riverside, Box Springs Mountains; Riverside, 2 mi SE of University of California; RE: Lake Matthews, Gavilan Peak; Lake Matthews, 0.5 mi S of jct. La Vista Court and Descanso Dr.; Lake Matthews, 0.5 mi SE of jct. Lakeridge Rd. and Lake Matthews Rd.; Lake Matthews, 0.7 mi W of Lake Matthews Dam; Lake Matthews, 1 mi N of Harford Springs, Along Gavilan Rd.; Lake Matthews, 2 mi W of Lake Matthews Dam; Mockingbird Canyon, 2.4 mi NW of jct. Mockingbird Canyon Rd. and Cajalco Rd.; CA: Corona, Eagle Valley; Lake Matthews, Olsen Canyon, S of Cajalco Rd.; CE: Casa Blanca, 2 km E; Corona, 2 km W of Eagle Valley; Corona, 3 km W of Eagle Valley; Gypsum Canyon, 4 km NE of Walnut Canyon Reservoir; Lake Elsinore, Walker Canyon, 5 km NW of jct. I-15 and Hwy. 74; Lake Elsinore, 3 km NW of jct. I-15 and Hwy 74; Lake Elsinore, 4 km NW of jct. I-15 and Hwy 74; Lake Elsnore, 1.1 mi NNE of jct. Main St. and Railroad Canyon Rd.; Lake Mathews, 1.6 km N of Gavilan Peak; Moreno Valley, jct. Clark Street and Fresh Sky Rd; Moreno Valley, 0.25 mi E of jct. Pigeon Pass Rd. and Lawless Rd.; Murietta Hot Springs, Santa Gertrudis Cr., 0.25 mi SSW of jct. Borel Rd. and Leon Rd; Murrieta Hot Springs, Santa Gertrudis Cr., 8 km NE of jct. I-215 and Hwy79; Murrieta Hot Springs, 2 km N of Murrieta Hot Springs; Peralta Hills, 3.5 km W of Walnut Canyon Reservoir; Perris, University of California Motte Rimrock Reserve; Perris, 0.3 mi NW of Good Hope Mine; Perris, 4 km N of Steele Peak; Perris, 4 km NW of jct. I 215 and Hwy 74; Riverside, 1 km SW of Mockingbird Reservoir; Sunnymead, NW end of Poorman Reservoir; Sunnymead, 2 km NW of Poorman Reservoir; Temecula, 0.5 km W of jct. I-15 and Hwy. 79; Temecula, 0.5 mi N of Pechanga Indian Reservation; Temecula, 2 km NW of jct. I-15 and Hwy. 79; Temescal Valley, jct. I-15 and Indian Truck Trail; Temescal Valley, 5 km W of Monument Peak.

SAN BERNARDINO CO. — HA: Colton; Colton, Reche Canyon; Colton, Slover Mtn.; Fontana, Mt. Jurupa; Redlands; San Bernardino; San Bernardino, Devil's Canyon; San Bernardino, Lytle Creek Wash; San Bernardino, near California State San Bernardino; San Bernardino, Reche Canyon; CA: San Bernardino, confluence of Cajon Wash and Lytle Creek Wash; San Bernardino, Reche Canyon, S of Loma Linda near Riverside Co. line.

SAN DIEGO CO. — HA: Bonita; Chula Vista; Dehesa; El Cajon; Escondido; Escondido, Lake Hodges; Grossmont; La Mesa; La Mesa, 2 Mi E; Lakeside; Mt. Soledad; National City; Pauma Valley; Ramona; San Diego; San Diego, Point Loma; San Pasqual; San Pasqual, Hammer Canyon; San Pasqual, Oaks Ranch; San Pasqual Valley, Oaks Ranch; Sweetwater Dam; Sweetwater Valley, 2 mi N of Sweetwater Reservoir; RA: Bonita, 0.8 km W of Southwestern Community College; La Mesa, 3 km W of Murray Reservoir; Oceanside; San Diego, Sweetwater Reservoir; San Luis Rey, NW of El Camino High School; RE: Bonsall, 1 km NE of Bonsall; Camp Pendleton MCB Marine Corps Base (=Camp Pendleton MCB), 1 mi W of Whelan Lake and 0.5 Mi S of Ysidora Basin; Carlsbad, Agua Hedionda Creek, S of jct. Kelly Dr. and Park Dr.; Carlsbad, E end of Batiquitos Lagoon, W of El Camino Real; Carlsbad, La Costa, 0.2 mi NNE of jct. Alga Rd. and Estrella Del Mar; Carlsbad, La Costa, 0.2 mi SE of jct. Alga Rd. and Alicante Rd.; Carlsbad, 0.5 mi NW of Laguna Riviera Park; Chula Vista, N of jct. Proctor Valley Rd. and Rancho Janal Drive; Chula Vista, N of Otay River at jct. Otay River and Otay Valley Rd.; El Cajon, 0.2 km N of Singing Hills Country Club; El Cajon, 0.2 km S of Singing Hills Country Club; El Cajon, 1.4 mi E of jct. La Cresta Rd. and Greenfield Dr.; El Cajon, 1.5 km E of Glenview; El Cajon, 3 mi ESE of El Cajon, jct. Burris Dr. and Brust St.; El Cajon, 7.5 km NE of jct. Hwy. 54 and Willow Glen Dr.; Encinitas, 0.7 mi SW of jct. Rancho Santa Fe and Manchester Ave.; Escondido, jct. Mt. Israel Rd. and Del Dios Hwy.; Escondido, 0.2 mi ESE of jct. San Pasqual Rd. and Bear Valley Parkway; Escondido, 0.2 mi NW of jct. Via Ranco Parkway and I-15; Escondido, 0.3 mi S of jct. Bear Valley Parkway and Hwy. 78; Escondido, 1 mi ESE of jct. San Pasqual Rd. and Bear Valley Parkway; Escondido, 1.5 mi N of jct. Cloverdale Rd. and San Pasqual Valley Rd.; Escondido, 1.5 mi SW of jct. Bear Valley Parkway and Hwy 78; Escondido, 2 mi NNE of Lake Hodges Dam; Escondido, 2.3 km NW of Bernardo Mt.; Escondido, 3 mi NE of Lake Hodges Dam; Imperial Beach, Tijuana River, 1 km E of Border Field State Park; Jamacha, 2.7 mi NE of Jamacha Jct.; La Mesa, San Diego River at Oak Canyon; La Mesa, 2.7 km NE of Gillespe Field; La Mesa, 3 km NE of Gillespe Field; La Presa, N side of Sweetwater Reservoir; Lakeside, W of Lakeview Rd., 1 mi SW Lake Jennings; Miramar Naval Air Station (=Miramar NAS), N of Clairemont Mesa Blvd and W of Santo Rd.; Miramar NAS, Tierra Santa and Camp Elliot Parcels; Oceanside, Fire Mountain, W of Eternal Hills Memorial Park; Oceanside, 0.1 mi SW of Capistrano City Park; Oceanside, 0.2 mi N of Mira Costa College; Oceanside, 0.3 mi N of jct. Hwy. 78 and Jefferson Rd.; Oceanside, 0.8 mi WSW of jct. Oceanside Blvd. and College Blvd.; Oceanside, 1.6 mi SSE of Guajome Lake; Poway, entrance to Lake Poway Recreation Area; Poway, 0.8 km E of Twin Peaks; Poway, 1 km NE of Twin Peaks; Ramona, jct. San Vincente Rd. and Wildcat Canyon Rd.; Rancho Bernardo, Coche Viejo, Espola Rd. 1.5 mi E of jct. with Pomerado Rd.; Rancho Bernardo, Mule Hill, SE of jct. San Pasqual Rd. and Bear Valley Parkway; Rancho Bernardo, 0.3 mi E of jct. Pomerado Rd. and Highland Valley Rd.; Rancho Bernardo, 0.5 mi N of jct. Hwy. 78 and San Pasqual Rd.; Rancho Bernardo, 0.5 mi NW of San Pasqual Battlefield Historic Park; Rancho Bernardo, 1.3 mi SW of jct. Hwy. 78 and San Pasqual Rd.; Rancho Santa Fe, immediately N of jct. San Dieguito Cr. and

Lusardi Cr.; Rancho Santa Fe, 1.4 km N of San Dieguito Reservoir; Rancho Santa Fe, 1.5 km NW of San Dieguito Reservoir; Rancho Santa Fe, 2 MI NE of Rancho Santa Fe, N and W of Lake Hodges; Rancho Santa Fe, 2.2 km NW of San Dieguito Reservoir; San Diego, 0.1 mi NE of jct. Frazee Rd. and Friars Rd.; San Diego, 0.4 mi NE of jct. Friars Rd. and I-15; San Miguel Mountain; San Pasqual, 1.7 km N of San Pasqual; San Ysidro, S of Otay Mesa Rd., 0.7 mi NE San Ysidro; San Ysidro, 0.5 mi S of Otay Valley Rd, Immediately E of I-805; San Ysidro, 0.8 mi WSW of jct. Hollister St. and Monument Rd.; Santee, E of northernmost Santee Lakes Pond; Santee, 2.7 km NE of Gillespe Field; Santee, 3 km NE of Gillespe Field; Spring Valley, La Presa, N end of San Bernardino Ave.; Spring Valley, 2 km N of Peak of San Miguel Mtn.; CA: Carlsbad, S of Agua Hedionda, in transmission corridor; Miramar, Los Penasquitos Canyon; Miramar NAS, between San Clemente Canyon and US International University; Rancho Santa Fe, West of Rancho Rd. East; CE: Bonita, Sweetwater River between Willow St. and I-805; Bonita, 2.5 km NE of Glen Abbey Cemetary; Camp Pendleton MCB, Cockleburr Canyon, 0.2 km W of I-5; Camp Pendleton MCB, jct. Stuart Mesa Rd. and Las Flores Creek; Camp Pendleton MCB, mouth of French Canyon, 0.8 km NE of Public Rest Area; Camp Pendleton MCB, NE Shore of O'Neill Lake; Camp Pendleton MCB, SE Shore of O'Neill Lake; Camp Pendleton MCB, 0.8 km SW of jct. Vandergrift Blvd. and Stuart Mesa Rd.; Camp Pendleton MCB, 1 km NW of Morro Hill; Camp Pendleton MCB, 1.2 km NE of Jofegan; Camp Pendleton MCB, 1.2 km NE of San Onofre Nuclear Plant; Camp Pendleton MCB, 1.2 km NE of Windmill Lake; Camp Pendleton MCB, 1.2 km NW of jct. Stuart Mesa Rd. and Las Pulgas Rd.; Camp Pendleton MCB, 1.2 km SW of jct. Stuart Mesa Rd. and Las Pulgas Rd.; Camp Pendleton MCB, 1.3 km NE of jct. Stuart Mesa Rd. and Las Flores Creek; Camp Pendleton MCB, 1.4 km SE of jct. Vandergrift Blvd. and Stuart Mesa Rd.; Camp Pendleton MCB, 1.4 km SE of Jofegan; Camp Pendleton MCB, 1.5 km NE of Agra; Camp Pendleton MCB, 1.5 km NE of jct. Stuart Mesa Rd. and Las Flores Creek; Camp Pendleton MCB, 1.5 km SE of Agra; Camp Pendleton MCB, 1.8 km W of Morro Hill; Camp Pendleton MCB, 1.8 km W of Morro Hill; Camp Pendleton MCB, 2 km NNW of jct. Stuart Mesa Rd. and Flores Creek; Camp Pendleton MCB, 2 km NW of Oceanside Harbor; Camp Pendleton MCB, 2.5 km E of O'Neill Lake; Camp Pendleton MCB, 2.5 km N of San Onofre Nuclear Plant; Camp Pendleton MCB, 2.6 km NW of San Onofre Nuclear Plant; Camp Pendleton MCB, 2.8 km NNE of Windmill Lake; Camp Pendleton MCB, 2.8 km NW of Morro Hill; Camp Pendleton MCB, 2.8 mi SW of Morro Hill; Camp Pendleton MCB, 3 km NNE of jct. Stuart Mesa Rd. and Las Flores Creek; Camp Pendleton MCB, 3 km W of Whelan Lake; Camp Pendleton MCB, 3.5 km SE of Jofegan; Camp Pendleton MCB, 4 km NE of Windmill Lake; Camp Pendleton MCB, 4 km NW of Whelan Lake; Camp Pendleton MCB, 4.2 km W of Whelan Lake; Camp Pendleton MCB, 5 km NNW of Whelan Lake; Camp Pendleton MCB, 5 km SE of San Onofre Nuclear Power Plant; Camp Pendleton MCB, 5.5 km SW of jct. Basilone Rd. and Las Pulgas Rd.; Camp Pendleton MCB, 6.5 km NW of Whelan Lake; Camp Pendleton MCB, 6.5 km W of Whelan Lake; Camp Pendleton MCB, 7 km NNW of Whelan Lake; Carlsbad, E end of Batiquitos Lagoon, W of El Camino Real; Carlsbad, La Costa, 0.2 mi NNE of jct. Alga Rd. and Estrella Del Mar; Carlsbad, 1 km s of Cerro de la Calavera; Carlsbad, 2 km NW of Calevera Lake; Chula Vista, Bonita, near end of Camino Elevado along Otay Lakes Rd.; Chula Vista, mouth of Sweetwater River; Chula Vista, Rice Canyon; Chula Vista, Rice Canyon; Chula Vista, Salt Creek 0.7 km W of Lower Otay Reservoir; Chula Vista, Telegraph Canyon, 2 km E of jct. I-805 and Telegraph Cr.; Del Mar, Del Mar; Del Mar, Zanja Canyon, 4 km W of Black Mtn.; Dulzura, 2.5 km NW of Dulzura; El Cajon, Amber Ridge, 2 km E of Hillsdale; El Cajon, Amber Ridge, 2 km SSE of Hillsdale; El Cajon, Amber Ridge, 2.5 KM SE of Hillsdale; El Cajon, 1 km E of Hillsdale; El Cajon, 2.5 km NW of McGinty Mtn.; El Cajon, 4 km S of Suncrest; Encinitas, 0.6 mi NW of jct. Manchester Ave. and I-5; Encinitas, 3.5 km NW of San Dieguito Reservoir; Escondido, 2 mi NNE of Lake Hodges Dam; Eucalyptus Hills, jct. Sycamore Canyon and Clark Canyon; Eucalyptus Hills, 3 mi NW Santee; Fallbrook Naval Weapons Station (=Fallbrook NWS), DeLuz; Fallbrook NWS, 0.5 km E of Fallbrook; Fallbrook NWS, 1 km E of O'Neill Lake; Fallbrook NWS, 1 km S of Fallbrook Airport; Fallbrook NWS, 1.5 km NE of O'Neill Lake; Fallbrook NWS, 2 km E of Fallbrook; Fallbrook NWS, 2.5 km E of O'Neill Lake; Fallbrook NWS, 3 km E of Fallbrook; Fallbrook NWS, 3 km NE of O'Neill Lake; Fallbrook NWS, 3 km S of Fallbrook Airport; Fallbrook NWS, 4 km E of Fallbrook; Fallbrook NWS, 4 km E of O'Neill Lake; Fallbrook NWS, 4.5 km NE of O'Neill Lake; Hillsdale, jct. Hillsdale Rd and Jamacha Rd, N of Valhalla High School; Kearny Mesa, 0.2 km W jct. Hwy 163 and Costa Mesa Blvd.; Kearny Mesa, 1 km W of jct. Hwy 163 and Costa Mesa Blvd.; La Mesa, Little Sycamore Canyon; La Mesa, SE side of of Murray Reservoir; La Mesa, 0.5 mi S of jct. East Ridge Dr. and Cinnabar; La Mesa, 0.5 mi S of jct. Normal Ave. and Olive Ave.; La Presa, SE side of Sweetwater Reservoir; Lake San Marcos, 1.5 km W of Lake San Marcos; Lake San Marcos, 2 km SE of Lake San Marcos; Lake San Marcos, 2 km SW of Lake San Marcos; Lake San Marcos, 3 km SW of Lake San Marcos; Lake San Marcos, 3 km SW of Lake San Marcos; Lakeside, San Diego River at Cactus County Park; Lakeside, San Diego River between Hwy 67 and Channel Rd.; Lakeside, 1 km NE of Lake Jennings; Leucadia, S of Batiquitos Lagoon; Leucadia, 0.3 mi ESE of jct. I-5 and Palomar Airport Rd.; Leucadia, 0.5 km N of Batiquitos Lagoon; Leucadia, 1 km E of Batiquitos Lagoon; Leucadia, 1.5 km N of Batiquitos Lagoon; Leucadia, 2.5 km E of Batiquitos Lagoon; Leucadia, 4.5 km E of Leucadia; Leucadia, 4.5 km NW of San Dieguito Reservoir; Lower Otay Reservoir, 3 km SE of Mother Miguel Mt.; Mira Mesa, NW of jct. Mira Mesa Blvd. and Camino Ruiz; Miramar, San Clemente Canyon at second San Diego aquaduct; Miramar, 4 km N of Miramar; Miramar NAS, Rose Canyon, 0.2 mi N of jct. Sidewinder Rd. and Tiger Ave.; Miramar NAS, San Clemente Canyon, S of Ammo Rd, E of Kearby Villa, W I-15; Miramar NAS, West Sycamore Canyon; Miramar NAS, 0.2 km W of Miramar Reservoir; Moosa Canyon; Oceanside, Fire Mountain, W of Eternal Hills Memorial Park; Oceanside, 1.6 mi SSE of Guajome Lake; Otay Mesa, mouth of O'Neal Canyon; Otay Mesa, 1.5 km SW of S end of Lower Otay Reservoir; Otay Mesa, 2 km N of Brown Field; Otay Mesa, 2 km NW of Brown Field; Otay Mesa, 3 km NE of Brown Field; Otay Mesa, 4 km N of Brown Field; Poway, 1.5 mi E of Battle Mtn.; Poway, 2.1 mi SE of Battle Mtn.; Poway, 2.5 mi E of

Battle Mtn.; Poway, 3 mi SE of Battle Mtn.; Poway, 4 km W of Poway; Ramona, 4 km SE of Ramona; Rancho Bernardo, San Pasqual Battlefield Historic Park; Rancho Bernardo, 0.3 mi E of jct. Pomerado Rd. and Highland Valley Rd.; Rancho Bernardo, 0.4 mi NW of Rancho Bernardo Community Park; Rancho Bernardo, 0.5 mi N of jct. Hwy. 78 and San Pasqual Rd.; Rancho Bernardo, 2.5 km NW of Rancho Bernardo; Rancho de Otay, Little Cedar Canyon at Dulzura Creek; Rancho de Otay, N end Lower Otay Reservoir; Rancho de Otay, NW side of Upper Otay Reservoir; Rancho de Otay, S end of Lower Otay Reservoir; Rancho de Otay, 0.5 km W of Lower Otay Reservoir; Rancho de Otay, 0.8 km N of Lower Otay Reservoir; Rancho de Otay, 0.8 km NE of Lower Otay Reservoir; Rancho de Otay, 1 km W of Lower Otay Reservoir; Rancho Santa Fe, La Zanja Canyon; Rancho Santa Fe, 0.5 km N of San Elijo Lagoon State Park; Rancho Santa Fe, 2 km E of San Dieguito Reservoir; Rancho Santa Fe, 3 mi E of Rancho Santa Fe, immediately N of Del Dios Hwy.; San Diego, Balboa Park; San Marcos, 2 km S of Lake San Marcos, S end of Cerro de las Posas; San Miguel Mountain; San Pasqual, 0.5 km S of Starvation Mtn.; San Ysidro, Dennery Canyon; San Ysidro, Dennery Canyon; San Ysidro, near jct. I-805 and Hwy 117; San Ysidro, Spring Canyon at Mexican border; Solano Beach, San Elijo Lagoon; Solano Beach, San Elijo Lagoon; Soledad Valley, 2.5 km SW of Carmel Mtn; Spring Valley, Calavo Gardens; Spring Valley, Dictionary Hill; Spring Valley, 0.2 km S of jct. Jamacha Rd. and Hwy. 54; Spring Valley, 1 km N of Dictionary Hill; Spring Valley, 2 km W of jct. Jamacha Rd. and Hwy. 54; Spring Valley, 2.4 km S of jct. Jamacha Rd. and Hwy. 54; Spring Valley, 2.4 km SSW of jct. Jamacha Rd. and Hwy. 54; Spring Valley, 3 km W of jct. Jamacha Rd. and Hwy. 54; Spring Valley, 4 km SW of jct. Jamacha Rd. and Hwy. 54; Spring Valley, 5.2 km SW of jct. Jamacha Rd. and Hwy. 54; Vista, end of Estrelita Dr.; Vista, Guajome Lake; Vista, W end of Calavera Lake; Vista, 2 km E of Guajome Lake; Vista, 4 km W of Guajome Lake; Vista, 5 km SE of Bonsall.

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Effects of Urban Encroachment on Wildlife in the Santa Monica Mountains

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Abstract. Land development is threatening biological diversity in many areas by encroaching upon and subdividing remaining natural habitats. Successful conservation strategies require understanding of how these activities influence natural communities. We are addressing these issues in an on-going project investigating the effects of urban encroachment on resident chaparral birds and small mammals in the Santa Monica Mountains of southern California. Specifically, we are (1) determining if development proximity alone can explain community-level changes in natural areas near urbanization, (2) examining the role of human-caused habitat disturbance on community patterns, (3) investigating how habitat isolation may affect observed patterns, and (4) assessing conservation implications of our results. Initial data collected in 1991 in twelve sites exposed to varying amounts of urban development and in four human-altered sites address the first two objectives. Results indicate that variation in bird and mammal species richness and relative abundance were not correlated with proximity to development. However, in disturbed sites, reduced abundance and a different species composition of small mammals were observed, suggesting the important role of human-caused habitat disturbance. In general, chaparral habitat very close to urban development can have important conservation value. In addition, direct habitat alteration may impact wildlife more than urban proximity alone. We discuss interpretations of our results and directions for future work.

Keywords: Birds; chaparral; edge effects; disturbance; habitat fragmentation; nature reserves; Santa Monica Mountains; small mammals; urbanization.

Introduction

Rapidly expanding human development is encroaching upon and subdividing natural areas worldwide (Murphy 1988). In many regions, formerly continuous natural habitats are now composed of smaller remnant patches, isolated from one another by human-modified areas (Wilcove et al. 1986). Such large scale landscape changes can pose serious threats to biological diversity. In particular, encroachment by development reduces the amount of available habitat for native species, partially or wholly isolates some species, and increases the amount of exposure natural habitats face to potential impacts from surrounding developed lands (Noss 1987). Overall, these effects have been recognized as some of the most serious threats facing biological diversity today (Wilcox and Murphy 1985; Lubchenco et al. 1991). As development activities continue, successful conservation strategies will depend upon increased understanding of how encroaching development affects natural habitats. This is especially true in southern California, where rapid urbanization has placed development immediately adjacent to or completely surrounding much of the remaining natural habitat.

Understanding and mitigating threats from encroaching development and habitat fragmentation has been traditionally approached by applying island biogeographic theory (MacArthur and Wilson 1967) towards the design and management of remaining habitats or nature reserves (e.g., Diamond et al. 1976; Wilson and Willis 1975; Wilcox 1980). Using this approach, remaining habitats (reserves) are treated as "islands" within a "sea" of human disturbance, and species losses in these areas are attributed primarily to increased extinction and decreased colonization rates from small size and greater isolation (e.g., Galli et al. 1976; Harris 1984; Lynch and Whigham 1984; Soule et al. 1988). Although this approach has resulted in some important discussions regarding optimal reserve design (e.g. Simberloff and Abele 1976, 1982; Diamond et al. 1976; Gilpin and Diamond 1980; Higgs 1981; Quinn and Robinson *in press*), it has been less useful for understanding how individual reserves interact

with surrounding lands (Saunders et al. 1991). Because most remaining natural areas and reserves are not islands, but instead interact with adjacent human-modified lands (Janzen 1983; Schonewald-Cox and Bayless 1986), the conservation of biological diversity in these areas will require understanding how natural communities are affected by nearby development.

Such external influences, or edge effects, can be broadly defined as any changes that occur in the diversity, abundance, or spatial distribution of previously undisturbed communities as a result of their proximity to human-modified areas (Lovejoy et al. 1986; Yahner 1988). Edge effects of this type are receiving increasing attention because of concern that natural habitats may be exposed to a variety of threats from adjacent human-altered areas (Janzen 1983; Schonewald-Cox and Bayless 1986; Forman and Godron 1986; Buechner 1987; Murphy 1988). For example, clear community-level differences have been demonstrated between highly developed (urbanized) areas and corresponding natural sites (Guthrie 1974; Emlen 1974; Bessinger and Osborne 1982; Tweit and Tweit 1986; Mills et al. 1989). More importantly, though, these human-induced ecological changes may penetrate into nearby "undisturbed" natural areas. For example, Lovejoy et al. (1986) suggest that a variety of biotic and abiotic influences enter undisturbed tropical forest fragments, influencing natural communities near deforested edges. In another example, rising oak tree mortality in Santa Rosa National Park, Costa Rica may be due to desiccating winds and insolation from adjacent deforested lands (Janzen 1986). In addition, several temperate zone studies have demonstrated that increased predation and brood parasitism near natural habitat edges may reduce the nesting success of some bird species, leading to population declines in fragmented areas (Gates and Gysel 1978; Brittingham and Temple 1983; Wilcove 1985; Andren and Angelstam 1988). Documenting the community-level consequences and conservation implications of such human-caused habitat encroachment has become widely recognized as a top research priority in conservation biology (Yahner 1988; Soule and Kohm 1989; Saunders et al. 1991).

This paper discusses an on-going research project we have initiated to address some of these issues. In particular, we are investigating how urban encroachment may influence wildlife communities in nearby natural habitat within the Santa Monica Mountains of southern California. Because this work was recently initiated and is still on-going, we are only able to present initial results based on data collected during the summer 1991 field season. However, because of the urgent need for information to guide conservation efforts throughout much of southern California, we believe that presenting our methods and initial results now could benefit these efforts. We stress, however, that conclusions here must be regarded as somewhat preliminary, reflecting only our findings to date. We have only just begun and anticipate conducting more analyses and collecting much more field data over the next two years on this project.

Objectives and Methods

Focal species

To evaluate the effects of urban encroachment on wildlife in the Santa Monica Mountains, we have chosen to measure the diversity and relative abundance of full-time resident bird and small mammal species in habitat areas near urbanization. These focal species include both native chaparral birds and small mammals and year-round resident species often associated with urban environments (Tables 1 and 2).

We chose to focus on these species for several reasons. First, because of their dependence upon local habitat resources on a year-round basis and their relatively short generation times, resident birds and small mammals should serve as good indicators of possible environmental changes in natural areas near urban development. Second, many of these species, particularly the small mammals, form the prey base for many of the larger carnivore species. These larger, more "charismatic" species are often of great interest to conservation efforts but are much more difficult to study directly. Insights into the ecological conditions faced by the larger animals could be provided through our work on the smaller species on which they depend. Finally, small mammals and resident birds can be easily sampled over wide areas, providing substantial information at relatively low cost.

Research objectives

Using these focal species, we hope to address four specific objectives regarding the effects of urban encroachment on wildlife. First, we are testing the hypothesis that development proximity or "urban exposure" is sufficient by itself to result in community-level changes for these species. This work was initiated in 1991 and most of our current data addresses this aspect of the project. Second, we are assessing the effects of direct human-caused habitat alteration on resident bird and small mammal community patterns. We obtained some preliminary data addressing this issue in 1991 and expect to collect much more as the project progresses. Our third objective is to investigate how habitat isolation may affect observed community patterns. Up to now we have only collected data within a very large (more

Table 1. Resident bird species in 12 urban exposure sites during the summer of 1991. Sites are arranged from least exposed to most exposed to urban development. Relative abundance codes: P = present in transect (1-5 individuals observed on at least one morning census); C = common (6-15 individuals observed); and A = abundant (over 16 individuals observed). * = birds often associated with urban environments.

	Species Abundance by Site (Least urban exposed site ----------> Most urban exposed)											
Bird Species	1	2	3	4	5	6	7	8	9	10	11	12
California quail	P		P		A		C	P	P	C	C	P
* Mourning dove	P	P	C	P	C		P		P	P	P	P
* Rock dove									P			C
Anna's hummingbird		P	P	P			P	P	P	P	P	P
Scrub jay	C	P	P	P	P	P	P	C	P	P	P	P
Common raven		P			P				P		P	P
Plain titmouse						P						
Bushtit		C		A	C	A				A		
Bewick's wren	P	P	P	P	P	P		P	P	P	P	P
Wrentit	C	C	C	C	P	C	C	P	P	P	P	P
* Northern mockingbird	P	P	P		P		P		P		P	C
California thrasher		P	P	P	P	P	P	P			P	
Rufous-sided towhee	P	C	P	P	P	P	P	P	P	P	P	P
* Brown towhee	P	P	P	P	P	P	P	P		P	P	
* House finch	A	C	A	C	A	C	A	C	A	C	A	A
Total Species	10	11	12	10	12	8	10	9	12	9	12	11

than 5000 ha), continuous habitat area. In the future we would like to also sample in smaller isolated sites that are completely surrounded by urban development. The fourth and final objective of our work is to continually assess the conservation planning and policy implications of our research. To this end, upon completing the project, we will be providing several local agencies and other interested organizations with conservation management recommendations based on our results.

As discussed above, currently our data only address the first two objectives. As a result, this paper focuses primarily upon these two specific issues of urban exposure and human-caused habitat disturbance. The final two objectives, isolation effects and conservation implications, are necessarily discussed in much less detail here.

Study area

Field data for the project were collected in the Santa Monica Mountains, near Los Angeles, California (Fig. 1). This area provides a particularly appropriate location for investigating urban encroachment effects on wildlife. In the Santa Monica Mountains, rapid urbanization has encroached upon and fragmented remaining protected habitats, potentially impacting native wildlife in many areas. Although there is significant public concern to maintain viable natural communities within and around the Santa Monica Mountains, very little information is currently available about how nearby development may affect conservation efforts. As federal, state, local, and private park agencies continue their conservation and

Table 2. Small mammal species in 12 urban exposure sites during the summer of 1991. Sites are arranged from least exposed to urban development to most exposed to urban development. Individual cell entries indicate the number of individuals captured at each transect site.

	Species Abundance by Site (Least urban exposed ----------> Most urban exposed)											
Mammal Species	1	2	3	4	5	6	7	8	9	10	11	12
Neotoma fuscipes	1	4	9	9	3	3	5		8	4	8	1
Neotoma lepida		1	3				1	3			8	
Peromyscus boylii	1	1	6	5			2	5		2	1	
Peromyscus californicus		1	1	1				1				1
Peromyscus truei										1		
Peromyscus maniculatus										1		
Perognathus californicus						1						1
Dipodomys agilis					1							
Juvenile sp.	1		1				1	1	1		3	1
Total Species	2	4	4	3	2	2	3	3	1	4	3	3
Total Individuals	3	7	20	15	4	4	9	10	9	8	20	4

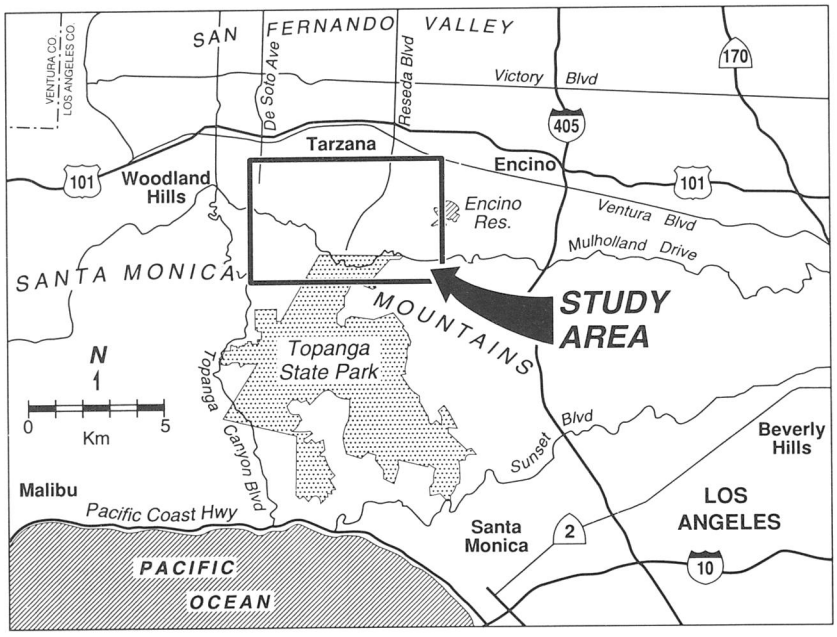


Figure 1. Location map for the Sant Monica Mountains.

park planning activities amid the threat of increasing development, an understanding of the potential effects of urban encroachment has become vital.

In our work to date, we have focused on a specific study area within the Santa Monica Mountains located along the southern edge of the San Fernando Valley near Topanga State Park (Fig. 1). This area includes approximately 400 ha of primarily chaparral habitat and is located adjacent to urbanization towards the north and additional natural habitat towards the south. Natural vegetation consists of a number of commonly associated chaparral species, including chamise (*Adenostoma fasciculatum*) and black sage (*Salvia mellifera*), with vegetation heights ranging from 1-3 m. When this study area is included with the more than 5000 ha of protected habitat to the south, it represents one of the largest reserved areas of open space in the Santa Monica Mountains and, indeed, within a city limits anywhere.

Sampling methods

To investigate the effects of urban encroachment on wildlife communities, full-time resident birds and small mammals were sampled along a number of 150 m long transects located throughout the study area. For bird species, early morning transect counts were used to estimate species richness and relative abundance at each transect site (Emlen 1971; Ralph and Scott 1981). Small mammals were sampled using Sherman live traps, with relative abundance estimated using mark and recapture techniques. Small mammals were captured, identified to species, measured, marked using nontoxic fur dyes, and released unharmed at the site of capture. Relative abundance of small mammals was estimated based on relative trapping success per transect and other population estimation methods (reviews in Davis 1982; Wakeley 1982; Cooperrider et al. 1986; Ludwig and Reynolds 1988).

In addition to wildlife sampling, we also collected data on vegetation cover and structure within each transect site using line point methods (Heady et al. 1959). A range pole was dropped at 100 points along the transect, and the vegetation height and all perennial shrub touches noted. Percentage cover of grasses and forbs was also recorded, however, our focus was primarily on perennial woody species.

To specifically investigate the relationship between proximity of development and wildlife community changes, birds and small mammals were sampled along twelve transects exposed to differing amounts of urban development. Locations for these urban exposure sites were identified using land use maps, aerial photographs, and field surveys (Fig. 2). All twelve transect sites were located within relatively undisturbed chaparral habitat and placed along north-south trending paths to facilitate access into the thick chaparral vegetation. To control for possible vegetation differences associated with slope and aspect variation, all sites were located along ridgelines or moderate slopes, with no transects placed in canyon bottoms. Overall, these criteria allowed us to identify twelve sites which were similar in vegetation conditions but varied in their degree of proximity or exposure to urban development. The twelve sites ranged from low

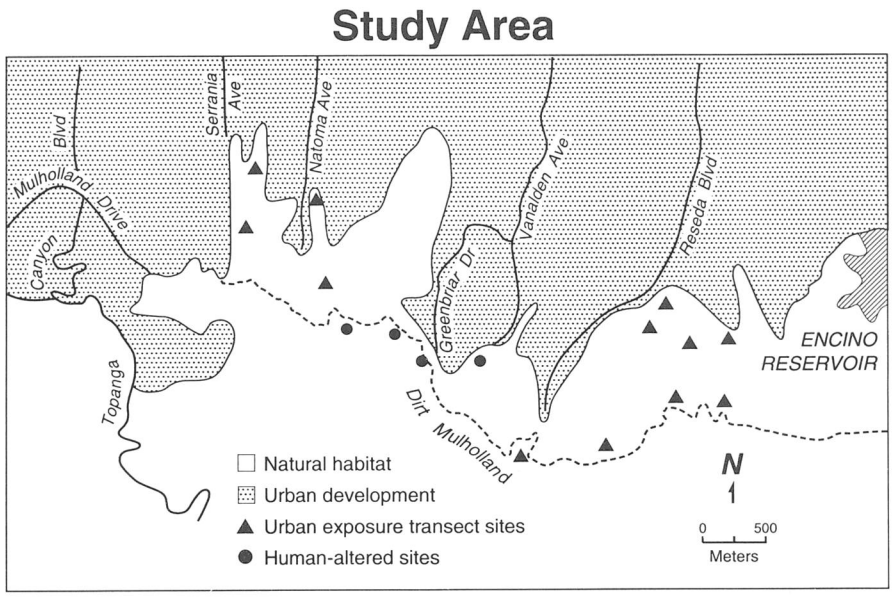


Figure 2. Study area map indicating the location of twelve urban exposure and four human-altered transect sites sampled in 1991.

levels (more than 1000 m from urban development) to high levels (adjacent to development) of urban exposure.

From 5 July through 27 August, 1991, birds and small mammals were sampled at each of the twelve urban exposure sites. Birds were sampled for five consecutive mornings at each site, resulting in a total of 60 morning bird samples in 1991. Small mammals were sampled at each transect site using two parallel trap lines of 25 traps each. Baited traps were set in the evening and checked every morning over a five-day period in each transect site. Individual traps were spaced 6 m apart parallel to the transect trail and placed 5 m from the trail edge in undisturbed chaparral. Using this design, each transect site was sampled for 250 trap-nights, with a total of 3000 small mammal trap-nights completed over all the sites in 1991. Vegetation characteristics were also recorded by dropping a range pole at 50 points located 5 m into the vegetation along each side of the transect trails. This resulted in the collection of 1200 vegetation point measurements between all of urban exposure sites over the 1991 season.

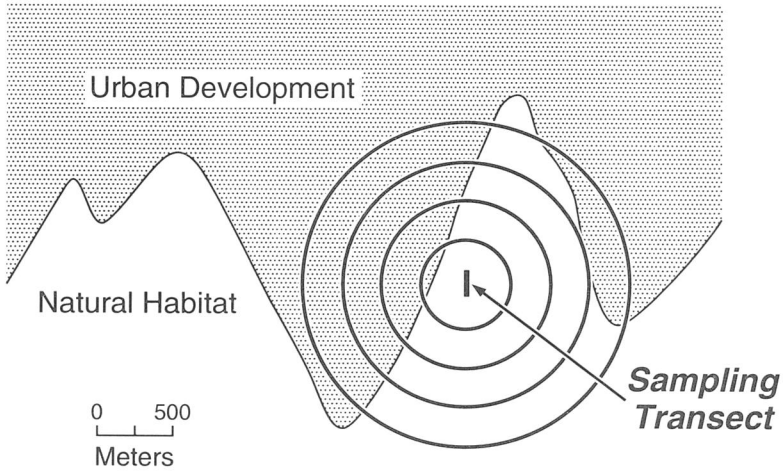


Figure 3. Urban exposure was measured by calculating the area of urban development surrounding each transect site within four concentric zones. Zone 1 was located from 0 to 250m, zone 2 from 250 to 500 m, zone 3 from 500 to 750 m, and zone 4 from 750 to 1000 m from the center of the transect site. A weighted sum of the amount of urban development within these zones (with zone 1 weighted by 1.00, zone 2 by 0.75, zone 3 by 0.50, and zone 4 by 0.25) was used as an estimate of urban exposure for each transect site.

To quantify the degree of development proximity around the transect sites, an urban exposure index was calculated that considers both the distance to and area of development surrounding each transect (Fig. 3). Using this technique, a weighted mean for urban exposure was calculated for each site, with the closest development weighted more heavily than development further away. This method allowed us to incorporate the variable shape of the urban edge when we quantified urban exposure, enabling us to take into account urban exposure differences expected between sites surrounded by nearby development on several sides versus those that had nearby development in only one direction, even if the distance to development was similar in both cases. In our analysis, we used these measures of urban exposure to investigate the relationship between development proximity and wildlife community changes for birds and small mammals.

Finally, in addition to the twelve "undisturbed" urban exposure sites, four human-altered sites were also sampled during the 1991 summer season (Fig. 2). Sampling methods identical to those used in the twelve urban exposure sites were used in the four disturbed sites to estimate the diversity and relative abundance of resident birds and small mammals, and to measure vegetation characteristics. The four disturbed sites were identified based on two primary criteria: clear evidence of past or current vegetation damage and an established history of human-caused alteration resulting in the observed habitat changes. The four disturbed sites ranged from 60 m to more than 600 m from urban development, encompassing a range from moderate to low urban exposure. Our interest in these sites was to examine specifically the effects of direct habitat alteration on wildlife community patterns rather than the effects of just urban proximity *per se*.

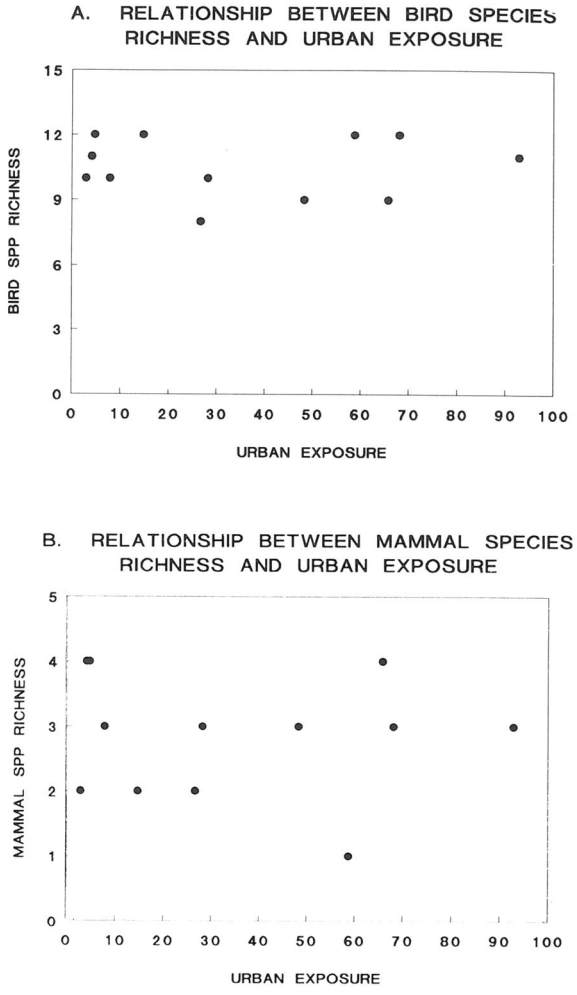


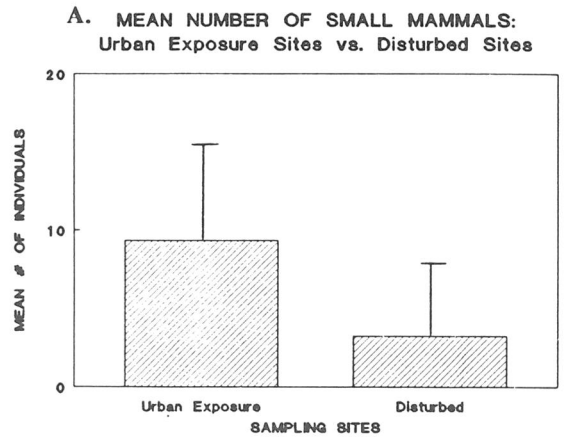
Figure 4. The relationship between species richness and urban exposure for (A) resident birds and (B) small mammals from 1991 data collected in the twelve urban exposure transect sites. Regression analysis indicated no significant relationship between the variables with r-squared values of less than one percent for both birds and mammals.

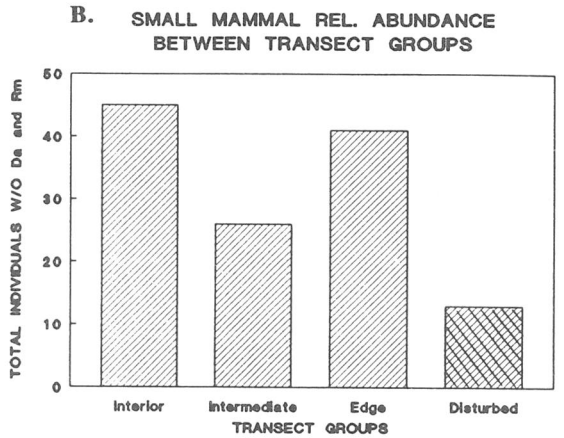
Results

Urban exposure sites

Resident bird species data collected in the summer of 1991 from the twelve urban exposure transect sites suggest that urban proximity by itself does not account for observed variations in bird species diversity and relative abundance (Table 1). In particular, many of the bird species were observed in every transect site, regardless of the level of urban exposure. For those bird species, which did not occur all sites, no clear relationship emerges with respect to changes in bird distribution and amounts of urban exposure. Instead, it appears more likely that some birds were patchily distributed or "rare" among the transect sites (e.g., common raven), resulting in varying patterns. This general pattern is reflected in all the bird species, including both habitat-specific chaparral birds and urban-associated species (Table 1), and, thus, it is not surprising that there is no relationship between bird species richness and urban exposure (Fig 4a). Relative abundance data for bird species are similar, with species-specific variation not associated with urban exposure changes. Instead, it appears that relative abundance data illustrate species-specific differences in population densities throughout the study area rather than differences associated with proximity to development (Table 1).

Small mammal data collected in the twelve sites in 1991 indicate similar results (Table 2 and Fig. 4b). Although high variation was observed in both small mammal species richness and relative abundance be-





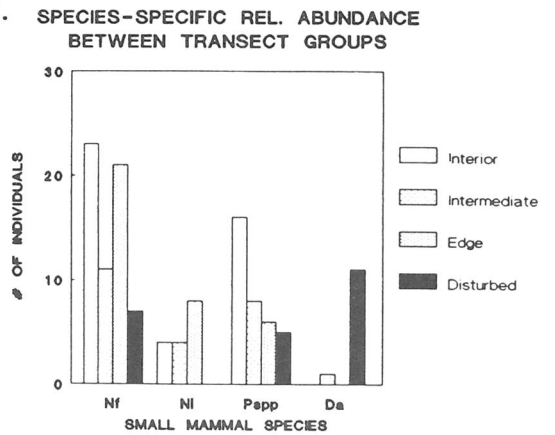


Figure 5. Small mammal relative abundance between the four disturbed sites and the twelve urban exposure sites based on 1991 data. Pacific kangaroo rats (*Dipodomys agilis*) and harvest mice (*Reithrodontomys megalotis*) (one individual was captured) are not included in these figures because these species were almost exclusively associated with disturbed sites (see text). (A) The mean munber of small mammal individuals per site was significantly less in disturbed sites than in the undisturbed urban exposure sites ($T = 3.12$, $p = 0.0081$). (B) When transect sites are divided into groups of four including the least urban exposed sites (interior), the moderately urban exposed sites (intermediate), the most urban exposed sites (edge), and the human-disturbed sites, a decline in the total number of individual small mammals observed in each of the transect groups is evident in the disturbed areas. This pattern is significantly different from the null hypothesis of an even distribution among the four transect groups ($G = 11.36$, $P<0.01$). (C) A similar decline is observed at the species-specific level between transect groups for intact chaparral species. Pacific kangaroo rats, on the other hand, show a large increase in disturbed sites. (Nf - *Neotoma fuscipes*, Nl = *Neotoma lepida*, Pspp = *Peromyscus* sp., and Da = *Dipodomys agilis*).

tween the transect sites, this variation was not found to be associated with changes in urban exposure. In general, many sites had woodrats (*Neotoma* sp.) and brush mice (*Peromyscus boylii*), with several additional species represented by single individuals captured throughout the study area (Table 2). No species-specific patterns appear to indicate a preference or avoidance of highly urban-exposed sites. In addition, no introduced, urban-associated small mammals such as house mice (*Mus musculus*) or black rats (*Rattus rattus*) were captured in any transect sites, even in areas immediately adjacent to established urban development.

Vegetation sampling results from 1991 indicate that transect sites were both physiognomically and floristically similar for woody species, with chamise and black sage dominant in most areas. Despite this general similarity, though, we did observe some differences in species composition and cover dominance between sites. However, preliminary analysis of this vegetation variation in combination with information on bird and small mammal species distributions suggest that vegetation differences account for little of the variability observed in the animal data.

Disturbed sites

Results from 1991 data collected in the four human-disturbed sites suggest that direct habitat alteration may be a more significant factor to consider than urban exposure alone. Although we are limited to data from only four sites, preliminary results suggest that small mammal abundance is lower for most species and that species composition varies in disturbed areas relative to undisturbed urban exposure sites. In particular, for the small mammals, disturbed areas included many more Pacific kangaroo rats (*Dipodomys agilis*), species that

tend to prefer more open areas, and harbored fewer of the chaparral mammal species that were relatively common in the twelve undisturbed urban exposure sites. Significant declines were found for small mammals, both in terms of the mean number of individuals captured per site and in the total number of individuals captured in disturbed versus undisturbed areas (Fig. 5a, b). In addition, this pattern of small mammal decline in disturbed sites was consistent among the species that occur in intact chaparral, with only the Pacific kangaroo rat showing marked preference for more open, disturbed areas (Fig 5c). Significant differences between the four disturbed and twelve "undisturbed" sites were not observed for bird species data collected in 1991, although it is possible that additional data collected in the future will indicate changes.

Vegetation data from the four disturbed sites are indicative of the human-caused habitat alterations that occurred in these sites during the past. Specifically, disturbed sites exhibited significantly less vegetation cover and structural diversity, shorter vegetation, and more forbs, grasses, and introduced species relative to the twelve undisturbed urban exposure sites. Although these habitat differences were a direct consequence of human disturbance, the disturbed sites sampled in 1991 are no longer being impacted by the most damaging human activities (e.g., illegal off-road-vehicle activities, dumping and trampling, etc.).

Discussion

Perhaps the most interesting result we have observed is the apparent lack of community-level changes associated with urban exposure for resident birds and small mammals. Specifically, changes in resident bird and small mammal species richness and relative abundance were not found to be associated with variations in proximity to development (urban exposure). Our 1991 data suggest that these species will continue to utilize intact chaparral habitat, even if it is located immediately adjacent to urbanization. This was found to be true for sites adjacent to well-established development (e.g., 20 to 40 years old) as well as for sites near recently graded but still undeveloped areas. Thus, our 1991 summer data do not support the hypothesis that intrusive impacts associated with urban exposure result in community-level edge effects; at least not for these species at this scale.

These results seem less surprising, however, when one considers that the potential edge effect mechanisms that could result in community-level changes for these species may not be operating in this system. For example, the threat of predation from urban-associated animals such as house cats may be mitigated by a high density of native predators such as coyotes and bobcats known to occur in the area. In our work, we have never observed a house cat in the chaparral habitat of the study area and a number of homeowners have told us that predation on cats by native predators is common. Thus, it is possible that a high density of native predators prevents "mesopredator release" (Soule et al. 1988), reducing the potential for impacts to native birds and small mammals from urban-associated predators.

A second factor that could help explain a lack of observed edge effects in this system is that all of the sampled sites are ultimately connected to a large, continuous habitat area of more than 5000 ha. To the extent that rapid recolonization of urban edge habitat is important in this system, there is clearly a large source pool from which potential bird and small mammal colonists could arrive. Thus, if declines or local extinctions were to occur near the edge, individuals from nearby natural habitats could recolonize the sites and "rescue" the declining populations. This type of "rescue effect" has been described by Brown and Kodric-Brown (1977) in the context of island biogeography. It is possible that without these continuous habitat connections in our study area, population declines would have occurred without corresponding recolonization in areas near urbanization. Extinction events have been documented in similar habitats for many of these same species in very small habitat isolates surrounded by urban development (Soule et al. 1988; Bolger et al. 1991). In our work, we hope to begin examining the effects of isolation on community patterns by sampling in small habitat areas (e.g., 10 ha) completely surrounded by urban development.

Finally, and perhaps most importantly, all twelve urban exposure sites were located within areas of relatively undisturbed, intact chaparral. Because we were interested in examining the role of urban proximity *per se*, we intentionally chose undisturbed sites that varied only in their degree of exposure to urban development. As a result, we may have prevented ourselves from observing any of the more apparent human-caused "edge effects" because our sites exhibited few obvious human impacts to the habitat. It is likely that many of the resident birds and small mammals we examined respond primarily to habitat structural integrity and resource conditions. Unless there are disturbances to these habitat characteristics, the focal species we have chosen may persist, despite being immediately adjacent to urban development.

Our preliminary data from the four disturbed sites support this conclusion. In particular, declines in small mammal abundance and species composition changes suggest that in areas of human-caused habitat damage, one should expect to see community-level changes. It is

this type of direct habitat alteration that we believe may be very important and a more significant factor to consider than just proximity to development alone. This conclusion may be especially relevant in chaparral communities where the resistive nature of the habitat may allow high quality, intact chaparral to persist, even adjacent to extensive human modified areas. We found that in many areas near urbanization, the thick chaparral habitat presented a formidable wall that could help reduce the impacts from nearby human activities from moving into the natural habitat. At the same time, however, areas within the chaparral that were subjected to concentrated human disturbances did show obvious habitat changes. Location of these disturbed sites were not correlated with urban proximity, but instead appeared to reflect historic human use and access patterns in the study area. It is within these human-disturbed sites that we believe community-level impacts will be most severe.

In future work, we will be investigating both urban proximity and habitat disturbance in much greater detail. In particular, we will continue to monitor our urban exposure sites and we are planning to sample in a large number of sites altered by human activities. Using these additional data, we hope to obtain more definitive conclusions regarding the relative importance of urban proximity and direct human disturbance on wildlife communities.

A final component of this project involves developing conservation recommendations based on our results. Although we do not yet have sufficient data to discuss specifics, several general comments are possible. First, our results to date suggest that natural habitat very close to urban development can have important conservation value. The prevalence of a variety of species persisting in high quality, but highly urban exposed areas suggests that proximity to development alone should not be a justification for claiming habitat degradation. Second, direct habitat alteration likely impacts wildlife more than urban proximity and may often be the key factor to consider. A clear recommendation from this conclusion is to minimize habitat disturbance in areas reserved for wildlife. Third, we believe that restoration of disturbed sites may provide significant wildlife benefits. Our data from the four disturbed sites indirectly support this because all of these areas are slowly and successfully recovering from past disturbance. Much of the habitat and wildlife species in these areas now have probably recently been reestablished as natural recovery continues to take place and additional human disturbances are prevented. Carefully planned restoration activities in these, and similar areas, could accelerate this process and result in the more rapid reestablishment of native flora and fauna. Under current conditions, what is occurring is similar to the "rescue effect" described above, with recovery in the disturbed sites clearly benefiting by connection to intact habitat. However, it is possible that smaller isolated areas, following disturbance events, could take much longer to recover, if recovery occurrs at all.

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Initial Effects of Housing Construction on Woodland Birds along the Wildland Urban Interface

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Abstract. California's avifauna inhabit some of the fastest growing suburban areas in the country. There are, however, opportunities to study and perhaps compensate for the processes causing species extirpations. Bird species respond differently to human disturbance; studying these differences during habitat conversion can help to understand the nature of species declines. I used a large-scale development project (525 ha) to study woodland-bird population changes during construction. Populations were estimated using spot- maps on four plots; two plots were isolated by construction, with two adjacent (natural) plots used for comparison. Data on species extirpations caused by construction are equivocal because of natural variability in bird occurrence and abundance. A large number of individuals moved onto the study plots after habitat conversion occurred on adjacent lands, however, these population surpluses were lost one year later.

Keywords: Edge effects; fragmented habitats; human disturbance; woodland birds.

Introduction

The diverse fauna of southern California inhabit some of the fastest growing suburban areas of the nation. This juggernaut of land development has led to a pessimism among biologists that the region's human population is growing too fast to maintain the plant and animal species in its path. Such pessimism, however, overlooks the remaining opportunities to study and perhaps compensate for the processes causing species declines and ecosystem degradation. Although the conservation prognosis for southern California avifauna is not good, knowledge about the effect of human actions on birds could conserve at least half of the region's species.

Current patterns of bird species occurrence and abundance (Garrett and Dunn 1981) show that the process of species decline is neither simple nor inevitable. Almost half of the bird species in southern California have suffered significant (>20%) losses of habitat since 1940 (see Grinnell and Miller 1944), but clear cases of extirpation are (as yet) uncommon. Despite wide-scale losses of habitat, species such as the golden eagle (*Aquila chrysaetos*), California gnatcatcher (*Polioptila californica*), and yellow-breasted chat (*Icteria virens*) still maintain populations in the region. Other birds, such as the corvids, have increased their densities and distributions by exploiting human-altered habitats (Anonymous 1990). Given this range of outcomes, it should be possible to explain some of the differences in bird species response to human activity by examining the natural history of individual species in urbanizing areas, particularly during times of habitat change.

Studies of urban birds have identified differences in bird community ecology between cities and adjoining wildlands (Catterall et al. 1989). Most researchers conclude that species richness is lower in urban areas, although biomass is typically higher than adjacent wildlands. Exotic species dominated most study areas in both number and biomass; however, a proportion of the native (and assumed original) species were found in urban habitats of all studies. Authors have attributed this persistence to the size of remnant habitat areas (Soulé et al. 1988; Blake and Karr 1987), population size (Bolger et al. 1991), volume of native vegetation (Mills et al. 1989), foliage height-diversity (Emlen 1974), habitat selectivity (Caterall et al. 1989), and food habits (Lancaster and Rees 1979).

In addition to the commonly studied features of habitat fragments (size, habitat quality, connectivity; see Blake and Karr 1987; Rosenberg and Raphael 1986; Lynch and Saunders 1991), woodland fragments in urban landscapes exhibit a unique set of edge conditions (Start 1991) associated with human habitats; introduced predators (Soulé et al. 1988), exotic species (Catterall et al. 1991), new source's of food and water (Emlen 1974), and outflows of pollutants and other materials (Low 1991; Start 1991). Urban areas concentrate these features in a manner that encourages diffusion into adjoining habitat

fragments, greatly increasing the importance of edge characteristics (Kelly and Rotenberry 1993). Finally, habitat conversion to urban uses is almost always a one-way process; relatively few areas ever return to anything close to original conditions (as might be expected in some forms of deforestation).

The importance of edge effects in urbanizing areas is exemplified by the pattern of habitat conversions in western Riverside County. Aerial photographs show that land development has occurred in over 300 separate patches, averaging 160 ha (Scott unpublished data). Had this development (50,000 ha) occurred as contiguous clusters around the five cites of the region, it would have created about 150 to 250 km of edge. Instead, the combined length of edge between these urban patches and adjacent wildlands is about 1600 km, roughly the distance from San Diego to Denver.

The most common configurations of woodland fragments are linear stream courses and patches of woodlands embedded with other vegetation types (see Scott 1991; Scott unpublished data). In the past it was often uneconomical to build on these locations because slopes were too steep for construction. At present, public demands for amenity value and recreation are forcing land development projects to avoid conversion of woodland areas. In either case, most woodlands left in urban landscapes are highly integrated with human habitats. Because of these factors, we can assume that edge effects may play a far greater role in urban landscapes than in rural forms of deforestation.

Most researchers studying urban birds have concentrated on the outcome of urbanization on bird populations, with studies undertaken 30 to 70 years after habitat conversion. These time lags kept most studies from examining any of the processes that lead to species extirpation. A few studies have used before- and-after surveys (Aldrich and Coffin 1980); but over a time interval and scale that preclude any examination of specific changes in habitat use. There is a need to study specific changes in bird species abundance and occurrence as urbanization occurs. These types of studies present the opportunity to examine both the evolutionary ecology of birds in changing environments and to identify bird sensitivities to human disturbance.

The construction of a large-scale housing project in Orange County, California, provided an opportunity to look for the specific changes in habitat-use and populations of oak woodland birds during the process of conversion. This project will convert 525 ha of oak woodland and sage scrub habitats into urban land-uses over a period of 6 years (1989-1995). This paper is a description of the initial response of the area's bird species to brushing, slope grading, and construction of houses.

Study Area

The woodlands used in this study were located in two adjacent canyons in southeastern Orange County, California. The final plan for land development called for the removal of oak woodland and sage scrub vegetation from the hills along both canyons. The canyon slopes and bottoms (110 ha total) were designated open space and became linear habitat fragments (Fig. 1). The canyons are separated by about 1000 m (350 ha of converted habitats). All areas to the south of the canyons were also cleared of vegetation. Areas to the north, however, were placed into a regional wilderness park. Thus, the woodlands of both canyons were divided in half; the downstream portion of each canyon (15 and 25 ha respectively) were to be surrounded by housing while the upstream portions (about 15 and 30 ha) remained in

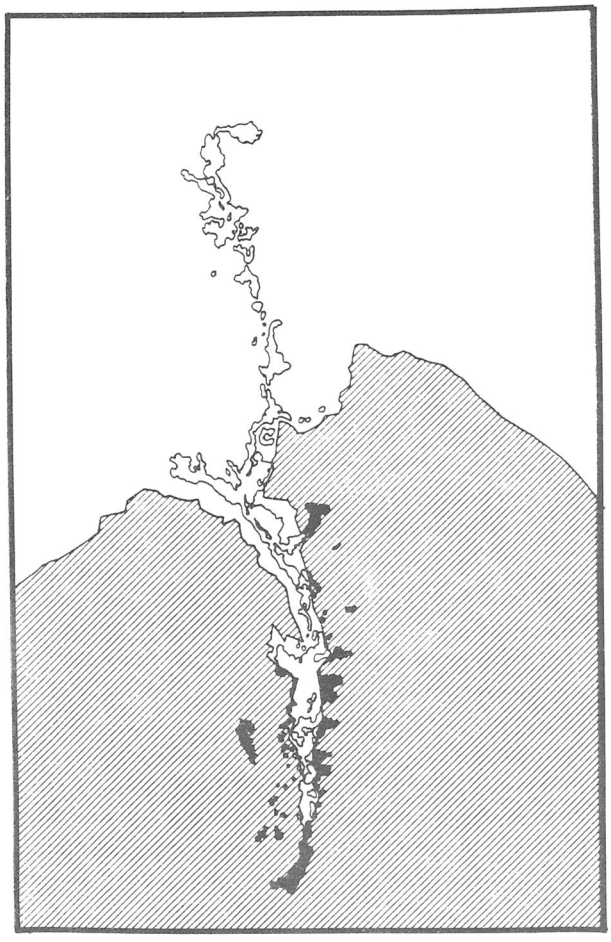

Figure 1. Map of the Borrego Canyon woodland area. Open polygons represent the remaining oak canopy area after vegetation was cleared in 1989. The black (solid) polygons represent oak canopy lost during clearing. The cross-hatched area represents the extent of area where vegetation was cleared. Scale = 1:20,000.

a matrix of native vegetation. This configuration of park and linear open space created four plots: two linear remnants of woodlands to be bordered by development and two comparison plots. The entire area of open space was used in linear plots; a equal sized area (15 and 25 ha, centered along the stream course) for each of the park plots. The linear plots in Lower Borrego and Lower Serrano were used-to record change in bird use of woodlands as development occurred; the park plots in Upper Borrego and Upper Serrano were used to compare changes at the linear plots to naturally occurring fluctuations in bird use of the region's woodlands.

Methods of Bird Observation

Starting in 1989, bird species occurrence and abundance in each of the four plots was measured using spot maps (Kendeigh 1944; Anonymous 1970) of singing males, with observations recorded on 1:800 scale maps. Each plot was censused 10 times per breeding season (March through June), with a 7 day interval between surveys. All surveys were made from sunrise to 5 hr past sunrise. Observers also recorded sightings of females and young, and made systematic nest searches to confirm breeding locations.

Results and Discussion

Effects of construction on woodlands

Housing construction requires the removal of vegetation, the leveling of slopes, and the building of structures, capital facilities, and landscaping. Thedirect impacts of these activities is relatively easy to describe (eg., loss of habitat), however, indirect or inadvertent impacts are much more insidious, unpredictable, and difficult to describe.

Environmental reviews typically assess the impacts that can be envisioned before a project is undertaken. Environmental biologists must estimate these impacts *a priori*,because they are seldom paid to observe impacts as they occur. Effects of construction that are poorly documented during environmental reviews include: trespass prior to and during construction, the timing of brushing and grading, and unauthorized use of open space areas.

Trespass in the canyons had become a chronic problem when we began fieldwork in 1989. Observers encountered 5 to 10 persons a day in the study area during the first two years of the project. After 1990, the only natural areas remaining on the construction site were the open space areas, so trespassers funneled down both canyons. More critical, the woodlands became overflow use areas for construction activities. The open space area was used for incidental storage of construction materials, workers used the woodlands as a toilet and dumping ground. Roads in the woodlands were used for access to construction sites.

The majority of habitat conversion was completed by the winter of 1990. Vegetation brushing around Lower Borrego began in 1989 and was completed by fall of that year. Most of the area (400 ha) between the two canyons was cleared by the end of 1989. Work continued through the 1990 breeding season on 3 to 5 ha next to Lower Serrano. Finally, one side of Lower Serrano was left in a disturbed condition (after 1990) but had not been cleared of vegetation by the end of the 1991 breeding season.

About 30% (3 ha) of the woodland canopy was removed in Lower Borrego; 15% (2.5 ha) in Lower Serrano. As of spring 1991, the former had 7 ha of woodland and 8 ha of sage scrub after clearing and the latter had 15 ha of woodland and 10 ha of sage scrub within the plot (area not graded by 1991). Streambeds of both areas were 15 to 30 m lower than elevation of surrounding housing pads. Debris from construction (such as paint, trash, concrete, and lumber) flowed downhill into these areas. Materials used to fill and level construction sites (limestone) eroded into the canyons, causing sediment deposition which in some places covered plants and soils to a depth of 20 cm.

Response of bird populations

A total of 43 bird species (18 families, Table 1) were confirmed as breeding on the plots between 1989 and 1991. Three species were absent for one year; two were absent for 2 years. In all cases these species were relatively rare (< 3 pairs per year per plot) in the study area and their absence could be the result of sampling error alone. Species richness in the study area was 40 species in 1989 and 1991, and 41 species in 1990 (Table 2).

Individual plots had fewer species than the entire area, ranging from 63% to 85% of total species recorded in any year. Plots shared 24 breeding species (56% of total); 28 (65%) bred in at least three plots, and 37 (81%) bred in at least 2 plots. Only 17 (40%) species bred in all 4 plots in all 3 years. Eight species bred on only one plot: 4 of these species bred in only one year.

All plots showed slightly lower numbers of species in 1991 than in 1989 (Table 2), but data on species extirpation are equivocal (Table 3). Plots embedded in natural vegetation lost 12 species and gained 3 species over the two years of the surveys.

Table 1. Total number of bird species breeding in the study plots (1989-1991), grouped by family.

Family	Number of species/year 1989	1990	1991
Accipitridae	3	3	3
Phasianidae	0	1	1
Columbidae	1	1	1
Tytonidae	1	1	1
Strigidae	1	1	1
Trochilidae	3	3	3
Picidae	3	3	3
Tyranidae	4	4	4
Corvidae	2	2	2
Paridae	1	1	1
Aegithalidae	1	1	1
Troglodytidae	3	3	3
Muscicapidae	2	3	2
Mimidae	1	2	2
Ptilogonatidae	1	1	1
Sturnidae[1]	1	1	1
Vireonidae	1	1	1
Emberizidae	11	9	9

[1] exotic family (European Starlings)

Table 2. Number of breeding bird species in the 4 study plots in southern Orange County, from 1989 to 1991.

Location	Number of species/year 1989	1990	1991
Lower Borrego	28	28	25
species lost		1	3
species gained		1	0
Upper Borrego	34	34	30
species lost		1	5
species gained		1	1
Lower Serrano	27	27	27
species lost		3	3
species gained		3	3
Upper Serrano	32	29	29
species lost		5	1
species gained		2	1
Study Area	41	40	41
species lost		2	0
species gained		3	1

The linear woodlands lost 9 species and gained 6 species. Three species, the red-tailed hawk (*Buteo jamaicensis*) and red-shouldered hawk (*Buteo lineatus*), and Pacific-slope flycatcher (*Empidonax difficilis*), bred in 1989 and 1991 (at Upper and Lower Serrano), but were not recorded in 1990. Changes in these species were counted twice (lost then regained) because the species did not nest onsite in the intervening year. Two species, the California quail (*Callipepla californica*) and mourning dove (*Zenaida macroura*) bred on both linear plots in 1990 where they had not been recorded in 1989 (gained), but were not present in 1991 (lost). Species reoccupation of plots only occurred at Lower and Upper Serrano.

Table 4 shows changes in the spot-mapping estimates of population size of 17 species that bred in all 4 plots. Ten of the 17 species (60%) experienced large population increases (54% to 355%) in 1990 followed by declines (to 1989 levels) in 1991. Five species experienced little population change (0 to 20%) over the three years; one species experienced an increase and one species declined. The greatest changes occurred at Lower

Table 3. Bird species lost and gained from individual plots in southern Orange County, between 1989 and 1991.

Species lost in 1990

American kestrel (US)[1] (*Falco sparverius*)
Red-shouldered hawk (LS,US) (*Buteo lineatus*)
Red-tailed hawk (US) (*Buteo jamaicensis*)
Pacific-slope flycatcher (LS) (*Empidonax difficilis*)
Northern mockingbird (LB) (*Mimus polyglottos*)
Lazuli bunting (UB) (*Passerina amoena*)
Blue grosbeek (US) (*Passerina caerulea*)
Grasshopper sparrow (US) (*Ammodramus savannarum*)

Species gained in 1990

California quail (LS) (*Callipepla californica*)
Mourning dove (LS) (*Zenaida macroura*)
Black-chinned hummingbird (UB,US) (*Archilochus alexandri*)
California thrasher (LS) (*Toxostoma redivivum*)

Species lost in 1991

California quail (LS)[2]
Mourning dove (LB)[2]
Costa's hummnigbird (UB) (*Archilochus costae*)
Ash-throated flycatcher (UB) (*Myiarchus cinerascens*)
Common raven (US)[2] (*Corvus corax*)
California gnatcatcher (LS) (*Polioptila californica*)
American robin (US) (*Turdus migratorius*)
Northern mockingbird (UB)
Phainopepla (UB) (*Phainopepla nitens*)
Hutton's vireo (LS) (*Vireo huttoni*)
Orange-crowned warbler (LS,LB) (*Vermivora celata*)
Northern oriole (LB) (*Icterits galbula*)

Species gained in 1991

American kestrel (LS)
Red-shouldered Hawk (LS)[3]
Red-tailed hawk (US)[3]
Pacific-slope flycatcher (LS)[3]
Common raven (UB)
American robin (LB)

[1] Lower Borrego (LB), Upper Borrego (UB), Lower Serrano (LS), Upper Serrano (US).
[2] Species that bred in 1990, but were not observed in 1989 or 1991.
[3] Species that bred in 1989 and 1991, not observed in 1990.

Serrano (16 species increased in 1990, 10 declined in 1991), followed by Lower Borrego (15 species increased in 1990, 10 species declined in 1991). Fewer species (11) increased at Upper Borrego; Upper Serrano, which was the farthest from development, had more species decline than increase in 1990. The estimated number of all 17 species on all plots increased from 475 in 1989 to 730 in 1990, and decreased to 472 in 1991.

Anecdotal observations during brush removal from the adjacent hills suggest that birds were able to escape slow-moving machinery. However, brush clearing continued through the breeding seasons of 1989, 1990, and 1991, destroying nests, eggs, and young.

To date, the loss of any species from the linear woodlands could be attributed to either natural variation or human activities. The plots embedded in the natural vegetation matrix lost twice as many species as the linear woodlands. The number of species at Lower Borrego dropped by three but remained unchanged at Lower Serrano. These levels of species richness, however, were maintained through substitution of species, rather than species persistence.

One of the species that moved into Lower Borrego was the mourning dove, a native species commonly associated with human habitats and cities (Tweit and Tweit 1986; Mills et al. 1989; Bessinger and Osborne 1982). However, it was lost from Lower Serrano during the same period and was rare on all plots. The northern mockingbird (*Mimus polyglottis*), another urban-associated species, was lost from both Borrego plots between 1989 and 1991. Other native species (Table 3) that moved onto the plots are relatively common in the region (Garrett and Dunn 1981) but are not usually found in urban settings. All plots had populations of European starlings (*Sturnus vulgaris*), however, another common exotic, the house sparrow (*Passer domesticus*), was not detected during any survey of any plot.

Table 4. Annual change (1989-1991) in spot-map counts on four woodland study plots in southern Orange County. LB (Lower Borrego) and LS (Lower Serrano) are linear plots bordered by housing pads; UB (Upper Borrego) and US (Upper Serrano) are plots located in wildland park. Annual changes in the number of singing males are recorded in five categories: greater than 2x increase (++), increase (+), no change (0), decrease (-), and decrease by > 50% (--).

	1989 to 1990				1990 to 1991			
	LB	UB	LS	US	LB	UB	LS	US
Woodland Species[1]								
Nuttall's woodpecker (*Dendrocopos nuttallii*)	+	++	++	0	0	-	0	0
Anna's hummingbird (*Archilochus anna*)	++	+	++	-	--	0	-	0
Pacific-slope flycatcher (*Empidonax difficilis*)	++	+	++	-	0	0	0	++
Ash-throated flycatcher (*Myiarchus cinerascens*)	++	+	++	-	--	-	--	--
Plain titmouse (*Parus inornatus*)	++	+	++	+	--	--	0	-
House wren (*Troglodytes aedon*)	++	++	++	0	-	0	-	0
European starling (exotic) (*Sturnus vulgaris*)	+	--	++	--	-	-	0	0
Hutton's vireo (*Vireo huttoni*)	0	0	++	-	0	+	--	-
Orange-crowned warbler (*Vermivora celata*)	++	++	0	--	0	--	++	++
Srub/Woodland Species								
Wrentit (*Chamaea fasciata*)	++	++	++	++	--	--	0	--
Common bushtit (*Psaltriparus minimus*)	++	0	++	-	0	0	0	+
Bewick's wren (*Thryomanes bewickii*)	+	+	++	+	--	-	-	-
Cactus wren (*Campylorynchus brunneicapillus*)	+	-	++	+	0	-	--	--
California towhee (*Pipilo fuscus*)	++	++	++	-	--	--	-	-
Rufous-sided towhee (*Pipilo erythrophthalmus*)	++	++	++	+	--	--	-	--
Lesser goldfinch (*Carduelis psaltria*)	--	-	++	0	+	--	-	-
House finch (*Carpodacus mexicanus*)	++	+	++	-	--	--	--	--

[1] see Zeiner et al. (1991) for description of species habitat-associations.

All of the species recorded as lost or gained were relatively rare on the plots. Their breeding status on the plots may fluctuate as a function of source populations in the vicinity of the canyons (Pulliam 1989), or because they are always rare and experience local extirpations (Shaffer 1987). As such, it is premature to associate human causes with the annual changes observed in species composition. Nevertheless, the woodlands have a high potential for species extirpation because so many of the species occur in small populations (Bolger et al. 1991). This danger is exemplified by the number of changes in bird occurrence on the plots; which averaged 2.75 losses and 1.5 gains of species per plot per year.

The changes in population size of individual species were far less equivocal. Species commonly associated with scrub habitats, particularly those using both scrub and woodland habitats, experienced sharp increases at Lower Borrego and Lower Serrano after the adjacent areas were cleared of vegetation. Upper Borrego, which was the next closest plot to the disturbed area, also had increases in species number, but with fewer species. Only two species increased at Upper Serrano, which was the farthest away from disturbance (by 300 m).

Species increases and decreases suggest that 1990 breeding populations may have been augmented by individuals from adjacent, converted scrub habitats. Birds that use a combination of scrub and woodland habitats showed the most consistent patterns of post-conversion increases. Birds that occur primarily in woodlands (Table 4), which suffered less conversion of area, experienced far smaller increases across the study area. Nevertheless, some of these species also experienced sharp increases in the linear plots. These population increases follow a pattern identified by Whitcomb et al. (1982) and Lovejoy et al. (1986), where bird populations in habitat fragments increase as adjacent habitats are lost, then decrease because of crowding and lack of resources.

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Habitat Linkages in an Urban Mountain Chain

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Abstract. Records were obtained from agencies responsible for the pick up of road kills in the Puente Hills, the adjacent portion of northwestern Orange County, and the Verdugo Mountains in the northeastern portion of the San Fernando Valley in an effort to identify existing wildlife corridors linking these locations with other open space. Both the Puente Hills and the Verdugo Mountains contain substantial acreage of existing open space, maintain existing populations of wildlife, and are virtually surrounded by urban development. Adjacent communities recognize the importance of preserving these open space areas, but the habitat value of these large expanses of open space is seriously compromised unless habitat linkages can be preserved. Records of road kills picked up include the date, type of animal removed, and location of pick up. Mapping these records identified potential linkages on the basis of road kill frequency and provided information on species utilizing linkages and seasonal changes in movement. Analysis of changes in linkages over time were included where records were available. Monitoring such records over time may result in more accurate analysis of impacts associated with increasing habitat fragmentation.

Keywords: Carbon Canyon; Chino Hills; Coyotes; habitat fragmentation; habitat linkages; land conversion; muledeer; oppossum; Puente Hills; raccoons; road kills; San Rafael Hills; skunks; Verdugo Mountains; Whittier Hills.

Introduction

Land conversion in California in general has resulted in the decline of populations of animal and plant species largely as a result of accelerating habitat loss (Jensen et al. 1990). However, the number of populations perched on the edge of extinction is not simply the result of direct habitat loss. An even larger threat to existing biodiversity in California is the fragmentation of remaining open space (Soule et al. 1988). This fragmentation is not only the result of large scale development, such as the creation of new communities taking place in southwestern Riverside County and northern San Diego County; fragmentation also results from smaller scale land conversion, particularly at the wildland-open space interface. Until recently, environmental review of biological issues centered on housing density and the presence of sensitive plant and animal species. As the emerging discipline of conservation biology has begun to bring long term effects of fragmentation, localized extinctions leading to larger scale extinctions, into focus, the overall extent of open space loss and the effect on regional wildlife are gaining importance in the environmental review process. Consequently, wildlife corridors or habitat linkages, areas of open space linking other areas of open space, have gained importance in discussions of impacts associated with land conversion.

The primary theoretical framework underlying the attempt to understand the problem of habitat fragmentation resulting from land conversion is the modelling of equilibrium numbers of species on islands (MacArthur and Wilson 1967; Diamond 1975). Briefly stated, MacArthur and Wilson's model predicts that the number of species occupying an island is a function of size of the island and distance from the colonizing source. Distance from a colonizing source influences the arrival of new individuals, therefore species, on the island, and the size of the island influences the likelihood of extinction since smaller areas support smaller populations which in turn are more at risk of localized extinction (MacArthur and Wilson 1967). The equilibrium number of species present on an island is determined by the balance between the processes of extinction and immigration, so while the species composition of an island may change over time, the total number of species present will remain constant at the equilibrium number (Mac Aruthur and Wilson 1967).

Land conversion effectively isolates areas of open space supporting natural communities leaving patches of open space in a matrix of urban development (Noss 1983). For this reason, the principles of MacArthur and Wilson's model have been used to predict rates of extinction on shrinking islands of open space (Diamond 1975) Extinction within isolated small patches of open space occurs because such patches are incapable of supporting self-sustaining populations of larger predators which require a large resource base (Frankel and Soule 1981; Beier 1991) and also because certain species are good colonizers but are not good persisters (Soule et al. 1988). For example, birds in general are good colonizers (Case and Cody 1987); however, populations are often transient, and turnover is high. In this situation, isolated patches of habitat are likely to loose such populations permanently since an avenue for recolonization no longer exists (Soule et al. 1988; Boger et al. 1991). A third factor in localized extinction in small isolated patches may be the elimination of habitat heterogeneity, but the isolation resulting from fragmentation plays an important role in localized extinction. Without connections to larger open space areas, even large expanses such as the Santa Monica Mountains, large predators are doomed to localized extinction in a matter of hundreds of years (Edelman 1990). Habitat linkages effectively increase the total habitat value of existing open space by connecting it to other open space (Edelman 1992). These connections increase the total amount of open space available to any given individual by allowing movement between open space patches which in turn increases population size and decreases the probability of extinction. Such connections also provide a conduit for gene exchange between populations occupying different patches of open space, and allow recolonization of patches in which localized extinction has occurred (Frankel and Soule 1981; Soule and Simberloff 1986; Liberstein 1989) In particular, the determination of impacts associated with loss of open space in this region should take into consideration the impact of fragmentation as well as the impact of habitat loss for the entire area encompassing the Chino, Puente, and San Jose Hills. Since these areas are potentially interconnected, loss of space in one area could impact all areas even if the impacts are not associated with loss in a "core" area actually connecting these open space areas.

Unfortunately, what exactly constitutes a habitat linkage is not as well understood as the benefits of habitat linkages (Liberstein 1989; Soule and Simberloff 1986). In an ideal world, habitat linkages would extend from ridgeline to ridgeline encompassing a diverse array of habitats in a wide corridor of open space. However, reality usually finds individuals and agencies arguing over how big and where a habitat linkage should be. Radio collaring and intensive photography or tracking are the most reliable methods of determining animal movement, but the use of these methods in standard environmental assessments is unrealistic given time and budget constraints. One alternative is the use of records of road kills to document areas animals are using to cross roadways bisecting open space. Records of road kills are kept by the Los Angles and Orange County Departments of Animal Regulation and the Los Angeles Department of Sanitation. These records provide the date, type of animal removed and the location of the pick up, and some of these records date back ten years which allows examination of changes in the frequency of road kills of species of interest over time. These records were used in an effort to assess wildlife movement in the Puente Hills, northwestern Orange County, and the Verdugo Mountains north of Burbank.

There are a number of assumptions inherent in the use of this type of data to attempt to identify movement corridors. First, road kill data only identifies areas where animals are attempting to cross roads. We are assuming that these roadway crossing areas are not only linking open spaces on either side of the road but are part of a potential corridor system. In other words the fact that wild animals are being hit on every major thoroughfare crossing the Puente Hills reflects the potential for movement throughout the hills, not just across streets. Second, road kills reflect species density to some extent; however, behavioral differences between species probably contributes heavily to the likelihood of making a successful crossing. One major limitation in the use of this type of data is that the lack of recorded road kills does not *a priori* translate into a lack of animal movement since traffic density, to some extent, dictates the frequency of road kills. A corollary to this limitation is that the presence of increased road kills may not be a function of changing movement patterns or populations, but instead may be a function of increased traffic density.

Our objectives for this study were to address the following questions:

1. Do road kill records provide information about regional wildlife movement

2. Is it possible to identify habitat linkages by mapping recorded road kills?

3. Is it possible to use road kill information to gain an historical perspective documenting shifts in wildlife activity as a consequence of regional development?

Methods

Records of road kills were obtained from the Los Angeles County Department of Sanitation, the Orange County Department of Animal Control and The Los Angeles Department of Animal Control. The records were scanned and records of road kills of wild animals were noted and the species, date, and address of the road kill pick up were recorded. Records for 1990 and 1986 were mapped using the Thomas Brothers maps of Los Angeles County. Point symbols were used to indicate locations where two or more road kills of target animals (coyotes, foxes, deer) occurred.

Results

Verdugo Mountains

Locations of coyote and deer road kills in the western portion of the Verdugo Mountains are shown in Figure 1. No road kills were reported from the portion of Foothill Blvd. crossing the Tujunga Wash. This probably reflects the presence of bridges which the animals use as under passes to cross the 210 Freeway as well as Foothill Blvd. Two road kills, a deer and a coyote are reported from the vicinity of the intersection of Wentworth and Mary Bell Avenue in a 24 month period spanning 1990 and 1991 (Fig. 1). In addition, a number of animals, (four coyotes) were killed attempting to cross Sunland Blvd. between Sunland Way and the Green Verdugo Fire Road. Road kills are not reported from elsewhere in the vicinity; for example it is unlikely that animals are moving underneath the 210 Freeway at Sunland Boulevard, since road kills of deer and coyotes are not reported in this area (Fig. 1). This suggests that the area between Wentworth Street and Sunland Blvd. may be an important conduit for animals moving between the San Gabriel Mountains and the Verdugo Mountains (Fig. 2,3).

The species composition of 1990/1991 road kills in the eastern edge of the Verdugo Mountains also known as the San Rafael Hills differs from that of the the western edge (Fig. 4). There are no records of road kills of coyotes or deer in the portion of La Canada Flintridge between the 210 Freeway and the San Rafael Hills, but there are records of deer and coyote road kills in neigh-

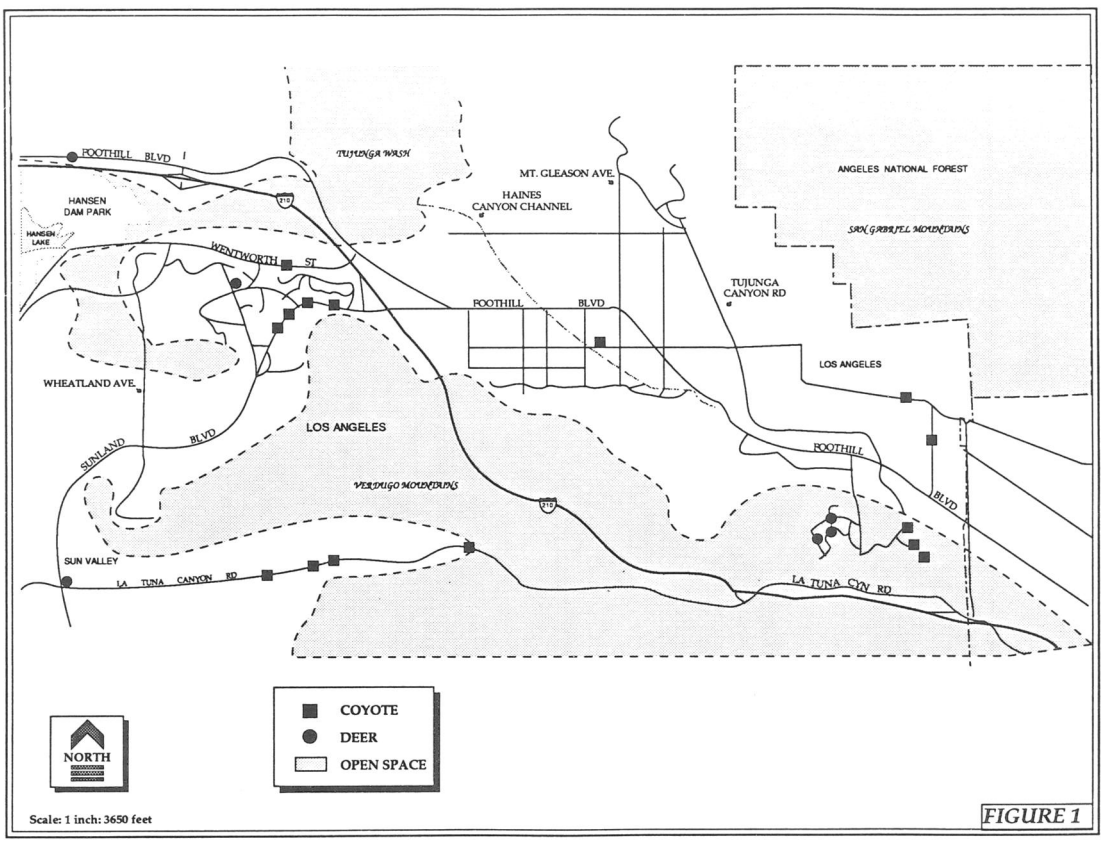

Figure 1. Locations of large mammal road kills during 1990 and 1991 at the western edge of the Verdugo Mountains.

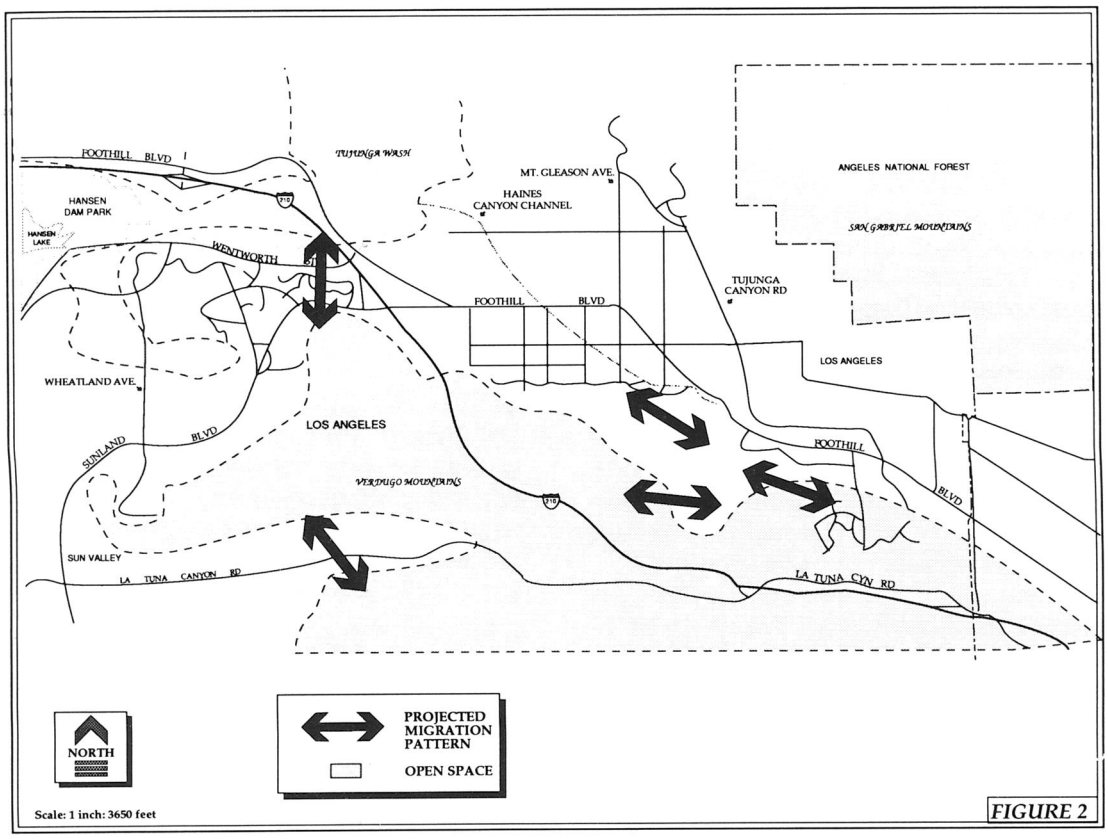

Figure 2. Projected corridors of movement based on locations of road kills of large mammals at the western edge of the Verdugo Mountains.

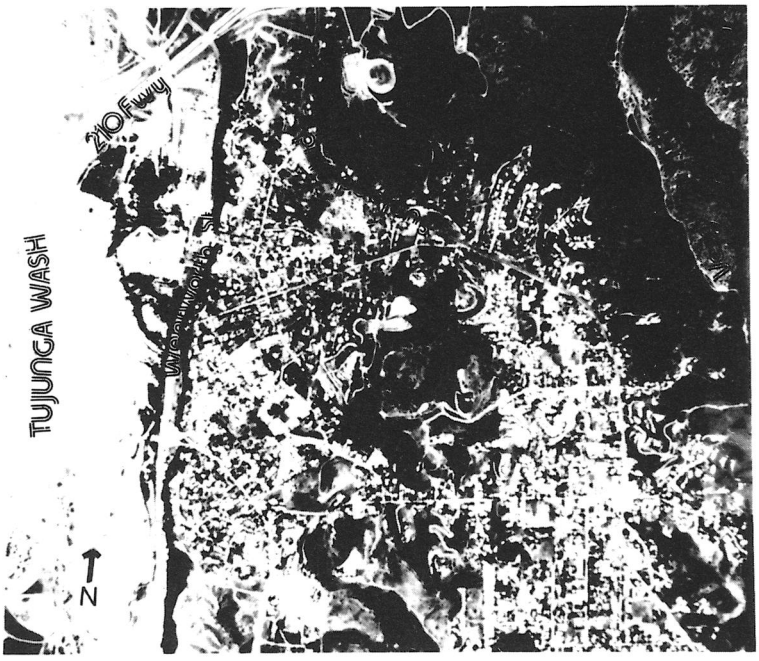

Figure 3. Areal photograph taken in 1991 of the western edge of the Verdugo Mountains showing the Tujunga Wash.

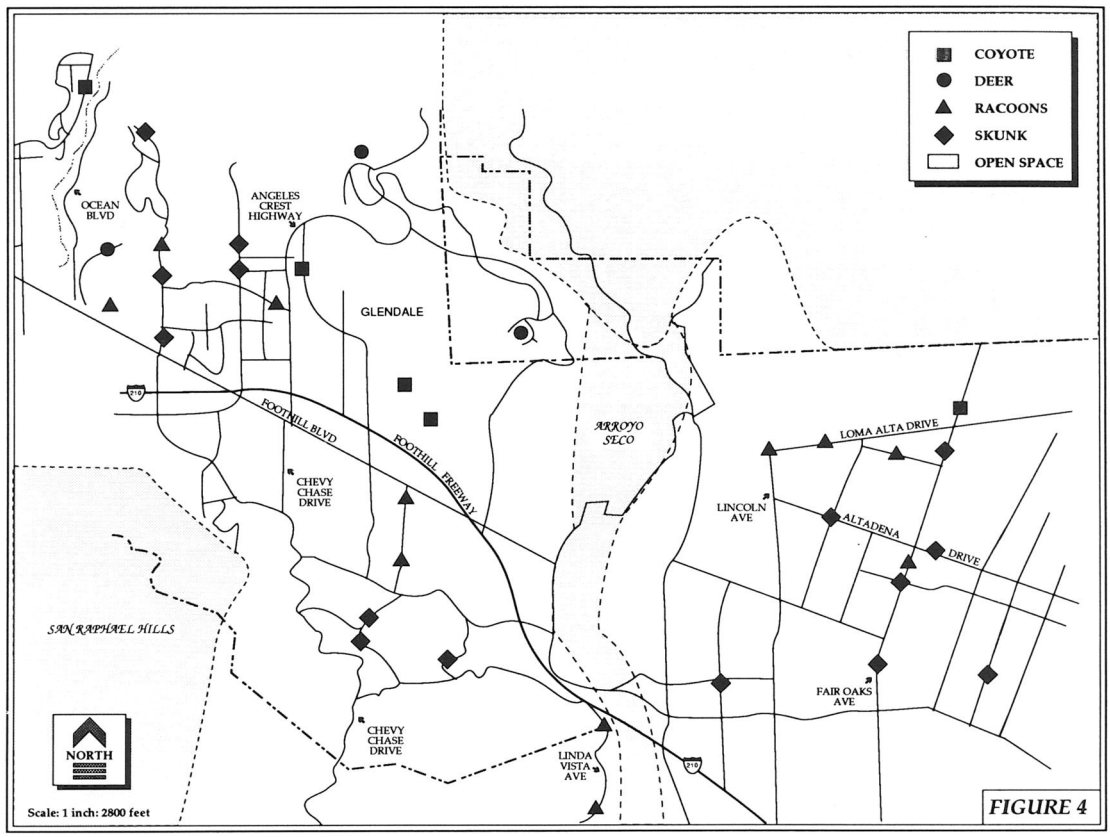

Figure 4. Locations of large mammal road kills in the San Rafael Hills during 1986 and 1990/91.

borhoods north of the 210 Freeway. The total number of coyote and deer road kills reported in the area bounded by Lincoln Avenue to the east, the 2 Freeway to the west and the San Rafael Hills and San Gabriel Mountains to the south and north respectively declined from 1986 to 1990/91 although the incidents of raccoon road kills increased over this same time period (Table 1).

Puente Hills

The Puente Hills extend northeast from Whittier, encompassing the Whittier Hills, to the edges of Walnut and Diamond Bar which border the San Jose Hills to the northeast (Fig. 5). The Chino Hills, which connect to the Santa Ana Mountains, begin to the east of the San Jose Hills. In 1986, a total of four coyotes were recorded hit

Table 1. Road kills reported from the western Verdugo Mountains, eastern San Rafael Hills, and Whittier Hills.

Location	Year	Skunk	Deer	Raccoon	Coyote	Owl	Fox	Snake
		Number of individuals per month						
Verdugo Mts.	1990/1991	0.08	0.21	0.33	0.625	0.12	-----	0.312
San Rafael Hills	1986	2.0	0.23	0.53	0.23	-----	-----	-----
	1990/91	1.56	0.125	1.56	0.187	0.12	-----	0.312
Whittier Hills	1986	3.30	0.615	0.46	0.53	-----	-----	0.92
	1990/91	1.37	0.31	1.06	0.81	-----	0.06	0.31

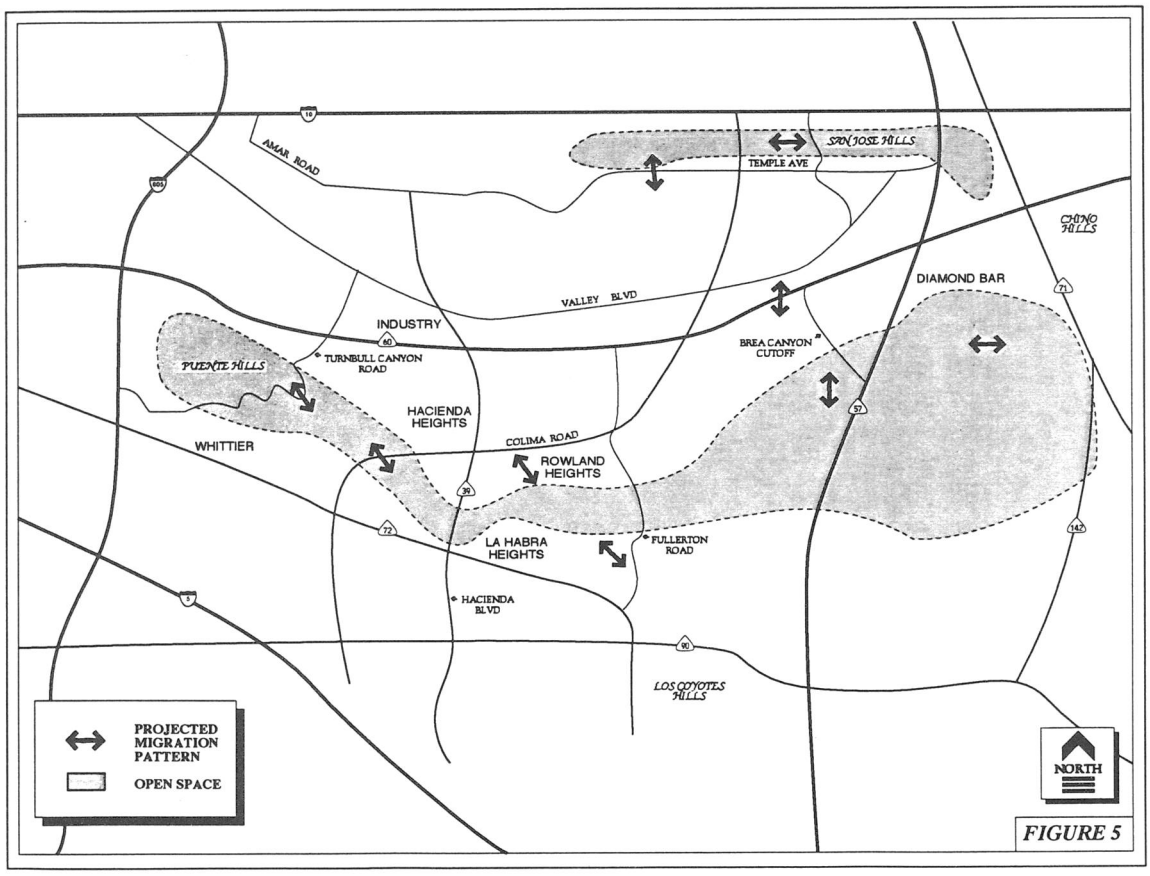

Figure 5. Regional map showing projected corridors based on locations of multiple large mammal road kills in the Puente Hills, San Jose Hills and the Chino Hills.

on Colima, and a deer was hit crossing Turnbull Canyon (Fig. 6). In 1990, a single coyote was reported from Turnbull Canyon and a fox and a coyote were hit crossing Colima in the same vicinity as the 1986 road kills (Fig. 6). These data indicate that animals were attempting to move across major roadways in the southwestern portion of the Puente Hills from Hacienda Heights to Workman Mill Road. This is a substantial area of open space when considered as whole, but is less substantial when considered to be separate patches of open space bisected by at least two major north/south thoroughfares (Fig 5, 6).

The community of Diamond Bar lies between the Puente Hills to the south, the Chino Hills to the east and the San Jose Hills to the north (Fig. 5). Numerous road kills, mainly skunks, were recorded as road kills in 1986; however, during 1990 and 1991 no road kills were reported in the same area (Fig. 7). This is understandable in light of the difference in population density indicated by the 1990/1991 Thomas Brothers map showing build out of streets northeast and east of the concentrated area of road kills as compared to the 1986 Thomas Brothers map (Fig. 7). While a number of coyote and deer road kills are reported in the area east of the Orange Freeway in 1986, road kills of these larger animals are present in fewer areas in 1990/1991 and are limited to the western side of the Orange Freeway .

The total number of road kills was similar in 1990/1991 and 1986 in the Walnut area; however, the area of concentration shifted to the southwest mirroring increased density in 1990/1991 as evidenced by road expansion to the northeast and southwest of the area encompassing reported road kills in 1986 (Fig. 8). The differences between the 1990/1991 and 1986 patterns of road kills suggests that as open space contracted and traffic patterns changed, the patterns of road kills also changed in such a way that large animals such as coyotes and deer (indicated by arrows) were hit attempting to cross roadways bisecting the remaining open space in 1990/1991 (Fig. 8). Overall, recorded road kill incidents of skunks, deer and snakes were lower in 1990/1991 relative to 1986; however recorded road kills of coyotes and raccoons were higher in 1990/1991.

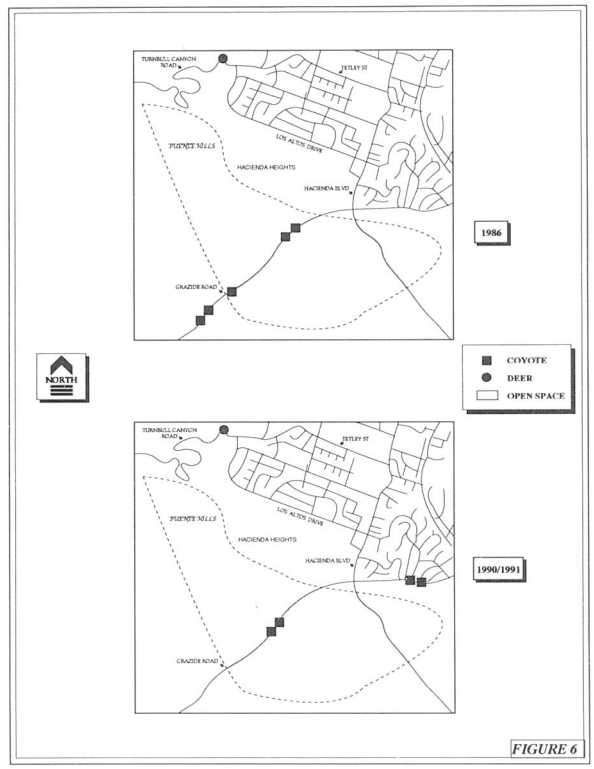

Figure 6. Locations of large mammal road kills in the Whittier Hills during 1986 and 1990 through 1991.

Northwestern Orange County

Figure 9 identifies locations of road kills in northwestern Orange County. This is a portion of the county where large open space areas are divided by major roadways. Because the data acquired from Orange County has some gaps (months where officers did not record the locations of road kills) it is not possible to generate documentation of shifts over time in the locations of kills in this part of the county. However, it is fairly evident that habitat linkages exist in the area of La Vida Hot Springs, where three deer and two coyotes were killed on Carbon Canyon Road between 1990 and February of 1992, and along Brea Canyon near the 57 Freeway, where two deer were killed in 1991. Also of interest were a cluster of road kills of smaller mammals and snakes in the developing area along Fairmont Boulevard, south of Chino Hills Regional Park.

Discussion

This study provides the beginning of an in-depth look at wildlife movement in the Verdugo Mountains, the Puente Hills, and adjacent portions of northwestern Orange County. These areas have been under heavy development pressure in the last decade, yet both of these areas still encompass large amounts of open space. The results of this study suggest that movement of large animals such as coyotes and deer throughout the remaining open space area included in the Puente Hills is possible since animals are being hit crossing roadways bisecting this total area. If the Puente Hills present an uninterrupted swath of open space to the Chino Hills which are connected to the Santa Ana Mountains, then the value of these hills as wildlife habitat increases as a result of total size and connectivity (Fig. 5).

The presence of existing interconnected cores of open space providing habitat including chaparral, coastal sage and riparian woodland in the Puente Hills, Chino Hills, and San Jose Hills brings up some important considerations. Presently there are several large parks in

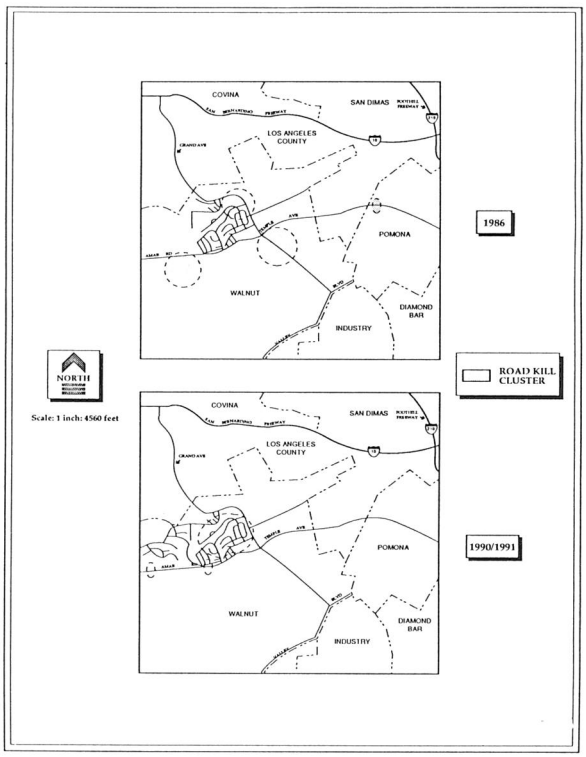

Figure 7. Locations of large mammal road kills reported from the vicinty of Walnut during 1986 and 1990 through 1991.

this area. The value of these areas, Hellman Park in the Whittier portion of the Puente Hills, Schabarum Regional Park in Rowland Heights, and Galster Regional Park, as habitat for native species will be compromised if fragmentation proceeds in the existing core of open space (Fig. 5). If conservation of open space for wildlife habitat within these areas is a target consideration, connections between the Chino Hills, San Jose Hills, and Puente Hills need to be preserved in order to maximize viable open space available to wildlife. In addition, the link to the Chino Hills is critical since the Chino Hills connect to the Santa Ana Mountains which likely act as colonizing "source". Conservation of open space within these areas without the maintenance of connectivity will provide little in the way of valuable wildlife habitat. Since the results of this study suggest that movement of large mammals such as deer, coyotes, and fox is taking place across major roadways bisecting this area, potential connections for these patches of existing open space may exist. The cumulative impacts of land conversion interrupting the core of open space then become greater as movement throughout the entire region may be disrupted (Fig. 5). In particular, the determination of impacts associated with loss of open space in this region should take into consideration the impact of fragmentation as well as the impact of habitat loss for the entire area encompassing the Chino, Puente, and San Jose Hills. Since these areas are potentially interconnected, loss of space in one area could impact all areas even if the impacts are not associated with loss in a "core" area actually connecting theses open space areas.

The Verdugo Mountains appear to still be at least partially connected to the San Gabriel Mountains via the Tujunga Wash and the area of low density housing lying between Wentworth Street and Sunland Blvd. This connection is tenuous at best especially in light of proposed development in the Tujunga Wash and along the Wentworth Street corridor. Examination of the area reveals that there is an alternative link between the wash and the Verdugo Mountains which is south of the area of concentration of road kills (Fig. 3). The lack of road kills in this area may indicate that this is a more viable link; however, the cluster of road kills in the vicinity of Green Verdugo Road and Sunland Boulevard certainly indicates a crossing.

Particularly disturbing is the absence of road kills of large animals such as coyote and deer in the San Rafael Hills which comprise the eastern edge of the Verdugo

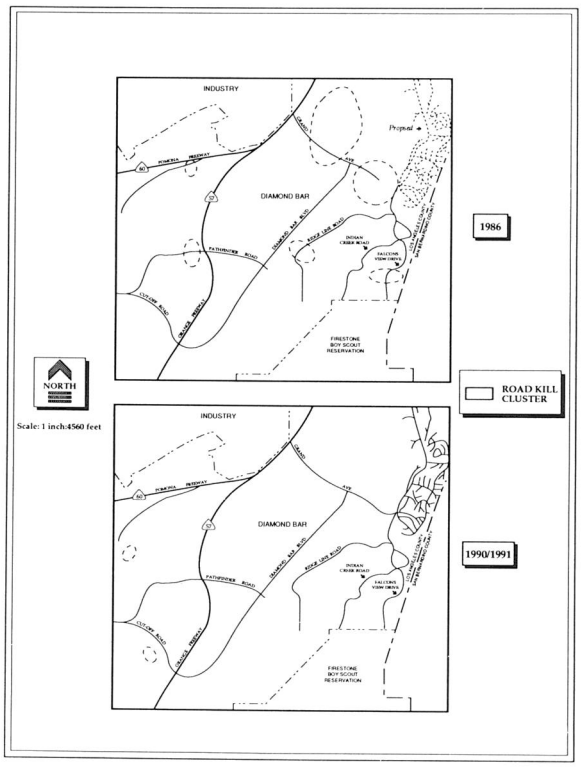

Figure 8. Locations of large mammal road kills reported from the vicinity of Diamond Bar during 1986 and 1990 through 1991.

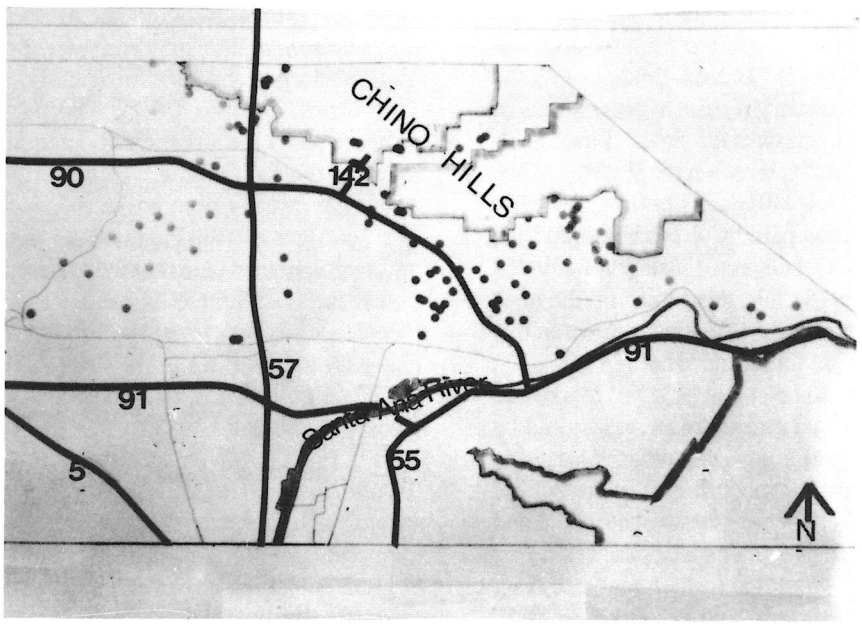

Figure 9. Locations of road kills reported from northwestern Orange County during 1990 through 1991.

Mountains. The absence of road kills in this area brings up the possibility that few, if any, animals are moving through this area from the San Gabriel Mountains via the Arroyo Seco. If a connection to the San Gabriel Mountains is not present at this eastern end of the Verdugo Mountains the only remaining connection is across the 2 Freeway from the Verdugo Mountains on the western side of the freeway. The San Rafael Hills and western Verdugo Mountains still encompass large expanses of open space. The Verdugo Mountains include a number of canyons several of which have permanent water and most of which have seasonal water and support riparian woodland (Swift 1992). In addition, there are two large parks in the Verdugos, Brand and Sunset, which include large amounts of preserved natural open space. However, the value of this habitat which is already protected may be compromised unless a linkage to the San Gabriel Mountains is maintained (Fig. 3).

This study suggests that road kill records do provide inexpensive and easily obtained information about regional wildlife movement and existing corridors of movement that serve as habitat linkages. Road kill data suggests that animals are moving from the Tujunga Wash through the Wentworth Street/Sunland Blvd. corridor, and these animals appear to be moving from the Tujunga wash to cross Wentworth where open space, the Sunland Tujunga ponds wildlife refuge, abuts Wentworth Street (Figs. 3,4). In addition, the study points to the need for increased attention to movement from the San Gabriel Mountains into the San Rafael Hills. Both deer and coyote kills are reported from the areas of La Canada and Altadena which border the Angeles National Forest in the San Gabriel Mountains (data not shown) yet road kills of these species are not reported from the edges of the San Rafael Hills (Fig. 1). Habitat linkages into the San Rafael Hills are necessary if present populations of large mammals are to persist over time. Finally, although regional connections between the Whittier Hills, San Jose Hills and Chino Hills are not unequivocally identified, the continuous pattern of large animal kills suggests that movement between these semi-isolated patches is possible (Fig. 5), and this brings up the need for consideration of regional wildlife movement in this area when environmental impacts are being evaluated.

Results of the study suggest that road kill records do provide a "window" to the past that can and should be used when environmental analysis addresses issues of impacts. The shift in road kill occurrences in the Walnut area from 1986 to 1990/1991 suggests that build out and increasing population density had significant effects on animal movement that should be further analyzed (Fig. 8). Although an exact cause and effect relationship between shifts in animal movement evidenced by road kill records and population density changes is difficult to make, there are several possibilities these data suggest. First, while it is true that 1986 was a wet year compared to 1991 leaving open the argument that shifts in movement are attributable to increased numbers of moving animals, it is unclear why absolute numbers of animals killed did not change significantly in the Walnut area but did in the Diamond Bar area (Fig. 7). If changes in wild animal population density are solely responsible for shifts in road kill occurrences, then why doesn't the Walnut area show a similar trend to Diamond Bar with respect to both abundance of animals hit and locations of road kills in 1990/1991 compared to 1986? Second, locations and types of road kills in the Turnbull Canyon and Colima corridors are similar in 1986 and in 1990/1991 (Fig. 6), but numbers of animals killed were lower in 1990/1991 (Table 1). If differences in population density attributable to increases in rainfall explain the lower numbers of animals killed in 1990/1991, such differences in climate do not explain the similarity in locations of road kills between these two years. Potential corridors of movement in the Whittier Hills were identified in an environmental report prepared for the City of Whittier in 1979. The report identified wildlife corridors on the basis of existing ridgelines and riparian corridors (L.D. King 1979). Results of this study suggest that movement is occurring in areas not identified in the report. This may be due to changes in movement patterns or to an insufficient database available to the preparers of the EIR.

Arguments concerning the value of wildlife corridors to the effort of preserving biodiversity point out that corridors are difficult to define since species differ in behavior (Soule and Simberloff 1986). In addition, conservation dollars may be better spent on the acquisition of larger, more ecologically significant areas (Simberloff and Cox 1987). This purpose of this study is not to advocate a particular position on the allocation of conservation funding. However, if substantial open space has already been preserved, such as is the case in the Verdugos and the Puente Hills, then spending a little more money to ensure that these already protected areas retain their usefulness as habitat makes sense. At the very least, any assessment of impacts of further development should take this point into consideration.

The limitations of these data are clear, but this study does suggest that road kill records may provide clues about animal movement at relatively little expense. While road kill records will never replace tracking, photography and trapping, the information can be used to pinpoint areas that require further information when regional planning issues are under consideration. Most importantly, road kill records have the advantage of providing a low cost regional look at habitat linkages that are often guessed at in the environmental review process.

With the accelerating loss of open space and the accompanying fragmentation road kill records should become an important data base in the evaluation of impacts on habitat quality carried out during the environmental review process.

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Habitat Corridors in the Eastern Santa Monica Mountains

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Abstract. Wildlife migration corridors provide the only natural means of maintaining regional genetic diversity within many isolated, fragmented habitats. It is generally agreed that species migration between source areas along such migration corridors will not readily occur if corridors are not environmentally similar to the source areas.

The easternmost Santa Monica Mountains, inclusive of Griffith Park, has become a fragmented habitat because the San Diego and Hollywood Freeways effectively cut the area off from the rest of the mountain range.

This paper focuses on the historic Mulholland Drive overpass on the Hollywood Freeway. This is the remaining link connecting the Mulholland Scenic Corridor (and the rest of the mountain range south of the bridge) with Griffith Park and its surrounding open space areas to the north. This area is in danger of being completely cut off from the rest of the Santa Monica Mountains unless habitat linkages are provided and maintained.

Introduction

The Santa Monica Mountain Range parallels the Pacific Coast of southern California in Ventura and Los Angeles Counties at 34°05'N latitude (Fig. 1). It stretches for a distance of about 70 km from Ventura County into the heart of the city of Los Angeles. At its western extent it measures 15 km in width, narrowing to about 4 km at its eastern boundary. Much of the southern boundary is delineated by the Pacific Ocean, and the northern boundary by the Ventura Freeway (Hwy 101).

The densely populated eastern section of the mountain range is separated from the more rugged and more sparsely populated central and western sections by the San Diego Freeway (Hwy 405) and Sepulveda Boulevard which parallels the freeway either on the east or west. Getty Center Drive, Sepulveda Boulevard, and Bel Air Crest Road N underpasses, as well as Skirball Center Drive and Mulholland Drive overpasses, provide the only potential migration corridors. However, Bel Air Crest Road N, the best of the above-named five migration routes, is presently blocked by the Bel Air Crest gated community. Also, the southernmost underpass, the entrance to the Getty Fine Arts Center now under construction, will be permanently obstructed, while Skirball Cultural Center and Hebrew University will act as a barrier to the open space west of Skirball Center Drive. This will leave Sepulveda Boulevard underpass and Mulholland Drive overpass as the only potential wildlife migration routes.

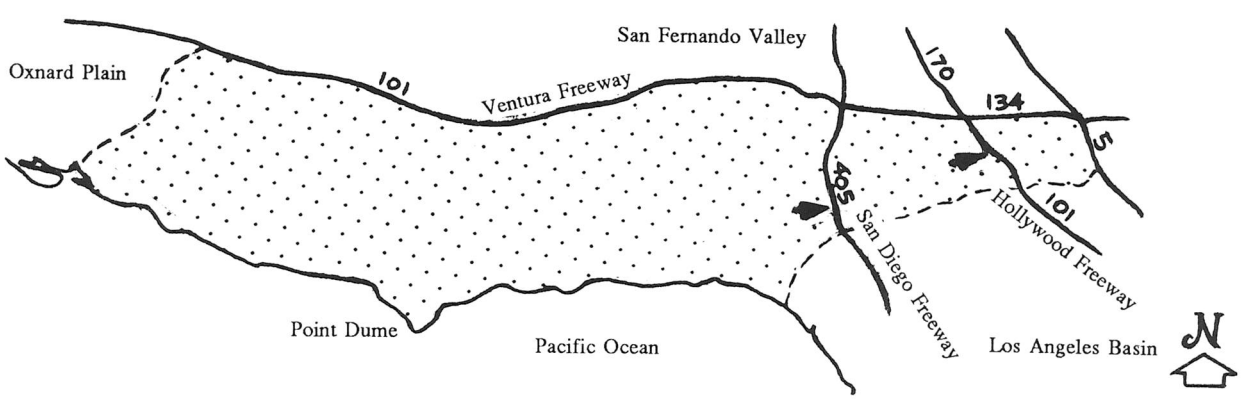

Figure 1. The Santa Monica Mountain Range in Los Angeles County.

With cooperative planning by all private landowners, as well as public agencies, all five potential corridors could again become effective wildlife corridors. This, however, is not the case in the far eastern end of the mountain range, which is the focus of this paper. As the mountain range narrows, it is further dissected by the Hollywood Freeway (Hwy 101) as it changes directions from east to south and cuts through Cahuenga Pass on its approach into Hollywood and downtown Los Angeles. The freeway forms another formidable wildlife migration barrier and (except for a fragile link) effectively isolates the easternmost section of the Santa Monica Mountains from the rest of the mountain range. The fragile link is the historic Mulholland Drive overpass over the Hollywood Freeway and it presents a bottleneck to migration. The overpass connects the Mulholland Scenic Corridor (and open space) in the eastern Santa Monica Mountain range with a still-undeveloped wildlife corridor leading to Griffith Park near the heart of the city of Los Angeles.

Wildlife Migration Corridors and Habitats

Habitat linkages or wildlife migration corridors are the only natural means of maintaining regional genetic diversity of plant and animal life within many isolated, fragmented habitats. Migration assures that species can maintain populations without inbreeding and that gene pool diversity is at least occasionally introduced.

Manion (1989), in reviewing the literature on habitat linkages in relation to habitat reserves (parklands) in the Santa Monica Mountains, stressed that each reserve must have a network of multiple habitat linkages of continuous native vegetation and natural topography. Habitat linkages should also be composed of entire watersheds, as wide as possible, and must connect riparian habitats. Species migration between source areas along migration corridors will not readily occur if corridors are not environmentally similar to the source areas. This assures that the corridors do not selectively discriminate against any species.

Edelman (1990) evaluated critical wildlife corridor/habitat linkages between the Santa Monica Mountains, the Santa Susanna Mountains, and the Simi Hills. He found that equestrian tunnels can provide effective habitat linkages for target species such as mule deer, bobcat, and coyote, although passage by grey fox and ringtail cat were rare. Mule deer and coyotes also represented the highest number of road kills along freeways, with bobcats a distant third.

Harrison (1992) pointed out that mammals will be most likely to utilize inter-refuge corridors during natal dispersal because once dispersal is completed by most adult mammals (with the exception of migrators), they will remain in the general area for the duration of their lives. Thus, he believes that examination of movement patterns during natal dispersal can provide insights into the requirement for corridors.

The minimum size of a habitat fragment required by a species depends on many factors, primary among these being its size, mobility, and location within the food chain. Mountain lions, requiring a large range and an ample supply of food such as deer, are therefore extinct in the eastern as well as most of the central Santa Monica Mountains. Foxes and ring-tail cats are also becoming extinct in many fragmented areas, while bobcats seem to hold their own for a longer period of time. Based on the observations by this author in living for over 15 years at the wildland/urban interface in the Santa Monica Mountains, deer, coyotes, raccoons, and wood rats have been able to maintain themselves in shrinking habitats, as they have learned to co-exist with, and even benefit from, human developments.

Birds with low metabolic rates, and plants, generally can persist much longer in shrinking territories than mammals and birds with high metabolic rates.

The terms "wildlife corridor" and "open space," while often used interchangeably, are not necessarily the same. A wildlife corridor presumes habitat linkages for a cross section of the wildlife present in an area, whereas open space describes any area not directly impacted by development, even though habitat destruction may have occurred, such as through required brush and weed clearance.

Development Impacts: A Case Study

The remaining open space area just to the east of the historic Mulholland Drive overpass (and adjacent to the Upper Hollywood Reservoir) is recognized by environmental groups not only as the only viable wildlife corridor still connecting Griffith Park with the Mulholland Scenic Corridor but also as an environmentally sensitive habitat area (ESHA).

A developer owning 176.5 acres of open space/wildland areas proposed (Proposal "N") to grade 105.1 acres for the construction of 64 single-family houses on one-acre lots within a private, gated community (City of Los Angeles 1990). The EIR acknowledged that the key open-space corridors would be seriously affected and that a large section of the ESHA consisting of about 38% of the California black walnut woodland and about 87% of the riparian areas on site would be eliminated.

The project site is described by environmental groups as the only viable wildlife corridor still connecting the Mulholland Scenic Corridor with Griffith Park via its only possible link, the Mulholland Drive overpass over the Hollywood Freeway (Hwy 101). Thus, the area is of

significance both as a wildlife migration artery and as a potential link for the cross-mountain trail system connecting the Mulholland Scenic Corridor with Griffith Park. Without proper implementation of a feasible wildlife corridor as well as a connecting section to the trail system, such links will be permanently severed by this development.

This author was asked to accomplish the latter tasks within the limitations of proposal "N". He cautioned that despite his mitigation proposals, scaling down the proposal to preserve key habitat areas should not be ruled out and that the alternative of preserving the area as permanent open space through public purchase may be the only alternative to truly retain the area as a functional wildlife migration artery.

Creating A Functional Wildlife Corridor

Key concepts of a proposed wildlife corridor are a circular flow system that is protected through permanent wildlife corridor and circulation easements. The circular flow system enables migrating animals approaching the Mulholland overpass from the north to backtrack in several key areas, as well as between many lots. This prevents animals from being trapped and also creates more variety of habitats.

Specifically, the proposal consists of the following:

1. *Wildlife corridor easements.* All areas within the tracts not taken up by developed lots, paved areas, and streets shall be designated through deed restrictions as wildlife corridor easements and can not be fenced nor otherwise obstructed in perpetuity. Specifically, such easements shall consist of all common areas and private parcels beyond the developed areas, as well as a 100-foot-wide and approximately 4,900-foot-long Department of Water and Power Public Utility easement corridor. The latter easement extends into the open space areas leading to Cahuenga Peak and Griffith Park on the north, and terminates near the Hollywood Freeway, approximately 500 feet southeast of the Mulholland Drive freeway overpass. This easement provides about 10 acres of open space.

2. *Circulation easements.* All small strips of common areas, as well as two small Department of Water and Power easements between private lots, shall be designated as wildlife corridor easements. Such easements provide for circulation within smaller open space areas as well as between cul-de-sac residential settings and open space areas and can not be fenced nor obstructed in any way, even by landscaping such as thick, dense hedges that may make passage by wildlife difficult.

   Since the development is proposed as a guarded community, it is important that easements also protect ingress and egress and movement through the development. Gates and fences will unnecessarily restrict the flow of wildlife, and if used, must be limited through deed restrictions strictly to paved streets for control of vehicular access.

3. *Tunnels.* Three-foot-high half culverts (six feet in diameter cut in half) will provide tunnels under streets for small mammals. Such half culverts and a few 12-inch to 18-inch full culverts placed in key locations and protected through effective screen planting can create cover and wildlife habitat niches.

4. *Ponds and Woodlands.* Shallow ponds shall be created in key areas and screened with dense woodland cover preferred by mule deer. Such areas will provide cover and water as well as resting and staging areas for the movement of deer and other wildlife along the wildlife corridor.

5. *Trail connection.* A 10-foot-wide hiking and equestrian trail easement in line with the primary wildlife corridor route shall connect Mulholland Scenic Corridor via the Mulholland Drive overpass with the open space areas north of the tract leading to Griffith Park. The trail easement will also serve as part of the circular trail network within the development and can be used as an occasional service road for the maintenance of the wildlife corridor and open space areas.

6. *Lighting.* The potentially adverse effects of lighting on any part of the wildlife corridors shall be mitigated by (a) low-intensity street lamps throughout the development and along its boundaries, as well as along the Mulholland Drive overpass; (b) low-elevation light poles where feasible; (c) shielding by internal silvering of the globe or external opaque reflectors; (d) use of directional lighting; and (e) prohibition of lighted tennis courts.

7. *Dog leash law.* All dogs must be leashed in the wildlife corridor and open space areas, inclusive of trails.

8. *Passive sports.* Only passive sports, such as hiking or jogging, are permitted within the wildlife corridor easements.

9. *Restoration.* In the final analysis, the extent of the restoration efforts will be dictated by the amount of riparian habitat and woodland habitat destroyed through grading. It is hoped that such destruction can be minimized, as any attempt to restore or replace the amount and quality of riparian and woodland habitats destroyed will be a time-consuming and costly process with no guarantee of success.

10. *Bonding of restoration project.* A 10-year restoration bond must be required to assure that the goals of every aspect of the restoration project are met and that professional management and maintenance will assure the long-term success of the project. This bond should also provide for replacement and accumulated management costs of any tree or shrub that dies within this time period for any reason (beyond a 10% attrition rate).

11. *Screen planting.* Effective landscaping inclusive of screen plantings shall be used along the sections of the wildlife corridor paralleling streets and roads, as long as it does not obstruct line-of-sight views required for safe driving.

12. *Department of Water and Power easement.* Wherever the Department of Water and Power easement crosses streets, unauthorized access by motorized vehicles may have to be blocked off with gates engineered for equestrian crossing (see Los Angeles City Parks and Recreation specifications). In one instance a wildlife overpass should be provided.

Key aspects of a functional wildlife corridor will hinge on the immediate establishment of limited but functional habitat niches through relocation of specimens of native oaks, walnuts, and tree-like shrubs, as well as the addition of ponds. The creation of oak woodlands, walnut woodlands, and oak savannahs in a more artificial setting, the limited restoration of riparian habitats, and the largely dry-site restoration of grassland, coastal scrub, and chaparral plant communities will take more time. Wildland fire safety regulations and slope stability landscape ordinances may be opposing factors to large-scale reestablishment of dry-site plant communities.

Summary

Several freeways have become effective wildlife migration barriers in the eastern Santa Monica Mountains. Continuous developments on either side of freeways are often characterized by massive grading that destroys riparian and woodland habitats and further chokes off the remaining wildlife migration arteries such as freeway overpasses and underpasses.

This author proposed measures to mitigate the destruction of the only remaining wildlife corridor linking Griffith Park with the eastern Santa Monica Mountains via the Mulholland Drive overpass should massive grading, and the destruction of most of the riparian and woodland habitats, become a reality. While many of the measures proposed are innovative and will benefit targeted species, such as mule deer, coyote, raccoons and opossums, they do not replace the riparian and woodlands destroyed. If not properly funded and managed, the proposed wildlife corridor may also degenerate into just an open space area over time. Thus, the author cautioned, that despite his mitigation proposals, scaling down the proposal to preserve key habitat areas should not be ruled out and that the alternative of preserving the area as permanent open space through public purchase may be the only alternative to truly retain the area as a functional wildlife migration artery.

The City of Los Angeles does not police the deed restrictions as proposed for the tracts and individual lots to assure a viable wildlife corridor in perpetuity. Thus, the burden of enforcement and policing would necessary fall on private citizen and public interest groups.

Acknowledgments. The author wishes to acknowledge the personal communication initiated by Animal Control Officer/Wildlife Specialist Larry Morales of the Los Angeles City Department of Animal Regulation regarding the author's proposed mitigation measures to Proposal N. Mr. Morales subsequently issued a report with general recommendations that raised many questions of concern to both parties and led to further reviews of the site by an independent biologist as part of the City's public review process. It is likely that major changes in the project's proposed design will emphasize reduction of the massive grading proposed and will thus preserve a much greater portion of the area's riparian habitats and subsequently key sections of the wildlife corridor.

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The Need For Corridors Between Coastal Wetlands and Uplands in Southern California

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Abstract. Endangered lightfooted clapper rails (*Rallus longirostris levipes*) were surveyed during the breeding season, 1980 - 1992, throughout their United States range which is in California from Santa Barbara County south to the Mexican border. In 1992 there were 275 pairs detected in 13 coastal marshes and 87% of them were in three, Upper Newport Bay (136 pairs) and Seal Beach National Wildlife Refuge (36 pairs) in Orange County, and Tijuana Marsh National Wildlife Refuge (67 pairs) in San Diego County. Of these three, Upper Newport Bay is the only one that has consistently maintained a large number of clapper rails, up to 71% of the state total, and it is threatened by isolation and the same consequences of mesopredator release that nearly extirpated the rail from Seal Beach National Wildlife Refuge. Coyote (*Canis latrans*) visitation to Seal Beach National Wildlife Refuge was interrupted and population irruptions occurred in the smaller predators, particularly non-native red foxes (*Vulpes vulpes*). Heavy predation reduced the clapper rails to 5 pairs in 1986 on Seal Beach National Wildlife Refuge. Over 300 foxes were trapped off Seal Beach National Wildlife Refuge and environs and the rails rebounded to 159 individuals by fall 1992. At Upper Newport Bay, coyotes were monitored with radio telemetry to better understand their habitat use and to plan for their continued presence. The need for interconnectedness between southern California's small coastal wetlands and much larger habitat blocks is emphasized. The alternative is intensive perpetual management or the loss of native wildlife including endangered species.

Keywords: Coastal marsh; corridor; coyote; endangered species; lightfooted clapper rail; mesopredator release; southern California.

Introduction and Objectives

Habitat fragmentation and isolation, particularly following its significant reduction, has resulted in extirpations of species (Burgess and Sharpe 1981; Diamond 1972; Lynch and Whigham 1984; Soule et al. 1979; Soule et al. 1988; Terborgh 1975; Whitcomb et al. 1981; Wilcove et al. 1986). This faunal collapse or reduction is often associated with mesopredator release; in the absence of native top carnivores, populations of smaller omnivores and predators explode (are released), becoming 4 to 10 times more abundant than normal (Eisenberg et al. 1979; Emmons 1984; Glanz 1982; Terborgh and Winter 1980). Mesopredator release has been implicated in local bird extinctions in several localities (Whitcomb et al. 1981; Wilcove et al. 1986) and has been implicated in scrub habitat patches in coastal southern California (Soule et al. 1988).

Exclusion of the coyote (*Canis latrans*) has led to explosions in feral cat (*Felis domesticus*) and fox (*Urocyon cinereoargenteus*) numbers and the associated high levels of predation in southern California shrubland islands (Soule et al. 1988). Coyotes occupy large home ranges (Bekoff 1977) and a variety of habitats (Ingles 1965). Given the fragmentation of much of southern California's remaining open space and habitats, it is quite likely that a variety of areas and habitats are only accessed sporadically by coyotes.

Coastal wetlands are of particular concern in this regard because of the concentrations of associated wildlife, including many endangered species. The southern California coastal marshes that still support listed and candidate species range in size from about 5 ha to 530 ha (U.S. Fish and Wildlife Service 1979). Most have been greatly reduced by human development and are isolated, or nearly so, from inland habitats. The larger marshes are separated from one another by about 22 km to 110 km with intervening development from Goleta Slough in Santa Barbara County to Tijuana Marsh on the Mexican border, as defined by the historic range of two of their endangered inhabitants. In addition, a complacency exists concerning self-maintenance of coastal marshes and even the endangered species dependent upon them, perhaps due to the visible abundance of wildlife.

The objective of this paper is to document the effects of mesopredator release on an endangered resident of a coastal marsh and present a preliminary accounting of coyote movements affecting the marsh that contains the largest remaining population of this endangered bird, the lightfooted clapper rail (*Rallus longirostris levipes*).

Study Area and Methods

The Seal Beach National Wildlife Refuge in Orange County covers 369 ha of the 2,024 ha Seal Beach Naval Weapons Station. About 299 ha of the refuge lands are subject to regular inundation by the tides. There are about 229 ha of salt marsh vegetation, 24 ha of mudflats that are exposed daily, and 46 ha of channel and open water. The wetlands are fully tidal, with a range of about 0.5 m to + 2.2 m MLLW (Mean Lower Low Water), and very productive with a high diversity and abundance of wildlife.

Upper Newport Bay is an Ecological Reserve of the California Department of Fish and Game, located in Orange County approximately 22 km downcoast of the Seal Beach National Wildlife Refuge. Approximately 304 ha are fully tidal, including 105 ha of marsh. The bay is flanked by bluffs 9-18 m high and surrounded by houses and roads. There are approximately 100 ha of shrublands remaining undeveloped on the edge of the wetlands and two local drainages with some cover along them coursing into the bay.

The Tijuana Marsh National Wildlife Refuge is located much further south in San Diego County.

Lightfooted clapper rails were censused annually during the onset of breeding by call counts, 1980 1992, throughout their California range from Santa Barbara County south to the Mexican border (Zembal 1992). In addition, counts were done each year by direct observations during postbreeding high tides in the fall or winter on the Seal Beach National Wildlife Refuge. Mesopredators were trapped and removed by personnel of the USDA Animal Damage Control beginning in 1986 (U.S. Fish and Wildlife Service and U. S. Navy 1990). Most of the trapping was with padded leghold traps in the spring and summer.

Coyotes were monitored at Upper Newport Bay through observation of indirect sign, night-lighting, and radio telemetry in 1990 and 1991. There were five trapping sessions with padded leghold traps, 10 April — 27 October 1990 with 4—35 traps set for 57 nights and 682 trap-nights; and 4—20 traps were set on 52 nights, 4 June—24 July 1991 for a total of 466 trap-nights. In addition, two large box-traps were set for approximately 80 trap-nights, 2 February—7 June 1991. Two 30 lb male coyotes were collared, one each year, and 4 pups were released unmarked. Ninety-nine locations were accrued on the first animal, 12 June—16 November 1990, with daily locations on the second animal 24-31 July 1991.

Results and Discussion

With the disappearance of the coyote from the Seal Beach Naval Weapons Station and Seal Beach National Wildlife Refuge in the 1970s, nonnative red foxes (*Vulpes vulpes*), feral cats, and striped skunks (*Mephitis mephitis*) proliferated, resulting in heavy predation on lightfooted clapper rails and other wildlife (U.S. Fish and Wildlife Service and U.S. Navy 1990). The foxes were a most difficult problem due to their preference for eggs, eggcaching behavior, hunting ability, non-hesitation to hunt in the marsh, abundance, avoidance of traps, and the lack of appropriate defensive responses by the rails. The clapper rails on the Seal Beach National Wildlife Refuge were reduced from 30 breeding pairs in 1980 (15% of the state total) to 5 pairs (3.5% of the state total) in 1986.

The trapping of red foxes from the Seal Beach National Weapons Station peaked in 1988 with the removal of 128 foxes during that one summer. The clapper rail's response to the reduction in predation pressure was manifest during the post-breeding high tide counts (Fig. 1). The 1992 count of 159 clapper rails is the highest single count ever recorded for the Seal Beach National Wildlife Refuge. Since 72 (73%) of the 98 clapper rails counted in the Fall of 1991 survived the winter to breed in 1992, it is predicted that, with similar winter conditions, 58 pairs will breed in 1993.

Since 1980, up to 71% of the state's breeding population of lightfooted clapper rails has been detected at Upper Newport Bay. In 1992, 136 pairs or 49.4% of the state's 275 breeding pairs were present, compared to the 36 pairs in the Seal Beach National Wildlife Refuge. Breeding clapper rails were detected in only 13 marshes in 1992 and most of those not residing at Upper Newport Bay or Seal Beach National Wildlife Refuge were found in the Tijuana Marsh National Wildlife Refuge (67 pairs). Upper Newport Bay has consistently maintained a large breeding population of clapper rails over the past decade (Zembal 1992). Although each of the inhabited wetlands is important, Upper Newport Bay is perhaps most critical to the continued survival and eventual recovery of the lightfooted clapper rail.

Coyotes are being monitored at Upper Newport Bay to help avoid a repeat of the circumstances that nearly extirpated the clapper rail from the Seal Beach National Wildlife Refuge. Concerns for the potential of mesopredator release occurring at Upper Newport Bay already exist. There are recent red fox sightings on one

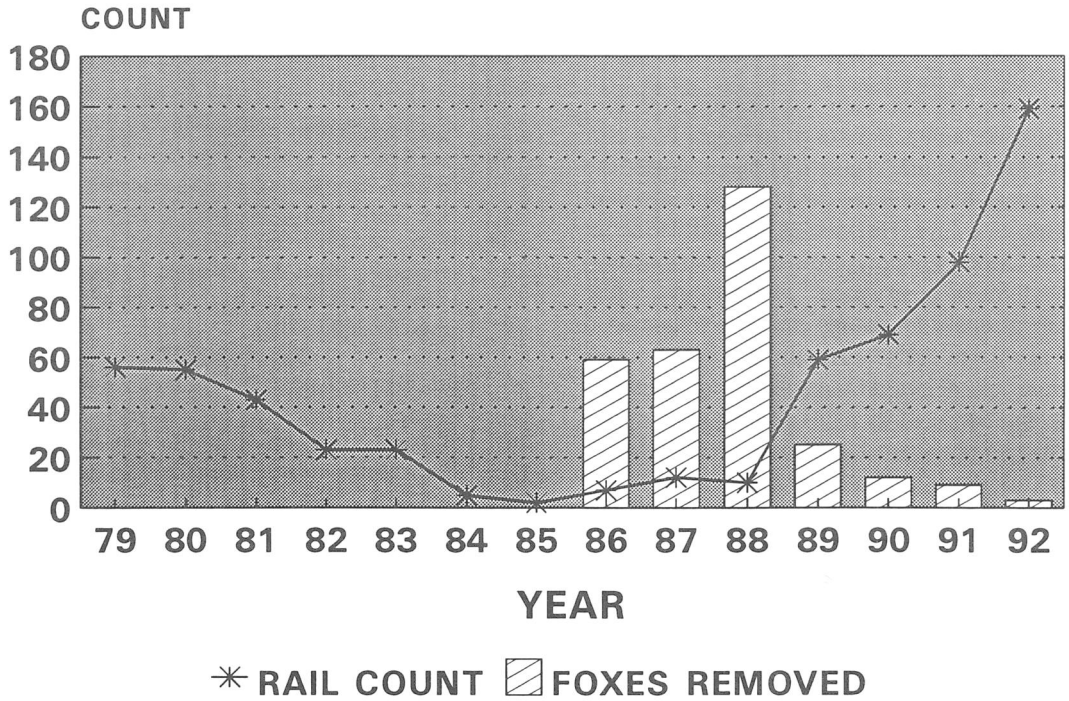

Figure 1. Clapper rails counted and red foxes removed from Seal Beach National Wildlife Refuge. Counts are high tide surveys.

side of the bay where there is little coyote activity. Also, coyote visitation has been highly variable for the past several years. For example, the two coyote trapping stations were at either end of Upper Newport Bay, about 4 km apart, and visitation by coyotes varied from daily to every four days. Additionally, at least four coyotes of this potentially small suburban population were killed during the study period. Two, including a lactating female, were hit by cars and two were shot. Lastly, the corridors into, and the uplands along, Upper Newport Bay are already narrow and limited with the potential of continued reduction for additional development.

The coyote collared in 1990 (Rick) traveled the length of Upper Newport Bay and along the Big Canyon corridor to about 4 km out of the site on Spyglass Hill. About 40% of this animal's daytime locations were in the remaining uplands along the edge or within 1 km of the bay. His daybeds and activities were in coastal sage scrub on the bluffs bordering the east side of Upper Newport Bay, or in coastal sage scrub and ornamentals along an exclusive golf course/housing development in Big Canyon, 2 to 3.5 km from Upper Newport Bay. He bedded 42% of the time in riparian woodlands in Big Canyon or on the edge.

Rick's 1990 den was in a 32 ha field on the east side of Upper Newport Bay at the oceanward end. There was also a den on the edge of this field in 1991. This is as far down the bay as any site provides good cover, food, water, and relatively little disturbance; the Newporter Resort, Newport Dunes Recreation and Conference Area, and a marina are just beyond.

Rick was never detected in the marsh or on the west side of the remaining uplands. However, even including the entire Upper Newport Bay area in his potential range results in a total of less than 1,000 ha. This is small for a coyote and is probably only a fraction of his year-round travels. The Big Canyon corridor he traveled is nearly dead-ended. A jaunt through a cemetery and along a busy 4-lane road for about 0.5 km could get him to additional fragmented open space and eventually to several thousands of hectares. Whether this connection is ever made requires additional study.

The more viable corridor into Upper Newport Bay, along San Diego Creek at the inland edge of Upper Newport Bay, provides a wider, less interrupted connection to the thousands of hectares of open space remaining in this part of Orange County. San Diego Creek is up to about 150 m wide and provides a mix of low herbaceous growth, a shrubby riparian thicket 40 m wide, and a small low-flow water course. The current corridor is as much as 1.2 km wider than the creek bed with adjacent open space and habitat including a freshwater wetland, ponds, open water treatment facility, and a golf course along the first 2.2 km. However, there are bottlenecks at three major roads crossing this section. The creek bends north at the golf course and the major connection with most of the open space is through a 150 ha, 3.5 km long park to the east. There is limited cover along the first 1.1 km of

park. Rick was never found along this corridor, but the 1991 coyote (Ster) was.

Upon release, Ster ran from San Diego Creek to a daybed in riparian habitat 0.5 km from Upper Newport Bay. By the time his signal was lost, he had traversed the park, using riparian as daytime cover on four days and shrublands on three days. He traveled gradually to about 9 km out from Upper Newport Bay in the midst of several thousand hectares of open space. Interestingly, while tracking him at dusk, he passed under a 4 lane road through a 5 m x 2.5 m culvert. About 1 km of travel was along a narrow belt of riparian and uplands, no more than 250 m wide and bordered by a busy 4 lane road on one side and a steep slope topped by houses on the other.

Coyotes continue to access Upper Newport Bay and recent sign indicates that visitation may have increased. It is also encouraging to note the use of remnant habitat, as along Big Canyon. However, as major pieces of habitat continue to be lost, and the number of coyotes is reduced, fewer would be expected to continue to venture through narrow, marginal habitat areas. The wider and more viable the corridors are kept and the more uplands that are retained at Upper Newport Bay, the more assurance that coyote presence will be maintained in the long term.

Habitat viability and carrying capacity, even for endangered species in coastal wetlands, can be greatly reduced by significant losses of interconnectedness with other habitats. Because our remaining coastal wetlands are so small, viable corridor connections must be maintained or restored between them and much larger parcels of upland habitats. Based upon preliminary evidence in Orange County, it is apparent that coyotes will deal effectively with bottlenecks, given large and frequent enough habitat bulges along the way and at either end. Since a coastal wetland is liable to be the terminus of continuous open space used by coyotes, it is vital that their requisites be maximally met as near the wetland as possible to maintain their regular presence. This highlights the importance of adjacent upland buffers with enough low level disturbance that denning sites are kept available. Some of the effects of diminution and fragmentation of southern California's coastal wetlands must be compensated by the provision of enough open space in viable habitat to maintain a strong presence by our native top carnivores. Otherwise, those wetlands cannot be expected to maintain large populations of native wildlife, including endangered species, without intensive perpetual management.

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