APPENDIX A-8

Population Viability Analysis for the Coastal Cactus Wren within the MSCP Study Area

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Prepared for
The Clean Water Program
City of San Diego

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1.0 INTRODUCTION

The objective of a population viability analysis (PVA) is to identify all threats, natural and human-caused, to a population and determine if these threats endanger the continued existence of the population. This PVA is closely related in scope and format to the PVA prepared for the California gnatcatcher (Ogden 1992). These two bird species co-occur in sage scrub habitats, but the cactus wren has the additional requirement of nesting only in tall cactus. This report discusses the the status, biology, and threats to the coastal cactus wren within the Multiple Species Conservation Program (MSCP) study area.

2.0 BACKGROUND

2.1 DECLINE IN POPULATION OF THE COASTAL CACTUS WREN

The coastal cactus wren (*Campylorhynchus brunneicapillus*) represents a disjunct population of a widely distributed and relatively common desert-dwelling species. The distributional limits of the coastal cactus wren are entirely within the coastal slope of southern California and northern Baja California, including the counties of Ventura, San Bernardino, Los Angeles, Orange, and San Diego. The southern portion of the coastal cactus wren population has been proposed as a distinct subspecies (*C. b. sandiegensis*; Rea and Weaver 1990). The southern California coastal slope has undergone extensive agricultural and urban development over the past 150 years, which is the primary cause for the decline and fragmentation of this regional wren population (Grinnell and Miller 1944). Rea and Weaver (1990) documented a loss of 33% of the 78 sites known to be occupied by the presumptive *C. b. sandiegensis* in southern Orange County and San Diego County during the mid-1980s. Cactus wren subpopulations elsewhere in the coastal slope are known to have been severely impacted by development (K. Garrett, B. McKernan, L. Salata, F. Roberts pers. comms.).

There is limited information on the abundance of coastal cactus wrens within each county. Gross population estimates by local ornithological experts place the total population estimate at between 1,900 and 2,500 pairs, with the majority of birds (65%) in Orange County (Table 1).

Table 1

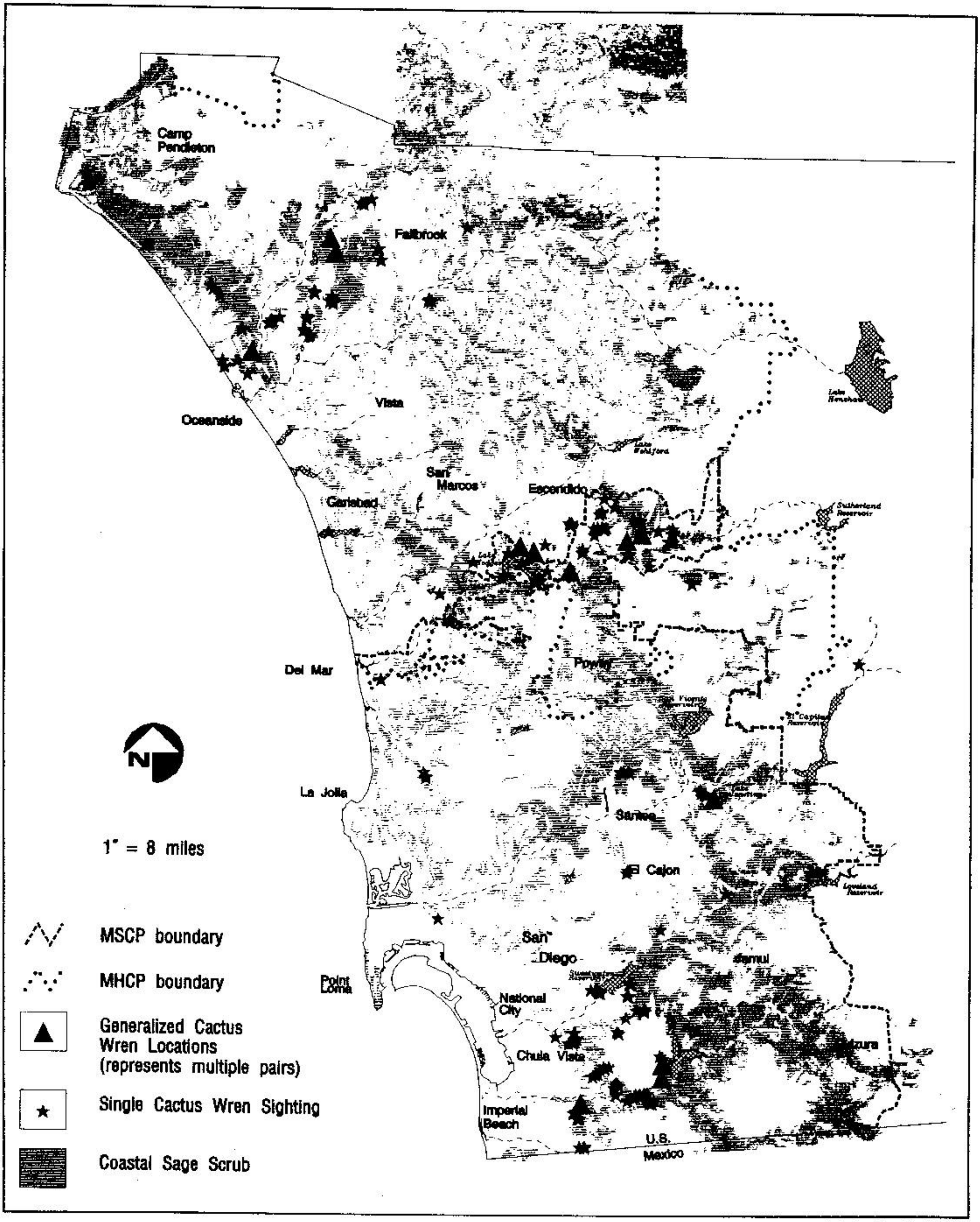
POPULATION ESTIMATES OF COASTAL CACTUS WREN WITHIN EACH COUNTY

County	Number of Pairs	Source
Ventura	< 200	Garrett 1992
Los Angeles	125 - 160	Garrett 1992
San Bernardino	40 - 70	B. McKernan pers. comm.
Riverside	65 - 150	B. McKernan pers. comm.
Orange	1200 - 1600	L. Salata pers. comm.
San Diego	< 270	Rea and Weaver 1990
Total	1900 - 2450	

Within San Diego County, the coastal cactus wren is limited to five substantial subpopulations having more than 20 pairs (Figure 1): Camp Pendleton (70 pairs), Lake Hodges/Wild Animal Park (90 pairs), Santee/Lake Jennings (25 pairs), Sweetwater River (25 pairs), and Otay River/Mesa (55 pairs). Other isolated localities each support less than 5 pairs (Rea and Weaver 1990). The MSCP study area encompasses about 200 pairs of cactus wren distributed primarily in four subpopulations. The potential for undocumented subpopulations of substantial size within San Diego County is believed to be low (K. Weaver pers. comm.).

2.2 CURRENT STATE AND FEDERAL STATUS OF THE COASTAL CACTUS WREN

To date, the coastal cactus wren is not listed by federal or state resource agencies. It is presently a federal Candidate 2 species and a state Species of Special Concern. On September 18, 1990, the San Diego Biodiversity Project and Palomar Audubon Society filed a petition with the Portland Regional Office of the U.S. Fish and Wildlife Service



OGDEN

Known Distribution of Coastal Cactus Wren
in San Diego County

FIGURE

1

(USFWS) to list the coastal cactus wren as endangered under the federal Endangered Species Act of 1973. A preliminary decision on whether listing of this species may be warranted is pending and is expected soon (L. Salata pers. comm.).

2.3 CACTUS WREN BIOLOGY

There are few quantitative studies of cactus wren biology for the coastal population (Rea and Weaver 1990), but some focused studies have recently been initiated (L. Szijj, B. McKernan pers. comms.). The majority of the species' biology is known from studies of desert populations in Arizona, New Mexico, and California (e.g., Anderson and Anderson 1973, Marr and Raitt 1983, Miles 1990, Simons and Simons 1990, Simons and Martin 1990).

The coastal cactus wren is a moderate-sized (33 to 47 g), year-round resident species that is dependent upon tall (>1 m in height) cactus (*Opuntia* spp.) for nest and roost sites. Arboreal cacti provide the firm support required for the wren's large, pouch-shaped nests used for roosting and breeding. In San Diego County, three *Opuntia* cactus species are used by wrens (*O. prolifera*, *O. littoralis*, and *O. oricola*). Wren-occupied cactus patches are found within coastal sage scrub, a habitat considered to be increasingly rare and threatened by development (O'Leary 1990). Rea and Weaver (1990) suggest that coastal cactus wrens tend to prefer sage scrub that is dominated by California sagebrush (*Artemisia californica*) and flat-top buckwheat (*Eriogonum fasciculatum*). Rea and Weaver (1990) found the size of 13 coastal cactus wren territories ranged from 2 to 5 acres (mean = 3.25 acres). In Arizona, Anderson and Anderson (1973) found wren territories ranged from 3 to 10 acres (mean = 4.75 acres).

Cactus wrens are primarily insectivorous birds, especially during the breeding season. Beal (1907) reported an examination of 41 cactus wren stomachs taken from the region between Los Angeles and San Bernardino during July to January. Insects and vegetable matter (fruit pulp and seeds) constituted 83% and 17%, respectively, of the diet. In Arizona, Simons and Martin (1990) reported a nestling diet of insects, consisting of mostly caterpillars, grasshoppers, and moths. Two quantitative studies of cactus wren foraging ecology have been conducted (Parker 1986, Miles 1990). Cactus wrens appear to forage on both the ground and in vegetation (Ricklefs and Hainsworth 1968, Parker 1986, Anderson and Anderson 1973, L. Szijj, K. Garrett pers comms.). Foraging substrate and technique of the cactus wren appear to vary seasonally (Miles 1990).

The breeding season for the coastal cactus wren extends from late February to August (Unitt 1984, L. Szijj, B. McKernan pers. comms.). Unitt reported egg dates ranging from March 14 to July 10 (n = 42 clutches) for coastal San Diego County. The domed nest is composed of grasses, leaves, small twigs, and rootlets (Harrison 1979) and is typically placed about 1 m above the ground (Rea and Weaver 1990). Cactus wrens typically build 4 to 6 nests within their year-round territory (Anderson and Anderson 1973). Cactus wrens often orient the entrance of the nest to take advantage of convective ventilation provided by prevailing winds (Austin 1974, Facemire et al. 1990).

During favorable years, cactus wrens can fledge two or three successful broods. Clutch size ranges from 2 to 5 eggs, with 3 or 4 eggs being the most common clutch sizes (Anderson and Anderson 1973, Marr and Raitt 1983, Simons and Martin 1990). The male has the primary responsibility of territorial defense while the female incubates the eggs. Both parents participate in feeding and protecting the young. The incubation period lasts for 16 days and nestlings fledge at 19 to 23 days of age (Anderson and Anderson 1973, Ricklefs 1975). Fledglings are dependent on their parents for food for four to six weeks and often remain within their parents' territory for several months thereafter. During this time, juveniles often participate in territorial disputes and occasionally help to feed siblings in second broods (Anderson and Anderson 1973).

Cactus wren nesting success is highly variable between years, being dependent upon fall/winter precipitation and predation rates (Anderson and Anderson 1973, Marr and Raitt 1983, Simons and Martin 1990). The coefficient of variation (CV) of annual productivity in a six-year study by Anderson and Anderson (1973) was 26.5% (overall mean = 4.3 fledglings per pair). No nest success data are available for coastal cactus wren populations.

Information on annual survival of adult and juvenile cactus wrens is limited to studies conducted in Arizona. Anderson and Anderson (1973) reported an overall adult survival rate of 50.6% for a six-year study. The coefficient of variation in annual adult survival rate for four years that had sufficient number of banded birds was 42.7% (mean survival rate = 52.8% per year). Less than 11% (n = 74 adult birds) of the Andersons' banded population survived to breed more than three seasons. The annual adult survival rate reported for cactus wrens is consistent with that of other songbird species (Karr et al. 1990).

Information on juvenile survival is much more difficult to obtain since mortality and emigration can not be easily distinguished. Survival of juvenile songbirds is considered to be typically low (e.g., Sullivan 1989). Anderson and Anderson (1973) reported juvenile cactus wren survival to be about 50% after one month of age and less than 15% beyond two months of age (n = 55 juvenile birds). Simons and Martin (1990) reported survival rates of juveniles after 4 to 6 weeks to be 48.5% (n = 33) and 20.5% (n = 34) during two different years. Fledging mass of surviving juveniles was significantly greater than that of nonsurvivors during the year with lower juvenile survival. For both of these studies, there is a potential to categorize birds that dispersed away from the study area as nonsurvivors. Known and potential predators of cactus wren include snakes, domestic cats, Cooper's hawks, American kestrels, roadrunners, and woodrats (Anderson and Anderson 1973, Austin et al. 1972, L. Szijj pers. comm.).

Information on the dispersal capacity of cactus wrens is very limited. Adult cactus wrens are considered to be highly sedentary, remaining in their territory for their entire adult life (Anderson and Anderson 1973). Sightings of vagrant individuals away from suitable habitat are rare (Unitt 1984, K. Garrett pers. comm.). Limited data from Arizona suggest that juvenile female cactus wrens tend to disperse farther away from their natal territories than juvenile males (Anderson and Anderson 1973).

3.0 POPULATION VIABILITY ANALYSIS

3.1 THREATS TO THE VIABILITY OF THE COASTAL CACTUS WREN POPULATION

3.1.1 Direct Human Impacts

In general, the greatest threats to wildlife populations come as a direct consequence of human activities such as habitat destruction, degradation, and fragmentation. The first two directly reduce population size while the impact of fragmentation may be more subtle (Wilcox and Murphy 1985, Rolstad 1991, Sanders et al. 1991). The fragmenting of habitat has been shown to lead to increased mortality in animal populations through a number of mechanisms. Fragmentation causes edges. Ecological edges can be beneficial to some species. However, edges between natural habitat and human-modified habitat are often associated with negative impacts that are generally detrimental to the value of the habitat to wildlife (Harris 1988, Laurence and Yensen 1991).

The physical conditions along the edge of a patch of vegetation are different from those in the center. For this and other reasons, the composition of the vegetation is usually different at the edge than at the core of the patch. Also, human impacts penetrate across the edge. Trampling of vegetation and path creation is usually greater near the edge of a patch. Exotic plants and animals invade the native habitat at edges. Introduced predators, such as domestic cats, and enhanced predators (natural predators whose densities are enhanced by the presence of humans) penetrate the native vegetation along edges (Churcher and Lawton 1987, Yahner et al. 1989). Because more of the area of a small patch is close to an edge than that of a large patch, the usefulness of small patches as wildlife habitat is reduced. Direct impacts can be controlled through habitat preservation and proper management, including buffers between the habitat and development.

3.1.2 Loss of Genetic Variation in Isolated Populations

Genetic drift refers to random changes in gene frequencies that occur from generation to generation in any finite breeding population of organisms (Crow 1986). Ultimately, genetic drift can lead to fixation at all genetic loci and a total loss of genetic variability (Franklin 1980). Small populations are expected to lose genetic variation as a consequence of genetic drift more rapidly than large populations, since sampling from a smaller number of genes each generation leads to greater fluctuations in gene frequencies than sampling from a larger number of genes. In small populations, the contribution of mutation is negligible and genetic drift is probably the most important factor affecting levels of genetic variation (Lacy 1987).

Loss of genetic variation in small populations has both short and long-term consequences (Lacy 1987). Over the short term, it will result in increased levels of genetic uniformity (homozygosity) within subpopulations and high levels of genetic differentiation between isolated subpopulations (e.g., Leberg 1991). In many organisms, increases in homozygosity can lower survival and diminish reproductive output. Elevated levels of homozygosity have been associated with poor growth, higher frequencies of disease, and decreased survival during periods of stress (Franklin 1980; Soulé and Simberloff 1986). Although information on the degree of inbreeding that negatively affects long-term survival is lacking for most species, even small amounts of inbreeding can have deleterious effects (Soulé and Simberloff 1986). Because birds and mammals have relatively low rates of reproduction, they may be more sensitive than other organisms to the negative effects of inbreeding (Franklin, 1980).

Over the long-term, depletion of genetic variation in small populations could lead to an inability to adapt to changing environmental conditions, rendering organisms more vulnerable to new predators, parasites, and diseases (Lacy, 1987). Genetic models suggest that the amount of genetic variation maintained in a population is inversely related to its rate of extinction (Lande and Barrowclough 1987). In a review article on evolutionary change in small populations, Franklin (1980) concludes that in order to preserve genetic variation for complex traits, effective population size in the short-term should not be less than 50 individuals and in the long-term should not be less than 500 individuals. Loss of genetic variation in fragmented cactus wren subpopulations is of concern since the dispersal capability of juvenile cactus wrens is unknown and is likely to be a constraining factor.

3.1.3 Demographic Variability

Demographic variability is simply chance events that independently affect the survival and reproduction of individuals within a population. These are most important in very small populations. For instance, in a very small population (e.g., ten individuals), it is possible all ten individuals would die in a single year, independent of any climatic effects; or all offspring born during a given period would be the same sex and if all the adults died, the population would be doomed because it would consist solely of the same sex. These types of demographic anomalies are only a serious threat to very small, isolated populations (<20 individuals; Soulé 1983, Pimm et al. 1988, Tracy and George 1992), as is the case for some of the cactus wren subpopulations in San Diego County.

3.1.4 Environmental Variability

Environmental variability is simply the natural vagaries of climate. Year to year variation in temperature, precipitation, and food supply affect the survival and reproduction of organisms. The viability of most populations decreases with greater environmental variation. A long series of bad years (i.e., years in which survival and/or birth rate are low) may threaten the existence of a population. In the absence of human-caused threats, environmental variation is probably the greatest threat to population viability.

Annual fluctuations in birth rate and survival are the expression of environmental variability. Long-term population data, which would allow us to directly estimate survival, fecundity, and how population density varies from year to year, do not exist for the coastal cactus wren population. Instead, we have considered the information that is available in the literature on fluctuations in songbird populations.

The traditional view is that annual variation in reproductive rate of arid region songbird populations is driven by variation in precipitation. It has been amply demonstrated that primary productivity in arid zone vegetation is tightly correlated with precipitation. The density of plant-feeding insects upon which insectivorous birds, such as the cactus wren, feed also varies with precipitation. Thus, the level of precipitation determines the amount of food resources available to the birds. This should affect not only birth rate, but survival as well. Some empirical work has shown a linkage between precipitation and reproductive parameters (e.g., Marr and Raitt 1983).

More recent studies have shown a more complex interaction between precipitation, nest predator density, and avian reproductive success. Rotenberry and Wiens (1989) found weak and often insignificant correlations between precipitation and reproductive parameters, including fledgling number, in three species of arid zone birds. They also found nest predation rates to be the most important factor determining the number of offspring successfully fledged. The density of nest predators, primarily snakes and ground squirrels, did not respond directly to annual precipitation, but rather the response was lagged in time. Ground squirrels at one of their sites reached very high levels only after two consecutive wet years. Based on their results, it appears that nest predation is an important determinant of reproductive success, and the relationship between precipitation and birth rate may be lagged in time. Nest predation has been shown to be an important determinant of reproductive success in a number of passerine birds. Nest failure due to predation is common in cactus wren populations (Anderson and Anderson 1973, Simons and Simons 1990). It is likely that cactus wren densities do respond to changes in precipitation; however, the relationship is likely to be a complex interaction with variation in predation pressure (Rotenberry and Wiens 1989, Lima 1987).

3.2 SPATIAL ARRANGEMENT OF COASTAL CACTUS WREN POPULATIONS IN THE MSCP STUDY AREA

The four major subpopulations of coastal cactus wren within the MSCP study area may be isolated from each other due to the relatively large distances between subpopulations (Figure 1). The Otay River and Sweetwater River subpopulations are sufficiently close to expect an occasional exchange of individuals. Population exchange between other subpopulations appears less likely due to greater dispersal distances and the presence of intervening development. Ornithological experts have suggested that coastal cactus wrens may be highly sedentary (K. Garrett, B. McKernan pers. comms.). Anderson and Anderson (1973) also believed that their study populations were sedentary. The lack of quantitative information on dispersal capability of cactus wrens makes evaluating connectivity between subpopulations problematical.

The distribution of suitable cactus habitat within the coastal slope may historically have been patchily distributed prior to extensive development (Grinnell and Miller 1944). Therefore, one might expect that the dispersal capability of coastal cactus wrens may be sufficient to allow for a moderate (e.g., about 1%) migration rate between adjacent populations. Cactus wrens are believed to have respectable flying abilities, and their moderate body size allows for sufficient storage of energy reserves necessary for dispersing between suitable habitat patches. Cactus wrens are similar to acorn woodpeckers in body size and both are dependent on nesting habitat that is patchily distributed at the landscape scale. Stacey and Taper (1992) report on their long-term study of an acorn woodpecker population in New Mexico they presumed to be isolated. They suggest that this woodpecker population is maintained by a regular influx of immigrating juveniles, presumably from the nearest subpopulation located nearly 27 miles away.

3.3 METAPOPULATION SIMULATION MODEL

As with our California gnatcatcher PVA (Ogden 1992), we are employing a metapopulation simulation program, RAMAS/space (Exeter Software, Setauket, New York) to explore the metapopulation dynamics of the cactus wren population within the MSCP study area. The RAMAS/space model is a generalized model that is best used in a comparative fashion to explore the relative merits of alternative reserve designs. The uncertainty of the model's input parameter values (i.e., long-term population averages and variances) and the degree of approximation required to realistically represent cactus wren population dynamics make this relatively simple population model primarily a heuristic tool. The MSCP cactus wren population is limited to about 200 pairs distributed unevenly between four subpopulations having more than 20 pairs each (Figure 1). An additional 200 pairs occur in two subpopulations north of the study area (Camp Pendleton and southern Orange County). The simulation model was run with and without these northern subpopulations.

We configured the wren model in a similar manner as the California gnatcatcher model; however, information on survival and dispersal capability are lacking for the coastal cactus wren. For survival rate, we used either the typical passerine value of 55% (Karr et al. 1990) or a conservative value of 40% in this set of simulations. Since the issue of connectivity between subpopulations is unclear, we set the migration rate between adjacent subpopulations at either zero (no migration) or one percent. The average rate of population increase (R) was set at either 1.15 or 1.2 with a 30% variance of R since these appear to be appropriate values for passerine populations (Ogden 1992). The carrying capacity of each subpopulation was assumed to be 1.2 times the initial subpopulation size. All simulations were run with a 200-year horizon for 100 iterations.

4.0 VIABILITY OF THE COASTAL CACTUS WREN METAPOPULATION IN THE MSCP STUDY AREA

The current coastal cactus wren population in the MSCP study area is about 200 pairs. We simulated this population size, divided into four subpopulations. The results of the simulation presented in Table 2 suggest the viability of the current MSCP population may be marginal. This outcome is likely due to the relatively small size of each subpopulation and the perceived low rate of exchange between adjacent subpopulations. The addition of two large subpopulations north of the MSCP study area to the model increased the metapopulation viability, meeting a 5% extinction threshold even without migration between subpopulations, except for one scenario.

The simulation model appears to be moderately sensitive to the annual survival rate parameter. The extinction rate was halved by increasing the survival parameter from 40% to 55%. Allowing for a small amount of exchange between subpopulations also reduced the extinction rate, although to a lesser degree (Table 2).

Table 2

SIMULATION OF CURRENT CACTUS WREN POPULATION WITHIN THE MSCP STUDY AREA (200 PAIRS) AND WITH TWO ADDITIONAL SUBPOPULATIONS (400 PAIRS)

Metapopulation Size (No. subpopulations)	R	%SD	% Survival	Migration Rate (%)	Extinction Probability (%)
200 (4)	1.15	30	40	1	35
200 (4)	1.15	30	55	0	23
200 (4)	1.15	30	55	1	14
200 (4)	1.2	30	40	1	16
200 (4)	1.2	30	55	0	9
200 (4)	1.2	30	55	1	7
400 (6)	1.15	30	40	1	8
400 (6)	1.15	30	55	0	1
400 (6)	1.15	30	55	1	1
400 (6)	1.2	30	40	1	2
400 (6)	1.2	30	55	0	1
400 (6)	1.2	30	55	1	< 1

5.0 CONCLUSIONS

Simulation models of metapopulations have shown that the dynamics of simulated metapopulations are driven to a large extent by the larger subpopulations. Smaller, more peripheral subpopulations can be lost without greatly affecting the viability of the entire system, but the loss or serious reduction of one or more of the larger subpopulations has a great effect on the overall viability of the model system (Gilpin MS). Unfortunately, the coastal cactus wren metapopulation within the MSCP study area is relatively small and severely fragmented.

The simple simulation model presented here used input values (e.g., survival and migration rates) that were educated estimates based on studies of desert-dwelling cactus wren populations and other songbird species. A more detailed simulation model requires population studies of the metapopulation of concern. This simple model suggests that small population size and habitat fragmentation constrain the long-term viability of the coastal cactus wren metapopulation within the MSCP study area in the absence of direct and indirect effects of development. Incorporating the adverse effects of development would further reduce the metapopulation viability. The largest subpopulations of coastal cactus wren occur in Orange County which may be interconnected with the Camp Pendleton subpopulation. However, the connectivity between the Pendleton subpopulation and the nearest MSCP subpopulation (Lake Hodges/Wild Animal Park) is likely to be low to nonexistent due to distance and existing development.

To achieve the MSCP objective of maintaining viable populations of the coastal cactus wren, most or all of the existing cactus wren subpopulations and habitat within the MSCP study area must be conserved and expanded by creation of additional cactus habitat. New subpopulations would also need to be established between currently occupied localities to further increase the size of the metapopulation and to promote connectivity between subpopulations. Information is required on coastal cactus wren demographics, especially in terms of dispersal capacity and annual variation in reproductive success and survival. A study of coastal cactus wren population genetics with regard to levels of homozygosity is also recommended.

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